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An Evaluation and Analysis of the Johnson and Ettinger (1991) Soil Gas Transport Model

by

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A thesis submitted to the Faculty of Graduate Studies and Research in partial fulfillment of the requirements for the degree of Master of Science

in

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The undersigned certify that they have read, and recommend to the Faculty of Graduate Studies and Research for acceptance, a thesis entitled An Evaluation and Analysis of the Johnson and Ettinger (1991) Soil Gas Transport Model submitted by Romalynn Marie de Guzman in partial fulfillment of the requirements for the degree of Master of Science in Environmental Science.

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Dr. Marvin Dudas

Sept. 23, 2001

Dr. Ted Nason

Good health and good sense are two of life's greatest blessings.
-Publius Syrus (42 B.C.)

To my Mother and Father
who have always encouraged me to
expand my horizons

Abstract

Indoor air quality may be impacted by contaminated soil gas migrating into building structures via advective and diffusive transport mechanisms. Petroleum hydrocarbon soil contamination from contaminant spills and leakage are of concern due to the human health impacts of petroleum hydrocarbon VOCs.

The Johnson and Ettinger screening model was developed to predict indoor air contaminant concentrations, which considers advective and diffusive transport, as well as contaminant, soil and building properties. A sensitivity analysis was conducted on the Johnson and Ettinger model to identify those parameters having the greatest contribution to variance in model output. A further analysis was conducted to determine which input parameters influence advective transport.

Remediation objectives for soil and groundwater were derived using CCME default values assigned to model input parameters. A review of field screening and analytical methods for detection of volatile petroleum hydrocarbon in the subsurface and indoor air environment is also included.

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α	attenuation coefficient
A_b	building area
A_c	basement crack area
A_v	building volume
ACH	air exchange rate
BSC	background soil concentration
C_i	contaminant concentration in phase i
C_{sat}	soil saturation concentration
C_{so}	vapour concentration at the contaminant source
C_s	soil gas concentration near the structure
D	Diffusion coefficient
D^{air}	Diffusivity of a component in air
D^{H_2O}	Diffusivity of a component in water
D_T^{eff}	overall effective porous media diffusion coefficient
GWQG	groundwater quality guideline
H	Henry's law constant
η	gas viscosity
ϵ_m	moisture filled porosity
ϵ_v	vapour filled porosity
ϵ_T	total soil porosity
ET	exposure term
g	acceleration due to gravity
K_{oc}	organic carbon partition coefficient
k_v	soil permeability to vapour flow
L_T	distance from contaminant source to foundations
ΔP	indoor-outdoor pressure difference
Pe	Peclet number
Q_s	soil gas flow rate into the building
R	universal gas constant
RfC	Reference Concentration
RSD	Risk Specific Dose
r_c	radius of an ideal cylinder
ρ_a	density of air
ρ_b	soil bulk density
SAF	soil allocation factor
SQG	soil quality guideline
μ	vapour viscosity
V_b	volume of structure
X_c	total floor-wall-seam perimeter distance
Z_c	basement floor depth below the surface

Chapter 1

Introduction

Indoor air quality may be affected by contaminant vapours present in surrounding soils by migration of soil gas from the contaminant source through cracks and openings in a housing foundation. This migration occurs by the processes of advection and diffusion. This project evaluates the Johnson and Ettinger (1991) vapour transport model in terms of developing risk-based guidelines for volatile petroleum hydrocarbons in soil. The Johnson and Ettinger (1991) model is a screening level algorithm used to predict indoor air contaminant concentrations based on advective and diffusive transport of soil gas. This transport process has been described earlier by Nazaroff et al. (1985) in a study of radon concentrations in homes and later extended to contaminant vapours in soil as a result of contaminant spills and landfill gas migration (Nazaroff et al., 1987; Garbesi & Sextro, 1989; Hodgson et al., 1992).

A number of cases of contaminant vapour migration into building structures have been reported in the literature. In 1992 the Canadian Mortgage and Housing Corporation (CMHC) conducted a nationwide survey of soil gas entry problems into houses affected by hazardous lands (Fugler et al., 1992). Three general categories of environmental pollutants entering houses were defined and included petroleum vapours, methane, and assorted volatile organic compounds (VOCs). Sources for these environmental pollutants include industrial sites, mine gases, leakage from fuel storage tanks, landfills, as well as methane contamination due to peat bogs and swamps. It was concluded that soil gas infiltration into houses by environmental pollutants is a widely occuring phenomenon in Canada (Fugler et al., 1992).

In the state of Massachusetts, releases of petroleum products, dry cleaning fluids, and chemicals from industrial sites have impacted the soil and groundwater at thousands of sites (Fitzpatrick & Fitzgerald, 1996). A study conducted by the Massachusetts Department of Environmental Protection (MADEP) identified and reviewed reports of 47 contaminated sites concerned with VOC partitioning and indoor air quality impacts (Fitzpatrick & Fitzgerald, 1996). This study compared indoor air quality and VOC levels in groundwater to MADEP groundwater standards, which were developed using the generic conditions and assumptions made in the Johnson and Ettinger (1991) model. This study found orders of magnitude difference in modeled contaminant attenuation observed between chlorinated and non-chlorinated VOC contaminants. Furthermore, MADEP groundwater standards underpredicted indoor air concentrations for chlorinated VOCs, but predictions for non-chlorinated VOCs fell near or below groundwater standards. It was concluded that attenuation predicted by the model and used for developing groundwater standards may not be protective of indoor air exposure for chlorinated VOCs. However, groundwater quality guidelines for non-chlorinated VOC contaminated sites may provide adequate protection for indoor air exposure if building and site parameters are similar to parameters used to derive the attenuation coefficient (Fitzpatrick & Fitzgerald, 1996).

Another study investigated soil gas contamination and mechanisms of entry of VOCs into a house with a basement adjacent to a municipal landfill and contaminated soil (Hodgson et al., 1992). High concentrations of VOCs found in the landfill site were also found in soil gas near the house, but at lower concentrations compared to landfill gas concentrations. The study found that the presence of soil gas contamination was not sufficient to result in significant exposures of VOCs in indoor air. The researchers concluded that limited entry of VOCs into the indoor environment was the result of low soil permeability, and relatively low below-grade leakage area of the house.

A residence and elementary school in the Midwest United States was found to be impacted by petroleum contamination (Moseley & Meyer, 1992). The school and residence were adjacent to a large petroleum and liquid-fertilizer tank farm and a gasoline station. Explosive levels of hydrocarbon vapours were found in the residence after complaints of gasoline odours. As a result of complaints of noxious odours by teachers and students, the school was evacuated and shut down and monitoring of air, soil gas, and groundwater was undertaken. The results of monitoring determined that a plume of subsurface hydrocarbon contamination was located beneath a leaking tank, portions of the school property, and the school building.

1.1 Petroleum Hydrocarbon VOCs

Volatile organic compounds (VOCs) are defined as having a boiling-point range from 50–100 °C to 240–260 °C (Maroni et al., 1995). Petroleum hydrocarbons contain VOCs, which have the number of carbon atoms in the range of C_6–C_{16} for aliphatic petroleum hydrocarbons and $C_{>8}$–C_{16} for aromatic petroleum hydrocarbons (CCME, 2000). Also included in these ranges are the BTEX group of compounds (benzene, toluene, ethylbenzene, o-xylene, m-xylene, and p-xylene).

Indoor air contamination by hydrocarbon contaminants in soil is of concern to human health due to the risks associated with exposure. The indoor air inhalation pathway is of importance since the majority of time spent by the population is in the indoor environment. A survey of Canadians reported that, on average, 88.6% of their time was spent indoors, with 65.9% of that time spent in their home environment (Leech et al., 1996).

Hydrocarbon vapour intrusion into a building from the subsurface can result from contaminant spills and leakage of petroleum products combined with advective transport into the building structure. VOCs can also originate in the indoor environment. Sources of VOCs prevalent indoors include consumer and commercial products, paints and associated supplies, adhesives, cosmetic and personal care products, hobby supplies, furnishings and clothing, building materials, heating, ventilating and air-conditioning systems, combustion appliances, and tobacco smoke (Maroni et al., 1995).

Hydrocarbon vapour exposure may result in both acute and chronic health effects under defined conditions of exposure. Many VOCs have narcotic effects resulting in central nervous system depression. Irritation of the eyes and respiratory tract as well as sensitization reactions may occur. Exposure to high concentrations of VOCs have been demonstrated to result in liver and kidney damage (Maroni et al., 1995). Edwards et al. (1997) discuss toxicity of Total Petroleum Hydrocarbons (TPHs). Table 1.1 lists reported health effects of equivalent carbon number fractions for TPHs and BTEX compounds. The Reference Concentration (RfC) associated with the compounds are also listed in Table 1.1. The RfC is "an estimate (with uncertainty spanning perhaps an order of magnitude) of continuous inhalation exposure to the human population, including sensitive subgroups, that is likely to be without appreciable risk of deleterious effects during a lifetime" (USEPA, 1996).

Equivalent Carbon Number	Critical Toxicity Effect	RfC (mg/m^3)
Aliphatic Fractions		
$C_{>5}$-C_6 and $C_{>6}$-C_8	nephrotoxicity, hepatoxicity (neurotoxicity if >53% n-hexane) [a]	18.4
$C_{>8}$-C_{10}, $C_{>10}$-C_{12} and $C_{>12}$-C_{16}	hepatic and hematological changes[a]	1.00
$C_{>16}$-C_{35}	hepatic (foreign body reaction) granuloma[a]	NA[b]
Aromatic Fractions		
$C_{>5}$-C_7 (benzene)	carcinogen; respiratory tract irritant[c]	1.82E-03[d]
$C_{>7}$-C_8 (toluene)	hepatotoxicity, nephrotoxicity[c]	1.0
$C_{>8}$-C_{10}, $C_{>10}$-C_{12} and $C_{>12}$-C_{16}	decreased body weight[c]	0.2
$C_{>16}$-C_{21} and C_{21}-C_{35}	nephrotoxicity[c]	
BTEX		
Benzene	carcinogen; respiratory tract irritant[c]	1.82E-03[d]
Toluene	narcotic; may cause anaemia[c]	1.0
Xylenes	narcotic,irritant; affect heart, liver, kidney, and nervous system[c]	0.7
Ethylbenzene	severe irritation to eye and respiratory tract, Central Nervous System effects [e]	0.4

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[a] Weisman, 1999

[b] NA - Not Available

[c] Maroni et al. 1995

[d] for risk level of 10^{-5}

[e] Risk Assessment Information System, 2000

Table 1.1: Toxicity of Equivalent Carbon Number Hydrocarbon Fractions and BTEX

1.2 Project Description

This study evaluated the Johnson and Ettinger (1991) model in the context of predicting risk-based standards for volatile petroleum hydrocarbons via the soil to indoor air pathway. The general objective was to increase knowledge and understanding of the model and provide information to assist Alberta Environment and industry in making improved risk-based decisions for managing volatile hydrocarbon contaminants in the subsurface. The specific objectives were to:

1. Undertake a sensitivity analysis of the Johnson and Ettinger (1991) model with the goal of identifying the most sensitive input parameters. The analysis would be performed under situations intended to portray variation of site parameters representative of Alberta conditions.

2. Provide recommendations and guidance on approaches to reduce uncertainty in model predictions with values from existing research literature.

3. Identify types of field measurements that can be collected on any site under a Tier II site-specific risk assessment approach. The intent of these data was to improve predictions of indoor vapour concentration and to modify or derive risk-based remediation objectives.

4. Review and describe field methods for measurement of soil gas, including field-based screening methods using flame- and photo-ionization detection and lab-based GC/MS analytical methods.

5. Derive risk-based objectives based on the Johnson and Ettinger (1991) model for selected hydrocarbon vapours measured in the subsurface using collection and analysis methods described above. The approach could be similar to that provided by MADEP using a multi-level assessment. This approach first applies conservative field-based vapour screening objectives and progresses towards a more complex form of quantification such as soil gas GC/MS lab analysis and indoor air quality measurements. Factors affecting variability of data, such as seasonal/temperature differences, measurement error and time to peak concentrations should be examined and discussed in the context of the type of evidence required for risk-based decisions using vapour measurements.

6. Provide ranges of soil and groundwater risk-based remediation objectives for BTEX and relevant carbon range fractions and analytes using the Johnson and Ettinger (1991) model. These ranges are intended to represent land use differences, changes in soil/groundwater properties, changes in contaminant depth, and other sensitive parameters. The method of application of these values should be structured for easy application or modification with typical Phase II Environmental Site Assessment information gathered from the field.

Chapter 2

Johnson and Ettinger(1991)Model

Johnson and Ettinger (1991) proposed a heuristic model to predict contaminant vapour intrusion into buildings for screening level calculations. The Canadian Council Ministers of the Environment (CCME) selected this model to predict Tier I standards based on the soil vapour to indoor air pathway. Alberta Environment is also planning to use this model to update its Petroleum Storage Tank Guidelines with the objective of providing improved guidance for making risk-based decisions at petroleum hydrocarbon contaminated sites. The Johnson and Ettinger (1991) model is used as a screening level calculation, and integrates both advective and diffusive transport, in addition to contaminant soil and building properties. This model has also been used by the USEPA to develop soil screening levels to address vapour intrusion into buildings. Spreadsheets have been developed by the USEPA for site specific application of the Johnson and Ettinger (1991) model for soil and groundwater contamination (USEPA, 1997).

2.1 Model Scenario

Figure 2.1 illustrates a contaminant vapour source located some distance below the basement floor or slab of an enclosed structure. In the region closest to the contaminant source, diffusion is the primary mode of vapour transport through the vadose zone to the building zone of influence. Advection is the main transport mechanism of soil gas within the building zone of influence. In this region, soil gas is swept into the building through cracks in the foundation as a result of the net negative pressure between the building and the soil, which is induced by wind and stack effects resulting from heating and ventilation.

The model is based on the following assumptions (Johnson & Ettinger, 1991):

1. chemical and biological transformations are insignificant
2. steady state conditions
3. contaminant vapours enter structures primarily through cracks and openings in the walls and foundation
4. air inside the building structure is completely mixed
5. no other contaminant sources or sinks exist inside the building structure
6. vapour velocities decrease rapidly with increasing distance from the structure

7. vapour phase diffusion is the dominant mechanism for transporting contaminant vapours from contaminant sources located away from the foundation to the soil region near the foundation

8. all contaminant vapours originating from directly below the basement will enter the basement unless the floors and walls are perfect vapour barriers

9. the soil is homogenous within any horizontal plane with respect to effective diffusion coefficients

10. advective vapour flow in the region near the building is uniform.

2.2 Mechanisms of Contaminant Migration

Soil gas transport through soil matrices can be modeled by the following advection-diffusion transport equation (Johnson & Ettinger, 1991). Equation 2.1 below states that the total amount of contaminant in a volume of soil is influenced by the total amount of contaminant flowing in the volume of soil per unit time by the mechanisms of advection and diffusion and by the formation rate of the contaminant in a given phase.

$$\frac{\partial \sum_i \epsilon_i C_i}{\partial t} + \sum_i \mathbf{u}_i \cdot \nabla C_i = \sum_i D_i^{eff} \nabla C_i + \sum_i R_i \tag{2.1}$$

where, i is the subscript that specifies the phase (v = vapour, s = sorbed, f = free phase or precipitate, m = soil moisture), t is time (s), ϵ_i is the volume fraction of phase i $[\frac{\text{volume of phase i}}{\text{volume of soil}}]$, C_i is the contaminant concentration

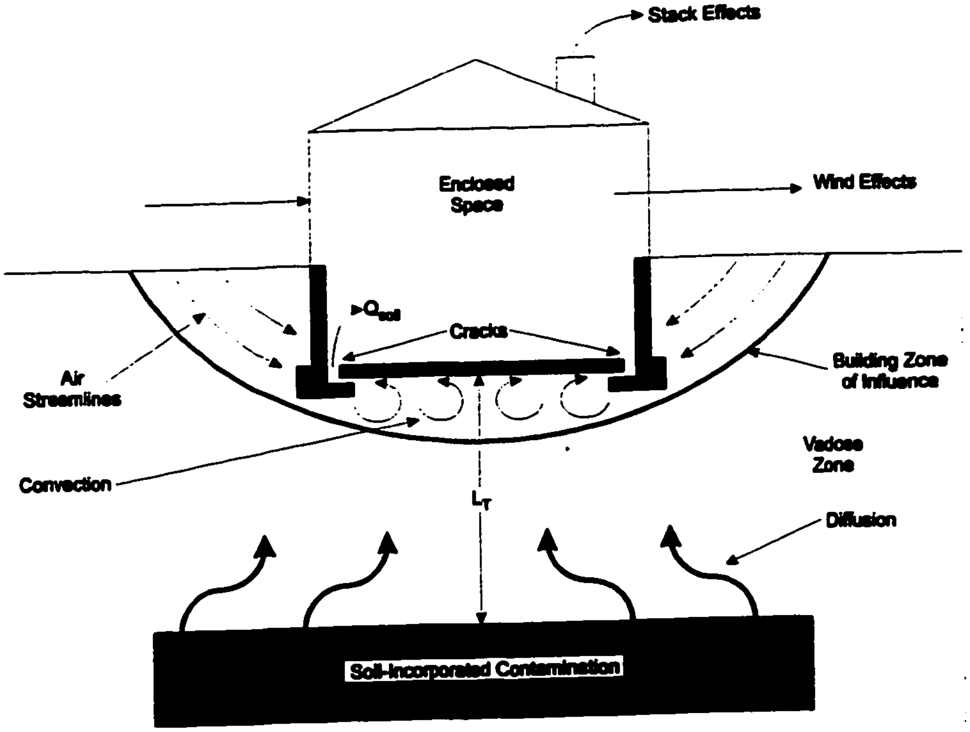

Figure 2.1: Conceptual Diagram of Soil Contamination (USEPA, 2000)

in phase i [$\frac{\text{mass}}{\text{volume}}$ of phase i, $\frac{g}{cm^3}$], $\mathbf{u}_i$ is the Darcy velocity vector associated with phase i [$\frac{cm}{s}$], ∇ is the del operator [$\frac{1}{cm}$], D_i^{eff} is the effective porous medium diffusion coefficient of contaminant in phase i [$\frac{cm^2}{s}$], and R_i is the formation rate of contaminant in phase i [$\frac{g}{cm^3 s}$].

2.2.1 Diffusion

Diffusion is the mass flux of a contaminant toward a decreasing concentration gradient and results in an increase in spreading of the contaminant concentration front with time and is described by Fick's first law of diffusion (Eq 2.2) (Grathwol, 1998):

$$F = -D\frac{\partial C}{\partial x} \qquad (2.2)$$

where, D is the diffusion coefficient (cm^2/s), which measures the rate molecules spread, and C is the solute concentration (g/cm^3), which is dependent on time and distance (x).

Johnson and Ettinger (1991) assumed that diffusive transport is significant only in the vapour and soil moisture phases of the soil matrices. Contaminant concentrations in the soil moisture and vapour phases are assumed to be at equilibrium and are related by the dimensionless Henry's law constant, H (cm^3 of $H_2O/cm^3$of vapour), which describes their partitioning. This relationship can be described as:

$$C_v = HC_m \quad . \qquad (2.3)$$

From the above assumption Eq 2.1 can be rewritten as:

$$\sum_i \nabla \cdot D_i^{eff}\nabla C_i = \nabla \cdot (D_v^{eff} + \frac{D_m^{eff}}{H})\nabla C_v = \nabla \cdot D^{eff}\nabla C_v \quad . \qquad (2.4)$$

In this equation, Fick's first law of diffsion is modified to account for tortuousity in soil pores by D_i^{eff}. D_i^{eff} is the effective porous medium diffusion coefficient based on vapour phase concentrations and is a function of chemical-specific molecular diffusion coefficients in air and water as well as porous medium and chemical properties (Johnson et al., 1998). The effective diffusion coefficient is used to account for irregular channels in soil by describing the tortuous path in vapour movement (Venkatraman et al., 1998). The effective porous medium diffusion coefficients for moisture and vapour phases (D_m^{eff} and D_v^{eff}) can be related to the pure component molecular diffusivities in water and air (D^{H_2O} and D^{air}), total soil porosity (ϵ_T), and vapour and moisture filled porosity (ϵ_v and ϵ_m) by the Millington-Quirk expression in Eq 2.5 and Eq 2.6 (Johnson & Ettinger, 1991):

$$D_m^{eff} = \frac{D^{H_2O}\epsilon_m^{3.33}}{\epsilon_T^2} \tag{2.5}$$

$$D_v^{eff} = \frac{D^{air}\epsilon_v^{3.33}}{\epsilon_T^2} \tag{2.6}$$

where, ϵ_m and ϵ_v are related to ϵ_T, the soil moisture content (θ_m) and the soil bulk density (ρ_b) by:

$$\epsilon_m = \theta_m \rho_b \tag{2.7}$$

$$\epsilon_v = \epsilon_T - \theta_m \rho_b \quad . \tag{2.8}$$

Knowledge of the gaseous diffusion coefficient for the substance in question is necessary to properly model gas movement in porous media as predictive model ouputs can be very sensitive to small changes in the effective diffusion coefficient (Jin & Jury, 1996; Johnson et al., 1998). Diffusive transport dominates when the vapour pressure and molecular weight of the chemical are low or at low concentrations and there is only a small pressure differential (Jin & Jury, 1996).

2.2.2 Advection

Advection is the bulk movement of a fluid resulting from pressure differentials. Advective transport of soil gas into buildings is due to the pressure gradients resulting from temperature differences between indoors and outdoors, wind loading on the building superstructure, barometric pressure changes due to the weather, precipitation and mechanical ventilation (Little et al., 1992; Fischer & Uchrin, 1996). Advective transport dominates soil vapour transport under conditions of large pressure differentials across a building envelope and in circumstances where soil gas contaminants have high vapour pressures.

Advective transport is considered in the model by assuming that significant advective transport takes place only in the vapour phase and is described by Darcy's law:

$$\mathbf{u}_{\mathrm{v}} = -\frac{\mathrm{k_v}}{\mu}\frac{\mathrm{d}}{\mathrm{dx}}\frac{(\mathrm{P} + \rho_{\mathrm{a}}\mathrm{gh})}{\theta_{\mathrm{a}}} \tag{2.9}$$

where, $\mathrm{k_v}$ is the soil permeability to vapour flow (cm^2), μ is the vapour viscosity (g/cm-s), P is the pressure in the vapour phase (g/cm-s^2), ρ_{a} is the density of air (g/cm^3), g is the acceleration due to gravity (cm/s^2), h is the elevation (cm), θ_{a} is the soil air-filled porosity, and $\mathbf{u}_{\mathrm{v}}$ is the vapour-phase mass-average velocity (cm/s)(Lyman et al., 1992).

2.2.3 Soil Gas Transport

To determine if either advection or diffusion dominates Eq 2.4 and Eq 2.9 are inserted into Eq 2.1 and the resulting equation is nondimensionalized (Johnson & Ettinger, 1991):

$$\frac{\partial \sum_i \epsilon_i C_i^*}{\partial t^*} - \frac{L_P}{L_D}(\nabla^* P^*) \cdot (\nabla^* C_i^*) = \nabla \frac{D^{eff} \mu L_P}{k_v \Delta P_r L_D} + \sum_i {R_i}^* \tag{2.10}$$

where, ΔP_r is the reference indoor-outdoor pressure difference and * denotes the following nondimensional variables:

$$C_i^* = \frac{C_i}{C_r} \tag{2.11}$$

$$\nabla^* = L_D \nabla \tag{2.12}$$

$$P^* = \frac{P}{\Delta P_r} \tag{2.13}$$

$$t^* = t \frac{k_v \Delta P_r}{L_P L_D \mu} \tag{2.14}$$

$$R_i^* = \frac{R_i L_P L_D \mu}{C_r k_v \Delta P_r} \tag{2.15}$$

where, C_r is the characteristic concentration, L_D is the diffusion pathway length, and L_P is the convection pathway length, which allow the dependent concentration variable and derivatives of C_i^* and P^* magnitudes of order unity (Johnson & Ettinger, 1991).

The following dimensionless group determines the relative significance of advective and diffusive transport mechanisms (Johnson & Ettinger, 1991):

$$Pe = \frac{k_v \Delta P_r L_D}{D^{eff} \mu L_P} \tag{2.16}$$

where, Pe is the Peclet number, which relates the effectiveness of advective transport to the effectiveness of diffusive transport (Fetter, 1993). Here, it is expressed in terms of the driving pressure, ΔP_r. When Pe $\gg$ 1 advective transport dominates. When Pe $\ll$ 1 diffusive transport dominates.

2.2.4 Contaminant Diffusion from Source to Building Zone of Influence

The rate of contaminant vapour transport from the source to the building zone of influence can be estimated by the following equation:

$$E_l = \frac{A_b(C_{so} - C_s)D_T^{eff}}{L_T} \tag{2.17}$$

where, E_l is the mass-transport rate toward the structure (g/s), A_b is the cross-sectional area through which vapours pass (cm^2), C_{so} is the vapour concentration at the contaminant source (g/cm^3), C_s is the soil gas concentration near the structure (g/cm^3), L_T is the distance from the contaminant source to the foundation (cm), and D_T^{eff} is the overall effective porous media diffusion coefficient based on vapour-phase concentrations for the region between the source and foundation (cm^2/s) (Johnson & Ettinger, 1991).

When equilibrium between vapour and moisture phases is assumed, an overall D_T^{eff} can be determined for multiple discrete soil layers by:

$$D_T^{eff} = \frac{L_T}{\sum_{i=0}^{n} \frac{L_i}{D_i^{eff}}} \tag{2.18}$$

where, L_i is the thickness of soil layer i (cm) and D_i^{eff} is the sum of Eq 2.5 and Eq 2.6 for each soil layer such that:

$$D_i^{eff} = D^{air}\frac{\epsilon_{v,i}^{3.33}}{\epsilon_T^2} + \frac{D^{H_2O}}{H}\frac{\epsilon_{m,i}^{3.33}}{\epsilon_T^2} \tag{2.19}$$

where, D_i^{eff} is the effective diffusion coefficient across soil layer i (cm^2/s), D^{air} is the diffusivity of a component in air (cm^2/s), D^{H_2O} is the diffusivity of a component in water, ϵ_v is the soil vapour filled porosity of layer i (cm^3/cm^3), ϵ_m is the soil moisture filled porosity of layer i (cm^3/cm^3), ϵ_T is the total soil porosity, and H is the dimensionless Henry's law constant (Johnson et al., 1998).

For compounds with dimensionless Henry's Law constants greater than 10^{-3} [(mg/L-vapour)/(mg/L-H_2O)], the contribution due to diffusion through soil moisture is negligible compared to vapour phase diffusion, except in high moisture content zones (Johnson et al., 1998; Johnson & Ettinger, 1991).

2.2.5 Transport of Soil Gas into the Building Structure

Vapour transport adjacent to a building foundation occurs by a combination of advective and diffusive transport mechanisms. The steady state, one dimensional solution to Eq 2.1 for vapour transport through a crack in a foundation or a porous medium with a constant uniform advective velocity is used to derive the equation for the rate of contaminant vapour intrusion into a building (Johnson & Ettinger, 1991).

To derive a differential equation from Eq 2.1, it is assumed that only the vapour phase is significant, steady state conditions (i.e. $\frac{dC_v}{dt} = 0$), and no reactions are taking place ($R_i = 0$). From these assumptions, Eq 2.1 becomes:

$$\mathbf{u}_i \cdot \nabla C_v = D_c^{eff} \nabla^2 C_v \quad . \tag{2.20}$$

We also assume that C_v is a function of the z direction only, and a constant uniform advective velocity

$$\mathbf{u} = \frac{\mathrm{Q_s}}{\mathrm{A_c}}\mathbf{e_z} \tag{2.21}$$

where, $\mathbf{u}$ is the advective velocity (cm/s), $\mathrm{A_c}$ is the basement crack area (cm^2), $\mathbf{e_z} = (\mathbf{0}, \mathbf{0}, \mathbf{1})$, and $\mathrm{Q_s}$ is the soil gas flow rate (cm^3/s), which is defined as (after Nazaroff, 1988):

$$\mathrm{Q_s} = \frac{2\pi\Delta \mathrm{P k_v X_c}}{\mu \ln(\frac{2\mathrm{Z_c}}{\mathrm{r_c}})} \quad \frac{\mathrm{r_c}}{\mathrm{Z_c}} \ll 1 \tag{2.22}$$

where, $\Delta\mathrm{P}$ is the indoor-outdoor pressure difference (g/cm-s^2), $\mathrm{k_v}$ is the soil vapour permeability (cm^2), $\mathrm{X_c}$ is the total floor-wall-seam perimeter distance (cm), $\mathrm{Z_c}$ is the basement floor depth below the surface (cm), and $\mathrm{r_c}$ is the radius of an ideal cylinder (cm), which is defined as:

$$\mathrm{r_c} = \frac{\mathrm{A_c}}{\mathrm{X_c}} \quad . \tag{2.23}$$

With the above assumptions Eq 2.20 becomes:

$$\frac{\mathrm{Q_s}}{\mathrm{A_c}}\frac{\mathrm{dC_v}}{\mathrm{dz}} = \mathrm{D_c^{eff}}\frac{\partial^2 \mathrm{C_v}}{\partial \mathrm{z}^2} \quad . \tag{2.24}$$

Rearranging gives:

$$\mathrm{D_c^{eff}}\frac{\partial^2 \mathrm{C_v}}{\partial \mathrm{z}^2} - \frac{\mathrm{Q_s}}{\mathrm{A_c}}\frac{\mathrm{dC_v}}{\mathrm{dz}} = 0 \quad . \tag{2.25}$$

The boundary conditions used to solve Eq 2.25 are:

$$\mathrm{C_v} = \mathrm{C_s} \quad \text{at} \quad \mathrm{z} = 0 \tag{2.26}$$

where, $\mathrm{C_s}$ is the contaminant vapour concentration in the soil near the structure and:

$$\mathrm{C_v} = \mathrm{C_b} \quad \text{at} \quad \mathrm{z} = \mathrm{L_c} \tag{2.27}$$

where, C_b is the contaminant vapour concentration in the building. The steady state one dimensional solution to Eq 2.1 for vapour transport through a crack with a constant uniform advective velocity is:

$$0 = D_c^{eff} \frac{\partial^2 C_v}{\partial z^2} - uC_v \quad . \tag{2.28}$$

Equation 2.28 can now be inserted into the following flux equation (Paul C. Johnson, personal communication, June 2000):

$$E = A_c(uC_v - D_c^{eff} \frac{dC_v}{dz}) \tag{2.29}$$

and integrated for C_v between C_s and C_b and for z between 0 and L_c, and rearranged to give:

$$E = Q_s C_s - \frac{Q_s(C_s - C_b)}{1 - \exp(\frac{Q_s L_c}{D_c^{eff} A_c})} \tag{2.30}$$

where, E is the entry rate of contaminant into the building (g/s), Q_s is the volumetric flow rate of soil gas into the building (cm^3/s), D_c^{eff} is the effective vapour-pressure diffusion coefficient through the crack in the foundation (cm^2/s), L_c is the foundation thickness (cm), C_b is the contaminant vapour concentration in the building (g/cm^3), and A_c is the area of cracks/openings through which contaminant vapours enter the building (cm^2) (Johnson & Ettinger, 1991).

The soil gas concentration (C_s) can be determined by assuming that the flux rate in soil (Eq 2.17) and the flux rate through concrete (Eq 2.30) are equal at steady state. Setting these equations equal and solving for C_s gives:

$$C_s = \frac{C_{so} \frac{D_T^{eff} A_b}{Q_s L_T} \exp \frac{Q_s L_c}{D_c A_c} + C_b}{\frac{D_T A_b}{Q_s L_T}(\exp \frac{Q_s L_c}{D_c A_c} - 1) + \exp \frac{Q_s L_c}{D_c A_c}} \quad . \tag{2.31}$$

Equation 2.31 can then be substituted into Eq 2.30 to determine the rate of contaminant entry into a building through the foundation to give:

$$\mathrm{E} = \frac{\frac{\mathrm{D_T^{eff} A_b C_{so}}}{\mathrm{L_T}} \exp\frac{\mathrm{Q_s L_c}}{\mathrm{D_c A_c}} - \frac{\mathrm{C_b}}{\mathrm{C_{so}}}}{\frac{\mathrm{D_T^{eff} A_b}}{\mathrm{Q_s L_T}}\left(\exp\frac{\mathrm{Q_s L_c}}{\mathrm{D_c A_c}} - 1\right) + \exp\frac{\mathrm{Q_s L_c}}{\mathrm{D_c A_c}}} \tag{2.32}$$

where, E is the product of the steady-state diffusive rate of contaminant transport from the source to a bare dirt foundation and a factor containing a number of dimensionless groups which are discussed in the following section (Johnson & Ettinger, 1991).

2.2.6 Attenuation Coefficient

Equation 2.32 can be used to give a definitive expression for indoor contaminant vapour concentration. If it is assumed that there are no other contaminant sinks or sources and the building is well-mixed then:

$$\mathrm{Q_b C_b = E} \tag{2.33}$$

where, $\mathrm{Q_b}$ is the building ventilation rate ($\mathrm{cm^3/s}$), which is defined as

$$\mathrm{Q_b} = \frac{\mathrm{V_b ACH}}{\mathrm{3600s}} \tag{2.34}$$

and, $\mathrm{V_b}$ is the volume of the structure ($\mathrm{cm^3}$) and ACH is the air exchange rate ($\mathrm{h^{-1}}$). Substituting Eq 2.32 into Eq 2.33 and rearranging gives:

$$\mathrm{C_b} = \frac{\mathrm{C_b^*} \times \exp\frac{\mathrm{Q_s L_c}}{\mathrm{D_c A_c}}}{\exp\frac{\mathrm{Q_s L_c}}{\mathrm{D_c A_c}} + \frac{\mathrm{D_T^{eff} A_b}}{\mathrm{Q_b L_T}} + \frac{\mathrm{D_T^{eff} A_b}}{\mathrm{Q_s L_t}}\left(\exp\frac{\mathrm{Q_s L_c}}{\mathrm{D_c A_c}} - 1\right)} \tag{2.35}$$

where,

$$\mathrm{C_b^*} = \frac{\mathrm{D_T^{eff} A_b C_{so}}}{\mathrm{Q_b L_T}} \tag{2.36}$$

and, C_b^* describes the indoor vapour concentration when vapours diffuse from the source to a bare soil foundation. Equation 2.35 can be rearranged to yield the attenuation coefficient (α):

$$\alpha = \frac{C_b}{C_{so}} \tag{2.37}$$

or

$$\alpha = \frac{\frac{D_t^{eff} A_b}{Q_b L_T} \times \exp\frac{Q_s L_c}{D_c A_c}}{\exp\frac{Q_s L_c}{D_c A_c} + \frac{D_T^{eff} A_b}{Q_b L_T} + \frac{D_T^{eff} A_b}{Q_s L_T}(\exp\frac{Q_s L_c}{D_c A_c} - 1)} \tag{2.38}$$

Within Eq 2.38, the following dimensionless groups exist:

$$\frac{Q_s L_c}{D_c A_c} \tag{2.39}$$

$$\frac{D_T^{eff} A_b}{Q_b L_T} \tag{2.40}$$

$$\frac{Q_s}{Q_b} \tag{2.41}$$

Equation 2.39 represents the equivalent Pe number for transport through the foundation. Eq 2.40 is the attenuation coefficient for diffusion dominated transport from an infinite source to a bare dirt foundation. Eq 2.41 represents the attenuation coefficient for convective transport from a source located adjacent to the building (Johnson & Ettinger, 1991).

Johnson and Ettinger (1991) analyzed these three dimensionless groups under limiting conditions. Equation 2.42 below represents a scenario where advection is the dominant transport mechanism through the basement floors and walls. When $(\frac{Q_s L_c}{D_c A_c}) \rightarrow \infty$, Eq 2.38 becomes:

$$\alpha = \frac{\frac{D_T^{eff} A_b}{Q_b L_t}}{\frac{D_T^{eff} A_b}{Q_s L_T} + 1} \quad . \tag{2.42}$$

Equation 2.43 represents a scenario where diffusion is the dominant transport mechanism through the basement floor and walls. This equation is used to represent the attenuation coefficient when $(\frac{Q_s L_c}{D_c A_c}) \rightarrow 0$.

$$\alpha = \frac{\frac{D_T^{eff} A_b}{Q_b L_T}}{1 + \frac{D_T^{eff} A_b}{Q_b L_T} + \frac{D_T^{eff} A_b L_c}{D_C A_c L_T}} \tag{2.43}$$

For Eq 2.41, the limit $Q_b \rightarrow 0$ corresponds to a perfectly sealed basement. In this case, Q_s must also approach zero. Therefore $\alpha \rightarrow 1$, where the indoor air concentration approaches the contaminant vapour concentration in the soil gas (Johnson & Ettinger, 1991).

Chapter 3

Sensitivity Analysis

A sensitivity analysis was performed on Eq 2.38 to determine those parameters which have the greatest influence on the attenuation coefficient (α). An additional sensitivity analysis was performed on the equivalent Pe number (Eq 2.39), since the value generated here determines whether advective or diffusive transport dominates overall contaminant mass transport.

3.1 Methodology

Two methods were initially considered for conducting the sensitivity analysis. The first method involved varying one input parameter at a time while keeping all other variables constant and plotting the results. This method will be referred to as the graphical method after the approach used by Hers (1999). The other method that was considered involved the use of Crystal Ball™ software, an add-in program for Microsoft Excel (Decisioneering, 1998). Crystal Ball™ uses Monte Carlo simulation, which generates random numbers to measure the effects of uncertainty in a spreadsheet model. The Monte Carlo method is used in Crystal Ball™ to generate random numbers for selected model parameters similar to real-life possibilities. The set of random numbers simulates a hypothetical scenario for the spreadsheet model. During the simulation, the model is recalculated for each scenario and the results are displayed (Decisioneering, 1998).

The Crystal Ball™ method was chosen to conduct the sensitivity analysis because the graphical method does not consider relationships between model parameters. The graphical method only provides a one-dimensional view of a multi-dimensional model. Therefore, in such an analysis, any perturbations in the model output as a result of parameter interactions are not considered. Since Crystal Ball™ can simultaneously vary all model parameters, a better indication of model sensitivity can be obtained.

Equation 2.38 and Eq 2.39 were prepared for the sensitivity analysis by identifying a total of eleven parameters and assigning each a distribution. These parameters and their assumptions are listed in Table 3.1. Uniform distributions were assigned to most model input parameters to indicate that all possible input values have an equal probability of occuring. The distribution and range of vapour filled porosity and total porosity were derived from data obtained from the Agricultural Region of Alberta Soil Inventory Database Version 1.0 (AGRASID) (Coen & Marciak, 1998). For this analysis, only soil data from upper (A) horizons were used to derive a distribution in order to achieve conservatism in the analysis. As discussed in the previous section, when $(\frac{Q_s L_c}{D_c A_c}) \rightarrow \infty$, Eq 2.38 reduces to Eq 2.42. For the sensitivity analysis, the model was set so that when $\frac{Q_s L_c}{D_c A_c} > 709$, Eq 2.38 defaults to

Eq 2.42 because Microsoft Excel 97 cannot calculate beyond exp(709). A total of 5000 calculations (trials) were run for the analysis of both Eq 2.38 and Eq 2.39.

Sensitivity is computed in Crystal Ball™ by determining rank correlation coefficients between each assumption (input parameter) and forecast (output) cell during the simulation. Correlation coefficients measure the degree to which assumptions and forecasts change together. Decisioneering (1998) defines a high correlation coefficient as ± 1.0. If there is no relationship between the forecast and assumptions, the correlation coefficient is much less than ± 1.0. As the strength of the relationship increases between the forecast and assumption so does the correlation coefficient. A high correlation coefficient indicates an assumption has a significant impact on the forecast (both through its uncertainty and its model sensitivity). Positive coefficients indicate an increase in the assumption is associated with an increase in the forecast. Negative coefficients imply the reverse situation. The larger an absolute value of a correlation coefficient, the stronger the relationship (Decisioneering, 1998).

Parameter	Symbol	Units	Minimum	Likeliest	Maximum	Distribution
Building Area	A_b	cm^2	7.4E+05	-	1.98E+06	uniform
Building Volume	A_v	cm^3	1.60E+08	-	7.32E+08	uniform
Air changes per hour[a]	ACH	h^{-1}	0.05	-	2.00	uniform
Pressure Differential[a]	ΔP	$g/cm\text{-}s^2$	10	-	150	uniform
Distance from source to foundation	L_T	cm	1	-	1000	uniform
Total Floor/Wall Seam Perimeter Distance	X_c	cm	3.45E+03	-	4.00E+03	uniform
Basement Floor Depth Below Surface	Z_c	cm	150	-	220	uniform
Area of Cracks	A_c	cm^2	9.90E+02	-	1.50E+04	uniform
Soil Vapour Permeability[b]	k_v	cm^2	1E-11	-	1E-03	uniform
Vapour filled Porosity[c]	θ_v	-	0.2	0.35	0.56	triangular
Total Porosity[c]	n	-	0.36	0.52	0.65	triangular

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[a] Fugler, 1997

[b] Nazaroff, 1992

[c] Coen & Marciak, 1988

Table 3.1: Model Input Parameters used in Sensitivity Analysis of Eq 2.38 and Eq 2.39

3.2 Results of Sensitivity Analysis

Table 3.2 illustrates the results of the sensitivity analysis of Eq 2.38. As can be seen from Table 3.2 the distance from the source to the foundation exhibited the highest contribution to variance followed by the air exchange rate. L_T has a negative correlation coefficient, indicating that the further away the contaminant source is from the building, the lower indoor contaminant air concentrations will be. As the distance from the source to the foundation exhibited the highest contribution to variance in Eq 2.38, it is recommended that this parameter be measured in the field for model input. The air exchange rate (ACH) enters into the attenuation coefficient equation by Eq 2.34. ACH is defined as the rate indoor air is replaced by outdoor air and is measured by the number of room volumes exchanged over one hour. ACH also exhibited a negative correlation coefficient indicating that as the ACH increases, indoor air contaminant concentrations decrease. Soil vapour filled porosity (ϵ_v) was ranked third in sensitivity and enters into the attenuation coefficient equation through calculation of the overall effective diffusion coefficient (D_T^{eff}) by Eq 2.18. Soil vapour filled porosity exhibited a positive correlation coefficient indicating that as this value increases, so does the model output.

Table 3.3 illustrates the results of the sensitivity analysis of Eq 2.39. As illustrated in Table 3.3, the results of the sensitivity analysis of Eq 2.39 indicated that soil vapour permeability exhibited the highest contribution to variance. Soil vapour permeability was also found to be a contributing factor influencing advective transport of soil gas in a study of VOC migration into basements (Fischer & Uchrin, 1996). Soil vapour permeability exhibited a positive correlation coefficient indicating that as this value increases, so does the model output. Soil vapour filled porosity was second highest in sensitivity followed by the indoor-outdoor pressure differential. The indoor-outdoor pressure differential is the result of depressurization of the building substructure with respect to the surrounding soil. The indoor-outdoor pressure differential exhibited a positive correlation coefficient with respect to model output.

The output of the Crystal Ball sensitivity analyses is presented in Appendix A.

Parameter	Contribution to Variance	Correlation Coefficient
L_T	33.6%	-0.54
ACH	28%	-0.49
ϵ_v	24%	0.45
V_b	7.8%	-0.28
A_b	3.8%	0.18
ϵ_T	2.7%	-0.15
Z_c	0%	0.02
k_v	0%	0.02
X_c	0%	0.01
ΔP	0%	-0.01

Table 3.2: Results of Sensitivity Analysis of Eq 2.38

To reduce uncertainty in model predictions for site-specific cases, the parameters identified as exhibiting a relatively greater contribution to variance in model output are recommended for data collection under a Phase II Environmental Site Assessment.

Parameter	Contribution to Variance	Correlation Coefficient
k_v	35.7%	0.56
ϵ_v	24.3%	-0.46
ΔP	24%	0.46
A_c	12.4%	-0.33
ϵ_T	3.1%	0.16
X_c	0.2%	0.04
V_b	0.1%	-0.03
L_t	0.1%	0.02
A_b	0%	0.01
Z_c	0%	0.01

Table 3.3: Results of Sensitivity Analysis of Eq 2.39

Chapter 4

Considerations for Site-Specific Assessments

Site specific measurements of soil properties used to predict indoor air quality concentrations may be easily merged with typical Phase II site assessments through the use of soil cores, soil moisture and soil gas measurements (Johnson et al., 1999). The most sensitive parameters (L_T, ACH, D_T^{eff}, k_v, ΔP) identified in the sensitivity analysis for Eq 2.38 and Eq 2.39, are recommended for data collection. Other site-specific factors such as time to steady state concentrations, seasonal variations and subsurface variations should also be considered in contaminanted site assessments as these factors can influence predictions of indoor air contaminant vapour concentrations and soil gas measurements.

4.1 Distance from Source to Foundation (L_T)

The distance from the source to the foundation (L_T) exhibited the greatest contribution to variance in the attenuation coefficient. L_T has a negative correlation coefficient, which means that the further away the contaminant source is from the building foundation, the lower the indoor contaminated air concentration will be. To reduce uncertainty in model output, representative data of L_T should be measured in the field for model input. Such measurements can be easily obtained during subsurface investigations. L_T is an important parameter because the closer the source is to the building foundation, the higher indoor air contaminant concentrations will be. Furthermore, L_T is needed to predict the time required for a contaminant to reach steady state concentrations. This concept is discussed further in Section 4.6.

4.2 Air Exchange Rate (ACH)

The air exchange rate (ACH) is defined as the rate indoor air is replaced by outdoor air and is measured by the number of room volumes exchanged over one hour. ACH is governed by infiltration, natural ventilation, and internal air movement. ACH is an important factor as lower air exchange rates contribute to poor indoor air quality due to limited dilution of indoor air pollutants (Fellin & Otson, 1994). The ACH attributed to natural ventilation, infiltration, and internal air movements are due to factors such as the surface area of gaps and openings to the outdoors and the temperature difference or the differential pressure between the indoor and outdoor air (Samimi, 1995).

ACH also contributes to uncertainty in model predictions and could be measured by dynamic and passive tracer gas techniques. Air exchange rates vary seasonally and are also dependent on building construction. Warmer temperatures generally coincide with higher air exchange rates since windows and doors are more likely to be open for longer periods of time in comparison to when cooler temperatures abound. Recently built energy efficient homes are likely to exhibit lower air exchange rates in comparison to older homes (Table 4.1).

Dynamic and passive tracer techniques can be used to measure air exchange and are described below. However, air exchange data can be expensive to collect and disruptive to occupants in the building. Table 4.1 was proposed by The Canada Mortgage and Housing Corporation (Fugler, 1997) and can be used as an estimate of air exchange rates during the heating (winter) season for Canadian prairie homes when calculating the attenuation coefficient. Air exchange rates for commercial and industrial buildings are expected to be higher due to occupancy requirements and increased pedestrian traffic. Air exchange rates for actual situations may have to be adjusted from values listed in Table 4.1 in some situations where normally ventilated houses have low ventilation rates (i.e. houses whose windows are closed during warm weather and have air conditioning). In this case, the values in Table 4.1, column 2 could be reduced by 50% to represent typical annual values. In temperate climates, ventilation rates may be lower than those shown in Table 4.1. Fugler (1997) recommended that for different climate zones the values in Table 4.1, column 2 should be multiplied by the factors listed in Table 4.2 (Fugler, 1997).

House Type	Estimated Heating Season Natural Ventilation Rate	Estimated Mechanical Ventilation Rate
Wood Frame pre-1945	0.5-1.0 hr^{-1}	none
Wood Frame 1946-1960	0.2-0.4 hr^{-1}	none
Wood Frame 1961-1980	0.15-0.3 hr^{-1}	maybe 0.2 hr^{-1} intermittent
Airtight New House	0.05-0.1 hr^{-1}	0.3 hr^{-1} installed capacity

Table 4.1: Typical Air Exhange Rates for Canadian Prairie Homes (Fugler, 1991)

Climate Zone	Natural Ventilation Multiplier
Mild	0.6
Moderate	0.8
Severe	1.0

Table 4.2: Adjustments for Climate Zone (Fugler, 1991)

Dynamic Tracer Tests

Dynamic tracer tests use a unique gas which is introduced into the indoor space or ventilation system. After adequate mixing of the tracer with indoor air, the decay of the tracer gas can be used to calculate the air exchange rate by tracking the tracer over time to produce an exponential decay curve. The air exchange rate can be derived from a plot of the log concentration vs. time. Sulfur hexafluoride (SF_6) is the most commonly used and ideal tracer gas since it is non-reactive, non-toxic, and can be measured at low concentrations using a gas chromatograph and electron-capture detector. This method can be used for acquiring short-term and immediate measurements of air exchange rates. Sampling may be carried out with grab samples using plastic bags, syringes, or evacuated tanks (Yocom & McCarthy, 1991).

Passive Tracer Technique

Permeation tubes that emit known rates of perfluorocarbon tracers can be used for measuring air exchange rates. Passive monitors are placed throughout the building to measure integrated concentrations of tracer gases and are analyzed by gas chromatography. This method allows determination of long-term air exchange rates (i.e. 6 days - 6 months) (Yocom & McCarthy, 1991). The Air Infiltration Measurement System (AIMS) available through the National Association of Home Builders (NAHB) uses a source emitter, which releases a perfluorocarbon tracer. A capillary adsorption tube serves as the passive monitor and is analyzed by a GC/ECD and a uniform concentration is determined in a short period of time (USEPA, 1991).

4.3 Effective Diffusion Coefficient (D_T^{eff})

The effective diffusion coefficient contributes to model variance through the vapour-filled porosity term (ϵ_v). Algorithms that model gas movement in porous media are typically sensitive to the effective diffusion coefficient (Jin & Jury, 1996; Johnson et al., 1998). The effective diffusion coefficient can be determined empirically for each layer between the source and foundation by Eq 2.19 if soil porosity data are known from field measurements or soil surveys. The overall effective diffusion coefficient for the region between the source and foundation can be calculated by Eq 2.18. However, empirically derived values for D_T^{eff} may also contribute an unacceptable amount of uncertainty in model predictions (Johnson et al., 1998). D_T^{eff} can be calculated by measuring sulfur hexafluoride (SF_6) as a tracer gas in field experiments. Johnson et al. (1998) outline a protocol for this method. A limitation of this method is its use in low permeability or wet soils. Furthermore, radial symmetry is assumed and therefore may not be appropriate for regions where large changes in soil properties occur over small distances (Johnson et al., 1998). These conditions are typical of many contaminanted sites where soils have been disturbed for the placement of foundations or other activities.

4.4 Soil Vapour Permeability (k_v)

Soil vapour permeability (k_v) exhibits the highest contribution to variance in the calculation of the equivalent Pe number (Eq 2.39). Soil vapour permeability was also found to be a major factor influencing advective transport of soil gas in a study of VOC migration into basements (Fischer & Uchrin, 1996). Because soil vapour permeability is sensitive to variations in soil structure, interpretation of field data is difficult. Despite its limitations, the use of soil core samples for laboratory measurement of soil vapur permeability is preferred (Grant & Groenevelt, 1993). Soil vapour permeability can be measured in the field or it can be measured in the lab by the constant pressure gradient method or by the constant flux method described below. Figure 4.1 may also be used as a preliminary estimate of soil permeability based on general knowledge of soil texture.

Constant Pressure Gradient Method

The constant pressure gradient laboratory method involves exposure of a soil core, placed in a cylindrical ring, to a large volume of air under a constant pressure and above atmospheric pressure. The constant pressure gradient method is ideal for testing highly permeable samples or for samples held at water potentials close to zero. The volume of air that passes through the soil core with time is measured and soil vapour permeability can be calculated for small air pressures by the following equation:

$$\frac{Q}{A} = k_v \frac{\rho_w g}{\eta} \frac{\Delta h}{L} \tag{4.1}$$

where, Q is the volume of air measured at the inlet of the soil core with units of (m^3/s), A is the cross-sectional area of the soil core (m^2), L is the length of the soil core (m), Δh is the air pressure difference over the soil core between the air inlet side and the air outlet side at atmospheric pressure (m), g is the gravitational constant (m^2/s) and η is the viscosity of the gas (kg/m-s) (Grant & Groenevelt, 1993).

Constant Flux Method

The constant flux laboratory method imposes a constant air flux across a soil core sample and measures the resulting pressure difference across the sample. The tools required for this method are easily obtained since a small cylinder and flow meter supplying compressed air are sufficient for analysis. During analysis, steady state air flow has a chance of occurring; in this situation, the pressure gradient can become very large (i.e. $\Delta h > 0.2$ m H_2O) and soil vapour permeability can be calculated by the following equation:

$$\frac{Q}{A} = K\frac{\Delta h}{L}[1 - \frac{\Delta h}{2H_1}] \tag{4.2}$$

where,

$$K = \rho_w g\frac{k_v}{\eta} \quad [m/s] \tag{4.3}$$

where, ρ_w is the density of water, and η is the gas viscosity and all other parameters have been described previously.

The main advantages of the constant flux method are that the apparatus is simple, and measurements can be made on samples of low permeability using either steady state or transient state flow conditions (Grant & Groenevelt, 1993). Grant and Groenevelt (1993) provide detailed methodology for determining soil permeability in the laboratory.

Field Measurement of Soil Vapour Permeability

Soil vapour permeability can be measured in the field through the use of soil probes. In this method, a steady flow of soil gas is drawn from the soil probe while measuring the disturbance pressure difference between the probe and the soil surface (Garbesi et al., 1993). If Darcy flow of gas is assumed, then soil permeability (m^2) can be determined by the following relationship:

$$k_v = \frac{Q\mu}{S\Delta P} \tag{4.4}$$

where, Q is the rate of soil air drawn from the probe (m^3/s), μ is the dynamic viscosity of air (1.75×10^{-5} Pa s at ambient conditions), ΔP is the disturbance pressure difference between the probe tip and the soil surface (in Pa) and S is the shape factor; a factor describing the shape and size of the porous section of the probe (Garbesi et al., 1993; Mosley et al., 1996).

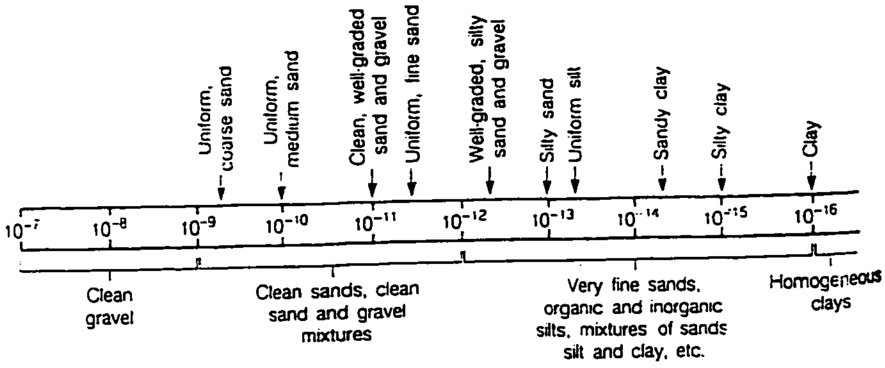

Figure 4.1: Permeability (m^2) of representative soil textures. (Nazaroff, 1992)

4.5 Indoor-Outdoor Pressure Differential (ΔP)

Pressure differentials are an important parameter in predicting if soil gas transport is dominated by advection. Pressure driven soil gas entry into buildings is the result of depressurization of the building substructure with respect to the surrounding soil. The main causes of basement depressurization include thermal differences between indoors and outdoors, wind loading on the building superstructure, imbalanced building ventilation, and barometric pressure fluctuations (Garbesi et al., 1993).

Stack effects result in pressure differentials and occur as a result of warm air rising, which escapes to upper levels and results in a low pressure in the basement causing soil air to enter into the house. Mechanical ventilation systems, fireplaces, oil and gas furnaces also contribute to pressure differentials since air that has been exhausted may be replaced by soil gases that are drawn from the basement. Furthermore, wind loading on houses and

House Type	Typical ΔP Mild Winter	Typical ΔP Moderate Winter	Typical ΔP Severe Winter
Slab-on-grade (no chimney)	10	20	30
Slab-on-grade (chimney)	30	40	50
1 or 2 storey (no chimney)	40	50	60
1 or 2 storey (chimney)	80	90	100
3 storey (no chimney)	70	80	90
3 storey(chimney)	130	140	150

Table 4.3: Maximum Pressure Difference Across Below Grade Building Envelopes (g-cm/s^2) (adapted from Fugler, 1997.)

changes in barometric pressure can also influence soil gas flow into buildings (Fugler, 1997). Because pressure differentials are uncontrolled and temporally variable, detailed studies at occupied sites are difficult to carry out due the invasive nature of the instrumentation needed to probe the site for information on soil, structure, and atmospheric and meteorological conditions (Garbesi et al., 1993).

Table 4.3 gives values for typical maximum winter pressure differentials for houses. The Canadian Mortgage and Housing Corporation suggests that the following adjustments should be made to the values in Table 4.3. If the house in question has a fresh air intake or a combustion air supply, then the pressure differentials should be reduced by 20 g-cm/s^2. If the house in question has a fireplace, central exhuast system or other large exhaust equipment, then the differential pressures given in Table 4.3 should be increased by 20 g-cm/s^2 (Fugler, 1997).

4.6 Time to Reach Steady State Concentrations

For risk based predictions of future indoor air quality impacts, consideration of near steady state soil gas concentrations is desirable. The time, τ_{ss} (days), required to reach near-steady vapour concentrations and fluxes at any distance L_T (m) from a source increases with the square of the distance from the source and is affected by contaminant chemical properties. τ_{ss} may be estimated by the following equation (Johnson et al., 1999):

$$\tau_{ss} \approx \frac{R_v \epsilon_v L_T^2}{D^{eff}} \tag{4.5}$$

where, R_v is the soil vapour-phase retardation factor (unitless), ϵ_v is the vapour filled porosity (m^3 vapour/ m^3 soil), and D^{eff} is the effective vapour diffusion coefficient (m^2/d). Different chemicals have different soil vapour-phase retardation factors. Monoaromatic hydrocarbons have vapour-phase retardation factors in the range of $10<R_v<100$; therefore, different chemicals approach near-steady concentrations at different times (Johnson et al., 1999).

For long-term predictions of indoor air quality at a contaminated site, measurement of soil gas concentrations near the source are most representative of near-steady state conditions; whereas, soil-gas measurements taken further away from the source may not be representative of near-steady conditions. If biodegradation and weathering of contaminants are taken into account, the time required to reach near steady conditions may be faster. Significant advection also decreases the time required to reach near steady conditions. Such a case could involve advective flow through karst formations and other permeable conduits (Johnson et al., 1999).

In cases where a site has been recently impacted by petroleum hydrocarbons, measurement of soil gas concentrations near the source are recommended if the subsurface geology is well known. If soil gas measurements are taken near the surface or foundation which are several meters from the contaminant source, the time since contaminant release must be taken into account (Johnson et al., 1999).

4.7 Seasonal Variations

Seasonal and temperature variations influence soil gas concentrations measured in the field. The time of year that soil gas measurements are taken should be considered when surveying a contaminated site. Sampling during Winter or early Spring is believed to represent a worst case evaluation due to building depressurization as a result of combustion furnaces and stack effects, lack of building ventilation, presence of frost layer impeding diffusion to the atmosphere, and/or presence of a high groundwater table (Spring). Soil gas VOC concentrations in the vadose zone may be affected by temperature fluctuations due to the Henry's Law constant dependence on temperature (Thomson et al., 1997). Although temperature does cause VOC concentrations to fluctuate, calculations using the Clapeyron equation comparing concentrations of volatile compounds at 10 °C and 25 °C demonstrated only small concentration differences (O. Bose, personal communication, November 2000). The Clapeyron equation is used to estimate the dimensionless form of the Henry's law constant at the system temperature (USEPA, 1997).

Soil moisture also varies seasonally; as soil moisture increases, diffusion of gas through the soil is slowed since water-filled pores increases diffusional path length, which decreases the concentration gradient (Washington, 1996). Increasing soil temperature leads to increased soil moisture losses resulting in increased gas-filled porosity available for soil gas exchange with atmospheric air (Chen & Thomas, 1994). During the night and morning, the near-surface soil becomes less sorptive and releases VOCs to the atmosphere. This is due to moisture that returns to the soil via water transport from below and from the presence of dew and increased atmospheric humidity. This daily cycling phenomenon may affect the atmospheric transport of VOCs, exposures to air pollutants and surface air fluxes (Batterman et al., 1995). Meteorological changes can also influence soil gas flux through changes in barometric pressure, temperature, and moisture content. High pressure systems present at the time of sampling may result in a lower volatile flux; therefore a lower soil gas concentration may be measured than during a low pressure system (Tillman et al., 1989b).

4.8 Subsurface Variations

Variability in subsurface conditions can also influence measured soil gas concentrations resulting in false positive or false negative results with respect to underlying contamination (Kerfoot, 1990). Karst formations and man made high permeability zones in the subsurface can serve as conduits for soil vapour transport and must be considered in site specific assessments. Such karst formations or cracks in the subsurface may result in detection of high soil gas concentrations which are not representative of near source vapour concentrations.

Clay lenses, impermeable frost layers, perched water tables, and paved surfaces can cause build up of subsurface vapour concentrations. If soil gas measurements are taken above such impermeable layers, then soil vapour detection will likely underpredict actual concentrations. Soil gas measurements taken below such impermeable layers may overpredict actual concentrations.

4.8.1 Biological and Chemical Processes

Two of the processes not addressed in the Johnson and Ettinger (1991) model are biological and chemical transformations. A study by the Massachusets Department of Environmental Protection found that soil quality guidelines and groundwater quality guidelines developed by the Johnson and Ettinger (1991) model underpredicted indoor air quality levels of chlorinated VOCs, but were conservative (i.e. overpredicted) for indoor air quality levels for petroleum hydrocarbons. Since chlorinated VOCs do not undergo biodegradation as readily as petroleum hydrocarbon VOCs, this result is not surprising, but is an important factor to consider.

Chapter 5

Risk-based Screening Levels and Remediation Objectives

In this section, risk-based screening levels for volatile petroleum hydrocarbons in soil were derived using the Johnson and Ettinger (1991) model. Chemical and physical characteristics of volatile petroleum hydrocarbon fractions were based on those referenced by Canadian Council Ministers of the Environment (CCME) (CCME, 2000). These screening levels and remediation objectives are recommended for use as decision-making criteria during Phase II environmental site assessments of petroleum hydrocarbon contaminated sites in Alberta. An overview of a suggested preliminary screening assessment and refined assessment approaches are shown in Table 5.1.

Preliminary Assessment	Soil and Groundwater (Tables 5.13 to 5.15)	Soil and/or groundwater remediation objectives based on CCME input parameter conditions for L_T,D_T^{eff}, ACH, and other parameters
	Soil Vapour (Tables 5.10 to 5.11)	Field vapour screening of soil gas samples taken near surface and/or subfoundation based on CCME input parameter conditions for L_T,D_T^{eff}, ACH, and other parameters
		Laboratory vapour screening of soil gas samples taken near surface and/or subfoundation based on CCME input parameter conditions for L_T,D_T^{eff}, ACH, and other parameters
Refined Assessment	Soil and Groundwater (comparisons made to look-up figures in Appendix F to Appendix M reflecting site-specific conditions)	Soil and/or groundwater remediation objectives look-up figures for L_T, D_T^{eff}, and ACH and default CCME assumptions for other parameters
	Indoor Air (Table 5.12)	Screening of indoor air samples taken inside structure at a representative location (within 3" of foundation) based on CCME input parameter conditions for L_T,D_T^{eff}, ACH, and other parameters

Table 5.1: Overview of Preliminary and Refined Assessment Framework Approach for Petroleum Hydrocarbon Contaminated Sites in Alberta

5.1 CCME Model Parameters

Risk based screening levels and remediation objectives for petroleum hydrocarbon VOCs were derived based on values proposed for model input parameters by CCME (CCME, 2000). These values were adopted to be consistent with CCME in the development of the Canada-Wide Standards for Petroleum Hydrocarbons in Soil. The most sensitive parameters identified by the sensitivity analysis are also evaluated in terms of those values proposed by CCME and recommendations for improvement are contained in this section. Tables 5.2 to 5.7 list CCME proposed input values used to derive screening levels and remediation objectives.

It should be noted that the total porosity value of 0.3 proposed for fine-grained soils by CCME is incorrect. The correct value for total porosity for fine-grained soils is 0.47 based on a soil bulk density of 1.4 g/cm^3 and a particle density of 2.65 g/cm^3. It is recommended that CCME reexamine the rationale for choosing this value.

Input Parameter	Symbol	Coarse-Grained Soil	Fine-Grained Soil
Organic Carbon Fraction (g/g)	f_{oc}	0.005	0.005
Soil Bulk Density (g/cm^3)	ρ_b	1.7	1.4
Total Soil Porosity	ϵ_T	0.4	0.3
Vapour-filled Porosity	ϵ_v	0.281	0.132
Moisture-filled Porosity	ϵ_w	0.119	0.168
Soil Vapour Permeability to Vapour Flow (cm^2)	k_v	10^{-8}	10^{-9}
Distance from structure to soil contamination (cm)	L_T	30	30

Table 5.2: CCME Assumed Soil and Site Characteristics (CCME, 2000)

Input Parameter	Symbol	Toddler	Adult
Relevant land use(s)		agricultural residential commercial	industrial
Body Weight (kg)	BW	16.5	70.7
Exposure Time (residential)	ET_R	1	1
Exposure Time (commercial)	ET_C	$\frac{10}{24} * \frac{5}{7} * \frac{48}{52}$	$\frac{10}{24} * \frac{5}{7} * \frac{48}{52}$
Exposure Time (industrial)	ET_I	$\frac{10}{24} * \frac{5}{7} * \frac{48}{52}$	$\frac{10}{24} * \frac{5}{7} * \frac{48}{52}$
Exposure Frequency (events/d)	EF	1	1
Inhalation Rate (m^3/d)	IR	9.3	16.2

Table 5.3: CCME Assumed Receptor Characteristics (CCME, 2000)

Input Parameter	Symbol	Residential (with basement)	Residential (slab-on-grade)	Commercial (slab-on-grade)
Building Length (cm)	LB	1225	1225	2000
Building Width (cm)	WB	1225	1225	1500
Building Area (cm^2)	AB	1.50E+06	1.50E+06	3.00E+06
Building Height including basement (cm)	HB	488	488	7300
Thickness of Building Foundation (cm)	L_c	11.25	11.25	11.25
Area of Cracks (cm^2)	A_c	994.5	994.5	1846
Radius of Idealized Cylinder (cm)	r_c	0.2	0.2	0.26
Length of Idealized Cylinder (cm)	X_c	4900	4900	7000
Distance below grade to Idealized Cylinder (cm)	Z_c	244	11.25	11.25
Effective vapour pressure diffusion coefficient through the crack (cm^2/s)	D_c	4.50E-04	4.50E-04	4.50E-04
Air Exchanges per Hour (h^{-1})	ACH	1	1	2
Pressure Differential across building envelope ($g/cm\text{-}s^2$)	ΔP	40	40	20

Table 5.4: CCME Assumed Building Characteristics Indoor Infiltration Pathway (CCME, 2000)

BTEX	Benzene	Toluene	Ethylbenzene	o-Xylene	m-Xylene	p-Xylene
Reference Concentration (RfC) (mg/m^3)	1.82E-03[a]	1.0E+00[b]	4.0E-01[b]	7.0E-0[c]1	7.0E-01[c]	7.0E-01[c]
Background Indoor/ Outdoor Air Concentration(C_a) (mg/m^3)	7.40E-03	3.60E-02	1.10E-02	7.60E-03	2.60E-02	2.60E-02 [d]
Henry's Law Constant[e] (H) (atm-m^3/mol)	5.50E-03	6.64E-03	7.88E-03	5.19E-03	7.34E-03	7.66E-03
Henry's Law Constant[b](H) (dimensionless)	2.25E-01	2.74E-01	3.58E-01	2.28E-01	2.95E-01	2.33E-01
Organic Carbon Partition Coefficient [b](K_{oc}) (mL/g)	5.89E+01	1.82E+02	3.63E+02	3.63E+02	4.07E+02	3.89E+02
Diffusion Coefficient in Air[b] (D^{air}) (cm^2)	8.80E-02	8.70E-02	7.50E-02	8.70E-02	7.00E-02	7.69E-02
Diffusion Coefficient in Water (D^{H_2O}) (cm^2)	n/c	n/c	n/c	n/c	n/c	n/c
Vapour viscosity[f] (μ) (g/cm-s)	7.57E-05	7.12E-05	6.47E-05	5.85E-05	5.59E-05	5.23E-05
Molecular weight (g/mol)	7.81E+01	9.21E+01	1.06E+02	1.06E+02	1.06E+02	1.06E+02
Solubility[b] (S) (mg/L)	1.75E+03	5.26E+02	1.69E+02	1.78E+02	1.61E+02	1.85E+02

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[a] based on a 70.7 kg adultand an inhalation rate of 16.2 m^3/d

[b] Gustafson, 1997

[c] Scorecard, 2000

[d] Otson & Fellin, 1991

[e] Chemfate Database

[f] Yaws,1999

Table 5.5: Assumed Physical and Chemical Parameters for BTEX Compounds

Aromatics	$C_{>8}$-C_{10}	$C_{>10}$-C_{12}	$C_{>12}$-C_{16}
Reference Concentration (RfC) (mg/m^3)	2.00E-01	2.00E-01	2.00E-01
Background Indoor/ Outdoor Air Concentration (C_a) (mg/m^3)	3.75E-02	0	0
Henry's Law Constant (H) (atm-m^3/mol)	1.20E-02	3.40E-03	1.30E-04
Henry's Law Constant (H) (dimensionless)	4.98E-01	1.41E-01	5.39E-03
Organic Carbon Partition Coefficient (K_{oc}) (mL/g)	1.58E+03	2.51E+03	5.01E+03
Diffusion Coefficient in Air (D^{air}) (cm^2)	5.00E-02	5.00E-02	5.00E-02
Diffusion Coefficient in Water (D^{H_2O}) (cm^2)	n/c	n/c	n/c
Vapour viscosity (μ) (g/cm-s)	1.73E-04	1.73E-04	1.73E-04
Molecular weight (MW) (g/mol)	1.20E+02	1.30E+02	1.50E+02
Solubility (S) (mg/L)	6.50E+01	2.50E+01	5.80

Table 5.6: Assumed Physical Chemical Parameters for Aromatic Hydrocarbon Fractions (CCME, 2000)

5.1.1 Distance from Source to Foundation

The distance from the source to the foundation (L_T) was set to 30 cm for all building structure types. This value forces conservatism in the development of soil screening and remediation objectives for a contaminant source located further from the building because it would result in lower allowable levels of indoor air contaminants. Soil gas measurements taken at the source combined with an input of $L_T = 30$ cm will provide a conservative estimate of indoor contaminant vapour concentrations. CCME has made an arbitrary assumption that if $L_T \gg 30$ cm, the condition that $\frac{r_c}{Z_c} \ll 1$ would be fulfilled. This condition is derived from Nazaroff, 1988 for the calculation of Q_s (Eq 2.22). The model should not be applied for $L_T < 30$ cm because it is uncertain if boundary conditions for the model are still applicable.

Aliphatics	C_6-C_8	$C_{>8}$-C_{10}	$C_{>10}$-C_{12}	$C_{>12}$-C_{16}
Reference Concentration (RfC) (mg/m^3)	1.84E+01	1.00E+00	1.00E+00	1.00E+00
Background Indoor/ Outdoor Air Concentration (C_a) (mg/m^3)	9.11E-02	3.88E-02	0	0
Henry's Law Constant (H) (atm-m^3/mol)	1.20	1.90	2.90	1.25E+01
Henry's Law Constant (H) (dimensionless)	4.98E+01	7.88E+01	1.20E+02	5.18E+02
Organic Carbon Partition Coefficient (K_{oc}) (mL/g)	3.98E+03	3.16E+04	2.51E+05	5.01E+06
Diffusion Coefficient in Air (D^{air}) (cm^2)	5.00E-02	5.00E-02	5.00E-02	5.00E-02
Diffusion Coefficient in Water (D^{H_2O}) (cm^2)	n/c	n/c	n/c	n/c
Vapour viscosity (μ) (g/cm-s)	1.73E-04	1.73E-04	1.73E-04	1.73E-04
Molecular weight(MW) (g/mol)	1.00E+02	1.30E+02	1.60E+02	2.00E+02
Solubility (S) (mg/L)	5.4	4.30E-01	3.40E-02	7.60E-04

Table 5.7: Physical Chemical Parameters for Aliphatic Hydrocarbon Fractions (CCME, 2000)

5.1.2 Air Exchange Rate

An air exchange rate of 1 h^{-1} or 2 h^{-1} is recommended by CCME in the calculation of the attenuation coefficient. However, for newer, energy efficient homes in northern climates with prolonged heating seasons, the air exchange rate may be as low as 0.05–0.1 h^{-1} (Fugler, 1997). A study of air exchange rates in homes in the Pacific Northwest found that the average air exchange rates are estimated to be between 0.40 and 0.45 h^{-1} during the heating season (Parker et al., 1990). Overall air exchange rates for homes in the United States have been reported to have an arithmetic mean of 2.0 h^{-1} with a standard deviation of 3.3 h^{-1}, and a range of 0.04 to 45.6 h^{-1}. In this study, homes in the Northwest had an arithmetic mean of 0.42 h^{-1} with a standard deviation of 0.3 h^{-1} (Pandian et al., 1993). If higher air exchange rates are used in making model predictions, indoor air contaminant concentrations may be underpredicted resulting in soil screening and remediation objectives, which may lead to indoor air quality levels exceeding Reference Concentrations.

