A NORTHERN GREAT PLAINS SOIL RESPONSE TO A ONE-TIME COMPOST APPLICATION

by

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of the requirements for the degree

of

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in

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ABSTRACT

Many organic cropping systems depend on organic amendments for maintaining crop yields and soil fertility. Weed control is often achieved through tillage, which can degrade soil and lead to a loss of soil organic matter (SOM). The objective of this two-year study was to determine how wheat (Triticum aestivum L.) grain yield, protein concentration, above ground biomass (AGB), and indices of soil health including SOM, substrate-induced respiration (SIR), and field soil respiration (FSR) measurements would respond to compost applications in a Northern Great Plains dryland cropping system on Farnuf loam soils (fine-loamy, mixed, superactive, frigid Typic Argiustolls) near Geraldine, Montana. Cattle manure compost was applied once in spring of 2019 at 0 (Control), 14 (Low), 46 (Medium), and 137 (High) Mg dry weight ha^{-1} to a spring wheat and cover crop rotation. Compost did not significantly impact wheat yield, AGB, or SOM content ($p > .05$). Differences were detected for the Low and High compost treatments compared to the Control in overall SIR rates and concentrations but not during any of four measurement periods (0, 7, 24 and 96 hours). The highest (median ± interquartile range [IQR]) SIR concentrations occurred at 24 hours (Control [25 ± 2], Low [28 ± 15], Medium [23 ± 5], and High [28 ± 5] mg CO_2-C g^{-1} SOC). Cumulative FSR, however, was significantly higher (2.5 Mg C ha^{-1}) for soils receiving the Medium compost rate compared to all other treatments including the Control, which respired ~2 Mg C ha^{-1} over the four-month measurement period. Together, these results indicate that when cattle manure compost is applied to fine-textured dryland soils, wheat grain yields and protein concentration, as well as soil properties like SOM, do not appear to be enhanced in the short-term. These results also indicate that, one year after application, compost treatments may be detectable via soil health indices such as SIR and FSR, but these results can be inconsistent. As demand for organically grown, dryland crops increase, future research should clarify the most sensitive, robust, and cost-effective indices of soil health capable of guiding improved agricultural practices, including compost amendments.

CHAPTER ONE

A NORTHERN GREAT PLAINS SOIL RESPONSE TO A ONE TIME COMPOST APPLICATION

Introduction

Modern agriculture faces a number of intersecting threats. From soil degradation to climate change, nourishing today's world cannot compromise our ability to nourish tomorrow's world. Global food caloric demand could grow from ~12 to ~18 quadrillion kilocalories by 2050 (Tian, 2021). Against this backdrop, dryland grain farms are vulnerable to soil degradation and losses of soil organic carbon (SOC) through the widespread use of tillage fallow[1] (Jarecki, 2003; Kirschenmann, 2010; LaCanne, 2018; Sanderman, 2017). Soil organic carbon is a keystone soil element that increases soil aggregation (Jarecki, 2003), water retention (Khaleel, 1981; Whalen, 2003), nutrient retention and cycling (Duong, 2012; Stukenholtz, 2002), and biological diversity (Mazumder, 2021). These qualities impart benefits to plants such as disease resistance (Schrumpf, 2013) and increased plant productivity (Costa and Silva, 2017).

[1] Tillage fallow is a farm practice, most often used in arid and semiarid environments, that maintains bare land through tillage for 14- to 21-months. It is intended to reserve soil water for successive crops by reducing water losses that have been found to occur through plant evapotranspiration (Machado 2015).

Fallow and tillage practices have been found to release SOC to the atmosphere as carbon dioxide (CO_2) (Sanderman, 2017; Schrumpf, 2013) and lower soil water holding capacity (Machado, 2015; O'Dea, 2015). Organic cropping systems in the Northern Great Plains are most limited by water and soil nutrient availability (Lehnhoff, 2017). Applying compost has been found to increase SOC immediately (< 1 year) (Oldfield, 2017; Ryals, 2014), increase soil nutrient availability (Atoloye, 2021; Duong, 2012; Mazumder, 2021), and increase soil water holding capacity (Jarecki, 2003; Khaleel, 1981; Whalen, 2003). However, on-farm use of compost is rare, even in areas that have manure surpluses (Viaene et al., 2016). Purchasing, transportation, and short-term return on investment are barriers to use of compost inputs (Reeve, 2012). The inherent variability of soil (McConkey, 2020) and compost quality (Endelman, 2009), along with complex interactions with seasonal and interannual weather variability, make it difficult to predict the outcomes of compost applications in agronomic systems.

Beyond the physiochemical benefits that compost may provide, compost additions are generally expected to improve yields. Stukenholtz et al. (2002) found dryland winter wheat yields increased from 2 to 4 Mg ha^{-1} between the control (no treatment) and a one-time 75 Mg ha^{-1} dry mass DM[2] compost application during the first growing season . In Devon, UK, winter wheat grain yields increased from 5.7 to 6.7 Mg ha^{-1} after a 100 Mg ha^{-1} compost application

[2] All references to compost applications are in dry weight unless otherwise stated.

(Tilston, 2005). Yet, Calderon et al. (2018) saw no change in winter wheat grain yields following compost applications from 20 and 99 Mg ha^{-1} in eastern Colorado. Short-term yield responses aside, compost adds nutritive and non-nutritive benefits (Stukenholtz, 2002) that can persist in the soil for >16 years after one application (Mäder, 2007; Reeve, 2012).

The non-nutritive benefits of compost (e.g. increased water holding capacity, aggregation, microbial diversity, and functioning) could be especially valuable in reparation and maintenance of degraded farm soils (Oldfield, 2017). The nutrient concentrations in compost depend primarily on the feedstocks that were used and their time-release as dictated, in part, by the carbon to nitrogen ratio (C:N) of the compost and the soil, and soil moisture (Endelman, 2009). In semiarid systems, 0-20% of the N in compost can become available for plant uptake in the first year and 20-40% during the second year (Gale, 2006). Chelation and mineralization of nutrients from compost can reduce nutrient leaching and improve nutrient uptake in plants (Reeve, 2012; Tejada, 2003).

Depending on the recalcitrance (e.g., high C:N) level of the compost and SOC content, compost can either accelerate (positive priming) or attenuate (negative priming) microbial respiration and biomass turnover (Basile-Doelsch, 2020; Bastida, 2019; Klammer, 2006). Soil respiration (SR) is a function of microbial activity, plant root growth, and soil fauna; variation occurs across and within landscapes based on soil and plant type, precipitation, temperature, and land management techniques (Maier, 2011). SOC turnover rates typically are shorter in surface soil horizons that receive substantial amounts of exogenous carbon materials relative to deep soil horizons (Basile-Doelsch, 2020). Soil respiration measurements are well correlated with microbial activity, indicating the ability of a soil's microbial community to mineralize nutrients

within residues and amendments (Moebius-Clune, 2017; Sherrod, 2012; Zhong, 2016). Laboratory-induced respiration experiments can complement in situ respiration measurements and indicate the decomposition rate of C substrates in a climate-controlled environment (Schädel, 2020). The substrate induced respiration (SIR) method indexes the portion of the microbial community capable of rapid respiration with an easily degradable C substrate (Wardle, 1995). Soil in an incubation experiment has been disturbed, which can potentially release C from the protection of aggregates and increases its susceptibility for further microbial breakdown. For these reasons, SIR experiments may overestimate C respiration relative to in situ measurements (Saloma, 2010).

In situ measurements document the combined SR from living plant roots and soil fauna that are unique to the soil habitat and it's climactic variations (Chen, 2019; Schlesinger, 1977; Zhong, 2016). Soil respired CO_2 diffuses through biopores and soil cracks at various rates. Rates of diffusion are dictated by changes in volumetric pressure from rain, temperature, soil texture and density (Chen, 2019; Davidson, 1998; Raich, 2000). The spatial and temporal variability of SR flux can be measured with long-term soil respiration chambers. These measures can inform on how substrate additions and management techniques influence a soil's capacity to store and release C.

The soil habitat, abiotic factors, and compost quality determine how much of the compost C and nutrients becomes available to crops or becomes part of the SOM pool. Evaluating short-term metrics such as crop yield and quality, soil elemental content, and SR are highly variable, but continued research on the outcomes is important since dryland cropping systems are particularly vulnerable to C losses from tillage and fallow. The objective of this study was to

76 determine if compost applications would increase wheat yield and quality, soil elemental content
77 and soil respiration in a dryland organic cropping system. It was not expected that significant
78 changes in SOC would occur within the timeline of this study, as they can often take >10 years
79 to detect (Jarecki, 2003). The primary hypothesis was that compost would increase spring wheat
80 yield and SR relative to forgoing a compost application.

CHAPTER TWO

SPRING WHEAT YIELD AND SOIL ELEMENTAL CONTENT ONE YEAR AFTER COMPOST APPLICATION

Introduction

Applying compost to agricultural soils has been found to increase SOC immediately (< 1 year) (Oldfield, 2017; Ryals, 2014). However, compost applications are rare in arable systems, even in areas that have manure surpluses (Viaene et al., 2016). In certified organic production systems, tillage is the primary weed control practice (Gattinger, 2012). It releases soil organic carbon (SOC), contributed as atmospheric carbon dioxide (CO_2) (Sanderman, 2017). SOC is a keystone soil element that enhances resilience to climactic extremes by increasing soil aggregation (Jarecki, 2003), water retention (Khaleel, 1981; Whalen, 2003), nutrient retention and cycling (Duong, 2012; Stukenholtz, 2002), and biological diversity (Mazumder, 2021). These qualities impart benefits to plants such as disease resistance (Schrumpf, 2013) and increased plant productivity (Costa and Silva, 2017). Organic dryland cropping systems are most limited by water and nutrients and can benefit from increases in SOC (Lehnhoff, 2017). The results of using compost as a mitigation strategy for soil degradation in organic dryland systems are not always predicable because of the inherent variability of soil (McConkey, 2020), compost quality (Endelman, 2009), and their complex interactions.

Methods

Site Characteristics

A compost-response study was initiated on a United States Department of Agriculture (USDA) certified organic farm in the spring of 2019 near Square Butte, Montana (Lat. 47.523611, Long -109.984167, elev. 1,071-m). Prior to organic certification in 2018, the field was managed in a conventional, no-till, dryland winter wheat (Triticum aestivum L.) -fallow rotation where weeds were controlled with synthetic herbicides during a 14-month fallow phase. Beef cattle (Bos taurus) grazed wheat stubble post-harvest during the crop phase. Sainfoin (Onobrychis viciifolia Scop.) was grown in the field for three years prior to initiation of this study.

Thirty years of interpolated data for this area indicate that mean monthly temperatures range from -4.5 ˚C in January to 20.2 ˚C in July; mean annual precipitation is 382-mm, with ~45% occurring between May-July (PRISM Climate Group, 2020). During the experiment the annual precipitation was 456 mm in 2019, ~20% more than the corresponding 30-year average. In 2020, annual precipitation was 353-mm, which was ~ 8% lower than the 30-year average. Moisture during the SW growing phase (May -July) was 8% and 10% higher for both years than the average; 227 mm in 2019 and 232 mm in 2020. The mean air temperatures for May-July were 15.8 and 17.0 ˚C, respectively, about 7% lower than the corresponding 30-year average (Figure 1). The site's soils are mapped as the Farnuf series, or fine-loamy, mixed, superactive, frigid Typic Argiustolls (Soil Survey Staff, 1999). The 0-15-cm depth had a mean bulk density of 2 g cm^{-3}, pH of 6.2, and SOM of 5% when the baseline soil samples for the experiment site were combined (n =96, ± 1 standard deviation; Table 1).

Experimental Design

Compost treatments in the experiment were arranged in a randomized complete block design and replicated four times; individual treatment plots were 3-m x 6-m and were separated by 3-m wide buffer strips. A 12.5-m buffer was established around the outside perimeter of the experiment to reduce field-edge effects. The buffers and experiment plots were planted with the same plant species. One-time compost treatments were applied at four rates: 0 (Control), 14 (Low), 46 (Medium), and 137 (High) Mg ha^{-1} (Figure 2a).

The site was tilled to an 8-cm depth using 40-cm wide duck foot sweeps mounted onto curved shanks attached to a tool bar (i.e., field cultivator). Tillage occurred on November 2018, January and May 2019, primarily to control cheatgrass (Bromus tectorum L.) and to terminate the sainfoin prior to establishing the field experiment. Certified-organic Gunnison hard red spring wheat (SW; T. aestivum emend. Thell.) was planted across the field site at 80 kg ha^{-1} (16 pure live seed m^{-2}) at a 25.4-cm row-spacing to a 2.54-cm depth using a John Deere no-till disk drill. Compost application, tillage, and SW seeding occurred on 3 June, 2019. SW was harvested on 9 September, 2019 and immediately planted with a three-species cover crop polyculture, with equal proportions by weight (68 kg ha^{-1}) of winter triticale (Triticosecale), hairy vetch (Vicia villosa), and Austrian winter pea (Pisum sativum ssp arvense [L.] Poir.). On 22 May, 2020, the soil was tilled, and SW was planted at an 18-cm row spacing to a 2.54-cm depth using a custom-built plot seeder with disk openers. SW was harvested on 22 August in 2020. Weeds were hand-pulled and removed from the site opportunistically during each growing season.

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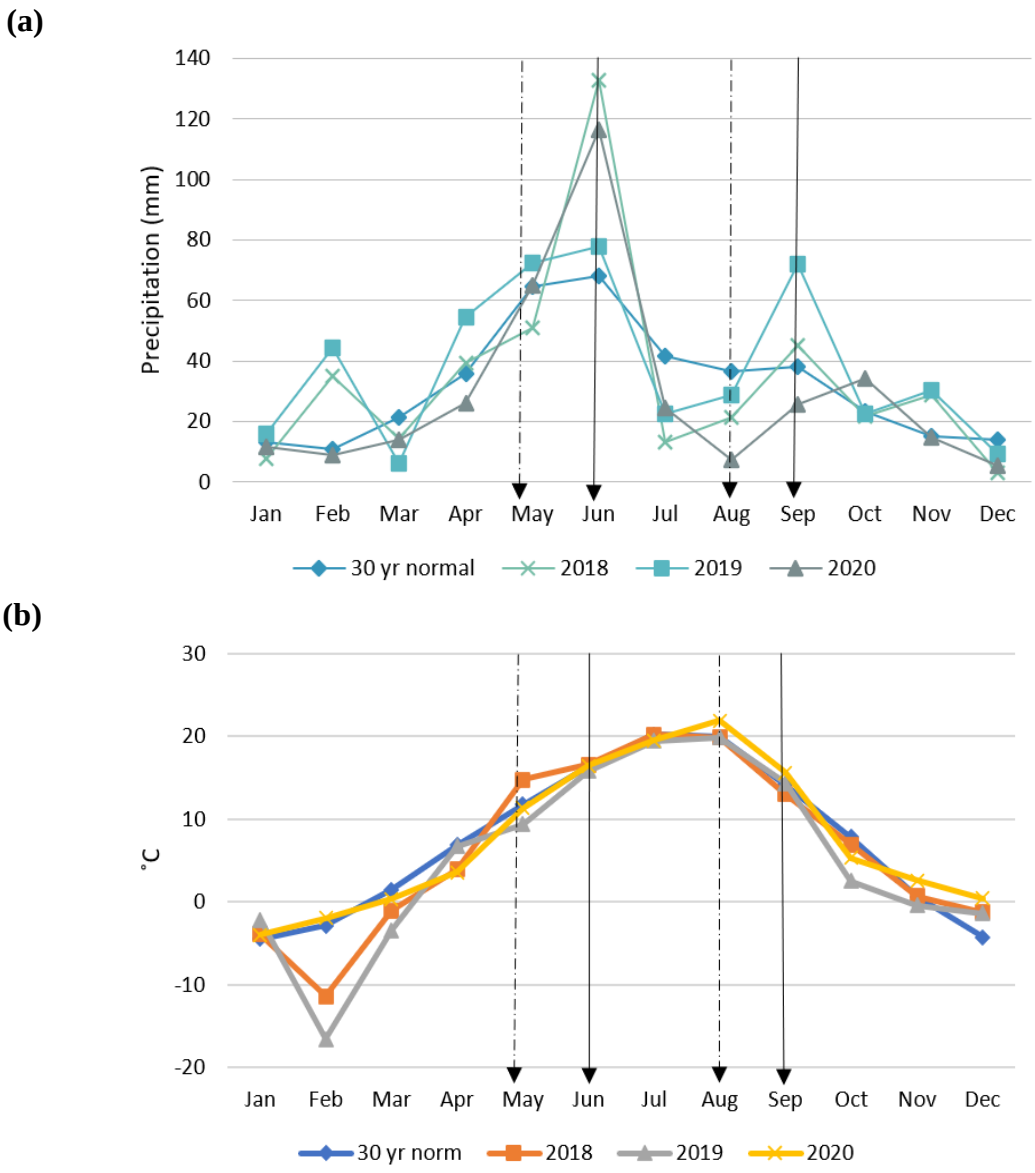


Figure 1. Mean monthly **(a)** precipitation and **(b)** air temperatures for the study site. The vertical lines represent planting and harvest dates in 2019 (solid lines), or 2020 (dashed lines) (PRISM Climate Group, 2020).

