REPORT #
RRTAC OF-7

Oil Sands Tailings

Preliminary Ecological Risk Assessment

by

Golder Associates Ltd.

Prepared for

Oil Sands Reclamation Research Program

ALBERTA CONSERVATION AND RECLAMATION MANAGEMENT GROUP
(Reclamation Research Technical Advisory Committee)

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Alberta's Reclamation Research Program

Regulating surface disturbances in Alberta is the responsibility of the Conservation and Reclamation Management Group. The Chairman is from Alberta Environmental Protection. The Group oversaw a reclamation research program, established in 1978, to identify the most efficient methods for achieving acceptable reclamation in the province. Funding for the research program was provided by Alberta's Heritage Savings Trust Fund, Land Reclamation Program. Funding ended in March of 1994.

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DISCLAIMER

This report is intended to provide government and industry staff with up-to-date technical information to assist in the preparation and review of Conservation and Reclamation Approvals, and development of guidelines and operating procedures. This report is also available to the public so that interested individuals similarly have access to the most current information on land reclamation topics.

The opinions, findings, conclusions, and recommendations expressed in this report are those of the authors and do not necessarily reflect the views of government or industry. Mention of trade names or commercial products does not constitute endorsement, or recommendation for use, by government or industry.

REVIEWS

This report was reviewed by RRTAC and the Oil Sands Reclamation Research Program committee. The report was also reviewed by Mike MacKinnon (Syncrude), John Gulley (Suncor), Tom Dereniwski (Imperial Oil) and Richard Nelson (AOSTRA).

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This project was funded by the Alberta Heritage Savings Trust Fund, Land Reclamation Program through the Alberta Conservation and Reclamation Management Group. Syncrude, Suncor and OSLO also provided funding for the study.

CHAIRMAN'S SUMMARY

This study takes the chemistry data collected from various oil sands soil-tailings mixtures (see report OF-4) and uses the values to determine the potential ecological risk that could be posed by having these materials at the surface of a reclaimed landscape. The intent of the report is to provide an overview of a methodology that could be used to help evaluate the risks posed by various reclamation options. The risks identified in this report are solely for dry landscapes; other papers prepared by industry scientists and consultants have reviewed the potential risks posed by wet landscapes.

The authors have identified, in bold face, the assumptions that went into the risk calculations. Readers should very carefully review these assumptions and understand their implications to the final results.

A copy of the detailed information on the chemical exposure limits of the various chemicals under study has been placed in the Alberta Environmental Protection library as an appendix to this report. The library is located on the 6th Floor, Bramalea Building, 9920 - 108 Street, Edmonton.

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ABSTRACT

This report outlines a conceptual and analytical framework for quantifying risks to terrestrial wildlife that might be exposed to solid-phase materials potentially associated with oil sands reclamation. The initial phase of the assessment involved screening the soil-tailings against published criteria to produce a short list of 10 constituents (8 organic and 2 inorganic) that pose a potential risk to terrestrial biota. After the 10 constituents were identified, a probabilistic model was developed that: (1) simulated exposure doses to three representative terrestrial wildlife receptors (deer mouse, white-tailed deer, American kestrel), (2) computed the probability of exceeding a chemical exposure limit for each of the receptors, and (3) summarized the relative contribution of the different exposure pathways (i.e., water and food ingestion, incidental soil ingestion, inhalation) to the total exposure dose. **Due to the paucity of data, a number of conservative assumptions were applied to this study that precluded firm conclusions with respect to potential risks associated with each of the soil-tailings mixtures. Nonetheless, the findings of this study provide useful information for directing future ecological risk assessments to assist in reclamation planning for the oil sands sites.**

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Chemical limits were developed by CanTox Inc., under the direction of Ms. Deirdre Treissman.

1. INTRODUCTION

The oil sands mining and extraction industry is currently evaluating alternatives for reclaiming large quantities of tailings generated by their operations. Some of the reclamation alternatives involve incorporating tailings into the landscape to produce a trafficable surface capable of sustaining a productive terrestrial ecosystem. A number of different options are currently under investigation for treating tailings (physical and/or chemical) to form an appropriate material for site reclamation or for disposal in pits. Industry and regulators have initiated a number of studies to examine potential impacts associated with these various reclamation materials. Recently completed studies that measured physical, chemical, and toxicological properties of several of these mixtures are of particular relevance (EMA 1993; EnviroTest 1993). In addition, a study is currently underway to evaluate the characteristics of water that might leach out of these mixtures into groundwater. Data collected from these studies plus information from the open literature can be used to provide **preliminary estimates** of risk to biota as a result of exposure to specific constituents associated with these tailings materials.

The objectives of this study are to identify those constituents associated with various tailings materials that pose a potential risk to terrestrial ecological receptors and, conversely, to eliminate from further analysis those constituents that pose little or no threat to these receptors. The methods used in this study provide a framework for quantifying ecological risks associated with alternative tailings materials. In particular, risks to terrestrial biota that might be exposed to tailings materials incorporated into near-surface, dry landscape units are assessed. Risks associated with other reclamation options, e.g., wet landscapes or in deeper in-pit disposal, or other receptors such as aquatic biota, plants and ecosystem function endpoints (e.g., nitrification, decomposition, nutrient cycling) were not evaluated. Nonetheless, the findings of this study provide preliminary information on potential risks associated with different tailings mixtures and can be used to help direct future ecological risk assessments.

2. APPROACH

This ecological risk assessment followed a phased approach (Figure 1). First, all constituents detected in the test samples were screened using chemical-specific environmental criteria to identify those that pose little or no hazard so that these constituents were not carried over to the more detailed (and time consuming) tasks of estimating exposure concentrations, calculating doses and determining species-specific toxic thresholds. This screening against published criteria was used to produce a short list of 10 constituents that pose a potential risk to terrestrial biota. **(This study was restricted to 10 constituents due to time and budget constraints).**

Three terrestrial receptors were selected for use in this risk assessment: an herbivore (white-tailed deer), an omnivore (deer mouse), and a predator (American kestrel). These receptors were selected to encompass organisms from a range of trophic levels and are common to these sites. It is recognized that this list of receptors is not comprehensive and does not include other potentially important receptors such as aquatic biota, plants, or indicators of ecosystem function, e.g., nutrient cycling in soils. Potential exposure routes for the selected terrestrial receptors that were quantified in this study include consumption of tainted water and food, incidental ingestion of contaminated soils, and inhalation of contaminated air vapours.

Exposure doses were computed using a probabilistic approach so that uncertainties in these estimates could be quantified. The estimated exposure doses were then compared to chemical exposure limits to calculate risks, where risks were defined as the probability with which a dose of a specific constituent is expected to exceed the effects threshold.

The following sections of the report outline, in detail, the approach used for the screening of constituents, exposure assessment, toxicity assessment, and risk characterization.

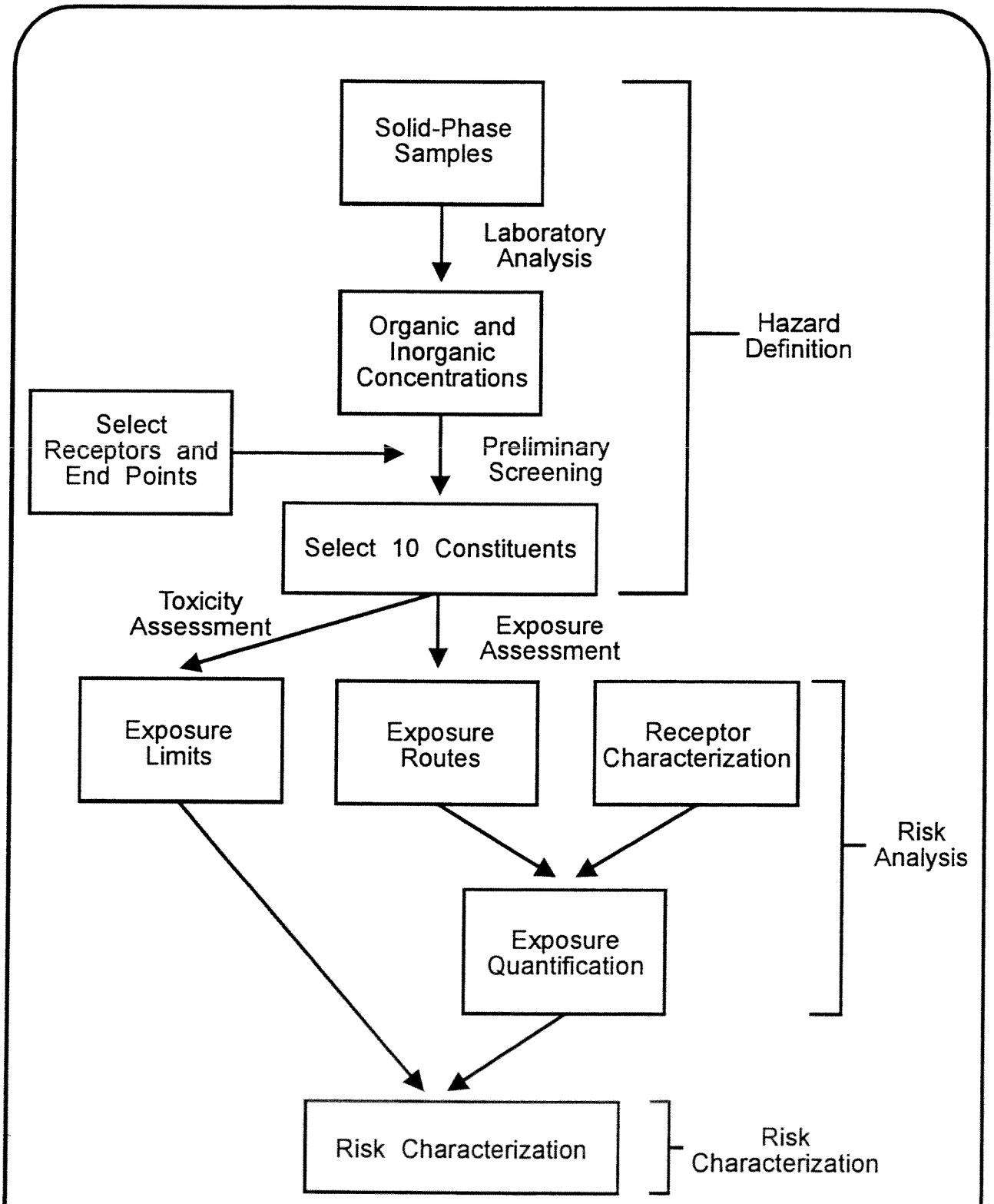

FIGURE 1 General Approach for Ecological Risk Assessment of Oil Sands Tailings

3. PRELIMINARY SCREENING OF CONSTITUENTS

The constituents considered for this investigation of ecological risk were those analyzed from 11 different natural soil, oil sands and tailings materials obtained from Syncrude, Suncor and OSLO (Table 1). The constituents include target priority pollutant PAHs, target substituted PAHs, metals and trace elements, which were analyzed from both solids phase material and leachate (Appendix 10.1; EnviroTest 1993).

The 11 samples were screened by comparing measured concentrations of organics and metals to values derived from published criteria. These Preliminary Levels of Concern (PLC) were based on the minimum values from the following published criteria:

- Alberta Environment (n.d.): Alberta Tier 1 Criteria for Contaminated Soil Assessment and Remediation; and

- CCME (1991): Interim Canadian Environmental Quality Criteria for Contaminated Sites.

Salts, nutrients and trace elements such as boron were not included in this screening as they are of less concern with respect to wildlife toxicity than organics or metals. The PLCs derived for this study are given in Table 2 (organics) and Table 3 (inorganics). The parent compound and methyl and C_n-substituted forms were added together (for all PAHs) to compute soil concentrations (Appendix 10.1). In total, 17 constituents were detected in one or more of the tailings samples at levels above the PLC, including 4 inorganic and 13 organic compounds (Table 4).

As noted in Section 2, quantification of ecological risks was limited to 10 constituents of most potential concern. We subdivided the 10 constituents to include two inorganic and eight organic constituents. This is roughly consistent with the proportion of inorganics and organics detected at

Table 1. Description of test samples.

Treatment		Description of Solid Phase Samples
This Report[a]	EMA (1993)[b]	
1		Fine Tails: Fresh (Suncor)
2		Fine Tails: Fresh (Syncrude)
3	2	Overburden Clay Shale (KCa)
4	11	Tailings Sand (Syncrude Plant 5 Beach)
5	7	Tailings Sand (Suncor Beach)
6	8	Tailings Sand + Fine Tails (Syncrude)
7	10	Fine Tails: Air Dried (Syncrude)
8	4	Fine Tails: Dry Pond 1 test pit (Suncor)
9	3	Oil Sand (Syncrude)
10		Oil Sand (Suncor)
11		Tailings Sand (OSLO OHWE)[c]

[a] Reference number used by EnviroTest (1993) and in this report.
[b] Reference number used by EMA (1993).
[c] Tailings produced at Syncrude using the Batch Extraction Unit, collected January 1993 (M. MacKinnon, 1994, person. commun.).

