

LIFE-CYCLE FRAMEWORK FOR ASSESSMENT OF SITE REMEDIATION OPTIONS: CASE STUDY

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Abstract—An adaptation of life-cycle assessment for contaminated site remediation activities was used to examine a lead-contaminated site remediated by excavation and disposal. The study indicated emissions and impacts associated with energy consumption (transportation of contaminated soil to hazardous and nonhazardous sites), solid waste production (contaminated soil and sludge), land use at four sites (the contaminated site, two disposal sites, and a borrow pit for clean fill), and potential toxicity from emissions and contaminants remaining on-site. These impacts were drawn from an inventory compiled mainly from proprietary consultants' reports, and related to impacts through several stressor-impact models; the process-related indicators of global warming potential, solid waste burden, and ecological and human toxicity potential; and site-related indicators of land use and residual human toxicity burden. Ecological and human toxicity potentials were estimated using the Mackay level III model for southern Ontario, Canada, adapted for metals. The life-cycle approach proved useful for identifying potential impacts occurring at local, regional, and global scales, over all activities and locations affected by the remediation.

Keywords—Life-cycle inventory Life-cycle impact assessment Site remediation Contaminated sites Toxicity assessment

INTRODUCTION

Contaminated sites can present short- and long-term risks to environmental and human health. The choice of remediation approaches to mitigate these risks depends on factors such as duration of remediation, cost, and effectiveness of the method used for the contaminants involved. Associated with each remediation option, however, are inherent burdens that may also contribute to environmental and human health risks.

To examine the potential environmental and human health implications of remediation options, a life-cycle framework (LCF) has been developed by Diamond et al. [1]. The framework is systematic, is based on life-cycle thinking, and includes two complementary approaches of differing complexity: a simpler, qualitative life-cycle management (LCM) and a more rigorous, quantitative life-cycle assessment (LCA) approach. Diamond et al. [1] illustrated the LCM approach by application to six generic remediation technologies. That analysis provided insight into sources of potential impacts (e.g., life-cycle stages or activities) associated with each technology and broadened consideration of impacts beyond the contaminated site and time horizon of major clean-up activities. They also described modifications to existing LCA methods [2-4] for application to contaminated site remediation that yield a systematic and quantitative analysis of remediation activities from cradle-to-grave.

This paper analyzes remediation activities at a contaminated site using the LCA-based approach. Our objective was first, to illustrate and evaluate LCA for site remediation and second,

to draw attention to impacts not usually considered when using particular remediation methods.

The case study considered in this paper was a parcel of land contaminated predominantly with lead. The remediation approach examined is known as excavation and disposal or dig and haul. A small area was also treated using the risk management approach of capping. This case study was chosen because it uses common remediation methods and case study data of high quality and completeness were accessible. Excavation and off-site disposal is used frequently because it is cost effective, convenient, reduces the mobility of contaminants through landfill containment, and is generally acceptable to the community surrounding the remediation site. However, this option does not reduce contaminant toxicity or volume, and dust and fugitive air emissions may be generated [5]. Capping contains contaminants on-site but does not reduce contaminant volume. Investigating the case study from a life-cycle perspective provides new insights into these and other aspects of a widely used remedial approach.

METHOD

The assessment approach used here is based on existing LCA methods [2-6] with boundaries, process descriptions, data issues, life-cycle stages, and impact assessment modifications described by Diamond et al. [1]. The analysis is presented according to the LCA components of initiation, inventory assessment, impact assessment, and interpretation [2].

Briefly, initiation involves defining the goal of the study, bounding the scope of the study, describing the remediation activities through process flow diagrams, explaining data issues, and describing assumptions. Inventory assessment assembles data on raw materials and energy consumption, air-

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borne and waterborne emissions, and solid waste associated with the remediation and related activities. Impact assessment links inventory items with impact indicators in order to estimate potential process-related and site-related impacts. Finally, interpretation analyzes the remediation activities that contribute most to the environmental burden of the process(es) as estimated by the selected impact indicators. A full analysis of the approach and case study is contained in a report by Diamond et al [6].

INITIATION

Goal definition

The two main purposes of this study were to examine a site remediation process from a life-cycle perspective, and assess and evaluate the LCA method itself. Specifically, we illustrate the application of life-cycle thinking to remediation activities, determine the inventory associated with these activities, and link the inventory with potential impacts, with the objective of broadening the environmental and human health considerations beyond the site itself.

Boundaries and scope of study

The site, located in the greater Toronto, Ontario, Canada, area, was extensively contaminated with lead and, secondarily, with arsenic, cadmium, and polyaromatic hydrocarbons (PAHs). In addition to the contaminated soil, on-site buildings and paving were contaminated. Most of the site was decommissioned to residential standards; one lead-contaminated area was capped with asphalt and access to a PAH-contaminated section was permanently restricted. Contaminated soil and sludge were discarded in hazardous and nonhazardous waste disposal facilities located in southern Ontario. Clean backfill was obtained from another location in southern Ontario. The duration of site remediation, from the beginning of soil excavation to final backfilling, was approximately 75 weeks, including a shut-down period.

The geographic boundary of the study is southern Ontario, encompassing all sites affected by clean-up activities. Temporally, the life cycle for this case study begins with remediation activities and extends over a 25-year time horizon, which is intended to capture longer term effects that could arise from various disposal scenarios and allow for inclusion of potential impacts regardless of time dependency [1].

The scope of the study is limited to an investigation of specific stages of the site remediation life cycle, including site processing, raw materials acquisition, waste treatment, and transportation. We have not elaborated on post-site processing and monitoring activities because of their minor role overall nor have we included site decommissioning activities such as decontaminating building surfaces and dismantling and demolishing on-site building structures.

Data issues, assumptions, and peer review

For this study, the data were largely facility-specific and proprietary, coming from primary sources. The consultants' reports, in particular the final project completion report, were the major sources of the data. Personal communications with key individuals involved in the remediation project were another important source of data. Because the data are from a specific remediation project, deviations or variations in data have not been smoothed out. Publicly accessible data from secondary sources are noted as appropriate. Secondary data sources (e.g., transportation emission factors) reflect the geo-

graphic specificity of southern Ontario and the time period of 1992 to 1993. The consultants' reports and analyses were assumed to be complete and accurate.