Table 1. The soil properties for the field experiment site. Sampling occurred in spring 2019 for the 0-15 and 15-30 cm soil depths (n=96, ± 1 standard deviation). Asterisks indicate water soluble elemental measurements.

		0-15 cm	15-30 cm
Property	**Unit**	**Mean ± SD**	**Mean ± SD**
Bulk density	g cm^3	2.15 ± 0.1	2.25 ± 0.1
pH	unitless	6.2 ± 0.4	6.3 ± 0.2
SOM	%	4.95 ± 1.1	3.5 ± 0.9
Total C	%	1.8 ± 0.6	0.84 ± 0.2
Total N	%	0.2 ± 0.1	0.10 ± 0.1
C:N	ratio	10.1 ± 0.8	8.4 ± 1.1
NH_4^+	mg kg^{-1}	4.8 ± 2.5	3.44 ± 0.9
NO_3^-	mg kg^{-1}	5.9 ± 3.2	6.31 ± 3.7
P*	mg kg^{-1}	56 ± 22	11 ± 9.7
K*	mg kg^{-1}	241 ± 85	85 ± 46

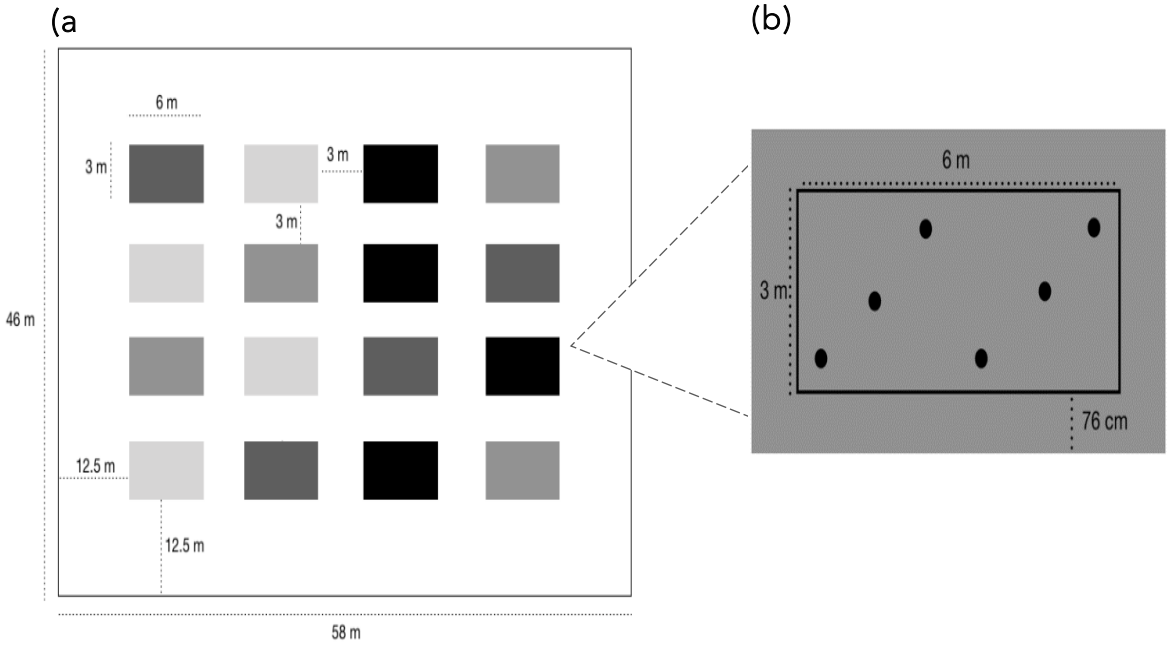


Figure 2. The field site randomized complete block design. Treatment rates (0, 14, 46, 137 Mg dry weight ha^{-1}) are represented by color gradient, black received the highest treatment rate, the lightest grey was the Control; **(b)** The soil sampling schematic. Each point represents a location where 2 soil samples were taken from 0-15 cm and composited, and 2 samples were taken from 15- 30 cm depths and composited (n=6 $plot^{-1}$ $depth^{-1}$).

Soil Sampling and Analysis

Soil samples were obtained from freshly plowed soil prior to compost applications in 2019 and prior to SW planting in 2020. Six georeferenced sample points were located 76-cm from the outer perimeter of each plot to reduce edge effects (Figure 2b). A 20-cm (diam) x 15-cm (deep) hole was dug with a close-backed Sharpshooter drain spade (manufacturer). The bulk soil was placed to the south of the hole. A 2.5-cm (diam) x 15-cm (length) soil probe was used to extract two samples from 0- 15 cm and 15- 30 cm depths from inside the hole. A ledge was cut from the side of the hole (~6-cm wide by 15-cm deep) to provide access to a non-compacted 15-30 cm depth for accessing the bulk density samples (Figure 3).

Each pair of soil samples was composited by sample point and depth into plastic bags, yielding n=6 $depth^{-1}$ $plot^{-1}$ for soil elemental analysis. Bulk density samples were taken in 2019 only, from the east and west halves of each plot for 0-15 and 15-30 cm depths (n=2 $depth^{-1}$ $plot^{-1}$). The soil probes and shovels were cleaned with water and dried with a towel after samples had been taken from each treatment plot. Baseline sampling (25 May, 2019) occurred during heavy rain and soils were near saturation in the north half of the plots. Concerns about sample integrity led to the suspension of sampling, which then resumed for the southern half of the plots the following day when saturated soils had drained for 24 hours. The reason soil sampling was not delayed was because sampling and compost application had to occur before spring wheat planting and the farm seeding equipment was going to move to a different location ~3 days from the soil sampling date. Soils were removed from the plastic bags, air dried in a shelter at ~21˚C for 62 hours, and stored in paper bags at 10 ˚C. In 2020, soils were sampled on May 8, 2020 and air-dried in a similar fashion as in 2019.

Bulk density was determined by dividing the weight of the oven dried soil by the volume of the soil core (76-cm^3) All soil samples were homogenized, sieved (2-mm) and then analyzed by Montana State University's Environmental Analytical Laboratory (EAL; Bozeman, Montana) in 2019 and by Ward Laboratory (Kearny, Nebraska) in 2020. The reason for changing the lab between years is associated with laboratory result turnaround time where the EAL took ~ 9 months to deliver results and Ward took 2 weeks. Discrepancies between laboratory procedures are present in this study for pH and loss-on-ignition (LOI %). Otherwise, the labs used the same analytical procedures for water-soluble phosphorus (P) and potassium (K, mg kg^{-1}; water, ICP), ammonium (NH_4^+) and nitrate (NO_3^-, mg kg^{-1}; 2M, KCL, Lachat), total nitrogen (TN) and total carbon (TC%, combustion, Costech). The EAL methods were: pH (1:2 soil:water ratio using a Mettler Toledo SevenEasy pH meter, Accumet 13-620-108A pH probe), and LOI (2 hours at 450˚C). The Ward Laboratory analytical procedures were: pH (1:1 soil:water ratio (Labfit, Dixie Scientific L.L.C.), and LOI (2.25 hours at 360˚C).The laboratory analytical procedures are in Appendix B.

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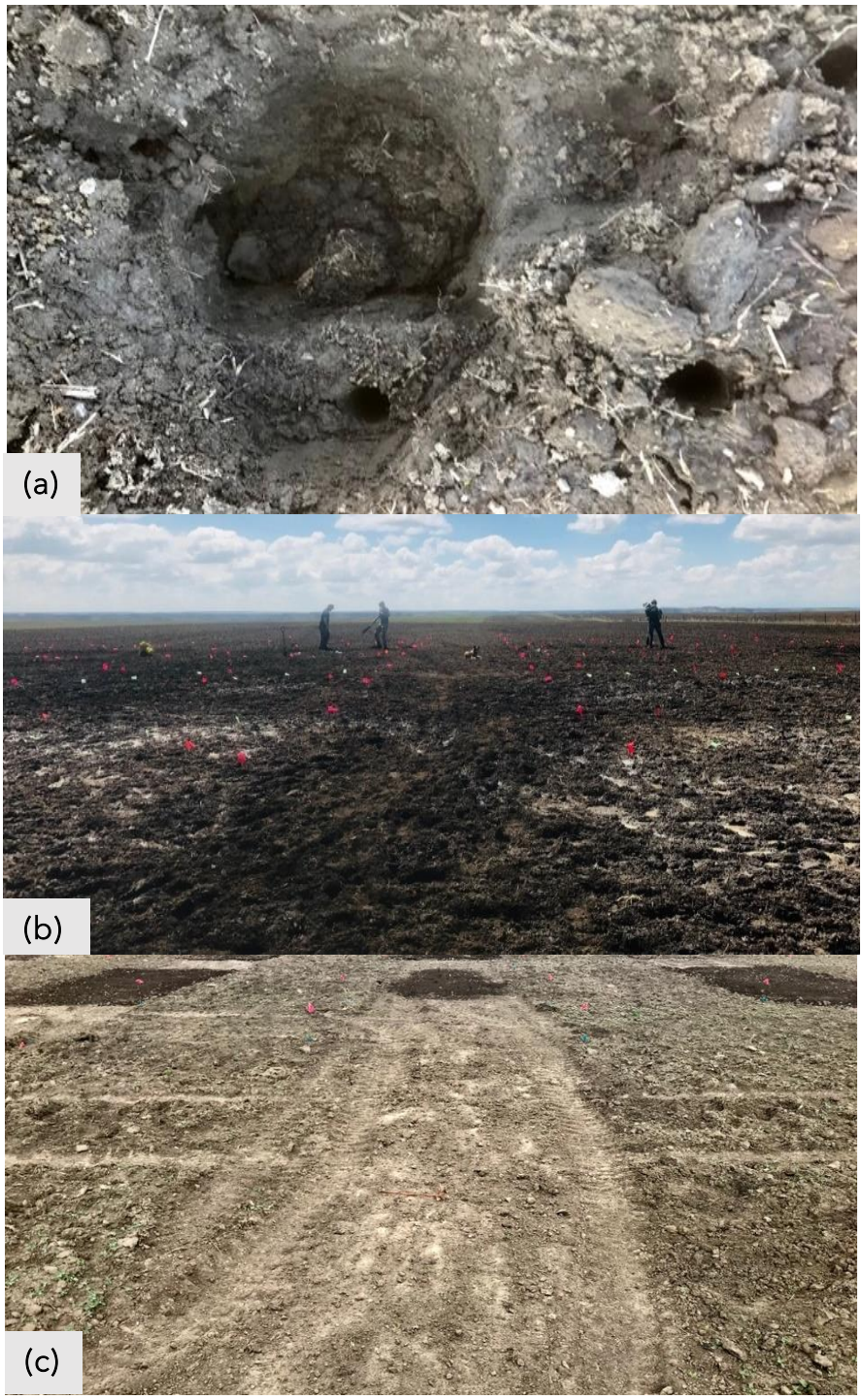


Figure 3. May 28, 2019 experiment field site **(a)** the soil sampling hole with ledge to extract bulk density sample; **(b)** experiment block saturated with water one day after baseline sampling was initiated, May 25, 2019; **(c)** graded soil surface made by a skid steer bucket for an evenly distributed compost application.

193 Compost Production and Application

194 The compost was prepared on-farm following USDA National Organic Program (NOP)
195 5021 compost regulations (USDA, 2011). Briefly, in July 2018, rough weight compost feedstock
196 proportions of 60% cattle manure, 20% hay, and 20% wheat straw were homogenized with a
197 compost windrow turner (Backhus 17 series, Germany). Moisture was maintained at ~50%
198 during the thermophilic phase (44-52°C) which ended 15 days after the initial turning and 4
199 weeks into the curing phase (< 54°C). The windrowed compost cured outside, uncovered,
200 through one Montana winter, and was applied 9 months after reaching ambient temperature.
201 Compost samples (n=3) were analyzed by Ward Laboratories in 2020 (Table 1).

202 A skid steer bucket was used to level the surface of the soil, which had 9- to 18-cm deep
203 imprints from the baseline soil sampling that occurred in soils that were at field capacity, for ease
204 of accomplishing an even compost application (Figure 3c). The treatments were spread by hand
205 for Low, and with a skid steer for Medium and High, application rates. A 16-tine welded gravel
206 rake (Groundwork),was used to evenly spread the treatments. The site was tilled to an 8-cm
207 depth using 40-cm wide duck foot sweeps mounted onto curved shanks attached to a tool bar
208 (i.e., field cultivator). Plowing displaced the compost up to ~1.7 m in the direction of plowing;
209 the highest concentrations of compost appeared to lie ~0.3 m outside the plot boundaries.
210 Similarly, unamended soils from outside the plots were brought into the plots via tillage. To
211 mitigate this potential confounding factor, the tillage direction was reversed in spring 2020.
212

Table 1. Properties of compost as analyzed and the equivalent amounts applied (Mg dry weight ha^{-1}) at three treatment rates (Low =14, Medium =46, High =137). The compost was made from 60% cattle manure, 20% straw, and 20% hay and was at ambient temperatures nine months prior to analysis; sample size n=3, ± 1 standard deviation (SD).

Parameter	Unit	Analytical Result	Quantity Applied by Treatment (kg ha^{-1})		
		Mean ± SD	Low	Medium	High
pH	unitless	8.9 ± <0.01	--	--	--
Moisture	%	36 ± 5	5.04	16.56	49.32
Dry matter	%	76 ± 5	10.58	34.78	103.57
C:N	ratio	12.4 ± 0.1	--	--	--
Total C	%	12.4 ± 2.0	1700.5	5587.5	16641.0
Total N	%	1.0 ± 0.2	140.1	460.3	1370.8
Organic N	%	0.9 ± 0.1	129.0	423.9	1262.4
Ammonium	%	0.1 ± 0.1	10.2	33.4	99.4
Nitrate	%	<0.01 ± <0.01	0.9	3.0	9.0
P_20_5	%	0.7 ± 0.1	95.3	313.2	932.7
K_20	%	1.5 ± 0.2	201.0	660.4	1966.9
Sulfur	%	0.2 ± 0.03	30.0	98.6	293.6
Calcium	%	1.7 ± 0.3	232.6	764.3	2276.3
Magnesium	%	0.7 ± 0.1	94.9	311.6	928.1
Sodium	%	0.1 ± 0.01	19.2	62.9	187.4
Zinc	ppm	124 ± 27	2	5	16
Iron	ppm	15329 ± 2249	212	698	2078
Manganese	ppm	497 ± 177	7	23	68
Copper	ppm	32 ± 4	1	2	7
Boron	ppm	25 ± 4	0	2	5
EC	dS/m	9.2 ± 0.4	--	--	--

Spring Wheat Harvest and Protein Concentration

Grain and total above-ground biomass (AGB) were hand harvested in 2019 from a 0.10-m^2 quadrat in six randomly selected areas in each plot. The cover crop did not grow to a harvestable height (< 3-cm) prior to 2020 spring wheat planting. In 2020, the grain was harvested

from a 9-m^{-2} area in the center of each plot with a plot combine. The yield in 2019 was considered "clean-weight" because wheat grain was hand harvested and cleaned which reduced the losses that tend to occur when harvest is performed with a combine. The yield in 2020 was considered "dirty-weight" because a plot combine was used because other materials can be brought into the machine that would contribute to the yield (e.g., insect parts, straw, weed seeds). Any differences between years could be due to a variety of factors, including differences in planting/harvesting methods. Thus, comparisons across years for yield were not made in this study. SW grain was stored in paper bags at 10 ˚C and 50% relative humidity, in cool dark conditions. Grain protein and moisture (%) was analyzed for both years at the Montana State University's Grain Quality Lab using near infrared transmittance **(**Montana State University, 2021**).**

Statistical Analysis

Yield, protein, and soil elemental content statistical analyses were performed using the R software package (version 4.0.5) (R Core Team, 2020). Models were designed to evaluate the statistically significant differences (SSD) of the overall effects conditional on the survey design. The model residuals versus fitted values and the normal Q-Q plot for the residuals were visually assessed for fit and normality. In the case of soil elemental content, residuals that violated the assumption of constant variance or had a non-normal distribution were transformed (natural log (log base e) transformation). C:N, NO_3^- and NH_4^+ were not improved by transformation and statistical analysis was performed on the untransformed data to maintain simplicity for the overall analysis. Linear mixed models were fit using the lmer function in R (Bates, 2015). Potential relationships between explanatory and response variables were assessed using Type II

Analysis of Variance with Satterthwaite's method (Kuznetsova, 2017). Models were refined using backwards step-wise regression, sequentially removing model terms with $p > .05$. Tukey's pairwise comparisons of treatment means were made with Kenward-Roger degrees-of-freedom method (family-wise confidence level 0.95) using least-squares means (Lenth, 2021). SSD between combinations of explanatory variables, corrected for multiple comparisons, were determined at $p < .05$.