Table 2. Preliminary levels of concern for organic constituents (mg constituent per kg sample).

Constituent	Alberta Environment (mg/kg)	CCME (mg/kg)	Preliminary Levels of Concern (mg/kg)
Naphthalene	0.1	0.1	0.1
Acenaphthylene	0.1	-[a]	0.1
Acenaphthene	0.1	-[a]	0.1
Fluorene	0.1	-[a]	0.1
Dibenzothiophene	0.1[b]	-[a]	0.1
Phenanthrene	0.1	0.1	0.1
Anthracene	0.1	-[a]	0.1
Fluoranthene	0.1	-[a]	0.1
Pyrene	0.1	0.1	0.1
Benzo(a)anthracene/chrysene[c]	0.1	0.1	0.1
Benzo(b&k)fluoranthene	0.1	0.1	0.1
Benzo(a)pyrene	0.1	0.1	0.1
Indeno(c,d-123)pyrene	0.1	0.1	0.1
Dibenzo(a,h)anthracene	-[a]	0.1	0.1
Benzo(g,h,i)perylene	0.1	-[a]	0.1
Biphenyl(s)	0.1[a]	-[a]	0.1

[a] No criterion
[b] Set to equal criterion for non-chlorinated PAHs.
[c] Based on criteria for benzo(a)anthracene.

Table 3. Preliminary levels of concern for inorganic constituents.

Inorganic Constituent	Alberta Environment (mg/kg)	CCME (mg/kg)	Preliminary Level of Concern (mg/kg)[a]
Arsenic	10	20	10
Selenium	2	2	2
Antimony	-[b]	20	20
Mercury	0.2	0.8	0.2
Barium	400	500	400
Beryllium	5	4	4
Cadmium	1	3	1
Chromium	100	250	100
Copper	80	100	80
Lead	50	375	50
Molybdenum	4	5	4
Nickel	40	100	40
Vanadium	50	200	50
Zinc	120	500	120
Cobalt	20	40	20

[a] Most conservative criterion used.
[b] No criterion.

Table 4. Constituents detected at concentrations (mg/kg) above the preliminary level of concern (PLC).

Constituent	PLC	1 Fresh Tails Suncor	2 Fresh Tails Syncrude	3 Clay Shale	4 Tailings Sand Syncrude	5 Tailings Sand Suncor	6 Tailings Sand/ Fine Tails	7 Air-Dried Fine Tails Syncrude	8 Air-Dried Fine Tails Suncor	9 Oil Sands Syncrude	10 Oil Sands Suncor	11 Tailings Sand OSLO
Naphthalene[a]	0.1	11.76	2.1	0.49	0.01	<0.01	0.81	5.4	7.97	7	9.52	0.61
Acenaphthene[a]	0.1	0.51	1.7	<0.01	<0.01	<0.01	0.03	0.12	0.52	0.36	1.1	0.01
Fluorene[a]	0.1	2.21	0.67	<0.01	<0.01	<0.01	0.59	1.51	2.24	3.12	3.14	0.24
Dibenzothiophene[a]	0.1	96.27	22.43	0.24	0.65	0.8	12.81	68.5	140.6	83.4	110	4.98
Phenanthrene[a]	0.1	96.9	27.4	0.15	0.44	0.56	10.68	50.3	88.8	76	144.7	4.68
Anthracene[a]	0.1	95	25.3	<0.01	0.4	0.54	10.1	48.9	87	72	140	4.44
Fluoranthene[b]	0.1	4.77	1.21	<0.01	<0.01	0.01	0.15	0.83	2.09	1.01	1.69	0.06
Pyrene[b]	0.1	5.08	1.58	<0.01	0.01	0.05	0.26	1.48	4.5	2.07	3	0.1
Benzo(a)anthracene/ Chrysene[a]	0.1	18	5.3	<0.01	0.38	0.65	2.04	13.7	40.7	24.3	38	1
Benzo(b&k)fluoranthene[c]	0.1	5.9	1.14	<0.01	0.1	0.23	0.77	3.57	12	5.21	2.23	0.27
Benzo(a)pyrene	0.1	1.2	0.5	<0.01	0.03	<0.01	0.17	0.18	0.79	0.92	0.92	<0.01
Benzo(g,h,i)perylene	0.1	0.33	0.17	<0.01	0.01	<0.01	0.05	0.2	<0.01	<0.01	<0.01	0.02
Biphenyl[a]	0.1	<0.01	<0.01	<0.01	<0.01	0.01	0.11	0.48	2.31	0.73	<0.01	0.07
Arsenic	10	5.35	5.44	15.8	1.07	0.63	1.49	6.85	7.88	1.75	1.55	1.08
Mercury	0.2	0.04	0.05	0.05	0.11	0.03	<0.01	0.06	0.11	4.62	0.02	<0.01
Nickel	40	39	23	30	3	2	6	27	51	16	15	2
Vanadium	50	75.1	28.9	15.1	3.4	2.8	6.7	30.3	111	22.5	26.2	2.5

Note: Shaded box indicates exceedance of PLC.

[a] Total methyl and carbon substituted forms.

[b] Total fluoranthene and methyl fluoranthene/pyrene.

[c] Total methylated and C_n-substituted benzo(b&k)fluoranthene and benzo(b&k)fluoranthene/benzo(a)pyrene.

levels above the PLC, and simply provides a demonstration of the risk assessment protocol for a wide variety of constituents and fates. This split is not meant to imply that any one constituent or group of constituents is more or less problematic than any other.

Of the four inorganics with levels above the PLCs, mercury and nickel were selected for additional analysis as these metals are toxic to wildlife and may accumulate in both animal and plant tissue (Eisler 1987; CCREM 1987). (There is little information on toxicity and bioaccumulation of vanadium, and arsenic was detected at concentrations above the PLC only in the overburden clay shale sample).

Criteria used to select the eight organic constituents of most potential concern were based on readily available data:

1. number of samples in which exceedences over PLCs were recorded, as an indicator of the prevalence of the constituent in various mixtures;

2. ratio of measured concentration to PLC, which reflects the potential hazard of the constituents;

3. solubility, which reflects the potential mobility of the constituent from the site;

4. animal bioconcentration factor, an indicator of the potential for food-chain accumulation;

5. decay rate in soils, an indicator of the potential for long-term effects; and

6. acute toxicity to wildlife.

Relative rankings based on these criteria are given in Table 5. Sample 3, overburden clay shale, was excluded as it is a natural sample that is not indicative of tailings waste. The 13

constituents were then assigned scores, from 1 to 13, based on their position within each of the categories listed in Table 5. For example, within each column the constituent with the most deleterious attribute received the highest score (13) and the one with the least deleterious attribute received the lowest score (1). All categories were equally weighted. Thus, the sum of the scores provides an indication of the potential hazard of each constituent (Table 6). The top eight organic constituents were then carried over, along with nickel and mercury, for the detailed assessment of ecological risk. Dibenzothiophene was selected over fluoranthene on the basis of its prevalence in the tailing-soil mixtures, i.e., column one of Table 5. Hence, the following 10 constituents were selected:

- benzo(b&k)fluoranthene
- chrysene
- pyrene
- phenanthrene
- anthracene
- benzo(g,h,i)perylene
- biphenyl
- dibenzothiophene
- nickel
- mercury.

Table 5. Relative ranking for selection of organic constituents.

Number of Samples Exceeding PLC[a]	Relative Concentration[b]	Solubility[c]	BCF Animal[c]	Decay Rate (Soil)[d]	Wildlife Toxicity Acute[c]
Anthracene*	Anthracene	Naphthalene	Benzo(g,h,i)perylene	Pyrene	Benzo(b&k)fluoranthene[e]
Benzo(a)anthracene/ Chrysene*	Phenanthrene	Biphenyl	Benzo(b&k)fluoranthene[e]	Benzo(b&k)fluoranthene[e]	Benzo(g,h,i)perylene*
Phenanthrene*	Benzo(a)anthracene/ Chrysene[e]	Acenaphthene	Biphenyl	Benzo(a)anthracene/ Chrysene[e]	Benzo(a)pyrene*
Dibenzothiophene*	Dibenzothiophene	Fluorene	Fluoranthene	Benzo(g,h,i)perylene	Benzo(a)anthracene/ Chrysene[e]
Benzo(b&k)fluoranthene*	Benzo(b&k)fluoranthene[e]	Dibenzothiophene	Pyrene	Benzo(a)pyrene	Naphthalene
Naphthalene**	Pyrene	Pyrene	Benzo(a)anthracene/ Chrysene[e]	Anthracene	Acenaphthene
Fluorene**	Fluorene	Phenanthrene	Phenanthrene	Fluoranthene	Phenanthrene
Benzo(a)pyrene***	Biphenyl	Fluoranthene	Anthracene	Dibenzothiophene	Pyrene
Pyrene***	Fluoranthene	Anthracene	Fluorene	Phenanthrene	Biphenyl
Acenaphthene****	Acenaphthene	Benzo(b&k)fluoranthene	Benzo(a)pyrene	Acenaphthene	Fluoranthene**
Fluoranthene****	Benzo(a)pyrene	Benzo(a)anthracene/ Chrysene	Naphthalene	Fluorene	Fluorene**
Biphenyl	Naphthalene	Benzo(a)pyrene	Acenaphthene	Naphthalene	Dibenzothiophene
Benzo(g,h,i)perylene	Benzo(g,h,i)perylene	Benzo(g,h,i)perylene	Dibenzothiophene	Biphenyl	Anthracene

Note: Ties within a column are indicated with asterisks (e.g., naphthalene and fluorene are tied in column 1).

[a] Highest to lowest (Table 4).

[b] Based on maximum concentration ÷ PLC (Table 4).

[c] Highest to lowest (Appendix 10.2).

[d] Lowest to highest (Appendix 10.2).

[e] Based on an average of the two constituents.

Table 6. Ranking of organic constituents.

Constituent	Score	Rank
Benzo(b&k)fluoranthene*	59	1
Benzo(a)anthracene/chrysene*	53	2
Pyrene*	51	3
Phenanthrene*	48	4
Anthracene*	43	5
Benzo(g,h,i)perylene*	40	6
Biphenyl*	39	7
Dibenzothiophene*	38	8
Fluoranthene	38	8
Naphthalene	37	9
Benzo(a)pyrene	36	10
Fluorene	36	10
Acenaphthene	35	11

* Consituents selected for ecological risk assessment

4. EXPOSURE ASSESSMENT

Exposure analysis is the process of converting a source term (i.e., constituent concentration) into estimates of doses to the endpoint organisms and involves the following steps:

1. identification of significant exposure routes for the contaminants of concern;

2. selection of target species and derivation of variables required for computing exposure doses, e.g., ingestion rates, body weight;

3. measurement/prediction of contaminant concentrations in all pertinent environmental media; and

4. quantification of exposure doses to target species for contaminants of concern.

Each of these steps is described in detail below.

4.1 EXPOSURE ROUTES

Exposure routes included in this study are consumption of contaminated water and food, incidental ingestion of contaminated soils, and inhalation of contaminated air vapours. Other exposure routes such as dermal contact and inhalation of fugitive dust emissions are generally considered negligible for terrestrial wildlife. (EVS Consultants 1992).

4.2 RECEPTOR CHARACTERIZATION

Receptor characterization was directed toward identification of appropriate target organisms from three different trophic levels, i.e., herbivore, omnivore and predator. The white-tailed deer, deer mouse and American kestrel were chosen as target species for each of the trophic levels, respectively. Organisms were selected as receptors rather than broader ecosystem components, e.g., communities, because quantitative expression of observed effects can be readily derived for specific

organisms (e.g., relevant toxicity tests) compared with information required for population, community and ecosystem endpoints (e.g., abundance, biomass, diversity, energy cycling).

4.2.1 White-tailed Deer

The white-tailed deer (*Odocoileus virginianus*) was selected as the target species for the herbivore trophic level. White-tailed deer are almost exclusively herbivores, deriving the bulk of their diet from grasses, forbs, and the leaves and buds of woody vegetation. Eventually, all these plant food types could be available to deer in the dry landscape tailings sites.

White-tailed deer are typically mobile, with home ranges extending between 59 and 520 hectares (ha). Therefore, deer resident to the study area would likely spend only a portion of their time in contact with dry landscape units. However, because of their recreational importance as a big game animal, this species was selected as the target herbivore species.

The critical pathways and exposure routes for white-tailed deer on dry landscape units potentially include:

- ingestion of contaminated soil,
- ingestion of contaminated vegetation, and
- ingestion of contaminated surface water.