An initial analysis of mass contribution [2] was conducted to determine the relevance of the various material streams. The analysis revealed that the system was dominated by the hazardous soil excavated and clean fill brought in (>35% and >47% of total mass contribution, respectively). However, the mass contribution analysis alone was not used as a method for determining data inclusion. Rather, we investigated most aspects of excavation and disposal because it is difficult to judge the environmental relevance of various remediation aspects at this formative stage. For example, special consideration of lead was made despite its mass contribution, because of its toxicity and role as the major on-site contaminant.

As discussed by Diamond et al [1], several alternative functional units exist for an analysis of site remediation activities; however, none are ideal. For this study, the data are presented on a per site basis. Other functional units that could be used include land area, volume or mass of treated soil, with the choice of functional unit largely dependent on the goal of the study.

The information contained within this paper has benefited from extensive peer review during the process of conducting the study. Remediation experts and consultants associated with the case study examined the inventory data, LCA practitioners reviewed the modified LCA method, and toxicology experts critiqued the environmental and human health toxicity assessment approach.

INVENTORY ASSESSMENT

Process flow diagrams

Processes and related activities for excavation and disposal are shown in a simplified process flow chart (Fig 1); the numerical references in the text refer to the numbered flow chart modules. The remedial activities presented in the process flow diagram have been identified according to life-cycle stages described by Diamond et al [1]. The contaminated site boundary and the overall system boundary have been highlighted, illustrating how this assessment broadens consideration of activities beyond the contaminated site itself.

The main remediation activities were soil excavation (1) and transportation to hazardous (15) or nonhazardous (14) waste sites according to the extent of contamination. A small portion of the hazardous soil was first combined with fly ash (6) to stabilize the soil for transportation (11), followed by deposition in a hazardous landfill (15). The excavated areas were backfilled (2) with clean backfill from native sources (5) as the remediation progressed. During excavation, a dust mitigation program was in effect (9). Water, including groundwater and surface water runoff from the dust mitigation program, was treated by coagulation followed by settling of the contaminated suspended particulates (13). The water treatment chemicals used included hydrated lime (8) and some poly aluminum chloride (PAC) (7). Highly contaminated water was removed to a liquid disposal facility (16). The water treatment sludge (15) was stabilized and sent to a hazardous waste landfill. Most of the site was decommissioned to residential status, one lead-contaminated area was capped (3) with asphalt (4), and access to a PAH-contaminated section was permanently restricted. Air emissions were monitored during site excavation and backfilling (10), and contaminant levels in the water treatment system (13) were measured.

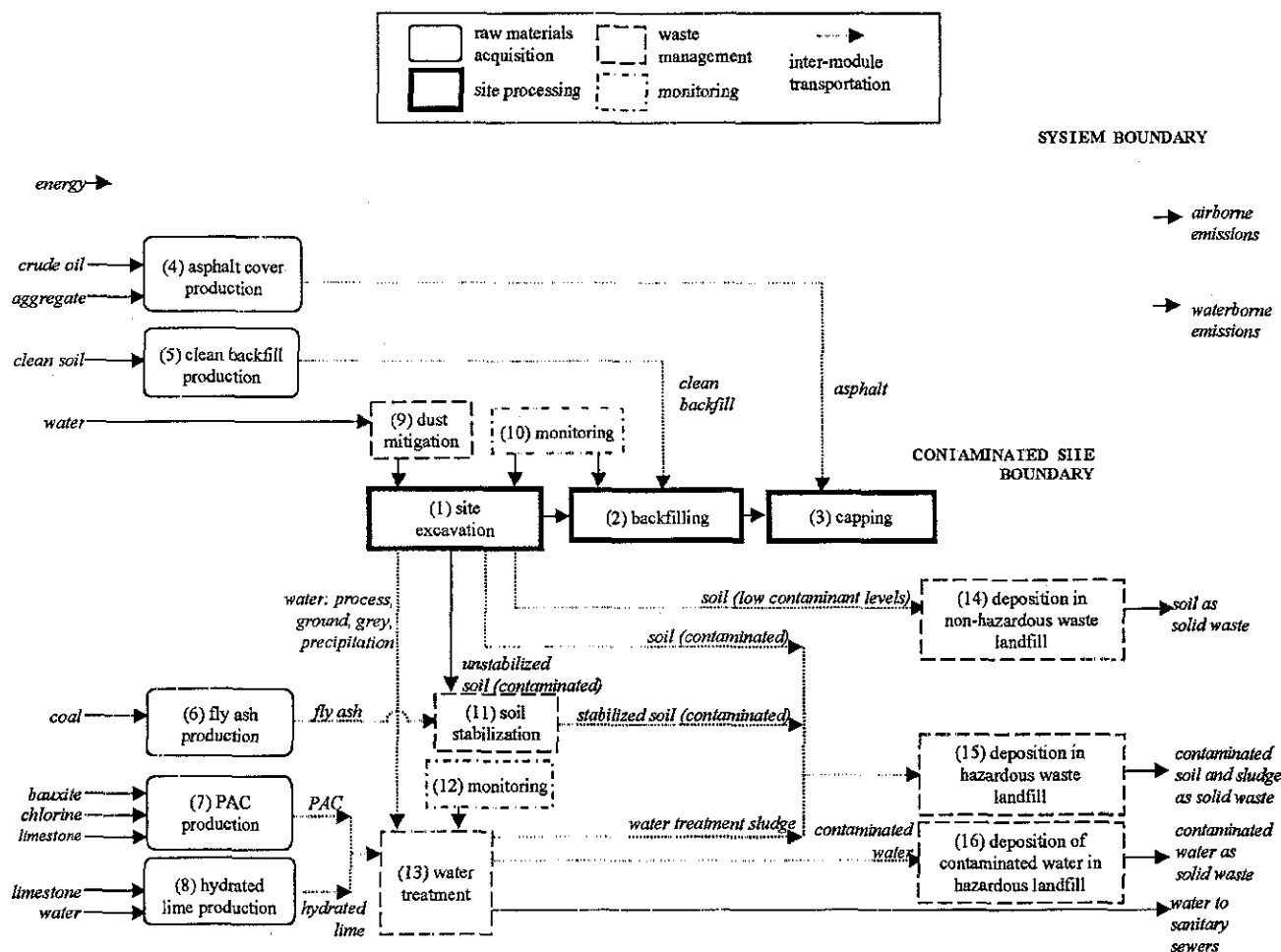


Fig 1 Schematic of case study remediation activities

Inventory results and discussion

Items associated with all activities included within the flow diagram (Fig 1) were included in the inventory with the exception of the following, because of a lack of data or their minor contribution: PAC production (7), and consequently the inputs of bauxite, chlorine, and limestone, with associated emissions and energy consumption; emissions associated with asphalt laying are addressed qualitatively only; inputs and outputs, including energy use associated with monitoring site excavation and backfilling (10); and water treatment (12). Although environmental releases from land disposal facilities may be included [3], emissions from hazardous and nonhazardous waste landfills were not included because of a lack of appropriate data

