UNDERSTANDING THE IMPACT OF IRRIGATION ON CO_2 EMISSIONS FROM SOILS WITH DIFFERENT PHYSICAL AND CHEMICAL PROPERTIES

by

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Abstract

Agricultural practices contribute to CO_2 emissions. Previous studies in our lab showed that application of irrigation water containing dissolved Ca^{2+}, Mg^{2+}, and HCO_3^- contributed to CO_2 emissions and, at the same time, enriched soil inorganic carbon (SIC) content (i.e., soil carbonates). However, it is unclear how soil properties affect the quantity of CO_2 emitted when irrigation water containing dissolved inorganic carbon (e.g., bicarbonates) is applied to the soil surface. I conducted a series of laboratory experiments to examine how soil moisture, texture, organic matter (OM) content, and temperature affect the release of CO_2 during irrigation of soils with water drawn from Okanagan Lake, which contains dissolved inorganic carbon (DIC). A Picarro cavity ring-down spectrometer was used to monitor changes in CO_2 production and ^{13}C of the emitted CO_2 following irrigation events. Organically derived CO_2 is depleted in ^{13}C compared with DIC-derived CO_2, hence $\delta^{13}CO_2$ measurement was used to determine the source of CO_2 emitted from the surface of soil columns. As hypothesized, a greater proportion of DIC-derived CO_2 was released during irrigation of wetter soils than drier soils. Among the five gravimetric soil moisture levels tested (10%, 15%, 20%, 25%, 30%), CO_2 derived from DIC in irrigation water was detected only in soils maintained at 15% and 25% moisture. A greater proportion of DIC-derived CO_2 was released during irrigation of a silt loam (finer-textured) than a sandy loam (coarser-textured) soil; this effect was probably related to differences in the water holding capacity in the two soil types. Neither soil temperature (21 °C, 30 °C) nor soil OM content (7.7% and 3.3%) affected the proportion of DIC-derived CO_2 that was released during irrigation. Based on these findings, farmers who irrigate with DIC-rich water could use less frequent irrigation to reduce the production of DIC-derived CO_2. Nevertheless, the quantity of DIC-derived CO_2 released from these soils during irrigation was less than 10% of the total CO_2 emitted; this CO_2 production is far outweighed by the CO_2 removed from the atmosphere by irrigated crops via photosynthesis.

Lay Summary

The Okanagan Valley, BC, is a semi-arid region, where irrigation water is drawn from Okanagan Lake. This lakewater contains high concentrations of dissolved Ca^{2+}, Mg^{2+}, and HCO_3^- (bicarbonate) ions, which promote the formation of soil carbonates and CO_2 when added to the soil. I wanted to know whether soil properties affect how much CO_2 is released from dissolved inorganic carbon (e.g., bicarbonates) when this lakewater is applied to the soil surface. During a series of laboratory experiments, I found that more CO_2 was released from dissolved inorganic carbon (DIC) when lakewater was applied to wetter soils than drier soils and when lakewater was applied to finer-textured soils than coarser-textured soils. Soil temperature and soil organic matter content did not affect release of CO_2 from DIC in the lakewater. Overall, the CO_2 released from DIC dissolved in Okanagan Lake water was less than 10% of the total CO_2 emitted from the soil during irrigation.

Preface

I performed a preliminary experiment before starting my main experiments. I obtained the soils for this preliminary experiment from Dr. Craig Nichol of the University of British Columbia Okanagan (UBC O), who had collected the soil in October 2013 and October 2014 from several plots in an Okanagan apple orchard at Agriculture and Agri-food Canada's Summerland Research and Developmental Centre (SuRDC). I ran four main experiments, dealing with four variables of interest: soil moisture, soil temperature, soil texture, and soil organic matter. Dr. Kirsten Hannam collected the soils for the first three experiments from SuRDC. Dr. Andrew Midwood (UBC O) provided data that informed soil selection. For the soil organic matter experiment, I performed the soil sampling. Water samples were collected from Okanagan Lake, once by Naomi Yamaoka, and three times by me. The experimental design for the experiments was developed collaboratively by Drs. Kirsten Hannam, Melanie Jones, Andrew Midwood, Craig Nichol, and myself. The soil collars and chambers were designed and built by Dr. Andrew Midwood. Also, the gas sampling strategy and use of the EGM-4 (Environmental Gas Monitor for CO_2), and Picarro cavity ring-down spectrometer (which measures the concentration of CO_2 and associated ^{13}C) were guided by Dr. Andrew Midwood. Data collection, data analyses, and thesis writing were conducted by myself under the supervision and guidance of Dr. Melanie Jones and Dr. Kirsten Hannam. All the raw data has been uploaded to the Open Science Framework (OSF) under Melanie Jones Lab. Dr. Melanie Jones can be contacted for access to the data.

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Dedication

To my parents, siblings, and husband

Chapter 1: Introduction

Since the 1750s, combustion of fossil fuel and land-use changes have increased atmospheric CO_2 concentration by 31% (Lal, 2004). Consequently, global warming and climate change have become a major concern in today's world. Therefore, a better understanding of global carbon fluxes is needed to mitigate global warming and climate change effects (Bughio et al., 2015). Soil is the largest terrestrial carbon (C) pool, containing about 2500 Pg C, and is ranked as the third largest C reservoir globally (Eswaran et al., 2000). Soil has been recognized as one of the biggest sources of global CO_2 emissions (Schlesinger and Andrews, 2000).

The agricultural sector, which covers 37% of the earth's surface area (Smith et al., 2008), is a large source of CO_2 emissions (Clark et al., 2020; Rosenzweig et al., 2020). It is estimated that about 20% of annual anthropogenic CO_2 emissions could be offset by changes in agricultural practices (Smith et al., 2008). CO_2 emissions from soil, including agricultural soils, result from the breakdown of organic matter by soil microbes, as well as geochemical reactions involving inorganic carbon. While the effects of various management practices on soil organic carbon have been studied intensively, much less is known about the role of soil inorganic carbon in C fluxes from agricultural soils.

1.1 Soil Organic Carbon

Soil organic C (SOC) consists of microbial cells, and plant and animal residues at various stages of decomposition (Paul, 2016). The concentration of organic C in the soil varies widely with 1) climate, e.g., temperature and precipitation, 2) soil physical and chemical properties, and 3) quantity and quality of C inputs to the soil (Luo et al., 2017). Plants add C to the soil in two major ways: 1) plant roots release carbon-containing exudates and organic substances directly into the soil (Cheng et., 1993; Paul and Clark, 1996; Kuzyakov and Demin, 1998; Kuzyakov and Domanski, 2000), and 2) dead or senescing plant roots and aboveground biomass (e.g., shoots, branches, leaves, fruit) are deposited into or on top of the soil, which augment soil C stocks as they are broken down by soil macrofauna (De Silva et al., 2010) or undergo microbial decomposition (Schlesinger, 1977; Paul and Clark, 1996). Only a portion of SOC is rapidly metabolized and released into the atmosphere; therefore, management tools that increase the rate

at which SOC enters the soil or slow the rate at which SOC is released back to the atmosphere may help mitigate anthropogenic CO_2 emissions (Yu et al., 2010; Lorenz and Lal, 2018). A large number of studies have examined methods of increasing SOC stocks in agricultural systems (Paustian et al., 2000; Abdalla et al, 2016; Abdalla et al, 2019). Some of the most important include no-tillage, using cover crops and green manure, selecting crops that have the potential to sequester C, increasing cropping frequency, and increasing the use of perennial forages as part of crop rotations. By contrast, there are very few studies that have focused on increasing soil inorganic carbon stocks (SIC) or reducing emissions from SIC.

1.2 Soil Inorganic Carbon

Soil inorganic C refers to the mineral forms of carbon in the soil, which consist primarily of carbonates (Schlesinger, 1985). Lithogenic carbonate (LC) and pedogenic carbonate (PC) are the two main types of SIC found in nature (Monger et al., 2015). There is another type, called biogenic carbonate, which is quite a small fraction of SIC and, therefore, is not considered as significant as LC and PC (Zamanial et al., 2016).

Lithogenic carbonate originates from original parent material (Zamanian et al., 2016). It is also referred to as non-pedogenic carbonate (Kraimer and Monger, 2009), geogenic carbonate (Kraimer and Monger, 2009; Zamanian et al., 2016), or detrital carbonate (Magaritz and Amiel, 1980). It consists of rock fragments of different sizes that include sand, silt, gravel, cobbles, stones, and boulders (Monger et al., 2015). Calcite ($CaCO_3$) and dolomite [$CaMg(CO_3)_2$] are the two most common chemical forms of LC (Lorenz and Lal, 2018).

Pedogenic carbonate is also known as secondary carbonate (Tamir et al., 2011; Bughio et al., 2015; Kowalska et al., 2019). It is the precipitated form of carbonate following weathering/dissolution of LC or preexisting PC (Wu et al., 2009). Calcite is the most common chemical form of PC in soil, while other forms are dolomite, siderite ($FeCO_3$), and aragonite (another crystal form of $CaCO_3$) (Ming, 2006).

The CO_2 or HCO_3^- required for PC formation are derived from both biotic and abiotic pathways. In addition to carbonate mineral dissolution (i.e., dissolution of either LC or preexisting PC),

they also derive from root and microbial respiration (Rovira and Vallejo, 2008) or atmospheric deposition through diffusion, rainfall (Schlesinger, 1982), and irrigation water (Sanderman, 2012; Hannam et al., 2016; Lorenz and Lal, 2018). The Ca^{2+}or Mg^{2+} for PC formation may derive from either weathering of LC and preexisting PC or exogenous sources such as calcium silicates, biosolids, mineral fertilizers (Nordt et al., 2000; Bughio et al., 2015), and irrigation water (Sanderman, 2012; Hannam et al., 2016; Lorenz and Lal, 2018).

In general, SIC has a less negative $\delta^{13}C$ value than SOC (Smith and Epstein, 1971; Dzurec et al., 1985; Pawellek and Veizer, 1994; Stevenson and Verburg, 2006; Bertrand et al., 2007; Tamir et al., 2011), with LC tending to have a less negative and smaller range of $\delta^{13}C$ values than PC (Table 1.1, Figure 1.1). These differences are caused, in part, by the fact that PC can be formed from soil-respired CO_2 (depleted in ^{13}C relative to LC), from atmospheric CO_2, and/or from dissolved inorganic carbon (e.g., DIC) originating from LC (relatively enriched in ^{13}C) (Figure 1.1). Ultimately, the final $\delta^{13}C$ value of newly precipitated carbonate depends on 1) the source of CO_2 consumed and 2) isotopic fractionation during carbonate precipitation (Cerling et al., 1989), which increases the $^{13}C/^{12}C$ ratio in the precipitated carbonate by 13.5 to 16.5 ‰, depending on the soil temperature (Rovira and Vallejo, 2008). Usually, an average fractionation factor of 15 ‰ is applied to the PC precipitation process. Therefore, the $\delta^{13}C$ value of the newly precipitated PC can be assumed to be the $\delta^{13}C$ of the respired soil CO_2 + 15 ‰ (Figure 1.1).

Table 1. 1 Isotopic values for various forms of carbon (Smith and Epstein, 1971; Dzurec et al., 1985; Pawellek and Veizer, 1994)

Isotopic values, $\delta^{13}C$ (‰)			
Soil organic C (SOC)			
Necromass of C_3 plants	-23 to -34‰	Necromass of C_4 plants	-9 to -17‰
Soil Inorganic C (SIC)			
Lithogenic Carbonate (LC)	-5 to +2‰	Pedogenic Carbonate (PC)	-12.5 to +2‰

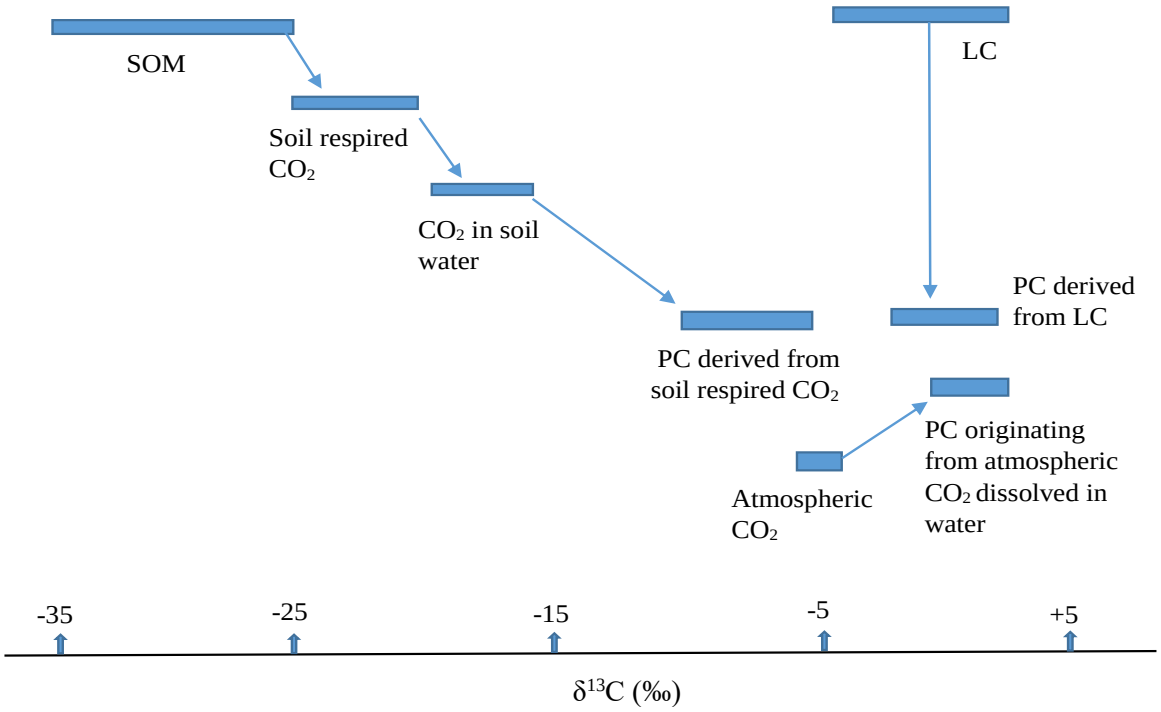

Figure 1. 1 Isotopic composition ($\delta^{13}C$ in ‰) of various forms of C in soil (adapted from Pawellek and Veizer, 1994). SOM = Soil organic matter, LC = Lithogenic carbonate, and PC = Pedogenic carbonate.

The amount of C estimated to be present in SIC stocks in the top 1 meter of soil is 700-1700 Pg (1 Pg=10^{15} g = 1 Gt) globally (Lorenz and Lal, 2018). However, SIC is a higher proportion of total soil C in arid and semiarid regions, which represent 30% of Earth's land area (Lal and Kimble, 2000). In these regions, SIC can be 10 times greater than the SOC stocks (Schlesinger, 1982). Furthermore, Diaz-Hernandez (2010) found large amounts of SIC below a 1-meter depth in the Guadix-Baza basin, a semiarid region of southern Spain, which indicates that the total amount of SIC may be much higher than most current estimates. According to Landi et al. (2003), SIC content ranged between 100.8 and 181.5 kg m^{-2} (up to 120 cm soil depth) in five different zones of boreal grassland and forest regions of Saskatchewan, Canada. Therefore, to understand the global C cycle, further investigation of the actual amount and dynamics of the SIC pool, especially in arid and semi-arid regions, is required (Zamanian et al., 2016).

There are very few studies that have focused on increasing SIC stocks for climate change mitigation. One reason for this is that SIC stocks are concentrated deeper in the soil profile than SOC stocks (Wang et al., 2010). The SIC pool has a longer mean residence time (~85,000 years) (Schlesinger, 1985) than the SOC pool (maximum ~3000 years) (Hsieh, 1992), which means SIC exchanges more slowly with the atmospheric CO_2 and thereby, has usually been assumed to contribute little to atmospheric CO_2 emissions (Schlesinger, 1985). There are exceptions to these general findings. For example, a study conducted in New Mexico found that the mean residence time of SIC can be as short as 120 years (Monger and Gallegos, 2000). Bertrand et al. (2007) demonstrated that carbonate dissolution may contribute up to 35% of total C efflux from five agricultural soils in France. Another recent study conducted in New Mexico by Wang et al. (2016) showed that, in a favorable soil environment, SIC can accumulate in months to years, rather than centuries to millennia. Cardinael et al. (2019) conducted a study examining the contribution of SIC to CO_2 emissions from a silty loam soil situated at Montpellier, France. According to their study, 20% of the total CO_2 emissions were contributed by SIC in the topsoil, and in the subsoil, it was 60%. Clearly, more work needs to be done to understand the SIC cycle.

1.2.1 Impacts of irrigation on pedogenic carbonate formation and CO_2 efflux

Inorganic carbonate cycling is more active in agricultural soils than in non-agricultural soils (Bughio et al., 2015). Irrigation, fertilization and liming, as well as the application of organic amendments, are some common activities on agricultural fields that can influence the rate of inorganic carbon cycling in agricultural fields. Among these practices, irrigation is one of the most important inputs, especially in arid and semi-arid regions. Based on previous studies, Entry et al. (2004) estimated that about one-third of the yield and one-half of the value of crops in the United States result from irrigated agriculture, particularly in arid and semiarid climatic zones. Considering the importance of irrigation to crop productivity, irrigated farming is being expanded worldwide. In 2007, approximately 257 million hectares of agricultural land were irrigated (Alexandratos and Bruinsma, 2012). It is predicted that additional land will be converted to irrigated farming in the near future, as the world population is growing rapidly (Wu et al., 2008; Alexandratos and Bruinsma, 2012). The global area converted to irrigated farming

may reach 318 million hectares in 2050 (FAO, 2011). Given the expanding use of irrigation across the globe, the effects of irrigation on SIC dynamics require further investigation.

The effects of irrigation on SIC formation vary widely. In many cases, the evidence of irrigation-induced changes can be found deep in the profile rather than at the surface layer (Entry et al., 2004). For example, Khokhlova et al. (1997) found that 30 years of irrigation had not changed carbonate levels in the upper part of the profile of soils on the semi-arid Russian steppe, but had caused formation of a compact carbonate horizon in the lower part (120-155cm) of the profile. By contrast, more carbonates were found in the top 1 m of an irrigated field (>30 years) compared to an adjacent field under native sagebrush vegetation in the Western USA (Entry et al., 2004). Margaritz and Amiel (1981) also detected a higher rate of $CaCO_3$ transformation (dissolution-recrystallization) in some irrigated soils of Israel compared to adjacent undisturbed soils. Recently, a reduction in carbonate concentration was observed in the finest fractions of the soil (0-20 cm depth) situated in the Ebro Basin of Iberian Peninsula after the implementation of irrigation (De Sotto et al., 2017).

Decreasing surface SIC stocks due to irrigation indicate the dissolution of carbonates (Equation 1 to the left) and subsequent downward movement of HCO_3^- ions. The newly dissolved HCO_3^- ions have two possible fates: 1) they may re-precipitate with Ca^{2+} or Mg^{2+} ions lower in the soil profile releasing CO_2 (Equation 1 to the right), which results in no net change in atmospheric CO_2 concentration, or 2) they may enter groundwater or migrate to the ocean where they can remain for a longer period. This latter fate is generally considered C sequestration (Sanderman, 2012).

$$Ca^{2+}_{(aq)} + 2HCO_3^-{}_{(aq)} \longleftrightarrow CaCO_{3\,(s)} + H_2O_{(l)} + CO_{2(g)} \quad (1)$$

Irrigation water quality plays a significant role in driving changes in SIC cycling on irrigated sites. For example, greater SIC accumulation and CO_2 release have been observed in fields irrigated using water with higher electrical conductivity (EC) than with a lower EC (Wu et al. 2008; Sanderman, 2012). Irrigation with treated effluent can also promote the accumulation of total and clay-sized carbonates compared with irrigation using freshwater (Eshel et al. 2007); in this case, carbonate dissolution may have been inhibited by the presence of higher concentrations of dissolved organic matter in the effluent (Lebron and Suarez, 1996).

In arid and semi-arid regions, irrigation water contains a large amount of dissolved Ca^{2+}, Mg^{2+} and inorganic carbon in the form of HCO_3^- ions, which play a significant role in neo-formation of CO_2 and PC in agricultural soils if leaching is not sufficient (Equation 1 to the right; Suarez, 2000; Sanderman, 2012; Bughio et al., 2015; Hannam et al., 2016). Hannam et al. (2016) recently conducted a study on a 10-year-old drip-irrigated apple orchard soil (non-calcareous) where water from Okanagan Lake, which contained a significant amount of Ca^{2+}, Mg^{2+}, and HCO_3^-, had been used for irrigation. They found a higher concentration of carbonates directly under the drippers; the isotopic signature of the evolved CO_2 from this soil indicated a strong presence of C from HCO_3^- in the irrigation water. In a related field study, between 9 and 15% of total evolved CO_2 was believe to have originated from dissolved HCO_3^- present in the irrigation water (Hannam et al., 2019). Based on their findings, they hypothesized that carbonate precipitation might be elevated with increasing concentrations of Ca^{2+}, Mg^{2+}, and HCO_3^- ions in the irrigation water.

If irrigation water contains high levels of basic cations (Ca^{2+}, Mg^{2+}) and HCO_3^-, then increasing the quantity of water applied during irrigation may also increase the formation of PC and release of CO_2 from the soil surface. For example, Entry et al. (2004) predicted that furrow irrigation may lead to greater neoformation of PC than sprinkler or drip irrigation because furrow irrigation applies more water over the soil surface. The opposite may occur in fields irrigated with water having low EC. In such cases, drip or sprinkler irrigation may encourage the accumulation of more PC in the top layer of the soils compared to furrow irrigation. Furrow irrigation with water having low EC encourages downward percolation of larger volumes of water, which are more likely to dissolve the carbonates present near the soil surface, leaching them downward within the profile or into the groundwater (Aase et al., 1998). Therefore, the surface layer of the soil may have a reduction of PC. Clearly, irrigation water quality and method of application both play important roles in determining whether irrigation will dissolve carbonates and leach them downward or promote the precipitation of carbonates near the soil surface as well as the associated CO_2 emission (Denef et al., 2008; Wu et al., 2008; Sanderman, 2012). However, it is unclear how physical and chemical properties of soil affect the quantity of PC and CO_2 production from the dissolved Ca^{2+} and HCO_3^- ions present in the irrigation water. Therefore, in my M.Sc. thesis, I decided to examine if soil properties have any impact on the proportion of inorganic carbon dissolved in the irrigation water that is released as CO_2.

1.2.2 Effect of soil properties on soil inorganic C (SIC) dynamics and their interactions with irrigation

Very few studies have been found that demonstrate the effect of soil properties on SIC dynamics and vice versa. Soil properties such as soil moisture, texture, temperature, and organic carbon content may affect PC accumulation and associated CO_2 formation.

Soil moisture

The accumulation and distribution of carbonates in soil are influenced by soil moisture (Egli and Fitze, 2001). Regions with <500 mm mean annual precipitation and high evapotranspiration generally experience a high amount of PC accumulation near the soil surface because low precipitation and high evapotranspiration limit carbonate dissolution and leaching deeper into the soil profile (Landi et al., 2003). With an increase in precipitation, the PC content in the upper soil layers decreases as HCO_3^- ions are leached downward. Similarly, when low EC irrigation water is applied to soils, the LC and PC present in the soil surface undergoes dissolution (Equation 1 to the left) and if there is sufficient downward percolation of water through the soil, the resultant ions move downward. The ions may reprecipitate as calcite ($CaCO_3$) (Equation 1 to the right) or dolomite [$CaMg(CO_3)_2$] deeper into the soil profile if conditions are favorable (Magaritz and Amiel, 1981; Khokhlova et al., 1997; Wang et al., 2010; Sanderman, 2012). In natural settings, water uptake by plants from the root zone acts to concentrate the soil water solution at the base of the root zone, and allows for the precipitation of calcite or dolomite. If the irrigation water itself contains dissolved Ca^{2+} and HCO_3^- ions, the ions react together to produce PC and CO_2 (Equation 1 to the right). This neoformation of PC and CO_2 is expected to be lower with increasing soil moisture content as soil moisture promotes biological CO_2 production by stimulating microbial respiration (Linn and Doran, 1984; Doran et al., 1990; Dilustro et al., 2005; Liu et al., 2009; Dong et al., 2014; Schimel, 2019; Butcher et al., 2020; Wang et al., 2020). The higher pCO_2 in wetter soil would discourage equation 1 to move to the right and thereby, would produce less irrigation water dissolved inorganic carbon (DIC)-derived CO_2 compared to drier soil.

Soil temperature

According to Amit et al. (2011), the effect of temperature on SIC content is complex. In the presence of sufficient moisture (Yuste et al., 2003; Chang et al., 2014), higher soil temperatures promote microbial activity in the soil, which enhances organic CO_2 production (Davidson et al., 1998; Fang et al., 1998; Lin et al., 1999; Lal and Kimble, 2000; Fang and Moncrieff, 2001; Qi and Xu, 2001; Dilustro et al., 2005; Hamdi et al., 2011; Litton et al., 2011; Schindlbacher et al., 2011; Ferrea et al., 2012; Karhu et al., 2014; Hicks Pries et al., 2017; Robinson et al., 2017; Bamminger et al., 2018; Bhanja et al., 2019; Chen et al., 2021). As a consequence, the soil solution might be expected to become enriched in dissolved CO_2. However, the solubility of CO_2 also decreases with temperature and, therefore, an increase in soil temperature leads to off-gassing of CO_2 and a decline in the concentration of CO_2 in the soil solution. Moreover, the rate of reaction between the Ca^{2+} and the HCO_3^- ions, provided by the irrigation water, can be expected to be faster at higher temperatures (equation 1 to the right). Therefore, applications of irrigation water are expected to cause the release of more CO_2 at warmer temperatures than at cooler temperatures (Equation 1 to the right).

