Exponent®

Operable Unit 1
Ecological Risk Assessment
Ventron/Velsicol Site
Wood-Ridge/Carlstadt,
New Jersey

Prepared for

Velsicol Chemical Corporation
Rosemont, Illinois

Morton International, Inc.
Chicago, Illinois

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April 2001

Doc. No. 8600B3N.001 0403 0301 DS01

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Acronyms and Abbreviations

ACOE	U.S. Army Corps of Engineers
BAF	bioaccumulation factor
BEE	baseline ecological evaluation
BsedAF	biota-to-sediment accumulation factor
CERCLIS	Comprehensive Environmental Response, Compensation and Liability Information System
CoPC	contaminant of potential concern
DEHP	bis[2-ethylhexyl]phthalate
DQO	data quality objective
EPA	U.S. Environmental Protection Agency
ERA	ecological risk assessment
ER-L	environmental effects range-low
ER-M	environmental effects range-medium
ERA	ecological risk assessment
ERAGS	Ecological Risk Assessment Guidance for Superfund
ESA	environmentally sensitive area
HHRA	human health risk assessment
HMD	Hackensack Meadowlands District
HMDC	Hackensack Meadowlands Development Commission
IM	intermuscular
IVA	Indicator Value Assessment
LEL	low-effect level
LOAEL	lowest-observed-adverse-effect level
NHP	Natural Heritage Program
NEC	no-effect concentration
NJDEP	New Jersey Department of Environmental Protection
NOAEL	no-observed-adverse-effect level
NPL	National Priorities List
OC	organic carbon
OU1	Operable Unit 1
OU2	Operable Unit 2
PAH	polycyclic aromatic hydrocarbon
PCB	polychlorinated biphenyl
POTW	publicly owned treatment works
Resolution	Resolution of the Berry's Creek/Wood-Ridge Site Action Committee
RI	remedial investigation
RI/FS	remedial investigation and feasibility study
SAP	sampling and analysis plan
SMDP	scientific/management decision point
SQ	screening quotient
SRP	New Jersey Site Remediation Program
SSL	soil screening level

Stipulation	Stipulation and Supplementary Order Approving Cooperative Agreement for Remedial Investigation and Feasibility Study and Amending Procedural Order Involving Remedy
SVOC	semivolatile organic compound
TRV	toxicity reference value
VAST	validation, analysis, and storage tool
VOC	volatile organic compound
WET	Wetland Evaluation Technique

Executive Summary

On behalf of Morton International and Velsicol Chemical Corporation, Exponent has prepared a draft ecological risk assessment (ERA) for Operable Unit 1 (OU1) of the Ventron/Velsicol site located in Wood-Ridge and Carlstadt, New Jersey. The remedial investigation (RI) report and the human health risk assessment (HHRA) were submitted under separate cover. The risk assessments are part of the remedial investigation and feasibility study (RI/FS) required by the "Resolution of the Berry's Creek/Wood-Ridge Site Action Committee" (Resolution) with the New Jersey Department of Environmental Protection (NJDEP), executed on August 15, 1996. The Ventron/Velsicol site is designated as a National Priorities List (NPL) site identified by U.S. Environmental Protection Agency (EPA) number NJD980529879, and bearing Comprehensive Environmental Response, Compensation and Liability Information System (CERCLIS) ID number 02C7.

This ERA evaluates the 26-acre portion of the site designated as OU1. OU1 consists of developed and undeveloped areas, associated groundwater, the onsite basin, and the West Ditch. The ERA follows EPA's *Ecological Risk Assessment Guidance for Superfund*: *Process for Designing and Conducting Ecological Risk Assessments, Interim Final* (U.S. EPA 1997a) (ERAGS). This ERA specifically addresses the first three steps of EPA eight-step process. This ERA is also intended to satisfy NJDEP's requirements for a Baseline Ecological Evaluation (BEE), and additional tasks suggested by NJDEP guidance.

As outlined in this guidance, the site was described with respect to its ecological potential. Due to its location in a developed area and its disturbed habitat, OU1 has marginal habitat for ecological receptors. Media were screened to assess whether there is a complete exposure pathway to ecological receptors. Based on this pathway analysis, chemicals in deep soil samples (greater than 1 ft) and those from the developed areas were eliminated from further screening. The latter are primarily beneath pavement or crushed stone in railroad beds, precluding current exposure to ecological receptors. The maximum concentrations of the chemicals in the remaining media (groundwater, surface soils from the undeveloped area, surface water, and sediments from an onsite basin and West Ditch) were then compared to conservative screening values or benchmarks recommended by NJDEP. In accordance with screening protocols, chemicals were retained as contaminants of potential concern (CoPCs) if the maximum concentration exceeded the benchmark, if they were detected and there was no available benchmark, or if the chemical was not detected but the detection limit exceeded a screening benchmark. A number of chemicals in all media were still retained as CoPCs after the initial screening, which corresponds to Step 2 of the eight-step process.

In Step 3, the remaining CoPCs were re-evaluated in a number of ways to refine the estimate of risk. Chemical concentrations were compared to background concentrations or alternate screening benchmarks, and screened under less conservative exposure scenarios. While many of the CoPCs could be dismissed under less conservative scenarios, several could not. Notably, mercury is retained as a CoPC in all media, several metals are still CoPCs in soils, and metals, polycyclic aromatic hydrocarbons (PAHs), and polychlorinated biphenyls (PCBs) are still CoPCs in sediments.

Although contaminants in surficial soils of the developed area pose no current exposure risk to ecological receptors, these data were screened for ecological risk in Step 3. This analysis was conducted to evaluate potential risks in a hypothetical future risk scenario in which the site is allowed to naturalize and the asphalt, buildings, and stone are removed. Most metals would be retained as CoPCs for this future risk scenario with more conservative screening methods. Under less conservative scenarios, these future risks from contaminants in developed area soils were nominal except for mercury. The risks from mercury were due largely to several samples with mercury concentrations in excess of 1,000 mg/kg.

Assessment endpoints were proposed based on a refined conceptual site model and the mechanism of toxicity of remaining CoPCs. Proposed assessment endpoints include aquatic benthos, fish, consumers of both fish and aquatic benthos, consumers of soil invertebrates, and top predators. Potential risk to these assessment endpoints was then considered with food chain models. Under most conservative exposure scenarios (e.g., 100-percent residence, exposure based on the maximum concentration), screening quotients (SQs) were greater than 1.0 for almost all of the assessment endpoints. Risks were primarily due to mercury, inorganic mercury in terrestrial food chains, and methylmercury in aquatic food chains. Metals such as lead also had SQs above 1.0 in terrestrial ecosystems. As more realistic exposure assumptions were employed, risks diminished (i.e., SQs declined to 1.0 or less). However, risks to consumers of soil invertebrates, such as woodcocks and shrews, could not be dismissed even if most conservative assumptions were relaxed. These risks were primarily due to mercury and lead. Risks to aquatic benthos from mercury and, to a lesser extent, other metals, also could not be dismissed even when most conservative assumptions were relaxed.

Both the EPA eight-step process and the NJDEP BEE are intended to be iterative processes in which risk assessors and risk managers interact periodically to determine whether information is sufficient to conclude 1) that risk is not likely, 2) that risk is certain, or 3) that more information is necessary. Exponent believes that the potential for ecological risk cannot be dismissed based on the available information. The next step for the risk assessment process is a scientific management decision point (SMDP) meeting to gather input from risk managers (i.e., EPA and NJDEP) concerning the selection of assessment endpoints and need for further analyses.

1 Introduction

On behalf of Morton International, Inc. and Velsicol Chemical Corporation (Velsicol), Exponent has prepared a draft ecological risk assessment (ERA) for Operable Unit 1 (OU1) of the Ventron/Velsicol site located in Wood-Ridge and Carlstadt, New Jersey. A human health risk assessment (HHRA) has also been prepared under separate cover (Exponent 2001). These risk assessments are part of the remedial investigation and feasibility study (RI/FS) required by the "Resolution of the Berry's Creek/Wood-Ridge Site Action Committee" (Resolution) with the New Jersey Department of Environmental Protection (NJDEP), executed on August 15, 1996. The Resolution is an amendment to the October 26, 1984 "Stipulation and Supplementary Order Approving Cooperative Agreement for Remedial Investigation and Feasibility Study and Amending Procedural Order Involving Remedy" (Stipulation). The Stipulation covers the approximately 38-acre Ventron/Velsicol site and the areas of Berry's Creek potentially affected by industrial activity at the site, while the Resolution provides for implementation of a separate RI/FS for the Ventron/Velsicol site. The Ventron/Velsicol site is designated as a National Priorities List (NPL) site identified by U.S. Environmental Protection Agency (EPA) number NJD980529879, and bearing the Comprehensive Environmental Response, Compensation and Liability Information System (CERCLIS) ID number 02C7.

This document presents the results of the ERA for OU1 of the Ventron/Velsicol site as defined in Section 2.1.1 according to NJDEP (Zervas 1999a). The ERA comprises Section 7 of the RI report for OU1 that was submitted in draft form to NJDEP in September 2000 (Exponent 2000). This ERA was conducted in accordance with the requirements of the approved 1996 work plan (CRA 1996) as clarified and modified in recent communications with the agency. The ERA follows EPA's national guidance entitled *Ecological Risk Assessment Guidance for Superfund: Process for Designing and Conducting Ecological Risk Assessments, Interim Final (U.S. EPA 1997a*). The national guidance has been referred to as "ERAGS" or the "EcoRisk Process Document."

The ERA process described in ERAGS proceeds according to the following eight steps (U.S. EPA 1997a):

1. **Screening-level problem formulation and ecological effects evaluation:** During this step, a preliminary conceptual model is developed for the site that includes the environmental setting and contaminants known or expected to be found at the site, contaminant transport and fate mechanisms, mechanisms of ecotoxicity associated with contaminants and potential receptors, complete exposure pathways, and selection of endpoints to screen for ecological risk (chronic no-observed-adverse-effect levels [NOAELs] based on conservative assumptions).

2. **Screening-level exposure estimate and risk calculation:** The second step of the ecological risk screening includes the exposure estimate and risk calculation. Risk is estimated based on maximum exposure concentrations compared to ecotoxicity screening values from Step 1, and screening

quotients of contaminants of potential concern [CoPCs] are presented). A screening quotient less than 1 indicates the contaminant alone is unlikely to cause adverse ecological effects.

3. **Baseline ecological risk assessment problem formulation:** The results of the screening assessment, in coordination with site-specific data, are used to assess the scope and goals of the ERA. The following should be completed at the end of this step: refine preliminary CoPCs; further characterize ecological effects; review and refine information on contaminant transport and fate, exposure pathways, and ecosystems potentially at risk; select assessment endpoints; develop conceptual model with testable hypotheses; and analyze uncertainties associated with the risk assessment.

4. **Study design and data quality objective process:** The conceptual model is completed during this step of the ERA, and measurement endpoints are developed based on the model. The conceptual model is used to determine the study design and the data quality objectives (DQOs). The products of this step include a work plan and sampling and analysis plan (SAP), detailing the data analysis methods, exposure parameters, data reduction and interpretation methods, and statistical analyses.

5. **Field verification of sampling design:** The sampling design, testable hypotheses, exposure pathway models, and measurement endpoints are examined to ensure they are appropriate and that they can be implemented.

6. **Site investigation and analysis phase:** This step includes all of the field sampling and surveys that are part of the ERA. The data collected during this phase are evaluated on existing and potential exposure and ecological effects outlined in Steps 1–5.

7. **Risk characterization:** This step consists of risk estimation and risk description. Data on exposure and effects are used to characterize risk based on assessment endpoints. The product of this step is the identification of a threshold for effects on the assessment endpoint(s) as concentrations ranging from levels found to pose no ecological risk to levels likely to produce adverse ecological effects.

8. **Risk management:** This phase involves balancing risk reductions associated with remediation of the site with the potential effects of the remediation itself.

This eight-step process is designed to be an interactive process among all involved parties. To promote communication among the risk assessors, risk managers, and associated stakeholders during the process, scientific/management decision points (SMDPs) follow Steps 2, 3, 4, 5, 6, and 8. These SMDPs allow input by stakeholders on the course of action in subsequent steps. This document covers Steps 1 through 3 of the ERAGS process. Upon review of this document, an SMDP is required before proceeding with the ERA.

The ERA also follows the baseline ecological evaluation (BEE) guidelines developed by the New Jersey Site Remediation Program (SRP). The BEE is Tier I of the process that the SRP uses to conduct ecological evaluations and risk assessments (Hamill and Demarest 1997). The objective of the BEE is to assess the site for: (1) contaminants of potential concern, referred to as CoPCs in this report, (2) environmentally sensitive areas, (3) potential contaminant migration pathways to environmentally sensitive areas, and (4) need for further investigation. These components are covered in Sections 2.2, 2.1.2.1, and 3.10 of the ERA.

Environmentally sensitive areas were identified, described, and mapped according to the N.J.A.C. 7:1E-4.10 guidance. This guidance requires identification of environmentally sensitive areas that exist on or adjacent to the site. For environmentally sensitive areas on the site, adjacent to the site, or under the influence of the site, the risk assessment includes a qualitative description of land use and major ecological habitats (Hamill and Demarest 1997). The site and adjacent area were evaluated to determine whether any of the following environmentally sensitive areas were present: wetlands and wetland transition areas; bay islands and barrier island corridors; dunes; areas designated as wild, scenic, recreational or developed recreational rivers; surface waters; water supply; beaches; and breeding areas and migratory stopover areas (N.J.A.C, 7:1E-4.10). A discussion of environmentally sensitive areas that were found on or adjacent to OU1, as specified by the BEE guidance, has been included in Section 2.1.2.8.

2 Preliminary ERA—Steps 1 and 2 of the ERAGS Process

The following represents the analyses that comprise Steps 1 and 2 of the ERAGS process.

2.1 Preliminary Problem Formulation

The preliminary problem formulation describes the general site characteristics and ecological receptors that could be exposed to site chemicals. A preliminary conceptual site model that considers complete exposure pathways is also described.

2.1.1 Site Location and Description

The Ventron/Velsicol site is located in Bergen County, New Jersey, within the boroughs of Wood-Ridge and Carlstadt (Figure 1). In accordance with instructions in an April 1, 1999 letter from NJDEP (Zervas 1999a, pers. comm.), the site has been divided into two operable units: OU1 and OU2 (Figure 2), of which only OU1 is addressed here. The two operable units together comprise an irregularly shaped, approximately 38-acre area within an industrialized area of northeastern New Jersey. Approximately 15.7 of the 38 acres are within the Borough of Wood-Ridge, and the remaining 22.6 acres are within the Borough of Carlstadt. The entire site is generally within the Hackensack Meadowlands area, and the portion in Carlstadt is within the jurisdiction of the Hackensack Meadowlands Development Commission (HMDC). The site is bordered to the east by Berry's Creek; to the west by the West Ditch, the Diamond Shamrock/Henkel and Randolph Products properties, and Park Place East; to the south by Diamond Shamrock/Henkel Ditch (south) and Nevertouch Creek, and to the north by Ethel Boulevard and a railroad track (Figure 2). Two active commercial/industrial facilities and an empty lot, on which a publicly owned treatment works (POTW) was formerly located, lie immediately north of Ethel Boulevard and the railroad track. The railroad crosses Berry's Creek at the northeast corner of the site and continues south along the eastern side of Berry's Creek.

Land use in the immediate vicinity of the site is primarily commercial/industrial. Teterboro Airport is approximately 0.6 miles to the north, State Highway 17 is approximately 500 ft to the west, and the Meadowlands Sports complex is approximately 1 mile to the south. The immediately adjacent Diamond Shamrock/Henkel property is undergoing an active remediation program under the NJDEP Environmental Cleanup Responsibility Act. The closest residential area is approximately 750 ft to the north. Additional information on topography and surface features, climate and meteorology, geologic setting, soils, hydrology, hydrogeology, ecology, demography, and land use is available in the RI report (Exponent 2000).

As indicated above, the site is divided into two operable units, OU1 and OU2, and only OU1 is evaluated here. OU1 includes two areas: a developed and an undeveloped area (Figure 2). The developed portion of OU1 covers approximately 7 acres and includes two active warehouses, the Wolf and U.S. Life Warehouses (Figure 2). The former mercury processing facility was

located on the portion of OU1 that is now occupied by these warehouses. The remainder of the developed area of OU1 is covered with asphalt pavement or gravel, which forms the bed for railroad tracks located immediately behind the warehouses. Drainage from the developed area is generally directed between the two warehouses and the Randolph Products property and it flows in the West Ditch (Figure 2) along the western property boundary.

The undeveloped area of OU1 lies generally south of the developed area and includes approximately 19 acres of land that were filled but not developed. This portion of OU1 is bordered to the north by the railroad track, to the south by the Diamond Shamrock/Henkel Ditch (north), to the west by the West Ditch, and to the east by Berry's Creek (Figure 2). The undeveloped filled area of OU1 is characterized by mixed vegetation and a variety of surficial debris. Much of this area is relatively flat, but the northeast portion of this area has uneven terrain. Two surface features are a small pit that may include remnants of an access structure for the drainage system from the Plant area that extended to Berry's Creek, and, in the undeveloped filled area, a small basin, hereafter discussed as the onsite basin. The onsite basin may have been or may be a remnant of a settling basin for discharges from the plant area or the Randolph Products property (Figure 2). The east and south perimeters of this area are steep stream banks adjacent to Berry's Creek and the Diamond Shamrock/Henkel Ditch (north), respectively. The north and west perimeters of the area are fenced; additional fencing to the east prevents site access via the tide gate.

The remaining 12 acres of the site are within OU2, south of the undeveloped filled area (Figure 2) and are not considered further here. A detailed history, including site background, operating history, and site characterization, is discussed in Section 1.3 of the RI report (Exponent 2000).

2.1.2 Ecological Setting

Section 3.7 of the RI report (*Ecological Description*) contains detailed discussions of the site ecology and habitat characterization (Exponent 2000). The ecological isolation and disturbed nature of the site affect its wildlife resources. Primary local land uses are industrial, and a substantial transportation infrastructure is present (a railroad bed adjoins the site, and municipal roadways and a state highway are present within a few hundred yards). The site was significantly disturbed by historic filling, regrading, vehicular traffic, ditch construction, and material disposal from 1940 through 1974.

2.1.2.1 Terrestrial Habitat

The developed area is almost entirely covered by pavement and two large warehouses; consequently, there is no ecological habitat in this area. The undeveloped filled area is dominated by upland vegetation with plant species consisting primarily of non-native opportunistic trees and shrubs and herbaceous plants characteristic of urban regions. These species are typical of a disturbed area in an industrial location. There are two distinct types of vegetation in the undeveloped filled area: a canopy of relatively small tree-of-heaven with a

weedy herbaceous layer; and an area without a canopy dominated by dense early-season annuals, including common reed.

2.1.2.2 Wetlands

Wetlands were delineated in 1997, as discussed in Sections 2.6 and 3.7.4.1 of the RI. The wetland delineation report was prepared and submitted as a separate document (Shisler 1997). According to the map produced as part of the wetland delineation, there are approximately 0.767 acres of wetlands in OU1 (Shisler 1997). In 1986–87, EPA, HMDC, the U.S. Army Corps of Engineers (ACOE), and NJDEP conducted a functional assessment of the Hackensack Meadowlands District (HMD), which includes the onsite wetlands, using a modified Wetland Evaluation Technique (WET) called the Indicator Value Assessment (IVA) method (U.S. EPA and ACOE 1995). According to this assessment, the HMD wetlands have the following values for the following wetland characteristics:

- Aquatic diversity and abundance—low value
- General wildlife habitat effectiveness—low value
- General fish habitat effectiveness—moderate value
- General waterfowl habitat effectiveness—high value.

For additional information, see Section 3.7.4.2 of the RI. Based on best professional judgment, Shisler (1997) drew a similar conclusion regarding wildlife habitat for the wetlands on OU1. Shisler gave an overall rating of 2.2 (out of 10) in his assessment of the overall wildlife attributes of OU1's wetlands.

The wetlands on the site have been classified as both Section 10 and Section 404 wetlands. The West Ditch is the only area of tidal wetlands in OU1 (Shisler 1997). This area is dominated by a dense monoculture of common reed (*Phragmites australis*). There are two small sections of nontidal wetlands on the site (upstream of the tide gate and the onsite basin). According to Shisler (1997), the nontidal wetlands are dominated by arrow arum, pickerel weed, and jewelweed.

2.1.2.3 Open Water Habitat

Open water habitat at OU1 is limited to the onsite basin and the West Ditch, which restricts the presence of aquatic species. Based on site reconnaissance, killifish (*Fundulus spp.*) are the only species of fish found in abundance in the West Ditch; however, other species of fish associated with estuarine creeks in New Jersey may periodically be present. Regionally common species include herrings (*Clupeids*), catfish (*Ameiurus sp.* and *Ictalurus punctatus*), silversides (*Menidia sp.*), eels (*Anguilla rostrata*), temperate basses (*Morone sp.*), sunfish (*Centrarchidae*), minnows (*Cyprinidae*), bluefish (*Pomatomus saltatrix*), and weakfish (*Cynoscion regalis*). While these other species of fish may occasionally migrate into the West Ditch of OU1, their occurrence is expected to be limited. Fish were not observed in the onsite basin; given its small size and isolated nature, it was assumed that fish were absent from the basin. The assemblage of aquatic

macroinvertebrates present in OU1 is likely to include such taxonomic groups as *Gastropoda* (snails), *Bivalvia* (clams and mussels), *Oligochaeta* (worms and leeches), *Polychaeta* (bristle worms), *Crustacea* (crabs and shrimp), and *Insecta* (insects).

2.1.2.4 Mammals

Common urban mammals have been observed at the site (e.g., woodchuck [*Marmota monax*], Norway rat [*Rattus norvegicus*], opossum [*Didelphis virginiana*], cottontail rabbit [*Sylvilagus floridanus*], and muskrat [*Ondatra zibethicus*]). These are listed in Table 3-4 of the RI report (Exponent 2000).

2.1.2.5 Birds

Birds that are characteristic of disturbed landscapes are commonly observed at the site. Species observed included redwing blackbird (*Agelaius phoeniceus*), robin (*Turdus migratorious*), common grackle (*Quiscalus quiscula*), starling (*Sturnus vulgaris*), English sparrow (*Passer domesticus*), mourning dove (*Zenaidura macroura*), mockingbird (*Mimus polyglottos*), catbird (*Dumetella carolinensis*), blue jay (*Cyanocitta cristata*), and others listed in Table 3-4 of the RI. Migratory species, including a number of warblers (*Parulidae*) and flycatchers (*Tyrannidae*), were observed on the site in the spring of 1997. Under baseline conditions, individual migrants likely are present for a few days to weeks in the spring and autumn. Other birds that are characteristic of the Hackensack Meadowlands as a whole may be present as transients. Herons, egrets, hawks, sandpipers, and plovers may be expected to forage in the site vicinity, although nearby human activity and lack of onsite habitat for these species probably restricts foraging.

2.1.2.6 Rare, Threatened, or Endangered Species

No rare, threatened, or endangered plant species have been identified or are expected to occur at the site (Shisler 1997).[1] Lists of endangered, threatened, rare, or uncommon species for the site vicinity, from the Natural Heritage Program (NHP) database for Bergen County, have been reviewed. No wildlife management areas have been identified in the immediate vicinity of the site.

2.1.2.7 Habitat Resource Value

As discussed in Section 3.7.3 of the RI report, the quality and resource value of both terrestrial and aquatic habitats has been compromised by several physical factors. Ecological isolation of the site (due to surrounding industrial and commercial land use) limits recruitment for many species and impairs diversity. Furthermore, the long history of physical disturbances has created conditions favorable for opportunistic vegetation that is characteristic of waste areas (e.g., tree-of-heaven, knotweed, and common reed). These physical factors have resulted in

[1] In 1997, a letter was sent to the NJDEP Natural Heritage Program (NHP) requesting a data search of the Natural Heritage Database on rare species at the site. On December 18, 2000, a followup letter was submitted to the NHP requesting an additional database search. In both of these letters, the NHP stated the "Database does not have any records for rare plants, animals, or natural communities on the Site" (NJDEP 1997b; NJDEP 2001).

fragmented or impaired conditions that reduce habitat quality. The limited habitat quality will, in turn, discourage wildlife from establishing territories for nesting and foraging.

