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The Impacts of Compost Application on Urban Soil Health and Carbon Storage:
Investigating the Role of Managed Organic Waste Products in Supporting City Greenspaces

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Peer reviewed|Thesis/dissertation

UNIVERSITY OF CALIFORNIA, MERCED

The Impacts of Compost Application on Urban Soil Health and Carbon Storage:
Investigating the Role of Managed Organic Waste Products in Supporting City
Greenspaces

A dissertation submitted in partial satisfaction of the requirements
for the degree Doctor of Philosophy

in

Environmental Sciences

by

Zachary David Malone

Dissertation Committee:

Professor Rebecca Ryals
Professor Asmeret Asefaw Berhe
Professor Sally Brown
Professor Colleen Naughton
Professor Teamrat Ghezzehei

2026

Chapter 2 Reproduced from Malone, Z., Berhe, A.A., & Ryals, R. (2023). Impacts of organic matter amendments on urban soil carbon and soil quality: A meta-analysis. *Journal of Cleaner Production*. © 2023 The Authors. Published by Elsevier. Distributed under the terms of the Creative Commons license.

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University of California, Merced
2026

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List of Symbols

BD: Bulk Density

C: Carbon

CH₄: Methane

CO₂: Carbon Dioxide

DF: Density Fractionation

EC: Electrical Conductivity

ESM: Equivalent Soil Mass

FD: Fixed Depth

FLF: Free Light Fraction

GHG: Greenhouse Gas

HF: Heavy Fraction

K: Potassium

MBC: Microbial Biomass Carbon

MBN: Microbial Biomass Nitrogen

N: Nitrogen

NDVI: Normalized Difference Vegetation Index

NH₄⁺: Ammonium

N₂O: Nitrous Oxide

NO₃⁻: Nitrate

NO_x: Nitric Oxide

OLF: Occluded Light Fraction

OMA: Organic Matter Amendment

P: Phosphorus

SOC: Soil Organic Carbon

SOM: Soil Organic Matter

VWC: Volumetric Water Content

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“ 'Cause I'm a real tough kid, I can handle my shit. They said, "Babe, you gotta fake it till you make it" and I did.”

— Taylor Swift

To the city of Merced: “It's the kinda place you walk into and everybody knows your name. It's the kinda place you walk out of and never feel the same.”

— Sammy Rae

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Curriculum Vita

Zachary Malone

EDUCATION

PhD Candidate - University of California, Merced

Aug 2020 – May
2026

Environmental Systems – 3.95 GPA

Biogeosciences Concentration

Advisor: Dr. Rebecca Ryals

May 15, 2020

BS - University of California, Merced

Earth System Science Major – 3.80 GPA, High Honors

RESEARCH EXPERIENCE

PhD Candidate at UC Merced – Ryals Agroecology Lab

Aug 2020 – May
2026

Center for Accelerated Mass Spectrometry Graduate Intern at
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May 2024 - Aug
2024

Research Assistant at UC Merced – Ryals Agroecology Lab

Sept 2019 - May
2020

Research Assistant at UC Merced – Moran Plant Lab

Aug 2018 - May
2019

Research Assistant at UC Merced – Dawson Marine Lab

Sept 2017- May
2019

PROFESSIONAL EXPERIENCE

UC Merced Yosemite Wilderness Education Ranger	Oct 2018 - May 2020
Yosemite Education Ranger President of the UCM Natural Resource and Ecology Association	May 2019 - Aug 2019 July 2019 - May 2020
Interpretation Intern, Yosemite National Park	May 2018 - Aug 2018
University of California, Merced, Trails Intern	Dec 2017 - May 2018

PUBLICATIONS

Published in the peer-reviewed literature

1. Defeo, S., Erickson, S., Perez Mendoza, M.F., Cooper, A., Barrios, B., **Malone, Z.**, Baxter, R.D., Ghosh, S. and Harmon, T.C. (2025). Making nanomaterial-enabled nitrate sensors useful for real water systems: User-centric design perspectives. *Frontiers in Sensors*, 6, 1513701. <https://doi.org/10.3389/fsens.2025.1513701>
2. McDonald, M. D., Owusu-Ansah, C., Ellenbogen, J. B., **Malone, Z. D.**, Ricketts, M. P., Frolking, S. E., ... & Weissman, J. L. (2023). What is microbial dormancy?. *Trends in Microbiology*. <https://doi.org/10.1016/j.tim.2023.08.006>
3. **Malone, Z.**, Berhe, A. A., & Ryals, R. (2023). Impacts of organic matter amendments on urban soil carbon and soil quality: A meta-analysis. *Journal of Cleaner Production*, 419, 138148. <https://doi.org/10.1016/j.jclepro.2023.138148>
4. T Longbottom, L Wahab, A Jurusik, K Moreland, M Dolui, K Min, T Thao, M Gonzales, Y Perez Rojas, J Alvarez, **Z Malone**, J Yan, T Ghezzehei, and AA Berhe. *What's Soil Got to do With Climate Change?* GSA Today. (2022). <https://doi.org/10.1130/GSATG519A.1>

Non Referred Report:

1. Hicks, Akiebia S., **Zachary Malone**, Megan A. Moore, Roslynn Powell, Austin Thompson, Patricia A. Whitener, and Rowan Williams. (2021). Green inequities:

Examining the dimensions of socioenvironmental injustice in marginalized communities. *Parks Stewardship Forum* 37(3): 534–540.
<https://doi.org/10.5070/P537354736>

Public Publications:

1. **Zachary Malone.** "Science funding cuts by Trump administration impact Merced County." Opinion in *The Merced Sun-Star*, November 11, 2025.

PRESENTATIONS

ORAL:

1. **Malone, Z.,** Lee, S., & Ryals, R. 2026, February. *Impacts of Compost on Urban Soil Profile Health and its Climate Impacts.* United States Composting Council Annual Meeting.
2. **Malone, Z.,** Lee, S., & Ryals, R. 2025, August. *Interplay of compost and irrigation on urban soil health and associated climate impacts.* Ecological Society of America Annual Meeting.
3. **Malone, Z.,** Lee, S., & Ryals, R. 2025, June. *Impacts of Compost on Turfgrass Soil Health.* **Invited talk** at Compost Application Workshop, Pleasanton, CA.
4. **Malone, Z.,** Lee, S., & Ryals, R. 2024, November. *Effects of Compost on Soil Carbon and Nutrient Dynamics in Urban Parks.* ASA, CSSA, SSSA International Annual Meeting, San Antonio, TX.
5. **Malone, Z.,** Lee, S., & Ryals, R. 2024, May. *Compost Impacts on Urban Soil Carbon, Soil Health, and Climate.* State of the Alameda Creek Watershed Conference.
6. **Malone, Z.,** & Ryals, R. 2023, October. *Impacts of Compost on Urban Soils Greenhouse Gas and NO_x Emissions.* In ASA, CSSA, SSSA International Annual Meeting. ASA-CSSA-SSSA.
7. **Malone, Z.** 2023. *Turning City Waste into Green Space; How to use Compost to Fight Climate Change in Cities.* SNRI Research Symposium: Climate Action Research.
8. **Malone, Z.** 2022. *Impacts Of Organic Matter Amendments On Urban Soil Carbon Sequestration And Soil Health: A Review.* Research Committee W4170: Beneficial Use of Residuals to Improve Soil Health and Protect Public and Ecosystem Health 2022 Annual Meeting.

POSTERS:

1. **Malone, Z.** and Ryals, R. 2025, December. *Effects of Compost, Recycled Water Use, and Baseline Soil Carbon on Urban Soil Greenhouse Gas and Nitrogen Trace Gas Emissions.* In AGU International Annual Meeting.
2. **Malone, Z.,** Lee, S., McFarlane, K., and Ryals, R. 2024, December. *Impacts of long-term compost application to turfgrass on soil carbon stocks, density fractions, and radiocarbon age.* In AGU International Annual Meeting.

3. **Malone, Z.**, Lee, S., and Ryals, R. 2023, December. *Impacts of Compost Use on Urban Soil Health and Climate, with Implications for Public Health*. In AGU International Annual Meeting.
4. **Malone, Z.**, Schoonmaker, K., Probert, T., Sung, Michelle., Young, Shannon., and Ryals, R. 2023, December. *Partnerships between local government and scientists to achieving California's landmark methane reduction policy (SB 1383)*. In AGU International Annual Meeting.
5. **Malone, Z.**, & Ryals, R. 2023, October. *Impacts of Compost on Urban Soils Greenhouse Gas and NOx Emissions*. In ASA, CSSA, SSSA International Annual Meeting. ASA-CSSA-SSSA.
6. **Malone, Z.**, & Ryals, R. 2023, September. *Impacts of Compost on Soil Health and Gas Emissions*. FIRA USA California Agricultural Robotics and Technology Forum
7. **Malone, Z.**, Berhe AA, Ryals R, 2022. *Meta-Analysis of Organic Matter Amendments for Urban Soil Carbon Sequestration and Urban Soil Health*. 8th International Meeting on Soil Organic Matter, Seoul, South Korea.
8. **Malone, Z.**, Berhe AA, Ryals R. 2021. *Meta-analysis of Organic Matter Amendments for Urban Soil Carbon Sequestration and Soil Health Improvements*. Ecological Society of America Conference.
9. **Malone, Z.** 2018. *Merced Trails Master Plan Community Outreach*. Student Success Internship Poster Symposium, University of California, Merced.

GRANTS AND FELLOWSHIPS

LLNL Center for Accelerator Mass Spectrometry Graduate Intern	2024 – \$30,000
UC Merced Vista Farms, Food, Future Fellowship	2023 – \$35,000
UC Merced CONDESA NSF NRT Fellowship	2022 – \$35,000
UC Merced Environmental Systems Travel Grant	2022 – \$1,000
SNRI Research Symposium Travel Award	2023 – \$500
UC Merced Graduate Student Association Travel Grant	2022 – \$400

AWARDS

UC Merced Grad Slam – Second Place 2026	A three minute public presentation competition about one’s dissertation research.
UC Merced Grad Slam – Second Place 2025	A three minute public presentation competition about one’s dissertation research.
Life and Environmental Sciences Graduate Student Best Paper Award 2024	Awarded to a graduate student with a first author paper published with an LES advisor.
Second Place in the poster competition presented by the Soil Science Society of America Urban and Anthropogenic Soils Division – SSSA 2023	Poster: “ <i>Impacts of Compost on Urban Soils Greenhouse Gas and NOx Emissions</i> ”
AGU Voices for Science – Policy Track 2023	Conducted outreach to local state and federal congresspeople on science policy
NSF EMERGE BII Summer Program 2022	Summer program training in: Multidisciplinary science of how ecosystems are responding to climate change, Science of team science, Mentor training, Diversity, Equity, & Inclusion, and Science communication. The final product was a group review paper on microbial dormancy published in trends in microbiology in 2024.
Ecological Society of America Katherine S. McCarter Graduate Student Policy Award 2022	Training in science policy and participation in a virtual congressional visit day

UC Merced John and Victoria Elia Fellowship	Graduate award to recruit an incoming PhD student who attended UC Merced
Outstanding Graduating Student Award	Graduating undergraduate recognized by faculty for excellence in regard to academics, research, and service to UC Merced
Chancellor's List 2017-2018, 2018-2019, 2019-2020 Academic Years	Maintained a 3.5 or better GPA for both fall and spring semesters.
Dean's List Spring 2017	Earned a 3.5 or better GPA for the spring 2017 semester.

TEACHING AND MENTORSHIP

Co-Instructor of Record – Fundamentals of Soil Science	Joint instructor of record at UC Merced for the <u>Fundamentals of Soil Science</u> course (ESS170) with Dr. Asmeret Berhe.	Jan 2026 – May 2026
Undergraduate Student Mentor	Mentor for an undergraduate UC Merced student (Leonardo Medina) on soil processing for various projects.	Oct 2025 – May 2026
Graduate Excel Peer Mentor Environmental Systems Program	Peer mentor for eight incoming first year graduate students in the environmental systems graduate program	Aug 2025 – Dec 2025

Teaching Assistant – Fundamentals of Soil Science	Teaching assistant at UC Merced for three sections of <u>Fundamentals of Soil Science</u> and its associated lab (ESS170)	Jan 2025 – May 2025
Graduate Excel Peer Mentor Environmental Systems Program	Peer mentor for eight incoming first year graduate students in the environmental systems graduate program	Aug 2024 – Dec 2024
Mentor for Research Technician	Mentor for a research technician (Samuel Lee) on our Dublin and Pleasanton urban soil compost projects	Aug 2023 – Aug 2025
Undergraduate Student Mentor	Mentored one undergraduate UC Merced student (Jesus Ceron Gonzalez) on urban soils and organic matter amendments. This student was taught and assisted in lab work ranging from soil processing, microbial biomass fumigation, to KCl extractions	Jan 2025 – Aug 2025
Level 3: Advanced Pedagogy Series Certificate	Awarded by the UCM Center of Engaged Teaching and Learning	Spring 2022
Teaching Fellow – Aristotle House Living Learning Community	Instructor of record for the Aristotle living learning community focused on leadership, education, and service. I taught and created curriculum for a two semester long seminar each with 30 student sections comprised of first year, primarily first generation students.	Aug 2021 – May 2022

Undergraduate Student Mentor	Mentored one undergraduate UC Merced student (Isabel Hernandez) on urban soils and organic matter amendments. Student was taught and assisted in literature review of organic matter amendment use in urban systems.	Jan 2022 – May 2022
Evidence-Based Teaching Practices Certificate	Awarded by the UCM Center of Engaged Teaching and Learning	Spring 2021
Teaching Assistant – Fundamentals of Soil Science Spring of 2021	Teaching assistant at UC Merced for one section of <u>Fundamentals of Soil Science</u> and associated lab (ESS170)	Jan 2020 – May 2021
Teaching Assistant – Agroecology and Ecosystems of CA Fall of 2020	Teaching assistant at UC Merced for a section of <u>Sustainability of Agricultural Ecosystems</u> (ESS/BIO172) and <u>Ecosystems of California</u> (ESS050)	Aug 2020 – Dec 2021
Preparing to Teach in the Online Environment Certificate	Awarded by the UCM Center of Engaged Teaching and Learning	Summer 2020

SERVICE AND OUTREACH

UC Merced MIST 251 Seminar Speaker	March 2026
UC Merced ESS190 Undergraduate Seminar Speaker	Nov 2025
StopWaste Recycling Board Members Presentation	Oct 2025
AGU Thriving Earth Exchange Scientist for Zero Waste Lab	July 2025 – Present
Panelist at ESA 2025 for “I Got My Bachelor's Degree, Now What?”	Aug 2025
Environmental Systems Student Seminar Lead	May 2025 – May 2026

UC Merced ESS190 Undergraduate Seminar Speaker	Nov 2024
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Dissertation Abstract

The Impacts of Compost Application on Urban Soil Health and Carbon Storage:
Investigating the Role of Managed Organic Waste Products in Supporting City
Greenspaces

by

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This dissertation investigates the impact of organic matter amendments (OMAs), with a focus on compost, on urban soil health and carbon storage. This research addresses key questions surrounding the impacts of OMAs not only in urban areas, but on soils generally, over month to decade timescales on soil biogeochemistry and management-relevant soil characteristics. Chapter two is a meta-analysis, focusing on how soil organic matter (SOM) and soil nutrients, such as nitrogen (N) and phosphorus (P), respond to different kinds of OMAs. We found that of the OMAs tracked – biochar, biosolids, and compost – biochar increased SOM% the most, by 6.5 percentage-points, while compost increased SOM% by 3.1. While biosolids saw the lowest SOM gains at 2.1, they had significantly greater gains in soil N and P. This chapter highlighted the urban OMA work done to this point, but stressed the importance of long-term and deep soil sampling, as most studies were 30cm or shallower and were monitored for less than three years, both of which informed our future work, particularly chapters three and four.

Chapter three examines the short-term impacts of compost application on turfgrass soil biogeochemistry. A three-month greenhouse study was conducted to assess how recycled water and compost additions impact soil greenhouse gas emissions and plant growth. We found compost increased root biomass by over 50%, while recycled water increased aboveground biomass by up to 300%. These changes in biomass were partially driven by changes in available P and N, which significantly increased with compost and recycled water respectively.

These results compliment chapter four, which delves into the multi-annual effects of compost application. We conducted a two-year field trial assessing soil greenhouse gas emissions and bulk soil properties over two annual compost applications. We found a no changes to soil C or N, potentially due to high decomposition rates of the applied compost, evident by an increase of 10% in cumulative CO₂ emissions from our compost plots. Compost plots also saw reduced N₂O emissions, important for whole system

nutrient budgets. However, this study highlighted the need to monitor compost application on longer timescales, leading to our chapter four project.

Chapter five dives into the long-term impacts of compost application in urban areas, specifically looking at changes in soil C to one meter and in different C fractions. We found bulk soil C to be increased by 50% with long-term compost additions, especially in the top 10cm and 75-100cm depth ranges. This increase was consistent in the mineral-associated fraction as well, which is thought to be slower to change and contain highly persistent soil C. In addition, the molecular makeup of this mineral fraction became more similar to compost, indicating the direct incorporation of compost C into even deep mineral C pools.

In conclusion, we found significant gains in whole profile bulk soil C and soil C persistence with long-term compost application, and that short-term compost application can reduce N₂O emissions while improving turfgrass growth.

Chapter 1: Introduction

Creating organic matter products is integral to the future of waste management. In California, approximately a third of landfilled wastes are organics, accounting for 30% of the total waste stream (CalRecycle 2021). One way to divert these materials from landfills is to supervise their decomposition and create organic matter amendments ranging from compost, biological charcoal (biochar), or treated wastewater solids (biosolids). These amendments can then be applied to improve soil health; a soils continued ability to support and maintain life. Compost in particular is an excellent well-rounded soil amendment fit for many scenarios (Malone et al., 2023). In addition to improving soil health and function, diverting these organic materials from landfills would reduce methane emissions, with landfills being the largest point-source of methane in California and the third largest in the country (Duren et al., 2019; US EPA, 2023). By reducing landfill methane emissions and increasing soil carbon, composting could be an important mitigator of climate change, in addition to cobenefits from improving soil health.

Urban areas are particularly well suited for compost additions as a large share of organic wastes, such as food waste and yard trimmings, are generated in cities. By creating and using compost in these areas it could help to close the waste loop. Despite this benefit, most composting facilities are far from urban centers, due to lack of facilities and to avoid air pollution in close communities, and decentralized composting practices may be needed to best support urban compost use (Harrison et al., 2024). However, even moving compost from outside the city in for application can be feasible, especially as urban areas account for 5% of the United States land area and thus have a lot of soil that could potentially be amended (Liu et al., 2014). In urban systems, turfgrass fields alone are 2% of the United States land area (Milesi et al., 2005a). Turfgrass fields, such as soccer fields, and other urban landscapes are already managed for irrigation and fertilization and compost additions could help to supplement or even reduce the need for other management strategies. However, most study of compost has been done in managed agriculture fields, with far less knowledge of the impacts on urban soils, especially throughout time and depth of the soil (Malone et al., 2023). We know that turfgrass soils can have high baseline levels of soil carbon and nitrogen, but research is still needed to confirm if compost can further improve these already well maintained soils, and to what depth benefits may be extended to (Phillips et al., 2023; Qian & Follett, 2012).

This questions exists as many studies have investigated urban soil health and found high baseline levels of carbon in the topsoil, but far fewer studies exist assessing urban soil below 30cm (Malone et al., 2023). Approximately 70% of bulk soil carbon is stored below this depth, thus, neglecting this portion of the soil profile can underscore total soil carbon and misrepresent changes over time (Jackson et al., 2017; Pries et al., 2023). Temporal changes are possible in turfgrass soils that receive plentiful irrigation, which allows for enhanced carbon transit in shorter periods of time than their natural system analogs. This carbon can then be used by microbes deeper in the soil and/or become adsorbed to deeper mineral interfaces, where it can

become mineral associated organic matter (Cotrufo et al., 2013). Associations with minerals facilitate long term carbon storage as the bonds between minerals, such as iron and aluminum oxides, and organic matter can be very strong (Lavallee et al., 2020). A large portion of the amount of carbon that is mineral associated also exists below 30cm, further emphasizing the value in studying deeper soil carbon pools to understand changes in soil carbon and a soil's climate change mitigation capacity (Sokol et al., 2022). Prior studies have found long term organic matter amendments can increase soil carbon even up to one meter deep. Villa et al. found that soil carbon increased with 20 years of biosolid applications, even in the mineral fraction, down to one meter (Villa et al., 2024). Increases in deep soil carbon are notable not only for the difficulty in improving them in short time frames, but for increasing carbon storage and deeper soil health.

In addition to benefits in soil carbon and climate change broadly, compost additions can influence soil nutrient availability, leading to changes in soil microbial activity, greenhouse gas (GHGs) emissions, and plant production. One reason for this is that while inorganic fertilizers increase soil nutrients more readily, compost releases nutrients into the soil more slowly, leading to benefits spread out over a long-term period (Ryals et al., 2015). As compost decomposes this soil microbial activity may increase soil leading to biogenic CO₂ emissions, improving the biological health of the soil (Wang et al., 2022). As this compost breaks down over time, nitrogen is released in the soil for both plant and microbial use. Understanding the fate of this nitrogen is important as it could lead to changes in N₂O emissions, an important GHG 300 times more potent than CO₂ (IPCC, 2021). In addition to microbial and GHG responses, understanding plant responses to compost are important as land managers, both at farms and urban parks alike, focus on healthy, productive plants as an end goal. Compost can improve plant production through increases in nutrients such as nitrogen and phosphorus, as well as through general improvements in soil health (Cooper & DeMarco, 2023; Ryals et al., 2016). Improved plant productivity is important for both yield in agriculture and for park quality in urban spaces, and has important implications for feeding back into soil health. Increases in root productivity in particular can improve the soil microbiome and increase carbon storage, and plants with higher root biomass may utilize resources from compost application more effectively (Junaidi et al., 2018; Yang et al., 2023). Understanding how soil nutrients and soil biology response to organic matter amendment application together is vital not only for the end-goals of management, but for understanding how we can improve the health and resilience of a soil system.

This dissertation aims to assess how waste management strategies, chiefly compost use, impact urban soil health on various time scales and throughout the soil profile. Through work spanning from the computational to the greenhouse, this research focuses on changes in not only soil carbon amounts, but the composition and distribution of soil carbon changes with compost use. Traditional measures of soil health such as nutrient availability and soil greenhouse gas emissions are also studied. This work gives insights into the potential for organic matter amendments to improve

soil carbon storage and improve overall soil health, not only for urban systems, but for any managed systems considering organic matter amendment use.

Chapter two assesses the literature for the impacts of different organic matter amendments in various urban settings to understand where amendments may be best used and what soil property each amendment is best suited to improve. We discuss how compost, biochar, and biosolids are all important avenues for waste management that each have unique use cases. In addition, different urban settings (landscaping vs urban gardens, for example) have different needs and will respond differently when amended. We highlight research gaps in the growing space of urban soils research both to understand how to best manage urban soil health and to inform our future dissertation work.

Chapter three focuses on the impacts of compost and irrigation source on soil biology, examining both greenhouse gas emissions and changes in biomass of both grass and microbes. We discuss how these variables are impacted in the short term (3 months) around the interactions of compost application, watering type (recycled water or filtered), and baseline carbon levels. This study highlights how various management and baseline variables can be the controlling narrative on how compost applications may impact a soil system.

Chapter four explores how compost impacts an urban soil system over several years of application with a focus on whole system climate accounting. Here we looked at how soil carbon, nitrogen, and soil greenhouse gas emissions change to assess the carbon and nitrogen cycles holistically. To compliment these data points, we also looked at changes in microbial biomass and available nitrogen over time. This chapter compares compost's impact in a medium-term scale relative to the short-term and long-term view in chapters three and five, respectively, and serves as the field study companion to chapter three.

Chapter five dives into the impacts of long-term compost application on urban park soils, focusing on how carbon and nitrogen changes throughout the soil profile and in distinct carbon cycling pools. We discuss how carbon and nitrogen change up to one meter and focus on how the mineral fraction changes in these depths. We assess carbon concentration, carbon stocks, density fractionation, radiocarbon age, and organic matter composition. We also assess the impact that the park age and management have on soil properties, particularly in those that do not receive compost applications. This study is an essential companion to our previous chapters in understanding how repeated decadal applications of compost will impact soil carbon over time.

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Chapter 2: Impacts of Organic Matter Amendments on Urban Soil Carbon and Soil Quality: A Meta-analysis

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Abbreviations: BD = Bulk density, K = Potassium, N = Nitrogen, OM = Organic matter, P = Phosphorus

Abstract

Organic matter amendment application is an important avenue of beneficial waste diversion and is used to improve soil quality in agricultural and urban settings. In urban regions, amendments are used to support local food production, maintain vegetation for landscaping and recreational use, and reclaim disturbed soils. Urban regions generate large quantities of wasted organic resources for potential application aiding in creating a circular nutrient economy. There is a growing interest in understanding the effects of amendments such as compost, biosolids, and biochar on soil properties in agricultural settings. Gaps remain, however, in assessing their effects in urban land uses. We conducted a literature review to assess the effects of compost, biochar, and biosolids on soil carbon and soil quality of urban soils managed for gardening, landscaping, recreation, and reclamation. Application of organic matter amendments led to an average increase of 3.6% in soil organic matter (SOM). Compost and biochar improved SOM the most, by 3.1% and 6.5%, respectively. Biosolids resulted in the smallest increase in SOM% but had greater nutrient benefits than other amendments. Parameters related to chemical and physical soil quality improved with the application of amendments. Gaps in the literature remain, such as assessing urban gardens, soil to depths greater than 30 cm, and the persistence of SOM in amended soils. This meta-analysis proposes that organic matter amendments are a powerful means to improve soil quality in urban regions, providing vital cobenefits to surrounding communities, and increase soil carbon storage.

Keywords:

- Urban soil
- Urban soil quality
- Organic matter amendments

- Soil organic matter
- Soil carbon sequestration
- Meta-analysis

1. Introduction

Many forms of organic matter amendments, such as compost, plant residues, animal manure, and biosolids, are used to improve soils (Pane et al., 2015, Anwar et al. 2015). The addition of organic matter amendments to soil is an established practice to improve soil health and support plant production. However, our understanding of the effects of amendments used in urban settings is lacking when compared to agricultural use (Fitzpatrick et al., 2005). While used in both settings, amendment use is dominated by agricultural settings due to focuses on production and large-scale soil health movements (Hargreaves et al., 2008). In agriculture three of the most common organic matter amendments are compost, biosolids, and biochar. Biochar is made from the pyrolysis of feedstocks, typically woody and plant based. Biochar is well positioned for increasing soil carbon and for reducing leaching of nutrients and pollutants in soils (Albert et al., 2021; J. Wang et al., 2016). However, biochar's high C:N can also lead to the immobilization of nutrients such as nitrogen (Nguyen et al., 2017). Biosolids are akin to a faster releasing fertilizer, typically made from treated municipal solid wastewater, and are very high in nutrients such as nitrogen and phosphorus. Biosolids often strongly increase soil nitrogen and phosphorus, but increase soil carbon to a lesser degree (Sharma et al., 2017; Wijesekara et al., 2021). Composts are a middle ground of the three amendments and they can be created from many organic materials. The carbon and nutrient soil gains of compost fall between biochar and biosolids by achieving moderate gains in both categories, and there are also often large improvements in microbial biomass and structure (Adugna, 2018; Kallenbach & Grandy, 2011; Siedt et al., 2021). All of these amendments are used to improved soil quality and health broadly, but which may be best is context dependent. Despite varying contexts, increases in soil quality factors such as nitrogen and phosphorus by 20 - 100% or more have been measured in agricultural meta-analyses with organic matter amendments (Gravuer et al., 2019; Ma et al., 2021). When comparing these results to urban soils, similar impacts are often seen in individual studies, though there is far less study and specification, leading to uncertainty.

While much of the use of organic matter amendments is focused on soil fertility and plant production, there is recent movement to further organic matter amendments use to combat climate change specifically. From a climate perspective, organic matter amendments contribute to climate change mitigation by tightening the waste loop and sequestering carbon in the soil. Soils have a high capacity to store carbon – holding significantly more carbon than the atmosphere and terrestrial biomass combined (Scharlemann et al., 2014). In agriculture these amendments can have sizeable carbon benefits, biochar is technologically capable of one petagram CO₂eq per year globally and compost can regularly increases SOC by 45% or more

(Aguilera et al., 2013; Griscom et al., 2017). However, these SOC increases are not permanent sinks and these estimates commonly only assess agricultural and natural soils, leaving urban soils out of the picture.

Specifically, urban soils are an underutilized reservoir for organic matter amendments due to the sheer amount of organic waste generated in urban regions, the proximity of their end use, and the potential for large carbon gains. Urban soils have been degraded from the placement of impermeable surfaces, soil erosion during construction, and improper management (S. Brown et al., 2012). Due to this, urban soils often contain less carbon than natural spaces – though some urban soils are very well managed - but this gap could be closed with soil health management, such as applying organic matter amendments (Chien & Krumins, 2022; Pouyat et al., 2006). Use of amendments could help to provide healthier soils and better urban soil carbon storage, however, the extent to which urban soils could contribute to city, state, national, and international climate goals is not well constrained. These lands constitute a nontrivial area; the United States is over 3% urban and this figure is expected to rise to 8% by 2050, and 2-3% of land globally is urban (Z. Liu et al., 2014; Nowak & Walton, 2005). Illustrating the magnitude of carbon loss due to urbanization, approximately 0.21 Pg of the 1.72 Pg total soil carbon contained in current urban areas was lost in the southern United States from 1945-2007 due to rapid urbanization from in this region (Zhang et al., 2012). Losses like this could potentially be negated and turned into sinks with the application of organic matter amendments. Additional external losses of carbon could be mitigated by replacing non-sustainable amendments and horticulture products such as urban peat use with organic matter amendments generated from urban organic waste streams. Many commercial bagged amendments used in urban settings consist of this peat, which is an important carbon resource that is destructively harvested, but this could be displaced by organic matter amendments (Alexander et al., 2008; Fascella, 2015; Schmilewski, 2008). However, there has been recent shifts away from peat use, especially in areas of Europe that have focused on reductions in peat use and restoration of peatlands, to urban green waste composts instead (Andersen et al., 2017; Boldrin et al., 2010; Kazamias et al., 2017). These organic matter amendments are prime for urban use as large quantities of organic resources are generated by humans in urban areas - 20 kg of dry matter biosolids and 22 kg of dry matter food waste per capita annually – and could be transformed and applied directly as urban soil amendments (Kim et al., 2017; Parthiba Karthikeyan et al., 2018). Current research supports the benefits of organic matter amendments in urban systems, but estimates are not well constrained and studies are often limited to single amendment types or singular urban land-uses.

In addition to sequestering soil carbon, organic matter amendments can contribute to methane reduction from landfill avoidance and have important implications for circular resource economies. In California, landfilling is a common fate for organic wastes, which account for 41% of the state's human-generated waste (CalRecycle 2015). As a result, landfills are the largest point-source of methane emissions in the state (Duren et al., 2019). Several recent and emerging policies target organic diversion from landfills as methane reduction strategy (CA SB 1383, 2016;

Reynolds et al., 2020), and some of these policies include requirements for the procurement and use of compost in urban spaces (e.g. CA SB 1383, Statutes of 2016).

When diverted from landfills and recycled to the land, organic matter amendments give societal and environmental co-benefits that are not directly monetized, often making them less financially attractive compared to the immediate and traditional results of synthetic alternatives. The use of organic matter amendments in agricultural ecosystems have increased transportation costs and less immediate benefits compared to synthetic fertilizers, which can lead to financial barriers in using organic matter amendments, though some organic matter amendments are supported by subsidies to combat this (Harrison et al., 2020; Kan et al., 2010; H. Wang et al., 2008). However, when farmers invest in organic matter amendments they recoup increased upfront cost with long term soil health benefits and can aid in becoming organic farmers (Hijbeek et al., 2019; Horrocks et al., 2016; Viaene et al., 2016). In addition to organic matter amendment application to agricultural soils, application to urban soils may present an opportunity to further the use of organic matter and decrease costs. Organic matter amendment application to urban soils may also be more cost-effective and feasible due to the resources used to create amendments being generated in urban settings, even though processing of organic material may occur beyond the direct urban boundary (Lehec, 2020; Zurbrügg et al., 2004). However, processing outside this urban boundary can still result in efficient urban soil application, especially when the production of compost of biosolids is municipality based (McIvor et al., 2012). There are also instances of direct urban center processing such as the cost-efficient use of home composting or city park departments conducting their own composting processes (Morrow & Davies, 2022; Vázquez & Soto, 2017). By using a portion of organic matter amendments from urban waste resources for use in urban settings, regions can tighten nutrient cycles, decreasing transportation costs and emissions.

Urban organic matter amendments may also have additional societal benefits, such as affordable and local urban food production. Using organic matter amendments on preexisting urban garden soils can help increase food security through increased produce production (Barthel & Isendahl, 2013). Higher degrees of productivity can also be achieved when organic matter amendments are used to reclaim vacant and polluted urban soils for food production (Kumar & Hundal, 2016). However, there are concerns over contaminants for sites with dangerous prior uses, such as leaded gasoline and paint exposure (Imperato et al., 2003; Laidlaw et al., 2017). Remediation and monitoring of these lands using organic matter amendments could increase urban food production, lowering the need for imported foods and providing significant shares of the food needed in urban areas, particularly in lower-income communities and countries (Algert et al., 2014; Badami & Ramankutty, 2015). Thus, organic matter amendments used in urban agriculture will contribute to lower greenhouse gas emissions from imported food and increased food security in addition to their soil benefits (Kulak et al., 2013).

While much is known about organic matter amendments benefits in agricultural settings and their potential social impacts, the study of their use and

impact on soils in urban settings is less developed, and there is a lack of a comprehensive meta-analysis. This meta-analysis synthesized current literature on organic amendment use in urban environments and quantitatively assesses the impacts of their use on soil properties. The questions this meta-analysis focused on are: **1.** To what degree do organic matter amendments affect urban soil carbon? Furthermore, which amendments are the most effective at increasing soil carbon stocks? **2.** To what degree do organic matter amendments improve urban soil quality indicators? Which amendments are the most effective at improving soil quality? **3.** What literature gaps remain in the study of these amendments in urban settings? In assessing these questions this paper quantitatively synthesizes the impacts of various organic matter amendments on soil properties, discusses the co-benefits for applying organic matter amendments, and highlights the knowledge gaps that require additional research focus.

2. Materials and Methods

2.1 Data Compilation and Literature Review

Data on soil responses to organic matter amendments in urban settings were compiled searching for peer-reviewed published works primarily via Google Scholar, Web of Science, African Journals Online, and Dialnet databases up until July 2023. Urban environments were defined as areas dominated by anthropogenic influences in urban and suburban spaces with relatively high population density and impervious surface cover, primarily excluding soil ecosystems such as rural agriculture and natural forests (Hobbie & Grimm, 2020). Keywords used in the literature search focused on urban soil + amendment type (Table S2-1) and forward and backward searches used these keywords as well. Studies were selected using the criteria below: **(1)** Included one or more organic matter amendment (compost, biosolids, biochar, or a combination); **(2)** Reported an urban environment or urban soil used for ex-situ studies; **(3)** Included data from one or more variables of interest, carbon (C), nitrogen (N), phosphorus (P), potassium (K), bulk density (BD), or pH (Table S2-2); and **(4)** Included a control urban soil in conjunction with amended soils within the study. Soil organic carbon was assessed (excluding inorganic carbon), and soil P and K data were classified as “extractable” and/or “available” while soil N was typically collected as total N. Additional properties such as soil microbial abundance, communities, and dynamics are important aspects of urban soil health that could be affected by organic matter amendments (Kranz et al., 2020; Oldfield et al., 2014; Sax et al., 2017). However, there was insufficient data on biological indicators of soil health to include in the meta-analysis and this is why we henceforth refer to overall changes in soil quality instead of changes in soil health.

2.2 Data Organization and Calculation

We found 50 studies in our literature assessment that fit our inclusion criteria, with a total of 322 reported sample values, 104 control and 218 treatment values.

Studies had four to five treatments on average with most studies including a control and more than one treatment, varying based on amendment types, application amounts, soil textures, and/or settings. From this, studies on average held 3 to 8% of our study's total samples for each soil property overall. The exception is for potassium which had only 13 studies reported, leading to studies contributing greater than 8% of the total potassium data on average (Table S2-3). The share percentages presented in Table 1 represent the percent of the total sample pool that any one study contained. Thus, despite one study containing 29% of an interested property's data (Bulk density), there is still independence in measurements due to variations in treatments within larger studies.

Table 2-0-1: ANOVA and regression results for models of soil properties by both urban land use and treatment. N refers to total data points across studies. Level of significant is: ns: $p > 0.05$, *: $p \leq 0.05$, **: $p \leq 0.01$, ***: $p \leq 0.001$, ****: $p \leq 0.0001$.

Soil Property	Independent Variables	Sample Size	Study Size	Adjusted R ²	Significance
SOM%	Setting	223	38	0.15	****
	Amending	223	38	0.10	****
	Setting + Amending	223	38	0.26	****
N (g/kg)	Setting	189	31	0.0	p=0.36
	Amending	189	31	0.04	**
	Setting + Amending	189	31	0.04	*
Total P (mg/kg)	Setting	142	24	0.0	p=0.55
	Amending	142	24	0.06	**
	Setting + Amending	142	24	0.06	*
Total K (mg/kg)	Setting	51	13	0.01	p=0.31
	Amending	51	13	0.18	**
	Setting + Amending	51	13	0.22	**
Bulk Density (g/cm ³)	Setting	149	21	0.22	****
	Amending	149	21	0.18	****
	Setting + Amending	149	21	0.41	****
pH	Setting	147	33	0.08	**
	Amending	147	33	0.0	p=0.57
	Setting + Amending	147	33	0.07	**

For regression models, urban land-uses were placed into four categories: gardening, reclamation, recreation, and landscaping. Gardening consists of urban gardens, arboretums, and a horticulture center. Reclamation consists of roadside plots, vacant lands, and former industrial sites. Recreation consists of urban parks, while landscaping consists of managed areas adjacent to buildings, passive turfgrass fields not used for recreation, and pot trials using urban soil (Figure 2-1B). Pot trials were placed in this category as these studies used soil from landscaping settings and aimed to use amendments in landscaping scenarios. These land-use categories were designated to help distinguish the impacts on unique soil systems with different baselines and management goals. In addition to land-use type, a geographic distribution of studies was assessed using ArcGIS (Figure 2-1A).

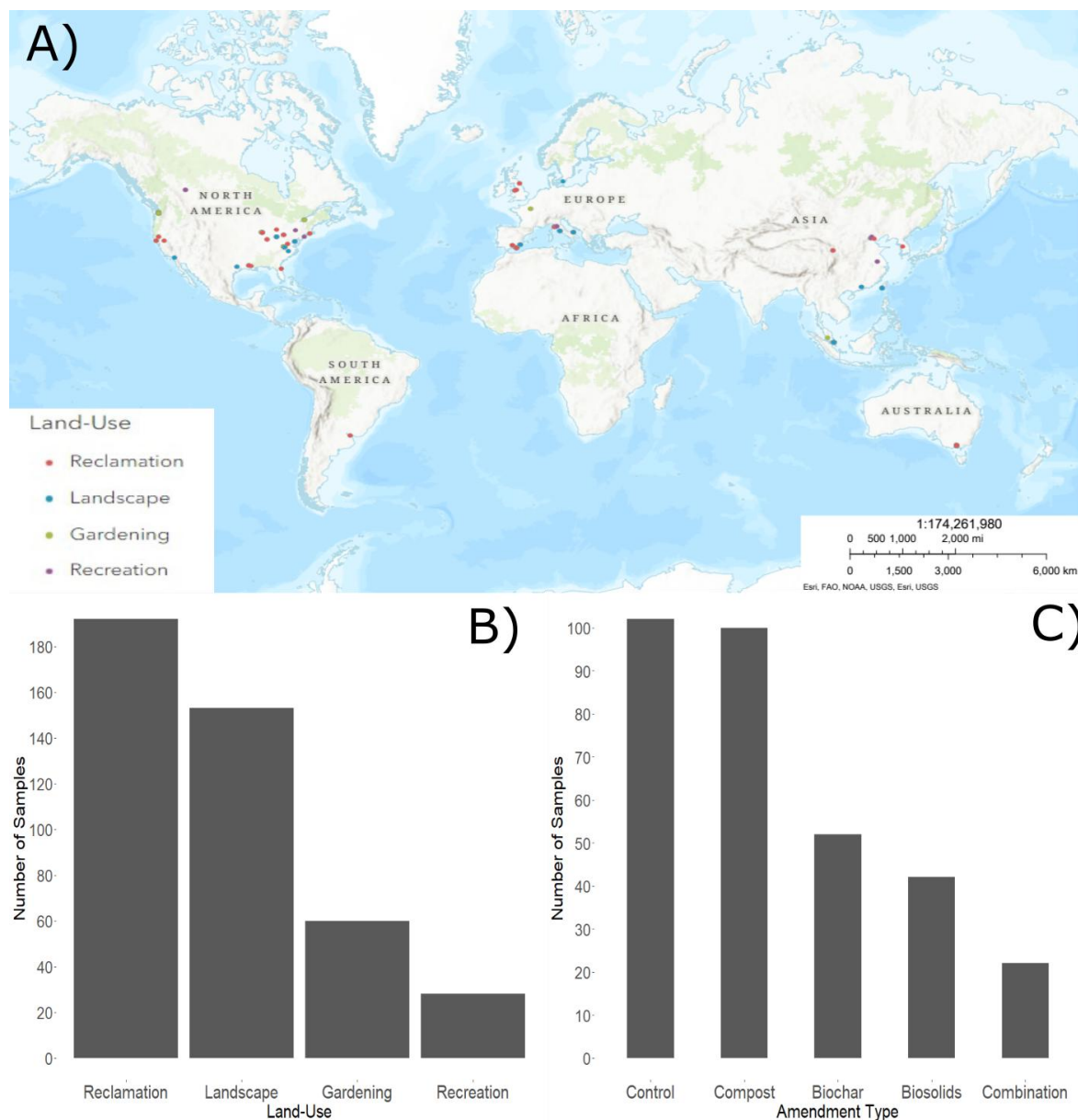


Figure 2-1: A) Geographic distribution of the samples used in this study. B) List of the total number of reported sample values for each amendment type. C) Number of samples present in this meta-analysis based on land-use of urban site.

Regression models were run on different amendment types, such as compost, biochar, biosolids, and combination to compare varying amendment impacts. The compost category was dominated by compost studies of green waste and also included composted biosolids. Biosolids include samples labeled as biosolids and sewage sludge, primarily from municipal wastewater treatment. Biochar is composed of woody waste with samples identified as biochar, fine char, and coarse char. The combination category included studies that looked at a combination of the above treatments, consisting of compost and biochar or compost with sewage sludge (Figure 2-1C).

Units used to report results were not consistent across all studies. Therefore, we converted to common units whenever possible. For example, conversions occurred if soil organic carbon percent (SOC%) was reported but not SOM%. SOM% was used in calculations due to its dominance in the respective literature and in its data availability. This conversion was done by multiplying SOC% by 1.72, which while not an exact conversion for all soils, can provide a baseline for the few studies that required conversion (Allison, 1965; Pribyl, 2010). In addition, soil carbon can vary based on measurement method, in our case elemental analysis was primarily used, but soil carbon can also be measured in other methods such as loss on ignition, which can cause discrepancies between studies, especially for biochars produced at high temperatures.

Calculations for estimating the technical potential for urban soil carbon sequestration at the global scale were made with several assumptions: **(1)** 8-26% of diverted organic wastes could be applied to urban lands in the United States (Harrison et al., 2020); **(2)** There are 146 million tons of municipal solid waste in the US and 51.4% is organic (US EPA, 2017). **(3)** Compost is assumed to be the organic matter amendment applied, at approximately 76 Mg/ha to increase SOM% by 1.2 and Mg/ha by 22.2 – the most efficient rate to increase SOM in our study when including associated bulk density changes (Figure 2-4); **(4)** Approximately 30% of carbon is lost from municipal solid waste during composting (H. Guo et al., 2019); **(5)** Soil carbon changes from amendments are focused in the top 30cm (no studies included this meta-analysis go beyond 50cm, most stop at 20 to 30cm); and **(7)** OM% is approximately 58% carbon (Allison, 1965).

2.3 Statistical Analyses

R software was used for statistical analyses and visualization (R Core Team, 2020). Analysis of variance (ANOVA) and linear regressions were used to assess statistically significant relationships. Mean values are reported along with $\pm$ standard error. An alpha level of 0.05 was given for significance in ANOVA testing. Visualization was conducted in R using the “ggplot2” and “ggpubr” packages in R for plots (Kassambara, 2020; Wickman, 2016, p. 2). Effect sizes were calculated using

the “metafor” package for calculating and presenting forest plots (Viechtbauer, 2010). The effect size used was the log ratio of the means of treatments and controls. Only studies that reported a standard error or standard deviation were included in this assessment, as a standard deviation is necessary for calculation. If only the standard error was reported, the standard deviation was calculated by multiplying the standard error by the square root of the sample size (Altman & Bland, 2005).

Partial dependence plots were created to evaluate the effects of application amount on soil properties. The “pdp” package was used to create the partial dependence plots shown in this paper. Partial dependence plots allowed the assessment of the marginal effect of prediction using the partial function. This function is useful for assessing how changes in organic matter amendment application amount influenced results. Application rate data in the literature were reported in both Mg/ha and as a percentage of the total soil profile with Mg/ha being used in this paper as it is a more standard metric. Percent amendment application conversions to Mg/ha were made when possible using measurements such as bulk density, soil area, and volume.

3. Results

3.1 Sample Overview

Studies included in this analysis encompassed a wide range of amendment types, land-uses, and geographic locations. Compost was the most common amendment type followed by biosolids and biochar (Figure 2-1C). These amendment types also varied in their C:N with an average value of 28, 10, and 108 for compost, biosolids, and biochar respectively (Table S2-4). The most common land use category was reclamation followed by landscaping. After this there is a sharp drop off in reported samples assigned to the remaining two categories: gardening and recreation (Figure 2-1B). Studies were not uniformly distributed geographically; North America and Europe dominated reported studies with several studies present in Asia (Figure 2-1A). International journals were searched for studies in Africa and South America but only one study were found to fit our inclusion criteria and/or to be fully available. Studies also varied by soil texture with sandy loam soils being the most common followed by loam, silt, and clay soils (Table S2-5). Overall, 79% of our samples had a reported soil texture (253 of 322).

Missing values in the dataset were reported to assess gaps in the literature data. To assess literature gaps, only rows that contained at least one of the following properties were included in this statical analysis: SOM%, total N, P, K, pH, or bulk density. SOM% was the most reported value in the literature, followed by the remaining factors except for soil K, which had a significantly higher missing percentage of data than the other soil properties reported (Figure S2-1). In addition to missing reporting of some soil properties, missing depth assessment (> 30 cm) is present in the literature as well. The majority of studies used for analysis assess soils to 30 cm or less, with few going to 50 cm, and none deeper. Studies were limited

temporally as well as studies on average lasted only two to three years and were concentrated in the last 10 years (Figure S2-2).

3.2 Soil Carbon

Soil carbon measurements were reported in a total of 38 studies. Prior to the addition of organic matter amendments, SOM% averaged $3.5\% \pm 0.4$ ($n = 67$) across all land use categories, while SOM% averaged $7.1\% \pm 0.4$ for amended samples ($n = 156$; $P < 0.05$). When amended, the effect size of SOM% was 0.7 ± 0.4 (study $n = 19$; Figure 2). SOM% varied by both amendment type and the land use of application ($P < 0.05$). All amendment types increased SOM% with the largest shifts seen in compost and biochar ($P < 0.05$) with a smaller increase due to biosolids ($P > 0.05$; Table 2). Differing application amounts also caused variances in SOM% as well as by amendment type in studies. Increased application amount increased SOM% for all amendment types, until between 300 and 400 Mg/ha, after which SOM% begins to increase much more slowly (Figure 2-4). Both biosolids and biochar have strong increases in SOM% with application amount, with biosolids plateauing before biochar and biochar starting from a higher SOM% baseline (Figure 2-4). Land use category differences in SOM% are minimal in the control except for gardening, which has a higher baseline SOM% (Figure 2-3). The largest increases in SOM% were observed in the landscape, recreation, and reclamation by land use category, with a smaller increase in gardening. (Figure 2-3).

