The Effect of Turfgrass-to-Meadow Restoration on Carbon Storage and Sequestration in an Urban Environment

by

Hayden Epp

A thesis

presented to the University of Waterloo

in fulfilment of the

thesis requirement for the degree of

Master of Science

in

Biology

Waterloo, Ontario, Canada, 2026

© Hayden Epp 2026

Author's Declaration

This thesis consists of material all of which I authored or co-authored: see Statement of Contributions included in the thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

I understand that my thesis may be made electronically available to the public.

Statement of Contributions

This thesis work is my intellectual property, and I take sole responsibility for it. I am the sole author of chapters 1 and 4, which are not intended for publication. I intend to publish chapters 2 and 3 as standalone manuscripts co-authored with Dr. Rebecca Rooney, Dr. Lilhac Medina and Dr. Lyndsay Cartwright. The work for these chapters was co-conceptualized with Dr. Rebecca Rooney and Dr. Lyndsay Cartwright. They also acquired the funding for this project. I completed the field work, data curation, analysis, visualization and writing the original drafts. Dr. Lilhac Medina also helped with field work. Dr. Rebecca Rooney and Dr. Lilhac Medina assisted with project administration, data analysis, and editing the original draft of this thesis document. Dr. Rebecca Rooney was the supervisor for this project and Dr. Lyndsay Cartwright, representing the TRCA, was also responsible for supervision in the field and will contribute to editing the manuscripts for publication. To recognize these contributions, I use “we” throughout the document.

Abstract

Since 2013, Toronto and Region Conservation Authority has been restoring a 16 km hydro corridor in Scarborough, Ontario, actively removing turfgrass and replacing it with meadow. The primary intention of this project, known as The Meadoway, is to improve biodiversity and habitat connectivity in the city, but there is the potential for other co-benefits such as enhanced carbon storage and sequestration. We made use of the section-by-section restoration timeline in a space-for-time substitution to evaluate changes with time since restoration in three categories: unrestored turfgrass, recently restored meadow (restored in 2020-2024), and older restored meadow (restored in 2013-2016). First, we measured CO_2 and CH_4 fluxes biweekly to make estimates of daily net ecosystem exchange, gross primary production, and ecosystem respiration. We found that, cumulatively, both recent (-12.5 ± 19.1 mol m^{-2} $season^{-1}$) and old meadow (-12.3 ± 6.44 mol m^{-2} $season^{-1}$) acted as carbon sinks while unrestored turfgrass ($+14.1 \pm 17.0$ mol m^{-2} $season^{-1}$) acted as a net carbon source over the course of our sampling season. Next, we conducted a 366-day plant litter transplant experiment to investigate whether decomposition rates differ due to litter quality, site conditions, or a combination of both. We found that meadow litter decomposes 33.6% more slowly than turfgrass litter regardless of site, but all litter decomposed 34.7% more at recent meadow sites compared to turfgrass sites. We measured biomass production over the growing season by clipping aboveground biomass and using a modified soil ingrowth core method belowground. We found 272% greater aboveground biomass standing stock, and 213% higher belowground plant production in old meadow compared to turfgrass in 2025. Finally, we measured belowground carbon stocks, using a loss-on-ignition method to measure soil organic matter, soil carbonate content, and root organic matter. We found largely similar belowground carbon stocks in restored meadow as in turfgrass, with the important

exception that the old meadow category had 22.4% lower surface soil organic matter compared to turfgrass. We lack baseline data on pre-restoration soil organic matter and suspect this is attributable to antecedent differences between the sections. In total, we provide convincing evidence that urban turfgrass-to-meadow restoration provides valuable climate mitigation co-benefits and could be considered a nature-based solution to climate change.

Acknowledgements

My thesis work would not have been possible without the help and guidance of many important people. First, I want to thank Dr. Rebecca Rooney for allowing me to stick around in the lab and work on this project. I have grown so much as a scientist under her supervision, and I am deeply grateful to have had this opportunity. Thank you also to my committee members, Dr. Roland Hall, Dr. Tonya DelSontro and Dr. Lyndsay Cartwright for their guidance, questions and feedback throughout this process.

Thanks also to all the great folks at Toronto and Region Conservation Authority who I have gotten to work with on the Meadoway team especially Dr. Lyndsay Cartwright, Sarah Kotsopoulos, and Katie Turnbull. I am also grateful for our partners at Trent University who have been great connections to have throughout this process: Dr. Autumn Watkinson, Jennifer Fedak and Lucy Martin-Johnson. What a cool partnership and project we have been part of! Thank you to the National Sciences and Engineering Research Council of Canada for funding this project.

A huge thank you to all the members of the Waterloo Wetland Lab. Thank you for being such a supportive group! The biggest thank yous go to Dr. Lilhac Medina and Megan Jordan who each spent countless hours helping me with my field and lab work. Lilhac has also been a great mentor throughout this process - thank you for answering my many questions! Thanks also to Jordyne Wong, Ryan Graham, Robyn Larkin, Wade Vanzant and Andrii Kramarenko for your help with field work and to many volunteers who have helped me with lab work. Thanks to Ashlyn DeBoer and Molly Lu for filling out the Meadoway team in our lab - I'm glad to have had the chance to work with you both!

Thank you to all my friends and family for your support over the last two years! You sustain me to keep going on this work and in general. Special shoutout to my housemate, friend and fellow ecologist, Jordan Li, for the many conversations about carbon and greenhouse gas fluxes at home (which our other housemates obviously love).

On the same day that I initially submitted my thesis for distribution to my committee, my Opa, Robert Konrad, went into hospice and passed away a few days later on May 21st, 2026. Opa had a uniquely large impact on who I have become. I don't think I'd be here today doing this type of work if not for the countless hours we spent together in his garden in my growing up years, for example. I'm very grateful for the 26 years I got to spend with him.

Land Acknowledgement

As an ecologist, I want to express my gratitude for the land which sustains me and is the subject of the work presented in this document.

My field work was conducted in Scarborough, Ontario which is the traditional territory of the Mississaugas of the Credit, the Anishinaabeg, the Chippewa, the Haudenosaunee and the Wendat peoples. I also acknowledge Treaty 13 signed with the Mississaugas of the Credit, and the Williams Treaties signed with multiple Mississaugas and Chippewa bands.

As a student at University of Waterloo, I live and work in Waterloo, Ontario which is the traditional territory of the Chonnonton, Anishinaabeg, and Haudenosaunee peoples. My home and school are located on the Haldimand tract which is the land promised to Six Nations in 1784 that includes six miles on either side of the Grand River. I am grateful to have called this land home for much of the last eight years.

I will also name that I have been shaped deeply by the land where I grew up and return to often in Leamington, Ontario – the traditional territory of the Caldwell First Nation and the Three Fires Confederacy of First Nations comprising the Ojibwe, Odawa and Potawatomi people. I would not be who I am without this land and in that way, this land has also shaped my work here.

I recognize that as a settler, I have specific responsibilities in working towards reconciliation and I hope to do so with humility and gratitude to my Indigenous neighbours, past and present.

Table of Contents

Author's Declaration ii

Statement of Contributions iii

Abstract iv

Acknowledgements vi

Land Acknowledgement viii

List of Figures xiii

List of Tables xiv

List of Abbreviations xv

1.0 General Introduction 1

1.1 Urban Green Spaces and Nature-based Solutions 1

1.2 Turfgrass as UGS 2

1.3 Meadows as UGS 3

1.4 The Meadoway 4

1.5 Carbon Sequestration in Turfgrass and Meadow Ecosystems 5

1.6 Study Design and Research Objectives 8

2.0 Greenhouse Gas Fluxes and Litter Decomposition Rates in The Meadoway 11

2.1 Introduction 11

2.2 Methods 16
2.2.1 Site Selection 16
2.2.2 Gas Flux 17
2.2.3 Plant Litter Transplant Experiment 20
2.2.4 Statistical Analysis 21
2.3 Results 23
2.3.1 Gas Flux 23
2.3.2 Plant Litter Transplant Experiment 31
2.4 Discussion 33
2.4.1 Gas Flux 34
2.4.2 Plant Litter Transplant Experiment 39
2.5 Conclusions and Next Steps 43
3.0 Plant Production and Belowground Carbon Stocks in The Meadoway 45
3.1 Introduction 45
3.2 Methods 48
3.2.1 Site Selection 48
3.2.2 Aboveground Biomass Production 50
3.2.3 Belowground Carbon Stocks 51
3.2.4 Belowground Biomass Production 52
3.2.5 Statistical Analysis 53
3.3 Results 54
3.3.1 Aboveground Biomass Production 54

3.3.2 Belowground Carbon Stocks 55
3.3.3 Belowground Biomass Production 59
3.4 Discussion 60
3.4.1 Aboveground Biomass Production 61
3.4.2 Belowground Carbon Stocks 63
3.4.3 Belowground Biomass Production 66
3.5 Conclusions and Next Steps 67
Conclusion 69
4.1 Thesis Overview 69
4.2 Thesis Summary 70
4.3 Overall Discussion and Recommendations 72
References 77
Appendices 93
Appendix A – Imputation of Missing Flux Variable Data 93
Appendix B – Sample Gas Flux Calculations 94
Appendix C – Ancillary Analysis for Litter Bag Content Ratio 96
Appendix D – Two-parameter Decay Model 97
Appendix E – Diagnostic Plots for Flux Models 98
Appendix F – Diagnostic Plots for Litter Decay & Mass Lost 105
Appendix G – Collection Dates for Chapter 3 107

Appendix H – Diagnostic Plots for Aboveground Biomass 108

Appendix I – Diagnostic Plots for Soil Cores.....111

Appendix J – Diagnostic Plots for Ingrowth Cores 131

Appendix K – Peak Aboveground Biomass Determination 133

Appendix L – Raw Data..... 134

List of Figures

Figure 1.1 Conceptual diagram 10

Figure 2.1 Map of The Meadoway 17

Figure 2.2 Cumulative NEE, GPP and ER from June 10th to November 20th, 2025 24

Figure 2.3 Scaled Daily NEE, GPP and ER by sampling week in 2025 26

Figure 2.4 Mean soil water content (%) by sampling week 27

Figure 2.5 Relationship between soil water content and each of NEE, GPP and ER 29

Figure 2.6 Scaled daily methane flux by sampling week in 2025 30

Figure 2.7 Mass remaining over time throughout year-long litter transplant experiment by litter category and site category 32

Figure 2.8 Soil Water Content and Soil Temperature over the course of the season for the subset of sites used in the litter bag transplant experiment 33

Figure 3.1 Map of The Meadoway 49

Figure 3.2 Aboveground biomass reported as dry weight (g/m^2) in 2024 and 2025 54

Figure 3.3 Belowground carbon stocks in the top 10 cm from 2024 and 2025 separated into soil organic matter, soil carbonate content, root dry weight and root organic matter 56

Figure 3.4 Belowground carbon stocks from 2025 reaching 30 cm deep separated into soil organic matter, soil carbonate content, root dry weight and root organic matter 58

Figure 3.5 Belowground plant production in the top 30 cm from the 2025 growing season presented as total root dry mass and root organic matter only 60

Figure 4.1 Conceptual diagram showing the different carbon pools with arrows representing the fluxes between pools for each restoration age category 72

List of Tables

Table 2.1 List of sites and their year of restoration 16

Table 2.2 Summary of seasonal greenhouse gas flux models 28

Table 3.1 List of sites and summary of soil core collection, soil ingrowth core installation and collection, and aboveground biomass collection 50

Table 3.2 Summary of t-test results comparing aboveground biomass in 2024 and 2025 for each restoration category 55

Table 3.3 Summary of ANOVA results for belowground carbon fractions 57

Table 3.4 Summary of two-way ANOVA results for belowground carbon fractions with restoration category and depth as covariates 59

List of Abbreviations

ER	Ecosystem respiration
GPP	Gross primary production
LOI	Loss-on-ignition
NbS	Nature-based solutions
NEE	Net ecosystem exchange
PAR	Photosynthetically active radiation
RDW	Root dry weight
ROM	Root organic matter
SCC	Soil carbonate content
SOC	Soil organic carbon
SOM	Soil organic matter
TRCA	Toronto and Region Conservation Authority
UGS	Urban green spaces

1.0 General Introduction

1.1 Urban Green Spaces and Nature-based Solutions

Approximately 75% of Canadians live in urban areas and urbanization continues to be on the rise across the country (Statistics Canada, 2026). The "triple planetary crisis" of biodiversity loss, climate change and pollution also continues to intensify (Miao & Nduneseokwu, 2024; UNFCCC, 2022). Urban areas are particularly connected to the triple planetary crisis as they are areas of relatively low biodiversity and high pollution (Grimm et al., 2008). They are also uniquely intertwined with climate change being sources of greenhouse gas emissions (Jasek-Kamińska et al., 2020) and disproportionately facing consequences such as increased flooding and urban heat (Gaur et al., 2019; Y. Wang et al., 2016). In addition to a myriad of well-documented human health and well-being benefits (Kondo et al., 2018), urban green spaces (UGS) provide a variety of ecosystem services that can be leveraged as nature-based solutions (NbS) with ecological, social and economic benefits (Pereira et al., 2023).

UGS is a broad term that includes all natural, semi-natural, and artificial ecological systems within and around a city (Cilliers et al., 2013). UGS are variable in their quality, with different levels of ecological integrity, functionality, and service provision (Aronson et al., 2014). NbS are increasingly implemented in urban areas to address interconnected challenges of urbanization and the triple planetary crisis through UGS restoration, often offering multiple co-benefits (Cohen-Shacham et al., 2016; Frantzeskaki, 2019). The Government of Canada has committed to NbS implementation as part of their commitment to reach net-zero greenhouse gas emissions by 2050 (Government of Canada, 2025). This commitment involves funding NbS projects through the Nature Smart Climate Solutions Fund and specifically supporting Indigenous-led initiatives (Government of Canada, 2025).

1.2 Turfgrass as UGS

Turfgrass is a ubiquitous vegetation type in many urban landscapes. UGS maintained as turfgrass is desired primarily for its aesthetic, recreational and social value (Ignatieva et al., 2015; Monteiro, 2017). Lawns are a significant part of the culture around desired home aesthetics in most of the Western world (Ignatieva et al., 2015). Ongoing maintenance is typically a feature of urban turfgrass, but the specific maintenance regime depends on the use and context of the turfgrass (Ignatieva et al., 2015). Lawn maintenance typically involves regular mowing and sometimes watering, fertilizer use, weeding, and/or pesticide use depending on the local climate and the values of the landowner (Ignatieva et al., 2015). Turfgrass is also a feature of urban parks, golf courses, and infrastructure corridors with different typical maintenance regimes for each (Ignatieva et al., 2015; Kotsopoulos et al., 2024).

There is increasing interest in quantifying the ecosystem services provided by urban turfgrass ecosystems (Ignatieva et al., 2015; Townsend-Small & Czimczik, 2010). These include urban heat mitigation, water infiltration, and carbon sequestration, among others (Monteiro, 2017; Thompson & Kao-Kniffin, 2017). It is difficult to make general claims about turfgrass ecosystems as ecosystem service provision can be heavily moderated by different maintenance regimes. For example, lawns have often been shown to be carbon sinks, though carbon dynamics will vary depending on mowing and watering frequency as well as fertilizer use (Selhorst & Lal, 2013; Trémeau et al., 2024). Complicating this further, emissions associated with maintenance could negate the carbon sequestration benefits provided by the turfgrass ecosystem (Ignatieva et al., 2015). Despite the ecosystem services provided by UGS maintained as turfgrass, compared to other ecosystem types, highly managed turfgrass is generally considered to be of low value, especially from a biodiversity perspective (Marshall et al., 2023; Winkler et al., 2024). Other

grassland ecosystem types, such as meadows, can often provide similar or improved ecosystem services, without the necessary maintenance of turfgrass ecosystems (Bretzel et al., 2024; Marshall et al., 2023; Paudel & States, 2023).

1.3 Meadows as UGS

Wildflower meadows are gaining attention as NbS especially to the biodiversity crisis, improving not only the diversity of plants but also attracting a more diverse community of invertebrates and microbes compared to turfgrass (Norton et al., 2019). Meadows are semi-natural grasslands which prominently feature native forbs and are mown infrequently (Kotsopoulos et al., 2024; Marshall et al., 2023; Norton et al., 2019). They are an especially compelling urban NbS option as they offer a "win-win" in situations where conservation needs might otherwise be at odds with the public's desire for spaces they consider aesthetically pleasing (Bretzel et al., 2024). Meadow establishment in urban areas typically involves replacing turfgrass with meadow (Ignatieva et al., 2020). There are a variety of methods for turfgrass-to-meadow restoration, ranging from passively managing lawns less intensively to actively removing turfgrass and planting native meadow, depending on the intention of the restoration (Paudel & States, 2023). Regardless of the method, turfgrass-to-meadow restoration is likely to improve biodiversity and generally enhance ecosystem services (Norton et al., 2019). Additionally, they typically require greatly reduced maintenance compared to turfgrass which, at the very least, gets mowed several times each growing season (Marshall et al., 2023; Norton et al., 2019).

1.4 The Meadoway

Since 2013, Toronto and Region Conservation Authority (TRCA) has been restoring a 16 km hydro corridor in Toronto, Ontario, replacing turfgrass with meadow (Toronto and Region Conservation Authority, 2023). Dubbed “The Meadoway”, this restored linear UGS improves habitat quality and connectivity in the city, connecting the Rouge National Urban Park with several other UGS in the city including ravines and city parks (Toronto and Region Conservation Authority, 2023). Such a restoration has the potential to enhance biodiversity and other ecosystem services (Kotsopoulos et al., 2024; Russell et al., 2018; Wagner et al., 2019).

The Meadoway is divided into seven sections, each having been restored at a different time. The seed mixes and exact preparation methods for restoration have varied from site to site as TRCA adjusts their methods based on restoration success (Toronto and Region Conservation Authority, 2025). For example, TRCA started using multiple rounds of tillage to prepare sites in newer sites (restored in 2020 or later), but this did not take place in older sites (restored in 2016 or earlier). Regardless, restoration was always active and intentional with turfgrass being removed completely before seeding with native seeds. Shrub nodes and turfgrass borders have been added alongside the main restored meadow. As of 2024, each section planned for restoration has been initiated. TRCA monitors the success of restoration in each section, and they will continue to maintain and adaptively manage The Meadoway as an urban natural heritage feature. Turfgrass borders get mowed 6 times per growing season and the meadow sections get mowed every 3-4 years to prevent shrub encroachment and simulate natural disturbances such as grazing or fire (Kotsopoulos et al., 2024; Toronto and Region Conservation Authority, 2025).

The Meadoway restoration project is among the largest linear UGS in Canada (Toronto and Region Conservation Authority, 2023). As it is a unique project, TRCA has created “The

Meadoway Manual", a "how-to" guide on urban turfgrass-to-meadow restoration to share their knowledge and lessons learned from the project for application across North America and beyond in similar contexts (Toronto and Region Conservation Authority, 2025). To deepen that knowledge, TRCA has partnered with universities in Ontario to help quantify the benefits of turfgrass-to-meadow restoration using The Meadoway as a study system. Our project is part of that partnership, looking to quantify the effect of turfgrass-to-meadow restoration on carbon storage and sequestration. If shown to be successful in this regard, The Meadoway could be a model of a high-quality UGS for other cities in North America, which could have a much larger impact on nation-wide climate change mitigation through NbS implementation.

1.5 Carbon Sequestration in Turfgrass and Meadow Ecosystems

There are a variety of reasons to expect enhanced carbon sequestration following turfgrass-to-meadow restoration. Turfgrass ecosystems have been shown to sequester carbon, though this can vary greatly depending on factors such as local climate, species composition and maintenance regimes (Selhorst & Lal, 2013; R. Wang et al., 2022). There is evidence linking diversity of primary producers to increased standing biomass production (Cardinale et al., 2011; Onandia et al., 2019). In general, ecosystems with more diverse plant communities show higher productivity and carbon storage (Chen et al., 2020; Fornara & Tilman, 2008; Y. Yang et al., 2019). We would expect meadow to be an improvement over turfgrass because they are more heterogenous in terms of plant functional types, containing a variety of forbs and grasses.

More than just a diversity effect, annual aboveground biomass production is expected to be higher in meadows compared to turfgrass, especially when regular mowing in turfgrass is

considered. Belowground biomass production is also expected to be higher in meadows due to the presence of deeper rooting species (Brown et al., 2010; Trémeau et al., 2024; Weaver, 1968). Plant biomass production is important for carbon: the rate of net primary production determines the rate of carbon sequestration by vegetation, accumulating in the vegetation's standing crop biomass. Because meadow plants are mainly herbaceous, this standing crop is a temporary stock of carbon that is released when plant matter senesces and oxidizes. Some fraction of the plant biomass may resist oxidation and enter the soil as organic matter (SOM), increasing the long-term store of carbon in the meadow soil. Additionally, more aboveground biomass should result in cooler soils due to shading (Byrne et al., 2008). These site conditions could result in slower rates of decomposition, increasing SOM accumulation (Kirschbaum, 1995; Pouyat & Carreiro, 2003; D. Wang et al., 2016).

There is mixed evidence that site-specific microbial communities may specialize in decomposing local litter, resulting in a "home-field advantage" where litter is more easily decomposed by its home community (Ayres et al., 2006; Gholz et al., 2000; Strickland et al., 2009). There is evidence that this effect could depend on litter quality (Pugnaire et al., 2023). Meadow litter is more heterogenous in composition compared to turfgrass and is expected to contain more recalcitrant components. Litter mixtures have been shown to have non-additive effects on litter decomposition rates (Gartner & Cardon, 2004), but more recalcitrant litter has been shown to decompose more slowly in a mixture (L. Wang et al., 2022). Thus, meadow litter is generally expected to decompose more slowly than turfgrass litter, making it more likely to contribute to lasting carbon stocks.

Plant production and decomposition rates will affect greenhouse gas fluxes in these ecosystems. Higher plant production should correspond with increased gross primary production

(GPP). Slower rates of decomposition should reduce ecosystem respiration (ER). Thus, net ecosystem exchange (NEE) is expected to be lower, signifying carbon sequestration if NEE is negative. Two studies have compared greenhouse gas fluxes between turfgrass and meadow-like ecosystems and found that turfgrass sequestered more carbon. First, Thienelt & Anderson (2021) compared CO_2 fluxes between a lawn and a xeric tallgrass prairie in Denver, Colorado, USA. They note that management factors such as irrigation and fertilizer in the lawn enhanced its carbon sequestration compared to the unwatered and unfertilized meadow (Thienelt & Anderson, 2021). Trémeau et al. (2024) studied CO_2 fluxes in Helsinki, Finland and also found that management regime enhanced the carbon sequestration by lawn. Importantly, unlike the lawns in these two prior studies, the Meadoway turfgrass is not watered or fertilized, and so we expect it to sequester less carbon than reported in these studies. Neither prior study quantified underlying mechanisms like biomass production and decomposition to complement their flux measurements. If these mechanisms work as we have already described here, we expect different CO_2 flux results than those presented in these two studies.

Aerated terrestrial soils are generally expected to be methane sinks due to the presence of methanotrophic bacteria (Le Mer & Roger, 2001), so there is generally no reason to expect a difference in methane flux between turfgrass and meadow, unless nitrogen fertilizer application is part of the ecosystem's management regime (Mosier et al., 1991). Trémeau et al. (2024) found no difference in methane flux following lawn-to-meadow conversion. However, there is some evidence that usually well-drained prairie soils may become methane sources if flooded, such as after spring melt (F. L. Wang & Bettany, 1995). It is possible that the same effect could be found in urban turfgrass and meadow ecosystems, though it is unclear how or if it would differ between these ecosystem types.

If meadow is sequestering more carbon than turfgrass, belowground carbon stocks should be greater in meadow than turfgrass as organic matter accumulates with time. Belowground carbon stocks can take the form of soil organic and inorganic carbon (Pouyat et al., 2009; Wilsey et al., 2020), as well as root stocks. A few studies have compared soil organic carbon stocks between turfgrass and meadow-like ecosystems. Studies from Helsinki and Denver both found higher soil carbon stocks in turfgrass ecosystems compared to meadow or shortgrass steppe ecosystems (Malmsten, 2023; Pouyat et al., 2009). Malmsten (2023) highlights the importance of studying meadow sites which used to be lawns because it is possible that the lawns in their study were established on healthier soil than the meadows. Marshall et al. (2023) compared soil carbon stocks between a well-established lawn and a newly established wildflower meadow portion of the lawn at King's College in Cambridge, England. They found no difference in soil organic carbon (SOC) stocks between the lawn and the meadow, though they only sampled the top 10 cm and highlight that deeper rooting species in the meadow could result in increased soil carbon deeper than 10 cm (Marshall et al., 2023). The meadow in their study was also just two years old when they sampled and they note that it is difficult to detect changes in soil carbon on such a short timescale (Marshall et al., 2023; Smith, 2004). It is possible that a change in SOC will be detectable with more time.

1.6 Study Design and Research Objectives

The overall objective of our project is to quantify the effect of turfgrass-to-meadow restoration on carbon storage and sequestration using The Meadoway as a study site. As previously mentioned, The Meadoway was restored in seven sections over time from 2013-2024. More specifically, there were two "batches" of restorations with the oldest sections restored from

2013-2016 and newer sections restored from 2020-2024. Baseline data on soil organic matter content or carbon flux rates was not collected, but patches of unrestored turfgrass still exist throughout The Meadoway as one large field as well as borders separating the meadow fields from paths and roads. The sequence of section restoration through time sets up a chronosequence which allows us to study the effect of meadow age on carbon dynamics using a space-for-time substitution. Specifically, we will make comparisons between three categories: unrestored turfgrass, recently restored meadow (restored in 2020-2024), and older restored meadow (restored in 2013-2016).

Our objective is to compare various aspects of carbon dynamics between the three restoration age categories to assess whether meadow performs better than mowed turfgrass when it comes to carbon sequestration and long-term storage. In Chapter 2, we will study plant litter decomposition using a reciprocal litter transplant experiment to partition the effect of litter chemical composition from the effect of site conditions on decay rates. We will also directly measure CO_2 and CH_4 fluxes to assess net carbon uptake or emission from meadow and turfgrass sites. In Chapter 3, we will measure annual aboveground and belowground biomass production by clipping aboveground biomass at its peak and using a modified soil ingrowth core method, respectively. We will also compare belowground carbon stocks among turfgrass, newly restored meadow and fully restored meadow sections, by taking soil cores and using a loss-on-ignition method to quantify soil and root organic matter and soil carbonate content. Figure 1.1 helps to visualize the different components of the overall carbon budget that we measured and will present in this thesis. We will compare results for each of these between the three restoration age categories to assess any age-related effects of restoration, keeping in mind that older restored

meadow is likely more representative of the long-term condition for restored urban meadow, at least from a carbon perspective.

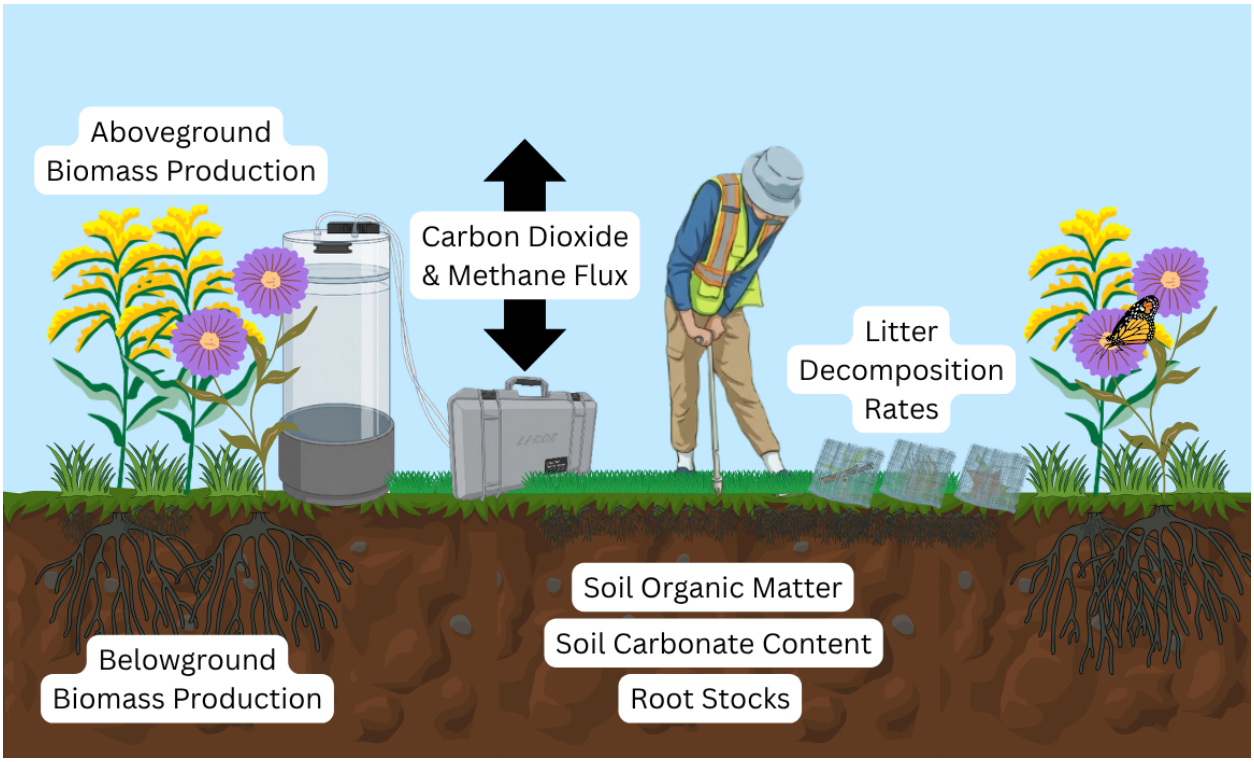


Figure 1.1 Conceptual diagram showing the different parts of the overall carbon budget that we measured for this thesis. Created using BioRender.com & Canva.com.

2.0 Greenhouse Gas Fluxes and Litter Decomposition Rates in The Meadoway

2.1 Introduction

Approximately 75% of Canadians live in urban areas and urbanization continues to be on the rise across the country (Statistics Canada, 2026). Meanwhile, the "triple planetary crisis" of biodiversity loss, climate change and pollution also continues to intensify (Miao & Nduneseokwu, 2024; UNFCCC, 2022). Urban areas are uniquely intertwined with climate change, being net sources of CO_2 (Jasek-Kamińska et al., 2020) and disproportionately facing consequences such as increased flooding and urban heat (Gaur et al., 2019; Y. Wang et al., 2016). Thus, UGS are gaining attention as potential NbS in the face of the triple planetary crisis with ecological, social and economic benefits (Cohen-Shacham et al., 2016; European Commission: Directorate-General for Research and Innovation, 2015).

UGS are variable in their quality, with different levels of ecological integrity, functionality and service provision (Niemelä et al., 2010). Turfgrass as UGS is ubiquitous in many parts of the urban landscape from home lawns, urban parks, sports fields, roadsides and infrastructure corridors (Ignatieva et al., 2015; Kotsopoulos et al., 2024). Turfgrass maintenance is varied, but typically involves regular mowing and sometimes watering, fertilizer application, weeding and/or pesticide use depending on land use, local climate and landowner values (Ignatieva et al., 2015). Nevertheless, mowed turfgrass is generally considered to be lower quality UGS compared to alternatives such as native plant meadow (Bretzel et al., 2024; Marshall et al., 2023).

Meadows are semi-natural grasslands which prominently feature native forbs (Kotsopoulos et al., 2024; Marshall et al., 2023; Norton et al., 2019). Wildflower meadows are an

effective NbS for the biodiversity crisis, improving not only the diversity of plants but also attracting a more diverse community of invertebrates and microbes compared to turfgrass (Norton et al., 2019). Meadows also require less maintenance, watering, fertilizer, and mowing than turfgrass, although some level of maintenance such as mowing or controlled burns every few years is typical to simulate natural disturbance (Auestad & Rydgren, 2026; Huhta et al., 2001; Quinlan et al., 2003). In urban areas, meadow establishment typically involves replacing turfgrass with meadow. It is important to understand the shift in ecosystem services that comes with turfgrass-to-meadow restoration to assess the efficacy of meadow as a NbS to the triple planetary crisis. The biodiversity benefits of turfgrass-to-meadow restoration are clear (Norton et al., 2019), but more work is needed to quantify the climate change mitigation potential of urban meadow establishment.

Turfgrass has been shown to sequester carbon, though results depend on local climate, turfgrass species and maintenance regimes (Selhorst & Lal, 2013; R. Wang et al., 2022). Additional emissions due to mowing and fertilizer application can, in some cases, completely negate the net carbon uptake of the turfgrass system itself (Selhorst & Lal, 2013). In general, ecosystems with more diverse plant communities show higher productivity and carbon storage (Chen et al., 2020; Fornara & Tilman, 2008; Y. Yang et al., 2019). We would expect meadow to be an improvement over turfgrass because meadows are more heterogeneous in terms of plant functional types, containing a diversity of forbs and grasses.

Taller and thicker vegetation in meadows should result in cooler soils due to shading (Byrne et al., 2008). These site conditions could result in slower rates of decomposition, increasing SOM accumulation (Kirschbaum, 1995; Pouyat & Carreiro, 2003; D. Wang et al., 2016). Additionally, site-specific microbial communities will influence decomposition rates.