2.1.2.8 Onsite or Adjacent Environmentally Sensitive Areas

N.J.A.C. 7:26E-3.11(a) 2 requires the identification of environmentally sensitive areas (ESAs), as defined in section 1E-1.8(a). Based on site reconnaissance, the wetlands delineation and two habitat characterizations, the following environmentally sensitive areas were identified within OU1 or adjacent to OU1:

Wetlands and wetland transition areas—The wetlands, the West Ditch, and the onsite basin have been classified as tidal wetlands and nontidal, open-water/emergent wetlands (Section 3.7.4.1 of the RI report). As noted above in Section 2.1.2.2, the onsite wetlands have limited habitat potential. The Berrys Creek wetlands are adjacent to OU1.

Breeding areas for forest-area nesting species, colonial water birds, or aquatic furbearers—The undeveloped area is thinly wooded and will provide some habitat for forest-nesting birds. The limited onsite wetland areas may provide breeding areas for furbearing mammals, as will the adjacent Berrys Creek wetlands.

Migratory stopover areas for migrant shorebirds, raptors, or passerines—The onsite upland and wetland areas may provide limited habitat to migrant birds. The adjacent Berrys Creek wetlands will provide more extensive habitat to migratory aquatic birds.

Forest areas, including prime forestland and unique forestland—The area is neither prime nor unique forestland. The undeveloped filled area is partially forested and provides habitat, breeding, and foraging opportunities to species living on the site. The adjacent areas are not forested.

2.1.3 Preliminary Conceptual Site Model and Assessment of Exposure Pathways

Chemicals associated with the site were found in surficial and deep soils, groundwater, surface water, and sediments at OU1. Chemicals in most of these media pose potential exposure risks to ecological receptors. For example, although chemicals in groundwater pose no current exposure to ecological receptors, there is a potential exposure pathway to ecological receptors in downgradient surface waters (e.g., the Diamond Shamrock/Henkel Ditch [north] and Berrys Creek). Chemicals in sediments may pose direct toxicity to aquatic benthos, or indirect toxicity via the food chain to predators of aquatic benthos. Chemicals in surface water have similar potential for direct toxicity to water column species and indirect toxicity to predators of those species. Lastly, chemicals in soils can pose direct toxicity to plants and soil organisms as well as indirect food chain exposure to herbivores and predators of soil organisms.

However, chemicals in two classes of media—surficial soils from the developed area of OU1, and deep soils—do not pose significant current exposure to ecological receptors. The surficial soil samples from OU1 were taken from below pavement or from the railroad bed that is

covered with crushed stone. These surfaces will prevent a complete exposure pathway. In addition, chemicals in deep soils (greater than 1 ft deep) are too deep to pose significant exposure to ecological receptors.

Based on the preliminary consideration of exposure pathways, chemicals in deeper soils (greater than 1 ft.) and surficial soils from the developed area will not be screened in Step 2. The exceptions to this are the data from the NJDEP borehole samples, which were taken from 0 to 2 ft. Chemicals in the other media will be screened. After this screening, the conceptual site model and assessment of exposure pathways will be revisited and refined in Step 3.

2.2 Ecological Screening Process

This section sets the foundation for screening CoPCs, in accordance with ERAGS (U.S. EPA 1997a) and the SRP BEE guidance. The preliminary screening of CoPCs is a component of Steps 1 and 2 in the ERAGS process. Data collected from surficial soils from the undeveloped area, sediment, groundwater, and surface water during the Phase I and Phase IA investigations were compiled as part of the RI and entered into a database created by Exponent. These data were collected by Exponent from 1997–2000 (Exponent 2000) and NJDEP in 1990-91 (NJDEP 1993). The maximum and mean values of each detected constituent in these media were then compared to ecological screening values (described in Sections 2.2.1.1, 2.2.1.2, and 2.2.1.3). Detected constituents that exceeded the screening values, undetected constituents with detection limits that exceeded the screening values, or constituents with no corresponding ecological screening values were tabulated and considered preliminary CoPCs.

2.2.1 Steps 1 and 2—Methods of Screening of Primary Media

Data from primary media (surficial soils, sediment, groundwater, and surface water) were compared to screening values for use at hazardous waste sites. Media-specific screening values were selected from documents recommended and prioritized by NJDEP (Demarest 2000, pers. comm.). The documents reviewed included those for media-specific screening values in the BEE guidelines (Hamill and Demarest 1997). The list of guidance documents was submitted to NJDEP in January 2001 (Hock 2001, pers. comm.)

Screening values are based on constituent levels associated with very low probability of unacceptable risk to ecological receptors. Specifically, the criteria are based on sensitive endpoints, sensitive species, and a conservative use of other ecological effects data (e.g., ingestion rates). Screening values are designed for use as a preliminary screening tool to determine whether there is potential ecological risk at a site and to assess the need to conduct further investigations (U.S. EPA 1999b).

Discussions were conducted between Exponent and NJDEP regarding the use of freshwater vs. marine/estuarine screening values. In a letter from NJDEP to Exponent dated June 25, 1999, the agency agreed that water data should be screened against freshwater screening values (Zervas 1999b, pers. comm.). This decision was based on the salinity at the site (average of about

2.5 ppt), which was not high enough to warrant the use of marine/estuarine screening values (Langseth 1999, pers. comm.).

The data collected from 1997–2000 are maintained in Exponent's Validation, Analysis, and Storage Tool (VAST) database. In the screening, results of duplicate analyses were averaged and compared to the screening values. The means in the screening tables (Tables A-2 through A-6) were calculated based on detected values and one-half the detection limit for undetected values. Screening quotients (SQ) were calculated according to the following equation:

SQ = maximum concentration/screening value.

2.2.1.1 Soil Screening

Ecological screening benchmarks for surface soil were selected from the following documents in the following order of preference:

- *Preliminary Remediation Goals for Ecological Endpoints* (Efroymson et al. 1997a)
- *Toxicological Benchmarks for Contaminants of Potential Concern for Effects on Soil and Litter Invertebrates and Heterotrophic Process: 1997 Revision* (Efroymson et al. 1997b)
- *Toxicological Benchmarks for Screening Contaminants of Potential Concern for Effects on Terrestrial Plants* (Efroymson et al. 1997c)
- *Toxicological Benchmarks for Wildlife: 1996 Revision* (Sample et al. 1996)
- *Contaminant Hazard Reviews* (Eisler 1987a,b)
- *Evaluating Soil Contamination* (Beyer 1990)
- *Cleanup Standards for Contaminated Sites, N.J.A.C. 7:26*
- *Ecological Soil Screening Level Guidance Draft* (U.S. EPA 2000).

2.2.1.2 Sediment Screening

Screening benchmarks for sediments criteria were selected from the following documents in the following order of preference:

- *Guidance for Sediment Quality Evaluations* (NJDEP 1998)
- *Guidelines for the Protection and Management of Aquatic Sediment Quality in Ontario* (Ontario Ministry of the Environment 1993)
- *Screening Quick Reference Tables* (NOAA 1999)

- *The Development of Canadian Marine Environmental Quality Guidelines* (MacDonald et al. 1992).

2.2.1.3 Water Screening (Surface Water and Groundwater)

Ecological benchmarks for surface water and groundwater were selected from the following documents in the following order of preference. The benchmarks selected from these documents were based on chronic freshwater exposure:

- *Surface Water Quality Criteria Applicable to New Jersey* (NJDEP 1997)
- *National Recommended Water Quality Criteria—Correction* (U.S. EPA 1999a)
- *Toxicological Benchmarks for Screening Potential Contaminants of Concern for Effects on Aquatic Biota: 1996 Revision* (Suter and Tsao 1996)
- *Preliminary Remediation Goals for Ecological Endpoints* (Efroymson et al. 1997a)
- *Screening Quick Reference Tables* (NOAA 1999).

The chronic freshwater screening criterion for mercury has recently changed from 0.012 ppb to 0.77 ppb (U.S. EPA 1999a). In the NJDEP (1997) document, the criterion is listed as 0.012 ppb, which is based on the former EPA chronic value. However, 0.012 ppb was actually a human health criterion, originally promulgated to prevent bioaccumulation of mercury in fish to levels that are hazardous to human health. EPA has since promulgated 0.77 μg/L (ppb) as a chronic value that is protective of aquatic life (U.S. EPA 1999a). Therefore, 0.77 ppb was used in the ERA to assess potential effects on aquatic life. Dr. Edward Demarest at NJDEP was notified about this issue on November 30, 2000 (Pearlman 2000, pers. comm.)

2.2.1.4 Historical Data

Historical groundwater and surficial soil data were collected in 1990–1991 by NJDEP (NJDEP 1993). As part of the NJDEP (1993) investigation, 12 wells were installed; groundwater samples were collected from the monitoring wells and soil samples were collected from the boreholes. This dataset is incomplete, as only detected values were reported. The NJDEP surficial soil data are also problematic because they come from the top 2 ft. Generally, ecological risks are minimal for chemicals located more than 1 ft below ground surface. As a result, these data were not incorporated into the dataset collected for the RI report. These data were, however, screened against soil and freshwater surface water screening values (Tables A-7 and A-8).

Aside from the NJDEP data described above, data from previous investigations at the site were not incorporated into the historical dataset nor were they screened. These data are of unknown quality. Exponent was unable to find information supporting the assessment of the data quality from these prior investigations. Therefore, Exponent has not attempted to validate or assess the

quality of this earlier data, some of which may not be of suitable quality to combine directly with data collected for the RI.

2.2.2 Results of Screening (Step 2 of ERAGS Process)

The comparison of site concentrations against NJDEP-accepted screening values represents the preliminary screening-level problem formulation of the ERA. The results are presented for evaluation in the SMDP following completion of Step 2 of the ERAGS process. Tables A-2, A-3, A-4, A-5, and A-6 show the comparison of site data to the screening values. Tables A-1, A-9, and A-10 provide summary results of the screening and a list of the preliminary CoPCs. The preliminary list of CoPCs includes those constituents whose maximum concentrations exceeded the screening values in one or more media (Table A-1), those constituents that were detected, but for which no screening value was available (Table A-9), and those constituents for which the screening value was less than the detection limit (Table A-10).

2.2.2.1 Onsite Surface Soils

This section discusses concentrations of CoPCs in surface soils from the following sources: 1) 200-ft intervals along a control grid established on the site; 2) surface soil intervals for the boreholes in which the three Phase IA monitoring wells (MW-13, MW-14, and MW-15) were installed. Seventeen metals (aluminum, antimony, arsenic, barium, cadmium, chromium, copper, iron, lead, manganese, mercury, nickel, selenium, silver, thallium, vanadium, and zinc), methylmercury, and one semivolatile organic compound (SVOC) (bis[2-ethylhexyl]phthalate [DEHP]) were identified as preliminary CoPCs based on screening against NJDEP-recommended ecological screening values. Table A-2 summarizes the screening results for onsite surface soil CoPCs. A complete listing of all these data can be found in Appendix B of the RI report. The surface soil borehole data were also screened against the ecological screening benchmarks. Only data from the 0-2 ft depth interval were screened because the subsurface pathway is not applicable to the ERA. Fourteen metals (arsenic, barium, cadmium, chromium, copper, iron, lead, manganese, mercury (total), nickel, selenium, silver, thallium, and zinc) exceeded the screening values. All of these were also selected as CoPCs from the surficial soil samples. Table A-3 summarizes the CoPCs that exceed screening values in surface borehole soils from OU1. A complete listing of all these data can be found in Appendix B of the RI report.

2.2.2.2 Sediment

In an initial round of sampling, the top 15 cm of sediment was sampled at each of the water sampling locations. A second grab sample of sediments was collected from the upper 0 to 2 cm at each Phase I sample location and analyzed only for mercury. Nine metals (arsenic, cadmium, chromium, copper, lead, mercury, nickel, silver, and zinc), fourteen SVOCs (acenaphthene, acenaphthylene, anthracene, benz[a]anthracene, benzo[a]pyrene, benzo[g,h,i]perylene, benzo[k]fluoranthene, chrysene, dibenz[a,h]anthracene, di-*n*-butyl phthalate, fluoranthene, indeno[1,2,3-cd]pyrene, phenanthrene, and pyrene), and two polychlorinated biphenyl (PCB) Aroclor® mixtures (1248 and 1260) exceeded their respective screening values and are

considered preliminary CoPCs. With the exception of di-*n*-butyl phthalate, all of the SVOCs are polycyclic aromatic hydrocarbons (PAHs). Table A-4 summarizes the exceedances of sediment screening values. A complete listing of all these data can be found in Appendix B of the RI report.

2.2.2.3 Groundwater

Groundwater samples were collected from 15 monitoring wells located on the site. The data from these samples were screened against surface water screening values based on the highly conservative assumption that these concentrations would prevail, without dilution or fate processes, at their point of discharge into surface water. Twelve metals (barium, cadmium, cobalt, copper, iron, lead, manganese, mercury, selenium, thallium, vanadium, and zinc), methylmercury, three volatile organic compounds (VOCs) (benzene, carbon disulfide, and toluene), and one SVOC (naphthalene) exceeded the screening values and were identified as CoPCs. Table A-5 provides a summary of chemical exceedances in groundwater. A complete listing of all these data can be found in Appendix B of the RI report.

2.2.2.4 Surface Water

Surface water samples were collected from two locations in the onsite basin during Phase I and three locations in the West Ditch during Phase IA. Samples designated for dissolved metals analyses were filtered in the laboratory. In Phase IA, only unfiltered whole water samples were analyzed.

Hardness was measured in the onsite basin twice. Both samples yielded hardness concentrations of 440 mg/L. Hardness was not measured in the West Ditch, but was measured in the samples immediately downstream (SW-05, SW-06, SW-07) in the Diamond Shamrock/Henkel Ditch (north). Hardness values at these three sampling sites were 450 mg/L, 920 mg/L, and 890 mg/L. A hardness value of 450 mg/L $CaCO_3$ was used to calculate hardness-dependent water quality criteria (cadmium, chromium, copper, lead, nickel, zinc) for use as screening values.

Five metals (barium, iron, manganese, mercury, and zinc) exceeded the screening benchmarks and were identified as CoPCs. Table A-6 is a summary of chemical exceedances in surface water. A complete listing of all these data can be found in Appendix B of the RI report.

2.2.3 Summary of Exceedances

The following describes the exceedances of the conservative screening benchmarks by each chemical.

Aluminum—There were no exceedances in surface borehole, sediment, groundwater, or surface water samples. However, aluminum exceeded the screening value in all of the soil samples.

Antimony—There were no exceedances in surface borehole, sediment, groundwater or surface water samples. Antimony exceeded the screening value in 5 out of 24 samples in soil with a maximum SQ of 10.7 and a mean SQ of 1.1.

Arsenic—There were no exceedances in groundwater or surface water. There were five exceedances out of 24 samples in soil with a maximum SQ of 1.4, and a mean SQ of 0.7. There was one additional exceedance in surface borehole soil (maximum SQ of 2.7, mean SQ of 1.8). In sediment, arsenic exceeded the screening value once out of five samples (maximum SQ of 1.5, mean SQ of 0.8).

Barium—There were no exceedances in sediment. Exceedances occurred in soil, surface borehole soil, groundwater, and surface water samples. In soil, barium exceeded the screening value in 13 of the 24 samples (maximum SQ of 2.1 and mean SQ of 1.0.) and in one of two samples in surface borehole soil (maximum SQ of 1.1 and mean SQ of 1). All of the groundwater and surface water samples exceeded the freshwater screening value.

Cadmium—Cadmium exceeded the screening value in all the media analyzed except for surface water. In soil, cadmium exceeded the screening value in 6 of 24 samples (maximum SQ of 5.3, mean SQ of 0.9) and in one of two samples in surface borehole soil (maximum SQ of 1.2 and a mean SQ of 1). Cadmium concentrations in sediment exceeded the screening value in all five samples (maximum SQ of 15.2, mean SQ of 6.2). Groundwater concentrations of cadmium exceeded the screening value 11 of 27 samples (maximum SQ of 5.7, mean SQ of 1).

Chromium—There were no exceedances for chromium in groundwater or surface water. However all of the samples in soil, surface borehole soil, and sediment exceeded the respective screening values.

Cobalt—In all four media, there was only one exceedance of 12 samples for cobalt in groundwater, with a maximum SQ of 1.3 and mean SQ of 0.5.

Copper—There were copper exceedances in all media except surface water. In soil, 21 of the 24 samples exceeded the screening values (maximum SQ of 16.8, mean SQ of 4.0). All of the surface borehole data (maximum SQ of 123.7 and mean SQ of 63.7) and sediment samples collected exceeded the criteria (maximum SQ of 12.1, mean SQ of 8.9). Three of the 27 samples in groundwater exceeded the screening values (maximum SQ of 32.4, mean SQ of 1.6).

Iron—Iron exceeded the screening values in soil, surface borehole soil, groundwater, and surface water. All of the soil and surface borehole soil samples exceeded the screening values (maximum SQ of 610, mean SQ of 135 and maximum SQ of 178 and a mean SQ of 149, respectively). Twenty-two of 27 samples in groundwater exceeded the screening values (maximum SQ of 37.5, mean SQ of 12). In surface water, three of the five samples exceeded the screening values (maximum SQ of 2.6, mean SQ of 1.7).

Lead—Lead exceeded the screening values in four of the five media. In soil, 21 of 24 samples exceeded the screening values (maximum SQ of 106.7, mean SQ of 19.8) and two of two samples in surface borehole data exceeded the screening values (maximum SQ of 9.7 and a mean SQ of 8.6). All five of the sediment samples exceeded the screening values (maximum

SQ of 15.1, mean SQ of 9). Six of 27 samples of lead in groundwater exceeded the screening values (maximum SQ of 5.6, mean SQ of 0.9).

Manganese—Manganese exceeded the screening values in all media except sediment. In soil, 21of the 24 samples exceeded the screening values (maximum SQ of 30.9, mean SQ of 4.5) and in surface borehole data, both samples exceeded the screening values (maximum SQ of 2.6 and a mean SQ of 2.5). Twenty-four of 27 samples of manganese in groundwater exceeded screening values (maximum SQ of 82.3, mean SQ of 16.9). There were exceedances of both filtered and unfiltered manganese concentrations in surface water. Filtered manganese exceeded the screening value in two of two samples (maximum SQ of 4.7, mean SQ of 4.5) and unfiltered manganese exceeded the screening value in five of five samples (maximum SQ of 5.2, mean SQ of 3.6).

Mercury—There were exceedances of mercury in all of the media analyzed. In surface soil, mercury was detected in all 24 of the samples and was in exceedance of the screening values (maximum SQ of 1,074,510, mean SQ of 235,294). Both samples in surface borehole soil exceeded the screening values (maximum SQ of 4,400,000 and mean SQ of 2,200,000). In sediment, all samples exceeded the mercury screening level (maximum SQ of 6,450 and mean SQ of 3,450). Mercury was analyzed as dissolved and unfiltered in groundwater. All three of the dissolved (filtered) samples exceeded the screening values (maximum SQ of 11, mean SQ of 5.2). Ten of 30 unfiltered samples exceeded the screening values (maximum SQ of 70, mean SQ of 6.5). In surface water, mercury was also analyzed as dissolved and unfiltered; however there were only two exceedances of the five unfiltered samples (maximum SQ of 22.9, mean SQ of 6.5).

Nickel—Nickel exceeded the screening value in soil, surface borehole soil, and sediment. In soil, 15 of 24 samples exceeded the screening values (maximum SQ of 2.7, mean SQ of 1.2) and one of two samples in surface borehole data exceeded the screening values (maximum SQ of 2.9 and a mean SQ of 1.8). In sediment, four of five samples exceeded screening levels (maximum SQ of 1.8, mean SQ of 1.5).

Selenium—There were no exceedances of selenium in sediment or surface water; however, there were exceedances in soil, surface borehole soil, and groundwater. Nine of the 24 samples in soil exceeded the screening values (maximum SQ of 9.5, mean SQ of 3.9) and both of the samples in surface borehole soil exceeded the screening values (maximum SQ of 7.1 and mean SQ of 6.2). In groundwater, four of 27 samples exceeded the screening values (maximum SQ of 2.7, mean SQ of 0.6).

Silver—The only exceedances for silver were in soil and sediment. In soil, 11 of the 24 samples exceeded the screening values (maximum SQ of 46.9, mean SQ of 3.3) and in surface borehole soil, one of two samples exceeded the screening values (maximum SQ of 4.8, mean SQ of 2.9). In sediment, three of five samples exceeded the screening values (maximum SQ of 4.3, mean SQ of 1.6).

Thallium—Thallium exceeded the screening value in soil, borehole surface soil, and groundwater. In soil, 2 of 24 samples exceeded the screening values (maximum SQ of 21.9, mean SQ of 2.2) and the one of two samples in surface borehole soil also exceeded the

screening values (maximum SQ of 5.4, mean SQ of 2.8). One of 27 groundwater samples exceeded the screening values (maximum SQ of 1.5, mean SQ of 0.4).

Vanadium—All 24 of the vanadium samples exceeded the screening value in soil (maximum SQ of 87.5, mean SQ of 25); however, only 1 of 12 samples in groundwater exceeded the screening values (maximum SQ of 2.7, mean SQ of 0.4).

Zinc—Zinc exceeded the screening value in all of the media analyzed. Eighteen of 24 samples had concentrations above the screening level (maximum SQ of 2,988.2, mean SQ of 314) and both the samples in surface borehole soil exceeded the screening values (maximum SQ of 248.2 and a mean SQ of 154.2). All five of the sediment samples exceeded the screening values (maximum SQ of 29.5, mean SQ of 11.9). Six of 27 samples in groundwater exceeded the screening values (maximum SQ of 8.0, mean SQ of 1.3), while one of five samples in surface water exceeded the screening values (maximum SQ of 1.1, mean SQ of 0.4).

Methylmercury—Methylmercury exceeded the screening value in both soil and groundwater. The soil samples had maximum SQ of 2.4 and mean SQ of 0.3. Samples in groundwater were analyzed as unfiltered and 14 of 27 samples exceeded the screening values (maximum SQ of 10.9, mean SQ of 2.2).

Aroclor® 1248 and 1260—Both Aroclor® 1248 and 1260 exceeded the screening value in sediment. Two of two samples exceeded screening values (maximum SQ of 8 [1248] and 98 [1260], mean SQ of 7 [1248] and 74 [1260]).

PAHs—A number of PAHs exceeded the screening values in sediment. Table 3.1 lists the specific compounds and the number of exceedances. With the exception of acenaphthene, where there was only one exceedance, each of the compounds exceeded the screening value at least twice and had maximum and mean SQs greater than 1.0.

Di-*n*-butyl phthalate—The one detection of di-*n*-butyl phthalate in sediment exceeded the screening values (maximum and mean SQ of 1.5).

***Bis*[2-Ethylhexyl]phthalate—**Bis[2-ethylhexyl]phthalate exceeded the screening value in soil only. There was 1 exceedance out of 24 samples (maximum SQ of 1.05, mean SQ of 0.1).

Benzene—There was one exceedance of 27 samples of benzene in groundwater (maximum SQ of 3, mean SQ of 0.2). Because benzene is highly volatile and undergoes rapid microbial degradation, benzene is expected to degrade to concentrations below screening levels.

Carbon disulfide—There was only one detection and one exceedance of carbon disulfide out of 12 groundwater samples (maximum SQ of 17.4, mean SQ of 6.5). Carbon disulfide is very volatile and will most likely degrade to below the screening level.

Naphthalene—Naphthalene exceeded the screening value for groundwater once out of 13 samples (maximum SQ of 4.2, mean SQ of 0.4).

Toluene—Two out of 27 groundwater samples for toluene exceeded the screening values (maximum SQ of 13.1, mean SQ of 0.6). Because toluene is highly volatile and undergoes rapid microbial degradation, it is expected to degrade to concentrations below screening levels.

The following conclusions were reached based on screening. The majority of the CoPCs in surface soils are metals, notably mercury, chromium, copper, lead, silver, thallium, vanadium, and zinc. All of these compounds had maximum SQs greater than 10 and mean SQs greater than 1.0 (Table A-2). Of the CoPCs, mercury had the highest SQ values. Many of the same metals are CoPCs in sediment. Mercury again has the highest SQ values in sediments, with maximum and mean SQs greater than 1,000. Zinc and PCBs also had SQs greater than 25. PAHs also had SQ values greater than 1.0, but SQs for these compounds are more moderate (i.e., less than 10).