Higher temperature also promotes acidifying processes, which release H^+ ions (Tan et al., 2018; Nguyen et al., 2019). Hydrogen ions can directly dissolve pre-existing PC (Equation 2 to the right) or can react with the HCO_3^- ions formed from the PC or dissolved in the irrigation water to produce CO_2 (Equation 3 to the right) (Bertrand et al., 2007; Tamir et al., 2011). Chevallier et al. (2016) examined the effect of temperature on SIC-derived CO_2 from a calcareous soil of northwestern Tunisia and found that the release of CO_2 from SIC increased with increasing temperature.

$$CaCO_{3(s)} + H^+_{(aq)} \longleftrightarrow Ca^{2+}_{(aq)} + CO_{2(g)}) + H_2O_{(l)} \quad (2)$$

$$H^+_{(aq)} + HCO_3^-{}_{(aq)} \longleftrightarrow H_2CO_{3(aq)} \longleftrightarrow H_2O_{(l)} + CO_{2(g)} \quad (3)$$

Soil texture

Soil texture influences the formation and accumulation of PC largely by affecting water holding capacity (Chadwick et al., 1989; Tahir and Marschner, 2017; Upadhyay and Raghubanshi, 2020). Fine-textured soils have a larger number of pore spaces, but with smaller diameters, than coarse-textured soils; these small pore spaces hold the water tightly in the soil. As long as the soil does not become anaerobic, the high water-holding capacity of a fine-textured soil promotes microbial respiration and hence, produces more biological CO_2 than coarse-textured soil (Kowalenko and Ivarson, 1978; Bouma and Bryla, 2000; Dilustro et al., 2005). Therefore, if irrigation water containing Ca^{2+} and HCO_3^- ions (dissolved) are applied, a fine-textured soil is expected to produce less DIC-derived PC and CO_2 (Equation 1 to the right) compared to coarse-textured soil.

Soil organic carbon content

Very few studies have examined the relationship between SIC and SOC. Nevertheless, there is a strong interaction between these two carbon pools. For example, soil carbonates can stabilize the SOC pool by coating individual particles of soil organic matter (SOM) (Mendoza-Vega and Messing, 2005). In this case, a positive relationship between SOC and SIC stocks can be expected. A positive relationship between SOC and SIC may also arise from the production of CO_2 during SOC decay, and its subsequent precipitation as Ca and/or Mg carbonates (Clough and Skjemstad, 2000). This mechanism explains the observations of Rovira and Vallejo (2008), who found that the enhanced microbial activity caused by applications of plant material to soils also elevated the precipitation of soil carbonates. Clearly, in the presence of sufficient Ca^{2+} or Mg^{2+} ions, SOC can be converted into SIC, causing a positive relationship between these two carbon pools (Clough and Skjemstad, 2000) (if the SOC itself is a source of Ca^{2+} or Mg^{2+} ions, more SIC can be produced). Furthermore, plant roots themselves may excrete Ca^{2+} and HCO_3^- ions that can promote carbonate precipitation under alkaline conditions (Kuzyakov et al., 2006).

A negative relationship can also be found between SOC and SIC. When the SOC is solely a source of carbon dioxide, it leads to SIC dissolution. Root and rhizosphere microbial respiration increase the partial pressure of CO_2 of soil air (Ferrea et al., 2012; Thangarajan et al., 2013; Chen et al., 2021), promoting the production of carbonic acid in the soil solution (relatively acidic soil;

Equation 3 from right to middle) and, in turn, increasing carbonate solubility (Andrews and Schlesinger, 2001). Indeed, active roots have been found to increase carbonate dissolution by five to ten times (Zamanian et al., 2016). In addition, carboxylic acids and H^+ ions are released by roots, which further depresses rhizosphere soil pH and increases carbonate dissolution (Andrews and Schlesinger, 2001). Accordingly, the higher ion concentrations resulting from carbonate dissolution near the root lead to greater re-precipitation of carbonates in adjacent root-free soils (Kuzyakov et al., 2006). Dong et al. (2017) observed a significant positive correlation between SOC and SIC content in bulk high pH soils, but in the clay fraction, they found a negative correlation. This can be attributed to the high surface area of the clay fraction, which easily reacts with carbonic acids produced during organic C mineralization, reducing the SIC pool (Tamir et al., 2011).

Soil microorganisms such as bacteria and fungi also play a role in PC formation. In the presence of Ca^{2+} ions in the solution, bacteria can form visible carbonates on their external surfaces within a few days (Monger et al., 1991). Even bacterial cell walls may act as nuclei for PC formation (Perito et al., 2014). The carbonates of these biotic origins are known as biogenic carbonates (see Section 1.2) (Zamanian et al., 2016). Soil organisms such as earthworms actively secrete calcium carbonates, which have been shown to increase soil organic C stabilization (Zhang et al., 2013).

Based on the above discussion, if irrigation water containing dissolved Ca^{2+} and HCO_3^- ions is applied to soil high in organic matter (SOC is proportional to soil organic matter) but neutral in pH, it is expected to produce less irrigation-derived PC and CO_2 compared to soil low in organic matter (OM). This is because the increased partial pressure of biological CO_2 in the soil high in OM would decrease the rate of reaction between Ca^{2+} and HCO_3^- ions in the applied irrigation water and therefore, reduce the production of new PC and CO_2 (Equation 1 to the right).

1.3 Research Objectives and Hypotheses

Based on the knowledge gaps described above, my MSc thesis research will investigate the effect of several soil characteristics on the interactions between DIC-rich irrigation water and CO_2 efflux. Equation 1 clearly shows how the Ca^{2+} and HCO_3^- ions dissolved in irrigation water generate CO_2 when they react to produce $CaCO_3$ (Equation 1 to the right). Soil carbonates (PC)

may cause a further release of CO_2 by reacting with H^+ ions present in the soil solution (Equation 3 to the right). Therefore, the C from the HCO_3^- in the irrigation water is often released as CO_2. My objective was to examine how the chemistry of irrigation water, as well as some physical and chemical properties of soil (soil moisture, texture, temperature, and organic matter), affects the release of CO_2 from DIC immediately after irrigation. My experiments used several soils from the semi-arid Okanagan Valley of British Columbia. Because of the hot dry summers, most crops in the Okanagan Valley are irrigated during the summer season (Wittneben, 1986). The irrigation water used in this region is often drawn from Okanagan Lake (K.D. Hannam, pers. comm.), which contains a relatively high concentration of HCO_3^- ions. I based my experiments on the chemistry of this water. Specifically, my thesis work was built on the finding of Hannam et al. (2016, 2019) that DIC in irrigation water can be released as CO_2 during irrigation events. My specific hypotheses were as follows:

- The C from the HCO_3^- in irrigation water applied to soils would contribute to CO_2 production (Equation 1 to the right; section 1.2.1).
- When irrigated with DIC-rich water, drier soil would release more DIC-derived CO_2 compared to wetter soil. My rationale was that lower moisture in drier soil would produce less biological CO_2 than wetter soil and thereby, encourage the neoformation of PC and CO_2 from the added Ca^{2+} and HCO_3^- ions in the irrigation water (Equation 1 to the right; section 1.2.2).
- At higher temperature, the overall CO_2 efflux from the DIC in irrigation water would be greater compared to the efflux at room temperature. My rationale was that higher temperatures would lead to the off-gassing of CO_2. Moreover, the rate of reaction between the Ca^{2+} and the HCO_3^- ions, provided by the irrigation water, would be faster at higher temperature than at room temperature; as a consequence SIC would be produced more readily, releasing CO_2 in the process (Equation 1 to the right). Higher temperatures also promote acidifying reactions, releasing H^+ ions, which would react with HCO_3^- ions applied in the irrigation water to produce H_2O and CO_2 (Equation 3 to the right). Together, these processes would result in a greater release of DIC-derived CO_2 at higher temperatures compared to room temperature (section 1.2.2).

- Soil texture is related to many factors, which made it difficult to predict its effects on CO_2 production. However, considering the effects of soil texture on water-holding capacity and microbial activity, I predicted that, after treatment with the irrigation water, coarser-textured soil would result in the release of more DIC-derived CO_2 compared to the finer-textured soil. My rationale was that coarser-textured soil would experience lower microbial activities (due to the lower water-holding capacities) resulting in less CO_2 than the finer-textured soil. Therefore, the neoformation of PC and CO_2 from the DIC in irrigation water would be greater in coarser-textured soil than the finer-textured soil (Equation 1 to the right; Section 1.2.2).
- Soil lower in organic matter (OM) would release more DIC-derived CO_2 compared to the soil higher in OM. My rationale was that a higher amount of OM would raise the pCO_2 (p = partial pressure) in the soil environment, which would discourage the neoformation of PC and CO_2 from the irrigation water (Equation 1 to the right; Section 1.2.2).

Chapter 2: The Effect of Irrigation Water Chemistry and Soil Moisture on CO_2 Production

2.1 Synopsis

Chapter 1 described how soil moisture is likely to play a role in regulating the release of CO_2 derived from DIC (e.g., bicarbonate ions) dissolved in irrigation water. For example, Hannam et al. (2019) reported that a greater volume of DIC-derived CO_2 was released from DIC-containing irrigation water applied to drier soils than to wetter soils. Equation 1 clearly shows that soil moisture is expected to play a role in determining the proportion of dissolved bicarbonate ions that are released as CO_2 from irrigation water.

$$Ca^{2+}_{(aq)} + 2HCO_3^-{}_{(aq)} \longleftrightarrow CaCO_{3\,(s)} + H_2O_{(l)} + CO_{2(g)} \quad (1)$$

One of my objectives was to develop a better understanding of the role of soil moisture in determining the proportion of irrigation water-derived DIC that is released during and after the application of irrigation water. For this chapter, I tested two hypotheses. First, I predicted that the Ca^{2+} and HCO_3^- in irrigation water applied to soils would contribute to CO_2 production. Second, in support of the observations of Hannam et al. (2019), I predicted that drier soil would release more CO_2 from the irrigation water compared to wetter soil when irrigated with DIC-containing water. My rationale was that the higher moisture in wetter soil would stimulate microbial respiration and hence, produce more biogenic CO_2 (Linn and Doran, 1984; Doran et al., 1990; Dilustro et al., 2005; Liu et al., 2009; Dong et al., 2014; Schimel, 2019; Butcher et al., 2020; Wang et al., 2020) than the drier soil, where the biogenic CO_2 production would be limited due to low moisture content (Davidson et al., 1998; Epron et al., 1999; Yuste et al., 2003; Dilustro et al., 2005; Ford et al., 2007; Butcher et al., 2020). The greater amount of biogenic CO_2 produced in the wetter soil would discourage precipitation of $CaCO_3$ (Pal et al., 2000) and, thus, suppress the production of CO_2 from the HCO_3^- ions in the irrigation water compared to the drier soil.

This experiment was conducted in the BRAES Soils Laboratory, at the Okanagan campus of the University of British Columbia (UBC O), in Kelowna. The CO_2 efflux rate (μmol $m^{-2}sec^{-1}$) and the $\delta^{13}CO_2$ (‰) were measured before and after irrigation water of three different chemistries was applied to soil collars packed with sandy loam soil and maintained at five different soil moisture levels.

2.2 Methodology

2.2.1 Preliminary experiment

Before doing the main experiment, a preliminary experiment was run to determine how long it took for CO_2 efflux to stabilize after air-dried soils were re-moistened. For this experiment, around 41 kg of frozen Osoyoos loamy sand soils were obtained from Dr. Craig Nichol of UBC O, who had collected the soil between 15th May and 22nd October 2013, and between 8th April and 29th October 2014 from several plots in an Okanagan apple orchard. In the laboratory, the samples were thawed, dried at 65 to 70 °C for 3 to 4 days, and homogenized, but not sieved. Afterward, three soil collars, which were made by cutting PVC irrigation pipe (10 cm long, 10.5 cm diameter), were packed with the dried soil to 7 cm depth, leaving 3 cm on the top of the soil surface to allow room for application of irrigation water. The collars were packed to a bulk density of 1.45 g cm^{-3}, which lies within the range of bulk density values that had been measured in May 2014 for unsieved soils from the same plots (K. Hannam, pers. comm). A piece of fine nylon fabric with 0.33 mm openings (E. Packard, pers. comm.) was fixed around the bottom of each soil collar using a zip tie. The mesh prevented soil from washing out of the collar while allowing drainage.

After packing the collars, deionized water was added to each soil collar to achieve one of three gravimetric water contents (10%, 15%, 20%). Soil collars were then placed on a rack that ensured air flow around the top and bottom of the collars, and maintained at room temperature for a 10-day stabilization period. During this period, the collars were re-weighed on a daily basis and deionized water added, as required, to maintain the desired moisture content. The CO_2 efflux of each chamber was measured using an EGM-4 Environmental CO_2 Gas Monitor (model 4.18, Amesbury, MA, USA). Using this system, an SRC-1 (Soil Respiration Chamber) connected to the EGM-4 was secured to the top of the soil collar to analyse CO_2 released from the soil; each efflux measurement took approximately 5 minutes. Measurements were repeated daily for 12 days. After 10 days, the CO_2 efflux had stabilized (Figure 2.1), so this stabilization period was used in future experiments using similar types of soil.

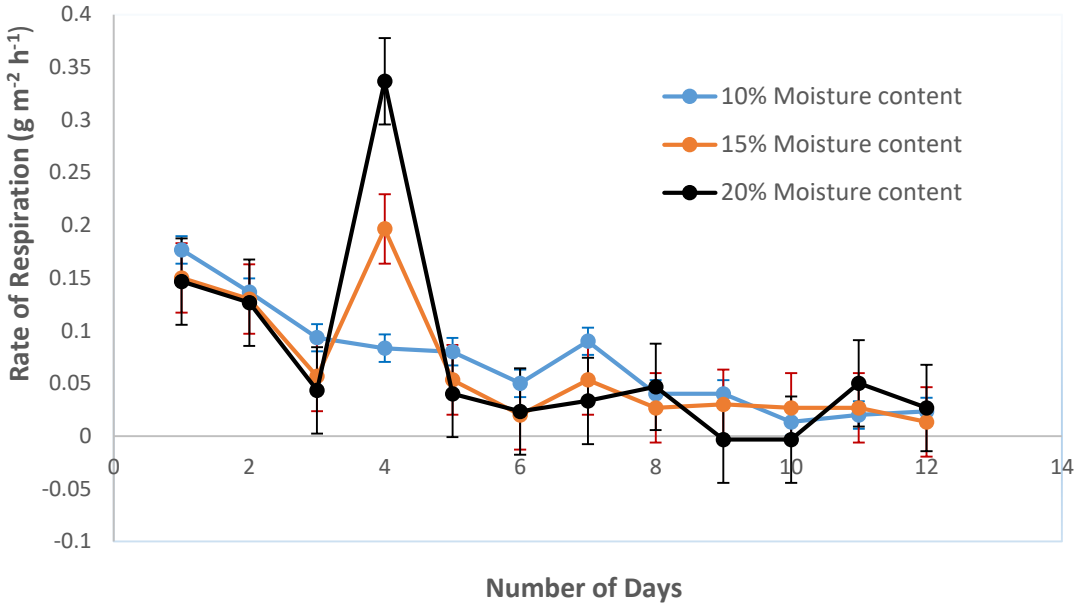

Figure 2. 1 Rate of CO_2 efflux (g m^{-2} h^{-1}) measured over 12 days from soil collars containing Osoyoos loamy sand soils maintained at 10%, 15%, or 20% gravimetric moisture content. The solid circles on the graph represent the treatment means (n=3); the error bars show the standard deviations.

2.2.2 Experiment 1: The effect of irrigation water chemistry and soil moisture on CO_2 production

Collection, processing, and storage of soil samples

The soil used in this study was a Skaha soil collected from the Agriculture and Agri-food Canada Summerland Research and Developmental Centre, which is located in the southern part of the Okanagan Valley, Canada. Taxonomically, this soil is classified as an Orthic Brown Chernozemic soil (Wittneben, 1986). Skaha soil tends to have a sandy loam or loamy sand texture, with a moderate to low water holding capacity, low organic C content, and a pH of 7.3 to 8.3 within the top 0 to 50 cm depth. Grasses (*Poaceae*), sagebrush (*Artemisia tridentata*), rabbitbrush (*Ericameria nauseosa*), and scattered ponderosa pine (*Pinus ponderosa*) are some naturally occurring vegetation on this soil type. However, tree fruits, grapevines, forage crops, and vegetables are grown in most of the cultivated areas with this soil type in the Okanagan Valley (Wittneben, 1986).

On March 25, 2019, soil samples were collected using a shovel (approximate depth, 20 cm) from a former experimental apple block. Samples were collected from plots that had received one of two treatments: 'alfalfa mulch' or 'black plastic mulch' (Neilsen et al. 2014). Plots had been established in 2006 and maintained until 2015, after which the trees were pulled from the soil and the area left fallow. For use in my experiment, around 25 kg of soil was collected from two replicate plots from each treatment. The collected samples were homogenized by hand, large roots and stones were removed, and then the samples were taken to the BRAES Soils Laboratory at UBC O. In the laboratory, the soils from the 'alfalfa mulch' and the 'black plastic mulch' treatments were composited into one bulk sample, oven-dried at 65 to 70 ℃ for 3 to 4 days, sieved through a 2 mm sieve, and stored at room temperature (21 ℃) until use.

Collection, preparation, and storage of water

Three types of water were used for this experiment: (i) Okanagan Lake water (ii) artificial lake water (iii) 'control' water. There is no consistency in the depth from which Okanagan Lake water is drawn for irrigation purposes, nor how it is treated prior to application. Therefore, for this experiment, Okanagan Lake water was collected from the lake shore on Cadder Avenue, Kelowna, on June 20, 2019. The water was collected using a plastic gallon bucket. The concentration of HCO_3^- ions found in Okanagan Lake water typically ranges from 130 to 190 mg L^{-1}, and the dissolved Ca^{2+} and Mg^{2+} ions usually range from 20 to 35 mg L^{-1} and from 9 to 12 mg L^{-1}, respectively (Mackie, 2010). The artificial lake water was prepared using sodium bicarbonate ($NaHCO_3$) and calcium chloride ($CaCl_2$) to attain a concentration of 150 mg HCO_3^- L^{-1} and 50 mg Ca^{2+} L^{-1} (Table 2.1). These values were chosen to approximate the chemical composition of Okanagan Lake water, while eliminating the effect of other dissolved materials in the natural Okanagan Lake water (e.g., dissolved organic C). For water used in the control treatment, NaCl was added to deionized water (Table 2.1) to control for the Na^+ and Cl^- ions added during the preparation of the artificial lake water (Equation 4). All the water samples were stored in plastic containers at room temperature (21 ℃) for use in experiments. No headspace gas was used while storing the water samples. In future studies, headspace gas (eg., nitrogen) should be used to preserve the chemical composition of the water.

$$CaCl_2 + NaHCO_3 \longrightarrow Ca(HCO_3)_2 + 2NaCl \quad (4)$$

Table 2. 1 Chemicals used to prepare 500-mL quantities of artificial lake water, and the resulting concentrations of Na^+, HCO_3^-, Cl^-, and Ca^{2+} ions dissolved in each solution

Type of irrigation water	**Weights of chemicals added to 500 mL deionized water (g)**	**Concentration of ions**			
		Na^+	HCO_3^-	Cl^-	Ca^{2+}
Okanagan Lake water	NA	*0.0005 M	0.002 M to 0.003 M	*0.0002 M	0.0005 M to 0.001 M
Artificial lake water	0.1 g $NaHCO_3$ 0.07 g $CaCl_2$	0.002 M	0.003 M	0.002 M	0.001 M
Control water	0.07 g NaCl	0.002 M		0.002 M	

*The concentration of Na^+ and Cl^- ions presented in the table were obtained from the ALS Environmental laboratory report 2107, Kamloops, BC (unpublished report).

Preparation of soil collars

On May 19, 2019, three PVC collars with fine nylon fabric fixed at one end (as described above for the preliminary experiment) were packed with sieved and dried Skaha soil (~ 780 g) to a bulk density of 1.50 g cm^{-3}, leaving a 4-cm space between the top of the soil surface and the rim of the collar. Although the original bulk density of the Skaha soil used for this study was not known, it has the same texture as Osoyoos soils in a nearby orchard, which had bulk densities ranging from 1.28 to 1.65 g cm^{-3}.

Analysis of CO_2-C respiration rate and ^{13}C isotopic composition

After packing the collars, the soils were randomly assigned to one of five gravimetric moisture contents (10%, 15%, 20%, 25%, or 30%) and moistened accordingly, using deionized water. As described above for the preliminary experiment, soils were incubated at room temperature for a 10-day stabilization period. During this time, moisture contents were maintained by periodically re-weighing each collar and adding deionized water as required.

After the 10-day stabilization period had ended, a Perspex^{TM} chamber was secured to the top of each soil collar (Figure 2.2). The top of each chamber had three holes. One hole was connected

to a fully depressed BD 60 ml Luer Slip Tip Syringe, and the other two holes were connected to a single Kynar bag (Keika Ventures, Chapel Hill, NC, USA) filled with CO_2-free air; both the syringe and the Kynar bag were connected to the holes in the top of the chamber via Bev-a-line tubing. The connections between the syringe and the chamber, and the Kynar bag and the chamber, could be opened or closed using two-way valves, but both were closed when the chamber was secured in place. After the chamber was attached to the soil collar, one of the two holes connected to the Kynar bag was opened and a line of CO_2-free air was inserted into the hole. Atmospheric air was removed from the chamber through the second hole. After one minute, the hole was closed again and the collar + chamber was then incubated at room temperature for 15 hours (pre-irrigation period).

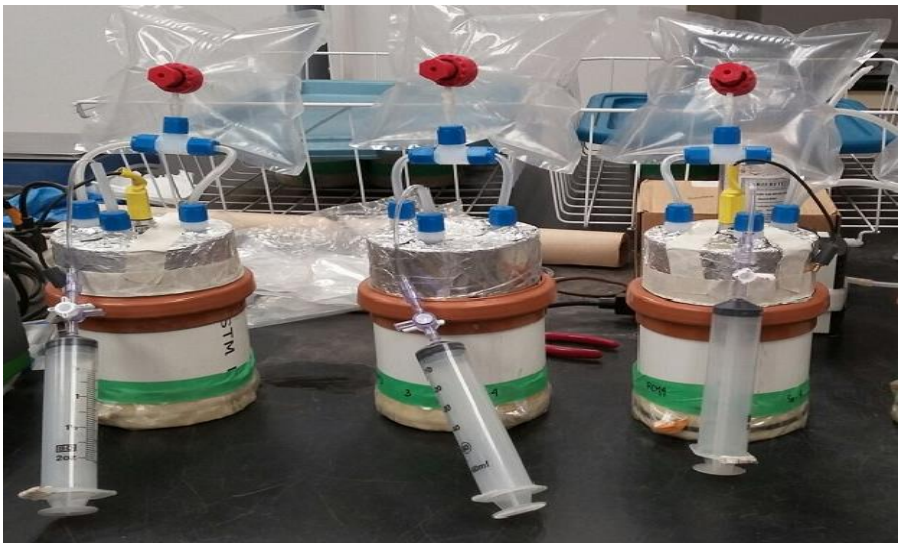

Figure 2. 2 Experimental set-up, showing foil-covered Perspex chambers secured on top of PVC soil collars, with syringes and air-filled Kynar bags in place.

After the pre-irrigation period had passed, the valves between the air-filled Kynar bag and the chamber and between the syringe and the chamber were opened, and a 10-ml gas sample from the chamber air-space was collected using the syringe attached to the top of the chamber. The air in the Kynar bag was used to prevent the build-up of negative pressure within the chamber during sample removal, and consequent suction of atmospheric air into the chamber through the

bottom of the collar. After gas sampling was complete, the valves connecting the chamber to the Kynar bag and the syringe were re-closed.

The 10-ml gas sample was injected into an environmental CO_2 gas monitor (EGM-4) to determine the CO_2 concentration. This value was used to calculate the baseline (pre-irrigation) CO_2 efflux rate (µmol m^{-2} s^{-1}). To measure the $\delta^{13}CO_2$ (‰) of the chamber air-space, an additional gas sample was collected from the chamber using the method described above. The volume of the sample varied from 36 ml to 125 ml, depending on the value given by the EGM-4. Because the Picarro G2131-*i* wavelength-scanned cavity ring-down spectrometer (Picarro, Santa Clara, CA, USA) cannot accurately measure the $\delta^{13}CO_2$ (‰) of highly concentrated samples, this second gas sample was diluted to obtain a target concentration of approximately 400 ppm, via injection of the sample gas into a second Kynar bag that had been pre-filled with CO_2 free air. The bag containing the diluted gas sample was connected to the Picarro for analysis of the $\delta^{13}CO_2$.

Once pre-irrigation measurements of CO_2 were complete, the chambers were flushed again with the CO_2-free air for one minute, as described above. Sixty millilitres of one of three water types (Okanagan Lake water, artificial lake water, or control water) was applied to the soil within the chamber using the syringe; all of the three collars in a run received the same type of irrigation water [note: one replicate of 25% soil moisture treatment (single soil collar) was run separately both during the Okanagan Lake water and the artificial lake water treatments (Table 2.2)]. It should be noted that the displaced soil air caused by the addition of irrigation water was not compensated. In future research, this should be accommodated. After adding water, the collars were incubated for an additional eight hours (post-irrigation) at room temperature.

After the post-irrigation period was complete, gas samples were collected and analyzed as described above. The difference between the pre-irrigation and post-irrigation measurements indicated the effect of the addition of water on the CO_2 efflux rate and $\delta^{13}C$ composition. Given the long time between soil collar set up, stabilization, and final measurements, and the fact that only three soil collars could be accommodated at one time, this experiment required 4.5 months to complete (May 19 to September 30, 2019) (Table 2.2). It should also be noted that the

original plan had been to test three moisture levels (10%, 15%, and 20%) but after observing initial results, 25% and 30% moisture contents were added to the experimental design.

Table 2. 2 The duration and number of replicates for each level of soil moisture and type of water tested

Water type	**Gravimetric soil moisture (%) before irrigation**	**Gravimetric soil moisture (%) after irrigation**	**Number of replicates**	**Measurement dates**
Okanagan Lake water	10	17.7	5	June 24 to July 26, 2019
	15	22.7	5	
	20	27.7	8	
	25	32.7	4	
	30	37.7	3	
Artificial lake water	10, 15, 20, and 30	17.7, 22.7, 27.7, and 37.7	3	May 19 to August 1, 2019
	25	32.7	4	
Control water	10, 15, 20, 25, and 30	17.7, 22.7, 27.7, 32.7, and 37.7	3	July 31 to September 30, 2019

Data analyses

Change in CO_2 efflux and $\delta^{13}CO_2$ after irrigation

The net change in CO_2 efflux rate and its $\delta^{13}CO_2$ composition were calculated by subtracting the pre-incubation measurement from the post-incubation measurement for each chamber such that values below zero indicated that applications of irrigation water had caused the $\delta^{13}C$ or CO_2 efflux rate to decrease and values above zero indicated that applications of irrigation water had caused the $\delta^{13}C$ or CO_2 efflux rate to increase.

