**Assessment of the Significance of Emissions During *Ex-situ* Remedial Activities
compared to Landfill Disposal
2012 Update**

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Meridian Project No. 11441.1

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Executive Summary

Under the authorization of the Petroleum Technology Alliance Canada (PTAC), Meridian Environmental Inc. (Meridian) has conducted an assessment of the environmental significance of vapour emissions during *ex-situ* remediation activities. The work was conducted under the Alberta Upstream Petroleum Research Fund (AUPRF), under the direction of the Canadian Association of Petroleum Producers (CAPP) and the Small Explorers and Producers Association of Canada (SEPAC), as indicated in a letter of approval from PTAC dated August 24, 2010, and a subsequent letter of approval dated February 3, 2012. The following report documents the results of the assessment, and has been updated to reflect work conducted in 2012.

The purpose of this study was to conduct a life cycle analysis of emissions from *ex-situ* remediation as compared to landfill disposal for soils contaminated with volatile petroleum hydrocarbons (e.g. condensate). Specifically, the study involves the development of a modelling approach that can be used to evaluate potential health and environmental effects from toxic emissions, effects on air quality from criteria air contaminants, and greenhouse gas emissions. *Ex-situ* remediation and landfill disposal are compared using approaches as similar as possible in order to ensure the comparison is meaningful. The objective of the project is to be able to meaningfully assess the relative human health and environmental impacts of the two approaches in order to determine whether *ex-situ* remediation may be an appropriate approach at a specific site.

A literature review was conducted to identify relevant models to estimate emissions of volatile organic chemicals (VOCs), criteria air pollutants and particulates from both *ex situ* remediation and landfill disposal. The most appropriate available model for each process was selected based on scientific basis, defensibility, regulatory acceptance, applicability to Canadian conditions, and ability to meet project objectives.

The selected models were combined into a single recommended modelling approach capable of predicting life cycle emissions of all emissions evaluated, as well as short-term emissions of VOCs for the evaluation of human health and environmental effects using regulatory air dispersion models. A test of the modelling approach was conducted using data from an example site where ambient air monitoring had been conducted.

Work conducted in 2012 included:

- Further research into the use of biofilters with ex situ remediation, including evaluating the lifecycle emissions from ex situ remediation with and without biofilters, including consideration of the production and fate of the biofilters; and the effects of biofilters on ambient air quality.
- Collection of real-world emissions data from ex situ remediation to calibrate and confirm the modelling assumptions, in particular from highly contaminated soils.
- Development of a simple spreadsheet-based calculation tool for screening-level evaluations using the modelling approach described herein.

1   INTRODUCTION

1.1   General

Under the authorization of the Petroleum Technology Alliance Canada (PTAC), Meridian Environmental Inc. (Meridian) has conducted an assessment of the environmental significance of vapour emissions during *ex-situ* remediation activities. The work was conducted under the Alberta Upstream Petroleum Research Fund (AUPRF), under the direction of the Canadian Association of Petroleum Producers (CAPP) and the Small Explorers and Producers Association of Canada (SEPAC), as indicated in a letter of approval from PTAC dated August 24, 2010, and a subsequent letter of approval dated February 3, 2012. The following report documents the results of the assessment, and has been updated to reflect work conducted in 2012. The activities conducted in 2012 are also documented separately (Meridian, 2012).

1.2   Background

Onsite *ex-situ* soil remedial technologies involve the excavation of contaminated material and treatment onsite, potentially resulting in the release of volatile chemicals to ambient air. Volatile emissions resulting from remedial activities may contribute to greenhouse gas emissions through fugitive emissions (Valsaraj et al., 1994), as well as potentially impacting ambient air for humans or sensitive ecological receptors on or near remedial sites. Other potential chemicals of concern may also be emitted from soil as a result of biological processes during soil treatment. Fugitive emissions, whether from industrial processes or remedial efforts in the oil and gas industry are often subject to considerable uncertainty surrounding their quantity and composition, in part due to the inconsistent use of measurement systems, and the limitations therein. Additionally, fugitive emissions from remedial activities are often not considered at all due to measurement limitations, as well as the inconsistency of emissions and the deemed negligible permanent effects. Exposure to volatile contaminants that workers, nearby residents, or sensitive ecological receptors may receive during remedial soil processes had not been quantified. Currently, data gaps exist about the risks receptors may be receiving.

The United Nations Framework Convention on Climate Change (UNFCCC) is a binding international legal instrument directly dealing with climate change. Under this framework Canada, an Annex I Party, must report a national annual inventory of human-induced greenhouse gas emissions that are consistent with the intergovernmental panel on climatic change (IPCC) guidelines. Within the IPCC guidelines, fugitive emissions from remedial activities are not specifically addressed; however, fugitive emissions from the oil and natural gas industry as a whole are addressed (Section 4.2, Fugitive Emission, IPCC, 2006) and remedial emissions are grouped under venting 'disposal of waste associated gas at oil production facilities' (IPCC, 2006). Tier 3, the most comprehensive, bottom-up sum of emission estimates considers source type at the individual facility level including: venting, flaring, fugitive equipment leaks, evaporation losses and accidental leaks. Currently, there is no guidance given or methods recommended for use in quantifying emissions from remedial activities; furthermore vented volumes are considered highly suspect since these values are typically estimates and not based on actual measurements. Currently the IPCC guidelines state that audits and reviews should be conducted by industry representatives to determine if all vented and flared volumes are actually reported, that the reported volumes are actually vented or flared, or to develop appropriate apportioning of venting relative to flaring. Forthcoming ERCB directives are also expected to require reporting of these emissions.

As an Annex I Party member, Canada is required to continuously improve the quality of our national greenhouse gas inventory. Fugitive emissions from the energy industry accounted for approximately 10.6% of total emissions in 2007, not including any releases from *ex-situ* remediation (Environment Canada, 2007). Emissions from *ex-situ* remedial activities have been recognized as contributing significant impacts to ambient air, and accepted technologies to minimize volatile losses have been recommended for over a decade (US EPA, 1997). However, there are also emissions associated with other soil remediation processes, including landfill disposal.

There are three general approaches to estimating exposure concentrations in air: (1) ambient air monitoring, (2) emission measurements coupled with dispersion modelling, and (3) emission modelling coupled with dispersion modeling (USEPA, 2006). Mine surface emissions in oil sands exploration have been monitored by both Syncrude and Suncor using soil emission flux chamber measurements to calculate site-specific emission factors. This surface emission estimation accounts for area, time exposed, ambient temperature, and surface type; however, this estimation differs from remedial emissions as soil handling during remedial activities aims to increase the volatility of contaminants. The development of a scientifically defensible model to quantify remedial emissions during *ex-situ* activities will allow upstream oil and gas to more accurately estimate their emissions, and define the associated risks.

1.3   Purpose/Objectives

The purpose of this study was to conduct a life cycle analysis of emissions from *ex-situ* remediation as compared to landfill disposal for soils contaminated with volatile petroleum hydrocarbons (e.g. condensate). Specifically, the study involves the development of a modelling approach that can be used to evaluate potential health and environmental effects from toxic emissions, effects on air quality from criteria air contaminants, and greenhouse gas emissions. *Ex-situ* remediation and landfill disposal are compared using approaches as similar as possible in order to ensure the comparison is meaningful. The objective of the project is to be able to meaningfully assess the relative human health and environmental impacts of the two approaches in order to determine whether *ex-situ* remediation may be an appropriate approach at a specific site.

1.4   Scope of Work

The scope of work was based on a proposal submitted by Meridian on July 23, 2009, modified based on subsequent discussions with the 2010 CAPP project manager, Steve Kullman of Husky Energy Inc. and members of the AUPRF Steering Committee.

In general, the scope included the following components:
- Conduct a literature review to identify and assess existing models used to estimate emissions to ambient air from *ex-situ* remediation and landfill disposal, and to identify key factors that would influence contaminant volatility and modelling outcomes.
- Conduct a critical review of available models based on applicability, scientific defensibility and regulatory acceptance.
- Develop a modelling approach for evaluating volatile and particulate emissions and air concentrations from *ex-situ* remediation, and life-cycle emissions of volatile organic compounds (VOCs), criteria air contaminants and greenhouse gasses from *ex-situ* remediation and landfill disposal.
- Determine appropriate default model input parameters.

- Document the results of the project in a report.

The work scope was expanded in 2012 to include the collection of site data to calibrate the model, research into lifecycle emissions from biofilters, and development of a user spreadsheet for the model. This work is detailed in a separate report (Meridian, 2012); key results have been incorporated into this report.

2   METHODS

2.1   General Approach to Life Cycle Assessment

Life cycle assessment is often applied as a technique to assess each and every impact associated with all the stages of a particular process. One key aspect in this application is that it compares potential environmental impacts of different options for providing the same service, in this case remediation of a site to meet guideline levels. Life cycle assessment is usually a comprehensive tool, covering a wide range of environmental impacts, which enables the estimation of direct as well as indirect environmental impacts related to the remediation activities being investigated. Its strengths are rooted in the basic engineering principles of material and energy balances, where some form or accounting is usually conducted. This can be achieved by compiling an inventory of relevant environmental releases, and evaluating the potential associated impacts.

Common categories of assessment in environmental life cycle assessment include greenhouse gases, acidification, smog, ozone layer depletion, eutrophication, energy use, toxicological pollutants, habitat destruction, desertification, land use, and depletion of mineral and fossil fuels. It is typical to categorize energy-environmental considerations in terms of various products of energy operations that flow into and potentially harm the environment. One way that energy systems interact with the environment is through the release of solid, liquid and gaseous phase by-products that are known to be hazardous to human health or to threaten ecosystems. Material by-products of energy consumption are often referred to as pollutants, hazardous emission, effluents and waste and are the focus of this life cycle assessment.

A life cycle assessment typically considers both primary and secondary impacts, where primary impacts are those local toxic impacts related to remedial activities, and secondary impacts are created due to the remediation process as well as upstream and downstream processes. Tertiary impacts refer to post-remediation phase impacts, such as site reuse restrictions, and will not be considered in this assessment due to a high level of associated uncertainty.

Often, to meet remedial objectives, and protect receptors from potential exposure, a volume of contaminated soil is disposed in a landfill disposal facility. Onsite *ex-situ* soil remedial technologies are alternative methods that involve the excavation of contaminated material and treatment onsite, resulting in the release of volatile chemicals and particulates to ambient air. These remedial emissions may affect ambient air quality in the vicinity of the site and contribute to overall emissions of greenhouse gases, total particulate emissions, and volatile organic compounds (VOCs). However, landfill disposal also generates and contributes to the emissions of greenhouse gases, total particulate emissions and volatiles; VOCs may also be released during excavation and transport.

A literature review of environmental life-cycle assessments related to remedial technologies indicates that landfill emissions are generally neglected as part of the assessment due to the

lack of data, or short time frame of the assessment; however, transport of equipment, material and soil to and from the site is generally considered (Lemming *et al.*, 2010). Often the focus of these *ex-situ* life assessments in the literature, although based on the life cycle principles, is to facilitate remedial comparisons and only considers energy use and impacts during the remediation project and related emissions. As the focus of this current report is the comparison, in a life cycle approach, of landfill and *ex-situ* remedial emissions the consideration of landfill emissions is required.

To complete this assessment a life cycle approach was used considering assessment damages that included the accounting of predicted pollutant emissions including greenhouse gases, and criteria air contaminants. To simplify the comparison, only aspects of the techniques that were quantifiable, comparable and specifically related to the remediation processes were considered. The assessment was considered a 'gate to grave' assessment and the environmental impacts of fuel production, equipment production, preceding site visits etc. were not considered, but rather the time scale began when remedial activities began on-site.

2.2   Contaminant Fate Modelling

Contaminant fate and transport models have historically been used in a variety of applications including risk assessment, analysis of remedial system performance, cost-benefit assessment and contaminant life cycle analysis (Benson et al., 1993; Poulsen et al., 2001). Models used in these applications attempt to simulate the chemical and physical processes affecting both the release and movement of multi-compound chemical mixtures. To enable a quantifiable comparison, kinetic or mass transfer equations are often applied because of their conceptual and mathematic simplicity. However, more complex modelling is often required to make estimations with defined timescales. The environmental effects associated with petroleum hydrocarbons are encountered at different length and times scales, so the various categories of human health and other adverse impacts are often divided into a more manageable length-time scale. For the purposes of this life cycle assessment, the time scale considered begins when remedial activities commence and concludes when the mass of contaminant is completely removed from the soil media. Environmental impacts assessment will benefit by disaggregating impacts of interest according to more defined processed of their impacts, as a single modelling platform would have little hope of reliably capturing all the effects (Lemming, 2010).

One approach is to solve the equations that govern the transport and transformation of pollutants to compute the contribution of a given source to the concentration of a pollutant at its impact location (Seinfeld and Pandis, 1998). Applying a chemical mass balance, which allocates a mass through the chemical analysis of pollutants measured at the source and can assume all or a portion of that mass is transferred to another media, is also frequently done in life cycle assessments (Tester, 2005). Greenhouse gas emissions models 'work' by solving the mathematic equations that express the laws of conservation of material and energy, and are therefore similar in theory to mass balance models where the mass in one media is assumed to be transferred to another media, but in the case of greenhouse gas models the mass is calculated as carbon dioxide equivalents. Mass balance, greenhouse gas and equipment emission modelling are the tools used to enable a quantifiable comparison in this life cycle approach of overall impacts from these two remedial scenarios.

While it is recognized that modelling also introduces uncertainty and that not every factor can be reduced to a number and inserted into a model, the use of modelling in this application is not

necessarily to provide a precise outcome value, but rather to allow for a relative comparison between the two scenarios. The application of mathematical models to comparative situations is often criticized, since studies can easily be swayed in favour of one product of process over another based on varying parameters, assumptions and differences in available data. To counter this, equivalent assumptions were made. The development of a modelling approach to enable a relative comparison is the basis of the work presented herein.

2.3   Atmospheric Fate of Petroleum Hydrocarbons

Contaminants found in the soil, representing a variety of compounds with varying carbon number, released into the atmosphere through volatilization will undergo additional transformation. Compounds released to air will have varying half-lives, and chemical specific transformation, often dependent on the availability of other compounds. To provide some context for the lifetimes of volatile compounds of main concern to human health a brief review of the atmospheric fate of key petroleum hydrocarbon compounds (benzene, ethylbenzene, toluene and xylenes) is provided.

Benzene released to the atmosphere will be found in the vapour phase, and reacts with photo-chemically produced hydroxyl radicals, with a predicted half-life of 13.4 days. However, in the presence of nitrogen oxides or sulphur dioxide, as would be expected at contaminated sites, this half-life is accelerated to 4-6 hours (US EPA, 2002b). The range of estimated half-lives for benzene is from 0.1 to 21 days, with photo-oxidation by-products including phenol, nitrophenol, nitrobenzene, formaldehyde, carbon dioxide and carbon monoxide, depending on the oxidant (Government of Canada, 1993a).

Ethylbenzene released to the atmosphere will exist predominately in the vapour phase, and also be broken down through photochemical degradation by reaction with hydroxyl radicals (US EPA, 1991). Ethylbenzene has also been found to undergo transformation reacting with NO_3 radicals and atomic oxygen (ATSDR, 2010). The predicted half-life ranges from 0.5 to 2 days through reaction with hydroxyl radicals (US EPA, 1991; ATSDR, 2010). Oxidation by-products from the reaction with hydroxyl radicals and nitrogen include ethylphenols, benzaldehyde acetophenone and nitro-ethylbenzenes (ATSDR, 2010), which are then themselves degraded further.

Toluene released to the atmosphere will also degrade by reacting with photo-chemically produced hydroxyl radicals yielding cresols, benzaldehyde, and a number of other products that are then themselves degraded further. The minimum half-life for toluene has been calculated to be 4.5 hours, but northern latitudes in winter are expected to result in longer half-lives (Government of Canada, 1992).

Xylenes in gas phase in the atmosphere will also photo-oxidize relatively quickly in a reaction with hydroxyl radicals yielding tolualdehydes, methyl glyoxal, methylbenzylnitrate, dimethyphenols and nitroxylenes, which over time are themselves degraded further. The range of lifetimes for xylenes has been estimated to be from 0.5 to 1.5 days (Government of Canada, 1993b).

The fate of these four contaminants, constituent components of residual condensate contamination, suggests that once released to air the predicted transformation times generally range from hours to days. Less data are available regarding the atmospheric fate and half-lives of other compounds found in volatile petroleum hydrocarbon mixtures. However, it is predicted

that photochemical oxidation and/or biological metabolism will play a role in the decomposition of such compounds as they have similar oxidative and enzymatic access points for cleavage; the time horizons for these activities are predicted to be similar. Overall, these volatile hydrocarbon compounds are not predicted to remain in the atmosphere longer than a few weeks before transformation to a breakdown product, which will also be further degraded. The resulting breakdown products either undergo additional atmospheric oxidation, or settle and undergo biological transformation. Either process will result in the eventual release of the constituent carbon as carbon dioxide or carbon monoxide; carbon monoxide eventually oxides atmospherically to carbon dioxide (Australian Government, 2009). Therefore, the carbon contained within contaminants released to the atmosphere is expected to eventually form carbon dioxide, and will be accounted for as such.

2.4   Processes Considered and Conceptual Models

For purposes of modelling, a conceptual model is required for each scenario to represent the processes and sources involved.

2.4.1   *Ex-situ Remediation*

Ex-situ remedial activities are varied but all include the physical disturbance of contaminated media for treatment. Soil disturbance will significantly increase the surface area of contaminated material to ambient air; often the aim of ex-situ remediation is aiding volatile contaminant partitioning from soil to air. Soil disturbance will also transform a significant portion of the contaminated media from anaerobic to aerobic conditions, affecting soil microbial carbon metabolism. Native microbial communities metabolize organic carbon sources, including hydrocarbons, often in cascade-type molecule breakdown. During these processes, substances that were previously non-volatile may be degraded to semi-volatile or volatile compounds and released to ambient air.

Ex-situ remediation aims to remove contaminant mass from soil media through the physical manipulation of the contaminated soil volume, encouraging partitioning from soil to air by exposing soil particle surfaces to air. Increasing soil exposure to air through physical manipulation, is the minimum requirement to be considered *ex-situ* remediation, however, this is often coupled with other techniques to encourage volatilization. The main techniques often used in combination with physical soil disturbance include biological treatment, chemical reduction or oxidation, and thermal desorption; for a more detailed overview of these techniques see Appendix A.

Ex-situ remediation is considered to involve the following processes:
- Mobilization/demobilization of equipment to the site.
- Excavation of contaminated soils.
- Onsite treatment of contaminated soils and replacement in the excavation.
- Long-term *in situ* degradation of residual contamination.

2.4.2   *Landfill Disposal*

As with the *ex-situ* remedial scenario, only residual contaminant mass contained within the soil media for the contaminant life-cycle will be reviewed, and the process and transport emissions will be considered using mathematic modelling and presented below.

Hydrocarbon contaminants disposed in a landfill will undergo both aerobic and anaerobic biological transformation. The rate of gas production is a function of waste composition, age of waste (time since emplacement), climate, moisture content, particle size, compaction, nutrient availability, and buffering capacity (USEPA, 2005). Depending on these variables biochemical processes can produce a variety of volatile organic compounds in varying volume; it can therefore be difficult to predict the specific gas species that will be produced, or the volume generated especially for a class II or industrial landfill where little gas generation data is available. However, landfills will experience distinct phases in contaminant breakdown, as seen below.
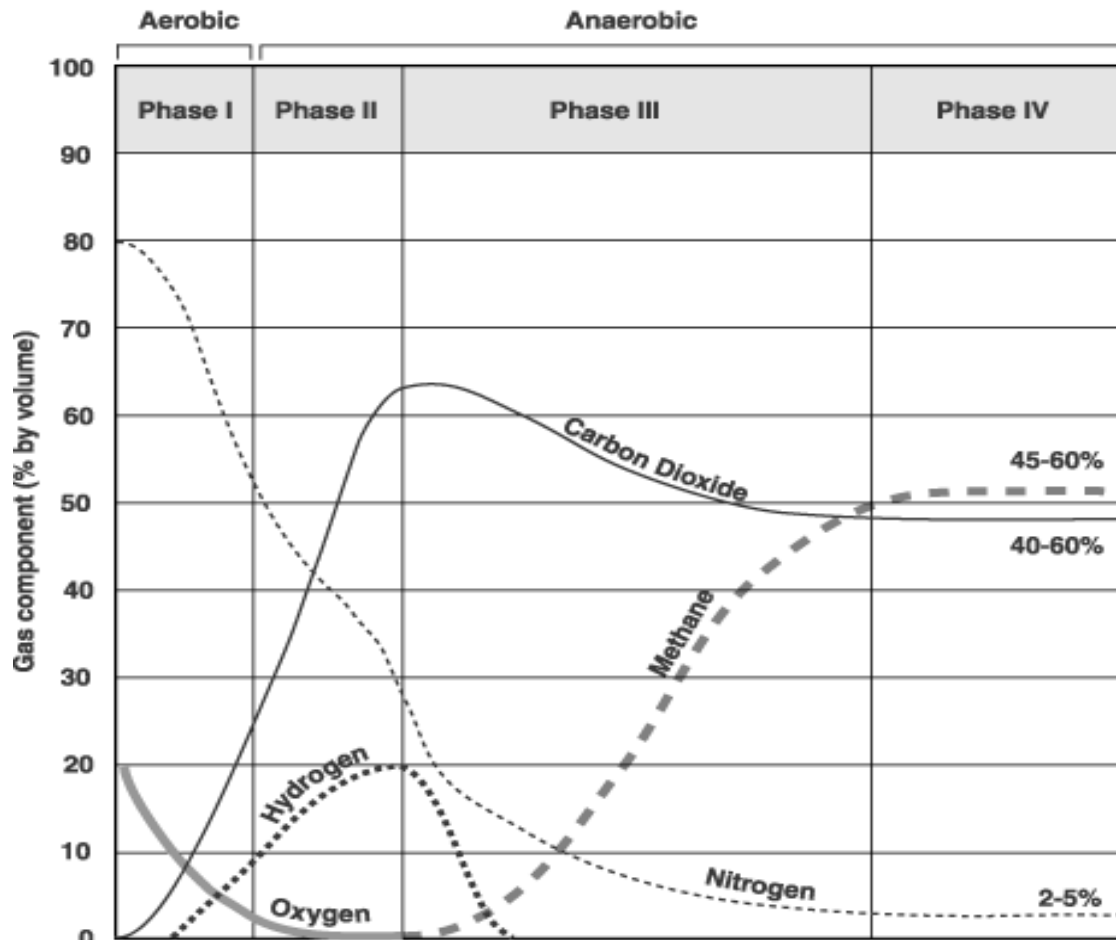
Note: Phase duration time varies with landfill conditions

Source: EPA 1997

Figure 1. Landfill gas evolution

Nitrous oxide, despite having a high greenhouse gas potential, is not an expected breakdown product of hydrocarbon degradation. Nitrous oxide is primarily produced from the bacterial breakdown of nitrogen in soils, and expected to be primarily released during the first phase as a result of soil spreading. Nitrous oxide, as a greenhouse gas, is not considered in the contaminant life-cycle.

Under aerobic conditions most soil microbes can use oxygen as an electron acceptor and release carbon dioxide into the atmosphere, where denitrifiers released nitrous oxide under anaerobic conditions. Long-term anaerobic conditions will deplete major oxidants resulting in methanogen proliferation and the release of methane (Changsheng, 2007). Both nitrous oxide (298x) and methane (25x) have a higher global warming potential when compared to carbon dioxide (1x), the by-product of aerobic metabolism (IPCC, 2007).

Anaerobic degradation of petroleum hydrocarbons by microorganisms has been shown to occur at negligible rates in comparison to aerobic breakdown, however in the absence of oxygen studies have demonstrated the breakdown of saturated and aromatic hydrocarbon compounds, with methane as a by-product (Leahy and Colwell, 1990; Heider et al., 1999; Widdel and Rabus, 2001). This is relevant to landfill disposal, as the carbon mass contained in hydrocarbons disposed of in a landfill will most likely be transformed through biological processes and be released to the atmosphere, over the life-time of the contaminant, as methane.

