

Drivers of Spatial Variation in Methane Emission and The Role of Vegetation in Methane Oxidation from Capped Landfills in Southern Ontario

by

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Author's Declaration

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

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Abstract

Landfills are a major source of methane (CH_4), accounting for 24% of Canada's total anthropogenic CH_4 emissions. This study investigates the environmental and meteorological factors driving temporal and spatial CH_4 flux variation and the role of vegetation in controlling CH_4 oxidation in landfill cover soils. This study was conducted across two capped landfill sections: the Original Landfill Area (OLA) (an older site without active gas collection system) and the Northern Expansion Area (NEA) (a relatively younger capped site with active gas collection) at a landfill in Southern Ontario. Both field data monitoring across the seasons and controlled incubation were carried out to critically assess CH_4 oxidation behavior in landfill cover soil by integrating vegetation treatments, environmental controls, and site specific controls. Field study revealed that the highest rate of CH_4 oxidation was observed with dry (<10%) and warmer (>25 °C) soil conditions between late summer and early fall, coinciding with reduced rainfall and higher air temperatures. CH_4 uptake was suppressed in wetter and colder conditions in winter and spring, correlating to cold and wetter soil conditions. Meteorological factors such as precipitation and air temperature significantly affected flux variations while wind speed and atmospheric pressure had a negligible impact. Presence of vegetation in landfill covers improved soil characteristics like bulk density, porosity, organic matter content, nitrate, phosphate, potassium and ammonium concentration, though it did not consistently alter the CH_4 fluxes. CH_4 fluxes were higher from the OLA's cover soils with slightly elevated fluxes in vegetated soil, while comparably lower fluxes were observed from NEA's cover soils, possibly due to its active gas collection system. Occasional CH_4 hotspots were observed, markedly at NEA even with active gas collection, suggesting localized structural failures in landfill cap or potential subsurface gas leaks. Quantified CH_4 oxidation rates on a per gram of cover soil basis indicated higher oxidation capacity in NEA, likely influenced by hotspots, providing favorable substrate supply to sustain efficient microbial CH_4 oxidation in the landfill cover.

The incubation results demonstrated that optimal CH_4 oxidation occurred at 20–40% soil moisture and 25 °C under vegetated and non-vegetated soil covers while CH_4 consumption markedly declined at cool and hot thermal conditions (5 °C and 35 °C), and in soil with higher moisture content (60–80%), reflecting microbial sensitivity to O_2 restrictions and thermal stress. There was

no significant vegetation effect, even under controlled incubation conditions, suggesting its influence may be indirect or more context specific (determined by soil structure and microclimate differences).

These findings emphasize that vegetation alone cannot stand as a mitigation strategy, but it formed favorable soil characteristics for CH₄ oxidation. A complex interplay between cap maturity, soil physical and chemical conditions, gas transport, CH₄ and O₂ availability, microbial community activity, and weather conditions determined CH₄ dynamics in capped landfill covers across seasons. Adaptive landfill covers should be designed for proper moisture control through drainage to control excessive water retention, incorporating soil amendments rich with organic matter to enhance soil properties for gas diffusivity and microbial habitat to maximize CH₄ oxidation capacity of cover soil. Thus, passive CH₄ mitigation approaches should be combined with engineered infrastructure facilities such as gas extraction systems for better CH₄ mitigation in capped landfill covers. Results from this study will also contribute to national greenhouse gas (GHG) inventories for reporting and accounting and to the development of control strategies or policies towards Canada's CH₄ emission reduction targets.

Keywords: CH₄ oxidation, Landfill cover soil, CH₄ flux, Methanotrophic activity, Soil moisture, Soil temperature, Gas diffusion, Mitigation strategy

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List of Abbreviations

CH ₄	Methane
CO ₂	Carbon Dioxide
O ₂	Oxygen
N ₂	Nitrogen
H ₂	Hydrogen
ANME	Anaerobic Methan Oxidizing Archaea
GHG	Greenhouse Gas
LFG	Landfill Gas
NH ₄ ⁺	Ammonium
NO ₃ ⁻	Nitrate
PO ₄ ³⁻	Phosphate
K ⁺	Potassium
EC	Electrical Conductivity
DOC	Dissolved Organic Carbon
NPOC	Non-Purgeable Organic Carbon
NDIR	Non-Dispersive Infrared
φ	Porosity
BD	Bulk Density
WFPS	Water-Filled Pore Space
LMMs	Linear Mixed Effect Models
UAVs	Unpiloted Aerial Vehicles
DIAL	Differential Absorption Lidar
NEA	Northern Expansion Area
OLA	Original Landfill Area
MSW	Municipal Solid Waste
Veg	Vegetated
Non-Veg	Non-Vegetated

Chapter 1

Literature Review

1.1 Introduction

Methane (CH_4) is responsible for 30% of the global rise in temperature and the reason for half a million premature deaths per year worldwide due to methane's indirect role in ground-level ozone formation (ECCC, 2021). The global warming potential of CH_4 is 28 times higher than CO_2 over 100-year timescales, and 86 times greater than CO_2 over 20 years (IPCC, 2013). Currently, atmospheric CH_4 levels consistently rise by 0.2-1% annually, emphasizing its impact on global warming. Solid waste in landfills undergoes complex physicochemical and biological interactions, leading to landfill gas (LFG) emissions (Chen et al., 2008). Landfill gas emissions contain CH_4 and therefore tend to be crucial for global warming (Christophersen et al., 2001). In fact, landfills contribute 10–19% of annual global CH_4 emissions (Kumar et al., 2004; USEPA, 2006), while about 24% of CH_4 emissions in Canada originate from landfills (Government of Canada, 2023). Landfill gas generation continues even after landfill closure, making CH_4 emissions a long-term issue (Bruce et al., 2018). CH_4 is produced in landfills through anaerobic decomposition of organic materials (Zhao et al., 2019). CH_4 oxidation in landfill cover soils plays a crucial role in reducing CH_4 emissions (Spokas & Bogner, 2011); improved understanding of controls on oxidation can help inform CH_4 mitigation strategies. This study investigates CH_4 oxidation rates from capped landfills under varying vegetation, temperature and moisture conditions. The findings will contribute to improving landfill cover designs and CH_4 mitigation strategies by providing insights into the key environmental drivers controlling microbial CH_4 oxidation.

1.2 Methane Production in Landfills

Landfilling is described as the dumping, compaction, and embankment filling of waste at suitable locations. In many situations, landfills are the only feasible method to dispose of all types of waste as it is easy to use, flexible, and less expensive than other waste disposal methods (Vaverková, 2019). One of the products generated by biological processes in municipal solid waste (MSW) landfills is biogas, which is produced because of the biological breakdown of organic materials (Shen et al., 2018). Organic materials in landfills comprise (usually) 50% cellulose, 15% lignin, 10% hemicellulose, and 5% protein, as well as starch, pectin, and other soluble carbohydrates

(Barlaz et al., 1992). Following the placement of MSW, easily broken-down organic substances in landfills undergo aerobic oxidation (Zhao et al., 2019). When the oxygen (O_2) supply is depleted, anaerobic digestion develops, and this is the primary bioreaction in landfills (Jardine et al., 2006). CH_4 and CO_2 , as well as a variety of trace gases including hydrogen sulfide, carbon monoxide, and nitrous oxide, are produced in significant quantities during the anaerobic degradation of organic materials (Bogner et al., 1999). Less CH_4 is created during the aerobic (with O_2) first stage of MSW breakdown in a landfill.

In general, anaerobic conditions emerge in less than a year, and CH_4 -producing microbes (Methanogenic Archaea) start to ferment waste degradation products and produce CH_4 (EPA, 2023). The production of LFG begins one to two years following the placement of waste in a landfill and lasts for 15–25 years (Thomassen et al., 2019).

1.3 Typical Landfill Gas Generation Phases

Landfill waste is decomposed by microbes in distinct phases as shown in Figure 1.1. In a landfill, waste may be going through several phases of the decomposition process simultaneously and gas composition varies with each of them. Phase I is aerobic, whereas Phases II - IV are anaerobic (Rovers and Farquhar, 1973).

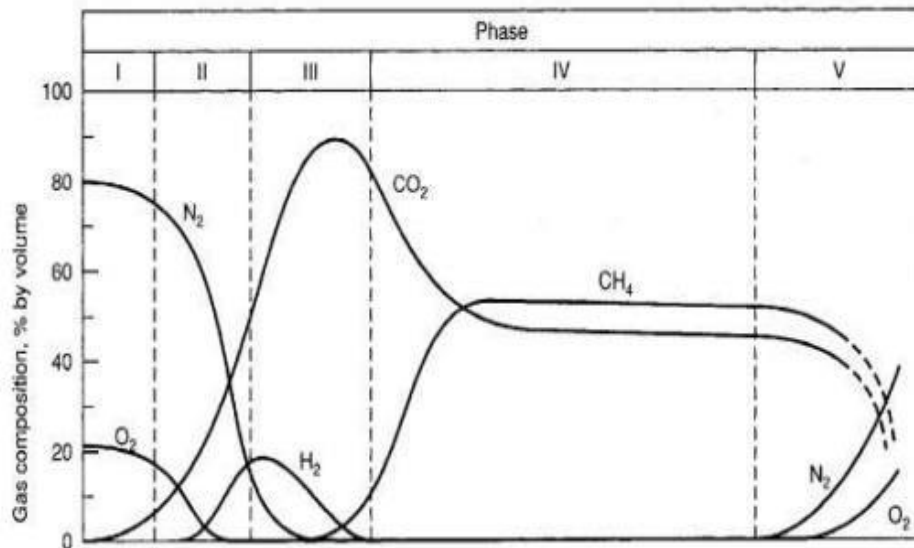


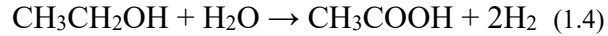
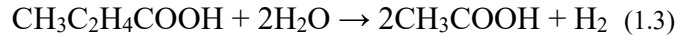
Figure 1.1: Generalized phases in the generation of landfill gases (Khalil et al., 2014).

In phase I (Aerobic Hydrolysis), aerobic bacteria utilize O_2 while breaking down the lengthy molecular chains of complex carbohydrates, proteins, and lipids that form organic waste during the initial phase of decomposition. CO_2 is the primary product formed by this process (ATSDR, 2001; Khalil et al., 2014). Although glucose is the primary outcome of this breakdown activity, secondary molecules such as sulfur and nitrogen are produced as an outcome of the oxidation of other components. During this period, the massive amount of CO_2 produced causes a depletion of O_2 and nitrogen (N_2). The creation of huge amounts of CO_2 causes a reduction of O_2 via microbial respiration and N_2 proportionally reduced because of displacement. It is accompanied by a substantial drop in redox potential. The rapid rise in landfill temperature reflects the massive energy release. Phase I degradation may occur over a few days or months (Zhao et al., 2019), depending on the amount of O_2 in the landfill at the time that waste is disposed of. The amount of O_2 present in waste depends on how loose or compacted it was once buried and the nature of the waste (Rovers and Farquhar, 1973; ATSDR, 2001; Khalil et al., 2014).

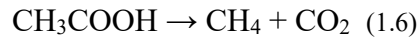
Phase II (Acidogenesis) is recognized as the transitional period, during which anaerobic conditions start to arise, and O_2 is reduced. Anaerobic bacteria transform substances produced by aerobic bacteria into acetic, lactic, and formic acids as well as alcohols like methanol and ethanol (reactions 1.1 and 1.2). The waste site turns highly alkaline. Hydrogen (H_2) and CO_2 are the gaseous byproducts produced in phase II reactions (Rovers and Farquhar, 1973; ATSDR, 2001; Khalil et al., 2014; Zhao et al., 2019).



In phase III (Acetogenesis), acetogenic microorganisms ferment organic acids, eventually reducing those into acetate (reactions 1.3 and 1.4). CH_4 producing organisms can establish themselves in the landfill because of this process as this makes the ecosystem more neutral (Aik Heng Lee et al., 2010). CH_4 and acid-producing microbes have a mutually beneficial interaction (Rovers and Farquhar, 1973; ATSDR, 2001; Khalil et al., 2014; Zhao et al., 2019).



During phase IV (Methanogenesis), methanogenic archaea produce CH_4 by either reducing CO_2 with H_2 or breaking down acids (CH_3COOH) to produce CH_4 and CO_2 as shown by the following reactions (1.5) and (1.6) (Zhao et al., 2019).



The production rates and composition of landfill gas remain largely unaltered. Usually, phase IV is reached in less than a year. In this stage, gas production stabilizes and lasts for around 20 years. Nonetheless, gas emissions persist for at least 50 years following the disposal of waste in a landfill (ATSDR, 2001). In this phase, landfill gas typically has a CH_4 content of 45% to 60% by volume, 40% to 60% CO_2 , and 2% to 9 % of other gases, such as sulfides (ATSDR, 2001; Khalil et al., 2014; Crawford and Smith, 2016; Zhao et al., 2019).

Phase V (Restoration or Maturation Phase) allows air to diffuse back to the landfills once accessible organic materials decompose to CO_2 and CH_4 . The precise quantity of bioconversion that takes place is determined by several factors, including the distribution and movement of moisture. Moisture is needed to create a biofilm over the waste matrix, which supports the growth of bacteria by hydrolysis of substances into substrates. Additionally, it permits nutrients to be distributed, chemical reactions to occur, and waste to be removed. A second aerobic phase may emerge in the top layer of very old landfills when only a greater amount of refractory organic carbon content is present in the landfilled wastes. During this phase, air continues to diffuse from the atmosphere, enabling the formation of aerobic zones and a redox potential that is too high to generate CH_4 . Therefore, less CH_4 is produced during this stage (Khalil et al., 2014).

Throughout the existence of a landfill, the content and rate of biogas generation changes. The combined model of microbial waste decomposition and the model of landfill gas's long-term

behavior from dated waste repositories determine how landfill gases are produced (Zhao et al., 2017).

1.4 Environmental Drivers of CH₄ Production

Machado et al. (2009) suggest that the most important factors influencing CH₄ production in landfills are waste composition (overall mixture of buried waste including both organic and inorganic waste), temperature, moisture content, residue age, and the presence of rapidly degrading organic material (labile carbon sources such as food waste, sugary residue). Leachate recycling, sludge addition, moisture content, and particle size are among the variables that have been researched to increase CH₄ generation in the study of CH₄ production from municipal refuse conducted by Barlaz et al. (1992). Zhao et al. (2019) found that waste generation, waste composition, waste management, and environmental variables were the four primary groups of factors that influenced LFG generation and emissions. The physical and chemical properties of landfill cover materials, moisture content, bacterial composition in landfills, seasonal variation in CH₄ oxidation rates, amount of organic material deposited in landfills, the thickness of landfill cover, and the percentage of anaerobic decomposition of organic matter in MSW all contribute significantly to the generation of landfill gas.

Organic Content

ATSDR (2001) report that the amount of landfill gas that bacteria produce during decomposition (CO₂, CH₄, nitrogen, and hydrogen sulfide) increases with the amount of organic waste in the landfill. It was also demonstrated that the age of the waste impacted CH₄ production. Waste buried for less than 10 years generated more landfill gas than waste older than 10 years due to volatilization, chemical reactions, and bacterial degradation. The highest level of gas production usually happens five to seven years after the garbage is landfilled.

Moisture Content

CH₄ production increases with higher moisture content up to a particular level (Pohland, 1982). In a landfill, moisture (in unsaturated conditions) promotes bacterial breakdown, which promotes gas production. Also, gas-producing chemical processes may be facilitated by moisture (ATSDR, 2001). Moisture creates aqueous conditions that facilitate nutrients and bacterial transfer inside the

waste. This transport mechanism enables microorganisms' access to their substrates, allowing gas generation and stabilization (El-Fadel and Khoury, 2000).

Temperature

Heat is found to be the main by-product of the MSW disposal process, and the biodegradation of MSW rises with temperature up to a certain value (between 34 and 41 °C) (Mata-Alvarez and Martinez-Viturtia, 1986; Yeşiller et al., 2005). Bacterial activity increases with rising landfill temperature, leading to higher gas production. The rates of chemical reactions and volatilization may also increase with temperature (ATSDR, 2001). The optimal temperature range for gas production in waste is between 40 and 45 °C (Rees, 1980).

pH

Since methanogenic archaea are active in a pH range of 6.8 and 7.4 (Gurijala and Sulflita, 1993), the MSW pH level significantly impacts biological degradation. Without methanogenic archaea, the ecosystem of the landfill would be unable to convert waste to gas and liquid (Rovers, 1973).

Oxygen Availability

CH₄ will not be built up in the landfill until all O₂ has been depleted. CH₄ producers are negatively impacted by O₂, even at very low O₂ concentrations (ElFadel and Khoury, 2000).

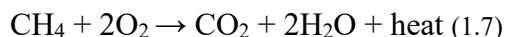
1.5 CH₄ Oxidation Mechanisms in a Landfill

Methanotrophic microorganisms gain energy by oxidizing CH₄ in oxic and anoxic environments using a variety of electron acceptors (Guerrero-Cruz et al., 2021). In 1906, it was initially discovered that methanotrophy is an oxygen-dependent biological mechanism within Proteobacteria (Whittenbury et al., 1970; Hanson & Hanson, 1996). However, recent research findings have uncovered microbes outside Proteobacteria and even within the archaeal domain that can anaerobically oxidize CH₄ using alternative electron acceptors such as sulfate, nitrite, nitrate, iron and manganese (Guerrero-Cruz et al., 2021).

Methanotrophic bacteria, a physiological type of bacteria known as methanotrophs that consume CH₄ as their carbon source, are primarily responsible for CH₄ oxidation in landfill-cover soil

(Scheutz et al., 2004). CH₄ is converted to CO₂ by the metabolism of methanotrophs in landfill-cover soils when it seeps from the landfill core and passes through the aerated cover soil (Hanson and Hanson, 1996). Engineered capping systems in landfill covers are primarily designed to act as a barrier against precipitation infiltration, limiting leachate production and regulating or controlling LFG production and emission into the atmosphere (Simon et al., 2004). A higher proportion of CH₄ is transported to the soil cover in the landfills without gas collection systems, often older and smaller sites. CH₄ oxidizing bacteria may then partially oxidize CH₄ (Stern et al., 2007).

Sadasivam and Reddy (2014) mentioned that proteobacterial methanotrophs are usually divided into two main categories, Type I and Type II, both of which are found in landfill soil. This classification also considers their physiological and taxonomic characteristics (Hanson and Hanson, 1996). Type I methanotrophs rely on the ribulose monophosphate pathway for carbon assimilation (Auman et al., 2001), prefer alkaline pH range and are adapted to low CH₄ levels (Hanson & Hanson, 1996). On the other hand, Type II methanotrophs utilize the serine pathway for carbon assimilation, are found in neutral pH environments and inhabit areas with higher CH₄ concentrations (Hanson and Hanson, 1996). Additionally, Anaerobic Methane Oxidizing Archaea (ANME) oxidize CH₄ under anoxic conditions via a reverse methanogenesis pathway (Cassarini, 2019). Overall, methanotrophic bacteria can oxidize CH₄ to microbial biomass, CO₂, and water (H₂O) in the presence of natural conditions (Abushammala et al., 2014). As discussed in Pawlowska and Stepniewski (2006) and Chiemchaisri et al. (2012), the chemical kinetics related to CH₄ oxidation are shown in the below equation (1.7).



1.6 Factors Influencing CH₄ Oxidation in Landfills

The oxidation of CH₄ in landfill cover soil is influenced by several factors, including temperature, air pressure, air-filled pore spaces, nutrients, pH, water content, and concentrations of CH₄ and O₂. The three most important factors are generally water content, O₂, and ammonium content (He et al., 2011). Börjesson et al. (2001) and Wilshusen et al. (2004) suggested that the soil texture,

organic content, moisture content, temperature, pH, nutrients, and O₂ and CH₄ concentrations are the key environmental variables regulating the CH₄ oxidation process in landfill cover soils.

Presence of Oxygen

Oxygen is one of the key reactants and limiting factors regulating the CH₄ oxidation process in landfill cover soils (Berger et al., 2005). The O₂ content fluctuates with soil depth and is controlled by a variety of factors, such as gas composition, meteorological conditions, the degree of microbial CH₄ oxidation, soil texture, and cover thickness (Abushammala et al., 2014). According to Hrad et al. (2012), the CH₄/O₂ mixing ratio is essential for regulating the oxidation activity of the soil microbial population, especially methanotrophs. At the point of maximum CH₄ oxidation, the gradients of CH₄ and O₂ levels in a soil profile overlap, and this depth is the optimal depth for maximum CH₄ oxidation (Abushammala et al., 2014). Aerobic CH₄ oxidation is observed primarily in the methanotrophic active zone, which is in the top 10-20 cm of the soil profile and decreases with depth (Pawlowska and Stepniewski 2006; He et al., 2011), while Jones and Nedwell (1993) found that the highest CH₄ oxidation occurred at depths ranging from 10 to 30 cm.

Soil Moisture Content

Soil moisture promotes microbial activity in landfill covers. There are different sources of water in landfill soil cover, notably surface water penetration, precipitation, water from manmade sources (leachate recirculation), and the decomposition process inside the soil cover (Abushammala et al., 2014). To maintain a balanced environment in the cover soil for CH₄ oxidation, an ideal soil moisture content of 10-20% w/w is required (Spokas and Bogner, 2011) since the oxidation capacity decreases considerably when moisture content falls below 5%. Tecle et al. (2008) also found a similar observation that the ideal moisture content to maximize CH₄ oxidation activities is 11-25% by volume (v/v). Wang et al. (2025) observed a moisture threshold of 6.2% v/v for in situ CH₄ oxidation in the alpine grasslands due to coarse texture of soil in these drylands that improved gas diffusion and promoted CH₄ uptake (Jørgensen et al., 2024). Lower soil moisture levels reduce CH₄ oxidation capacity by reducing biological activity in the soil cover (Tecle et al., 2008). As mentioned by Scheutz et al. (2009a, 2009b), high moisture level also prevents O₂ from penetrating through soil layers and reducing empty pore space in the soil for gas transport and diffusion.

Soil Temperature

According to Borjesson et al. (2004) and Spokas and Bogner (2011), the majority of methanotrophs grow at temperatures between 25 and 35 °C, whereas Type I methanotrophs can oxidize CH₄ at lower temperatures between 2 and 10 °C. The optimum operating temperature of methanotrophs is between 25 and 35 °C. As reported by Scheutz and Kjeldsen (2004), increasing soil temperature from 2 to 25 °C enhanced CH₄ oxidation dramatically; at 30 °C, the highest rate of CH₄ oxidation occurred, and at 40 °C, the rate of oxidation began to fall.

pH

The pH of a landfill soil alters CH₄ oxidation activities (Hutsch et al., 1994). In testing CH₄ oxidation capability in different types of soils with pH values varying from 3 to 7.5, Saari et al. (2004) identified that the optimal pH for CH₄ oxidation ranged from 4 to 7.5.

Inorganic N

The amount of inorganic N in the landfill cover soil impacts its capacity to oxidize CH₄. Methanotrophs have a high N-demand, but high levels of NH₄⁺ competitively inhibit CH₄ oxidation since ammonia is structurally similar to CH₄ and is a known inhibitor of the CH₄ monooxygenase enzyme (Borjesson et al., 2004; He et al. 2011).

Soil Texture

The texture of the soil influences LFG transport and atmospheric O₂ penetration. It thereby regulates both CH₄ release and oxidation rates. According to Gebert and Grongroft (2009), coarse-textured soils with more than 17% air-filled pores by volume, such as sands, loamy sands, and some coarsely textured loams, are suitable for CH₄ oxidizing biocovers. The rate at which CH₄ is transferred across a water medium is 104 times slower than that of an air medium (Dubey, 2005). A soil cover with a high proportion of pores larger than 50 µm is preferred to enhance the transfer of CH₄ through cover soils under wet circumstances (Scheutz et al., 2009a;2009b).

1.7 Vegetation Effects on CH₄ Dynamics

Tree planting in closed landfill sites is a generally applied technique that aims to improve aesthetics, reduce water percolation, and improve carbon sequestration (EPA, 2003; Venkatraman

and Ashwath, 2009). Typically, native plants spread over the landfill with the landfill closure; vegetation coverage rates may rise to 60-70% within four years (Xiaoli et al., 2010). The vegetation characteristics of closed landfills are still mostly unknown. Enhancing comprehension of the correlation between plant conditions, cover soil properties, and LFG emissions would be beneficial for landfill management and for preventing the release of harmful GHGs (Xiaoli et al., 2010). Cover soil and the vegetation cover are major factors in determining soil-mediated CH₄ oxidation, and it can affect methanotroph activity and development in several ways (Reay et al., 2005; Wang et al., 2008; Berger et al., 2005; Scheutz et al., 2004). With the stabilization of a closed landfill, an increase in the vegetation cover, vegetation heights, and plant species growth was observed. The environment became more favorable for the growth of vegetation due to the lower release of LFG in the closed landfill. Vegetation growth could potentially be limited by high levels of LFG while only tolerant plants could survive (Xiaoli et al., 2010).