There is mixed evidence that local microbial communities may specialize in decomposing local litter, resulting in a "home field advantage" where litter is more easily decomposed by its "home" community (Ayres et al., 2006; Gholz et al., 2000; Strickland et al., 2009). It is unclear whether this effect would be detectable when comparing turfgrass to meadow, but there is evidence that this effect may depend on litter quality (Pugnaire et al., 2023). Litter quality will also influence decomposition rates regardless of site conditions. Turfgrass litter is homogenous and known to be labile and relatively quick to decompose (Hendrickson et al., 2001; Printz & Hendrickson, 2015; Shi et al., 2006). Meadow litter, on the other hand, is more heterogenous and expected to contain more recalcitrant components. Litter mixtures have been shown to have non-additive effects on litter decomposition rates (Gartner & Cardon, 2004), but more recalcitrant litter has been shown to decompose more slowly in a mixture (L. Wang et al., 2022). Thus, meadow litter is generally expected to decompose more slowly than turfgrass litter, making it more likely to contribute to lasting carbon stocks.

Turfgrass-to-meadow restoration is also expected to alter CO_2 fluxes. For one, meadows are expected to produce more biomass both above and belowground due to increased species diversity (Cardinale et al., 2011; Onandia et al., 2019) which should correspond with increased GPP. Additionally, slower rates of decomposition, as we have already discussed, should reduce ER (Kirschbaum, 1995; D. Wang et al., 2016). Higher GPP and lower ER should result in more negative NEE, signifying increased carbon sequestration. Few studies have directly compared CO_2 fluxes in urban turfgrass and meadow ecosystems. Two studies that have made similar comparisons found lawns to be greater carbon sinks than meadows/prairies (Thienelt & Anderson, 2021; Trémeau et al., 2024). However, these results would have been influenced by

local climate, species composition and management practices such as irrigation and/or fertilizer application (Thienelt & Anderson, 2021; Trémeau et al., 2024).

Aerated terrestrial soils in ecosystems such as turfgrass and meadow are generally methane sinks due to the presence of methanotrophic bacteria (Le Mer & Roger, 2001). Thus, there is generally no reason to expect a difference in methane flux between turfgrass and meadow, unless nitrogen fertilizer application is part of the ecosystem's management regime (Mosier et al., 1991). There is some evidence that usually well-drained prairie soils may become methane sources if flooded, such as after spring melt (F. L. Wang & Bettany, 1995), but again this effect is not expected to differ between turfgrass and meadow.

Since 2013, TRCA has been restoring a 16 km hydro corridor in Scarborough, Ontario, replacing turfgrass with meadow in seven sections over time with the most recent sections completed in 2024 (Toronto and Region Conservation Authority, 2023). Dubbed "The Meadoway", this restored linear UGS improves habitat quality and connectivity in the city, connecting the Rouge National Urban Park with several other UGS in the city including ravines and city parks (Toronto and Region Conservation Authority, 2023). The TRCA adjusted restoration methods and seed mixes over time as they tested different options, but restoration was always very active and intentional with turfgrass being removed completely before seeding native species. Ongoing maintenance involves mowing sites every 3-4 years to mimic natural disturbances and controlling invasive species as needed through herbicide application and hand-pulling (Toronto and Region Conservation Authority, 2025). Patches of turfgrass still exist in The Meadoway as one full field as well as borders along public paths and roadsides. Turfgrass maintenance is minimal, being mowed ~6 times per season and nothing else.

The Meadoway is among the largest linear UGS in Canada. As it is a unique project, TRCA has created “The Meadoway Manual”, a “how-to” guide on urban turfgrass-to-meadow restoration to share their knowledge and lessons learned from the project for application across North America and beyond in similar contexts (Toronto and Region Conservation Authority, 2025). To deepen that knowledge, TRCA has partnered with universities in Ontario to help quantify the benefits of turfgrass-to-meadow restoration using The Meadoway as a study system. Our project is part of that partnership, looking to quantify the effect of turfgrass-to-meadow restoration on carbon sequestration.

We hypothesize that turfgrass-to-meadow restoration will enhance carbon sequestration. We predict that compared to turfgrass, restored meadow will show lower (more negative) NEE, signifying greater carbon sequestration. We predict increased GPP and lower ER in meadow compared to turfgrass, corresponding with increased plant production and decreased decomposition rates. We predict no significant difference in methane flux between restored meadow and turfgrass. We also hypothesize that restored meadow will show slower decomposition rates than turfgrass due to differences in site conditions and/or the production of more chemically diverse and recalcitrant litter.

To test these predictions, we used a space-for-time substitution to make comparisons between three restoration age categories in The Meadoway: unrestored turfgrass, recently restored meadow (restored in 2020-2023), and older restored meadow (restored in 2013-2016). We include two meadow age categories in order to assess any age-related effects of restoration, keeping in mind that older restored meadow is likely more representative of the long-term condition for restored urban meadow.

2.2 Methods

2.2.1 Site Selection

To evaluate the effect of meadow restoration on the rate of carbon sequestration, we carried out a manual static chamber-based gas flux study and a reciprocal plant litter transplant experiment in The Meadoway. We selected a total of 24 sites spanning the length of The Meadoway for the gas flux study split evenly between three categories representing different restoration timelines: “old” (restored in 2013-2016; n = 8), “recent” (restored in 2020-2023; n=8), and “unrestored” (turfgrass; n=8) (Figure 2.1, Table 2.1). We used a subset of these for the litter transplant experiment (Figure 2.1, Table 2.1; n = 3 for each category).

Table 2.1 List of sites and their year of restoration. A third column indicates the subset of sites used in a litter transplant experiment. Sites starting with “t” are unrestored turfgrass sites, sites starting with “rm” are recent meadow sites and sites starting with “om” are old meadow sites.

Site Name	Year Restored	Litter Bags?
t1		
t2		X
t3		
t4		
t6		X
t7		
t8		
tb		X
rm1	2020	
rm2	2020	X
rm3	2020	
rm4	2020	
rm5	2023	X
rm6	2023	
rm7	2022	
rm8	2023	X
om1	2015	
om2	2013	
om3	2013	
om4	2013	
om5	2014	X

Site Name	Year Restored	Litter Bags?
om6	2014	X
om7	2015	X
om8	2016	

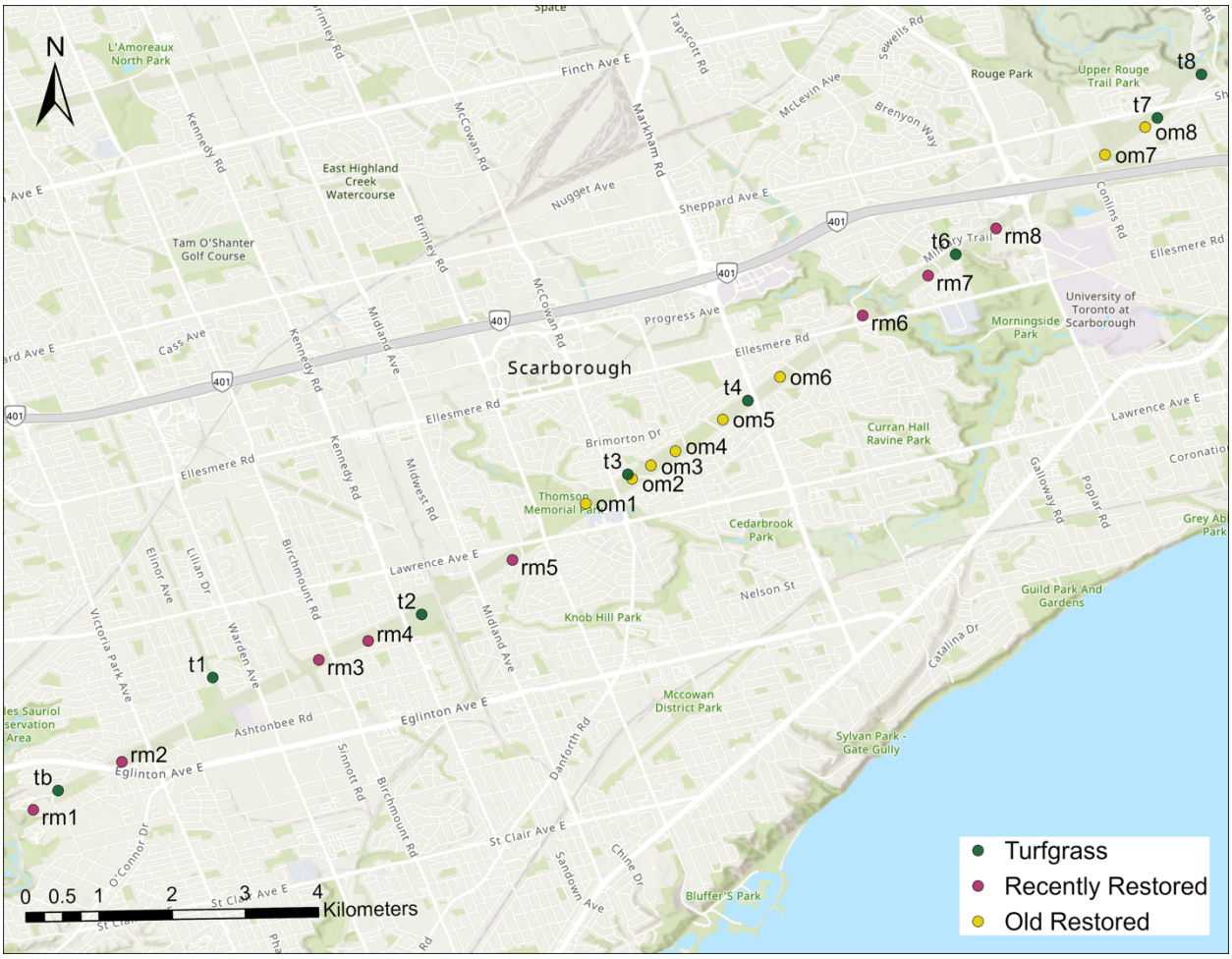


Figure 2.1 Map of The Meadoway with all sites labeled. Sites starting with "t" are unrestored turfgrass sites (n = 8), sites starting with "rm" are recent meadow sites (n = 8) and sites starting with "om" are old meadow sites (n = 8).

2.2.2 Gas Flux

In late April and early May 2025, we installed one 20 cm deep and 30 cm in diameter PVC collar at each of the 24 sites (Figure 2.1). We buried them around sprouting vegetation, minimizing disturbance as much as possible. Starting the week of June 9th, we returned biweekly

to each site with custom built clear acrylic chambers and a $CH_4/CO_2/H_20$ Trace Gas Analyzer (LI-COR LI-7810). These chambers were in 45 or 35 cm tall sections that stacked onto the collars that we sealed with a bead of water. A 10 cm tall top piece included a fan (Wathai 12V 92×25 mm Brushless Cooling Fan, 1800 RPM) to ensure adequate mixing within the chamber, as well as ports to connect the gas analyzer to the chamber stack with PVC tubing. This design allowed the chambers to vary in height depending on the height of the vegetation growing within the collars.

At each site, each time we sampled, we ran four 2-minute flux measurements: two in the light and two in the dark, using a shade cloth to block the sunlight (85-90% light blocking). An adjustment period of ~2 min was afforded between the application of the shade cloth and start of the dark flux measurements. Light fluxes measure NEE and dark fluxes measure ER (Bansal et al., 2023). Methane (CH_4) flux was measured at the same time. We recorded the start time for each flux, the air temperature before and after (Extech SD700), and took a measurement of photosynthetically active radiation (PAR) before and after (LI-COR LI-1500). We took the PAR measurement for the dark fluxes with the sensor beneath the shade cloth. At each site, we measured the height of the chamber at three points around the chamber since it varied depending on the number of chamber pieces used, the height of the collars above the ground, and how level the collar sat. We also recorded the barometric pressure (Extech SD700), as well as soil temperature, soil water content, and soil pore water conductivity (Delta-T Devices HH2 Moisture Meter & WET-2 Sensor).

Air temperature and barometric pressure are essential for flux calculations. Occasional missing air temperature and barometric pressure values were imputed using regression relationships we built using field measurements and alternate data sources (Appendix A).

Additionally, the vegetation inside the collars at some sites did not recover from the disturbance of collar installation, resulting in the final unbalanced site distribution of eight unrestored turfgrass, seven recent meadow and five old meadow sites. We excluded data from an additional old meadow site from August 19th onward due to an unplanned disturbance at the site. Other occasional fluxes were missing or excluded due to equipment malfunctions or the site being temporarily inaccessible.

To ready the flux data for analysis, we visualized each flux by graphing the concentration of gaseous CO_2 and CH_4 over a 90 second time interval starting at the start time we recorded in the field, making any adjustments needed to that start time to isolate the clearest flux possible. Then, following Bansal et al. (2023), we converted the concentrations of both CO_2 and CH_4 to moles using the ideal gas law for that 90 second flux interval. Following this, we fit a line to the concentration of each gas over time to get the rate of change over the flux interval (in mol s^{-1}). Then we divided the rate of change by the surface area of the collars to get the final flux value for each gas (in mol m^{-2} s^{-1}). This process was repeated for each of the 878 fluxes included in our analysis.

Once each light (NEE) and dark (ER) flux was determined, we calculated GPP using the relationship

Equation 2.1 $NEE = ER - GPP$

where NEE is net ecosystem exchange, GPP is gross primary productivity and ER is ecosystem respiration. For each visit to each site, we calculated GPP based on the first light and the first dark flux, then again for the second light and second dark flux. Since ER happens all day long, but GPP only happens in the light, we scaled ER and GPP to their daily totals (in mol m^{-2} day^{-1}).

We scaled ER to 24 hours and GPP to the number of daylight hours on the day of the flux, taken from National Research Council Canada's sunrise/sunset calculator set to Toronto (National Research Council Canada, n.d.). We used the daily GPP and ER flux values to back calculate an estimate of daily NEE (Equation 2.1). A sample flux calculation can be found in Appendix B. We then averaged the duplicate fluxes to yield a mean daily NEE, GPP and ER per visit to each site to use in further calculations and analysis.

We calculated cumulative NEE, GPP and ER for each site by interpolating linearly between the biweekly visits and integrating to get cumulative flux totals (in mol m^{-2} $season^{-1}$). We did this integration over the same period for every site. Since we were missing data from one old meadow site after its unplanned disturbance on August 19th, this site was excluded from the determination of cumulative totals.

2.2.3 Plant Litter Transplant Experiment

To understand rates of decay in The Meadoway and to partition the effect of litter chemical composition from the effect of vegetation on the environmental conditions that influence decay rates, we conducted a reciprocal litter transplant experiment. We made litterbags out of 2 mm fibreglass mesh and filled them with 10.00 g (Sartorius QUINTIX2102-1S Toploading Balance) of litter from biomass we collected between August 27th and September 12th, 2024. The 10 g of litter in each bag comprised about 7 g of stems and 3 g of leaf litter, representing the mass ratio of vegetation tissues found in the contents of the 2024 samples (Appendix C). We deployed the litterbags on April 16th, 2025 in sets of three replicates each filled with litter from one of the three restoration age categories: newly restored, old restored,

and unrestored turfgrass. We deployed 11 of these sets of three litter bags at three representative sites from each category (Table 2.1). To ensure good soil contact, we pinned litterbags to the ground using garden staples. We deployed three sets of three litter bags alongside the other litter bags, one at a site from each restoration category, and recollected them the same day to serve as travel blanks to account for travel loss in analysis.

Over the course of the following year, we returned to these sites to collect sets of three litter bags, one of each type at each site. We collected litter bags on day 0 (travel blanks), 14, 29, 61, 91, 120, 153, 182, 216, 292 (2 sites inaccessible), and 366. On the last collection date, we collected all remaining litter bags from the sites (n = 2 or 3). After each collection date, we dried the contents of each litter bag at 105ºC for 24 hours and then weighed to the nearest 0.01 g (Sartorius QUINTIX2102-1S Toploading Balance). This repeated collection over time allowed us to construct decay curves to visualize and analyze decomposition over time by site type and by litter type.

2.2.4 Statistical Analysis

We analyzed the gas flux data both by the cumulative totals and using the scaled daily rates throughout the season including some environmental predictors in the model. To analyze the cumulative totals, we used ANOVAs to assess differences between the three restoration categories in NEE, GPP and ER. We used Tukey's Honestly Significant Difference (HSD) tests to further investigate differences which were statistically significant. To analyze the scaled daily flux rates throughout the season, we fit linear mixed effect models for each of daily GPP, ER and NEE to assess differences between the three restoration categories. Alongside restoration

category, we included PAR (for GPP & NEE only), soil temperature, soil water content, as well as the interaction between restoration category and soil water content as predictors in the models. We included the category × moisture interaction term to investigate the possibility that meadow is more drought resistant than turfgrass. We included site as a random effect to account for both between-site variation and the correlation of repeated observations within each site throughout the sampling season. A linear model was fit for methane flux, including restoration category, soil temperature, soil water content and the interaction between restoration category and soil water content as predictors since a linear mixed effects model for methane flux produced a degenerate random effect.

To analyze plant litter decay rates, we calculated the proportion of litter mass remaining for each litter bag collected as

Equation 2.2 $$proportion\ remaining = \frac{measured\ mass\ (g)}{initial\ mass\ (g) - travel\ loss\ for\ litter\ type\ (g)}$$

where measured mass (g) is the dry mass of litter measured after collection, initial mass (g) is the mass of litter initially placed in the litter bag (always 10.00 g), and average travel loss (g) being the mean mass loss of 3 travel blanks of that litter type deployed and recollected on day 0.

Similar to Yuckin et al. (2023), we fit the proportion of mass remaining over time for each site × litter type combination using two models: a single exponential model (Olson, 1963) and a two-parameter decaying coefficient model (Ágoston-Szabó & Dinka, 2008; Godshalk & Wetzel, 1978). In this chapter, we report the results of the single exponential decay model coefficient analysis. Models for the two-parameter decaying coefficient model can be found in Appendix D. We tested the effects of site restoration category, litter restoration category, and their interaction in a general linear model with the fitted k parameter as the response variable. We also fit a similar model with proportion mass lost as the response variable to test for an effect of

the same predictors on cumulative mass lost in addition to their effect on the rate of mass loss. We used variance partitioning to determine the relative influence of each site category, litter category and their interaction on the response variable.

We used R v4.5.3 (R Core Team, 2026) in RStudio v2026.04.0+526 (Posit team, 2026) for all statistical analysis. We used trapz() from the package "pracma" for the integration required for the cumulative flux analysis (Borchers, 2025). We implemented general linear models using lm() in the stats package and linear mixed effects models using the packages "lme4," "lmerTest" and "MuMIn" (Bartoń, 2026; Bates et al., 2026; Kuznetsova et al., 2026). We assessed the assumptions of normality and homogeneity of variation of the residuals visually, examining Q-Q plots and plots of residuals versus fitted values (Appendices E & F). We used nlsLM() from the "minpack.lm" package to fit the single-exponential model using the Levenberg-Marquardt nonlinear least-squares algorithm (Elzhov et al., 2023). We used the package "dplyr" (Wickham, François, et al., 2026) for data manipulation, as well as "ggplot2" (Wickham, Chang, et al., 2026) and "patchwork" (Pedersen, 2025) for data visualization. We used the package "multcompView" to visualize Tukey's HSD results (Graves et al., 2026).

2.3 Results

2.3.1 Gas Flux

Figure 2.2 shows cumulative NEE, GPP and ER between June 10th and November 20th, 2025. We found moderate evidence that cumulative NEE differed by restoration category (ANOVA, $F_{2,16} = 6.02$, $p = 0.011$) with unrestored turfgrass acting as a net seasonal CO_2 source ($+14.1 \pm 17.0$ mol m^{-2} season^{-1}) and both recent (-12.5 ± 19.1 mol m^{-2} season^{-1}) and old meadow (-12.3 ± 6.44 mol m^{-2} season^{-1}) acting as sinks, representing a difference of approximately 26.5

mol m^{-2} $season^{-1}$ relative to unrestored sites, on average (Figure 2.2A). We did not find evidence that cumulative GPP differed among the categories (ANOVA, $F_{2,16} = 0.42$, $p = 0.666$; Figure 2.2B), with mean seasonal totals of 93.1 ± 32.2, 98.4 ± 26.1, and 82.8 ± 13.4 mol m^{-2} $season^{-1}$ for unrestored, recent, and old meadow respectively. We found only weak evidence of a trend in cumulative ER (ANOVA, $F_{2,16} = 2.78$, $p = 0.092$), where old meadow shows lower cumulative respiration than unrestored turfgrass (70.5 ± 7.0 vs. 107.2 ± 36.1 mol m^{-2} $season^{-1}$, a difference of 36.6 mol m^{-2} $season^{-1}$, on average).

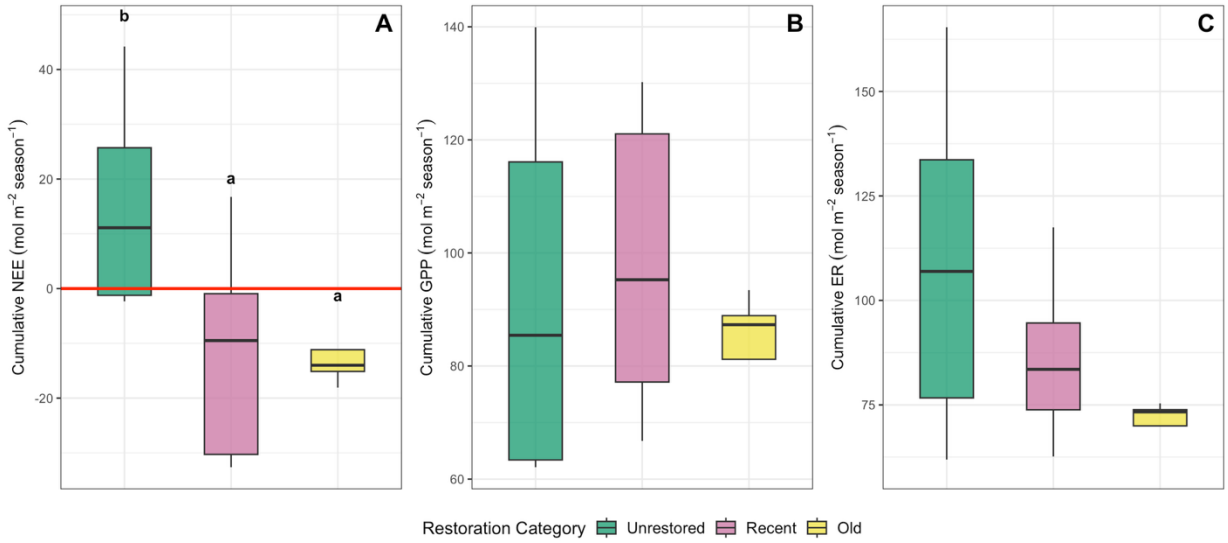


Figure 2.2 Cumulative NEE (A), GPP (B) and ER (C) from June 10th to November 20th, 2025 by restoration category: unrestored turfgrass (n = 8), recently restored meadow (n = 7) and old restored meadow (n = 4). Boxplots show medians with 25% quartiles and whiskers display extremes within 1.5 times the spread beyond the quartile. Lowercase letters show results of Tukey's HSD post-hoc tests for ANOVA results with $p < 0.05$. Categories with different letters are significantly different.

Figure 2.3 shows the biweekly variation in NEE, GPP and ER. Figure 2.3A shows that turfgrass acted as a carbon source between the weeks of June 23rd and September 15th. This coincides with a period of relatively low soil water content overall (Figure 2.4). Old and recent

meadow were CO_2 sources overall the weeks of August 4th and August 18th, though they were more often CO_2 sinks. Figure 2.3C shows that turfgrass was often respiring more than the meadow categories, though not always. This pattern is especially clear in the fall months. Looking at Figure 2.3B, turfgrass also had higher GPP in the fall months. Turfgrass GPP was lower in the dry summer months, corresponding generally with the higher (more positive) NEE in turfgrass during those months already mentioned. There were also differences in soil water content between the categories overall with recent meadow generally showing higher soil water content compared to old meadow throughout the season (Figure 2.4).

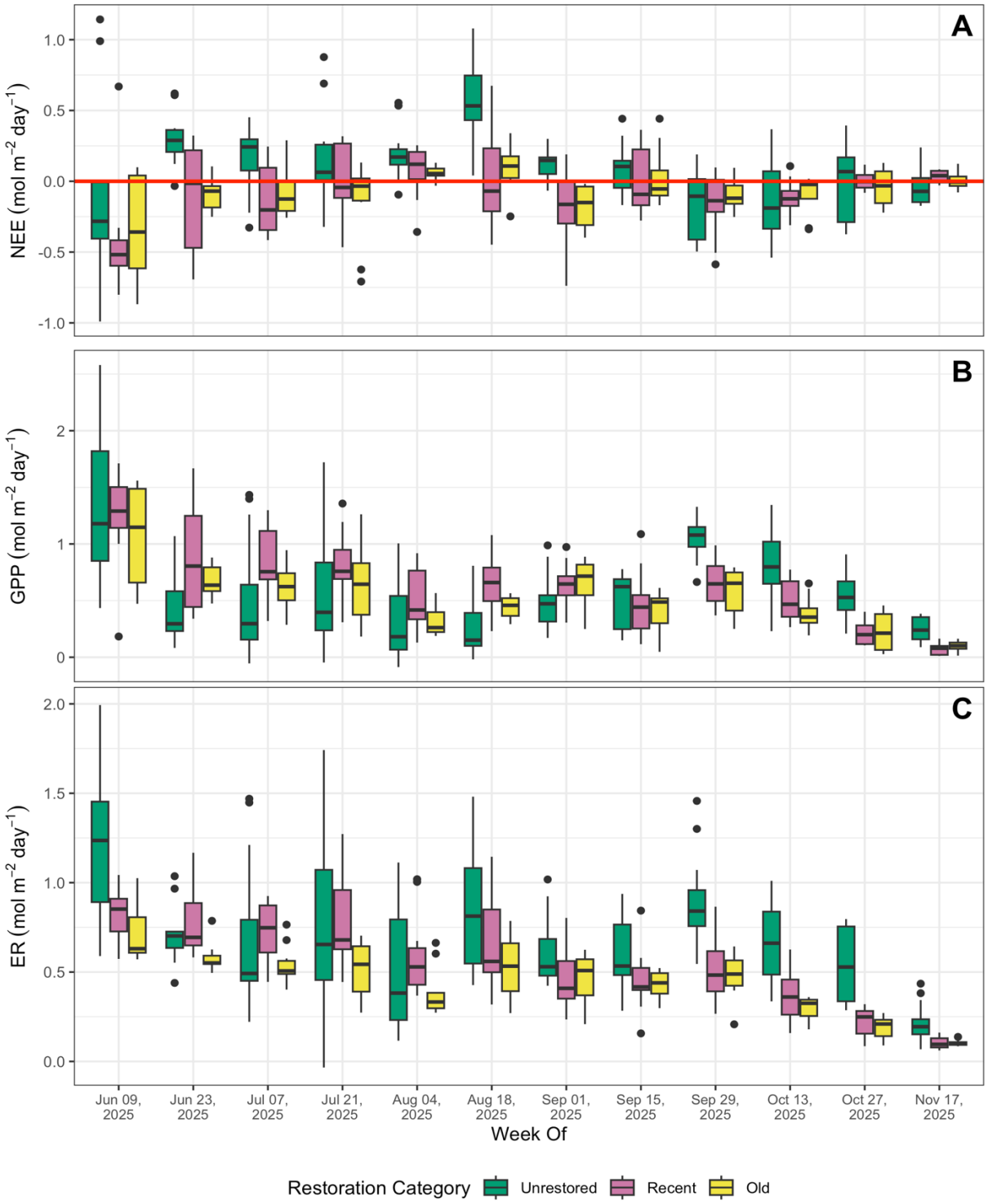


Figure 2.3 Scaled Daily NEE (A), GPP (B) and ER (C) by sampling week in 2025 by restoration category: unrestored turfgrass (n = 8), recently restored meadow (n = 7) and old restored meadow (n = 5, n = 4 after August 19th). The red line in panel A is at NEE = 0. Values above this line represent net CO_2 efflux and values below this line represent net CO_2 influx. Positive GPP represents CO_2 uptake and positive ER represents CO_2 efflux. Boxplots show medians with 25% quartiles and whiskers display extremes within 1.5 times the spread beyond the quartile.

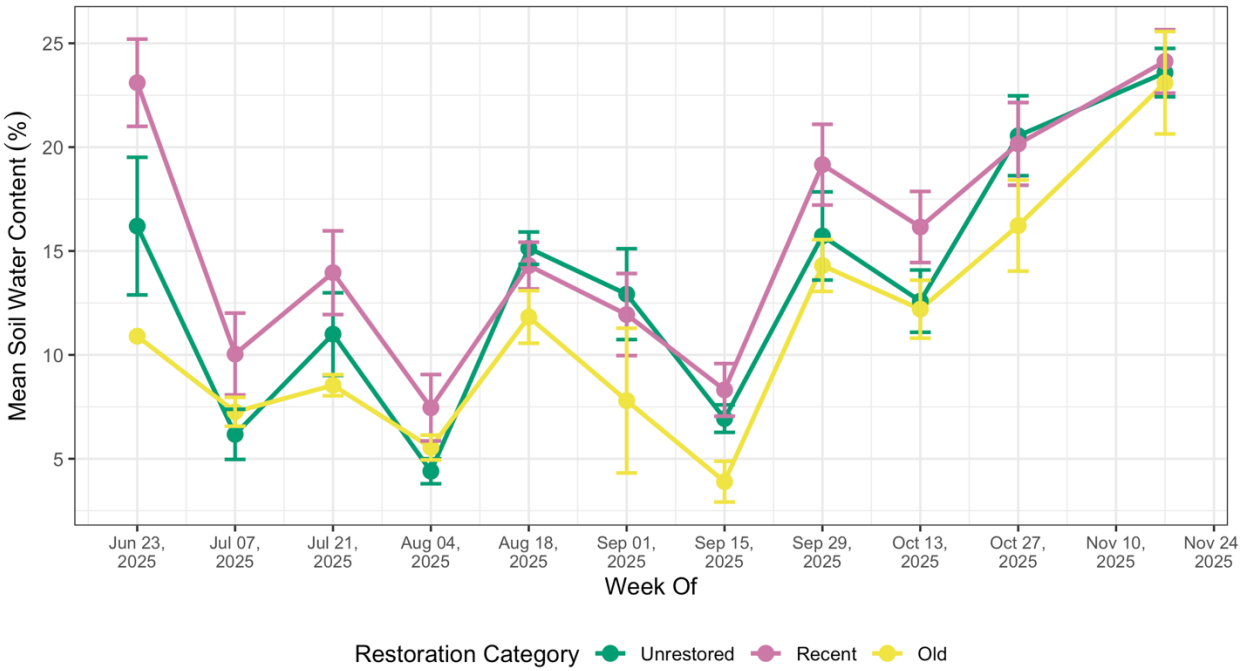


Figure 2.4 Mean soil water content (%) by sampling week by restoration category: unrestored turfgrass (n = 8), recently restored meadow (n = 7) and old restored meadow (n = 5, n = 4 after August 19th).

We found moderate evidence that scaled daily NEE was influenced by restoration category with old meadow showing lower NEE compared to turfgrass (Table 2.2). However, recent meadow did not differ from turfgrass (Table 2.2). We found strong evidence that soil temperature influenced daily NEE and weak evidence that PAR also had an effect (Table 2.2). Soil water content did not influence daily NEE, nor did interactions between restoration category and soil water content (Table 2.2).

Table 2.2 Summary of seasonal greenhouse gas flux models. NEE, GPP and ER were fit as linear mixed-effects models with site included as a random effect. Degrees of freedom and p-values use Satterthwaite's approximation. CH_4 was fit as a linear model since the mixed model produced a degenerate site random effect with 0% variance. The fixed factor Category compares to unrestored turfgrass. Marginal R^2 is the variance explained by the fixed effects while conditional R^2 is the variance explained by the fixed and random effects. For CH_4, just the standard R^2 is shown. Site random effect is not applicable to CH_4 model.

Response Variable	Fixed Effect	df	t-value	p-value	Site Random Effect (% Variance)	R^2 (marginal / conditional)
NEE	Category (Recent Meadow)	81.9	-1.62	0.109	9.8%	0.166 / 0.248
	Category (Old Meadow)	82.6	-2.20	0.030		
	PAR	174.6	-1.85	0.066		
	Soil Temperature	184.0	3.96	<0.001		
	Soil Water Content	184.8	0.64	0.520		
	Category (Recent Meadow) × Soil Water Content	183.1	0.12	0.904		
	Category (Old Meadow) × Soil Water Content	180.7	1.25	0.213		
GPP	Category (Recent Meadow)	75.6	2.11	0.039	15.3%	0.129 / 0.263
	Category (Old Meadow)	70.4	1.33	0.187		
	PAR	173.9	2.69	0.008		
	Soil Temperature	182.2	1.61	0.109		
	Soil Water Content	183.2	2.66	0.008		
	Category (Recent Meadow) × Soil Water Content	185.0	-2.31	0.022		
	Category (Old Meadow) × Soil Water Content	178.7	-2.15	0.033		
ER	Category (Recent Meadow)	42.9	0.85	0.400	39.7%	0.355 / 0.611
	Category (Old Meadow)	37.1	-0.34	0.739		
	Soil Temperature	177.4	9.82	<0.001		
	Soil Water Content	177.5	4.96	<0.001		
	Category (Recent Meadow) × Soil Water Content	180.8	-2.91	0.004		
	Category (Old Meadow) × Soil Water Content	175.0	-1.76	0.080		
CH_4	Category (Recent Meadow)	187	0.46	0.643		0.084
	Category (Old Meadow)		-0.36	0.722		
	Soil Temperature		2.37	0.019		
	Soil Water Content		1.98	0.049		
	Category (Recent Meadow) × Soil Water Content		0.39	0.695		
	Category (Old Meadow) × Soil Water Content		0.25	0.803		

We also found moderate evidence that recent meadow had higher scaled daily GPP compared to unrestored turfgrass while old meadow did not differ from turfgrass (Table 2.2). We found strong evidence that both PAR and soil water content positively influenced GPP, though the effect of soil water content was moderated by restoration category (Table 2.2). Compared to turfgrass, there was strong evidence of an interaction between restoration category and soil water content for both old and recent meadow (Table 2.2). Figure 2.5B shows that GPP decreases with increasing soil water content in the meadow categories but increases in the turfgrass.

