Some Future Implications For Greenhouse Gas Mitigation Policy Frameworks: The Example Of Methane Oxidation By Soils In A Temperate Oak Woodland

by

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ABSTRACT

At the end of the 1990s, atmospheric concentrations of methane, a contributor to global warming, approached 1.8 parts per million by volume -- nearly double pre-industrial levels. This is due not only to increasing emissions, but also to inhibition of natural sinks. One of these sinks occurs in soils. Two distinct groups of soil bacteria, the methanotrophs and the nitrifiers, are capable of methane oxidation. The highest rates of methane uptake occur in soils inhabited by methanotrophs, while the lowest rates are characteristic of nitrifying bacteria: ammonium fertilization tends to encourage dominance by nitrifiers.

Short-term chamber experiments were carried out in a variety of different terrestrial environments in Victoria, British Columbia, Canada. Results were consistent with those obtained by other investigators for temperate forest sites elsewhere. Uptake rates of 0.059-0.082 $mg \cdot m^{-2} \cdot h^{-1}$ were measured at the Garry oak (*Quercus garryana*) woodland, while the closed-canopy mixed forest (*Acer macrophyllum* and *Pseudotsuga menziesii*) had values in the 0.032-0.042 $mg \cdot m^{-2} \cdot h^{-1}$ range. Modified environments such as lawns had significantly lower uptake rates. An abandoned hayfield sampled for this study showed intermediate values. Other researchers have shown that it can take years or decades for environments to recover after reversion to low-nitrogen regimes, which is consistent with a long-term shift in bacterial community composition.

Given that changes in land use affect soil processes which are intimately linked to atmospheric trace gas regimes, these issues will likely grow in importance over this century. Although current international legislation emphasizes the sequestration of atmospheric carbon dioxide in biomass, not all greenhouse gases follow this model. If mitigative policies are to be extended to other compounds such as methane and nitrous oxide, better understanding of non-sequestration sinks, e.g. soil uptake of CH_4, and the processes regulating them is essential. More flexible "adaptive management" strategies are desirable, to accommodate changes in environmental conditions and scientific knowledge.

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CHAPTER 1 - INTRODUCTION

1.1 Trace gases and environmental change

More than 99% of the Earth's atmosphere consists of two components: molecular nitrogen (N_2), and oxygen (O_2). Among the other compounds which make up the remaining 0.97% are argon (Ar), carbon dioxide (CO_2), and methane (CH_4) (Christopherson 1994). Many of these atmospheric trace gases only exist in miniscule concentrations, yet they have become increasingly important in environmental research.

Using archived data and air samples, and proxy sources such as ice cores, scientists have attempted to learn more about trace gas concentrations prior to the Industrial Revolution (Khalil and Rasmussen 1983, Wigley 1983, Chappellaz *et al.* 1990, Lorius *et al.* 1990). The consensus is that significant atmospheric changes have occurred over the past two centuries, as a result of human activities (Table 1.1). Changes in land use patterns due to agriculture, forestry, and urbanization, in addition to industrial emissions, have been implicated in the rising concentrations of trace gases such as methane (Watson *et al.* 1990, Keller *et al.* 1991). Although methane only accounts for around 15-20% of the radiative forcing associated with anthropogenic greenhouse gas emissions, it is still viewed as a significant part of the problem (Dlugokencky *et al.* 1998, Dauncey and Mazza 2001).

Conditions	**CO_2**	**CH_4**	**N_2O**	**CFC-12**
Pre-industrial (ppbv)	280 000	700	275	zero
Circa 1992 (ppbv)	355 000	1 714	311	0.503
Avg. change for 1980s (ppbv/a)	+ 1 500	+ 13	+ 0.75	+ 0.018

Table 1.1 - Changes in key greenhouse gas concentrations
(ppbv = parts per billion volume)
Source: Prather *et al.* 1995

By admitting sunlight while preventing the escape of longwave radiation into space, the atmosphere keeps the average planetary temperature at around 15°C, approximately 35 degrees higher than it would be in a vacuum (Nullet 1992). The analogy of the panes of glass in a greenhouse or hothouse is not strictly correct, since these structures work primarily by restricting convection (Oke 1978, Lewis 1992), but the concept of the "greenhouse effect" has stuck in the public consciousness.

Table 1.2 lists some characteristics of the most prominent "greenhouse gases", as they relate to radiative forcing. Assessment of a particular gas's strength in terms of its ability to influence climate is somewhat ambiguous, since it could be interpreted as the overall effect of each gas in the atmosphere, or alternatively, as the ability of individual molecules to absorb and re-radiate energy. "Global Warming Potential" is frequently used to compare different greenhouse gases, with CO_2 being allocated a value of 1, so all other compounds are expressed relative to it. Given that the estimated atmospheric lifetimes of these compounds can range from less than a decade to more than a century, there are also significant differences in the amount of energy which a given molecule would be expected to absorb and re-emit, depending on the time horizon; for this reason, GWP is calculated based on a uniform period of 100 years (Isaksen *et al.* 1992).

Gas	**Global Warming Potential (100 a)**	**Est. lifetime in years**	**Est. contribution to 1980s warming**
CO_2	1	120? (variable)	55%
CH_4	17-32	9-17	15%
N_2O	310-320	120-132	6%
CFC-11, CFC-12	4000-8500	45-102	24% combined

Table 1.2 - Relative strengths of radiatively-active trace gases

Sources: Watson *et al.* 1990; Hengeveld 1991; Isaksen *et al.* 1992; Intergovernmental Panel on Climate Change 1995, 1996

The concept of biogeochemical cycling is crucial to understanding the budgets of these chemicals. The accumulation of carbon dioxide in the atmosphere, for example, can be seen not just in the context of a one-time reaction which turns solid carbon into CO_2, but as part of a vast network of sources and sinks, with matter and energy in continuous motion in between. Carbon is transformed into various compounds as it moves through the system from the lithosphere to the atmosphere, or from the atmosphere into vegetation. Particularly over the past quarter-century, scientists have been investigating the relative importance of different sources and sinks, and the flows which link them.

1.2 Land use change and climate

Over the past decade, policymakers have made a distinction between reservoirs and sinks. In this context, carbon reservoirs refer to sequestration in soil, oceans, and organic matter; or in the longer (geological) term, fossil fuels and carbonates. Sinks are the actual processes which remove trace gases from the atmosphere, such as photosynthesis for CO_2 (Anderson *et al.* 2001). Both sinks and reservoirs can be affected by changes in land use, such as deforestation, conversion of agricultural land into urban areas, or the abandonment of pasture or farmland. Large regions of the planet have experienced significant surface changes, particularly over the past century (Kates *et al.* 1990).

Land use was discussed as part of greenhouse gas abatement strategies during the preliminary work on the United Nations Framework Convention on Climate Change (UNFCCC), signed in 1992. Parties to the UNFCCC promised to work for "conservation and enhancement" of "sinks and reservoirs of all greenhouse gases", "including biomass, forests and oceans." As international negotiations to fill in the framework with specific strategies and targets continued, land use emerged as a vital issue. Not only has past land use change been implicated as a major contributor to rising levels of atmospheric trace gases, but management of landscapes and the biophysical processes which take place in them is being presented as a way to mitigate global warming.

1.3 Planning issues

Proposals for avoiding atmospheric change by reducing emissions of trace gases are either directed at cutting back sources, or expanding sinks. Due to widespread dependence on fossil fuels, combined with the belief that conserving or switching energy sources will be detrimental to the economy, many countries and regions are unwilling or unable to decrease hydrocarbon use, at least not in the short term. The alternative is to focus on the sinks. This is not necessarily detrimental with regards to global environmental change issues, since preserving or enhancing sinks can complement broader land use objectives which are also becoming a concern in many areas: biological diversity and landscape conservation, and more equitable planning and development.

A sink-based strategy which is being advocated for measures such as forestry-related carbon offsets, where emission "credits" are granted to governments, agencies, or corporations, based on the amount of forest being protected or re-planted. The World Bank, the World Wildlife Fund, and automotive manufacturers such as Mazda and Fiat are among the groups which have purchased offsets over the past few years (Dauncey and Mazza 2001).

One of the problems faced by researchers and policymakers is the fact that not all carbon sinks are alike. In the past it has been expedient to assume that they function in the same way, *e.g.* by storing or sequestering carbon. The "vault" or "lockbox" model, although adequate for carbon dioxide, is not necessarily applicable to other carbon species, nor to other biogeochemical cycles. The real situation is far more complicated, and in order to develop successful policies and legislation, a broader understanding of the physical, chemical, and biological processes will need to be incorporated.

In addition, the problems inherent in trying to develop a planning framework for a situation where social, economic, and environmental conditions are shifting rapidly -- and scientific knowledge will also change as new discoveries are made -- increase difficulties and uncertainty. An analogy would be attempting to inhabit a house which is under construction, even before the foundations have solidified and the walls have been raised. Already concerns have been raised about attempting to implement significant, far-reaching legislation based on initial assumptions which may be incorrect, or subject to variability.

CHAPTER 2 - GLOBAL LEGISLATION ON GREENHOUSE GASES

2.1 The situation from 1992-2002

The United Nations Framework Convention on Climate Change, signed at Rio in 1992, also set in motion a regular series of annual meetings, the Conferences of the Parties (COPs), with the mandate to negotiate further additions to the UNFCCC. Meanwhile, the Intergovernmental Panel on Climate Change, set up in 1988 by the World Meteorological Organization and the United Nations Environment Programme, was continuing to co-ordinate international research into climate change, assess the science, evaluate potential impacts, and develop response strategies.

COP1, the first conference of the parties to the UNFCCC, was held in Berlin in 1995. It set a 1997 deadline for negotiating a binding agreement to cover emissions from industrialized countries after 2000 (Jepma and Munasinghe 1998). COP3, which took place in late 1997, resulted in the first international greenhouse gas treaty with clear commitments -- the Kyoto Protocol. This document also formalized the view that carbon sinks, as well as emission reduction at the source, both have roles to play in greenhouse gas mitigation. Essentially, signatories would be rewarded for protecting reservoirs or enhancing sinks. Granting emission reduction credits and debits for changes in carbon sequestration, due to "Land Use, Land Use Change, and Forestry" (LULUCF) efforts under the Kyoto provisions, were debated and analysed during the lead-up to that meeting, and have been an intense topic of discussion ever since (Watson *et al.* 1998).

As specified in the Protocol, the first commitment period is set for 2008-2012. It applies to industrialized nations listed in Annex B, whose respective annual emissions are referenced to the "base year", which is 1990 in most cases (Watson *et al.* 1998). Canada's annual commitment is 6% below 1990 emissions, while the U.S. is 7% below, and the European Union is 8% below (Anderson *et al.* 2001). While some authorities have been critical of reliance on sinks, seeing this as a loophole to avoid cutting emissions at the source (Godrej 2001), others felt it was a necessary compromise in order to bring in the nations which account for most of the emissions. Some countries had been lobbying for inclusion of projects initiated earlier, such as reforestation. Critics argued that this would be contrary to the entire purpose of the agreement: for example, a controversial Canadian proposal would have allowed the industrialized nations to avoid any measures to curb their own CO_2 (Anderson *et al.* 2001). At COP4 in Buenos Aires in 1998, delegates agreed that credit would not be given for activities initiated prior to 1990.

With regards to LULUCF, the most relevant parts of the agreement occur in the third article of the Kyoto Protocol. Article 3.3 states that participating countries will be credited or debited for verifed changes in sequestered carbon due to "direct human-induced land use change and forestry activities" -- in this context, afforestation, reforestation, and deforestation (Watson *et al.* 1998, Anderson et *al.* 2001). Article 3.4 states that for later (post-2012) commitment periods, "additional human-induced activities" in the agricultural soils and forestry categories will be included (United Nations Framework Convention on Climate Change 2002).

Sinks continued to be contentious at COP6, held in The Hague late in 2000. This was one of the issues which stalled the negotiations. Some members of the Umbrella Group (specifically, Canada, the U.S., Japan, and Australia) wanted to count forest and agricultural sinks as part of their Kyoto commitments, while the European Union felt that the credits should be strictly limited. It had been suggested that large countries with low population density might be able to fulfill "up to 100% or more" of their Kyoto commitments solely through carbon sequestration (Topfer 2001), which some felt was open to abuse (Anderson *et al.* 2001). A later attempt to reconcile the positions in Ottawa also failed. Early the following year, the new U.S. administration rejected the Kyoto Protocol, and serious doubts were expressed about the viability of the agreement without the co-operation of the world's leading greenhouse gas emitter.

In July 2001, an additional session was convened in Bonn, to attempt to resolve the impasse before COP7 in Marrakesh. This extra meeting, variously called COP6b, CoP6.5, or the Sixth Session (Part Two), succeeded in reaching international agreement on the most public issues (Pew Center on Global Climate Change 2001). Under the compromise, more activities can be included under for carbon sink credits post-2012, including forest and cropland management, and revegetation. There is no overall cap on sink credits, but various categories have country-specific limits (Canada's is 10% of base-year emissions). Credits for cropland management, grazing land management, and revegetation will not be capped, but countries can only be credited if sequestration increases beyond 1990 levels (Pew Center on Global Climate Change 2001).

COP7, held in Morocco in November 2001, succeeded in producing an agreement -- the Marrakesh Accords. The goal was to finalize enough of the operational details, or "rulebook", to raise the level of certainty and enable more governments to ratify the Kyoto Protocol. As a result, the beginnings of an international system to deal with this problem are now in place. Despite ongoing science and policy issues, the Kyoto Protocol appears to be heading towards implementation (Bodansky 2002), particularly since its ratification by industrialized countries such as Germany, the UK, Japan, and Canada.

One of the tasks which had to be addressed at COP7 was the need to agree on terminology, such as what constitutes a "forest", and the difference between "afforestation" and "reforestation." The resulting definition was elaborate in its specification of canopy cover percentages, tree height potential, and stand area -- but this degree of precision is important, given the confusion that resulted in earlier COPs when some delegates assumed that the older Intergovernmental Panel on Climate Change (IPCC) definitions would automatically be used (Schlamadinger and Marland 2000).

Not surprisingly, the LULUCF components required much of this attention to detail at Bonn, Marrakesh, and the subsidiary meetings. The initial version of the agreement, meant to apply to land use, land-use change, and forestry project activities, in practice is limited to afforestation and reforestation (at least for the first decade). Sinks from management of cropland and grazing land will eventually be included, but not until later (Schlamadinger and Marland 2000).

Even before Canada's ratification of the Kyoto Protocol in December 2002, the federal government had been working on a proposed greenhouse gas mitigation plan, combining emission reductions with offsets -- possibly involving the Clean Development and Joint Implementation Mechanisms -- and enhancement of domestic sinks (Government of Canada 2002). As of early 2003, only one more country -- Russia -- is needed to ratify the Protocol in order to bring it into effect. Given that even jurisdictions in countries which are opposed to the Protocol, such as Australia and the

United States, are proceeding with their own reduction plans, it now appears likely that these kinds of strategies are going to be more widespread in the future.

2.2 Possible future developments

The UNFCCC, Kyoto Protocol, and subsequent agreements are open to the possibility of interactions with other compounds involved in the carbon cycle. However, legislation on other radiatively-active trace gases has been postponed. Most of the policies specifically addressing global warming have focused on carbon dioxide, particularly at the national and international level. The Kyoto Protocol also applies to five other greenhouse gases (methane, nitrous oxide, hydrofluorocarbons, perfluorocarbons, and sulphur hexafluoride), but they are expressed in terms of carbon dioxide equivalents, the concentration of CO_2 which would create identical radiative forcing (Jepma and Munasinghe 1998). The Emission Reduction Unit (ERU) is becoming the "currency" for global warming initiatives -- equal to a tonne of CO_2, as calculated using Global Warming Potential values for each gas.

If CO_2 accounts for 53% of the overall climate change, it seems reasonable to make this compound the primary focus; however, others have argued that the other 47% of the problem should not be overlooked (Dauncey and Mazza 2001). If the "basket" of six gases is figuratively broken open, and each gas treated separately, with different reduction goals, it may make mitigation strategies more flexible.

In order to incorporate these other gases into a management regime, allowances will have to be made for the fact that they are different compounds, and do not behave the same way as carbon dioxide. For example, CO_2 sinks emphasize sequestration/removal of carbon from the system, in long-term (geological) reservoirs such as carbonates, or shorter-term (biological) reservoirs such as biomass and soil carbon. However, nitrous oxide (N_2O) is not "fixed" by plants during the photosynthetic process, therefore a LULUCF sequestration model would not be applicable. Methane (CH_4), a reduced form of carbon, is another trace gas responsible for global warming. It provides an interesting example of some of the similarities and differences compared to the "standard" model of carbon dioxide, as will be explored in subsequent chapters.

It is already apparent that current sequestration-based legislation will require considerable changes to make it suitable for most other greenhouse gases, biogeochemical processes, and local habitats. For example, open woodlands, scrublands, and grasslands accumulate far less biomass per unit area than forests -- yet these ecosystems cover a third of the planet's terrestrial surface (Brown and Gibson 1983). In addition, many of these environments are important in their own right. They are not as species-rich as tropical rainforests, but nonetheless provide habitat for unique organisms, and are important components of the biosphere as a whole. The provisions of the Marrakesh Accord recognize that they should not be sacrificed by converting them into "carbon plantations" via the introduction of fast-growing tree species, in order to mitigate CO_2 emissions.

CHAPTER 3 - METHANE, LAND USE, AND POLICY

3.1 Methane in the atmosphere

Methane (CH_4) is one of the simplest hydrocarbons, consisting of four atoms of hydrogen joined by single covalent bonds to one atom of carbon, in a tetrahedral formation. The characteristics and origins of this compound are of interest to the planetary and biological sciences, since it occurs in many environments on Earth (Watson *et al.* 1990), and is a major atmospheric component of the outer planets (Wood 1979). Since CH_4 is produced by a variety of biotic and abiotic processes (Schoell 1988), its presence has been interpreted in different ways, some of them quite controversial. The ongoing strife over "deep gas" (Cole 1996, Evans 1996, Kelley and Fruhgreen 1999) is a recent example.

In the last quarter of the 20th century, methane became linked with a key environmental issue: atmospheric change. Along with compounds such as carbon dioxide and water vapour, methane was identified as a contributor to global warming. These radiatively-active gases, transparent to incoming visible light, are efficient absorbers of longwave infrared wavelengths emitted by the Earth's surface. Among anthropogenic trace gases, methane's potential impact on global warming is perceived as being second only to carbon dioxide, in terms of radiative forcing (Penner 1994): it is believed to account for about 15-20% of current radiative forcing, or approximately one-quarter of the influence of CO_2 (Smith 1995, Dauncey and Mazza 2001).

3.2 Methane sources and sinks

Land use can have an indirect effect on trace gas emissions, by impairing natural removal processes (sinks), *e.g.* through physical modifications such as clearing vegetation, disturbing soil, or altering local hydrology (Rodhe and Svensson 1995, Nevison *et al.* 1996, Steudler *et al.* 1996b). Understanding the processes which generate and destroy trace gases means finding out more about the roles played by surface environments, and how they interact with the atmosphere. Methane is particularly well-suited to a geographical approach, since it is so strongly related to ecosystem characteristics and land use. Table 3.1 describes some of the known sources and sinks for atmospheric methane, on a global scale.

NATURAL SOURCES	**Teragrams of methane per year**
Wetlands	115
Oceans	10
ANTHROPOGENIC SOURCES	**Teragrams of methane per year**
Natural gas	40
Coal mines	30
Petroleum industry	15
Enteric fermentation (livestock)	85
Rice paddies	60
Biomass burning	40
Landfills	40
Animal waste	25
Domestic sewage	25
SINKS	**Teragrams of methane per year**
Tropospheric hydroxl reactions	445
Stratosphere	40
Soils	30

Table 3.1 - Estimates of known sources and sinks for methane

Source: Prather *et al.* 1995

Atmospheric methane concentrations, which varied between 0.6 and 0.8 ppm during the past 3 000 years, have more than doubled since the mid-1800s to reach their present concentration of 1.7-1.8 ppm (Moore 1988, Tyler *et al.* 1990). By comparison, CO_2 only increased by 25% during the same time interval (Smith 1995). In terms of overall magnitude, human activities are believed to be responsible for at least 60% of current methane emissions (Prather *et al.* 1995).

Given that methane has a relatively brief residence time in the atmosphere -- on the order of a decade or two compared to CFC-13 ,which may persist for several centuries (Intergovernmental Panel on Climate Change 1995) -- methane concentrations would be expected to respond more quickly than longer-lived trace gases to changes in source, sink, and transport conditions (Khalil and Rasmussen 1990, Intergovernmental Panel on Climate Change 1995, Watson *et al.* 1998). Prather *et al.* (1995) estimated a lag time on the order of 11 to 17 years, an adjustment to compensate for the relatively small biological sink. This may already be happening, since Dlugokencky *et al.* (1994a, 1998) suggest that the annual rate of increase in atmospheric methane slowed in the early 1990s, at least for the northern hemisphere. Earlier, Khalil and Rasmussen (1986) proposed that the Southern Oscillation may have contributed to low CH_4 concentrations measured at Cape Meares on the Oregon coast, and at other locations.

The major sink reported for methane is in the atmosphere itself, and is attributed to the hydroxyl abstraction reaction (Watson *et al.* 1990, Fung *et al.* 1991). Hydroxyl radicals ($OH\cdot$) are produced by the photodissociation of water molecules, and are also an important component of ozone photochemistry (Study of Man's Impact on Climate 1971, Killops and Killops 1993). Hydroxyls are the dominant oxidizing compound in the troposphere, capable of destroying not only methane, but most other gases which contain hydrogen (Prinn *et al.* 1995), such as ethane (C_2H_6) and methyl chloroform (CH_3CCl_3). Higher up in the stratosphere, methane oxidation is a removal mechanism for chlorine radicals, which are responsible for damaging the ozone layer (Khalil and Rasmussen 1983, Tyler 1986, Gupta et al. 1996). Recent work by Prinn *et al.* (2001) suggests that global concentrations of hydroxls have declined by an average of 10% over the past two decades, weakening the ability of the atmosphere to cleanse itself.

The other methane sink is due to microbial activity in soils, and also in freshwater and marine environments. Aquatic systems are beyond the scope of this study -- the global significance of methane uptake by soils will be discussed in this and subsequent chapters. Although oxidation in soils worldwide is estimated to account for only 5-15% of the total sink term (Fung *et al.* 1991, Smith 1995, Liptay *et al.* 1998), it is nonetheless a significant part of the cycle. Since the methane uptake capabilities of many environments have yet to be fully evaluated, it is possible that the size of this sink has been underestimated.

Research to date has demonstrated that the atmosphere is a complex system. Studying the origin and fate of trace gases like methane contributes to our understanding of the atmosphere as being active and continuously in flux, with close linkages to processes occurring at the planet's surface. Given that soils play an important role in land use change processes, their relationship with atmospheric trace gases should be examined more closely.

3.2.1 Microbial oxidation of methane

Microorganisms which inhabit soil, water, and sediments play a key role in biogeochemical cycling. The importance of nitrogen-fixing bacteria, *e.g. Rhizobium*, which lives in nodules on the roots of plants, is widely acknowledged (Delwiche 1970, Killops and Killops 1993); but until almost twenty years ago the ability of microorganisms to break down sulphur compounds and hydrocarbons tended to remain unexamined outside of fields such as air pollution meteorology, microbiology, and geochemistry. It is only during the past few decades that a coherent picture has begun to develop of the roles which microbes play in biogeochemical cycles, and how they are affected by environmental factors such as temperature, moisture, substrate type, or oxygen availability. Field measurements have often revealed interesting phenomena which may not have been recognized in previous laboratory or numerical modelling studies.

Using methane as an example, the production of CH_4 by methanogenic bacteria under anaerobic (anoxic) conditions is known to occur in both terrestrial and aquatic environments. Some soil types, *e.g.* the gleysols and histosols, are saturated for all or part of the year, and experience low oxygen availability due to restricted diffusion.

In the 1970s, at the beginning of the current wave of climate change research, there was a widespread assumption that soils would act as a source of atmospheric trace gases: investigations focused on the possible magnitude of that source. Measurements in non-agricultural wetlands suggested annual soil CH_4 emissions of

150 Tg, more than 25% of estimated global totals at that time (Khalil and Rasmussen 1983). These findings encouraged researchers to look at environments known to be generating methane, such as the bogs and peatlands of northern Canada. The Boreal Ecosystem Atmosphere Study (BOREAS) was one such initiative that continued into the 1990s. Researchers were attempting to quantify the size of trace gas fluxes to the atmosphere from a variety of different ecosystems, which in turn would contribute to knowledge of the carbon cycle, and increase the accuracy of climate models. As work progressed, the findings indicated that emissions could vary considerably over time, depending on factors such as temperature, and fluctuations in the water table (Harriss *et al.* 1985, Mooney *et al.* 1987).

At this point, the story was incomplete: methane can be oxidized by other types of bacteria known as methanotrophs, in a reaction summarized by Smith (1995):

$$CH_4 + O_2 + 2\mathbf{X} \dashrightarrow CO_2 + 2\,XH_2 \qquad (3.1)$$

where **X** represents an electron acceptor. In fact the actual process is more complicated, transforming CH_4 through a series of carbon compounds, including methanol, formaldehyde, and formic acid, until it becomes CO_2 (Davidson and Schimel 1995, Roslev *et al. 1997).* In addition, some microorganisms, known as methylotrophs, can utilize the methyl groups (CH_3) which occur in various organic substances such as hydrocarbons; however, not all of these microbes are capable of assimilating CH_4 (Davidson and Schimel 1995). In soils, most methane-consuming bacteria require aerobic conditions to function (Born *et al.* 1990). Some sulphate-reducers can carry out anaerobic oxidation of methane (Jorgensen 1983, cited in Killops and Killops 1993), although this situation would not apply to most freshwater or terrestrial ecosystems, which contain substantially less sulphate than marine environments such as ocean sediments.

There is some evidence that methane uptake rates are highest not at the soil surface, where oxygen is most readily available, but at a depth of several centimetres (Adamsen and King 1993, Roslev *et al.* 1997). In most cases this is deeper than the litter and humus of the organic ("O") horizon (Figure 3.1), but still well within the "A" horizon which contains the most biological activity (Whittow 1984, Soil Classification Working Group 1998). Using radioisotope-labelled methane, $^{14}CH_4$, Roslev *et al.* (1997) traced the uptake of atmospheric methane in a forest soil. They were able to identify the zone of maximum methane oxidation, the A2 horizon, which in this case was 15-18 cm below the surface (Roslev *et al.* 1997).

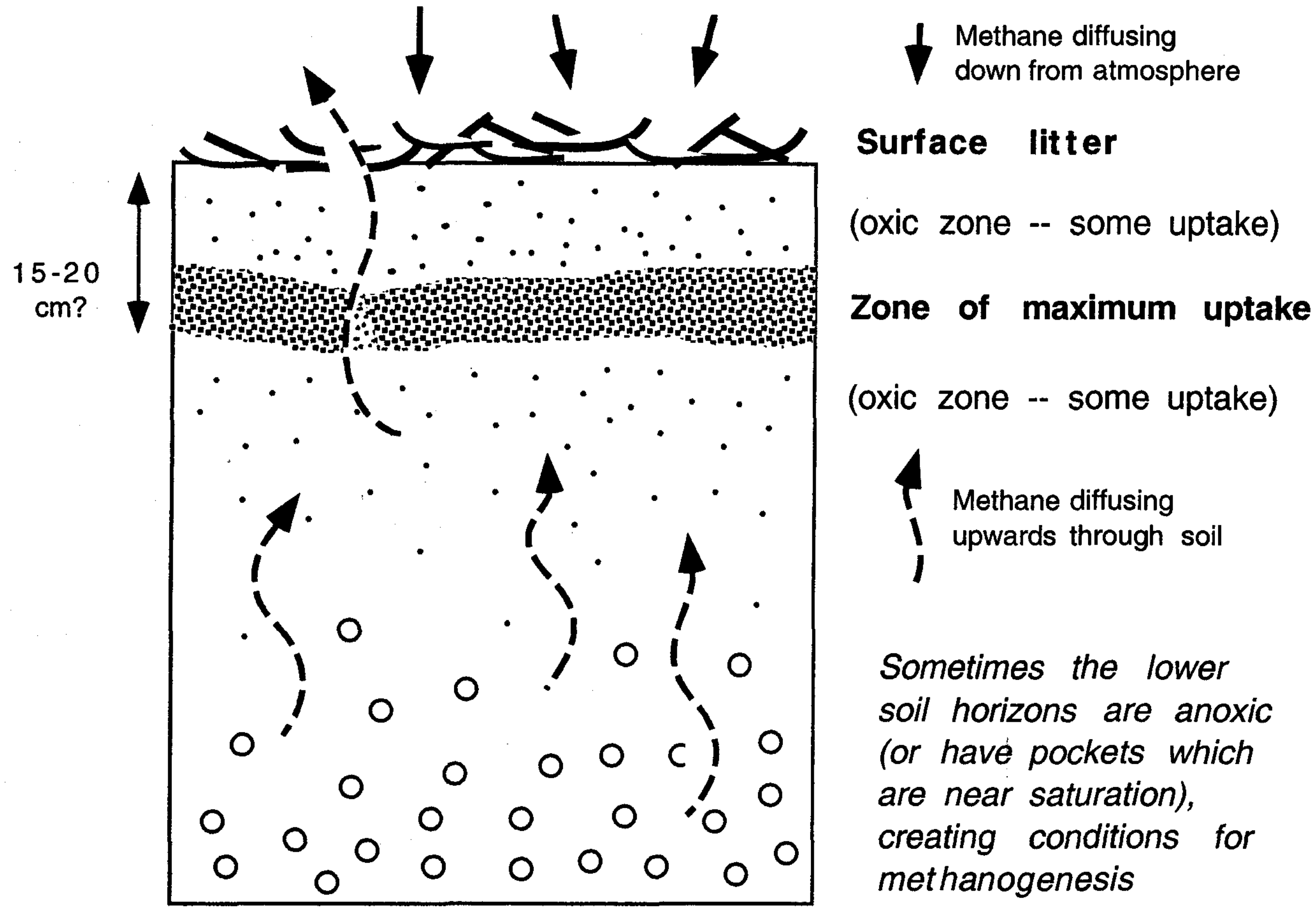

Figure 3.1 - Methane uptake by the soil

Most of the early work on methanotrophy was carried out by microbiologists. In their 1981 review article, Higgins *et al.* state that although microbial oxidation of methane had been recognized for most of the century, its biogeochemical implications had generally been overlooked, with more attention being paid to its potential commercial applications, *e.g.* mass production of proteins and polymers. Some air pollution meteorologists did take note, however, and as early as 1971, Abeles *et al.* had suggested that the soil could be an important sink for various pollutants, including ethylene and other hydrocarbons. Since uptake was inhibited by autoclaving, chemical sterilization, and absence of oxygen, microbial activity was proposed as the main mechanism.

At around the same time, the Study of Man's Impact on Climate (SMIC) concluded that "it is fairly certain that most [methane] is also destroyed again by microbiological action under aerobic conditions on the surface" (1971, p. 242). However, the potential size of the reservoirs and fluxes was not addressed, and the authors even stated that "because CH_4 has no direct effects on the climate or the biosphere, it is considered to be of no importance for this report" (SMIC 1971, p. 242).

In the subsequent decade, laboratory studies of methane oxidizing bacteria were carried out (Coleman *et al.* 1981). Harriss *et al.* (1982) were among the first researchers to discover field evidence of methane uptake: they had been sampling a wetland environment, traditionally assumed to be a methane source, and observed reversal of the flux during drought conditions.