Plants showed dual roles regarding CH₄ dynamics. Plants can improve CH₄ oxidation by providing O₂ to the soil. As explained by Xiaoli et al. (2010), the computed CO₂ to CH₄ concentration ratios indicated that the vegetated cover soil had a higher CO₂ to CH₄ ratio than the bare cover soil. An explanation is that plant roots can facilitate the diffusion of O₂ into the soil cover and elevate the levels of soil O₂, establishing an ideal habitat for bacteria that oxidize CH₄ (Abichou et al., 2006). Plants may also emit CH₄ (Carmichael et al., 2014). Plants function as gas conduits because of plant parenchymatous structures that boost the release of CH₄ (Xin et al., 2016; Raven, 2022). Trees, especially in wetland and upland environments, may provide potential pathways for CH₄ to move to the atmosphere from subsurface sources (Maier et al., 2018; Pangala et al., 2013). When tree roots route CH₄ from below ground in landfills, the portion of CH₄ oxidized is reduced which leads to higher CH₄ emissions (Fraser-McDonald et al., 2022). In addition, CH₄ oxidation may be negatively affected by the competition for O₂ of root respiration of plants (Wang et al., 2008). This emphasizes the significance of understanding the impact of certain plant species planted in landfills.

Plant organic matter, such as root exudates and root litter may act as a carbon substrate for methanogens in wetland systems (Chanton et al., 1993; Whiting & Chanton, 1993; Bellisario et al., 1999; Saarinen, 1996). Instead of plants in general, a significant correlation between CH₄

emissions and sedge cover has been observed because the large root surface and air filled aerenchyma structure of sedges create an effective gas collection system with a rapid gas diffusion path (Whiting & Chanton, 1993). Plant mediated gas transport mechanisms are varied by species due to differences in carbon distribution and physiological transport capacity within plants (Schimel 1995; Ström et al., 2003). Given that cover soils in landfills are generally unsaturated and that LFG is produced below the cap, these wetland findings may not apply unless sedge roots penetrate the landfill cap.

Cugler et al. (2024) reported that grasses in drier areas exhibited higher CH₄ emissions than those from tree stems, while grasslands emit 100-fold higher CH₄ emissions than tree stems. The well-developed root network of grasses fuel CH₄ production by supplying carbon substrates for methanogens in anaerobic soil (Fry et al., 2017). Moreover, root systems of grasses extend up to 30-50 cm providing effective gas transport channels at shallower depths compared to trees whose roots penetrate several meters deep into the soil (Fender et al., 2013; Xiong et al., 2024). Dense turf layers of grasses hinder O₂ penetration into the soil and deplete existing O₂ through root respiration at the same time, thereby forming anaerobic soil pockets that promotes methanogenesis (van Delden et al., 2018). Clay-rich grasslands hold higher moisture contents, even during dry periods, resulting in wetter and lower soil O₂ levels beneath the thick grass covers with ideal micro-environment for CH₄ production and release (Fry et al., 2017; Kleine et al., 2020). Whether these conditions can also arise in landfill cover soils requires further investigation.

1.8 Seasonal Variation in CH₄ Emissions

CH₄ emissions also vary seasonally (Börjesson and Svensson, 1997). Monitoring over different seasons provides more accurate yearly landfill gas emission estimates. In winter, CH₄ emissions may increase due to the inhibition of CH₄ oxidation activity with the low temperatures and soil moisture content. Late summer CH₄ oxidation activities are found to be lower due to a combination of summer dryness (warm summer temperatures) and heat from CH₄ oxidation (Maurice and Lagerkvist, 2003). As studied by Maurice and Lagerkvist (2003), snow cover functions as an insulator, limiting ice reaching into the deep soil. In their test cell site, the impact on temperature was visible down to around 3-m depth and the waste temperature only fluctuated by a few degrees, and the LFG generation is anticipated to remain relatively stable. The fact that CH₄ emissions

occurred exclusively in late winter when the ground was frozen, shows that CH₄-oxidizing organisms are suppressed over this time.

Börjesson and Svensson (1997) and Christophersen et al. (2001) discovered similar seasonal trends of CH₄ and CO₂ emissions, reporting higher CH₄ emissions in winter and higher CO₂ emissions in summer, suggesting soil temperatures are the key controlling factor. Moreover, soil moisture content and pressure gradients were recognized as the critical variables influencing CH₄ emissions, particularly temperature-dependent CH₄ oxidation processes (Rachor et al., 2013). The opposite seasonal CH₄ pattern was observed by Jones and Nedwell (1993), with higher CH₄ emissions in summer influenced by soil moisture content. Their finding demonstrated that an increase in soil moisture content in winter lowered gas permeability resulting in lower CH₄ emissions. Higher CH₄ emissions in winter (moisture and cooler circumstances) and lower emissions in summer (drier and warmer circumstances) were discovered during seasonal campaigns (Rachor et al., 2013). These observations underline the functions of temperature and moisture in regulating CH₄ emission through seasonal changes. CH₄ emissions in cold regions are further complicated by long winter seasons and snow cover, making monitoring and quantification more challenging. Yet, monitoring emissions throughout the year is needed to obtain accurate yearly emission estimations (Maurice and Lagerkvist 2003). Under regions with diverse seasons, the rate of methanotrophic oxidation in landfill soil cover varies greatly, emphasizing the need for multiple measurements to account for this variability. Overall, cooler and dryer climates are predicted to exhibit reduced CH₄ oxidation (Klusman and Dick, 2000).

1.9 Methane Emission Monitoring Techniques

Mønster et al. (2019) described that techniques for measuring landfill emissions may be categorized into two groups: surface emission factor methods and mass emission methods. Surface emission factor method is based on measuring the emissions coming from a portion of the landfill surface and calculating an emission factor that is then applied to the remaining sections of the landfill (NASEM, 2018). Conversely, mass emission approaches entail measuring the emissions from the landfill to a greater region or the entire landfill site (Mønster et al., 2019). Quantitative methodologies deploy field monitoring of CH₄ emissions and qualitative reconnaissance

techniques are used to identify landfill "hotspots" or elevated surface emission trends (Mønster et al., 2019).

1.9.1 Qualitative Reconnaissance Techniques for Detecting “Hotspots” and Elevated Surface Emissions

A portable CH₄ analyzer is used in this technique to detect hotspots by systematic or random walkover surveys (EA 2010; Wilkinson et al., 2018; Mønster et al., 2019). This method can produce either qualitative or semi-quantitative readings of surface CH₄ concentrations directly above the landfill (Mønster et al., 2019). The advantage of this technique is that it is quick and easy to use when deployed in the field, resulting in a comparatively inexpensive option (Mønster et al., 2019). However, the correlations between surface fluxes of CH₄ and measured concentrations are not always consistent. This occurs because site topography, soil, climatic conditions and conflicting sources of CH₄ can all impact the above surface CH₄ concentration (Mønster et al., 2019).

CH₄ concentrations can also be monitored higher above the surface using unpiloted aerial vehicle (UAV) surveys. Because they can fly at lower speeds while making takeoffs and landings simpler, rotary UAVs are frequently chosen (Shaw et al., 2021). Positive aspects of this method are survey flights are substantially faster than walkover surveys because they may be pre-programmed. The safety rules governing UAV usage at landfills, battery life requirements for rotary UAVs, and meteorological conditions (wind and rain) all place limitations on the use of this method (Mønster et al., 2019; Shaw et al., 2021).

Field infrared surveys detect locations or regions with high temperatures using thermal infrared technology. Depending on the scale and necessary sensitivity, sensors are either deployed on UAVs or aircraft or onsite (inspection using portable devices) (Mønster et al., 2019). Lewis et al. (2003) mentioned that the advantages of this method include quick and easy field implementation, including using UAVs for aerial measurements, comparatively cheap (depending on the scope and complication of the field measurement project), and dealing for locating more extensive localized gas leakage. The disadvantage of this method is that mechanisms other than the release of warmed LFG might be the cause of thermal anomalies. Further, thermal anomalies caused by emission hotspots might not be noticeable in warm areas (Fjelsted et al., 2019; Tanda et al., 2017).

Both an auditory and visual examination of the landfill cover is called the visual inspection method. The smell of trace elements in LFG (Hitchin,1998), alterations in vegetation, and anomalies in the cover (cracks and fissures) (Scheutz et al., 2011b) can all be used to identify emission hotspots (Mønster et al., 2019). Strengths of this technique is that it is a quick and easy first method for standard inspection processes, whereas constraints are that observations can arise from a wide range of potential effects, such as landfill cover maintenance, site topography, soil moisture, and wind direction and speed (Hitchin,1998; Mønster et al., 2019).

1.9.2 Quantitative Techniques for the Field Measurement of CH₄ Landfill Emissions

Flux chamber methods (i.e., closed chambers) have been the most used technique for measuring surface CH₄ emissions from landfills (Mackie and Cooper, 2009; Mønster et al., 2019). In closed surface flux chambers, observations are made at a single location by combining surface-mounted chambers with time-series analysis of changing gas concentration within the chamber (Mønster et al., 2019). The rate at which concentration changes may be used to compute the emission flux. The closed flux chamber has been commonly used to measure gas emissions from landfill surfaces, where emissions are impacted by pressure as well as concentration (Lucernoni et al., 2016; Rachor et al., 2013; Abichou et al., 2011). The method detects CH₄ fluxes, which are typically positive (CH₄ emission) but can also be negative (absorption of atmospheric CH₄, indicating oxidation). The flux chamber technique is a well-established and suitable approach for assessing CH₄ oxidation at a specific spot on the surface or analyzing environmental variables influencing CH₄ oxidation.

Simple, direct measurements and minimal surface restrictions on application are benefits of flux chamber methods (Oonk, 2010). Due to the ability to assess spatially mixed and homogenized gas concentrations in the air, such methods provide benefits in addressing the diversity of surface CH₄ releases (Jeong et al., 2019). The chamber method is appropriate for emission process investigation (Mønster et al., 2019). A disadvantage of closed chamber method is that to cover the whole waste surface, measurement takes a significant amount of effort and time. Because it measures from a small number of unique places, it requires an acceptable scale-up of recorded fluxes to estimate emissions from the entire landfill. Since this method of estimation does not accurately capture the

fluctuation in surface emissions, it typically results in a considerable bias in total estimation (Borjesson et al., 2001; Spokas et al., 2003).

The turbulent transport of gases between the ground surface and the atmosphere is measured through the eddy covariance technique (Mønster et al., 2019). The advantage of this method is that it provides a comprehensive measure of fluxes at the landscape scale (Zhu et al., 2023). This method is suitable for measuring temporal emission variabilities and runs automatically for extended periods (Mønster et al., 2019; Baldocchi, 2020). Further, it allows for long-term and temporary research under changing atmospheric scenarios where emission routes are not affected (Fowler et al., 1995, Laurila et al., 2005; Xu et al., 2014). However, eddy covariance is difficult to implement in changeable terrain, with competing CH₄ sources (Mønster et al., 2019) and unsteady environmental conditions (Baldocchi, 2003).

Mass balance using aerial measurements is a method in which an aircraft is used to measure the quantities of CH₄ in the downwind plume from the landfill (Gasbarra et al., 2019; Lunt et al., 2025). The horizontal flow of CH₄ through the 2D plane can be computed (Peischl et al., 2013; Cambaliza et al., 2017; Allen et al., 2019), providing an integrated emission from the entire landfill without requiring access, the major advantage of this method. Limitations of this method are that sources that are close together might be challenging to differentiate, covering the whole plume is required, and precise weather information and highly precise aircraft equipment are required for implementation (Mønster et al., 2019).

The Differential Absorption Lidar (DIAL) method monitors the backscattered laser portion after a laser pulse is emitted into the atmosphere (Collis and Russell, 2005). In landfills, DIAL measurements are suitable for locating CH₄ absorption lines (Mønster et al., 2019). This method provides a modeled 3D downwind concentration plane linked with wind direction and wind speed data (Innocenti et al., 2017). This technique provides emissions for both specific areas and the entire site, it has minimal disruption from additional surrounding CH₄ sources and, it can provide comprehensive details on emission trends (Innocenti et al., 2023). The drawbacks are the massive and costly analytical setup of DIAL and the intricate data processing required for the procedure (Babilotte et al., 2010; Robinson et al., 2011; Bourn et al., 2019).

1.10 Mitigation Strategies

Landfill CH₄ emissions have been mitigated in two main ways. The first strategy recovers or burns LFG using gas collecting systems, whereas the second strategy relies on various strategies, such as waste recycling, incineration, and composting to reduce emissions (Abushammala et al., 2014). Apart from that, microbial CH₄ oxidation in bio mitigation systems (such as bio-covers and bio-windows) has been also found as an efficient passive way to control CH₄ emissions (Chavan et al., 2018) even along with active landfill gas extraction facilities (Scheutz et al., 2023).

1.10.1 Gas Collection and Recovery Systems

The present standard LFG management systems employ a landfill gas collection system, a landfill cover system, or a combination of each of them. Many engineered landfill sites have been designed with gas extraction systems that consist of vertical wells and horizontal collectors (Barlaz et al., 2009). Following that, the CH₄ gas is extracted and either flared or used to recover energy. Applying negative pressure along the pipe's collection point is the concept underlying LFG capture by extraction process (Park and Shin 2001; Barlaz et al., 2009). In landfills that are still actively operating, gas collecting systems are more beneficial as waste decomposition produces a significant volume of CH₄ gas. This is because high CH₄ content reduces the cost of gas flaring or energy recovery (Chiemchaisri et al., 2012). Abushammala et al. (2014) highlighted that the gas collection is more affordable for larger sanitary landfills, where it is more often used. It is thought to be excessively expensive and impractical for smaller, older landfills with substantially lower CH₄ emission rates. Huber-Humer et al. (2008) mentioned that gas emissions may also leak partially from and around wells, and they may migrate along the paths of installed landfill equipment, making collecting systems inefficient; also, collecting systems may function for less than half the time that LFG is generated, because they are only economically viable when the level of CH₄ is significant and when CH₄ concentration is below 35–40% and with 30 - 50 m³ h⁻¹ gas production rates (Haubrichs & Widmann 2006). Since minimal gas production might last for decades or even a century, the net accumulation of CH₄ during non-collection periods has a significant impact on global warming (Huber-Humer et al., 2008). Figure 1.2 displays the generation and recovery of CH₄ during the lifetime of a landfill. CH₄ emissions (CH₄ not caught) are depicted as a function of cover type and do not account for CH₄ oxidation removal.

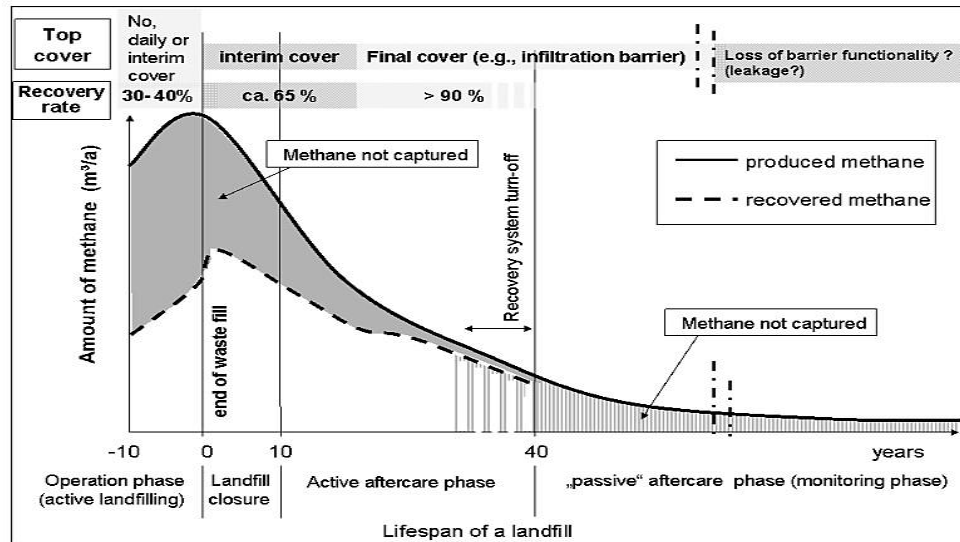


Figure 1.1: CH₄ Generation and Recovery Based on Time over the Lifespan OLA Landfill (Spokas et al., 2006; Huber-Humer et al., 2008).

1.10.2 Bio-Based Systems

Microbial CH₄ oxidation in landfill cover soil can also be an efficient way to control CH₄ emissions (Abashamala et al., 2013a, Huber-Humer et al., 2008, Stern et al., 2007). Conventionally, landfills are covered using soil cover systems to reduce leachate formation and landfill gas emissions. As LFGs pass through the cover soil it promotes the microbial oxidation of CH₄ (Sadasivam and Reddy, 2014). Recent innovative efforts aimed at further reducing CH₄ emissions contributed to a development in the usage of bio-based covers as a substitute for soil coverings worldwide. A biocover is a specific system designed to promote CH₄ oxidation and uniformly distribute the LFG across the area to accomplish efficient oxidation (Stern et al., 2007; Huber-Humer et al., 2008; 2009; Park et al., 2010). The oxidation layer is typically made up of organically rich materials such as compost, sewage sludge, peat, etc. (Scheutz et al., 2009a). These materials can be added to the cover soil as amendments or used alone to continuously encourage and promote the development of methanotrophs by creating favorable conditions for improved CH₄ oxidation (Sadasivam and Reddy, 2014). According to Huber-Humer et al. (2008), researchers have so far suggested four different types of bio-based systems to reduce LFG namely, biocovers, biofilters, biowindows, and biotarps.

A biotarp is a daily cover or a short-term remedy for managing CH₄ emissions throughout the active life of the landfill cell, whereas biocover, biowindow, and biofilter systems are final coverings that minimize CH₄ emissions from a landfill. Biotarps are typically at least 15 cm thick, detachable geotextile structures that have been impregnated with methanotrophs (Sadasivam and Reddy, 2014). A biotarp reduces the amount of early-stage CH₄ emissions from freshly placed garbage and serves as a daily cover or short-term fix for CH₄ releases that occur during the active part of a landfill (Huber-Humer et al., 2008).

Huber-Humer et al. (2009) describe a landfill biocover as a top cover designed to improve the methanotroph environment and biotic CH₄ consumption. A biocover needs the selected organic material, resembling soil cover, to cover the whole landfill (Sadasivam and Reddy, 2014). In biofilter applications, for biofilters to function correctly, an engineered biotic system must be installed in pre-selected landfill areas that have been excavated to a suitable depth (Abushammala et al., 2014). Since filter material can sorb contaminated gas and maintain bacterial cultures, it is regarded as the most crucial component of a bio-filter system. Peat, compost made from bio-waste, heather, shredded bark, and sawdust are examples of bio-filter materials that are mainly biological in source (Huber-Humer et al., 2008). Biowindows are another method for reducing CH₄ emissions from landfills into the atmosphere. To maximize the effectiveness of CH₄ oxidation by increased microbial activity by CH₄ oxidation bacteria, composted material can be used as biowindow media (Huber-Humer et al., 2008). This approach works effectively when there are no gas collection methods at the landfill (Sadasivam and Reddy, 2014).

1.11 Research Problem and Research Questions

Determining and measuring the worldwide sources and sinks of CH₄ is essential for the creation of efficient climate change mitigation plans (Kirschke et al., 2013; Nisbet et al., 2020). Reducing anthropogenic CH₄ emissions is an important step in mitigating global warming, given that CH₄ has a substantial climate-forcing potential, estimated to be 40% that of CO₂ over the last 150 years (Venkatraman and Ashwath, 2009). Landfill gas generation continues even after the closure of a landfill due to the anaerobic decomposition of organic waste, making landfills a continuous source of CH₄ (Bruce et al., 2018). This highlights the importance of establishing long-term mitigation approaches. Christophersen et al. (2001) state that landfill CH₄ and CO₂ emissions have generally been the subject of limited field studies, and temporal variability has attracted little attention.

Several variable factors impact CH₄ oxidation in landfill cover soil, including methanotroph activities and community structures, depth and features of cover soil, and climatic conditions (Serrano-Silva et al., 2014). Climate, soil cover characteristics including water content, nutrient availability, pH, texture, porosity, fissures, and fractures, as well as the age and distribution of waste, may all significantly affect CH₄ emissions spatially and temporally (Gonzalez-Valencia et al., 2016). Due to the close connection between these variables and plant physiology, estimates of CH₄ emissions from vegetated landfills are not completed unless climatic conditions are included (Xin et al., 2016). The characteristics of the vegetation in closed landfills are currently the subject of little information. On the other hand, barely any study has been done on how plant cover affects the ability of landfill cover soil to oxidize CH₄ and reduce CH₄ emissions as a result. A greater understanding of the connection between plant condition, cover soil features, and landfill gas emission would be beneficial for landfill site management related to reduction of GHG emissions (Xiaoli et al., 2011). Due to variations in soil temperature, moisture content, and ambient pressure, the emissions behavior of the landfill site changes significantly across time and space. Klusman and Dick (2000) emphasized that it is crucial to collect data across several seasons to accurately evaluate the effect of CH₄ produced by landfills.

Huber-Humer et al. (2009) also mention that even after a landfill closure, CH₄ generation continues as fugitive emissions which has the potential to last for up to a century. Furthermore, according to Huber-Humer et al. (2008), "hot spots" created by the LFGs escaping via the vents around the collecting systems increase the quantity of emissions into the environment. The installation of an LFG extraction system is a primary means of controlling emissions in contemporary designed landfills. However, the need for controlling fugitive emissions that the extraction systems are unable to capture is growing (Sadasivam and Reddy, 2014). Canada declared its support for the Global Methane Pledge, to reduce CH₄ emissions by 30% below 2020 levels by 2030 (Environment and Climate Change Canada, 2021). However, there is a lack of comprehensive and timely research that considers the complex interrelationships between environmental factors, the influence of plant communities on CH₄ emissions, and the effects of changing seasons on CH₄ gas emissions in the context of CH₄ emissions from capped landfills in southern Ontario.

One of the main unknowns in calculating national or international CH₄ emissions from landfills is the quantification of CH₄ oxidation (Bogner and Spokas, 1993). It has been found that frequently

changing environmental conditions challenge our understanding of the soil oxidation capacity (Spokas & Bogner, 2011). It is evident that methanotrophic bacteria in landfill cover soil lower the quantity of CH₄ released by certain landfills, while it is yet unclear how significant this is quantitatively. In older sites where the quantity of CH₄ in LFG is too low for flare or energy recovery, manipulating landfill cover soils to optimize their CH₄ oxidation capability may offer a different approach for controlling CH₄ emissions (Röwer et al., 2011). As mentioned in Kightley et al. (1995), quantification of microbial CH₄ oxidation in landfill cover soil is needed with special concern to older sites where flaring is not practical. Although previous studies have discussed the role of vegetation and soil conditions in CH₄ dynamics, the specific processes which drive CH₄ flux across field and controlled laboratory experiments are still not well understood.

Thus, this research aims to study the effect of vegetation, environmental drivers and microbial CH₄ oxidation capacity on CH₄ dynamics by integrating field measurements with incubation trials to support more effective landfill cover designs and develop CH₄ mitigation management strategies in capped landfills. The following general and specific objectives provide the framework of this study.

1.12 General and Specific Objectives

The overall aim of this study is to establish a comprehensive understanding of the interrelationships among vegetation, soil conditions, and meteorological factors in influencing CH₄ dynamics of capped landfills in southern Ontario by integrating both field observations and lab-based experiments. This will support the examination of existing and suitable conditions for microbial CH₄ oxidation in landfill cover soil, a key passive mechanism for CH₄ mitigation.

The specific objectives were to:

1. Evaluate the effect of vegetation on soil properties and effect of vegetation treatments on CH₄ flux in landfill cover soils (Chapter 2).
2. Examine the role of soil moisture, soil temperature, season and meteorological variables in regulating CH₄ dynamics (Chapter 2).
3. Quantify CH₄ oxidation rates in both vegetated and non-vegetated landfill soils under natural field conditions (Chapter 2).

4. Quantify CH₄ oxidation rates in vegetated and non-vegetated soils under controlled soil temperature and soil moisture conditions (Chapter 3).
5. Determine the optimal soil moisture and temperature levels for CH₄ oxidation in landfill cover soils and whether this varies with vegetation cover (Chapter 3).

1.12.1 Organization of Thesis

The above-mentioned objectives will be critically addressed by two primary research chapters in this thesis. Chapter 2: Effect of Vegetation Cover and Environmental factors on Methane Flux in a Capped Landfill in Southern Ontario addresses Objectives 1, 2, and 3. This chapter explores a CH₄ emissions and oxidation based on field investigations incorporating the influence of vegetation, soil conditions, and meteorological variables across two capped landfill sections of various ages. Chapter 3: Assessing CH₄ Oxidation Responses to Controlled Soil Moisture and Temperature: A Landfill Soil Incubation Study addresses objectives 4 and 5. This chapter details a laboratory-based incubation study conducted under controlled conditions to examine the optimal soil moisture and temperature conditions for CH₄ oxidation rates in vegetated and non-vegetated landfill soils.