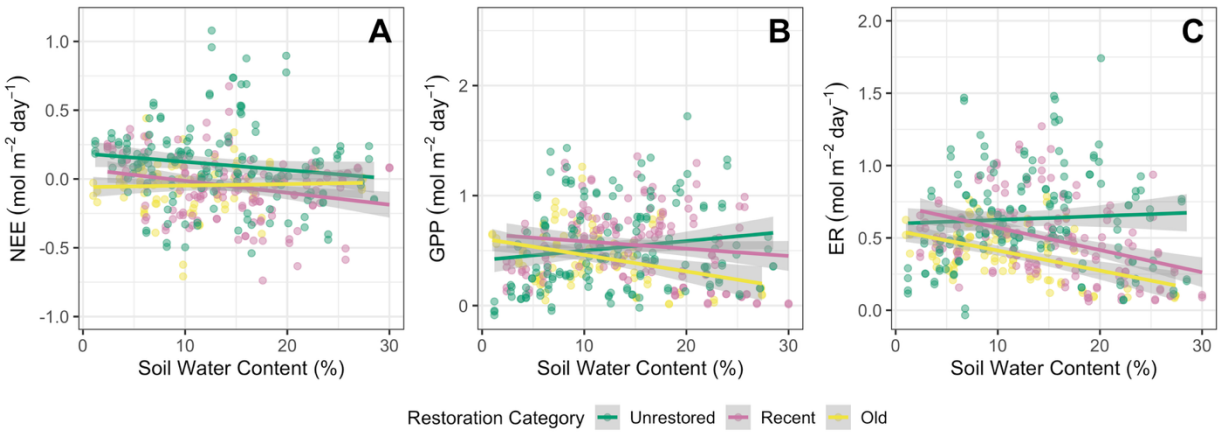


Figure 2.5 Relationship between soil water content and each of NEE (A), GPP (B) and ER (C) by restoration category (unrestored turfgrass, recent meadow, old meadow)

We found no evidence that scaled daily ER was influenced by restoration category when analyzed alongside environmental predictors (Table 2.2). There was strong evidence that both soil temperature and soil water content were positive predictors of ER, though the effect of soil water content was moderated by restoration category (Table 2.2). Compared to turfgrass, there was strong evidence of an interaction between restoration category and soil water content for both recent meadow and weak evidence for old meadow (Table 2.2). Figure 2.5C shows that like

with GPP, ER decreases with increasing soil water content in the meadow categories but increases in the turfgrass.

Overall, all restoration categories were methane sinks throughout the sampling season (Figure 2.6). Neither recent nor old meadow differed from turfgrass overall (Table 2.2). There was moderate evidence that both soil temperature and soil water content predicted daily methane flux (Table 2.2). There was no evidence of any interaction between restoration category and soil water content (Table 2.2).

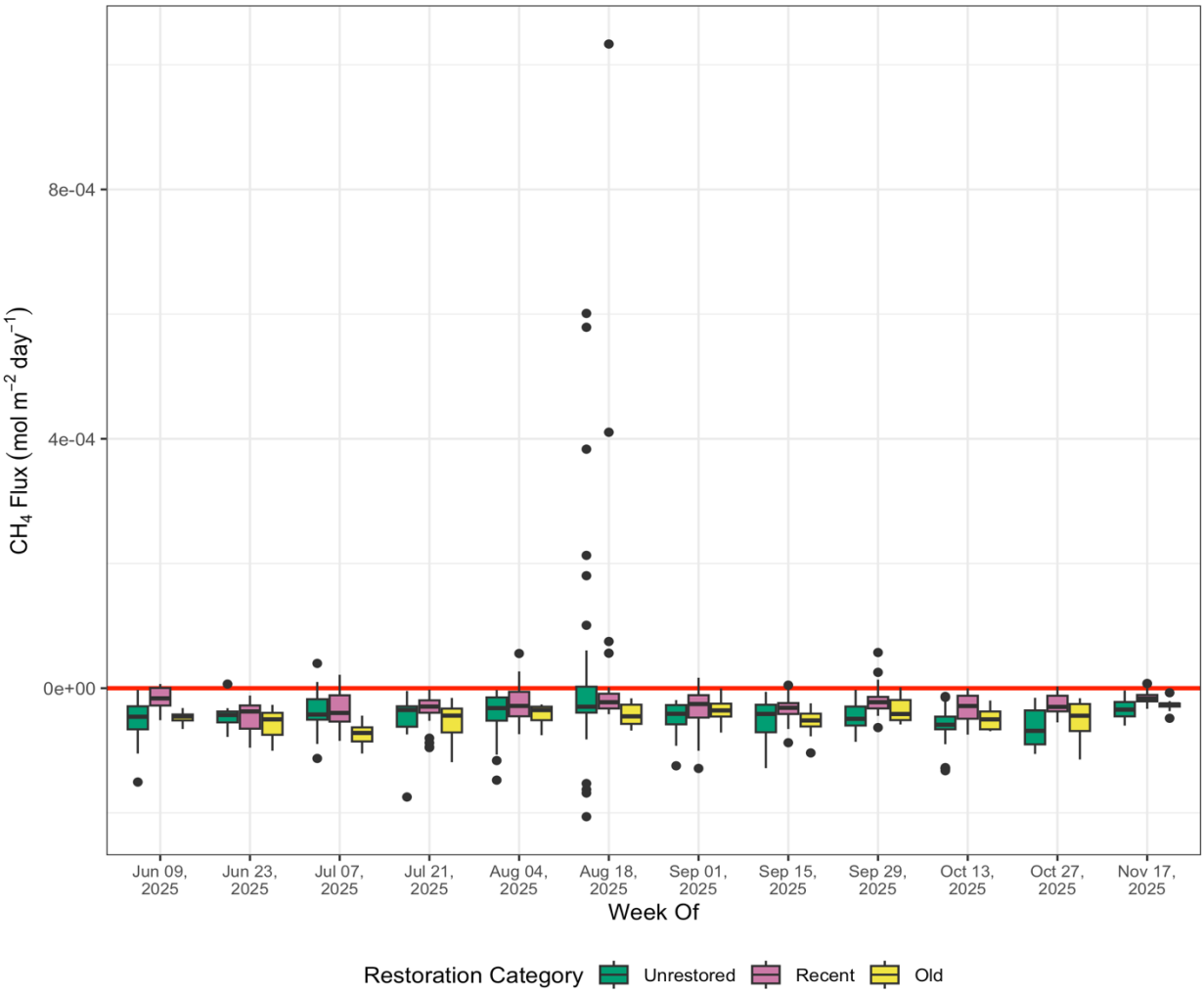


Figure 2.6 Scaled daily methane flux by sampling week in 2025 by restoration category. Positive values represent methane efflux and negative values represent methane influx. Boxplots show medians with 25% quartiles and whiskers display extremes within 1.5 times the spread beyond the quartile.

2.3.2 Plant Litter Transplant Experiment

Figure 2.7 shows that the decomposition rate (k) for turfgrass litter was considerably higher than the two meadow litter types (A), but litter decomposition rates were higher at recently restored meadow sites compared to turfgrass sites (B). Corroborating this pattern, we found strong evidence that plant litter decomposition rates differed by litter category (ANOVA, $F_{2,18} = 14.23$, $p < 0.001$) and moderate evidence that they differed by site category (ANOVA, $F_{2,18} = 3.83$, $p = 0.041$). Overall, the model explained ~68% of the variance in the single exponential parameter for decay rate (k), of which ~51% can be attributed to litter category and ~14% can be attributed to site category. There was moderate evidence that decomposition rates were lower for litter from older restored meadow compared to turfgrass ($t_{18} = -2.18$, $p = 0.043$). There was weak evidence that decomposition rates were lower for litter from recently restored meadow ($t_{18} = -1.78$, $p = 0.093$). Regardless of litter type, we found weak evidence that decay rates were higher at recently restored meadow sites ($t_{18} = 1.92$, $p = 0.071$). There was no evidence of an interaction between litter category and site category on the litter decomposition rate (k) (ANOVA, $F_{4,18} = 0.44$, $p = 0.781$).

We also found strong evidence that the proportion of litter mass lost by the end of the 366-day experiment differed overall both by litter category (ANOVA, $F_{2,50} = 7.43$, $p = 0.001$) and site category (ANOVA, $F_{2,50} = 11.35$, $p < 0.001$). We found strong evidence that litter decomposed more at recent meadow sites compared to turfgrass ($t_{50} = 2.78$, $p = 0.008$) and weak evidence that litter decomposed more at old meadow sites compared to turfgrass ($t_{50} = 1.81$, $p = 0.077$). Despite the evidence of an overall effect, there was no evidence that recent or old meadow litter decomposed less than turfgrass litter ($t_{50} = -0.86$, $p = 0.395$ & $t_{50} = -1.24$, $p = 0.221$). We found no evidence of an overall interaction between litter category and site category

(ANOVA, $F_{4,50} = 0.89$, $p = 0.476$), but there was weak evidence that old litter decomposed less than turfgrass litter when placed at old sites ($t_{50} = -1.82$, $p = 0.075$). Overall, the model explained ~45% of the variance in the proportion mass lost, of which ~16% can be attributed to litter category and ~23% can be attributed to site category. Figure 2.8 shows that site conditions (soil temperature and water content) were similar at all three restoration site categories throughout the main growing season.

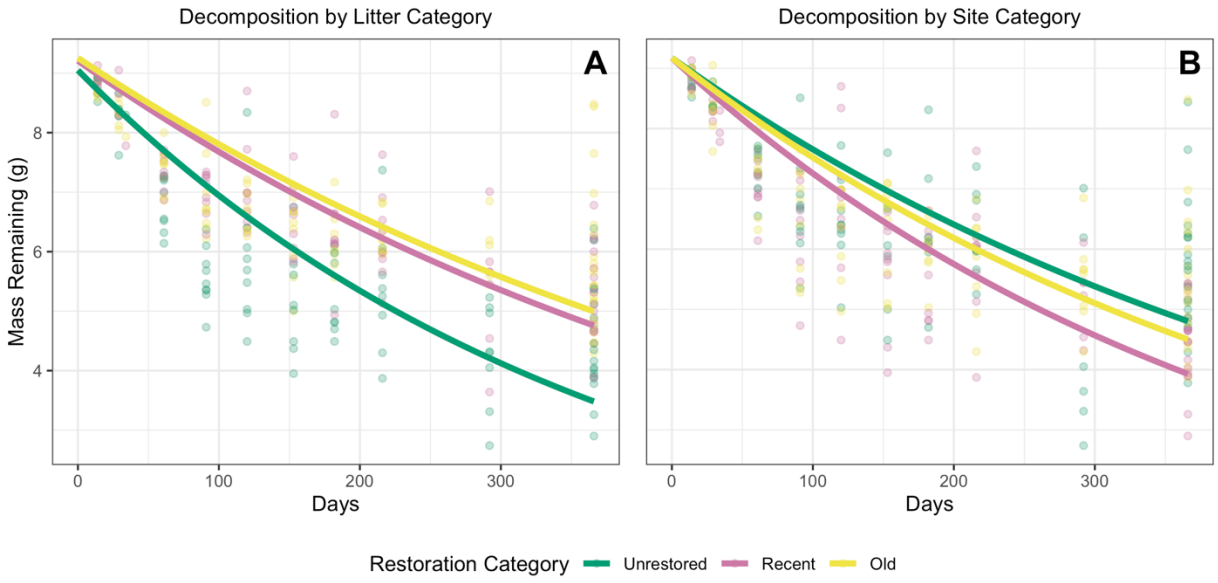


Figure 2.7 Mass remaining over time throughout year-long litter transplant experiment by litter category (A) and site category (B) with fitted single exponential model for each category.

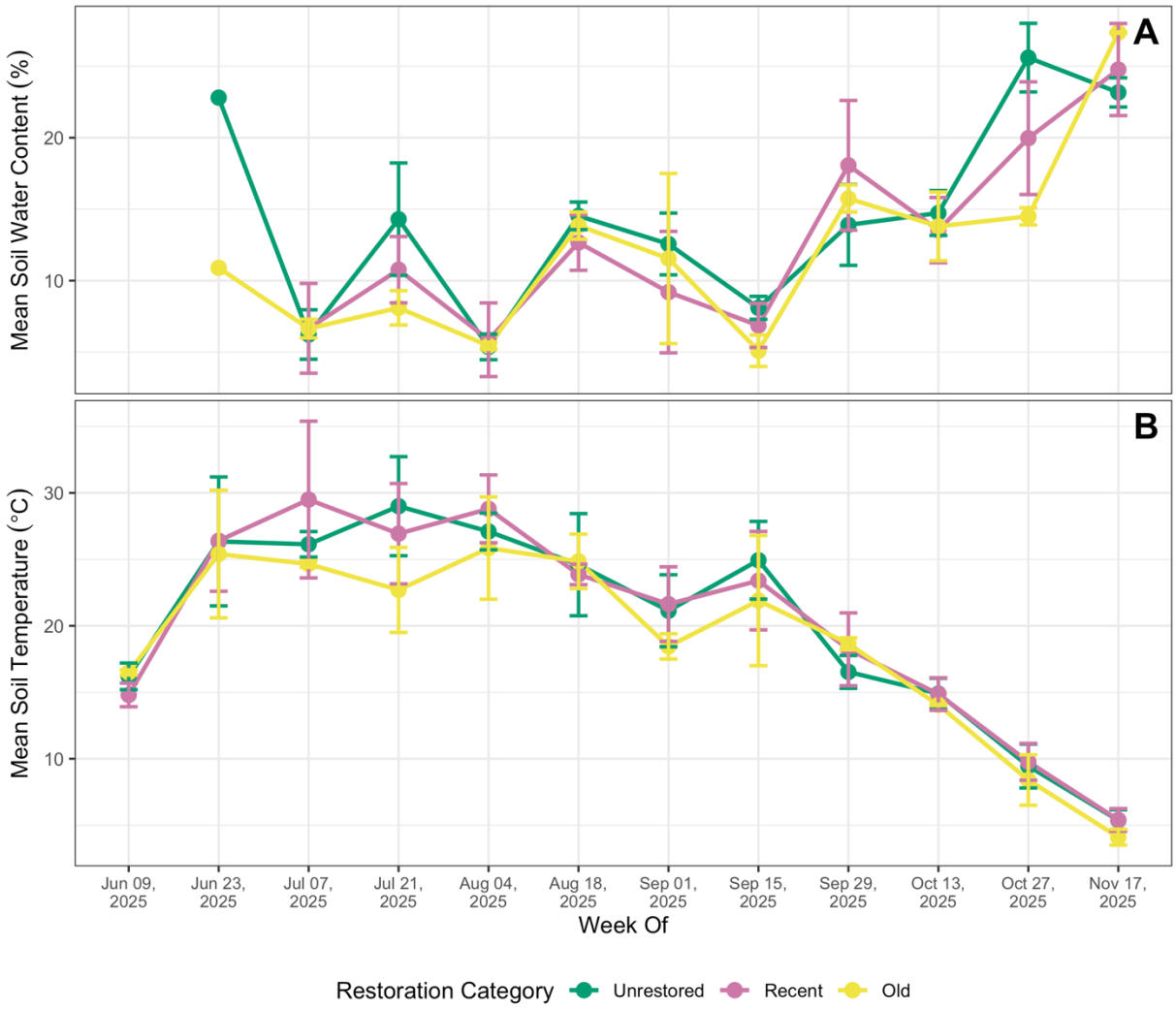


Figure 2.8 Soil Water Content (%; A) and Soil Temperature (ºC; B) over the course of the season for the subset of sites used in the litter bag transplant experiment displayed by restoration category. Soil Water Content data is missing for the week of June 9th and incomplete for the week of June 23rd due to sampling challenges.

2.4 Discussion

In this chapter, we hypothesized that turfgrass-to-meadow restoration would enhance carbon sequestration. We predicted that meadow would have more net CO_2 influx, and we found that both restored meadow categories had more negative cumulative NEE compared to turfgrass

which had positive cumulative NEE overall. NEE was significantly more negative only for old meadow compared to turfgrass when analyzed alongside environmental predictors. There was no difference in cumulative GPP between the restoration categories, but there was weak evidence that old meadow had less cumulative ER compared to turfgrass, suggesting that lower ER may explain the difference in NEE. Considered alongside environmental predictors, we found higher GPP in recent meadow compared to turfgrass but no difference in ER between the categories. We predicted that methane flux would not differ between restoration age categories, and we found that neither old nor recent meadow differed from turfgrass. We did not see any major seasonal variation in methane flux over our sampling period, which we attribute to low soil moisture content during that time. We hypothesized that plant litter would decompose more slowly in the meadow than turfgrass, either because of differences in the quality of the litter generated by the plant communities growing in these different site categories, due to differences in site conditions (lower temperature, higher moisture, different microbial community), or because of a combination of litter quality and site condition differences. We found that turfgrass litter did decompose more quickly than meadow litter, but all litter types decomposed more in the recently restored meadow site. Below, we expand on these findings in detail.

2.4.1 Gas Flux

The cumulative seasonal NEE estimates show clearly that both recent and old restored meadow were carbon sinks while turfgrass was a carbon source over the growing season in 2025 (Figure 2.2). Of course, the cumulative fluxes are simply estimates based on linear interpolation between daily flux values at each site, which themselves are estimates of daily GPP, ER and NEE that were scaled assuming that the ER we measured mid-day is consistent throughout the 24 h

day, whereas true ER rates vary throughout the day (Gilmanov et al., 2005). Our goal was to make comparisons between meadow and turfgrass, not to necessarily determine the true daily flux values. While the values themselves may be rough estimates, the pattern is clear.

Contrary to our predictions, we did not find differences in cumulative GPP between the categories. However, Figure 2.2B shows that the range of cumulative GPP reaches the lowest in unrestored turfgrass. There was only weak evidence that cumulative ER differed between the categories, but Figure 2.2C shows that cumulative ER trends lower with increasing meadow age, which aligns with our predictions. These results suggest that the lower cumulative NEE in restored meadow is driven more by lower ER rather than higher GPP.

Seasonally, we see that turfgrass was a carbon source between the weeks of June 23rd and September 15th (Figure 2.3A). Old and recent meadow were also carbon sources occasionally during this period, but far less consistently than turfgrass (Figure 2.3A). We see that the consistently positive NEE in turfgrass is due to low GPP, as ER stayed largely consistent with the meadow categories during this period (Figure 2.3). Into the fall (starting the week of September 29th), turfgrass had higher GPP and similar ER, resulting in NEE similar to the two meadow categories at this time. Thus, it seems that differences in total NEE are largely driven by the summer period when turfgrass was not very productive relative to the meadow. This pattern of low productivity in the summer months was likely caused by the dry conditions experienced in summer 2025, classified in the Canadian Drought Monitor as abnormally dry (1 in 3 years) to moderate drought (1 in 5 years) (Agriculture and Agri-Food Canada, National Agroclimate Information Service, 2025). Although not extreme, this likely created moisture stress, especially for the turfgrass which appeared dormant for much of the summer. Under higher moisture availability, turfgrass may have had higher GPP and lower NEE overall. Future sampling in

wetter summers would test this hypothesis. However, as climate change intensifies and extreme weather like heat waves and drought become more common (Gampe et al., 2021; Seneviratne et al., 2014), our results demonstrate that meadow is more climate-resilient than turfgrass in urban areas.

We analyzed the seasonal flux data alongside environmental predictors to better isolate the effect of restoration age category. We know that cumulatively, NEE was similarly negative in both old and recent meadow. However, when analyzed alongside environmental predictors, we found that just old meadow had lower NEE compared to turfgrass (Table 2.2). Alongside the effect of category, we found both PAR and soil temperature to be predictors of NEE (Table 2.2). Similarly, despite finding no difference in cumulative GPP, we found that recent meadow had higher GPP compared to turfgrass with PAR and soil water content also predicting GPP (Table 2.2). We found weak evidence that cumulative ER differed among the categories, but alongside environmental predictors, ER in both old and recent meadow were not different from turfgrass.

Due to a lack of baseline data, it is not possible to say whether differences in environmental conditions (soil temperature and water content) are due to preexisting spatial differences throughout the 16 km hydro corridor or due to the restoration itself. For example, Figure 2.4 shows that soil water content differed between the categories with recent meadow having higher soil water content compared to old meadow throughout the season. These differences could be simply because more recently restored sites happened to be moister to begin with, due to topography or other spatial differences in soil quality and surface runoff. Alternatively, the difference in soil water content could be the result of the restoration, for example, if shading from taller vegetation reduces evaporation from the soil or if the plant community uses less water at the surface in recently restored meadow.

Another complicating factor is differences in restoration methods used by TRCA for restoring the older meadow compared to the more recent meadow. The site preparation methods, seed mixes, seeding methods, ongoing maintenance regimes all differ between sites as TRCA fine-tune methods and respond to site-specific challenges such as invasive species. One difference is the use of tillage for site preparation. Restoration for recent meadow sites typically involved multiple rounds of tillage, which is a common restoration method used to deplete non-native propagules from the seedbank (Kotsopoulos et al., 2024; Wolf & Young, 2016; Xie & MacIvor, 2025). This method was not used for older restored sites. A consequence of tillage is that SOM is mixed and stable aggregates are broken up and exposed, which results in increased SOM decomposition (Bilen et al., 2020; Gencer et al., 2024; Linsler et al., 2013). While we did not find that recent meadow had statistically different ER compared to turfgrass or old meadow, it does appear that cumulative ER in recent meadow was slightly elevated compared to old meadow (Figure 2.2). It is possible that this is due to residual effects of tillage in the recent meadow, though it is not possible to separate this effect from the effect of more time passing to reach an equilibrium in the old meadow. A future study could look at the effects of several rounds of tillage on soil carbon stocks in an ecosystem restoration context to help managers weigh the benefits of tillage for invasive species management against the potential loss of soil carbon stocks. In Chapter 3, we measure SOM stocks to further investigate this idea.

We also found evidence that both GPP and ER were predicted by interactions between soil water content and restoration category. Figure 2.5 shows that GPP and ER decrease in meadow categories but increase in turfgrass as soil water content increases. We interpret this to mean that turfgrass was water limited between the weeks of July 7 and September 15, when soil moisture averaged only 9.5±5.5%. Turfgrass was able to become more productive in the late

summer and early fall as soil moisture became more available, averaging 18.0±6.2% during the weeks of September 29 to November 17 (Figure 2.4). As we have discussed, meadow plants maintained higher GPP through this dry period, likely reflecting their greater drought tolerance compared to turfgrass species. Rather than consider the negative association between soil moisture and GPP in the restored meadow sites as growth suppression due to soil saturation, we interpret this trend as spurious, reflecting the coincident timing of fall senescence and increasing moisture availability in the fall. We note that the soil was not saturated: soil moisture in the meadow sites reached a maximum of 30.0% at site rm2 on November 18, 2025.

Trémeau et al. (2024) compared CO_2 flux in lawns and meadows in Helsinki, Finland. They report, on average, mean daily GPP rates throughout their sampling season from May to August of irrigated lawns (0.555 mol CO_2 m^{-2} day^{-1}), non-irrigated lawns (0.380 mol CO_2 m^{-2} day^{-1}), mesic meadow (0.543 mol CO_2 m^{-2} day^{-1}) and dry meadow (0.198 mol CO_2 m^{-2} day^{-1}). We might expect our unrestored turfgrass and non-irrigated restored meadow GPP values to resemble those of Trémeau et al. (2024)'s non-irrigated lawn and dry meadows, yet GPP in our old and newly restored meadow averaged 0.512 and 0.621 mol CO_2 m^{-2} day^{-1}, whereas in turfgrass GPP averaged 0.581 mol CO_2 m^{-2} day^{-1}, more resembling their wetter conditions. Based on modelling these ecosystems from 2005-2021, Trémeau et al. (2024) conclude that the lawns had more negative NEE compared to the meadows, which is the opposite of what we found. However, many differences between our study systems may account for this discrepancy such as differences in climate, restoration and management methods, species composition, and sampling effort. Kucharik et al. (2006) report peak ER in a 65-year-old restored prairie in Southern Wisconsin of 1.1±0.3 mol CO_2 m^{-2} day^{-1} and 0.9±0.1 mol CO_2 m^{-2} day^{-1} in an adjacent natural prairie remnant (both scaled assuming consistent daily ER as we do), which are consistent with

our higher ER values despite differences in restoration age (Figure 2.3C). These studies show that our results align with expected flux rates for turfgrass and meadow-like ecosystems, despite their conclusions which may differ from our own.

On our last sampling week in late November, NEE, GPP and ER were all approaching zero reflecting that plants had mostly senesced and providing confidence that we did not miss major carbon uptake (Figure 2.3). Snowpack prevented using our static chamber method to continue measuring carbon fluxes through the winter. Urban lawns and meadows have been shown to be slight carbon sources during the winter months (Jasek-Kamińska et al., 2020), and future work over winter should compare ER between turfgrass and restored meadow.

As we predicted, methane flux did not vary greatly throughout the sampling season (Figure 2.6). However, our sampling did not start until June 9th – well after the spring melt when methane emissions are most likely (F. L. Wang & Bettany, 1995). Sampling earlier in the season would have allowed us to capture methane flux during spring melt. It is possible that some sites could have been methane sources during this period, though this would more likely be related to topography, redox potential and other site-specific factors than to age since restoration (F. L. Wang & Bettany, 1995). Soil water content was not high enough at any point in our sampling season to expect significant methane emissions from any of the vegetation types we sampled in The Meadoway.

2.4.2 Plant Litter Transplant Experiment

The results of our plant litter transplant experiment reveal that the effect of litter quality and site conditions on decomposition rate and litter mass lost were inverse. We found lower

decomposition rates for litter from older meadow compared to turfgrass, regardless of whether litter bags were deployed in meadow or turfgrass sections of the Meadoway. However, we found weak evidence that all litter types decomposed faster when deployed in recently restored meadow than turfgrass and strong evidence that more litter mass was lost in recently restored meadow than turfgrass. Recall that recent meadow sites had higher soil water content overall over the course of our gas flux season, between the weeks of July 7th to November 27th, 2025 (Figure 2.4). One thought may be that the higher soil water content we found in recent meadow creates more optimal conditions for decomposers, accelerating decomposition rates. However, the subset of recent meadow sites we used for the litter transplant experiment were, on average, drier than the recently restored meadow sites overall, showing similar soil water content in all three site categories (Figure 2.8). Thus, higher soil water content cannot explain the greater litter decomposition in recent meadow sites.

As soil temperature was also similar in all three categories in our subset of sites used for the litter transplant experiment (Figure 2.8), it seems most likely that the higher mass lost over the course of the year at recently restored meadow sites is due to differences in the microbial community which is able to decompose any litter more quickly at recent sites. Recall that recent meadow sites were tilled repeatedly during restoration. It is possible that this disturbance temporarily shifted the microbial communities at recent sites. For example, tillage has been shown to shift the microbial community in favour of a higher bacteria to fungi ratio (Simmons & Coleman, 2008). Interestingly, fungi have typically been considered to be more important for lignin decomposition, though there is some evidence that bacteria can also break down lignin (Bugg et al., 2011). Thus, it is possible that tillage reduced decomposition rates for recalcitrant litter, but increased decay rates for the more labile fraction of the litter in the recent meadow.

Future work should focus on characterizing the composition and activity of soil decomposer communities to further understand these site dynamics.

Variance partitioning shows that litter quality (explaining ~51% of the variance) matters more than site conditions (explaining ~14 % of the variance) for the decay rate analysis, suggesting the greater decomposability of turfgrass litter may matter more than the potentially higher decay rate in recently restored meadow. However, for litter mass lost during the one-year litterbag deployment, the importance of litter quality and site conditions was more equivalent: site conditions explained ~23% of the variance and litter quality explained ~16% of the variance. Given this study occurred during a mild to moderate drought, we infer that a different microbial community in the recent meadow enabled greater litter mass lost during a period of drought-related stress. The litterbag transplant experiment was valuable in allowing us to tease apart the opposing effects of litter type and site conditions on the decomposition of litter in The Meadoway. Future research could expand on this by analyzing the chemical composition of the litter, in addition to characterizing the soil decomposer communities.

The importance of litter quality for decay rates suggests that the meadow litter contains more recalcitrant components than turfgrass litter. This makes sense as turfgrass litter is known to be labile and relatively quick to decompose (Hendrickson et al., 2001; Printz & Hendrickson, 2015; Shi et al., 2006). Chemical composition affects decomposition in complex ways, often with multiple compounds interacting. Ratios of lignin:cellulose or lignin:nitrogen have been shown to predict litter decomposition rates (Aerts, 1997). Talbot & Treseder (2012) showed that lignin reduced litter decomposition in part by decreasing the loss of cellulose, but cellulose content may increase lignin decomposition. Further, nitrogen was shown to increase decomposition of litter partially by increasing the loss of lignin (Talbot & Treseder, 2012).

Nitrogen could come from legumes or, in the case of some more managed turfgrass ecosystems, from artificial fertilizer. These interactions are complex and difficult to predict in turfgrass and meadow systems such as The Meadoway. In addition, different microbial communities will interact with these compounds differently. Future research could expand on our study by analyzing the chemical composition of the litter, as well as by characterizing the composition and activity of soil microbial communities.

There was no evidence of an interaction between litter category and site category, suggesting no major home-field advantage. The weak evidence we found that old meadow litter decomposed less than turfgrass litter when placed at old sites is actual contrary to the home-field advantage hypothesis. This could mean that the microbial communities did not differ between turfgrass and meadow, though we did not study the microbial community directly. However, the home-field advantage hypothesis is not consistently supported in the literature (Ayres et al., 2006; Gholz et al., 2000; Strickland et al., 2009). A study by Pugnaire et al. (2023) did show that home-field advantage effects were complex and moderated by both site conditions and litter composition. Specifically, they found home-field advantages for litter types which were more recalcitrant, but not for more labile litter which can be readily broken down by most microbial communities (Pugnaire et al., 2023). As already mentioned, turfgrass is known to be labile and quick to decompose (Hendrickson et al., 2001; Printz & Hendrickson, 2015; Shi et al., 2006), which fits with our results. Meadow litter is more chemically diverse and can be expected to have more recalcitrant components relative to turfgrass, but as we did not analyze the chemical makeup of the litter, we could not predict its relative recalcitrance or lability. Thus, it is possible that herbaceous meadow litter is still relatively labile overall and, as a result, potential effects of a home-field advantage were not detectable in our experiment. A separate study focusing on the

lignin, cellulose, and nitrogen content of the litter produced in turfgrass compared to newly and older restored meadow could reveal the influence of turfgrass-to-meadow restoration on the litter chemistry-decay relationship in the context of the home-field advantage hypothesis.

2.5 Conclusions and Next Steps

Overall, between the gas flux and litter decomposition results, we have evidence that turfgrass-to-meadow restoration has resulted in enhanced carbon sequestration. NEE was negative in both old (-12.3 ± 6.44 mol m^{-2} $season^{-1}$) and recent meadow (-12.5 ± 19.1 mol m^{-2} $season^{-1}$) compared to turfgrass which was positive ($+14.1 \pm 17.0$ mol m^{-2} $season^{-1}$), and there was evidence that litter decomposition rates were 33.6% lower for restored meadow litter compared to turfgrass. We did also have evidence that litter decomposed 34.7% more at recently restored meadow sites. However, variance partitioning suggests that litter quality is the more important factor, at least for the decay rate analysis (explaining ~51% of the variance, compared to just ~14% for site category). Although we have no reason to believe the recently restored meadow sites differed meaningfully from the sites selected for restoration earlier, we do not have baseline flux data or pre-restoration litter decomposition rates to rule out antecedent differences as factors that could explain the greater litter mass loss observed in recently restored meadow. Still, our findings here can inform future work on urban ecosystem restoration. Future work should focus on quantifying the chemical composition of litter and the composition and activity of soil decomposers to further tease these effects apart. Additionally, future studies could directly investigate the effect of tillage for site preparation on carbon stocks in an ecosystem restoration context, as well as the home-field advantage hypothesis in the context of turfgrass-to-meadow

restoration. Finally, in chapter 3, we will measure carbon stocks to further investigate differences between site categories.

3.0 Plant Production and Belowground Carbon Stocks in The Meadoway

3.1 Introduction

In light of the ongoing "triple planetary crisis," UGS are increasingly being leveraged as NbS with ecological, social and economic benefits (Miao & Nduneseokwu, 2024; UNFCCC, 2022). UGS are variable in their quality, with different levels of ecological integrity, functionality and service provision (Niemelä et al., 2010). UGS maintained as turfgrass is ubiquitous in many parts of the urban landscape (Ignatieva et al., 2015), yet mowed turfgrass is generally considered to be lower quality UGS compared to alternatives such as native plant meadows which increase ecosystem service provision (Bretzel et al., 2024; Marshall et al., 2023).