In their major 1983 paper, Khalil and Rasmussen proceeded from the assumption that reaction with hydroxl radicals, and "some irrecoverable loss of CH_4 to the stratosphere" were the dominant sinks for atmospheric methane; still, soils were discussed only in the context of CH_4 emissions. However, methane oxidation was being detected in a growing number of environments: temperate forest (Keller *et al.* 1983); tropical savanna (Seiler *et al.* 1984); and even habitats previously assumed to be methanogenic, such as boreal wetlands (King *et al.* 1989).

Research on methane uptake was not confined to terrestrial settings. During the 1970s and 80s, numerous studies had reported finding methanotrophic bacteria in sulphate-reducing environments, and near undersea hydrothermal vents and cold water seeps, along with other chemoautotrophic organisms (Conway 1994). Whalen *et al.* (1990), echoing SMIC (1971), stated that "Methane oxidation must be an important modulator of atmospheric CH_4 flux; roughly half of the organic carbon degraded anaerobically is converted to CH_4, yet CH_4 release to the atmosphere represents only 0.5% of the total carbon turnover."

By the late 1980s, many investigators were aware of methanotrophy, even if they only mentioned it in passing (Mooney *et al.* 1987, Ford and Naiman 1988, Moore 1988, Nisbet 1989). A few years later, methane uptake estimates had become a standard component of methane flux studies, *e.g.* B.H. Levelton & Associates 1991, Lessard *et al.* 1994, Roulet *et al.* 1992, Smith 1995, and Yavitt *et al.* 1995. Comparing the major international reports from the Intergovernmental Panel on Climate Change (IPCC), the update by Prather *et al.* (1995) differs from the earlier version (Watson *et al.* 1990). Although the estimate for the global soil sink remained at 30 (plus or minus 15) Tg/a of CH_4, the discussion of microbial uptake was expanded, with more attention devoted to its role in modulating methane fluxes, and the presumed impact of human activities on soil conditions. Recently, Holmes *et al.* (1999) increased the estimate of methane taken up by soils to 20-60 Tg/a, and stated that this sink's sensitivity to disturbance had contributed to increasing atmospheric methane concentrations during the 20th century.

Meanwhile, field investigations have continued in a variety of environments. Monitoring of trace gas fluxes over months and years revealed that as the seasons change, a given site may oscillate back and forth between methane production and oxidation, depending on environmental factors such as soil moisture and temperature (Steudler *et al.* 1989, Adamsen and King 1993, Keller *et al.* 1993). Researchers are particularly interested in the ability of methanotrophs to survive for long periods of time in sub-optimal conditions (Yavitt *et al.* 1995), where concentrations of methane are low and free oxygen may be scarce or absent. Once believed to be obligate aerobes (Steudler *et al.* 1989), the abilities of these bacteria to survive anaerobic starvation as spores, or even maintain minimum requirements through limited anaerobic glucose metabolism (Roslev and King 1995), demonstrate remarkable resilience.

Due to the relative sizes of the habitats and organisms involved, the scale of the processes can reach all the way down to micrometres. The intricate microtopography created by the orientation of individual clods and grains of soil, combined with the fluctuation of the water table in many soils during the year, means that it is possible to have oxic and anoxic microsites in close proximity to each other within the same soil profile, or even the same horizon. As a result, conditions can be highly localized, with considerable variability at a particular location (Lessard et al. 1994).

In practical terms, even a shallow layer of aerobic soil on top of a saturated horizon can act as a filter: the resident methanotrophs can intercept and oxidize the methane diffusing from below, before it reaches the atmosphere (Whalen *et al.* 1990, Rodhe and Svensson 1995, Kightley *et al.* 1995, Bergamaschi *et al.* 1998b). Experiments with methyl fluoride (CH_3F), a chemical known to inhibit methanotrophs, revealed that oxidation can remove as much as 90% of available methane (Oremland and Culbertson 1992).

3.2.2 Methanotrophs and nitrifiers

Given that it can be difficult to identify and culture methane-oxidizing bacteria, the microorganisms and processes at work are still not completely understood. In the 1990s, more research was initiated to understand the conditions under which methane-oxidizing bacteria (MOB) live, their taxonomy, population dynamics, and the biochemical pathways which they utilize (Holmes *et al.* 1999). So far, it appears that two different enzymes are capable of oxidizing methane: CH_4 monooxygenase, which is produced by methanotrophs; and NH_3 monooxygenase, which occurs in nitrifiers (Davidson and Schimel 1995).

Methane and ammonium (NH_4^+) molecules are both tetrahedral in shape, and of similar mass and size. As a result, ammonia monooxygenase can also oxidize CH_4 (Davidson and Schimel 1995). The ability of certain enzymes to catalyze reactions for both carbon and nitrogen compounds has also been documented for nitrogenase, which is essential for nitrogen fixation by microbes such as Azotobacter. Nitrogenase normally converts atmospheric nitrogen to ammonia, but can also reduce acetylene (C_2H_2) to ethylene (C_2H_4) (Killham 1994).

While nitrifiers are capable of oxidizing methane using NH_3 monooxygenase, Wood (1989) states that under laboratory conditions, some methanotrophs can oxidize NH_4^+ using methane monooxygenase: the extent to which this occurs in natural soils is still uncertain. Work by Roy and Knowles (1994), on polluted freshwater sediments in Hamilton Harbour, suggests that high concentrations of CH_4 are necessary for this to occur.

Considerably more research has been done on nitrifying bacteria than on methanotrophs. Nitrifiers are known to be obligate chemoautotrophs and obligate aerobes (Davidson and Schimel 1995), *i.e.* they obtain all their energy from the nitrification reaction. Also, in order to fix CO_2 as organic carbon, they need free oxygen; they cannot work under anaerobic conditions, using sulphates as electron acceptors in the manner of some other types of bacteria.

Two main physiological groups of nitrifiers have been isolated (Richards 1974). The first group, *e.g. Nitrosomonas*, oxidizes ammonium to nitrite (NO_2^-):

$$2NH_4^+ + 3O_2 \dashrightarrow 2NO_2^- + 4H^+ + 2H_2O \quad (3.2)$$

Since the oxidation of ammonium to NH_2OH is an endergonic process, the NH_3 monooxygenase enzyme is essential for that first step (Wood 1989).

The second group, *Nitrobacter* for example, converts nitrite to nitrate, completing the oxidation process:

$$2NO_2^- + O_2 \dashrightarrow 2NO_3^- \quad (3.3)$$

Richards (1974) notes that this reaction serves an important function in the environment, since nitrites are toxic to many plant species. However, the energy yield from the oxidation of nitrite is quite low (only 71 kJ/mole), and only a few bacterial species are able to subsist on it (Wood 1989). By comparison, the oxidation of a mole of ammonium provides 272 kJ. This is almost four times as much energy, but still less than 10% of what is released from oxidizing glucose (Wood 1989).

Interestingly, the ability of *Nitrobacter* to oxidize acetate (CH_3COOH) as well as nitrite has been known since at least the early 1970s. Richards (1974) proposed the term "facultative autotroph", as a way of recognizing the ability to switch between utilizing carbon or nitrogen as an electron acceptor. In retrospect, the fact that nitrite-oxidizing bacteria are also capable of using acetate as an energy source suggests that nitrifiers could also be involved in the oxidation of other carbon compounds. Richards (1974) speculates that this could be a useful survival trait under low-nitrite conditions. If nitrifiers are capable of oxidizing methane, this implies that CH_4 uptake could occur wherever nitrifying bacteria are found; and these varieties of microbes are widespread, occurring even in Antarctica (Wilson *et al.* 1997).

Steudler *et al.* (1996a) discuss a lab assay technique which distinguishes between the two microbial communities involved in methane oxidation, methanotrophs and nitrifiers. There are significant differences between nitrifiers and methanotrophs, in terms of their overall ability to oxidize methane, as well as in the biochemical pathways used. Investigations of several sites appear to indicate that the highest rates of methane uptake occur in soils inhabited by methanotrophs. Oxidation is significantly lower in soils with mixed communities (both methanotrophs and nitrifiers), and lowest of all when only nitrifiers are present (Steudler *et al.* 1996a).

In effect, the situation for methane-oxidizing bacteria combines the attributes of two different types of sinks. While living organisms are responsible (as in the vegetation sink for atmospheric carbon dioxide), most of the methane is not sequestered as bacterial biomass, but released as CO_2. This situation is even more pronounced in (abiotic) atmospheric sinks, where the compounds in question are transformed, but not stored.

3.3 Current approaches to methane abatement

Methane is not the major cause of global climate change, but it fits in well with broader land use and anti-pollution strategies. Anthropogenic methane emissions are intimately connected with the resources which we use, and the way in which we utilize them. Accidental leakage during fossil fuel extraction and transportation demonstrates our dependence on these energy sources. Methane is also a product of waste generation and disposal, in the form of sewage, livestock effluent, landfills, and incineration. In the context of the movement to make our society more sustainable, CH_4 emissions are a key indicator of efficiency. For example, since methane can itself be used to produce energy, the recovery and use of landfill or livestock CH_4 has been cited as a way to save money and protect the environment (Liptay *et al.* 1998, Dauncey and Mazza 2001).

Besides carbon dioxide, five other greenhouse gases are mentioned by the Kyoto Protocol, including methane. While land use sinks have been widely discussed for CO_2, *e.g.* forests, this approach has not yet been applied to methane. Instead, the focus has been on cutting emissions at the source. Projects proposed and implemented so far have involved capturing CH_4 from landfill or agricultural sources, as in

Vermont's climate change action plan (Anderson *et al.* 2001, Dauncey and Mazza 2001, Environmental Protection Agency 2001).

3.4 Considerations for future policy development

As of yet, removal mechanisms for tropospheric methane have not figured in policy discussions. However, there may soon be concerns about protecting both the atmospheric and soil sinks. If the decline in worldwide hydroxl concentrations noted by Prinn *et al.* (2001) continues, there could be implications not only for greenhouse gas concentrations, but other pollutants, such as the precursors to smog. In addition, soils in many parts of the world are being damaged by erosion, agricultural or industrial contamination, and nutrient depletion, which may among other things have an adverse effect on their ability to oxidize methane.

Existing legislation would have to be modified, since there is a significant difference between sinks for methane, and for CO_2, as they are presently recognized in the Kyoto Protocol. Article 3.3 refers to situations where carbon is being sequestered, so it would not apply to methane. Because all of the methane which reacts with hydroxl radicals is transformed into carbon dioxide , rather being incorporated into living matter, the atmospheric sink does not serve as a long-term reservoirs for CH_4. In effect, it could be termed a "non-sequestration sink". Methane is transformed into carbon dioxide: still a greenhouse gas, but molecule-for-molecule, a less potent one. If the CO_2 situation is analogous to a vault with a time-release lock, the atmospheric methane sink, which accounts for the greatest proportion of the CH_4 in the system, is more akin to a filter: or better yet, a catalytic converter.

Regarding the soil sink, although some of the CH_4 taken up by methane-oxidizing bacteria is transformed into microbial biomass, most of it (57 to 69%) is oxidized immediately (Roslev *et al.* 1997). More work needs to be done to determine the longevity of microbially-fixed CH_4 in the soil, after its transformation into proteins, nucleic acids and polysaccharides, and lipids -- experiments have determined that some (1-3% per day) is respired as CO_2 relatively quickly, and that microbial predation by other organisms may also be contributing to carbon turnover (Roslev *et al.* 1997). Carbon sequestration models which are based on the decay rates of different organic compounds, *e.g.* lignin in woody vegetation, may not be adequate to simulate these processes.

Theoretically, modifications could be made via Article 3.4, which allows other land use and land use change activities to be added. In the case of methane, this would presumably be done with reference to Article 3.7, which refers to non-CO_2 greenhouse gases in terms of "CO_2-equivalents." This would make monitoring and evaluation of atmospheric trace gas fluxes more comprehensive, but correspondingly more complex.

Even though methane is not as significant to the overall greenhouse effect as carbon dioxide, its shorter residence time in the atmosphere could provide a valuable model system for testing mitigation policies, since the time lag for changes in methane sources and sinks to be reflected in atmospheric concentrations is also much briefer.

For this particular example, science and policy appear to be asynchronous. Arguably, the problem for most environmental issues has been that policy lags far behind the science -- the concerns about acid rain in the 1970s and Persistent Organic Pollutants (POPs) in the 1990s are examples. But in this case, the gap is much narrower, and the policy framework may even be getting ahead in some cases.

Research on methane sources and sinks in aquatic, marine, and terrestrial environments has proceeded apace for the past three decades or more. However, despite the contributions of Schoell (1988), Seiler *et al.* (1984), Conrad and Rothfuss (1991), Fung *et al.* (1991), and many others, there is still much to be learned about the global methane budget. There are many uncertainties about how environmental changes may alter the processes which create and destroy CH_4 -- the impacts of other pollutants such as anthropogenic nitrogen compounds, or even rising temperatures (*e.g.* Luo *et al.* 2001), have yet to be fully investigated.

Similar problems have been noted for CO_2 itself. The global carbon cycle is still being mapped out, with more work to be done on the fate of carbon in the oceans and soil. Many environments are still being investigated. The finding that old-growth forests can sequester more carbon than new plantations, for example, has significant implications for both climate change and resource management policies (Schulze *et al.* 2000). Formulating clear (and enforceable) legislation, whether at the international or local level, requires information which is accurate as well as specific. In the absence of a baseline or reliable measurement techniques, it will be difficult to assess variables such as carbon sequestration -- or in the case of atmospheric methane, oxidation -- and even more difficult to assign economic values to them, *e.g.* for emissions trading purposes. In addition, research on how particular ecosystems may react to projected changes over the next century, such as higher temperatures, altered rates of disturbance, and shifts in the hydrological regime, is far from complete.

CHAPTER 4 - SOIL UPTAKE OF METHANE IN AN OAK WOODLAND

4.1 The Victoria Case Study

Current policies on greenhouse gas mitigation frequently incorporate or rely heavily upon carbon dioxide sequestration. While this is a convenient approach to the problem, it also simplifies the situation to the extent that may encourage a "one size fits all" type of mindset, which may not be suitable for other gases. The following example looks at methane, and the soil sink associated with one particular type of ecosystem, a temperate deciduous open woodland.

The main focus of this field research was to evaluate methane uptake rates for several terrestrial environments representing conditions in the Pacific coastal temperate zone. An attempt was also made to consider different land use regimes, and the amount of time required for soils to return to their pre-cultivation flux values after abandonment. Two upland forest types characteristic of southern Vancouver habitats were sampled: a closed-canopy mixed forest and a more open deciduous oak woodland. An abandoned field in the same vicinity had likely supported vegetation from both these environments before being cleared of trees sometime in the 1880s (Jupp 1980). The field may originally have been used for pasturing cattle at the beginning of the 20th century, but was later put into cultivation, and produced crops of hay up until the 1960s (pers. comm. Tony James 1998). A groomed and irrigated portion of the campus lawns was also sampled.

Measurements representing a wide variety of environmental types are necessary in any attempt to quantify regional or global methane budgets. This area includes habitats which have not been investigated in terms of methane fluxes, to the same degree as temperate and tropical wetlands, or subarctic forests and tundra. Although West Coast ecosystems occupy less geographical area than some of these other habitat types, their vegetation, soil, and climate characteristics are similar to regions elsewhere in North America, and on other continents. Hence these local settings could serve as analogs for future research into global environmental change. In addition, many areas along the Pacific coast have come under increasing development pressures during the past century: Vancouver Island, the Lower Mainland, and Puget Sound have all experienced rapid urban growth, and significant alterations in land use patterns since the early 1900s. Given the findings of earlier research into methane and soils, it is important to consider the effects of different management practices. By including cultivated land which has been abandoned, or converted to other uses, the data may indicate the degree to which human impacts have influenced methane uptake in local ecosystems, and allow an assessment of potential recovery times.

4.2 Environmental description

The city of Victoria, British Columbia, is located on the west coast of Canada, on the south-eastern tip of Vancouver Island (48° 27' N, 123° 18' W), at the base of the Saanich Peninsula. It is separated from the Lower Mainland by the Strait of Georgia, and from Puget Sound and the Olympic Peninsula by the Juan de Fuca Strait (Figure 4.1).

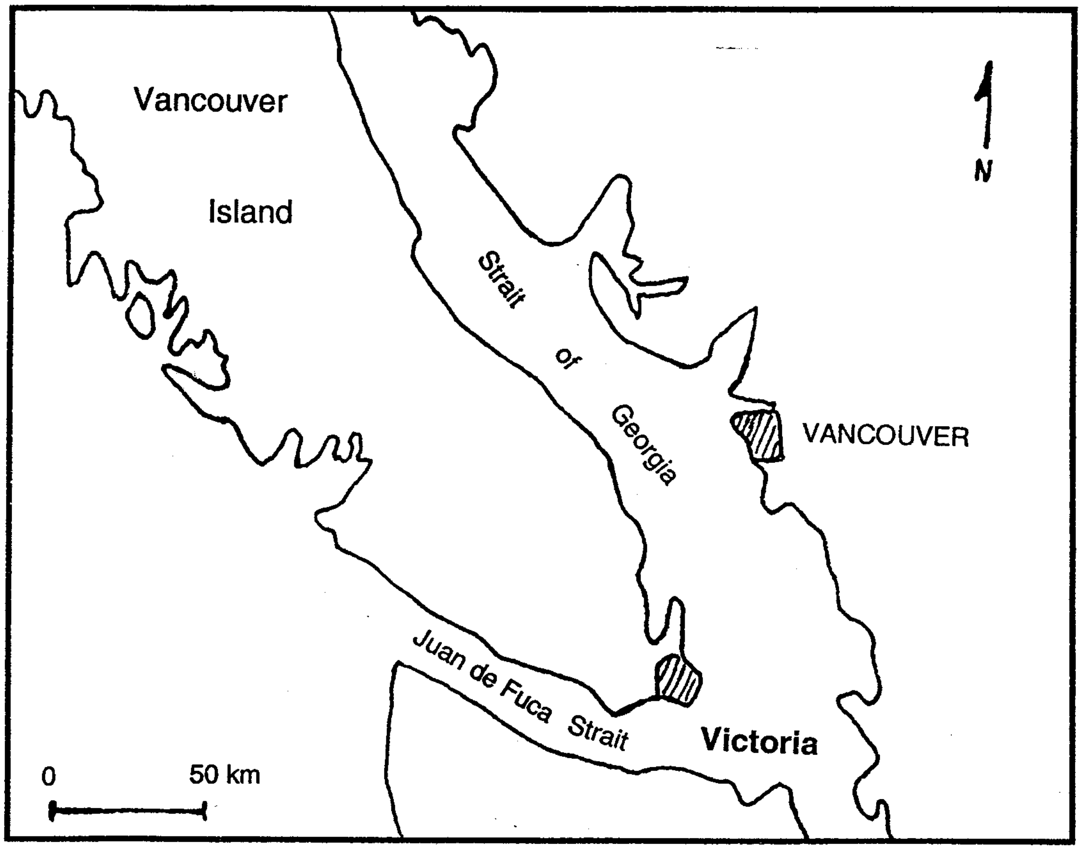

Figure 4.1 - The Victoria B.C. region

4.2.1 Regional climate

Similar to more southerly Pacific coastal environments, Vancouver Island experiences a summer precipitation minimum and accompanying water deficit conditions. For a few weeks each July, Victoria is one of the driest locations in Canada (Pavlick 1986). The onset of water deficiency in soils begins around June 2, assuming a standard storage capacity equivalent to 3 inches (7.6 cm) of precipitation: timing can vary by approximately two weeks either way (Day *et al.* 1959). In Victoria the water deficit period lasts 4 months, the longest in the province (Klinka *et al.* 1979). Figure 4.2 depicts monthly temperature and insolation measured at Victoria International Airport near Sidney, while Figure 4.3 shows monthly precipitation. Both graphs are based on the 1961-1990 climate normals (Environment Canada 1993).

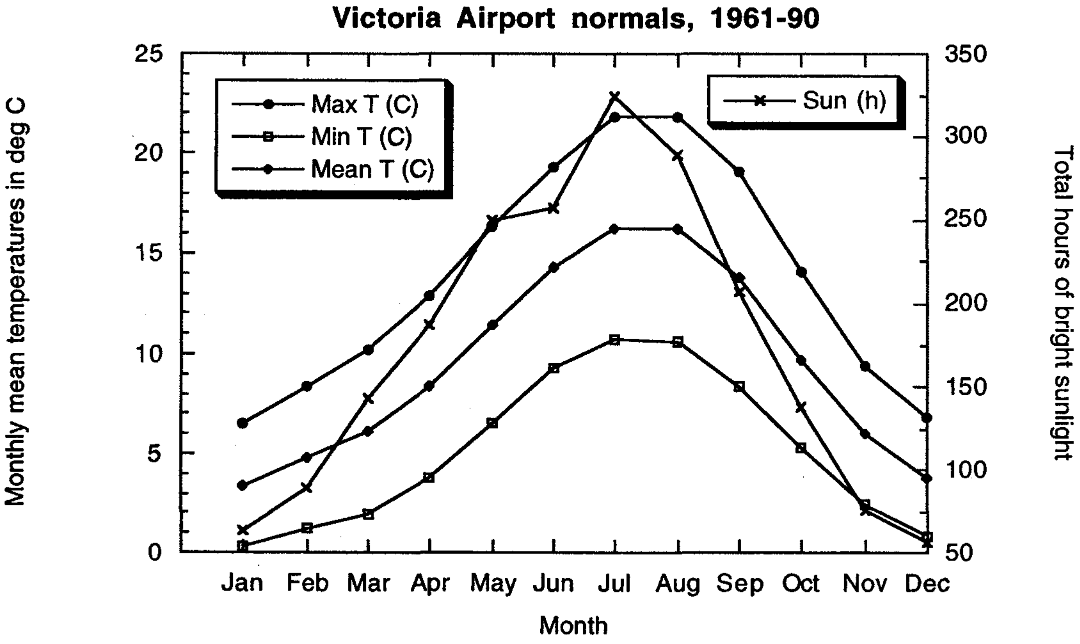

Figure 4.2 - Victoria Airport mean monthly temperature normals

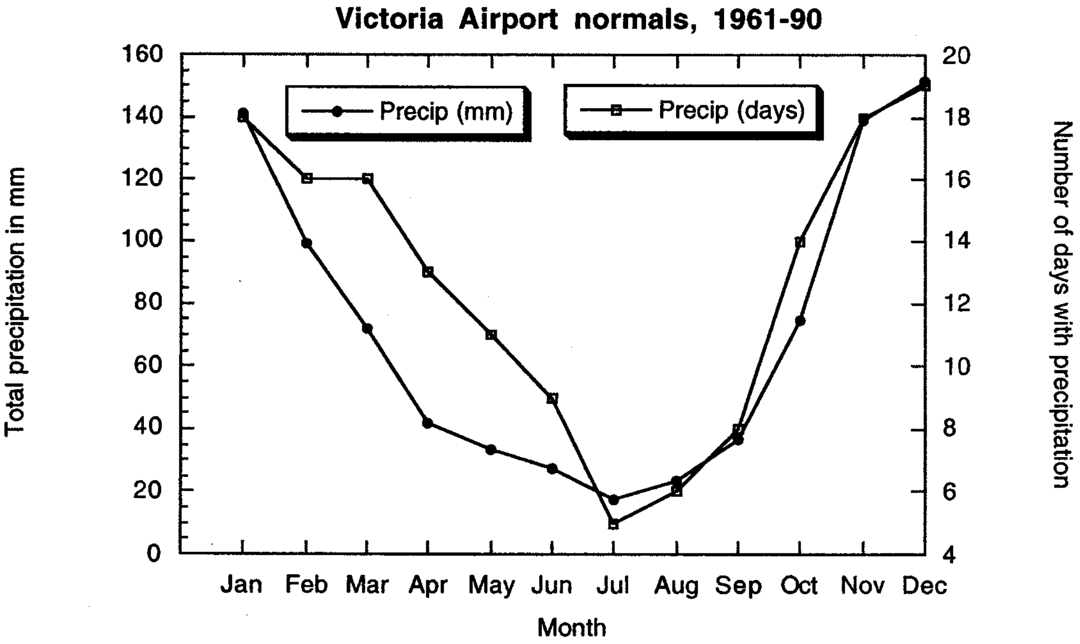

Figure 4.3 - Victoria Airport monthly precipitation normals

There are two main reasons for this seasonal anomaly: topography, and global climate patterns. The entire west coast of the continent, including B.C., is dominated by a system of mountain ranges. This creates high variability in precipitation, over local and regional scales (Pittock 1977, Tuller 1987). Due to its position on the windward side of the mountains, Vancouver experiences orographic effects, resulting in average annual precipitation in excess of 1 400 mm (Kendrew 1953). However, much of the southern tip of Vancouver Island lies within rainshadow areas: in the lee of the Vancouver Island Range, sheltered from the prevailing northwesterlies (Tuller 1987), while the mountains of the Olympic Peninsula serve as another barrier to the south (Pavlick 1986, Chilton 2000). As a result, this site is atypical of climates in coastal B.C., and in terms of precipitation may be more similar to areas further south.

North and east of Victoria, the rainshadow effect diminishes with distance: average precipitation at the Victoria International Airport, 25 km away, routinely exceeds downtown totals by more than 200 mm per year (Environment Canada 1993). This extra moisture is sufficient to delay onset of the summer water deficit by a week (Day *et al.* 1959).

Superimposed over topographic factors are broad-scale climate patterns, caused by planetary circulation. Dry summers and wet winters are characteristic of west coast climates, although the relative lengths of these periods vary, depending on latitude. Elsewhere in North America, *e.g.* the Great Lakes basin, precipitation is evenly distributed throughout the year (Kendrew 1953).

In coastal British Columbia, two major circulation features, the Aleutian Low and the Hawaiian (North Pacific) High, dominate seasonal weather patterns. The North Pacific High, located over the Pacific between 35°-45° N (Hordon 1967), is a component of Hadley cell circulation in the lower latitudes. Air flowing down from the high undergoes adiabatic warming, becoming hotter and drier as it descends, and creating the warm, clear conditions typical of summertime Mediterranean climates, *e.g.* coastal California and Oregon. Southeast Vancouver Island represents the highest-latitude occurrence of true Mediterranean climate in the world (Day *et al.* 1959).

When the North Pacific High reaches its maximum strength in July and August, its influence extends as far north as the Alaskan panhandle. After the onset of the autumnal equinox in September, the North Pacific High wanes and precipitation increases as low-pressure systems from the Gulf of Alaska sweep down, following the westerly flow at the southern edge of the Arctic front. In Victoria the average number of days with precipitation is 18 for December, compared with only 5 in July (Environment Canada 1993).

4.2.2 Local vegetation

The study area is classified as part of the coastal Douglas-fir (CDF) biogeoclimatic zone (Klinka *et al.* 1979, Pavlick 1986), within the drier maritime climate subzone (van Vliet *et al.* 1987). This region, including Victoria and the Saanich Peninsula, experiences the drier conditions characteristic of eastern Vancouver Island. These differences are reflected in the local vegetation, which is sparser than the temperate rainforest on the western side of the Island.

Two main types of natural vegetation are found near the University of Victoria campus: open woodland or parkland, consisting of deciduous Garry oaks *(Quercus garryana)*, and sometimes arbutus *(Arbutus menziesii)* scattered over grassy uplands;

and a closed-canopy mixed forest of broadleaf maple *(Acer macrophyllum)* and red alder *(Alnus rubra)* interspersed with conifers such as Douglas-fir *(Pseudotsuga menziesii)*, western red cedar *(Thuja plicata)*, western hemlock *(Tsuga heterophylla)*, and grand fir *(Abies grandis)*, with undergrowth consisting mainly of small shrubs like Oregon grape *(Mahonia nervosa)* and salal *(Gaultheria shallon)*, and clumps of ferns. Also, a narrow band of moisture-loving trees, *e.g.* western red cedar and western hemlock, is found inside Mystic Vale, a deep ravine which runs through the university grounds. A small grove of mature Garry oaks is located on campus, near the intersection of Gordon Head Road and Cedar Hill X-Road, but due to its limited area and the amount of foot traffic which the campus grove experiences, the representative Garry oak site was selected from the more extensive woodland nearby on Mt. Tolmie. These trees appear stunted in comparison to the campus specimens, but this seems to be typical of Garry oaks growing in exposed areas, *e.g.* Gonzales Hill and Mt. Douglas.

Some species, such as Douglas-fir, have a broad ecological niche which allows them to grow in either habitat, although in this area they are more evident in the closed-canopy forests than in the Garry oak meadows. The part of the forest selected for sampling, at the edge of Mystic Vale on the eastern side of the campus, was characteristic of areas which are more mesic than the well-drained sediments surrounding rock outcrops like Mt. Tolmie, but drier than the valleys and depressions. Douglas-fir and broadleaf maple dominated, while water-loving species like western red cedar and red alder were scarce.

Two kinds of ground cover were observed under the forest canopy: moss with small ferns, and dry litter consisting mainly of conifer twigs and needles. These were both accounted for in the sampling procedure. Canopy height was estimated to be at least 5-10 metres in the mixed forest, while the individuals and loose clusters of oaks on Mt. Tolmie were considerably shorter, usually in the 3-6 metre range. Both of these types of settings would appear to meet the crown cover and height criteria for forest environments, as specified in the Marrakesh Accords (UNFCCC 2002b), except for the exposed bedrock areas on Mt. Tolmie.

The Mediterranean climate has a significant impact on both the mixed forest and open oak woodland environments. Brix (1972) noted that tree growth slows with the onset of the summer drought period, resulting in the formation of darker latewood. The water deficit is even more obvious in the Garry oak meadows: as mid-June approaches, the last of the flowering forbs set seed, and the grass begins to turn yellow. The meadows produce new growth when the winter rains return in late October.

4.2.3 Local soils

The most recent Canada Soil Inventory (Agriculture Canada 1992) identifies the Victoria area (Cell 820), comprising approximately 62 square kilometres, as predominantly sombric brunisols, with some gleysols. The parent material is classified as gravelly loam, originating from glacial moraine. Regional studies focusing on Vancouver Island soils, *e.g.* van Vliet *et al.* (1987), describe a variety of soil series, not only brunisols and gleysols, but luvisols, humisols, and podzols. The last comprehensive soil survey covering the Capital Regional District, including urban Victoria, was carried out in the late 1950s.

Although some of these designations may be obsolete by now, the soils documented by Day *et al.* (1959) will be used for descriptive purposes. In and around the study area, two soil series in particular are identified: Langford (surrounding the rock outcrop of Mt. Tolmie), and Tolmie (believed to constitute most of the main University of Victoria campus). Day *et al.* (1959) classified the Tolmie series as a "Dark Grey Gleysol", a term which was changed to "Humic Gleysol" in the 1960s (Soil Classification Working Group 1998). Van Vliet *et al.* (1987) add more details about the Tolmie series, which is also found on the Gulf Islands: they refine the classification to orthic humic gleysol. Also, they note that it is frequently colonized by red alder. Individuals and small groves of this species are evident around the campus, suggesting that despite years of agriculture and extensive human modification, mesic conditions are still widespread.

Langford, along with two other local soil series (Cadboro and Esquimalt), is categorized as belonging to the "Black" group (Day *et al.* 1959). This name is problematic, since the term "black earth" is traditionally used to describe chernozem soils (Whittow 1984), which are characteristic of midlatitude grassland environments. Day *et al.* (1959) mention the existence of an oak-grass association several times. However, this suggestion appears to have been revised in the past few decades: although the dark, crumbly surface horizons may appear similar at first glance, the resemblance is superficial. The dark grey to greyish-brown upper ("A") soil horizon is physically, chemically, and biologically different from true chernozem soils.

