CCME

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de l'environnement

Canadian Soil Quality Guidelines
TRICHLOROETHYLENE
Environmental and Human Health Effects

Scientific Supporting Document

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ABSTRACT

This scientific supporting document provides the background information and rationale for the derivation of human health and environmental soil quality guidelines for trichloroethylene (TCE). Canadian Soil Quality Guidelines for the protection of environmental health for TCE were originally published by the Canadian Council of Ministers of the Environment (CCME) in 1997 (CCME 1997). These underwent minor revisions and were re-published in 1999 in *Canadian Environmental Quality Guidelines* (CCME 1999). This supporting document updates the TCE ecological assessment using information available to November 2004. In addition, calculations of TCE soil quality guidelines protective of human health have been added and reflect the recent Health Canada deliberations on human toxicological thresholds for TCE. Guidelines in this scientific supporting document were calculated using the most recent (2003) draft of the CCME protocols for the derivation of Canadian Soil Quality Guidelines. It is anticipated that this draft protocol document will be finalized in 2005.

This document contains an updated review of information on the chemical and physical properties of trichloroethylene, a review of sources and emissions in Canada, the distribution and behaviour of trichloroethylene in the environment, and the toxicological effects of trichloroethylene on microbial processes, plants, and animals – including humans. This information is used to derive soil quality guidelines for trichloroethylene to protect human health and ecological receptors in four land uses: agricultural, residential/parkland, commercial, and industrial.

Trichloroethylene (CAS no. 79-01-6) is encountered as a soil and particularly groundwater contaminant in Canada, owing to its extensive use as a degreasing solvent and subsequent release to the environment. TCE has a relatively high vapour pressure and volatility, and also a relatively high Henry's law coefficient and solubility. As a result of these properties, TCE may be present in significant concentrations in both the dissolved and vapour phases. Human exposure to TCE may occur through the ingestion of contaminated drinking water, and through occupational exposures, especially based on inhalation and/or dermal absorption.

The environmental soil quality guideline (SQG_E) that have been derived for trichloroethylene for all of the four land uses, based on potential for groundwater mediated transfer to adjacent water bodies that contain aquatic life are 0.05 mg/kg in coarse textured soils and 0.16 mg/kg in fine-textured soils.

The human health soil quality guideline (SQG_{HH}) that has been derived is 0.01 mg/kg, based on protection of potable groundwater. The same guideline applies to both soil types and to all four land uses.

The Canadian Soil Quality Guideline for trichloroethylene for the protection of environmental and human health is 0.01 mg/kg for both soil types and all land uses.

It is noted that vinyl chloride is a potential degradation product of TCE, and may be more toxic than TCE. Accordingly, it is imperative that an assessment of vinyl chloride concentrations be made whenever TCE is present in the environment.

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1. INTRODUCTION

Canadian Environmental Quality Guidelines are intended to protect, sustain, and enhance the quality of the Canadian environment and its many beneficial uses. They are generic numerical concentrations or narrative statements that specify levels of toxic substances or other parameters in the ambient environment that are recommended to protect and maintain wildlife and/or the specified uses of water, sediment, and soil. These values are nationally endorsed through the Canadian Council of Ministers of the Environment (CCME) and are recommended for toxic substances and other parameters (e.g., nutrients, pH) of concern in the ambient environment.

The development of Canadian Soil Quality Guidelines was initiated through the National Contaminated Sites Remediation Program (NCSRP) in 1991 by the CCME Subcommittee on Environmental Quality Criteria for Contaminated Sites. In response to the urgent need to begin remediation of high priority “orphan” contaminated sites, an interim set of soil quality criteria was adopted from values that were in use in various jurisdictions across Canada (CCME 1991a). Although the NCSRP program officially ended in March of 1995, the development of soil quality guidelines was pursued under the direction of the CCME Soil Quality Guidelines Task Group because of the continued need for national soil quality guidelines for the management of soil quality (with a particular focus on remediation of contaminated sites).

Canadian Soil Quality Guidelines are developed according to procedures that have been described by CCME (CCME 1996a; reprinted in 1999, and currently under revision). The soil quality guidelines for TCE in the current report were developed based on the most recent available draft version of the revised protocol, referenced as CCME (2003). It is noted that any changes that may be made in finalizing this draft (CCME 2003) protocol are not reflected in the current report. According to this protocol, both environmental and human health soil quality guidelines are developed for four land uses: agricultural, residential/parkland, commercial, and industrial. The lowest value generated by the two approaches for each of the four land uses is recommended by CCME as the Canadian Soil Quality Guideline. Guidelines for a number of substances were developed using this protocol and released in a working document entitled *Recommended Canadian Soil Quality Guidelines* (CCME 1997). The guidelines originally published in that document have since been revised and were superseded by the Canadian Soil Quality Guidelines for the protection of environmental and human health published by CCME in October of 1999 (CCME 1999). Guidelines produced since 2000 generally reflect a third major round of improvements to the derivation protocols and are based on new scientific information available since the mid 1990s.

This scientific supporting document provides the background information and rationale for the derivation of environmental soil quality guidelines for trichloroethylene. This document contains a review of information on the chemical and physical properties of trichloroethylene, a review of sources and emissions in Canada, the distribution and behaviour of trichloroethylene in the environment, the toxicological effects of trichloroethylene on biota (microbial processes, terrestrial plants, soil invertebrates) as well as humans and mammalian species, and Canadian soil quality guidelines protective of human and ecological health that are based on the above information.

Comprehensive reviews of the environmental sources, fate and effects of trichloroethylene have been produced in other non-Canadian jurisdictions, and interested readers are referred to these other major compilations for relevant specific information; in particular, WHO (1985), ATSDR (1997), and others.

The Canadian Soil Quality Guidelines presented in this document are intended as general guidance. Site-specific conditions should be considered in the application of these values. The reader is referred to CCME (1996b) for further generic implementation guidance pertaining to the guidelines. Soil quality guidelines are derived to approximate a "no- to low-" effect level (or threshold level) based only on the toxicological information and other scientific data (fate, behaviour, etc.) available for the substance of concern, and they do not consider socioeconomic, technological, or political factors. These non-scientific factors are to be considered by site managers at the site-specific level as part of the risk management process. Because these guidelines may be used and applied differently across provincial and territorial jurisdictions, the reader should consult the laws and regulations of the jurisdiction they are working within for applicable implementation procedures.

2. BACKGROUND INFORMATION

2.1 Physical and Chemical Properties

The physical and chemical properties of TCE are summarized in Table 2. Trichloroethylene (1,1,2-trichloroethene; TCE; CAS Registry No. 79-01-6) is a clear, colourless, non-viscous liquid with a characteristic, slightly sweet odour (McNeill 1979). It is an unsaturated, chlorinated aliphatic compound (chemical formula C_2HCl_3) with a low molecular weight (131.4 $g{\cdot}mole^{-1}$), and is a powerful solvent for a large number of natural and synthetic substances (Schaumburg 1990). Trichloroethylene is a volatile liquid at room temperature (melting point -83.5°C, boiling point 86.7°C) with a higher density (1.46 $g{\cdot}mL^{-1}$ at 20°C) and a lower surface tension (0.029 N/m) than water. Its vapour is heavier than air (Eisenreich et al. 1981; ATSDR 1989).

Hsieh et al., (1994) conducted an in-depth review of the published literature on the physical and chemical properties of TCE. Their methodology involved surveying the literature for published measured values of each parameter, and taking the arithmetic mean. Their results for a number of key parameters follow. Where necessary, the units have been changed.

Solubility (S)

TCE has a moderate aqueous solubility of 1,450 mg/L (arithmetic mean of 7 values; coefficient of variation 15%; Hsieh et al. 1994).

Vapour Pressure (VP)

Hsieh et al. (1994) indicate that the vapour pressure of TCE at 25°C is 9,700 Pa (arithmetic mean of 5 measured values; coefficient of variation 2%). They also used data from Kirk-Othmer (1964) together with an Antoine equation, to generate the following relationship between the vapour pressure of TCE in Pa (VP) to the temperature in degrees centigrade (T):

$$Log(VP) = 10.128 - \frac{1830.4}{273 + T} \qquad (1)$$

Substituting T = 25°C in this equation yields 9,700 Pa, consistent with the mean of the measured values noted above. However, at a temperature of 5°C (more typical of subsurface conditions in Canada), the vapour pressure estimated using equation (1) is 3,500 Pa.

Henry's Law Constant (H and H')

Trichloroethylene has a Henry's law constant at temperatures in the range of 20 - 25°C of 890 Pa-m^3/mol (arithmetic mean of 12 values; coefficient of variation 18%; Hsieh et al 1994). The dimensionless Henry's law constant is calculated using the following equation:

$$H' = \frac{H}{RT} \qquad (2)$$

Where:

H’ = dimensionless Henry’s law constant (-);
H = Henry’s law constant (890 Pa-m^3/mol);
R = ideal gas constant (8.314 Pa-m^3·mol^{-1}K^{-1}); and,
T = temperature (298 K).

Substituting these values in the above equation yields a value of the dimensionless Henry’s law constant of 0.36 at 25°C.

The Henry’s law constant can also be estimated using the formula

$$H = \frac{VP}{S} \qquad (3)$$

Where:

H = Henry’s law constant (Pa-m^3/mol);
VP = Vapour pressure (Pa); and,
S = solubility (mol/m^3).

Using equations (2) and (3), together with the vapour pressure at 5°C calculated above (3,500 Pa) allows H and H’ to be estimated at 5°C as 320 Pa-m^3·mol^{-1} and 0.14, respectively.

Organic Carbon – Water Partition Coefficient (K_{oc})

Hsieh et al. (1994) estimate the K_{oc} of trichloroethylene to be 86, based on the arithmetic mean of 13 measured values (K_{oc} is unitless). The coefficient of variation of these data is 46%, reflecting the greater variation associated with measuring this parameter. This may be a result of variations in the composition of the organic carbon in the soils in the various studies.

2.2 Impurities and Stabilizers

Trichloroethylene produced for chemical reagent uses has a minimum purity of 99.85%. The commercial product may contain impurities as shown in Table 3. Under conditions of normal use, trichloroethylene is considered nonflammable and moderately stable, but requires the addition of stabilizers (up to 2% v/v) in commercial grades. Stabilizers commonly added to commercial grades of trichloroethylene are summarized in Table 3. In the absence of stabilizers, trichloroethylene is slowly oxidized by air or photolyzed by light.

2.3 Analytical Methods

One of the principal reference sources for analytical methods for soils (and other materials) is U.S. EPA Document SW-846: "*Test Methods for Evaluating Solid Wastes – Physical/Chemical Methods*" (U.S. EPA 2004b). EPA Methods referred to below are sourced from this document. Most techniques for the analysis of trichloroethylene in soil include the following two elements:

1. sample preparation; and,
2. separation, followed by detection and quantification of the volatile compounds.

EPA Methods for Sample Preparation

EPA methods for sample preparation are summarized below.

- EPA Method 5035 "Closed-system purge-and-trap and extraction for volatile organics in soil and waste samples" involves heating an aqueous solution of the sample to 40°C purging with inert gas and collecting the purged volatiles in a trap before injection into the GC.

- EPA Method 5021 "Volatile organic compounds in soils and other solid matrices using equilibrium headspace analysis" involves heating an aqueous solution of the sample to 85°C in a headspace vial, equilibrating for 50 minutes, and introducing a known amount of headspace vapour into the GC.

- EPA Method 5032 "Volatile organic compounds by vacuum distillation" involves subjecting the soil sample to a vacuum of 10 Torr (the vapour pressure of water), and cryogenically trapping the distillate before introducing it into the GC.

EPA Methods for Separation and Detection/Quantification

Methods for separation and detection/quantification include the following:

- EPA Method 8021B "Aromatic and halogenated volatiles by gas chromatography using photoionization and/or electrolytic conductivity detectors" provides details of a methodology involving gas chromatographic separation and a photoionization (PID) detector for aromatic compounds in series with an electrolytic conductivity detector for halogenated compounds.

- EPA Method 8260B "Volatile organic compounds by gas chromatography/mass spectrometry" provides details of a methodology involving gas chromatographic separation and identification/quantitation using mass spectrometry. The estimated quantitation limit (EQL) of Method 8260 for an individual compound is somewhat instrument dependent and also dependent on the choice of sample preparation/introduction method. Using standard quadrapole instrumentation and the purge-and-trap technique, limits should be approximately 0.005 mg/kg (wet weight) for soil/sediment samples. Somewhat lower limits may be achieved using an ion trap mass spectrometer or other instrumentation of improved design (U.S. EPA 2004b).

Methods Used in Canadian Laboratories

The methods most commonly used in Canadian Environmental laboratories are probably EPA 5035 (purge-and-trap) or EPA 5021 (headspace) followed by EPA 8260B (GC-MS), or slight modifications of these methods. Some laboratories may extract soil samples into methanol prior to purge-and-trap or headspace techniques. One well-respected laboratory located in Edmonton, Alberta quoted method detection limits (MDLs) of 0.01 and 0.004 mg/kg for headspace and purge-and-trap techniques, respectively.

2.4 Production and Uses

Trichloroethylene is generally produced via chlorination of ethylene or ethylene dichloride (WHO 1985; ATSDR 1991). Trichloroethylene was produced in Canada at two plants, C-I-L (now I-C-I Canada) and Venchem, both in Shawinigan, Quebec. Canadian production of trichloroethylene peaked during the mid 1970s with a high of 22.5 kilotonnes produced in 1976. However, both Canadian plants were closed by 1985, due primarily to decreasing domestic demand. Canadian usage of TCE has been steadily decreasing since the mid-1970s, as a result of increasing environmental and health concerns along with the introduction of tighter equipment specifications, closed system cleaning and degreasing technologies, and recycling systems. As of 2001, the total domestic demand in Canada for TCE was 2.4 kilotonnes, which was met entirely through imports (CIS 2002).

Data from the 2002 National Pollutant Release Inventory (NPRI) include 46 facilities that handled TCE and met the NPRI reporting requirements. In 2002, 748 tonnes were reported as having been released (as emissions, effluent or spills) on-site, 114 tonnes were sent for off-site disposal, and 40 tonnes were recycled at off-site facilities (902 tonnes total). Data from the 2001 reporting year indicate values of 754 tones released on-site, 91 tonnes disposed off-site, and 100 tonnes recycled (945 tonnes total). Thus, total amounts of TCE reported as released exceed the Health Canada (2004) usage estimates. The majority of facilities that reported in 2002 were in Ontario (31 facilities) while Quebec, Alberta and Nova Scotia had the remainder at 13, 1 and 1 facilities each, respectively. NPRI reported emissions of TCE have decreased by 17% between 1995 and 2000.

The major use of trichloroethylene in Canada, greater than 85% of domestic consumption, is vapour degreasing and cold cleaning of fabricated metal parts, which is closely associated with the automotive and metals industries (ATSDR 1991; CIS 2002). Health Canada (2004) estimated that about 90% of total domestic use of TCE is used in metal degreasing applications. Minor uses include the production of adhesives and co-polymers, household and industrial dry cleaning, textile manufacturing, cleaning of electronic components, petroleum industry processes involving refining catalysts, paint removers, coatings, vinyl resins, and laboratory reagent/solvent applications. Consumer products that may contain trichloroethylene include typewriter correction fluids, paint removers/strippers, adhesives, spot removers, and rug-cleaning fluids (WHO 1985; Frankenberry et al. 1987; ATSDR 1989; Bruckner et al. 1989; ATSDR 1991). In the vast majority of these uses, TCE is not destroyed, but dispersed into the environment. The total domestic demand is nearly exclusively used for replacing emission losses and for distribution in end products. Prior to the closure of the two Canadian production plants, trichloroethylene was also used in the synthesis of tetrachloroethylene (also known as perchloroethylene or PERC) (CPI 1986)

On August 13, 2003, the *Solvent Degreasing Regulations* (SOR/2003-283) were published in Part II of the Canada Gazette. The regulations are aimed at reducing the use of TCE and PERC by the degreasing industry, by setting allowable consumption units based on an operation's calculated average historical use. Use of TCE will be limited to the calculated allowable consumption unit for the years 2004-2006. In 2007, it will be required that each degreasing operation reduce the use of TCE and PERC by 65% of the calculated average consumption unit. This regulation applies only to degreasing operations using more than 1,000 kg of TCE/PERC per calendar year. It is estimated that the regulations will prevent the release of 10.2 kilotonnes of TCE and PERC into the atmosphere over the 2004-2021 time period.

2.5 Trichloroethylene in the Environment

2.5.1 Emissions

There are only limited data available regarding releases of trichloroethylene to the Canadian environment. However, since nearly all of the nationwide usage of TCE is dispersive, the potential release of TCE to the Canadian environment can be estimated as equal to the Canadian net domestic consumption [e.g. 2.4 kilotonnes in 2001 (CIS 2002)].

Pandullo et al. (1985) have reported that metal degreasing operations are the major industrial sources of trichloroethylene emission in the U.S. where eventually most of the TCE is released to the atmosphere, despite internal recycling. Canadian metal degreasing operations consume over 90% of the domestic trichloroethylene supply (Health Canada 2003). A similar situation is therefore expected to exist in Canada. Other emission sources identified in the U.S. that may apply to Canada include accidental and intentional industrial discharges, evaporation from dry cleaning operations, sewage treatment plants, waste disposal sites, waste incineration, and use of trichloroethylene-containing products (e.g. glues, paints, shoe polish, spot removers, paint removers, upholstery cleaners, and adhesives) (Pandullo et al. 1985, Schaumburg 1990; ATSDR

1991). Very little trichloroethylene is released during its manufacture or its use as a chemical intermediate (Colborn 1990). Soil, air and groundwater contamination can also occur from leaking underground storage tanks, landfills, accidental spills, while smaller amounts may leak through septic tanks and septic tank cleaners, old drain and pipe cleaners (Wang et al. 1985, Muraoka and Hirata 1988). Furthermore, trichloroethylene can also be formed in groundwater as a biodegradation product of tetrachloroethylene contamination (Major et al. 1991).

2.5.2 Atmosphere

Mean levels (24 hours) of trichloroethylene in air sampled between 1988 and 1990 in eleven Canadian cities ranged from 0.07 to 0.98 μg/m^3, with a maximum 24-hour concentration of 19.98 μg/m^3 measured in Pointe aux Trembles, Montreal, Quebec in 1990 (Dann and Wang 1992). The OMOE (1988) reported levels near twelve Canadian homes that ranged between non-detectable (detection limit not stated) and 2 μg/m^3. The spring mean was 0.8 μg/m^3 and the winter mean was 0.2 μg/m^3. At the only rural monitoring site in Canada (Walpole Island, Ont.), mean trichloroethylene concentrations of 0.19 μg/m^3 were measured during 1989-1990, with a maximum concentration of 0.46 μg/m^3 (Dann and Wang 1992).

Recent U.S. data are in the range of levels measured in Canada. In 1998, ambient air measurement data from 115 monitors located in 14 states indicated that TCE levels ranged from 0.01 to 3.9 μg/m^3, with a mean of 0.88 μg/m^3. Mean TCE air concentrations (1985-1998) for rural, suburban, urban, commercial and industrial land uses were 0.42, 1.26, 1.61, 1.84 and 1.54 μg/m^3, respectively (Health Canada 2003). Results for rural U.S. sites indicate that concentrations of trichloroethylene in air ranged between 0.006 and 1.9 μg/m^3 (HWC 1990). Similar studies of forested areas in Germany carried out in 1987 and 1988 indicated atmospheric levels ranging from 0.2 to 1.1 μg/m^3 (average 0.5 μg/m^3) (Frank et al. 1989, Frank 1989). Average trichloroethylene levels of 0.04 to 0.05 μg/m^3 were detected in Arctic air between 1982 and 1983 (Khalil and Rasmussen 1983; Hov et al. 1984).

Trichloroethylene levels in air above hazardous waste and landfill sites can be higher than those in ambient rural or urban air. No data were available for Canadian landfill or hazardous waste sites. However, data from both active and abandoned sites in New Jersey revealed that mean trichloroethylene concentrations ranged between 0.43 and 15.5 μg/m^3 (LaRegina et al. 1986; Harkov et al. 1983), with maximum recorded values of 108 μg/m^3 and 66.5 μg/m^3, respectively. In Germany, levels ranging from 800 to 10500 μg/m^3 were measured (Koch et. al. 1990). Proximate to industrial point sources, TCE levels may also be very high, although no Canadian data were found. However, levels as high as 1460 μg/m^3 have been recorded 0.5 km from a TCE production and storage site in the United States (U.S. EPA 1977).