The final chapter offers an integrated discussion for findings from both field and incubation studies, delivering key insights on CH₄ dynamics in capped landfills, and providing directions on landfill management strategies, practices and further research for CH₄ emission mitigation.

Chapter 2

Effect of Vegetation Cover and Environmental factors on Methane Flux in a Capped Landfill in Southern Ontario

2.1 Introduction

Methane (CH_4) is the second most important greenhouse gas (GHG) after carbon dioxide (CO_2) and is responsible for 40% of global warming during the past 150 years (Lelieveld et al., 1998). Estimated global CH_4 emission levels based on atmospheric inversion modeling during 2000 to 2019 was averaged to $575 \text{ Tg CH}_4 \text{ yr}^{-1}$ and roughly 65% ($\sim 369 \text{ Tg CH}_4 \text{ yr}^{-1}$) of this total were generated from anthropogenic sources such as waste, agriculture, burning biomass, and fossil fuel extraction and use (Saunois et al., 2025). Landfills are commonly known as significant anthropogenic CH_4 emission sources (Chen et al., 2008). Landfills are the most common method of disposing of waste in Canada, accounting for 97% of Municipal Solid Waste (MSW), and the country has over 2000 operational landfills, 880 of which are in the province of Ontario (CCME, 2014). In 2020, Canadian landfills accounted for 24% of Canada's total anthropogenic CH_4 emissions (Government of Canada, 2023). Canada aims to reduce CH_4 emissions by 30% below 2020 levels by 2030 (ECCC, 2021) and management of landfill emissions can play a role in reaching this target. Landfill gas generation continues even after the closure of a landfill due to the anaerobic decomposition of MSW, making landfills a continuous source of CH_4 (Bruce et al., 2018). This highlights the importance of establishing long-term mitigation approaches.

Many countries are actively working to mitigate landfill CH_4 emissions. For example, as mandated by the Danish landfill directive, landfill gas should be controlled via techniques such as installing gas collection and energy utilization systems, flaring, or bio-mitigation approaches like microbial CH_4 oxidation in the landfill cover soils and constructed biofilters (Miljøministeriet, 2019). However, low and diminishing gas production rates were observed at legacy landfills in Denmark leading to approximately 50% reduction across 23 studied landfills (Duan et al., 2022a). Extracting landfill gas with high CH_4 content through gas collection facilities is not effective at aging landfills with declining gas production. Implementing biocovers is considered as an environmentally friendly approach to replace gas collection and recovery systems in older landfills to mitigate persistently escaping CH_4 emissions during and after the gas collection period has ended.

Therefore, maintaining long-term high levels of CH₄ oxidation efficiencies in biocovers is essential to meaningfully reduce GHG emissions throughout the landfill's lifetime (Scheutz et al., 2023).

Methane consumption in cover soils arises from microbial CH₄ oxidation by methanotrophs (Scheutz et al., 2004). Landfill cover soil shows high temporal and spatial variability in CH₄ oxidation depending on climate, soil temperature, moisture content, nutrient availability, pH, texture, porosity, fissures, and fractures, as well as the age and distribution of waste (Albanna et al., 2007; Gonzalez-Valencia et al., 2016). Changes in environmental factors directly influence the biological process of CH₄ oxidation and physical gas transport processes (Rachor et al., 2013). Temperature is a crucial factor for microbial activities in landfill soils. The optimal temperature range for CH₄ oxidation in cover soil was found to be 31–36 °C according to laboratory experiments (Whalen et al. 1990; Czepiel et al. 1995). Soil moisture content is also an important regulator of the rate of CH₄ oxidation in the upper soil surfaces of the landfill cover. The presence of ample moisture is needed for the physiological activities of methanotrophs (Rachor et al., 2013). On the contrary, CH₄ oxidation rate is reduced by the high levels of moisture content due to the restrictions of CH₄ and oxygen (O₂) diffusion (Klusman & Dick, 2000). Thus, moisture is a crucial factor influencing CH₄ emission from landfills as it plays two distinct roles in boosting and restricting gas production. Organic matter-rich substrates can enhance the microbial oxidation of CH₄ in landfill cover soil (Pariatamby et al., 2015). The results of Chanton and Liptay (2000), who observed an increase in CH₄ oxidation activity in soils rich in organic matter, are consistent with this observation. According to Christensen et al. (1996), organic matter serves as a transporter for microorganisms, improving the substrate and soil qualities. Soil texture regulates the movement of landfill gases and the O₂ infiltration of the landfill cover soil, thereby affecting CH₄ oxidation activities. Soil with coarse texture rich with air-filled pore spaces promotes CH₄ oxidation (Gebert and Grongroft, 2009). Soil with abundant large pores (>50 µm) facilitates gas transport under moist conditions due to CH₄ diffusion through air 100 times faster compared to diffusion through air (Scheutz et al., 2009a;2009b).

Meteorological factors, such as temperature, air pressure, wind, and precipitation, all impact landfill CH₄ emissions (Scheutz et al., 2009a). Monitoring CH₄ emissions over different seasons provides a more accurate estimate of yearly landfill gas emissions. In winter, CH₄ emission may increase due to inhibition of CH₄ oxidation activity with the low temperatures and moist soil

conditions (Maurice and Lagerkvist, 2003; Rachor et al., 2013). Late summer CH₄ oxidation activities have also been found to be lower due to summer dryness and heat from CH₄ oxidation (Maurice and Lagerkvist, 2003). Börjesson and Svensson (1997) and Christophersen et al. (2001) discovered similar seasonal trends of CH₄ and CO₂ emissions, finding greater CH₄ emissions in winter and higher CO₂ emissions in summer. Klusman and Dick (2000) underscored the need for comprehensive year-round monitoring of gas fluxes to capture CH₄ production and accurately quantify its impact on atmospheric gas dynamics. Geberth et al. (2011) mentioned that pressure differences induced by the wind caused higher CH₄ emissions at specific sites, highlighting the connection between CH₄ concentration changes in topsoil and wind's impact on soil gas movements (Poulsen and Møldrup, 2006). Notably, CH₄ concentrations considerably decreased as wind speed increased, indicating wind interferes with the physical movement of gases in the soil (Geberth et al., 2011). Furthermore, wind-driven gas transport in soil is mainly determined by the soil's air permeability and the magnitude of the wind-driven pressure variations (Poulsen and Møldrup, 2006). Variations in barometric pressure significantly influence landfill CH₄ emissions by suppressing CH₄ emissions from landfills with rising barometric pressure, while releasing CH₄ with falling barometric pressure. This phenomenon is commonly described as barometric pumping (Xu et al., 2014). Katy et al. (2009) found that vertical gas migration rates were increased with dropping barometric pressure. Inversely, rising barometric pressure allows air to reach the ground and dilute the soil gas concentrations. The rate of change in the barometric measure is a significant factor that should be considered when establishing migration controls (when optimizing properties of soil matrix such as permeability, moisture content and soil compaction levels).

Plants can affect CH₄ dynamics in several ways. Plants can improve CH₄ oxidation by providing O₂ to the soil (Abichou et al., 2006). Plants may also emit CH₄ from soils (Carmichael et al., 2014), acting as gas conduits because of plant aerenchyma cells that may boost the release of CH₄ (Xin et al., 2016). When tree roots route CH₄ from below ground in landfills, the portion of CH₄ oxidized is reduced, possibly leading to higher CH₄ emissions (Fraser-McDonald et al., 2022). Therefore, planting trees on closed landfill sites increases CH₄ emissions (Maier et al., 2018; Pangala et al., 2013). On the other hand, plant roots can facilitate O₂ diffusion, provide root exudates, and promote CH₄ oxidation by enhancing microbial activities (Abichou et al., 2006). For example, Xiaoli et al. (2010) observed a higher CO₂:CH₄ ratio in vegetated soil relative to bare cover soil, explaining elevated CH₄ oxidation with vegetation.

Several variable factors impact CH₄ oxidation in landfill cover soil, including methanotroph activities and community structures, depth and features of cover soil, and climatic conditions (Serrano-Silva et al., 2014). Due to the close connection between these variables and plant physiology, CH₄ emissions from vegetated landfills are not completely assessed unless climatic conditions are included (Xin et al., 2016). The characteristics of the vegetation in closed landfills and its role in CH₄ release are currently poorly understood. Similarly, few studies have been done on how plant cover improves the ability of landfill cover soil to oxidize CH₄ and reduce CH₄ emissions as a result. A greater understanding of the connection between vegetation cover, soil cover features, and landfill gas emission would be beneficial for landfill site management as well as the reduction of GHG emissions (Xiaoli et al., 2011).

Therefore, the objectives of this study were to: i) Evaluate the effect of vegetation on soil properties in landfill cover soils; ii) Study the influence of vegetation treatment on CH₄ dynamics in the landfill covers; iii) Examine the effect of meteorological variables (e.g., precipitation, air temperature, wind speed) and soil moisture and soil temperature in regulating CH₄ flux; iv) Analyze the seasonal variation in CH₄ flux across treatments; v) Quantify CH₄ oxidation rates in both vegetated and non-vegetated landfill soils in the field conditions.

2.2 Materials and Methods

2.2.1 Site Location and Study Design

This study was conducted across two capped landfill sections at a Southern Ontario landfill (43°26' N 80°34' W). The total area of the site is 197 ha, about 71.1 ha of which are used for the disposal of solid waste to the maximum allowed capacity of 20,000,000 m³ of non-hazardous commercial, industrial, and domestic waste. Currently, this landfill maintains both active and capped landfill sections. The two capped landfill sections selected for this research were the Original Landfill Area (OLA) and Northern Expansion Area (NEA), which were opened in 1972 and 1994 and closed and capped in 2001 and 2016, respectively. A leachate collection system is installed in both capped landfill sections, and an active landfill gas collection system is operating in NEA at present. For the final cover of OLA, 0.9 m of till capping with 0.1 m of topsoil on top was placed. NEA was covered with a 0.9 m low-permeable clay cover followed by a 0.15 m layer of vegetated topsoil (Landfill Management Annual Reporting, anonymity requested). Woody vegetation is present in

the upper section, and dense herbaceous plants, mainly grasses with scattered shrubs spread over the bottom section of OLA. The entire NEA is filled with herbaceous plants dominated by grasses, and that vegetation layer is thin compared to OLA (Figure 2.1).

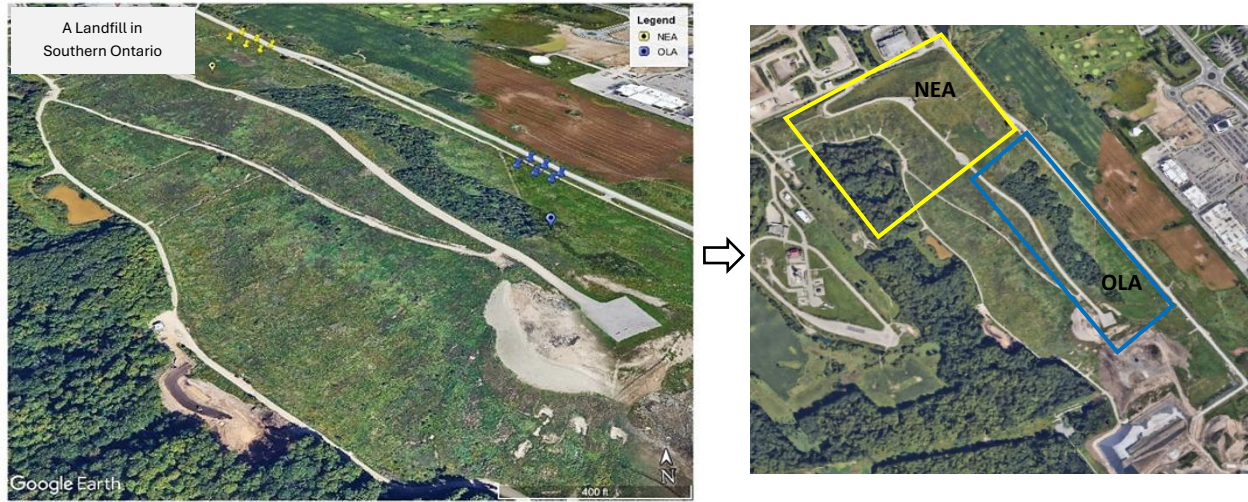


Figure 2.1: Satellite View of a Landfill in Southern Ontario Showing Vegetated and Non-Vegetated Collar Locations in the North Expansion Area (NEA) and the Original Landfill Area (OLA) (Google Earth Pro, 2025) (Left). Methane flux sampling locations are marked with pins (NEA = yellow, OLA = blue).

Six sampling locations were selected on each site with three of them located near the edge of the landfill at the base of the slope and the rest located in the middle of the sloping landfill surface; 20 m distance was maintained between each location. A pair of 1×1 m plots were made in each study location for vegetated and non-vegetated treatments with 50 cm between plots within the pair. The above ground vegetation was manually removed and maintained throughout the study time (Figure 2.2) in the non-vegetated plots. Fluxes were measured approximately 2-5 minutes after vegetation removal. Collars for CH_4 flux measurements were installed in the centre of the 1 m^2 plot by inserting them ~ 5 cm into the topsoil and were left in place throughout the study period.



Figure 2.2: Vegetated (Left) and Non-Vegetated (Right) Plots with Installed Collars in the Landfill Cover Soil

2.2.2 Closed Chamber Fluxes and Environmental Variables

An acrylic chamber (height: 40.5 cm, diameter: 21 cm) equipped with a fan and thermocouple was used to measure CH₄ fluxes (Chen et al., 2008). The chamber was placed on the permanently installed collar, and water was added to the collar rim to prevent air leakage. LI-7810 CH₄/CO₂/H₂O trace gas analyzer (Licor, USA) was then connected to the closed chamber, and the CH₄ concentration in the headspace was recorded every second for three minutes. The WET-2 sensor connected to an HH2 reader (Delta-T Devices, UK) was used to record soil moisture and soil temperature in the top 10 cm of soil. Field data were collected once or twice per week throughout summer and fall and once per week to twice per month during winter and spring, from April 2024 to April 2025, except for two winter months.

CH₄ flux (mg CH₄ / m² / d) was calculated based on the linear rate of increase in CH₄ concentration ($\mu\text{mol CH}_4 \text{ mol air}^{-1} \text{ s}^{-1}$) in the headspace of the closed chamber using a temperature-corrected volumetric approach. The equation incorporated the molar mass of CH₄ ($16.04 \times 10^3 \text{ mg mol}^{-1}$), the chamber headspace volume (L), the average internal chamber headspace temperature (K), and the standard molar volume of air (22.4 L mol^{-1}). CH₄ flux was normalized to the surface area of the chamber (m²) and converted to a daily rate. Extremely high fluxes were treated as hotspots by

visually inspecting the CH₄ concentrations-time curves and comparing them with other flux measurements. During quality control measurements, curves started with higher concentrations (typically, concentration readings start within 14,000 - 328,000 ppb/s) and then suddenly dropping were excluded while curves with weak linear fits ($R^2 < 0.75$) were also removed. To further quality control the data for other possible outliers, a filtering threshold range of $\pm 25 \text{ mg CH}_4 / \text{m}^2 / \text{d}$ was used and fluxes within this range were taken into the analysis of non-hotspot fluxes.

Weather data for precipitation, air temperature, wind speed and atmospheric pressure were obtained from the Kitchener/Waterloo Station (43°27'39"N, 80°22'43"W; ECCC, 2024). Precipitation data were collected as cumulative precipitation over the 48 hours before each flux measurement, whereas other variables were selected for the same hour of flux collections. Climate normals of precipitation and air temperature values from 1991-2020 were obtained from the same station (ECCC, 2025) and used as a baseline to compare weather patterns during the sampling period from April 2024 to April 2025.

2.2.3 Soil Chemical and Physical Properties

Selected soil chemical properties were analyzed by using air dried and 2 mm sieved soil from all vegetative and non-vegetative plots in OLA and NEA sections and analyzed at the Natural Resources Analytical Laboratory, University of Alberta.

To determine available ammonium (NH₄⁺), nitrate (NO₃⁻), phosphate (PO₄³⁻) and potassium (K⁺), 5 g of air dried, sieved, homogenized soil was shaken with 50 ml of extraction solution for 30 minutes and filtered. The 2.0 M KCl extraction method was used for available NH₄-N and NO₃-N with concentrations determined by colorimetric methods. Modified Kelowna extraction method was used for available PO₄-P and K, analyzed by colorimetric methods and ICP-OES, respectively (Maynard et al., 2008).

pH and electrical conductivity (EC) were measured using a Fisher Scientific AR 20 meter. Soil samples were mixed with water in specific ratios (1:5 soil-to-water ratios), and readings were adjusted to 25 °C (Hendershot et al., 2007; Miller & Curtin, 2007; Karla, 1995).

A Shimadzu TOC-L CPH Total Organic Carbon Analyzer at the Natural Resources Analytical Laboratory, University of Alberta was used to analyze dissolved organic carbon (DOC) concentration, followed by the non-purgeable organic carbon (NPOC) method. Aqueous soil

extracts were prepared, and samples were filtered through a 0.45 μm membrane and then acidified with 1 M HCl. Purgeable organic and inorganic carbon were removed, and the non-volatile carbon was combusted at 720 $^{\circ}\text{C}$ in the presence of a platinum catalyst. The produced CO_2 was detected using a non-dispersive infrared (NDIR) detector. Certified potassium hydrogen phthalate standards were used for quantification. DOC values were determined by subtracting inorganic carbon from total carbon present in the samples (Shimadzu Corporation, 2001; Williams, 2000).

Dry bulk density was determined by oven drying a known volume of soil at 105 $^{\circ}\text{C}$ until it reached a constant weight. The bulk density was calculated as the ratio between dried soil mass and volume of the soil sample (Panagos et al., 2024) as shown in equation (2.1).

$$\text{Bulk Density (g/cm}^3\text{)} = \frac{\text{Dry Mass of Soil (g)}}{\text{Total Soil Volume (cm}^3\text{)}} \quad (2.1)$$

To determine porosity, a known volume of oven-dried (V_s) soil was sampled and water (V_l) was added to the pore spaces until saturated (V_g). Total sample volume was also measured (V). Porosity (ϕ) is determined by a portion of pore space volume to the total volume (Flint & Flint, 2002) according to below equations (2.2 and 2.3),

$$V = V_s + V_l \quad (2.2)$$

$$\phi = \frac{V_l}{V} \quad (2.3)$$

2.2.4 Quantification of Oxidized CH_4 Per Mass of Landfill Cover Soil

The amount of oxidized CH_4 per gram soil of vegetated and non-vegetated treatments was estimated by scaling the average CH_4 flux rate to the mass of soil at each plot. The soil mass of 1 $\times$ 1 m area was determined by multiplying cover soil volume (assuming 1 m $\times$ 1 m surface area and a topsoil depth of 0.10 m for OLA and 0.15m for NEA, resulting a total volume of 0.10 m^3 and 0.15 m^3 , respectively) by soil bulk density. Then, the scaled CH_4 flux was normalized by this mass to express oxidized CH_4 in $\mu\text{g CH}_4/\text{g soil/d}$ by using below formula (2.4).

$$\text{CH}_4 \text{ oxidized (}\mu\text{g CH}_4/\text{g soil/d)} = \frac{\text{CH}_4 \text{ flux (}\mu\text{g CH}_4/\text{m}^2/\text{d)} \times \text{Surface area (m}^2\text{)}}{\text{Soil volume (cm}^3\text{)} \times \text{Bulk Density (g/cm}^3\text{)}} \quad (2.4)$$

2.2.5 Statistical Analysis of Data

All data were analyzed using R version 4.3.1 (R Core Team, 2023) and RStudio version 2023 (RStudio Team, 2023) using a significance level of $\alpha = 0.05$. The ggplot2 package (Wickham et al., 2016) was used to create graphs. CH₄ flux data were log-transformed to meet the assumptions of normality, then visually confirmed through Q-Q plots and residuals.

Differences in soil physical and chemical properties such as bulk density, porosity, DOC, soil nutrient contents and pH between vegetated and non-vegetated treatments and landfill sections (i.e., sites) were evaluated using linear models and the 'lm' function in base R whereas the effect of treatment, site and their interaction on soil moisture and soil temperature were analyzed using the linear mixed effect models (LMMs) (lme4 package; Bates et al., 2015) and ANOVA output with Type III sum of squares from the lmerTest package (Kuznetsova et al., 2014) were employed. When the main effect was significant, emmeans package (Lenth et al., 2018) was used to obtain estimated marginal means, then Tukey's HSD test was used with Tukey-adjusted pairwise comparisons to control multiple comparisons.

LMMs were again employed to evaluate the effect of treatment and site and their interactions on CH₄ flux, while considering repeated measurements from collar locations by including collar as a random effect. A similar analysis was conducted to study monthly variations in CH₄ flux by treatment, and the effect of weather variables including precipitation (48-hour), air temperature, wind speed, and atmospheric pressure on variation in CH₄ flux, with collar kept as the random effect. To determine the significance of fixed effects in the built models Type III ANOVA was used. Estimated marginal means were then obtained using emmeans package and Tukey's HSD test (Lenth et al., 2018) was used with Tukey-adjusted pairwise comparisons to control for multiple comparisons.

To determine the effect of soil moisture, soil temperature, treatment and their interaction on CH₄ flux a series of linear mixed effect models were used, while treating soil moisture, soil temperature, treatment and their interactions as the fixed effects and considering collars as a random factor in the initial model developed. I then performed model selection by removing non-significant terms one by one starting with the term with the higher p-value based on the Type III ANOVA output. It revealed that vegetation treatment had no significant impact on CH₄ flux, nor did its interaction

with soil moisture or soil temperature. As the next step, non-significant effects were excluded through a stepwise reduction process and only soil moisture and soil temperature remained in the final model.

2.3 Results

2.3.1 Meteorological Conditions

The study period from April 2024 to April 2025 (Table 2.1) received 812.2 mm of precipitation, lower than the historical 30-year average of 828.8 mm. Monthly average precipitation was highest in July 2024 (149 mm), followed by April 2024 (108 mm) and May 2024 (102.1 mm) with these months exceeding historical norms, whereas little precipitation was observed in September 2024 (23.4 mm), then started increasing gradually until December 2024. Precipitation declined to the lowest levels in early 2025, with 8.7 mm (January 2025). Average monthly air temperature across the monitoring period was 9.93 °C, slightly warmer than the 30 years historic average of 9.04 °C. Except for July 2024, air temperature values through the study period were warmer compared to their 30-year monthly averages with higher air temperatures recorded in the summer and early fall. Aligned with air temperature fluctuations, soil temperature variations showed a seasonal trend (Table A3). The highest soil temperatures were observed in both vegetated plots and non-vegetated plots in July 2024 at 34.1 °C and 33.1 °C, respectively. During the winter, a significant decline in soil temperature was observed with December 2024 having the lowest temperature at 5.84 °C in vegetated and 3.44 °C in non-vegetated plots (Table A3). Seasonal shifts in air temperature and precipitation were reflected in soil moisture level fluctuations. Peak moisture conditions occurred in April 2025, with average values of 38.1% in vegetated and 37.2% in non-vegetated plots. Conversely, soil moisture declined to 7.2% (vegetated plots) and 3.9% (non-vegetated plots) in August 2024, followed by similarly low levels in September and October 2024 due to minimal precipitation and elevated air temperatures. Atmospheric pressure remained at a narrow range from 97.2 kPa to 99.0 kPa, being highest in April 2024 and the lowest in May 2024, during the sampling period. Wind speed varied more markedly, spanning monthly from 11.3 to 29.1 km/h, peaking in April 2025 and being lowest in July 2024. Even though some month-to-month trends were observed in both variables, neither of these demonstrated a clear seasonal pattern (Table A3).

Table 2.1: Average Monthly Values of Air Temperature and Precipitation Over the Study Period Compared To 1991–2020 Historical Normals^a

Year 2024					
Month	Year	Air Temperature (°C)	Historical Air Temp (°C) (1991-2020)	Precipitation (mm)	Historical Precipitation (mm) (1991-2020)
Apr	2024	7.80	5.9	108.0	81.2
May	2024	15.04	12.6	102.1	80.2
Jun	2024	18.80	17.8	81.5	80.5
Jul	2024	18.38	20.2	149.0	96.2
Aug	2024	19.24	19.1	103.5	67.2
Sep	2024	16.72	15.2	23.4	75.2
Oct	2024	10.11	8.8	32.0	71.0
Nov	2024	5.27	2.6	43.6	74.9
Dec	2024	-1.90	-2.8	67.4	54.9
Average		12.16	11.71	710.5	681.3
Year 2025					
Month	Year	Air Temperature (°C)	Historical Air Temp (°C) (1991-2020)	Precipitation (mm)	Historical Precipitation (mm) (1991-2020)
Jan	2025	-7.50	-6.3	8.7	66.3
Feb	2025	-6.50	-5.9	35.7	46.1
Mar	2025	1.40	-0.8	15.0	57.0
Apr	2025	6.62	5.9	43.6	81.2
Average		-1.49	-1.77	102	250.6

a. Data obtained from ECCC, 2025

2.3.2 Influence of Vegetative and Non-Vegetative Treatments on Soil Properties in Landfill Covers

As shown by Table 2.2 and 2.3, vegetation treatment had a notable effect on $\text{NH}_4^+\text{-N}$ levels in the soil, whereas site-level differences between OLA and NEA significantly influence bulk densities, EC, $\text{PO}_4\text{-P}$, and soil temperature. Other properties, including porosity, $\text{NO}_3\text{-N}$, K^+ , DOC, and pH, exhibited minimal variation between treatments or sites. Soil properties were generally altered and improved by vegetation treatment across the sites, though the differences between treatments were not always statistically significant.