4.2.2 Deer Mouse

The deer mouse (*Peromyscus maniculatus*) was selected as the target species for the omnivore trophic level. The deer mouse is likely the most abundant mammal in the province (Smith 1993), and as such, provides an important prey base for predators such as weasels, hawks and owls. In Alberta, deer mice diets consist primarily of seeds and insects (Appendix 10.3). Their home ranges are small, usually ranging between 0.02 and 0.64 ha. Therefore, large numbers of deer mice could spend their entire lives within dry landscape units. Additionally, they often nest in burrows in the ground (Burt and Grossenheider 1964), and thereby could be exposed to contaminated air vapours.

The critical pathways and exposure routes for deer mice on the dry landscape tailings sites potentially include:

- ingestion of contaminated soil,
- ingestion of contaminated insects and vegetation,
- ingestion of contaminated surface water, and
- inhalation of contaminated air vapours.

4.2.3 American Kestrel

The American kestrel (*Falco sparverius*) was selected as the target species for the carnivore trophic level. Species at this trophic level are most susceptible to being exposed to toxicants that can bioconcentrate. The most important prey of the kestrel are mice (e.g., deer mice), voles and large insects such as grasshoppers. Kestrels are summer residents of Alberta and their home ranges in the province are suspected to be relatively small (about 10 to 15 ha). Therefore, several kestrels could conceivably have their entire home ranges within a single dry landscape unit.

The critical pathways and exposure routes for American kestrels on the dry landscape units potentially include:

- ingestion of contaminated soil,
- ingestion of contaminated insects and mice, and
- ingestion of contaminated surface water.

4.3 EXPOSURE QUANTIFICATION

Reclamation plans for the oil sands sites have not been finalized so it is not known how the various reclamation materials would be incorporated into dry landscape units. For example, some material may be buried below a layer of clean soil. This would reduce direct contact with plants and animals, which would likely reduce exposures through certain pathways, e.g., ingestion of soils and plants.

As a very conservative (worst-case) assumption for this preliminary ecological risk assessment, we have assumed that all material evaluated would be in direct contact with the surface environment, and that the dry landscape unit would cover 1 km^2. As reclamation planning commences, these assumptions can be changed to more accurately reflect expected conditions. Nonetheless, because the same assumption was applied to all tailings materials, the findings of this study can be used to identify: (1) the relative risks associated with the different tailings materials, (2) the chemical constituents and pathways that are of potential concern, and (3) the chemical constituents that are unlikely to pose a risk to terrestrial wildlife.

Terrestrial organisms are most likely to be exposed to contaminants through ingestion of contaminated drinking water, soil or food. Deer mice spend a substantial portion of their life underground in burrows, so it is probable that they also could be exposed to contaminated air vapours. Therefore, inhalation of air vapours was considered for deer mice. Inhalation of dust and dermal uptake was presumed to be negligible for all species.

Total daily intake (*TDI*, mg/kg-BW/d) of contaminants by terrestrial organisms was computed from:

$$TDI = EDI_{soil} + EDI_{water} + EDI_{food} + EDI_{inh} \qquad (1)$$

where EDI_{soil}, EDI_{water}, EDI_{food} and EDI_{inh} are the estimated daily intakes due to ingestion of contaminated soil, drinking water, food, and inhalation of contaminated air (for deer mouse only), respectively, in units of milligrams (mg) contaminant retained per kilogram (kg) body weight (*BW*) per day. TDI represents the average daily exposure rate expected during the adult lifetime of the receptor species. Daily intake rates for each of these routes were computed according to:

$$EDI_{soil} = \frac{C_{soil} R_{soil} F_{soil} BA_{ing}}{BW} \qquad (2)$$

$$EDI_{water} = \frac{C_{water} R_{water} F_{water} BA_{ing}}{BW} \quad (3)$$

$$EDI_{food} = \frac{(C_{plant} R_{plant} F_{plant} + C_{animal} R_{animal} F_{animal}) BA_{ing}}{BW} \quad (4)$$

$$EDI_{inh} = \frac{C_{air} R_{air} F_{air} BA_{inh}}{BW} \quad (5)$$

where BA_{ing} BA_{inh} are the bioavailability factors for ingested or inhaled chemicals, respectively, i.e., the amount of chemical retained within the body and is assumed to represent the amount of chemical available to produce adverse health effects. C_{soil}, C_{water}, C_{plant}, C_{animal} and C_{air} are the contaminant concentrations in soil (mg contaminant per kg dry weight soil), drinking water (mg/L), food (plants and animals, in units of mg contaminant per kg dry weight), and air (mg/m^3), respectively. Note that C_{animal} is broken down into either insects or mice, depending upon the receptor. Concentrations for all media were estimated as discussed in Section 4.4.

R_{soil}, R_{water}, R_{plant} and R_{animal} are the average ingestion rates of soil, water, plant tissue, and animal tissue, respectively (kg dry weight per day, except water, L/d), and R_{air} is the inhalation rate of air (m^3 per day). These rates are based on information from the scientific literature (Appendix 10.3).

F_{soil}, F_{water}, F_{plant}, F_{animal}, and F_{air} are the fractions of soil, water, vegetation, prey, and air, respectively, derived from the site over the course of a year (dimensionless). For this study, it was assumed that the various tailings mixtures were incorporated into a 1 km^2 dry landscape unit.

Exposure pathways considered in this study are summarized in Figure 2. The values of the variables for computing daily intake were set as noted in Appendix 10.3 and are summarized in Table 7.

4.4 CONTAMINANT FATE ANALYSIS

Calculation of exposure doses requires estimates of contaminant concentrations in all environmental media, i.e., C_{soil}, C_{water}, C_{plant}, C_{insect}, and C_{mouse}, as indicated in Equations (2) to (5). There are, however, few direct measures of concentrations from these environmental media. For example, a single measure of soil concentration (C_{soil}) is available for most test samples, and no direct measures of soil water (C_{water}) are available that corresponds to soil-tailings samples (acid extractions have been done on some of the samples, which provides an indication of the upper bound expected for water in equilibrium with these mixtures). No direct measures of other media are available from the study site. Therefore, a simple modelling approach was used to estimate contaminant concentrations in these environmental media. As discussed in Section 2, a probabilistic approach was utilized so that uncertainties in these estimates could be accounted for in the predicted exposure concentrations. As additional data are obtained, uncertainty in the predicted concentrations will be reduced, which might in turn reduce uncertainty in the computed exposure doses.

Contaminant concentrations measured in the test samples (C_{soil}, in mg/kg dry weight soil) were used to predict concentrations in soil water (C_{water}, in mg/L) according to:

$$C_{water} = \frac{C_{soil}}{K_d} \tag{6}$$

where K_d is the soil-water partition coefficient derived from:

$$K_d = K_{oc} f_{oc} \tag{7}$$

where K_{oc} is the constituent-specific organic carbon-water partition coefficient and f_{oc} is the fraction of organic carbon of the soil mixture (g organic C per g soil; Karickhoff et al. 1979).

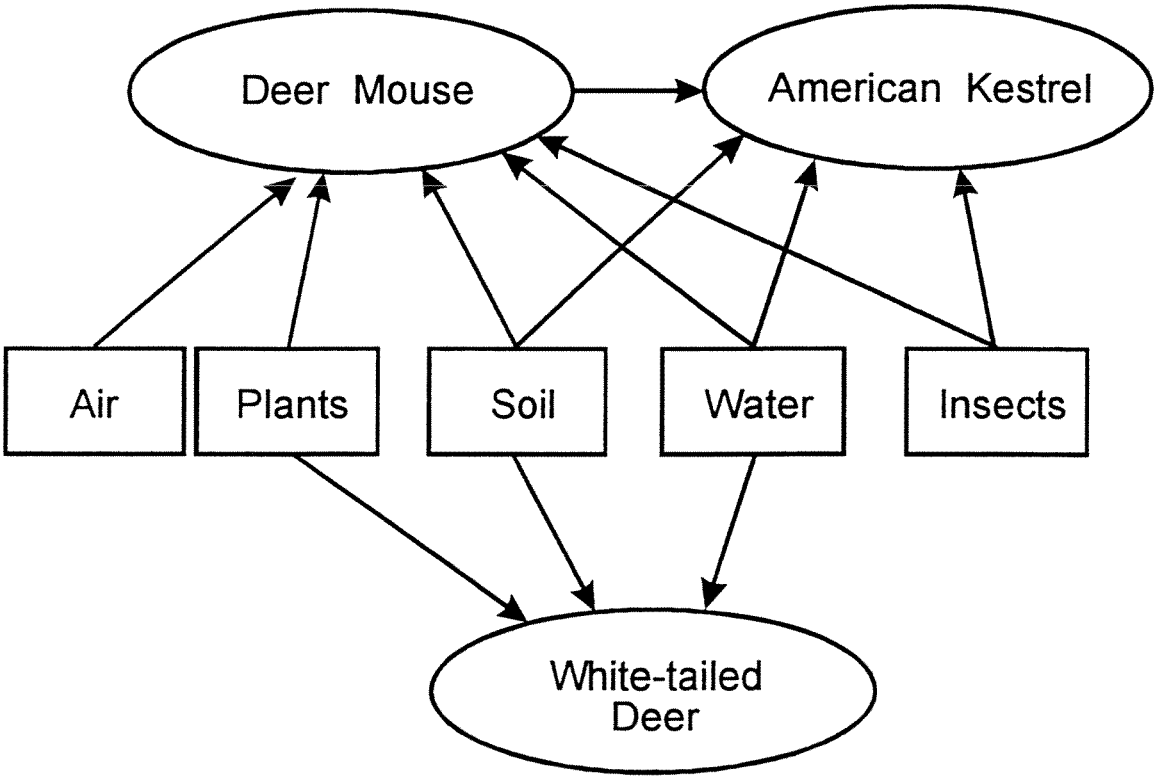

FIGURE 2 Exposure Pathways

Table 7. Receptor-specific data for computing exposure doses.

Variable	Unit	Deer Mouse		White-tailed Deer		American Kestrel	
		Distribution Type	Value(s)[a]	Distribution Type	Value(s)[a]	Distribution Type	Value(s)[a]
Body weight	kg	Normal	0.0187, 0.0043	Triangular	57,70,90	Normal	0.137, 0.0057
Number days on site per year	days	Fixed	365	Fixed	365	Fixed	130
Air inhalation rate	L/d	Uniform	8.98, 31.4	N/A	N/A	N/A	N/A
Fraction food/ water/air derived from site	dimensionless	Fixed	1.0	Uniform	0.19, 1.0	Fixed	1.0
Invertebrate prey ingestion rate	kg/d[b]	Triangular	0.0012, 0.0015, 0.0018	N/A	N/A	Triangular	0.0016, 0.0024, 0.0032
Vertebrate prey ingestion rate	kg/d[b]	N/A	N/A	N/A	N/A	Triangular	0.007, 0.0105, 0.0142
Plant ingestion rate	kg/d[b]	Triangular	0.0037, 0.0046, 0.0049	Triangular	1.4, 2.1, 3.5	N/A	N/A
Water ingestion rate	L/d	Triangular	0.001, 0.002, 0.003	Triangular	2.3, 3.5, 6.0	Triangular	0, 0.004, 0.012
Soil ingestion rate	kg/d[b]	Triangular	0, 0.001, 0.002	Uniform	0.016, 0.032	Triangular	0, 0.0023, 0.0046

[a] Normal: Mean, SD; Triangular: Minimum, Most Probable, Maximum; Uniform: Minimum, Maximum (Note: See Appendix 10.3 for sources of information.)

[b] kg dry weight per day.

We assumed that soil water would be available as a source of undiluted drinking water to wildlife, e.g., as a result of discharge via groundwater springs into ditches, wetlands, etc. This is obviously a conservative worst-case assumption as it does not account for (1) potential fates within soils (e.g., microbial decay, sorption) that might lower groundwater concentrations prior to discharge, or (2) dilution of water from uncontaminated sources, e.g., precipitation, surface runoff.

Values for C_{soil} are given in Appendix 10.1 and for K_{oc} and f_{oc} in Table 8. Since C_{soil} was based on a single measured value, we arbitrarily assumed that soil concentrations would follow a triangular distribution with the measured value representing the most probable value and the minimum and maximum values set at half and twice the measured value. Values for the partition coefficient were assumed to follow a uniform distribution with the minimum and maximum values set to encompass the range derived from: (1) values calculated with measured soil and leachate chemistry and (2) values computed based on equilibrium partitioning based on chemical and soil characteristics (Table 8).

Contaminant concentrations in the air of the soil voids (C_{air} , mg/m^3) were estimated according to an equation given by Scott and Hetrick (1993):

$$C_{air} = \frac{C_{water}\ H \cdot 10^{-3}}{R(T+273)} \qquad (8)$$

where: H is Henry's Law constant (m^3-atm/mol);
R is the gas constant (8.2 x 10^{-5} m^3-atm/mol/K);
T is the soil temperature (°C); and
10^{-3} converts C_{water} units from mg/L to mg/m^3.

It was assumed that soil temperatures follow a triangular distribution, varying from 0 to 15 °C, with a most probable value of 6 °C (Clayton et al. 1977).