According to the 3rd edition of the Canadian System of Soil Classification (Soil Classification Working Group 1998), among the current requirements for a chernozemic soil are the following points of agreement: an A horizon more than 10 cm deep, high carbon content (1-17%), and a C:N ratio less than 17. Some of Day *et al.*'s "Black" soils only appear to meet the first two criteria. The C:N ratio for Langford loam can exceed 19 (Day *et al.* 1959). Also, the pH of "Black" A-horizons is generally under 6.0 (*Day et al.* 1959), and while some chernozemic soils, *e.g.* eluviated brown chernozem, have moderately acidic upper horizons, the combination of low pH and a high C:N ratio would tend to suggest a brunisol (Soil Classification Working Group 1998). This may have something to do with the fact that both the climate and vegetation around Victoria bear a closer resemblance to coastal California than to the continental temperate grasslands.

One characteristic which both grasslands and Garry oak woodlands have in common is that they tend to accumulate humus in the Ah horizon over time, which increases soil moisture capacity (Erickson 1996). The typical soils for Garry oak habitats are orthic sombric brunisols, with a surface layer of "mull" (well-decomposed leaves), and a higher base saturation than soils in Douglas-fir dominated environments. In contrast, conifers tend to have "moder" or "mor" layers, exhibiting less decomposition due to chemical and structural differences in the litter (van Vliet *et al.* 1987). Douglas-fir habitats are often dystric brunisols, and being more acidic, may show a greater tendency towards podzolization. This trend is more evident in closed-canopy forests dominated by trees such as western red cedar: true podzols have been identified farther north on Vancouver Island.

The soil at the Mt. Tolmie research site is probably an orthic sombric brunisol similar to the Langford series (van Vliet *et al.* 1987). The on-campus sites, including the forest near Mystic Vale, the manicured lawns, and the campus community garden, appear to be orthic humic gleysols, possibly of the Tolmie soil series (van Vliet *et al.* 1987). With poorer drainage than the Mt. Tolmie site, the soil is frequently saturated during the wet season, and excavations in the community garden did reveal mottles below 30 cm, which are evidence of anoxic, reducing conditions for at least part of the year (Soil Classification Working Group 1998). The abandoned hayfield site on

Gordon Head Road likely consists mainly of orthic sombric brunisols, based on its close proximity to Mt. Tolmie and the conditions in the upper soil horizons.

4.3 Overview of campus and Mt. Tolmie sites

All measurements were carried out in the vicinity of the University of Victoria main campus, located at approximately 48°27'45" N, 123°18'45"W, elevation approximately 52 m above mean sea level (Figure 4.4). Sampling methodology and laboratory procedures are described in the appendix. Three main on-campus sites were selected for repeated measurements using small non-steady-state plastic portable chambers:

(1) Abandoned field behind the Visual Arts complex, near the intersection of Gordon Head Road and Cedar Hill X-Road

(2) Mixed forest just outside Ring Road, adjacent to Mystic Vale, along jogging path

(3) Manicured lawn within the inner ring, between the MacLaurin and Cornett buildings

In addition, two other campus sites were occasionally sampled, to add more variety:

(4) The campus community garden, across McKenzie Avenue from the tennis courts, fenced-in area behind Parking Lot 7

(5) Turfed areas adjacent to E-Hut and J-Hut (in close proximity to the lab)

The other sampling sites were located in Mount Tolmie Park (6), across Cedar Hill X-Road from the university. Mt. Tolmie (48°27'30"N, 123°19'15"W, elevation approximately 125 m) is the nearest upland environment which has not been modified by agriculture or construction, and is currently maintained by the Municipality of Saanich. The selected habitat was situated midway up the northeastern face of the hill, about 60 m into the park (measuring from the start of the footpath at the intersection of Cedar Hill X-Road and Gordon Head Road).

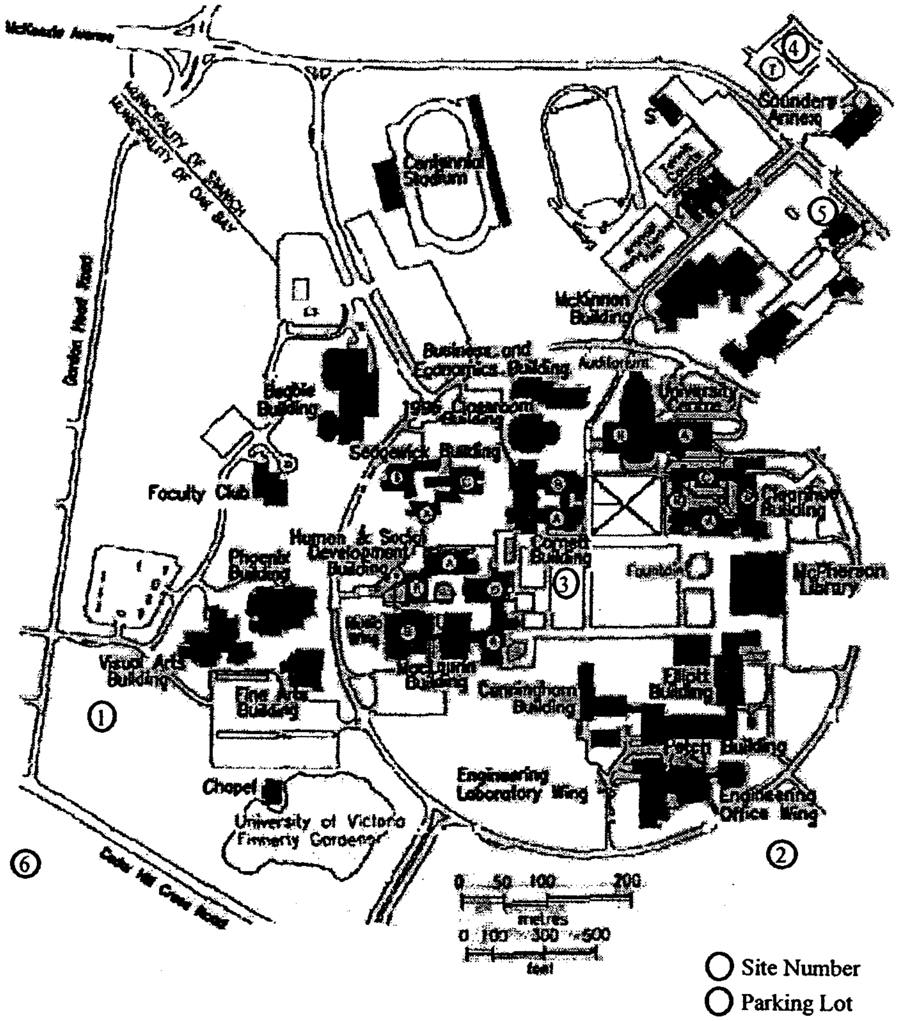

Figure 4.4 - Locations of main sampling sites

Source: Facilities Management, University of Victoria

Site Index:

(1) Abandoned hayfield near Gordon Head Road
(2) Mixed closed-canopy forest
(3) Groomed campus lawn between Garry oaks
(4) Campus community garden
(5) E-Hut lawn, and new turf on former J-Hut
(6) Garry oak woodland and meadow on Mt. Tolmie

Three sites (C, E, and F) were located in a clump of Garry oak trees 10 m off the footpath, and another one (D) was approximately 8 m away, in a flat open area. Site B was representative of the thin moss-covered soil layer on top of one of the many rock outcrops in the park, a couple of metres from the oak trees. All of these sites were marked, so the rings could be re-used. Early in the study period, one-time samples were taken in grassy areas on the other side of the path: one brown and desiccated, the other greener and more mesic.

The abandoned field site adjacent to Gordon Head Road was only sampled in 1996. The manicured lawn between the MacLaurin and Cornett buildings, near the mature Garry oak trees, was sampled in both 1996 and 1997. All other sites were visited only in 1997. Figures 4.5-4.17 are photographs of the six main locations, taken in early December 1999. The small white cylinders visible in most of the photographs are sampling chambers included to show relative scale (height: approximately 14 cm).

Figure 4.5 - Photo of old hayfield site (GHf)

Figure 4.6 - Photo of campus woods understory (CW)

Figure 4.7 - Photo of campus woods moss site (CWm)

Figure 4.8 - Photo of campus woods litter site (CWl)

Figure 4.9 - Photo of campus lawn site by oaks (CLo)

Figure 4.10 - Photo of campus lawn by MacLaurin (CLg,s)

Figure 4.11 - Photo of campus garden plot site (CGs)

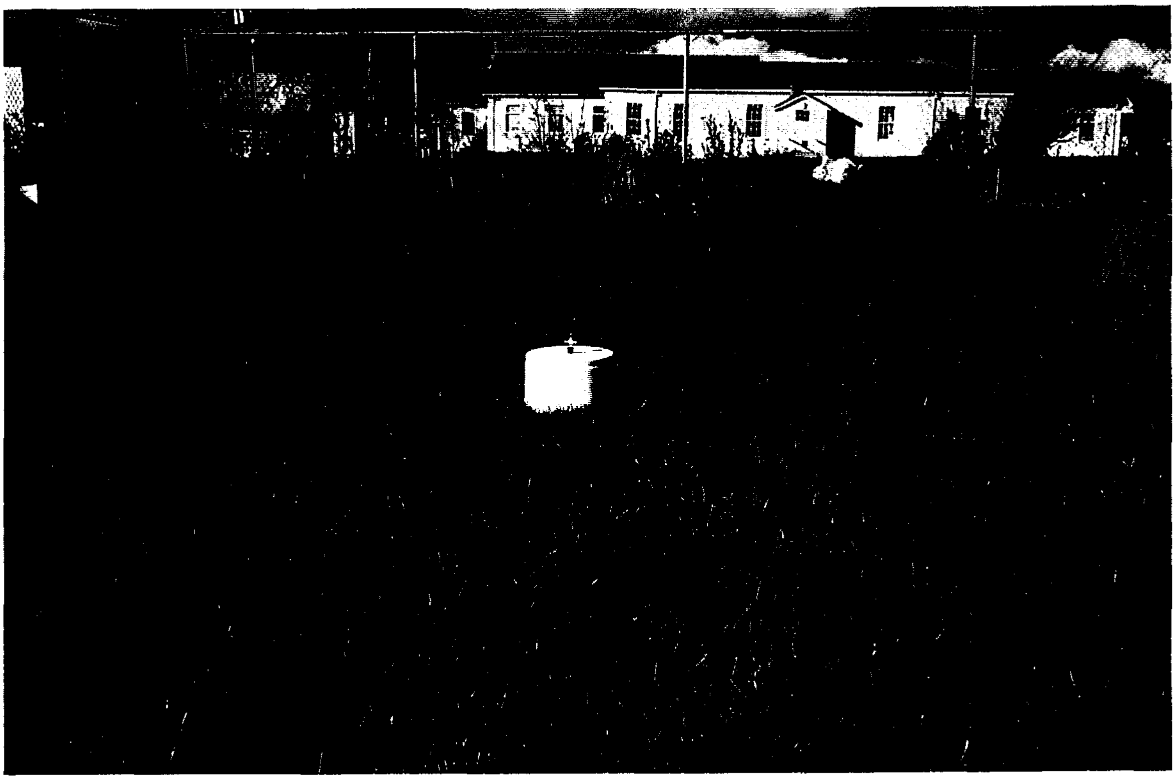

Figure 4.12 - Photo of campus garden grass site (CGg)

Figure 4.13 - Photo of new turf on former J-Hut (CLj)

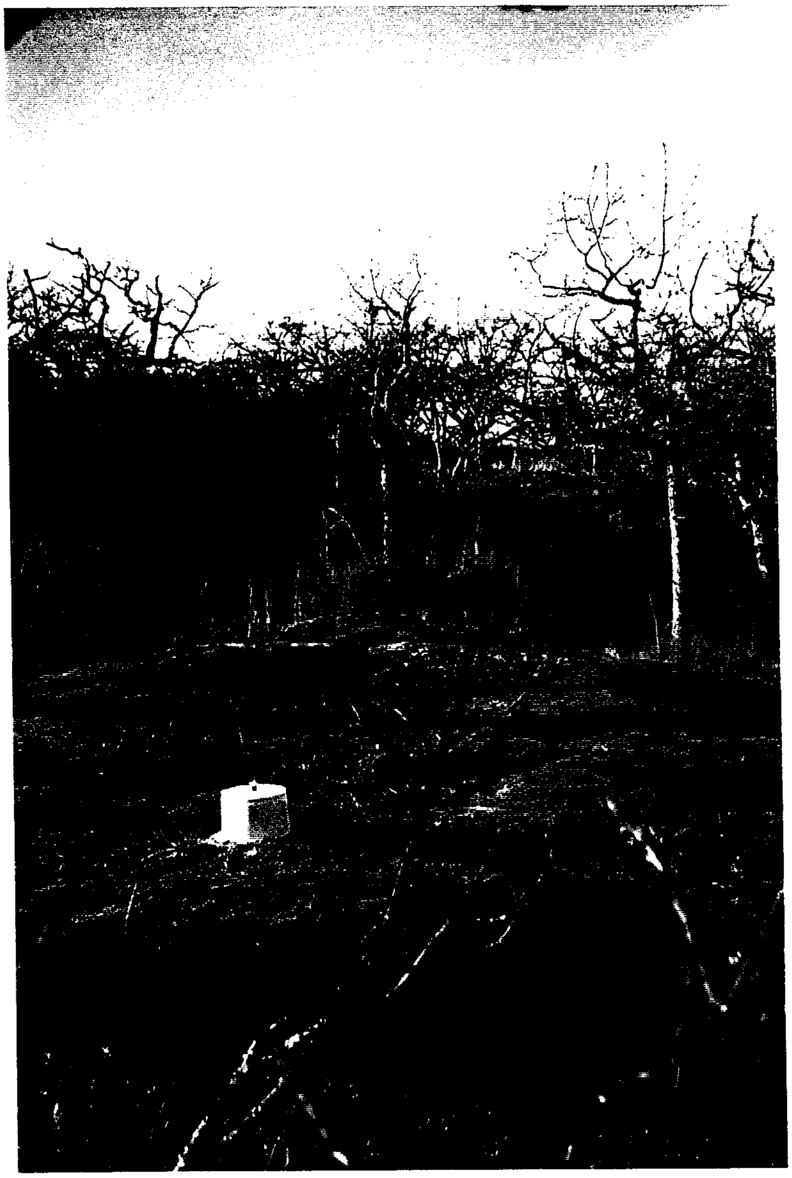

Figure 4.14 - Photo of Mt. Tolmie open oak woodland (MTB)

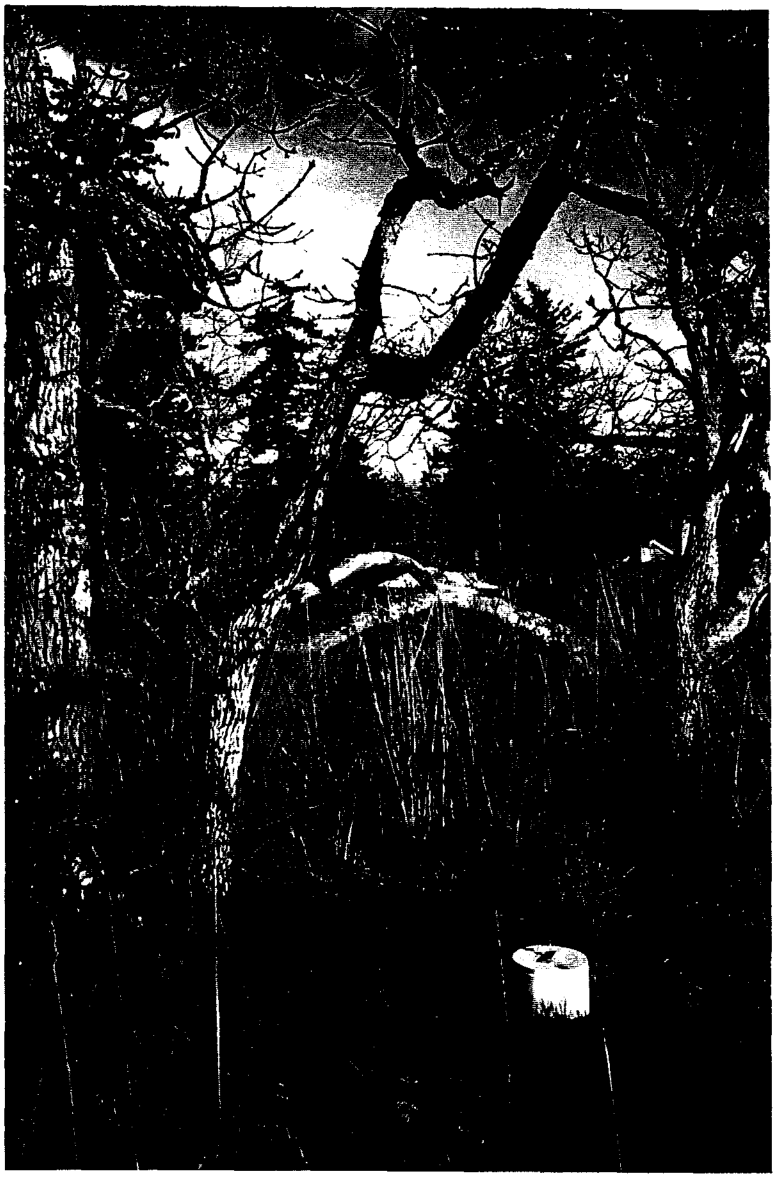

Figure 4.15 - Photo of Mt. Tolmie site under oaks (MTC)

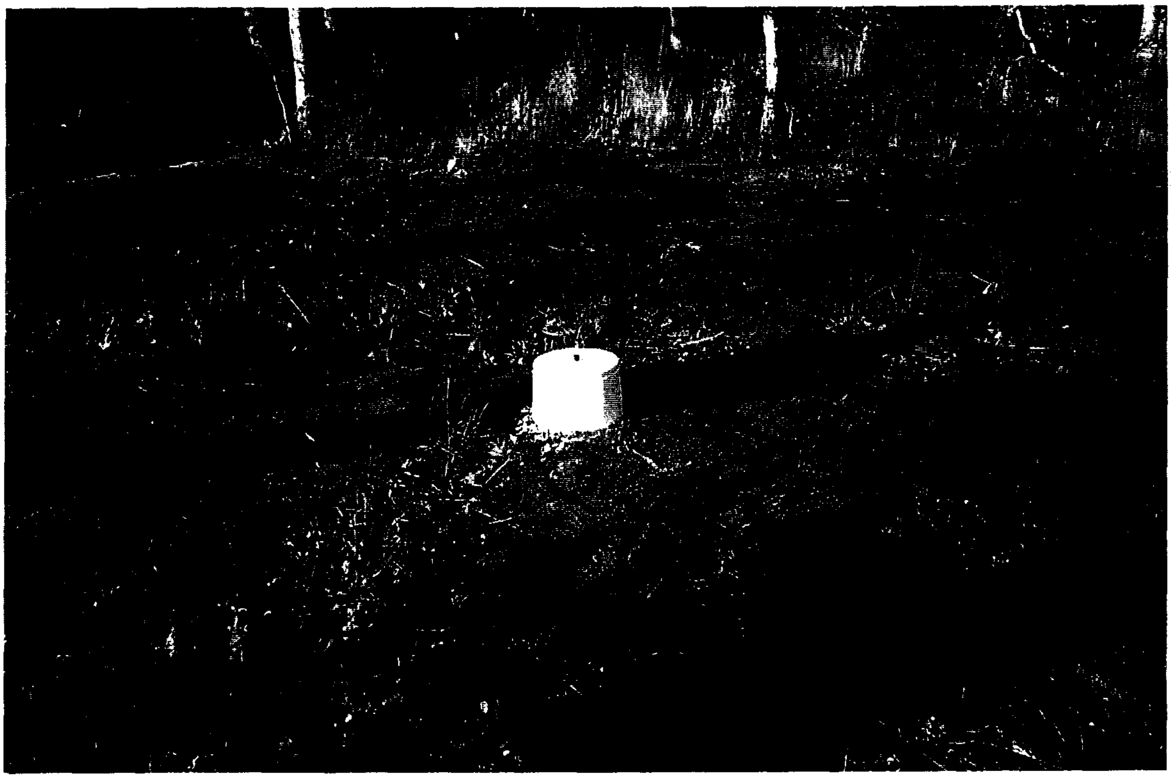

Figure 4.16 - Photo of Mt. Tolmie rock outcrop (MTB)

Figure 4.17 - Photo of Mt. Tolmie open meadow (MTD)

4.4 Results

In almost all the cases, methane concentration decreased over time within the sampling chambers: values at or below 1.2 ppm, substantially less than average atmospheric values, were measured on several occasions. The only locations which appeared to show increasing methane were a few recently-watered sites.

Table 4.1 presents the grouped hourly uptake rates (based on the 30-minute samples) for the various landscape types, expressed in micrograms$\cdot$m$^{-2}\cdot$h^{-1}, including medians and ranges where possible. "MT" indicates Mt. Tolmie, "GH" is the old hayfield site on Gordon Head Road, "CW" is the campus mixed woods, "CL" is the campus lawn, and "CG" is the campus community garden. An upper-case suffix, *e.g.* "C" in "MTC", signifies measurements from a particular site (and marked ring) on Mt. Tolmie. A lower-case suffix indicates that if there were multiple measurements, the chambers were installed in the same area, although not in the same ring as before. Approximate error values, utilizing the estimates worked out in the appendix, are +/- 32 mg$\cdot$m$^{-2}\cdot$h^{-1} for the Type A (1996) chambers, and +/- 48 mg$\cdot$m$^{-2}\cdot$h^{-1} for the Type B (1997) chambers.

Site	**Descript**	**Type**	**Median**	**Range**	**n**
MTF (1997)	Mt. Tolmie under oaks	unmodified	81.5	18.7-115.7	10
MTE (1997)	Mt. Tolmie under oaks	unmodified	79.2	58.7-112.6	7
MTg (1997)	Mt. Tolmie, green area	unmodified		71.9	1
GHf (1996)	old hayfield	abandoned	63.2	54.9-71.3	4
MTC (1997)	Mt. Tolmie under oaks	unmodified	59.1	11.5-96.9	10
CWl (1997)	mixed woods, litter	unmodified	41.6	23.2-72.5	6
MTb (1997)	Mt. Tolmie, brown area	unmodified		41.5	1
CWm (1997)	mixed woods, moss	unmodified	32.2	23.9-118.7	4
MTD (1997)	Mt. Tolmie open meadow	disturbed?	20.3	12.0-49.3	7
CGm (1997)	Garden, between plots	watered		19.7	1
CGg (1997)	Garden grass	abandoned	16.4	7.5-27.9	4
CLo (1997)	MacL-Cornett between oaks	watered	14.9	4.0-36.3	5
CLg (1997)	lawn near MacL	watering		6.7	1
CLo (1996)	MacL-Cornett between oaks	watered		6.3	1
CLd (1997)	lawn near Cunn.	dry		1.3	1
CLe (1997)	E-Hut lawn	watered		-1.5	1
CLs (1997)	lawn near MacL	watered		-6.7	1
MTB (1997)	Mt. Tolmie on rock	unmodified (poor seal?)		-3.1-11.5	2 (t=20 min)

CLj (1997)	J-Hut new turf	watered	-15.7	1

Table 4.1 - Summarized hourly methane uptake rates
(sampled at t=30 min, expressed as micrograms per square metre per hour)

Sites identified as being unmodified by human activity have the highest uptake rates in this study, while sites which are presently under some form of management are the lowest. Among the unmodified sites, the Garry oak meadow habitats generally have higher uptake rates than the mixed Douglas-fir/broadleaf maple forest on campus. Other authors have noted that broadleaf deciduous forests and woodlands tend to exhibit stronger methane oxidation properties than conifer-dominated environments (*e.g.* Steudler *et al.* 1989). It is already known that vegetation composition can influence soil processes: for example, litter from evergreens contributes to podzolization (Whittow 1984), due to high concentrations of organic acids. Also, differences in macrofauna (Brumme and Borken 1999) and bacterial activity have been observed between habitat types: there is less nitrification in coniferous forest soils (Killham 1994), especially under litter composed of "mor" (undecomposed twigs). Some connection between nitrifiers and methanotrophs has already been suggested.

The differences between sites which are currently being watered and fertilized, and those which are not, are less clear. Watered sites account for both high and low methane uptake within the managed environments. Some of these results might be explained by the fact that the campus lawn site near the MacLaurin building, which appears to exhibit a positive methane flux, had been thoroughly watered earlier that day. If conditions were at or near saturation, this could be inhibiting the methanotrophs due to lower oxygen diffusion rates, and allowing methane produced in deeper horizons to reach the surface. As will be discussed later, lack of water can also inhibit soil microbial activity (Roslev *et al.* 1997), so methane uptake rates may be highest within an optimal range of soil moisture.

There also may be some fertilization effects, since the garden plot which had been treated with chicken manure earlier that year exhibited a low uptake rate, relative to the adjacent area which had also received water but no fertilizer. However, caution is necessary before drawing conclusions, since the comparison area had been mown but not stripped of turf.

Some exceptions to the broad patterns of methane uptake are the "B" site on Mt. Tolmie, which sampled a thin coating, frequently less than 5 cm deep, of moss and soil on a rock outcropping. Although this environment has an even higher probability of the others of being unaltered by human activities, it has a low uptake rate. This may be due to its tendency to dry out for long periods of time, and the scarcity of soil. Also on Mt. Tolmie, the "D" site, which is dissimilar from the others located nearby, may not be as pristine as originally assumed. The surrounding area is unusually flat, compared to other areas on the hillside, and this might have been caused by human influence in the past (brush clearing, digging, or perhaps foot or vehicle compaction caused by hikers, joggers, or mountain bikers straying off the marked paths). Although natural "grass balds" do occur on the summits of hills around Victoria (Pavlick 1986), this may not be one of them, despite its open appearance.

Finally, the abandoned field by Gordon Head Road, behind the Fine Arts buildings, is a human-modified environment which has methane uptake rates comparable to undisturbed sites. It has had more than three decades to recover from agriculture, and since it was last used for haymaking in the 1960s (pers. comm. Tony James 1998), it may have experienced less tillage and fertilization, relative to more labour-intensive crops such as vegetables or soft fruit. The larger volume of the chambers used for measurement also results in an increased error estimate (about 1 mg/m^2 higher?), but even when this is taken into account, its ranking is little affected.

Based on the previous set of tables, the box plots in Figure 4.18 show the distribution of hourly methane uptake values for the major sites measured in 1997, based on 30-minute sampling intervals. The "MT" label indicates Mt. Tolmie sites (C-F), "CWl" is for litter surfaces in the campus mixed woods, and "CLo" is the site on the groomed campus lawn between the Cunningham and Cornett buildings, near the large Garry oaks.

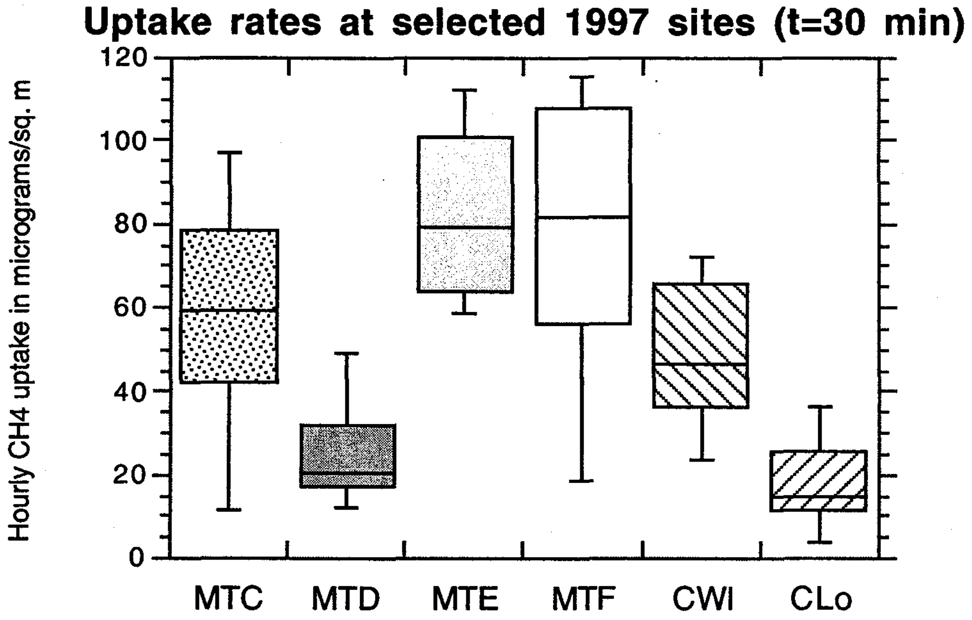

Figure 4.18 - Box plots of hourly methane oxidation rates

MTC = Mt. Tolmie under Garry oaks, C-ring

MTD = Mt. Tolmie open meadow, D-ring

MTE = Mt. Tolmie under Garry oaks, E-ring

MTF = Mt. Tolmie under Garry oaks, F-ring

CWl = Campus mixed closed-canopy woods, litter

CLo = Campus lawn, groomed area between oaks, Cunningham/Cornett

Figure 4.19 shows time-series graphs of the uptake rates calculated over 30-minute sampling intervals for the "C", "D", "E", and "F" rings on Mt. Tolmie, and for the CWl (campus woods, litter substrate) site, over several weeks. These sites were selected because the number of separate measurements was high enough to examine trends over time, and also because they were receiving all of their moisture in the form of precipitation. (The groomed campus lawn sites were being watered by sprinklers at least 3 times a week.)

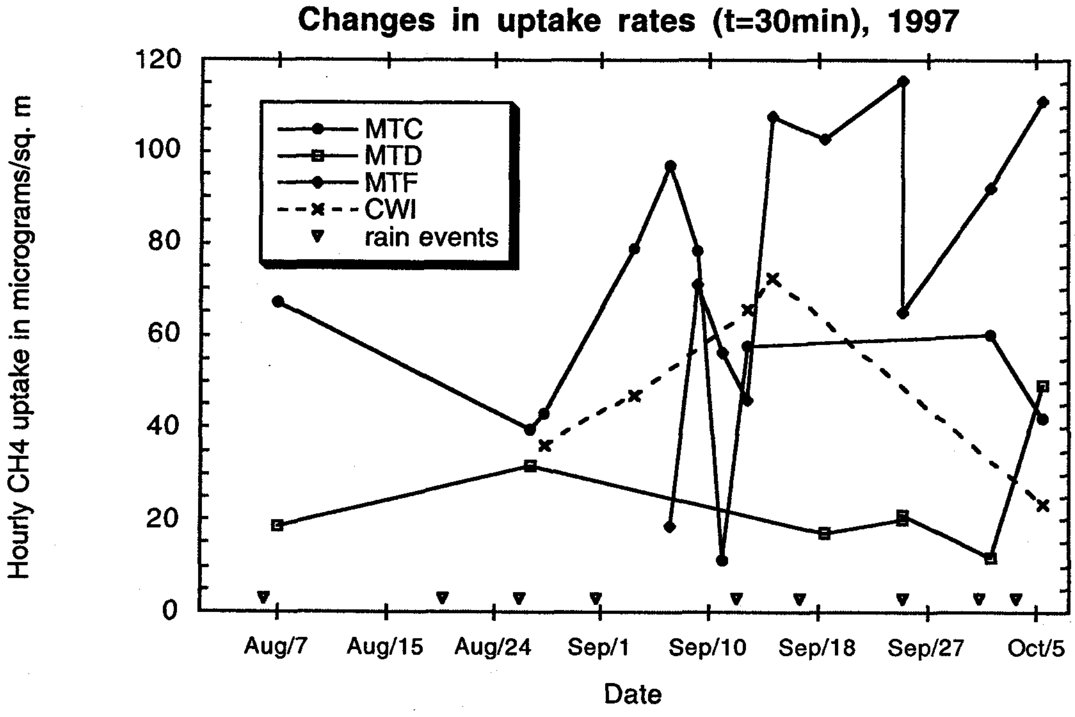

Figure 4.19 - Changes in uptake rates through summer 1997

From these examples, a pattern is apparent: a rapid decrease in methane uptake within 24 hours of a rainfall event, then recovery over the next few days. After more than a week without additional moisture, rates begin to decline again. For example, "C" exhibits its lowest uptake rates around 10 days after the last major rain event, when the surface horizon was observed to be quite dry. While these data are not really extensive enough to permit comparison of the oak woodland and mixed forest environments over the field season, it is interesting to note what appears to be a time lag at the forest site (peaking some time after the Mt. Tolmie sites). Also, over the same dates, variability in uptake rates (at least for "C", "E", and "F") seems to be more pronounced in the oak woodland than in the forest. Precipitation sufficient to dampen the surface layer of soil was observed on 3 days during the 1996 measurements (22 September; 2 and 14 October), and on 10 days during the 1997 season (21 July; 6, 20, and 26 August; 1, 12, 17, and 25 September; 1 and 4 October).

