

**Canadian Soil Quality Guidelines for
Trichloroethylene: Environmental
Supporting Document — Final Draft
December 1996**

NOTICE

This final draft document provides the information supporting the derivation of environmental soil quality guidelines for trichloroethylene. Development of these soil quality guidelines was initiated through the National Contaminated Sites Remediation Program (NCSRSP) which officially ended in March 1995. Given the need for national soil quality guidelines for contaminated sites management and many other applications, development was pursued under the direction of the CCME Soil Quality Guidelines Task Group after the end of the NCSRSP.

This document is a working document that was released shortly after the publication of "A Protocol for the Derivation of Environmental and Human Health Soil Quality Guidelines" (CCME 1996). The CCME recognizes that some refinements or changes to the Protocol may become necessary upon application and testing. If required, amendments to the Protocol will be made and the guidelines will be modified accordingly. For this reason guidelines are referred to in this document as CCME Recommended Guidelines. Readers who wish to comment or provide suggestions on the Protocol or on the guidelines presented in this document should send them to the following address:

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This document is a supporting technical document. It is available in English only. A French Abstract is given on page vii.

Ce document technique de soutien n'est disponible qu'en anglais avec un résumé en français présenté à la page vii.

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ABSTRACT

Canadian environmental quality guidelines, developed under the auspices of the Canadian Council of Ministers of the Environment (CCME), are numerical concentrations or narrative statements recommended to support and maintain designated resource uses. CCME Canadian soil quality guidelines can be used as the basis for consistent assessment and remediation of contaminants at sites in Canada.

This report was prepared by the Guidelines Division of the Science Policy and Environmental Quality Branch (Environment Canada), which acts as Technical Secretariat for the CCME Soil Quality Guidelines Task Group. The Guidelines were derived according to the procedures described in *A Protocol for the Derivation of Environmental and Human Health Soil Quality Guidelines* (CCME 1996).

Following the introduction, chapter 2 presents chemical and physical properties of trichloroethylene and a review of the sources and emissions in Canada. Chapter 3 discusses trichloroethylene's distribution and behavior in the environment while chapter 4 reports the toxicological effects of trichloroethylene on microbial processes, plants, and animals. These informations are used in chapter 5 to derive soil quality guidelines for trichloroethylene to protect environmental receptors in four types of land uses: agricultural, residential/parkland, commercial, and industrial.

The following soil quality guidelines are recommended by the CCME based on the available scientific data. For trichloroethylene, the available data were insufficient/inadequate to derive soil quality guidelines according to the CCME 1996 Protocol. Therefore, the Provisional Method, described in CCME (1997) and hereafter was used to derive provisional environmental soil quality guidelines for trichloroethylene. The provisional environmental soil quality guideline (SQG_E —provisional) for agricultural and residential/parkland land uses is $3 \text{ mg}\cdot\text{kg}^{-1}$ soil whereas for commercial and industrial land uses it is $31 \text{ mg}\cdot\text{kg}^{-1}$ soil.

RÉSUMÉ

Les recommandations canadiennes pour la qualité de l'environnement, élaborées sous les auspices du Conseil Canadien des Ministres de l'Environnement (CCME), sont des concentrations ou des énoncés décrivant les limites recommandées dans le but d'assurer le maintien et le développement durable d'utilisations désignées des ressources. Les recommandations canadiennes pour la qualité des sols proposées par le CCME peuvent être utilisées comme base pour l'uniformisation des processus d'évaluation et d'assainissement des terrains contaminés au Canada.

Le présent document a été préparé par la Division des Recommandations de la Direction de la Qualité de l'Environnement et de la Politique Scientifique (Environnement Canada), qui agit comme secrétaire technique pour le Groupe de Travail du CCME sur les Recommandation pour la Qualité des Sols. Les Recommandations ont été élaborées selon les procédures décrites dans le *Protocole d'élaboration de recommandations pour la qualité des sols en fonction de l'environnement et de la santé humaine* (CCME 1996).

Faisant suite à une brève introduction, le chapitre 2 présente les propriétés physiques et chimiques du trichloroéthylène de même qu'un survol des sources et des émissions au Canada. Le chapitre 3 discute du devenir et du comportement de cette substance dans l'environnement alors que le chapitre 4 rapporte ses effets toxicologique sur les processus microbiens, les plantes et les animaux. Ces informations sont utilisées au chapitre 5 afin d'élaborer des recommandations pour la qualité des sols relatives au trichloroéthylène en vue de la protection de l'environnement dans le cadre de quatre types d'utilisations de terrains: agricole, résidentiel/parc, commercial et industriel.