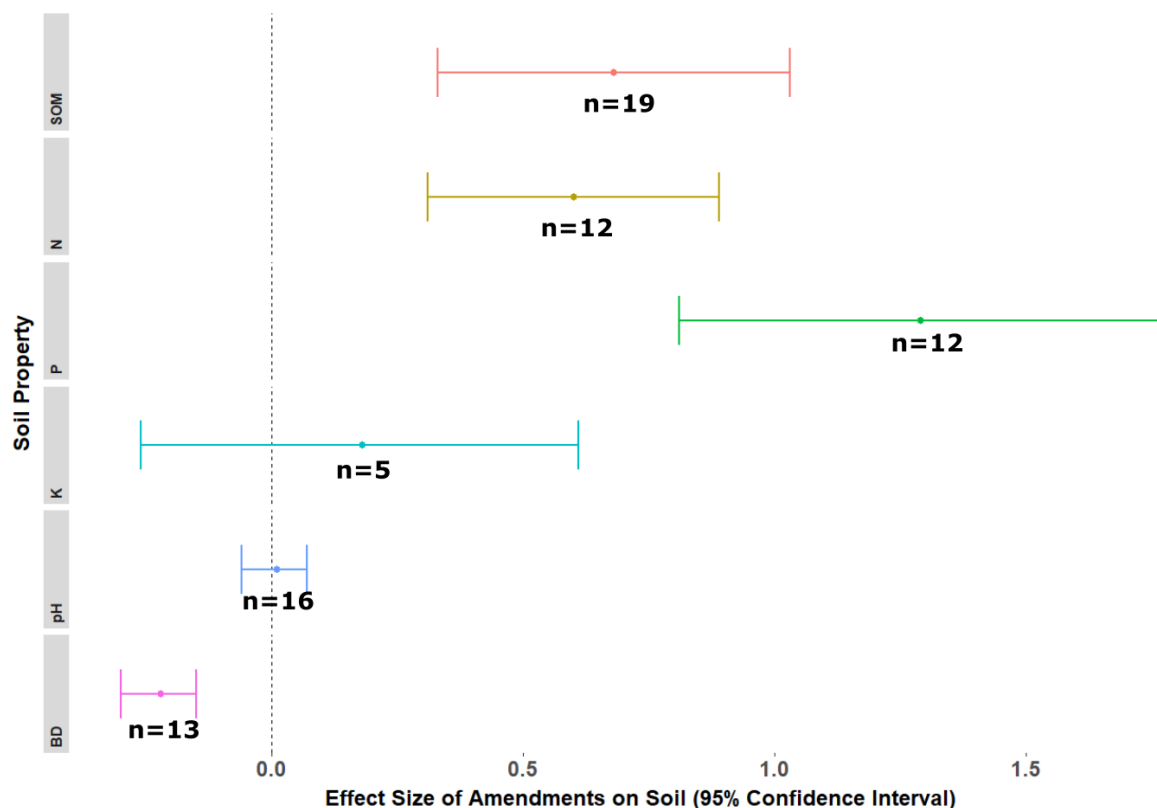


Figure 2-2: Forest plot of log ratio effect size of amendments on different soil properties. Upper and lower bounds represent the 95% confidence interval. N

represents the number of studies in the forest plot for that soil property.

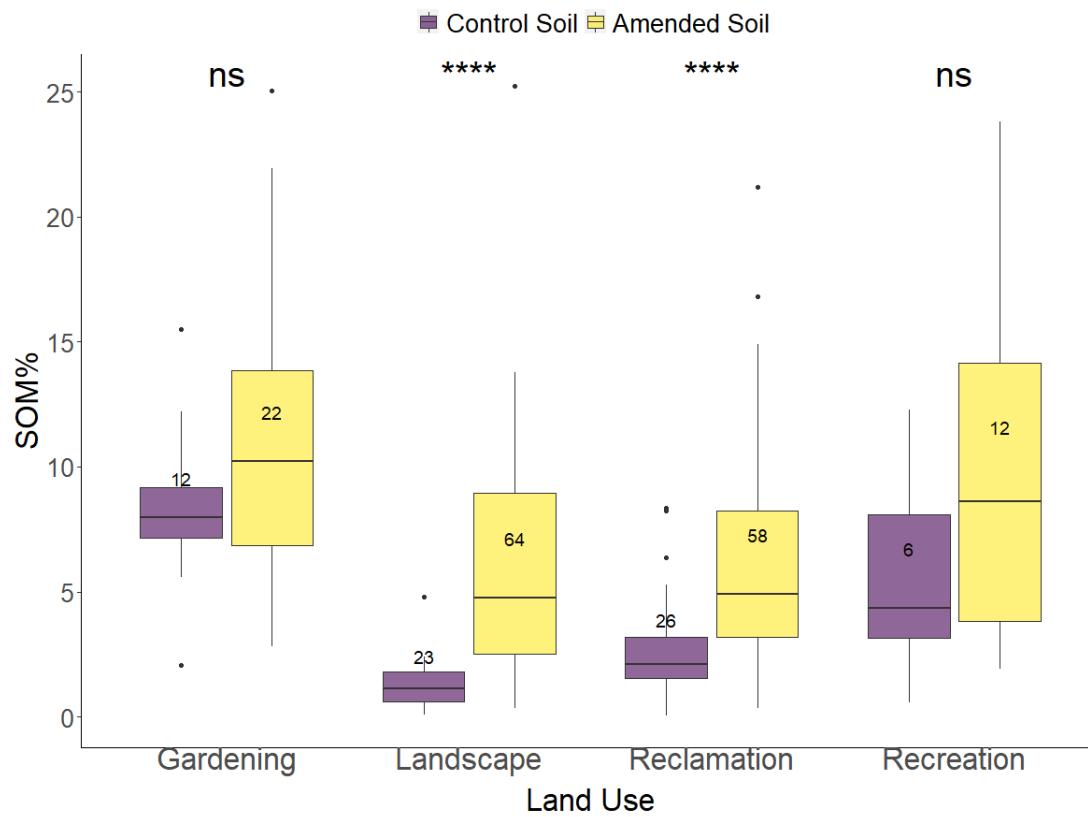


Figure 2-3: Box plots of SOM% by land use category, with control in purple and amended sites in yellow. Number represent sample size. Level of significant is: ns: $p > 0.05$, *: $p \leq 0.05$, **: $p \leq 0.01$, ***: $p \leq 0.001$, ****: $p \leq 0.0001$.

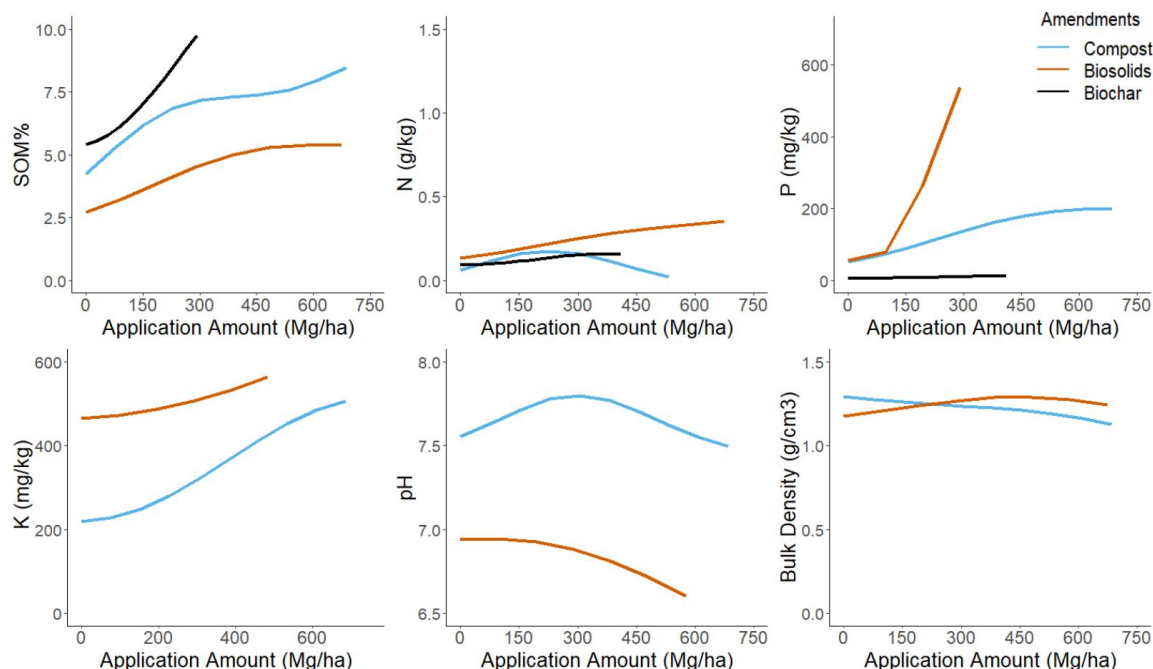


Figure 2-4: Partial dependence plots of amendment type and application amount (Mg/ha) on soil properties.

3.3 Soil Total Nitrogen

Soil total nitrogen measurements were reported in 31 studies. Soil total nitrogen in controls averaged $1.4 \text{ g/kg} \pm 0.2$ ($n = 56$) and $3.1 \text{ g/kg} \pm 0.4$ for amended samples ($n = 133$; $P < 0.05$). The effect size for nitrogen presents increasing nitrogen with amending with an effect size of 0.6 ± 0.3 (study $n = 12$; Figure 2-2). Nitrogen varied by amending ($P < 0.05$) but not significantly by land use category overall ($P > 0.05$; Table 2-1). All amendment types increased total soil nitrogen, with larger shifts due to compost and biochar ($P < 0.05$; Table 2-2) compared to biosolids ($P > 0.05$). Varying application amounts influenced nitrogen levels with increasing application amounts sharply increasing nitrogen for biosolids with smaller increases and a short plateau for compost and biochar (Figure 2-4). Soil total nitrogen in controls were relatively similar across the land use categories landscaping, reclamation, and recreation (means = 1.0 g/kg , 0.9 g/kg , 1.7 g/kg , respectively). Organic matter amendments increased total soil nitrogen significantly in these land use categories (Figure 2-5). In contrast, no significant differences in total soil nitrogen were observed within the gardening land use category, which had significantly higher soil total nitrogen in controls compared to other land use categories ($P > 0.05$, mean = 3.1 g/kg ; Figure 2-5).

Table 2-2: ANOVA relationships between amended and control plots for all soil properties reported. Sample size indicates amended and control samples of a treatment type. N refers to total data points across studies. Level of significant is: ns: $p > 0.05$, *: $p \leq 0.05$, **: $p \leq 0.01$, ***: $p \leq 0.001$, ****: $p \leq 0.0001$.

Soil Property	Amendment Type	Sample Size	Study Size	R	Significance
SOM%	Compost	129	31	3.05	***
	Biochar	58	12	6.49	***
	Biosolids	30	8	2.14	p=0.29
N (<i>g/kg</i>)	Compost	90	23	2.45	*
	Biochar	56	8	1.72	*
	Biosolids	34	8	1.25	p=0.13
Total P (<i>mg/kg</i>)	Compost	64	18	33.39	p=0.22
	Biochar	52	7	44.13	p=0.28
	Biosolids	22	7	403.61	**
Total K (<i>mg/kg</i>)	Compost	27	10	120.75	p=0.09
	Biochar	14	4	127.59	p=0.16
	Biosolids	3	1	193.50	p=0.08
Bulk_Density (<i>g/cm³</i>)	Compost	88	19	-0.25	****
	Biochar	20	5	-0.33	****
	Biosolids	23	5	-0.24	p=0.23
pH	Compost	92	28	-0.04	p=0.82
	Biochar	26	8	0.60	p=0.11
	Biosolids	22	7	-0.33	p=0.32

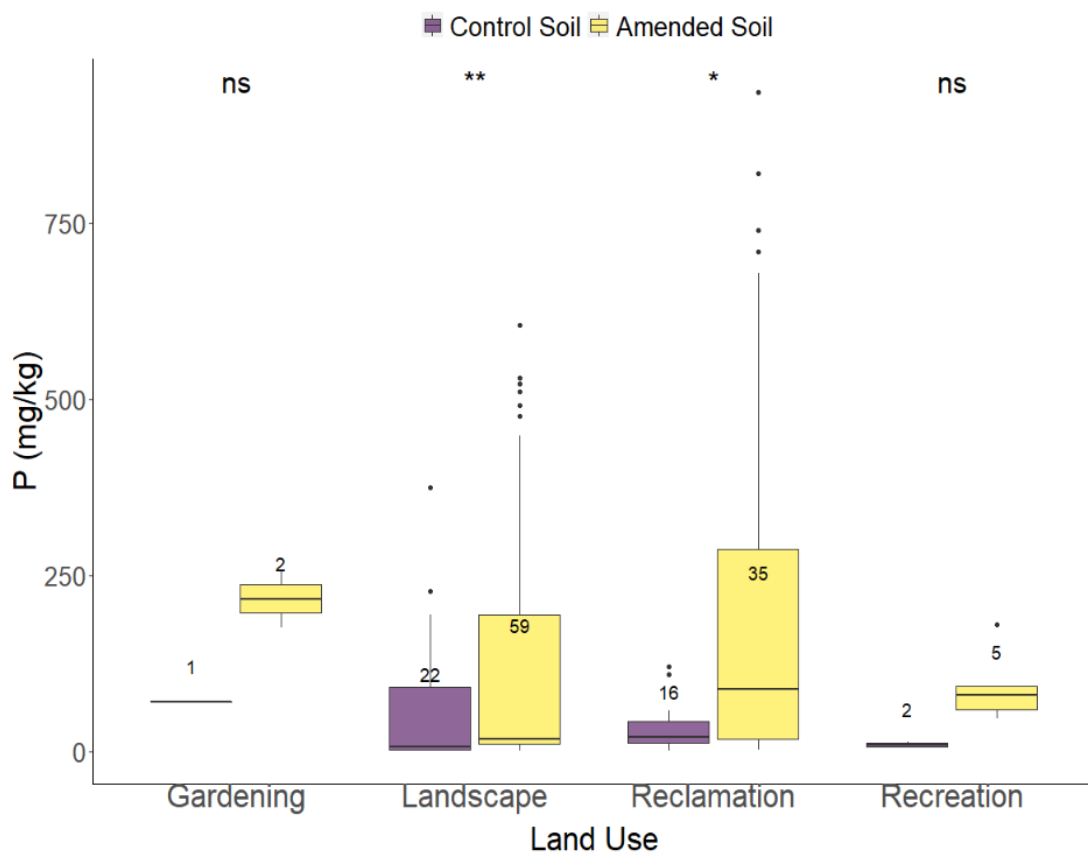
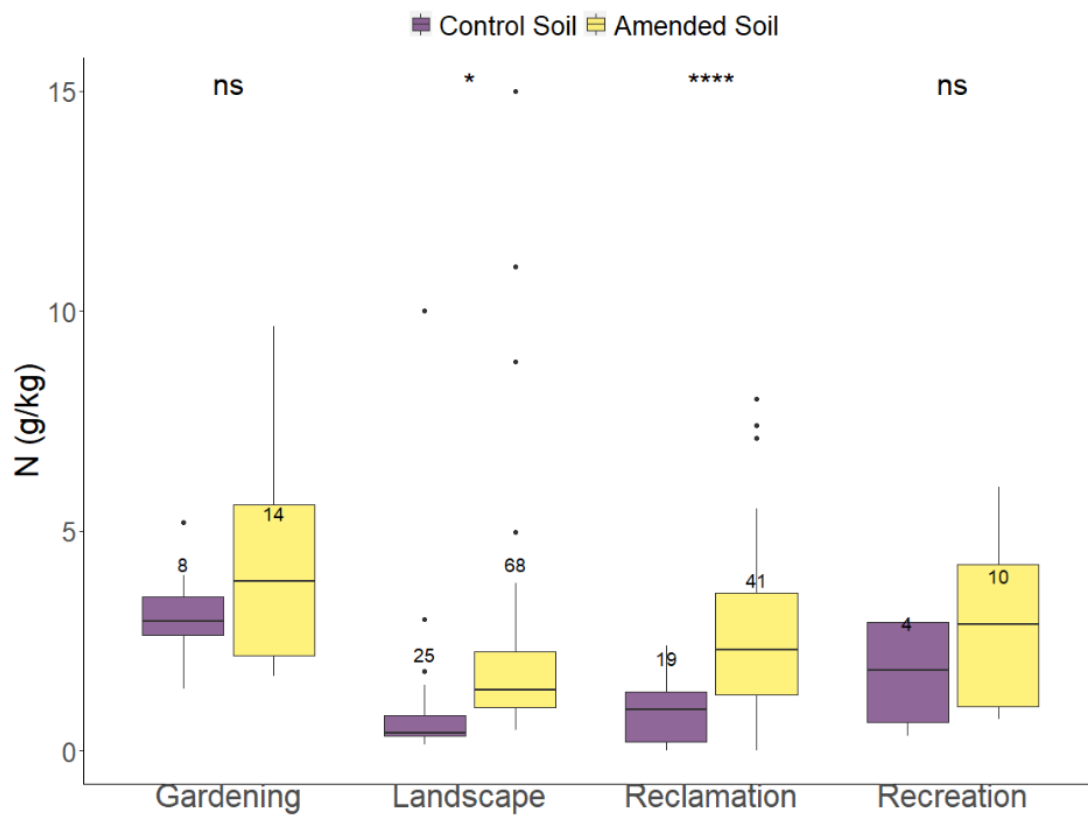


Figure 2-5: Box plots of N (**g/kg**) and P (**mg/kg**) by setting with control in purple and amended in yellow. Number represent sample size. Level of significant is: ns: $p > 0.05$, *: $p \leq 0.05$, **: $p \leq 0.01$, ***: $p \leq 0.001$, ****: $p \leq 0.0001$.

3.4 Soil Phosphorus

Soil phosphorus was reported in 24 studies. Prior to the addition of organic matter amendments soil phosphorus averaged $49.4 \text{ mg/kg} \pm 12$ ($n = 41$) while it averaged $156 \text{ mg/kg} \pm 21$ for amended samples ($n = 101$; $P < 0.05$). The effect size for phosphorus showed increasing phosphorus with amending with an effect size of 1.3 ± 0.5 (study $n = 12$; Figure 2). All amendments led to increases in phosphorus, with an order of magnitude increase in biosolids amended soils ($P < 0.05$; Table 2), compared to compost and biochar ($P > 0.05$). Application amount sharply increased soil phosphorus for biosolids with much smaller increases for compost and little increase from biochar (Figure 2-4). When amended, all land-uses saw sizeable increases in soil phosphorus, though only landscaping and reclamation sites were statistically significant (Figure 2-5).

3.5 Soil Potassium

Soil potassium was reported in 13 studies. Prior to the addition of organic matter amendments soil potassium averaged $130.6 \text{ mg/kg} \pm 32$ ($n = 19$) while it averaged $301.2 \text{ mg/kg} \pm 33$ for amended samples ($n = 32$; $P < 0.05$). The effect size for potassium concentration showed increasing potassium with amending with an effect size of 0.2 ± 0.4 , but a study size of only five and wide range (Figure 2-2). Potassium concentrations increased with amending ($P < 0.05$) but did not vary significantly by site type ($P > 0.05$; Table 2-1). All treatments increased potassium, with the largest increases in biosolids ($P > 0.05$) and compost ($P > 0.05$), compared to biochar ($P > 0.05$; Table 2-2). Application amount also influenced potassium concentration change, sharply increasing potassium for compost with smaller increases for biosolids (Figure 2-4). Potassium concentrations increase in all reported sites, with reclamation having the largest increases (Figure S2-4). While all potassium concentrations increase, it is important to note that potassium levels in soil can be very feedstock and ecosystem dependent due to its high solubility (Mikkelsen, 2007).

3.6 Soil Bulk Density

Bulk density was reported in 21 studies. Prior to the addition of organic matter amendments soil bulk density averaged $1.5 \text{ g/cm}^3 \pm 0.04$ ($n = 55$), while it averaged $1.2 \text{ g/cm}^3 \pm 0.03$ for amended samples ($n = 94$; $P < 0.05$). The effect size for bulk density was -0.2 ± 0.1 , signaling decreasing density with amending (study $n = 13$; Figure 2). In addition to amending, bulk density varied by setting ($P < 0.05$; Table 2-1). All amendments lower bulk density but only compost and biochar decreased bulk density significantly ($P < 0.05$; Table 2-2), with the largest decrease due to biochar. Bulk density steadily decreased with increasing application of compost with less change with application of biosolids (Figure 2-4). When assessed by land use category, gardening had the lowest bulk density in both treatment and control, with all sites showing decreased bulk densities in the amended sites (Figure 2-6).

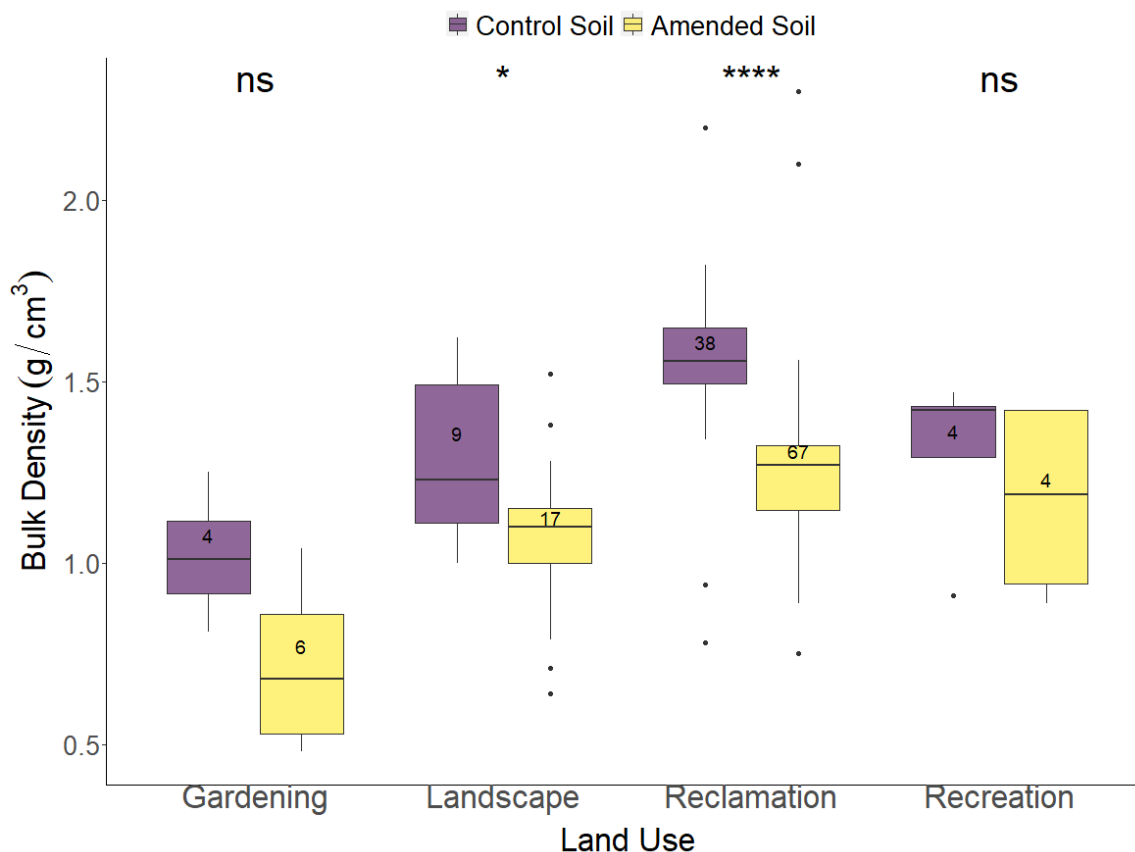


Figure 2-6: Box plots of bulk density (g/cm³) by setting with control in purple and amended in yellow. Numbers correspond to sample size. P-value level of significant is: ns: $p > 0.05$, *: $p \leq 0.05$, **: $p \leq 0.01$, ***: $p \leq 0.001$, ****: $p \leq 0.0001$.

3.7 Soil pH

Soil pH was reported in a total of 33 studies. It is important to note that pH was measured predominantly in a water solution but also in a CaCl_2 and potentially a KCl solution (as several studies did not mention pH methods), which could lead to variation in measurements between studies. For controls, soil pH averaged 7.2 ± 0.2 ($n = 50$), while it averaged 7.3 ± 0.1 for amended samples ($n = 97$; $P > 0.05$). The effect size for pH was minimal at 0.01 ± 0.1 , consistent with the changes overall between control and amended pH (study $n = 16$; Figure 2-2). This is due to contrasting effects of amendments such as compost and biosolids (Table 2-2); pH increased consistently by application amount of compost until around 400 Mg/ha while there was a consistent decrease when applied with biosolids (Figure 2-4). pH varied strongly by site ($P < 0.05$), but weakly by amending overall ($P > 0.05$; Figure S2-4).

4. Discussion

4.1 Urban Soil Carbon

Urban soils are understudied for carbon sequestration, especially compared to agricultural and natural ecosystems (Chien & Krumins, 2022). Natural and agricultural lands are commonly thought of in a carbon context, while this is not often true for urban soils (Lorenz & Lal, 2015; Vasenev et al., 2013). These soils, however, have the potential to store large quantities of carbon given proper management strategies (Brown et al., 2012; Pouyat et al., 2006). We found that organic matter amendments increase SOM% by an average of 3.6% SOM across a wide range of land uses, climates, and edaphic characteristics. While this represents a doubling of SOM concentration across amending, soil BD must be considered when assessing soil carbon change (Lee et al., 2009; VandenBygaart & Angers, 2006). When accounting for changes in both BD and SOM% from amending, we found that soil carbon storage increased by 60%. This effect represents a sizeable increase in urban soil carbon that could be used for climate change mitigation.

It is important to note that soil carbon increases varied by the applied amendment type applied. The greatest increases in soil carbon resulted from biochar applications, followed by compost, and then biosolids (Table 2-2). This finding is consistent with the nature of these materials as biosolids have lower carbon concentrations than both compost and biochar, and is a faster-releasing amendment than the two as well (Lima et al., 2009; Scharenbroch et al., 2013). Overall, having this explicate data from aggregated studies helps to properly assess which amendment is best to use based on the goals of application and to support prior assumptions of soil impacts from these amendment types.

Despite representing the largest impact on SOM, biochar may be more difficult to produce directly from urban settings. Biochar is often created with woody agricultural “waste” materials which are currently less utilized in urban settings – though this woody share currently constitutes 10% of the organic urban waste share (Zhao et al., 2019). However, biochar can also potentially be produced from sewage sludge treated with gasification or pyrolysis, a potentially plentiful feedstock for urban application (CalRecycle 2015). A biochar conversion pathway for sewage sludge represents an emerging strategy to organic resource recovery and reuse that could benefit urban soils. In using urban materials directly, fractions of green waste and sewage sludge could be used to create biochar instead of compost, diversifying the organic matter amendment pool. Despite this, creation of biochar may prove difficult as large scale infrastructure for biochar production is currently nascent and compost may confer more soil quality benefits, specifically with available nitrogen and phosphorus, making compost more attractive (J. Liu et al., 2012; Sánchez-Monedero et al., 2019). Biochar may prove to be better for some urban sites that have high fertility and are focusing specifically on improving soil carbon or can simply act as a smaller portion of the urban waste conversion puzzle due to the logistics of composting versus biochar.

Materials to create compost are plentiful in urban regions as cities generate large amounts of green waste, food waste, and biosolids (CalRecycle 2015; Gomez et al., 2008). With some regions focusing on increasing compost creation and use in agriculture, competition could arise between urban and rural users in using this waste resource to create compost. However, a recent study assessed the example of California's SB 1383 policy and found that at least 8-26% of California's diverted organic waste could then be applied to urban lands without compromising the ability for agriculture to substantially tap into this compost resource (Harrison et al., 2020). This leaves a sizeable portion of the total resource to apply in Californian urban regions — 6.3×10^5 to 1.9×10^6 metric tons of compost — which could be applied to urban Californian soils, despite the high demand for compost in the agriculturally intense state (Harrison et al., 2020). Less agriculturally intense states may have less of a need to distribute organic matter amendments to agricultural sites, leaving even higher shares of their region's organic waste available to be transformed into organic matter amendments for urban areas. Utilizing this share of urban waste as compost in urban areas specifically would help to increase soil carbon efficiently, reduce waste loads, and reduce harmful indirect emissions such as those from landfills and transportation (Adhikari et al., 2010; Keng et al., 2020). These benefits would be plentiful as large amounts of waste would be converted to compost and applied with large gains in SOM% up to 400 Mg/ha (Figure 2-4). High amounts of carbon from composts are soaked up by these urban soils when applied reaping strong benefits - such as increased urban soil fertility, improved urban food production, and climate change mitigation.

Biosolids offer modest increases in SOM% but represent another common resource in urban systems. While their use as a carbon sequestration technique may be reduced compared to compost and biochar, they present an opportunity to reuse human waste and increase soil nutrients. Using and treating biosolids can also help to improve sanitation standards in many regions and improve the ecosystem services of amended locations (Trimmer et al., 2019, 2020). In addition to addressing important societal inequalities, land application of biosolids have the largest increases in soil phosphorus and potassium of any of the three major amendments in urban soil systems (Table 2-2). However, biosolids are often low in K and usually only garner improvements in soil N and P, so biosolid K gains could be an artifact of low urban biosolids K sample size (S. Brown et al., 2020). Nevertheless, by increasing nutrients, biosolids are useful in combination with other treatments or in soils limited by additional nutrients for carbon sequestration (Awasthi et al., 2017; Kirkby et al., 2014). In a goal to maximize SOM% while achieving some of the benefits of biosolids use, one could use composted biosolids which would add less available nitrogen but more carbon than uncomposted biosolids (Boen & Haraldsen, 2011; McIvor et al., 2012).

One important detail when assessing organic matter amendments influence on SOM% is that not all changes in SOM may come from the direct amendment itself, but through indirectly influencing other processes. Increases in plant productivity from amending may increase inputs to the soil and increase the SOM pool over time (Bolan et al., 2013; McClelland et al., 2022). In addition to this, other indirect

mechanisms of organic matter amendment application can influence the SOM pool such as negative priming from biochar application reducing carbon mineralization rates (Jones et al., 2011; Reed et al., 2017). It is important to acknowledge that impacts to the soil from organic matter amendments are more complex than just the direct input of carbon and nutrients from the amendments themselves.

Among urban land-uses, the largest shifts in SOM% were seen in reclamation and landscaping, partially due to having the lowest baseline values (Figure 2-3). Reclamation sites have the most to gain as they consist of degraded soils such as roadsides and vacant lands. Similarly, amendments in landscaping contexts increase carbon, potentially due to degraded lands nearby buildings and construction. Lower relative increases in carbon occur in gardening and recreation sites likely due to prior management strategies. For example, urban gardens are often sustainably managed for food production leading to higher baseline carbon levels, while recreation sites may be managed with nutrient and water additions to ensure parks remain green. Regardless, all sites improved soil carbon when organic matter amendments are applied.

Over all amendments and land-uses, the average reported increase in SOM% across studies was 3.6% in SOM%, with an average application of once over two to three years. To assess large scale changes in soil carbon more analysis is needed, however. In calculating these large scale changes in carbon storage we assumed 8-26% of urban municipal waste was transformed into an organic matter amendment and applied to urban soils annually (Harrison et al., 2020). This 8-26% would correspond to 22,191 to 72,125 ha of urban application when comparing to the application rate of the most efficient carbon increase, 76.2 Mg/ha for 1.2 SOM% increase, and the total US municipal solid waste load of 146 million tons, with it being 51.4% organic (US EPA, 2017). If amendments were to be widespread applied on urban soils this increase in carbon in the top 30 cm could represent an increase of 1.1×10^{-3} Pg C to 3.6×10^{-3} Pg C, in the United States, equivalent to 0.08% to 0.26% of the United States' emissions in 2020 (US EPA, 2022). This would stem from an average increase of 22.2 Mg of carbon per hectare after 76.2 Mg/ha of amendment application. Globally, in addition to soil carbon sequestration, there is a large potential for avoided landfill emissions by transforming organic waste into organic matter amendments, as landfill emissions constitute up to 8-11% of global greenhouse gas forcing (Duren et al., 2019; Saunio et al., 2016). These numbers present untapped potential for carbon sequestration and carbon management in urban areas and policy measures should take this into account for regional and global carbon action plans. However, further study is needed on the longevity of the SOM increases due to these amendments in urban systems as most studies focus on short term single application of amendments. With focuses on short term application of amendments and the limit of 8-26% of urban municipal waste being used as an organic matter amendment, applying these amendments to urban regions would still be a multi-year project if we were to apply to all urban soils, leading to gains over time. Even if the same locations we reapplied annually or after several years there would be additional avoided landfill emissions, abated inorganic fertilizer emissions, and further soil carbon benefits, though reapplication may reduce the soil carbon increase due to

carbon saturation limits (Stewart et al., 2007). However, these limits have largely not been reached as non-permafrost soils average holding 42% and 31% of their mineral carbon capacity in the top and subsoils respectively, with the highest carbon sequestration rates in soils at one-tenth of their carbon capacity (Feng et al., 2014; Georgiou et al., 2022). Thus, prioritizing the use of organic matter amendments for restoring and utilizing vacant and degraded urban land plots is the strongest path forward for increasing soil carbon.

While these urban lands in the United States represents only 3 to 5% of the country's total area, this figure will increase to 8% by 2050, and the potential C that could be stored in urban soils and the organic waste used to do so will continue to increase (Nowak & Walton, 2005). As the urban footprint grows it will be important to do so sustainability, such as by maintaining soil carbon and soil health and by utilizing organic resources such as food losses to create a circular economy. In addition to future sprawl, many current urban soils are degraded and could have substantial increases in carbon if organic matter amendments are applied, up to a doubling of SOM% in these soils (Table 2-1; Figure 2-2). Carbon increases are one vital benefit in these soils when amendments are applied, but there can be social benefits as well, such as increased capacity to support urban food production and security.

4.2 Urban Soil Quality

The application of organic matter amendments improve soil properties such as soil nitrogen, phosphorus, potassium, and bulk density (Table 2-1; Salomon et al., 2020). These soil properties improve regardless of amendment type, but soil amendment type influences soil quality in distinct ways (Table 2-2). Biosolids have the most dramatic increases in soil nutrient levels for P and K, but when compared to compost's and biochar's improvements in N the changes are lackluster (Table 2-2). This is contrary to what one might expect, but the large difference between compost and biosolids is partially driven by composted biosolids being grouped with compost overall, thus increasing the compost applied nitrogen levels. There is still improvements in soil N with biosolids, though lower than expected with an average C:N of 10 (Table S2-3), and may be due to biosolids often being applied in lower amounts than compost. As for biochar, it is important to contextualize how biochar saw more increases than biosolids for N (Table 2-2). While studies have shown biochar to increase soil P, this large of an increase in soil N was unexpected (Gao et al., 2019). Approximately 1 g/kg of biochar's predicted 1.7 g/kg increase in soil N was driven by a study with pyrolyzed sewage sludge rather than woody material, Yue et al. 2017, which saw an increase of 0.8 g/kg to 10 and 20 g/kg, double the closest study. Even when including these pyrolyzed biosolids, biosolids alone are likely the favored amendment for high nutrient levels and faster mineralization compared to biochar and compost (Hagemann et al., 2017; Lu et al., 2012). Co-use of high nutrient biosolids with slower-releasing and higher-carbon compost or biochar could be used to effectively restore degraded urban soils, supplying carbon and nutrients, and could use further study (S. L. Brown et al., 2003).

Among land-uses, the largest improvements in nutrient levels are in reclamation and landscaping sites (Figures 2-3, 2-5, 2-6, S2-4). These land-uses likely see the largest gains as they have the lowest baseline values, allowing for the most potential improvement (Figure 2-5). In comparison, only modest shifts are seen in gardening sites. This may be because urban garden sites are already managed for factors such as nitrogen and thus, do not have as much to gain as a vacant land site. However, garden sites may require more regular maintenance due to harvest and could be an effective location for continued nutrient-focused amendment application. Biosolids may then be appealing in gardening spaces where crop productivity is a main objective as these gardening spaces may not have as much room to grow in their SOM%, making them optimal locations for biosolids use to focus on improving soil nutrients instead (Figure 2-3). All amendments types and land-uses see improved urban soil quality with amending, but as seen in gardening sites, aligning individual strengths to specific application and land-use goals will allow for the greatest potential benefit.

4.3 Societal Co-Benefits

Organic matter amendments can also provide broad societal benefits to urban populations. Increased greenery can improve mental health, and the productivity of green spaces can be improved with soil amendments (South et al., 2018). Organic amendments are also important components of urban gardening, which improves local food security and provides a location for communities to learn and recreate (Barthel & Isendahl, 2013; Diekmann et al., 2020; Langemeyer et al., 2018). Encouraging organic matter amendment use in these areas could also lower the need for gardening peat, a destructively harvested carbon resource (Alexander et al., 2008; Schmilewski, 2008). However, a potential barrier to widespread use of some organic matter amendments in urban environments are the real or perceived risks of contaminants. However, the use of these amendments has actually been found to reduce the bioavailability of harmful compounds such as lead and arsenic in both soil and in plant tissue (Yang et al., 2018; Yue et al., 2017). In addition, plant concentrations of helpful nutrient such as nitrogen and phosphorus can increase, potentially making urban garden plants improved for human consumption with amendment application (Antonious et al., 2014; Una et al., 2022).

In addition to these benefits, compost application can reduce NOx emissions of soils by 80% (Meijide et al., 2007). Using California as an example, this could decrease total NOx emissions by approximately 7% as California soils emit 30% of state NOx emissions, with 30% of those emissions coming from developed land soils, leading to improvements in urban air quality (Almaraz et al., 2018; L. Guo et al., 2020). Therefore, additional science and outreach should be prioritized to reduce perceptions of risk and emphasize these potential public health benefits from adding organic matter amendments (Beecher et al., 2005; Whitehouse et al., 2022).

4.4 Literature Gaps and Future Research

Based on our findings, we recommend seven priority focuses for future research in urban soil systems. First, future studies should further assess the effects of biochar and biosolids as the current literature is dominated by compost studies,

despite the fact that feedstocks for biochar and particular biosolids are prevalent in urban systems. Second, future studies should assess recreation and gardening land-uses as they are underreported in the literature, are likely managed with fertilizer that could be supplemented by organic matter amendments, and provide many societal benefits. Third, future research should assess carbon to depth as reporting is done almost exclusively above 30cm in urban systems with few observations to 50cm or 1m, potentially missing large stocks of carbon (Edmondson et al., 2012; Kranz et al., 2020). Fourth, studies should investigate the changes in soil carbon persistence to understand how changes in urban soil carbon stocks from organic matter amendments may last. Fifth, future research should evaluate how organic matter amendments influence urban soil greenhouse gas emissions. Several studies have found increases in CO₂ fluxes in urban soil systems but changes in nitrous oxide and methane soil emissions from organic matter amendment application – generally finding reduced fluxes - have overwhelmingly been assessed solely in agricultural contexts and without a systems perspective (Beesley, 2014; Pierre et al., 2016). Sixth, additional research should identify how urban soil microbial communities adapt to organic matter inputs, particularly as they are unique and highly diverse compared to other soil microbiomes (Guilland et al., 2018; Muth, 2021). Lastly, there is a need for longer term study as the vast majority of literature in this field is conducted on a one to three year timescale. Incorporating the recommendations above will allow for a more wholistic and long-term understanding of the urban soil carbon implications after the application of organic matter amendments.

5. Conclusion

Results from this study indicate the ability of organic matter amendments to strongly increase urban soil carbon, with SOM% doubling in amended sites. All urban land use types showed increases in SOM%, a finding that was consistent regardless of amendment type. Biochar and compost produced the largest increase in SOM%, while biosolids led to more pronounced increases in soil nutrients. Additional soil properties improve with amendments such as soil nitrogen and decreased bulk density. In addition to improving soil quality, amending presents many co-benefits. Urban planners and policy makers should prioritize the use of these urban organic matter amendments to reduce municipal waste loads, improve public health, and improve regional carbon planning. Organic matter amendments in urban settings hold untapped potential to help improve waste streams, improve urban soil quality, and increase soil carbon storage.

Data Availability Statement

Datasheets used in this manuscript for analysis were generated via scholarly journal searches and the datasheets used and analyzed during the current study are available from the corresponding author on reasonable request. Data is available on Dryad (<https://doi.org/10.5061/dryad.h70rxwdsml>).

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Chapter 3: Recycled Water Increases Overall Biomass and Nitrogen While Compost Increases Belowground Biomass and Phosphorus, Depending on Initial Soil Health

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Abstract

Turfgrass is one of the most intensively managed and common crops in the United States. These systems are often heavily irrigated and fertilized, which can lead to both high soil carbon stocks and soil greenhouse gas emissions. Therefore, management practices can alter whether a turfgrass system is a net carbon source or sink. Many municipalities are now looking to urban soils in their climate action plans, to reduce their dependence on external inputs as cost-saving measures, or to improve their waste management strategies. Compost application and recycled water use are two strategies that may achieve these goals. Compost can be created from green waste while recycled water is non-potable treated wastewater, with both management strategies used to supply important soil nutrients. To evaluate nutrient dynamics and climate impacts of these management strategies, both on their own and their interaction, we conducted a year-long nutrient mineralization study and a three-month, multifactorial greenhouse experiment with two compost application rates, two irrigation sources, and two initial soil carbon contents. In the greenhouse experiment, we found that the compost additions did not increase carbon dioxide (CO₂) or methane (CH₄) emissions, regardless of initial soil carbon conditions. We did, however, find increases in nitrous oxide (N₂O) emissions in soils watered with recycled water. In addition, recycled water led to a large increase in aboveground biomass while compost additions increased belowground biomass. The increases in biomass may have been supported by increased soil nutrients, which we observed in the mineralization experiment results through increased total phosphorus and phosphorus mineralization rates. These results suggest benefits to belowground carbon inputs with compost additions, while recycled irrigation water provides aboveground production benefits at the cost of increased soil nitrous oxide emissions.

Key Words: compost, recycled water, soil organic matter, urban soil, turfgrass, greenhouse gas emissions, mineralization

1. Introduction:

As policymakers and researchers aim to foster circular bioeconomies that recycle nutrient waste streams as a source of soil fertility, two potential pathways are compost application and the use of recycled water. Compost application is an important practice that reduces landfill methane (CH_4) emissions, improves the nutrient loop, and enhances soil health. Soil nutrients can also be added through recycled water use, the growing practice of using treated wastewater for irrigation in order to improve system-wide water use efficiency and supply nutrients to soil systems. Both compost application and recycled water irrigation can reduce nutrient loss to waste processes, but also have large impacts on the soil health and climate impact of managed systems that require further study. Compost application and recycled water use can both influence soil greenhouse gas emissions, an effect which is vital to understanding the net climate impact of managed ecosystems. These are particularly apparent for turfgrass systems, which require intense management and are rapidly up taking both compost and recycled water use as a means to manage ecosystem health.

Turfgrass systems often have high levels of soil carbon (C), which along with intensive management strategies can lead to variable soil greenhouse gas emissions. In a vacuum, increasing soil C stocks can lead to increased soil CO_2 emissions, which can reflect fast cycling of biogenic carbon or priming of native soil organic matter (Rastogi et al., 2002; Zhang et al., 2013). Thus, if compost increases soil organic matter there is likely to be increased biogenic CO_2 emissions and the potential of native C priming. Even in systems where compost does not increase soil carbon, the addition of exogenous C that is then decomposed will increase CO_2 emissions in the short-term (Beesley, 2014; Wong et al., 2023). In addition in CO_2 , compost could impact CH_4 emissions by impacting soil carbon, water status, or microbial community composition, though most studies in non-flooded soils find no significant effect on CH_4 emissions with compost use (Ryals & Silver, 2013). In turfgrass systems, the result of compost application is unclear as CH_4 emissions in turfgrass soils are highly variable and sensitive to soil water conditions. Compost application to turfgrass systems can be a source or a sink of CH_4 depending on management factors (Braun et al., 2023; Law et al., 2021).

As turfgrasses also receive high amounts of irrigation and fertilization, there is the potential for high levels of reactive N species emissions, such as from nitrous oxide (N_2O), ammonia (NH_3), or nitrogen oxide (NO_x) emissions. These emissions are influenced by pools of available inorganic N, soil water content, microbial community activity and structure, and pH (Butterbach-Bahl et al., 2013, Lü et al., 2023; Smith et al., 2018). The application of inorganic fertilizer often leads to increased emissions of N_2O , with the creation and use of fertilizer being the largest anthropogenic source of N_2O (Davidson, 2009; Zhang et al., 2012). Compared to inorganic fertilizer, compost and other organic-based fertilizers release N slowly, which can contribute to reduced N_2O emissions (Ball et al., 2004; Ding et al., 2013a). This reduction in N_2O emissions comes from a lower rate of N mineralization with organic-based fertilizers compared to inorganic fertilizers, even though the total soil

N in an organic fertilizer system may be higher (Ding et al., 2013b; Lazicki et al., 2020a). With higher total soil N, higher rates of denitrification are common in organic-fertilizer systems, but complete denitrification to N_2 dominates, which is why N_2O emissions do not increase on a similar scale relative to inorganic fertilizers (Tatti et al., 2013).

Within organic-based fertilizers, N mineralization rates of green waste compost are low compared to manure-based composts due to a higher C:N ratio, leading to more sustained available N and longer term improvements in soil health even after just a one-time application with green waste composts (Atoloye et al., 2022; Geisseler et al., 2021a). In some cases amendments with high C:N ratios can lead to net nitrogen immobilization, decreasing overall nutrient availability (Kaleem Abbasi et al., 2015). Organic fertilizers can also improve other soil properties such as phosphorus (P) availability and microbial biomass, leading to more holistic improvements in soil health compared to inorganic fertilizer additions (Golabi et al., 2007; Krause et al., 2022). Even when mineral fertilizers are used, organic-based fertilizers can be added in combination to mitigate emissions or harmful impacts of fertilization such as N_2O and nitrate leaching (Ding et al., 2013c; Plošek et al., 2017).

The impacts of compost amendments also vary based on native soil characteristics, chiefly soil organic matter concentrations. Soils with high concentrations of soil organic matter may receive diminishing marginal returns in soil health improvements with compost additions, relative to low organic matter soils (Nicoloso et al., 2016; West & Six, 2007). This is because soils with adequate soil organic carbon, estimated to be 2% or higher, may already be supplying nutrients to plants at an adequate level for their growth (Oldfield et al., 2019). Many turfgrass systems already have high C levels, often exceeding 4% soil organic carbon, and understanding how compost additions may impact these high C soils is an open question (Phillips et al., 2023; Qian & Follett, 2012). In addition to high carbon, these soils are often intensively managed with inorganic fertilizer additions, irrigation, and frequent mowing. These management practices and inputs impact soil N pools and N_2O production, and compost additions may further impact N cycling and loss.

The type of irrigation used can also influence soil quality, greenhouse gas emissions, and plant production. In many arid regions, cities are increasing their use of recycled water to irrigate turfgrass fields for water conservation and fertilization. Recycled water is treated wastewater that is often high in N and other micronutrients compared to irrigation with potable water. However, excess nutrient application from recycled water can lead to high N loading in these systems, especially when inorganic fertilizer is still applied (Chiou, 2008). Some studies have found that recycled water can lead to increases in N_2O and NH_3 emissions from soils when compared to potable water irrigation (Chi et al., 2023; Zhu et al., 2023). While emissions may increase, plant growth responses can vary, as recycled water provides a source of nutrients to increase plant production but can also lead to salinization, reducing plant growth (Phogat et al., 2020).

Recycled water may increase available soil nutrients, but adding compost to these fields may also mitigate nutrient losses by building soil organic matter and

increasing nutrient retention. However, the interactive effects of using recycled water with compost additions is understudied. Current research shows that adding compost and recycled water together could have a synergistic effect on plant production by increasing available N, P, and potassium (Clarke et al., 2019; Regkouzas et al., 2025). However, recycled water may also increase the rate of compost decomposition and lead to further N_2O emissions through the supply of additional N (Chi et al., 2024; Clarke et al., 2019). These impacts could vary by initial soil C level as well, as varying soil C and N levels will influence nutrient availability and microbial activity. Specifically, changes can range from priming differences, which are often stronger in soils with lower carbon, to differences in the amount of nitrifying and denitrifying bacteria, influencing N gas emissions (Duan et al., 2018; Elrys et al., 2021; Sun et al., 2019). Overall, both compost and recycled water can vastly impact soil nutrient and C cycling, and how they do so may vary by the initial soil quality.

Our objective was to assess the short-term impacts of compost application and recycled water irrigation, separately and combined, on plant production, soil carbon, soil N, and soil gas emissions in a turfgrass system. As cities begin to implement these management strategies both alone and combined, understanding how they interact is vital. This is particularly true as few studies have wholistically assessed these management impacts in high quality and well managed soils, like a turfgrass field, where their potential benefits are not as clear. We hypothesized that 1) Compost additions will have P mineralization as the compost has more available P than the soil alone, but lead to low N immobilization, due to the relatively high C:N (16) of applied compost compared to soil alone (10.5). 2) Recycled water would increase N_2O emissions, available N, and plant biomass relative to potable water due to the recycled water's high level of N and micronutrients. 3) The impacts of compost additions and recycled water irrigation would be most pronounced in the low-C initial soils compared to the high-C initial soils due to significantly lower starting C, N, and overall soil health in the low-C initial soils.

2. Materials and Methods:

We conducted a mineralization assay and greenhouse experiment to estimate the potentially available nutrients following compost addition and the combined impacts of compost, irrigation water type, and initial soil C concentration on plant production and greenhouse gas emissions. The mineralization assay estimated short- and long-term mineralization rates of inorganic N and P for a total of 52 weeks. Moist soils were incubated in the lab with two levels of compost application and two irrigation types. The greenhouse experiment was conducted with living turfgrass with two levels of compost application, two irrigation types, and two levels of initial soil quality. Running these experiments together allows for understanding both the isolated nutrient dynamics (mineralization) and how the system as a whole (greenhouse) would respond to compost application, irrigation types, and initial soil quality.

2.1 Nitrogen and phosphorus mineralization assays

N and P mineralization rates were determined using a plant-free mineralization assay. This consisted of cups with 50 g of 2mm sieved soil per cup. The soils were classified as fine, smectitic, thermic Xeric Endoaquerts from the Clear Lake soil series with a sandy loam texture, and were sourced from the top 10 cm at a recreational park with turfgrass cover in Dublin, CA (Emerald Glen Park, Dublin, CA, 37°42'38"N, 121°52'33"W). Cups were covered with lids with 4 holes drilled to allow for air flow and then placed in a blackout chamber. Sprouts and roots were removed when found.

Treatments included three compost application rates (0, 14.3, and 28.5 Mg/ha dry mass) each watered with either filtered deionized water or recycled water. Additional cups were included with compost only (25 g) to determine the mineralization rate of compost without soil. The low application mineralization cups received 2.4 g and high application rate cups received 4.8 g of air-dried compost to the top of the soil, both scaled to field application rates (Table S3-1). The compost used was a green waste compost (18.5% C, 1% N, 17.9 C:N, pH of 7.8) was sourced from Pacific Landscape Supply in Vernalis, CA. Cups were watered once per week with either filtered deionized water or recycled water to maintain field capacity. Recycled water was sourced from the city of Dublin, CA. The recycled water contained 30 mg/L of NH_4^+ and had a pH of 7.1. This equated to 60 μg of N applied per week with recycled water, for a total of 3.1 mg of N applied by week 52.