5.1.3 Soil Vapour Permeability

Soil vapour permeability (k_v) was found to have the greatest contribution to variance for determining whether advective of diffusive transport dominates (Eq 2.39). The soil vapour permeability chosen by CCME to represent coarse-grained soils (10^{-8} cm^2) is representative of permeabilities for well-graded, silty sands and gravels (CCME, 2000). The fine-grained soil vapour permeability assumed by CCME (10^{-9} cm^2) is representative of silty sands (Nazaroff, 1992).

A much more conservative soil vapour permeability for coarse-grained soils would be in the order of 10^{-5} to 10^{-3} cm^2, which is typical of clean gravels. The CCME parameters for coarse-grained soils result in an equivalent Pe of 229 for a building with a basement and 154 for a slab on grade structure. CCME parameters for fine-grained soils result in an equivalent Pe of 22.9 for a building with a basement and 15.4 for a building with a slab-on-grade foundation.

If soil permeabilities for coarse-grained soils typical of clean gravels were used for a building with a basement, an equivalent Pe of 230 000 and 23 000 000 is generated for soil permeabilities of 10^{-5} cm^2 and 10^{-3} cm^2, respectively. For slab on grade structures, an equivalent Pe of 150 000 and 15 000 000 is calculated for soil permeabilities of 10^{-5} cm^2 and 10^{-3} cm^2, respectively. If soil permeabilities for fine-grained soils ranging from sandy clays ($k_v = 10^{-10}$ cm^2) to clays ($k_v = 10^{-12}$ cm^2) were used for a building with a basement, an equivalent Pe ranging from 2.3 to 0.023 would result. For slab on grade structures, an equivalent Pe of ranging from 1.5 to 0.015 would result for fine-grained soils ranging from silty sands to clays.

The soil permeabilities assigned to coarse-grained soils by CCME for developing soil quality guidelines do not represent clean gravels and therefore may not provide sufficient conservatism for this soil type. Furthermore, the fine-grained soil permeability value used by CCME is of a coarse texture and does not warrant the use of Eq 2.43. Values for fine-grained soil parameters assigned by CCME result in an equivalent Pe number > 1 indicating that advective transport is still very much the dominant transport mechanism. As stated in Section 2.2.6, diffusive soil gas transport dominates when $\frac{Q_s L_c}{D_c A_c} \rightarrow 0$. Clearly, this is not the case for the CCME soil vapour permeabilities assigned to fine-grained soils.

5.1.4 Total Soil Porosity

CCME recommended total soil porosity values may also not be representative of environmental soil conditions at actual sites. Total porosity values for coarse-grained soils are higher ($n = 0.4$) compared to fine-grained soils ($n = 0.3$) in the CCME recommended values. However, in nature, coarse-textured soils generally have lower total porosities than fine-textured soils (Hillell, 1982). Although the sensitivity analysis did not identify total soil porosity as a highly sensitive parameter, it is still important that these values reflect site conditions.

5.2 Screening Level Objectives

As a preliminary assessment, near surface and subfoundation vapour measurements can be used to identify if there are any immediate indoor air impacts to a structure in the vicinity of a contaminated site. Soil gas sample collection can be performed with ease using vapour sampling probes and a Flame Ionization Detector (FID) or Photoionization Detector (PID).

The attenuation coefficient was calculated for Fraction 1 (C_6–C_{10}), Fraction 2 ($C_{>10}$–C_{16}) aliphatic and aromatic petroleum hydrocarbons, and BTEX compounds using CCME recommended parameters for chemical, building, soil, site, and receptor characteristics (Tables 5.2 to 5.7) and based on a contaminant source located in the soil 30 cm from the foundation. For comparison, both Eq 2.38 and Eq 2.43 were used to calculate the attenuation coefficient for fine-grained soils. Table 5.8 illustrates calculated attenuation coefficient values for CCME defined coarse-grained and fine-grained soils using Eq 2.38. Table 5.9 illustrates the attenuation coefficient values for fine-grained soils using Eq 2.43. Use of Eq 2.43 to estimate the attenuation coefficient for fine-grained soils (Table 5.9) results in values more than 100 times smaller than use of Eq 2.38 (Table 5.8) for the same CCME input parameter values. Given that Peclet numbers for the fine-grained soils case indicate that advection is still very much the dominant transport mechanism (see Section 5.1.3) attenuation coefficient values based on Eq 2.38 are a better representation of the CCME input parameter conditions. For the case where the Pe number ($\frac{Q_s L_c}{D_c A_c}$) $\rightarrow 0$ (or $0 \ll 1$) one would get the same result for the attenuation coefficient using Eq 2.39 and Eq 2.38. This is definitely not the case for the CCME input parameter conditions (i.e. CCME input parameter conditions for the fine-grained soil case does not reflect situations where diffusion is the rate-controlling mass transport step and advective transport is negligble). Therefore, one should not use Eq 2.43 for the fine-grained soils case for CCME input parameter conditions. However, for site-specific cases; Eq 2.43 may be used where it can be shown that $(\frac{Q_s L_c}{D_c A_c}) \ll 1$, i.e. ≤ 0.1 for the fine-grained soil case.

	coarse-grained soils Commercial Slab on Grade	coarse-grained soils Building with Basement	fine-grained soils Commercial Slab on Grade	fine-grained soils Building with Basement
BTEX				
Benzene	3.93E-04	6.75E-04	4.25E-05	7.53E-05
Toluene	4.11E-04	7.00E-04	4.46E-05	7.85E-05
Ethylbenzene	4.22E-04	7.00E-04	4.65E-05	8.02E-05
o-Xylene	4.73E-04	7.90E-04	5.20E-05	9.01E-05
m-Xylene	4.54E-04	7.37E-04	5.10E-05	8.60E-05
p-Xylene	4.90E-04	7.98E-04	5.49E-05	9.28E-05
Aliphatics				
C_6-C_8	1.82E-04	3.21E-04	1.94E-05	3.51E-05
$C_{>8}$-C_{10}	1.82E-04	3.21E-04	1.94E-05	3.51E-05
$C>_{10}$-C_{12}	1.82E-04	3.21E-04	1.94E-05	3.51E-05
$C>_{12}$-C_{16}	1.82E-04	3.21E-04	1.94E-05	3.51E-05
Aromatics				
$C_{>8}$-C_{10}	1.82E-04	3.21E-04	1.94E-05	3.51E-05
$C>_{10}$-C_{12}	1.82E-04	3.21E-04	1.94E-05	3.51E-05
$C>_{12}$-C_{16}	1.82E-04	3.21E-04	1.94E-05	3.51E-05

Table 5.8: Attenuation Coefficients Used to Develop Soil Gas Screening Levels Using Eq 2.38

5.2.1 Field Soil Vapour Screening Levels

An acceptable screening-level soil vapour concentration protective of human health in the indoor inhalation pathway was calculated by the following equation:

$$C_{sg} = \frac{(RfC - C_a)}{\alpha} \tag{5.1}$$

	Slab on Grade	Building with Basement
BTEX		
Benzene	1.48E-07	1.96E-07
Toluene	1.48E-07	1.96E-07
Ethylbenzene	1.48E-07	1.95E-07
o-Xylene	1.48E-07	1.96E-07
m-Xylene	1.48E-07	1.95E-07
p-Xylene	1.48E-07	1.95E-07
Aliphatics		
C_6-C_8	1.48E-07	1.95E-07
$C_{>}8$-C_{10}	1.48E-07	1.95E-07
$C>_{10}$-C_{12}	1.48E-07	1.95E-07
$C>_{12}$-C_{16}	1.48E-07	1.95E-07
Aromatics		
$C_{>}8$-C_{10}	1.48E-07	1.95E-07
$C>_{10}$-C_{12}	1.48E-07	1.95E-07
$C>_{12}$-C_{16}	1.48E-07	1.95E-07

Table 5.9: Attenuation Coefficients Using CCME Approach for Diffusion Limited Case (Eq 2.43)

where, C_{sg} is soil gas screening level beneath the foundation (mg/m^3), RfC is the reference concentration for the target analyte (mg/m^3), C_a is the background indoor/outdoor air concentration of the target analyte (mg/m^3), and α is the attenuation coefficient calculated using Eq 2.38 for both CCME defined coarse- and fine-grained soils. The method used is similar to the Massachusetts Department of Environmental Protection (MADEP) approach to demonstrate no adverse impacts to indoor air (MADEP, 1997b).

Soil gas screening levels for carcinogens (i.e. benzene) were calculated by:

$$C_{sg} = \frac{(\mathrm{RSD})(\mathrm{BW})}{(\mathrm{IR})(\alpha)} \tag{5.2}$$

where, BW is the body weight (70.7 kg), IR is the inhalation rate (16.2 m^3/d) of an adult receptor, and RSD is the risk specific dose. The RSD which is determined by the following equation:

$$\mathrm{RSD} = \frac{\mathrm{ER}}{q_1^*} \tag{5.3}$$

where, ER (Excess Risk) is equal to a specific risk level of 1 in 100,000 lifetime risk of cancer, and q_1^* is the oral slope factor for benzene, which is equal to 1.5 x 10 $^{-2}$ to 5.5 x 10 $^{-2}$ mg/kg/day (USEPA, 2000a). Using the range of q_1^* above, a middle value was calculated for the benzene RSD equal to 4.2 x 10 $^{-4}$ mg/kg/day. For a 70.7 kg adult with an inhalation rate of 16.2 m^3/day, the resulting RfC for benzene is equal to 1.8 $\mu g/m^3$. The recommended RfC for benzene at a risk level of 1 in 100 000 is 1.3 to 4.5 $\mu g/m^3$ (USEPA, 2000a).

Table 5.10 illustrates soil gas screening levels for coarse-grained soils estimated using Eq 2.38. Table 5.10 can be used to screen for potential impacts to indoor air quality as a result of contamination in coarse-grained soils. Soil gas concentrations above values listed in Table 5.10 may imply impacts to indoor air quality. However, enclosed spaces with lower air exchange rates (such as newer, energy efficient homes) may have greater indoor air concentrations. The values in Table 5.10 were designed to screen for levels of Fraction 1 and Fraction 2 hydrocarbons and BTEX compounds using total organic vapour detectors such as Flame Ionization Detectors (FID) and Photoionization Detectors (PID); therefore it must be assumed that each fraction represents 100% of the total organic vapour reading. However, it is important to note that since PIDs can only distinguish aromatic hydrocarbons may underestimate petroleum hydrocarbon concentrations. Section E.1 in Appendix E provides a discussion of FID and PID instrumentation for screening level analyses.

	Building with Basement[a]	Slab on Grade Structure[a]	Building with Basement[b]	Slab on Grade Structure[b]
BTEX				
Benzene	9.11E-01	1.56+00	8.16E+00	1.45E+01
Toluene	1.38E+02	2.35E+02	1.23E+03	2.17E+03
Ethylbenzene	3.26E+02	5.41E+02	2.84E+03	4.90E+03
o-Xylene	2.02E+02	3.38E+02	1.77E+03	3.07E+03
m-Xylene	2.11E+02	3.42E+02	1.81E+03	3.05E+03
p-Xylene	1.95E+02	3.17E+02	1.67E+03	2.83E+03
Aliphatics				
C_6-C_8	1.40E+04	2.45E+04	1.28E+05	2.31E+05
$C_{>8}$-C_{10}	5.64E+02	9.91E+02	5.15E+03	9.31E+03
$C>_{10}$-C_{12}	4.77E+02	8.38E+02	4.35E+-3	7.87E+03
$C>_{12}$-C_{16}	3.81E+02	6.70E+02	3.48E+03	6.30E+03
Aromatics				
$C_{>8}$-C_{10}	1.03E+02	1.82E+02	9.43E+02	1.71E+03
$C>_{10}$-C_{12}	1.17E+02	2.06E+02	1.07E+03	1.94E+03
$C>_{12}$-C_{16}	1.02E+02	1.79E+02	9.29E+02	1.68E+03

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[a] coarse-grained soils

[b] fine-grained soils

Table 5.10: Soil Gas Screening Levels (ppm(v))

5.2.2 Laboratory Soil Vapour Screening Levels

If the potential for indoor air impacts cannot be excluded from preliminary field soil gas screening methods, then further laboratory analytical testing is recommended. Laboratory soil vapour screening levels were developed simply by converting values obtained for field soil vapour screening levels by Eq 5.1 and 5.2 from ppm to mg/m^3.

Soil gas measurements for laboratory analysis can be sampled by a variety of methods, which are discussed in Appendix B. Methods employed for preconcentration of VOCs include sorbent sampling and collection in stainless steel canisters with subsequent analysis by GC. EPA methods TO-1, TO-2, TO-14, and TO-17 describe these methods in detail (USEPA, 1999a). However, for soil gas sampling, sample introduction into analytical equipment

	Building with Basement[a]	Slab on Grade Structure[a]	Building with Basement[b]	Slab on Grade Structure[b]
BTEX				
Benzene	2.91E+00	4.99E+00	2.61E+01	4.62E+01
Toluene	5.20E+02	8.86+02	4.64E+03	8.17E+03
Ethylbenzene	1.41E+03	2.35E+03	1.23E+04	2.13E+04
o-Xylene	8.76E+02	1.46E+03	7.69E+03	1.33E+04
m-Xylene	9.14E+02	1.48E+03	7.84E+03	1.32E+04
p-Xylene	8.45E+02	1.37E+03	7.26E+03	1.23E+04
Aliphatics				
C_6-C_8	5.71E+04	1.00E+05	5.22E+05	9.43E+05
$C_{>8}$-C_{10}	3.00E+03	5.27E+03	2.74E+04	4.95E+04
$C_{>10}$-C_{12}	3.12E+03	5.48E+03	2.85E+04	5.15E+04
$C_{>12}$-C_{16}	3.12E+03	5.48E+03	2.85E+04	5.15E+04
Aromatics				
$C_{>8}$-C_{10}	5.07E+02	8.91E+02	4.63E+03	8.37E+03
$C_{>10}$-C_{12}	6.24E+02	1.10E+03	5.70E+03	1.03E+04
$C_{>12}$-C_{16}	6.24E+02	1.10E+03	5.70E+03	1.03E+04

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[a] coarse-grained soils

[b] fine-grained soils

Table 5.11: Soil Gas Fractional Concentrations (mg/m^3)

can be modified to direct injection/desorption (MADEP, 1997b).

The concentration of each fraction can be compared to the soil gas action levels. Table 5.11 can be used to screen for soil gas in coarse-grained soils; measurements above these levels may indicate impacts to indoor air quality.

5.2.3 Indoor Air Quality Screening Levels

Indoor air quality measurements are recommended if field and laboratory soil gas analysis is unable rule out potential indoor air quality impacts. Table 5.12 provides screening levels for indoor air quality. These values were derived for Fraction 1 and Fraction 2 hydrocarbons and TEXs by the following equation

$$\mathrm{RfC} - \mathrm{C_a} \tag{5.4}$$

and for benzene by

$$\frac{\mathrm{RSD} \times \mathrm{BW}}{\mathrm{IR}} \tag{5.5}$$

where, RSD is the Risk Specific Dose (mg/kg) and BW is the body weight of an adult (70.7 kg), and IR is the inhalation rate of an adult (16.2 m^3/d).

Samples can be collected by preconcentration on sorbent tubes or collection in canisters as described in USEPA methods TO-1, TO-2, TO-14, and TO-17. Sample analysis should be performed using GC procedures. Appendix C provides a discussion on indoor air sampling methods.

5.3 Risk-Based Remediation Objectives

Remediation objectives for soil and groundwater were derived by following procedures based on CCME soil quality guidelines for human health in the indoor inhalation pathway for petroleum hydrocabon fractions in soil (CCME, 2000). Soil Quality Guidelines for agricultural, residential, commercial, and industrial land uses were determined using CCME input parameters (Tables 5.2 to 5.7). Eq 2.38 was used to calculate the attenuation coefficient for both CCME defined coarse- and fine-grained soils.

5.3.1 Soil Remediation Objectives

Soil remediation objectives were determined by calculating the soil quality guideline (SQG) for the indoor inhalation pathway (CCME, 2000):

$$\mathrm{SQG_{ii,b}} = \frac{(\mathrm{RfC} - \mathrm{C_a})[\epsilon_\mathrm{m} + (\mathrm{K_{oc}})(\mathrm{f_{oc}})(\rho_\mathrm{b}) + \frac{\mathrm{H}}{\mathrm{RT}}\epsilon_\mathrm{v}](\mathrm{SAF})(\mathrm{DF_i})(10^3\,\frac{\mathrm{g}}{\mathrm{kg}})}{\frac{\mathrm{H}}{\mathrm{RT}}(\rho_\mathrm{b})(\mathrm{ET})(10^6\,\frac{\mathrm{cm^3}}{\mathrm{m^3}})} + \mathrm{BSC} \tag{5.6}$$

where, $\mathrm{SQG_{ii,b}}$ is the soil quality guideline by indoor infiltration for volatile petroleum hydrocarbons using RfC (mg/m^3), RfC is the reference air concentration (mg/m^3), C_a is the background indoor/outdoor air concentration (mg/m^3), SAF is a soil allocation factor (0.5, unitless) used to reflect the fact that only up to 50% of total inhalation exposure to a compound is allocated to soil (after CCME, 2000), ϵ_m is the moisture filled porosity (unitless), ϵ_v is the vapour filled porosity (unitless), K_{oc} is the organic carbon partition coefficient (mL/g), f_{oc} is the fraction organic carbon (g/g), ρ_b is the dry bulk density (g/cm^3), H is the Henry's Law Constant (atm-m^2/s), R is the universal gas constant (8.2×10^{-5} atm-m^3/mol-K), T is the absolute temperature (294.15 K), DF_i is a dilution factor, which is the inverse of the attenuation coefficient (α, unitless), ET is the exposure term (unitless), and BSC is the background soil concentration (mg/kg). Tables 5.13 to 5.14 illustrate the SQG derived using CCME input parameters.

Soil remediation objectives for carcinogens (i.e. benzene) were determined by

$$\mathrm{SQG}_{ii,b} = \frac{(\mathrm{RSD})(\mathrm{BW})[\epsilon_m + (K_{oc})(f_{oc})(\rho_b) + \frac{H}{RT}\epsilon_v](\mathrm{DF}_i)(10^3\frac{g}{kg})}{(\mathrm{IR})\frac{H}{RT}(\rho_b)(\mathrm{ET})(10^6\frac{cm^3}{m^3})} + \mathrm{BSC} \tag{5.7}$$

where, BW is the body weight (70.7 kg) and IR is the inhalation rate (16.2 m^3/d) of an adult receptor. The SAF was not included in the calculation of SQG for benzene because 50% absorption via inhalation has been accounted for in the calculation of the oral slope factor (USEPA, 2000a).

Because the maximum concentration of a chemical in soil is limited by saturation, the soil saturation concentration (C_{sat}) was calculated. C_{sat} is the chemical concentration in soil corresponding to a point where the sorption limit of soil particles, solubility limits of soil pore water, and saturation of soil pore air have been reached. Therefore, if health risks are acceptable at C_{sat}, then they will not increase above C_{sat}. C_{sat} was calculated by (Gustafson et al., 1997):

$$C_{sat} = \frac{S}{\rho_b} \times [H\theta_a + \theta_w + K_{oc}f_{oc}\rho_b] \tag{5.8}$$

where, S is the water solubility (mg/L) and all other parameters have been defined previously.

5.3.2 Groundwater Remediation Objectives

Groundwater remediation objectives were determined by the following equation (after MADEP, 1997a):

$$GWQG = \frac{(RfC - C_a)(AF)}{(\alpha)(H)(ET)} \times 0.001\frac{m^3}{L} \tag{5.9}$$

where, GWQG is the groundwater quality guideline (mg/L), RfC is the reference concentration (mg/m^3), C_a (mg/m^3) is the background indoor/outdoor air concentration (mg/m^3), AF is an allocation factor (0.5)(after CCME, 2000), α is the attenuation coefficient, H is the dimensionless Henry's Law Constant, and ET is the exposure term (unitless).

For carcinogens (i.e. benzene), groundwater remediation objectives were determined by:

$$GWQG = \frac{(RSD)(BW)}{(IR)(\alpha)(H)} \times 0.001\frac{m^3}{L} \quad . \tag{5.10}$$

Table 5.15 illustrates the groundwater quality guidelines derived using CCME input parameters.

5.3.3 Look-up Figures

To encourage the use of soil remediation objectives in the field based on the indoor infiltration pathway, look-up figures were developed for use under Phase II Environmental Site Assessments. Soil remediation objectives were generated using CCME parameters for soil, building, contaminant, and receptor properties. Those parameters identified as contributing to uncertainty in the prediction of Eq 2.38 (i.e., L_T, D_T^{eff}, ACH) were varied with respect to the SQG or GWQG. Eq 2.38 was used to calculate the attenuation coefficient for both coarse- and fine-grained soils.

For both coarse- and fine-grained soils, the air exchange rate, and $\frac{D_T^{eff}}{L_T}$ result in appreciable differences in estimated guideline values. Figure 5.1 illustrates a look-up figure for the SQG for C_6–C_8 aliphatic hydrocarbons for coarse-grained soils (building with a basement and an agricultural/residential exposure scenario). Look-up figures for remaining conditions modeled are located in Appendices G–K. These look-up figures can be used in the field if the air exchange rate of a building is known, and if $\frac{D_T^{eff}}{L_T}$ can be calculated based on site specific measurements of related parameters. For example, in Figure 5.1, for an air exchange rate of 0.83 h^{-1} and $\frac{D_T^{eff}}{L_T}$ of 1E-04, the soil quality guideline for C_6–C_8 aliphatic hydrocarbons for the scenario described in Figure 5.1 equals 15 mg/kg.

BTEX	
Benzene	1.8E-03[a]
Toluene	3.64E-01
Ethylbenzene	9.89E-01
o-Xylene	6.92E-01
m-Xylene	6.74E-01
p-Xylene	6.74E-01
Aliphatics	
C_6-C_8	1.83E01
$C_{>8}$-C_{10}	9.61E-01
$C>_{10}$-C_{12}	1.00E+00
$C>_{12}$-C_{16}	1.00E+00
Aromatics	
$C_{>8}$-C_{10}	1.63E-01
$C_{>10}$-C_{12}	2.00E-01
$C_{>12}$-C_{16}	2.00E-01

Table 5.12: Indoor Air Quality Action Levels (mg/m^3)

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[a]based on a 70.7 kg adult with an inhalation rate of 16.2 m^3/d

BTEX	Coarse Soils	C_{sat}	Fine Soils	C_{sat}
Benzene	2.09E-03	6.20E+02	3.34E-03	5.80E+02
Toluene	9.69E-01	3.79E+02	8.895E+00	4.09E+02
Ethylbenzene	4.19E+00	2.13E+02	3.71E+01	2.42E+02
o-Xylene	3.91E+00	2.22E+02	3.49E+01	2.52E+02
m-Xylene	3.24E+00	2.23E+02	2.81E+01	2.55E+02
p-Xylene	2.75E+00	2.45E+02	2.39E+01	2.80E+02
Aliphatics				
C_6-C_8	1.62E+01	8.29E+01	1.30E+02	1.10E+02
$C_{>8}$-C_{10}	3.26E+00	4.24E+01	2.88E+01	5.27E+01
$C_{>10}$-C_{12}	1.65E+01	2.54E+01	1.50E+02	3.10E+01
$C_{>12}$-C_{16}	7.57E+01	1.12E+03	6.90E+02	1.37E+01
Aromatics				
$C_{>8}$-C_{10}	4.10E+00	3.16E+02	3.75E+01	1.33E+02
$C_{>10}$-C_{12}	2.80E+01	1.89E+02	2.56E+02	3.78E+02
$C_{>12}$-C_{16}	1.45E+03	8.65E+01	1.33E+04	2.27E+02

Table 5.13: SQG for a Building with Basement, Agricultural/Residential Scenario (mg/kg)

BTEX	Coarse Soils	C_{sat}	Fine Soils	C_{sat}
Benzene	3.26E-02	6.20E+02	3.27E-01	5.80E+02
Toluene	6.12E+00	3.79E+02	5.80E+01	4.09E+02
Ethylbenzene	2.58E+01	2.13E+02	2.37E+02	2.42E+02
o-Xylene	2.42E+01	2.22E+02	2.24E+02	2.52E+02
m-Xylene	1.95E+01	2.23E+02	1.76E+02	2.55E+02
p-Xylene	1.66E+01	2.45E+02	1.50E+02	2.80E+02
Aliphatics				
C_6-C_8	1.05E+02	8.29E+01	8.67E+02	1.10E+02
$C_{>8}$-C_{10}	2.12E+01	4.24E+01	1.93E+02	5.27E+01
$C_{>10}$-C_{12}	1.08E+02	2.54E+01	1.00E+03	3.10E+01
$C_{>12}$-C_{16}	4.92E+02	1.12E+03	4.62E+03	1.37E+01
Aromatics				
$C_{>8}$-C_{10}	2.67E+01	3.16E+02	2.51E+02	1.33E+02
$C_{>10}$-C_{12}	1.82E+02	1.89E+02	1.72E+03	3.78E+02
$C_{>12}$-C_{16}	9.46E+03	8.65E+01	8.91E+04	2.27E+02

Table 5.14: SQG for a Commercial Slab on Grade Building, Commercial/Industrial Scenario (mg/kg)

	coarse-grained soils[a]	fine-grained soils[a]	coarse-grained soils[b]	fine-grained soils[b]
BTEX				
Benzene	4.84E-01	4.35E+00	2.22E-02	2.05E-01
Toluene	9.49E-01	8.46E+00	5.99E+00	5.52E+01
Ethylbenzene	1.97E+0	1.72E+01	1.21E+01	1.10E+02
o-Xylene	1.92E+00	1.69E+01	119E+01	1.08E+02
m-Xylene	1.55E+00	1.33E+01	9.31E+00	8.30E+01
p-Xylene	1.81E+00	1.56E+01	1.09E+01	9.76E+01
Aliphatics				
C_6-C_8	5.73E-01	5.24E+00	3.73E+00	3.51E+01
$C_{>8}$-C_{10}	1.90E-02	1.74E-01	1.24E-01	1.16E+00
$C_{>10}$-C_{12}	1.30E-02	1.19E-11	8.46E-02	7.95E-01
$C_{>12}$-C_{16}	3.01E-03	2.75E-02	1.96E-02	1.84E-01
Aromatics				
$C_{>8}$-C_{10}	5.09E-01	4.65E+00	3.31E+00	3.11E+01
$C_{>10}$-C_{12}	2.21E+00	2.02E+01	1.44E+01	1.35E+02
$C_{>12}$-C_{16}	5.79E+01	5.29E+02	3.77E+02	3.54E+03

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[a] Building with Basement, Agricultural/Residential Scenario

[b] Slab on Grade Structure, Commercial/Industrial Scenario

Table 5.15: GWQG for Various Exposure Scenarios (mg/L)

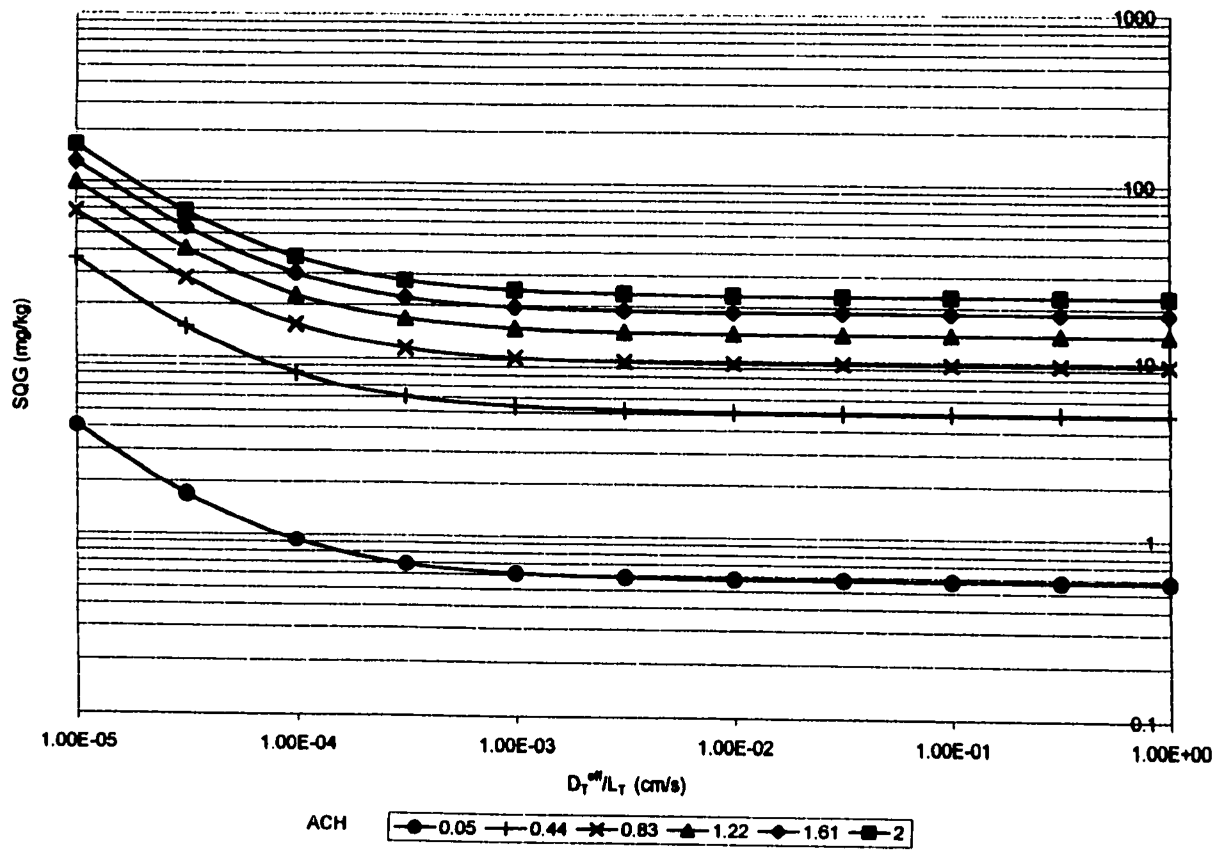

Figure 5.1: SQG for C_6–C_8 aliphatic hydrocarbons in Coarse-Grained Soils for a Residential Building with a Basement and an Agricultural/Residential Exposure Scenario vs. $\frac{D_t^{eff}}{L_T}$ for various air exchange rates

As can be seen in Figure 5.1, both the air exchange rate and $\frac{D_T^{eff}}{L_T}$ the influence soil quality guideline. This phenomenon was discussed by Johnson et al. (1999), where it was explained that an increase in the attenuation coefficient, which corresponds to a decrease in guideline values, is due to increasing diffusional resistance through soil as the contaminant source moves further away from the building structure. The lower asymptote (i.e. right hand side of Figure 5.1) in the guideline values represent the case where the source is close to the building and soil gas transport is dominated by advection through the foundation and any attenuation is due to dilution through mixing within the building structure (Johnson et al., 1999). These look-up figures should only be used for screening level purposes and are recommended for use as an aid in decision-making for petroleum hydrocarbon contamination management.

Chapter 6

Discussion and Recommendations

Indoor air quality may be impacted if a contaminant source is located in the subsurface within the vicinity of a building structure. Soil gas can migrate from a contaminant source located a distance below the surface to the building zone of influence by the processes of advection and diffusion. This phenomenon has been described in the studies of radon gas by Nazaroff et al. (1985). Hydrocarbon contamination of soils may result from leakage of underground storage tanks and contaminant spills and is a human health concern due to the risks associated with VOC exposure. Furthermore, the indoor inhalation pathway is of concern since almost 90% of the time spent by Canadians is spent in the indoor environment (Leech et al., 1996).

Johnson and Ettinger (1991) developed a heuristic model to predict indoor air concentrations based on soil gas concentrations for screening level calculations. The Canadian Council Ministers of the Environment chose this model to predict Tier I standards based on the soil vapour to indoor air pathway (CCME, 2000). Alberta Environment also plans to use this model in updating its Petroleum Storage Tank Guidelines. This model has also been used by the United States Environmental Protection Agency to develop soil and groundwater screening levels (USEPA, 1997).

The Johnson and Ettinger (1991) model integrates both advective and diffusive transport as well as contaminant, soil, and building properties to predict indoor air concentrations. The key algorithms of the model are

$$\alpha = \frac{\frac{D_t^{eff} A_b}{Q_b L_T} \times \exp \frac{Q_s L_c}{D_c A_c}}{\exp \frac{Q_s L_c}{D_c A_c} + \frac{D_T^{eff} A_b}{Q_b L_T} + \frac{D_T^{eff} A_b}{Q_s L_T}(\exp \frac{Q_s L_c}{D_c A_c} - 1)} \tag{2.38}$$

$$\frac{Q_s L_c}{D_c A_c} \tag{2.39}$$

$$\alpha = \frac{\frac{D_T^{eff} A_b}{Q_b L_t}}{\frac{D_T^{eff} A_b}{Q_s L_T} + 1} \tag{2.42}$$

$$\alpha = \frac{\frac{D_T^{eff} A_b}{Q_b L_T}}{1 + \frac{D_T^{eff} A_b}{Q_b L_T} + \frac{D_T^{eff} A_b L_c}{D_C A_c L_T}} \quad . \tag{2.43}$$

Equation 2.38 is the attenuation coefficient. When Eq 2.39, the equivalent Peclet number, $(\frac{Q_s L_c}{D_c A_c}) \rightarrow \infty$, Eq 2.38 becomes Eq 2.42. When $(\frac{Q_s L_c}{D_c A_c}) \rightarrow 0$, diffusion is the dominant mode of soil gas transport and Eq 2.43 can be used.

A sensitivity analysis was performed on Eq 2.38 to identify those parameters having the greatest influence on model output. Eq 2.39 was also subjected to a sensitivity analysis since the value generated here determines if advective transport dominates. The results of the sensitivity analysis on Eq 2.38 indicated that the distance from the source to the foundation (L_T)

had the highest contribution to variance, followed by the air exchange rate (ACH), via the building ventilation rate (Q_b), and the soil vapour filled porosity (ϵ_v), via the effective diffusion coefficient (D_T^{eff}). The results of the sensitivity analysis on Eq 2.39 indicated that soil vapour permeability (k_v) had the highest contribution to variance, followed by soil vapour filled porosity (ϵ_v) and the indoor-outdoor pressure differential (ΔP).

To reduce the uncertainty in model predictions, L_T, ACH, D_T^{eff}, k_v, and ΔP (identified by the sensitivity analysis) are recommended for field measurement. In addition to the sensitive parameters, other site specific factors must be considered when undertaking a Phase II Environmental Site Assessment. Such factors include the time to steady state concentrations, seasonal variations, subsurface variations and biological and chemical processes. The time required to reach near steady state concentrations is of importance because this will predict when the maximum contaminant concentration will be reached; this value increases with the square of the distance from the source. For long term predictions of indoor air quality, soil gas concentrations measured near the source will provide the best estimate of near steady state concentrations (Johnson et al., 1999). Seasonal variations, subsurface variations and biological and chemical processes must also be considered because they influence soil gas measurements. To increase conservatism in model predictions, soil gas measurements should be taken in the winter time, which represents a worst case scenario due to increased indoor-outdoor pressure differentials, lower air exchange rates, as well as possible accumulation of VOCs in the subsurface.

Risk based screening levels and remediation objectives for volatile petroleum hydrocarbons were developed using input parameters defined by CCME. The risk based screening levels that were developed follow a multi-level assessment framework where conservative field based screening level measurements are first implemented and then progresses to soil gas lab analysis and indoor air quality measurements. As comparative values for these measurements were developed using CCME parameters for building, soil, site, and contaminant characteristics it is important to note that the screening levels may not represent all situations. If the most sensitive parameters (i.e. L_T, ACH, D_T^{eff}, k_v, and ΔP) for an actual site differ greatly from default values used by CCME, the screening levels may be more or less conservative depending on the site specific characteristics. The screening levels, as their name suggests, should only be used as a preliminary assessment of a contaminated site. Furthermore, since VOCs are prevalent in the indoor air environment (e.g. consumer and commercial products, paints, adhesives, furnishings and clothing, building materials, ventilating and air-conditioning systems, combustion appliances, tobacco smoke) VOCs detected by indoor air quality measurements may not represent VOC concentrations as a result of contaminated soil gas transport.

Remediation objectives for soil were estimated by the soil quality guideline algorithm used by CCME (2000). The soil saturation concentration (C_{sat}) was also derived. C_{sat} is the chemical concentration in soil corresponding to a point where the sorption limit of soil particles, solubility limits of soil pore water, and saturation of soil pore air have been reached. Therefore, if health risks are acceptable at C_{sat}, then they will not increase above C_{sat}. Groundwater quality guidelines were also derived using CCME parameters.

Look up figures were developed for use under Phase II environmental assessments. Guideline values were varied by $\frac{D_T^{eff}}{L_T}$ and ACH for various scenarios. Because CCME fine-grained soil parameters are not representative of conditions where diffusive soil gas transport dominates, Eq 2.38 was used to develop the figures for both coarse- and fine-grained soils. The look up figures can be used as a preliminary estimate of soil quality guidelines for specific sites. A government distributed computer spreadsheet program similar to the USEPA (1997) could be implemented to assist users in determining remediation guidelines for site specific cases.

Limitations of the CCME approach include the arbitrary assumption that if L_T is greater than or equal to 30 cm to fulfill the condition that $\frac{r_c}{Z_c} \ll 0$ in the calculation of Eq 2.22. This is not an issue for structures with basements, but may affect slab on grade structures. Therefore, the model should not be applied for L_T <30 cm because it is uncertain if boundary conditions for the model are still applicable.

The soil vapour permeabilities assigned to coarse-grained soils by CCME should be reexamined. Soil vapour permeabilities for CCME defined coarse-grained soils are representative of well-graded, silty sands and gravels and therefore may not be representative of sites where the predominant material is composed of clean gravels. The fine-grained soil vapour permeability assumed by CCME is representative of silty sands and is of a coarse texture, which does not warrant the use of Eq 2.43 - the diffusion limited equation. Soil vapour permeabilities for fine-grained soils assigned by CCME result in an equivalent Peclet number greater than one indicating that advective transport is still very much the dominant transport mechanism. Since diffusive soil gas transport dominates when $\frac{Q_s L_c}{D_c A_c} \rightarrow 0$, Eq 2.43 should not be used for soil vapour permeabilities for fine-grained soils defined by CCME and should only be used if the equivalent Peclet number is found to be much less than one. Given that the equivalent Peclet numbers generated for the CCME fine-grained soils case indicate that advection is a dominant transport mechanism, the attenuation coefficient determined based on Eq 2.38 is a better representation of CCME input parameter conditions. Therefore, Eq 2.38 should be used for CCME defined fine-grained soils rather than Eq 2.43. However, if it can be shown that the equivalent Pe number is much less than one for a fine-grained soil case by assigning more realistic soil vapour permeabilities, Eq 2.43 may be used.

The Johnson and Ettinger (1991) model is thought to be a conservative predictor of indoor air concentrations based on subsurface contamination since biodegradation and other losses are not taken into consideration. However, a study undertaken by the Massachusetts Department of Environmental Protection found that groundwater quality guidelines based on the Johnson and Ettinger (1991) model underpredicted indoor air concentrations of chlorinated VOCs. Indoor air concentrations of non-chlorinated VOCs were reported to be near the predicted value (Fitzpatrick & Fitzgerald, 1996). The fact that indoor air concentrations of chlorinated VOCs were underestimated can be explained since such compounds are not readily amenable to biodegradation. However, because indoor air concentrations of non-chlorinated VOCs fell close to the predicted value does not indicate model conservatism. Further limitations of the Johnson and Ettinger (1991) model are that underground pathways for soil gas migration are not accounted for. Such pathways include utility lines, karst formations, and other preferential soil gas migration pathways, which may account for indoor air contamination from nearby contaminated soils (Altshuler & Burmaster, 1997).

The Johnson and Ettinger (1991) model has been gaining widespread use as a regulatory tool (USEPA, 1997; CCME, 2000; MADEP, 1997b). However, field validation of this model is necessary to verify that the model adequately describes the soil gas to indoor air transport pathway. Difficulties with field validation include influences on soil gas transport that cannot be controlled (such as seasonal influences, biological and chemical processes, building construction and construction materials, as well as variability in soil properties). Unfortunately, field validation studies are expensive and therefore few field validation studies have been undertaken. Regardless, further study of this model is necessary to ascertain if this model adequately predicts the attenuation coefficient. Comparison of field validation studies and modeling results will also help identify those parameters which may be influencing soil gas transport that may have been ignored (Altshuler & Burmaster, 1997).

A further analysis on model sensitivity using mathematical techniques would be useful in validating the Crystal Ball™ sensitivity analysis of the Johnson and Ettinger model. It would be desirable to analytically establish bounds on the growth of the variables so that the sensitivity of the Johnson and Ettinger (1991) model could be determined once and for all.

The Johnson and Ettinger (1991) model is only a first step in modeling the soil vapour to indoor air pathway. More realistic models should be investigated through collaboration with experts in the field of fluid dynamics. Through field validation and testing of such models, a refined model of the soil vapour to indoor air pathway may evolve and a better understanding of vapour transport through porous media established.

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Appendix A

Results of Sensitivity Analyses

A.1 Sensitivity Analysis of Eq 2.38 - Measured by Contribution to Variance

Crystal Ball Report
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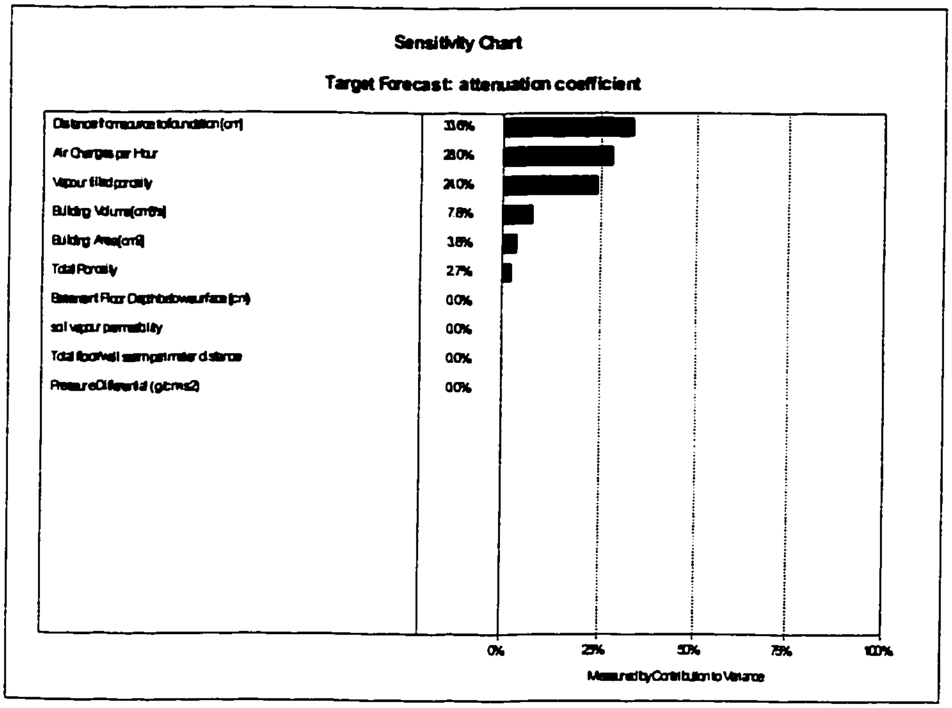
Simulation stopped on 1/25/01 at 11:33:54



A.2 Sensitivity Analysis of Eq 2.38 - Measured by Rank Correlation

Crystal Ball Report
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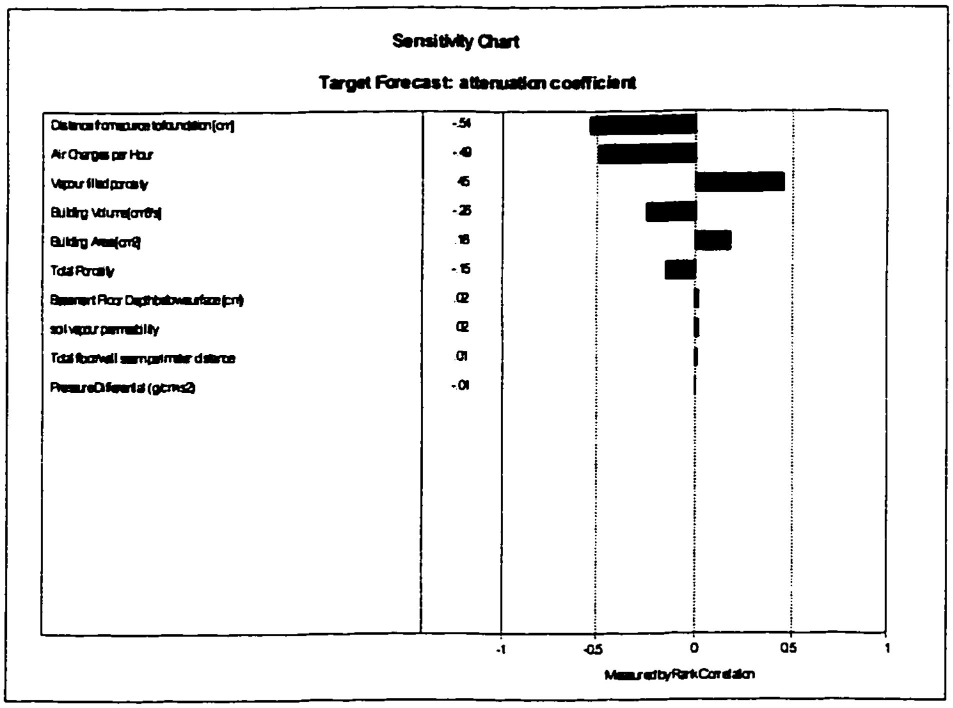

A.3 Sensitivity Analysis of Eq 2.39 - Measured by Contribution to Variance

Crystal Ball Report
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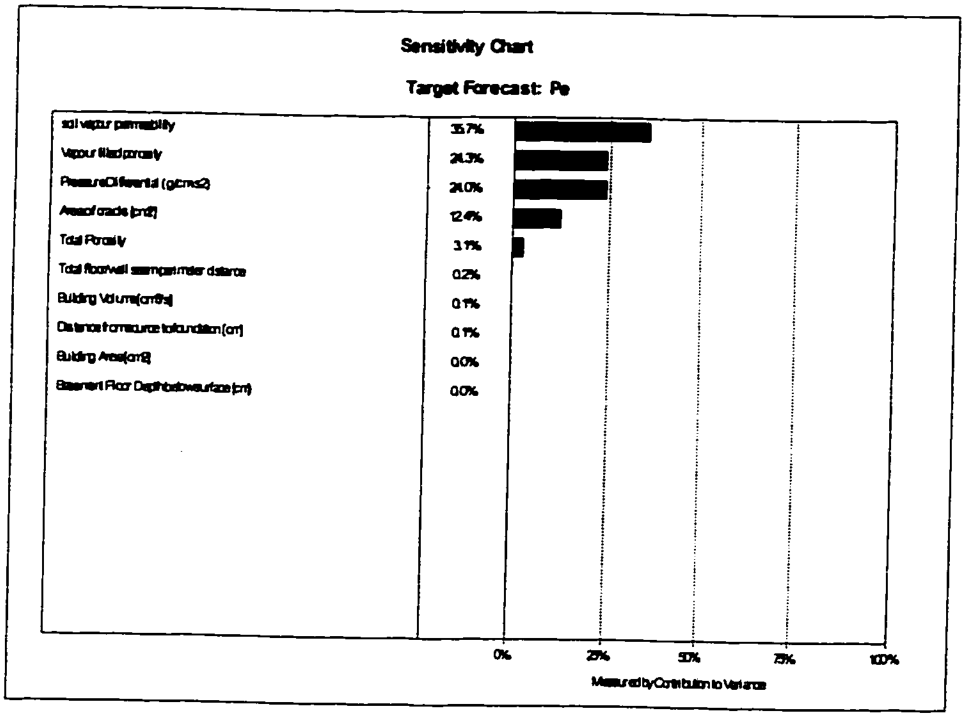
Simulation stopped on 1/25/01 at 11:33:54



A.4 Sensitivity Analysis of Eq 2.39 - Measured by Rank Correlation

Crystal Ball Report
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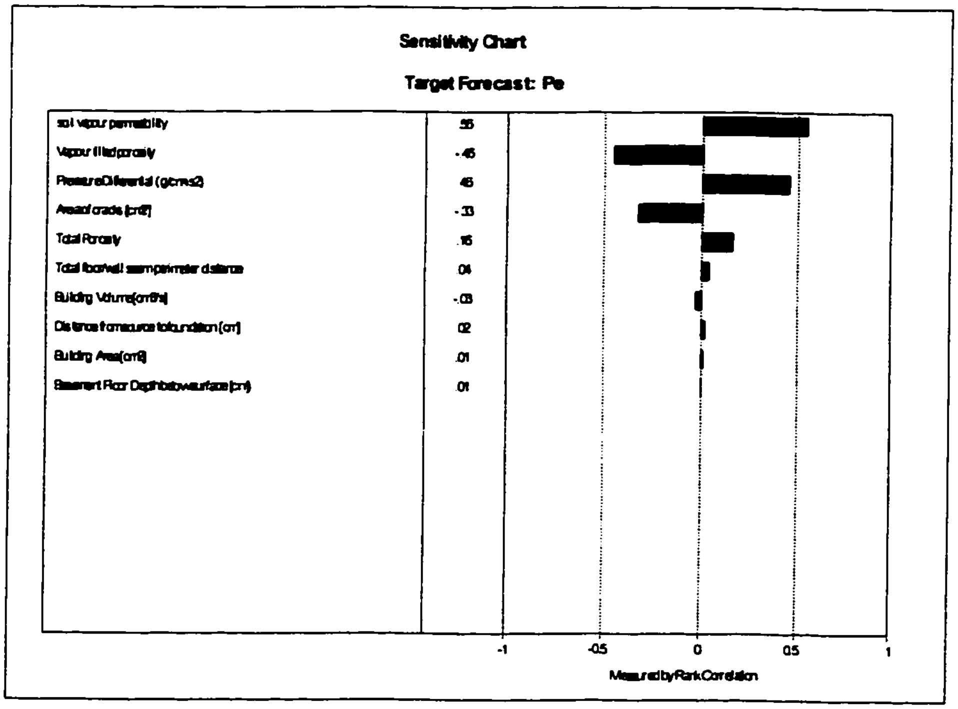
Simulation stopped on 1/25/01 at 11:33:54



Appendix B

Soil Gas Sampling Methods

Soil gas sampling can be used to assess upward migration of contaminant vapours towards buildings for risk assessments. In the tiered framework for volatile hydrocarbon contaminated site assessment outlined in Section 5.2 (Table 5.1) field screening of soil gas is the first step in the process. If potential indoor air impacts cannot be ruled out by initial field soil gas screening, laboratory analytical soil gas sampling methods can be undertaken. Soil gas can be sampled for VOCs by a number of methods including dynamic sampling, diffusion wells, headspace extraction, and purge and trap techniques.

B.1 Dynamic Soil Gas Sampling

Dynamic soil gas samples are collected from a stream of soil gas, which is pumped through a probe in the soil, or from permanently installed tubes. A hollow probe may be used with this method, which is manually or pneumatically driven into the soil or inserted into boreholes, permanently installed tubes, or ground water monitoring wells (USEPA, 1993).

A vacuum is then applied to the rods or to the tubing inside the rods for sample collection on solid sorbent tubes or Tedlar bags and stainless steel canisters to allow sample preconcentration when VOC levels are below analytical detection limits. Section C.1 describes these gas sample collection devices in more detail. Soil gas concentrations can be analyzed in the field using portable analytical instruments or at a remote laboratory using gas chromatography and an appropriate detector.

Dynamic soil gas sampling is best suited to conditions where contaminant concentrations are expected to be high and the vadose zone has a high vapour permeability (Ullom, 1995). High levels of soil moisture can render dynamic soil gas sampling efforts ineffective. However, such conditions can be overcome by minimizing sample volume, increasing air volume around the tip, and allowing time for equilibrium (Hitzig et al., 1997). The best results are achieved when soil gas samples are taken near a contaminant source and when soils are porous enough to result in good gas transport (Selby, 1991).

Probes used in dynamic sampling may destroy natural soil permeability near the probe due to soil compaction and can be a severe limitation in moist, heavy clay soils (Ullom, 1995). In very dry soils, radial fractures can result from probe insertion resulting in dilution of the soil atmosphere leading to lower contaminant concentrations, which may be below analytical detection limits. Subsurface barriers to vapour migration can result in contaminant vapour accumulation resulting in misleading results. Likewise, sampling on steep slopes, near animal burrows, or over utility corridors may lead to false negative results (Ullom, 1995).

Different probe tip sizes are available for soil gas sampling. Although the use of probe tips with diameters greater than the probe rods result in reduced friction during probe advancement, this practice is not advisable due to an increased likelihood of sample contamination with atmospheric gases and dilution of soil gas concentrations (Hitzig et al., 1997). Small diameter probes are least likely to disturb the soil and can provide a representative soil gas sample. Once a probe has been placed, continuous pumping of the system is not advised since sample dilution may occur (Tillman et al., 1989a).

Dynamic sampling methods are highly depth dependent. Sampling depths should be based on site-specific factors (i.e. moisture conditions, air filled porosity, VOC concentrations, and depth to groundwater) and compound-specific factors (i.e. solubility, volatility, and degradability) (Marrin & Kerfoot, 1988). Sampling at depths between one to two metres minimizes impacts from minor surface contamination as well as the effects of barometric pressure, temperature, rainfall, and air pollution (Tillman et al., 1989a). Soil gas samples taken less than 60 cm from the surface may be subject to atmospheric dilution (Tillman et al., 1989b).

Advantages of dynamic soil gas sampling are that samples can be taken in a short time, and results are available immediately when combined with field analytical tools. Limitations include depth dependent concentrations, and local VOC concentrations may be disturbed as a result of pumping to retrieve the sample (USEPA, 1993; Marrin & Kerfoot, 1988). Near-surface soil gas measurements may also increase susceptibility to sampling errors due to short-circuiting along sampling probes and the presence or absence of surface barriers, which can affect near-surface vapour concentrations. Furthermore, the time of sampling may not coincide with near-steady state soil gas concentrations at the surface resulting in lower soil gas concentrations than what could be expected in the future. Consideration of near-steady state soil gas concentrations may addressed by sampling at the depth near the source zone (Johnson et al., 1999).

Table B.1 lists some soil gas sampling systems and their manufacturers. Standard methods and guidelines for dynamic sampling can be found in the ASTM Standard Guide for Soil Gas Monitoring in the Vadose Zone (D5314-91 Vol. 4.08) and in Characterization of Hazardous Waste Sites - A Methods Manual, Vol. II: Available Sampling Methods (EPA/600/4-840076) (USEPA, 1993)

Instrument	Features	Manufacturer
AMS Gas Vapor Probe System	Asphalt and concrete drill for pavement penetration Electric rotary hammer drill and/or an AMS Slide Hammer Hardened steel tip drive end Hand-operated retrieval jack Hand-operated vacuum pump 10 AMS Gas Vapour Points 76 m Teflon Tubing	Art's Manufacturing & Supply, Inc 105 Harrison American Falls, ID 83211-1230 United States Tel: (800) 635-7330 or (208) 226-2017
AMS Heavy Duty Gas Vapor Probe Kit	Includes everything in Gas Vapor Probe System plus 5 cm AMS Flighted Augers with two 91 cm extensions SDS Max Auger Adapter Carbide Insert Tip 1.5" Concrete Bit	Art's Manufacturing & Supply, Inc
AMS Hollow Stem Auger	Portable hollow stem augers designed for sampling soils, soil gas and shallow groundwater	Art's Manufacturing &Supply, Inc
AMS Flighted Auger	Continuous flighted augers enable drilling of a small diameter hole at virtually any angle in suitable soils	Art's Manufacturing & Supply, Inc
Membrane Interface Probe	Detects VOCs Consists of a thin film membrane impregnated into a stainless steel screen on the probe face Carrier gas transfers sample to PID or FID at the surface Operates in both vadose and saturated zones Can detect dissolved hydrocarbons down to 1 ppm Usable to a depth of 60 feet	Geoprobe Systems Salina, Kansas 67401 Tel: (800) 436-7762 or (785) 825-1842

Table B.1: Soil Gas Sampling Instrumentation and Manufacturers

B.2 Diffusion Wells

Diffusion wells may be used to obtain soil gas samples as an alternative to dynamic soil gas sampling. Figure B.1 illustrates the design of some diffusion wells. With this sampling strategy, air inside the well is allowed to equilibrate with soil gas at the inlet through diffusion. The time to reach equilibrium depends on the diffusion constant of the soil, the cross section of the inlets and the length of the well. Samples can be withdrawn from gas wells using evacuated test tubes or plastic syringes (Farrell et al., 1993).

B.3 Gas Headspace Extraction

The gas headspace extraction method is used at contaminated sites to collect VOCs in soils and ground water for identification and quantification. Gas headspace extraction uses dead space to collect gases moving through water or soil. The gas headspace method for VOC analysis, involves the transfer of soil core samples to 500 ml glass containers fitted with modified lids including a tape sealed probe-hole on top of the lid. Samples can be broken by hand to increase sample surface area and to encourage vapour release. Samples should be quickly transferred to the container to minimize vapour losses. Furthermore, the sample container should only be half-filled to allow for adequate headspace for vapour accumulation. Vapour concentrations in the headspace can be analyzed by inserting a hydrocarbon vapour analyzer probe such as a FID or PID calibrated to known hydrocarbon standards into the probe-hole tape seal (AENV, 1994). Multiple headspace extraction is sometimes used because not all vapours will degas the first time. However,

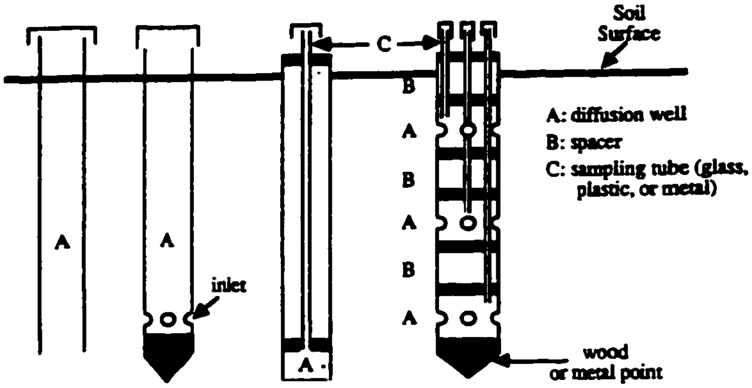


Figure B.1: Diffusion Well Designs (Farrel et al. 1993) full extraction of volatiles from soil or ground water is not always possible (USEPA, 1993).

The advantages of this method are reduced biodegradation of samples, rapid analysis and turnaround time, and opportunity for field analysis (Roe et al., 1989). Other advantages of the manual headspace method include ease of sample collection and low operational costs. However, matrix effects have been observed using this method for some soil types and sensitivity and dynamic range using this technique are limited (Parr et al., 1991). Gas headspace extraction may also result in measurement errors through vapour dilution by drawing air into the headspace while sampling and conducting headspace screening in a manner such that concentration-dependent factors are not well controlled (e.g., temperature, volume of headspace to sample, encapsulation time) (Robbins et al., 1989).

The headspace method can be used for preparation of VOCs in soil for screening with total vapour analyzers and analysis by gas chromatography (GC) or by gas chromatography/mass spectrometry (GC/MS). This method is applicable for a variety of organic compounds with high enough volatility to be adequately removed from soil samples using an equilibrium headspace procedure (USEPA, 2000b). The method detection limits using GC/MS are compound, matrix, and instrument dependent and vary from about 0.1 to 3.1 μg/kg. The concentration range of analyte that can be discerned from this method is about 10 to 200 μg/kg (USEPA, 2000b). Detection limits are generally higher for headspace analysis than for purge and trap analysis (Weisman, 1998). However, instrument flow rate and response characteristics vary, especially in headspace analysis using small sampling containers

(Fitzgerald, 1989).

B.4 Purge and Trap Methods

Purge and trap is used for the analysis of VOCs in solid materials such as soil and sediments. This method involves the use of a sample vial with an air tight seal to minimize VOC losses during sample transport, handling, and analysis (USEPA, 2000b). Purge and trap techniques involve pushing a gas such as helium through a soil sample to entrain VOCs. Subsequently, entrained volatiles are injected into the analytical instrument or can be used in combination with a sorbent trap to concentrate samples for later analysis (USEPA, 1993). Figure B.2 illustrates a schematic of a purge and trap device.

This method is advantageous because it provides better recovery compared to the headspace extraction and dynamic sampling methods. Limitations of this method include the use of more complex equipment than headspace extraction, the use of a purge gas, and more specialized training is required compared to gas headspace extraction and dynamic sampling. This method is commonly used for mobile laboratory analysis of volatiles in soil and ground water (USEPA, 1993).

The applicable concentration range for this method falls between 0.5 to 200 μg/kg. However, high concentration samples (i.e. greater than 200 μg/kg) can be analyzed using USEPA SW-846 Method 5030 (Purge and Trap for Aqueous Samples). Purge and trap is suitable for most VOCs with boiling points below 200 °C that are insoluble or slightly soluble in water. Volatile, water soluble compounds can be extracted using this technique but

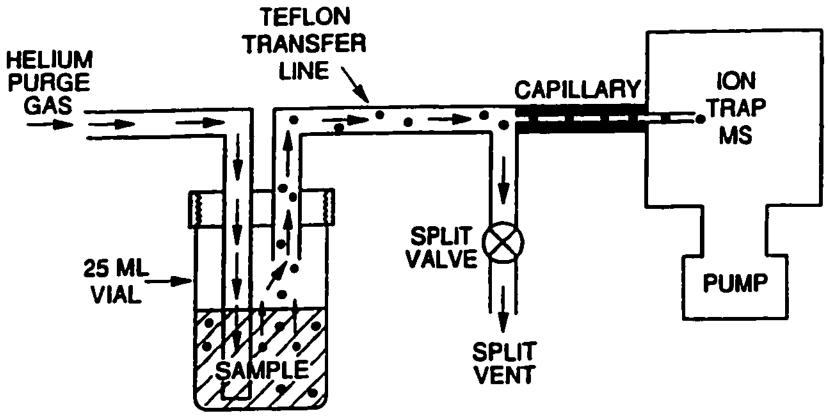


Figure B.2: Schematic of a Purge and Trap Device (USEPA, 1993) quantitation limits can be up to ten times greater due to poor purging efficiency (USEPA, 2000b).

Analysis of samples extracted by purge and trap can be accomplished by Gas Chromatography/Flame Ionization Detection (GC/FID) for light aliphatic hydrocarbon fractions and by Gas Chromatography/Photoionization Detection (GC/PID) for aromatic fractions. A complete analysis of gasoline fractions can be determined using GC/PID/ELCD (Electron Capture Detector) in series with GC/FID (USEPA, 2000b).

Appendix C

Indoor Air Sampling Methods

Indoor air sampling can be used to assess indoor air quality in the tiered framework for volatile hydrocarbon contaminated site assessments if soil gas analysis is unable to rule out potential indoor air quality impacts (Table 5.1). Indoor air can be sampled for VOCs by a variety of methods including dynamic and passive sampling. In dynamic sampling, air is actively pumped into a collection device. Passive sampling involves diffusion of air into a sampling cylinder and adsorption onto a sorbent (typically activated carbon).

The transfer of pollutants from the sampling equipment to separation and identification instruments is an important part of ambient air sampling and analysis because of its influence on the volume of air sampled and the overall analytical method. For indoor air sampling, influences from other indoor VOC sources as well as seasonal changes may affect analytical results. Direct measurement of indoor air concentrations may be of limited application due to a number of factors, which include confounding by indoor VOCs emanating from furnishings, carpets, and household cleaning products.

Indoor air sampling may cause undue emotional stress to occupants of buildings and residences. Therefore, unless flammable subsurface concentrations have been identified or there are odour complaints which may indicate a short-term threat, direct indoor air sampling is not recommended as a primary mode of investigation. Indoor air sampling is not always suitable for a site especially when indoor vapour concentrations are not representative of long-term steady state conditions. Furthermore, some assessments may involve sites where a building or enclosed space does not exist but predictions of potential future impacts are of interest (Johnson et al., 1999).

Portable detectors are useful in determining presence of VOCs in indoor air and soil gas. Guidelines for collection of whole air samples for VOC detection and measurement in SUMMA passivated stainless steel canisters and collection on solid adsorbents and cryogenic preconcentration techniques have been developed by the USEPA (USEPA, 1999b; Woolfendent & McClenny, 1999; Riggin, 1984a; Riggin, 1984b).

C.1 Integrated Dynamic Grab Samplers

Dynamic samplers mechanically draw air through a sampling device. This method involves the intake of a known volume of sample air into a collection device over a specified time period. Volatile and semi-volatile organic compounds can be collected on solid sorbents or stainless steel canisters and analyzed using GC/FID, /ECD, or /MS. Use of dynamic grab sampling is advantageous due to the large number of configurations and methods of pollutant collection available, adjustments of sample flow and sampling time can increase sensitivity, and multiple units can be combined into a sequential sampler. Some limitations of dynamic sampling include long set up procedures, use of corrosive or toxic solutions, and risk of sample contamination during handling and transport (Yocom & McCarthy, 1991).

C.1.1 Active Sampling onto Sorbent Tubes

Both indoor air and soil gas can be sampled for VOCs at 0.5 to 25 ppb(v) levels using sorbent tube/thermal desorption/gas chromatograph-based methods. GC/MS is the preferred method of analysis since it provides a broad spectrum of analysis in addition to identification of target compounds (USEPA, 1991). A number of solid adsorbents are available commercially and may be used independently or in multisorbent packings. In the former case, sorbents are packed in order of increasing sorbent strength to improve quantitative retention and desorption of VOCs over a large volatility range (Woolfendent & McClenny, 1999). A combination of sorbents allows for efficient sampling and thermal desorption of a wide range of VOCs (Maroni et al., 1995).

Active sampling onto sorbent tubes involves the intake of sample air into a glass or stainless steel tube packed with a selected sorbent. Flow rates and sample volumes used for sampling vary and are determined by the analyte and sampling time. After adsorption, the sample is thermally desorbed with an inert gas such as helium. The inert gas transfers VOCs from the cartridge onto a cold trap. The desorbed sample is then introduced into a high resolution (e.g. capillary column) GC with an accompanying detector for identification and quantification (Riggin, 1984b).

Limitations of solid sorbents include the thorough cleanup and thermal conditioning required to ensure contaminant-free cartridges, careful handling and storage is necessary to prevent contamination, small breakthrough volumes of some compounds prevent quantitative analysis, and concentrations may be underestimated if sorption is not complete or if the sorbent becomes saturated (USEPA, 1993; USEPA, 1991). Interference occurs due to the formation of artifacts in the presence of NO_x, from long term storage of blank tubes, and artifacts generated from sampling and sample storage. Water collection on sorbent tubes can also cause interference; therefore, a hydrophobic sorbent for the sample tube should be selected to minimize water interference. Sample splitting and dry purging are other methods available to reduce water interference. High concentrations of particulates can also cause interference, which may be remedied by a particulate filter connected to the sampling end of the tube (Woolfendent & McClenny, 1999). Despite its limitations, adsorption onto solid sorbents has several advantages including the small size of the samplers, portability of the sampling devices, placement of sorbent packing as the first part of the sampling train to reduce contamination from upstream sources, wide selection and availability of sorbents and thermal desorption systems, ease of use, and the ability to collect integrated samples over a period of 8 to 12 hours (USEPA, 1991; Woolfendent & McClenny, 1999).

Two critical parameters for active sampling include breakthrough volume and sample flow rate. Breakthrough volume is the volume of air that can be passed through a trapping medium before the analyte is detected in a back-up sampler. For solid sorbents, breakthrough volume depends on amount of sorbent, air pollutant concentration, and the physico-chemical characteristics of both the sorbent and pollutants. High sample flow rates may reduce breakthrough volume; therefore, sample flow rates should be low enough to allow adequate contact time between the air and the sorbent for pollutant diffusion to the sorbent surface (Maroni et al., 1995). Flow rates above 10 mL/min are usually used to minimize errors due to penetration of VOCs by diffusion (Woolfendent & McClenny, 1999). The typical working range for sorbent packing is between 0 and 40 °C. Generally, a temperature increase of 10 °C will reduce breakthrough volume of sorbent packings by a factor of two (Woolfendent & McClenny, 1999).

Detection limits depend on minimum artifact levels, GC detector selection, and volume of air sampled. The volume of air sampled also depends on safe sampling volumes, pump flow rate limitations and time weighted-average monitoring time constraints. Detection limits can range from sub-part-per-trillion for halogenated species such as CCl_4 and the freons using an electron capture detector to sub-ppb for volatile hydrocarbons in 1 L air samples using the GC/MS operated in the full scan mode. Detection limits depend on proper management of water for GC capillary analysis of VOCs in air using sorbent technology (Woolfendent & McClenny, 1999). Precision and accuracy varies depending on the contaminant, and other factors such as breakthrough volumes, structural isomers, cartridge contamination, and factors related to the analytical instrument and skill of the analyst. Therefore, it is important that quality assurance/quality control procedures be adhered to in order to minimize such problems (Riggin, 1984b).

For sorbent sampling, land 4 L air sample volumes are recommended if consistent with safe sampling volumes. Flow rates can be adjusted to accommodate low safe sampling volumes. Detection limits of (< 0.5 ppb per analyte) can be accomplished using 300 mL samples with full scan mass spectrometry detection. Sensitivity can be magnified by at least ten-fold if conventional GC detectors or selected ion monitoring are applied (USEPA, 1993).

Table C.1 lists sorbents and, where available, safe sampling volumes suitable for aliphatic, aromatic and cyclic hydrocarbons less volatile than ethane and more volatile than n-C_{20} using a single sorbent bed $\frac{1}{4}$ inch outside diameter stainless steel tubes and compatible with thermal desorption-capillary GC analytical procedures (Woolfendent & McClenny, 1999). Table C.2 lists some guidelines for sorbent selection.