Compost treatment at the plot level was a fixed effect in analysis of variance (ANOVA) model used for grain yield and protein concentration in 2019. Compost treatment blocks (4) and the number of plots nested within block (16) were considered random effects; there were up to 96 observations (Figure 4a). In 2020, compost treatment was the fixed effect and block the

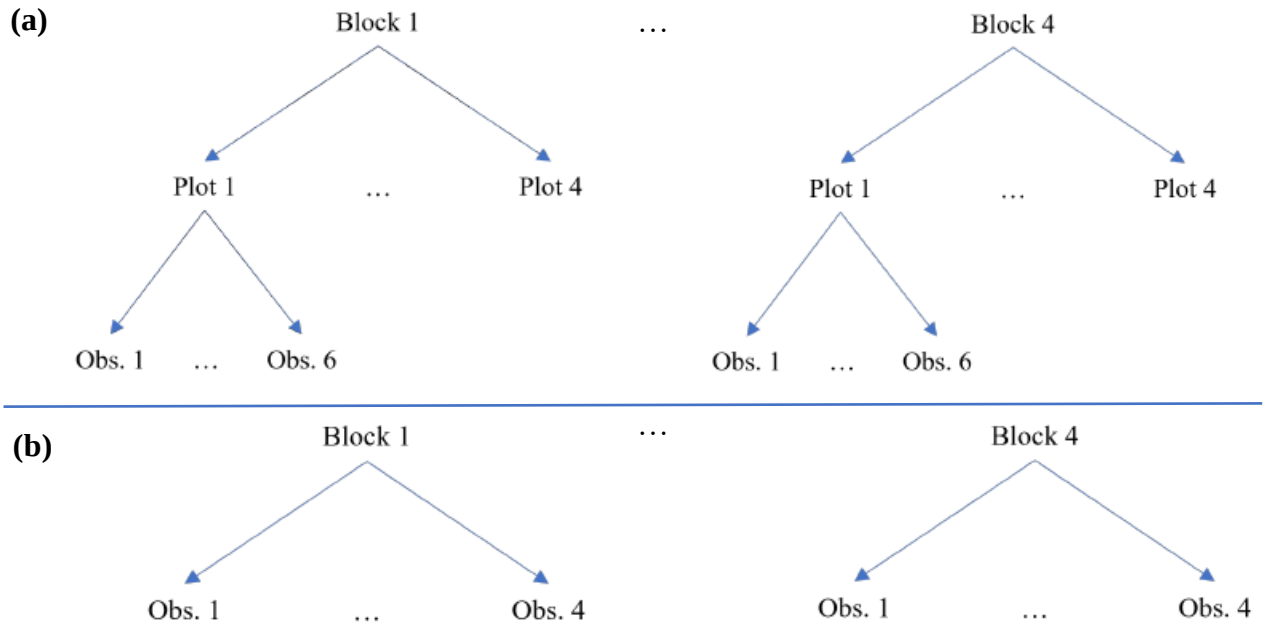


Figure 4. A depiction of the data structures used in the **(a)** 2019 spring wheat grain, above ground total biomass and protein concentration. The fixed effects were treatment at the plot level (6), and the random effects were treatment blocks (4 levels) and plot within treatment (16 levels) with an error of up to 96 observations; **(b)** 2020 spring wheat grain and protein content where treatment at the plot level was the fixed effect and the random effect was treatment block (4 levels) for a total of 16 observations.

random effect with 16 total observations (Figure 4b). Yields from each compost-amended plot were compared to yields from unamended plots for respective years. The raw data, conversions, and code for yield parameters are in Appendix A.

The soil elemental content fixed effects were: treatment at the plot level, year at the pre/post compost application level (year was aliased with the lab that analyzed the data because of is potential confounding factors), and depth (0-15, 15-30-cm). The random effects were plot (16 levels), location in plot (96 levels), and time in location in plot (192 levels), resulting in an error of 384 observations. A location point variable was created to identify the points within each plot to ensure that the levels of the categorical variables, which were used for random effects, uniquely identified the combination of the random effects. The location variable was made from the plot and point variables, to uniquely identify the sample source-location for each plot, resulting in 96 levels (16 plots x 6 sample points). The sample location and year variables were used to create a "time" random effect variable which identified the pre/post compost application status for each sampling source-location from each plot, resulting in 192 levels (2 years x 16 plots x 6 sample points in a plot). The soil elemental content was compared between treatments in each year and between years, after correcting for depth and the random effects structure. The raw data, conversions, and code for soil and compost elemental contents are in Appendix B.

The data discussed in this chapter had nonparametric distribution and so the median ± 1 median absolute value (MAD; Wilcox (2017)) are used to describe statistical values. Statistically significant differences (SSD) and the term "significant" indicate $p < .05$. The compost-treated plots will be referred to as +C, in these cases, Control is referred to as -C.

Results

Spring Wheat Yield and Protein Content

Compost did influence SW grain yields in either 2019 or 2020 (Table 2). However, a non-significant trend ($p = .06$) was detected in 2019. SW yields (median ± IQR) for -C plots were unexpectedly higher (6.7 ± 1 Mg ha^{-1}) than +C plots (Low [4.8 ± 3.8], Medium [3.3 ± 1.6], and High [3.3± 1.6] Mg ha^{-1}; Figure 5a). Above-ground biomass followed a similar trend ($p = .06$) where Control was the highest (19 ± 5 Mg ha^{-1}) and High was the lowest (12 ± 3 Mg ha^{-1}). In 2020, median grain yields across all treatments were ~1.1 ± 0.1 Mg ha^{-1} (Figure 5b). Grain protein concentrations in both years ranged between 12 ± 2% and 14 ± 3% with a moisture content of 7%; there were no SSD in either year .

Soil Properties

Compost additions did not affect any soil element in either year (Table 3). Overall, median (± IQR) elemental content in the 0-15 cm depth increased significantly between years for C:N (10.2 ± 1 to 10.7 ± 0.4), TN (0.17 ± 0.06 to 0.25 ± 0.08), NO_3^- (5.4 ± 3.9 to 12.0 ± 5.3), and water-soluble P (59 ± 35 to 129 ± 53). No significant changes were detected for SOM (~4.7 ± 1.2), TC (~2.3 ± 0.9), NH_4^+ (~4.6 ± 1.8), or water-soluble K (~94 ± 44) between years. The soil properties were significantly ($p < .001$) higher in the 0-15 cm depth in comparison to the 15- 30 cm depth, except for NO_3^-. Significant increases occurred between years in the 15-30 cm depth for C:N (8.4 ± 1.2 to 9.2 ± 0.5), TN (0.10 ± 0.02 to 0.12 ± 0.02), and water-soluble P (8 ± 8 to 26 ± 34) but SOM significantly decreased (3.7 ± 1.3 to 2.5 ± 0.2). No significant changes occurred for TC (~1.0 ± 0.2), NH_4^+ (~3.5 ± 1.3), NO_3^- (~5.3 ± 3.6) or water-soluble K (93 ± 95) in this lower depth.

Table 2. Model analysis for 2019 and 2020 spring wheat aboveground biomass (AGB), grain yields and protein content from spring applied cattle manure compost at rates of 0 (C), 14 (L), 46 (M), and 137 (H) Mg ha^{-1} dry weight. Tukey's pairwise comparisons, adjusted for multiple comparisons of treatment mean (n=4) made with Kenward-Roger degrees-of-freedom method (family-wise confidence level 0.95) using least-squares means was used to determine significance ($p < .05$) of compost treatments compared to C.

				Model Analysis		
Year	**Measure**	**Treat**	**Median (MAD)**	**F - value**	**DF**	**p value**
2019	**Grain (Mg ha-1)**	**C**	6.7 (1.0)	3.62	3, 9.00	0.06
		L	4.8 (3.8)			
		M	3.3 (1.6)			
		H	3.3 (1.6)			
	AGB (Mg ha-1)	**C**	19.0 (4.5)	3.63	3, 9.00	0.06
		L	16.0 (5.9)			
		M	12.0 (3.0)			
		H	12.0 (3.0)			
	Protein (%)	**C**	11.8 (1.6)	1.44	3, 8.28	0.30
		L	11.8 (1.7)			
		M	11.8 (1.1)			
		H	13.8 (1.4)			
2020	**Grain (Mg ha-1)**	**C**	1.1 (0.1)	0.25	3, 9.00	0.86
		L	1.1 (<0.01)			
		M	1.1 (0.1)			
		H	1.1 (0.1)			
	Protein (%)	**C**	13.8 (1.7)	0.17	3, 9.00	0.90
		L	13.2 (1.9)			
		M	12.5 (2.3)			
		H	12.5 (1.6)			

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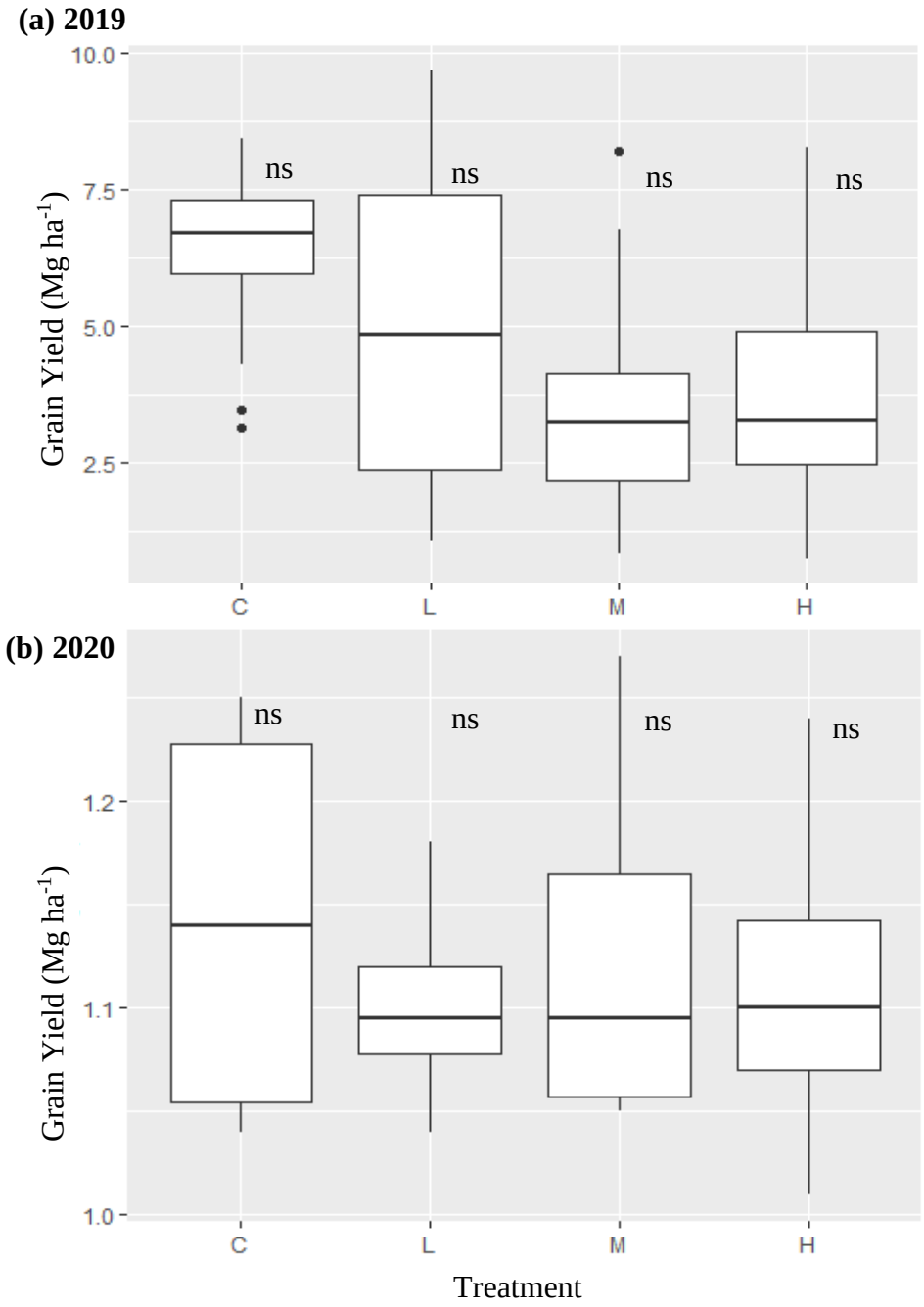


Figure 5. Spring wheat grain yield from **(a)** 2019 quadrat-harvest and **(b)** 2020 combine harvest which are not directly comparable to each other. Compost was applied in spring of 2019 at 0 (C), 14 (L), 46 (M), and 137 (H) Mg ha^{-1} dry weight. No significant (ns) differences were detected when compost applied treatments were compared to the Control. Statistical analysis used Tukey's pairwise comparisons of treatment mean (n=4) made with Kenward-Roger degrees-of-freedom method (family-wise confidence level 0.95) using least-squares means, significance was determined at $p < .05$.The horizontal lines indicate median values.

Table 3. Soil properties for the 0-15 and 15-30-cm depth pre (2019), and post (2020) treatment application of 0, 14, 46, 137 Mg ha^{-1} dry weight cattle manure compost. Results are the median of 96 samples ± median absolute deviation (MAD); significance ($p < .05$) is indicated by different letters next to property values as determined by Tukey test adjusted for multiple pairwise comparisons. Properties with an asterisk (*) were not log-transformed for statistical analysis.

		Content (Median (MAD))			
		0-15 cm		**15-30 cm**	
Soil Property	**Unit**	**2019**	**2020**	**2019**	**2020**
* C:N	ratio	10.16 (0.98) [a]	10.73 (0.44) [b]	8.36 (1.20) [c]	9.20 (0.52) [d]
Soil organic matter	%	4.78 (1.37) [a]	4.60 (1.10) [a]	3.66 (1.28) [c]	2.50 (0.23) [d]
Total carbon	%	1.73 (0.78) [a]	2.66 (0.91) [a]	0.79 (0.22) [b]	1.15 (0.23) [b]
Total nitrogen	%	0.17 (0.06) [a]	0.25 (0.08) [b]	0.10 (0.02) [c]	0.12 (0.02) [d]
* Ammonium	mg kg^{-1}	4.09 (1.55) [a]	5.21 (2.05) [a]	3.29 (0.72) [b]	3.73 (1.77) [b]
* Nitrate	mg kg^{-1}	5.40 (3.91) [a]	11.92 (5.28) [b]	5.40 (4.95) [a]	5.26 (2.30) [a]
Potassium (water-soluble)	mg kg^{-1}	242 (144) [a]	151 (124) [a]	70 (42) [b]	116 (105) [b]
Phosphorus (water-soluble)	mg kg^{-1}	59 (35) [a]	129 (53) [b]	8 (8) [c]	26 (34) [d]

Discussion

No significant changes were detected in SW yields, grain protein concentration, or soil elemental content as a result of compost applications in this experiment. Several factors could be responsible for this: (i) the compost had no effect on soil properties, (ii) the displacement of compost, from using farm sized equipment, may have diluted the soil in the plots, or (iii) changes in analytical laboratories between years. In some cases the variation between labs is not any greater than the inherent variability found in the soil (Ross, 2015).

Similarly, organic dryland winter wheat yields in Colorado were not improved after three compost applications of ~20 and ~100 Mg ha^{-1} over 5 years compared with forgoing compost (Calderon, 2018). Others have detected yield increases resulting from compost applications over the short- (<1 year) (Cherif, 2009; Miller, 2018; Oldfield, 2017; Stukenholtz, 2002) and long-term (>4 years) (Reeve, 2012; Tilston, 2005). Some of the most significant increases in yield have resulted from combining applications of synthetic N fertilizer and compost (Klammer, 2006; Tejada, 2003).

Keeling et al. (2003) found significant increases in grain yield relative to a standard fertilizer application as the control. In their study, winter wheat grain yields increased by 26% from an application of 20% mature (12 -month aged) greenwaste compost + 80% ammonium nitrate meeting 20% of calculated plant available N (PAN) requirements. When compost fulfilled 100% of calculated PAN requirements, however, yields decreased by 36% relative to the control. Keeling et al. (2003) compared the effects of compost maturity on plant growth and found that the 6-months cured compost, while not significantly different, had mean higher yields in

comparison to the 12-month cured compost. These effects did not hold true once the synthetic fertilizer was added to the compost, but this outcome points to the nuance of compost maturity and its effect on crop yields. The nutritive qualities that compost can impart to a soil/plant system are highly variable; they depend on abiotic factors such as temperature and moisture (Calderon, 2018), soil attributes such as bulk density, texture, and microbial communities (Duong, 2012), and compost quality (Gondek, 2020; Gravuer, 2019).