Meadows are semi-natural grasslands which prominently feature native forbs and are mown infrequently (Kotsopoulos et al., 2024; Marshall et al., 2023; Norton et al., 2019). In urban areas, meadows typically replace turfgrass ecosystems, providing ecosystem services such as relatively high biodiversity and habitat quality, as well as urban heat and flood mitigation (Meenar et al., 2022; Norton et al., 2019; Paudel & States, 2023; Zölch et al., 2019). There is also the matter of carbon sequestration, which remains poorly understood in the context of turfgrass-to-meadow restoration. Turfgrass has been shown to sequester carbon, though results are varied and depend on climate, species composition and maintenance regimes (Selhorst & Lal, 2013; R. Wang et al., 2022). Additional emissions due to mowing and fertilizer application can, in some cases, completely negate the net carbon uptake of the turfgrass system itself (Selhorst & Lal, 2013). Generally, ecosystems with more diverse plant communities show higher productivity and carbon storage (Chen et al., 2020; Fornara & Tilman, 2008; Y. Yang et al., 2019). Meadows, with their variety of native grasses and forbs, are more diverse than turfgrass

and can be expected to sequester more carbon as a result. We have previously demonstrated greater net carbon uptake and slower litter decomposition rates in meadow compared to turfgrass (Chapter 2), but we have not measured soil or biomass carbon stocks.

Our results from Chapter 2 suggest that meadow aboveground biomass will decompose more slowly than turfgrass litter which should result in greater carbon stocks in meadow in the long term. Plant biomass production is important for assessing the carbon sequestration potential of ecosystems both as carbon stores in standing crop biomass and as a rate of net primary production which can contribute to long lasting belowground stocks. There is evidence linking diversity of primary producers to increased standing biomass production (Cardinale et al., 2011; Onandia et al., 2019), which suggests that meadows would produce more biomass than turfgrass. Combined with the fact that meadow ecosystems are mowed less regularly than turfgrass, annual aboveground standing crop biomass is expected to be higher in meadows. Belowground biomass production is also expected to be higher in meadows due to the presence of deeper rooting species (Brown et al., 2010; Trémeau et al., 2024; Weaver, 1968). This, too, should result in greater belowground carbon stocks over time.

To determine the consequences of different rates of carbon sequestration between meadow and turfgrass ecosystems, we must measure how much carbon is stored belowground. Belowground carbon stocks can take the form of soil organic and inorganic carbon (Pouyat et al., 2009; Wilsey et al., 2020), as well as root stocks. Several studies that have compared soil organic carbon (SOC) between turfgrass and meadow ecosystems have found no difference or higher SOC in lawns (Malmsten, 2023; Marshall et al., 2023; Pouyat et al., 2009). However, turfgrass and meadow ecosystems in these studies all vary in species composition, climate, preexisting soil quality, and management regimes which could influence SOC stocks over time. We have

previously found evidence of both slower decomposition rates and lower NEE (signifying net carbon uptake) in meadow compared to turfgrass (Chapter 2). Combined with our prediction that annual biomass production will be greater in meadow compared to turfgrass, there is reason to expect more carbon stock accumulation following turfgrass-to-meadow restoration.

Soil inorganic carbon or carbonate has long been assumed to cycle much more slowly than organic carbon, but more recently it has been shown to be an active pool that can respond to shifts in vegetation in less than ten years (Wilsey et al., 2020). Soil carbonate is expected to be most influenced by parent material, but species composition and land use history can also be important. In urban contexts, urban fill can also be an anthropogenic source of carbonate depending on land use history (J.-L. Yang & Zhang, 2015). In a turfgrass-to-meadow restoration context, it is possible that deeper root systems will result in more carbonate dissolution over time (Wen et al., 2021; Wilsey et al., 2020). However, the relative impact of this potential carbonate dissolution on total carbon stocks remains difficult to predict.

Since 2013, TRCA has been restoring a 16 km hydro corridor in Toronto, Ontario, replacing turfgrass with meadow in seven sections over time with the most recent sections completed in 2024 (Toronto and Region Conservation Authority, 2023). Dubbed “The Meadoway”, this restored linear UGS improves habitat quality and connectivity in the city, connecting the Rouge National Urban Park with several other UGS in the city including ravines and city parks (Toronto and Region Conservation Authority, 2023). TRCA adjusted restoration methods and seed mixes over time as they tested different options, but restoration was always very active and intentional with turfgrass being removed completely before seeding with native seeds. Ongoing maintenance involves mowing sites every 3-4 years to mimic natural disturbances and controlling invasive species as needed through herbicide application and hand-

pulling (Toronto and Region Conservation Authority, 2025). Patches of turfgrass still exist in The Meadoway as one full field as well as borders along public paths and roadsides. Turfgrass maintenance is minimal, being mowed ~6 times per season. No overseeding, irrigation, or fertilization is undertaken, differentiating Meadoway turfgrass from high-maintenance residential lawns, sports fields, or golf courses.

Our goal in this chapter is to quantify carbon stocks and annual biomass production in both meadow and turfgrass in The Meadoway. We hypothesize that in the long term, turfgrass-to-meadow restoration will result in greater SOM stocks due, in part, to increased plant production. Thus, we predict increased plant production both above and belowground in restored meadow compared to turfgrass. We also predict that old meadow sites will have more SOM and root stocks compared to turfgrass, while recent meadow may be similar or lower than turfgrass. Finally, we predict that old meadow will have lower carbonate content compared to turfgrass and recent meadow due to increased carbonate dissolution over time by deeper rooting species.

To test our hypothesis, we use a space-for-time substitution to make comparisons between three restoration age categories in The Meadoway: unrestored turfgrass, recently restored meadow (restored in 2020-2023), and older restored meadow (restored in 2013-2016). We include two meadow age categories to assess any age-related effects of restoration, keeping in mind that older restored meadow is likely more representative of the long-term condition for restored urban meadow especially with regard to soil carbon stocks.

3.2 Methods

3.2.1 Site Selection

Making use of The Meadoway's chronosequence, we selected eight sites in each of three age-related categories: "old" (restored in 2013-2016; n = 8), "recent" (restored in 2020-2023; n=8), and "unrestored" (turfgrass; n=8; Figure 3.1). One unrestored site from 2024 was inaccessible in 2025 due to construction (t5), so we replaced it with a new site (tb). Two sites (om8 and t7) were affected by an unexpected disturbance in August 2025, resulting in them being excluded from aboveground biomass collection in 2025 (Table 3.1). A full list of sites, their year of restoration, and the components of this chapter they were used for can be found in Table 3.1.

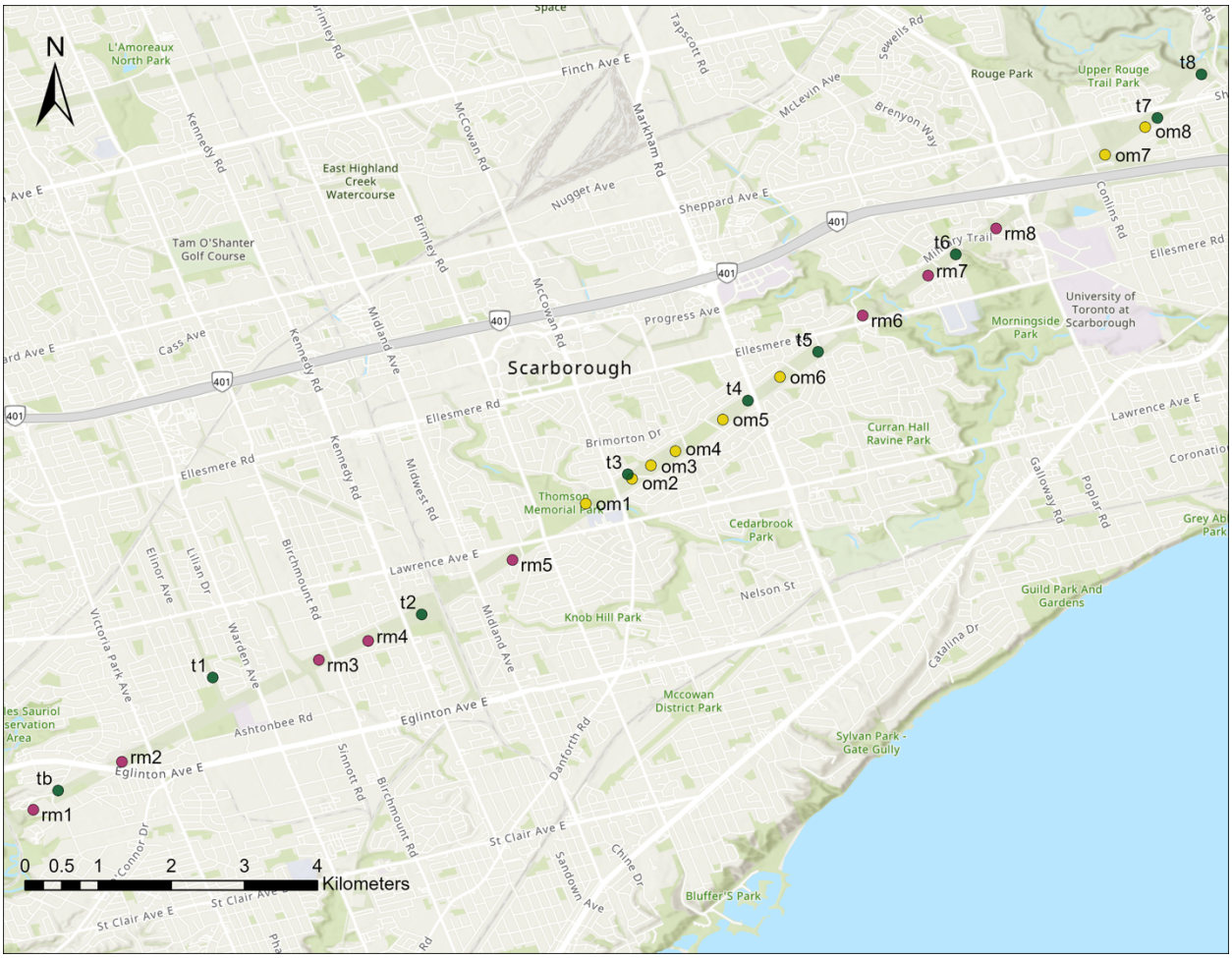


Figure 3.1 Map of The Meadoway with all sites labeled. Sites starting with "t" are unrestored turfgrass sites, sites starting with "rm" are recent meadow sites and sites starting with "om" are old meadow sites.

Table 3.1 List of sites and summary of soil core collection, soil ingrowth core installation and collection, and aboveground biomass collection. Sites starting with "t" are unrestored turfgrass sites, sites starting with "rm" are recent meadow sites and sites starting with "om" are old meadow sites.

Site Name	Year Restored	2024		2025		
		Soil Cores	Aboveground Biomass	Soil Cores	Soil Ingrowth Cores	Aboveground Biomass
t1		X	X	X	X	X
t2		X	X	X	X	X
t3		X	X	X	X	X
t4		X	X	X	X	X
t5		X	X			
t6		X	X	X	X	X
t7		X	X	X	X	
t8		X	X	X	X	X
tb				X	X	X
rm1	2020	X	X	X	X	X
rm2	2020	X	X	X	X	X
rm3	2020	X	X	X	X	X
rm4	2020	X	X	X	X	X
rm5	2023	X	X	X	X	X
rm6	2023	X	X	X	X	X
rm7	2022	X	X	X	X	X
rm8	2023	X	X	X	X	X
om1	2015	X	X	X	X	X
om2	2013	X	X	X	X	X
om3	2013	X	X	X	X	X
om4	2013	X	X	X	X	X
om5	2014	X	X	X	X	X
om6	2014	X	X	X	X	X
om7	2015	X	X	X	X	X
om8	2016	X	X	X	X	

3.2.2 Aboveground Biomass Production

To measure annual rates of aboveground plant production, we harvested five 0.5 m × 0.5 m quadrats of aboveground biomass at each of the meadow and turfgrass sites in both 2024 and 2025, similar to previously used methods (Kucharik et al., 2006; Yuckin et al., 2023). We included only biomass from the active growing season. Collection dates can be found in

Appendix G. These samples were dried at 105ºC for 24 hours and weighed to the nearest 0.01 g (Sartorius QUINTIX2102-1S Toploading Balance).

3.2.3 Belowground Carbon Stocks

To measure carbon stocks belowground, we collected soil cores at each site in both 2024 (5 cm diameter, 10 cm length) and 2025 (5 cm diameter, 30 cm total length in 10 cm increments). Sampling took place in mid-June in 2024 and in mid-May in 2025. Specific collection dates at each site are in Appendix G.

Both years, we froze the cores immediately upon return to campus. We stored and processed the three stacked cores from each site in 2025 separately, allowing for analysis by depth. After thawing each core, we separated roots from soil, picking roots and rhizomes with forceps until no more could be seen. In this chapter, we will refer to the whole belowground biomass fraction (roots and rhizomes) as just "root" for simplicity. We washed root samples, then dried both soil and root samples at 105ºC for 24 hours and weighed to determine dry weights for each. We homogenized samples and took subsamples (1 g for soil and 0.5 g for roots ±0.0100 g; Mettler Toledo Analytical Balance NewClassic MF MS204S). In some instances, there was less than 0.5 g of roots in the total soil core. In these cases, we used the entire available root fraction as the subsample. We used a loss-on-ignition (LOI) method, ashing subsamples in a muffle furnace (Thermo Scientific, Thermolyne F6010) for 4 hours at 550ºC, followed by 1 hour at 950ºC to gravimetrically determine DW_{550} and DW_{950}. We calculated LOI_{550} and LOI_{950} following Heiri et al. (2001) as

Equation 3.1 A $LOI_{550} = \frac{DW_{550} - DW_{105}}{DW_{105}}$ **B** $LOI_{950} = \frac{DW_{550} - DW_{950}}{DW_{105}}$

where LOI_{550} and LOI_{950} represent LOI at 550ºC and 950ºC, DW_{105} is the initial dry weight of the subsample after drying at 105ºC, DW_{550} is the dry weight of the subsample after ashing at 550ºC, and DW_{950} is the dry weight of the subsample after ashing at 950ºC. Using these values and total sample dry weight values (including root dry weight, abbreviated as RDW in this chapter), we calculated values representing soil organic matter (SOM), soil carbonate content (SCC) and root organic matter (ROM) for each site (in g/m^2).

3.2.4 Belowground Biomass Production

To measure annual rates of belowground plant production, we installed soil ingrowth cores in tandem with soil core collection in 2025. We used a modified soil ingrowth method, filling the soil core holes with vermiculite and recollecting them later in the season (Neill, 1992; Yuckin et al., 2023). Vermiculite is used because it is nutritionally neutral and thus should not affect root growth (Yuckin et al., 2023). We installed hardware cloth (Quest Better Barriers Plastic Rigid Utility Mesh, Hardware Cloth, 3 × 15-ft; cut in roughly 8.5 cm × 8.5 cm pieces) over top of each ingrowth core to prevent tampering by animals. We recollected the ingrowth cores in mid-August. Specific recollection dates can be found in Appendix G. We froze the ingrowth cores upon return to campus each day and thawed them in small batches to be processed over time.

We processed the ingrowth cores as one 30 cm deep core for each site, rather than in 10 cm sections like the soil cores because it was not possible to preserve the separation by depth when recollecting the samples. We separated the roots from the vermiculite. Then we washed the roots, dried them at 105ºC for 24 hours, and weighed them (Sartorius QUINTIX2102-1S

Toploading Balance). We also used the same loss-on-ignition method to measure ROM, which we report in g/m^2.

3.2.5 Statistical Analysis

To test the effect of turfgrass-to-meadow restoration on carbon storage, we used ANOVAs to make comparisons between restoration age categories. We tested for differences in aboveground biomass among the restoration categories using the dry weight of live plant material. We tested for differences in belowground carbon stocks among the three restoration categories, considering the top 10 cm cores collected in 2024 and each 10 cm increment of the 30 cm cores from 2025 separately. We tested for differences in SOM, SCC, RDW and ROM for each soil core increment. Additional two-way ANOVAs were used to analyze the 30 cm soil cores from 2025 by depth in each belowground carbon fraction. We tested for differences in root growth rate using the RDW and ROM from the 30 cm soil ingrowth cores collected in 2025. Additionally, we conducted t-tests to compare aboveground biomass production in 2024 to 2025 in each of the restoration categories.

We used R v4.5.3 (R Core Team, 2026) in RStudio v2026.04.0+526 (Posit team, 2026) to implement ANOVAs with aov() in the "stats" package in base R. We assessed the assumptions of normality and homogeneity of variation of the residuals visually, examining Q-Q plots and plots of residuals versus fitted values (Appendices H, I & J). When ANOVA results were significant, we used Tukey's Honestly Significant Difference (HSD) tests to further investigate those differences using TukeyHSD() in the "stats" package in base R. We created all figures using the packages "ggplot2" (Wickham, Chang, et al., 2026) and "patchwork" (Pedersen, 2025). We used the package "multcompView" to visualize Tukey's HSD results (Graves et al., 2026).

3.3 Results

3.3.1 Aboveground Biomass Production

We found strong evidence that restoration category affected aboveground biomass production in 2024 (ANOVA, $F_{2,117} = 35.41$, $p < 0.001$) and 2025 (ANOVA, $F_{2,107} = 44.10$, $p < 0.001$). Both years, there was more aboveground biomass in both meadow categories compared to turfgrass (Figure 3.2). We found strong evidence of a difference between sampling years for each category (Table 3.2). Compared to 2024, in 2025 there was, on average, 34% less aboveground biomass in old meadow, 32% less in recent meadow and 44% less in unrestored turfgrass (Table 3.2).

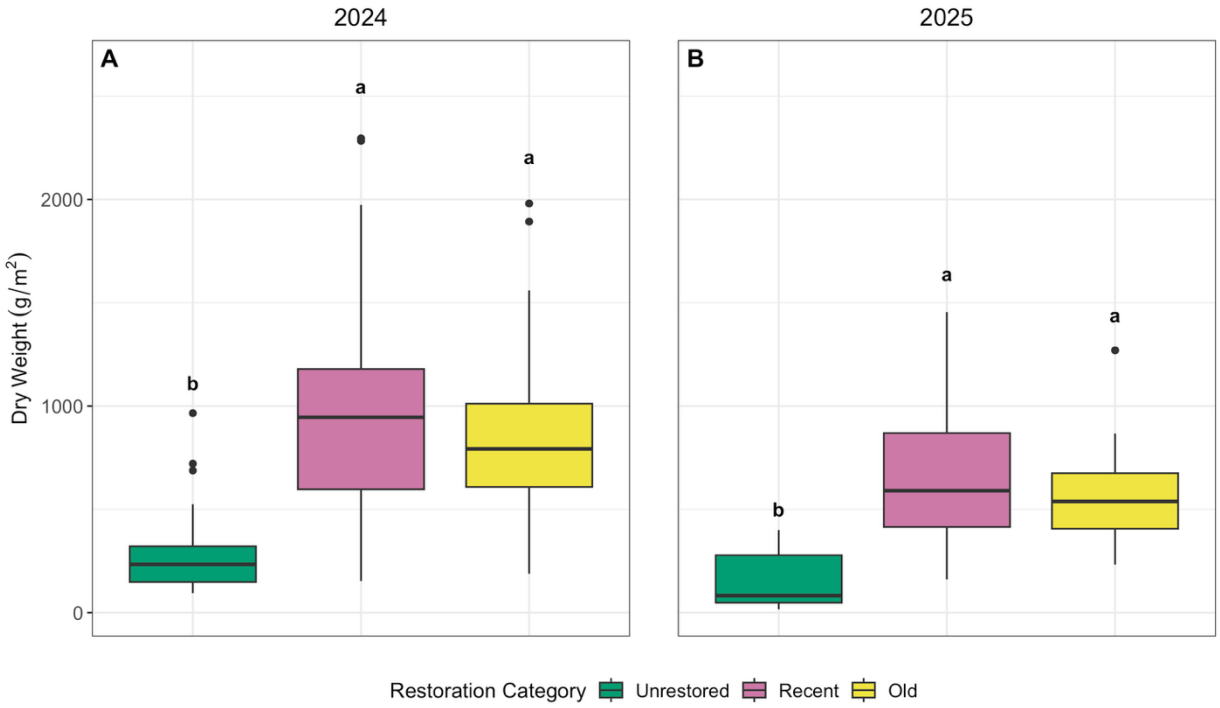


Figure 3.2 Aboveground biomass reported as dry weight (g/m^2) in 2024 (A) and 2025 (B). Boxplots show medians with 25% quartiles and whiskers display extremes within 1.5 times the spread beyond the quartile. Points indicate potential outliers. Lowercase letters show results of Tukey's HSD post-hoc tests for significant ANOVA results. Categories with different letters are significantly different.

Table 3.2 Summary of t-test results comparing aboveground biomass in 2024 and 2025 for each restoration category: unrestored turfgrass, recent meadow and old meadow

Category	2024 Mean ± SD (g/m^2)	2025 Mean ± SD (g/m^2)	Difference	t-value	p-value
Unrestored turfgrass	267.75 ± 181.99	149.28 ± 123.73	118.47	3.33	0.001
Recent meadow	955.42 ± 521.76	647.19 ± 320.15	308.24	3.32	0.001
Old meadow	840.52 ± 331.58	555.61 ± 241.17	284.91	3.79	<0.001

3.3.2 Belowground Carbon Stocks

We found no evidence of any differences among restoration categories in either SOM or SCC in the 2024 soil core data (Table 3.3). However, there was weak evidence of differences in both RDW and ROM among the restoration categories (Table 3.3). Figure 3.3E,G shows that recent meadow had less RDW and ROM compared to both unrestored and old meadow, but the greatest difference in each case was between recent meadow and unrestored turfgrass.

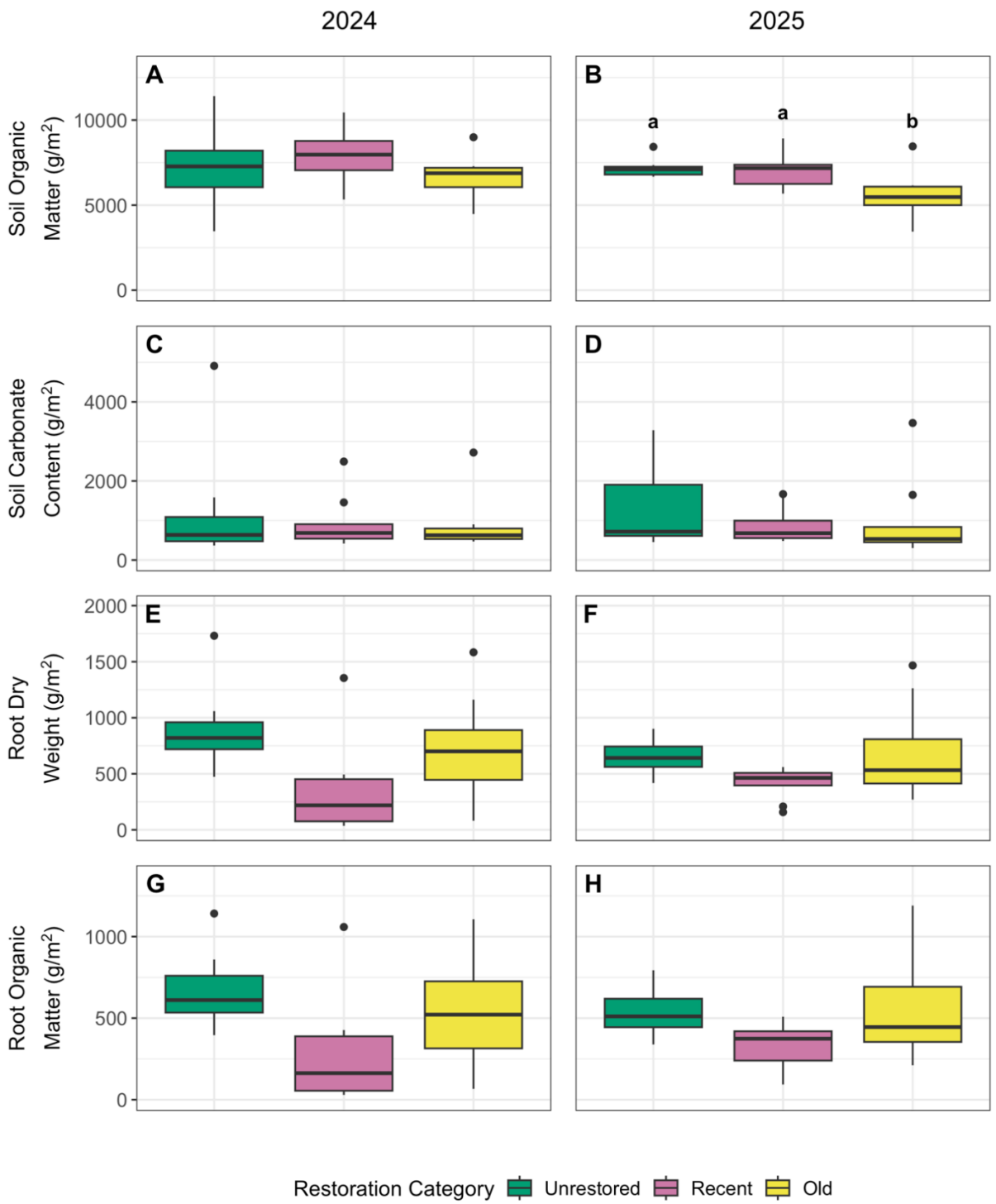


Figure 3.3 Belowground carbon stocks in the top 10 cm from 2024 (left) and 2025 (right) separated into soil organic matter (A-B), soil carbonate content (C-D), root dry weight (E-F) and root organic matter (G-H). "Root" here refers to the entire belowground biomass fraction. Boxplots show medians with 25% quartiles and whiskers display extremes within 1.5 times the spread beyond the quartile. Points indicate potential outliers. Lowercase letters show results of Tukey's HSD post-hoc tests for ANOVA results with $p < 0.05$. Categories with different letters are significantly different.

Table 3.3 Summary of ANOVA results for belowground carbon fractions by restoration category at each soil depth for both 2024 (0-10 cm only) and 2025 (all depths)

Depth (cm)	Carbon Fraction	Degrees of Freedom	F-value	p-value
0-10 (2024)	SOM	2, 21	0.99	0.387
	SCC	2, 21	0.27	0.769
	RDW	2, 21	3.28	0.058
	ROM	2, 21	3.28	0.058
0-10 (2025)	SOM	2, 21	5.10	0.016
	SCC	2, 21	0.54	0.590
	RDW	2, 21	2.10	0.147
	ROM	2, 21	2.48	0.108
10-20 (2025)	SOM	2, 21	3.05	0.069
	SCC	2, 21	1.06	0.364
	RDW	2, 21	2.08	0.149
	ROM	2, 21	1.15	0.337
20-30 (2025)	SOM	2, 21	0.07	0.930
	SCC	2, 21	1.42	0.265
	RDW	2, 21	1.38	0.274
	ROM	2, 21	1.54	0.237

The top 10 cm cores from 2025 showed largely similar patterns to the 2024 cores, as seen in Figure 3.3. However, we found no evidence that RDW or ROM differed among the categories (Table 3.3). In the 2025 soil cores, we found moderate evidence that old meadow had less SOM compared to both unrestored turfgrass and recent meadow in the top 10 cm, weak evidence in the 10-20 cm depth, and no evidence in the 20-30 cm depth fraction (Table 3.3, Figure 3.4).

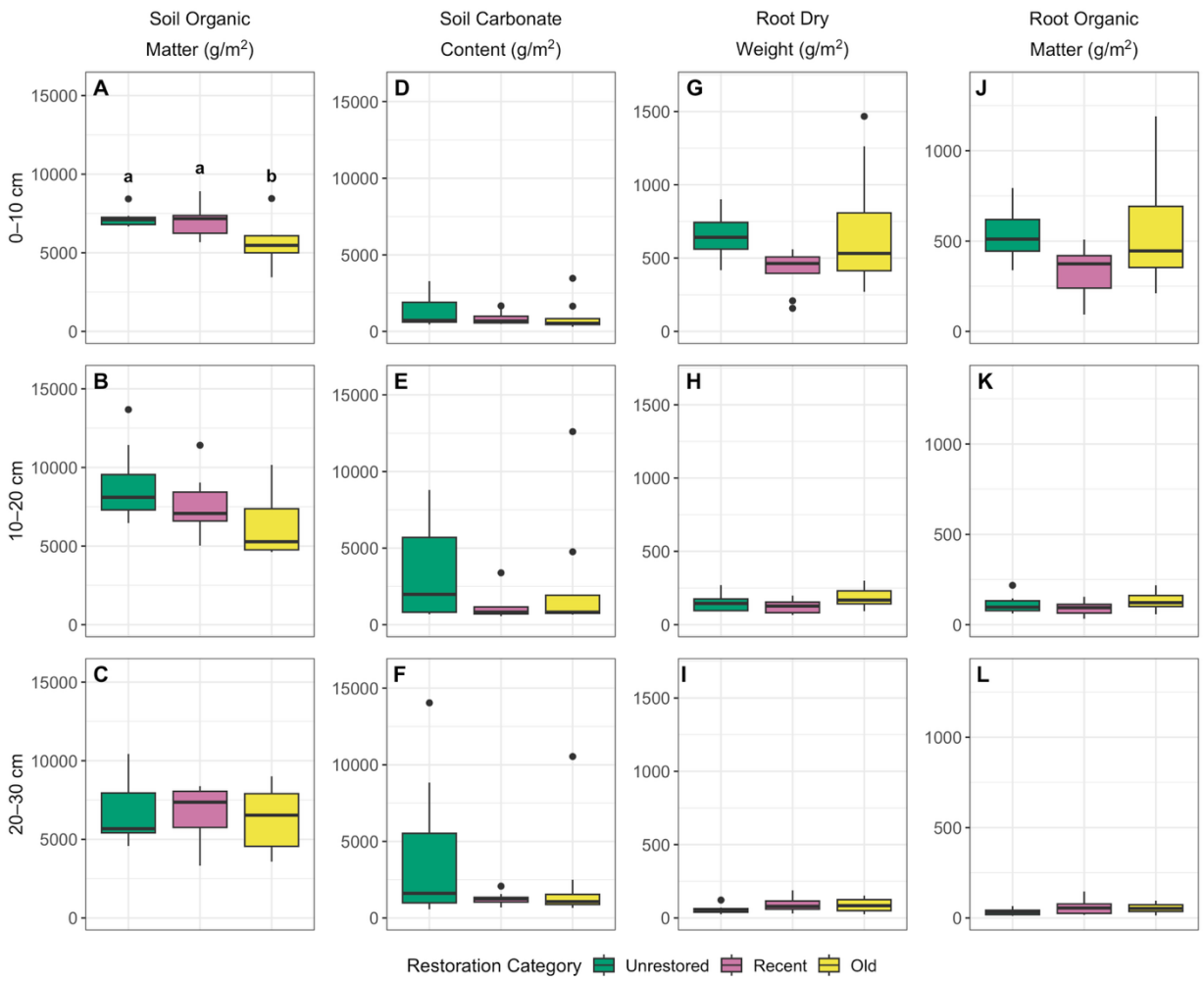


Figure 3.4 Belowground carbon stocks from 2025 in three 10 cm depths reaching 30 cm deep separated into soil organic matter (A-C), soil carbonate content (D-F), root dry weight (G-I) and root organic matter (J-L). Boxplots show medians with 25% quartiles and whiskers display extremes within 1.5 times the spread beyond the quartile. Lowercase letters show results of Tukey's HSD post-hoc tests for ANOVA results with $p < 0.05$. Categories with different letters are significantly different.

When we tested for effects of category alongside depth and their interaction, there continued to be moderate evidence of a difference in SOM among the restoration categories in the 2025 soil cores, but we did not find an effect of depth or of the interaction between depth and category (Table 3.4). We found weak evidence of differences in SCC by both depth and restoration category, but no evidence of an interaction (Table 3.4). We also found moderate

evidence that the restoration categories had different patterns of ROM with depth (Table 3.4): namely, ROM showed less of a decline with depth in meadow categories whereas it dropped off more rapidly in turfgrass (Figure 3.4). RDW shows the same pattern, but the statistical evidence of an interaction is weaker (Table 3.4). However, the magnitude of both RDW and ROM in the bottom 10 cm especially is quite small in the context of overall belowground carbon stocks, with ROM in the bottom 10 cm making up <1% of the total belowground stocks (SOM, SCC and ROM combined) in any category.

Table 3.4 Summary of two-way ANOVA results for belowground carbon fractions with restoration category and depth as covariates (2025 only)

Carbon Fraction	Effect	Degrees of Freedom	F-value	p-value
SOM	Category	2, 66	4.23	0.019
	Depth	1, 66	0.00	0.981
	Category × Depth	2, 66	0.57	0.569
SCC	Category	2, 66	2.80	0.068
	Depth	1, 66	3.78	0.056
	Category × Depth	2, 66	0.81	0.450
RDW	Category	2, 66	2.01	0.142
	Depth	1, 66	86.60	<0.001
	Category × Depth	2, 66	2.78	0.069
ROM	Category	2, 66	2.13	0.127
	Depth	1, 66	87.38	<0.001
	Category × Depth	2, 66	3.23	0.046

3.3.3 Belowground Biomass Production

In the 30 cm deep soil ingrowth cores, old meadow had, on average, higher values than the unrestored turfgrass and the recent meadow was intermediate regardless of whether we consider dry mass or organic matter (Figure 3.5). We found moderate statistical evidence to

support this. RDW production (ANOVA, $F_{2,22} = 4.29$, $p = 0.027$) and ROM production (ANOVA, $F_{2,22} = 4.25$, $p = 0.028$) both differ by restoration category.

