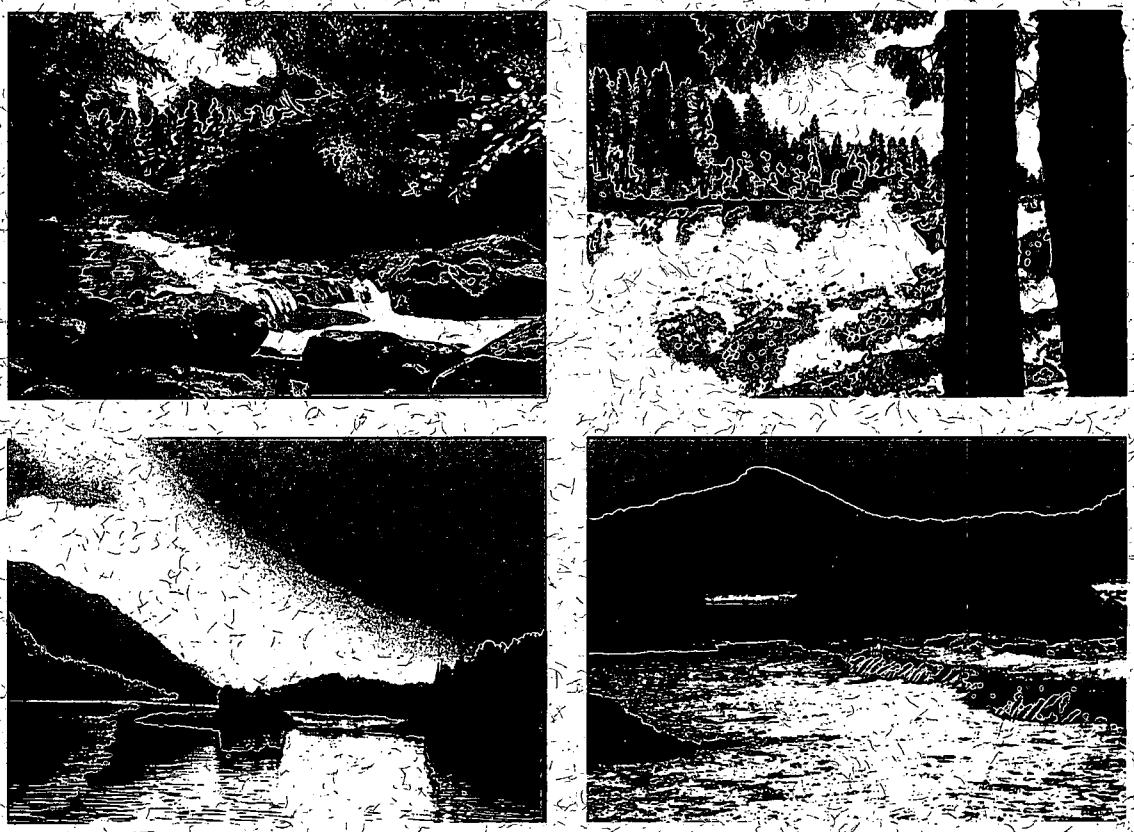
Environment Canada
Conservation and Protection

Environnement Canada
Conservation et Protection

Canadian Water Quality Guidelines for Trichloroethylene

D.R.J. Moore, S.L. Walker and R. Ansari



SCIENTIFIC SERIES NO. 183

INLAND WATERS DIRECTORATE
WATER QUALITY BRANCH
OTTAWA, ONTARIO, 1991

(Disponible en français sur demande)

Canada

Environment Canada
Conservation and Protection

Environnement Canada
Conservation et Protection

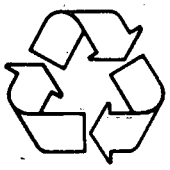
Canadian Water Quality Guidelines for Trichloroethylene

D.R.J. Moore, S.L. Walker and R. Ansari

SCIENTIFIC SERIES NO. 183

INLAND WATERS DIRECTORATE
WATER QUALITY BRANCH
OTTAWA, ONTARIO, 1991

(Disponible en français sur demande)



Printed on paper that contains recovered waste

Published by authority of
the Minister of the Environment

Cat. No. En 36-502/183E
ISBN 0-662-19123-4

Contents

Contents (cont'd)

Tables

Illustrations

Abstract

A literature review was conducted of the available information on the physical and chemical properties, environmental concentrations, environmental fate and persistence, bioaccumulation potential, and toxic effects of trichloroethylene (TCE) on freshwater and marine biota. The information is summarized in this publication. From this information, water quality guidelines are recommended for the protection of specific water uses in Canada.

Résumé

On a examiné la documentation sur les données relatives au trichloroéthylène, à ses propriétés physiques et chimiques, à ses concentrations dans l'environnement, à son devenir et sa persistance dans l'environnement, à sa capacité de bioaccumulation et aux effets de sa toxicité dans l'eau douce et le biote marin. Cette publication présente les résultats de la recherche entreprise, laquelle a permis d'adopter des recommandations sur la qualité de l'eau pour la protection de l'eau et pour ses usages particuliers au Canada.

Preface

Toxicological and environmental concerns have recently led to both reductions in usage of trichloroethylene (TCE) and placement of the compound on the Canadian Environmental Protection Act (CEPA) Priority Substances List (Canada Gazette, 1989). According to this Act, substances on this list must be assessed to determine whether they could have immediate or long-term adverse effects on the environment. The purpose of this report is to develop Canadian water quality guidelines for TCE that ensure the protection and maintenance of the five major water uses: raw water for drinking water supplies; recreation and aesthetics; freshwater and marine aquatic life; livestock watering and irrigation supplies; and industrial water supplies.

Canadian Water Quality Guidelines for Trichloroethylene

D.R.J. Moore, S.L. Walker, and R. Ansari

INTRODUCTION AND BACKGROUND

Trichloroethylene (TCE) (1,1,2-trichloroethylene) is an unsaturated, low molecular weight C_2 compound with the formula $Cl_2C{=}CHCl$ (Fig. 1). The CAS Registry Number of TCE is 79-01-06. Other common and trade names by which TCE is known include trichloroethene, acetylene-trichloride, chlorelen, chloroethylen, petzinol, triasol, and narcogen (Love and Eilers, 1982). TCE is used primarily as a degreasing solvent in the metal cleaning industries, although it has also been used as a household and industrial dry cleaning solvent, in textile manufacturing, in paint stripping, as an extractive solvent in foods, as an anaesthetic agent during some surgical procedures, and as a fumigant (U.S. EPA, 1978; Cogswell *et al.*, 1982; Bruckner *et al.*, 1989).

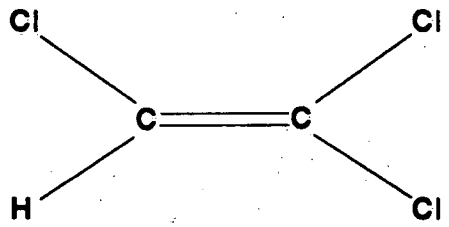


Figure 1. Structural formula for trichloroethylene.

Production

Commercially, TCE is manufactured by the chlorination of ethylene and dichloroethane. In the early 1970s, total annual production of TCE in the United States was 277 100 t (McNeill, 1979). However, by the late 1970s, this had declined to 118 000 t annually due primarily to state-imposed restrictions on TCE emissions (ATRG, 1988).

In Canada, C-I-L and Venchem, both of Shawinigan, Quebec, were the only domestic manufacturers of TCE; in 1976, the two plants had a total potential capacity of 38 000 t. In 1976, domestic production of TCE was 22 500 t, whereas imports were relatively small (200 t). Total domestic demand for TCE in 1976 was 12 700 t, of which 10 000 t were consumed by the metal cleaning industry, 2.3 t were consumed in the production of perchloroethylene, and the remainder was consumed for miscellaneous uses (400 t), export sales (6500 t), and inventory adjustments (3300 t). Since 1976, continued environmental pressures have led to decreased usage of TCE. Also, tighter equipment specifications have led to reduced emissions and greater recycling of TCE, further reducing demand. Since the closure of C-I-L's perchlor/trichlor (TCE) plant in 1985, imports have become the sole supply source. In 1988, total TCE imports were 3000 t, of which 2900 t were consumed by the metal cleaning industry. The forecast for domestic consumption of TCE in 1992 indicates further reduced demands, with consumption estimated to be 2200 t (CPI, 1988).

TCE in the Environment

TCE is not known to occur as a natural product. It has been estimated that 60% of the total world production to date has been released to the environment (U.S. EPA, 1979). TCE has been detected in air, soil, food, and human tissues (Pearson and McConnell, 1975; Bruckner *et al.*, 1989). Its detection in rivers and lakes, municipal water supplies, the sea, and aquatic biota (see Environmental Concentrations) indicates that TCE is widely distributed in the aquatic environment (U.S. EPA, 1978). TCE is present in up to 34% of water supplies tested in the United States (Conglio *et al.*, 1980; Westrick *et al.*, 1984) and is the chemical most often detected at Superfund sites (Abelson, 1990).

Review of TCE Guidelines

The U.S. EPA (1978, 1980, 1986) has prepared documents on ambient water quality with respect to TCE, but, because their minimum data base requirements were not met, no numerical limits were set. However, the U.S. EPA found that the acute freshwater lowest-observed-effect level (LOEL) occurred at 45.0 $mg{\cdot}L^{-1}$, and the chronic freshwater LOEL occurred at 21.9 $mg{\cdot}L^{-1}$. Several U.S. states have proposed or set TCE guidelines for the protection freshwater aquatic life at lower levels than those proposed by the U.S. EPA. For instance, Michigan

has set a guideline level of 0.094 $mg \cdot L^{-1}$ as the highest concentration of TCE that theoretically will produce no adverse effects on important aquatic organisms (and their progeny) exposed continuously for a lifetime (Zugger, 1989). In Canada, CCREM (1987) concluded that there were insufficient data available at the time of writing to recommend guidelines for TCE.

PHYSICAL AND CHEMICAL PROPERTIES

Properties

The physical and chemical properties of TCE are shown in Table 1. TCE is a liquid at room temperature (boiling point = 87°C) and is heavier (specific gravity 20°C/4°C = 1.47) and has a lower surface tension, (0.029 $N \cdot m^{-1}$) than water (McNeill, 1979; Muraoka and Hirata, 1988). TCE is moderately soluble in water (1100 $mg \cdot L^{-1}$) (Pearson and McConnell, 1975), has a high vapour pressure (8.0 kPa at 20°C) (Verschueren, 1983), and has a measured log 1-octanol/water partition coefficient of 2.29 (U.S. EPA, 1979). Field trials have shown that TCE does not selectively partition to aquatic sediments (Pearson and McConnell, 1975), nor is it sorbed to coarse gravel (McConnell *et al.*, 1975). However, sediments of high organic content have a high adsorptive capacity for TCE. The physical and chemical properties of TCE suggest that volatilization will be an important process in the removal of TCE from the aquatic environment, although small amounts may be found in other compartments of the aquatic environment, particularly biota and sediment.

Analytical Aspects of TCE

TCE is not included in routine monitoring of surface waters by Water Quality Branch personnel at Environment Canada. However, several provinces (e.g., Ontario, Alberta) and the National Water Quality Laboratory and National Water Research Institute in Burlington have conducted studies to determine TCE levels in selected surface waters.

A commonly employed technique for determining TCE levels in water is to use a Hewlett-Packard gas chromatograph (GC) with a 25-m fused silica capillary column and hydrogen as the carrier gas. The column, equipped with a ^{63}Ni electron capture (EC) detector, is programmed from 20°C to 80°C at $4°C \cdot min^{-1}$ with an initial 2-min hold (Kaiser and Oliver, 1976; Comba and Kaiser, 1983). Recovery of TCE-spiked samples using GC/EC techniques was found to be 51% ± 15% (Kaiser and Valdmanis, 1979) with a detection limit of 1 $ng \cdot L^{-1}$ (Kaiser *et al.*, 1983).