Destructive sampling for N and P availability occurred at 1, 4, 12, 36, and 52 weeks ($n=3$ per timepoint). At each timepoint, soils were extracted for both inorganic N (NO_3^- and NH_4^+) and P (soluble inorganic P, organic P, labile inorganic P, and stable mineral P), while gravimetric water content was measured to express units on a dry mass basis. N mineralization rates were calculated by subtracting the initial soil NH_4^+ and NO_3^- from the NH_4^+ and NO_3^- content at each time point and then dividing by the time point (Geisseler et al., 2019). P mineralization rates were calculated in the same way, but separated into each of the four P pools.

2.2 Greenhouse experimental design

A greenhouse experiment was conducted to determine the separate and combined effects of compost and irrigation water type on soils with different initial soil C concentrations. Pots (diameter = 10 cm, volume = 864 cm^3) were filled with 400 g of soil on top of 100 g of sand and a cheesecloth at the bottom to promote drainage. The soils were sourced from the same location as the mineralization assay soils, a recreational park with turfgrass cover (Emerald Glen Park, Dublin, CA, 37°42'38"N, 121°52'33"W). Two sampled depths were used to create the initial soil treatment with 0-10 cm (a sandy loam with 5.2% C, 0.5% N, and pH of 7.6) representing high-C initial soils and 10-30 cm (a sandy clay loam 1.9% C and 0.16% N, and pH of 8.4) representing the low-C initial soils. Roots were removed from soils prior to the experiment start. The compost added to the turfgrass pots was 16 g for the low compost application rate and 32 g for the high compost application rate, scaled to the same field rates used for the mineralization assays (Table S3-1). The compost applied in the greenhouse experiment is the same as the compost used in the

mineralization assay (green waste, 18.5% C, 1% N, 17.9 C:N, pH of 7.8) which was sourced from Pacific Landscape Supply in Vernalis, CA. Before compost application, 105 turfgrass seeds (50% blue fescue, 50% Kentucky bluegrass at a rate of 2.9 kg per 100 m²) were added and grown for one month for the greenhouse trial prior to compost application, for a total of four months post seed sowing. Pots received two different kinds of irrigation, either deionized water or recycled water sourced from the city of Dublin, CA (30 mg/L of NH₄⁺, pH of 7.1). Pots were watered two to three times per week after greenhouse gas measurements to hold soils at 70% water holding capacity. Across all recycled water pots, an average of 72 mg of N was added to each recycled water pot. A total of 60 pots were started to ensure plant growth, which were then reduced to 36 pots to begin the gas incubation, with 3 replicates for each treatment (irrigation type, compost type, and initial soil quality). The greenhouse facility was located at the University of California, Merced was kept at 24°C and 70% relative humidity for the duration of the experiment from February to June 2025. Pots were monitored for three months to collect adequate soil greenhouse gas emission data that was both far enough removed from the initial compost addition system shock and similar in length to a season to observe variance in emissions over time common in turfgrass soils (Salinas, 2024). This also allowed enough time for both above and belowground biomass to respond to treatments on a management time scale, as turfgrass can respond rapidly to change (Shi et al., 2006; Wen et al., 2025).

2.3 Soil Analysis

At the end of the greenhouse experiment, soils were sieved to 2 mm. Fine roots were extracted by hands and rinsed free of soil. Sieved soils were air-dried and ground with mortar and pestle. Soils and biomass were analyzed for C and N concentration on an elemental analyzer (Costech 4010; Costech Analytical Technologies, Inc., Valencia, CA) at the Stable Isotope Ecosystem Laboratory of the University of California, Merced. Fresh soil samples collected at the end of the experiment were analyzed for extractable N, extractable P, and microbial biomass carbon.

2.4 Greenhouse Gas Sampling

Soil CO₂, N₂O, and CH₄ fluxes were sampled using a closed-circuit, static chamber connected to a cavity ringdown spectrometer (Picarro G2508; Picarro, Inc., Santa Clara, USA). Chambers consisted of a 3D printed chamber top (520 cm³) fit to the soil-plant system in a 3.6 L plastic pot. Soil gas fluxes were measured immediately after compost application, the following day, and then twice a week for the next three months. Measurements occurred between 7 am and 2 pm of each sampling day. Pots were rotated after each sampling event to remove placement bias in the greenhouse. During individual soil gas flux measurements, concentrations of CO₂, N₂O, and CH₄ were measured continuously for five minutes. Fluxes were calculated in the Picarro Soil Flux Processor using a linear model or the Hutchinson and Mosier exponential model when nonlinear changes in concentration were observed (Hutchinson &

Mosier, 1981). Cumulative gas fluxes were estimated by linearly interpolating between sampling timepoints.

2.5 Plant Biomass

Plants were cut to a height of 9 cm weekly to mimic mowing events. Clipped biomass was collected along with the final aboveground biomass to calculate the total aboveground net primary production during the experiment for each pot. At the end of the experiment, belowground biomass was harvested by washing soil from roots for 15 minutes. This allowed for a consistent time-constrained and efficient comparison of root biomass between pots while still allowing for the collection of most root biomass (Addo-Danso et al., 2016). All biomass was then dried in ovens at 65°C for 48 hours before being weighed. Dried aboveground biomass samples were then cut using a Wiley-Mill (Thomas Scientific, Swedesboro, NJ) using the 40 mesh (0.4 mm) while belowground biomass was ground with a mortar and pestle. Samples were analyzed for C and N concentrations on an elemental analyzer (Costech 4010; Costech Analytical Technologies, Inc., Valencia, CA) at the Stable Isotope Ecosystem Laboratory of the University of California, Merced.

2.6 Microbial Biomass

Microbial biomass carbon (MBC) and microbial biomass N (MBN) was measured for initial soils (n=5) to test for pre-treatment differences and then for each pot at the end of the experiment using the chloroform fumigation method (Brookes et al., 1985). For each sample, 10 g of soil was placed in an oven at 105°C for 48 hours to determine moisture content. Then, 20 g of soil was shaken with 80 ml of 0.5M K₂SO₄ for one hour at 200 rpm, filtered through a Whatman #42 filter paper, and frozen at -20°C until analysis. Another 20 g of soil was placed in a 50 ml beaker for chloroform fumigation. Beakers of soil were placed in a vacuum desiccator along with and a 100 ml beaker with boiling chips and 30 ml of chloroform. The desiccator was then evacuated and vented three times in the fume hood with the last boiling for 60 seconds without venting. These soils were then placed in the dark for five days, and then extracted with 0.5M K₂SO₄ as described above. Samples were thawed and diluted by four times with deionized water and analyzed on a total organic carbon analyzer with a total nitrogen module (Shimadzu TOC-L, Kyoto, Japan) at the UC Merced Environmental Analytical Lab. Microbial biomass C was calculated as the difference between C content of nonfumigated and fumigated soils. A similar approach was used to calculate MBN. An extraction efficiency factor of 0.45 was used in the final calculation (Beck et al., 1997).

2.7 Soil Available Nitrogen

Soil nitrate (NO₃⁻) and ammonium (NH₄⁺) concentrations were assessed using a KCl extraction procedure. Approximately 75 ml of 2M KCl was added to 20 g of fresh soil and then shaken for one hour at 200 rpm. The slurry was poured through a Whatman #1 filter, and the filtrate was stored at -20°C until it was analyzed on a microplate reader (Agilent BioTek Gen5 Microplate Reader, Agilent Technologies, Santa Clara,

CA, USA) using the hydrazine and sodium nitroprusside spectrophotometric methods for NO_3^- and NH_4^+ , respectively (Downes, 1978; Mulvaney, 1996). A 10 g subsample of each soil was placed at 105°C for 48 hours to assess gravimetric water content and used to express inorganic pools on a dry weight basis.

2.8 Soil Available Phosphorus

Phosphorus was measured using the biologically based-P (BBP) method (DeLuca et al., 2015). Four extractants were used; 0.01M calcium chloride, 0.02 EU ml⁻¹ phosphatase enzyme solution in 50 mM sodium acetate buffer, 0.01M citric acid, and 1M hydrochloric acid. These were used to represent soluble inorganic P, organic P, labile inorganic P, and stable mineral inorganic P, respectively. Ten ml of extractant was added to 1 g of soil and then shaken for three hours at 200 rpm. After shaking, tubes were refrigerated for 30 minutes to allow for particles to settle. Then, the solution was filtered through a Whatman #1 filter. Citric acid, Enzyme P, and hydrochloric acid filtrates required 10-80 times dilution to run on the plate reader. The filtrate was then stored in the freezer at -20°C . The filtrate was analyzed on a microplate reader using the Malachite green method (Baykov et al., 1988). 50 ul of mixed reagent (sulfuric acid, ammonium molybdate tetrahydrate, tween, and deionized water) was then added to 200 ul of sample, allowed to sit for one hour at room temperature, and then read for absorbance at 630 nm.

2.9 Statistical Analyses

All statistical analyses and all data visualization was done in R (R Core Team, 2020). All data was tested for normality and outliers were assessed using Z-scores, with a value of three or more connotating an outlier. Visual inspection was used to confirm, and outliers were only detected for CH_4 data, where three data points were removed (Cousineau & Chartier, 2010). A three-way analysis of variance (ANOVA) was used to assess statistically significant relationships among compost application rates, watering type, and sampling time point in the mineralization assay. For the greenhouse experiment, three-way ANOVAs were used to assess statistically significant relationships among initial soil quality, compost application rates, and watering type. After running an ANOVA, Pairwise Tukey's Honestly Significant Difference (Tukey HSD) post hoc tests were used to discern significances between independent variables using the *emmeans* function in the *emmeans* package. An alpha value of 0.05 was used for significance testing. Mean values are reported alongside their $\pm$ standard error. Visualization was conducted in R using the "ggplot2" and "ggpubr" packages in R for plots (Kassambara, 2020; Wickman, 2016, p. 2).

3. Results:

3.1 Nitrogen and P mineralization rates

Soil NH_4^+ made up the majority of the total available N pool among all treatments and in the soil-free compost comparison but total available N pools did not vary by any treatment type (Figure S3-1). When incubated in moist conditions without soil or

plants, compost had positive net N mineralization rates for at least 12 weeks. However, in weeks 36 and 52, there was net N immobilization (Figure S3-2). Weeks four and 12 had stable mineralization rates, 0.68 and 0.58 mg N/ kg compost / week respectively, at for both watering types. When assessed over the 52 week period the mineralization rate was -0.05 mg N/ kg compost / week. When compost was added to soil, there was net N immobilization across all treatment types and timepoints, except for a net N mineralization for all recycled water treatments in week 36 (Figure 3-1). Across all compost application rates and dates, watering type marginally increased the mineralization rates with a rate of -0.013 mg N/ kg soil / week and -0.011 mg N/ kg soil / week for deionized and recycled water respectively when assessed at 52 weeks ($p < 0.1$).

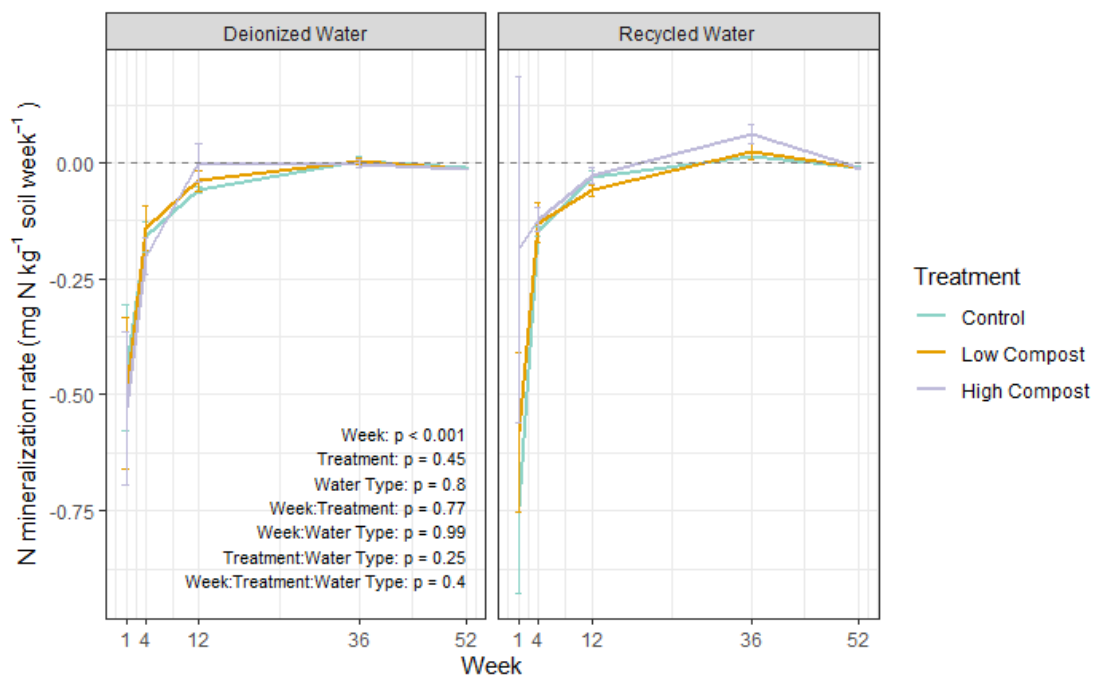


Figure 3-1: Nitrogen mineralization rate between compost application rates and watering types in the mineralization. The ANOVA tables denote statistical differences between application rate, water type, and initial soil C.

Phosphorus mineralization rates increased with the high level of compost addition compared to controls for soluble inorganic P and stable mineral P ($p < 0.05$, Figure 3-2A, Figure 3-2D). For organic P and labile inorganic P, all compost treatments increased mineralization rates, with higher levels of compost having significantly higher rates compared to the low application rate ($p < 0.05$, Figure 3-2B, Figure 3-2C). For all P mineralization, rates were highest in weeks one and four before stabilizing in weeks 36 and 52 (Figure 3-2). In the compost only treatment, P mineralized over the 52 weeks in the soluble organic, organic, and labile inorganic pools while it immobilized in the stable mineral pool (Figure S3-3). There was no impact of water type on P mineralization rates in any scenario ($p > 0.05$). When looking at overall concentration, compost application significantly increased the concentration of

soluble inorganic P from week 36 onward, organic P after week 1, labile inorganic P in all weeks, and stable mineral P was only significantly increased by compost application in weeks 36 and 52 (Figure S3-4). Compost only P pools increased in the first month for soluble inorganic P and organic P, but were generally stable overtime after this point. The two exceptions were for decreases in week 36 labile inorganic P, which rebounded in week 52, and a decrease from week four to week 12 for stable mineral P ($p < 0.05$, Figure S3-5).

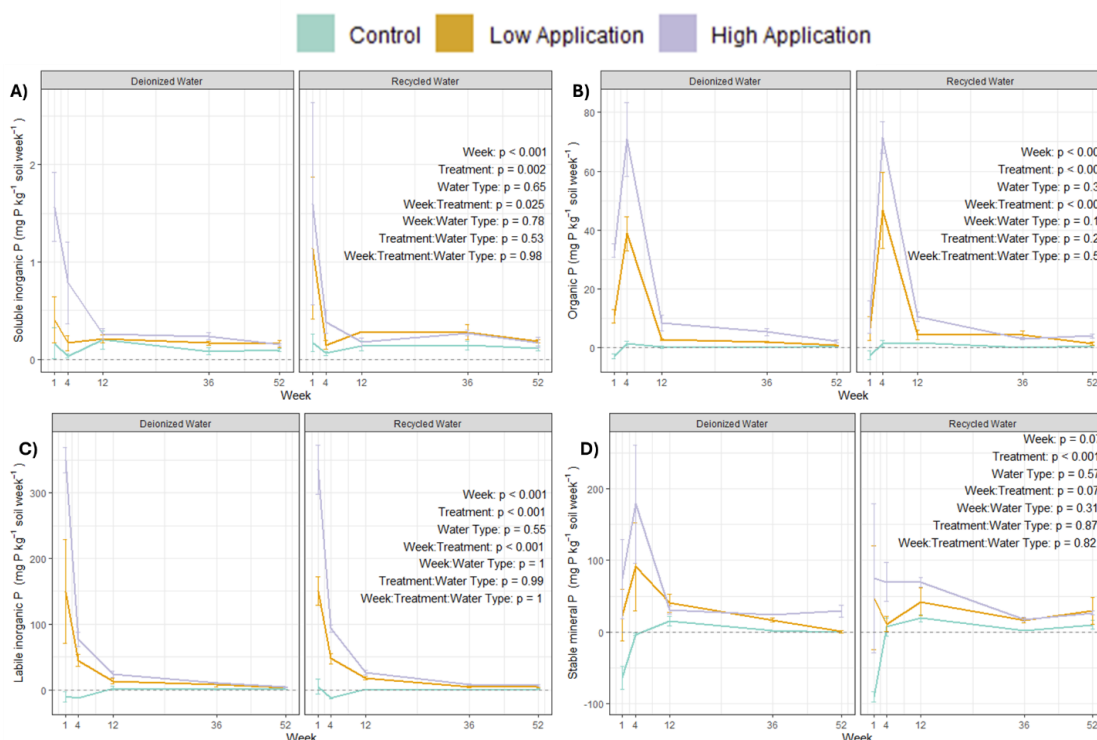


Figure 3-2: Soil phosphorus mineralization rates for pools of soluble inorganic P (A), organic P (B), labile inorganic P (C), and stable mineral P (D) between treatments over the mineralization experiment. The ANOVA tables denote statistical differences between application rate, water type, and initial soil C.

3.2 Changes to soil nitrogen and phosphorus pools in the greenhouse experiment

In the greenhouse experiment, compost level, water type, and initial-C of the soil had statistically significant impacts on total soil available N (Figure S3-1A). Compost decreased available N concentrations with all compost compared to controls ($p < 0.05$), but not when compared within the three treatments ($p > 0.05$). The low-C initial soils had lower available N than the high-C initial soils, and within the low-C initial soils recycled water had more had significantly more available N ($p < 0.05$).

Soil P significantly increased by compost addition among all pools ($p < 0.05$) except soluble inorganic P ($p > 0.05$). Differences in application rate only significantly impacted labile inorganic P and organic P in the high-C initial soil ($p < 0.05$) with the higher application rate increasing soil P more than the low application rate. There was no impact of watering type on any P pool ($p > 0.05$) and only in soluble inorganic P and stable mineral P was there an impact of initial soil C, with higher-C initial soils containing more P than low-C initial soils ($p < 0.05$, Figure 3-3).

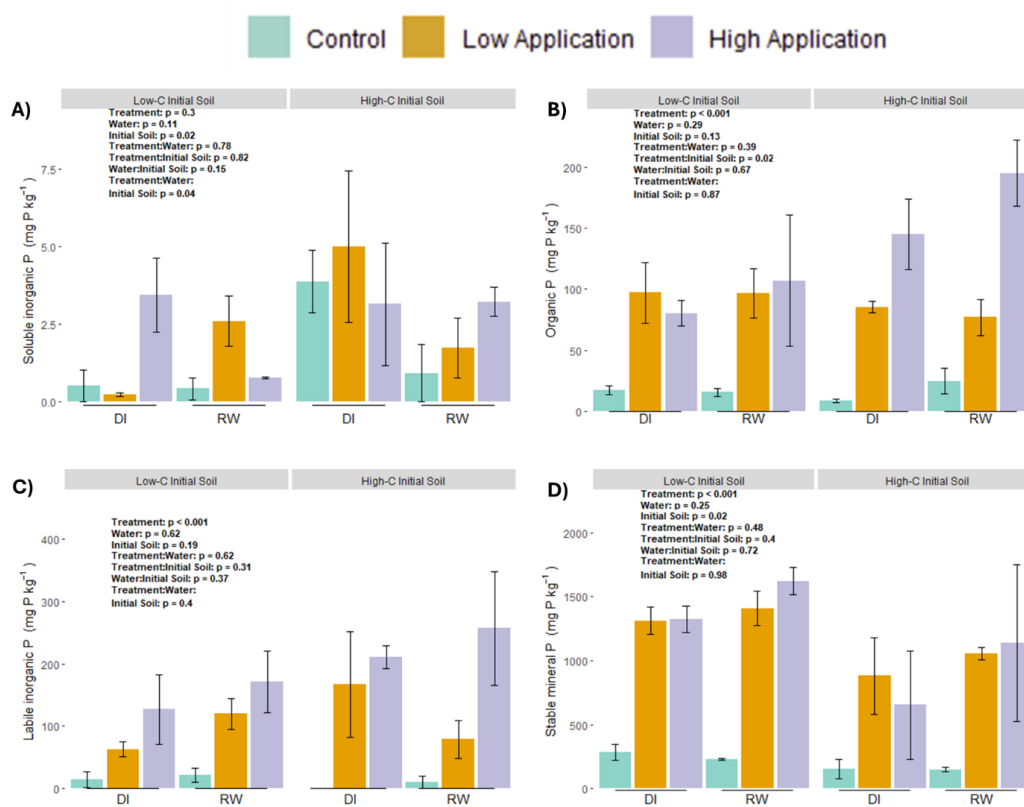


Figure 3-3: Soil P pools of soluble inorganic P (A), organic P (B), labile inorganic P (C), and stable mineral P (D) between treatments over the greenhouse experiment. The ANOVA tables denote statistical differences between application rate, water type, and initial soil C.

3.3 Soil Carbon and Nitrogen

Soil C and N concentrations were significantly impacted by compost, watering type, and initial soil conditions (Figure 3-4, $p < 0.05$). Both levels of compost additions led to increased soil C and N concentrations ($p < 0.05$), but here was no difference between low and high compost application rates ($p > 0.05$). High-C initial soil had significantly higher soil C and N than low-C initial soil ($p < 0.05$). Recycled water decreased both soil C and N concentrations relative to deionized water ($p < 0.05$). Both compost applications rates increased the C:N ratio of the soil by over one unit compared to controls ($p < 0.05$, Figure S3-6), while the low-C soil had a C:N ratio one

unit higher than the high-C initial soil ($p < 0.05$, Figure S3-6). Watering type had no impact on soil the C:N ratio ($p > 0.05$).

MBC, MBN, and the ratio of MBC:MBN did not change with any compost treatment, but MBC and MBN were higher in the high-C initial soil (Figure S3-7, Figure S3-8, $p < 0.05$). Recycled water decreased the MBC:MBN ratio, and this relationship was strongest in the low-C initial soil ($p < 0.05$, Figure S3-8).

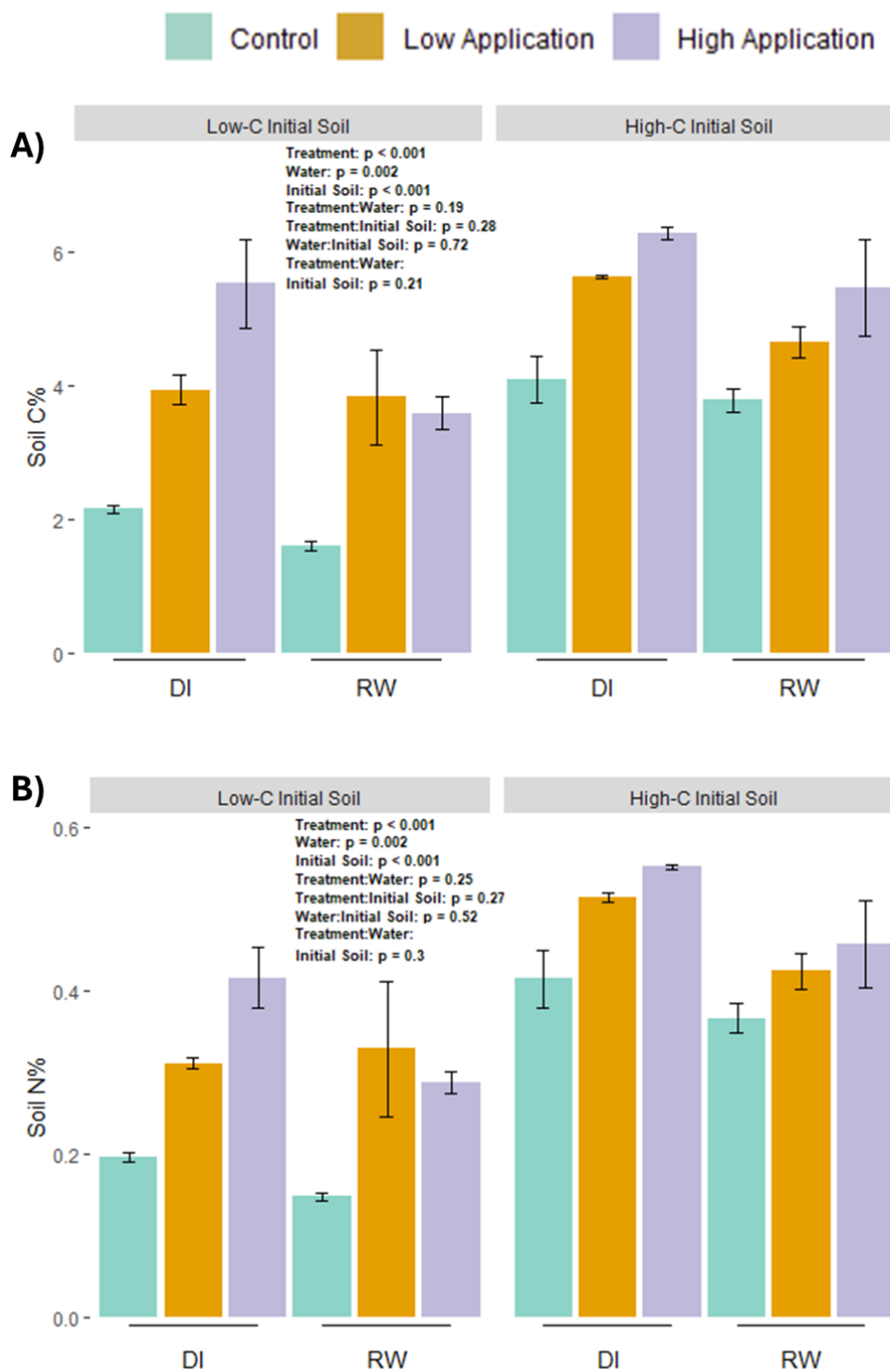


Figure 3-4: Effects of compost, depth, and water type on soil C and N concentrations. The ANOVA tables denote statistical differences between application rate, water type, and initial soil C.

3.4 Soil Greenhouse Gas Emissions

Soil CO₂ fluxes were higher with compost additions in the first 20 days of the experiment, but then were similar to control values (Figure S3-9). Cumulative CO₂ emissions were not significantly different among all treatments ($p > 0.05$, Figure 3-5A). There was no difference in cumulative soil N₂O emissions by compost or initial soil C ($p > 0.05$), but soil N₂O had a marginally significant increase with recycled water compared to deionized water ($p < 0.1$, Figure 3-5B). Soil CH₄ emissions were significantly increased with the high application of compost ($p < 0.05$, Figure 3-5C). The overall soil CO₂-eq was marginally significant with recycled water and higher initial soil C increasing CO₂-eq ($p < 0.1$), but when excluding CO₂, the CO₂-eq was not statistically different between treatments. ($p > 0.05$, Figure 3-5D).

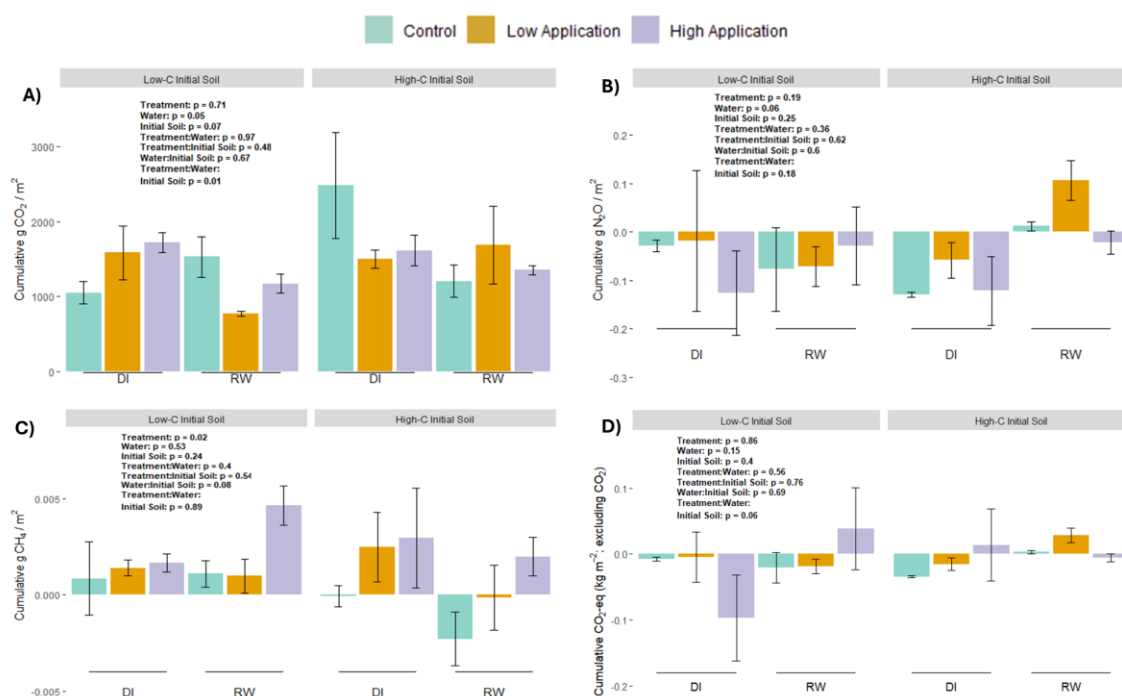


Figure 3-5: Bar chart for cumulative CO₂(A), N₂O (B), CH₄(C), and CO₂-eq without CO₂ (D) emissions over three months based on compost, water type, and initial soil C. The ANOVA tables denote statistical differences between application rate, water type, and initial soil C.

3.5 Soil Biological Responses

We found that the high initial-C soil had significantly more aboveground (Figure 3-6, $p < 0.05$) and belowground biomass ($p < 0.05$) than the lower initial-C soil pots. However, root:shoot increased in the low-C initial soils ($p < 0.05$, Figure S10). Recycled water also significantly increased aboveground ($p < 0.05$) and belowground biomass ($p < 0.05$) while increasing the N concentration within both the roots and shoots compared to deionized water pots (Figure 3-7C, 3-7D, $p < 0.05$). The increase

in both biomass types was associated in a decrease in root:shoot in recycled water soils compared to deionized water soils ($p < 0.05$, Figure S3-10). Both compost rates did not impact aboveground biomass ($p > 0.05$) but did increase belowground biomass ($p < 0.05$), root:shoot ($p < 0.05$), and shoot C concentration ($p < 0.05$, Figure 3-6, Figure 3-7, Figure S3-10) relative to unamended controls. There was no difference between the rates of compost application on any of these biomass properties ($p > 0.05$).

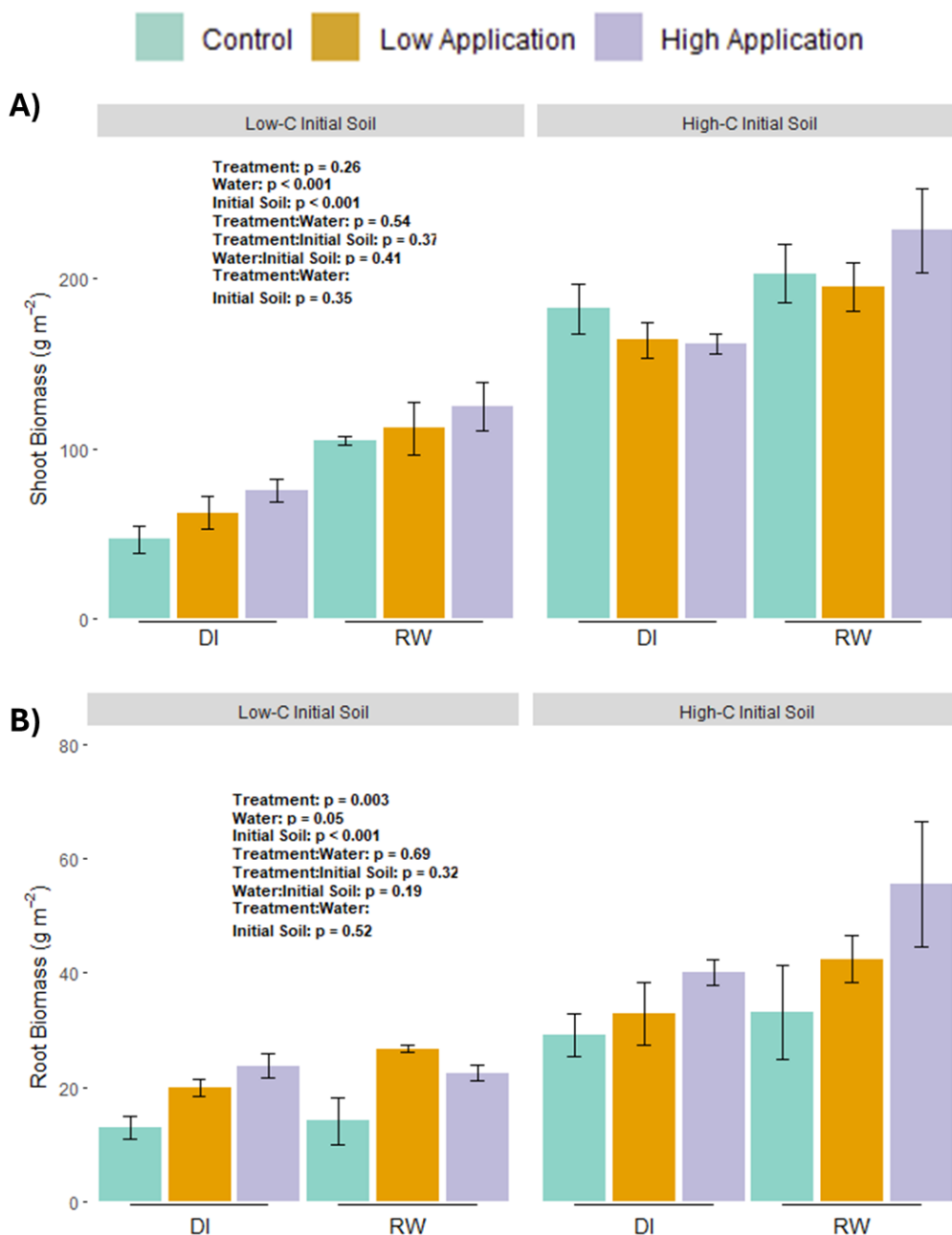


Figure 3-6: Effect of compost, depth, and water type on above (A)- and belowground (B) biomass. The ANOVA tables denote statistical differences between application rate, water type, and initial soil C.

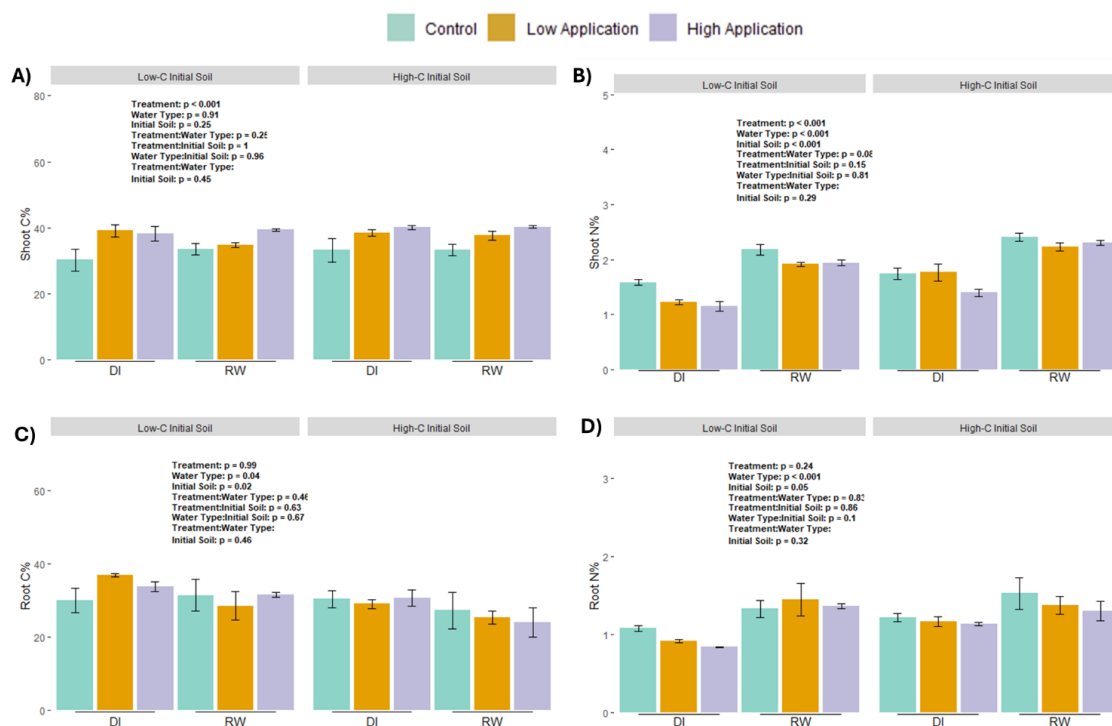


Figure 3-7: Effect of compost, depth, and water type on above and below ground biomass C and nitrogen concentration. The ANOVA tables denote statistical differences between application rate, water type, and initial soil C.

4. Discussion:

4.1 Effects of compost amount and water type on nitrogen and phosphorus mineralization rates

Both recycled water and compost are potential sources of nutrients to the soil system. We anticipated that the largest changes to N mineralization of any treatment would come from the addition of recycled water, though we saw no impact on mineralization rates or total concentrations (Figure 3-1, Figure S3-1). In the laboratory assay, there was net N immobilization until week 12, followed by very low rates of net N mineralization in weeks 36 and 52. Compost additions also had no impact on N mineralization rates compared to unamended control soils, even though compost incubated without soil had a net mineralization rate for weeks 4 and 12 at 0.68 and 0.58 mg N/ kg compost / week (Figure S3-2). It is possible that once the compost was applied to the soil that the microbes from the soil used the compost as a C source, thereby using N from the soil leading to immobilization (Curtright et al., 2026; Lazicki et al., 2020b). This response was mirrored in control soils due to wet-

up activating microbes that began to take up inorganic N as they restarted their metabolism in the early weeks of the experiment (Zhang et al., 2017). Conversely, compost incubated without soil had net N mineralization in early weeks likely due to quick mineralization of organic matter and ammonification of the compost as if it were a wetted compost pile (Gil et al., 2024; Mondini et al., 2002). By week 36 there was minimal change in available N rates in the compost treatment, when a compost pile would expect to be more stable (Geisseler et al., 2021b).

The P mineralization rates significantly increased with compost additions (Figure 3-2). These changes were within the first 12 weeks of the experiment, similar to the N time series switch from immobilization to no net change in N availability. The gains in P mineralization were largest in organic P, which is important as it constitutes bioavailable labile organic P that is readily available for soil biota and roots (DeLuca et al., 2015). This increase in organic P spanned from less than 50 mg P / kg soil to the 100 to 200 mg P / kg soil range in the highest compost application and high soil (Figure 3-3B). While this and the increases seen in other pools are sizable, our pools of P (outside of the stable mineral P which is not easily available) were still lower than the original paper by DeLuca et al. which found levels near 1000 mg P / kg across all four pools and indicated no net P loss from the soil system (DeLuca et al., 2015). This can reassure that despite the increase in soil P in our system, we likely did not have rates that would have contributed to P leaching and pollution. The mineralization rates support this theory as well, as P mineralization rates from the compost were slower releasing than inorganic fertilizers. Mineralization rates remained elevated through week 12, but were no longer different by weeks 36 or 52 (Geisseler et al., 2021b; Grigatti et al., 2019). Thus, compost was effective at increasing rates and pools of P for 12 weeks, with minimal impact of N mineralization.

4.2 Effects of compost amount and water type on soil nitrogen and phosphorus pools in turfgrass growth experiment

For the greenhouse experiment, we found that pools of total available N increased with recycled water compared to deionized water treatments, particularly in the low-C initial soil (Figure S3-1). This likely explains why aboveground biomass increased with recycled water with 0.91 g/pot and 1.27 g/pot in the deionized water and recycled water pots respectively (Figure 3-6A). The gains that recycled water brought in aboveground biomass, an increase of 12.2 mg N in biomass, were not able to account for the total amount of N applied (72 mg) from recycled water over the course of the experiment, nor was the increase in N₂O emissions with recycled water (0.8 mg N₂O-N, Figure 3-5B). It is possible a large portion was lost through leaching, through immediate volatilization of N gases after recycled water was applied, complexation with C, or complete denitrification (Rocha et al., 2019).

The increase in available N from the greenhouse was not observed in the mineralization experiment at week 12, the same timepoint as the greenhouse

experiment length, but this result fits with other findings that recycled water increases soil N availability as all of the N in the reclaimed water was plant available (Sakadevan et al., 2000). The larger response by the low-C initial soil is likely driven by the fact that low-C initial soils contained less available N concentrations than the high-C initial soils, leading to diminishing marginal returns from added N in the high-C initial soils (Ferreira et al., 2017). Initial soil quality appears to play a large role in the response to exogenous material as recycled water can be framed as an effective nutrient management strategy for soils with low organic matter, but there is a tradeoff with increased N₂O emissions when recycled irrigation water is used on high organic matter soils (Chi et al., 2025).

Compost led to decreased available N concentrations overall, though there were no unique interactions from this effect (Figure S3-1). As the compost had a higher C:N than the soil an N-limiting environment may be created leading to the immobilize of available N into microbial biomass. The MBC:MBN ratios increased in compost treatments compared to control soils ($p < 0.1$, Figure S8), which could be an indicator of microbial N immobilization (Hu & Zhu, 2025; Mooshammer et al., 2014). The MBC:MBN ratio was unchanged in the recycled water treatments, suggesting that recycled water and compost treatments may balance each other out in their impact on the MBC:MBN ratios which coincides with the reduced available N for combined recycled water and compost treatments (Figure S3-8).

4.3 Recycled water and initial soil quality impact all biomass measures, while compost impacts are focused on belowground

Compost, recycled water use, and initial soil organic matter content would be expected to improve overall biological activity. Recycled water and higher initial soil quality both leading to increases in both aboveground and belowground plant biomass (Figure 3-6). In addition, both recycled water and high-C initial soils increased the N concentrations of aboveground and belowground plant tissues which could signal healthier and more photosynthetically capable turfgrasses (Field & Mooney, 1986). Compost did not impact aboveground biomass, though there was a decrease in aboveground biomass N concentrations likely due to net N immobilization after the application of compost to soils (Figure 3-6, Figure 3-7). As both soils were already near the proposed 2% to 4% soil C upper limit for a positive plant response from increasing soil C, at 1.9% and 5% C respectively, the impact on aboveground biomass may have been minimal despite the large difference in initial soil carbons (Ma et al., 2023; Oldfield et al., 2019). While compost additions did not elicit an increase in aboveground plant biomass, the different initial soil carbon treatments did. The high-C initial soils had significantly above and belowground biomass compared to the low-C initial soils (Figure 3-6), which are theoretically within the limit of positive plant response (Oldfield et al., 2019). While compost increased the soil carbon, it is likely

that three months was too short a time for the compost to incorporate into the soil to provide a aboveground plant benefit in the low-C initial soils (Evanylo et al., 2008)

Unlike in the aboveground biomass, compost additions increased root biomass with compost additions despite no aboveground biomass change (Figure 3-6). While this could signal increased belowground plant health through increased root production, it could come at the expense of aboveground growth. This may be necessitated by the need for roots, with more roots supporting more exudation and more microbial transformations, to support the decomposition of compost to acquire nutrients in an immobilizing environment as compost can “lock up” nutrients (Dijkstra et al., 2013; Donn et al., 2014; Junaidi et al., 2018). Conversely, while recycled water increased belowground biomass, it significantly decreased the root:shoot ratio, especially in the low-C initial soils (Figure S3-10). With the addition of recycled water in low-C initial soils the need for microbes to mine the soil or compost for nutrients decreased as there was significantly more available N in recycled water pots (Evanylo et al., 2010). Removing the need to mine the soil for N would signify that increased root biomass with recycled water is more a product of overall biomass increase. When recycled water was added the root:shoot decreased, highlighting that plants spent proportionally less resources belowground when recycled water was added, but because aboveground biomass greatly increased it still led to a raw increase in belowground biomass. Changes in the root:shoot ratio and belowground biomass have important implications for future soil C levels, as belowground biomass can more easily become future soil C (Sokol et al., 2019; Sokol & Bradford, 2019). Thus, compost could not only increase soil C directly in our system, but also by supporting root biomass through increased biomass and a shifted C allocation strategy by turfgrasses.

While each variable had an impact on plant biomass, only initial soil quality had a significant impact on MBC and MBN, with high-C soils having significantly increased levels over the low-C initial soils (Figure S3-7). However, there was a shift in the MBC:MBN ratio with watering type as recycled water led to a lower MBC:MBN ratio compared to deionized water, particularly in the low-C initial soil (Figure S3-8). A shift in the MBC:MBN ratio could signify a change in the fungi to bacteria ratio in the soil, as fungi have a higher MBC:MBN ratio than bacteria and are efficient at breaking down the complex organic matter in compost (Babur et al., 2022; Wang et al., 2022). Shifts in relative bacteria and fungi abundance in the soil could explain why there was a significant interaction between our compost and water treatments, with compost increasing the MBC:MBN ratio with deionized water but not impacting recycled water pots MBC:MBN. Increased MBC:MBN ratio with deionized watering is indicative of a shifting microbiome toward fungi that mined the complex organic matter in compost to acquire nutrients that were not available, while recycled water decreased this mining need with additional available N supply (Fu & He, 2024; Leite et al., 2017). Understanding how the activity and abundance of denitrifiers (such as *nirS* and *nirK*) and NH_4^+ oxidizers (such as *amoA*), which are important for determining N_2O emissions, are an important for future studies (Lazcano et al., 2021).

Soil CO₂ fluxes can also give us insight into the biological activity of the soil. While we did not observe changes in the cumulative CO₂ emissions by treatments, soil CO₂ fluxes increased significantly during the first 20 days of the experiment in the high compost application treatments, potentially due to an initial rapid decomposition rapid immediately after compost application ($p < 0.05$, Figure S3-9). The largest early difference in fluxes came from higher CO₂ fluxes in the high-C initial soil compared to low-C initial soils. The high-C initial soils had significantly MBC both prior to experiment start and at the end, signaling that the high-C initial soils had a greater response in soil respiration as there were already more microbes available to engage in decomposition than in the low-C soils (Bastida et al., 2019).

4.4 Recycled water increases soil GHGs while compost use has no impact

In addition to supporting plant production by providing sources of essential nutrients, the use of compost has the ability to support climate change mitigation. We found stepwise increases with higher compost applications in both soil C and N across both water types and initial soil qualities (Figure 3-4). There was no significant difference in how the initial soil C responded to compost additions. This signals to land managers that in the short-term, compost can increase soil C regardless of starting soil quality. Compost did not change cumulative CO₂ emissions, those emission from compost are biogenic which are assumed to have no climate impact (Vergara & Silver, 2019). Despite no change in soil respiration, belowground biomass and root:shoot increased with compost additions signaling a shift in C allocation belowground. Increased belowground allocation could lead to more efficient increases in soil carbon in the future, making these gains particularly valuable (Sokol & Bradford, 2019; Yang et al., 2023).

When assessing recycled water use, we saw improvements in aboveground biomass which could lead to future incorporation in the soil C pool. However, we saw negative direct climate impacts from recycled water use in the gas emissions measured. Recycled water treatments had more cumulative N₂O and CO₂-eq than DI treatments, which correspond to the increase in available N pools we observed with recycled water use ($p < 0.1$). Contrasting this, we found no increase in cumulative N₂O or CO₂-eq from compost treatments. This is important as one concern is that compost additions may stimulate N₂O emissions through increased N stocks as well as by increasing soil water content. A lack of increasing N₂O emissions from compost is a consistent with stabilized or decreased N₂O emissions from compost application, and is important to contextualizing the total climate impact (Ryals et al., 2015). It is possible that compost additions could indirectly increase N₂O by increasing NH₃ volatilization and future N deposition, though NH₃ volatilization from green waste compost is typically low (Nicholson et al., 2017). It is more likely that if our system were a source of NH₃ volatilization and future indirect N₂O emissions it would be from recycled water use due to the high losses, up to 30%, of irrigation water's NH₃ content, through volatilization in the hours after irrigation (Saez et al., 2012).

When we assess the other soil GHG emissions, we found an increase in soil CH₄ emissions when compost was applied, leading to a negative climate impact of

compost application in this system. Despite this increase in CH₄ with compost, there was no change to the cumulative CO₂-eq when excluding CO₂ emissions, which are biogenic, with any level of compost application (Figure 3-5D). Surprisingly, initial soil-C did not impact any soil greenhouse gas emission or the overall CO₂-eq but watering type did have a marginal impact.

When we compare the compost climate results to the associated field study, the field work found N₂O decreased with compost emissions over two-years (Malone et al. 2026 Chapter 4). Because we measured GHGs before watering, there is a chance we missed immediate spikes in GHG emissions from recycled water, particularly in N₂O. Measurements directly after recycled water irrigation would be helpful as emissions of N₂O and other N gases can be higher in the hours following irrigation or fertilization (Recio et al., 2018). To get a fuller picture of N gas emissions, measuring N₂ and NO_x emissions and adding in leaching data would help close the overall N cycle. With frequent irrigation it is likely a sizable but unknown portion of the applied N in the system is leached (Sakadevan et al., 2000). It is also possible that the long-term impacts of recycled water use may show a clearer trend in N₂O emissions than our short-term study, but these studies are lacking in the current literature.