2.5.3 Indoor Air

Concentrations of trichloroethylene in indoor air in approximately 750 homes from across Canada were up to 165 μg/m^3, with an overall mean value of 1.4 μg/m^3, based on preliminary results of a pilot study (Otson *et al*. 1992). In two homes tested, it was reported that showering with well water containing extremely high levels of TCE (40 mg/L) increased levels of TCE in bathroom air from <0.5 to 67.81 mg/m^3 in less than 30 minutes (Health Canada 2003). Similar indoor air levels to the Otson *et al*., (1992) values have been reported in smaller surveys in Toronto (Chan *et al*. 1990; Bell *et al*. 1991) and extensive surveys in the U.S. (Wallace *et al*. 1991; U.S. EPA 1987; Pellizari *et al*. 1989; Shah and Singh 1988).

2.5.4 Surface Water

Reported levels of TCE in Canadian surface waters ranged from below the detection limit (<0.001 μg/L) to 90 μg/L, with the highest observed levels being reported for sites in Quebec and Ontario. In Ontario, trichloroethylene has been detected in 78% of sewage treatment plant effluents and in 82% of water samples collected in the area of the St. Clair River where significant industrial activity occurs (OMOE 1984). In surface water samples in the St. Lawrence River, Lum and Kaiser (1986) found levels of trichloroethylene as high as 90 μg/L at the mouth of the Yamaska and St. François Rivers in Lac St. Pierre near Sorel, Quebec, and levels at several stations below Cornwall and in Lac St. Louis as high as 2.8 to 20 μg/L. Generally, levels in the St. Lawrence River were in the 0.01 to 0.05 μg/L range. Trichloroethylene has also been found in other Ontario rivers; the Niagara River at Niagara-on-the-Lake (0.008 to 0.12 μg/L) (Strachan and Edwards 1984), the Welland river (below detection limit to 0.75 μg/·L) (Kaiser and Comba 1983), and the St. Clair river (0.01 to 0.10 μg/L) (Kaiser and Comba 1986a: detection limit 0.001 μg/L). In a contaminant plume in the St. Clair River, levels as high as 42 μg/L were measured adjacent to industrial sewer outfalls (COARGLWQ 1986). Also, samples from the Great Lakes contained trichloroethylene concentrations ranging from below detection limit to 0.033 μg/L (mean 0.0025 μg/L) in Lake Ontario (Kaiser et al. 1983); 0.006 to 0.168 μg/L in Lake Erie (Kaiser and Valdmanis 1979), and below detection limit to 0.036 μg/L (mean 0.0094 μg/L) in Lake St. Clair (Kaiser and Comba 1986b).

In Ontario and Quebec, several monitoring studies have detected trichloroethylene in leachates from various landfills at levels ranging from 0.29 to 67 μg/L (Barker 1987; Lesage et al. 1989). Levels in the discharge from three non-contact cooling water systems at the Olin Corporation in Niagara Falls (US) ranged from 826 to 2553 μg/L (Hang and Salvo 1981). Levels of trichloroethylene measured in effluent from Dow Chemical in Ontario to the St. Clair River ranged from below the detection limit (1 μg/L) to a maximum of 780 μg/L for a point source discharge to the St. Clair River (Dow Scott Road Landfill effluent - after carbon filtration) (COARGLWQ 1986). The Rockwood Propellant plant of Bristol Aerospace Limited, located north of Winnipeg, MB (UMA 1992a) had trichloroethylene concentrations in building effluent discharges of 1.9 µg/L (boiler cooling water from a stagnant ditch) to approximately 1,100 to 1,350 µg/L (raw water, laboratory waste holding tank, ditch samples), with a maximum of 1,555 µg/L in a compressor cooling water sample.

Records of surface water quality in Alberta indicated that trichloroethylene is only rarely present. Of 3,405 samples for which trichloroethylene concentration information was available, it was detected in only 16 samples, all of which had less than or equal to 1 µg/L trichloroethylene.

Six accidental discharges and spills of trichloroethylene in Canada were reported voluntarily between 1981 and 1988. They resulted from material failure, equipment damage, or transportation accidents (NATES 1992; DGAIS 1992). Release volumes ranged from less than one litre to 5.3 tonnes.

2.5.5 Groundwater

The highest levels of trichloroethylene in groundwater are associated with leaching from specific sources, such as landfill waste disposal sites. Trichloroethylene is one of the most frequently observed volatile organic compounds found in municipal sewage entering public treatment works in the U.S. (Burns and Roe Industrial Services Corp. 1982).

The highest concentrations of trichloroethylene in Canadian water have been recorded in groundwater near waste disposal sites. Groundwater samples near and from the Ville Mercier landfill in Quebec, had trichloroethylene levels ranging from 102 µg/L to 12,950 µg/L. An extremely high concentration of 181 x 10^5 µg/L was also found in a leachate oil sample (Pakdel et al. 1989; Lesage et al. 1989). Lesage et al. (1990) found trichloroethylene levels ranging from below the detection limit (<1 µg/L) to 2,480 µg/L in groundwater collected in May 1988 near a municipal landfill in Gloucester, Ontario, where chlorinated solvents were disposed of between 1969 and 1980. At a contaminated industrial site in Vancouver, groundwater levels from 60 to 21,900 µg/L were detected with a mean concentration of 771 µg/L (Golder Associates Ltd. 1989).

High levels of trichloroethylene have been detected in groundwater at and surrounding a contaminated industrial site in Manitoba (UMA 1992a). The groundwater came from an aquifer known to discharge to the surface. Trichloroethylene concentrations in groundwater below the site reached levels as high as 13,200 µg/L. Wells fed by the same aquifer located several kilometres from the site and used for human and livestock water consumption and irrigation contained trichloroethylene at concentrations up to 490 µg/L.

Trichloroethylene has been detected in groundwater at concentrations of around 2,000 µg/L at an industrial site in Alberta, where this chemical had been used for degreasing over a number of years (*pers. comm.*, John Horgan, Alberta Environment).

Trichloroethylene has also been detected at and beyond the site of a PCB storage and transfer facility operated at Smithville, Ontario between 1978 and 1985. Although PCBs were the major contaminants of the overburden, bedrock and groundwater underlying the site, relatively high levels of trichloroethylene were also present in the dissolved chemical plume (Feenstra 1992). This was a result of the fact that the contaminating dense, non-aqueous phase liquid (DNAPL), contained approximately 2% trichloroethylene in addition to approximately 45% PCBS and 40% mineral oils. The higher solubility of TCE resulted in its dissolution into groundwater and its migration farthest from the site. At monitoring wells situated 75 m and 300 m beyond the farthest extent of DNAPL migration, trichloroethylene concentrations averaged 1,000 and 350 µg/L, respectively (Feentstra 1992).

In Amherst, Nova Scotia, trichloroethylene has been detected in several municipal and private wells at concentrations ranging from 5 to 84 µg/L (NAQUADAT 1991). Trichloroethylene was detected in two groundwater drinking water samples from Prince Edward Island at 1.5 and 1.6 µg/L (detection limit 1.0 µg/L) in 1986. However, nine groundwater samples (non-drinking water) from Nova Scotia did not contain detectable levels of trichloroethylene in 1988 (NAQUADAT 1991).

Data from New Brunswick (1994-2001), Alberta (1998-2001), the Yukon (2002), Ontario (1996-2001) and Quebec (1985-2002) for raw (surface water and groundwater), treated and distributed water indicated that more than 99% of samples contained TCE at concentrations less than or equal to 1.0 µg/L, with a maximum concentration of 81 µg/L. Of those samples with detectable TCE concentrations, most were from groundwater. A 2000 survey of 68 First Nations community water supplies (groundwater and surface water) in Manitoba found that TCE concentrations were non-detectable (<0.5 µg/L) (Yuen and Zimmer 2001).

In 2000, concentrations of TCE exceeding the guideline for Canadian drinking water quality (which was 50 µg/L at the time) were detected in private wells in Beckwith Township, Ontario; the source of the TCE was an abandoned landfill site. In 1997, high levels of TCE were detected in an aquifer under the Valcartier military base in Quebec, and three years later in private wells in Shannon, a town close to the Valcartier base. A study conducted in 2001 found that concentrations of TCE near the source at Valcartier were as high as 13,500 µg/L, while maximum concentrations at the boundary between the base and town of Shannon were from 260 to 340 µg/L (Lefebvre et al. 2003). In 1995, a survey of TCE occurrence across Canada was conducted for 481 municipal/communal and 215 private/domestic groundwater supplies. It was found that 93% of sites had non-detectable levels (detection limits varied from 0.01 to 10 µg/L), 3.6% had a maximum concentration of <1 µg/L, 1.4% had a maximum of 1-10 µg/L, 0.43% had a maximum of 10-100 µg/L and 1.3% had a maximum of >100 µg/L (Raven Beck Environmental Ltd. 1995; Health Canada 2005).

2.5.6 Biota

Levels of trichloroethylene in Canadian biota are not available. Trichloroethylene was identified, but not quantified in adult herring gulls from Pigeon Island near Kingston Harbour, Lake Ontario, and from the Kingston landfill site (Hallett et. al. 1982). Trichloroethylene was detected in marine animal tissue collected in 1980-81 near the discharge zone of a Los Angeles County waste treatment plant. Concentrations were 17 µg/L in the effluent and 0.3-7 µg/kg wet weight in various marine animal tissues (Wu and Schaum 2000).

2.5.7 Sediment

Few data exist on the levels of trichloroethylene in Canadian sediments. Trichloroethylene levels have been measured in St. Clair River sediment following a TCE spill that occurred in 1985 near Sarnia, Ontario. Levels in bottom sediments ranged from below the detection limit (< 0.01 µg/kg) to 1.1×10^5 µg/kg with a mean concentration of 21 µg/kg. After a second industrial spill in the St. Clair River, TCE was measured in 45 of 68 sediment samples taken between September and December (detection limit not stated). The mean concentration was 21 µg/kg, and the maximum was 110 µg/kg (COARGLWQ 1986). It should be noted that at these high levels, such contaminated sediments can act as chronic sources of TCE to overlying surface waters, potentially causing harmful effects to aquatic organisms.

2.5.8 Soil

Soil samples collected throughout Ontario from undisturbed old urban and rural parklands not impacted by local point sources of pollution were analyzed for a variety of chemicals to determine average background concentrations known as "Ontario Typical Range" (OTR_{98}) (OMEE 1993). These OTR_{98} values correspond to the 98th percentile of the sample population analyzed. For trichloroethylene, the OTR_{98} values are 0.63 µg/kg (6.3×10^{-4} mg/kg) and 0.028 µg/kg (2.8×10^{-5} mg/kg) for old urban parkland and rural parkland respectively.

In Vancouver, B.C., 14 of 21 soil samples from a former chemical warehouse and distribution facility were found to have trichloroethylene concentrations ranging from trace to 4.5 mg/kg (mean: 0.036 mg/kg) (Golder Associates Ltd. 1989).

In Manitoba, an investigation on trichloroethylene soil contamination due to disposal practices of spent solvent was done at the Rockwood propellant plant of Bristol Aerospace, focusing on areas of suspected contamination (UMA 1992b). Concentrations of trichloroethylene were highest around metal cleaning operations buildings. Discharge from building floor drains to a regional road side ditch resulted in soil concentrations of trichloroethylene ranging from 140 to 1,000 mg/kg. Other reported localized soil contamination occurred beside solvent burn-off areas, where trichloroethylene concentrations ranged from 0.1 to 890 mg/kg (UMA 1992b).

2.5.9 Food

Trichloroethylene may be present in foodstuffs as a residue from its use as a solvent in food processing or as the result of environmental contamination. A study conducted by McConnell at al. (1975) provided the trichloroethylene content of some common foodstuffs. Values in dairy products ranged from 0.3 μg/kg in fresh milk to 10 μg/kg in English butter. Concentrations measured in meat ranged from 12 to 22 μg/kg and in fruit and vegetables from 3 to 5 μg/kg. However, in recent decades, severe restrictions have been placed on the use of TCE in food processing in North America, and the disposal of TCE is more carefully controlled in other industrial sectors (Health Canada 2003). The US EPA (2001) concluded that exposure to TCE from food was probably low and that there were insufficient food data for reliable estimates of exposure. The daily intakes of TCE in food for Canadian adults (20-70 years old) and children (5-11 years old) were estimated to range from 0.004 to 0.01 μg/kg bw per day and from 0.01 to 0.04 μg/kg bw per day, respectively (Department of National Health and Welfare 1993). These numbers were based on TCE concentrations from U.S. food surveys from the mid- to late 1980s as well as Canadian food consumption data. There is no reason to suppose that these values would have increased in the interim (Health Canada 2003).

2.5.10 Estimated Daily Intake for Canadians

The Government of Canada (1993), under the Canadian Environmental Protection Act (CEPA), has published estimated daily intakes of trichloroethylene for Canadians for ambient air, indoor air, drinking water, food and total intake. These data are reproduced in Table 4. The total intake of trichloroethylene is estimated to be 0.48-0.53 μg/kg bw per day for Canadian toddlers and 0.39-0.41 μg/kg bw per day for Canadian adults.

2.5.11 Extent of TCE Contaminated Sites Issue in Canada

A brief survey was undertaken on the extent of TCE contaminated sites that have been encountered within various Canadian jurisdictions in recent years. Available information is summarized below.

One of the few federal sites contaminated with TCE in central-eastern Canada is the Valcartier military base in Quebec.

Ontario's contaminated sites framework formally addresses movement of TCE from both soil and groundwater to indoor air (Marius Marsh, *Pers. Com.*). Ontario Ministry of Environment (OMOE) records would not include all sites where TCE has been identified as a contaminant, as proponents who clean-up to generic criteria would not necessarily inform OMOE of the original concentrations. In addition, records of TCE contaminated sites that are received are on file at the District or Regional offices. The database on sites that have undergone site-specific risk assessment contains information on 9 sites since 2001 (the database does not go back further in time) where TCE in groundwater occurred at concentrations above the generic criteria, 5 of which exhibited TCE contaminated soil. There are some sites that have significant TCE problems that are affecting a wider area of influence than a single site. These include a site in Peterborough and the Berwick landfill site in Eastern Ontario.

In Manitoba, the only TCE-contaminated site on record was the Rockwood Bristol Plant that was addressed in 1992, as described above.

Alberta has adopted use of the Johnson and Ettinger model to assess the vapour intrusion pathway. It was estimated that Alberta Environment AENV deals with on the order of 20-30 sites with TCE issues at any given time. It was estimated that AENV has dealt with around 50-60 TCE contaminated sites over the last five years. Of these, about ten sites may have had soil vapour intrusion issues that were addressed using a site-specific risk assessment approach.

In British Columbia, the protocols for establishing generic soil standards within the BC Contaminated Sites Regulation did not include consideration of contaminated soil vapour intrusion into buildings; however, this issue has been addressed at number of contaminated sites through site-specific risk assessments. BC is currently revising a number of procedures under the Contaminated Sites Regulation, including how volatile substances such as TCE are handled.

Limited data on trichloroethylene in soil in the Yukon are available. TCE was detected in Yukon soil at a concentration of 2.6 mg/kg in a pit that had been used for approximately 20 years as a disposal site for waste oil, waste solvents, restaurant grease and likely also dry cleaning wastes (Ruth Hall, Yukon Environment, *pers. comm.*).

2.6 Existing Guidelines and Criteria for Trichloroethylene in Various Media

Soil, water and groundwater quality guidelines and criteria established for trichloroethylene in various jurisdictions are summarized in Table 5.

3. ENVIRONMENTAL FATE AND BEHAVIOUR

3.1 Atmosphere

Due to its high vapour pressure, TCE in the atmosphere is expected to be present in the vapour phase, rather then sorbed to particles (Wu and Schaum 2000). In the atmosphere, TCE is destroyed by photooxidation, with a half-life of 3–8 days during the summer months and approximately 2 weeks in cold climates during the winter.

Wet deposition is an important removal process for trichloroethylene, although TCE can soon re-volatilize back to the atmosphere after being deposited.

Mackay et al. (1993) determined the half-life of trichloroethylene to be 170 hours using the level two fugacity model. The relatively short atmospheric half-life (in the order of days) generally precludes significant long-range transport of TCE (Class and Ballschmiter 1986; Bunce 1992). However, under favourable conditions such as high winds and cloud cover, TCE will undergo short- and medium-range atmospheric transport (Mackay 1987).

Chloroacetic acids (especially dichloroacetic acid: DCA) are possible atmospheric degradation products of TCE, and Peters (2003) measured DCA in European soil and plant samples as a means of examining the extent of wet or dry deposition of trichloroethylene. Vegetation concentrations of DCA were approximately 20-fold higher than in soil samples, and were in the range of 4.7 to 17 µg per kg dry weight of vegetation.

3.2 Water

Trichloroethylene volatilizes rapidly from the top layers of surface water, with rates varying according to temperature, water movement and depth, air movement, and other factors (ATSDR 1991). Due to its high volatility, TCE concentrations are normally less than 1 µg/L in surface water (Health Canada 2003) (see also Section 2.5.4). The instantaneous concentration depends on the rate of input of fresh TCE sources relative to removal rates through volatilization and possibly biodegradation.

Estimated volatilization half-lives from a pond, a lake, and a river are 11 days, 4-12 days, and 1-12 days, respectively (Smith et al. 1980). Measured seasonal volatilization half-lives for trichloroethylene in experimental marine ecosystems ranged from 13 to 28 days (Wakeham et. al. 1983). Mackay et al. (1993) determined the half-life of trichloroethylene in water to be 550 hours using the level two fugacity model. The estimated half-lives for photo-oxidation and hydrolysis are 10.7 months and 30 months respectively (Dilling et al. 1975; Pearson and

McConnell 1975). Neither reaction is therefore considered significant to the environmental fate of trichloroethylene.

Concentrated or continuous discharges of TCE to surface and groundwaters can lead to the formation of free product accumulations (non-aqueous phase liquids: NAPLs) due to the density and relatively low water solubility of TCE (Schwille 1988). TCE has a higher specific gravity than groundwater or surface water and free-phase accumulations of TCE tend to sink down until retarded by the underlying substratum. Releases of TCE at concentrations approaching or in excess of solubility limits, therefore, can result on the presence of a dense non-aqueous phase liquid (DNAPL). It should be noted that via these DNAPL accumulations in river or lake bottoms, aquatic organisms, especially benthic organisms, may be exposed to point sources of very high concentrations of often pure TCE, a fact not yet considered in toxicity testing. These DNAPL accumulations can also represent a chronic source of TCE to some aquatic species. Similarly, under certain circumstances, localized high TCE concentrations in water can remain for several days, therefore also causing a chronic exposure for organisms with shorter life cycles.

3.3 Groundwater

Groundwater is a significant recharge source to some surface waters in Canada, particularly during winter and dry summer months. As well, it provides drinking water for wildlife through surface springs. High concentrations are frequently observed in contaminated groundwater where volatilization and biodegradation are greatly limited (Schwille 1988). The highest concentrations of TCE in Canadian water have been recorded in groundwater, suggesting that in specific circumstances Canadian groundwater may contaminate surface waters through recharging.

In groundwater, biodegradation may be the most important transformation process for TCE, although it does not appear to occur rapidly. Various aerobic and anaerobic biodegradation studies in the field and laboratory found TCE to be resistant or only slowly biodegraded with half-lives of several months to years (Roberts et al. 1982; Rott et al. 1982; Wilson et al. 1983a; 1983b; 1986; Wakeham et al. 1983). Other studies noted more rapid biodegradation, depending on the local conditions, induction, and artificial nutrient enrichment with half-lives on the order of a few months (Tabak et al. 1981; Parsons et al. 1984; Wilson and Wilson 1985; Barrio-Lage et al. 1988). These results indicate that TCE in groundwater can undergo biodegradation, but at removal rates much slower than would occur where volatilization is possible.