Table 1. Chemical and Physical Properties of Trichloroethylene

Property	Value
Physical state:	Colourless liquid[1]
Odour:	Chloroform-like[1]
Molecular weight:	131.5[1]
Melting point:	-87°C
Boiling point:	87°C
Vapour pressure:	2.7 kPa (0°C)[1] 8.0 kPa (20°C)[1] 12.7 kPa (30°C)[1]
Solubility in water:	1100 $mg \cdot L^{-1}$ (25°C)[2]
Specific gravity:	1.47 (20°C/4°C)[3]
Surface tension:	0.029 $N \cdot m^{-1}$ (20°C)[4]
Log 1-octanol/water partition coefficient	2.29[5]

[1]Verschueren (1983)
[2]Pearson and McConnell (1975)
[3]McNeill (1979)
[4]Muraoka and Hirata (1988)
[5]U.S. EPA (1979)

ENVIRONMENTAL CONCENTRATIONS

Sources

TCE can enter the aquatic environment through a number of routes, including industrial discharges, landfill leaching, septic tank leaks, accidental spills, leaky storage tanks, and disposal from individual households. Volatilization during production and use with subsequent atmospheric transport is likely to be the most significant route of entry of TCE to the aquatic environment in nonindustrial areas (U.S. EPA, 1979).

Residues

Groundwater

Low molecular weight compounds like TCE, which are volatile and have a moderate to high solubility in water, are candidates for transport to groundwater (Connor, 1984). Once present in the groundwater, TCE has the potential to persist because volatilization is an ineffective removal process from this medium (Zoeteman *et al.* 1980). In the Netherlands, TCE con-

centrations as high as 1.10 mg•L^{-1} have been detected in groundwater during routine surveys (Zoeteman *et al.* 1980); in Vero Beach, Florida, TCE concentrations as high as 8.98 mg•L^{-1} were detected after a leak from an underground storage tank (Wang *et al.*, 1985). Even after an intensive effort to decontaminate the groundwater at the Vero Beach site (by dewatering of the aquifer), TCE levels of 0.128 mg•L^{-1} were detected 32 months after the leak.

In Canada, few studies have been conducted to determine TCE levels in groundwater. Lesage *et al.* (1990) found that groundwater located near a Gloucester, Ontario, landfill had TCE levels that ranged from below the detection limit (DL = 1 µg•L^{-1}) to 0.583 mg•L^{-1}. In Ville-Mercier, Quebec, groundwater from a site near a landfill was contaminated with TCE at levels ranging from 4.40 to 7.20 mg•L^{-1} (S. Lesage, 1989, National Water Research Institute, pers. com.). In Amherst, Nova Scotia, TCE has been detected in several municipal and private wells at concentrations ranging from 5 to 84 µg•L^{-1} (N.S. DOE, 1983).

Surface Water

TCE has been detected in both fresh and marine waters from a number of sites worldwide, particularly near industrial and urban areas (U.S. EPA, 1979). The presence of trace levels of TCE in nonindustrial areas may be due to atmospheric transport (Pearson and McConnell, 1975; McConnell *et al.*, 1975). Pearson and McConnell (1975) found that levels of TCE in marine waters of Liverpool Bay, England, ranged from below the detection limit (DL = 0.01 µg•L^{-1}) to 3.6 µg•L^{-1}. Wang *et al.* (1985) found that TCE levels in the Indian River, near the Vero Beach, Florida, TCE leak, ranged from below the detection limit (DL = 0.1 µg•L^{-1}) to 9.9 µg•L^{-1}. Both of these areas were located near industrial sites that were known point sources of TCE, and thus the observed levels of TCE in these surface waters are indicative of relatively contaminated sites.

Reported levels of TCE in surface waters of Canada ranged from below the detection limit (DL = 0.001 µg•L^{-1}) to 90.0 µg•L^{-1}, with Quebec and Ontario having the highest observed levels (Table 2). In surface water samples taken from monitoring stations in the St. Lawrence River near Sorel, Lum and Kaiser (1986) found concentrations of TCE up to 90.0 µg•L^{-1}. Industrial effluents near Maitland, Montreal, and Sorel were identified as point sources of TCE discharges to the St. Lawrence River. TCE has also been found in Ontario surface water samples from the St. Clair (Kaiser and Comba, 1986), Niagara (Kaiser *et al.*, 1983), and Welland (Kaiser and Comba, 1983) rivers and in Lakes Ontario, Erie, and St. Clair. Surface water samples collected from the Oldman, Bow, Red Deer, Athabasca, and North Saskatchewan rivers in Alberta contained no detectable levels of TCE (DL = 0.2 µg•L^{-1} for 1984–86; DL = 1.0 µg•L^{-1} for 1987–88) (AEC, 1989).

Sediments

Cohen and Ryan (1985), using a multimedia compartment model, predicted that TCE concentrations in aquatic sediments would be similar to those occurring in the overlying water. Pearson and McConnell (1975) found that levels of TCE in the sediments of Liverpool Bay, England, ranged from below the detection limit (DL = 0.01 µg•kg^{-1}) to 9.9 µg•kg^{-1}, levels that were similar to those found in water samples from the bay. A similar relationship was observed in the Indian River in an area close to the Vero Beach, Florida, TCE leak, where sediment levels of TCE ranged from below the detection limit (DL = 0.5 µg•kg^{-1}) to 1.7 µg•kg^{-1} (Wang *et al.*, 1985). Prior to the discharge of TCE into the Indian River, TCE levels in the sediments were below the detection limit (Wang *et al.*, 1985).

There are few data available on levels of TCE in aquatic sediments in Canada. In a document prepared under the auspices of the Canada – Ontario Agreement Respecting Great Lakes Water Quality (COARGLWQ, 1986), TCE levels were reported in 45 of 68 sediment samples (detection limit not stated) taken in 1985 from the St. Clair River near Sarnia, Ontario. The maximum concentration detected was 110.0 mg•kg^{-1}; this value is several orders of magnitude above the level found in the relatively contaminated Liverpool Bay, England, and Vero Beach, Florida, sites. In the only sediment sample taken from an Alberta river in 1985, TCE did not occur above the detection limit (detection limit not stated) (AEC, 1989).

Biota

Neely *et al.* (1974) showed that the bioaccumulation potential of a compound is directly related to its octanol/water partition coefficient. The log of the measured 1-octanol/water partition coefficient (K_{ow}) for TCE is 2.29. This indicates that bioaccumulation is possible for TCE but not likely to the same extent as has been observed with other more nonpolar

Table 2. Trichloroethylene Levels in Canadian Surface Waters

Location	Range ($\mu g \cdot L^{-1}$)	Dection limit ($\mu g \cdot L^{-1}$)	Reference
St. Lawrence River, Que.	0.011–90.0	0.001	Lum and Kaiser, 1986
Lower Niagara River, Ont.	ND–0.011	0.001	Kaiser *et al.*, 1983
Lake Ontario, Ont.	ND–0.033	0.001	Kaiser *et al.*, 1983
Lake Erie, Ont.	0.003–0.063*	0.001	Kaiser and Valdmanis, 1979
Lake St. Clair, Ont.	ND–0.036	0.001	Kaiser and Comba, 1986
Lower St. Clair River, Ont.	ND–0.023	0.001	Kaiser and Comba, 1986
St. Clair River, Ont.	ND–42.0	1.0	COARGLWQ, 1986
Welland River, Ont.	ND–0.75	0.001	Kaiser and Comba, 1983
Oldman River, Alta.	ND	0.2(1984–86); 1.0 (1987–88)	AEC, 1989
North Saskatchewan River, Alta.	ND	0.2(1984–86); 1.0 (1987–88)	AEC, 1989
Bow River, Alta.	ND	0.2(1984–86); 1.0 (1987–88)	AEC, 1989
Red Deer River, Alta.	ND	0.2(1984–86); 1.0 (1987–88)	AEC, 1989
Athabasca River, Alta.	ND	0.2(1984–86); 1.0 (1987–88)	AEC, 1989

*Surface water and bottom water samples.

organochlorines, such as PCBs (Nimmo *et al.*, 1975; Subramanian *et al.*, 1986). Pearson and McConnell (1975) determined TCE levels in the tissues of a wide range of marine biota from different trophic levels in Liverpool Bay, England. In the livers of fish, they found that levels of TCE ranged from below the detection limit (DL < 0.1 $\mu g \cdot kg^{-1}$ wet weight) in mackerel (*Scomber scombrus*) to 30.0 $\mu g \cdot kg^{-1}$ wet weight in dab (*Limanda*). Based on their survey, Pearson and McConnell (1975) concluded that bioaccumulation of TCE was relatively minor (bioaccumulation factors < 100), and that there was no evidence of food chain biomagnification (Fig. 2). Similarly, Wang *et al.* (1985) found that estuarine oysters (*Crassostrea virginica*) in the Indian River near the Vero Beach TCE leak showed little evidence of TCE bioaccumulation. Concentrations in the oysters ranged from below the detection limit (DL = 1.0 $\mu g \cdot kg^{-1}$) to 10.8 $\mu g \cdot kg^{-1}$ wet weight. Ofstad *et al.* (1981) found that TCE levels in Norwegian fish ranged from below the detection limit (DL = 2.5–100 $\mu g \cdot kg^{-1}$, depending on fish fat content) for cod and salmon fillets to 400 $\mu g \cdot kg^{-1}$ fat weight in cod livers (species names were not given). Higher TCE levels in the latter study were associated with fish caught in waters contaminated by industrial effluents from pulp and paper mills and chemical plants.

No data were available on TCE levels in Canadian aquatic biota.

ENVIRONMENTAL FATE AND PERSISTENCE

Behaviour of TCE in the Freshwater Environment

Multimedia models based on the chemical and physical properties of TCE have been developed to help predict the environmental fate and transport be-

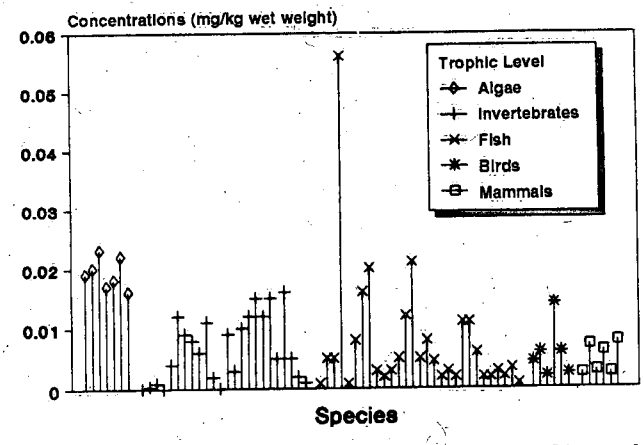


Figure 2. Trichloroethylene levels in marine biota from Liverpool Bay, England (from Pearson and McConnell, 1975). Trichloroethylene levels in the surface waters ranged from below the detection limit (DL = 0.01 $\mu g \cdot L^{-1}$) to 3.6 $\mu g \cdot L^{-1}$.

haviour of this compound (Cohen and Ryan, 1985; Mackay *et al.*, 1985; Mackay, 1987). Generally, these models predict that, because of its high volatility, TCE will be lost to the atmosphere upon release to the environment, with only minor amounts partitioning to the terrestrial and aquatic environments (Figs. 3 and 4). In the atmosphere, TCE is relatively short-lived, and a major proportion of the compound is rapidly removed through photooxidation and subsequent hydrolysis (Gay *et al.*, 1976; U.S. EPA, 1979). Some TCE will also undergo long-range atmospheric transport, and rain scavenging may deposit TCE in terrestrial and aquatic systems in areas without local sources (Mackay, 1987).