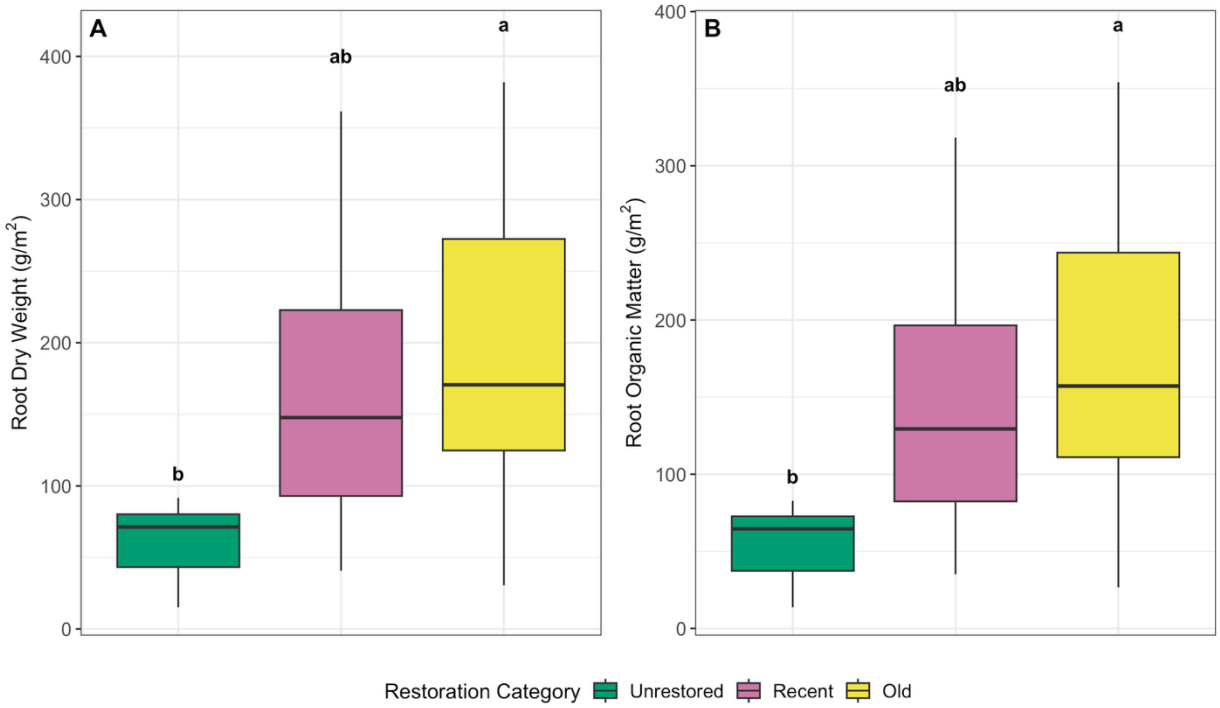


Figure 3.5 Belowground plant production (roots and rhizomes) in the top 30 cm from the 2025 growing season (mid-May to mid-August) presented as total root dry mass (A) and root organic matter only (B). Boxplots show medians with 25% quartiles and whiskers display extremes within 1.5 times the spread beyond the quartile. Points indicate potential outliers Lowercase letters show results of Tukey's HSD post-hoc tests for ANOVA results with $p < 0.05$. Categories with different letters are significantly different.

3.4 Discussion

Our hypothesis that turfgrass-to-meadow restoration will result in greater SOM stocks due, in part, to increased plant production was partially supported by our results. We predicted increased plant production both above and belowground in restored meadow compared to turfgrass. We found strong evidence that both old and recently restored meadow produced more biomass in both 2024 and 2025, and that old meadow produced more belowground biomass compared to turfgrass with recent meadow in an intermediate position. We also predicted that old

meadow sites would have more SOM and root stocks compared to turfgrass, while recent meadow may have similar or lower SOM stocks than turfgrass. This prediction was not supported by our results as we found little statistical evidence of differences among the three categories in 2024 or 2025, apart from weak evidence of differences in root stocks in the 2024 cores. The one major difference in belowground stocks was in SOM in the top 10 cm where, in complete contrast to our prediction, old meadow showed significantly less SOM in 2025 compared to recent meadow and unrestored turfgrass. Finally, we predicted lower SCC in old meadow compared to turfgrass and recent meadow, but we found no difference in SCC between any of the categories.

3.4.1 Aboveground Biomass Production

We measured aboveground biomass to serve both as a stock of live biomass as well as an annual rate of aboveground plant production. As we predicted, we found greater aboveground biomass in the meadow compared to turfgrass both in 2024 and 2025. The average dry weight of aboveground biomass did differ in each category between the years with 2024 having more biomass overall compared to 2025 (Table 3.2). Our sampling did differ between the two years. Most significantly, perhaps, is the difference in sampling dates. In 2025 we sampled a few weeks earlier than we did in 2024. However, in 2024, we sampled weekly to determine if there was a distinct peak aboveground biomass during this period and we did not find one (Appendix K). We also did have one site change (t5 in 2024 and tb in 2025) and two sites not sampled in 2025 due to an unrelated disturbance at those sites (t7 and om8). These site differences could not be responsible for the difference we found between the years. Instead, the difference between years is almost certainly because summer 2025 was particularly dry, classified in the Canadian

Drought Monitor as abnormally dry (1 in 3 years) to moderate drought (1 in 5 years; Agriculture and Agri-Food Canada, National Agroclimate Information Service, 2025). It is well established that plants produce less aboveground biomass when stressed by drought (Kröel-Dulay et al., 2022). Despite producing less aboveground biomass in 2025, we previously showed that meadow still had more net carbon uptake compared to turfgrass under those conditions, suggesting that meadow is more drought-tolerant than mowed turfgrass (Chapter 2). From the perspective of UGS, this is a major advantage of meadow over turfgrass as climate change intensifies.

One complication when it comes to measuring aboveground biomass is that turfgrass is mowed regularly. Because of this, aboveground biomass sampling late in the growing season is not representative of aboveground net primary production for turfgrass. In Chapter 2, we measured CO_2 flux and found more negative NEE in both meadow categories compared to turfgrass, suggesting that turfgrass is indeed less productive than meadow.

We did not measure standing dead biomass in either meadow or turfgrass. Carbon from dead biomass would be expected to cycle differently in these ecosystems. The regular mowing in turfgrass would result in more frequent dead biomass input. However, as we found in Chapter 2, turfgrass litter decomposes faster than meadow litter. Turfgrass clippings are also smaller and softer pieces than a lot of the standing dead biomass from the end of season in the meadow, making turfgrass litter more likely to make good contact with soil, which should speed up decomposition (Douglas et al., 1980). Putting these pieces together, we would expect that dead biomass would contribute less to lasting carbon stocks in the turfgrass compared to meadow.

3.4.2 Belowground Carbon Stocks

Belowground carbon stocks were largely similar between the three restoration age categories, with a few important exceptions. The biggest of these is the significantly lower SOM in the top 10 cm of the 2025 samples from old meadow compared to recent meadow and turfgrass. This result is opposite our predictions and, on the surface, does not easily fit together with the results from Chapter 2. There was no evidence of a difference in the 2024 data. Differences between the years could be due to differences in sampling, or differences in environmental conditions before sampling each year. Finding no difference between the categories, as we did in 2024, would have been contrary to our predictions, but not unprecedented in the literature. For example, Marshall et al. (2023) also found no difference in belowground carbon stocks in the top 10 cm between a newly restored meadow and a well-established, highly managed lawn.

Lower belowground carbon stocks alongside evidence of greater carbon sequestration (as we found in other parts of this chapter and Chapter 2) is unexpected and difficult to explain. Perhaps the most likely explanation is the potential for antecedent differences between site categories unrelated to restoration. We do not have baseline soil carbon stock data from pre-restoration. It is entirely possible that old meadow sites happened to have lower SOM to begin with, especially given that the old meadow sites are more clustered in one section of The Meadoway than other categories (Figure 3.1). This is simply a reality of the history of The Meadoway restoration. There may be spatial differences in topography or other spatial factors that affect carbon dynamics differently in the areas where the earliest meadow restoration happens to have taken place.

There may also be differences in restoration methods between the categories. The site preparation methods, seed mixes, seeding methods, ongoing maintenance regimes all differ between sites as TRCA fine-tune methods and respond to site-specific challenges such as invasive species. For example, site preparation for recent meadow sites typically involved multiple rounds of tillage, which is a widely used restoration method with benefits for non-native seedbank depletion (Kotsopoulos et al., 2024; Wolf & Young, 2016; Xie & MacIvor, 2025). This method was not used when restoring sites in the old meadow category. A consequence of tillage is that SOM is mixed and stable aggregates are broken up and exposed, which results in increased SOM decomposition (Bilen et al., 2020; Gencer et al., 2024; Linsler et al., 2013). Thus, tillage is not a plausible explanation for the differences we observed in SOM because tillage of the recently restored meadow should have resulted in lower SOM, not higher SOM than older restored meadow where no tillage took place. Considering just the comparison between recent meadow and unrestored turfgrass, it is possible that SOM was depleted somewhat by tillage and have only just recovered and may yet exceed the turfgrass. Resampling in five or ten years would confirm whether carbon is accumulating more in the restored meadow compared to turfgrass, as we expect from our chapter 2 results and the higher biomass production demonstrated in this chapter.

We found weak evidence that recent meadow had lower RDW and ROM in the top 10 cm in 2024 but not 2025. There were lower root stocks in recent meadow in 2024 which makes sense because less time has passed to build up stocks (Figure 3.3E,G). Also, several other factors may differ between categories (as we previously discussed) that could affect root stocks such as use of tillage, species composition (different seed mixes), and soil moisture. We did not find any

evidence of this trend in 2025, though recent meadow does appear to be on the lower end in that data too (Figure 3.3F,H).

When we analyzed the 2025 soil core data by depth, we found weak evidence that SCC differed by both depth and restoration category. Carbonate content in The Meadoway is expected to be determined by calcareous parent material, land use history and species composition. Depending on land use history, urban fill could be an anthropogenic carbonate source in some sites, which would be difficult to predict (J.-L. Yang & Zhang, 2015). Figure 3.4 shows that turfgrass has more variance in SCC at all depths, which may have to do with urban fill and other anthropogenic carbonate sources because turfgrass sites more often border the road and pathways in The Meadoway, making these soils more prone to human interference. Plant community-mediated carbonate dynamics are complex, making it difficult to predict a timescale on which we may see effects. It has often been assumed that carbonates cycle much more slowly than organic carbon, but there is also evidence that carbonate pools can shift in less than ten years following a change in the plant community (Wilsey et al., 2020). Deeper, more productive root systems in meadow compared to turfgrass could result in more carbonate dissolution over time (Trémeau et al., 2024; Wilsey et al., 2020). We did find that while there seems to be less roots with increasing depth overall, RDW and ROM decreased less in the meadow categories compared to turfgrass. This was expected as native grasses and forbs tend to root deeper compared to (non-native) turfgrass species (Brown et al., 2010; Trémeau et al., 2024; Weaver, 1968). Roots on their own are not a large belowground carbon stock compared to soil stocks, but they are likely to affect both SOM and SCC.

3.4.3 Belowground Biomass Production

In line with our hypothesis, we found evidence of greater belowground plant production in the old meadow compared to turfgrass, with recent meadow in an intermediate position when looking at either RDW or ROM (Figure 3.5). Combined with the aboveground biomass results, this shows that meadow is more productive above and belowground compared to turfgrass. This is particularly perplexing in light of the unexpected SOM results because root production is associated with SOM accumulation (Rasse et al., 2005).

Interestingly, belowground production in recent meadow shows an intermediate effect, but aboveground production did not. Combining the 2025 ingrowth core data which represents one season's belowground production and the 2025 soil core stocks data, it appears that only a fraction of the root stock is produced each season (Figures 3.4 & 3.5). This shows that root stocks do build up over time, but not evenly in each category. In 2025, turfgrass produced only 7.1±3.4% of the total stock. Meanwhile, old meadow produced 20.0±12.7% of the total stock, and recent meadow produced 26.8±16.9% of the total stock. It makes sense that recent meadow shows the highest proportion of root stocks produced in one season because it would be expected to have the least root stock built up. There was weak evidence that recent meadow root stocks were lower than other categories in 2024 in the top 10 cm. We did not find statistical evidence of this trend in 2025, but Figure 3.4 shows lower root stocks in recent meadow than in old meadow, especially in the top 10 cm and 10-20 cm depths.

Like the aboveground biomass, the soil ingrowth cores were undoubtedly affected by the dry conditions we experienced in summer 2025 (Agriculture and Agri-Food Canada, National Agroclimate Information Service, 2025). However, unlike with the aboveground biomass, we do

not have another year to compare to. Thus, we cannot say for sure to what extent it was affected. However, in Chapter 2 we found evidence that the meadow might be more drought-tolerant than turfgrass. Certainly, GPP was consistently higher in the meadow than the turfgrass during the period in which the ingrowth cores were deployed (Figure 2.2). It is likely that in a different year without any drought, similar to the aboveground biomass, there would be more overall belowground plant production in all categories. However, if we are right that meadow is more drought-tolerant, it is possible that the difference would be smaller between turfgrass and meadow. Even if this is true, the meadow provides a climate-resilience that turfgrass cannot.

3.5 Conclusions and Next Steps

Despite not completely supporting our main hypothesis, this chapter largely provides evidence that turfgrass-to-meadow restoration enhances carbon storage and sequestration. Building on our work in Chapter 2, here we show that old meadow had 214% and 272% more aboveground biomass stock than turfgrass in 2024 and 2025 respectively, and 213% higher belowground plant production compared to turfgrass. With time, this increased plant production is expected to result in more carbon accumulation. Contrary to our prediction, we found 22.4% lower SOM in the old meadow surface soils (upper 10 cm) in 2025. Given our other findings in this chapter and in Chapter 2, it is unlikely that this result is related to age since restoration, though we cannot test this without pre-restoration data. The Meadoway is a long, complex ecosystem with a varied restoration history. Our biggest recommendation would be to resample belowground carbon stocks in five to ten years to more accurately quantify trends in carbon accumulation at each site regardless of complicating factors like differences in topography and

restoration history. Still, our findings here can go on to inform future work on turfgrass-to-meadow restoration.

Conclusion

4.1 Thesis Overview

UGS maintained as turfgrass are ubiquitous in the urban landscape (Ignatieva et al., 2015), yet are of low quality compared with alternatives such as native plant meadow (Bretzel et al., 2024; Marshall et al., 2023). Meadows are semi-natural grasslands which prominently feature native forbs and are mown infrequently (Kotsopoulos et al., 2024; Marshall et al., 2023; Norton et al., 2019). They function as an effective NbS to the biodiversity crisis (Norton et al., 2019) and as with many NbS, they have the potential to offer multiple co-benefits, impacting carbon storage and sequestration as well. There are a variety of reasons to expect that turfgrass-to-meadow restoration will enhance carbon storage and sequestration, yet few studies have sought to quantify this effect. Comparisons between turfgrass and meadow-like ecosystems are made more complex by the large variation in maintenance regimes that exist for both turfgrass and meadow ecosystems.

The Meadoway project began in 2013 when TRCA began restoring a 16 km hydro corridor in Scarborough, Ontario, actively removing pre-existing turfgrass and replacing it with meadow. In order to share their knowledge and lessons learned from the project, TRCA has created “The Meadoway Manual”, a “how-to” guide on urban turfgrass-to-meadow for application across North America and beyond in similar contexts (Toronto and Region Conservation Authority, 2025). To deepen that knowledge, TRCA has partnered with universities in Ontario to help quantify the benefits of turfgrass-to-meadow restoration using The Meadoway as a study system. The work presented in this thesis is part of that partnership, with the objective of quantifying the effect of turfgrass-to-meadow restoration on carbon storage and sequestration. Using the chronosequence created by The Meadoway restoration timeline, we made comparisons

between three restoration age categories: unrestored turfgrass, recently restored meadow (restored in 2020-2023), and older restored meadow (restored in 2013-2016). We set out to compare various aspects of carbon dynamics between these categories such as greenhouse gas fluxes, decomposition rates, plant production and belowground carbon stores.

4.2 Thesis Summary

In Chapter 2, we measured fluxes of CO_2 and CH_4 to directly measure carbon sequestration in The Meadoway. We measured light (NEE) and dark (ER) fluxes biweekly between the weeks of June 9th and November 17th, 2025. We calculated GPP and scaled up NEE, GPP and ER to represent rough daily rates for comparison between the three age categories. We also calculated estimates of cumulative NEE, GPP and ER over the course of the sampling season. We found evidence that restored meadow (-12.5 ± 19.1 and -12.3 ± 6.44 mol m^{-2} $season^{-1}$) had more negative cumulative NEE compared to turfgrass ($+14.1 \pm 17.0$ mol m^{-2} $season^{-1}$), signifying greater carbon sequestration. We found that neither recent nor old meadow differed from turfgrass when it comes to CH_4 flux. In the same chapter, we also report on a 366-day plant litter transplant experiment where we tested whether decomposition rates would differ due to litter quality, site conditions, or a combination of litter quality and site conditions. We found both litter quality and site conditions to have strong effects, depending on whether we analyzed by the overall decay rate or based only on the total mass lost over the one-year experiment. Meadow litter decomposed 33.6% more slowly than turfgrass regardless of where it is placed, but all litter decomposed 34.7% more at recent meadow sites compared to turfgrass sites. There was no interaction between litter category and site category in either model. The gas flux results are

good evidence that meadow is sequestering more carbon than turfgrass. The litter bag experiment confirms that litter from turfgrass decomposes more rapidly than litter from meadow sites and that litter quality plays a bigger role than site conditions in explaining the variance in the decay rate analysis. However, the mass lost analysis shows that more litter decayed in recent meadow sites compared to turfgrass, suggesting site conditions vary among the categories and also impact decomposition rates.

In Chapter 3, we measured above and belowground plant production representing both a standing carbon stock and a rate of seasonal net primary production, part of which may resist decomposition and remain as part of long-term belowground carbon stores. We found 214-334% more aboveground biomass in restored meadow compared to turfgrass between both years that we sampled. We also found 213% greater belowground plant production in old meadow compared to turfgrass, with recent meadow in an intermediate position. Greater above and belowground plant production represents more carbon temporarily stored in plant tissue and greater input for potential long-term storage. We also measured belowground carbon stocks, quantifying soil organic matter and carbonate content, as well as root stocks. Belowground carbon stocks were largely not different between the three categories with the important exception that SOM was 22.4% lower in the older restored meadow compared to turfgrass. Without baseline samples from before the meadow restoration, it is not possible to determine whether this difference is due to antecedent differences in soil quality among the different Meadoway sections or is a consequence of the restoration; however, given the greater aboveground vegetation standing crop, slower litter decomposition rates and more negative cumulative NEE, it seems likely that SOM will accumulate in the restored meadow more rapidly than in turfgrass.

A summary of our findings from chapters 2 and 3 can be seen in Figure 4.1. This figure shows that turfgrass overall seems to be emitting more carbon compared to restored meadow which is taking up more carbon. Recent meadow shows some more intermediate features, but it ultimately had similarly negative cumulative NEE as old meadow, suggesting greater carbon sequestration. Over time, recent meadow may reach an equilibrium more similar to the old meadow.

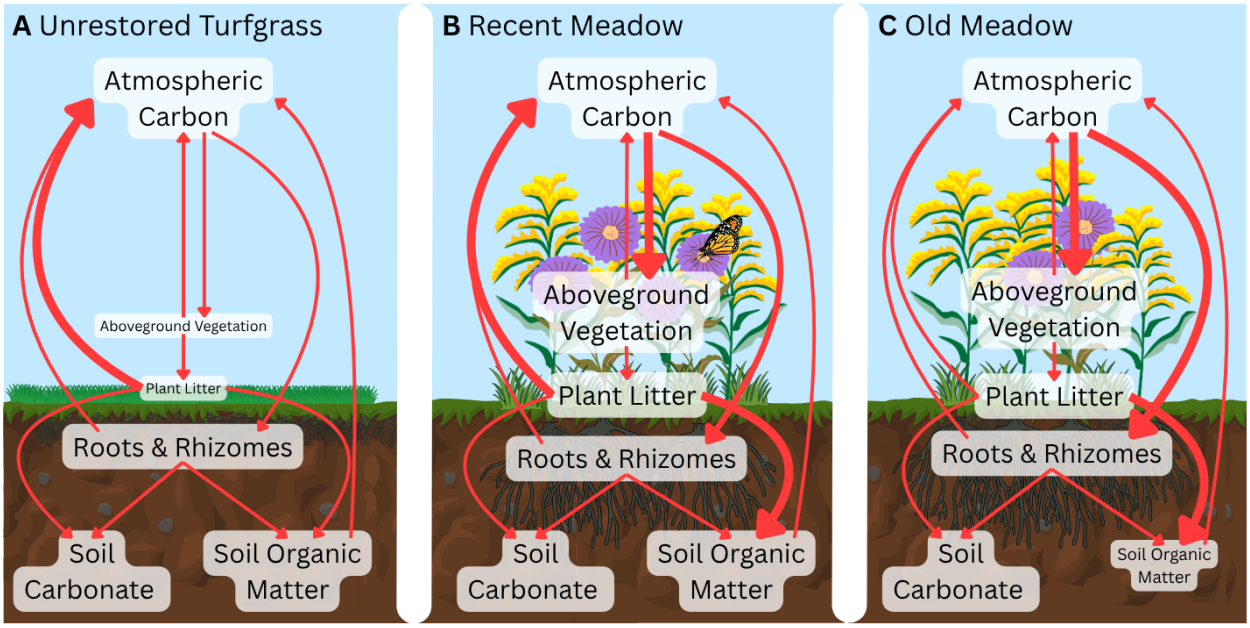


Figure 0.1 Conceptual diagram showing the different carbon pools with arrows representing the fluxes between pools for each restoration age category: unrestored turfgrass (A), recently restored meadow (B) and old restored meadow (C). The size of the text boxes and arrows roughly represents the relative size of the pool/flux between the categories based on our results. Made using Canva.com.

4.3 Overall Discussion and Recommendations

At 16 km in length, restoring the entire Meadoway at once was logistically and financially impossible. The sequence of restoration of sections across the Meadoway was beneficial in providing TRCA with the opportunity for continuous improvement in restoration

practice through learning-by-doing. It also resulted in a chronosequence of restoration, enabling us to contemporaneously test the effects of restoration with a space-for-time substitution. The main limitation of our space-for-time substitution is that we do not have baseline data on carbon stores and fluxes from before restoration took place to confirm that antecedent conditions were equivalent across the sections. Given the more negative NEE, the greater aboveground and belowground plant biomass, and the more decay-resistant litter produced by restored meadow, we would expect the SOM to increase with time since restoration. Assuming that all sections of the Meadoway began with the same SOM levels, we would thus expect SOM to be highest in the older meadow. Our finding that SOM was actually lower in the older restored meadow is plausible if, by coincidence, the first sections targeted for restoration happened to have lower SOM than the sections restored later and the remnant turf areas. If we could recommend just one thing, it would be to take soil samples again in five or ten years to investigate trends in SOM accumulation (or depletion) with time since restoration.

There are potential spatial factors that could differ among the age categories we used in this thesis such as topography, soil composition and structure, and water table depth. Without baseline data which could allow us to correct for the starting point of each site, we cannot know the effect of such differences, especially when thinking about the belowground carbon stocks. Further, although adaptive management undoubtedly improved the outcome of restoration efforts in the Meadoway, it meant restoration methods like site preparation and invasive species management evolved through time and consequently differ between the older and more recent restoration categories.

A potentially consequential example of this is the use of tillage in recent sites but not old. One consequence of tillage is that SOM is mixed, resulting ultimately in increased SOM

decomposition (Bilen et al., 2020; Gencer et al., 2024; Linsler et al., 2013). Tillage in recent meadow but not old may have mobilized buried SOM that is still decomposing. Indeed, the upper 10 cm of soil from recently restored meadow had higher SOM than the equivalent soil from older restored meadow.

Recent meadow sites also had higher soil water content most of the season than turfgrass or old meadow (Figure 2.3), possibly as a consequence of this higher surficial SOM since SOM contributes to the water holding capacity of soils (Hudson, 1994). We discussed how this may have supported greater microbial activity during an abnormally dry summer, with drier sites being moisture limited. It could be that the tilling undertaken to deplete invasive plants from the seedbank before seeding with native meadow plants promoted greater soil moisture and brought buried SOM to the surface, facilitating the decay of that SOM and the faster decay rates and greater mass loss we observed of all litter types in the recently restored meadow sites.

Whether in summers with greater moisture availability decay rates in the recently restored meadow would remain proportionately higher than in older meadow and turfgrass is unclear. Our results in chapter 2 are based on work conducted during just one field season – one which was characterized by mild-to-moderate drought (Agriculture and Agri-Food Canada, National Agroclimate Information Service, 2025). While not extreme, these conditions undoubtedly affected our results. We were able to show that meadow is more resilient to drought than turfgrass, but it is not clear how fluxes and litter decomposition might differ with different weather. Thus, one recommendation would be to repeat the work of that chapter over multiple years in order to have a better idea of long-term trends in carbon sequestration post-restoration.

The goal of The Meadoway restoration was to improve biodiversity and habitat connectivity. Like many NbS, The Meadoway offers additional co-benefits, one of which is

enhanced carbon sequestration, as we have demonstrated in this thesis. However, there is potential for trade-offs between these benefits which restoration practitioners must face. One such trade-off comes from the use of tillage for restoration methods. Tillage is known to deplete otherwise stable SOM stores (Bilen et al., 2020; Gencer et al., 2024; Linsler et al., 2013), but has benefits for invasive species management during site preparation (Kotsopoulos et al., 2024; Wolf & Young, 2016; Xie & MacIvor, 2025). A future study focusing on tillage for ecosystem restoration site preparation could give practitioners the information they need to properly weigh the costs and benefits of tillage.

Because of the varied ways in which turfgrass and meadow ecosystems are managed, the limited number of studies previously published on this topic generally find that meadow is the same or less effective than turfgrass at sequestering and storing carbon (Malmsten, 2023; Marshall et al., 2023; Pouyat et al., 2009; Thienelt & Anderson, 2021; Trémeau et al., 2024). Despite some limitations, the work presented in this thesis does show that urban turfgrass-to-meadow restoration can enhance carbon sequestration. This is significant because it means that, at least in some contexts, there is not a trade-off between the biodiversity benefits of urban turfgrass-to-meadow restoration and climate change mitigation. Of course, future work is needed to better understand which specific restoration practices are likely to improve or worsen a carbon sequestration co-benefit. Still, the methods used by TRCA which are outlined in “The Meadoway Manual” (Toronto and Region Conservation Authority, 2025) have resulted in an ecosystem that seems to be functioning to sequester more carbon than adjacent turfgrass. There is no shortage of UGS managed as turfgrass in the urban landscape across the Western World (Ignatieva et al., 2015), so there is much opportunity to improve urban ecosystem service provision by replacing turfgrass with meadow. Our work shows that the insights shared by TRCA in The Meadoway

Manual can be effective for implementing NbS with multiple co-benefits addressing the joint biodiversity and climate crisis through urban turfgrass-to-meadow restoration.

References

Aerts, R. (1997). Climate, leaf litter chemistry and leaf litter decomposition in terrestrial ecosystems: A triangular relationship. *Oikos*, *79*(3), 439–449. https://doi.org/10.2307/3546886

Ágoston-Szabó, E., & Dinka, M. (2008). Decomposition of *Typha angustifolia* and *Phragmites australis* in the littoral zone of a shallow lake. *Biologia*, *63*(6), 1104–1110. https://doi.org/10.2478/s11756-008-0154-4

Agriculture and Agri-Food Canada, National Agroclimate Information Service. (2025). *Canadian drought monitor* [Dataset]. https://agriculture.canada.ca/en/agricultural-production/weather/canadian-drought-monitor

Aronson, M. F. J., La Sorte, F. A., Nilon, C. H., Katti, M., Goddard, M. A., Lepczyk, C. A., Warren, P. S., Williams, N. S. G., Cilliers, S., Clarkson, B., Dobbs, C., Dolan, R., Hedblom, M., Klotz, S., Kooijmans, J. L., Kühn, I., MacGregor-Fors, I., McDonnell, M., Mörtberg, U., … Winter, M. (2014). A global analysis of the impacts of urbanization on bird and plant diversity reveals key anthropogenic drivers. *Proceedings of the Royal Society B: Biological Sciences*, *281*(1780), 20133330. https://doi.org/10.1098/rspb.2013.3330

Auestad, I., & Rydgren, K. (2026). Keep mowing: Predictions show that successful grassland restoration depends on post-restoration management. *Ecosphere*, *17*(3), e70538. https://doi.org/10.1002/ecs2.70538

Ayres, E., Dromph, K. M., & Bardgett, R. D. (2006). Do plant species encourage soil biota that specialise in the rapid decomposition of their litter? *Soil Biology and Biochemistry*, *38*(1), 183–186. https://doi.org/10.1016/j.soilbio.2005.04.018

Bansal, S., Creed, I. F., Tangen, B. A., Bridgham, S. D., Desai, A. R., Krauss, K. W., Neubauer, S. C., Noe, G. B., Rosenberry, D. O., Trettin, C., Wickland, K. P., Allen, S. T., Arias-Ortiz, A., Armitage, A. R., Baldocchi, D., Banerjee, K., Bastviken, D., Berg, P., Bogard, M. J., … Zhu, X. (2023). Practical guide to measuring wetland carbon pools and fluxes. *Wetlands*, *43*(8), 105. https://doi.org/10.1007/s13157-023-01722-2

Bartoń, K. (2026). *MuMIn: Multi-Model Inference* (Version 1.48.19) [Computer software]. https://cran.r-project.org/web/packages/MuMIn/index.html

Bates, D., Maechler, M., Bolker [cre, B., aut, Walker, S., Christensen, R. H. B., Singmann, H., Dai, B., Scheipl, F., Grothendieck, G., Green, P., Fox, J., Bauer, A., simulate.formula), P. N. K. (shared copyright on, Tanaka, E., Jagan, M., Boylan, R. D., & Ly, A. (2026). *lme4: Linear Mixed-Effects Models using "Eigen" and S4* (Version 2.0-1) [Computer software]. https://cran.r-project.org/web/packages/lme4/index.html

Bilen, S., Çelik, A., & Altıkat, S. (2020). Effects of strip and full-width tillage on soil carbon IV oxide-carbon (CO_2-C) fluxes and on bacterial and fungal populations in sunflower. *African Journal of Biotechnology*, *9*(38). https://avesis.atauni.edu.tr/yayin/f3f21f82-e7ff-4119-8a88-287d94c48057/effects-of-strip-and-full-width-tillage-on-soil-carbon-iv-oxide-carbon-co2-c-fluxes-and-on-bacterial-and-fungal-populations-in-sunflower

Borchers, H. W. (2025). *pracma: Practical Numerical Math Functions* (Version 2.4.6) [Computer software]. https://cran.r-project.org/web/packages/pracma/index.html

Bretzel, F., Vannucchi, F., Pezzarossa, B., Paraskevopoulou, A., & Romano, D. (2024). Establishing wildflower meadows in anthropogenic landscapes. *Frontiers in Horticulture*, *2*. https://doi.org/10.3389/fhort.2023.1248785

Brown, R. N., Percivalle, C., Narkiewicz, S., & DeCuollo, S. (2010). Relative rooting depths of native grasses and amenity grasses with potential for use on roadsides in New England. *HortScience*, *45*(3), 393–400. https://doi.org/10.21273/HORTSCI.45.3.393

Bugg, T. D. H., Ahmad, M., Hardiman, E. M., & Rahmanpour, R. (2011). Pathways for degradation of lignin in bacteria and fungi. *Natural Product Reports*, *28*(12), 1883–1896. https://doi.org/10.1039/C1NP00042J

Byrne, L. B., Bruns, M. A., & Kim, K. C. (2008). Ecosystem properties of urban land covers at the aboveground–belowground interface. *Ecosystems*, *11*(7), 1065–1077. https://doi.org/10.1007/s10021-008-9179-3

Cardinale, B. J., Matulich, K. L., Hooper, D. U., Byrnes, J. E., Duffy, E., Gamfeldt, L., Balvanera, P., O'Connor, M. I., & Gonzalez, A. (2011). The functional role of producer diversity in ecosystems. *American Journal of Botany*, *98*(3), 572–592. https://doi.org/10.3732/ajb.1000364

Chen, X., Chen, H. Y. H., Chen, C., Ma, Z., Searle, E. B., Yu, Z., & Huang, Z. (2020). Effects of plant diversity on soil carbon in diverse ecosystems: A global meta-analysis. *Biological Reviews*, *95*(1), 167–183. https://doi.org/10.1111/brv.12554

Cilliers, S., Cilliers, J., Lubbe, R., & Siebert, S. (2013). Ecosystem services of urban green spaces in African countries—Perspectives and challenges. *Urban Ecosystems*, *16*(4), 681–702. https://doi.org/10.1007/s11252-012-0254-3

Cohen-Shacham, E., Walters, G., Janzen, C., & Maginnis, S. (Eds.). (2016). *Nature-based solutions to address global societal challenges*. IUCN International Union for Conservation of Nature. https://doi.org/10.2305/IUCN.CH.2016.13.en

Douglas, C. L., Allmaras, R. R., Rasmussen, P. E., Ramig, R. E., & Roager, N. C. (1980). Wheat straw composition and placement effects on decomposition in dryland agriculture of the Pacific Northwest. *Soil Science Society of America Journal*, *44*(4), 833–837. https://doi.org/10.2136/sssaj1980.03615995004400040035x

Elzhov, T. V., Mullen, K. M., Spiess, A.-N., & Bolker, B. (2023). *minpack.lm: R interface to the Levenberg-Marquardt nonlinear least-squares algorithm found in MINPACK, plus support for bounds* (Version 1.2-4) [Computer software]. https://cran.r-project.org/web/packages/minpack.lm/index.html

European Commission: Directorate-General for Research and Innovation. (2015). *Towards an EU research and innovation policy agenda for nature-based solutions & re-naturing cities – Final report of the Horizon 2020 expert group on "Nature-based solutions and re-naturing cities" – (full version)*. Publications Office. https://doi.org/10.2777/765301

Fornara, D. A., & Tilman, D. (2008). Plant functional composition influences rates of soil carbon and nitrogen accumulation. *Journal of Ecology*, *96*(2), 314–322. https://doi.org/10.1111/j.1365-2745.2007.01345.x

Frantzeskaki, N. (2019). Seven lessons for planning nature-based solutions in cities. *Environmental Science & Policy*, *93*, 101–111. https://doi.org/10.1016/j.envsci.2018.12.033

Gampe, D., Zscheischler, J., Reichstein, M., O'Sullivan, M., Smith, W. K., Sitch, S., & Buermann, W. (2021). Increasing impact of warm droughts on northern ecosystem productivity over recent decades. *Nature Climate Change*, *11*(9), 772–779. https://doi.org/10.1038/s41558-021-01112-8

Gartner, T. B., & Cardon, Z. G. (2004). Decomposition dynamics in mixed-species leaf litter. *Oikos*, *104*(2), 230–246. https://doi.org/10.1111/j.0030-1299.2004.12738.x

Gaur, A., Gaur, A., Yamazaki, D., & Simonovic, S. P. (2019). Flooding related consequences of climate change on Canadian cities and flow regulation infrastructure. *Water*, *11*(1), 63. https://doi.org/10.3390/w11010063

Gencer, M., Gök, M., & Celik, İ. (2024). Sustaining soil biological activity: The role of extended reduced and no-tillage techniques. *Kahramanmaraş Sütçü İmam Üniversitesi Tarım ve Doğa Dergisi*, *27*(6), 1443–1458. https://doi.org/10.18016/ksutarimdoga.vi.1428787

Gholz, H. L., Wedin, D. A., Smitherman, S. M., Harmon, M. E., & Parton, W. J. (2000). Long-term dynamics of pine and hardwood litter in contrasting environments: Toward a global model of decomposition. *Global Change Biology*, *6*(7), 751–765. https://doi.org/10.1046/j.1365-2486.2000.00349.x

Gilmanov, T. G., Tieszen, L. L., Wylie, B. K., Flanagan, L. B., Frank, A. B., Haferkamp, M. R., Meyers, T. P., & Morgan, J. A. (2005). Integration of CO_2 flux and remotely-sensed data for primary production and ecosystem respiration analyses in the Northern Great Plains: Potential for quantitative spatial extrapolation. *Global Ecology and Biogeography*, *14*(3), 271–292. https://doi.org/10.1111/j.1466-822X.2005.00151.x

Godshalk, G. L., & Wetzel, R. G. (1978). Decomposition of aquatic angiosperms. II. Particulate components. *Aquatic Botany*, *5*, 301–327. https://doi.org/10.1016/0304-3770(78)90074-8

Government of Canada. (2025, June 3). *Nature-based climate solutions*. https://www.canada.ca/en/services/environment/our-environment/nature-based-climate-solutions.html

Graves, S., Piepho, H.-P., Selzer, L., & Dorai-Raj, S. (2026). *multcompView: Visualizations of Paired Comparisons* (Version 0.1-11) [Computer software]. https://cran.r-project.org/web/packages/multcompView/index.html

Grimm, N. B., Faeth, S. H., Golubiewski, N. E., Redman, C. L., Wu, J., Bai, X., & Briggs, J. M. (2008). Global change and the ecology of cities. *Science*, *319*(5864), 756–760.