For both years of the Montana study, the mean grain protein content was ~13% across all treatments, which could indicate that yield and protein were compromised by under-fertilization (Jones, 2012). The C:N ratio can provide insight into PAN release from a compost. Roughly 5% of PAN is released during the first season following application when compost has a C:N ratio between 10-20, TN between 1-2%, and NO_3^- between 100-1,000 ppm, (Sullivan, 2018). The C:N ratio of the compost applied in this experiment was 12.5 and TN was 1%, but NO_3^- was only ~0.05 ppm, which implies there was not adequate N for increases in plant production. None of the N forms measured indicated that the compost supplied N to the soil because their increases occurred across all experiment plots. This finding may reflect a legacy of the N-fixing sainfoin that had been established and tilled into the soil prior to the initiation of this experiment.

Conclusion

Results of this two-year study did not indicate that there were changes in wheat yield, grain quality, or soil elemental content from compost applications in a dryland spring wheat system. This could be attributed to any number of factors: (i) the displacement of soil via using a skid steer to create an even surface for compost application, (ii) compost displacement that resulted from using farm-scale equipment;

(iii) the change in harvest methods and laboratories between years; and/or (iii) legacy effects from the sainfoin that had been established for three years prior to this experiment. While the benefits of adding compost do not appear to be immediate in the dryland setting selected for this experiment, it is also possible that any yield, quality, and/or soil elemental benefits associated with compost amendments might have simply required a longer duration study to be quantifiable.

CHAPTER THREE

SOIL RESPIRATION ONE YEAR AFTER COMPOST APPLICATION TO SEMIARID SOIL

Introduction

Soil respiration (SR) is an indicator of activity by soil microbes, plant roots, and soil fauna which vary across and within landscapes based on soil and plant type, precipitation, temperature, and land management techniques (Maier, 2011). SR correlates with microbial activity (Anderson, 1978; Beare, 1990), such as microbial mineralization of nutrients within residues and amendments (Sherrod, 2012). Supplying soil with exogenous carbon-(C) based substrates has been found to either increase (Beesley, 2014; Borken, 2002) or decrease (Zhong, 2016) SR. Detecting differences in respiration in situ can be challenging (Chen, 2019). Laboratory experiments can complement in situ measurements by quantifying the metabolically active component of the soil community and the decomposability of C substrates under conditions in which the climate, inputs, and disturbance are kept constant (Beare, 1990; Rui, 2016; Schädel, 2020). Laboratory experiments enable cross-comparisons of soils that have unique C sources and concentrations (Schädel, 2014).

Microbial communities in soil are present in various physiological states (e.g., active, dormant, replicating, desiccating, and so forth; Blagodatskaya (2013)). Moreover, the physiological state of a microbial community is not constant in space or time. Similarly, soil organic matter (SOM) is a catch-all term that belies a staggering diversity of chemical

compounds (Angst, 2021) in various stages of decomposition by different microbial communities (Schmidt, 2011). Short-duration incubation experiments lasting from 24 hours to a few months can inform our understanding of soil C dynamics (Schädel, 2020). Incubation studies that supply water only indicate the breakdown of inherent SOC (Schädel, 2014). Substrate-induced respiration (SIR) experiments supply a C substrate and measure the fraction of a microbial community that can quickly digest, and replicate from the added substrate (Fierer, 2003; O'Dea, 2015). When substrates, e.g., yeast or glucose, are added to soil and respired CO_2 is measured, microbial community biomass is estimated to be relatively proportional to the respired CO_2 (Anderson, 1978). The SIR method indexes the portion of the microbial community capable of rapid respiration with an easily degradable C substrate (Wardle, 1995). While compost in its entirety is not considered to be an easily decomposable substrate (Calderon, 2018), the C addition could alter the active microbial population (Bowles, 2014; Chang et al., 2007), which may be reflected by higher respiration in compost treated soils compared to the control. To some degree, all soils in an incubation experiment have been disturbed through soil sampling, root-picking, sieving, and re-wetting. These types of disturbances can potentially release C from the protection of aggregates and increase its susceptibility for further microbial breakdown. Because of these reasons, SIR experiments may overestimate C respiration relative to in situ measurements (Saloma, 2010).

The objectives of this short-term incubation study were to determine if (1) SR would increase because of compost applications (0 (Control), 14, (Low), 46 (Medium), 137 (High) Mg ha^{-1}) made the previous year, and (2) there were differences in SR across treatment application rates. We hypothesized that SIR treatments had higher respiration rates than the water only

treatment. If the field soil's microbial community and C food sources were altered by compost applications, it's expected that the differences would appear in both laboratory treated soils by higher respiration rates. Identifying a compost application rate that does not have different respiration rates compared to the Control would indicate a potential mitigation strategy for soil C losses that can occur through tillage production methods.

Methods

Experimental Design

The soil used in the SIR experiment was obtained from the field experiment at the Sandy Arrow Ranch (Lat. 47.523611, Long-109.984167, elev. 1,071-m) near Geraldine, Montana in spring 2020, one year following compost applications of 0 (Control), 14 (Low), 46 (Medium), and 137 (High) Mg dry weight ha^{-1}. Unless otherwise noted, all references to compost application rates and soil in this document will be in dry weight (dw). The soil had been tilled three days prior to soil sampling. Six georeferenced sample points were located 76 cm inside plot boundaries to reduce edge effects. A 2.5-cm diameter soil probe was used to pull two soil samples from each sample point from the 0-15 cm depth interval; each pair of two samples were composited into plastic bags until the end of the sampling day when they were transferred to paper bags (n=6 $plot^{-1}$). This approach yielded 96 samples, each ~150 grams. The soil samples were dried and stored for ~9 months at 18.3˚C and 50% relative humidity prior to this experiment. Two weeks before the SIR experiment, three soil samples were randomly selected from the six collected from each of the 16 field plots. Each sample was individually homogenized in a stainless-steel electric grain mill (C. Goldenwall, Taiwan), sieved (< 2 mm),

and further divided into six technical replicates, each ~25 grams. The mill was cleaned with isopropyl alcohol and air dried between samples.

The SIR method was modified from Fierer et al. (2003). Briefly, 8.6 g of oven dw equivalent soil was weighed into individual 50 ml centrifuge tubes, tamped to a uniform bulk density (~1 g cm^3), and adjusted to 25% water-filled pore space as part of the substrate addition. The water adjustment was determined via a separate mini-experiment. The water + substrate (WS) treatment received 1.5 ml of yeast extract solution (12 g autolyzed yeast in 1 L H_2O; Difco Laboratories, Detroit, MI); the water only treatment (W) received an equivalent volume of water (n=3 $plot^{-1}$ $treatment^{-1}$). The treatments and soil were homogenized with a sterilized, stainless-steel rod for 30 seconds. The tubes were sealed with gas-tight lids fitted with rubber septa and incubated at 21.5˚C. Headspace CO_2 concentrations were measured from each tube using an infrared gas analyzer (Licor 820, Lincoln, NE, USA) at Day 0, 0.3, 1, and 4 (D0, D0.03, D1, D4) by injecting 5 mL of laboratory air into the tube, extracting 5 mL of headspace air, and injecting the headspace sample into the gas analyzer injection port. After each measurement the rubber septa were removed for 15 minutes to supply fresh oxygen to the sample and replaced. Soil-less centrifuge tubes served as blank controls to determine the laboratory air and headspace CO_2 concentrations at each time point.

Peak CO_2 concentrations were recorded and converted to CO_2 concentrations with a 5-point standard curve for calibration (0; 500; 1000; 10,000; and 100,000 parts per million volume CO_2). Because most peak CO_2 readings were <10,000 ppm, the slope of the truncated four-point calibration was used: 76.4 (R^2=0.9999). We used the fixed soil weights and centrifuge tube volumes as well as variable SOC levels to convert headspace CO_2 concentrations (ppmV as µL

463 CO_2 per L) to two final unit systems: (i) concentrations as mass of carbon-equivalent CO_2-C on a
464 mass SOC basis (mg CO_2-C [g SOC]$^{-1}$) or (ii) rates as mass of CO_2-C per mass SOC basis per
465 unit day [mg CO_2-C [g SOC]$^{-1}$ d^{-1}). Briefly, we followed a 5-step process:

466 1. the standard temperature-sensitive conversion for ppmV CO_2 (volume:volume) to
467 mg CO_2 m^{-3} (mass:volume) is 1.7959 for CO_2 at 25°C (44/24.5; Kuo 2014:8),
468 2. the standard molar conversion of CO_2.to C is 0.2727 (12 g C mol^{-1} [44 g CO_2
469 mol^{-1}]$^{-1}$),
470 3. the m^3 to L conversion is 0.001,
471 4. the per volume (L)$^{-1}$ to per mass (g soil)$^{-1}$ conversion requires the quotient of the
472 centrifuge tube volume (0.05L) to oven dw of soil used (8.6 g; 0.0058), and
473 5. the conversion of per mass oven dw soil to per mass of SOC requires the fraction
474 SOC for each soil, which ranged from 1.1% (0.011) to 3.3% (0.033), averaging
475 (±1SD, n=48) 2.3±0.5% (0.023) so a multiplicand of ~1/0.023=43.4783

476

477 An example calculation follows for a headspace CO_2 measurement of 120,000 ppmV or 12%,
478 close to the actual average of 121,353 ppmV on D1 (n=288 for both water only and substrate
479 treatments combined):

480 $$120{,}000\ \text{ppmV} \cdot \frac{44\ \text{mg CO}_2}{24.5\ \text{m}^3} \cdot \frac{12\ \text{mg C}}{44\ \text{mg CO}_2} \cdot \frac{1\ \text{m}^3}{1000\ \text{L}} \cdot \frac{0.05\ \text{L}}{8.6\ \text{g soil}} \cdot \frac{1\ \text{g soil}}{0.023\ \text{g SOC}}$$

481 $$= \frac{14.9\ \text{mg CO}_2 - \text{C}}{\text{g SOC}}$$

482 Because some prior studies have reported SIR on a per dw mass soil basis, not per SOC
483 basis, dividing the concentration values we report by the step 5 multiplic and (e.g., 43.4783)

yields results on a per dw mass basis. In this example, 120,000 ppmV would yield a converted concentration value of 0.342 mg CO_2-C g^{-1} or 342 µg CO_2-C g^{-1}; since this ppmV was obtained after 1 day of incubation, we could also re-express this result, after subtracting the average Time 0 value, as 342-11=331 µg CO_2-C g^{-1} d^{-1} or 14 µg CO_2-C g^{-1} h^{-1}. Here, SIR results are expressed on a mass per mass SOC basis (mg CO_2-C g^{-1} SOC) or a mass per unit mass per unit day rate (mg CO_2-C g^{-1} SOC day^{-1}). Any C contribution that was made by the compost applications would have been in the individual sample results. Rates for D0.3, D1, and D4 were calculated by dividing CO_2-C concentration by the incubation period relative to a 24-hr cycle. For example, D0.3 values were divided by 0.3 and D4 values were divided by 4. The final values were then divided by initial SOC concentrations and expressed on a $gram^{-1}$ basis. Appendix C contains the raw data, conversions, and model results for the laboratory SR methods.

Statistical Analysis

A general linear model (GLM) that included Day, Field Treatment (FieldTreat: Control, Low, Medium, High), Laboratory Treatment (LabTreat: W, WS). The nested effects were plot identification within Field Treatment (Plot_ID:FieldTreat), and Field Treatment within Day (FieldTreat:Day). Plot_ID:FieldTreat was designed to evaluate the potential for within treatment variability between plots and FieldTreat:Day was designed to test the interactions between treatments within measurement periods (D0, D0.3, D1, D4). An additional GLM, separating W and WS, further examined the significant effects. Statistically significant differences (SSD) were determined at a $p < .05$ using the Type III Sum of Squares on the natural log-transformed (ln) means of replicated plot respiration values ($n = 9$ $plot^{-1}$). The mean of the plot replicates was applied towards the final mean treatment respiration values ($n=4$ $treatment^{-1}$) because evaluation

of statistical models with all data indicated a large variance component from the within-plot replicates for the laboratory treatments (W, WS). The potential for average differences were determined by linear contrasts comparing compost applied field treatments (+C) to the Control (-C). In addition, a Duncan's Multiple Range Test (MRT, significance at alpha = 0.05) was applied to ln terms for W and WS to compare across and between Day respiration measurements.

Results

SIR concentrations were significantly greater for Low ($F^1 = 10.19$, $p < 0.01$) and High ($F^1 = 5.11$, $p = .03$) compost treatments compared to Control in the overall experiment, however, no treatment differences were detected during any of four measurement periods (D0, D0.3, D1, D4; Table 4). Maximum headspace concentrations occurred on D1 for WS- and W-treated soils (Figure 6). All measurement periods were significantly different for WS, but D1 and D4 were not different from each other in W-treated soils (Figure 7). Median (± IQR) WS headspace concentrations were ~ 400% higher and had greater variability than for W. For example, Control, Low, Medium, and High WS concentrations on D1 were 21±7, 25±13, 26±16, and 26±8 mg CO_2-C g^{-1} SOC, respectively; corresponding W concentrations were 4±3, 6±3, 5±2, and 6±3 mg CO_2-C g^{-1} SOC, respectively (Table 5). The substrate-induced and temporal differences were far greater than those associated with compost treatments. This interpretation is confirmed by the F values for Day ($F^3 = 415.08$) and Lab Treatment (W vs WS; $F^1 = 82.88$), which were relatively larger than that of Field Treatment ($F^3 = 3.18$; Table 4). Both W and WS, respectively, had significant interactions for field plot nested within field treatment ("Plot_ID:FieldTreat" [W] $F^{12} = 2.04$, $p = .05$; [WS] $F^{12} = 2.25$, $p = .03$). "Plot_ID" as a non-nested effect was not significant for either W or WS (Figure 8).

The model output for respiration rates (not shown) was the same as the concentration model output, the difference being the Type III SS values, thus, the respiration rates followed similar trends as concentration. Compost levels did not significantly change the mg CO_2-C g^{-1} SOC day^{-1} respiration rates when soils were amended with W, but they did increase respiration when soils were amended with WS, though not within any of the individual three measurement

periods (D0.3, D1, D4; Table 4). Again, the linear contrasts indicated that when +C treatments were compared to Control, Low (F^1 = 10.19, p = .003) and High (F^1 = 5.11, p = .03) respiration rates were significantly higher but Medium (F^1 = 2.19, p = .15) were not. Respiration rates were highest for WS on D1 but occurred on D0.3 for W. Median (±IQR) control, low, medium, and high rates for WS were 21 ± 2, 28 ± 15, 23 ± 5, and 28 ± 5 mg CO_2-C g^{-1} SOC day^{-1}, respectively (Table 5). In contrast to the greatest variability occurring for Medium headspace concentrations, Low had the greatest variability in respiration rates. The MRT indicated that D0.3 and D1 rates were not significantly different from each other but they were higher than D4 for W and that all measurement periods were different in WS with D1 having the highest respiration rate and D4 having the lowest respiration rate (Figure 9).