The major biodegradation products of TCE in groundwater are dichloroethylene, chloroethane, and vinyl chloride (Smith and Dragun 1984, Vogel and McCarty 1985, Baek and Jaffe 1989). Fliermans et al. (1988) reported optimal TCE biodegradation occurring at a slightly basic pH (range 7.0 to 8.1) with optimal temperatures observed between 22 and 37 °C, and with little degradation at temperatures below 12 °C or above 60 °C. An aerobic degradation study of TCE in seawater showed that 80% of TCE was degraded in eight days (Jensen and Rosenberg 1975). In anaerobic environments, reductive dechlorination of TCE occurred at redox potentials between -50 and -150 mV in the presence of excess substrate (Kastner 1991).

TCE is not used as the sole carbon source under aerobic conditions (Henry and Grbic-Galic 1991). Substrates found to stimulate aerobic TCE degradation by bacteria include acetate, glucose, phenol, formate (Fliermans et al. 1988; Semiprini et al. 1990), methane, methanol (Little et al. 1988, Berwanger and Barker 1988, Strandberg et al. 1989), toluene, *o*-cresol, *m*-cresol (Nelson et al. 1987), ammonia (Arciero et al. 1989), propane (Wackett et al. 1989), and a natural gas mixture (Wilson and Wilson 1985).

3.4 Sediments

Limited field measurements suggest that TCE does not partition to aquatic sediments to any appreciable degree (Pearson and McConnell 1975). However, sediments with a high organic content were shown to have a high adsorptive capacity for TCE (McConnell et al. 1975, Lay et al. 1984, Smith et al. 1990). Using a level two fugacity model, Mackay et al. (1993) determined that only 0.0028% of a known quantity of TCE is thought to partition to suspended and bottom sediments.

Limited biodegradation may occur in sediments. Methane-utilizing bacteria isolated from sediment degraded 630 ng/mL of TCE to 200 ng/mL in 4 days at 20°C. Trichloroethylene was converted to carbon dioxide, but was not found to degrade to dichloroethylene or vinyl chloride (Fogel et al. 1986).

3.5 Soil

Generally, the majority of TCE released to soil surfaces will volatilize to the atmosphere. However, significant accumulation of the chemical in saturated and unsaturated zones may result where TCE penetrates the surface before evaporation (Schwille 1988). In most cases, TCE enters the soil media as an undiluted solution from spills or leaking storage tanks, as leachate from landfill sites, or by wet deposition in rain and snow from the atmosphere (Muraoka and Hirata 1988). It is highly mobile in the subsurface environment and is susceptible to leaching (Schwille 1988). The half-life of TCE in soil as determined by the level two fugacity model is 1,700 hours (Mackay et al. 1993).

Transport processes in the soil include gaseous and liquid diffusion, gaseous and liquid dispersion as pure liquid or as solute in water, and advection throughout the headspaces within porous soils (Peterson et al. 1988; Cho and Jaffé 1990). The major routes of transport within the soil occur via the vapour phase (Peterson et al. 1988; Smith et al. 1990), or in the liquid phase via vertical migration until an impenetrable layer is encountered (Schwille 1988). All of these processes are dependent on the hydrological and geochemical characteristics of the medium (Schwille 1988; Colborn 1990). Pools of TCE may be retarded by saturated zones, but the limited solubility of TCE will allow some direct transport of pure TCE droplets within the water phase of aquifers. The degree of water infiltration into the soil may also influence TCE transport.

The ability of soils to retain TCE is governed by partitioning to organic matter or sorption onto mineral surfaces (Stauffer and MacIntyre 1986; Ong and Lion 1991b). TCE partitioning to soil will be the dominant mechanism of soil retention even at organic carbon contents of 0.1% (Ong and Lion 1991b). Partitioning to organic matter within soils is a function of the hydrophobicity of TCE (Lesage et al. 1990). Partitioning processes are influenced by moisture content (Chiou et al. 1988), soil composition, types of organic matter and organic matter content (Grathwhol 1990) and to a slight degree pH (Stauffer and MacIntyre 1986). Garbarini and Lion (1986) suggested a relationship between sorption and composition of the organic matter, with sorption decreasing as the proportion of oxygen in organic matter increases.

High hydrogen to oxygen ratios may indicate relatively few oxygen containing functional groups within the soil material. A soil having this characteristic may show relatively low polarity and high hydrophobicity and stronger TCE partitioning (Grathwohl 1990). Adsorption is also controlled by the moisture content of the soil. Water tends to suppress TCE adsorption by competitively sorbing to clay surfaces providing a polarized shield to TCE (Rao et al. 1989). Trichloroethylene increasingly partitions to soil with increasing organic matter content (Garbarini and Lion 1986; Seip et al. 1986; Stauffer and MacIntyre 1986). In some subsurface soils, TCE sorption and desorption can be slow, and thus, subsurface liquid TCE can continue to contaminate groundwater aquifers and soils long after pollution sources have been eliminated and remedial actions have been performed (Smith et al. 1990).

Surface soils having higher organic carbon content than deeper soils are likely to have significant TCE adsorption capacities and effectively act as a barrier to volatilization losses (Ong and Lion 1991a). Fuentes et al. (1991) concluded that soil moisture content can negatively affect TCE vapour phase diffusion. Diffusion coefficients for TCE have been observed to be 0.0237 to 0.0292 cm^2/s at 1 to 3% soil moisture and 0.0067 to 0.0070 cm^2/s under wet conditions of 13 to 15% soil moisture (Fuentes et al. 1991). Reduced diffusion is a consequence of increased water sorption and a reduction in penetrating air volume.

TCE within the saturated soil zones is relatively immobile (Marrin and Thompson 1987). It is suggested that a diffusive breakthrough time for a mobile organic, such as TCE, in the 1 m thick clay liner, as used by regular municipal landfill sites, may be less than 10 years (Johnson et al.

1989). The measured apparent in-soil diffusion coefficients for TCE through soil columns ranged from 0.254 x 10^{-3} to 1.986 x 10^{-3} cm^2/s, with the larger values being associated with higher soil porosity levels (Hutter et al. 1992).

Significant movement of TCE in soil was demonstrated by soil infiltration systems in which TCE was observed to leach rapidly into groundwater (Giger et al. 1983; Schwarzenbach et al. 1983). Desorption or leaching from soils occurs as an initial fast phase lasting hours followed by a slow phase which may take from days to months depending on the degree of equilibrium and on soil exposure time to TCE (Pavlostathis and Jagial 1991). Long-term soil exposure to TCE produces a fraction of sorbed contaminant that is relatively resistant to desorption. This is due to the time period necessary (on the order of months to years) for TCE to reach equilibrium within soil systems. TCE may continue to contaminate aquifers by desorption long after remediation actions have been performed (Smith et al. 1990).

Biodegradation is not an efficient TCE loss mechanism. Evaporative loss may be a much more effective removal process due to the slow degradation reaction rates in natural soils. While in a natural soil mixture no degradation of TCE by anaerobic soil microorganisms has been reported, even after 16 weeks (Wilson et al. 1981), artificial nutrient enrichment and induction in the same soil led to extensive aerobic biodegradation (Wilson and Wilson 1985). The slow biodegradation rates in natural soils are due to nutrient limitations, competition for resources with other non-degrading microbes, and a lack of proper biological inducing conditions. Walton and Anderson (1990) found that in previously exposed (i.e. induced) soils, microbial degradation occurs faster in vegetated than in the non-vegetated soils. In organic soils, sorption processes in which TCE becomes buried within the micropores of soil aggregates, may also make TCE unavailable to microorganisms capable of degradation (Pavlostathis and Jagial 1991). Very high concentrations of TCE, as found in some contaminated plumes, may also exhibit a toxic effect on microbial populations which will inhibit degradation.

Canadian wetlands are potentially the ecosystems at greatest risk when posed with contamination from dense non-aqueous phase liquids, such as trichloroethylene. In these habitats, shallow surface waters represent a barrier to TCE movement away from surface dwelling organisms. Trichloroethylene sorption to these organic carbon rich soils (18-20% organic carbon) may provide a significant exposure route to terrestrial omnivores and detritivores from the direct ingestion of trichloroethylene contaminated soil or invertebrates.

3.6 Biota

There is little evidence to suggest substantial bioaccumulation of TCE in living tissues. Both the moderate *n*-octanol/water partition coefficient of TCE and various field studies from different trophic levels indicate that bioaccumulation of TCE is a minor process (Pearson and McConnell

1975; Dickson and Riley 1976; Kawasaki 1980; Barrows et al. 1980; Ofstad et al. 1981; Wang et al. 1985; Freitag et al. 1985; Smets and Rittmann 1990). Bioaccumulation factors measured ranged from <3 for muscle tissue of marine and freshwater birds to approximately 100 for fish livers (Pearson and McConnell 1975).

According to ATSDR (1997), TCE has been detected in small amounts in fruits and vegetables, suggesting a potential for limited phytoaccumulation. Laboratory studies with carrot and radish plants and radioactively labelled TCE (Schroll et al. 1994) lead to calculations of bioconcentration factors (BCFs) in the range of 4.4 to 63.9. As noted by the authors, however, these may not have represented true soil-plant BCFs, since the experiment indicated that uptake occurred mainly through the foliage as opposed to the roots in these plants, with subsequent translocation throughout the plant tissues.

Hsieh et al. (1994) were unable to find any field or laboratory-based data for the partitioning of TCE from soil to plants, food products such as whole milk, eggs or fresh meat, human breast milk, or other terrestrial biota tissues. This document provides estimation techniques in the absence of such data; however, these should be used cautiously and only with a full appreciation of their limitations.

Ma and Burken (2002; 2003) demonstrated that TCE is taken up by hybrid poplars and volatilized to the atmosphere. The diffusion of TCE along the transpiration pathways was the dominant process, although volatilization also occurred through the stems and leaves. Laboratory and field studies concluded that TCE transpiration rates decreased with elevation (tree height) and in the radial direction, providing fundamental evidence for diffusion. Poplar cuttings showed no signs of toxicity or inhibition in these short-term experiments at a concentration up to 50 ppm. No leaf wilting, chlorosis, or water usage reduction was observed. Partitioning coefficients of TCE between water, air, and biomass are determined by the physicochemical characteristics of the contaminant, such as Henry’s law constant and vapour pressure (Ma and Burken 2002).

Hsieh et al. (1994) summarized bioconcentration factors (BCFs) reported by two researchers for TCE transfers from water to fish (Veith et al 1980, as reported in Hsieh et al.: Bluegill sunfish; Freitag et al. 1985, as reported in Hsieh et al.: Golden Ide). The bluegill sunfish log BCF was reported as 19 (unitless) and the Golden Ide BCF was 90 (unitless).

3.6.1 Fate Modelling - The Fugacity (LEVEL III) Model

The advanced Level III fugacity model incorporates expressions for intermedia transport rates by various diffusive and non-diffusive processes: deposition, diffusion, evaporation, run-off, resuspension, and includes dispersive phases within media, e.g. air-aerosol, water-suspended particles and fish, soil solids-air, soil solids-water, and sediments solids-pore water (MacKay et al. 1985). Advection velocity and chemical decay half-lives are also included in the Level III fugacity model. In order to simplify the required calculations, four primary media, soil, air, water and sediment are used. The emission rate of TCE utilized was 1,000 kg/hr. The results of the model indicated that for a TCE release to soil, the intermedia-transport rates for TCE for soil-air were the highest (965.3 kg/hr) while the transport rate between soil and water was second highest (5.08 kg/hr). This indicates that the soil-air is the dominant pathway of the environmental movement of TCE released to the soil. Soil-water, air-soil, water-air are less significant pathways and air-water, water-sediment and sediment to water transfer values were relatively insignificant (MacKay et al. 1993).

4. BEHAVIOUR AND EFFECTS IN BIOTA

The available information on the toxicological effects of trichloroethylene on soil microbial processes, terrestrial plants and invertebrates, as well as mammals and birds has been reviewed and summarized in this chapter in support of the derivation of environmental soil quality guidelines.

Most of the animal toxicity data comes from laboratory studies with surrogate species such as rats, mice, and chickens with trichloroethylene often dosed in water or air. Additional toxicity and pharmacokinetic data using soil sorbed trichloroethylene are required for terrestrial invertebrate and plant species.

It has been recognized that damage to an ecosystem may be caused by a total environmental burden and not by a single contaminant. Most chlorinated aliphatics have a similar mode of action in organisms and thus the toxicity of mixtures of these contaminants needs detailed examination. The necessity for evaluating mixtures is underscored by the occurrence of complex contaminant mixtures at contaminated sites. Research should focus on common and relevant industrial mixtures such as trichloroethylene and tetrachloroethylene, decay products of these compounds, and stabilisers commonly added to commercial preparations of these compounds (Table 3).

4.1 Microbial Processes

The toxicity of trichloroethylene varies widely among microbial organisms. Trichloroethylene toxicity to methanotrophic bacteria and other bacteria capable of degrading trichloroethylene was speculated to occur via TCE degradation intermediates. Fliermans et al. (1988) suggested that the biological tolerance to TCE in contaminated environments generally appears to be in the range of 200 to 300 mg/L. Inamori et al. (1989), reported an EC_{50} of 330 mg TCE/L as determined by a reduction in the consumption of dissolved oxygen by unspecified soil microorganisms. Very tolerant populations, like *Rhodococcus erythropolis* JE 77 can survive and partially degrade trichloroethylene at concentrations of up to 1,000 mg/L (Ewers et al. 1990).

Inhibition of trichloroethylene decay by the bacterial species *Methylosinus trichosporium* OB3b was reported to occur at a concentration of 9.2 mg/L and 26.3 mg/L in a cell suspension of 420 mg of cells per L (Oldenhuis et al. 1989). Marinucci and Chervu (1985) reported that complete inhibition of microbial activity occurred in mixed cultures at concentrations of 5,000 mg/kg in soil and 1,000 mg/L in water. Reductions in soil microbial biomass, measured as ATP content, were reported as acute effects occurring at 10 mg/L (Kanazawa and Filip 1986).

Effects of chronic exposure to trichloroethylene on microbial communities may include decreased microbial culture vitality, lowered ability to utilize substrate, and inhibited enzyme activity. Cultures obtained from groundwater consisting of mixed methanotrophs and purified *Methlomonas* sp. demonstrated a three-fold decrease in their ability to utilize methane following a 2h incubation with 6 mg/L (Henry and Grbic-Galic 1991). The viability of *Methlomonas* cultures was also reduced by trichloroethylene whereby exposed cultures had lower cell counts (2 ±1·105 CFU/mL) relative to the controls (5 ± 2·106 CFU/mL) (Henry and Grbic-Galic 1991). Kanazawa and Filip (1986) reported reduced proteinase activity from 5.58 nmol/g/min (controls) to 2.69 nmol/g/min at trichloroethylene concentrations of 10 mg/kg. Phosphatase and phosphodiesterase activities were also reduced by 25% and 41%, respectively, after 28 days exposure to trichloroethylene.

4.2 Terrestrial Plants

Consulted plant toxicity tests with trichloroethylene are presented in Table 7.

Radish (*Raphanus sativus*) was exposed to concentrations of trichloroethylene in an artificial soil following the method of Greene et al. (1988) (Environment Canada 1995). Ten seeds in 30 g of artificial soil and 30 g of sand were exposed for 72 h to concentrations (nominal) ranging from 0 to 3,661 mg TCE per kg soil. Both nominal and measured soil concentrations are reported in Table 7. The discrepancy between corresponding pairs of nominal and measured data indicates that significant volatile losses were incurred between spiking the soil and analytical measurements being taken. Accordingly, the measured concentrations are assumed to be a better estimate of exposure concentrations than the nominal values. Values for the NOEC, LOEC EC_{25}, and EC_{50} for seedling emergence based on measured data were 9, 16, 14, and 53 mg/kg. Similarly, Environment Canada (1995) reported the effect of trichloroethylene on lettuce (*Lactuca sativa*) seedling emergence. The test duration was 120 h. Values for the NOEC, LOEC EC_{25}, and EC_{50}, for seedling emergence based on measured data were 16, 48, 26, and 37, mg/kg.

Several previous studies of TCE effects on plants in spiked soil experiments (Pestemer and Ausburg 1989; Kordel et al. 1984; Ballhorn et al. 1984) failed to account for the rapid losses from soil of TCE during preparation of test soils and during the exposure period, as a result of the high volatilization rate. Recent experience in Canadian toxicity testing laboratories with benzene, petroleum hydrocarbons, MTBE, and other highly volatile substances has shown that nominal (spiked) concentrations cannot be used to estimate the exposure concentration, and that modified experimental protocols are required to derive meaningful dose-response results for laboratory or field mesocosm tests.

Native Canadian fir trees (*Abies alba*), non-native tree species Norway spruce (*Picea abies*) and

European beech (*Fagus silvatica*), and other trees growing in regions with low SO_2 and NO_X concentrations have exhibited increasing incidence of chlorosis, necrosis, and premature needle and leaf loss over the last two decades in the Northern Hemisphere, especially Germany, Finland, and North America (Frank 1989; 1991). This tree damage has been attributed to exposure to chlorinated ethenes, namely trichloroethylene and tetrachloroethylene, rather than classic air pollutants such as acid precipitation, NO_X, SO_2, or O_3 (Frank 1989; Figge 1990; Frank et al. 1991; Frank et al. 1992a; 1992b). Trichloroethylene and tetrachloroethylene are converted by photochemical reaction to highly reactive intermediates (radicals) which are believed to cause the damage (Frank 1989). Several studies have explored the reaction pathways producing these intermediates (see Frank and Frank 1985, Frank 1991). Thus, photoactivation of trichloroethylene and tetrachloroethylene is required to cause the observed phytotoxic effects. The ambient trichloroethylene concentrations in damaged forest areas range from below the detection limit (0.1 µg TCE/m^3) to 0.7 µg TCE/m^3, with averages around 0.2 to 0.5 µg TCE/m^3 (Frank 1989, Frank et al. 1991). However, it was shown that air concentrations can undergo rapid and extreme fluctuations, depending on the local meteorological conditions (Ohta et al. 1977; Frank et al. 1991).

In a field experiment in southwestern Germany, continuous and simultaneous exposure of a 10-year old Serbian spruce (*Picea omorica*) for approximately seven months to levels of trichloroethylene and tetrachloroethylene averaging 4.6 µg/m^3 and 11.8 µg m^3, respectively led to chlorosis, necrosis, and premature loss of the sun-exposed needles. The observed damage intensified after periods of several clear, sunny days. A hornbeam shrub (*Carpinus betulus*), located about two metres downwind developed the same symptoms (Frank and Frank 1985).

In a laboratory experiment, needles from five year old spruce trees (*Picea abies*) were exposed to trichloroethylene at 180 ppbv (approximately 1 mg/m^3) and irradiated with light in the visible and UV range for five hours (Frank and Frank 1986). The exposed needles showed a strong alteration in their apparent colour, changing from the natural dark-green to a dirty brown-green. Subsequent HPLC analysis revealed a significant decrease in the photosynthetic pigment concentrations with chlorophyll-alpha and beta-carotene being the most severely affected. Chlorophyll-alpha and beta-carotene levels were down to 33% and 43% of control levels respectively. This artificially-induced damage was similar to the field damage observed in areas with high levels of trichloroethylene in the atmosphere.

4.3 Terrestrial Invertebrates

Consulted toxicity tests on invertebrates are presented in Table 8.

The survival of the earthworm *Eisenia foetida* exposed to trichloroethylene was reported by Environment Canada (1995). The test followed the protocol for short term toxicity testing by

Greene et al. (1988). Applied nominal trichloroethylene concentrations ranged from 0 to 7,321 mg/kg and the measured concentrations ranged from 0 to 440 mg/kg. Both nominal and measured soil concentrations are reported in Table 8. The discrepancy between corresponding pairs of nominal and measured data indicates that significant volatile losses were incurred between spiking the soil and analytical measurements being taken. Accordingly, the measured concentrations are assumed to be a better estimate of exposure concentrations than the nominal values. Values for the NOEC, LC_{25}, LC_{50} and LOEC, based on measured concentrations, were 60, 79, 106 and 159, mg/kg, respectively. It should be noted that at the LOEC in this study, 100% mortality of earthworms was observed. Therefore, the LC_{25} and LC_{50} were estimated by interpolating between the NOEC and LOEC.