The inventory items, listed as stressors in Table 1, are subdivided by impact categories of pollution, disturbance, and depletion, according to the potential impact checklist [1] The inventory items are also listed by four life-cycle stages: raw materials and energy acquisition, site processing, waste management, and transportation. Note that in Figure 1 "transportation" involves only the activities facilitating movement between modules and is represented by the broken lines joining selected modules (e.g., trucking clean backfill from clean backfill production to backfilling) Distribution, or on-site transportation activities, includes transportation within each module. For example, site excavation involved the operation of

backhoes, loaders, and bulldozers, with associated inputs and emissions being grouped under site processing The inventory items associated with energy acquisition have been grouped under "fossil fuel use and energy consumption" and were, therefore, allocated to their respective life-cycle stage (e.g., precombustion emissions and energy associated with backfill production were grouped under the raw materials acquisition life-cycle stage).

Pollution These stressors were grouped under three categories, acid emissions, photochemical smog precursors, and greenhouse gases; contaminants and particulates emitted to air; and contaminants discharged to water.

Emissions of acid precursors, (e.g., SO₂, NO_x), air pollutants (e.g., particulates), photochemical smog precursors (e.g., volatile organic compounds [VOCs], NO_x), and greenhouse gases (e.g., CO₂, CO) were largely related to fuel combustion for site processing, raw materials acquisition, and transportation life-cycle stages. These emissions were calculated from energy consumption values (see fossil fuel use under depletion) using emission factors from several sources, appropriate for the 1992 through 1993 time frame [7-12]. Overall, most emissions (72-78% for each stressor) were associated with off-site (i.e., between-module) transportation activities Within these transportation activities, the major contributors to pollution were from transportation of hazardous soil to the hazardous landfill site and clean backfill to the excavated site (75 and

Table 1. Summary of life-cycle inventory items^a

Stressor category	Stressor	Life-cycle stages			
		Raw materials acquisition	Site processing	Waste management	Transportation
Pollution					
Acid emissions/photochemical smog precursors	NO _x (kg)	3,120	2,950	—	20,700
	SO ₂ (kg)	800	776	—	5,550
	Particulates (kg)	285	276	—	1,950
	VOC (kg)	615	597	—	3,900
Greenhouse gases	CO ₂ (kg)	334,000	265,000	—	1,880,000
	C _x H _y (kg)	270	240	—	1,280
	CO (kg)	7,260	7,040	—	50,400
	NO _x (kg) ^b	3,120	2,950	—	20,700
Contaminants/particulates to air	Coarse dust (kg)	—	1,400	—	—
	Lead (kg)	—	3.57	—	—
Contaminants in surface and groundwater	Fluorine (kg)	—	—	4.51	—
	Silver (kg)	—	—	0.05	—
	Aluminum (kg)	—	—	116	—
	Arsenic (kg)	—	—	5.81	—
	Cadmium (kg)	—	—	1.60	—
	Chromium (kg)	—	—	0.40	—
	Copper (kg)	—	—	13.5	—
	Iron (kg)	—	—	550	—
	Lead (kg)	—	—	91.1	—
	Phosphorus (kg)	—	—	6.11	—
	Zinc (kg)	—	—	13.0	—
	BOD (kg)	—	—	60.0	—
	TSS (kg)	—	—	4,530	—
Disturbance					
Aquifer quality stressors	Groundwater (kg)	—	—	8,000,000	—
Depletion					
Fossil fuel use/energy consumption	Diesel (GJ)	3,560	3,450	—	16,700
	Oil (GJ)	85.4	—	—	—
	Natural gas (GJ)	11.5	—	—	—
	Coal (GJ)	0.814	—	—	—
	Nuclear (GJ)	1.64	—	—	—
	Hydro (GJ)	0.357	—	—	—
	Precombustion (GJ)	401	380	—	1,830
	Other fuel (GJ)	—	1.05	—	—
	Coal (raw material) (kg)	300,000	—	—	—
	Crude oil (raw material) (kg)	26,100	—	—	—
Solid waste	Nonhazardous soil (kg)	—	—	12,400,000	—
	Hazardous soil (kg)	—	—	50,380,000	—
	Hazardous sludge (kg)	—	—	95,000	—
	Hazardous water (kg)	—	—	108,000	—
	Mineral waste (kg)	215	169	—	5,340
	Ash (kg)	207	192	—	1,030
	Inert chemicals/industrial waste (kg)	42.9	36.8	—	197
	Drillings/cuttings (kg)	36.3	—	—	—
	Water (kg)	195	—	3,430,000	—
	Mineral/soil use	Aggregate (kg)	235,000	—	—
Clean soil (kg)		67,500,000	—	—	—
Limestone (kg)		1,080	—	—	—

^a VOC = volatile organic compound; ISS = total suspended solids; GJ = gigajoule

^b Listed again for illustrative purposes

20% of the transportation life-cycle stage CO₂ emissions, respectively) Transportation of nonhazardous waste, water treatment sludge, and other materials contributed negligible amounts.

Within the raw materials acquisition life-cycle stage, the majority of air emissions were related to clean backfill production (e.g., 82% of total CO₂; 97% of NO_x and C_xH_y; 100% of CO and particulates) Asphalt production, however, contributed 42% of methane emissions, but hydrated lime production contributed negligibly Potential emissions from asphalt-laying (e.g., benzene, PAHs, lead, total particulates and

particulate matter less than 10 μm diameter [PM₁₀], CO, SO₂ [13,14]) have not been quantified For site processing, the majority (>96%) of pollution emissions were related to site excavation activities

Excavation and backfilling activities contributed to particulate emissions to air (e.g., coarse dust and lead associated with airborne particles). These emissions were measured as dustfall and total lead in dustfall, and reported on a monthly basis over the duration of the remediation activities, including the shut-down period

In the discharged wastewater, the concentrations of several

contaminants were determined, for example, BOD and total suspended solids, aluminum, arsenic, iron, and lead. Water from several sources was collected and treated together: water for dust mitigation (e.g., wetting the excavation site, washing trucks), gray water (e.g., showers and laundry), rainfall, and groundwater. It was not possible to distinguish between water used for remediation versus decommissioning activities, with the latter being outside the process boundary. Thus, the total amount of contaminants discharged in the water is potentially greater than that associated with only excavation and disposal activities.