Table 8. Organic carbon partition coefficients (Log K_{oc}).

Compound	Sample								Average[d]	Literature[b]
	4 Tailings Sand Syncrude	5 Tailing Sand Suncor	6 Tailings Sand/Fine Tails	7 Air-Dried Fine Tails Syncrude	8 Air-Dried Fine Tails Syncrude	9 Oil Sands Syncrude	10 Oil Sands Suncor	11 Tailings Sand OSLO		
Dibenzothiophene	7.43	7.60	7.51	7.34	7.24	7.33	7.44	8.32	7.53	3.72
Phenanthrene	7.17	7.45	5.90	6.27	6.14	6.15	6.44	6.93	6.56	3.70
Anthracene	7.22	7.43	6.24	6.45	6.46	6.35	6.57	7.11	6.73	3.69
Pyrene	5.62	6.40	4.97	5.67	5.75	5.73	5.88	6.62	5.83	4.00
Benzo(b&k)fluoranthene	6.62	7.06	6.28	6.05	6.18	6.13	5.75	7.05	6.39	5.16
Benzo(g,h,i)perylene	5.62		5.10	4.80				5.92	5.36	5.70
Biphenyl(s)		5.70	5.44	5.18	5.46	5.27		6.46	5.59	3.20
Chrysene	7.20	7.51	6.20	6.64	6.71	6.73	6.98	7.62	6.95	4.57
Nickel[c]	5.18	5.00	4.74	4.64	5.14	5.06	3.13	4.70	4.70	
Mercury[c]	3.34	2.78		3.08	3.34	4.97	2.60		3.35	
Fraction organic carbon[e] (g/g)	0.0006	0.0005	0.0100	0.0790	0.2000	0.0970	0.1000	0.0007		

[a] $C_{soil}/C_{water}/f_{oc}$; where C_{soil} and C_{water} are given by EnviroTest (1993). Leachate was not analyzed from soil mixtures 1 to 3; f_{oc} given by EMA (1993), except samples 10 and 11 (M. MacKinnon, 1994, person. commun.). Blanks indicate that both C_{soil} and C_{water} were below detection limits.

[b] Appendix 10.2.

[c] Values for nickel and mercury not corrected for organic carbon (i.e., expressed as log K_d)

[d] Input distributions assumed to be triangular with the most probably number set as the average of the observed values; the minimum value set as the minimum recorded or literature value, whichever was lower; and the maximum value set as the maximum recorded.

[e] Distributions assumed to be triangular with most probable value equal to measured value and minimum and maximum value set at 0.5x and 2x, respectively, the measured value. Distributions were set for each soil mixture, and mixtures 1 to 3 assumed similar to #7.

The approach used here to estimate exposure concentration in biota was based largely on partitioning modelling using bioconcentration factors. This is a standard approach for predicting bioaccumulation in both plants and animals (USEPA 1988). Tissue concentrations in plants were based on partitioning directly with the soil (i.e., two-phase process):

$$C_{plant} = BCF_{plant} C_{soil} \quad (9)$$

where BCF_{plant} is given in Table 9.

Bioconcentration factors calculated from plant tissue concentrations from the Syncrude site were remarkably similar to those based on structural activity relationships (SAR; Table 9). For this study it was assumed that BCF_{plant} followed triangular distributions with the most probable value equal to the SAR-computed value (Table 9) and minimum and maximum values set ± one order-of-magnitude from the most probable value. Tissue concentrations in insects (C_{insect}, mg/kg dry weight plant) were based on concentrations in the soil water (i.e., a three-phase process), with soil water computed as in Eq. (6) and C_{insect} computed as:

$$C_{insect} = BCF_{insect} C_{water} \quad (10)$$

BCF_{insect} was based on structural-activity relationships as described in Table 9. Tissue concentrations in deer mouse (C_{mouse}, mg/kg dry weight mouse) were based on partitioning between lipids and contaminant concentrations in food (i.e., a five-phase process):

$$C_{mouse} = \frac{(C_{insect} f_{insect} + C_{plant} f_{plant}) \bullet BCF_{mouse}}{L} \quad (11)$$

where f_{insect} and f_{plant} are the average annual fraction of insects and plants ingested, respectively, for deer mice, BCF_{mouse} was based on SAR predictions (Table 9) and L is the lipid content (fraction of total dry weight) of deer mouse (Appendix 10.3). For both BCF_{insect} and BCF_{mouse}, triangular distributions were assumed with the most probable value set as in Table 9 and the minimum and maximum value set at ± one order-of-magnitude from the most probable value.

Table 9 Bioconcentration factors (log *BCF*).

Compound	Plants		Invertebrates[c]	Mice[d]
	Measured[a]	SAR[b]		
Dibenzothiophene	-1.54	-1.01	3.89	-0.10
Phenanthrene	-1.12	-0.99	3.86	-0.03
Anthracene	-1.11	-0.98	3.85	0.26
Pyrene	-1.22	-1.23	4.28	-0.05
Benzo(b&k)Fluoranthene	-2.55	-1.95	5.52	0.31
Benzo(g,h,i)Perylene	-2.30	-2.59	6.63	0.44
Biphenyl(s)	-1.68	-0.059	3.16	-0.76
Chrysene	-1.67	-1.68	5.06	0.29
Nickel[e]	-0.50	N/A	2.0	1.00
Mercury[f]	-0.48	N/A	3.5	2.52

[a] Based on concentrations in wetland plants rooted in fine-tails/soil mixtures on Syncrude lease; data supplied by M. MacKinnon (Syncrude Canada Ltd.).

[b] Based on structural-activity relationship (SAR) given by Travis and Arms (1988) for terrestrial plants:

$$\log BCF = 1.588 - 0.578 \log K_{ow}$$

[c] Based on SAR reported for earthworms (Connell and Markwell, 1990):

$$\log BCF = \log K_{ow} - 0.6$$

[d] Based on mean of two SARs for rodent lipids (Garten and Trabalka, 1983):

$$\log BCF = -3.849 + 0.617 \log K_{ow}$$

$$\log BCF = 0.527 - 0.538\ (WS)$$

where *WS* is water solubility (mg/L)

[e] Little information on accumulation in terrestrial animals. *BCF* for invertebrates assumed the same as for aquatic biota (Appendix 10.2) and log *BCF* for mice set at 1.0 to account for some biomagnification.

[f] *BCF* for invertebrates assumed the same as for aquatic biota (Appendix 10.2) and for mice set at mean value measured in cattle, sheep and chicken (CCME, 1987).

4.5 EXPOSURE MODEL

A computer simulation model was used to compute exposure concentrations and doses for the three target species. All pertinent input variables to the models were expressed probabilistically, and a Monte Carlo technique was used to compute probabilities associated with exposure concentrations and doses (Appendix 10.4).

The model is a stand-alone computer program coded in C++. It is flexible to allow easy changes to input variable distributions and assumptions. Output from the model consists of: (1) doses for each contaminant, pathway and receptor, (2) total daily doses for each contaminant and receptor, (3) summary statistics for these data, and (4) probabilities of exceeding the chemical exposure limits for each contaminant and receptor.

5. CHEMICAL EXPOSURE CRITERIA

Chemical exposure limits are the daily exposure rates that could occur over a lifetime of an animal species without causing any measurable, adverse population effect. Exposure limits for the 10 constituents of concern were derived according to the rationale given in Appendix 10.5. **Briefly, limits for chemicals that exhibit a dose-response threshold (i.e., dose-response relationships are highly non-linear and the chemical does not damage genetic material) were based on a No Observable Adverse Effect Level (NOAEL) divided by an uncertainty (safety) factor. This gives a reference dose (RfD) below which no adverse effect is expected.**

Daily exposure limits for chemicals not considered to have a dose-response threshold (e.g., carcinogens) were based on a mathematical model-unit risk estimation approach. This gives a risk-specific dose (RsD) that equates to the daily dose associated with a one-in-one hundred risk of a receptor developing cancer during its lifetime. This 1:100 risk level was set lower than that typically used for humans (1-in-one-million) because: (1) the limits developed here are set to protect populations of wildlife (i.e., average individuals) rather than the most sensitive individual in a population (as is done for humans), (2) a potential loss of one percent of the individuals in a population is low relative to that lost due to natural causes and hunting pressures, and (3) a loss of one percent of individuals in a population (for the receptors used here) is not expected to cause any adverse impacts on the long-term viability of those populations.

The exposure limits and dose-response types for the 10 constituents of interest are summarized in Table 10.

Table 10. Chemical exposure limits and dose-response type.

Chemical	Exposure Limit (μg/kg body weight/day)	Type
Polycyclic Aromatic Hydrocarbons		
anthracene	500,000	RfD[a]
benzo[b&k]fluoranthene[b]	35.8	RsD[c]
benzo[g,h,i]perylene	358	RsD
biphenyl	25,000	RfD
chrysene	13.8	RsD
dibenzothiophene	62,500	RfD
phenanthrene	20,000	RfD
pyrene	37,500	RfD
Metals		
mercury	900	RfD
nickel	65	RfD

See Appendix 10.5 for derivation of limits and text for definition of terms.

[a] Reference dose

[b] Based on benzo[b]fluoranthrene

[c] Risk-specific dose

6. RISK CHARACTERIZATION

This preliminary ecological risk assessment was based on a number of conservative assumptions that will preclude the use of these results to extrapolate to ecological risks posed by specific reclamation plans. For example, all materials were assumed to be incorporated into landscapes without a capping layer of clean soil. This is an unrealistic assumption given that capping depths of up to 1.2 m would be expected at these sites. Thus, our estimates of exposure concentrations in plants and prey (insects and mice) are considerably higher than would be expected under a more realistic scenario. Additionally, the drinking water source was assumed to consist solely of contaminated groundwater. In reality, drinking waters would likely be a mixture of groundwater diluted with surface runoff and precipitation.

In addition to these conservative assumptions, there was considerable uncertainty in the predicted risks as a result of the lack of measured exposure concentrations. For instance, concentrations in all environmental media were estimated from a single measure of contaminant concentrations in the tailings mixture samples. In addition, the concentrations of the parent compound were set as the sum of the parent compound plus all C_n-substituted forms, and this "total" was assumed to have the same chemical and toxicological properties as the parent compound. Obviously, additional information on soil concentrations plus confirmation of the predicted exposure concentrations is necessary to decrease uncertainty in predicted exposure concentrations.

The probabilistic simulation method used here resulted in a large number of independent estimates of exposure doses. These estimates of exposure doses were summarized as complementary cumulative probability functions, and the probability of exceeding chemical exposure limits were determined directly from these figures (e.g., Figure 3). Probabilities are summarized for each test sample and each receptor in Tables 11 to 13.

For this study, a high-end risk was classified as one for which the 95th percentile of the exposure dose population distribution exceeded the exposure limit. This descriptor is intended to

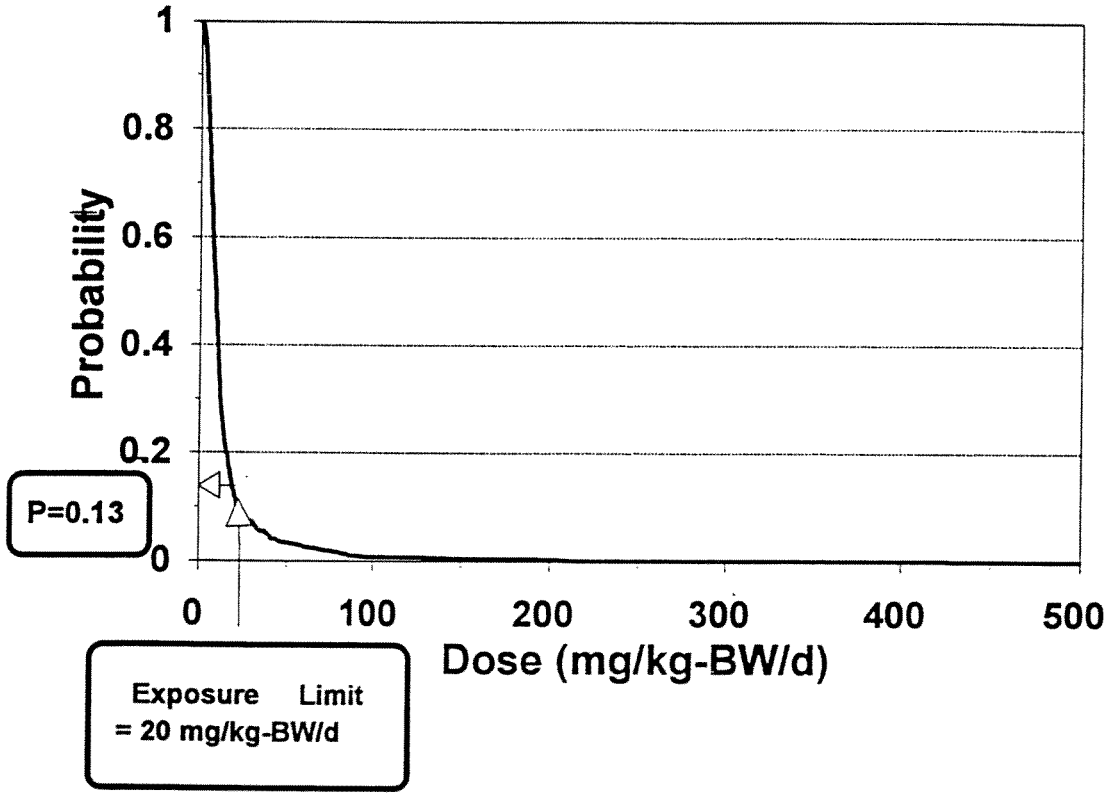

FIGURE 3 Probability Distribution for Sample 1: Dose of Phenanthrene to Deer Mice

Table 11. Probability of exceeding exposure criteria - deer mouse.