Sources of TCE to the aquatic environment include direct discharge, soil runoff, and atmospheric deposition (Lay *et al.*, 1984). In the aquatic environment, TCE is moderately soluble and tends to remain in

solution; such that little selective adsorption to the suspended or bottom sediments occurs. Volatilization is the primary removal process of TCE from the aquatic environment (Dilling *et al.*, 1975; Jensen and Rosenberg, 1975; Dilling, 1977), whereas other possible removal processes such as oxidation, hydrolysis, and microbial degradation are not considered important (Jensen and Rosenberg, 1975). The following sections describe the possible fate processes of TCE in the aquatic environment.

Volatilization

Dilling *et al.* (1975) and Dilling (1977) found that a 1.0 $mg \cdot L^{-1}$ aqueous solution of TCE in an open beaker was lost with a half-life of 17.7–23.5 min. The solution was kept moving with constant stirring at 200 rpm. However, when the solution was intermittently stirred (for 15 s every 5 min), the half-life increased to more than 90 min. These results led Dilling *et al.* (1975) to conclude that volatilization appears to be the major pathway by which TCE is lost from water. However, the results also point out the difficulties encountered in extrapolating from laboratory results to natural aquatic ecosystems, where TCE concentrations would be much lower and parameters such as surface agitation would be highly variable. When components such as clay, limestone, sand, or peat moss were added to the open test container, TCE volatilization rates were only slightly reduced and did not change by more than a factor of 2.

Jensen and Rosenberg (1975) found that TCE concentrations declined by 80% after 8 d in a partly open aquarium kept in the light (half-life = 3.44 d). When the aquarium was kept closed in the light or dark, less than 50% of the initial TCE concentration disappeared after 8 d. These results indicated that volatilization of TCE proceeds much more rapidly than photolysis, hydrolysis, or oxidation in aquatic systems.

No field studies were found in the literature that measured TCE volatilization rates in isolation from other possible fate processes in natural aquatic systems. Lay *et al.* (1984) found that the half-life of TCE added to a small pond was 2.7 d. Although no evidence was found that the bottom sediments acted as a sink for TCE, it is possible that fate processes other than volatilization contributed to the relatively rapid disappearance of TCE from the pond.

Once TCE enters the atmosphere, the carbon–carbon double bond is attacked by hydroxyl radicals, resulting in the formation of dichloroacetyl chloride and phosgene as the primary products (Gay *et al.*, 1976; U.S. EPA, 1979). These compounds are then rapidly hydrolyzed, resulting in the formation of HCl, CO, CO_2, and a carboxylic acid (Gay *et al.*, 1976)

Photolysis, Oxidation, and Hydrolysis

Wang and Tan (1987) found that TCE concentrations were reduced by 96.8% when irradiated in a platinum-catalyzed water photolysis system for 12 h, with ethane as the major degradation product. When zinc was added to the system, the degradation rate was enhanced. However, the environmental significance of this reaction is not likely to be important, because this reaction could not occur under natural conditions. Without the platinum catalyst, Wang and Tan (1987) found that no photolysis of TCE occurred.

Dilling *et al.* (1975) found possible evidence of photooxidation of TCE in aerated water in the presence of sunlight. The amount of TCE remaining in aerated water stored in the dark for 12 months in a closed system was approximately 20% greater than the amount remaining in a duplicate system exposed to sunlight. The more rapid removal of TCE in sunlight was attributed to free radical oxidation. However, the observed experimental half-lives of TCE in the dark (resulting from oxidation and hydrolysis) and in sunlight (resulting from photooxidation and hydrolysis) were 10.7 and 10.1 months, respectively. Similarly, Jensen and Rosenberg (1975) found no significant difference in TCE removal from closed aquaria exposed to either daylight or darkness for 8 d, thus indicating that no direct photolysis occurred. Further, the TCE removal rates from these treatments (5% over 8 d) were well below the rate found when the aquaria were open (80% over 8 d). These studies indicate that photolysis, oxidation, and hydrolysis are not important processes for the removal of TCE from the aqueous environment when compared with volatilization.

Sorption

Dilling *et al.* (1975) explored TCE removal by sorption to clay, limestone, and peat moss in three closed-system experiments. They found that bentolite clay at a concentration of 375 $mg \cdot L^{-1}$ adsorbed 10% of the TCE in solution in 10 min, whereas a concentration of 750 $mg \cdot L^{-1}$ of clay adsorbed 22% of the TCE in solution after 30 min. Dolomitic limestone at a concentration of 500 $mg \cdot L^{-1}$ caused a 50% depletion

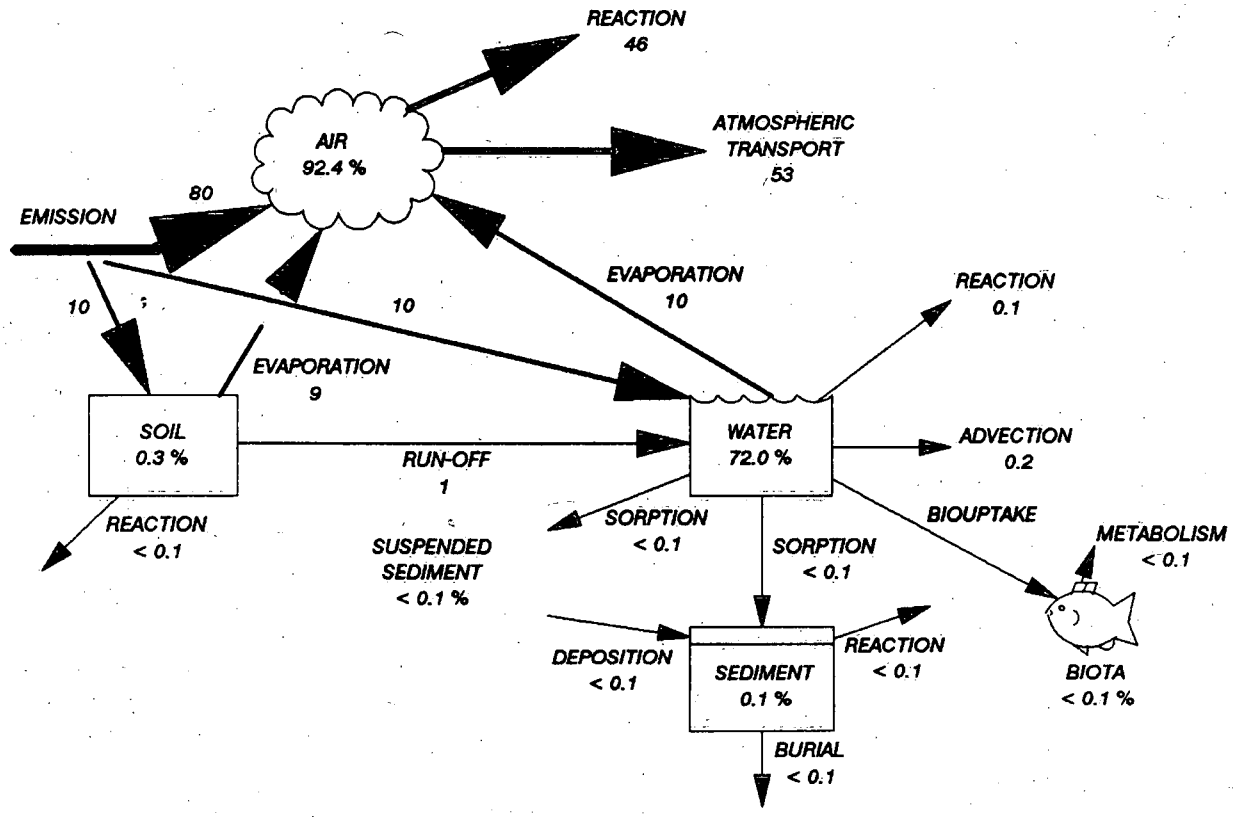


Figure 3. Environmental fate diagram for trichloroethylene (from Mackay, 1987).

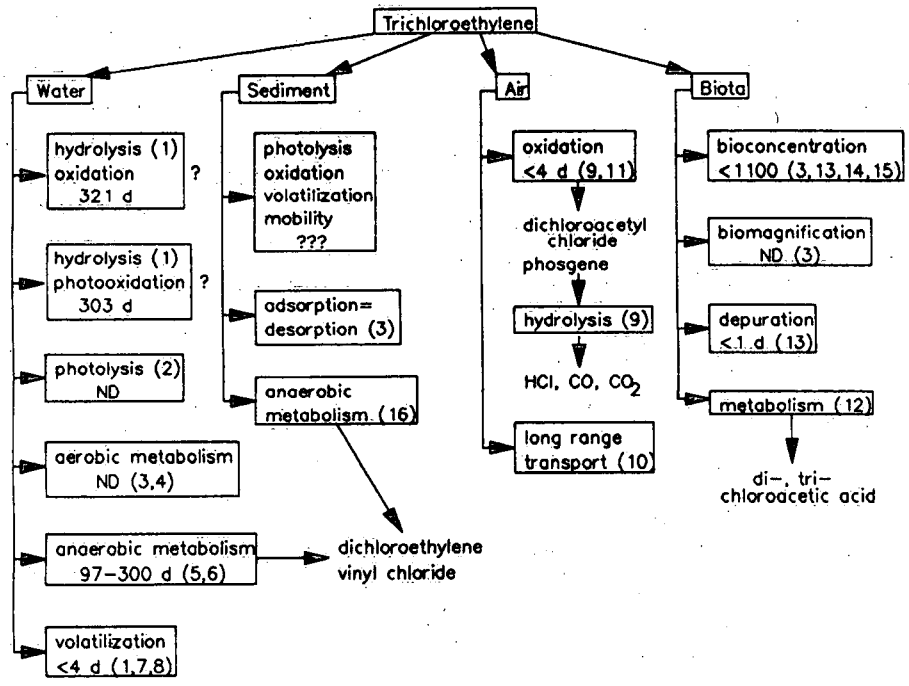


Figure 4. Fate processes for trichloroethylene.

of TCE in approximately 20 min and a 90% depletion in approximately 70 min. Peat moss (500 $mg \cdot L^{-1}$), added to simulate a high content of organic material in water, caused a 40% TCE depletion in 10 min, after which no further depletion was measured.

In a field study by Pearson and McConnell (1975) in Liverpool Bay, England, no evidence for selective concentration of TCE in marine sediments was found, indicating that adsorption in natural marine ecosystems is not likely to be an important removal process when compared with other removal mechanisms.

Biodegradation

A few organisms have been shown to degrade TCE by reductive dechlorination under anaerobic conditions conducive to methanogenesis (Kleopfer *et al.*, 1985; Parsons *et al.*, 1985; Barrio-Lage *et al.*, 1988). The half-life for microbial degradation of TCE was found to be 97.1–105.9 d in an anaerobic microcosm experiment (Barrio-Lage *et al.*, 1988) and 300 d in a field study of spiked groundwater in Palo Alto, near San Francisco (Roberts *et al.*, 1982). In the former study, methane added as a nutrient to the microcosms decreased the TCE half-life to 39.0 d. These results indicated that biodegradation of TCE in groundwater can occur, but at rates much slower than would occur where volatilization is possible. Baek and Jaffe (1989) showed that a symbiotic nonmethanogenic fermentation process during methanogenesis enhanced the anaerobic degradation of trichloroethylene in mixed methanogenic cultures. TCE degradation products have been found in groundwater of the Gloucester, Ontario, and the Ville-Mercier, Quebec, landfill sites (Pakdel *et al.*, 1989; Lesage *et al.*, 1990). Degradation has also been shown to occur in anaerobic landfill leachate. Of concern, however, are the products of TCE anaerobic biodegradation. These include the toxic substances chloroethane, dichloroethylene, and vinyl chloride, which is carcinogenic (Bouwer *et al.*, 1981; Kleopfer *et al.*, 1985; Parsons *et al.*, 1985; Vogel and McCarty, 1985; Baek and Jaffe, 1989; Lesage *et al.*, 1989, 1990; Pakdel *et al.*, 1989).