The vegetation removal treatment did not have a significant effect on porosity ($F_{1,20} = 0.00$, $p = 0.961$) or bulk density ($F_{1,20} = 1.20$, $p = 0.2854$) (Table 2.2; Table 2.3). Nevertheless, vegetated soil had a slightly higher porosity (0.91% in OLA and 0.90% in NEA), whereas non-vegetated soil had a porosity of 0.89% at both NEA and OLA sites, although these differences were not significant. As shown by Table 2.3, in OLA, vegetated (0.81 g cm^{-3}) and non-vegetated soil (0.85 g cm^{-3}) had higher bulk density compared to vegetated (0.75 g cm^{-3}) and non-vegetated soil (0.76 g cm^{-3}) at NEA, resulting in a significant site effect for bulk density.

Vegetated soil exhibited marginally higher DOC levels in both OLA and NEA (relative to the non-vegetated soil), but there was no significant effect of site ($F_{1,20} = 1.51$, $p = 0.2333$), treatment ($F_{1,20} = 1.38$, $p = 0.2547$), or their interaction ($F_{1,20} = 0.56$, $p = 0.4615$) (Table 2.2; Table 2.3). Soil pH ranged between 7.78–7.85 regardless of treatment or sites (all $p > 0.51$). OLA and NEA vegetated plots had similar soil $\text{NH}_4\text{-N}$ concentrations that were significantly higher compared to their non-vegetative counterpart ($F_{1,20} = 8.23$, $p = 0.0095$), and neither site nor interaction terms were significant ($F_{1,20} = 0.08$, $p = 0.7805$; $F_{1,20} = 0.17$, $p = 0.6824$, respectively).

Table 2.2 and Table 2.3 show a significant difference between sites with elevated EC values in the OLA site compared to lower EC values in NEA ($F_{1,20} = 7.39$, $p = 0.0133$), though it did not show any significant vegetation or vegetation: site interaction effects ($F_{1,20} = 1.52$, $p = 0.2324$; $F_{1,20} = 0.00$, $p = 0.9526$). $\text{PO}_4\text{-P}$ exhibited no significant treatment or interaction effects, but it showed a significant difference between sites ($F_{1,20} = 5.64$, $p = 0.0277$) as $\text{PO}_4\text{-P}$ levels were relatively higher in OLA than NEA. Despite vegetated soils containing higher levels of $\text{NO}_3\text{-N}$ and K^+ than non-vegetated soils, both showed no significant differences between treatments, sites, or their interactions (Table 2.2; Table 2.3).

Vegetated plots in both OLA and NEA sections showed slightly elevated soil moisture levels, 19.13% and 18.15%, respectively, compared to soil moisture levels in non-vegetated plots at OLA and NEA, 17.93% and 15.18% accordingly (Table 2.3), but the differences among treatments, sites, and site: treatment were not significant (all $p > 0.05$). Soil temperature varied significantly between sites ($F_{1,20} = 6.96$, $p = 0.0083$), with higher values recorded in the OLA sections, but there was no statistical effect of treatment or treatment: site interaction on soil temperature. The OLA section was warmer with higher mean soil temperatures in both vegetation (23.99°C) and non-vegetated

soils (22.41 °C), whereas NEA showed overall lower soil temperatures (20.85 °C in vegetated soil and 20.55 °C in non-vegetated soil) (Table 2.3).

Table 2.2: Main Effects of Soil Physical and Chemical Properties Compared across Sites and Treatments

Model Description	Effect	DF	F / Chisq	p-value
Effect of Site and Vegetation on Bulk Density (g/cm ³)	Site	1,20	4.60	0.0445*
	Treatment	1,20	1.20	0.285
	Site × Treatment	1,20	0.10	0.754
Effect of Site and Vegetation on Percent Porosity (%)	Site	1,20	0.06	0.816
	Treatment	1,20	0.00	0.961
	Site × Treatment	1,20	0.02	0.882
Effect of Site and Vegetation on DOC (mg/L)	Site	1,20	1.51	0.233
	Treatment	1,20	1.38	0.255
	Site × Treatment	1,20	0.56	0.462
Effect of Site and Vegetation on EC (μS/cm)	Site	1,20	7.39	0.013*
	Treatment	1,20	1.52	0.232
	Site × Treatment	1,20	0.00	0.953
Effect of Site and Vegetation on Soil pH	Site	1,20	0.43	0.517
	Treatment	1,20	0.14	0.710
	Site × Treatment	1,20	0.00	0.947
Effect of Site and Vegetation on NH ₄ -N (mg/kg)	Site	1,20	0.08	0.781
	Treatment	1,20	8.23	0.009**
	Site × Treatment	1,20	0.17	0.682
Effect of Site and Vegetation on NO ₃ -N (mg/kg)	Site	1,20	2.09	0.164
	Treatment	1,20	0.02	0.894
	Site × Treatment	1,20	0	0.989
Effect of Site and Vegetation on PO ₄ -P (mg/kg)	Site	1,20	5.64	0.028*
	Treatment	1,20	0.02	0.901
	Site × Treatment	1,20	0.1	0.758

Model Description	Effect	DF	F / Chisq	p-value
Effect of Site and Vegetation on K (mg/kg)	Site	1,20	1.43	0.245
	Treatment	1,20	0.14	0.716
	Site × Treatment	1,20	0.01	0.909
Effect of Site and Vegetation on Soil Moisture (%) (LMM)	Site	1,21	0.63	0.429
	Treatment	1,21	2.19	0.139
	Site × Treatment	1,21	0.30	0.581
Effect of Site and Vegetation on Soil Temperature (°C) (LMM)	Site	1,670	6.96	0.008**
	Treatment	1,670	0.06	0.801
	Site × Treatment	1,670	0.57	0.450

2.3.3 Descriptive Analysis of CH₄ Flux

A total of 705 CH₄ flux observations were considered, with 17 (2.41%) hotspot records and 688 (97.59 %) non-hotspot records across both OLA and NEA capped landfill study areas. Descriptive statistics indicated a large variability in CH₄ flux; average CH₄ flux across both landfill sections with hotspots and non-hotspot data included was 2446 mg CH₄ / m² / d (standard deviation (SD) = 29344), while mean CH₄ flux dropped to -1.12 mg CH₄ / m² / d excluding hotspot data irrespective of treatment (Table A5; Figure 2.3). In order to assess the effect of vegetation and other environmental variables on CH₄ dynamics in the soils of the landfill cap, hotspot data and other outliers were removed to a ± 25 mg CH₄ / m² / d filtering threshold range (i.e., any points higher than 25 or lower than -25 mg CH₄ / m² / d were omitted), retaining 675 measured fluxes (~95% of dataset) after excluding hotspots, fluxes between hotspots and upper limit of threshold and fluxes lower than lower limit of the threshold.

In general, flux values fluctuated around zero and monthly flux variations were visible with a weak seasonal trend where variation was higher in warm months and median CH₄ flux was lower, indicating greater uptake (Figure 2.6). The difference in CH₄ flux between vegetated and non-vegetated treatments was minimal throughout the study period.

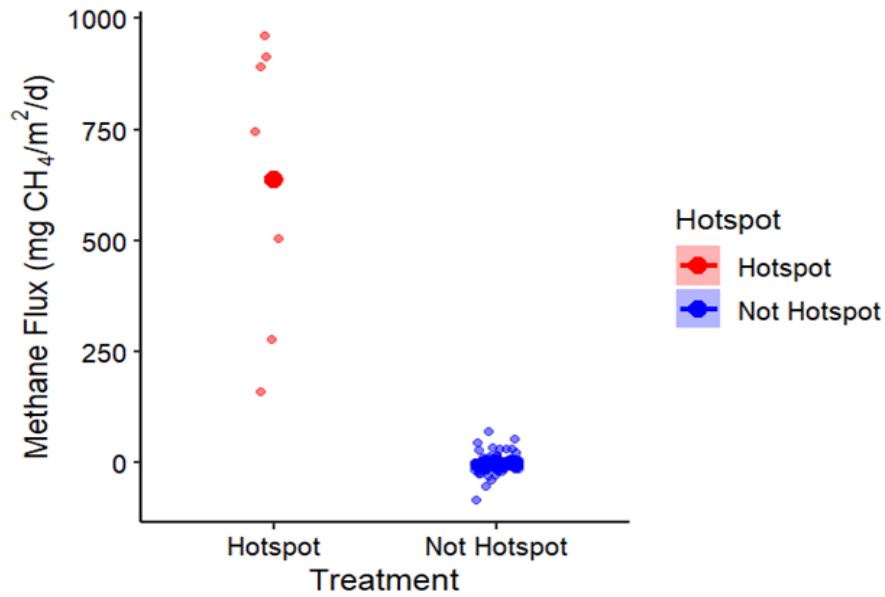


Figure 2.3: Mean CH₄ Flux across Sites including Hotspots and Non-Hotspots.

Table 2.3: Descriptive Statistics of CH₄ Flux and Soil Properties by Site by Treatment. Tukey's HSD Test Was Applied and Letters Not Shared Between Groups Denote Statistically Significant Differences. Each Value Shows the Mean and Standard Deviations in Brackets.

	OLA Section		NEA Section	
Variable	Veg. Treatment	Non-Veg. Treatment	Veg. Treatment	Non-Veg. Treatment
CH ₄ Flux (mg CH ₄ / m ² / d)	-0.59 (SD=3.69) ^a	-0.96 (SD=4.73) ^a	-1.88 (SD=3.35) ^a	-1.73 (SD=3.44) ^a
Soil Temperature (°C)	23.99 (SD=11.87) ^b	22.41 (SD=10.96) ^{ab}	20.85 (SD=10.57) ^{ab}	20.55 (SD=9.83) ^b
Soil Moisture (%)	19.13 (SD=11.72) ^a	17.93 (SD=12.37) ^a	18.15 (SD=11.55) ^a	15.18 (SD=11.93) ^a
Bulk Density (g/cm ³)	0.75 (SD=0.064) ^a	0.76 (SD=0.053) ^{ab}	0.81 (SD=0.042) ^{ab}	0.85 (SD=0.046) ^b
Porosity (%)	0.27 (SD=0.01) ^c	0.23 (SD=0.01) ^{ab}	0.25 (SD=0.02) ^b	0.22 (SD=0.01) ^a
DOC (mg/L)	92.60 (SD=20.97) ^a	89.55 (SD=8.68) ^a	78.55 (SD=16.19) ^a	66.79 (SD=17.94) ^a
pH	7.85 (SD=0.13) ^a	7.82 (SD=0.07) ^a	7.80 (SD=0.12) ^a	7.78 (SD=0.12) ^a
NO ₃ -N (mg/kg)	2.61 (SD=0.67) ^a	2.50 (SD=0.50) ^a	3.93 (SD=2.18) ^a	3.81 (SD=2.16) ^a
PO ₄ -P (mg/kg)	42.43 (SD=27.40) ^{ab}	46.03 (SD=28.90) ^b	14.92 (SD=4.20) ^{ab}	13.53 (SD=1.51) ^a

K ⁺ (mg/kg)	167.96 (SD=35.54) ^a	156.83 (SD=11.55) ^a	192.95 (SD=40.51) ^a	185.25 (SD=46.75) ^a
NH ₄ -N (mg/kg)	17.70 (SD=4.23) ^b	13.50 (SD=1.61) ^{ab}	17.20 (SD=4.00) ^b	11.90 (SD=1.78) ^a
EC (μS/cm)	356.80 (SD=53.3) ^b	325.20 (SD=41.7) ^b	283.80 (SD=45.6) ^{ab}	251.60 (SD=53.6) ^a

2.3.4 Effect of Vegetated and Non-Vegetated Treatments on CH₄ Flux

CH₄ flux did not significantly differ by treatment or interaction between treatment and site (all $p > 0.05$) (Table 2.4), but relatively lower CH₄ fluxes (i.e., high CH₄ uptake) were observed in the NEA section with -1.88 mg CH₄ / m² / d at vegetated plots and -1.73 mg CH₄ / m² / d in non-vegetated plots. At OLA, mean fluxes were -0.59 mg CH₄ / m² / d at vegetated plots and -0.96 mg CH₄ / m² / d at non-vegetated plots (Figure 2.4). Therefore, CH₄ flux was marginally significantly different by site (Table 2.4), with higher CH₄ uptake from the NEA landfill section with an active landfill gas collection system and a comparably younger landfill cap due to site-specific conditions regardless of the vegetation treatment.

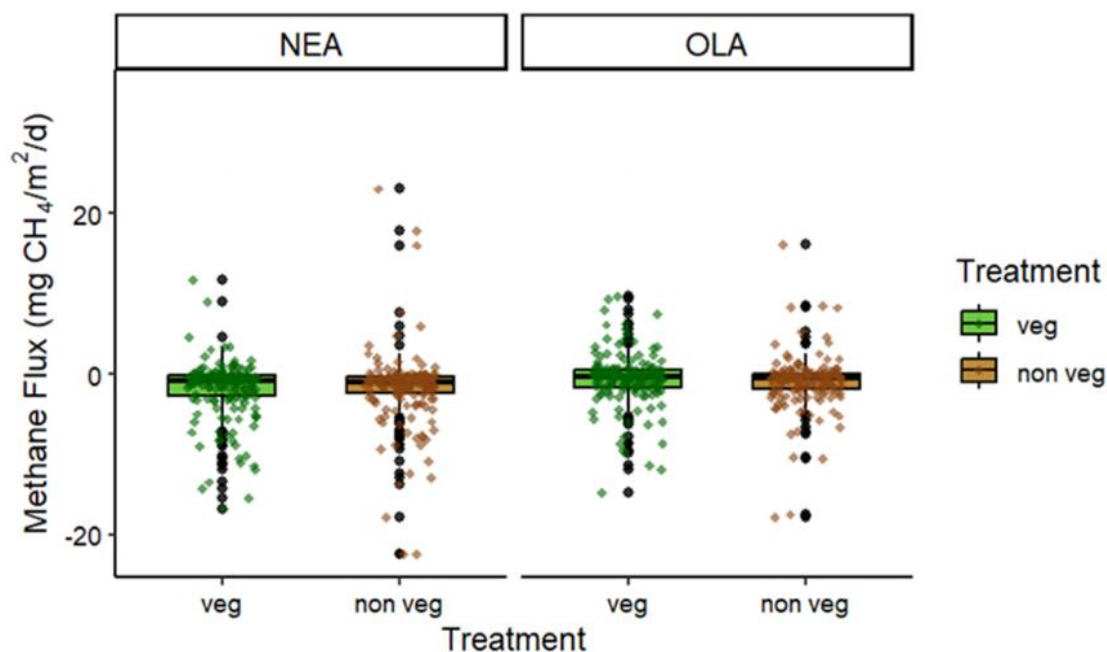


Figure 2.4: Mean CH₄ Flux for Vegetated and Non-Vegetated Plots across NEA and OLA.

Table 2.4: Summary of Type III ANOVA Results for Models Evaluating CH₄ Flux and Soil Variables

Model Description	Effect	NumDF	DenDF	F value	Pr(>F)	Marginal R ²	Conditional R ²
Effect of Vegetation and Site on CH₄ Flux	Treatment	1	15	0.074	0.789	0.159	0.746
	Site	1	15	4.256	0.056		
	Treatment × Site	1	15	0.075	0.787		
Soil Moisture and Soil Temperature Effects	Soil Temperature	1	643	0.999	0.318	0.023	0.051
	Soil Moisture	1	615	10.18	0.001 **		
Seasonal Variation in CH₄ Flux by Treatment	Treatment	1	30	0.247	0.623	0.168	0.905
	Months	8	573	130.87	<0.0001 ***		
	Treatment × Months	8	573	0.076	0.999		
Weather Variables Predicting CH₄ Flux	Precipitation (48 hrs)	1	560	5.161	0.024 *	0.178	0.397
	Wind Speed	1	574	0.050	0.823		
	Atm. Pressure	1	562	2.966	0.086		
	Air Temperature	1	568	4.967	0.026 *		

2.3.5 Effect of Soil Moisture and Soil Temperature on CH₄ Flux

A series of linear mixed effect models were developed including treatment, site, soil moisture, soil temperature, and their interactions. However, Type III ANOVA revealed that vegetation treatment had no significant impact on CH₄ flux, nor did its interaction with soil moisture or soil temperature.

Thus, these non-significant effects were systematically eliminated, retaining soil moisture and soil temperature as the sole main effects in the reduced model which are taken into this interpretation.

The final model identified that CH₄ flux was significantly influenced by soil moisture (Table 2.4), while soil temperature did not significantly explain variation in CH₄ flux. As shown by Figure 2.5, regardless of the vegetation or sites, CH₄ flux had a consistent positive trend with soil moisture, whereas a mild, but not statistically significant, negative connection was observed between soil temperature and CH₄ flux. Despite some minor differences in soil temperature and soil moisture because of vegetation removal (Section 2.3.2), these differences between treatments did not lead to a clear effect on CH₄ oxidation.

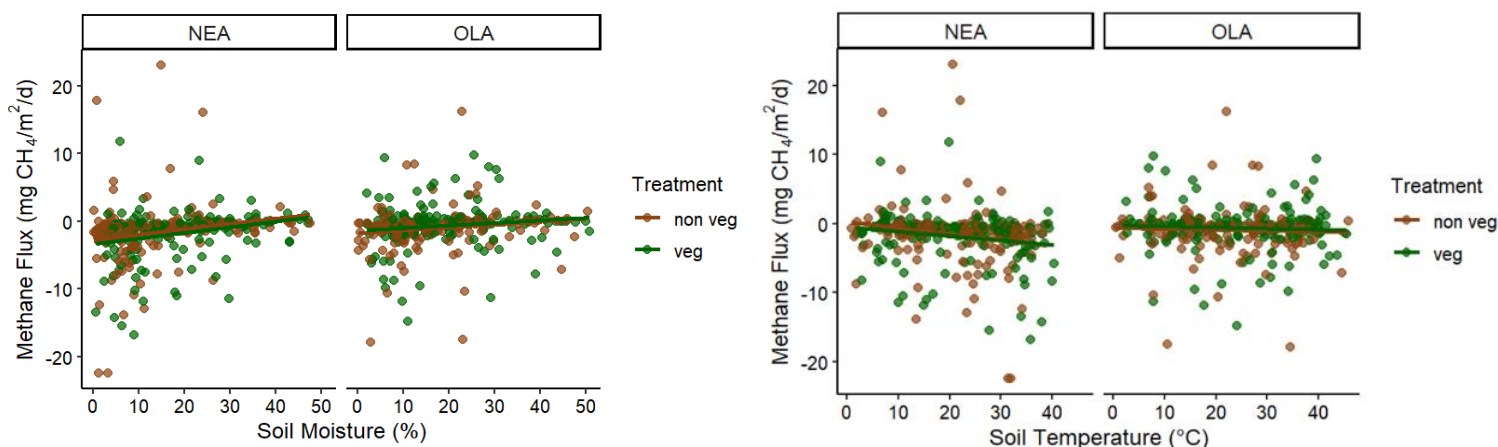


Figure 2.5: Effect of Soil Moisture and Temperature on CH₄ Flux across Vegetated and Non-Vegetated Plots at NEA and OLA Sites.

2.3.6 Monthly Variation in CH₄ Flux and the Role of Vegetation

CH₄ oxidation tended to increase from May to September 2024, with the highest uptake recorded in September 2024 in warmer and drier soil with peak soil temperatures of 31.2 °C and moisture dropping as low as 3.9% (Figure 2.6; Figure A3; Figure A4). Moreover, between October and December 2024 CH₄ uptake tended to weaken with cooler (soil temperature dropped to 3.4 °C) and moist (up to 24.5%) soil conditions. CH₄ uptake further declined from January to April 2025 with the increase of soil moisture (up to 37%) and relatively low soil temperatures (~7.3°C). In general, CH₄ fluxes tended to decrease from spring to summer with drier and warmer soil condition

and the trend shifted between fall to winter when declining CH_4 oxidation under wetter and cooler soil conditions resulted in higher CH_4 flux (Table A2).

CH_4 flux significantly varied across months ($F_{8,573} = 130.88$, $p < 0.0001$) while neither treatment ($F_{1,30} = 0.25$, $p = 0.6229$) and nor its interaction with months ($F_{8,573} = 0.076$, $p = 0.9997$) had a significant impact. Therefore, the seasonal trend in CH_4 dynamics was similar across treatments (Table 2.4). Pairwise comparisons among months indicated that the only difference was a significantly higher CH_4 flux in April 2025 than all other measured months.

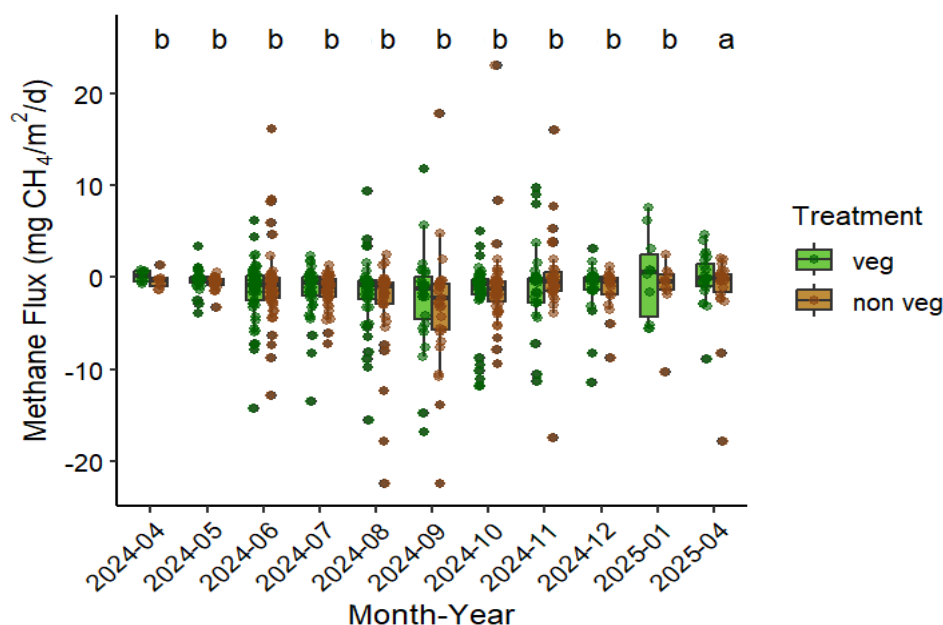


Figure 2.6: Monthly Variation in CH_4 Flux across Vegetated and Non-Vegetated Plots from April 2024 to April 2025. Tukey's HSD test was applied and Letters not shared between groups denote statistically significant differences. Each value shows the mean and standard errors in brackets. Statistical significance was set at $p < 0.05$ for all tests.

2.3.7 Influence of Meteorological Variables on CH_4 Flux

Analysis of the effect of weather variables on CH_4 dynamics revealed that precipitation had a positive ($F_{1,560} = 5.16$, $p = 0.0235$) significant effect on CH_4 flux and air temperature had a significant negative effect ($F_{1,568} = 4.97$, $p = 0.0262$) (Table 2.4, Table A1). During April 2025, CH_4 uptake dropped to $-0.72 \text{ mg CH}_4 / \text{m}^2 / \text{d}$, aligning with the one of the highest received short term precipitation (48 hour-precipitation) levels of 183.2 mm, whereas increased CH_4 uptake was recorded with minimal precipitation received August 2024 (1.8 mm) and September 2024 (0.6

mm), with fluxes of -2.25 and -2.79 $\text{mg CH}_4/\text{m}^2/\text{d}$, respectively. These high CH_4 uptake rates in August and September 2024 were also associated with the driest (5.5%) and warm ($\sim 30^\circ\text{C}$) soil conditions, coinciding with the warmer air temperature ($\sim 24.5^\circ\text{C}$), while lower CH_4 uptake was observed in April 2024 (-0.126 $\text{mg CH}_4/\text{m}^2/\text{d}$) and May 2024 (-0.447 $\text{mg CH}_4/\text{m}^2/\text{d}$) with cooler air temperature values (11.5°C and 15.5°C , respectively). Atmospheric pressure showed a marginally negative influence on CH_4 flux (Table 2.4; Figure 2.7). For instance, minimal CH_4 uptake (-0.716 $\text{mg CH}_4/\text{m}^2/\text{d}$) was observed with high atmospheric pressure in April 2025 (98.6 kPa) whereas relatively higher CH_4 release (-0.572 $\text{mg CH}_4/\text{m}^2/\text{d}$) was recorded in January 2025 with dropped atmospheric pressure (97.9 kPa), but this pattern was not consistent always. Wind speed had no significant influence on variation in CH_4 flux (Table 2.4; Table A1, Table A3).