Although I had initially planned to examine the interactive effect of water quality and soil moisture on DIC-derived CO_2 efflux, it was not appropriate to perform a two-way ANOVA

using these data because the water quality treatments had not been randomized during each run. Consequently, after confirming that assumptions were met (log transformation was required for the $\delta^{13}CO_2$ data of the Okanagan Lake water treatment), separate one-way ANOVAs were used to determine the individual effects of the five soil moisture levels on the net change in CO_2 efflux rate and its $\delta^{13}CO_2$ composition for the Okanagan Lake and control water treatments. Assumptions of normality and homogeneity of variance were tested using the Shapiro-Wilk (Q-Q plots) and Levene's tests, respectively. To determine differences among the treatments, Tukey tests were performed. Differences between treatment means were considered statistically significant at $P<0.05$. Statistical analysis was not performed for the artificial lake water treatment because runs for this water type occurred over a long period of time (Table 2.2) and the soil moistures were evaluated at very different times. Other factors, such as room temperature and performance of the analytical equipment, may also have varied over this time. Therefore, the data from the artificial lake water treatment are not presented.

To test whether the change in CO_2 efflux rate and its $\delta^{13}CO_2$ composition after irrigation was greater than or less than zero, individual one-tailed t-tests were performed separately for all of the five soil moisture levels irrigated with control or Okanagan Lake water.

CO_2 efflux from HCO_3^- in irrigation water

The proportion of total soil surface CO_2 efflux that was emitted from DIC in the irrigation water from Okanagan Lake (P_{bicarb}) was determined using equation 5, derived from Hannam et al. (2019), where the $\delta^{13}C_{bicarb}$ of the Okanagan Lake water was assumed to be -2.6‰ (Hannam et al., 2016), the $\delta^{13}C_{baseline}$ was the $\delta^{13}CO_2$ of the chamber gas space after the pre-irrigation period, and the $\delta^{13}C_{efflux}$ was the $\delta^{13}CO_2$ of the chamber gas space after the post-irrigation period (i.e., after the application of Okanagan Lake water). An example calculation demonstrating how equation 5 was used to calculate the quantity of CO2 released from irrigation water-derived DIC can be found in Appendix 1.

$$P_{bicarb} = (\delta^{13}C_{efflux} - \delta^{13}C_{baseline}) / (\delta^{13}C_{bicarb} - \delta^{13}C_{baseline}) \quad (5)$$

By using this mass balance calculation, we have assumed conservative mixing. This is an over-simplication, which is unlikely to be completely accurate, given that the added water (whether Okanagan Lake water or control water) is undersaturated with respect to DIC and, therefore, likely to absorb soil CO_2. In cases where the amount of CO_2 emitted in the treatment is greater than the baseline flux, however, these calculations provide a mass-weighted $\delta^{13}C$ value for the amount of 'extra' CO_2 released. It should also be noted that the amount of DIC in the soil water solution prior to irrigation was not accounted for, because Hannam et al. (2016) found very little water-extractable inorganic carbon in surface soils collected from similar sites.

A one-way ANOVA was performed to determine the effect of soil moisture on the proportion of Okanagan Lake DIC released as CO_2. Individual one-tailed t-tests were performed for each of the soil moisture levels irrigated with Okanagan Lake water to test whether the proportion of the DIC in the Okanagan Lake water that was detected in CO_2 efflux was greater than or less than zero. All analyses were performed using R 3.6.1 (Ihaka and Gentleman, 1992).

Results

Change in CO_2 efflux and $\delta^{13}CO_2$ after irrigation

Irrigation caused larger increases in CO_2 efflux in drier soils than in wetter soils (Figure 2.3; Table 2.3). When soils were irrigated with Okanagan Lake water, efflux increased more in soils at 10% soil moisture than in soils at 15% or 30% soil moisture; changes in efflux following application of Okanagan Lake water to soils at 20% and 25% soil moisture were intermediate. According to the one-tailed t-tests, irrigation with Okanagan Lake water resulted in a significant increase in CO_2 efflux rate in soils at 10% ($p < 0.001$), 15% ($p < 0.001$), 20% ($p < 0.001$), and 25% ($p = 0.041$) moisture levels, but not at 30% moisture ($p = 0.129$). When soils were irrigated with control water, differences in efflux increased in the order: 10% = 15% = 20% > 25% > 30%. According to the one-tailed t-tests, however, the CO_2 efflux rate increased compared to pre-irrigation in soils at 10% ($p = 0.001$), 15% ($p < 0.001$), and 20% ($p = 0.003$) moisture levels. The CO_2 efflux did not change in soils at 25% ($p = 0.068$) and 30% ($p = 0.080$) moisture levels when irrigated with control water.

Before irrigation, the $\delta^{13}CO_2$ value was -19 ± 1.4‰ for soil with 10% moisture, -18.3 ± 1.8‰ for soil with 15% moisture, -19.8 ± 2.1‰ for soil with 20% moisture, -20.2 ± 2.2‰ for soil with 25% moisture, and -22.6 ± 1.2‰ for soil with 30% moisture (Appendix 2). When the soils were irrigated with Okanagan Lake water, the net change in $\delta^{13}CO_2$ was more positive in soils maintained at 15%, 20%, 25%, and 30% moisture than in soil at 10% moisture (Figure 2.4; Table 2.4; Appendix 2). According to the one-tailed t-test, the change in $\delta^{13}CO_2$ was less than 0 in the soil at 10% ($p = 0.011$). The changes in $\delta^{13}CO_2$ did not differ among the 15%, 20%, 25%, and 30% moisture levels. According to the one-tailed t-tests, however, the changes in $\delta^{13}CO_2$ were greater than 0 only in soils at 15% ($p = 0.003$) and 25% ($p < 0.001$) moisture levels. The greatest increase in $\delta^{13}CO_2$ (by approximately 4.5‰) was observed in soil maintained at 25% moisture. In soil maintained at 15% soil moisture, the $\delta^{13}CO_2$ increased by approximately 1.3‰. According to the one-tailed t-tests, when the soils were irrigated with control water, the $\delta^{13}CO_2$ declined in soils maintained at 10% ($p = 0.009$), 15% ($p = 0.001$), and 20% (0.005) soil moisture, but increased by approximately 5.5‰ in soil maintained at 30% ($p = 0.008$) moisture. The $\delta^{13}CO_2$ did not change in soil maintained at 25% moisture ($p = 0.110$).

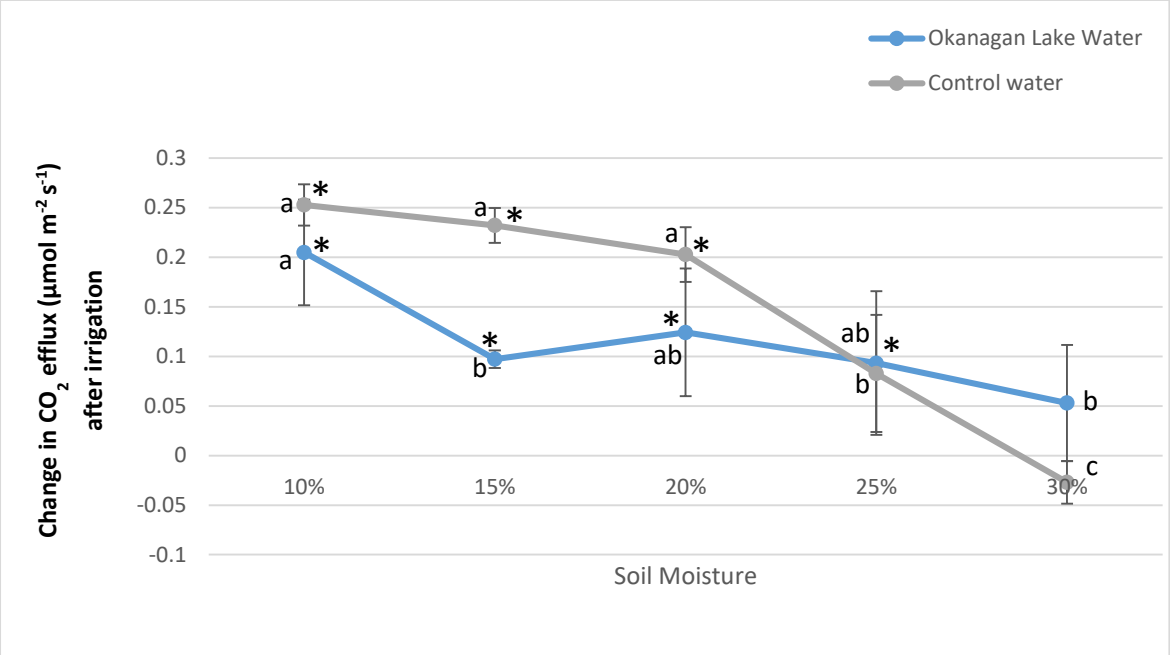

Figure 2. 3 The effect of soil moisture on the change in CO_2 efflux (µmol m^{-2} s^{-1}) after application of Okanagan Lake water or control water. Solid circles represent the treatment means (n = 3 to 8) and error bars show the standard deviations. Within irrigation water treatments, different letters indicate significant differences ($p<0.05$) among soil moisture treatments (Tukey test). Asterisks indicate that the changes in CO_2 efflux (µmol m^{-2} s^{-1}) were significantly greater than zero, according to one-tailed t-tests.

Table 2. 3 Results of one-way ANOVAs on the effects of soil moisture on the change in CO_2 efflux (µmol m^{-2} s^{-1}) after application of Okanagan Lake water or control water

Water quality	**df**	**F**	**p**
Okanagan Lake water	4	4.346	**0.011**
Control water	4	38.55	**<0.001**

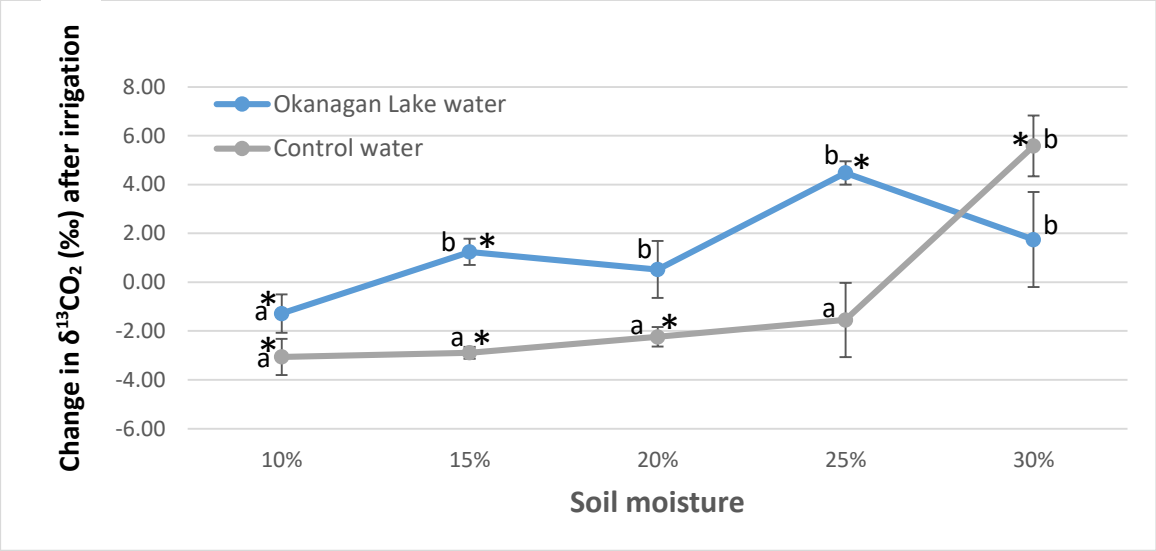

Figure 2. 4 The effect of soil moisture on the change in $\delta^{13}CO_2$ (‰) after application of Okanagan Lake water or control water. Solid circles represent the treatment means (n = 3 to 8) and error bars show the standard deviations. Within irrigation water treatments, different letters indicate significant differences ($p<0.05$) among soil moisture treatments (Tukey test). Asterisks indicate that the changes in $\delta^{13}CO_2$ (‰) were significantly greater than or less than zero, according to one-tailed t-tests.

Table 2. 4 Results of one-way ANOVAs on the effects of soil moisture on the change in $\delta^{13}C$ (‰) of CO_2 efflux after application of Okanagan Lake water or control water

Water quality	**df**	**F**	**p**
Okanagan Lake water	4	9.671	**<0.001**
Control water	4	42.9	**<0.001**

CO_2 efflux from HCO_3^- in irrigation water

In general, the proportion of DIC in the Okanagan Lake water released as CO_2 during the measurement period increased with increasing soil moisture (Figure 2.5). Values were greatest in soils maintained at 25% moisture, lowest in soils maintained at 10% soil moisture, and intermediate in soils maintained at 15% and 20% soil moisture. However, according to one-tailed t-tests, CO_2 derived from DIC in the irrigation water was detected only in soils maintained at 15% ($p = 0.003$) and 25% ($p = 0.010$) soil moisture. Overall, an average of 1.8 % of the DIC dissolved in the Okanagan Lake water was released as CO_2. The Okanagan Lake water is not

saturated with respect to calcite according to PHREEQC. Therefore, it is unlikely to produce PC and CO_2 from the water when applied to the soil. However, previous field experiments conducted by Hannam et al. (2016, 2019) and this laboratory experiment clearly show that some of the CO_2 released after irrigation is derived from DIC in the water drawn from Okanagan Lake.

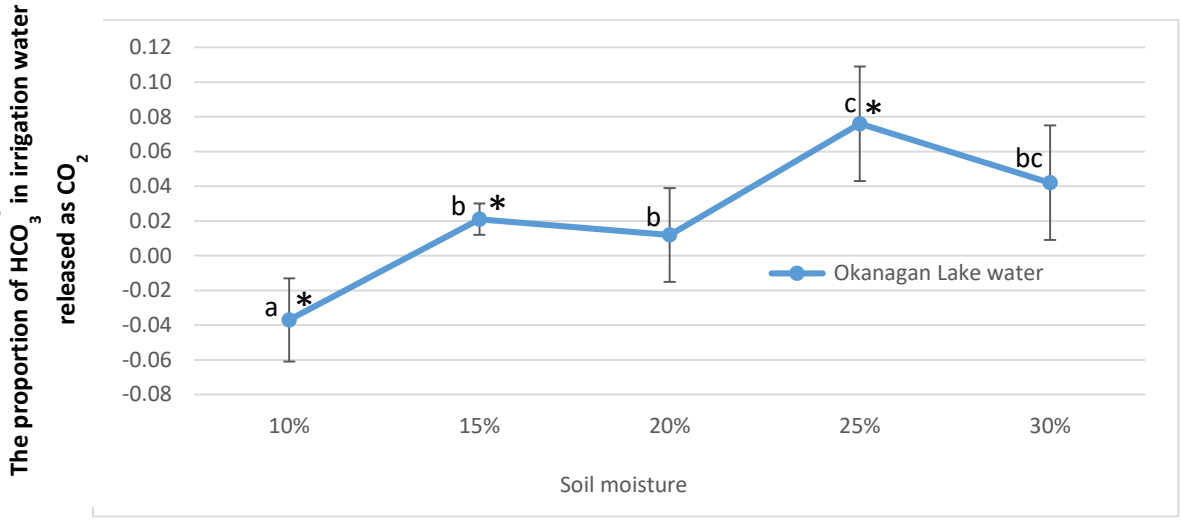

Figure 2.5 The effect of soil moisture on the proportion of DIC in Okanagan Lake irrigation water that was released as CO_2-C (one-way ANOVA with soil moisture as the fixed factor: df = 4, F = 11.36, and $p < 0.001$). Solid circles represent the treatment means (n = 3 to 8) and error bars show the standard deviations. Different letters indicate significant differences ($p<0.05$) among the soil moisture treatments (Tukey test). Asterisks indicate that the proportion of DIC that was released as CO_2-C was significantly greater than or less than zero, according to one-tailed t-tests.

Discussion

Changes in the rate and ^{13}C composition of soil CO_2 efflux following the addition of irrigation water depend on a complex interplay between biological processes (e.g., soil organic matter decay and microbial respiration), chemical processes (e.g., dissolution of CO_2 into soil solution, dissolution of PC ($CaCO_3$)), and physical processes (e.g., downward movement of soil solution through the soil column, physical blockage of soil pores by excess soil moisture, expulsion of soil air from soil pore space by percolating irrigation water). The relative importance of each of these processes in determining the response of soil CO_2 efflux to irrigation events depends on a

variety of factors, including soil moisture and the chemical composition of the water added during irrigation.

The efflux rate tended to increase after irrigation; however, post-irrigation increases in efflux rate tended to be greater in dryer soils than in wetter soils (Figure 2.3; Table 2.3). With respect to the $\delta^{13}CO_2$ data, irrigation with Okanagan Lake water tended to cause ^{13}C enrichment of CO_2 efflux, particularly with wetter soils. Irrigation with control water tended to cause ^{13}C depletion of CO_2 efflux, particularly in dryer soils. Regardless of water type, however, irrigation of soils at 30% moisture caused ^{13}C enrichment of CO_2 efflux (Figure 2.4; Table 2.4). These patterns did not support my original hypothesis. I proposed that drier soil would release more irrigation-derived CO_2 than wetter soil because drier soil would have a lower amount of organically derived CO_2 dissolved in the soil solution than wetter soil upon the addition of irrigation water. I hypothesised that the lower concentrations of dissolved biogenic CO_2 in drier soil would encourage the release of CO_2 from the HCO_3^- ions applied in the irrigation water (Equation 1 to the right).

When considering the effects of soil moisture on soil CO_2 efflux it is particularly important not to overlook biological CO_2 production. Application of irrigation water to soils with only 10% moisture may have caused a sudden increase of microbial activity and hence, a rapid increase in CO_2 production. The addition of water to dry soils may also have expelled relatively more biogenic CO_2 from air-filled soil pore spaces than from wetter soils. Therefore, a relatively large amount of CO_2 (derived from organic sources) may have accumulated inside the chamber in response to irrigation of dry soils, causing the apparently large increase in ^{13}C-depleted CO_2 efflux (Figure 2.3; Figure 2.4; Table 2.4). Increased microbial activity and expulsion of biogenic CO_2 from soil pores also occurred in soils maintained at 15%, 20%, and 25% moisture, but the effect was less strong (Figure 2.3) because the microbes already had enough water to respire and because more pores were already occupied by water compared to the soils maintained at 10% soil moisture.

The accumulation of ^{13}C-depleted CO_2 following application of control water to soils maintained at 10%, 15%, and 20% moisture (one-tailed t-tests; Figure 2.4) suggests that the control water stimulated the release of biogenic CO_2 [biogenic CO_2 is relatively depleted in ^{13}C (Smith and Epstein, 1971; Dzurec et al., 1985; Pawellek and Veizer, 1994; Stevenson and Verburg, 2006;

Bertrand et al., 2007; Tamir et al., 2011)]. However, the lack of change in $\delta^{13}CO_2$ for soil maintained at 25% moisture content (one-tailed t-test; Figure 2.4) indicates that some ^{13}C-enriched CO_2 may have been released from the dissolution of pre-existing PC, which could have counteracted the effect of ^{13}C depleted biogenic CO_2 on the overall change in $\delta^{13}CO_2$ after irrigation.

Regarding the Okanagan Lake water treatment, the increase in CO_2 efflux rate (one-tailed t-tests; Figure 2.3) but accumulation of ^{13}C-depleted CO_2 following irrigation in soil maintained at 10% (one-tailed t-tests; Figure 2.4) suggests that the Okanagan Lake water stimulated the release of relatively large quantities of biogenic CO_2. On the other hand, the increase in CO_2 efflux rate (one-tailed t-tests; Figure 2.3), and ^{13}C enrichment of the CO_2 following irrigation in soil maintained at 15% and 25% moisture contents (one-tailed t-tests; Figure 2.4) indicates that the CO_2 released from the soil surface was derived at least in part from dissolved DIC applied in the water. The application of Okanagan Lake water also increased the CO_2 efflux rate from soil maintained at 20% moisture content (one-tailed t-test; Figure 2.3), however, the $\delta^{13}CO_2$ did not change following irrigation (one-tailed t-test; Figure 2.4); some ^{13}C-rich CO_2 may have been released from pre-existing PC but this may have been counteracted by the release of ^{13}C-poor biogenic CO_2, resulting in no net change in $\delta^{13}CO_2$.

Given that both the CO_2 efflux rate and its $\delta^{13}CO_2$ composition increased after Okanagan Lake water was applied to the soils maintained at 15% and 25% moisture contents (one-tailed t-tests; Figure 2.3 and 2.4), the proportion of DIC released as CO_2 from Okanagan Lake water was also determined in soils maintained at these two moisture contents (one-tailed t-tests; Figure 2.5). Contrary to my hypothesis, the proportion of DIC released was higher at 25% soil moisture than at 15% soil moisture (Figure 2.5). The 25% moisture content of the experimented soil corresponded to ~ 63% water-filled pore space, which went up to ~ 76% after irrigation water was applied. The 15% moisture content, on the other hand, corresponded to ~ 42% water-filled pore space, which went up to ~ 58.5% after irrigation. According to Doran et al. (1990), microbial respiration increases with increasing water-filled pore space and reaches the maximum at 61% water-filled porosity (in coarse-textured soils) after which the respiration rate starts to decrease. Therefore, more biogenic CO_2 can be assumed to be produced when irrigation water was applied to the soil maintained at 15% moisture than to the soil maintained at 25% moisture

in the present study. This may explain why the proportion of DIC released from irrigation water was higher at 25% soil moisture than at 15% soil moisture. Nevertheless, the reason for no CO_2 from DIC in irrigation water when applied to soil maintained at 10% moisture and 20% moisture is unknown.

With respect to the 30% soil moisture treatment, no CO_2 derived from DIC in irrigation water was detected (Figure 2.5). Two factors may explain this observation. First, the release of CO_2 from the soil surface may have been disrupted due to the blockage of soil pores caused by excess moisture after irrigation water was applied to the soil (Davidson et al., 1998; Bouma and Bryla, 2000). Furthermore, the respiration of soil biota may have been severely hampered due to the lack of O_2 when irrigation water of any type was added to the wettest soil treatments. This finding corroborates Sharma et al. (2011), who found lower rates of soil respiration when experimental fields were irrigated with supra-optimum/excessive water. According to Linn and Doran (1984) and Doran et al. (1990), soil respiration falls to a minimum level when water-filled porosity exceeds 80%. When irrigation water in the present experiment was added to the soil with 30% moisture (~ 71.5% water-filled porosity), the moisture content went up to ~ 38%, which caused ~ 83% of the soil pores to be filled with water leaving only ~ 17% air-filled pores for the soil biota to respire. As a result, the change of CO_2 efflux values was almost zero when irrigated with Okanagan Lake water (negative for control water) (Figure 2.3). That being said, the reason for the increase in $\delta^{13}CO_2$ after irrigation with control water of soil maintained at 30% soil moisture (Figure 2.4) is unknown.

The overall findings of this experiment are not fully in agreement with the field experiment conducted by Hannam et al. (2019). In the present experiment, DIC in the applied irrigation water contributed to ~10% of the total CO_2 emissions during the irrigation events, which is supported by Hannam et al. (2019) as they found a similar contribution of the DIC (between 9 and 15%) to the total CO_2 evolved during irrigation. According to their findings, however, a greater amount of irrigation-derived DIC was released as CO_2 from the drier soil away from the dripper (~ 10% gravimetric moisture content) than the wetter soil under the dripper (~25% gravimetric moisture content), which did not agree with the findings of this experiment. The reason for this discrepancy might be due to a few differences in the experimental set-up between this laboratory experiment and their field experiment. Although the characteristics of the soils

(e.g., soil pH, soil texture, and organic matter content) used in both experiments were similar, the soil in this experiment was oven-dried at 65 to 70 ℃ and sieved through a 2 mm sieve prior to experimentation. As the soil was oven-dried, deionized water was applied to artificially moisten the soil to selected gravimetric water content before irrigation treatment was initiated. In this experiment, the water in the soil collar was unable to move downward because a fine nylon mesh was attached to the bottom of each collar and the collars were placed on a bench (Figure 2.2). This was not the case in the study carried out by Hannam et al. (2019), where the collars were inserted into intact soil and the bottom of the collars underneath the soil was open to allow the free flow of irrigation water (Figure 2.7).

Figure 2. 6 Experimental set-up (Hannam et al., 2019)

As the Okanagan Valley soils had been irrigated with Okanagan Lake water for many years, there were already more carbonates accumulated at the soil surface under dripper compared to the soil away from dripper (Hannam et al., 2016). Therefore, when 60 ml of Okanagan Lake water was applied to the soil under the dripper, the pre-existing carbonates might have discouraged the neoformation of carbonates and CO_2 from the irrigation water. By contrast, the lack of carbonates at the surface in the drier soil (away from the dripper) compared to that in the wetter soil might have encouraged the neoformation of carbonates and CO_2 (Equation 1 to the right) when 60 ml Okanagan Lake water was applied to the soil away from the dripper.

Moreover, the shortage of soil moisture in the drier soil might have discouraged the downward leaching and precipitation of the DIC that came from irrigation water or the dissolved carbonates, which might also have resulted in more DIC-derived CO_2 to be released from the drier soil than the wetter soil.

The DIC-derived CO_2 that was released from the soil maintained at 25% soil moisture in the present experiment, was greater than that released from the soil with 25% moisture used by Hannam et al. (2019). Again, this might be due to the difference in soil conditions between the two experiments. As the soil used in this experiment was sieved, it did not have any visible organisms except a few pieces of very fine dead plant roots. By contrast, the soil under the dripper (wetter soil) in Hannam et al. (2016) had living plant roots, microbes, and macro-organisms (e.g., earthworm). Therefore, the soil under the dripper probably had more biogenic CO_2 than the soil away from the dripper, which might have caused a smaller amount of CO_2 release from the DIC in irrigation water. The lower pCO_2 in the sieved soil in this experiment might have encouraged equation 1 to move to the right and hence, resulted in more DIC-derived CO_2 from irrigation water when compared to that in the field soil experimented by Hannam et al. (2019).