The earthworm *Eisenia foetida* was exposed to trichloroethylene in a contact filter paper test for 48 hours (Neuhauser et al. 1985). An LC_{50} of 105 mg TCE/cm^2 was reported. It was noted that the filter paper based contact test does not accurately represent what would likely occur in soil systems (Neuhauser et al. 1985).

4.4 Terrestrial Birds and Mammals

No information on the toxicity of TCE to avian or mammalian wildlife was found. Therefore, studies on laboratory mammals have been used as the basis for the discussion of possible toxicity to wildlife. These studies are reviewed in Section 5. Most studies on TCE have used inhalation as the route of exposure. Trichloroethylene may be taken up by wildlife by three main routes: dermal absorption of contaminated air and/or soil; oral ingestion of contaminated soils; inhalation of contaminated air.

The mammalian toxicity data used to derive human toxicity reference values (Section 5) can also be used to estimate threshold effects levels for other mammalian species, allowing for various forms of uncertainty when extrapolating between species with broadly different evolutionary and life histories, foraging strategies, gut physiology, behaviours, and so on.

Inhalation is expected to be a minor route of exposure for wildlife. However, subsurface areas contaminated by high concentrations of TCE (e.g. in the case of spills or industrial discharges) may act as a localized source of high inhalation and dermal exposure for certain wildlife, especially burrowing mammals, reptiles, and amphibians. There are no available direct soil contact exposure data for vertebrates. Similarly certain burrowing animals and grazing ungulates may incidentally ingest contaminated soil during daily activities, especially during foraging.

5. BEHAVIOUR AND EFFECTS IN HUMANS AND MAMMALIAN SPECIES

5.1 Overview

The mammalian oral toxicology of trichloroethylene has been recently and thoroughly reviewed by Health Canada (2004), who identified an oral tolerable daily intake and cancer unit risk for trichloroethylene in support of the revised Canadian drinking water guideline for trichloroethylene. Health Canada have not published an inhalation reference concentration; however an inhalation RfC is calculated in a draft U.S. EPA (2001) document. These exposure limits, and the rationale for the values adopted in this document are discussed further in Section 5.9.

It is not the role or the intention of this document to critically re-evaluate the human toxicology of trichloroethylene. This has already been done by agencies responsible for protecting human health in Canada and other jurisdictions. Accordingly, Sections 5.2 to 5.8 below represent just a brief summary of trichloroethylene toxicology with the objective of putting the exposure limits (Section 5.9) in context. This summary is largely based on Health Canada (2004) with additional information included as appropriate.

5.2 Classification

Both cancer and non-cancer endpoints are significant in the toxicological evaluation of trichloroethylene. Health Canada (2004) classifies trichloroethylene as Group II (probably carcinogenic to humans). This classification has been confirmed by the International Agency for Research on Cancer (IARC 1995), who list trichloroethylene as Group 2A (probably carcinogenic to humans). A draft report by the U.S. EPA (2001) classifies trichloroethylene as “highly likely to produce cancer in humans”. For the purpose of calculating soil quality guidelines for the protection of human health in this document, both cancer and non-cancer endpoints are evaluated separately for each exposure pathway, and the lower of the two values is used as the applicable guideline.

5.3 Pharmacokinetics

5.3.1 Absorption

TCE is rapidly and extensively absorbed by all routes of environmental exposure—ingestion, inhalation, and skin contact (U.S. EPA 2001). Significant inter- and intra-species variability in TCE absorption following all routes of exposure has been well documented (Health Canada 2004).

5.3.2 Distribution

Once absorbed, TCE is distributed via the circulatory system throughout the body, where it can accumulate in fat and other tissues. Storage of TCE in fat represents an internal source of exposure that can later release TCE again into the circulation (U.S. EPA 2001). Estimated half-lives for TCE in adipose tissue are in the range of 3.5 to 5 hours, as opposed to other compartments of the body where half-lives can be just a few minutes (Lash et al. 2000). Studies in humans have found trichloroethylene or its metabolites in most major organs and tissues. Primary sites of distribution include the lungs, kidneys, liver, and central nervous system (Health Canada 2004).

5.3.3 Metabolism

Many of the resulting metabolites of TCE in humans (e.g., trichloroacetic acid, dichloroacetic acid, chloral hydrate, and trichloroethanol, among others) are thought to be responsible for much of the toxic effect of TCE (ATSDR 1997). A state-of-the science review of trichloroethylene metabolism is available in Lash (2000), and the interested reader is directed there for detailed information. A summary of the metabolism of TCE follows.

Trichloroethylene metabolism occurs primarily in the liver (Health Canada 2004). Trichloroethylene is metabolized by two main pathways: oxidation by cytochrome P-450, and conjugation with glutathione by glutathione-S-transferases. Exposure to trichloroethylene results in exposure of tissues to a complex mixture of metabolites (U.S. EPA 2001).

The initial step in the oxidative metabolic pathway is thought to be the formation of an unstable epoxide (trichloroethylene oxide). The predominant pathway is then spontaneous rearrangement to chloral (2,2,2-trichloroacetaldehyde; CCl_3-CHO), followed by hydration to chloral hydrate (2,2,2-trichloro-1,1-ethanediol; CCl_3-$CH(OH)_2$). Chloral hydrate is then metabolized to trichloroacetic acid (CCl_3-COOH) and other metabolites. Trichloroacetic acid is the primary trichloroethylene metabolite in blood (Health Canada 2004).

The glutathione metabolic pathway starts by the conjugation of trichloroethylene with glutathione to form S-(1,2-dichlorovinyl) glutathione, and may then proceed via cysteine conjugation, N-acetylation and other pathways to yield a complex series of metabolites. Glutathione conjugation of trichloroethylene occurs more slowly than the cytochrome P-450 oxidation pathway (Health Canada 2004).

5.3.4 Elimination

Trichloroethylene clearance is well characterized both in humans and animals. Elimination kinetics of trichloroethylene and its metabolites vary by route of exposure, however, elimination pathways appear to be similar for ingestion and inhalation (Health Canada 2004). Trichloroethylene is eliminated i) unchanged in expired air, or ii) as metabolites, primarily in the urine.

5.4 Acute Exposure

The ability of TCE to cause neurotoxic effects is well established. In general, TCE produces a "solvent narcosis" that may be related to effects on membrane fluidity and may include anesthetic effects. TCE was formerly used as a general anesthetic and induces this effect at about 2,000 ppm (ATSDR 1997).

Acute human oral exposure to trichloroethylene has been shown to result in symptoms ranging from gastrointestinal effects and cardiovascular anomalies to hepatic failure and death (ATSDR 1997). Studies on laboratory animals have yielded oral LD_{50} values of 2,402 mg/kg bw per day for mice (Tucker et al. 1982) and 7,208 mg/kg bw per day for rats (Smyth et al. 1969). Other effects of acute oral exposure of laboratory animals to trichloroethylene include reduced body weight gain, increased liver and kidney weights, and neurological symptoms (ATSDR 1997).

Acute inhalation of TCE has been associated with dizziness, headache, sleepiness, nausea, confusion, blurred vision, and weakness in several human studies cited by U.S. EPA (2001). Acute inhalation exposure to high concentrations of trichloroethylene in humans has been observed to result in a variety of symptoms including nausea, vomiting, central nervous system depression, and death (ATSDR 1997). In laboratory animals, acute inhalation exposure to death, effects on the cardiovascular system and effects on the lungs and kidneys (ATSDR 1997). The LC_{50} for Sprague-Dawley rats in a 4 hour exposure test was 12,500 ppm (Siegel et al. 1971).

5.5 Sub-Chronic and Chronic Exposure

5.5.1 Oral Exposure

A significant number of animal studies have investigated the effects of longer term oral exposure to TCE. ATSDR (1997) reviewed 22 oral studies of intermediate duration and 13 studies of chronic duration (excluding cancer studies). Renal and hepatic effects have been well documented in longer term oral exposure, but effects in other systems (respiratory, gastrointestinal, hematological, immunological, and neurological) have also been noted. Studies on reproductive and developmental toxicity are discussed separately in Section 5.6. Note that, due to the limited solubility of TCE, many of the oral studies were carried out by gavage in a corn oil carrier. These studies are confounded by the fact that corn oil has been found to alter the pharmacokinetics of TCE and to affect lipid metabolism and other pharmacodynamic processes (Health Canada 2004).

Renal Effects

The lowest level at which renal effects were reported (ATSDR 1997) was in a study by Maltoni et al. (1986). Sprague-Dawley rats were exposed to TCE by gavage in corn oil 5 days a week for 52 weeks. This study revealed renal tubular nucleocytosis in 50% of male rats exposed to TCE at 250 mg/kg bw per day (LOAEL). The NOAEL for male rats in this study was 50 mg/kg bw per day. NTP (1988; 1990) and other groups also found renal effects at doses in the range 500 – 1,000 mg/kg bw per day.

Hepatic Effects

The lowest level at which hepatic effects were reported (ATSDR 1997) was in a study by Buben and O'Flaherty (1985). Swiss-Cox mice were exposed to TCE by gavage in corn oil 5 days a week for 6 weeks. This study showed a dose related progression of hepatic alterations in male rats with increasing doses of trichloroethylene, beginning with an increase in the relative liver weight at 100 mg/kg bw per day and enlarged liver cells and decreased DNA concentration at ≥400 mg/kg bw per day. In another study, a dose-related effect was seen in male mice treated with trichloroethylene for 3 weeks (Stott et al. 1982). At 250 and 500 mg/kg bw per day, there were slight increases in cytoplasmic eosinophilic staining indicative of changes in hepatocyte organelles.

5.5.2 Inhalation Exposure

A significant number of animal studies have investigated the effects of longer term inhalation exposure to TCE. ATSDR (1997) reviewed 20 inhalation studies of intermediate duration and 1 study of chronic duration (excluding cancer studies). Consistent with the oral studies, renal and hepatic effects were observed in some inhalation studies. In addition, neurological effects were well documented.

Renal Effects

Maltoni et al. (1986) exposed Sprague-Dawley rats to TCE by inhalation for 7 hours/day, 5 days/week for 104 weeks. Kidney effects (renal tubule meganucleocytosis) were observed in male rats at 300 ppm (approximately 1,600 mg/m^3) (LOAEL). The NOAEL for male rats in this study was 100 ppm (approximately 550 mg/m^3).

Hepatic Effects

Kjellstrand et al. (1983) exposed NMRI mice to TCE by inhalation continuously for 30 days. They observed liver effects (increased plasma butyrylcholinesterase (BuChE) activity and increased liver weight) in male rats at 75 ppm (approximately 410 mg/m^3) (LOAEL). The NOAEL for male rats in this study was 37 ppm (approximately 200 mg/m^3). Liver weight, liver histology, and serum BuChE activity returned to normal 4 months later, indicating reversibility of the hepatic effects. The study authors suggested that the effects were not toxicologically significant. No other intermediate or chronic studies finding hepatic effects were reported in ATSDR (1997).

Neurological Effects

The lowest level at which neurological effects were reported (ATSDR 1997) was in a study by (Haglid et al. 1981). Mongolian gerbils were exposed to TCE by inhalation continuously for 3 months followed by a recovery period of 4 months. Exposure to 60 ppm TCE (approximately 330 mg/m^3) resulted in increased brain S100 protein content, consistent with astroglial hypertrophy and proliferation. Exposure to 320 ppm (approximately 1,750 mg/m^3) produced significantly elevated deoxyribonucleic acid (DNA) content in the cerebellar vermis and sensory motor cortex. It is not known whether such effects reflect adverse changes. Arito et al. (1994) exposed Wistar rats to TCE by inhalation for 8 hours/day, 5 days/week for 6 weeks. At 50 ppm (approximately 270 mg/m^3) decreased wakefulness was observed during the exposure. Effects remaining at 22 hours after the end of the 6-week exposure included decreased heart rate during sleep at 50 ppm and decreased wakefulness after exposure of 100 ppm (approximately 550 mg/m^3). Other studies found a range of neurological effects at higher concentrations.

5.6 Reproductive Effects and Teratogenicity

There is some evidence that exposure to trichloroethylene in drinking water may cause certain types of birth defects in humans. However, this body of research is still far from conclusive and there is insufficient evidence to determine whether or not there is an association between exposure to TCE and developmental effects (ATSDR, 1997).

A number of studies in rats and mice have shown a range of reproductive and developmental effects including increased prenatal loss, impaired copulatory behaviour, decreased sperm motility, decrease in the number of myelinated fibers in the hippocampus, decreased neonatal survival, and increased perinatal mortality (ATSDR 1997).

The most sensitive endpoint observed in these studies was in a study by Dawson et al., (1993). Dawson et al. (1993) exposed groups of 9-39 female rats to trichloroethylene in drinking water (1.5 or 1,100 ppm) either before pregnancy (for 3 months prior to mating), before and during pregnancy (2 months prior plus 21 days into gestation), or during pregnancy only (21-day gestation). Maternal toxicity was not observed in any of the exposure groups. Fetal heart defects were not observed in fetuses from dams exposed only before pregnancy. Abnormal fetal heart development was observed at both concentrations in dams exposed before and during pregnancy (3% in controls; 8.2% at 0.18 mg/kg/day; 9.2% at 132 mg/kg/day). This was based on examination of 2,037 hearts from litters of l-20 live fetuses. In dams exposed only during pregnancy, fetal heart defects were observed only at the higher dose (10.4% versus 3% in controls).

5.7 Carcinogenicity and Genotoxicity

As noted in Section 5.2, Health Canada (2004) and several other agencies classify trichloroethylene as a probable human carcinogen. This is supported by both human epidemiological studies and laboratory animal carcinogenicity studies. The U.S. EPA (2001) has reviewed and calculated slope factors from 5 human studies and 4 animal studies. The human studies were used to calculate slope factors based on liver cancer, kidney cancer, and non-Hodgkin's lymphoma. The animal studies were used to calculate slope factors based on liver cancer and kidney cancer.

Data regarding the genotoxicity of trichloroethylene suggest that it is a very weak, indirect mutagen. The potential for heritable gene mutations and the mechanisms of carcinogenicity are not known (ATSDR 1997).

5.8 Toxicological Limits

5.8.1 Oral Exposure – Non-Cancer Effects

Health Canada (2004) calculated a tolerable daily intake (TDI) for oral exposure for non-cancer effects of 0.00146 mg/kg bw per day. The critical study (Dawson *et al. 19*93) on which this TDI was based was a developmental study in which female rats were exposed to trichloroethylene in their drinking water before and/or during pregnancy. This study yielded a LOAEL of 0.18 mg/kg bw per day, based on an increased incidence of fetal heart defects in the young of dams exposed to trichloroethylene at this level before and during pregnancy. The critical study was selected based on the seriousness of the effect (heart defects), the presence of similar effects in epidemiological studies, as well as the observation of similar malformations in studies of TCE metabolites. Since this study did not identify a NOAEL, Health Canada (2004) calculated a benchmark dose (BMD_{10}), which was an estimate of the dose that would result in a low incidence (10%) of fetal heart malformations over background. In this study the background incidence of fetal heart defects was 3%, see Section 5.6, and so the BMD_{10} is an estimate of the dose that would result in a 3.3% incidence of fetal heart defects. The BMD_{10} calculated by Health Canada (2004) was 0.146 mg/kg bw per day. Health Canada (2004) applied an uncertainty factor of 100 to this value and calculated a TDI of 0.00146 mg/kg bw per day. The Health Canada (2004) value (0.00146 mg/kg bw per day, Table 9) is used in this report to calculate guidelines protective of non-cancer endpoints in humans for soil and food ingestion exposure pathways.

The Health Canada (2004) TDI is higher than the oral reference dose (RfD) for non-cancer effects calculated by the U.S. EPA (2001) of 0.2 μg/kg bw per day; however, it is based on a more sensitive toxicological endpoint. The U.S. EPA (2001) RfD was based on liver toxicity in rats and mice. U.S. EPA (2001) selected a departure point of 1 mg/kg bw per day based on a subchronic NOAEL at 1 mg/kg-d in mice (Tucker et al. 1982), a subchronic LOAEL at 1 mg/kg-d in mice (Buben and O'Flaherty 1985), and a subchronic LED_{10} of 0.6 mg/kg-d in rats (Berman et al. 1995). The selected departure point was considered to be a dose where adverse liver effects could begin to be observed in two species after subchronic dosing. A composite uncertainty factor of 5,000 was obtained by multiplying factors of: 50 for intra-individual variability, $10^{1/2}$ for animal-to-human uncertainty, $10^{1/2}$ for using subchronic instead of lifetime studies, $10^{1/2}$ for using a point of departure where adverse effects have been observed, and, a modifying factor of $10^{1/2}$ to reflect background exposures to trichloroethylene and its metabolites. Dividing the 1 mg/kg-d point of departure by a composite uncertainty factor of 5,000 yielded an RfD of 0.2 μg/kg bw per day. This RfD has since been withdrawn by the U.S. EPA.

5.8.2 Inhalation Exposure – Non-Cancer Effects

Health Canada (2004) focuses on developing a Guideline for Canadian Drinking Water Quality for trichloroethylene, and does not consider inhalation studies in detail or calculate an inhalation tolerable concentration (TC). However, an inhalation TC for non-cancer effects was required in this document in order to properly evaluate appropriate soil quality guidelines protective of indoor vapour inhalation.

The inhalation TC used in this document was 40 $\mu g/m^3$, based on the RfC of the same value in U.S. EPA (2001), although the U.S. EPA RfC has since been withdrawn. The RfC calculated in U.S. EPA (2001) was based on a consideration of non-cancer effects on the central nervous system, kidney, liver, and endocrine system in inhalation studies. A point of departure of 38 mg/m^3 was identified as representing the lower end of the range of concentrations at which signs of central nervous system toxicity were observed in several occupational studies (Rasmussen et al. 1993; Ruitjen et al. 1991; Vandervort et al. 1973; Okawa and Bodner 1973). This level was supported by an LEC_{10} of 27 mg/m^3 for heart rate and electroencephalographic changes in rats (Arito et al. 1994). A composite uncertainty factor of 1,000 was obtained by multiplying factors of: 10 for intra-individual variability, 10 for starting from subchronic instead of lifetime studies, and 10 for starting from effect levels instead of NOAELs. Dividing the 38 mg/m^3 point of departure by a composite uncertainty factor of 1,000 yielded an RfC of 40 $\mu g/m^3$ (Table 9).

It is noted that Health Canada plans to review the inhalation toxicity of TCE in 2005/6 (Mark Richardson, *pers. comm.*). Once this review is complete, it may be desirable to revise the TC value noted above and the corresponding guideline values.

5.8.3 Oral Exposure - Cancer Effects

Health Canada (2004) reviewed the available trichloroethylene cancer studies and selected kidney tumours as the endpoint on which to base their cancer risk assessment. National toxicology program studies (NTP 1988; 1990) were used to calculate unit risks for tubular cell adenomas and adenocarcinomas of the kidneys in four strains of rats following oral exposure to trichloroethylene for 103 weeks. A study by Maltoni et al., (1986), in which rats were exposed to trichloroethylene by inhalation for 104 weeks, was used to calculate unit risks for renal tubular adenocarcinomas in rats. Using the pooled unit risks for kidney tumours in the male rats from the cancer potency assessment, an amortized unit risk was calculated by Health Canada (2004) and converted to a human equivalence value as 8.11 x 10^{-4} (mg/kg bw per day)$^{-1}$. In this report, a risk-specific dose is calculated for an excess cancer Risk Level of 10^{-6}, using the following equation:

$$RsD = \frac{Risk\ Level}{Unit\ Risk}$$

The risk-specific dose calculated was 0.00123 mg/kg bw per day (Table 9) and this value is used in this report to calculate guidelines protective of cancer endpoints in humans for soil and food ingestion exposure pathways.

The Health Canada (2004) unit risk values may be put in context by considering the slope factors calculated by U.S. EPA (2001). Note that unit risk and slope factor are equivalent terms. The U.S. EPA (2001) recommend a range for the slope factor for trichloroethylene of 2 x 10^{-2} to 4 x 10^{-1} (mg/kg bw per day)$^{-1}$. This range is calculated by considering nine estimates of slope factor based on studies in humans rats and mice. The two highest values (human studies) and the two lowest values (one rat and one mouse study) were rejected to yield the range noted above.