Table C.1: Suitable Sorbents for Hydrocarbon Detection (USEPA, 1999)

Compound	Sorbents and Safe Sampling Volumes
n-Butane	CS III, C 1000, Combination Tubes 1,2, or 3 or Spherocarbon (SSV 820 L)
n-Pentane	CSIII, C1000, Spherocarb (SSV 30 000L), Combination Tubes 1,2, or 3 or Chromosorb 106 (SSV 5.5L)
n-Hexane	Carbopack^{TM} B, Combination Tubes, 1, 2 or 3 or Chromosorb 106 (SSV 30 L)
Benzene	Carbopack^{TM} B, Combination Tubes, 1, 2, 3 or Chromosorb 106 (SSV 30 L) or Tenax (SSV 6 L)
n-Heptane	Carbopack^{TM} B, Tenax (SSV 17L), Combination Tubes 1,2 or 3 or Chromosorb 106 (SSV 160L)

continued on next page

Table 28: Suitable Sorbents for Hydrocarbon Detection

Compound	Sorbents and Safe Sampling Volumes
Toluene	Carbopack™ B, Tenax (SSV 38L), Combination Tubes 1, 2, 3 or Chromosorb 106 (SSV 80L)
n-Octane	Carbopack™ B, Tenax (SSV 700 L) Combination Tubes 1, 2, 3 or Chromosorb 106 (SSV 360 L)
Ethylbenzene	Carbopack™ B, Tenax (SSV 180 L), Combination Tubes 1, 2, 3 or Chromosorb 106 (SSV 360 L)
all Xylenes	Carbopack ™ B, Tenax (SSV 300 L), Combination Tubes 1, 2 or 3 or Chromosorb 106 (SSV 7000 L)
n-Nonane	Carbopack™ C/B, Tenax (SSV 700L), Combination Tubes 1, 2, or 3 or Chromosorb 106 (SSV 7000 L)
Styrene	Carbopack™ C/B, Tenax (SSV 300L) or Combinaton Tubes 1, 2, or 3
Isopropylbenzene	Carbopack™ C/B, Tenax (SSV 480 L) or Combination Tubes 1, 2, or 3
n-Propylbenzene	Carbopack™ C/B, Tenax (SSV 850 L) or Combination Tubes 1, 2, or 3
1-Methyl-3-ethylbenzene	Carbopack™C/ B, Tenax (SSV 1000L) or Combination Tubes 1, 2, or 3
1-Methyl-4-ethylbenzene	Carbopack™ C/B, Tenax (SSV 1000L) or Combination Tubes 1, 2, or 3
1,3,5-Trimethylbenzene	Carbopack™ C/B, Tenax (SSV 1800 L), Combination Tubes 1, 2, or 3 or Chromosorb 106 (SSV 2800)
Methylstyrene	Carbopack™ C/B, Tenax (SSV 1200 L), Combination Tubes 1, 2 or 3 or Chromosorb 106 (SSV 2800 L)
Methyl-2-ethylbenzene	Carbopack™ C/B, Tenax (SSV 1000L) or Combination Tubes 1, 2, or 3
1,2,4-Trimethylbenzene	Carbopack™ C/B, Tenax (SSV 1800 L) or Combination Tubes 1, 2, or 3
n-Decane	Carbopack™ C/B, Tenax (SSV 2100 L) or Combination Tubes 1, 2 or 3 or Chromosorb 106 (SSV 37 000L)
1,2,3-Trimethylbenzene	Carbopack™ C/B, Tenax (SSV 1800 L) or Combination Tubes 1, 2, or 2
n-Undecane	Carbopack™ C/B, Tenax (SSV 12 000L) or

continued on next page

Table 28: Suitable Sorbents for Hydrocarbon Detection	
Compound	Sorbents and Safe Sampling Volumes
	Combination Tubes 1, 2, or 3
n-Dodecane	Carbopack™ C/B, Tenax (SSV 63 000L) or Combination Tubes 1, 2, or 2

Sample Tube Sorbent	Approximate Analyte Volatility Range	Max. Temp (°C)
CarbotrapC CarbopackC Anasorb GCB2	n-C_8–n-C_{20}	>400
Tenax GC	bp 80 °C–200 °C	
Tenax TA	bp 100 °C–400 °C n-C_7–n-C_{30}	350
Teanx GR	bp 100 °C–450 °C n-C_7–n-C_{30}	350
Carbotrap CarbopackB Anasorb GCB1	(n-C_4) n-C_5–nC_{14}	>400
Choromosorb 102	bp 50 °C–200 °C	250
Chromosorb 106	bp 50 °C–200 °C	250
Porapak Q	bp50 °C–200 °C	250
Porapak N	bp 50 °C–150 °C n-C_5–n-C_8	180
Spherocarb	-30 °C–150 °C C_3–n-C_8	>400
Carboseive SIII Carboxen 1000 Anasorb CMS	-60 °C–80 °C	400
Zeolite Molecular Seive 13X	−60 °C–80 °C	350
Coconut Charcoal (rarely used)	−80 °C–??°C	>400

Table C.2: Guidelines for Sorbent Selection (USEPA, 1999)

C.1.2 Active Sampling into SUMMA Passivated Stainless Steel Canisters

Another variation of active sampling of indoor air and soil gas is the collection of whole air samples in SUMMA™ passivated stainless steel canisters coated with inert chrome-nickel oxide. This method involves drawing air into an evacuated SUMMA^{TM} canister via a sampling train. The sampling train includes a particle filter, mass flow controller, electronic timer, and Magnelatch valve. Sample analysis involves attaching the canister to a GC/MS and VOCs are cooled in a cryogenic trap and revolatilized prior to injection into the GC/MS. This method can detect ppb(v) concentrations of a variety of contaminants including BTEX. The advantages of this method include its sensitivity to a large number of VOCs. Furthermore, several samples may be collected over a prolonged period to provide a long term integrated sample. Limitations to this method are the cost of analysis by an outside laboratory or developing in house analytical capability, high skill level required by analytical personnel, and the potential for loss of samples or accidental contamination (USEPA, 1991). Another disadvantage to this method is the complex instrumentation required compared to sorbent methods (Yocom & McCarthy, 1991).

C.1.3 Cryogenic Preconcentration Techniques

Highly volatile compounds with boiling points in the range of -10–200 °C can be determined using cryogenic preconcentration. This method employs the use of a collection trap submerged in liquid oxygen or argon to preconcentrate air samples. Sampling is performed by withdrawing air into the trap via a sampling valve while simultaneously measuring air sample volume. The sample is transported to a GC, whose column oven is cooled to subambient temperatures (−50 °C), via the carrier gas. The liquid cryogen is simultaneously removed and the trap heated to facilitate sample transfer to the GC. Limitations of the cryogenic technique lies with interference in the GC apparatus from other compounds. A significant drawback of this technique is condensation of moisture in the collection trap. Ice may also plug the trap, stopping air flow. Water transferred to the capillary column may also stop flow and cause damage to column materials. Permaselective Nafion tubing can be used in-line before the cryogenic trap to avoid these problems. However this tubing must be used with caution due to possible losses of some compounds. Contamination from the Nafion tubing may also result (USEPA, 1999b).

Detection limits of 1–5 ng are found using the FID with cryogenic preconcentration techniques. Precision levels of ± 5% can be achieved at concentrations ten times the detection limit, and accuracy is estimated to be within ± 10% (USEPA, 1999b).

C.2 Passive Samplers

Passive samplers rely on diffusion to sample air pollutants. Passive sampler devices come in the form of badges, canisters, and tubes. Advantages of passive samplers are that they are simple to operate, are relatively inexpensive, and are unobtrusive. Limitations of passive samplers are that results are expressed as the measured concentration during a given time period (ppm-hr), long exposure times are required (e.g. 30 days), interference from other pollutants may occur, and results obtained may be affected by air currents and temperature changes (USEPA, 1991; Yocom & McCarthy, 1991).

For passive samplers, the effective sampling rate is defined by Fick's first law of diffusion.

$$\text{Sampling rate} = \frac{\text{D} \times \text{A}}{\text{L}} \qquad \text{(C.1)}$$

where, D is the diffusion coefficient for the compound being sampled, A is the cross-sectional area, and L is the diffusion path length. Passive samplers have low flow rates and need sufficient air movement across the sampling area to maintain constant sampling rates (Lewis & Gordon, 1996).

C.2.1 Sorbent Badges

Sorbent badges contain liquid or solid sorbents. Liquid sorbent badges are composed of a liquid-filled chamber with a bound-liquid membrane on the sampling interface. Air diffuses through the diffusion barrier and interacts with the liquid membrane. Flow rates range from 0.5 to 50 mL/min with a collection capacity range of 5 to 26 mL. Gas chromatography can be used for sample analysis. Solid sorbent badges can be used alone or with a backup, which is recommended due to the finite capacity of solid sorbents. After sampling, samples are sent to a laboratory for analysis by a number of methods including gas chromatography (USEPA, 1991).

C.2.2 Passive Bubbler

The passive bubbler consists of a glass vial filled with a sorbing solution and sealed with a Teflon faced Knudsen diffusion disk and cap. This type of sampling device is quite versatile as the vial can be attached to a subject's clothing or mounted on a stand to collect an area sample. Samples can be readily analyzed using GC procedures as well as High Performance Liquid

Chromatography, ion specific electrodes or colorimetry (USEPA, 1991)

Appendix D

Sample Handling and Transport

Sampling handling and transport is an important consideration especially for samples with low VOC concentrations. This is a critical period due to the opportunity for loss or gain of contaminants from or to sample containers. Sample loss can occur through contaminant condensation in the sampling system, sorption, thermal degradation, and leakage to the atmosphere. Sample contamination may also occur through insufficient decontamination procedures. The time between sample collection and analysis should be minimized to reduce sample loss and contamination. Integration of sampling and analytical systems can reduce problems associated with sample handling and transport. However, cross contamination may occur and can be attributed to tubing, flow meters, and analyzer components (Ullom, 1995).

Sample transfer from containers or analytical devices may be executed in a number of ways and is largely dependent on the sample volume recovered.

Small volume samples can be retrieved by a syringe for injection into an analyzer or a small volume container. Larger volume samples can be recovered by hand or mechanical pumps. Tubing is generally used when recovering large volumes of soil gas. However, for detection of low concentrations, tubing may pose a cross-contamination problem if not replaced often (Ullom, 1995).

The type of sample container selected is based on the physical properties of contaminants sampled, volume of sample recovered, physical properties of the contaminants, sampling system, anticipated sample holding time prior to analysis, and the analytical method chosen. The same container type should be used for each survey since container type can bias data due to sorptive or desorptive processes related to container type. Whole air samples may be contained in any device that can satisfy survey, handling, transport, and analytical requirements (Ullom, 1995).

Gas samples can be collected in stainless steel canisters, glass bulbs, Tedlar bags, syringes, and synthetic polymer traps (Hitzig et al., 1997; Tillman et al., 1989a). Glass and stainless steel containers provide the most representative samples and yield the best shelf life for VOCs (Tillman et al., 1989a). Although stainless steel canisters are durable, they are difficult to decontaminate and glass bulbs are easy to decontaminate but are breakable and may leak (Hitzig et al., 1997). Atmospheric sampling bags such as Tedlar bags are easy to use but may pose challenges due to leakage that can occur after repeated use (Ullom, 1995;Tillman et al., 1989a). In addition, contaminants may readily sorb onto the Tedlar bags. However, this is not a problem for primary gasoline constituents. Syringes are good sample containers as they are inexpensive and allow for easy collection of samples. Nevertheless, syringes have short handling times and can be difficult to decontaminate (Hitzig et al., 1997). Synthetic polymer traps can be used to concentrate volatiles to enhance detectability of certain compounds. A limitation of polymer traps is that it produces less representative data due to competition for adsorptive sites. The sample containers chosen should be made of non-reactive material and protected from light and kept tightly sealed to prevent dilution, contamination, or compositional changes (Tillman et al., 1989a).

Appendix E

Field Screening and Analytical Methods

Field screening methods can indicate the presence or absence of a chemical or whether the chemical is above or below a threshold, but cannot provide chemical specific concentration information. Analytical methods comprise those analyses that provide chemical-specific, quantitative data in the field, laboratory or non-laboratory scenario. Field analytical techniques are usually faster and less expensive than simlar chemical analyses undertaken in a laboratory (USEPA, 1993).

Soil gas and indoor air can be screened for VOCs by a number of methods including portable VOC detectors (i.e. FIDs and PIDs) and gas chromatography. Portable VOC detectors are easily transported in the field, require minimal operator skill, and can provide immediate data (Ullom, 1995). VOC analysis by GC methods provide the best analysis but are costly. Portable GCs are available but do not provide the high resolution results available from a remote laboratory (Tillman et al., 1989a). Selection of a detector depends on a number of variables including the analyte, budget constraints, required detection limits, potential interferences, project objectives, equipment, and operator capability. Precision, accuracy, sensitive and potential interferences vary depending on the analyte (USEPA, 1991).

E.1 Field Screening Methods

On-site soil gas monitoring equipment can be used to screen contaminated sites. Field based screening methods for the detection of soil gas and indoor air allow for a site to be surveyed for VOCs in a short time period. A number of portable analytical instruments contain or consist of a PID or FID. Most units are designed solely as PID or FID response meters without chromatographic separation and provide data on total organic vapour (TOV) concentrations. Most portable field units are designed to analyze gas samples at ambient temperatures. Due to the lack of chromatographic separation, total organic vapour methods are limited to analysis of samples containing known contaminants (Fitzgerald, 1989).

A variety of environmental factors can affect TOV instrument response to organic vapours. Such factors include soil air composition, relative humidity of soil air and soil permeability. The composition of soil air may differ from the composition of the calibration standard used (Robbins et al., 1990). Total organic vapour analyzers may also detect naturally occuring volatiles not measured in a laboratory such as methane and are sensitive to changes in ambient conditions resulting in inaccurate readings (Robbins et al., 1990;

Tillman et al., 1989a).

Advantages of total detectors such as the FID and PID are that they are very portable and easy to use, are relatively inexpensive (about 5 000 USD), and have a rapid response time. Limitations of these detectors include their non-specificity (USEPA, 1993).

E.1.1 Photoionization Detector

A portable photoionization detector (PID) can be used in the field for the screening of soil gas levels at an affected site and can be adjusted to selectively detect aromatic compounds (Weisman, 1998). The PID uses an ultraviolet lamp as an ionizing source and responds to VOCs that have an ionization potential less than or equal to the lamp. Concentrations are reported as total ionizable compounds (USEPA, 1993).

The photoionization reaction takes place when photo-excitation occurs to such an extent that the electron is removed from its molecular orbital. Equation E.1 illustrates this reaction:

$$\mathrm{R} + \mathrm{h}\nu \rightarrow \mathrm{R}^{+} + \mathrm{e}^{-} \quad . \tag{E.1}$$

The available lamp energy for a PID ranges from 8.3 to 11.7 eV. Aromatic hydrocarbons or heteroatom containing compounds are best analyzed with a PID because these species have ionization potentials that are within the reach of available UV lamps (Chasteen, 2000b). Figure E.1 illustrates a schematic diagram of a PID.

Advantages of the PID are its portability and ease of use. The major disadvantage of the PID is its non-specificity (USEPA, 1993). The PID

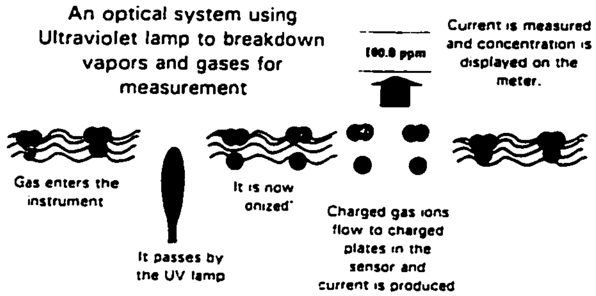


Figure E.1: Schematic of a Photo Ionization Detector (RAE Systems)

can be used to measure volatile aromatic compounds such as BTEXs, some oxygenates such as methyl-t-butyl ether (MTBE) and many olefins, some branched alkanes and cycloalkanes (Weisman, 1998). The response of the PID increases with the degree of compound separation. This results in a benzene response that is about five times greater than hexane when using a 10.2 eV lamp. Therefore with this selectivity, BTEX compounds can be focused on in a sample (Fitzgerald, 1989). Table F.1 in Appendix B lists some PIDs and their manufacturers.

E.1.2 Flame Ionization Detector

The Flame Ionization Detector (FID) uses a hydrogen flame to ionize organic vapours entering the detector and reports concentrations of total organics as the ppm equivalent to a calibration compound (usually methane) (USEPA, 1993). Organic vapours are ionized in a mixture of H_2 and air. Carbon atoms, with the exception of carbonyl and carboxyl carbons, produce CH- radicals, which then produce CHO^+ ions in the flame. Equation E.2 illustrates this reaction:

$$CH + O \rightarrow CHO^+ + e^- \tag{E.2}$$

CHO^+ produced in the flame is collected at the cathode located above the flame. The response to organic compounds is directly proportional to the solute mass over seven orders of magnitude. When organic solutes are absent, the current is almost zero (Harris, 1995). Unlike the PID, FIDs respond to methane and may read significantly higher total organic vapour concentrations in some media samples. Furthermore, the FID is sensitive to more VOCs that the PID (USEPA, 1993). Table F.1 in Appendix B lists FIDs and their manufacturers. Figure E.2 illustrates a schematic diagram of a FID.

E.2 Analytical Methods

Analytical methods can provide quantitative data in the field or in the laboratory. Gas chromatography is the most commonly used analytical technique and can be used both in the field and in a mobile or traditional laboratory setting.

E.2.1 Gas Chromatography

Gas chromatography (GC) involves the separation of gases on a stationary phase in a column, which is held on a solid support (USEPA, 1993). In gas chromatography, a volatile liquid or gas is injected through a rubber

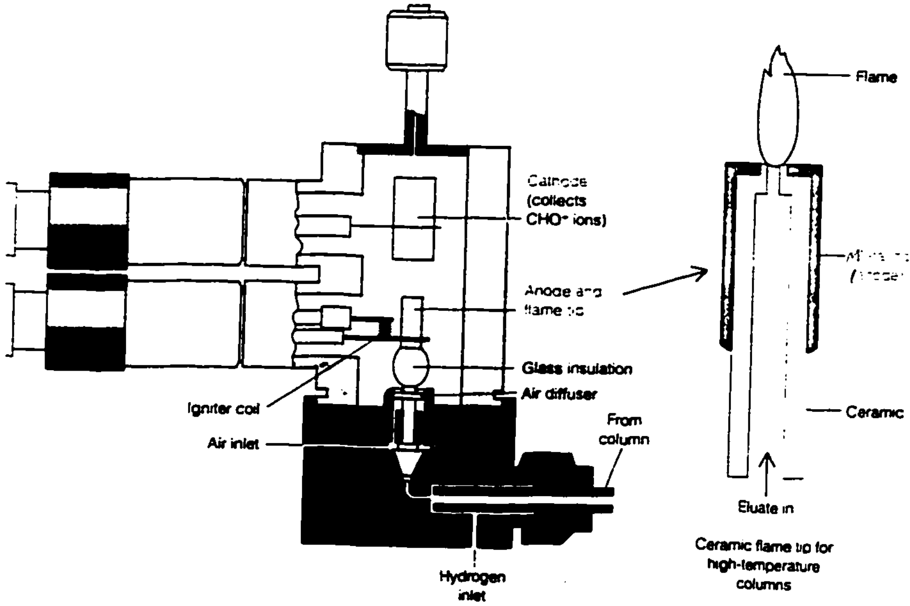


Figure E.2: Schematic of a Flame Ionization Detector (Harris, 1995)

septum into a heated port, which vapourizes the sample. The sample is then swept through the column by a He, N_2, or H_2 carrier gas, and the separated gases flow through a detector, whose response is displayed on a recorder or computer. The column should be hot enough to produce sufficient vapour pressure for each solute to be eluted in a reasonable time. The detector is maintained at a higher temperature than the column, so that all solutes are gaseous (Harris, 1995). A schematic diagram of a gas chromatograph is shown in Figure E.3.

Different detector types can be used with a gas chromatograph (GC). A flame ionization detector (FID) or a photoionization detector (PID) can be

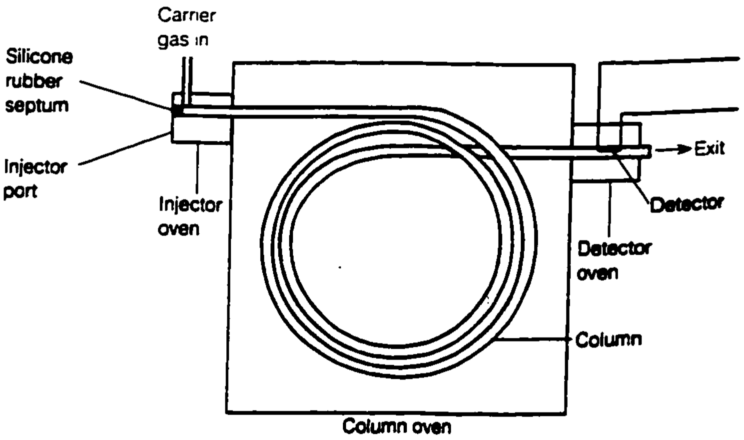


Figure E.3: Schematic of a Gas Chromatograph (Harris, 1995)

used to detect specific compounds after they have been separated in the GC. GC is also used as a sample preparation step for other types of instrumentation, such as the mass spectrometer (MS) (USEPA, 1993).

Advantages of using a GC are its portability, its specificity, and its sensitivity (ppb to ppm). Limitations of the GC are that it is less sensitive than mass spectrometers, it has a slower response time compared to mass spectrometers (tens of minutes vs. seconds), time consuming calibration steps, and a library of retention times is required to identify compounds and non-target compounds, making identification difficult if detected analytes are not in the library or the quality of the library match is too low to make positive identification (USEPA, 1993).

VOCs may be detected and quantified to different degrees by GC. The GC/FID is the most general method for detecting and quantifying hydrocarbons because the FID is sensitive to most organic compounds and retains a linear response over a wide range of conditions. The GC/PID is used for determination of aromatic and unsaturated compounds but is insensitive to other organics. GC/MS (mass spectrometry) is used for the most precise work since it allows better resolution of components in complex mixtures compared to MS alone (USEPA, 1993).

Portable gas chromatographs provide real-time data collection in the field. The major disadvantage is the high detection limit which are usually higher than concentrations found indoors (Yocom & McCarthy, 1991). A wide range of quality and accuracy exists for gas chromatographs. Because field GCs cannot provide the detailed high-resolution results of lab equipment, lab quality GCs are preferred for reliable results. Mobile laboratories with lab grade GCs can provide high quality results for field analysis when necessary (Tillman et al., 1989a). Table F.1 in Appendix F lists Portable GCs and their manufacturers.

E.2.2 Mass Spectrometry (MS) and GC/MS

Mass spectrometry involves the conversion of compounds in a sample into charged ionic particles consisting of the parent ion and ionic fragments of the original molecule. The characteristic mass/charge ratios allow for compound identification, and ion current magnitudes read at different mass settings can be related to concentration. The main components of a mass spectrometer include the inlet system, the ion source, the electrostatic accelerating system, and the detector and readout system that gives a mass spectrum recording the numbers of different kinds of ions. Mass spectrometers often are used in conjunction with gas chromatography (USEPA, 1993). A schematic diagram of a Mass Spectrometer is shown in Figure E.4.

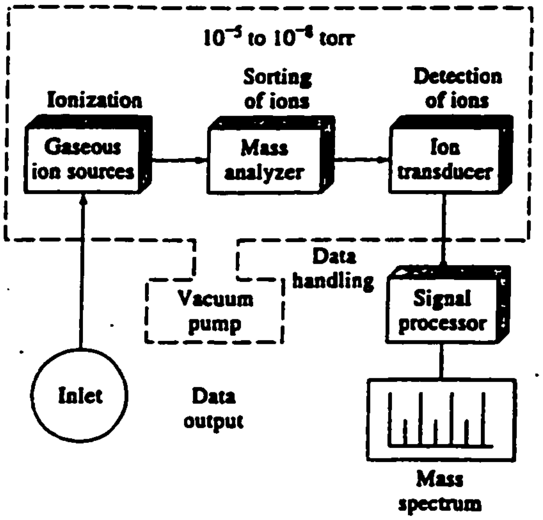


Figure E.4: Schematic of a Mass Spectrometer (Skoog et al., 1998)

A smaller version of a mass spectrometer can be coupled to a GC and is called gas chromatography/mass spectrometry (GC/MS). The mass detector is placed at the end of a chromatographic column similar to other GC detectors. However, the mass detector is more complicated in comparison to other detectors due to its complex requirements for the process of creating, separating, and detecting gas phase ions (Chasteen, 2000a).

In the GC/MS, volatile compounds are introduced into the GC. A temperature programmed column is used to separate analytes, which are detected with a MS interfaced to the GC. Analytes from the capillary column in the GC are introduced into the MS by a jet separator or direct connection. Analyte identification is achieved by comparing mass spectra with electron impact of standards. Analyte concentrations are determined by comparison of major ion responses to internal standards using a five-point calibration curve (USEPA, 2000b).

Method 8260B in SW-846 describes the procedure for analysis of VOCs by GC/MS. A list of compounds that can be determined by this method is also listed in SW-846. Analytes from solid samples can be introduced into the GC/MS by purge and trap (Method 5035), automated static headspace for solid samples (Method 5021), and closed system vacuum distillation (Method 5032) for solid samples. Gaseous analytes can be analyzed directly using a sample loop by sub-sampling from Tedlar bags (Method 0040). GC/MS can be used to quantify most VOCs with boiling points below 200 °C (USEPA, 2000b).

Advantages of a MS detector include its sensitivity (ppb range), rapid response time, small sample sizes (mg to μg) and it has good specificity in non-complex matrices. Limitations include its bulk and fragility, its expense (50 000 to 200 000 USD) and the requirement of a library of spectra (USEPA, 1993).

E.2.3 Gas Chromatography/Flame Ionization Detection

A gas chromatography/flame ionization detector (GC/FID) detects hydrocarbons that elute from the column and burn. GC/FID methods specify a certain portion of the chromatogram for quantification. The carbon number range will approximate that of the fuel of interest - gasoline, diesel, or heavier hydrocarbons. Volatile compounds that elute before the solvent peak (usually those $<C_6$) are typically not measured.

The advantages of GC/FID include quantitative measurement of all organic compounds in the gasoline range, ability to generate a finger print that can be used to provide supplemental information, is sensitive to hydrocarbons in the nanogram range, and a PID can be integrated in series with the GC/FID to measure volatile aromatics (Parr et al., 1991). The FID is sensitive to a larger number of VOCs than the PID such as methane, ethane, and certain toxic gases with high ionization potential, such as carbon tetrachloride and HCN (USEPA, 1993).

E.2.4 Gas Chromatography/Photoionization Detection

The gas chromatograph can be interfaced with a photoionization detector (GC/PID) to allow for selectivity and sensitivity in the analysis of soil gas. For light aromatics, typical detection limits are 0.5 μg/L in water and 5 μg/L in soil (Weisman, 1998). The PID is used to selectively determine aromatic hydrocarbons or organo-heteroatom species. The PID uses ultraviolet light to ionize analytes exiting from the GC column. The ions produced are collected by electrodes and the current generated is proportional to the analyte concentration (Chasteen, 2000b).

Appendix F

Field Screening and Analytical Instruments

Table F.1: Field Screening and Analytical Instruments and Distributors

Instrument	Detection Principle	Detection Range	Sensitivity	Canadian Distributor
OVM/GC	PID	0–2000ppm	0.1ppm with GC: 5 ppb	SSCAN-Grodyne 90 West Beaver Creek Rd. Richmond Hill, ON L4B 1E7 Tel: (800) 263-4701
HW-101	PID	-	-	HNU Canada 3 Robert Speck Pkwy, Suite 900 Mississauga, ON L4Z 2G5 Tel: (905) 949-0475
2020	PID	-	0.5 ppm	PE Photovac 330 Cochrane Drive Markham, ON L3R 8E5 Tel: (905) 477-8088
Gascorder	PID or FID	-	0.1 ppm	MSA Canada Inc. 148 Norfinch Dr. North York, ON M3N 1X8 Tel: (416) 667-9400
MicroTIP	PID	0–2000 ppm	0.1 ppm	CAN-AM Instruments 2495 Haines Rd. Mississauga, ON L4Y 1Y7 Tel: (905) 277-0331
Scentogun	PID	0–2000 ppm	0.1 ppm	CAN-AM Instruments 2495 Haines Rd. Mississauga, ON L4Y 1Y7 Tel: (905) 277-0331
DL-101	PID	-	-	HNU Canada 3 Robert Speck Pkwy, Suite 900

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continued from previous page

Instrument	Detection Principle	Detection Range	Sensitivity	Canadian Distributor
				Mississauga, ON L4Z 2G5 Tel: (905) 949-0475
UltraRAE	PID with vacuum pump and UV lamp	0–200 ppm (benzene)	0.1 ppm (benzene)	Nortech GSI Inc. 5821-51 Ave Beaumount, Alberta T4X 1B7 Tel: (780) 929-9583
MiniRAE	PID	0–999 ppm (isobutylene) 100-10 000 ppm (isobutylene)	0.1 ppm (isobutylene) 0.2–1 ppm (isobutylene)	Nortech GSI Inc. 5821-51 Ave Beaumont, Alberta T4X 1B7 Tel: (780) 929-9583
TVA-100	PID, FID	-	PID:0.1ppm FID:1 ppm	Nortech Control Equipment, Inc. 1131 Derry Road E. Mississauga, ON L5T 1P3 Tel: (905) 564-4700
Model 860 Hydrocarbon Vapor Monitor	FID	0–20 000 ppm	0.1 ppm	SSCAN-Grodyne 90 West Beaver Creek Rd Richmond Hill, ON L4B 1E7 Tel: (800) 263-4701
MicroFID	FID	0.1–50 000 ppm	-	PE Photovac 330 Cochrane Drive Markham, ON L3R 8E5 (tel) (905) 477-8088
Portable FID	FID (GC Option)	0–10 000 ppm	1 ppm	Novatech Controls Inc. 5600 Cypihot Ville Saint-Laurent, QC H4S 1V7 Tel: (514) 339-5371
Detecto-Pak IIM	FID	-	-	Health Consultants Canada 2085 Piper Lane

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Instrument	Detection Principle	Detection Range	Sensitivity	Canadian Distributor
				London, ON N5V 3S5 Tel:(519) 659-1144
Detecto-Pak III	FID	-	-	Health Consultants Canada 2085 Piper Lane London, ON N5V 3S5 Tel:(519) 659-1144
Pota-FID II	FID	0–5000 ppm	-	Health Consultants Canada 2085 Piper Lane London, ON N5V 3S5 Tel:(519) 659-1144
Century OVA 128	FID	0–100 ppm	-	Nortech Control Equipment Inc. 1131 Derry Road E. Mississauga, ON L5T 1P3 Tel: (905) 564-4700
Centry OVA 108/88	FID	1– 10 000 ppm (methane)	-	Nortech Control Equipment Inc. 1131 Derry Road E. Mississauga, ON L5T 1P3 Tel: (905) 564-4700
TVA-100	PID, FID	-	PID:0.1ppm FID: 1ppm	Nortech Control Equipment, Inc. 1131 Derry Road E. Mississauga, ON L5T 1P3 Tel: (905) 564-4700
Model 8610 BTEX GC	GC/ FID, PID, TCD, ECD, D FP	-	Dependent on detector and accessories chosen	Chromatographic Specialties Inc. PO Bag 1150 Brockville, ON K6V 5W1 Tel: (613) 342-4678

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Instrument	Detection Principle	Detection Range	Sensitivity	Canadian Distributor
MSI-301 Organic Vapour Monitor	GC/solid state	-	1 ppb	Nortech Control Equipment Inc. 1131 Derry Road East Mississauga, ON L5T 1P3 Tel: (905) 564-4700
Quad 4	GC/solid state	-	-	Nortech Control Equipment Inc. 1131 Derry Road E. Mississauga, ON L5T 1P3 Tel: (905) 564-4700
Scentograph	GC/ECD, /AID, , /PID,/TID	Detector Dependent	-	Geneq 7978 Jarry E Montreal, QC H1J 1H5 Tel: (514) 354-2511
Scentoscreen (Plus)	GC/ECD, /AID, /PID	-	1 ppb (benzene)	Geneq 7978 Jarry E Montreal, QC H1J 1H5 Tel: (514) 354-2511
Snapshot	GC/PID	-	-	CAN-AM Instruments 2495 Haines Rd. Mississauga, ON L4Y 1Y7 Tel: (905) 277-0331
10S Plus	GC/PID	-	50 ppb	CAN-AM Instruments 2495 Haines Rd. Mississauga, ON L4Y 1Y7 Tel: (905) 277-0331
EGC700A	GC/PID	-	1 ppb (benzene)	American Scientific Instrumentation 107 Hawthorne Ave.

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Instrument	Detection Principle	Detection Range	Sensitivity	Canadian Distributor
				Park Ridge, NJ USA 07656 Tel: (201) 391-0922
Mini-GC	GC,/TCD, FID, PID, ECD	-	-	Chromato-Sud 19, rue Nauville 33000 Bordeaux, France Tel: (33) 5624-4649
Model 311 GC	GC/PID (/ECD)	-	-	HNU Canada 3 Robert Speck Pkwy, Suite 900 Mississauga, ON L4Z 2G5 Tel: (905) 949-0475

Appendix G

Soil Quality Guidelines for Coarse-Grained Soils for Buildings with Basements, Agricultural/Residential Exposure Scenarios

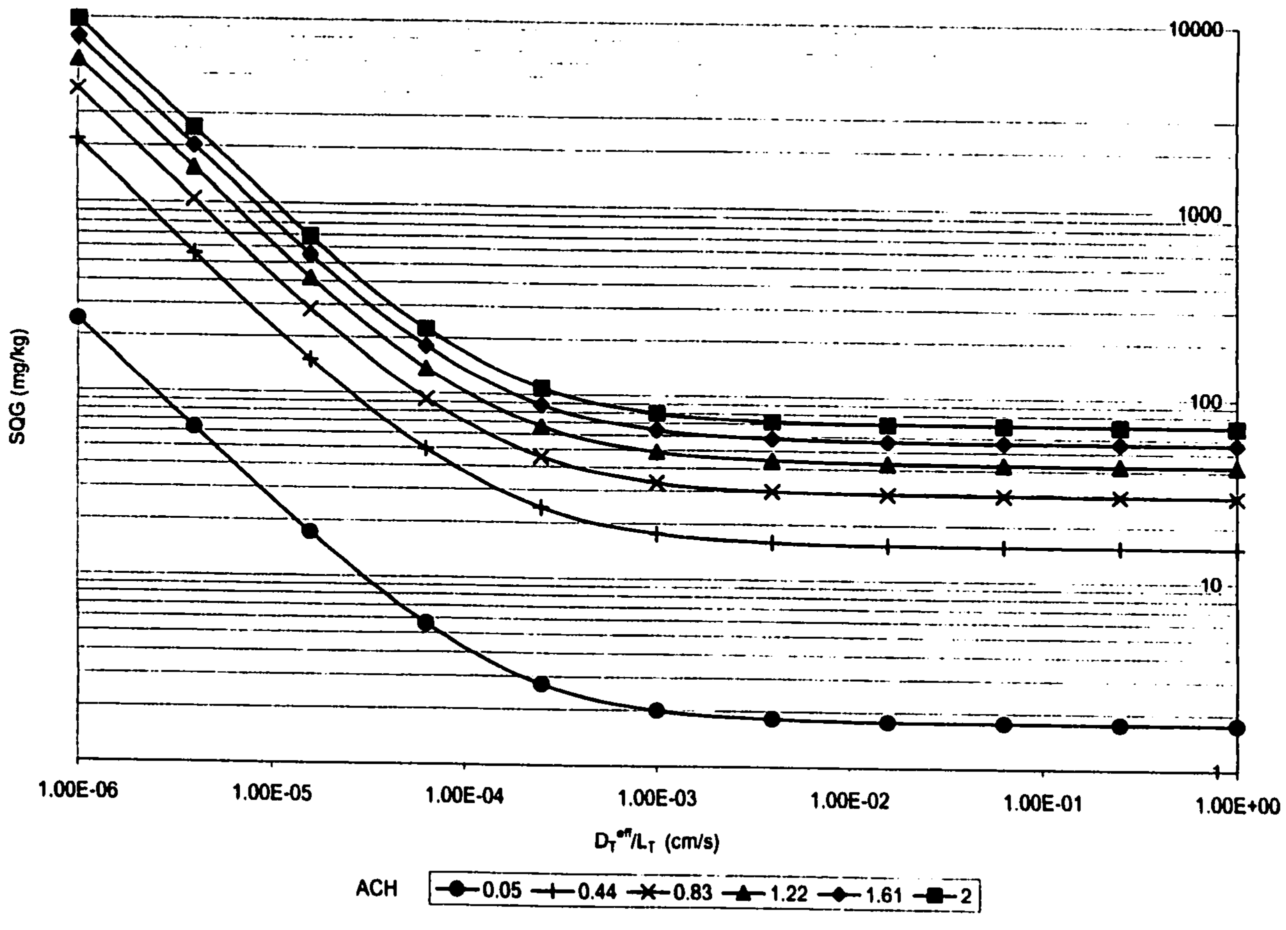

Figure G.1: SQG for Benzene in Coarse-Grained Soils for a Residential Building with a Basement for an Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

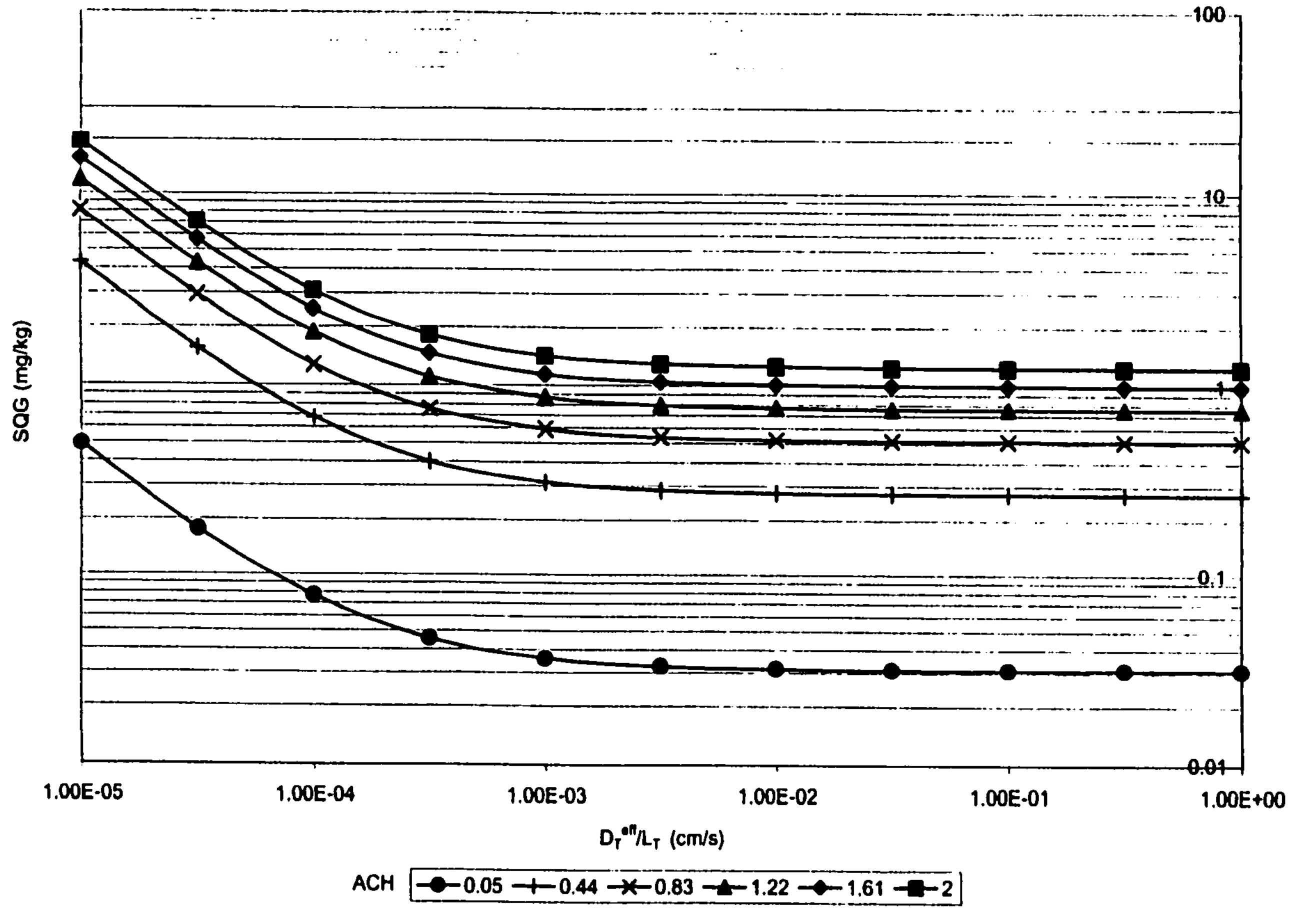

Figure G.2: SQG for Toluene in Coarse-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

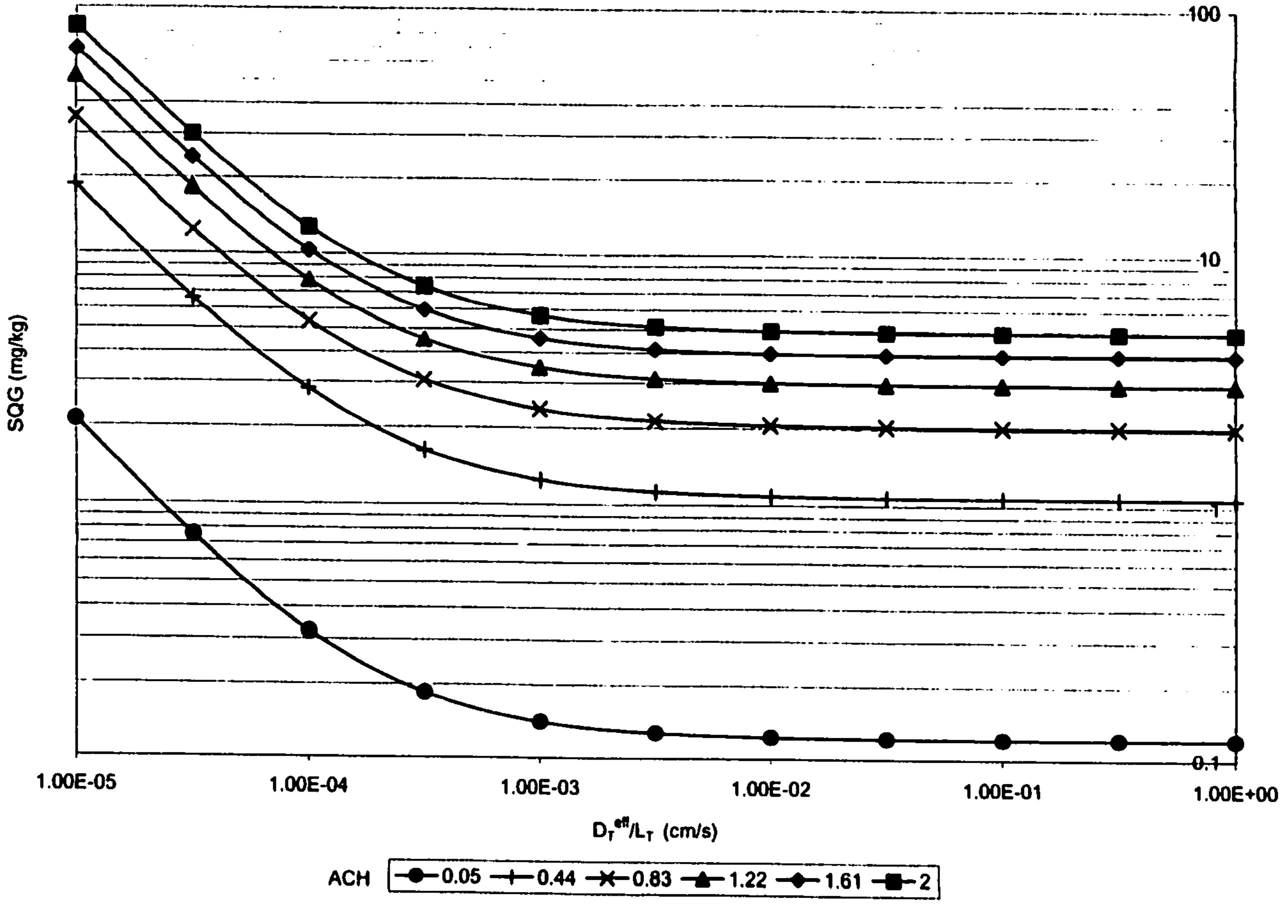

Figure G.3: SQG for Ethylbenzene in Coarse-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

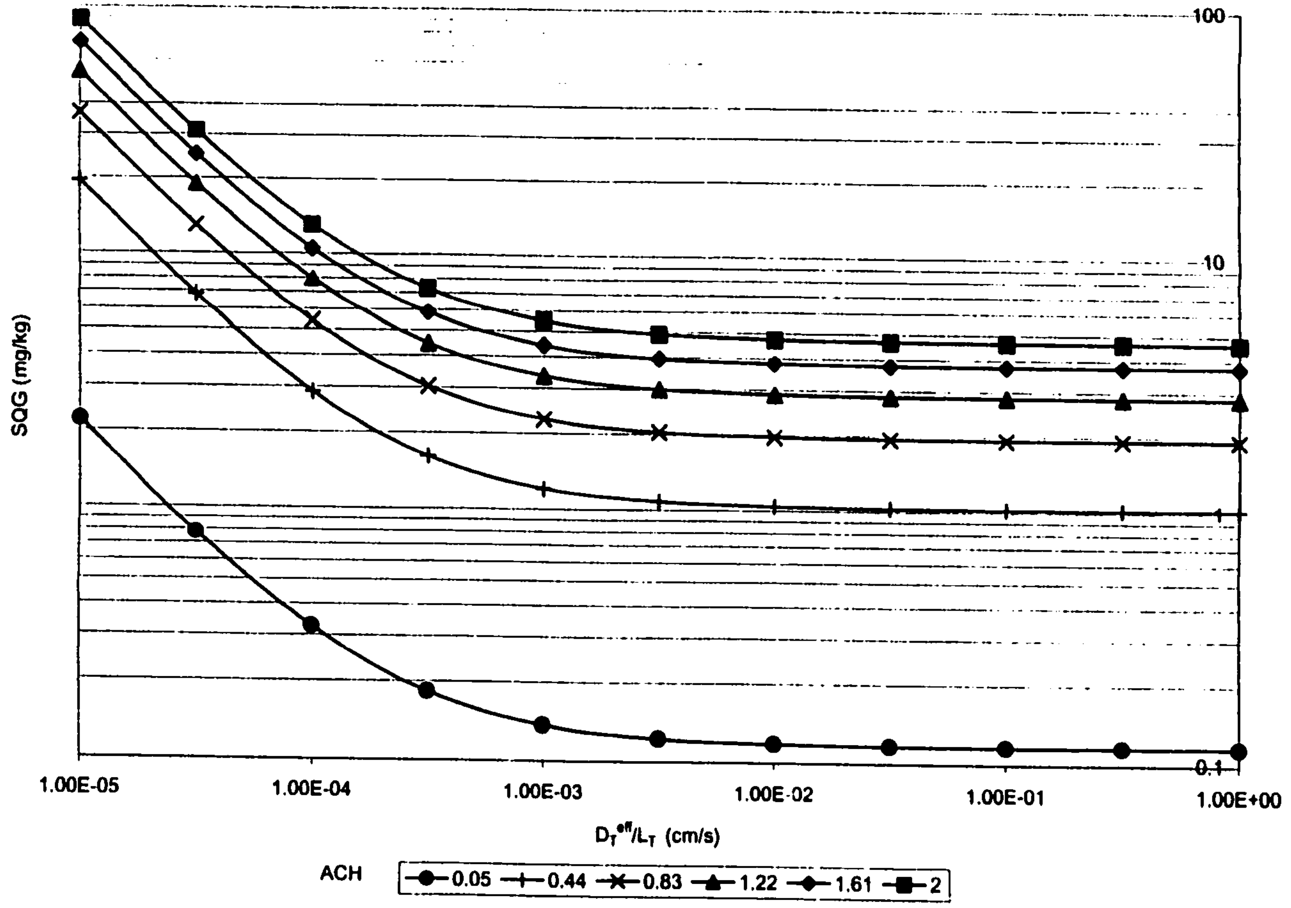

Figure G.4: SQG for o-Xylene in Coarse-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

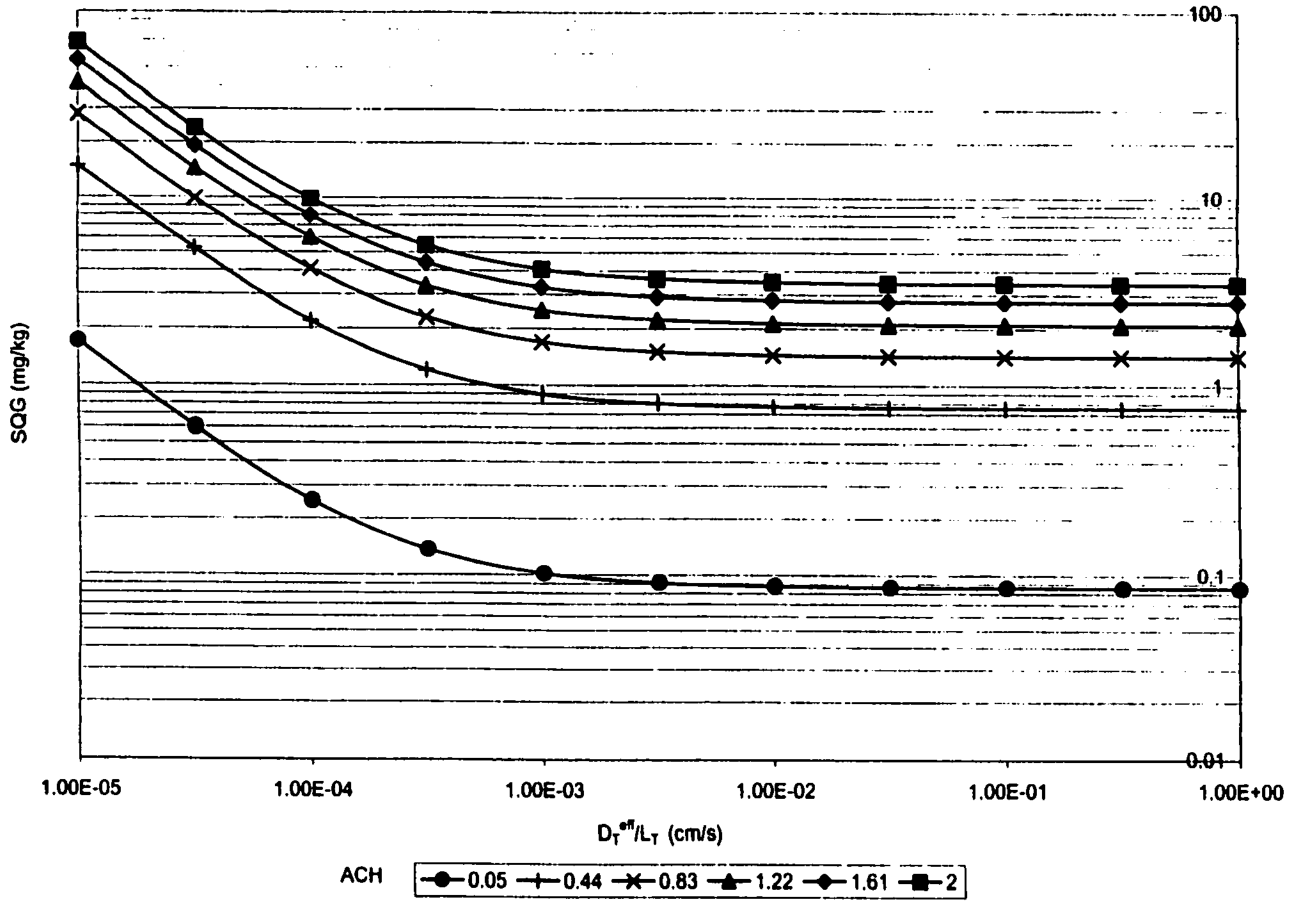

Figure G.5: SQG for m-Xylene in Coarse-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

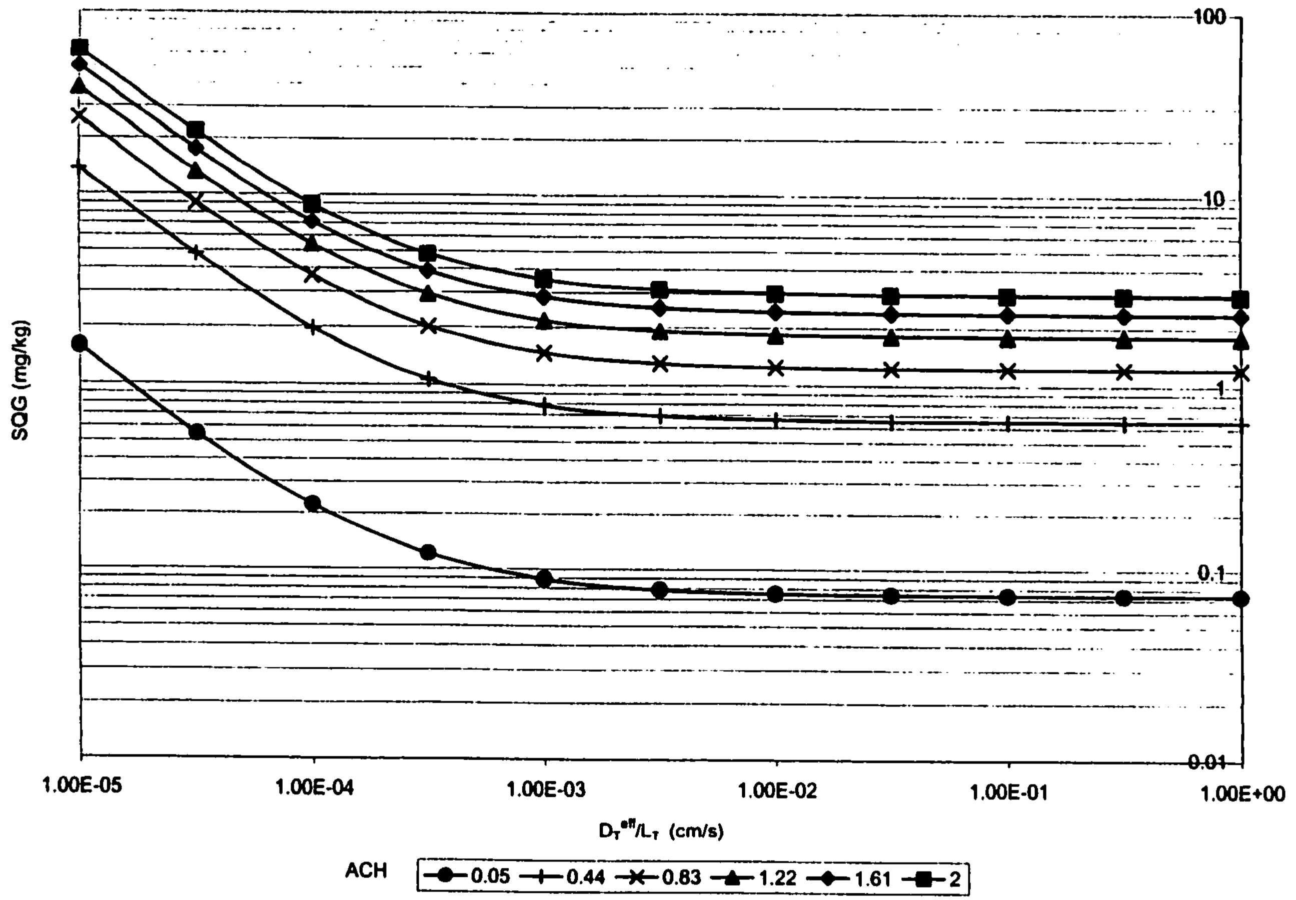

Figure G.6: SQG for p-Xylene in Coarse-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_t^{eff}}{L_T}$ for various air exchange rates

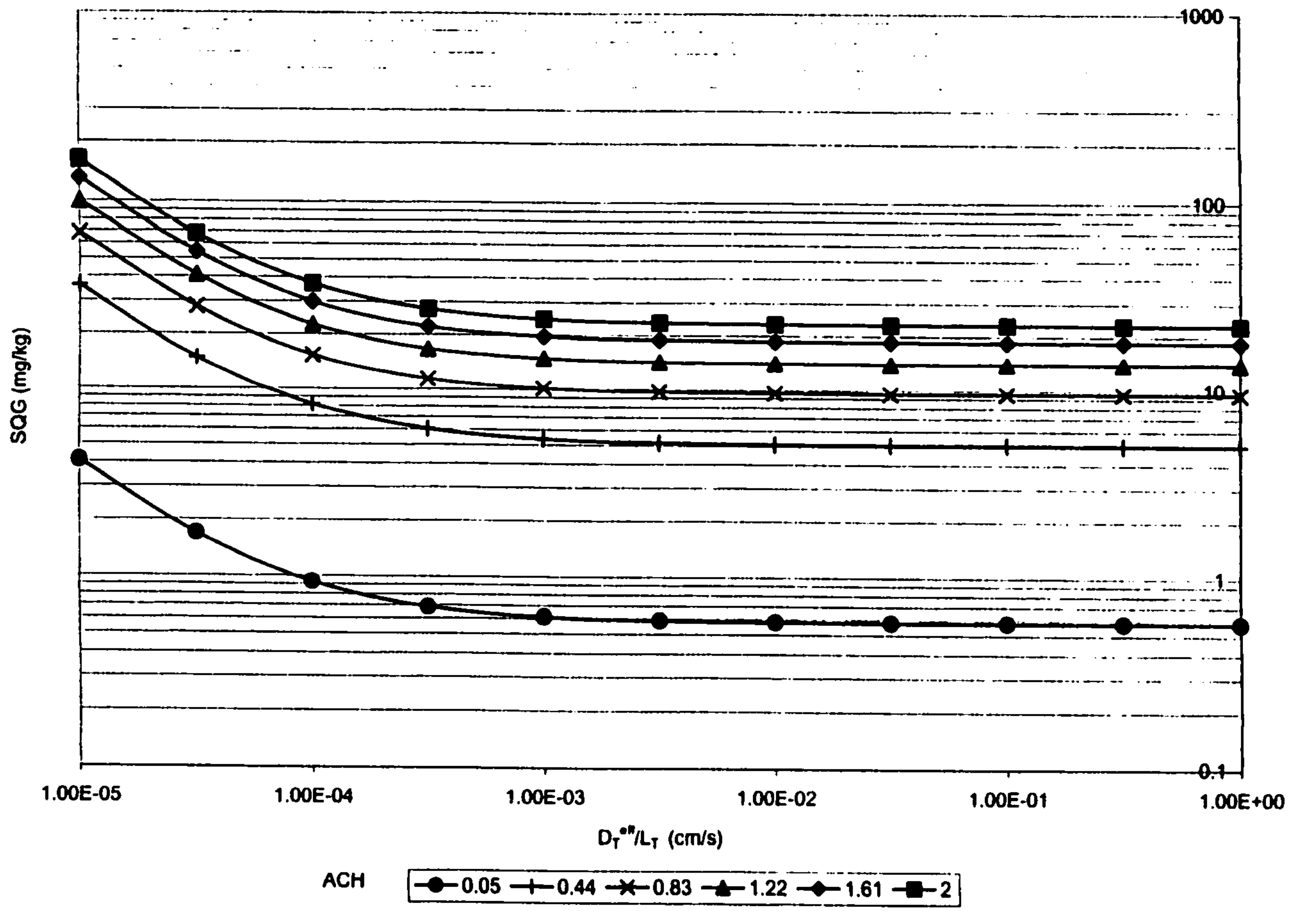

Figure G.7: SQG for C_6–C_8 aliphatic hydrocarbons in Coarse-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

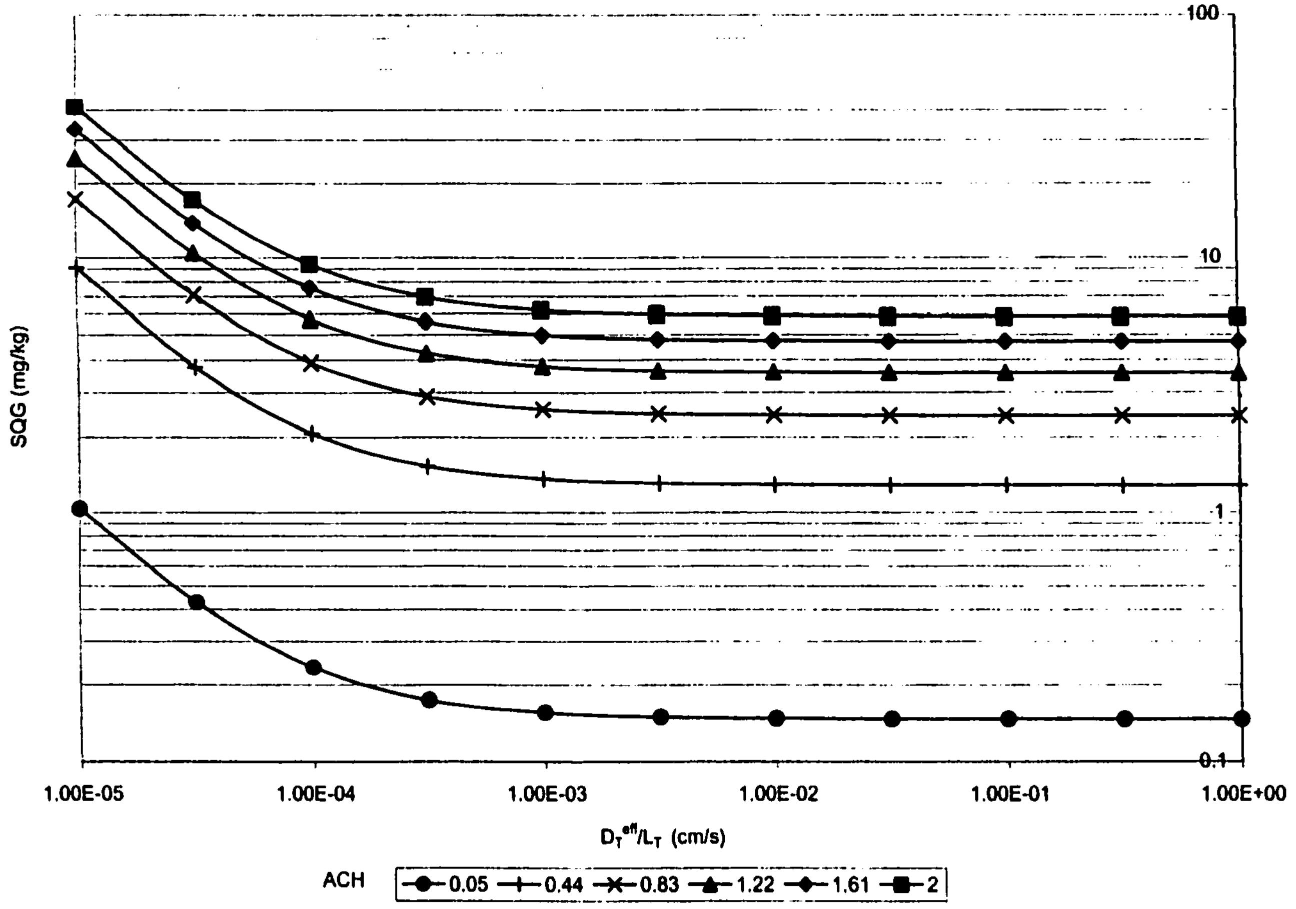

Figure G.8: SQG for $C_{>8}$–C_{10} aliphatic hydrocarbons in Coarse-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_t^{eff}}{L_T}$ for various air exchange rates

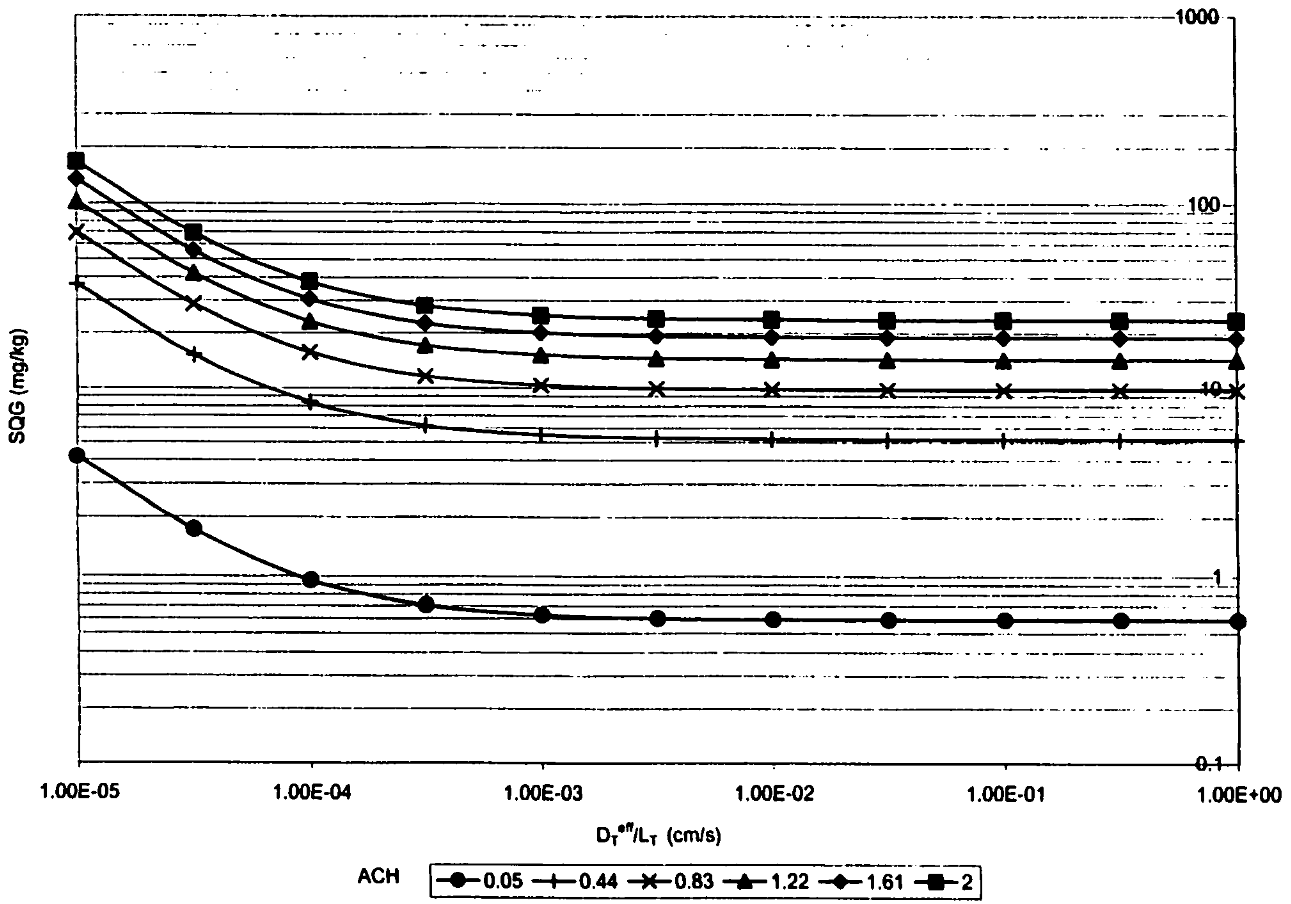

Figure G.9: SQG for $C_{>10}$–C_{12} aliphatic hydrocarbons in Coarse-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

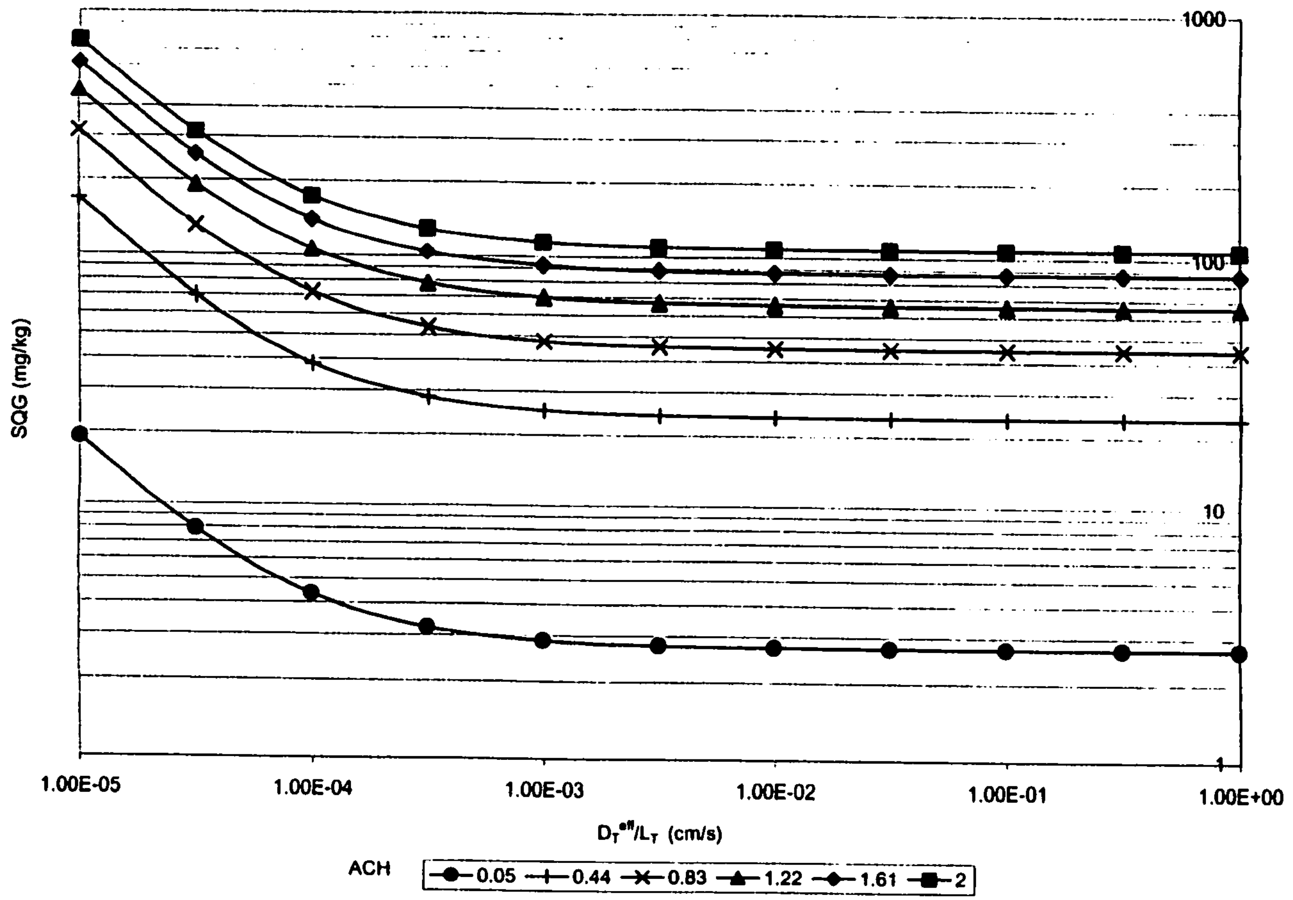

Figure G.10: SQG for $C_{>12}$–C_{16} aliphatic hydrocarbons in Coarse-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