Les recommandation pour la qualité des sols suivantes, proposées par le CCME, sont fondées sur les données scientifiques disponibles. Pour le trichloroéthylène, les données disponibles sont insuffisantes/inadéquates pour permettre l'élaboration de recommandations selon les procédures décrites dans le Protocole du CCME (1996). Ainsi, la Méthode provisoire, décrite dans un document du CCME (1997) et ci-après, a été utilisée pour élaborer des recommandations provisoires pour la qualité des sols relatives au trichloroéthylène en vue de la protection de l'environnement. La recommandation provisoire pour la qualité des sols en vue de la protection de l'environnement (RQS_E —provisoire) visant les terrains à vocation agricole et résidentielle/parc est de $3 \text{ mg} \cdot \text{kg}^{-1}$ de sol, alors que pour les terrains à vocation commerciale et industrielle, elle est de $31 \text{ mg} \cdot \text{kg}^{-1}$ de sol.

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1. INTRODUCTION

The Canadian Council of Ministers of the Environment's (CCME) Canadian Environmental Quality Guidelines are numerical limits for contaminants intended to maintain, improve, or protect environmental quality and human health. CCME Canadian Soil Quality Guidelines can be used as the basis for consistent assessment and remediation of contaminants at sites in Canada along with CCME guidelines issued for the protection of water quality, sediment quality and tissue quality. In response to the urgent need to begin remediation of high priority "orphan" contaminated sites, an interim set of criteria was adopted from values currently in use in various jurisdictions across Canada (CCME 1991). Many of the CCME interim soil remediation criteria do not have a complete supporting scientific rationale and are being updated based on current scientific information.

This report reviews the sources and emissions of trichloroethylene, its distribution and behaviour in the environment, and its toxicological effects on terrestrial mammals, plants and soil organisms. This information is used to derive guidelines for trichloroethylene to protect ecological receptors according to the processes outlined in *A Protocol for the Derivation of Environmental and Human Health Soil Quality Guidelines* (CCME 1996) for agricultural, residential/parkland, commercial and industrial land uses.

The values derived herein are environmental soil quality guidelines, and are intended as general guidance. Site specific conditions should be considered in the application of these values. The values may be applied differently in various jurisdictions, therefore, the reader should consult the appropriate jurisdiction before application of the values.

2. BACKGROUND INFORMATION

2.1 Physical and Chemical Properties

Trichloroethylene (1,1,2-trichloroethene ; TCE; CAS Registry No. 79-01-6) is a clear, colourless, nonviscous 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 ($131.4 \text{ g}\cdot\text{mole}^{-1}$), and is a powerful solvent for a large number of natural and synthetic substances (Schaumburg 1990). The physical and chemical properties of TCE are listed in Table 1.

Trichloroethylene is a volatile liquid at room temperature (boiling point 86.7°C) with a higher density ($1.46 \text{ g}\cdot\text{mL}^{-1}$ (20°C)) and a lower surface tension (0.029 N/m) than water. 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. In the absence of stabilizers, it is slowly oxidized by air or photolyzed by light. Trichloroethylene is moderately soluble in water ($1.1 - 1.4 \text{ g}\cdot\text{L}^{-1}$). It has a moderately low *n*-octanol/water partition coefficient ($\log K_{ow} 2.42$) and estimated soil/water partition coefficients (K_d) ranging from 0.30 to 383 depending upon soil type. Trichloroethylene has a high vapour pressure (8.0 kPa at 20°C) and its vapour is heavier than air (Eisenreich *et. al.* 1981; ATSDR 1989).

2.2 Analytical Methods

The National Contaminated Sites Remediation Program (NCSRP) guidance manual for sampling, analysis and data management of contaminants for all environmental media (CCME 1993a,b) recommend two methods for determining trichloroethylene levels in soil. US EPA method 8240B titled "VOCs by gas chromatography (GC) and/or mass spectrophotometry (MS)", and the second is US EPA method 8260A Rev. 1 titled, "Semivolatiles by capillary GC/MS [Part 2]". Details of these methods are found in CCME (1993a,b). Method 8240B has a detection limit of $0.005 \text{ mg}\cdot\text{kg}^{-1}$ wet soil, and method 8260A has a detection limit of $0.19 \text{ }\mu\text{g}\cdot\text{L}^{-1}$.