Microbial degradation of TCE under aerobic conditions has not been demonstrated (Pearson and McConnell, 1975; Bouwer *et al.*, 1981), except in the presence of growth substrates such as phenol, toluene, and cresol (Nelson *et al.*, 1987), methane and methanol (Little *et al.*, 1988; Berwanger and Barker, 1988; Strandberg *et al.*, 1989), or a natural gas mixture (Wilson and Wilson, 1985). These results may be useful for groundwater reclamation projects, but are likely of little environmental significance.

There is limited evidence from mammalian studies indicating that TCE can be metabolized by higher organisms to produce di- and trichloroacetic acids (Daniel, 1963). Leibman (1965) reported that TCE is converted to chloral by liver microcosms and provided evidence that the reaction was catalyzed by cytochrome P-450. Chloral, trichloroethanol, and trichloroacetic acid are the major metabolites of TCE detected *in vivo* (Defalque, 1961; Kimmerle and Eben, 1973).

BIOACCUMULATION

Geyer *et al.* (1984) found that the green alga *Chlorella fusca* var. *vacuolata*, when exposed to a TCE concentration of 50 $\mu g \cdot L^{-1}$ for 1 d, had a bioconcentration factor (BCF) of 1160. No other studies of bioaccumulation in plants or phytoplankton were found in the literature.

Barrows *et al.* (1980) found that bluegill sunfish (*Lepomis macrochirus*) exposed to TCE at a mean concentration of 3.4 $\mu g \cdot L^{-1}$ for a period of 14 d had a bioconcentration factor of 17. In this study, TCE levels had equilibrated in the fish tissues by the end of the 14-d exposure period, and therefore longer exposures would not have led to a higher bioconcentration factor. When the fish were moved to clean aquaria, TCE depuration was rapid, with a half-life of less than 1 d. In the only other study found in the literature of TCE bioaccumulation in fish, Freitag *et al.* (1985) found that golden ide (*Leucociscus idus melanotus*) had a bioconcentration factor of 90 after a 3-d exposure to TCE. It is not clear from this study whether TCE bioconcentration reached equilibrium in the fish during the 3-d exposure period.

Pearson and McConnell (1975) determined the level of TCE in tissues of a wide range of marine organisms in Liverpool Bay, England. Species selected were chosen to represent different trophic levels. The evidence suggested that there was little bioaccumulation by plankton (BCF < 10), and that, in invertebrates, fish, birds, and mammals, bioaccumulation was highly variable (BCF < 2–100) (see Fig. 2). There was no evidence of biomagnification of TCE in the aquatic food webs of Liverpool Bay, England.

TOXICITY TO AQUATIC BIOTA

Because standard protocols for toxicity testing may become outdated or are not always available or

followed, a great deal of variability exists in the quality of published toxicity data. To ensure a consistent scientific evaluation for each chlorinated ethane compound, the data used in deriving a guideline must meet certain criteria as outlined in CCME (1991). These criteria include information on test conditions/design (e.g., flow-through, static), test concentrations, temperature, water hardness, pH, experimental design (controls, number of replicates), and a description of the statistics used in evaluating the data. Each study is evaluated based on the above information and ranked as primary, secondary, or unacceptable (see CCME 1991, for a detailed description of the ranking criteria). All data included in the minimum data set must be primary in order for full guideline derivation to proceed. For interim guideline derivation, primary or secondary data may be used. Toxicity data that do not meet the criteria of primary or secondary data are unacceptable and cannot be used in either derivation procedure.

There are a large number of studies that have considered the acute toxicity of TCE in freshwater and marine biota (Fig. 5) (Tables B-1 and B-2). Fewer studies have examined the chronic toxicity of TCE in aquatic biota (Table B-3). Similarly, few studies have considered the effects of TCE on aquatic communities in microcosm or field studies (Tables B-1 and B-2). The purpose of this section is to describe the toxic effects of TCE on aquatic biota.

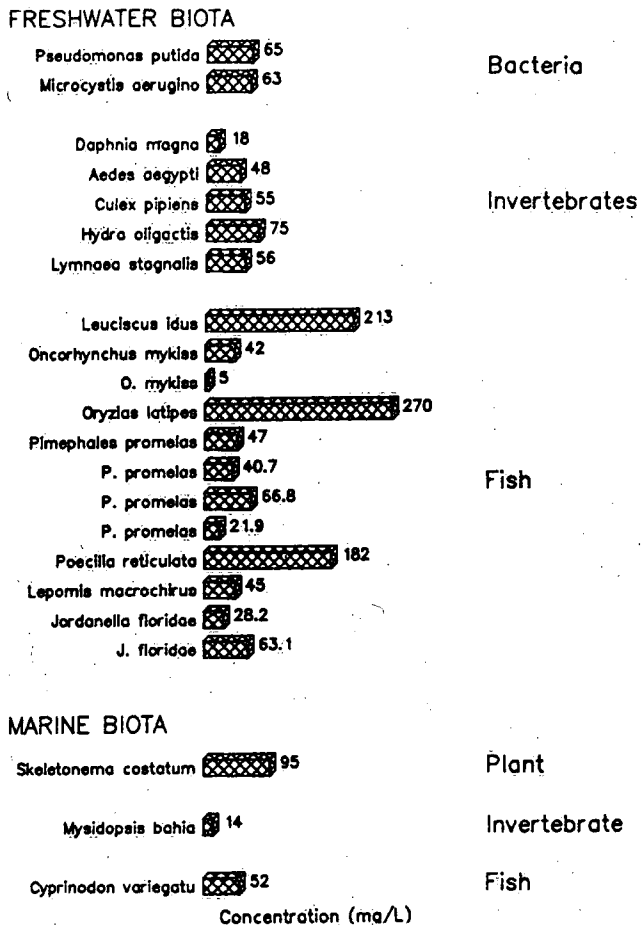


Figure 5. Observed significant responses in aquatic biota after acute exposures to TCE.

Acute Toxicity

Bacteria

The TCE toxic threshold (the concentration that causes initial inhibition of cell multiplication) for the bacterium *Pseudomonas putida* was 65.0 mg•L^{-1} after a 6-h exposure, and for the cyanobacterium *Microcystis aeruginosa*, the toxic threshold was 63.0 mg•L^{-1} after a 192-h exposure (Bringmann and Kuhn, 1980; Slooff *et al.*, 1983a). Both tests, ranked as secondary, were conducted under static conditions and with the reported TCE levels in the growth medium based on nominal (unmeasured) concentrations. The marine bacterium *Photobacterium phosphoreum*, which has also been used to test for toxicity in fresh waters, had a 50% light inhibition at a concentration of 120 mg•L^{-1} after a 15-min exposure; however, the study was deemed unacceptable because of insufficient reporting of test conditions (Kaiser and Ribo, 1988).

Phytoplankton

One marine species, the diatom *Skeletonema costatum*, was observed to have a 50% reduction in cell number after a 96-h exposure to TCE at a mean measured concentration of 95.0 mg•L^{-1} in a static test ranked as secondary (Ward *et al.*, 1986). In a static, unmeasured test with the unicellular marine alga *Phaeodactylum tricornutum*, a nominal TCE concentration of 8.0 mg•L^{-1} caused a 50% inhibition of ^{14}C uptake during photosynthesis (Pearson and McConnell, 1975). However, this study was deemed unacceptable because experimental conditions were not reported. In the only freshwater single-species study found, *Selenastrum capricornutum* was reported to have a no-observed-effect concentration (NOEC) for growth rate of 175 mg•L^{-1} following a 96-h exposure to TCE in a static test ranked as secondary because concentrations were not measured (Slooff *et al.*, 1983a).

No inhibitory effect on ^{14}C uptake during photosynthesis was observed in a natural estuarine phytoplankton assemblage during a 48-h exposure to

0.5, 1.0, and 2.0 mg•L^{-1} TCE in a flow-through test ranked as secondary because concentrations were not measured (Erickson and Hawkins, 1980). In a mixed culture containing the marine diatom *Thalassiosira pseudonana* and the marine alga *Dunaliella tertiolecta*, no effect on community composition was observed after a 96-h exposure to TCE at the two test concentrations of 0.05 and 0.10 mg•L^{-1} in a static test (Biggs *et al.*, 1979). However, this study was deemed unacceptable because experimental conditions were not reported.

Invertebrates

Several acute toxicity tests have been performed to determine the effects of TCE on *Daphnia* spp. *Daphnia magna* (4–6 d old) had a 48-h EC_{50} (immobilization) of 7.76 mg•L^{-1} TCE in a static, unmeasured test ranked as secondary (Abernethy *et al.*, 1986). In a static, unmeasured test ranked as secondary, young *D. magna* (<24 h old) had a 48-h LC_{50} of 18.0 mg•L^{-1} and a 48-h NOEC of 2.2 mg•L^{-1} TCE (LeBlanc, 1980). In studies from three laboratories with a similar experimental protocol, the corresponding 48-h LC_{50} was higher and ranged from 42.0 to 97.0 mg•L^{-1} TCE (Canton and Adema, 1978). The latter studies also found that young *D. pulex* (<24 h old) had a 48-h LC_{50} of 45.0 mg•L^{-1} TCE, and 11-d-old *D. cucullata* had a 48-h LC_{50} of 57.0 mg•L^{-1} TCE. These studies were deemed unacceptable because experimental conditions were not reported.

Invertebrate species from widely divergent taxonomic groups such as Oligochaeta, Diptera, Gastropoda, Amphipoda, and Ephemenoptera have also been tested for their response to short-term exposures of TCE (Pearson and McConnell, 1975; Slooff, 1983; Slooff *et al.*, 1983a; Ward *et al.*, 1986). The results from these tests suggest that differences in species' sensitivity to TCE may exist. For example, susceptibility to TCE ranged from a 96-h LC_{50} of 14.0 mg•L^{-1} for the marine mysid shrimp *Mysidopsis bahia* in a static, measured test ranked as secondary (Ward *et al.*, 1986) to a 48-h LC_{50} of 132.0 mg•L^{-1} for the freshwater oligochaete worms *Tubifex* and *Limnodrilus* in a static, unmeasured test deemed unacceptable because experimental conditions were not reported in sufficient detail (Slooff, 1983).

In a static test, Slooff *et al.* (1983b) found that the freshwater mussel *Dreissena polymorpha* experienced a significantly increased valve closure rate at a mean nominal TCE concentration of 9.7 mg•L^{-1}. The ecological significance of this response was not discussed by Slooff *et al.* (1983b). This study was deemed unacceptable because there was insufficient reporting of experimental conditions.

Fish

Two studies have reported LC_{50} values for TCE exposure of marine fish species. In the first study, sheepshead minnow (*Cyprinodon variegatus*) were found by Ward *et al.* (1986) to have a 96-h LC_{50} of 52.0 mg•L^{-1} in a static, measured test ranked as secondary. Pearson and McConnell (1975) found that *Limanda* had a 96-h LC_{50} of 16.0 mg•L^{-1} TCE in a flow-through, measured test. However, this latter study was considered unacceptable because test conditions were not reported in sufficient detail.