Disturbance Disturbance refers to on- and off-site disruption, degradation or restriction of land, and degradation of groundwater quality. Off-site land quantity (e.g., related to waste disposal), and water and backfill use are discussed under depletion. The disturbance stressor categories may be grouped into disturbances to land through nonremediation of land or application of an impervious surface, and aquifer stressors. At the contaminated site, regions remain contaminated and a portion of the site was capped. Overall, 28% of the site area has contaminants at depth and/or is paved. The major stressor to the aquifer was associated with the removal of approximately 8,000 m³ of groundwater and capping that restricts recharge.

Depletion More than 70% of all fossil fuel and energy consumption was associated with the transportation life-cycle stage. Of this, transporting hazardous soil consumed the greatest proportion of energy (75% of transportation energy). For off-site transportation (i.e., between modules), fossil fuel use was estimated using information from transportation logs (e.g., payload, vehicle type, and haulage route), energy consumption factors [8–10,15], and mode of travel considered. Most of the fossil fuel use and energy consumption for raw materials acquisition and site processing was associated with distribution (i.e., on-site transportation).

Throughout the project, diesel fuel was the main energy type used, followed by natural gas and electricity, with electricity coming from the Ontario grid (comprised of 45% nuclear, 32% hydro, 23% coal). Hydrated lime production uses energy to mine the limestone and raise it to decomposition temperature [16–18], using natural gas, transportation fuels, and electricity. Asphalt production requires energy to produce asphalt, mine mineral aggregates, and prepare asphalt-aggregate mixes [13,19]. The energy associated with the production of fly ash, as discussed below, was not included in the analysis.

Fly ash and asphalt are by-products of coal combustion [20] and crude oil refining [19,21], respectively. Here, the fly ash originated from a local utility, where approximately 0.073 tonnes of ash is produced per tonne of coal. Because the purpose of coal combustion is predominantly energy production, fly ash may be considered as a waste or coproduct. The total amount of coal used to produce fly ash was allocated on a mass basis [22], which appears in Table 1 as "coal (raw material)," with no assigned environmental burdens (i.e., zero allocation). Similarly, asphalt is one of many petroleum products produced from crude oil [19] and again, allocation of crude oil and the fraction of energy use allotted to asphalt production was made on a mass basis [22].

Solid waste was produced from site excavation, where 82% of the excavated soil was considered hazardous and deposited in hazardous landfill. In addition, contaminated wastewater and water treatment sludge were removed from the site and landfilled. Other solid wastes produced such as mineral waste, ash,

inert chemicals, drillings, and cuttings were associated with the precombustion of fossil fuels [7,10,18].

Process water, estimated at 3,430 m³, was used for many activities, including dust mitigation. In addition, water was used in the production of hydrated lime [16], although this amount is negligible in comparison to the total amount of process water used on-site.

Finally, mineral and soil use included the aggregate used for the asphalt cap [13], limestone used in the production of hydrated lime [16], and clean soil taken from a borrow pit to backfill the excavated site.

IMPACT ASSESSMENT

Life-cycle impact assessment (LCIA) is used to evaluate the potential impacts of stressors on ecosystems, human health, and natural resources. Quantitative impact indicators are used as surrogates of these impacts and are few [18] in comparison to the comprehensive coverage presented in the potential impact checklist [1]. In this study, the impact assessment includes process- and site-related impacts.

Process-related potential impacts

Global warming potential and solid waste burden impact indicators. Global warming potential (GWP) is an index reflecting the contribution of various greenhouse gas emissions to atmospheric global warming, weighted relative to carbon dioxide. For this study, radiative forcing values were derived from the Intergovernmental Panel on Climate Change (IPCC) [23], with a time horizon of 100 years and expressed as a CO₂-equivalent mass. The total GWP for the case study was 3.77 × 10⁶ kg CO₂-equivalent (i.e., on a per site basis).

Solid waste burden (SWB) is a cumulative measure of solid wastes emitted from all life-cycle stages, expressed in mass units. Although SWB does not relate directly to an impact and does not capture the number of sites affected, it is indicative of land consumption for disposal, total material use, and the potential for chemical emissions from disposal sites. Two methods for calculating SWB have been used for this study: SWB_A includes all solid wastes generated from all life-cycle stages; SWB_B excludes all soils (e.g., hazardous and nonhazardous soil) and sludges coming from the site itself, and includes only mineral waste, ash, inert chemicals, industrial waste, and drillings and cuttings. The SWB_A is approximately 63,000 tonnes, whereas SWB_B is only 7.46 tonnes.

Figure 2 illustrates the relative contributions of the life-cycle stages to GWP and SWB. The major contributor to GWP was emissions from transportation energy consumption (76%). The GWP is strongly correlated with gross energy consumption, which is a measure of the total quantity of energy used in processes or activities included within the system boundary, including both combustion and precombustion energy requirements [18]. The SWB_A, which includes all solid wastes, was dominated by the waste management life-cycle stage, whereas SWB_B, with soils and sludges excluded, revealed that the majority (88%) of these solid wastes were related to transportation activities.

Toxicity impacts. Life-cycle assessments are generic by nature, lacking a specific geographic (i.e., spatial) and temporal context. The analysis is often presented by functional unit and so represents an assessment of incremental, rather than cumulative environmental burden, from which actual impacts could be calculated. Consequently, the impact assessment cannot easily draw exposure information from the inventory data.