Potential ecological risk from CoPCs in surface water was dominated by total mercury and barium, which had maximum SQs of 22, and 49, respectively (Table A-6). Several other metals were nominally over their ecological benchmarks. However, the excessive SQs for these other metals are largely attributable to the conservative screening methods. SQs for metals were generated with total metals concentrations based on sampling at what is essentially a worst-case acute exposure scenario (low tide and minimal dilution). These concentrations were applied to chronic water quality criteria that are more accurately expressed as dissolved metals. This conservative comparison also fails to consider reduced bioavailability due to binding capacity of the dissolved organic carbon (OC) in the water. The perceived risk from barium is due largely to an overly conservative screening value, as will be discussed below in Section 3.2.2. Therefore, the dominant risk in surface water was judged to be largely due to mercury.

Many compounds were also were retained as CoPCs in groundwater (Table A-5). Several compounds had maximum SQ values greater than 10, including barium, manganese, copper, and mercury. However, the conservativeness of the screening methods should be stressed. As with the surface water samples, the groundwater screening applies concentrations from whole water samples to screening values, many of which are more properly based on dissolved concentrations that reflect bioavailable metals. The contribution of particulate and adsorbed compounds is especially problematic when screening groundwater samples. In addition, comparing groundwater concentrations to surface water criteria assumes that there is no dilution as the groundwater travels to the nearest surface water. These assumptions are also very conservative.

2.3 General Uncertainties Associated with Steps 1 and 2 of the ERAGS Process

The following uncertainties are associated with the first two steps of the ERAGS process:

- For constituents with screening values less than the detection limits, the number of exceedances could not be determined, making it difficult to assess the corresponding risk associated with those constituents. To be conservative, these constituents (Table A-10) were carried forward to Step 3

of the ERAGS process. The risk associated with constituents detected in measurable quantities, but for which there was no available screening value, is also uncertain. These constituents have been summarized in Table A-9 and will be carried forward to Step 3 of the ERAGS process.

- The risk assessment did not assess risks from chemicals in soils in the developed area because the area is not habitat and the exposure pathway from soil to ecological receptors is intercepted by asphalt, building, or crushed stone. Exclusion of these data introduces uncertainty because it is unknown how long the area will remain developed and how long the barriers will preclude exposure to soil chemicals into the future.

- Several metals were analyzed as total metals, yet were screened against the most toxic valence or complex state (i.e., total chromium in water was screened against chromium VI). As the actual chromium VI concentration is unknown, this approach was chosen because it is conservative, tending to exaggerate likely risk. Similarly, most of the data for metals in groundwater and surface water were based on analyses of whole water. In contrast, potential toxicity is more accurately based on dissolved metals concentrations. As the total metals concentration is generally greater than the dissolved concentration, this assumption also exaggerates risk.

- There is also considerable uncertainty associated with the choice of screening values. For some of the more common contaminants, benchmarks can be obtained from a wide variety of sources. These benchmarks can vary greatly in the underlying methods of calculation, the amount and type of external review, and the degree of conservatism, which ultimately affects the magnitude of the screening benchmark. For example, the soil benchmarks used in Step 2 (Efroymson et al. 1997a,b,c) are based on very conservative assumptions. Consequently, these benchmarks tend to be extremely conservative and several, notably the mercury value, contain significant errors[2]. Canada recently developed soil-screening benchmarks (CCME 1997) that were based on methods similar to but less conservative than those used by Efroymson et al. In addition, there are different Canadian guidelines for parklands and industrial areas, thereby formally accommodating differences in land use. EPA recently proposed draft soil screening levels (SSLs) using a process with a high degree of conservatism (U.S. EPA 2000). Finally, EPA promulgated de facto safe-soil concentrations for various metals with its

[2] The PRG from Efroymson et al. for mercury in soil contains two significant errors. First, the value is based on prediction of dry weight mercury concentrations in worms but is erroneously applied to a wet weight rate of food consumption. In addition, the soil benchmark is based on a methylmercury benchmark, and mercury in terrestrial food chains is overwhelmingly inorganic. These two errors greatly decrease the mercury levels. Correctly applying the ORNL methods (i.e., applying a wet weight worm concentration to a wet weight diet and using a TRV for inorganic mercury) produces a safe mercury concentration of about 37 mg/kg in soil, which is over 70,000 times higher than the PRG estimated by ORNL. It should be noted that neither method includes risk from inadvertent soil consumption.

sludge regulations (U.S. EPA 1993a). These latter benchmarks underwent extensive external public review during the rulemaking process.

Table 1 shows the variability of the benchmarks for some common metals and demonstrates the inverse relationship between perceived risk and the choice of screening benchmarks. Note that the Efroymson benchmarks, which were used in Step 2 screening analysis, are lower than those from other reliable sources. The Efroymson et al. (1997a) values for aluminum, chromium, mercury, selenium, and zinc appear to be especially unrealistic.

Table 1. Comparison of commonly used soil screening benchmarks

Compound	Efroymson et al. (1997a)	EPA SSLs (U.S. EPA 2000a)	Canadian Soil Guideline (CCME 1997)	Sludge Regulations (U.S. EPA 1993a)	Ratio Max. to Efroymson
Aluminum	50	*			>1,000*
Antimony	5	21			4.2
Arsenic	9.9	37	26	21	3.7
Barium	283		2,000		7.1
Cadmium	4.0	29	27	20	7.3
Chromium	0.4	21	87	1,504	3,760
Copper	60	61	100	752	12.5
Lead	40.5		400	150	9.9
Mercury	0.00051		30	9	58,823
Nickel	30		50	211	7.0
Selenium	0.21		10	50	238.1
Vanadium	2.0		130		65.0
Zinc	8.5	120	380	1,404	165.2

Note: Values in mg/kg

* According to U.S. EPA (2000), aluminum is not considered a potential risk at a pH above 5.5. Furthermore, aluminum has been found at background levels in natural soils at a concentration of about 50,000 mg/kg and at levels as high as 300,000 mg/kg (Shacklette et al. 1971). Based on this information, the Efroymson et al. (1997a) benchmarks could be 1,000 to 60,000 times lower than values EPA now considers non-problematic.

CCME - Canadian Soil Guidelines (Canadian Council of Ministers of the Environment)

EPA - U.S. Environmental Protection Agency

SSL - Soil screening levels

2.4 Conclusions of Preliminary Screening Assessment (Step 2)

The preliminary screening assessment identified CoPCs as chemicals that exceeded conservative ecological benchmarks (Table A-1), chemicals that were detected for which no screening values were obtained (Table A-9), and chemicals whose detection limits exceeded screening levels (Table A-10). According to the eight-step process (U.S. EPA 1997a), an SMDP occurs at the end of Step 2 to assess whether there is sufficient information to conclude that no ecological risks are likely. At this point, given the large number of compounds that are retained as CoPCs, and the level of exceedances of some chemicals compared to their screening benchmarks, it is concluded that potential risk exists. Therefore, the assessment proceeds to Step 3.

3 Baseline Problem Formulation (Step 3 in the ERAGS Process)

The results of the screening assessment, in coordination with site-specific data, are used to assess the scope and goals of the ERA. The following should be completed at the end of this step: refine preliminary CoPCs; further characterize ecological effects; review and refine information on contaminant transport and fate, exposure pathways, and ecosystems potentially at risk; select assessment endpoints; develop conceptual model with testable hypotheses; and analyze uncertainties associated with the risk assessment.

3.1 Methods

U.S. EPA (1997a) describes the Step 3 problem formulation as the "process for generating and evaluating preliminary hypotheses to determine whether ecological effects have occurred, or may occur, from human activities." The problem formulation process provides a "systematic approach for organizing and evaluating available information on stressors and possible effects" (U.S. EPA 1997a). The following components of problem formulation are the planning tools that focus the ERA and provide a basis for defining ecological risk:

- Refinement of CoPCs
- Information on contaminant transport and fate and biota potentially at risk
- Characterization of the modes of ecological effects/toxicity of CoPCs
- Characterization of exposure and refinement of the conceptual site model
- Selection of assessment endpoints and development of hypotheses to be addressed in the ERA
- Refined assessment of risk to assessment endpoints using simple food chain models
- Uncertainty analysis
- Final assessment of risks.

These tasks are described in the sections below. The simple food chain models, suggested in the sixth bullet, will be based on preceding tasks, specifically the refined conceptual site model, proposed assessment endpoints, and a more detailed description of modes of toxicity of CoPCs. Simple food chain models allow consideration of site-specific information and, thus, a better delineation of risk. In contrast, generic ecological screening values are sometimes inapplicable to site conditions and/or the chosen assessment endpoints.

3.2 Refinement of List of Chemicals of Potential Concern (CoPCs)

Step 2 of the ERAGS process (Section 2.2) identified preliminary CoPCs for which maximum concentrations exceeded conservative screening values. These constituents are summarized in Table A-1. Constituents with no established screening value were also carried forward by the screening process (Table A-9). Other constituents were carried forward as a conservative measure because their detection limits were higher than the applicable screening values (Table A-10).

The purpose of this section is to re-evaluate, and potentially refine, the list of CoPCs in order to identify those constituents that require further evaluation in the ERA. In accordance with ERA guidelines (U.S. EPA 1997a), the first part of Step 3 is the refinement of preliminary CoPCs in order to eliminate contaminants and exposure pathways that pose negligible risks. The refinement process ultimately considers factors in addition to the screening values. Specifically, the preliminary CoPCs were reevaluated according to the following attributes:

- Frequency of detection
- Frequency of exceedance of screening value
- Average exceedance of screening criterion
- Background concentrations
- Less conservative, but still defensible benchmarks.

These attributes are described below along with the results of the re-evaluation.

In addition to considering how the screening assessment may have exaggerated risk, it is equally important to consider how Steps 1 and 2 might have underestimated risk. Because current exposure to chemicals in soil from the developed area is negligible, the screening assessment did not screen these chemicals. However, the barriers to exposure may not be present in the future, so a future ecological risk scenario will be presented to refine the potential risks if the site is ever allowed to revert to nature.

3.2.1 Frequency of Detection, Frequency of Exceedance of Screening Value, and Average Exceedance of Screening Criterion

Ecological risk assessment pertains to potential effects on populations of plants and animals. Significant adverse effects on populations of biota will not generally occur for compounds that exceed benchmarks by small margins and/or in very localized areas. Therefore, the frequency of exceedance and the level of average exceedance provide useful information about the potential for ecological risk. Given the conservativeness of the screening benchmarks, incidental or nominal exceedance of a benchmark suggests little potential for risk. CoPCs identified in the conservative Step 2 screening will be reconsidered in terms of their frequency

and degree of exceedance. As a general rule, compounds that were detected less than 5 percent of the time, exceeded their benchmark less than 10 percent of the time, or had average SQ values less than 1.0 were not retained as CoPCs. Results of this rescreening for CoPCs in soil, sediment, groundwater, and surface water are found in Tables A-11, A-12, A-13, and A-14. A variety of CoPCs could be dismissed based on this rescreening process (see Table A-16 for summary).

3.2.2 Comparison to Background Concentrations

Some metal CoPCs may exceed screening levels even at naturally occurring concentrations. Therefore, this section compares CoPC concentrations to background concentrations. Background concentrations in soil were taken from NJDEP (1993) and Shacklette et al. (1971), in that order of priority. In accordance with the practice of EPA Region IV (Wellman 1997, pers. comm.), maximum concentrations in soil were compared to twice the background concentrations. Constituents whose maxima were less than twice background were eliminated as CoPCs. Aluminum in soil was the only compound eliminated from further consideration based on comparisons to background concentrations. Results of rescreening soil CoPCs vs. background concentrations are found in Table A-11.

3.2.3 Rescreening with Less Conservative Benchmarks

As discussed above in the Step 2 uncertainty section, there is considerable variability in screening values, which, in turn, produce variable lists of CoPCs. Very conservative screening benchmarks are useful in that they are very protective, but their usefulness is limited if they are so conservative that they fail to screen out non-problematic chemicals (i.e., those that pose little or no risk.) When non-problematic chemicals are not screened out, considerable effort is spent considering chemicals that have little or no potential to cause impacts. This diverts attention and resources from those chemicals that potentially pose a real risk. As a result, it is useful to rescreen CoPCs against screening benchmarks from other reliable sources. This rescreening indicates whether an exceedance is due to a conservative screening benchmark or to the potential for ecological risk.

For soils, CoPCs were rescreened against the most conservative of the three alternative sources of soil benchmarks presented above in Table 1. No alternative benchmarks are available for iron and manganese. Therefore, onsite data were compared to average soil concentrations in the U.S. (Shacklette et al. 1971). Rescreening of soil contaminants eliminated six compounds (antimony, iron, manganese, nickel, selenium, and vanadium as CoPCs based on SQs less than or only nominally above 1.0 (Table A-15).

For surface water and groundwater, alternate ecological benchmarks were generated in two ways. First, hardness-dependent criteria for groundwater were recalculated at hardness values likely to occur at the site. As no hardness data were available for groundwater, the original screening values for groundwater were calculated at a default hardness of 100 mg/L. However, hardness can be estimated for groundwater in the following manner. Hardness is the sum of divalent cations, primarily Ca^{++} and Mg^{++}, expressed in terms of an equivalent amount of

$CaCO_3$. Groundwater at the site averaged of total of 260 mg/L of calcium and magnesium. This would equal a total hardness of approximately 650 mg/L[3]. Thus, for groundwater, hardness-dependent criteria were recalculated at a hardness value of 650 mg/L as $CaCO_3$.

Second, screening benchmarks for remaining CoPCs were reviewed for reasonableness. Based on this review, the ORNL values for barium and carbon disulfide were rejected. The barium benchmark is not consistent with the available toxicological data[4]. The carbon disulfide screening benchmark is also at variance with the toxicological data in EPA's AQUIRE database. EPA Region V has recently developed ecological benchmarks (U.S. EPA V 2000b), and benchmark values from this source were used in rescreening barium and carbon disulfide in surface and groundwater.

Use of these less conservative benchmarks reduced the numbers of CoPCs for both groundwater and surface water (Tables A-12, A-13). In groundwater, all but iron, manganese, and various forms of mercury were eliminated as CoPCs. The same compounds also remained potentially problematic (i.e., their SQs were greater than 1.0) in surface water.

Table A-16 presents the results of the rescreening, the compounds that were dismissed and the rationale for dismissal, and the compounds that remain as CoPCs after the rescreening.

3.2.4 Future Risk Scenario for Developed Area Soils

As concluded in the Section 2.3.1, there is no reasonable potential for significant current exposure to chemicals in the soils of the developed area. Exposure pathways from soil to ecological receptors should also continue to be incomplete well into the future. The developed area has been developed for at least 75 years, currently contains two ongoing businesses, and represents prime commercial land near one of the largest cities in the world. There is reason to believe that the site will remain developed for the foreseeable future. In addition, the primary physical barriers between soil chemicals and ecological receptors are buildings and pavement, which are long-lived even without active maintenance. Without active removal, these barriers to exposure will remain for a long time.

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[3] Hardness is defined as the sum of calcium and magnesium ions, expressed in terms of $CaCO_3$. As calcium made up most of the sum of calcium and magnesium, for simplicity it can be assumed that it was all calcium. Calcium makes up 40 percent of the molecular weight of $CaCO_3$, which means that total hardness, expressed as $CaCO_3$ should be about 2.5 times the calcium concentration. Thus, the total hardness in groundwater is estimated to be about 650 mg/L.

[4] For example, consider the benchmark for barium of 4 μg/L. In the Gold Book (U.S. EPA 1986, *Quality Criteria for Water 1986*), EPA states that "experimental data indicate that soluble barium concentrations in fresh and marine water generally would have to exceed 50 mg/L before toxicity to aquatic life would be expected." A search of the AQUIRE database suggests that the screening benchmark of 4 μg/L is also much too conservative. In a series of acute bioassays, barium was barely toxic to a number of organisms—acute LC50 ranged from 44,686,695 μg/L to 25,687 μg/L, with a geometric mean value of 413,000 μg/L. Chronic toxicity to *Daphnia* was also low—reproduction was decreased by an ecologically marginal 16 percent at 5,800 μg/L. In one series of bioassays, the barium salt of boric acid was considerably more toxic to the harlequinfish, which had a 48-hour LC50 value of only 260 μg/L barium. As boric acid is itself a wide-spectrum pesticide, this aberrant value can be attributed to the boric acid as opposed to barium.

Nonetheless, many of the chemicals, especially the metals, under the asphalt and crushed stone are also likely to persist into the distant future. Therefore, it would be useful to determine the potential ecological risk if for some reason the developed area were allowed to revert to natural habitat in the future. (It should be noted that the potential for reversion can be minimized by deed restrictions.) To assess this, metals in soil in the developed area were screened against the benchmarks employed in the original screens as well as the alternative benchmarks used in the refined screening analyses. As illustrated in Table A-17, a few of the metals may be problematic (e.g., having SQs greater than 1.0) under more conservative scenarios (maximum concentrations applied to conservative benchmarks). Mercury has the highest SQ values. Under the least conservative scenario—average soil concentration applied to the alternative benchmark—few of the metals except mercury would appear problematic. Even under the less conservative scenario, mercury generates an SQ of almost 200.

Thus, this analysis suggests that risks due primarily to mercury could be significant in the future if the building and pavement are removed, and if there is no significant reduction in mass or bioavailability between now and then.

3.3 Fate and Transport Characteristics of CoPCs

To ensure that relevant modes of exposure for the CoPCs are considered during the ERA, the available literature on fate and transport of CoPCs was reviewed. In view of the very large number of CoPCs, the following description of fate and transport focuses on contaminants with the highest SQs: mercury, PAHs, PCBs, and miscellaneous metal CoPCs—cadmium, chromium, copper, lead, and zinc.

3.3.1 Fate and Transport—Mercury

Mercury is a naturally occurring element that exists in the environment in different chemical forms. The predominant species in water, soil, and sediment is ionic mercury (Hg^{2+}). Ionic mercury can exist in a free ionic form, but the majority is adsorbed or chemically bound to organic matter. In soils, mercury tends to be bound to organic matter and does not readily leach into groundwater. Although mercury in soils generally has low bioavailability, it can occasionally contaminate underlying groundwater. In aquatic environments, methylmercury (CH_3Hg^+; $(CH_3)_2$ Hg) represents a small fraction of total mercury (approximately 10 percent) in the water column (NOAA 1996). Methylmercury is formed primarily by sulfate-reducing bacteria that are able to use ionic mercury (Hg^{2+}) as an electron acceptor. However, methylmercury is highly toxic and tends to bioaccumulate and biomagnify in both terrestrial and aquatic food chains. The fate and transport of mercury in the environment is described in greater detail in Section 5.2 of the RI report.

3.3.2 Fate and Transport—PCBs

PCBs are sparingly soluble in water and exhibit a strong affinity for sediment and other organic matter. It is unlikely, however, that dissolved concentrations near this solubility limit would be

found even in highly contaminated systems because of the strong affinity of PCBs to adsorb to sediment or other organic matter. Although sorption and subsequent sedimentation immobilizes the bulk of PCBs in an aquatic system, PCBs stored in sediments may enter the aquatic food web and be accumulated in biota. In terrestrial ecosystems, PCBs also tend to sorb strongly to soils. Transport is generally associated with particle transport (e.g., overland water flows), and PCBs do not generally contaminate groundwater. As in aquatic environments, PCBs in terrestrial systems, if bioavailable, will bioaccumulate in biota.

3.3.3 Fate and Transport—PAHs

PAHs include a large number of compounds, which vary in terms of molecular weight, hydrophobicity, persistence, and toxicity. They are ubiquitous compounds with a large number of sources: incomplete combustion, asphalt, oil, etc. The fate and transport characteristics of PAHs are roughly correlated with molecular weight. With increasing molecular weight, aqueous solubility decreases, and melting point, boiling point, and the log K_{ow} (octanol/water partition coefficient) increase. In turn, these higher-molecular-weight compounds are increasingly soluble in fats, are less resistant to oxidation and reduction, and have a decreased vapor pressure (Eisler 1987a).

The higher-molecular-weight species are highly persistent in the environment. Fate and transport of PAHs in aquatic environments is dominated by sorption. Due to low water solubility and high K_{ow}, PAHs are primarily found adsorbed to sediments or suspended particles. The degree of adsorption of PAHs to sediments is a function of OC content and particle size.

Given their tendency to sorb strongly to particles, and low volatility, deposition of PAHs in the sediments is the primary loss process from the water column. Once deposited in the sediments, PAHs are subject to microbial degradation and burial, after which their bioavailability becomes limited. Compounds with four or fewer cyclic rings are most amenable to microbial degradation processes in aquatic environments. Volatilization is a relatively minor fate process, especially for the higher-molecular-weight PAHs.

PAHs in terrestrial systems are also generally tightly adsorbed to soils. Therefore, PAHs do not readily leach from soils or contaminate groundwater. Rather, PAH contamination tends to remain localized near spill areas unless transported elsewhere by overland flows that erode and transport the PAH-bound soil particles.

Despite their high lipid solubility, PAHs do not tend to bioconcentrate or biomagnify in either aquatic or terrestrial food chains because most upper level biota can rapidly metabolize PAHs. Fish, for example, can metabolize PAHs extensively and rapidly, which explains why the compounds are frequently undetected, or only detected in low concentrations in the liver (Eisler 1987a). However, metabolic processes are slower in some invertebrate taxa, and PAHs do bioaccumulate in some invertebrate taxa, especially shellfish (Tracey and Hansen 1995).

3.3.4 Fate and Transport—Miscellaneous Metals

Several metals—cadmium, chromium, copper, lead, and zinc—were found at elevated concentrations in sediments and soils. These metals vary in their solubility, leaching characteristics, and potential to contaminate groundwater. However, in soils, these metals tend for either form insoluble salts or adsorb to organic matter. Therefore, their transport from terrestrial systems to aquatic systems is primarily via overland flow and transport of particles. Once in aquatic systems, these compounds tend to precipitate to the sediments. Most of these metals do not readily bioaccumulate or biomagnify in food chains, although cadmium will tend to biomagnify somewhat in food chains.

3.4 Mechanisms of Toxicity and Selection of Toxicity Reference Values (TRVs) for CoPCs

The mechanisms of toxicity and relevant toxicological data are reviewed for each of the major CoPCs. TRVs are doses of a chemical shown to have minimal or no ecological effects on an organism. These values are necessary for the interpretation of the food chain models that will be conduced in subsequent sections.

3.4.1 Mechanisms of Toxicity and Selection of TRVs—Mercury

Methylmercury is more toxic than inorganic mercury and tends to bioaccumulate and biomagnify[5]. Methylmercury is primarily found in aquatic systems; thus, mercury toxicity tends to be most problematic to species higher up in aquatic food chains. Methylmercury in birds has been demonstrated to affect various organ systems, with young birds and embryos being more sensitive than adults (Eisler 1987b). Toxic effects of methylmercury to avian species include altered behavior, hepatic lesions, ataxia, weakness, muscular atrophy, and death. The most sensitive indicator of exposure appears to be reduced fecundity manifested primarily as a decline in fledgling rates. For birds, a value for methylmercury of 0.064 mg/kg-day from Heinz (1979) is often used as a lowest-observed-adverse-effect level (LOAEL). This was a multigenerational study in which ducks were fed a ration with approximately 0.5 ppm methylmercury. Reproduction was significantly suppressed for the second generation, but not the initial generation or the third generation. This LOAEL has been translated to a NOAEL with 10-fold (Sample et al. 1996; U.S. EPA 1997b) and 2-fold (U.S. EPA 1993c) uncertainty factors, producing very different NOAELs of 0.0064 mg/kg-day and 0.032 mg/kg-day. The smaller uncertainty factor was justified “because the LOAEL appeared to be very near the threshold for dietary effects” (U.S. EPA 1993c).

[5] Methylmercury includes monomethyl- and dimethylmercury. Monomethylmercury tends to bioaccumulate and biomagnify. Dimethylmercury is generated in aquatic systems, but its high vapor pressure tends to push it into the atmosphere. Dimethylmercury does not biomagnify, though it can bioaccumulate. The ongoing discussion will focus upon monomethylmercury.

As demonstrated above, the arbitrary choice of uncertainty factor significantly affects the final TRV and, consequently, the perceived risk. Close examination of the Heinz study suggests that U.S. EPA (1993c) is more scientifically defensible. Due to slight variations in feeding rate and the concentrations of mercury in the final ration, the actual dose of methylmercury varied slightly from generation to generation. Ducks in the second generation, whose reproduction was significantly reduced, were exposed to approximately 0.082 mg/kg-day. Those in the third generation, whose reproduction was not significantly reduced, were exposed to about 0.060 mg/kg-day. Consequently, it is apparent that EPA's (1993c) conclusion that the LOAEL from Heinz was close to a NOAEL was valid. Therefore, the NOAEL (0.032 mg/kg-day), from U.S. EPA (1993c), was chosen as the avian TRV for methylmercury.