In a flow-through test, Alexander *et al.* (1978) found that fathead minnow (*Pimephales promelas*) had a 96-h LC_{50} at a measured concentration of 40.7 mg•L^{-1} TCE. Similar test conditions (flow-through, measured) were found to yield an LC_{50} of 28.2 mg•L^{-1} TCE for juvenile flagfish (*Jordanella floridae*) (ATRG, 1988). Both of these studies were ranked as primary. Slooff *et al.* (1983a) found that rainbow trout (*Oncorhynchus mykiss*) experienced 50% mortality after a 96-h exposure to a nominal concentration of 42.0 mg•L^{-1} TCE in a static test ranked as secondary because concentrations were not measured. The no-observed-lethality concentration (NOLC) was 33.0 mg•L^{-1}. In a sublethality test by Slooff (1979), *O. mykiss* was found to have a significantly increased respiration rate after a 24-h exposure to 5.0 mg•L^{-1} TCE in a flow-through test also ranked as secondary because concentrations were not measured. The ecological significance of this response and the length of time in which the response was elevated were not discussed by Slooff (1979). Other reported LC_{50} values range from 45.0 mg•L^{-1} for young-of-the-year bluegill (*Lepomis macrochirus*) in a 96-h static, unmeasured test (Buccafusco *et al.*, 1981) to 270.0 mg•L^{-1} TCE for medaka (*Oryzias latipes*) in a 48-h static, unmeasured test (Slooff *et al.*, 1983a). Both of these tests were ranked as secondary because concentrations were not measured.

Amphibians

In 48-h static, unmeasured tests deemed unacceptable because insufficient information on experimental conditions was reported, the clawed toad (*Xenopus laevis*) and the Mexican axolotl (*Ambystoma mexicanum*) experienced 50% mortality

at 45.0 and 48.0 mg•L^{-1} TCE, respectively (Slooff and Baerselman, 1980). In this study, *X. laevis* and A. mexicanum had NOLC values of 41.0 and 29.0 mg•L^{-1} TCE, respectively (Slooff *et al.*, 1983a).

Chronic Toxicity

In a primary ranked study, the Aquatic Toxicity Research Group (ATRG, 1988) determined the chronic toxicity of TCE to American flagfish (*J. floridae*) and brook trout (*Salvelinus fontinalis*) in flow-through life-cycle tests employing four biological end points: embryo survival (egg hatchability), larval survival, fry survival, and fry growth. TCE significantly reduced 10-d larval survival and 28-d fry survival of flagfish at the highest concentrations tested—21.2 mg•L^{-1} and 20.9 mg•L^{-1}, respectively. However, these concentrations had no effect on embryo survival or fry growth. The NOECs for 10-d survival and 28-d fry survival were 5.76 and 10.6 mg•L^{-1} TCE, respectively. The data reported for brook trout embryo survival, survival of swim-up larvae, and 120-d fry survival did not indicate a consistent dose/response relationship and are therefore inconclusive. However, there was a consistent relationship between TCE concentration and fry growth; at the lowest concentration tested, 0.21 mg•$^{-1}$ TCE, there was a significant 5% decrease in 120-d fry weight. At TCE concentrations of 0.68, 1.96, 4.31, and 11.3 mg•L^{-1} TCE, 120-d fry weights decreased by 5%, 17%, 25%, and 44%, respectively.

Loekle *et al.*, (1983) found that at nominal concentrations of 1.0 and 5.0 mg•L^{-1} TCE, black molly (*Poecilia sphenops*) exhibited reduced growth rates and survival time during a 60-d exposure in a static test in which the test chemical was renewed every 14 d. However, these data are considered unacceptable because the tests were not replicated, critical test conditions were not stated (e.g., temperature and pH), and a static long-term test is not applicable with highly volatile chemicals such as TCE (ASTM, 1988).

In a field experiment on a natural pond community ranked as a secondary study, Lay *et al.* (1984) found that at a nominal concentration of 25.0 mg•L^{-1} TCE, *Daphnia* abundance and phytoplankton richness and abundance were significantly reduced at the conclusion of the 43-d observation period. These changes in community structure persisted to the end of the observation period despite the fact that 50% of the TCE had disappeared after 2.7 d and 98% had disappeared after 15 d. This study indicated the potential for long-term impacts on aquatic communities after accidental spills of TCE.

Carcinogenicity, Mutagenicity, and Teratogenicity

After reviewing the literature, Bull (1985) concluded that TCE is a probable carcinogen and mutagen that may pose a threat to humans in cases where drinking water is obtained from contaminated groundwater supplies. TCE has been shown to induce cancer in mice and rats (National Cancer Institute, 1976; National Toxicology Program, 1983, 1987; Fukuda *et al.*, 1983; Maltoni *et al.*, 1988). However, studies by other researchers did not show a carcinogenic response to TCE (Van Duuren *et al.*, 1979; Henschler *et al.*, 1984). Connor (1984) calculated that an additional 2.1–2.4 x 10^{-6} cancers in humans would result from a lifetime consumption of groundwater contaminated with 5.7–6.5 µg•L^{-1} TCE. The U.S. EPA (1978) has estimated that TCE concentrations of 21, 2.1, and 0.21 µg•L^{-1} TCE in water would result in additional human lifetime cancer risks of 10^{-5}, 10^{-6}, and 10^{-7}, respectively. No study has been found that documents the potential carcinogenic, mutagenic, or teratogenic effects of TCE on aquatic biota, and thus risk estimates cannot yet be determined.

SUMMARY AND RECOMMENDED GUIDELINES

Following for extensive evaluation of the published literature on trichloroethylene, Canadian water quality guidelines were derived (Table 3). The rationales for the development of the recommended guidelines are summarized below.

Table 3. Recommended Water Quality Guidelines for Trichloroethylene

Uses	Guidelines
Raw water for drinking water supplies	0.05 mg•L^{-1}*
Recreation and aesthetics	Insufficient data
Aquatic life	
Freshwater	0.02 mg•L^{-1}†
Marine	Insufficient data
Agriculture	
Livestock watering	0.05 mg•L^{-1}‡
Irrigation	Insufficient data
Industrial	Insufficient data

*Proposed by the Federal-Provincial Subcommittee on Drinking Water (Health and Welfare Canada, 1989).

†Interim guideline.

‡Canadian drinking water guideline (Health and Welfare Canada, 1989) adopted as interim Canadian livestock watering guideline.

Raw Water Sources for Drinking Water

The Federal–Provincial Subcommittee on Drinking Water has proposed a TCE guideline of 0.05 mg•L^{-1} for this water use (Health and Welfare Canada, 1989).

If, after 1 year, no evidence is presented that questions the suitability of the proposed value, it will be adopted as the guideline.

Recreation and Aesthetics

The odour threshold of TCE in water is 10 $mg \cdot L^{-1}$ (Verschueren, 1983). No thresholds for taste or fish tainting were found in the literature. Without these data, there is insufficient information to recommend a Canadian water quality guideline for recreational water to protect this water use.

Aquatic Life

Freshwater Aquatic Life

Figure 6 indicates that the minimum data requirements to derive a water quality guideline to protect freshwater aquatic life were not met for TCE. In particular, no primary studies were available for freshwater invertebrates or plants. However, the minimum data requirements for an interim guideline were met, and thus an interim guideline can be derived from the available litrature (CCME,1991).

The chronic toxicity data indicate that the most sensitive freshwater organism was brook trout (*Salvelinus fontinalis*), which experienced a significant 5% decrease in 120-d fry weight (LOEL) after exposure to 0.21 $mg \cdot L^{-1}$ TCE in a flow-through test (ATRG, 1988). An interim Canadian water quality guideline of 0.02 $mg \cdot L^{-1}$ TCE is recommended for the protection and maintenance of freshwater aquatic life. This level was derived by applying a safety factor of 1 order of magnitude, as specified in CCEM (1991), to the lowest chronic value (brook trout, ATRG, 1988) resulting from long-term exposure.

Marine Aquatic Life

A guideline or interim guideline for the protection and maintenance of marine aquatic life could not be calculated because there were no primary studies available for marine fish, invertebrate, or plant species, and because only one fish and one invertebrate acute study were available for marine biota (Ward *et al.*, 1986) (Fig. 7). At least two studies on both invertebrate and fish species are required as a minimum data base before an interim guideline derivation can proceed (CCEM, 1991).

Agriculture

Livestock Watering

The U.S. EPA (1978) determined that the major route for exposure of terrestrial mammals to TCE was by inhalation. No data could be found regarding the toxicity of TCE to domestic livestock. In the absence of such data for other chemicals, Canadian drinking water quality guidelines are adopted as interim livestock watering guidelines as a means of providing a margin of safety for livestock and preventing unacceptable residues in animal products. The TCE drinking water quality guideline of 0.05 $mg \cdot L^{-1}$ proposed by Health and Welfare Canada (1989) is recommended as an interim Canadian livestock watering guideline to protect this water use.

Irrigation

No studies were found in the literature that documented the effects of TCE on terrestrial macrophytes including crops. Therefore, it was not possible to calculate a water quality guideline for TCE for this water use.

Industrial Water Supplies

At present the necessary information is lacking to set water quality guidelines for most chemical compounds in industrial water supplies. A survey of industry water quality needs is being conducted; upon completion, it may be possible to set guidelines for many compounds, including TCE, to protect this water use.

DATA GAPS

Several research areas require further attention in order to reduce the level of uncertainty with regard to TCE toxicity and to allow development of TCE guidelines for all water uses. At present, there are insufficient data documenting TCE levels in Canadian groundwaters, sediments, and biota. These data are required for two purposes: to assess the potential hazard TCE currently poses to Canadian aquatic environments, and to determine if the environmental fate processes predicted for TCE in laboratory studies are accurately reflected by measured TCE levels in the environment. There is also a shortage of studies that have determined the bioaccumulation potential and depuration rates of TCE in aquatic biota. Thus

Water Use: Protection of Freshwater Aquatic Life

Compound: Trichloroethylene (TCE)

Canadian Water Quality Guideline Requirements: Minimum Toxicity Data Set

Aquatic Biota	Number of Studies Required	Primary	Resident in North America	Additional Requirements*	Reference
Fish	1.	x	x	COLD, CHRONIC	ATRG 1988
	2.	x	x	WARM, CHRONIC	ATRG 1988
	3.	x	x		Alexander et al. 1978
Invertebrates	1.	—	—		—
	2.	—	—		—
Plants	1.	—	—		—

* Fish: (i) at least one cold- and one warmwater species are required (COLD, WARM)
(ii) at least two chronic (partial or full lifecycle) studies are required (CHRONIC)

Invertebrates: (i) at least two chronic (partial or full lifecycle) studies are required (CHRONIC)
(ii) at least two invertebrate classes must be represented (CL 1, CL 2)
(iii) at least one species must be planktonic (PLK)

Scientifically Justified Exemptions to Above Requirements: Yes__ No x

Minimum Toxicity Data Set Requirements Met: Yes__ No x If no, go to interim guideline section

Canadian Water Quality Guideline Requirements: Minimum Environmental Fate Data Set

(1) Is the mobility of the compound and the compartments of the aquatic environment in which it is likely to be found known? Yes x No __

(2) Are the kinds of chemical and biological reactions that take place during transport and after deposition known? Yes x No __

(3) Are the eventual chemical metabolites known? Yes x No __

(4) Is the persistence of the compound in water, sediments and biota known? Yes x No __

If the answer is no to any of the above, go to interim guideline section

Canadian Water Quality Interim Guideline Requirements:

(1) Are there at least two acute and/or chronic studies for fish, and for invertebrates? Yes x No __

(2) Is one fish species a coldwater species resident in North America? Yes x No __

(3) Are there two invertebrate species from different classes and is one species planktonic and resident in North America. Yes x No __

If no to any of the above, than an interim guideline cannot be calculated. Please note that primary or secondary studies may be used for interim guideline derivation.