2.3 Production and Uses

Trichloroethylene is generally produced via chlorination of ethylene or ethylene dichloride (WHO 1985; ATSDR 1991). Canadian supply and demand of trichloroethylene peaked during the mid 1970s with a high of 22.5 kilotonnes produced in 1976. It has been steadily decreasing since then 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. Trichloroethylene was produced in Canada at two plants, C-I-L (now I-C-I Canada) and Venchem, both in Shawinigan, Quebec. However, these plants were closed by 1985, due primarily to decreasing domestic demand. Imports are therefore the only current supply of this chemical in Canada. Total Canadian imports of trichloroethylene equalled 1.6 kilotonnes during 1989 (CIS 1991). The bulk of this amount, 1.3 kilotonnes, was used by the metal cleaning industry, mostly to replace emission losses, while less than 0.2 kilotonnes were used for miscellaneous applications. The remaining 0.1 kilotonnes were exported. Estimated imports for 1990 as well as forecasts for 1993 do not differ significantly from 1989 (1.5 and 1.4 kilotonnes, respectively) (CIS 1991).

Although trichloroethylene use in Canada has declined in recent years, the demand for this solvent could increase significantly in the near future if proposals to control the global production and use of another solvent, 1,1,1-trichloroethane, are approved. 1,1,1-Trichloroethane is also used for metal cleaning purposes, and its market for this application is significantly greater than that for trichloroethylene (14.5 kilotonnes versus 1.3 kilotonnes in 1989, respectively) (CCB 1990; CIS 1991).

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 1991). 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 (CPI 1986).

2.4 Trichloroethylene in the Environment

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. 3.0, 2.5, and 1.5 kilotonnes in 1987, 1988, and 1989, respectively) (CIS 1991).

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 85% of the domestic trichloroethylene supply (CIS 1991). 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).

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 $\mu\text{g}\cdot\text{m}^{-3}$, with a maximum 24-hour concentration of 19.98 $\mu\text{g}\cdot\text{m}^{-3}$ 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 $\mu\text{g}\cdot\text{m}^{-3}$. The spring mean was 0.8 $\mu\text{g}\cdot\text{m}^{-3}$ and the winter mean was 0.2 $\mu\text{g}\cdot\text{m}^{-3}$. At the only rural monitoring site in Canada (Walpole Island, Ont.), mean trichloroethylene concentrations of 0.19 $\mu\text{g}\cdot\text{m}^{-3}$ were measured during 1989-1990, with a maximum concentration of 0.46 $\mu\text{g}\cdot\text{m}^{-3}$ (Dann and Wang 1992). Results for rural U.S. sites indicate that concentrations of trichloroethylene in air ranged between 0.006 and 1.9 $\mu\text{g}\cdot\text{m}^{-3}$ (HWC 1990). Similar studies of forested areas in Germany carried out in 1987 and 1988 indicated atmospheric levels ranging from 0.2 - 1.1 $\mu\text{g}\cdot\text{m}^{-3}$ (average 0.5 $\mu\text{g}\cdot\text{m}^{-3}$) (Frank *et al.* 1989, Frank 1989). Average trichloroethylene levels of 0.04 - 0.05 $\mu\text{g}\cdot\text{m}^{-3}$ 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 $15.5 \mu\text{g}\cdot\text{m}^{-3}$; LaRegina *et al.* 1986; Harkov *et al.* 1983), with maximum recorded values of $108 \mu\text{g}\cdot\text{m}^{-3}$ and $66.5 \mu\text{g}\cdot\text{m}^{-3}$, respectively. In Germany, levels ranging from 800 - $10,500 \mu\text{g}\cdot\text{m}^{-3}$ 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 $1460 \mu\text{g}\cdot\text{m}^{-3}$ have been recorded 0.5 km from a TCE production and storage site in the United States (U.S. EPA 1977).