4.5 Annual calculations: a few scenarios

Table 4.1 presented estimates for hourly methane uptake rates in selected Victoria environments: Mt. Tolmie, the old hayfield by Gordon Head Road, the campus woods (litter and moss surfaces), and the manicured lawn around the large Garry oak trees (between the Cunningham and Cornett university buildings). From these results, it is possible to generate some methane uptake projections for the region around Victoria, using the median group values for the 30-minute sampling intervals. Table 4.2 presents daily estimates for the environments surveyed, assuming that methane uptake continues at similar rates for 24 hours, day and night. Where the sample size "n" was 1, the single value was used; for n=2, the data were averaged; and for n>2, the median was taken.

Site	**n**	**Description**	**Daily uptake (mg/m^2)**	**Daily uptake ($mmol/m^2$)**
CGg (1997)	4	Garden tall grass	0.392	0.024
CGm (1997)	*1*	*Garden between plots*	*0.473*	*0.030*
CGs (1997)	*1*	*Garden plot soil*	*0.518*	*0.032*
CLd (1997)	*1*	*Cunn lawn dry*	*0.032*	*0.002*
CLe (1997)	*1*	*E-Hut lawn*	*-0.036*	*-0.002*
CLg (1997)	*1*	*MacL lawn groomed*	*-0.161*	*-0.001*
CLj (1997)	*1*	*J-Hut new turf*	*-0.376*	*-0.024*
CLo (1996)	*1*	*MacL-Cornett between oaks*	*0.152*	*0.009*
CLo (1997)	5	MacL-Cornett between oaks	0.358	0.022
CLs (1997)	*1*	*MacL lawn new-watered*	*0.307*	*0.019*
CWl (1997)	6	Mixed woods litter	0.998	0.062
CWm (1997)	4	Mixed woods moss	0.773	0.048
GHf (1996)	5	Old hayfield	1.517	0.095
MTb (1997)	*1*	*Mt. Tolmie dry*	*0.996*	*0.062*
MTB (1997)	*2*	*Mt. Tolmie rock*	*0.101*	*0.006*
MTC (1997)	10	Mt. Tolmie oaks	1.418	0.088
MTD (1997)	7	Mt. Tolmie meadow	0.486	0.030
MTE (1997)	7	Mt. Tolmie oaks	1.900	0.118
MTF (1997)	10	Mt. Tolmie oaks	1.957	0.122
MTg (1997)	*1*	*Mt. Tolmie green*	*1.730*	*1.108*

Table 4.2 - Estimated daily methane uptake rates
(italics indicate only 1 or 2 samples)

B.H. Levelton and Associates (1991) proposed a net methane uptake value of approximately 41 kilotonnes per year for the entire province of British Columbia. They based their calculations on an estimated area of 900 000 square kilometres of aerobic soils, and an average oxidation rate of 0.6 $mg \cdot m^{-2} \cdot d^{-1}$ for half the growing season (75 days), presumably to accommodate conditions in northern and alpine environments. The values listed in Table 4.2 varied above and below that estimate, depending on the habitat: similar for unmodified Douglas-fir forest, much lower for landscaped areas, and much higher for unmodified Garry oak woodland. In terms of B.C.'s anthropogenic CH_4, 41 kilotonnes is less than 8% of 1990 emissions (B.H. Levelton and Associates 1991). Table 4.3 lists annual uptake rates, based on 30-minute sampling intervals and a 180-day season, for a variety of different scenarios. Error values are +/- 207 milligrams per square metre: only the 1997 measurements from Type B chambers were used this time.

Scenario 1: coastal Douglas-fir biogeoclimatic zone, unmodified vs. irrigated lawns

Land area: 2 800 km^2

Site type	**CH_4 uptake ($mg/m^2/d$)**	**180-day total (tonnes)**
Mixed woods (litter)	1.00	503.0
Mixed woods (moss)	0.77	389.7
Irrigated lawn	0.36	180.5

Scenario 2: Garry oak habitat range, before and after development, unmodified vs. irrigated lawns

Land area: 100 km^2 "F" (before); 10 km^2 "F" + 90 km^2 lawn (after)

Site type	**CH_4 uptake (mg/m^2/d)**	**180-day total (tonnes)**
Oak woodland Mt. Tolmie "F", vs.	1.96	35.2 (100 km^2)
* *Mt. Tolmie "F"*	*1.96*	*3.5 (10 km^2)*
* *Irrigated lawn*	*0.36*	*5.8 (90 km^2)*
* Developed total		9.3 (100 km^2)
Oak woodland Mt. Tolmie "D", vs.	0.49	8.8 (100 km^2)
** *Mt. Tolmie "D"*	*1.96*	*0.9 (10 km^2)*
** *Irrigated lawn*	*0.36*	*5.8 (90 km^2)*
** Developed total		6.7 (100 km^2)

Scenario 3: "Black" soil type, before and after development, unmodified vs. irrigated lawns

Land area: 63 km^2 (Day et al. 1959)

Site type	**CH_4 uptake (mg/m^2/d)**	**180-day total (tonnes)**
Oak woodland Mt. Tolmie "F"	1.96	22.2
Oak woodland Mt. Tolmie "D"	0.49	5.5
Irrigated lawn	0.36	4.1

Scenario 4: Dark Grey Gleysol soil type, before and after development, unmodified vs. irrigated lawns

Land area: 246 km^2 (Day et al. 1959)

Site type	**CH_4 uptake (mg/m^2/d)**	**180-day total (tonnes)**
Mixed woods (litter)	1.00	44.2
Mixed woods (moss)	0.77	34.2
Irrigated lawn	0.36	15.9

Scenario 5: Tolmie soil series, before and after development, unmodified vs. irrigated lawns

Land area: 89 km^2 (Day et al. 1959)

Site type	**CH_4 uptake (mg/m^2/d)**	**180-day total (tonnes)**
Mixed woods (litter)	1.00	16.0
Mixed woods (moss)	0.77	12.4
Irrigated lawn	0.36	5.7

Table 4.3 - Projected annual methane uptake rates for various scenarios

The 180-day period used to calculate the annual values for Table 4.3 is just short of 6 months, approximating the time between early May and late October, which encompasses the greater part of the growing season in Victoria. It includes the June-September moisture deficit, and several weeks on either side when the soil is drying out or regaining moisture. By comparison, Steudler *et al.* (1989) chose a period of 120 days for boreal environments, and 200 days for temperate locations.

In many years, the onset of spring in coastal British Columbia may commence as early as April in some areas, since by that time the soil is warm and dry enough to permit successful seed germination of cool-weather crops. The measurements collected during this study are most likely to be applicable during the period from late spring until early fall. Conditions are frequently saturated during the winter rains, especially for poorly-drained soils such as gleysols.

4.5.1 Scenario 1 (CDF biogeoclimatic zone, unmodified vs. landscaped)

The coastal Douglas-fir (CDF) biogeoclimatic zone, in which Victoria is located, comprises 0.3% of the province's land area, or approximately 2 800 km^2 out of over 930 500 km^2 (Colombo 1978). If we assume that the uptake rates measured in the campus Douglas-fir/broadleaf maple forest are representative (1.00 mg·m^{-2}·d^{-1}, on a litter substrate), the amount of methane oxidized over a 180-day season would be approximately 503 tonnes. In comparison, 2 800 km^2 of irrigated lawns (0.36 mg·m^{-2}·d^{-1}) would only oxidize 181.4 tonnes, or less than half that amount. In Scenario 1, if the entire coastal Douglas-fir biogeoclimatic zone were converted into irrigated lawns, annual methane uptake by that area of land could be decreased by between 208.2 and 321.6 tonnes , or roughly 46 to 64%, depending on whether moss or forest litter substrate is dominant. However, it is unlikely that the entire zone would ever be completely landscaped, despite the rapid pace of development on the east coast of Vancouver Island.

4.5.2 Scenario 2 (*Q. garryana* range, unmodified vs. landscaped)

Scenario 2 is closer to reality, focusing on the conversion of Garry oak habitat. Only a small fraction of the coastal Douglas-fir zone is or has been oak woodland. Prior to European settlement, Garry oak habitat in this region may have occupied a hundred square kilometres, but has dwindled by as much as 90% (Erickson 1996). This scenario compares 100 km^2 of unmodified oak woodland to a land-use mixture approximating today's situation, 10 km^2 of oaks and 90 km^2 of irrigated lawns. When the maximum uptake values from Mt. Tolmie ("F" site), 1.96 mg·m^{-2}·d^{-1}, were used to represent undisturbed Garry oak habitat, estimated annual methane uptake dropped from 35.2 to 9.3 tonnes, a decrease of around 74%. However, if the minimum uptake values observed for that environmental type (Mt. Tolmie "D" site) were substituted, the decline would be an considerably less, dropping from 8.8 to 6.7 tonnes per year, by approximately 24%. This high amount of variation suggests that more research is needed into the heterogenity of soil environments, in this and other regions.

4.5.3 Scenario 3 (Regional "Black" soils, unmodified vs. landscaped)

For Scenario 3, the "Black" soils identified by Day *et al.* (1959), which were associated with Garry oak woodland, were assumed to have been entirely converted to irrigated grass. The soil series classified as "Black" (Cadboro, Esquimalt, and the Langford soil around Mt. Tolmie) were estimated to cover a total area of 15 600 acres, or 63 km^2 (Day *et al.* 1959). The change in CH_4 uptake would have gone from 22.2 tonnes per year to only 4.1 (a nearly 82% drop); or from only 5.5 down to 4.1 (-25%), if the conditions from the Mt. Tolmie D site are assumed to be more representative.

4.5.4 Scenario 4 (Regional "Dark Grey Gleysol" soils, unmodified vs. landscaped)

Scenario 4 examines another soil group, the Dark Grey Gleysols, which are more extensive than the "Black" soils, covering 7.4% of southeastern Vancouver Island and the Gulf Islands: 52 607 acres, equivalent to 246 km^2 (Day *et al.* 1959). Dark Grey Gleysols are now known as Humic Gleysols, in the Canadian System of Soil Classification (Soil Classification Working Group 1998). As in Scenario 1, the campus forest measurements were selected to represent unmodified conditions. After development, the decrease in methane uptake is estimated to be from 34.2 - 44.2 tonnes per year down to less than 16 -- a 54-64% drop, depending on whether the original conditions favour moss or forest litter.

4.5.5 Scenario 5 (Local Tolmie soil series, unmodified vs. landscaped)

In Scenario 5, only the Tolmie soil series is considered. It accounted for 21 930 acres, or 89 km^2 in Day *et al.*'s 1959 survey, and is the dominant type on the University of Victoria campus. More recently, the Tolmie series has been classified as an orthic humic gleysol (van Vliet *et al.* 1987). Under this scenario, annual methane uptake in tonnes would drop from 12.4 (moss) or 16.0 (forest litter), down to 5.7. In both cases, this would be less than half of the presumed original values (a decrease of 54-64%).

Given the relatively small amounts of land involved, all of these scenarios are minor, compared with the calculations from B.H. Levelton and Associates (1991), which estimated methane uptake at 41 000 tonnes per year for the entire province. Even when agricultural land on Vancouver Island is included, the numbers are still fairly small. Statistics Canada (1997a) assumes a total of 572 square kilometres of farmland, of which only about 188 km^2 is under crops: the rest is natural and seeded pasture, or summerfallow. If the measurements for watered lawns are used, the methane uptake of the total amount of farmland on Vancouver Island for a 180-day season would be around 34 tonnes. If the old hayfield measurements are used instead (although they would likely represent only the natural pasture component), uptake would be a maximum of 154.4 tonnes.

When all of these scenarios are compared to anthropogenic methane emissions within the province (Table 4.4), it becomes evident that Vancouver Island soils will not have much influence on net provincial totals. The estimated annual methane uptake of the entire CDF biogeoclimatic zone, plus the value for the original undisturbed Garry oak habitat range, would amount to less than 50% of the projected 2005 emissions of the smallest category, Stationary Combustion (1.2 kilotonnes). However, these findings do not include other environment types, or take conditions elsewhere in the province, *e.g.* the drier interior regions, into account. More research is needed, into the variability of methane oxidation rates between and within different environments.

Source	**1990 emissions (tonnes/a)**	**2005 estimate (tonnes/a)**
Landfills	290 405	282 891
Manure	58 315	72 657
Coal Mines	51 600	50 800
Oil and Natural Gas	60 843	83 657
Domesticated Animals	36 618	42 427
Prescribed Fires	17 736	12 415
Mobile Combustion	8 991	7 392
Incineration	6 795	6 788
Stationary Combustion	1 068	1 179
TOTAL	532 371	560 206

Table 4.4 - Anthropogenic B.C. methane emissions

Source: B.H. Levelton and Associates 1991

4.6 Methane uptake rates in other environments

The maximum uptake rates for this study were measured at the Garry oak woodland "F" site on Mt. Tolmie: an average value of 0.082 $mg \cdot m^{-2} \cdot h^{-1}$, using 30-minute sampling periods. The other natural vegetation type examined, the closed-canopy Douglas-fir/broadleaf maple forest on campus, had rates of around 0.042

mg·m^{-2}·h^{-1}, for a litter surface. Table 4.5 presents a summary of other researchers' findings on methane uptake rates, for various environments.

CH4 uptake (mg/m^2/h)	**Site type**	**Location**	**Source**
ALPINE			
0.002	ridge (snow)	Wyoming	Sommerfeld et al. 1993
0.004	meadow (snow)	"	"
0.005	meadow (bare)	"	"
0.091	steppe	"	"
TUNDRA			
0.050	sedge tundra	Alaska	King et al. 1989
0.083	dry tundra	N. Canada?	in Roulet et al. 1992
0.139	elevated tundra	N. Quebec	Adamsen & King 1993
0.011-0.065	spruce-lichen	"	"
TEMPERATE GRASSLAND			
0.208	wetland (drought)	Virginia	Harriss et al. 1982
~0-0.023	grassland	Germany	Born et al. 1990
0.035	shortgrass	Colorado	Mosier et al. 1991
0.020	shortgrass swale	"	"
0.020	***oak meadow (disturbed?)***	***Victoria B.C.***	***this study***
TEMPERATE FOREST			
0.028	beech-oak	Germany	Brumme et al. 1999
0.008	spruce	"	"
0.046	mixed woodland	N. Ontario	Lessard et al. 1994
0.010	hardwood, spodosol	New York	Yavitt et al. 1995
0.010-0.148	mixed forest	Germany	Born et al. 1990

0.112	pine/oak	Maine	Adamsen et al. 1993
0.115	hardwood	Mass.	Steudler et al. 1996a
0.097	hardwood (high N)	"	"
0.118	pine	"	"
0.068	pine (high N)	"	"
0.067	maple (regrowth)	"	"
0.028	locust (regrowth)	"	"
0.059-0.082	***oak woodland***	***Victoria B.C.***	***this study***
0.032-0.042	***mixed forest***	***Victoria B.C.***	***this study***
CULTIVATED			
0.005	cornfield	N. Ontario	Lessard et al. 1994
~0.000	cornfield	Mass.	Steudler et al. 1996a
0.021	pasture	Colorado	Mosier et al. 1991
~0-0.023	cultivated	Germany	Born et al. 1990
0.063	***ex-hayfield***	***Victoria B.C.***	***this study***
0.006-0.015	***irrigated lawn***	***Victoria B.C.***	***this study***

Table 4.5 - Summarized methane uptake rates in temperate environments

Sources: various authors cited

Steudler *et al.* (1996a) measured much higher methane oxidation rates for temperate forests in Massachusetts (0.115 mg·m^{-2}·h^{-1} for hardwoods, 0.118 for pine). The Victoria results are closer in magnitude to those reported for other midlatitude environments: German beech-oak forest, 0.001-0.028 mg·m^{-2}·h^{-1} (Brumme and Borken 1999); Colorado shortgrass prairie, 0.020-0.035 mg·m^{-2}·h^{-1} (Mosier *et al.* 1991); and Great Lakes/St. Lawrence Lowland forest near Ottawa, 0.002-0.046 mg·m^{-2}·h^{-1} (Lessard *et al.* 1994). While eastern Ontario experiences a slightly more continental climate regime than Massachusetts (Kendrew 1953), temperature and precipitation patterns for those eastern North America sites are more similar to each other than to the west coast -- other factors, *e.g.* soil conditions or measurement strategies, may provide a more valid explanation for the discrepancy.

The exposed meadow site on Mt. Tolmie, "D", which may have experienced some anthropogenic disturbance, had an average methane uptake value comparable to some other midlatitude grassland habitats (Mosier *et al.* 1991). The measurements for cultivated environments seemed to be in line with work by earlier researchers: the abandoned hayfield by Gordon Head Road had higher uptake rates than fields and pastures currently in use, while the measurements from the groomed campus lawns were midway between cornfield and pasture values (Mosier *et al.* 1991, Lessard *et al.* 1994, Steudler *et al.* 1996a).

In terms of climatic and edaphic factors, Vancouver Island and Lower Mainland environments are similar to areas further south along the coast, *i.e.* in Washington and Oregon (Klinka *et al.* 1979, Erickson 1996). The Garry oak parklands around Victoria bear some resemblance to the postglacial relict oak savannas on sandy terraces adjacent to the Great Lakes (Van Zant 1992), but they differ in species composition.

Not much is known about trace gas oxidation in open oak woodlands, although these ecosystems are attracting increasing attention from researchers interested in their composition, ecology, and structure (Stevens 1995). The oak communities in coastal B.C. are considered to be a component of the Pacific Lowland Mixed Forest province, which extends south from Puget Sound into Oregon. The estimated size of this ecological province is 38 600 km^2 (Bailey 1995), larger than the entire area of Vancouver Island (Colombo 1978). Open stands of oak also occur in California environments, specifically Coastal Chaparral and Coastal Steppe/Mixed Forest, bringing the total area of ecologically-similar habitat up to 77 200 km^2 (Bailey 1995). Using the Mt. Tolmie sites as an example, if methane uptake rates in those locations are comparable to Victoria's oak woodlands, these types of west coast environments could be oxidizing as much as 2 $kg{\cdot}km^{-2}{\cdot}d^{-1}$, which could amount to more than 4 500 tonnes of CH_4 per month, for the region between California and southern British Columbia.

Elsewhere on the continent, other oak species are found in the broadleafed forests of the east and midwest, and in the warm temperate environments of the Carolinas and Gulf of Mexico. These regions are all significant in terms of size, and biological diversity. Sometimes, due to factors such as local differences in precipitation patterns, soil drainage, or disturbance, these oak forests have open canopies, resulting in a structural resemblance to the west coast woodlands. Stevens (1995) notes that: "The oak savannas of the mid-continent .... constituted a vast landscape stretching down ... all the way from the Great Lakes to the Gulf Coast." Including only the environments where open forests of deciduous oak species are likely to occur -- Eastern Broadleaf Continental Forest, Central Appalachians, and Temperate Prairie Parkland -- these three ecological provinces comprise over 1.4 million km^2 (Bailey 1995), an area more than 18 times the size of the coastal woodland habitats discussed earlier. Undoubtedly there is a need for more information on greenhouse gas fluxes in oak communities, to get a clearer picture of the magnitude and variability involved. This is particularly important, given that many of these habitats are coming under increasing development pressures, ranging from pollution and water diversions to urban sprawl and invasive species (Stevens 1995).

CHAPTER 5 - INFLUENCES ON THE SOIL SINK

5.1 Factors influencing methane uptake capabilities

During this past century, geography has experienced a shift in how environments are perceived, from "faces" to "flows" (Kates *et al.* 1990, Turner and Meyer 1994). Earlier, there was more of an emphasis on landscape description and identification, using techniques such as cartography. This resulted in detailed yet static impressions of living systems. In later decades, technological advances provided the capability to monitor and record movement; for example, the exchange of energy, nutrients, and propagules in time and space (Odum 1963, Garrels and Lerman 1984, Westbroek 1991, and others). The concept of budgets, whether they be for energy, water, or sediments, is widespread in physical geography. These approaches have both benefited from and contributed to the emerging field of global change research (*e.g.* Thomas 1956, Turner *et al.* 1990).

Identification of linkages between climate and land uses is not a new area of research: in the 1970s, researchers concerned about climate change saw the alteration of surface characteristics as a potential problem (Study of Man's Impact on Climate 1971). Changes in surface albedo, due to urbanization or the removal of natural vegetation, were cited as factors which could alter the regional energy and water balance; also, the emission of particulates and other atmospheric contaminants by agriculture and industry was recognized as having global implications. Regarding methane in particular, a number of different characteristics are known to influence oxidation rates, among them the rate of microbiological activity, soil moisture content, previous disturbance, and the presence of nitrogen compounds in the environment.

5.1.1 Soil microbiology

In order to understand methane uptake in the environment, we need to learn about the composition and population characteristics of soil microorganism communities. As discussed earlier, methane can be oxidized by methanotrophs and by nitrifiers. Which type of bacteria dominates in a particular setting may be influenced by human activities: agriculture appears to favour nitrifiers (Steudler *et al.* 1996a), which can oxidize both ammonium and methane; but methane uptake rates for nitrifying bacteria are significantly lower than for methanotrophs.

There is still a great deal of uncertainty about soil microecology. Wood (1989) suggests that bacteria are not distributed evenly throughout the soil, but are clustered in colonies, often restricted to particular microhabitats. In effect, the soil consists of vast network of voids, root surfaces, and granules, forming a world unto itself. At any given time, microbes may only occupy 0.1% of the potential surface area in a soil sample (Wood 1989). Richards (1974) suggested that population densities of nitrifiers, such as *Nitrosomonas* and *Nitrobacter*, may be relatively low: 100 cells per gram of soil in low-pH habitats, 10 000 cells/g under more alkaline conditions (unless ammonium has been added to the system).

These families of bacteria may not be operating independently: ecological relationships could exist within the groups, or between methanotrophs and nitrifiers, or even with other types of microorganisms. Of particular interest are the ways in which individual bacteria or colonies interact with other bacterial species, and microorganisms in general. Complicated associations of photosynthetic, chemosynthetic, and decomposer bacteria can result in extensive bacterial communites, inhabiting soil horizons or different layers of the water or sediment column (Killops and Killops 1993). Numerous varieties of microbes can literally end up living on each other's waste products (Martens and Berner 1977, Fenchel and Finlay 1994). There is still a great deal to be learned about the composition and ecology of these communities (Volk 1994).

These types of associations are often cited as examples of mutualism: if essential to the survival of all species involved, symbiosis; or if favourable but non-obligatory, proto-cooperation (Pears 1985). The superorganism interpretation has been criticized as overly deterministic (*e.g.* Tivy 1982, Pears 1985, Budiansky 1995). Various authors have pointed out that groups of bacteria compete for resources, just as larger organisms do; the complexities of microbial interactions are still being investigated. In the context of methanogenesis. Lovley *et al.* (1982) proposed that competition between sulfate reducers and methanogens (methane-producing bacteria) for hydrogen can result in the inhibition of CH_4 production, provided that there is plenty of sulphate, and that the sulfate reducers can decrease hydrogen concentrations below the minimum level which the methanogens need to function. Roslev *et al.* (1997) suggested that microbial predation may be playing a role in the apparent stratification of methanotrophs, in particular soil layers.

In addition, scientists have known for some time that the availability of micronutrients can influence biological activity in the soil. Although ecological problems are seldom reducible to single limiting factors, there have been cases where this approach has yielded interesting results. Enzymes often contain minute but essential quantities of trace elements, such as manganese or cobalt. Delwiche (1970) cites the example of the nitrogen-fixing enzyme nitrogenase, which incorporates molybdenum: legume crops in some parts of Australia met with limited success, until trace amounts of molybdenum had been added to compensate for local deficiencies.

According to Wood (1989), the creation of NH_3 monooxygenase requires copper (Cu). Like other biologically-important trace elements, the presence of copper has been studied in many environments. Some soils are poor in this element, *i.e.* less than 5 ppm, particularly sites low in clay or high in organics (Committee on Medical and Biologic Effects of Environmental Pollutants 1977). Most British Columbia soils do not appear to be Cu-deficient, with amounts measured in the 10-100 ppm range; experiments on the Saanich Peninsula suggested that local Victoria soils have average to above-average copper concentrations (Day *et al.* 1959). However, this factor should be considered when looking at environments which exhibit lower methanotrophic activity than expected.

5.1.2 Soil moisture

Soil water content has both direct and indirect effects on bacterial gas fluxes. In situations when moisture is severely limited, dehydration can be an important factor: microorganisms experience physiological stress, which can inhibit enzyme activity (Stark and Firestone 1995, Schnell and King 1996). The data from the Mt. Tolmie sites (Figures 3.5 a and b) showed declining CH_4 uptake rates after more than a week without precipitation. Experiments by Schnell and King (1996) also resulted in decreased methane oxidation at low soil moisture potentials.

Soil moisture potential can also have an indirect influence, via the availability of reactants: in this case, methane and oxygen. Various authors (King 1990, Focht 1992, and others) have noted that diffusion rates pose a significant obstacle to methane uptake. Striegl (1993) observes that "The capacity of the methylotrophs to consume CH_4 commonly exceeds the potential of CH_4 to diffuse from the atmosphere to the consumers." Diffusion coefficients are also known to play a role in microbial reaction rates for marine environments (Iversen and Jorgensen 1993).

Tortuosity plays a major role in governing rates of gas diffusion within a soil: it is determined by physical characteristics, such as the size and arrangement of particles (Striegl 1993). Soil moisture content is crucial, since diffusion rates are significantly lowered if soil pores are filled with water: for example, the diffusion rate of oxygen in water is only one ten-thousandth of its diffusion rate in air (Wood 1989). Even soils which are not saturated may have considerable amounts of water within their matrices, in the form of thin coatings of water adhering to particle surfaces (Oke 1978). This has implications for the diversity of microbial communities within the soil column: anaerobic conditions favour methanogens over methanotrophs, which are obligate aerobes. If the soil environment is heterogeneous enough, both methane oxidizers and methane producers could survive in close proximity. A soil which appears to be dry and well-aerated may have air-filled voids or macropores, yet the much smaller pores within individual blocks or granules may still be saturated. Anaerobic microenvironments inside soil crumbs larger than 6 mm in diameter have been postulated (Wood 1989).

Given that physiological water stress will influence bacterial processes at low soil moisture potentials, while restrictions on gas diffusion will become important when water content is high, there should be an optimum range for methane uptake. Schnell and King (1996) reported maximum CH_4 oxidation rates when the soil moisture was around 25%, for a mixed broadleaf-conifer temperate forest ecosystem in New England: the uptake rates from other studies peaked at around 20%.

In addition to influencing biological activity and gas diffusion, precipitation could be having other effects on soil microbes. Water can transport compounds such as ammonium and nitrates, from the atmosphere or from surface applications, down into the soil. As will be discussed later in this chapter, nitrogen availability can have a significant impact on methane uptake rates.

Hutchinson and Brams (1992) found that precipitation caused a rapid increase in NH_4^+ concentrations in the surface layers of both fertilized and unfertilized pasture plots. In the case of the fertilized areas, rainwater dissolved the dry $(NH_4)_2SO_4$ which had been scattered over the plots more than a week before, and washed it down into the soil. The authors offer a couple of possible explanations for the sudden rise in NH_4^+ for the unfertilized areas: transportation of ammonium from the adjacent fertilized plots, either as volatilized gaseous NH_3 deposited downwind, or in solution with runoff; or "the burst of mineralization/nitrification activity that typically occurs immediately after wetting very dry soil" (Hutchinson and Brams 1992). This last suggestion is particularly interesting in the Victoria context, since the local soils do experience extended periods of drought during the summer. Hutchinson and Brams (1992) note that rainshowers of only a few millimetres are sufficient to cause a response in dry soils, detectable as a rapid surge in NO emissions, even if there is no measurable change in soil nitrogen concentrations.

Given what is already known about the relationship between methane uptake and nitrification, it is possible that the post-rainfall "nitrification burst" may also be affecting CH_4 oxidation. For example, in the Victoria data (especially on 25 Sept

1997), the rapid drop in methane uptake rates during the relatively sparse first hour of precipitation might correspond to enhanced $NH4^+$ oxidation, and not merely be a response to the filling of pore spaces by rainwater.

Another issue which requires further research is the role of artificial irrigation, and the ways in which it differs from precipitation events. In the summer months, many areas in Victoria are watered, sometimes daily, to prevent the grass from dying back. Individuals or groups who are reluctant to apply chemical fertilizers, herbicides, and pesticides, may still insist on regular watering during the dry season. This factor might be contributing to the discrepancies between the groomed and natural habitats observed in this study. Given that lawns, municipal parks, playing fields, and golf courses account for a considerable proportion of municipal greenspace, these sites may be reacting to significantly different conditions than unmodified environments such as Mt. Tolmie and Gonzales Hill.

5.1.3 Soil disturbance

Besides being affected by soil moisture content, tortuosity can be influenced by compaction and other forms of mechanical disturbance (Striegl 1993). Various authors have postulated that soil texture influences trace gas uptake and emission rates (Nevison *et al.* 1996, Steudler *et al.* 1996a, Ball *et al.* 1997, and others). Agricultural scientists have already described the impacts which can result from the misapplication of heavy equipment: a single passage by a tractor can cause a significant reduction in soil porosity, as deep as 30 cm below the surface (Cary 1979); likewise, ploughing when the soil is too wet can also reduce pore space (Finkl 1979). Priem and Christensen (1999) suggested that non-mechanized agricultural systems can also have a significant impact on soil structure and methane uptake. Even trampling by livestock (Goudie 1986) and human foot traffic (Liddle 1975), has been shown to alter soil structure.

Tilling increases both the aeration, and the average temperature of the soil, which in theory could enhance oxidation; however, it also causes significant disruption of the soil horizons (Volk 1994), which has implications for the methanotroph communities. Steudler *et al.* (1996a) found that forested sites which had regenerated from farmland still had a characteristic Ap soil horizon, and significantly lower methane uptake rates. Roslev *et al.* (1997) experimented with physical disturbance to soils: sieving and mixing decreased oxidation rates by more than 50%, although the carbon conversion efficiency (percentage of methane actually assimilated into bacterial biomass) was unaffected.

Like many other urbanized regions in British Columbia, the Gordon Head neighbourhood around the University of Victoria campus has experienced soil disruption, beginning with the clearing of natural vegetation, and conversion to agriculture (Forward 1976, Jupp 1980). Even natural-looking groves and meadows, *e.g.* around the Science and Fine Arts complexes, may have been subjected to grading and levelling. Aerial photographs showing the construction of the university, and of the military camp which predates it, imply that large areas of the campus were affected by construction and earthmoving (Smith 1993). These activities continue up to the present day: for example, the abandoned field site adjacent to Gordon Head Road and Cedar Hill X-Road, which last produced hay in the 1960s, was briefly used to store the soil from the Arts building site in 1992 (pers. comm. Tony James 1998). Excavation of plots at the community garden next to Parking Lot 7, near the Saunders Annex, revealed the remains of gravel drainage beds from at least two now-vanished temporary buildings, dating from the wartime use of the area as a military base.