Figure 6. The minimum data set work sheet for the derivation of a Canadian water quality guideline for TCE to protect freshwater aquatic life.

Water Use: Protection of Marine Aquatic Life

Compound: Trichloroethylene (TCE)

Canadian Water Quality Guideline Requirements: Minimum Toxicity Data Set

Aquatic Biota	Number of Studies Required	Primary	Temperate Species	Chronic Study	Two Classes Represented	Reference
Fish	1.	—	—	—		—
	2.	—	—	—		—
	3.	—	—			—
Invertebrates	1.	—	—	—	—	—
	2.	—	—	—		—
Plants	1.	—	—			—

Scientifically Justified Exemptions to Above Requirements: Yes___ No _x_

Minimum Toxicity Data Set Requirements Met: Yes___ No _x_ If no, go to interim guideline section

Canadian Water Quality Guideline Requirements: Minimum Environmental Fate Data Set

(1) Is the mobility of the compound and the compartments of the aquatic environment in which it is likely to be found known? Yes_x_ No ___

(2) Are the kinds of chemical and biological reactions that take place during transport and after deposition known? Yes_x_ No ___

(3) Are the eventual chemical metabolites known? Yes_x_ No ___

(4) Is the persistence of the compound in water, sediments and biota known? Yes_x_ No ___

If the answer is no to any of the above, go to interim guideline section

---

Canadian Water Quality Interim Guideline Requirements:

(1) Are there at least two acute and/or chronic studies for marine fish, and for marine invertebrates? Yes___ No _x_

(2) Is one fish species a temperate species? Yes_x_ No ___

(3) Are the two invertebrate species from different classes and is one of the species temperate? Yes___ No _x_

If no to any of the above, than an interim guideline cannot be calculated. Primary or secondary studies may be used for the interim guideline data requirements.

Figure 7. The minimum data set work sheet for the derivation of a Canadian water quality guideline for TCE to protect marine aquatic life.

far, only one species of phytoplankton and two species of fish have been studied with regard to TCE bioaccumulation potential. No bioaccumulation studies have been conducted with invertebrate species. Moreover, only one unpublished study (ATRG, 1988) has been conducted to determine the effects of long-term TCE exposure on aquatic biota. In order to upgrade the freshwater aquatic life TCE interim guideline to guideline status, at least two primary studies examining the response (preferably a life-cycle measurement) of an invertebrate species to chronic exposures of TCE must be conducted. To derive a marine aquatic life guideline, the following TCE toxicity studies are required: (i) one primary acute exposure study on a marine fish species; (ii) two primary chronic exposure studies (nonlethal end points) on marine fish species, each of which includes the early life stages of a temperate species; (iii) two chronic exposure studies (nonlethal end point) on marine invertebrate species, which include the early life stages of a temperate species; and (iv) one primary toxicity study on a temperate plant or algal species. An interim marine aquatic life guideline could be derived with one additional acute exposure study on both a marine invertebrate and a marine fish species.

Although further TCE toxicity information is desirable for livestock, it is likely that a water quality guideline for this water use is unnecessary because water is not an important TCE exposure route for livestock. There is no information available to assess the tolerance of crops to TCE in irrigation water. Given that TCE is volatile and nonpersistent, it is unlikely that this compound could pose a toxic threat to Canadian crops. However, further fate studies to determine TCE persistence in crops and soil and toxicity studies on crops are both desirable and necessary for the calculation of a guideline for this water use.

ACKNOWLEDGMENTS

The authors gratefully acknowledge the helpful comments and suggestions of the members of the Canadian Council of Ministers of the Environment (CCME) Task Force on Water Quality Guidelines. The critical reviews of M.C. Taylor, R.A. Kent, M.P. Wong, R.C. Pierce, and B.D. Pauli of the Water Quality Branch, K.L.E. Kaiser and S. Lesage of the National Water Research Institute, B. Elliott of the Commerical Chemicals Branch, and L.E. Fiddler of MacDonald Environmental Sciences Ltd. greatly improved this manuscript.

REFERENCES

Abelson, P.H. 1990. Volatile contaminants of drinking water. Science, 247: 141.

Abernethy, S., A.M. Bobra, W.Y. Shiu, P.G. Wells, and D. Mackay. 1986. Acute lethal toxicity of hydrocarbons and chlorinated hydrocarbons to two planktonic crustaceans: The key role of organism–water partitioning. Aquat. Toxicol. 8: 163–174.

AEC (Alberta Environment Centre). 1989. Trichloroethylene data from Oldman, Bow, Red Deer, Athabasca, Peace and North Saskatchewan River basins (1984–1988) Unpublished report, Alberta Environmental Protection Services.

Alexander, H.C., W.M. McCarty, and E.A. Bartlett. 1978. Toxicity of perchloroethylene, trichloroethylene, 1,1,1-trichloroethane, and methylene chloride to fathead minnows. Bull. Environ. Contam. Toxicol., 20: 344–352.

ASTM. 1988. Standard practice for conducting acute toxicity tests with fishes, macroinvertebrates, and amphibians. E 729-88. Philadelphia, Pa. American Society for Testing and Materials.

ATRG. 1988. Aquatic toxicity of multiple organic compounds. II. Chlorinated ethanes and chlorinated ethylenes. A summary report prepared for the Ontario Ministry of the Environment by the Aquatic Toxicity Research Group, Lakehead University, Thunder Bay, Ontario.

Baek, N.H., and P.R. Jaffe. 1989. The degradation of trichloroethylene in mixed methanogenic cultures. J. Environ. Qual., 18: 515–518.

Barrio-Lage, G.A., F.Z. Parsons, and P.A. Lorenzo. 1988. Inhibition and stimulation of trichloroethylene biodegradation in microaerophilic microcosms. Environ. Toxicol. Chem., 7: 889–895.

Barrows, M.E., S.R. Petrocelli, K.J. Macek, and J.J. Carroll. 1980. Bioconcentration and elimination of selected water pollutants by bluegill sunfish (Lepomis macrochirus). *In* Dynamics, Exposure and Hazard Assessment of Toxic Chemicals, ed. R. Haque. Ann Arbor, Michigan: Ann Arbor Science Publ.

Berwanger, D.J., and J.F. Barker. 1988. Aerobic biodegradation of aromatic and chlorinated hydrocarbons commonly detected in landfill leachates. Water Pollut. Res. J. Can., 23: 460–474.

Biggs, D.C., R.G. Rowland, and C.F. Wurster. 1979. Effects of trichloroethylene, hexachlorobenzene and polychlorinated biphenyls on the growth and cell size of marine phytoplankton. Bull. Environ. Contam. Toxicol., 21: 196–201.

Bouwer, E.J., B.E. Rittmann, and P.L. McCarty. 1981. Anaerobic degradation of halogenated 1- and 2-carbon organic compounds. Environ. Sci. Technol., 15: 596–602.

Bringmann, G., and R. Kuhn. 1980. Comparison of the toxicity thresholds of water pollutants to bacteria, algae, and protozoa in the cell multiplication inhibition test. Water Res., 14: 231–241.

Bruckner, J.V., B.D. Davis, and J.N. Blancato. 1989. Metabolism, toxicity, and carcinogenicity of trichloroethylene. CRC Crit. Rev. Toxicol., 20: 31–50.

Buccafusco, R.J., S.J. Ells, and G.A. LeBlanc. 1981. Acute toxicity of priority pollutants to bluegill (*Lepomis macrochirus*). Bull. Environ. Contam. Toxicol., 26: 446–452.

Bull, R.J. 1985. Carcinogenic and mutagenic properties of chemicals in drinking water. Sci. Total Environ., 47: 385–413.

Canada Gazette. 1989. Priority Substances List. Extract Canada Gazette, Part I, February 11, 1989, Ministry of Supply and Services Canada.

Canton, J.H., and D.M.M. Adema. 1978. Reproducibility of short-term and reproduction toxicity experiments with *Daphnia magna* and comparison of the sensitivity of *Daphnia magna* with *Daphnia pulex* and *Daphnia cucullata* short-term experiments. Hydrobiologia, 59: 135–140.

CCME (Candian Council of Ministries of Environment). 1991. A Protocol for the Derivation of Water Quality Guidelines for the protection of Aquatic life. *In* Canadian Water Quality Guidelines (1987). Prepared by the task Force on Water Quality Guidelines of the Canadian Council of Ministers of the Environment, Appendix IX. 24pp.

CCREM. 1987. Canadian Water Quality Guidelines. Prepared by the Task Force on Water Quality Guidelines of the Canadian Council of Resource and Environment Ministers.

COARGLWQ. 1986. Canada–Ontario Agreement Respecting Great Lakes Water Quality. St. Clair River Pollution Investigation (Sarnia area).

Cogswell, S.A., J. Bakker, and A. Kitai. 1982. CEH Marketing Research Report, C_2 Chlorinated Solvents. Chemical Economics Handbook. New York: SRI International.

Cohen, Y., and P.A. Ryan. 1985. Multimedia modelling of environmental transport: Trichloroethylene test case. Environ. Sci. Technol., 19: 412–417.

Comba, M.E., and K.L.E. Kaiser. 1983. Determination of volatile contaminants at the ng/L level in water by capillary gas chromatography with electron capture detection. Int. J. Environ. Anal. Chem., 16: 17–31.

Conglio, W.A., K. Miller, and D. Mackeever. 1980. The occurrence of volatile organics in drinking water. Criteria and Standards Division, Science and Technology Branch, U.S. Environmental Protection Agency, Washington, D.C. (Cited in Bruckner *et al.*, 1989.)

Connor, M.S. 1984. Comparison of the carcinogenic risks from fish vs. ground water contamination by organic compounds. Environ. Sci. Technol., 18: 628–631.

CPI (Canadian Process Industries). 1988. Trichloroethylene. Corpus Information Services, Don Mills, Ontario.

Daniel, J.W. 1963. Biochem. Pharmacol., 12: 795–802. (Cited in Pearson and McConnell, 1975.)

Defalque, R.J. 1961. Clin. Pharmacol. Ther., 2: 665–688. (Cited in Miller and Guengerich, 1982.)

Dilling, W.L. 1977. Interphase transfer processes. II. Evaporation rates of chloro methanes, ethanes, ethylenes, propanes, and propylenes from dilute aqueous solutions. Comparisons with theoretical predictions. Environ. Sci. Technol., 11: 405–409.

Dilling, W.L., N.B. Tefertiller, and G.J. Kallos. 1975. Evaporation rates and reactivities of methylene chloride, chloroform, 1,1,1-trichloroethane, trichloroethylene, tetrachloroethylene, and other chlorinated compounds in dilute aqueous solutions. Environ. Sci. Technol., 9: 833–838.

Erickson, S.J., and C.E. Hawkins. 1980. Effects of halogenated organic compounds on photosynthesis in estuarine phytoplankton. Bull. Environ. Contam. Toxicol., 24: 910–915.

Freitag, D., L. Ballhorn, H. Geyer, and F. Korte. 1985. Environmental hazard profile of organic chemicals. An experimental method for the assessment of the behaviour of organic chemicals in the ecosphere by means of simple laboratory tests with ^{14}C labelled chemicals. Chemosphere, 14: 1589–1616.

Fukuda, K., K. Takemoto, and H. Tsuruta. 1983. Inhalation carcinogenesis of trichloroethylene in mice and rats. Ind. Health, 21: 243–254.

Gay, B.W., P.L. Hanst, J.J. Bufalini, and R.C. Noonan. 1976. Atmospheric oxidation of chlorinated ethylenes. Environ. Sci. Technol., 10: 58–67.

Geyer, H., G. Politzki, and D. Freitag. 1984. Prediction of ecotoxicological behaviour of chemicals: Relationship between n-octanol/water partition coefficient and bioaccumulation of organic chemicals by alga *Chlorella*. Chemosphere, 13: 269–284.