Chemical	Sample 1 Fresh Tails Suncor	2 Fresh Tails Syncrude	3 Clay Shale	4 Tailings Sand Syncrude	5 Tailings Sand Suncor	6 Tailings Sand/ Fine Tails	7 Air-Dried Fine Tails Syncrude	8 Air-Dried Fine Tails Suncor	9 Oil Sands Syncrude	10 Oil Sands Suncor	11 Tailings Sand OSLO
Dibenzothiophene	0.029	<0.002	<0.002	0.015	0.028	0.025	0.015	0.017	0.016	0.027	0.017
Phenanthrene	0.131	0.022	<0.002	0.039	0.057	0.056	0.053	0.068	0.082	0.241	0.125
Anthracene	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002
Pyrene	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.008
Benzo(b&k)fluoranthene	1.000	0.928	0.010	1.000	1.000	0.997	0.999	1.000	1.000	0.978	1.000
Benzo(g,h,i)perylene	0.902	0.762	0.038	0.995	1.000	<0.002	0.802	0.006	0.022	0.527	0.902
Biphenyl	<0.002	<0.002	<0.002	0.599	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002
Chrysene	1.000	1.000	0.011	0.997	1.000	0.998	1.000	1.000	1.000	1.000	1.000
Nickel	0.990	0.948	0.977	0.082	0.024	0.350	0.960	1.000	0.832	0.995	0.934
Mercury	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	0.095	<0.002	<0.002

Note: Shaded boxes indicate high-end risk ($P > 0.05$).

Table 12. Probability of exceeding exposure criteria - white-tailed deer.

Chemical	Sample										
	1 Fresh Tails Suncor	2 Fresh Tails Syncrude	3 Clay Shale	4 Tailings Sand Syncrude	5 Tailings Sand Suncor	6 Tailings Sand/ Fine Tails	7 Air-Dired Fine Tails Syncrude	8 Air-Dried Fine Tails Suncor	9 Oil Sands Syncrude	10 Oil Sands Suncor	11 Tailings Sand OSLO
Dibenzothiophene	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002
Phenanthrene	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002
Anthracene	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002
Pyrene	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002
Benzo(b&k)fluoranthene	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002
Benzo(g,h,i)perylene	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002
Biphenyl	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002
Chrysene	0.033	<0.002	<0.002	<0.002	<0.002	<0.002	0.021	0.191	0.064	0.174	<0.002
Nickel	0.066	0.010	0.025	<0.002	<0.002	<0.002	0.023	0.107	0.005	0.052	0.058
Mercury	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002

Note: Shaded boxes indicate high-end risk ($P \geq 0.05$).

Table 13. Probability of exceeding exposure criteria - American kestrel.

Chemical	Sample										
	1 Fresh Tails Suncor	2 Fresh Tails Syncrude	3 Clay Shale	4 Tailings Sand Suncor	5 Tailings Sand Suncor	6 Tailings Sand/ Fine Tails	7 Air-Dried Fine Tails Syncrude	8 Air-Dried Fine Tails Suncor	9 Oil Sands Syncrude	10 Oil Sands Suncor	11 Tailings Sand OSLO
Dibenzothiophene	0.003	<0.002	<0.002	<0.002	0.004	0.003	<0.002	<0.002	<0.002	<0.002	0.002
Phenanthrene	0.024	0.008	<0.002	0.014	0.021	0.020	0.004	0.011	0.014	0.031	0.020
Anthracene	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002
Pyrene	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002
Benzo(b&k)fluoranthene	0.941	0.621	<0.002	0.969	0.998	0.920	0.882	0.956	0.915	0.744	0.941
Benzo(g,h,i)perylene	0.721	0.572	0.027	0.929	0.947	<0.002	0.618	0.008	0.017	0.340	0.723
Biphenyl	<0.002	<0.002	<0.002	0.303	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002
Chrysene	0.994	0.842	0.005	0.952	0.990	0.872	0.990	0.996	0.995	1.000	0.994
Nickel	0.824	0.726	0.768	0.140	0.087	0.273	0.765	0.891	0.591	0.846	0.843
Mercury	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	<0.002	0.080	<0.002	<0.002

Note: Shaded boxes indicate high-end risk ($P \geq 0.05$).

represent the risks that might be expected to occur in small but definable segments of the receptor population, and is consistent with definitions applied by the USEPA (1992).

Total daily exposure doses to white-tailed deer are expected to be low relative to most chemical exposure limits; probabilities of exceeding the exposure limits are <0.002 for all compounds except chrysene and nickel. For chrysene, probabilities of exceeding exposure limits range from <0.002 for several samples to 0.191 for air-dried tailings from Suncor's Dry Pond 1 test pit. For nickel, the probabilities range from <0.002 in several samples to 0.107 in air-dried tailings from Suncor's Dry Pond 1 test pit (Table 12).

In contrast to white-tailed deer, estimated exposure doses to deer mouse and American kestrels are expected to exceed exposure limits for several chemicals based on the scenario simulated here (Tables 11 and 13). In particular, exposure to the carcinogens benzo(b&k)fluoranthene, benzo(g,h,i)perylene and chrysene are of potential concern as probabilities of exceeding exposure limits for these chemicals are high for virtually all samples tested, with the notable exception of the overburden clay-shale sample. In addition, doses of nickel exceed exposure limits for all samples, including the overburden clay-shale sample.

In addition to total daily exposure doses, the simulation model provided estimates of daily doses derived from the various exposure pathways (e.g., food, water, air, soil). For most test samples and receptors, the most important exposure pathway was ingestion of food. Incidental ingestion of soil was of secondary importance, and ingestion of drinking water and inhalation of air vapours were of little concern for these receptors. A representative example of the relative contribution from the different exposure pathways is given in Table 14. The only notable exception to the pattern shown in Table 14 is for the tailings sand samples from Syncrude and Suncor (samples 4 and 5, respectively). For these samples, drinking water would be a significant pathway for exposure of most chemical to white-tailed deer, but not to deer mouse or American kestrel. The cause of the high drinking water exposures was a direct result of the extremely low organic carbon content of these mixtures. This is because, as the amount of organic carbon within any particular solid-phase sample decreases, the predicted concentration in the water phase increases as is evident from Equations (6) and (7).

Table 14. Contribution of different exposure pathways to total dose (*EDI/TDI*, as %), test sample #1.

Chemical	Air	Soil	Water	Food
Mouse				
Dibenzothiophene	0.4	39.4	0.0	60.2
Phenanthrene	0.0	34.7	0.0	65.3
Anthracene	0.0	36.2	0.0	63.8
Pyrene	0.0	16.5	0.0	83.5
Benzo(b&k)fluoranthene	0.0	4.7	0.0	95.4
Benzo(g,h,i)perylene	0.0	0.2	0.0	99.8
Biphenyl	2.7	24.2	0.0	73.1
Chrysene	0.0	15.8	0.0	84.2
Nickel	0.0	32.5	0.0	67.5
Mercury	0.0	18.2	0.0	81.2
Deer				
Dibenzothiophene	0.0	6.0	0.1	93.9
Phenanthrene	0.0	6.2	0.1	93.8
Anthracene	0.0	6.2	0.1	93.8
Pyrene	0.0	10.7	0.2	89.1
Benzo(b&k)fluoranthene	0.0	38.2	0.1	61.7
Benzo(g,h,i)perylene	0.0	71.9	0.7	27.4
Biphenyl	0.0	2.6	0.2	97.3
Chrysene	0.0	24.9	0.1	75.0
Nickel	0.0	2.2	0.0	97.8
Mercury	0.0	1.9	0.1	97.9

Continued

Table 14. Concluded.

Chemical	Air	Soil	Water	Food
Kestrel				
Dibenzothiophene	0.0	21.2	0.0	78.8
Phenanthrene	0.0	15.5	0.0	84.5
Anthracene	0.0	14.5	0.0	85.5
Pyrene	0.0	4.5	0.0	95.5
Benzo(b&k)fluoranthene	0.0	1.1	0.0	98.9
Benzo(g,h,i)perylene	0.0	0.0	0.0	100.0
Biphenyl	0.0	20.8	0.0	79.2
Chrysene	0.0	3.9	0.0	96.1
Nickel	0.0	3.1	0.0	96.9
Mercury	0.0	2.7	0.0	97.3

One of the objectives of this study was to rank the samples with respect to potential risks to terrestrial wildlife. There are a number of approaches that could be used to accomplish this task; we followed a relatively simple approach whereby the number of chemicals classed as high-end risk for each soil-tailings mixture were summed for all three receptors, giving an indicator of relative risk. For example, for sample 1 (fresh tailings from Suncor), five chemicals exceeded the 0.05 probability level for deer mouse, one for white-tailed deer, and four for American kestrel, giving a total of 10 high-end risks. Note that this method of computing relative risks assumes equal weighting of all probabilities and chemicals. Based on this ranking system, the samples of greatest relative risk to terrestrial wildlife are oil sands and fresh tailings from Suncor and tailings sand from Syncrude and OSLO. The sample with the lowest risk is posed by the overburden clay-shale sample (Table 15).

Table 15. Relative rankings based on number of chemicals with high-end risks.

Rank (Worst to Best)	Score	Sample Number	Material
1	10	1	Fresh Fine Tails (Suncor)
1	10	4	Tailings Sand (Syncrude)
1	10	10	Oil Sand (Suncor)
1	10	11	Tailings Sand (OSLO OHWE)
2	9	7	Air-Dried Fine Tails (Syncrude)
2	9	8	Air-Dried Fine Tails (Suncor)
2	9	9	Oil Sand (Syncrude)
3	8	2	Fresh Fine Tails (Syncrude)
3	8	5	Tailings Sand (Suncor)
4	7	6	Tailings Sand + Fine Tails (Syncrude)
5	2	3	Overburden Clay Shale

7. CONCLUSIONS

This report outlines a framework for quantifying risks to terrestrial wildlife that might be exposed to soil-tailings mixtures associated with oil sands reclamation. A probabilistic model was developed that simulated exposure doses to three receptors (deer mouse, white-tailed deer, American kestrel), computed the probability of exceeding a chemical exposure limit, and summarized the relative contribution of the different exposure pathways to the total exposure dose.

Notwithstanding the limited data set and the conservative assumptions applied in this study, the findings of this study provide useful information for directing future ecological risk assessments. For example, risks to white-tailed deer (and probably to other herbivores) are predicted to be low. In contrast, risks to omnivores and carnivores are potentially high as a result of ingestion of tainted prey and incidental ingestion of contaminated soil. Other exposure routes such as drinking water and inhalation of air vapours will not likely pose a risk for the chemicals evaluated in this study.

8. RECOMMENDATIONS

This report presents the results of a preliminary assessment of potential risks to terrestrial wildlife associated with various soil-tailings mixtures. A number of conservative assumptions were applied to this study, and the paucity of data precludes firm conclusions with respect to potential risk to wildlife. Listed below are a number of recommendations for future work in this area that would increase the utility of risk assessment for reclamation planning:

1. The exposure doses were based on a single measure of chemical concentrations in each of the test samples. Additional samples need to be analyzed to quantify the variability of contaminant concentrations in these material.

2. The soil mixtures analyzed did not include any samples that represent potential capping material, thus, the exposure doses computed in this study could not be compared to a background sample.

3. Sensitivity analysis should be used to identify the most important input variable(s) controlling the variability in predicted exposure doses, e.g., how important is BCF_{insect} compared to the ingestion rate of insects? Future studies can then be planned to reduce uncertainty in the most important variables.

4. Exposure modelling is a valuable tool for estimating exposure concentrations. However, all exposure models are simplifications of reality, and the predicted exposure concentrations need to be compared to measured concentrations. For example, given that ingestion of insects and mice is the single most important exposure route for omnivores and carnivores, small-scale field experiments, which mimic a proposed reclamation plan, could be designed to monitor tissue concentrations in prey, to calculate *BCF*s in terrestrial plants, etc.

5. Estimates of exposure doses for each PAH were based on the total concentration (i.e. concentrations of parent compound plus substituted forms) rather than simply on measured concentrations of the parent compound. The validity of summing these different forms (with respect to chemical and toxicological properties) needs to be addressed.