Most hydrocarbons disposed in landfill are predicted to breakdown during the last phase, or the steady methanogenic phase, however it is unclear what proportion of the waste would be emitted as carbon dioxide or methane. Landfill disposal facilities monitored during the methanogenic phase, emit landfill gas that comprises roughly 45-60% methane (CH_4) (IPCC, 2001; ESRD, 2008). Methane and carbon dioxide are the primary constituents of landfill gas but oxygen, nitrogen gas, and water vapour are also produced (US EPA, 2005). An overview of landfill gas composition during the methanogenic stage of the landfill can be seen in Table A.

Table A. Typical Landfill Gases Production by Volume and Characteristics

Component	% Volume	Characteristic
CH_4	45 to 60	Methane is a naturally occurring, colourless, and odourless gas. Its concentration in ambient air of ~0.0002%. Landfills are the single largest source of man-made methane emissions
CO_2	40 to 60	Carbon dioxide is a colourless and slightly acidic gas that occurs naturally at a small concentration (0.03%) in the atmosphere
N_2	2 to 5	Nitrogen gas comprises approximately 79% of the atmosphere. It is odourless, tasteless, and colourless.
O_2	0.1 to 1	Oxygen gas comprises approximately 21% of the atmosphere. It is odourless, tasteless, and colourless.
Ammonia	0.1 to 1	Ammonia is colourless gas with a pungent odour. Atmospheric concentrations are less than 0.0001%
Non-methane organic compounds	0.01-0.6	Non-methane organic compounds, excluding methane may occur naturally or be formed by synthetic chemical processes.
Sulfides	0 to 1	Sulfides (e.g., hydrogen sulfide, dimethylsulfide, mercaptans) are naturally occurring gases that give the landfill gas mixture its rotten egg smell. Sulfides can cause unpleasant odours even at low concentrations. Ambient air concentrations are less than 0.001%
Hydrogen	0 to 0.2	Hydrogen gas is an odourless and colourless gas. Atmospheric concentrations are less than 0.001%.
CO	0 to 0.2	Carbon monoxide is an odourless and colourless gas. Atmospheric concentrations are less than 0.00001%.

US EPA, 2005

Overall, the fate of the carbon contained within the hydrocarbon mass disposed in a landfill will be emitted as either carbon dioxide or methane and therefore regarded as having a higher greenhouse gas potential than *ex-situ* emissions, which are not predicted to generate methane emissions. However, the appropriate fraction of emissions predicted as methane from hydrocarbon waste is unknown as most published information on the fraction of methane in landfill gas relates to waste with a large volume of degradable organic carbon like municipal and agricultural wastes. Industrial contaminants, including petroleum contaminants, differ in their susceptibility to biological decomposition; often taking decades to degrade through cascade type biochemical reactions involving numerous microbial species, and certain high molecular weight aromatics may not break down at all (Das and Chandran, 2010). Based on the time scale of degradation, it is expected that most, if not all, degradation will take place during landfill end-stage anaerobic methanogenic processes. Most predictions of roughly half the composition of the landfill gas comprising methane may underestimate methane emissions strictly from hydrocarbon contamination but in the absence of appropriate class 2 landfill data, or industrial landfill emission data, this value may be appropriate for adoption.

Landfill disposal is considered to include the following processes:
- Mobilization/demobilization of equipment to the site.
- Excavation of contaminated soils.
- Transport of contaminated soils to a landfill and placement in the landfill.
- Long-term degradation of contamination at the landfill.

2.5   Defining Pollutant Emissions

A review of life cycle assessments in the literature related to *ex-situ* remedial techniques reveals a difference in their fundamental approach and goals (Lemming, 2010). Some life cycle assessments appear to be a more detailed retrospective approach, while the approach taken in this report is to be more prospective, intending to serve as a decision-support in the choice between different remedial options and relies primarily on a modelling approach. The goals of a life cycle assessment are often defined by a functional unit, which in this case will be considered as the remedial volume of soil treated to effectively remove all contaminant mass. The removal of all contaminant mass may not be accomplished in *ex-situ* remediation, since only the achievement of applicable soil remediation guidelines is sought. However, remaining contaminant mass may continue to attenuate in soil after the remediation activities are completed. The impact categories considered include pollutants released to air including criteria air contaminants and greenhouse gases.

2.5.1   Volatile Organic Compounds

Volatile organic compounds (VOCs) are considered to include chemicals that contain carbon and hydrogen with boiling points roughly in the range of 50-250°C. This definition includes thousands of naturally occurring and synthetic chemicals. VOCs expected to be encountered during remedial activities at a site with residual natural gas condensate include primarily volatile aromatic and aliphatic hydrocarbons. These compounds are categorized in CCME (2008) as petroleum hydrocarbon sub-fractions 1 and 2 and consist of:

Fraction 1, $C_6 - C_{10}$:
- Benzene, toluene, ethylbenzene and xylenes (BTEX)
- Aromatics $C_{>8}-C_{10}$

- Aliphatics C_6-C_8, $C_{>8}$-C_{10}

Fraction 2, $C_{>10} - C_{16}$:
- Aromatics $C_{>10}$-C_{12}, C_{12}-C_{16}
- Aliphatics $C_{>10}$-C_{12}, $C_{>12}$-C_{16}

The present work focuses on natural gas condensate in soil. However, the same approach for quantifying the carbon in the contaminant mass can be applied to other volatile petroleum hydrocarbons. Only volatile compounds that can partition into air are considered; while it has been shown that *ex-situ* techniques encourage aerobic metabolism and breakdown of larger chain hydrocarbons influencing volatility (Leahy and Colwell, 1990), data gaps surrounding this process make it difficult to quantify and large chain compounds, which are non-volatile and are not considered herein.

2.5.2   *Criteria Air Contaminants*

Environment Canada defines criteria air contaminants as nitrogen (NO_x), sulphur dioxide (SO_2), carbon monoxide (CO), volatile organic compounds (VOCs – treated separately above), and particulate matter including total particulate matter (TPM), as well as particle matter less than 2.5 microns ($PM_{2.5}$) and 10 microns (PM_{10}) (Environment Canada, 2010). While ground level ozone and secondary particulate matter are also considered to be criteria air contaminants, they were not considered..

NO_x is the generic term for mono-nitrogen oxides NO and NO_2, which are produced during combustion and found in engine exhaust. SO_2 is naturally occurring in petroleum products, and despite usually being removed to a large part in the refining process, the combustion of petroleum products releases measurable SO_2. Carbon monoxide is produced during the incomplete combustion of fossil fuels and biomass, including wildfires and controlled burns. Carbon monoxide is also produced indirectly from the photochemical oxidation of methane and other VOCs in the upper atmosphere (ATSDR, 2009).

2.5.3   *Greenhouse Gasses*

Greenhouse gases (GHG) included for consideration as part of this study will include methane and carbon dioxide. Methane, although not a volatile source onsite will be produced over the life cycle of the contaminant breakdown in the landfill (see Contaminant Life cycle below). The International Panel on Climate Change (IPCC) defines both of these compounds as greenhouse gases and can be seen below in Table B with their associated global warming potentials.

Table B. IPCC global Warming Potentials for Relevant Greenhouse Gases

Greenhouse Gas	Global Warming Potentials [a]
Carbon dioxide	1
Methane	25

a – IPCC Fourth Assessment report, (2007), 100-year time horizon

2.6   Vapour Capture and Disposal

Any combination of hoods and/or ventilation systems that captures or contains organic vapours so they may be directed to an abatement or recovery device is known as a vapour capture system, also referred to as soil vapour recovery systems. The objectives of vapour recovery

systems used in remedial work may be varied but are generally utilized to prevent harmful exposures to workers, prevent unacceptable emissions, or potentially captured volatiles for condensation, refinement and reuse. In many cases where vapours are captured to prevent ambient exposures the volatile compounds are bound to an active carbon filter. These filters are then disposed of, ending up in landfill disposal facilities. In such cases, the disposal of the carbon contained within the captured vapour will undergo the same processes as other carbon sources disposed in landfill facilities. A portion of this carbon will be metabolized though microbial catabolism emitted as carbon dioxide, and a portion of the carbon mass will be emitted as methane. As noted above the proportion of methane emitted from hydrocarbon sources is uncertain, this is also true for hydrocarbon bound in carbon filters, as well as the carbon represented in the filters. However, some mass of methane will be produced from this disposal otherwise not predicted to be produced in *ex-situ* remediation.

2.6.1   *Biofilters*

Background

A typical biofilter uses a three-phase system, consisting of a gas phase transporting the target chemicals through the reactor, a liquid phase which contains an aqueous biofilm, and a solid phase substrate on which the microorganisms are present (Ikemoto *et al*. 2006).

The gas phase often consists of ambient air, with the typical open biofilter being exposed to the atmosphere and limited by the rate of ascending gas flow (Delhomenie and Heitz 2005). The speed of the biofiltration process is often limited by the rate of diffusion of contaminants inside the biofilm layer. A prototype biofilter using a convective flow biofilm to speed up this process was tested by Fang and Govind (2007) and found to be more effective than a diffusive flow biofilter; however, the majority of biofilters use diffusive flow.

Biofilms are populations of microorganisms attached to the solid biofilter substrate surfaces through polymeric substances, and the biofilm matrix will contain: water, microbial cells, polymers, absorbed nutrients, and metabolic byproducts (Singh *et al*. 2006). Establishment of a biofilm within a biofilter requires an initial microbial population and sufficient nutrient flow to maintain that population (Singh *et al*. 2006).

The solid substrate, or filter bed, is generally in the form of inert packing material, and is required to increase the available area for surface transfer between phases (Delhomenie and Heitz 2005). Microorganisms are immobilized on this surface, which is often also infused with a nutrient solution (Delhomenie and Heitz 2005). A filter bed should have: high surface area, high porosity to allow for gas transport, good water retention, nutrients to support microbes, and diverse indigenous microflora (Delhomenie and Heitz 2005). Options for filter bed material are peats, soils, or compost, which can also have additional material added for structural support, such as wood chips. Typical biofilter substrates have a porosity of 40 to 50%, a surface area of 1 to 100 m^2/g, and contain 1 to 5% organic matter (Nelson and Bohn 2011). Compost typically has a higher porosity (50 to 80%) and organic matter content (50 to 80%) along with greater microbial density (Nelson and Bohn 2011).

Use of other organisms besides bacteria, such has fungi, has also been attempted. A study by Harms *et al*. (2001) identified potential fungal species capable of degrading PHCs; however, there were not considered to be ideal for biofilters as they were less resilient and grew slower

than similar bacteria. Hasan and Sorial (2010) also looked at using fungi to enhance bioremediation in acidic environments, and found that adding fungi to a biofilter increased the available surface area, increasing the gas-phase uptake and elimination diffusion as a rate limiting step.

At this time most commercial biofilters operate using indigenous microbial populations.

Relevant Parameters

The parameters of greatest importance to biofilter operation are generally considered to be: air flow rate, temperature, pH, moisture, and substrate organic content (Delhomenie and Heitz 2005). The treatment rates are often limited by the mass transfer of target compounds to the biofilm phase, and mass transfer rates are most influenced by: target chemical properties, biofilter substrate properties, gas flow behavior, phase surface area, wettability of biofilter substrate, and environmental conditions (Kraakman *et al.* 2011).

During biofilter operation, moisture content often requires regular monitoring. The ideal moisture content is considered to be between 10 and 25% for soil substrate based biofilters, and between 20 and 20% for compost substrate based biofilters (Nelson and Bohn 2011). In order to maintain optimal humidity levels, active systems such as passing inlet air through a water column may be required depending on environmental conditions (Rani 2009).

Treatment Rates

Removal efficiencies over 99% have been obtained for BTEX in lab tests (Mathur *et al.* 2007, Pandey *et al.* 2010) and complete removal of hexane in a compost biofilter (Zamir *et al.* 2011); however, lower removal efficiencies closer to 80% are often encountered in practice (Namkoong *et al.* 2003). Treatment rates are typically limited by the rate of mass transfer between phases in a biofilter, and are influenced by the initial concentration of microbes and the air flow rate through the biofilter (Delhomenie and Heitz 2005). Mass transport in biofilms is described by Fick's Law (Delhomenie and Heitz 2005), and soil water partitioning in biofilters is generally the limiting factor in treatment rates, whereas air-water partitioning tends to reach equilibrium at typical flow rates (Massabo *et al.* 2007).

Remediation rates are typically proportional to environmental temperature and are inversely proportional to the molecular weight of the target compounds (Sanscartier *et al.* 2011). Microbial activity is linked to ambient temperature, with an optimum between 20 and 40 °C; however, degradation processes can cause a temperature gradient to occur within the filter which can hinder maintenance of the filter (Delhomenie and Heitz 2005). Achieved treatment efficiency will vary depending on input concentrations of target compounds and the biofilters internal conditions; however, addition of silicon rubber pellets in a bioreactor or other compounds that can absorb/desorb gaseous components can be used to smooth out input concentrations, preventing possibly damaging high transient loadings of contaminants (Littlejohns and Daugulis 2008) and maintaining a more consistent treatment rate. Despite its relatively low molecular weight, removal rates of benzene are commonly the lowest among BTEX and other petroleum hydrocarbons (Namkoong *et al.* 2001) as it is easier to degrade water soluble compounds (Nelson and Bohn 2011).

Other chemical parameters of target compounds also influence the treatment rates of biofiltres. Degradation rates tend to increase with the presence of compounds with double bonds, and higher oxygen, nitrogen, and sulphur content (Nelson and Bohn 2011). The presence of certain contaminants can also interfere with the biodegradation of target compounds. For example, Hasan and Sorial (2009) found the degradation of n-hexane to be negatively impacted by the presence of benzene. Shim *et al*. (2006) determined that the presence of methyl tert-butyl ether (MTBE) decreased the elimination capacity for benzene, toluene and xylenes, with benzene degradation most strongly inhibited by MTBE. It was also found that MTBE could not be degraded alone, as it was co-metabolically degraded with benzene, toluene, or xylene.

The materials used in biofilter construction can also significantly treatment rates. The use of granulated activated carbon can be used to enhance gas adsorption (Mathur *et al*. 2007), and has been show to benefit treatment efficiency, depending on humidity (Nabatilan and Moe 2011). Use of higher porosity substrates can increase removal rates while reducing plugging issues (Nikakhtari and Hill 2008). When trying to remediate hydrophobic substances that will not readily dissolve into the biofilm phase, the use of surfactants has been found to enhance solubility and increase biofilter effectiveness (Hasan and Sorial 2010), this result has not been found to be significant in all cases involving PHCs (Lee *et al*. 2011).. The use of organic solvents has also been attempted to achieve this effect (Kraakman *et al* 2011)

Operational and maintenance considerations are also important when trying to determine treatment rates. High conversion rates of target compounds can result in microbial growth rates high enough to result in filter clogging (Kraakman *et al*. 2011). Shim *et al*. (2006) found that the presence of specific pollutants, such as MTBE, can significantly alter the established bacteria communities and the expected treatment rates for other target compounds. Additionally, high inlet concentrations of target parameters may also damage the degrading biomass, Barona *et al*. (2007) found this to be the case with even low concentrations of toluene.

Applicability to Emissions Model

Design of biofilters based on models or small-scale testing often runs in to difficulties due to uncertainties in determining the rate-limiting step in larger systems (Kraakman *et al*. 2011). While treatment rates can often be established based on easily measurable parameters, such as biofilter design or flow rate in lab scale tests, determining the actual treatment rate of a biofilter under field conditions would require some form of validation.

However, for all biofilters, there is a range of operational parameters over which biofiltration is effective, dependent on the filter design, microbial community, and contaminants of concern (Fang and Govind 2007). Therefore, it is possible that a standard removal efficiency credit could be applied to the emission model if a generic biofilter design was confirmed to be operating within the defined operational range. This credit would also be contingent on some basic site conditions, such as the absence of contaminants that may limit microbial growth, or adverse site conditions (such as extremes of temperature or humidity).

While overall removal efficiencies greater than 95% are possible and efficiencies of 80% or greater are expected to be achieved, actual removal efficiency will vary between contaminants and result in changes to the composition of volatile contaminants. Therefore application of an

overall removal credit should be based on the lowest expected removal efficiency, in order to account for difficult to degrade chemicals and the potential for inhibitory interactions between compounds. Based on a review of published literature on biofilter treatment of petroleum hydrocarbons, a minimum expected efficiency of 60% is considered reasonable for difficult to treat compounds such as toluene (Barona *et al.* 2007; Fang and Govind 2007; Zmir et al. 2011), n-hexane (Hasan and Sorial 2010; Zamir et al. 2011; Zmir et al. 2011), methyl tert-butyl ether (Mudliar et al. 2010), or benzene (Namkoong et al. 2001; Pandey et al. 2010). A 60% reduction factor has been applied to long-term air concentration exposure during ex-situ remediation, the biofilter reduction factor was not applied to the short-term air concentration.

A case-study, similar to the case used for the original emission model in Meridian (2011) was used to demonstrate the effect of the biofilter reduction factor on long term exposure concentrations. As discussed above, presence of a biofilter is assumed to reduce long term exposure concentrations by a factor directly proportional to the biofilter treatment efficiency. Use of a biofilter also increases engine combustion emissions of criteria air contaminants by a small factor due to transport of the biofilter to and from the site. Model inputs and outputs for the example site are included in Appendix E.

Lifecycle Emissions

Lifecycle emissions for biofilters can be considered similarly to how landfill emissions are handled in the Meridian (2011) model, which involves determining the mass of carbon entering the biofilter and then applying a standard conversion to CO_2 factor. As biofilters will be operated aerobically, the calculation of emissions from ex-situ remediation does not change with the presence of a biofilter unit and it is assumed that 100% of contaminants will be converted to carbon dioxide.

Additional consideration for biofilters that require some active component, such as a fan to increase airflow or systems to maintain temperature or humidity has been incorporated into the 'hours of power generation' input ex-situ remediation.

The lifecycle emissions from construction and maintenance of biofilters are expected to be minimal, as soil or waste material is often used for the filter substrate, and indigenous microbes can be used under most conditions (Massabo *et al.* 2007), providing that an acclimatization period with the targeted contaminants is factored into the treatment rate (Delhomenie and Heitz 2005). For calculation purposes, it is assumed that the filter substrate is comprised of natural or waste materials and makes up the bulk of the biofilter mass. The remainder of the biofilter, consisting of the outer shell and and monitoring equipment, is assumed to be reused. Maintenance of the biofilter is necessary as bacteria have a tendency to accumulate at phase boundary interfaces (Hanzel *et al.* 2012); however, maintenance activities are not expected to represent a significant source of emissions. In order to account for additional transportation of the biofilter to the site and removal afterwards, if a biofilter is being used it is assumed by default that an additional truck is required to mobilize to the site twice based on the typical size of a biofilter.

2.7   Review/Selection of Emissions Models

All processes thought to be directly involved in contributing emissions from *ex-situ* remedial activities, including criteria air contaminants and greenhouse gases, were considered. As discussed above, model selection considered multiple aspects of the model, its developmental purpose and potential application to estimating emissions from remediation. As no individual model was ideally suited to all aspects of the remediation scenarios, multiple models were utilized that were best suited to specific processes.

The basic approach for quantifying life-cycle emissions for both scenarios involved considering the emissions from contaminant breakdown, and the process and equipment emissions separately. Quantification of emissions from contaminant breakdown applied a mass balance approach, where the mass of carbon contained within the contaminants was calculated and assumed to completely volatilize over the course of the soil treatment. For the *ex situ* scenario the mass of the carbon is assumed to entirely transform to carbon dioxide, which was calculated. This approach was also conducted for the landfill disposal, however; a fraction of the emissions were assumed to be methane and calculated as such. Modelling was used to predict an emission concentration, or air concentration, as a function of time applicable to on-site worker exposure during an ex-situ remedial scenario, or for application in air dispersion modelling to potential near-by receptors.

Models with some form of regulatory acceptance, particularly federal regulatory acceptance that would be broadly applicable were preferred. The Environment Canada vehicular transport particulate emission model was taken from the NPRI toolbox and is currently a federally recommended model for the quantification of particulate emissions in industrial requirements. The CCME biosolids model is a national guidance tool with a certain level of regulatory recognition, and the GHGenius model was developed by scientists for Natural Resources Canada. The national acceptance, publication or endorsement of these models was considered in the model selection.

Models that specifically considered Canadian conditions, or were developed or endorsed by Canadian regulators, were also preferentially considered. For example, the GHGenius model applies Canadian fuel specifications, the Environment Canada NPRI toolbox model includes corrections for Canadian climatic conditions.

Another consideration was the ability of the model to quantify a variety of emissions using the same input assumptions. For instance, the GHGenius model was able to predict combustion product emissions for all criteria air contaminants, including particulates, as a function of emissions factors. The metrics of the emission were also important to the model selection; models that quantified an emission mass were preferred as this allowed a direct sum and comparison between processes. Another major consideration in the model selection was the data requirements of the model, models that require data inputs that would need to be based on scenario assumptions or site specific information as a major inputs were avoided.

Models were reviewed related to the partitioning and mixing of volatile contaminants to calculate air concentrations, equipment and process emissions of particulates, and separately calculated equipment and process emissions of criteria air contaminants. The reviewed models can be viewed in Appendix B.

2.8   Establishment of Input Parameters

Many model input parameters will vary on a site-specific basis (e.g. distance to landfill, chemical concentrations in soil). Others may vary on a site-specific basis but may not be routinely measured at contaminated sites (e.g. certain soil properties). A third group of model parameters either will not vary on a site-specific basis (e.g. some parameters related to landfills) or may not be readily known ahead of time (e.g. fuel efficiency of equipment and vehicles).

In order for the modelling approach to be manageable, model input parameters for the selected models are divided into three categories:

- Key site-specific parameters, which are always established on a site-specific basis. These variables are readily measured and have a significant effect on model results.
- Optional site-specific parameters, for which default values are provided but which may be adjusted on a site-specific basis where suitable data are available. These variables are less readily measured or have only minor effects on model results.
- General parameters, for which default values are provided and it is expected they would rarely, if ever, be adjusted on a site-specific basis. These variables are generally not readily measured.

For all except the key site-specific parameters, default values were selected based on values previously recommended by major regulatory agencies, recommendations of model authors, and scientific support.

3   RESULTS

3.1   Selected Modelling Approaches

A list of selected models for emission quantification requirements can be seen below in Table C. A summary of the relevant equations and model input parameters is provided in Appendix D.

Table C. Selected Models for Various Emissions Quantification Applications

Emission	Model
Ex-situ contaminant emission	Mass Transfer
Landfill gas emission	CCME Biosolids Emission Estimation
Combustion emissions (particulates, criteria air contaminants and greenhouse gases)	GHGenius NRC Model
Equipment process particulate emissions	US EPA Construction Particulate Emissions, 1993
Vehicular transport particulate emissions	Environment Canada NPRI Toolbox Estimating Road Dust Emissions

The relevance of any modelling results is based on the accuracy of the data inputs. In order to ensure that the modelling results were as accurate as possible, recent data was applied for fuel usage and combustion emission analysis for emission factors, and internationally recognized uncertainties and assumptions often applied as correction factors. Both the CCME biosolids model and the GHGenius model have been published in the past year, and the NPRI toolbox vehicle particulate emission model was published or partially updated in 2009.

For some of the processes few models are available to quantify emissions, as in the case of the particulate process emissions. The US EPA particulate generation and emission equations are generally regarded as the definitive model for this application and many other regulatory agencies including those in Canada reference these models. In this case, the rational for the model selection became about current availability.

3.1.1   *Estimating Volatile Organic Compound Emissions*

VOC emissions occur from several different processes, some of which apply to both *ex situ* remediation and landfill disposal. In addition to the mass balance calculation used to estimate total potential emissions from the contaminants in the soil, the calculation of potential exposure to receptors either onsite, such as workers or nearby offsite receptors, may provide an additional consideration to augment the lifecycle analysis and is relevant to remedial decision making.