5. Conclusions

Our study revealed that compost and recycled water use on urban park soils have interacting impacts on urban soils that lead to improved nutrient cycling and plant production. Both treatments had unique impacts with recycled water increasing soil N and total plant biomass, while compost increased P and root biomass, with interactions between the two leading to the greatest system response. Improvements in soil nutrient cycling were particularly prevalent in the lower quality soils compared to higher quality starting soils. Thus, prioritization of these management strategies for lower quality soils and parks may be the most efficient way to scale the use and demonstrate the value of these resources. In addition to improving soil quality, compost additions and recycled water use can present many co-benefits. Urban land managers and policy makers should attempt to make use of compost and/or recycled water to reduce municipal waste loads, reduce fertilizer dependence, and improve C and park health.

CRediT Authorship Contributions Statement

Zachary Malone: Conceptualization, Writing – original draft, Writing – review & editing, Formal analysis, Investigation, Data curation, Visualization, Software.

Rebecca Ryals: Conceptualization, Writing – review & editing, Supervision, Funding acquisition.

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Conflict of Interest Statement

The authors declare no conflicts of interest.

Data Availability Statement

Data and code available will be made available on Dryad upon manuscript publication.

Supporting Information

Additional supporting information and figures can be found online in the supporting information section of this article.

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Chapter 4: Short-term compost application reduces reactive nitrogen emissions, but does not increase soil carbon, in high carbon turfgrass soils

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Abstract

Compost is increasingly considered to improve soil health and reduce the amount of organic materials in landfills. Managed urban areas, such as recreation and sports parks, are an ideal land use for compost application due to the need to maintain high quality grass and the proximity to urban waste streams that can be composted. These parks are typically maintained for high grass productivity and quality. As a result, park soils often have high levels of soil carbon, leading to questions of whether or not additional soil carbon sequestration can be achieved through improved management, like compost additions. To assess the benefits of compost application to sports parks, we applied a green waste compost to a public sports park at two application rates for two consecutive years. We measured soil biological, chemical, and physical properties annually and soil gas measurements every two weeks for 24 months. There was no significant change in total soil carbon and nutrient contents with compost addition, however there were significant effects on soil greenhouse gas. Carbon dioxide emissions increased with compost over two years, particularly at high application rates ($28.5 \text{ Mg ha}^{-1} \text{ y}^{-1}$), signaling rapid decomposition of the compost after application. Compost decreased soil nitrous oxide (N_2O) emissions by 25% over the two year experiment, despite no change in soil extractable nitrogen or volumetric water content, providing a climate benefit to the system. Our results indicate that compost application in urban soils with high baseline soil carbon concentration provides little short-term gain in additional soil carbon but can yield climate benefits from soil N_2O reductions.

Abbreviations:

- C Carbon
- DOC Dissolved Organic Carbon
- EC Electrical Conductivity
- MBC Microbial Biomass Carbon
- N Nitrogen
- NDVI Normalized Difference Vegetation Index
- OM Organic Matter

- TOC Total Organic Carbon
- VWC Volumetric Water Content

1. Introduction

Compost is an effective management practice to promote soil health and sequester carbon (C) in agroecosystems. Less research has been conducted on the effects of compost use in urban land uses, which account for approximately 5% of the US land area and can vary in management intensity and use of external inputs (Liu et al., 2014; Milesi et al., 2005). Compost is most commonly used in urban farms and gardens to achieve multiple goals, including supplying plant nutrients, improving soil water infiltration and retention, providing infrastructure for local organics recycling, and in the construction of raised beds to remediate or mitigate heavy metal contamination (Sæbø & Ferrini, 2006; Small et al., 2023). Compost is also used to revitalize the soils within brownfields or construction areas for similar reasons in urban regions (Megharaj & Naidu, 2017; Sæbø & Ferrini, 2006). In contrast, urban land uses covered with turfgrass, which constitute 2% of US land area (Liu et al., 2014; Milesi et al., 2005), tend to receive less or no compost inputs. Many of these soils receive extensive fertilizer and irrigation inputs which lead to high baseline levels of soil organic matter resulting from extensive root biomass (Wang et al., 2022). Compost application to turfgrass systems may provide another pathway for soil C sequestration and soil health improvements, however the question remains if compost can further improve these already well maintained soils (Phillips et al., 2023; Qian & Follett, 2012). Previous research has shown even with high baseline levels C these soils often see increases in soil C and nitrogen (N) with compost addition, though data is often limited to the top 30 cm and is missing the impacts of compost to other C cycling properties, such as greenhouse gas emissions (Malone et al., 2023). Soil carbon dioxide (CO₂) and methane (CH₄) emissions typically increase with compost additions as exogenous C fuels the soil C cycle, and turfgrass soils often have high respiration rates due to active C cycling even without compost additions (Kaye et al., 2004; Xu et al., 2025). In some cases, compost leads to a priming effect in turfgrass soils, leading to net loss of native soil organic matter (Azeem et al., 2020; Beesley, 2014). Understanding if turfgrass soils are approaching a C saturation point or if they can be managed as an urban nature-based climate solution is an important next step in turfgrass management.

Compost application to turfgrass systems can increase soil organic matter and drive changes in microbial activity. These effects, in turn, can alter plant production and soil nutrient cycling, water retention, and C stability (Azeem et al., 2020; Hale et al., 2021). The direction of these changes is heavily influenced by compost feedstock type and the resulting C:N ratio and other physiochemical qualities of the finished compost (Cambardella et al., 2003; Hale et al., 2021). There is an opportunity for widespread use of composts in urban areas due to the volume of green waste generated within these regions. However, in most regions there is a dearth of composting facilities to handle the volume of green waste generated in these urban centers (Harrison et al., 2024). In California, for example, there are procurement and

landfill diversion requirements for urban areas, and approximately 8-26% of generated green waste could be diverted and applied as compost to urban lands within the state (Harrison et al., 2020). There is a need to better understand the impact that compost application may have in these areas once these materials are diverted from landfills and land applied in order to make informed decisions about urban waste management.

Compost use in turfgrass systems can also have large impacts on N cycling and loss in urban soil systems. Many turfgrass systems, particularly in arid regions, receive inorganic N from recycled water and/or synthetic fertilizer for plant growth (Serba et al., 2022; Shaddox et al., 2022). With fertilizer additions and frequent wetting, the denitrification potential of turfgrass systems is high, leading to high reactive N losses (Lu et al., 2015). Reactive N gas emissions, including nitrous oxide (N_2O), and nitric oxides (NO_x), have a cascading effect in the environment (Fowler et al., 2013; Galloway et al., 2003) and contribute to the transgression of a planetary boundary (Richardson et al., 2023). Soil NO_x emissions can impair local air quality through the formation of tropospheric ozone and have an indirect impact on climate forcing (IPCC, 2021a). In California, soils account for 25 to 41% of state level NO_x emissions, with up to 30% coming from turfgrass systems despite turfgrass taking up only 5% of the state's land area (Almaraz et al., 2018; Guo et al., 2020). In addition to NO_x emissions, turfgrass N_2O emissions can be high. One study conducted in Larimer County, CO finding that 30% of regional N_2O emissions came from the 6.4% of land area occupied by urban lawns (Kaye et al., 2004). While there are large variations across turfgrass systems in gaseous N due to highly variable management strategies, biweekly sampling can effectively capture variation within a site (Groffman et al., 2009; Li et al., 2013). Across sites with recycled water irrigation several studies have found increases in N_2O emissions even when N loading rates are similar to mineral fertilized fields, no studies have taken direct measurements of these gaseous losses with recycled water application in a turfgrass system (Chi et al., 2024, 2025).

One potential method of reducing soil N_2O and NO_x emissions is compost application. Several studies in both agricultural and urban systems demonstrate reduced gaseous N loss with compost additions attributed to higher rates of immobilization from increased soil C:N (Meijide et al., 2007; Pierre et al., 2016). However, other studies have found increases in soil N_2O emissions with compost additions relative to unfertilized controls, particularly those in sandy soils and with low C:N composts (Ding et al., 2013; Zhu et al., 2014; Zhu-Barker et al., 2015). Compost with a low C:N ratio may reduce gaseous reactive N emissions in situations where synthetic fertilizer is applied since the N in the compost will slowly mineralize (Meijide et al., 2007). When recycled water is used as a source of water for irrigation, it can introduce a large amount of readily available N even when turfgrass fields do not receive fertilizer applications. There has been little work done to assess N gaseous losses from turfgrass systems managed with both recycled water and compost.

The objective of this study was to assess the impacts of compost application to a turfgrass system irrigated with recycled water on soil C content, reactive N gas emissions, and soil health indicators. We hypothesized that 1) Soil C, N, and microbial biomass concentrations would increase slightly with high rates of compost additions. These changes would be particularly concentrated in the top 30 cm with minimal changes in the subsoil due to the additional of exogenous C from the compost. 3) Soil CO₂ emissions would increase as compost aerobically decomposed, with the aerobic status of the soil allowing for minimal change in CH₄ emissions. 4) Soil emissions of N₂O and NO_x would decrease as compost application would lead to N immobilization. 5) Compost application would provide a climate benefit, driven primarily by decreases in soil N₂O emissions.

2. Materials and Methods

2.1 Study site and experimental design

The experiment was conducted in the city of Dublin, California (37.7159° N, 121.9101° W) on a publicly managed athletic field (Emerald Glen Park). This park is used for both soccer play and general recreational field use. At the beginning of each year (between January-March), renovations are conducted on the field. Renovations consist of soil aeration, the addition of mineral fertilizer (15-15-15-9 N-P-K-S, with 50 kg NH₄⁺ N/ha), turf reseeding in high-use areas (50% blue fescue, 50% Kentucky bluegrass at a rate of 2 to 3 kg / 100 m²), and the addition of a 2 cm layer of sandy loam to the soil surface. The site is irrigated using recycled water (920 mm/yr) five times a week between March and October. An addition of 110 kg N/ha/yr entered the soil throughout the year through irrigation. Irrigation is concentrated most in the summer due to the Mediterranean climate, characterized by warm, dry summers and cool, wet winters, with precipitation falling as rain solely in the winter months. Mean annual precipitation is 500 mm/y and mean annual temperature is 15.5 °C at the site. Soils are classified as fine, smectitic, thermic Xeric Endoaquerts from the Clear Lake soil series that have a sandy loam texture at 0-10 cm and sandy clay loam below 10 cm.

We established a controlled, replicated experiment on an active athletic field. We used a random block design including three blocks, each with 3 m x 5 m treatment plots separated by 0.5 m buffer strips. Treatments consisted of unamended controls and compost applied for two consecutive years at two application rates: 14.3 Mg/ha (hereafter referred to as “low application rate”) and 28.5 Mg/ha (hereafter referred to as “high application rate”) on a dry mass basis. All treatment plots received the same management during the renovation period and the same amount of irrigation water. Green waste compost was sourced from Pacific Landscape Supply in Vernalis, CA (Table 4-1).

Table 4-1: Characteristics of the compost (n=5) applied to the field site.

	C%	N%	C:N	pH	EC ($\mu\text{S}/\text{cm}$)
Compost	18.48	1.04	17.91	7.8	863.2

Ratio of 1:5 compost: deionized water on a mass basis used for pH and EC.

2.2 Soil carbon and total nitrogen

Soils were collected in August 2022 prior to compost application at sampling depths of 0-10, 10-30, 30-50, 50-75, and 75-100 cm. A 3.8 cm diameter auger was used for baseline sampling while a 5.5 cm auger was used in 2024 and 2025, and each plot was sampled in three locations, with a single replicate to 1 m depth and two replicates to 50 cm depth. Pretreatment soils were analyzed for total soil C and N concentrations, microbial biomass carbon, pH, EC, and texture. Compost was applied on March 23, 2023, and March 20, 2024. Soils were sampled approximately one year after each application of compost on March 19, 2024 (year 1) and April 9th, 2025 (year 2).

All soils were prepared for soil total C and N analysis by air-drying, sieving to 2 mm, and grinding to a fine powder with mortar and pestle. Carbonate presence was tested by adding 2M HCl to the processed soil, with 30-100 cm samples exhibiting a slight effervescent response. C and N concentrations were measured on an elemental analyzer (Costech ECS 4010 CHNS-O Elemental Analyzer coupled with a ThermoFisher Delta-V Plus continuous flow isotope ratio mass spectrometer, Costech Analytical Technologies, Inc., Valencia, CA) at the Stable Isotope Ecosystem Laboratory of UC Merced. Elemental concentrations were converted to content by using the equivalent soil mass method (ESM) which used a cubic spline function to estimate cumulative soil C content in the absent of bulk density measurements (Wendt & Hauser, 2013). For each sample year the ESM used the soil masses from that sample year as its reference soil mass values below 30 cm. For the 0-10 cm and 10-30 cm depths, bulk density measurements from 2025 were used for 2024 and 2025 soil layer reference masses, as bulk density measures only existed for 2025. For the baseline 2022 soils masses from that sample year were used as its reference as bulk density was not taken prior to compost additions. This was done to correct for the difference in auger sizes and soil masses collected in years 2024 and 2025 compared to 2022.

Inorganic C was not detected in these soils at determined by the addition of 4 N HCl and lack of observed effervescence. Soil pH, electrical conductivity, and texture were measured on subsamples that were composited by depth and plot. Soil pH and electrical conductivity were measured using probes in a 1:2 soil:deionized water ratio and 1:5 compost:deionized water ratio (McLean, 1982). Texture was determined using the hydrometer method (Gee & Bauder, 1986).

2.2 Microbial biomass carbon

Soil microbial biomass (MBC) was measured pretreatment and at the year one and year two timepoints at 0-10 cm and 10-30 cm. MBC was assessed using the chloroform fumigation method (Brookes et al., 1985). Once sampled, soils were immediately placed into a dry ice container for transit. Samples were processed within 24 hours after collection. For each sample, 10 g of soil was placed in a 105°C oven for 48 hours to determine gravimetric moisture content. Then 20 g was placed in a beaker for chloroform extraction and 20 g placed in specimen cups for TOC determination. For TOC, 80 ml of 0.5M K₂SO₄ was added then shaken for one hour at 200 rpm before filtering with a Whatman #42 filter and frozen at -20°C until analyzed. The remaining 20 g of soil was placed in a vacuum desiccator with 30 ml of chloroform and evacuated three times. These soils were then placed in the dark for five days, and then filtered in the same manner as the TOC solution above. When ready for analysis, samples were diluted by four times with deionized water and analyzed on a total organic C analyzer (Shimadzu TOC-L, Kyoto, Japan) at the UC Merced Environmental Analytical Laboratory. Microbial biomass C was calculated as the difference between the C content of the nonfumigated and fumigated soils, using an extraction efficiency factor of 0.45 (Beck et al., 1997).

2.3 Available nitrogen

Three soil samples from each plot were collected and analyzed for available N (NO₃⁻ and NH₄⁺) once per week in the month following each compost application, and once a month throughout the rest of the year. Samples were collected from the top 10 cm using a 3 cm diameter probe and were transported in a cooler with ice to the lab where they were extracted by combining 75 ml of 2M KCl and 20 g of soil and then placed on a shaker table at 200 rpm for one hour. Once the specimen cups finished shaking, the solution was poured through a Whatman #1 filter. Filtrate was then stored at -20°C until analysis on a microplate reader (Agilent BioTek Gen5 Microplate Reader, Agilent Technologies, Santa Clara, CA, USA). Concentrations of NO₃⁻ were quantified using the hydrazine method, and concentrations of NH₄⁺ were measured using the sodium nitroprusside spectrophotometric method (Downes, 1978; Mulvaney, 1996). A 10 g subsample of each soil was placed at 105°C for 48 hours to quantify gravimetric water content and used to express available N on a dry mass basis.

2.4 Soil greenhouse gases and NO_x gas fluxes

Soil fluxes of carbon dioxide (CO₂), nitrous oxide (N₂O), and methane (CH₄) were measured using a cavity ring-down laser spectrometer (Picarro G2508, Picarro, Inc., Santa Clara, CA) connected to a closed system static chamber (26 cm diameter by 15 cm tall) made of polyvinyl chloride. Soil NO_x fluxes were measured using a chemiluminescent analyzer (Serinus 40 NO_x analyzer, Acoem, Richmond, VA) using a closed chamber technique. The PVC collar for each chamber was inserted

approximately 3 cm in the soil at the beginning of each sampling day, and chamber tops were placed on collars during flux measurements.

Two baseline gas sampling days were conducted, one in January 19, 2023, and one on March 22, 2023, the day before the first compost application. No pre-treatment differences in soil gas fluxes were detected during these timepoints. Following each of the annual compost application events, soil gas fluxes were measured the next day, twice per week for four weeks, and then biweekly thereafter. Soil gas fluxes were measured in two locations within each plot. Both soil NO_x and greenhouse gas emissions were measured at each sampling spot for five minutes. Measurements occurred between 9 am and 1 pm on each sampling day, and the starting gas sampling position was shuffled each sampling date to reduce systematic impacts of weather on gas emission data. Measurements did not occur during irrigation timing, with most irrigation events occurring before 8am.

Greenhouse gas fluxes were calculated using Picarro's Soil Flux Processor software using a linear model or the Hutchinson and Mosier exponential model when changes in concentration were nonlinear (Hutchinson & Mosier, 1981). Soil NO_x fluxes were calculated by fitting a linear curve to concentrations over time. Cumulative gas emissions were calculated by linear interpolation between sample timepoints and summation over the study period. CO₂-eq was calculated using the IPCC Sixth Assessment Report values for global warming potential at 100 years of CO₂ = 1, CH₄ (biogenic/non-fossil) = 27, N₂O = 273 (IPCC, 2021b).

2.5 Turfgrass NDVI and Soil VWC, EC, and temperature

On each gas sampling day, every sampling plot was measured in triplicate for the NDVI, VWC, EC, and soil temperature at 12 cm and 20 cm depth. Turf greenness and NDVI was measured by placing a sensor (FieldScout TCM500, Spectrum Technologies) flush with the turfgrass surface. VWC, EC, and soil temperature were measured using a FieldScout TDR350 time domain reflectometry moisture meter.

2.6 Statistical analysis

Statistical analyses and data visualization were conducted using the R programming language in RStudio (R Core Team, 2020). Linear-mixed effects models were used to incorporate changes in properties with respect to blocks and temporal variation using the *emmeans* function in the *emmeans* package while Pairwise Tukey's Honestly Significant Difference (Tukey HSD) *post hoc* tests were used to discern significances between treatments after analysis of variances (ANOVA) were run among all treatments. An alpha value of 0.05 was used for significance testing. Visualization

was performed using the “ggplot2” and “ggpubr” packages in R for plots (Kassambara, 2020; Wickman, 2016).

3. Results

3.1 Soil health indicators

There was no significant pre-treatment differences across plots or within depths, with an average C and N concentration of $3.9\% \pm 0.2\%$ and $0.38\% \pm 0.02\%$ respectively, in the 0-10 cm (Table S4-1). There were no significant pre-treatment differences in soil C, N, or MBC. After two years of compost application at two different rates, there was no significant difference in soil C or N concentration at any depth (Table S4-1, Figure S4-1). The C to N ratio (C:N) did not differ by treatment or year, and increased with depth (Figure S4-2).

There was no difference in soil bulk density in 2025 for 0-10 cm or 10-30 cm by treatment ($p > 0.05$). Across all treatments, there was significantly lower bulk density at the 0-10 cm with an average of $1.04 \text{ g/cm}^3 (\pm 0.05)$ compared to $1.27 \text{ g/cm}^3 (\pm 0.06)$ for the 10-30 cm depths ($p < 0.05$). Soil C stocks increased over time for the control ($p < 0.05$) and low application rate ($p < 0.1$) in the 0-10 cm from baseline to 2024, and in the high application rate at 50-75 cm ($p < 0.1$) and 75-100 cm ($p < 0.05$) from baseline to 2025 (Figure 4-1). Over the whole profile, there was no difference in total soil profile C stocks in any year by treatment ($p > 0.05$), or between years when aggregated across all treatments ($p > 0.05$, Figure S4-3). Soil N stocks followed the same patterns as soil C stocks as C:N did not vary by treatment (Figure S4-4).

MBC was significantly higher at the 0-10 cm depth compared to the 10-30 cm depth for all years ($p < 0.05$) across all treatments. Control soils had higher MBC at 10-30 cm than both high compost plots in baseline data and low compost plots in 2025 within the same respective years (Figure S4-5, $p < 0.05$).

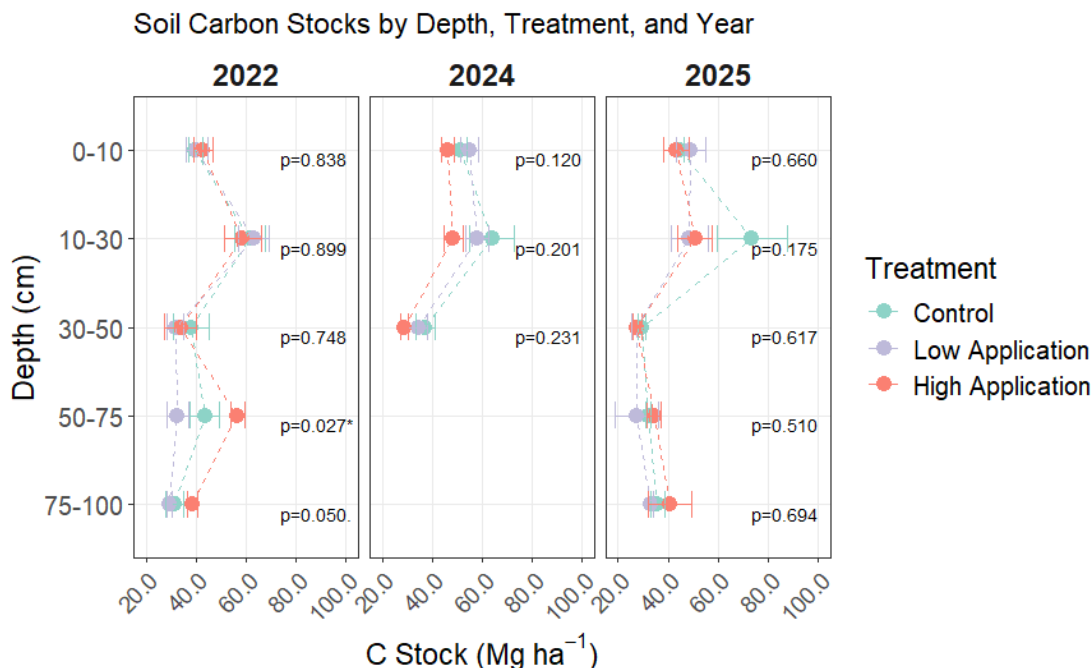


Figure 4-1: Soil carbon stocks (Mg C /ha) by treatment, depth, and year.

The only significant impact on soil available N came from recycled water irrigation impacts. Soil NO_3^- decreased ($p < 0.05$) in the two months after recycled water irrigation ceased in year one, near day 200, for both compost treatments compared to the control plots (Figure S4-6).

Soil pH did not vary by treatment or between year one and two ($p > 0.05$, Figure S4-7). Across all years and treatments, soil pH was lowest in the top 10 cm at 7.6, and then dramatically rose and stabilized between 8 and 8.5 for the 10-100 cm depths. Soil EC was significantly increased in the top 50 cm in baseline sampling compared to year one and two sampling, likely due to sampling at the end of the irrigation season (for 2022) compared to the beginning (2024 and 2025, Figure S4-8). There was no difference by treatment or between years one and two among all depths for EC.

We found no significant treatment effect on real-time VWC, EC, or soil temperature, all measured at both 12 cm and 20 cm, over two years (Figures S4-9, S4-10, and S4-11). Composted soils did have an observable increase in EC in the week after compost application, but this was short-lived (Figure S4-10). The only difference observed for NDVI was seasonal where larger decreases were seen in control plots compared to both compost treatments in the summer of 2024, beginning 400 days and 40 days after the first and second application of compost, respectively (Figure S4-12). This gap lasted through October of 2024 (approximately day 600), being present for most of the summer and early fall of 2024 (Figure S4-12).

3.2 Soil gas emissions

Soil CO₂ flux increased with seasonal soil temperature increases, particularly in year two (Figure S4-13). The summer of year two is where we see the biggest treatment differences, with CO₂ fluxes increasing with application rate (Figure S4-13). Cumulative soil CO₂ flux over the two-year period after compost applications was highest in high application rate (40.7 kg/m², $p < 0.05$), followed by low application rate (37.8 kg/m²) and control (37.5 kg/m²) (Figure 4-2A).

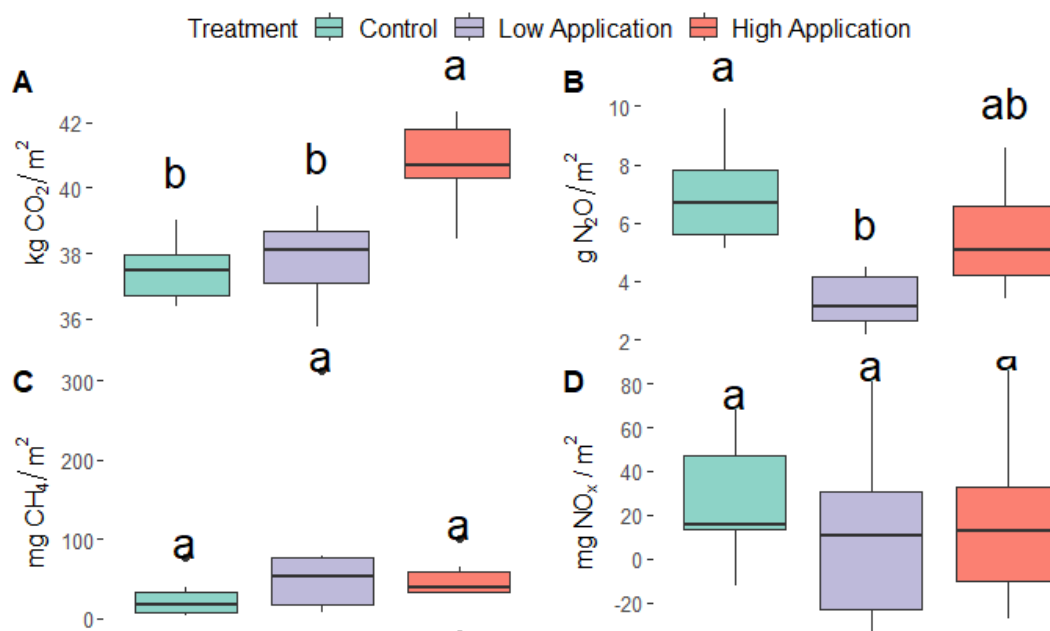


Figure 4-2: Cumulative soil emissions of (A) CO₂, (B) N₂O, (C) CH₄, and (D) NO_x between control (green), low application rate (purple), and high application rate (red) treatments.

There was no significant treatment difference in cumulative CH₄ emissions between control ($26.1 \text{ mg/m}^2 \pm 11.3$), low compost application rate ($86.2 \text{ mg/m}^2 \pm 46.8$), or the high compost application rate ($43.3 \text{ mg/m}^2 \pm 16.4$, Figure 4-2C). Soil N₂O fluxes were characterized by hot moments, with high variability and a pulse in emissions in the late summer and early fall of year two and during inorganic fertilization events, particularly in control plots (Figure S4-14). Cumulative soil N₂O post-application was significantly less from all compost plots compared to control plots ($p < 0.05$). The emissions were highest in the control plots ($7 \text{ g N}_2\text{O/m}^2$), followed by high application rate ($5.5 \text{ g N}_2\text{O/m}^2$, $p > 0.05$) and low application rate ($3.3 \text{ g N}_2\text{O/m}^2$, $p < 0.05$, Figure 2B).

Cumulative emissions expressed as CO₂-eq was significantly higher in the high compost application plots (42.3 ± 0.7 kg CO₂-eq/m²) compared to the low application rate (38.7 ± 0.5 kg CO₂-eq/m²) and control plots (39.4 ± 0.6 kg CO₂-eq/m², $p < 0.05$). Across all treatments CO₂ emissions accounted for 96% of the CO₂-eq across all treatments with N₂O accounting for 3.6%, the rest (0.4%) accounted for by CH₄ (Table S4-2). Across both rates of application, compost plots had a significantly higher share of their CO₂-eq derived from CO₂ emissions (95.2% vs 97.1%, $p < 0.05$) and a significantly lower share of CO₂-eq derived from N₂O emissions (4.8% vs 3%, $p < 0.05$, Table S4-2) compared to control plots. When excluding CO₂ emissions, the CO₂-eq was significantly lower for the low compost application rate plots (0.9 ± 0.1 kg CO₂-eq/m²) compared to the control (1.9 ± 0.2 kg CO₂-eq/m², $p < 0.05$) and high application rate (1.5 ± 0.2 kg CO₂-eq/m², $p < 0.1$) plots, while there was no difference between high application rate and control plots ($p > 0.05$). Differences in CO₂-eq when excluding CO₂ for the low application rate plots were driven entirely by differences in N₂O emissions and not CH₄ emissions (Table S4-2).

Cumulative soil NO_x emissions post-application for the control (25.9 ± 12.4 mg NO_x/m²), high application rate (17.6 ± 17.9 mg NO_x/m²), and low application rate (11.8 ± 16.7 mg NO_x/m²) plots were not significantly different across treatments (Figure 4-2D).

4. Discussion

4.1 Effects of compost on plant health, microbial properties, and soil health indicators

Compost additions to managed soils often results in observable benefits to multiple biological, physical, and soil health indicators (Tully & McAskill, 2020). As a source of organic matter, compost can improve nutrition retention, provide an energy source for microorganisms, and improve porosity and aggregation of soil (Albiach et al., 2000; Annabi et al., 2007). However, improvements to soil health indicators are heavily influenced by baseline soil conditions. Turfgrass ecosystems are intensively managed and tend to have high levels of soil organic matter due to extensive root biomass (Phillips et al., 2023; Qian et al., 2010). We found that the additional benefits to soil health from compost additions to turfgrass systems were marginal. Soil VWC was not found to vary with either treatment level, despite common claims to compost's ability to improve soil water capacity, (Carter, 2007; Serra-Wittling et al., 1996). It is possible that with the high level of native OM, these sandy loam soils may have already been near their improvable limit for water storage, but more specific research into high OM soils physical properties and structure post-compost application is needed (Minasny & McBratney, 2018).

Relatedly, we observed no effect of compost application on turfgrass health as indicated by repeated field-based NDVI measurements (Figure S4-12). Turfgrass NDVI was high among all treatments before and after application, so it is possible that the turf greenness was already at the ideal color. This field site was likely at the

peak coloration due to fertilizer and recycled water irrigation supplying plentiful N to the soil system for chlorophyll synthesis (Mangiafico & Guillard, 2007). However, it is possible above or belowground biomass may have increased with compost, as other turfgrass studies have found, but we were unable to directly measure net primary production due to the active use of our field site (Evanylo et al., 2016).

The microbial biomass content of the soils did not change with compost additions though values with high overall (Figure S4-5). For context, in a grassland study with 20 years of biosolids additions MBC averaged 517 mg C/ kg soil and 157 mg C/ kg of soil in the 0-10 cm and 10-30 cm respectively, while we found that MBC averaged 608 ± 57 mg C/ kg of soil and 235 ± 23 mg C/ kg of soil (Villa & Ryals, 2021). Finding this high of an MBC pool, without even observing any treatment differences, showcases how active and large the microbial biomass of turfgrass fields may be prior to compost additions and compared to other systems (Shi et al., 2006).

While MBC did not change with compost additions, we found soil CO₂ emissions increased with compost. Increasing CO₂ emissions could occur without a change in MBC due to more active soil microbes or community composition changes (Calderón et al., 2001; Xiao et al., 2019). It is also possible that changes to MBC may have occurred during specific moments we did not capture, such as shortly after compost additions when CO₂ fluxes in compost plots were highest (Kraut-Cohen et al., 2023a). While cannot confirm that MBC was higher shortly after compost additions, increased soil CO₂ fluxes immediately after compost additions indicated higher levels of microbial activity. High microbial activity is further supported as this site received plentiful N from fertilizer and irrigation, allowing microbes to quickly decompose the added compost OM as there was likely not nutrient limitations. Microbial activity may have been limited by C, rather than N, prior to compost addition (Mooshammer et al., 2014). The hypothesis that microbes were C limited can be supported by the observed decrease in N₂O emissions with compost, pointing to a tighter N cycle in the face of unchanging MBC. Changes in these N₂O emissions could also be explained from changes in microbial composition, which we did not measure, such as by increasing the proportion of denitrifiers. While MBC did not vary in any year or treatment, changes in both CO₂ and N₂O emissions point to altered microbial activity within compost plots.

4.2 Effects of compost additions on soil carbon and nitrogen

We observed no change in soil C and N concentration or stocks between the treatments to 1 m. Total C stocks to 1 m in year two were 216 ± 9 Mg C/ha, 182 ± 22 Mg C/ha, and 197 ± 3 Mg C/ha for control, low compost, and high compost plots respectively ($p > 0.05$, Figure S4-3). Changes in bulk density and soil mass per layer within the ESM calculations can propagate within C concentrations and stocks as ESM relies on mass-depth relationships. In this study, we saw no difference in bulk density between treatments in year two. However, there are possible treatment effects on bulk density trends from baseline to year two that we did not capture that could

lead to uncertainty in our ESM C stocks. Despite this, our year two measure bulk density values were consistent and within the expected range for urban turfgrass soil bulk density values, which suggests deviations from our ESM calculations are likely to be small (Malone 2023). Across all treatment types the average C stock of 192 ± 6 Mg C/ha represents a relatively high level of soil C even for a turfgrass systems, as soil organic C stocks of 110 and 127 Mg C/ha were estimated Baltimore and Denver turfgrass systems, respectively (Pouyat et al., 2009). These high levels of soil C stocks are without any observed gains from compost applications. As there were not changes in soil C stocks, it possible that the compost was decomposed rapidly, evident by a spike in CO₂ flux over the first 80 days after the second year of application in the high compost plots ($p < 0.05$, Figure S4-13). The greatest increase in soil CO₂ emissions occurred with the high rate of compost application, suggesting greater microbial activity and decomposition that led to the lack of increase in soil C despite the addition of compost C. This rapid-level of decomposition was likely possible from the high level of baseline soil MBC and from the use of recycled water, in excess of 100 Kg N / ha / yr, which enables microbes to quickly decompose the applied compost (Azeem et al., 2020; Gill et al., 2022). Azeem et al. (2020) found that N-induced respiration in turfgrass soils increased by 94% in compost amended soils compared to control soils, explaining the quick loss of the applied compost OM in our N rich study system.

4.3 Compost Impacts on Soil Nitrogen Emissions

While the compost did not change soil total or extractable N, cumulative N₂O emissions decreased with compost additions while cumulative NO_x emissions did not change (Figure 4-2B, 4-2D). By application rate, cumulative N₂O was significantly lower in the low compost plots relative to control plots but not in the high application rate. Soil N₂O emissions are variable with time and in this system short-lived pulses of N₂O were observed following the annual application of synthetic fertilizer (Molodovskaya et al., 2012). Control plots had significantly more N₂O emissions than both compost plots beginning after the synthetic fertilizer addition in year two (January 2024) prior to the second compost application, which lasted until the recycled water additions stopped for the year (October 2024, Figure S14). The increase in N₂O emissions in the month(s) after fertilizer additions and in the months with frequent recycled water N additions were likely what drove the differences in cumulative N₂O emissions from high application rate versus control plots. Compost studies have found that compost alone or compost in addition to inorganic fertilizer can reduce N₂O emissions relative to fertilizer alone (Alluvione et al., 2010; Ding et al., 2013). However, these studies are often based on applying the same amount of total N between treatment, and our compost plots received more total N loading due to all treatments continuing their standard recycled water irrigation and annual fertilization.

Compost application can reduce N₂O emissions by relieving microbial C limitations which allows for less “leaky” loss of N₂O during decomposition, or by increasing the proportion of nitrifying and denitrifying microbes leading to more complete denitrification (Firestone & Davidson, 1989; Lazcano et al., 2021). As compost allowed microbe to meet their C needs, less N₂O may have been lost through denitrification and nitrification processes (Firestone & Davidson, 1989; Mooshammer et al., 2014). Another pathway for N₂O emissions that was unmeasured is NH₃ volatilization, which can lead to N₂O emissions elsewhere through future N deposition. However, NH₃ volatilization from green waste compost is typically low, so this is likely to be a recycled water mediated process in our study system as recycled water can lose up to 20% of the water’s NH₃ content through volatilization (Nicholson et al., 2017; Saez et al., 2012). Most of the N₂O impact in our system is likely to come from direct N₂O emissions, and if compost can reduce the C need in N rich environments then there exist important implications for long-term gas emissions in systems with exogenous N additions (Kraut-Cohen et al., 2023b; Lazcano et al., 2021; Ros et al., 2006).

Understanding how recycled water irrigation and compost application – both management strategies many urban areas are considering – impact N₂O emissions is vital for supporting both plant production and soil health (Chen et al., 2015). The decrease in N₂O emissions occurred without a change in real-time measured VWC or available N. By not decreasing VWC or available N while reducing the N₂O emissions, compost was able to continue the soil’s support of plant production without increasing its global warming impact.

4.4 Changes in Net Climate Impact Through Compost Application

To assess the climate impact of compost application we assessed changes in both soil gas emissions and soil C stocks in our study. Compost plots had higher CO₂-eq than control plots, but this was driven by an increase in CO₂ emissions, which in our system are biogenic (Table S4-2). Cumulative CO₂ emissions were 10% higher in high compost plots, accounting for 75 kg of CO₂ over two years per high compost plot. On the plot level, this works out to be 24.6 kg C emitted as CO₂ per high application plot over the two years of this study. During this time, these plots had compost (with 19% C) applied at 28.5 Mg/ha annually for two years, equating to an application of 11 kg C per high application plot over the two years of this study. In these plots, the CO₂-C emitted can account for the compost C applied, supporting the idea that rapid decomposition of the applied compost led to an increase in CO₂ emissions. The remaining increase in CO₂ emissions could be due to increased root respiration, though we did not observe increased plant quality or growth via NDVI, or the priming effect (Li et al., 2024; Sun et al., 2019). Loss of native SOC can occur when exogenous inputs with moderate to high C:N are added to the system, leading to mining of the native SOC for additional nutrients (Sun et al., 2019; Xu et al., 2024). However, we saw no significant change in soil C stocks by treatment, supporting that the increase in CO₂ emissions is likely driven by compost breakdown and plant

respiration, and not through a significant level of native SOC priming (Ogle et al., 2015). When we assess overall carbon GHGs, we saw no change in soil CH₄ emissions after compost addition. We did observe a 20-50% reduction in cumulative soil N₂O emissions when compost was applied to these turfgrass systems. Additionally, a smaller proportion of net CO₂-eq emissions were composed of N₂O in composted plots relative to controls (Figure 4-2B, Table S4-2). Observing a decrease in soil N₂O emissions after compost additions is highly relevant to soil management and assessment of climate solutions in managed systems. While models often predict increases in soil N₂O fluxes with compost additions, field-based measurements from agricultural and rangeland systems provide evidence for the contrary. Many compost additions experiments observed no change or a reduction in soil N₂O emissions following compost addition (Lazcano et al., 2021; Ryals & Silver, 2013). Studies in agricultural systems have shown that benefits to soil health last, such as the decrease in N₂O emissions, both after a one-time application and after consistent longer-term compost additions, but it is unknown if they do in a higher C and N systems like turfgrass (Atoloye et al., 2022; Nichols et al., 2024).

From this assessment, short-term application of compost in turfgrass soil may be a net climate benefit due to reduced N₂O emissions, not from any gains in soil C stocks. Gains in soil C stocks have been observed in other high C baseline turf studies, but they often have significantly lower starting soil C stocks than our site (Azeem et al., 2020; Beesley, 2012; Wang et al., 2022). Even without gains in soil C, which are often the focus of compost studies, decreases in N₂O emissions are significant for climate benefit assessments.

5. Conclusions

This study supports the ability for compost to mitigate climate change through reduced nitrous oxide emissions in highly managed soils. While compost increased soil CO₂ emissions, these emissions are biogenic and the finding of reduced nitrous oxide emissions is vital for climate change mitigation in soils with high nitrogen loading. There was also no change in NO_x, despite no change in nitrogen availability or volumetric water content. These are important for supporting compost's ability to continue to support plant production, without increasing the amount of nitrogen lost as N₂O. However, we saw no change in other soil and plant health properties such as soil carbon stocks, microbial biomass, or turf greenness. It is possible that in a system with high baseline soil health, more than two years of compost additions are required to accrue soil health benefits. Despite the lack of increasing soil carbon stocks, N₂O emission reduction and many potential co-benefits, such as reduced landfill emissions, exist with compost application to turfgrass systems. Understanding if this reduction in N₂O emissions and lack of soil carbon stock change continues after additional years of compost applications will be vital to better constrain the impacts on compost application in the long-term. In the short term, our findings show that compost provides an important avenue to reuse of organic wastes and to mitigate climate change within public parks.

CRedit Authorship Contributions Statement

Zachary Malone: Conceptualization, Writing – original draft, Writing – review & editing, Formal analysis, Investigation, Data curation, Visualization, Software. **Samuel Lee:** Writing – review & editing, Investigation. **Rebecca Ryals:** Conceptualization, Writing – review & editing, Supervision, Funding acquisition.

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Conflict of Interest Statement

The authors declare no conflicts of interest.

Data Availability Statement

Data and code available will be made available on Dryad upon manuscript publication.

Supporting Information

Additional supporting information and figures can be found online in the supporting information section of this article.

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Chapter 5: Long-term compost application increases carbon stocks and mineral-associated organic carbon in deep urban soils

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Abstract

Compost is used to improve agricultural soil health and for its potential to sequester carbon. Emerging climate policies aim to divert organic material from landfills to prevent methane emissions, increase composting as a circular nutrient strategy, and incentivize the use of compost as a soil carbon amendment to enhance soil carbon sequestration. Intensively managed urban soils covered in turfgrass, such as recreational parks, may be strategic land uses for increased compost application. However, these soils typically have high initial soil carbon concentrations, and the potential to further increase soil carbon is unknown. Additionally, there are few studies on the long-term compost application and the impacts to deep soil carbon cycling. We sampled 14 fields in recreational parks in Pleasanton, CA; eight with long-term compost application (20 and 30 years) and six that had never received compost application. We found that unamended control sites had high carbon stocks (239 Mg C ha⁻¹ to 100 cm). Soil carbon stocks were 23-29% higher in parks that received long term compost applications, with 294 Mg ha⁻¹ and 309 Mg ha⁻¹ of soil carbon at the parks receiving compost for 20 and 30 years, respectively ($p < 0.05$). These gains were concentrated in the 0-10 cm and 75-100 cm depths, highlighting the ability for surface-applied compost to improve deep soil C over time. We found increases of up to 70% in the C content of the mineral fraction with in the 75-100 cm with compost additions compared to control soils ($p < 0.05$). These results show that long-term compost application can increase persistent soil carbon to at least 1 m in high organic matter soil systems and that new C from compost application can enter the mineral fraction on decadal timescales.

1. Introduction

Compost is a slow-release fertilizer used to increase soil carbon (C) and improve soil health (Goldan et al., 2023; Hargreaves et al., 2008; Malone et al., 2023). However, most research on compost and other exogenous organic matter (OM) additions occur on short timescales (<5 years) which limits our understanding of long-term and persistent changes in soil C. Some studies on the long-term (>20 years) impacts of OM applications have shown soil C increases not only in the topsoil, but also at depth and in the mineral associated pool (Mayer et al., 2022; Tian et al., 2017; Villa et al., 2024). Several mechanisms lead to deep soil transit of C including irrigation, tillage, and deeply rooted plants (Sokol et al., 2022a). Once C arrives at depth, C can be more readily stabilized in the mineral associated pool than in upper layers as the fraction of fine silt, clay, and Fe/Al oxides increases throughout the profile (Rumpel & Kögel-Knabner, 2011; Villa et al., 2024). These findings suggest that compost additions influence long-term C dynamics in some managed ecosystems, but significant work remains as few replicated studies have been conducted.

There are limited data on the impacts of long-term compost additions throughout a deep soil profile (up to 1 meter) across most ecosystems. While intensively managed urban soils are a strategic place for compost use, previous studies have been short (typically three or fewer years) and limited to shallow soils (Malone et al., 2023). Urban land is approximately 5% of total US land area. A large portion of urban land is intensively managed with turfgrass cover, constituting 2% of total US land area (16,000,000 ha) (Z. Liu et al., 2014; Milesi et al., 2005). Turfgrass systems often have high soil C concentrations, particularly in the topsoil, due to their rapid plant growth above and belowground which is commonly aided by fertilization and irrigation (Phillips et al., 2022; Qian & Follett, 2012). Soil organic C concentrations in mature turfgrass soils often exceed 5% in the top 10 cm with increases in the first 30 years after establishment, reaching an apparent limit around 50 years post-establishment (Phillips et al., 2022). Given relatively high soil C content, there is uncertainty in how much additional C mature turfgrass soils may store under compost additions. Understanding soil C saturation limits C accrual, even if limits are soft slowdowns rather than hard caps on soil C, is important to understand total soil C capacity, particularly as cities incorporate C sequestration into climate action plans (Georgiou et al., 2022, 2025).

Studies that find a limit in turfgrass soil C accrual, however, have only analyzed bulk soil C. This omits both how the persistence of soil C may change with long term turfgrass management as well as changes in the long cycling mineral fraction, despite mineral associated OM constituting 65% of global soil C (Sokol et al., 2022b). One concern is that the C gains observed in turfgrass soils may be due to increases in particulate organic matter (POM), the least persistent pool, due to accumulation of undecomposed root and grass fragments from mowing (Kerek et al., 2002). In one of the few studies of mineral-associated organic matter in urban turfgrass ecosystems, Kerek et al. (2002) observed increases in mineral-associated organic matter of turfgrass soils over time (30 years) in the top 15 cm of the soil, but

to a lesser extent than gains in POM (Kerek et al., 2002). Across land uses, even high quality soils are not close to their mineral-associated C capacity, though gains of C in this pool may be more difficult to detect than gains in the POM pool (Georgiou et al., 2022). This makes the study of mineral-associated organic matter vital in long-term turfgrass systems.

Deep soil (>30 cm) C dynamics are greatly understudied compared to top soils (<30 cm), especially in urban soils (Malone et al., 2023; Pries et al., 2023; Yost & Hartemink, 2020). This gap persists despite knowledge that over 50% of soil C is stored below 30 cm and that deep soils can be actively managed to sequester C. C cycling in deep soils is governed principally by mineralogy and inputs from shallower soil layers. When the topsoil is managed for high plant and microbial biomass, vertical C transit supports organo-mineral formation at depth (Pries et al., 2023). This potential for topsoil organic C translocation is notable in urban parks that receive long-term irrigation, which drastically changes the residence time and storage of C in the soil system (Han et al., 2010; Jobbágy & Jackson, 2000; Skadell et al., 2023). Compost application could further increase deep C storage in soils with long-term irrigation as it allows for additional dissolved organic C (DOC) transit, with Wright et al. (2008) observing increased DOC for 29 months after compost application in turfgrass soils (Stergiadi et al., 2016; Wright et al., 2008). Thus, deep soil sampling is vital in compost applied turfgrass systems where irrigation may aid transit. Restricting soil sampling at 30 cm or less would lead to an incomplete understanding of deep soil C dynamics in a soil system that has high C translocation potential.

Our objective was to assess the impacts of long-term (20 and 30 years) additions of compost on total soil and mineral-associated organic C in urban turfgrass systems to a 1 m depth. We hypothesized that 1) unamended soils would have high soil organic C concentrations (>3%), with older parks containing more C due to additional years of high levels of plant OM inputs. 2) Turfgrass soils with long-term compost additions would have greater amounts of total soil C, as well as greater amounts of mineral-associated organic matter, relative to unamended soils. Compost itself will supply additional OM into the soil system and support additional plant production that further increases total C input to the soil. With irrigation this will lead to more C transit, where C is more likely to be associated with minerals. 3) Organic matter from compost will be incorporated into mineral associated organic C pools, including at depth, as there is a high ability for translocation due to long-term irrigation.

2. Materials and Methods:

2.1 Study sites and experimental design

The study sites are located in the city of Pleasanton, California (37.6604° N, 121.8758° W) on publicly managed athletic parks with turfgrass cover. A total of 14 fields were sampled (six unamended and 8 compost applied fields). Three fields had 20 years of compost application while five fields had 30 years of compost application. Compost was derived from green waste feedstocks and was sourced from the same

vendor (Recology Blossom Valley Organics, Vernalis, CA). The 20 year site is estimated to have received a total of 68.6 Mg C / ha (2.8 Mg C / ha / yr) of compost additions while the 30 year site is estimated to have received 42.7 Mg C / ha (1.4 Mg C / ha / yr) from compost additions. Unamended parks have no record of receiving compost for at least 30 years. All parks receive additional nutrients either from mineral fertilizer or recycled water. Each park receives irrigation three times a week, herbicides and pesticides annually, and share similar grass species with seed mixes that are applied at similar rates. All parks are used for soccer, except for one which is used for baseball outfields. The park age ranges from approximately 59 years to 15 years (Table 5-1). These parks are also placed throughout the city, leading to four soil orders and numerous soil textures and taxonomic classes being represented in the sample set (Table 5-1). The factors within soil taxonomic classes in urban soils are important in determining soil function, but can be less informative in urban systems due to heavy anthropogenic disturbance and management (Ajmone-Marsan et al., 2016).

A total of 384 soil samples were collected between March 4 and March 12, 2024. Soil was sampled at six locations within each field at various depths. Sampling locations within each field were randomly selected with the criteria that each sampling location was at least 10 m away from the next closest sampling location. Soil samples were collected with a 5.5 cm diameter auger at the following depth intervals: 0-10, 10-30, 30-50 for three of six sampling locations and 0-10, 10-30, 30-50, 50-75, and 75-100 cm for three additional sampling locations. At each location that was sampling to 100 cm, an additional sample was collected for microbial biomass C determination at depths of 0-10 cm and 10-30 cm. The first sampling location of each field was sampled with a quantitative bulk density corer at depths of 0-10 and 10-30 cm.

Table 5-1: Description of field site and management practices. Soil sample total represented by n.