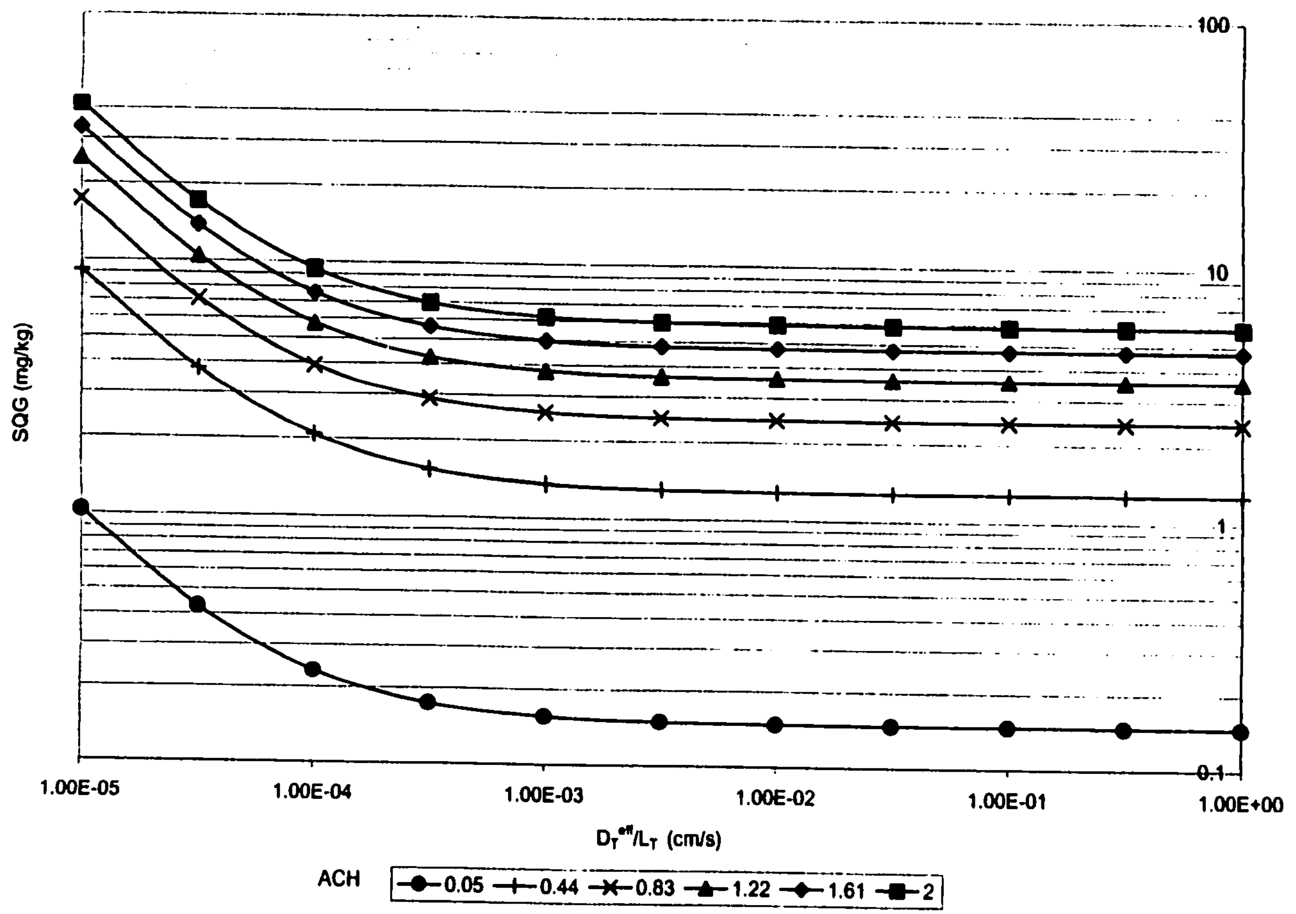

Figure G.11: SQG for $C_{>8}$-C_{10} aromatic hydrocarbons in Coarse-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_t^{eff}}{L_T}$ for various air exchange rates

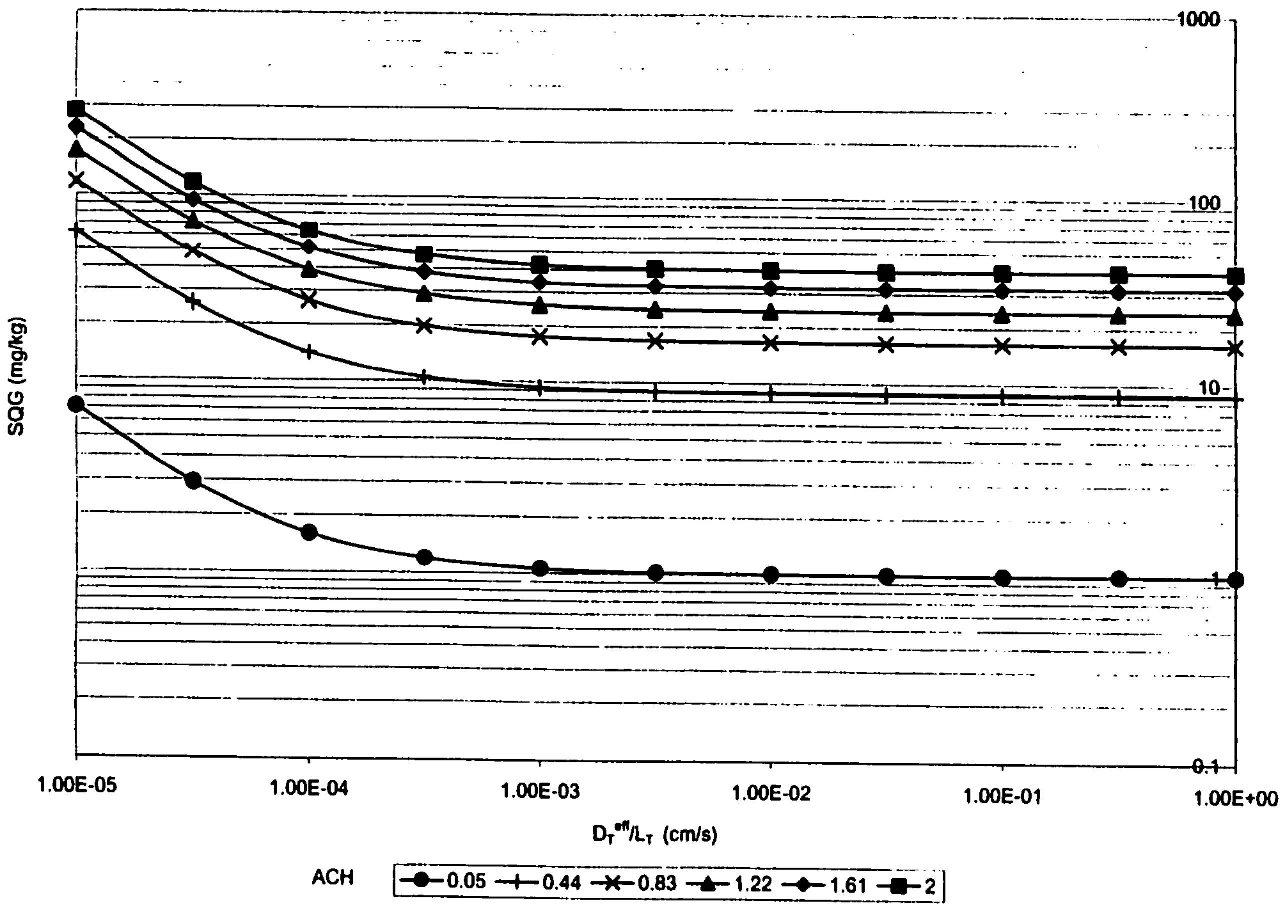

Figure G.12: SQG for $C_{>10}$–C_{12} aromatic hydrocarbons in Coarse-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_t^{eff}}{L_T}$ for various air exchange rates

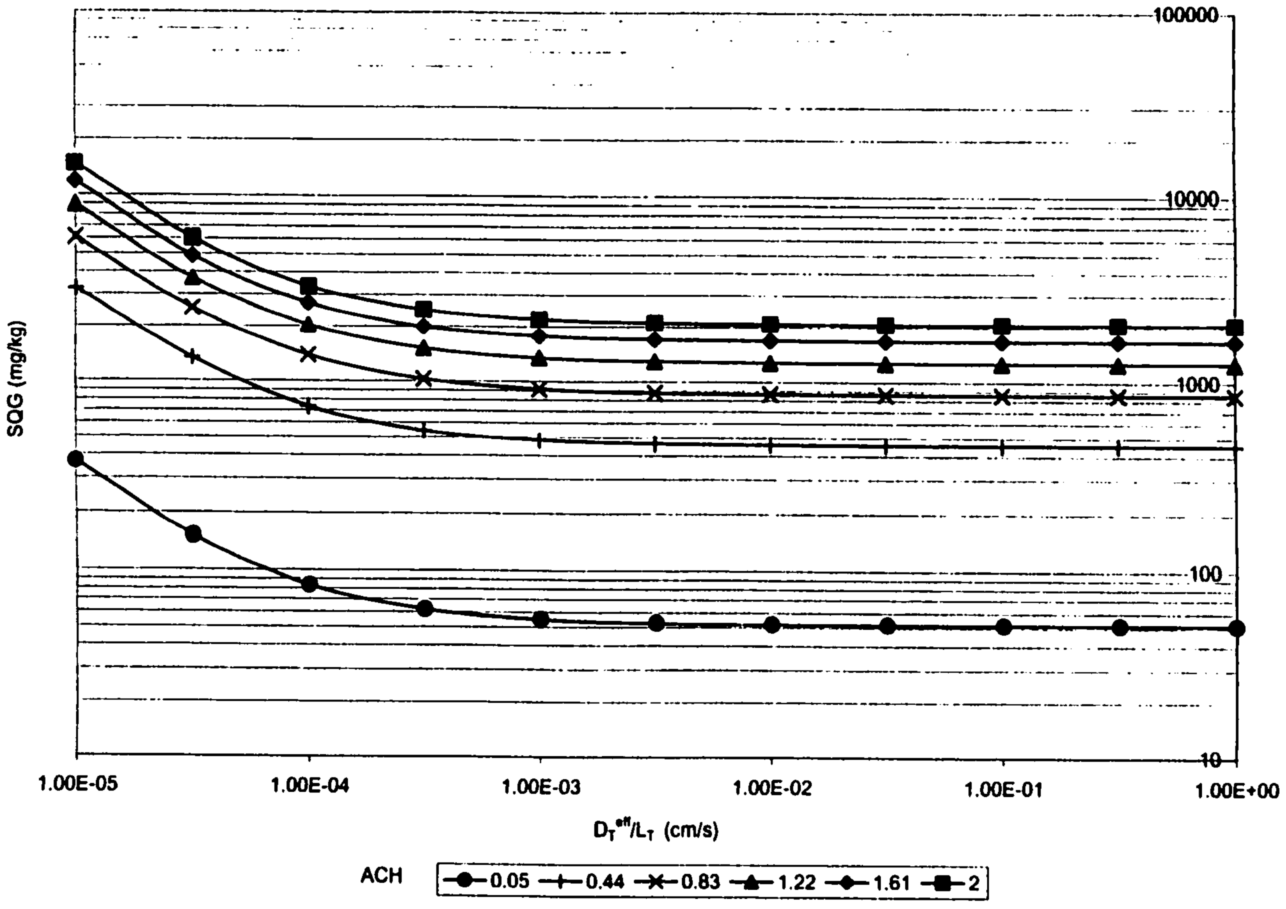

Figure G.13: SQG for $C_{>12}$–C_{16} aromatic hydrocarbons in Coarse-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

Appendix H

Soil Quality Guidelines for Coarse-Grained Soils for Slab on Grade Structures, Commercial/Industrial Exposure Scenarios

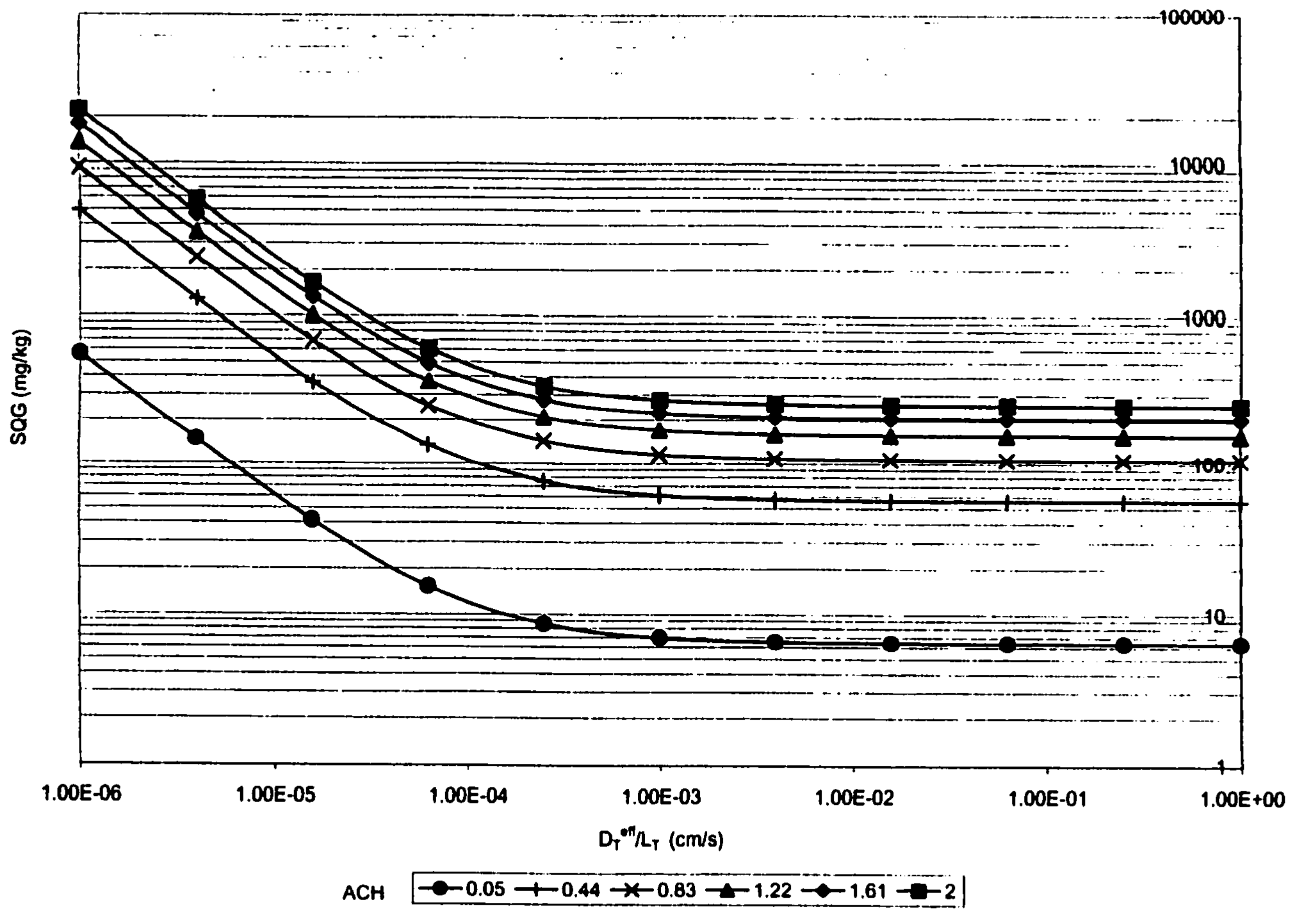

Figure H.1: SQG for Benzene in Coarse-Grained Soils for a Slab on Grade Structure, Commercial/Industrial Exposure Scenario vs. $\frac{D_t^{eff}}{L_T}$ for various air exchange rates

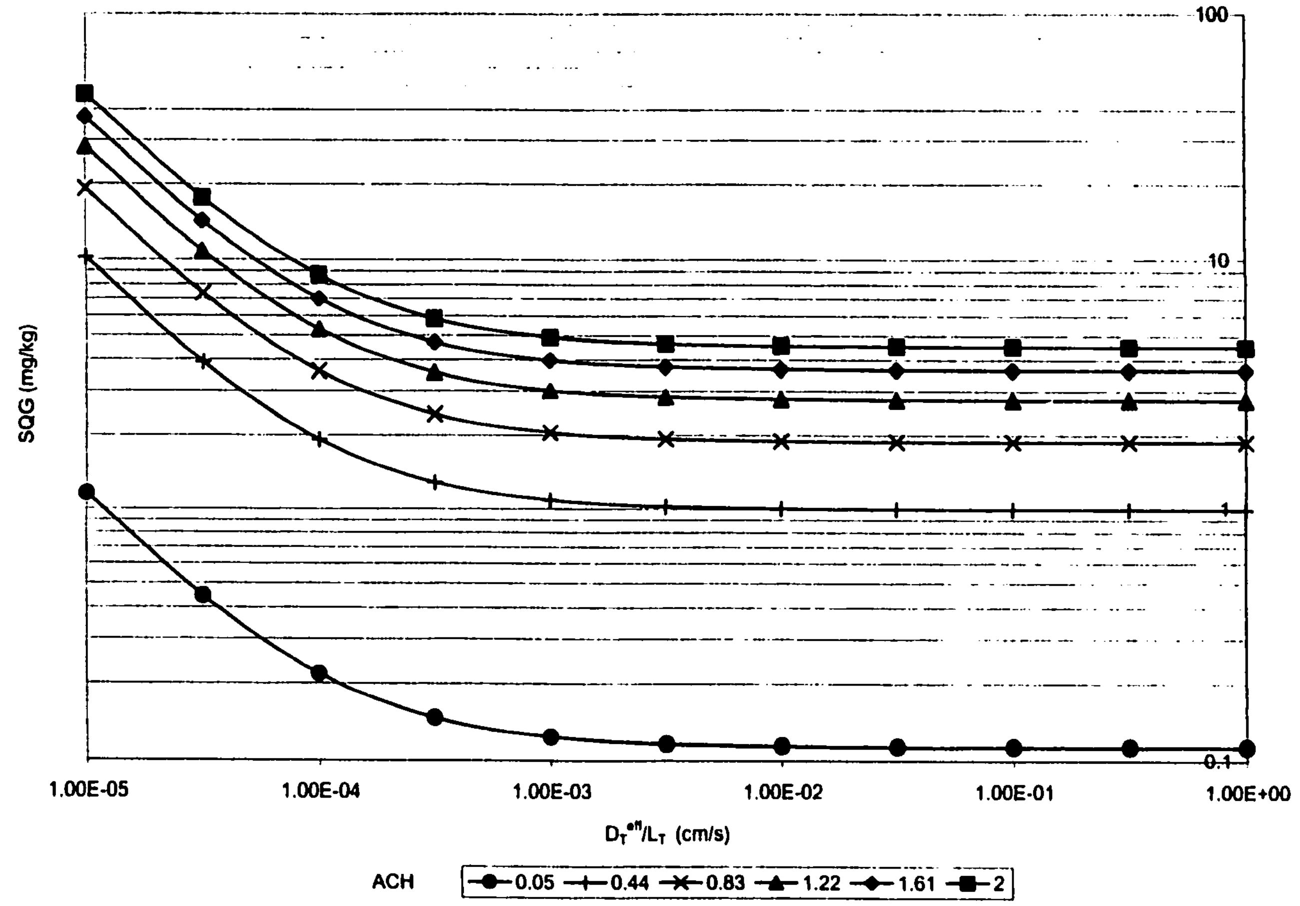

Figure H.2: SQG for Toluene in Coarse-Grained Soils for a Slab on Grade Structure, Commercial/Industrial Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

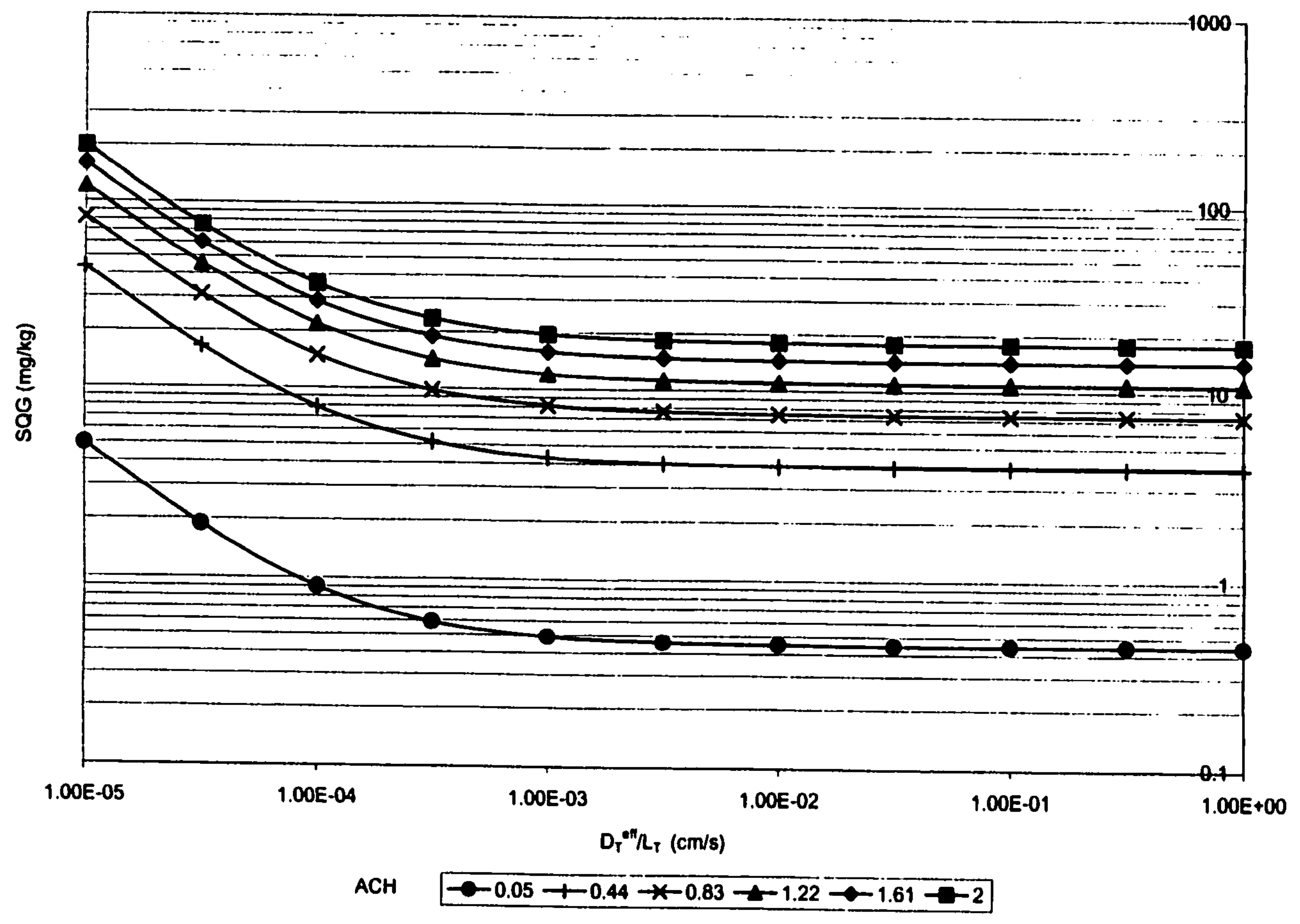

Figure H.3: SQG for Ethylbenzene in Coarse-Grained Soils for a Slab on Grade Structure, Commercial/Industrial Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

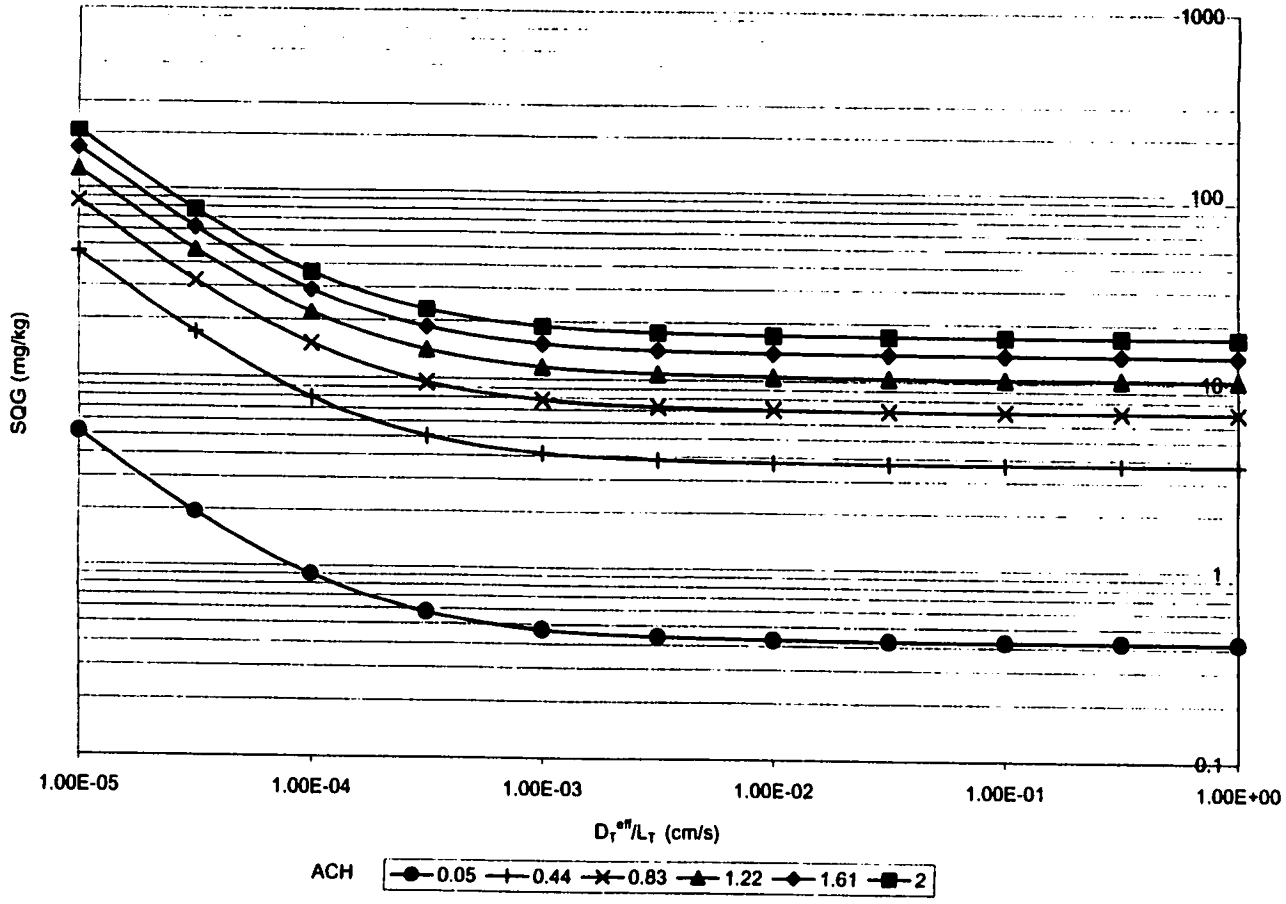

Figure H.4: SQG for o-Xylene in Coarse-Grained Soils for a Slab on Grade Structure, Commercial/Industrial Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

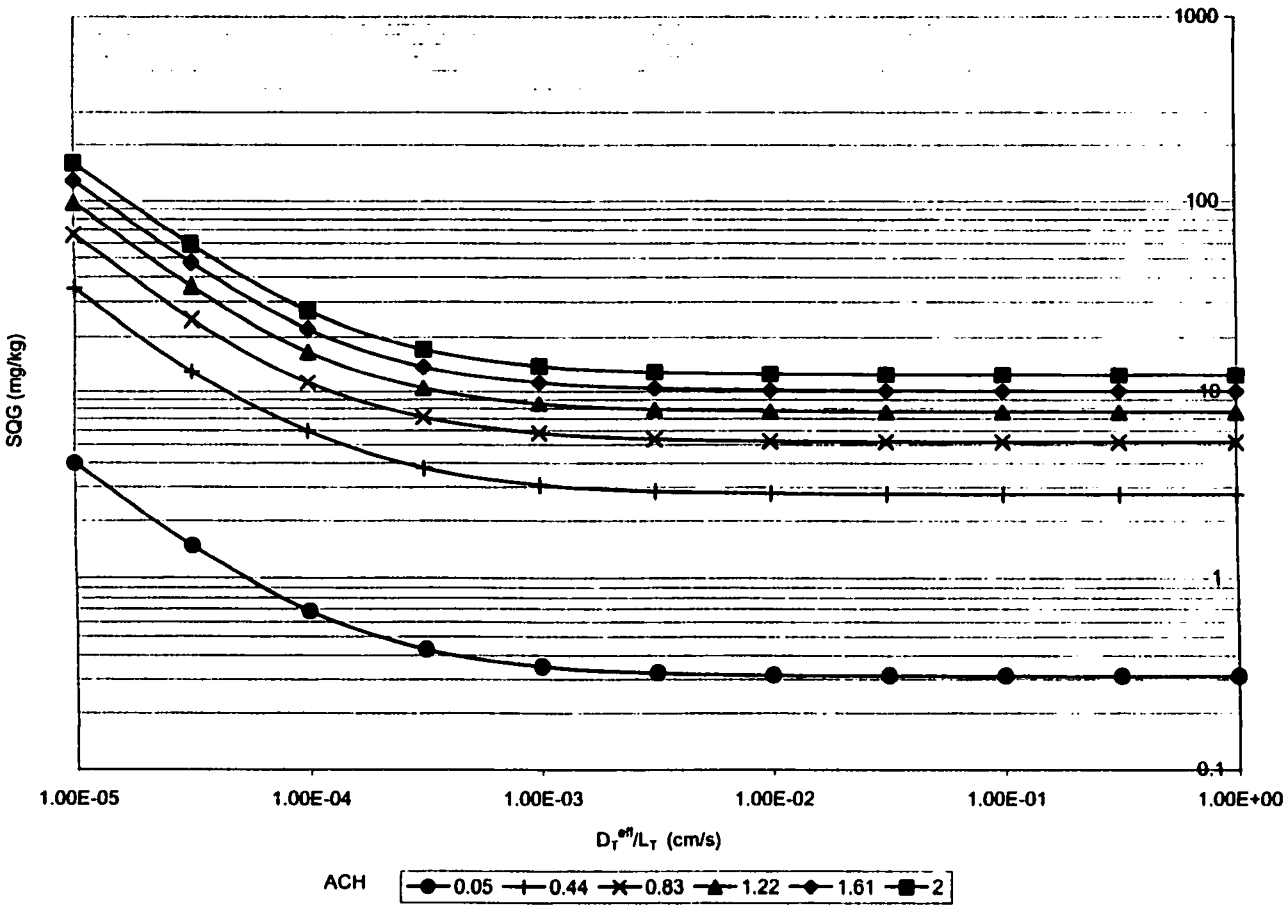

Figure H.5: SQG for m-Xylene in Coarse-Grained Soils for a Slab on Grade Structure, Commercial/Industrial Exposure Scenario vs. $\frac{D_t^{eff}}{L_T}$ for various air exchange rates

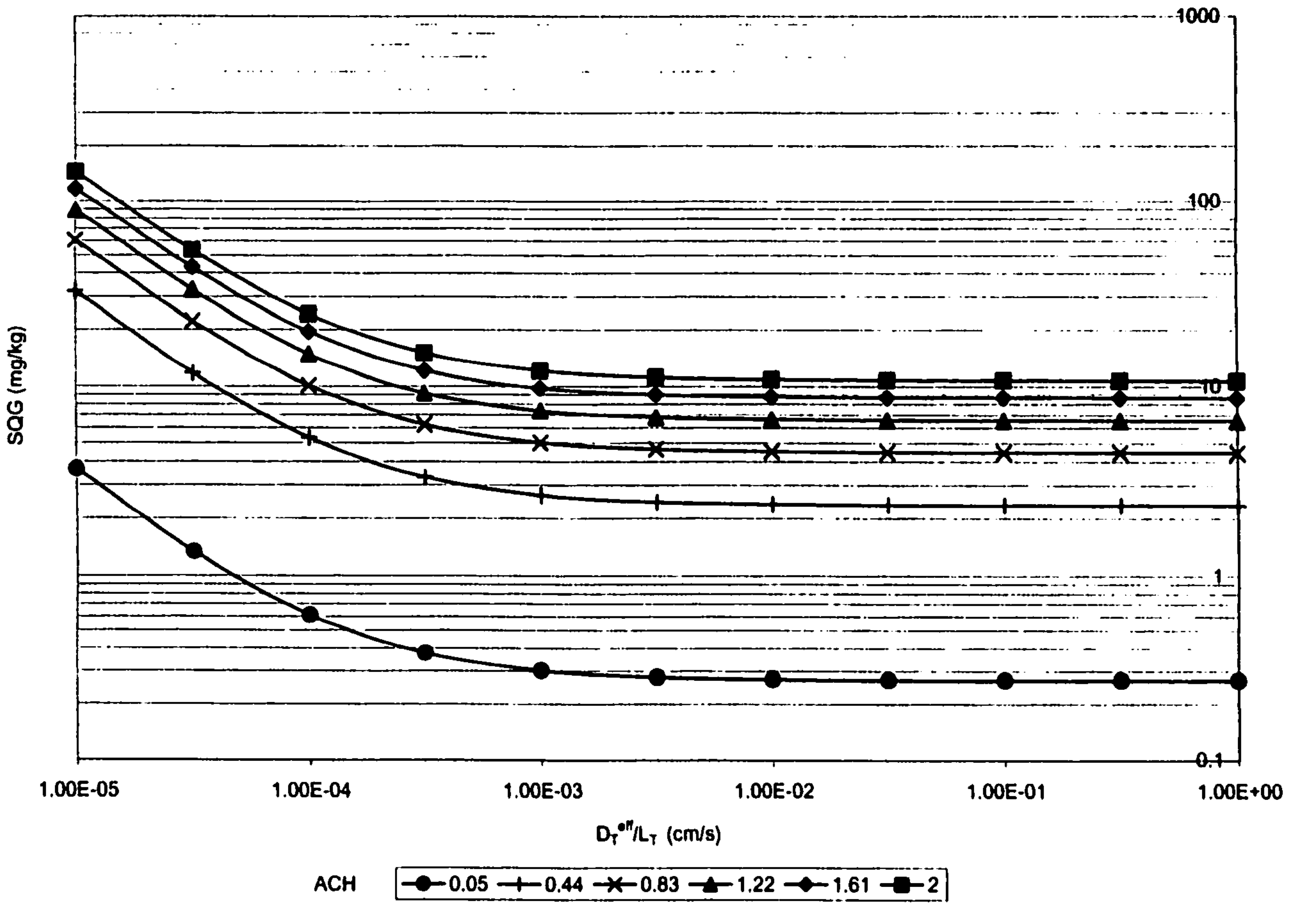

Figure H.6: SQG for p-Xylene in Coarse-Grained Soils for a Slab on Grade Structure, Commercial/Industrial Exposure Scenario vs. $\frac{D_t^{eff}}{L_T}$ for various air exchange rates

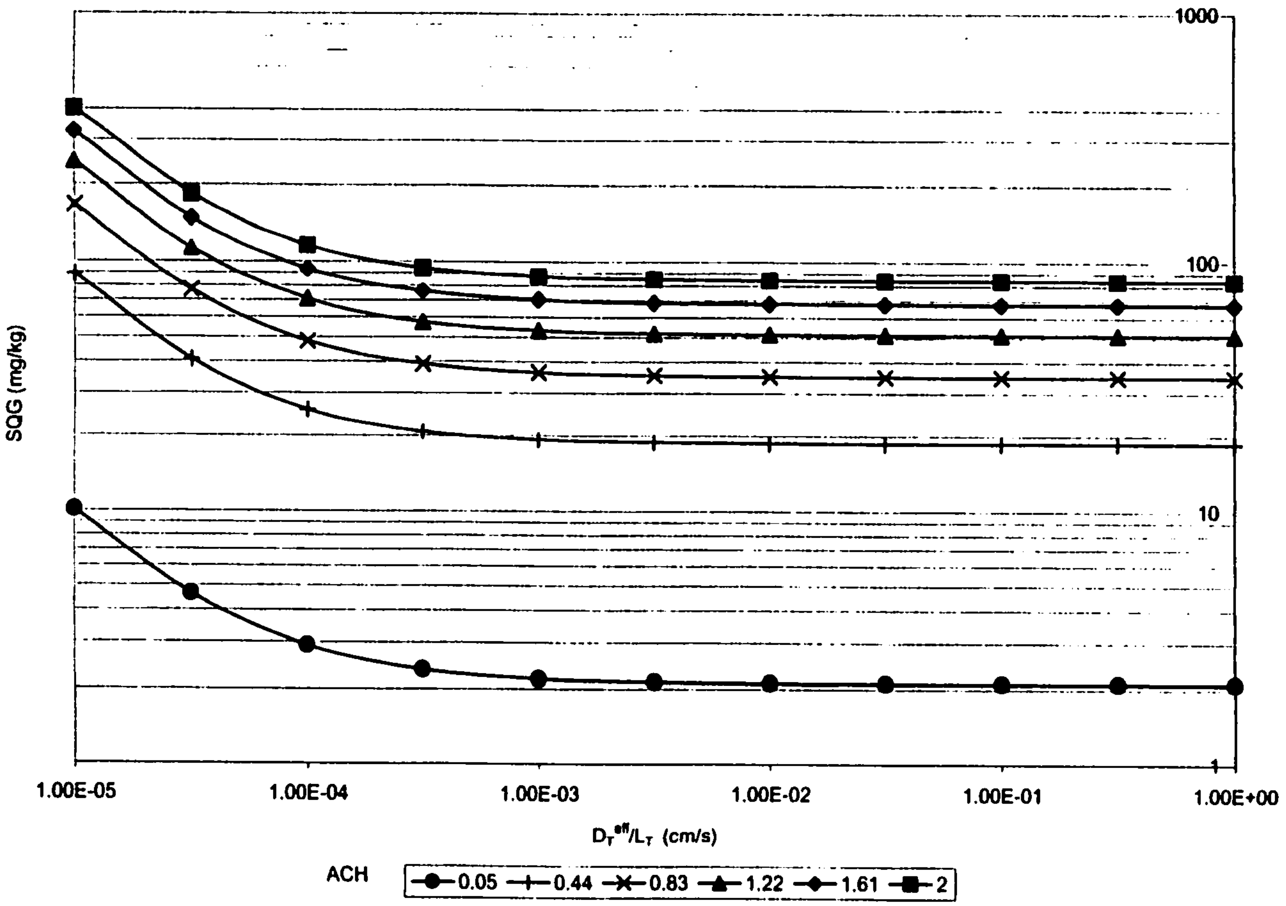

Figure H.7: SQG for C_6–C_8 aliphatic hydrocarbons in Coarse-Grained Soils for a Slab on Grade Structure, Commercial/Industrial Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

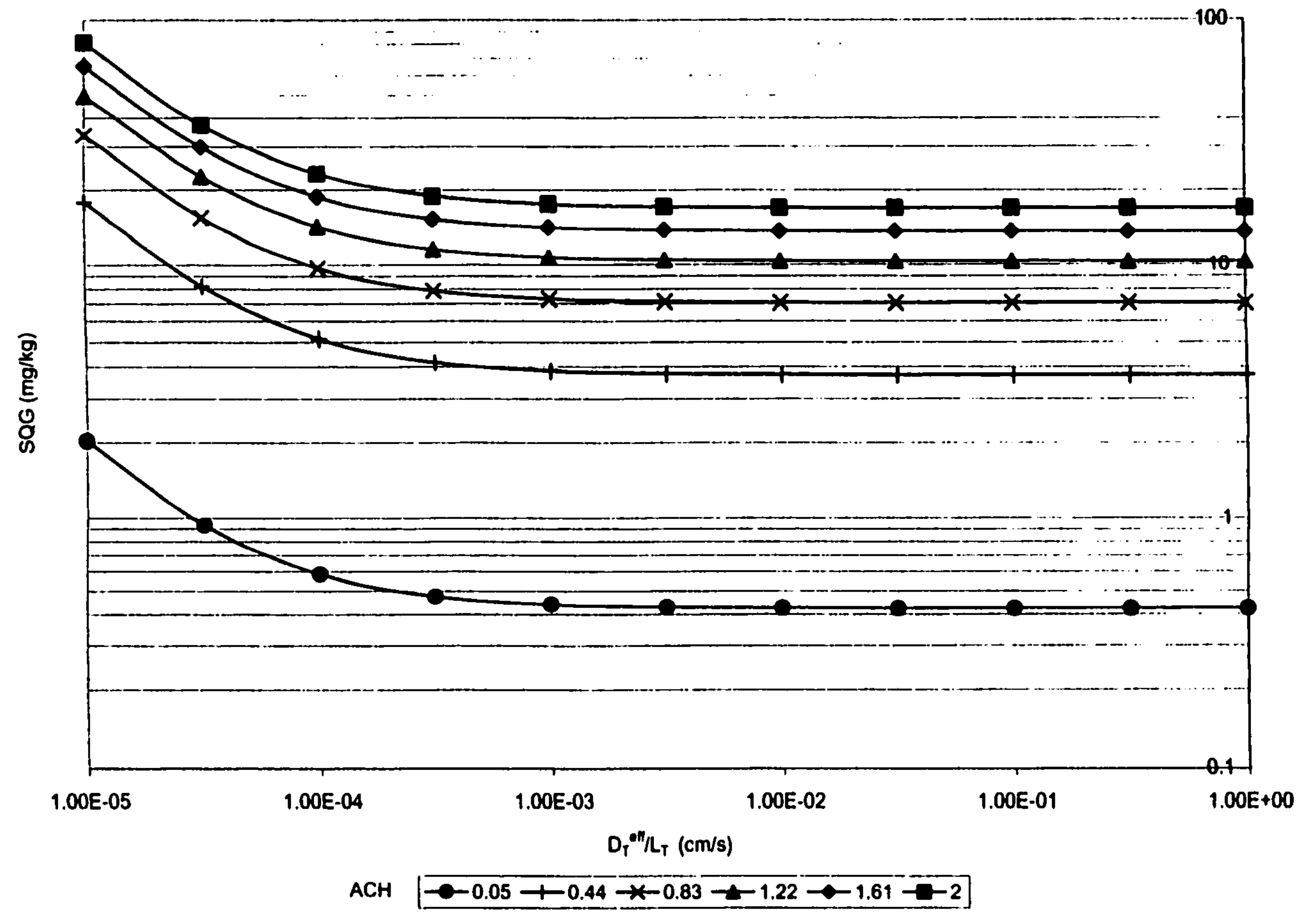

Figure H.8: SQG for $C_{>8}$–C_{10} aliphatic hydrocarbons in Coarse-Grained Soils for a Slab on Grade Structure, Commercial/Industrial Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

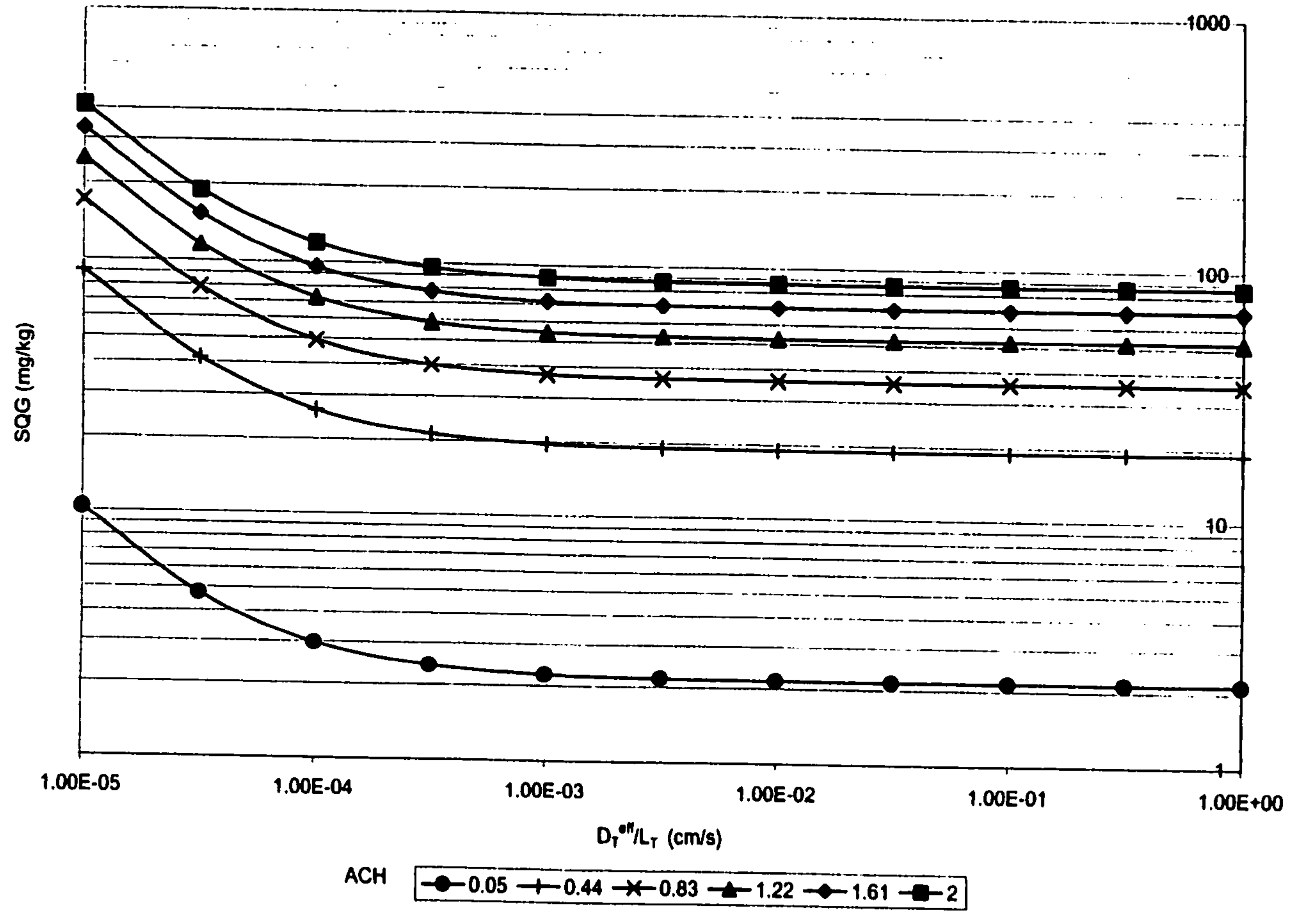

Figure H.9: SQG for $C_{>10}$–C_{12} aliphatic hydrocarbons in Coarse-Grained Soils for a Slab on Grade Structure, Commercial/Industrial Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

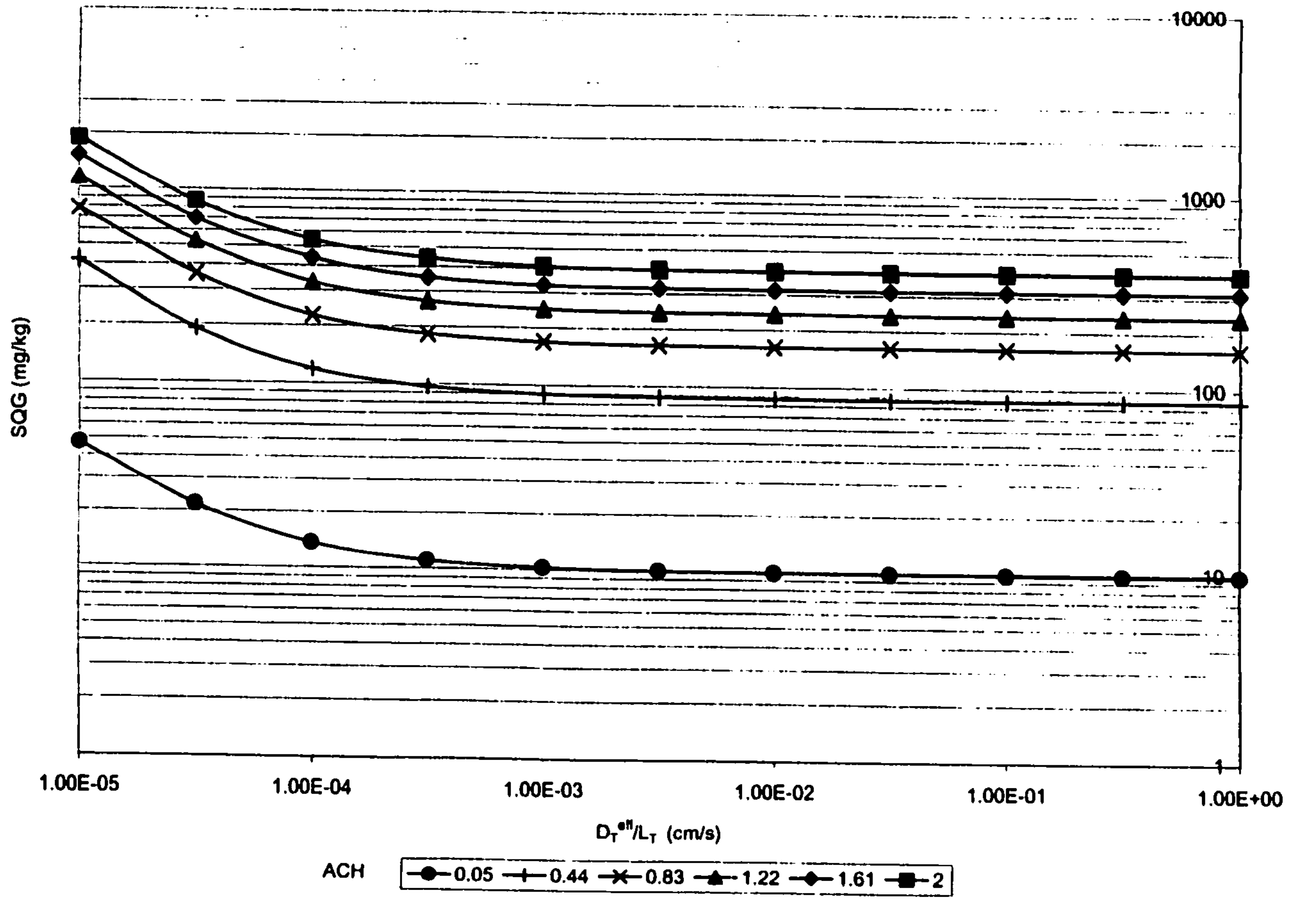

Figure H.10: SQG for $C_{>12}$–C_{16} aliphatic hydrocarbons in Coarse-Grained Soils for a Slab on Grade Structure, Commercial/Industrial Exposure Scenario vs. $\frac{D_t^{eff}}{L_T}$ for various air exchange rates

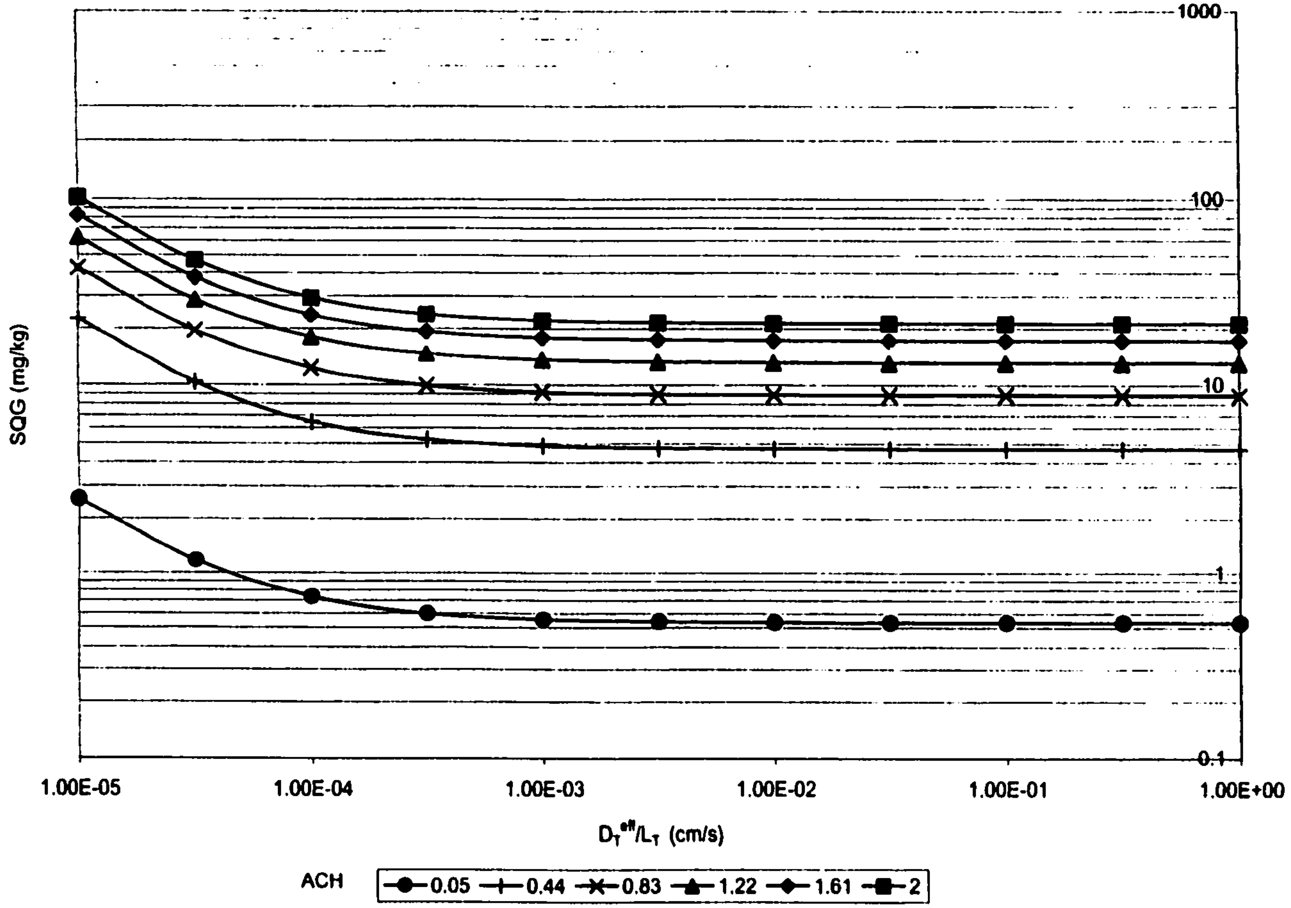

Figure H.11: SQG for $C_{>6}$–C_{10} aromatic hydrocarbons in Coarse-Grained Soils for a Slab on Grade Structure, Commercial/Industrial Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

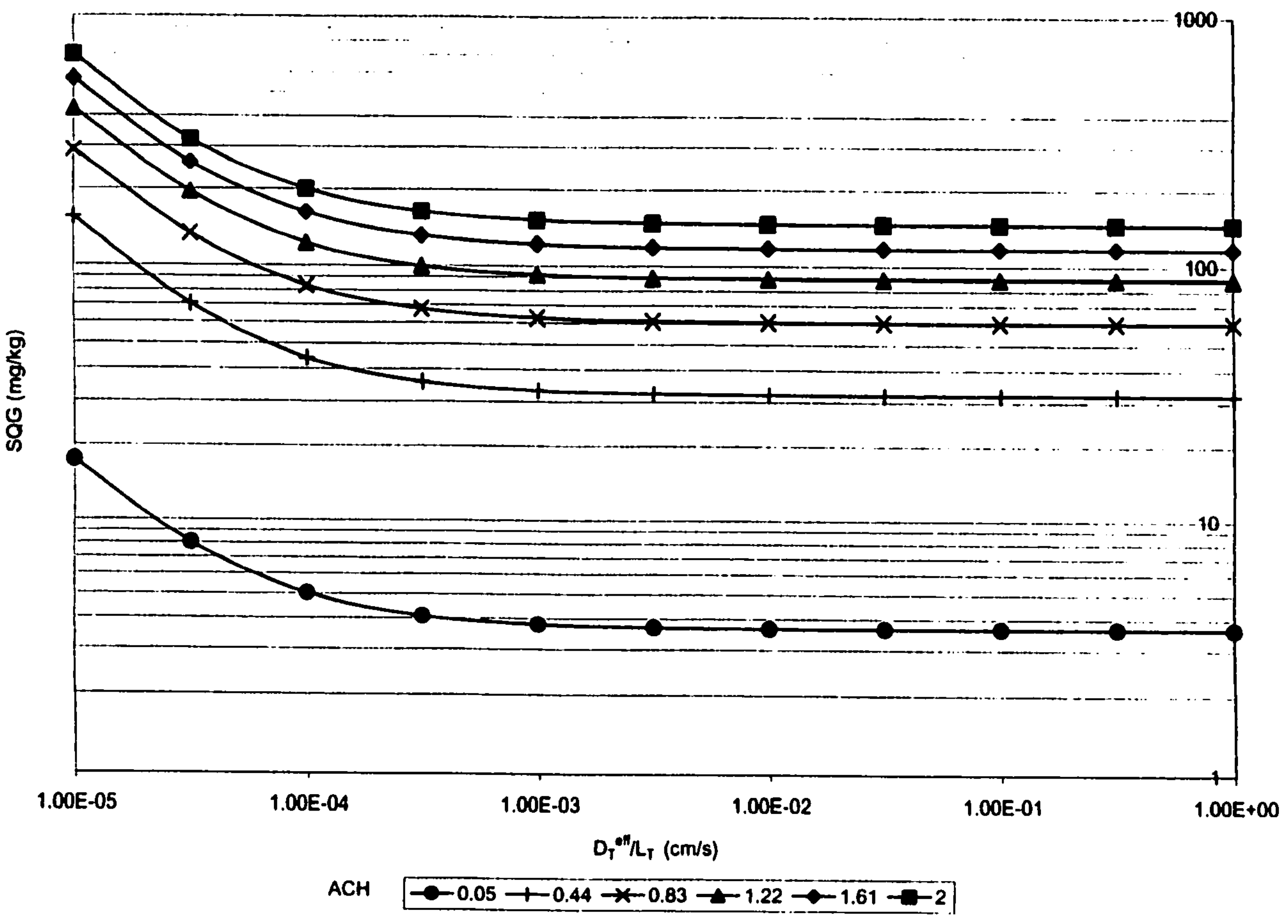

Figure H.12: SQG for $C_{>10}$-C_{12} aromatic hydrocarbons i in Coarse-Grained Soils for a Slab on Grade Structure, Commercial/Industrial Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

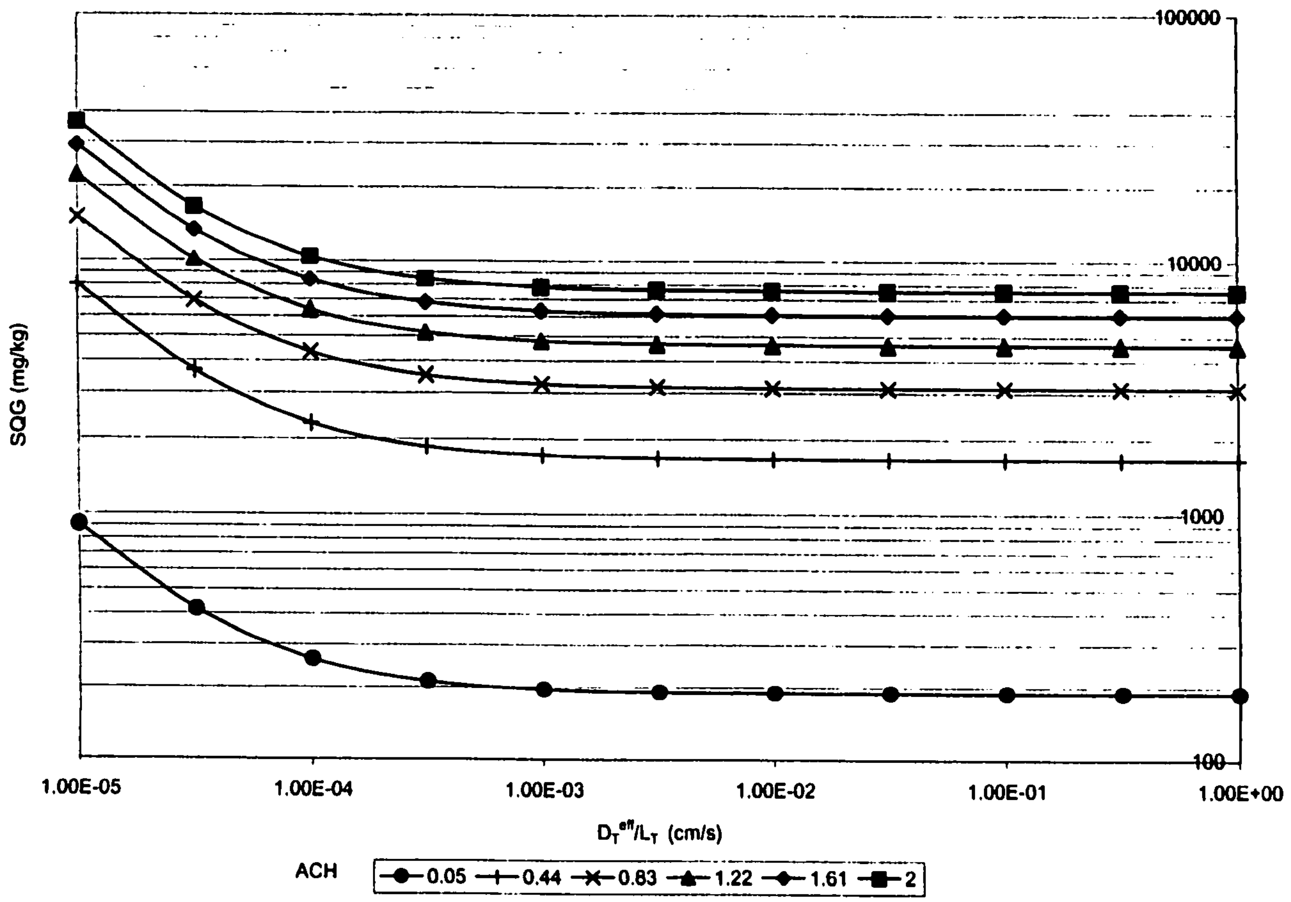

Figure H.13: SQG for $C_{>12}$–C_{16} aromatic hydrocarbons i in Coarse-Grained Soils for a Slab on Grade Structure, Commercial/Industrial Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

Appendix I

Groundwater Quality Guidelines for Coarse-Grained Soils for Buildings with Basements, Agricultural/Residential Exposure Scenarios

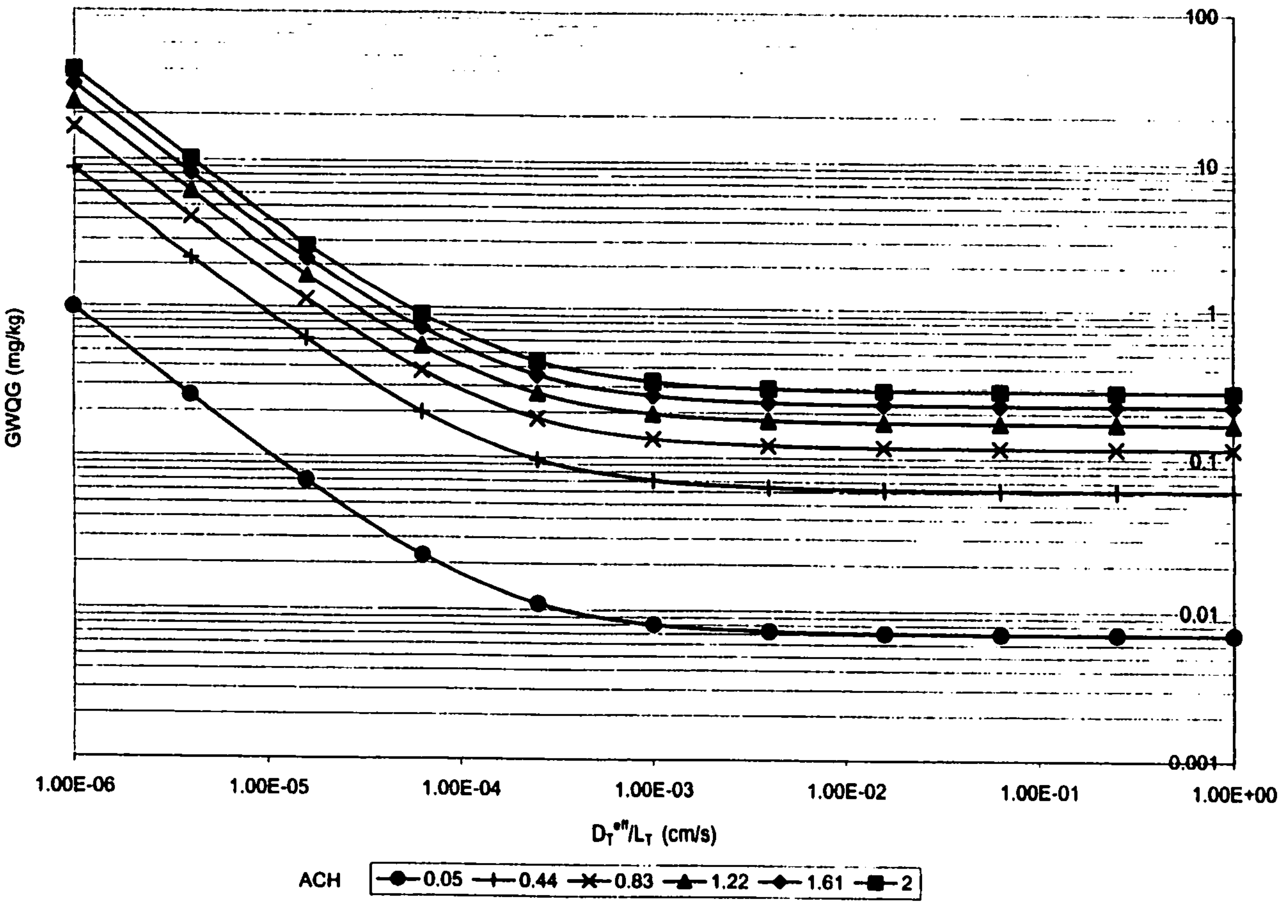

Figure I.1: GWQG for Benzene in Coarse-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

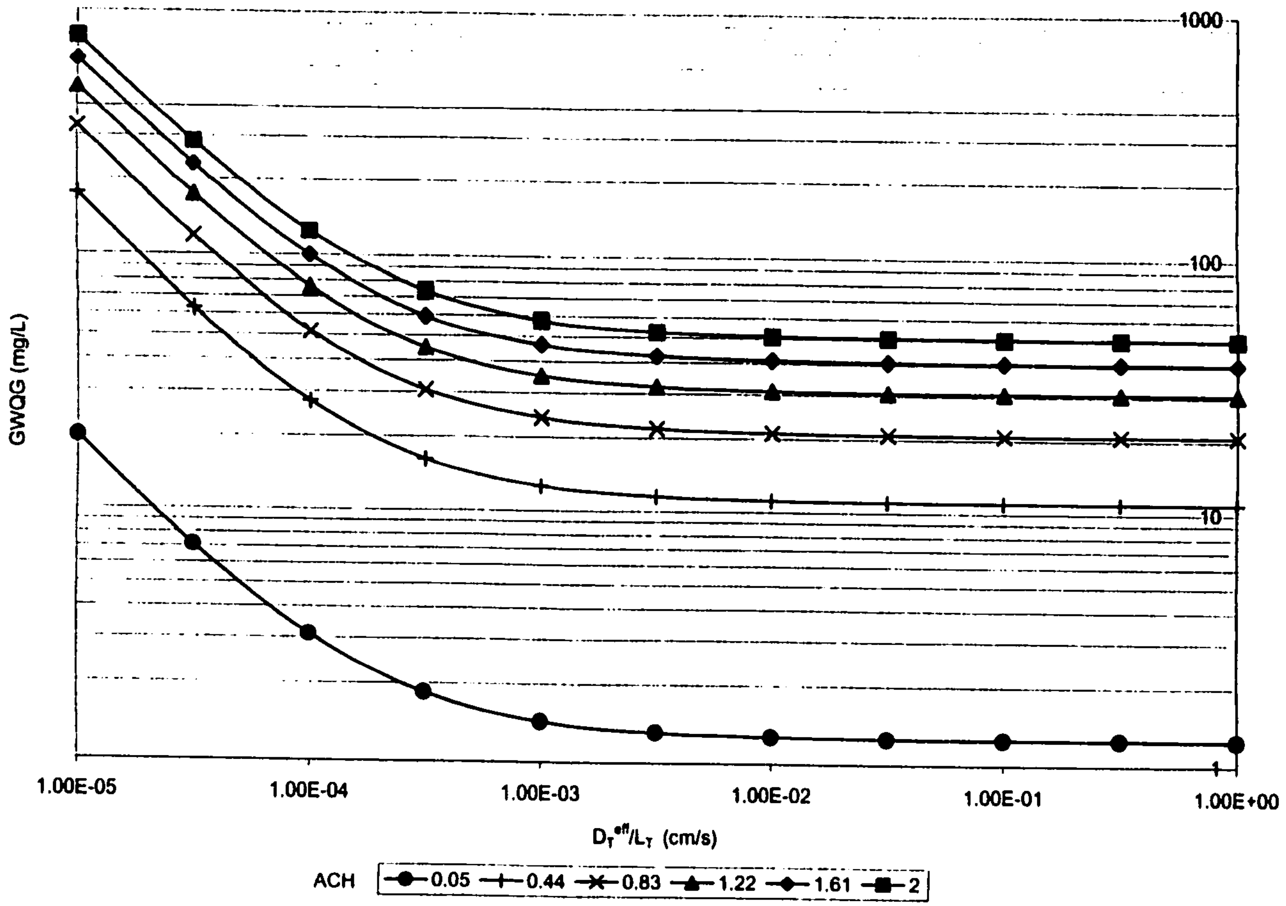

Figure I.2: GWQG for Toluene in Coarse-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

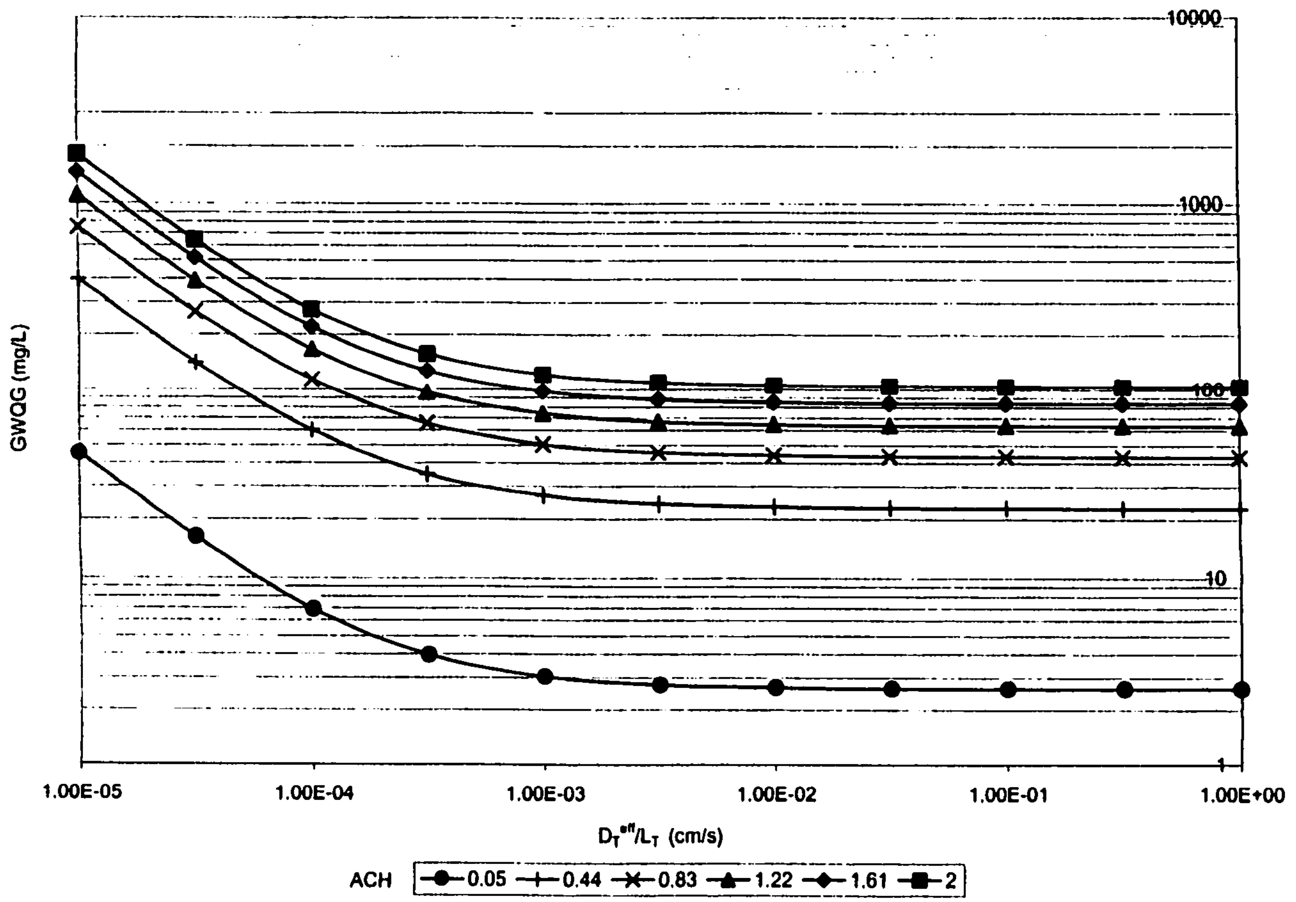

Figure I.3: GWQG for Ethylbenzene iin Coarse-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_t^{eff}}{L_T}$ for various air exchange rates