Surface Water

Reported levels of TCE in Canadian surface waters ranged from below the detection limit ($0.001 \mu\text{g}\cdot\text{L}^{-1}$) to $90.0 \mu\text{g}\cdot\text{L}^{-1}$, 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 $90.0 \mu\text{g}\cdot\text{L}^{-1}$ at the mouth of the Yamaska and St. Francois Rivers in Lac St. Pierre near Sorel, Quebec, and levels at several stations below Cornwall and in Lac St. Louis as high as $2.8 \mu\text{g}\cdot\text{L}^{-1}$ to $20 \mu\text{g}\cdot\text{L}^{-1}$. Generally, levels in the St. Lawrence river were in the 0.01 - $0.05 \mu\text{g}\cdot\text{L}^{-1}$ range. Trichloroethylene has also been found in other Ontario rivers; the Niagara River at Niagara-on-the-Lake (0.008 to $0.12 \mu\text{g}\cdot\text{L}^{-1}$) (Strachan and Edwards 1984), the Welland river (below detection limit to $0.75 \mu\text{g}\cdot\text{L}^{-1}$) (Kaiser and Comba 1983), and the St. Clair river (0.01 - $0.10 \mu\text{g}\cdot\text{L}^{-1}$) (Kaiser and Comba 1986a: detection limit $0.001 \mu\text{g}\cdot\text{L}^{-1}$). In a contaminant plume in the St. Clair River, levels as high as $42 \mu\text{g}\cdot\text{L}^{-1}$ were measured adjacent to industrial sewer outfalls (COARGLWQ 1986). Also, samples from the Great Lakes contained trichloroethylene concentrations ranging from below detection limit to $0.033 \mu\text{g}\cdot\text{L}^{-1}$ (mean $0.0025 \mu\text{g}\cdot\text{L}^{-1}$) in Lake Ontario (Kaiser *et al.* 1983); 0.006 - $0.168 \mu\text{g}\cdot\text{L}^{-1}$ in Lake Erie (Kaiser and Valdmanis 1979), and below detection limit to $0.036 \mu\text{g}\cdot\text{L}^{-1}$ (mean $0.0094 \mu\text{g}\cdot\text{L}^{-1}$) 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 $0.29 \mu\text{g}\cdot\text{L}^{-1}$ to $67 \mu\text{g}\cdot\text{L}^{-1}$ (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 $2553 \mu\text{g}\cdot\text{L}^{-1}$ (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 ($1 \mu\text{g}\cdot\text{L}^{-1}$) to a maximum of $780 \mu\text{g}\cdot\text{L}^{-1}$ 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 (UMA 1992a) had trichloroethylene concentrations in building effluent discharges of $1.9 \mu\text{g}\cdot\text{L}^{-1}$ (boiler cooling water from a stagnant ditch) to approximately $1,100$ - $1,350 \mu\text{g}\cdot\text{L}^{-1}$ (raw water, laboratory waste holding tank, ditch samples), with a maximum of $1555 \mu\text{g}\cdot\text{L}^{-1}$ in a compressor cooling water sample.

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.

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 $102 \mu\text{g}\cdot\text{L}^{-1}$ to $12,950 \mu\text{g}\cdot\text{L}^{-1}$. An extremely high concentration of $181 \times 10^5 \mu\text{g}\cdot\text{L}^{-1}$ 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 (D.L. = $1 \mu\text{g}\cdot\text{L}^{-1}$) to $2480 \mu\text{g}\cdot\text{L}^{-1}$ 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 $21,900 \mu\text{g}\cdot\text{L}^{-1}$ were detected with a mean concentration of $771 \mu\text{g}\cdot\text{L}^{-1}$ (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 $13,200 \mu\text{g}\cdot\text{L}^{-1}$. 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 $490 \mu\text{g}\cdot\text{L}^{-1}$.

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, nonaqueous-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 1000 and $350 \mu\text{g}\cdot\text{L}^{-1}$, respectively (Feenstra 1992).

In Amherst, Nova Scotia, trichloroethylene has been detected in several municipal and private wells at concentrations ranging from 5 to $84 \mu\text{g}\cdot\text{L}^{-1}$ (NAQUADAT 1991). Trichloroethylene was detected in two groundwater drinking water samples from Prince Edward Island at 1.5 and $1.6 \mu\text{g}\cdot\text{L}^{-1}$ (detection limit $1.0 \mu\text{g}\cdot\text{L}^{-1}$) in 1986. However, nine groundwater samples (non-drinking water) from Nova Scotia did not contain detectable levels of trichloroethylene in 1988 (NAQUADAT 1991).

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).

Sediment

Little data exists 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 ($0.01 \text{ ng}\cdot\text{g}^{-1}$) to $1.1 \times 10^5 \text{ ng}\cdot\text{g}^{-1}$ with a mean concentration of $21 \text{ ng}\cdot\text{g}^{-1}$. 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 $21 \text{ ng}\cdot\text{g}^{-1}$, and the maximum was $110 \mu\text{g}\cdot\text{g}^{-1}$ (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.

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" (OTR₉₈) (OMEE 1993). These OTR₉₈ values correspond to the 98th percentile of the sample population analyzed. For trichloroethylene, the OTR₉₈ values are $0.63 \text{ ng}\cdot\text{g}^{-1}$ and $0.028 \text{ ng}\cdot\text{g}^{-1}$ 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 $4.5 \mu\text{g}\cdot\text{g}^{-1}$ (mean: $36 \text{ ng}\cdot\text{g}^{-1}$) (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 $1000 \mu\text{g}\cdot\text{g}^{-1}$. Other reported localized soil contamination occurred beside solvent burn-off areas, where trichloroethylene concentrations ranged from 0.1 to $890 \mu\text{g}\cdot\text{g}^{-1}$ (UMA 1992b).

2.5 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 2.

3. ENVIRONMENTAL FATE AND BEHAVIOUR

Atmosphere

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, cloud cover, TCE will undergo short- and medium-range atmospheric transport (Mackay 1987).