Regarding the drier soil, DIC-derived CO_2 was not detected from the soil maintained at 10% moisture in the present experiment (Figure 2.5) whereas, Hannam et al. (2019) found a significant amount of DIC in the irrigation water to be released as CO_2 from their drier soil (~ 10% moisture). The temperature in the field ranged from 24.4 to 37.4 ℃ (Environment Canada, 2018) when Hannam et al. (2019) conducted their study whereas, the temperature in the lab was 21 ℃ during this experimentat. This might have led to CO_2 off-gassing immediately after applications of irrigation water in the field, which would have been less likely in this lab study.

Finally, the post-irrigation values in this experiment were recorded 8 hours after irrigation whereas Hannam et al. (2019) monitored the changes of $\delta^{13}CO_2$ and CO_2 efflux every second for 50 minutes, starting immediately after irrigation. Hannam et al. (2019) observed a spike in $\delta^{13}CO_2$ within five minutes of irrigation, after which the $\delta^{13}CO_2$ started to decrease and returned to the pre-irrigation value after 50 minutes. As there was a decreasing trend, the $\delta^{13}CO_2$ could have decreased further with time, which may have corroborated what was found after 8 hours of

irrigation in the present experiment (Figure 2.4). However, as that had not been monitored, it is difficult to draw any firm conclusion. Further study is required to understand how the HCO_3^- in irrigation water affects the release of CO_2 from dry soil over time.

One drawback of the present experiment is that only the soil moisture treatments, and not irrigation water types, were randomized during each 'run' of three collars. Using only one type of irrigation water per 'run' might have caused systematic errors associated with changes in the chemistry of the water sources or changes in the experimental conditions during that 'run', e.g., room temperature, functioning of the analytical equipment. Ideally, both soil moisture and irrigation water chemistry treatments would have been randomly applied during each run or, even better, one replicate of all moisture content x irrigation water chemistry treatments would have been measured during each 'run'; each 'run' could then have been treated as a 'block' in the statistical analysis. Furthermore, it would have been better if a 100% deionized water treatment was also used as control, given that Okanagan Lake water typically contains a negligible amount of Na^+ and Cl^- ions (Table 2.1). However, it should be noted that the Skaha soil used in this experiment was also used in the second experiment described in Chapter 3, with 100% deionized water as control and the changes in $\delta^{13}C$ of CO_2 following irrigation with the 100% deionized water were similar to those observed after irrigating with NaCl-containing water in the current soil moisture experiment.

Another potential drawback of this experiment is the possible intrusion of atmospheric air into the soil collar through the fine nylon fabric, which was fixed at the bottom of the collar. This might be the reason that some CO_2 values were unexpectedly enriched in ^{13}C after the pre-incubation period. In subsequent experiments, a plastic sheet was fixed at the bottom of the collar, in addition to the fine nylon fabric, to prevent intrusion of ^{13}C-rich atmospheric air.

In conclusion, although I could not conclude that DIC in the irrigation water were released as CO_2 at 10%, 20%, and 30% moisture levels, due to high variability in the data for those treatments, some DIC was released as CO_2 at 15% and 25% soil moisture levels (Figure 2.5). According to Fentabil et al. (2016), irrigated Okanagan Valley soils often have 42, 53, or 63% water-filled porosities, which are equivalent to 15, 20, and 25% gravimetric moistures, respectively. Clearly, irrigation with Okanagan Lake water can contribute to CO_2 efflux from

Okanagan Valley soils, and the effect is mediated by soil moisture content. Therefore, the frequency of irrigation and the quality of irrigation water are two important considerations when implementing efficient irrigation practices that reduce CO_2 emissions from agricultural fields.

Chapter 3: The Effect of Irrigation Water Chemistry and Soil Temperature on CO_2 Production

3.1 Synopsis

Soil temperature is expected to play an important role in determining the proportion of irrigation water-derived DIC that is released from the soil surface as CO_2. The relationship between soil temperature and soil CO_2 efflux is particularly complex (Amit et al., 2011) because both biotic and abiotic processes play a role.

$$Ca^{2+}_{(aq)} + 2HCO_3^-{}_{(aq)} \longleftrightarrow CaCO_{3\,(s)} + H_2O_{(l)} + CO_{2(g)} \quad (1)$$

$$CaCO_{3\,(s)} + H^+_{(aq)} \longleftrightarrow Ca^{2+}_{(aq)} + CO_{2(g)\,)} + H_2O_{(l)} \quad (2)$$

$$H^+_{(aq)} + HCO_3^-{}_{(aq)} \longleftrightarrow H_2CO_{3\,(aq)} \longleftrightarrow H_2O_{(l)} + CO_{2(g)} \quad (3)$$

Higher soil temperatures typically increase microbial activity and, consequently, biological CO_2 production (Davidson et al., 1998; Fang et al., 1998; Lin et al., 1999; Lal and Kimble, 2000; Fang and Moncrieff, 2001; Qi and Xu, 2001; Dilustro et al., 2005; Hamdi et al., 2011; Litton et al., 2011; Schindlbacher et al., 2011; Ferrea et al., 2012; Karhu et al., 2014; Hicks Pries et al., 2017; Robinson et al., 2017; Bamminger et al., 2018; Bhanja et al., 2019; Chen et al., 2021) as long as soil moisture is adequate (Yuste et al., 2003; Chang et al., 2014). However, higher soil temperatures can also promote off-gassing of CO_2 from solution. Furthermore, the rate of reaction between the Ca^{2+} and HCO_3^- ions provided by the irrigation water (Equation 1 to the right) is expected to increase at higher temperature, leading to greater and more rapid production of DIC-derived CO_2 during precipitation of carbonates. H^+ ions released from organic acids or from other acidifying reactions (e.g., nitrification) are also promoted by higher temperature (Tan et al., 2018; Nguyen et al., 2019) and could directly dissolve pre-existing PC (Equation 2 to the right) or could react with HCO_3^- ions formed from the PC or applied in the irrigation water to produce H_2O and CO_2 (Equation 3 to the right) (Bertrand et al., 2007; Tamir et al., 2011). Taking all of these processes together, the release of CO_2 from PC dissolution and irrigation water is expected to increase with increasing temperature.

In Chapter 2, I discussed an experiment that examined the effect of soil moisture on the proportion of DIC released as CO_2, and found that a greater proportion of DIC from the

irrigation water was released as CO_2 from wetter soils than drier soils. In the experiment described in Chapter 3, my objective was to examine the role of soil temperature in determining the proportion of DIC in Okanagan Lake water that was released as CO_2. I predicted that the release of CO_2 from irrigation water-derived DIC would be higher at warmer temperature than in the soil at cooler temperature. The CO_2 efflux rate (μmol m^{-2} s^{-1}) and $\delta^{13}CO_2$ (‰) were measured before and after irrigation water of two different chemistries was applied to soils that had been maintained at 10% gravimetric moisture content at either room temperature (21 ℃) or 30 ℃.

3.2 Methodology

Collection and storage of soil and water samples

Oven-dried and sieved Skaha soil, similar to that used in the experiment described in Chapter 2, was used for the current study. Two types of water were applied as irrigation treatments: Okanagan Lake water or deionized water (as the control). Unlike the experiment described in Chapter 2, no NaCl was added to the deionized water used as the control treatment. The Okanagan Lake water was collected from the lake shore on Cadder Avenue, Kelowna, on 25 October 2019. Both the Okanagan Lake water and the deionized water were stored in glass bottles and kept in a walk-in refrigerator (5 ℃) until use (no headspace gas was used).

Preparation of soil collars

On October 17, 2019, four PVC collars were packed with Skaha soil, as described in Section 2.2.2 except that both a fine nylon fabric (0.33 mm openings) and a piece of plastic sheet cut from a food storage bag (Ziploc™ Freezer Bags) were fixed to the bottom of each soil collar using masking tape. The nylon fabric was used to hold the soil in the collar and the plastic sheet was used to prevent the intrusion of atmospheric air into the soil from the bottom of the soil collar.

Analysis of CO_2-C respiration rate and ^{13}C isotopic composition

After packing the collars, the soils were moistened with deionized water to achieve a 10% gravimetric moisture content. Each collar was assigned to one of four treatments: i. 'low temp' soil and deionized water; ii. 'low temp' soil and Okanagan Lake water, iii. 'high temp' soil and deionized water, iv. 'high temp' soil and Okanagan Lake water. During each 10-day stabilization period, the two collars assigned to treatments i. and ii. were placed on a rack at room temperature (21 ℃) and the two collars assigned to treatments iii. and iv. were kept in an incubator at 30 ℃. Soil collars were re-weighed every day over this period, and deionized water was added to maintain a 10% moisture content.

The soil CO_2 efflux and $\delta^{13}C$ composition were measured following the pre- and post-irrigation periods, as described in Section 2.2.2. The experiment comprised six runs, each taking approximately 11 days to complete. One replicate of each of the four experimental treatments was included in each experimental run. The collars were re-packed with fresh soil for each run, and allowed to stabilise for 10 days prior to taking pre-irrigation measurements. The experiment was started on October 28, 2019, and completed on December 10, 2019.

Data analyses

Change in CO_2 efflux and $\delta^{13}CO_2$ after irrigation

The net change in CO_2 efflux rate and its $\delta^{13}CO_2$ composition was calculated by subtracting the pre-incubation measurements from the post-incubation measurements, as described in Section 2.2.2. Assumptions of normality and homogeneity of variance were tested using the Shapiro-Wilk (Q-Q plots) and Levene's tests, respectively. When the assumptions were satisfied (log transformation was required for the CO_2 efflux data), two-way ANOVA was used to determine the effects of soil temperature, water quality, and their interactions on the net change in CO_2 efflux rate and its $\delta^{13}CO_2$ composition. When interactive effects were statistically significant, one-way ANOVAs on the effects of soil temperature or water quality on measured variables were performed. 'Run' was included as a random factor (i.e., analogous to a 'block' effect) in

both the two-way and one-way ANOVAs. Differences between treatment means were considered statistically significant at $p<0.05$.

Individual one-tailed t-tests were performed separately for both of the 'high temp' soil and 'low temp' soil irrigated with deionized or Okanagan Lake water to test whether the change in CO_2 efflux rate and its $\delta^{13}CO_2$ composition after irrigation was greater than or less than zero.

CO_2 efflux from HCO_3^- in irrigation water

The proportion of DIC dissolved in the irrigation water from Okanagan Lake that was subsequently detected as CO_2 efflux was determined using a simple isotopic mass balance model, as described in Section 2.2.2. A one-way ANOVA was then performed to test for an effect of soil temperature on the proportion of Okanagan Lake DIC released as CO_2. 'Run' was considered as a random factor. Individual one-tailed t-tests were performed for both the 'high temp' soil and the 'low temp' soil irrigated with Okanagan Lake water to test whether the proportion of DIC in the Okanagan Lake water that was detected as CO_2 efflux was greater than or less than zero. All analyses were performed using R 3.6.3 (Ihaka and Gentleman, 1992).

3.3 Results

Change in CO_2 efflux and $\delta^{13}CO_2$ after irrigation

CO_2 efflux rates increased after irrigation at both temperatures and with both types of irrigation water (one-tailed t-tests: $p < 0.001$ for both the 'high temp' and 'low temp' soils irrigated with either water). Regardless of irrigation water quality, the CO_2 efflux rate increased more when irrigation water was applied to the 'high temp' soil than to the 'low temp' soil (Figure 3.1; Table 3.1).

Before irrigation, the $\delta^{13}CO_2$ value was -23.7 ± 0.7‰ for 'high temp' soil and -21.1 ± 0.9‰ for the 'low temp' soil (Appendix 3). The effect of soil temperature on the $\delta^{13}C$ of CO_2 efflux depended on the chemistry of the water applied during irrigation of the cores (temperature X irrigation water interaction $p < 0.001$). Thus, one-way ANOVAs were used to examine the

individual effects of temperature or water quality on the $\delta^{13}C$ of soil CO_2 efflux. Irrigation with deionized water caused a greater decline in the $\delta^{13}CO_2$ in 'low temp' soil than in 'high temp' soil (Figure 3.2, Table 3.2). By contrast, temperature did not affect the change in $\delta^{13}CO_2$ when irrigated with Okanagan Lake water (Figure 3.2, Table 3.3).

The $\delta^{13}C$ of CO_2 efflux was affected by water quality at both temperatures (Figure 3.2, Table 3.4, Table 3.5). In 'low temp' soil, the $\delta^{13}C$ of CO_2 efflux declined by approximately 2.7‰ when deionized water was applied to the soil surface but showed no change when Okanagan Lake water was applied (one-tailed t-tests: $p < 0.001$ when irrigated with deionized water; $p = 0.389$ when irrigated with Okanagan Lake water). In 'high temp' soil, the $\delta^{13}C$ of CO_2 efflux declined by approximately 1.1 ‰ when deionized water was applied to the soil surface but, again, showed no change when Okanagan Lake water was applied (one-tailed t-tests: $p < 0.001$ when irrigated with deionized water; $p = 0.254$ when irrigated with Okanagan Lake water).

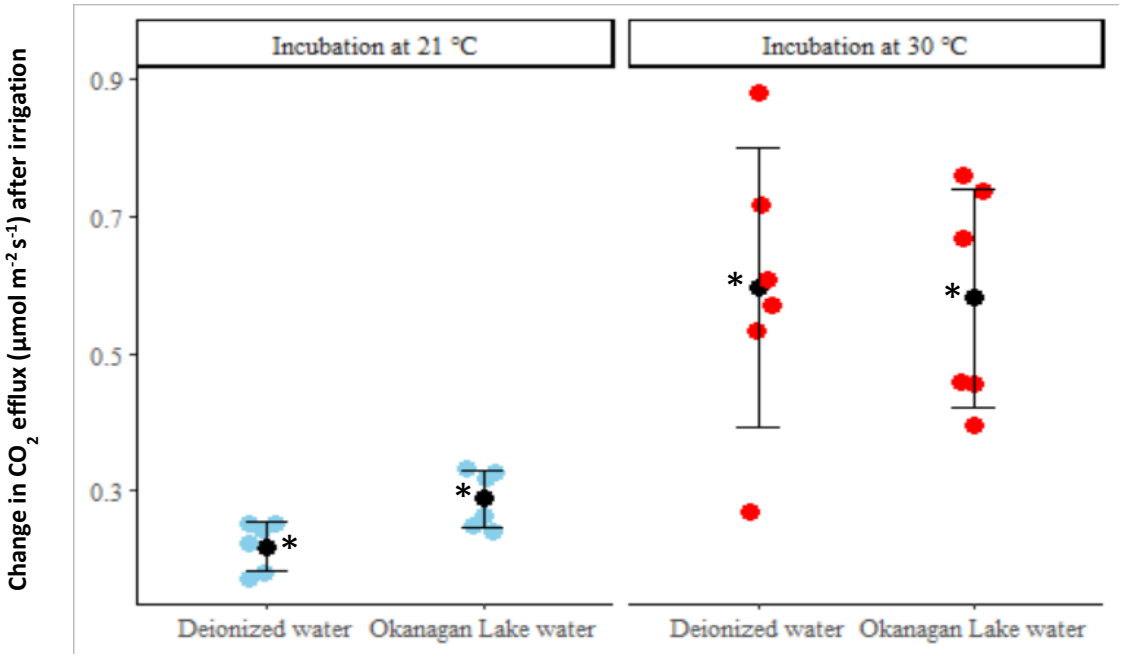

Figure 3. 1 The effect of soil temperature on the change in CO_2 efflux (µmol m^{-2} s^{-1}) after application of Okanagan Lake water or deionized water. Coloured circles represent individual data points for 'low temp' soil (blue circles) or 'high temp' soil (red circles). Black circles represent the treatment means (n = 6) and error bars show the standard deviations. Asterisks indicate that the treatment means of the change in CO_2 efflux were significantly greater than zero, according to one-tailed t-tests.

Table 3. 1 Results of two-way ANOVA of the effects of soil temperature and irrigation water quality on the change in CO_2 efflux (µmol m^{-2} s^{-1})

Factor	**Df**	**F**	**p**
Block	1	0.066	0.800
Soil temperature	1	52.424	**<0.001**
Water quality	1	1.563	0.226
Soil temperature x water quality	1	1.521	0.233

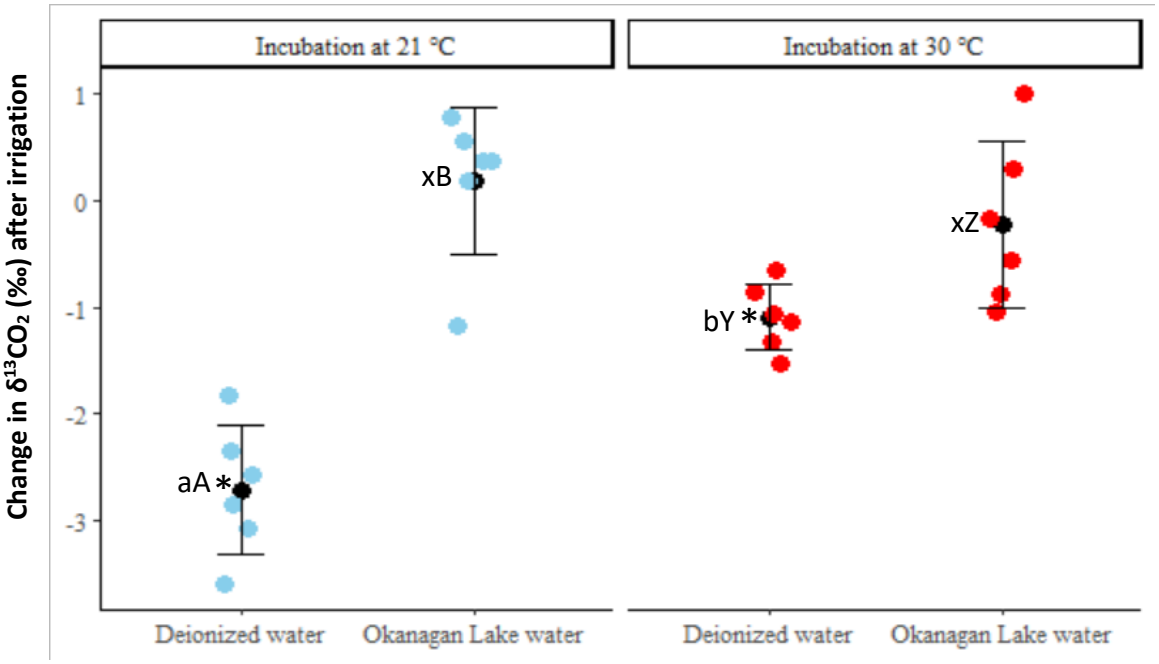

Figure 3. 2 The effect of soil temperature on the change in $\delta^{13}CO_2$ (‰) after application of Okanagan Lake water or deionized water. Coloured circles represent individual data points for 'low temp' soil (blue circles) or 'high temp' soil (red circles). Black circles represent the treatment means (n = 6) and error bars show the standard deviations. Asterisks indicate that the treatment means of the change in $\delta^{13}CO_2$ (‰) were significantly less than zero, according to one-tailed t-tests. Within irrigation water treatments, different lower-case letters indicate significant differences ($p<0.05$) between soil temperature treatments. Within soil temperature treatments, different upper-case letters indicate significant differences ($p<0.05$) between irrigation water treatments.

Table 3. 2 Results of separate one-way ANOVA of the effect of soil temperature on the change in $\delta^{13}CO_2$ (‰) after application of deionized water

Water quality	**Df**	**F**	**p**
Block	1	0.516	0.491
Deionized water	1	31.862	< **0.001**

Table 3. 3 Results of separate one-way ANOVA of the effect of soil temperature on the change in $\delta^{13}CO_2$ (‰) after application of Okanagan Lake water

Water quality	**Df**	**F**	**p**
Block	1	4.569	0.061
Okanagan Lake water (treatment differences are not statistically significant)	1	1.249	0.293

Table 3. 4 Results of separate one-way ANOVA of the effect of irrigation water quality on the change in $\delta^{13}CO_2$ (‰) in 'low temp' soil

Soil	**Df**	**F**	**p**
Block	1	1.348	**0**.275
'low temp' soil	1	61.073	< **0.001**

Table 3. 5 Results of separate one-way ANOVA of the effect of irrigation water quality on the change in $\delta^{13}CO_2$ (‰) in 'high temp' soil

Soil	**Df**	**F**	**p**
Block	1	3.234	0.106
'high temp' soil	1	7.949	**0.020**

CO_2 efflux from HCO_3^- in irrigation water

Soil temperature did not affect the proportion of DIC in the Okanagan Lake water that was released as CO_2 during the measurement period (Figure 3.3; one-way ANOVA: $p = 0.411$). According to one-tailed t-tests, no CO_2 derived from DIC in irrigation water was detected at either of the soil temperature levels.

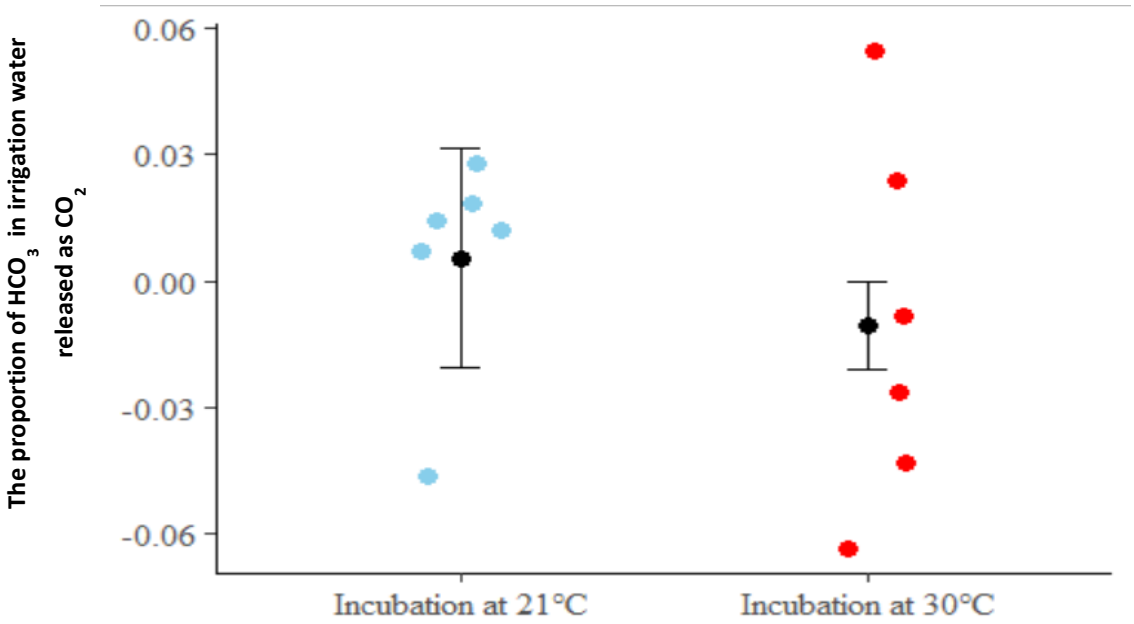

Figure 3. 3 The effect of soil temperature on the proportion of DIC in the Okanagan Lake irrigation water that was released as CO_2-C (one-way ANOVA with soil temperature as the fixed factor: df = 1, F = 0.744, and p = 0.411). Coloured circles represent individual data points for 'low temp' soil (blue circles) or 'high temp' soil (red circles). Black circles represent the treatment means (n = 6) and error bars show the standard deviations.

3.4 Discussion

Contrary to expectations, I found no evidence that a greater proportion of DIC was released from Okanagan Lake water applied to 'high temp' soil than to 'low temp' soil (Figure 3.3). This was supported by the data presented in Figure 3.2, where the change in $\delta^{13}CO_2$ did not differ between the soil temperature levels when irrigated with Okanagan Lake water.

The application of either deionized water or Okanagan Lake water caused an increase in CO_2 efflux rate from both the 'high temp' and 'low temp' soils (One-tailed t-tests; Figure 3.1). This may be due to the relatively dry moisture contents (10%) at which the soil collars were maintained during the pre-incubation period. Such low moisture contents correspond to only ~ 30% water-filled pore space. Given that biogenic CO_2 is relatively depleted in ^{13}C (Smith and Epstein, 1971; Dzurec et al., 1985; Pawellek and Veizer, 1994; Stevenson and Verburg, 2006; Bertrand et al., 2007; Tamir et al., 2011), the increase in CO_2 efflux (One-tailed t-tests; Figure

3.1) and accumulation of ^{13}C depleted CO_2 following application of deionized water (One-tailed t-tests; Figure 3.2) suggests that applications of deionized water stimulated microbial respiration. Addition of water to such dry soils may also have expelled some biogenic CO_2 out of the air-filled soil pore spaces and into the chamber atmosphere. Regardless of whether the CO_2 released in response to the application of deionized water was generated by the active soil microbial population or was released from air-filled soil pores, it appears to be primarily biological in origin.

Application of deionized water to 'high temp' soil induced a greater increase in efflux rate than the application of deionized water to 'low temp' soil (Figure 3.1; Table 3.1). Again, this is not altogether surprising, because higher temperatures tend to increase chemical and biological reaction rates in soil. However, the $\delta^{13}C$ of CO_2 released from 'high temp' soil was less negative than the $\delta^{13}C$ of CO_2 released from 'low temp' soil (Figure 3.2). This suggests that some dissolution of pre-existing PC did occur and that this process was accelerated by incubation at the higher temperature. This finding agrees with Chevallier et al. (2016), who examined the effect of temperature on SIC-derived CO_2 from a calcareous soil of northwestern Tunisia. They found that both the release of CO_2 and the $\delta^{13}C$ of the emitted CO_2 increased with increasing temperature. Finneran and Morse (2009) also found that the rate of calcite dissolution (in KCl and NaCl solutions) increased with increasing temperature. Alternatively, incubation at higher temperature in the present experiment may have promoted acidifying processes, such as nitrification (Tan et al., 2018; Nguyen et al., 2019) which releases H^+ ions. The H^+ ions may then have dissolved pre-existing PC or reacted with HCO_3^- ions dissolved in the soil solution (e.g., due to the dissolution of PC) and produced more ^{13}C-rich CO_2 (Bertrand et al., 2007; Tamir et al., 2011). Although Okanagan Lake water treatment did not cause any significant changes in $\delta^{13}CO_2$ (One-tailed t-tests; Figure 3.2), the CO_2 was more enriched in ^{13}C when compared with deionized water treatment (Figure 3.2; Table 3.4; Table 3.5). The reason for this enrichment is unclear as the CO_2 released from the DIC in the Okanagan Lake water was not significantly different than zero, though it should be noted that the data were quite variable (Figure 3.3).