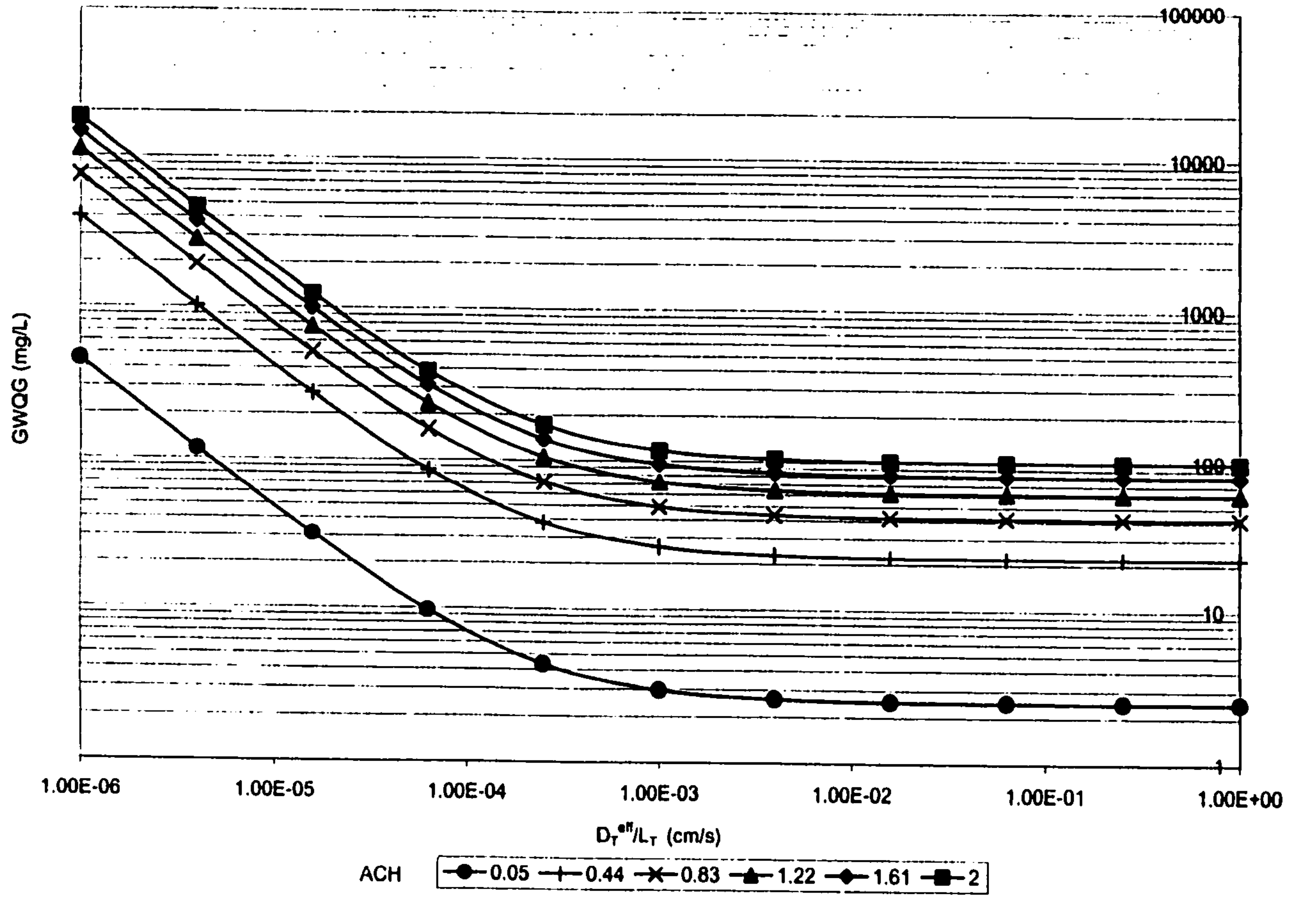

Figure I.4: GWQG for o-Xylene in Coarse-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

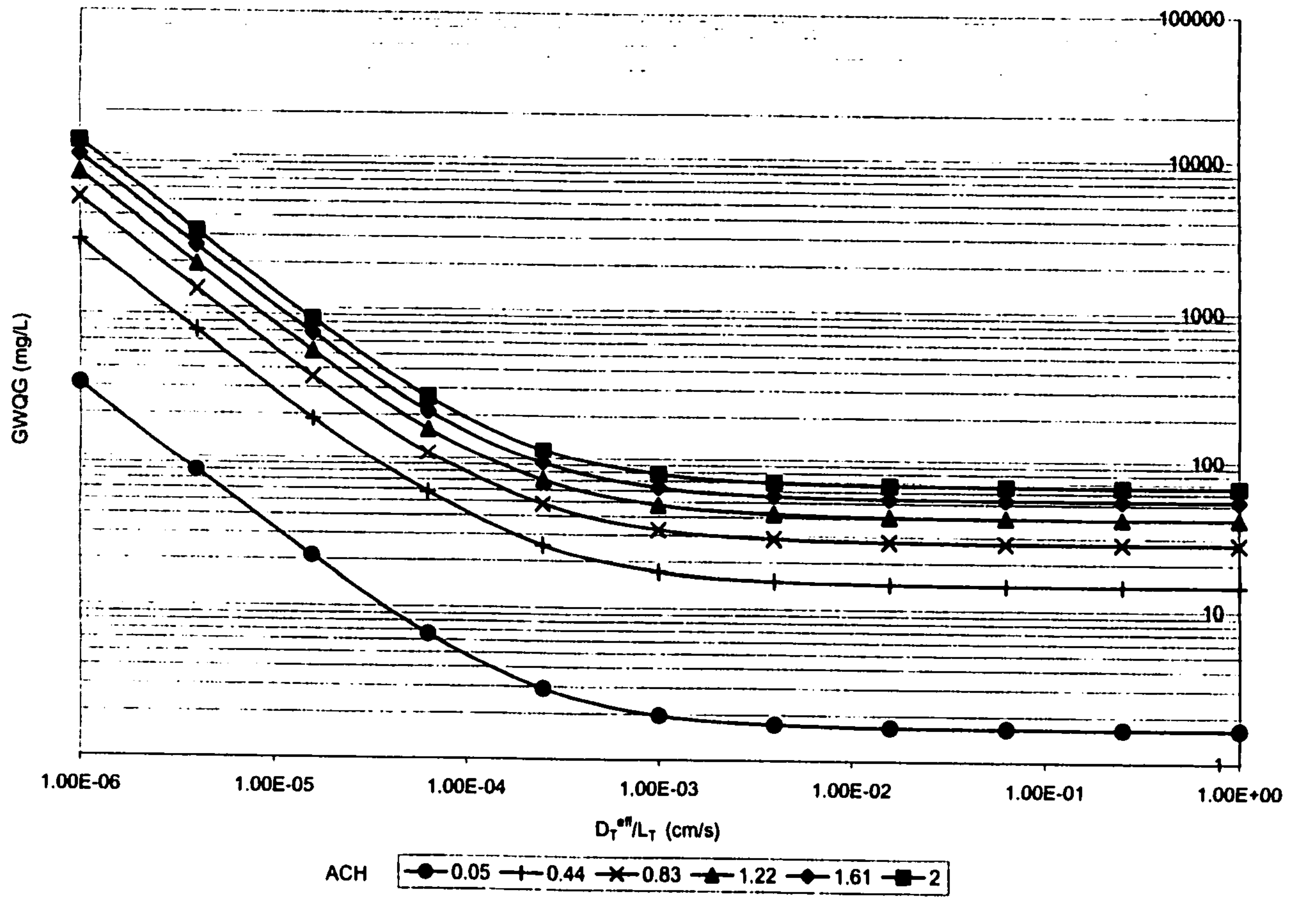

Figure I.5: GWQG for m-Xylene in Coarse-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_t^{eff}}{L_T}$ for various air exchange rates

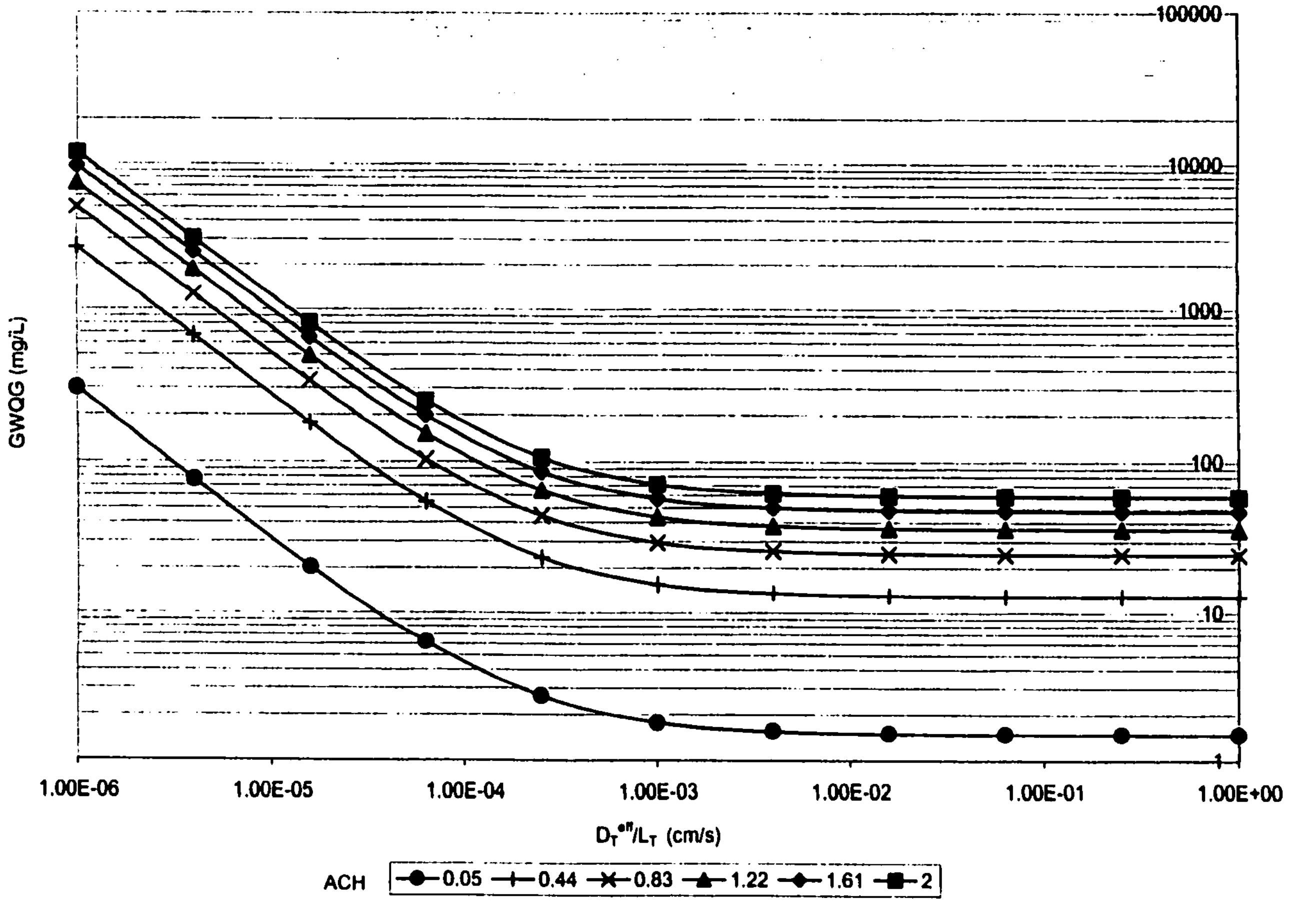

Figure I.6: GWQG for p-Xylene in Coarse-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

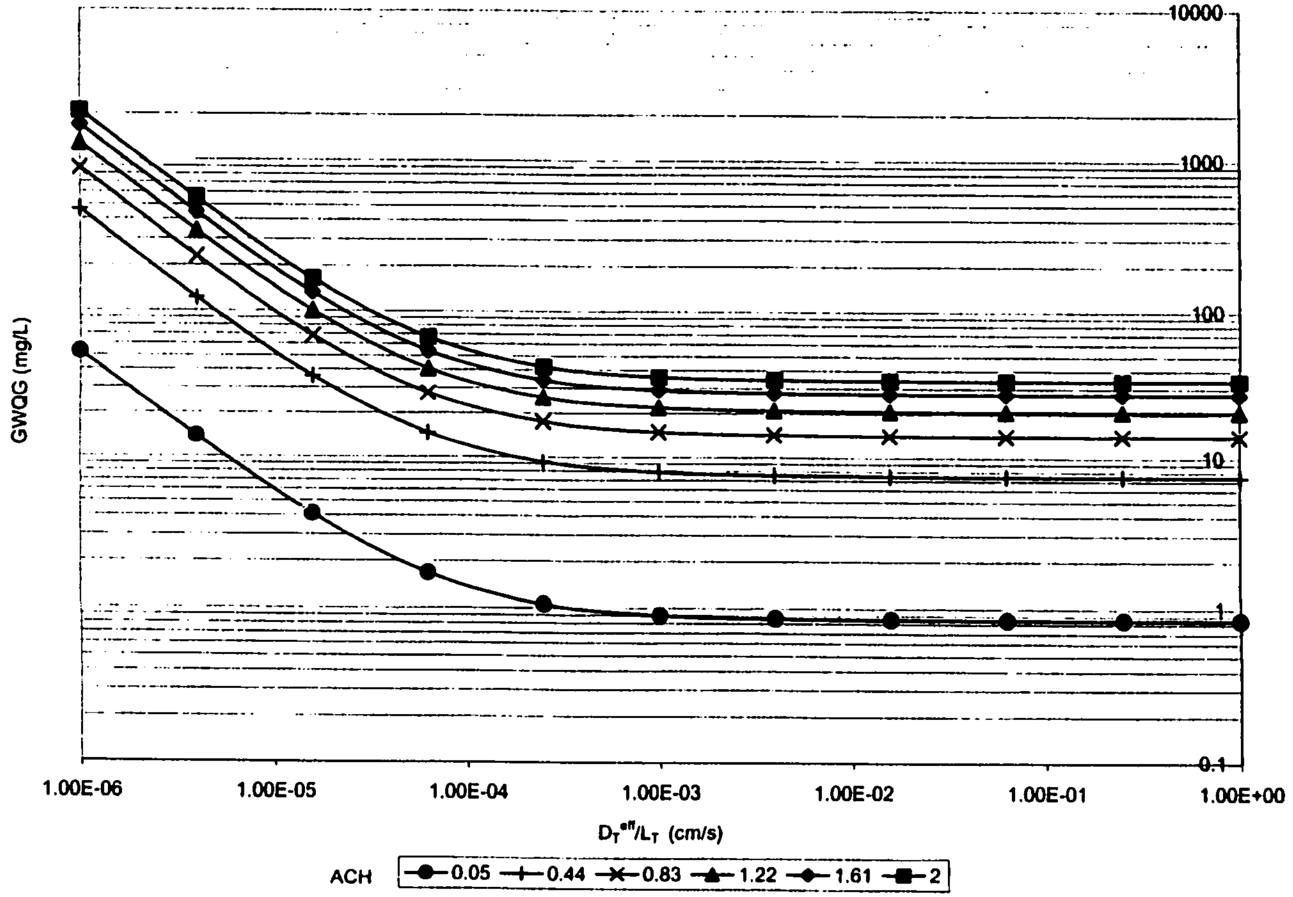

Figure I.7: GWQG for C_6–C_8 aliphatic hydrocarbons in Coarse-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_t^{eff}}{L_T}$ for various air exchange rates

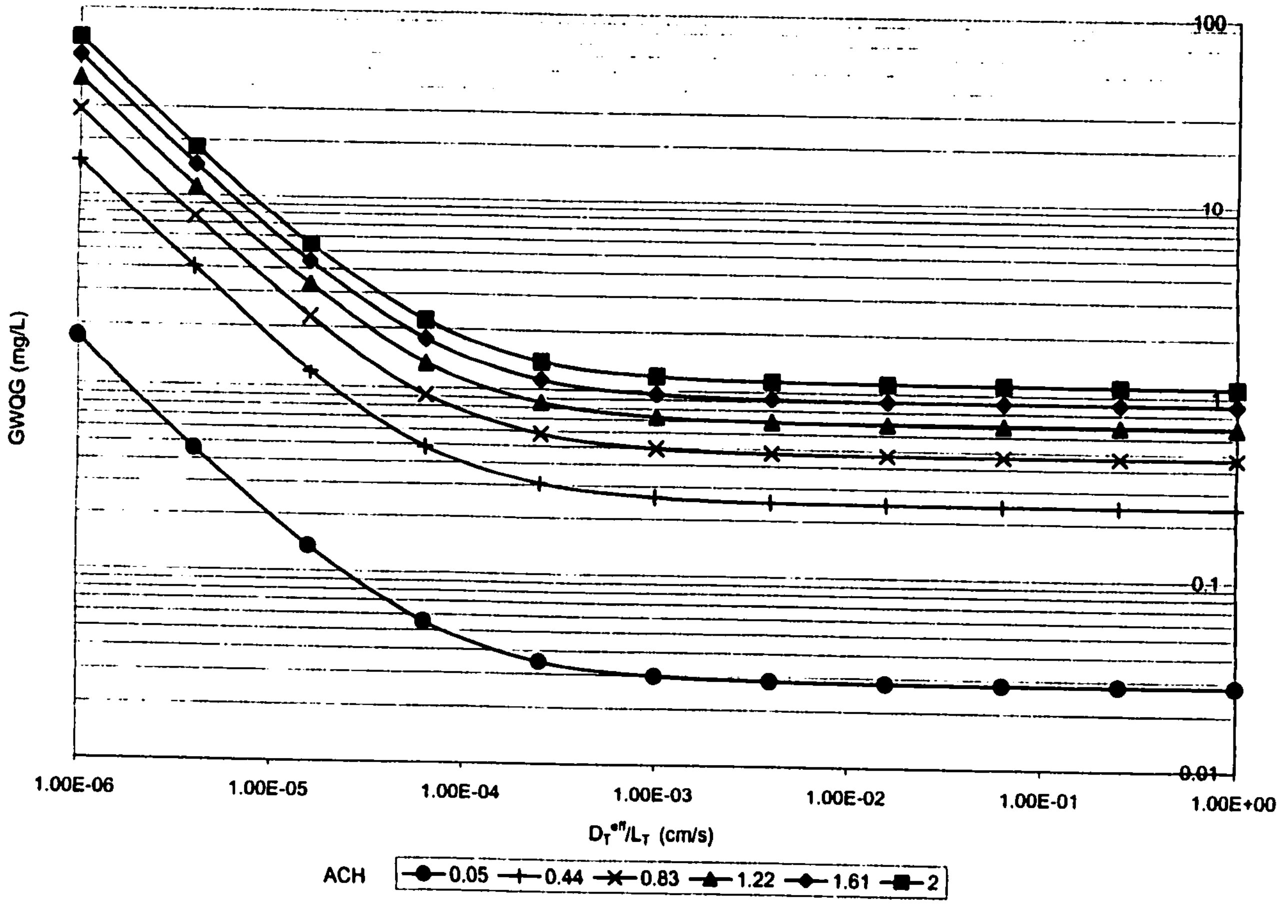

Figure I.8: GWQG for $C_{>8}$-C_{10} aliphatic hydrocarbons in Coarse-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

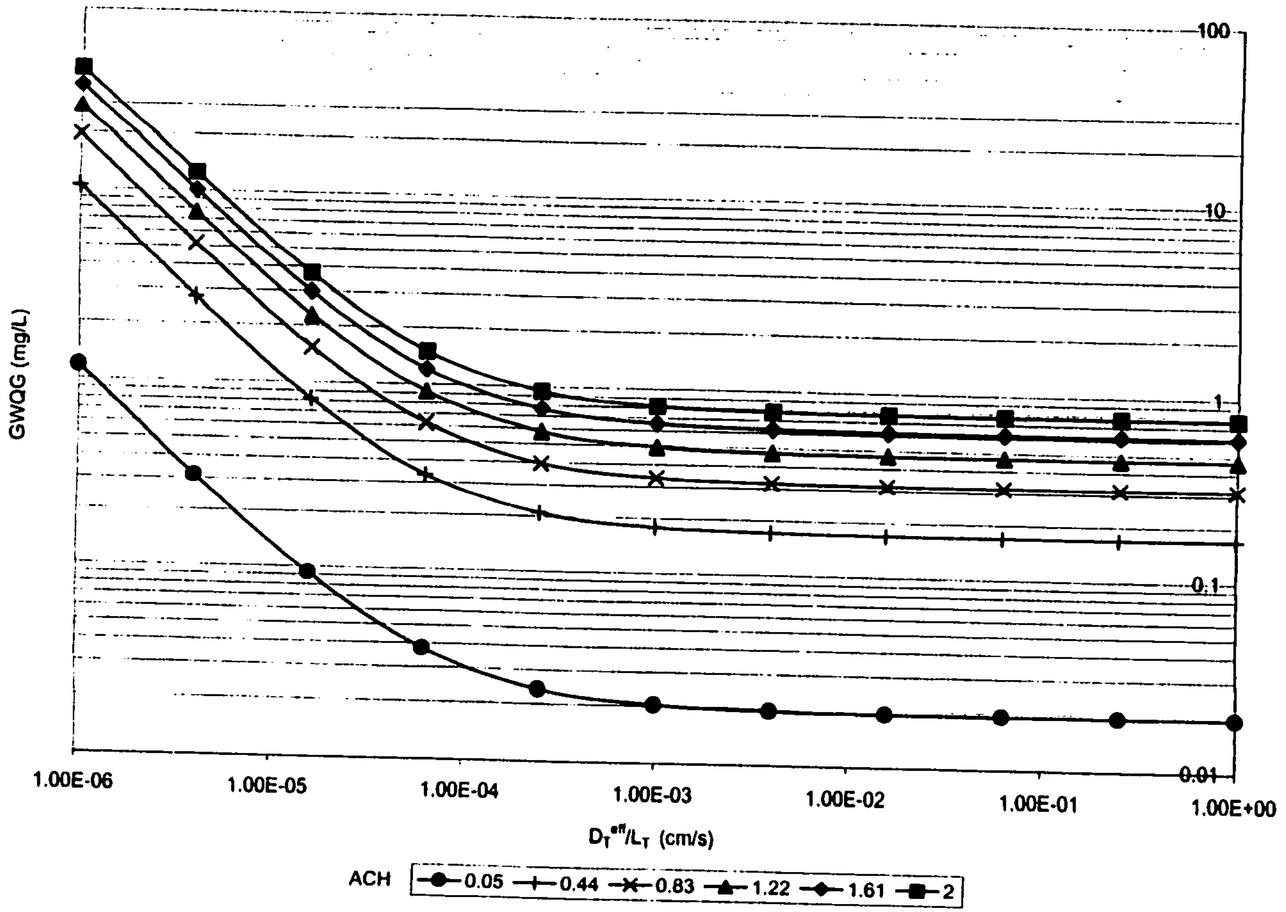

Figure I.9: GWQG for $C_{>10}$-C_{12} aliphatic hydrocarbons in Coarse-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

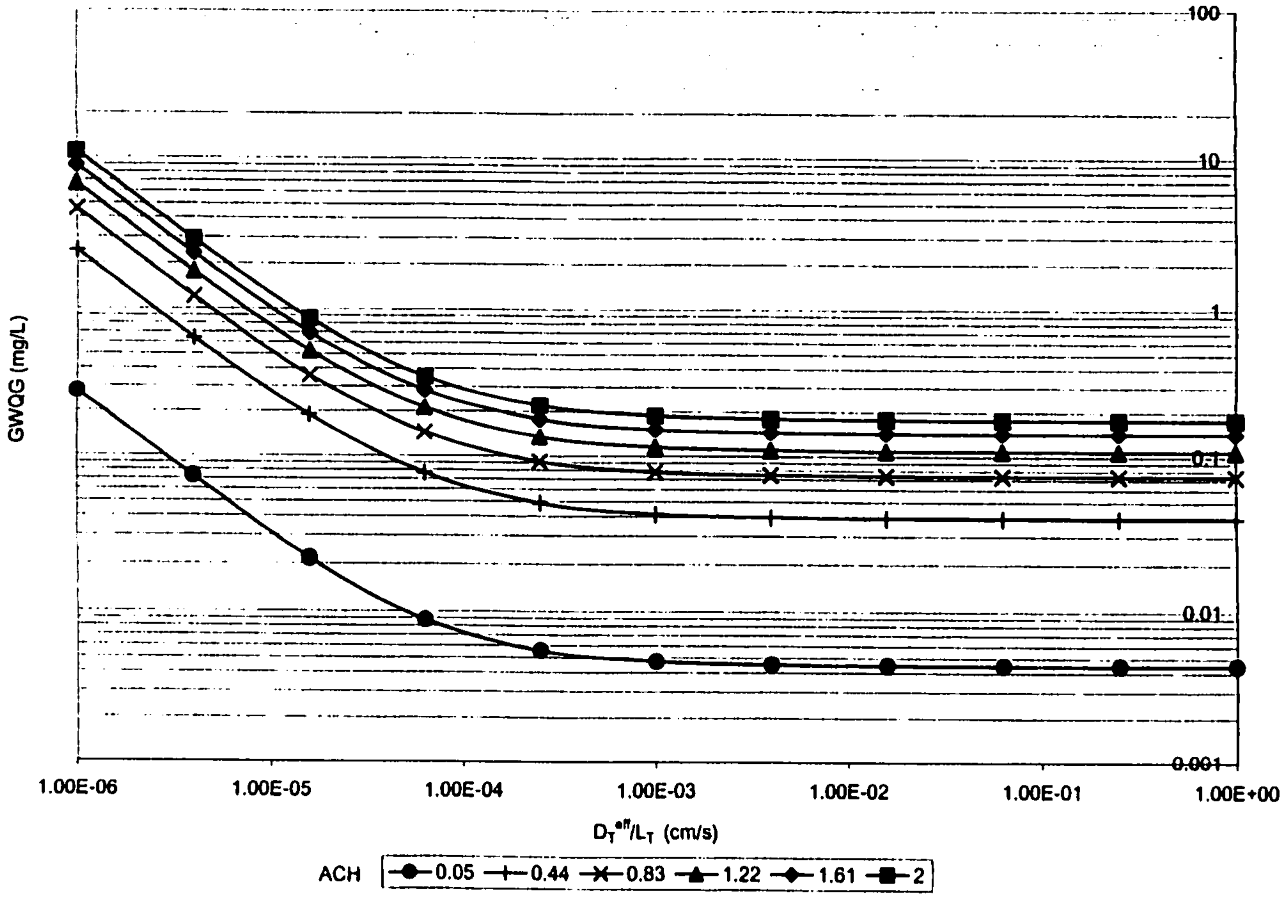

Figure I.10: GWQG for $C_{>12}$-C_{16} aliphatic hydrocarbons in Coarse-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_t^{eff}}{L_T}$ for various air exchange rates

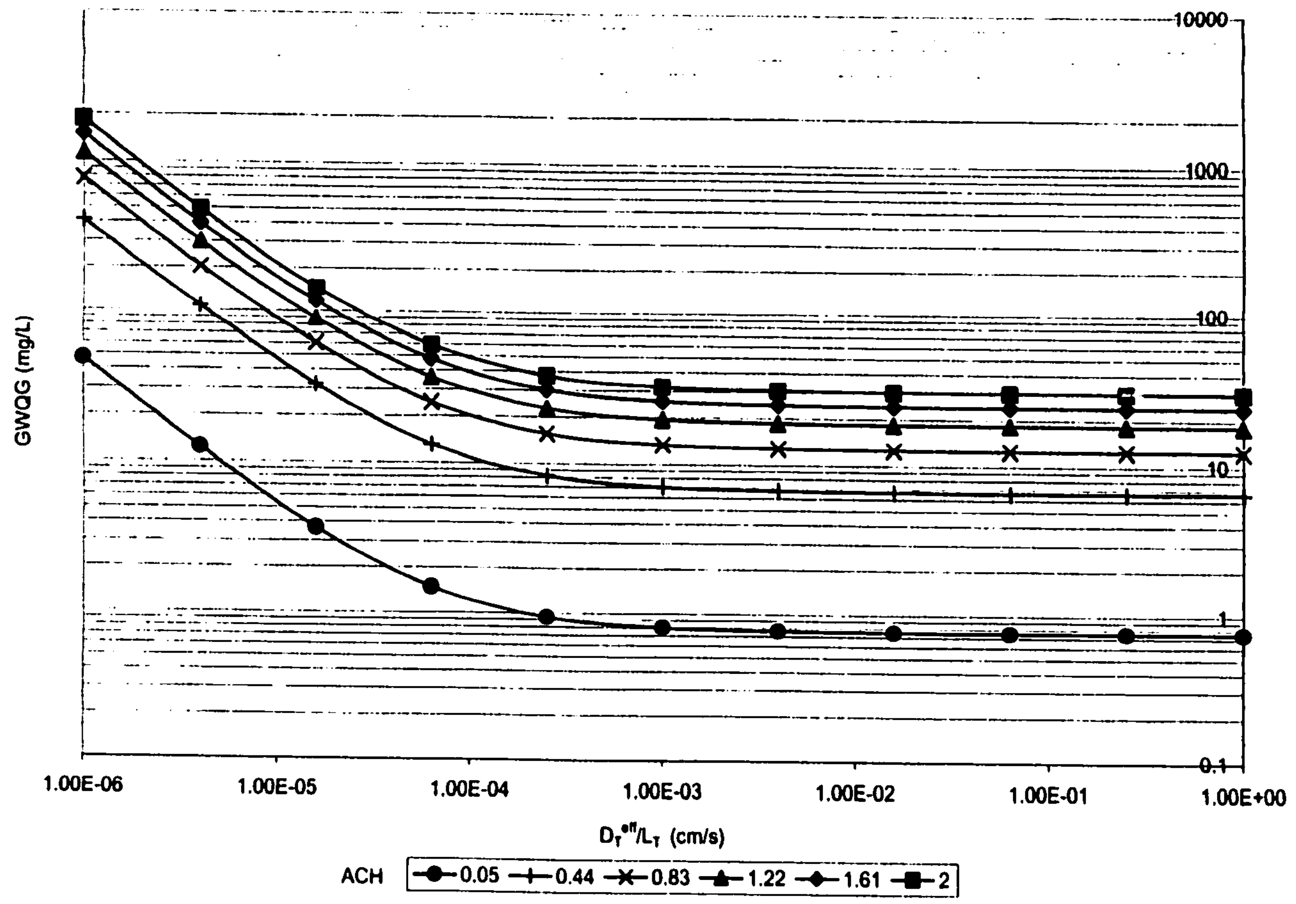

Figure I.11: GWQG for $C_{>8}$–C_{10} aromatic hydrocarbons in Coarse-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

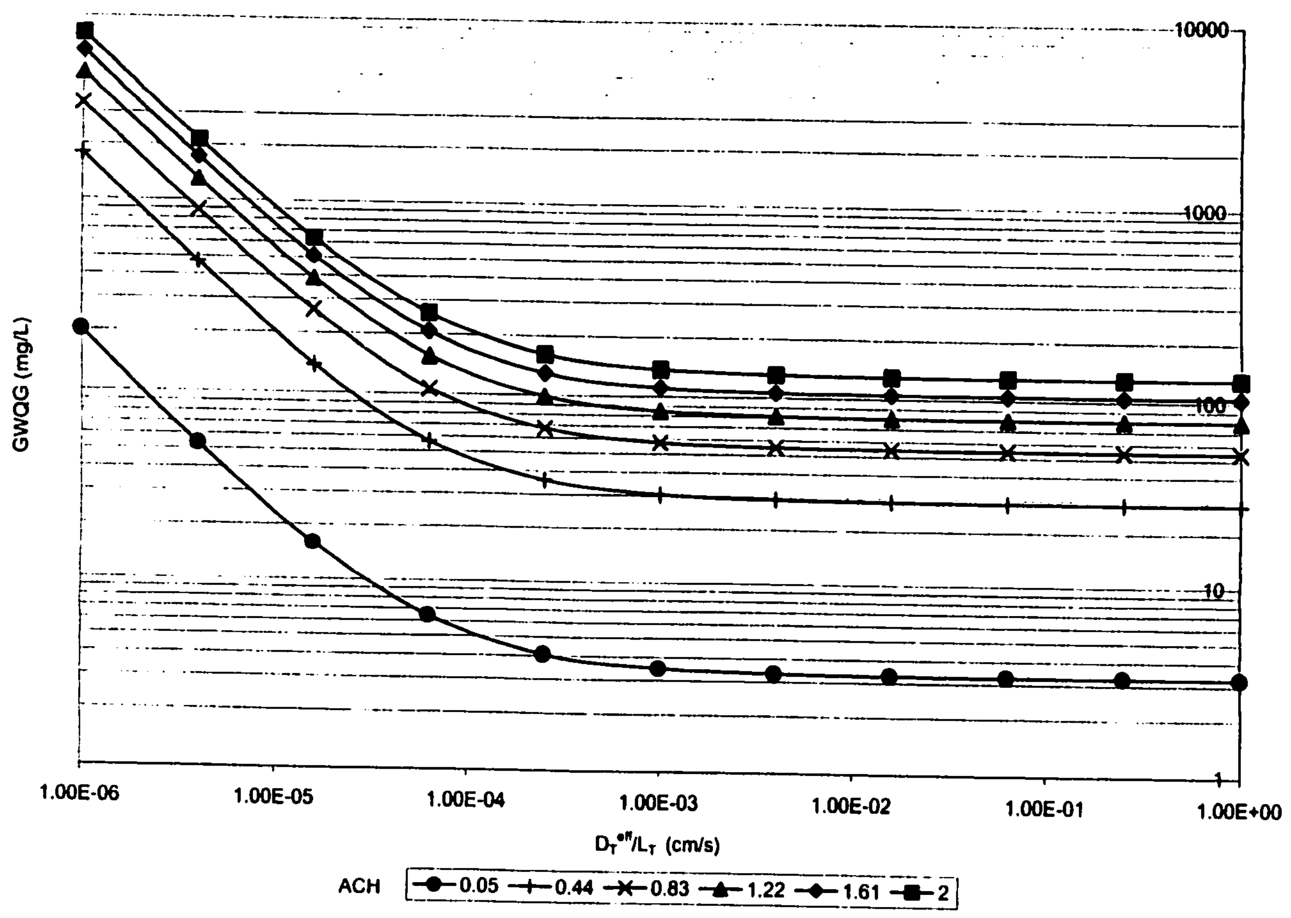

Figure I.12: GWQG for $C_{>10}$-C_{12} aromatic hydrocarbons in Coarse-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

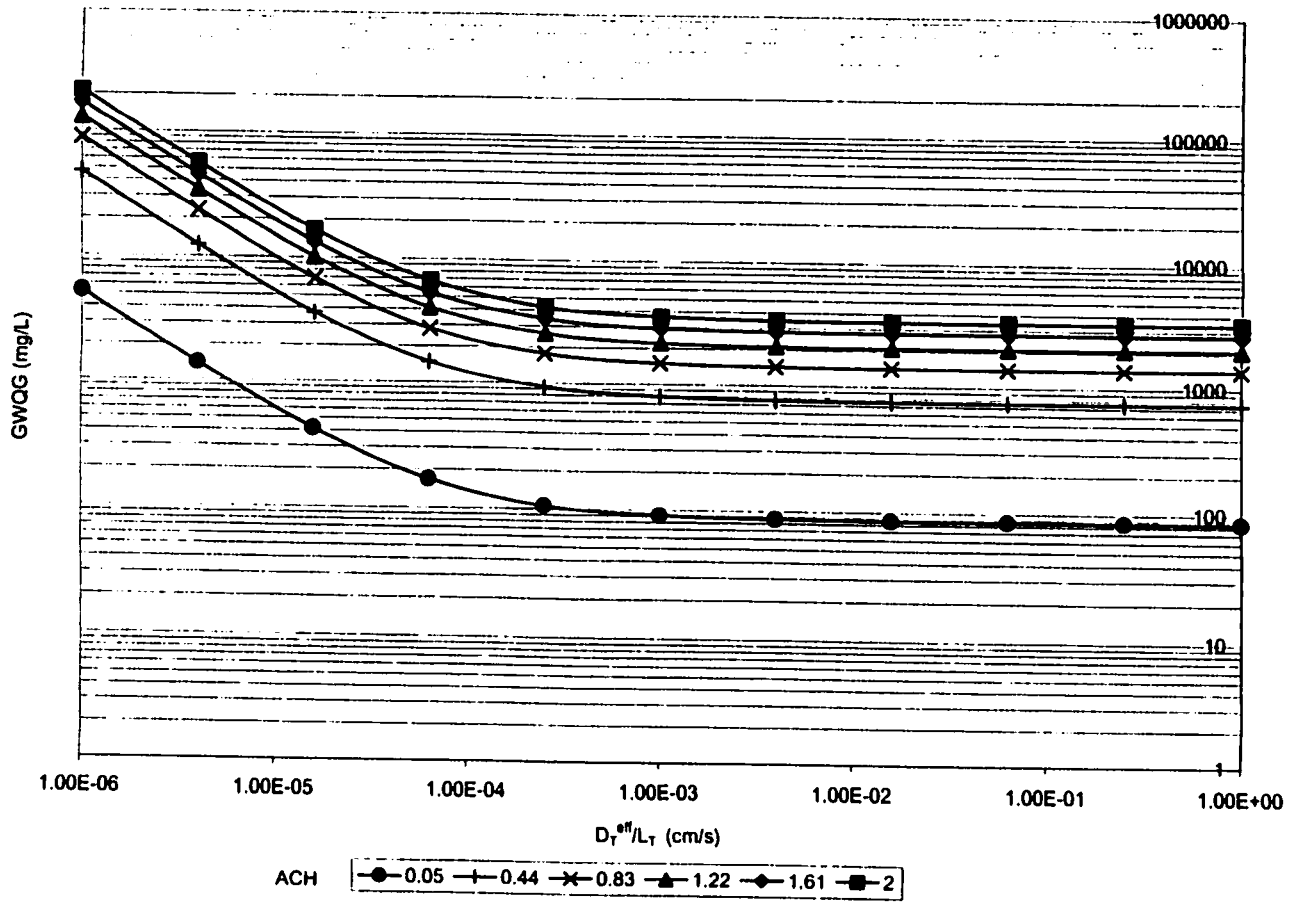

Figure I.13: GWQG for $C_{>12}$–C_{16} aromatic hydrocarbons in Coarse-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

Appendix J

Groundwater Quality Guidelines for Coarse-Grained Soils for Slab on Grade Structures, Commercial/Industrial Exposure Scenarios

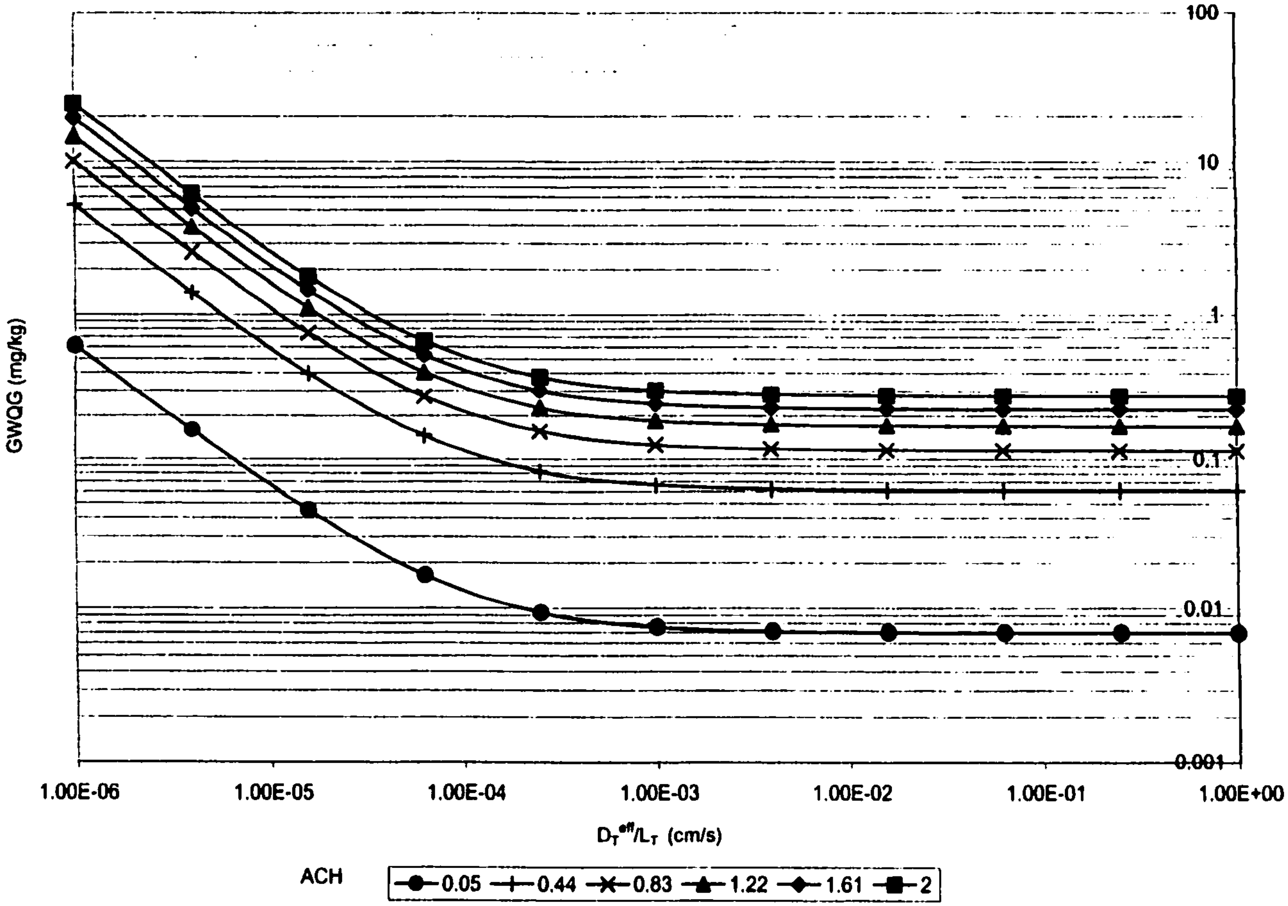

Figure J.1: GWQG for Benzene in Coarse-Grained Soils for a Slab on Grade Structures, Commercial/Industrial Exposure Scenario vs. $\frac{D_t^{eff}}{L_T}$ for various air exchange rates

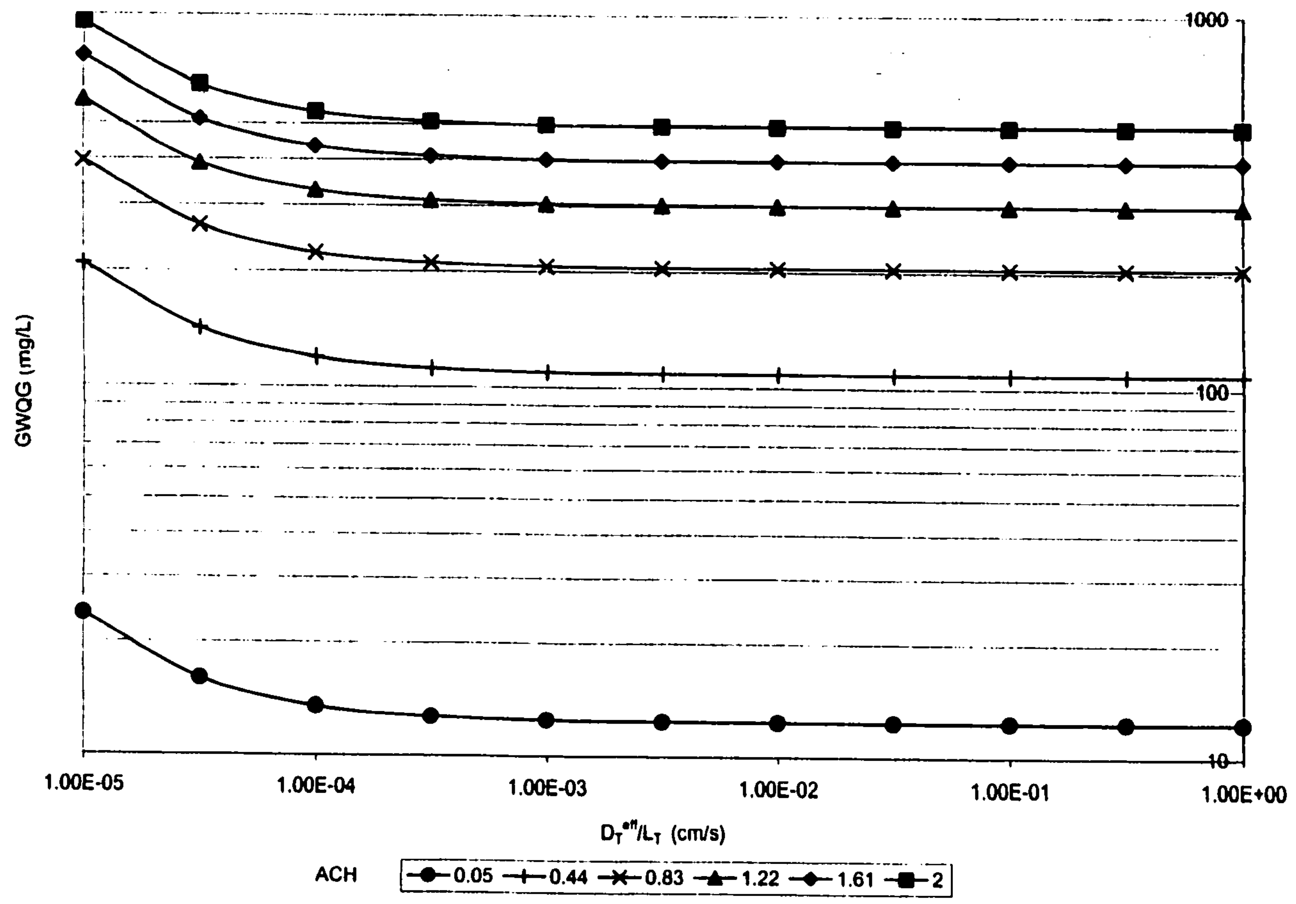

Figure J.2: GWQG for Toluene in Coarse-Grained Soils for Slab on Grade Structures, Commercial/Industrial Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

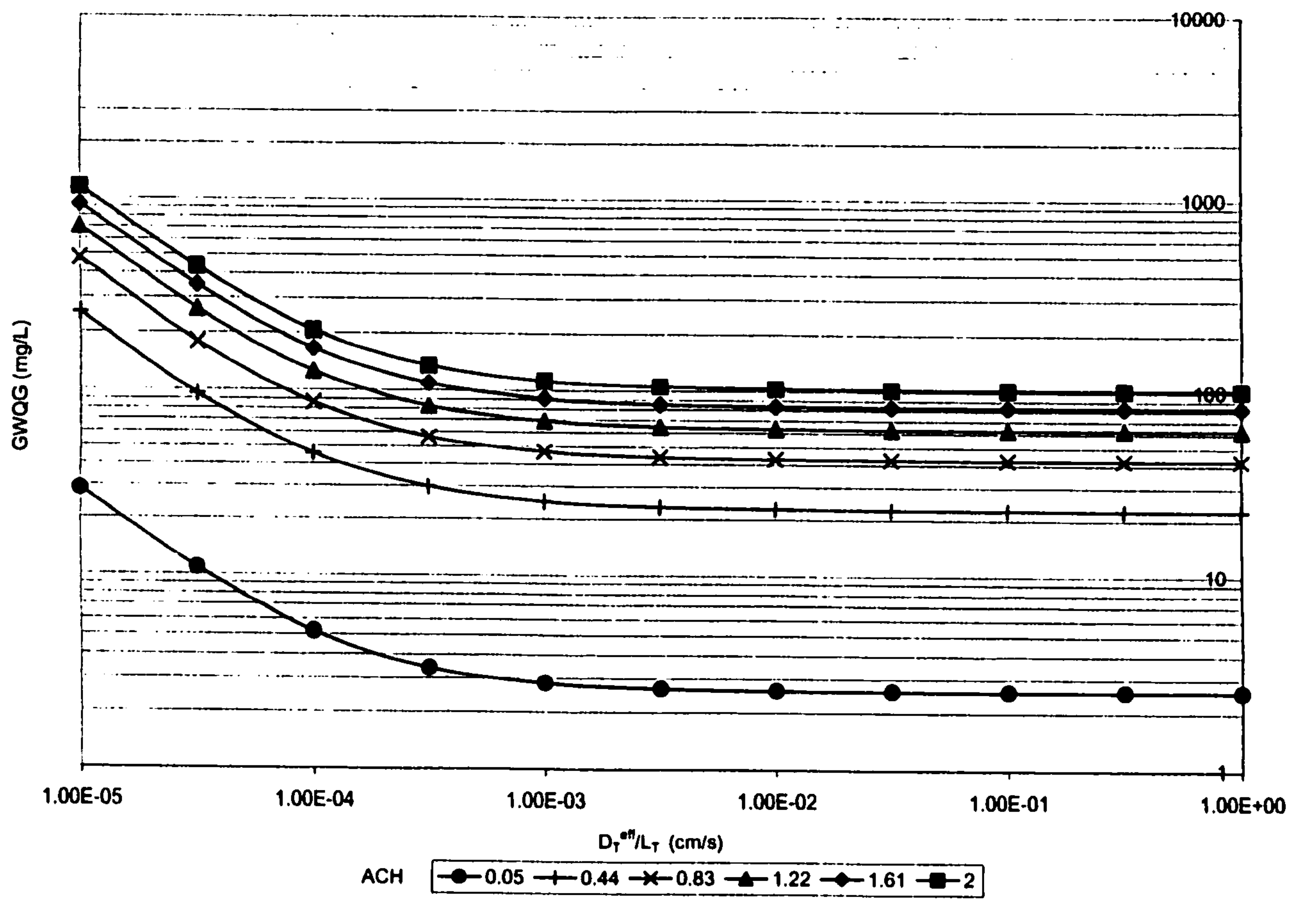

Figure J.3: GWQG for Ethylbenzene in Coarse-Grained Soils for Slab on Grade Structures, Commercial/Industrial Exposure Scenariovs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

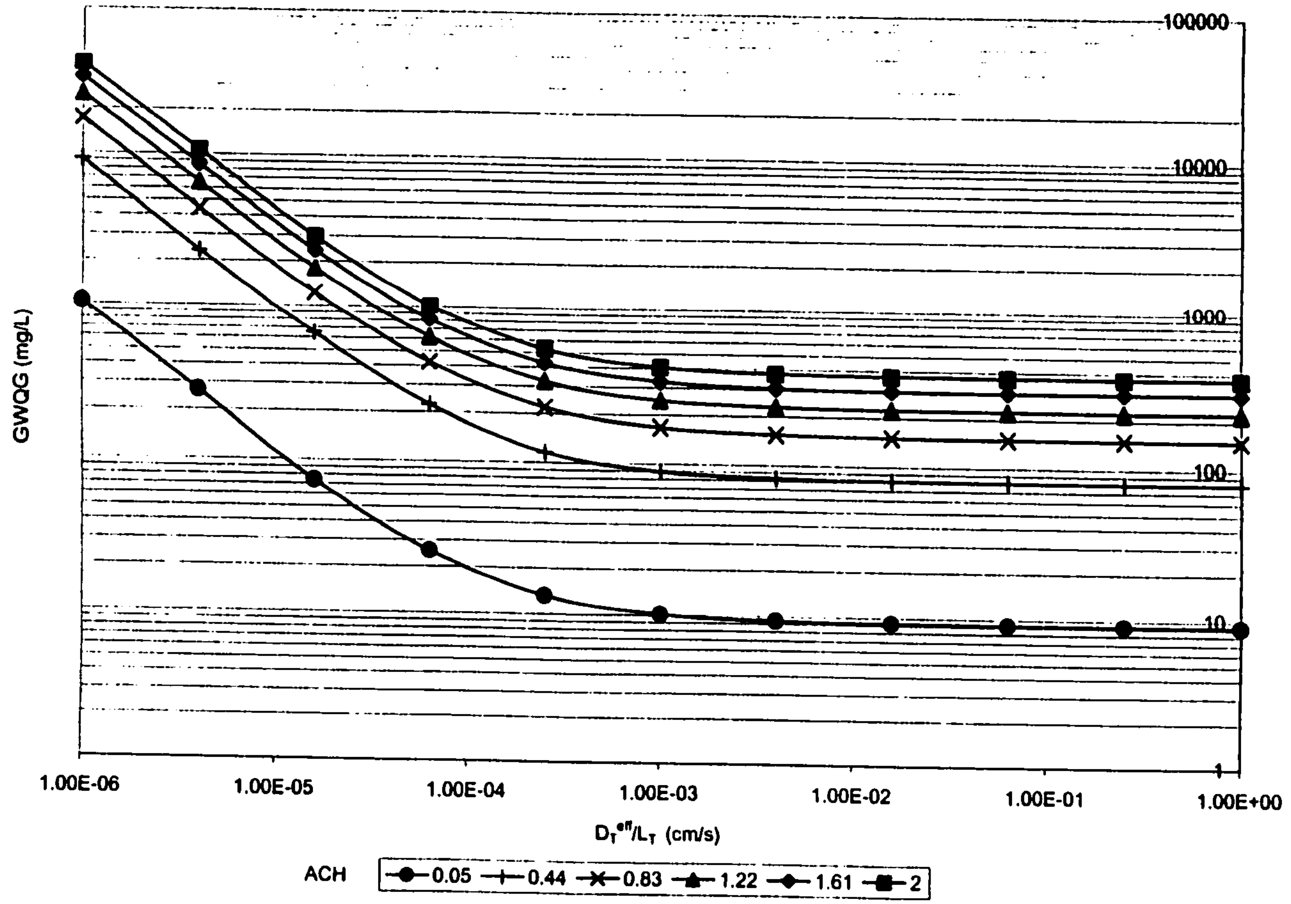

Figure J.4: GWQG for o-Xylene n Coarse-Grained Soils for Slab on Grade Structures, Commercial/Industrial Exposure Scenariovs. $\frac{D_t^{eff}}{L_T}$ for various air exchange rates

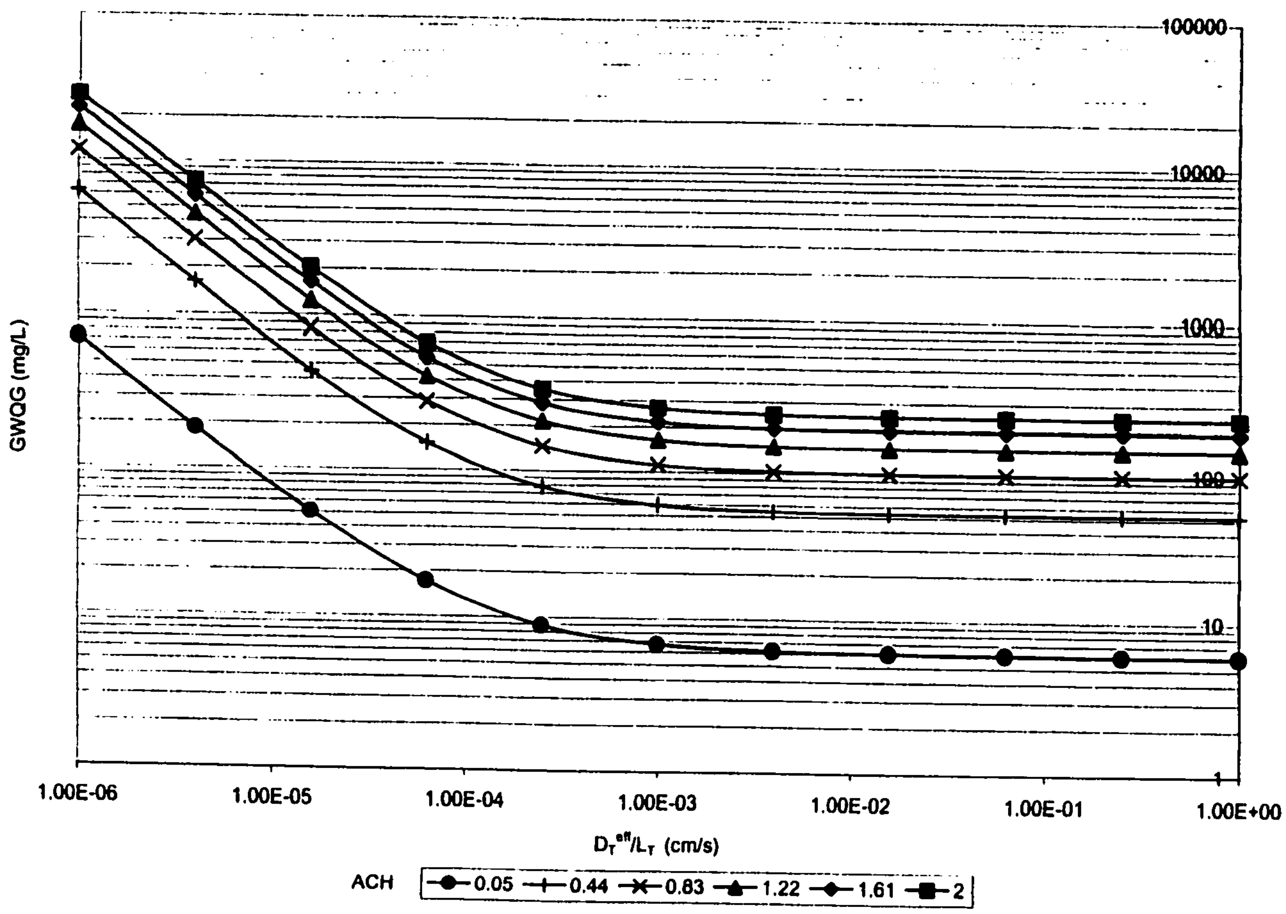

Figure J.5: GWQG for m-Xylene in Coarse-Grained Soils for Slab on Grade Structures, Commercial/Industrial Exposure Scenario vs. $\frac{D_t^{eff}}{L_T}$ for various air exchange rates

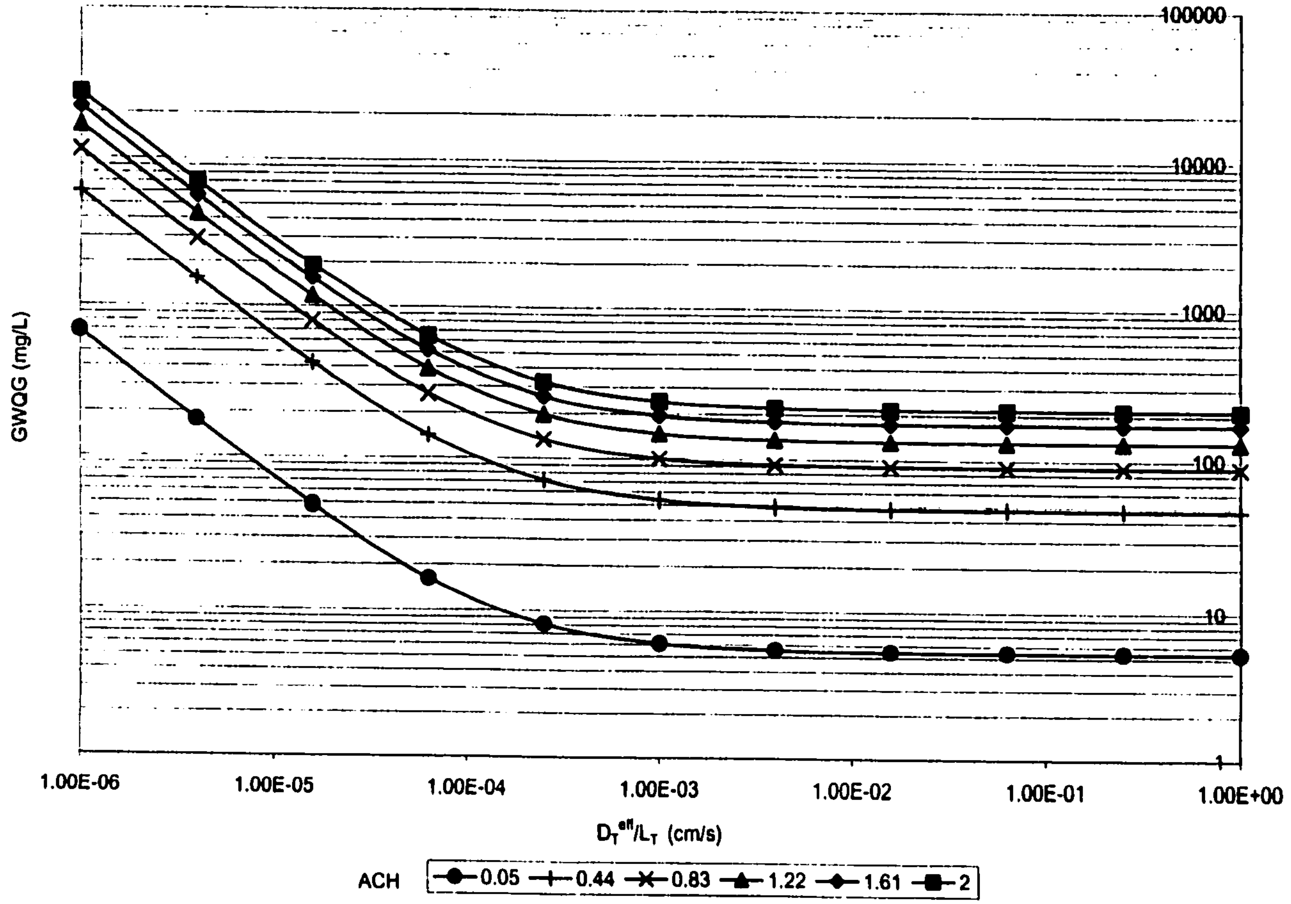

Figure J.6: GWQG for p-Xylene in Coarse-Grained Soils for Slab on Grade Structures, Commercial/Industrial Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

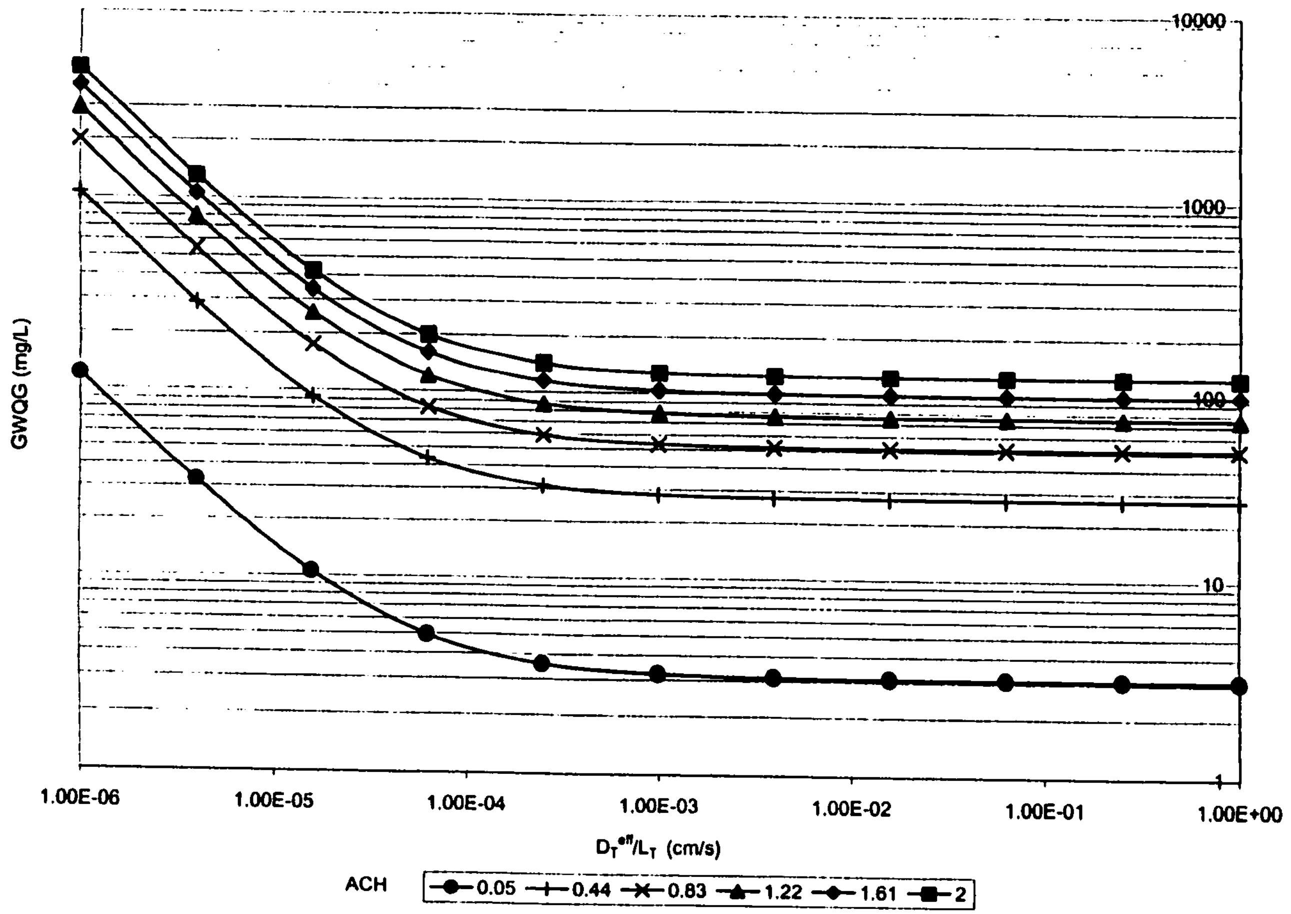

Figure J.7: GWQG for C_6–C_8 aliphatic hydrocarbons in Coarse-Grained Soils for Slab on Grade Structures, Commercial/Industrial Exposure Scenario vs. $\frac{D_t^{eff}}{L_T}$ for various air exchange rates

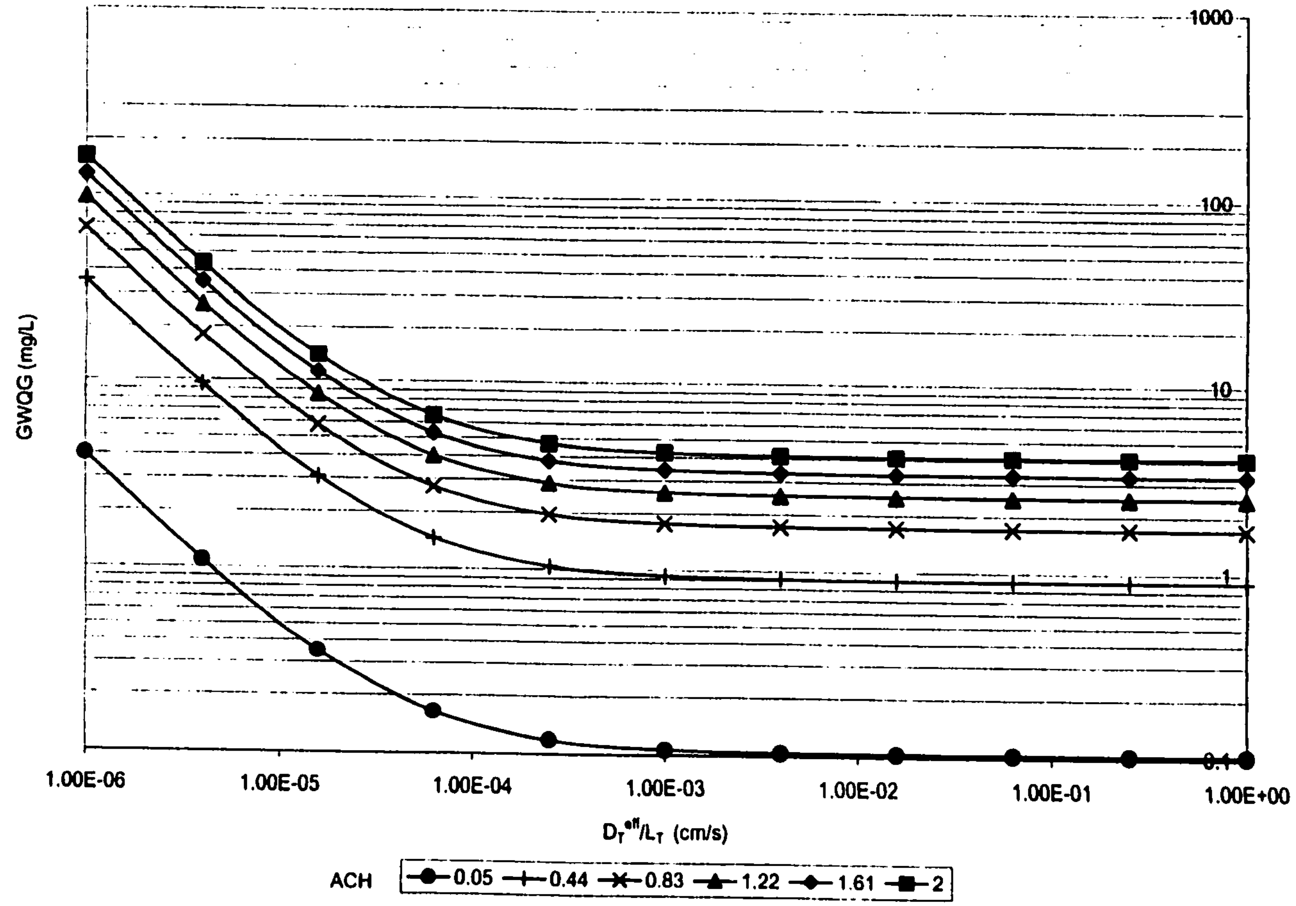

Figure J.8: GWQG for $C_{>8}$–C_{10} aliphatic hydrocarbons in Coarse-Grained Soils for Slab on Grade Structures, Commercial/Industrial Exposure Scenario vs. $\frac{D_t^{eff}}{L_T}$ for various air exchange rates

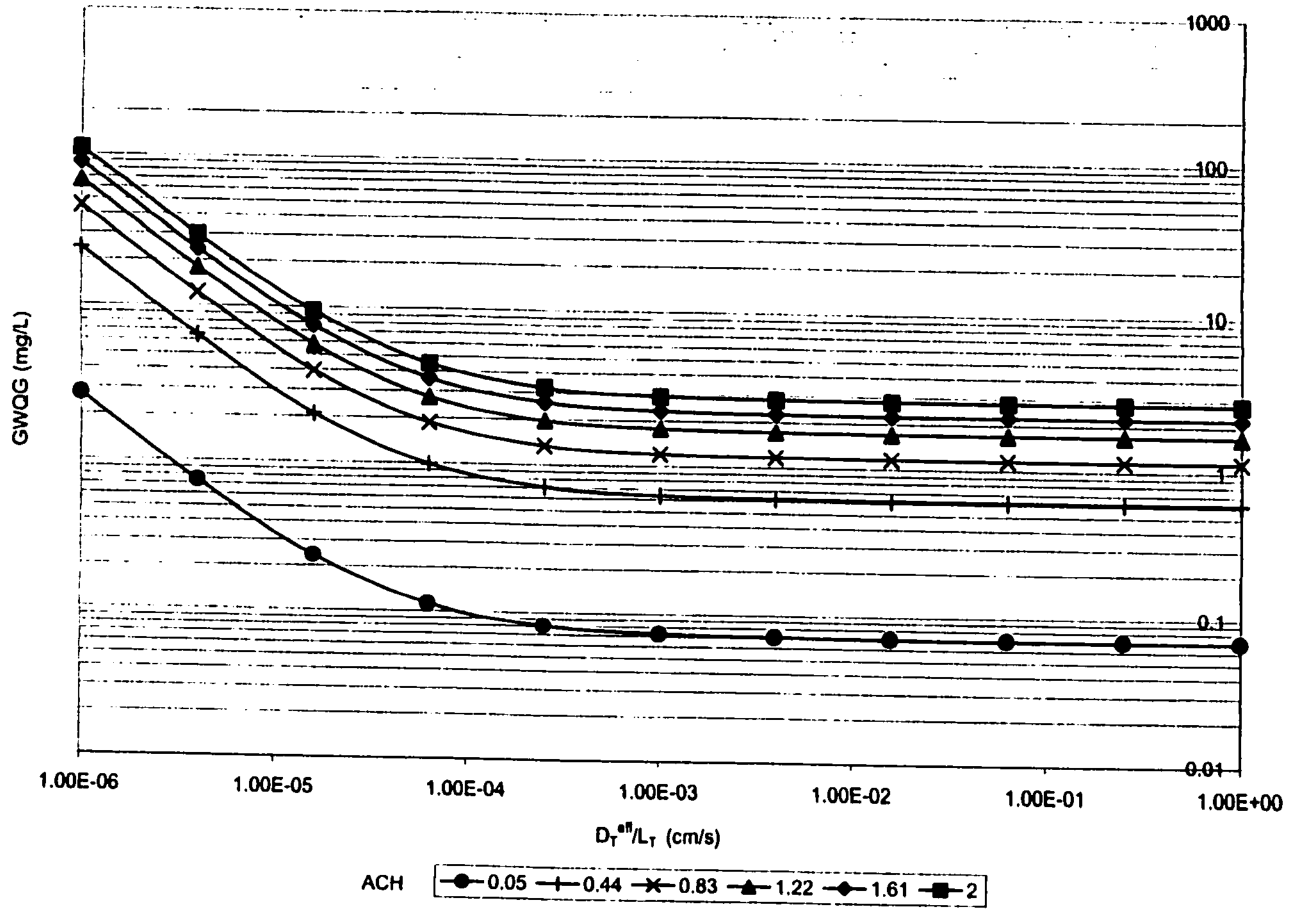

Figure J.9: GWQG for $C_{>10}$–C_{12} aliphatic hydrocarbons iin Coarse-Grained Soils for Slab on Grade Structures, Commercial/Industrial Exposure Scenario vs. $\frac{D_t^{eff}}{L_T}$ for various air exchange rates