5.8.4 Inhalation Exposure - Cancer Effects

Health Canada (1996) has published an inhalation TC_{05} for trichloroethylene of 82 mg/m^3, based on their assessment of trichloroethylene as a Group II carcinogen (probably carcinogenic to humans). A TC_{05} is the concentration of a chemical expected to cause a 5% incidence of cancer. The TC05 may be extrapolated to a risk-specific concentration (RsC) evaluated at an excess cancer risk of 10^{-6} by dividing by 50,000 to give an RsC of 0.00164 mg/m^3 (Table 9). This is the value used in this report to calculate soil quality guidelines protective of indoor infiltration and inhalation for cancer endpoints.

It is noted that Health Canada plans to review the inhalation toxicity of TCE in 2005/6 (Mark Richardson, *pers. comm.*). Once this review is complete, it may be desirable to revise the RsC value noted above and the corresponding guideline values.

5.9 Toxicity of Environmental Degradation Products

As noted in Section 3.3, the major biodegradation products of TCE in groundwater are dichloroethylene, chloroethane, and vinyl chloride. The potential for TCE to generate vinyl chloride in the environment is of particular concern, since vinyl chloride is a potent carcinogen.

Health Canada (1996) has not evaluated the carcinogenicity of vinyl chloride, however, the U.S. EPA (2000) assessed vinyl chloride as Group A ("known human carcinogen") by both the inhalation and oral routes. Based on information in U.S. EPA (2000), the risk specific dose (RsD) (evaluated at an excess cancer risk of 1 in 10^6) can be calculated to be 7×10^{-7} mg/kg bw per day, and the risk specific concentration (RsC) (again evaluated at an excess cancer risk of 1 in 10^6) is 0.00023 mg/m^3. Based on this information, vinyl chloride is a more potent carcinogen than TCE.

Accordingly, whenever TCE is present in the environment, it is possible that vinyl chloride is present, and moreover that the risk to human health from vinyl chloride could exceed the risk from TCE. Thus it is imperative that vinyl chloride concentrations be assessed whenever TCE is found or suspected to be present.

6. DERIVATION OF ENVIRONMENTAL SOIL QUALITY GUIDELINES

Procedures for deriving environmental soil quality guidelines were developed to maintain important ecological functions that support activities associated with the identified land uses. The derivation of environmental soil quality guidelines for trichloroethylene is outlined in the following sections for four land uses: agricultural, residential/parkland, commercial, and industrial. These guidelines have been developed according to the protocol described by CCME (CCME 1996a; reprinted in 1999, and subsequently revised in 2003 with acceptance of the latest revisions anticipated in 2005). The information presented in this chapter includes the revisions made to the environmental soil quality guidelines for trichloroethylene that were last revised in 1999. According to the protocol, these environmental soil quality guidelines for trichloroethylene were considered along with the human health guidelines in making final recommendations for Canadian Soil Quality Guidelines for the protection of environmental and human health.

The environmental soil quality guidelines for trichloroethylene are derived using the available toxicological data to determine the threshold level of effects for key ecological receptors. Exposure from direct soil contact is the primary derivation procedure for environmental guidelines for residential/parkland, commercial and industrial land uses. Another procedure, exposure from contaminated soil and food ingestion, may be considered for certain land uses if there is adequate data. For agricultural land use, if both derivation procedures are used, the lowest-value result is considered the environmental soil quality guideline.

6.1 Direct Soil Contact Guideline (SQG_{SC}) for the Protection of Soil Invertebrate and the Plant Community

Three options are available for deriving guidelines for agricultural and residential/parkland land use in consideration of direct soil contact and the maintenance of soil ecological functioning. Depending on the availability and quality of data on toxicity of substances to soil invertebrates and plants, a guideline can be calculated using –

1) Weight-of-Evidence (WOE) approach to calculate a “Threshold Effects Concentration” (TEC), provided minimum data requirements are met; or

2) Estimation of a TEC (for agricultural and residential/parkland land uses) or an ECL (for commercial and industrial land uses) from lowest observable adverse effect concentrations (LOAECs) in soil, provided appropriate LOAEC endpoints are available; or,

3) Estimation of a TEC (for agricultural and residential/parkland land uses) or an ECL (for commercial and industrial land uses) from median effective concentrations (EC_{50}s) or medial lethal concentrations (LC_{50}s).

For the WOE approach, available effects data, ideally standardized to a 25% effect size, are gathered or interpolated from a critical examination of the available studies (EC_{25}, LC_{25}). Other types of effects data, EC_X or LC_X, may be included depending on the value of 'X' and on professional judgment about the value of the data in describing toxicological thresholds for various taxa. At least ten data points from not less than three studies is required for the WOE approach, with data for at least two species of both soil invertebrates and plants. The WOE approach as specified in the CCME (2003) protocol revision also suggests that the plant and soil invertebrate toxicity data should be initially treated separately, since there is often a difference in the sensitivity of the two major taxa.

No new data has been produced since 1999 on the toxicity of TCE to plants or soil invertebrates, based on exposures in soil (Tables 7 and 8). The data are insufficient to calculate a SQG_{SC} using a WOE approach.

Environment Canada (1996) developed NOEC and LOEC estimates for lettuce, (*Lactuca sativa)*, and radishes, (*Raphanus sativus)* seedling emergence (Table 7). The estimated LOECs for lettuce and radish were 48 mg/kg and 16 mg/kg, respectively, when expressed on the basis of measured as opposed to spiked (nominal) TCE concentrations in the soil units. Environment Canada (1996) also developed a LOEC for mortality using the earthworm species *Eisenia foetida*, of 159 mg/kg TCE based on measured exposure concentrations. Mortality is generally a far less sensitive endpoint in earthworms such as *E. foetida* or *Lumbricus terrestris* than reproductive output in longer term exposures.

According to CCME (2003), a minimum of three studies reporting LOEC endpoints must be considered to use the 2nd of the three procedures described above. Requirements also include at least one terrestrial plant and one soil invertebrate study.

Ideally, the available data should meet a number of other quality tests. In particular, the soil pH, texture, and other potentially important characteristics should have been described in the presentation of the results. The EC/LCx, NOEC and LOEC endpoints should have been derived using statistical techniques that stand up to scientific peer review, and other aspects of experimental design should lead to confidence in the validity of the test results for establishing ecologically protective thresholds, as well as the reproducibility of the results.

Unfortunately, very few toxicity studies meet the minimum reporting requirements. In the specific case of TCE, while some of the quality requirements for the three test species have not been met, the most important requirements have. A standardized test soil was used to control for the influences of different soil types, and – perhaps more importantly – the dose-response relationships were examined on the basis of measured versus nominal concentration estimates. This is extremely important for a substance such as TCE that is rapidly lost from test units even before introduction of the test organisms.

Because the available data marginally meet these requirements, a provisional SQG_{SC} was calculated as follows:

The TEC is calculated using the following equation:

$$\text{TEC} = \text{lowest LOEC/UF}$$

where TEC = threshold effects concentration (mg/kg soil)
LOEC = lowest observed effect concentration (mg/kg soil)
UF = uncertainty factor (if needed)

The ECL is calculated according to the equation:

$$\text{ECL} = (\text{LOEC}_1 \text{ x LOEC}_2 \text{ x ... x LOEC}_n)^{1/n}$$

where ECL = effects concentration low (mg/kg)
LOEC = lowest observed effect concentration (mg/kg)
n = the number of available LOECs

For TCE:

$$\text{TEC} = \text{lowest LOEC/UF}$$

$$= 16 \text{ mg/kg} / 5$$

$$= 3.2 \text{ mg/kg} \approx 3 \text{ mg/kg}$$

An uncertainty factor (UF) of 5 was chosen within an optional range of 1-5. This is because data are available for only 3 species and in one soil type. In addition, the LOECs were derived from short term exposures, and may not reflect more sensitive effects types.

$$\text{ECL} = (\text{LOEC}_1 \text{ x LOEC}_2 \text{ x ... x LOEC}_n)^{1/n}$$

$$= (16 \text{ mg/kg x } 48 \text{ mg/kg x } 159 \text{ mg/kg})^{1/3}$$

$$= 49.6 \text{ mg/kg} \approx 50 \text{ mg/kg}$$

Based on the above, the SQG_{SC} for agricultural and residential/parkland sites is set at **3 mg/kg**, based on the TEC calculation shown above. The SQG_{SC} for commercial and industrial sites is set at **50 mg/kg**, based on the ECL calculation shown above.

6.2 Derivation of Soil Quality Guidelines for Soil and Food Ingestion by Livestock and Wildlife (SQG_I)

The procedure for deriving soil quality guidelines for ingestion of soil and food (SQG_I) by grazing livestock and wildlife is described in CCME (2003). This exposure scenario is intended to apply in agricultural settings, specifically for those substances that have the potential to affect herbivorous livestock that may ingest either contaminated surface soil or plants that have bioaccumulated the contaminant. Because this procedure is limited to a herbivore food chain, chemicals that bioaccumulate in the tissues of plants and that can be transferred to herbivores are of primary importance.

In order to confidently assign a SQG_I, information is required on (i) soil – plant partitioning, (ii) average daily or annual soil ingestion rates by a livestock or wildlife species of interest, (iii) plant (or forage) daily or annual ingestion rates, and (iv) a threshold-effects dose (or similar form of toxicity reference value; TRV) for the toxicant in the livestock or wildlife species of interest. According to CCME (2003):

> “In view of the data requirements and model parameters used to estimate generic guidelines for soil and food ingestion, it is only possible to derive ingestion guidelines where data are sufficient to keep model parameter uncertainty at a minimum and also reduce the need for large inter-species extrapolations. Therefore, until more data are available for other receptors, it is the opinion of SQGTG that guidelines for soil and food ingestion should only be derived for grazing herbivores on agricultural lands.”

For TCE, limited data (Schroll et al. 1994) suggest that plants can bioaccumulate TCE (see Section 3.6); however, the principle route of uptake appears to be via volatilization from soils followed by foliar uptake, and subsequently by translocation. In light of this, there is a very high degree of uncertainty about whether TCE is taken up in the root zone by plants, and regarding soil-plant bioaccumulation factors (BAFs). In addition to this, virtually all laboratory or controlled mammalian toxicity studies available to the present time have been conducted on mice or rats (ATSDR 1997; HC 2003). The relevance of such studies to ruminant livestock and wildlife species is uncertain.

In light of (i) the expected limited tendency of TCE to persist in surface soils or be taken up into plants, (ii) the degree of uncertainty about soil-plant BAFs, and (iii) the minimum data requirements of one avian study and two mammalian studies, only one of which may be a laboratory rodent study, not being met, a SQG_I was not calculated for TCE. In addition to this, TCE release sites are expected to occur primarily in commercial and industrial settings. While there is some potential for soil contamination in agricultural settings, the primary mechanism would likely be off-site migration from an adjacent industrial site, or landfill. In such cases, the off-site migration of TCE would best be examined by a more focused site-specific risk assessment, followed by the appropriate risk management actions.

6.3 Derivation of Soil Quality Guidelines for the Protection of Freshwater Life (SQG_{FL})

TCE present in soil can migrate to groundwater. If there are surface water bodies (streams, rivers, lakes, etc.) nearby, then aquatic life in these surface water bodies may be affected by the contamination, particularly if there is a permeable, unconfined aquifer connecting the TCE contamination with the surface water body.

SQG_{FL} is calculated for both coarse and fine soils, and is the same for all land uses. The overarching concern is off-site migration to an adjacent water body that could support aquatic life in the absence of the contamination. The on-site land-use designation, therefore, is less important than the actual site hydrogeology, potential for groundwater fluxes to surface water bodies, and the distance of any ecologically important surface water bodies from the TCE contaminated soil. These factors are very important for influencing the level of soil contamination beyond which aquatic life might be negatively impacted; however, it is necessary to make several simplifying and near 'worst-case' assumptions about the site properties in order to derive a SQG_{FL} that is generically applicable (sufficiently protective) when applied at the major portion of Canadian sites.

The protection of groundwater model presented in CCME (2003) is the same as that used by CCME (2000), which, in turn is based on BC MELP (1996) as adapted from Domenico (1987) and U.S. EPA (1996). The model considers four processes:

1. partitioning from soil to leachate;
2. transport of leachate from base of contamination to water table;
3. mixing of leachate and groundwater; and,
4. groundwater transport down-gradient to a discharge point.

For each of these four processes, a dilution factor was calculated (DF1 through DF4, respectively). The reader is referred to CCME (2003) and AENV (2001) for greater detail.

DF1 has units of (mg/kg)/(mg/L) or L/kg. The other three dilution factors are dimensionless [units of (mg/L)/(mg/L)]. The overall dilution factor is used to calculate the soil concentration that is protective of freshwater aquatic life using the following equations:

$$SQG_{FL} = CWQG_{FL} \text{ x } DF$$

$$DF = DF1 \text{ x } DF2 \text{ x } DF3 \text{ x } DF4$$

where:

SQG_{FL} = soil quality guideline protective of groundwater for aquatic life (mg/kg);

$CWQG_{FL}$ = Canadian water quality guideline for the protection of freshwater aquatic life (mg/L);

DF = **overall dilution factor** (L/kg);

DF1 = **dilution factor 1** (L/kg): ratio of the concentration of a contaminant in soil to the concentration in leachate that is in contact with the soil. This “dilution factor” represents the three phase partitioning between contaminant sorbed to soil, contaminant dissolved in pore water (*i.e.*, as leachate) and contaminant present as soil vapour.

DF2 = **dilution factor 2** (dimensionless): ratio of the concentration of a contaminant in leachate that is in contact with the soil, to the concentration in pore water just above the groundwater table. This dilution factor reflects a decrease in concentration as dissolved contaminant moves downwards from the base of contamination through the unsaturated zone to the water table. The decrease in concentration is due to two processes: i) dispersion in the unsaturated zone, and ii) biodegradation in the unsaturated zone. For Tier 1 calculations, the contaminant is assumed to extend to the water table, and so there is no decrease in concentration due to these processes and DF2 = 1.0.

DF3 = **dilution factor 3** (dimensionless): ratio of the concentration of a chemical in pore water just above the groundwater table, to the concentration in groundwater beneath the source. This dilution factor reflects a decrease in concentration as leachate mixes with uncontaminated groundwater. DF3 is a function of groundwater velocity, infiltration rate, source length, and mixing zone thickness. The mixing zone thickness was calculated as being due to two processes: i) mixing due to dispersion, and ii) mixing due to infiltration rate.

DF4 = **dilution factor 4**, (dimensionless): dilution factor 4 (DF4) accounts for the processes of dispersion and biodegradation as groundwater travels down-gradient from beneath the source of contamination, and is the ratio of the concentration of a chemical in groundwater beneath the source, to the concentration in groundwater at a distance (10 m for Tier 1) down-gradient of the source. The equation used to calculate this dilution factor is based on an analytical groundwater transport model (Domenico 1987).

Assumptions implicit in the model include the following:

- the soil is physically and chemically homogeneous;
- moisture content is uniform throughout the unsaturated zone;
- infiltration rate is uniform throughout the unsaturated zone;
- decay of the contaminant source is not considered (*i.e.*, infinite source mass);
- flow in the unsaturated zone is assumed to be one dimensional and downward only (vertical recharge) with dispersion, sorption-desorption, and biological degradation;
- contaminant is not present as a free phase product;
- maximum possible concentration in the leachate is equivalent to the solubility limit of the chemical in water under the defined site conditions;
- groundwater aquifer is unconfined;
- groundwater flow is uniform and steady;
- co-solubility and oxidation/reduction effects are not considered;
- attenuation of the contaminant in the saturated zone is assumed to be one dimensional with respect to sorption-desorption, dispersion, and biological degradation;
- dispersion in groundwater is assumed to occur in the longitudinal and transverse directions only and diffusion is not considered;
- mixing of the leachate with the groundwater is assumed to occur through mixing of leachate and groundwater mass fluxes; and
- dilution of the plume by groundwater recharge down-gradient of the source is not included.

Parameter Values

Default (CCME 2003) values are used for all generic site parameters unless otherwise noted.

Canadian Water Quality Guidelines for the Protection of Freshwater Life for TCE have previously been endorsed by the Water Quality Task Group of CCME. The Guideline value for TCE is 21 μg/L, or 0.021 mg/L (Table 5).

Following the latest guidance from the Soil Quality Guidelines Task Group of CCME, and in anticipation of changes in the upcoming (2005) version of the SQG protocol, the value of hydraulic conductivity for groundwater-bearing zones associated with fine and coarse soil are 32 m/year and 320 m/year for both human and ecological groundwater pathways. Note that this represents a departure from values used in the PHC CWS, where the hydraulic conductivity for fine soils was 0.32 m/year for the ecological pathway (and 32 m/year for potable groundwater).

The CCME (2003) protocols allow for the inclusion of contaminant biodegradation half-life estimates, and degradation in the unsaturated zone and saturated zone is considered. TCE loss rates from soil or groundwater are discussed in Sections 3.5 and 3.3. While volatilization is an important loss process for TCE in some environments, it cannot be assumed to occur at a significant rate at a large subset of sites where the depth to groundwater and soil permeability above it result in only slow losses via the upward advection and diffusion of soil vapour. As noted in Section 3.3, biodegradation is an important process for TCE, but biodegradation half-lives in the saturated zone have been estimated to be in the range of a few months to years. For the purpose of deriving SQG_{FL}, a saturated zone biodegradation half-life of 800 d (2.2 y) was assumed (Table 2).

Some jurisdictions within Canada and elsewhere have adopted policy decisions that allow for a minimum amount of dilution of groundwater that enters a surface water body, and then potentially interacts with aquatic life. Given differences in policies between jurisdictions in Canada, however, no allowance for dilution of groundwater near the groundwater outflow face in the receiving environment has been made herein. The resulting soil quality guidelines, therefore, are lower than is supported in jurisdictions that have adopted a groundwater-surface water 10-fold or other dilution allowance. Some jurisdictions have explicitly rejected such a policy, since benthic invertebrates and other sediment, lakebed, or streambed associated organisms might experience perfusion of undiluted groundwater at the outflow face.

6.3.1 Dilution Factor 1

Dilution factor 1 (DF1) is the ratio of the concentration of a contaminant in soil to the concentration in leachate that is in contact with the soil. This “dilution factor” represents the three phase partitioning between contaminant sorbed to soil, contaminant dissolved in pore water (*i.e.*, as leachate), and contaminant present as soil vapour. DF1 is calculated using the following equation:

$$DF1 = K_{oc} \times f_{oc} + \frac{(\theta_w + H' \times \theta_a)}{\rho_b}$$

where:

DF1 = dilution factor 1 (L/kg);
K_{oc} = organic carbon-water partition coefficient (86 L/kg; Table 2);
f_{oc} = fraction organic carbon (0.005 g/g; CCME 2003);

θ_w = water filled porosity (0.119 (coarse); 0.168 (fine) CCME 2003);
H' = dimensionless Henry's Law constant (0.36; Table 2);
θ_a = air filled porosity (0.241 (coarse); 0.302 (fine) CCME 2003); and,
ρ_b = dry soil bulk density (1.7 g/cm^3 (coarse); 1.4 g/cm^3 (fine); CCME 2003).

Substituting these values in the above equations yields values for DF1 of 0.551 L/kg and 0.628 L/kg for coarse and fine soils, respectively.

6.3.2 Dilution Factor 2

Dilution factor 2 (DF2) is the ratio of the concentration of a contaminant in leachate that is in contact with the soil, to the concentration in pore water just above the groundwater table. DF2 takes the value 1.00 (*i.e.,* no dilution) for generic guidelines because it is assumed at Tier 1 that the contaminated soil extends down to the water table.