Regarding the temperature effect, application of Okanagan Lake water to 'high temp' soil resulted in higher CO_2 efflux than the 'low temp' soil (Figure 3.1; Table 3.1). This might be due to the higher rate of biogenic and PC derived CO_2 production in 'high temp' soil compared to

that in 'low temp' soil. However, the ^{13}C enriched CO_2 derived from PC could have counteracted the decline in ^{13}C released from the biogenic CO_2, resulting in no net change in $\delta^{13}CO_2$ (Figure 3.2).

The changes in $\delta^{13}C$ of CO_2 that occurred after applications of deionized water or Okanagan Lake water in this experiment were smaller than those observed for the 10% soil moisture treatment described in Chapter 2. This may be due to slight differences in soil collar construction between the two experiments. In the experiment described in Chapter 2, the bottom of the soil collars was enclosed with a nylon mesh, which could have allowed atmospheric air to enter the chamber atmosphere, particularly in dry soils. By contrast, in the experiment described in this chapter, a plastic bag was also affixed to the bottom of the collar, to prevent the influx of atmospheric air. Diffusion of ^{13}C-rich atmospheric air into the chamber from the bottom of the soil collar during the 15 hour-long pre-incubation period may explain why the changes (decreases) in $\delta^{13}CO_2$ observed after irrigation in the previous experiment were slightly higher than those observed in the present experiment.

The proportion of DIC released as CO_2 from Okanagan Lake water was extremely variable, particularly for 'high temp' soil (Figure 3.3). In this experiment, there was a closed chamber on top of the soil collar so that CO_2 and soil moisture (as water vapour) could not escape to the outside atmosphere. Even so, it was expected that the higher temperature would increase the rate at which the CO_2 gas and the water vapour left the soil surface and accumulated in the atmosphere inside the chamber. Under the dry soil conditions imposed in this experiment (soils were maintained at 10% prior to application of the irrigation treatment), a lack of dissolved CO_2 in the soil solution would promote the release of CO_2 from irrigation water. However, the 30 ℃ temperature might not be high enough to increase the rate of evaporation and CO_2 off-gassing from the soil surface, which might have discouraged the release of CO_2 from irrigation water (Equation 1 to the right). Given the high variability in the data, however, it is difficult to draw any firm conclusions about the effect of soil temperature on the proportion of irrigation water-derived DIC released as CO_2.

In conclusion, the proportion of DIC dissolved in the irrigation water that was released as CO_2 was not affected by soil temperature levels, although the data were extremely variable,

particularly in ‘high temp’ soil (Figure 3.3). This high variability may be explained by the heterogenous nature of soils, uneven distribution of water in soils, etc. In addition, the effect of the two experimental factors (soil temperature and water quality) may have been better understood if a greater range of soil temperatures were tested at a wider range of soil moisture contents.

Chapter 4: The Effect of Irrigation Water Chemistry and Soil Texture on CO_2 Production

4.1 Synopsis

Soil texture is expected to regulate the release of CO_2 derived from DIC in irrigation water largely by affecting water holding capacity (Chadwick et al., 1989; Tahir and Marschner, 2017; Upadhyay and Raghubanshi, 2020). Fine-textured soils have a larger number of pore spaces, but with smaller diameters than coarse-textured soils; the small pore spaces in fine-textured soils hold the water tightly (Tahir and Marschner, 2017). Fine-textured soils also have larger surface areas, which increase the water-holding capacity and also likely the kinetics of reactions compared to coarse-textured soils (Tahir and Marschner, 2017). The high water-holding capacity of a fine-textured soil is expected to enhance microbial respiration and, hence, produce more biological CO_2 than coarse-textured soil (Kowalenko and Ivarson, 1978; Bouma and Bryla, 2000; Dilustro et al., 2005). The resulting build-up of biologically derived CO_2 is likely to discourage the production of PC and CO_2 from the irrigation water containing Ca^{2+} and HCO_3^- ions (Equation 1 to the right).

$$Ca^{2+}_{(aq)} + 2HCO_3^-{}_{(aq)} \longleftrightarrow CaCO_{3\,(s)} + H_2O_{(l)} + CO_{2(g)} \quad (1)$$

As described, in Chapter 2, I found that a greater proportion of DIC in the irrigation water was released as CO_2 from wetter soils than from drier soils. In the experiment described in Chapter 4, my objective was to examine the role of soil texture in determining the proportion of DIC in Okanagan Lake water that is released as CO_2. For this experiment, I used two soils that had similar chemical properties (Table 4.1) but differed in their texture types: sandy loam soil and silt loam soil. Therefore, any difference in the formation of PC and the associated release of CO_2 (Chadwick et al., 1989) was likely through an effect of soil texture. I predicted that the release of CO_2 from irrigation water-derived DIC would be higher in the coarser-textured soil than in the fine-textured soil.

The CO_2 efflux rate (μmol m^{-2} s^{-1}) and the $\delta^{13}CO_2$ (‰) were measured before and after irrigation water of two different chemistries was applied to soils of two different textures (sandy loam and silt loam) that had been maintained at 15% gravimetric moisture content. Although the moisture

contents were the same, silt loam soil was expected to hold more water nearer to the soil surface compared to the sandy loam soil.

4.2 Methodology

Collection, processing, and storage of soil samples

The two soils (sandy loam and silt loam) used in this study were Penticton soils collected from the Agriculture and Agri-food Canada Summerland Research and Developmental Centre. Taxonomically, Penticton soil is classified as an Orthic Brown Chernozem, which tends to have a high water-holding capacity, well to moderately well drained, moderate organic C content, and a pH of 7.1 to 7.9 in the top 0 to 30 cm depth (Wittneben, 1986). Grasses (Poaceae), sagebrush (*Artemisia tridentata*), rabbitbrush (*Ericameria nauseosa*), and scattered ponderosa pine (*Pinus ponderosa*) are some naturally-occurring vegetation on this soil (Wittneben, 1986). However, tree fruits and vineyards are grown in most of the cultivated areas with this soil type in the Okanagan Valley. Some features of the selected Penticton soils are presented in Table 4.1.

In July 2019, soil samples were collected using a shovel (approximate depth, 15 cm) from the rows of an irrigated apple orchard (sandy loam) and from an adjacent unirrigated uncultivated (silt loam) site. Approximately 25 kg of soil was collected from two locations at each site. Each soil type was combined and homogenized by hand; large roots and stones were removed, and the samples were taken to the BRAES Soils Laboratory at the Okanagan campus. In the laboratory, the soils were oven-dried at 65 to 70 ℃ for 3 to 4 days, sieved through a 2-mm sieve, and stored at room temperature (21 ℃) until use.

Table 4. 1 Some features of the selected Penticton soils (Wittneben, 1986; A.J. Midwood, L.A. Phillips, unpublished)

	Sandy loam soil (Row: 0-15 cm)	**Silt loam soil (Alley: 0-15 cm)**
Taxonomic Name	Orthic Brown Chernozem	Orthic Brown Chernozem
Cultivated crops	Apple	Uncultivated
Type of Irrigation	Drip	Unirrigated
Soil pH	7.6	7.3
OM (%)	2.4	2.6
Inorganic C (%)	0.04	0.06
CEC (meq/100 g)	11.8	9.3
Sand (%)	**44**	**21**
Silt (%)	**46**	**70**
Clay (%)	**10**	**8.9**
Exchangeable Ca (%)	72.3	70.7
Exchangeable Mg (%)	20	17
Exchangeable K (%)	7	11.2

Collection, preparation, and storage of water

Two types of water were used for this experiment: Okanagan Lake water or deionized water (control). The Okanagan Lake water was collected from the lake shore on Cadder Avenue, Kelowna, on 21 September 2019. Both the Okanagan Lake water and the deionized water were stored in glass bottles and kept in a walk-in refrigerator (5 ℃) until use (no headspace gas was used).

Preparation of soil collars

On September 13, 2019, two PVC collars were packed with sandy loam soil to a dry bulk density of 1.41 g cm^{-3}, leaving a 3-cm space on top of the soil surface. The other two collars were packed with silt loam soil to a dry bulk density of 1.22 g cm^{-3}, leaving a 2-cm space above the soil surface. Since soil bulk density decreases with increasing porosity of the soil (Chaudhary et al.,

2013), the bulk density of the silt loam soil was slightly lower than the sandy loam soil. The two types of soils were packed to a different height to allow similar soil weights in each collar. Approximately, 850 g of soil was added to each of the four collars. As described in Section 3.2, both a fine nylon fabric (0.33 mm openings) and a piece of plastic sheet were fixed to the bottom of each soil collar.

Analysis of CO_2-C respiration rate and ^{13}C isotopic composition

After packing the collars, the soils were moistened with deionized water to achieve a 15% gravimetric moisture content. Collars packed with soil of the same texture were assigned to one of two irrigation treatments: deionized water or Okanagan Lake water. Thus, there were four experimental treatments in total: i. sandy loam soil and deionized water; ii. sandy loam soil and Okanagan Lake water, iii. silt loam soil and deionized water, iv. silt loam soil and Okanagan Lake water. To determine the CO_2 efflux stabilization period for Penticton soil, the CO_2 efflux of each chamber was measured daily for 8 days using the EGM-4 Environmental CO_2 Gas Monitor (model 4.18, Amesbury, MA, USA). After 7 days, the CO_2 efflux had stabilized. Soil collars were re-weighed every day over this stabilization period, and deionized water was added, as required, to maintain 15% gravimetric moisture content.

The soil CO_2 efflux and $\delta^{13}CO_2$ composition were measured following the pre- and post-irrigation periods, as described in Section 2.2.2. The experiment comprised six runs, each taking approximately 8 days to complete. One replicate of each of the four experimental treatments was included in each experimental run. The collars were re-packed with fresh soil for each run. The experiment was started on September 24, 2019, and completed on November 13, 2019.

Data analyses

Change in CO_2 efflux and $\delta^{13}CO_2$ after irrigation

The net change in CO_2 efflux rate and its $\delta^{13}CO_2$ composition were calculated by subtracting the pre-incubation measurements from the post-incubation measurements, as described in Sections 2.2.2 and 3.2. Assumptions of normality and homogeneity of variance were tested using the

Shapiro-Wilk (Q-Q plots) and Levene's tests, respectively. When the assumptions were satisfied, two-way ANOVAs were used to determine the effects of soil texture, water quality, and their interactions on the net change in CO_2 efflux rate and its $\delta^{13}CO_2$ composition. 'Run' was included as a random factor (i.e., analogous to a 'block' effect). Differences between treatment means were considered statistically significant at $p < 0.05$. Individual one-tailed t-tests were performed for both of the soil textures irrigated with deionized or Okanagan Lake water to test whether the change in CO_2 efflux rate and its $\delta^{13}CO_2$ composition after irrigation was greater than zero. All of the analyses were performed using R 3.6.3 (Ihaka and Gentleman, 1992).

CO_2 efflux from HCO_3^- in irrigation water

The proportion of DIC in the irrigation water from Okanagan Lake that was subsequently detected in the CO_2 efflux was determined using a simple isotopic mass balance model, as described in Section 2.2.2. A one-way ANOVA was performed to test for an effect of soil texture on the proportion of Okanagan Lake DIC released as CO_2. 'Run' was included as a random factor. Individual one-tailed t-tests were performed for both of the soil textures irrigated with Okanagan Lake water to test whether the proportion of DIC in the Okanagan Lake water that was detected in CO_2 efflux was greater than zero. All analyses were performed using R 3.6.3 (Ihaka and Gentleman, 1992).

4.3 Results

Change in CO_2 efflux and $\delta^{13}CO_2$ after irrigation

In the sandy loam soil, CO_2 efflux rate increased after irrigation with Okanagan Lake water (one-tailed t-test: $p = 0.007$) but did not change after irrigation with deionized water (one-tailed t-test: $p = 0.144$). CO_2 efflux rate in the silt loam soil increased with both types of irrigation water (one-tailed t-test: $p = 0.001$ when irrigated with Okanagan Lake water; $p = 0.034$ when irrigated with deionized water). Regardless of the soil texture, the CO_2 efflux rate increased more when soils were irrigated with Okanagan Lake water than when they were irrigated with deionized water (Figure 4.1; Table 4.2).

Before irrigation, the $\delta^{13}CO_2$ value was -21.7 ± 1.3‰ for sandy loam soil and -22.7 ± 1.2‰ for the silt loam soil (Appendix 4). While only a borderline effect of irrigation water quality (p = 0.055) on the $\delta^{13}C$ of CO_2 efflux was detected in the two-way ANOVA (Figure 4.2; Table 4.3), Okanagan Lake water caused an increase in $\delta^{13}C$ of CO_2 efflux released from either soil type (one-tailed t-tests: p = 0.040 for sandy loam soil; p = 0.013 for silt loam soil), whereas deionized water did not (one-tailed t-tests: p = 0.270 for sandy loam soil; p = 0.388 for silt loam soil). When Okanagan Lake water was applied, the average increase in $\delta^{13}CO_2$ was 3.7‰ for the silt loam soil and 2‰ for the sandy loam soil, but the effect of soil texture was not statistically significant (Figure 4.2 and Table 4.3).

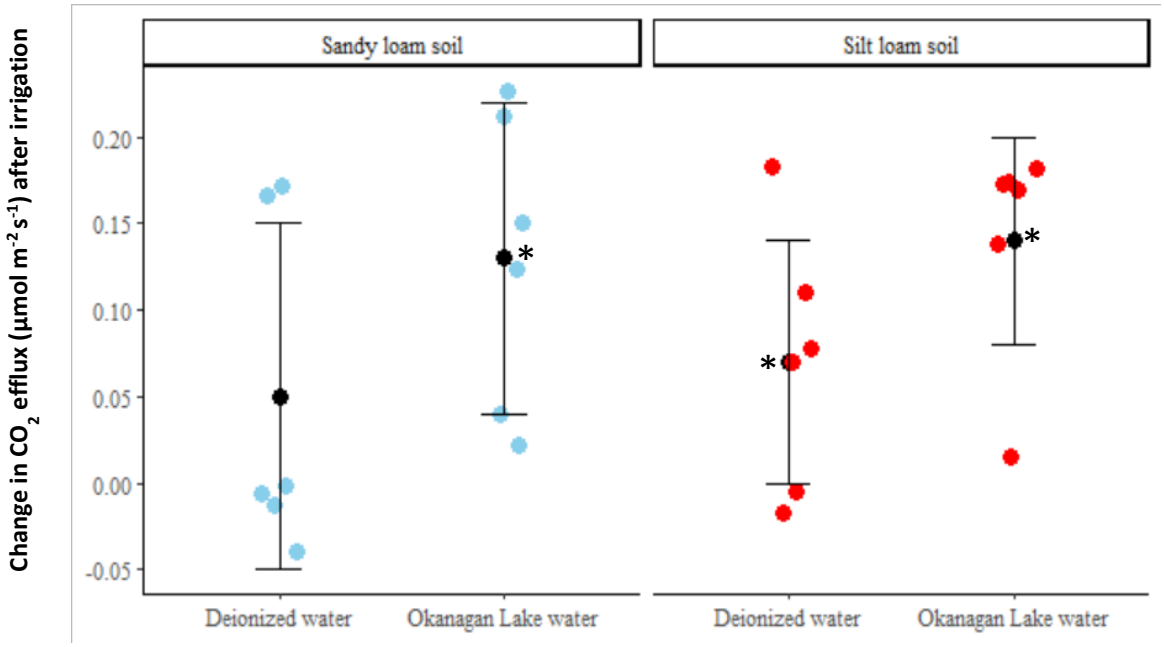

Figure 4. 1 The effect of soil texture on the change in CO_2 efflux (µmol m^{-2} s^{-1}) after application of Okanagan Lake water or deionized water. Coloured circles represent individual data points for sandy loam soil (blue circles) or silt loam soil (red circles). Black circles represent the treatment means (n = 6) and error bars show the standard deviations. Asterisks indicate that the treatment means of the change in CO_2 efflux were significantly greater than zero, according to one-tailed t-tests.

Table 4. 2 Results of two-way ANOVA of the effects of soil texture and irrigation water quality on the change in CO_2 efflux (µmol m^{-2} s^{-1})

Factor	Df	F	p
Block	1	3.446	0.079
Soil texture	1	0.344	0.565
Water quality	1	6.148	**0.023**
Soil texture x Water quality	1	0.028	0.870

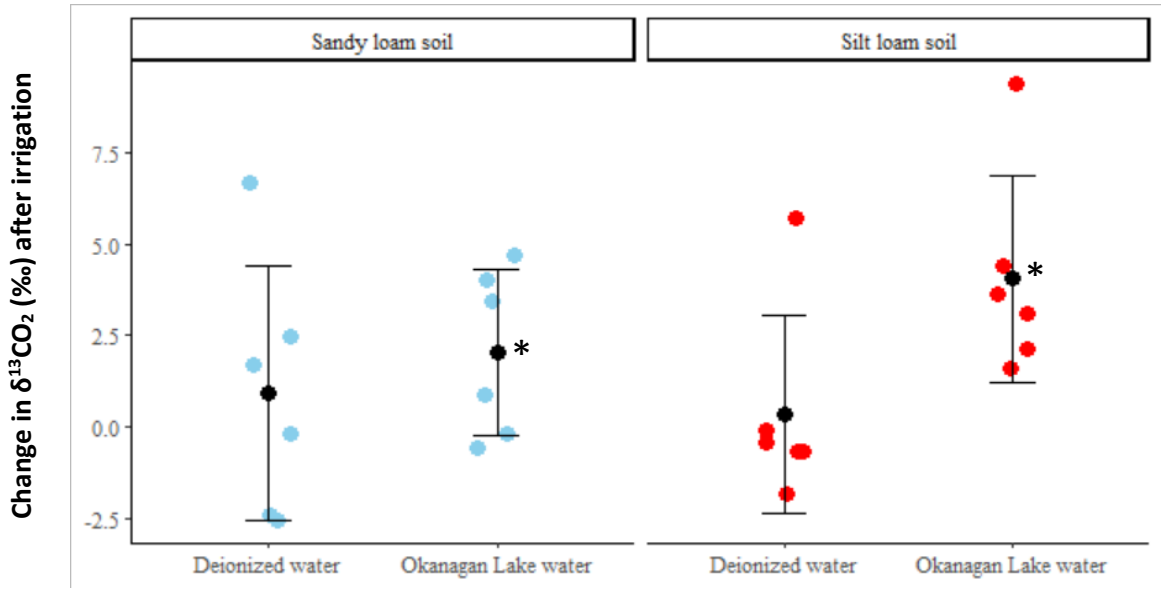

Figure 4. 2 The effect of soil texture on the change in $\delta^{13}CO_2$ (‰) after application of Okanagan Lake water or deionized water. Coloured circles represent individual data points for sandy loam soil (blue circles) or silt loam soil (red circles). Black circles represent the treatment means (n = 6) and error bars show the standard deviations. Asterisks indicate that the treatment means of the change in $\delta^{13}CO_2$ (‰) were significantly greater than zero, according to one-tailed t-tests.

Table 4. 3 Results of two-way ANOVA of the effects of soil texture and irrigation water quality on the change in $\delta^{13}C$ (‰) of CO_2 efflux

Factor	Df	F	p
Block	1	3.716	0.069
Soil texture	1	0.237	0.632
Water quality	1	4.167	0.055
Soil texture x Water quality	1	1.072	0.313

CO_2 efflux from HCO_3^- in irrigation water

A greater proportion of Okanagan Lake water-derived DIC was released as CO_2 (one-way ANOVA; p = 0.033) when water was applied to the silt loam soil than to the sandy loam soil (Figure 4.3). According to one-tailed t-tests, CO_2 derived from DIC in the irrigation water was detected in the efflux from both soil types (p = 0.047 for sandy loam soil; p <0.001 for silt loam soil). Overall, an average of 4.9 % of the DIC in the Okanagan Lake water was released as CO_2.

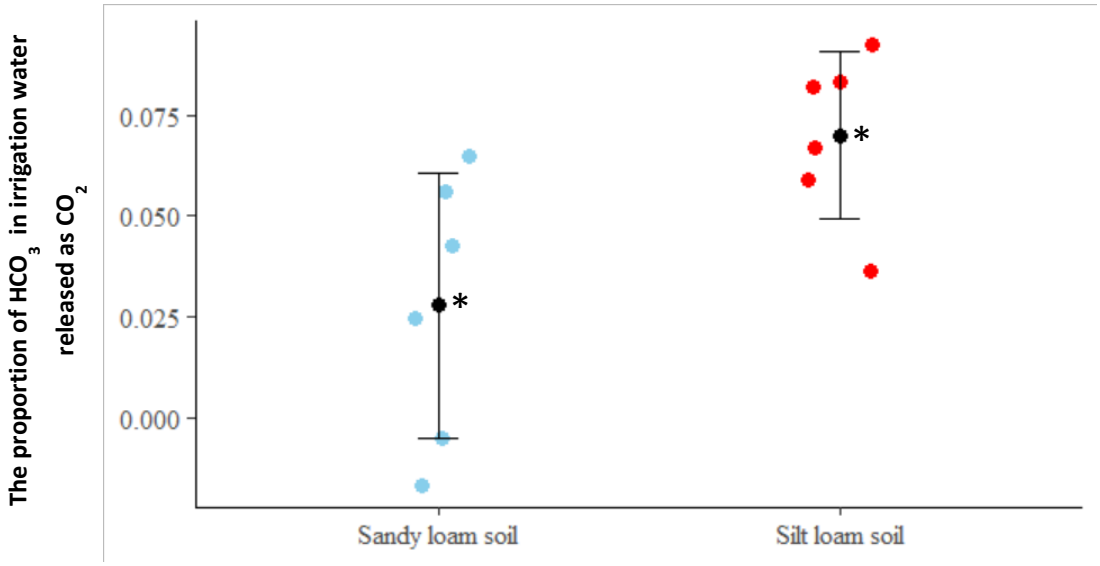

Figure 4. 3 The effect of soil texture on the proportion of DIC in the Okanagan Lake irrigation water that was released as CO_2-C (one-way ANOVA with soil texture as the fixed factor: df = 1, F = 6.294, p = 0.033). Coloured circles represent individual data points for Sandy loam soil (blue circles) and Silt loam soil (red circles). Black circles represent the treatment means (n = 6) and error bars show the standard deviations. Asterisks indicate that the treatment means of the proportion of HCO_3^- in irrigation water released as CO_2 were significantly greater than zero, according to one-tailed t-tests.

4.4 Discussion

Contrary to my expectation, a higher proportion of Okanagan Lake water-derived DIC was released after irrigation of finer-textured (silt loam) soil than coarse-textured (sandy loam) soil (Figure 4.3). I found no evidence that CO_2 efflux or $\delta^{13}CO_2$ increased more in the silt loam soil than the sandy loam soil following irrigation with Okanagan Lake water. However, the higher proportion

of DIC released from the silt loam soil compared with the sandy loam soil following irrigation with Okanagan Lake water is apparent from the mathematically higher average increase in $\delta^{13}CO_2$ observed in the silt loam soil (3.7‰) than in the sandy loam soil (2‰) (Figure 4.2).

The application of deionized water to sandy loam soil did not affect the CO_2 efflux rate (one-tailed t-test; Figure 4.1). This lack of treatment effect may be the result of the relatively high gravimetric moisture contents (15%) at which the soil collars were maintained during the pre-incubation period. In the sandy loam soil, 15% moisture content corresponds to ~ 37% water-filled pore space; consequently, microbes in these soils may not have been water-stressed. By contrast, the sandy loam/loamy sand used in Experiment 2 (Chapter 3) was maintained at 10% gravimetric soil moisture content and, consequently, had fewer water-filled pore space (~ 30%); as a result, applications of irrigation water probably caused a greater stimulation of microbial activity and expulsion of biogenic CO_2 from soil pores compared with the sandy loam soil used in this experiment.

The application of deionized water to the silt loam soil in this experiment did cause an increase in CO_2 efflux rate (one-tailed t-test; Figure 4.1). According to da Costa et al. (2013), fine-textured soil often contains a higher amount of organic matter (OM) than coarse-textured soil. Indeed, the OM content of the silt loam soil used in this experiment was slightly higher than the sandy loam soil (Table 4.1). Therefore, soil microbes may have been more active in the silt loam soil than in the sandy loam soil (Gasch and DeJong-Hughes, 2019). Moreover, the 15% moisture content of the silt loam soil corresponds to only ~ 29% water-filled pore space. Thus, the addition of irrigation water may have caused a greater stimulation of microbial respiration and expulsion of biogenic CO_2 from air-filled soil pore spaces compared with the sandy loam soil used in this experiment. Given that biogenic CO_2 is depleted in ^{13}C compared to the CO_2 derived from inorganic sources (Smith and Epstein, 1971; Dzurec et al., 1985; Pawellek and Veizer, 1994; Stevenson and Verburg, 2006; Bertrand et al., 2007; Tamir et al., 2011), the increase in CO_2 efflux rate (one-tailed t-test; Figure 4.1) but lack of change in $\delta^{13}CO_2$ following application of deionized water (one-tailed t-test; Figure 4.2) suggests that some ^{13}C-enriched CO_2 may have been released from the dissolution of pre-existing PC, which could have counteracted the effect of ^{13}C depleted biogenic CO_2 on the overall change in $\delta^{13}CO_2$ after irrigation.

Application of Okanagan Lake water to either sandy loam soil or silt loam soil resulted in a higher CO_2 efflux rate than that observed after application of deionized water (Figure 4.1; Table 4.2). The ^{13}C enrichment of CO_2 released after irrigation of either soil type with Okanagan Lake water (one-tailed t-tests; Figure 4.2) indicates that the additional CO_2 released from these soils was derived from the DIC applied in the water.

Soil texture did not significantly affect the change in CO_2 efflux rate following irrigation (Figure 4.1, Table 4.2). This was unexpected. Bouma and Bryla (2000) examined the effects of soil texture on CO_2 concentration (biogenic) in soil and CO_2 efflux from the soil surface. Although they observed that following irrigation, CO_2 was more concentrated in finer textured soil than in sandy soil, they also found that CO_2 efflux from the soil surface was more restricted in finer textured soil than in sandy soil. This was due to the higher water holding capacity of the finer-textured soil, which decreased the diffusivity of CO_2 through the soil after irrigation water was applied (Bouma and Bryla, 2000). This might help explain why CO_2 efflux from the silt loam soil in the present experiment was lower than expected after any type of irrigation water was applied.