Table 4. Model statistics and linear contrasts for respired mg CO_2-C g^{-1} SOC day^{-1} from the 0-15 cm depth, 1-year post field treatment application. Field treatments, in Mg ha^{-1} dry weight compost rates were: Control (C) = 0, Low (L) = 14, Medium (M) = 46, and High (H) = 137. Statistically significant differences ($p < .05$) between treatments were determined by ANOVA Type III SS and linear contrasts. A natural log transformation was applied to mean field treatment (n=4) over the mean of plot replicates (n = 9)

Model I (mg CO_2-C g^{-1} SOC day^{-1})

Effect	DF	Type III SS	Mean Square	F Value	Pr > F
Lab Treatment	1	15.77	15.77	82.88	<.0001
FieldTreat	3	1.82	0.61	3.18	0.03
Day	3	237.00	79.00	415.08	<.0001
Plot_ID:FieldTreat	12	5.08	0.42	2.22	0.02
FieldTreat:Day	9	0.61	0.07	0.35	0.95

Model II

	Water (W)					Water + Substrate (WS)				
Effect	**DF**	**Type III SS**	**Mean Square**	**F Value**	**Pr > F**	**DF**	**Type III SS**	**Mean Square**	**F Value**	**Pr > F**
FieldTreat	3	0.74	0.25	1.73	0.18	3	1.16	0.39	3.65	0.02
Day	3	82.31	27.44	192.39	<.0001	3	163.06	54.35	511.83	<.0001
Plot_ID	3	0.74	0.25	0.13	0.94	3	0.86	0.29	0.18	0.91
Plot_ID:FieldTreat	12	3.49	0.29	2.04	0.05	12	2.87	0.24	2.25	0.03
FieldTreat:Day	9	0.60	0.07	0.46	0.89	9	0.16	0.02	0.17	1.00

Linear Contrasts

	Water (W)					Water + Substrate (WS)				
Contrast	**DF**	**Contrast SS**	**Mean Square**	**F Value**	**Pr > F**	**DF**	**Contrast SS**	**Mean Square**	**F Value**	**Pr > F**
C vs L	1	0.52	0.52	3.66	0.06	1	1.08	1.08	10.19	0.00
C vs M	1	0.01	0.01	0.07	0.79	1	0.23	0.23	2.19	0.15
C vs H	1	0.32	0.32	2.27	0.14	1	0.54	0.54	5.11	0.03

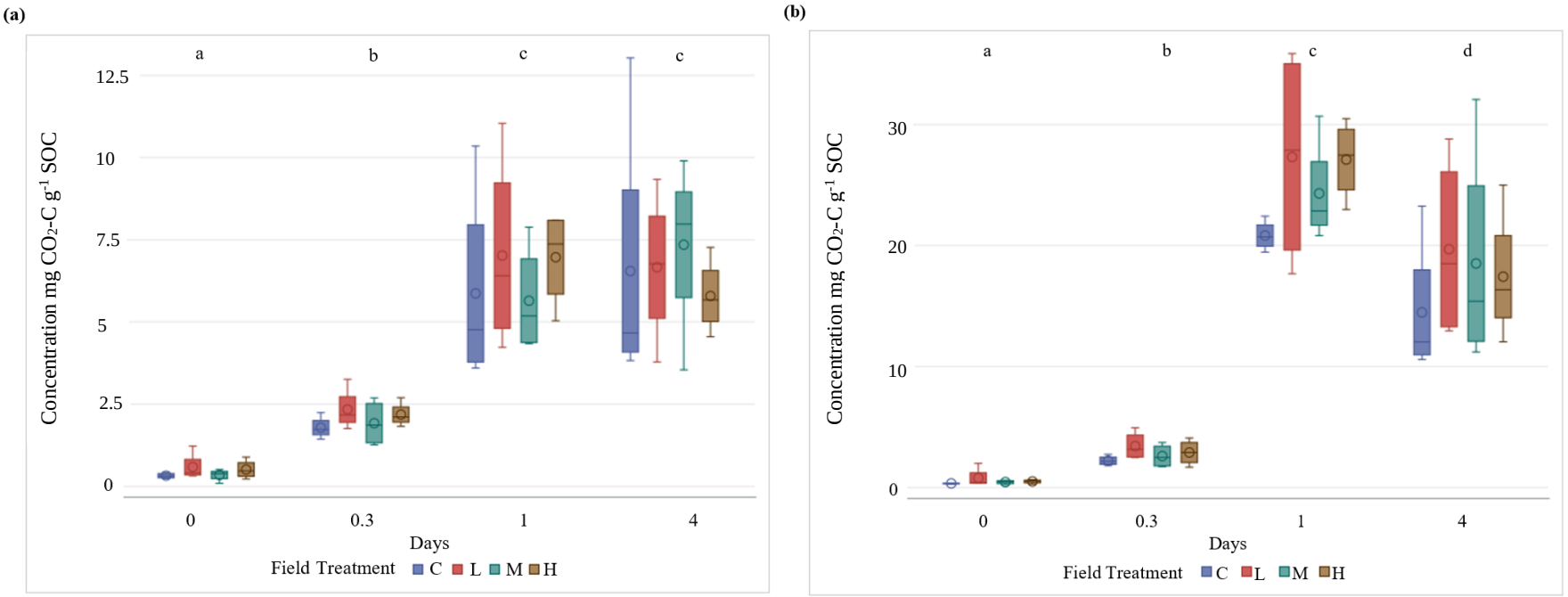


Figure 6. Box-and-whisker plots of **(a)** Water and **(b)** water + substrate CO_2-C respiration concentrations expressed per gram of SOC and measured at 4 time points (Day). Note the ~3X difference in y maxima between plots. Field treatment rates, in Mg ha^{-1} dry weight, were: Control (C) = 0, Low (L) = 14, Medium (M) = 46, and High (H) = 137. Different letters indicate statistically significant differences (SSD) ($p < .05$; ANOVA Type III SS, linear contrasts) between incubation duration (n = 12 Day^{-1}) when the model controlled for field treatments nested withing Day ($p < .0001$). No SSD between field treatments were detected within any given Day, and so are not indicated. Circles indicate treatment mean, and horizontal lines inside boxes indicate treatment median values (n=4). The y-axis between charts **(a)** and **(b)** are distinct.

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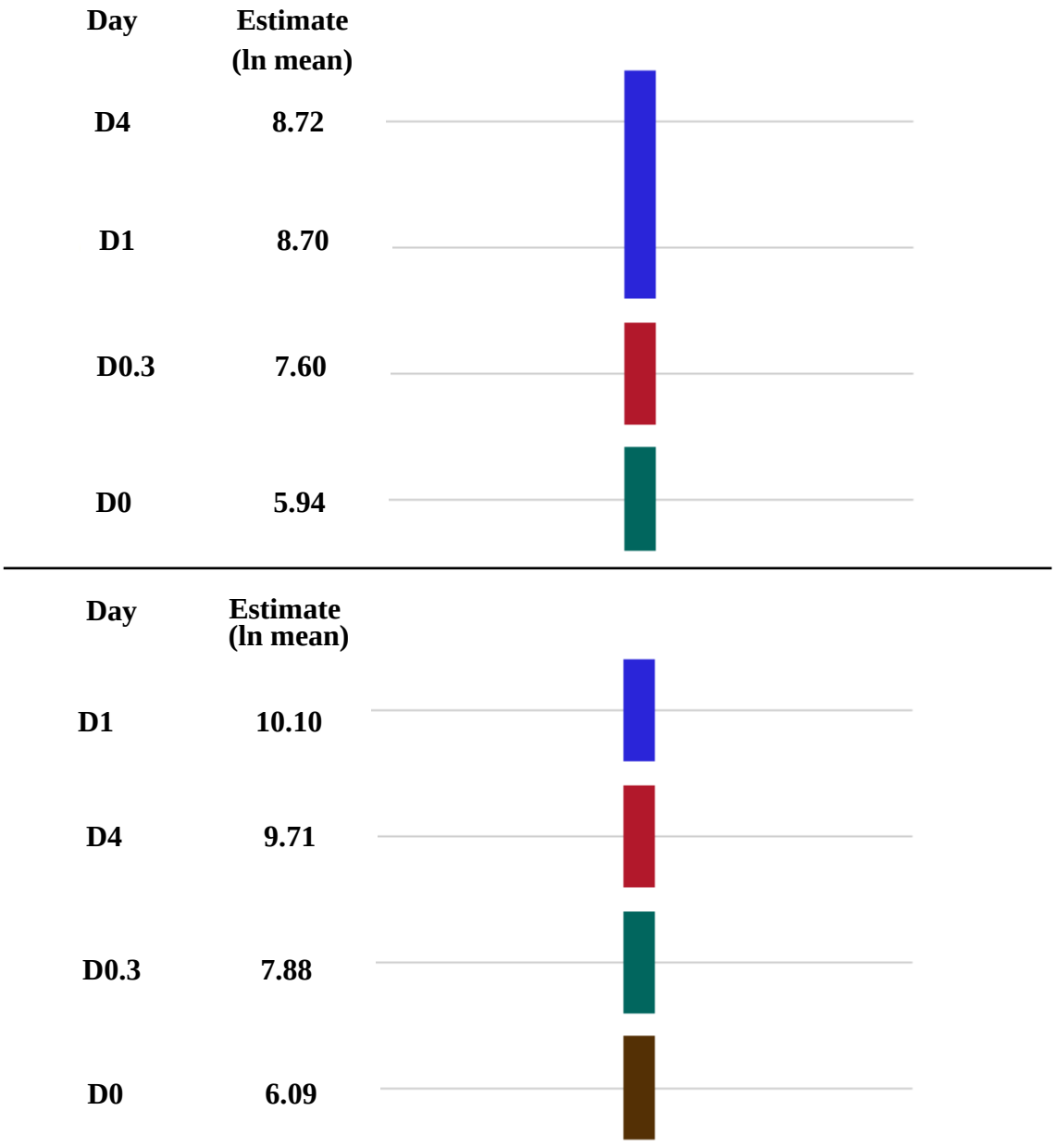


Figure 7. The Duncan's Multiple Range Test comparing the natural log-transformed (ln) means of mg CO_2-C g^{-1} SOC concentrations between measurement periods for **(a)** water and **(b)** water + substrate treated soils. Means covered by the same bar are not significantly different (alpha = 0.05, Degrees of Freedom = 36).

Table 5. Median and IQR for CO_2-C concentration and rates from water and water + substrate soils amended with compost (Field Treat) one year prior to analysis. Field treatments, in Mg ha^{-1} dry weight, were: Control (C) = 0, Low (L) = 14, Medium (M) = 46, and High (H) = 137.

		Median ± IQR			
		Concentration mg CO_2-C g^{-1} SOC		**Rate mg CO_2-C g^{-1} SOC day^{-1}**	
Field Treat	**Day**	**Water (W)**	**Water + Substrate (WS)**	**Water (W)**	**Water + Substrate (WS)**
C	D0	0.3 ± 0.3	0.4 ± 0.2	--	--
	D0.3	2 ± 0.4	2 ± 0.4	6 ± 2	7 ± 2
	D1	4 ± 3	21 ± 7	5 ± 2	21 ± 2
	D4	5 ± 4	13 ± 7	1 ± 1	3 ± 2
L	D0	0.4 ± 0.4	0.4 ± 0.7	--	
	D0.3	2 ± 1	3 ± 2	8 ± 7	11 ± 7
	D1	6 ± 3	25 ± 13	6 ± 4	28 ± 15
	D4	5 ± 4	16 ± 14	2 ± 0.8	5 ± 3
M	D0	0.4 ± 0.3	0.5 ± 0.3	--	--
	D0.3	2 ± 1	2 ± 2	6 ± 4.1	9 ± 6
	D1	5 ± 2	25 ± 16	5 ± 2.5	23 ± 5
	D4	6 ± 3	16 ± 14	2 ± 0.8	4 ± 3
H	D0	0.4 ± 0.4	0.5 ± 0.4	--	--
	D0.3	2 ± 0.6	3 ± 2	7 ± 2	10 ± 6
	D1	6 ± 3	26 ± 8	7 ± 2	28 ± 5
	D4	5 ± 4	17 ± 13	1 ± 0.4	4 ± 2

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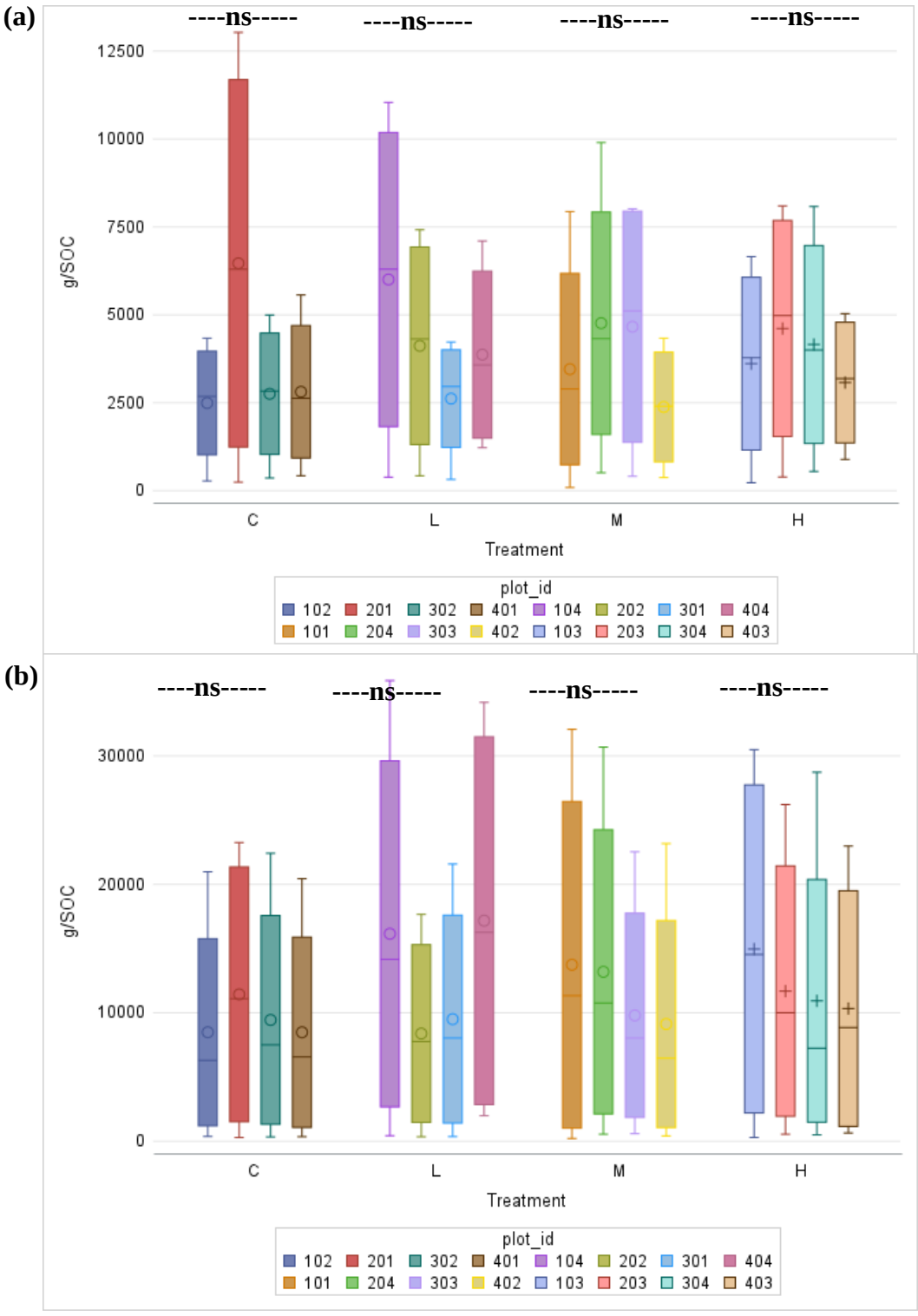


Figure 8. Within-plot treatment variability for **(a)** water and **(b)** water + substrate laboratory induced respiration concentrations from soil one-year after compost applications (Control (C) = 0, Low (L) = 14, Medium (M) = 46, and High (H) = 137). There were no significant (ns) differences indicated within treatments (ANOVA Type III SS, n=9 plot^{-1}).