Site	Compost Application Rate	Fertilizer	Irrigation Type	Year of Opening	Taxonomic Class	n
Amaral	0	Yes	Regular	1997	Fine-silty, mixed, superactive, nonacid, thermic Mollic Xerofluvents	24
Creekside	0	No	Recycled Water	2002	Fine-silty, mixed, superactive, nonacid,	24

						thermic Mollic Endoaquepts	
Muirwood	0		Yes	Regular	1970	Fine, smectitic, thermic Xeric Endoaquerts	24
Upper Pleasanton	0		Yes	Regular	1965	Fine, smectitic, thermic Mollic Palexeralfs	24
Val Vista	0		Yes	Recycled Water	2004	Fine, smectitic, thermic Xeric Endoaquerts	24
Bernal	0		Yes	Regular	2009	Fine, thermic Typic Calcicquolls	24
Amador	20 204 total	Years, Mg/ha	Yes	Regular	1970	Fine-silty, mixed, superactive, nonacid, thermic Mollic Endoaquepts	72
Ken Mercer	30 153 total	Years, Mg/ha	No	Recycled Water	1970	Fine, thermic Typic Calcicquolls	120

2.2 Soil processing, pH, EC, and soil texture

Soil pH, electrical conductivity, and texture were measured on air dried subsamples that were composited by depth and plot. Soil pH and electrical conductivity were measured using probes in a 1:2 soil:deionized water ratio (McLean, 1982). Soil texture was assessed using the PARIO plus particle size analyzer, which uses the integral suspension method (PARIO, Meter Group, 2021). On day one 30 ml of 30% hydrogen peroxide (H₂O₂) and 30 ml of DI water was added to 40 g of soil and left to react overnight to destroy organic matter. Soils that had a high SOM% (5-12%) had this process repeated (Acevedo et al., 2021)..These samples were placed on a hot plate at 60°C for four to six hours before adding 100 ml of sodium

hexametaphosphate and 250 ml of DI water. The following day, the samples were mixed for five minutes and then added into the sedimentation cylinder where the PARIO device was used to monitor the pressure change over 2.5 hours. After this a subsample of the suspension was taken while the remaining solution was wet sieved using 2000 – 500 μm , 500 – 250 μm , and 250 – 53 μm sieves. After drying in an oven at 105°C for 48 hours, these beakers were weighed, had their tare masses subtracted, and the remaining dry solid masses were input into the PARIO software to assess the particle size distribution.

Soils were also air-dried, sieved at 2 mm, and finely grounded with mortar and pestle to prepare for C and nitrogen (N) concentration and isotopic analysis. Carbonate presence was tested by adding 2M HCl to the soil, with no sites exhibiting an effervescent response. An elemental analyzer (Costech ECS 4010 CHNS-O Elemental Analyzer coupled with a ThermoFisher Delta0V Plus continuous flow isotope ratio mass spectrometer, Costech Analytical Technologies, Inc., Valencia, CA) at the Stable Isotope Ecosystem Laboratory of UC Merced was used for C and N analysis. Elemental concentrations were converted to content by using the equivalent soil mass method (ESM) which used a cubic spline function to estimate cumulative soil C content in the absent of bulk density measurements (Wendt & Hauser, 2013). Reference soil layer masses were calculated for each field to control for differences in soil layer masses and soil density between different parks.

2.3 Microbial biomass C

Microbial biomass C (MBC) was performed using the chloroform fumigation method on the 0-10 cm and 10-30 cm depth samples for three of six replicates per site (Brookes et al., 1985). Once sampled, soils were immediately placed into a cooler with dry ice and transported to UC Merced. For each sample, 10 g of soil was placed in a 105°C oven for 48 hours to determine gravimetric moisture content. Then, 20 g was placed in a 50 ml beaker for chloroform extraction and 20 g placed in 150 ml specimen cups for total organic C (TOC) and total N (TN) determination. For the 150ml specimen cups, 80 ml of 0.5 M K_2SO_4 was added to each cup, then each were shaken for one hour at 200 rpm. After shaking for one hour, the solution was filtered through a Whatman #42 filter paper (2.5 μm pore size) and filtrate was frozen at -20°C until analysis.

The remaining 20 g of soil in the 50 ml beaker was placed in a vacuum desiccator along with a 100 ml beaker with 30 ml of chloroform and boiling chips. The desiccator was then evacuated three times with the last boiling for 60 seconds without venting. The desiccator was then placed in the dark for five days, and then extracted with 80 ml of 0.5 M K_2SO_4 , filtered, and frozen. Prior to analysis, samples were thawed and diluted to a 1:5 ratio (sample: water) and analyzed on a Shimadzu total organic C analyzer (Shimadzu TOC-L, Kyoto, Japan) at the UC Merced Environmental Analytical Lab for both TOC and TN. Microbial biomass C was calculated as the difference between the C content of the nonfumigated and fumigated soils, using an extraction efficiency factor of 0.45 (Beck et al., 1997).

2.4 Density fractionation

Density fractionation was performed on a subset of samples; one sample of the 0-10, 30-50, and 75-100 cm depths was fractionated at each unamended site while three samples per depth was fractionated at each of the long-term composted fields. Density fractionation was done in a similar manner to Swanston 2005 and Villa 2024 with three pools of C fractionated (Swanston et al., 2005; Villa et al., 2024). The free light fraction (FLF) and occluded light fraction (OLF) constitute the POM pool while the heavy fraction (HF) makes up the mineral-associated organic matter pool. Samples were density separated using a 1.85 g/cm^3 solution of sodium polytungstate (SPT). 75 ml of SPT was added to 20 g of air-dried soil which was then inverted, allowed to sit for 45 minutes, and then centrifuged at 3470 rpm for 1 hour. The FLF was then aspirated, filtered through a $0.8 \mu\text{m}$ membrane filter under vacuum, and rinsed five times with 100 ml of deionized water through the filter to remove SPT particles from the FLF. FLF was then placed on an aluminum dish in a 55°C oven for 48 hours, weighed, and transferred to a glass vial for storage. The remaining solution was then mixed at 1700 rpm with a benchtop mixer and sonicated at 280 J/ml (282 J/ml , $\text{sd} = 16.1$) to separate the OLF from the HF. Cylinders were placed in an ice bath during sonication to avoid overheating the soil-SPT solution. After sonication, the cylinder sat for 45 minutes and was then centrifuged at 4000 rpm, aspirated, rinsed, and dried similar to the FLF. To remove the HF, 150 ml of deionized water was added to the cylinder and shaken vigorously, centrifuged for 5 minutes at 3470 rpm, and then the sides were rinsed. After this, the solution was centrifuged at 3470 rpm for 20 minutes and then the solution was aspiration down to the remaining soil pellet. This was done 5 times to rinse SPT from the HF, after which the HF was moved to an aluminum dish and placed in a 105°C oven for 24 hours. The overall mass recovery averaged $85\% (\pm 1)$ and this did not vary by treatment or depth (Table S5-1, $p > 0.05$). The estimated C recovery was $70\% (\pm 10)$ for density fractionated soils, with no variation by treatment (Table S5-1, $p > 0.05$), but with more mass loss at depth (Table S5-1, $p < 0.05$). This larger estimated C loss at depth is within reported literature values for deeper soils which contain a greater proportion of mineral-associated organic matter that could be susceptible to physical dispersion during fractionation and rinsing steps, leading to the loss of colloidal or dissolved organic carbon (Crow et al., 2007; Helbling et al., 2021). The soil C stocks for each fraction were calculated by taking the ESM soil layer stocks and multiplying them by the proportion of C within each fraction for the same corresponding field. If a field had not been density fractionated, then it was removed from the average for the purposes of calculating DF C stocks.

2.5 Radiocarbon analysis

Radiocarbon (^{14}C) analysis was conducted on the HF of density fractionated soils for the depth intervals of 0-10 cm, 30-50 cm, and 75-100 cm. HFs containing 0.5 to 1 mg C were placed in 6 mm quartz tubes with roughly 250 mg of copper oxide to provide oxygen for combustion and a small amount of silver to eliminate sulfur interference. This tube was then placed in a 9 mm quartz tube, placed on a vacuum line to remove

atmospheric gases, and flame sealed with a hydrogen torch. After sealing, tubes were placed in an oven at 900°C for at least five hours to convert all soil C to CO₂ gas. The CO₂ gas was graphitized, using H₂ as a reducing agent (Vogel et al., 1984). Samples were then run on the NEC Compact Accelerator Mass Spectrometer at Lawrence Livermore National Lab for ¹⁴C analysis (Broek et al., 2021). Radiocarbon isotopic values are corrected for mass dependent fractionation using measured $\delta^{13}\text{C}$ values and reported in Fraction Modern (Fm) (“Discussion,” 2004; Reimer et al., 2004) and in $\Delta^{14}\text{C}$ notation (Stuiver and Polach, 1977).

2.6 Diffuse reflectance infrared Fourier transform spectroscopy (DRIFTS)

Diffuse reflectance infrared Fourier transform spectroscopy (DRIFTS) was performed on the HF samples obtained from the density fraction procedure. This was done using a Fourier transform infrared (FTIR) spectrophotometer with an attached Praying Mantis DRIFTS attachment to capture spectra. Finely ground potassium bromide (KBr) powder was used as a background reference material prior to analysis and ran after every 12 samples. Mineral fraction samples were ground finely with a mortar and pestle and oven dried at 105°C prior to analysis. We performed 300 scans per sample at 4000 to 400 cm⁻¹ that were averaged and background corrected. Assessment of the spectra and operation of the instrument was performed using the OPUS software (version 7.5.18; Bruker Optik GmbH, Ettlingen, Germany). The resulting absorbance peak areas within spectra were then integrated within three function groups by wavenumber ranges; C–H stretching compounds representing simple plant OM (2976–2898 cm⁻¹ and 2870–2839 cm⁻¹), aromatic C = C compounds representing complex plant OM (1550–1500 cm⁻¹), and a carboxylic C = O stretch group representing microbial OM (1660– 1580 cm⁻¹) (Parikh et al., 2014, Mainka et al., 2022).

2.7 Statistical analyses

All statistical analyses and data visualization was done using RStudio (R Core Team, 2020). Data was tested for normality using the Shapiro-Wilks test. Pairwise Tukey’s Honestly Significant Difference (Tukey HSD) *post hoc* tests were used to discern significances between compost treatments after analysis of variances (ANOVA) were run among all treatments. An alpha value of 0.05 was used for significance testing. Visualization was performed using the “ggplot2” and “ggpubr” packages in R for plots (Kassambara, 2020; Wickman, 2016, p. 2). Correlation matrices were constructed using the “corrplot” package (Wei & Simko, 2024). Code to assess DRIFTS spectra was originally written by Dr. Steven Hall (Hall et al., 2018).

3. Results:

3.1 Soil texture, EC, and pH

Across unamended sites, there was no significant change in the content of any size fraction between depths, with large variability overall ($p > 0.05$, Figure 5-1, Table S5-1). For amended samples, there was a decrease in sand content as depth progressed ($p < 0.05$), accompanied by an increase in fine silt and total silt ($p < 0.05$) with subsequent depths (Figure 5-1, Table S5-1). Coarse silt was only higher when comparing 10-30 cm to 30-50 cm and 30-50 cm to 50-75 cm ($p < 0.05$). The clay content was significantly higher in the 0-10 cm than in both the 10-30 cm and 30-50 cm ($p < 0.05$), while the 75-100 cm depth had significantly more clay content than the 10-30 cm and 30-50 cm depths ($p < 0.05$), but not the 0-10 cm ($p > 0.05$). Between the two compost application lengths, there was significantly more total sand from 0-50 cm at 30 year site while the inverse was true at 75-100 cm ($p < 0.05$). Both coarse and fine silt content were roughly doubled at the 30-50 cm for the 20 year site ($p < 0.05$) and overall silt content was higher from 10-75 cm for the 20 year site ($p < 0.05$). Clay content was higher at the 20 year site for the 10-30 cm ($p < 0.05$), but was higher from 50-100 cm at the 30 year site ($p < 0.1$). Mirroring these depth trends, combined silt and clay content had a 20% relative increase in mass at the 20 year site from 10-50 cm ($p < 0.05$), but had a 10% relative increase in mass from 75-100 cm at the 30 year site ($p < 0.05$).

Comparing treatments, the total sand content was higher in unamended soils at both the 50-75 cm (41% vs 22% total soil mass) and 75-100 cm (37% vs 14% total soil mass) depths ($p < 0.05$). This was accompanied by increased coarse silt (at 0-10 cm and 50-75 cm), fine silt (at 50-75 cm and 75-100 cm), and total silt (at 50-75 cm and 75-100 cm) ($p < 0.05$). Clay content had marginally significant higher value in the 0-10 cm depth at amended sites at 29% compared to 21% ($p < 0.1$). Combined silt and clay content was significantly higher at the amended sites for 50-75 cm (59% vs 77% total soil mass) and 75-100 cm (62% vs 86% total soil mass) compared to unamended samples ($p < 0.05$).

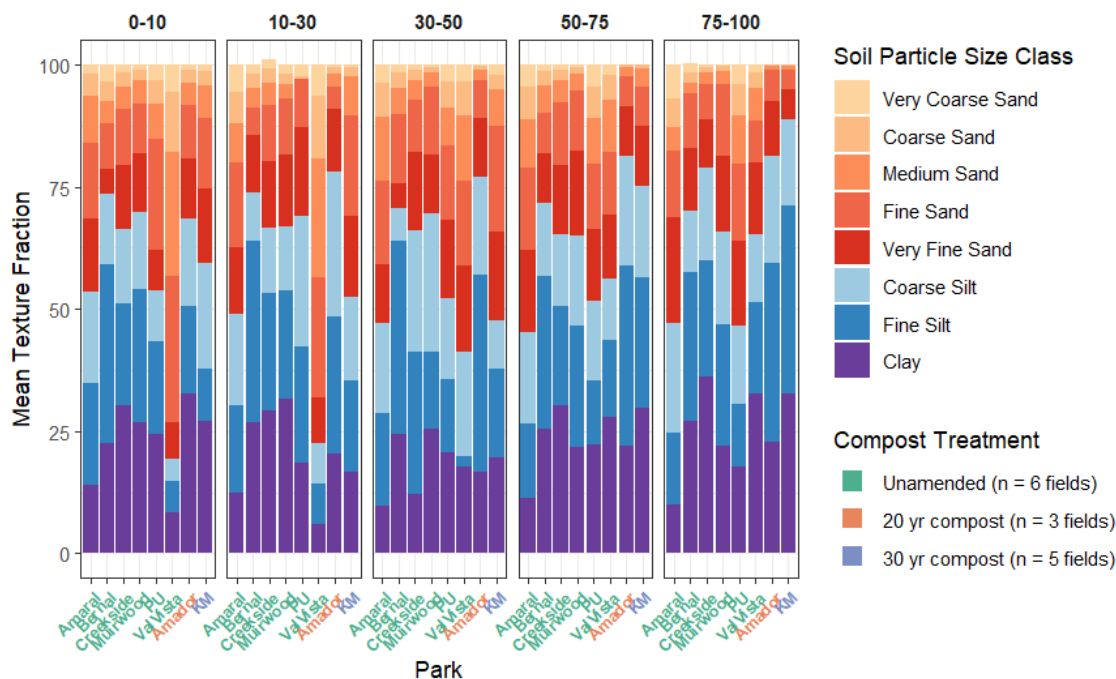


Figure 5-1: Soil particle size class distribution by park, compost treatment, and depth. Color on the x-axis tick labels correspond to green as unamended, red as 20 years of compost, and blue as 30 years of compost additions. Color of soil particle sizes have reds corresponding to sand classes, blues as silts, and purple as clay.

Across all sites, pH increased with soil depth from 7.09 ± 0.07 at 0-10 cm to 8.2 ± 0.09 at 75-100 cm. The 30 year site had significantly higher pH at 10-30 cm (8.27 ± 0.05) than the unamended (7.73 ± 0.11 , $p < 0.05$) and the 20 year samples (7.78 ± 0.06 , $p < 0.05$) but there were no other differences by treatment (Figure S5-1A). EC was highly variable and relatively consistent to 1 m within unamended plots ($p > 0.05$, Figure S1B). Within compost treatments, the EC was lower when comparing the 0-10 cm to all lower depths for both compost sites ($p < 0.05$, Figure S5-1B). The EC within compost sites had a marginally significant decrease when comparing the 30-50 cm depth to the 75-100 cm depth ($p < 0.1$, Figure S5-1B). Despite these decreases at depth within the compost profile, there was no significant difference by treatment effect for within depth comparisons ($p > 0.05$).

3.3 Soil C and N

Bulk soil C concentrations were significantly higher at 0-10 cm at the 20 year site and amended sites overall and higher in the 75-100 cm at both compost amended compared to unamended soils ($p < 0.05$, Table S5-2). From 10-75 cm the 20 year site had a significantly higher soil C concentration than the 30 year site, and significantly higher compared to unamended soils from 30-75 cm ($p < 0.05$, Table S5-2).

Soil C stocks were significantly higher at the 30 year site compared to unamended soils in the 0-10 and 75-100 cm depths ($p < 0.05$, Figure 5-2). This held true when

comparing all composted soils to all unamended soils, as there was significantly more C stocks in the 0-10 cm ($p < 0.05$) and in the 75-100 cm ($p < 0.05$) in composted soils. The remaining depths were not significantly different between treatments. Cumulative soil organic C to 100 cm was least in the unamended soils ($239 \pm 15.3 \text{ Mg C ha}^{-1}$), followed by 20 years of compost application ($294.1 \pm 12.8 \text{ Mg C ha}^{-1}$) and then 30 years of compost ($309.7 \pm 28.7 \text{ Mg C ha}^{-1}$). These differences were driven both by increased soil C concentrations (Table 5-2) and changes in soil mass (Figure S5-2), though bulk density directly measured in the 0-10 and 10-30 cm were not significantly different by treatment ($p > 0.05$). The 0-10 and 75-100 cm depths had a significant increase in C% in overall compost plots and in the 30 year soils compared to unamended ($p < 0.05$), while the 30 year site also had significantly increased soil mass in the 0-10 cm over both 20 years and unamended samples ($p < 0.05$). Soil mass was significantly lower in the 75-100 cm depth at both compost sites compared to unamended ($p < 0.05$), despite stocks being significantly higher at the depth for the 30 year site ($p < 0.05$).

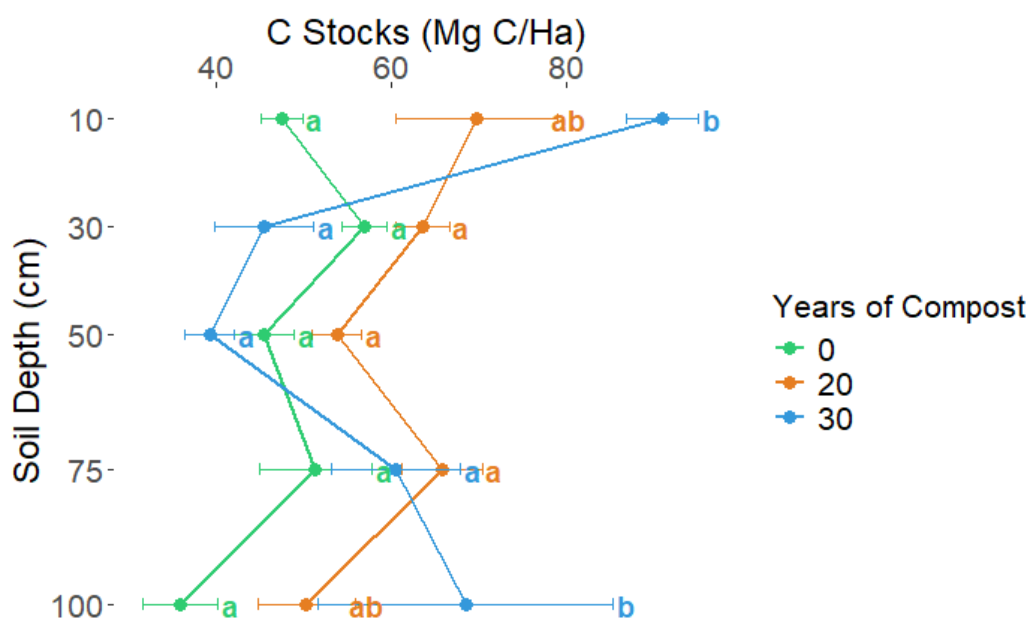


Figure 5-2: Soil C stock (Mg C/Ha) by years of compost application and depth. Letters indicate Tukey HSD significances while error bars represent SE.

Soil N was significantly higher in the sites amended with compost for 30 years compared to the unamended parks at 0-10 cm ($p < 0.05$) and 75-100 cm ($p < 0.05$) depths, but similar to the 20 year compost parks (Table 5-2). When comparing all compost soils to unamended soils, compost soils had significantly more N in only the 0-10 cm depth ($p = 0.08$). The $\delta^{15}\text{N}$ was significantly higher at both the 20 and 30 year site compared to unamended soils in the 0-50 cm depths ($p < 0.05$) and marginally higher when comparing all amended samples to unamended samples in the 50-75 cm ($p < 0.1$, Table 2).

The soil C:N ratio increased with depth at the unamended and 20 year composted sites, but showed no trend by depth at the 30 year site (Table 5-2). Between treatments, the C:N was significantly lower from 0-30 cm at 30 years compared to unamended soils ($p<0.05$) while it was significantly higher from 50-100 cm for the 20 year site compared to unamended soils ($p<0.05$).

Table 5-2: Table of C, N, and C:N by depth and years of compost application. SE indicated by $\pm$.

Treatment	Depth (cm)	Carbon (g/kg soil)	Nitrogen (g/kg soil)	C:N
Unamended	0-10 cm	49.5 $\pm$ 2.1	4.8 $\pm$ 0.2	10.3 $\pm$ 0.13
	10-30 cm	17.7 $\pm$ 1.6	1.7 $\pm$ 0.2	10.2 $\pm$ 0.14
	30-50 cm	11.1 $\pm$ 0.7	1.1 $\pm$ 0.1	10.6 $\pm$ 0.20
	50-75 cm	8.9 $\pm$ 1.1	0.8 $\pm$ 0.1	10.7 $\pm$ 0.46
	75-100 cm	7.2 $\pm$ 0.8	0.7 $\pm$ 0.1	10.7 $\pm$ 0.73
20 Years	0-10 cm	68.4 $\pm$ 3.2	6.7 $\pm$ 0.3	10.1 $\pm$ 0.11
	10-30 cm	20.6 $\pm$ 0.9	2.1 $\pm$ 0.1	9.9 $\pm$ 0.25
	30-50 cm	14.6 $\pm$ 779	1.3 $\pm$ 0.1	11.3 $\pm$ 0.35
	50-75 cm	15.0 $\pm$ 1.0	1.2 $\pm$ 0.1	12.7 $\pm$ 0.57
	75-100 cm	12.5 $\pm$ 1.0	1.0 $\pm$ 0.1	13.1 $\pm$ 0.79
	0-10 cm	55.0 $\pm$ 2.8	5.6 $\pm$ 0.3	9.7 $\pm$ 0.07
	10-30 cm	14.4 $\pm$ 1.7	1506 $\pm$ 0.2	9.6 $\pm$ 0.09

30 Years	30-50 cm	7.7 ± 0.5	0.7 ± 0.1	10.6 ± 0.23
	50-75 cm	9.4 ± 0.7	0.9 ± 0.1	10.6 ± 0.22
	75- 100 cm	13.2 ± 1.5	1.3 ± 0.1	10.1 ± 0.20

3.5 Density fractionation

In the 0-10 cm depth, there was marginally more FLF C ($p < 0.1$) and significantly more N stocks at the 30 year site compared to the unamended and 20 year site ($p < 0.05$, Figure 5-4) with a slightly higher percentage of the bulk soil mass found in the FLF at the 30 year relative to the 20 year site ($p < 0.1$, Table S5-3). There was marginally more OLF C stocks across both compost sites compared to unamended soils ($p < 0.1$) and significantly more N stocks compared to unamended soils ($p < 0.05$, Table S4). For the heavy fraction, there was marginally more N stocks at 30 years compared to unamended ($p < 0.1$, Table S5-4) but there were no differences for the 20 year site.

There were no differences in the FLF, OLF, or HLF in the 30-50 cm among any of the C and N variables measured among treatments (Table S5-3, S5-4, S5-5).

The FLF at the 75-100 cm depth had a significantly higher N concentration compared to both the unamended and 20 year soils ($p < 0.05$, Table S5-3) but no difference in FLF N stocks or C:N. There was no difference among treatments in the OLF fraction at the 75-100 depth (Table S5-4), but there were many in the HF (Table S5-5). The C concentration of the HF was marginally higher in the 30 year vs unamended soils ($p < 0.1$) and significantly higher when all compost samples were compared with unamended ($p < 0.05$, Table S5-5). The N concentration of the HF was significantly higher in the 30 year vs unamended soils ($p < 0.05$) and when all compost samples were compared with unamended ($p < 0.05$, Table S5-5). This also led to marginally more N stocks in the 30 year site vs unamended soils ($p < 0.1$) though there was no significant difference in C stocks or C:N among treatments ($p > 0.05$, Table S5-5, Figure 5-4).

The proportion of C stored within each fraction had no significant differences by treatment within any depth ($p > 0.05$). When averaging the fractions in each depth across all treatments, the proportion of FLF C had a marginally significant increase at 0-10 cm compared to 30-50 cm ($p < 0.1$), and a significant increase when comparing 0-10 cm to 75-100 cm ($p < 0.05$, Figure 5-3). There was no significant change in the proportion of soil C within the OLF fraction by depth, but the proportion of C in the HF had a marginally significant decrease in the 0-10 cm compared to the 30-50 cm and 75-100 cm depths ($p < 0.1$, Figure 5-3).

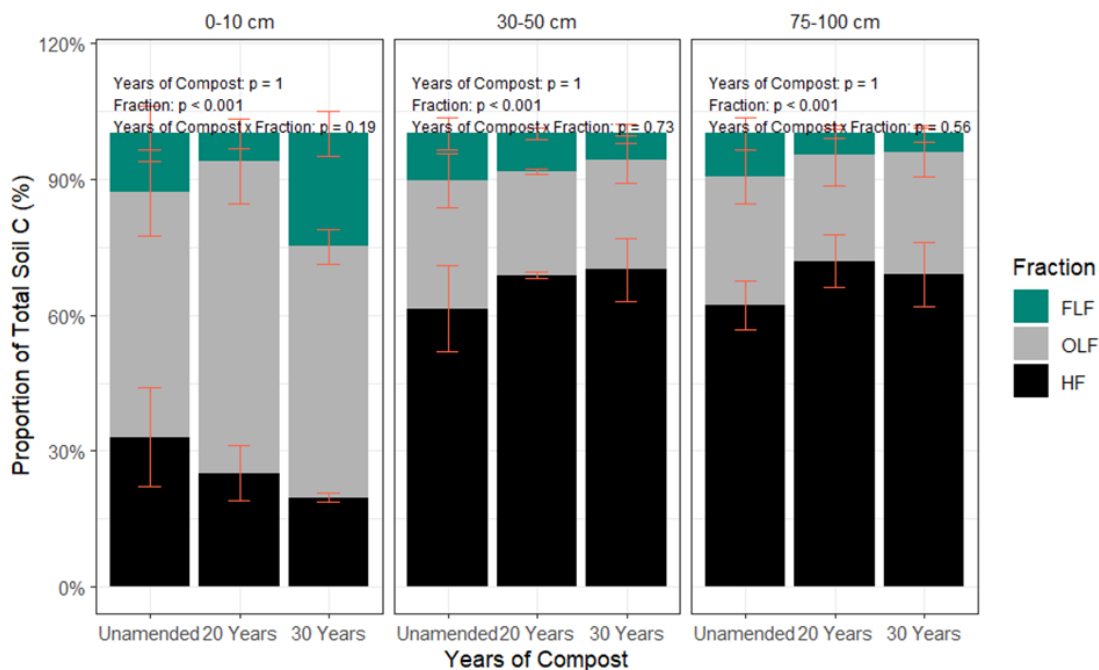


Figure 5-3: Proportion of density fractions with the FLF, OLF, and HF by depth and years of compost.

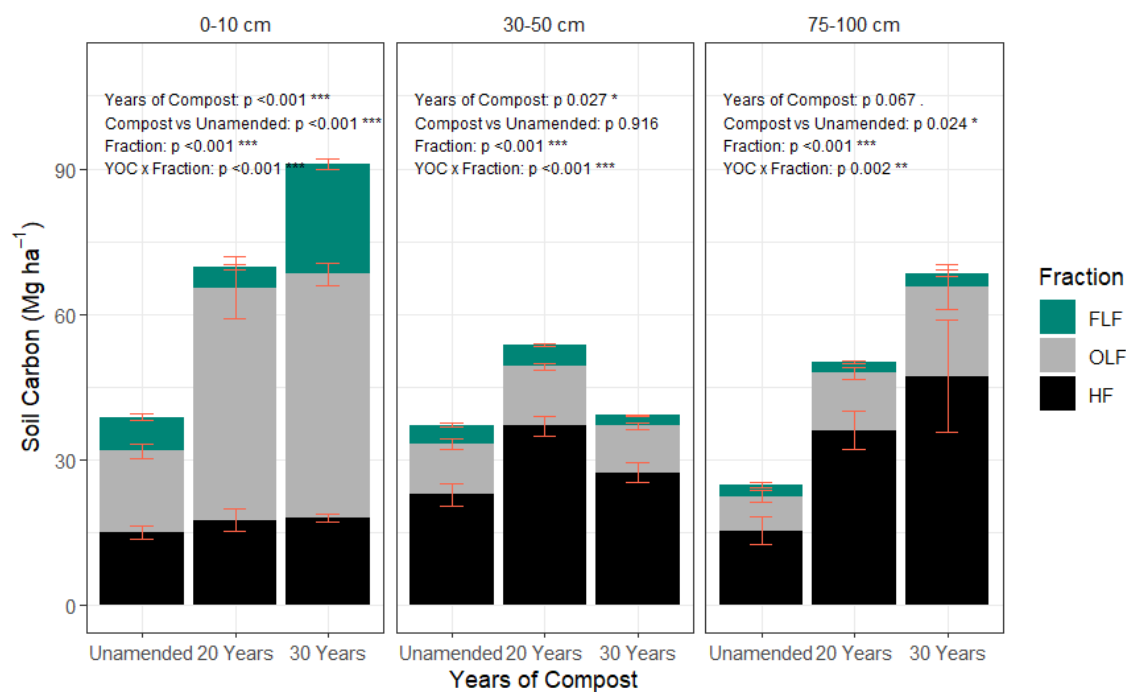


Figure 5-4: Soil C stocks (Mg C/ha) by years of compost, depth, and density fraction. YOC, FLF, OLF, and HF represent years of compost, free light fraction, occluded light fraction, and heavy fraction respectively.

3.6 Mineral Fraction Analysis: Radiocarbon and DRIFTS FTIR

The fraction modern of the mineral-associated HF did not vary by treatment at any depth (Figure 5-5, $p > 0.05$). The fraction modern at 0-10 cm was significantly higher than the fraction modern at the 30-50 and 75-100 cm depths ($p < 0.05$).

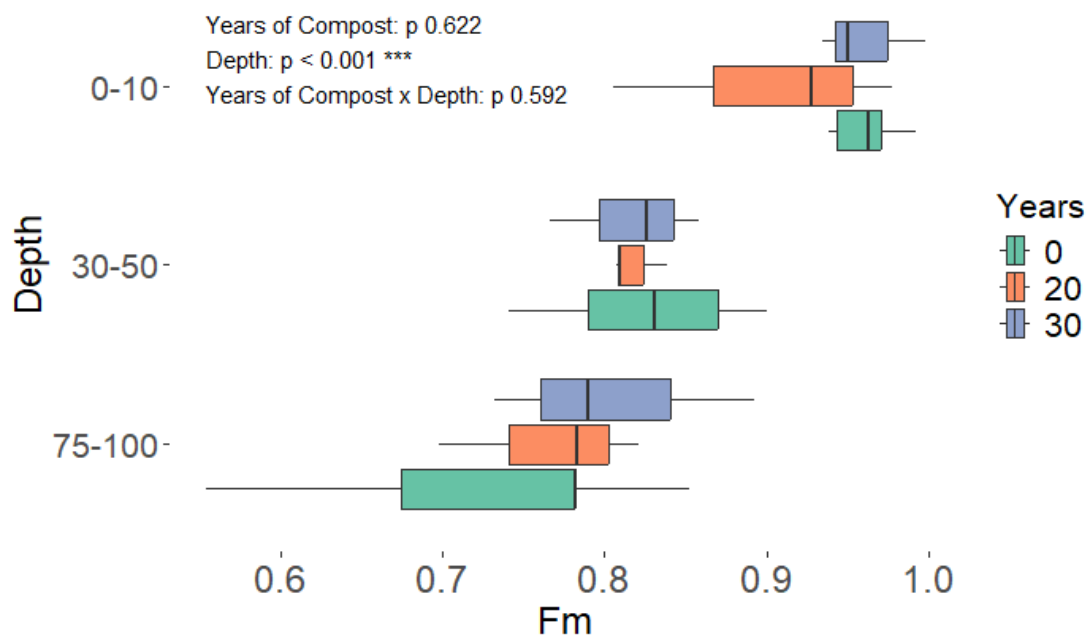


Figure 5-5: Soil fraction modern by depth and years of compost application.

To further assess changes to the mineral fraction, DRIFTS was used to molecular composition of the OM within the mineral fraction but both depths and the years of compost added (Figure 5-6). The simple plant OM to microbial OM ratio (simple:microbial) increased when assessed

d over all three depths when comparing the 20 year site to unamended soils ($p < 0.05$) and when comparing all compost sites to unamended soils ($p < 0.05$, Table S5-6). No specific depth was significant on their own, however. The simple plant OM: complex plant OM had a marginally significant increase across all compost samples compared to unamended samples ($p < 0.1$, Table S7) when assessed across all depths, but no individual depth was statistically significant. These changes shifted the OM signature of compost samples closer to that of compost alone (Figure 5-6). While there was no difference at any level for the complex plant OM: microbial ($p > 0.05$), there was for simple plant + complex plant OM: microbial OM across all depths as the 20 year and compost samples overall had a significantly higher simple plant + complex plant OM: microbial OM ($p < 0.05$), but there was no difference at the 30 year site ($p > 0.05$, Table S5-7).

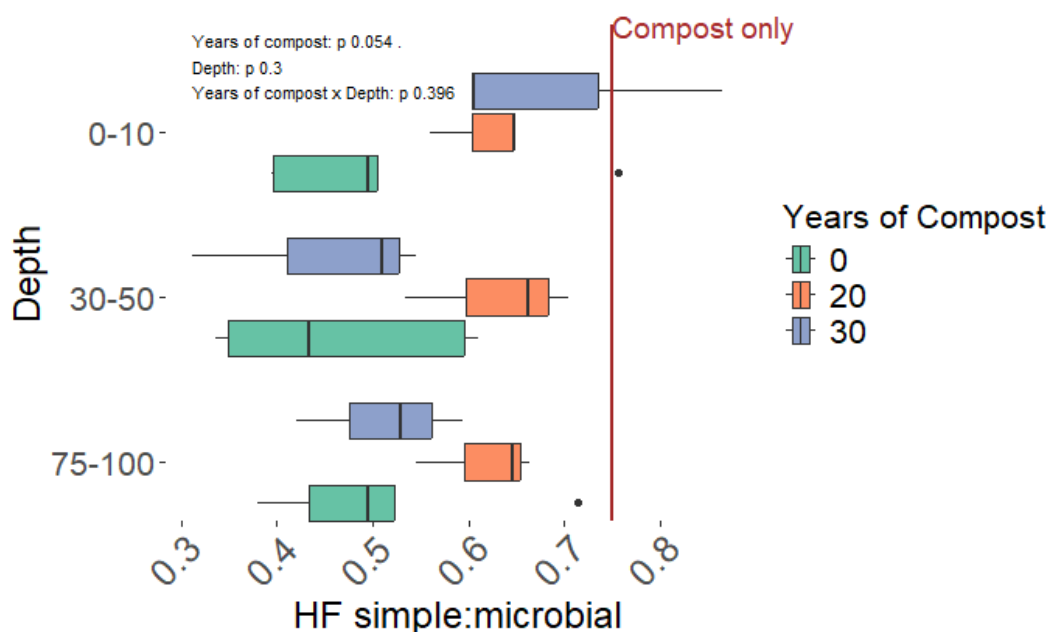


Figure 5-6: DRIFTS FTIR Simple OM to Microbial OM ratio between treatments and depths. The brown line represents the value of compost (n=3) by itself. P-values represent statistical significance of Tukey's HSD posthoc test between unamended (n=5), 20 years of compost (n=3), and 30 years of compost (n=3) treatments.

3.7 Microbial Biomass

There was no significant change in total MBC or MBN for either the 0-10 cm or 10-30 cm by years of compost additions or combined treatments relative to unamended samples ($p > 0.05$, Figure S5-3). Across all treatments, there was significantly more MBC and MBN in the 0-10 cm than the 10-30 cm ($p < 0.05$). There was a significant decrease from 5.9 to 5.1 in the MBC:MBN ratio from unamended to 20 year sites in the 0-10 cm ($p < 0.05$), with a marginally significant decrease when comparing all compost sites to unamended samples ($p < 0.1$, Figure S5-3). Across all treatments, the MBC:MBN ratio had a marginally significant increase of 5.7 to 6 from the 0-10 cm to the 10-30 cm depth ($p < 0.1$).

3.8 Relationships between C variables in fractionated soils

Correlation matrices were used to assess correlation between multiple variables. Within the 0-10 cm depth years of compost had a significant positive relationship with bulk soil N and park age ($p < 0.05$). FLF C content had a significantly positive relationship to FLF N and OLF C and N contents ($p < 0.05$) but not HF ($p > 0.05$). The MBN was negatively related to the HF N content ($p < 0.05$). While fraction modern was only done on heavy fractions, the fraction modern of the heavy fraction had a positive significant relationship to total C and N percent ($p < 0.05$). At 30-50 cm, the heavy fraction C and N contents had a positive significant relationship with bulk soil

C and N ($p < 0.05$). The FLF C had a positive significant relationship with FLF N and OLF C ($p < 0.05$), while the FLF N was positively correlated to the OLF C and OLF N ($p < 0.05$). None of the FLF or OLF C and N contents were correlated to the heavy fraction C and N ($p > 0.05$). At 75-100 cm both the bulk soil C and N concentrations were positively correlated with FLF N, OLF C and N, and HF C and N contents ($p < 0.05$) but not FLF C ($p > 0.05$). HF C and N were positively correlated to park age ($p < 0.05$) and all three of these variables, in addition to bulk soil N concentration, were positively correlated to years of compost ($p < 0.05$).

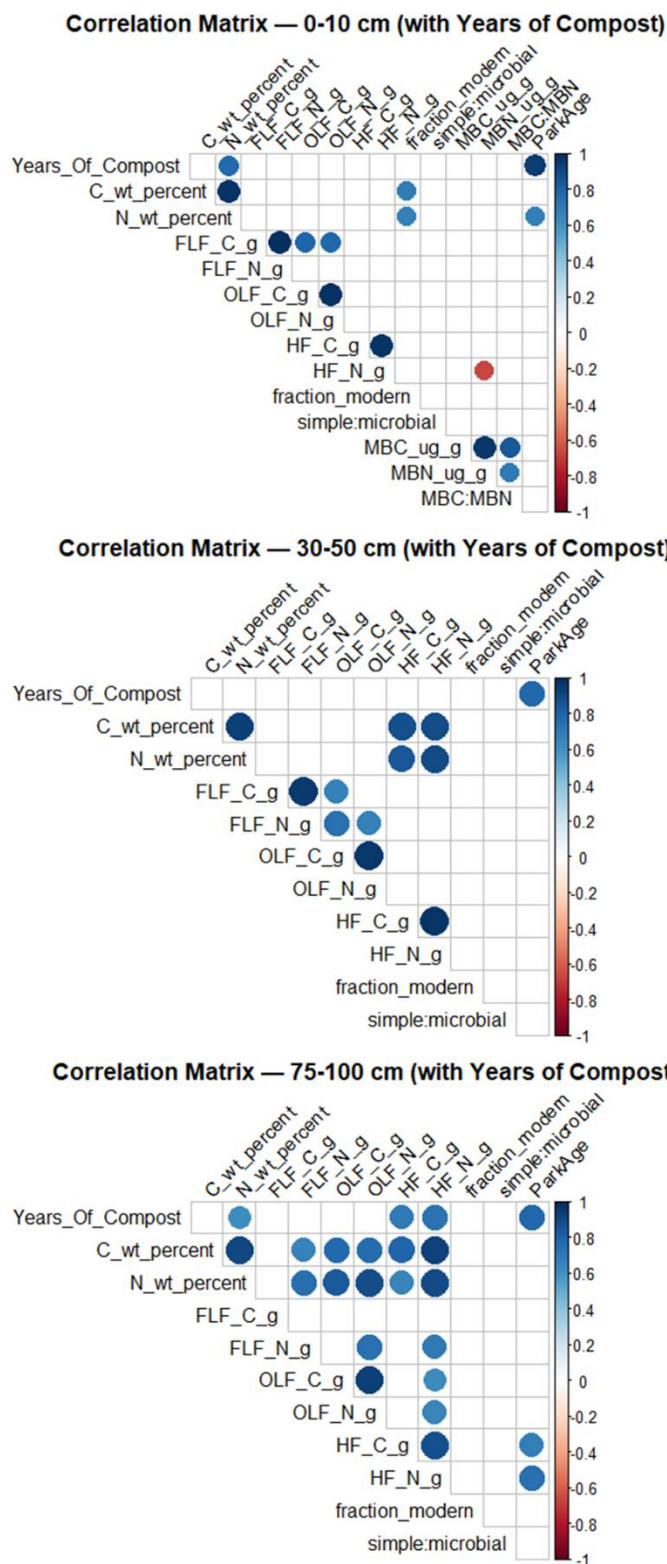


Figure 5-7: Correlation matrices in 0-10, 30-50, and 75-100 cm depths within the density fractionation dataset.

4. Discussion

4.1 Unamended Site Properties are Dependent on Soil Texture, Park Age, and Management

To understand baseline soil conditions in turfgrass systems and have a point of comparison for long-term compost sites, a focus of this study was understanding the soil health variation within parks that have never received compost amendments. The variable with the largest prediction ability for soil carbon stocks at unamended sites was combined silt and clay content, which when aggregated across all depths explained approximately 19% of the variation in whole profile soil carbon stocks ($p < 0.05$), with only marginal explanatory power from pH (4%, $p > 0.05$).

We saw no consistent impact of park age or management on soil pH, but both age and management played a role in soil EC. While the whole profile soil EC was significantly lower by park age, ($p < 0.05$), this was an artifact of irrigation strategy. The two oldest parks (average age of 55 compared to the other four average age of 21) do not receive recycled water irrigation. Recycled water contains significant amounts of salts which can dramatically raise soil EC (Toze, 2006). When we compare the four non-recycled water soils to the two-recycled water parks, we see that recycled water soils average 1531 mS/cm compared to 583 mS/cm over the whole profile ($p < 0.05$), though the only individual depth this is significant at is 75-100 cm ($p < 0.05$), suggesting solutes may be settling at the bottom of the soil profile rather than near the surface. These increases in soil EC with recycled water are to be expected, even to one meter, as recycled water contains many solutes. Qian and Lin found a decade of recycled water use to increase soil salinity by 27% (Qian & Lin, 2019). While soil EC increased, all soils were still below saline thresholds. Recycled water can change more than just salt content as the overall chemical makeup, such as increased N loading, is much different than traditional irrigation which could impact for future nutrient management and C storage (Chiou, 2008; Toze, 2006).

We found no impact of park age or management on whole profile C stocks or at individual depths above 75 cm ($p > 0.05$), though there was a negative trend at 75-100 cm with age ($p < 0.05$). This was driven by the relatively high C stocks in the topsoil of the youngest site (15 years), and that the oldest park (59 years) had the lowest stocks below 30 cm, potentially due to this site possessing the second highest amount of sand from 30-100 cm (Figure 5-1). With no overall change in total C stocks by age, this runs contrary to current literature which finds increasing C stocks with turfgrass age (Phillips et al., 2022; Qian & Follett, 2012). However, these previous studies have focused on the top 30 cm of the soil while our unamended sites show the greatest variation in C stocks across sites below 50 cm (Figure S5-4). These studies also focus on bulk soil C and we found that in the mineral fraction the oldest two sites did have the highest C concentration and content compared to all other older unamended samples, though it was not statistically significant ($p > 0.05$).

The two oldest unamended parks can act as strong references to our long-term compost parks as they are of similar age (59 and 54 years old vs 55 years for the compost parks) and management, except for the application of compost. These old

parks had a 20% relative increase in C stocks in the 10-30 cm depth ($p<0.1$) compared to the younger unamended soils, but there was no difference over the whole soil profile. When comparing these analogs to the amended parks, we observed a gain in soil C stocks with compost additions over the whole profile ($p<0.05$).

4.2 Effects of Long-Term Compost on Surface Soils

Compost impacts many aspects of soil health, especially when applied annually for decades. SOC is an important soil health indicator, as it influences biological, physical, and chemical soil functions. In the top soil, we found that long-term compost application increased soil organic C and N stocks in the 0-10 cm depths. The largest total gains in stocks at any depth were within the 0-10 cm depth, which are expected as compost has been surface applied. Without compost, unamended parks averaged 5% C while the 20 and 30 year compost sites averaged over 6% and 7% soil C respectively, which led to significantly higher C stock values at our compost sites (Figure 5-2). While there was sizable variation in C stocks the 10-75 cm depth range, there were no significant changes in soil C stocks with compost additions (Figure 5-2). This may be due to the decrease in silt and clay content in this range compared to the 0-10 cm and 75-100 cm, leading to decreased ability to increase soil C (Figure 5-1). When accounting for this change, silt and clay content were consistent predictors for soil C stocks across all soil treatments from 10-100cm ($p<0.1$), while compost is a strong predictor for soil C stocks in the surface soil (0-10 cm, $p<0.05$) and deep soil (75-100 cm, $p<0.05$). These results support a model where biological controls dominate SOC cycling in the surface horizons and mineral controls become increasingly important in subsoil layers (Pries et al., 2023).

While compost strongly increased soil carbon stocks in deep soil (75-100 cm), there was a lack of change in stocks in the mid-profile (10-75cm). Despite this lack of change, there is evidence for the translocation of compost C in the bulk soil's $\delta^{15}\text{N}$ content. The natural abundance of $\delta^{15}\text{N}$ has been used in soil compost systems to track the fate of compost N throughout the soil profile (Choi et al., 2003; Lynch et al., 2006). The composting process increases $\delta^{15}\text{N}$ with typical values ranging from +8 to +15 $\delta^{15}\text{N}$, though mature compost values are variable based on feedstock and management of a pile (Chalk et al., 2019). While we do not have data on the applied compost $\delta^{15}\text{N}$ from this study, we did observe a significant increase in the soil $\delta^{15}\text{N}$ at both compost sites relative to controls in the 0-50 cm ($p<0.05$), with a marginally significant increase at 50-75 cm ($p<0.1$, Table 5-2). It is likely that compost is changing the SOC within the mid-profile, even if it is not altering C stocks. The change in $\delta^{15}\text{N}$ can be combined with our C data in the top and deep soil to further suggest that soil profile changes are due to compost additions, not from baseline soil differences that originated from before compost additions started (Choi et al., 2003; Lynch et al., 2006).

Changes in soil $\delta^{15}\text{N}$ could occur from microbial transformations, though there were no significance differences in MBC and MBN contents ($p>0.05$). However, the MBC:MBN did decrease with compost additions in the 0-10 cm ($p<0.1$), particularly when comparing the 20 year site ($p<0.05$, Figure S3). The addition of exogenous C

may have shifted community composition towards bacteria rather than fungi (MBC:MBN of 6 vs 5 in unamended vs composted soils) while not actually increasing the overall microbial biomass, as it was already quite high prior to compost additions (Babur et al., 2022). The shift in MBC:MBN is consistent with an increase in carbon use efficiency (CUE) and decreased N limitation, though CUE was not directly measured (Sinsabaugh et al., 2016). Reduced N limitation by microbes could also increase microbial activity and turnover, potentially increasing necromass inputs to mineral-associated organic matter, which saw higher mineral C stocks at compost sites (Figure 5-4) (Sokol & Bradford, 2019). The decrease in MBC:MBN could point to additional N cycling activity and N uptake, and Ros et al. (2006) found that after 12 years of compost additions there were increases in bacterial diversity and ammonia oxidizing bacteria specifically, which are important for N cycling. In addition to N cycling microbes, Liu et al. (2022) found that overall bacterial richness increased with 25 years of compost additions as did the relative abundance of Proteobacteria and Actinobacteria, both characteristic of higher C environments. Both of these studies only assessed changes in microbial community in the top 20 and 30 cm, respectively, while we saw large relative increases in the C content up to 1 meter, which could lead to shifts in the community structure at depth. For example, studies have found that long term mineral fertilizer application can impact microbial community structure in the subsoil, up to three meters deep (Li et al., 2014; Wang et al., 2017). Assessing changes in the microbial community would give a better understanding of how changes in microbial characteristics could influence mineral associated organic matter (Sokol & Bradford, 2019).

Assessing minerally associated organic matter, and other C fractions, is a valuable tool to understand how different pools of C and N are cycling, giving more context than observing changes in bulk soil alone (Lavalley et al., 2020; Swanston et al., 2005). When we assess the mineral associated organic matter and other density fractions, we see that benefits were not evenly distributed among the three fractions: the FLF, OLF, and HF, or by depth (Figure 5-3, Figure 5-4). We found that there was significantly more C and N stocks in the 0-10 cm FLF at the 30 year site compared to both our unamended and 20 year sites, driven by increased mass of FLF rather than changing C concentration (Figure 5-3, Table S5-3). A lack of FLF C stock change, aside from the increase at 0-10 cm for 30 years of additions, could be explained by the FLF from compost application cycling into the other fractions. This could happen as turfgrass systems often have high microbial biomass and quickly cycling particulate organic matter pools, driven by turfgrass high allocation to the rhizosphere (Kaye et al., 2005; Thompson & Kao-Kniffin, 2019). We observed high MBC in our top soil, but there was no difference by compost treatment.