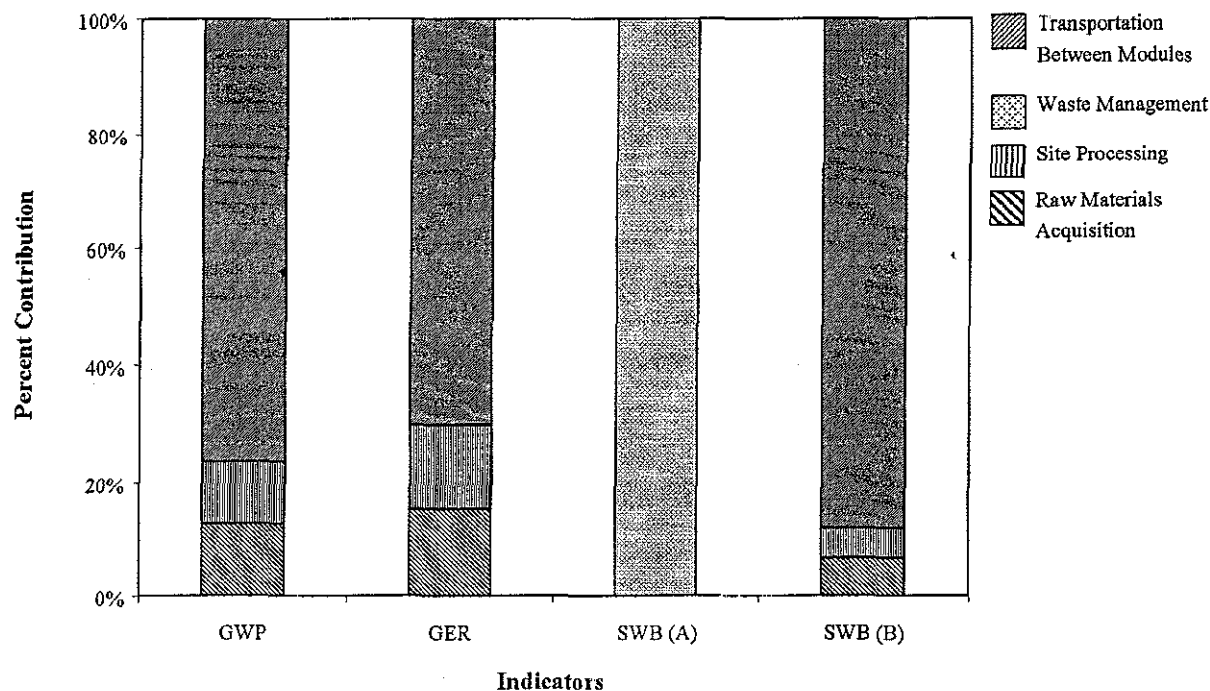


Fig 2 Distribution of indicators among life-cycle stages

for toxicologic evaluations. Therefore, LCIA is not equivalent to risk assessment, which attempts to quantitatively estimate actual, site-specific human and/or ecosystem risk. Assessing potential human and ecosystem impacts through LCIA requires numerous assumptions in order to estimate relative exposure and hence the toxicity potential of inventory items.

Many approaches exist for assessing human and environmental health impacts, including critical volume, direct valuation, health reference, and generic exposure [4,24,25]. We used a generic exposure approach similar to that proposed by Guinée and Heijungs [25] for assessing human toxicity and ecotoxicity. The approach involved estimating an exposure component using the Mackay level III multimedia model [26-29], and comparing it with an effect level. This model is suitable for estimating exposure leading to chronic or long-term effects of persistent compounds, but is not appropriate for dealing with acute toxicity or effects due to reactive, short-lived chemicals.

We have adapted the generic exposure approach in three important ways. First, the chemicals of concern were metals emitted to water after treatment (13 in Fig. 1). Because metals (with the exception of elemental mercury) are not volatile, the use of fugacity as an equilibrium criterion is not appropriate and the equilibrium criterion equivalence was used in its place [28]. Second, unlike LCA of products, for which an emission rate is difficult to determine, an emission rate was easily obtained by assuming all emissions occurred over the 75 weeks of remedial activities. Third, the geographic area of concern was southern Ontario, for which Mackay and coworkers parameterized their model [29].

Readers are referred to the literature mentioned above for details of the model. For illustrative purposes, we have examined the toxicity potential of five metals emitted to water. Table 2 contains the parameters for the main model compartments and subcompartment volume fractions [29]. Table 3 presents partition coefficients and Table 4 contains emission

Table 2 Parameters of major compartments of Mackay level III model of southern Ontario [27]

Parameters	Major compartments			
	Air	Water	Soil	Sediment
Density (kg/m ³)	1.19	1,000	1,500	1,420
Areas (m ²)	20 × 10 ¹⁰	8 × 10 ¹⁰	12 × 10 ¹⁰	8 × 10 ¹⁰
Depth (m)	2,000	50	0.1	0.01
Volume (m ³)	4 × 10 ¹⁴	4 × 10 ¹²	1.2 × 10 ¹⁰	8 × 10 ⁸
Volume fractions for subcompartments				
air	1.0	0	0.2	0
water	0	1.0	0.3	0.7
biota	0	1 × 10 ⁻⁶	0	0
solids	2 × 10 ⁻¹¹	5 × 10 ⁻⁶	0.5	0.3
Organic carbon fraction in solids				
	NA ^a	0.2	0.02	0.4

^a NA = not available

Table 3. Partition coefficients for selected metals [30,31]

	Air-water (dimensionless)	Particle-water (L/kg)	Fish-water (L/kg)	Soil-water (L/kg)	Sediment-water (L/kg)
Arsenic	1×10^{-15}	5×10^4	7.36×10^1	1.7×10^4	1×10^4
Cadmium	1.33×10^{-13}	3.3×10^5	9.26×10^2	7.8×10^2	2×10^4
Copper	8.75×10^{-14}	2×10^4	2.86×10^3	6.67×10^2	1×10^4
Lead	2.85×10^{-13}	6.68×10^5	7.69×10^2	6.67×10^4	3.33×10^5
Zinc	9.01×10^{-14}	1×10^6	8.38×10^2	1.0×10^3	5.2×10^4

rates. As expected, metals introduced to water tend to accumulate in sediment [30,31].

For human health, water concentrations are of concern because water is the main route of exposure. Estimated water concentrations were compared to a no-effect intake (NEI), in this case a subchronic oral reference dose [32-35] that was converted to a concentration assuming a 70-kg body weight and daily water intake rate of 1.5 L/d [36]. For ecosystem effects, estimated water and sediment concentrations were compared to no-adverse-effect concentrations (NECs) derived from the lowest observable effects levels (LOELs) [33].

$$IR_{ijk} = \frac{C_j}{NEI_{ijk}/70 \text{ kg} \cdot 1.5 \text{ L/d}} \text{ or } \frac{C_j}{NEC_{ijk}} \quad (1)$$

where

IR = toxicity ratio

i = receptor system considered (human, terrestrial, aquatic, sediment)

j = contaminant (e.g., As, Cd, Cu, Pb, Zn)

k = bulk compartment (e.g., water or sediment)

C = model-predicted concentration

NEI = no-effect intake

NEC = no-effect concentration

To account for differences among biological species used in toxicity testing versus relevant species in the ecosystem, we used extrapolation factors to compensate for the lack of variability in test species and number of studies [37]. We assumed complete bioavailability of metals in water and sediment, which provides a conservative estimate of potential tox-

icity [38]. We have not considered metal speciation and metal-metal interactions [39].