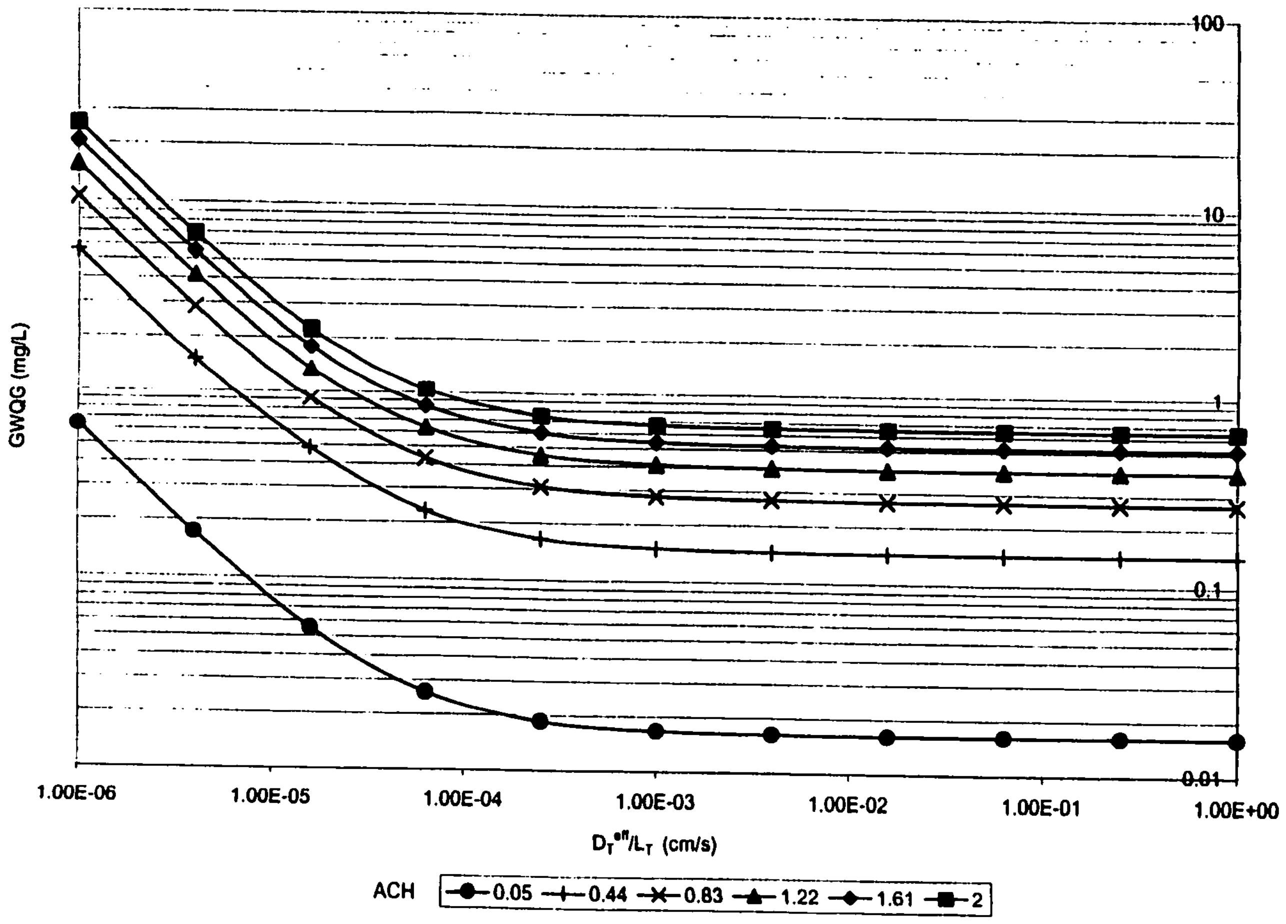

Figure J.10: GWQG for $C_{>12}$–C_{16} aliphatic hydrocarbons in Coarse-Grained Soils for Slab on Grade Structures, Commercial/Industrial Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

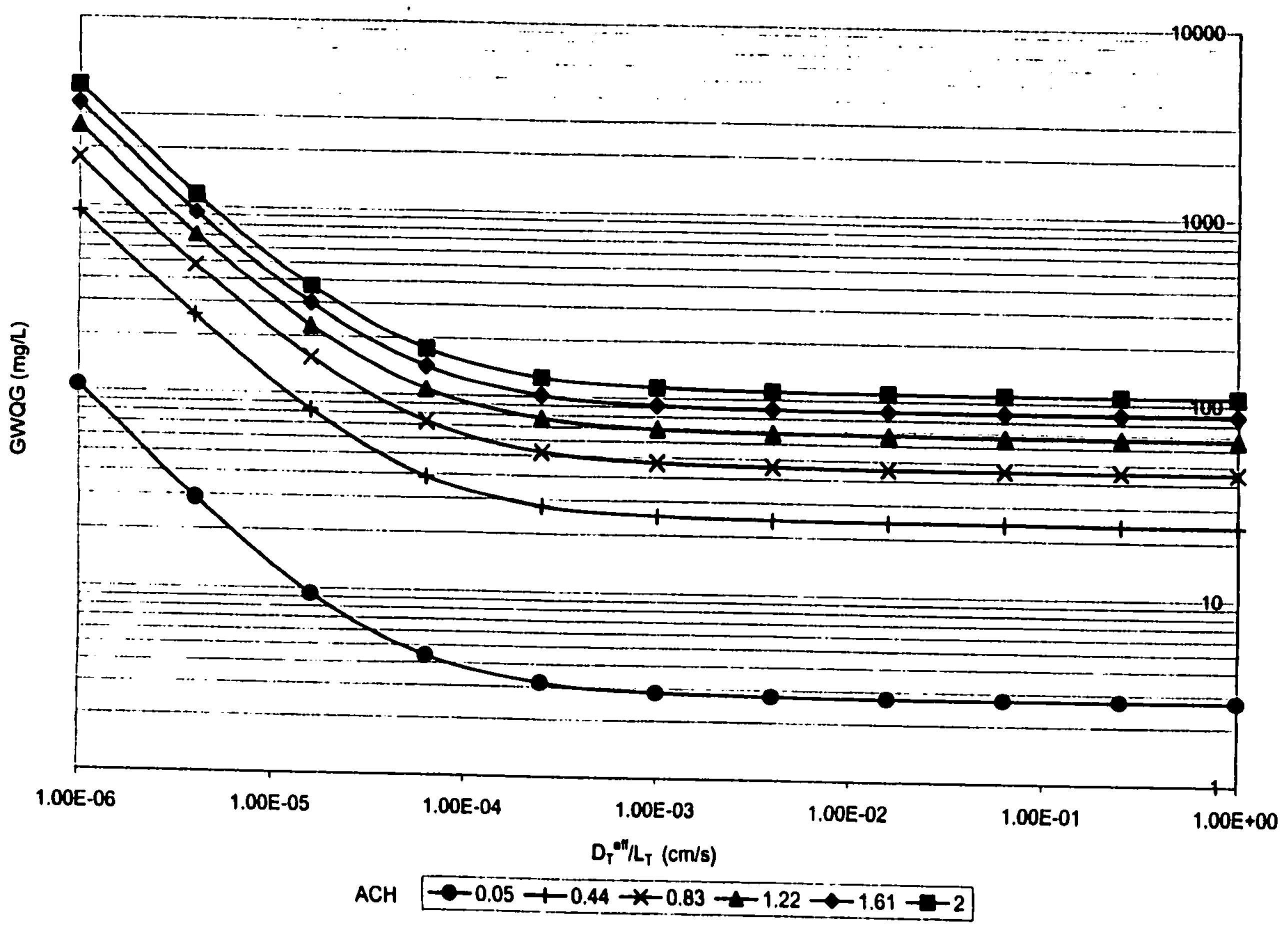

Figure J.11: GWQG for $C_{>6}$–C_{10} aromatic hydrocarbons in Coarse-Grained Soils for Slab on Grade Structures, Commercial/Industrial Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

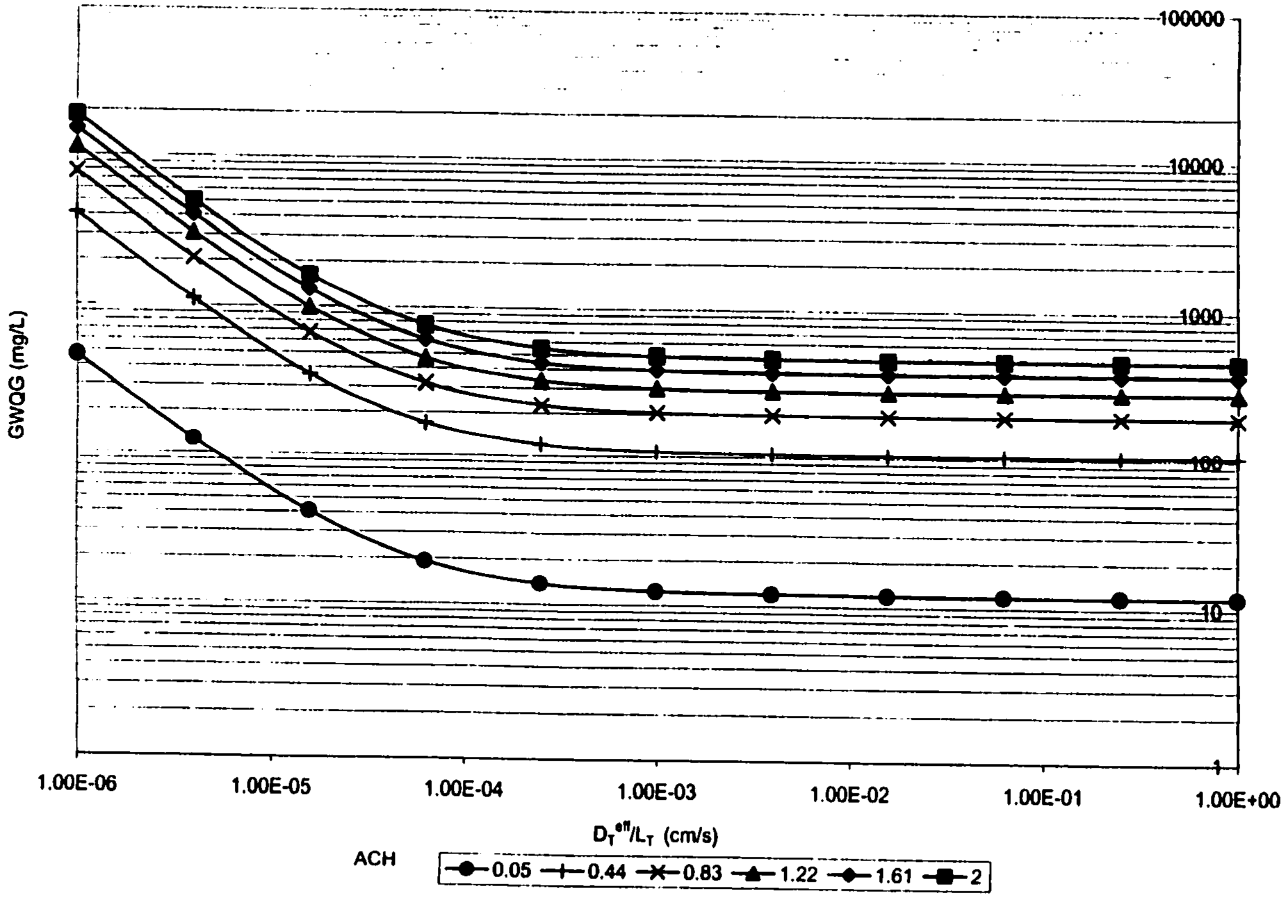

Figure J.12: GWQG for $C_{>10}$–C_{12} aromatic hydrocarbons in Coarse-Grained Soils for Slab on Grade Structures, Commercial/Industrial Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

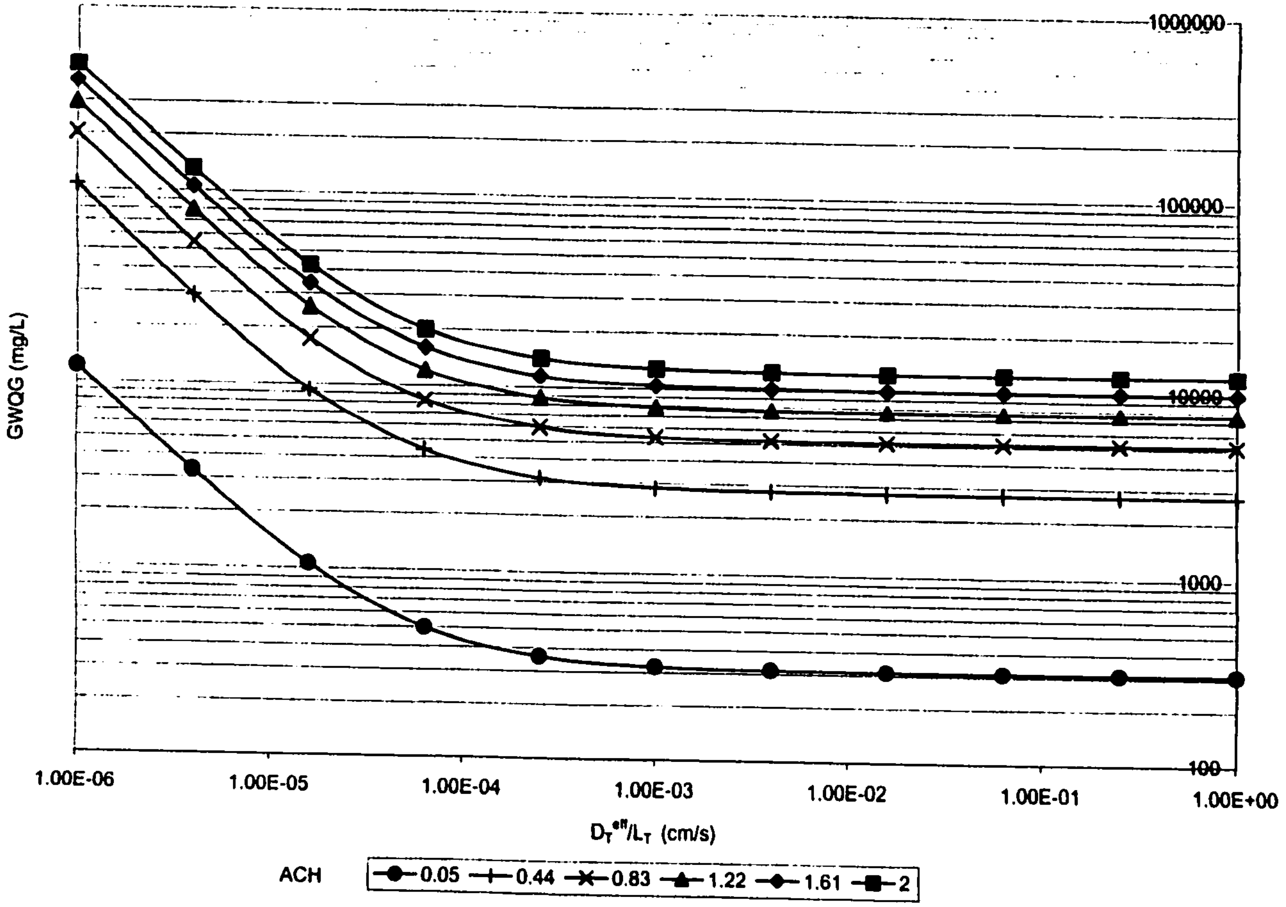

Figure J.13: GWQG for $C_{>12}$–C_{16} aromatic hydrocarbons in Coarse-Grained Soils for Slab on Grade Structures, Commercial/Industrial Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

Appendix K

Soil Quality Guidelines for Fine-Grained Soils for Buildings with Basements, Agricultural/Residential Exposure Scenarios

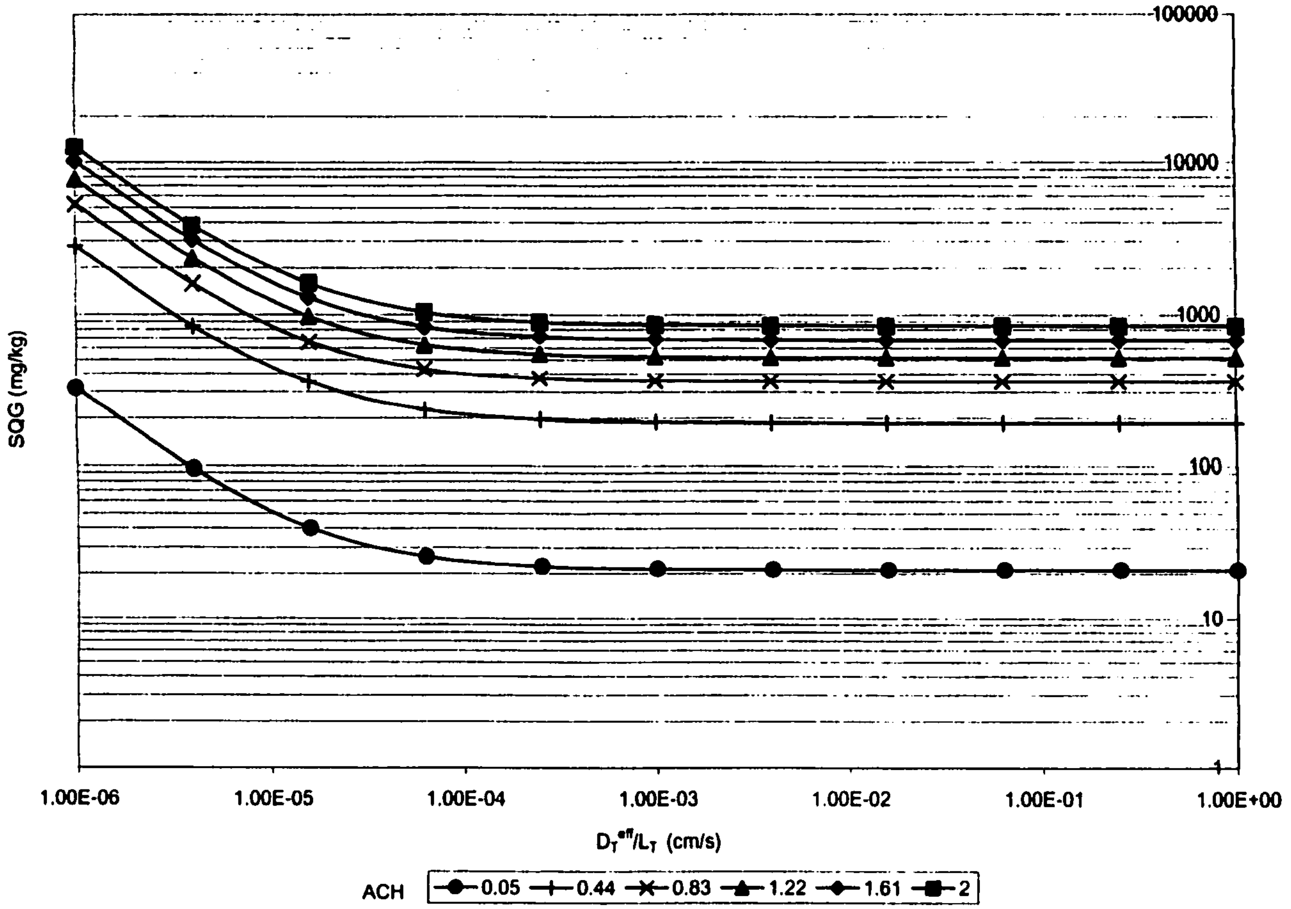

Figure K.1: SQG for Benzene in Fine-Grained Soils for a Building with Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_t^{eff}}{L_T}$ for various air exchange rates

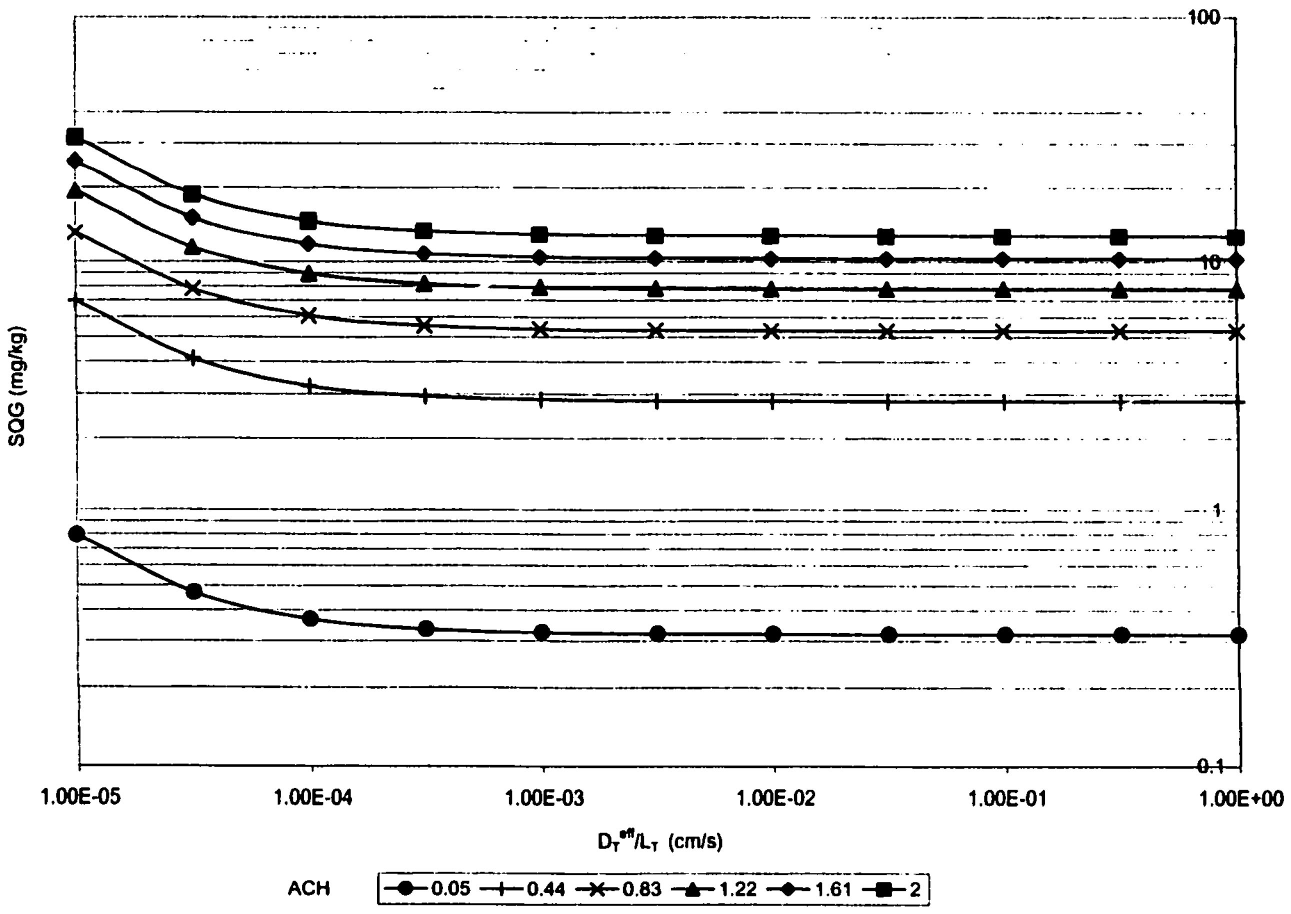

Figure K.2: SQG for Toluene in Fine-Grained Soils for a Building with Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_t^{eff}}{L_T}$ for various air exchange rates

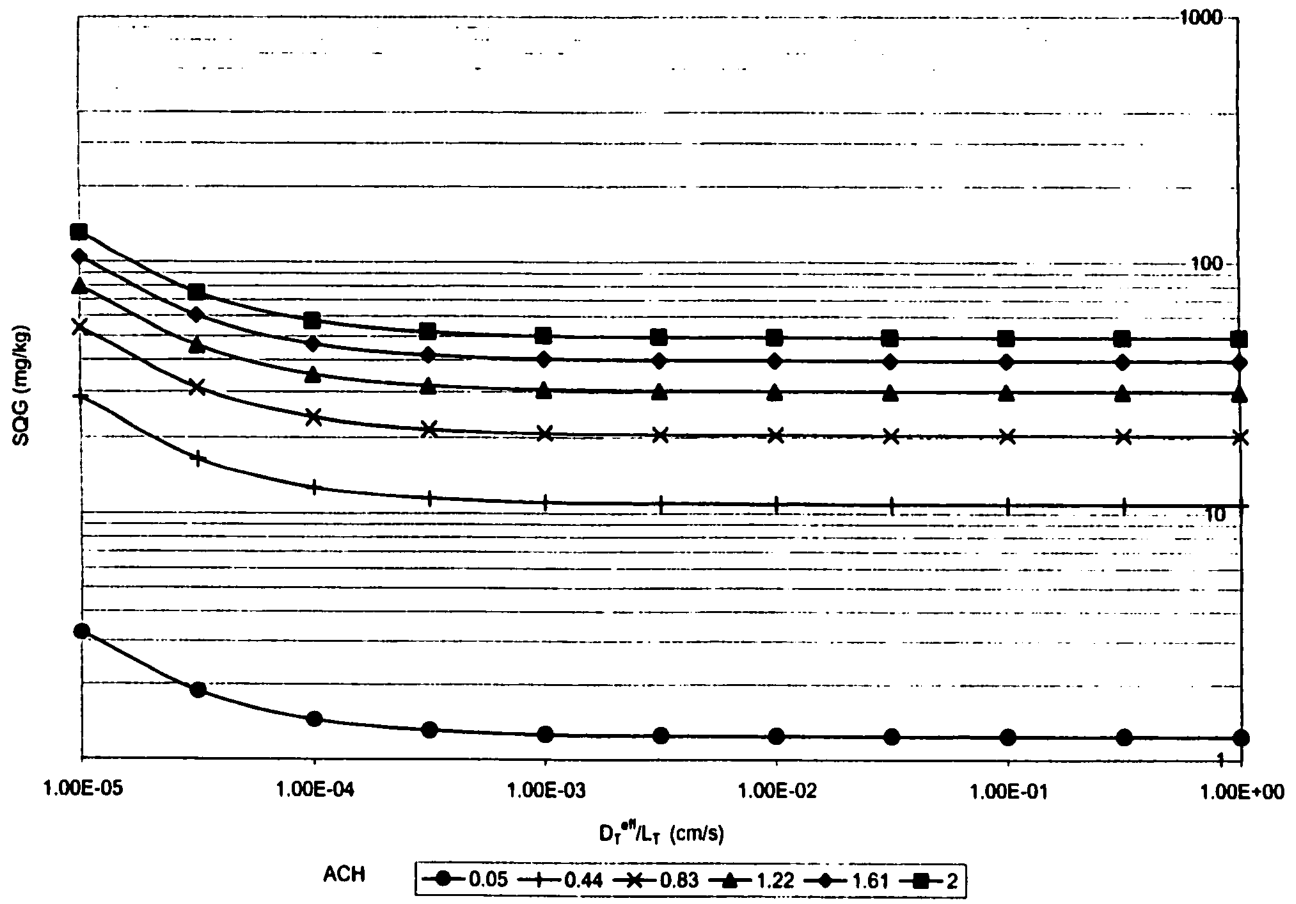

Figure K.3: SQG for Ethylbenzene in Fine-Grained Soils for a Building with Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

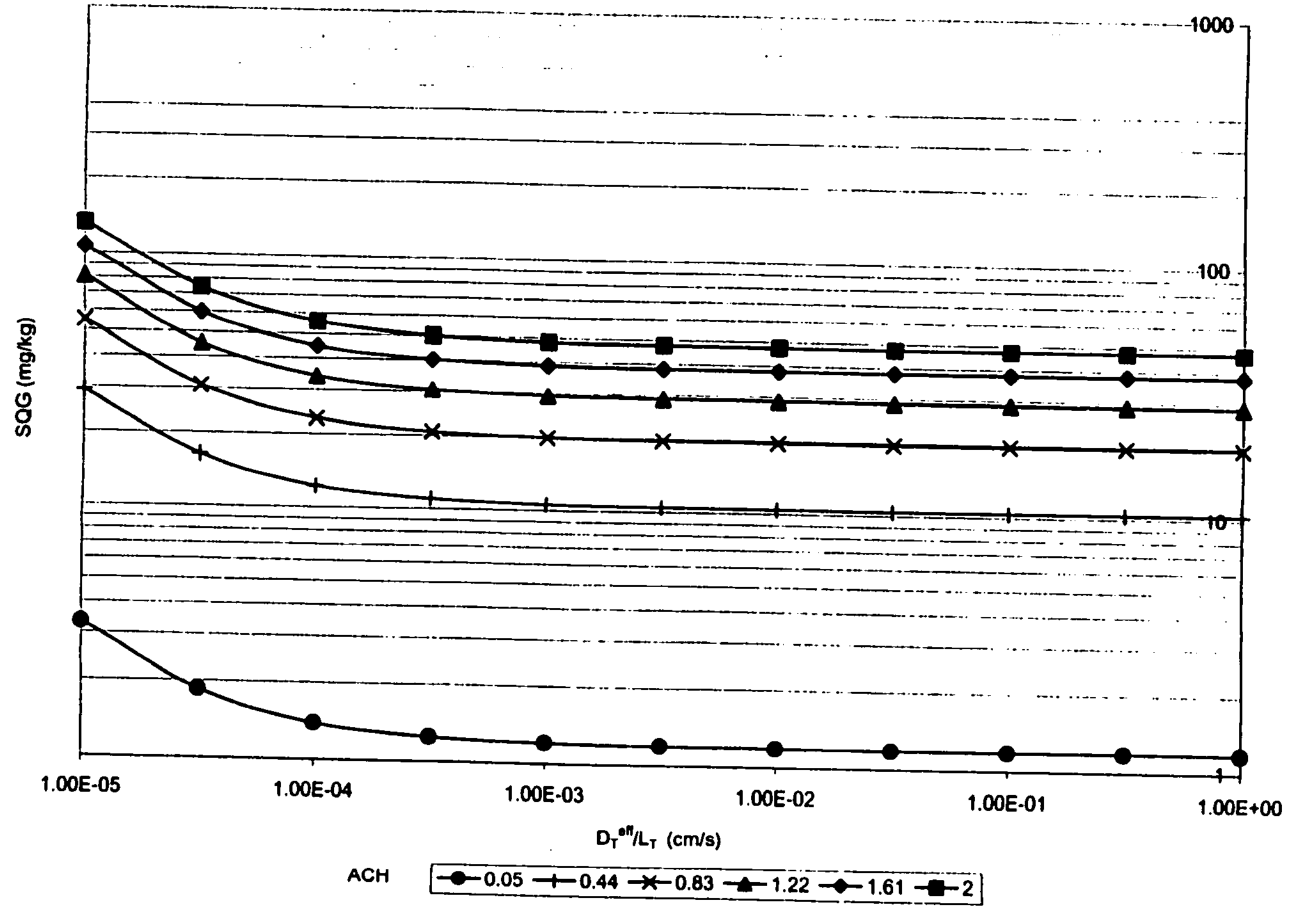

Figure K.4: SQG for o-Xylene in Fine-Grained Soils for a Building with Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

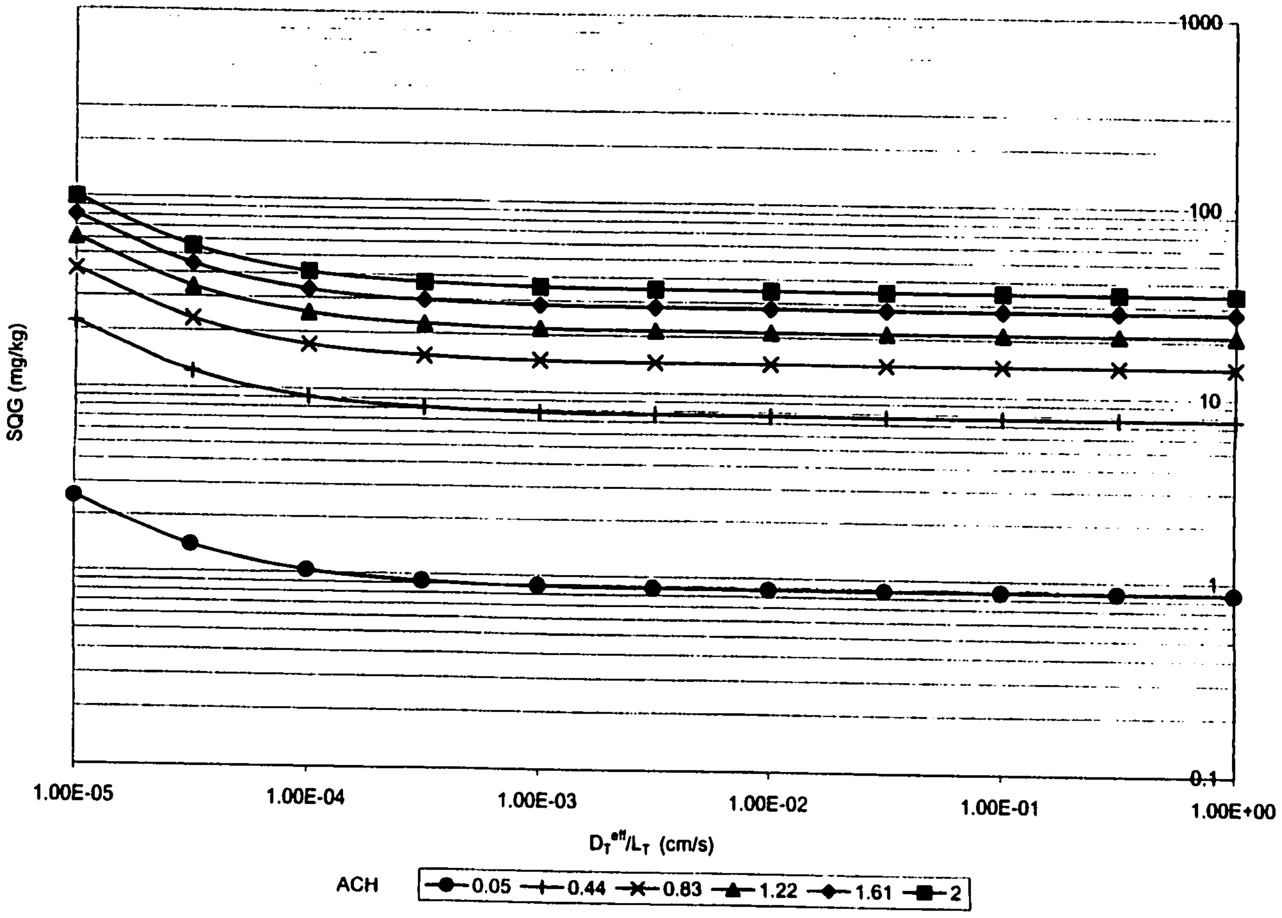

Figure K.5: SQG for m-Xylene iin Fine-Grained Soils for a Building with Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

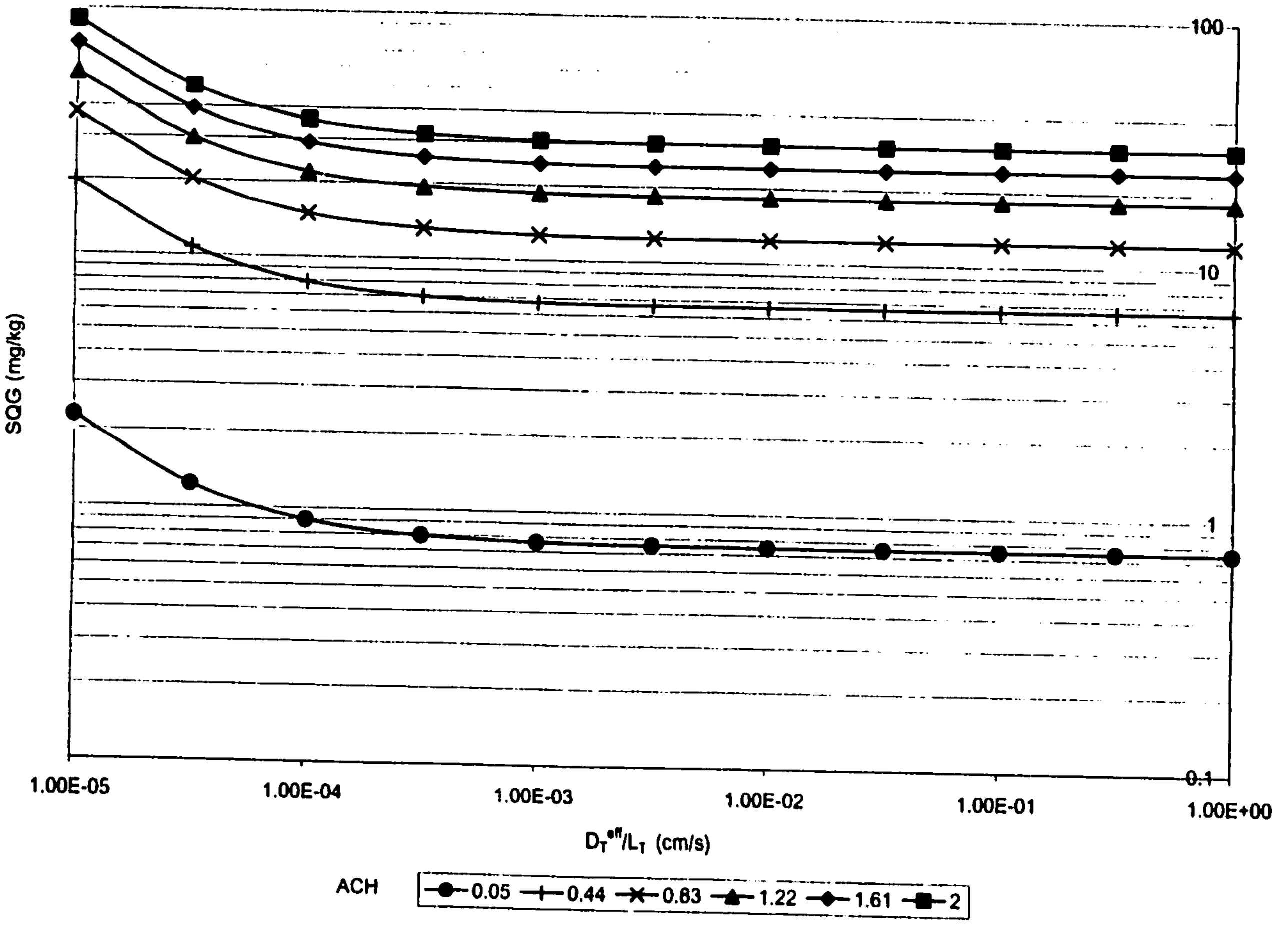

Figure K.6: SQG for p-Xylene in Fine-Grained Soils for a Building with Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

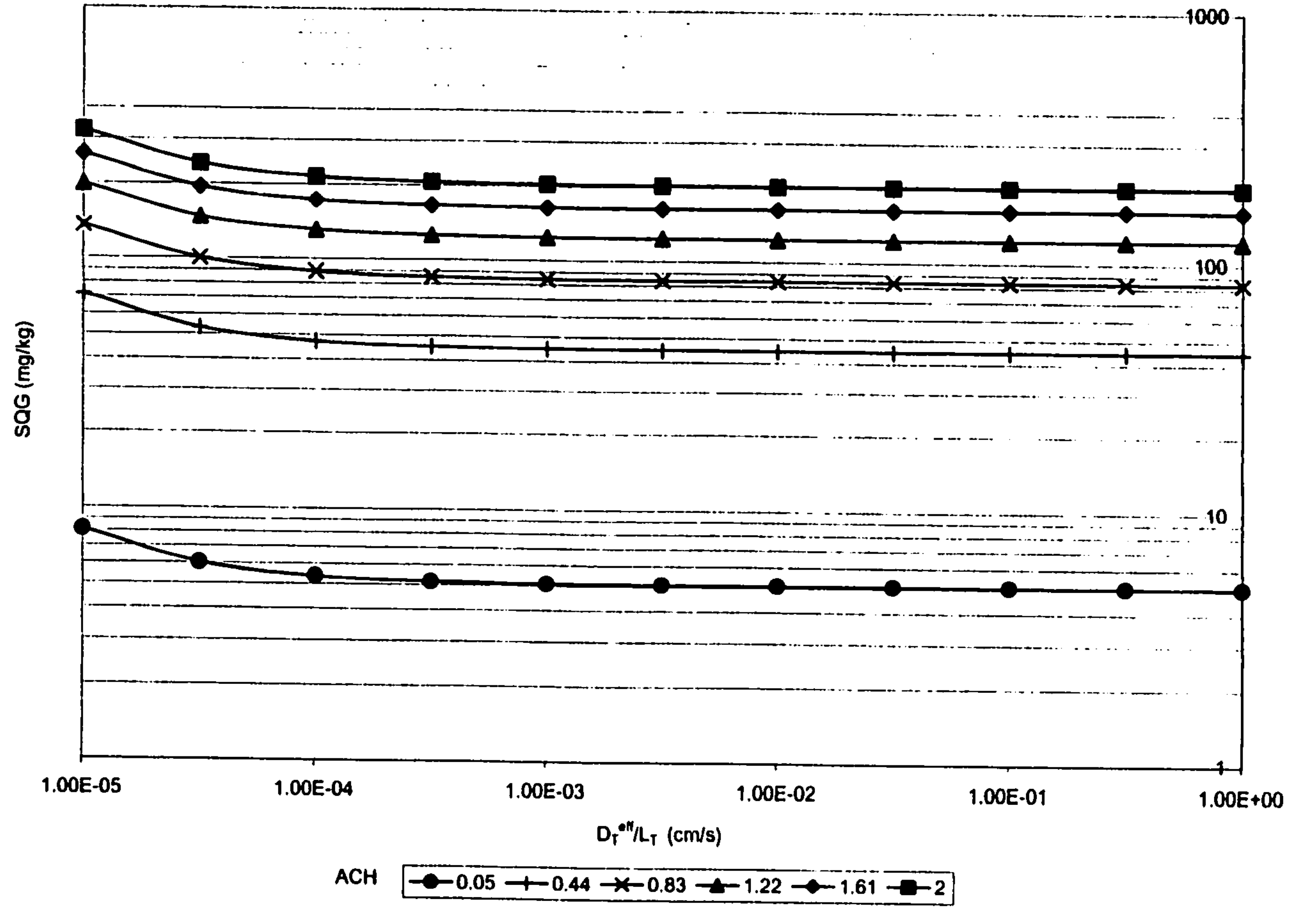

Figure K.7: SQG for C_6–C_8 aliphatic hydrocarbons in Fine-Grained Soils for a Building with Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

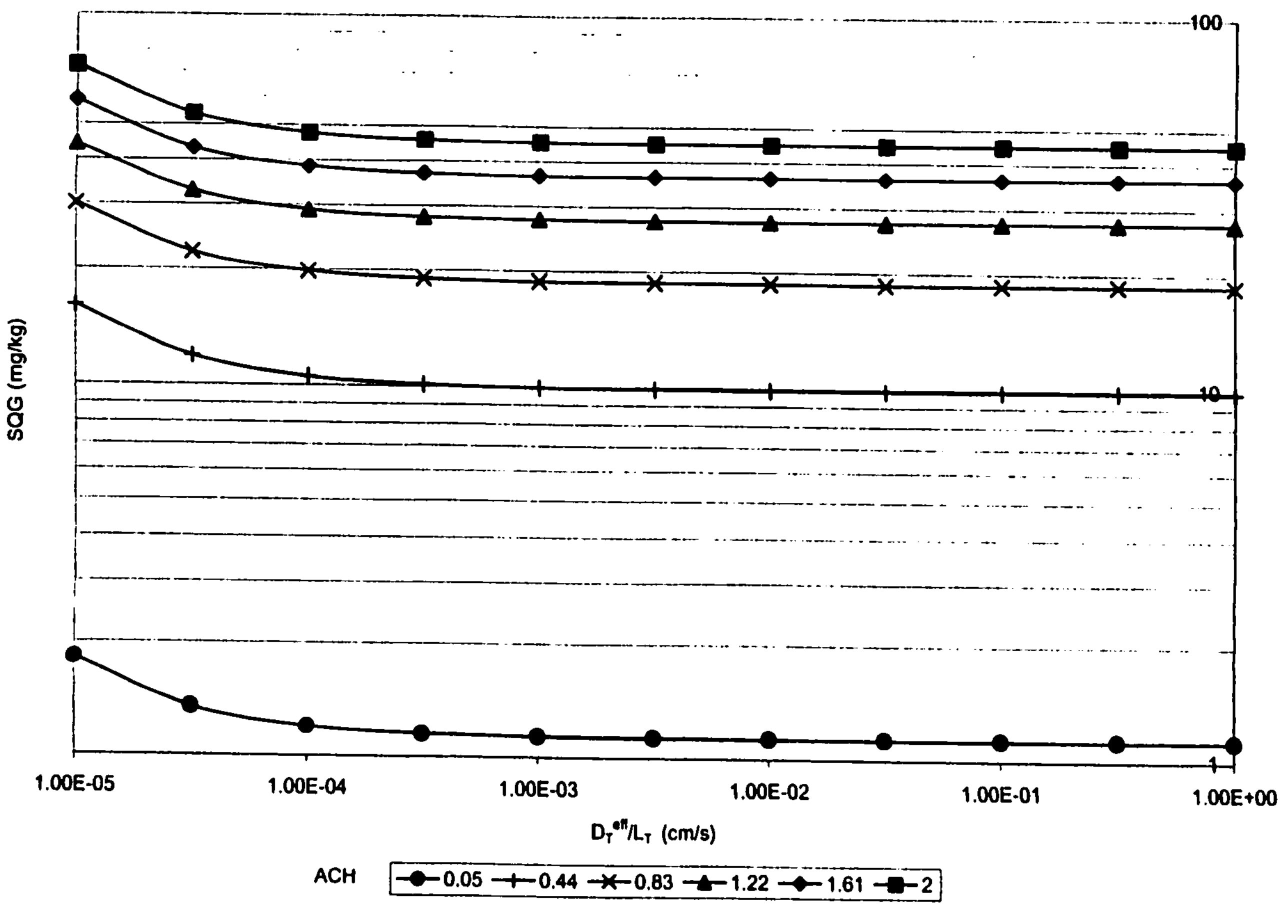

Figure K.8: SQG for $C_{>8}$–C_{10} aliphatic hydrocarbons in Fine-Grained Soils for a Building with Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

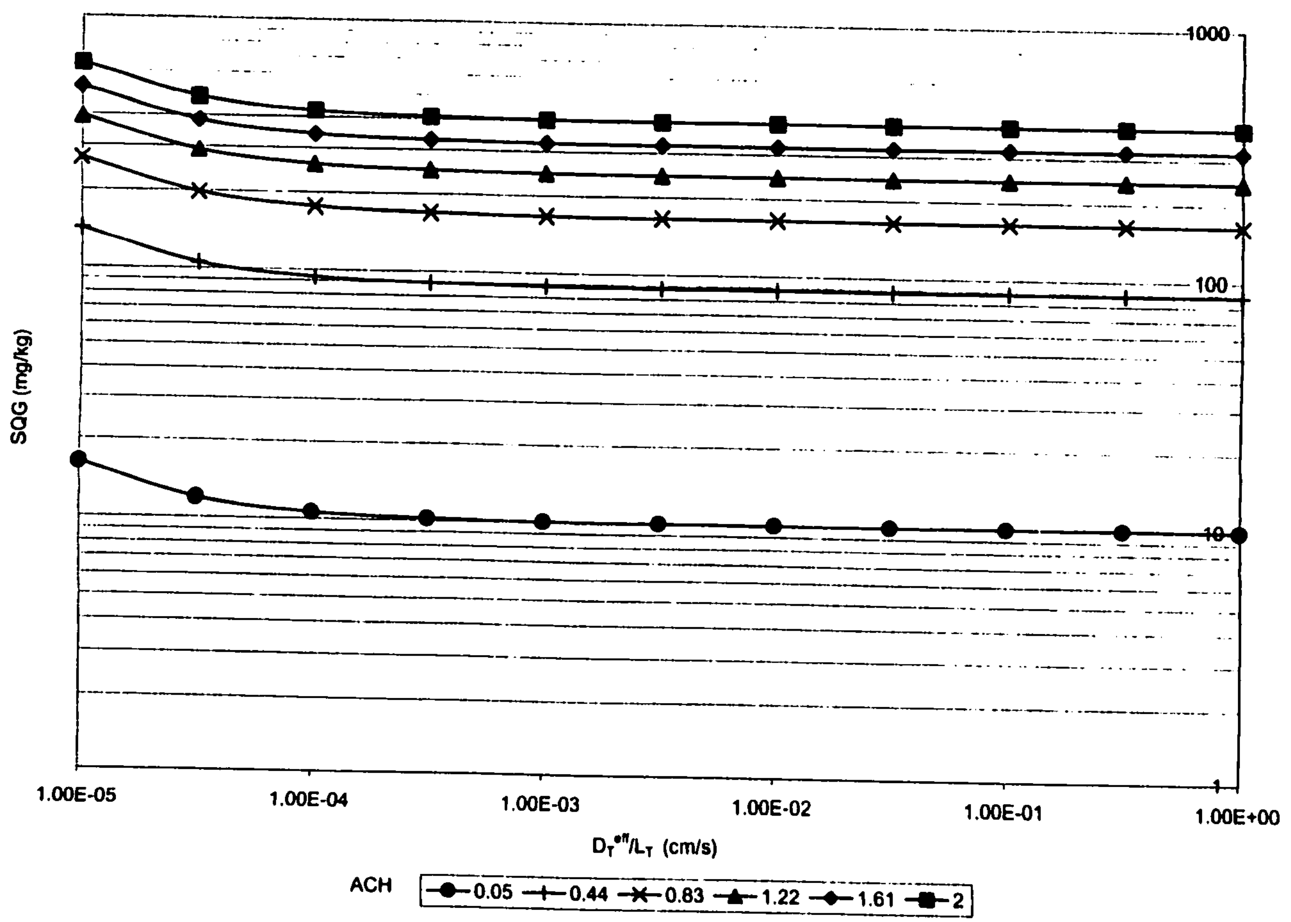

Figure K.9: SQG for $C_{>10}$–C_{12} aliphatic hydrocarbons in Fine-Grained Soils for a Building with Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

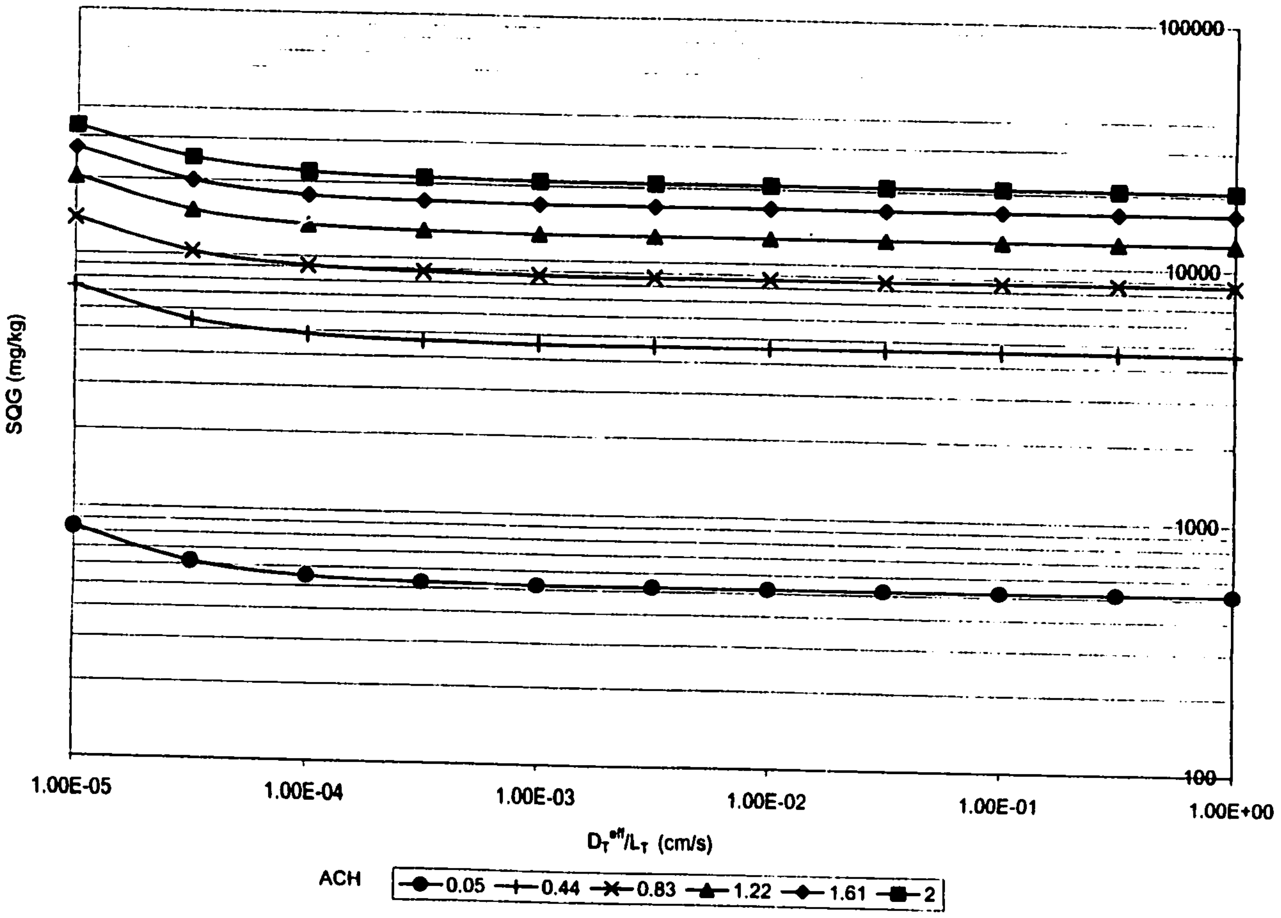

Figure K.10: SQG for $C_{>12}$–C_{16} aliphatic hydrocarbons in Fine-Grained Soils for a Building with Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

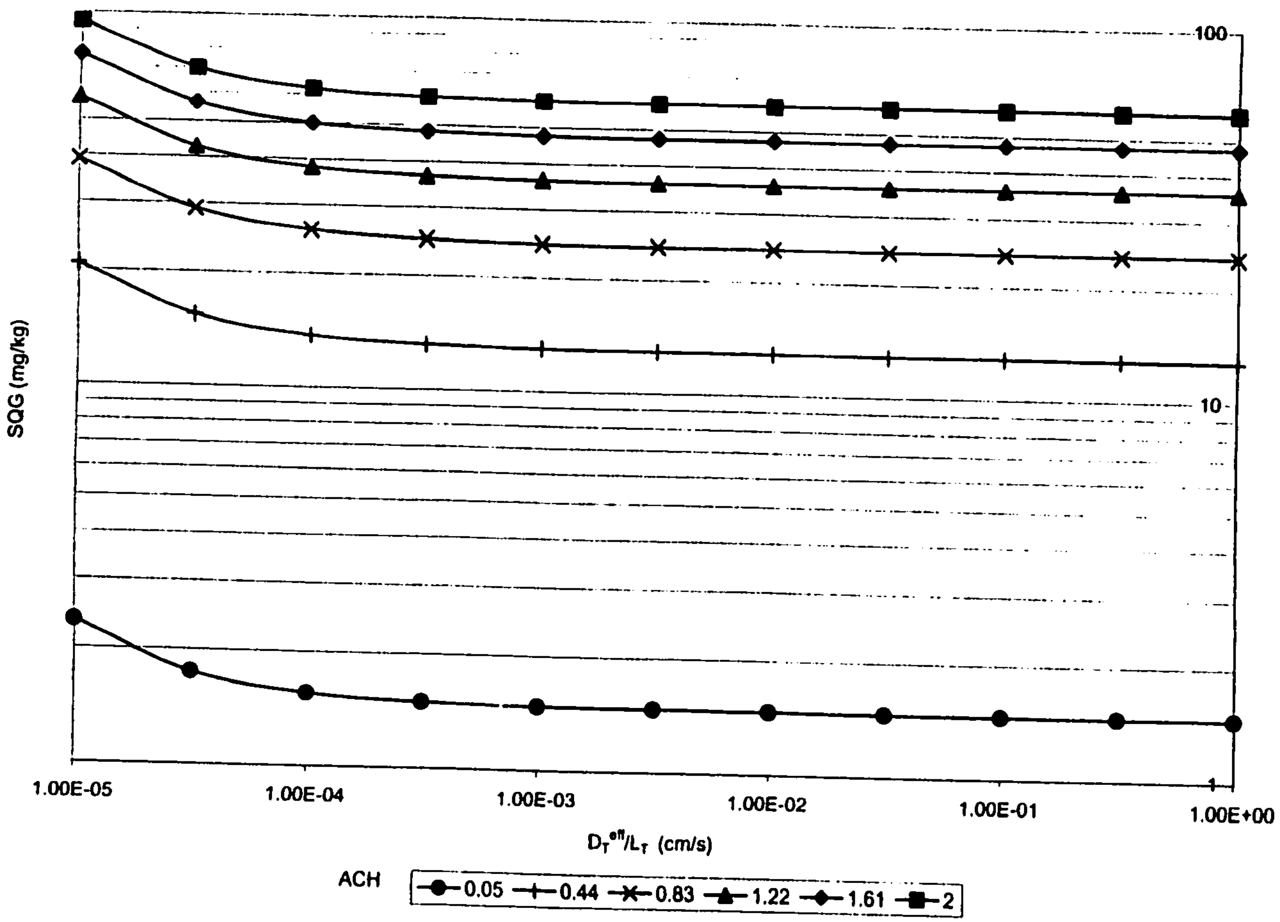

Figure K.11: SQG for $C_{>8}$–C_{10} aromatic hydrocarbons in Fine-Grained Soils for a Building with Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

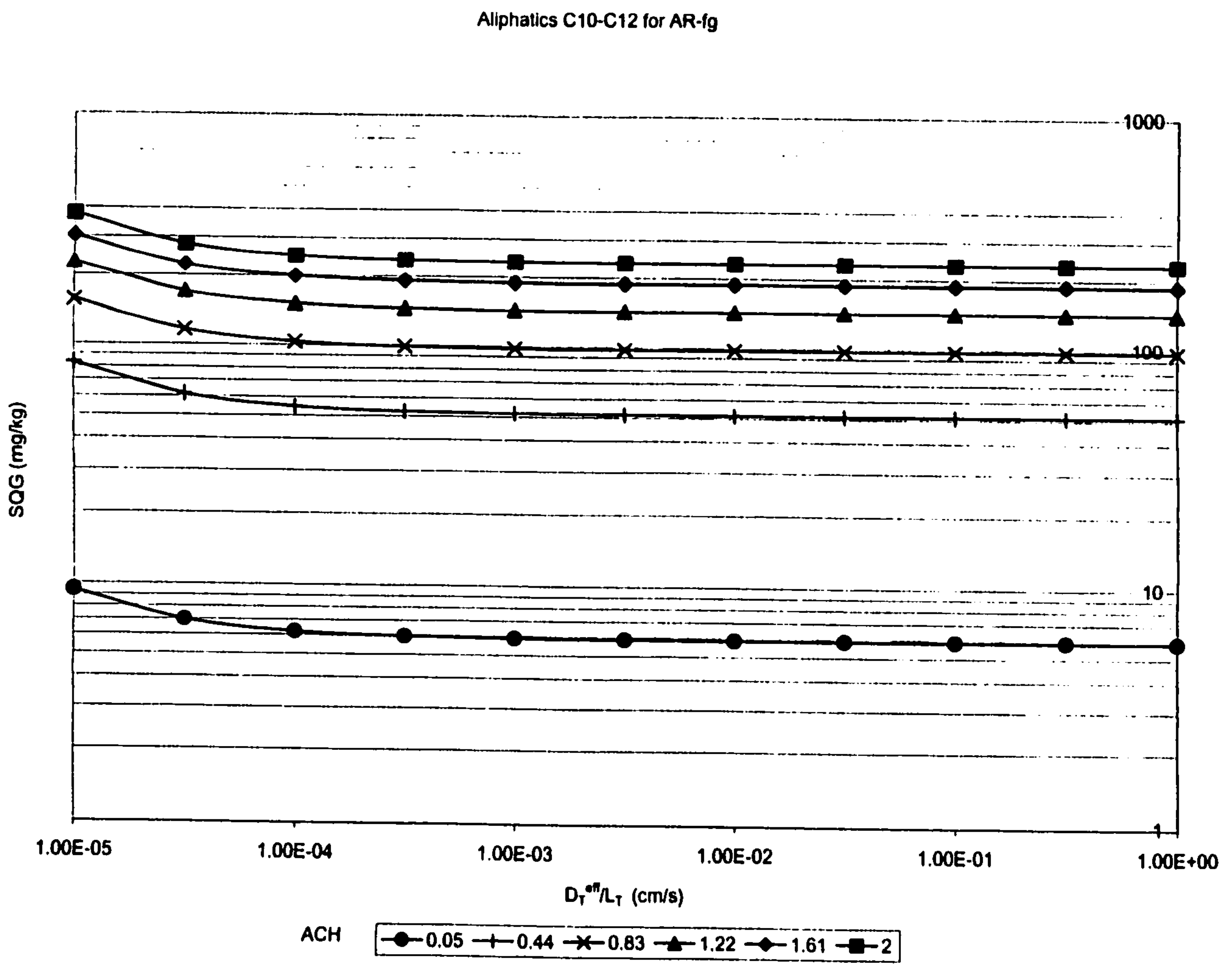

Figure K.12: SQG for $C_{>10}$–C_{12} aromatic hydrocarbons in Fine-Grained Soils for a Building with Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

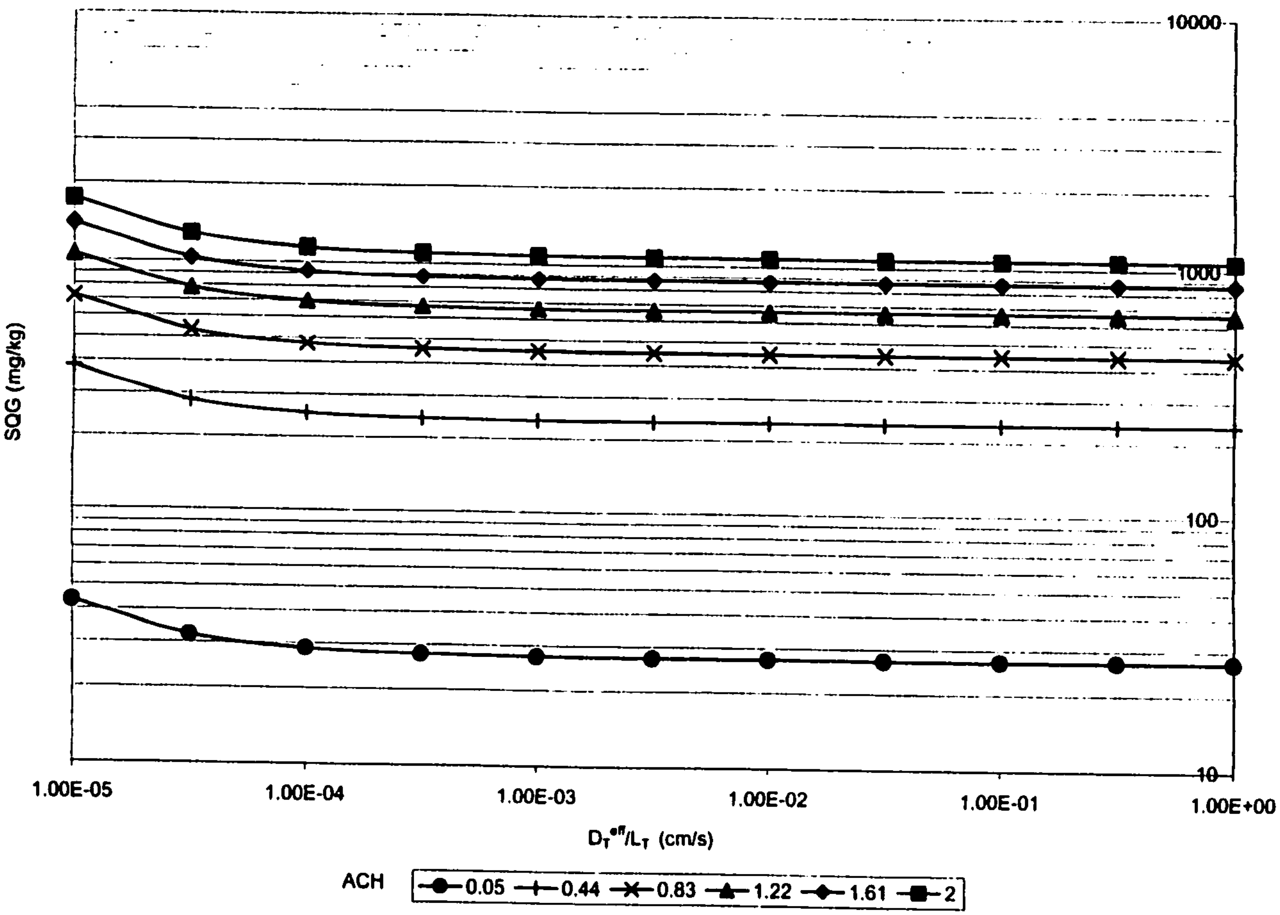

Figure K.13: SQG for $C_{>12}$–C_{16} aromatic hydrocarbons in Fine-Grained Soils for a Building with Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

Appendix L

Soil Quality Guidelines for Fine-Grained Soils for Slab on Grade Structures, Commercial/Industrial Exposure Scenarios

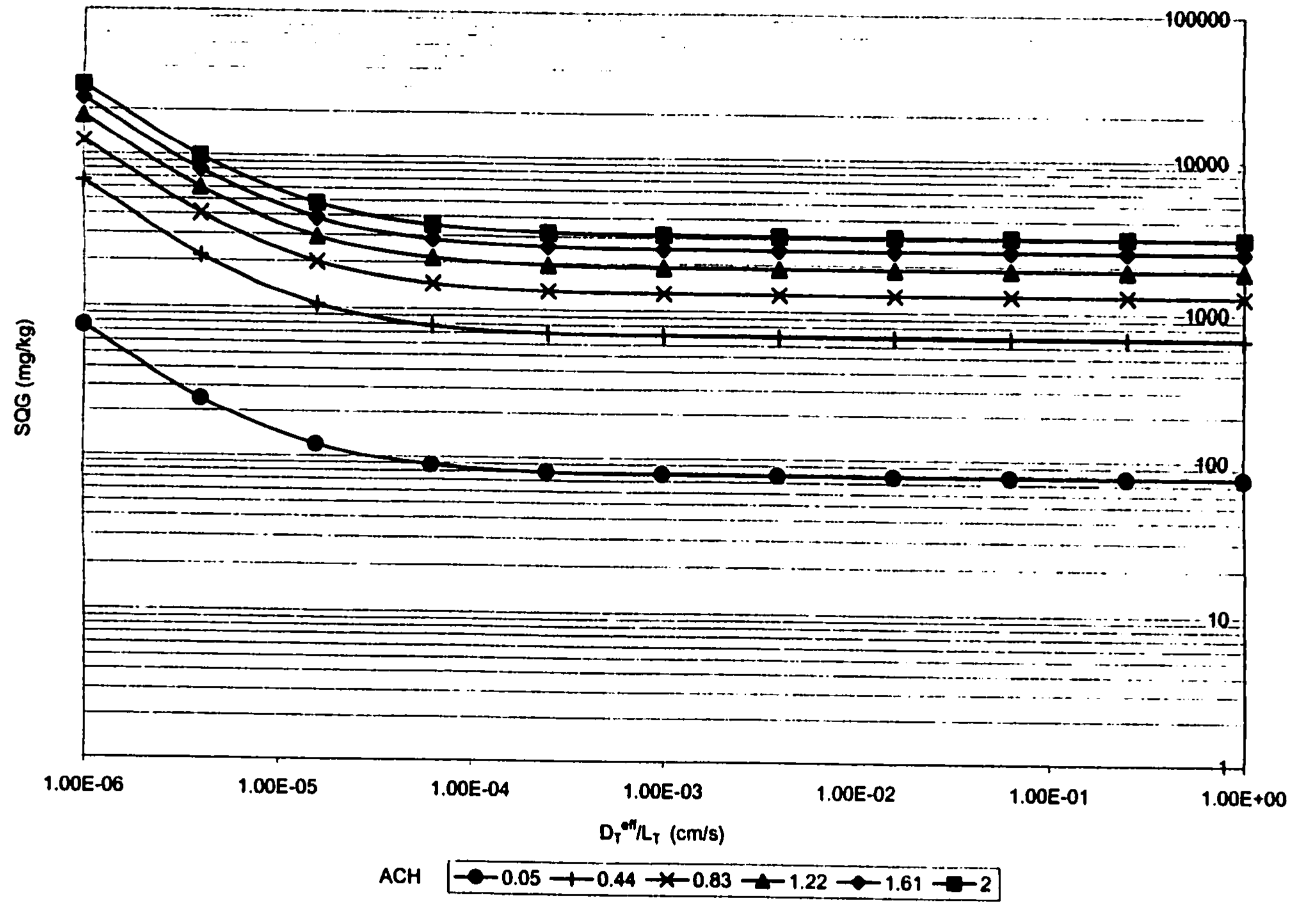

Figure L.1: SQG for Benzene in Fine-Grained Soils for Slab on Grade Structures, Commercial/Industrial Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

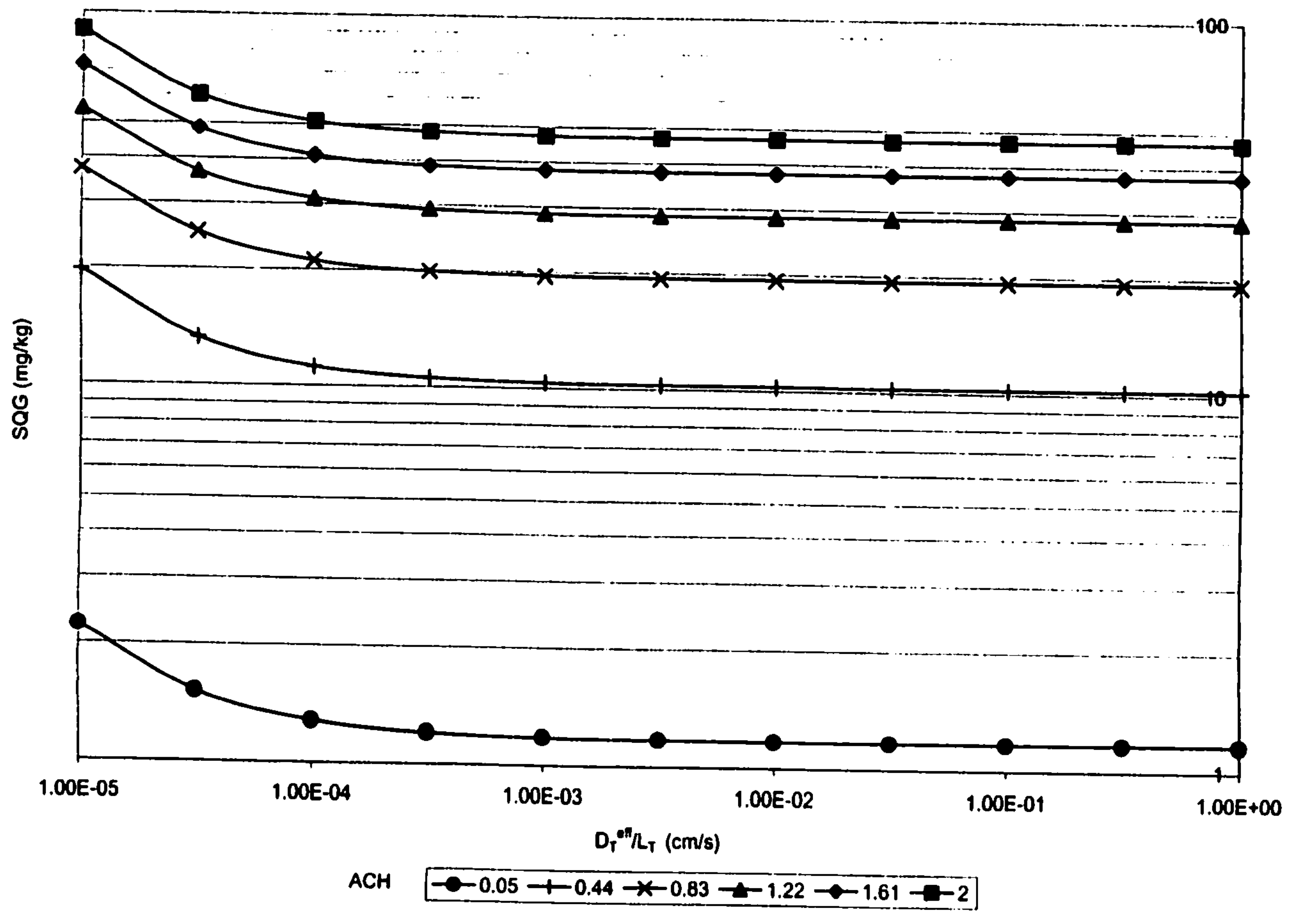

Figure L.2: SQG for Toluene in Fine-Grained Soils for Slab on Grade Structures, Commercial/Industrial Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