Fire is a form of disturbance which is widespread in open woodlands (Stevens 1995). Garry oak habitat, all the way down the west coast, is known to be fire-prone: in fact, a grass fire on the opposite side of Mt. Tolmie from the study site occurred in late July 1999, and came close to threatening the nearby residential area. Many oak savanna communities, including southeastern Vancouver Island, were managed for food production by First Nations peoples, using techniques which included deliberate burning (Forward 1976). Researchers have begun to investigate the long-term effects of fire on soils. In some cases, fine charcoal particles can increase water retention, even in well-drained, sandy areas (Moore 1996). Priem and Christensen (1999) observed that fire did appear to affect methane uptake in tropical savannas, although the relationship was complicated by time since burning.

For all of the types of disturbance discussed, it is reasonable to assume that varying intensities and frequencies of soil perturbation can affect microbial populations, just as they influence the distribution of larger organisms. Previous research has shown that different disturbance regimes can result in ecological communities which have disparate life strategies (Connell 1978, Sousa 1979, Sprugel 1991). The substitution of agriculture for wildfire, or of mowing for tilling, could have resulted in significant changes in the types of species which inhabited these environments.

5.1.4 Role of nitrogen compounds

Besides being the major component of the Earth's atmosphere, nitrogen is a macronutrient: its complex biogeochemical cycle covers the entire biosphere (Delwiche 1970, Killops and Killops 1993). In its diatomic form (N_2) it is relatively inactive, but in the nitrogen cycle, it is transformed into ammonium (NH_4^+) via nitrogen fixation; then from ammonium into nitrites (NO_2^-) and nitrates (NO_3^-) through nitrification; and from nitrate back to atmospheric nitrogen, after denitrification (nitrate reduction).

Many nitrogen compounds have been studied in the context of pollution. Oxides of nitrogen (NO_x) from sources like automobile exhaust and biomass combustion can travel for considerable distances in the atmosphere, and have been implicated in acid rain (Oke 1978, Fisher and Oppenheimer 1991, Munger *et al.* 1998, Savarino and Legrand 1998). Runoff of nitrate and ammonium fertilizer contributes to the eutrophication of many rivers and lakes, by causing massive algal blooms. In terrestrial ecosystems, various authors have discussed the biogeochemical impacts of nitrogen overloading on forests (Magill *et al.* 1997, Ohrui and Mitchell 1997, Fenn *et al.* 1998) and grasslands (Hutchinson and Brams 1992).

Plants are capable of utilizing nitrogen in both its reduced and fully oxidized forms, but nitrifying bacteria are limited to oxidizing ammonium and nitrites (NO_2^-). King and Schnell (1994) believe that microbial methane oxidation involves complex interactions among different types of soil bacteria, and the various chemicals which they consume or produce (NH_4^+, NO_2^-, CH_4, etc.). They point out that even between bacteria which appear to carry out the same processes, there are significant differences: for example, Methylobacter albus appears to be more sensitive to ammonium and nitrite concentrations than Methylosinus trichosporium, although both are capable of oxidizing NH_4^+ to NO_2^-. In Roslev *et al.* (1997), not only did the addition of ammonium decrease the rate of methane oxidation, but it also appeared to suppress carbon conversion efficiency by 85 to 99%, so relatively little CH_4 was actually being converted into microbial biomass. Also, unlike the specialized methanotrophic bacteria, autotrophic nitrifiers which were oxidizing CH_4 instead of ammonia tended to convert relatively little of the methane into biomass, as low as 1-2% (Jones and Morita 1983, cited in Roslev *et al.* 1997).

By comparison, physical disturbance of the soil seems to have much less of an impact than the addition of ammonium compounds. Roslev *et al.* (1997) found that soil mixing also decreased methane oxidation rates, and therefore methane assimilation, but the carbon conversion efficiency was relatively unchanged even after sieving. Turnover rates (loss of methane which had been incorporated into microbial biomass) were likewise unaffected by soil disturbance, but increased more than tenfold after the addition of NH_4^+: also, after ammonia fertilization, more methane was incorporated into low molecular weight metabolites, and less into more complex compounds such as nucleic acids, polysaccharides, and proteins (Roslev *et al.* 1997). For both these reasons, the assimilated carbon may be less likely to remain in the soil. Oxidation rates seem to remain low for years after ammonium fertilization, even after NH_X concentrations have declined to pre-experimental levels (Schnell and King 1994). This suggests some kind of persistent, long-term ecological shift in the composition of the bacterial communities.

Killham (1994) acknowledges that certain fermentative bacteria can reduce nitrate to ammonium, in effect reversing the usual flow of the nitrogen cycle: this process is distinct from the denitrification reaction. Although little is known about the nature of this particular flux, local factors may play a role. There is some evidence that high-nitrate, carbon-poor environments may be more suited to denitrifiction, while dissimilatory reduction of NO_X^- to NH_4^+ may occur in carbon-rich environments, along with other types of fermentation reactions (Killham 1994). However, it is commonly believed that in most situations, nitrates and nitrites would not be converted to NH_4^+, but to NO, N_2O, and eventually N_2, by anaerobic denitrifying bacteria (Davidson and Schimel 1995). Atmospheric deposition of oxides of nitrogen seems to have different effects on methane oxidation, depending on the regional environmental characteristics (Brummer and Borken 1999). For these reasons, this discussion will focus on ammonium as a pollutant.

Munger *et al.* (1998) estimated that NH_X compounds received at the Harvard Forest site, both in precipitation and as dry deposition, amounted to about 8.1 $mmol{\cdot}m^{-2}{\cdot}a^{-1}$ (~150 $mg{\cdot}m^{-2}{\cdot}a^{-1}$), or around 15% of observed nitrate inputs: the authors state that these are relatively low concentrations. By comparison, a Japanese study on a forested watershed approximately 100 km northwest of Tokyo (Ohrui and Mitchell 1997) estimated NH_4^+ received in precipitation at around 500 $mg{\cdot}m^{-2}{\cdot}a^{-1}$.

A variety of sources for ammonia have been identified, including volatilization from sewage and animal wastes, geothermal steam, refrigerant leakage, and industrial activity, ranging from foundries and petroleum refineries to coke and fertilizer manufacturing (Graedel 1978). Mooney *et al.* (1987) cite intensive agriculture as a significant source of atmospheric ammonia. Bouwman *et al.* (1997) have compiled a global emissions inventory (Table 5.1).

Source of ammonia (NH_3)	**Global emissions (teragrams/a)**
Domesticated animals	21.6
Synthetic fertilizers	9.0
Oceans	8.2
Biomass burning	2.6
Households	2.6
Natural environments	2.4
Industries	0.2
Fossil fuels	0.1
Wildlife	0.1
TOTAL	54

Table 5.1 - Estimated global ammonia emissions, 1990
Source: Bouwman *et al.* 1997

Overall, they concluded that domestic animals, and their waste products, were responsible for an estimated 40% of ammonia released into the environment. Synthetic fertilizers were the next largest contributor, at around 17%.

In British Columbia, ammonia is one of the most common pollutants: in 1996, 1 802 tonnes were released, second only to methanol, at 3 501 tonnes (British Columbia Ministry of Environment, Lands and Parks 1998). However, there is considerable spatial heterogeneity in atmospheric deposition, based on prevailing wind patterns, and the dispersion of point and area sources. In the 1980s, ammonia manufacturing in B.C. was concentrated at Ocelot Chemicals Ltd. in Kitimat, and Cominco Fertilizers Ltd. in Trail (Concord Environmental Corporation 1991). Cominco has since ceased production, due to economic reasons (B.H. Levelton 1990).

NH_X emissions from other operations, such as the production and use of nitrogen-based explosives, are relatively low (Bouwman *et al.* 1997).

It is likely that any anthropogenic NH_X entering these sites originates from fertilizer applications. Worldwide, the use of synthetic chemical fertilizers dates from the 1820s, with the production of ammonium sulphate as a by-product of coal gas manufacturing; however, levels approaching present-day consumption were not reached until after the Second World War (Goudie 1986). Laboratory and field experiments by various researchers have demonstrated that methane oxidation in terrestrial soils can be decreased significantly by addition of nitrogen, particularly in the form of ammonium (Steudler *et al.* 1989, Conrad and Rothfuss 1991, and others). *Nitrosomonas europaea*, a common nitrifier, can oxidize CH_4 even when atmospheric mixing ratios are low, but this ability is suppressed when soil ammonium concentrations are in excess of 15 ppm (Steudler *et al.* 1989). Both NH_3 and NH_4^+ can enter the soil through direct fertilization or indirect runoff, or by way of the atmosphere, in precipitation or dry deposition (Munger *et al.* 1998).

A few decades ago, most scientific and popular concerns about anthropogenic emissions of "fixed" nitrogen focused on issues such as eutrophication in aquatic environments, or the role of NO_X in the formation of smog and acid rain. Soil microorganisms capable of nitrogen fixation were not recognized as a potentially sensitive component in the environment: in fact, a century ago scientists speculated that the recently-discovered denitrifiers (bacteria that can reduce nitrogen oxides back to NO, N_2O, or N_2) were a threat to soil fertility (Delwiche 1970).

NH_X fertilization can affect the atmosphere twice over: first, it impairs CH_4 uptake, so methane which would otherwise have been oxidized is allowed to diffuse out of the soil. Second, the oxidation of NH_X generates nitrous oxide (N_2O), which is a greenhouse gas, and also contributes to the breakdown of stratospheric ozone (Mooney *et al.* 1987, Bouwman *et al.* 1997). Combined with existing knowledge about the detrimental effects of eutrophication and acid deposition, research on nitrogen and trace gas fluxes has helped shape the growing consensus that nitrogen overloading poses a significant ecological threat (Turner and Meyer 1994, Vitousek *et al.* 1997, Fenn *et al.* 1998).

The Haber-Bosch process invented just prior to the First World War (Grubler 1994) allowed mass synthesis of ammonia, which was frequently added directly to the soil -- an early example of artificial nitrogen enhancement. While anhydrous (liquid) NH_3 is still used, in many areas it has been superseded by other forms of synthetic fertilizers, such as urea. In B.C., as of 1995, 3 610 km^2 of farmland was treated with commercial fertilizers, an increase of 9% since 1991 (Statistics Canada 1997a). Earlier in the decade, chemical fertilizer use, including anhydrous ammonia, urea, and ammonium salts, was estimated at almost 30 kilotonnes of N for the province (Concord Environmental Corporation 1991). On Vancouver Island, the total area treated was estimated at around 170 km^2, or over 29% of farmland (Statistics Canada 1997a).

The use of manure, a by-product of domestic livestock, remains popular: such organic fertilizers also release NH_X through microbial breakdown of urea and uric acid (Bouwman *et al.* 1997). The reaction which transforms urea into ammonia is called hydrolysis, and it is catalysed by urease, an enzyme which is produced by many soil microorganisms (Killham 1994):

$$CO(NH_2)_2 + H_2O \dashrightarrow H_2NCOONH_4 \dashrightarrow 2\ NH_3 + CO_2 \qquad (5.1)$$

In addition, animal wastes contain proteins which are converted into NH_3 through bacterial ammonification (Delwiche 1970). For landscaping applications, dry granular types of fertilizer are favoured, as there is less odour, and the nutrients are more concentrated. However, synthetic urea is also a major component of these dry fertilizers, and reacts just as readily. Particularly in warm climates, volatilization from both manure and urea fertilizers can result in the loss of significant amounts of NH_3 to the atmosphere (Killham 1994).

During this study, some inquiries were made regarding the composition and amounts of fertilizer used on campus. A representative from Facilities Management explained that mass fertilization of the campus lawns has not been carried out since the mid-1980s (pers. comm. Tony James 1998), presumably due to labour and budget constraints. At present, only the playing fields are regularly treated with chemical fertilizer: generally 30-6-10, with most of the nitrogen in the form of sulphur-coated urea, to delay release and reduce volatilization. This type of fertilizer is manufactured by Evergro of Delta, B.C., and is supplied by Cedar Hill Nursery (pers. comm. University of Victoria Purchasing 1999). Evergro also produces the 23-3-23 Total w/ESN mixture used by the City of Victoria's Parks Division (pers. comm. 1999).

The natural forested habitats on campus are left untouched, with the exception of the rhododendron and azalea plantings, which are hand-fertilized: nutrient spikes are sometimes installed at the base of deciduous trees in the groomed areas (pers. comm. Pat Green 1999). Outlying areas of the campus, such as the old hayfield, are mown every couple of months during the year, but are otherwise not modified (pers. comm. Tony James 1998). The hayfield itself likely has not been fertilized since the late 1960s. Given that in this region, hay crops do not require additional nutrients unless they are being harvested intensively, *i.e.* annually instead of every other year (pers.

comm. Colin Wood 1999), additions of nitrogen may have been low even before that time. The fenced-in plot across McKenzie from the tennis courts, once used for equipment storage but currently the site of the Campus Community Garden, was left as rough grass: fertilization, if any, would have occurred prior to 1975 (pers. comm. Tony James 1998).

Elsewhere in the Victoria region, the parks department of Oak Bay treats heavy-use areas with 1 lb/square foot of Evergro Organic Plus (24-4-12), four times during the May-September growing season (pers. comm. Dan Bell 1999). This works out to approximately 50 kg/ha, or just under 200 kg/ha per year. In comparison, Steudler *et al.* (1996a) described a Massachusetts cornfield which was treated with 155 $kg \cdot ha^{-1} \cdot a^{-1}$ of urea; therefore, in some situations the amount of chemical inputs for turf can be similar to agricultural crops. The municipality of Saanich also uses chemical fertilizers consisting mainly of urea on its playing fields: although the composition varies from 16-32-6 in the spring, to 25-3-12 in May-September, treatments are applied on a monthly basis too (pers. comm. Clyde Smith 1999).

Both Saanich and Oak Bay restrict fertilizer use to the groomed areas in their municipal parks. In locations where the grass is allowed to grow long, *e.g.* in Uplands Park (Oak Bay) and on Mt. Tolmie (Saanich), no nutrients are added. For a number of years, the Oak Bay parks department has experimented with natural management techniques, such as postponing mowing until after the wildflowers have finished their life cycle, restricting or eliminating watering, and mulching instead of adding chemical fertilizers. One of these areas is the triangular scrap of parkland at the junction of Midland Road and Beach Drive, in Oak Bay. These practices not only reduce landscaping costs (pers. comm. Dan Bell 1999), but also have significantly less impact on the environment. However, residents sometimes complain that the long grass appears untidy and may harbour weeds.

Another possible source of nitrogen compounds could be the presence of rabbits, *Oryctolagus cuniculus* and *Sylvilagus floridanus*, in certain areas. These non-native species were introduced to Vancouver Island after European settlement (Carl and Guiget 1958, Nagorsen 1986). Rabbits were not observed at the Mt. Tolmie site, although the nearby university campus is home to a population of several dozen, perhaps more than a hundred, gone feral from pet stock. Plentiful droppings and signs of burrows were evident at the old hayfield site behind the Fine Arts building.

The rabbits themselves are seen almost everywhere in the grassy areas of the campus, although the situation is not as extreme as on the grounds of the Victoria General Hospital (Willcocks 1998). Although such relatively small animals would appear to be inconsequential, highly-concentrated populations may have some impacts: Bouwman *et al.* (1997) assume that larger herbivores would produce approximately 6 mg of urea per gram of biomass consumed; if a 2 kg rabbit, during a day's grazing, consumes an amount equivalent to 10% of its body mass, it might release more than 1 000 mg of urea in the same timespan. This would likely not have a significant effect on soil nutrient status, except in areas where people regularly attract large numbers of rabbits by setting out food, *e.g.* around the campus residences and the Student Union building. In these cases, biogeochemical inputs might reach a level equivalent to amounts of NH_X received in precipitation, 100-500 $mg{\cdot}m^{-2}{\cdot}a^{-1}$ (Ohrui and Mitchell 1997, Munger *et al.* 1998). Yamulki *et al.* (1999) found that the urine patches left by larger herbivores (cattle) did not appear to decrease methane uptake, but their experimental site was already rich in nitrogen compounds. The situation might be different if the habitat is initially low in nitrogen. Adamsen and King (1993) observed that addition of ammonium inhibited methane uptake more strongly in subarctic than in richer, more temperate soils.

Other introduced species may have had a greater influence on the local environment around Victoria, this time through nitrogen fixation: vegetation in the study areas includes non-native legumes (Pea Family: *Fabaceae*), such as clover (*Trifolium* spp.) and Scotch broom *(Cytisus scoparius)*, which are hosts to nitrogen-fixing symbiotic bacteria. In these microbes, such as Azotobacter, the nitrogenase enzyme catalyzes the fixation of atmospheric nitrogen (N_2) into ammonia (Killham 1994). Wood (1989) suggests that nitrogen fixation by white clover *(Trifolium repens)*, a species naturalized in many habitats around the world including Vancouver Island, can be as high as 4.5-67.3 $g \cdot m^{-2} \cdot a^{-1}$. Changes can be even more pronounced in cases of deliberate planting: Delwiche (1970) estimates that nitrogen inputs from densely-planted monocultures of leguminous crops can be in the neighbourhood of 350 kg/hectare annually. This is 50 to 100 times the amount of fixed nitrogen generally available in ecosystems, such as grasslands, which have few native legumes. Around 8% of Vancouver Island farmland is presently occupied by alfalfa, another legume (Statistics Canada 1997a): in addition, both clover and alfalfa are frequently added to seed mixtures to improve the quality of the grazing, or the hay harvest (pers. comm. Colin Wood 1999).

Steudler *et al.* (1996a) found that methane uptake in a stand of black locust *(Robinia pseudoacacia)*, a leguminous tree, was lower than all of the other sites, except a fertilized cornfield; however, since it had been cultivated at some time in the past, it was difficult to determine whether these anomalies were due to the presence of the locust trees, or to past soil disturbance and fertilization.

In the Victoria region, broom and other invasive exotics have been the focus of public attention in recent years (Norbury 1998). In their 1972 report, Szczawinski and Harrison estimated that for the Saanich Peninsula, the number of introduced floral taxa (species and subspecies) stood at 333, compared with 691 indigenous ones, or just over 48%. The same authors gave the ratio of indigenous to introduced *Leguminosae (Fabaceae)* as 29:28. If anything, the number of introduced species in B.C. has risen in the intervening quarter-century (Douglas *et al.* 1999). Similar patterns have been observed in many other environments, such as Hawaii and New Zealand (Vitousek *et al.* 1996). Invasion rates tend to be higher for environments which are naturally depauperate in species, due to isolation (such as islands), or particular edaphic conditions, *e.g.* nitrogen-poor bogs.

The worldwide change in species composition since European settlement, the "Columbian Exchange" has been something of a revolution (Crosby 1986). It has had far-reaching effects in terms of ecological interactions and even soil chemistry (Wedin and Tilman 1996, Mlot 1997). Imported nitrogen-fixers have played a significant role: once levels of biologically-available nitrogen increase, other invasive exotics are able to gain a competitive advantage over native vegetation. Grasses, in particular, are subsequently able to modify the environment even further by forming dense root mats that can exclude other species. Some researchers have suggested that excess nitrogen has resulted in "the terrestrial equivalent of eutrophication", as weed populations explode and outcompete native vegetation (Raloff 1996). These transformations are not restricted to the New World: Mlot (1997) cites evidence that European heathland ecosystems, characteristically nutrient-poor, have become susceptible to invasion by Eurasian grass species as a result of fertilization.

Another way in which vegetation can change the availability of nitrogen is through a change from coniferous to deciduous vegetation, or vice versa. These shifts have been known to happen naturally (Day et al. 1959), or due to logging and silviculture. In coniferous forests, most nitrogen is in the form of NH_X, rather than nitrate (Killham 1994); along with soil pH, methane oxidation rates are also lower (Brumme and Borken 1999). The shift towards ammonium may be due to allelopathic interactions among plant species. Low rates of nitrification are characteristic of coniferous forest litter, while nitrifiers proliferate in clearcuts (Killham 1994).

5.2 Prospects for recovery

Over the past century a global pattern of land use change has emerged. As areas become industrialized, rural/urban distinctions become blurred. Cities rapidly expand in size, and the surrounding countryside is transformed by the growth of suburban communities, transportation routes, and zones for heavy and light industry (Douglas 1994). Most of the world's land surface is being utilized or modified by human activities, to varying degrees (Richards 1990).

Locally, the population of Victoria has increased from just under 121 000 in 1956 (Day *et al.* 1959) to almost 287 900 in 1991, and over 304 200 just 5 years later (Statistics Canada 1997b). By 1996, the Census Metropolitan Area (CMA) for Victoria had expanded to around 633 km^2, occupying a substantial portion of the Saanich Peninsula. A half-century earlier much of the Peninsula was being intensively farmed, for specialty and market crops such as flowers, soft fruit, potatoes, and vegetables (Day *et al.* 1959). Due to population growth in subsequent decades, accompanied by a tremendous expansion in infrastructure, many of those farms were converted into commerical or residential use, despite the efforts of the provincial government to restrict urban sprawl through legislation such as the Agricultural Land Reserve (Siddon 1995). Many jurisdictions around Victoria are experiencing similar development pressures: the Gulf Islands, the east coast of Vancouver Island as far north as Courtney and Comox, the Lower Mainland, and the area around Puget Sound have all experienced rapid growth in the past few decades.

The rural population has dwindled in many areas, but impacts on the land often do not, due to intensive mechanized farming practices, and high inputs of synthetic fertilizers and pesticides. Some communities have experienced a switch in economic activities, from agriculture and resource extraction to tourism and recreation. In these cases, there may be significant impacts resulting from increasing numbers of visitors (transportation infrastructure, waste, etc.), and from the development of landscaped facilities such as hotels and resorts, even within protected habitats. For example, golf courses have been subjected to recent criticism: Siddon (1995) argues that contrary to popular assumptions, they are not a rural/agricultural land use, since they entail substantial changes to the environment such as the addition of topsoil, farmed turf, and continued inputs of water and chemicals.

With regards to patterns of land use change, several factors should be considered. The types of impacts which have already been experienced by a particular area must be taken into account, both direct (farming, ranching, resource extraction, or the building of infrastructure), and indirect, such as the long-distance transport of pollutants. New development can encroach on areas which until now have remained "natural", with little or no human modification; it can also affect environments which have already been influenced by activities such as agriculture or logging. The more accessible areas, which tend to be favoured for industrial or residential expansion, are generally not pristine, but may have been used for ranching or farming in earlier decades. As a result, the areas most directly affected by urban growth are adjacent to existing settlements, and may have already been modified by human activities: these land-use changes are typical of rural/urban fringe settings in North America (Siddon 1995).

In theory, the loss of these "sacrificial" lands may be less detrimental to the environment than degradation of a more pristine area, further away. However, many of these fringe settings contain important ecosystems in their own right, such as wetlands and coastal or floodplain habitats. The Garry oak meadows of southern and eastern Vancouver Island coincide with the centres of population, so the oaks which have escaped the plough are now endangered by subdivisions, warehouse-style retail stores, and other forms of urban sprawl. From a nutrient-cycling viewpoint, the locations where pollution is being generated by urban, industrial, and agricultural activities, also will have a lower capacity for absorption: nearby areas of land and water are overloaded, and are unable to deal with increasing volumes of sewage, or CO_2, or nitrogen compounds. Emissions end up being exported further afield, to affect other ecosystems (Folke *et al.* 1997).

Cities, particularly ones in well-off industrialized countries, do not exhibit continuous paved or built-over surfaces: they are interrupted by public and private greenspace, with ornamental plantings visible even along downtown streets (Douglas 1994). Urban greenspace has long been touted as desirable, for aesthetic and practical reasons: parks and trees ameliorate local microclimates, and remove pollutants, or at least spread out the sources so they are not concentrated in one location. The total area of Victoria's Capital Regional District is 236 000 ha (2 360 km^2), of which 80% is identified as greenspace (Westland Resource Group 1997). However, most of that amount consists of areas in the Agricultural Land Reserve, institutional grounds, golf courses, and sites which are currently being managed for forestry purposes. Depending on the location, they may have been heavily modified by human activities, so that the capacity of the soil to absorb pollution may already have been reduced, by fertilization or mechanical disturbance.

Research in both temperate and tropical ecosystems suggests that cultivation decreases the ability of the soil to oxidize methane, frequently by 60% or more (Priem and Christensen 1999). Existing data seem to indicate that recovery is relatively slow. Long after the vegetation has regenerated, methane uptake rates remain depressed. Steudler *et al.* (1996a) considered two environments, in addition to the mixed hardwood forest and pine plantation used in earlier studies. The first site consisted of mature sugar maples (Acer saccharum), on land which had been farmed from the mid-1800s until the early 1900s: the CH_4 oxidation rate was 0.067 $mg{\cdot}m^{-2}{\cdot}h^{-1}$, less than half of either the mixed hardwood or pine plantation controls. The second site was black locust *(Robinia pseudoacacia)* with some black cherry *(Prunus serontina)*, which had been in agriculture possibly as late as the 1950s, if the age of the trees is used as an indicator. The uptake rate at this site was even lower, 0.028 $mg{\cdot}m^{-2}{\cdot}h^{-1}$, or less than 25% of the controls. (Admittedly, natural nitrogen fixation may be taking place at that site, since black locust is a member of the legume family.)

Differences in CH_4 uptake were also apparent in the Victoria results. In the abandoned hayfield, which had last been farmed in the late 1960s, the methane uptake rate was around 0.063 $mg{\cdot}m^{-2}{\cdot}h$. In the uncultivated portion of the campus community garden, which has not been fertilized since the early to mid-1970s, the rate was 0.016 $mg{\cdot}m^{-2}{\cdot}h^{-1}$ while the groomed lawn next to the MacLaurin Building, which has been untreated for approximately a decade, showed a value of 0.015 $mg{\cdot}m^{-}$

$^{2} \cdot h^{-1}$. While the two latter sites had similar uptake rates despite the fact that the garden area was receiving much less water, the main difference between these areas and the former hayfield appears to be the decade or more since cultivation.

Long-term monitoring of sites as they recover from fertilization and other disturbances would be required for an adequate assessment of recovery times, since local heterogeneity in soil characteristics makes comparisons difficult, but so far it appears that a return to original conditions may take several decades, or even more than a century (Keller and Reiners 1994, Priem *et al.* 1997). Cessation of fertilizer use does seem to have a measurable effect over more than a quarter century: perhaps even as much as 50-75% recovery, if we assume that the CH_4 uptake values for the Mt. Tolmie oak woodlands and the campus mixed forest approximate the pre-disturbance situation for the lawns and the hayfield, respectively.

There is some uncertainty about the potential for lag effects, and the possible storage of nitrogen compounds within the soil. Through adsorption onto clay particles, soils have been known to retain water-soluble compounds such as ammonium (Richards 1974, Wood 1989, Richter and Markewitz 1995). Given the massive amounts of anthropogenic nitrogen fertilizers utilized during the late 20th century, it is possible that many areas will exhibit high nitrogen levels for decades to come.

Douglas (1994) notes that in the United States, more than three-quarters of population lives on only 2% of the land; actually, the area being affected by human impacts, in terms of resource extraction and waste inputs, is far greater than that occupied by settlements (Rees and Wackernagel 1996, Folke *et al.* 1997). There is the potential for significant changes to regional or even global nutrient cycles. Vitousek *et al.* (1997) note that anthropogenic inputs have doubled the amount of fixed nitrogen in the biosphere, with serious implications for soil and water chemistry, ecosystem composition, and the planet's atmosphere. In addition, much of the pressure is relatively recent: the amount of synthetic nitrogen fertilizer applied during the 1980s exceeds cumulative use during the previous two centuries (Vitousek *et al.* 1997).

5.3 Scope for future investigations

Biologists are exploring rainforests, isolated mountaintops, and deep sea hydrothermal vents, in hopes of discovering previously-unknown species: yet in terms of microorganisms, soil is the last easily accessible frontier. New finds are literally within reach of a shovel. Only 4 000 of an estimated 3 million species of bacteria, or around 0.1%, have been described by scientists (Hawksworth 1992). Microbes are now accepted as a major component of the biosphere, with the total population of prokaryotic organisms believed to exceed 1×10^{29} cells in the oceans, 2.5×10^{29} in terrestrial soils, and 40×10^{29} in subsurface sediments: they may contain 5×10^{5} teragrams of carbon, or roughly half of the amount believed to be sequestered in vegetation (Gallagher 1998).

The ecology of individual species is also of interest, when looking at the factors behind spatial distributions. There is evidence that some types of microorganisms are capable of movement within the soil matrix, mostly through passive transportation when adhering to soil macrofauna, *e.g.* earthworms, or borne along by water as it percolates down through the horizons (Wollum 1979). Dispersal rates may be important, when considering the colonization and destruction of soil microsites where moisture and oxygen availability can change relatively quickly.

Identifying which freshwater, marine, and terrestrial microbes are capable of methanotrophy, and under which conditions, is a considerable challenge. It is even difficult to keep many of these microorganisms alive outside of their accustomed habitats (Volk 1994). In addition there is currently some disagreement on defining the species concept for microbes, since boundaries between taxa are frequently less distinct than for larger organisms (Hawksworth 1992). For these reasons, considerable uncertainty is likely to remain about the composition of microbial communities in the soil, for the forseeable future. Holmes *et al.* (1999) attempted to characterize methane-oxidizing bacteria using DNA sequences, but found that many of the species belonged to an unknown grouping which had not yet been described. There is clearly a need for more taxonomic work in this area.

Some research has been done on determining whether methanotrophs or nitrifiers are carrying out methane oxidation in a particular environment, *e.g.* through the use of specific inhibitors, chemical compounds which can prevent either CH_4 or NH_X from being oxidized (Oremland and Culbertson 1992, Davidson and Schimel 1995). Another technique involves measuring the oxidation of carbon monoxide, in addition to methane, since the $CO{:}CH_4$ ratios are different for nitrifiers and methanotrophs (Davidson and Schimel 1995). We need to learn more about how these groups of bacteria interact in microbial communities, just as we study the ecological relationships among much larger organisms (*e.g.* Huston 1979). It is already known that the addition of common pollutants, such as sulphates and benzene, can influence the results of competition between methanogens and other types of bacteria (Lovley *et al.* 1982, van Vlaardingen and van Beelen 1992).

Recent refinements of techniques such as stable isotope analysis, and the use of radioactive carbon-14 tracers, could help in tracing the origins and fate of atmospheric methane (*e.g.* Whiticar *et al.* 1990, Merritt *et al.* 1995, Trumbore *et al.* 1995, Hornibrook *et al.* 1997, Roslev *et al* 1997, Sansone *et al.* 1997, Bergamaschi *et al.* 1998). In addition, more sophisticated tools are enabling researchers to examine soils in situ, in order to minimize disruption due to handling (*e.g.* Niemann and Edgell 1993). Analysis of DNA traces may provide a means to identify particular bacteria strains in situ. Advances in both lab and field instrumentation should enable some of these questions to be addressed in more detail.

5.4 Some implications of findings

Research on the oxidation of methane and other pollutants by soil microorganisms has immediate practical applications. Various investigators have suggested that this may provide another option for treatment of landfill gas emissions. Under the right conditions, high methane consumption rates in landfill cover soils can result in significantly lower CH4 emissions, or even a negative flux in some cases (Kightley *et al.* 1995, Liptay *et al.* 1998). Bacterial uptake has fewer potential hazards than flaring (burning off combustible gases), and is cheaper, since it requires less infrastructure and maintenance. These factors could make the enhancement of soil oxidation desirable, particularly for rural or developing regions, compared to the expensive methane collection and flaring systems currently under consideration by some municipalities (Westland Resource Group 1997).