Although the change in $\delta^{13}CO_2$ did not significantly differ between the two soil textures (Table 4.3), the average increase in $\delta^{13}CO_2$ was mathematically higher following application of Okanagan Lake water to the silt loam soil than to the sandy loam soil (Figure 4.2). Both experimented soils had similar chemical properties (including CEC and exchangeable Ca) and were maintained at 15% gravimetric moisture content, the hypothesis was formed based on the higher water holding capacity of the finer-textured soil near the soil surface. However, water-filled porosity after irrigation was higher in sandy loam soil (~ 51%) than the silt loam soil (~ 39%). Considering the water-filled porosity, microbial respiration was probably lower in silt loam soil than the sandy loam soil; Doran et al. (1990) demonstrated that the microbial respiration of a coarser-textured soil with ~ 50% water-filled porosity is higher than a medium-textured soil with ~ 40% water-filled porosity. Lower microbial respiration in silt loam soil may have resulted in less biological CO_2 accumulation than in the sandy loam soil solution, in combination with slower rates of infiltration, might have accelerated the release of CO_2 from the DIC in irrigation water (Equation 1 to the right) when compared with the sandy loam soil.

In conclusion, although the rate of CO_2 efflux was not affected by soil texture, the proportion of DIC released from the Okanagan Lake irrigation water was affected by soil texture: more DIC was released as CO_2 from silt loam soil than from sandy loam soil. That being said, the data were extremely variable (Figure 4.3), which may be due to the heterogenous nature of soils, uneven distribution of water in soils, etc. The effects of the two experimental factors (soil texture and water quality) may have been better understood if a greater range of soil textures were tested at a wider range of soil moisture contents.

Chapter 5: The Effect of Irrigation Water Chemistry and Soil Organic Matter (SOM) on CO_2 Production

5.1 Synopsis

Soil organic matter (SOM) is expected to regulate the release of CO_2 derived from DIC in irrigation water because SOM activates soil microbes by providing an energy source, nutrients, and habitat (Gasch and DeJong-Hughes, 2019). Therefore, it can be presumed that soil with a higher amount of organic matter (OM) would produce more biological CO_2 than soil with a lower amount of OM (Ferrea et al., 2012; Thangarajan et al., 2013; Chen et al., 2021). The build-up of biologically-derived CO_2 is likely to discourage equation 1 to move to the right. Therefore, soil with a lower amount of OM is expected to produce more irrigation-derived PC and CO_2 than soil with a higher amount of OM.

$$Ca^{2+}_{(aq)} + 2HCO_3^-{}_{(aq)} \longleftrightarrow CaCO_{3\,(s)} + H_2O_{(l)} + CO_{2(g)} \quad (1)$$

As described in Chapter 2, I found that a greater proportion of DIC in the irrigation water was released as CO_2 from wetter soils than from drier soils, while in Chapter 4, I found that a greater proportion of DIC was released from silt loam soil than from sandy loam soil. I concluded that the lower microbial respiration associated with the water-filled porosities of soils causes a greater proportion of CO_2 to be released from irrigation water-derived DIC. In the experiment described here in Chapter 5, my objective was to examine the role of SOM in determining the proportion of DIC in Okanagan Lake water that is released as CO_2. For this experiment, I used two soils that had similar physical and chemical properties (Table 5.1) but differed in their OM contents: the 'high OM' soil had an OM content of 7.7% and the 'low OM' soil had an OM content of 3.3%. Therefore, any difference in the formation of pedogenic carbonates (PC) and associated release of CO_2 (Chadwick et al., 1989) was likely through an effect of the amount of OM content in the two soil types. I predicted that the release of CO_2 from irrigation water-derived DIC would be higher in the 'low OM' soil than in the 'high OM' soil. The CO_2 efflux rate ($\mu mol\ m^{-2}\ s^{-1}$) and $\delta^{13}CO_2$ (‰) were measured before and after irrigation water of two different chemistries was applied to 'high OM' and 'low OM' soils that were maintained at 15% gravimetric moisture content.

5.2 Methodology

Collection, processing, and storage of soil samples

The two soils used in this study were chosen because they had different organic matter (OM) contents but their other soil chemical and physical properties were very similar. The 'high OM' soil was characterized as a Rutland soil (7.7% OM), collected from a cherry orchard inWest Kelowna, Okanagan Valley, Canada. Taxonomically, this soil is classified as an Orthic Dark Brown Chernozem, which tends to have a low water-holding capacity, rapid drainage capacity, moderate organic C content, and a pH of 6.6 to 7.4 within the top 0 to 20 cm depth (Wittneben, 1986). Grasses (Poaceae), sagebrush (*Artemisia tridentata*), rabbitbrush (*Ericameria nauseosa*), and scattered ponderosa pine (*Pinus ponderosa*) are the native vegetation on this soil (Wittneben, 1986). Among the cultivated and irrigated crops with this soil type, tree fruits and grapes are the most common.

The 'low OM' soil was characterized as an Osoyoos soil (3.3% OM), collected from an apple orchard in Lake Country, Okanagan Valley, Canada. The soil is taxonomically classified as an Orthic Brown Chernozem, which tends to have a low to very low water-holding capacity, rapid drainage capacity, low to very low organic C content, and a pH of 6.3 to 7.8 within the top 0 to 50 cm depth (Wittneben, 1986). Most of this soil is used for agriculture (Government of Canada, 2019) and therefore, the native vegetation is not known. Some features of the selected Rutland and Osoyoos soils are presented in Table 5.1.

On September 25, 2019, soil samples were collected using a shovel (approximate depth, 15 cm) from the alleys of each orchard. Approximately 25 kg of soil was collected from three locations at each site. The soils collected from each site were combined and homogenized by hand; large roots and stones were removed, and the samples were taken to the BRAES Soils Laboratory at the Okanagan campus. In the laboratory, the soils were oven-dried at 65 to 70 ℃ for 3 to 4 days, sieved through a 2 mm sieve, and stored at room temperature (21 ℃) until use.

Table 5. 1 Some features of the selected Rutland and Osoyoos soils (Wittneben, 1986; A.J. Midwood, L.A. Phillips, unpublished)

	Rutland soil (Alley: 0-15 cm)	**Osoyoos soil (Alley: 0-15 cm)**
Taxonomic Name	Orthic Dark Brown Chernozem	Orthic Brown Chernozem
Cultivated crops	Cherry	Apple
Type of Irrigation	Micro-spray	Micro-spray
Soil Texture	Sandy loam, Loamy sand	Sandy loam, Loamy sand
Soil pH	6.6	6.5
OM (%)	***7. 7***	***3.3***
CEC (meq/100 g)	15.1	15.8
Exchangeable Ca (%)	63.8	62.5
Exchangeable Mg (%)	15	19.7
Exchangeable K (%)	8	6.8

Collection, preparation, and storage of water

Two types of water were used for this experiment: Okanagan Lake water and deionized water (control). The Okanagan Lake water was collected from the lake shore in Cadder Avenue, Kelowna on 6 December 2019. Both the Okanagan Lake water and the deionized water were stored in glass bottles and kept in a walk-in refrigerator (5 ℃) until use (no headspace gas was used).

Preparation of soil collars

On November 29, 2019, two PVC collars were packed with 'high OM' soil (~ 715 g) to a dry bulk density of 1.18 g cm^{-3}, leaving a 3-cm space on top of the soil surface. The other two collars were packed with 'low OM' soil (~ 830 g) to a dry bulk density of 1.37 g cm^{-3}, leaving the same 3-cm space above the soil surface. Since soil bulk density decreases with increasing OM content (Pri and Quimet, 2008), the bulk density of the 'high OM' soil was slightly lower than the 'low OM' soil. The weight of the soils added to the collars was different for the two soil types to ensure that a similar 3-cm space remained between the top of the soil surface and the rim of the

collar. As described in Sections 3.2 and 4.2, both a fine nylon fabric (0.33 mm openings) and a piece of plastic sheet were fixed to the bottom of each soil collar.

Analysis of CO_2-C respiration rate and ^{13}C isotopic composition

After packing the collars, the soils were moistened with deionized water to achieve a 15% gravimetric moisture content. Each collar with the same soil type (i.e., the same amount of OM) was assigned to one of two irrigation treatments: deionized water or Okanagan Lake water. Thus, there were four experimental treatments in total: i. 'high OM' soil and deionized water; ii. 'high OM' soil and Okanagan Lake water, iii. 'low OM' soil and deionized water, iv. 'low OM' soil and Okanagan Lake water. To determine the CO_2 efflux stabilization period for the 'high OM' and 'low OM' soils, the CO_2 efflux of each chamber was measured daily for 8 days using the EGM-4 Environmental CO_2 Gas Monitor (model 4.18, Amesbury, MA, USA). After 7 days, the CO_2 efflux of both soil types had stabilized. Soil collars were re-weighed every day over this stabilization period, and deionized water was added, as required, to maintain 15% gravimetric moisture content.

The soil CO_2 efflux and $\delta^{13}C$ composition were measured following the pre- and post-irrigation periods, as described in Section 2.2.2. Because the soils used in this experiment were higher in OM content compared to the soils used in the previous experiments, two runs with all four experimental treatments were carried out to examine if the duration of the eight-hour post-incubation period (used in the four previous experiments) could be reduced. During these runs, a 10 ml gas sample was collected from each chamber after 3 hours of post-incubation. The collected gas sample was injected into the environmental CO_2 gas monitor (EGM-4) to determine the CO_2 concentration. The average gas concentration (3331 ppm for 'high OM' soil; 2462 ppm for 'low OM' soil) was found to be more than the minimum (400 ppm) required for analysis by the Picarro G2131-*i* wavelength-scanned cavity ring-down spectrometer (Santa Clara, CA, USA). Considering that, a new set of all four experimental treatments was run using a 3-hour post-incubation period instead of the 8-hour post-incubation period used in the previous experiments. The new experiment with 3 hours of post-incubation comprised six runs, each taking approximately 9 days to complete. One replicate of each of the four experimental treatments was included in each experimental run. The collars were re-packed with fresh soil for

each run, and allowed to stablise for 7 days. The experiment was started on December 19, 2019, and completed on February 04, 2020.

Data analyses

Change in CO_2 efflux and $\delta^{13}CO_2$ after irrigation

The net change in CO_2 efflux rate and its $\delta^{13}CO_2$ composition were calculated by subtracting the pre-incubation measurements from the post-incubation measurements, as described in Section 2.2.2. Assumptions of normality and homogeneity of variance were tested using the Shapiro-Wilk (Q-Q plots) and Levene's tests, respectively. When the assumptions were satisfied, two-way ANOVA was used to determine the effects of soil organic matter (SOM), water quality, and their interactions on the net change in CO_2 efflux rate. 'Run' was included as a random factor (i.e., analogous to a 'block' effect). Differences between treatment means were considered statistically significant at $p<0.05$.

With respect to the $\delta^{13}CO_2$ data, Bartlett's test, in addition to Levene's test, was used to confirm the homogeneity of variance of the data. Although the $\delta^{13}CO_2$ data were normally distributed, the assumption of homogeneity of variance was violated, even after log transformation. Consequently, a regression analysis, equivalent to one-way Welch's ANOVA (assuming unequal variance) was performed to determine the differences among the effects of the four SOM x water quality combinations on the net change in $\delta^{13}CO_2$ composition. Again, 'run' was considered as a random factor. Differences between treatment means were considered statistically significant at $p<0.05$.

Individual one-tailed t-tests were performed separately for both of the 'high OM' soil and 'low OM' soil irrigated with deionized or Okanagan Lake water to test whether the change in CO_2 efflux rate and its $\delta^{13}CO_2$ composition after irrigation was greater than or less than zero.

CO_2 efflux from HCO_3^- in irrigation water

The proportion of DIC in the irrigation water from Okanagan Lake that was subsequently detected in the CO_2 efflux was determined using a simple isotopic mass balance model, as described in Section 2.2.2. A one-way ANOVA was performed to determine the effect of soil organic matter (SOM) on the proportion of Okanagan Lake DIC released as CO_2. 'Run' was included as a random factor. Individual one-tailed t-tests were performed for both the 'high OM' soil and the 'low OM' soil irrigated with Okanagan Lake water to test whether the proportion of DIC in the Okanagan Lake water that was detected in CO_2 efflux was greater than zero. All analyses were performed using R 3.6.3 (Ihaka and Gentleman, 1992).

5.3 Results

Change in CO_2 efflux and $\delta^{13}CO_2$ after irrigation

CO_2 efflux rates increased after irrigation in both soils and with both types of irrigation water (one-tailed t-test: $p < 0.001$ for the 'high OM' soil with either water source; 'low OM' soil, $p = 0.007$ for deionized water, $p = 0.003$ for Okanagan Lake water). Regardless of the organic matter content, CO_2 efflux rate increased more when soils were irrigated with Okanagan Lake water than when they were irrigated with deionized water (Figure 5.1; Table 5.2).

Irrigation water quality, but not soil OM, also affected the $\delta^{13}C$ of CO_2 efflux (Figure 5.2; Table 5.3). Before irrigation, the $\delta^{13}CO_2$ value was -23.1 ± 1.0‰ for 'high OM' soil and -23 ± 1.0‰ for the 'low OM' soil (Appendix 5). Okanagan Lake water caused an increase in the $\delta^{13}C$ of CO_2 efflux released from either soil type (one-tailed t-tests: $p = <0.001$ for 'high OM' soil; $p = 0.020$ for 'low OM' soil). Deionized water caused a decrease in the $\delta^{13}C$ of CO_2 efflux released from 'high OM' soil but did not cause any change in the $\delta^{13}C$ of CO_2 efflux released from the 'low OM' soil (one-tailed t-tests: $p = <0.001$ for 'high OM' soil; $p = 0.060$ for 'low OM' soil).

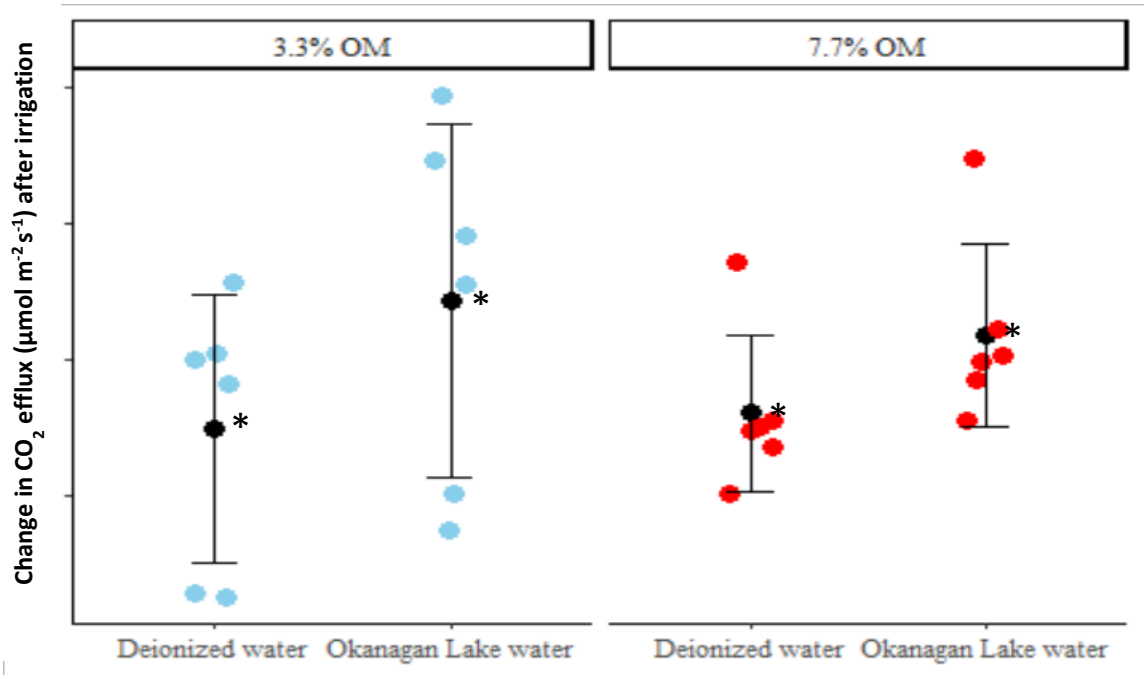

Figure 5. 1 The effect of soil organic matter (SOM) on the change in CO_2 efflux (µmol m^{-2} s^{-1}) after application of Okanagan Lake water or deionized water. Coloured circles represent individual data points for 'low OM' soil (blue circles) or 'high OM' soil (red circles). Black circles represent the treatment means (n = 6) and error bars show the standard deviations. Asterisks indicate that the treatment means of the change in CO_2 efflux were significantly greater than zero, according to one-tailed t-tests.

Table 5. 2 Results of two-way ANOVA of the effects of soil organic matter (SOM) and irrigation water quality on the change in CO_2 efflux (µmol m^{-2} s^{-1})

Factor	**df**	**F**	**p**
Block	1	35.550	**<0.001**
SOM	1	0.093	0.764
Water quality	1	10.969	**0.004**
SOM x Water quality	1	0.662	0.426

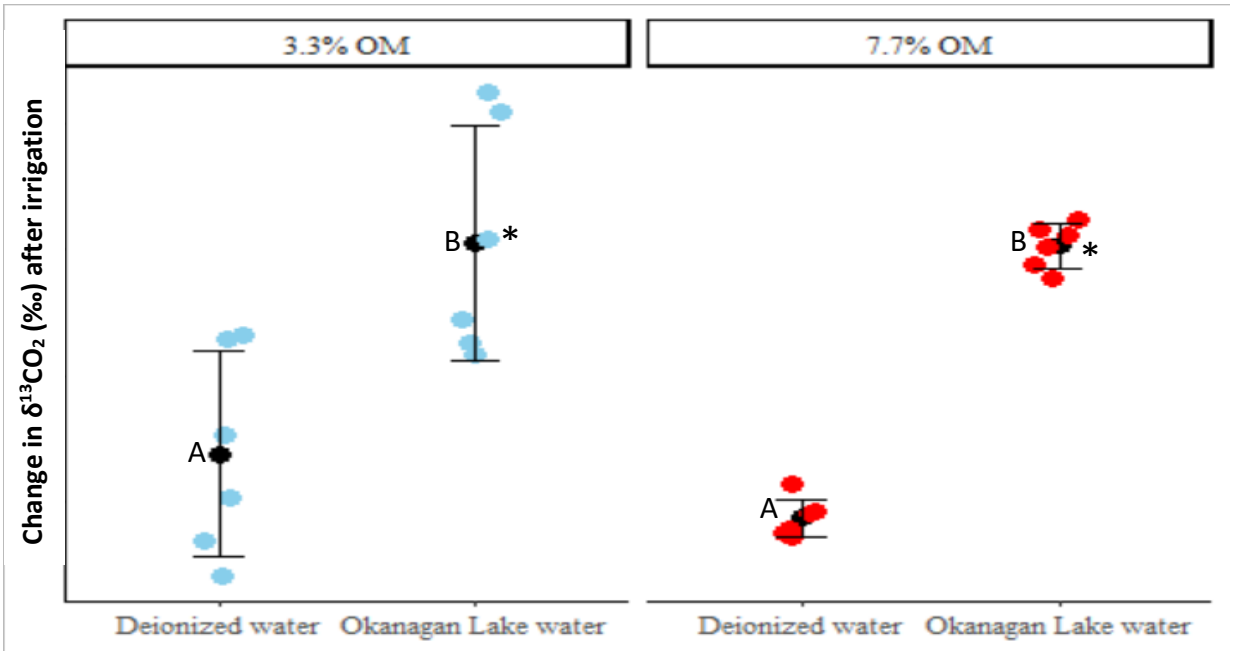

Figure 5. 2 The effect of soil organic matter (SOM) on the change in $\delta^{13}CO_2$ (‰) after application of Okanagan Lake water or deionized water. Coloured circles represent individual data points for 'low OM' soil (blue circles) or 'high OM' soil (red circles). Black circles represent the treatment means (n = 6) and error bars show the standard deviations. Different letters indicate significant differences ($p<0.05$) between the treatment means. Asterisks indicate that the treatment means of the change in $\delta^{13}CO_2$ (‰) were significantly greater than or less than zero, according to one-tailed t-tests.

Table 5. 3 Results of regression analysis (equivalent to one-way Welch ANOVA) of the effects of soil organic matter (SOM) x irrigation water quality combinations on the change in the $\delta^{13}C$ (‰) of CO_2 efflux over the incubation period.

Factor	**Estimate**		**Std. Error**	**t**	**p**
Block	-0.347		0.132	-2.624	**0.017**
low OM, Deionized → high OM, Deionized	-1.015		0.638	-1.591	0.128
low OM, Deionized → low OM, Okanagan Lake	3.347		0.638	5.245	**<0.001**
high OM, Deionized → high OM, Okanagan Lake	4.322		0.638	6.773	**<0.001**
low OM, Okanagan Lake → high OM, Okanagan Lake	-0.04		0.638	-0.063	0.951

OM = Organic matter

CO_2 efflux from HCO_3^- in irrigation water

The organic matter content of the soil did not affect the proportion of DIC in the Okanagan Lake water that was released as CO_2 (Figure 5.3; one-way ANOVA: $p = 0.613$). According to one-tailed t-tests, CO_2 derived from DIC in the irrigation water was detected in the efflux from both soils ($p < 0.001$ in 'high OM' soil; $p = 0.009$ in 'low OM' soil). An average of 7.3 % of the DIC dissolved in the Okanagan Lake water was released as CO_2 over the incubation period.

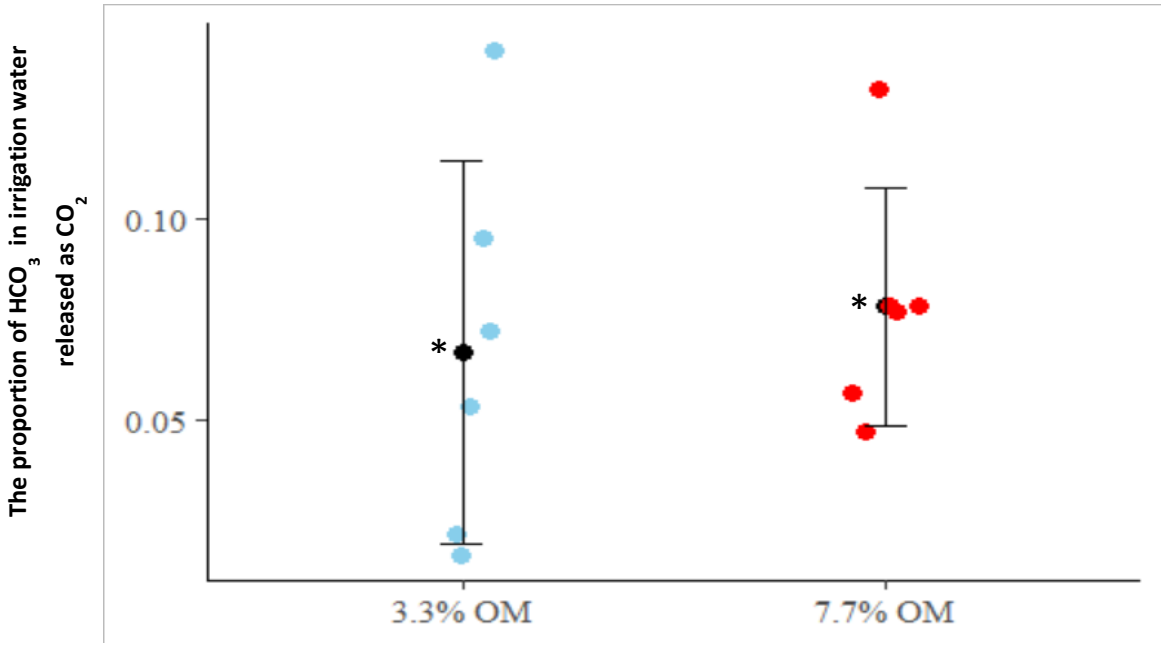

Figure 5. 3 The effect of soil organic matter (SOM) on the proportion of DIC in the Okanagan Lake irrigation water that was released as CO_2-C (one-way ANOVA with soil organic matter as the fixed factor: df = 1, F = 0.274, and p = 0.613). Coloured circles represent individual data points for 'low OM' soil (blue circles) or 'high OM' soil (red circles). Black circles represent the treatment means (n = 6) and error bars show the standard deviations. Asterisks indicate that the treatment means of the proportion of HCO_3^- in irrigation water released as CO_2 were significantly greater than zero, according to one-tailed t-tests.

5.4 Discussion

Contrary to expectations, I found no evidence that a greater proportion of DIC was released from Okanagan Lake water applied to 'low OM' soil than to 'high OM' soil (Figure 5.3). This was supported by the data presented in Figures 5.1 and 5.2, where neither the change in CO_2 efflux nor the $\delta^{13}CO_2$ differed between the soil types.

Unlike Experiment 3, where irrigation with deionized water did not change the CO_2 efflux rate in the sandy loam soil (Chapter 4), the CO_2 efflux rate did increase following irrigation with deionized water in the sandy loam soils in this experiment (one-tailed t-tests; Figure 5.1). Because in both Experiments 3 and 4, a 15% gravimetric moisture content was maintained during the pre-incubation period, the differing responses might be due to the higher OM content of the soils used in this experiment (soils with 7.7% and 3.3% OM) compared to the 2.4% OM content of the sandy loam soil used in Experiment 3. Moreover, the water-filled porosities of the soils (~27% for 'high OM' soil; ~35.5% for 'low OM' soil) in this experiment were slightly lower than the sandy loam soil (~ 37%) used in Experiment 3. Therefore, the addition of deionized water to the soils used in this experiment may have triggered an increase in CO_2 efflux that was not observed in Experiment 3 because of a greater stimulation of microbial respiration and the expulsion of more biogenic CO_2 from air-filled soil pore spaces.

Application of Okanagan Lake water to either 'high OM' soil or 'low OM' soil in this experiment resulted in a higher CO_2 efflux rate than those observed after application of deionized water (Figure 5.1; Table 5.2). Moreover, the CO_2 released from both soil types was enriched in ^{13}C following the application of Okanagan Lake water (one-tailed t-tests; Figure 5.2). This suggests that the additional CO_2 released from the soil surface after the application of Okanagan Lake water was derived primarily from dissolved DIC applied in the water. By contrast, the increase in CO_2 efflux rate (one-tailed t-test; Figure 5.1) and accumulation of ^{13}C-depleted CO_2 following application of deionized water (one-tailed t-test; Figure 5.2) to 'high OM' soil suggests that the application of deionized water stimulated the release of biogenic CO_2. The increase in CO_2 efflux rate (one-tailed t-test; Figure 5.1) but lack of change in $\delta^{13}CO_2$ following application of deionized water (one-tailed t-test; Figure 5.2) to 'low OM' soil suggests that some ^{13}C-enriched CO_2 may have been released from the dissolution of pre-existing PC, which could have

counteracted the effect of ^{13}C depleted biogenic CO_2 on the overall change in $\delta^{13}CO_2$ after irrigation.