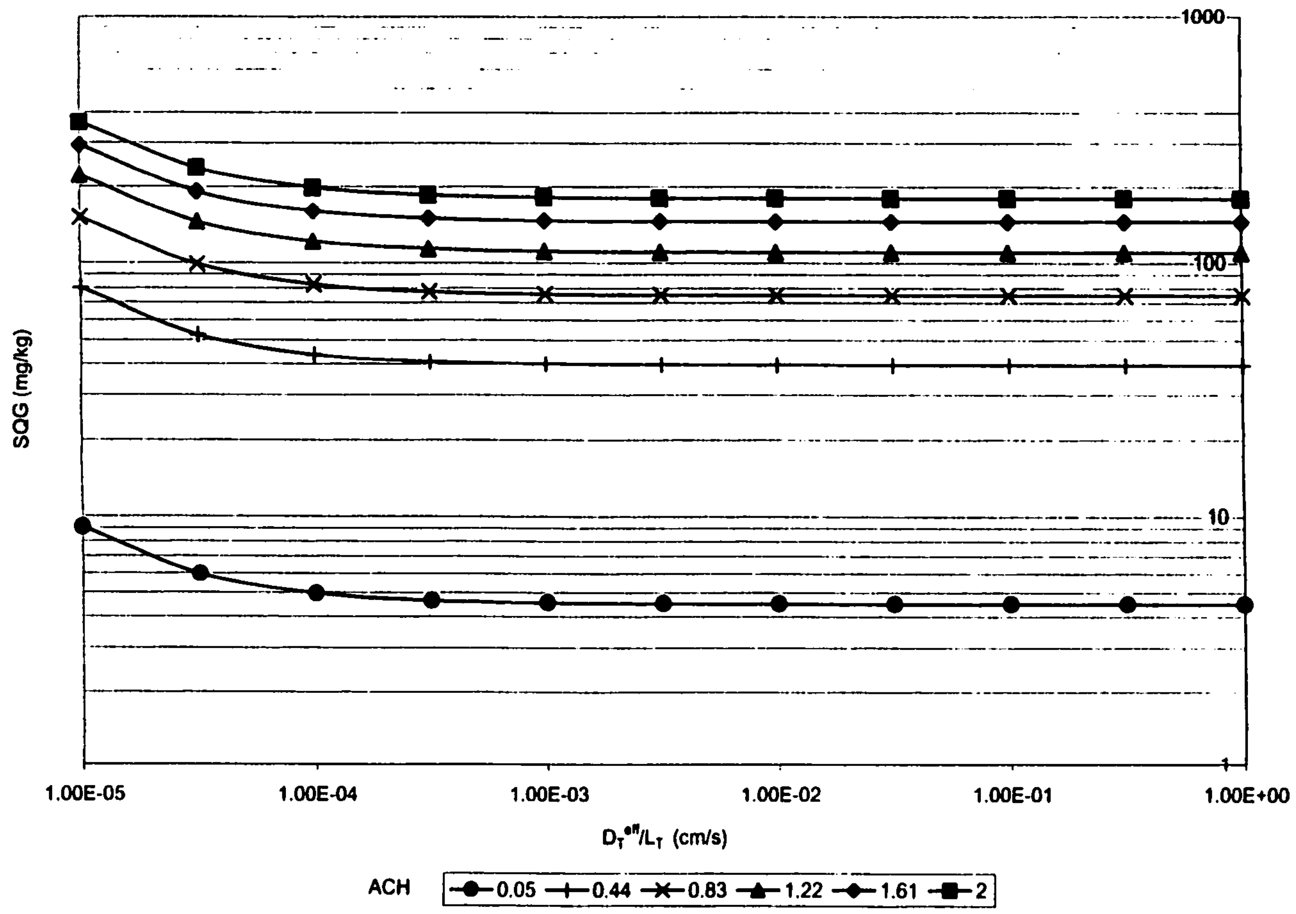

Figure L.3: SQG for Ethylbenzene in Fine-Grained Soils for Slab on Grade Structures, Commercial/Industrial Exposure Scenario vs. $\frac{D_t^{eff}}{L_T}$ for various air exchange rates

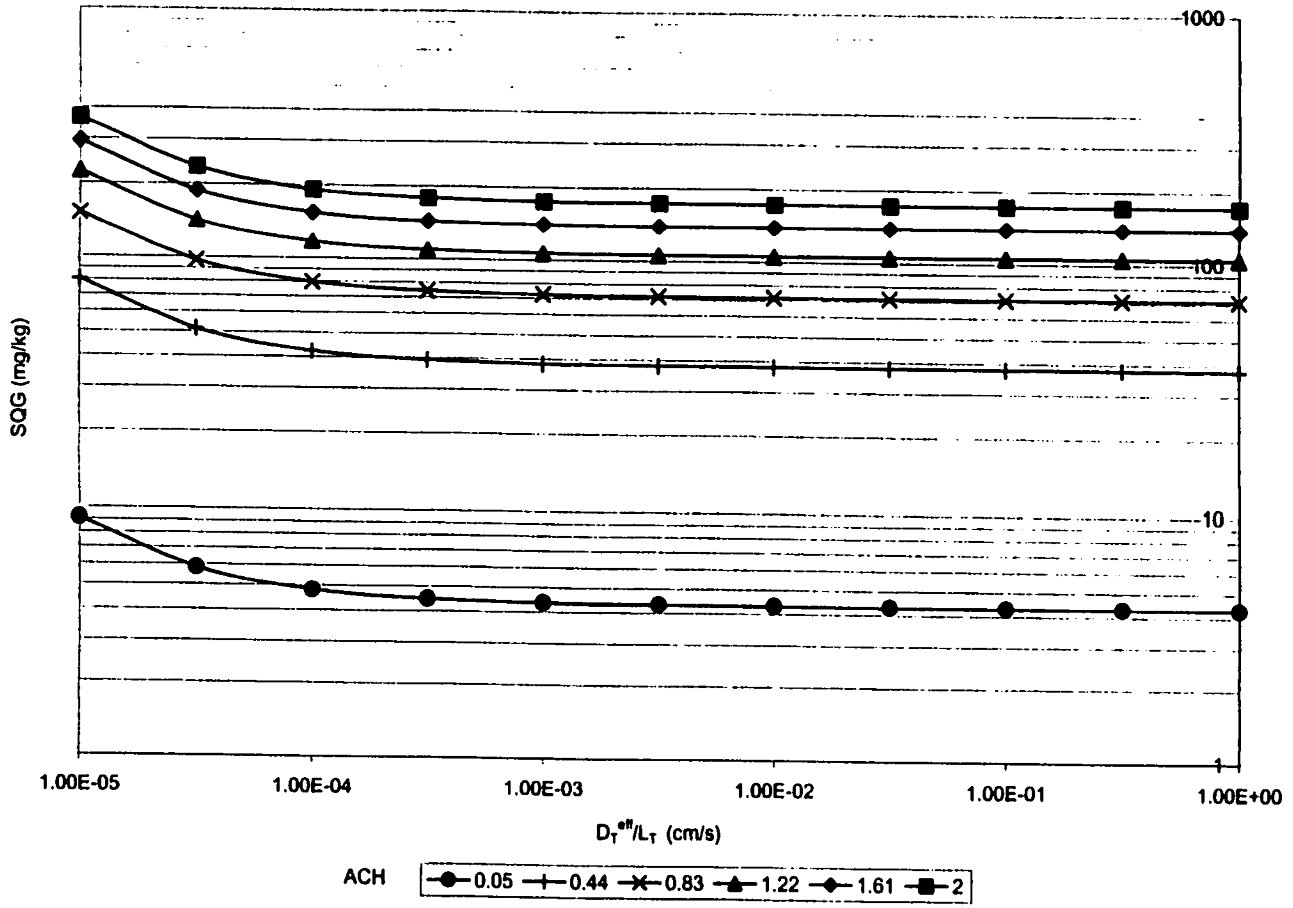

Figure L.4: SQG for o-Xylene in Fine-Grained Soils for Slab on Grade Structures, Commercial/Industrial Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

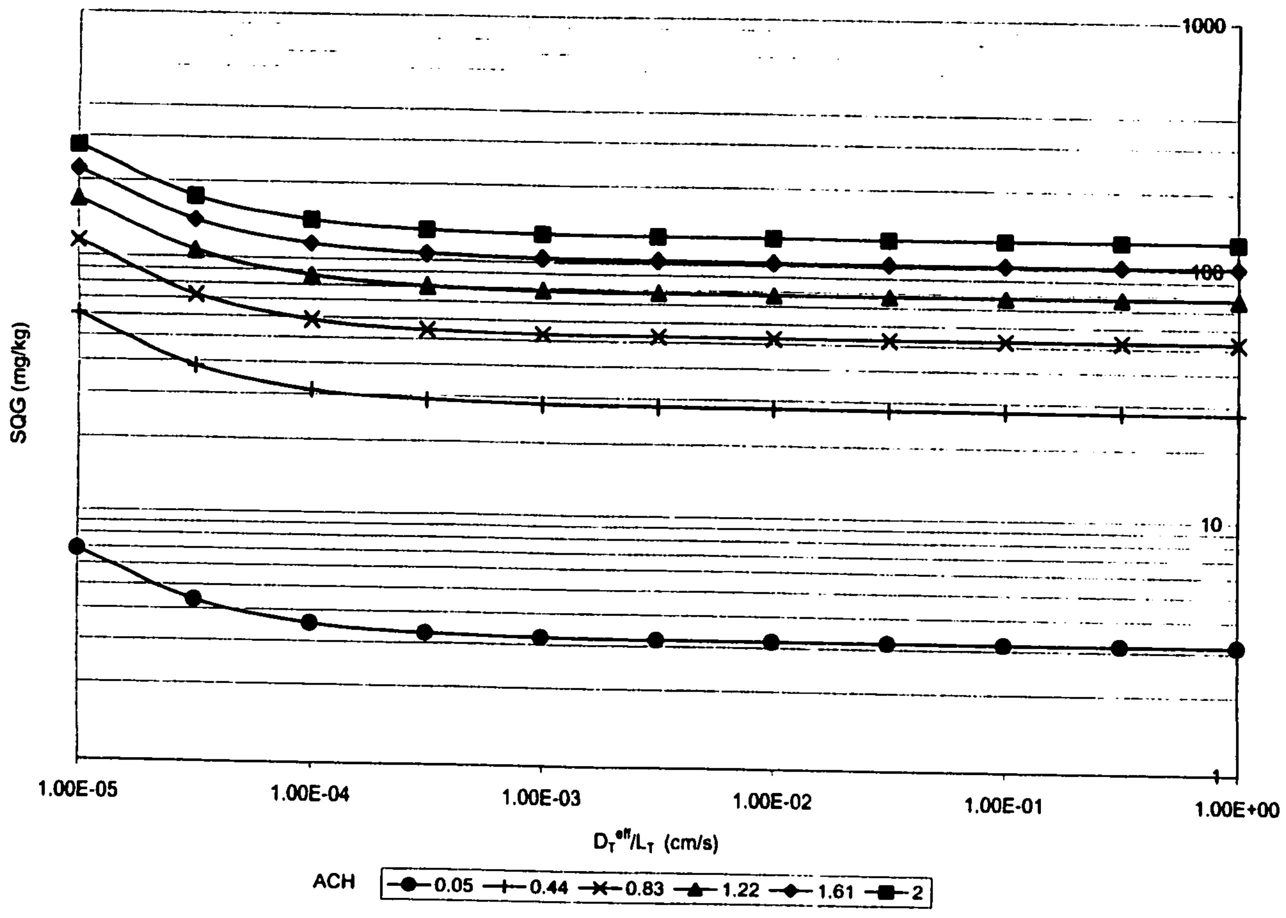

Figure L.5: SQG for m-Xylene in Fine-Grained Soils for Slab on Grade Structures, Commercial/Industrial Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

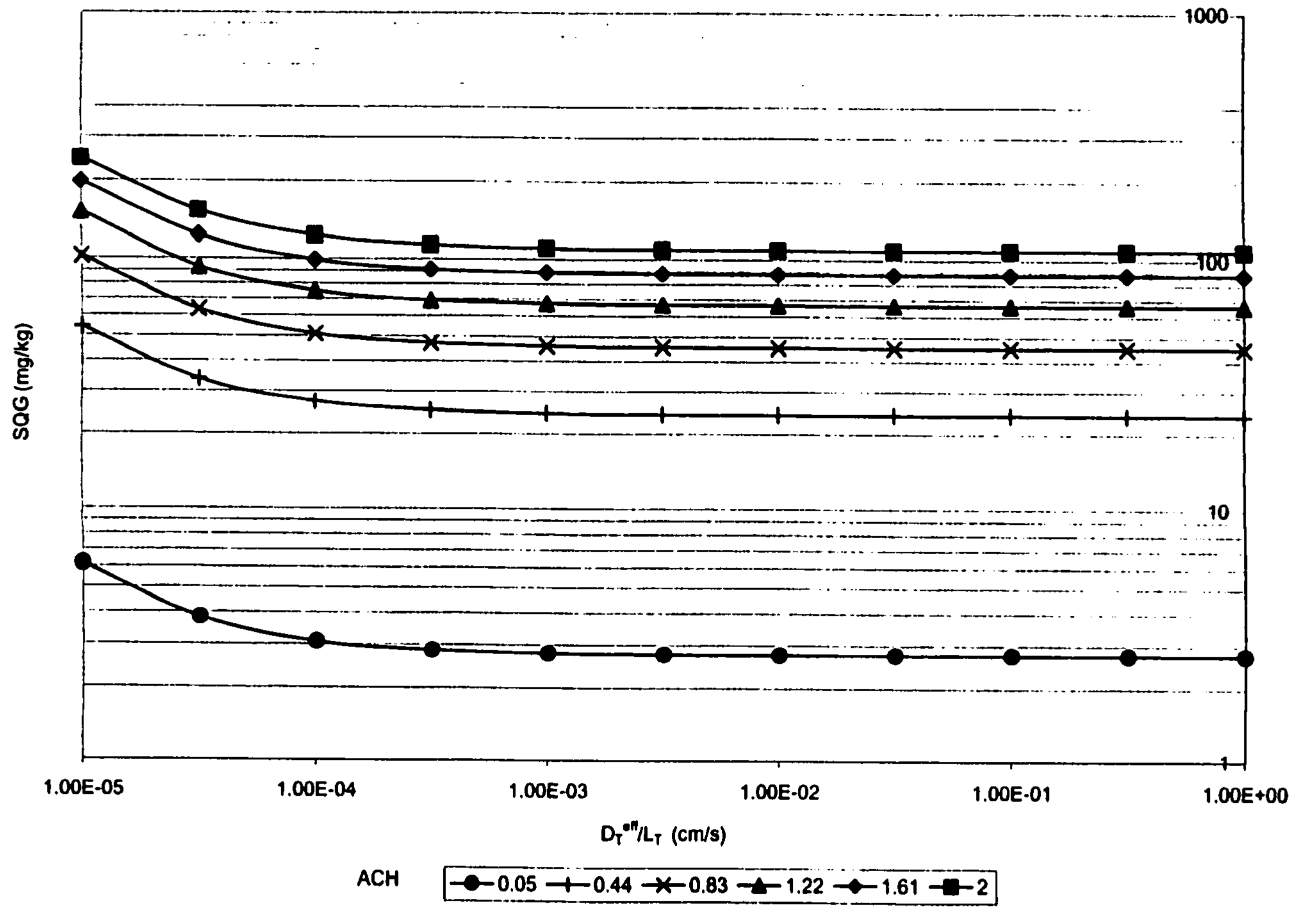

Figure L.6: SQG for p-Xylene in Fine-Grained Soils for Slab on Grade Structures, Commercial/Industrial Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

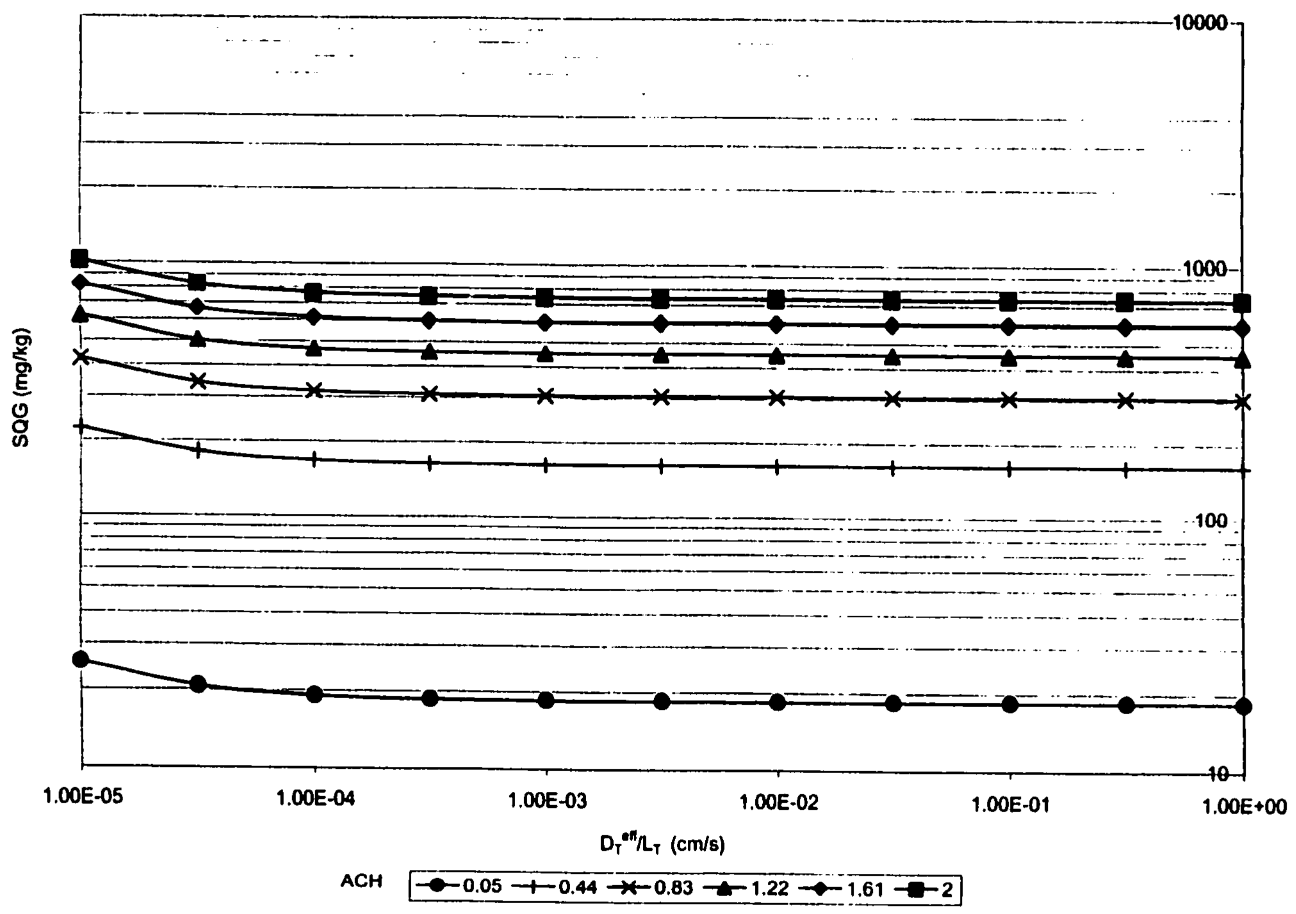

Figure L.7: SQG for C_6–C_8 aliphatic hydrocarbons in Fine-Grained Soils for Slab on Grade Structures, Commercial/Industrial Exposure Scenario vs. $\frac{D_t^{eff}}{L_T}$ for various air exchange rates

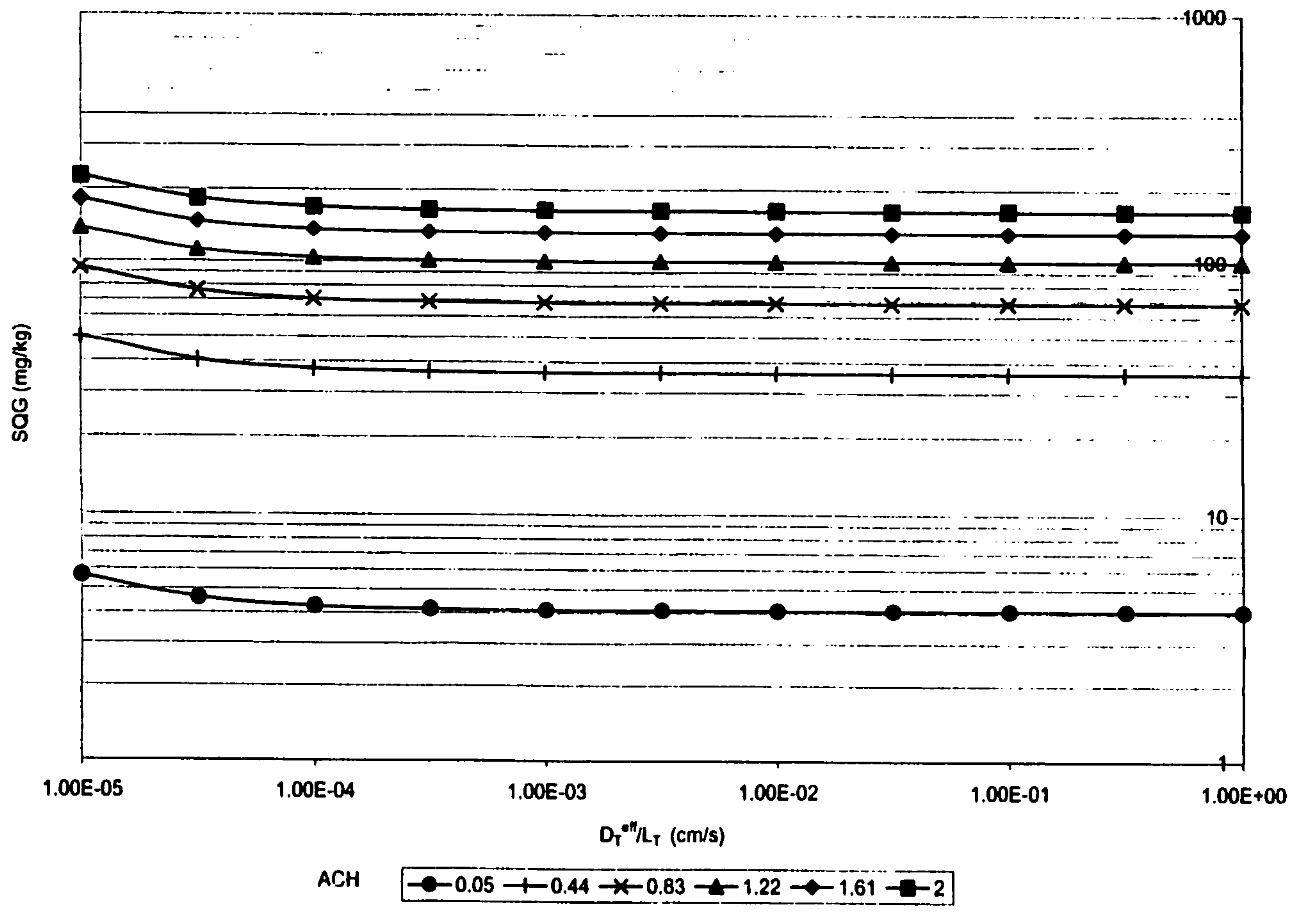

Figure L.8: SQG for $C_{>8}$–C_{10} aliphatic hydrocarbons in Fine-Grained Soils for Slab on Grade Structures, Commercial/Industrial Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

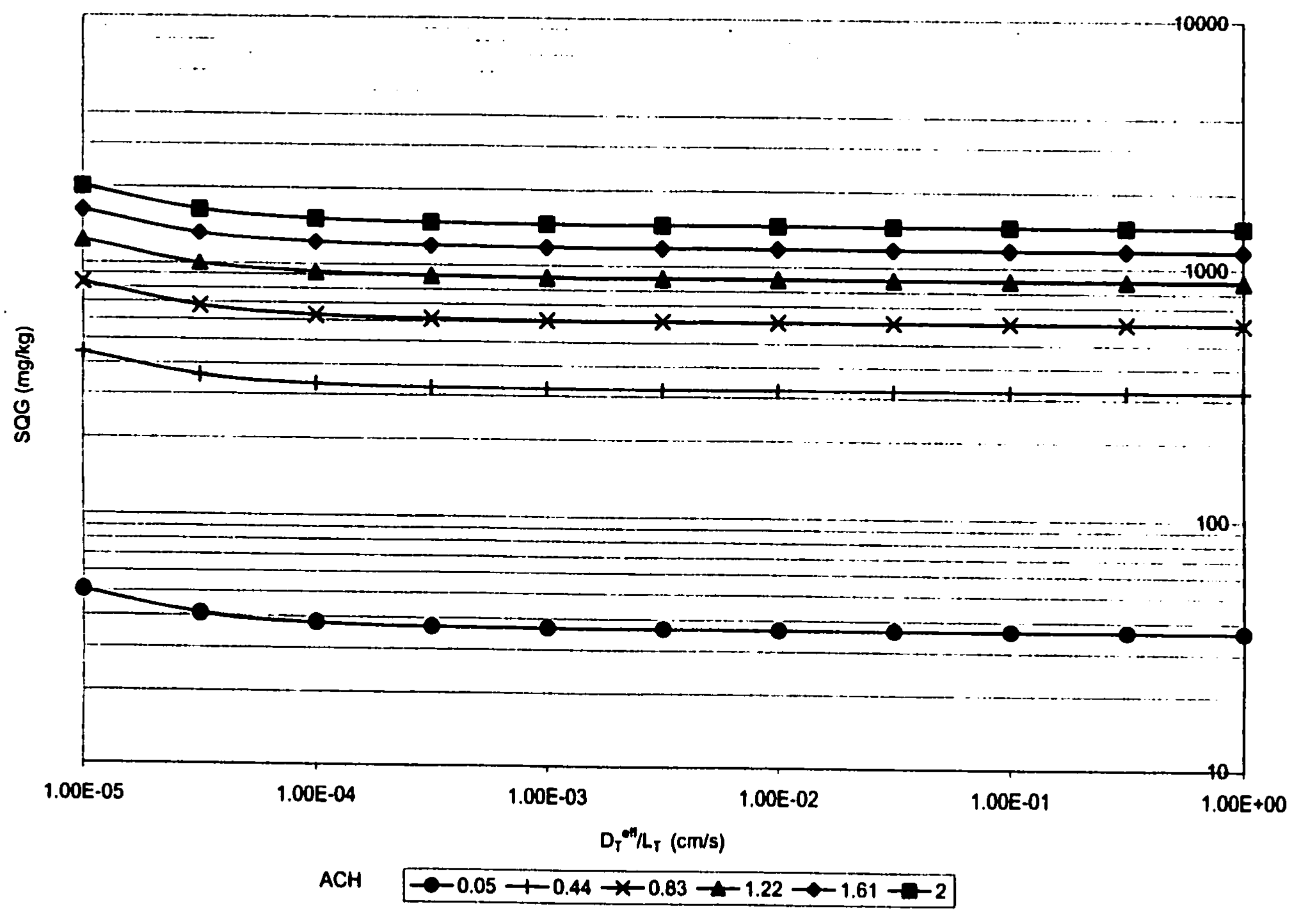

Figure L.9: SQG for $C_{>10}$–C_{12} aliphatic hydrocarbons i in Fine-Grained Soils for Slab on Grade Structures, Commercial/Industrial Exposure Scenario vs. $\frac{D_i^{eff}}{L_T}$ for various air exchange rates

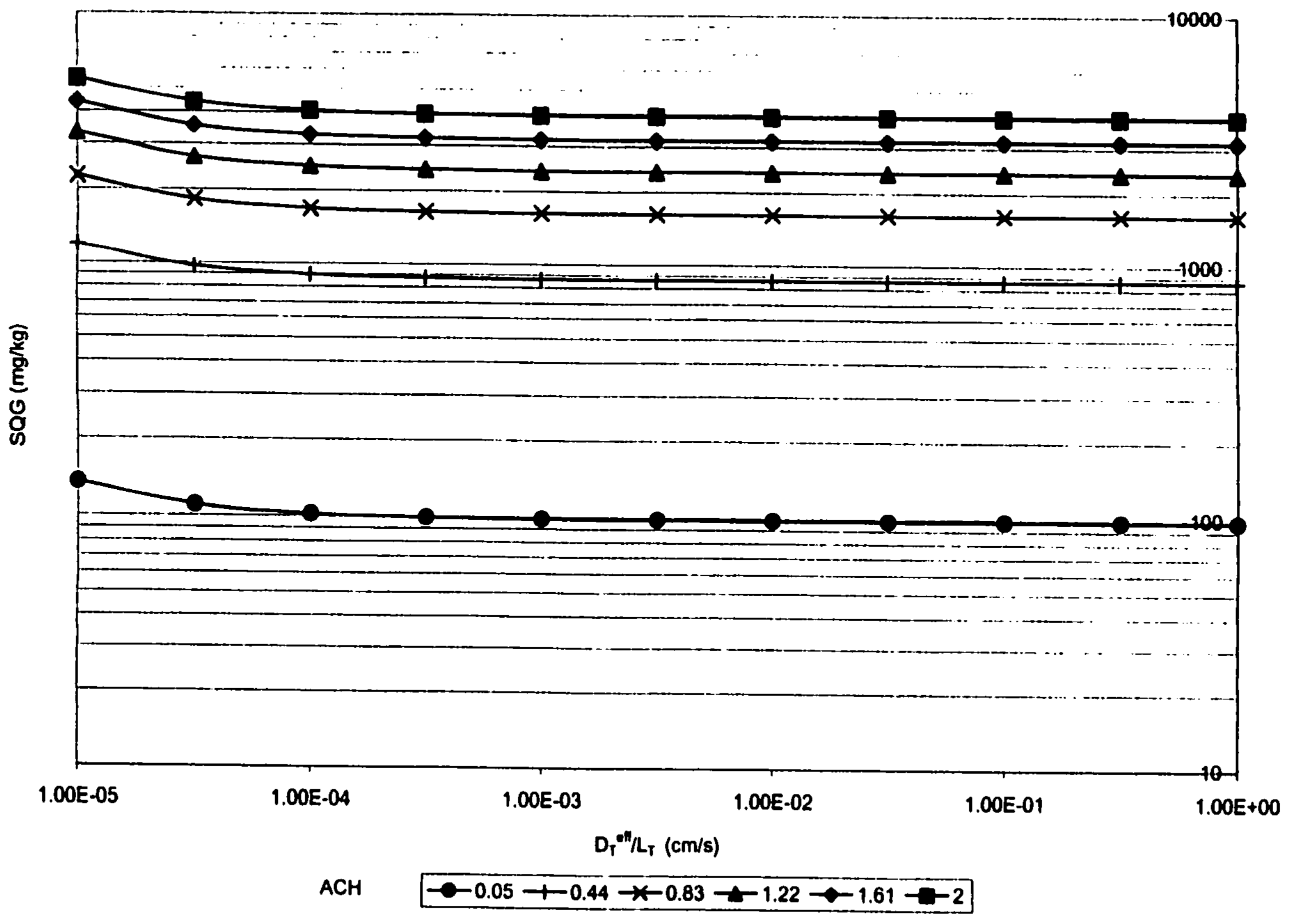

Figure L.10: SQG for $C_{>12}$–C_{16} aliphatic hydrocarbons in Fine-Grained Soils for Slab on Grade Structures, Commercial/Industrial Exposure Scenario vs. $\frac{D_t^{eff}}{L_T}$ for various air exchange rates

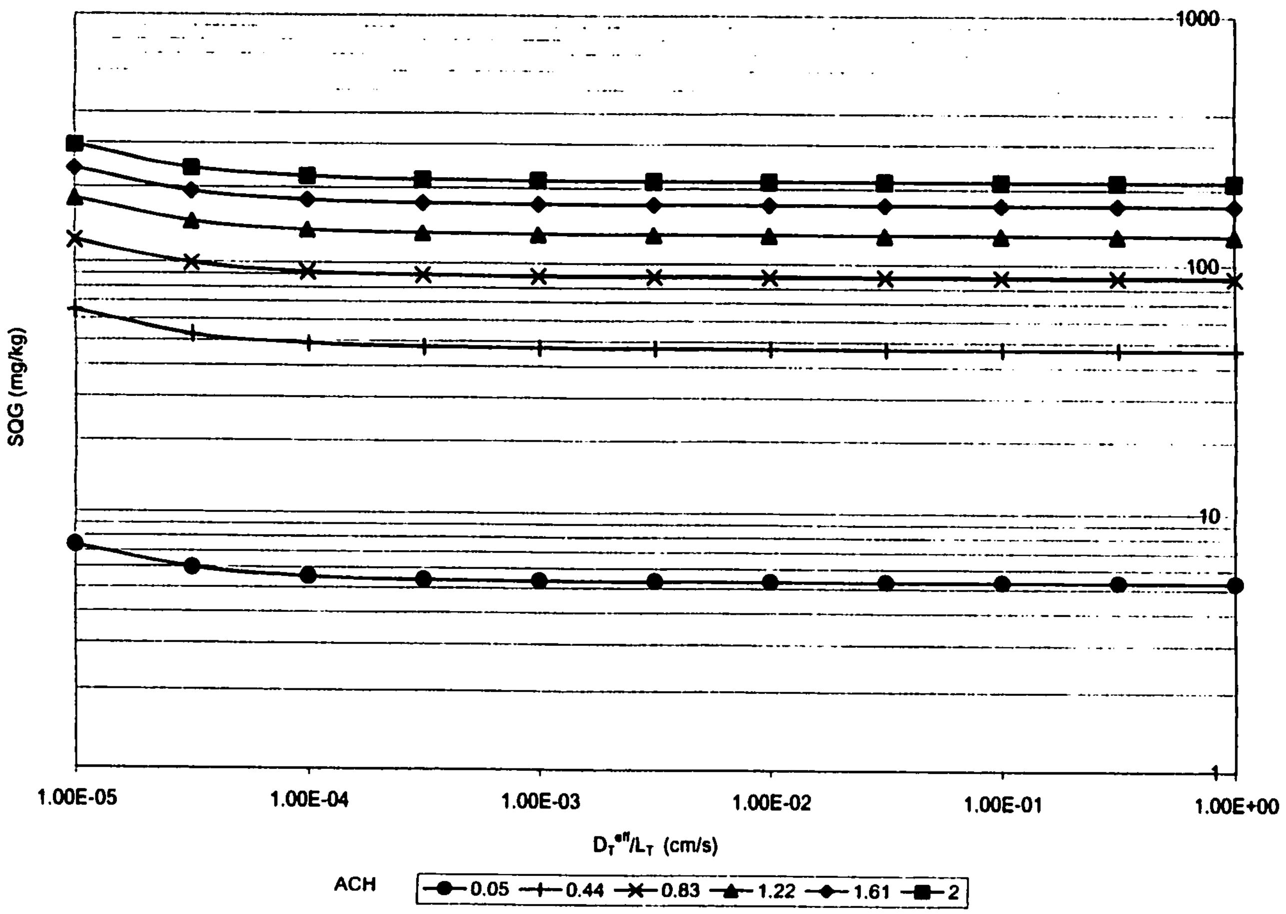

Figure L.11: SQG for $C_{>6}$–C_{10} aromatic hydrocarbons in Fine-Grained Soils for Slab on Grade Structures, Commercial/Industrial Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

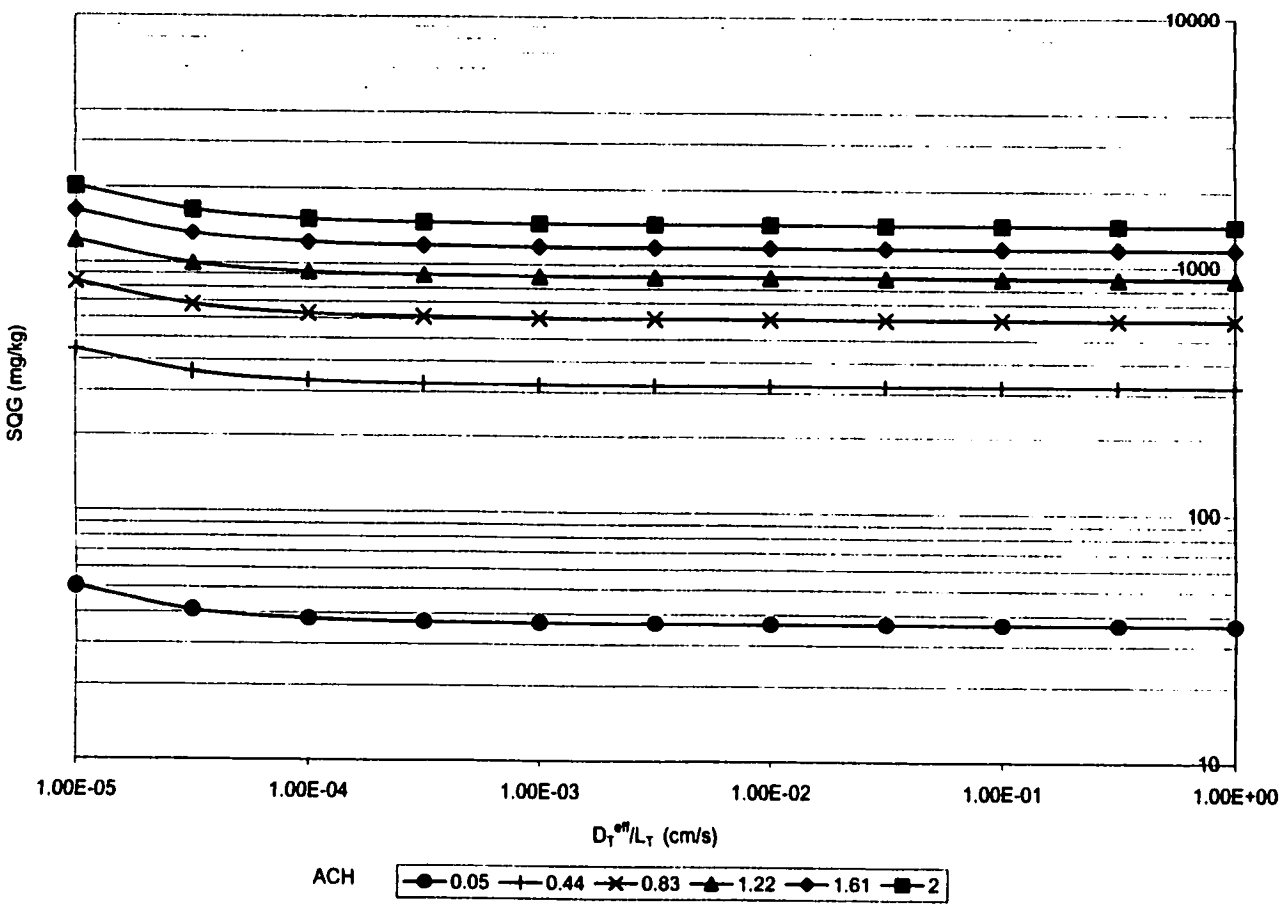

Figure L.12: SQG for $C_{>10}$–C_{12} aromatic hydrocarbons in Fine-Grained Soils for Slab on Grade Structures, Commercial/Industrial Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

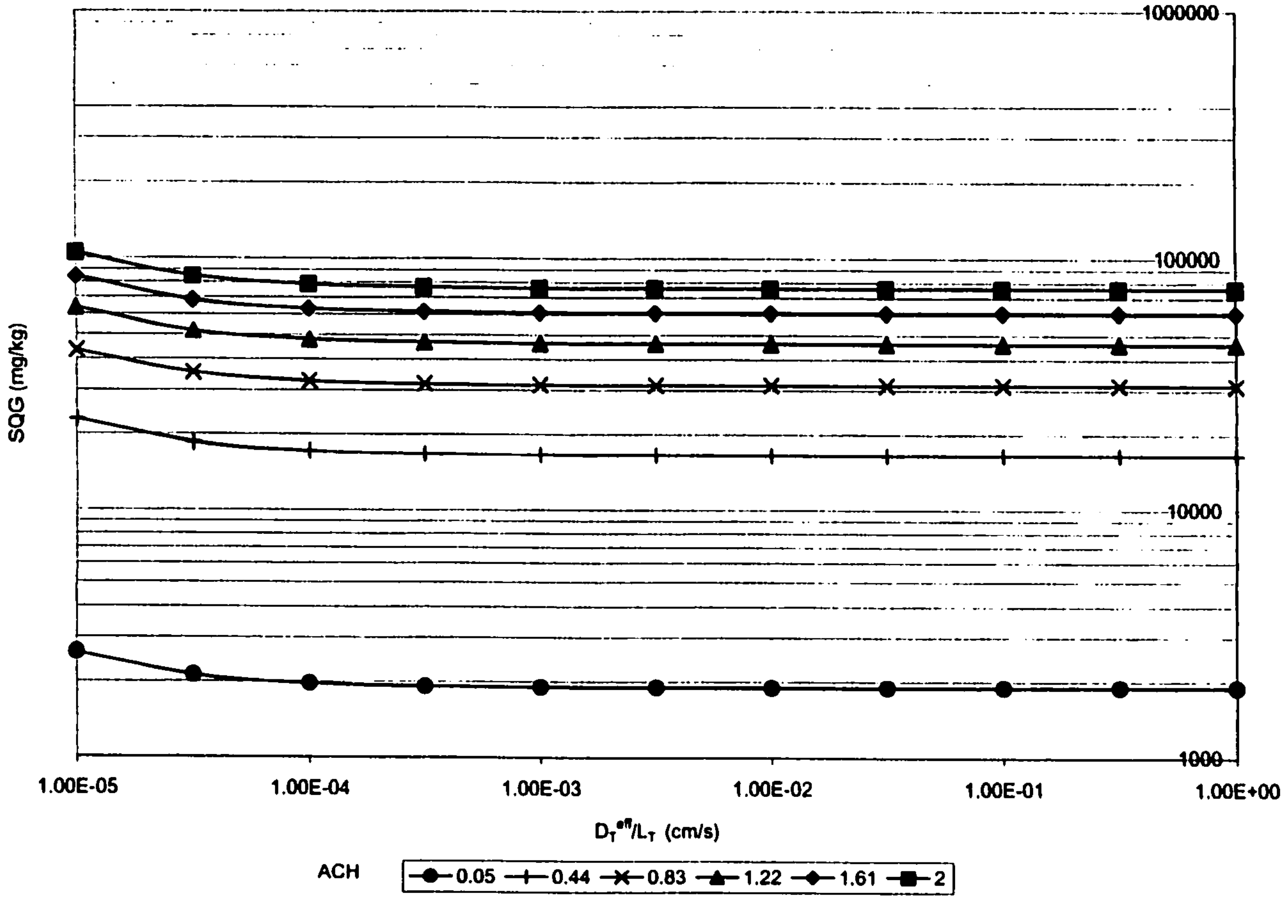

Figure L.13: SQG for $C_{>12}$–C_{16} aromatic hydrocarbons in Fine-Grained Soils for Slab on Grade Structures, Commercial/Industrial Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

Appendix M

Groundwater Quality Guidelines for Fine-Grained Soils for Buildings with Basements, Agricultural/Residential Exposure Scenarios

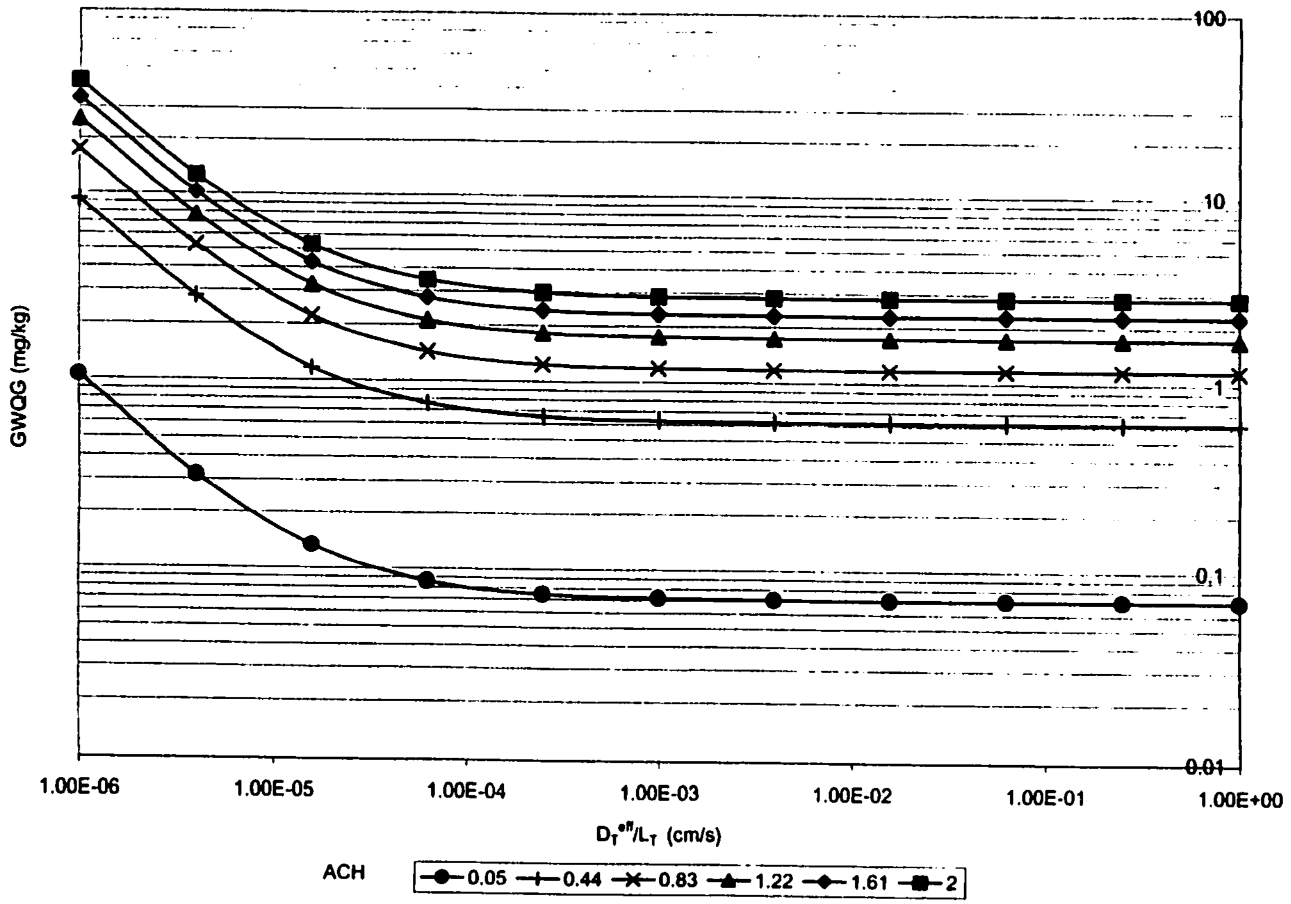

Figure M.1: GWQG for Benzene in Fine-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

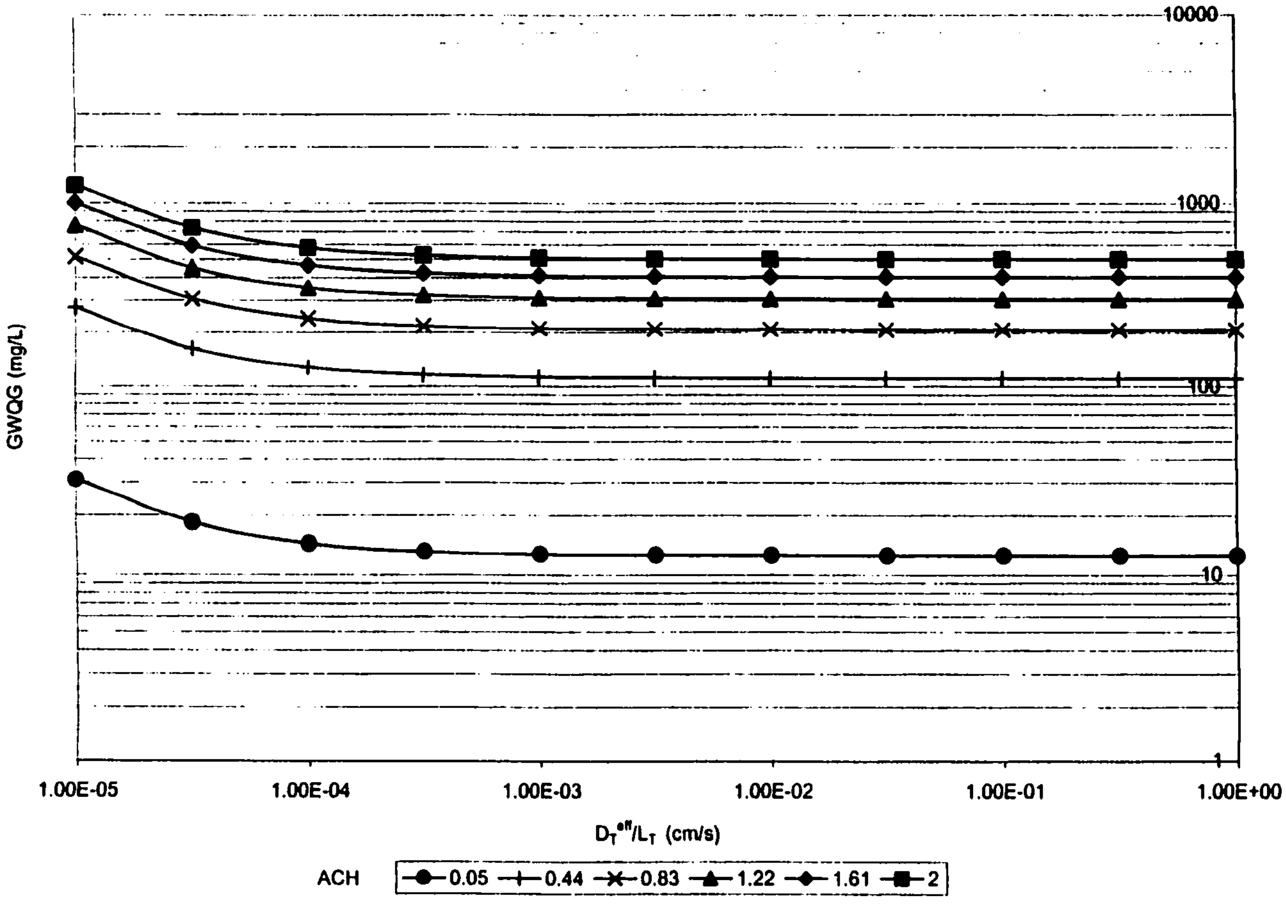

Figure M.2: GWQG for Toluene in Fine-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

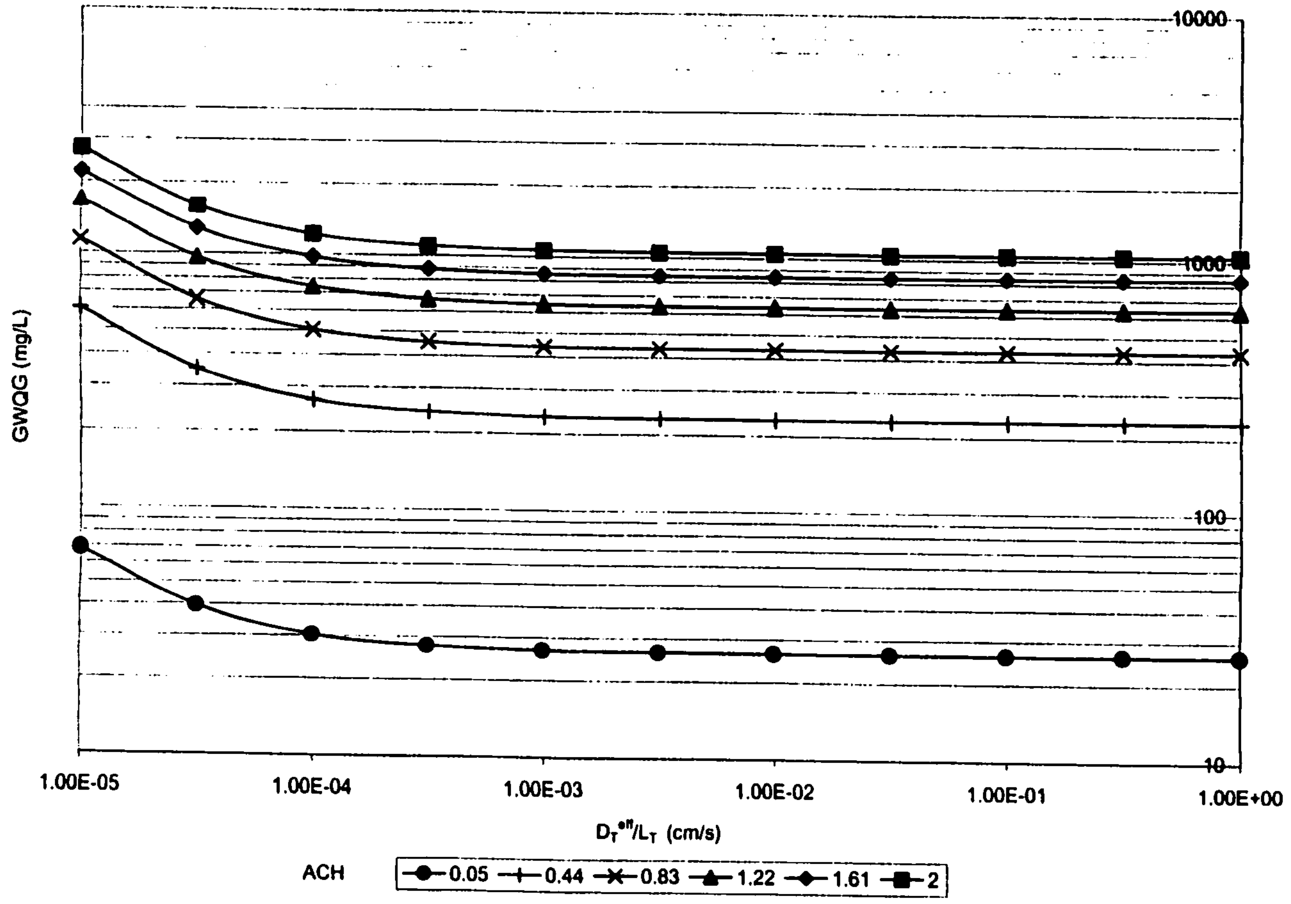

Figure M.3: GWQG for Ethylbenzene in Fine-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_t^{eff}}{L_T}$ for various air exchange rates

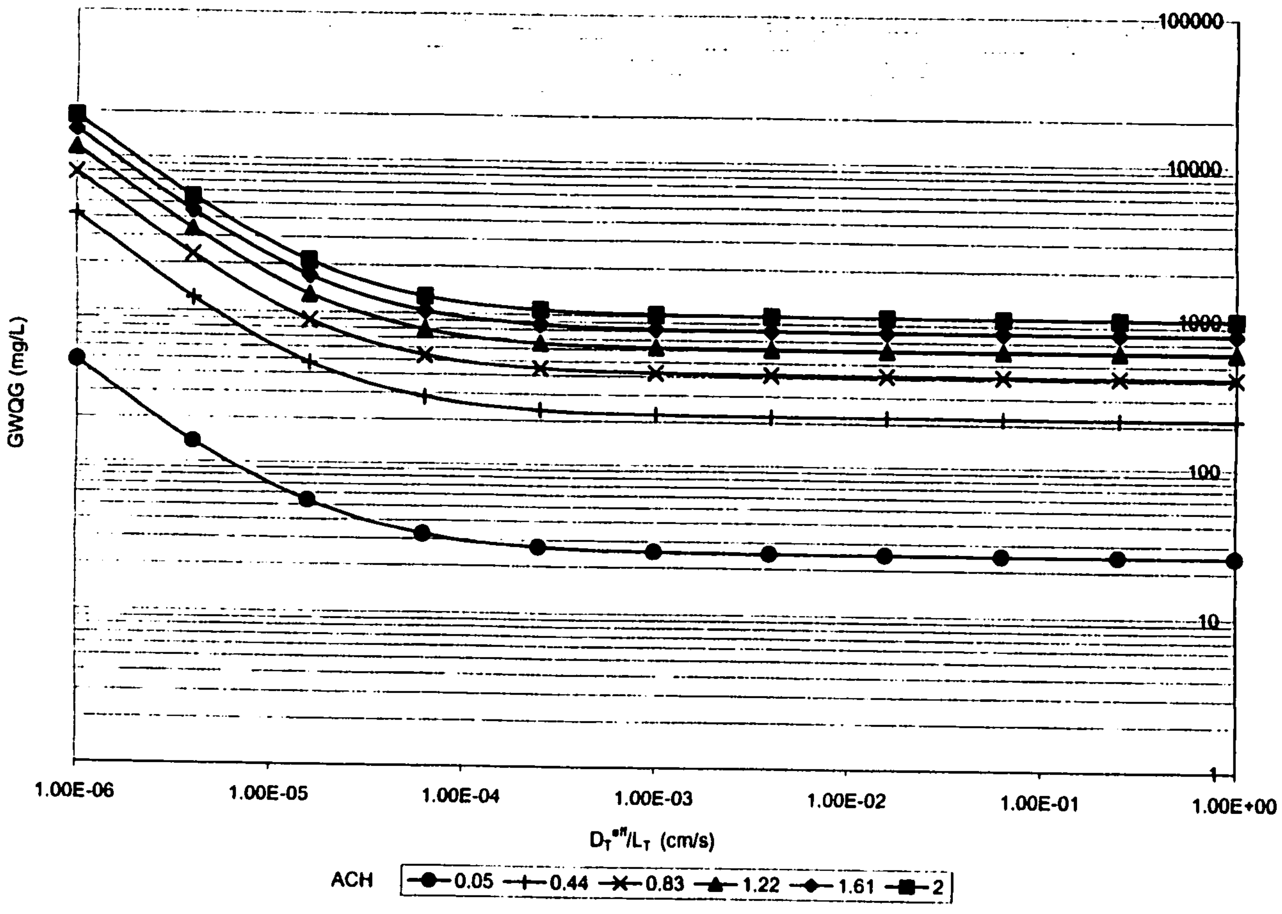

Figure M.4: GWQG for o-Xylene iin Fine-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

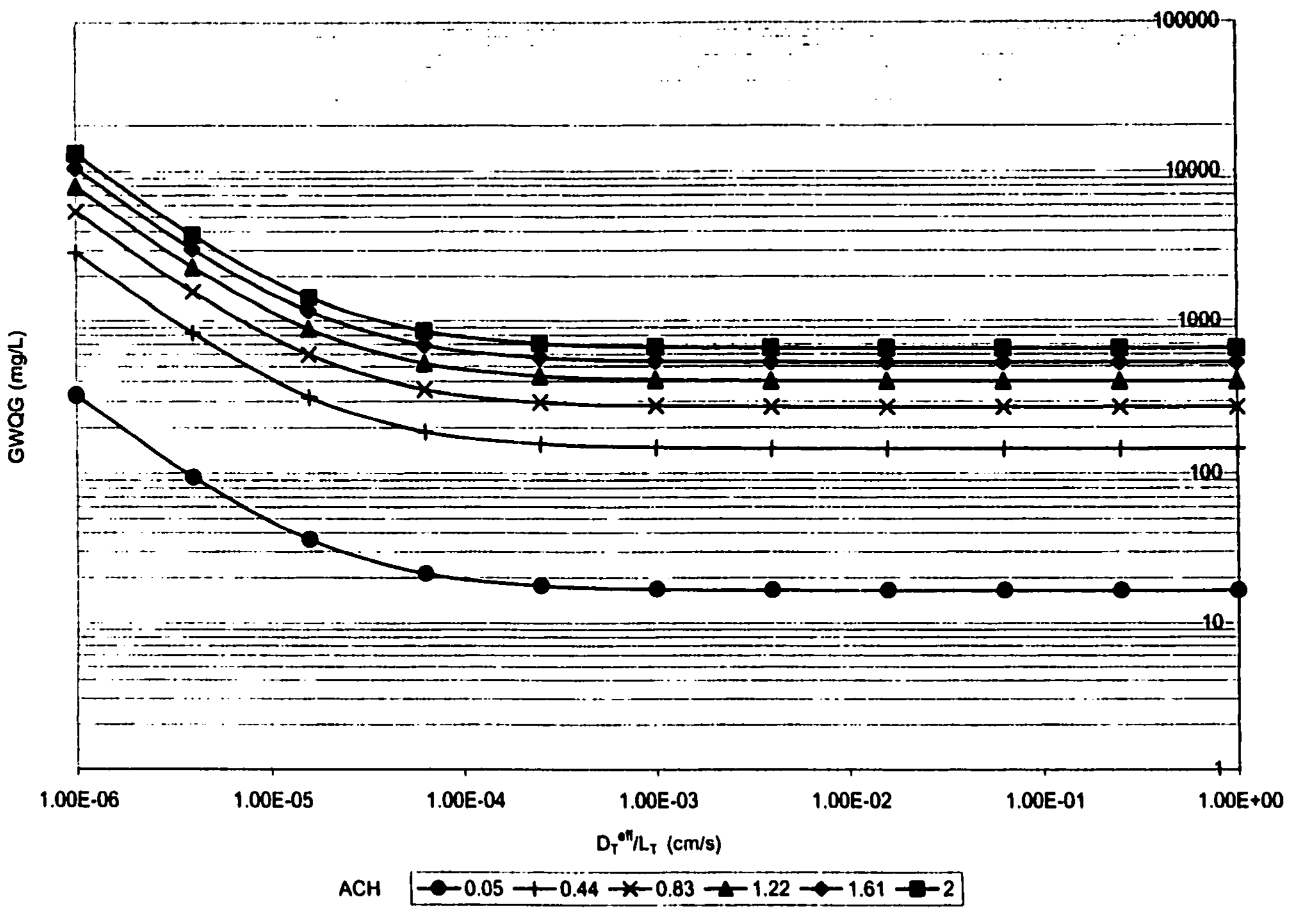

Figure M.5: GWQG for m-Xylene in Fine-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

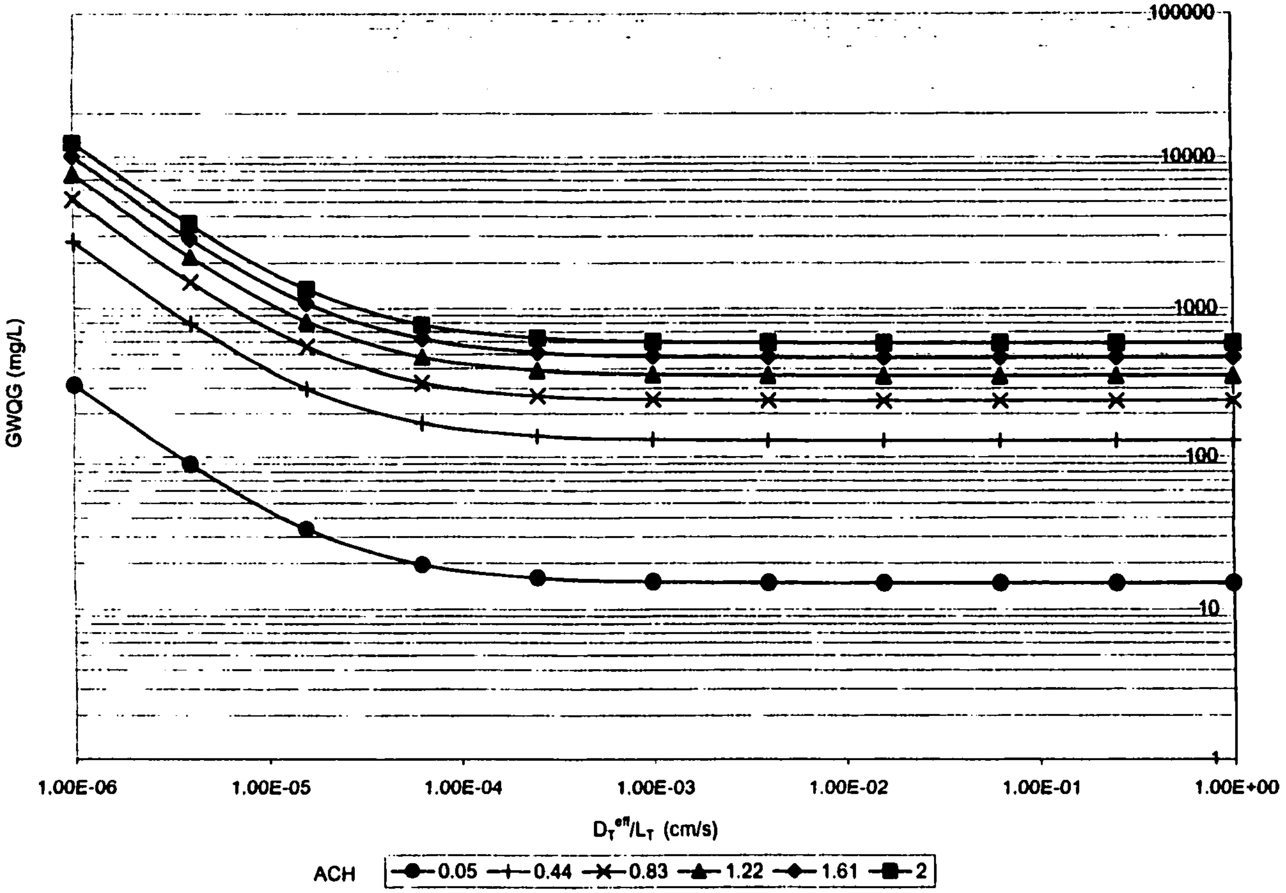

Figure M.6: GWQG for p-Xylene in Fine-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

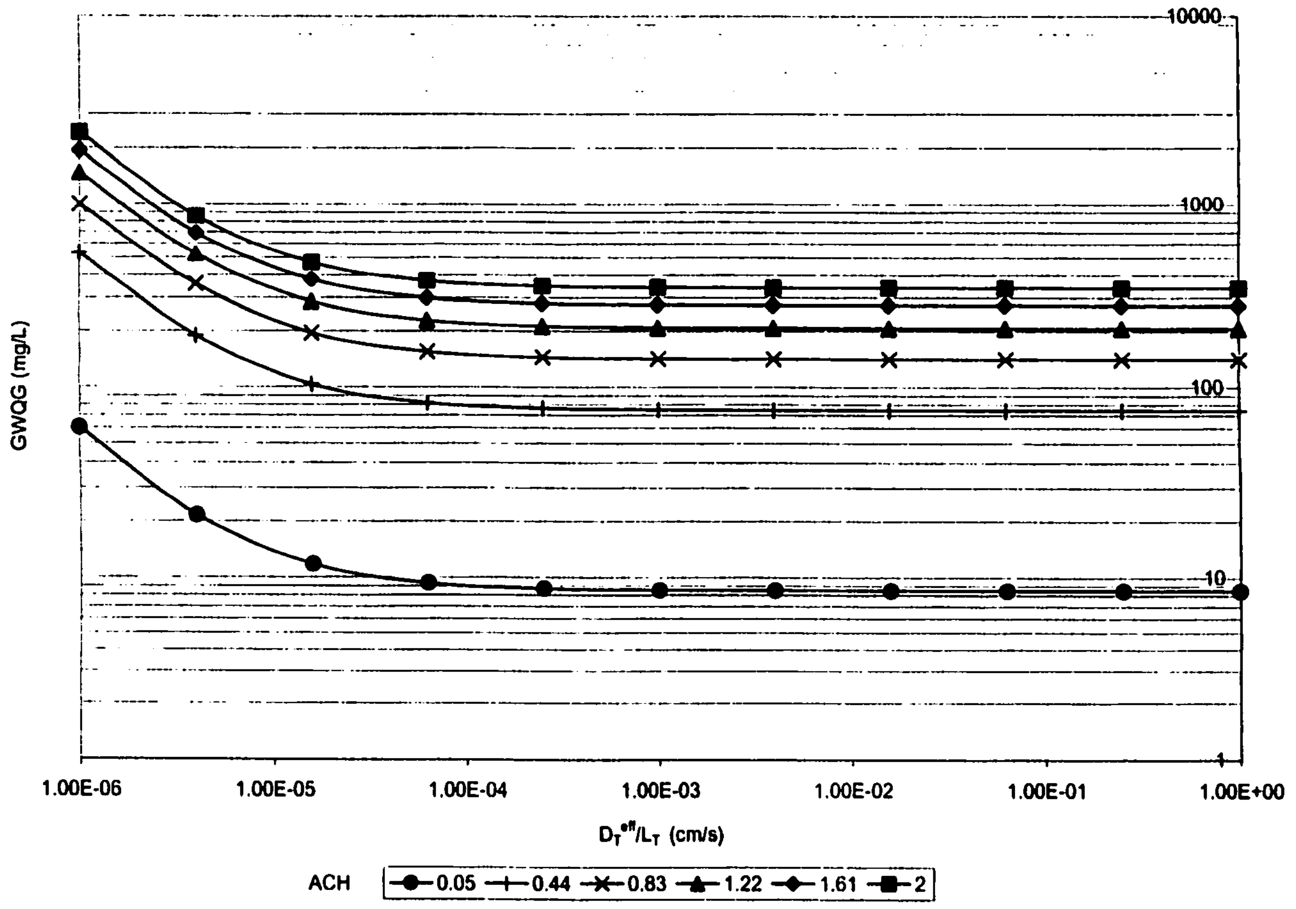

Figure M.7: GWQG for C_6–C_8 aliphatic hydrocarbons in Fine-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