Evidence for FLF cycling into other fractions was observed in the 0-10 cm, where the OLF had a significant increase in both C and N stocks at both compost sites compared to unamended soils (Table S5-4, S5-6). As OLF cycles on a decadal scale, changes in this fraction may be expected as several decades of applied compost OM break down and have opportunities to form and/or be enclosed by soil aggregates. This OLF pool can also be highly varied as it is possible for relatively undecomposed OM from compost to be directly incorporated into the OLF relatively quickly, so long

as aggregation potential exists (Wagai et al., 2009). At the composted sites, there was a decrease in silt and clay fractions in the 10-75 cm depth range compared to top and deep soil, leading to less aggregation, more leaching potential, and a higher rate of mineral fraction C formation if this OM made it to the 75-100 cm depth range (Cotrufo et al., 2013). This could explain the lack of change within the FLF and especially OLF fractions below 10 cm.

When assessing the HF, we saw significantly more C stocks in the mineral pool at the 20 year site compared to unamended and 30 years at both the 0-10 cm and 30-50 cm depths (Figure 5-4). As the compost was surface-applied compost, we might expect large changes in the makeup of the top soil mineral fraction pool, and the 20 year site may change more in the surface as it received more total compost than the 30 year site. However, we did see that the molecular properties of the mineral fraction changed across both compost sites in the top 50 cm

Within the HF we also saw that the molecular properties of the mineral fraction changed within the top soil, even where the total stock of C did not. While no individual depth was significant on its own, we saw that the simple plant OM: microbial OM, simple plant OM: complex plant OM, and all plant OM: microbial OM ratios increased across all compost samples when pooled for the 0-50 cm depths ($p < 0.1$, Table S5-7). These values shifted to become more similar to the ratios found within finish compost compared to the ratios found in unamended soils. Thus, despite a lack of change in HF C stocks in some treatments in the top soil, such as the 30 year site, the HF is still being altered through long-term compost application. Another paper that assessed the molecular signature of top soil HF after 40 years of amendment additions found that that long-term compost application resulted in the mineral fraction having less OM decomposition, becoming more similar to a compost's OM signature (Mayer et al., 2022). However, they also observed an increase in topsoil mineral content, which we did not. Because our top soil mineral C and N stocks had little change at 30 years, despite observing the inclusion of compost materials in the mineral fraction through DRIFTS OM signature change, there must be mechanism(s) leveling out the exogenous C inputs. Potential explanations could range from an increased turnover rate to the translocation of C into the mineral fraction facilitated by the compost addition. In support of the latter, changes in the OM signature were present across the profile within the HF at the 30 year site and compost sites overall (Table S5-4, S5-6).

4.3 Effects of Long-Term Compost on Deep Soils

Long-term compost additions lead to dramatic changes in the deep soil (50cm+), focused on the 75-100 cm depth range. Bulk soil C and N stocks at 75-100 cm were significantly higher at the 30 year site and compost sites overall than unamended sites ($p < 0.05$), but not at the 20 year site compared to unamended soils ($p > 0.5$, Figure 5-2). This change was not evenly distributed among the three fractions of soil C. The FLF had double the N concentration, but no C change, at the 30 year site compared to unamended and 20 year samples at 75-100 cm (Table S5-6, S5-3). This increase at depth could point to the translocation of some FLF from compost

additions, but all other FLF factors at depth were unchanged. The N concentration could change alone if the compost had a low C:N but at an estimated 16:1, it did not. Another potential explanation is if there was sizable nitrate leaching. We lack data on the compost nitrate or overall N mineralization for confirmation, but the 30 year site has begun to receive recycled water irrigation while the 20 year site does not, which could explain the increase in N stocks at depth (Geisseler et al., 2021).

While slower cycling than the FLF pool, we saw more change in the OLF pool at 75-100 cm. The OLF stocks were increased when comparing both compost sites and the 30 year site alone to unamended soils (Figure 5-4). The OLF likely increased more than the FLF as these soils have sizable silt and clay content at this depth leading to a large capacity for aggregate protection of C where to reach the subsoil (Cotrufo et al., 2013).

While the mineral fraction is typically described as the slowest-cycling and slowest-changing pool, there is recent evidence that short-term changes can impact the pool greatly (Jilling et al., 2025; Torn et al., 2013). This can be exaggerated when assessed at depth as deeper soil C pools are traditionally thought of as slower cycling (Pries et al., 2023). In this study, we found large changes in soil C and N concentrations and stocks in the mineral fraction, specifically in the 75-100 depth range (Figure 5-3, Table S5-4, S5-6). While compost additions ($p < 0.1$), pH ($p < 0.05$), and combined silt and clay were strong predictors at the profile scale ($p < 0.05$), these relationships were depth dependent. Within depths, the only significant predictor of HF stocks was found in the 75-100 cm range and was caused by compost additions ($p < 0.05$), which was independent of silt and clay ($p > 0.05$) or pH impacts ($p > 0.05$). In contrast to the bulk soil C, which showed a significant increase in surface C (0-10 cm) due to compost additions, the relationship was not significant within the mineral fraction.

Despite these depth-specific effects in the mineral fraction, bulk C stocks drive much of the variation in HF C ($p < 0.05$), as the proportion of C distributed between the three fractions did not change with compost additions (Figure S5-3). This indicates that increases in mineral C from compost additions are primarily reflecting proportional increases in total SOC, rather than shifts in the partitioning of carbon within fractions. While HF C stocks changed they were not accompanied by a change in the HF organic matter signature by treatment within any depth (Figure 5-4, Table S5-7). This is consistent with the idea that deep soil C stocks may be fueled both by direct compost C and primed top soil native mineral C that then translocated downward, which would increase deep soil C stocks while having minimal impact on the OM signature (Borken et al., 2004; Sun et al., 2019). This loss of C through translocation may explain the relatively sustained stocks of C and N in the top soil despite compost additions. This translocation with frequent irrigation could also explain why gains in the subsoil are greatest in the 30 year site, which received less cumulative compost than the 20 year site, as there have been 10 additional years of potential compost C translocation. However, we do not see a change in HF radiocarbon values in top soil that one may expect if the pool is continually receiving younger C via compost and new plant production while leaching other “native” C to

depth (Figure 5-5). To better understand this phenomenon, future studies should assess how DOC amount and molecular makeup throughout the profile may change with long term compost additions. This would allow for better understanding of how compost-derived DOC, recently produced plant DOC, or native soil C that may be displaced and become DOC are contributing to changes in the mineral fraction makeup. In addition, running these DOC pool(s) and other fraction samples for radiocarbon could be used in conjunction to assess the age/source of DOC being moved and altered throughout the profile as radiocarbon in the mineral fraction alone was not definitive in this study for piecing together how the makeup of mineral fraction C changed.

Whether by DOC or another mechanism, compost is likely moving both itself and native soil C downward, explaining the relative lack of top soil mineral C increase, gain of C in the mineral fraction at depth, and why the deep C in composted sites has started to more closely resemble top soil mineral C and/or compost properties (such as in bulk ^{15}N , Table S5-2). Understanding these impacts of long-term compost additions is vital for both land managers and our climate.

4.4 Management Implications of Long-Term Turfgrass Management and Compost Use

In a high C baseline system of turfgrass, we still observed gains to one meter depth with 30 years of annual compost additions, even in the deep mineral fraction. Similar to our results, Villa et al. (2024) found that 20 years of biosolids applications in irrigated pastures increased soil C over the whole profile, and in the mineral fraction at the 75-100 cm depth. The majority of the soil studies, especially those in turfgrass systems, have focused on top soil C where a broader consensus exists for the increase in soil C that occurs with compost additions (Malone et al., 2023). Few urban soil studies have assessed soil C stocks to 1 m, and most of these are short-term, assessing soil responses to 5 years or less of OM additions. Contextualizing the C stocks present in both the unamended and compost amended sites is important for understanding how C rich these soils begin as, and just how much more C they may gain. Pouyat et al. (2009) measured SOC stocks without OM additions to 1 m in urban turfgrass systems in Baltimore, MD and extrapolated them to 1 m in Denver, CO (soil sampling ended at 30 cm in Denver, CO). They estimated turfgrass soil C stocks to be 110 and 127 Mg C ha⁻¹ to 1 m respectively, while our unamended sites averaged 239 Mg C ha⁻¹. When assessing C distribution throughout the soil, we found that the proportion of C below 20 cm at our sites were significantly more than estimated in the aforementioned paper. We estimate that across all sites and treatments approximately 65% of soil C stocks are below 20 cm (53% below 30 cm) in our study compared to 53.5% below 20 cm in Pouyat et al. (2009). This could indicate a large degree of C translocation occurs in irrigated turfgrass systems, and that deep soil C is driving some of the difference between study estimates of turfgrass soil C stocks, independent of differences in quality or management of the system (Pouyat et al., 2009). This comparison also denotes the importance of deep soil sampling, as much of the data at depth in Pouyat 2009 was based on a small sample size of extrapolated C values using grassland proportions of soil C distribution,

potentially underestimating urban turfgrass total soil C stocks (Jobbágy & Jackson, 2000; Pouyat et al., 2009).

When comparing the C stock change with OM amendment use at our sites, we see a larger increase in our study than others have previously reported for long-term study. Ryals 2015 modeled long-term compost additions impact on soil C (up to one meter), soil gas emissions, and plant production in a grassland and found a benefit of approximately $84 \pm 10 \text{ g CO}_2\text{-eq}\cdot\text{m}^{-2}\cdot\text{yr}^{-1}$, which would be roughly equal to $0.9 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$, over 30 years (Ryals et al., 2015). This modeled work was based on a shorter term study which found a benefit of 4.6 to 5.9 Mg C/ha/yr in net ecosystem C storage over three years after a single high rate (14 Mg C ha^{-1}) of compost application in the top 10 cm (Ryals & Silver, 2013). Our observed rates of C increase fall between these at $2.75 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$ at our 20 year site and $2.33 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$ at our 30 year site, though our measured benefits come from soil C gains alone as we are not able to quantify impacts to plant production or soil greenhouse gas emissions. These studies also had a high baseline level of soil C, though likely significantly lower than our control sites which have high baseline soil C, which could dampen our soils responses to OM amendments as higher soil C leads to less C accrual, though this is not a uniform response (Georgiou et al., 2025; Stewart et al., 2007; Wu et al., 2024). Despite high baseline soil C stocks, our soil showed a strong ability to increase C over decades of relatively standard rates of compost application (1.4 to $2.8 \text{ Mg C ha}^{-1} \text{ yr}^{-1}$ of compost). Despite a lower total amount and rate of compost application, the 30 year site had higher level of soil C stocks. Thus, consistency may be more important than application rate in improving soil C stocks with amendment additions (Evanylo et al., 2008).

When we assess the soil C characteristics of our turfgrass system across all treatments compared to other land-uses, we find unique impacts in management implications. This played out clearly in the age and distribution of C among the fractions compared to other systems. Across a global average 65% of SOC is found in the mineral fraction, while in our study the average was 58% across treatments (Sokol et al., 2022b). Our 30-50 cm and 75-100 cm samples were remarkably consistent with values from 61% to 71% across all treatments and both depths. Relative to these values, our top soil saw a lower proportion of C within the mineral fraction across all treatments at only 43%, though this measurement had the highest within depth variation (Figure 5-3). The low average was primarily driven by unamended ($43\% \pm 12$) and 30 year ($20\% \pm 1.7$) sites with lower proportions, while the 20 year ($66\% \pm 32.8$) site was closer to the average, albeit with large variation. The proportion of mineral C can vary widely by ecosystem, though grasslands average 70% for both the top and subsoil, as high microbial biomass limits particulate OM accumulation (Bastida et al., 2021; Sokol et al., 2022a). With a decreased contribution of mineral C to the topsoil C pool, it seems likely that even before factoring in compost application, the management of a soil as a turfgrass systems leads to increased proportion of C within the OLF (Figure 5-3). This proportion of C as OLF in our study across all treatments and textures was $38\% \pm 9$ in the top soil (0-10 cm), which is at the upper end of values compared to other managed systems, even those that also had multiple years of OM amendments applied (10% - 25%) (Ryals et al., 2014; Villa

et al., 2024). A shift from mineral fraction to occluded fraction C in the top soil of turfgrass soils signal a more actively cycling C pool than other systems, though protections still exist within the aggregates (Cotrufo et al., 2013). In turfgrass systems the roots are concentrated in the top 10 cm and have very high turnover, contributing to active soil C cycling (Qian & Follett, 2002; R. Wang et al., 2022). Grassland roots are known to shape the depth distribution of both light fractions, highlighting why turfgrass systems could lead to increased proportions of FLF and/or OLF in the top soil (Schrumpf et al., 2013; McFarlane et al., 2024). While this change could occur in either or both light fractions, we observed similar levels of FLF contribution to total C in the top soil ($18.6\% \pm 4.5$ across treatments) compared to other studies, highlighting the change being accounted for by the OLF within our study (Ryals et al., 2014; Villa et al., 2024). An important note is that these stocks by fraction were still significantly higher than most other systems, even if the proportion of our soil C fractions in the 0-10 cm is changing (Pouyat et al., 2009; Villa et al., 2024).

Soil C age and turnover rate of the HF can also be useful in understanding how C is cycling through a system. Mayer et al. (2023) found that 40 years of compost additions increased the turnover rate (decreased the ^{14}C -based mean residence time). However, we saw no change in our top soil radiocarbon activity with compost additions and radiocarbon activity only varied when assessed across all treatments over depth (Figure 5-5). This may reflect the low sample size for radiocarbon data between treatments, but also highlights the potential for a large proportion of C stored post-bomb in the mineral fraction. All turfgrass parks within this study were established post-bomb after the high turnover of soil C may have begun, leading to signals within the mineral C that are difficult to resolve with radiocarbon alone (Schrumpf et al., 2013; Torn et al., 1997). A dual-isotope approach could resolve differences between treatments; however, the $\delta^{13}\text{C}$ values were unavailable for the applied composts as no archival compost samples or data exist and $\delta^{13}\text{C}$ values for current applied compost was not distinct from soil $\delta^{13}\text{C}$ values.

Assessing radiocarbon alone signals to a generally fast turnover time across all turfgrass mineral soil C, but without confirmation of changing turnover times due to compost additions. In the mineral fraction, we observed $\Delta^{14}\text{C}$ values of $-63 \pm 16\text{‰}$, $-186 \pm 14\text{‰}$, and $-247 \pm 28\text{‰}$ in our 0-10 cm, 30-50 cm, and 75-100 cm depths respectively while an analysis of the international soil radiocarbon database by Li et al. (G. Li et al., 2023) found that average values over 295 diverse soil profiles were $-3 \pm 5\text{‰}$, $-168.03 \pm 14.91\text{‰}$ and $-358 \pm 29\text{‰}$ for the 0-30 cm, 30-60 cm, and 60-100 cm depths respectively. Even taking into consideration the fact that our samples were collected more recently than those already in the database, when comparing our results to the Li et al. study, we see diverging trends. Surface soil $\Delta^{14}\text{C}$ values are significantly more negative than the database average. Conversely, the 75-100 cm depth is significantly higher than the database values at 60-100 cm. Thus, it is likely that turfgrass soils, regardless of compost additions, may have older C in the top soil and younger C at depth relative to most other ecosystems (Heckman et al., 2022; Torn et al., 1997). Turfgrass soil management may lead to the priming of “younger” HF C in the top soil and irrigation may facilitate the translocation of this C, and other young C, to depth, leading to both older top soil C and younger subsoil HF C – a

phenomenon that has been observed in other ecosystems with high root inputs (Schiedung et al., 2023). This also provides evidence to the power of translocation in these heavily irrigated systems, as there is a significant fraction of “young” C within the 75-100 cm HF at our sites compared to global averages (G. Li et al., 2023). Even in systems that do not apply amendments to improve soil C, heavy use of irrigation may increase deep soil C storage, potentially at the expense of top soil C. It is important for land managers to see that top soil C loss could be offset through compost additions, allowing for top soil C maintenance or gain, while receiving the soil C gains at depth from irrigation use (McClelland et al., 2022).

5. Conclusions

We assessed the impacts on C cycling from long-term compost additions to urban soils, where we observed large increases in the soil C stocks of urban parks up to 1 m. These gains were concentrated in the top soil and at the bottom of the soil profile, highlighting the capacity for surface-applied compost to improve deep soil C on a decadal timescale. Soil C gains were largest in the mineral fraction, particularly at depth, pointing to fast incorporation of C into the mineral fraction. Within the mineral fraction, we saw a shift from microbial to simple plant OM, potentially from the direct incorporation of compost C into the mineral fraction at depth. Despite high levels of initial C managed turfgrass ecosystems, decades of compost application significantly increased and altered the makeup of soil C in urban soils. These findings suggest that long-term compost additions have the ability to increase whole profile soil carbon stocks.

CRedit Authorship Contributions Statement

Zachary Malone: Conceptualization, Writing – original draft, Writing – review & editing, Formal analysis, Investigation, Data curation, Visualization, Software.

Samuel Lee: Writing – review & editing, Investigation. **Karis McFarlane:**

Investigation, Methodology, Resources, Supervision, Writing – review & editing

Rebecca Ryals: Conceptualization, Writing – review & editing, Supervision, Funding acquisition.

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Conflict of Interest Statement

The authors declare no conflicts of interest.

Data Availability Statement

Data and code available will be made available on Dryad upon manuscript publication.

Supporting Information

Additional supporting information and figures can be found online in the supporting information section of this article.

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Chapter 6: Conclusions

This research conducted in this dissertation showcase the ability for OMAs, with a focus on compost, to serve as a powerful soil health enhancer on the scales of three months to thirty years. The focus is specifically on how OMAs improve urban soil health, which have wide-ranging soil qualities and are often heavily managed. The use of OMAs in urban areas provide a means to improve urban soil health, urban climate change mitigation, landfill diversion, and public greenspace quality.

One key finding of this work is that the different classes of OMAs have distinct impacts on soil health. Soil carbon increased in order of the carbon content of amendments from biochar, compost, to the least improvement with biosolids. Nutrient improvements were in the opposite order (biosolids, compost, biochar) corresponding with amendment nutrient content. Compost provided a middle-ground of effectively boosting both urban soil carbon and soil nutrients. These amendments showed the largest relative improvements in soil carbon and nutrients in land uses with the lowest baseline soil quality. All urban land uses, even those with SOM% close to 10, still saw improvements in soil quality with amendment application.

The remaining research in this dissertation focused compost impacts to turfgrass systems over three months to 30 year timescales. In short term greenhouse and lab experiments, we found that compost led to increased soil carbon and nitrogen concentrations, phosphorus mineralization, and belowground plant biomass. Greater increases in both above and belowground biomass were seen with a recycled water treatment, though this also led to increased N₂O and CO₂-eq emissions. Most of these benefits occurred regardless of initial soil quality, those higher initial soil quality on its own did increase most measures of soil and plant quality.

When we assessed several years of compost applications in the field we found contrasting results. Compost additions did not impact soil carbon or nitrogen stocks, and plant quality measured through NDVI was stable. Unlike in the greenhouse, we found the high rate of compost application increased cumulative soil CO₂ emissions while low rates of compost additions (and compost overall) decreased cumulative N₂O emissions. Increasing soil respiration could signal a healthier, more active soil microbiome with compost additions, though we saw no change in microbial biomass annually. A decrease in N₂O emissions with compost additions supports the case for compost as a climate change mitigator, despite the lack of increase in soil carbon stocks over several years.

When we assess compost on the decadal timescale, the benefits to soil carbon stocks from consistent compost application become apparent. While unamended turfgrass parks had high soil carbon stocks relative to other ecosystems, at 239 Mg C/ha to a meter, sites with 20 and 30 years of compost additions had 294 Mg/ha and 309 Mg/ha of C, respectively, to one meter. The majority of these benefits were seen in the top soil and very bottom of the profile. In the 0-10 cm, the increase in carbon stocks is hugely important for supporting soil quality and function. Gains in the

topsoil were focused in the free light and occluded light fractions, pools of carbon important for soil function. The proportion of soil carbon within these pools across all treatments was found to be higher than other ecosystems, which may display that the high amount of soil carbon in most turfgrass systems is within the light or particulate soil fractions. We still observed increases in the occluded light fraction, which show some ability for compost to increase more resilient pools of soil carbon in the top soil. At depth, the increases in carbon stocks at 75-100 cm showcase the ability for compost to increase carbon throughout the soil profile, where it may be more persistent and contribute to deeper soil quality. Gains in carbon at depth were primarily in the heavy mineral fraction of carbon. Both bulk and mineral soil carbon at depth across all sites and treatments were relatively high compared to other ecosystems, and mineral carbon had higher fraction modern values at 75-100 cm. Higher radiocarbon activity in deep mineral carbon pools provide evidence for translocation playing a large role in deep soil carbon storage, and may explain why compost was able to greatly increase deep soil carbon storage. The increase in deep mineral carbon and increased relative radiocarbon activity show that the mineral pool can be active and increased over decades, displaying that this pool may be more responsive to management than previously thought.

Future work should continue the study of compost additions in urban areas, particularly at depth and on decadal timescales. Many questions remain about how long-term compost additions impact soil greenhouse gases over decades, how microbial communities may shift, and what mechanisms fuel increases in deep soil carbon. Lifecycle assessments would be prudent to combine the long-term carbon stock changes, measures in soil greenhouse gas emissions, and measurements of compost application and transportation emissions to provide whole system climate accounting for municipalities that may begin to apply compost applications to their public parks.

In conclusion, this dissertation provides strong evidence for OMAs to improve urban soil health. The OMAs studied, ranging from biosolids to compost, were found to provide benefits that are unique with length of application. These benefits can be seen in as little as three months after one application to continual improvement even after thirty years of consecutive annual compost additions. By finding benefits to soil health with multiple OMAs, in multiple land-uses, and at many time-scales, this work serves as a strong evidence base for using OMAs to improve soil health in urban contexts. Since many urban systems have agricultural similarities, such as being intensely managed with irrigation and fertilization, these findings can be insightful for land-uses outside of city spaces. Thus, these projects show that almost any land-use can find a soil benefit from amendments additions, and amendments continue to improve soil health even after decades of use.

Appendices

Appendix A: Supplemental Information for Chapter 2

Table S2-1: This table displays the keywords used to search the literature for relevant studies. All terms used in combination with opposite category to find relevant studies.

Land-Use Search Terms	Amendment Search Terms
Urban	Amendment OR Amendments
Urban Soil	Amendment Restoration
City	Amendment Application
City Soil	Sewage Sludge
Urban Garden	Biosolids
Urban Recreation	Compost
Urban Park	Composted Biosolids
Roadside	Biochar
Roadside Restoration	

Table S2-2: This table displays the papers and some of their characteristics used in the quantitative analysis of amendments effects on OM%, N%, P, K, bulk density, and pH.

Reference	Amendment(s)	Properties Analyzed	Study Length (years)	Application Amount	Vegetation
Alvarez-Campos et al. (2019)	Compost, Biosolids	N	0.5		Tall fescue
Alvarez-Campos et al. (2019)	Compost, Biosolids	OM, N	2		
Attanayake et al. (2015)	Compost, Biosolids	OM, P, pH	1.8	440 Mg/ha	Carrots, Ryegrass, Tomato, Collard Greens
Badzmierowski et al. (2019)	Compost, Biosolids	OM, N, P, BD, pH	3	0.16–23.8 Mg/ha	Tall Fescue

Basta et al. (2016)	Compost, Biosolids	OM, N, P, pH	3	137, 202, 404 Mg/ha	
Beesley et al. (2012)	Compost	OM, BD	0.16	15%, 30%, 45%, 50%	
Beniston et al. (2016)	Compost, Biochar	OM, N, P, K, BD, pH	2	150 Mg/ha	Tomato, Swiss Chard, Sweet Potatoes, Ryegrass
Brose et al. (2016)	Biosolids	OM, N, P. pH	4	0-50%	Turfgrass
Brown et al. (2011)	Compost, Biosolids	OM, N, P, K, pH	1-15	157, 224 Mg/ha	Turfgrass and Shrubs
Brown et al. (2011)	Compost, Biosolids	OM, P, K	3	20%, 50%	Grasses
Bryan et al. (2011)	Compost	BD	0.75	30%	Live Oak
Chan et al. (2008)	Compost	OM, N, pH	1.5	60, 120 Mg/ha	Broccoli, Eggplant, Cabbage
Civeira et al. (2010)	Compost	OM, N, P, BD, pH	1.5	20, 40, 70 Mg/ha	Grasses
Curtis et al. (2009)	Compost	OM, N, BD	0.75	540 Mg/ha	
De Lucia et al. (2015)	Compost	N, P, pH	2	15, 30, 45%	Mediterrane an Shrubs
De Lucia et al. (2013)	Compost	OM, N, P, pH,	0.583	15, 30, 45%	Hedges
Dubelko et al. (2022)	Compost	P, K, pH, BD	1		Shrubs
Dunifon et al. (2011)	Compost	OM, N, P, K, pH	2.5		Fescue

Ghosh et al. (2018)	Compost	OM, N, P, K, pH	0.5	3, 5 kg/m ³	Calophyllum soulattri and Syzygium myrtifolium
Ghosh et al. (2015)	Compost, Biochar	OM, N, P	0.5		Suregada multiflora
Ghosh et al. (2012)	Biochar	OM, N, P	0.1		
Japakumar et al. (2021)	Biochar, Compost	OM, pH	0.25		
Jusselme et al. (2019)	Compost	OM, N, pH	2	5cm layer	Ornamental perennials
Kim et al. (2021)	Biochar	BD	0.27		
Li et al. (2023)	Compost	OM, P, K, BD, pH	1		Trees
Lord et al. (2019)	Compost	OM, N, K	10	250, 500, 750 Mg/ha	
McIvor et al. (2012)	Biosolids	N, P, BD, pH	1.5	200 Mg/ha	Onion, Beet, Carrot, Potato, Beans
Murray et al. (2011)	Compost	OM, pH	0.22	9%, 25%	
Pengcheng et al. (2008)	Compost	OM, N, P, BD, pH	2.33	15, 30, 60, 120 Mg/ha	Perennial Ryegrass
Pierre et al. (2016)	Compost	BD	3	2.5cm ³ / 100 cm ³	Shrubs
Pini et al. (2012)	Compost	OM, BD	0.67	30 Mg/ha	Wildflowers
Richnau et al. (2016)	Compost	OM, N, pH	1.5	10 cm layer	Shrubs (Quercus)
Rojas et al. (2021)	Compost	OM, N, K, pH	2	20%	Trees and Shrubs

Rojas et al. (2022)	Compost	OM, N, K, pH	2	20%, 50%	
Sax et al. (2017)	Compost	OM, P, BD, pH	12	33%	Trees and Shrubs
Scharenbroch et al. (2013)	Compost, Biochar	OM, pH	1.5	25 Mg/ha	A. saccharum and G. triacanthos
Scharenbroch et al. (2022)	Biochar	OM, N, P, K, pH	3	37.5 Mg/ha	Trees
Shiu et al. (2022)	Compost, Biochar	OM, N, P, pH	1		Trees
Somerville et al. (2020)	Compost, Biochar	BD	2.5	10%, 20%, 25%	Corymbia maculata
Tian et al. (2019)	Biochar	OM, N, P, K, pH		5%	Turfgrass
Una et al. (2022)	Compost, Biosolids	OM, N, BD	2	30, 56, 61.5 Mg/ha	
Vannucchi et al. (2015)	Compost	OM, N, BD, pH	3	20 Mg/ha	Herbacious species
Ward et al. (2021)	Compost	OM, N, BD, pH	6		Trees
Weir et al. (1997)	Compost	OM, pH	0.25	2.2, 25, 50 Mg/ha	Mustard and Tomato
Wu et al. (2017)	Biosolids	OM, N, P, K, BD, pH	0.25	15, 30, 60%	A. ruellia, S. podophyllum, S. odorata, A. macrorrhiza, and D. ensifolia
Yue et al. (2017)	Biochar	OM, N, P, K, pH		1%, 5%, 10%, 20%, 50%	Grasses
Zhou et al. (2021)	Biochar	OM, N, pH	0.22	2%	Photinia serrulata and

					Rhododendr on
Zhou et al. (2022)	Compost	OM	2	5cm layer	Trees

Table S2-3: Total studies, samples, and share percentages of treatment only samples by study for each soil property in this meta-analysis (Not including control values).

Soil Property	Study n	Sample n	Average Samples	Highest Share for one study	Average Share for a study	Lowest Share for a study
SOM%	38	159	4.18	15.1%	3.1%	0.6%
N (g/kg)	27	134	4.96	17.9%	3.7%	0.7%
Total P (mg/kg)	24	117	4.88	20.5%	4.2%	0.9%
Total K (mg/kg)	13	33	2.54	15.1%	7.7%	3.0%
Bulk Density (g/cm ³)	21	96	4.57	29.2%	4.8%	1.0%
pH	33	99	3.09	8.1%	3.5%	1.0%

Table S2-4: Table of average C:N's of amendments and the C:N ratios of all studies used in this meta-analysis that reported them in text.

Amendment	Study Size	C:N (± SD)	Min and Max C:N
Compost	24	27.9 ± 34.9	8 / 156.6
Biosolids	12	9.8 ± 3.8	2.3 / 23.9
Biochar	4	108.4 ± 150	8.3 / 350
Author	Year	Amendment	C:N
De Lucia	2013	Biosolids	10.2
Ghosh	2015	Compost	13
		Biochar	8.3
Finni	2016	Compost	30.66
Mclvor	2012	Biosolids	13.8
		Compost	23.9
Scharenbroch	2013	Compost	10

		Compost	22
		Biochar	160
		Biosolids	10
Basta	2016	Compost	15.94
		Biosolids	12.8
Badzmierowski	2019	Biosolids	7
		Biosolids	15
		Biosolids	15
		Compost	13
Alvarez-Campos	2019	Biosolids	8.1
		Biosolids	12.7
		Biosolids	13.2
		Compost	12
		Biosolids	7.8
Ghosh	2012	Biochar	13.2
		Biochar	10.5
De Lucia	2015	Biosolids	10.2
Badzmierowski	2020	Biosolids	7.9
		Biosolids	13
De Ona	2009	Biosolids	2.26
		Compost	12.69
Changanti	2014	Compost	37
		Compost	10
Dunifon	2011	Compost	16.9
Brown	2011	Biosolids	6.7
		Compost	64
Somerville	2020	Compost	34.6
		Biochar	350
Beniston	2016	Compost	14.3
Wiseman	2012	Compost	18
Bakr	2012	Compost (wood based)	139.4
Attanayake	2015	Compost	13.8
		Biosolids	7.7
		Compost	13.6
		Compost	38.6
De Ona	2011	Biosolids	3.4
Chan	2008	Compost	19
Richnau	2016	Compost	14.8
Pini	2012	Compost	17.9

Vannucchi	2015	Compost	9.2
Dubelko	2022	Compost	8
Shiu	2022	Compost/Biochar Mix	25
Civeira	2010	Compost	156.6
Zhou	2022	Compost/wood chip	14.71
Li	2023	Compost	26.51

Table S2-5: The soil textures reported in this meta-analysis with the amount of individual samples corresponding to each texture type.

Soil Texture	Number of Occurrences
Sand	24
Sandy Loam	41
Sandy Clay Loam	39
Silt	7
Silt Loam	18
Silt Clay Loam	12
Loam	30
Loamy Sand	44
Clay	21
Clay Loam	17

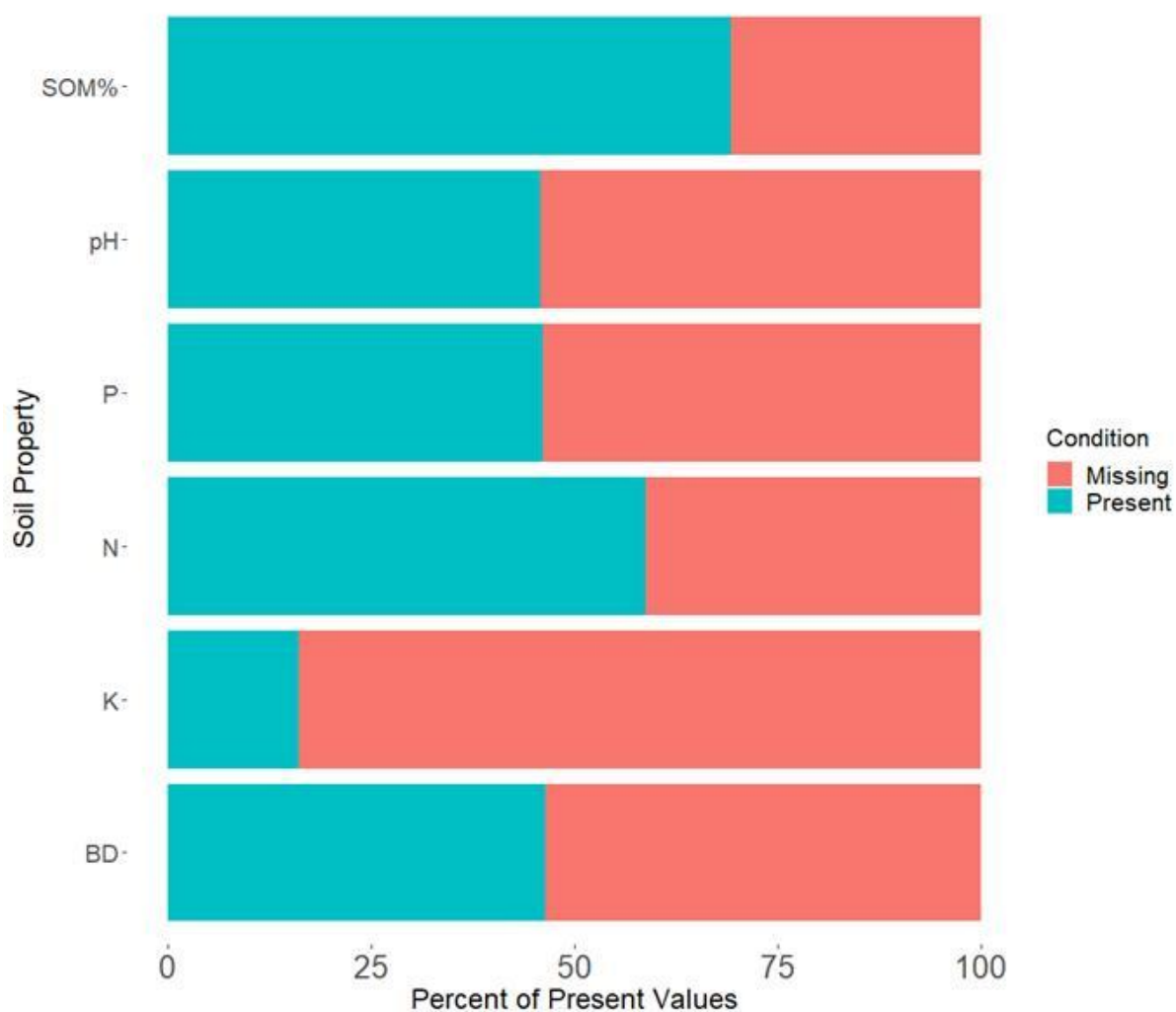


Figure S2-1: Present and Missing data in the data set used for analysis. The calculation is only based on rows where at least one of the factors above (SOM%, N, P, BD, pH, K) is reported. This figure is supplemental to understanding the reporting statistics of soil properties in the studies used for this analysis.

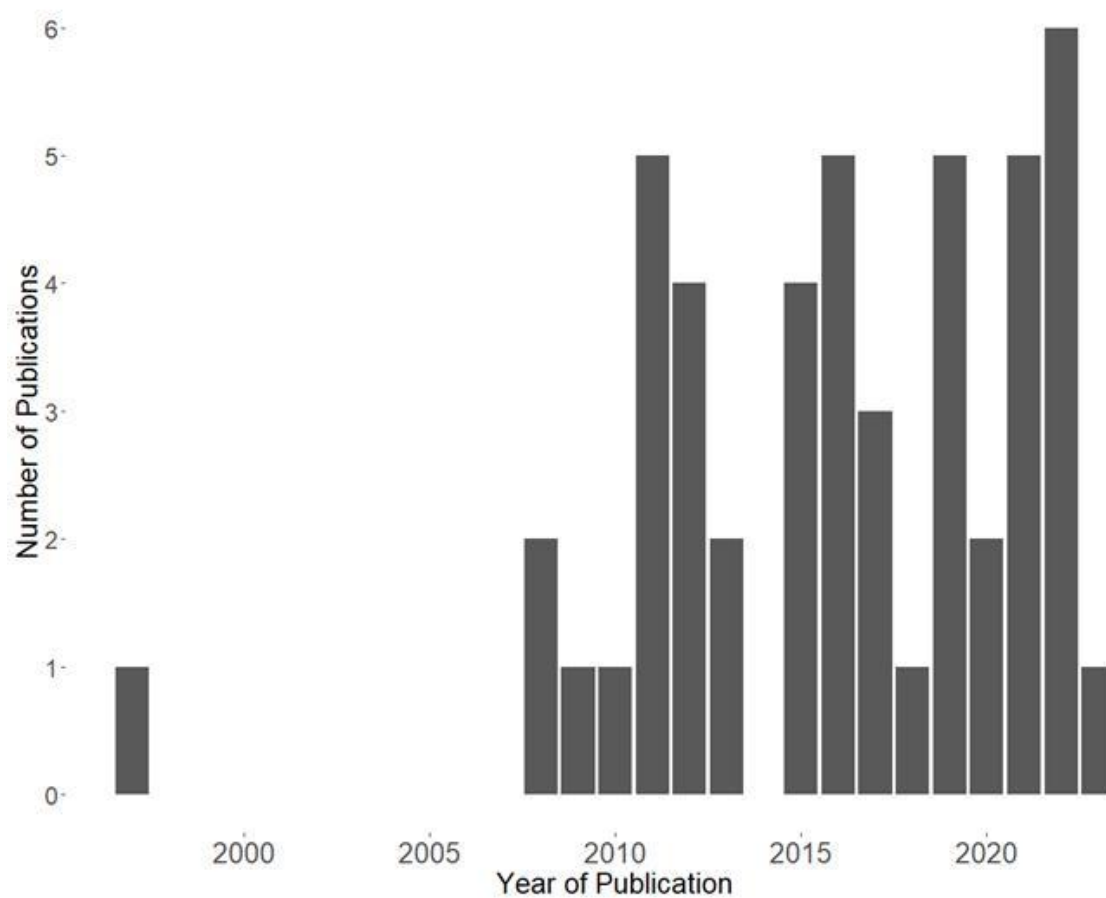


Figure S2-2: Bar chart of publications per year for the studies included in this meta-analysis.

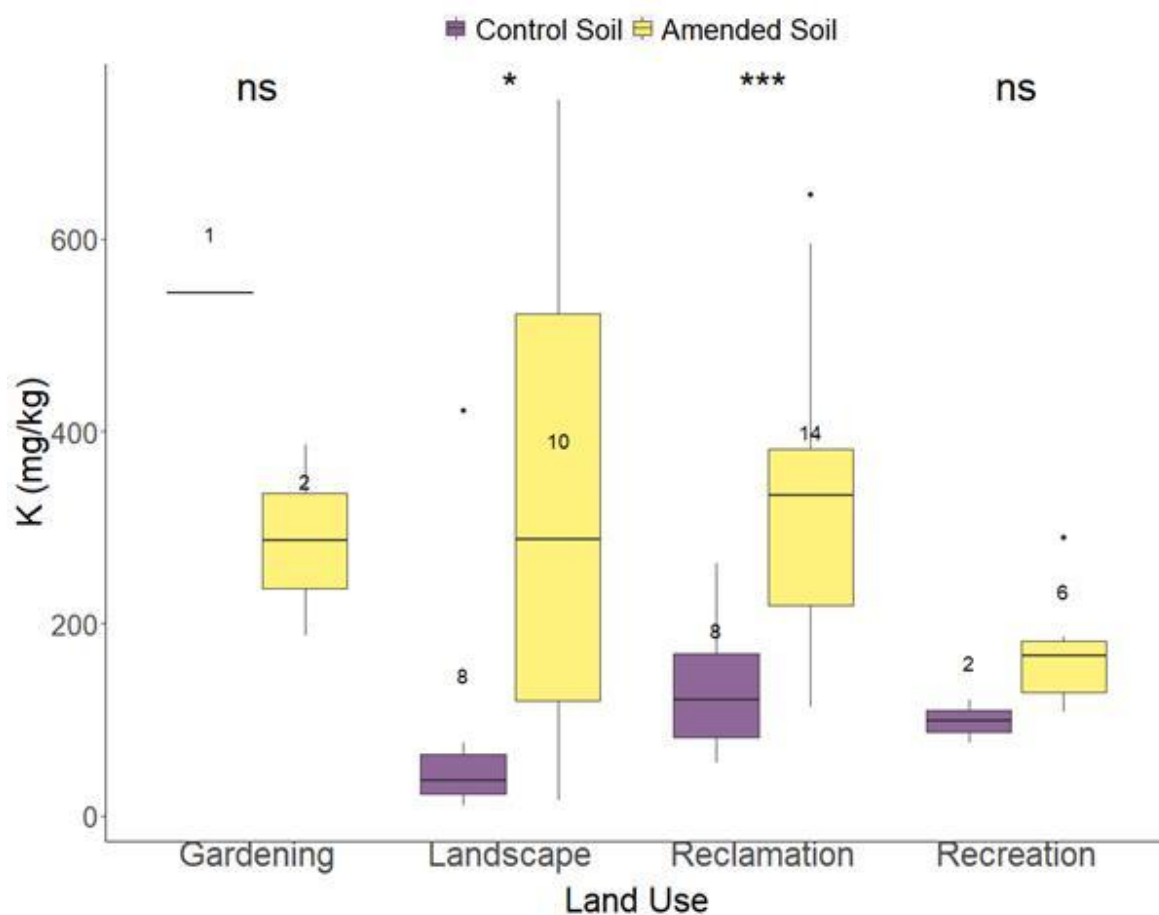


Figure S2-3: Box plots of potassium by setting with control in purple and amended in yellow. Numbers correspond to sample size. Level of significant is: ns: $p > 0.05$, *: $p \leq 0.05$, **: $p \leq 0.01$, ***: $p \leq 0.001$, ****: $p \leq 0.0001$.

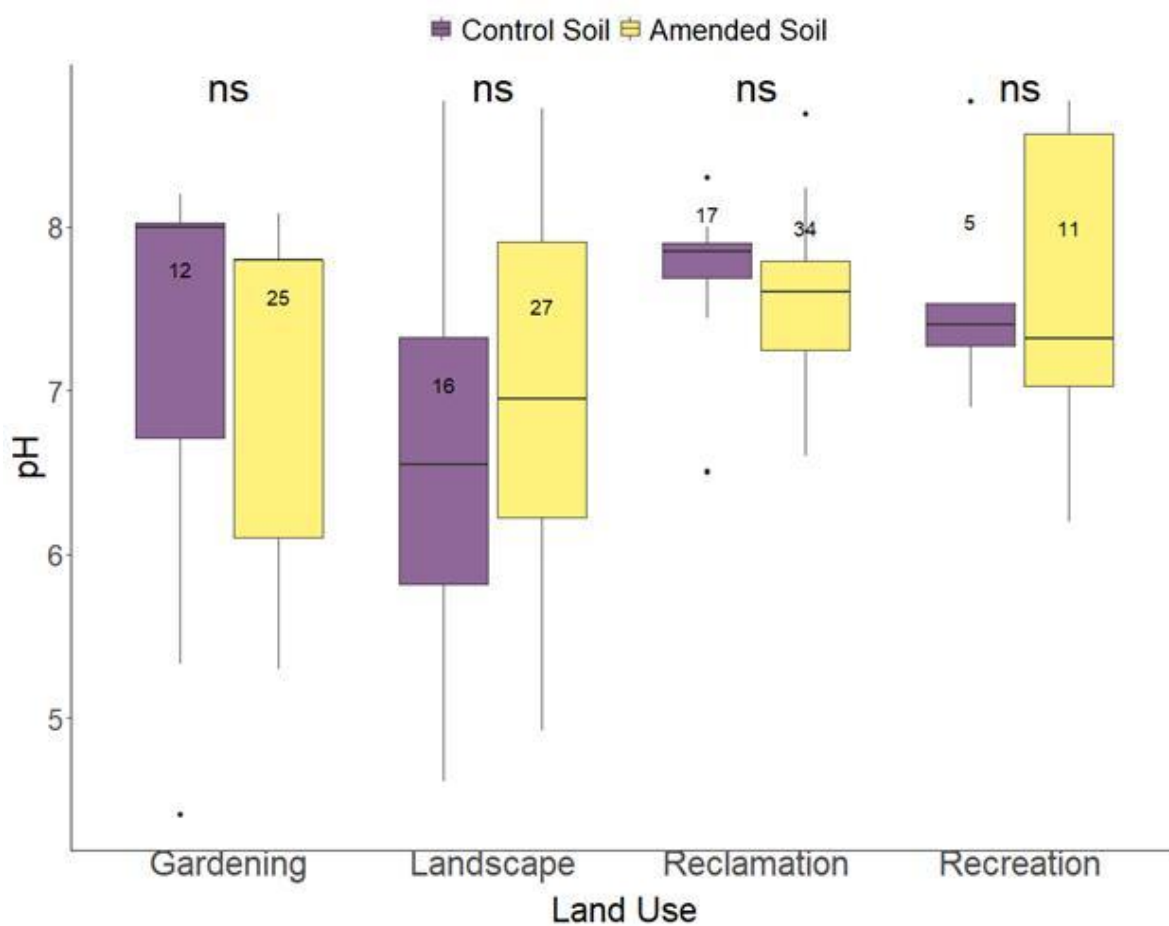


Figure S2-4: Box plots of pH by setting with control in purple and amended in yellow. Numbers correspond to sample size with pink dots representing control samples and blue dots represented amended samples. Level of significant is: ns: $p > 0.05$, *: $p \leq 0.05$, **: $p \leq 0.01$, ***: $p \leq 0.001$, ****: $p \leq 0.0001$.

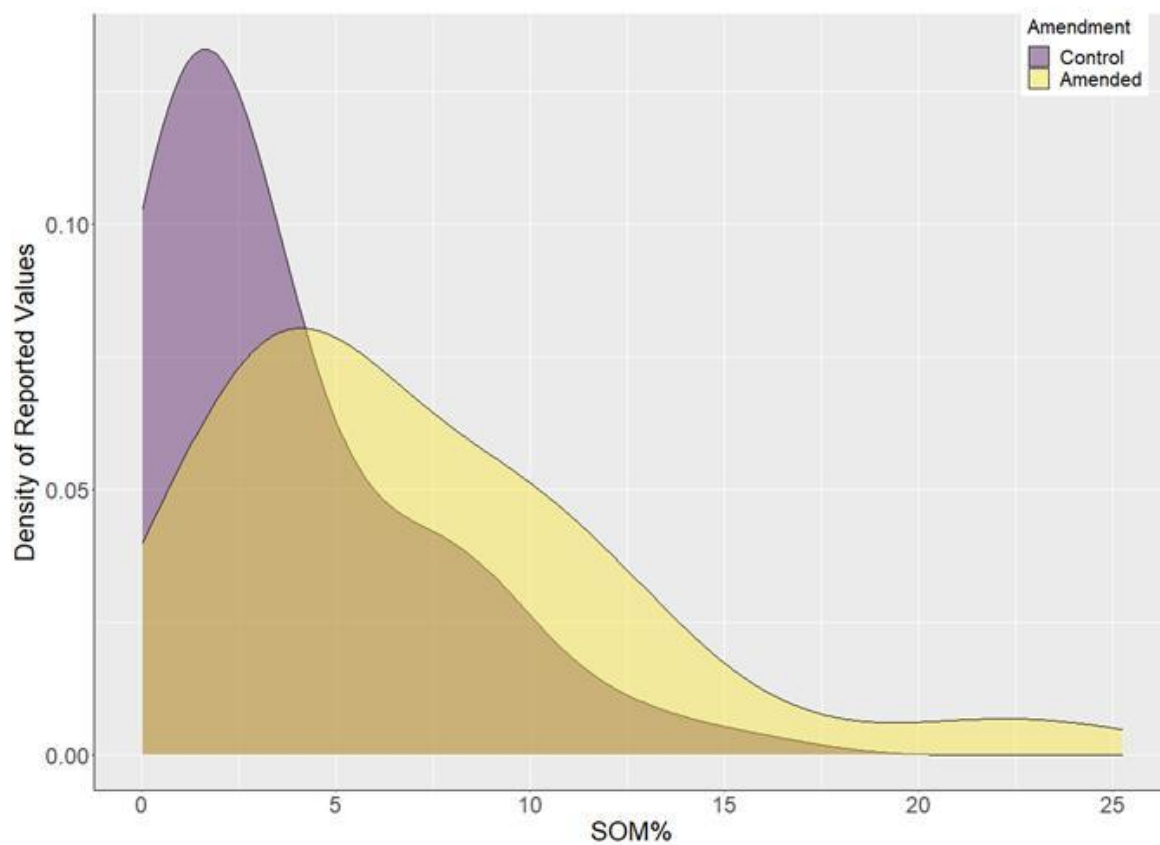


Figure S2-5: Density plot of soil organic matter percent values for amended and control samples in this meta-analysis.