Given that soil organic matter (SOM) content affected neither the CO_2 efflux rate (Table 5.2) nor its $\delta^{13}CO_2$ composition (Table 5.3), the proportion of DIC released as CO_2 from Okanagan Lake water was also not affected by SOM content (Figure 5.3). The alleys of the cherry orchard, from which the 'high OM' soil was collected, were covered with grass. Inputs of C from dense grass roots near the soil surface probably explain why this soil was very high in OM content (Weissert et al., 2016; Panettieri et al., 2017). Our observation of a similar response in CO_2 efflux rate following irrigation of soils with strongly contrasting SOM contents corroborates the findings of Weissert et al. (2016), who also observed similar efflux rates in SOC-rich grass-dominated loamy soils (0-10 cm) in New Zealand.

The lack of effect of SOM on the CO_2 efflux rate in the present experiment might be caused by the stabilization of some organic C in the grass-covered 'high OM' soil. This argument is supported by a laboratory experiment conducted by Panettieri et al. (2017) on loamy-textured soils in western France. Some soil C pools (e.g., lignin) were preserved more efficiently and showed a longer residence time in the soils under grassland than in soil under maize, suggesting that plant litter quality and SOC stabilization may play a role in determining the susceptibility of some SOC pools to decay.

Regarding the present experiment, earthworms could also play a role in SOC stabilization. A large number of earthworms was found in the 'high OM' soil collected from the cherry orchard site. Earthworms actively secrete calcium carbonates, which increase soil organic C stabilization (Zhang et al., 2013) by coating individual particles of SOM (Mendoza-Vega and Messing, 2005). As a consequence, the organic C in the 'high OM' soil may not have been as susceptible to mineralization even after irrigation water was added and, hence, did not produce CO_2. This might be a reason for the lack of effect of SOM on the release of DIC-derived CO_2.

Another possible reason for the lack of treatment effects might be the high levels of organic matter (OM) in both soils used in this experiment. According to Murphy et al. (2012), more than 2.5% OM in sandy loam soil and more than 2% OM in loamy sand soil is considered a very high level of OM. Therefore, both the 'high OM' soil and 'low OM' soil used in this experiment were

relatively OM-rich. The impact of SOM may have been more apparent if one of the two soils had contained less than 2% OM.

Differences in the amount of soil packed into the collars may also have masked differences in the effects of OM content on the release of irrigation water-derived DIC. The bulk density of the 'high OM' soil was slightly lower (1.18 g cm^{-3}) than the 'low OM' soil (1.37 g cm^{-3}). As a consequence, a smaller amount of 'high OM' soil (~ 715 g) was packed into the assigned collars than that of the 'low OM' soil (~ 830 g) to maintain the same 3-cm space between the top of the soil surface and the rim of the collar. The smaller quantity of 'high OM' soil packed into these collars may have resulted in lower than expected rates of production of biogenic dissolved CO_2 in the soil solution which, in turn, caused the release of similar amounts of CO_2 from the irrigation water in both soil types. It may have been better to pack a similar amount of soil into both the 'high OM' and 'low OM' soil collars to better understand the effect of SOM on the release of CO_2 from irrigation water-derived DIC.

In conclusion, the proportion of DIC dissolved in the irrigation water that was released as CO_2 was not affected by soil organic matter (SOM). However, the data were extremely variable, particularly in the 'low OM' soil (Figure 5.3). As mentioned in Experiment 2 and Experiment 3, soil heterogeneity and uneven distribution of water in soil could have caused the data variability.

Chapter 6: Conclusion

My research was performed to give a better understanding of soil inorganic carbon (SIC) dynamics in the agro-environment. In contrast to a large number of studies examining soil organic carbon (SOC) cycling, very few studies have focused on SIC cycling in agricultural systems. This is because SIC stocks are generally concentrated deeper in the soil profile than SOC stocks (Wang et al., 2010) are believed to have a longer mean residence time (~85,000 years) (Schlesinger, 1985) than the SOC pool (maximum ~3000 years) (Hsieh, 1992). However, Monger and Gallegos (2000) found that the mean residence time of SIC can be as short as 120 years. Recent studies demonstrated that SIC may contribute to CO_2 emissions from agricultural soils (Bertrand et al., 2007; Cardinael et al., 2019). There is a growing understanding that irrigation, which is one of the most important agricultural inputs in arid and semi-arid regions, can change SIC budgets. The impacts of irrigation on SIC accumulation in soil can differ by depth. A study conducted by Khokhlova et al. (1997) demonstrated that 30 years of irrigation had resulted in a compact carbonate horizon in the lower part (1.2-1.6 m) of the soil profile in the semi-arid Russian steppe. By contrast, Entry et al. (2004) found that carbonates were accumulated in the top 1 m of a soil in the Western USA that had been irrigated for more than 30 years. According to Margaritz and Amiel (1981), the rate of carbonate transformation (dissolution-recrystallization) is higher in irrigated soil than in undisturbed soil. Recently, a study revealed that irrigation caused a reduction in carbonate concentration in the finest fractions of a soil (0-20 cm depth) located in the Iberian Peninsula (De Sotto et al., 2017). Reduction in soil surface SIC stocks following irrigation indicates the dissolution of the carbonates (Equation 1 to the left) and subsequent downward movement of the resulting ions (Ca^{2+}, Mg^{2+}, and HCO_3^-). The ions may re-precipitate deeper into the soil profile releasing CO_2 (Equation 1 to the right) (no net change in atmospheric CO_2 concentration) or enter groundwater and migrate to the ocean to remain for a longer period (C sequestration) (Sanderman, 2012).

The quality of irrigation water is a factor that affects SIC cycling. Irrigation with treated effluent promotes the accumulation of carbonates compared with irrigation using freshwater (Eshel et al., 2007); the presence of a higher amount of dissolved organic matter in the effluent may inhibit carbonate dissolution in soil (Lebron and Suarez, 1996). According to Wu et al. (2008) and Sanderman (2012), irrigation water with high electrical conductivity (EC) causes greater PC

accumulation and CO_2 release compared to irrigation water having low EC. Irrigation water in arid and semi-arid regions contains a large amount of dissolved Ca^{2+}, Mg^{2+} and HCO_3^- ions, which contribute to CO_2 and PC formation when applied to soil (Equation 1 to the right; Suarez, 2000; Sanderman, 2012; Bughio et al., 2015; Hannam et al., 2016). A recent study conducted by Hannam et al. (2019) also revealed that DIC in irrigation water can be released as CO_2 during irrigation events.

Increasing the quantity of irrigation water containing Ca^{2+}, Mg^{2+}, and HCO_3^- ions, may increase the neoformation of PC and CO_2. Therefore, irrigation method is another factor that can affect SIC cycling. Furrow irrigation applies more water over the soil surface than drip and sprinkler irrigation and hence, is expected to produce more PC and CO_2 (Entry et al., 2004). However, furrow irrigation with water having low EC may cause a reduction of PC in the topsoil as the large volume of water in the furrow irrigation is likely to dissolve the PC and moves the ions down the soil profile or into the groundwater. In such cases, drip or sprinkler irrigation may cause a greater accumulation of PC in the topsoil compared to furrow irrigation (Aase et al., 1998).

Based on the above discussions, the quality and quantity of irrigation water both play important roles in determining the fate of DIC once it is applied to the soil. However, little is known about the effect of soil properties on SIC dynamics in irrigated agriculture. In my research, I examined how irrigation water, containing a high amount of Ca^{2+} and HCO_3^- ions, affected the release of CO_2 from Okanagan Valley soils having different physical and chemical properties. In the laboratory, I used an EGM-4 (Environmental Gas Monitor) and Picarro cavity ring-down spectrometer to determine changes in CO_2 production and $^{13}CO_2$ composition that were caused by irrigation. CO_2 released from organic sources is depleted in ^{13}C compared with the CO_2 derived from inorganic sources (Smith and Epstein, 1971; Dzurec et al., 1985; Pawellek and Veizer, 1994; Stevenson and Verburg, 2006; Bertrand et al., 2007; Tamir et al., 2011); hence, $\delta^{13}CO_2$ was used to trace the sources of CO_2.

$$Ca^{2+}_{(aq)} + 2HCO_3^-{}_{(aq)} \longleftrightarrow CaCO_{3\,(s)} + H_2O_{(l)} + CO_{2(g)} \quad (1)$$

My first hypothesis was that the C from the HCO_3^- in irrigation water would contribute to CO_2 production when applied to soils. In support of my hypothesis, I found that DIC from irrigation water is evolved as CO_2 from most of the experimental soils. Out of 157.33 µmol DIC in 60 ml irrigation water applied to the soil surface (0.009 m^2), an average of -0.41 to 11.42 µmol was released as CO_2 depending on the experiments. My second hypothesis, evaluated in Chapter 2, was that the CO_2 efflux from the DIC in irrigation water would be higher in drier soils than the wetter soils. My rationale was that the higher moisture content of the wetter soils would increase organic CO_2 production in soil (microbial respiration) (Linn and Doran, 1984; Doran et al., 1990; Dilustro et al., 2005; Liu et al., 2009; Dong et al., 2014; Schimel, 2019; Butcher et al., 2020; Wang et al., 2020), which would discourage the production of PC and CO_2 (Equation 1 to the right) from the applied irrigation water containing Ca^{2+} and HCO_3^- ions. Contrary to my hypothesis, a greater proportion of DIC in irrigation water was released as CO_2 from wetter soils than drier soils.

Overall, the findings of this experiment are not in agreement with the field experiment conducted by Hannam et al. (2019). According to their findings, a higher amount of DIC in irrigation water was released as CO_2 from drier soil away from the dripper (~ 10% gravimetric moisture content) than the wetter soil under the dripper (~25% gravimetric moisture content). Although the soils used in both experiments were similar in characteristics (e.g., soil pH, soil texture, and organic matter content), a few differences in the experimental setup might have caused the discrepancy in the results. In my experiment, a fine nylon mesh was fixed at the bottom of the collars and the collars were seated on the desk. In the study carried out by Hannam et al. (2019), the collars were inserted into the soil and the bottom of the collars was open to allow the free flow of irrigation water.

According to Hannam et al. (2016), the wetter soil under the dripper in the Okanagan Valley contains more PC compared to the drier soil away from the dripper. In the experiment run by Hannam et al. (2019), the pre-existing PC in the wetter soil (under the dripper) might have discouraged Equation 1 to move to the right and thereby, limited the neoformation of CO_2 and PC from the irrigation water. Moreover, HCO_3^- ions added by the irrigation water or from the dissolved pre-existing PC might have leached downward to the soil profile through the open bottom of the collars. On the other hand, the lack of carbonates in the drier soil (away from the

dripper) might have promoted the neoformation of CO_2 and carbonates from the irrigation water (Equation 1 to the right). In addition, the deficiency of soil moisture in the drier soil might have discouraged the downward leaching of DIC (derived from irrigation water or PC), which might have promoted the production of CO_2 and PC from the irrigation water The soil under dripper also generally has a higher number of plant roots, macro- and micro-organisms than the soil away from the dripper. Therefore, more biogenic CO_2 was expected in the soil under dripper than the soil away from the dripper.

Differences in temperature conditions may be another reason that the two experiments found such contrasting results. The average temperature in the field was ~ 31 ℃ whereas, the temperature in the lab was 21 ℃. The higher temperature in the field might have caused off-gassing of CO_2 from the applied irrigation water when it made contact with the warm soil surface.

A final reason for the discrepancy in results between the two studies might be the long post-irrigation period (8 hours) in my experiment. According to Hannam et al. (2019), they found a sharp increase in $\delta^{13}CO_2$ within 5 minutes of irrigation after which the value started to decrease. If they had continued to monitor the value until 8 hours they may have found similar results as I found in my experiment. However, further research is required to draw a firm conclusion.

Regarding the contribution of irrigation-derived DIC to the total CO_2 efflux from the soil, the DIC in irrigation water in the experiment run by Hannam et al. (2019) contributed to between 9 and 15% of the total CO_2 efflux during irrigation. This supports my findings as I found a similar contribution of the DIC (~10%) to the total CO_2 emissions during the irrigation events.

My third hypothesis, evaluated in Chapter 3, was that the CO_2 efflux from the DIC in irrigation water would be higher in soil incubated at a higher temperature compared to the soil incubated at room temperature. My rationale was that the higher temperature would lead to off-gassing of CO_2 from the soil surface and promote the neoformation of PC and CO_2 from the irrigation water (Equation 1 to the right). Also, I hypothesized that the accelerated rate of acidifying reactions at higher temperature (Tan et al., 2018; Nguyen et al., 2019) would facilitate the production of CO_2 from DIC in the irrigation water. Unexpectedly, none of the DIC in the irrigation water appeared to be released at either of the soil temperatures (21 ℃, 30 ℃). Although there was a closed

chamber on top of the soil collar to prevent CO_2 and soil moisture (as water vapour) from escaping to the outside atmosphere, it was expected that the rate at which the CO_2 gas and the water vapour left the soil surface and accumulated in the chamber atmosphere would be accelerated at the higher temperature. However, the 30 ℃ temperature might not be high enough to increase the rate of evaporation and CO_2 off-gassing from the soil surface, which might be a reason for the lack of temperature effect on the release of CO_2 from the DIC in irrigation water.

Next, I hypothesized that the release of CO_2 from the DIC in irrigation water would be higher in coarser-textured soil than in the finer-textured soil. My rationale was that finer-textured soil would produce more biogenic CO_2 than the coarser-textured soil because of greater water retention (Kowalenko and Ivarson, 1978; Bouma and Bryla, 2000; Dilustro et al., 2005) at the soil surface. The elevated biogenic CO_2 in the finer-textured soil would discourage CO_2 production from the DIC in irrigation water (Equation 1 to the right). Contrary to my hypothesis, a greater proportion of DIC in irrigation water was emitted as CO_2 from the finer-textured soil (silt loam) than the coarser-textured soil (sandy loam). This might be due to the higher water-filled porosity in the sandy loam soil (~ 51%) than the silt loam soil (~ 39%) following irrigation, which caused greater microbial respiration and infiltration of water in the sandy loam soil. The lower pCO_2 in the silt loam soil might have encouraged a greater amount of CO_2 formation from the DIC in irrigation water compared to the sandy loam soil (Equation 1 to the right).

My final hypothesis was that a higher amount of DIC in irrigation water would be released from soil low in organic matter (OM) than the soil high in OM. My rationale was that higher OM content would increase the production of biological CO_2 (Ferrea et al., 2012; Thangarajan et al., 2013; Chen et al., 2021), which would discourage the neoformation of CO_2 from the DIC in irrigation water (Equation 1 to the right). However, my data did not support the hypothesis. Although a significant proportion of DIC in irrigation water was released as CO_2 from both the 'high OM' and 'low OM' soil, the OM content of the soil did not affect the proportion of DIC that was released.

After analyzing the data from all of the four experiments, soil moisture appears to be the most important soil factor influencing CO_2 release from DIC in irrigation water. Moisture in soil determines the water-filled porosity of soil, depending on the texture of the soil, and thereby,

controls microbial respiration and subsequent dissolution of the biogenic CO_2, which influences the release of CO_2 from DIC in irrigation water. In my first experiment with Skaha soil, more DIC-derived CO_2 was detected in soil maintained at 25% moisture than the soil maintained at 15% moisture, which might be due to the less microbial respiration in the wetter soil caused by excess moisture upon irrigation. Although smaller in quantity, CO_2 derived from DIC in irrigation water was detected in soil maintained at 15% gravimetric soil moisture. This finding supports my third and fourth experiments, where I used different soils but the soils were maintained at 15% gravimetric moisture content before irrigation; CO_2 from DIC in irrigation water was detected from all of the soils. In my first experiment with Skaha soil, no DIC-derived CO_2 was detected in soil maintained at 10% gravimetric moisture. My second experiment, where I also used Skaha soil and maintained the soil at 10% gravimetric moisture content, no CO_2 from DIC in irrigation water was detected at either of the soil temperatures (21 ℃, 30 ℃). As mentioned earlier, the reason for no DIC-derived CO_2 when applied to soils with 10% gravimetric moisture is unknown. According to Yuste et al. (2003), soil respiration was not affected by temperature when the volumetric water content of the soil was less than 15%. This corroborates my finding because the 10% gravimetric moisture content of the soil used in my experiment is equivalent to ~13% volumetric water content.

One of the key strengths of my research was that I was able to vary only the soil characteristic of interest. In Experiment 1, where I tested the effects of soil moisture on CO_2 emission from irrigation-derived DIC, I used one type of soil but moistened the treated samples to five gravimetric moisture levels before the experimentation. Therefore, any differences in the formation of CO_2 were likely through the effects of the soil moisture. Similarly, in Experiment 2, I used one type of soil but kept soil collars at two different temperatures so that only the effects of soil temperature could be determined. While testing the effects of soil texture on DIC-derived CO_2 emissions, I used Penticton soils from two different sites. They were similar in chemical properties but differed in their texture type. In my final experiment, I used two different soils that had similar physical and chemical properties but differed in their OM contents so that the data reflects only the effects of soil organic matter (SOM). Thus, in each of my experiments, I used soils that differed in the factor (soil property) of interest but were the same or similar in other properties so that only the effects of the soil factor that I was interested in could be detected.

Another key strength of my research was the use of $\delta^{13}CO_2$ to distinguish the response of inorganic CO_2 from biogenic CO_2; inorganic CO_2 has less negative $\delta^{13}CO_2$ value than biogenic CO_2 (Smith and Epstein, 1971; Dzurec et al., 1985; Pawellek and Veizer, 1994; Stevenson and Verburg, 2006; Bertrand et al., 2007; Tamir et al., 2011). This is quite a unique aspect of my research as very few other researchers have used this approach. Moreover, my work is one of the few to examine the role of SIC in CO_2 efflux from agricultural soils. Previously, most studies of CO_2 efflux from the agricultural soils have focused on the contribution of SOC.

A few limitations, however, became apparent after the completion of Experiment 1. First, I could not compare the results between the two irrigation water types because only the soil moisture treatments, not the water types, were randomized during each 'run' of the experiment. Second, the fine nylon fabric, which was fixed at the bottom of the soil collars, might have allowed ^{13}C-rich atmospheric air into the collar. Lastly, the water I used as a control was not 100% deionized; NaCl was added to the deionized water in order to be a better control for the artificial lake water. However, changes were made in the subsequent experiments to address these concerns; both the soil factor and the irrigation water chemistry treatments were randomly applied during each run, a plastic sheet was fixed at the bottom of each soil collar, in addition to the fine nylon fabric, to prevent the possible intrusion of atmospheric air into the collar, and 100% deionized water was used as control. I used the 100% deionized water assuming that differences in CO_2 emissions between deionized water and Okanagan Lake water treatments were only the results of DIC dissolved in the Lake water. However, there might have been other factors (i.e., other chemicals and organic C dissolved in the Okanagan Lake water), which could affect the release of irrigation-derived DIC by influencing the biogenic CO_2 production in soil. This might be a limitation of my study. In Experiments 2, 3, and 4, I used two soils maintained at one moisture level to test the effects of soil property and water quality on CO_2 emissions from DIC in irrigation water. The use of the limited number of soil and soil moisture content is another limitation of my research.

For future studies, I would add all the substances (except the Ca^{2+} and HCO_3^- ions), typically found in the Okanagan Lake water, to the deionized water to better understand the contribution of DIC to CO_2 efflux. I would test a greater range of the selected soil properties at a wider range of soil moisture contents to better understand the interacting effects of the experimental factors

(soil property and water quality). Moreover, my overall research findings generated two new research questions in my mind. What was the amount of soil inorganic carbon (i.e., soil carbonates) that was precipitated in the soil after irrigation? How would long-term accumulation of soil carbonates affect soil properties? In future studies, I would, therefore, like to address these questions by determining 1) the effects of long-term irrigation (water containing $Ca^{2+/}Mg^{2+}$and HCO_3^-) on the accumulation of soil carbonates deeper into the soil profile and 2) the effects of long-term accumulation of carbonates on soil physical, chemical, and biological properties as well as the associated crop responses.

The potential application of my research is for farmers who use DIC-rich water to irrigate their fields. Based on my overall findings, which reflect the importance of soil moisture on CO_2 emissions from DIC in irrigation water, farmers who use DIC-rich water for irrigation should irrigate their fields less frequently. This is because the less frequent irrigation keeps soil moisture lower compared to more frequent irrigation although the total amount of applied water to the field in both cases is the same (Neilsen et al., 2010). However, crop yield should also be considered in this regard. Some studies conducted in the Okanagan Valley, BC, demonstrated the effects of irrigation frequency on crop yield, crop quality, and greenhouse gas emissions. According to Neilsen et al. (1997), low irrigation frequency decreased the growth and yield of apple trees compared to a higher irrigation frequency. The finding was supported by the previous study of Neilsen et al. (1995) where the growth of apple trees increased when irrigated more frequently, although leaf nitrogen concentration tended to decrease with the high frequency of the irrigation. Recently, Hannam et al. (2013) observed that reduced quantity of irrigation increased the grape juice YAN (yeast-assimilable nitrogen); however, reduced irrigation frequency did not affect the grape juice YAN but did increase crop yield. According to Neilsen et al. (2010), sweet cherry production was increased with less frequent irrigation compared to the more frequent irrigation. Considering the effect of irrigation frequency on greenhouse gas emission, Fentabil et al. (2016) conducted a study in a drip-irrigated apple orchard. They found a reduction in N_2O emissions from the soil with less frequent irrigation. Based on the above discussions, less frequent irrigation according to crop water demand is expected to reduce CO_2 emissions from the DIC in irrigation water, however, the growers should also consider the possible effects of less frequent irrigation on crop production, which sometimes may negative.

In spite of the above discussion, the CO_2 released from DIC in irrigation water in the present study added very little (<10%) to overall CO_2 emissions (Figure 6.1), although I did not track the fate of this DIC beyond a few hours after irrigation (as the soil dried out). Under natural conditions, when the growing season ends and irrigation is stopped, dissolved DIC can be washed deeper into the soil and subsequently can enter into long-term storage in the groundwater or the ocean (Sanderman, 2012). Therefore, the amount of CO_2 released from the DIC during irrigation is very small compared to the extra CO_2 that irrigated plants absorb from the atmosphere during photosynthesis and, subsequently, add to the soils as organic C. Plants absorb CO_2 from the atmosphere and convert it into organic compounds during the process of photosynthesis. Over time, this C is added to the soil in two ways: 1) root exudates (Cheng et., 1993; Paul and Clark, 1996; Kuzyakov and Demin, 1998; Kuzyakov and Domanski, 2000), and 2) decomposition/breakdown (by microbes or other soil fauna) of dead or senescing plant roots and aboveground biomass (e.g., shoots, branches, leaves, fruit) deposited on top of the soil (Schlesinger, 1977; Paul and Clark, 1996; De Silva et al., 2010). The transport of CO_2 from the atmosphere to soil is expected to be elevated in irrigated agriculture as irrigation increases crop productivity (Boreux et al., 2016; Surendran et al., 2016; Lu et al., 2018; Smidt et al., 2019; Chilundo et al., 2020). According to the FAO (2000), irrigation increases crop productivity by more than 100%. Recently it has been found that irrigated agriculture nearly doubled the organic matter content of the Okanagan Valley soils (Midwood et al. 2021). Thus, DIC is not a large source of CO_2 during irrigation and is hugely outweighed by the atmospheric C taken up by irrigated plants.

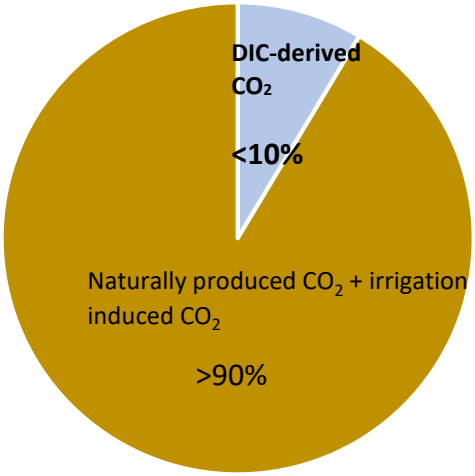

Figure 6. 1 The amount of CO_2 (%) released from different sources during irrigation

As the world population is growing very rapidly, irrigated agriculture is being expanded worldwide to meet the increasing food demand (Wu et al., 2008; FAO, 2011; Alexandratos and Bruinsma, 2012). My research findings have improved our understanding of the effects of irrigation water chemistry on CO_2 emissions from different soils of the Okanagan region. The amount of CO_2 emitted from the DIC in irrigation water, although small compared to the total CO_2 emitted during irrigation, indicates that irrigation can cause changes in the SIC budgets of agroecosystems in arid and semi-arid regions. If research in this field continues, the agricultural sector will benefit, as the findings will help growers implement efficient irrigation management strategies across BC and in other arid and semi-arid environments. This will enhance the long-term productivity of good quality crops by improving the soil quality and, potentially, serving as an effective mitigation tool for global warming.