As detailed in Appendix B, the Jury model as implemented by US EPA or Ontario Ministry of Environment is suitable for estimating VOC emissions from excavations and from soil piles. The model calculates emission rates as a function of time. The highest emission rates (and hence the worst-case exposure) occur immediately after soil disturbance or handling, as the highest concentration of volatile contaminants will be present in soil and begin partitioning into the vapour filled pore space and moisture filled pore space before contaminant particles will move into the ambient air (Figure 2). As time passes, the contaminant mass in the soil will decrease, resulting in lower volatile emissions and ambient air exposure concentrations.

For estimation of risks from short-term exposure, calculations are conducted based on average emissions from initial exposure of the soil to the time used for the exposure limit. For example, for comparison with a 1-hour exposure limit or air quality guideline, emission rates are calculated from time = 0 to time = 1 hour. Total or long-term average emissions are evaluated based on the anticipated length of time a soil pile is present at the site.
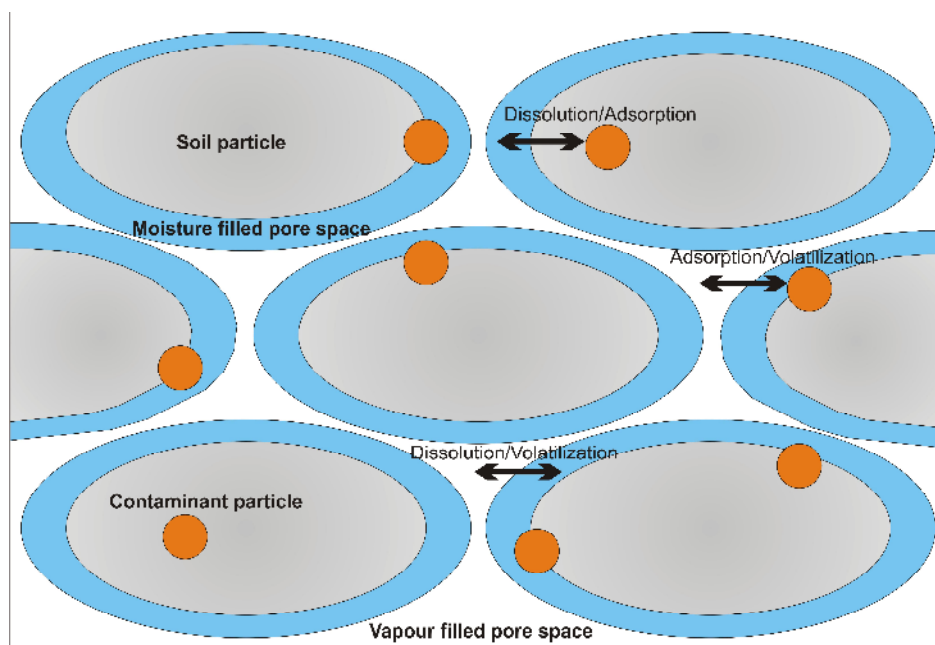
Figure 2: Contaminant partitioning

Data on emissions during *ex situ* remediation were very limited and no existing models were identified; therefore a mass balance approach was used, in many respects similar to a US EPA (1992a) model for emissions from soil handling during a remedial excavation. For estimating potential short-term maximum exposures to workers, the maximum chemical concentration in soil is used. For estimating total emissions and long-term average exposures, average concentrations are more appropriate.

VOC emissions predicted for any of these processes are then used with regulatory air dispersion models to predict offsite air quality. For screening purposes, AERSCREEN is recommended as a screening-level air dispersion model with regulatory acceptance; for site-specific evaluations AERMOD or CALPUFF could also be used, as appropriate.

3.1.2   *Estimating Particulate Matter Emissions*

Particulate emissions, regardless of whether or not the emitted material is contaminated, are recognized as harmful to human health. Canada Wide Standards (CCME, 2000) have been published for particulate matter and recognize a fine fraction $PM_{2.5}$ (airborne particles that are 2.5 microns or less), as well as larger particulates PM_{10} (airborne particles that are 10 micron or less in diameter); of which the fine particles are recognized as having the greatest effect on human health. The Canada Wide Standard for $PM_{2.5}$ is 30 µg/m^3 over a 24-hour averaging time. Alberta Environment has also published objectives for total suspended particulates of 100 µg/m^3 over a 24-hour averaging period.

Contributions of total particulate emissions in both landfill and *ex-situ* remedial activities are a result of a variety of processes such as road traffic on paved and unpaved roads, equipment and vehicle exhaust, material transfer operations, mixing and tilling, soil spreading and is influenced by environmental factors like wind, soil moisture and particle size. Many of these processes and variables will fluctuate substantially over different phases of the remedial process and introduce significant uncertainty in trying to estimate particulate emissions over the course of the entire remedial process. To help address some of these uncertainties available guidance on estimating particulate emissions from heavy construction or soil moving operations recommends that emissions are to be estimated for each particular construction process (US EPA, 1995). Models reviewed to estimate particulate emissions related to these activities are presented in Appendix B; the final equations are presented in Appendix D.

3.1.3   *Estimating Emissions of Criteria Air Contaminants*

Few models are available to quantify emission of criteria air contaminants from fuel specifically used in Canada; fuel specifications vary regionally and emissions are related to the composition of the fuel used. Recently, Natural Resources Canada has published a spreadsheet tool referred to as the GHGenius model which was developed as a life cycle assessment tool that calculates greenhouse gases and criteria air contaminants generated from the time a fuel is extracted to the time that it is converted in a motive energy vehicle to produce power, considering internal combustion in various equipment sources. Due to the Canadian focus of this model and the regulatory acceptance, this model was deemed the most appropriate and used in the calculation of equipment engine emissions. This emission generation model is based on fuel use, and calculates the mass of pollutant emitted per GJ of fuel used by device. The amount of energy potential in GJ per L of petroleum diesel was adopted from the National Energy Board Energy (2010) conversion tables. Equipment fuels use was obtained from a recent comprehensive field study of fuel usage by equipment type (Frey et al., 2010), reflective of equipment utilized during both *ex-situ* remedial activities and landfill disposal.

3.1.4   Estimating Landfill Gas Generation

Models used to estimate landfill gas generation were reviewed and summarized in Appendix B. However, most available models are related to annual municipal landfill reporting requirements and calculate methane conversion as a function of time. For the purposes of this project, total emissions or conversion to methane over the life-cycle of the waste in the landfill are of more interest.

CCME has recently developed a biosolids emission estimation model which is intended to be used to calculate the greenhouse gas emissions from carbon contained in biosolids being disposed on in landfills. A modified version of this model was adopted, with adjustments made to the default assumptions related to methane generation, and uncertainty factors applied to the calculation of landfill methane generation from carbon inputs into a landfill.

3.2   Exposure Limits

Calculated concentrations in outdoor air (onsite or offsite) should be compared to appropriate limits. For worker exposure onsite, occupational exposure limits would apply. For offsite air quality impacts, ambient air quality guidelines/objectives and health-based limits apply. Suggested limits are summarized in Appendix C.

3.3   Recommended Approach Equations and Default Model Parameters

The modelling approach and equations recommended to complete the lifecycle analysis can be seen in Appendix D. Default model inputs used to complete the modelling are also documented within this appendix.

The modelling approach detailed in Appendix D is flexible enough to be adapted to a range of scenarios, since individual processes are modelled separately and the results combined. The approach yields both total lifecycle emissions of contaminants and predicted concentration in onsite and offsite ambient air.

3.4   Example Site Calculations

An example site scenario was created and calculations were completed based on site data provided by Husky Energy Inc. in order to test the modelling approach and compare results against site air quality measurements. The example calculations are detailed in Appendix E, the tables in this appendix detail the findings of these calculations, the inputs used, as well as the assumed site and chemical characteristics.

3.5   Model Sensitivity

Certain input parameters will have more of an impact on the outcome numbers than others. Modelling greenhouse gas emissions from landfill disposal is highly dependent on the fraction of methane assumed in landfill gas. This parameter is highly uncertain with respect to methane generation from petroleum hydrocarbons.

The most sensitive variable input parameters in the calculation of contaminant flux from excavations and soil piles, and thus air concentrations include the fraction of organic carbon, bulk density, and soil porosity. The final calculation of air concentration is also highly influenced by the source length and wind speed. Concentrations predicted directly above a soil pile or within an excavation are inversely related to the wind speed; during a windy day air concentrations would be lower. Wind speed, allu bucket dimensions and allu bucket height are also sensitive parameters in the calculation of air concentrations during *ex-situ* allu soil treatment.

The calculation of particulate emission onsite from equipment processes is most influenced by soil moisture content, where higher soil moisture content results in lower particulate emissions. Wind speed is also and important factor for particulate emissions. In the calculation of particulates from vehicular traffic, the mass of the vehicle followed by the percent silt content are important to the calculation of a particulate emission factor. However, the distance travelled is the most influential factor on the overall particulate emissions from vehicular traffic. Emissions of volatiles from engine combustion are most influenced by the equipment specific fuel use and operational hours, as these emissions are based on fuel use and emission factors.

4   DISCUSSION

The modelling approach developed herein and summarized in Appendix D is intended to compare lifecycle emissions from alternative remediation scenarios, as well as evaluate potential human health and environmental impacts. There is considerable uncertainty in the overall lifecycle emissions; the modelling approach is intended primarily to enable a relative comparison between alternative scenarios. From this relative comparison, some broad conclusions can be made. Due to the anaerobic transformation in landfill disposal, a mass of methane will be emitted that will not be produced in the *ex-situ* remedial scenario, with a higher relative greenhouse gas potential. The transport/mobilisation distance and subsequent fuel used will influence the engine emissions of criteria air contaminants, which will be increased with increased distance to a landfill/site and could be an important consideration when making remedial decisions. Particulate emissions will be dependent on site characteristics including soil moisture, soil particle size and wind conditions.

The VOC emission models are believed to be conservative in most cases, and likely over-predict ambient air concentrations of VOCs.

4.1   Uncertainty assessment

The results of this life-cycle assessment are subject to considerable uncertainty due to a variety of assumptions that have to be made to conduct the comparison. These assumptions have varying influences on the outcomes but some of the major assumptions and sources of uncertainty include the following.

Some contaminant mass will be volatilized to the atmosphere during transport and incorporation into the landfill cell; however, quantifying this loss is subject to considerable uncertainty and would assumed to be accounted for in the landfill transformation uncertainty factor. During phase 1, the aerobic phase, some volatile losses may also be expected; however, data quantifying volatile hydrocarbon losses during this phase are limited.

Information related to landfill gas generation from industrial and hazardous waste landfills is generally unavailable. The appropriateness of applying methane generation factors from municipal landfill gas generation to petroleum hydrocarbon wastes specifically is unknown. The model assumes landfill gas is not collected and the landfill environment transitions from an aerobic environment to an anaerobic environment. Both scenarios consider the complete breakdown and atmospheric emission of bound carbon as either methane or carbon dioxide, the appropriateness of this assumption over the life-time of the contaminant is unknown.

These more qualitative aspects, although important, are not always amenable to comparison. Consideration of landfill gas collection or other *ex-situ* remedial soil options was not explicitly considered as uncertainties associated with these techniques are difficult to account for.

4.2   Next Steps

While this project is considered to be substantially complete, the field data collected (Meridian, 2012) could not be used to reliably calibrate the model since the volatile hydrocarbon concentrations were relatively low and concentrations did not change significantly between pre- and post- treatment samples. Therefore, there may be value in collecting further field data from sites with high levels of volatile hydrocarbon (ideally condensate) contamination undergoing *ex situ* remediation.

5   CLOSURE

This report has been prepared under the Alberta Upstream Petroleum Research Fund (AUPRF), under the direction of the Canadian Association of Petroleum Producers (CAPP) and the Small Explorers and Producers Association of Canada (SEPAC), as indicated in a letter of approval from PTAC dated August 24, 2010, and a subsequent letter of approval dated February 3, 2012. Quantitative and qualitative environmental modelling and fate analysis involves a number of uncertainties and limitations. As a consequence, the use of the results presented herein to develop site management strategies may either be overly protective or may not necessarily provide complete protection of human and environmental receptors or prevent damage of property in all circumstances. The work presented herein was conducted in accordance with generally accepted protocols. Given the assumptions used herein, the modelling approach is believed to provide a conservative estimate of the risks involved. The services performed in the preparation of this report were conducted in a manner consistent with the level of skill and care ordinarily exercised by professional engineers and scientists practising under similar conditions.

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APPENDIX A
EX-SITU REMEDIATION TECHNIQUES

APPENDIX A: EX SITU REMEDIATION TECHNIQUES

DEFINING EX-SITU REMEDIATION

Ex-situ remedial techniques aim to remove contaminated mass from media using a variety of technique that all include at a minimum physical soil manipulation. During this initial process, soil is broken down into agglomerates resulting in a significant increase in surface area, encouraging contaminant partitioning from soil to air. The volatilization rate, or emission flux, is dependent on a chemical concentration gradient between the soil and ambient air, the diffusive resistance within the soil matrix, and on convective mass transfer rates in the air adjacent to the soil (Devaull, 2001). To further encourage contaminant breakdown and liberation often multiple techniques are used (i.e. excavation and exposure, in conjunction with chemical reduction/oxidation). Techniques that are employed onsite and do not require the transport of soil will eliminate transport emissions and but include the equipment and process emission from onsite activities.

Excavation & Exposure

The excavation and removal of contaminated soils is necessary for all ex-situ remedial techniques. The impacted medium is physically extracted and either transported to the process unit (assumed on-site), treated, and the treated soil may be used as fill at the site. Soils are treated aboveground, and this technique is sometimes referred to as land farming. Soil handling generally involved in this process, influencing volatility, includes dumping, grading, sizing and feeding soil into treatment processes. Procedural elements of this process will also influence the volatility, like the duration and vigorousness of the soil handling, rate of excavation, the dumping drop height, the shape of the piles, and the size of the equipment used (smaller bucket size will increase surface area exposure). The main emission points for soil disturbance includes exposed wastes in pits or trenches, material as it is physically processed (i.e. dumped from the excavation bucket or passed through an aeration auger) and soil storage units or biopiles (US EPA, 1997). Often soil may be physically aerated through tilling, or auguring to improve aeration and volatilization. Importantly, contaminants are also exposed to ultra-violet light enhancing contaminant transformation. Particulate emissions will be higher depending on the aggressiveness of the physical treatment process.

Volatilization through Soil Manipulation

Physical soil manipulation and processing to encourage contaminant volatilization by increasing the total exposed area of the soil often uses mechanical processors to pulverise and aerate the soil material. One mechanical processor commonly applied is an allu bucket, which usually contains an internal rotating drum, or some form of an internal crushing/screening system, generally controlled by the operator. This physical processing pulverizes the soil matrix, removing any hardened soil aggregates, and separates any large rocks, roots or other large items of organic matter. This pulverized soil is then passes through the screen and released some distance to the ground, generally 2-5 meters or the height of the bucket.

As the soil is released to the ground, which aerates the soil and encourages volatilization, the soil volume also dries and warms to ambient air temperatures. The drying and warming of the soil, in addition to the reduced aggregate size and increased surface area, encourage contaminant volatilization reducing contaminant concentrations in the soil. Little information is available on allu processing efficiencies, however, the

required number of soil passes through the allu bucket to volatilize a sufficient mass of contaminant will depend on the soil moisture content, the amount of organic carbon present in the soil, the ambient air temperatures and the initial contaminant concentrations.

Biological Treatment

This treatment technique refers to a broad range of technologies such as bio-treatment cells, soil piles, composting, and prepared treatment beds. These treatments usually involve placing excavated soil into above ground enclosures or spreading it over treatment beds that may include aeration of leachate systems. These systems allow for increased control on environmental factors influencing biodegradation processes and rates. Often in addition to the physical treatment design (i.e. treatment cells) the addition of oxygen, nutrients, water or microorganisms may be done to enhance the breakdown of contaminants. Active aeration, also using tilling, dumping, auguring, or even forced air flow with fans improves soil aeration and contact between the microorganisms and the contaminants; nutrients, microorganisms or soil amendments are routinely added at this time. Emission points are similar to excavation and removal and generally regarded as area emissions, but the main emission points would include exposed waste in excavation pit or pile and material as it is dumped from the excavation bucket (US EPA, 1997). Emissions tend to increase with an increase in surface turbulence due to wind or mechanical agitation. Temperature affects emissions through its influence on microbial activity, as well as increasing chemical volatility. Microbial degradation of contaminants to encourage volatile losses takes more time than exposure alone.

Chemical Reduction/Oxidation

As with the other ex-situ techniques, this soil treatment technique involves the extraction of contaminated material and the addition of reducing/oxidizing agents to chemically reduce contaminant concentrations and create more stable, less mobile or inert compounds, as well as stimulating aerobic microbial degradation through the presence of oxygen released during decomposition (Goi et al., 2009). Commonly used reducing agents are hydrogen peroxide, hypochlorites, chlorine, and chlorine dioxide (Environment Canada, 2002). This technique will generally require less soil disturbance than excavation and exposure alone and will therefore be expected to have less particulate matter emissions.

Thermal Desorption

Low and high temperature thermal desorption involves exposing the excavated material to heat, encouraging the volatilization of water and organic contaminants. Low temperature thermal desorption utilizes temperatures from 95 – 315 ⁰C, while high temperature thermal desorption applies temperatures ranging from 315 – 540 ⁰C, and these higher temperatures facilitate the removal of less volatile compounds. Both treatments control the residence time of heat application, designed to volatilize selected contaminants but not oxidize them, and not incinerate the soil. These techniques, in comparison to the other ex-situ techniques reviewed, would require fuel addition to generate the required desorption temperatures creating additional site emissions, in addition to the excavation equipment emissions and soil volatile losses.

APPENDIX B
REVIEW OF EMISSIONS MODELS

Appendix B:
Review of Emissions Models

MODELS TO ESTIMATE AIR CONCENTRATIONS FROM CONTAMINANT PARTITIONING

Models based on contaminant partitioning and diffusion are appropriate for estimating the emissions of VOCs from stationary soil. This would include emissions from excavations and from stockpiled contaminated soils.

Jury Model

The original Jury model was published in 1983 and updated with a simplified equation in 1990. It is used to calculate volatilization losses for both infinite and finite sources. In general, both versions of the model describe the vapour-phase diffusion of the contaminants to the soil surface and loss by volatilization to the atmosphere, establishing the relationship between vapour and solute diffusion and adsorption by defining total phase concentration partitioning as it relates to the effective diffusion coefficient. Each model predicts an exponential decay curve over time once equilibrium is achieved; determined by the rate at which contaminants diffuse upward. The simplified equation applies the following assumptions (US EPA, 1996b):

- uniform soil properties,
- instantaneous linear adsorption,
- linear liquid-vapour partitioning
- soil concentration is below saturation limit,
- no stagnant air layer, or boundary layer thickness,
- no water evaporation or leaching,
- no chemical reactions, including biodegradation or photolysis, and
- diffusion occurs simultaneously across the upper boundary and the lower boundary.

The model is therefore limited to surface contamination extending to a known depth and does not account for subsurface contamination covered by a layer of clean soil. Both models do not account for the high initial rate of volatilization before equilibrium is attained and will tend to under-predict emissions during this period (Environmental Quality Management, 1995). The model does also not consider mass flow of contaminants due to water movement in the soil profile, or the volatilization rate of saturated soils or non-aqueous phase liquids.

US EPA commissioned a study to validate the relative accuracy of the Jury volatilization models using experimental emission flux data (Environmental Quality Management, 1995). From the results of this study, it was concluded that for the compounds included in the experimental data, both models showed good agreement with measured data given the conditions of each test. Each model demonstrated a high agreement with bench-scale measured values, and to a lesser extent the infinite source model showed reasonable agreement with pilot-scale data, and overall these models are expected

make reasonable estimates of loss through volatilization at the soil surface given the boundary conditions of each model.

The simplified finite source model as implemented by US EPA (1996b, 2002) is shown below. The apparent diffusivity is calculated first, followed by contaminant flux. Finally, once the contaminant flux is calculated a volatilization factor is calculated. US EPA has created a publically available model largely based on the Jury model called EMSOFT which could be used to calculate volatilization factors.

$$J_s = C_o \left(\frac{D_A}{\pi t}\right)^{\frac{1}{2}} \left[1 - \exp\left(\frac{-d_s{}^2}{4 D_A t}\right)\right]$$

where:

J_s	= contaminant flux at ground surface (g/cm^{2}sec)
C_o	= uniform contaminant concentration at t=0 (g/cm^3)
D_A	= apparent diffusivity (cm^2/s)
d_s	= depth of uniform soil contamination at t=0 (cm)
t	= time (seconds)

$$D_A = \frac{\left[\dfrac{\theta_a{}^{10/3} D_i H' + \theta_w{}^{10/3} D_w}{n^2}\right]}{\left(\rho_b K_d + \theta_w + \theta_a H'\right)}$$

where:

D_A	= apparent diffusivity (cm^2/s)
θ_a	= air filled porosity
n	= total soil porosity
θ_w	= water-filled soil porosity
D_w	= diffusivity in water (cm^2/s)
D_i	= diffusivity in air (cm^2/sec)
H'	= dimensionless Henry's Law constant
ρ_b	= soil dry bulk density (g/cm^3)
K_d	= soil-water partition coefficient (cm^3/g)

$$VF = \left(\frac{Q}{C}\right) \times \left(\frac{C_o}{\rho_b}\right) \times \left(\frac{1}{J_s}\right) \times 10^{-4}$$

where:

VF	= volatilization factor (m^3/kg)
J_s	= contaminant flux at ground surface (g/cm^{2}sec)
C_o	= uniform contaminant concentration at t=0 (g/cm^3)

ρ_b = soil dry bulk density (g/cm^3)

$\dfrac{Q}{C}$ = inverse concentration factor for air dispersion (g/m^2 per kg/m^3). If an area emission rate of 1 g/m^{2}s is assumed, then (Js x 10^{-4} cm^2/m^2) = 1, and the equation simplifies to simply the inverse of the maximum contaminant air concentration (kg/m^3).

American Society for Testing and Material Model

The American society for Testing and Materials (ASTM, 2004) has published equations to develop risk based screening levels for contaminant volatilization from soil and migration to air. Two separate scenarios can be modelled with these equations: one assuming contamination is exposed to air, and the second assuming that there is clean soil overlying the contamination. Both scenarios require calculation of a dispersion factor for air. Separate approaches are used to calculate a volatilization factor from soil to air depending on whether contamination is directly exposed to air.

The surface soil model includes the following assumptions:
- The contaminant is uniformly distributed in the affected soils.
- Partitioning between sorbed, dissolved and vapour phases is based on linear equilibrium partitioning.
- The chemical diffuses through the surficial soil layer.
- There is no biodegradation or other loss of the chemical.
- Vapours are well mixed in the atmosphere as modelled by a box model.
- If the time-averaged flux exceeds what would occur if the entire mass of contaminant was volatilized over the averaging time, then the volatilization factor can be determined using a mass balance relationship instead.