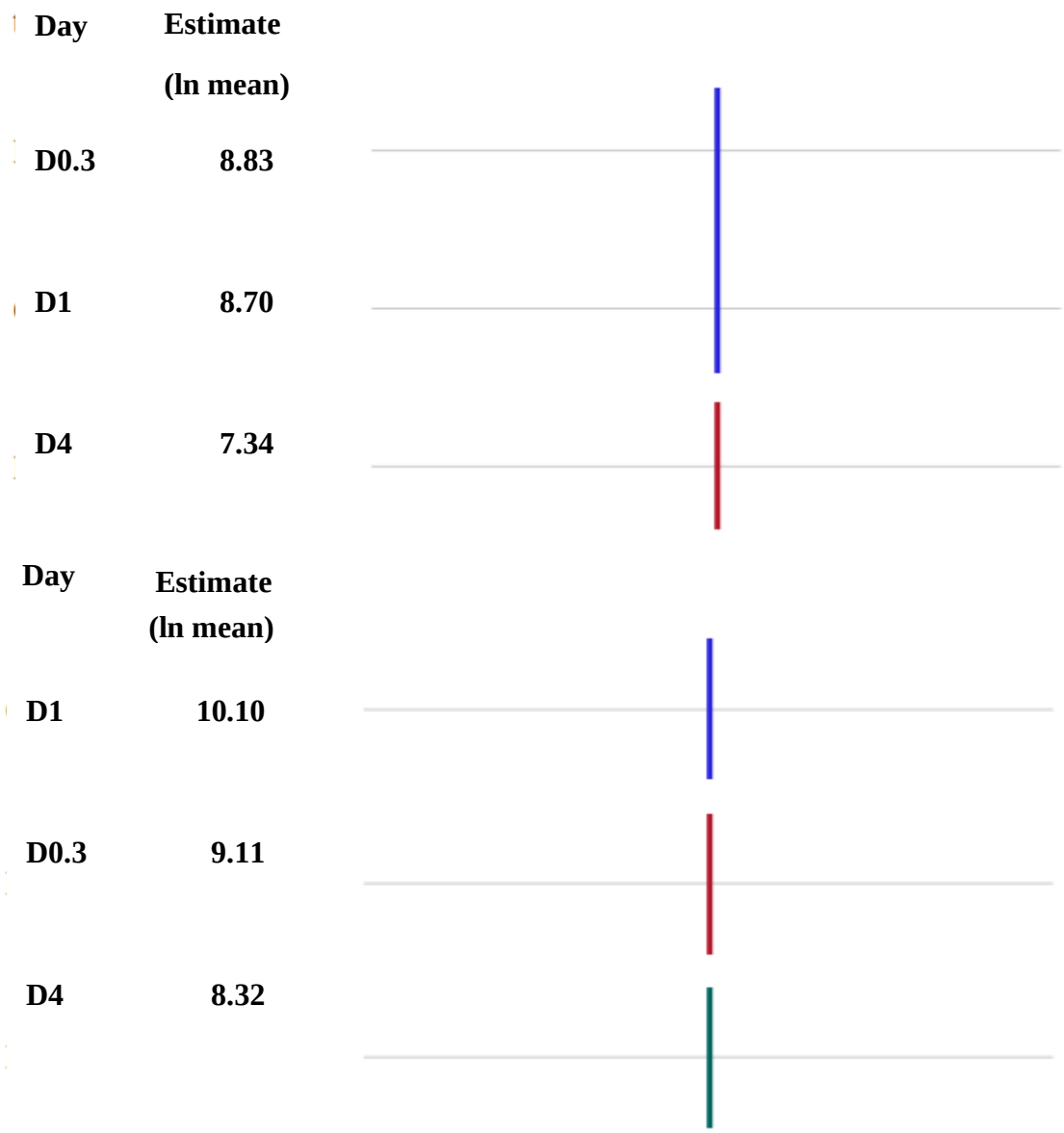


Figure 9. The Duncan's Multiple Range Test comparing the natural log-transformed (ln) means of mg CO_2-C g^{-1} SOC day^{-1} rates between measurement periods for **(a)** water and **(b)** water + substrate treated soils. Means covered by the same bar are not significantly different (alpha = 0.05, Degrees of Freedom = 36).

Discussion

The findings in this experiment indicate an SIR treatment effect from compost one-year after application. The model indicated significantly higher SIR concentrations and rates for Low and High compared to -C, but not for Medium. There is no biological explanation for why Medium was not different from -C; the variability in CO_2-C concentration is the most likely explanation for the nonsignificant results. Similarly, +C soils had greater variability than -C soils which could explain why the rate of application appeared to be significant but only when the model controlled for incubation period ("Day"). For soils that received water only, no significant treatment differences were detected.

These results are different from other incubation studies that have found increases in SR relative to exogenous substrate inputs. In a water incubation experiment, Chalwe et al. (2019), plowed in (10-cm depth) various rates of cattle manure compost to a peanut (Arachis hypogaea L.) cropping soil (chromic luvisol, pH = 4.22, sandy loam, 0.7% SOM). They found a linear relationship between water induced respiration relative to the amount of compost applied (0, 12, 35 Mg ha^{-1}). One hundred days after the compost was applied, they measured SR rates of 31, 45, and 66 mg CO_2-C g^{-1} SOC day^{-1}. The respiration rates reported by Chalwe et al. (2019) were ~45% higher than those measured in the present study. Chalwe et al. (2019) attributed the increase in respiration to the fresh C substrate of the compost, its inherent microbial population, and the combined effects integrating with the native soil community. In the present study, no SSD occurred in W +C treatments relative to -C, which may be due to the lapse of time that occurred (one-year) between compost application and SR measurements. During this time, the labile C that may have been detectable via SR could have been mineralized.

The decomposition of SOM occurs in stages and at differing rates, depending on its chemical make up, the actively digesting microbial population, and the soil habitat. In a microcosm experiment that used greenwaste and spent mushroom compost, Pérez-Piqueres et al. (2006) found that a greenwaste compost and water incubated soil had significantly higher respiration compared to the control (no treatment) in a sandy loam soil but had no effect in a clay soil. The spent mushroom compost used in their work elicited SSD responses from both soils. They attributed the different results to the relative lability of the mushroom compost as compared to the relative recalcitrance of the greenwaste compost. Their work highlights that clay soils may not respond to compost additions in the same way as lighter-textured, sandier soils. The soil in the present study was a silty clay (>40% clay) with a relatively high bulk density (~2 g cm^3), characteristics which are known to reduce soil arability and could impact SR.

SIR rates depend on the shifting microbial community as well as ever-changing oxygen levels, matric potentials, and dynamics of the original SOM pool, all of which, in turn, can slow or prime mineralization losses of SOM (Kuzyakov, 2006). Detecting a soil priming dynamic was beyond the scope of this study but respiration rates obtained following substrate additions are not dependent on the substrate alone. There are several factors that could result in significant increases in respiration from +C in WS and but have no +C effect in the W soils. The compost could have altered a C-responsive soil microbial community, one that would not respond to a water addition due to a shortage of available C substrates in the inherent soil but would respond to a WS addition. While SR increased in this experiment by ~400% when substrates were added, the increase was not necessarily meaningful because of the variability within treatments. Furthermore, the -C soils had relatively less variability than the +C soils. This could be due to

distribution of compost across and within the soils. As mentioned in Chapter 2, a smearing effect from plowing may have displaced the compost up to ~1.7 m in the direction of plowing; the highest concentrations of compost appeared to lie ~0.3 m outside the plot boundaries. The particle size of the compost would also relate to the +C soil variability because nutrient accessibility would depend on the surface area available for microbial breakdown. Laboratory procedures such as inadequate mixing of soil and substrate can increase variability in SR (Birge, 2013).

Conclusion

The present study indicates that compost applications made to a dryland SW setting are likely to have highly variable SIR concentrations and rates one year after compost applications. This variability can make it difficult to detect differences between treatments. This work confirms that SR increases significantly when fresh substrates, such as yeast, are added but that respiration may not directly correlate with more complex substrate additions such as compost. The persistence of C in any soil system is not entirely dependent on the intrinsic qualities of the C but also on the biological and physiochemical influences in the soil habitat, both of which can ultimately accelerate the loss of or protect C from degradation (Liang, 2017; Schmidt, 2011). The balance of SOC is influenced by management practices, its losses are tied to soil disturbance (Sanderman, 2017) and a loss of productivity (LaCanne, 2018). While C inputs can increase SR for a period, even one- year after application, in general the long-term benefits are considered to make up for the short-term losses (Borken, 2002; Sherrod, 2005).

CHAPTER FOUR

SOIL RESPIRATION ONE-YEAR AFTER COMPOST APPLICATION TO SEMIARID SOIL

Introduction

Soil represents the largest surficial terrestrial carbon (C) reservoir (~ 2.5 billion Mg to 1-m depth globally; Scharlemann (2014)). The top meter of soils contains approximately four times as much C as all the world's plants and animals and approximately three times as much as our atmosphere, even with annual increases due to fossil fuel emissions (Schumana, 2002). While these comparisons imply an impressive storage capacity, no form of soil C is ever permanently sequestered: soils release C, and "if this were not so, any completely resistant fraction would by now cover the surface of the earth (Hayes, 2018). Soils storing more C than they release are referred to as "C sinks"; conversely, soils releasing more C than they store are referred to as "C sources." With global warming, there is a risk that warmer and drier conditions could lead to a positive feedback, resulting in an amplification of soil C losses, which would feed back to even warmer and drier conditions (IPCC., 2021).

Lal (1998) estimated that ~ 50-75% of inherent SOC has been lost from tillage, fallow, and residue burning in agricultural fields . Just as past management has led to losses of previously stored soil C (Sanderman, 2017), future management that shifts the balance from C

sources to C sinks can have significant global impacts, even if small changes (i.e., 0.5- 1%) were consistent across global agricultural landscapes which represent ~5 billion ha (FAO, 2020).

As discussed in Chapter 1, applications of compost to agricultural lands have been shown to increase soil C concentrations and provide associated benefits (Calderon, 2018; Klammer, 2006; Reeve, 2012; Ryals, 2013), such as increased water holding capacity (Whalen, 2003), nutrient cycling (Mazumder, 2021), and disease resistance (Tilston, 2005). However, applications of exogenous substrates often increase soil respiration (SR), which releases soil C as CO_2 into the atmosphere (Beesley, 2014; Borken, 2002; Chalwe, 2019; Fontaine, 2007).

Soil respiration is the second largest terrestrial C flux (60 Gt $year^{-1}$) after oceanic respiration (90 Gt $year^{-1}$; (Kuzyakov, 2006; Washington State University, 2013). Approximately 10% of atmospheric CO_2 will cycle through soil C pathways annually, which is ~10 times more than the CO_2 released via fossil fuel combustion (Raich, 2000). In situ SR comes from living plant roots and soil fauna (Chen, 2019; Schlesinger, 1977; Zhong, 2016). Its movement through the soil is significantly impacted by soil type, density, moisture, and temperature (Chen, 2019; Davidson, 1998; Raich, 2000). SR flux varies substantially in both space and time and this variability can be measured in situ using SR chambers multiplexed to an infrared gas analyzer.

There is a dearth of studies available that have measured in situ SR following compost applications in dryland cropping systems. For those that do exist, most measurements occur shortly after compost application. The goal of this work was to determine if the addition of C would illicit SR with detectable differences relative to the control by measuring SR one-year following compost applications. Based on the findings of others (Borken, 2002; Chen, 2019; Fabrizio, 2009; Iovieno et al., 2009), it was expected that SR (μmol CO_2 m^{-2} s^{-1}) would increase

with higher compost rates and that cumulative growing season flux (Mg C ha^{-1}) would follow the same trend.

Methods

Experimental Design

The site description, field preparation, compost production details, and compost application rates can be found in Chapter 2 (Experimental Design). In 2019, compost was applied at rates of 0 (Control), 14 (Low), 46 (Medium), and 137 (High) Mg dw ha^{-1} to a certified organic dryland spring wheat field. Flux measurements were obtained pre- and post-harvest nearly continuously from May-September 2020. The respiration chambers were arranged in two configurations to measure between-treatment variability ("All" configuration) and within-plot variability ("North" configuration). The "All" configuration consisted of one chamber in each of the 16 plots for ~28 days: May 28-31, June 1-16, July 24, and August 18-26 (n=1 plot $^{-1}$, n=4 treatment $^{-1}$; Figure 11a). The "North" configuration consisted of two chambers in each of the eight northern-most plots of the experiment block for ~33 days: May 15-28, July 24-25, August 4-5, and 26-31, and September 1-13, (n=2 plot $^{-1}$, n=2 treatment $^{-1}$) (Figure 11b). Coupled temperature (˚C, model LI- 8150-203) and volumetric water content (VWC, model Decagon EC-5) soil sensors were attached to 8 of the 16 chambers and installed at an ~8-cm depth. To ensure the moving parts of the chamber were not encumbered, plants growing inside the chamber collars and ~ 5 cm outside of the chamber were cut to 2.54 cm and removed from the site.

The LI-8100A has 3 main weather proofed components: an infrared gas analyzer and control unit, the LI-8150 16-port multiplexer, and the 16 long-term automated chambers (8100-104). Chamber set up followed the LI-COR manual protocols (LI-COR, 2010). In brief, the

chambers were placed over 20-cm diam. polyvinyl chloride (PVC) sewer pipe sections, each of which was ~18 cm tall. The pipes were inserted into the soil to a depth of ~9 cm and remained undisturbed throughout preharvest measurements. The collars were removed for harvest and replaced for the remainder of the season. Because the soil chambers were moved between North and All configurations, it was advantageous to use the soil collars because repeated measures could be taken by returning to precisely the same location without any fresh disturbance. Chamber height offsets were calculated between the soil surface and the chamber baseplate. The final chamber offset value was the mean of 3 measurements, taken from the north, east, and west directions in each soil collar. Observation lengths ranged between 50-90 seconds with a 20-second pre-purge and a 30-second post purge. Each flux measurement was the interpolated slope of the increase in chamber CO_2 over a 60-second sequence of measurements (Figure 10).

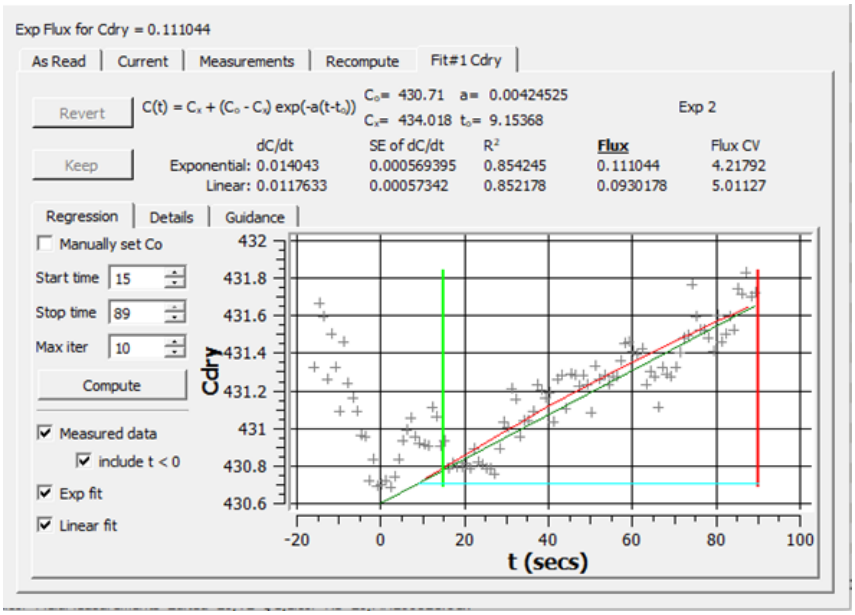


Figure 10. An example of the output from LI-COR's Soil Flux Pro software showing individual data points initially descending while the chamber is open (-20 to 0 seconds). After the chamber closes at 0 seconds, values begin to rise; green (15 seconds) and red vertical lines (90 seconds) indicate period over which slope of increasing concentrations are calculated. Red-fit line indicates the exponential fit (R^2=0.854, flux 0.11 µmol CO_2 m^{-2} s^{-1}) while green-fit line indicates the linear fit (R^2=0.852, flux 0.09 µmol CO_2 m^{-2} s^{-1}).

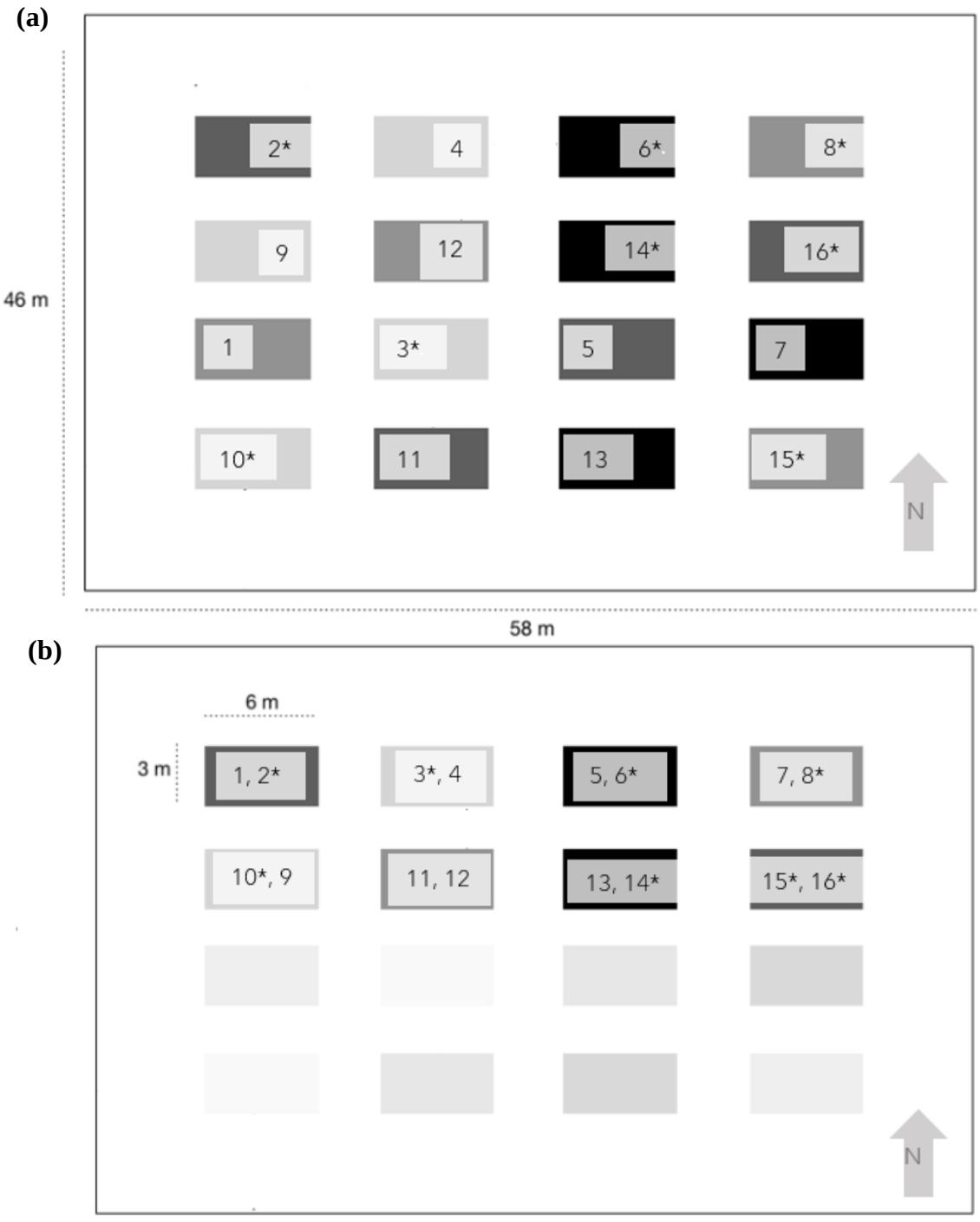


Figure 11. **(a)** All (n=1 plot $^{-1}$, n=4 treatment $^{-1}$) soil respiration chamber configurations. **(b)** North (n= 2 plot $^{-1}$, n= 2 treatment $^{-1}$) and Chamber numbers with asterisks indicate coupled temperature and moisture sensors. The plots received compost 1-year prior to soil respiration measurements. Treatment applications were, in Mg dry weight ha^{-1}, Control=0, Low=14, Medium=46, High=137) and are represented by color gradient, the lightest grey is Control and black is High.