The results indicate that, for human toxicity and ecotoxicity, lead is of greatest concern. Lead has the greatest emission flux and partitions strongly into sediment. In addition, lead has low NECs for humans and aquatic species.

For a more complete analysis, effects due to lead emissions to air should be calculated; however, that analysis must account for the pulsed emissions that were deposited to land and water. A level III model assuming a subchronic inhalation effect would be inappropriate here. Finally, one might also consider indirect exposure of metals deposited on, or bioaccumulated in, foodstuffs [39].

Site-related impacts

Land use assessment. The movement of soil was integral to the treatment option examined here: contaminated soil was excavated and removed from the site, and clean soil was excavated from a borrow pit to fill the excavation. The main purpose of remediating the site was to render the site area acceptable for future use at the detriment of land consumed for the borrow pit and landfill (hazardous and nonhazardous) sites. In order to consider the land use and consumption associated with the entire project over its life cycle, the land area at the site, borrow pit, and landfill sites should be compared.

At the contaminated site, approximately 10,850 m² of land was remediated to acceptable residential contaminant levels. Approximately 4,130 m² of the site area was restricted through application of an impervious surface or the presence of high contaminant levels. At the hazardous and nonhazardous landfill sites, the total area required to accommodate all solid waste was 2,150 m², assuming an average depth of 12.2 m [40]. The

Table 4. Summary of inputs and results of Mackay level III model and toxicity analysis

	Arsenic	Cadmium	Copper	Lead	Zinc
Emission rate to water (kg/week)	0.077	0.021	0.180	1.215	0.173
Concentrations and distribution					
Water (g/m ³)	1.40×10^{-9}	1.82×10^{-10}	6.54×10^{-9}	8.70×10^{-9}	1.24×10^{-9}
(%)	16	8	26.7	7	6.8
Sediment (g/m ³)	3.80×10^{-5}	1.00×10^{-5}	8.90×10^{-5}	5.90×10^{-4}	8.40×10^{-5}
(%)	84	92	73.3	93	93.2
No-effect concentrations [32-34]					
Human (g/m ³)	1.4×10^{-2}	2.3×10^{-2}	2.3×10^0	3.5×10^{-2}	1.4×10^1
Aquatic (g/m ³)	4.8×10^{-3}	1.1×10^{-3}	2.3×10^{-3}	3.2×10^{-3}	1.1×10^{-3}
Sediment (g/m ³)	8.5×10^0	8.5×10^{-1}	2.3×10^1	4.4×10^1	1.7×10^2
Toxicity ratios					
Human	1.0×10^{-7}	7.8×10^{-9}	2.8×10^{-9}	2.5×10^{-7}	8.9×10^{-11}
Aquatic	2.9×10^{-8}	1.6×10^{-7}	2.8×10^{-6}	2.7×10^{-6}	1.1×10^{-8}
Sediment	4.5×10^{-6}	1.2×10^{-6}	1.0×10^{-5}	6.9×10^{-5}	9.9×10^{-6}

Table 5. Residual toxicity burden calculation data and results

	Arsenic	Cadmium	Lead
Residual contaminant concentration in soil ($\mu\text{g/g}$)			
Area 1	0.35	0.95	8.5
Area 2	4.2	1.5	475
Area 3	0.8	2.1	2,280
Typical background concentration in Ontario soil ($\mu\text{g/g}$) [32]			
	17	1	120
Toxicity based reference concentration ($\mu\text{g/g}$)			
	1.05	1.75	2,625
Residual toxicity burden			
Area 1	-16	-0.03	-42
Area 2	-12	0.29	135
Area 3	-15	0.63	823

total area required for backfill was approximately 560 m², assuming an average depth of 50 m at the borrow pit

Overall, the total area of remediated land was greater than the total area consumed elsewhere. This difference is due to the shallow depth of the contaminated site in comparison to that of the landfill and borrow pit. The use of different depths to calculate areas raises allocation issues concerning the landfill and borrow pit. The depths used were based on the assumption that the borrow pit and landfill sites were established facilities and, therefore, the maximum depths could be accessed. However, if the borrow pit were accessed at its initial phases of use, the depth to area ratio would be very low. Finally, simply summing the total "positive" remediated land area with the total "negative" land consumptions is suspect. Although the areas are known, their relative value in economic, land use, and ecological (e.g., habitat) terms was not considered.

Residual human toxicity burden. The residual toxicity burden (RTB) accounts for contaminants remaining in the soil and is intended to capture the potential toxicity to humans associated with these contaminants should they resurface (e.g., impacts from future disturbance of the site). The RTB also accounts for the difference between a site with contaminants remaining at depth and a completely clean site.

The RTB (Eq. 2) is dimensionless and similar to a toxicity ratio. It is calculated using toxicologic data such as reference dose (RfD) or risk-specific dose (RsD), depending on the contaminant, and converted to a concentration using body mass and standard values for daily intake of soil [36].

$$\text{RTB}_i = \frac{C_i - C_{\text{background},i}}{C_{\text{REF},i}} \quad (2)$$

where

C_i = mean concentration of contaminant i in soil

$C_{\text{background},i}$ = typical background soil concentration of contaminant i

$C_{\text{REF},i}$ = toxicity-based reference concentration of contaminant i derived from RsD or RfD values

Values of RTB greater than one indicate future adverse health risk with the possible reemergence of contaminants, with the level of concern being positively related to the magnitude of the value. Negative values indicate background concentrations higher than C_{REF} . Table 5 summarizes the RTB results for three areas of the case study site that differed in residual contaminant concentrations. The results indicate areas 2 and 3 pose concern for potential human health effects be-

cause of elevated lead concentrations. Soil in these areas would require further remediation should human exposure become likely through future land use (e.g., excavation). Arsenic and cadmium concentrations at the site should not cause concern should they resurface given negative values of RTB or values less than one. A similar analysis could be undertaken for terrestrial ecosystem receptors; however, suitable NECs for a variety of organisms must be determined.