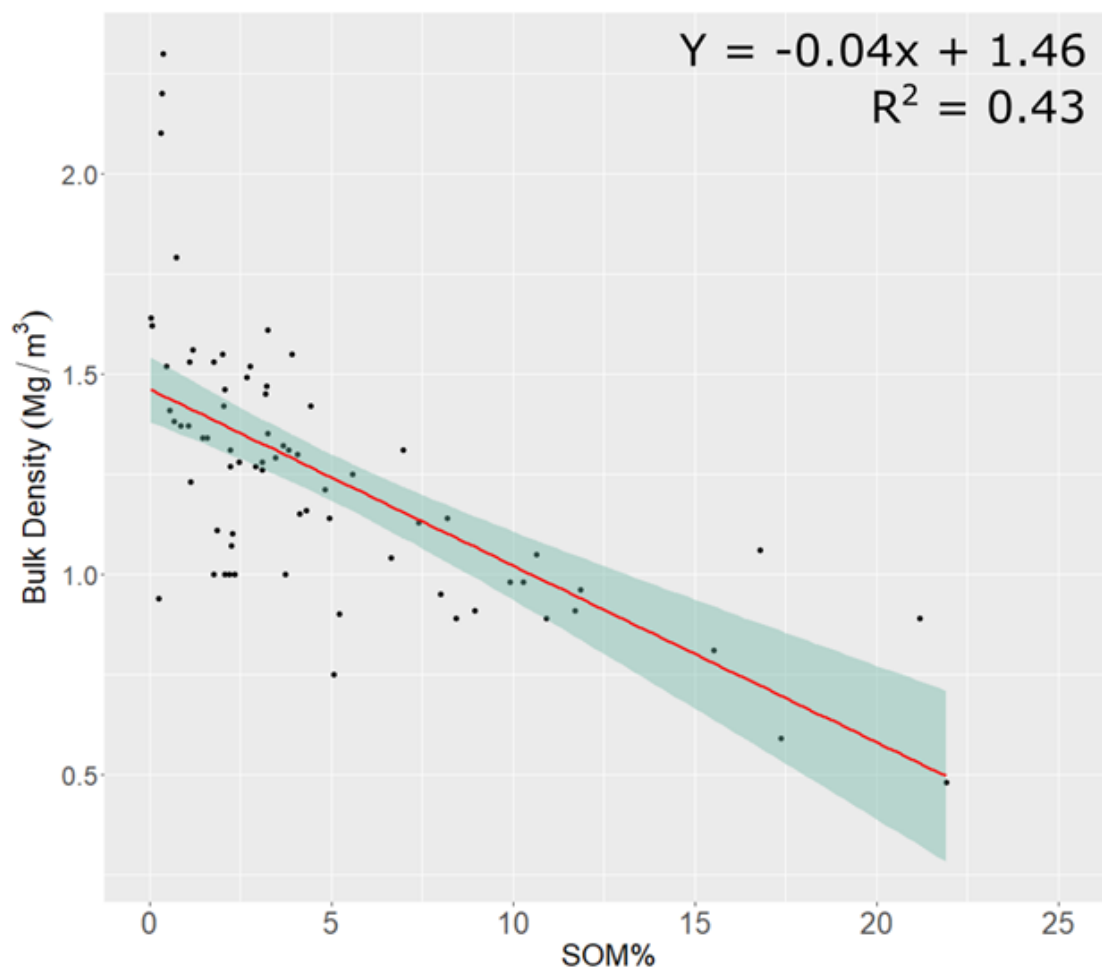


Figure S2-6: Scatterplot of organic matter percent and bulk density (Mg/m³) measurements with a 95% confidence interval. Regression equation and R^2 values are placed in the top right corner of the plot.

Appendix B: Supplemental Information for Chapter 3**Table S3-1:** Characteristics of the compost (n=5) applied.

	C%	N%	C:N	pH	EC (us/cm)
Compost	18.48	1.04	17.91	7.8	836.3

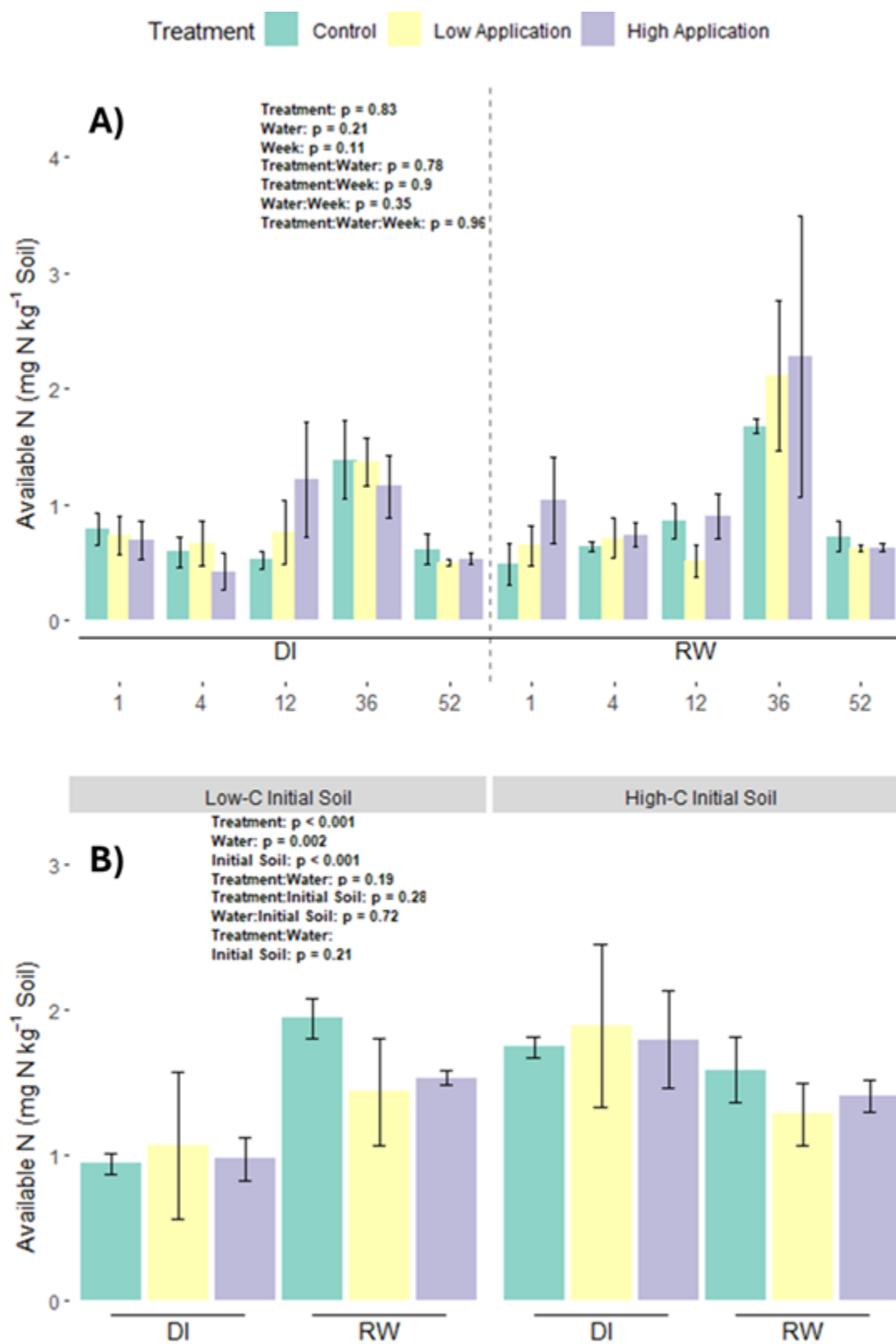


Figure S3-1: Available nitrogen pools between compost application rates and watering types in the mineralization (A) and greenhouse experiment (B). The ANOVA tables denote statistical differences between application rate, water type, and initial soil C.

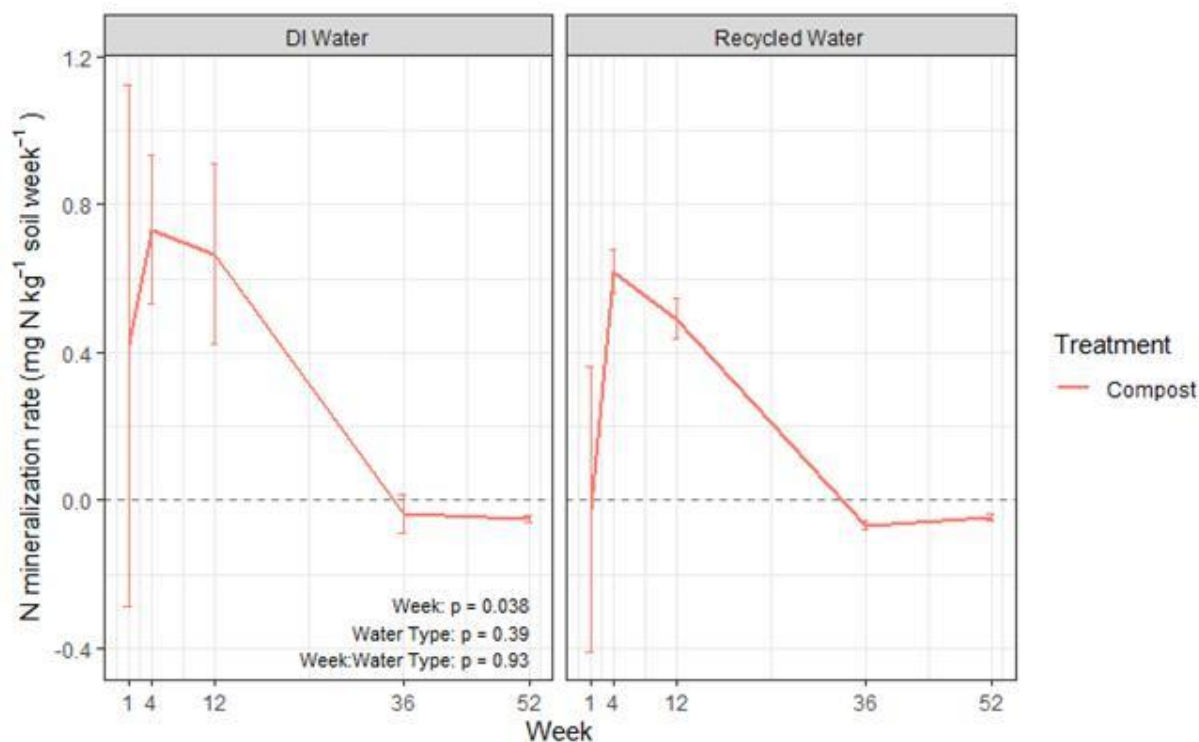


Figure S3-2: Available nitrogen mineralization rate by time point and watering type in the mineralization experiment for compost only treatments. The ANOVA tables denote statistical differences between application rate, water type, and initial soil C.

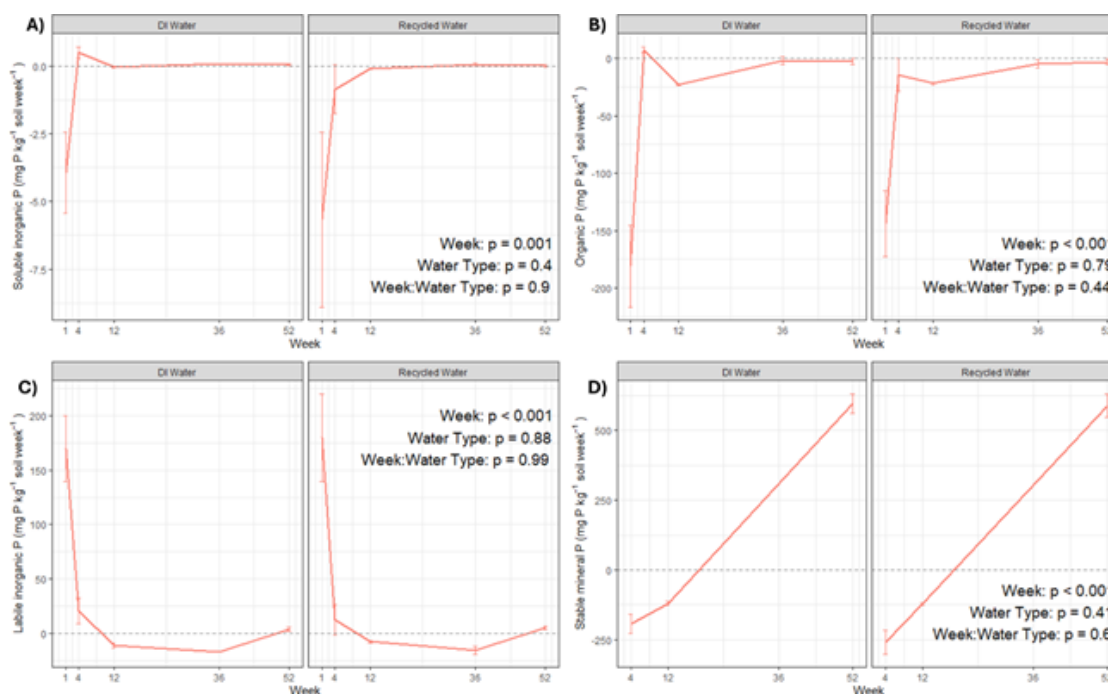


Figure S3-3: Soil phosphorus mineralization rates for pools of soluble inorganic P (A), organic P(B), labile inorganic P (C), and stable mineral P (D) by time point and watering type in the mineralization experiment for compost only treatments. The ANOVA tables denote statistical differences between application rate, water type, and initial soil C.

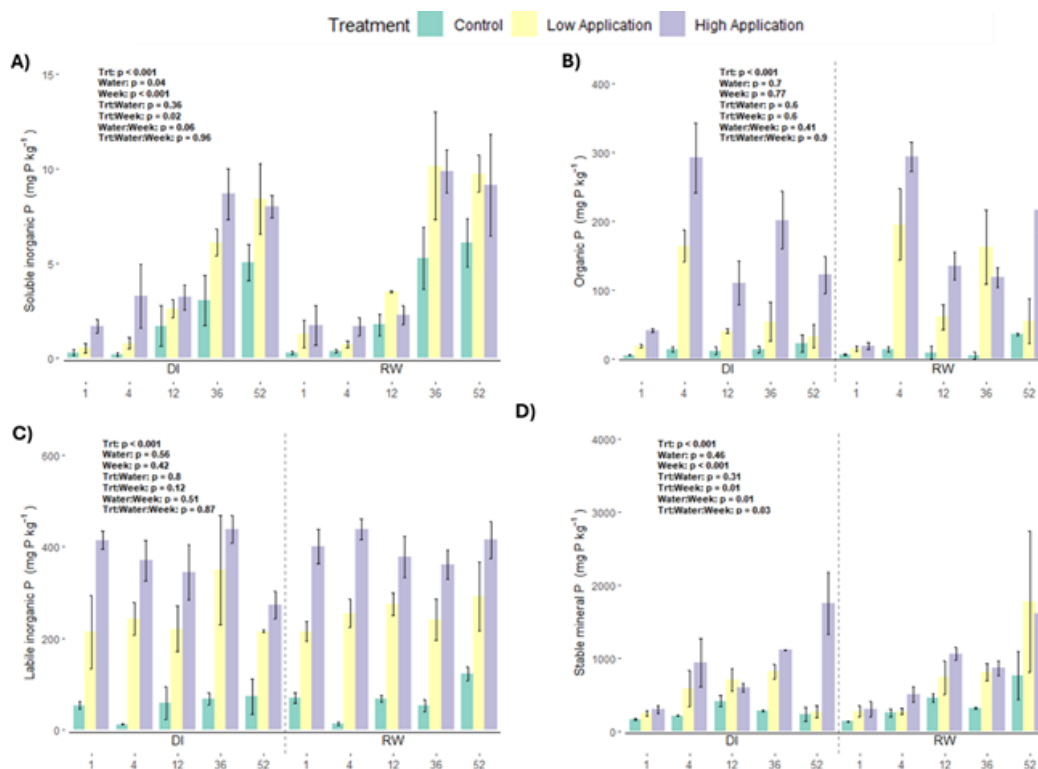


Figure S3-4: Soil phosphorus pools of soluble inorganic P (A), organic P(B), labile inorganic P (C), and stable mineral P (D) between treatments in the mineralization experiment. The ANOVA tables denote statistical differences between application rate, water type, and initial soil C.

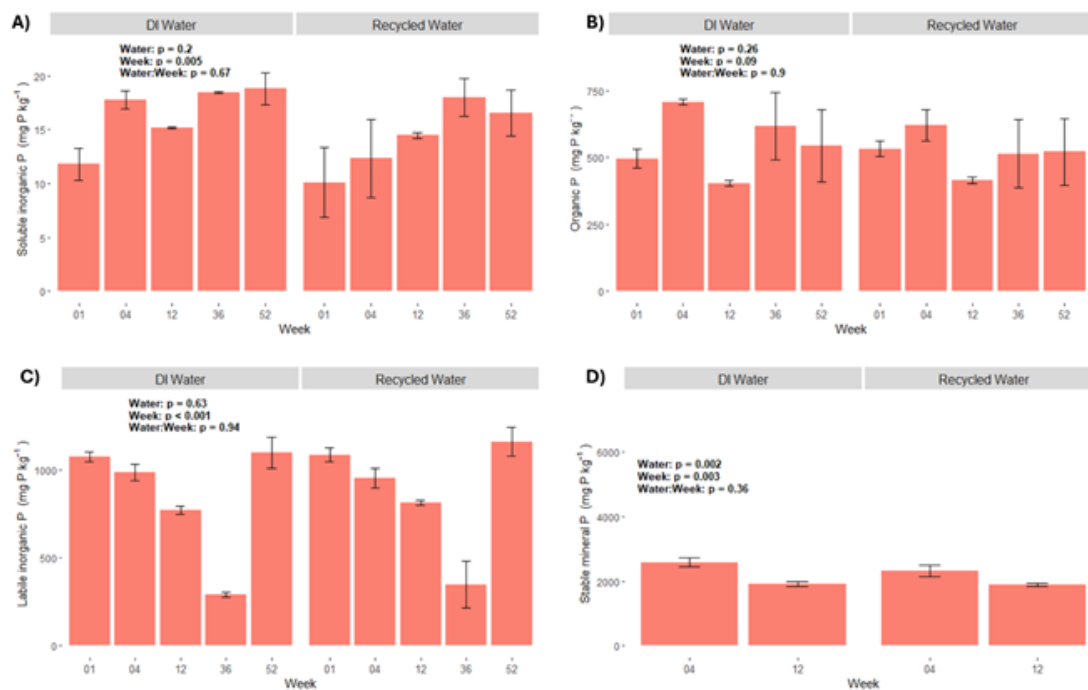


Figure S3-5: Phosphorus pools of soluble inorganic P (A), organic P(B), labile inorganic P (C), and stable mineral P (D) for the compost only treatment in the mineralization experiment. The ANOVA tables denote statistical differences between application rate, water type, and initial soil C.

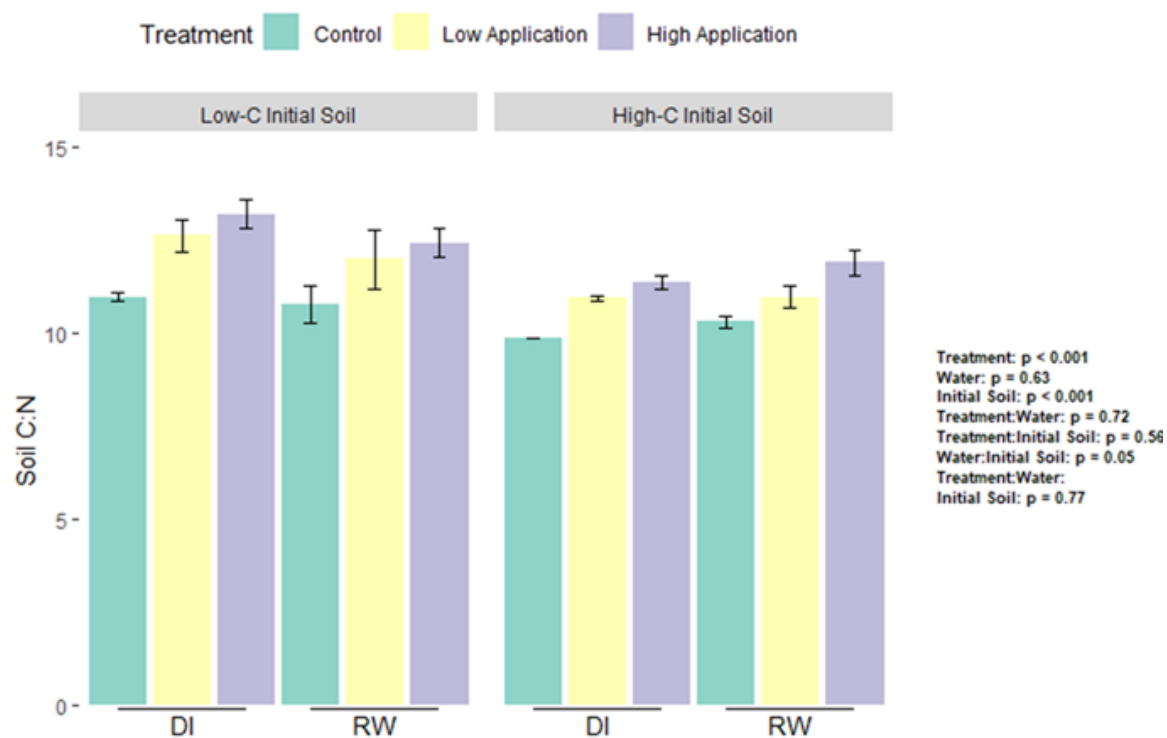


Figure S3-6: Bar chart of effect of compost, initial soil C, and water type on C:N. The ANOVA tables denote statistical differences between application rate, water type, and initial soil C.

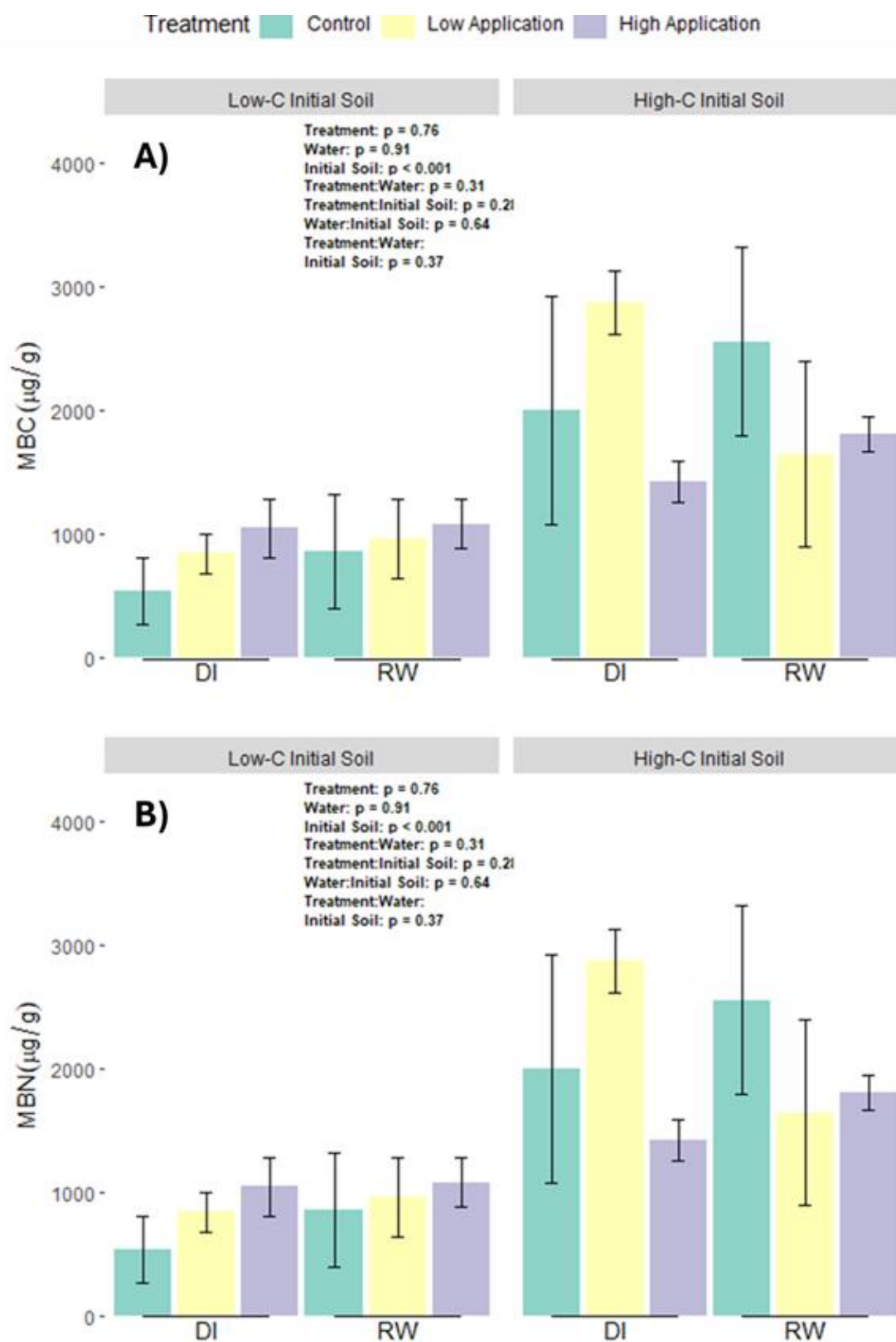


Figure S3-7: Bar chart on the effect of compost, depth, and water type MBC (A) and MBN (B). The ANOVA tables denote statistical differences between application rate, water type, and initial soil C.

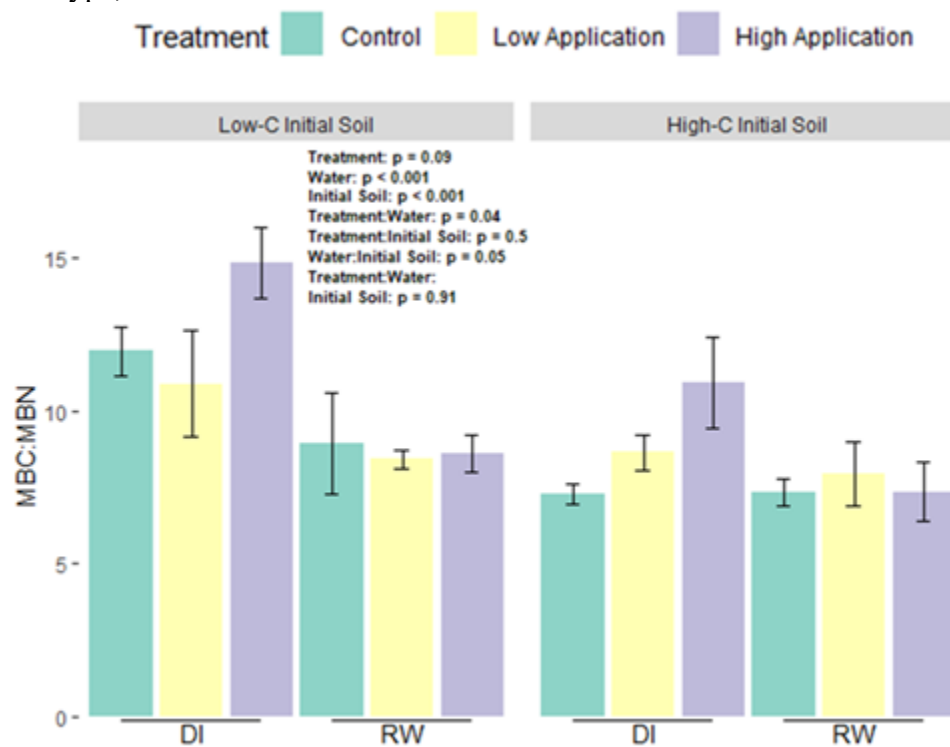


Figure S3-8: Bar chart on the effect of compost, depth, and water type MBC:MBN. The ANOVA tables denote statistical differences between application rate, water type, and initial soil C.

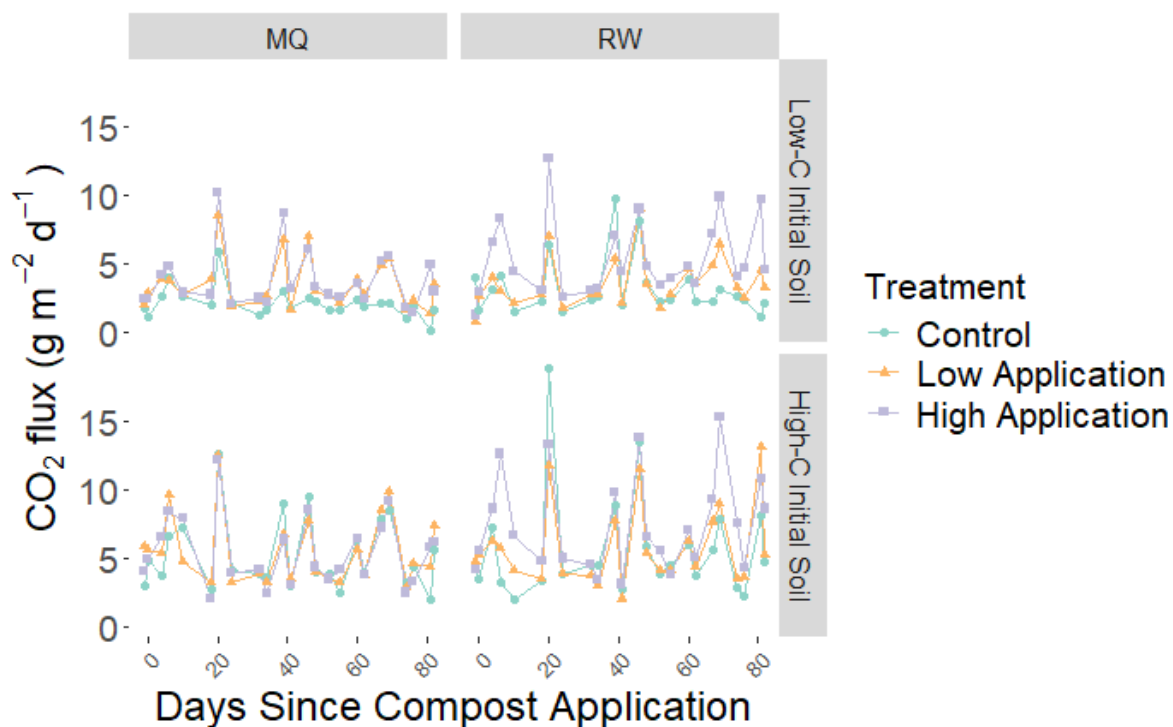


Figure S3-9: Line graph for CO₂ flux over three months based on compost, water type, and depth.

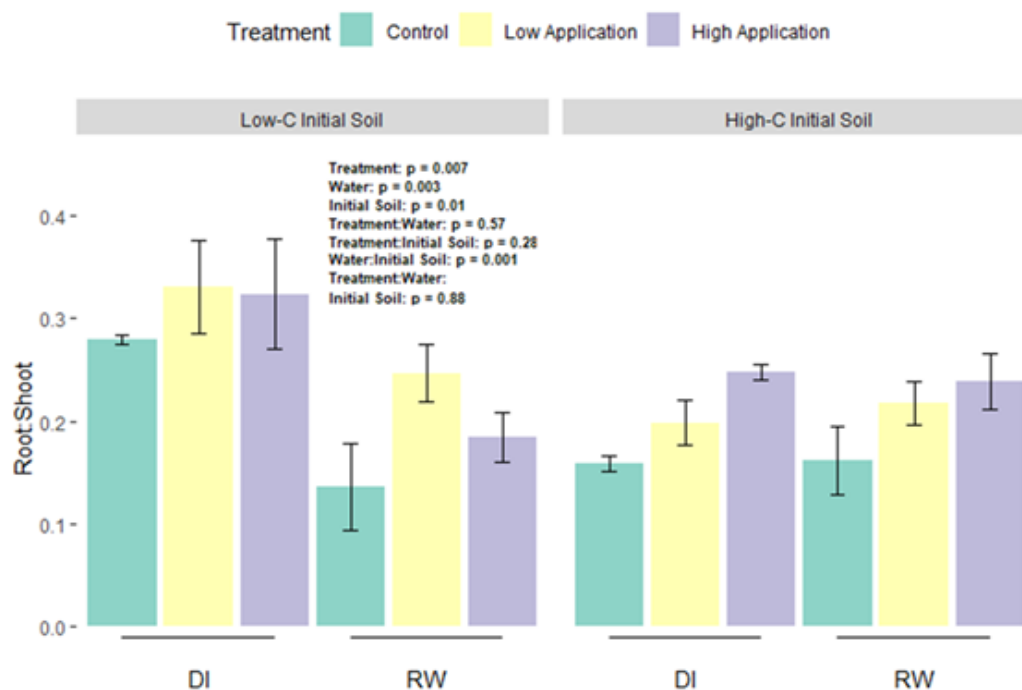


Figure S3-10: Bar chart on the effect of compost, depth, and water type root:shoot. The ANOVA tables denote statistical differences between application rate, water type, and initial soil C.

Appendix C: Supplemental Information for Chapter 4

Table S4-1: Table of C%, N%, and C:N by depth in baseline soils.

Depth	Carbon	Nitrogen	C:N
0-10cm	3.9% $\pm$ 0.02	0.38% $\pm$ 0.02	10.2 $\pm$ 0.08
10-30cm	2.2% $\pm$ 0.13	0.21% $\pm$ 0.01	10.6 $\pm$ 0.27
30-50cm	1.3% $\pm$ 0.12	0.11% $\pm$ 0.009	12.0 $\pm$ 0.23
50-75cm	0.8% $\pm$ 0.07	0.07% $\pm$ 0.007	12.9 $\pm$ 0.50
75-100cm	0.7% $\pm$ 0.04	0.06% $\pm$ 0.004	13.0 $\pm$ 0.39

Table S4-2: Table of the percentage contribution of each cumulative gas emission to the CO₂-eq for each treatment. ANOVA and Tukey HSD used for statistical significance with significance codes: <0.001 ***, <0.01 **, <0.05 *, <0.10 .

Ga s	Control Mean $\pm$ SE (%)	High Applicat ion	Low Applicatio n	ANOVA Treatmen t	Tukey Contro l vs Low	Tukey Contro l vs High	Tukey Low vs High
CO 2	95.2 $\pm$ 0.5	96.4 $\pm$ 0.5	97.7 $\pm$ 0.3	0.003**	0.002* *	0.114	0.129
N ₂ O	4.8 $\pm$ 0.5	3.6 $\pm$ 0.5	2.3 $\pm$ 0.3	0.003**	0.002* *	0.114	0.129
CH 4	0 $\pm$ 0	0 $\pm$ 0	0 $\pm$ 0	0.333	0.333	0.940	0.510

GWP100 values from IPCC AR6: N₂O = 273, CH₄ = 27.

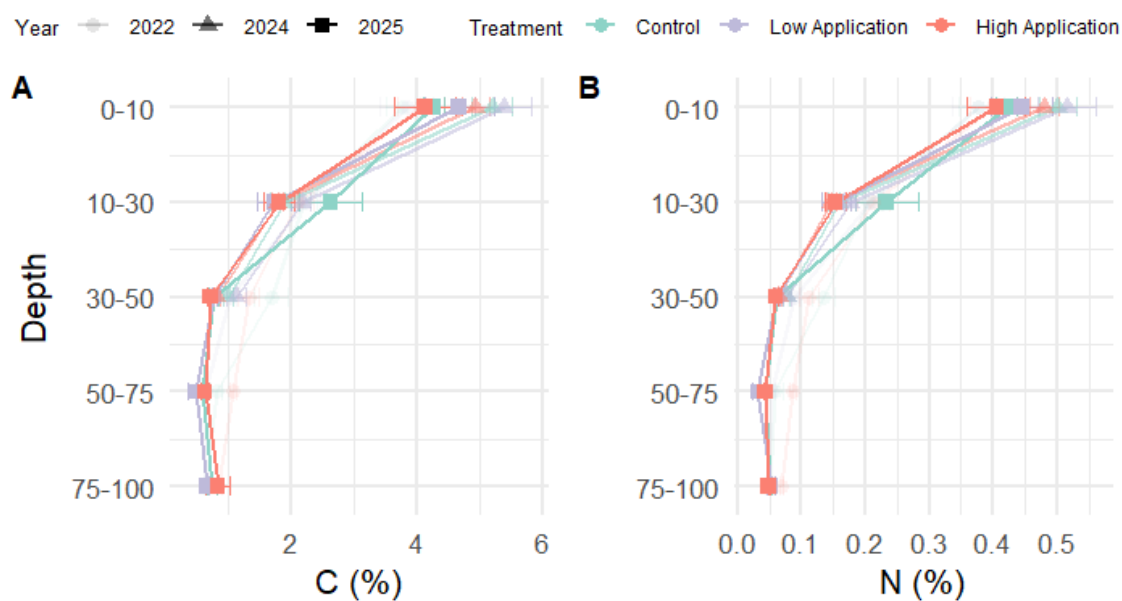


Figure S4-1: Soil carbon and nitrogen percent by treatment, depth, and year.

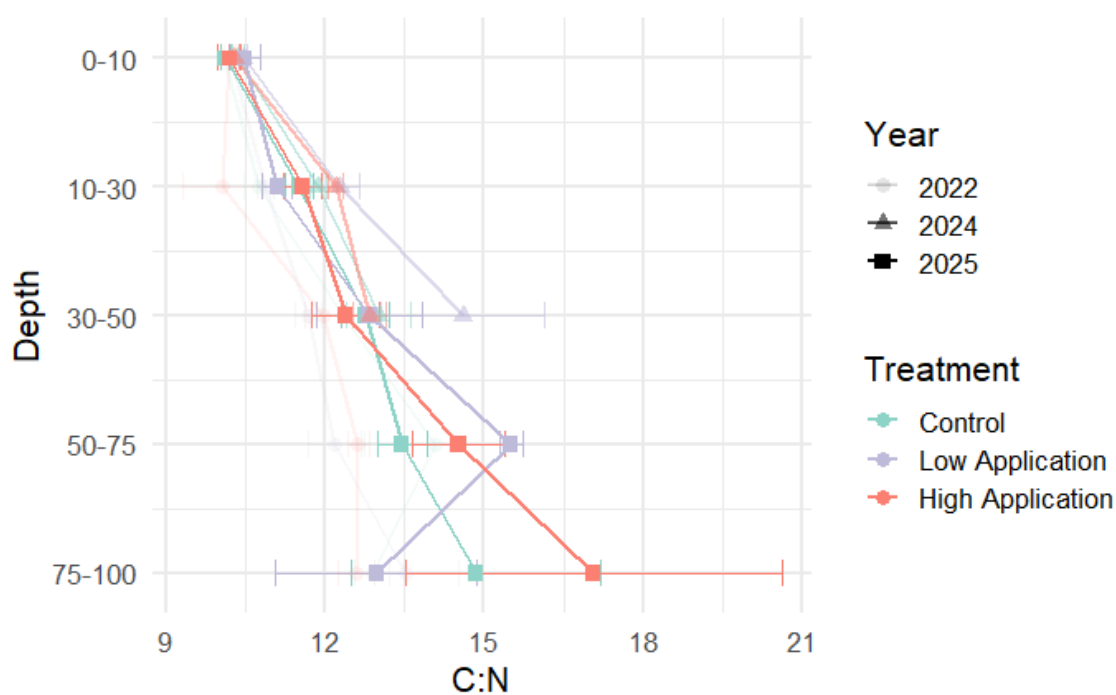


Figure S4-2: Soil carbon to nitrogen ratio by treatment, depth, and year.

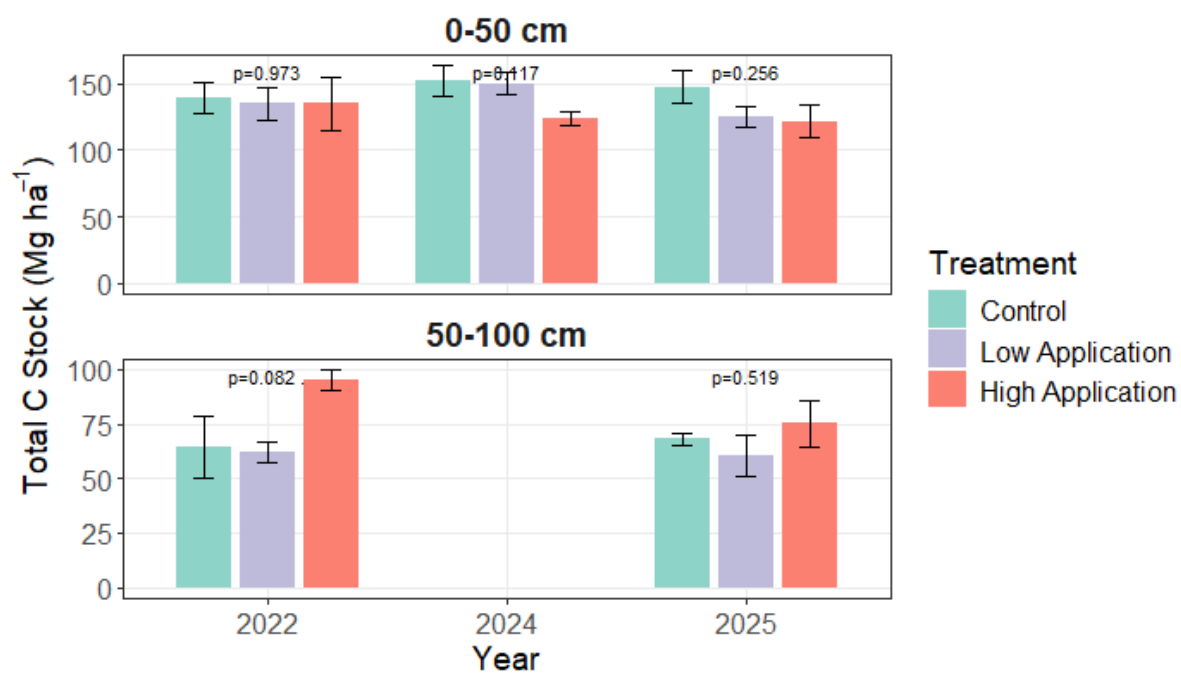


Figure S4-3: Total soil carbon stocks (Mg C /ha) by treatment, depth, and year in the 0-50 cm and 50-100 cm depth intervals.

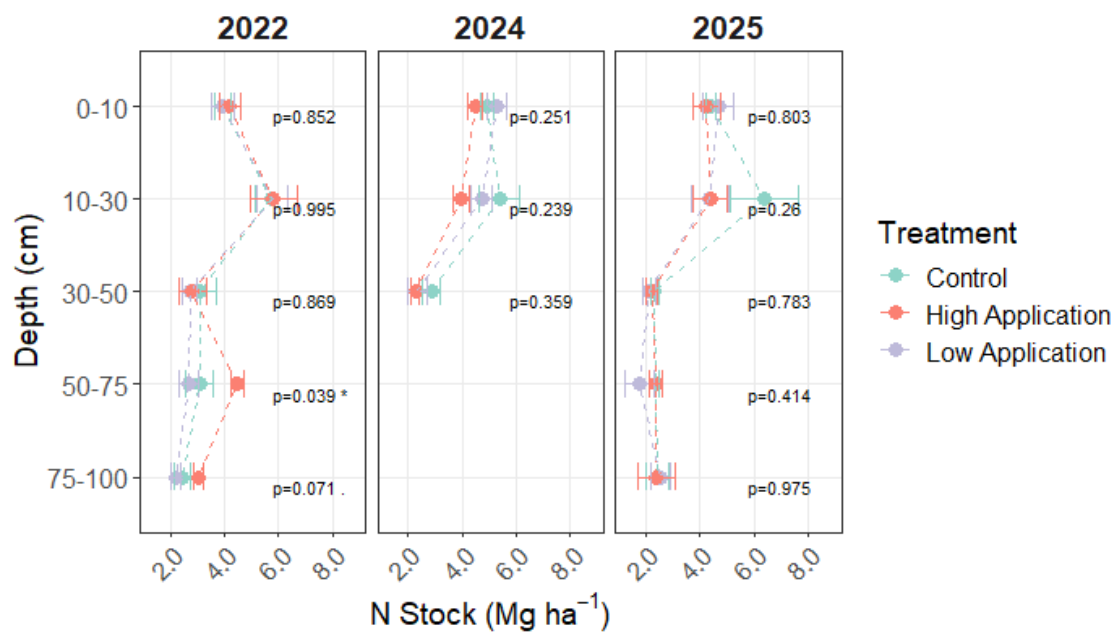


Figure S4-4: Total soil nitrogen stocks (Mg N /ha) by treatment, depth, and year.

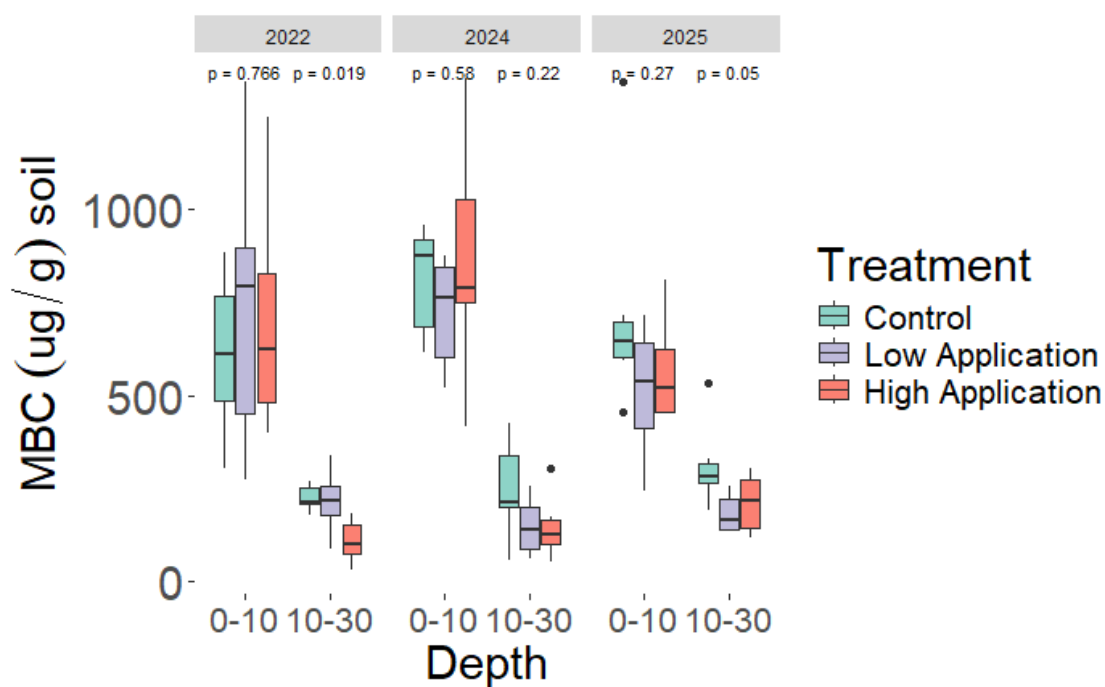


Figure S4-5: MBC (ug / g soil) by depth, year, and treatment.

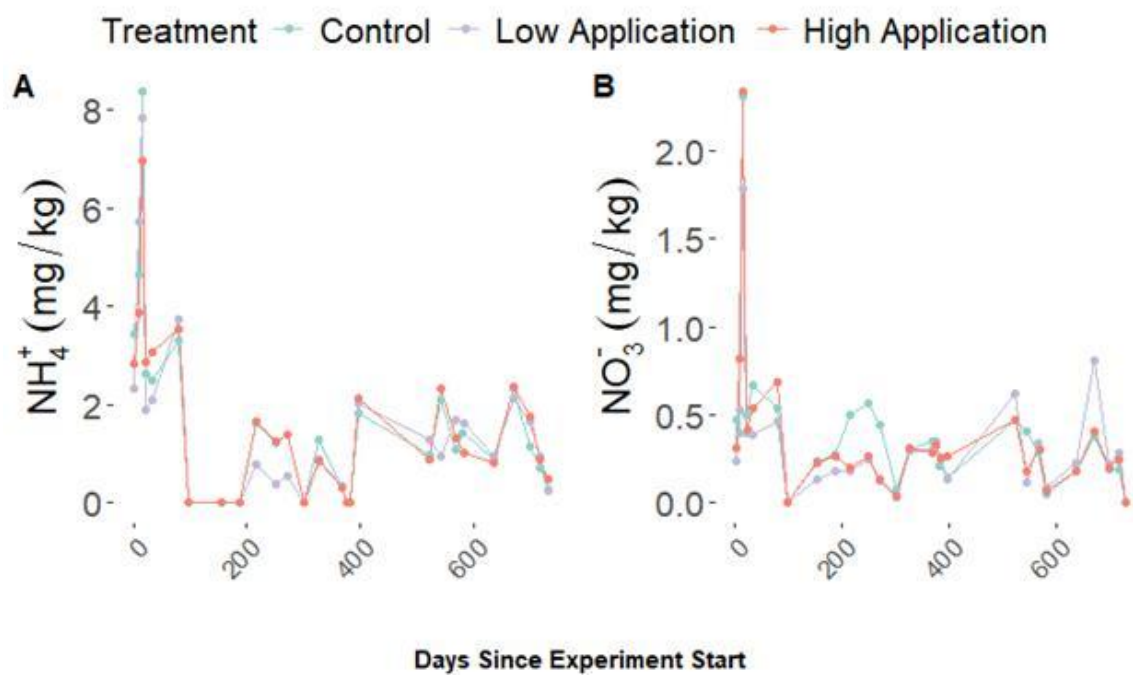


Figure S4-6: Soil ammonium (A) and nitrate (b) levels since the first compost application by treatment.