Experiments with mammals show them to be about as sensitive as birds to effects of methylmercury. In sub-chronic feeding studies with mink, Wobeser et al. (1976) found no significant ecological effects on mink at a feeding rate of a 0.15 mg/kg-day methylmercury. In addition, Halbrook et al. (1999) found no effects on mink at a methylmercury dose of 0.023 mg/kg-day. Furthermore, there were no impacts of methylmercury on mink at a dose of about 0.08 mg/kg-day (Kirk 1971, as cited by Halbrook et al. 1999) and 0.16 mg/kg-day (Wren et al. 1987). Thus, the Wobeser et al. (1976) value was used in this risk assessment as the mammalian TRV for methylmercury. An uncertainty factor of 3, as in U.S. EPA (1993c), was applied for sub-chronic to chronic, resulting in a mammalian TRV of 0.05 mg/kg-day.

Mercury exposure in terrestrial ecosystems is dominated by inorganic mercury. An avian TRV for inorganic mercury was obtained from toxicity studies conducted on the Japanese quail (Hill and Schaffner 1976). This study found a NOAEL dose of 0.45 mg/kg-day. A mammalian TRV was based on a mink NOAEL of 1.01 mg/kg-day inorganic mercury (Aulerich et al. 1974).

In summary, a methylmercury TRV of 0.05 mg/kg-day was chosen for mammals, while 0.032 mg/kg-day of methylmercury was chosen for birds. The inorganic mercury TRVs chosen are 0.45 mg/kg-day for birds and 1.01 mg/kg-day for mammals.

3.4.2 Mechanisms of Toxicity and Selection of TRVs—PCBs

Because of their hydrophobicity, miscibility with organic compounds, and resistance to metabolic breakdown, PCBs have the potential to bioaccumulate and biomagnify in food chains, indicating that high-trophic-level wildlife may have higher exposure levels. PCBs can elicit a broad range of toxic effects in laboratory animals; however, adverse reproductive effects (e.g., litter size, offspring survival) appear to be the most sensitive endpoints of PCB toxicity (Golub et al. 1991; Rice and O'Keefe 1995; Hoffman et al. 1996). Therefore, although PCBs can cause acute toxicity from direct high-level exposure, the primary ecological concerns associated with PCBs are the potential reproductive effects in higher-trophic-level wildlife resulting from chronic low-level dietary exposure. Reproductive success in fish, mammals, and birds can be affected directly by toxic action on the differentiated reproductive tract or indirectly on systems that regulate reproduction (e.g., endocrine and central nervous systems). In laboratory studies, PCBs have been reported to elicit a broad range of direct and indirect effects that could conceivably lead to decreased reproductive function. For example, the liver is one of the primary targets of PCB toxicity and changes in the activity of liver enzymes can result in

modulation of steroid hormone levels, suggesting a mechanism by which PCBs could alter reproductive function. PCBs have also been implicated in the modulation of other systems important for reproduction, such as the central nervous system, adrenal gland, and thyroid hormone levels. Direct effects on the gonads and the female reproductive tract have also been reported (Fuller and Hobson 1986; Peakall 1986; Barron et al. 1995). Despite the extensive amount of information on the effects of PCBs on reproduction in mammals and birds, the precise mechanism or mechanisms by which PCBs cause these effects remains unclear.

Mink are widely regarded as among the most sensitive piscivores to PCBs. Aulerich and Ringer (1977) derived a NOAEL for PCBs of 1 ppm in food. A TRV for mammals of 0.16 mg/kg-day was derived from this study. A TRV of 0.41 mg/kg-day was derived for avian species from the NOAEL from the chronic feeding study of PCBs to screech owls (McLane and Hughes 1980).

3.4.3 Mechanisms of Toxicity and Selection of TRVs—PAHs

Discussion of the ecotoxicology of PAHs is complicated because they are a class of compounds that differ in terms of relative toxicity and modes of action. Due to their generally high hydrophobicity, plants do not readily incorporate PAHs. In addition, most higher plants can catabolize benzo(a)pyrene, and possibly other PAHs (Eisler 1987a). Therefore, plant tissue PAH residues are low and PAHs do not generally pose risk to plants or to most herbivores.

Although PAHs are known to bioaccumulate in biota, they are rapidly metabolized and excreted by most species and, therefore, biomagnification is not significant in biota in either terrestrial or aquatic ecosystems (Eisler 1987a). Toxic effects of PAHs are generally most significant to organisms low in the food chain, via direct exposure to water, soil, and sediments.

The effects of PAHs on avian and mammalian species are demonstrated by the following experiments. In a study with mallards, PAH concentrations of 4,000 mg/kg in the diet were administered for a 7-month period and resulted in no mortality or visible signs of toxicity. However, liver weight increased and blood flow to the liver increased by 30 percent compared to controls (Eisler 1987a). At doses of 10 mg/kg-day, the offspring of rats exposed to benzo(a)pyrene during gestation suffered a number of problems, including impaired fertility (Mackenzie and Angevine 1981).

The evaluation of PAH toxicity to mammals was based on a study by Mackenzie and Angevine (1981) that examined the reproductive effects of benzo(a)pyrene on mice. Female CD-1 mice were exposed to benzo(a)pyrene ranging from 10 to 160 mg/kg through daily intubation. Treatment commenced on day 7 after the best estimated time of conception and continued through day 16 of gestation. Mean pup weight was observed to be significantly reduced in the 10 mg/kg treatment group. This treatment was considered to be a LOAEL. The estimation of the TRV was therefore based on the application of a 10-fold level of uncertainty to the toxicity estimate, producing a TRV of 1 mg/kg-day.

Studies on the chronic toxicity of PAHs to birds, particularly with regard to impacts on reproduction or other ecologically relevant endpoints, are not available. There are essentially two chronic studies. As noted above, Patton and Dieter (1980) report a feeding study in which

ducks were fed 4,000 ppm (= dose of about 400 mg/kg-day) in food for 7 months, with little or no effect noted. Hough et al. (1993) examined the effects of benzo(a)pyrene on pigeons. Three- to six-month-old pigeons were administered a dose of 10 mg/kg intermuscular (IM) injections weekly for a period of 5 months. The treatment birds were reported to have suffered complete reproductive failure and an associated gross alteration in ovarian structure.

Thus, two studies looking at birds yield an ecological NOAEL of about 400 mg/kg-day versus and ecological LOAEL of 1.43 mg/kg-day. Much of the difference is likely due to difference in exposure method. PAHs taken in food will travel directly from the stomach to the liver, where most all of the PAHs will be sequestered, broken down, and then excreted. Internal concentrations of PAHs from dietary exposure, and attendant tissue exposure, are generally unmeasureable except for the liver. The IM injection bypasses the liver. Thus, PAH concentrations faced by internal tissues are higher via IM injection than dietary exposure. The second difference is the type of PAHs. The PAHs in the food fed to the mallards were largely lower-molecular-weight PAHs—naphthalenes, naphthenes, and phenanthrene—which probably have reduced toxicity compared to higher-molecular-weight PAHs. In contrast, the IM injections were of benzo(a)pyrene, which is one of the more toxic PAHs. Thus, the Hough et al. study exaggerates likely toxicity from total PAHs both by its mode of exposure and by the use of a most toxic PAH. The Patton and Deiter study underestimates potential toxicity by use of less toxic PAHs.

Given this wide range of potential endpoints, the mammalian TRV of 1 mg/kg-day was applied to birds without an application factor. The rationale for this is that the birds and mammals both metabolize and excrete PAHs rapidly. In addition, the mammalian TRV is based on benzo(a)pyrene. This exaggerates likely toxicity of total PAHs, which includes significant amounts of lower-molecular-weight, presumably less toxic PAHs. In addition, this TRV (1 mg/kg-day) is lower than the Hough et al. LOAEL of 1.43 mg/kg-day, providing a margin of safety from this already very conservative TRV.

Thus, the final TRVs for total PAHs are 1 mg/kg-day for both mammals and birds.

3.4.4 Mechanisms of Toxicity and Selection of TRVs—Miscellaneous Metal CoPCs

Several metals (cadmium, chromium, copper, lead, and zinc) were found in elevated concentrations in sediment and soils. Although these chemicals do not tend to biomagnify in food chains, they are bioaccumulated to some extent by organisms low in the food chain. Therefore, predators can receive toxic levels from their prey and from incidental soil or sediment ingestion. TRVs for these chemicals were generally obtained from Sample et al. (1996) without modification (Table A-20), except for the following: The chromium TRV for birds was set equal to 1.6 mg/kg-day, as presented in U.S. EPA (2000a). The zinc TRV for birds was set equal to 130.9 mg/kg-day from Stahl et al. (1990). Sample et al. mistakenly identify this as a LOAEL, but review of that work indicates that there were no significant effects at this dose.

3.5 Exposure Characterization

3.5.1 Complete Exposure Pathways and Ecological Receptors

A conceptual site model depicts the primary and secondary exposure pathways that exist on the site (Figure 3). Complete exposure pathways require that a contaminant travel from the source to the ecological receptor and be taken up by the receptor via one or more exposure routes. Potential exposure pathways to biota in OU1 include direct contact with surface water, sediment, suspended particles, and soil; ingestion of water, sediment, suspended particles, and soil; and uptake from food containing bioaccumulated chemicals.

For terrestrial receptors, the available information suggests that the primary risks to site-related chemicals occur through two idealized pathways: soil to worm to worm-predator, and soil to plant to small vertebrate to top predator. These pathways are depicted in Figure 3. As noted above, these pathways tend to be idealized. Most worm predators will also prey on other invertebrates. Similarly, small vertebrates such as mice and voles may bioaccumulate soil contaminants from the consumption of plants, invertebrates, and incidental soil ingestion. Both types of predators may also be exposed to contaminants via ingestion of water. Complete exposure pathways to aquatic receptors include direct toxicity of surface water and sediments to fish and aquatic invertebrates. In addition, several of the CoPCs bioaccumulate in aquatic systems; therefore, there is a complete exposure pathway from contaminants in water and sediments to fish to fish-eating wildlife. An allied pathway of concern is from sediments to macroinvertebrates to consumers of those macroinvertebrates. These pathways are also depicted in Figure 3.

The separation into terrestrial and aquatic pathways is also an idealized distinction that is blurred in the real world. Many organisms are linked in both dimensions. For example, water snakes will take both fish and small terrestrial mammals, and will, in turn, be eaten by both herons and fox.

Complete exposure pathways from groundwater to ecological receptors at OU1 are limited to any groundwater that discharges to the West Ditch or the onsite basin. These potential risks are more directly addressed by the surface water samples taken from those areas. Groundwater under OU1 may also pose risk to receptors in OU2 as that groundwater discharges to Berry's Creek and the Diamond Shamrock/Henkel Ditch (north). However, that risk is more properly addressed in the risk assessment for OU2. Therefore, groundwater will not be considered further.

3.5.2 Assessment Endpoints

Assessment endpoints are defined as "explicit expressions of the actual environmental value that is to be protected, operationally defined by an ecological entity and its attributes" that "focus a risk assessment on particular components of the ecosystem that could be adversely affected" by CoPCs (U.S. EPA 1997a). The selection of assessment endpoints depends on "1) the contaminants present and their concentrations; 2) mechanisms of toxicity; 3) ecologically

relevant receptor groups that are potentially sensitive or highly exposed to the contaminant and attributes of their natural history; and 4) potentially complete exposure pathways" (U.S. EPA 1997a).

The assessment endpoints were selected for the following reasons: 1) they reflect important ecosystem components of the site; 2) they have practical measurement endpoints that have been reported in the literature; and 3) they are most likely to be affected by the chemicals present at the site. For a selected assessment endpoint to provide a meaningful measure of effect, a testable hypothesis and measurement endpoint must be practical and possible to implement in a field or laboratory study. Thus, the endpoints listed below were selected to assess the effects of the CoPCs on OU1 of the Ventron/Velsicol ecosystem.

The selected assessment endpoints for the site are:

1. Maintenance of a healthy, reproducing community of worm-eating birds and mammals. Mercury and some other metals are found in high concentrations in onsite soils. Once bioaccumulated in worms and other soil macroinvertebrates, these chemicals may affect consumers of these soil macroinvertebrates.

2. Maintenance of a healthy, reproducing community of top predators or terrestrial prey. Mercury and some other metals are found in high concentrations in onsite soils. Once bioaccumulated into small prey, these chemicals may affect predators such as hawks and foxes.

3. Maintenance of a healthy, reproducing benthic invertebrate community. Several metals, PCBs, and PAHs were found at levels above screening criteria in aquatic sediments. These chemicals may have toxic effects on the benthic invertebrates that inhabit those sediments.

4. Maintenance of a healthy, reproducing insectivorous bird and bat community. Mercury, PCBs, and several other contaminants were found in aquatic sediments where larval aquatic insects may bioaccumulate these compounds from the sediment. These bioaccumulated chemicals may pose a risk to predators that consume emergent aquatic insects from the site.

5. Maintenance of healthy, reproducing populations of bird and mammal species that feed on aquatic benthos. The chemicals found in OU1 sediments are bioaccumulated by aquatic benthos. These compounds may then pose a risk to consumers of aquatic benthos from the site.

6. Maintenance of a healthy, reproducing community of wildlife feeding on fish from the West Ditch. Mercury was found in aquatic sediments of the West Ditch, and PCBs, while not measured, may also occur there. They were also found in the onsite basin and immediately downstream in the Diamond Shamrock/Henkel Ditch (north). These compounds biomagnify in aquatic food chains and, subsequently, may pose risks to consumers of fish. The onsite basin has no connection to other surface water bodies. It is assumed to

not be fish habitat, and the potential pathway from chemicals in the onsite basin to piscivores is assumed to be incomplete. Fish do occur in the West Ditch (at higher tides), so the pathway from sediments to piscivores is potentially complete for the West Ditch.

In addition to the assessment endpoints listed above, other species and biotic communities are exposed to site-related chemicals that could, in turn, pose ecological risk. For example, onsite levels of metals pose potential risk to onsite plants , soil invertebrates, and indirectly via the food chain to small and large herbivores such as mice and deer. OU1 will also be habitat to reptiles and amphibia, which may also be exposed to site-related chemicals. Per EPA guidance (U.S. EPA 1997a), however, the assessment endpoints proposed above focus upon a subset of potential endpoints that are most likely to be affected by site-related chemicals and those most likely to be of concern to society.

3.5.3 Testable Hypotheses

The following testable hypotheses were developed from the conceptual site model and pathway analysis:

- Are insectivorous, worm-eating birds and mammals feeding on the site adversely affected by levels of CoPCs through the food chain?
- Are top predators feeding on terrestrial prey from the site adversely affected by levels of CoPCs from the food chain?
- Is the structure and function of the benthic community of the ditch and onsite basin adversely affected by the levels of CoPCs in sediments at the site?
- Are predators that feed on the aquatic invertebrates from the ditch adversely affected by levels of CoPCs from the food chain?
- Are predators that feed on fish that inhabit the ditch adversely affected by levels of CoPCs from the food chain?

Consideration of these hypotheses will help assess the degree of risk, if any, to the health of the OU1 ecosystem.

3.6 Refinement of Risk to Assessment Endpoints

The benchmarks used in the conservative and refined screening analyses conducted in Steps 2 and 3 apply to a wide range of receptors, not necessarily to the site-specific assessment endpoints chosen for the risk assessment or to the specific modes of action of site-related chemicals. For example, some of the soil benchmarks pertain to plants or to soil microflora, as opposed to the assessment endpoints selected above. On the other hand, the sediment benchmarks largely pertain to protection of aquatic benthos, not to consumers of those benthos.

These benchmarks could be over- or underprotective for predators feeding on the benthos, and this potential can be evaluated with simple food chain models. The following represents a refined assessment of risk specific to the assessment endpoints.

3.6.1 Potential Risks to Aquatic Benthos

In the initial screening, concentrations of chemicals in sediment samples were compared to conservative ecological benchmarks, mostly environmental effects range-low (ER-L) and low-effect levels (LEL), to screen for ecological effects. Based on this screening analysis, it would appear that there is high potential for risk from a number of metals, as well as PAHs and PCBs. To interpret what these exceedances mean, it is necessary to understand the derivation of the LEL and ER-L values (Long and Morgan 1991). The LEL and ER-L benchmarks are based on the observed co-occurrence between a level of a chemical and a potential impact on benthic invertebrates. However, there is generally high covariance among toxic chemicals in environmental samples. That is, high levels of cadmium tend to co-occur with high levels of copper, lead, zinc, PAHs, PCBs, etc. In addition, many of these toxic chemicals tend to co-occur, in the field, with high levels of naturally occurring toxins such as ammonia and hydrogen sulfide. Consequently, it is acknowledged that these benchmarks have not established causality (Peddicord and Lee 1998; O'Connor 1999), and even the authors of the benchmarks agree that they are not applicable to estimation of risk for individual chemicals (Long and McDonald 1999). In addition to the problem with assigning causality to any one chemical, both the ER-Ls and LELs have a high potential for false positives, about 90 percent. Therefore, while failure to exceed an ER-L or LEL indicates with certainty that that a compound is unlikely to cause toxicity, it is difficult to interpret what an exceedance of an LEL or ER-L really means.

Given the difficulty interpreting ER-L and LEL exceedances, NOAA also reported environmental effects range-medium (ER-M) values, which are concentrations in which about half of sediments showed impacts (Long and Morgan 1991). Compared to exceedance of an ER-L value, exceedance of an ER-M value suggests a higher probability of potential impacts. Impacts of some of the compounds, especially the metals, are also thought to be additive. Some analysts have quantified this relationship, summing SQs based on multiple chemicals to account for the potential additivity of toxicity. In a detailed analysis comparing chemistry and bioassay mortality for Great Lakes sediments, Canfield et al. (1996) found that bioassay toxicity generally did not occur when the sum of eight SQs[6], based on the ER-Ms, was below about 6.0 (or averaged about 0.75 per SQ.)

To better understand the potential risk to benthos, sediment CoPCs were rescreened against ER-Ms, and the SQs obtained with ER-M values were summed (Table A-18). Summing the SQs for the maximum sediment concentrations of the eight compounds produces a sum of SQ of 14.3, about 2½ times the threshold below which toxicity was generally not observed. It should also be stressed that this summed SQ does not include the SQ for mercury, which has the highest SQ of any compound. If mercury is included in the sum of ER-M values, the maximum and average concentrations of chemicals produce average ER-M SQ values of 87.9 and 46.8. These

[6] Concentrations of eight chemicals were compared to ER-M values: cadmium, chromium, lead, nickel, zinc, chrysene, benzo(a)pyrene, and benzo(g,h,i) perylene.

values are well above the average ER-M SQ value, about 0.75, below which toxicity was rarely observed. Thus, the chemical data indicate that a number of chemicals in sediments, especially mercury, exceed even less conservative ER-M values.

As discussed above, exceedance of an ER-M or a summation of ER-Ms only suggests that there might be impacts, not that impacts are certain. In a companion analysis to Canfield et al. (1996), Ingersoll et al. (1996) also produced no-effect concentrations (NECs), which are the highest concentrations that co-occurred with no toxicity[7]. NEC values demonstrate that no toxicity may occur at higher concentrations than the ER-M levels. In addition, conditions at OU1 are those in which toxicity is unlikely to be expressed: probable domination by relatively insensitive infauna such as midge larvae and worms (oligochaetes) and very high levels of OC and potentially sulfides. It is notable that the very high mercury concentration that occurred in the onsite basin co-occurred with very high OC concentrations of 18 percent and 9.7 percent, which will tend to reduce bioavailability of the CoPCs.

Notwithstanding these modifying factors, the level of exceedance for mercury alone is high. Even when using the less conservative ER-M benchmarks, the maximum and mean SQs for mercury alone are roughly 2,000 and 1,000. By comparison, Canfield et al. (1996) always observed toxicity when any compound had an ER-M SQ greater than about 40. The high mercury concentrations and high SQ values were based on samples from the onsite basin, where mercury concentrations averaged over 1,000 mg/kg and habitat value is low. SQ values are considerably lower for the West Ditch, which had much lower mercury concentrations. Nonetheless, SQ values based on ER-M values for mercury concentrations from the West Ditch are still about 100.

3.6.2 Estimation of Potential Risk to Consumers of Aquatic Invertebrates

In addition to the risks to the benthos themselves, the CoPCs in sediment may pose risks to predators of the benthos if significant bioaccumulation occurs. Potential predators of benthos include birds and raccoons that could feed on the mud flats during low tides and fish and ducks foraging at high tide. After emergence, adults of the benthic insects also pose a complete exposure pathway from chemicals in sediments to bats, swallows, and redwing blackbirds. Potential risks to these sentinel species will be examined below using simple food chain models.

Concentrations of metal CoPCs in aquatic insects were predicted from regression models produced by ORNL (Bechtel Jacobs 1998a). Models based on all data from Table 3 of the ORNL reference were used to predict metals concentrations. The ORNL models predict dry weight concentrations. These were converted to wet weight benthos concentrations by assuming that benthos were 75 percent water (Table A-19).

PCB and PAH concentrations in aquatic benthos were predicted from Tracey and Hansen (1995). Tracey and Hansen expressed their biota-to-sediment accumulation factors (BSedAF) values normalized to OC in the sediments and lipid in the benthos, both on a dry weight basis.

[7] The NEC values are similar to Apparent Effects Threshold (AET) values produced elsewhere.

Median BSedAFs for PAHs and PCBs were 0.29 (gram lipid per gram OC) and 1.11 (gram lipid per gram OC), respectively. Aquatic sediments in the onsite basin averaged 14 percent OC. Sediments in the West Ditch were not analyzed for OC, but nearby locations in the Diamond Shamrock/Henkel Ditch (north) averaged 26 percent organic carbon[8]. Thus, a reasonable assumption is that the West Ditch soils had an average of 20 percent OC. Aquatic invertebrates have lipid levels of about 2.0 percent (Oliver and Niimi 1988). Thus, the final BSedAF for PAHs was estimated as follows:

0.29 * 2% lipid / 20% OC = 0.029 wet weight biota to dry weight sediment

Likewise, the final BSedAF for PCBs was estimated as:

1.11 * 2% lipid / 20% OC = 0.11 wet weight biota to dry weight sediment.

Four different sentinel receptors were evaluated with respect to risks of chemicals in benthic organisms. Redwing blackbirds and brown bats were selected to represent consumers of adult aquatic insects after emergence. Raccoons and mallard ducks were selected to represent consumers of larval aquatic insects and aquatic benthos. Consumption rates for the four sentinel species were estimated from U.S. EPA (1993b). It was conservatively assumed that species ate only aquatic organisms from the site. It was also conservatively assumed that all of the mercury in aquatic invertebrates was methylmercury, although a more reasonable value appears to be about 25 percent (Becker and Bigham 1995). Incidental soil ingestion was assumed to be zero for the consumers of adult insects. Incidental soil ingestion for raccoons and ducks was based on U.S. EPA (1993b). As the incidental soil ingestion could be a significant source of exposure to inorganic mercury, the duck and the raccoon food-chain analyses included potential risks from inorganic mercury as well as methylmercury. Given the conservativeness of the assumptions and relative insignificance of exposure to drinking water, exposure via drinking water was ignored.

The food-chain analyses suggest that mercury poses potential risk to all of the potential receptors, although none of the SQs was greater than 10 (Tables A-20 and A-21).

3.6.3 Estimation of Risks to Piscivorous Wildlife—West Ditch Only

Of the CoPCs found in the West Ditch water and sediment, only mercury is sufficiently bioaccumulative to pose risks to piscivorous wildlife. Mercury concentrations in small fish were estimated as follows. A bioaccumulation factor (BAF) of 1,600,000 L/kg for trophic level 3 fish (from U.S. EPA 1997b) was applied to the maximum and mean concentrations of methylmercury obtained from the water in the West Ditch. These water concentrations produced estimates of maximum and mean mercury concentration of 4.4 and 3.0 mg/kg in fish. These predicted values are likely to significantly overestimate concentrations in West Ditch fish for several reasons. First, the fish that inhabit this ditch will be small and only intermittently resident during high tides. Mercury bioaccumulation will likely be lower for the smaller and

[8] SD-05 had 36.4 percent, SD-06 had 7.6 percent, and SD-08 had 34 percent organic carbon.

transient fish inhabiting the West Ditch. In addition, the BAF is applicable to dissolved methylmercury concentrations. No dissolved methylmercury concentrations were measured in the West Ditch, but only about 40 percent of the methylmercury was dissolved in the water column of the onsite basin. Applying this value to the West Ditch would reduce estimated fish concentrations by 60 percent.