Hendrickson, J. R., Wienhold, B. J., & Berdahl, J. D. (2001). Decomposition rates of native and improved cultivars of grasses in the Northern Great Plains. *Arid Land Research and Management*, *15*(4), 347–357. https://doi.org/10.1080/153249801753127633

Hudson, B. D. (1994). Soil organic matter and available water capacity. *Journal of Soil and Water Conservation*, *49*(2), 189–194. https://doi.org/10.1080/00224561.1994.12456850

Huhta, A.-P., Rautio, P., Tuomi, J., & Laine, K. (2001). Restorative mowing on an abandoned semi-natural meadow: Short-term and predicted long-term effects. *Journal of Vegetation Science*, *12*(5), 677–686. https://doi.org/10.2307/3236908

Ignatieva, M., Ahrné, K., Wissman, J., Eriksson, T., Tidåker, P., Hedblom, M., Kätterer, T., Marstorp, H., Berg, P., Eriksson, T., & Bengtsson, J. (2015). Lawn as a cultural and ecological phenomenon: A conceptual framework for transdisciplinary research. *Urban Forestry & Urban Greening*, *14*(2), 383–387. https://doi.org/10.1016/j.ufug.2015.04.003

Ignatieva, M., Haase, D., Dushkova, D., & Haase, A. (2020). Lawns in cities: From a globalised urban green space phenomenon to sustainable nature-based solutions. *Land*, *9*(3), 73. https://doi.org/10.3390/land9030073

Jasek-Kamińska, A., Zimnoch, M., Wachniew, P., & Różański, K. (2020). Urban CO_2 budget: Spatial and seasonal variability of CO_2 emissions in Krakow, Poland. *Atmosphere*, *11*(6), 629. https://doi.org/10.3390/atmos11060629

Kirschbaum, M. U. F. (1995). The temperature dependence of soil organic matter decomposition, and the effect of global warming on soil organic C storage. *Soil Biology and Biochemistry*, *27*(6), 753–760. https://doi.org/10.1016/0038-0717(94)00242-S

Kondo, M. C., Fluehr, J. M., McKeon, T., & Branas, C. C. (2018). Urban green space and its impact on human health. *International Journal of Environmental Research and Public Health*, *15*(3), 445. https://doi.org/10.3390/ijerph15030445

Kotsopoulos, S., Turnbull, K., Cormack, C., Cartwright, L. A., Hayes, S. M., Ford, B., Shachak, R., & Watkinson, A. (2024). The Meadoway: Native meadow creation in underutilized transmission line corridors. *Botany*, *102*(10), 421–427. https://doi.org/10.1139/cjb-2023-0157

Kröel-Dulay, G., Mojzes, A., Szitár, K., Bahn, M., Batáry, P., Beier, C., Bilton, M., De Boeck, H. J., Dukes, J. S., Estiarte, M., Holub, P., Jentsch, A., Schmidt, I. K., Kreyling, J., Reinsch, S., Larsen, K. S., Sternberg, M., Tielbörger, K., Tietema, A., … Peñuelas, J. (2022). Field experiments underestimate aboveground biomass response to drought. *Nature Ecology & Evolution*, *6*(5), 540–545. https://doi.org/10.1038/s41559-022-01685-3

Kucharik, C. J., Fayram, N. J., & Cahill, K. N. (2006). A paired study of prairie carbon stocks, fluxes, and phenology: Comparing the world's oldest prairie restoration with an adjacent remnant. *Global Change Biology*, *12*(1), 122–139. https://doi.org/10.1111/j.1365-2486.2005.01053.x

Kuznetsova, A., Brockhoff, P. B., Christensen, R. H. B., & Jensen, S. P. (2026). *lmerTest: Tests in Linear Mixed Effects Models* (Version 3.2-1) [Computer software]. https://cran.r-project.org/web/packages/lmerTest/index.html

Le Mer, J., & Roger, P. (2001). Production, oxidation, emission and consumption of methane by soils: A review. *European Journal of Soil Biology*, *37*(1), 25–50. https://doi.org/10.1016/S1164-5563(01)01067-6

Linsler, D., Geisseler, D., Loges, R., Taube, F., & Ludwig, B. (2013). Temporal dynamics of soil organic matter composition and aggregate distribution in permanent grassland after a single tillage event in a temperate climate. *Soil and Tillage Research*, *126*, 90–99. https://doi.org/10.1016/j.still.2012.07.017

Malmsten, A. M. (2023). *Comparing soil carbon content in urban meadows and managed lawns in the Helsinki Metropolitan Area* [University of Helsinki]. http://hdl.handle.net/10138/562777

Marshall, C. A. M., Wilkinson, M. T., Hadfield, P. M., Rogers, S. M., Shanklin, J. D., Eversham, B. C., Healey, R., Kranse, O. P., Preston, C. D., Coghill, S. J., McGonigle, K. L., Moggridge, G. D., Pilbeam, P. G., Marza, A. C., Szigecsan, D., Mitchell, J., Hicks, M. A., Wallis, S. M., Xu, Z., … Eves-van den Akker, S. (2023). Urban wildflower meadow planting for biodiversity, climate and society: An evaluation at King's College, Cambridge. *Ecological Solutions and Evidence*, *4*(2), e12243. https://doi.org/10.1002/2688-8319.12243

Meenar, M., Heckert, M., & Adlakha, D. (2022). "Green enough ain't good enough:" Public perceptions and emotions related to green infrastructure in environmental justice communities. *International Journal of Environmental Research and Public Health*, *19*(3), 1448. https://doi.org/10.3390/ijerph19031448

Miao, Q., & Nduneseokwu, C. (2024). Global environmental leadership: Addressing the triple planetary crisis. In *Environmental Leadership in a VUCA Era: An Interdisciplinary Handbook* (pp. 37–119). Springer Nature. https://doi.org/10.1007/978-981-96-0324-4_2

Monteiro, J. A. (2017). Ecosystem services from turfgrass landscapes. *Urban Forestry & Urban Greening*, *26*, 151–157. https://doi.org/10.1016/j.ufug.2017.04.001

Mosier, A., Schimel, D., Valentine, D., Bronson, K., & Parton, W. (1991). Methane and nitrous oxide fluxes in native, fertilized and cultivated grasslands. *Nature*, *350*(6316), 330–332. https://doi.org/10.1038/350330a0

National Research Council Canada. (n.d.). *Sunrise/sunset calculator*. National Research Council Canada. Retrieved April 16, 2026, from https://nrc.canada.ca/en/research-development/products-services/software-applications/sunrisesunset-calculator

Neill, C. (1992). Comparison of soil coring and ingrowth methods for measuring belowground production. *Ecology*, *73*(5), 1918–1921. JSTOR. https://doi.org/10.2307/1940044

Niemelä, J., Saarela, S.-R., Söderman, T., Kopperoinen, L., Yli-Pelkonen, V., Väre, S., & Kotze, D. J. (2010). Using the ecosystem services approach for better planning and conservation of urban green spaces: A Finland case study. *Biodiversity and Conservation*, *19*(11), 3225–3243. https://doi.org/10.1007/s10531-010-9888-8

Norton, B. A., Bending, G. D., Clark, R., Corstanje, R., Dunnett, N., Evans, K. L., Grafius, D. R., Gravestock, E., Grice, S. M., Harris, J. A., Hilton, S., Hoyle, H., Lim, E., Mercer, T. G., Pawlett, M., Pescott, O. L., Richards, J. P., Southon, G. E., & Warren, P. H. (2019). Urban meadows as an alternative to short mown grassland: Effects of composition and height on biodiversity. *Ecological Applications*, *29*(6), e01946. https://doi.org/10.1002/eap.1946

Olson, J. S. (1963). Energy storage and the balance of producers and decomposers in ecological systems. *Ecology*, *44*(2), 322–331. https://doi.org/10.2307/1932179

Onandia, G., Schittko, C., Ryo, M., Bernard-Verdier, M., Heger, T., Joshi, J., Kowarik, I., & Gessler, A. (2019). Ecosystem functioning in urban grasslands: The role of biodiversity, plant invasions and urbanization. *PLOS ONE*, *14*(11), e0225438. https://doi.org/10.1371/journal.pone.0225438

Paudel, S., & States, S. L. (2023). Urban green spaces and sustainability: Exploring the ecosystem services and disservices of grassy lawns versus floral meadows. *Urban Forestry & Urban Greening*, *84*, 127932. https://doi.org/10.1016/j.ufug.2023.127932

Pedersen, T. L. (2025). *patchwork: The composer of plots* (Version 1.3.2) [Computer software]. https://cran.r-project.org/web/packages/patchwork/index.html

Pelmorex Corp. (2025). *The Weather Network* (Version 6.6) [iOS]. https://apps.apple.com/us/app/the-weather-network/id473299958

Pereira, P., Yin, C., & Hua, T. (2023). Nature-based solutions, ecosystem services, disservices, and impacts on well-being in urban environments. *Current Opinion in Environmental Science & Health*, *33*, 100465. https://doi.org/10.1016/j.coesh.2023.100465

Posit team. (2026). *RStudio: Integrated development environment for R*. Posit Software, PBC. http://www.posit.co/

Pouyat, R. V., & Carreiro, M. M. (2003). Controls on mass loss and nitrogen dynamics of oak leaf litter along an urban-rural land-use gradient. *Oecologia*, *135*(2), 288–298. https://doi.org/10.1007/s00442-003-1190-y

Pouyat, R. V., Yesilonis, I. D., & Golubiewski, N. E. (2009). A comparison of soil organic carbon stocks between residential turf grass and native soil. *Urban Ecosystems*, *12*(1), 45–62. https://doi.org/10.1007/s11252-008-0059-6

Printz, J. L., & Hendrickson, J. R. (2015). Impacts of Kentucky bluegrass Invasion (*Poa pratensis* L.) on ecological processes in the Northern Great Plains. *Rangelands*, *37*(6), 226–232. https://doi.org/10.1016/j.rala.2015.10.006

Pugnaire, F. I., Aares, K. H., Alifriqui, M., Bråthen, K. A., Kindler, C., Schöb, C., & Manrique, E. (2023). Home-field advantage effects in litter decomposition is largely linked to litter quality. *Soil Biology and Biochemistry*, *184*, 109069. https://doi.org/10.1016/j.soilbio.2023.109069

Quinlan, A., Dale, M. R. T., & Gates, C. C. (2003). Effects of prescribed burning on herbaceous and woody vegetation in northern lowland meadows. *Restoration Ecology*, *11*(3), 343–350. https://doi.org/10.1046/j.1526-100X.2003.00219.x

R Core Team. (2026). *R: A language and environment for statistical computing*. R Foundation for Statistical Computing. https://www.R-project.org/

Rasse, D. P., Rumpel, C., & Dignac, M.-F. (2005). Is soil carbon mostly root carbon? Mechanisms for a specific stabilisation. *Plant and Soil*, *269*(1), 341–356. https://doi.org/10.1007/s11104-004-0907-y

Russell, K. N., Russell, G. J., Kaplan, K. L., Mian, S., & Kornbluth, S. (2018). Increasing the conservation value of powerline corridors for wild bees through vegetation management: An experimental approach. *Biodiversity and Conservation*, *27*(10), 2541–2565. https://doi.org/10.1007/s10531-018-1552-8

Selhorst, A., & Lal, R. (2013). Net carbon sequestration potential and emissions in home lawn turfgrasses of the United States. *Environmental Management*, *51*(1), 198–208. https://doi.org/10.1007/s00267-012-9967-6

Seneviratne, S. I., Donat, M. G., Mueller, B., & Alexander, L. V. (2014). No pause in the increase of hot temperature extremes. *Nature Climate Change*, *4*(3), 161–163. https://doi.org/10.1038/nclimate2145

Shi, W., Muruganandam, S., & Bowman, D. (2006). Soil microbial biomass and nitrogen dynamics in a turfgrass chronosequence: A short-term response to turfgrass clipping addition. *Soil Biology and Biochemistry*, *38*(8), 2032–2042. https://doi.org/10.1016/j.soilbio.2006.01.005

Simmons, B. L., & Coleman, D. C. (2008). Microbial community response to transition from conventional to conservation tillage in cotton fields. *Applied Soil Ecology*, *40*(3), 518–528. https://doi.org/10.1016/j.apsoil.2008.08.003

Smith, P. (2004). How long before a change in soil organic carbon can be detected? *Global Change Biology*, *10*(11), 1878–1883. https://doi.org/10.1111/j.1365-2486.2004.00854.x

Statistics Canada. (2026, January 14). Canada's population estimates: Subprovincial areas, 2025. *The Daily*. https://www150.statcan.gc.ca/n1/daily-quotidien/260114/dq260114a-eng.htm

Strickland, M. S., Lauber, C., Fierer, N., & Bradford, M. A. (2009). Testing the functional significance of microbial community composition. *Ecology*, *90*(2), 441–451. https://doi.org/10.1890/08-0296.1

Talbot, J. M., & Treseder, K. K. (2012). Interactions among lignin, cellulose, and nitrogen drive litter chemistry–decay relationships. *Ecology*, *93*(2), 345–354. https://doi.org/10.1890/11-0843.1

Thienelt, T. S., & Anderson, D. E. (2021). Estimates of energy partitioning, evapotranspiration, and net ecosystem exchange of CO_2 for an urban lawn and a tallgrass prairie in the Denver metropolitan area under contrasting conditions. *Urban Ecosystems*, *24*(6), 1201–1220. https://doi.org/10.1007/s11252-021-01108-4

Thompson, G. L., & Kao-Kniffin, J. (2017). Applying biodiversity and ecosystem function theory to turfgrass management. *Crop Science*, *57*(S1), S-238-S-248. https://doi.org/10.2135/cropsci2016.05.0433

Toronto and Region Conservation Authority. (2023). *The Meadoway – Community powered green spaces*. The Meadoway. https://themeadoway.ca/

Toronto and Region Conservation Authority. (2025). *The Meadoway manual: A blueprint for greening infrastructure corridors*. Toronto and Region Conservation Authority (TRCA). https://themeadoway.ca/app/uploads/2022/10/Meadoway-Manual-2022-FA-Web_v2.pdf

Townsend-Small, A., & Czimczik, C. I. (2010). Carbon sequestration and greenhouse gas emissions in urban turf. *Geophysical Research Letters*, *37*(2). https://doi.org/10.1029/2009GL041675

Trémeau, J., Olascoaga, B., Backman, L., Karvinen, E., Vekuri, H., & Kulmala, L. (2024). Lawns and meadows in urban green space – a comparison from perspectives of greenhouse gases, drought resilience and plant functional types. *Biogeosciences*, *21*(4), 949–972. https://doi.org/10.5194/bg-21-949-2024

UNFCCC. (2022, April 13). What is the triple planetary crisis? *United Nations Framework Connvention on Climate Change*. https://unfccc.int/news/what-is-the-triple-planetary-crisis

VandenBygaart, A. J., & Kay, B. D. (2004). Persistence of soil organic carbon after plowing a long-term no-till field in Southern Ontario, Canada. *Soil Science Society of America Journal*, *68*(4), 1394–1402. https://doi.org/10.2136/sssaj2004.1394

Wagner, D. L., Metzler, K. J., & Frye, H. (2019). Importance of transmission line corridors for conservation of native bees and other wildlife. *Biological Conservation*, *235*, 147–156. https://doi.org/10.1016/j.biocon.2019.03.042

Wang, D., He, N., Wang, Q., Lü, Y., Wang, Q., Xu, Z., & Zhu, J. (2016). Effects of temperature and moisture on soil organic matter decomposition along elevation gradients on the Changbai Mountains, Northeast China. *Pedosphere*, *26*(3), 399–407. https://doi.org/10.1016/S1002-0160(15)60052-2

Wang, F. L., & Bettany, J. R. (1995). Methane emission from a usually well-drained prairie soil after snowmelt and precipitation. *Canadian Journal of Soil Science*, *75*(2), 239–241. https://doi.org/10.4141/cjss95-033

Wang, L., Zhou, Y., Chen, Y., Xu, Z., Zhang, J., Liu, Y., & Joly, F.-X. (2022). Litter diversity accelerates labile carbon but slows recalcitrant carbon decomposition. *Soil Biology and Biochemistry*, *168*, 108632. https://doi.org/10.1016/j.soilbio.2022.108632

Wang, R., Mattox, C. M., Phillips, C. L., & Kowalewski, A. R. (2022). Carbon sequestration in turfgrass–soil systems. *Plants*, *11*(19), Article 19. https://doi.org/10.3390/plants11192478

Wang, Y., Berardi, U., & Akbari, H. (2016). Comparing the effects of urban heat island mitigation strategies for Toronto, Canada. *Energy and Buildings, SI: Countermeasures to Urban Heat Island*, *114*, 2–19. https://doi.org/10.1016/j.enbuild.2015.06.046

Weaver, J. E. (1968). *Prairie plants and their environment ; a fifty-year study in the Midwest*. University of Nebraska Press.

Wen, H., Sullivan, P. L., Macpherson, G. L., Billings, S. A., & Li, L. (2021). Deepening roots can enhance carbonate weathering by amplifying CO_2-rich recharge. *Biogeosciences*, *18*(1), 55–75. https://doi.org/10.5194/bg-18-55-2021

Wickham, H., Chang, W., Henry, L., Pedersen, T. L., Takahashi, K., Wilke, C., Woo, K., Yutani, H., Dunnington, D., Brand, T. van den, Posit, & PBC. (2026). *ggplot2: Create elegant data visualisations using the grammar of graphics* (Version 4.0.2) [Computer software]. https://cran.r-project.org/web/packages/ggplot2/index.html

Wickham, H., François, R., Henry, L., Müller, K., Vaughan, D., Software, P., & PBC. (2026). *dplyr: A grammar of data manipulation* (Version 1.2.0) [Computer software]. https://cran.r-project.org/web/packages/dplyr/index.html

Wilsey, B., Xu, X., Polley, H. W., Hofmockel, K., & Hall, S. J. (2020). Lower soil carbon stocks in exotic vs. Native grasslands are driven by carbonate losses. *Ecology*, *101*(7), e03039. https://doi.org/10.1002/ecy.3039

Winkler, J., Pasternak, G., Sas, W., Hurajová, E., Koda, E., & Vaverková, M. D. (2024). Nature-based management of lawns—Enhancing biodiversity in urban green infrastructure. *Applied Sciences*, *14*(5), 1705. https://doi.org/10.3390/app14051705

Wolf, K. M., & Young, T. P. (2016). Summer water in a restored native grassland flushes annual grass seed bank but fails to increase native perennial cover. *Ecosphere*, *7*(6), e01309. https://doi.org/10.1002/ecs2.1309

Xie, G., & MacIvor, J. S. (2025). Evaluating rehabilitation practices to inform invasion resistance in designed urban meadows. *Landscape and Urban Planning*, *261*, 105386. https://doi.org/10.1016/j.landurbplan.2025.105386

Yang, J.-L., & Zhang, G.-L. (2015). Formation, characteristics and eco-environmental implications of urban soils – A review. *Soil Science and Plant Nutrition*, *61*(sup1), 30–46. https://doi.org/10.1080/00380768.2015.1035622

Yang, Y., Tilman, D., Furey, G., & Lehman, C. (2019). Soil carbon sequestration accelerated by restoration of grassland biodiversity. *Nature Communications*, *10*(1), 718. https://doi.org/10.1038/s41467-019-08636-w

Yuckin, S. J., Howell, G., Robichaud, C. D., & Rooney, R. C. (2023). Phragmites australis invasion and herbicide-based control changes primary production and decomposition in a freshwater wetland. *Wetlands Ecology and Management*, *31*(1), 73–88. https://doi.org/10.1007/s11273-022-09902-3

Zölch, T., Rahman, M. A., Pfleiderer, E., Wagner, G., & Pauleit, S. (2019). Designing public squares with green infrastructure to optimize human thermal comfort. *Building and Environment*, *149*, 640–654. https://doi.org/10.1016/j.buildenv.2018.12.051

Appendices

Appendix A – Imputation of Missing Flux Variable Data

Due to missing or malfunctioning equipment, we did not collect air temperature, barometric pressure and soil temperature data at the beginning of our sampling season using the same equipment we used the rest of the season. Air temperature (ºC) was imputed for dates before July 21st, 2025 and atmospheric pressure (hPa) was imputed for dates before June 25th, 2025 based on values noted from The Weather Network app while in the field (Pelmorex Corp., 2025). Soil temperature values (ºC) were imputed for dates before June 27th, 2025 based on a more basic temperature probe for which the make and model is unknown. For each of these, data was collected using the preferred method alongside the alternate method for some time once we got the preferred method working. From here, we used linear models in order to impute values as we have already mentioned. The models are presented in the table here.

Table A.1 Linear models for imputing missing data used for gas flux calculations

Variable	Equation	Slope *p*-value	Intercept *p*-value	R^2	RSE	*n*
Air Temperature (ºC)	y = −1.321 + 1.329x	<0.001	0.276	0.659	2.346	296
Atmospheric Pressure (hPa)	y = −22.956 + 1.006x	<0.001	0.535	0.881	1.785	106
Soil Temperature (ºC)	y = −27.627 + 2.252x	<0.001	<0.001	0.801	2.264	128

Appendix B – Sample Gas Flux Calculations

Presented here is the process for gas flux calculations following values from site rm1 on June 27, 2025.

Step 1: Convert each CO_2 concentration (ppm) to moles using ideal gas law $n = \frac{PV}{RT}$ where P is the partial pressure of CO_2 (kPa), V is the chamber volume (L), n is moles of CO_2, R is the Ideal Gas Constant (8.314 $\frac{L \cdot kPa}{mol \cdot K}$), T is the temperature (K).

Following the initial concentration of CO_2 in ppm,

$$V = \pi r^2 h = \pi(0.15)^2(0.605) = 0.0428\, m^2 = 42.8L$$

$$P = \frac{443.2281}{1\,000\,000} \times 100.4\, kPa = \ 0.0445\, kPa$$

$$T = 21.3ºC + 273.15 = 294.45K$$

$$n = \frac{PV}{RT} = \frac{0.04450\, kPa \times 42.8\, L}{8.314\, \frac{L \cdot kPa}{mol \cdot K} \times 294.45\, K} = 0.000778\, mol$$

Step 2: Linear regression on converted values from 90 second flux interval to get rate of change

```
Call:
lm(formula = flux$molesco2noveg ~ flux$elapsed_seconds)

Residuals:
       Min         1Q     Median         3Q        Max
-1.350e-06 -3.981e-07  4.700e-09  4.274e-07  1.651e-06

Coefficients:
                      Estimate Std. Error t value Pr(>|t|)
(Intercept)           7.763e-04  1.308e-07  5934.8   <2e-16 ***
flux$elapsed_seconds -1.108e-06  2.510e-09  -441.2   <2e-16 ***
---
Signif. codes:  0 ‘***’ 0.001 ‘**’ 0.01 ‘*’ 0.05 ‘.’ 0.1 ‘ ’ 1

Residual standard error: 6.29e-07 on 89 degrees of freedom
Multiple R-squared:  0.9995,    Adjusted R-squared:  0.9995
F-statistic: 1.947e+05 on 1 and 89 DF,  p-value: < 2.2e-16
```

Here the estimated slope is $-1.108 \times 10^{-6}\, mol/s$.

Step 3: Calculate flux rate

$$flux = \frac{slope}{SA} = \frac{-1.108 \times 10^{-6}\ mol/s}{\pi(0.15)^2} = -1.567 \times 10^{-5}\ mol\ m^{-2}s^{-1}$$

Step 4: Calculate GPP

$$GPP = ER - NEE = 7.671 \times 10^{-6}\ mol\ m^{-2}s^{-1} - (-1.567 \times 10^{-5}\ mol\ m^{-2}s^{-1})$$
$$= 2.33 \times 10^{-5}\ mol\ m^{-2}s^{-1}$$

Step 5: Scale ER and GPP to daily rates, with GPP scaled according to daylight hours from National Research Council Canada's sunrise/sunset calculator set to Toronto (National Research Council Canada, n.d.)

$$ER_{daily} = 7.67 \times 10^{-6}\ mol\ m^{-2}s^{-1} \times 3600\frac{s}{h} \times 24\frac{h}{day} = 0.663\ mol\ m^{-2}day^{-1}$$

$$GPP_{daily} = 2.33 \times 10^{-5}\ mol\ m^{-2}s^{-1} \times 3600\frac{s}{h} \times 15.43\frac{daylight\ hours}{day}$$
$$= 1.29\ mol\ m^{-2}day^{-1}$$

Step 6: Back calculate daily NEE

$$NEE_{daily} = ER_{daily} - GPP_{daily} = 0.663\ mol\ m^{-2}day^{-1} - 1.29\ mol\ m^{-2}day^{-1}$$
$$= -0.627\ mol\ m^{-2}day^{-1}$$

Appendix C – Ancillary Analysis for Litter Bag Content Ratio

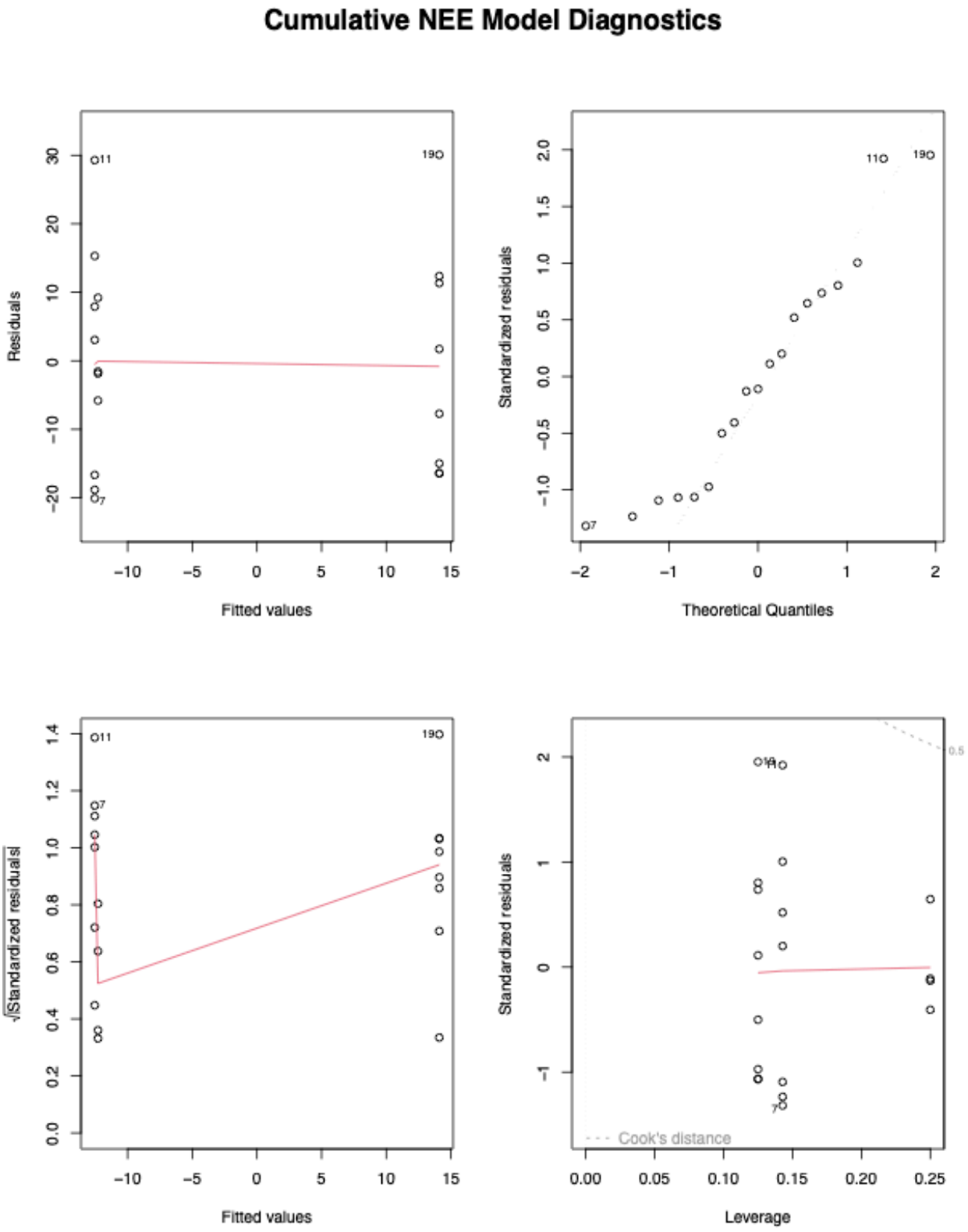
As mentioned in the main body of chapter 2, we used a rough ratio of 7 g of stem tissue to 3 g of leaf tissue when filling each litter bag with biomass. We determined this approximate ratio by mixing a few samples from each old and recent meadow separately, then separating the loose leaves from the stems with attached leaves. We weighed each fraction and in both old and recent meadow, there was roughly a 7:3 ratio of stem to leaf mass. Thus, for simplicity and consistency, we roughly used this ratio to fill the litter bags from these categories. Turfgrass litter is more homogenous, so no similar analysis was done for that category.

Appendix D – Two-parameter Decay Model

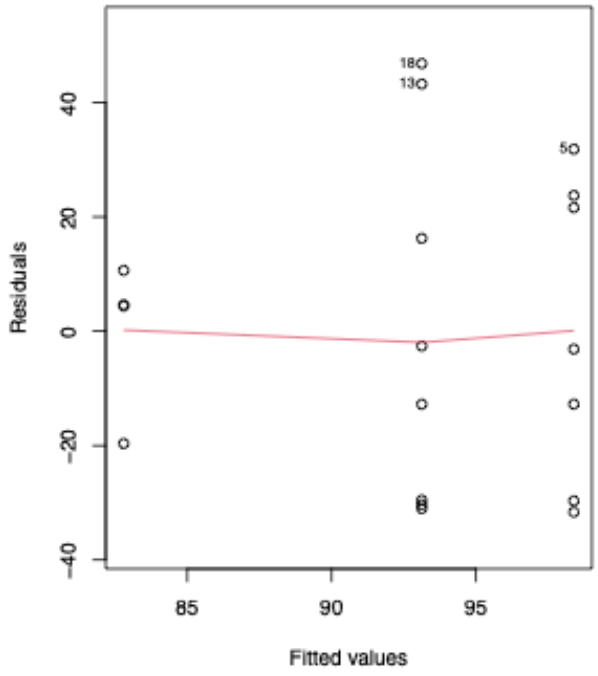
Table D.1 Summary of four models which explain the litter decomposition data presented in chapter 2: a single exponential decay rate model and a proportion mass lost model which were presented and discussed in the main body of this thesis, alongside linear models for each of two-parameters which represent the initial decay rate as well as a transitional rate which modifies the initial rate.