Depth to source can be adjusted to reflect the depth from the base of the excavation to the source. Alternatively, when groundwater is pooled in the trench a volatilization factor can be calculated accounting for loss from groundwater instead of soil. Risk-based screening levels from soil can be calculated using the volatilization factor and a target concentration in air.

$$DF_{amb} = \frac{U_{Air} \times W \times \delta_{Air}}{A}$$

where:

DF_{amb} = dispersion factor for ambient air from trench surface (cm/s)

U_{Air} = ambient air velocity in mixing zone (cm/s)

W = width of source-zone area (cm)

δ_{Air} = mixing zone height (cm)

A = source-zone area (cm^2)

$$D_{eff-vadose} = D_{air} \times \left(\frac{\theta_{air}^{3.33}}{n^2} \right) + D_{water} \times \left(\frac{\theta_{water}^{3.33}}{H_{eff} \times n^2} \right)$$

where:

$D_{eff-vadose}$ = dispersion factor for ambient air from trench surface (cm²/s)

D_{air} = molecular diffusion coefficient in air (cm²/s)

D_{water} = molecular diffusion coefficient in water (cm²/s)

H_{eff} = effective Henry's Law coefficient (dimensionless)

θ_{air} = volumetric air content of soil (dimensionless)

θ_{water} = volumetric water content of soil (dimensionless)

n = total soil porosity (dimensionless)

VF_{ss} = the lower of $VF_{ss,1}$ or $VF_{ss,2}$:

$$VF_{ss,1} = \frac{\rho_b}{DF_{amb}} \sqrt{\frac{4 \times DF_{eff-vadose} \times H_{eff}}{\pi \times t \times 31536000 \, sy^{-1} \times K_d \times \rho_b}}$$

$$VF_{ss,2} = \frac{\rho_b \times L_{ss}}{DF_{amb} \times t \times 3153600 \, sec/yr}$$

where:

VF_{ss} = volatilization factor, surficial soil to ambient outdoor air (g/cm³)

DF_{amb} = dispersion factor for ambient air (cm/sec)

ρ_b = soil bulk density (g/cm³)

L_{ss} = thickness of surficial soils (cm)

t = average time for surface emission vapour flux (yrs)

$D_{eff-vadose}$ = dispersion factor for ambient air from trench surface (cm²/s)

H_{eff} = effective Henry's Law coefficient (dimensionless)

K_d = soil to water partition coefficient (cm³/g)

$$VF_{sub} = \frac{1}{\left(1 + \frac{DF_{amb} \times L_s}{D_{eff-vadose}} \right) \times \frac{K_d}{H_{eff}}}$$

where:

VF_{sub} = volatilization factor, subsurface soil to ambient outdoor air (g/cm³)

DF_{amb} = dispersion factor for ambient air (cm/sec)

$D_{eff-vadose}$ = dispersion factor for ambient air from trench surface (cm²/s)

H_{eff} = effective Henry's Law coefficient (dimensionless)

L_s = depth to subsurface soils (cm)

K_d = soil to water partition coefficient (cm^3/g)

$$RBSL_s = \frac{RBSL_{air}}{VF} \times 10^{-3}$$

where:

$RBSL_s$ = risk-based screening level for soil (mg/kg)

$RBSL_{air}$ = risk-based screening level for air inhalation (mg/m^3)

VF = volatilization factor, soil to ambient air (g/cm^3)

Note: the VF used in the above equation will correlate to either the subsoil or surface soil volatilization; only one will be used to calculate the risk-based soil screening level.

This model assumes that vapour concentrations remain constant over the duration of exposure and all inhaled chemicals are absorbed. The calculation of the diffusion coefficient from the vadose zone assumes homogeneous soil layers. These equations published by ASTM (2004) are part of a body of equations used to develop risk-based screening levels and generally have a high level of regulatory acceptance. The application of these equations to a trench exposure scenario can be done through the manipulation of model inputs to reflect the source area and mixing zone of the trench, but are essentially applied without change.

The basis of the equations is not thoroughly documented. There is a reference to the US EPA (1988) Superfund Exposure Assessment Manual, but the specific equations published by ASTM (2004) do not appear to be in that document.

RTI Land Treatment Emission Model

The US EPA has published a model estimating emissions from the application of contaminated material onto land surfaces and during land treatment, developed by Clark Allen and referred to as the RTI model. The model, considered to be best suited for waste-piles by the US EPA due to a sound scientific basis, reasonable available input data, consideration of evaporative loses and diffusion through air-filled pore spaces. When contaminated soil is disturbed, volatile compounds have the potential for partitioning into four different phases: a vapour phase, an oil phase where volatiles are dissolved in the oil, a water phase where volatiles are dissolved in the soil moisture, and a soil phase where volatile material is absorbed by organic carbon within the soil (US EPA, 1994).

The RTI model is based on Fick's second law of diffusion (concentration depletion is proportional to the curvature of the concentration gradient). Multiple solutions are available for various time intervals in both approaches considering short, or long term emission lengths. When material containing volatile organics is applied onto or tilled into soil, the maximum rate of air emissions will occur immediately after application, and the highest exposure concentration is predicted during this period.

The two approaches are given to calculate short-term emission, first to calculate an instantaneous emission rate, and second to calculate the fraction of total emissions at a given time. Both of these approaches assume that emissions from the surface of the soil are limited by the diffusion of vapours though the pore spaces in the soil and further assumes that an equilibrium concentration of organic vapours exists at all times within pore spaces.

Short-term instantaneous emission rate equation:

$$E = \frac{M_o}{l} \left(\frac{1}{\dfrac{q_a}{K_{ea}k_G} + \left(\dfrac{pt}{D_e K_{ea}} \right)^{0.5}} \right)$$

Where:

E	=	emission rate of compound (g/cm^2/s)
M_o	=	initial loading of compound present (g/cm^3)
l	=	depth to which compound is found in soil (cm)
$_a$	=	air-filled porosity (unitless)
K_{eq}	=	equilibrium coefficient of compound in the soil (unitless)
k_g	=	gas phase mass transfer coefficient (cm/sec)
D_e	=	effective diffusion coefficient of compound in soil (cm^2/s)
t	=	time after initial disturbance (s)

Using:

$$K_{eq} = \frac{H_c}{Rt} 10^6 \frac{q_a}{Waste_{fraction}}$$

Where:

H_c	=	Henry's Law constant for constituent (atm cm^3/g mol)
R	=	Ideal gas constant (82.1 atm cm^3/g mol k)
t	=	temperature of vapour in soil, (K)
$_a$	=	air-filled porosity (unitless)
$Waste_{fraction}$	=	volume fraction of compound in contaminated soil volume

$$k_G = 4.82(10^{-3})U^{0.78} \frac{U_a}{D_a^2}^{-0.67} de^{-0.11}$$

Where:

k_G	=	gas phase mass transfer coefficient (cm/sec)
U	=	wind speed (m/s)
de	=	effective diameter of land treatment area (m)
U_a	=	viscosity of air (g/cm s)
D_a	=	density of air (g/cm^3)

Fraction of emitted mass at short times:

$$F_{at} = \left(\frac{K_{eq}D_e t}{\rho l^2}\right) 2\left(1 - \frac{1}{3}\frac{t}{t_b}\right)$$

Where:

F_{at}	=	fraction of constituent emitted to the air at time t (unitless)
K_{eq}	=	equilibrium coefficient of compound in the soil (unitless)
k_g	=	gas phase mass transfer coefficient (cm/sec)
D_e	=	effective diffusion coefficient of compound in soil (cm^2/s)
t	=	time after initial disturbance (s)
t_b	=	time constant for biological decay of constituent (unitless)

Evaluation

The Jury model (either the original version, the US EPA implementation or the Ontario implementation) and the ASTM model can address the expected scenarios for worst-case exposures to a worker at an ex-situ exposure situation. These allow for the consideration of a soil source and allow for an excavation directly in contact with contamination. The Jury model and ASTM model both have at least some degree of regulatory acceptance and review. Model field-testing has also been conducted by the US EPA to validate the relative accuracy of the Jury model; concluding sufficient agreement between the model and measured soil flux for compounds testing and has been applied to predict exposure concentrations herein. The Jury model first calculates a soil flux or rate of contaminant volatilization per unit of soil and time. This soil flux rate can then be used to calculate an air concentration as a function of time, with the concentration in air decreasing as contaminant mass is removed from soil.

The Jury model also importantly considers the loss of contaminant mass from the source over time, allowing the more accurate estimation of long-term exposures applicable to life cycle assessment considerations. Overall, based on its regulatory acceptance, field validation, and ability to predict emissions from both soil piles and excavations, the Jury model is recommended for the estimation of VOC emissions from soil piles and excavations.

ESTIMATING EX-SITU SOIL HANDLING EMISSIONS

Calculation of Emission Rates

In addition to estimating overall volatile exposures from the contaminant source, as explored above, estimating the emissions as a result of soil processing is also important to remedial exposure and life cycle emission quantification. Excavation and removal of soils contaminated with volatile compounds will increase the air exchange with soil-pore gas and the atmosphere, increasing emissions. The magnitude volatile emissions are

predicted to be a function of activity intensity, porosity and moisture content of the soil, as well as the initial concentration of VOCs.

Limited guidance is available related to the emissions from ex-situ soil processing specifically. US EPA has published related guidance on estimating air impacts during the excavation of contaminated soil (US EPA, 1992a), which outlines the calculation of an average long-term emission rate. This rate calculation uses the equation:

$$ER_{Long-term} = \frac{S_v \times C \times b \times 1}{t_R}$$

where:

$ER_{Long\text{-}term}$	= long-term emission rate (g/s)
S_v	= volume of soil moved (m^3)
C	= average contaminant concentration in soil (µg/g)
	= bulk density (g/cm^3)
1	= constant ($g/10^6$ µg x $10^6\,cm^3/m^3$)
t_r	= duration of allu bucket operation (s)

This equation calculates a total emission potential as a function of remedial activity time, assuming that soil contamination is homogeneous and that volatilization of contaminants occurs at a constant rate. As this equation considers entire contaminant loss over time it is also referred to as the calculation of a long-term emission rate. While this model was not necessarily developed to predict worst case exposures, as the contaminant volatilization losses are distributed evenly over the activity time, it can quantify overall volatile losses from soil processing. This equation was applied using average soil concentrations to calculate long-term life cycle emission rates per compound from soil processing.

A short-term emission rate using the above equation can also be calculated. This modified equation removes the concentration deemed acceptable by an appropriate soil quality guideline for the subject site, calculated as:

$$ER_{Short-term1} = \frac{(C - SQG)\,S_v \times b \times 1}{t_R}$$

where:

SQG	= soil quality guideline (µg/g)

Short-term emission rates during ex-situ soil processing, particularly at the start of soil processing, are predicted to be higher than long-term rates, which are a function of the entire processing time. Higher emission rates, resulting in higher air concentrations, are more appropriate for the calculation of worst-case worker exposure during soil treatment. Soil processing with an allu bucket was explored; this particular technique compared to other ex-situ techniques may result higher exposures due to increased aeration as soil is passed through the allu bucket and dropped to the ground. The calculation of an emission rate from soil pore space that is emitted to the atmosphere during an excavation was used (US EPA, 1992). To account for soil processing times, an allu

processing rate, which was assumed to be 25% slower than excavation rate to account for soil pulverizing and dropping, was adopted. The ExC, soil-gas to atmosphere exchange rate, was also adjusted to a value of 1, which conservatively assumes that allu treatment releases the entire fraction in pore space to the atmosphere. The above equation is based on the assumption that soil gas is saturated for each compound, and therefore may over-predict the emission rate.

$$ER_{Short-term2} = \frac{P \times MW \times 10^6 \times q_a \times Q \times ExC}{R \times T}$$

where:

$ER_{Short-term2}$	= short-term emission rate (g/s)
P	= chemical specific vapour pressure (mm Hg)
MW	= molecular weight of compound emitted (g/g-mol)
q_a	= soil air-filled porosity (cm^3/cm^3)
Q	= allu processing rate (m^3/s)
ExC	= soil-gas to atmosphere exchange rate (m^3/sec)
R	= gas constant (mmHg-cm^3/g-mol°K)
T	= temperature (°K)

The calculation of an emission rate considering soil concentration and partitioning to soil vapour was also explored to compare the emission rates with the above equation. Soil vapour was calculated using the following equation (CCME, 2006):

$$C_{Vapour} = \frac{H' \times C_{Soil} \times r_b}{q_w + K_d \times r_b + H' \times q_a}$$

where:

C_{Vapour}	= soil vapour concentration (g/cm^3)
H'	= henry's law constant (unitless)
C_{Soil}	= soil concentration (g/g)
r_b	= soil dry bulk density (g/cm^3)
Kd	= soil-water partition coefficient (cm^3/g)
q_a	= soil air-filled porosity (cm^3/cm^3)

Soil vapour, as calculated above was then substituted into a modified version of the equation for the calculation of short emissions ($ER_{Short-term}$). The equation was adjusted to remove the consideration of soil vapour based on vapour pressure, and include the calculated soil vapour concentration based on soil partitioning. This modified equation calculated and short-term emission rate based on soil partitioning as:

$$ER_{Short-term3} = C_{Vapour} \times 10^6 \times q_a \times Q \times ExC$$

where:

$ER_{Short-term3}$	= short-term emission rate (g/s)

C_{Vapour} = soil vapour concentration (g/cm^3)
θ_a = soil air-filled porosity (cm^3/cm^3)
Q = allu processing rate (m^3/s)
ExC = soil-gas to atmosphere exchange rate (m^3/sec)

Estimation of Ambient Air Concentrations

Ambient air, or outdoor air concentrations, that a worker may be exposed to during ex-situ soil processing can be estimated by applying a box model to estimate outdoor air concentrations. Conceptually, the emitting source will contribute emissions to a defined box area, at the rate defined by the emission rate, with pollutants removed from this theoretical box by wind. To estimate emission to a worker, the box was defined as the length of the emitting source and the height of the allu bucket. Wind speed will act as a lateral dispersion force. This box model is predicted to conservatively estimate ambient air concentrations, as it does not account for dispersion influences and applies conservative limits on the box area. Ambient air concentrations were calculated as:

$$C_{Air\_ST} = \frac{ER_{Short-term} \times 1000}{U \times Allu_h \times L}$$

where:

C_{Air-ST} = short-term air concentrations (mg/m^3)
$ER_{Short-term}$ = short-term emission rate (g/s)
U = wind speed (m/s)
$Allu_h$ = height of allu bucket (m)
L = source length (m)

Long-term ex-situ air concentrations can be estimated for the life cycle assessment substituting the long-term emission rate in place of the short-term emission rate.

Evaluation

The calculation of a short-term emission rate reflective of more realistic worker exposures is based on an emission rate that considers an allu processing time ($ER_{Short-term2}$). This short-term emission rate may over-predict the emission rate, as it assumes vapour saturation in soil pore spaces. Soil vapours are not expected to be saturated during soil handling operations at most sites when contaminant concentrations in soil are below saturation limits, and therefore the partitioning version is likely more appropriate than assuming saturated vapours. Where soil concentrations are high enough to result in saturated soil pore space, the partitioning relationship will result in saturated vapours. The predicted emission rates from the calculation of $ER_{Short-term3}$ considering soil partitioning and $ER_{Short-term1}$ considering soil quality guideline remediation levels should be compared, with the highest value used for exposure calculations.

The calculation of this short-term emission rate assumes that remedial activity using an allu soil treatment method will occur up to the guideline value only.

MODELS TO ESTIMATE LANDFILL EMISSIONS

Landfill emission models, commonly used in greenhouse gas accounting were reviewed. However, most of these models are based on the annual reporting requirements for municipal landfills and do not consider the total contaminant lifetime emissions but rather emissions as a function of time.

Models used for the estimation of landfill gas emissions typically focus on the generation of methane through biodegradation of the initial waste substrate. This process occurs at a rate described by Monod's equation:

$$\frac{dC}{dt} = \frac{KxC}{K_c + C}$$

where:
C = concentration of substrate at time t
x = concentration of micro-organisms
K = maximum rate of substrate utilization
K_c = substrate concentration when rate is half of maximum rate

Zero order models assume generated landfill gas is constant and does not consider age or type of waste, first order models consider the depletion of carbon in the waste over time and are affected by waste characteristics, and second order models also consider different reaction rates of a number of degradation reactions (Kamalan, 2011).

Zero Order Models

Zero order models are straight forward and require minimal input regarding temporal changes to the landfill material, and are most appropriate for substrates with very high concentrations of contaminants and consistent communities. Examples of zero order models include EPER Germany, SWANA, and IPCC. The generic form of a zero order model is:

$$Q = M \times R$$

where:
Q = methane generation rate (mass/time)
M = mass of waste (mass)
R = carbon to methane conversion rate (1/time)

Models may also account for various inefficiencies in the biodegradation process with additional factors for carbon content, landfill gas recovery, lag time in methane generation etc. as appropriate.

First Order Models

First order models are the most common landfill emission models as they can consider the effect of biodegradation reducing the overall carbon content in the waste material. These models are more appropriate for contaminated material with a lower concentration of organic material and, much like zero order models, assume a constant microbial community. Examples of first order models include: SWANA, TNO, LandGEM, GasSim, Afvalzord, EPER France, Mexico FLG, and LFGGEN. The generic form of a first order model is:

$$Q = M \times R \times e^{-kt}$$

where:
Q   = methane generation rate (mass/time)
M   = mass of waste (mass)
R   = carbon to methane reaction rate (1/time)
k   = degradation rate constant (1/time)
t   = time

A common feature of these models is the use of broad waste categories to describe the carbon content in the initial material and generic descriptors of landfill conditions accounting for moisture and biodegradation rates.

US EPA Model (LandGEM)

The US EPA has included a first order model for determining methane emissions from municipal solid waste landfills in the *Mandatory Reporting of Greenhouse Gases; Final Rule* (2009). This is intended as a tool to determine yearly greenhouse gas emissions, but could easily be modified to calculate total emissions.

The formula used is as follows:

$$G_{CH4} = \sum_{x=S}^{T-1}\left[W_x L_{0,x}\left(e^{-k(T-x-1)} - e^{-k(T-x)}\right)\right]$$

where:
G_{CH4} = methane generation rate (mass methane/time)
x   = year in which waste was disposed
S   = start year of calculation
T   = reporting year for which emissions are calculated
W_x = quantity of waste disposed of (mass waste)
L_0 = methane generation potential (mass methane/mass waste)
k   = rate constant (1/time)

Methane generation potential is the mass of methane generated from a specified mass of waste and is calculated as follows:

$$L_0 = MCF \times DOC \times DOC_F \times F \times (16/12)$$

where:

L_0 = methane generation potential (mass methane/mass waste)
MCF   = methane correction factor (unitless, default value is 1)
DOC   = degradable organic carbon (mass carbon/mass waste)
DOC_F = fraction of DOC dissimilated (unitless, default value is 0.5)
F   = fraction of methane in landfill gas (unitless, default value is 0.5)

Further calculations are available to account for methane recovery units and oxidation of methane in cover material. The adjustment for oxidation is done using the following formula:

$$Q = G_{CH4} \times (1 - OX)$$

where:
Q   = adjusted methane generation rate (mass methane/time)
G_{CH4} = methane generation rate (mass methane/time)
OX   = oxidation fraction (unitless, default value is 0.1)

Default values are provided for all necessary parameters, with additional values for the degradable organic carbon and methane generation rate constants that are based on waste composition. As this model is intended for municipal landfills no values have been provided specifically for contaminated soils. This model is publically available as a spreadsheet tool.

CCME Biosolids Emission Estimation Model

The Canadian Council of Ministers of the Environment (CCME) recently published an updated version of the Biosolids emission estimation model, intended for the calculation of greenhouse gases generated from the transformation of biosolid contained carbon disposed of in municipal landfills. This model, although it does not consider petroleum waste specifically, considers the disposal of organic carbon mass applying conversion and uncertainty factors to the generation of carbon dioxide and methane. The default inputs referenced in this model as well as the general approach were reviewed and thought to be appropriate based on the current publically recognized inputs and national scope of the publication.

Within the references and assumptions sheet of this spreadsheet tool, inputs used for the landfill disposal scenario are listed including the proposed fraction of methane in landfill gas (50%), a methane correction factor for semi-aerobic landfills (50%) and a model correction factor of (90%) to account for model uncertainties, which were applied. While this scenario is intended to consider an anaerobic landfill scenario, the landfill will transition from aerobic to anaerobic and this correction factor of 50% for semi-aerobic landfills was thought appropriate. Conversion factors converting carbon to methane or carbon dioxide were also applied of 1.3 and 3.667 respectively.

While this model considers the organic carbon present in the soil as available for decomposition this was not considered as this organic carbon will too eventually break down in the ex-situ scenario and introduced unnecessary error and uncertainty.

Landfill methane emissions were calculated using the following formula:

$$CH_4 emissions = C_{mass} \times Fraction_{CH_4} \times Fraction_{uncerta \operatorname{int} y} \times Correction_{CH_4} \times CF$$

where:

Cmass	= total mass of carbon from contaminants
Fraction $_{CH4}$	= fraction of methane in landfill gas
Fraction$_{uncertainty}$	= model correction factor to account for uncertainties
CF	= conversion factor, carbon to methane
Correction$_{CH4}$	= Methane correction factor, to account for uncertainties in landfill conversion from aerobic to anaerobic gas generation

Once the predicted methane emissions were calculated with the equation above, the remaining carbon that was calculated remaining in the landfill and not converted to methane was assumed to be eventually emitted to the atmosphere as carbon dioxide and accounted for as such.

Evaluation

The CCME Biosolids Emission Estimation model was selected to calculate greenhouse gas emission from landfill disposal. This model was published by a Canadian regulatory agency, applied the most current application accepted model correction factors from the International Panel on Climate Change, was macro in focus and is not limited to a specific time period. This model is considered appropriate for the life cycle assessment that did not consider the influences of time, but rather considered the contaminant life cycle, which was assumed completed during the emission of contaminant breakdown products carbon dioxide and methane.

MODELLING PARTICULATE EMISSIONS

For the purposes of this life cycle assessment, particulate emissions from vehicular road traffic, non-road equipment processes such as excavation as well as particulate emissions from combustion by-products are required. There are a number of applicable established emission models published by both by regulatory agencies, and industry for activity specific emission calculations.

Particulate Emissions from Combustion Engines

For non-road engines used during remedial activities, including bulldozers, excavators, backhoes, and generators it can often be difficult to attempt emission estimations partly because the power output of these engines is a function of the fuel used and thus emissions. Emission factors applied in the calculation of exhaust emissions are frequently used.

The NONROAD model applies an emission factor that is a function of horse power use per hour, assuming a constant power output and the emissions calculation is a function of time (rather than fuel use). The NONROAD model does not consider Canadian fuel characteristics and because of the constant power output that is assumed may not be applicable to remedial situations which encompass variable horse power outputs.

More recently, studies have been conducted to measure non-road equipment emissions in the field and generate equipment specific emission factors and fuel usage based on equipment model and specific soil handling activity; data are presented in both fuel and time-based emission rates (Frey et al., 2010). Fuel-based emission rates tend to have less variability and higher certainty than time-based emission rates. Therefore, the development of emission inventories based on quantifying fuel consumed, rather than time of operation, is preferred where possible (Frey et al., 2010). The results from these field measurements indicate that the results are highly dependent on the type of soil handling activity that is being performed, and may be a preferred method for emissions estimations in ex-situ situations. Alternatively for vehicular road traffic, it may be more applicable to apply an emission factor based on distance travelled for vehicular road traffic, rather than a fuel use alone because of the dependence on distance travelled on particulate matter generation and that generally engine power output is more consistent for highway travel.

GHGenius Model

In Canada, the GHGenius model has been developed for Natural Resources Canada, capable of analyzing emissions from conventional internal combustion engines for light duty vehicles, and class 3 - 8 heavy-duty trucks for both diesel and gasoline combustions. This comprehensive model considers many aspects of life-cycle emissions relating to vehicle operation, fuel dispensing, distribution and storage and considers all criteria air contaminants, greenhouse gases, particulate matter emissions, and non-methane organic compounds, and can be applied to an individual vehicle perspective. This model is based on an algorithm from a US EPA model, MOBILE 6.2, subsequently adapted to Canadian vehicles and environment by Environment Canada, and appears to be the most current model for calculating in-use emission from light and heavy duty motor vehicles considering Canadian fuel composition, climatic conditions, and road types. Data are obtained from Canadian sources including Statistics Canada, Natural Resources Canada, Environment Canada and the National Energy Board; Canadian Association of Petroleum Producers (CAPP) and the Canadian Gas Association (CGA) have also been used as sources of data. This model is publically available at www.ghgenius.ca

The usable portion of this comprehensive model includes the light-duty and heavy-duty emissions in g/km for diesel and gasoline engines, under the exhaust emission tab, as well as the Emissions Factors by device under Tab 'n'. Using distance travelled to estimate emissions may not accurately estimate emissions during remedial activities as equipment is often running but not travelling, and stationary equipment is therefore not considered. An estimation of emissions based on fuel input may be more appropriate, which can be seen in Table 5. The US EPA (1996a) AP-42 Emission Factors document establishes emissions factors for diesel and gasoline engines in both power output and fuel input (lb/mm Btu), similar to the emission factors produced by the GHGenius model. Based on the consideration of Canadian fuel characteristics and the variety of equipment

considered in the GHGenius model this spreadsheet tool was used in the calculation of emissions from all associated equipment. Fuel usage by equipment model and type was taken from Frey et al., 2007 and 2010, which is referenced in the model.