6.3.3 Dilution Factor 3

Dilution factor 3 (DF3) is the ratio of the concentration of a chemical in pore water just above the groundwater table, to the concentration in groundwater beneath the source. This dilution factor reflects a decrease in concentration as leachate mixes with uncontaminated groundwater. DF3 is a function of groundwater velocity, infiltration rate, source length, and mixing zone thickness. The mixing zone thickness is calculated as being due to two processes: i) mixing due to dispersion, and ii) mixing due to infiltration rate. The equations used are as follows:

$$DF3 = 1 + \frac{Z_d \times V}{I \times X}$$

$$Z_d = r + s$$

$$r = 0.01 \times X$$

$$s = d_a \left\{ 1 - exp\left(\frac{-2.178 \times X \times I}{V \times d_a} \right) \right\}$$

$$V = K \times i$$

where:

DF3 = dilution factor 3 (dimensionless);

Z_d = average thickness of mixing zone (0.467 m (coarse); 2.20 m (fine) calculated above);
V = Darcy velocity in groundwater (16 m/year (coarse); 1.6 m/year (fine); calculated above);
I = infiltration rate (0.28 m/year (coarse); 0.20 m/year (fine); CCME 2003);
X = length of contaminated soil (10 m; CCME 2003);
r = mixing depth due to dispersion (0.10 m; calculated above);
s = mixing depth due to infiltration rate (0.367 m (coarse); 2.10 m (fine); calculated above);
d_a = unconfined aquifer thickness (5 m; CCME 2003);
K = aquifer hydraulic conductivity (320 m/year (coarse); 32 m/year (fine); CCME 2003); and,
i = lateral hydraulic gradient in aquifer (0.05; CCME 2003).

Substituting these values in the above equations yields values for DF3 of 3.67 and 2.76 for coarse and fine soil, respectively.

6.3.4 Dilution Factor 4

Dilution factor 4 (DF4) accounts for the processes of dispersion and biodegradation as groundwater travels downgradient from beneath the source of contamination, and is the ratio of the concentration of a chemical in groundwater beneath the source, to the concentration in groundwater at a distance (10 m for Tier 1) downgradient of the source. DF4 was calculated using the following equations:

$$DF4 = \frac{4}{exp(A) \times erfc(B) \times [erf(C) - erf(D)]}$$

$$A = \frac{x}{2D_x}\left\{1 - \left(1 + \frac{4L_s D_x}{v}\right)^{1/2}\right\}$$

$$B = \frac{x - vt\left(1 + \frac{4L_s D_x}{v}\right)^{1/2}}{2(D_x vt)^{1/2}}$$

$$C = \frac{y + Y/2}{2\left(D_y x\right)^{1/2}}$$

$$D = \frac{y - Y/2}{2\left(D_y x\right)^{1/2}}$$

$$L_s = \frac{0.691}{t_{1/2s}} \times exp\left(-0.07d\right)$$

$$v = \frac{V}{\theta_t R_s}$$

$$R_s = 1 + \frac{\rho_b K_{oc} f_{oc}}{\theta_t}$$

$$D_x = 0.1x$$

$$D_y = 0.01x$$

where:

DF4 = dilution factor 4 (dimensionless);
erf = the error function;
erfc = the complimentary error function;
A = dimensionless group A (-0.17 (coarse); -1.49 (fine); calculated above);
B = dimensionless group B (-19.7 (coarse); - 7.52 (fine); calculated above);
C = dimensionless group C (7.50; calculated above);
D = dimensionless group D (-7.50; calculated above);
x = distance to source (10 m; CCME 2003);
D_x = dispersivity in the direction of groundwater flow (1.0 m; calculated);
L_s = decay constant (0.256/year; calculated above);
v = velocity of the contaminant (14.7 m/year (coarse); 1.49 m/year (fine); calculated above);
t = time since contaminant release (100 years; CCME 2003);
y = distance to receptor perpendicular to groundwater flow (0 m; CCME 2003);
Y = source width (30 m; CCME 2003);
D_y = dispersivity perpendicular to the direction of groundwater flow (0.10 m; calculated above);
$t_{1/2s}$ = decay half-life of TCE in aquifer (2.2 years; Table 6);
d = water table depth (3 m; CCME 2003);

V = Darcy velocity in groundwater (16 m/year (coarse); 1.6 m/year (fine); calculated);
θ_t = total soil porosity (0.36 (coarse); 0.47 (fine); CCME 2003);
R_s = retardation factor in saturated zone (3.03 (coarse); 2.28 (fine); calculated above);
ρ_b = dry soil bulk density (1.7 g/cm^3 (coarse); 1.4 g/cm^3 (fine); CCME 2003);
K_{oc} = organic carbon partition coefficient (86 mL/g; Table 2); and,
f_{oc} = fraction organic carbon (0.005 g/g; see CCME 2003).

Substituting these values in the above equations yields values for DF4 of 1.19 and 4.44 for coarse and fine soil, respectively.

Based on the above, the modelled soil concentration for TCE based on freshwater life protection is –

1. Coarse soils: SQG_{FL} = 0.05 mg/kg.

2. Fine soils: SQG_{FL} = 0.16 mg/kg.

6.4 Microbial (Nutrient and Energy Cycling) Check

Soil processes such as decomposition, respiration and organic nutrient cycles are important components of the ecological function of soil. These processes may be affected by the presence of TCE (Section 4.1). According to CCME (2003):

> "The microbial ecology relevant to the cycling of organic nutrients indicates that contaminant data from nitrogen fixation, nitrification, nitrogen mineralization, decomposition, and respiration studies are all potentially acceptable for use in a checking role against guidelines derived from single species bioassay. … …Of these, N-fixation and nitrification data are preferred, but carbon cycling and nitrogen mineralization measures may be used when the former are unavailable or insufficient for guideline derivation."

The available studies (Section 4.1) tended to examine microbial community responses when exposed to a nominal concentration of TCE in a slurry suspension (often with toxicity endpoints expressed in mg TCE/L) or as added soil concentrations but without actually measuring the realized exposure concentrations. In light of this, the studies reviewed were not deemed to be sufficient in order to calculate a Nutrient and Energy Cycling Check value.

6.5 Off-Site Migration (SQG_{OM-E})

The soil quality guideline for off-site migration (SQG_{OM-E}) is not required for volatile chemicals (CCME 2003) and accordingly is not calculated for TCE.

7. DERIVATION OF HUMAN HEALTH SOIL QUALITY GUIDELINES

Soil quality guidelines protective of human health were calculated based on the CCME (2003) draft protocol. Trichloroethylene can cause both cancer and non-cancer effects. Guidelines are calculated separately for cancer and non-cancer effects; the lower of the guidelines calculated is retained for each exposure pathway. All equations and parameter values are from CCME (2003) unless otherwise noted. Guideline values are summarized in Table 10. Details of the calculations are provided in the following sections. Note that the results of intermediate calculations are presented below to 3 significant figures, but that the guideline values (presented to 2 significant figures) are always calculated using the full available precision. Accordingly, substituting the values provided in the equations given may occasionally result in slightly different results from the guideline values presented.

7.1 Parameter Values

Most parameter values used in calculating the soil quality guidelines are standard values from CCME (2003), or chemical-specific values discussed in Section 2 and listed in Table 2. However, some parameters may require a little further explanation. The values for these parameters are discussed below.

Background Soil Concentration

While trichloroethylene has certainly been detected in soil in the Canadian environment, its presence in soil is likely limited to local areas where this chemical has been used or released. It is anticipated that the majority of soil with which Canadians come into contact will not have measurable concentrations of trichloroethylene, and accordingly, the background soil concentration (BSC) for trichloroethylene is set at 0 mg/kg.

Background Indoor Air Concentration

The Government of Canada (1993) considers 0.0014 mg/m^3 to be representative of typical concentrations of trichloroethylene in indoor air in Canada, based on the mean value from an unpublished study of 750 homes from 10 provinces across Canada (Section 2.5.3). This value was used in the calculation of the SQG_I.

Estimated Daily Intake

The estimated daily intakes of trichloroethylene for Canadian toddlers and adults used to calculate the soil quality guidelines in this report are 0.53 and 0.41 μg/kg bw per day, for toddlers and adults, respectively. These values are based on the upper end of the ranges provided in Government of Canada (1993) (Section 2.5.10 and Table 4).

Soil Allocation Factor

The soil allocation factor (SF) is the relative proportion which it is allowable for soil to constitute in the total exposure from various environmental pathways (air, soil, food, water, consumer products). By default, SF is set to 0.2, based on the assumption that a given chemical may be present in all five of the above-noted environmental media. CCME (2003) allows SF to be increased in cases where the chemical in question is unlikely to be in all the five media. Based on the information in Section 2.5, trichloroethylene can be present in all the five media (air, soil, food, water, consumer products), and accordingly SF is set at a value of 0.2.

Absorption Factors

The equations in Section 7.2 require values for the absorption factors for gut, lung, and skin (AF_G, AF_L, and AF_S, respectively). However, a value other than 1 is only required if the corresponding exposure limit was based on absorbed, rather than administered dose). The oral and inhalation exposure limits used in this document are based on administered dose (Health Canada 2004, and U.S. EPA 2001), and accordingly the absorption factors for these routes are set at 1. For dermal absorption, Poet et al. (2000) found that a maximum of 4.25% was absorbed from water-borne TCE under a patch after 2 hours exposure (a patch reduces evaporation). For soil, the maximum absorbed was 0.8%. Health Canada (2003) suggest a dermal absorption factor for TCE of 10%. In this document, the Health Canada (2003) guidance was followed, and the dermal absorption factor, AF_S, was set at 0.1.

Soil Dermal Contact Rate

Soil dermal contact rate (SR) was introduced in CCME (2003) and is the mass of contaminated soil which is assumed to contact the skin each day. This parameter is calculated as follows:

$$SR = \{(SA_H \times DL_H) + (SA_O \times DL_O)\} \times EF$$

Where:

SR = soil dermal contact rate (kg/day);
SA_H = exposed surface area of hands (m^2);
DL_H = dermal loading of soil to hands (kg/m^2 per event);
SA_O = area of exposed body surfaces other than hands (m^2);
DL_O = dermal loading of soil to other surfaces (kg/m^2 per event); and,
EF = exposure frequency (events/day).

Substituting standard (CCME 2003) values for the above parameters in this equation gives the following values for SR: 6.88×10^{-5} kg/day for toddlers, and 1.14×10^{-4} kg/day for adults.

Effective Diffusion Coefficient in Cracks

The effective diffusion coefficient in cracks (D_{crack}) provides an indication of the ease with which contaminants can diffuse through basement cracks. No guidance is provided in CCME

(2003) as to selecting values for this parameter. In CCME (2000), D_{crack} was set based on the assumption that floor cracks were filled with coarse soil. This same assumption is made here, and accordingly, D_{crack} in both coarse and fine soil is set equal to the values of D_T^{eff} calculated for coarse soil (5.29×10^{-3} cm^2/s; see Section 7.4.3)

7.2 Direct Soil Exposure Pathways

The direct soil exposure pathways include ingestion, dermal contact, and particulate inhalation. However, particulate inhalation is not expected to be significant for a volatile chemical such as trichloroethylene (CCME 2003), and is not calculated (contact rate set to zero in the equations below). Calculations of the guidelines for cancer and non-cancer effects are provided in Sections 7.2.1 and 7.2.2, respectively. Guidelines for these two sets of effects are compared in Section 7.2.3, and the lower value selected. The rationale for the values used for certain parameters is provided in Section 7.1. Note that the CCME (2003) equations implicitly assume that the oral TDI can be used to assess the dermal contact pathway.

7.2.1 Cancer Effects

For cancer effects, the CCME (2003) protocol considers an adult exposed over their entire lifetime to develop soil quality guidelines for all land use settings. The human health soil guideline for cancer effects was calculated using:

$$PSQG_{HH} = \frac{RsD \times BW}{\left[\left(AF_G \times SIR\right) + \left(AF_L \times IR_S\right) + \left(AF_S \times SR\right)\right] \times ET} + \left[BSC\right]$$

Where:

$PSQG_{HH}$ = preliminary human health-based soil quality guideline (mg/kg);
RsD = risk-specific dose (1.23×10^{-3} mg/kg bw per day; Table 9);
BW = adult body weight (70 kg; CCME 2003);
AF_G = absorption factor for gut (1; Section 7.1);
AF_D = absorption factor for lung (1; Section 7.1);
AF_S = absorption factor for skin (0.1; Section 7.1);
SIR = soil ingestion rate for adult (0.00002 kg/day; CCME 2003);
IR_S = inhalation of particulate matter re-suspended from soil (0 kg/day; see above);
SR = soil dermal contact rate for adults (1.14×10^{-4} kg/day; Section 7.1);
ET = exposure term (1 for all land uses for carcinogenic effects CCME 2003); and,
BSC = background soil concentration (0 mg/kg; Section 7.1).

Substituting these values in the above equation and rounding to 2 significant figures yields 2,700 mg/kg, which is the preliminary human health-based soil quality for cancer effects for all land uses.

7.2.2 Non-Cancer Effects

For non-cancer effects, the CCME (2003) protocol considers a toddler (the most sensitive life-stage for this exposure pathway) for all land uses except industrial, in which an adult is considered. The human health soil guideline for non-cancer effects was calculated using:

$$PSQG_{HH} = \frac{(TDI - EDI)\times SAF \times BW}{[(AF_G \times SIR) + (AF_L \times IR_S) + (AF_S \times SR)] \times ET} + [BSC]$$

Where:

$PSQG_{HH}$ = preliminary human health-based soil quality guideline (mg/kg);
TDI = tolerable daily intake (1.46×10^{-3} mg/kg bw per day; Table 9);
EDI = estimated daily intake (0.53×10^{-3} and 0.41×10^{-3} mg/kg bw per day for toddlers and adults, respectively; Section 7.1);
SAF = soil allocation factor (0.2; Section 7.1);
BW = body weight (13 kg: toddlers, and 70 kg: adults ; CCME 2003);
AF_G = absorption factor for gut (1; Section 7.1);
AF_D = absorption factor for lung (1; Section 7.1);
AF_S = absorption factor for skin (0.1; Section 7.1);
SIR = soil ingestion rate (0.00008 kg/day: toddlers; and 0.00002 kg/day: adults; CCME 2003);
IR_S = inhalation of particulate matter re-suspended from soil (0 kg/day; see above);
SR = soil dermal contact rate (6.88×10^{-5} kg/day: toddlers; and, 1.14×10^{-4} kg/day: adults; Section 7.1);
ET = exposure term (1.0: agricultural and residential/parkland uses and 0.2747: commercial and industrial land uses; CCME 2003); and,
BSC = background soil concentration (0 mg/kg; Section 7.1).

Substituting these values in the above equation and rounding to 2 significant figures yields 28 mg/kg (agricultural and residential), 100 mg/kg (commercial), and 1,700 mg/kg (industrial), which are the preliminary human health-based soil quality for non-cancer effects.

7.2.3 Comparison of Guidelines for Cancer and Non-Cancer Effects

The soil quality guidelines calculated in Section 7.1.2 for non-cancer effects are lower than the guidelines for cancer effects calculated in Section 7.1.1 for all land uses. Accordingly the non-cancer guidelines: 28 mg/kg (agricultural and residential), 100 mg/kg (commercial), and 1,700 mg/kg (industrial) are adopted for human exposure to direct soil contact for trichloroethylene Table 10).

7.3 Ingestion of Groundwater as Drinking Water Pathway

The soil quality guidelines for the ingestion of groundwater as drinking water pathway (SQG_{DW}), were calculated using the draft CCME (2003) protocol. Note that the calculation is significantly different from the calculation in the former (CCME 1996a) protocol and also differs from the procedure used to calculate this guideline in the Petroleum Hydrocarbon Canada-Wide Standard (CCME 2000). The CCME (2003) protocol calculates SQG_{DW} using the same model as is used in both CCME (2000) and CCME (2003) to calculate the soil quality guideline for the protection of freshwater aquatic life (SQG_{FL}), with the exception that the drinking water extraction is assumed to occur adjacent to the contaminated site, rather than 10 m downgradient from it.

SQG_{DW} is calculated for both coarse and fine soils, and is the same for all land uses. There was no need to evaluate cancer and non-cancer effects of trichloroethylene separately for this exposure pathway because SQGDW is based on the (draft) Canadian drinking water guideline for trichloroethylene (0.005 mg/L) and Health Canada (2004) has already considered both cancer and non-cancer effects in deriving this guideline.

The model used to calculate SQG_{FL} considers four processes:

1. partitioning from soil to leachate;
2. transport of leachate from base of contamination to water table;
3. mixing of leachate and groundwater; and,
4. groundwater transport downgradient to the receptor.

Each of these processes results in the sequential dilution of the contaminant from the source to the receptor, and the degree of dilution is characterized by 4 dilution factors, DF1 to DF4, respectively. In the case of generic guidelines for SQG_{DW}, it is assumed that the base of the contaminated zone is at the water table, and hence there is no dilution in process #2, and DF2 takes the value 1. In addition, no offset is assumed between source and receptor, and DF4 is also 1. Even though DF2 and DF4 do not affect the overall calculation for this pathway, they are retained to maintain consistency with the terminology used in the calculation of SQG_{FL}. The following equations are used to calculate SQG_{DW}; guideline calculations for both coarse and fine soil are provided below.

$$SQG_{DW} = WQG_{DW} \times DF$$

$$DF = DF1 \times DF2 \times DF3 \times DF4$$

where:

SQG_{DW} = soil quality guideline protective of potable groundwater (mg/kg);
WQG_{DW} = drinking water quality guideline (0.005 mg/L; Health Canada 2004);
DF = overall dilution factor (2.02 L/kg (coarse); 1.73 L/kg (fine));
DF1 = dilution factor 1, (0.551 L/kg (coarse); 0.628 L/kg (fine));
DF2 = dilution factor 2, (1.00 (coarse and fine));
DF3 = dilution factor 3, (3.67 (coarse); 2.76 (fine)); and,
DF4 = dilution factor 4, (1.00 (coarse and fine)).

Substituting these values in the above equations and rounding yields 0.01 mg/kg for both coarse and fine soils. This value is the soil quality guideline for protection of potable groundwater (Table 10). The calculations for the individual dilution factors are reproduced below.

7.3.1 Dilution Factor 1

Dilution factor 1 (DF1) is the ratio of the concentration of a contaminant in soil to the concentration in leachate that is in contact with the soil. This "dilution factor" represents the three phase partitioning between contaminant sorbed to soil, contaminant dissolved in pore water (*i.e.*, as leachate), and contaminant present as soil vapour. DF1 is calculated using the following equation:

$$DF1 = K_{oc} \times f_{oc} + \frac{(\theta_w + H' \times \theta_a)}{\rho_b}$$

where:

DF1 = dilution factor 1 (L/kg);
K_{oc} = organic carbon-water partition coefficient (86 L/kg; Table 2);
f_{oc} = fraction organic carbon (0.005 g/g; CCME 2003);
θ_w = water filled porosity (0.119 (coarse); 0.168 (fine) CCME 2003);
H' = dimensionless Henry's Law constant (0.36; Table 2);
θ_a = air filled porosity (0.241 (coarse); 0.302 (fine) CCME 2003); and,
ρ_b = dry soil bulk density (1.7 g/cm^3 (coarse); 1.4 g/cm^3 (fine); CCME 2003).

Substituting these values in the above equations yields values for DF1 of 0.551 L/kg and 0.628 L/kg for coarse and fine soils, respectively.

7.3.2 Dilution Factor 2

Dilution factor 2 (DF2) is the ratio of the concentration of a contaminant in leachate that is in contact with the soil, to the concentration in pore water just above the groundwater table. DF2 takes the value 1.00 (*i.e.,* no dilution) for generic guidelines because it is assumed at Tier 1 that the contaminated soil extends down to the water table.

7.3.3 Dilution Factor 3

Dilution factor 3 (DF3) is the ratio of the concentration of a chemical in pore water just above the groundwater table, to the concentration in groundwater beneath the source. This dilution factor reflects a decrease in concentration as leachate mixes with uncontaminated groundwater. DF3 is a function of groundwater velocity, infiltration rate, source length, and mixing zone thickness. The mixing zone thickness is calculated as being due to two processes: i) mixing due to dispersion, and ii) mixing due to infiltration rate. The equations used are as follows:

$$DF3 = 1 + \frac{Z_d \times V}{I \times X}$$

$$Z_d = r + s$$

$$r = 0.01 \times X$$

$$s = d_a \left\{ 1 - exp\left(\frac{-2.178 \times X \times I}{V \times d_a} \right) \right\}$$

$$V = K \times i$$

where:

DF3 = dilution factor 3 (dimensionless);
Z_d = average thickness of mixing zone (0.467 m (coarse); 2.20 m (fine) calculated above);
V = Darcy velocity in groundwater (16 m/year (coarse); 1.6 m/year (fine); calculated above);
I = infiltration rate (0.28 m/year (coarse); 0.20 m/year (fine); CCME 2003);
X = length of contaminated soil (10 m; CCME 2003);
r = mixing depth due to dispersion (0.10 m; calculated above);
s = mixing depth due to infiltration rate (0.367 m (coarse); 2.10 m (fine); calculated above);

d_a = unconfined aquifer thickness (5 m; CCME 2003);
K = aquifer hydraulic conductivity (320 m/year (coarse); 32 m/year (fine); CCME 2003); and,
i = lateral hydraulic gradient in aquifer (0.05; CCME 2003).