Given the problems with the BAF approach, mercury concentrations in small fish were also estimated with a food chain multiplier. According to ORNL regression models, invertebrate

prey in the West Ditch should have a maximum and mean concentration of 0.28 and 0.21 mg/kg of total mercury[9]. According to U.S. EPA (1993b), mercury concentrations increase by about 128 percent between trophic level 2 and trophic level 3. Applying this food chain multiplier to the predicted benthos concentrations suggests maximum and mean fish concentrations of 0.36 and 0.27 mg/kg.

The geometric mean of the two estimation methods was chosen as a best estimator, producing estimates of 1.2 mg/kg as the maximum mercury concentration and 0.9 mg/kg as the mean mercury concentration in West Ditch fish.

PCBs were not sampled in the West Ditch. However, these compounds were found in the onsite basin. PCB concentrations in the West Ditch were assumed to equal those in the onsite basin. Body burdens of PCBs in small fish were based on BSedAF values provided in Tracey and Hansen (1995). Small killifish were assumed to have a lipid concentration of 4 percent, a reasonable value for small fish. Based on BSedAF value and OC concentrations assumed for the West Ditch (20 percent), the wet-weight-fish-to-dry-weight-sediment concentration BSAF can be estimated as follows.

1.11 * 4% lipid / 20% OC = 0.27 wet weight biota to dry weight sediment.

Based on the observed maximum and mean sediment concentrations of 0.73 mg/kg and 0.58 mg/kg, this BSAF estimates 0.20 mg/kg and 0.16 mg/kg PCBs wet weight in small fish. The mink and kingfisher were chosen as sentinel species to model the risk to piscivores. Risks to these species were assessed using the default, very conservative assumptions: 100-percent residence, 100-percent diet of contaminated species, and SQs based on NOAELs. Based on these very conservative assumptions, it was estimated that there are potential impacts of mercury on both mink and kingfishers (Table A-22).

3.6.4 Potential Risks to Consumers of Terrestrial Invertebrates

Onsite soils in OU1 have elevated concentrations of a number of CoPCs, and several of these compounds exceeded conservative and less conservative screening values. After rescreening, seven metals (chromium, copper, lead, mercury, silver, thallium, and zinc) remained as CoPCs

[9] These concentrations are different from those presented in Table A-19 because the estimated mercury concentrations for benthos presented in Table A-19 are based on the mercury data from sediments from the onsite ditch and the West Ditch. The estimates above are based only on West Ditch data.

(Table A-16). These CoPCs can bioaccumulate in soil invertebrates and potentially be problematic to consumers of those invertebrates.

Potential food chain exposure was modeled based on ORNL models for earthworm bioaccumulation (Sample et al. 1998a). This is a conservative assumption because earthworms tend to have higher bioaccumulation rates of soil chemicals than other soil macrobiota. Worm bioaccumulation was based on the regressions of the entire datasets (Table 12 of Sample et al. 1998a) except for chromium and mercury. In the case of chromium, the regression had a negative slope, which produced higher estimated worm concentrations at the mean soil concentration than at the maximum. In this case, the higher of the two estimates was applied to both exposure scenarios based on mean and maximum soil concentrations. In the case of mercury, the regression from the original dataset was employed (Table A-23). The latter exception produced higher (i.e., more conservative) estimates of mercury in worms. ORNL models predict dry weight concentrations in worms. These were converted to wet weight worm concentrations by dividing by 6.25, based on the U.S. EPA (1993b) data that worms are 84-percent water[10].

Risks to avian and mammalian consumers of soil invertebrates were modeled with a conservative diet consisting of 100-percent worms and incidental soil. Shrews and woodcocks were chosen as sentinel species for this assessment endpoint. Consumption rates for each organism were based on information provided in U.S. EPA (1993b). Based on U.S. EPA (1993b), incidental soil ingestion for the woodcock was assumed to be 10.4 percent of the total dry weight food consumption. Appling the conversion factor for a worm diet produced an incidental soil ingestion rate of 1.6 percent of total consumption, on a wet weight basis. No data on incidental soil ingestion were found for the shrew. It was assumed to have the same incidental soil consumption as the woodcock.

The results of the screening assessment are presented in Table A-24. As can be seen, the simple food chain analysis suggests that risks will exist for both species. For the scenario based on the maximum soil concentrations, almost all of the metals have SQs greater than 1.0 for both receptors. Mercury and lead pose the greatest risks, but chromium and zinc may also pose risks to both receptors. It must be stressed that these SQs are based on a number of very conservative assumptions. The potential effects of these conservative assumptions are considered below in Section 3.6.6.

3.6.5 Potential Risks to Top Predators in the Terrestrial Food Chain

Once bioaccumulated by resident rodents and other small prey, contaminants in site soils could affect top predators feeding in these areas. Bioaccumulation by small vertebrate prey of the soil CoPCs addressed above was estimated with ORNL regression models (from Table 8 of Sample et al. [1998b], which relate dry weight soil concentrations to dry weight concentrations in small rodents. These values were applied to the soil chemicals remaining as CoPCs (Table A-23).

[10] If the worm is 84-percent water, the dry weight mass is 16 percent of the total. Total wet weight is therefore 100 percent/16 percent, or 6.25 times dry weight.

Unfortunately, there are no models for silver and thallium, so the concentrations of these chemicals in small mammals could not be estimated.

While there is a regression model for mercury (Sample et al. 1998b), this model is based on limited data, is not statistically significant, and provides neither useful nor sufficiently conservative predictions[11]. In view of these problems, Sample et al. (1998b) recommends a constant uptake factor be applied, but this recommendation also does not produce useful predictions. The uptake rate of metals by small mammals (metals concentration in mammal/metals concentration in soil) decreases as soil concentrations increase. Consequently, a constant uptake factor greatly exaggerates likely uptake. This is especially true for the uptake factor for mercury, in which the uptake factor is calculated at concentrations very different from the site concentrations.

Therefore, bioaccumulation of mercury in small mammals was modeled to be a function of consumption, as in U.S. EPA (1999c). Bioaccumulation of mercury in small prey was assumed to follow the same slope as that for plants, for which a statistically significant regression equation does exist (Bechtel Jacobs 1998b).[12] The rationale for this method is as follows. Total intake of mercury by small mammals is the sum of the exposure from consumption of plants, small invertebrates, and incidental soil. Assuming an archetypal omnivorous diet of 50 percent vegetation, 50 percent earthworms, and 0.6 percent incidental soil ingestion[13], the resulting total intake at different concentrations of soil mercury can then be estimated with regression equations for plants (Bechtel Jacobs 1998b) and earthworms (Sample et al. 1998a). According to these ORNL regressions, the proportion of mercury uptake due to invertebrates falls as mercury concentrations in soil rise, and the uptake from plants is the dominant source of dietary mercury at higher soil concentrations. Therefore, the total dietary uptake tends to follow the slope of the plant uptake vs. soil.

Based on this assumption that mercury body burdens in small mammals versus soil concentrations will have the same slope as uptake by plants from soil, the body burdens of mercury at the maximum and mean concentrations (548 mg/kg and 120 mg/kg) can be estimated to be about 45 and 20 times, respectively, the body burdens at 0.5 mg/kg in soil. The latter value was chosen because the available data on paired soil-small mammal mercury concentrations are clustered around this soil concentration. Data presented in Sample et al. (1998b) suggest that small mammals inhabiting soils with about 0.5 mg/kg mercury will have body burdens of about 0.035 mg/kg mercury, dry weight. Small mammal body burdens at 548 mg/kg and 120 mg/kg can then be estimated to be about 1.6 mg/kg and 0.70 mg/kg, dry weight[14]. These two values convert to 0.4 mg/kg and 0.17 mg/kg as wet weight mercury concentrations.

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[11] This model predicts that concentrations of mercury in small mammals living at high soil concentrations will be lower than those at low soil concentrations of mercury.

[12] The equation is
ln plant conc, (mg/kg, dry weight) = -.996+ 0.544 ln soil concentration (mg/kg., dry weight).

[13] This is the incidental soil ingestion rate for meadow voles reported in U.S. EPA (1993b).

[14] That is, the small mammal concentration at 548 mg/kg is estimated by the plant regression equation to be 45 times the small mammal concentration at a soil concentration of 0.5 mg/kg. The latter is about 0.035 mg/kg, dry weight. Forty-five times this value produces an estimate of 1.6 mg/kg, dry weight.

The risk to two predators of the small mammals (red-tailed hawk and red fox) was then assessed based on these estimated concentrations in small mammals. Incidental soil ingestion was assumed to be 2.4 percent of total diet on a dry weight basis for the fox, based on information presented in U.S. EPA (1993a). No information could be found for the red-tailed hawk, so the value from the fox was assumed for the hawk. Based on the default standard conservative assumptions (100-percent residence, 100-percent consumption of prey, 100-percent absorption of chemicals in prey and soil), no compound poses any significant potential for risk to the top predators (Table A-25). Moreover, these assessments contain a number of very conservative assumptions (100-percent residence, 100-percent absorption, concentrations based on maximum soil concentrations).

On the other hand, as is clear from the discussion above, there is significant uncertainty concerning the estimation of mercury levels in small mammal prey. Using a constant uptake factor of 0.014, wet weight mammal to dry weight soil, as recommended by Sample et al. (1998b), produces estimated wet weight concentrations of 7.4 mg/kg and 1.6 mg/kg in small mammals at the maximum and mean soil concentrations, respectively. Risk to upper predators would occur under most scenarios if these mercury levels actually occurred in their small mammal prey.

3.6.6 Sensitivity of Screening Quotients to Conservative Assumptions

While the food chain analyses described above often generated SQs greater than 1, thereby indicating risk, it is important to understand the effects of the multiple conservative assumptions that were applied in these analyses. These assumptions and their potential effects are discussed below. When possible, the effect of the conservative assumption has been quantified to illustrate the likely effect of relaxing that assumption on the final SQ (Table A-26).

Assumption 1. 100-percent residence. The food chain analyses assumed that the sentinel species obtained 100 percent of their food and incidental soil from OU1. This is a realistic assumption only for the shrew. The lifetime foraging ranges of larger animals and small migratory animals are larger than OU1. A more realistic assumption is that the proportion of food obtained from OU1 is the ratio of its area compared to a representative home range for the animal. The latter was obtained from U.S. EPA (1993a) or elsewhere. For example, the mallard has a median home range of about 750 acres. To be conservative, it was assumed that the published home ranges included both the actual wetland foraging areas and contiguous terrestrial areas. In contrast, the entire undeveloped area of OU1 is about 19 acres. Therefore, the average duck would be expected to spend about 2.5 percent of its time foraging in OU1. Following the conservative assumption of 100 percent residence exaggerated likely exposure about 40-fold. For migratory birds (heron, redwing blackbird, mallard), this ratio was multiplied by 2 to reflect the lack of exposure during winter. Table A-26 contains the results of relaxing this assumption.

Assumption 2. 100-percent consumption of contaminated prey: The food chain analyses also assumed that the species ate only contaminated prey. For example, ducks were assumed to only eat benthos, bats were assumed to only eat insects of aquatic origin, etc. In fact, many of the species are omnivorous or, in the case of the mink and bat, eat terrestrial as well as aquatic

prey. The reported diet of each species was then compared to the conservative assumption to estimate the quantitative effect of this estimate. Thus, for example, bats were assumed to eat 20 percent non-aquatic prey, while mink were assumed to eat 50 percent non-aquatic prey. The effect of the conservative assumption was estimated as the 1.0 divided by the percent of the diet made up of other types of food. Thus, for example, mink are reported to eat about 50 percent terrestrial prey, so the effect of assuming 100 percent aquatic prey was estimated to be 2.0, or a 200 percent overestimation of likely exposure.

Assumption 3. 100-percent methylmercury in aquatic prey: Mercury concentrations in benthic invertebrates and fish were assumed to be 100-percent methylmercury for risk assessments considering risk from methylmercury. A more likely value for aquatic benthos is about 25 percent (Becker and Bigham 1995). This conservative assumption, therefore, increases the SQ by about 400 percent. On the other hand, the mercury in fish is generally 95 percent or more methylmercury, so this assumption had marginal effect on the risk assessment for piscivores.

Assumption 4. 100-percent absorption of CoPCs from soil and sediment. The food chain analyses assumed that 100-percent of the CoPCs contained in incidentally consumed soils were absorbed. Recorded absorption rates for most metals are much lower. For example, absorption of elemental mercury is essentially zero and absorption of ionic mercury is also very low. Under even the best of conditions—relatively soluble HgCl salts in water—absorption is 7 percent in humans and 15 percent in mice. Absorption of other forms of mercury (i.e., HgS) in other media (e.g., bound to organic matter in soil and sediments) can be expected to be considerably less, especially given the tight binding between mercury and OC. Consult the RI report (Exponent 2000) for more detailed discussion and references. If a more realistic value of 10-percent absorption from soil is assumed, the risk for several of the sentinel species decreases considerably. The effect listed in Table A-26 is the average effect on the SQ_{max} and SQ_{mean} of assuming 10-percent absorption versus the conservative default value of 100 percent.

Assumption 5. Simple Average Sediment Concentrations vs. Area-Weighted Average. The risks to consumers of aquatic benthos were estimated using the arithmetic average of the seven samples taken from the wetland areas of OU1. This simple average is dominated by four very high mercury concentrations from the onsite basin. The average was less affected by three samples from the West Ditch, which had much lower concentrations. The simple average of samples is not a realistic predictor of average exposure because exposure is a function of the relative area represented by each sample. Weighting the samples by relative area of the two systems would produce an average total mercury sediment concentration of about 200 mg/kg[15], considerably lower than the simple average value of 686 mg/kg. Use of this more realistic average reduces risk considerably for the raccoon, and to a lesser extent for the bat, redwing blackbird, and duck.

[15] The open water in the ditch and onsite basin are about the same width, but the West Ditch is about 7 times as long as the basin. The average concentration for the basin was 1,152 mg/kg, whereas that for the ditch was only 65 mg/kg. Weighting these by relative area produces an average concentration of 200 mg/kg compared to almost 700 mg/kg for the simple average of all samples.

The estimated effects of the conservative assumptions are listed in Table A-26 along with the maximum and average SQ based on the default conservative assumptions. Examination of this table allows the reader to determine which conservative assumptions drive the SQ values and perceived risk. This information is critical to understanding the legitimacy of the SQs—i.e., does an SQ greater than 1 really imply a significant potential for impact? This information is also important to determine which site-specific data could be collected to better refine the risk estimates. For example, the assumed percentage of methylmercury in aquatic invertebrates significantly affects the risk to consumers of the benthos. It should also be noted that many, but not all, of the factors are totally or largely independent of each other. Therefore, the total safety factor of two conservative assumptions is sometimes the product of the effect values in the same row. For example, the exposure to redwing blackbirds will be reduced, compared to the conservative default assumptions, by the bird's limited residence at OU1, its omnivorous diet, and the likelihood that much of the mercury in the aquatic insects will be inorganic mercury. The net effect of simultaneously assuming all three conservative assumptions is the product of the three safety factors, i.e., 16-fold overestimation of likely risk.

By comparing the relative magnitude of the SQ values and the estimated effects of the conservative assumptions, one can assess whether impacts on that receptor are likely or more probably an artifact of the conservatism of the risk assessment. Safety factors whose magnitude exceeds the mean SQ are shown in bold typeface in the tables. Relaxation of these conservative assumptions alone would produce mean SQs less than 1.0. For example, Table A-26 shows that the excessive SQs found with the larger receptors (mallards, raccoons, hawks, red-tailed hawk, mink, heron) could be reduced below 1.0 by either more realistic assumptions of percent-residence or by more realistic assumptions about bioavailability of metals in incidentally consumed soil and sediment. Given the low likelihood of either conservative assumption being true, the risks to these species can be dismissed as unlikely.

Table (A-26) suggests that the risks to bats, which can have very large foraging ranges, are also not likely to be significant. On the other hand, impacts on redwing blackbirds are more likely, as it would be necessary to relax most all of the conservative assumptions to obtain a NOAEL SQ of less than 1.0. The risks from mercury to the worm predators—the shrew and the woodcock—are also more difficult to discount.

3.7 Uncertainty Analysis

In addition to the factors discussed above, a number of uncertainties are involved in the assessment of ecological risks. Given the conservative nature of the risk assessment process, most uncertainties were dealt with conservatively, i.e., in a manner that will likely increase perceived risks:

- A major source of uncertainty is the extrapolation of laboratory-derived data to the natural environment and to sentinel species. Many factors that will influence a toxicological response are encountered in the real world, and cannot be predicted in the laboratory. Moreover, while the TRVs selected for the risk assessment were based on most sensitive species, most sentinel

species will likely be less sensitive. On the other hand, the toxicological data for these CoPCs is limited; thus, it is possible that sentinel species may be more sensitive than the most sensitive of the small number of species tested in the laboratory. Lastly, there is significant uncertainty concerning extrapolation across different-sized animals. This risk assessment applied a constant TRV dose across different-sized animals. Since small animals consume more per unit body weight than large animals, use of a constant TRV effectively suggests that smaller species are more vulnerable to the same ambient concentrations of a chemical than are large species. As many of the CoPCs (e.g., the metals) are naturally occurring chemicals, it would seem more likely that species would have become adapted to similar concentrations, as opposed to similar doses.

- There is considerable uncertainty regarding site-specific bioavailability of the CoPCs at the site. Site sediments are very rich in OC, which will greatly reduce the CoPC bioavailability to benthos and consumers of those benthos. Similarly, the sediments are likely to be low in oxygen, conditions that favor formation of sulfides, which form insoluble complexes with many metals. On the other hand, site soils appeared to be fill as opposed to well-developed natural soils with ample OC (although fill can sometimes contain appreciable amounts of carbon). These conditions might enhance bioavailability for chemicals in soil.

- The methods used to predict concentrations of CoPCs in earthworms, aquatic benthos, and small mammals all entail significant uncertainty. In most cases, the predictive regressions were based on soil and sediment concentrations less than those encountered at OU1. Therefore, predictions were based on the outer ends of regression lines where uncertainty is highest. In addition, bioaccumulation of mercury and other CoPCs in all of the biota will depend on many site-specific factors, which are currently not measured.

- The actual site use by ecological receptors is another source of uncertainty. The terrestrial part of the study area is a disturbed area within an industrialized area. Likewise, the onsite basin is an isolated, non-natural feature. Overall exposure of chemicals to terrestrial wildlife may be significantly limited by the limited habitat structure of the site and its environs. The aquatic areas also offer limited habitat value. The West Ditch is tidal. Water column species are necessarily intermittent, and benthic species are limited to those than can withstand the rigors of tidal existence.

- The risk assessment effectively applies impacts on individuals or small numbers of individuals to assessment of ecological risk. In fact, ecological risk pertains to the population and community level. Even severe effects on a small number of individuals may have no effect at all on the local population or community.

3.8 Final Assessment of Risks

Despite a refined screening analysis in which conservative assumptions were relaxed, several compounds, notably mercury, still may pose risk to ecological receptors (Table A-15). For example, SQs for mercury in sediment still exceeded 1,000, even when compared to less conservative ER-M values (Table A-18). Other compounds, notably chromium, lead, zinc, PAHs, and PCBs may also pose risk to aquatic benthos. Mercury concentrations in surface water were also considerably higher than the chronic aquatic life criterion, suggesting potential effects on aquatic life. Under most conservative exposure scenarios, mercury toxicity also poses risk to most of the sentinel species evaluated (Table A-26). Worst-case exposure scenarios sometimes produced SQ values above 10 for the worm predators. For most of the sentinel species considered, however, the perceived risk appears less likely when more realistic exposure scenarios are assumed, with the exception of the worm predators. Even under less conservative scenarios, worm predators still may face potential risks from onsite mercury and lead. In addition, risks from mercury to terrestrial receptors would be higher if the developed portion of the site is allowed to revert to natural lands, without either natural attenuation or active remediation.

Thus, the available data suggest that a number of CoPCs, notably mercury, pose risk to a number of receptors, notably benthic invertebrates, other aquatic life, and worm predators. It is also important to note that many chemicals remain as CoPCs because they lack sufficient toxicological information or because their analytical detection limits exceeded screening benchmarks. These chemicals were not considered in any detail, but they might also contribute to the total risk.

Several factors will mitigate any potential impacts on the ecological receptors. For one, the most significant risks pertain to the benthos, primarily the benthos in the onsite basin. This is a small, non-natural, isolated, potentially semi-aquatic system with limited habitat value. Other potentially significant risks pertain to consumers of soil invertebrates. Here, again, potential impacts will be limited by the site's poor habitat value, which should limit exposure to site-related chemicals. Another mitigating factor is the very high levels of OC found in these sediments, which will reduce bioavailability of most site contaminants.

3.9 Conclusions for the First Three Steps of the ERAGS Process

The potential for risk from several site related chemicals to several ecological receptors cannot be dismissed with currently available information. Of potential risks, risks to aquatic invertebrates, aquatic life, and worm predators appear to be most significant and most likely. The primary contaminant of concern is mercury, although other compounds, notably chromium, lead, and zinc, are also potentially problematic. There were also a number of compounds that remained as CoPCs because they lacked benchmarks or because they were undetected at detection limits above benchmarks. At this juncture, the information cannot be interpreted to suggest that no impacts are possible.

Therefore, an SMDP should occur at the end of Step 3. It is necessary to solicit input from the risk managers concerning the selection of assessment endpoints and need for further analyses. Alternatively, potential risks identified in this ERA can be addressed in the feasibility study through risk management.

3.10 Conclusions for New Jersey BEE Process

According to New Jersey regulations concerning baseline ecological evaluations, further analysis is warranted if all of the following are true:

- Contaminants of ecological concern exist on the site
- An environmentally sensitive natural resource area exists on or immediately adjacent to the site
- Potential contaminant migration pathways to an environmentally sensitive natural resource area exist or an impact to a sensitive area is observed.

All three criteria are satisfied. There are several contaminants of ecological concern on the site, notably mercury, but also a number of metals, PAHs, and PCBs. Two environmentally sensitive areas exist on the site (the wetlands and the forested area). Both of these areas are degraded habitat. The site is also adjacent to a sensitive natural resource area, Berry's Creek and its associated wetlands. The contaminants of concern are already in the onsite sensitive natural resource areas, and there are potential migration pathways for the contaminants of ecological concern to Berry's Creek. According to NJDEP guidance, further analysis is warranted. Alternatively, potential risks can be addressed in the feasibility study through risk management.