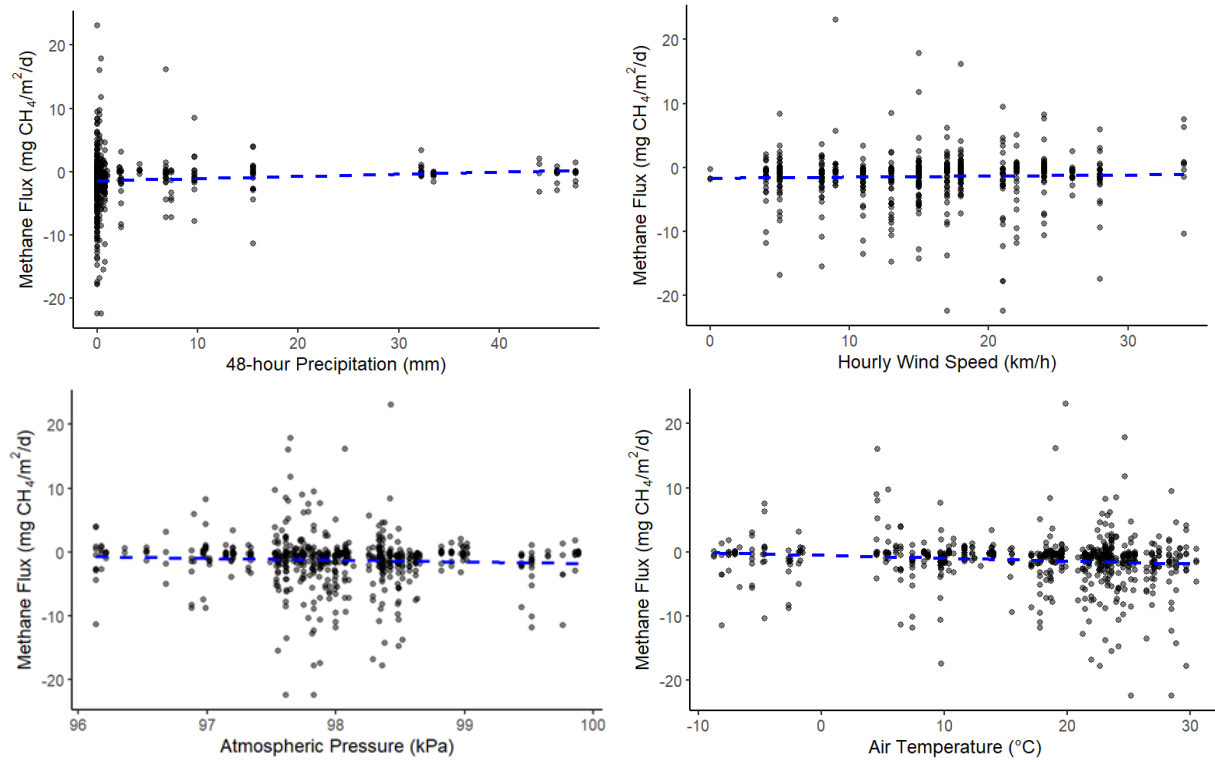


Figure 2.7: Scatterplots Showing the Relationships Between CH_4 Flux and Meteorological Variables:(a) 48-Hour Precipitation (mm), (b) Hourly Wind Speed (km/h), (c) Atmospheric Pressure (kPa), and (d) Air Temperature ($^\circ\text{C}$).

2.3.8 Quantification of CH₄ Oxidation Rates in Landfill Cover Soil

CH₄ oxidation rates in landfill cover soil with and without vegetation treatment were calculated by scaling the collected chamber fluxes to an estimated mass of soil (g) in topsoil layers in 1 m² of cover soil surface area which is derived from site specific bulk density values. As shown in Figure 2.8, NEA had higher estimated CH₄ oxidation rates under both vegetated (0.0156 µg CH₄ /g /d) and non-vegetated treatments (0.0138 µg CH₄ /g /d) compared to the OLA site oxidation rates of 0.0080 and 0.0126 µg CH₄ /g /d in vegetated and non-vegetated treatments, respectively. The influence of vegetation played only a minimal role in regulating CH₄ oxidation in cover soil in both NEA and OLA sites.

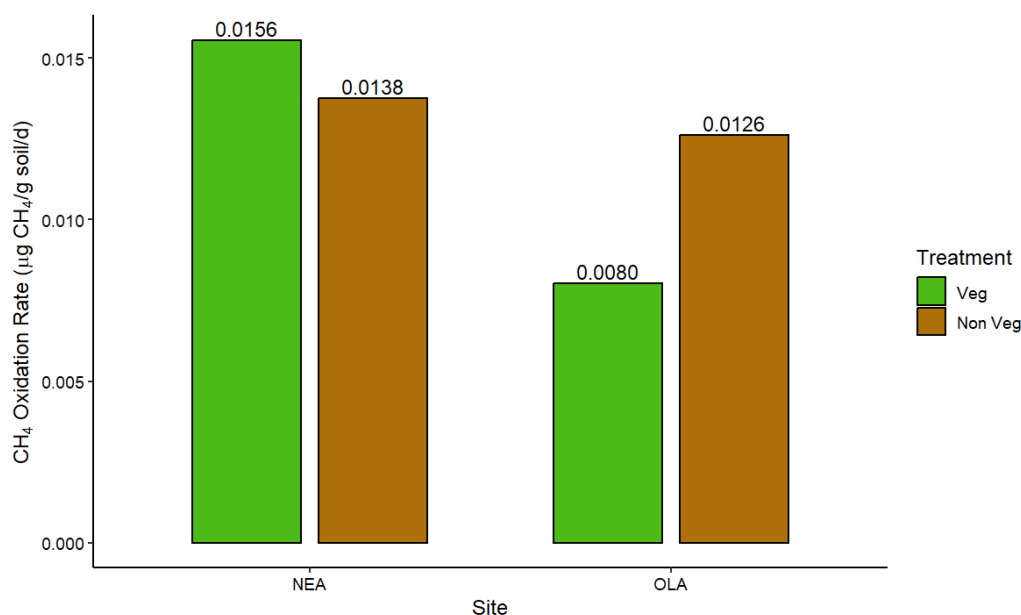


Figure 2.8: CH₄ oxidation Rates by Site by Treatment.

2.4 Discussion

2.4.1 The role of hotspots for CH₄ emissions in Landfill Covers

During the sampling period significant CH₄ flux variations were observed across the capped landfill covers due to the hotspots (high CH₄ emission spots). About 2.4% of gathered fluxes were identified as hotspots pushing average flux rates rose up to 2,446 mg CH₄ / m² / d when including these hotspots (some emissions exceeding 100,000 mg CH₄ / m² / d were also noted). There are

areas with weak structures in the landfill covers (such as cracks, fissures, slopes and cell intersections), leaks in leachate collection systems or pipe systems, which complicate the detection of localized hotspots and quantifying emissions due to their unpredictable nature (Fredenslund et al., 2010; Scheutz et al., 2011b). Identified hotspots were excluded as to quality control the data and resulting average flux significantly dropped to $-1.12 \text{ mg CH}_4 / \text{m}^2 / \text{d}$. Furthermore, to focus clearly on the influence of environmental factors and biological factors on CH_4 oxidation, a threshold of $\pm 25 \text{ mg CH}_4 / \text{m}^2 / \text{d}$ was used to filter data to improve the sensitivity of findings. In general, a net CH_4 uptake was observed in the landfill covers during the sampling months which aligns with the previous research observations, explaining that CH_4 oxidation activities are present in landfill caps due to methanotrophic action when environmental conditions are in a favorable range (Schuetz et al., 2009a; Spokas et al., 2006). Thus, CH_4 emission reduction efforts should focus on eliminating these hotspots while working to maximize CH_4 oxidation in cover soils.

2.4.2 Influence of Vegetative and Non-Vegetative Treatments on Soil Properties and CH_4 Flux Behavior in Cover Soil

According to the analysis conducted, vegetation treatment influenced the $\text{NH}_4^+\text{-N}$ content in the soil the most compared to other considered soil properties. The observed higher $\text{NH}_4^+\text{-N}$ levels in vegetated soils than non-vegetated soils was likely due to regular litterfall, root turnover, root exudates and microbial activities in rhizosphere as it sustains mineralized $\text{NH}_4^+\text{-N}$ in the soil despite active plant uptake (Schimel & Bennett, 2004; Kuzyakov & Xu, 2013; Zhu et al., 2021). In non-vegetated soils, this pathway is disrupted by shoot removal which is similar to biomass-removed systems that restrict the return of nitrogen rich plant residues to the soil (Robertson & Groffman, 2015). Under this condition, microbes will lock available nitrogen in their biomass for their growth (microbial immobilization) without releasing nitrogen via rapid mineralization (Paul & Juma, 1981), thus reducing available nitrogen species in the soil (Zhu et al., 2021). Though treatment did not significantly alter the $\text{NO}_3^-\text{-N}$ levels, higher $\text{NO}_3^-\text{-N}$ levels were found in vegetated soils, maybe reflecting partial nitrification of a greater NH_4^+ stock due to aerated soil conditions and enhanced microbial mineralization activities with vegetation (Robertson & Groffman, 2015; Schimel & Bennett, 2004). $\text{PO}_4\text{-P}$ concentrations were significantly different by site with markedly higher values in the OLA section, while no treatment effects related to vegetation removal were observed. Higher organic matter content in older soils can support

retention of phosphorus in the soil for long periods by fixing phosphorus onto mineral soil particles and reducing its accessibility to plants (Koopmans et al., 2004b; van der Salm et al., 2009). Vegetated soil exhibited marginally higher K^+ levels, reflecting root turnover and litter decomposition (Qiu et al., 2018). Qiu et al. (2018) mentioned that plant growth, biomass accumulation and species diversity have been associated with K^+ availability in soil. When the soil is disturbed, K^+ is readily lost via leaching and erosion (Sparks, 2003). K^+ plays a key role in supporting plant physiology including drought adaptation and enzyme activation (Wang et al., 2013; Marschner, 2012). EC values were higher in vegetated soil, even though EC varied similarly across treatments or sites. Ni et al. (2020) found that, in vegetated soil, ion accumulation in the root zone and greater water retention can result in higher EC levels, where EC variability mainly depends on soil moisture, whereas EC is more sensitive to temperature in bare soil.

Bulk density had a significant difference between sites but not between treatments, with higher bulk density values reported in the OLA section, and comparably lower bulk density levels observed in vegetated soils in both sections. Site-level and treatment level differences in bulk density values may be due to differences in soil compaction, soil organic matter levels, and age (maturity) of landfill covers (Wickramarachchi et al., 2011). Porosity significantly differed between sites or treatment, but slightly higher porosity was observed with vegetated soils, reflecting root growth and increased bioturbation that passively improve pore connectivity and soil structure, thus enhancing soil gas exchange to promote methanotrophic activity (Rachor et al., 2011; Meurer et al., 2020). DOC levels also varied across sites and treatments with elevated DOC concentration observed in the OLA section and vegetated soil in both landfill sections. Availability of root exudates, decaying plant litter and microbial activity likely contributed to elevated DOC concentrations with vegetation presence, supplying labile carbon sources to microbes in the cover soil (Khatoon et al., 2017; Panchal et al., 2022). The more mature cover soil with dense vegetation may have caused the observed higher DOC values at OLA compared to NEA.

Soil pH values were consistent across OLA and NEA sites and treatments, falling within a narrow range from 7.78 to 7.85 pH. This pH window ranged from neutral to alkaline, which has been identified as suitable for methanotrophic activity. Previous studies mentioned that at pH levels between 6.7-8.2, type I methanotrophs are active mostly, whereas Type II methanotrophs are abundant at pH 5-6.5 (Maanoja, 2021). Since the recorded pH range (7.78 -7.85 pH) of both landfill

covers across both treatments in this study remained stable and within the above favorable pH windows for methanotrophs, pH may have a limited role in restricting microbial CH₄ oxidation rates (Scheutz and Kjeldsen, 2004). Soil moisture did not differ significantly between treatments or sites, but vegetated soil consistently showed slightly elevated soil moisture levels at both OLA and NEA, suggesting vegetation improves soil water retention via organic matter input and root induced soil aggregation, thereby improving overall soil structure (Boyle et al., 1989; Angers & Caron, 1998; Lal, 2020). The influence of vegetated treatment on soil temperature variation was minimal in this study, though vegetative covers are known to act as buffers regulating soil temperature via evaporation and shading (Armson, 2012). Furthermore, vegetation intercepts solar radiation and regulates heat absorption at the ground surface (Lozano-Parra et al., 2018). The above-discussed facts affect ecohydrological processes, soil physical and chemical properties, and thermal exchange mechanisms (Lozano-Parra et al., 2018) in landfill soil cover.

Though vegetation treatment improved soil properties in a way to promote CH₄ oxidation activities, it did not affect flux dynamics in this study. At NEA, lower CH₄ fluxes were observed in both treatments due to the presence of an active landfill gas collection system that can reduce CH₄ loading into the near-surface soil layers (Chanton et al., 2011; Duan et al., 2022). Elevated CH₄ fluxes in vegetated plots at the OLA section were observed. The OLA section has a thick grass cover with a dense root system which can potentially hold water, limiting O₂ diffusion and creating anaerobic microsites in shallow soil layers. However, relatively dry soil conditions throughout most of the study period suggest that soil would generally be well aerated. Saturated microenvironments promote CH₄ production through supplying substrates and required soil conditions for microbial CH₄ production processes (Fry et al., 2017; Xiong et al., 2024), emphasizing that there is potential for grass covers to promote CH₄ emissions in vegetated soils in OLA.

Apart from that, elevated CH₄ fluxes in vegetated soil across sites may be explained by the availability of nitrogen species in the soils discussed under soil physical and chemical property improvement with vegetation treatment. CH₄ oxidation is known to be inhibited when NH₄⁺ competes with CH₄ for binding sites in methane monooxygenase enzymes in methanotrophic bacteria (Bédard & Knowles, 1989; Bridgham et al., 2013). Anaerobic methane-oxidizing archaea (ANME) also oxidize CH₄ in anaerobic conditions and may be susceptible to similar competitive

effects for similar substrates, thereby suppressing CH₄ oxidation accordingly (Boetius et al., 2000). Reduced CH₄ oxidation in vegetated soil with higher NH₄⁺ levels may be influenced by this competitive inhibition created by NH₄⁺, along with the composition of localized microbial communities and site-specific environmental conditions (Bodelier & Laanbroek, 2004; Borrel et al., 2011).

The half-year-long vegetation removal trial revealed that vegetation treatment has improved soil properties to some extent, though these benefits might not clearly contribute to higher CH₄ oxidation rates with the interference of site-specific factors like gas extraction systems, CH₄ supply to cover, cover type, and environmental conditions in soil covers. Longer-term vegetation removal experiments in the field or greenhouse experiments in which the CH₄ supply rate is controlled could provide further insight into the role of vegetation for CH₄ oxidation in landfill cover soils.

2.4.3 Meteorological Conditions, Soil Moisture and Soil Temperature as Drivers of CH₄ Flux

In both OLA and NEA sites, soil moisture and CH₄ flux had a positive relationship (Von Fischer & Hedin, 2007), reflecting increased CH₄ production and reduced CH₄ uptake with wetter soil conditions especially in the OLA vegetated plots with higher moisture levels. The association of higher soil moisture with higher CH₄ release from vegetated plots has been observed in previous studies where shrubs increased soil water retention via soil-plant interactions following rainfall (Huang et al., 2009; Liu et al., 2023). Wang et al. (2008), Zhang et al. (2011) and Mensah et al. (2015) also mentioned that soil with dense vegetation improves moisture-holding capacity due to the litter cover, organic matter and fine textured soil. Wang et al. (2024) further explained that alteration of bulk density, porosity, and microclimate by vegetation enhances soil's water holding ability. A significant positive effect between CH₄ flux and precipitation (recorded over the previous 48 hours of sampling) reflected a low oxidation rate ($-0.716 \text{ mg CH}_4 / \text{m}^2 / \text{d}$) recorded in April 2025, coinciding with higher soil moisture (23.78%). Soil water content increased with the intense rainfalls, reducing gas diffusion and O₂ availability, suppressing CH₄ oxidation temporarily until soils desaturate post rain (Zhang et al., 2013; Bian et al., 2020). At high soil moisture, O₂ availability is reduced, limiting CH₄ oxidation and promoting methanogenesis by creating anaerobic soil conditions (Börjesson & Svensson, 1997; Klusman & Dick, 2000), enhancing CH₄ release. Favorable moisture conditions are essential for nutrient and bacterial translocation, which

enables bacteria to access their substrates, results in gas oxidation and stabilization (El-Fadel and Khoury, 2000), but CH₄ oxidation can be maintained even under very low soil moisture contents (Wen et al., 2024). Therefore, CH₄ oxidation in cover soils is generally enhanced under dry soil conditions due to improved O₂ diffusion and microbial respiration (Bian et al., 2020) (Figure 2.7, Table A3). For instance, CH₄ oxidation was greatest in dry months of August and September 2024 (−2.25 and −2.79 mg CH₄ / m² / d, respectively), with minimal precipitation (<12 mm) and the lowest reported soil moisture levels (5.52–8.57%).

In contrast, soil temperature was not a key control on CH₄ flux dynamics in this study, though there was some evidence of a negative correlation with CH₄ flux, which was also reflected in the air temperature relationship. Slightly elevated temperatures were recorded in vegetated plots compared to vegetation removal plots in both OLA (around 24.0 °C) and NEA (around 21.0 °C), but CH₄ flux remained unaffected, indicating that CH₄ flux patterns were more influenced by other factors such as soil moisture and local site conditions (Wu et al., 2010). Abichou et al. (2015) also noticed that soil temperature differences between their vegetated and non-vegetated cover soils were not statistically significant and Ndanga et al. (2015) also stated that vegetation treatment in their study did not show a noticeable effect on soil temperature. Despite this limited association, with the rising air temperatures, higher CH₄ oxidation was evident (i.e., lower CH₄ flux). Lowest CH₄ fluxes were measured with soil temperature ranging from 25–26 °C during warmer months (air temperature increased up to 30.4 °C in September 2024) (Figure 2.7, Table A3). This observation aligns with the similar favorable soil temperature range of 25–35 °C for higher microbial CH₄ oxidation activity found in previous studies (Börjesson, 1997; Börjesson et al., 2004; Scheutz et al., 2009a). Furthermore, Bian et al. (2020) found maximum CH₄ oxidation when air temperature ranged between 10–25 °C which is partly supported by the favorable soil temperature conditions created by heat generation of microbial CH₄ oxidation itself, but beyond this temperature level oxidation is suppressed due to thermal stress on microbes and lower gas retention time due to faster gas diffusion. Notably, CH₄ uptake declined below 10 °C when oxidation is suppressed due to reduced microbial enzyme activity at low temperatures (Scheutz & Kjeldsen, 2004; Spokas & Bogner, 2011).

The effect of other metrological factors was less consistent. Atmospheric pressure had a marginal negative effect, implying its role in driving gas movements in landfill cover soils, though the

observed effect of atmospheric pressure on CH₄ dynamics was not consistent across the sampling period. For instance, the highest atmospheric pressure (99.0 kPa) was recorded in April 2024, coinciding with a weak CH₄ uptake rate ($-0.13 \text{ mg CH}_4 / \text{m}^2 / \text{d}$), illustrating this inverse relationship between atmospheric pressure and the flux (Table A3). A similar observation was reported by Xu et al. (2014), explaining that gas release is suppressed when atmospheric pressure rises and vice versa. Additionally, gas movement in landfills with active gas collection systems will be highly reliant on the negative pressure produced by the gas collecting system, leading to reduced surface emissions that are less susceptible to fluctuations in barometric pressure (Xu et al., 2014). Although pressure changes can alter gas transport mechanisms, including diffusion or advection, their effect is highly site specific, and inconsistent CH₄ fluxes observed with pressure fluctuations may be due to O₂ availability and moisture content (or soil permeability) (when pressure rises it brings more O₂ into the oxidation layers when it is not damp and compacted) (Bian et al. 2020). Furthermore, the same study showed that CH₄ oxidation is limited to near-surface soil layers (<5 cm) where advection becomes the primary gas transport mechanism when atmospheric pressure is low, whereas diffusion-driven gas movements are dominant with higher pressures, allowing more efficient CH₄ oxidation with improved O₂ penetration beyond surface layers.

Wind speed showed no meaningful correlation with CH₄ flux across the study period (the lowest wind speed (11.3 km/h) recorded in July 2024 and January 2025, reaching the highest at 29.1 km/h in January 2025, with no noticeable variations in CH₄ flux (Table A3). These findings suggest that the effect of wind was not a key predictor, likely because the closed chamber method used may isolate fluxes from the wind conditions. Gebert et al. (2011) mentioned that CH₄ concentrations considerably decreased as wind speed increased, indicating wind interferes with the physical movement of gases in the soil; they found that wind-driven pressure changes did not consistently impact CH₄ fluxes, though it has the ability to influence gas transport. Wind-induced gas movements are strongly associated with localized soil permeability and wind-driven pressure conditions (Poulsen and Møldrup, 2006).

The low marginal R² (0.02) of the soil conditions model indicates that soil moisture and soil temperature alone explained 2% of CH₄ flux variability whereas the conditional R² of 0.05 accounting for collar effects slightly improved the overall model performance. Similarly, fixed effects (weather variables) of the weather inclusive model accounted for 18% of variation in CH₄

flux. A 40% (conditional R^2) of flux variation was explained, when random effect (collar variability) was added to the weather inclusive model, reflecting collar level spatial heterogeneity factors explained most of the variation. Given that CH_4 flux from landfill cover soils is also linked to the ability of CH_4 to reach the surface through the underlying cap, this may explain the high degree of spatial variation among the collars. For example, we often observed higher CH_4 emissions from one plot in NEA (i.e., it was often a hotspot); localized conditions like this example make it difficult to detect the specific role of other environmental drivers of flux.

Overall, soil moisture was found as the primary driver of CH_4 flux while precipitation and air temperature indirectly shaped fluxes by controlling moisture and temperature conditions in the cover soil in a way to promote microbial CH_4 oxidation capacity in the soil by maintaining effective O_2 / CH_4 diffusion. Soil temperature had a secondary role, promoting higher oxidation in warmer conditions. Atmospheric pressure and wind speed had no significant impact on CH_4 dynamics.

2.4.4 Seasonal Variation of CH_4 Flux Dynamics

Few studies have monitored CH_4 flux from temperate landfills throughout an entire annual cycle. Monthly CH_4 flux variation observed in the present study showed distinct changes, but vegetation treatment had no significant effect on these trends. In April 2024 (spring), CH_4 fluxes were slightly higher (especially in vegetated soils) with wet (30% in vegetated and 28% in non-vegetated treatment) and cool soil (~ 14.1 – 15.8 °C) due to wetter soil conditions with snowmelt, which limits O_2 supply and suppresses CH_4 oxidation. CH_4 flux generally declined from spring to summer, aligning with rising microbial oxidation rates due to the influence of progressively drier and warmer soil conditions (Maurice and Lagerkvist, 2003) (Table A2). Soil moisture declined from $\sim 30\%$ in spring to below 10% in late summer and early fall, causing the highest CH_4 uptake rates in September 2024. These findings fit with the previous findings of the observed higher CH_4 uptake during late summer to early fall under similar soil conditions (Borken et al., 2006; Rachor et al., 2013) due to greater CH_4 uptake associated with dry and warm soil conditions as it improves gas diffusivity and forms suitable moisture conditions for methanotrophic activity (Whalen et al., 1990; Börjesson & Svensson, 1997). Then, from late fall to winter, CH_4 uptake weakened steadily with the return of cooler and wetter soil conditions, while in January 2025, CH_4 fluxes continued to be suppressed due to continued cold (7.9 – 9.6 °C) and moist (23.5–27.5%) soils (Figure A2 &

A4). During winter, suppression of methanotrophic activities can occur due to inadequate supply of O₂ along with soil water saturation and low soil temperatures in the cover soil (Scheutz & Kjeldsen, 2004; Einola et al., 2007). On the other hand, snow accumulation on the landfill cover was observed during December 2024 and January 2025, and this likely insulated the soil while restricting gas diffusion or exchange and CH₄ consumption (Maurice & Lagerkvist, 2003). Site access restrictions limited measurements in February and March 2025, but CH₄ uptake was slightly reduced by April 2025 despite higher soil moisture (38.0 and 37.2% in vegetated and non-vegetated plots, respectively) and moderately warmer soil temperatures (13.2 °C at vegetated, 16.0 °C at non-vegetated) (Figure 2.6; A2; A4). CH₄ oxidation was likely again limited by low O₂ availability with excessive water in soil pores from rapid snowmelt and rainfall (Whalen et al., 1990; Börjesson & Svensson, 1997).