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10. APPENDICES

APPENDIX 10.1 TEST SAMPLE CONCENTRATIONS

Table A.1 Organic constituent concentrations measured from the 11 samples.

Organic Analytes (μg/g)	1	2	3	4	5	6	7	8	9	10	11
Naphthalene[a]	11.76	2.1	0.49	0.01	<0.01	0.81	5.4	7.97	7	9.52	0.61
Acenaphthylene[a]	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Acenaphthene	0.51	1.7	<0.01	<0.01	<0.01	0.03	0.12	0.52	0.36	1.1	0.01
Fluorene[a]	2.21	0.67	0.05	<0.01	<0.01	0.59	1.51	2.24	3.12	3.14	0.24
Dibenzothiophene[a]	96.27	22.43	0.24	0.65	0.8	12.81	68.5	140.6	83.4	110	4.98
Phenanthrene[a]	96.9	27.4	0.15	0.44	0.56	10.68	50.3	88.8	76	144.7	4.68
Anthracene[a]	95	25.3	0.09	0.4	0.54	10.1	48.9	87	72	140	4.44
Total Anthracene & Phenanthrene	96.9	27.4	0.15	0.44	0.56	10.68	50.3	88.8	76	144.7	4.68
Fluoranthene[b]	4.77	1.21	<0.01	<0.01	0.01	0.15	0.83	2.09	1.01	1.69	0.06
Pyrene[b]	5.08	1.58	<0.01	0.01	0.05	0.26	1.48	4.5	2.07	3	0.1
Total Fluoranthene & Pyrene	5.65	1.79	<0.01	0.01	0.05	0.32	1.72	5.09	2.51	3.69	0.12
Benzo(a)anthracene/Chrysene[a]	18	5.3	<0.01	0.38	0.65	2.04	13.7	40.7	24.3	38	1
Benzo(b&k)fluoranthene[c]	5.9	1.14	<0.01	0.1	0.23	0.77	3.57	12	5.21	2.23	0.27
Benzo(a)pyrene	1.2	0.5	<0.01	0.03	<0.01	0.17	0.18	0.79	0.92	0.92	<0.01
Indeno(c,d-123)pyrene	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Dibenzo(a,h)anthracene	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01	<0.01
Benzo(g,h,i)perylene	0.33	0.17	<0.01	0.01	<0.01	0.05	0.2	<0.01	<0.01	<0.01	0.02
Biphenyl[a]	<0.01	<0.01	<0.01	<0.01	0.01	0.11	0.48	2.31	0.73	<0.01	0.07

Data from Enviro-Test (1993).
[a]Total methylated organics
[b]Total fluoranthene and methyl fluoranthene/pyrene
[c]Total methylated benzo(b&k)fluoranthene and benzo(b&k)fluoranthene/benzo(a)pyrene

Table A.2 Inorganic constituent concentrations measured from the 11 samples.

Inorganic Analytes (μg/g)	1	2	3	4	5	6	7	8	9	10	11
Arsenic	5.35	5.54	15.8	1.07	0.63	1.49	6.85	7.88	1.75	1.55	1.08
Selenium	0.1	0.09	0.74	<0.02	<0.02	<0.02	0.1	0.1	<0.02	<0.02	<0.02
Antimony	0.06	<0.05	0.06	<0.05	<0.05	<0.05	0.05	<0.05	<0.05	<0.05	<0.05
Mercury	0.04	0.05	0.07	0.11	0.03	<0.01	0.06	0.11	4.62	0.02	<0.01
Aluminum	2690	3440	10500	433	172	911	4350	5890	789	748	302
Barium	59.7	77.9	219	8.5	4.9	17.4	78.2	81.9	16.9	18.7	6.2
Beryllium	1.5	1.1	1	0.1	<0.1	0.2	1.2	2.3	0.4	0.4	<0.1
Cadmium	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3	<0.3
Chromium	6.8	15	5.1	1.4	<0.5	3.2	18.1	17.6	5.6	2	1
Copper	13.5	8.5	25.1	0.7	<0.5	2.5	17.5	12.6	3.4	2	<0.5
Iron	18100	13800	23400	2040	3350	3560	16800	18000	4320	7450	1300
Lead	9	7	10	<2	<2	<2	9	14	<2	<2	<2
Manganese	518	472	117	74.4	56.5	121	581	498	153	217	45.4
Molybdenum	<2	<2	<2	<2	<2	<2	<2	<2	<2	<2	<2
Nickel	39	23	30	3	2	6	27	51	16	15	2
Vanadium	75.1	28.9	15.1	3.4	2.8	6.7	30.3	111	22.5	26.2	2.5
Zinc	41.2	37.5	72.7	8.3	5.8	12.2	47.1	49.1	12.8	10.7	6
Cobalt	12	11	12	2	2	4	13	16	4	4	1

Data from Enviro-Test (1993).

Table A.3 Key to Calculation of PAH Concentrations Reported by Enviro-Test (1993).

Naphthalene = naphthalene + methyl naphthalenes + C2 + C3 + C4 subst'd naphthalenes
Acenaphthylene = acenaphthylene
Acenaphthene = acenaphthene + methyl acenaphthene
Fluorene = fluorene + methyl fluorene + C2 subst'd fluorene
Dibenzothiophene = dibenzothiophene + methyl dibenzothiophene + C2 + C3 + C4 substit'd dibenzothiophenes
Phenanthrene = phenanthrene + methyl phenanthrene/anthracene + C2 + C3 + C4 substit'd phenanthrene/anthracenes
Anthracene = anthracene + methyl phenanthrene/anthracene + C2 + C3 + C4 substit'd phenanthrene/anthracenes
Total phenanthrene & anthracenes = phenanthrene + anthracene + methyl phenanthrene/anthracene + C2 + C3 + C4 substit'd phenanthrene/anthracenes + 1-methyl-7-isopropylphenanthrene
Fluoranthrene = fluoranthene + methyl fluoranthene/pyrene
Pyrene = pyrene + methyl fluoranthene/pyrene
Total Fluoranthene & pyrene = fluorene + pyrene + methyl fluoranthene/pyrene
Benzo(a)Anthracene/Chrysene = Benzo(a)Anthracene/chrysene + methyl benzo(a)anthracene/chrysene
Benzo(b&k)fluoranthene = benzo(b&k)fluoranthene + methyl benzo(b or k)fluorathen/methyl benzo(a)pyrene + C2 substit'd (b or k) fluoranthene/benzo(a)pyrene
Benzo(a)pyrene
Indeno(c,d-123)pyrene
Dibenzo(a,h)anthracene
Benzo(g,h,i)perylene
Biphenyls = biphenyl + methyl biphenyl + C2 substit'd biphenyl

APPENDIX 10.2 CHEMICAL CHARACTERISTICS

Table A.4 Chemical Characterization - Organics

Compound	K_{ow} log	Solubility mg/L @25°C	Vapour Pressure mm/Hg	MW	*BCF*[a] (animals) log	*BCF*[b] (plants) log
Acenaphthene	3.92	3.57E+00	2.50E-03	154.21	2.59	-0.68
Anthracene	4.45	4.34E-02	2.67E-06	178.23	3.13	-0.98
Benzo(a)anthracene	5.66	9.40E-03	1.05E-07	228.29	2.54	-1.68
Benzo(b)fluoranthene	6.12	1.50E-03	5.00E-07	252.32	4.38	-1.95
Benzo(a)pyrene	5.97	1.62E-03	5.49E-09	252.32	2.96	-1.25
Chrysene	5.66	2.00E-03	6.23E-09	228.29	4.07	-1.68
Fluoranthene	4.95	2.06E-01	1.23E-08	202.26	3.60	-1.27
Fluorene	4.18	1.98E+00	6.33E-04	166.22	3.11	-0.83
Naphthalene	3.30	3.10E+01	8.50E-02	128.18	2.63	-0.32
Phenanthrene	4.46	1.15E+00	1.12E-04	178.23	3.25	-0.99
Dibenzothiophene	4.49	1.47[c]	2.02E-03[c]	184.27	1.65[d]	
Benzo(a)chrysene	n/a	n/a	n/a	n/a	n/a	n/a
Benzo(k)fluoranthene	6.84	0.0141	5.03E-07[e]	252	4.12[d]	
Benzo(g,h,i)perylene	7.23	0.00026[c]	n/a	276	7.06[f]	
Biphenyl	3.76	8.5[c]	2.56E-02[e]	154.19	3.96[f]	
Pyrene	4.88	1.35E+00	4.59E-06	202.26	3.43	-1.23

Values obtained from USEPA's Environmental Fate Database (ENVIROFATE) unless otherwise noted.

[a] Animal *BCF* based on concentration in tissue vs. water.

[b] Vegetation bioconcentration factor (dry wt. basis; uptake from soil), based on the regression equation given in Travis and Arms (1988); experimentally measured *BCF*s given for benzo(a)pyrene.

[c]Verschueren (1983) Handbook of Environmental Data on Organic Chemicals.

[d]CanTox (1992a).

[e]CanTox (1992b).

[f]Calculated as log $BCF = 0.893(\log K_{ow})+0.607$, from Suter (1993).

n/a - not available

Table A.5 Half Lives

Compound	Half-Lives (days)			
	Soil	Air	Surface Water	Ground Water
Acenaphthene	12.3-102	0.037-0.366	0.125-12.5	24.6-204
Anthracene	50-460	0.024-0.071	0.024-0.071	100-920
Benzo(a)anthracene	102-680	0.125-0.042	0.125-0.042	204-1360
Benzo(b)fluoranthene	360-610	0.06-0.596	0.363-30	720-1220
Benzo(a)pyrene	57-530	0.015-0.046	0.015-0.046	114-1060
Chrysene	371-1000	0.033-0.334	0.183-0.542	742-2000
Fluoranthene	140-440	0.084-0.842	0.875-2.6	280-880
Fluorene	32-60	0.284-2.8	32-60	64-120
Naphthalene	16.6-48	0.123-1.233	0.5-20	1-258
Phenanthrene	16-200	0.084-0.837	0.125-1.042	32-400
Dibenzothiophene[a]	294	n/a	n/a	n/a
Benzo(a)chrysene[b]	n/a	n/a	n/a	n/a
Benzo(k)fluoranthene	909-2139	0.05-0.46	0.16-20.8	1821-4270
Benzo(g,h,i)perylene	590-650	0.013-0.13	590-650	1168-1314
Biphenyl	1.5-7	0.325-4.6	1.5-7	3-14
Pyrene	210-1900	0.028-0.085	0.028-0.085	420-3800

Values given in Howard et al. (1991)
[a]CanTox (1992a).
[b]this compound is likely similar to Benzo(a)anthracene
n/a - not available

Table A.6 Preliminary estimate of acute toxicity to wildlife.

Compound	USFWS[a] mg/kg	Toxicity Tests[b]			
		Acute mg/kg	Chronic mg/kg	Test	Organism
Acenaphthene	-	600	-	LD50	Rat
Anthracene	-	17000	-	LD50	Mouse
Benzo(a)anthracene	-	200	-	LD50	Rat
Benzo(b)fluoranthene	-	50	5	TDlo	Rat
Benzo(a)pyrene	-	150[h]	15	TDlo	Rat
Chrysene	-	320	-	LD50	Mouse
Fluoranthene	-	2000	-	LD50	Rat
Fluorene	-	2000	-	LD50	Mouse
Naphthalene	-	400	-	LDlo	Dog
Phenanthrene	-	700	-	LD50	Mouse
Dibenzothiophene[c]	-	2650[h]	265	TDlo	Mouse
Benzo(a)chrysene[c,d]	-	200	-	-	-
Benzo(k)fluoranthene[c,e]	-	50	-	-	-
Benzo(g.h.i)perylene[c,f]	-	150	-	-	-
Biphenyl[c,g]	-	2000	-	Lethality	Rat
Pyrene	-	800	-	LD50	Mouse

NOAEL - No observable adverse effect level; LD50 - lethal dose for 50% of test population; LDlo - lethal dose for any members of test population; TDlo - lowest dose reported to produce any toxic effect.

[a] US Fish and Wildlife Service Contaminant Hazard Review Reports; lowest proposed criteria for wildlife, unless noted otherwise.

[b] Most conservative value from US National Institute of Occupational Safety and Health's RTECS database.

[c] CanTox (1992a).

[d] likely similar to benzo(a)anthracene.

[e] this compound is 10 x less toxic than Benzo(a)fluroanthene.

[f] see Benzo(a)pyrene for similar effects.

[g] biphenyls are readily metabolized by mammalian physiological systems.

[h] assumed to be 10 x higher than chronic.

APPENDIX 10.3 ECOLOGICAL AND PHYSIOLOGICAL ASSUMPTIONS FOR WILDLIFE RECEPTORS

ECOLOGICAL AND PHYSIOLOGICAL ASSUMPTIONS FOR WHITE-TAILED DEER

BODY WEIGHT

70 kg for an adult female (range between 57 and 90 kg). Average and maximum live mass reported by Stelfox (1993) and range of values derived from data given by Smith (1993) and Soper (1964).