Health and Welfare Canada. 1989. Guidelines for Canadian Drinking Water Quality. Prepared by the Federal–Provincial Subcommittee on Drinking Water of the Federal–Provincial Advisory Committee on Environmental and Occupational Health.

Henschler, D., H. Elsasser, W. Romen, and E. Eder. 1984. Carcinogenicity study of trichloroethylene, with and without epoxide stabilizers in mice. J. Cancer Res. Clin. Oncol., 104: 149.

Jensen, S., and R. Rosenberg. 1975. Degradability of some chlorinated aliphatic hydrocarbons in seawater and sterilized water. Water Res., 9: 659–661.

Kaiser, K.L.E., and M.E. Comba. 1983. Volatile contaminants in the Welland River watershed. J. Great Lakes Res., 9: 274–280.

Kaiser, K.L.E., and M.E. Comba. 1986. Tracking river plumes with volatile halocarbon contaminants: The St. Clair River–Lake St. Clair example. Environ. Toxicol. Chem., 5: 965–976.

Kaiser, K.L.E., and D.G. Oliver. 1976. Determination of volatile halogenated hydrocarbons in water by gas chromatography. Anal. Chem., 48: 2207–2209.

Kaiser, K.L.E., and J.M. Ribo. 1988. *Photobacterium phosphoreum* toxicity bioassay. II. Toxicity data compilation. Toxic Assess., 3: 195–237.

Kaiser, K.L.E., and I. Valdmanis. 1979. Volatile chloro- and chlorofluorocarbons in Lake Erie—1977 and 1978. J. Great Lakes Res. 5: 160–169.

Kaiser, K.L.E., M.E. Comba, and H. Huneault. 1983. Volatile halocarbon contaminants in the Niagara River and in Lake Ontario. J. Great Lakes Res., 9: 212–223.

Kimmerle, G., and A. Eben. 1973. Arch. Toxikol., 30: 115-126. (Cited in Miller and Guengerich, 1982.)

Kleopfer, R.D., D.M. Easley, B.B. Haas, T.G. Diehl, D.E. Jackson, and C.J. Wurrey. 1985. Anaerobic degradation of trichloroethylene in soil. Environ. Sci. Technol., 19: 277–280.

Lay, J.P., W. Schauerte, and W. Klein. 1984. Effects of trichloroethylene on the population dynamics of phyto- and zooplankton in compartments of a natural pond. Environ. Pollut. Ser. A, Ecol. Biol., 33: 75–91.

LeBlanc, G.A. 1980. Acute toxicity of priority pollutants to water flea (*Daphnia magna*). Bull. Environ. Contam. Toxicol., 24: 684–691.

Leibman, K.C. 1965. Mol. Pharmacol., 1: 239–246. (Cited in Miller and Guengerich, 1982).

Lesage, S., P.G. Riemann, and R.A. McBride. 1989. Degradation of organic solvents in landfill leachate. *In* Proceedings of the Ontario Ministry of the Environment Technology Transfer Conference 2: 88–97.

Lesage, S., R.E. Jackson, M.W. Priddle, and P. Reiemann. 1990. Occurrence and fate of organic solvent residues in anoxic ground-water at the Gloucester landfill, Canada. Environ. Sci. Technol., 24(4): 559–566.

Little, C.D., A.V. Palumbo, S.E. Herbes, M.E. Lidstrom, R.L. Tyndall, and P.J. Gilmer. 1988. Trichloroethylene biodegradation by a methane-oxidizing bacterium. Appl. Environ. Microbiol., 54: 951–956.

Loekle, D.M., A.J. Schecter, and J.J. Christian. 1983. Effects of chloroform, tetrachloroethylene, and trichloroethylene on survival, growth, and liver of *Poecilia sphenops*. Bull. Environ. Contam. Toxicol. 30: 199–205.

Love, O.T., and R.G. Eilers. 1982. Treatment of drinking water containing trichloroethylene and related industrial solvents. J. Am. Water Works Assoc., 74: 413–425.

Lum, K.R., and K.L.E. Kaiser. 1986. Organic and inorganic contaminants in the St. Lawrence River: Some preliminary results on their distribution. Water Pollut. Res. J. Can. 21: 592–603.

Mackay, D. 1987. The holistic assessment of toxic organic chemicals in Canadian waters. Can. Water Res. J., 12: 14–22.

Mackay, D., S. Patterson, B. Cheung, and W. Neely. 1985. Evaluating the behaviour of chemicals with a Level III Fugacity Model. Chemosphere, 14: 335–374.

Maltoni, C., G. Lefemine, G. Cotti, and G. Perino. 1988. Long-term carcinogenicity bioassays on trichloroethylene administered by inhalation to Sprague-Dawley rats and Swiss and B6C3F1 mice. Ann. N.Y. Acad. Sci., 534: 316–342.

McConnell, G., D.M. Ferguson, and C.R. Pearson. 1975. Chlorinated hydrocarbons and the environment. Endeavor 34: 13–18.

McNeill, W.C. 1979. Trichloroethylene. *In* Encyclopedia of Chemical Technology, Vol. 5, ed. I. Kirk and D.F. Othmer. New York: John Wiley and Sons.

Miller, R.E., and F.P. Guengerich. 1982. Oxidation of trichloroethylene by liver microsomal cytochrome P-450: Evidence for chlorine migration in a transition state not involving trichloroethylene oxide. Biochemistry, 21: 1090–1097.

Muraoka, K., and T. Hirata. 1988. Hydraulic behaviour of chlorinated organic compounds in water. Water Res., 22: 485–489.

National Cancer Institute. 1976. Carcinogenesis bioassay of trichloroethylene. CAS No. 79-01-6. NCI-CG-TR-2. Washington, D.C.

National Toxicology Program. 1983. Carcinogenesis bioassay of trichloroethylene (without epichlorohydrin) in F344/N rats and B6C3F1/N mice (gavage studies). Tech. Rep. 243, NIH Publ.

B6C3F1/N mice (gavage studies). Tech. Rep. 243, NIH Publ. No. 83–1799. U.S. Department of Health and Human Services, Washington, D.C. (Cited in Bruckner *et al.*, 1989.)

National Toxicology Program. 1987. Toxicology and carcinogenesis studies in four strains of rats. (ACI, August, Marshall, Osborne-Mendell). NTP TR 273, NIH Publ. No. 88-2529. U.S. Department of Health and Human Services, Bethesda, Md.

Neely, W.B., D.R. Branson, and G.E. Blau. 1974. Partition coefficient to measure bioconcentration potential of organic chemicals in fish. Environ. Sci. Technol., 8: 1113–1115.

Nelson, M.J.K., S.O. Montgomery, W.R. Mahaffey, and P.H. Pritchard. 1987. Biodegradation of trichloroethylene and involvement of an aromatic biodegradative pathway. Appl. Environ. Microbiol., 53: 949–954.

Nimmo, D.R., D.J. Hansen, J.A. Couch, N.R. Cooley, P.R. Parrish, and J.I. Lowe. 1975. Toxicity of Aroclor 1254 and its physiological activity in several estuarine organisms. Arch. Environ. Contam. Toxicol. 3: 22–39.

N.S. DOE. 1983. The occurrence of trichloroethylene and tetrachloroethylene in well water in Amherst, N.S. Unpubl. doc., Nova Scotia Department of Environment.

Ofstad, E.B., H. Drangsholt, and G. Carlberg. 1981. Analysis of volatile halogenated organic compounds in fish. Sci. Total Environ., 20: 205–215.

Pakdel, H.S., S. Lesage, P. Geilinas, and C. Roy. 1989. Toxic chemicals in soil and ground water at the contaminants site of Ville-Mercier, P.Q. Presented at the 39th Canadian Chemical Engineering Conference, Hamilton, Ont., October 1–4, 1989.

Parsons, F., G. Barrio-Lage, and R. Rice. 1985. Biotransformation of chlorinated organic solvents in static microcosms. Environ. Toxicol. Chem., 4: 739–742.

Pearson, C.R., and G. McConnell. 1975. Chlorinated C_1 and C_2 hydrocarbons in the marine environment. Proc. R. Soc. London, Ser. B, 189: 305–332.

Roberts, P.V., J. Schreiner, and G.D. Hopkins. 1982. Field study of organic water quality changes during ground water recharge in the Palo Alto baylands. Water Res., 16: 1025–1035.

Slooff, W. 1979. Detection limits of a biological monitoring system based on fish respiration. Bull. Environ. Contam. Toxicol., 23: 517–523.

Slooff, W. 1983. Benthic macroinvertebrates and water quality assessment: Some toxicological considerations. Aquat. Toxicol., 4: 73–82.

Slooff, W., and R. Baerselman. 1980. Comparison of the usefulness of the Mexican axolotl (*Ambystoma mexicanum*) and the clawed toad (*Xenopus laevis*) in toxicological bioassays. Bull. Environ. Contam. Toxicol., 24: 439–443.

Slooff, W., J.H. Canton, and J.L.M. Hermens. 1983a. Comparison of the susceptibility of 22 freshwater species to 15 chemical compounds. I. (Sub)Acute toxicity tests. Aquat. Toxicol., 4: 113–128.

Slooff, W., D. de Zwart, and J.M. Marquenie. 1983b. Detection limits of a biological monitoring system for chemical water pollution based on mussel activity. Bull. Environ. Contam. Toxicol., 30: 400–405.

Strandberg, G.W., T.L. Donaldson, and L.L. Farr. 1989. Degradation of trichloroethylene and trans-1,2-dichloroethylene by a methanogenic consortium in a fixed-film, packed-bed bioreactor. Environ. Sci. Technol., 23: 1422–1425.

Subramanian, A., S. Tanabe, H. Hidaka, and R. Tatsukawa. 1986. Bioaccumulation of organochlorines (PCBs and p, p'-DDE) in Antarctic Adelie penguins *Pygoscelis adeliae* collected during a breeding season. Environ. Pollut. Ser. A., Ecol. Biol., 40: 173–189.

U.S. EPA. 1978. Ambient water quality criteria: Trichloroethylene. U.S. Environmental Protection Agency PB 292 443.

U.S. EPA. 1979. Water-related environmental fate of 129 priority pollutants. II. Halogenated aliphatic hydrocarbons, halogenated ethers, monocyclic aromatics, phthalate esters, polycyclic aromatic hydrocarbons, nitrosamines, and miscellaneous compounds. U.S. Environmental Protection Agency EPA 440/4 79-029b.

U.S. EPA. 1980. Ambient water quality criteria for trichloroethylene. U.S. Environmental Protection Agency EPA 440/5-80-077.

U.S. EPA. 1986. Water quality criteria summary. Criteria and Standards Division, U.S. Environmental Protection Agency, Washington, D.C.

Van Duuren, B.L., B.M. Goldschmidt, G. Loewengart, A.C. Smith, S. Melchionne, I. Seidman, and D. Roth. 1979. Carcinogenicity of halogenated olefinic and aliphatic hydrocarbons in mice. J. Natl. Cancer Inst., 63: 1433–1439.

Verschueren, K. 1983. Handbook of Environmental Data on Organic Chemicals. New York: Van Nostrand Reinhold Company.

Vogel, T.M., and P.L. McCarty. 1985. Biotransformation of tetrachloroethylene to trichloroethylene, dichloroethylene, vinyl chloride, and carbon dioxide under methanogenic conditions. Appl. Environ. Microbiol., 49: 1080–1083.

Wang, T.C., and C.K. Tan. 1987. Photodegradation of trichloroethylene in microheterogeneous aqueous systems. Environ. Int., 13: 359–362.