This model was also used to estimate other criteria air contaminants other than particulate matter including N_2O, NO_x, SO_x, volatile organic compounds, CH_4, CO and CO_2. The same process where fuel usage by equipment type was applied to the emissions factors provided by the GHGenius model for Canadian fuel specifications was used.

Equipment Process Particulate Matter Emission Modelling

Few models are available for the estimation of particulate emissions specifically from remedial processes and many of the applicable available models do not consider the adsorbent properties of petroleum hydrocarbons influencing particulate emissions during soil disturbance and processing. US EPA, Environment Canada and the Ontario Ministry of Environment all use a set of applicable equations for various remedial processes based on the publically available US EPA methods. To estimate equipment emissions of particulate from various soil handling processe are generally accepted as the definitive source of construction related particulate emissions, and have been applied for applications in this report.

Particulate emissions from all transfer operations (i.e. adding to or removing from piles, conveyor belts, truck dumping etc.), US EPA (1993):

$$E = \frac{k(0.0016)M\left(\dfrac{U}{2.2}\right)^{1..3}}{\left(\dfrac{q_w}{2}\right)^{1.4}}$$

Where:
E   = emissions (g)
k   = particle size multiplier (unitless)
M   = mass of waste handled (Kg)
U   = mean wind speed (m/sec)
 $_w$ = soil moisture content (%)
0.0016 = empirical constant (g/kg)
2.2   = empirical constant (m/sec)

Particulate emissions during mixing and tilling (waste incorporation and cultivation), US EPA (1993):

$$E = k(0.00538)SA \times 10^{-4}(s)^{0.6}$$

Were:

E   = emissions (g)
k   = particle size multiplier
10^{-4} = conversion factor (hectare/m^2)
SA   = area treated (m^2)
s   = percent silt content (%)
0.00538   = empirical constant (g/hectare)

Percent silt content was estimated from typical soil texture composition in glacial till, which generally comprises sand, clay and till representing a one third portion each. Based on this a default silt content was estimated to be 30%, however, site specific information related to soil texture would help to more accurately define this proportion and is recommended where available.

Particulate emissions during soil being spread by a bulldozer or any other tractor with a blade can be calculated as a function of time with the following equation, US EPA (1993):

$$ER_{PM10} = \frac{0.094 \times s^{1.5}}{q_w^{1.4}}$$

Where:
ER_{PM10} = PM$_{10}$ emission rate (g/sec)
s   = percent silt content (%)
$_w$ = percent moisture content (%)
0.094   = empirical constant (g/sec)

In addition to these equations, there is also published guidance on emission factors from area sources that may be applicable if the requirements for field data cannot be met and default input variables seem inappropriate. Published applicable emissions factors are available for activities relevant to this study such as: loading of excavated material into trucks, truck dumping, truck transport, debris and soil handling, soil haulage, scrapers unloading topsoil, and vehicular traffic (US EPA, 1995).

Vehicular Transport Particulate Emission Modelling

Dust emissions for both paved and unpaved roads vary with the weight of the vehicle, and the silt content (or silt loading content in paved roads); unpaved road dust emissions will also be dependent on the vehicle speed, the number of vehicle wheels, and in Canada the number of days where dust can be liberated (no snow cover). Environment Canada has published guidance on estimating road dust from industrial unpaved surfaced, adopted from US EPA (2010), with an adjustment for snow covered days, no guidance is provided for particulate emissions from paved road traffic. Ontario (1999) has published guidance on particulate emissions from vehicular traffic on paved roads, which is adopted from US EPA (2010). Particulate emissions from paved roads are due to direct emissions from vehicles in the form of exhaust, brake wear, tire emissions, and resuspension of loose material on the road surface.

Particulate emissions from resuspension of loose material on the road surface due to vehicle traffic on a dry paved road, US EPA (2010). Note that in the equation below, the weight is modified by a factor of 2.72 instead of 3 to allow for metric vehicle weights, as per Environment Canada (2009) guidance:

$$EF_{Paved1} = k \left(\frac{sL}{2} \right)^{0.98} \times \left(\frac{W}{2.72} \right)^{0.53} \times \left(\frac{S}{30} \right)^{0.16}$$

The above equation is an update, and Ontario (1999) modified an older equation to reflect snow and precipitation influences:

$$EF_{Paved2} = k \left(\frac{sL}{2} \right)^{0.65} \times \left(\frac{W}{2.72} \right)^{1.5} \times \left(\frac{365-p}{365} \right)$$

The new US EPA equation could be updated to include precipitation influences becoming:

$$EF_{Paved3} = k \left(\frac{sL}{2} \right)^{0.98} \times \left(\frac{W}{2.72} \right)^{0.53} \times \left(\frac{s}{30} \right)^{0.16} \times \left(\frac{365-p}{365} \right)$$

Where:
EF	= particulate emission factor (g/vehicle km travelled)
k	= particle size multiplier (g/vehicle km travelled)
sL	= road surface silt surface loading (g/m^2)
S	= average speed of vehicle (km/hr)
s	= percent silt content of road surface (%)
W	= mean vehicle weight (tonnes, metric)
w	= mean number of wheels per vehicle
S	= average speed (km/hr)
p	= number of days with <0.01 inches precipitation

Days with <0.01 precipitation are assumed to have no particulate emissions from the resuspension of particles. Based on this the US EPA, 2010, equation maybe the most applicable given the recent update to equation factors and the total calculation of emissions.

Particulate emissions from vehicular traffic can also be estimated using Environment Canada's, NPRI (National Pollutant Release Inventory) Toolbox, guidance on estimating road dust emissions from industrial unpaved sources. While models are available from the US EPA and the Ontario MOE, a trial run of these models produced emissions considerably higher than the model from the NPRI toolbox and was deemed inappropriate with our data inputs. Considering that the Environment Canada model is currently used to calculate national industrially relevant reportable emissions, it was applied here to calculate vehicular traffic particulate emissions.

Particulate emissions from vehicular traffic on unpaved industrial surfaces, Environment Canada, (2009):

$$EF_{Unpaved} = k \times \left(\frac{s}{12}\right)^a \times \left(\frac{W}{2.72}\right)^b$$

$$VKT = Total\ count\ of\ vehicle\ passes \times Length\ of\ Unpaved\ Road \times Operating\ Days$$

$$E_x = VKT \times EF_{Unpaved} \times ADJ$$

Where:
EF	= emission factor (g/vehicle km travelled)
k	= particle size multiplier (g/vehicle km travelled)
a, b	= numerical constant dependent on road dust particle size
s	= percent silt content of road surface (%)
W	= mean vehicle weight (tonnes, metric)
w	= mean number of wheels per vehicle
S	= average speed (km/hr)
p	= number of days with <0.01 inches precipitation
VKT	= total vehicle km travelled
ADJ	= Adjustment factor for precipitation, snow cover and frozen days (p + Snow Covered + Frozen Days)
E_x	= Particulate emissions (kg)

Modified by the Ontario Ministry of Environment (1999), accounting for number of wheels and speed:

$$EF_{Unpaved} = k \times 1.7 \times \left(\frac{s}{12}\right) \times \left(\frac{S}{48}\right) \times \left(\frac{W}{2.7}\right)^{0.7} \times \left(\frac{w}{4}\right)^{0.5} \frac{(365-p)}{365}$$

Evaluation

The models provided for particulate emissions from soil processing (US EPA, 1993) were adopted for the life cycle assessment with the rationale that currently they are really the only accepted published models for such calculations.

Both the GHGenius model and the particulate emissions from vehicular traffic model published by Environment Canada were adopted for their Canadian focus and national applicability. However, the vehicular emission models from the US EPA and the Ontario Ministry of Environment were also calculated to give context to the choice. While the basic approach is the same between the US EPA model and the Environment Canada model, the Environment Canada model applies numerical constants to the calculation dependent on road dust particle size. Environment Canada also provides useful

guidance on appropriate default values for unpaved silt road content, and further recommends the use of Environment Canada's Climate Data Online, although weather is not predicted to be an important consideration for the life-cycle assessment as most remedial activities are conducted during dry conditions. The US EPA model only calculated one number for particulate emissions (PM_{10}) and did not consider smaller particulate emission ($PM_{2.5}$), which is defined as a criteria air contaminant and was therefore not considered further. The Ontario model was published 10 years prior to the Environment Canada model, with no real change and calculated numbers are considerably higher than both the US EPA and Environment Canada model. Due to these consideration the Environment Canada particulate emissions from vehicular traffic model was selected.

The GHGenius model was used to estimate equipment emissions from engine combustion applying an emission factor. This model applied Canadian fuel specifications and applied data from Statistics Canada Data. This model was developed in consultation with Natural Resources Canada, Environment Canada and the National Energy Board; Canadian Association of Petroleum Producers (CAPP) and the Canadian Gas Association (CGA) and due to the regulatory agency consideration, the industry inclusion and Canadian focus this model was selected as appropriate for life-cycle considerations.

APPENDIX C
AMBIENT AIR QUALITY OBJECTIVES AND
OCCUPATIONAL EXPOSURE LIMITS

APPENDIX C
AMBIENT AIR QUALITY OBJECTIVES AND OCCUPATIONAL EXPOSURE LIMITS

C.1   INTRODUCTION

Ambient air exposure may result in unacceptable human health risks for occupationally exposed individuals during both ex-situ remediation and landfill disposal of contaminated soil. The largest releases of landfill gas to ambient air occur when a permeable landfill cover is used, or when an impermeable cover has been breached. Onsite human exposure occurs as a function of the actual landfill gas emission rates, while offsite exposure occurs through atmospheric dispersion of emissions.

Two main types of exposure limits in ambient air are briefly summarized here:
- Ambient air quality objectives, which are maximum concentrations in ambient air deemed acceptable by regulatory agencies, most often expressed as a 1-hour average concentration but sometimes as 15-minute, 24-hour or annual average.
- Occupational exposure limits, which are used to evaluate risks to onsite workers; normally specified as either an 8-hour average concentration or a short-term exposure limit.

In addition to these limits, tolerable concentrations and risk-specific concentrations, which are health-based limits established for long-term (chronic) exposure, may also be applicable in some cases. In general these limits would only be applied for long-term exposure or in the absence of ambient air quality objectives, however.

C.2   AMBIENT AIR QUALITY OBJECTIVES

Within Canada, many of the provinces (Alberta, Quebec, Ontario, Newfoundland and Labrador, Nova Scotia, and Nunavut) have published provincial ambient air quality objectives or standards, while other provinces (Manitoba, Yukon, Saskatchewan, the Northwest Territories, and Prince Edward Island) rely on national air ambient air quality objectives published by the Government of Canada. The number of compounds included in the ambient air objectives varies significantly by jurisdiction, and the most comprehensive documents are provided by Alberta Environment, the Ontario Ministry of Environment, and the Quebec Ministère du Développement durable, de l'Environnement et des Parcs. National AAQOs published by Health Canada are available for carbon monoxide, ground-level ozone, and particulate matter.

Alberta Environment has adopted AAQO that are protective of the environment and human health. The AAQO are used as measures of air quality, as well as for planning and management of emissions from industrial facilities.

C.3   OCCUPATIONAL EXPOSURE LIMITS

Occupation exposure limits differ from ambient air objectives in both intent and application. Occupational exposure limits are considered to be acceptable for repeated exposure of workers using a well-defined exposure term, whereas ambient air quality objectives are intended to be protective of sensitive receptors in the general population under continuous exposure. Many of the provinces have based their occupational exposure limits on threshold limit values for atmospheric contaminants published by the American Conference of Governmental Industrial Hygienists (ACGIH). These values are not standards or objectives but refer to airborne concentrations of substances under which '*it is believed that nearly all workers may be repeatedly exposed day after day without adverse health effects*' (ACGIH, 1994), referred to as threshold limit values (TLV). The ACGIH also notes that it may often be economically or technically infeasible to meet some TLVs under certain conditions. The basis of the published TLVs includes peer reviewed scientific literature as a primary source, with reviewed articles and unpublished studies as secondary sources (ACHIH, 2006). TLVs are generally expressed as an average concentration per cubic metre of air measured over a specified exposure period, and are not derived for continuous exposure situations or other exposure durations. TLVs are not intended for use as ambient air objectives, but in many cases the data reviewed to develop the TLV have been used as a starting point to from which to derive provincial ambient air objectives.

The Canadian Centre for Occupational Health and Safety (CCOHS) has published chemical profiles for 27 compounds, many of which have air exposure limits based on toxicological responses derived by other organizations. The CCHOS air exposure limits are based solely on human health endpoints and do not consider environmental effects of the chemicals. These values can also be used as a starting point to develop occupational exposure limits in the absence of published guidelines. The basis for occupational exposure limits in various provinces are described in Table C.1.

Table C.1. Basis of Provincial Occupational Exposure Limits

Alberta	Occupational exposure limits (OELs) are under the Occupational Health and Safety Act, Chemical Hazard Regulation, which requires the employer to ensure that workers are not exposed above the limits.
British Columbia	The Industrial Health and Safety Regulations set legal requirements for most of British Columbia industry, which refer to the current schedule of TLVs for atmospheric contaminants published by the American Conference of Governmental Industrial Hygienists (ACGIH).
Manitoba	The Department of Environment and Workplace Safety and Health is responsible for legislation and its administration concerning the OELs. The guidelines currently used to interpret risk to health are the ACGIH TLVs with the exception that carcinogens are given a zero exposure level "so far as is reasonably practicable".
New Brunswick	The applicable standards are those published in the latest ACGIH issue and, in case of an infraction, it is the issue in publication at the time of infraction that dictates compliance.
Northwest Territories	The Northwest Territories Safety Division of the Justice and Service Department regulates workplace safety for non-federal employees under the latest edition of the ACGIH TLVs.
Nova Scotia	The list of OELs is the same as that of the ACGIH as published in 1976 and its subsequent amendments and revisions.
Ontario	Regulations for a number of hazardous substances are enforced under the Occupational Health and Safety Act, published each in a separate booklet that includes the permissible exposure level and codes for respiratory equipment, techniques for measuring airborne concentrations and medical surveillance approaches.
Quebec	Permissible exposure levels are similar to the ACGIH TLVs and compliance with the permissible exposure levels for workplace air contaminants is required.

Taken from ILO, 2007

In Alberta an 8-hour OEL refers to the maximum concentration, averaged over an eight hours exposure period, which a sample (obtained over a period of 60 minutes) cannot exceed. The 15-minute OEL, or short-term exposure limit, cannot be exceeded over any 15-minute period. Elevated short-term exposure cannot occur more than 4 times per 8 hour shift. It may be possible to extend the OEL to a 10-hour, or extended work shift, depending on the contaminant and associated health effects.

C.3   ALBERTA AIR CONCENTRATION LIMITS

Alberta ambient air quality objectives and occupational exposure limits for key compounds are summarized in Table C.2 below.

Table C.2. Alberta Ambient Air Quality Objectives and Occupational Exposure Limits (mg/m^3)

Chemical	Ambient Air Quality Objective[a]	Occupational Exposure Limit[b]
Benzene	0.003 (1-hour average)	1.6 (8-hour average) 8 (15-minute OEL)
Toluene	1.88 (1-hour average) 0.4 (24-hour average)	188 (8-hour average)
Ethylbenzene	2 (1-hour average)	434 (8-hour average) 543 (15-minute OEL)
Xylenes	2.3 (1-hour average) 0.7 (24-hour average)	434 (8-hour average) 651 (15-minute OEL)
n-Hexane	21 (1-hour average) 7 (24-hour average)	176 (8-hour average)

a – Alberta Ambient Air Quality Objectives
b – Alberta Occupational Health and Safety Code (2009)

APPENDIX D
RECOMMENDED MODELLING APPROACH AND EQUATIONS

APPENDIX D: RECOMMENDED MODELLING APPROACH AND EQUATIONS

D.1 INTRODUCTION

This appendix presents the details of the equations used to facilitate a comparison between emissions of concern through contaminant disposal in a landfill, or contaminant removal through *ex situ* remediation. Emissions of concern included the emission of greenhouse gases, volatile organic compounds, particulate matter, NO_x, and SO_x, from remedial activities. This life cycle comparison considered emissions from the various component processes involved including equipment combustion and process emissions, as well as the direct emissions from release or transformation of contaminants. Details of the models and input parameters used to estimate emissions are presented in the sections below, and the default model input parameters applied can be seen in Table D-1.

D.2 GREENHOUSE GASE EMISSIONS FROM SOURCE CONTAMINANTS

The available carbon, eventually assumed released to the atmosphere was calculated by first estimating the mass of carbon bound within the contaminant, and was calculated as follows:

$$Carbon_{Mass} = \frac{(Carbon_{fraction} \times Soil_{Mass} \times Soil_{Concentration})}{10^6}$$

Where:

$Carbon_{Mass}$ = Mass of carbon emitted from contaminant in soil (Tonnes)
$Carbon_{fraction}$ = Relative mass portion of carbon (fraction)
$Soil_{Mass}$ = Mass of contaminated soil (kg)
$Soil_{Concentration}$ = Maximum measured soil concentration of compound (mg/kg)
10^6 = Unit conversion factor

Where:

$$Soil_{Mass} = (rb \times 1000) \times Soil_{volume}$$

$Soil_{Mass}$ = Mass of contaminated soil (kg)
b = Fine-grained, dry soil bulk density (g/cm^3)
$Soil_{Volume}$ = Volume of contaminated soil (m^3)

Where:

$$Carbon_{fraction} = \frac{(MW_{carbon} \times Carbon_{count})}{MW_{contaminant}}$$

Carbon$_{Fraction}$ = Relative mass portion of carbon (fraction)
MW$_{carbon}$ = Molecular weight of carbon (g/mol)
Carbon$_{count}$ = Number of carbon molecules in compound (g/mol)
MW$_{Contaminant}$ = Molecular weight of compound (g/mol)

From this mass of carbon, that is assumed to be released as carbon dioxide in both *ex situ* and landfill disposal remedial scenarios, the mass of carbon dioxide was calculated. In the landfill scenario the entire calculated mass of carbon is assumed to be transferred

$$CO_{2Mass} = \frac{Carbon_{mass}}{0.2727}$$

Where:

CO$_{2\,Mass}$ = Mass of carbon dioxide emitted (Tonnes)
Carbon$_{Fraction}$ = Mass of carbon emitted from contaminant in soil (Tonnes)
0.2727   = Mass percentage of carbon in carbon dioxide

D.3 GREENHOUSE GASES FROM LANDFILL EMISSIONS

Landfill methane emissions were calculated using the following formula:

$$CH_4 emissions = Carbon_{mass} \times Fraction_{CH_4} \times Fraction_{uncerta\,int\,y} \times Correction_{CH_4} \times CF$$

where:

CH$_4$emissions   = total mass of methane emitted (tonnes)
Carbon$_{Mass}$ = total mass of carbon from contaminants (tonnes)
Fraction $_{CH4}$ = fraction of methane in landfill gas
Fraction$_{uncertainty}$ = model correction factor to account for uncertainties
CF   = conversion factor, carbon to methane (1.34)
Correction$_{CH4}$ = methane correction factor, to account for uncertainties in landfill conversion from aerobic to anaerobic gas generation

Once the predicted methane emissions were calculated with the equation above, the remaining carbon from contaminants which was not converted to methane was assumed to be eventually emitted to the atmosphere as carbon dioxide and accounted for as such.

D.4 CALCULATION OF VOLATILE EXPOSURE AND EMISSIONS

D.4.1 Exposure from Contaminant Source After Disturbance

A variation of the Jury model was used to estimate air concentrations of volatiles potentially released during remedial activities. The US EPA version of the Jury model (1990) is shown below, calculating a contaminant flux at ground surface using a calculated effective diffusivity.

$$J_s = C_o \left(\frac{D_{Eff}}{\rho t} \right)^{\frac{1}{2}} \left[1 - \exp\left(\frac{-d_s^2}{4 D_{Eff} t} \right) \right]$$

where:

J_s = contaminant flux at ground surface (g/cm^2/s)
C_o = uniform contaminant concentration at t=0 (g/cm^3)
D_{Eff} = effective diffusivity (cm^2/s)
d_s = depth of uniform soil contamination from surface at t=0 (cm)
t = time (seconds)

$$D_{Eff} = \frac{\left[\dfrac{\theta_a^{10/3} D_{air} H' + \theta_w^{10/3} D_w}{n^2} \right]}{\left(\rho_b K_{oc} F_{oc} + \theta_w + \theta_a H' \right)}$$

where:

D_{eff} = effective diffusivity (cm^2/s)
θ_a = air filled porosity
n = total soil porosity
θ_w = water-filled soil porosity
D_w = diffusivity in water (cm^2/s)
D_{air} = diffusivity in air (cm^2/s)
H' = dimensionless Henry's Law constant
ρ_b = soil dry bulk density (g/cm^3)
K_{oc} = organic carbon coefficient (cm^3/g)
f_{oc} = fraction of organic carbon

To calculate a concentration in air, a box model was considered in the following equation to

conservatively estimate concentrations of volatile losses above the soil disturbance in a defined volume. Theoretically, the box works as follows, emissions are assumed to volatilize directly into a defined volume of air, this volume experiences air exchanges, which are analogous to mixing through wind influences. The use of a box model approach is conservative and the defined box volume will directly influence outcome numbers. Therefore, a realistic volume that is appropriately conservative without inflating the exposure numbers is the goal when defining the box volume. The first eight minutes were omitted from the eight-hour average as considerable uncertainty is predicted during this initial disturbance, however volatilization losses during the first 8 minutes are considered in the 15-minute exposure.

An instantaneous rate calculated using a modified equation from the Jury model, (US EPA, 1990) to omit considerations related to air exposure in a trench was conducted as follows:

$$C_{air} = \frac{J_s 10^6 A_c}{(U \times Box_h \times L)}$$

where:

C_{air}	= concentration in air ($\mu g/m^3$)
J_s	= contaminant flux at ground surface ($g/cm^2/s$)
A_c	= contaminated area (cm^2)
U	= average wind speed (m/s)
Box_h	= height of assumed box (m)
L	= length of site perpendicular to wind direction (m)

D.4.2 Exposure During Ex-Situ Allu Bucket Soil Processing

The approach for deriving an allu soil processing emission rate was to calculate two separate emission rate and adopt the highest, one rate based on soil partitioning and the other based on contaminant mass lost distributed over the allue processing time. The later calculates the contaminant emission rate, or a short-term emission rate, from soil processing using an allu bucket a function of the soil volume, concentration less the soil quality guideline and the bulk density was distributed over the Allu processing time. This equation assumes that the entire mass of contaminants, less the contaminant mass remaining in soil at the guideline concentration, is volatilized at a constant rate during the course of Allu bucket operation. The maximum contaminant concentration was be used to determine the short-term emission rate.

$$ER_{Short-term1} = \frac{(C-SQG)S_v \times b \times 1}{t_R}$$

where:

$ER_{Short-term1}$	= long-term emission rate (g/s)
SQG	= soil quality guideline ($\mu g/g$)
S_v	= volume of soil moved (m^3)

C = average contaminant concentration in soil (µg/g)

 = bulk density (g/cm^3)

1 = constant (g/10^6 µg x 10^6 cm^3/m^3)

t_r = duration of allu bucket operation (s)

A short-term emission rate calculated based on soil partitioning was also calculated, and the lower of this and ER$_{Short-term1}$ was adopted. A soil gas concentration was used to calculate the emission rate based on soil partitioning and calculated with the equation:

$$C_{Vapour} = \frac{H' \times C_{Soil} \times r_b}{q_w + K_d \times r_b + H' \times q_a}$$

where:

C_{Vapour} = soil vapour concentration (g/cm^3)

H' = henry's law constant (unitless)

C_{Soil} = soil concentration (g/g)

$_b$ = soil dry bulk density (g/cm^3)

Kd = soil-water partition coefficient (cm^3/g)

$_a$ = soil air-filled porosity (cm^3/cm^3)

This calculated soil vapour was then applied in the equation to calculate a short-term emission rate based on soil partition, calculated as:

$$ER_{Short-term2} = C_{Vapour} \times 10^6 \times q_a \times Q \times ExC$$

where:

ER$_{Short-term3}$ = short-term emission rate (g/s)

C_{Vapour} = soil vapour concentration (g/cm^3)

$_a$ = soil air-filled porosity (cm^3/cm^3)

Q = allu processing rate (m^3/s)

ExC = soil-gas to atmosphere exchange rate (m^3/sec)

The higher of the two calculated short-term emission rates is then applied to calculate a short-term ex-situ air concentration as calculated below.