With these possibilities in mind, it would be useful to gain a greater understanding of the characteristics which favour methanotrophy. Since land use seems to play an important role in the balance between methane oxidizers and nitrifiers, this adds a new dimension to existing concerns about soil and surface water management. In the Victoria region, which has a national reputation for being garden-obsessed, there is already public concern about environmental damage caused by excessive use of water and chemicals (Westland Resource Group 1997). More natural strategies have been advocated: xeriscaping, which utilizes vegetation adapted to dry conditions, has met with success in other jurisdictions which experience water shortages, such as California and Arizona (McPherson and Haip 1989). Public education, which attempts to reshape popular attitudes regarding the desirability of smooth green lawns year-round, is ongoing.

Even if many residents and visitors have become accustomed to well-manicured gardens, it may be possible to reduce complaints about the "unsightliness" of less intensive park maintenance, as in the Oak Bay example, through conservation arguments. For instance, reduction of fertilizer use could accomplish at least three important things: less pollution to damage salmon streams and wetlands; lower emissions of N_2O, a greenhouse gas; and less impairment of soil oxidation, which should help counteract emissions of CH_4, another greenhouse gas. A similar "2-for-1" approach was used to advocate banning CFCs, based on the argument that this would reduce both ozone depletion and global warming.

This leads into a broader picture of the relationship between human societies and global environmental change. There is increasing awareness that Earth habitats are being affected by human activities -- whether directly, as in the clearing of land for agriculture or urban expansion -- or indirectly, through the emission of pollutants (Richards 1990, Meyer 1995, Dale 1997). Large areas of the planet, even distant environments which may not be regarded as socially or economically important, are absorbing and breaking down our waste products, and returning them to the system. These habitats are supporting vital processes, such as water collection and purification, flood and erosion control, and waste recycling. The economic value of such activities, described as ecosystem services (van Wilgen *et al.* 1996), has been estimated conservatively at $US 33 trillion per year, or almost twice the output of the global human economy (Costanza *et al.* 1997).

Steudler *et al.* (1996a) have found evidence that environmental conditions may be shaping the microbial composition of the soil: disturbance, and the addition of nitrogen in the form of NH_X, both favour the establishment of vigorous nitrifying communities. In habitats where human activities are less intensive, or where many decades (or even centuries) have elapsed since the soil was modified, both nitrifiers and methanotrophs exist; whereas only methanotrophs are evident in areas of low disturbance and low fertilization. This is similar to the situation described by Connell (1978) and Sousa (1979), where the intermediate disturbance hypothesis predicts maximum diversity in habitats which experience neither high nor low, but moderate amounts of disturbance. In terms of oxidation potential, the highest rates of methane uptake have been measured in soils inhabited by methanotrophs: rates are significantly lower in soils with mixed communities (both methanotrophs and nitrifiers), and lowest of all in cultivated land which is regularly fertilized, where only nitrifiers are present (Steudler *et al.* 1996a).

These results are consistent with the hypothesis offered by Schnell and King (1994), that the relationship between ammonium and methane uptake is complicated: possibly involving substrate competition between different groups of microorganisms, whose populations vary depending on both environmental and ecological conditions. Matson and Harriss (1995), when referring to the study of nitrification processes in various soils, note that the interaction of distal (overarching) and proximal (immediate) factors in regulating biogenic trace gas fluxes must be considered.

Studying soil nitrogen availability in itself poses some difficult problems. Concentrations of NH_4^+ and NO_3^- have been observed to vary considerably over scales of a few metres, and can change relatively quickly (Cain *et al.* 1999): a variety of different factors, such as the amount and type of litterfall, rates of deposition and leaching by precipitation, and differences in nitrogen uptake by plant species, may be responsible.

In general, learning more about soil processes, although they can be difficult to measure and evaluate, is essential for a clearer understanding of nutrient cycling. Raich and Schlesinger (1992) point out that relative to the combustion of fossil fuels, soil respiration in terrestrial ecosystems accounts for the release of ten to fifteen times as much CO_2; hence, learning more about the physical, chemical, and biological factors at work will contribute to our understanding of the carbon cycle, as well as to our ability to predict potential global warming feedbacks. Tans and White (1998) add that "the carbon cycle is a tightly interacting system -- improvements in knowledge of one reservoir help constrain ideas about the others." According to Schimel and Potter (1995), of the processes influencing atmospheric CH_4, researchers are only confident about their understanding of production, and much less so about oxidation and transport. More detailed information, particularly about environments which experience spatial/temporal transition between aerobic and anaerobic conditions, is required before more complex modelling can proceed.

5.4.1 Implications for soil oxidation sink evaluation

It appears that in general, monitoring soil sinks will come under a different set of constraints from the terrestrial vegetation sink currently being investigated for CO_2 abatement. Because soil processes are not taking place aboveground, techniques such as satellite monitoring or forest mensuration, which could be used to evaluate changes in net primary productivity, are not directly applicable. In addition, for compounds which are being transformed into other gases, rather than being fixed in solid form (as is the case for most of the CH_4 being oxidized by methanotrophs), chemical analysis of the soil, *e.g.* for carbon content, may not be adequate on its own. Techniques such as stable isotope analysis may be able to indicate how much methane has already been processed, through shifts in the isotope ratios.

While atmospheric trace gas sinks could theoretically be assessed by monitoring compounds which play important roles in pollutant removal, such as hydroxl radicals -- at present, direct evaluation, *e.g.* through chamber deployment, may be more suitable for estimating soil oxidation of methane or other gases. As Roslev *et al.* (1997) showed, tracer experiments could yield valuable information about what happens to the methane once it has been assimilated into bacterial biomass, what transformations it undergoes, and how long it remains in the soil -- with particular attention to how much becomes part of the soil carbon reservoir.

Presumably, if methane uptake by soils is to be included under future greenhouse gas agreements, this would require periodic measurements at selected sites, in order to ground-truth habitat mapping or remote sensing data on different environmental types within a particular jurisdiction. The Victoria study demonstrated the heterogeneity of soil conditions within similar types of environments, even at close proximity to each other. Particularly if this is to be used for official accounting purposes, in order to allocate emission reduction credits, considerable care in sampling would be required to ensure accurate values.

It might be simpler and more reliable to consider indirect measures, *e.g.* variables which are believed to have a negative impact on methane oxidation, such as the application of ammonia-based fertilizers. This would have the added bonus of focusing more attention on the nitrogen glut, which is known to contribute to other environmental problems, such as eutrophication.

CHAPTER 6 - GENERAL POLICY IMPLICATIONS

6.1 Recent changes in atmospheric methane

The average increase in methane concentrations during 1983-91, for both the northern and southern hemispheres, was just over 11 parts per billion volume (ppbv) annually (Dlugokencky *et al.* 1994a), or approximately 1% per year (Tyler *et al.* 1990). As of the late 1990s, the global rate of increase appears to be levelling out. The decline in the growth curve for atmospheric methane concentrations is more noticeable in the northern hemisphere, polewards of 30°N (Prather *et al.* 1995), possibly due to declining fossil fuel production in the former Soviet Union after the economic collapse earlier in the decade. Dlugokencky *et al.* (1998) predict that, should the situation remain unchanged, mixing ratios for atmospheric methane will stabilize at around 1.8 ppm over the next few decades; however, because the reasons behind this are still unclear, future projections are far from certain.

The implication seems to be that if these trends continue, the actual contribution of methane to enhanced global warming may be less important than the insight it can give us into biogeochemical processes, particularly since changes in atmospheric CH_4 have a shorter lag time than most other trace gases. If we can find out more about what happens to methane during its "life cycle", and the magnitude of the sources and sinks involved, it will lead to a better understanding of the global carbon budget, and help improve the accuracy of climate change projections.

6.2 Methane sinks and policy considerations

Aside from the caveat expressed earlier, that methane sinks do not function as carbon storage "vaults", there are some other factors which should be taken into account before incorporating soil and atmospheric oxidation of CH_4 into climate change policy. The atmospheric methane sink in particular requires more investigation, in order to document the spatial and temporal variability in hydroxl distribution, and account for why it appears to be declining (Prinn *et al.* 2001). A better understanding of what maintains particular sinks is essential before any meaningful policies to protect them can be developed, let alone incorporate them into pollutant mitigation efforts.

There are many obstacles to incorporating the uptake of methane by soils into any policy framework. First, there may be potential conflicts with proposed CO_2 abatement strategies. Nitrogen fertilization of boreal and temperate forests to raise productivity and speed up carbon sequestration has been suggested (Watson *et al.* 1998). However, in environments that have relatively high methane uptake rates, this could amount to trading off the CH_4 soil sink for enhanced CO_2 fixation. Some authors have expressed doubts that fertilizers will be of much use in most terrestrial environments: unless they are properly applied, runoff and denitrification will contribute to the existing nitrogen pollution problem, with only minor temporary gains in carbon storage (Anderson *et al.* 2001). Nitrous oxide (N_2O), another potent greenhouse gas, may also be generated by improper fertilizer use.

Anderson *et al.* (2001) noted that concern has been expressed, primarily by the European Union and the Alliance of Small Island States, that too many additions under Article 3.4 of the Kyoto Protocol could weaken the agreement. They argue that there would be too much emphasis placed on impermanent short-term sinks, upon which the effects of climate change are still unknown.

In addition, there might be the unintentional incentive to replace native ecosystems, such as "low carbon content scrub" with faster-growing species in managed carbon plantations, in order to maximize CO_2 credits. In the context of Article 3.3 of the Kyoto Protocol, "afforestation" has been defined as planting trees on land which has not been forested for more than half a century: it excludes reforestation efforts, *e.g.* after logging (Anderson *et al.* 2001). Given that there already are cases where rare relict savanna and open woodland environments have been "reforested" with exotics by well-intentioned government agencies and conservation groups (Stevens 1995), some protection must be included for these types of ecosystems. In the Victoria context, the oak woodlands where part of this study was carried out have recently been recognized as being in urgent need of preservation.

There is an increasing feeling that if carbon credits are to be issued for land use change, the activities should be compatible with long-term social, economic, and environmental benefits. Anderson *et al.* (2001) favour efforts such as increasing soil carbon storage by restoring cropland to continuous forage (grassland). Although it will take more than half a century for humus to develop and any significant carbon sequestration to occur, they feel that this type of project could ultimately be more effective and permanent than other management techniques, since it does not require annual intervention. Also, it tends to complement existing soil conservation initiatives.

Expanding the Kyoto Protocol to include sinks for other greenhouse gases, such as CO and N_2O, is not likely to occur in the immediate future -- at least until the first commitment period is well underway. But if the trend continues, the next step is to look beyond carbon sequestration to the processes involved. Methane sinks could help provide a model for extending the agreement to cover non-sequestration situations, in order to take more parts of the carbon cycle, and other biogeochemical cycles, into account.

Before then, we need to learn more about how these processes may be affected by global climate change. Particularly when living organisms are involved, such as vegetation and soil microorganisms, there may be unforseen effects due to climatic and ecological changes -- many of which may be exacerbated by global warming. Feedbacks within the system could complicate efforts to estimate a "safe" level for greenhouse gas emissions (Godrej 2001), essential for structuring future climate change mitigation policies. Development of greenhouse gas "budgets" requires better baseline data, and more detailed information on biogeochemical cycling characteristics in more ecosystems. In addition, human societies will be adding another layer of complexity to the mix. The Marrakesh Declaration, issued by the heads of delegations which met for COP7 (UNFCCC 2002b), expressed concern that achieving sustainable development would be complicated by the "synergies" between the policies and legislation already enacted to safeguard biological diversity and combat desertification.

Another factor which could potentially determine the outcome of these initial global efforts is the role of international trade, particularly the World Trade Organization's General Agreement on Tariffs and Trade. A carbon or energy tax that discriminates against goods that used large amounts of petroleum to travel to assembly points or to their final markets would, under current international legislation, be susceptible to challenge (Shrybman 1999).

6.3 Response strategies

The precautionary principle laid out in Article 3.3 of the UNFCCC, emphasizes the desirability of preventative strategies, such as anticipating and mitigating -- or at least minimizing -- the suspected causes of climate change as early as possible (Jepma and Munasinghe 1998). Mitigation has involved developing solutions which are either focused on limiting greenhouse gas sources, or on protecting and enhancing sinks.

However, some scientists and policymakers are also calling for adaptation: technological measures, or changes in economic structures and behaviour, to limit the vulnerability of human infrastructure, society, and managed ecosystems. As costly as these initiatives would be, *e.g.* constructing shoreline protection to guard coastal metropolitan areas against storm surges, they have the advantage that construction can be put off for a few years, until sea level rise or other global warming impacts become more apparent. Some jurisdictions, such as the United States, have recently announced that they will focus on adaptive responses to global climate change (U.S. Department of State 2002).

Among more proactive observers, concerns have been expressed that adopting a "wait and see" attitude, and deciding to rely primarily on adaptation, will not contribute to addressing this issue. If enough countries follow the "free-riding" strategy, it could set back greenhouse gas abatement efforts worldwide (Jepma and Munasinghe 1998). This would result in delays in cutting back emissions, which could lead to more pronounced climate change in the future.

Decision-making for preventative measures, or mitigation, is frequently more complex and risky than for adaptation, since it involves planning well in advance around situations which may not turn out as they were envisioned. The costs are harder to estimate, and the effects of decisions may not be immediately noticeable. There is still a perception in some quarters that international environmental agreements are so complicated and involve so many different interests as to be unworkable (Godrej 2001). Even with a rapidly-developing base of expertise in policy and economic modelling (*e.g.* Jepma and Munasinghe 1998, Newell and Pizer 2001, and others), supporters of the Kyoto Protocol have found it a challenge to gain support from all groups in the economy, particularly the powerful coal, petroleum, and heavy industry sectors. Combined with the fact that the techniques for assessing compliance are still being developed, these factors have made many nations reluctant to commit funds to mitigation, feeling that adaptation, at least for the short-term, would be more cost-effective.

Increasingly the view is being expressed that both mitigation and adaptation are needed now -- as we try to prevent future changes, or at least reduce their severity, we must also do something to safeguard people and their livelihoods, in case mitigation is not enough to avert increased climatic instability. However, acceptance of the need for both does not lessen the potential for conflict between the two approaches. They may be in competition for limited funds, especially with the recent downturn in the global economy.

Critics have complained that the Kyoto Protocol, the first international agreement of its kind, is "fatally flawed"; some are using it as an excuse to discard climate legislation altogether. Defenders of the process argue that while there are flaws, they can be rectified by a better understanding of the processes involved.

Schlamadinger and Marland (2000), in their report on LULUCF to the Pew Center, noted: "It is inevitable that a system cannot be optimized by treating only a portion of that system. The interests and concerns of negotiators have created a framework for carbon sinks but with many issues still to be decided."

Other international agreements, such as the Convention on Trade in Endangered Species (CITES), and the Montreal Protocol, have evolved over years or decades to accommodate changing conditions. Some potential problems have been anticipated by the framers of the Kyoto Protocol and Marrakesh Accords: hence the specification that traditionally non-forested ecosystems should not be converted to fast-growing tree plantations.

Analysts have identified several improvements which they feel need to be made to the Kyoto Protocol. First, Dauncey and Mazza (2001) have called attention to points which conflict with other international treaties, such as encouraging the use of hydrofluorocarbons as chlorofluorocarbon substitutes in the Montreal Protocol, even though HFCs are also in the "basket" of greenhouse gases covered by Kyoto. Second, due to interlacing biogeochemical cycles, pollutants can have interrelated histories. Ammonia is connected with methane, because certain types of methanotrophs feed on both CH_4 and NH_3. Methane is linked to CFCs, because hydroxl radicals are a removal mechanism for them, and for other pollutants as well.

Ideally, a greenhouse gas mitigation strategy should be responsive, flexible, and capable of adapting to new scientific findings. As the century unfolds, we will likely need to resolve as-yet-unforseen conflicts between different objectives of global treaties: climate protection, safeguarding biodiversity, preventing desertification, encouraging international trade.

With regards to non-CO_2 greenhouse gases, Schlamadinger and Marland (2000) admit that at present, the Kyoto Protocol is "not very clear regarding greenhouse gases other than CO_2." Methane and nitrous oxide, for example, are only viewed in the context of agricultural emissions -- not forest environments, a significant omission (Schlamadinger and Marland 2000). Even though other sections of Kyoto which apply to LULUCF seem to be including non-CO_2 gases, "verifiable changes in stocks" is still associated with sequestration of carbon dioxide in biomass and the soil. In the case of atmospheric CH_4, the "catalytic converter" analogy would be more suitable than the "lockbox", for describing a non-sequestration sink. Soils, which transform some methane into microbial biomass and release the rest as CO_2, utilize aspects of both models.

6.4 Models for planning and managemen

The Third Assessment Report (IPCC 2001) has described some of the problems inherent in developing broadly-inclusive plans for situations with uncertain outcomes. Not only is the projected change in global temperatures subject to a high degree of variability, but the extent of environmental disruption from the emissions released to date is still unknown, due to lags in the system. In addition, changes in ecosystems, technology, and human societies will continue to occur, meaning that the "ground" will constantly be shifting as policymakers attempt to develop strategies. For example, recent findings suggest that increased intensity of ultraviolet radiation has contributed to the photochemical destruction of CH_4 molecules, and that the (apparently successful) introduction of chlorofluorocarbon abatement measures aimed at protecting the ozone layer could decrease the atmospheric methane sink (IPCC 2001). These unexpected conflicts between existing environmental policies (and between environmental, trade, and social issues) will complicate future planning efforts.

Traditionally, long-range planning has focused on the comprehensive rational (or "synoptic") planning model. In its preference for understanding a situation completely, before choosing and implementing an optimal solution, it emphasizes management and control. Unfortunately, it tends to ignore both complexity and uncertainty, which are precisely the attributes which characterize issues in global environmental change, and resource management in general (Mitchell 2002). Adjustments to the comprehensive rational model, such as incorporating "feedback loops" to allow adjustments at a later date -- or the development of alternative strategies such as "disjointed incrementalism" or "mixed scanning" to cope with unclear problems and conflicting objectives -- may also not be adequate. Concern has been expressed about the setbacks that could result from "locking into" a strategy prematurely, resulting in delays to the implementation of new innovations (IPCC 2001). (An analogy might be committing to purchase expensive new computer equipment at a time of rapid technological developments, which could render the investment obsolete in matter of months.)

"Hedging" is an approach which has been proposed, as a way to deal with uncertain and rapidly-changing conditions (IPCC 2001). Mitchell (2002) defines hedging as a deliberate strategy to "avoid the worst consequences" by selecting the option with the "least bad worst outcome", which is in agreement with the precautionary principle (itself a part of established international environmental treaties such as the Montreal Protocol and the Stockholm Convention). However, hedging could itself lead to "premature adoption of new technologies" as a way of solving problems -- and being inherently conservative, it tends to uphold the status quo, rather than encouraging fundamental changes (Mitchell 2002). "Flexing", on the other hand, incorporates monitoring of a particular situation, and learning from mistakes. Unlike hedging, it increases the range of available options, provided that decisions can be altered or reversed before damage occurs (Mitchell 2002).

The concept of adaptive environmental management may be useful when developing climate change policies, since it attempts to come to terms with planning when faced with a high degree of uncertainty in natural systems (Mitchell 2002). This type of approach would be more responsive to changes in the environment and society, and accommodate a wide range of strategies -- for example, new policies that could protect and enhance alternative sinks for pollutants such as methane, rather than focusing entirely on carbon sequestration. Another advantage to adaptive environmental management is that it is designed to take advantage of experiments, and is not expected to be "correct" the first time. Critics of the Kyoto Protocol have already branded it a "failure" for not achieving its original targets for ratification or compliance, although this is the international community's first attempt to tackle climate change on such a scale. Shifting perspectives so that these efforts are seen as ongoing, rather than a one-time event like adopting a new technical innovation, would be a more realistic portrayal of the situation. Given that a long-term commitment will be needed from all parties to decrease and stabilize greenhouse gas emissions, encouraging a change in attitudes about not just human impacts on the environment, but planning itself, is required.

There are disadvantages to adaptive environmental management -- in particular the growing need for more and better-quality data, the limits of scientific theories, and surprises that confound predictions (Mitchell 2002). However, these are all issues with which the natural sciences have been familiar for decades, even if they have only recently been acknowledged by policymakers. Admitting uncertainty may be seen as a challenge to traditional planning styles, but it is more likely to result in future successes.

In conclusion, widespread industrialization has not only increased the emission of atmospheric trace gases, but has decreased the ability of environments to absorb these pollutants. We are still attempting to understand the impacts which these changes are having on the vast and complicated nutrient cycles that maintain the biosphere. To date, efforts to mitigate global climate change have, out of necessity, focused on some of these flows and reservoirs while omitting others. However, a more broadly-based approach will likely be needed this century, to make these strategies more effective and equitable.

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APPENDIX

Methodology for trace gas exchange measurement

Recent decades have seen major technological advances in atmospheric flux research, such as the introduction of new types of sensors for real-time data collection (Blaha *et al.* 1994), and airborne sampling of large areas (Greenberg *et al.* 1990, Tohjima *et al.* 1996). However, due to logistical constraints, most atmospheric trace gas exchange studies still rely on ground-based measurements. Eddy correlation flux estimates, based on micrometeorological techniques for quantifying energy and water exchanges, use sensors installed on towers to quantify local boundary-layer conditions (Oke 1978, Matson and Harriss 1995). On even smaller scales, artificial enclosures are frequently deployed to measure near-surface fluxes. While all of these methods have advantages and drawbacks, enclosure studies continue to be popular due to their low cost, relative simplicity, and usefulness for discrete measurements of intra-site variability (Livingston and Hutchinson 1995).

The near-surface exchange of trace gases is influenced primarily by two processes: advection, or mass flow; and the molecular transport of gases by diffusion (Livingston and Hutchinson 1995). (The mixed layer, further away from the Earth's surface, is dominated by turbulent processes). While diffusion rates are affected by differences in concentration -- and by the physical properties of the compounds involved, including the permeability of the medium -- advective transport of gas between the soil and the atmosphere is driven by pressure gradients at or near the surface (Livingston and Hutchinson 1995). Both diffusivity and advection must be taken into account when considering enclosure type and sampling strategy.

Design of chamber experiments

The general approach to chamber experiments involves trapping a small parcel of air next to the surface using an enclosure, and observing what happens to it over time. Specialized enclosure design for trace gas flux measurements in various environments are discussed in detail by Matson and Harriss (1995) and Livingston and Hutchinson (1995). The chambers used for this particular study were of the simplest type (Figure A.1): non-steady-state, non-flow-through, and non-vented, with a single sample port (Livingston and Hutchinson 1995). Due to these limitations, the size and frequency of the samples could not be as great, and the monitoring period as long as for steady-state chambers, which allow air to circulate and maintain a known trace gas concentration gradient (and diffusion rate). Measurement was avoided on windy days, since the turbulence created by strong winds is believed to influence pressure gradients in the soil-atmosphere interface zone (Livingston and Hutchinson 1995).

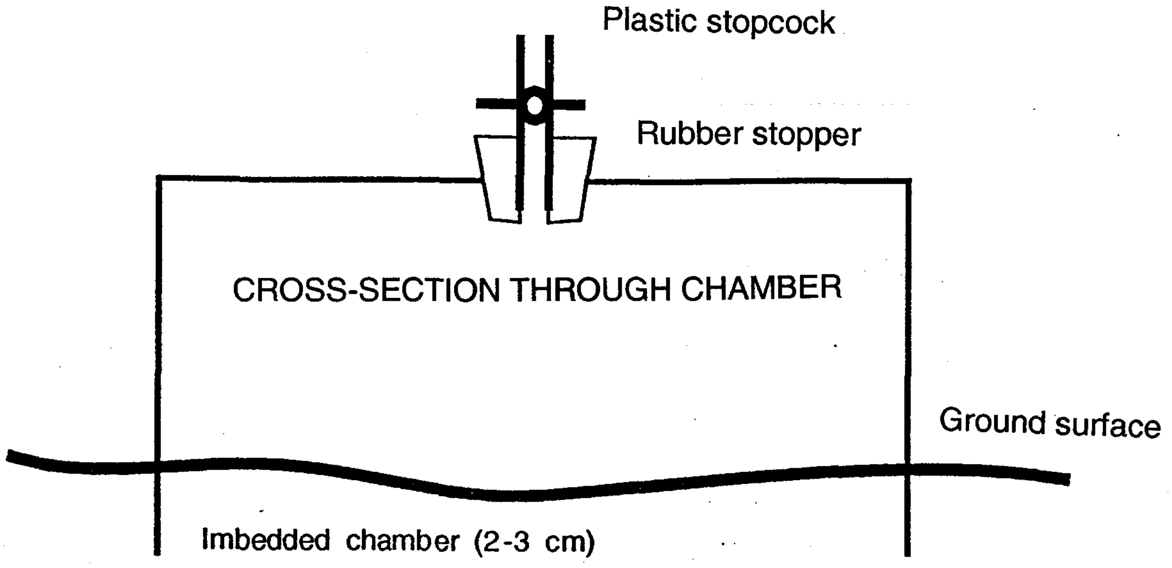


Figure A.1 - Diagram of chamber design

Admittedly, there are some potential problems with using chambers. The very act of sealing a parcel of air off from contact with the rest of the atmosphere, suppressing diffusion and convection, can create unnatural conditions. When gas concentrations reach levels above or below normal, a different set of processes may come into effect. For example, if there is vegetation inside the chamber, the plants will continue photosynthesis in the presence of adequate sunlight, lowering the CO_2 concentration until the chloroplasts are forced to shut down for lack of substrate. In terms of microclimate, chambers made of transparent materials can become miniature greenhouses, even if they are not in direct sunlight: unless the experiment was intended to examine the effects of higher temperatures, this is undesirable. Matson *et al.* (1990) noted that damage to the root mat during installation of enclosures can alter gas exchanges, resulting in a substantial buildup of N_2O after only a couple of hours. In extreme cases, if a chamber is left in place for too long, it may simulate an environment which is quite unlike its surroundings, due to the death and decomposition of vegetation over days or weeks.

For these reasons, this study was designed to measure changes over a period of 60 minutes or less. Livingston and Hutchinson (1995) have suggested that non-steady-state chambers are appropriate for short deployment periods of this nature, particularly if fluxes are expected to be low.

The first set of chambers (Type A), constructed in 1996, consisted of three sturdy cylindrical plastic planters: off-white in colour, in order to reduce heating due to excessive light penetration. The sides were approximately 4 mm thick with an inner diameter of 25 cm (Table A.1). Since enclosures with large volumes relative to their basal areas take longer to exhibit measurable changes in gas concentration (Livingston and Hutchinson 1995), the V:A (volume:basal area) ratio was reduced by cutting the chambers down to a height of 15 cm. After trimming, their interior volume was around 7.4 L.

Chamber type	**Inset (cm)**	**Height (cm)**	**Area (cm^2)**	**Volume (L)**
1996-Type A	0	15	491	7.4
1996-Type A	2	13	491	6.4
1997-Type B	0	14	410	5.6
1997-Type B	3	11	410	4.4

Table A.1 - Sampling chamber dimensions
("Area" = approximate area of ground covered by chamber)

The sample port consisted of a circular hole 11 mm in diameter, drilled in the top of the chamber, and fitted with a black rubber stopper (size "00") which had a pre-existing hole. A clear plastic Cole-Parmer one-way stopcock, with the male luer slip tip (P-30600-01 in LABCOR 1999-2000 catalogue), was inserted tightly into the stopper. The valve was kept in a closed position, except when sampling.

Trial experiments revealed that it was necessary to imbed chambers to a depth of at least 2 cm, to avoid leakage to and from the outside air. Chambers were carefully pushed into the litter or turf, assisted by cutting straight downwards with a flat-handled knife. Twisting the chambers into the ground was avoided, to minimize damage to plant roots and the possible release of trace gases (Bleakley and Tiedje 1982, Hutchinson and Brams 1992, Livingston and Hutchinson 1995).

For the 1997 field season, thinner (<2 mm) chambers (Type B) were constructed from white plastic bulk food buckets: they were lighter and more easily installed than the planters. The inner diameter of the buckets was 22.6 cm, and they were cut down to 14 cm deep: the same stopper/valve arrangement was retained. This time the interior volume was around 5.6 L. The basal areas of both types of chambers were well within the suggested range of 175 cm^2 to 1 m^2 (Livingston and Hutchinson 1995): a larger volume-to-area ratio would have increased the amount of time needed to detect a concentration change and resulted in an unacceptably long sampling interval; while a smaller such ratio might have disrupted the near-surface pressure gradients with repeated sampling.

The enclosures were judged to be firmly seated, at depths of 2 cm for the Type A chambers, and 3 cm for the Type B chambers: marks scored on the outside of the plastic were used for orientation. Where possible, these cut-in rings were re-used for later measurements, except where they had grown over again, or were damaged during removal of the chambers. Re-using rings at the forest sites was not always possible, since the crumbly soil and lack of root mat resulted in unavoidable disturbance to the soil surface whenever a chamber was removed at the end of a session.

Sampling procedure

Sampling was carried out in September and October of 1996, and July through October of 1997. The procedure utilized plastic syringes, both 3 mL and 5 mL in volume. The 5 mL syringes were preferred for samples at the end of a measurement interval, since they provided adequate sample sizes for duplicate analyses. All syringes were fitted with the same type of plastic valve installed in the chamber sample ports.

To take a sample, the syringe valve was opened and the syringe evacuated. Then, the tip of the syringe valve was inserted snugly into the protruding end of the stopper valve. The seal was tested by pulling on the syringe plunger: if there was no air intake from outside the chamber, it was assumed that the seal was complete. The stopper valve was then opened, and the syringe was filled. Afterwards, both valves were closed, and the syringe removed from the sample port. Syringes were stored with an elastic band holding down the end of the plunger, providing partial compression of the gas inside, in order to reduce the chances of contamination by leakage from the outside air.

Samples were extracted at t=0 minutes, and at regular intervals afterwards. The exact timing of sample intervals was adjusted later in the study, to focus on the first half-hour: plans to use a curve based on later measurements were scrapped, since it was felt that it might overestimate uptake rates. Occasionally, open-air samples were taken, at 1 m and 2 m above the ground, for comparison purposes. Full syringes were taken back to the lab for analysis, usually that day, or at most within 48 hours. Steudler *et al.* (1996a) found that changes in air stored in similar syringes were minor even after a week had elapsed.

Methane and the gas chromatograph

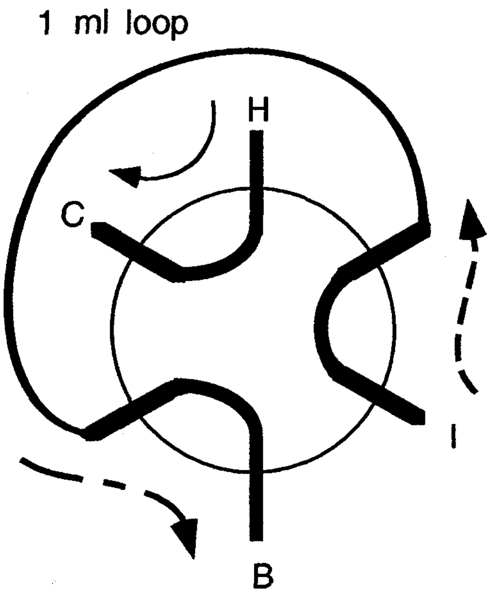
Methane concentrations were measured on an SRI gas chromatograph (Series 8610B), equipped with a flame ionization detector (FID). The use of flame ionization to measure methane is discussed in Crill *et al.* (1995): it is one of the most popular methods, due to its high reproducibility and precision. The column used in the gas chromatograph (GC) consisted of stainless steel tubing with an outer diameter of 0.125

inches, 2 m in length, 80/100 mesh, filled with Porapak "Q". The carrier gas was H_2, supplied by a hydrogen generator. The PeakSimple software package, Version 2.0, provided system control and peak evaluation.