Overall, measurements over a full annual cycle indicate that CH₄ oxidation peaked in late summer to early fall under warm and dry soil conditions, while CH₄ fluxes were higher in winter and spring, corresponding to cold and wet or saturated soil conditions, with vegetation playing a minimal role in the overall trend.

2.4.5 Quantification of CH₄ Oxidation Rates in Landfill Cover Soil

CH₄ Oxidation rates were higher at the NEA site (veg: 0.0156 µg CH₄/g /d and non-veg:0.0138 µg CH₄/g /d) relative to the lower rates at OLA (veg: 0.0080 and non-veg:0.0126 µg CH₄/g /d), emphasizing CH₄ oxidation capacities of per gram of cover soil of landfill caps depending on maturity, soil properties, microbial activities, CH₄ supply and distribution with the cover and available site infrastructure, rather than presences of vegetation (Chanton et al., 2011).

NEA has an active landfill gas collection system in place that should intercept high loads of CH₄ before they reach aerobic near-surface soil layers. This lower supply could limit methanotroph activity. However, since the NEA section is relatively younger (capped in 2016), a considerable level of CH₄ production can be expected. Despite the gas collection system, the average CH₄ flux at NEA was several orders of magnitude higher than OLA due to several very high CH₄ emissions (hotspots). Hotspots in capped landfills are associated with abundant methanotroph communities (Roy et al. 2025), and this likely contributed to the higher measured CH₄ oxidation at NEA. Moreover, this further highlights the importance of introducing or maintaining passive CH₄

oxidation applications along with active gas collection systems to further eliminate uncaptured CH₄ emissions (Ahooghalandari & Cabral, 2017; de Almeida et al., 2024).

Similarly, the lower oxidation rates recorded in the OLA section may be due to a diminished supply of CH₄ to the cover as CH₄ production has gradually declined over time since it was capped in 2001, despite no active gas collection system currently in place. At lower CH₄ loading rates, the methanotrophic community is also limited (Chanton et al., 2011), and this may also contribute to observed reduced oxidation rates in OLA's cap.

Vegetation treatment across both sites played a secondary role in CH₄ oxidation relative to site-specific controls such as presence of gas collection systems, soil physical conditions, O₂ and CH₄ loading to the cover, though plant roots enhance soil aeration and provide labile carbon sources for methanotrophic communities (van Verseveld and Gebert, 2020; Reddy et al., 2021).

2.4.6 Conclusion

In conclusion, CH₄ dynamics across the studied capped landfill sections were primarily controlled by soil moisture and site-specific factors other than by vegetation alone. Even though vegetation had no significant impact on CH₄ flux, its presence has changed (or improved) soil properties, showing higher NH₄⁺-N, DOC, EC, NO₃-N, K⁺, porosity, soil moisture, and lower bulk density, all conditions that support enhanced microbial activity (Cameron et al., 2013). Therefore, although vegetation removal did not significantly affect CH₄ flux, the presence of vegetation on capped landfills creates soil conditions that are favorable to CH₄ oxidation (Abichou et al., 2011). Soil moisture emerged as the strongest driver of CH₄ flux with suppression of CH₄ oxidation under wetter conditions, while soil temperature was not a significant predictor, although the observed negative trend between CH₄ flux and temperature indicates more oxidation with the rising temperature in the soil. Further, the greatest CH₄ oxidation (lowest CH₄ flux) was recorded around 25–30 °C based on the field data, aligning with literature (Scheutz et al., 2009a). These trends illustrate that when environmental conditions are favorable, landfill cover soil serves as an efficient CH₄ sink (Borken et al., 2006).

Observed trends reflect a clear seasonal shift in CH₄ dynamics, higher CH₄ uptake recorded in late summer to early fall, coinciding with falling soil moisture levels below 10 % and rising soil temperatures above 25 °C (Börjesson & Svensson, 1997), whereas lower uptake in early spring

and winter with wetter, cooler soil conditions (Maurice & Lagerkvist, 2003). Precipitation and air temperature turned out to be the significant weather variables, regulating CH₄ flux behaviors. Atmospheric pressure had an inconsistent effect on CH₄ flux, and wind speed was also found to be insignificant, reflecting gas dynamics mainly regulated by below-ground gas movements rather than surface-level air turbulence (Bian et al., 2020).

Notably, site-level CH₄ flux differences were observed with relatively higher CH₄ oxidation rates at NEA with an active landfill gas collection system, reduce supplying extremely high CH₄ concentrations to the cover soils, promoting efficient CH₄ oxidation rates per gram of cover soil (Abichou et al., 2006). These findings underscore how cover soil properties (or material), gas residence time, adequate substrate supply, and site infrastructure affect gas transport and CH₄ oxidation in the landfill covers while highlighting how passive CH₄ oxidation works even with gas collection systems further abate fugitive CH₄ emissions.

The substantial plot-level variations in the studied models emphasize the need to consider the highly distributed, large amount of spatial data to capture microclimatic conditions, microtopography, root distribution, and localized soil heterogeneity, which modulates oxidation capacity to accurately study landfill CH₄ flux dynamics. Although hotspot emissions were rare (2.4% of total fluxes), due to their magnitudes, mean flux values were skewed. Therefore, excluding these anomalies is important when studying drivers of CH₄ oxidation and quantifying CH₄ oxidation rates in cover soils. Usually, extreme or higher CH₄ releases can be observed in areas with structural defects such as cracks, fissures, pipe interfaces, and root holes (Scheutz et al., 2011b). CH₄ uptake was dominant when high emission values were taken out, further illustrating that landfill cover soils work as a CH₄ sink when leaks from the cap are minimized.

Chapter 3

Assessing CH₄ Oxidation Responses to Controlled Soil Moisture and Temperature: A Landfill Soil Incubation Study

3.1 Introduction

Landfill gas generation continues even after landfill closure, making methane (CH₄) emissions a long-term issue (Bruce et al., 2018). Gas collection systems are commonly used to recover or burn landfill gases to control landfill gas emissions (Scheutz et al., 2004; EPA, 2015). Even with the presence of landfill gas collection systems in large landfills, some CH₄ escapes to the atmosphere or moves through the landfill cover soil (Abushammala et al., 2014). Conventional gas recovery systems approximately trap 50–90% of landfill gas emissions (Augenstein and Pacey, 1996). Landfill gas mitigation approaches that included microbial CH₄ oxidation processes have been considered as effective methods of controlling fugitive CH₄ emissions from landfill cover (Abushammala et al., 2013; Huber-Humer et al., 2008).

Soil uptake of atmospheric CH₄ due to microbial oxidation happens in soils in natural systems with activities of several groups of bacteria that demand oxygen (O₂) for the oxidation process. This microbial oxidation process can be applied to control uncaptured or fugitive CH₄ emissions from landfill cover with or without a gas recovery system on the landfill site (Abushammala et al., 2013; Huber-Humer et al., 2008). Jugnia et al. (2008) stated that field studies and laboratory experiments indicate that the average percentage oxidation capacity in landfill soil is nearly 40%, varying between 0 to 100%. According to the Intergovernmental Panel on Climate Change (IPCC) guidelines, the recommended landfill cover soil CH₄ oxidation value of between 0 to 10% of landfill emissions should be applied for national inventories (IPCC, 2006).

Seasonal variation, soil texture, soil organic carbon content, soil moisture, soil temperature, nutrients, pH, and O₂ and CH₄ concentrations in landfills determine the CH₄ oxidation capacity of landfill cover soils (Börjesson et al., 2001; Wilshusen et al., 2004; Albanna et al., 2007; Einola et al., 2007; Tecle et al., 2008; Abushammala et al., 2013). Temperature is an important variable determining CH₄ oxidation rates in laboratory incubation studies (Börjesson et al., 2004) since soil temperature variation affects the diversity and activity of CH₄-oxidizing consortia (Mohanty et al.,

2007). Scheutz and Kjeldsen (2004) observed that CH₄ oxidation increased rapidly with temperature increases from 2 to 25 °C, and the highest oxidation rate was reported at 30 °C, whereas oxidation declined at 40 °C. Microbial CH₄ consumption is temperature-dependent, with the maximum temperature occurring between 15 to 30°C (Mor et al., 2006). Soil moisture is also a main controlling factor of CH₄ oxidation activities of microorganisms (Jugnia et al., 2008), and nearly 78% of the variation of CH₄ oxidation rates is determined by soil moisture variations (Castro et al., 1994). Availability of soil pore spaces for gas exchange is changed under different moisture levels, thus making this a significant factor for CH₄ oxidation rates in landfill cover soil (Hilger and Humer 2003; Chapter 2). Early studies reported that optimum volumetric soil moisture for CH₄ uptake in landfill cover soils was 10–20% (Borjesson et al., 1997; Whalen and Reeburgh, 1996), while maximum CH₄ oxidation rate was recorded at volumetric moisture content of 20-35% in another study (Bender and Conrad, 1995). At lower soil moisture levels, CH₄ oxidizing capacity decreases due to microorganisms' physiological response to water stress (Czepiel et al., 1996).

Mosier and Parkin (2006) stated that potential biological demand and gas diffusion are the main drivers controlling CH₄ oxidation, whereas CH₄ diffusion rates are controlled by physical environmental factors, while biological demand is influenced by both chemical and physical environments. CH₄ oxidizing efficiency and capacity differ broadly in landfill cover soil, as mentioned in the literature. Albanna et al. (2007) demonstrated that CH₄ oxidation was positively impacted by soil moisture and nutrient availability; with added nutrients at 30% moisture content, oxidation increased from 75% to 81%, emphasizing the interactive role of moisture and nutrient availability.

Vegetation growing on landfill cover provides O₂ and organic substrates to promote soil-mediated CH₄ oxidation in several ways (Reay et al., 2005; Wang et al., 2008). Plant roots can improve O₂ diffusion into the soil cover and enhance O₂ availability in the root zone, thereby supporting activities of methanotrophs (Abichou et al., 2006). On the other hand, root exudates and plant litter may supply carbon sources and stimulate the growth of microbial communities (Chanton et al. 1993; Whiting & Chanton, 1993; Bellisario et al., 1999). Thus, the presence of vegetation is expected to alter the rate of CH₄ oxidation in the cover soil by influencing soil physical and chemical properties and microbial community biomass and composition.

The quantification of CH₄ oxidation in landfill soil is a major uncertainty in estimating CH₄ emissions from landfills at the national or international level (Bogner and Spokas, 1993). Constantly shifting environmental conditions further intensify efforts to evaluate the soil oxidation capacity (Spokas & Bogner, 2011). The exact quantity of CH₄ being oxidized by methanotrophic bacteria in landfill cover remains uncertain. Notably, in older landfill sites, enhancing the CH₄ consumption capacity of landfill soil has been identified as a promising alternative method to control CH₄ emissions since gas concentrations are insufficient for flaring or recovery (Röwer et al., 2011). As mentioned in Kightley et al (1995), further research for measuring microbial CH₄ oxidation in landfill cover soil is required, particularly in such aged systems where conventional gas management is unfeasible. The lack of a standard methodology to calculate oxidation rates, limits data availability regarding CH₄ oxidation capacities of landfill cover soil (Abushammala et al., 2014). Thus, this chapter reports on a laboratory incubation study conducted for landfill cover soil from vegetated and non-vegetated treatments under controlled soil moisture and soil temperature conditions to assess CH₄ oxidation capacities and optimal environmental conditions for maximum oxidation. The objectives of this study were to: i) Quantify CH₄ oxidation rates in landfill cover soils collected from vegetated and non-vegetated treatments applied for 6 months before soil collection under controlled soil temperature and soil moisture conditions; ii) Determine the optimal moisture and temperature levels for CH₄ oxidizing activity in landfill cover soils under vegetated and non-vegetated treatments. Findings of this study will provide critical insight into CH₄ dynamics in landfill cover and help refine the landfill cover system to effectively reduce CH₄ emissions.

3.2 Materials and Methods

3.2.1 Study Site and Experimental Design

Soil samples from the top 10 cm of the soil cap were collected from vegetated and non-vegetated plots in two capped landfill sections: Original Landfill Area (OLA) and Northern Expansion Area (NEA) at a Landfill in Southern Ontario (43°26' N 80°34' W). OLA was opened in 1972 and closed and capped in 2001 while NEA operated from 1994 until its capping in 2016. An active gas collection system is in place at only the NEA section whereas both sections have leachate collection systems. OLA is capped with 0.9 m of till and 0.1 m of topsoil, while NEA has a 0.9 m

low-permeability-clay cap with 0.15 m of vegetated topsoil (Landfill Management Annual Reporting, anonymity requested). Six study locations were selected from each section; half were at the base of the landfill slope, and the remaining half were positioned at the center sloping area, with 20 m between each study location at each site. Each study area had paired 1 × 1 m plots where vegetated and non-vegetated treatments were applied. Vegetation removal was done manually and maintained regularly from April 2024 to October 2024. In late September 2024, soil samples were collected for the incubation study. Homogenized and composite soil samples from vegetated and non-vegetated treatments of both OLA and NEA sections were prepared by removing large debris, thoroughly mixing, followed by sieving at 4 mm.

The study design was a full factorial experiment considering four levels of soil moisture (20%, 40%, 60%, 80%), four levels of temperature (5°C, 15°C, 25°C, 35°C), two levels of vegetation treatment (vegetated and non-vegetated) and two sites (NEA and OLA). Each combination was incubated in triplicate; a total of 96 incubation jars were used. Deionized (not sterilized) water was added to adjust soil moisture levels.

Incubation under each temperature level was completed separately, and jars with soils from the vegetated and non-vegetated treatments from both study sites were prepared at the desired moisture contents as described below, with samples incubated for three days before measurements. CH₄ oxidation rate was then determined by injecting 50 ml of 2500 ppm CH₄ standard into the jar headspace, and gas samples were collected immediately after the injection (T₀) and 5 minutes following the injection (T₁) and three injection rounds conducted to obtain triplicate measurements with the same initial CH₄ concentration in the headspace maintained during each injection round. Jars were taken out of the incubators immediately before the injection and remained outside at room temperature until all injection rounds were finished. The CH₄ concentrations were analyzed on a gas chromatograph (SCION-456-GC in the Schiff Lab in the Department of Earth and Environmental Science, University of Waterloo), where a total of 576 CH₄ oxidation rates were obtained testing collected gas samples.

The CH₄ oxidation rate was calculated based on the change in concentration over time using equation 3.1 (Lam, 2025):

$$F_{\text{gas}} = \frac{(C_{T_1} - C_{T_0}) \cdot V_{\text{headspace}}}{R \cdot T \cdot \Delta t \cdot M_{\text{soil}}} \quad (3.1)$$

Where F_{gas} is the rate of CH_4 oxidation, C_{T0} is the concentration of CH_4 immediately after standard injection and C_{T1} is the concentration after 5 minutes of the injection, V is the headspace volume (L), Δt is the sampling time interval (5 minutes), M_{soil} is the incubated soil mass in the jar (g), T is the incubation temperature (K), and R ($0.0821 \text{ L} \cdot \text{atm} \cdot \text{mol}^{-1} \cdot \text{K}^{-1}$) is the ideal gas constant.

3.2.2 Analysis of Soil Physical Properties and Experimental Set-up

Air-dried soil was placed in labelled incubation jars, weighed, and then adjusted to the desired moisture content based on soil physical properties and residual moisture content. Water-Filled Pore Space (WFPS) is the percentage of soil pore space filled with water, accounting for both soil moisture and porosity of soil. Experimental moisture levels were calculated based on WFPS as described by Lam (2025), and soil moisture adjusted to 20%, 40%, 60%, and 80%. Soil temperatures were set for 5°C, 15°C, 25°C, and 35°C, and soil samples were kept open in incubators for 3 days.

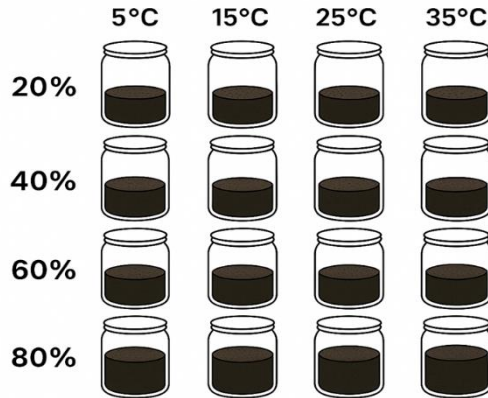


Figure 3.1: Experimental Set-up Used for Incubation Study with Different Soil Moisture and Soil Temperature Levels.

Soil was incubated in narrow-mouth glass septa bottles to fill half of the volume of the bottles. The mass of soil to be incubated (M) was determined based on the half volume of the glass bottle (V) and the bulk density (BD) of soil, using the equation below,

$$M = V \times BD \quad (3.2)$$

Bulk density

Bulk density (BD) was measured by oven-drying a known volume of soil at 105 °C until it reached a constant weight. Dry bulk density was then calculated as the ratio between dried soil mass and volume (Panagos et al., 2024).

$$\text{Bulk Density (g/cm}^3\text{)} = \frac{\text{Dry Mass of Soil (g)}}{\text{Total Soil Volume (cm}^3\text{)}} \quad (3.3)$$

Porosity

Porosity was determined using a known volume of oven-dried (V_s) soil. Water (V_l) was added to the pore spaces until saturated. Total sample volume was also measured (V). Porosity(ϕ) is determined by the proportion of pore space volume to the total volume (Flint & Flint, 2002).

$$V = V_s + V_l \quad (3.4)$$

$$\phi = \frac{V_l}{V} \quad (3.5)$$

Soil Moisture Adjustment for Incubation Setup

The volume of water required to achieve the desired WFPS (V_w) in the incubated soil (M) was calculated using bulk density (BD) and porosity (ϕ) as shown in equation 3.6 (Lam, 2025).

$$V_w = \left[\frac{(\text{Desired WFPS} \div 100) \times \phi}{BD} \right] \times M \quad (3.6)$$

The existing moisture content in the air-dried soil was quantified before water addition (to adjust moisture content to obtain desired WFPS levels) to prevent exceeding target moisture levels. The following steps were used to carry out measurements. The existing moisture content (MC_g) of soil

was determined by the difference between air-dried (M_a) and oven-dried (M_o) soil mass as shown below,

$$MC_g = M_a - M_o \quad (3.7)$$

Water was then added to the incubation jars to reach the target moisture contents. Further details of all calculations used to determine water additions for adjustment to desired WFPS levels are given in the Appendix.

3.2.5 Statistical Analysis of Data

Prior to statistical analysis, quality control was applied to the GC data; observed outliers in incubation data such as unusual peaks, inconsistent concentration readings, or injection errors were removed. Samples that met the required standard (decline in concentrations over time) were used in the analysis (from a total of 576 CH_4 consumption rates, approximately 184 data remained for further analysis).

Statistical analyses were carried out using R version 4.3.1 (R Core Team, 2023) and RStudio version 2023 (RStudio Team, 2023). A series of linear models was conducted by using the `lm()` function in the base R to evaluate the effect of the soil moisture, soil temperature, and treatment on CH_4 consumption rates ($\mu g CH_4 / g / d$) (CH_4 oxidation rates). Shapiro-Wilk test and Q-Q plots were used to assess the residual normality, and the significance was determined at $\alpha = 0.05$. The `ggplot2` package (Wickham et al., 2016) in R was used for the visualization of gathered data. Due to the loss of some replicates during quality control, the full four-way ANOVA could not be assessed. As our interest was largely in the effect of vegetation removal on CH_4 consumption and whether this altered the rate of CH_4 consumption under different moisture and temperature conditions, we started with an initial model considering treatment, soil moisture and soil temperature and all two-way and three-way interactions; this full factorial model yielded no significant terms ($p > 0.25$ for all), indicated potential overfitting. Based on the results of this initial model, a systematic model simplification method was performed, eliminating non-significant interaction terms with the highest p-values one-by-one using Type III ANOVA to simplify the model. As no significant interactions were observed, we present here the results from a more focused model including only the main effects of soil moisture, soil temperature, and treatment. When significant effects were present, `emmeans` package was used to obtain estimated marginal

means, then Tukey's HSD test (Lenth et al., 2018) was used with Tukey-adjusted pairwise comparisons to control multiple comparisons.

3.3 Results

3.3.1 Summary of Soil Physical Properties

Table 3.1 shows the variability of soil physical characteristics and CH₄ consumption rates by treatment and locations. The highest average CH₄ consumption rate was observed at non-vegetated soils, followed by vegetated soils, though this difference lacked statistical significance (Table A4).

Bulk density values in both treatments in the OLA section were similar, but in NEA, vegetated soil exhibited the greatest bulk density, and the lowest in its non-vegetated soils. OLA shows similar porosity values across treatments, whereas porosity was marginally higher in vegetated soils compared to non-vegetated soil in the NEA section (Table 3.1), thus showing inconsistent soil properties across treatments in prepared soil samples for the incubation study.

Table 3.1: Soil Physical Properties and Average CH₄ Consumption Rates by Site and Treatment

Variable	OLA Section		NEA Section		Both Sections	
	Veg. Treatment	Non-Veg. Treatment	Veg. Treatment	Non-Veg. Treatment	Veg. Treatment	Non-Veg. Treatment
Porosity	0.32 (SD = 0.002)	0.33 (SD = 0.006)	0.37 (SD = 0.005)	0.33 (SD = 0.006)	0.34 (SD=0.026)	0.33 (SD=0.006)
Bulk Density (g/cm ³)	0.75 (SD = 0.010)	0.75 (SD = 0.007)	0.79 (SD = 0.011)	0.72 (SD = 0.007)	0.77 (SD=0.026)	0.735 (SD=0.021)
Avg. CH ₄ Consumption (μg CH ₄ /g /d)	17.7 (SD = 8.25)	27.0 (SD = 15)	25.2 (SD = 15)	20.6 (SD = 10.51)	21.4 (SD = 12.5)	23.8 (SD = 13.2)

3.3.1 The Effect of Vegetation Removal on CH₄ Consumption Under Varying Soil Moistures and Temperatures

CH₄ consumption rates varied between 1.5 to 57.4 μg CH₄ /g /d across all temperature and moisture treatments. Although the effect of vegetation removal differed between NEA and OLA (Table 3.1), variation under each treatment was high within sites. As our main interest was to evaluate the impact of vegetation in capped landfills in general, we combined data from both sites for further analysis as described in the methods (Section 3.2.5).

The final linear model retained only soil moisture, soil temperature, and treatment as the main effects, and both soil moisture and temperature were statistically significant, whereas treatment was not found as a significant predictor of CH₄ consumption (Table 3.2). Thus, CH₄ oxidation was primarily controlled by soil moisture and soil temperature conditions during incubation.

Table 3.2: Type III ANOVA Results for CH₄ Consumption Model Including Treatment, Temperature and Moisture

Effect	Sum Sq	Df	F value	Pr(>F)
Treatment	619.7	1	1.1581	0.2865
Temperature	11705.5	3	7.2923	0.0003 ***
Moisture	27441.3	3	17.0953	<0.0001 ***

CH₄ Consumption Rates by Moisture Conditions

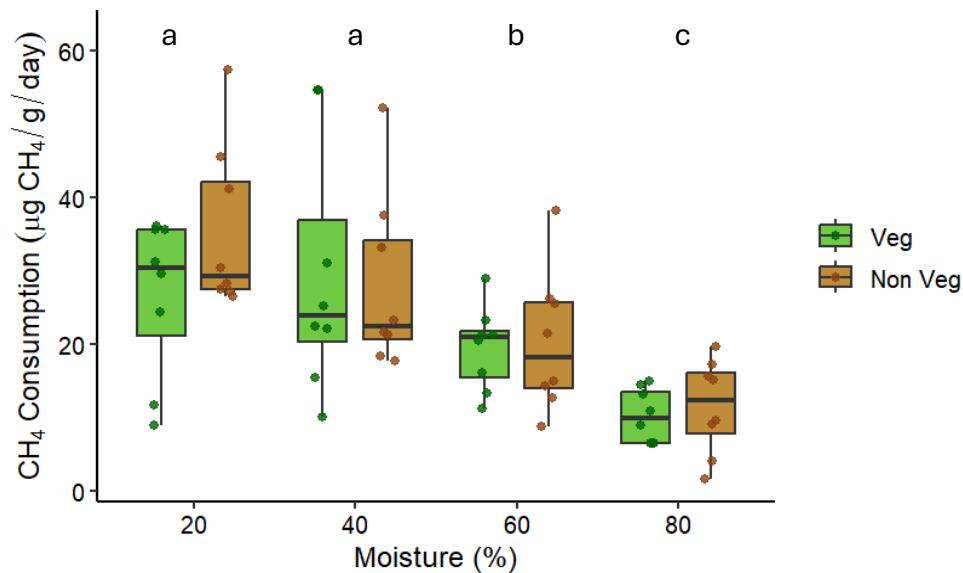


Figure 3.2: CH₄ Consumption Rates in Vegetated and Non-Vegetated Soils under Different Moisture Conditions.