DISCUSSION

Case study

The investigation of the excavation and disposal case study, through the quantitative detailing of remediation and related activities, has highlighted impacts usually hidden from traditional analyses conducted for contaminated sites such as risk assessment, with its sole focus on potential toxicity impacts. In addition to toxicity, the LCA emphasized the relative importance of energy consumption, solid waste production, and land use impacts resulting from clean-up activities. For example, energy consumption, with resultant resource depletion and air pollution impacts, was primarily due to off-site transportation. The magnitude of these impacts was a function of the mass of material hauled and distance, where three sites required consideration (hazardous and nonhazardous disposal facilities, and the borrow pit). In addition to transportation-related impacts, solid waste was a concern because of resulting land use impacts at two sites. Thus, the life-cycle analysis of these activities has emphasized the cycle of related land use disturbances from the excavation of a backfill region (i.e., borrow pit), to the transfer of clean fill to the excavation site, whose soil was, in turn, deposited elsewhere. Despite these off-site disturbances, the use of the contaminated site itself remains restricted because of pockets of contaminated soil at depth and, consequently, complete land use has not been restored at the remediated site. Because the life-cycle approach does not discount the value of various lands (e.g., particular communities, land with high or low rent potential, land with distinct aesthetic value), this assessment clearly demonstrates the disruption of three distinct areas.

In traditional applications of LCA, a final check of the inventory data consists of determining whether the overall system balances (i.e., inputs equal outputs). For contaminated sites, however, the processes considered are not continuous and balancing the inventory is not a criterion for the system boundary. Consequently, the difference between inputs and outputs provides useful information about the system. In this case, the inventory revealed net removal of groundwater from the site, reflecting local depletion of the aquifer. A second

example is that more clean materials were used to remediate the site than waste removed, indicating a net depletion of land and mineral resources

Although every attempt was made at constructing a complete analysis, several omissions in the inventory and impact assessment may limit the final conclusions drawn from this analysis. For example, many of the environmental effects resulting from the remediation method have long-term effects that were not fully captured by the analysis. Although hazardous waste containment may require limited resources and have few effects in the short term, these effects, and resources required to minimize the effects, occur over the long term. Similarly, disturbance at the backfill site and contaminants remaining at the remediated site will result in long-term impacts. No plans were in effect for restoring the asphalt cap in future, although this will undoubtedly be necessary in order to minimize potential toxicity impacts. Other inventory omissions have been noted above, and rationalized by an analysis of mass contribution (e.g., emissions related to laying asphalt, inputs and outputs associated with monitoring site excavation and backfilling). Estimating all omitted inventory items, including a nonlinear extrapolation over the 25-year time boundary, was beyond the scope of this project, and remains an area for further work. At a minimum, this exercise has resulted in the qualitative assessment of these activities, which is an improvement over assessment methods that do not evaluate impacts throughout the life cycle of an activity.

As with most LCAs, the interpretation of the inventory through the impact assessment was constrained by the limited number of stressor-impact models [18,41]. For example, further work is required to develop an indicator(s) related to land use that imparts information on habitat alteration, the viability of land to support various functions, and land stagnation [1]. We have addressed, but not adequately captured these impacts through the land use and SWB metrics. As a contribution to the impact assessment, the categories of process- and site-related stressor impacts were developed to clarify the nature and source of impacts, along with toxicity measures suitable for each category.

LCA method

This study has revealed several challenges in using an LCA-based approach for contaminated sites. In the initiation or scoping phase, determining particular activities and processes to be included within the system boundary required considerable judgment; however, the goal of the study and definitions of life-cycle stages proved useful for guiding decisions (e.g., inclusion or exclusion of decommissioning activities that were numerous and often interrelated with the soil remediation activities). The definition of the system boundary is an essential step that is necessary before the inventory component is attempted, yet it requires an in-depth understanding of all related remediation activities that often comes through conducting the inventory analysis. The time horizon of 25 years was intended to capture longer term effects; however, these effects proved difficult to estimate and were either not included or inadequately included within this study (e.g., emissions from the hazardous waste landfill).

The life-cycle stages developed by Diamond et al. [1] for contaminated site remediation activities were adequate for describing the case study. Post-site processing and monitoring were not developed because of their minor role in this study. Although monitoring played a minor role in the inventory, it

provided essential information (e.g., concentrations of contaminants in air, dustfall measures) from which inventory data were derived.

Site-specific remediation data, as mentioned, were taken largely from consultants' reports. Although the entire inventory and subsequent interpretation were reviewed by remediation consultants and a panel of experts, the process of summarizing data for the public forum, such as this paper, obscures the transparency of the data, which is an unfortunate, yet typical, feature of LCA studies relying on proprietary data. For example, certain reagents were proprietary and thus it was not possible to determine constituents, let alone inputs and outputs for production. This lack of information, although noted, contributed to gaps in the overall inventory, which again is typical of many LCA case studies. The presentation of inventory data according to the checklist format proved useful for linking inventory items to stressors and, conceptually, to their potential impacts and clarifies the distribution of the inventory items relative to life-cycle stage.

The impact assessment was restricted to a limited number of potential impact indicators that have been developed: GWP, SWB, multimedia contaminant fate and toxicity, land use assessment, and RIB. These impact indicators, although few, provide essential information to help interpret the inventory. Here, we have contributed to developing multimedia contaminant fate and toxicity estimates for metals, and have distinguished these effects from those due to contaminants remaining on-site, for which the RIB is introduced. The multimedia assessment accounts for the environmental fate of persistent contaminants and represents a more realistic evaluation of contaminant impacts than a nonfate indicator model such as critical volumes. Residual toxicity burden accounts for contaminants remaining on-site, and reflects their potential toxicity should they resurface.

In summary, this paper presents an application of an LCA-based method to site remediation activities that has highlighted important aspects associated with the remedial option. The LCA has produced a quantitative assessment upon which decisions and choices can be made and opportunities for improvement can be identified. The relative contributions of the life-cycle stages to the overall impacts of the entire remediation approach have been clarified. The case study illustrates the utility and feasibility of taking a life-cycle approach to analyze a process such as site remediation.

Overall, the case study results indicated that remediation of localized contaminated soil has resulted in burdens on local (e.g., contaminants remaining on-site, aquifer depletion), regional (e.g., land or space consumption, mineral consumption, air pollution, potential ecotoxicity and human health impacts), and global scales (e.g., acid rain, global warming, ozone depletion, energy source depletion). Thus, the effects of the excavation and disposal remediation option extend beyond the contaminated site itself, and only become evident when analyzed from a life-cycle perspective.