The LI-COR software calculates water-corrected mass CO_2 flux (μmol CO_2 m^{-2} s^{-1}) and descriptive statistics. It fits linear and exponential regressions through the measurements and compares, for each measurement, the normalized sums of the squares of the residuals of the linear and the exponential fits to find the better fit. The LI-COR gas analyzer assumes Fick's first law of diffusion (Cook et al., 1998; Davidson, 2002). As the chamber CO_2 concentration builds up, the driving gradient for the flux becomes smaller because the difference between the soil and atmosphere concentrations is diminished. Thus, the exponential model is typically a better fit because the actual flux at any point in time is decreasing as the measurement period is extended. Linear fits are not uncommon to find in relatively high bulk density and compacted soils (Gorres, 2016). While ~ 64% of the data from the North configuration had a linear fit, exponential flux values were used for the analysis of the present study for ease of cross comparison to others research findings.

Statistical Analysis

The large dataset of over 21,000 flux measurements (~60 day^{-1} * 24 hours day^{-1} * 60 minutes $hour^{-1}$ * 60 seconds $minute^{-1}$) required considerable wrangling. Quality control was performed in LI-COR's data analysis software, SoilFluxPro (Version 4.2.1), by: (i) removing measures with missing column data (as occurred when stopping a measurement program before a chamber measurement was complete) (ii) correcting for pre- and post- harvest chamber offsets, (iii) replacing negative chamber temperatures with the values from the next closest chamber, (iv)

removing data with Cdry[3] values >450 ppm (Price, 2020) and (v) removing data with chamber time closing <10 and >130 seconds after the initiation of flux measurements. The respective datasets were then imported into SAS (SAS Institute Inc, 2013.) and fluxes were discarded from the two "All" or "North" datasets by applying the following criteria: (i) negative exponential flux values, (ii) CV >5%, and (iii) R^2 <0.85. Chamber 5 flux values in the North configuration High treatment were removed due to mechanical malfunction. After quality control measures were applied, ~70% of the North data remained (n =8,829 fluxes) and 84% remained for All (n= 12,375).

Mean monthly values were derived and because the data were nonparametric for both configurations, values were natural log (ln)-transformed for statistical analyses. Statistically significant differences (SSD) were determined via an ANOVA at a $p < .05$. The potential for average differences were determined by ANOVA linear contrasts and pairwise comparisons were made by the Duncan's Multiple Range Test (MRT) with a significance level determined at alpha = 0.05 (Bewick, 2004). The figures and tables in this chapter are made from the raw data values but the statistical differences were determined using ln values.

[3] Cdry is the initial value for CO2 ppm that has been corrected for water vapor, which should be around the ambient level. When Cdry is outside of the ambient levels it implies equipment error such as impediments to the pathway of the infrared gas analyzer (e.g., water or creatures).

Seasonal cumulative fluxes (Mg C ha^{-1} $season^{-1}$) were calculated by converting each µmol CO_2 m^{-2} s^{-1} flux values to Mg C ha^{-1} s^{-1} since 1 mol of C weighs 12 grams and there are 10,000 m^2 ha^{-1}. Average monthly fluxes (Mg C ha^{-1} $month^{-1}$) were calculated for May, June, July, August, and September and summed to provide the cumulative respiration for each treatment for the 5 month measurement period (Yang, 2019). Appendix D contains the raw data, conversions and model output for field soil respiration.

Results

The Medium compost rate had significantly higher fluxes than any of the other treatments, median (± IQR) fluxes were 2.3 ± 0.9 µmol CO_2 m^{-2} s^{-1} ($F^1 = 7.27$, $p < .01$; Table 6, 12a). Low and High amended and unamended (Control) plots were not significantly different from each other (Figure 13a); median fluxes were 2.1 ± 0.6, 1.8 ± 0.6, and 2.3 ±1.1 µmol CO_2 m^{-2} s^{-1}, respectively (Table 7). No differences were detected between treatments within any month ($F^9 = 0.34$, $p = .96$). Over the four-month measurement period, Medium respired significantly ($F^1 = 6.21$, $p = .03$) more C (median ± IQR, 2.5 ± 2.4 Mg C ha^{-1}) than other +C treatments and the Control (2.3 ± 2.4 Mg C ha^{-1};

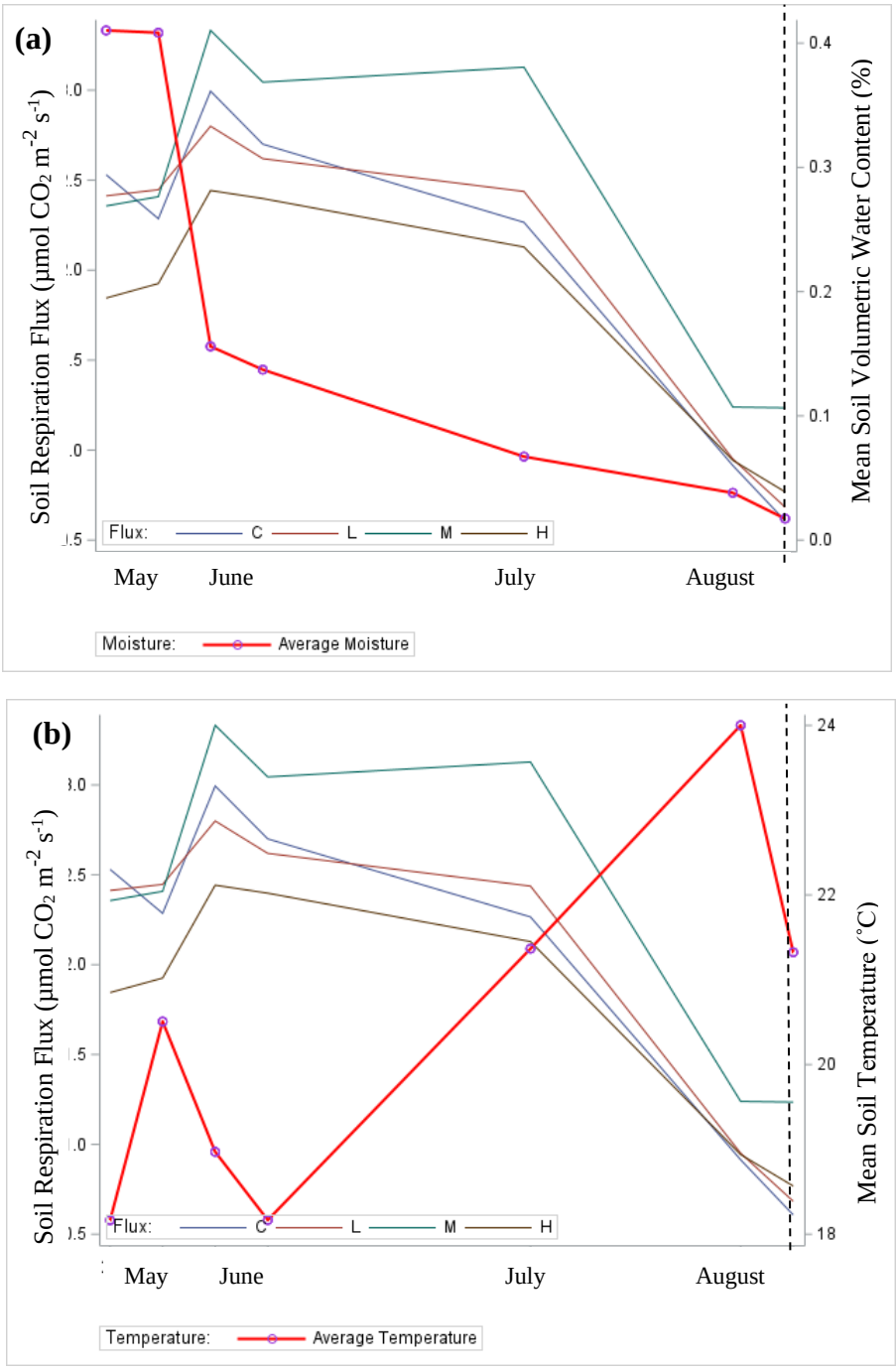
Table 6). Low and High +C plots and unamended (Control) plot median (± IQR) flux rates of 2.0 ± 2.33, 2.0 ± 1.7 and 2.3 ± 2.4 Mg C ha^{-1} were not significantly different (Figure 13b).

Across all treatments, intermediate soil moisture and temperature conditions in July were generally associated with the highest median (± IQR) fluxes: 3.1 ±0.9 µmol CO_2 m^{-2} s^{-1}. During cooler portions of the growing season (e.g., May), fluxes were 50-80% of these maxima; during hotter and drier portions of the growing season (e.g., August), fluxes were 25-50% of these maxima. As temperatures and VWC continued to decline into September, fluxes also generally declined (Table 7, 12). The effects of treatment and month were significant, but these nested terms were not significant.

There was no indication that within plot variability ("Plot_ID") was significant in the North configuration µmol CO_2 m^{-2} s^{-1}, however, there was significant within treatment plot variability ("Replicate:Treatment"; $F^{11} = 2.95$, $p < .01$) between the Low compost treated plots (Table 8, Figure 15).

Table 6. All configuration statistical models and linear contrasts for µmol CO_2 m^{-2} s^{-1} soil flux and Mg C ha^{-1} flux rates over a four-month measurement. The natural log transformed monthly means were used to determine statistical significance ($p < .05$). Measurements were from a dryland spring wheat soil treated with compost in 2019 at 0 (C), 14 (L), 36 (M), 137 (H) Mg ha^{-1}.

Model I Flux (umol CO_2 m^{-2} s^{-1})					
Effect	**DF**	**Type III SS**	**Mean Square**	**F Value**	**p Value**
Treatment	3	0.33	0.11	5.12	0.003
Harvest	1	3.42	3.42	157.37	<.0001
Month:Harvest	3	3.22	1.07	49.26	<.0001
Replicate:Treatment	12	0.87	0.07	3.32	0.001
Model II Flux (umol CO_2 m^{-2} s^{-1})					
Effect	**DF**	**Type III SS**	**Mean Square**	**F Value**	**Pr > F**
Treatment	3	0.28	0.09	6.19	0.001
Month	3	3.22	1.07	72.1	<.0001
Treatment:Month	9	0.01	0.01	0.34	0.96
Replicate:Treatment	12	1.00	0.08	5.59	<.0001
Linear Contrasts					
Contrast	**DF**	**Contrast SS**	**Mean Square**	**F Value**	**Pr > F**
C vs L	1	<.001	< .0001	0.04	0.85
C vs M	1	0.16	0.16	7.27	< .01
C vs H	1	0.02	0.02	0.96	0.33
Model I Cumulative Flux (Mg C ha^{-1} $season^{-1}$)					
Effect	**DF**	**Type III SS**	**Mean Square**	**F Value**	**Pr > F**
Treatment	3	0.14	0.04	4.93	0.03
Month	3	3.38	1.13	120.11	<.0001
Linear Contrasts (Mg C ha^{-1} $season^{-1}$)					
Contrast	**DF**	**Contrast SS**	**Mean Square**	**F Value**	**Pr > F**
C vs L	1	0.00	0.00	0.09	0.77
C vs M	1	0.06	0.06	6.21	0.03
C vs H	1	0.01	0.01	1.35	0.28



12. All configuration mean **(a)** volumetric water content and **(b)** soil temperature and (red overlays; n = 7) to ~8-cm depth for 2020 soil respiration flux measurements (n= 4 treatment^{-1}) taken over a four-month period. The black, dashed, vertical line indicates the post-harvest date. Measurements were from a dryland spring wheat soil treated with compost in 2019 at 0 (C), 14 (L), 36 (M), 137 (H) Mg dry weight ha^{-1}.

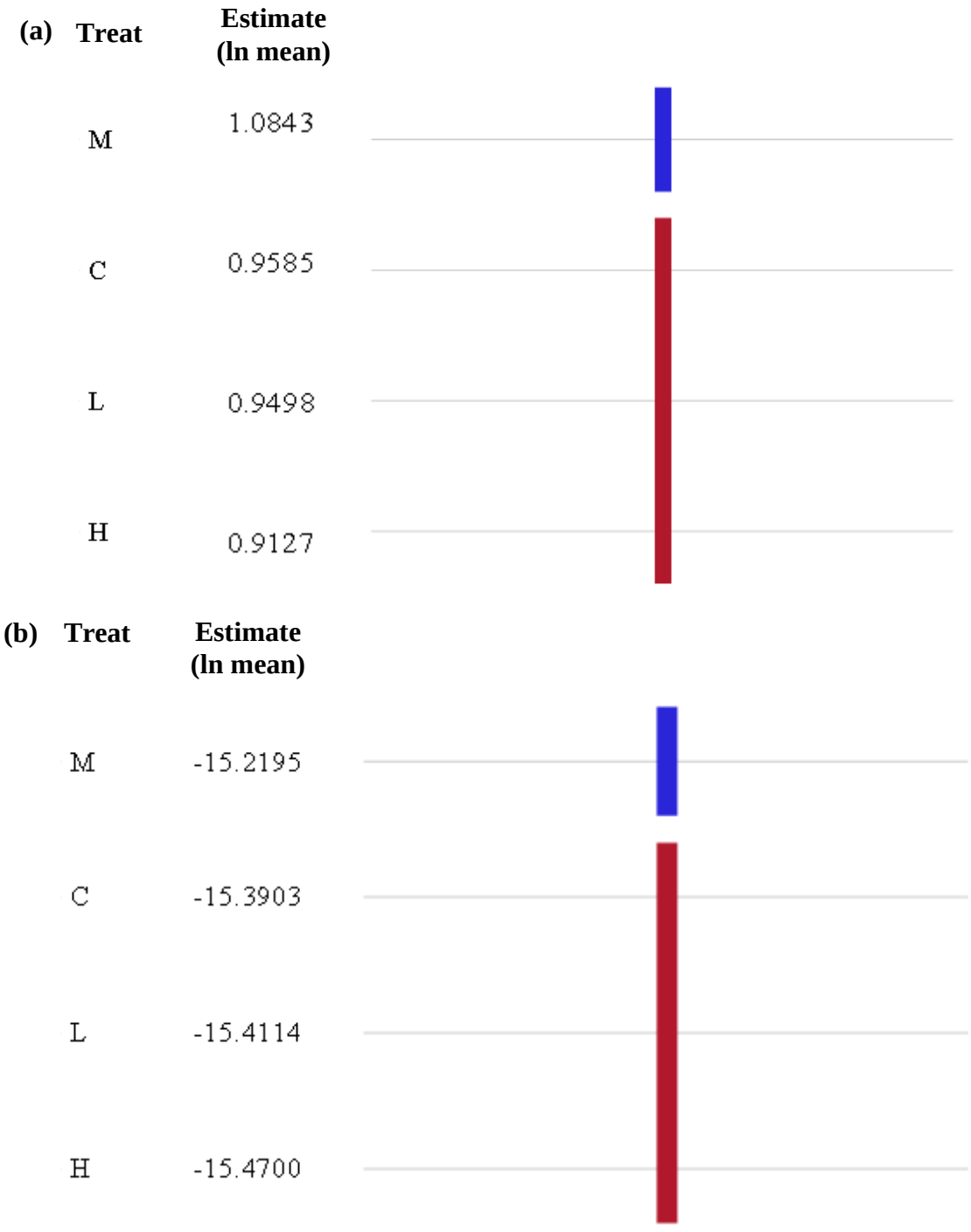


Figure 13. The Duncan's multiple range test (significance at alpha = 0.05) comparing soil fluxes for the All configuration **(a)** µg CO_2 m^{-2} s^{-1}, and **(b)** the Mg C ha^{-1} respiration rates over a four-month period. Measurements occurred one-year after applications (Treat) of 0 (C), 14 (L), 46 (M), 137 (H) Mg ha^{-1} compost. The natural log-transformed (ln) means covered by the same bar are not significantly different.