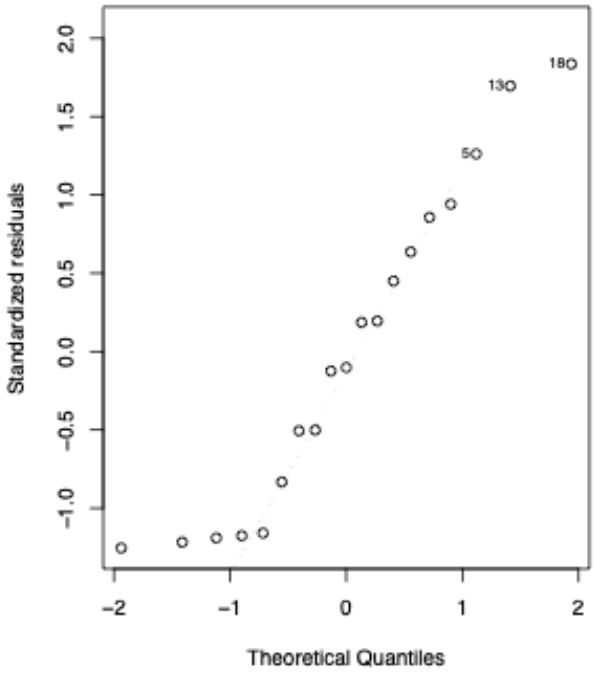
Model	Factor	df	F value	p-value
k (single k)	Site category	2	3.83	0.041
	Litter category	2	14.23	<0.001
	Site category × litter category	4	0.44	0.781
k1 (initial rate)	Site category	2	0.03	0.967
	Litter category	2	5.45	0.014
	Site category × litter category	4	0.27	0.893
k2 (transition rate)	Site category	2	1.46	0.259
	Litter category	2	1.31	0.294
	Site category × litter category	4	0.31	0.864
Proportion Mass Lost	Site category	2	11.35	<0.001
	Litter category	2	7.43	0.001
	Site category × litter category	4	0.89	0.476

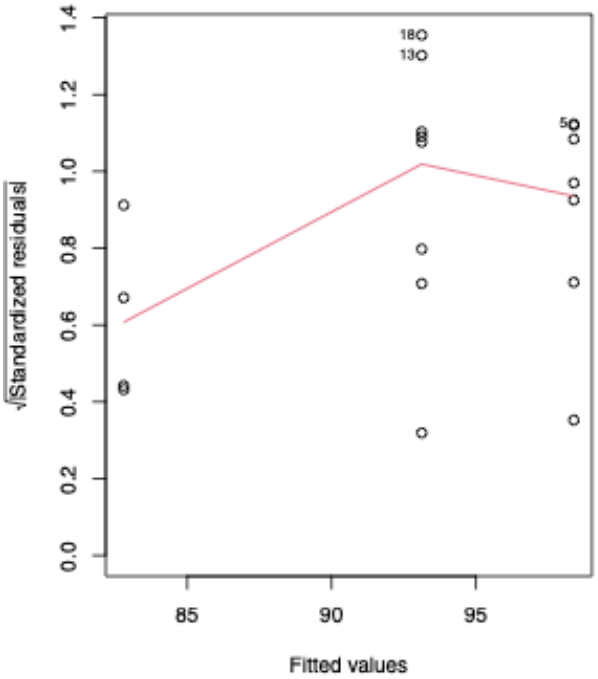
Appendix E – Diagnostic Plots for Flux Models

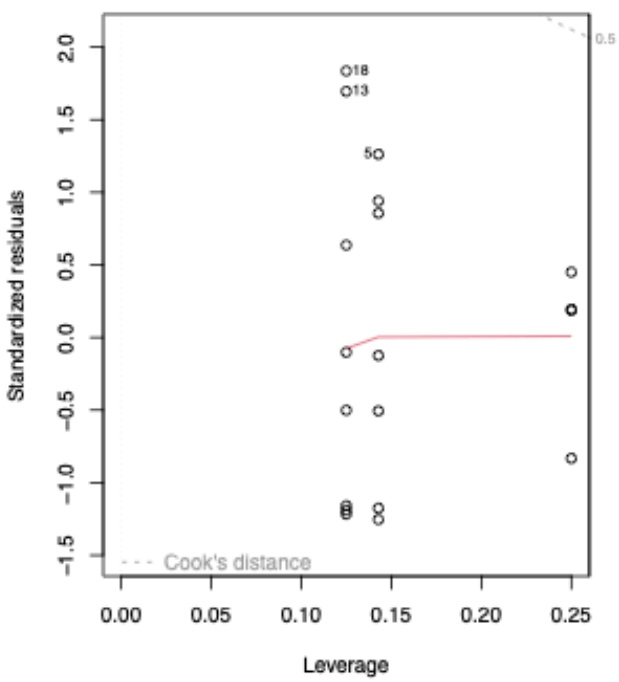


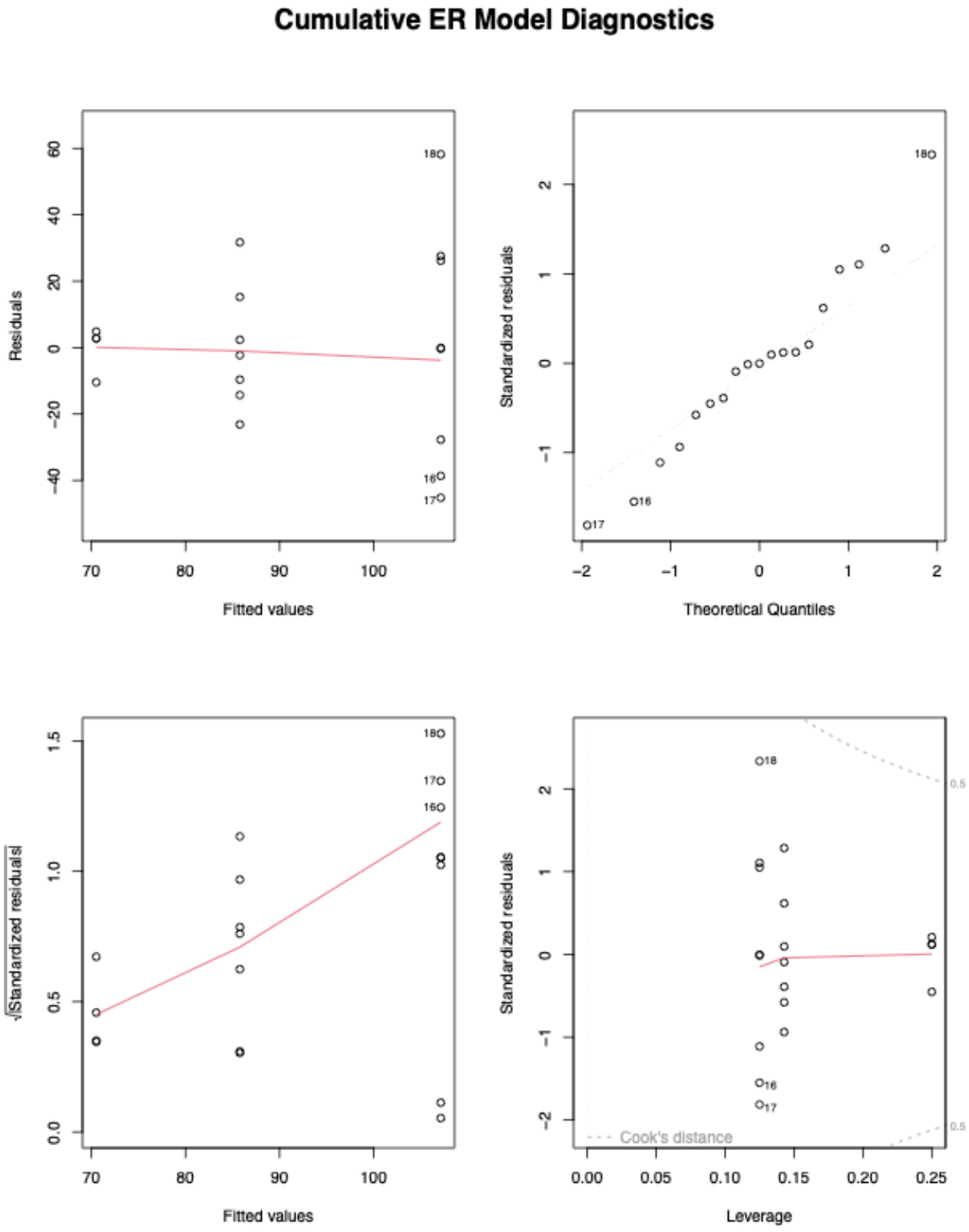
Cumulative GPP Model Diagnostics



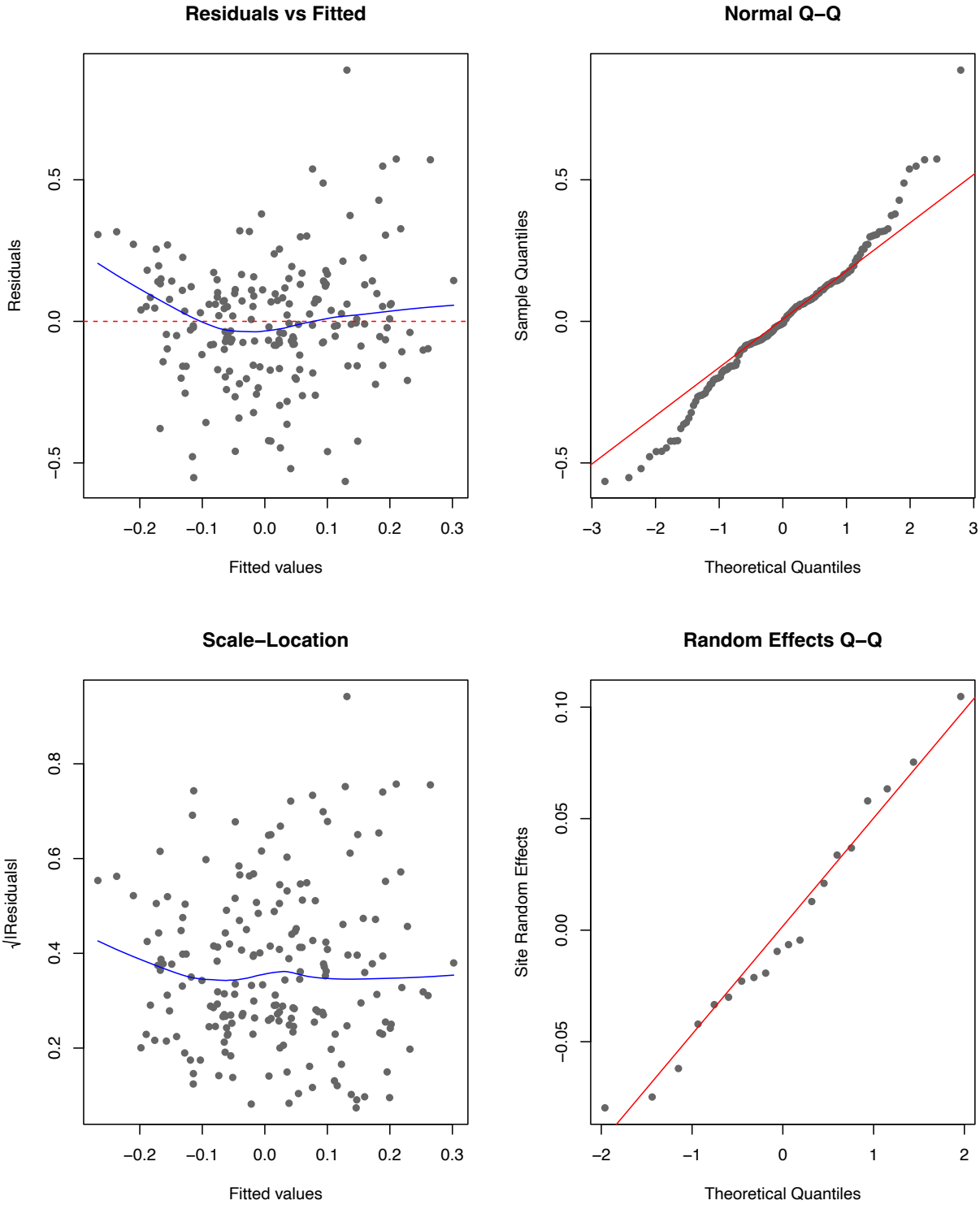




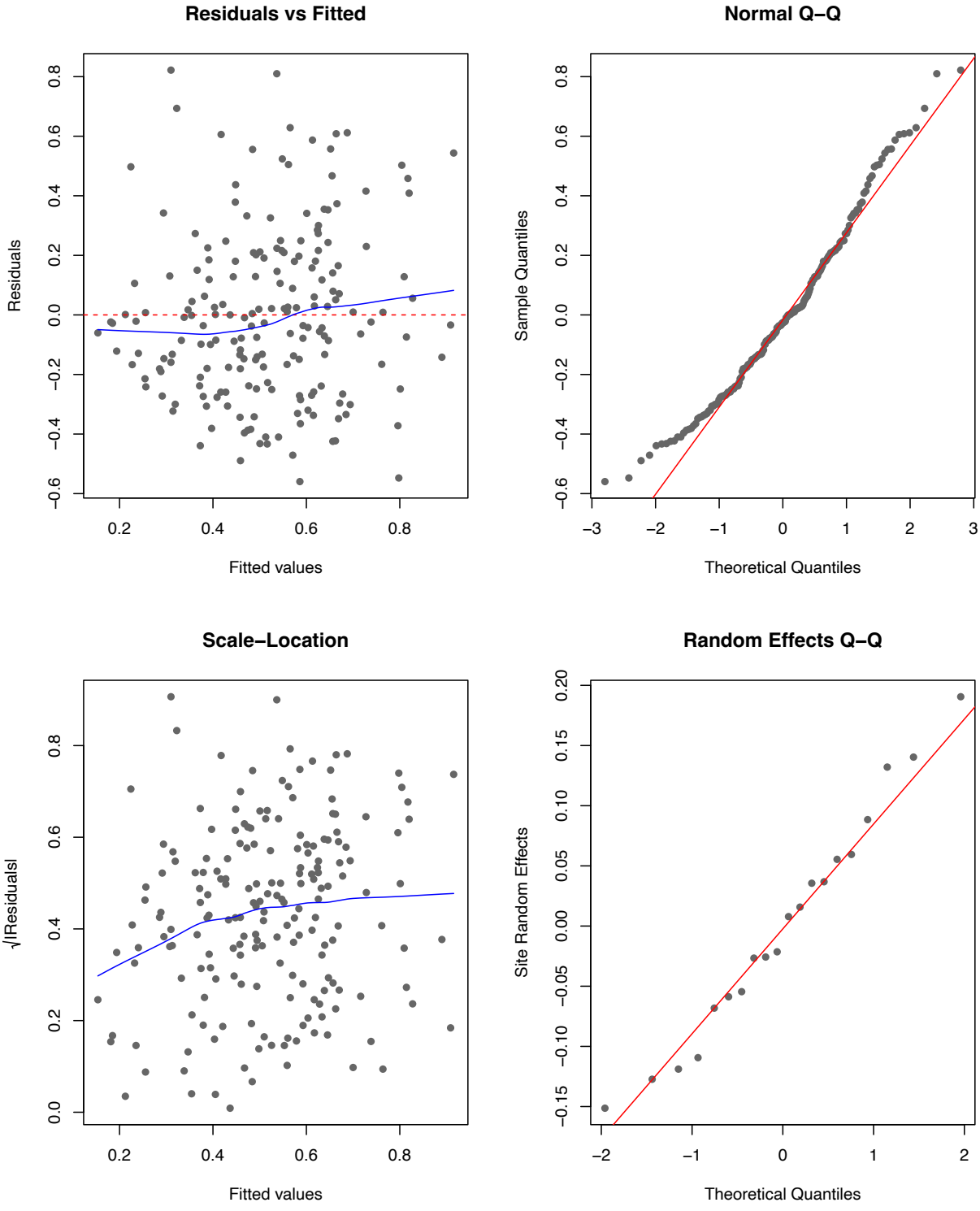



Cumulative ER Model Diagnostics
Residuals
Fitted values
Standardized residuals
Theoretical Quantiles
√|Standardized residuals|
Fitted values
Standardized residuals
Leverage
Cook's distance

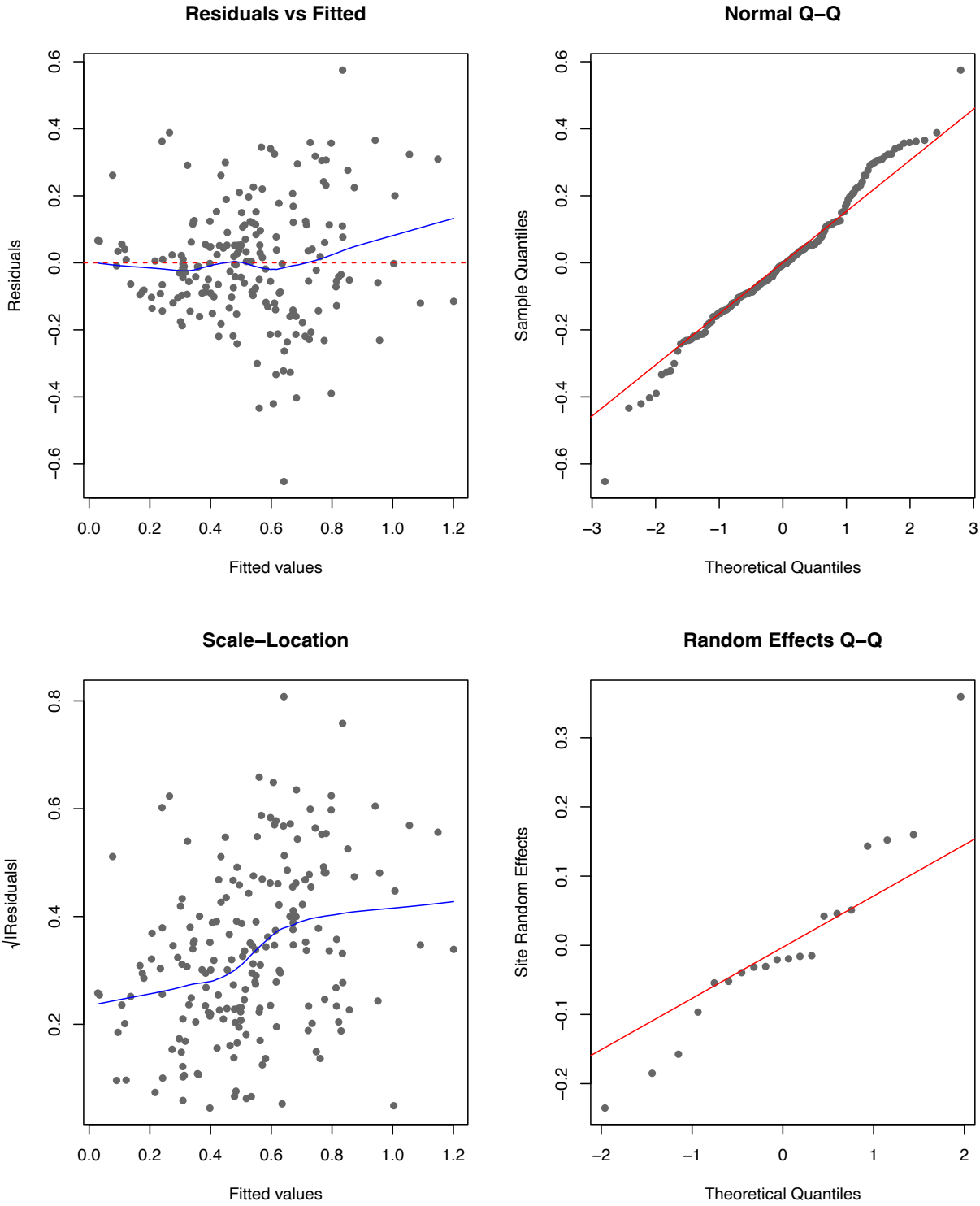
NEE Model Diagnostics



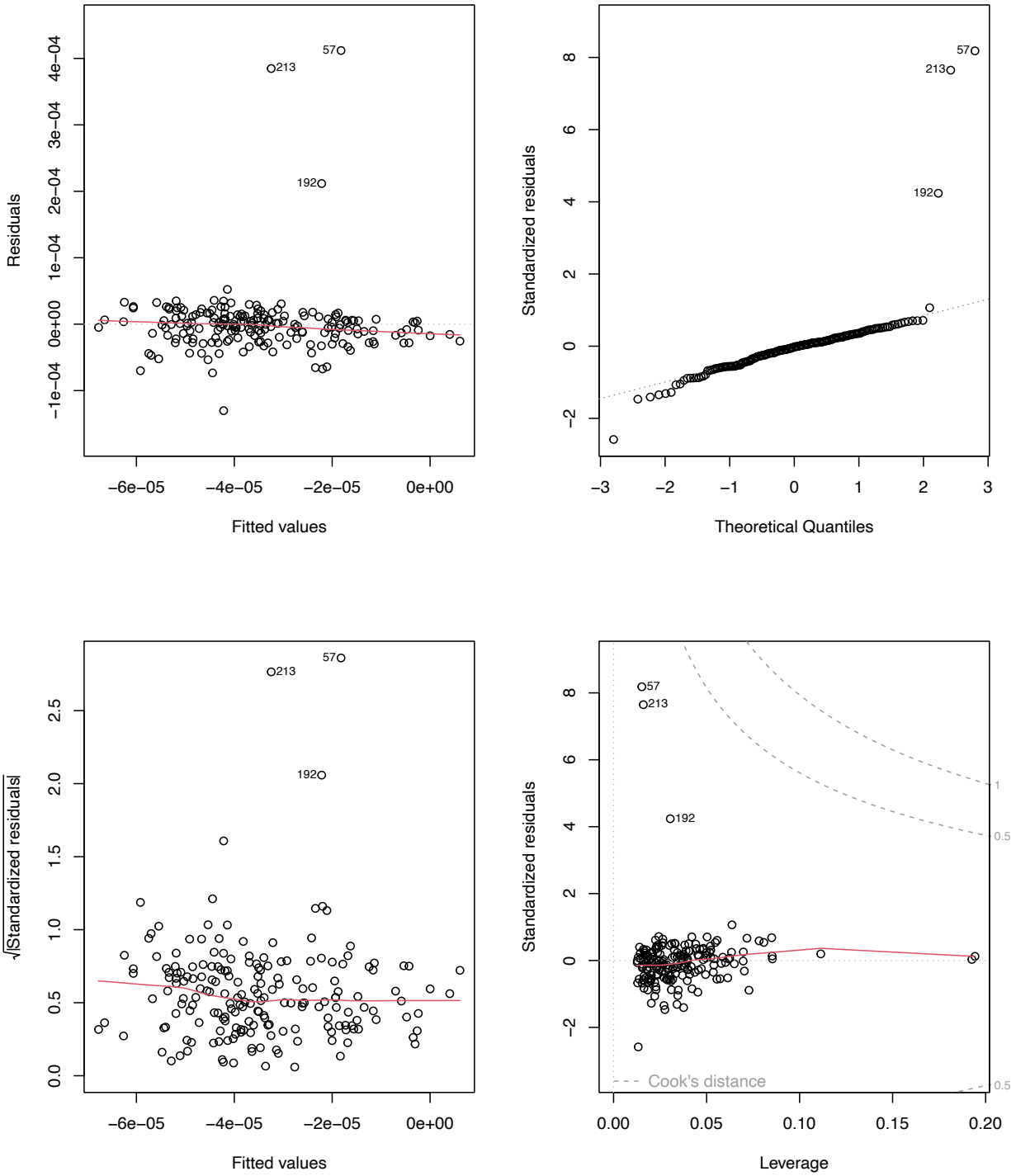
GPP Model Diagnostics



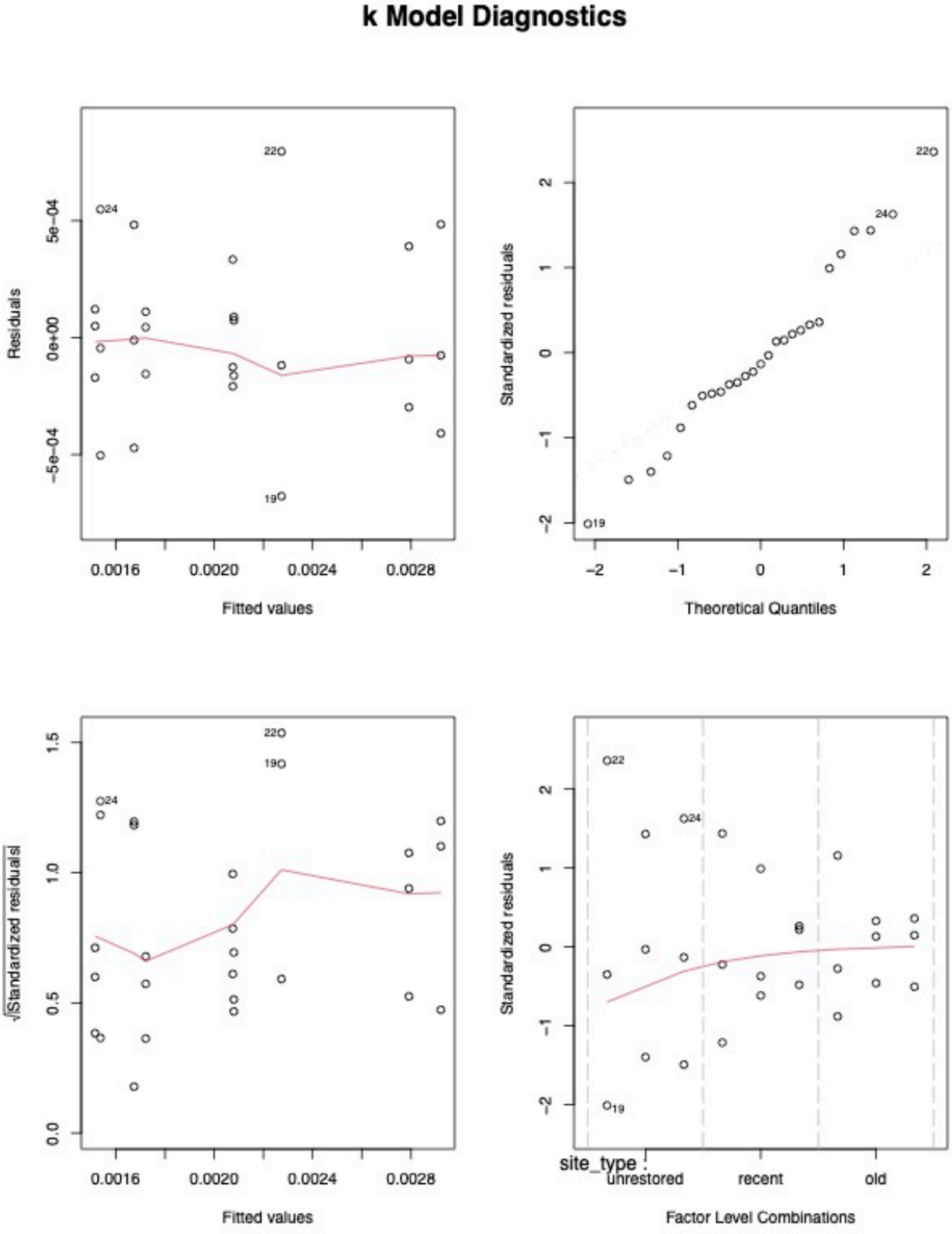
ER Model Diagnostics



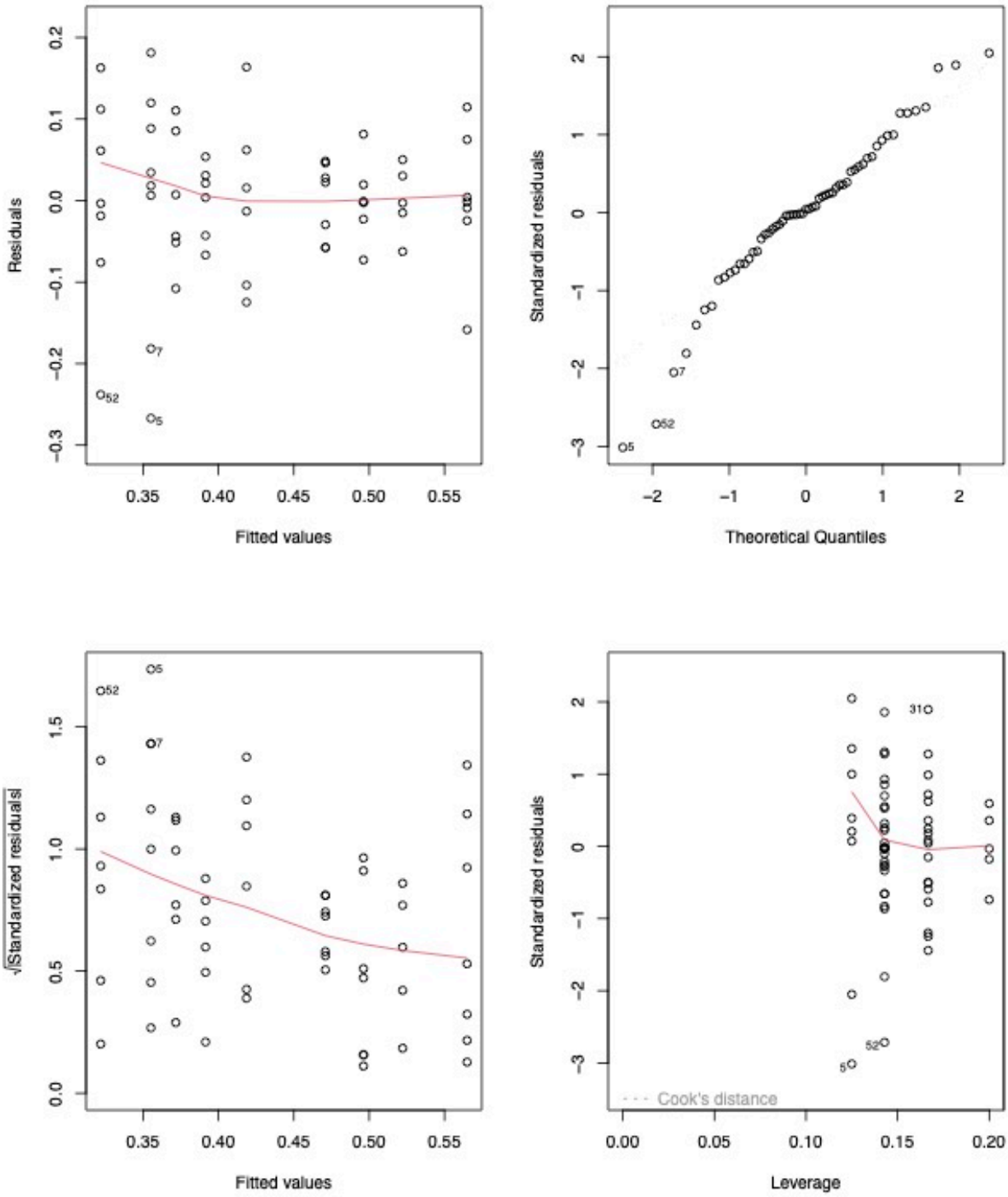
CH4 Model Diagnostics



Appendix F – Diagnostic Plots for Litter Decay & Mass Lost



Proportion Mass Lost Model Diagnostics



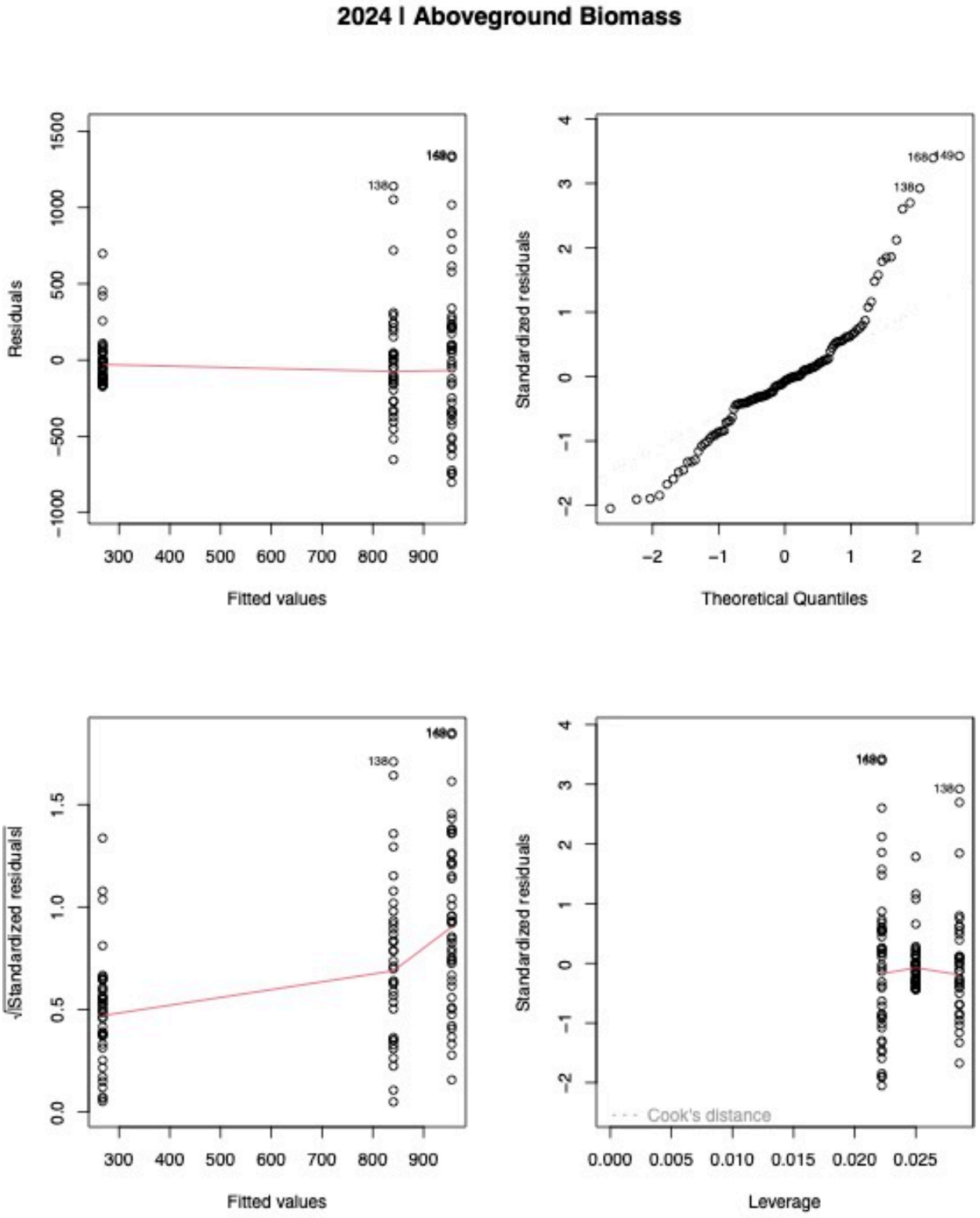
Appendix G – Collection Dates for Chapter 3