The highest CH_4 consumption was recorded at 20-40% WFPS with no significant difference between vegetated and non-vegetated treatments (Figure 3.2). Notably, oxidation rates significantly declined at the 60% moisture level in both vegetation treatment types compared to the 20% and 40% levels. The lowest CH_4 consumption rates were observed in both treatments at 80%, reflecting a statistically significant reduction from all previously tested moisture levels. There was no significant difference between treatments at 20% and 40% moisture levels, though mean oxidation rates differed with moisture level changes, with the highest rates at 20% WFPS for soils from non-vegetated plots and 40% WFPS for vegetated plots.

CH_4 Consumption Rates by Temperature Conditions

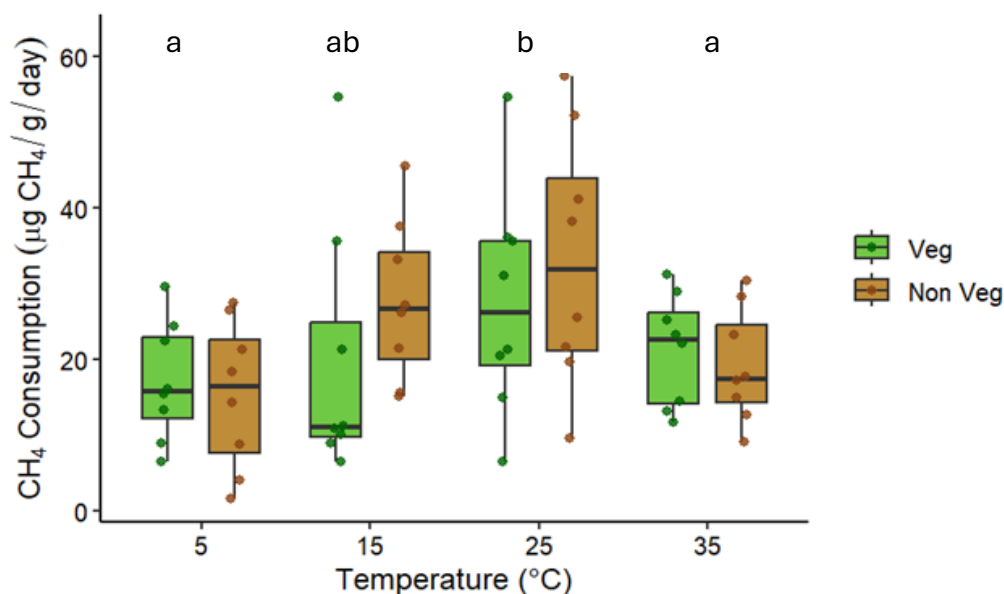


Figure 3.3: CH_4 Consumption Rates in Vegetated and Non-Vegetated Soils under Different Temperature Conditions.

As shown in Figure 3.3, the highest CH_4 oxidation rates were recorded at 25 °C in both vegetated and non-vegetated soil, which were significantly greater than oxidation rates at 5 °C and 35 °C. The lowest CH_4 oxidation was measured at 5 °C, followed by 35 °C. An intermediate effect was observed at 15 °C, and oxidation rates were moderate in both vegetated and non-vegetated soil, indicating no significant difference with other temperature levels. While the treatment effect was

not statistically significant, the ANOVA results, non-vegetated soil generally had higher oxidation rates, particularly at 15 and 25 °C.

3.3.2 Identification of Optimal Conditions for CH₄ Oxidation

Tukey post hoc test confirmed that these values were significantly greater than those at 5 °C and 35 °C, establishing 25°C as the optimal temperature level for CH₄ oxidation for both treatment types (Figure 3.3). Notably, the highest CH₄ oxidation rate occurred at 40% moisture in vegetated soil (29.4 µg CH₄ /g /d), whereas the highest CH₄ oxidation in non-vegetated soil was observed at 20% moisture (35.4 µg CH₄ /g /d). According to Tukey's post hoc analysis, there was no significant difference between CH₄ oxidation rates occurring at 20% and 40% moisture within either treatment, implying a common optimal moisture content between 20 - 40 % across treatments (Figure 3.3; Table A4). Together, the results indicate that the greatest rates of CH₄ oxidation will occur in soils of capped landfills when the temperature is between 15 and 25 °C and the moisture content is 20 - 40% WFPS.

3.4 Discussion

3.4.1 Impact of Vegetation Presence on Soil Physical Properties on CH₄ Oxidation Rates

Vegetation can indirectly enhance CH₄ oxidation by modifying soil structure, aeration, and microbial activity (Abichou et al., 2015). Soils sampled directly from the studied vegetation removal experiment also followed this pattern (Chapter 2), with soils from the vegetation removal plot having significantly lower bulk density, higher NH₄⁺-N, DOC, EC, NO₃-N, K⁺, porosity, and soil moisture. In contrast, the measured bulk density and porosity of samples used in this incubation experiment did not have any significant difference between vegetated and non-vegetated soil. Combining plot-level soil samples to prepare composite samples, root removal, and sieving to get homogenized samples likely changed soil structure, pore networks, and disrupted soil macro-aggregates that are known to support in situ CH₄ oxidation and gas movements (Abichou et al., 2011; Prajapati and Jacinthe, 2014). Abichou et al. (2011) and Prajapati and Jacinthe (2014) further explained that field oxidation rates measured on homogenized soils underestimate those from the intact soil samples, as it disrupts preferential gas transport pathways and the natural structure of the soil matrix important for methanotrophic activity. Moreover,

incubation-based CH₄ oxidation reflects more on the influence of controlled soil moisture and soil temperatures (Huber-Humer et al., 2009) than on the gas transport or soil physical properties (Prajapati and Jacinthe, 2014). This study lacked detailed soil property characterization of organic matter content, nutrient availability, and microbial abundance to further explain treatment effects and oxidation potential of soil. Additionally, use of intact cores or non-destructive samples would mimic natural field conditions (Gebert et al., 2003; Scheutz et al., 2009a; Abichou et al., 2015) and should be considered for future studies (sampling activities such as soil core and soil gas collection were not allowed below 10 cm in the studied landfill caps, so we did not carry out any sampling activities beyond that depth to avoid damaging site's cap systems).

The above-mentioned factors likely contributed to the lack of a significant vegetation treatment effect on CH₄ oxidation rates. Ndanga et al. (2015), also found limited vegetation effect on CH₄ oxidation rates compared to bare soil. However, there was also no significant difference in CH₄ flux between vegetated and non-vegetated plots in the field study (Chapter 2), suggesting that other factors are more important than vegetation presence for controlling CH₄ cycling in landfill cover soils. Further, as there was no significant interaction between treatment and soil moisture or temperature in the incubation, there is little evidence that the vegetation removal treatment substantially altered the function of the CH₄-oxidizing microbial community. This could be due to the relatively short period of vegetation removal (i.e., 6 months). Higher CH₄ oxidation activity was observed along with lower surface emissions in older landfill caps (7-16 years) compared to the active landfill section due to improved oxidation capacity as the cover matures (Kong et al., 2013). Vegetation may enhance O₂ diffusion and release root exudates, which support sustaining higher oxidation capacities and microbial communities over a long time (Kong et al., 2013; Ding et al., 2005; Bohn et al., 2011). When the incubation temperature was at ≤ 30 °C, Type I (Methylobacter) was dominant, while it shifted to Type II (Methylocaldum-thermotolerant) when the temperature rose above ≥ 40 °C (Eshinimaev et al., 2004; Bodrossy et al., 1997). Similarly, Type I methanotrophs dominate at lower temperatures (5-10°C), whereas the presence of Type II methanotrophs increased under CH₄-rich, warmer, and O₂ deficient micro-environments (Horz et al., 2005; Mohanty et al., 2007). Therefore, identification of shifts in microbial communities with soil temperature variations in CH₄ oxidation experiments has become necessary (Kallistova et al., 2007; Reddy et al., 2019).

In this study, non-vegetated soil exhibited effective CH₄ oxidation under drier and warmer (20%, 25 °C) soil conditions, likely due to better O₂ diffusion and efficient microbial utilization of O₂ (Gebert et al., 2003; Scheutz et al., 2009). Meanwhile, vegetated soil performed greater CH₄ oxidation with moist (40%) and warm (25 °C) conditions, coincide with favorable conditions for microbial optima zones (Whalen et al., 1990; Börjesson et al., 2004).

3.4.2 CH₄ Consumption Response to Soil Moisture and Soil Temperature

The incubation study revealed that soil temperature and soil moisture content were the primary drivers of CH₄ oxidation. Both are known to influence the activity of methanotrophic communities and gas diffusivity (Scheutz et al., 2009a; Spokas & Bogner, 2011). Low CH₄ consumption was recorded at 5 °C in both vegetated (17.1 µg CH₄ /g /d) and non-vegetated (15.3 µg CH₄ /g /d) soils, reflecting reduced microbial enzyme activity at low temperatures (Scheutz & Kjeldsen, 2004; Spokas & Bogner, 2011). CH₄ consumption gradually rose when the temperature increased, and a moderate consumption was observed at 15 °C, reflecting improved methanotrophic activity. At 25 °C, maximum CH₄ consumption rates (veg: 27.6 µg CH₄ /g /d, non-veg: 33.1 µg CH₄ /g /d) were obtained, aligning with the similar optimal temperature range of 25–35 °C for higher enhanced oxidation activity found in previous studies (Börjesson, 1997; Börjesson et al., 2004; Scheutz et al., 2009a). CH₄ consumption rates were reduced at 35 °C (veg: 21.2 µg CH₄ /g /d and non-veg: 19.2 µg CH₄ /g /d), reflecting the suppressed microbial activity under temperatures above the mesophilic range (Christophersen et al., 2000). Czepiel et al. (1996) and Boeckx et al. (1996) also observed that microbial activity is negatively affected after a certain thermal threshold, resulting in lower oxidation rates.

The greatest rate of CH₄ consumption was exhibited around 20–40% with vegetated soil having the highest CH₄ uptake at 40% (29.4 µg CH₄ /g /d) and non-vegetated soil peaking at 20% (35.4 µg CH₄ /g /d). Despite the peak oxidation moisture content differing by treatment, rates were statistically similar at 20% and 40% in both treatments. These findings are consistent with previous studies where moderate soil moisture improved CH₄ consumption, which enhanced microbial activity and maintained adequate air-filled-pore spaces for gas exchange (Whalen et al., 1990; Ball et al., 2013). On the other hand, CH₄ consumption rates declined drastically at high moisture levels of 60–80% (veg: 19.5-10. µg CH₄ /g /d, non-veg: 20.2-11.5 µg CH₄ /g /d). At higher moisture

contents, water-filled pores restrict gas movements and availability of CH₄ and O₂ to methanotrophs to function (Scheutz et al., 2003; Fan et al., 2014).

Based on the incubation results, the optimal conditions for higher CH₄ oxidation were at 20-40% soil moisture and 25 °C temperature for both vegetation treatments, with consistent oxidation rates under aerobic incubation conditions. These conditions could be targeted in the development of CH₄ biological control systems for CH₄ emissions from capped landfills for optimal performance.

3.4.3 Conclusion

The incubation study explored the optimal soil moisture and soil temperature conditions for maximum CH₄ oxidation in landfill cover soils under vegetated and non-vegetated treatments. Results revealed that CH₄ oxidation was primarily governed by soil moisture and soil temperature, as vegetation treatment did not show a significant impact alone or for altering the response to temperature and moisture under laboratory-controlled conditions. Use of intact cores or non-destructive samples would mimic natural field conditions (Gebert et al., 2003; Scheutz et al., 2009; Abichou et al., 2015) and soil characteristics related to vegetated soil will remain intact in such an intact core experimental set up.

The most favorable moisture content for CH₄ oxidation fell between 20–40% where vegetated soil achieved peak oxidation at 40% and non-vegetated soil at 20%. Nevertheless, no significant difference was noted between the treatments within this moisture window, suggesting enhanced methanotrophic activity within this moisture threshold. Soil temperature also played a key role. Oxidation rates gradually increased from 5 °C to 25 °C, peaked at 25 °C, and then decreased at 35 °C, consistent with mesophilic temperature optimum and limited enzymatic activity beyond this range (Börjesson & Svensson, 1997; Christophersen et al., 2000).

To conclude, optimal conditions for CH₄ oxidation in landfill cover soils were determined as 25 °C and 20–40% soil moisture, independent of vegetation. Such evidence offers valuable insight for landfill management to improve microbial CH₄ mitigation strategies.

Chapter 4

Summary, Conclusions and Recommendations

4.1 Summary of Main Findings

This study examined how vegetated and non-vegetated soil covers, soil conditions, and meteorological variables affect CH₄ dynamics across two capped landfill sections in the landfill in Southern Ontario, incorporating both year-long field observations and controlled incubations. Soil moisture emerged as the key driver of CH₄ flux in the field study, and soil temperature remained as a secondary factor. Vegetation presence on soil covers did not alone alter CH₄ dynamics significantly under field or laboratory conditions, but vegetation led to some improvements on soil chemical and physical conditions (lower bulk density, lower PO₄-P, higher porosity, higher DOC, and higher K⁺, NH₄⁺-N, and NO₃-N). Moreover, the potential influence of site-specific landfill cover soil structure, composition, age of the landfill, and infrastructure available, such as gas collection systems, likely affected how much CH₄ was available in the landfill cover, gas transport, and flux behavior, even though we did not directly examine these influential factors in detail in this study.

Field observations revealed that CH₄ flux showed a seasonal trend where the highest CH₄ oxidation, as indicated by the lowest CH₄ flux, occurred between late summer to early fall when the soil was dry (<10% moisture) and warm (>25 °C), aligning with lower rainfall and higher air temperature. These conditions enable better gas diffusion and favorable aerobic soil conditions for CH₄ uptake. In contrast, in wetter and colder months, particularly in winter and early spring, CH₄ oxidation slowed down due to limited O₂ availability and microbial CH₄ oxidation activities coinciding with higher precipitation, snow cover, and low soil temperatures. These findings underline how weather conditions and seasonal soil temperature and soil moisture shaped CH₄ oxidation patterns, emphasizing how seasonal hydrology and climate regulated gas fluxes (Christophersen et al., 2001). Meteorological variables, air temperature, and precipitation through clear links to soil moisture and soil thermal conditions explained variation in CH₄ flux, whereas wind speed and atmospheric pressure had negligible effects.

There was a notable difference in CH₄ dynamics between NEA and OLA sites. Considering only non-hotspot CH₄ flux measurements, NEA had lower CH₄ fluxes and higher quantified CH₄ oxidation rate per gram of landfill cover soil across treatments. Despite an active gas collection system, a higher CH₄ supply linked to more frequent hotspot emissions likely enhanced CH₄ oxidation activity. On the other hand, the OLA site exhibited higher CH₄ fluxes reflecting reduced substrate availability to oxidize due to diminished CH₄ production from decomposition in aged waste. Alternatively, physical deterioration of soil covers and dense grass covers that could introduce anaerobic microsites or bypassed oxidation could also contribute to greater surface emissions at OLA.

The incubation experiment (Chapter 3) revealed that when soil moisture and temperature were isolated under a controlled environment, the greatest CH₄ oxidation occurred at 25 °C and 20–40% moisture in both vegetated and non-vegetated soils where CH₄ oxidation sharply dropped at 60–80% moisture due to O₂ limitation and at 5°C and 35°C temperatures, much lower CH₄ oxidation rates were observed due to thermal stress on microbial communities. These results coincide with field observations, especially with soil temperature ranges for higher or lower CH₄ uptake. Vegetation treatment had no significant effect under controlled conditions, likely because homogenized soils lacked the structural and microbial differences present in the field (Prajapati & Jacinthe, 2014).

Localized hotspots were observed in both capped landfill sections with different cap maturity and gas control facilities. At OLA, the legacy site, moderately elevated fluxes were monitored linked to availability of preferential gas flow paths whereas notably higher intermittent CH₄ releases were captured at NEA even with an active landfill gas collection system and an engineered relatively young cap structure pointing to structural damages in cover or subsurface gas leaks. Moreover, quantified CH₄ emission was substantially higher when hotspots data were included, as it accounted for most of the total CH₄ release from the landfill cover. This suggests the need for long-term continual monitoring even with adequate mitigatory measures in place.

In conclusion, the findings emphasize that CH₄ flux in landfill cover soils is regulated by several interacting factors such as environmental conditions, soil texture, cap age, cap material, gas transport processes, available infrastructure (gas collection, leachate collection systems), and site-specific management practices. Vegetation plays an indirect role in controlling CH₄ flux patterns

despite its ability to enhance soil properties to promote microbial activity (Abichou et al., 2015) and oxidation, and this influence can be highly context dependent. Thus, vegetation cannot be a sole strategy. Therefore, passive CH₄ mitigation approaches should prioritize optimizing favorable soil moisture conditions, confirming proper gas diffusion, soil texture, soil composition, and the overall landfill cap conditions.

4.2 Significance of Study

Landfill CH₄ emissions are recognized as a major contributor to GHG emissions and a critical climate concern; thus, it has been proposed that CH₄ emissions from Canadian landfills be reduced by 30% from 2020 levels by 2030 (ECCC, 2021). This research demonstrates that vegetated landfill covers act as a passive CH₄ oxidation strategy, particularly for smaller and older capped landfills that lack active gas collection systems. Residual emissions may persist even at well-managed landfills with active gas collection systems in place, suggesting the potential use of vegetated landfill soil covers as a secondary mitigation approach (Duan et al., 2022).

Although, presence of vegetation did not consistently affect CH₄ uptake, it can improve soil biological, physical and chemical properties that not only to enhance CH₄ oxidation capacity of landfill cover soil but also to enhance nutrient cycling and ecosystem rehabilitation along with sustained CH₄ oxidation over time in the landfill covers (Abichou et al., 2015; Qiu et al., 2018). Apart from that, cap age or maturity, structure and composition of landfill cover, cracks or damages in the cap, and subsurface gas transport conditions also contribute to CH₄ emissions and uptake in landfill cover soil in capped landfills. For instance, the NEA site with an active gas collection system, vegetated soil cover can act as an additional barrier to capture residual emissions, despite the limited CH₄ supply received by its landfill cover to be oxidized (Abichou et al., 2015). Therefore, regardless of in-situ infrastructure (e.g., gas collection systems) present in the landfill cover, it is essential to enhance critical soil properties such as porosity, bulk density, organic matter levels, and moisture holding capacity of soil, to maintain effective CH₄ oxidation rates. A year-long field observation and control laboratory incubation study together confirmed that warmer and drier conditions (20–40% and ~25 °C) are ideal for optimal oxidation.

This study is one of the few studies to monitor a year-round CH₄ flux behavior at capped landfill sections with or without gas collection system in southern Ontario, offering a broader view of

seasonal shifts in soil conditions and weather conditions while addressing seasonal challenges that hinder CH₄ oxidation efficiency such as summer excessive dryness and winter frost (Berenjkar, 2021). Reductions in CH₄ oxidation in winter highlight the importance of adaptive landfill cap (bio-cover) designs that can perform effectively year-round under seasonal changes.

4.3 Recommendations, Limitations, and Future Directions

According to the changing seasonality and future climate projections, weather conditions in Southern Ontario are predicted to have changes in precipitation patterns, warmer temperature conditions along with more extreme weather events due to climate change. This alteration will likely shift the soil moisture and temperature conditions that influence CH₄ production and oxidation activities in landfill covers by controlling gas flux dynamics in landfill covers. Freez-thaw cycles and extreme summer dryness (may create surface cracks) can intensify or shift temporally, affecting oxidation capacity and gas transport in landfill covers.

Based on projected climate change scenarios, CH₄ emissions and oxidation efficiency could change, emphasizing the need to introduce adaptive landfill cap designs and management practices. Thus, this study suggests a few practical recommendations to improve CH₄ mitigation in the landfill cover systems, including: (1) enhance drainage and maintain effective irrigation to avoid oversaturation of cover soil and maintain aerobic conditions by controlling soil moisture within 20-40% WFPS; (2) select cover soil mixtures with high gas permeability, limited soil compaction and high organic matter content, as opposed to compacted clay, to enhance gas diffusivity of CH₄ and O₂ and microbial oxidation potential; (3) design climate-resilient landfill covers that maintain performance under more variable moisture and temperature conditions; (4) develop dynamic monitoring strategies that account for future seasonal extremes; (5) integrate future emission scenarios into national carbon accounting frameworks, ensuring that targets remain realistic as environmental conditions evolve. The outlined approaches based on the regional findings of this study would further provide insights to bridge these observations to suggest long-term policy planning and stress both current and future need of proactive mitigation strategies to face anticipated future climate changes.

As with all studies, certain limitations of the present research exist that can be improved upon in future research. Even though year-round data collection was carried out, it may not capture long-term flux dynamics in landfill cover. To capture landfill conditions more broadly, flux dynamics

from multiple capped landfill sites should be studied to strengthen and generalize the findings. Meteorological data were drawn from a nearby station rather than on-site measurements; therefore, microclimatic variability may have been missed. Furthermore, the incubation study used homogenized soils that lack intact soil structure (such as cracks, fissures, vents, etc.), reducing comparability with in-situ observations. A complete year-long CH₄ flux measurements could not gather due to restrictions on site access during February and March 2025. In Chapter 3, considerable portion of collected data were not aligned reduction in consumption rates over time between time intervals may be due to insufficient O₂ conditions in the headspace of jars since gas samples were collected continuously for 15 minutes in sealed environment with limited headspace volume and handling constraints (such as injection errors and multiple punctures degraded the septa allowing minor gas leakages). Maintained adequate O₂ levels in the headspaces by increasing headspace volume, minimizing sample handling outside the incubators to reduce potential temperature drifts in soil and prevent leaks using leak-proof valves.

Future research related to the role of vegetation in CH₄ dynamics in landfill cover soils should focus on an analysis of methanotrophic microbial communities to establish the connection between microbial activity and oxidation patterns while characterizing microbial activity and methanotrophic population structure in the vegetated and non-vegetated landfill covers (Kallistova et al., 2007; Reddy et al., 2019). It is important to monitor landfill cap performance continuously over different landfill cap ages to capture shifts in flux dynamics in response to climatic and seasonal variabilities (Abichou et al., 2011). Furthermore, a comprehensive quantification of CH₄ oxidation efficiency and CH₄ emission over multiple capped landfills in this studied region, southern Ontario, will effectively contribute to improving GHG inventories and to adaptation for optimal cap and cover design, emission reduction, and sustainable mitigation measures (Bogner et al., 2008). Integrating capped landfills' CH₄ emission measurements into national carbon accounting frameworks will further improve the accuracy of inventories, helping to achieve national reduction targets and compliance with climate policy commitments.

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Appendix

Table A1: Fixed Effects and Significance of Vegetation and Monthly Variation on CH₄ Flux and Soil Properties

Model Description	Effect	Estimate	Std. Error	df	t value	Pr(>t)
Effect of Vegetation and Site on CH ₄ Flux	(Intercept)	3.0440	0.0320	13.14	95.22	<0.0001 ***
	Treatment (veg)	-0.0030	0.0417	16.30	-0.072	0.9434
	Site (OLA)	0.0562	0.0456	13.59	1.232	0.2389
	Treatment × Site	0.0185	0.0620	14.89	0.298	0.7696
Soil Moisture and Soil Temperature Effects	(Intercept)	2.9614	0.0580	371.09	51.03	<0.0001 ***
	Soil Temperature (°C)	-0.0017	0.0017	643.15	-1.000	0.3179
	Soil Moisture (%)	0.0050	0.0016	615.91	3.191	0.0015 **
Seasonal Variation in CH ₄ Flux by Treatment	(Intercept)	1.3578	0.6644	32.57	2.044	0.0491 *
	Treatment (veg)	0.2813	0.8628	33.34	0.326	0.7465
	May	3.5307	0.2097	645.10	16.84	<0.0001 ***
	June	3.5716	0.1759	645.01	20.30	<0.0001 ***
	July	3.5937	0.1860	645.03	19.32	<0.0001 ***
	August	3.5746	0.1889	645.04	18.92	<0.0001 ***
	September	3.5779	0.2050	645.06	17.45	<0.0001 ***
	October	3.6034	0.1871	645.04	19.26	<0.0001 ***
	November	3.5965	0.1998	645.04	18.00	<0.0001 ***
	December	3.6201	0.2310	645.06	15.67	<0.0001 ***
	January	3.5654	0.2868	645.02	12.43	<0.0001 ***
	Treatment × May	0.1253	0.2902	645.12	0.432	0.6661

Model Description	Effect	Estimate	Std. Error	df	t value	Pr(>t)
Weather Variables Predicting CH ₄ Flux	Treatment × June	0.1272	0.2475	645.02	0.514	0.6076
	Treatment × July	0.1176	0.2618	645.03	0.449	0.6534
	Treatment × August	0.1744	0.2667	645.16	0.654	0.5133
	Treatment × September	0.2114	0.3036	646.82	0.696	0.4866
	Treatment × October	0.1940	0.2753	647.09	0.705	0.4813
	Treatment × November	0.1935	0.2909	646.12	0.665	0.5062
	Treatment × December	0.0950	0.3157	645.13	0.301	0.7637
	Treatment × January	0.1251	0.4057	645.08	0.308	0.7579
	Intercept	49.57	29.63	563.36	1.67	0.0950
	Precipitation(48hrs)	0.033	0.0146	560.40	2.27	0.0235*
	Wind Speed	-0.0057	0.0255	574.67	-0.22	0.8230
	Pressure	-0.513	0.2979	562.76	-1.72	0.0856
	Air Temperature	-0.0477	0.0214	568.58	-2.23	0.0262*

Table A2: Seasonal Variation in CH₄ Emissions and Soil Conditions across treatments. Tukey's HSD test was applied and Letters not shared between groups denote statistically significant differences. Each value shows the mean and standard errors in brackets. Statistical significance was set at $p < 0.05$ for all tests.