Table 7. All configuration statistical measures of soil respired monthly fluxes in µmol CO_2 m^{-2} s^{-1} and cumulative Mg C ha^{-1} over a four-month measurement period. Measurements occurred one-year after 0 (C), 14 (L), 46 (M), and 137 (H) Mg dry weight ha^{-1} compost applications. Different lowercase italicized letters in the month and treatment (Treat) column indicate statistically significant differences between natural log-transformed mean monthly plot values (Duncan's Multiple Range Test, alpha = 0.05). Treatments in August with an asterisk (*) indicate post-harvest measures.

		Flux (µmol CO_2 m^{-2} s^{-1})							
Month	**Treat**	**n**	**Min**	**Q1**	**Median**	**Mean**	**Q2**	**Max**	**IQR**
May a	**C**	619	0.8	1.9	2.3	2.5	3.0	4.7	1.1
	L	523	0.5	1.2	2.3	2.4	3.5	7.6	2.3
	M	615	0.8	1.7	2.3	2.4	2.9	5.0	1.1
	H	597	0.4	1.5	1.8	1.9	2.2	3.8	0.8
June b	**C**	1388	0.9	1.9	2.5	2.8	3.4	6.8	1.5
	L	1306	0.4	1.9	2.4	2.7	3.3	8.1	1.4
	M	1261	0.7	1.9	2.6	3.0	3.9	8.8	2.0
	H	1387	0.5	1.7	2.1	2.3	2.8	6.0	1.2
July a,b	**C**	31	1.4	1.6	2.3	2.3	2.8	3.8	1.2
	L	29	1.4	1.8	2.1	2.4	2.4	11.8	0.6
	M	30	2.0	2.6	3.1	3.1	3.5	5.5	0.9
	H	29	1.5	1.8	2.0	2.1	2.4	2.9	0.6
August c	**C**	434	0.2	0.7	0.9	1.0	1.1	3.5	0.4
	L	456	0.3	0.8	0.9	1.0	1.1	3.5	0.3
	M	429	0.3	0.9	1.1	1.3	1.4	8.2	0.5
	H	416	0.4	0.8	0.9	1.0	1.1	3.1	0.4
	***C**	708	0.3	0.5	0.6	0.6	0.8	1.9	0.3
	***L**	724	0.2	0.5	0.6	0.7	0.8	1.8	0.3
	***M**	657	0.3	0.5	0.7	1.2	1.2	11.8	0.7
	***H**	735	0.4	0.6	0.7	0.8	0.9	2.0	0.2
		Cumulative Flux (Mg C ha^{-1} $season^{-1}$)							
	Treat	**n**	**Min**	**25 %**	**Median**	**Mean**	**75%**	**Max**	**IQR**
	C a	3,180	0.26	1.1	2.3	2.5	3.5	8.7	2.4
	L a	3,038	0.32	1.1	2.0	2.4	3.4	15.1	2.3
	M b	2,992	0.36	1.4	2.5	2.9	3.8	15.1	2.4
	H a	3,164	0.45	1.2	2.0	2.2	2.8	7.7	1.7

Table 8. A statistical model for the North configuration examining within-plot, between-treatment plots, and between treatment variability for µmol CO_2 m^{-2} s^{-1} soil respiration using natural log-transformed mean monthly values (n = 2 $plot^{-1}$).

Effect	DF	Type III SS	Mean Square	F Value	Pr > F
Plot_ID	7	0.70	0.10	1.53	0.18
Treatment	3	0.03	0.01	0.40	0.75
Replicate:Treatment	11	0.78	0.07	2.95	<0.01
Replicate:Plot_ID	7	0.11	0.02	0.24	0.97

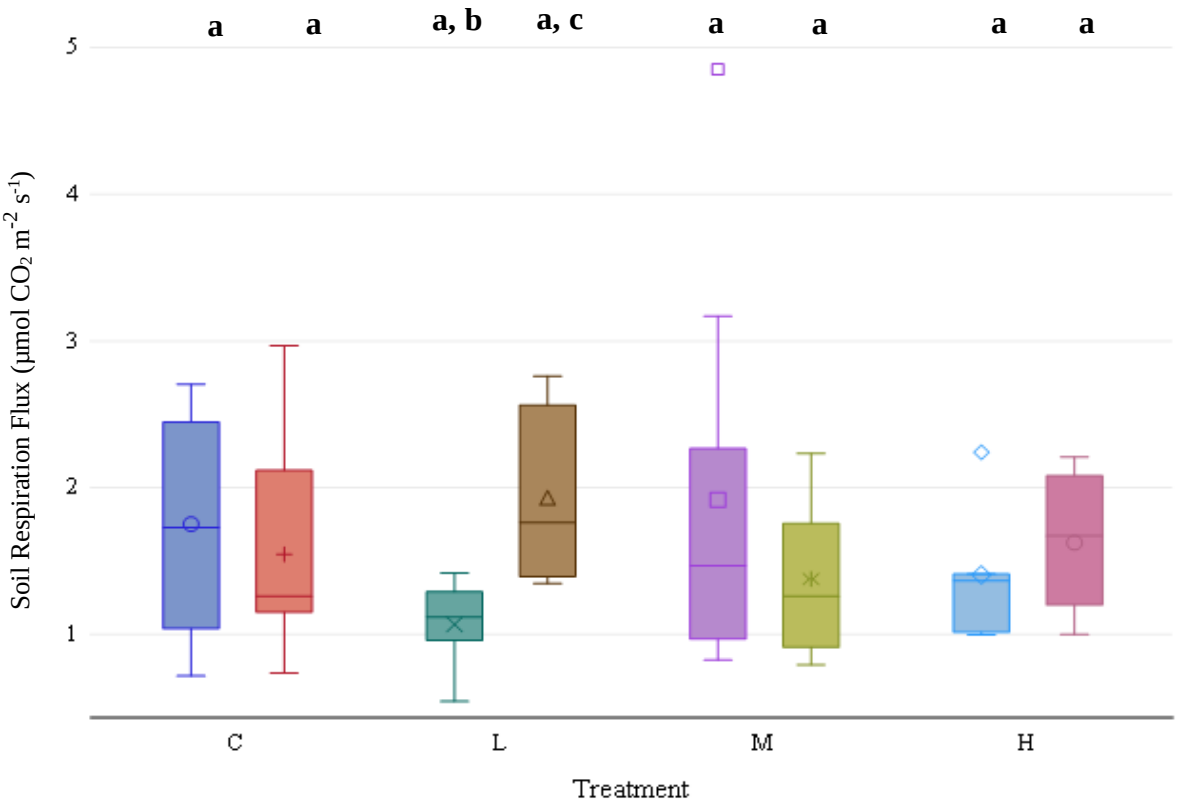


Figure 14. The North configuration soil fluxes from within treatment plots (n= 2 chambers $plot^{-1}$). Significance was determined by Duncan's multiple range test (significance at alpha = 0.05) comparing the natural log-transformed between plot_id mean µg CO_2 m^{-2} s^{-1}soil fluxes. Measurements occurred one-year after of 0 (C), 14 (L), 46 (M), 137 (H) Mg dry weight ha^{-1} compost applications. Horizontal lines are medians and shapes inside boxes are the mean.

Discussion

Median μmol CO_2 m^{-2} s^{-1} and Mg C ha^{-1} season-1 soil respiration was higher for Medium compost treated soils compared to other +C and untreated (Control) plots, as was variability in the rate. No differences were detected in soil respiration between Control, Low and High treatments. Based on the North configuration, within plot variability and between treatment variability did not appear to have a significant impact on the results. Many studies have detected higher SR from increasing rates of exogenous substrate additions (Chen, 2019; López-López, 2012), which was not the case in this experiment. Similar to the laboratory incubation experiment in Chapter 3, there is no biological explanation for why Medium had higher respiration rate than all other +C treatments and the Control. Interestingly, the results in that laboratory incubation experiment indicated the Medium had the lowest respiration rate in comparison to other treatments. One explanation for these contrasting results is that the labile, easy to digest, C substrates provided by the compost may have been mineralized in the first year following application and so any treatment effects might have been transient and therefore missed by our measurement campaign. Many studies attribute respiration responses to fresh C substrate addition.

Fabrizio et al. (2009) measured SR 150 days after plowing 50 and 85 Mg ha^{-1}compost into the top 20 cm of a maize (Zea mays L.) field. While no SSD's were evident, the mean flux values from the larger treatment rate were higher (6.6 ± 0.6 μmol CO_2 m^{-2} s^{-1}) in comparison to the lower treatment rate fluxes (~6.1 ± 0.6 μmol CO_2 m^{-2} s^{-1}) and the control (4.4 ± 1.3 μmol CO_2 m^{-2} $s^{-1)}$. Fabrizio et al. (2009) associated the increases in flux rates with more labile forms of C accessible to the soil community from the compost treatments. The control SR increased over

the season from 2.46 ± 0.18 to 7.77 ± 2.59 μmol CO_2 m^{-2} s^{-1} and showed more variability than the compost-treated soils. In the present study, compost treatment variability of respired μmol CO_2 m^{-2} s^{-1}, ranked low to high by IQR. was : High (0.2-1.2), Low (0.3-2.3), Control (0.3-1.5), Medium (0.5-1.1) and variability decreased as the season progressed with the lowest variability occurring in August.

An extreme example of a high compost application rate, which is not likely to occur for largescale agriculture, is from work by Beesley et al. (2014). Significant increases in SR from an application of ~1,140 Mg ha^{-1} of greenwaste compost (~30 cm deep) to an urban soil planted with turf-grass were detected. The treated soil had fluxes 1.5 μmol CO_2 m^{-2} s^{-1} higher than the control (no greenwaste compost). The exact duration of time that had passed since applying compost was not clear, but it did not exceed 3-yr from compost application. The compost rate used by Beesley et al. (2014) was ~730% higher than the highest rate applied in the present study. Variability was greater in the compost treated soil (IQR ± 2 Beesley et al. (2019 attributed their results to increased substrate availability.

Ray et al. (2020) measured SR from a cropped (collard greens) field in Texas immediately following organic amendment applications of chicken litter, dairy manure, and sewage waste, respectively, at three target application rates (168, 336, 672 kg N ha^{-1}). By comparison, compost treatments in the present study were ~ 10, 40, and 110 kg N ha^{-1} (Chapter 2,Table 1). Ray et al. (2020) found that all amended soils had significantly ($p < .01$) higher SR rates compared to the control (no compost) and that SR increased with increasing treatment rates. The largest chicken litter application rate had the highest CO_2 emission (4.4 μmol CO_2 m^{-2} s^{-1} in season 1, and 5.4 μmol CO_2 m^{-2} s^{-1} in season 2), and the lowest dairy manure application had the

lowest SR emission (2.3 µmol CO_2 m^{-2} s^{-1} in season 1, and 2.7 µmol CO_2 m^{-2} s^{-1} in season 2). These treatments were applied annually for two years and, on average, the second growing season had higher SR than the first, except for treatments that received the dairy compost at medium application rates. The increases were directly proportional to the rate of substrate application for all three treatments. The highest flux from Ray et al. chicken litter compost in year 1 was 48% higher than the highest Median flux in this experiment. Their lowest flux from cattle manure compost in year 1 was 21% higher than the lowest detected in this experiment. This could be attributed to the one-year lapse in time that occurred before our measurements. Furthermore, their measurements occurred on Wockley fine sandy loam which would have less bulk density in comparison to the , in contrast to the Farnuf loam soils that hosted this experiment.

Conclusion

Results indicate that soil flux from the Medium compost application (46 Mg ha^{-1}) was significantly higher than flux rates from either a 3.3X-lower (14 Mg ha^{-1}) or 3X-higher (137 Mg ha^{-1}) application rate. Variability in flux appeared greater for Medium in comparison to other compost treatments but in the North configuration, variability was not a significant effect. The results from this study indicate that one-year after compost applications, soil flux may be detectable but that increasing rates relative to substrate additions may not occur. Still, inherent SR rates change as soil biological activity is impacted by climactic shifts (Ishaq, 2020). Increasing the productivity of our landscapes is imperative towards supplying food, fiber, and fuel for present and future generations. One way to address this challenge is to work towards

replacing the soil C that has been lost. If compost is used to mitigate soil degradation, it is unlikely to increase CO_2 emissions in dryland agricultural systems.

CHAPTER FIVE

CONCLUSION

The objective of this two-year study was to determine how wheat (Triticum aestivum L.) grain yield, protein concentration, and above ground biomass (AGB), and indices of soil health including SOM, laboratory substrate-induced respiration (SIR), and field soil respiration (FSR) measurements would respond to compost applications in a Northern Great Plains dryland cropping system on Farnuf loam soils (fine-loamy, mixed, superactive, frigid Typic Argiustolls). Single compost applications (0, 14, 46, and 137 Mg ha^{-1}) that were made to a spring wheat and cover crop rotation did not impact wheat yield, AGB, or SOM content ($p > 0.05$). The 46 Mg ha^{-1} (Medium rate) compost application elicited a higher in situ SR (2.5 Mg C ha^{-1} $season^{-1}$) than other +C and -C treatments but also had the lowest SIR (25 mg CO_2-C g^{-1} SOC) of +C treatments in a laboratory incubation experiment. These results indicate that when cattle manure compost is applied to fine-textured dryland soils, wheat grain yields and protein concentration, as well as soil properties like SOM, do not appear to be enhanced in the short-term. These results also indicate that, one year after application, compost treatments may be detectable via soil health indices such as SIR and FSR, but these results can be inconsistent. Future research should clarify the most sensitive, robust, and cost-effective indices of soil health capable of guiding improved agricultural practices, including compost.

Several profound learning outcomes have resulted from engaging in this graduate thesis. Broadly, these outcomes have to do with understanding the importance of consistency, organization, communication, and self-care through experience. The “wrong” choices made at

the beginning of this experiment have had a legacy effect lasting for duration of this project. This serves as a reminder to write often and share with others having research experience prior to engaging in new experiments.

Because nature is inherently variable, consistency in research methodology is essential. This is point is made clear early on and throughout a scientists career, but when the resource of time or finance diminishes, maintaining consistency is not always possible. As the, mostly, nonsignificant results began to arrive in my email inbox, I immediately thought of the inconsistencies (e.g., using different soil analytical labs, planting and harvesting methods, and statistical coaches between years) that I had allowed into my experiment and I wondered how much noise they added to what is already inherent in nature.

The most surprising outcome of this work was that detecting a compost signal through the noise of the soil habitat was more difficult than anticipated. The nonsignificant downward trend in yields from 2019, no change in yields between treatments in 2020, and finally the Medium compost rate having the lowest SIR but the highest in situ respiration are outcomes that I had not expected nor had hoped for. It is because of the inconsistencies that I allowed into my experiment, that I now have a profound understanding of why replication, randomization, and nonattachment to specific analytical outcomes are linchpins of science.

The contribution that this body of work provides to the agricultural community is that high-rate, one-time compost applications made to a fine textured soil in a dryland setting are not likely to provide short-term responses such as increases in yield and protein concentrations or soil health indicators such as soil carbon. Nonetheless, cycling “waste” resources back into the soil habitat has been shown, in other studies, to improve soil health indicators in the long-term.

Agriculture is responsible for ~ 24% of anthropogenic greenhouse gas emissions. It is widely accepted that humans have significantly disrupted the Earth's global cycles and that these impacts are likely to have long-lasting, potentially irreversible, repercussions. If humans halt the exhaustive, extraction-based approaches that have been degrading the soil resource and engage in a new paradigm of land management strategies that support diversity and aim for resiliency, the quality of life on this planet will improve.

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APPENDICES

APPENDIX A

Yield Chapter 2

APPENDIX B

Elemental Content Chapter 3

APPENDIX C

Substrate Induced Respiration Chapter 3

APPENDIX D

Field Soil Respiration Chapter 4