Substituting these values in the above equations yields values for DF3 of 3.67 and 2.76 for coarse and fine soil, respectively.

7.3.4 Dilution Factor 4

Dilution factor 4 (DF4) accounts for the processes of dispersion and biodegradation as groundwater travels downgradient from beneath the source of contamination, and is the ratio of the concentration of a chemical in groundwater beneath the source, to the concentration in groundwater at a distance downgradient of the source. As noted earlier, for SQG_{DW}, the distance downgradient is assumed to be zero, and accordingly there is no dilution from this process and DF4 takes the value 1.

7.4 Volatilization of Contaminants to Indoor Air

Calculations of the guidelines for cancer and non-cancer effects are provided in Sections 7.4.1 and 7.4.2, respectively. Dilution factor Calculations are provided in Section 7.4.3. Guidelines for these two sets of effects are compared in Section 7.4.4, and the lower value selected. The rationale for the values used for certain parameters is provided in Section 7.4.5.

7.4.1 Cancer Effects

Soil quality guidelines are calculated in this Section for the indoor infiltration and inhalation pathway (SQG_I) for both coarse and fine soils considering cancer effects only. The calculation for SQG_I for trichloroethylene in a residential setting with slab-on-grade construction for both coarse and fine soil is provided as an example. The following equation was used.

$$SQG_I = \frac{RsC \times [\theta_w + (K_{oc} \times f_{oc} \times \rho_b) + (H' \times \theta_a)] \times DF_i \times 10^3}{H' \times \rho_b \times ET \times 10^6} + BSC$$

Where: SQG_I = soil quality guideline for indoor infiltration (mg/kg);
RsC = risk-specific concentration (0.00164 mg/m^3; Table 9);
θ_w = moisture-filled porosity (0.119 (coarse soil); 0.168 (fine soil); CCME 2003);
K_{oc} = organic carbon partition coefficient (86 mL/g; Table 2);
f_{oc} = fraction organic carbon (0.005 g/g; CCME 2003);

ρ_b = dry soil bulk density (1.7 g/cm^3 (coarse soil); 1.4 g/cm^3 (fine soil); CCME 2003);
H’ = dimensionless Henry’s Law Constant (0.36; Table 2);
θ_a = vapour-filled porosity (0.241 (coarse soil); 0.302 (fine soil); CCME 2003);
DF$_i$ = dilution factor from soil gas to indoor air (14,300 (coarse soil) or 130,000 (fine soil); see derivation below);
10^3 = conversion factor from kg to g;
ET = exposure term for residential (1; CCME 2003);
10^6 = conversion factor from m^3 to cm^3; and,
BSC = background soil concentration (0 mg/kg; Section 7.1).

Substituting these values in the above equations and rounding to 2 significant figures yields values of 0.036 mg/kg (coarse soil) and 0.37 mg/kg (fine soil) which are the guidelines for human indoor vapour inhalation of trichloroethylene in a residential scenario in a building with slab-on-grade construction (SQG$_I$) protective of cancer effects. Guidelines for other scenarios were calculated by substituting standard (CCME 2003) parameters for these scenarios in the above equations. Note that the exposure term for cancer effects is 1 for all exposure scenarios.

7.4.2 Non-Cancer Effects

Soil quality guidelines are calculated in this Section for the indoor infiltration and inhalation pathway (SQG$_I$) for both coarse and fine soils considering non-cancer effects only. The calculation for SQG$_I$ for trichloroethylene in a residential setting with slab-on-grade construction for both coarse and fine soil is provided as an example. The following equation was used.

$$SQG_I = \frac{(TC - C_a) \times [\theta_w + (K_{oc} \times f_{oc} \times \rho_b) + (H' \times \theta_a)] \times SAF \times DF_i \times 10^3}{H' \times \rho_b \times ET \times 10^6} + BSC$$

Where:
SQG$_I$ = soil quality guideline for indoor infiltration (mg/kg);
TC = tolerable concentration (0.040 mg/m^3; Table 9);
C$_a$ = background air concentration (0.0014 mg/m^3; Section 7.1);
θ_w = moisture-filled porosity (0.119 (coarse soil); 0.168 (fine soil); CCME 2003);
K$_{oc}$ = organic carbon partition coefficient (86 mL/g; Table 2);
f$_{oc}$ = fraction organic carbon (0.005 g/g; CCME 2003);
ρ_b = dry soil bulk density (1.7 g/cm^3 (coarse soil); 1.4 g/cm^3 (fine soil); CCME 2003);
H’ = dimensionless Henry’s Law Constant (0.36; Table 2);

θ_a = vapour-filled porosity (0.241 (coarse soil); 0.302 (fine soil); CCME 2003);
SAF = soil allocation factor (0.2; Section 7.1);
DF_i = dilution factor from soil gas to indoor air (14,300 (coarse soil) or 130,000 (fine soil); see derivation below);
10^3 = conversion factor from kg to g;
ET = exposure term for residential (1; CCME 2003);
10^6 = conversion factor from m^3 to cm^3; and,
BSC = background soil concentration (0 mg/kg; Section 7.1).

Substituting these values in the above equations and rounding to 2 significant figures yields values of 0.17 mg/kg (coarse soil) and 1.8 mg/kg (fine soil) which are the guidelines for human indoor vapour inhalation of trichloroethylene in a residential scenario in a building with slab-on-grade construction (SQG_I) protective of non-cancer effects. Guidelines for other scenarios were calculated by substituting standard (CCME 2003) parameters for these scenarios in the above equations.

7.4.3 Dilution Factor Calculation

This section presents the equations that were used to calculate the dilution factor. Data for the residential scenario with slab-on-grade construction for coarse and fine soil are provided as examples. The dilution factor (DF_i) was calculated as follows:

$$DF_i = \frac{1}{\alpha}$$

Where: DF_i = dilution factor from soil gas concentration to indoor air concentration (unitless); and,
α = attenuation coefficient (unitless; see derivation below).

Calculation of α

The attenuation coefficient, α, was calculated using the following equation:

$$\alpha = \frac{\left(\frac{D_T^{eff} A_B}{Q_B L_T}\right) exp\left(\frac{Q_{soil} L_{crack}}{D_{crack} A_{crack}}\right)}{exp\left(\frac{Q_{soil} L_{crack}}{D_{crack} A_{crack}}\right) + \left(\frac{D_T^{eff} A_B}{Q_B L_T}\right) + \left(\frac{D_T^{eff} A_B}{Q_{soil} L_T}\right)\left[exp\left(\frac{Q_{soil} L_{crack}}{D_{crack} A_{crack}}\right) - 1\right]}$$

where:

α = attenuation coefficient (dimensionless);
D_T^{eff} = effective porous media diffusion coefficient ($5.29\text{x}10^{-3}$ cm^2/s, coarse soil; and, $6.58\text{x}10^{-3}$ cm^2/s, fine soil; calculated below);
A_B = building area ($1.5\text{x}10^6$ cm^2; CCME 2003);
Q_B = building ventilation rate (203,000 cm^3/s; calculated below);
L_T = distance from contaminant source to foundation (30 cm; CCME 2003);
Q_{soil} = volumetric flow rate of soil gas into the building (15.1 cm^3/s, coarse soil; and, 1.51 cm^3/s, fine soil; calculated below);
L_{crack} = thickness of the foundation (11.25 cm; CCME 2003);
D_{crack} = effective vapour diffusion coefficient through the crack ($5.29\text{x}10^{-3}$ cm^2/s; Section 7.1); and,
A_{crack} = area of cracks through which contaminant vapours enter the building (995 cm^2; CCME 2003).

Substituting these values in the above equations yields values for α in coarse and fine soil of $7.01\text{x}10^{-5}$ and $7.68\text{x}10^{-6}$, respectively, corresponding to dilution factors of 14,300 and 130,000, respectively.

Calculation of D_T^{eff}:

$$D_T^{eff} \approx D_a \times \left(\frac{\theta_a^{10/3}}{\theta_t^2} \right)$$

Where:
D_T^{eff} = overall effective porous media diffusion coefficient based on vapour-phase concentrations for the region between the source and foundation (cm^2/s);
D_a = diffusion coefficient of trichloroethylene in air (0.0787 cm^2/s; Table 2);
θ_a = soil vapour-filled porosity (0.241, coarse soil; and, 0.302, fine soil; CCME 2003); and,
θ_t = soil total porosity (0.36, coarse soil; and, 0.47, fine soil; CCME 2003).

Substituting these values in the above equations yields a value for D_T^{eff} of 0.00529 cm^2/s (coarse soil) and 0.00658 cm^2/s (fine soil).

Calculation of Q_B:

$$Q_B = \frac{L_B W_B H_B ACH}{3{,}600}$$

Where: Q_B = building ventilation rate (cm^3/s);
L_B = building length (1,225 cm; CCME 2003);
W_B = building width (1,225 cm; CCME 2003);
H_B = building height (488 cm; CCME 2003);
ACH = air exchanges per hour (1 h^{-1}; CCME 2003); and,
3,600 = conversion factor from hours to seconds.

Substituting these values in the above equations yields a value for Q_B of 203,000 cm^2.

Calculation of Q_{soil}:

$$Q_{soil} = \frac{2\pi \Delta P k_v X_{crack}}{\mu \, ln\left[\frac{2Z_{crack}}{r_{crack}}\right]}$$

Where Q_{soil} = volumetric flow rate of soil gas into the building (cm^3/s);
ΔP = pressure differential (40 g/cm·s^2; CCME 2003);
k_v = soil vapour permeability to vapour flow (1.0 x 10^{-8} cm^2, coarse soil; 1.0 x 10^{-9} cm^2, fine soil; CCME 2003);
X_{crack} = length of idealized cylinder (4,900 cm; CCME 2003);
μ = vapour viscosity (0.000173 g/cm·s; CCME 2000);
Z_{crack} = distance below grade to idealized cylinder (11.25 cm; CCME 2003); and,
r_{crack} = radius of idealized cylinder (0.2 cm; CCME 2003).

Substituting these values in the above equations yields values for Q_{soil} in coarse and fine soil of 15.1 cm^3/s and 1.51 cm^3/s, respectively.

7.4.4 Comparison of Guidelines for Cancer and Non-Cancer Effects

The soil quality guidelines calculated in Section 7.4.1 for cancer effects are lower than the guidelines for non-cancer effects calculated in Section 7.4.2 for all land uses. Accordingly the cancer-based guidelines for volatilization of contaminants to indoor air are adopted as the SQG_I for trichloroethylene (Table 10).

7.5 Consumption of Contaminated Produce, Meat, and Milk

Calculations of the guidelines for the consumption of contaminated produce, meat, and milk (SQG_{FI}) for cancer and non-cancer effects are provided in Sections 7.5.1 and 7.5.2, respectively. Guidelines for these two sets of effects are compared in Section 7.5.3, and the lower value selected. The rationale for the values used for certain parameters is provided in Section 7.1.

Trichloroethylene is not expected to biomagnify, based on its relatively low Kow (320, Table 2), and corresponding very low biotransfer factors into meat and milk (see below). Accordingly, the meat, produce, and milk ingestion pathway is treated as a check value, and professional judgement is required before using the result as a generic guideline (CCME 2003).

7.5.1 Cancer Effects

For cancer effects, the CCME (2003) protocol considers an adult exposed over their entire lifetime to develop a soil quality guideline for this exposure pathway. The human health soil guideline for cancer effects was calculated using:

$$SQG_{FI} = \frac{RsD \times BW}{(P_h \times P_c \times B_v) + (M_h \times M_c \times B_p \times SIR_c) + (MK_h \times MK_c \times B_m \times SIR_c)} + [BSC]$$

Where:

SQG_{FI} = soil quality guideline for the consumption of contaminated produce, meat and milk (mg/kg);
RsD = risk-specific dose (1.23×10^{-3} mg/kg bw per day; Table 9);
BW = adult body weight (70 kg; CCME 2003);
P_h = proportion of produce home grown (0.5 for agricultural; 0.1 for residential; CCME 2003);
P_c = produce consumption rate (0.25 kg/day for adults; CCME 2003);
B_v = biotransfer factor for produce (1.37 kg(soil)/kg(produce); calculated from $\log B_v = 1.59 - 0.58 \log K_{ow}$; CCME 2003);
M_h = proportion of meat home produced (0.5 for agricultural; 0 for residential; CCME 2003);
M_c = meat consumption rate (0.25 kg/day for adults; CCME 2003);
B_p = biotransfer factor for meat (8.0×10^{-6} day/kg(meat); calculated from $\log B_p = -7.6 + \log K_{ow}$; CCME 2003);
SIR_c = soil ingestion rate for cattle (0.9 kg/day; CCME 2003);
MK_h = proportion of milk home produced (1.0 for agricultural; 0 for residential; CCME 2003);
MK_c = milk consumption rate (0.23 kg/day for adults; CCME 2003);
B_m = biotransfer factor for milk (2.5×10^{-6} day/kg(milk); calculated from $\log B_m = -8.1 + \log K_{ow}$; CCME 2003); and,
BSC = background soil concentration (0 mg/kg; Section 7.1).

Substituting these values in the above equation and rounding to 2 significant figures yields 0.50 mg/kg and 2.5 mg/kg which are the soil quality guidelines for consumption of contaminated produce, meat, and milk (SQG_{FI}) in agricultural and residential settings, respectively, for cancer effects.

7.5.2 Non-Cancer Effects

For non-cancer effects, the CCME (2003) protocol considers a toddler (the most sensitive life-stage for this exposure pathway) for both agricultural and residential land uses. The SQG_{FI} for non-cancer effects was calculated using:

$$SQG_{FI} = \frac{(TDI - EDI) \times SAF \times BW}{(P_h \times P_c \times B_v) + (M_h \times M_c \times B_p \times SIR_c) + (MK_h \times MK_c \times B_m \times SIR_c)} + [BSC]$$

Where:

SQG_{FI} = soil quality guideline for the consumption of contaminated produce, meat and milk (mg/kg);

TDI = tolerable daily intake (1.46×10^{-3} mg/kg bw per day; Table 9);

EDI = estimated daily intake (0.53×10^{-3} mg/kg bw per day for toddlers; Section 7.1);

SAF = soil allocation factor (0.2; Section 7.1);

BW = adult body weight (70 kg; CCME 2003);

P_h = proportion of produce home grown (0.5 for agricultural; 0.1 for residential; CCME 2003);

P_c = produce consumption rate (0.25 kg/day for adults; CCME 2003);

B_v = biotransfer factor for produce (1.37 kg(soil)/kg(produce); calculated from $\log B_v = 1.59 - 0.58 \log K_{ow}$; CCME 2003);

M_h = proportion of meat home produced (0.5 for agricultural; 0 for residential; CCME 2003);

M_c = meat consumption rate (0.25 kg/day for adults; CCME 2003);

B_p = biotransfer factor for meat (8.0×10^{-6} day/kg(meat); calculated from $\log B_p = -7.6 + \log K_{ow}$; CCME 2003);

SIR_c = soil ingestion rate for cattle (0.9 kg/day; CCME 2003);

MK_h = proportion of milk home produced (1.0 for agricultural; 0 for residential; CCME 2003);

MK_c = milk consumption rate (0.23 kg/day for adults; CCME 2003);

B_m = biotransfer factor for milk (2.5×10^{-6} day/kg(milk); calculated from $\log B_m = -8.1 + \log K_{ow}$; CCME 2003); and,

BSC = background soil concentration (0 mg/kg; Section 7.1).

Substituting these values in the above equation and rounding to 2 significant figures yields 0.028 mg/kg and 0.14 mg/kg which are the soil quality guidelines for consumption of contaminated produce, meat, and milk (SQG_{FI}) in agricultural and residential settings, respectively, for non-cancer effects.

7.5.3 Comparison of Guidelines for Cancer and Non-Cancer Effects

The soil quality guidelines calculated in Section 7.5.2 for non-cancer effects are lower than the guidelines for cancer effects calculated in Section 7.5.1 for all land uses. Accordingly the non-cancer guidelines: 0.028 mg/kg (agricultural), 0.14 mg/kg (residential) are adopted as the SQG_{FI} (Table 10).

7.6 Off-Site Migration

The soil quality guideline for off-site migration (SQG_{OM}) is not required for volatile chemicals (CCME 2003) and accordingly is not calculated for trichloroethylene.

7.7 Final Human Health Soil Quality Guideline

Trichloroethylene is a soluble, volatile chemical, that is not expected to biomagnify, and as such (CCME 2003) the required exposure pathways for the development of the soil quality guideline for human health (SQG_{HH}) are direct contact ($PSQG_{HH}$), drinking water (SQG_{DW}), vapour inhalation (SQG_I), produce, meat and milk (SQG_{FI}) (the last is only considered a check mechanism for this chemical). All of the above guidelines and checks were calculated for trichloroethylene, and thus a full final SQG_{HH} can be calculated. The SQG_{HH} is set at the lowest of the human health guidelines which is the drinking water guideline (SQG_{DW}) for all land uses and soil types (Table 10). The SQG_{DW} is 0.01 mg/kg.

8. RECOMMENDED CANADIAN SOIL QUALITY GUIDELINES FOR TCE

Table 10 provides a collation of all soil quality guideline and check values developed in Chapters 6 and 7. The CCME (2003) protocol further specifies the nomination of the most sensitive risk-based guideline for the protection of each of human and environmental health, as the final CCME soil quality guidelines.

According to the formal CCME protocol (CCME 1996a), both environmental (SQG_E) and human health (SQG_{HH}) soil quality guidelines are developed for four land uses: agricultural, residential/parkland, commercial and industrial. The lowest value generated by the two approaches for each of the four land uses is recommended by CCME as the Canadian Soil Quality Guideline.

Revised SQG_E based on direct soil contact and protection of soil invertebrates and plants are similar to values provisionally proposed under the 1999 revision to the TCE SQG. The SQG_{SC} values are 3 mg/kg for agricultural and residential/parkland land uses and 50 mg/kg for commercial and industrial settings. Freshwater life protective check values are much lower than this (0.05 and 0.16 mg/kg in coarse and fine soils, respectively), and protection goals are expected to be far more stringent at TCE-contaminated sites where there is a potential for groundwater-mediated transport of TCE to adjacent water bodies with aquatic life.

For humans, the SQG_{HH} associated with direct soil ingestion was calculated to be 28, 100 and 1700 mg/kg for exposures in an agricultural or residential/parkland, commercial, or industrial setting respectively. The acceptable soil concentrations for TCE for indirect exposure pathways are estimated to be several orders of magnitude lower than this. The most stringent of the indirect pathways is the protection of potable groundwater, with a guideline value of 0.01 mg/kg for both soils types and all four land uses.

The environmental soil quality guidelines (SQG_E) that have been derived for trichloroethylene for all of the four land uses, based on potential for groundwater mediated transfer to adjacent water bodies that contain aquatic life are 0.05 mg/kg in coarse textured soils and 0.016 mg/kg in fine-textured soils.

The human health soil quality guideline (SQG_{HH}) that has been derived is 0.01 mg/kg for both soil types and all four land uses.

The Canadian Soil Quality Guideline for Trichloroethylene for the protection of environmental and human health is: 0.01 mg/kg for both soils types and all four land uses.

In practice, those assessing and remediating contaminated sites within Canada have tended to avail themselves of the broader range of developed preliminary SQG_{HH} and SQG_E when interpreting risks at a site, and developing risk management or risk reduction strategies. The larger set of preliminary SQG provide pathway-specific estimates of protective thresholds, which can be used along with an implicit or explicit qualitative environmental risk assessment approach. In particular, this avoids the oft-seen mistake of failing to account for risks associated with groundwater-mediated contaminant transfer, since these check values may not be captured in the final nominated SQG for a substance.