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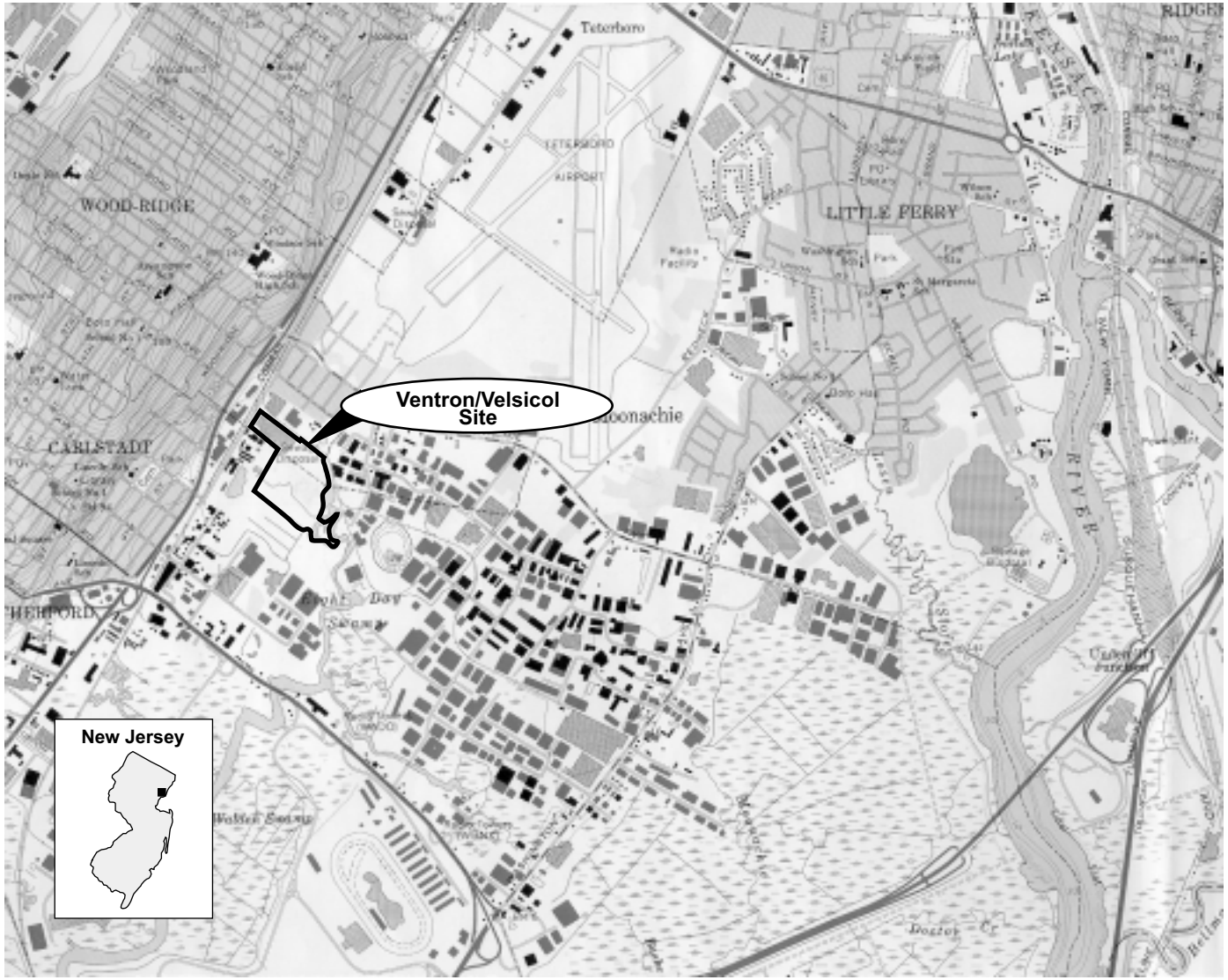
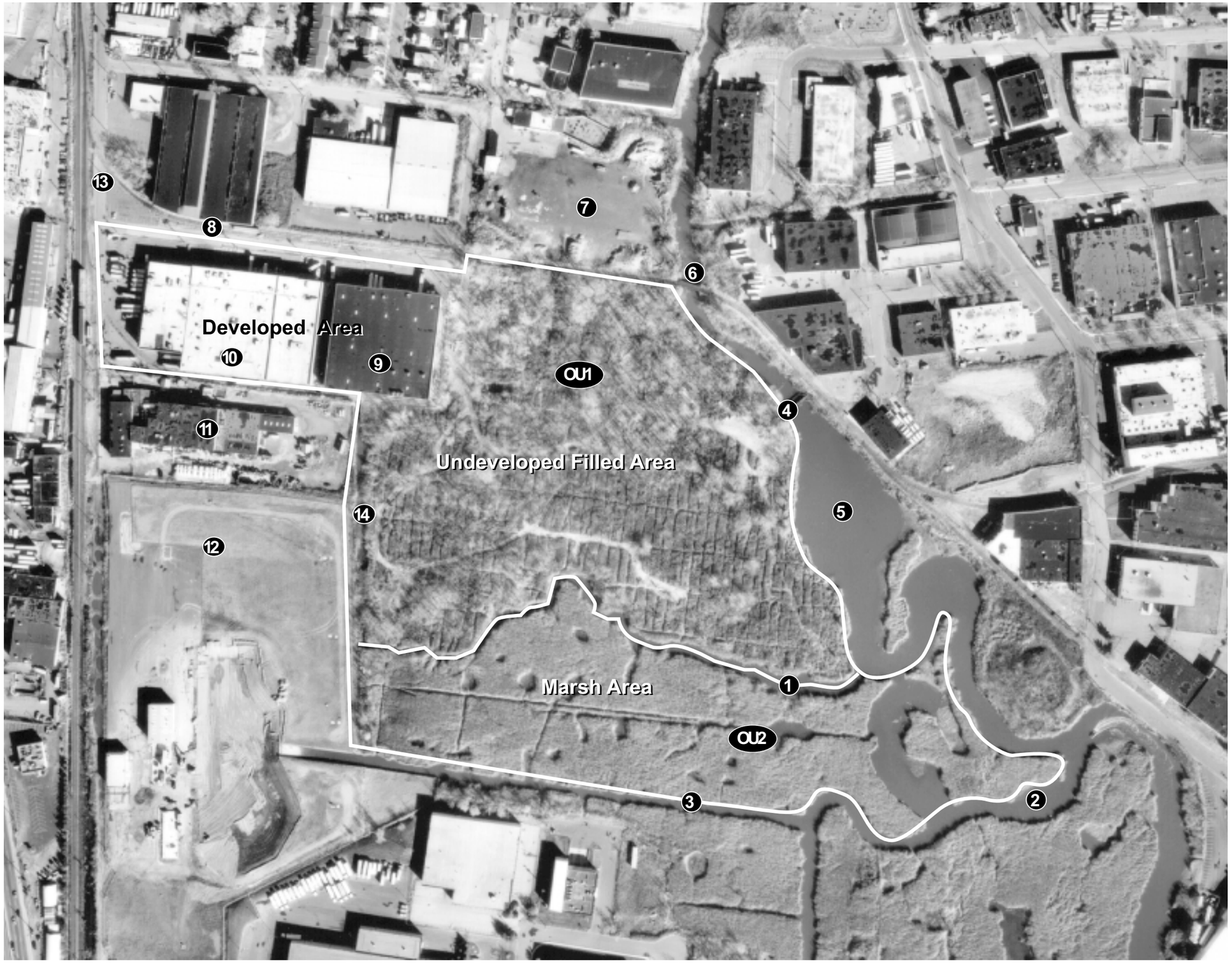
Figures



0 2000 4000 feet

Base map source: USGS Weehawken, NJ-NY

Figure 1. Site location map.



Photograph source: James Stewart, Inc. (November 29, 1997)

LEGEND

1. Diamond Shamrock/Henkel ditch (north)
2. Nevertouch Creek
3. Diamond Shamrock/Henkel ditch (south)
4. Tide gate
5. Berry's Creek
6. Railroad bridge
7. Former POTW
8. Ethel Boulevard
9. Wolf warehouse
10. U.S. Life warehouse
11. Randolph Products
12. Diamond Shamrock/Henkel Property
13. Park Place East
14. West ditch

Site boundary shown as white line

OU1 Operable Unit 1

OU2 Operable Unit 2

0 200 feet
approximate scale

Figure 2. Site layout map.

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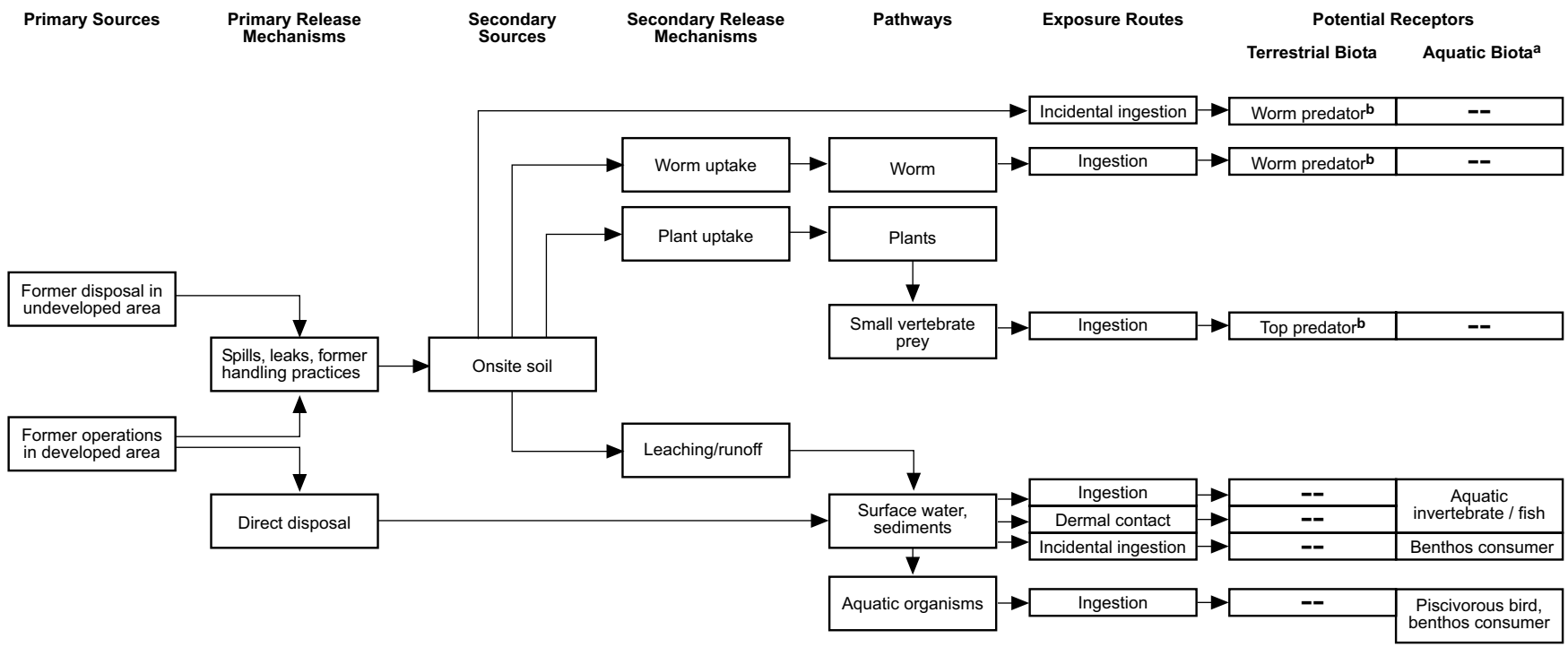

LEGEND

→ Potential transport or exposure pathway

-- Incomplete exposure pathway

a Surface water discussed here is from Diamond Shamrock/Henkel Ditch (north), West Ditch, and the onsite basin. Other surface water, including from Berry's Creek, will be evaluated with OU2.

b This pathway will include exposure via surface water consumption.

Figure 3. Ecological conceptual site model

Appendix A

Screening Tables

Table A-1. Summary of preliminary ecological CoPCs by medium

Analyte	Soils[a]	Sediment	Groundwater	Surface Water
Metals				
Aluminum	X			
Antimony	X			
Arsenic	X	X		
Barium	X		X	X
Cadmium	X	X	X	
Chromium	X	X		
Cobalt			X	
Copper	X	X	X	
Iron	X		X	X
Lead	X	X	X	
Manganese	X		X	X
Mercury (total)	X	X	X	X
Nickel	X	X		
Selenium	X		X	
Silver	X	X		
Thallium	X		X	
Vanadium	X		X	
Zinc	X	X	X	X
Methylmercury				
Methylmercury	X		X	
PCBs				
Aroclor® 1248		X		
Aroclor® 1250		X		
Semivolatile Organic Compounds				
Acenaphthene		X		
Acenaphthylene		X		
Anthracene		X		
Benz[a]anthracene		X		
Benzo[a]pyrene		X		
Benzo[ghi]perylene		X		
Benzo[k]fluoranthene		X		
Bis[2-ethylhexyl]phthalate	X			
Chrysene		X		
Dibenz[a,h]anthracene		X		
Di-*n*-butyl phthalate		X		
Fluoranthene		X		
Naphthalene			X	
Indeno[1,2,3-cd]pyrene		X		
Phenanthrene		X		
Pyrene		X		
Volatile Organic Compounds				
Benzene			X	
Carbon disulfide			X	
Toluene			X	

Note: These chemicals had maximum concentrations that exceeded a screening value.
CoPC - contaminant of potential concern
PCB - polychlorinated biphenyl
[a]CoPCs identified in screening of surface borehole soil are included here.

Table A-2. Screening of constituents detected in surface soil against ecological screening criteria

Constituent	Unit	Screening Criterion Value	Screening Criterion Source	Detection Frequency	Number of Exceedances	Minimum Detected Value	Maximum Detected Value	Arithmetic Mean[a]	Maximum Hazard Quotient[b]	Average Hazard Quotient[c]
Aluminum	mg/kg	50	Efroymson et al. (1997c)	24/24	24	3,580	11,000	6,000	220.0	120.0
Antimony[d]	mg/kg	5	Efroymson et al. (1997a)	18/24	5	1	54	5	10.7	1.1
Arsenic	mg/kg	10	Efroymson et al. (1997a)	20/24	5	4	14	7	1.4	0.7
Barium	mg/kg	283	Efroymson et al. (1997a)	24/24	13	33	608	280	2.1	1.0
Cadmium	mg/kg	4	Efroymson et al. (1997a)	15/24	6	0	21	4	5.3	0.9
Chromium	mg/kg	0.4	Efroymson et al. (1997a)	24/24	24	11	1,150	120	2,875.0	300.0
Copper	mg/kg	60	Efroymson et al. (1997a)	24/24	21	23	1,010	240	16.8	4.0
Iron	mg/kg	200	Efroymson et al. (1997b)	24/24	24	5,530	122,000	27,000	610.0	135.0
Lead	mg/kg	40.5	Efroymson et al. (1997a)	24/24	23	39	4,320	800	106.7	19.8
Manganese	mg/kg	100	Efroymson et al. (1997b)	24/24	21	66	3,090	450	30.9	4.5
Mercury (total)	mg/kg	0.00051	Efroymson et al. (1997a)	24/24	24	1	548	120	1,074,509.8	235,294.1
Methylmercury	ng/g	132	Sample et al. (1996)	9/9	1	1	322	40	2.4	0.3
Nickel	mg/kg	30	Efroymson et al. (1997a)	22/24	15	11	82	36	2.7	1.2
Selenium[d]	mg/kg	0.21	Efroymson et al. (1997a)	9/24	9	1	2	1	9.5	3.9
Silver	mg/kg	2	Efroymson et al. (1997a)	21/24	11	1	94	7	46.9	3.3
Thallium[d]	mg/kg	1	Efroymson et al. (1997a)	2/24	2	15	22	2	21.9	2.2
Vanadium	mg/kg	2	Efroymson et al. (1997a)	24/24	24	10	175	50	87.5	25.0
Zinc[d]	mg/kg	8.5	Efroymson et al. (1997a)	18/24	18	192	25,400	2,665	2,988.2	313.5
Bis[2-ethylhexyl] phthalate	μg/kg	363,000	Sample et al. (1996)	13/24	1	80	380,000	27,244	1.0	0.1

Note: [a] Mean calculated based on detected values and one-half the detection limit for undetected values.
[b] Maximum concentration/screening criterion.
[c] Arithmetic mean concentration/screening criterion.
[d] Screening criterion is less than detection limit.

Table A-3. Screening of constituents detected in surface borehole soils against ecological screening criteria

Constituent	Unit	Screening Criterion Value	Screening Criterion Source	Detection Frequency	Number of Exceedances	Minimum Detected Value	Maximum Detected Value	Arithmetic Mean[a]	Maximum Hazard Quotient[b]	Average Hazard Quotient[c]
Arsenic	mg/kg	9.9	Efroymson et al. (1997a)	2/2	1	8.4	26.4	17.4	2.7	1.8
Barium	mg/kg	283	Efroymson et al. (1997a)	2/2	1	270	304	287	1.1	1
Cadmium	mg/kg	4	Efroymson et al. (1997a)	2/2	1	3.4	4.74	4.07	1.2	1
Chromium	mg/kg	0.4	Efroymson et al. (1997a)	2/2	2	34.4	131	82.7	328	207
Copper	mg/kg	60	Efroymson et al. (1997b)	2/2	2	220	7,420	3,820	124	63.7
Iron	mg/kg	200	Efroymson et al. (1997b)	2/2	2	23,900	35,600	29,800	178	149
Lead	mg/kg	40.5	Efroymson et al. (1997a)	2/2	2	307	393	350	9.7	8.6
Manganese	mg/kg	100	Efroymson et al. (1997b)	2/2	2	241	262	252	2.6	2.5
Mercury (total)	mg/kg	0.00051	Efroymson et al. (1997a)	2/2	2	24	2,250	1,140	4.41×10^{6}	2.23×10^{6}
Nickel	mg/kg	30	Efroymson et al. (1997a)	2/2	1	21.8	87.8	54.8	2.9	1.8
Selenium	mg/kg	0.2	Efroymson et al. (1997a)	2/2	2	1.1	1.5	1.3	7.1	6.2
Silver	mg/kg	2	Efroymson et al. (1997a)	2/2	1	1.9	9.6	5.75	4.8	2.9
Thallium	mg/kg	1	Efroymson et al. (1997a)	1/2	1	5.4	5.4	2.77	5.4	2.8
Zinc	mg/kg	8.5	Efroymson et al. (1997a)	2/2	2	512	2,110	1,310	248	154

Note: [a] Mean calculated based on detected values and one-half the detection limit for undetected values.
[b] Maximum concentration/screening criterion.
[c] Arithmetic mean concentration/screening criterion.

Table A-4. Screening of constituents detected in sediment against ecological screening criteria

Constituent	Unit	Screening Criterion Value	Screening Criterion Source	Detection Frequency	Number of Exceedances	Minimum Detected Value	Maximum Detected Value	Arithmetic Mean[a]	Maximum Hazard Quotient[b]	Average Hazard Quotient[c]
Arsenic	mg/kg	6	NJDEP (1998)	5/5	1	2.6	8.8	5	1.5	0.8
Cadmium	mg/kg	0.6	NJDEP (1998)	5/5	5	0.9	9.1	3.7	15.2	6.2
Chromium	mg/kg	26	NJDEP (1998)	5/5	5	55.4	156	103	6	4
Copper	mg/kg	16	NJDEP (1998)	5/5	5	94	194	143	12.1	8.9
Lead	mg/kg	31	NJDEP (1998)	5/5	5	188	469	279	15.1	9
Mercury (total)	μg/g	0.2	NJDEP (1998)	7/7	7	18.95	1,290	690	6,450	3,450
Nickel	mg/kg	16	NJDEP (1998)	5/5	4	14.2	29.2	24.7	1.8	1.5
Silver[d]	mg/kg	1	NJDEP (1998)	3/5	3	1.1	4.3	1.6	4.3	1.6
Zinc	mg/kg	120	NJDEP (1998)	5/5	5	434	3,540	1,430	29.5	11.9
Aroclor® 1248	μg/kg	30	NJDEP (1998)	2/2	2	190	240	210	8	7
Aroclor® 1260	μg/kg	5	NJDEP (1998)	2/2	2	260	490	370	98	74
Acenaphthene[d]	μg/kg	16	NJDEP (1998)	1/5	1	100	100	440	6.3	27.5
Acenaphthylene[d]	μg/kg	44	NJDEP (1998)	2/5	2	180	270	360	6.1	8.2
Anthracene	μg/kg	220	NJDEP (1998)	5/5	4	170	350	280	1.6	1.3
Benz[a]anthracene	μg/kg	320	NJDEP (1998)	5/5	3	230	1,700	700	5.3	2.2
Benzo[a]pyrene	μg/kg	370	NJDEP (1998)	5/5	3	300	1,600	700	4.3	1.9
Benzo[ghi]perylene	μg/kg	170	NJDEP (1998)	5/5	5	270	1,200	600	7.1	3.5
Benzo[k]fluoranthene	μg/kg	240	NJDEP (1998)	5/5	5	280	660	430	2.8	1.8
Chrysene	μg/kg	340	NJDEP (1998)	5/5	4	330	1,600	800	4.7	2.4
Dibenz[a,h]anthracene[d]	μg/kg	60	NJDEP (1998)	5/5	5	91	320	200	5.3	3.3
Di-*n*-butyl phthalate[d]	μg/kg	110	NOAA (1999)	1/2	1	160	160	160	1.5	1.5
Fluoranthene	μg/kg	750	NJDEP (1998)	5/5	3	510	2,800	1,400	3.7	1.9
Indeno[1,2,3-cd]pyrene	μg/kg	200	NJDEP (1998)	5/5	5	220	1,200	500	6	2.5
Phenanthrene	μg/kg	560	NJDEP (1998)	5/5	2	180	1,800	800	3.2	1.4
Pyrene	μg/kg	490	NJDEP (1998)	5/5	4	380	2,900	1,200	5.9	2.4

Note: [a] Mean calculated based on detected values and one-half the detection limit for undetected values.
[b] Maximum concentration/screening criterion.
[c] Arithmetic mean concentration/screening criterion.
[d] Screening criterion is less than detection limit.

Table A-5. Screening of constituents detected in groundwater against ecological screening criteria

Constituent	Unit	Screening Criterion Value	Screening Criterion Source	Detection Frequency	Number of Exceedances	Minimum Detected Value	Maximum Detected Value	Arithmetic Mean[a]	Maximum Hazard Quotient[b]	Average Hazard Quotient[c]
Barium	μg/L	3.9	Suter and Tsao (1996)	27/27	27	22.7	934	320	240	82.1
Cadmium	μg/L	1	NJDEP (1997)	13/27	11	0.89	5.7	1	5.7	1
Cobalt	μg/L	3	Suter and Tsao (1996)	10/12	1	0.94	3.9	1.6	1.3	0.5
Copper[d]	μg/L	11	NJDEP (1997)	10/27	3	1.7	356	18	32.4	1.6
Iron	μg/L	1,000	U.S. EPA (1999)	25/27	22	152	37,500	12,000	37.5	12
Lead[d]	μg/L	2.5	NJDEP (1997)	9/27	6	0.8	13.9	2.3	5.6	0.9
Manganese	μg/L	80	Suter and Tsao (1996)	27/27	24	7.2	6580	1350	82.3	16.9
Mercury (filtered)	ng/L	770	U.S. EPA (1999)	3/3	3	924	8,470	4,030	11	5.2
Mercury (unfiltered)	ng/L	770	U.S. EPA (1999)	23/30	10	10.8	54,200	5,000	70.4	6.5
Methylmercury (unfiltered)	ng/L	3	Suter and Tsao (1996)	27/27	14	0.12	32.7	6.7	10.9	2.2
Selenium	μg/L	5	NJDEP (1997)	6/27	4	2.34	13.4	2.8	2.7	0.6
Thallium[d]	μg/L	9	Efroymson et al. (1997a)	4/27	1	4.9	13.5	3.2	1.5	0.4
Vanadium	μg/L	19	Suter and Tsao (1996)	10/12	1	2.3	50.7	8.5	2.7	0.4
Zinc	μg/L	100	NJDEP (1997)	19/27	6	9.5	803	132	8	1.3
Naphthalene	μg/L	24	Suter and Tsao (1996)	2/13	1	9	100	10	4.2	0.4
Benzene	μg/L	46	Suter and Tsao (1996)	8/27	1	1.2	140	9	3	0.2
Carbon disulfide[d]	μg/L	0.9	Efroymson et al. (1997a)	1/12	1	16	16	6	17.4	6.5
Toluene	μg/L	130	Suter and Tsao (1996)	2/27	2	330	1,700	80	13.1	0.6

Note: [a] Mean calculated based on detected values and one-half the detection limit for undetected values.
[b] Maximum concentration/screening criterion.
[c] Arithmetic mean concentration/screening criterion.
[d] Screening criterion is less than detection limit.

Table A-6. Screening of constituents detected in surface water against ecological screening criteria

Constituent	Unit	Screening Criterion Value	Screening Criterion Source	Detection Frequency	Number of Exceedances	Minimum Detected Value	Maximum Detected Value	Arithmetic Mean[a]	Maximum Screening Quotient[b]	Average Screening Quotient[c]
Barium (filtered)	µg/L	3.9	Suter and Tsao (1996)	2/2	2	189	190	190	48.7	48.7
Barium (unfiltered)	µg/L	3.9	Suter and Tsao (1996)	5/5	5	40	189	107	48.5	27.4
Cadmium (unfiltered)	µg/L	3.7	NJDEP (1997)	1/5	0	1.2	1.2	0.4	0.3	0.1
Iron (unfiltered)	µg/L	1,000	U.S. EPA (1999)	5/5	3	653	2,620	1,740	2.6	1.7
Lead (filtered)	µg/L	21.6	NJDEP (1997)	1/2	0	4.7	4.7	2.6	0.2	0.1
Lead (unfiltered)	µg/L	21.6	NJDEP (1997)	5/5	0	2	19	7	0.9	0.3
Manganese (filtered)	µg/L	80	Suter and Tsao (1996)	2/2	2	351	373	362	4.7	4.5
Manganese (unfiltered)	µg/L	80	Suter and Tsao (1996)	5/5	5	141	413	287	5.2	3.6
Mercury (total) (unfiltered)	ng/L	770	U.S. EPA (1999)	5/5	2	402	17,600	5,000	22.9	6.5
Zinc (unfiltered)	µg/L	379	NJDEP (1997)	4/5	1	35.9	403	151	1.1	0.4

Note: [a] Mean calculated based on detected values and one-half the detection limit for undetected values.
[b] Maximum concentration/screening criterion.
[c] Arithmetic mean concentration/screening criterion.