FOOD

Ingestion Rate:

2.1 kg/day (dry weight) (range 1.4 to 3.5 kg/day).

Halls (1978) estimated the daily food consumption (dry-weight) for deer is approximately 2 to 4% of live body weight. Therefore we chose 3% as the average value with conservative minimum and maximum values of 2% and 5% respectively. For example, a 70 kg deer would consume an average of 2.1 kg of food (dry-weight) per day (70 kg x 0.03%/day = 2.1 kg/day).

Diet:

In northern Alberta, Rhude and Hall (1978) (in Stelfox 1993) reported that the diet composition of white-tailed deer varies with season as follows:

	Grasses/Forbs	Shrubs/Trees
Spring/Summer	72%	28%
Fall	18%	72%
Winter	89%	11%

Assumption: Stelfox did not supply information on summer diet composition. However, it is likely that the diet is most similar to the spring diet. Therefore, we estimated summer diet composition to be the same as spring diet composition.

Fraction of Food Derived From Site:

19% to 100%

Dryland tailings sites will be assumed to cover 1 km^2 (or 100 ha).

Average home range size = 290 ha (range 59 to 520 ha).

Home range was calculated using the equation provided in Hudson (1985:7):

$$\text{Home range (ha)} = 6.06 \times (\text{body weight in kg})^{0.91} \quad (r = 0.80, \text{estimated})$$

For example, using the above equation, the home range of a 70 kg white-tailed deer was estimated to be 290 hectares.

Smith (1991) reported the range of values for home range sizes of white-tailed deer is between 59 and 520 hectares. Interestingly, the midpoint of this range is equal to our calculated value of 290 ha for a 70 kg deer. Therefore, we believed that the most biologically meaningful and conservative approach was to have the home range equal to 290 ha with a range of 59 to 520 ha.

Therefore, at the low end of the scale, the dryland tailings site will comprise 19% of a white-tailed deer's home range (i.e., site = 1 km^2 = 100 ha; and 100 ha/520 ha home range = 19%). At the high end of the scale, the dryland tailings site will comprise 100% of a white-tailed deer's home range (i.e., site = 1 km^2 = 100 ha; and 100 ha/59 ha home range = 169%).

Assumption: Deer derive food equally from their entire home range. For example, if only 25% of a deer's home range overlaps onto a tailings site, we assumed that only 25% of their total daily intake was taken from the site.

Duration:

Assumption: Deer consume food from the site year-round.

WATER

Ingestion Rate:

3.5 L/day (range 2.3 to 6.0 L/day).

Lautier et al. (1988) reported that deer with an average weight of 54 kg consumed an average of 3.6 L/day while on a dry pelleted diet. This represents a consumption rate of 6.7% of the live body weight per day.

Nichols (1936) reported a 50% decrease in water intake for deer on a succulent diet. Therefore, 50% of 6.7% is equal to a daily intake value of 3.35% of the live body weight.

Thus, the range of values for water volume intake was estimated to be between 3.35% and 6.7% of the live body weight per day, with a midpoint of 5%. Therefore, for a 70 kg deer,

water intake would range between 2.3 and 4.7 L/day. We estimated the average intake of 3.5 L/day with a conservative range of 2.3 to 6.0 L/day.

Fraction of Water Derived From Site:

19 to 100%. See above discussion and assumptions for fraction of food derived from site.

Duration:

Assumption: Deer drink potable water from the site from April 1 to November 1.

SOIL

Ingestion Rate:

16 to 32 g/day (dry weight).

Weston (1989 in ESE 1993) reported an estimated daily intake of 16 g/day dry weight. As a conservative approach, we doubled this value to obtain a maximum figure of 32 grams/day dry weight.

16 to 32 g/day dry weight represent an intake range of 0.0178 to 0.0561% of the live body weight (i.e., 32 g/57 kg = 0.0561% and 16 g/90 kg = 0.0178%).

Assumption: Deer ingest soil from the site from April 1 to November 1.

Fraction of Soil Derived From Site:

19 to 100%. See above discussion and assumptions for fraction of food derived from site.

ECOLOGICAL AND PHYSIOLOGICAL ASSUMPTIONS FOR AMERICAN KESTREL

BODY WEIGHT

Mean weight for an adult female is 137 g (n = 73, SD = 5.7, S.E. = 0.67 g) reported in Bortolotti et al. (1991) for a population in north-central Saskatchewan.

FOOD

Ingestion Rate:

28.5 g/day mice (wet weight) (range 18.8 to 38.3 g/day).
9.5 g/day insects (wet weight) (range 6.3 to 12.8 g/day)

however, given that mice = 63% water (based on Robbins 1983;18) and that
crickets = 75% water (Rudolph 1982),

10.5 g/day mice (dry weight) (range 7.0 to 14.2 g/day)
2.4 g/day insects (dry weight) (range 1.6 to 3.2 g/day)

The equation given by Walsberg (1980) for the calculation of DEE for birds:

$$DEE = 189.3W^{0.61}$$ (Correlation Coefficient not available)

where: DEE = daily energy expenditure (kcal/day) and W = body weight in kg. Therefore, for a 137 g adult female American Kestrel DEE = 189.3 x $(.137)^{0.61}$ = 56.3 kcal/day.

Using the equation given by Aschoff and Pohl (1970) to calculate basal metabolic rate for birds :

$BMR = 73.5W^{0.734}$ (Correlation Coefficient not available)

where: BMR = basal metabolic rate (kcal/day) and W = body weight in kg. Therefore, for a 137 g bird, $BMR = 73.5(.137)^{0.734} = 17$ kcal/day. Robbins (1983:136) reported that DEE is equal to 2 to 4 times the BMR, therefore the DEE for an 137 g American Kestrel is between 34 and 68 kcal/day.

The DEEs calculated by the Walsberg equation and the Aschoff and Pohl (modified by Robins) equation match fairly well (i.e., 56 kcal/day [Walsberg equation] is close to the midpoint between 34 and 68 kcal/day [51 kcal/day]). Thus, for a 137 g bird, we assumed a DEE = 51 kcal/day with a range between 34 and 68 kcal/day.

A 20 g deer mouse (wet weight) has a gross energy value of 30 kcal (Rudolph 1982). Therefore, a deer mouse has an energetic density = 30 kcal/20 g = 1.5 kcal/g. A 0.4 g cricket (wet weight) has a gross energy value of 0.6 kcal (Rudolph 1982). Therefore, a cricket has an energetic density of 0.6 kcal/0.4 g = 1.5 kcal/g, which is identical to that of the deer mouse.

If an American Kestrel's energy requirement is 51 kcal/day, then they require 51 kcal/day ÷ 1.5 kcal/g = 34 g/day (wet weight) of either mice or crickets. However, the digestive efficiency of American Kestrels consuming either rodents or invertebrate prey was estimated to be 89% (Robbins 1983:288); therefore, the daily intake = 34 g/day ÷ .89 = 38 g/day. Consequently, we estimated the average daily intake of a 137 g adult female American Kestrel to be 38 g/day of deer mice and crickets (with a range of 25 to 51 g/day).

Diet:

75% vertebrate, 25% invertebrate diet

One study estimated the amount of invertebrate and vertebrate prey in the diet to be 75% vertebrate and 25% invertebrate prey during the nestling period (Gard and Bird 1990).

Fraction of Food Derived From Site:

100%. The mean home range reported for American Kestrels in Quebec is 13.1 ha (n=15, S.E.=1.03, Gard and Bird 1990). Therefore, it is reasonable to assume that an American Kestrel could have a home range that is 100% within a dryland tailings site of 1 km^2 (or 100 ha).

Assumptions:

1. Deer mice make up 100% of the vertebrate diet, or if they don't, the other vertebrates consumed by American Kestrels have similar energetic densities and are digested with the same efficiency.

2. If other vertebrates are consumed, they have similar BCF's for the constituents of concern as deer mice.

3. Crickets provide an accurate model for the energetic density and digestibility of all invertebrates consumed by American Kestrels.

4. The diet composition is constant over the time period spent in northern Alberta.

WATER

Ingestion Rate:

4 g/day (range 0 to 12 g/day).

We calculated the mean volume of water required for a 137 g bird to be 32 g/day based on the three equations below:

$Y = 0.111W^{0.69}$ (Ohmart et al. 1970) (r = 0.97)

$Y = 0.203W^{0.81}$ (Thomas and Phillips 1975) (r = 0.92)

$Y = 0.119W^{0.75}$ (Walter and Hughes 1978) (r not given)

where: Y = litres of water required per day and W = mass in kg.

Using W = 0.137 kg, we obtained estimates of 28, 41, and 27 mL/day for the 3 respective equations. The average of these 3 values is 32 mL/day. The average correlation coefficient was r = 0.95.

Given: mice = 63% water (based on allometric equation in Robbins 1983:18)
crickets = 75% water (Rudolph 1982)

Then, based on a 45 g/day diet consisting of:

Scenario #:	Water obtained from diet
1. 100% deer mice	28 g/day

2.	75% deer mice and 25% crickets	30 g/day
3.	50% deer mice, 50% crickets	31 g/day
4.	25% deer mice, 75% crickets	32 g/day
5.	100% crickets	34 g/day

Therefore, based on the scenarios above, the most a 137 g bird would have to consume per day in potable water would be 4 g/day or 3% of its live body weight per day. Taking a conservative approach, we tripled this figure (i.e., potable water consumption of 9% of live body wt per day) for the upper bound of the range.

Fraction of Water Derived From Site:

100%. See home range information reported above.

SOIL

Ingestion Rate:

2.3 g/day (dry weight) (range 0 to 4.6 g/day).

Assumption: Incidental soil ingestion of 0 to 5% of daily food intake. Therefore, 5% of 45 g (average daily intake of 137 g bird) = 2.3 g/day. To be conservative, we adjusted the upper bound to be 2 x as much (i.e., 4.6 g/day).

ECOLOGICAL NOTES

Duration of their stay in Alberta is approximately 130 days between April 25 and August 31 (Semenchuk 1992).

ECOLOGICAL AND PHYSIOLOGICAL ASSUMPTIONS FOR DEER MICE

BODY WEIGHT

Adult females 18.7 g (SD = 4.3)

The mean weight of an adult pre-parous female from the Kananaskis region of Alberta is 18.7 g (n = 73, SD = 4.3, S.E. = 0.5; as reported in Millar et al. 1992).

FOOD

Ingestion Rate:

May-June: 6.6 g/day insects (dry weight) (range 5.25 to 8.0 g/day)
The average annual ingestion rate of invertebrate prey is 1.5 g (dry weight) per day (range 1.2 to 1.8 g/day). The average annual rate of plant material (seeds) is 4.6 g (dry weight) per day (range 3.7 to 4.9 g/day).

The above ingestion rates for the deer mouse (*P. maniculatus*) were calculated on an annual basis using the daily intake rate (kg/day dry weight) and the proportion of the year during which the animal consumed a particular food type at a particular rate (based on data given below).

Sample Calculation:

$$\textit{Invertebrate prey} \quad \frac{(0.0066 kg/d \times 62\ d)\ (0.00165 kg/d \times 92\ d)}{365\ d}$$

$$= \quad 0.0015 \text{ kg/d (average annual intake rate)}$$

May-June: 6.6 g/day insects (dry weight) (range 5.25 to 8.0 g/day)

July-Sept.: 1.65 g/day insects (dry weight) (range 1.25 to 2.0 g/day)
7.1 g/day seeds (dry weight) (range 5.5 to 8.5 g/day)

Oct.-April: 4.8 g/day seeds (dry weight) (range 4.0 to 6 g/day)

The average daily metabolic rate (ADMR) for rodents was calculated using the following equation provided by Robbins (1983:133):

$$ADMR = 85.65W^{0.54} \qquad \text{(r not given)}$$

where: ADMR = kcal/day and W = body weight in kg. Thus, for a 18.7 g deer mouse ADMR = 10 kcal/day.

Millar (1985) calculated the amount of food required by an adult female deer mouse for maintenance and to maintain an average litter size (5 pups) from southeastern Alberta. The total energy requirements for a female supporting herself and a litter of 5 was estimated to be 35 kcal/day, which is 3.5x the ADMR estimated using Robbins' allometric equation. We believed this to be a reasonably conservative (i.e., high) estimate of total daily energy expenditure.

Based on a diet of invertebrates (crickets = 1.5 kcal/g wet weight [Rudolph 1982]) and seeds [average gross energy value of several taxa of seeds = 4.2 kcal/g dry weight (Fredrickson and Taylor 1979)], we calculated the amount of food consumed per day. However, deer mice diets vary with the time of year. For example, during spring deer mice rely heavily on invertebrates. During summer, they largely consume seeds, and some insects; and throughout winter, it believed that deer mice rely entirely on cached and gathered seeds (pers. commun. S. Sharpe, B.C. Ministry of Environment, Prince George, B.C.). Based on this information, we assumed diets composed as reported below.