	CH ₄ Flux mg (CH ₄ /m ₂ /d)		Soil Moisture (%)		Soil Temperature (°C)	
Year-Month	Vegetated Treatment	Non-Vegetated Treatment	Vegetated Treatment	Non-Vegetated Treatment	Vegetated Treatment	Non-Vegetated Treatment
2024-April	0.15 (SD = 0.61) ^b	-0.4 (SD = 0.85) ^b	30.3 (SD = 6.0) ^{cde}	28.3 (SD = 8.3) ^{def}	14.1 (SD = 2.3) ^{bc}	15.8 (SD = 3.7) ^d
2024-May	-0.34 (SD = 1.25) ^b	-0.57 (SD = 0.73) ^b	31.0 (SD = 7.3) ^{ef}	29.7 (SD = 6.8) ^{de}	17.1 (SD = 4.1) ^c	16.8 (SD = 2.9) ^c
2024-June	-1.4 (SD = 3.09) ^b	-0.89 (SD = 3.99) ^b	16.3 (SD = 9.7) ^b	15.8 (SD = 10.5) ^b	31.8 (SD = 6.5) ^{de}	28.9 (SD = 5.2) ^f
2024-July	-1.48 (SD = 2.77) ^b	-1.47 (SD = 1.79) ^b	17.7 (SD = 12.3) ^{bc}	19.4 (SD = 12.8) ^{bc}	34.1 (SD = 5.8) ^{ff}	33.1 (SD = 4.6) ^{ff}
2024-August	-1.74 (SD = 4.09) ^b	-2.75 (SD = 4.83) ^b	7.2 (SD = 3.9) ^a	3.9 (SD = 3.6) ^a	31.2 (SD = 5.1) ^{efe}	29.7 (SD = 4.2) ^e
2024-September	-2.33 (SD = 5.54) ^b	-3.22 (SD = 6.68) ^b	10.6 (SD = 5.9) ^a	8.6 (SD = 5.5) ^a	26.6 (SD = 6.1) ^{ed}	25.8 (SD = 6.4) ^d
2024-October	-1.82 (SD = 3.80) ^b	-1.02 (SD = 4.74) ^b	11.2 (SD = 4.3) ^a	8.6 (SD = 3.9) ^a	16.3 (SD = 3.7) ^{cd}	15.3 (SD = 3.6) ^c
2024-November	-1.27 (SD = 3.18) ^b	-1.29 (SD = 3.47) ^b	23.8 (SD = 4.6) ^{cd}	21.7 (SD = 5.2) ^c	8.5 (SD = 2.4) ^c	8.4 (SD = 2.0) ^a
2024-December	-1.21 (SD = 3.03) ^b	-1.35 (SD = 2.37) ^b	24.5 (SD = 4.4) ^d	18.6 (SD = 7.8) ^{bc}	5.8 (SD = 4.0) ^a	3.4 (SD = 3.2) ^a
2025-January	0.07 (SD = 4.70) ^b	-1.21 (SD = 3.44) ^b	27.5 (SD = 5.8) ^{de}	23.5 (SD = 5.0) ^{bcd}	9.6 (SD = 2.9) ^{ab}	7.9 (SD = 1.6) ^{ab}
2025-April	-0.10 (SD = 2.80) ^a	-1.36 (SD = 4.25) ^a	38.0 (SD = 9.7) ^f	37.2 (SD = 9.8) ^e	13.2 (SD = 4.5) ^{bc}	16.0 (SD = 4.5) ^c

Table A3: Monthly Variation in Meteorological and Environmental Conditions and CH₄ Flux (2024–2025) from the Study Landfill. Each Value Shows the Mean and Standard Deviations in Brackets. Statistical significance was set at $p < 0.05$ for all tests.

Year	Month	Air Temperature (°C)	Atm. Pressure (kPa)	Wind Speed (km/h)	Precipitation (48hrs) (mm)	Soil Moisture (%)	Soil Temperature (°C)	CH ₄ Flux (mg CH ₄ /m ₂ /d)
2024	April	11.5 (SD=0.93)	99.0 (SD=0.02)	23.2 (SD=1.00)	0 (SD = 0)	29.25 (SD=7.08)	14.94 (SD=3.14)	-0.126 (SD=0.77)
2024	May	15.5 (SD=2.30)	97.2 (SD=0.08)	14.7 (SD=5.82)	32.7 (SD = 22.56)	30.38 (SD=7.02)	16.96 (SD=3.56)	-0.447 (SD=1.04)
2024	June	23.4 (SD=3.20)	97.8 (SD=0.43)	18.3 (SD=4.65)	17.3 (SD = 4.50)	15.77 (SD=10.06)	30.37 (SD=6.02)	-1.14 (SD=3.56)
2024	July	23.8 (SD=3.65)	98.0 (SD=0.37)	11.3 (SD=4.35)	45.1 (SD = 13.08)	18.36 (SD=12.54)	33.64 (SD=5.22)	-1.48 (SD=1.85)
2024	August	24.5 (SD=2.85)	98.1 (SD=0.44)	14.6 (SD=5.07)	1.8 (SD = 0.34)	5.52 (SD=4.11)	30.40 (SD=4.70)	-2.25 (SD=4.47)
2024	September	23.5 (SD=1.39)	98.1 (SD=0.39)	12.2 (SD=4.23)	0.6 (SD = 0.20)	8.57 (SD=5.97)	26.92 (SD=6.27)	-2.79 (SD=6.12)
2024	October	14.5 (SD=4.43)	98.6 (SD=0.56)	12.7 (SD=8.29)	0 (SD = 0)	9.91 (SD=4.32)	15.79 (SD=3.67)	-1.43 (SD=4.28)
2024	November	7.05 (SD=2.15)	97.3 (SD=0.75)	25.5 (SD=1.74)	18.1 (SD = 7.40)	22.34 (SD=4.98)	8.31 (SD=2.17)	-0.429 (SD=4.71)
2024	December	-4.93 (SD=2.84)	98.3 (SD=1.57)	16.1 (SD=2.88)	7.4 (SD = 0.85)	21.91 (SD=6.75)	4.79 (SD=3.85)	-1.27 (SD=4.06)
2025	January	-5.21 (SD=0.67)	97.9 (SD=0.08)	29.1 (SD=5.20)	0 (SD = 0)	25.5 (SD=5.67)	8.75 (SD=2.43)	-0.572 (SD=4.71)
2025	April	15.9 (SD=6.79)	98.6 (SD=0.27)	11.3 (SD=7.08)	183.2 (SD = 20.52)	23.78 (SD=3.54)	5.87 (SD=2.62)	-0.716 (SD=3.60)

Table A4: CH₄ consumption Rates by Moisture and Temperature Levels and Treatment. Tukey's HSD Test Was Applied and Letters Not Shared Between Groups Denote Statistically Significant Differences. Each Value Shows Mean and Standard Deviations in Brackets. Statistical significance was set at $p < 0.05$ for all tests.

CH ₄ consumption Rates by Moisture Level and Treatment				
Treatment	Moisture (%)	Mean CH ₄ Consumption (µg CH ₄ /g /d)	Min CH ₄ Consumption (µg CH ₄ /g /d)	Max CH ₄ Consumption (µg CH ₄ /g /d)
Veg	20	26.6 (SD = 10.8) ^a	8.9	36.1
Veg	40	29.4 (SD = 16.7) ^a	10.1	54.5
Veg	60	19.5 (SD = 5.7) ^b	11.2	28.9
Veg	80	10.2 (SD = 3.7) ^c	6.4	15.0
Non-Veg	20	35.4 (SD = 11.4) ^a	26.5	57.4
Non-Veg	40	28.1 (SD = 12.0) ^a	17.6	52.2
Non-Veg	60	20.2 (SD = 9.5) ^b	8.8	38.1
Non-Veg	80	11.5 (SD = 6.5) ^c	1.5	19.7
CH ₄ consumption Rates by Temperature Level and Treatment				
Treatment	Temperature (°C)	Mean CH ₄ Consumption (µg CH ₄ /g /d)	Min CH ₄ Consumption (µg CH ₄ /g /d)	Max CH ₄ Consumption (µg CH ₄ /g /d)
Veg	5	17.1 (SD = 7.9) ^a	6.4	29.6
Veg	15	19.9 (SD = 16.9) ^{ab}	6.5	54.5
Veg	25	27.6 (SD = 15.0) ^b	6.5	54.5
Veg	35	21.2 (SD = 7.4) ^a	11.7	31.1
Non-Veg	5	15.3 (SD = 9.8) ^a	1.5	27.5
Non-Veg	15	27.7 (SD = 10.6) ^{ab}	15.0	45.4
Non-Veg	25	33.1 (SD = 16.7) ^b	9.6	57.4
Non-Veg	35	19.2 (SD = 7.5) ^a	9.1	30.4

Table A5: Summary Statistics for All Data, Hotspot Vs. Non-Hotspot, and by Site.

Overall Summary					
Group	Avg. CH ₄ Flux (µg CH ₄ /g /d)	Total Flux	SD	N	Site(s)
All Data	2,446	1,724,308	29,344	705	NEA & OLA
No Hotspot	-1.12	-774	7.44	688	NEA & OLA
Only Hotspot	101,475	1,725,082	164,951	17	NEA & OLA
Site-Specific Summary					
Group	Avg. CH ₄ Flux (µg CH ₄ /g /d)	Total Flux	SD	N	Site
All Data – NEA	4,628	1,721,494	40,296	372	NEA
All Data – OLA	8.45	2,814	84	333	OLA
No Hotspot – NEA	-1.98	-712	8.09	359	NEA
No Hotspot – OLA	-0.19	-62.2	6.55	329	OLA
Only Hotspot – NEA	132,477	1,722,206	178,475	13	NEA
Only Hotspot – OLA	719	2,876	307	4	OLA

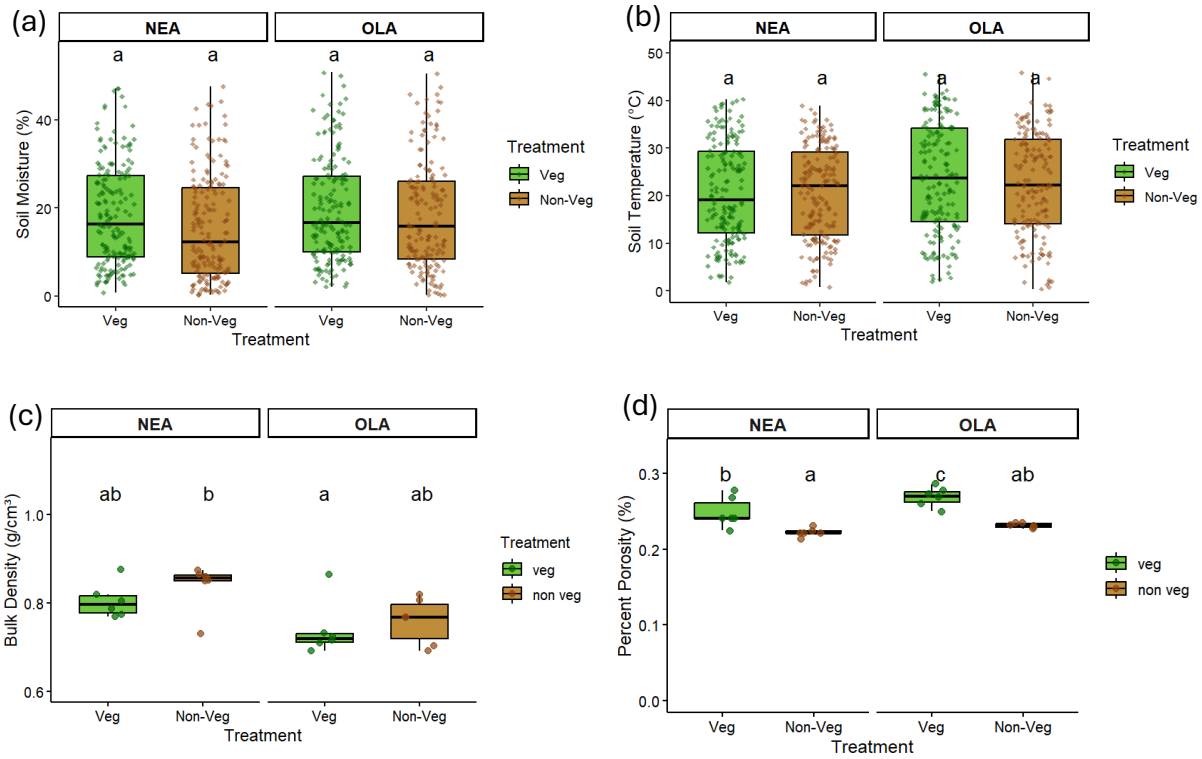


Figure A1: Soil Physical Properties by Site and Treatment. Boxplots show variation in (a) Soil moisture, (b) Soil temperature, (c) Bulk Density, and (d) Percent Porosity between vegetated and non-vegetated treatments at NEA and OLA sites. Tukey's HSD test was applied and Letters not shared between groups denote statistically significant differences. Statistical significance was set at $p < 0.05$ for all tests.

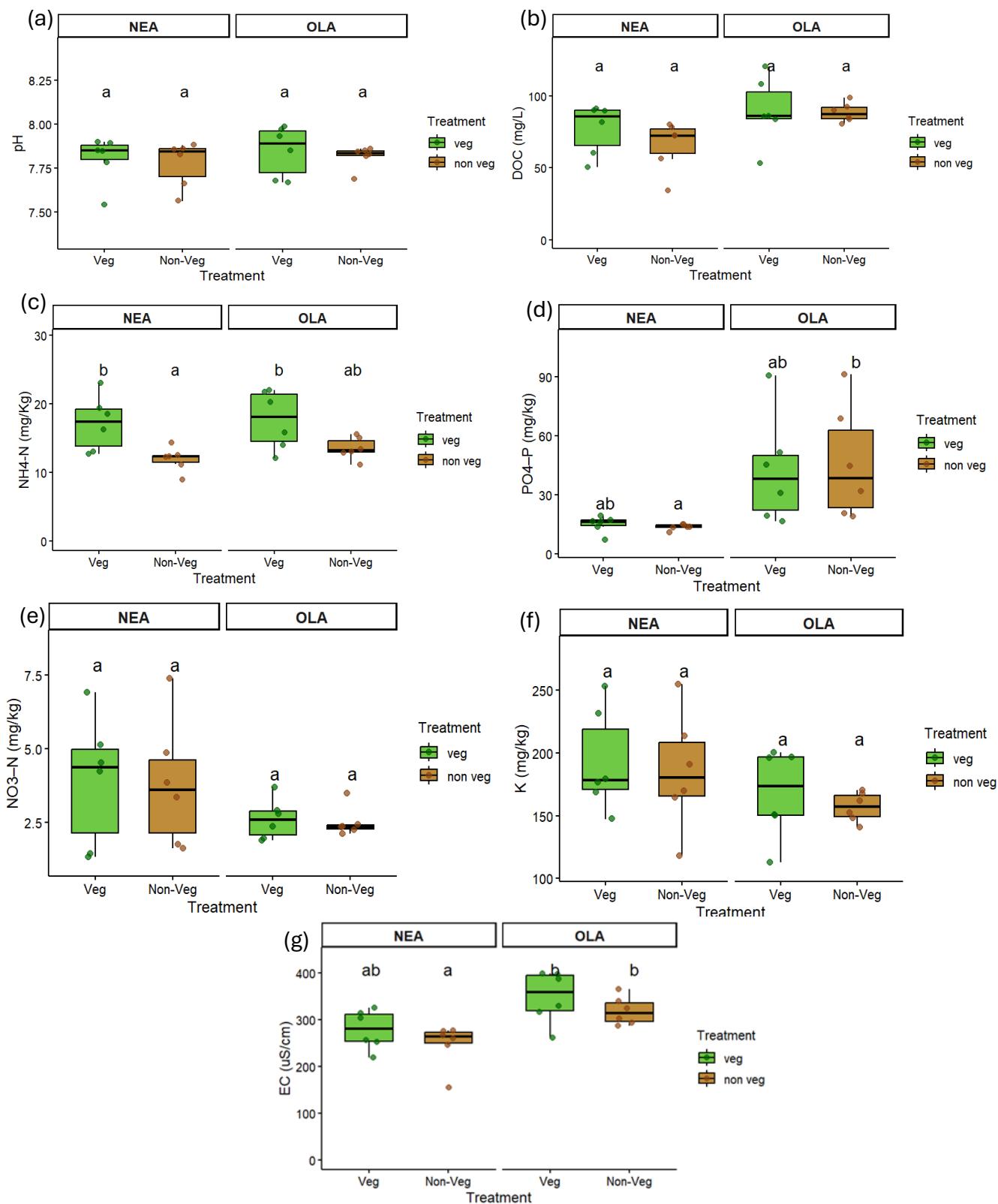


Figure A2: Soil Chemical Properties by Site and Treatment. Boxplots show (a) pH, (b) Dissolved Organic carbon (DOC), (c) Ammonium ($\text{NH}_4^+\text{-N}$), (d) Phosphate ($\text{PO}_4\text{-P}$), (e) Nitrate ($\text{NO}_3\text{-N}$) (f) K (mg/Kg) and (g) EC ($\mu\text{S/cm}$), concentrations across vegetated and non-vegetated soils at NEA and OLA sites. Tukey's HSD test was applied and Letters not shared between groups denote statistically significant differences.

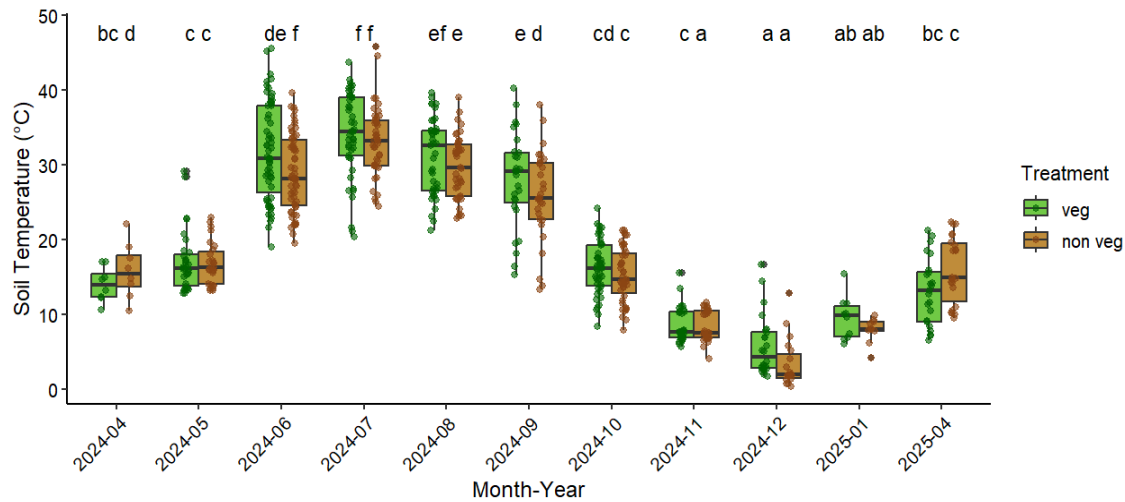


Figure A3: Monthly Variation in Soil Temperature (°C) Across Vegetated and Non-Vegetated Treatments from April 2024 to April 2025. Tukey's HSD Test Was Applied and Letters Not Shared Between Groups Denote Statistically Significant Differences.

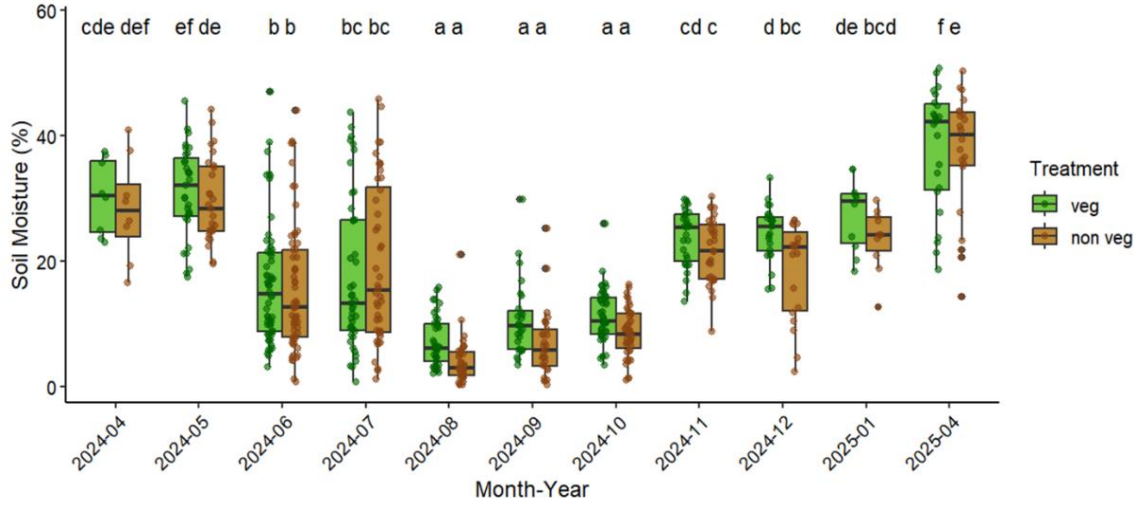


Figure A4: Monthly Variation in Soil Moisture (%) Across Vegetated and Non-Vegetated Treatments from April 2024 to April 2025. Tukey's HSD Test Was Applied and Letters Not Shared Between Groups Denote Statistically Significant Differences.

Calculations used to determine water additions for adjusted to desires WFPS levels in Incubation Experiment

Total Pore Space in Soil (P_s): The total pore space in the soil (P_s) is determined based on oven dried mass, bulk density (BD) and porosity (ϕ) values as follows.

$$P_s = \left(\frac{M_o}{BD} \right) \times \phi \quad (1)$$

Where:

P_s = Total pore space (ml)

M_o = Oven-dried soil mass (g)

BD = Bulk density (g/ml)

ϕ = Porosity

Existing WFPS (MC_{WFPS}): The existing moisture content in terms of WFPS (MC_{WFPS}) was computed by the ratio of existing mass of moisture (MC_g) to the total pore space in soil (P_s).

$$MC_{WFPS} = \frac{MC_g(g)}{P_s(ml)} \quad (2)$$

Where:

MC_{WFPS} = Existing water-filled pore space ratio

MC_g = Existing moisture content (g)

P_s = Total pore space (ml)

Total Existing Moisture in Incubated Soil Mass (MC_M): Then the total existing moisture content (MC_M) in the soil mass(M) was calculated by;

$$MC_M = \frac{M}{BD} \times MC_{WFPS} \quad (3)$$

Where:

MC_M = Moisture already contained in the incubated soil mass (ml)

M = Soil mass used in incubation (g)

BD = Bulk density (g/ml)

MC_{WFPS} = Existing water-filled pore space ratio

The above formulas were used according to the methodology outlined by Lam (2025).

Calculating Additional Water to be Added to Reach Target Moisture Level: Lastly, the amount of water to be added (W_{add}) sets the difference between the desired WFPS (V_w) and the moisture content already present in the incubated soil (MC_M).

$$W_{add} = V_w - MC_M \quad (4)$$

Where:

V_w = Water required for target WFPS (ml)

MC_M = Moisture already in the incubated soil mass (ml)

W_{add} = Volume of extra(additional) water to be added (ml)

The targeted(desired) moisture adjustments were calculated based on moisture adjustment method used by Lam (2025).

*AI and Grammarly were used in different places in this thesis as follow. The support of Grammarly was used throughout the work to improve the clarity of the sentences, to check grammar, and spelling. AI tools were used to generate R scripts(codes) for analyzing data, to make LaTeX functions in Microsoft Word to build formulars and equations in the appendix, chapter 2 and 3 and building tables throughout the work and to create schematic diagrams.