Table A-7. Monitoring well borehole data for MW-1 through MW-12 as reported by NJDEP (1993) compared against ecological soil screening criteria

	Screening Criterion		MW-1	MW-2	MW-3	MW-4	MW-5	MW-6	MW-7	MW-9	MW-10	MW-12
Analyte	Value	Source	0–2 ft	0–2 ft	0–2 ft	0–2 ft	0–2 ft	0–2 ft	0–2 ft	0–2 ft	0.5–2.5 ft	0–2 ft
Volatile Organic Compounds												
Benzene	523	Sample et al. (1996)										
2-Butanone (methyl ethyl ketone)	16,744	Sample et al. (1996)										
Ethylbenzene												
Toluene	200	Efroymson et al. (1997a)										
Vinyl chloride	6	Sample et al. (1996)										
Semivolatile Organic Compounds												
Acenaphthene	20	Efroymson et al. (1997a)							0.045			
Acenaphthylene									0.079			
Anthracene									0.17			
Benz[a]anthracene												
Benzo[a]pyrene	20	Sample et al. (1996)							0.33			
Benzo[b]fluoranthene									0.61			
Benzo[k]fluoranthene												
Benzoic acid									0.11			
Bis[2-ethylhexyl]phthalate	363	Sample et al. (1996)							0.48			
Butylbenzyl phthalate												
Chrysene									0.57			
Dibenzofuran									0.037			
Diethyl phthalate	100	Efroymson et al. (1997a)										
Di-*n*-butyl phthalate	200	Efroymson et al. (1997a)										
Di-*n*-octyl phthalate												
Fluoranthene									0.9			
Fluorene	30	Efroymson et al. (1997b)							0.1			
2-Methylnaphthalene									0.036			
Naphthalene												
Phenanthrene									0.81			
Phenol	30	Efroymson et al. (1997a)										
Pyrene									0.66			

Table A-7. (cont.)

Analyte	Screening Criterion Value	Screening Criterion Source	MW-1 0–2 ft	MW-2 0–2 ft	MW-3 0–2 ft	MW-4 0–2 ft	MW-5 0–2 ft	MW-6 0–2 ft	MW-7 0–2 ft	MW-9 0–2 ft	MW-10 0.5–2.5 ft	MW-12 0–2 ft
Metals												
Arsenic	9.9	Efroymson et al. (1997a)										
Chromium	0.4	Efroymson et al. (1997a)							136			
Copper	60	Efroymson et al. (1997a)							332			
Mercury (total)	0.00051	Efroymson et al. (1997a)	21.6	588	98.3	1.6	50.7	395	352	149	1,820	4.1
Nickel	30	Efroymson et al. (1997a)							193			
Silver	2	Efroymson et al. (1997a)							30			
Thallium	1	Efroymson et al. (1997a)							10			
Vanadium	2	Efroymson et al. (1997a)							245			
Zinc	8.5	Efroymson et al. (1997a)							10,600			
Pesticides/PCBs												
alpha-Chlordane												
gamma-Chlordane												
4,4'-DDT												
Aroclor® 1242												
Aroclor® 1248	0.37								4.4			
Aroclor® 1254												
Aroclor® 1260												

Note: Units in mg/kg
Boxed values exceed the screening criterion
No summary statistics were calculated because the data set contains only data above the detection limits
NJDEP - New Jersey Department of Environmental Protection
PCB - polychlorinated biphenyl

Table A-8. Monitoring well groundwater data for MW-1 through MW-12 as reported by NJDEP (1993) compared against ecological screening criteria

	Screening Criterion[a]													
Analyte	Value	Source	MW-1	MW-2	MW-3	MW-4	MW-5	MW-6	MW-7	MW-8	MW-9	MW-10	MW-11	MW-12
VOCs														
Chlorobenzene	130	Suter and Tsao (1996)				4	4							
Chloroethane									37					
1,2-Dichloroethane	910	Efroymson et al. (1997a)								84	58			
Toluene	130	Suter and Tsao (1996)		13	8	2								
Xylene isomers (total)														
SVOCs														
Acenaphthene	23	Suter and Tsao (1996)	2	1										
Bis[2-ethylhexyl]phthalate														
1,4-Dichlorobenzene	15	Suter and Tsao (1996)					3							
Diethyl phthalate	220	Suter and Tsao (1996)					2							
Isophorone				9										
2-Methylnaphthalene				23			1				17			
4-Methylphenol						13	2							
Naphthalene	24	Suter and Tsao (1996)		140			2				5			
N-nitrosodiphenylamine	210	Efroymson et al. (1997a)	3											
Phenanthrene	6	Suter and Tsao (1996)									2			
Metals (filtered)														
Antimony	30	NOAA (1999)								12				
Arsenic	190	NJDEP (1997)			2		4	4		6				3
Barium	3.9	Suter and Tsao (1996)	380	950	98	379	576	143	150	127	264	199	363	128
Beryllium	0.66	Efroymson et al. (1997a)												
Cadmium	1	NJDEP (1997)	2											
Calcium			300,000	220,000	559,000	237,000	368,000	175,000	68,500	72,400	95,300	73,800	88,400	303,000
Chromium	10	NJDEP (1997)												
Cobalt	3	Suter and Tsao (1996)												
Copper	11	NJDEP (1997)												
Iron	1,000	U.S. EPA (1999)	18,000	33,000	17,400	27,800	17,100	1,420	5,540	12,500	23,800	106	6,380	6,700
Lead	2.5	NJDEP (1997)						2						
Magnesium			38,600	66,300	49,400	42,000	47,200	48,000	22,300	30,900	35,300	10,100	10,400	11,700
Manganese	80	Suter and Tsao (1996)	850	853		655	843	1,370	444	4,040	7,930	201		429
Mercury (total)	0.77	U.S. EPA (1999)			19				0.44			0.32		
Nickel	160	NJDEP (1997)		7				22						
Potassium			12,200	55,000	10,400	17,000	20,800	16,600	12,700	2,020	3,350	1,130	773	2,910
Selenium	5	NJDEP (1997)	20	20		20		20	20	30				

Table A-8. (cont.)

Analyte	Screening Criterion[a] Value	Screening Criterion[a] Source	MW-1	MW-2	MW-3	MW-4	MW-5	MW-6	MW-7	MW-8	MW-9	MW-10	MW-11	MW-12
Metals (filtered) (cont.)														
Silver	0.12	NOAA (1999)												
Sodium			58,900	209,000	66,900	44,200	6	221,000	45,000	311,000	128,000	17,200	26,300	69,200
Thallium	9	Efroymson et al. (1997a)												
Vanadium	19	Suter and Tsao (1996)												9
Zinc	100	NJDEP (1997)					6	116	254	9	10	4	8	
Metals (unfiltered)														
Aluminum	87	U.S. EPA (1999)	1,910	1,010	4,420	1,560	1,520	8,620	41,200	3,700	2,570	1,770	1,670	5,120
Antimony	30	NOAA (1999)							54					14
Arsenic	190	NJDEP (1997)	2	5	6	2	5	23	6	7	4		2	3
Barium	3.9	Suter and Tsao (1996)	485	1,100	167	449	636	391	550	176	303	237	410	204
Beryllium	0.66	Efroymson et al. (1997a)						1	4					
Cadmium	1	NJDEP (1997)	6		3			2	7					
Calcium			297,000	230,000	554,000	250,000	370,000	182,000	89,400	75,300	96,500	74,800	92,900	294,000
Chromium	10	NJDEP (1997)	36		28	11		128	235	7	9	7	7	11
Cobalt	3	Suter and Tsao (1996)			4	4		14	43	4	6			5
Copper	11	NJDEP (1997)	43	12	42		11	163	312	21	45	9	14	18
Iron	1,000	U.S. EPA (1999)	28,400	42,700	28,600	34,700	22,800	29,700	114,000	25,400	31,700	3,380	12,700	19,500
Lead	2.5	NJDEP (1997)												
Magnesium			41,500	6,700	49,900	43,800	48,700	53,200	38,400	32,900	36,400	10,700	11,400	13,200
Manganese	80	Suter and Tsao (1996)	983	842	2,020	758	914	1,830	2,060		8,100	244	426	483
Mercury (total)	0.77	U.S. EPA (1999)	14	12	24	2	30	28	4,110	15	29	28	3	4
Nickel	160	NJDEP (1997)	13	11	9	10	9	58	163	11	9			11
Potassium			13,700	56,400	10,500	17,500	21,200	17,500	16,000	2,370	3,600	1,360	1,020	3,260
Selenium	5	NJDEP (1997)												
Silver	0.12	NOAA (1999)							31					
Sodium			66,100	211,000	65,500	45,300	66,900	236,000	46,900	317,000	127,000	16,900	26,100	71,300
Thallium	9	Efroymson et al. (1997a)	3											
Vanadium	19	Suter and Tsao (1996)							633	10		6	4	33
Zinc	100	NJDEP (1997)	225	123	292	112	74	586	11,300	54	56	18	30	68
Cyanide	5.2	NJDEP (1997)	40		71	15	24		40					12

Table A-8. (cont.)

Analyte	Screening Criterion[a] Value	Screening Criterion[a] Source	MW-1	MW-2	MW-3	MW-4	MW-5	MW-6	MW-7	MW-8	MW-9	MW-10	MW-11	MW-12
Pesticides/PCBs														
4,4'-DDT	0.0010	NJDEP (1997)	0.34	0.44	0.07		0.32		0.25					
Endosulfan	0.0560	NJDEP (1997)							0.28					
Endrin	0.0023	NJDEP (1997)									0.04			
Heptachlor	0.0038	NJDEP (1997)	0.38											
beta-BHC	5,000	Suter and Tsao (1996)												0.04
gamma-BHC (Lindane)	0.080	NJDEP (1997)				0.22								

Note: Units in μg/L
Boxed values exceed the screening criterion
No summary statistics were calculated because the data set contains only data above the detection limits
NJDEP - New Jersey Department of Environmental Protection
PCB - polychlorinated biphenyl
SVOC - semivolatile organic compound
VOC - volatile organic compound
[a] Based on dissolved value.

Table A-9. Summary of detected constituents with no ecological screening criteria

Analyte	Soil	Sediment	Groundwater	Surface Water
Metals				
Aluminum		X		
Antimony		X		
Barium		X		
Beryllium		X		
Calcium	X		X	X
Cobalt		X		
Iron		X		
Magnesium	X	X	X	X
Potassium	X	X	X	X
Selenium		X		
Sodium	X	X	X	X
Thallium		X		
Vanadium		X		
Methylmercury				
Methylmercury		X		
Semivolatile Organic Compounds				
1,2-Dichlorobenzene	X	X		
1,2-Dichloroethane		X		
1,2-Dichloropropane		X		X
1,3-Dichlorobenzene	X	X		
1,4-Dichlorobenzene		X		
1,2,4-Trichlorobenzene		X		
2-Chloronaphthalene	X	X	X	X
2-Chlorophenol	X	X	X	X
2-Methylnaphthalene	X		X	X
2-Methylphenol	X	X		
2-Nitroaniline	X	X	X	X
2-Nitrophenol	X	X	X	X
2,2'-Oxybis[1-chloropropane]	X	X	X	X
2,4-Dichlorophenol	X	X	X	X
2,4-Dimethylphenol	X	X	X	X
2,4-Dinitrophenol		X	X	X
2,4-Dinitrotoluene	X	X	X	X
2,6-Dinitrotoluene	X	X	X	X
2,4,5-Trichlorophenol		X		
2,4,6-Trichlorophenol		X	X	X
3-Nitroaniline	X	X	X	X
3,3'-Dichlorobenzidine	X	X	X	X
4-Bromophenyl-phenyl ether	X	X		
4-Chloroaniline	X	X	X	X
4-Chlorophenyl-phenyl ether	X		X	X
4-Chloro-3-methylphenol			X	X
4-Methylphenol		X	X	X
4-Nitroaniline	X	X	X	X

Table A-9. (cont.)

Analyte	Soil	Sediment	Groundwater	Surface Water
Semivolatile Organic Compounds (cont.)				
4-Nitrophenol		X		
4,6-Dinitro-2-methylphenol	X	X	X	X
Acenaphthylene	X		X	X
Anthracene	X			
Benz[a]anthracene	X			
Benzo[b]fluoranthene	X	X	X	X
Benzo[ghi]perylene	X		X	X
Benzo[k]fluoranthene	X		X	X
Bis[2-chloroethoxy]methane	X	X	X	X
Bis[2-chloroethyl]ether	X	X	X	X
Bis[2-ethylhexyl]phthalate		X		
Butylbenzyl phthalate	X	X		
Carbazole	X	X	X	X
Chrysene	X		X	X
Dibenz[a,h]anthracene	X		X	X
Dibenzofuran	X			
Diethyl phthalate		X		
Dimethyl phthalate		X	X	X
Di-*n*-octyl phthalate	X	X	X	X
Fluoranthene	X			
Hexachlorobutadiene	X	X	X	X
Hexachlorocyclopentadiene		X	X	X
Hexachloroethane	X	X		
Indeno[1,2,3-cd]pyrene	X		X	X
Isophorone	X	X	X	X
Naphthalene	X			
Nitrobenzene		X	X	X
N-nitroso-di-*n*-propylamine	X	X	X	X
N-nitrosodiphenylamine		X		
Pentachlorophenol		X		
Phenanthrene	X			
Pyrene	X		X	X
Volatile Organic Compounds				
1,1-Dichloroethane	X	X		
1,1-Dichloroethene	X	X		
1,2-Dichloroethene isomers (total)	X	X		
1,2-Dichloropropane			X	
1,1,1-Trichloroethane	X	X		
1,1,2,2-Tetrachloroethane	X	X		
1,1,2-Trichloroethane	X	X		
2-Butanone (methyl ethyl ketone)		X		
2-Hexanone	X	X		
4-Methyl-2-pentanone (MIBK)	X	X		
Acetone		X		
Bromodichloromethane	X	X	X	X
Bromoform	X	X	X	X
Bromomethane	X	X	X	X
Carbon disulfide	X	X		
Carbon tetrachloride		X		

Table A-9. (cont.)

Analyte	Soil	Sediment	Groundwater	Surface Water
Volatile Organic Compounds (cont.)				
Chlorobenzene		X		
Chloroethane	X	X	X	X
Chloroform		X		
Chloromethane	X	X	X	X
cis -1,3-Dichloropropene	X	X	X	X
Dibromochloromethane	X	X	X	X
Ethylbenzene	X			
Methylene chloride (dichloromethane)	X	X		
Styrene		X	X	X
Tetrachloroethene	X			
trans -1,3-Dichloropropene	X	X	X	X
Trichloroethene	X			
Vinyl chloride		X		
Xylene isomers[a]	X		X	X

Note: [a]Also detected in surface borehole soil.

Table A-10. Summary of constituents with ecological screening criteria less than detection limits

	Soil	Sediment	Groundwater	Surface Water
Metals				
Aluminum			X	X
Antimony	X			
Beryllium			X	
Copper			X	X
Lead			X	X
Selenium	X			
Silver		X	X	X
Thallium	X		X	
Zinc	X			
PCBs				
Aroclor® 1016		X		
Aroclor® 1254		X		
Semivolatile Organic Compounds				
1,2-Dichlorobenzene			X	
1,3-Dichlorobenzene			X	
1,4-Dichlorobenzene			X	
1,2,4-Trichlorobenzene			X	
2-Methylphenol			X	
4-Bromophenyl-phenyl ether			X	X
4-Nitrophenol	X		X	
2-Methylnaphthalene		X		
2,4,5-Trichlorophenol	X		X	
2,4,6-Trichlorophenol	X			
Acenaphthene		X	X	
Acenaphthylene		X		
Anthracene			X	X
Benz[a]anthracene			X	X
Benzo[a]pyrene			X	X
Bis[2-ethylhexyl]phthalate			X	
Butylbenzyl phthalate			X	
Dibenz[a,h]anthracene		X		
Dibenzofuran		X	X	
Diethyl phthalate			X	
Di-*n*-butyl phthalate		X	X	
Fluoranthene			X	X
Fluorene		X	X	X
Hexachlorobenzene		X	X	X
Hexachloroethane			X	
N-nitrosodiphenylamine			X	
Naphthalene		X		
Pentachlorophenol	X		X	X
Phenanthrene			X	X
Phenol		X	X	
Volatile Organic Compounds				
Carbon disulfide			X	X
Carbon tetrachloride			X	X

Note: PCB - polychlorinated biphenyl

Table A-11. Refined screening of constituents detected in surface soil—consideration of background, average screening quotient, and frequency of exceedance

Constituent	Screening Value	Units	Number of Analyses	Frequency of Detects	Frequency of Exceed-ance[a]	Minimum Detected Value	Maximum Detected Value	Arithmetic Mean Value
Aluminum	50	mg/kg	24	1.00	1.00	3,580	11,000	6,000
Antimony	5	mg/kg	24	0.75	0.21	1	54	5
Arsenic	10	mg/kg	24	0.83	0.21	4	14	
Barium	283	mg/kg	24	1.00	0.54	33	608	280
Cadmium	4	mg/kg	24	0.63	0.25	0	21	4
Chromium	0.4	mg/kg	24	1.00	1.00	11	1,150	120
Copper	60	mg/kg	24	1.00	0.88	23	1,010	240
Iron	200	mg/kg	24	1.00	1.00	5,530	122,000	27,000
Lead	40.5	mg/kg	24	1.00	0.96	39	4,320	800
Manganese	100	mg/kg	24	1.00	0.88	66	3,090	450
Mercury (total)	0.00051	mg/kg	24	1.00	1.00	1	548	120
Methylmercury (dry)	132	ng/g	9	1.00	0.11	1	322	40
Nickel	30	mg/kg	24	0.92	0.63	11	82	36
Selenium	0.21	mg/kg	24	0.38	0.38	1	2	1
Silver	2	mg/kg	24	0.88	0.46	1	94	7
Thallium	1	mg/kg	24	0.08	0.08	15	22	2
Vanadium	2	mg/kg	24	1.00	1.00	10	175	50
Zinc	8.5	mg/kg	24	0.75	0.75	192	25,400	2,665
bis[2-Ethylhexyl]phthalate	363,000	μg/kg	24	0.54	0.04	80	380,000	27,244

Table A-11. (cont.)

Constituent	Screening Value	Units	Maximum Screening Quotient	Average Screening Quotient	Back-ground Geo Means	Source	Basis for Elimi-nation
Aluminum	50	mg/kg	220	120	66,000	Shacklette et al.	B
Antimony	5	mg/kg	10.7	1.1			
Arsenic	10	mg/kg	1.4	0.0	5.49	NJDEP 1993b	A
Barium	283	mg/kg	2.1	1.0	554	Shacklette et al. 1971	A
Cadmium	4	mg/kg	5.3	0.9	0.5	NJDEP 1993b	A
Chromium	0.4	mg/kg	2,875	300	11.2	NJDEP 1993b	
Copper	60	mg/kg	16.8	4.0	32.8	NJDEP 1993b	
Iron	200	mg/kg	610	135	25,000	Shacklette et al. 1971	
Lead	40.5	mg/kg	106.7	19.8	113	NJDEP 1993b	
Manganese	100	mg/kg	30.9	5	283	NJDEP 1993b	
Mercury (total)	0.00051	mg/kg	1,074,509.8	235,294.1	0.2	NJDEP 1993b	
Methylmercury (dry)	132	ng/g	2.4	0.3			A
Nickel	30	mg/kg	2.7	1.2	14.1	NJDEP 1993b	
Selenium	0.21	mg/kg	9.5	3.9			
Silver	2	mg/kg	46.9	3.3	0.16	NJDEP 1993b	
Thallium	1	mg/kg	21.9	2.2	0.07	NJDEP 1993b	
Vanadium	2	mg/kg	87.5	25	76	Shacklette et al. 1971	
Zinc	8.5	mg/kg	2,988.2	313.5	116	NJDEP 1993b	
bis[2-Ethylhexyl]phthalate	363,000	μg/kg	1.05	0.1			A,C

Note: A - average screening quotient < 1
B - maximum did not exceed twice the background concentration
C - frequency of exceedance < 5%
[a]Number of exceedances/number of analyses

Table A-12. Refined screening of constituents detected in sediment—consideration of average screening quotient and frequency of exceedance

Constituent	Screening Value	Units	Number of Analyses	Number of Detections	Frequency of Detects	Number of Exceedances	Frequency of Exceedance[a]
Arsenic	6	mg/kg	5	5	1	1	0.2
Cadmium	0.6	mg/kg	5	5	1	5	1
Chromium	26	mg/kg	5	5	1	5	1
Copper	16	mg/kg	5	5	1	5	1
Lead	31	mg/kg	5	5	1	5	1
Mercury (total)	0.2	μg/kg	7	7	1	7	1
Nickel	16	mg/kg	5	5	1	4	0.8
Silver	1	mg/kg	5	3	0.6	3	0.6
Zinc	120	mg/kg	5	5	1	5	1
Aroclor® 1248	30	μg/kg	2	2	1	2	1
Aroclor® 1260	5	μg/kg	2	2	1	2	1
Acenaphthene	16	μg/kg	5	1	0.2	1	0.2
Acenaphthylene	44	μg/kg	5	2	0.4	2	0.4
Anthracene	220	μg/kg	5	5	1	4	0.8
Benz[a]anthracene	320	μg/kg	5	5	1	3	0.6
Benzo[a]pyrene	370	μg/kg	5	5	1	3	0.6
Benzo[ghi]perylene	170	μg/kg	5	5	1	5	1
Benzo[k]fluoranthene	240	μg/kg	5	5	1	5	1
Chrysene	340	μg/kg	5	5	1	4	0.8
Dibenz[a,h]anthracene	60	μg/kg	5	5	1	5	1
Di-*n*-butyl phthalate	110	μg/kg	2	1	0.5	1	0.5
Fluoranthene	750	μg/kg	5	5	1	3	0.6
Indeno[1,2,3-cd]pyrene	200	μg/kg	5	5	1	5	1
Phenanthrene	560	μg/kg	5	5	1	2	0.4
Pyrene	490	μg/kg	5	5	1	4	0.8

Table A-12. (cont.)