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). 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 puddles due to the density and moderate water solubility of TCE (Schwille 1988). It should be noted that through these puddles 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 puddles 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.

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 - 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).

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⁻¹ of TCE to 200 ng·mL⁻¹ 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).

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 1700 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 (Grathwohl 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 - 0.0292 cm²/s at 1-3% soil moisture and 0.0067 - 0.0070 cm²/s under wet conditions of 13-15% 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 0.254×10^{-3} to $1.986 \times 10^{-3} \text{ cm}^2/\text{s}$, 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.

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). No studies on bioaccumulation in higher plants or terrestrial biota were found.

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 include 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 the required calculations, four primary media, soil, air, water and sediment are used. The emission rate of TCE utilized was 1000 kg·h⁻¹. 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·h⁻¹) while the transport rate between soil-water was second highest (5.08 kg·h⁻¹). 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

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⁻¹. Inamori *et al.* (1989), reported an EC₅₀ of 330 mg TCE·L⁻¹ 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 1000 mg TCE·L⁻¹ (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⁻¹ and 26.3 mg·L⁻¹ in a cell suspension of 420 mg of cells·L⁻¹ (Oldenhuis *et al.* 1989). Marinucci and Chervu (1985) reported that complete inhibition of microbial activity occurred in mixed cultures at concentrations of 5 mg·g⁻¹ in soil and 1000 mg·L⁻¹ in water. Reductions in soil microbial biomass, measured as ATP content, were reported as acute effects occurring at 10 mg·L⁻¹ (Kanazawa and Filip 1986).

Effects on microbial communities, of chronic exposure to trichloroethylene, 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 *Methylomonas* sp. demonstrated a three-fold decrease in their ability to utilize methane following a 2h incubation with 6 mg TCE·L⁻¹ (Henry and Grbic-Galic 1991). The viability of *Methylomonas* cultures was also reduced by trichloroethylene whereby exposed cultures had lower cell counts ($2 \pm 1 \cdot 10^5$ CFU·mL⁻¹) relative to the controls ($5 \pm 2 \cdot 10^6$ CFU·mL⁻¹) (Henry and Grbic-Galic 1991). Kanazawa and Filip (1986) reported reduced proteinase activity from 5.58 nmol·g⁻¹/min (controls) to 2.69 nmol·g⁻¹/min at trichloroethylene concentrations of 10 µg·g⁻¹. 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 3.

Radish (*Raphanus sativa*) 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 3661 mg TCE·kg⁻¹. The nominal concentrations of trichloroethylene in soil for the NOEC, EC₂₅, EC₅₀ and LOEC were 344, 597, 1525 and 686 mg TCE·kg⁻¹. Similarly, Environment Canada (1995) reported the effect of trichloroethylene on lettuce (*Lactuca sativa*) seedling emergence. The test duration was 120 h. The nominal concentrations of trichloroethylene in soil for the NOEC, EC₂₅, EC₅₀, and LOEC were 686, 944, 1238, and 1372 mg TCE·kg⁻¹. The measured concentrations for both radish and lettuce varied between 0 and 220 mg TCE·kg⁻¹.

Native Canadian fir trees (*Abies alba*), non-native tree species Norway spruce (*Picea abies*), beech trees (*Fagus silvatica*), and other trees growing in regions with low SO₂ 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, SO₂, or O₃ (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 to 0.7 µg TCE·m⁻³, with averages around 0.2 - 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 TCE·m⁻³ and 11.8 µg TCE·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 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-*a* and β-carotene being the most severely affected. Chlorophyll-*a* and β-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 3.

The survival of the earthworm *Eisenia fetida* 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 0-7321 mg·kg⁻¹ and the measured concentrations ranged from 0-440 mg·kg⁻¹. The reported NOEC, LC₂₅, LC₅₀ and LOEC, based on nominal concentrations, were 1830, 2212, 2695, and 3661 mg TCE·kg⁻¹ respectively.

The earthworm *Eisenia fetida* was exposed to trichloroethylene in a contact filter paper test for 48 hours (Neuhauser *et al.* 1985). An LC₅₀ 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. 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.

In a two year study in which rats and mice were orally exposed to 500 and 1000 mg·kg⁻¹-bw·d⁻¹, respectively, the survival of male rats and mice was significantly reduced and toxic nephrosis was found in both rats and mice (NTP 1982, 1986). A sub-chronic NOEL of 793 mg·kg⁻¹-bw·d⁻¹ was reported in a 6-month study with mice exposed to drinking water containing TCE (ATSDR 1989).