Diet Composition:

May through June:	**100% insects**
July through Sept.:	**25% insects, 75% seeds**
Oct. through April:	**100% seeds**

From May through June, a deer mouse's energy requirement was estimated to be 35 kcal/day. Hence, they would require 35 kcal/day ÷ 1.5 kcal/g = 23.3 g/day (wet weight) of insects. However, the digestive efficiency of deer mice consuming either insects or seeds was conservatively estimated to be 88% (Robbins 1983:286-288); therefore, the daily intake = 23.3 g/day ÷ 0.88 = 26.5 g/day. Consequently, we estimated the average daily intake of a 18.7 g adult female deer mouse to be 26.5 g/day of insects. We set an arbitrary range of ± 20% to estimate the sensitivity of departures from the point estimate.

Between July and September, a deer mouse's energy requirement was estimated to be 35 kcal/day. Moreover, we estimated that their diet composition during this time period was comprised of 25% insects and 75% seeds. Hence, if dietary intake was 25% insects (1.5 kcal/g) and 75% grains (4.2 kcal/g), they would need to eat 5.8 g/day insects and 6.25 g/day of seeds. However, after accounting for digestive efficiency, daily insect intake = 5.8 g/day ÷ 0.88 = 6.6 g/day (wet weight) and daily seed intake = 6.25 g/day ÷ 0.88 = 7.1 g/day (dry weight). Consequently, we estimated the average daily intake of a 18.7 g adult female deer mouse to be 6.6 g/day (wet weight) of insects and 7.1 g/day (dry weight) of seeds. We set an arbitrary range of ± 20% to estimate the sensitivity of departures from the point estimates.

Between October and April, females would require energy for maintenance only, as they do not breed during this time period. Thus, they would require 17.8 kcal/day for self-maintenance (Millar 1985). By eating entirely seeds (4.2 kcal/day), they would need to eat 17.8 kcal/day ÷ 4.2 kcal/g = 4.2 g/day of seeds to meet their daily energy requirements. However, after accounting for digestive efficiency, the daily intake = 4.2 g/day ÷ 0.88 =

4.8 g/day. Consequently, we estimated the average daily intake of a 18.7 g adult female deer mouse to be 4.8 g/day of seeds. We set an arbitrary range of ± 20% to estimate the sensitivity of departures from the point estimate.

Fraction of Food Derived From Site:

100%. The home range size of the deer mouse ranges between 0.02 and 0.64 hectares (King 1968, Mullican 1988) and Banfield (1974) gives a mean home range size of 0.56 ha for Canada. Therefore, it is safe to assume that 100% of their home range will occur within a dryland tailing site of 1 km^2 (or 100 ha).

Assumptions:

1. Seeds make up 100% of the diet during winter, insects make up 100% of the diet during spring, and the July-September diet is comprised of 25% insects and 75% seeds.

2. Crickets provide an accurate model for the energetic density and digestibility of all invertebrates consumed by deer mice.

WATER

Ingestion Rate:

4 mL/day (range 2 to 6 mL/day).

4.0 g/day is the reported mean intake with a minimum of 2 g/day and a maximum of 6 g/day (King 1968). 4.0 g/day ÷ 18.7 g mouse = 21%.

Fraction of Water Derived From Site:

100%. See home range information reported above.

SOIL

Ingestion Rate:

1 g/day (dry weight) (range 0 to 2 g/day).

No information was obtained with respect to soil ingestion rates by deer mice. However, 5% of the live body weight per day was deemed to be a conservative estimate. We set an arbitrary range of $\pm$100% of this value to estimate the sensitivity of departures from the point estimate.

Fraction of Soil Derived From Site:

100%. See home range information reported above.

RESPIRATION

Daily Respiration Rate = 480 to 1,680 mL air/g-day

8.976 to 31.4 L/day based on 19 kg weight.

For mice with an average weight of 19 g,

ADMR = 4.2 mL O_2/g-h (French et al. 1976:204)

= 100.8 mL O_2/g-day

Converting to ml air/g-day:

1 ml air = 0.21 mL O_2

Daily Respiration Rate = 480 mL air/g-day

Total energy requirements for an adult female supporting herself and a litter of 5 was estimated to be 3.5x the ADMR (see FOOD section). Therefore, a meaningful upper range is:

3.5 x 480 mL air/g-day = 1,680 mL air/g-day

LIPID COMPOSITION OF PEROMYSCUS

non-breeding female 4.2% (s.d. = ±0.008%)
pregnant female 3.7% (s.d. = ±0.072%)
lactating female 2.4% (s.d. = ±0.403%)

Body fat composition of Peromyscus is required to calculate contaminant tissue concentrations. Millar (1975) calculated the body fat of non-breeding, pregnant and lactating females as follows:

reproductive condition	sample size (n)	mass (g)	body fat (g)	standard deviation (s.d.)
non-breeding	10	21.6	0.912	0.00018
pregnant	5	22.3	0.832	0.0161
lactating	4	20.1	0.486	0.081

ECOLOGICAL NOTES

Peromyscus maniculatus is active throughout the year in Alberta (Robinson and Bolen 1989).

APPENDIX 10.4 MONTE CARLO METHOD

Computer simulation models are tools designed to represent a simplified version of reality. Water quality models can, in theory, predict water quality conditions for a particular system, based on the system's physical properties coupled with chemical and biological processes that are known to occur in surface water environments. The reliability of model predictions depends upon how well the model approximates field conditions. However, simplifying assumptions must always be made to construct a model because field situations are much too complex to be simulated exactly. For example, traditional mathematical water quality models have been developed on the basis that the behaviour of surface water is deterministic. These deterministic models operate with the assumption that model parameters can be described fully by a unique set of values estimated from a limited set of field data. This implies that the water quality conditions for a particular system can be assessed definitively.

Deterministic models are well suited to applications in which they are used to explore the implications of various management decisions on systems for which they have been calibrated, particularly when uncertainties in model processes are small or negligible. They are less well suited for exploring implications of alternative assumptions about other systems, for projecting water quality conditions far into the future, or for predicting the effects of perturbations to the system. In addition, these models do not account for the inherent uncertainty resulting from random characteristics in the physical, biological and chemical processes that affect water quality.

Over the last decade models have been developed to take into account the uncertainties associated with water quality processes. These uncertainties arise from a number of sources, including errors in measuring (or lack of measuring) these processes and the randomness of the value of water quality processes, i.e., water quality processes are stochastic, that is, they may vary as a function of time or space or both, and these variations are partially or fully governed by the laws of chance.

Probabilistic models include an assessment of the effects of these uncertainties in model parameters on the confidence that can be placed on model output (Figure 1). Analysis of this variability is particularly important in a management context as it helps establish error bounds on the predictions, thus, the value of the information provided by these models can be assessed.

Two types of methods have been used in estimating errors in water quality models; analytical and numerical methods (e.g., Monte Carlo techniques). Analytical methods, such as first-order error analysis, are best suited for relatively simple models and they require much less computational time than numerical models. However, the analytical models require some simplifying assumptions about parameter distributions and cannot be integrated into a complex water quality model, such as that used here. Monte Carlo methods, on the other hand, are relatively simple to program but require considerable computational time.

Monte Carlo sampling consists of repeating simulations of the model a large number of times (iterations). Each iteration consists of a unique set of input values, which are specified by sampling the input parameters at random from their assumed probability distributions. This results in a large number of random simulations, each providing a unique set output values. The output is then examined statistically to define prediction error.

A critical aspect of probabilistic modelling is definition of appropriate probability distributions for input parameters. For some parameters, distributions can be defined accurately from evaluation of generic or site-specific data, e.g., rainfall, streamflow, etc. For other parameters where there are few or no data and/or high natural spatial and temporal variability, definition of distributions are more subjective. A variety of techniques have been developed to assist in defining distributions for such parameters. These techniques range from self-assessment by a single analyst to a formal evaluation with a group of experts (Roberds 1990).

A probabilistic model requires definition of the input parameters in the form of probability distributions. These distributions can take many different forms, ranging from a simple uniform

$$X = X_0 e^{-k(T)t}, \text{ where } k(T) = k(20^{\circ}C)\,\theta^{(T-20)}$$

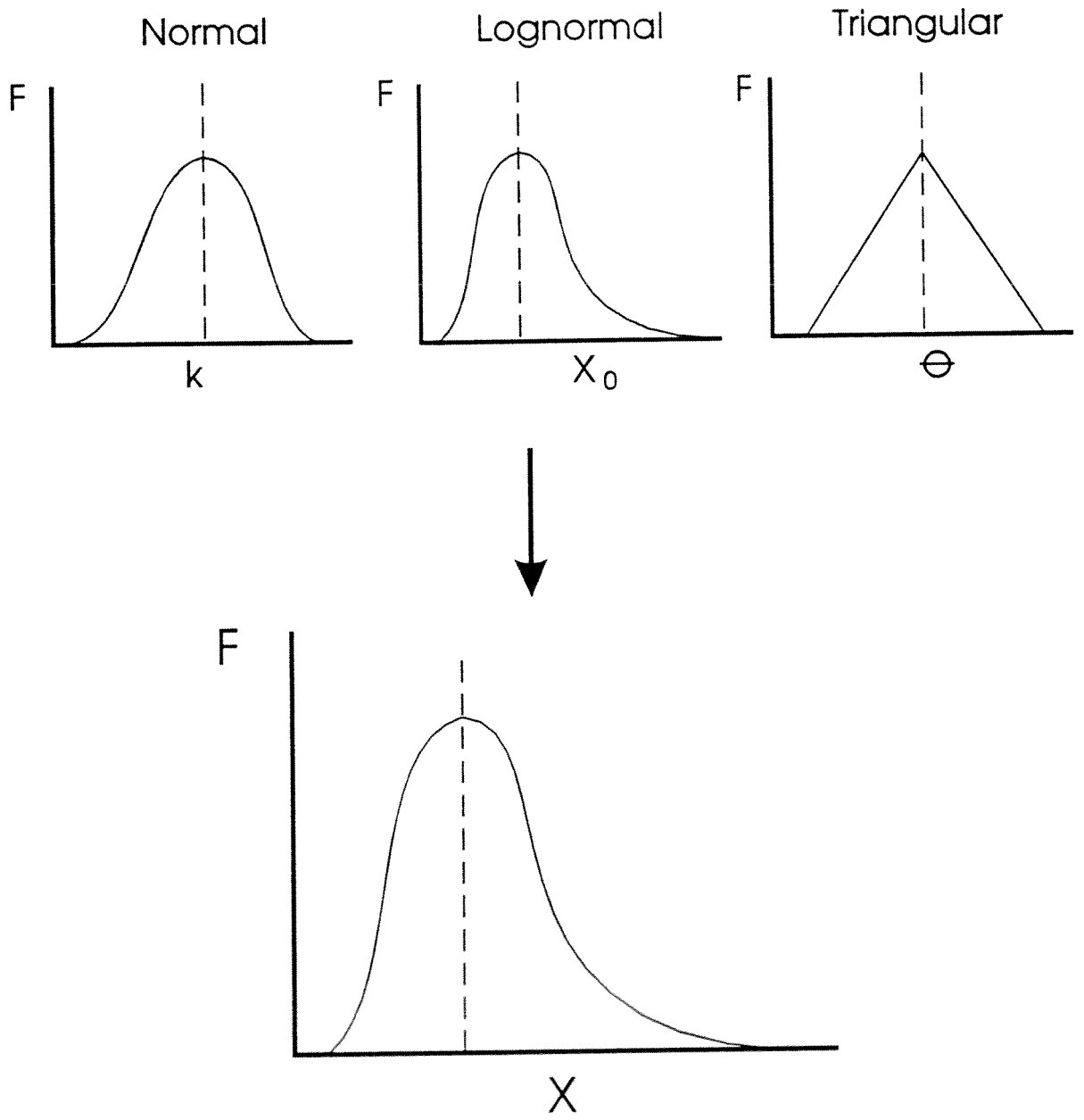

Figure 1 Example of a First-order Temperature Dependent Decay Model, Deterministic vs Probabalistic Methods.

distribution defined only by a minimum and maximum value to more complex ones like a truncated log-normal distribution, where log-normally distributed values are constrained within a specific range. In most cases, there is adequate information only to define simple uniform or triangular distributions (where minimum, most probable and maximum values are specified).

In addition to defining distributions for each of the sensitive parameters, it is important to determine whether there are correlations among the parameters. For example, within a single water body, mineralization rates (biodegradation of organic matter to inorganic forms) for phosphorus and nitrogen might be expected to be correlated as they are both a function of characteristics of the organic matter and the bacterial community within that water body. The effect of such correlations may be to either increase or decrease model output uncertainty, and correlation tends to assume increasing importance as the magnitude of the error in model inputs increases (Brown 1990).