For this study, measurements were carried out at below ambient temperature: if temperatures are too high, values would begin to drift. A variety of cooling options were tried. Initial measurements in 1996 used a tinfoil container of liquid nitrogen (LN_2). Pulverized dry ice (CO_2) lasted longer and was more cost-effective than LN_2, but care had to be exercised to prevent the dry ice from actually touching the SRI's measurement column, which would have concentrated other trace gases within the sample, increasing the difficulty of obtaining a clear CH_4 reading. In addition, the steep temperature gradients created by both LN2 and dry ice resulted in the condensation of water vapour within the SRI unit, which had not been designed with these conditions in mind.

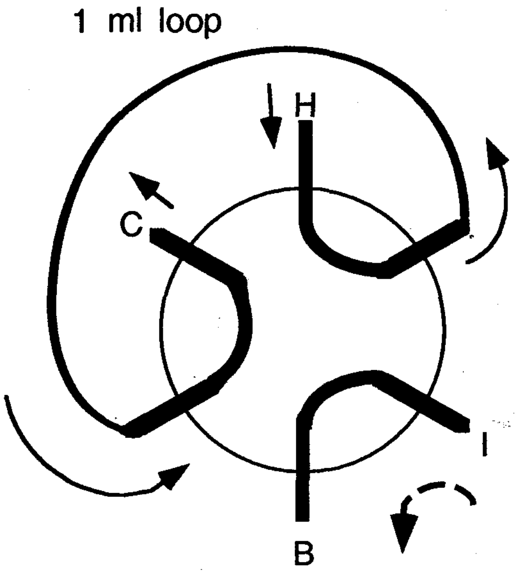
Repeated testing revealed that operating temperatures between 0 and 8°C were low enough to allow consistent CH_4 measurements. This was accomplished by placing a couple of blue-gel plastic freezer packs next to the column, then filling the remaining space with plastic-encased reusable ice cubes (which substantially reduced water leakage into the SRI unit, protecting the electronics). It was necessary to keep several batches of fresh coolant materials in the lab refrigerator, replacing them every couple of runs.

A Hamilton miniature inert plug valve (Type HVXL 6-6 loop valve) was installed on the side of the SRI, with a 1 mL plastic sample loop. Rotation of the valve provided two different flow paths: one that allowed the sample loop to be filled by docking a sample syringe and depressing the plunger; and a second option which closed off the sample loop and allowed the carrier gas to push the loop's contents into the SRI's column (Figure A.2). The 3 mL syringes were sufficient to flush the dead air space in the system and run the sample once; the 5 mL syringes could be run twice, provided at least 2 mL of sample remained.



**Position 1:
Filling the sample loop**

Sample: Injection port (I) to loop to bubbler.
Carrier gas (H) to column (C).



**Position 2:
Flow into column**

Carrier gas (H) pushes sample from loop into column.
Remains of sample injected out through bubbler (B).

Figure A.2 - Diagram of six-way valve

Data conversion and error estimates

After integration by the software, output from the SRI gas chromatograph was expressed in millivolt-seconds (mVs). These data had to be converted into more relevant concentration units, based on moles or mass of CH_4 per unit volume, or upon mixing ratios (parts per million or parts per billion). A calibration curve had already been constructed by earlier users, using the same sample loop, as part of lab support for the 1995 EOS 240 course (Figure I-3).

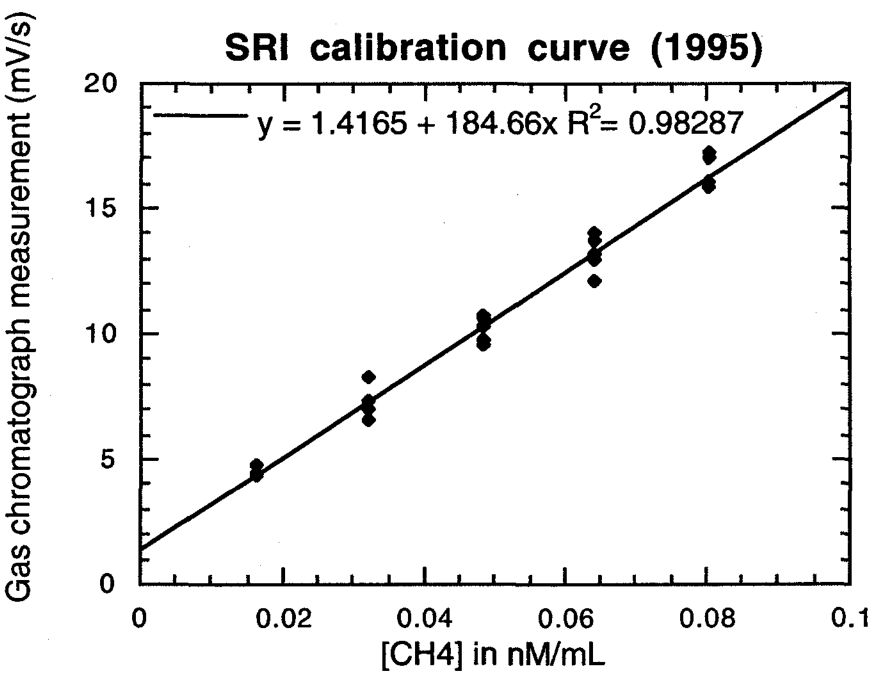


Figure A.3 - SRI gas chromatograph data calibration curve

Sources: pers. comm. Scott Harris 1998, pers. comm. Michael Whiticar 1999

This curve (solid line) describes the relationship between SRI peak area, and amounts of CH_4 in the sample:

$$y = 184.66x + 1.4165$$

where x represents [CH4] in nanomoles per mL since the 1 mL sample loop was used; y is the GC curve area in mV/s; 184.66 is the slope of the calibration curve in $\text{nmol}\cdot\text{mL}^{-1}\cdot\text{mV}^{-1}\cdot\text{s}^{-1}$; and 1.4165 is a constant, in mV/s.

Because the sample dilutions were carried out with ordinary air, not helium or pure nitrogen, the regression curve in Figure I-3 has a non-zero value for the y-intercept, representing methane present in the air mixture used at that time. The equation can be adjusted for this factor by subtracting the y-intercept value (1.4165 mV/s) so the curve passes through the origin point:

$$x = y / 184.66 \qquad (R2 = 0.98)$$

where x is in nmol/mL, y is in mV/s, and the constant is in $\text{nmol}\cdot \text{mL}^{-1}\cdot\text{mV}^{-1}\cdot\text{s}^{-1}$. The adjusted curve is shown in Figure I-4.

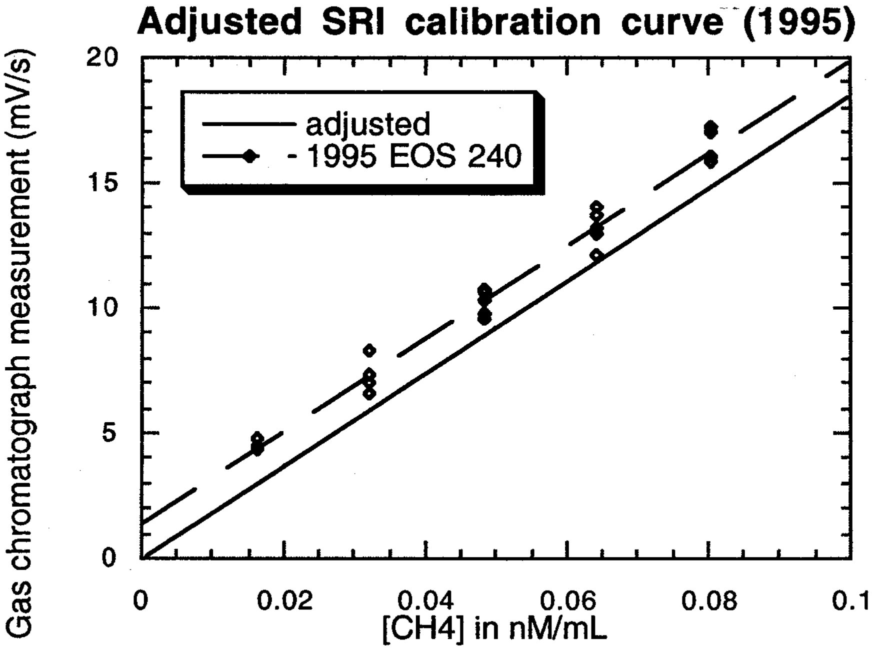


Figure A.4 - Adjusted SRI gas chromatograph calibration curve

Measurements taken during the 1997 field season in the vicinity of the sites, in the open air prior to chamber sampling (n=149), resulted in an average value of 14.21 mV/s. When substituted into the above equation, the average [CH_4] for that time period worked out to be 0.077 nmol/mL. Values expressed in nmol/mL can be converted into parts per million (ppm), using the general law for gases:

$PV = nRT$

1 mole of any gas at standard temperature and pressure occupies 22.414 L.

Therefore:

1 L = 1 / 22.414 moles
1 L = 0.045 moles
1 mole = 22.414 L
1 mmol = 22.414 mL
1 mmol/L = 22.414 mL/L

Multiplying by 1000 to convert to nanomoles and nanolitres per litre:
1000 nmol/L = 22414 nL/L

Dividing by 1000 to convert to nanomoles and nanolitres per millilitre:

1 nmol/mL = 22.414 nL/mL
Since 1 nmol/mL = 22.414 parts per million
Therefore 0.077 nmol/mL = (0.077 * 22.414) parts per million = 1.73 ppm.

Multiplying by 22.414 ppm/nmol/mL-1, 0.077 nmol/mL is equivalent to 1.72 ppm. If the error value is assumed to be plus or minus 2.14 mVs (double the standard deviation of that particular data set), it would translate into +/- 0.012 nmol/mL, or +/- 0.27 ppm. A set of open-air measurements for the 1996 field season, though smaller (n=77), yielded similar results: 14.57 (+/- 1.48) mV/s , or 0.079 (+/- 0.008) nmol/mL, or 1.77 (+/- 0.18) ppm.

The 1997 values can be double-checked in part, using estimates for the 1997 seasonal average atmospheric methane concentration for this region of North America. Quay *et al.* (1999), for sites at both Cape Meares, Oregon, and the Olympic Peninsula in Washington State, estimated the mean CH4 concentration for the mid-1990s at 1.775-1.810 parts per million, for the months of July through October. In fact, Dlugokencky *et al.* (1998) predicted that atmospheric concentrations of methane, which have been experiencing decreasing growth rates since the mid-1980s, should stabilize at around 1.8 ppm. Cape Meares is the more established of the two stations, and a time series from that site for average August values is shown in Figure I-5.

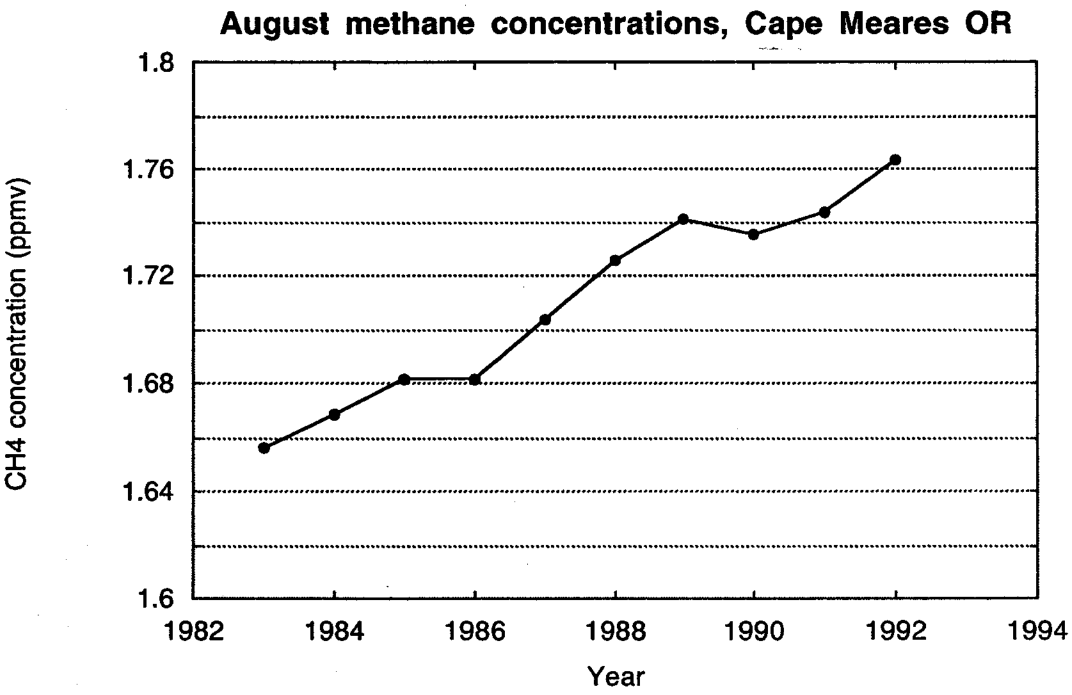


Figure A.5 - Cape Meares atmospheric CH4 August time series, 1983-92

August is the middle of the measurement period, for the 1997 University of Victoria data: most samples were taken in August, and the months preceding and following it. This curve shows a noticeable decrease in the rate of change after 1988. Assuming that this trend continued through the 1990s, as inferred by Dlugokencky *et al.* (1998), the 1988-92 portion of the curve was extrapolated using KaleidaGraph for the Macintosh (Version 3.08) to arrive at a regional estimate for atmospheric methane concentrations during the 1997 field season (Figure I-6).

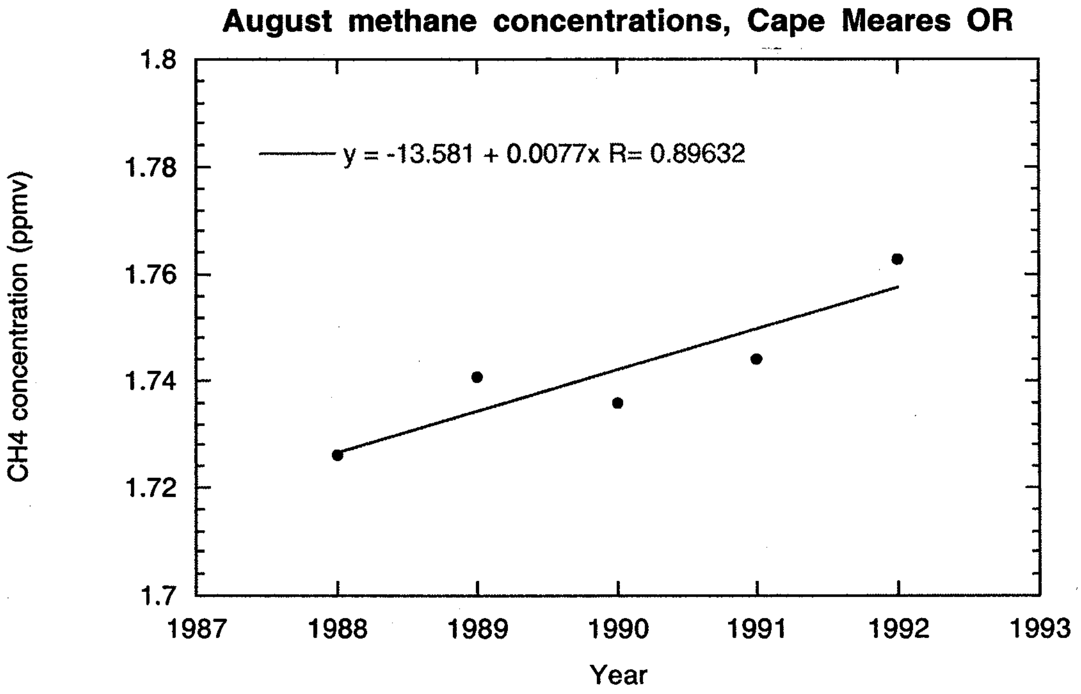


Figure A.6 - Cape Meares atmospheric CH_4 August trend, 1988-92

The linear regression equation derived is:

$$y = 0.0077x - 13.581 \quad (R2 = 0.80)$$

where y is CH_4 concentration in ppm, and x is the year. This equation can be used to estimate a value of 1.80 ppm, for the August 1997 CH_4 mixing ratio. This is within the range of 1.47 to 1.99 ppm worked out for the methane concentration of ambient air during the 1997 field season, previously calculated. Experimentation with the 1 mL loop on the SRI revealed a standard deviation of 0.004 nmol, for n=15. Two standard deviations encompasses the 95% confidence interval (0.008 nmol/mL, or 0.18 ppm). This is actually within the +/- 0.26 ppm margin of error calculated earlier.

Estimation of methane uptake rates

Converting from concentration values (nmol/mL) to uptake rates requires information on the sample chambers, specifically their internal volume when imbedded in the soil (Table A.1). To get the total amount of methane within the chamber, the sample concentration value is multiplied by the chamber volume (6.4 L for Type A, 4.4 L for Type B). Most methane uptake data are reported in terms of mass, so molar measurements can be converted by multiplying them by the molecular mass of 16.04 g/mol.

The uptake rate is calculated by taking the difference in the amount of methane, *i.e.* between t=0 min and t=30 min, and dividing it by the basal area of the chamber (0.05 m^2 for Type A, 0.04 m2 for Type B), to get the change in methane per unit area, over that span of time. A more elaborate procedure would have involved calculating the slope of the curve connecting all the points between t=0 and t=60, but the t=60 values were likely approaching the limits of reliability for non-steady-state, non-flow-through chamber designs, since the angle of the curve frequently flattened out after the first 30 minutes. The t=30 concentrations were chosen because they tended to be more conservative than uptake rates based on 10 or 15-minute values.

Although most enclosure studies of this type use 20-40 minute measurement periods (Livingston and Hutchinson 1995), results are usually reported as hourly or daily estimates. When results from different methane uptake studies are being compared, it may be valuable to know not only the design type, volume, and basal area, but also deployment period of the chambers. Table A.2 lists 30-minute concentration changes, and further information for selected sites, including landscape type classification, and the estimated number of days since the last noticeable rainfall event. All data collected are shown in Table A.3.

Campus Garden (tall grass)

Decrease in CH_4, from t=0 to t=30 minutes; 4.4 L chamber, 0.04 m^2

Site	nmol/mL	ng/mL	Date ddmmyy	Time	Days dry
CGg *	0.0028	0.045	310797	14:45	10
CGg	*0.0021	0.034	100997	15:30	9
CGg	0.0079	0.127	100997	15:30	9
CGg	0.0064	0.103	100997	15:30	9

Campus Garden (plot boundary)

Decrease in CH_4, from t=0 to t=30 minutes; 4.4 L chamber, 0.04 m^2

Site	nmol/mL	ng/mL	Date ddmmyy	Time	Days dry
CGm	0.0056	0.090	310797	14:45	irrig

Campus Garden (bare soil)

Decrease in CH_4, from t=0 to t=20 minutes; 4.4 L chamber, 0.04 m^2

Site	nmol/mL	ng/mL	Date ddmmyy	Time	Days dry
CGs	0.0061	0.098	310797	14:45	irrig

Mt. Tolmie (rock)

Decrease in CH_4, from t=0 to t=20 minutes; 4.4 L chamber, 0.04 m^2

Site	nmol/mL	ng/mL	Date ddmmyy	Time	Days dry
MTB*	-0.0025	-0.040	070897	15:00	1
MTB*	0.0032	0.051	270897	12:00	1

Mt. Tolmie (under oaks)

Decrease in CH_4, from t=0 to t=30 minutes; 4.4 L chamber, 0.04 m^2

Site	**nmol/mL**	**ng/mL**	**Date ddmmyy**	**Time**	**Days dry**
MTb	0.0118	0.189	010897	11:15	11
MTg	0.0204	0.327	010897	11:15	11
MTC*	0.0190	0.305	070897	15:00	1
MTC*	0.0113	0.181	270897	12:00	1
MTC*	0.0122	0.196	280897	12:00	2
MTC*	0.0224	0.359	040997	11:15	3
MTC*	0.0275	0.441	070997	16:00	6
MTC*	0.0222	0.356	090997	16:45	8
MTC*	0.0032	0.051	110997	15:45	10
MTC*	0.0164	0.265	130997	14:30	1
MTC*	0.0171	0.274	021097	16:30	1
MTC*	0.0119	0.191	061097	16:00	2
MTE*	0.0181	0.290	280897	12:00	2
MTE*	0.0285	0.457	040997	11:15	3
MTE*	0.0260	0.417	090997	16:45	8
MTE*	0.0221	0.354	110997	15:45	10
MTE*	0.0166	0.266	130997	14:30	1
MTE*	0.0224	0.359	150997	12:45	1
MTE*	0.0319	0.512	190997	16:00	2
MTF*	0.0053	0.085	070997	16:00	6
MTF*	0.0201	0.322	090997	16:45	8
MTF*	0.0160	0.257	110997	15:45	10
MTF*	0.0131	0.201	130997	14:30	1
MTF*	0.0305	0.489	150997	12:45	1
MTF*	0.0292	0.468	190997	16:00	2
MTF*	0.0328	0.526	250997	13:50	wet
MTF*	0.0185	0.297	250997	15:00	wet
MTF*	0.0261	0.419	021097	16:30	1
MTF*	0.0316	0.507	061097	16:00	2

Campus lawn

Decrease in CH_4, from t=0 to t=30 minutes; 6.4 L chamber, 0.05 m^2

Site	nmol/mL	ng/mL	Date ddmmyy	Time	Days dry
CLo	0.0015	0.024	091096	13:00	irrig

Campus lawn

Decrease in CH_4, from t=0 to t=30 minutes; 4.4 L chamber, 0.04 m^2

Site	nmol/mL	ng/mL	Date ddmmyy	Time	Days dry
CLe	-0.0004	-0.0006	250797	13:30	irrig
CLd	0.0004	0.006	080897	14:30	2
CLg	-0.0019	-0.030	080897	14:30	irrig
CLj	-0.0044	-0.070	250797	13:30	4
CLs	0.0036	0.058	080897	14:30	wet
CLo	0.0011	0.018	070997	17:00	irrig
CLo	0.0103	0.165	070997	17:00	irrig
CLo	0.0071	0.114	080997	17:00	irrig
CLo	0.0032	0.051	080997	17:00	irrig
CLo	0.0042	0.067	080997	18:00	irrig

Mt. Tolmie (meadow -- disturbed?)

Decrease in CH_4, from t=0 to t=30 minutes; 4.4 L chamber, 0.04 m^2

Site	nmol/mL	ng/mL	Date ddmmyy	Time	Days dry
MTD*	0.0053	0.085	070897	15:00	1
MTD*	0.0090	0.144	270897	12:00	1
MTD*	0.0049	0.078	190997	16:00	2
MTD*	0.0057	0.091	250997	13:50	wet
MTD*	0.0060	0.096	250997	15:00	wet
MTD*	0.0034	0.054	021097	16:30	1
MTD*	0.0140	0.224	061097	16:00	2

Gordon Head abandoned hayfield

Decrease in CH_4, from t=0 to t=30 minutes; 6.4 L chamber, 0.05 m^2

Site	nmol/mL	ng/mL	Date ddmmyy	Time	Days dry
GHf	0.0133	0.213	250996	14:00	3
GHf	0.0171	0.274	270996	17:00	5
GHf	0.0171	0.274	011096	14:30	9
GHf	0.0132	0.212	031096	15:30	1

Campus closed-canopy mixed woods

Decrease in CH_4, from t=0 to t=30 minutes; 4.4 L chamber, 0.04 m^2

Site	nmol/mL	ng/mL	Date ddmmyy	Time	Days dry
CWl	0.0072	0.115	280897	14:00	2
CWl	0.0103	0.165	280897	14:00	2
CWl	0.0133	0.213	040997	13:15	3
CWl	0.0186	0.298	130997	16:00	1
CWl	0.0205	0.329	150997	15:00	1
CWl	0.0066	0.106	061097	17:00	2
CWm	0.0075	0.120	040997	13:15	3
CWm	0.0308	0.494	061097	17:00	2
CWm*	0.0107	0.172	130997	16:00 1	
CWm*	0.0068	0.109	150997	15:00	1

Table A.2 - Changes in chamber CH_4 concentrations

** indicates exact same location for chambers at that particular site*

TABLE A.3 - CONCENTRATIONS FROM CHAMBER EXPERIMENTS

#	Sampled	Run	Location	Min	Time	Peak Area	nM CH4/mL	change	nmol/m2	mg/m2	ppm
263	Sept 25, 96	Sept 27, 96	GH meadow	t0	14:00	14.72	0.08274				1.85
270	Sept 25, 96	Sept 27, 96		t5		13.97	0.07868	-0.0041	-527.8320	-0.0085	1.76
273	Sept 25, 96	Sept 27, 96		t15		13.19	0.07446	-0.0083	-1076.7772	-0.0173	1.67
262	Sept 25, 96	Sept 27, 96		t30		12.27	0.06948	-0.0133	-1724.2511	-0.0277	1.56
266	Sept 25, 96	Sept 27, 96		t60		11.81	0.06699	-0.0158	-2047.9880	-0.0328	1.50
267	Sept 25, 96	Sept 27, 96		t150		11.13	0.06330	-0.0194	-2526.5557	-0.0405	1.42
286	Sept 27, 96	Sept 27, 96	GH meadow	t0	17:00	15.66	0.08783				1.97
296	Sept 27, 96	Sept 27, 96		t5		14.35	0.08074				1.81
294	Sept 27, 96	Sept 27, 96		t15		12.8	0.07235	-0.0155	-2012.7992	-0.0323	1.62
287	Sept 27, 96	Sept 27, 96		t30		12.51	0.07078	-0.0171	-2216.8942	-0.0356	1.59
306	Sept 27, 96	Sept 27, 96		t60		11.45	0.06504	-0.0228	-2962.8967	-0.0475	1.46
309	Sept 27, 96	Sept 27, 96		t120		10.12	0.05783	-0.0300	-3898.9188	-0.0625	1.30
337	Oct 1, 96	Oct 2, 96	GH meadow	t0	14:30	15.16	0.08513				1.91
335	Oct 1, 96	Oct 2, 96		t5		14.84	0.08339				1.87
327	Oct 1, 96	Oct 2, 96		t15		13.33	0.07522	-0.0099	-1287.9100	-0.0207	1.69
345	Oct 1, 96	Oct 2, 96		t30		12	0.06801	-0.0171	-2223.9320	-0.0357	1.52
338	Oct 1, 96	Oct 2, 96		t60		11.62	0.06596	-0.0192	-2491.3669	-0.0400	1.48
351	Oct 1, 96	Oct 2, 96		t120		10.8	0.06152	-0.0236	-3068.4631	-0.0492	1.38
375	Oct 3, 96	Oct 4, 96	GH meadow	t0	15:30	14.45	0.08128				1.82
383	Oct 3, 96	Oct 4, 96		t5		13.57	0.07652				1.72
390	Oct 3, 96	Oct 4, 96		t15		13.08	0.07386	-0.0074	-964.1731	-0.0155	1.66
369	Oct 3, 96	Oct 4, 96		t30		12.02	0.06812	-0.0132	-1710.1756	-0.0274	1.53
368	Oct 3, 96	Oct 4, 96		t60		12.2	0.06910	-0.0122	-1583.4959	-0.0254	1.55
372	Oct 3, 96	Oct 4, 96		t120		10.32	0.05892	-0.0224	-2906.5947	-0.0466	1.32
401	Oct 9, 96	Oct 11, 96	Campus lawn	t0	13:00?	14.51	0.08161				1.83
407	Oct 9, 96	Oct 11, 96		t5		14.63	0.08226				1.84
412	Oct 9, 96	Oct 11, 96		t15		15.53	0.08713	0.0055	717.8515	0.0115	1.95
405	Oct 9, 96	Oct 11, 96		t30		14.23	0.08009	-0.0015	-197.0573	-0.0032	1.80
410	Oct 9, 96	Oct 11, 96		t60		13.62	0.07679	-0.0048	-626.3606	-0.0100	1.72

411	Oct 9, 96	Oct 11, 96	t120		13.79	0.07771	-0.0039	-506.7187	-0.0081	1.74
421	Oct 16, 96	Oct 18, 96 Campus lawn	t0	14:45	14.36	0.08079				1.81
418	Oct 16, 96	Oct 18, 96	t15		12.93	0.07305	-0.0077	-1006.3996	-0.0161	1.64
428	Oct 16, 96	Oct 18, 96	t60		11.52	0.06542	-0.0154	-1998.7237	-0.0321	1.47
427	Oct 16, 96	Oct 18, 96	t120		10.23	0.05843	-0.0224	-2906.5947	-0.0466	1.31
438	Oct 16, 96	Oct 18, 96	t180		10.07	0.05756	-0.0232	-3019.1988	-0.0484	1.29
425	Oct 16, 96	Oct 18, 96 Campus lawn	t0	14:45	14.56	0.08188				1.84
439	Oct 16, 96	Oct 18, 96	t5		13.72	0.07733				1.73
436	Oct 16, 96	Oct 18, 96	t120		12.49	0.07067	-0.0112	-1456.8162	-0.0234	1.58
416	Oct 16, 96	Oct 18, 96	t180	14:45	13.12	0.07408	-0.0078	-1013.4374	-0.0163	1.66
447	Jul 25, 97	Jul 25, 97 J-hut turf	t0		14.5	0.08155				1.83
454	Jul 25, 97	Jul 25, 97	t5		15	0.08426				1.89
458	Jul 25, 97	Jul 25, 97	t15		14.81	0.08323	0.0017	184.7474	0.0030	1.87
446	Jul 25, 97	Jul 25, 97	t30	13:30	15.32	0.08599	0.0044	488.6868	0.0078	1.93
456	Jul 25, 97	Jul 25, 97	t120		15.37	0.08626	0.0047	518.4847	0.0083	1.93
449	Jul 25, 97	Jul 25, 97 asphalt EJ	t0		14.78	0.08307				1.86
455	Jul 25, 97	Jul 25, 97	t5		15.02	0.08437				1.89
451	Jul 25, 97	Jul 25, 97	t15		15.02	0.08437	0.0013	182.0208	0.0029	1.89
450	Jul 25, 97	Jul 25, 97	t30		15.5	0.08697	0.0039	546.0625	0.0088	1.95
453	Jul 25, 97	Jul 25, 97	t120		15.2	0.08534	0.0023	318.5365	0.0051	1.91
459	Jul 25, 97	Jul 25, 97 E-hut lawn	t0		14.65	0.08237				1.85
461	Jul 25, 97	Jul 25, 97	t5		14.85	0.08345				1.87
457	Jul 25, 97	Jul 25, 97	t15		14.8	0.08318	0.0008	89.3939	0.0014	1.86
448	Jul 25, 97	Jul 25, 97	t30		14.73	0.08280	0.0004	47.6768	0.0008	1.86
460	Jul 25, 97	Jul 25, 97	t120		14.87	0.08356	0.0012	131.1111	0.0021	1.87
468	Jul 31, 97	Jul 31, 97 Garden A-ring	t0	14:45	14.42	0.08112				1.82
484	Jul 31, 97	Jul 31, 97	t10		14.35	0.08074				1.81
482	Jul 31, 97	Jul 31, 97	t20		14.01	0.07890	-0.0022	-244.3434	-0.0039	1.77
467	Jul 31, 97	Jul 31, 97	t30		13.9	0.07830	-0.0028	-309.8989	-0.0050	1.76
475	Jul 31, 97	Jul 31, 97	t60		14.39	0.08096	-0.0002	-17.8788	-0.0003	1.81
492	Jul 31, 97	Jul 31, 97 Garden edge	t0		14.9	0.08372				1.88