Using the contaminant short-term emission rate ex-situ air concentration can be calculated. The emission rate is assumed to be occurring across the site simultaneously into the defined area of the source contaminant length and allu bucket height and dispersed with wind.

$$C_{air\_ST} = \frac{ER \times 1000 mg/g}{U \times L \times Allu_h}$$

C$_{air\_ST}$ = short-term ex-situ air concentration (mg/m^3)

ER$_{Short-term}$ = short-term contaminant emission rate (g/s)

U = average wind speed (m/s)

L   = site length perpendicular to wind direction (m)
$Allu_h$ = height of Allu bucket during operation (m)

A long-term emission rate, applicable to life-cycle assessment considerations applied the same equation but instead considered that the entire contaminant mass would be emitted. This calculation used average soil concentrations and did not consider soil quality guideline levels.

$$ER_{Long-term} = \frac{S_v \times C \times b \times 1}{t_R}$$

where:

$ER_{Long\text{-}term}$ = long-term emission rate (g/s)
S_v = volume of soil moved (m^3)
C   = average contaminant concentration in soil (µg/g)
                          = bulk density (g/cm^3)
1   = constant (g/10^6 µg x 10^6 cm^3/m^3)
t_r = duration of allu bucket operation (s)

Long-term ex-situ air concentrations from soil processing with an Allu bucket apply the same equation to calculate the short-term ex-situ air concentration with a direct substitution of the long-term emission rate for the short-term emission rate, all other inputs remain the same.

D.4.3 Exposure Reduction from Biofilters

The presence of a biofilter does not change any of the fundamental assumptions regarding the previous calculations of exposure concentration. The only change is a contaminant removal efficiency factor, the proportion of contaminants removed by the biofilter, which is applied directly to the long-term air concentration for ex-situ exposure.

$$C_{air\_BF} = C_{air} \times (1 - BEF)$$

where:

C_{air} = concentration in air (µg/m^3)
C_{air_BF} = concentration in air with biofilter (µg/m^3)
BEF = Biofilter efficiency factor (unitless)

D.5 CALCULATION OF PARTICULATE EMISSIONS

D.5.1 Particulate Emissions Generated from Vehicle Traffic

Environment Canada's NPRI (National Pollutant Release Inventory) Toolbox guidance on estimating road dust emissions from industrial unpaved sources was applied.

Particulate emissions from vehicular traffic on unpaved industrial surfaces, Environment Canada, (2009):

$$EF_{Unpaved} = k \times \left(\frac{s}{12}\right)^{a} \times \left(\frac{W}{2.72}\right)^{b}$$

Where:

EF   = emission factor (g/vehicle km travelled)
k   = particle size multiplier (g/vehicle km travelled)
s   = percent silt content of road surface (%)
W   = mean vehicle weight (tonnes, metric)
a   = numerical constant (see Table 1)
b   = numerical constant (see Table 1)

Once an emission factor has been calculated this can be applied to the total number of vehicle kilometers travelled, for this scenario it was assumed that no remedial transport activities would take place during the winter or during rain events and no dust control methods would be applied.

$$E_x = VKT \times EF$$

where:

E_x = Particulate emissions (kg)
VKT   = total vehicle kilometers travelled (km)
EF   = emission factor (g/vehicle km travelled)

D.5.2 Particulate Emissions Generated from Equipment Processes

Particulate emissions from all transfer operations (i.e. adding to or removing from piles, conveyor belts, truck dumping etc.), were calculated using US EPA (1993) as follows:

$$E = \frac{K(0.0016)M\left(\frac{U}{2.2}\right)^{1..3}}{\left(\frac{\theta_w}{2}\right)^{1.4}}$$

Where:

E   = emissions (g)
K   = particle size multiplier (unitless)
M   = mass of waste handled (kg)

U   = mean wind speed (m/s)

   w   = soil moisture content (%)
0.0016   = empirical constant (g/kg)
2.2   = empirical constant (m/s)

Particulate emissions during soil being spread by a bulldozer or any other tractor with a blade can be calculated as a function of time with the following equation from US EPA (1993):

$$ER = \frac{0.094 \times s^{1.5}}{q_w^{1.4}}$$

Where:

ER   = PM$_{10}$ emission rate (g/s)
s   = percent silt content (%)

   w   = percent moisture content (%)
0.094   = empirical constant (g/s)

Percent silt content was estimated from soil texture mapping in Alberta (Alberta Agriculture, 2005). The methodology to determine an appropriate silt default value for Alberta is as follows. The textural class grouping that covers the largest area of the province is the medium textural class. Within this textural class silt content is estimated to represent approximately 6.5%, not considering silty-loam or silty-clay loam. The silt textural class contains silt ranging in content from 70 - 100%. Based on this the silt content was estimated to be 6.5%, which conservatively assumes 100% silt content in the silt textural class, but does not consider potential silt contributions in other textural classes.

D.5.3 Particulate Emissions Generated from Engine Combustion

Particulates are also generated as a combustion by-product and were calculated using emission factors published by the GHGenius model, applied with fuel usage based on equipment type published in Frey et al. (2010). The calculation is as a follows:

$$EQ = \frac{EF \times F}{E_1 \times H}$$

where:

EQ   = particulate emissions from engine combustion (g)

EF   = equipment specific particulate emission factor based on fuel use (g/GJ)
F   = equipment specific fuel use (L/hr)
H   = hours of equipment operation (hr)
E_1 = petro-diesel required to generate 1 GJ energy (L/GJ)

Hours of equipment use were estimated based on professional experience.

D.6 CALCULATION OF COMBUSTION RELATED VOLATILE EMISSIONS

Equipment engine emissions of volatiles were also estimated using emission factors from the GHGenius model and applied with fuel use, the calculation is as follows:

$$E_{Engine} = \frac{F \times H \times EF \times E_2}{1000}$$

where:

E_{Engine} = emissions of criteria air contaminants as engine combustion products (kg)
F   = equipment specific fuel use (L/hr)
H   = hours of equipment operation (hr)
EF   = equipment specific criteria air contaminant emission factor based on fuel use (g/GJ)
E_2 = energy released per liter of petro-diesel (GJ/L)

D.7 INPUT PARAMETERS

Default input parameters that were used specifically for model related inputs, not necessarily site dependent can be found in Table D-1 below.

TABLE D-1
DEFAULT MODEL INPUTS

Vehicular Traffic and Equipment Process Particulate Emissions						
Parameter	**Symbol**	**Units**	**$PM_{2.5}$**	**PM_{10}**	**TPM**	**References / Notes**
Particle size multiplier - unpaved	k	kg/VKT	0.042	0.42	1.38	Environment Canada, 2009
Numerical constant	a	unitless	0.9	0.9	0.7	
Numerical constant	b	unitless	0.45	0.45	0.45	
Particle size multiplier - soil handling	K	dimensionless	0.11	0.35	-	US EPA, 1993
Parameter			**Default**	**Range**		
Mean wind speed	U	m/sec	4			Environment Canada, 2008
Soil silt content	s	%	30			Estimated based on glacial till texture portioning.
Surface road material silt content	S	%	6.65	3.2	12.9	US EPA, 2006 (see below)
Mean vehicle weight	W	Tonnes	23			Maximum Alberta commerical weight 53.5 tonnes, 23 is the max for a tridem axle, more probable.

Silt Content (%) in Surface Materal, Industrial Unpaved Roads						
Industry	**Road Use**	**N**	**Mean**	**Range**		
Municipal solid waste landfills	Disposal routes	20	6.4	2.2	21	US EPA, 2006 Table 13.2.2-1
Construction Sites	Scraper routes	20	8.5	0.56	23	
Western Surface coal mining	Plant road	2	5.1	4.9	5.3	
Sand and gravel processing	Plant road	3	4.8	4.1	6	
Taconite mining and processing	Service road	8	4.3	2.4	7.1	
Taconite mining and processing	Haul road	12	5.8	3.9	9.7	
Stone quarrying and processing	Plant road	10	10	2.4	16	
Stone quarrying and processing	Haul road	20	8.3	5	15	
Mean silt content	All	-	6.65	3.18	12.89	Adopted as default

Combustion Engine Particulate Emissions						
Energy content in petro-diesel	E1	L/GJ	25.6			GHGenius
Energy content in petro-diesel	E2	GJ/L	0.039			
Wheeled loader	EF1	g/GJ	42.27			
Off-road trucks	EF2	g/GJ	41.29			
Tractor	EF3	g/GJ	97.27			
Industrial Engine	EF4	g/GJ	80.10			
Mass excavation using an excavator	F1	L/hr	7.19			Frey et al., 2010
Material handling using a front-end loader	F2	L/hr	4.67			
Power generation using a generator	F3	L/hr	4.54			
Contaminated soil transport (km)	F4	L/hr	7.57			

Combustion Engine Criteria Air Contaminant Emissions[a]			**CH_4**	**CO**	**N_2O**	**NO_x (NO_2)**	**SO_x (SO_2)**	**CO_2**	**VOC**
Wheeled loader	EF5	g/GJ fuel used	1.38	138.58	28.60	223.30	6.54	68,158	27.02
Off-road trucks	EF6	g/GJ fuel used	1.38	79.46	28.60	280.42	6.54	68,253	27.02
Tractor	EF7	g/GJ fuel used	1.20	194.02	28.60	359.59	6.54	67,849	48.46
Industrial engine	EF8	g/GJ fuel used	4.15	199.55	28.60	439.41	6.54	67,889	45.74

a - Emission factors taken from GHGenius model

APPENDIX E
EXAMPLE SITE CALCULATIONS

APPENDIX E: EXAMPLE SITE CALCULATIONS

E.1 APPROACH

An example site and scenario was created to test the modeling approach for both an ex situ remediation, with and without a biofilter, and a landfill disposal scenario. While based on data from a real site where ambient air quality monitoring was conducted during ex situ remediation, for purposes of this report the site is not identified or considered to be at a specific location. This example site reflects soil hydrocarbon concentrations thought to be applicable to a range of contaminated sites, and many of the assumptions are attempted to be as widely applicable as possible. For the evaluation of landfill disposal, the site is assumed to be located approximately 100 km from the nearest appropriate landfill disposal facility, connected by gravel roads. Approximately 800 m^3 of contaminated soil are estimated to be present onsite, based on an affected area of 400 m^2 to a depth of 2 m. Soils were assumed to have the same characteristics as a fine-grained soil as define by the Canadian Council of Ministers for the Environment (CCME, 2006).

All site-related assumptions and equipment operational hours for each scenario are presented in Table E-1. The equipment assumed to be present during soil processing for both ex situ and landfill disposal, with respective emission factors and default model inputs are presented in Table E-2. Chemical-specific parameters used for the calculations are provided in Table E-3. Contaminant concentrations in soil are provided in Table E-4; the average concentrations were used to calculate lifecycle emissions of greenhouse gases, and maximum soil concentrations were used to calculate worst-case volatile exposures onsite.

For the ex situ remediation scenario, it was assumed that the contaminated soil volume, 800 m^2, would be first excavated using a backhoe, then passed through an allu bucket twice before being placed back into the excavation pit and leveled using a front-end loader to resemble pre-disturbance surface conditions. If a biofilter was used, it was assumed to be present over excavated material. For the landfill disposal scenario, it was assumed this same volume of soil would be excavated using a backhoe and placed directly into a haul truck for transport to a landfill disposal facility. Once at the landfill, it was assumed this soil volume would be incorporated into the landfill cell using a backhoe. Both scenarios were assumed to use a site generator.

Workers in both the landfill and ex situ scenario are predicted to be exposed to worst case air concentrations during the initial soil excavation; however, the ex situ worker may receive an increased relative exposure as this worker is predicted to be exposed longer.

E.2 WORKED EXAMPLE CALCULATIONS

Worked example calculations for greenhouse gas emissions, criteria air contaminants including particulates and volatile exposure can be found below.

E.2.1 Greenhouse Gas Emissions from Source contaminants - Benzene

$$Soil_{Mass} = \rho b \times Soil_{volume}$$

$$Soil_{Mass} = 1.4 \frac{g}{cm^3} \times 1000 \times 800 m^3 = 1120000 kg$$

Where:

$Soil_{Mass}$	= Mass of contaminated soil (kg)
b	= Fine-grained, dry soil bulk density (g/cm³)
$Soil_{Volume}$	= Volume of contaminated soil (m³)

$$Carbon_{fraction} = \frac{(MW_{carbon} \times Carbon_{count})}{MW_{contaminant}}$$

$$Carbon_{fraction} = \frac{(12.0107 g/mol \times 6)}{78.1 g/mol} = 0.92$$

$Carbon_{Fraction}$	= Relative mass portion of carbon (fraction)
MW_{carbon}	= Molecular weight of carbon (g/mol)
$Carbon_{count}$	= Number of carbon molecules in compound (g/mol)
$MW_{Contaminant}$	= Molecular weight of benzene (g/mol)

$$Carbon_{Mass} = \frac{(Carbon_{fraction} \times Soil_{Mass} \times Soil_{Concentration})}{10^6}$$

$$Carbon_{Mass} = \frac{(0.92 \times 1120000 kg \times 0.49 mg/kg)}{10^6} = 0.51 Tonnes$$

Where:

$Carbon_{Mass}$	= Mass of carbon emitted from contaminant in soil (Tonnes)
$Carbon_{fraction}$	= Relative mass portion of carbon (fraction)
$Soil_{Mass}$	= Mass of contaminated soil (kg)
$Soil_{Concentration}$	= Maximum measured soil concentration of benzene (mg/kg)
10^6	= Conversion factor

$$CO_{2 Mass} = \frac{Carbon_{mass}}{0.2727}$$

$$CO_{2 Mass} = \frac{0.51 tonnes}{0.2727} = 1.9 tonnes$$

Where:

$CO_{2\,Mass}$	= Mass of carbon dioxide emitted from benzene in soil (Tonnes)
$Carbon_{Fraction}$	= Mass of carbon emitted from benzene in soil (Tonnes)
0.2727	= Mass percentage of carbon in carbon dioxide

E.2.2 Greenhouse Gases from Landfill Emissions

Using the equation to calculate $Carbon_{Mass}$ for benzene shown above, the carbon mass for all contaminants was estimated and summed, giving 1556 tonnes. Landfill methane emissions were then calculated using the following formula:

$$CH_4 emissions = Carbon_{mass} \times Fraction_{CH_4} \times Fraction_{uncertainty} \times Correction_{CH_4} \times CF$$

$$CH_4 emissions = 1556 tonnes \times 0.5 \times 0.9 \times 0.5 \times 1.3 = 445 tonnes$$

where:

$Carbon_{Mass\_total}$	= total mass of carbon from contaminants (Tonnes)
$Fraction_{CH4}$	= fraction of methane in landfill gas
$Fraction_{uncertainty}$	= model correction factor to account for uncertainties
CF	= conversion factor, carbon to methane
$Correction_{CH4}$	= Methane correction factor, to account for uncertainties in landfill conversion from aerobic to anaerobic gas generation

E.2.3 Calculation of Volatile Emissions from Source- Benzene

$$D_{Eff} = \frac{\left[\dfrac{q_a^{10/3} D_{air} H' + q_w^{10/3} D_w}{n^2}\right]}{\left(r_b K_{oc} F_{oc} + q_w + q_a H'\right)}$$

$$D_{Eff} = \frac{\left[\dfrac{0.302^{10/3} 0.088 \times 0.225 + 0.168^{10/3} \times 0.0000098}{0.47^2}\right]}{\left(1.4 \times 79.4 \times 0.005 + 0.168 + 0.302 \times 0.225\right)} = 0.0021 cm^2 / s$$

where:

D_{eff} = effective diffusivity (cm²/s)

θ_a = air filled porosity (cm³/cm³)

n = total soil porosity (cm³/cm³)

θ_w = water-filled soil porosity (cm³/cm³)

D_w = diffusivity in water (cm²/s)

D_{air} = diffusivity in air (cm²/s)

H' = dimensionless Henry's Law constant (unitless)

ρ_b = soil dry bulk density (g/cm³)

K_{oc} = organic carbon coefficient (cm³/g)

f_{oc} = fraction of organic carbon

$$J_s = C_o \left(\frac{D_{Eff}}{\rho t} \right)^{\frac{1}{2}} \left[1 - \exp\left(\frac{-d_s^2}{4 D_{Eff} t} \right) \right]$$

Instantaneous contaminant flux at 1 second after disturbance:

$$J_s = 6.88 \times 10^{-7} \left(\frac{0.0021}{3.14 \times 1} \right)^{\frac{1}{2}} \left[1 - \exp\left(\frac{-200^2}{4 \times 0.0021 \times 1} \right) \right] = 1.78 \times 10^{-8} \, \text{g/cm}^2\text{s}$$

where:

J_s = contaminant flux at ground surface (g/cm²s)

C_o = uniform contaminant concentration at t=0 (g/cm³)

D_{Eff} = effective diffusivity (cm²/s)

d_s = thickness of uniform soil contamination at t=0 (cm)

t = time (seconds)

$$C_{air} = \frac{J_s 10^6 A_c}{(U \times Box_h \times L)}$$

Instantaneous air concentration at 1 second after disturbance:

$$C_{air} = \frac{1.78 \times 10^{-8} g/cm^2 s \times 10^6 \mu g/g \times 4000000 cm^2}{4m/s \times 2m \times 20m} = 4.45 \times 10^2 ug/m^3$$

where:

C_{air}	= concentration in air ($\mu g/m^3$)
J_s	= contaminant flux at ground surface ($g/cm^2 s$)
A_c	= contaminated area (cm^2)
U	= average wind speed (m/s)
Box_h	= height of assumed box (m)
L	= length of site perpendicular to wind direction (m)

E.2.4 Calculation of Volatile Emissions from Ex-Situ Soil Treatment with an Allu Bucket

Two emission rates were calculated as, shown for benzene:

$$ER_{Short-term1} = \frac{(C - SQG)S_v \times b \times 1}{t_R}$$

$$ER_{Short-term1} = \frac{(0.49mg/kg - 0.046mg/kg)800m^3 \times 1.4g/cm^3 \times 1}{7.1h \times 3600s/h} = 0.139g/s$$

where:

$ER_{Short-term1}$	= long-term emission rate (g/s)
SQG	= soil quality guideline ($\mu g/g$)
S_v	= volume of soil moved (m^3)
C	= average contaminant concentration in soil ($\mu g/g$)
	= bulk density (g/cm^3)
1	= constant ($g/10^6 \mu g \times 10^6 cm^3/m^3$)
t_r	= duration of allu bucket operation (s)

$$C_{Vapour} = \frac{H' \times C_{Soil} \times r_b}{q_w + K_d \times r_b + H' \times q_a}$$

$$C_{Vapour} = \frac{0.225 \times 0.00000317 g/g \times 1.4g/cm^3}{0.168 + 0.397 cm^3/g \times 1.4g/cm^3 + 0.225 \times 0.302} = 0.00000126 g/cm^3$$

where:

C_{Vapour} = soil vapour concentration (g/cm^3)
H' = henry's law constant (unitless)
C_{Soil} = soil concentration (g/g)
$_b$ = soil dry bulk density (g/cm^3)
Kd = soil-water partition coefficient (cm^3/g)
$_a$ = soil air-filled porosity (cm^3/cm^3)

$$ER_{Short-term2} = C_{Vapour} \times 10^6 \times q_a \times Q \times ExC$$

$$ER_{Short-term2} = 0.00000126 g/cm^3 \times 10^6 \times 0.302 \times 112.5 m^3/hr \times \frac{1hr}{3600s} \times 1m^3/s = 0.0119 g/s$$

where:

$ER_{Short-term3}$ = short-term emission rate (g/s)
C_{Vapour} = soil vapour concentration (g/cm^3)
$_a$ = soil air-filled porosity (cm^3/cm^3)
Q = allu processing rate (m^3/s)
ExC = soil-gas to atmosphere exchange rate (m^3/sec)

Based on the above calculations two equations calculating short-term emission rates, the higher $ER_{Short-term1}$ was adopted and applied to calculate an outdoor air concentration from ex-situ Allu bucket soil processing, which was calculated as:

$$C_{air\_ex-situ\_ST} = \frac{ER_{Short-term1} \times 1000}{U \times L \times Allu_h}$$

$$C_{air\_ex-situ\_ST} = \frac{0.139 g/s \times 1000 mg/g}{4m/s \times 20m \times 2m} = 0.868 mg/m^3$$

where:

$C_{air\_ex\text{-}situ\_ST}$ = short-term ex-situ air concentration (mg/m^3)
ER   = short-term contaminant emission rate (g/s)
U   = average wind speed (m/s)
L   = site length perpendicular to wind direction (m)
$Allu_h$ = assumed box height, equal to height of Allu bucket during operation (m)

A long-term emission rate was also calculated, more appropriate for life-cycle considerations and was calculated as:

$$ER_{Long-term} = \frac{S_v \times C \times \text{b} \times 1}{t_R}$$

$$ER_{Short-term1} = \frac{0.49 mg/kg \times 800 m^3 \times 1.4 g/cm^3 \times 1}{7.1 h \times 3600 s/h} = 0.0215 g/s$$

where:

$ER_{Long-term}$ = long-term emission rate (g/s)
S_v = volume of soil moved (m^3)
C   = average contaminant concentration in soil (µg/g)
     = bulk density (g/cm^3)
1   = constant (g/10^6 µg x 10^6 cm^3/m^3)
t_r = duration of allu bucket operation (s)

The calculation of outdoor air considering long-term soil emissions:

$$C_{air\_ex-situ\_LT} = \frac{ER_{Long-term} \times 1000}{U \times L \times Allu_h}$$

$$C_{air\_ex-situ\_LT} = \frac{0.0215 g/s \times 1000 mg/g}{4 m/s \times 20 m \times 2 m} = 0.134 mg/m^3$$

where:

$C_{air\_ex\text{-}situ\_LT}$ = long-term ex-situ air concentration (mg/m^3)
ER = long-term contaminant emission rate (g/s)
U = average wind speed (m/s)
L = site length perpendicular to wind direction (m)
$Allu_h$ = assumed box height, equal to height of Allu bucket during operation (m)

For the biofilter case, the long-term air concentration was modified using a biofilter efficiency factor, using the following calculation:

$$C_{air\_BF} = C_{air\_ex-situ\_LT} \times (1 - BEF)$$

$$C_{air\_BF} = 0.134 mg/m^3 \times (1 - 0.6) = 0.054 mg/m^3$$

where:

$C_{air\_ex\text{-}situ\_LT}$ = long-term ex-situ air concentration (mg/m^3)
C_{air_BF} = long-term ex-situ air concentration with a biofilter (mg/m^3)
BEF = Biofilter efficiency factor (BEF)

E.2.5 Calculation of Particulate Emissions

E.2.5.1 Particulate Emissions Generated from Vehicle Traffic

$$EF_{Unpaved} = k \times \left(\frac{s}{12}\right)^a \times \left(\frac{W}{2.72}\right)^b$$

Fine-grained, particulate matter 2.5 micrometers or less:

$$EF_{Unpaved} = 0.042 kg/VKT \times \left(\frac{6.5}{12}\right)^{0.9} \times \left(\frac{23}{2.72}\right)^{0.45} = 0.0645 kg/VKT$$

Where:

EF = emission factor (kg/vehicle km travelled)
k = particle size multiplier (kg/vehicle km travelled)
s = percent silt content of road surface (%)
W = mean vehicle weight (tonnes, metric)
a = numerical constant
b = numerical constant

$$E_x = VKT \times EF$$

$$E_x = 125km \times 0.0645kg/km = 8.06kg$$

where:

E_x = Particulate emissions (kg)
VKT = total vehicle kilometers travelled (km)
EF = emission factor (kg/vehicle km travelled)

E.2.5.2 Particulate Emissions Generated from Equipment Processes

$$E = \frac{K(0.0016)M\left(\dfrac{U}{2.2}\right)^{1..3}}{\left(\dfrac{\theta_w}{2}\right)^{1.4}}$$

Emissions from all transfer operations PM2.5:

$$E = \frac{0.052(0.0016g/kg)1120000kg\left(\dfrac{4.4m/s}{2.2m/s}\right)^{1..3}}{\left(\dfrac{0.168}{2}\right)^{1.4}}$$

$$E = \frac{0.11(0.0016g/kg)1120000kg\left(\dfrac{4m/s}{2.2m/s}\right)^{1..3}}{\left(\dfrac{0.168}{2}\right)^{1.4}} = 13700g$$

Where:

E = emissions from soil transfer operations (g)

K   = particle size multiplier (unitless)
M   = mass of waste handled (kg)
U   = mean wind speed (m/s)
$_w$ = soil moisture content (%)
0.0016   = empirical constant (g/kg)
2.2   = empirical constant (m/s)

$$ER = \frac{0.094 \times s^{1.5}}{q_w^{1.4}}$$

Emissions from soil spreading, PM10:

$$ER = \frac{0.094\,g\,/s \times 0.30^{1.5}}{0.168^{1.4}} = 0.19\,g\,/s$$

Where:

ER   = PM_{10} emission rate (g/s)
s   = percent silt content (%)
$_w$ = percent moisture content (%)
0.94   = empirical constant (g/s)

E.2.5.3 Particulate Emissions Generated from Engine Combustion

$$EQ = \frac{EF \times F}{E_1 \times H}$$

Excavator engine emissions during ex situ excavation and allu:

$$EQ = \frac{97.27g\,/GJ \times 7.19L\,/hr}{25.6L\,/GJ \times 24hr} = 655.7g$$

where:

EQ   = particulate emissions from engine combustion (g)
EF   = equipment specific particulate emission factor based on fuel use (g/GJ)
F   = equipment specific fuel use (L/hr)

H   = hours of equipment operation (hr)
E_1 = petro-diesel required to generate 1 GJ energy (L/GJ)

E.2.6 Calculation of Combustion Related Volatile emissions

Equipment engine emissions of volatiles were also estimated using emission factors from the GHGenius model and applied with fuel use, the calculation is as follows:

$$E_{Engine} = \frac{F \times H \times EF \times E_2}{1000}$$

Excavator ex situ methane emissions:

$$E_{Engine} = \frac{7.19 L/hr \times 24 hr \times 1.2 g/GJ \times 0.039 GJ/L}{1000} = 0.008 kg$$

where:

E_{Engine} = emissions of criteria air contaminants as engine combustion products (kg)
F   = equipment specific fuel use (L/hr)
H   = hours of equipment operation (hr)
EF   = equipment specific criteria air contaminant emission factor based on fuel use (g/GJ)
E_2 = energy released per liter of petro-diesel (GJ/L)

E.3 RESULTS

The results of the calculations are presented in Tables E-5 through E-9. Based on the calculations, more methane is produced in the landfill disposal scenario than in the *ex-situ* soil treatment scenario; landfill disposal is predicted to produce more particulate emissions, largely a result of distance to landfill, and landfill disposal is predicted to have higher criteria air contaminant emissions based on predicted higher equipment operational hours. Exposures from the soil source after disturbance, not considering soil processing, are very similar for both scenarios and highest immediately after soil disturbance. For this example site, the volatile exposures from ex-situ soil treatment with the allu bucket are not predicted to exceed occupational exposure limits, neither is exposure from the disturbed soil source for both the ex-situ or landfill disposal scenarios. Utilization of a biofilter is reduces long-term ex-situ exposure as a basic model assumption and always provides a reduction to long-term air concentration as long as the biofilter is operating. The use of a biofilter increased engine combustion emissions of criteria air contaminants, but the contribution of was equivalent to less than 1% of overall equipment emissions.

E.4 INDUSTRIAL HYGIENE ASSESSMENT

An industrial hygiene assessment was conducted at this example site, involving instantaneous spot measurements on equipment where workers would likely be exposed. Total volatile organic compounds ranged from 0.4 ppm to 24 ppm during the excavation, and no measurable concentrations of benzene were detected (<0.01 ppm). These results are below the modelled predicted exposures, suggesting that the modelling of volatile exposures is conservative and may over-predict exposures.

E.5 DISPERSION MODELLING AND PREDICTION OF OFFSITE CONCENTRATIONS

Parameters calculated by the emission model can be utilized as inputs for air dispersion modelling software, including models recommended by Alberta Environment such as AERSCREEN, CALPUFF, and AERMOD-PRIME. Use of these dispersion models allows for predictions of emission behaviour based on local topological and meteorological condition, as well as prediction of pollutant concentrations at critical receptor locations.

For demonstration purposes, results of the emission model from an allu bucket at the example site were used as inputs in the AERSCREEN dispersion model. This is a screening model that utilizes a generic set of site conditions and calculates a worst-case concentration at regular distance intervals from the site, as well as the maximum 1-hour, 3-hour, 8-hour, 24-hour, and scaled annual concentrations. The meteorological conditions that resulted in the maximum predicted concentrations at each distance are also determined and can potentially be used as a risk management tool. The required inputs included the rate of short-term benzene emissions from soil being processing using an Allu bucket (0.139 g/s, detailed in section 2.4), the volume of soil contained in the bucket (2 m^3), and the height Allu bucket emissions (2 m). Selected results are shown below:

Distance (m)	Maximum Predicted Benzene Concentration (mg/m^3)
3.2	5.2
50	1.6
100	0.93
250	0.42
500	0.21
1000	0.093
2000	0.038
5000	0.011

As this specific model is a screening tool, the results obtained are fairly generic. More detailed results can be obtained when refined dispersion models are utilized and additional site specific data is provided. Refined results could include determination of exposure concentrations, exceedances of short-term exposure limits at specific locations, maps of the emission plume, or

various other outputs. The required meteorological data sets to run refined models is publicly available, and detailed user guidance on running these models is provided by Alberta Environment and other regulatory organizations.

TABLE E.1
EXAMPLE SITE CHARACTERISTICS & PROCESS ASSUMPTIONS

Parameter	Symbol	Unit	Number	Reference
Volumetric air content or vapour-filled porostiy	a	cm^3/cm^3	0.302	Fine-grained soil CCME (2006)
Volumetric water content or soil moisture content	w	cm^3/cm^3	0.168	Fine-grained soil CCME (2006)
Total soil porosity	n	cm^3/cm^3	0.47	Fine-grained soil CCME (2006)
Soil dry bulk density	b	g/cm^3	1.4	Fine-grained soil CCME (2006)
Soil dry bulk density	b	kg/m^3	1400	Fine-grained soil CCME (2006)
Fraction of organic carbon	f_{oc}	unitless	0.005	Fine-grained soil CCME (2006)
Thickness of source soil	ds	cm	200	Assumed
Contaminated area	Ac	m^2	400	20 m by 20 m
Contaminated soil volume	V	m^3	800	site specific input
Mass of contaminated soil	Soil	kg	1120000	calculated
Contaminant thickness	Cd	cm	200	Assumed
Box defined area	Box_{area}	m^2	400	Assumed
Box defined height	Box_h	m	2	Assumed

Equipment Operational Inputs				
Process		Unit	Operational Hours Ex Situ Scenario	Operational Hours Landfill Disposal Scenario
Material Handling using a front end loader		hr	6	12
Mass excavation using a backhoe, including allu		hr	24	12
Power generation using a generator		hr	24	24
Contaminated soil transport (km)		km	25	125

Vehicular Traffic and Equipment Process Particulate Emissions						
Parameter	**Symbol**	**Units**	**PM$_{2.5}$**	**PM$_{10}$**	**TPM**	**References / Notes**
Particle size multiplier - unpaved	k	kg/VKT	0.042	0.42	1.38	Environment Canada, 2009
Numerical constant	a	unitless	0.9	0.9	0.7	
Numerical constant	b	unitless	0.45	0.45	0.45	
Particle size multiplier - soil handling	K	dimensionless	0.11	0.35	-	US EPA, 1993
Parameter			**Default**	**Range**		
Mean wind speed	U	m/sec	4			Environment Canada, 2008
Soil silt content	s	%	30			Estimated based on glacial till texture portioning.
Surface road material silt content	S	%	6.65	3.2	12.9	US EPA, 2006 (see below)
Mean vehicle weight	W	Tonnes	23			Maximum Alberta commerical weight 53.5 tonnes, 23 is the max for a tridem axle, more probable.
Silt Content (%) in Surface Materal, Industrial Unpaved Roads						
Industry	**Road Use**	**N**	**Mean**	**Range**		
Municipal solid waste landfills	Disposal routes	20	6.4	2.2	21	US EPA, 2006 Table 13.2.2-1
Construction Sites	Scraper routes	20	8.5	0.56	23	
Western Surface coal mining	Plant road	2	5.1	4.9	5.3	
Sand and gravel processing	Plant road	3	4.8	4.1	6	
Taconite mining and processing	Service road	8	4.3	2.4	7.1	
Taconite mining and processing	Haul road	12	5.8	3.9	9.7	
Stone quarrying and processing	Plant road	10	10	2.4	16	
Stone quarrying and processing	Haul road	20	8.3	5	15	
Mean silt content	All	-	6.65	3.18	12.89	Adopted as default
Combustion Engine Particulate Emissions						
Energy content in petro-diesel	E1	L/GJ	25.6			GHGenius
Energy content in petro-diesel	E2	GJ/L	0.039			
Wheeled loader	EF1	g/GJ	42.27			
Off-road trucks	EF2	g/GJ	41.29			
Tractor	EF3	g/GJ	97.27			
Industrial Engine	EF4	g/GJ	80.10			
Mass excavation using an excavator	F1	L/hr	7.19			Frey et al., 2010
Material handling using a front-end loader	F2	L/hr	4.67			
Power generation using a generator	F3	L/hr	4.54			
Contaminated soil transport (km)	F4	L/hr	7.57			

Combustion Engine Criteria Air Contaminant Emissions[a]			**CH$_4$**	**CO**	**N$_2$0**	**NO$_x$ (NO$_2$)**	**SO$_x$ (SO$_2$)**	**CO$_2$**	**VOC**
Wheeled loader	EF5	g/GJ fuel used	1.38	138.58	28.60	223.30	6.54	68,158	27.02
Off-road trucks	EF6	g/GJ fuel used	1.38	79.46	28.60	280.42	6.54	68,253	27.02
Tractor	EF7	g/GJ fuel used	1.20	194.02	28.60	359.59	6.54	67,849	48.46
Industrial engine	EF8	g/GJ fuel used	4.15	199.55	28.60	439.41	6.54	67,889	45.74

Biofilter Inputs									
Biofilter Efficiency Factor	BEF	%		60		Meridian, 2012			

TABLE E.3
CHEMICAL PARAMETERS

Parameter Name	Symbol	Units	Benzene	Toluene	Ethyl-benzene	Xylenes	Aliphatic F1 (C_6-C_8)	Aliphatic F1 (C_8-C_{10})	Aromatic F1 (C_8-C_{10})	Aliphatic F2 (C_{10} - C_{12})	Aliphatic F2 (C_{12} - C_{16})	Aromatic F2 (C_{10} - C_{12})	Aromatic F2 (C_{12} - C_{16})	Source
Henry's Law constant	H_i	atm-m^3/mol	5.50E-03	6.71E-03	8.75E-03	7.30E-03	1.22E+00	1.96E+00	1.20E-02	2.94E+00	1.27E+01	3.40E-03	1.30E-03	Health Canada (2009)
Dimensionless Henry's Law constant	H'	unitless	2.25E-01	2.74E-01	3.58E-01	3.00E-01	5.00E+01	8.00E+01	4.80E-01	1.20E+02	5.20E+02	1.40E-01	5.30E-02	Health Canada (2009)
Diffusivity in water	D_w	cm^2/s	9.80E-06	8.60E-06	7.80E-06	7.80E-06	1.00E-05	1.00E-05	1.00E-05	1.00E-05	1.00E-05	1.00E-05	1.00E-05	Health Canada (2009)
Diffusion coefficient in air	D_{air}	cm^2/s	8.80E-02	8.70E-02	7.50E-02	7.00E-02	5.00E-02	5.00E-02	5.00E-02	5.00E-02	5.00E-02	5.00E-02	5.00E-02	Health Canada (2009)
Molar mass of carbon in compound	Carbon Molar Mass	g/mol	72.0642	84.0749	96.0856	96.0856	108.0963	108.0963	108.0963	132.1177	168.1498	120.107	144.1284	Carbon count x Molar mass of carbon
Molecular weight of component i	MW_i	g/mol	7.81E+01	9.21E+01	1.06E+02	1.06E+02	1.00E+02	1.30E+02	1.20E+02	1.60E+02	2.00E+02	1.30E+02	1.50E+02	ORNL (2010) online RAIS database, Health Canada (2009b).
Relative mass portion of carbon	Carbon$_{Fraction}$	fraction	0.92	0.91	0.91	0.91	1.08	0.83	0.90	0.83	0.84	0.92	0.96	Calculated
Organic Carbon - Water Partition coefficient	K_{oc}	cm^3/g	7.94E+01	1.51E+02	2.95E+02	4.10E+02	3.98E+03	3.16E+04	1.58E+03	2.51E+05	5.01E+06	2.51E+03	5.01E+03	Health Canada (2009b)
log Octanol-water partition coefficient	log K_{ow}	unitless	2.13	2.69	3.13	3.12	4.11	5.21	3.69	6.3	7.95	3.93	4.29	Health Canada (2009b)
Fraction of Petroleum Hydrocarbon Subfraction in Soil	$F_{i(soil)}$	unitless					0.55	0.36	0.09	0.36	0.44	0.09	0.11	CCME, 2008
Maximum concentration of contaminant in soil	C_s	μg/g	3.22	0.09	227	246	1837	1202	301	16992	20768	4248	5192	Maximum value
Maximum concentration of contaminant in soil	C_o	g/cm^3	0.004508	0.000126	0.3178	0.3444	2.5718	1.68336	0.42084	23.7888	29.0752	5.9472	7.2688	Maximum value
Average concentration of contaminant in soil	C_s	μg/g	0.49	0.04	17	16	152.35	99.72	25	458	559.68	114.48	139.92	Average value
Average concentration of contaminant in soil	C_o	g/cm^3	0.000687831	0.000050225	0.0233977	0.022252588	0.21329	0.139608	0.034902	0.641088	0.783552	0.160272	0.195888	Average value

Notes   Not accounting for petroleum hydrocarbon subfraction contributions from soil vapour or groundwater contributions.

TABLE E.4
EXAMPLE SOIL ANALYTICAL RESULTS

Depth (m)	Benzene mg/kg	Toluene mg/kg	Ethylbenzene mg/kg	Xylenes mg/kg	F1-BTEX mg/kg	F2 mg/kg	F3 mg/kg	F4 mg/kg
Maximum	3.22	0.09	227	246	3340	47200	150000	19400
Average	0.49	0.04	17	16	277	1272	4029	1307
0.6-1.0	0.004	<0.005	<0.010	<0.010	<4	174	921	414
5.75-6.0	<0.004	<0.005	<0.010	<0.010	<4	14	118	26
1.0-1.5	<0.004	<0.005	<0.010	<0.010	12	437	2480	1060
5.75-6.0	<0.004	<0.005	<0.010	<0.010	<4	27	168	29
1.0-1.5	<0.004	<0.005	<0.010	<0.010	<4	86	1150	502
2.75-3.0	<0.004	<0.005	<0.010	<0.010	<4	19	47	<20
0.6-1.0	<0.004	<0.005	<0.010	<0.010	<4	20	<30	<20
2.75-3.0	<0.004	<0.005	<0.010	<0.010	<4	<10	<30	<20
0.6-1.0	<0.004	<0.005	<0.010	<0.010	<4	120	845	392
2.75-3.0	<0.004	<0.005	<0.010	<0.010	<4	<10	40	<20
1.0-1.5	<0.004	<0.005	<0.010	<0.010	<4	132	591	248
4.25-4.5	<0.004	<0.005	<0.010	<0.010	<4	132	267	37
0.3-0.6	<0.004	<0.005	<0.010	<0.010	<4	128	993	492
2.75-3.0	<0.004	<0.005	<0.010	<0.010	<4	<10	40	<20
0-0.3	<0.004	<0.005	<0.010	<0.010	5	201	2000	1010
2.75-3.0	<0.004	<0.005	<0.010	<0.010	<4	14	<30	<20
1.5	0.24	< 0.02	0.80	1.80	458	< 10	47	71
1	< 0.02	< 0.02	< 0.02	< 0.02	< 0.1	< 10	356	266
1.0	< 0.02	< 0.02	< 0.02	< 0.02	< 0.1	< 10	< 10	< 10
1.0	1.79	< 0.02	2.47	8.38	145	< 10	< 10	< 10
1.4	< 0.02	< 0.02	< 0.02	< 0.02	< 0.1	< 10	< 10	< 10
1.5	< 0.02	< 0.02	< 0.02	0.03	7.8	47200	150000	19400
1.0	< 0.02	< 0.02	0.03	0.11	25.9	< 10	20	< 10
1.0	< 0.02	< 0.02	1.01	0.24	254	1620	1090	437
1.25-1.4	<0.004	<0.005	<0.010	<0.010	<4	164	1230	116
2.15-2.5	<0.004	<0.005	<0.010	<0.010	<4	25	119	<20
0.6-1.0	<0.004	<0.005	<0.010	<0.010	<4	19	95	<20
0.6-1.0	0.036	0.09	1.33	6.93	401	<10	<30	<20
1.0-1.5	0.009	0.061	0.389	1.58	262	3520	10200	3940
0-0.3	<0.004	0.017	<0.010	0.03	132	716	3390	1020
1.25-1.4	<0.004	<0.005	<0.010	<0.010	<4	20	217	22
3-3.2	<0.004	<0.005	<0.010	<0.010	<4	<10	<30	<20
0-0.3	0.006	<0.005	<0.010	<0.010	8	2320	9230	4770
0.3-0.6	0.711	<0.005	0.248	2.52	14	<10	<30	<20
0.8	< 0.02	< 0.02	< 0.02	< 0.02	< 0.1	< 10	< 10	< 10
0.6	< 0.02	< 0.02	0.03	0.17	7.5	228	3920	2940
0.6	< 0.02	< 0.02	< 0.02	< 0.02	< 0.1	< 10	< 10	< 10
0.6	< 0.02	< 0.02	< 0.02	< 0.02	< 0.1	17	517	875
0.3	< 0.02	< 0.02	< 0.02	< 0.02	0.5	53	992	925
1.5	< 0.02	< 0.02	< 0.02	< 0.02	< 0.1	< 10	< 10	< 10
0.30-0.60	<0.004	<0.005	<0.010	<0.010	<4	25	56	<20
4	<0.004	<0.005	<0.010	<0.010	<4	74	239	65
2	<0.004	<0.005	<0.010	<0.010	<4	65	165	35
2	<0.004	<0.005	<0.010	<0.010	<4	31	76	<20
0.50 - 0.75	<0.004	<0.005	<0.010	<0.010	<4	10	49	27
0.50 -0.75	<0.004	<0.005	<0.010	<0.010	<4	20	130	53
3.5	<0.004	<0.005	<0.010	<0.010	<4	13	60	21
1	<0.004	<0.005	<0.010	<0.010	<4	24	58	<20
2	<0.004	<0.005	<0.010	<0.010	<4	56	152	35
1.5	<0.004	<0.005	<0.010	<0.010	<4	55	131	31
2.5	<0.004	<0.005	<0.010	<0.010	<4	56	148	34
1.20-1.50	<0.004	<0.005	<0.010	<0.010	<4	56	124	27
0.35-0.50	<0.004	<0.005	<0.010	<0.010	<4	60	261	115
0.50-0.65	<0.004	<0.005	<0.010	<0.010	<4	120	280	26
0.5	<0.004	<0.005	<0.010	<0.010	<4	36	140	60
1	0.011	0.009	0.012	0.05	6	313	2160	701
1.5	<0.004	<0.005	<0.010	<0.010	<4	151	353	30
0.00-0.15	<0.004	<0.005	<0.010	<0.010	<4	635	5530	2180
1.5	<0.004	<0.005	<0.010	<0.010	0	55	151	23
1	<0.004	<0.005	<0.010	<0.010	<4	<10	<30	<20
1	<0.004	<0.005	<0.010	<0.010	<4	49	135	<20
0.8-1.0	0.244	0.062	0.482	1.41	256	6280	25100	10200
1.5	0.039	0.024	0.144	0.48	163	2620	9870	4020
1.8-2.0	<0.004	<0.005	<0.010	<0.010	52	745	2820	1140
2.5	<0.004	<0.005	<0.010	<0.010	<4	23	36	<20
0.15-0.30	3.22	0.015	227	246	3340	5690	8450	3840
0.5	0.01	<0.005	0.017	0.28	<4	42	61	<20
1.5	<0.004	<0.005	<0.010	0.03	<4	32	46	<20
0.5	0.067	0.009	0.015	0.17	<4	52	96	22
0.00-0.30	<0.004	<0.005	<0.010	<0.010	<4	22	82	41
0.40-0.70	<0.004	<0.005	<0.010	<0.010	<4	43	96	26
0.30-0.60	<0.004	<0.005	<0.010	<0.010	<4	33	51	<20
0.45-0.70	<0.004	<0.005	<0.010	<0.010	<4	68	705	662
0.70-1.00	<0.004	<0.005	<0.010	<0.010	<4	50	691	171
0.50-0.75	<0.004	<0.005	<0.010	<0.010	<4	18	206	173

TABLE E.5
MODELLING AVERAGE POTENTIAL VOLATILE EXPOSURE FROM EXCAVATION

Calculated Parameter	Symbol	Unit	Benzene	Toluene	Ethyl-benzene	Xylenes	Aliphatic F1 (C_6-C_8)	Aliphatic F1 (C_8-C_{10})	Aromatic F1 (C_8-C_{10})	Aliphatic F2 (C_{10}-C_{12})	Aliphatic F2 (C_{12}-C_{16})	Aromatic F2 (C_{10}-C_{12})	Aromatic F2 (C_{12}-C_{16})
Calculation of VOC Exposure and Emissions													
Effective Diffusion Coefficient	D_{eff}	cm²/s	2.10E-03	1.53E-03	9.63E-04	5.64E-04	4.87E-03	1.37E-03	1.77E-04	2.81E-04	6.20E-05	3.31E-05	6.32E-06
Instantaneous soil flux	J_s	g/cm²s	1.78E-08	1.11E-09	4.10E-07	2.98E-07	8.40E-06	2.91E-06	3.67E-13	8.49E-12	4.87E-12	7.28E-13	3.89E-13
Air Concentration (t=0)	Ca^c	µg/m³	4.45E+02	2.77E+01	1.02E+04	7.45E+03	2.10E+05	7.28E+04	9.18E-03	2.12E-01	1.22E-01	1.82E-02	9.72E-03
Air concentration (t=15 minutes)	Ca	µg/m³	1.60E+02	2.99E+01	2.11E+04	8.03E+03	2.26E+05	7.85E+04	9.89E-03	2.29E-01	1.31E-01	1.96E-02	1.05E-02
8-Hour Average Ambient Air Concentrations After Soil Disturbance													
Day 1	Ca	µg/m³	4.77E+00	2.97E-01	1.49E+03	7.99E+01	2.25E+03	7.81E+02	9.84E-05	2.28E-03	1.31E-03	1.95E-04	1.04E-04
Day 2	Ca	µg/m³	1.40E+00	8.75E-02	4.39E+02	2.35E+01	6.63E+02	2.30E+02	2.90E-05	6.70E-04	3.85E-04	5.75E-05	3.07E-05
Day 3	Ca	µg/m³	1.03E+00	6.41E-02	3.22E+02	1.72E+01	4.86E+02	1.68E+02	2.12E-05	4.91E-04	2.82E-04	4.21E-05	2.25E-05
Day 4	Ca	µg/m³	8.50E-01	5.30E-02	2.66E+02	1.43E+01	4.01E+02	1.39E+02	1.75E-05	4.06E-04	2.33E-04	3.48E-05	1.86E-05
Day 5	Ca	µg/m³	7.41E-01	4.62E-02	2.32E+02	1.24E+01	3.49E+02	1.21E+02	1.53E-05	3.54E-04	2.03E-04	3.03E-05	1.62E-05
Regulatory Requirements													
Occupational Health and Safety Alberta													
8-hour Occupational Exposure Limit		ug/m³	1597	188000	434000	434000							
15 Minute Short-term Exposure Limit		ug/m³	7986		543000	651000							