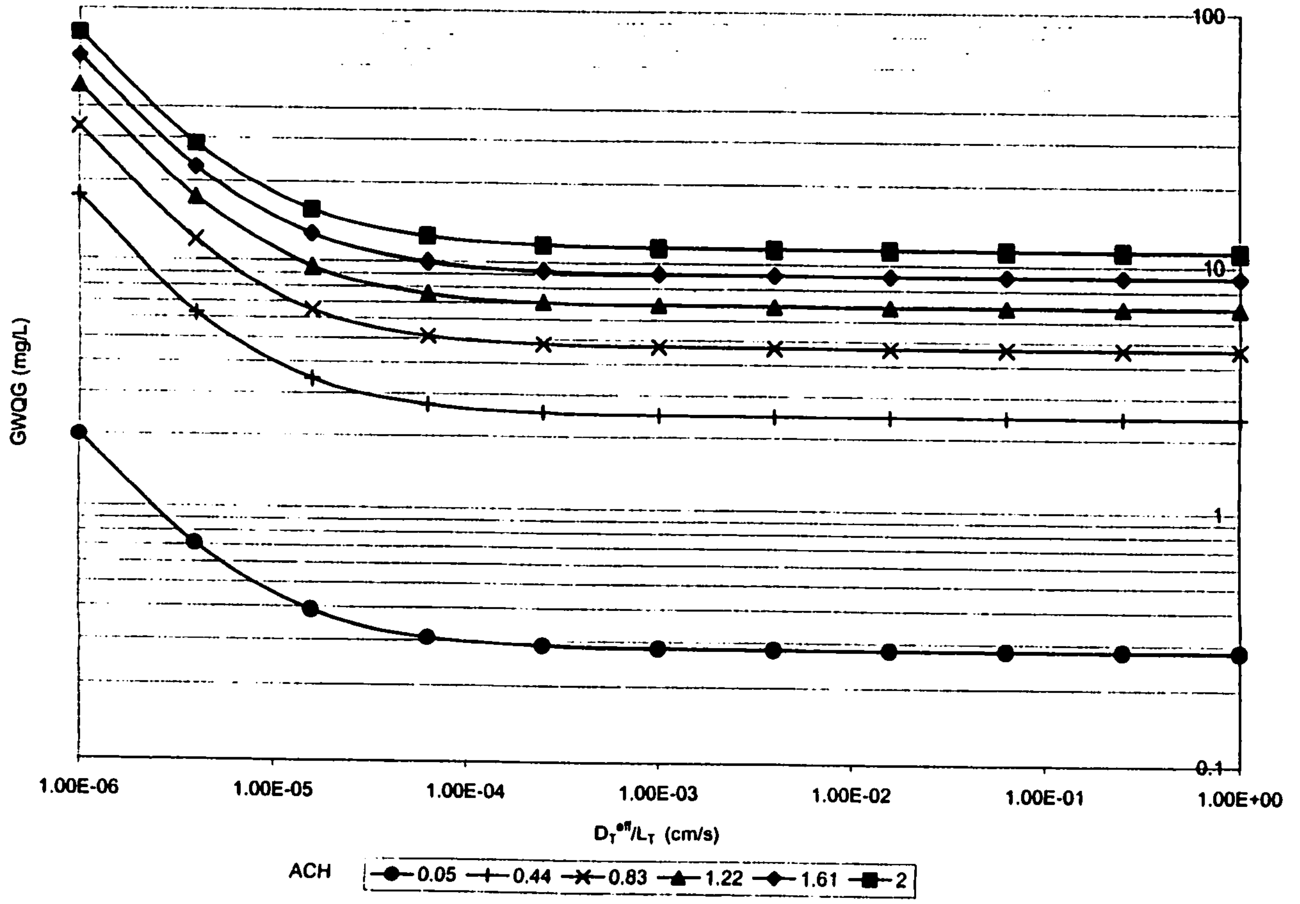

Figure M.8: GWQG for $C_{>8}$–C_{10} aliphatic hydrocarbons in Fine-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

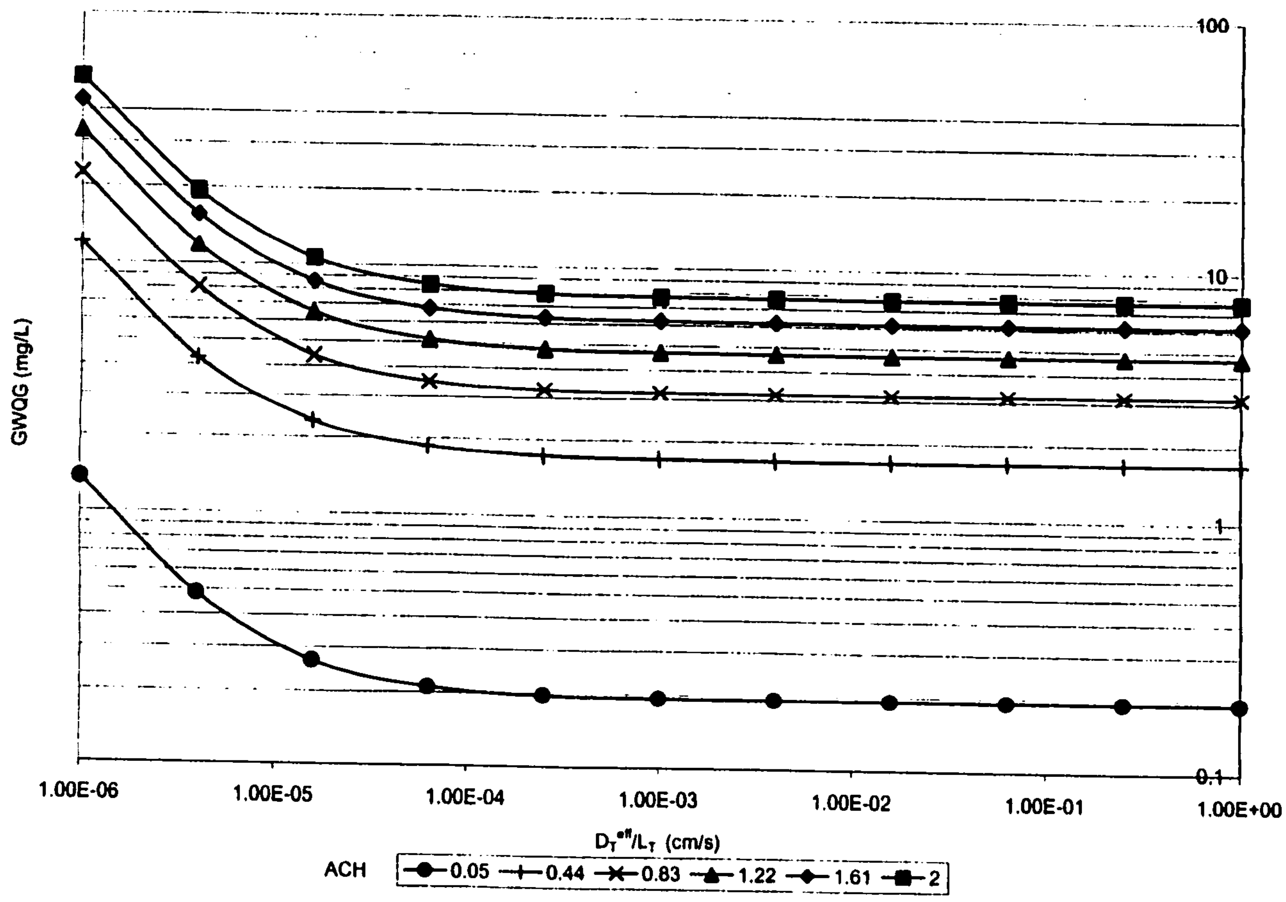

Figure M.9: GWQG for $C_{>10}$-C_{12} aliphatic hydrocarbons in Fine-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

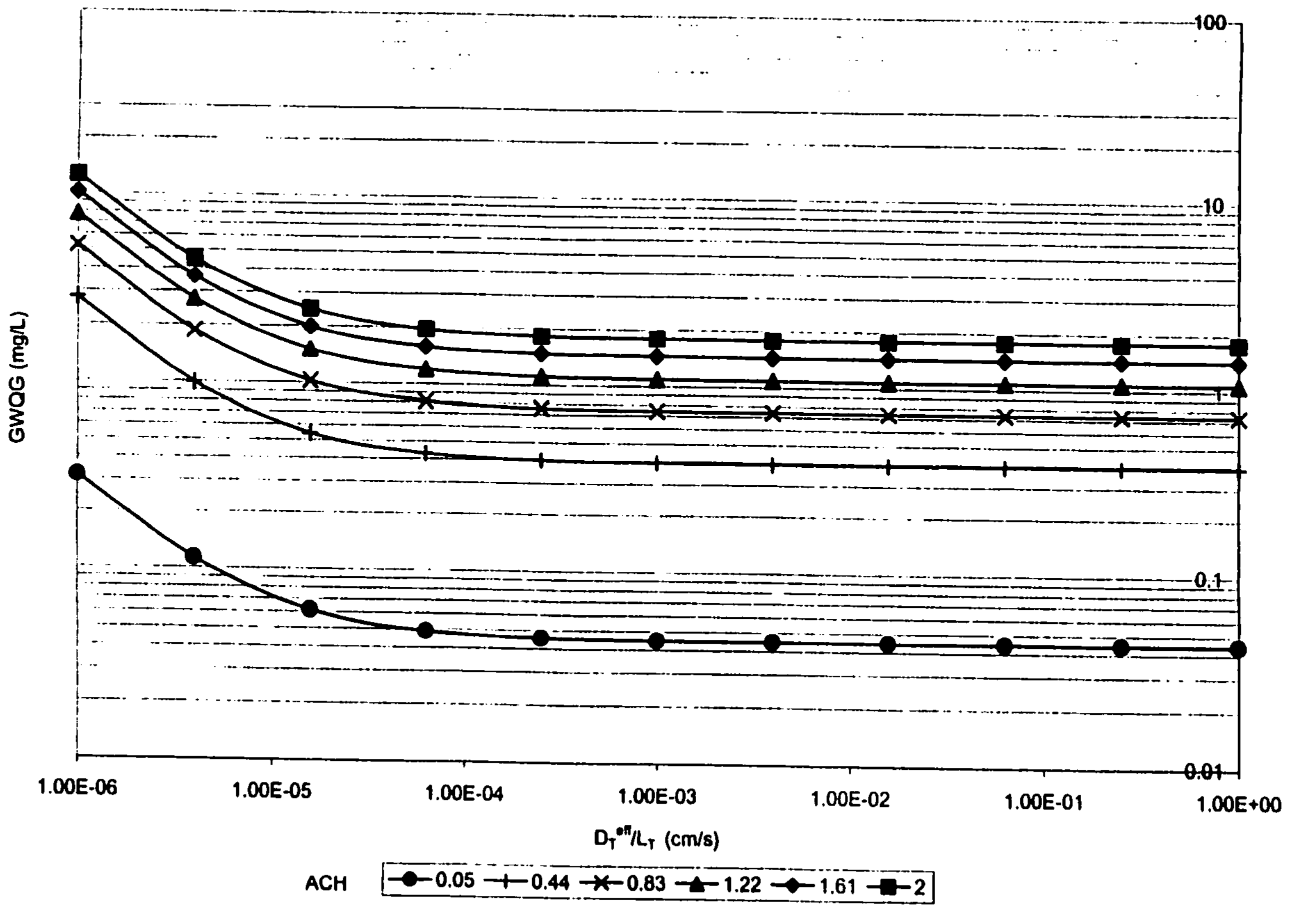

Figure M.10: GWQG for $C_{>12}$–C_{16} aliphatic hydrocarbons in Fine-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_t^{eff}}{L_T}$ for various air exchange rates

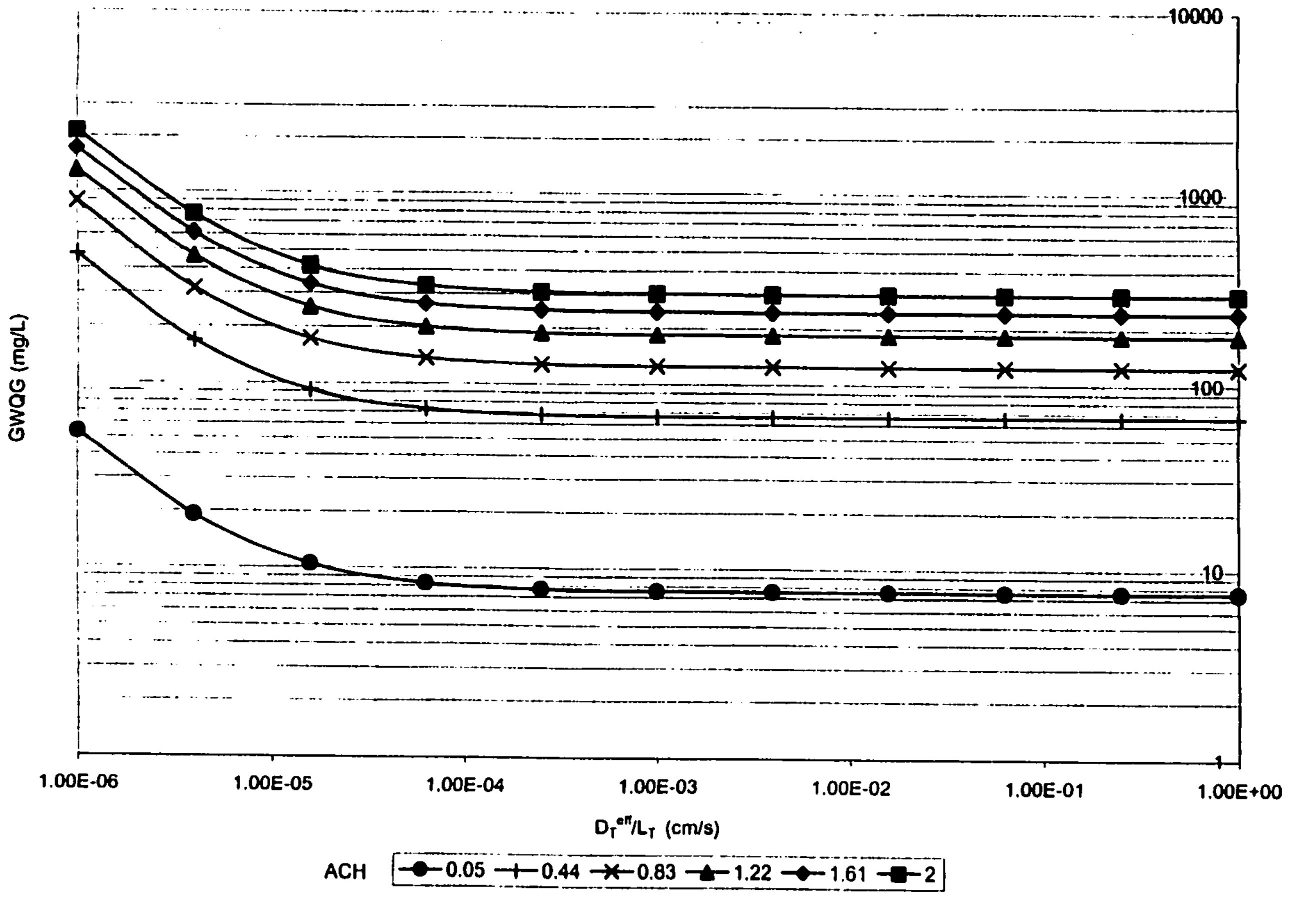

Figure M.11: GWQG for $C_{>8}$-C_{10} aromatic hydrocarbons in Fine-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

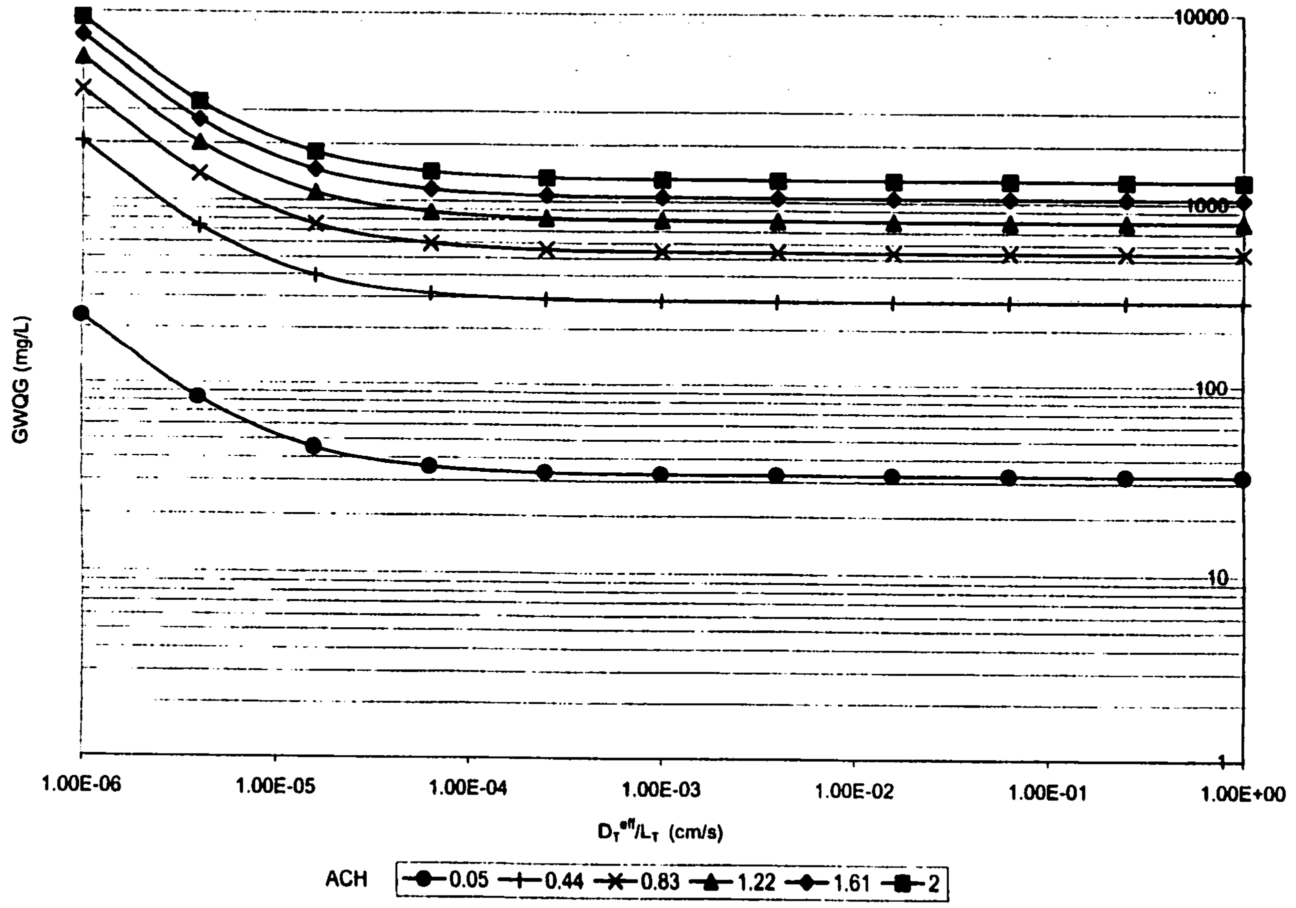

Figure M.12: GWQG for $C_{>10}$-C_{12} aromatic hydrocarbons in Fine-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

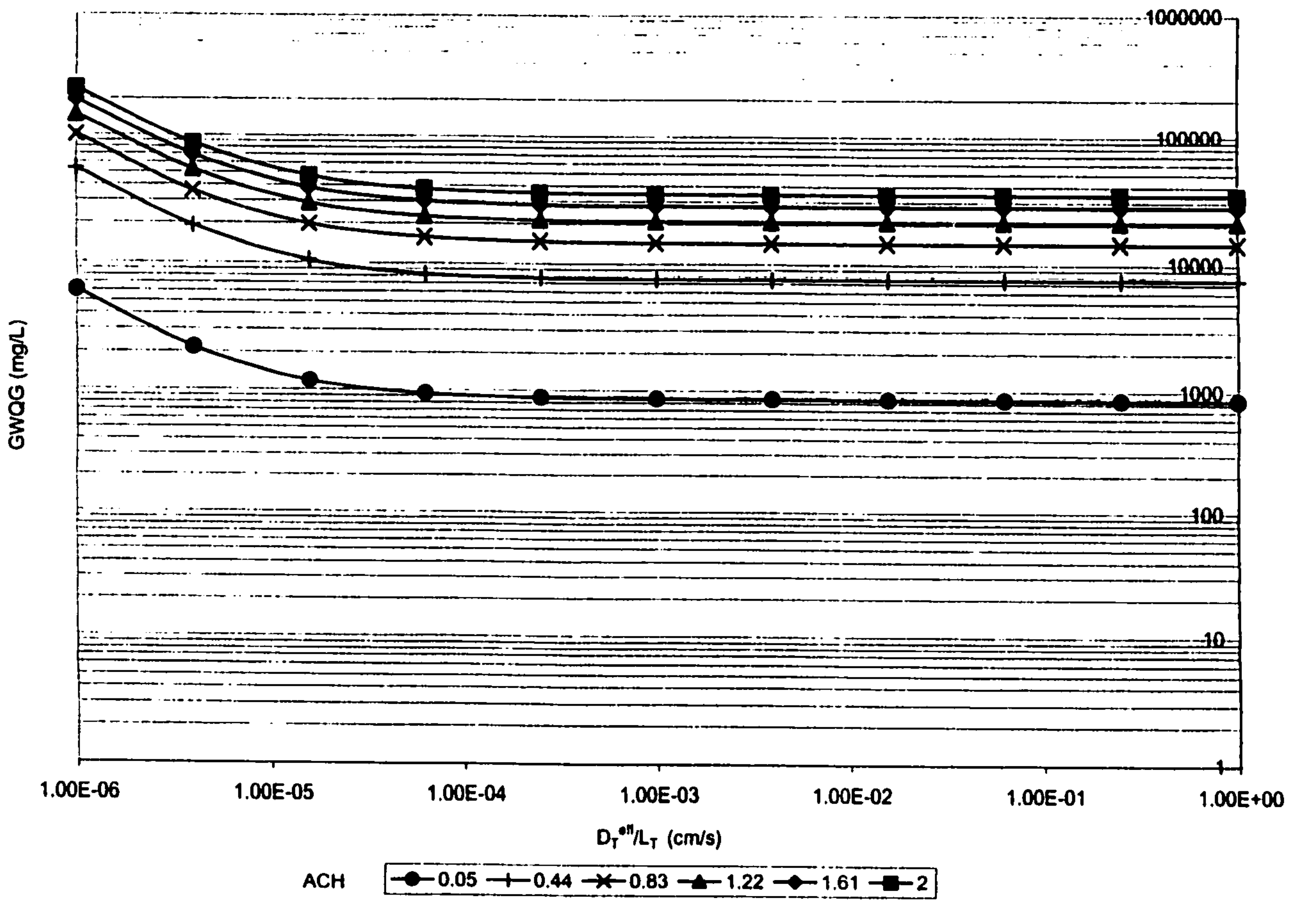

Figure M.13: GWQG for $C_{>12}$–C_{16} aromatic hydrocarbons in Fine-Grained Soils for a Building with a Basement, Agricultural/Residential Exposure Scenario vs. $\frac{D_t^{eff}}{L_T}$ for various air exchange rates

Appendix N

Groundwater Quality Guidelines for Fine-Grained Soils for Slab on Grade Structures, Commercial/Industrial Exposure Scenarios

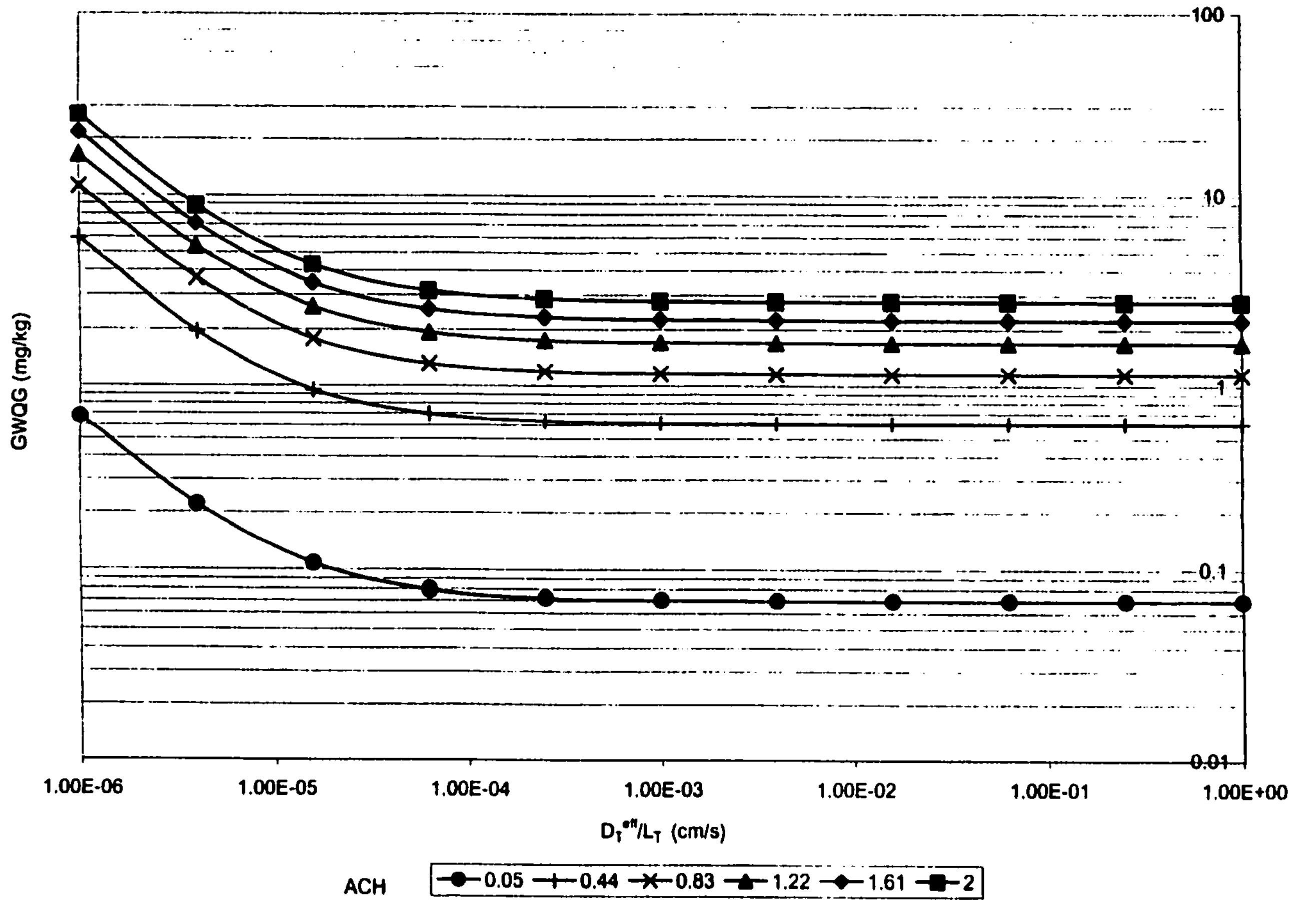

Figure N.1: GWQG for Benzene in Fine-Grained Soils for a Slab on Grade Structure, Commercial/Industrial Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

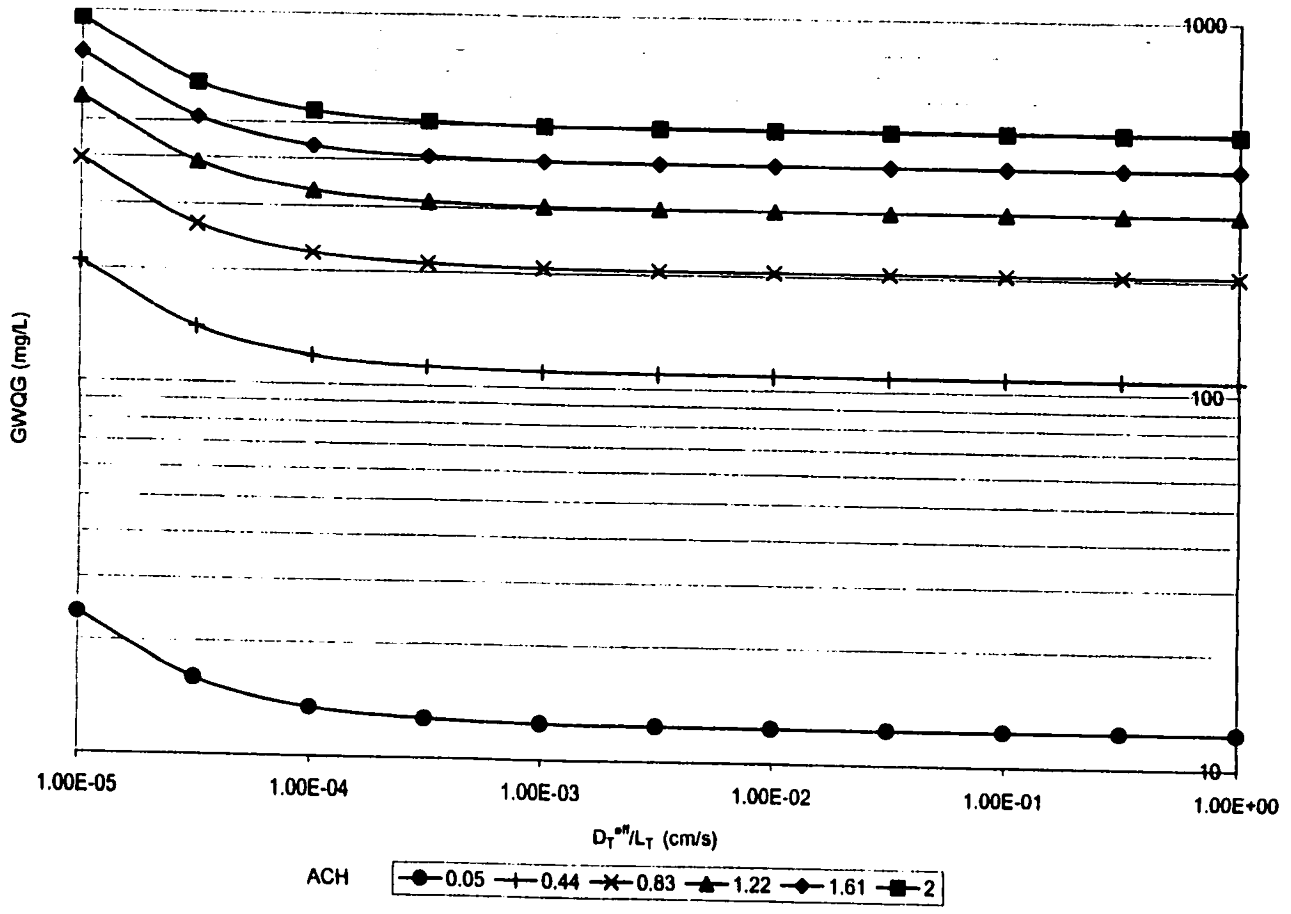

Figure N.2: GWQG for Toluene in Fine-Grained Soils for a Slab on Grade Structure, Commercial/Industrial Exposure Scenario vs. $\frac{D_t^{eff}}{L_T}$ for various air exchange rates

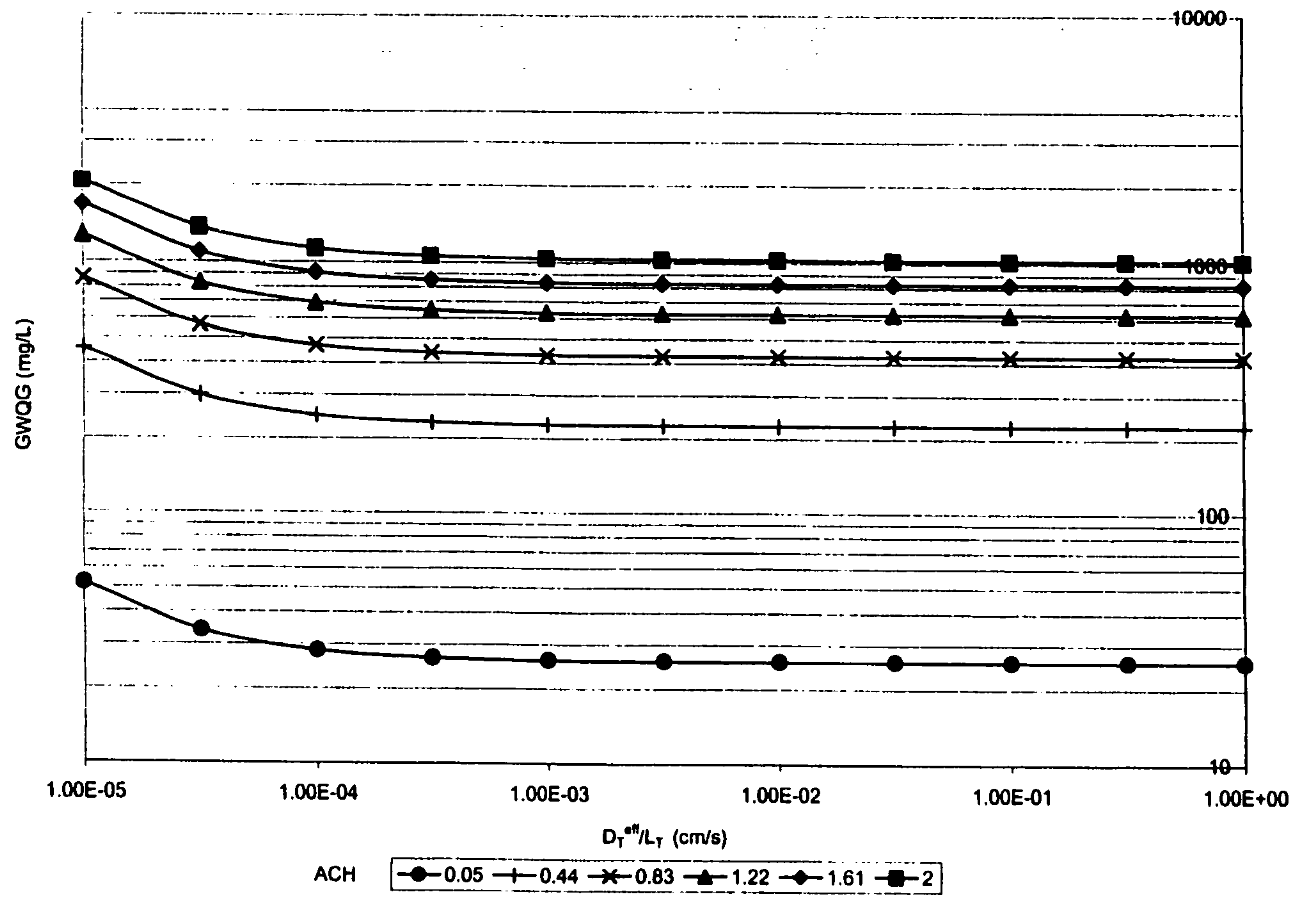

Figure N.3: GWQG for Ethylbenzene in Fine-Grained Soils for a Slab on Grade Structure, Commercial/Industrial Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

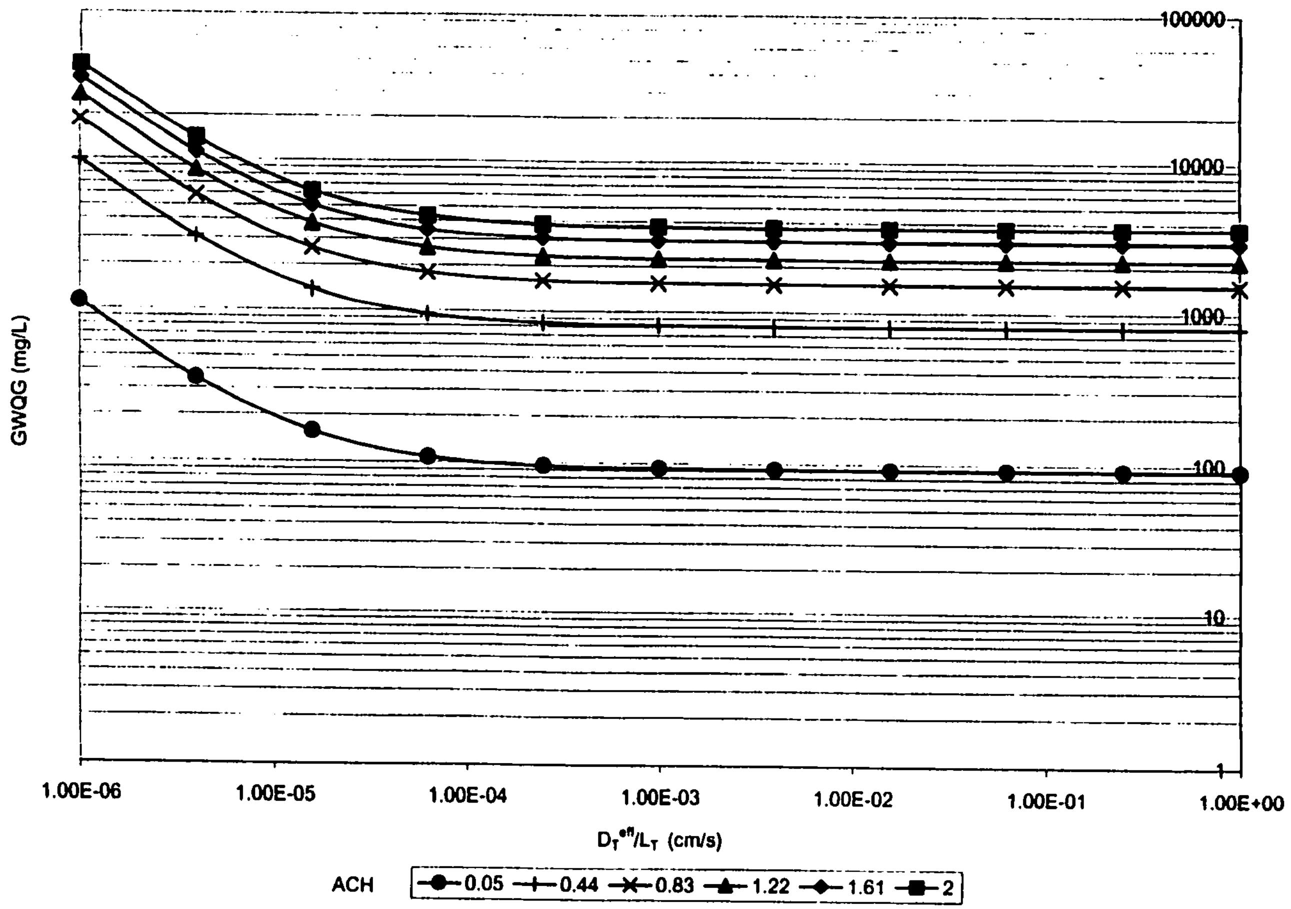

Figure N.4: GWQG for o-Xylene in Fine-Grained Soils for a Slab on Grade Structure, Commercial/Industrial Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

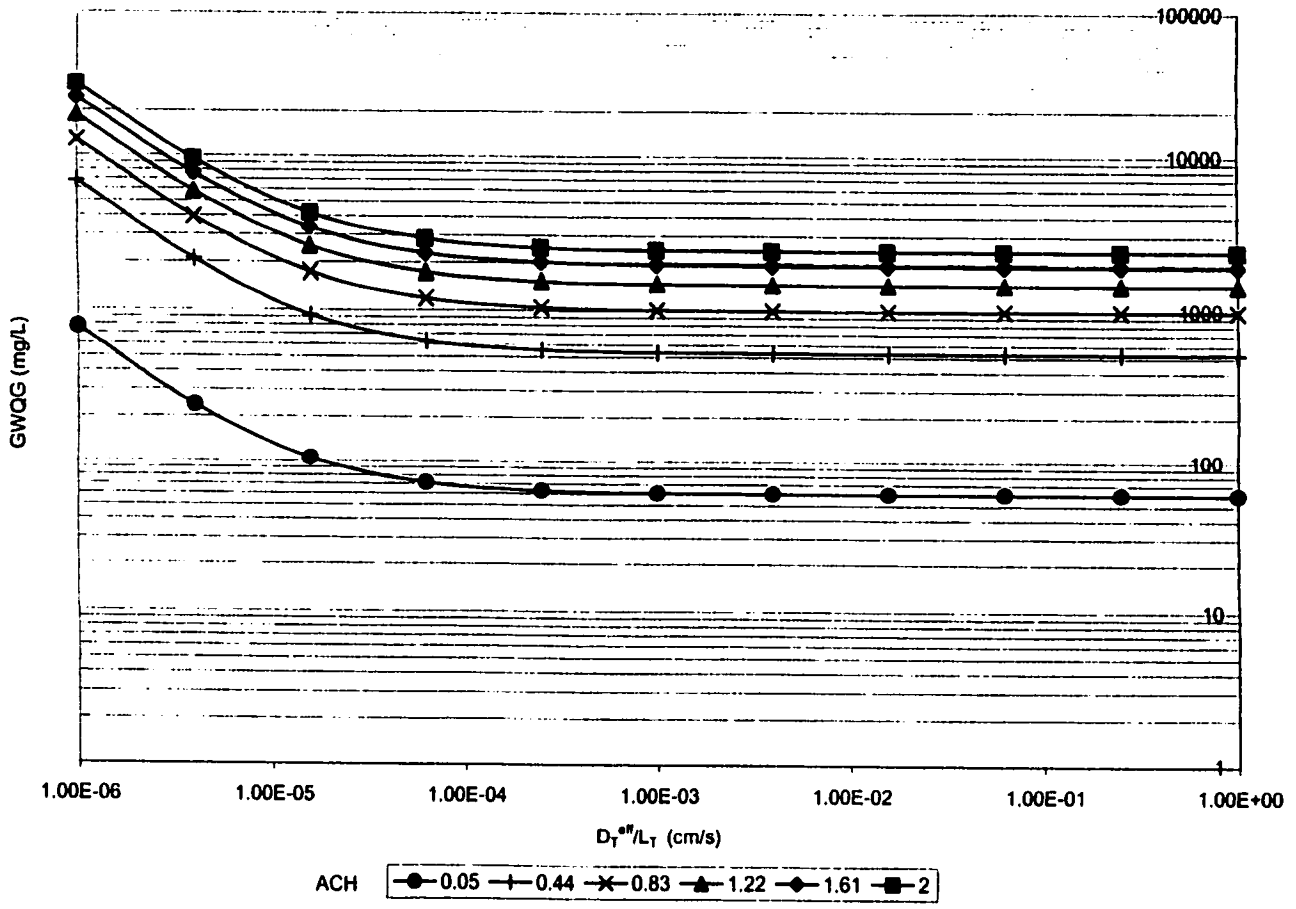

Figure N.5: GWQG for m-Xylene in Fine-Grained Soils for a Slab on Grade Structure, Commercial/Industrial Exposure Scenario vs. $\frac{D_t^{eff}}{L_T}$ for various air exchange rates

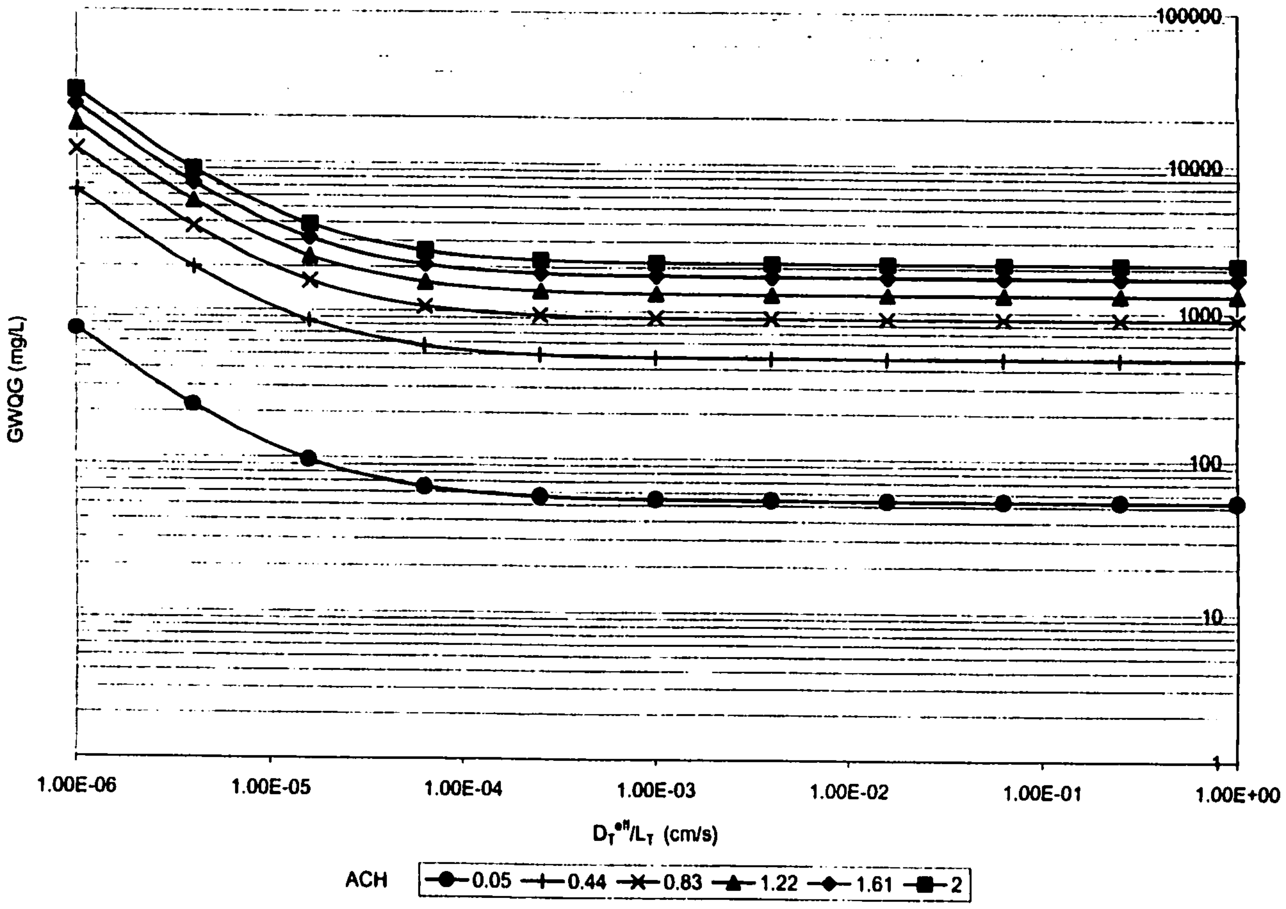

Figure N.6: GWQG for p-Xylene in Fine-Grained Soils for a Slab on Grade Structure, Commercial/Industrial Exposure Scenario vs. $\frac{D_t^{eff}}{L_T}$ for various air exchange rates

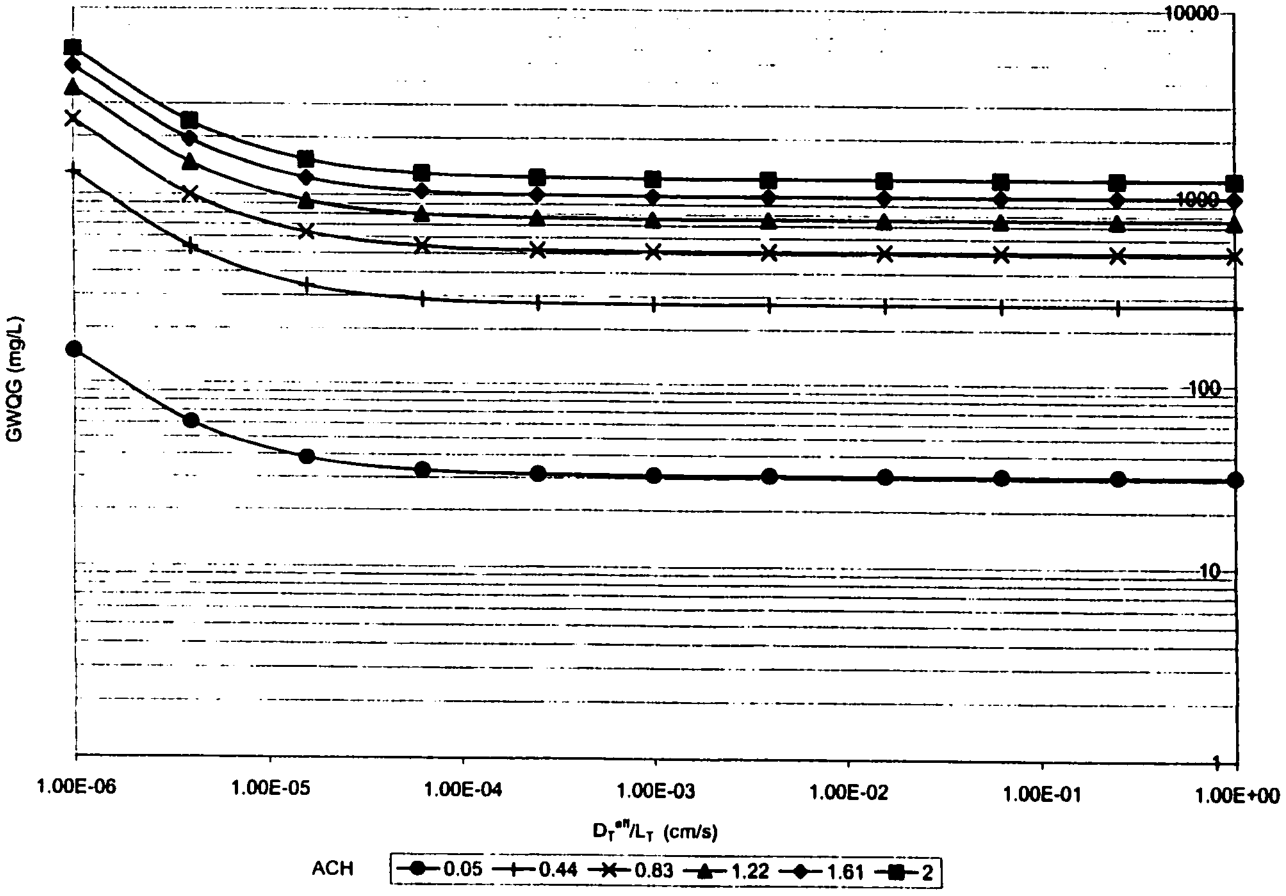

Figure N.7: GWQG for C_6–C_8 aliphatic hydrocarbons in Fine-Grained Soils for a Slab on Grade Structure, Commercial/Industrial Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

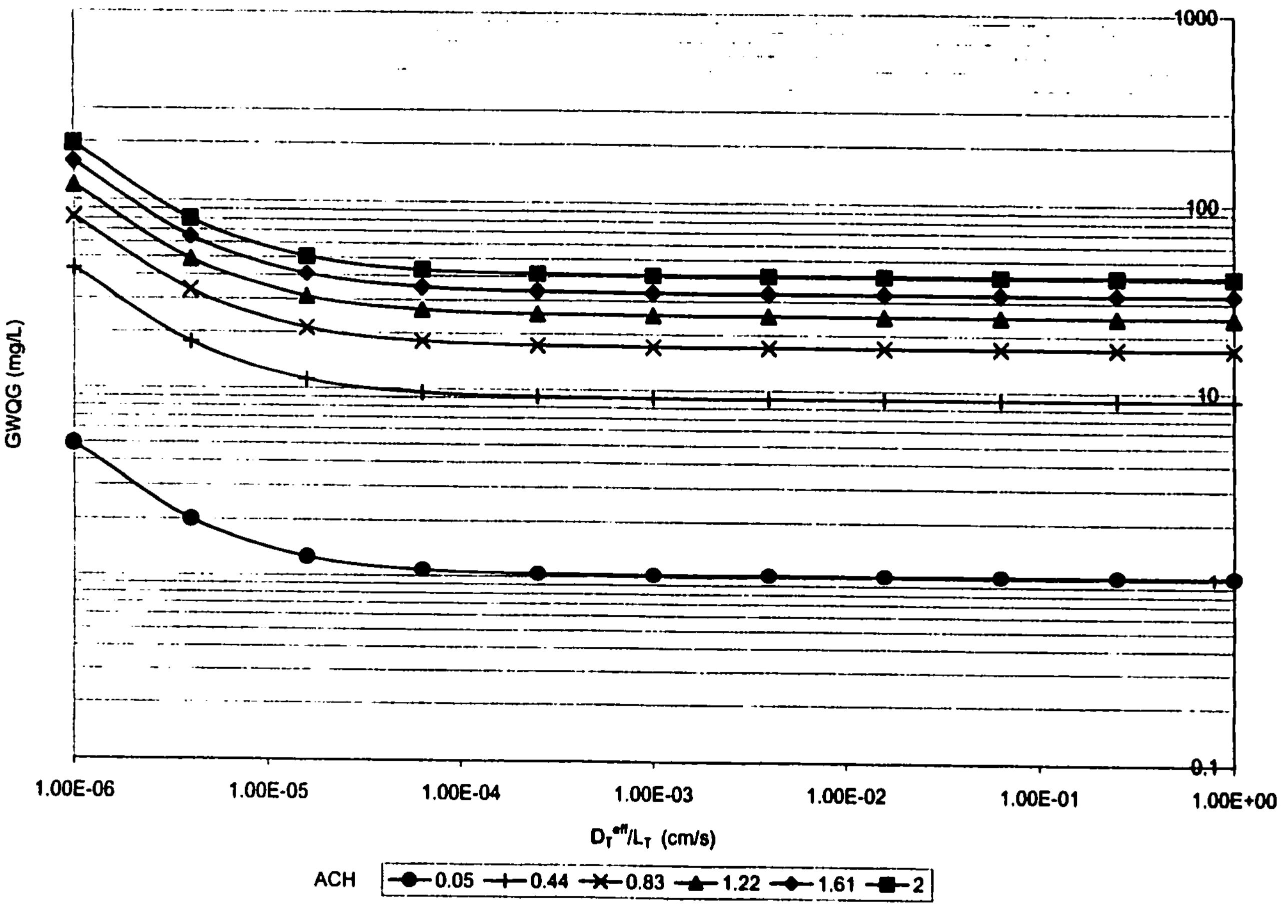

Figure N.8: GWQG for $C_{>8}$–C_{10} aliphatic hydrocarbons in Fine-Grained Soils for a Slab on Grade Structure, Commercial/Industrial Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

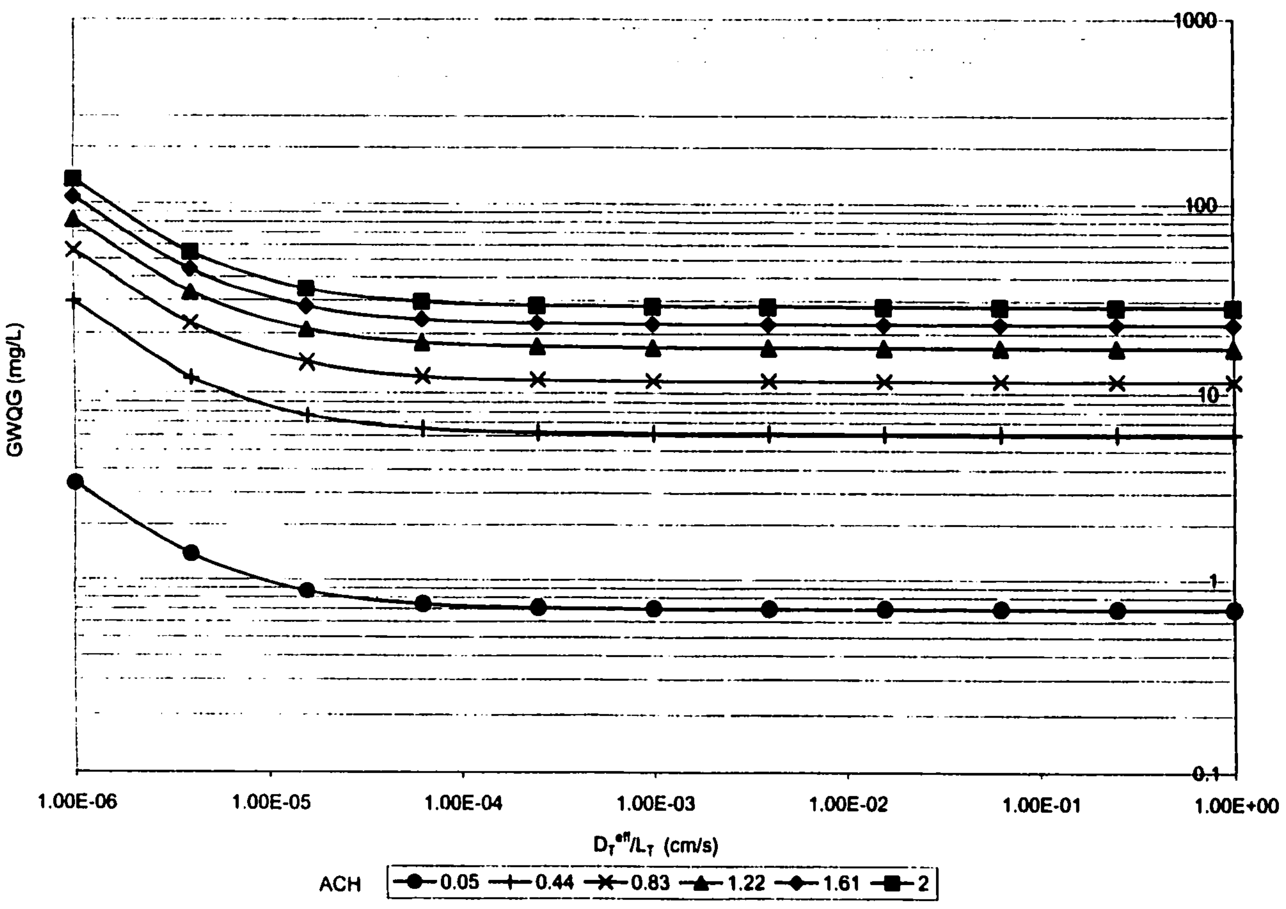

Figure N.9: GWQG for $C_{>10}$–C_{12} aliphatic hydrocarbons in Fine-Grained Soils for a Slab on Grade Structure, Commercial/Industrial Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

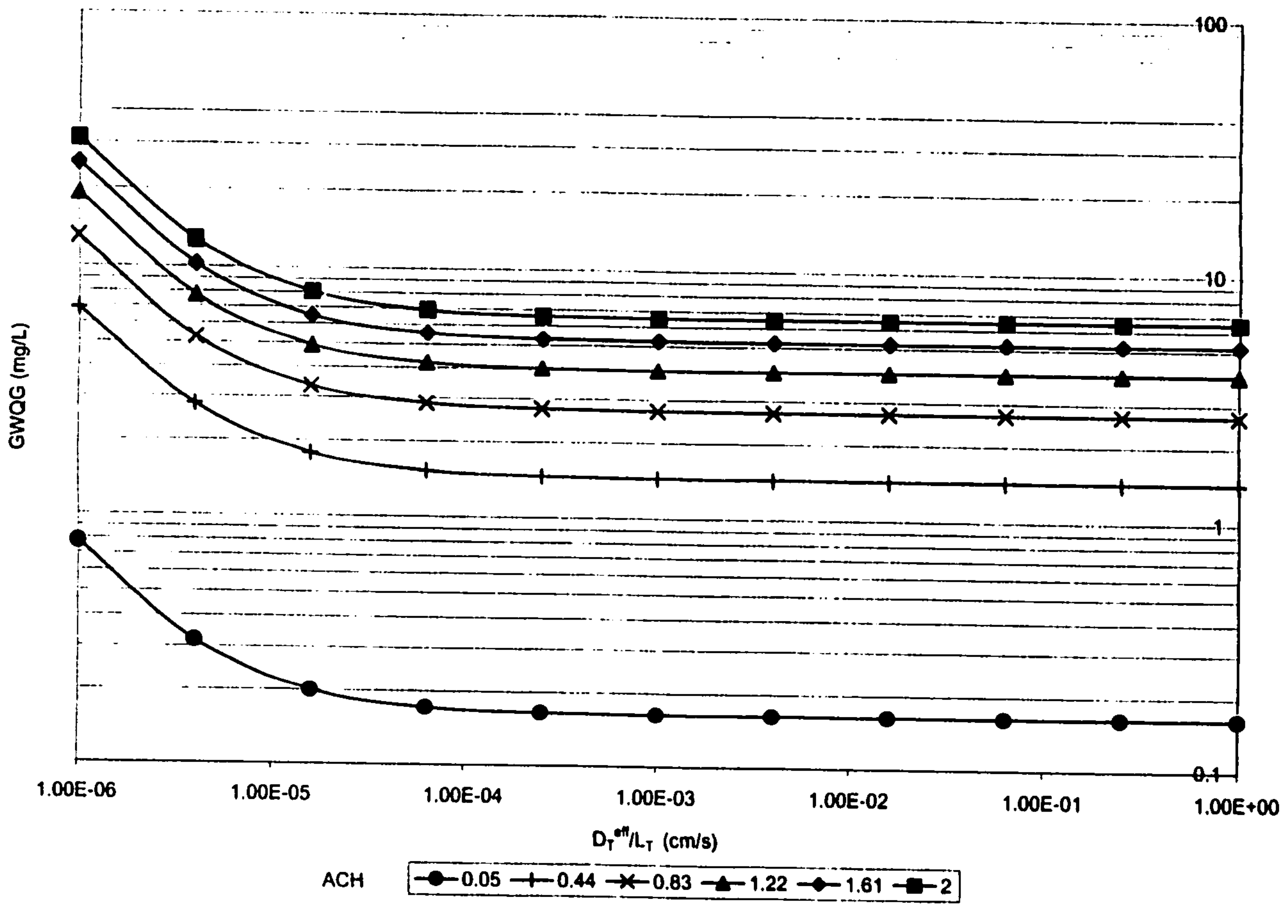

Figure N.10: GWQG for $C_{>12}$-C_{16} aliphatic hydrocarbons in Fine-Grained Soils for a Slab on Grade Structure, Commercial/Industrial Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

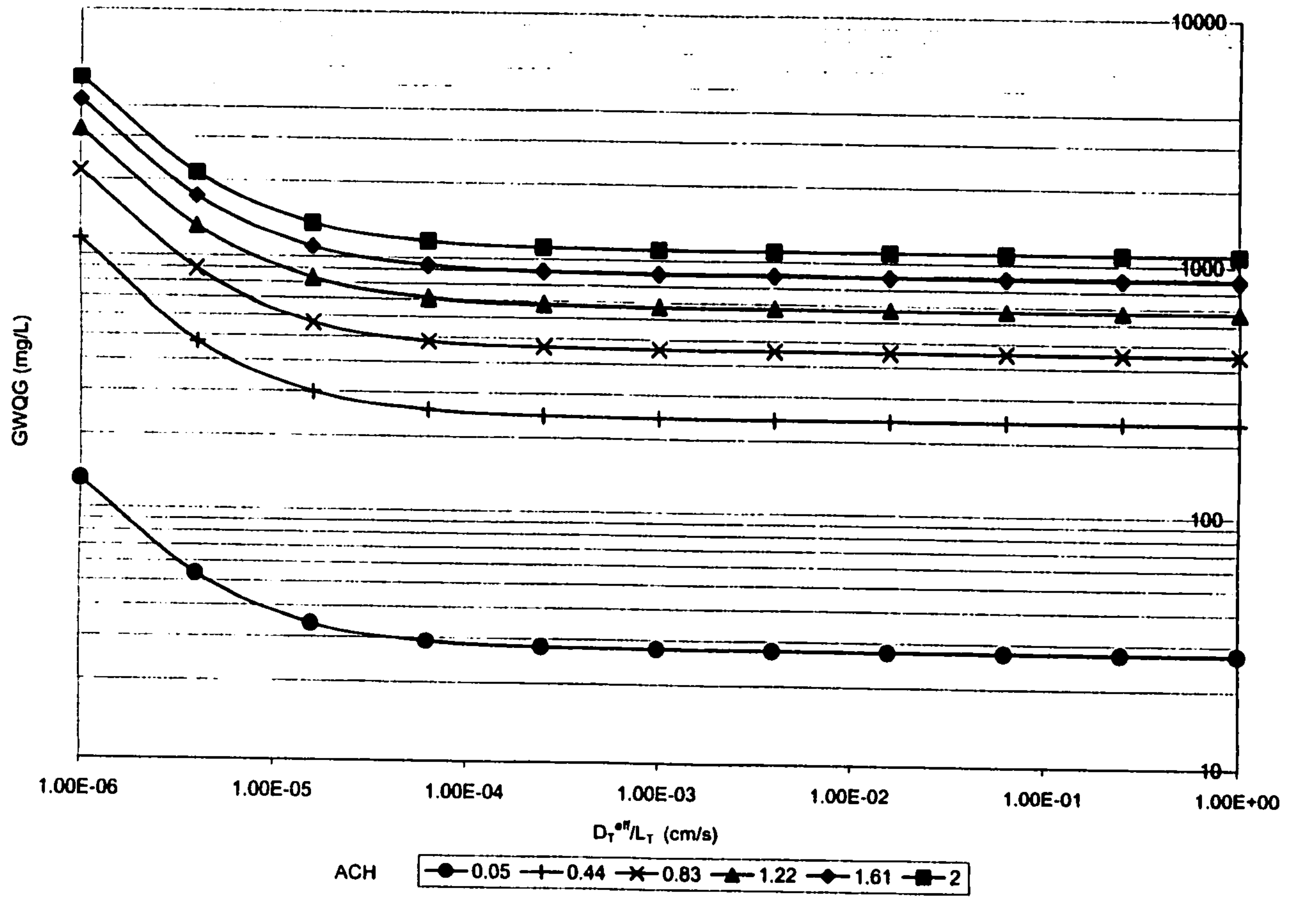

Figure N.11: GWQG for $C_{>6}$–C_{10} aromatic hydrocarbons i in Fine-Grained Soils for a Slab on Grade Structure, Commercial/Industrial Exposure Scenario vs. $\frac{D_i^{eff}}{L_T}$ for various air exchange rates

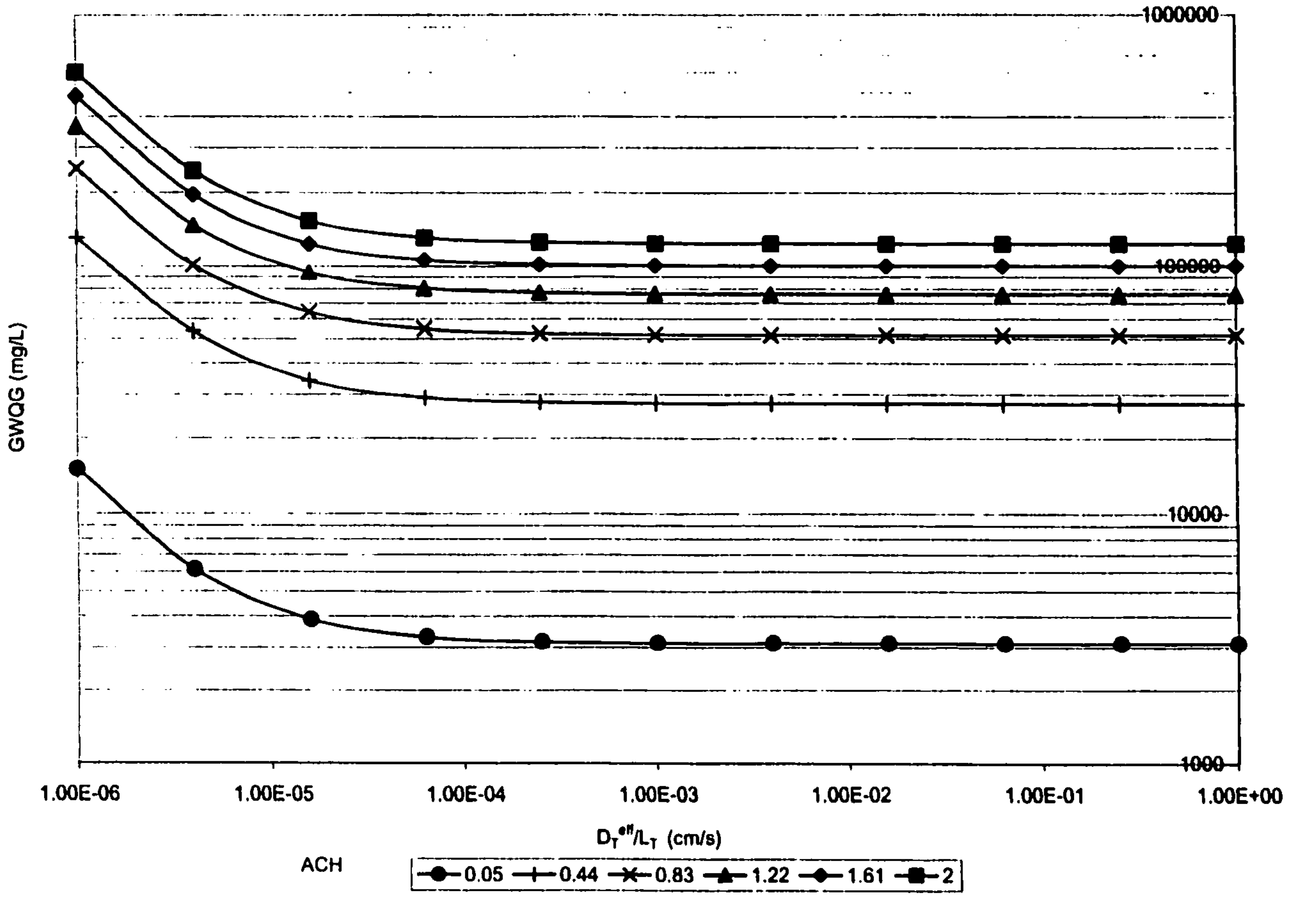

Figure N.12: GWQG for $C_{>10}$–C_{12} aromatic hydrocarbons in Fine-Grained Soils for a Slab on Grade Structure, Commercial/Industrial Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates

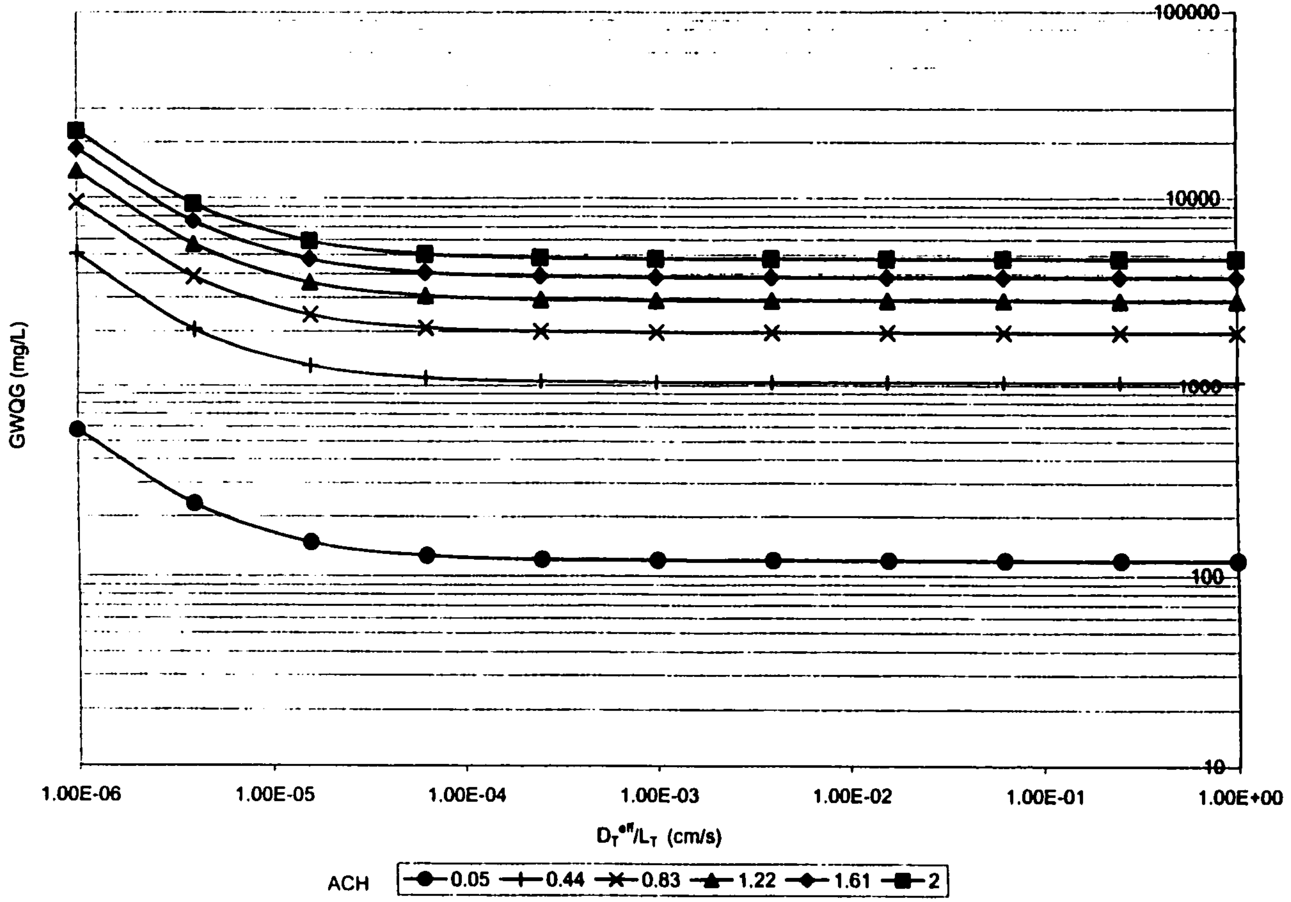

Figure N.13: GWQG for $C_{>12}$–C_{16} aromatic hydrocarbons in Fine-Grained Soils for a Slab on Grade Structure, Commercial/Industrial Exposure Scenario vs. $\frac{D_T^{eff}}{L_T}$ for various air exchange rates