In closing, it is re-emphasized that vinyl chloride is a potential degradation product of TCE, and may be more toxic than TCE. Accordingly, it is imperative that an assessment of vinyl chloride concentrations be made whenever TCE is present in the environment.

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Table 1. Common Synonyms and Trade Names for Trichloroethylene

Trichloroethylene – CAS# 79-01-6

TCE
1,2,2-trichloroethylene
1,1,2-trichloroethylene
1,1,2-trichloroethene
1,1-dichloro-2-chloroethylene
1-chloro-2,2-dichloroethylene
acetylene trichloride
ethylene trichloride
ethinyl trichloride
Algylen
Anamenth
Benzinol
Blacosolv
Blancosolv
Cecolene
Chlorilen
Chlorylea
Chlorylen
Chorylen
Circosolv
Crawhaspol
Densinfluat
Dow-Tri
Dukeron
Fleck-Flip
Flock Flip
Fluate
Gemalgene
Germalgene
Lanadin
Lethurin
Narcogen
Narkogen
Narkosoid
Nialk
Perm-A-Chlor
Perm-A-Clor
Petzinol
Philex
Threthylene
Trethylene
Tri
Triad
Trial
Triasol
Trichloran
Trichlorathane
Trichlorathane
Trichloren
Trichloroethene
Triciene
Tri-Clene
Trielene
Trieline
Triklone
Trilene
Triline
Trimar
Triol
Tri-Plus
Tri-Plus M
Vestrol
Vitran
Westrosol

Table 2. Physical and Chemical Properties of Trichloroethylene

Property	Unit	Value	CV [1]	Range	Comment	Reference
CAS No.	-	79-01-6				
Physical State (@ 25°C)	-	Liquid				WHO (1985)
Colour	-	Colourless				WHO (1985)
Odour	-	Ethereal				WHO (1985)
Molecular Weight	g/mol	131.4				Government of Canada (1993)
1 ppm equals	mg/m^3	5.48			At room temperature; calculated	Government of Canada (1993)
Melting Point	°C	-83.5				Hsieh (1994)
Boiling Point	°C	86.7				Government of Canada (1993)
Water Solubility	mg/L	1,450	0.15	1,100-1,818	Mean of 7 measured values	Hsieh et al. (1994)
Density, Specific Gravity	g/mL	1.46 at 20°C				Government of Canada (1993)
Surface Tension	N/m	0.029 at 20°C 0.0264 at 20°C			Liquid Vapour	Muraoka and Hirata (1988) McNeill (1979)
Henry's Law Constant	Pa-m^3/mol	890 at 25°C 320 at 5°C	0.18	683-1,186	Mean of 12 measured values Estimated; see text	Hsieh et al. (1994)
Henry's Law Constant (dimensionless)	-	**0.36 at 25°C** 0.14 at 5°C			Estimated; see text Estimated; see text	Hsieh et al. (1994) Estimated; see text
Vapour Pressure	Pa	9,700 at 25°C 3,500 at 5°C	0.02	9,466-9,986	Mean of 5 measured values Estimated; see text	Hsieh et al. (1994)
Octanol/Water Partition Coefficient (K$_{ow}$)	-	**320**	0.32	195-468	Mean of 6 measured values	Hsieh et al. (1994)
Organic Carbon/Water Partition Coefficient (K$_{oc}$)	L/kg	**86**	0.46	18.5-150	Mean of 5 measured values	Hsieh et al. (1994)
Diffusion Coefficient in Air	cm^2/s	**0.0787**			Estimated	Hsieh et al. (1994)
Diffusion Coefficient in Water	cm^2/s	1 x 10^{-5}			Estimated	Hsieh et al. (1994)

1. CV = coefficient of variation, defined as (standard deviation of data)/(mean of data) expressed as a percentage

Table 3. Possible Impurities and Stabilizers in Commercial Trichloroethylene

Impurities	Stabilizers
carbon tetrachloride	pentanol
chloroform	thymol
1,2-dichloroethane	triethanolamines
trans1,2-dichloroethylene	triethylamine
cis1,2-dichloroethylene	2,2,4-trimethylpentene
pentachloroethane	cyclohexene oxide
1,1,1,2-tetrachloroethane	*n*-propanol
1,1,2,2-tetrachloroethane	iso-butanol
1,1,1-trichloroethane	*n*-methylmorpholine
1,1,2-trichloroethane	diisopropylamine
1,1-dichloroethylene	*n*-methylpyrrole
bromodichloroethylene	methylethylketone
perchloroethylene	epichlorohydrin[a]
bromodichloromethane	
benzene	

Source: WHO (1985)

[a] *Now used to a much lesser extent commercially.*

Table 4. Estimated Daily Intake of Trichloroethylene by Canadians

Age Range (years)	(Unit)	Infant 0 - 0.5	Toddler 0.5 - 4	Child 5 - 11	Youth 12 - 19	Adult 20 - 70
Ambient air [a]	µg/kg bw per day	0.003 – 0.02	0.004 – 0.03	0.005 – 0.03	0.004 – 0.03	0.004 – 0.02
Indoor air [a]	µg/kg bw per day	0.33	0.45	0.52	0.43	0.38
Drinking Water [a]	µg/kg bw per day	0.02	0.01	0.007	0.005	0.004
Food [a]	µg/kg bw per day	0.02	0.02 – 0.04	0.01 – 0.04	0.006 – 0.02	0.004 – 0.01
Total Intake [a]	µg/kg bw per day	0.37 – 0.39	0.48 – 0.53	0.54 – 0.60	0.45 – 0.49	0.39 – 0.41
Value Used in This Document	**µg/kg bw per day**		**0.53**			**0.41**

a: source CEPA (1993)

Table 5. Existing Soil and Water Quality Guidelines for Trichloroethylene

Medium (units)	Jurisdiction	Criteria	Concentration	Reference
		Current Guidelines		
Soil (mg/kg)				
	Canada	Canadian Environmental Quality Guidelines:		CCME (1999)
		Agricultural land	0.1	
		Residential/parkland	3	
		Commercial/industrial land	31	
	Alberta	Tier I Criteria for Contaminated Soil Assessment and Remediation	0.1	AENV (1994)
	British Columbia	Matrix Numerical Soil Standards:		CSR (2003)
		Human soil intake:		
		Agricultural/	200	
		Residential/Urban Park	200	
		Commercial	600	
		Groundwater used for drinking water	0.15	
		Plants and Soil invertebrates:		
		Agricultural/	0.1	
		Residential/Urban Park	5	
		Commercial/Industrial	50	
		Groundwater for aquatic life	0.65	
		Groundwater for livestock watering	0.15	
	Ontario	Soil clean-up criteria, all land uses, all scenarios.	0.39	OMEE 1994a
	Netherlands	Target value	0.1	VROM (2000)
		Intervention value	60	

Table 5. Existing Soil and Water Quality Guidelines for Trichloroethylene (cont.)

Medium (units)	Jurisdiction	Criteria	Concentration	Reference
		Current Guidelines (cont.)		
Drinking Water (µg/L)	Canada	Maximum Acceptable Concentration (MAC)	5	Health Canada (2004)
	British Columbia	Drinking Water Guideline	50	CSR (2003)
	Ontario	Drinking-Water Quality Standards	50	SDWA (2002)
	United States	Maximum Acceptable Concentration (MAC)	5	USEPA (2004a)
		Human Health for the consumption of:		
		Water plus organism		USEPA (2002)
		Organism only	2.5	
			30	
	World	Guideline Value (Provisional)	70	WHO (2004)
Water for Aquatic Life (µg/L)	Canada	Protection of Freshwater Aquatic Life (Interim)	21	CCME (1999)
	Ontario	Provincial Water Quality Objective (Interim)	20	OMEE 1994b
Water for Livestock	Canada	Protection of Livestock Watering	50	CCME (1999)
	British Columbia	Protection of Livestock Watering	50	CSR (2003)
Groundwater (µg/L)	Ontario	Potable or non-potable groundwater	50	OMEE 1994
	The Netherlands	Target value	24	VROM (2000)
		Intervention value	500	

Table 5. Existing Soil and Water Quality Guidelines for Trichloroethylene (cont.)

Medium (units)	Jurisdiction	Criteria	Concentration	Reference
Former and Superseded Guidelines				
Soil (mg/kg)	Canada	Assessment Criterion	0.1	CCME 1991
		Remediation Criteria:		
		Agricultural land	0.1	
		Residential/parkland	5.0	
		Commercial/industrial land	50	
Water (General) (μg/L)	Canada	Assessment Criteria	0.1	CCME 1991
Water for Aquatic Life (μg/L)	Canada	Protection of Freshwater Aquatic Life (Interim)	20	CCREM 1987

Table 6. Trichloroethylene Degradation Rates

Property	Unit	Value	CV[1]	Range	Comment	Reference
Half-life in Air	days	3.5	0.11	2.8-4.0	Mean of 6 experimental values	Hsieh et al. (1994)
Half-life in Surface Water	days	120	0.88	7-325	Mean of 8 experimental values	Hsieh et al. (1994)
Half-life in Groundwater	days	800	1.5	128-2,888	Mean of 6 experimental values	Hsieh et al. (1994)
Half-life in Soil	days	760	1.4	33-2,888	Mean of 6 experimental values	Hsieh et al. (1994)
Half-life in Sediment	days	43				Mackay et al. (1993)

1. CV = coefficient of variation, defined as (standard deviation of data)/(mean of data) expressed as a percentage

Table 7. Toxicity of Trichloroethylene to Terrestrial Plants

Organism	Effect	End-point	Test Duration	Observed Response Mean	Observed Response Units	Media Type	Reference
Lactuca sativa (lettuce)	Seedling emergence	NOEC	Not indicated	690 nominal 16 measured	mg/kg	Artificial soil	Environment Canada (1995)
		LOEC		1,400 nominal 48 measured	mg/kg		
		EC_{25} (a)		940 nominal 26 measured	mg/kg		
		EC_{50}		1,200 nominal 37 measured	mg/kg		
Raphanus sativus (radish)	Seedling emergence	NOEC	Not indicated	340 nominal 9 measured	mg/kg	Artificial soil	Environment Canada (1995)
		LOEC		690 nominal 16 measured	mg/kg		
		EC_{25}		600 nominal 14 measured	mg/kg		
		EC_{50}		1,500 nominal 53 measured	mg/kg		
Consulted Data Not Suitable for Guideline Derivation							
Sinapis alba	Growth	EC_{50}	14 d	>1,000 (nominal)	mg/kg	Unknown (pH 6)	Pestemer and Auspurg (1989)
Brassica napus napus	Growth	EC_{50}	14 d	“	mg/kg	Unknown (pH 6)	
Brassica rapa - rapa	Growth	EC_{50}	14 d	“	mg/kg	Unknown (pH 6)	
Brassica chinensis	Growth	EC_{50}	14 d	“	mg/kg	Unknown (pH 6)	
Raphanus sativus	Growth	EC_{50}	14 d	“	mg/kg	Unknown (pH 6)	
Vicia sativa	Growth	EC_{50}	14 d	“	mg/kg	Unknown (pH 6)	
Vigna radiata radiata	Growth	EC_{50}	14 d	“	mg/kg	Unknown (pH 6)	
Trifolium pratense	Growth	EC_{50}	14 d	“	mg/kg	Unknown (pH 6)	
Lolium perenne	Growth	EC_{50}	14 d	“	mg/kg	Unknown (pH 6)	
Avena sativa	Growth	EC_{50}	14 d	“	mg/kg	Unknown (pH 6)	
Triticum aestivum	Growth	EC_{50}	14 d	“	mg/kg	Unknown (pH 6)	

Table 7. Toxicity of Trichloroethylene to Terrestrial Plants (cont.)

Sorghum bicolor bicolor	Growth	EC_{50}	14 d	“	mg/kg	Unknown (pH 6)	Pestemer and Auspurg (1989)
Lepidium sativum	Growth	EC_{50}	14 d	“	mg/kg	Unknown (pH 6)	
Lactuca sativa	Growth	EC_{50}	14 d	“	mg/kg	Unknown (pH 6)	
Avena sativa	Growth	EC_{50}	14 d	>1,000 (nominal)	mg/kg	Unknown	Kordel et al. (1984)
Brassica rapa	Growth	EC_{50}	14 d	>1,000 (nominal)	mg/kg	Unknown	
Avena sativa	Growth	LC_{50}	14 d	>1,000 (nominal)	mg/kg	Unknown	Ballhorn et al. (1984)
Brassica rapa	Growth	LC_{50}	14 d	>1,000 (nominal)	mg/kg	Unknown	
Nicotiana tabacum	Germination	ED_{25}	2 h	< 4	mg/l	Hydroponic	Schubert et al. (1995)
Nicotiana tabacum	Germination	ED_{25}	2 h	<730	ppm	Hydroponic	
Nicotiana tabacum	Germination	ED_{50}	2 h	31.7	mg/l	Hydroponic	
Nicotiana tabacum	Germination	ED_{50}	2 h	5,800	ppm	Hydroponic	
Medicago sativa sativa	Seed Germination	NOEC	1 d	1,300	mg/l	Culture Medium (fumigation)	Young (1929)
Medicago sativa sativa	Seed Germination	EC_{6}	1 d	650	mg/l	Culture Medium (fumigation)	
Hordeum vulgare	Seed Germination	NOEC	1 d	1,300	mg/l	Culture Medium (fumigation)	
Phaseolus lunatus	Seed Germination	NOEC	1 d	1,300	mg/l	Culture Medium (fumigation)	
Fabaceae	Seed Germination	NOEC	1 d	1,300	mg/l	Culture Medium (fumigation)	
Fagopyrum esculentum	Seed Germination	EC_{16}	1 d	1,300	mg/l	Culture Medium (fumigation)	
Fagopyrum esculentum	Seed Germination	NOEC	1 d	650	mg/l	Culture Medium (fumigation)	
Trifolium sp.	Seed Germination	NOEC	1 d	1,300	mg/l	Culture Medium (fumigation)	

Table 7. Toxicity of Trichloroethylene to Terrestrial Plants (cont.)

Trifolium sp.	Seed Germination	NOEC	1 d	650	mg/l	Culture Medium (fumigation)	Young (1929)
Zea mays	Seed Germination	EC_8	1 d	1,300	mg/l	Culture Medium (fumigation)	
Zea mays	Seed Germination	EC_{11}	1 d	650	mg/l	Culture Medium (fumigation)	
Vigna unguiculata unguiculata	Seed Germination	NOEC	1 d	1,300	mg/l	Culture Medium (fumigation)	
Vigna unguiculata unguiculata	Seed Germination	NOEC	1 d	650	mg/l	Culture Medium (fumigation)	
Avena sativa	Seed Germination	EC_{11}	1 d	1,300	mg/l	Culture Medium (fumigation)	
Avena sativa	Seed Germination	NOEC	1 d	650	mg/l	Culture Medium (fumigation)	
Secale cereale	Seed Germination	NOEC	1 d	1,300	mg/l	Culture Medium (fumigation)	
Helianthus annuus	Seed Germination	NOEC	1 d	1,300	mg/l	Culture Medium (fumigation)	
Phleum pratense	Seed Germination	NOEC	1 d	1,300	mg/l	Culture Medium (fumigation)	
Phleum pratense	Seed Germination	EC_5	1 d	650	mg/l	Culture Medium (fumigation)	
Daucus carota	Root Growth	NR	8 d	0.25	ppm	Vermiculite	Inderjit et al. (2003)
Daucus carota	Shoot Growth	NR	8 d	0.25	ppm	Vermiculite	
Daucus carota	Root Growth	NR	8 d	0.25	ppm	Filter Paper / soaked cotton	
Daucus carota	Shoot Growth	NR	8 d	0.25	ppm	Filter Paper / soaked cotton	

(a) The EC endpoints represent the effects concentration as calculated by the CCME from the data presented by the author(s).

NQ = the observe effect was not quantifiable

Table 8. Toxicity of Trichloroethylene to Terrestrial Invertebrates

Organism	Effect	Endpoint[(a)]	Concentration mg/kg nominal (measured)	Test Substrate	Analytical Method	References
Earthworm (*Eisenia foetida*)	Mortality	NOEC LOEC LC_{25} LC_{50}	1,830 (60) 3,661 (159) 2,212 (79) 2,695 (106)	Artificial Soil	EPA method 8240	Environment Canada 1995
Earthworm (*Eisenia foetida*)	Mortality	LC_{50}	(105 mg/cm)	Filter paper	EEC	Neuhauser et al. 1985

Notes:

(a) The EC endpoints represent the effects concentration as calculated by the CCME from the data presented by the author(s).

Table 9. Toxicity Reference Values for Trichloroethylene

Exposure	Effects	Exposure Limit	Unit	Value	Source
Oral	Non-cancer	Tolerable Daily Intake (TDI)	mg/kg bw per day	0.00146	Health Canada (2004)
Inhalation	Non-cancer	Tolerable Concentration (TC)	mg/m^3	0.040	USEPA (2001)
Oral	Cancer	Risk Specific Dose (RsD) [a]	mg/kg bw per day	0.00123	Health Canada (2004)
Inhalation	Cancer	Risk Specific Concentration (RsC) [a]	mg/m^3	0.00164	Health Canada (1996)

These are the values used in this report to calculate soil quality guidelines protective of human health.
See Section 5.9 for an explanation of these values
a: evaluated at an excess cancer risk of 1 in 10^6.

Table 10. Soil Quality Guidelines for Trichloroethylene

	Land Use			
	Agricultural ($mg \cdot kg^{-1}$)	**Residential/ Parkland ($mg \cdot kg^{-1}$)**	**Commercial ($mg \cdot kg^{-1}$)**	**Industrial ($mg \cdot kg^{-1}$)**
Recommended Guideline (coarse and fine soils)*:	**0.01**	**0.01**	**0.01**	**0.01**
Human health guidelines/check values				
Overall SQG_{HH} (coarse and fine soils)	0.01	0.01	0.01	0.01
Direct contact ($PSQG_{HH}$)	28	28	100	1,700
Inhalation of indoor air (SQG_I) :				
coarse soil	0.036(0.058) [a]	0.036(0.058) [a]	0.11	0.11
fine soil	0.37(0.55) [a]	0.37(0.55) [a]	0.92	0.92
Protection of potable water (SQG_{DW}) (coarse and fine soils):	0.01	0.01	0.01	0.01
Produce, meat and dairy products check (SQG_{FI})	0.028	0.14	—	—
Off-site migration check (SQG_{OM})	—	—	—	NC [b]
Environmental health guidelines/check values				
Overall SQG_E				
coarse soil	0.05	0.05	0.05	0.05
fine soil	0.16	0.16	0.16	0.16
Soil contact guidelines (provisional)	3 [c]	3 [c]	50 [c]	50 [c]
Soil and food ingestion	NC [d]	—	—	—
Nutrient and energy cycling check	NC [d]	NC [d]	NC [d]	NC [d]
Protection of freshwater life				
coarse soil	0.05	0.05	0.05	0.05
fine soil	0.16	0.16	0.16	0.16
Off-site migration check	—	—	—	NC [b]
Interim soil quality criterion (CCME 1991)	0.1	5	50	50

Notes:

Note that vinyl chloride is a potential degradation product of TCE that is a more potent carcinogen than TCE. Accordingly, it is imperative that vinyl chloride concentrations be assessed whenever TCE is found or suspected to be present.

SQG_{HH} = soil quality guideline for human health; SQG_E = soil quality guideline for environmental health; NC = not calculated; — = guideline/check value are not a part of the exposure scenario for that land use, or the pathway is not applicable, and therefore is not calculated.

Coarse Soils: soils which contain greater than 50% by mass particles greater than 75 µm mean diameter (D_{50} > 75 µm)

Fine Soils: soils which contain less than 50% by mass particles greater than 75 µm mean diameter (D_{50} .< 75 µm)

* *This guideline value may be less than the common limit of detection for trichloroethylene in some jurisdictions. Contact jurisdiction for guidance.*

[a] *First value is for slab-on-grade construction, value in parentheses is for construction with a basement.*

[b] *Calculation of this check value is not required for volatile compounds.*

[c] *Provisional guideline.*

[d] *Insufficient data available to calculate this guideline/check value.*