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Appendices

Appendix 1: Calculation of the proportion of HCO_3^- in irrigation water released as CO_2

The proportion of total soil surface CO_2 efflux that was emitted from the DIC dissolved in the irrigation water from Okanagan Lake (P_{bicarb}) was determined using equation 5 derived from Hannam et al. (2019), where the $\delta^{13}C_{bicarb}$ of the Okanagan Lake water was assumed to be -2.6‰ (Hannam et al., 2016), the $\delta^{13}C_{baseline}$ was the $\delta^{13}CO_2$ of the chamber gas space after the pre-irrigation period, and $\delta^{13}C_{efflux}$ was the $\delta^{13}CO_2$ of the chamber gas space after the post-irrigation period (i.e., after the application of Okanagan Lake water).

$$P_{bicarb} = (\delta^{13}C_{efflux} - \delta^{13}C_{baseline}) / (\delta^{13}C_{bicarb} - \delta^{13}C_{baseline}) \quad (5)$$

The quantity of DIC-derived CO_2 (µmol) was then estimated using equation 5a

$$Q_{bicarb} = P_{bicarb} \times ELW \times \pi r^2 \times T_{ir} \quad (5a)$$

[Q_{bicarb} was the quantity of DIC-derived CO_2 (µmol), ELW was the CO_2 efflux (µmol m^{-2} s^{-1}) after irrigation with Okanagan Lake water, πr^2 was the soil surface area (m^2), and T_{ir} was the post-irrigation period (s)]

The proportion of DIC that was released as CO_2 was subsequently determined using equation 5b

$$P_{bicarb\text{-}CO2} = Q_{bicarb} / D_{bicarb} \quad (5b)$$

[$P_{bicarb\text{-}CO2}$ was the proportion of DIC that was released as CO_2, D_{bicarb} was the quantity of DIC in the applied Okanagan Lake water (157.3 µmol)]

Calculation of 157.3 µmol:

The average concentration of HCO_3^- typically found in Okanagan Lake water is 160 mg L^{-1} = 160 mg per 1000 ml. Therefore, 60 ml of Okanagan Lake water (the amount applied to the soil) contains (160 mg × 60 ml)/1000 ml = 9.6 mg HCO_3^- = 9600 µg HCO_3^-

The molecular mass of the HCO_3^- is 61.02 g mol^{-1} = 61.02 µg $µmol^{-1}$. Therefore, the quantity of DIC in the applied Okanagan Lake water was 9600 µg /61.02 µg $µmol^{-1}$= 157.3 µmol.

Example from experiment:

$P_{bicarb} = (\delta^{13}C_{efflux} - \delta^{13}C_{baseline}) / (\delta^{13}C_{bicarb} - \delta^{13}C_{baseline})$

$= \{-16.18 - (-18.09) / \{-2.6 – (-18.09)\}$

$= 0.12$

[$\delta^{13}C_{efflux} = -16.18$, $\delta^{13}C_{baseline} = -18.09$, $\delta^{13}C_{bicarb} = -2.6$]

Now, $Q_{bicarb} = P_{bicarb} \times ELW \times \pi r^2 \times T_{ir}$

$= 0.12 \times 0.16$ µmol m^{-2} $s^{-1} \times 0.01$ $m^2 \times 28800$ s

$= 5.53$ µmol

[$ELW = 0.16$ µmol m^{-2} s^{-1}, $\pi r^2 = 3.14 (0.05 \text{ m})^2 = 0.01$ m^2, $T_{ir} = 8$ hours = 28800 s]

$P_{bicarb\text{-}CO2} = Q_{bicarb} / D_{bicarb}$

$= 5.53$ µmol/157.3 µmol

$= 0.04$

Appendix 2: The results of statistical analyses for Experiment 1

The Effect of Irrigation Water Chemistry and Soil Moisture on CO_2 Production

Table A.2. 1 ^{13}C-CO_2 and efflux rate before and after irrigation. Values are means with standard deviation in parentheses)

Soil moisture	Before irrigation		After irrigation			
			Okanagan Lake water		Control water	
	^{13}C(‰)	CO_2 efflux (µmol m^{-2} s^{-1})	^{13}C(‰)	CO_2 efflux (µmol m^{-2} s^{-1})	^{13}C(‰)	CO_2 efflux (µmol m^{-2} s^{-1})
10%	-19.0 (1.4)	0.08 (0.01)	-19.5 (1.4)	0.28 (0.06)	-23.3 (0.7)	0.33 (0.03)
15%	-18.3 (1.8)	0.08 (0.02)	-16.1 (0.3)	0.16 (0.01)	-22.8 (1.8)	0.33 (0.01)
20%	-19.8 (2.1)	0.10 (0.03)	-18.3 (3.0)	0.22 (0.09)	-23.3 (1.6)	0.33 (0.04)
25%	-20.2 (2.2)	0.11 (0.05)	-14.1 (1.0)	0.17 (0.06)	-23.9 (1.6)	0.23 (0.05)
30%	-22.6 (1.2)	0.13 (0.02)	-19.2 (4.1)	0.19 (0.07)	-16.8 (1.1)	0.09 (0.01)

Change in CO_2 efflux rate (µmol m^{-2} s^{-1}) after irrigation

Table A.2. 2 One-way ANOVA and Tukey test (soil moisture as the factor: Five moisture levels) when treated with Okanagan Lake water

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
moist	4	0.05503	0.013758	4.346	**0.0109** *
Residuals	20	0.06331	0.003166		

Signif. codes:	0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1				
Tukey multiple comparisons of means					
95% family-wise confidence level					
Fit: aov(formula = model)					
$`onewaylakeefflux$moist`					
	diff	lwr	upr	p adj	
15%-10%	**-0.107629354**	**-0.21411302**	**-0.001145686**	**0.0468046**	
20%-10%	-0.080680089	-0.17666317	0.015302992	0.1267783	
25%-10%	-0.111686679	-0.22462966	0.001256306	0.0535132	
30%-10%	**-0.151857646**	**-0.27481440**	**-0.028900898**	**0.0111005**	
20%-15%	0.026949265	-0.06903382	0.122932346	0.9147751	
25%-15%	-0.004057325	-0.11700031	0.108885660	0.9999669	
30%-15%	-0.044228293	-0.16718504	0.078728456	0.8163133	
25%-20%	-0.031006590	-0.13410896	0.072095778	0.8935534	
30%-20%	-0.071177558	-0.18516151	0.042806399	0.3649459	
30%-25%	-0.040170967	-0.16876221	0.088420278	0.8799574	

Table A.2. 3 One-tailed t-tests performed for each of the soil moisture levels when treated with Okanagan Lake water. Alternate hypothesis: true mean is greater than 0

Soil moisture	Df	Mean of X	p-value
10%	4	0.2048442	**0.0005033**
15%	4	0.09721489	**8.493e-06**
20%	7	0.1241642	**0.0004774**
25%	3	0.09315757	**0.04114**
30%	2	0.0529866	0.1287

Table A.2. 4 One-way ANOVA and Tukey test (soil moisture as the factor: Five moisture levels) when treated with control water

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
moist	4	0.16787	0.04197	38.55	**4.77e-06** ***
Residuals	10	0.01089	0.00109		

Signif. codes:	0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1				
Tukey multiple comparisons of means					
95% family-wise confidence level					
Fit: aov(formula = model)					
$`onewaydeionizedefflux$moist`					
	diff	lwr	upr	p adj	
15%-10%	-0.02076379	-0.1094283	0.06790074	0.9334417	
20%-10%	-0.05002606	-0.1386906	0.03863847	0.3961771	
25%-10%	**-0.17006180**	**-0.2587263**	**-0.08139727**	**0.0006470**	
30%-10%	**-0.27988021**	**-0.3685447**	**-0.19121567**	**0.0000087**	
20%-15%	-0.02926227	-0.1179268	0.05940226	0.8097230	
25%-15%	**-0.14929800**	**-0.2379625**	**-0.06063347**	**0.0017826**	
30%-15%	**-0.25911641**	**-0.3477809**	**-0.17045188**	**0.0000175**	
25%-20%	**-0.12003573**	**-0.2087003**	**-0.03137120**	**0.0084003**	
30%-20%	**-0.22985414**	**-0.3185187**	**-0.14118961**	**0.0000510**	
30%-25%	**-0.10981841**	**-0.1984829**	**-0.02115388**	**0.0148741**	

Table A.2. 5 One-tailed t-tests performed for each of the soil moisture levels when treated with control water. Alternate hypothesis: true mean is greater than 0 or less than 0

Soil moisture	Df	Mean of X	p-value
10%	2	0.252807	**0.001129**
15%	2	0.2320432	**0.0009696**
20%	2	0.2027809	**0.003066**
25%	2	0.08274519	0.06777
30%	2	-0.02707322	0.07999

Change in $\delta^{13}CO_2$ (‰) after irrigation

Table A.2. 6 One-way ANOVA and Tukey test (soil moisture as the factor: Five moisture levels) when treated with Okanagan Lake water (after log transformation)

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
moist	4	6.646	1.6616	9.671	**0.00016** ***
Residuals	20	3.436	0.1718		

Signif. codes:	0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1				
Tukey multiple comparisons of means					
95% family-wise confidence level					
Fit: aov(formula = model)					
$`onewaylake$moist`					
	diff	lwr	upr	p adj	
15%-10%	**1.0221646**	**0.23771012**	**1.8066192**	**0.0070623**	
20%-10%	**0.8896518**	**0.18255403**	**1.5967495**	**0.0095230**	
25%-10%	**1.5863384**	**0.75429869**	**2.4183780**	**0.0001227**	
30%-10%	**1.3609925**	**0.45518246**	**2.2668026**	**0.0018398**	
20%-15%	-0.1325129	-0.83961061	0.5745849	0.9792536	
25%-15%	0.5641737	-0.26786595	1.3962134	0.2886524	
30%-15%	0.3388279	-0.56698218	1.2446379	0.7946030	
25%-20%	0.6966866	-0.06285825	1.4562314	0.0820452	
30%-20%	0.4713407	-0.36836765	1.3110491	0.4677332	
30%-25%	-0.2253458	-1.17266466	0.7219730	0.9513475	

Table A.2. 7 One-tailed t-tests performed for each of the soil moisture levels when treated with Okanagan Lake water. Alternate hypothesis: true mean is greater than 0 or less than 0 (*Using untransformed data*)

Soil moisture	Df	Mean of X	p-value
10%	4	-1.282	**0.01097**
15%	4	1.242	**0.003355**
20%	7	0.98625	0.07257
25%	3	4.4175	**0.000109**
30%	2	3.506667	0.1056

Table A.2. 8 One-way ANOVA and Tukey test (soil moisture as the factor: Five moisture levels) when treated with control water

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
moist	4	158.38	39.59	42.9	**2.9e-06** ***
Residuals	10	9.23	0.92		

Signif. codes:	0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1				
Tukey multiple comparisons of means					
95% family-wise confidence level					
Fit: aov(formula = model)					
$`onewaydeionized$moist`					
	diff	lwr	upr	p adj	
15%-10%	0.1700000	-2.411425	2.751425	0.9994181	
20%-10%	0.8233333	-1.758091	3.404758	0.8271061	
25%-10%	1.5166667	-1.064758	4.098091	0.3606294	
30%-10%	**8.6400000**	**6.058575**	**11.221425**	**0.0000051**	
20%-15%	0.6533333	-1.928091	3.234758	0.9142613	
25%-15%	1.3466667	-1.234758	3.928091	0.4661956	
30%-15%	**8.4700000**	**5.888575**	**11.051425**	**0.0000061**	
25%-20%	0.6933333	-1.888091	3.274758	0.8965002	
30%-20%	**7.8166667**	**5.235242**	**10.398091**	**0.0000127**	
30%-25%	**7.1233333**	**4.541909**	**9.704758**	**0.0000293**	

Table A.2. 9 One-tailed t-tests performed for each of the soil moisture levels when treated with control water. Alternate hypothesis: true mean is greater than 0 or less than 0

Soil moisture	Df	Mean of X	p-value
10%	2	-3.06	**0.009476**
15%	2	-2.89	**0.001171**
20%	2	-2.236667	**0.005109**
25%	2	-1.543333	0.1104
30%	2	5.58	**0.008051**

The proportion of HCO_3^- in Okanagan Lake water that was released as CO_2

Table A.2. 10 One-way ANOVA and Tukey test (soil moisture as the factor) when treated with Okanagan Lake water

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
moist	4	0.03069	0.007673	11.36	**5.63e-05** ***
Residuals	20	0.01350	0.000675		

Signif. codes:	0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1				
Tukey multiple comparisons of means					
95% family-wise confidence level					
Fit: aov(formula = model)					
$`effluxfromirrumole$moist`					
	diff	lwr	upr	p adj	
15%-10%	**9.159691**	**1.4224042**	**16.896977**	**0.0155477**	
20%-10%	**7.707964**	**0.7336679**	**14.682260**	**0.0258640**	
25%-10%	**17.795454**	**9.5888223**	**26.002086**	**0.0000226**	
30%-10%	**12.422418**	**3.4881692**	**21.356667**	**0.0039258**	
20%-15%	-1.451727	-8.4260228	5.522569	0.9696365	
25%-15%	**8.635763**	**0.4291315**	**16.842395**	**0.0361597**	
30%-15%	3.262727	-5.6715215	12.196976	0.8081331	
25%-20%	**10.087490**	**2.5958947**	**17.579086**	**0.0052765**	
30%-20%	4.714454	-3.5678160	12.996725	0.4542216	
30%-25%	-5.373036	-14.7166970	3.970625	0.4443737	

Table A.2. 11 One-tailed t-tests performed for each of the soil moisture levels when treated with Okanagan Lake water. Alternate hypothesis: true mean is greater than 0 or less than 0

Soil moisture	Df	Mean of X	p-value
10%	4	-0.03707615	**0.01373**
15%	4	0.02114346	**0.002866**
20%	7	0.01191619	0.1296
25%	3	0.07603295	**0.009882**
30%	2	0.04188157	0.08005

Appendix 3: The results of statistical analyses for Experiment 2

The Effect of Irrigation Water Chemistry and Soil Temperature on CO_2 Production

Table A.3. 1 ^{13}C-CO_2 and efflux rate before and after irrigation. Values are means with standard deviation in parentheses)

Soil temperature	Before irrigation		After irrigation			
			Okanagan Lake water		Control water	
	^{13}C(‰)	CO_2 efflux (µmol m^{-2} s^{-1})	^{13}C(‰)	CO_2 efflux (µmol m^{-2} s^{-1})	^{13}C(‰)	CO_2 efflux (µmol m^{-2} s^{-1})
Low	-21.1 (0.9)	0.12 (0.02)	-21.6 (1.7)	0.42 (0.05)	-23.4 (0.7)	0.33 (0.03)
High	-23.7 (0.7)	0.24 (0.08)	-23.8 (0.5)	0.80 (0.24)	-24.9 (0.4)	0.86 (0.23)

Change in CO_2 efflux rate (µmol m^{-2} s^{-1}) after irrigation

Table A.3. 2 Two-way ANOVA: Soil temperature and water quality as factors (after log transformation)

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Block	1	0.005	0.005	0.066	0.800
temp	1	4.014	4.014	52.424	**7.14e-07 *** **
water	1	0.120	0.120	1.563	0.226
temp:water	1	0.116	0.116	1.521	0.233
Residuals	19	1.455	0.077		

Signif. codes:		0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1			

Table A.3. 3 One-tailed t-tests performed for each of the soil temperature levels when treated with deionized or Okanagan Lake water. Alternate hypothesis: true mean is greater than 0 (*Using untransformed data*)

	‘high temp’ soil (30 °C), Deionized	‘high temp’ soil (30 °C), Okanagan Lake	‘low temp’ soil (21 °C), Deionized	‘low temp’ soil (21 °C), Okanagan Lake
Df	5	5	5	5
Mean of X	0.5966667	0.5816667	0.2183333	0.2883333
p-value	**0.0004094**	**0.0001467**	**1.159e-05**	**7.308e-06**

Change in $\delta^{13}CO_2$ (‰) after irrigation

Table A.3. 4 Two-way ANOVA: Soil temperature and water quality as factors

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Block	1	1.450	1.450	4.389	**0.04980 ***
temp	1	2.190	2.190	6.630	**0.01856 ***
water	1	21.188	21.188	64.136	**1.65e-07 *****
temp:water	1	6.131	6.131	18.558	**0.00038 *****
Residuals	19	6.277	0.330		

Signif. codes:		0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1			

One-way ANOVA (Soil temperature as the factor):

Table A.3. 5 One-way ANOVA (soil temperature as the factor: two temperature levels: 21℃ and 30 ℃) when treated with deionized water

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Block	1	0.127	0.127	0.516	0.490976
temp	1	7.825	7.825	31.862	**0.000316 *****
Residuals	9	2.210	0.246		

Signif. codes:		0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1			

Table A.3. 6 One-way ANOVA (soil temperature as the factor: two temperature levels: 21℃ and 30 ℃) when treated with Okanagan Lake water

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Block	1	1.815	1.8149	4.569	0.0613 .
temp	1	0.496	0.4961	1.249	0.2927
Residuals	9	3.575	0.3972		

Signif. codes:		0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1			

One-way ANOVA (Water Quality as the factor):

Table A.3. 7 One-way ANOVA (water quality as the factor: Okanagan Lake water and deionized water) at room temperature (21℃)

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Block	1	0.553	0.553	1.348	0.275
water	1	25.056	25.056	61.073	**2.67e-05 *** **
Residuals	9	3.692	0.410		

Signif. codes:		0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1			

Table A.3. 8 One-way ANOVA (water quality as the factor: Okanagan Lake water and deionized water) at hot temperature (30 ℃)

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Block	1	0.9202	0.9202	3.234	0.1057
water	1	2.2620	2.2620	7.949	**0.0201 ***
Residuals	9	2.5611	0.2846		

Signif. codes:		0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1			

Table A.3. 9 One-tailed t-tests performed for each of the soil temperature levels when treated with deionized or Okanagan Lake water. Alternate hypothesis: true mean is less than 0

	'high temp' soil (30 ℃), Deionized	'high temp' soil (30 ℃), Okanagan Lake	'low temp' soil (21 ℃), Deionized	'low temp' soil (21 ℃), Okanagan Lake
Df	5	5	5	5
Mean of X	-1.093333	-0.225	-2.708333	-0.195
p-value	**0.0001769**	0.2541	**5.631e-05**	0.3885

The proportion of HCO_3^- in Okanagan Lake water that was released as CO_2

Table A.3. 10 One-way ANOVA (Soil temperature as the factor) when treated with Okanagan Lake water

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Block	1	0.004038	0.004038	4.047	0.0751 .
temp	1	0.000742	0.000742	0.744	0.4108
Residuals	9	0.008980	0.000998		

Signif. codes:		0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1			

Table A.3. 11 One-tailed t-tests performed for each of the soil temperature levels when treated with Okanagan Lake water. Alternate hypothesis: true mean is greater than 0 or less than 0

Soil	Df	Mean of X	p-value
'low temp' soil (21 ℃),	5	0.005279006	0.3203
'high temp' soil (30 ℃),	5	-0.0104511	0.2925

Appendix 4: The results of statistical analyses for Experiment 3

The Effect of Irrigation Water Chemistry and Soil Texture on CO_2 Production

Table A.4. 1 ^{13}C-CO_2 and efflux rate before and after irrigation. Values are means with standard deviation in parentheses)

Soil texture	Before irrigation		After irrigation			
			Okanagan Lake water		Control water	
	^{13}C(‰)	CO_2 efflux (µmol m^{-2} s^{-1})	^{13}C(‰)	CO_2 efflux (µmol m^{-2} s^{-1})	^{13}C(‰)	CO_2 efflux (µmol m^{-2} s^{-1})
Sandy loam	-21.7 (1.3)	0.14 (0.04)	-19.2 (2.9)	0.26 (0.11)	-21.2 (4.8)	0.20 (0.12)
Silt loam	-22.7 (1.2)	0.15 (0.03)	-19.0 (4.0)	0.30 (0.10)	-22.4 (3.5)	0.22 (0.10)

Change in CO_2 efflux rate (µmol m^{-2} s^{-1}) after irrigation

Table A.4. 2 Two-way ANOVA: Soil texture and water quality as factors

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Block	1	0.01997	0.01997	3.446	0.0790 .
texture	1	0.00199	0.00199	0.344	0.5646
water	1	0.03563	0.03563	6.148	**0.0227 ***
texture:water	1	0.00016	0.00016	0.028	0.8695
Residuals	19	0.11012	0.00580		

Signif. codes:	0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1				

Table A.4. 3 One-tailed t-tests performed for each of the soil texture types when treated with Deionized or Okanagan Lake water. Alternate hypothesis: true mean is greater than 0

	Sandy loam, Deionized	Sandy loam, Okanagan Lake	Silt loam, Deionized	Silt loam, Okanagan Lake
Df	5	5	5	5
Mean of X	0.0464955	0.1287313	0.0698917	0.1417792
p-value	0.1441	**0.007045**	**0.03445**	**0.001403**

Change in $\delta^{13}CO_2$ (‰) after irrigation

Table A.4. 4 Two-way ANOVA: Soil texture and water quality as factors

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Block	1	26.83	26.827	3.716	0.0690 .
texture	1	1.71	1.712	0.237	0.6318
water	1	30.08	30.083	4.167	0.0554 .
texture:water	1	7.74	7.741	1.072	0.3134
Residuals	19	137.17	7.219		

Signif. codes:		0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1			

Table A.4. 5 One-tailed t-tests performed for each of the soil texture types when treated with Deionized or Okanagan Lake water. Alternate hypothesis: true mean is greater than 0

	Sandy loam, Deionized	Sandy loam, Okanagan Lake	Silt loam, Deionized	Silt loam, Okanagan Lake
Df	5	5	5	5
Mean of X	0.9316667	2.035	0.33	3.705
p-value	0.2703	**0.0396**	0.3883	**0.01264**

The proportion of HCO_3^- in Okanagan Lake water that was released as CO_2

Table A.4. 6 One-way ANOVA (soil texture as the factor) when treated with Okanagan Lake water

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Block	1	0.000005	0.000005	0.005	0.9430
texture	1	0.005303	0.005303	6.294	**0.0334 ***
Residuals	9	0.007582	0.000842		

Signif. codes:		0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1			

Table A.4. 7 One-tailed t-tests performed for each of the soil texture types when treated with Okanagan Lake water. Alternate hypothesis: true mean is greater than 0

Soil texture	Df	Mean of X	p-value
Coarse	5	0.0277697	**0.04713**
finer	5	0.06981186	**0.0002119**

Appendix 5: The results of statistical analyses for Experiment 4

The Effect of Irrigation Water Chemistry and Soil Organic Matter (SOM) on CO_2 Production

Table A.5. 1 ^{13}C-CO_2 and efflux rate before and after irrigation. Values are means with standard deviation in parentheses)

SOM	Before irrigation		After irrigation			
			Okanagan Lake water		Control water	
	^{13}C(‰)	CO_2 efflux (μmol m^{-2} s^{-1})	^{13}C(‰)	CO_2 efflux (μmol m^{-2} s^{-1})	^{13}C(‰)	CO_2 efflux (μmol m^{-2} s^{-1})
High	-23.1 (1.0)	0.22 (0.09)	-21.0 (0.7)	1.3 (0.41)	-25.4 (1.2)	1.0 (0.38)
Low	-23.0 (1.0)	0.26 (0.09)	-20.8 (2.6)	1.5 (0.75)	-24.4 (2.1)	1.0 (0.53)

Change in CO_2 efflux rate (μmol m^{-2} s^{-1}) after irrigation

Table A.5. 2 Two-factor ANOVA: SOM content and water quality as factors

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Block	1	2.8080	2.8080	35.550	**9.73e-06 *** **
SOM	1	0.0073	0.0073	0.093	0.76365
Water	1	0.8664	0.8664	10.969	**0.00367 ** **
SOM:Water	1	0.0523	0.0523	0.662	0.42603
Residuals	19	1.5008	0.0790		

Signif. codes:	0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1				

Table A.5. 3 One-tailed t-tests performed for each of the soil types when treated with Deionized or Okanagan lake water. Alternate hypothesis: true mean is greater than 0

	‘high OM’ soil (7.66% OM), Deionized	‘high OM’ soil (7.66% OM), Okanagan Lake	‘low OM’ soil (3.34% OM), Deionized	‘low OM’ soil (3.34% OM), Okanagan Lake
Df	5	5	5	5
Mean of X	0.8033333	1.09	0.745	1.218333
p-value	**0.000522**	**0.0002609**	**0.006939**	**0.002934**

OM = Organic Matter

Change in $\delta^{13}CO_2$ (‰) after irrigation

Alternative of Two-way ANOVA (as the data are non-homogeneous):

Table A.5. 4 Regression analysis (using the untransformed data) equivalent to One-way Welch ANOVA (collapsing the two factors: SOM and water quality into one factor)

(Regression analysis was performed three times considering one of the four treatments as a reference each time)

	Estimate	Std. Error	t	p
Intercept	-0.03508	0.646001	-0.05431	0.957256
Block	-0.34664	0.132094	-2.62422	0.016698
low OM, deionized-high OM, deionized	-1.015	0.638074	-1.59072	0.128173
low OM, deionized-high OM, lake	3.306667	0.638074	5.182262	**5.30E-05**
low OM, deionized-low OM, lake	3.346667	0.638074	5.24495	**4.61E-05**

	Estimate	Std. Error	t	p
Intercept	-1.05008	0.646001	-1.62551	0.120525
Block	-0.34664	0.132094	-2.62422	0.016698
high OM, deionized-high OM, lake	4.321667	0.638074	6.772986	**1.81E-06**
high OM, deionized-low OM, deionized	1.015	0.638074	1.590724	0.128173
high OM ,deionized-low OM, lake	4.361667	0.638074	6.835675	**1.59E-06**

	Estimate	Std. Error	t	p
Intercept	3.311583	0.646001	5.126284	6.00E-05
Block	-0.34664	0.132094	-2.62422	0.016698
low OM,lake-high OM,deionized	-4.36167	0.638074	-6.83567	1.59E-06
low OM,lake-high OM,lake	-0.04	0.638074	-0.06269	0.950669
low OM,lake-low OM,deionized	-3.34667	0.638074	-5.24495	4.61E-05

Table A.5. 5 One-tailed t-tests performed for each of the soil types when treated with Deionized or Okanagan lake water. Alternate hypothesis: true mean is greater than 0 or less than 0 (*Using untransformed data*)

	'high OM' soil (7.66% OM), Deionized	'high OM' soil (7.66% OM), Okanagan Lake	'low OM' soil (3.34% OM), Deionized	'low OM' soil (3.34% OM), Okanagan Lake
Df	5	5	5	5
Mean of X	-2.263333	2.058333	-1.248333	2.098333
p-value	**4.538e-06**	**1.567e-05**	0.0595	**0.01982**

The proportion of HCO_3^- in Okanagan Lake water that was released as CO_2

Table A.5. 6 One-way ANOVA (SOM as the factor) when treated with Okanagan Lake water

	Df	Sum Sq	Mean Sq	F value	Pr(>F)
Block	1	0.002949	0.0029490	2.112	0.180
SOM	1	0.000382	0.0003823	0.274	0.613
Residuals	9	0.012565	0.0013961		

Table A.5. 7 One-tailed t-tests performed for each of the soil types when treated with Okanagan lake water. Alternate hypothesis: true mean is greater than 0

Soil	Df	Mean of X	p-value
'high OM' soil (7.66% OM)	5	0.07823763	**0.0006309**
'low OM' soil (3.34% OM)	5	0.06694943	**0.008979**