Wang, T.C., R. Lenahan, M. Kanik, and J. TenEyck. 1985. The removal of trichloroethylene contaminated ground water at Vero Beach, Florida. Arch. Environ. Contam. Toxicol., 14: 719–723.

Ward, G.S., A.J. Tolmsoff, and S.R. Petrocelli. 1986. Acute toxicity of trichloroethylene to saltwater organisms. Bull. Environ. Contam.Toxicol., 37: 830–836.

Westrick, J.J., J.W. Mello, and R.F. Thomas. 1984. The ground water supply survey. J. Am. Water Works Assoc., 76(5): 52–59.

Wilson, J.T., and B.H. Wilson. 1985. Biotransformation of trichloroethylene in soil. Appl. Environ. Microbiol., 49: 242–243.

Zoeteman, B.C.J., K. Harmsen, J.B.H.J. Linders, C.F.H. Morra, and W. Slooff. 1980. Persistent organic pollutants in river water and ground water of the Netherlands. Chemosphere, 9: 231–249.

Zugger, P.D. 1989. Memorandum from P.D. Zugger, Chief, Surface Water Quality Division. State of Michigan Department of Natural Resources, Rule 57 (2) Guideline Levels for January, 1989.

Appendix A
Literature Search

APPENDIX A

Literature Search

A literature search of the following data bases was conducted to retrieve any references that considered the effects of TCE on major water uses:

	Data base	Coverage period
1.	AQUAREF (Canadian Water Resources References)	1970 – Oct. 1988
2.	ASFA (Aquatic Sciences and Fisheries Abstracts)	1978 – Nov. 1988
3.	BIOSIS	1979 – Dec. 1988
4.	CAS Online (Chemical Abstracts Service)	1967 – Dec. 1988
5.	CODOC	1970 – Dec. 1988
6.	COMPENDEX	1970 – Dec. 1988
7.	ELIAS (Environment Libraries Automated System)	1976 – Dec. 1988
8.	ENVIROLINE	1970 – Oct. 1988
9.	EPB (Environmental Bibliography)	1974 – Apr. 1988
10.	FEDERAL REGISTER ABSTRACTS	1977 – Dec. 1988
11.	GEOREF (Geological Reference File)	1985 – Nov. 1988
12.	IRPTC (International Reference of Potentially Toxic Chemicals)	1976 – Dec. 1988
13.	MICROLOG	1979 – Sept. 1988
14.	NTIS (National Technical Information Service)	1964 – Dec. 1988
15.	POLLUTION ABSTRACTS	1970 – Sept. 1988
16.	SWRA (Selected Water Resources Abstracts)	1968 – Jan. 1989
17.	TOXLIT	1981 – Dec. 1988
18.	TOXICO	1974 – Dec. 1988

Several studies were also obtained by consulting review papers.

Appendix B
Toxicity Studies

Table B-1. Freshwater Acute Toxicity Studies

Organism	Test type	Test rank	Concentration ($mg \cdot L^{-1}$)	Effect	Reference
Bacteria					
Pseudomonas putida	S, U	SE	65.0	6-h toxicity threshold	Bringmann and Kuhn, 1980
Microcystis aeruginosa	S, U	SE	63.0	192-h toxicity threshold	Sloof *et al.*, 1983a
Phytoplankton					
Selenastrum capricornutum	S, U	SE	175.0	96-h NOEC	Slooff *et al.*, 1983a
Scenedesmus pannonicus	S, U	SE	>1000.0	192-h toxicity threshold	Slooff *et al.*, 1983a
Invertebrates					
Daphnia magna (<24 h old)	S, U	SE	22.0 18.0 2.2	24-h LC_{50} 48-h LC_{50} 48-h NOEC	LeBlanc, 1980
D. magna (<24 h old)	S, U	UN	42.0 56.0 97.0	48-h LC_{50} 48-h LC_{50} 48-h LC_{50}	Canton and Adema, 1978*
D. magna (<24 h old)	S, U	SE	54.0	48-h NOLC	Slooff *et al.*, 1983a
D. magna (4–6 d old)	S, U	SE	7.76	48-h EC_{50} immobilization	Abernethy *et al.*, 1986
D. pulex (<24 h old)	S, U	UN	45.0	48-h LC_{50}	Canton and Adema, 1978
D. pulex (<24 h old)	S, U	SE	25.0	48-h NOLC	Slooff *et al.*, 1983a
D. cucullata (11 d old)	S, U	UN	57.0	48-h LC_{50}	Canton and Adema, 1978
Aedes aegypti (3rd instar)	S, U	SE	48.0 32.0	48-h LC_{50} 48-h NOLC	Slooff *et al.*, 1983a
Culex pipiens (3rd instar)	S, U	SE	55.0 29.0	48-h LC_{50} 48-h NOLC	Slooff *et al.*, 1983a
Dugesia cf. *lugubris*	S, U	UN	42.0	48-h LC_{50}	Slooff, *et al.*, 1983a

*Results reported from three different laboratories

S = static test
U = unmeasured TCE concentration
F = flow-through test
M = measured TCE concentration
y-o-y = young-of-year
PR = primary study, which may be included in minimum data set for Canadian water quality guidelines
SE = secondary study, which may be included in minimum data set for Canadian water quality guidelines or interim guidelines
UN = unaceptable study, which cannot be included in minimum data set for Canadian water quality guidelines or interim guidelines

Table B-1. Continued

Organism	Test type	Test rank	Concentration ($mg \cdot L^{-1}$)	Effect	Reference
Hydra oligacatis (budless)	S, U	SE	75.0 62.0	48–h LC_{50} 48–h NOLC	Slooff *et al.*, 1983a
Lymnae stagnalis (3–4 wk)	S, U	SE	56.0 32.0	48–h LC_{50} 48–h NOLC	Slooff *et at.*, 1983a
Limnodrilus. Tubifex	S, U	UN	132.0	48–h LC_{50}	Slooff, 1983
Chironomus gr. thummi	S, U	UN	64.0	48–h LC_{50}	Slooff, 1983
Erpobdella octoculata	S, U	UN	75.0	48–h LC_{50}	Slooff, 1983
Asellus aquaticus	S, U	UN	30.0	48–h LC_{50}	Slooff, 1983
Dugesia cf. *lugubris*	S, U	UN	42.0	48–h LC_{50}	Slooff, 1983
Corixa punctata	S, U	UN	110.0	48-h LC_{50}	Slooff, 1983
Gammarus pulex	S, U	UN	24.0	48-h LC_{50}	Slooff, 1983
Ischnura elegans	S, U	UN	49.0	48-h LC_{50}	Slooff, 1983
Nemoura cinerea	S, U	UN	70.0	48-h LC_{50}	Slooff, 1983
Cloeon dipterum	S, U	UN	42.0	48-h LC_{50}	Slooff, 1983
Dreissena polymorpha	F, U	UN	9.7	Increased valve closure rate	Slooff *et al.*, 1983a
Fish					
Leuciscus idus	S, U	SE	213.0	48-h LC_{50}	Slooff *et al.*, 1983a
Oncorhynchus mykiss (5–8 wk)	S, U	SE	42.0 33.0	48-h LC_{50} 48-h NOLC	Slooff *et al.*, 1983a
O. mykiss	F, U	SE	5.0	Increased respiration rate	Slooff, 1979
Poecilia reticulata (3–4 wk)	S, U	SE	182.0 120.0	48-h LC_{50} 48-h NOLC	Slooff *et al.*, 1983a
Oryzias latipes (4–5 wk)	S, U	SE	270.0 220.0	48-h LC_{50} 48-h NOLC	Slooff *et al.*, 1983a
Pimephales promelas (3–4 wk)	S, U	SE	47.0 36.0	48-h LC_{50} 48-h NOLC	Slooff *et al.*, 1983a

Table B-1. Continued

Organism	Test type	Test rank	Concentration ($mg \cdot L^{-1}$)	Effect	Reference
P. promelas	F, M	PR	40.7	96-h LC_{50}	Alexander *et al.*, 1978
	S, U	SE	66.8	96-h LC_{50}	
	F, M	PR	21.9	96-h EC_{50} (any of equilibrium, narcosis, gill swelling, melanization)	
Lepomis macrochirus (y-o-y)	S, U	SE	45.0	96-h LC_{50}	Buccafusco *et al.*, 1981
Jordanella floridae (juvenile)	F, M	PR	28.2	96-h LC_{50}	ATRG, 1988
	S, U	SE	63.1	96-h LC_{50}	
Amphibians					
Xenopus laevis	S, U	UN	45.0	48-h LC_{50}	Slooff and Baerselman, 1980
Ambystoma mexicanum	S, U	UN	48.0	48-h LC_{50}	Slooff and Baerselman, 1980

Table B-2. Marine Acute Toxicity Studies

Organism	Test type	Test rank	Concentration ($mg \cdot L^{-1}$)	Effect	Reference
Bacteria					
Photobacterium phosphoreum	S, U	UN	120	15-min EC_{50}	Kaiser and Ribo, 1988
Phytoplankton					
Phaeodactylum tricornutum	S, U	UN	8.0	EC_{50} (^{14}C uptake)	Pearson and McConnell, 1975
Skeletonema costatum	S, M	SE	95.0	96-h EC_{50} (cell number)	Ward *et al.*, 1986
Thalassiosira pseudonana and *Dunaliella tertiolecta*	S, U	UN	0.05 0.10	72-h NOEC (cell size distribution)	Biggs *et al.*, 1979
Natural estuarine phytoplankton assemblage	F, U	SE	0.5, 1.0 2.0	27% increase in ^{14}C uptake during 48-h exposure 48-h NOEC (^{14}C uptake)	Erickson and Hawkins, 1980
Invertebrates					
Mysidopsis bahia (3 d old)	S, M	SE	14.0	96-h LC_{50}	Ward *et al.*, 1986
Elminius tricornutum (larvae)	S, M	UN	20.0	48-h LC_{50}	Pearson and McConnell, 1975
Fish					
Limanda limanda	F, M	UN	16.0	96-h LC_{50}	Pearson and McConnell, 1975
Cyprinodon variegatus	S, M	SE	52.0	96-h LC_{50}	Ward *et al.*, 1986

S = static test
U = unmeasured TCE concentration
F = flow-through test
M = measured TCE concentration
PR = primary study, which may be included in minimum data set for Canadian water quality guidelines
SE = secondary study, which may be included in minimum data set for Canadian water quality guidelines or interim guidelines
UN = unacceptable study, which cannot be included in minimum data set for Canadian water quality guidelines or interim guidelines

Table B-3. Freshwater Chronic Toxicity Studies

Organism	Test type	Test rank	Concentration ($mg \cdot L^{-1}$)	Effect	Reference
Fish					
Jordanella floridae	F, M	PR	21.1	10-d larval survival reduced by 58%	ATRG, 1988
			20.9	28-d fry LC_{100}	
			5.8	10-d larval survival NOEC	
			10.6	28-d fry survival NOEC	
Salvelinus fontinalis	F, M	PR	0.21	120-d fry weight reduced 5%	ATRG, 1988
Poecilia sphenops	S, U	UN	1.0, 5.0	Reduced growth and survival*	Loekle *et al.*, 1983
Field					
Natural pond community	S, U	SE	25.0	After 43 d, *Daphnia* abundance reduced, phytoplankton richness and abundance reduced	Lay *et al.*, 1984

*Tests were not replicated, static conditions deemed inappropriate for chronic study.

S = static test
U = unmeasured TCE concentration
F = flow-through test
M = measured TCE concentration
PR = primary study, which may be included in minimum data set for Canadian water quality guidelines
SE = secondary study, which may be included in minimum data set for Canadian water quality guidelines or interim guidelines
UN = unacceptable study, which cannot be included in minimum data set for Canadian water quality guidelines or interim guidelines