470	Jul 31, 97	Jul 31, 97	t10		14.27	0.08031				1.80
493	Jul 31, 97	Jul 31, 97	t20		14.19	0.07987	-0.0038	-423.1312	-0.0068	1.79
487	Jul 31, 97	Jul 31, 97	t30		13.87	0.07814	-0.0056	-613.8383	-0.0098	1.75
471	Jul 31, 97	Jul 31, 97	t60		13.38	0.07549	-0.0082	-905.8584	-0.0145	1.69
469	Jul 31, 97	Jul 31, 97 Garden soil	t0		15.17	0.08518				1.91
476	Jul 31, 97	Jul 31, 97	t10		14.19	0.07987				1.79
472	Jul 31, 97	Jul 31, 97	t20		14.04	0.07906	-0.0061	-673.4342	-0.0108	1.77
490	Jul 31, 97	Jul 31, 97	t60		14.24	0.08015	-0.0050	-554.2423	-0.0089	1.80
497	Aug 1, 97	Aug 1, 97 MT green	t0	11:15	16.99	0.09504				2.13
499	Aug 1, 97	Aug 1, 97	t10		14.01	0.07890				1.77
525	Aug 1, 97	Aug 1, 97	t20		13.14	0.07419	-0.0208	-2294.4440	-0.0368	1.66
521	Aug 1, 97	Aug 1, 97	t30		13.23	0.07468	-0.0204	-2240.8076	-0.0359	1.67
510	Aug 1, 97	Aug 1, 97	t60		12.77	0.07218	-0.0229	-2514.9490	-0.0403	1.62
498	Aug 1, 97	Aug 1, 97	t90		12.59	0.07121	-0.0238	-2622.2217	-0.0421	1.60
527	Aug 1, 97	Aug 1, 97 MT brown	t0		16.34	0.09152				2.05
508	Aug 1, 97	Aug 1, 97	t10		14.49	0.08150				1.83
504	Aug 1, 97	Aug 1, 97	t20		13.67	0.07706	-0.0145	-1591.2118	-0.0255	1.73
511	Aug 1, 97	Aug 1, 97	t30		14.17	0.07977	-0.0118	-1293.2320	-0.0207	1.79
515	Aug 1, 97	Aug 1, 97	t60		14.53	0.08172	-0.0098	-1078.6866	-0.0173	1.83
507	Aug 1, 97	Aug 1, 97	t90		13.99	0.07879	-0.0127	-1400.5047	-0.0225	1.77
534	Aug 7, 97	Aug 7, 97 MT B-ring	t0	15:00	15.02	0.08437				1.89
547	Aug 7, 97	Aug 7, 97	t10		14.79	0.08312				1.86
545	Aug 7, 97	Aug 7, 97	t20		14.86	0.08350	-0.0009	-95.3535	-0.0015	1.87
536	Aug 7, 97	Aug 7, 97	t30		15.48	0.08686	0.0025	274.1414	0.0044	1.95
560	Aug 7, 97	Aug 7, 97	t60		14.61	0.08215	-0.0022	-244.3434	-0.0039	1.84
558	Aug 7, 97	Aug 7, 97 MT C-ring	t0		15.34	0.08610				1.93
554	Aug 7, 97	Aug 7, 97	t10		13.18	0.07440	-0.0117	-1287.2725	-0.0206	1.67
556	Aug 7, 97	Aug 7, 97	t20		12.21	0.06915	-0.0170	-1865.3531	-0.0299	1.55
541	Aug 7, 97	Aug 7, 97	t30		11.83	0.06709	-0.0190	-2091.8177	-0.0336	1.50
562	Aug 7, 97	Aug 7, 97	t60		11.59	0.06579	-0.0203	-2234.8480	-0.0358	1.47
532	Aug 7, 97	Aug 7, 97 MT D-ring	t0		14.63	0.08226				1.84

539	Aug 7, 97	Aug 7, 97	t10		14.51	0.08161				1.83
542	Aug 7, 97	Aug 7, 97	t20		13.66	0.07700	-0.0053	-578.0807	-0.0093	1.73
551	Aug 7, 97	Aug 7, 97	t30		13.66	0.07700	-0.0053	-578.0807	-0.0093	1.73
544	Aug 7, 97	Aug 7, 97	t60		13.12	0.07408	-0.0082	-899.8988	-0.0144	1.66
575	Aug 8, 97	Aug 8, 97 Cunn dry	t0	14:30	14.89	0.08367				1.88
586	Aug 8, 97	Aug 8, 97	t10		14.28	0.08036				1.80
588	Aug 8, 97	Aug 8, 97	t20		14.66	0.08242	-0.0012	-137.0707	-0.0022	1.85
574	Aug 8, 97	Aug 8, 97	t30		14.82	0.08329	-0.0004	-41.7172	-0.0007	1.87
601	Aug 8, 97	Aug 8, 97	t60		14.75	0.08291	-0.0008	-83.4343	-0.0013	1.86
595	Aug 8, 97	Aug 8, 97 MacLau lawn	t0		14.38	0.08090				1.81
593	Aug 8, 97	Aug 8, 97	t10		14.72	0.08274				1.85
584	Aug 8, 97	Aug 8, 97	t20		14.91	0.08377	0.0029	315.8585	0.0051	1.88
600	Aug 8, 97	Aug 8, 97	t30		14.73	0.08280	0.0019	208.5858	0.0033	1.86
591	Aug 8, 97	Aug 8, 97	t60		14.61	0.08215	0.0012	137.0707	0.0022	1.84
579	Aug 8, 97	Aug 8, 97 MacLau wet	t0		15.37	0.08626				1.93
597	Aug 8, 97	Aug 8, 97	t10		14.2	0.07993				1.79
572	Aug 8, 97	Aug 8, 97	t20		14.5	0.08155	-0.0047	-518.4847	-0.0083	1.83
592	Aug 8, 97	Aug 8, 97	t30		14.7	0.08264	-0.0036	-399.2928	-0.0064	1.85
590	Aug 8, 97	Aug 8, 97	t60		14.77	0.08302	-0.0032	-357.5757	-0.0057	1.86
617	Aug 27, 97	Aug 27, 97 MT D-ring	t0	12:00	15.06	0.08459				1.90
620	Aug 27, 97	Aug 27, 97	t10		14	0.07885				1.77
618	Aug 27, 97	Aug 27, 97	t20		12.87	0.07273	-0.0119	-1305.1512	-0.0209	1.63
610	Aug 27, 97	Aug 27, 97	t30		13.39	0.07554	-0.0090	-995.2523	-0.0160	1.69
614	Aug 27, 97	Aug 27, 97	t60		13.21	0.07457	-0.0100	-1102.5250	-0.0177	1.67
619	Aug 27, 97	Aug 27, 97 MT C-ring	t0		13.97	0.07868				1.76
613	Aug 27, 97	Aug 27, 97	t10		13.25	0.07478				1.68
605	Aug 27, 97	Aug 27, 97	t20		12.53	0.07088	-0.0078	-858.1816	-0.0138	1.59
265	Aug 27, 97	Aug 27, 97	t30		11.89	0.06742	-0.0113	-1239.5957	-0.0199	1.51
611	Aug 27, 97	Aug 27, 97	t60		11.22	0.06379	-0.0149	-1638.8885	-0.0263	1.43
607	Aug 27, 97	Aug 27, 97 MT B-ring	t0		14.47	0.08139				1.82
608	Aug 27, 97	Aug 27, 97	t10		15.67	0.08789				1.97

616	Aug 27, 97	Aug 27, 97		t20		13.87	0.07814	-0.0032	-357.5757	-0.0057	1.75
621	Aug 27, 97	Aug 27, 97		t60		14.67	0.08247	0.0011	119.1919	0.0019	1.85
640	Aug 28, 97	Aug 28, 97	MT C-ring	t0	12:00	13.44	0.07581				1.70
631	Aug 28, 97	Aug 28, 97		t15		12.72	0.07191	-0.0039	-429.0908	-0.0069	1.61
648	Aug 28, 97	Aug 28, 97		t30		11.18	0.06357	-0.0122	-1346.8684	-0.0216	1.42
644	Aug 28, 97	Aug 28, 97		t60		11.07	0.06298	-0.0128	-1412.4239	-0.0227	1.41
632	Aug 28, 97	Aug 28, 97	MT C-ring	t0		13.94	0.07852				1.76
643	Aug 28, 97	Aug 28, 97		t15		12.42	0.07029	-0.0082	-905.8584	-0.0145	1.58
636	Aug 28, 97	Aug 28, 97		t30		10.59	0.06038	-0.0181	-1996.4642	-0.0320	1.35
646	Aug 28, 97	Aug 28, 97		t60		10.04	0.05740	-0.0211	-2324.2419	-0.0373	1.29
634	Aug 28, 97	Aug 28, 97	CW-litter	t0		15.14	0.08502				1.91
641	Aug 28, 97	Aug 28, 97		t15		14.49	0.08150	-0.0035	-387.3737	-0.0062	1.83
633	Aug 28, 97	Aug 28, 97		t30		13.81	0.07782	-0.0072	-792.6261	-0.0127	1.74
650	Aug 28, 97	Aug 28, 97		t60		14.19	0.07987	-0.0051	-566.1615	-0.0091	1.79
645	Aug 28, 97	Aug 28, 97	CW-litter	t0		14.23	0.08009				1.80
647	Aug 28, 97	Aug 28, 97		t15		13.03	0.07359	-0.0065	-715.1514	-0.0115	1.65
639	Aug 28, 97	Aug 28, 97		t30		12.33	0.06980	-0.0103	-1132.3230	-0.0182	1.56
649	Aug 28, 97	Aug 28, 97		t60		11.73	0.06655	-0.0135	-1489.8987	-0.0239	1.49
659	Sept 4, 97	Sept 4, 97	MT C-ring	t0	11:15	14.15	0.07966				1.79
673	Sept 4, 97	Sept 4, 97		t15		11.26	0.06401	-0.0157	-1722.3229	-0.0276	1.43
669	Sept 4, 97	Sept 4, 97		t30		10.02	0.05729	-0.0224	-2461.3126	-0.0395	1.28
662	Sept 4, 97	Sept 4, 97		t60		10.06	0.05751	-0.0221	-2437.4742	-0.0391	1.29
661	Sept 4, 97	Sept 4, 97	MT E-ring	t0		14.59	0.08204				1.84
665	Sept 4, 97	Sept 4, 97		t15		10.41	0.05940	-0.0226	-2491.1106	-0.0400	1.33
671	Sept 4, 97	Sept 4, 97		t30		9.33	0.05356	-0.0285	-3134.7468	-0.0503	1.20
670	Sept 4, 97	Sept 4, 97		t60		8.92	0.05134	-0.0307	-3379.0902	-0.0542	1.15
655	Sept 4, 97	Sept 4, 97	CW-moss	t0	13:15	14.69	0.08258				1.85
657	Sept 4, 97	Sept 4, 97		t15		13.95	0.07857	-0.0040	-441.0100	-0.0071	1.76
666	Sept 4, 97	Sept 4, 97		t30		13.3	0.07505	-0.0075	-828.3837	-0.0133	1.68
672	Sept 4, 97	Sept 4, 97		t60		13.02	0.07354	-0.0090	-995.2523	-0.0160	1.65
667	Sept 4, 97	Sept 4, 97	CW-litter	t0		15.83	0.08876				1.99

663	Sept 4, 97	Sept 4, 97	t15		14.36	0.08079	-0.0080	-876.0604	-0.0141	1.81
658	Sept 4, 97	Sept 4, 97	t30		13.38	0.07549	-0.0133	-1460.1007	-0.0234	1.69
656	Sept 4, 97	Sept 4, 97	t60		13.58	0.07657	-0.0122	-1340.9088	-0.0215	1.72
693	Sept 7, 97	Sept 7, 97 MT C-ring	t0	16:00	13.75	0.07749				1.74
683	Sept 7, 97	Sept 7, 97	t15		11.99	0.06796	-0.0095	-1048.8887	-0.0168	1.52
681	Sept 7, 97	Sept 7, 97	t30		8.68	0.05004	-0.0275	-3021.5145	-0.0485	1.12
684	Sept 7, 97	Sept 7, 97	t60		9.19	0.05280	-0.0247	-2717.5752	-0.0436	1.18
686	Sept 7, 97	Sept 7, 97 MT F-ring	t0		12.36	0.06996				1.57
679	Sept 7, 97	Sept 7, 97	t15		11.18	0.06357	-0.0064	-703.2322	-0.0113	1.42
695	Sept 7, 97	Sept 7, 97	t30		8.21	0.04749	-0.0053	-584.0403	-0.0094	1.06
680	Sept 7, 97	Sept 7, 97	t60		9.17	0.05269	-0.0173	-1901.1107	-0.0305	1.18
697	Sept 7, 97	Sept 7, 97 Campus lawn	t0	17:00	13.98	0.07874				1.76
690	Sept 7, 97	Sept 7, 97	t30		13.77	0.07760	-0.0011	-125.1515	-0.0020	1.74
682	Sept 7, 97	Sept 7, 97	t60		13.28	0.07495	-0.0038	-417.1716	-0.0067	1.68
688	Sept 7, 97	Sept 7, 97 Campus lawn	t0		13.93	0.07847				1.76
691	Sept 7, 97	Sept 7, 97	t15		13.45	0.07587	-0.0026	-286.0605	-0.0046	1.70
698	Sept 7, 97	Sept 7, 97	t30		12.03	0.06818	-0.0103	-1132.3230	-0.0182	1.53
685	Sept 7, 97	Sept 7, 97	t60		12.89	0.07283	-0.0056	-619.7978	-0.0099	1.63
706	Sept 8, 97	Sept 8, 97 Campus lawn	t0	17:00	12.58	0.07116				1.59
715	Sept 8, 97	Sept 8, 97	t15		13.69	0.07717	0.0060	661.5150	0.0106	1.73
710	Sept 8, 97	Sept 8, 97	t30		12.61	0.07132	0.0002	17.8788	0.0003	1.60
718	Sept 8, 97	Sept 8, 97	t60		11.33	0.06439	-0.0068	-744.9493	-0.0119	1.44
717	Sept 8, 97	Sept 8, 97 Campus lawn	t0		13.93	0.07847				1.76
708	Sept 8, 97	Sept 8, 97	t15		13.94	0.07852	0.0001	5.9596	0.0001	1.76
714	Sept 8, 97	Sept 8, 97	t30		13.33	0.07522	-0.0032	-357.5757	-0.0057	1.69
707	Sept 8, 97	Sept 8, 97	t60		12.98	0.07332	-0.0051	-566.1615	-0.0091	1.64
711	Sept 8, 97	Sept 8, 97 Campus lawn	t0	18:00	14.19	0.07987				1.79
722	Sept 8, 97	Sept 8, 97	t15		14.03	0.07901	-0.0009	-95.3535	-0.0015	1.77
720	Sept 8, 97	Sept 8, 97	t30		13.41	0.07565	-0.0042	-464.8484	-0.0075	1.70
703	Sept 8, 97	Sept 8, 97	t60		13.3	0.07505	-0.0048	-530.4039	-0.0085	1.68
744	Sept 9, 97	Sept 9, 97 MT C-ring	t0	16:45	14.24	0.08015				1.80

730	Sept 9, 97	Sept 9, 97	t15		11.98	0.06791	-0.0122	-1346.8684	-0.0216	1.52
740	Sept 9, 97	Sept 9, 97	t30		10.14	0.05794	-0.0222	-2443.4338	-0.0392	1.30
737	Sept 9, 97	Sept 9, 97	t60		10.05	0.05745	-0.0227	-2497.0702	-0.0401	1.29
746	Sept 9, 97	Sept 9, 97 MT F-ring	t0		13.52	0.07625				1.71
745	Sept 9, 97	Sept 9, 97	t15		11.49	0.06525	-0.0110	-1209.7977	-0.0194	1.46
732	Sept 9, 97	Sept 9, 97	t30		9.81	0.05616	-0.0201	-2211.0096	-0.0355	1.26
738	Sept 9, 97	Sept 9, 97	t60		8.93	0.05139	-0.0249	-2735.4540	-0.0439	1.15
739	Sept 9, 97	Sept 9, 97 MT E-ring	t0		14.47	0.08139				1.82
736	Sept 9, 97	Sept 9, 97	t15		11.13	0.06330	-0.0181	-1990.5046	-0.0319	1.42
741	Sept 9, 97	Sept 9, 97	t30		9.67	0.05540	-0.0260	-2860.6054	-0.0459	1.24
734	Sept 9, 97	Sept 9, 97	t60		8.6	0.04960	-0.0318	-3498.2821	-0.0561	1.11
755	Sept 10, 97	Sept 10, 97 Garden uncut	t0	15:30	14.38	0.08090				1.81
762	Sept 10, 97	Sept 10, 97	t15		14.14	0.07960	-0.0013	-143.0303	-0.0023	1.78
760	Sept 10, 97	Sept 10, 97	t30		12.92	0.07300	-0.0079	-870.1008	-0.0140	1.64
754	Sept 10, 97	Sept 10, 97	t60		11.66	0.06617	-0.0147	-1621.0098	-0.0260	1.48
767	Sept 10, 97	Sept 10, 97 Garden A-ring	t0		13.47	0.07598				1.70
756	Sept 10, 97	Sept 10, 97	t15		13.93	0.07847	0.0025	274.1414	0.0044	1.76
752	Sept 10, 97	Sept 10, 97	t30		13.08	0.07386	-0.0021	-232.4242	-0.0037	1.66
764	Sept 10, 97	Sept 10, 97	t60		12.53	0.07088	-0.0051	-560.2019	-0.0090	1.59
763	Sept 10, 97	Sept 10, 97 Garden uncut	t0		13.93	0.07847				1.76
758	Sept 10, 97	Sept 10, 97	t15		12.26	0.06942	-0.0090	-995.2523	-0.0160	1.56
769	Sept 10, 97	Sept 10, 97	t30		12.74	0.07202	-0.0064	-709.1918	-0.0114	1.61
766	Sept 10, 97	Sept 10, 97	t60		11.11	0.06320	-0.0153	-1680.6057	-0.0270	1.42
785	Sept 11, 97	Sept 11, 97 MT C-ring	t0	15:45	12.49	0.07067				1.58
794	Sept 11, 97	Sept 11, 97	t15		11.7	0.06639	-0.0043	-470.8080	-0.0076	1.49
784	Sept 11, 97	Sept 11, 97	t30		11.89	0.06742	-0.0032	-357.5757	-0.0057	1.51
778	Sept 11, 97	Sept 11, 97	t60		11.46	0.06509	-0.0056	-613.8383	-0.0098	1.46
795	Sept 11, 97	Sept 11, 97 MT F-ring	t0		13.73	0.07738				1.73
796	Sept 11, 97	Sept 11, 97	t15		11.82	0.06704	-0.0103	-1138.2826	-0.0183	1.50
786	Sept 11, 97	Sept 11, 97	t30		10.78	0.06141	-0.0160	-1758.0804	-0.0282	1.38
788	Sept 11, 97	Sept 11, 97	t60		9.37	0.05377	-0.0236	-2598.3833	-0.0417	1.21

781	Sept 11, 97	Sept 11, 97 MT E-ring	t0		14.06	0.07917				1.77
790	Sept 11, 97	Sept 11, 97	t15		13.99	0.07879	-0.0004	-41.7172	-0.0007	1.77
782	Sept 11, 97	Sept 11, 97	t30		9.97	0.05702	-0.0221	-2437.4742	-0.0391	1.28
792	Sept 11, 97	Sept 11, 97	t60		10.57	0.06027	-0.0189	-2079.8985	-0.0334	1.35
818	Sept 13, 97	Sept 13, 97 MT C-ring	t0	14:30	14.66	0.08242				1.85
815	Sept 13, 97	Sept 13, 97	t10		11.86	0.06726				1.51
817	Sept 13, 97	Sept 13, 97	t20		11.82	0.06704	-0.0154	-1692.5249	-0.0271	1.50
805	Sept 13, 97	Sept 13, 97	t30		11.63	0.06601	-0.0164	-1805.7572	-0.0290	1.48
820	Sept 13, 97	Sept 13, 97 MT F-ring	t0		13.47	0.07598				1.70
807	Sept 13, 97	Sept 13, 97	t15		12	0.06801	-0.0080	-876.0604	-0.0141	1.52
802	Sept 13, 97	Sept 13, 97	t30		11.06	0.06292	-0.0131	-1436.2623	-0.0230	1.41
804	Sept 13, 97	Sept 13, 97	t60		10.83	0.06168	-0.0143	-1573.3330	-0.0252	1.38
814	Sept 13, 97	Sept 13, 97 MT E-ring	t0		14.08	0.07928				1.78
813	Sept 13, 97	Sept 13, 97	t15		12.1	0.06856	-0.0107	-1179.9997	-0.0189	1.54
811	Sept 13, 97	Sept 13, 97	t30		11.01	0.06265	-0.0166	-1829.5956	-0.0293	1.40
809	Sept 13, 97	Sept 13, 97	t60		11.05	0.06287	-0.0164	-1805.7572	-0.0290	1.41
808	Sept 13, 97	Sept 13, 97 CW G-ring	t0	16:00	13.37	0.07543				1.69
821	Sept 13, 97	Sept 13, 97	t15		12.64	0.07148	-0.0040	-435.0504	-0.0070	1.60
806	Sept 13, 97	Sept 13, 97	t30		11.39	0.06471	-0.0107	-1179.9997	-0.0189	1.45
810	Sept 13, 97	Sept 13, 97	t60		11.4	0.06477	-0.0107	-1174.0402	-0.0188	1.45
812	Sept 13, 97	Sept 13, 97 CW-litter	t0		15.38	0.08632				1.93
816	Sept 13, 97	Sept 13, 97	t15		12.53	0.07088	-0.0154	-1698.4845	-0.0272	1.59
803	Sept 13, 97	Sept 13, 97	t30		11.95	0.06774	-0.0186	-2044.1410	-0.0328	1.52
801	Sept 13, 97	Sept 13, 97	t60		11.5	0.06531	-0.0210	-2312.3227	-0.0371	1.46
832	Sept 15, 97	Sept 15, 97 MT F-ring	t0	12:45	14.27	0.08031				1.80
837	Sept 15, 97	Sept 15, 97	t15		12.25	0.06937	-0.0109	-1203.8381	-0.0193	1.55
833	Sept 15, 97	Sept 15, 97	t30		8.64	0.04982	-0.0305	-3355.2518	-0.0538	1.12
851	Sept 15, 97	Sept 15, 97	t60		8.75	0.05041	-0.0299	-3289.6963	-0.0528	1.13
838	Sept 15, 97	Sept 15, 97 MT E-ring	t0		14.35	0.08074				1.81
835	Sept 15, 97	Sept 15, 97	t15		11.9	0.06747	-0.0133	-1460.1007	-0.0234	1.51
843	Sept 15, 97	Sept 15, 97	t30		10.21	0.05832	-0.0224	-2467.2722	-0.0396	1.31

845	Sept 15, 97	Sept 15, 97	t60		7.62	0.04430	-0.0364	-4010.8072	-0.0643	0.99
846	Sept 15, 97	Sept 15, 97 CW G-ring	t0		14.01	0.07890				1.77
852	Sept 15, 97	Sept 15, 97	t15		13.38	0.07549	-0.0034	-375.4545	-0.0060	1.69
840	Sept 15, 97	Sept 15, 97	t30		12.76	0.07213	-0.0068	-744.9493	-0.0119	1.62
847	Sept 15, 97	Sept 15, 97	t60		11.28	0.06412	-0.0148	-1626.9693	-0.0261	1.44
844	Sept 15, 97	Sept 15, 97 CW-litter	t0		13.4	0.07560				1.69
848	Sept 15, 97	Sept 15, 97	t15		11.1	0.06314	-0.0125	-1370.7068	-0.0220	1.42
842	Sept 15, 97	Sept 15, 97	t30		9.61	0.05507	-0.0205	-2258.6864	-0.0362	1.23
841	Sept 15, 97	Sept 15, 97	t60		11	0.06260	-0.0130	-1430.3027	-0.0229	1.40
876	Sept 19, 97	Sept 19, 97 MT F-ring	t0	16:00	14.57	0.08193				1.84
871	Sept 19, 97	Sept 19, 97	t15		11.29	0.06417	-0.0178	-1954.7471	-0.0314	1.44
864	Sept 19, 97	Sept 19, 97	t30		9.18	0.05274	-0.0292	-3212.2215	-0.0515	1.18
881	Sept 19, 97	Sept 19, 97	t60		7.38	0.04300	-0.0389	-4284.9486	-0.0687	0.96
880	Sept 19, 97	Sept 19, 97 MT E-ring	t0		15.03	0.08442				1.89
877	Sept 19, 97	Sept 19, 97	t15		10.76	0.06130	-0.0231	-2544.7469	-0.0408	1.37
862	Sept 19, 97	Sept 19, 97	t30		9.14	0.05253	-0.0319	-3510.2013	-0.0563	1.18
869	Sept 19, 97	Sept 19, 97	t60		7.18	0.04191	-0.0425	-4678.2818	-0.0750	0.94
878	Sept 19, 97	Sept 19, 97 MT D-ring	t0		13.89	0.07825				1.75
867	Sept 19, 97	Sept 19, 97	t15		13.98	0.07874	0.0005	53.6364	0.0009	1.76
879	Sept 19, 97	Sept 19, 97	t30		12.98	0.07332	-0.0049	-542.3231	-0.0087	1.64
875	Sept 19, 97	Sept 19, 97	t60		11.97	0.06785	-0.0104	-1144.2422	-0.0184	1.52
891	Sept 25, 97	Sept 25, 97 MT F-ring	t0	13:50	15.5	0.08697				1.95
905	Sept 25, 97	Sept 25, 97	t15		11.17	0.06352	-0.0234	-2580.5045	-0.0414	1.42
892	Sept 25, 97	Sept 25, 97	t30		9.45	0.05421	-0.0328	-3605.5548	-0.0578	1.21
888	Sept 25, 97	Sept 25, 97	t60		7.82	0.04538	-0.0416	-4576.9687	-0.0734	1.02
897	Sept 25, 97	Sept 25, 97 MT D-ring	t0		15.72	0.08816				1.98
890	Sept 25, 97	Sept 25, 97	t15		16.11	0.09027	0.0021	232.4242	0.0037	2.02
898	Sept 25, 97	Sept 25, 97	t30		14.66	0.08242	-0.0057	-631.7170	-0.0101	1.85
894	Sept 25, 97	Sept 25, 97	t60		13.16	0.07430	-0.0139	-1525.6562	-0.0245	1.67
895	Sept 25, 97	Sept 25, 97 MT F-ring	t0	15:00	14.77	0.08302				1.86
900	Sept 25, 97	Sept 25, 97	t15		11.32	0.06433	-0.0187	-2056.0602	-0.0330	1.44

903	Sept 25, 97	Sept 25, 97		t30		11.36	0.06455	-0.0185	-2032.2218	-0.0326	1.45
886	Sept 25, 97	Sept 25, 97		t60		8.97	0.05161	-0.0314	-3456.5649	-0.0554	1.16
899	Sept 25, 97	Sept 25, 97	MT D-ring	t0		15.23	0.08551				1.92
906	Sept 25, 97	Sept 25, 97		t15		13.99	0.07879	-0.0067	-738.9897	-0.0119	1.77
896	Sept 25, 97	Sept 25, 97		t30		14.12	0.07950	-0.0060	-661.5150	-0.0106	1.78
901	Sept 25, 97	Sept 25, 97		t60		12.68	0.07170	-0.0138	-1519.6966	-0.0244	1.61
912	Oct 2, 97	Oct 2, 97	MT C-ring	t0	16:30	13.23	0.07468				1.67
923	Oct 2, 97	Oct 2, 97		t15		12.48	0.07061	-0.0041	-446.9696	-0.0072	1.58
921	Oct 2, 97	Oct 2, 97		t30		10.08	0.05762	-0.0171	-1877.2723	-0.0301	1.29
925	Oct 2, 97	Oct 2, 97		t60		9.83	0.05626	-0.0184	-2026.2622	-0.0325	1.26
929	Oct 2, 97	Oct 2, 97	MT F-ring	t0		14.69	0.08258				1.85
915	Oct 2, 97	Oct 2, 97		t15		10.8	0.06152	-0.0211	-2318.2823	-0.0372	1.38
913	Oct 2, 97	Oct 2, 97		t30		9.87	0.05648	-0.0261	-2872.5246	-0.0461	1.27
919	Oct 2, 97	Oct 2, 97		t60		7.6	0.04419	-0.0384	-4225.3526	-0.0678	0.99
924	Oct 2, 97	Oct 2, 97	MT D-ring	t0		13.85	0.07803				1.75
930	Oct 2, 97	Oct 2, 97		t15		13.21	0.07457	-0.0035	-381.4141	-0.0061	1.67
928	Oct 2, 97	Oct 2, 97		t30		13.22	0.07462	-0.0034	-375.4545	-0.0060	1.67
917	Oct 2, 97	Oct 2, 97		t60		12.19	0.06904	-0.0090	-989.2927	-0.0159	1.55
939	Oct 6, 97	Oct 6, 97	MT C-ring	t0		12.81	0.07240				1.62
954	Oct 6, 97	Oct 6, 97		t15		12.07	0.06839	-0.0040	-441.0100	-0.0071	1.53
952	Oct 6, 97	Oct 6, 97		t30		10.61	0.06049	-0.0119	-1311.1108	-0.0210	1.36
950	Oct 6, 97	Oct 6, 97		t60		7.91	0.04587	-0.0265	-2920.2014	-0.0468	1.03
940	Oct 6, 97	Oct 6, 97	MT F-ring	t0		15.68	0.08794				1.97
936	Oct 6, 97	Oct 6, 97		t15		9.46	0.05426	-0.0337	-3706.8679	-0.0595	1.22
946	Oct 6, 97	Oct 6, 97		t30		9.85	0.05637	-0.0316	-3474.4437	-0.0557	1.26
938	Oct 6, 97	Oct 6, 97		t60		7.56	0.04397	-0.0440	-4839.1909	-0.0776	0.99
957	Oct 6, 97	Oct 6, 97	MT D-ring	t0		15.8	0.08859				1.99
941	Oct 6, 97	Oct 6, 97		t15		14.42	0.08112	-0.0075	-822.4241	-0.0132	1.82
944	Oct 6, 97	Oct 6, 97		t30		13.22	0.07462	-0.0140	-1537.5754	-0.0247	1.67
956	Oct 6, 97	Oct 6, 97		t60		11.6	0.06585	-0.0227	-2503.0298	-0.0401	1.48
949	Oct 6, 97	Oct 6, 97	CW-litter	t0	17:00	14.62	0.08220				1.84

955	Oct 6, 97	Oct 6, 97		t15	14.55	0.08182	-0.0004	-41.7172	-0.0007	1.83
951	Oct 6, 97	Oct 6, 97		t30	13.4	0.07560	-0.0066	-727.0706	-0.0117	1.69
935	Oct 6, 97	Oct 6, 97	CW-moss	t0	16.54	0.09260				2.08
945	Oct 6, 97	Oct 6, 97		t15	11.27	0.06406	-0.0285	-3140.7064	-0.0504	1.44
953	Oct 6, 97	Oct 6, 97		t30	10.85	0.06179	-0.0308	-3391.0094	-0.0544	1.38
947	Oct 6, 97	Oct 6, 97		t60	10.33	0.05897	-0.0336	-3700.9083	-0.0594	1.32
			error term (4.4 L chamber)		2.14	0.01462		1608.8522	0.0258	0.33
			error term (6.4 L chamber)		2.14	0.01462		1899.9136	0.0305	0.33