Constituent	Screening Value	Units	Minimum Detected Value	Maximum Detected Value	Arithmetic Mean Value	Maximum Screening Quotient	Mean Screening Quotient	Basis for Elimination
Arsenic	6	mg/kg	2.6	8.8	5	1.5	0.8	A
Cadmium	0.6	mg/kg	0.9	9.1	3.7	15.2	6.2	
Chromium	26	mg/kg	55.4	156	103	6	4	
Copper	16	mg/kg	94	194	143	12.1	8.9	
Lead	31	mg/kg	188	469	279	15.1	9	
Mercury (total)	0.2	μg/kg	19	1,290	686	6,450	3,430.0	
Nickel	16	mg/kg	14.2	29.2	24.7	1.8	1.5	
Silver	1	mg/kg	1.1	4.3	1.6	4.3	1.6	
Zinc	120	mg/kg	434	3,540	1,430	29.5	11.9	
Aroclor® 1248	30	μg/kg	190	240	210	8	7	
Aroclor® 1260	5	μg/kg	260	490	370	98	74	
Acenaphthene	16	μg/kg	100	100	440	6.3	27.5	
Acenaphthylene	44	μg/kg	180	270	360	6.1	8.2	
Anthracene	220	μg/kg	170	350	280	1.6	1.3	
Benz[a]anthracene	320	μg/kg	230	1,700	700	5.3	2.2	
Benzo[a]pyrene	370	μg/kg	300	1,600	700	4.3	1.9	
Benzo[ghi]perylene	170	μg/kg	270	1,200	600	7.1	3.5	
Benzo[k]fluoranthene	240	μg/kg	280	660	430	2.8	1.8	
Chrysene	340	μg/kg	330	1,600	800	4.7	2.4	
Dibenz[a,h]anthracene	60	μg/kg	91	320	200	5.3	3.3	
Di-*n*-butyl phthalate	110	μg/kg	160	160	160	1.5	1.5	
Fluoranthene	750	μg/kg	510	2,800	1,400	3.7	1.9	
Indeno[1,2,3-cd]pyrene	200	μg/kg	220	1,200	500	6	2.5	
Phenanthrene	560	μg/kg	180	1,800	800	3.2	1.4	
Pyrene	490	μg/kg	380	2,900	1,200	5.9	2.4	

Note: A - average screening quotient < 1
[a]Number of exceedances/number of analyses

Table A-13. Refined screening of constituents detected in groundwater—consideration of average screening quotient and alternative benchmarks

Constituent	Alternative Screening Value	Units	Source	Minimum Detected Value	Maximum Detected Value	Arithmetic Mean Value	Maximum Screening Quotient	Mean Screening Quotient	Basis
Barium	5,000	µg/L	EPA V	22.7	934	320	0.2	0.1	A
Cadmium	5	µg/L	NJDEP	0.89	5.7	1	1.2	0.2	A
Cobalt	3	µg/L	*	0.94	3.9	1.6	1.3	0.5	A
Copper	59	µg/L	NJDEP	1.7	356	18	6.1	0.3	A
Iron	1,000	µg/L	NJDEP	152	37,500	12,000	37.5	12	
Lead	34.5	µg/L	NJDEP	0.8	13.9	2.3	0.4	0.1	A
Manganese	80	µg/L	*	7.2	6,580	1,350	82.3	16.9	
Mercury (filtered)	770	ng/L	*	923.5	8,473.9	4,025.2	11	5.2	
Mercury (unfiltered)	770	ng/L	*	10.84	54,243	5,000	70.4	6.5	
Methyl mercury (unfiltered)	3	ng/L	*	0.12	32.73	6.7	10.9	2.2	
Selenium	5	µg/L	*	2.34	13.4	2.8	2.7	0.6	A
Thallium	9	µg/L	*	4.9	13.5	3.2	1.5	0.4	A
Vanadium	19	µg/L	*	2.3	50.7	8.5	2.7	0.4	A
Zinc	518	µg/L	NJDEP	9.5	803	132	1.6	0.3	A
Naphthalene	24	µg/L	*	9	100	10	4.2	0.4	A
Benzene	46	µg/L	*	1.2	140	9	3.0	0.2	A
Carbon disulfide	84.1	µg/L	EPA V	16	16	6	0.2	0.1	A
Toluene	130	µg/L	*	330	1,700	80	13.1	0.6	A

Notes: * - screening criterion unchanged from initial screening
A - average screening quotient < 1

Table A-14. Refined screening of constituents detected in surface water—consideration of alternative benchmarks and average screening quotient

Constituent	Screening Value	Units	Source	Minimum Detected Value	Maximum Detected Value	Arithmetic Mean Value	Maximum Screening Quotient	Mean Screening Quotient	Basis
Barium (dissolved)	5,000	µg/L	Region V	189	190	190	0.0	0.0	A
Barium	5,000	µg/L	Region V	40	189	107	0.0	0.0	A
Iron	1,000	µg/L	*	653	2,620	1,740	2.6	1.7	
Manganese (dissolved)	80	µg/L	*	351	373	362	4.7	4.5	
Manganese	80	µg/L	*	141	413	287	5.2	3.6	
Mercury	770.0	ng/L	*	402	17,600	5,000	22.9	6.5	
Zinc	379	µg/L	*	35.9	403	151	1.1	0.4	A

Note: Unless noted, results are for unfiltered analyses
* - screening criterion unchanged from initial screening
A - average screening quotient < 1

Table A-15. Refined screening of CoPCs in surface soil—consideration of alternative benchmarks

Constituent	Alternative Screening Value	Source	Minimum Detected Value	Maximum Detected Value	Arithmetic Mean Value	Maximum Screening Quotient	Average Screening Quotient	Basis
Antimony	5	*	1	54	5	10.7	1.1	A
Chromium	21	SSL	11	1,150	120	54.8	5.7	
Copper	61	SSL	23	1,010	240	16.6	3.9	
Iron	25,000	Background	5,530	122,000	27,000	4.9	1.1	A
Lead	150	Sludge	39	4,320	800	28.8	5.3	
Manganese	560	Background	66	3,090	450	5.5	0.8	A
Mercury (total)	9	Sludge	1	548	120	60.9	13.3	
Nickel	50	CCME	11	82	36	1.6	0.7	A
Selenium	10	CCME	1	2	1	0.2	0.1	A
Silver	2	*	1	94	7	46.9	3.3	
Thallium	1	*	15	22	2	21.9	2.2	
Vanadium	130	CCME	10	175	50	1.3	0.4	A
Zinc	120	SSL	192	25,400	2,665	211.7	22.2	

Note: Units in mg/kg
* - screening criterion unchanged from initial screening
A - average screening quotient < 1 or only nominally above
Background - Average background concentrations for U.S. from Shacklette et al.
CCME - Canadian Soil Guidelines (Canadian Council of Ministers of the Environment)
CoPC - contaminant of potential concern
Sludge - safe soil concentrations from Sludge Regulations
SSL - soil screening levels

Table A-16. Summary of rescreening results

Analyte	Still Soil COPC?	Reason	Still Sediment COPC?	Reason	Still Ground water COPC?	Reason	Still Surface Water COPC?	Reason
Metals								
Aluminum	No	B						
Antimony	No	A						
Arsenic	No	A	No	A				
Barium	No	A			No	AA	No	AA
Cadmium	No	A	**Yes**	**X**	No	AA		
Chromium	**Yes**	**X**	**Yes**	**X**				
Cobalt					No	A		
Copper	**Yes**	**X**	**Yes**	**X**	No	AA		
Iron	No	AA			**Yes**	**X**	**Yes**	**X**
Lead	**Yes**	**X**	**Yes**	**X**	No	AA		
Manganese	No	AA			**Yes**	**X**	**Yes**	**X**
Mercury (total)	**Yes**	**X**	**Yes**	**X**	**Yes**	**X**	**Yes**	**X**
Nickel	No	AA	Yes	X				
Selenium	No	AA			No	A		
Silver	**Yes**	**X**	**Yes**	**X**				
Thallium	**Yes**	**X**			No	A		
Vanadium	No	AA			No	A		
Zinc	**Yes**	**X**	**Yes**	**X**	No	AA	No	A
Methylmercury								
Methylmercury	No	A			**Yes**	**X**		
PCBs								
Aroclor® 1248			**Yes**	**X**				
Aroclor® 1250			**Yes**	**X**				
Semivolatile Organic Compounds								
Acenaphthene			**Yes**	**X**				
Acenaphthylene			**Yes**	**X**				
Anthracene			**Yes**	**X**				
Benz[a]anthracene			**Yes**	**X**				
Benzo[a]pyrene			**Yes**	**X**				
Benzo[ghi]perylene			**Yes**	**X**				
Benzo[k]fluoranthene			**Yes**	**X**				
Bis[2-ethylhexyl]phthalate	No	A,C						
Chrysene			**Yes**	**X**				
Dibenz[a,h]anthracene			**Yes**	**X**				
Di-*n*-butyl phthalate			**Yes**	**X**				
Fluoranthene			**Yes**	**X**				
Naphthalene					No	A		
Indeno[1,2,3-cd]pyrene			**Yes**	**X**				
Phenanthrene			**Yes**	**X**				
Pyrene			**Yes**	**X**				
Volatile Organic Compounds								
Benzene					No	A		
Carbon disulfide					No	AA		
Toluene					No	A		

Table A-16. (cont.)

Note: CoPCs which exceeded benchmarks (see Table A-1) after rescreening along with reason for dismissal from CoPC list. Compounds with an "X" still CoPCs after refinements.

No -- No longer CoPC after refinement

Yes -- Still CoPC after refinement

Rationale for dismissal from CoPC list

- A - Average screening quotient < 1
- B - maximum did not exceed twice the background concentration
- C - frequency of exceedance < 5%
- AA - Average screening quotient with alternative benchmark < 1.0
- X - None of the above true, compound retained as CoPC after rescreening

CoPC - contaminant of potential concern

PCB - polychlorinated biphenyl

Table A-17 Screening of constituents detected in developed areas surface soil against ecological screening criteria

Constituent	Screening Criterion Value	Screening Criterion Source	Detection Frequency	Minimum Detected Value	Maximum Detected Value	Arithmetic Mean[a]	Maximum Screening Quotient[b]	Average Screening Quotient[c]	Alternative Benchmark	Average Alt. SQ
Aluminum	50	Efroymson et al. (1997c)	11/11	2,950	12,000	6,000	240.0	120.0	66,000	0.1
Antimony[d]	5	Efroymson et al. (1997a)	0/11	0	0.0	0.0	0.0	0.0		
Arsenic	10	Efroymson et al. (1997a)	6/11	3	11	3	1.1	0.3		
Barium	283	Efroymson et al. (1997a)	11/11	25	190	90	0.7	0.3		
Cadmium	4	Efroymson et al. (1997a)	5/11	0.2	3.1	0.6	0.8	0.1		
Chromium	0.4	Efroymson et al. (1997a)	11/11	4	97	22	242.3	54.0	21	1.0
Copper	60	Efroymson et al. (1997a)	11/11	12	470	100	7.8	1.7	61	**1.6**
Iron	200	Efroymson et al. (1997b)	11/11	3,120	23,000	11,000	115.0	55.0	25,000	0.4
Lead	40.5	Efroymson et al. (1997a)	11/11	11	390	100	9.6	2.5	150	0.7
Manganese	100	Efroymson et al. (1997b)	11/11	110	540	260	5.4	2.6	560	0.5
Mercury (total)	0.00051	Efroymson et al. (1997a)	11/11	9.3	13,800	1,700	27,058,824	3,333,333	9	**188.9**
Nickel	30	Efroymson et al. (1997a)	9/11	3.6	72	19	2.4	0.6	50	0.4
Selenium[d]	0.21	Efroymson et al. (1997a)	1/11	0.7	0.7	0.3	3.3	1.6	10	0.0
Silver	2	Efroymson et al. (1997a)	6/11	0.6	8.0	1.8	4.0	0.9	*	0.9
Thallium[d]	1	Efroymson et al. (1997a)	6/11	0.6	8.0	1.8	8.0	1.8	*	**1.8**
Vanadium	2	Efroymson et al. (1997a)	0/11	0.0	0.0	0.0	0.0	0.0		
Zinc[d]	8.5	Efroymson et al. (1997a)	11/11	5.2	140	30	16.5	3.5	120	0.3

Note: Units in mg/kg.

Bold entries indicate screening quotient is greater than 1.

[a] Mean calculated based on detected values and one-half the detection limit for undetected values.

[b] Maximum concentration/screening criterion.

[c] Arithmetic mean concentration/screening criterion.

[d] Screening criterion is less than detection limit.

Table A-18. Assessment of risk to aquatic benthos—comparisons of sediment concentrations to ER-M values

Constituent	ER-M Value	Units	Maximum Value in Sediments	Mean Value in Sediments	Maximum ER-M SQ	Mean ER-M SQ
Arsenic	70	mg/kg	8.8	5	0.1	0.1
Cadmium	9.6	mg/kg	9.1	3.7	0.9	0.4
Chromium	370	mg/kg	156	103	0.4	0.3
Copper	270	mg/kg	194	143	0.7	0.5
Lead	218	mg/kg	469	279	2.2	1
Mercury	0.7	μg/kg	1,290	686	1,817	966.2
Nickel	52	mg/kg	29.2	24.7	0.6	0.5
Silver	4	mg/kg	4.3	1.6	1.2	0.4
Zinc	410	mg/kg	3,540	1,430	8.6	3.5
Aroclor® 1248	130	μg/kg	240	210	2	2
Aroclor® 1260	130	μg/kg	490	370	4	3
Acenaphthene	500	μg/kg	100	440	0	0.9
Acenaphthylene	640	μg/kg	270	360	0.4	0.6
Anthracene	1,100	μg/kg	350	280	0.3	0.3
Benz[a]anthracene	1,600	μg/kg	1,700	700	1.1	0.4
Benzo[a]pyrene	1,600	μg/kg	1,600	700	1.0	0.4
Benzo[ghi]perylene	NA	μg/kg	1,200	600		
Benzo[k]fluoranthene	NA	μg/kg	660	430		
Chrysene	2,800	μg/kg	1,600	800	0.6	0.3
Dibenz[a,h]anthracene	260	μg/kg	320	200	1.2	0.8
Di-*n*-butyl phthalate	NA	μg/kg	160	160		
Fluoranthene	5,100	μg/kg	2,800	1,400	0.5	0.3
Indeno[1,2,3-cd]pyrene	NA	μg/kg	1,200	500		
Phenanthrene	1,500	μg/kg	1,800	800	1.2	0.5
Pyrene	2,600	μg/kg	2,900	1,200	1.1	0.5
Sum of ERM SQ (Canfield et al., see text)					14.3	6.6
Average of ERM -SQ (Canfield et al.)					1.8	0.8
Sum of ERM SQ, all compounds					1845	982
Average of ERM SQ (all compounds)					87.9	46.8

Note: CoPC - contaminant of potential concern
ER-M - effects range-medium
NA - none available
SQ - screening quotient

Table A-19. Estimation of CoPC concentrations in aquatic benthos (wet weight)

Constituent	Units	Maximum Value in Sediments	Mean Value in Sediments	Predicted Max. in Benthos	Predicted Mean in Benthos
Arsenic	mg/kg	8.8	5	0.7	0.4
Cadmium	mg/kg	9.1	3.7	1.3	0.7
Chromium	mg/kg	156	103	2.6	2.2
Copper	mg/kg	194	143	13.3	12.2
Lead	mg/kg	469	279	5.8	3.8
Mercury	mg/kg	1,290	686	0.56	0.45
Nickel	mg/kg	29.2	24.7	1.8	1.8
Silver	mg/kg	4.3	1.6	No Estimate	No Estimate
Zinc	mg/kg	3,540	1,430	86.3	71.5
Total PCBs	μg/kg	730	580	75.9	60.3
Total PAHs	μg/kg	16,500	8,410	495	252

Note: CoPC - contaminant of potential concern
PAH - polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl
No Estimate - No regression equation available. See text.

Table A-20. Assessment of risk to consumers of adult aquatic insects: brown bat and redwing blackbird

Constituent	Toxicity Reference Value (mg/kg/day)	Feeding Rate (mg/kg/day)	Maximum Predicted Conc. in Biota	Mean Predicted Conc. in Biota	Max. Dose (mg/kg/day)	Mean Dose (mg/kg/day)	Max. SQ	Mean SQ
				Brown Bat				
Cadmium	1	0.62	1.26	0.68	0.78	0.42	0.78	0.42
Chromium	3.3	0.62	2.56	2.20	1.58	1.36	0.48	0.41
Lead	8	0.62	5.80	3.80	3.60	2.36	0.45	0.29
Methylmercury	0.05	0.62	0.56	0.45	0.35	0.28	**6.94**	**5.58**
Zinc	160	0.62	86.30	71.50	53.51	44.33	0.33	0.28
PCBs	0.16	0.62	0.08	0.06	0.05	0.04	0.29	0.23
PAH	1.0	0.62	0.50	0.25	0.31	0.16	0.31	0.16
				Redwing Blackbird				
Cadmium	1.45	0.5	1.26	0.68	0.63	0.34	0.44	0.23
Chromium	1.6	0.5	2.56	2.20	1.28	1.10	0.80	0.69
Lead	3.85	0.5	5.80	3.80	2.90	1.90	0.75	0.49
Methylmercury	0.03	0.5	0.56	0.45	0.28	0.23	**8.75**	**7.03**
Zinc	130.9	0.5	86.30	71.50	43.15	35.75	0.33	0.27
PCBs	0.41	0.5	0.08	0.06	0.04	0.03	0.09	0.07
PAH	1.0	0.5	0.50	0.25	0.25	0.13	0.25	0.13

Note: **Bold** entries indicate screening quotient is greater than 1.
PAH - polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl
SQ - screening quotient

Table A-21. Assessment of risk to consumers of aquatic benthos: raccoon and mallard

Constituent	Toxicity Reference Value (mg/kg/day)	Feeding Rate (mg/kg/day)	Maximum Predicted Conc. in Biota	Mean Predicted Conc. in Biota	Max. Dose (mg/kg/day)	Mean Dose (mg/kg/day)	Max. SQ	Mean SQ
				Raccoon				
Cadmium	1	0.19	1.26	0.68	0.28	0.15	0.28	0.15
Chromium	3.3	0.19	2.56	2.20	1.18	0.88	0.36	0.27
Lead	8	0.19	5.80	3.80	3.20	1.97	0.40	0.25
Mercury	1.01	0.19	0.56	0.45	5.87	3.15	**5.81**	**3.12**
Methylmercury	0.05	0.19	0.56	0.45	0.11	0.09	**2.14**	**1.72**
Zinc	160	0.19	86.30	71.50	32.20	19.97	0.20	0.12
PCBs	0.16	0.19	0.08	0.06	0.02	0.01	0.11	0.09
PAH	1	0.19	0.50	0.25	0.17	0.09	0.17	0.09
				Duck				
Cadmium	1.45	0.308	1.26	0.68	0.40	0.21	0.28	0.15
Chromium	1.6	0.308	2.56	2.20	1.03	0.84	0.64	0.52
Lead	3.85	0.308	5.80	3.80	2.51	1.60	0.65	0.42
Mercury	0.45	0.308	0.56	0.45	0.71	0.57	**1.58**	**1.27**
Methylmercury	0.03	0.308	0.56	0.45	0.17	0.14	**5.40**	**4.33**
Zinc	130.9	0.308	86.30	71.50	32.03	24.22	0.24	0.19
PCBs	0.41	0.308	0.08	0.06	0.02	0.02	0.06	0.05
PAH	1	0.308	0.50	0.25	0.18	0.09	0.18	0.09

Note: **Bold** entries indicate screening quotient is greater than 1.
PAH - polycyclic aromatic hydrocarbon
PCB - polychlorinated biphenyl
SQ - screening quotient

Table A-22. Assessment of risk to piscivorous wildlife: mink and belted kingfisher

Constituent	Toxicity Reference Value (mg/kg/day)	Feeding Rate (mg/kg/day)	Maximum Predicted Conc. in Fish	Mean Predicted Conc. in Fish	Max. Dose (mg/kg/day)	Mean Dose (mg/kg/day)	Max. SQ	Mean SQ
				Mink				
Methylmercury	0.05	0.15	1.10	0.90	0.17	0.14	**3.30**	**2.70**
PCBs	0.16	0.15	0.20	0.16	0.03	0.02	0.19	0.15
				Kingfisher				
Methylmercury	0.03	0.50	1.10	0.90	0.55	0.45	**17.19**	**14.06**
PCBs	0.41	0.50	0.20	0.16	0.10	0.08	0.24	0.20

Note: **Bold** entries indicate screening quotient is greater than 1.
PCB - polychlorinated biphenyl
SQ - screening quotient

Table A- 23. Estimation of chemical concentrations in terrestrial food chain

Constituent	Soil Maximum	Soil Mean	Earthworm Maximum	Earthworm Mean	Small Mammal Maximum	Small Mammal Mean
Chromium	1,150	120	1.4	1.4	10.2	1.9
Copper	1,010	240	5.3	3.6	5.2	4.3
Lead	4,320	800	110.5	28.3	10.9	5.2
Mercury	3,090	450	2.6	1.4	0.4	0.2
Silver	94	7	No estimate	No estimate	No estimate	No estimate
Thallium	22	2	No estimate	No estimate	No estimate	No estimate
Zinc	25,400	2,665	381.1	181.9	46.2	39.1

Note: Units in mg/kg; dry weight for soil and wet weight for biota
No estimate - No regression equation available. See text.

Table A-24. Assessment of risk to consumers of soil invertebrates: shrew and woodcock

Constituent	Toxicity Reference Value (mg/kg/day)	Feeding Rate (mg/kg/day)	Maximum Predicted Conc. In Biota	Mean Predicted Conc. In Biota	Max. Dose (mg/kg/day)	Mean Dose (mg/kg/day)	Max. SQ	Mean SQ
				Shrew				
Chromium	3.3	0.6	1.4	1.4	11.4	1.9	**3.5**	0.6
Copper	11.7	0.6	5.3	3.6	12.3	4.2	1.0	0.4
Lead	8.0	0.6	110.5	28.3	101.2	23.1	**12.7**	**2.9**
Mercury	1.0	0.6	2.6	1.4	6.5	1.9	**6.4**	**1.8**
Zinc	160.0	0.6	381.1	181.9	446.1	125.6	**2.8**	0.8
				Woodcock				
Chromium	1.6	0.8	1.4	1.4	15.8	2.6	**9.9**	**1.6**
Copper	47.0	0.8	5.3	3.6	17.0	5.9	0.4	0.1
Lead	3.9	0.8	110.5	28.3	140.4	32.1	**36.5**	**8.3**
Mercury	0.5	0.8	2.6	1.4	9.0	2.6	**20.0**	**5.7**
Zinc	130.9	0.8	381.1	181.9	618.9	174.2	**4.7**	**1.3**

Note: **Bold** entries indicate screening quotient is greater than 1.
SQ - screening quotient

Table A-25. Assessment of risk to top predators: fox and red-tailed hawk

Constituent	Toxicity Reference Value (mg/kg/day)	Feeding Rate (mg/kg/day)	Maximum Predicted Conc. in Biota	Mean Predicted Conc. In Biota	Max. Dose (mg/kg/day)	Mean Dose (mg/kg/day)	Max. SQ	Mean SQ
				Fox				
Chromium	3.4	0.1	10.2	1.9	1.8	0.3	0.5	0.1
Copper	11.7	0.1	5.2	4.3	1.2	0.6	0.1	0.1
Lead	8.0	0.1	10.9	5.2	4.1	1.1	0.5	0.1
Mercury	1.0	0.1	0.4	0.2	0.4	0.1	0.4	0.1
Zinc	160.0	0.1	46.2	39.1	22.4	5.8	0.1	0.0
				Red-tailed Hawk				
Chromium	1.6	0.1	10.2	1.9	1.8	0.3	**1.1**	0.2
Copper	47.0	0.1	5.2	4.3	1.2	0.6	0.0	0.0
Lead	3.9	0.1	10.9	5.2	4.1	1.1	**1.1**	0.3
Mercury	0.5	0.1	0.4	0.2	0.4	0.1	0.9	0.2
Zinc	131.0	0.1	46.2	39.1	22.1	5.7	0.2	0.0

Note: **Bold** entries indicate screening quotient is greater than 1.
SQ - screening quotient

Table A-26. Sensitivity analysis to conservative assumptions—estimation of effects of safety factors associated with default conservative assumptions

Receptor	CoPC	Max. SQ	Mean SQ	100% Residence	100% Diet	100% Methyl	Simple Sediment Mean	100% Absorption
Bat	Methylmercury	6.9	5.6	**298.7**	1.3	4.0	1.5	NA
Redwing Blackbird	Methylmercury	8.8	7.0	2.0	2.3	4.0	1.5	NA
Raccoon	Inorganic mercury	5.8	3.1	**19.8**	2.0	NA	**3.3**	**8.3**
Raccoon	Methylmercury	2.1	1.7	**19.8**	**2.0**	**4.0**	1.5	NA
Duck	Inorganic mercury	1.6	1.3	**79.9**	**2.0**	NA	**1.8**	**3.1**
Duck	Methylmercury	5.4	4.3	**79.9**	2.0	4.0	1.5	NA
Shrew	Chromium	3.4	0.6	**1.0**	**1.0**	NA	NA	**4.2**
Shrew	Lead	12.7	2.9	1.0	1.0	NA	NA	1.5
Shrew	Inorganic mercury	6.4	1.8	1.0	1.0	NA	NA	**2.7**
Shrew	Zinc	2.8	0.8	**1.0**	**1.0**	NA	NA	**1.6**
Woodcock	Chromium	9.9	1.6	**5.8**	1.0	NA	NA	**4.2**
Woodcock	Lead	36.5	8.3	5.8	1.0	NA	NA	1.5
Woodcock	Inorganic mercury	20.0	5.7	**5.8**	1.0	NA	NA	2.7
Woodcock	Zinc	4.7	1.3	**5.8**	1.0	NA	NA	**1.6**
Mink	Methylmercury	3.3	2.7	**6.4**	2.0	1.1	NA	NA
Kingfisher	Methylmercury	17.2	14.1	10.6	1.0	1.1	NA	NA
Fox	Inorganic mercury	0.4	0.1	**55.5**	**2.0**	NA	NA	**4.7**
Red-tailed Hawk	Chromium	1.1	0.2	**119.6**	**1.0**	NA	NA	**1.6**
Red-tailed Hawk	Lead	1.1	0.3	**119.6**	**1.0**	NA	NA	**2.4**

Note: **Bold** entries indicate effects that are greater than mean screening quotient.
CoPC - contaminant of potential concern
NA - not applicable
SQ - screening quotient