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. DERIVATION OF ENVIRONMENTAL SOIL QUALITY GUIDELINES

5.1 Introduction

Canadian soil quality guidelines are designed to protect four different land uses: agricultural, residential/parkland, commercial and industrial. The derivation procedures are described in *A Protocol for the Derivation of Environmental and Human Health Soil Quality Guidelines* (CCME 1996).

There were insufficient data available to meet the minimum data requirements described in the Protocol (CCME 1996), for the derivation of soil quality guidelines based on soil contact (SQG_{sc}), soil and food ingestion (SQG_I) or the nutrient and energy cycling check for any of the categories of land use. Studies were excluded from use because of one or more of the following reasons:

- soil pH was not recorded;
- soil pH was below 4 (as this is considered outside the normal pH range of most soils in Canada)
- no indication of soil texture was provided;
- inappropriate statistical analysis was used;
- test was not conducted using soil or artificial soil;
- test soil was amended with sewage sludge or a mixture of toxicants;
- test did not use controls.

Therefore, the Provisional Method, described in CCME (1997) and hereafter was used to derive Provisional Canadian Soil Quality Guidelines for trichloroethylene. Results are presented in Table 4.

5.2 Provisional Soil Quality Guidelines

Provisional Soil Quality Guidelines Derivation Method

Limitations in the availability of quality soil toxicity data hinder the derivation of soil quality guidelines since the methods described in the Protocol (CCME 1996) are all limited by minimum data requirements. However, literature searches often yield data which do not meet the requirements of the 1996 Protocol, but still provide some toxicity information. Also, toxicity tests using standard methodologies may produce data that do not meet the regular quality standards defined by toxicologists, due to difficulties in handling and evaluating certain substances such as volatile organic chemicals in the context of a soil contact test for example.

While acknowledging the need for toxicity data of the highest quality, it is the opinion of the Subcommittee that guidelines based on limited toxicity information are still more practical and scientifically defensible than the absence of guidelines or than guidelines that are not risk based. Thus, a provisional method for deriving environmental soil quality guidelines was proposed (CCME 1997). The guidelines derived using this method are called "provisional" to clearly indicate that the underlying data do not meet the requirements of the Protocol (CCME 1996) and that there is an urgent need for additional research. Also, given the increased uncertainty surrounding the

provisional soil quality guidelines, such provisional guidelines may not be used to raise an existing criteria or guideline, such as the 1991 interim criteria, but only to lower it if the provisional value indicates that the existing criteria or guideline is not adequately protective.

The guiding principles described in the Protocol (CCME 1996) are to be followed while applying relaxed data requirements in order to allow the derivation of provisional soil quality guidelines. When the data requirements of one of the derivation methods described in the Protocol (CCME 1996) are met, regular soil quality guidelines must be derived. However, when the available data do not meet those requirements, professional judgement is used to evaluate the available information and to recommend a provisional soil quality guideline. The derivation of provisional soil quality guidelines is directed by the following landmarks:

- be precautionary: use higher safety factors where degree of uncertainty is high;
- use soil contact data for plants and invertebrates preferentially over mammalian data, equilibrium partitioning data or other data;
- rely on equilibrium partitioning data only to justify additional safety factors and to assist in professional judgement;
- use microbial data as a check mechanism, follow the method described in the protocol (use geometric mean of soil contact number and microbial value where microbial value is lower);
- keep in mind that provisional soil quality guidelines for agricultural and residential/parkland land uses are intended to approximate no appreciable effect levels where those for commercial/industrial land use allow for a low level of effects;
- be consistent with the spirit of the 1996 Protocol.

Derivation of Provisional Soil Quality Guidelines

Table 4 presents a summary of the results of the Provisional derivation procedures and a comparison with the CCME (1991) interim remediation criteria.

The available data was insufficient to meet the minimum data requirements for the derivation of a guideline based on soil contact. However, the available data based on direct soil contact was used to derive a SQG_E—provisional. From the available data, only the reported EC₂₅ endpoints based on direct soil contact experiments, and for these endpoints, only the recovered (measured) concentrations were considered.

For Agricultural and Residential/Parkland land uses, the lowest of the considered values was divided by an uncertainty factor of 5 and the result was taken as the SQG_E—provisional. Thus, from Table 3, the available EC₂₅ endpoints based on direct soil contact experiments are: 14 mg·kg⁻¹ for radish seedling emergence, 26 mg·kg⁻¹ for lettuce seedling emergence, and 79 mg·kg⁻¹ for earthworm survival. Hence, the SQG_E—provisional for Agricultural and Residential/Parkland land uses is:

$$14 / 5 = 2.8 = 3 \text{ mg} \cdot \text{kg}^{-1}$$

For Commercial and Industrial land uses, the SQG_E —provisional was taken as equal to the geometric mean of the available EC_{25} endpoint values listed above, which are based on direct soil contact experiments. Hence, the SQG_E —provisional for Commercial and Industrial land uses is:

$$(14 \times 26 \times 79)^{1/3} = 30.6 = 31 \text{ mg} \cdot \text{kg}^{-1}$$

5.3 Groundwater Check

Soils are hydrologically linked to groundwater systems. A major concern with soil contamination is that it can and does lead to groundwater contamination. This check is designed to prevent unacceptable transfers of contaminants to groundwater to protect aquatic life. Thus, the lowest value between the Provisional guideline and the values derived using this check should be used in areas with underlying groundwater systems.