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REFERENCES

1. Diamond ML, Page CA, Campbell M, McKenna S, Lall R. 1999 Life-cycle framework for assessment of site remediation options: Method and generic survey. *Environ Toxicol Chem* 18:788-800
2. Canadian Standards Association. 1994 Environmental life cycle assessment. CAN/CSA-Z 760 Toronto, ON.
3. Society of Environmental Toxicology and Chemistry 1991 A technical framework for life-cycle assessment Workshop Report. SETAC, Pensacola, FL, USA.
4. Society of Environmental Toxicology and Chemistry 1993 A conceptual framework for life-cycle impact assessment Workshop Report. SETAC, Pensacola, FL, USA.
5. U S Environmental Protection Agency 1991 Selection of control technologies for remediation of lead battery recycling sites EPA/590/2-91/014. Washington, DC.
6. Diamond ML, Page CA, Campbell M, McKenna S. 1998 Life-cycle framework for assessment of site remediation options Final Report PIBS 3613E Ontario Ministry of Environment, Toronto, ON, Canada.
7. Jacques AP. 1992 Canada's greenhouse gas emissions: Estimated for 1990. EPS 5/AP/4 Environment Canada, Ottawa, ON.
8. Ontario Round Table on Environment and Economy. 1992. Sectoral task force report: Transportation ORTEE, Toronto, ON, Canada
9. Transport Concepts 1995. External costs of truck and train. Brotherhood of Maintenance of Way Employees, Ottawa, ON, Canada
10. National Energy Board. 1994. Canadian energy: Supply and demand 1993-2010 Technical Report Calgary, AB, Canada
11. Khan AM. 1991 Energy and environmental factors in freight transportation National Economic Research Branch, Ottawa, ON, Canada
12. Covery DB, Friend MJ, Shulhan GM. 1976 Energy equivalents for current and prospective automotive fuels in Canada National Research Council Canada, Ottawa, ON
13. Gerhartz W, ed. 1985. Asphalt and bitumen. In *Ullmann's Encyclopedia of Industrial Chemistry*, 5th ed Vol A3. VCH Publishers, Deerfield, FL, USA, pp 185-187.
14. U S Environmental Protection Agency. 1994. Evaluation of emissions from paving asphalt *U S EPA Control Technol Cent News* 6(3)
15. Garret IK. 1994. *Automotive Fuels and Fuel Systems*. Vol 2—Diesel. Society of Automotive Engineers, Warrendale, PA, USA.
16. McKetta J, ed. 1988. Lime: Calcination principles In *Encyclopedia of Chemical Processing and Design*. Vol 28 Marcel Dekker, New York, NY, USA, pp 119-138.
17. John Wiley & Sons 1978. Flocculating agents. In *Kirk-Othmer Encyclopedia of Chemical Technology* Vol 10, 3rd ed. New York, NY, USA, pp 496-499
18. Young SB. 1996 Assessment of environmental life-cycle approach for industrial materials and products Ph D thesis University of Toronto, Toronto, ON, Canada.
19. Meyers R, ed 1996 *Handbook of Petroleum Refining Processes* McGraw Hill. New York, NY, USA
20. McGraw-Hill. 1984 *Perry's Chemical Engineers Handbook* 5th ed. New York, NY, USA
21. John Wiley & Sons. 1992 Asphalt In *Kirk-Othmer Encyclopedia of Chemical Technology* Vol 3, 4th ed. New York, NY, USA, pp 715-716.
22. Henshaw CL. 1994 Co-product allocation in life-cycle assessment *Proceedings Annual Meeting Society of Environmental Toxicology and Chemistry—Europe*, Brussels, Belgium, February 24-25, pp 120-121.
23. Houghton JT, et al, eds 1995 *Climate Change 1994: Radiative Forcing of Climate Change and an Evaluation of the IPCC IS92 Emission Scenarios* Cambridge University Press, Cambridge, UK
24. International Life Sciences Institute. 1996 *Human Health Impact Assessment in Life Cycle Assessment: Analysis by an Expert Panel*. ILSI Health and Environmental Sciences Institute, Washington, DC
25. Guinée JB, Heijungs R. 1993. A proposal for the classification of toxic substances within the framework of life cycle assessment of products *Chemosphere* 26:1925-1944
26. Jia CQ, Di Guardo A, Mackay D. 1996 Toxics release inventories: Opportunities for improved presentation and interpretation *Environ Sci Technol* 30:86a-91a
27. Mackay D, Paterson S, Shiu WY. 1992. Generic models for evaluating the regional fate of chemicals. *Chemosphere* 24:695-717.
28. Mackay D, Diamond ML. 1989 Application of the QWASI (quantitative water air sediment interaction) fugacity model to the dynamics of organic and inorganic chemicals in lakes. *Chemosphere* 18:1343-1365.
29. Mackay D, Paterson S. 1991. Evaluating the multimedia fate of organic chemicals: Level III fugacity model *Environ Sci Technol* 25:427-436.
30. Ling H, Diamond ML, Mackay D. 1993. Application of the QWASI fugacity/equivalence model to assessing sources and fate of contaminants in Hamilton Harbour *J Great Lakes Res* 19:582-602
31. Diamond ML, Mackay D, Poulton DJ, Stride FS. 1994 Development of a mass balance model of the fate of 17 chemicals in the Bay of Quinte *J Great Lakes Res* 20:643-666
32. Livett EA. 1988. Geochemical monitoring of atmospheric heavy metal pollution: Theory and applications *Adv Ecol Res* 18:65-77
33. Ontario Ministry of Environment and Energy 1996 Rationale for the development and application of generic soil, groundwater and sediment criteria for use at contaminated sites in Ontario. Toronto, ON, Canada.
34. U.S. Environmental Protection Agency. 1992 HEAST (health effects assessment summary table) U S EPA Computer Database Washington, DC
35. U.S. Environmental Protection Agency 1993 IRIS (integrated risk information system) U S EPA Computer Database Washington, DC
36. Health Canada 1995. *Investigating Human Exposure to Contaminants in the Environment: Handbook for Exposure Calculations* Ottawa, ON.
37. U.S. Environmental Protection Agency 1984. Estimating concern levels for concentrations of chemical substances in the environment Environmental Effects Branch, Washington, DC
38. Campbell GC, Tessier A. 1996. Ecotoxicology of metals in the aquatic environment: Geochemical aspects In Newman MC, Jago CJ, eds, *Ecotoxicology A Hierarchical Treatment* Lewis, Chelsea, MI, USA.
39. Newman MC, McIntosh AW. 1991 *Metal Ecotoxicology: Concepts and Applications*. Lewis, Chelsea, MI, USA.
40. Laidlaw Environmental Services. 1995. Environmental assessment: Landfill service continuation. EA Document 4 Ottawa, ON, Canada.
41. Society of Environmental Toxicology and Chemistry. 1997 Life-cycle impact assessment: The state-of-the-art SETAC, Pensacola, FL, USA