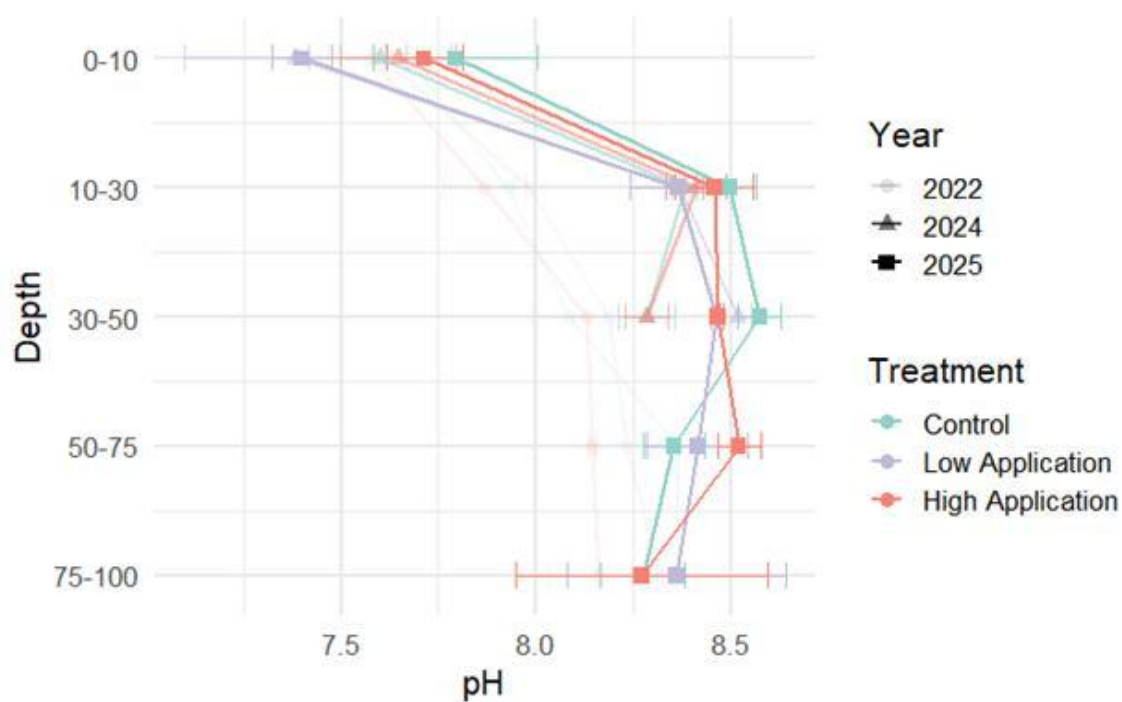


Figure S4-7: Soil pH by treatment, year of sampling, and depth.

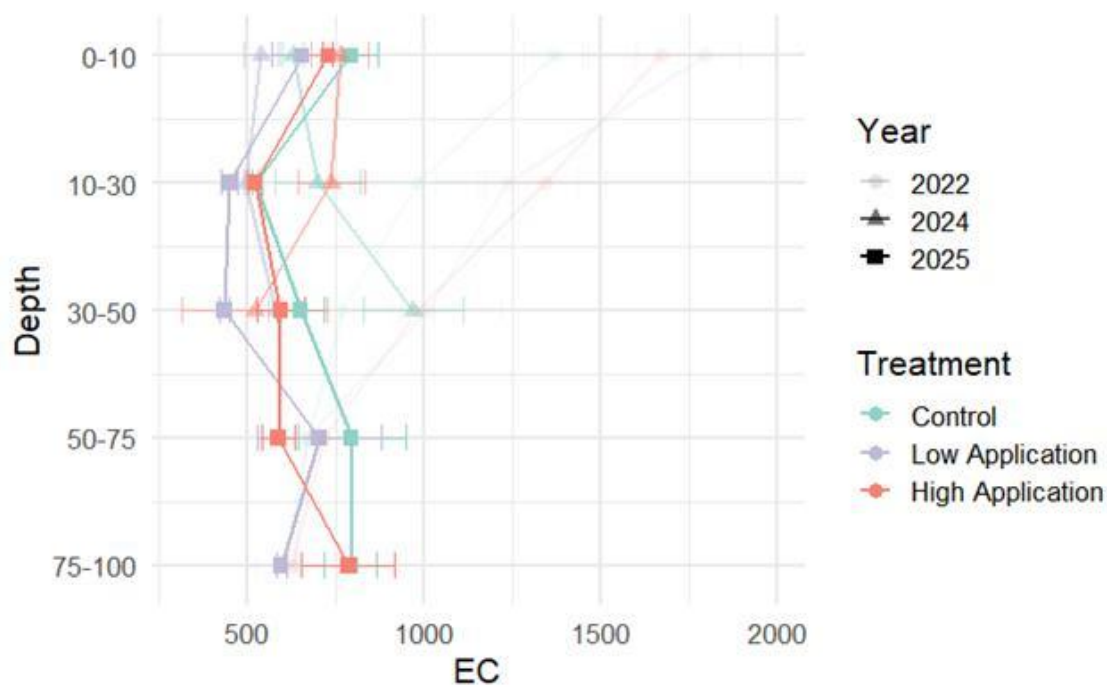


Figure S4-8: Soil electrical conductivity (EC, $\mu\text{S}/\text{cm}$) by treatment, year of sampling, and depth.

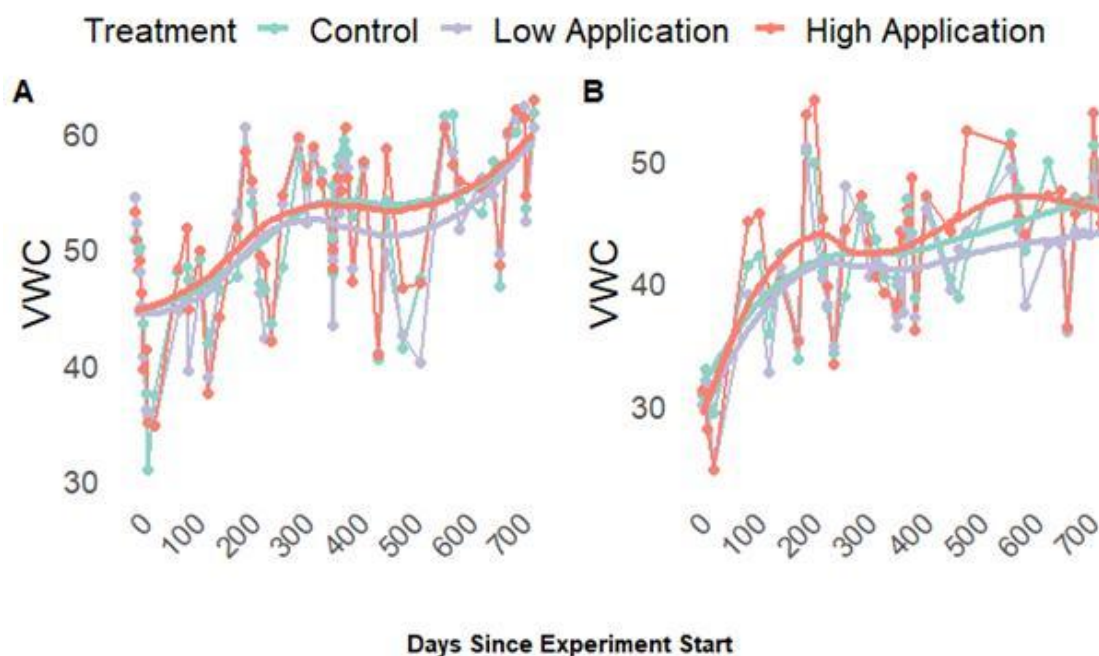


Figure S4-9: Line graph with a LOESS (Locally Estimated Scatterplot Smoothing) curve of Soil VWC at 12cm (A) and 20cm (B) by treatment throughout the course of the experiment.

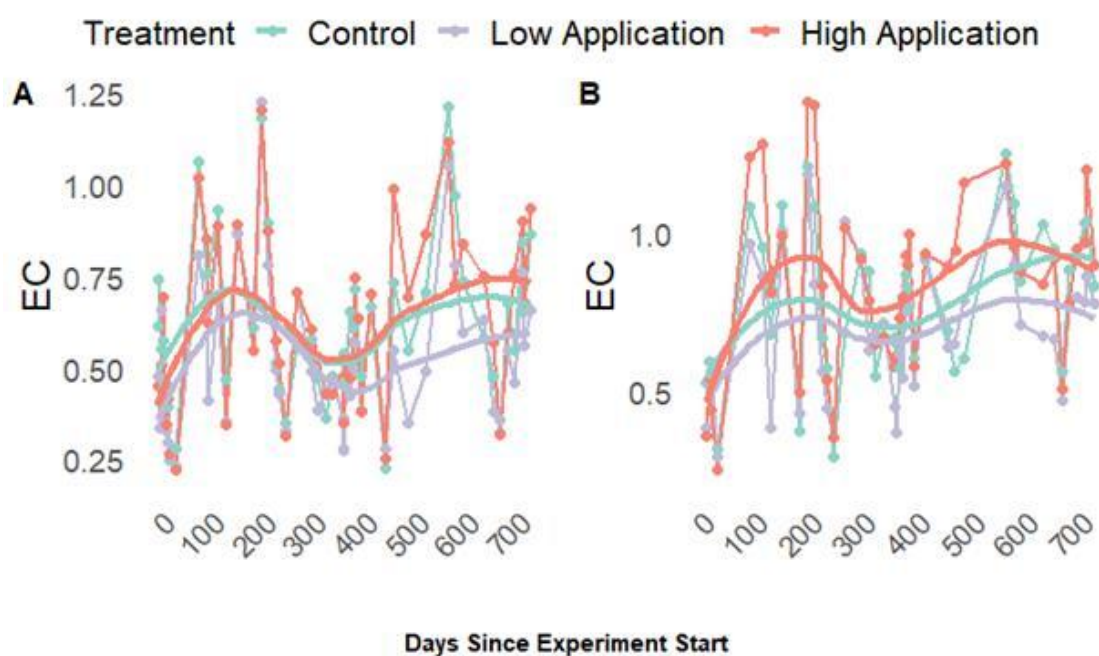
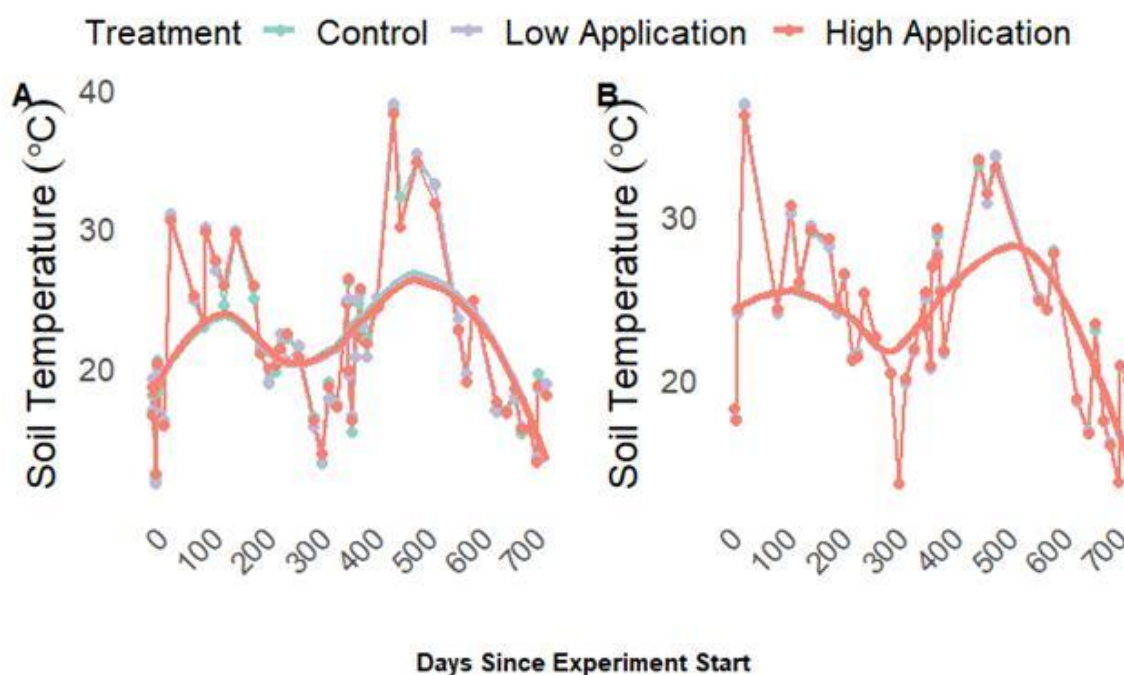


Figure S4-10: Line graph with a LOESS curve of Soil EC (mS/cm) at 12cm (A) and 20cm (B) by treatment throughout the course of the experiment.



SI Figure S4-11: Line graph with a LOESS curve of Soil temperature at 12cm (A) and 20cm (B) by treatment throughout the course of the experiment.

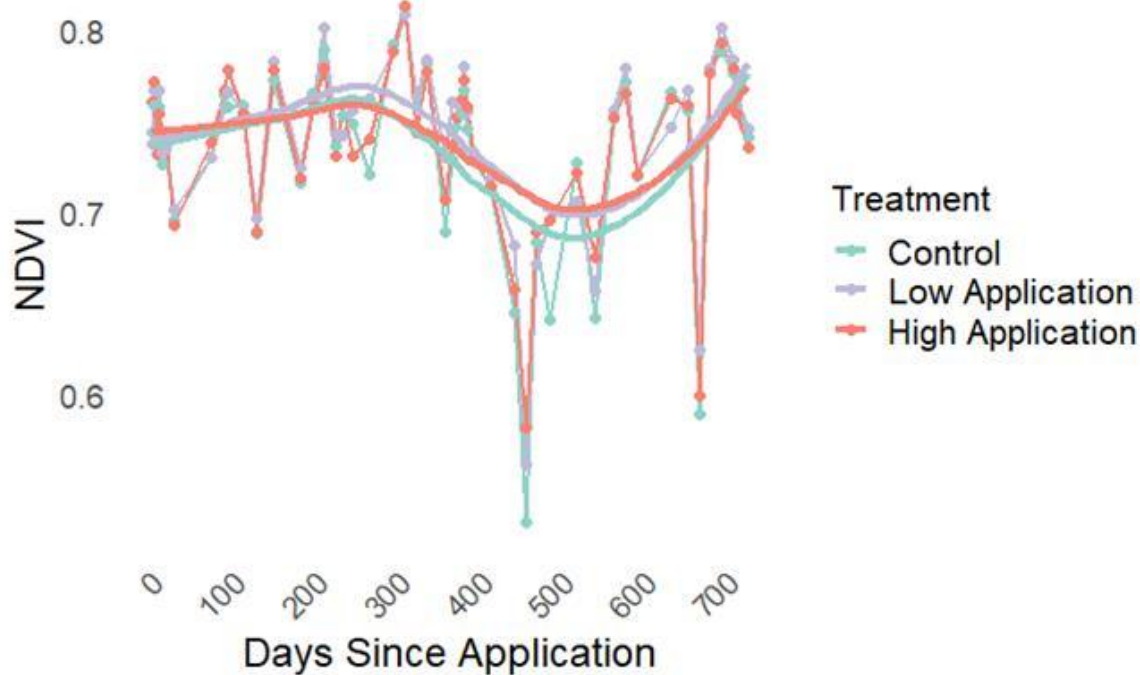


Figure S4-12: Line graph with a LOESS curve of NDVI by treatment throughout the course of the experiment.

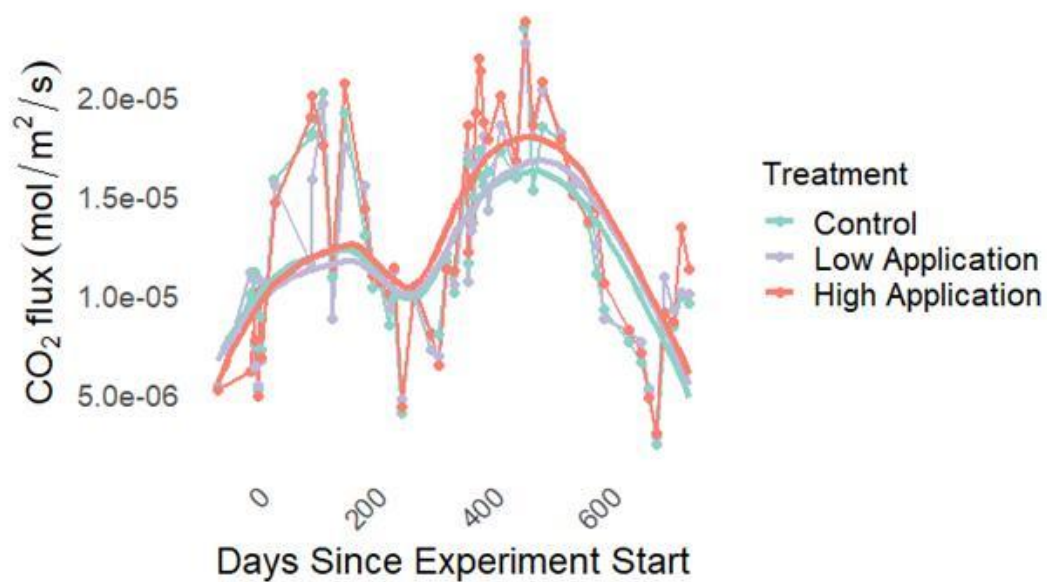


Figure S4-13: Soil CO₂ flux over the course of the experiment by treatment.

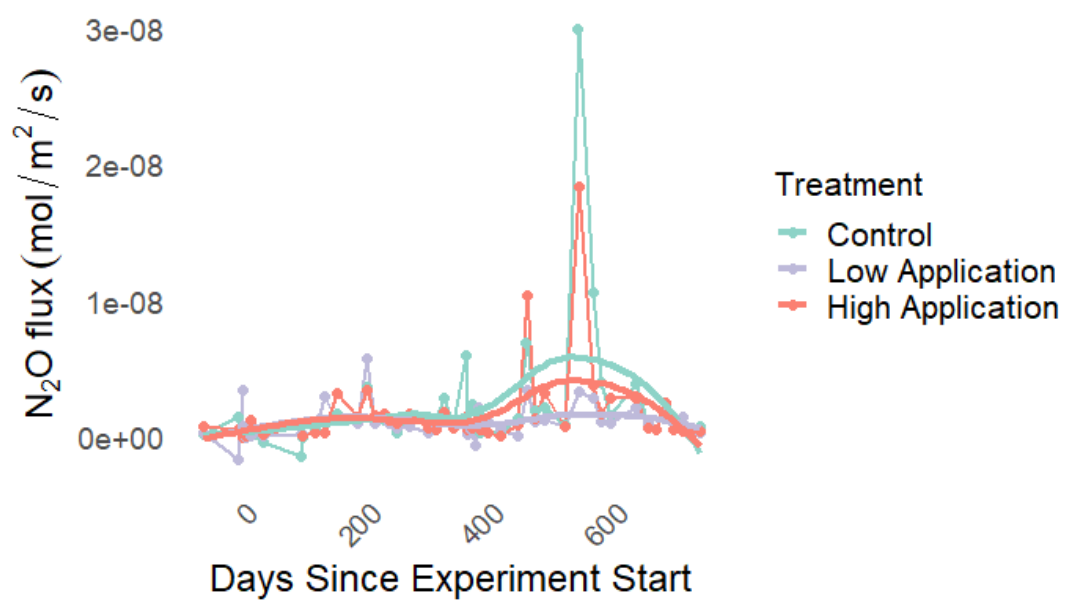


Figure S4-14: Soil N₂O flux over the course of the experiment by treatment.

Appendix D: Supplemental Information for Chapter 5

Table S5-1: Mass recovery and estimate carbon recovery of density fractioned soils by depth and years of compost. Sample size represented by n and $\pm$ represents standard error.

Depth	Years of Compost	n	Mass Recovery	Carbon Recovery
0-10	0	5	83.7% $\pm$ 3.2%	78.4% $\pm$ 5.4%
0-10	20	3	84.4% $\pm$ 5.4%	83.4% $\pm$ 7%
0-10	30	3	89.4% $\pm$ 1.3%	75.8% $\pm$ 0.9%
30-50	0	5	87.1% $\pm$ 1.1%	72.4% $\pm$ 2.1%
30-50	20	3	85.6% $\pm$ 0.7%	56.1% $\pm$ 6.1%
30-50	30	3	80.8% $\pm$ 4.2%	72.8% $\pm$ 4.3%
75-100	0	5	83.1% $\pm$ 2.4%	62.8% $\pm$ 7.8%
75-100	20	3	85.3% $\pm$ 1%	60.6% $\pm$ 4.5%
75-100	30	3	81.1% $\pm$ 4%	64.8% $\pm$ 1.4%

Table S5-1: Soil texture by park and depth. $\pm$ SE shown in parentheses for Amador and KM where field n > 1.

Depth (cm)	Park	Treatment	Very Coarse Sand (1000-2000 μ m)	Coarse Sand (500-1000 μ m)	Medium Sand (250-500 μ m)	Fine Sand (100-250 μ m)	Very Fine Sand (50-100 μ m)	Coarse Silt (20-50 μ m)	Fine Silt (2-20 μ m)	Clay (<2 μ m)
0-10	Amaral	Unamended	1.9	4.4	9.7	15.4	15.2	18.5	20.8	14.1
	Bernal	Unamended	3.6	3.9	4.5	9.5	4.9	14.4	36.7	22.5
	Creekside	Unamended	1.7	2.8	4.6	11.6	13	15.1	20.9	30.3
	Muirwood	Unamended	0.9	2.1	4.9	10.2	11.9	15.8	27.4	26.7
	PU	Unamended	3.2	4.8	7.3	22.6	8.3	10.3	19.2	24.3
	Amador	20 yr compost	1.3 (± 0.3)	2.4 (± 0.4)	4.7 (± 0.3)	11 (± 0.6)	12.3 (± 0.6)	17.8 (± 1.9)	17.9 (± 2.2)	32.7 (± 4.1)
	KM	30 yr compost	1.4 (± 0.2)	3 (± 0.4)	6.5 (± 1)	14.6 (± 1.3)	15.2 (± 1.1)	21.7 (± 5.2)	10.6 (± 3.9)	27.1 (± 0.4)
		Unamended	5.7	12.4	25.3	29.8	7.6	4.5	6.3	8.5
10-30	Amaral	Unamended	5.6	6.6	8	17.2	13.6	18.8	17.9	12.4
	Bernal	Unamended	2	2.8	3.9	5.8	11.6	10.1	36.9	26.9
	Creekside	Unamended	1.8	2.9	4.6	11.4	13.8	13.2	24.2	29.1
	Muirwood	Unamended	1.8	2.3	2.9	11.4	14.6	13.3	22.2	31.5
	PU	Unamended	2.4	0.7	0	9.7	18.3	26.8	23.8	18.4
	Amador	20 yr compost	0.6 (± 0.2)	1.3 (± 0.2)	2.7 (± 0)	4.7 (± 1.4)	12.8 (± 1.2)	29.7 (± 5.2)	28 (± 4.2)	20.4 (± 0.2)

30-50	KM	30 yr compost	0.6 (±0.2)	1.8 (±0.5)	7.9 (±1.3)	20.7 (±3.9)	16.5 (±1.2)	16.9 (±3.2)	18.7 (±2.3)	16.8 (±0.6)
		Unamended	6.5	12.8	24.3	24.5	9.5	8.2	8.2	6
	Amaral	Unamended	3.8	7.1	12.9	17.3	12	18.3	19	9.7
	Bernal	Unamended	1.8	3.1	5.3	14.1	5.3	6.5	39.6	24.4
	Creekside	Unamended	1	2.1	4.1	10.8	15.9	24.9	29	12.2
	Muirwood	Unamended	0.5	1.1	2.8	14	12	28.3	15.8	25.4
	PU	Unamended	3.7	5.3	7.6	15.4	15.9	16.7	14.8	20.7
	Amador	20 yr compost	0.2 (±0.1)	0.7 (±0.2)	2.2 (±0.6)	7.7 (±0.6)	12.1 (±1.6)	19.9 (±2.1)	40.5 (±1.8)	16.5 (±1.1)
	KM	30 yr compost	2.1 (±1.7)	2.8 (±1.1)	7.5 (±1.6)	21.7 (±3.4)	18.2 (±1.8)	9.7 (±1.6)	18.2 (±3.8)	19.6 (±2.8)
		Unamended	3.4	6.9	13.4	17.4	17.6	21.4	2.3	17.6
50-75	Amaral	Unamended	4.6	6.7	9.8	16.8	16.9	18.7	15.2	11.3
	Bernal	Unamended	1.5	3	5.6	8.2	10.2	15	31.2	25.4
	Creekside	Unamended	0.9	2.1	4.7	12.7	14.1	14.9	20.2	30.3
	Muirwood	Unamended	0.5	1.3	3.4	12.4	17.3	18.5	24.8	21.7
	PU	Unamended	4.4	6.4	9.4	13.4	14.7	16.2	13.1	22.3
	Amador	20 yr compost	0.1 (±0.1)	0.5 (±0.3)	1.7 (±1.2)	6.3 (±1.2)	10.1 (±1)	22.4 (±1.7)	37 (±2.6)	21.9 (±1.1)

75-100	KM	30 yr compost	0.2 (±0.1)	0.7 (±0.4)	3.8 (±1.9)	8 (±2.8)	12.2 (±4)	18.8 (±2)	26.7 (±6.5)	29.8 (±3.5)
		Unamended	2.2	5	10.6	12.9	13.2	12.5	15.8	27.8
	Amaral	Unamended	6.9	5.9	4.9	13.5	21.8	22.2	14.7	10.1
	Bernal	Unamended	1.9	2	2.2	11.1	13	12.5	30.5	27
	Creekside	Unamended	0.5	1.1	2.4	7.1	10.1	19	23.7	36.1
	Muirwood	Unamended	0.4	1	2.7	14.5	15.5	19	25	21.9
	PU	Unamended	4	6.3	9.9	15.7	17.3	16.1	12.8	17.8
	Amador	20 yr compost	0.1 (±0)	0.2 (±0)	0.7 (±0.1)	6.3 (±0.8)	11.3 (±1)	21.9 (±1)	36.5 (±1.5)	22.9 (±1.2)
	KM	30 yr compost	0.1 (±0)	0.2 (±0.1)	0.7 (±0.3)	4.2 (±1.2)	6 (±1)	17.7 (±4.6)	38.4 (±2.8)	32.7 (±3.1)
		Unamended	1.5	3.3	6.5	8.6	14.8	13.7	18.9	32.6

Table S5-2: Bulk soil p-values by treatment and depth. p-values are by treatment levels (Tukey HSD) and all compost samples aggregated (t-test). Significance codes: <0.001 ***, <0.01 **, <0.05 *, <0.10 .

Variable	Depth (cm)	0 vs 20 yrs	0 vs 30 yrs	20 vs 30 yrs	Control vs Amended (t-test)
C (%)	0-10	<0.001 ***	0.243	0.004 **	0.001 **
	10-30	0.469	0.277	0.046 *	0.635
	30-50	0.003 **	<0.001 ***	<0.001 ***	0.377

	50-75	<0.001 ***	0.918	0.003 **	0.062 .
	75-100	0.013 *	<0.001 ***	0.924	<0.001 ***
N (%)	0-10	<0.001 ***	0.034 *	0.011 *	<0.001 ***
	10-30	0.3	0.53	0.056 .	0.983
	30-50	0.03 *	0.001 **	<0.001 ***	0.255
	50-75	0.017 *	0.684	0.1	0.069 .
	75-100	0.131	<0.001 ***	0.156	<0.001 ***
C:N	0-10	0.594	<0.001 ***	0.077 .	0.006 **
	10-30	0.262	0.012 *	0.638	0.007 **
	30-50	0.155	0.964	0.114	0.464
	50-75	0.011 *	0.969	0.008 **	0.249
	75-100	0.044 *	0.735	0.012 *	0.551
d13C	0-10	<0.001 ***	0.996	<0.001 ***	<0.001 ***
	10-30	<0.001 ***	0.688	<0.001 ***	0.002 **
	30-50	<0.001 ***	0.002 **	0.068 .	<0.001 ***
	50-75	0.063 .	0.931	0.137	0.19
	75-100	0.306	0.128	0.01 **	0.574
d15N	0-10	<0.001 ***	<0.001 ***	0.317	<0.001 ***
	10-30	<0.001 ***	0.019 *	<0.001 ***	<0.001 ***
	30-50	0.001 **	0.079 .	0.199	0.002 **
	50-75	0.503	0.128	0.843	0.076 .
	75-100	0.52	0.795	0.858	0.358

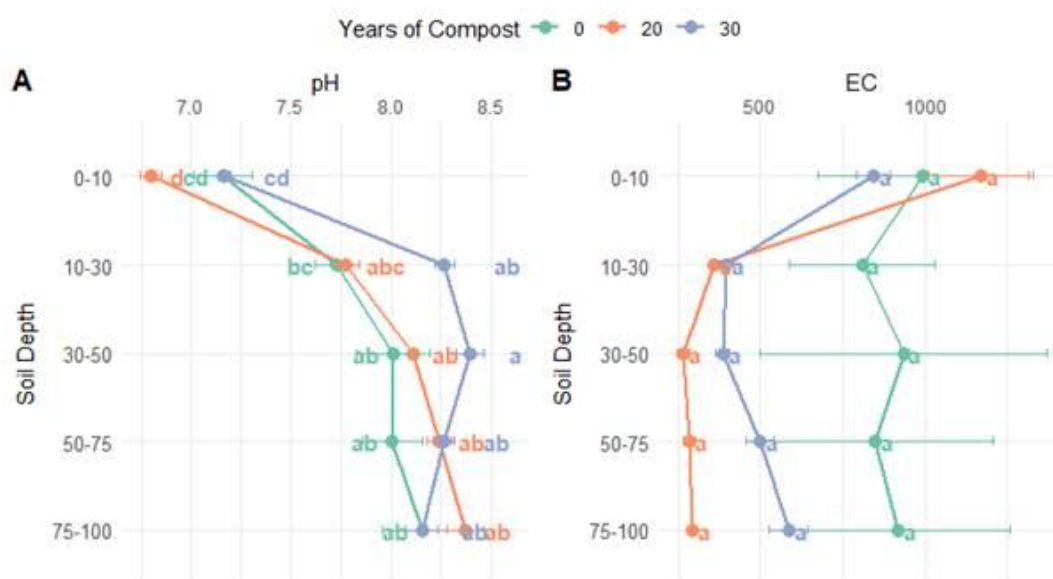


Figure S5-1: Soil pH (A) and EC (B) by years of compost application and depth.

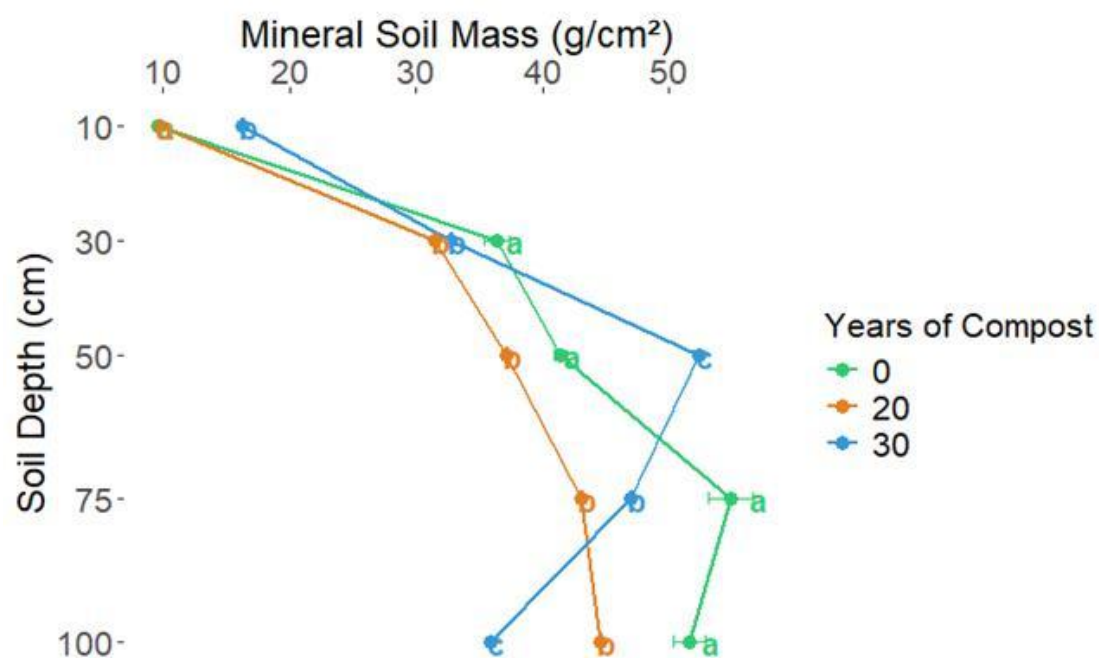


Figure S5-2: Soil mass (g/cm²) for each depth and treatment from the equivalent soil mass method.

Table S5-3: Mass, C, and N comparisons of FLF by depth and years of compost application. p-values are by treatment levels (Tukey HSD) and all compost samples aggregated (t-test). Significance codes: <0.001 ***, <0.01 **, <0.05 *, <0.10 .

Variable	Depth (cm)	0 vs 20 yrs	0 vs 30 yrs	20 vs 30 yrs	Control vs Amended (t-test)
Bulk soil mass (%)	0-10	0.627	0.243	0.095 .	0.698
	30-50	0.989	0.755	0.863	0.625
	75-100	0.93	0.956	0.997	0.706
C (%)	0-10	0.801	0.613	0.953	0.364
	30-50	0.992	0.827	0.804	0.764
	75-100	0.533	0.927	0.792	0.433
C (Mg C/ha)	0-10	0.713	0.017 *	0.011 *	0.3
	30-50	0.93	0.822	0.975	0.599
	75-100	0.756	0.734	0.999	0.429
N (%)	0-10	0.878	0.162	0.388	0.201
	30-50	0.536	0.864	0.861	0.362
	75-100	0.974	0.08 .	0.156	0.172
N (Mg C/ha)	0-10	0.73	0.018 *	0.012 *	0.298
	30-50	0.767	0.899	0.97	0.54
	75-100	0.988	0.894	0.856	0.844
C:N	0-10	0.928	0.84	0.691	0.907
	30-50	0.176	0.915	0.142	0.437
	75-100	0.669	0.348	0.853	0.238

Table S5-4: Mass, C, and N comparisons of OLF by depth and years of compost application. p-values are by treatment levels (Tukey HSD) and all compost samples aggregated (t-test). Significance codes: <0.001 ***, <0.01 **, <0.05 *, <0.10 .

Variable	Depth (cm)	0 vs 20 yrs	0 vs 30 yrs	20 vs 30 yrs	Control vs Amended (t-test)
Bulk soil mass (%)	0-10	0.346	0.455	0.974	0.127
	30-50	1	0.994	0.994	0.955
	75-100	0.944	0.723	0.603	0.794
C (%)	0-10	0.72	0.277	0.695	0.291
	30-50	0.581	0.873	0.886	0.335
	75-100	0.994	0.543	0.661	0.497
C (Mg C/ha)	0-10	0.423	0.198	0.841	0.078 .
	30-50	0.763	0.675	0.38	0.932
	75-100	0.771	0.96	0.923	0.583
N (%)	0-10	0.923	0.371	0.604	0.419
	30-50	0.17	0.995	0.203	0.316
	75-100	0.928	0.325	0.261	0.557
N (Mg C/ha)	0-10	0.358	0.136	0.776	0.051 .
	30-50	0.647	0.627	0.275	0.985
	75-100	0.698	0.984	0.66	0.711
C:N	0-10	0.261	0.118	0.849	0.113
	30-50	0.566	0.646	0.242	0.943
	75-100	0.991	0.308	0.435	0.396
d15N	0-10	0.436	0.174	0.782	0.168
	30-50	0.044 *	0.996	0.062 .	0.223
	75-100	0.892	0.588	0.425	0.77

Table S5-5: Mass, C, and N comparisons of HF by depth and years of compost application. p-values are by treatment levels (Tukey HSD) and all compost samples aggregated (t-test). Significance codes: <0.001 ***, <0.01 **, <0.05 *, <0.10 .

Variable	Depth (cm)	0 vs 20 yrs	0 vs 30 yrs	20 vs 30 yrs	Control vs Amended (t-test)
Bulk soil mass (%)	0-10	0.831	0.918	0.984	0.54
	30-50	0.874	0.163	0.392	0.161
	75-100	0.824	0.843	0.575	0.982
C (%)	0-10	0.251	0.996	0.352	0.364
	30-50	0.997	0.942	0.931	0.886
	75-100	0.227	0.054 .	0.659	0.031 *
C (Mg C/ha)	0-10	0.978	0.163	0.281	0.267
	30-50	0.974	0.58	0.531	0.653
	75-100	0.59	0.775	0.956	0.348
N (%)	0-10	0.408	0.798	0.808	0.3
	30-50	0.999	0.856	0.897	0.739
	75-100	0.399	0.029 *	0.279	0.033 *
N (Mg C/ha)	0-10	0.986	0.104	0.184	0.23
	30-50	0.985	0.38	0.38	0.493
	75-100	0.795	0.626	0.963	0.365
C:N	0-10	0.964	0.341	0.534	0.367
	30-50	0.981	0.962	0.997	0.797
	75-100	0.664	0.947	0.869	0.494

Table S5-6: Mass, C, and N comparisons of soil density fractions by depth and years of compost of compost application. p-values are by treatment levels (Tukey HSD) and all compost samples aggregated (t-test). Significance codes: <0.001 ***, <0.01 **, <0.05 *, <0.10 .

Fraction	Depth (cm)	Treatment	Percent of bulk soil mass	C (%)	C (Mg C/ha)	N (%)	N (Mg C/ha)	C:N
Free Light Fraction	0-10	Control	2.8 (1.19)	22.1 (1.28)	7.02 (2.755)	1.76 (0.076)	0.563 (0.2208)	12.6 (0.53)
		20 Years	1.2 (0.63)	23.3 (0.94)	3.1 (1.551)	1.81 (0.039)	0.235 (0.1214)	12.9 (0.75)
		30 Years	5.9 (1.51)	23.8 (1.18)	24.69 (5.699)	1.97 (0.067)	2.071 (0.5244)	12.1 (0.39)
	30-50	Control	0.2 (0.07)	30.8 (2.39)	3 (1.138)	1.77 (0.285)	0.166 (0.0723)	18.4 (1.86)
		20 Years	0.2 (0.05)	31.4 (2.55)	2.51 (0.545)	1.33 (0.117)	0.106 (0.0226)	23.6 (0.41)
		30 Years	0.1 (0.04)	27.5 (6.91)	2.18 (0.251)	1.57 (0.334)	0.128 (0.0158)	17.3 (2.35)
	75-100	Control	0.1 (0.06)	34.3 (3.28)	3.49 (2.045)	1.1 (0.165)	0.076 (0.0205)	36.5 (10.06)
		20 Years	0.1 (0.03)	29.8 (1.42)	1.72 (0.469)	1.17 (0.142)	0.069 (0.0231)	26 (1.83)
		30 Years	0.1 (0.06)	32.8 (1.18)	1.62 (0.889)	1.93 (0.393)	0.097 (0.0608)	18.7 (4.29)
Occluded Light Fraction	0-10	Control	9.5 (2.92)	27.5 (4.82)	28.37 (7.734)	2.37 (0.326)	2.53 (0.7126)	11.5 (0.54)
		20 Years	16.7 (5.25)	23.5 (1.95)	45.7 (14.728)	2.24 (0.123)	4.364 (1.4347)	10.5 (0.31)
		30 Years	15.6 (0.69)	19 (0.64)	53.61 (1.478)	1.88 (0.058)	5.284 (0.1418)	10.1 (0.06)
	30-50	Control	0.5 (0.08)	34.4 (0.73)	8.41 (1.674)	2.16 (0.107)	0.541 (0.1235)	16.1 (0.94)
		20 Years	0.5 (0.06)	30.6 (5.62)	6.61 (0.682)	1.74 (0.255)	0.377 (0.0194)	17.4 (0.91)
		30 Years	0.5 (0.12)	32.6 (0.56)	10.61 (2.5)	2.18 (0.082)	0.711 (0.1705)	15 (0.34)
	75-100	Control	0.5 (0.15)	37.1 (2.31)	11.27 (3.594)	1.74 (0.185)	0.586 (0.201)	22.7 (3.67)

Fraction	Depth (cm)	Treatment	Percent of bulk soil mass	C (%)	C (Mg C/ha)	N (%)	N (Mg C/ha)	C:N
Heavy Fraction	0-10	20 Years	0.4 (0.08)	36.7 (3.28)	7.92 (1.773)	1.65 (0.111)	0.357 (0.0767)	22.1 (0.54)
		30 Years	0.7 (0.2)	33.4 (0.59)	9.97 (3.335)	2.09 (0.078)	0.633 (0.2255)	16 (0.41)
		Control	71 (3.98)	1.5 (0.17)	13.92 (2.003)	0.17 (0.022)	1.505 (0.2472)	9.5 (0.57)
	30-50	20 Years	66.6 (9.47)	1.9 (0.08)	14.44 (1.692)	0.2 (0.012)	1.557 (0.2149)	9.4 (0.33)
		30 Years	68 (0.46)	1.6 (0.05)	19.2 (0.625)	0.18 (0.008)	2.262 (0.0923)	8.5 (0.08)
		Control	86.4 (1.17)	0.6 (0.2)	22.42 (7.86)	0.06 (0.015)	2.465 (0.6076)	8.4 (0.86)
	75-100	20 Years	84.9 (0.75)	0.5 (0.06)	20.02 (2.354)	0.06 (0.004)	2.305 (0.1389)	8.6 (0.67)
		30 Years	80.2 (4.19)	0.6 (0.18)	33.69 (9.39)	0.07 (0.02)	3.817 (1.0205)	8.7 (0.35)
		Control	82.5 (2.47)	0.4 (0.07)	20.51 (3.572)	0.05 (0.007)	2.605 (0.3128)	7.7 (0.83)
		20 Years	84.8 (0.92)	0.6 (0.08)	25.29 (3.469)	0.07 (0.004)	2.89 (0.194)	8.7 (0.75)
		30 Years	80.3 (4.08)	0.7 (0.03)	23.77 (1.003)	0.08 (0.01)	3.018 (0.3296)	8 (0.71)

Table S5-7: FTIR OM ratios of the HF by depth and years of compost application. p-values are by treatment levels (Tukey HSD) and all compost samples aggregated (t-test). Significance codes: <0.001 ***, <0.01 **, <0.05 *, <0.10 .

FTIR Ratio	Depth	Unamended (0 yr) Mean (±SE)	20 yr Compost Mean (±SE)	30 yr Compost Mean (±SE)	Tukey 0 vs 20 yrs	Tukey 0 vs 30 yrs	Tukey 20 vs 30 yrs	t-test Control vs Amended
Simple Plant OM: Microbial OM	0-10 cm	0.509 (±0.066)	0.619 (±0.029)	0.691 (±0.087)	0.517	0.201	0.783	0.108
	30-50 cm	0.465 (±0.059)	0.633 (±0.051)	0.456 (±0.073)	0.198	0.995	0.232	0.351
	Topsoil (0-50 cm)	0.487 (±0.042)	0.626 (±0.027)	0.574 (±0.073)	0.139	0.439	0.782	0.062 .
	75-100 cm	0.509 (±0.057)	0.618 (±0.037)	0.515 (±0.051)	0.374	0.997	0.481	0.422
	Pooled Depth	0.494 (±0.033)	0.624 (±0.02)	0.554 (±0.05)	0.043 *	0.480	0.453	0.036 *

	Measurements							
Simple Plant OM: Complex Plant OM	0-10 cm	1.07 (±0.177)	1.3 (±0.071)	1.705 (±0.4)	0.77 0	0.19 2	0.54 0	0.143
	30-50 cm	0.913 (±0.134)	1.253 (±0.091)	0.91 (±0.193)	0.27 7	1.00 0	0.34 1	0.377
	Topsoil (0-50 cm)	0.992 (±0.108)	1.277 (±0.053)	1.307 (±0.267)	0.39 5	0.32 4	0.99 1	0.090 .
	75-100 cm	1.005 (±0.159)	1.245 (±0.099)	0.957 (±0.121)	0.51 2	0.97 2	0.46 7	0.620
	Pooled Depth Measurements	0.996 (±0.086)	1.266 (±0.045)	1.191 (±0.185)	0.21 4	0.44 0	0.90 3	0.076 .
Complex Plant OM: Microbial OM	0-10 cm	0.485 (±0.014)	0.478 (±0.027)	0.428 (±0.053)	0.98 3	0.38 1	0.54 0	0.345
	30-50 cm	0.515 (±0.01)	0.506 (±0.031)	0.517 (±0.037)	0.96 6	0.99 7	0.95 4	0.906
	Topsoil (0-50 cm)	0.5 (±0.009)	0.492 (±0.02)	0.472 (±0.035)	0.95 9	0.60 1	0.80 8	0.426
	75-100 cm	0.52 (±0.022)	0.498 (±0.011)	0.542 (±0.016)	0.73 7	0.73 1	0.40 1	0.995
	Pooled Depth Measurements	0.506 (±0.01)	0.494 (±0.013)	0.495 (±0.026)	0.83 8	0.86 8	0.99 8	0.497
All Plant OM:	0-10 cm	0.994 (±0.053)	1.096 (±0.048)	1.119 (±0.059)	0.42 9	0.30 1	0.96 4	0.116

Microbial OM	30-50 cm	0.979 (±0.049)	1.14 (±0.074)	0.973 (±0.036)	0.154	0.997	0.191	0.309
	Topsoil (0-50 cm)	0.987 (±0.034)	1.118 (±0.041)	1.046 (±0.045)	0.068	0.537	0.485	0.052
	75-100 cm	1.029 (±0.037)	1.117 (±0.027)	1.056 (±0.035)	0.251	0.853	0.559	0.228
	Pooled Depth Measurements	1.001 (±0.026)	1.118 (±0.027)	1.049 (±0.031)	0.015 *	0.438	0.282	0.020 *

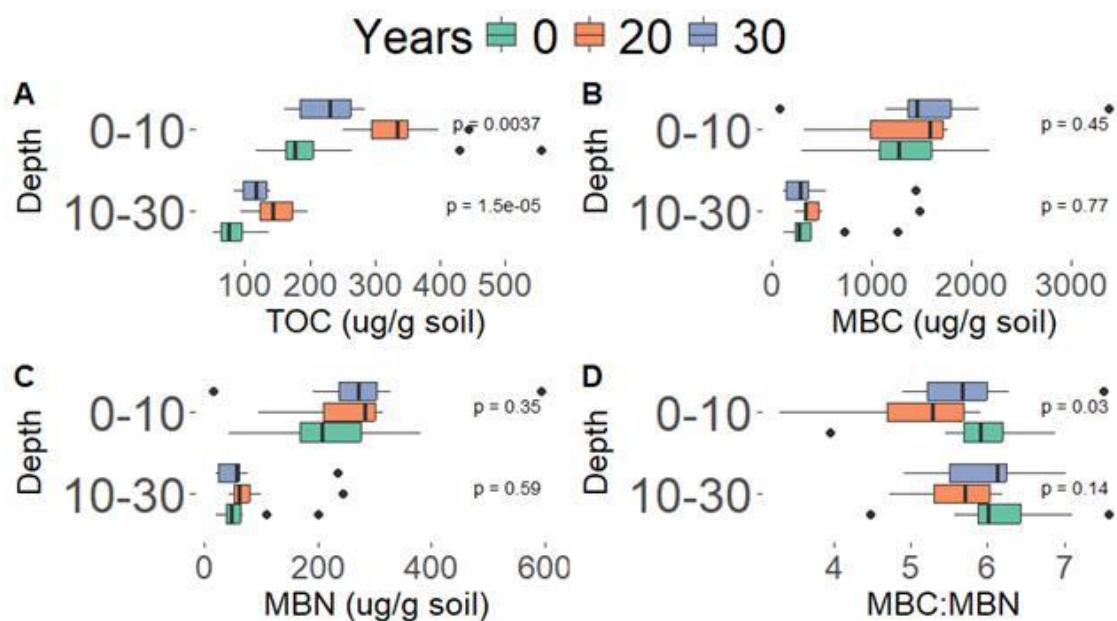


Figure S5-3: Total organic carbon (A), microbial biomass carbon (B), microbial biomass nitrogen (C), and MBC:MBN (D) by depth and years of compost application. P-values represent statistical significance of anova between control (n=15), 20 years of compost (n=9), and 30 years of compost (n=9) treatments.

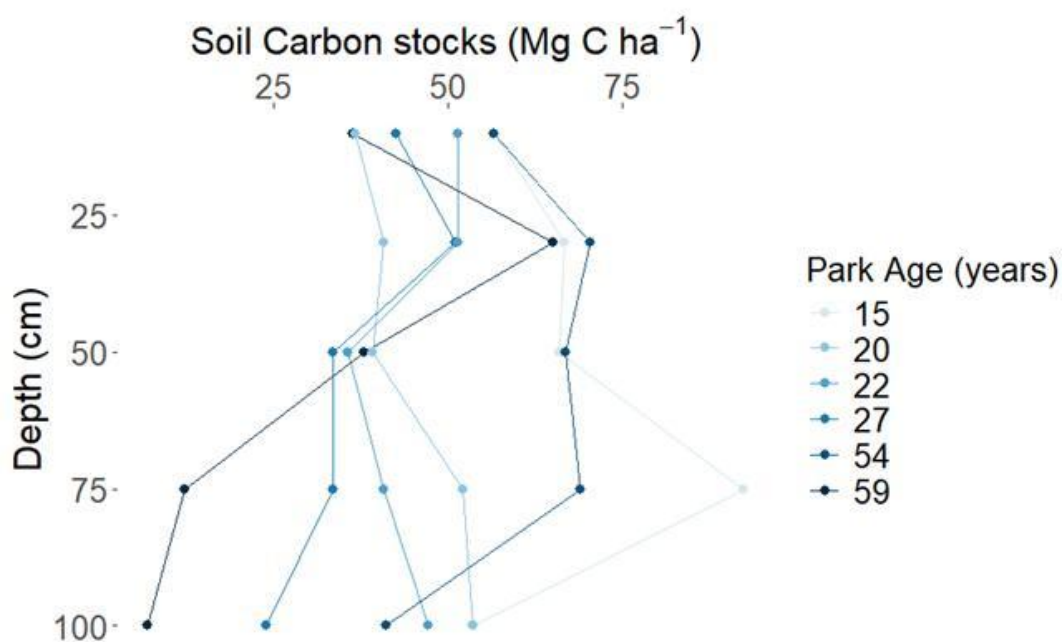


Figure S5-4: Soil carbon stocks (SOC g/cm²) for unamended soils by park age, soil order, and depth.