Prudent assumptions are that an aquifer underlying a remediated site may have the potential to enter surface water bodies. Therefore, the following equations are used to calculate the concentration in soil which will not cause an exceedence of groundwater concentrations above existing water quality guidelines for the protection of freshwater aquatic life. The groundwater check follows the rationale and calculation procedure recommended in *A Protocol for the Derivation of Environmental and Human Health Soil Quality Guidelines* (CCME 1996). It is based on equilibrium partitioning between solid and aqueous phases. For most nonionic organics, including trichloroethylene, the following equation is used:

$$Y = DF \times C_{wa} (K_d + \theta_m)$$

where,

Y = concentration of contaminants in soil ($\text{mg substance} \cdot \text{kg}^{-1}$ dry soil)

DF = dilution factor = 50 (default)

C_{wa} = water protection value

K_d = distribution coefficient

θ_m = soil moisture content = 0.1 (default)

The distribution coefficient (K_d) is directly related to the organic matter content and can be derived using the sorption partition coefficient (K_{oc}) and the organic fraction of dry soil (f_{oc}) as described by the following equation:

$$K_d = K_{oc} \times f_{oc}$$

A default value of 0.3% (or 0.003) is used for f_{oc} (CCME 1996). For consistency, the sorption partition coefficient should be evaluated using the octanol-water partition coefficient based on the following equation (Mackay *et al.* 1993):

$$K_{oc} = 0.41 \times K_{ow}$$

For TCE, $\log K_{ow}$ has been evaluated at 2.42 (ATSDR 1989) and a value of $0.02 \text{ mg}\cdot\text{L}^{-1}$ has been established for the protection of freshwater aquatic life in Canada.

Therefore,

$$K_{ow} = 263.03$$

$$K_{oc} = 107.84$$

$$K_d = 0.3235$$

For the protection of freshwater aquatic life:

$$Y = 50 \times 0.02 (0.3235 + 0.1) = 0.4 \text{ mg TCE}\cdot\text{kg}^{-1} \text{ dry soil.}$$

For TCE, the groundwater check indicates that for the protection of freshwater aquatic life, the level of trichloroethylene in soil should not exceed $0.4 \text{ mg TCE}\cdot\text{kg}^{-1}$.

6. DATA GAPS

This document highlights the overall lack of Canadian soil related trichloroethylene data. 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. Site specific assessments for the toxicity and detection of trichloroethylene in soil and terrestrial organisms earthworms, plants, small mammals and birds in areas contaminated with this priority pollutant are lacking.

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.

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, and on the decay products of these compounds.

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TABLES

Table 1. Physical and chemical properties of trichloroethylene

Property	Value
CAS No.	79-01-6
Physical State (@ 25°C)	Liquid (WHO 1985)
Colour	Colourless (WHO 1985)
Odour	Ethereal (WHO 1985)
Molecular Weight	131.4 g·mol ⁻¹ (CEPA 1993)
Boiling Point	86.7°C (CEPA 1993)
Water Solubility	1.1 to 1.4 g·L ⁻¹ (CEPA 1993)
Density, Specific Gravity	1.46 g·mL ⁻¹ @ 20°C (CEPA 1993)
Henry's Law Constant	0.020 atm·m ³ /mol at 20°C (ATSDR 1989) 0.011 atm·m ³ /mol at 25°C (ATSDR 1989)
Vapour Pressure	57.8 mm Hg at 20°C (CCOHS 1985) 74 mm Hg @ 25 °C (ATSDR 1989) 59 mm Hg @ 29°C (ATSDR 1989) 2.7 kPa @ 0°C (Verschuere 1983) 8.0 kPa @ 20°C (Verschuere 1983) 12.7 kPa @ 30°C (Verschuere 1983)
Log Octanol/Water Partition Coefficient (log K _{ow})	2.42 (ATSDR 1989)
log K _{oc}	2.03 - 2.66 (Garbarini and Lion 1986)
Bioconcentration Factor (log BCF)	1.20 to 3.06 (Mackay <i>et al.</i> 1993)
Half-Life: AIR WATER SOIL SEDIMENT GROUNDWATER	(Mackay <i>et al.</i> 1993) 0.24 -2.4 4320 -8640 <10 days; 4320 -8640 h; 43 days upper soil and 86 lower soil 43 days <10 days
Surface Tension	0.029 N/m @ 20 °C, liquid (Muraoka and Hirata 1988) 0.0264 N/m @ 20°C, vapour (McNeill 1979)

Table 2. Summary of existing soil and water quality criteria or guidelines for trichloroethylene.