Hours	Seconds	Calculated Parameter	Symbol	Unit	Benzene	Toluene	Ethyl-benzene	Xylenes	Aliphatic F1 (C_6-C_8)	Aliphatic F1 (C_8-C_{10})	Aromatic F1 (C_8-C_{10})	Aliphatic F2 (C_{10}-C_{12})	Aliphatic F2 (C_{12}-C_{16})	Aromatic F2 (C_{10}-C_{12})	Aromatic F2 (C_{12}-C_{16})
		Effective Diffusion Coefficient	D_{eff}	cm²/s	2.10E-03	1.53E-03	9.63E-04	5.64E-04	4.87E-03	1.37E-03	1.77E-04	2.81E-04	6.20E-05	3.31E-05	6.32E-06
0.00	1	Instantaneous Soil flux	J_s	g/cm²s	1.78E-08	1.11E-09	4.10E-07	2.98E-07	8.40E-06	2.91E-06	3.67E-13	8.49E-12	4.87E-12	7.28E-13	3.89E-13
0.13	480				8.12E-10	5.06E-11	2.54E-07	1.36E-08	3.83E-07	1.33E-07	1.68E-14	3.87E-13	2.22E-13	3.32E-14	1.77E-14
0.27	960				5.74E-10	3.58E-11	1.80E-07	9.62E-09	2.71E-07	9.40E-08	1.18E-14	2.74E-13	1.57E-13	2.35E-14	1.26E-14
149.60	538560				2.42E-11	1.51E-12	7.58E-09	4.06E-10	1.12E-08	3.97E-09	5.00E-16	1.16E-14	6.64E-15	9.92E-16	5.30E-16
149.73	539040				2.42E-11	1.51E-12	7.58E-09	4.06E-10	1.12E-08	3.97E-09	5.00E-16	1.16E-14	6.64E-15	9.92E-16	5.30E-16
149.87	539520				2.42E-11	1.51E-12	7.58E-09	4.06E-10	1.12E-08	3.97E-09	5.00E-16	1.16E-14	6.63E-15	9.91E-16	5.29E-16
150.00	540000				2.42E-11	1.51E-12	7.57E-09	4.06E-10	1.12E-08	3.96E-09	4.99E-16	1.16E-14	6.63E-15	9.91E-16	5.29E-16
0.00	1	Instantaneous Air Concentration	Ca	ug/m³	4.45E+02	2.77E+01	1.02E+04	7.45E+03	2.10E+05	7.28E+04	9.18E-03	1.22E-01	1.22E-01	1.82E-02	9.72E-03
0.13	480				2.03E+01	1.27E+00	6.35E+03	3.40E+02	9.58E+03	3.32E+03	4.19E-04	9.69E-03	5.56E-03	8.31E-04	4.44E-04
0.27	960				1.44E+01	8.95E-01	4.49E+03	2.41E+02	6.77E+03	2.35E+03	2.96E-04	6.85E-03	3.93E-03	5.87E-04	3.14E-04
0.40	1440				1.17E+01	7.30E-01	3.67E+03	1.96E+02	5.53E+03	1.92E+03	2.42E-04	5.59E-03	3.21E-03	4.80E-04	2.56E-04
7.60	27360				2.69E+00	1.68E-01	8.41E+02	4.51E+01	1.27E+03	4.40E+02	5.55E-05	1.28E-03	7.36E-04	1.10E-04	5.88E-05
7.73	27840				2.66E+00	1.66E-01	8.34E+02	4.47E+01	1.26E+03	4.37E+02	5.50E-05	1.27E-03	7.30E-04	1.09E-04	5.83E-05
7.87	28320				2.64E+00	1.65E-01	8.27E+02	4.43E+01	1.25E+03	4.33E+02	5.45E-05	1.26E-03	7.24E-04	1.08E-04	5.78E-05
8.00	28800				2.62E+00	1.63E-01	8.20E+02	4.39E+01	1.24E+03	4.29E+02	5.41E-05	1.25E-03	7.18E-04	1.07E-04	5.73E-05
		15 Minute average air concentration	Ca	ug/m³	160	30	21083	8033	2.26E+05	7.85E+04	9.89E-03	1.31E-01	1.96E-02	1.05E-02	
		Day 1: Average 8-hour air concentration	Ca	ug/m³	5	0	1492	80	2.25E+03	7.81E+02	9.84E-05	2.28E-03	1.31E-03	1.95E-04	1.04E-04
8.13	29280				2.60E+00	1.62E-01	8.13E+02	4.36E+01	1.23E+03	4.26E+02	5.36E-05	1.24E-03	7.12E-04	1.06E-04	5.68E-05
8.27	29760				2.58E+00	1.61E-01	8.06E+02	4.32E+01	1.22E+03	4.22E+02	5.32E-05	1.23E-03	7.06E-04	1.06E-04	5.64E-05
8.40	30240				2.56E+00	1.59E-01	8.00E+02	4.29E+01	1.21E+03	4.19E+02	5.28E-05	1.22E-03	7.01E-04	1.05E-04	5.59E-05
8.53	30720				2.54E+00	1.58E-01	7.94E+02	4.25E+01	1.20E+03	4.16E+02	5.24E-05	1.21E-03	6.95E-04	1.04E-04	5.55E-05
31.60	113760				1.32E+00	8.22E-02	4.12E+02	2.21E+01	6.22E+02	2.16E+02	2.72E-05	6.29E-04	3.61E-04	5.40E-05	2.88E-05
31.73	114240				1.32E+00	8.20E-02	4.12E+02	2.20E+01	6.21E+02	2.16E+02	2.71E-05	6.28E-04	3.60E-04	5.38E-05	2.88E-05
31.87	114720				1.31E+00	8.18E-02	4.11E+02	2.20E+01	6.20E+02	2.15E+02	2.71E-05	6.27E-04	3.60E-04	5.37E-05	2.87E-05
32.00	115200				1.31E+00	8.17E-02	4.10E+02	2.20E+01	6.18E+02	2.15E+02	2.70E-05	6.25E-04	3.59E-04	5.36E-05	2.86E-05
		Day 2: Average 8-hour air concentration	Ca	ug/m³	1.40E+00	8.75E-02	4.39E+02	2.35E+01	6.63E+02	2.30E+02	2.90E-05	6.70E-04	3.85E-04	5.75E-05	3.07E-05
32.13	115680				1.31E+00	8.15E-02	4.09E+02	2.19E+01	6.17E+02	2.14E+02	2.70E-05	6.24E-04	3.58E-04	5.35E-05	2.86E-05
32.27	116160				1.30E+00	8.13E-02	4.08E+02	2.19E+01	6.16E+02	2.14E+02	2.69E-05	6.23E-04	3.57E-04	5.34E-05	2.85E-05
32.40	116640				1.30E+00	8.12E-02	4.07E+02	2.18E+01	6.15E+02	2.13E+02	2.69E-05	6.21E-04	3.57E-04	5.33E-05	2.85E-05
32.53	117120				1.30E+00	8.10E-02	4.07E+02	2.18E+01	6.13E+02	2.13E+02	2.68E-05	6.20E-04	3.56E-04	5.32E-05	2.84E-05
55.60	200160				9.94E-01	6.20E-02	3.11E+02	1.67E+01	4.69E+02	1.63E+02	2.05E-05	4.74E-04	2.72E-04	4.07E-05	2.17E-05
55.73	200640				9.93E-01	6.19E-02	3.11E+02	1.66E+01	4.69E+02	1.63E+02	2.05E-05	4.74E-04	2.72E-04	4.06E-05	2.17E-05
55.87	201120				9.91E-01	6.18E-02	3.10E+02	1.66E+01	4.68E+02	1.62E+02	2.05E-05	4.73E-04	2.72E-04	4.06E-05	2.17E-05
56.00	201600				9.90E-01	6.17E-02	3.10E+02	1.66E+01	4.67E+02	1.62E+02	2.04E-05	4.73E-04	2.71E-04	4.05E-05	2.17E-05
		Day 3:Average 8-hour air concentration	Ca	ug/m³	1.03E+00	6.41E-02	3.22E+02	1.72E+01	4.86E+02	1.68E+02	2.12E-05	4.91E-04	2.82E-04	4.21E-05	2.25E-05
56.13	202080				9.89E-01	6.17E-02	3.09E+02	1.66E+01	4.67E+02	1.62E+02	2.04E-05	4.72E-04	2.71E-04	4.05E-05	2.16E-05
56.27	202560				9.88E-01	6.16E-02	3.09E+02	1.66E+01	4.66E+02	1.62E+02	2.04E-05	4.72E-04	2.71E-04	4.04E-05	2.16E-05
56.40	203040				9.87E-01	6.15E-02	3.09E+02	1.65E+01	4.66E+02	1.62E+02	2.04E-05	4.71E-04	2.70E-04	4.04E-05	2.16E-05
56.53	203520				9.86E-01	6.14E-02	3.08E+02	1.65E+01	4.65E+02	1.61E+02	2.03E-05	4.70E-04	2.70E-04	4.03E-05	2.15E-05
79.60	286560				8.31E-01	5.18E-02	2.60E+02	1.39E+01	3.92E+02	1.36E+02	1.71E-05	3.96E-04	2.28E-04	3.40E-05	1.82E-05
79.73	287040				8.30E-01	5.17E-02	2.60E+02	1.39E+01	3.91E+02	1.36E+02	1.71E-05	3.96E-04	2.27E-04	3.40E-05	1.81E-05
79.87	287520				8.29E-01	5.17E-02	2.59E+02	1.39E+01	3.91E+02	1.36E+02	1.71E-05	3.96E-04	2.27E-04	3.39E-05	1.81E-05
80.00	288000				8.29E-01	5.17E-02	2.59E+02	1.39E+01	3.91E+02	1.36E+02	1.71E-05	3.95E-04	2.27E-04	3.39E-05	1.81E-05
		Day 4: Average 8-hour concentration	Ca	ug/m³	8.50E-01	5.30E-02	2.66E+02	1.43E+01	4.01E+02	1.39E+02	1.75E-05	4.06E-04	2.33E-04	3.48E-05	1.86E-05
80.13	288480				8.28E-01	5.16E-02	2.59E+02	1.39E+01	3.91E+02	1.36E+02	1.71E-05	3.95E-04	2.27E-04	3.39E-05	1.81E-05
80.27	288960				8.27E-01	5.16E-02	2.59E+02	1.39E+01	3.90E+02	1.36E+02	1.71E-05	3.95E-04	2.27E-04	3.39E-05	1.81E-05
80.40	289440				8.26E-01	5.15E-02	2.59E+02	1.39E+01	3.90E+02	1.35E+02	1.71E-05	3.94E-04	2.26E-04	3.38E-05	1.81E-05
80.53	289920				8.26E-01	5.15E-02	2.58E+02	1.38E+01	3.90E+02	1.35E+02	1.70E-05	3.94E-04	2.26E-04	3.38E-05	1.81E-05
103.60	372960				7.28E-01	4.54E-02	2.28E+02	1.22E+01	3.42E+02	1.19E+02	1.50E-05	3.47E-04	1.99E-04	2.98E-05	1.59E-05
103.73	373440				7.28E-01	4.54E-02	2.28E+02	1.22E+01	3.42E+02	1.19E+02	1.50E-05	3.47E-04	1.99E-04	2.98E-05	1.59E-05
103.87	373920				7.27E-01	4.53E-02	2.28E+02	1.22E+01	3.42E+02	1.19E+02	1.50E-05	3.47E-04	1.99E-04	2.98E-05	1.59E-05
104.00	374400				7.27E-01	4.53E-02	2.27E+02	1.22E+01	3.42E+02	1.19E+02	1.50E-05	3.47E-04	1.99E-04	2.97E-05	1.59E-05
		Day 5: Average 8-hour air concentration	Ca	ug/m³	7.41E-01	4.62E-02	2.32E+02	1.24E+01	3.49E+02	1.21E+02	1.53E-05	3.54E-04	2.03E-04	3.03E-05	1.62E-05

c - Assuming emitted mass is bound in a box volume, bound by125 m in height x area of contamination (400m²)

TABLE E.6
EX SITU EXPOSURE DURING ALLU SOIL TREATMENT

Parameter	Symbol	Units	SM Bucket	M Bucket	LG Bucket	References / Notes		
Bucket volume	-	m^3	1	2	4	US EPA, 1993		
Soil processing rate	AR	m^3/hr	50	150	240	US EPA, 1993		
Allu processing rate	Q	m^3/hr	37.5	112.5	180	25% slower processing than excavation, assumed		
Excavation hours	-	hr	16	5.33	3.33	800 m^3		
Allu aeration hours	tr	hr	21.3	8.0	4.4	2 passes through allu		
Total soil mass		kg		1120000		Function of bulk density		
Wind Speed	U	m/s		4		Environment Canada, 2008		
Total soil volume	Vsoil	m^3		800		Assumed		
Source length	L	m		20		Assumed		
Source area	SA	m^2		400		Assumed		
Height of Allu bucket	Allu$_h$	m		2		Assumed		
Calculated Parameter	**Symbol**	**Unit**	**Benzene**	**Toluene**	**Ethyl-benzene**	**Xylenes**	**F1**	**F2**
AENV Tier 1 FG Commerical Soil Guideline	SQG	mg/kg	0.046	0.52	0.11	15	320	260
Average concentration of contaminant in soil	Cs	mg/kg	0.49	0.036	16.7	15.9	277	1272.0
Maximum Concentration of contaminant in soil	Cs	mg/kg	3.22	0.09	227	246	3340	47200
Short term emission rate to AENV Soil Guideline	ER	g/s	1.23E-01	BG	8.82E+00	8.98E+00	1.17E+02	1.83E+03
Cs)	ER	g/s	1.91E-02	1.40E-03	6.50E-01	6.18E-01	1.08E+01	4.95E+01
Short-term Ex Situ Air Concentration	C$_{air\_ex-situ}$	mg/m^3	7.71E-01	BG	5.51E+01	5.61E+01	7.34E+02	1.14E+04
Long-term Ex Situ Air Concentration	C$_{air\_ex-situ}$	mg/m^3	1.19E-01	8.72E-03	4.06E+00	3.86E+00	6.73E+01	3.09E+02
Long-term Ex Situ Air Concentration (with biofilter)	C$_{air\_BF}$	mg/m^3	4.78E-02	3.49E-03	1.62E+00	1.55E+00	2.69E+01	1.24E+02
8-hour Occupational Exposure Limit		mg/m^3	1.60	188	434	434	-	-
15 Minute Short-term Exposure Limit		mg/m^3	7.99	-	543	651	-	-

BG - below guideline

	Units	Benzene (C_6H_6)	Toluene (C_7H_8)	Ethyl-benzene (C_8H_{10})	Xylenes	Aliphatic F1 (C6-C8)	Aliphatic F1 (C8-C10)	Aromatic F1 (C8-C10)	Aliphatic F2 (C10 - C12)	Aliphatic F2 (C12 - C16)	Aromatic F2 (C10 - C12)	Aromatic F2 (C12 - C16)	VOC
Fraction of Petroleum Hydrocarbon Subfraction in Soil	unitless	-	-	-	-	0.55	0.36	0.09	0.36	0.44	0.09	0.11	-
Average measured soil concentration at an example site prior to remedial activities	mg/kg	0.4913077	0.035875	16.71264286	15.894706	152	100	25	458	560	114	140	
Calculation of Greenhouse Gas Emissions from Ex Situ													
Contaminant emission mass (kg)[a] assuming no transformation	kg	5.50E+02	4.02E+01	1.87E+04	1.78E+04	1.71E+05	1.12E+05	2.79E+04	5.13E+05	6.27E+05	1.28E+05	1.57E+05	1.77E+06
Molar mass of carbon in compound	g/mol	72.0642	84.0749	96.0856	96.0856	108.0963	108.0963	108.0963	132.1177	168.1498	120.107	144.1284	-
Molecular mass of contaminant	g/mol	7.81E+01	9.21E+01	1.06E+02	1.06E+02	1.00E+02	1.30E+02	1.20E+02	1.60E+02	2.00E+02	1.30E+02	1.50E+02	-
Relative mass portion of carbon	fraction	0.92	0.91	0.91	0.91	1.08	0.83	0.90	0.83	0.84	0.92	0.96	
Mass of carbon emitted	Tonnes	0.51	0.037	17	16	184	93	25	423	527	118	151	1556
Mass of carbon dioxide emitted[b]	Tonnes	1.9	0.13	62	59	676	341	92	1553	1933	434	552	5705
Calculation of Greenhouse Gas Emissions From Landfill Disposal													
Total Mass of Carbon from Contaminants	Tonnes	1556											
Predicted Landfill Methane Emission	Tonnes	455											
Predicted Landfill Carbon Dioxide Emissions	Tonnes	1284											

a - Assuming a uniform soil contaminant concentration and a soil volume of 800 m^3, an area of 400m^2 with a contaminant depth of 2 m

b - Assuming aerobic transformation of contaminants solely over the lifecycle of the emitted compound

* Concentrations adjusted for the fraction of petroleum hydrocarbon subfraction in soil.
Concentrations of F3 and F4 were not considered

TABLE E.8
REMEDIAL RELATED PARTICULATE EMISSIONS

Calculation	Symbol	Units	Value	Assumptions
Particle Emission Due to Soil Handling				
Emissions from all transfer operations, FG PM2.5	$E_{Transfer1}$	g	13748.9	
Emissions from all transfer operations, FG PM10	$E_{Transfer2}$	g	43746.5	
Emissions from soil spreading (i.e. bulldozer, tractor blade), FG	$E_{Spreading1}$	g/sec	0.19	
Particulate emissions From Vehicular Traffic				
Emissions from unpaved industrial surfaces (PM2.5)	$EF_{Unpaved1}$	g/VKT	64.5	
Emissions from unpaved industrial surfaces (PM10)	$EF_{Unpaved2}$	g/VKT	649.9	
Emissions from unpaved industrial surfaces (TPM)	$EF_{Unpaved3}$	g/VKT	2387.6	
Ex Situ Remedial Activities				
Emissions from all transfer operations, FG PM2.5		g	54995.6	Soil volume moved 4x (excavation, allu x2, soil replacement)
Emissions from all transfer operations, FG PM10		g	174986.1	Soil volume moved 4x (excavation, allu x2, soil replacement)
Total Emissions from soil spreading and levelling		g	2702	4 hours of levelling and spreading
Emissions from unpaved industrial surfaces (PM2.5)		g	1613.20	25 km travelled onsite, all equipment
Emissions from unpaved industrial surfaces (PM10)		g	16247.19	25 km travelled onsite, all equipment
Emissions from unpaved industrial surfaces (TPM)		g	59690.26	25 km travelled onsite, all equipment
Excavator engine emissions during excavation and allu		g	655.66	12 hr excavating, and 12 hr of alluing
Front-end loader engine emissions during levelling		g	106.47	6 hr of active levelling and compacting
Generator engine emissions		g	340.926	24 hr of generator power output
TOTAL PM2.5 from transfer operations and unpaved site traffic		**Kg**	**56.609**	
TOTAL PM10 from tranfer operations and unpaved site traffic		**Kg**	**191.2**	
TOTAL PARTICULATE EMISSIONS FROM ALL ON-SITE EX-SITU REMEDIAL ACTIVITIES		**Kg**	**238.5**	
Landfill Disposal				
Emissions from all transfer operations into and out of haul truck, FG PM2.5		g	27497.8	Soil volume moved 2x, into and out of haul truck only
Emissions from all transfer operations into and out of haul truck, FG PM10		g	87493.0	Soil volume moved 2x, into and out of haul truck only
Emissions from spreading and levelling into landfill cell		g	1351.2	2 hr to incorporate into landfill cell
Emission from unpaved industrial surfaces (PM2.5)		g	8066.0	25 km onsite equipment travel and 100 km to landfill
Emissions from unpaved industrial surfaces (PM10)		g	81235.9	25 km onsite equipment travel and 100 km to landfill
Emissions from unpaved industrial surfaces (TPM)		g	298451.3	25 km onsite equipment travel and 100 km to landfill
Excavator engine emissions during excavation		g	327.83	12 hr of active excavation
Front-end loader emission during levelling		g	212.93	12 hr of levelling and compacting into landfill cell
Generator engine emissions		g	340.926	24 hr of generator power output
TOTAL PM2.5 from transfer operations and unpaved onsite and offsite traffic		**Kg**	**35.56**	
TOTAL PM10 from tranfer operations and unpaved onsite and offsite traffic		**Kg**	**168.7**	
TOTAL PARTICULATE EMISSIONS FROM ALL LANDFILL DISPOSAL ACTIVITIES		**Kg**	**388.2**	

TABLE E.9
ENGINE COMBUSTION EMISSIONS OF CRITERIA AIR CONTAMINANTS (KG)

Process	L/hr[b]	Operational Hours Ex Situ Scenario	Operational Hours Landfill Disposal Scenario
Material Handling using a front end loader	4.67	6	12
Mass excavation using an excavator	7.19	24	12
Power generation using a generator	4.54	24	24
Contaminated soil transport (hr)	7.57	0.3125	1.5625

Equipment	CH_4	CO	N_2O	NO_x (NO_2)	SO_x (SO_2)	CO_2	VOC[c]
Wheeled loader	1.38	138.58	28.60	223.30	6.54	68,158	27.02
Off-road trucks	1.38	79.46	28.60	280.42	6.54	68,253	27.02
Tractor	1.20	194.02	28.60	359.59	6.54	67,849	48.46
Industrial engine	4.15	199.55	28.60	439.41	6.54	67,889	45.74
Ex-Situ							
Equipment	CH_4	CO	N_2O	NO_x (NO_2)	SO_x (SO_2)	CO_2	VOC[c]
2004 Front-end loader, 6 cylinder, 149 HP	0.002	0.151	0.031	0.244	0.007	74.481	0.030
2002 Generator, 4 cylinder, 108 HP	0.018	0.848	0.122	1.867	0.028	288.489	0.194
2001 Excavator 1, 6 cylinder, 254 HP	0.008	1.306	0.192	2.420	0.044	456.610	0.326
2005 On-highway truck, 6 cylinder, 306 HP	0.000	0.007	0.003	0.026	0.001	6.297	0.002
2005 On-highway truck, 6 cylinder, 306 HP (for biofilter transport)	0.000	0.007	0.003	0.026	0.001	6.297	0.002
Sum	**0.03**	**2.31**	**0.35**	**4.56**	**0.08**	**825.88**	**0.55**
Sum (with biofilter)	**0.03**	**2.32**	**0.35**	**4.58**	**0.08**	**832.17**	**0.56**
Landfill Disposal							
2004 Front-end loader, 6 cylinder, 149 HP	0.003	0.303	0.063	0.488	0.014	148.963	0.059
2002 Generator, 4 cylinder, 108 HP	0.018	0.848	0.122	1.867	0.028	288.489	0.194
2001 Excavator 1, 6 cylinder, 254 HP	0.004	0.653	0.096	1.210	0.022	228.305	0.163
2005 On-highway truck, 6 cylinder, 306 HP	0.001	0.037	0.013	0.129	0.003	31.485	0.012
Sum	**0.03**	**1.84**	**0.29**	**3.69**	**0.07**	**697.24**	**0.43**

a - taken from GHGenius for 2010 western Canada fuel specifications www.ghgenius.ca
b – Frey et al., 2010, rates paired with equipment and activity most applicable.
c – does not consider methane into equipment VOC total

Notes
Emissions applied from off-road trucks were applied to on-highway truck fuel use in lieu of a more appropriate input