Medium (units)	Criteria	Concentration	Reference
Soil (mg·kg ⁻¹)	Assessment Criteria	0.1	CCME 1991
	Remediation Criteria		
	Agricultural land	0.1	
	Residential/parkland	5.0	
	Commercial/industrial land	50	
	Surface soil clean-up criteria in a potable groundwater situation (soil pH 5-9):		OMEE 1994
	Agricultural	0.39	
	Residential/parkland	0.39	
	Commercial/industrial	0.39	
	Sub-surface soil clean-up criteria in a potable groundwater situation (soil pH 5-11):		
	Residential/parkland	0.39	
	Commercial/industrial	0.39	
	Surface soil clean-up criteria in a non-potable groundwater situation (soil pH 5-9):		
	Residential/parkland	0.39	
	Commercial/industrial	0.39	
	Sub-surface soil clean-up criteria in a non-potable groundwater situation (soil pH 5-11):		
	Residential/parkland	0.39	
	Commercial/industrial	0.39	
	Target value	0.001	The Netherlands HSPE 1994
	Intervention value	60	
Water (µg·mL ⁻¹)	Assessment Criteria	0.0001	CCME 1991
	Protection of Freshwater Aquatic Life (Interim)	0.02	CCREM 1987
Drinking Water (µg·mL ⁻¹)	For a drinking water contribution of 10% (Provisional)	0.07	WHO 1994
	Protection of Human Health (MAC)	0.05	HWC 1990
Groundwater (µg·mL ⁻¹)	Potable or non-potable groundwater	0.05	OMEE 1994
	Target value	0.00001	The Netherlands HSPE 1994
	Intervention value	0.5	

Table 3. Consulted plant and invertebrate toxicity studies for trichloroethylene

Organism	Effect	Endpoint ^(a)	Concentration mg·kg ⁻¹ nominal and (measured)	Test Substrate	Analytical Method	References
Lettuce (<i>Lactuca sativa</i>)	Seedling emergence	NOEC LOEC EC ₂₅ EC ₅₀	686 (16) 1372 (48) 944 (26) 1238 (37)	Artificial Soil	EPA method 8240	Environment Canada 1995
Radish (<i>Raphanus sativa</i>)	Seedling emergence	NOEC LOEC EC ₂₅ EC ₅₀	344 (9) 686 (16) 597 (14) 1525 (52)	Artificial Soil	EPA method 8240	Environment Canada 1995
Serbian spruce (<i>Picea omorica</i>)	Visible damage (NQ)	EC	4.6 µg·m ⁻³	Field experiment	—	Frank & Frank 1985
<i>Picea abies</i>	Visible damage (NQ)	EC	180 ppbv	None	—	Frank & Frank 1986
Earthworm (<i>Eisenia fetida</i>)	Mortality	LC ₅₀	(105 mg·cm ⁻²)	Filter paper	EEC	Neuhauser et al. 1985
Earthworm (<i>Eisenia fetida</i>)	Mortality	NOEC LOEC LC ₂₅ LC ₅₀	1830 (60) 3661 (159) 2212 (79) 2695 (106)	Artificial Soil	EPA method 8240	Environment Canada 1995

(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 4. Summary of environmental soil quality guidelines for trichloroethylene.

Guideline	Land Use		
	Agricultural (mg·kg ⁻¹)	Residential/ Parkland (mg·kg ⁻¹)	Commercial/ Industrial (mg·kg ⁻¹)
TEC or ECL ^(a)	Insufficient data	Insufficient data	Insufficient data
Microbial check	Insufficient data	Insufficient data	Insufficient data
SQG _{sc} ^(a)	Insufficient data	Insufficient data	Insufficient data
SQG _t	Insufficient data	Not applicable.	Not applicable.
Groundwater Check for the protection of aquatic life	0.4	0.4	0.4
Provisional SQG _e	3	3	31
CCME (1991) Interim Criteria	0.1	5	50

(a) As per the CCME (1996) protocol, the SQG_{sc} for Agricultural and Residential/Parkland land uses corresponds to the 25th percentile of the effects and no effects data distribution when using the Weight of Evidence method, while the SQG_{sc} for Commercial/Industrial land use corresponds to the 25th percentile of the effects only data distribution.