Table G.1 List of sites and summary of important sampling dates for soil core collection, soil ingrowth core installation and collection, and aboveground biomass collection. Sites starting with "t" are unrestored turfgrass sites, sites starting with "rm" are recent meadow sites and sites starting with "om" are old meadow sites. Cells are empty if we did not sample at that site.*

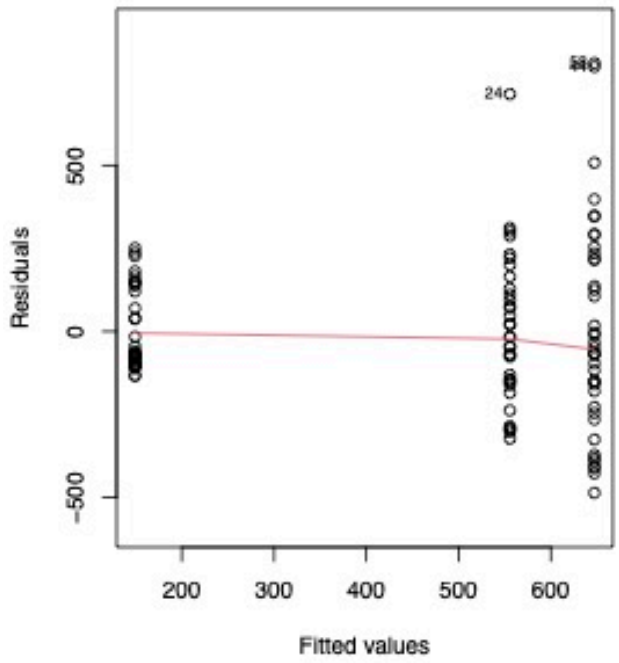
Site Name	2024		2025		
	Soil Core Collection	Aboveground Biomass Collection	Soil Core Collection / Soil Ingrowth Core Installation	Soil Ingrowth Core Collection	Aboveground Biomass Collection
t1	Jun 18, 2024	Sept 3, 2024	May 14, 2025	Aug 14, 2025	Aug 14, 2025
t2	Jun 18, 2024	Sept 3, 2024	May 15, 2025	Aug 11, 2025	Aug 11, 2025
t3	Jun 13, 2024	Sept 3, 2024	May 16, 2025	Aug 11, 2025	Aug 11, 2025
t4	Jun 13, 2024	Sept 3, 2024	May 16, 2025	Aug 12, 2025	Aug 12, 2025
t5	Jun 13, 2024	Sept 5, 2024			
t6	Jun 14, 2024	Sept 5, 2024	May 15, 2025	Aug 12, 2025	Aug 12, 2025
t7	Jun 18, 2024	Sept 5, 2024	May 14, 2025	Aug 12, 2025	
t8	Jun 14, 2024	Sept 5, 2024	May 14, 2025	Aug 12, 2025	Aug 12, 2025
tb			May 14, 2025	Aug 11, 2025	Aug 11, 2025
rm1	Jun 18, 2024	Aug 29, 2024	May 16, 2025	Aug 11, 2025	Aug 11, 2025
rm2	Jun 18, 2024	Aug 29, 2024	May 15, 2025	Aug 14, 2025	Aug 14, 2025
rm3	Jun 18, 2024	Aug 29, 2024	May 14, 2025	Aug 14, 2025	Aug 14, 2025
rm4	Jun 18, 2024	Aug 29, 2024	May 14, 2025	Aug 14, 2025	Aug 14, 2025
rm5	Jun 13, 2024	Aug 28, 2024	May 15, 2025	Aug 14, 2025	Aug 14, 2025
rm6	Jun 14, 2024	Aug 27, 2024	May 16, 2025	Aug 12, 2025	Aug 12, 2025
rm7	Jun 14, 2024	Aug 27, 2024	May 16, 2025	Aug 11, 2025	Aug 11, 2025
rm8	Jun 14, 2024	Sept 12, 2024	May 20, 2025	Aug 12, 2025	Aug 12, 2025
om1	Jun 13, 2024	Aug 28, 2024	May 15, 2025	Aug 12, 2025	Aug 12, 2025
om2	Jun 13, 2024	Aug 28, 2024	May 14, 2025	Aug 11, 2025	Aug 11, 2025
om3	Jun 13, 2024	Aug 28, 2024	May 16, 2025	Aug 12, 2025	Aug 12, 2025
om4	Jun 13, 2024	Aug 28, 2024	May 16, 2025	Aug 12, 2025	Aug 12, 2025
om5	Jun 13, 2024	Aug 28, 2024	May 14, 2025	Aug 11, 2025	Aug 11, 2025
om6	Jun 13, 2024	Aug 27, 2024	May 15, 2025	Nov 20, 2025	Aug 12, 2025
om7	Jun 25, 2024	Aug 27, 2024	May 14, 2025	Aug 11, 2025	Aug 11, 2025
om8	Jun 18, 2024	Aug 27, 2024	May 16, 2025	Aug 12, 2025	

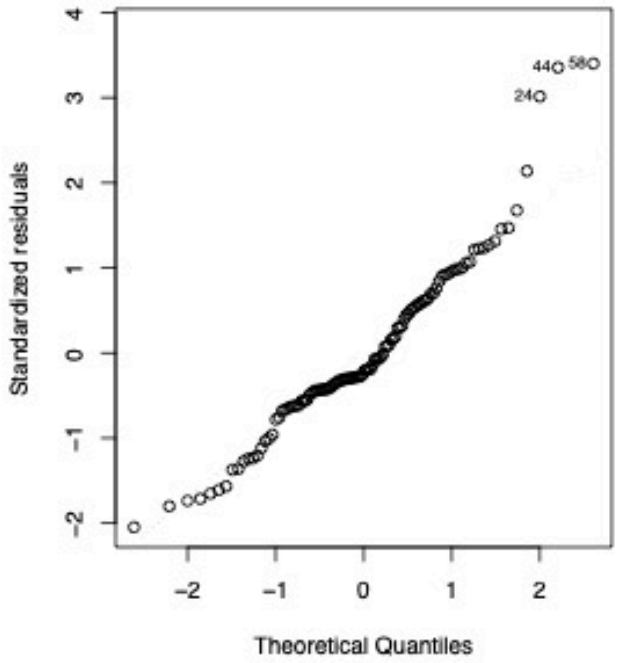
*In 2025, t5 became inaccessible due to construction, so it was replaced by tb for the second year. t7 and om8 were active sites in 2025, but were mowed unexpectedly outside of the regular TRCA maintenance regime just before aboveground biomass sampling and so we could not collect from those sites.

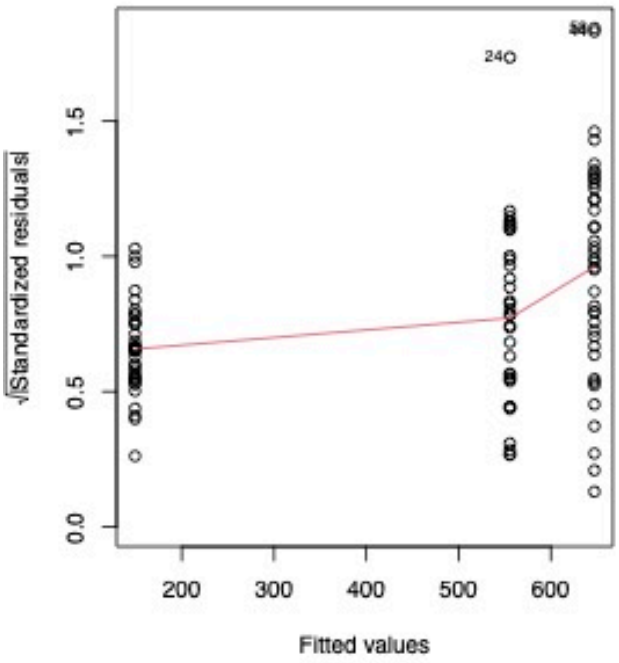
Appendix H – Diagnostic Plots for Aboveground Biomass

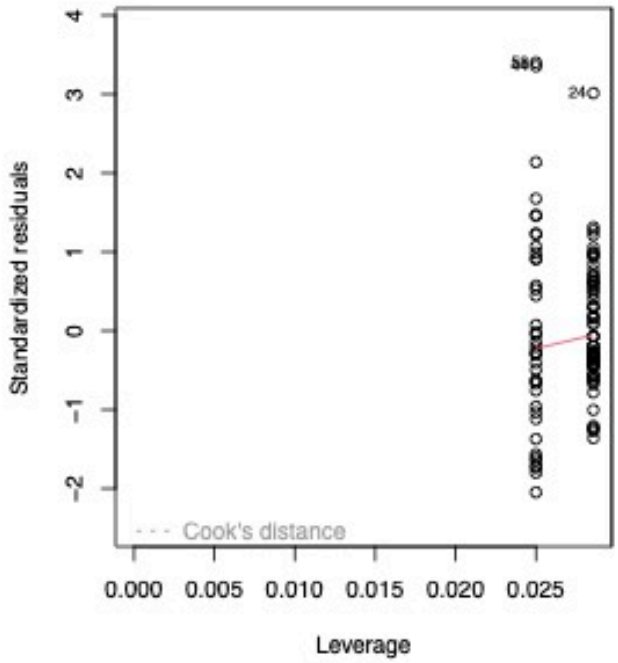


2025 | Aboveground Biomass

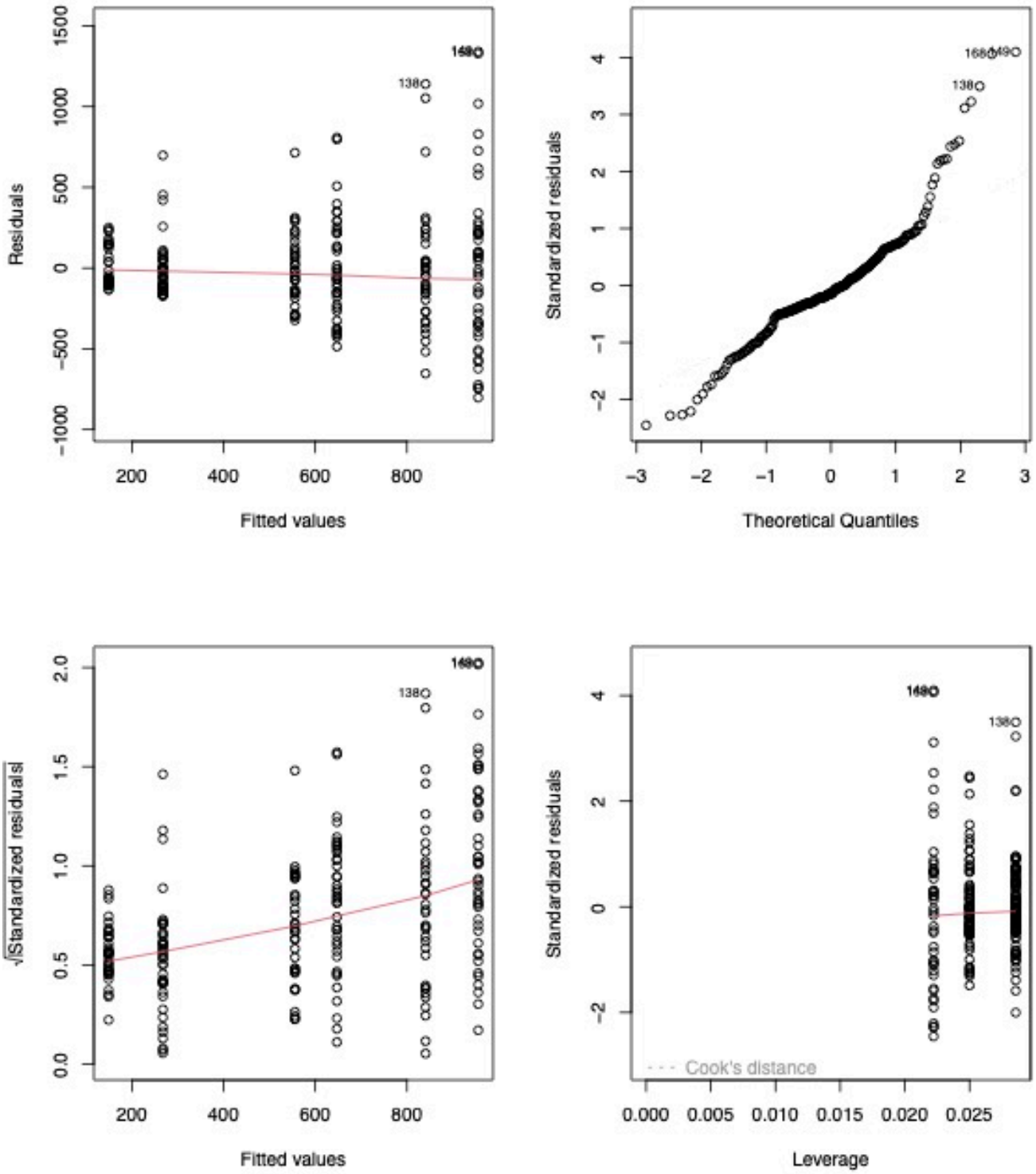




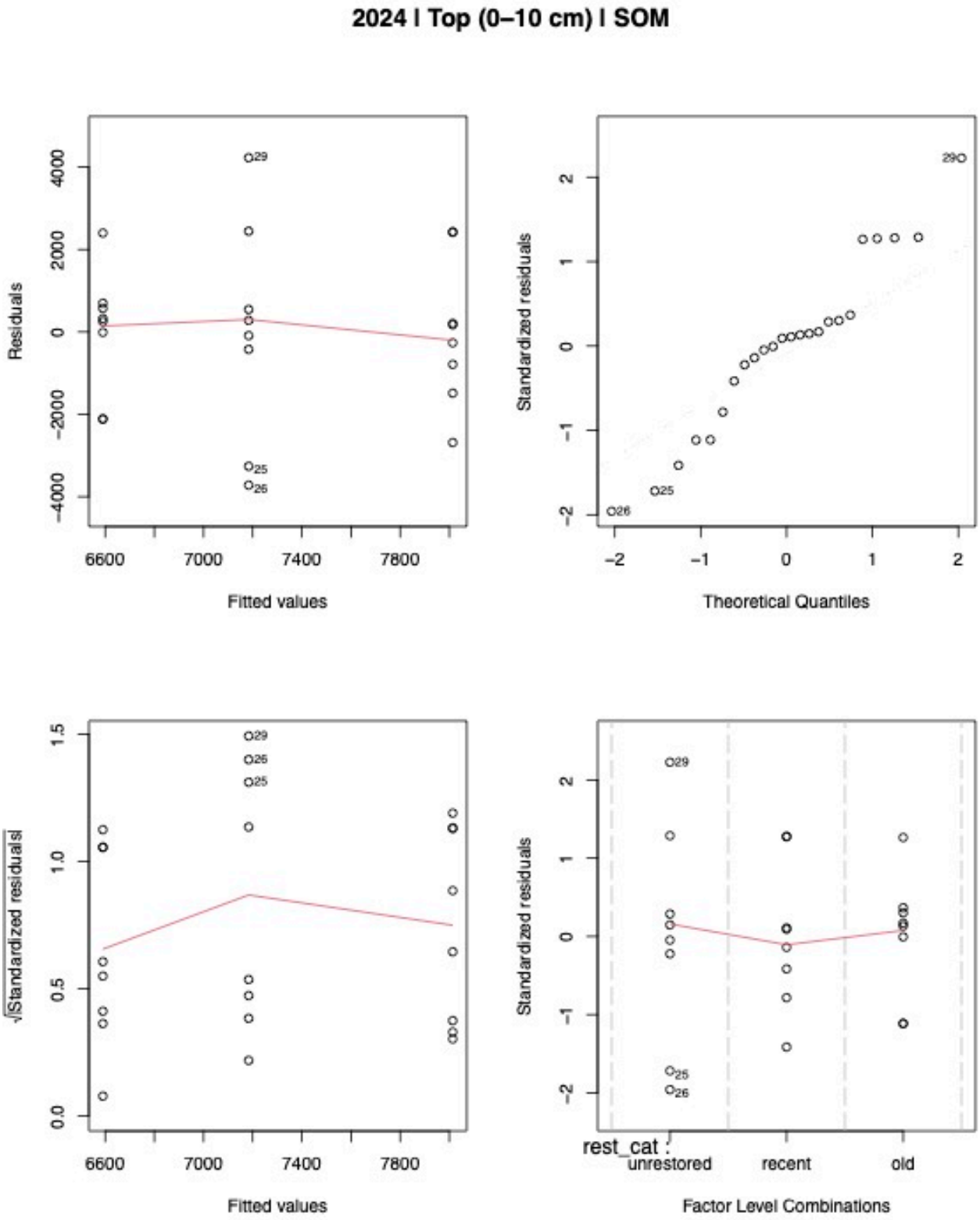




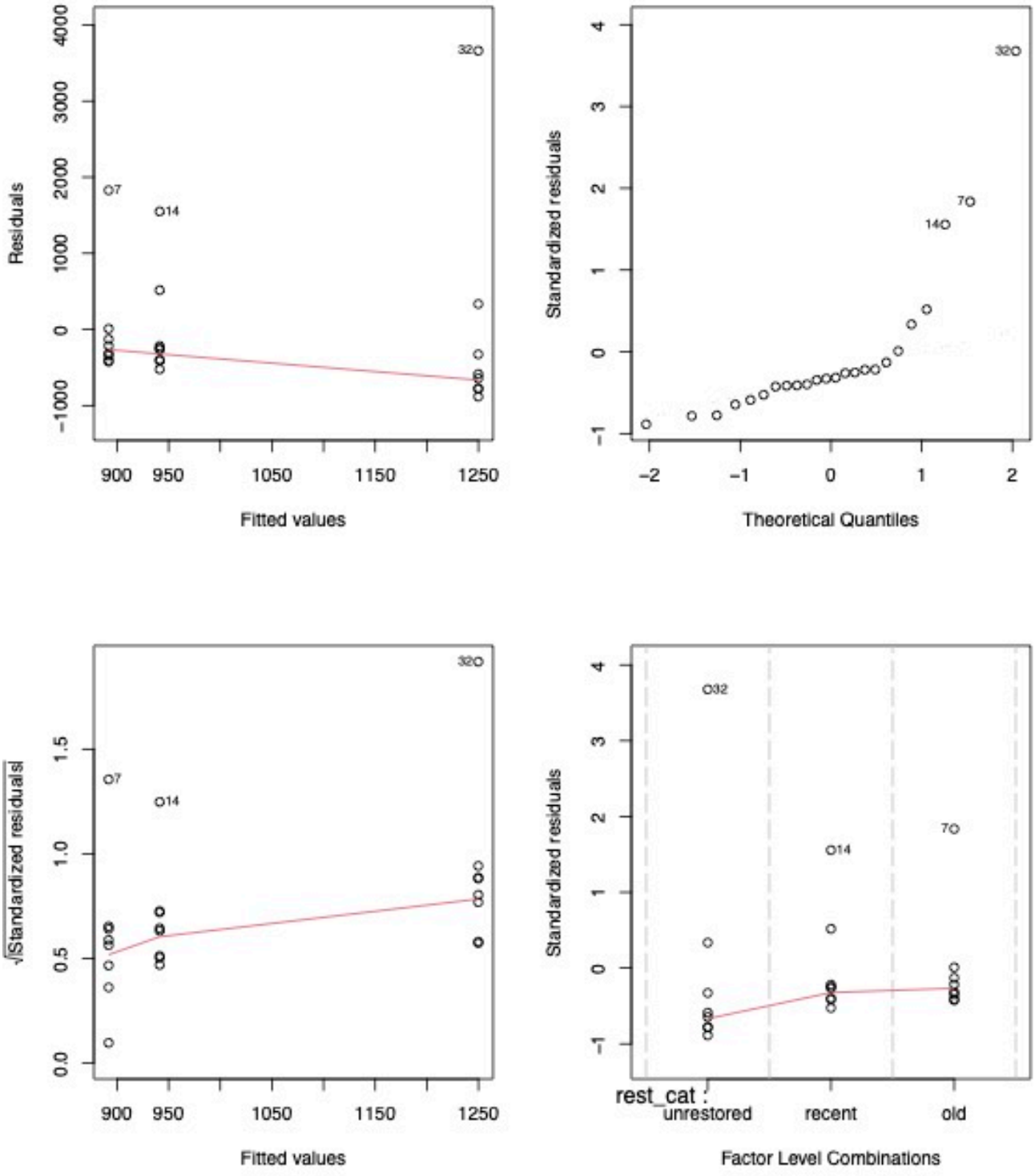
Two–Way (category x year) | Aboveground Biomass



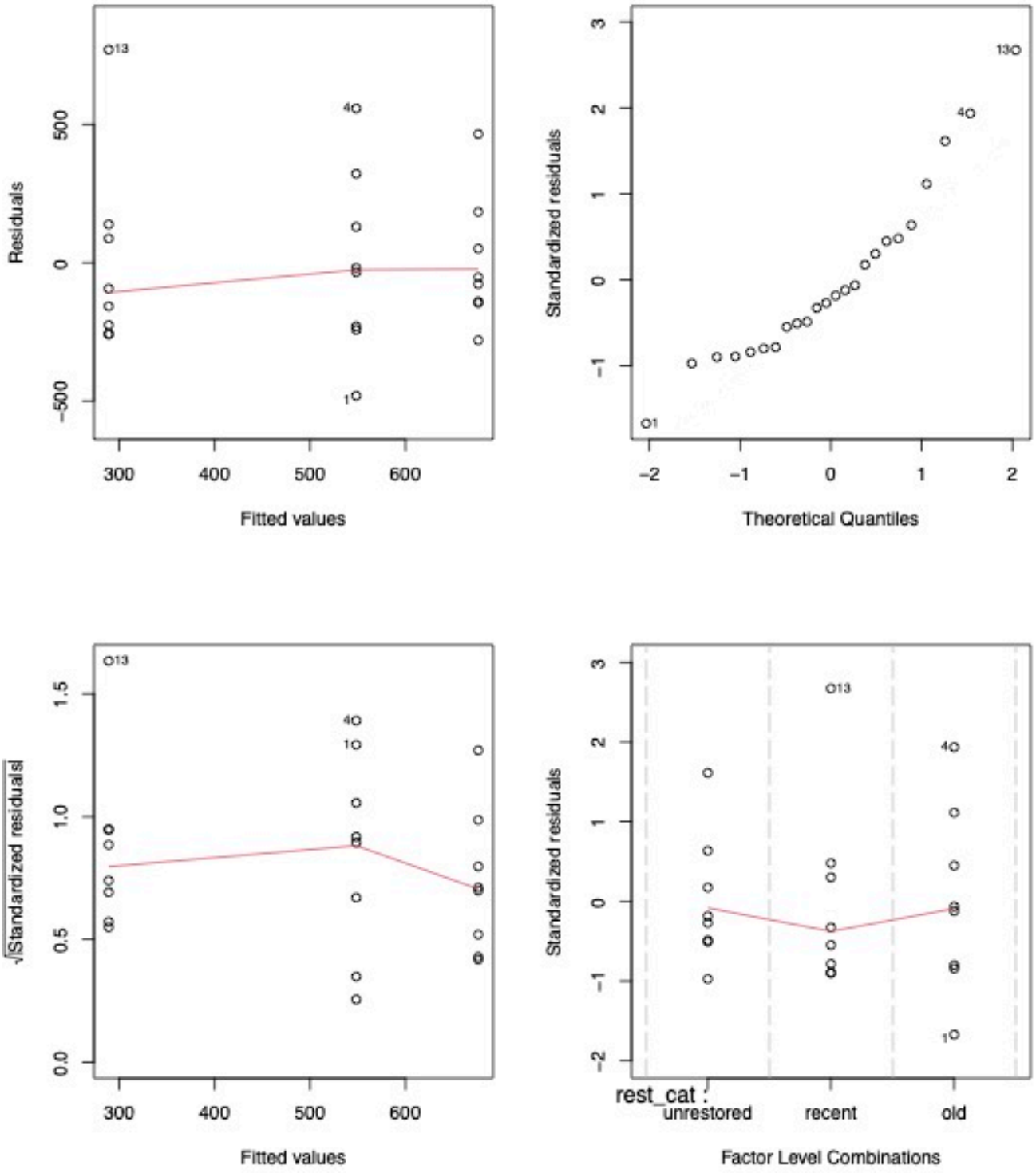
Appendix I – Diagnostic Plots for Soil Cores



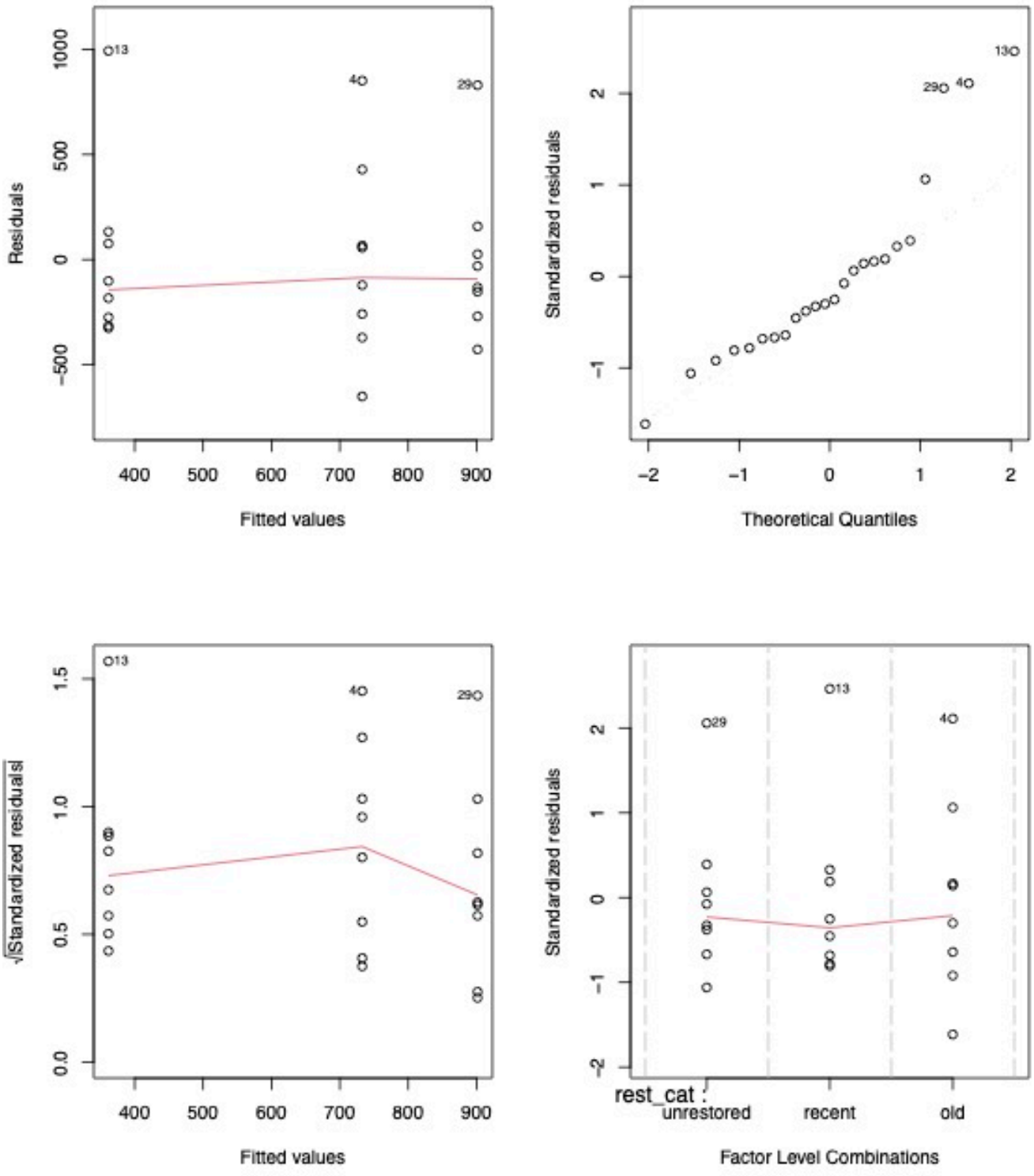
2024 | Top (0–10 cm) | SCC



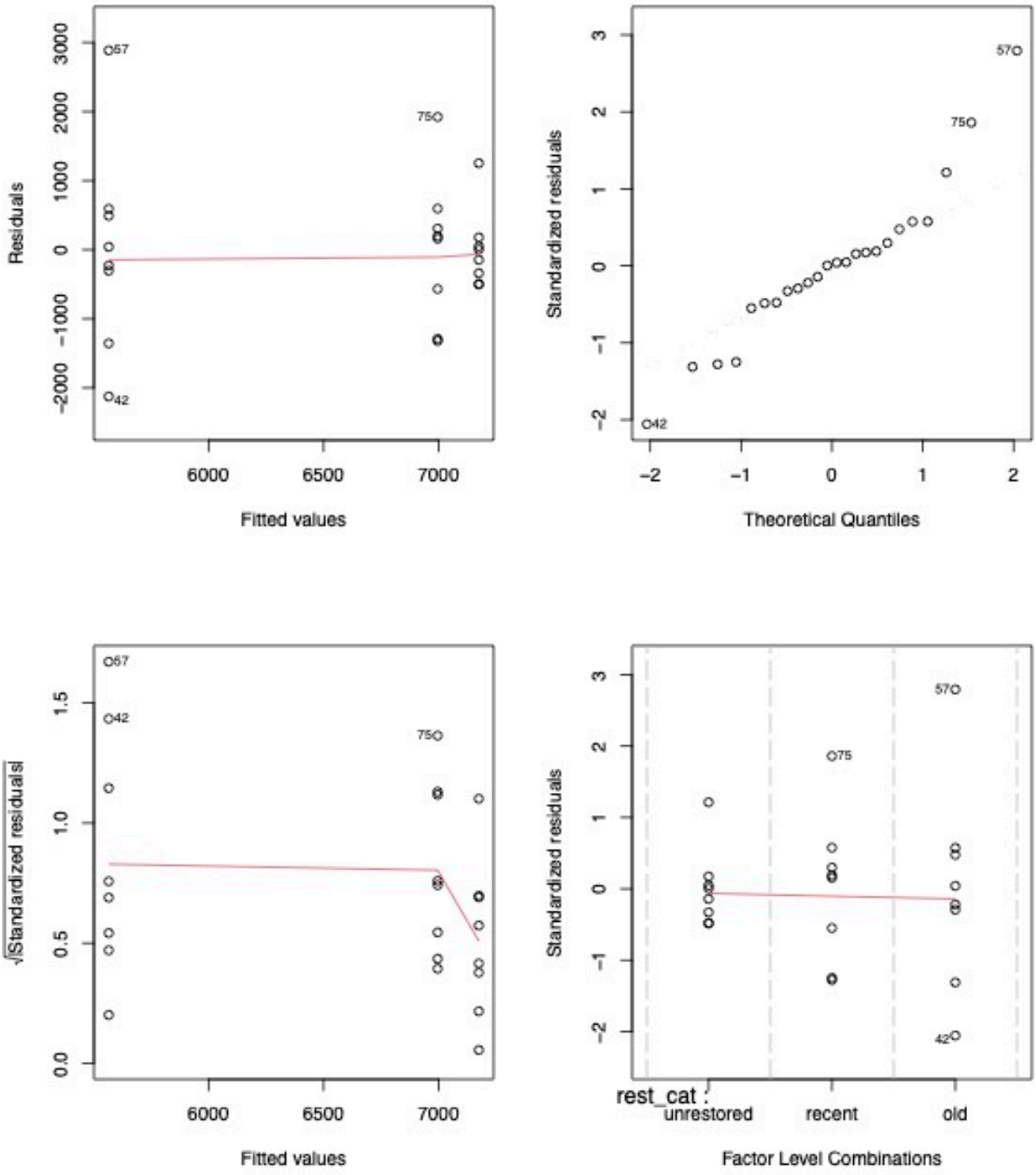
2024 | Top (0–10 cm) | ROM



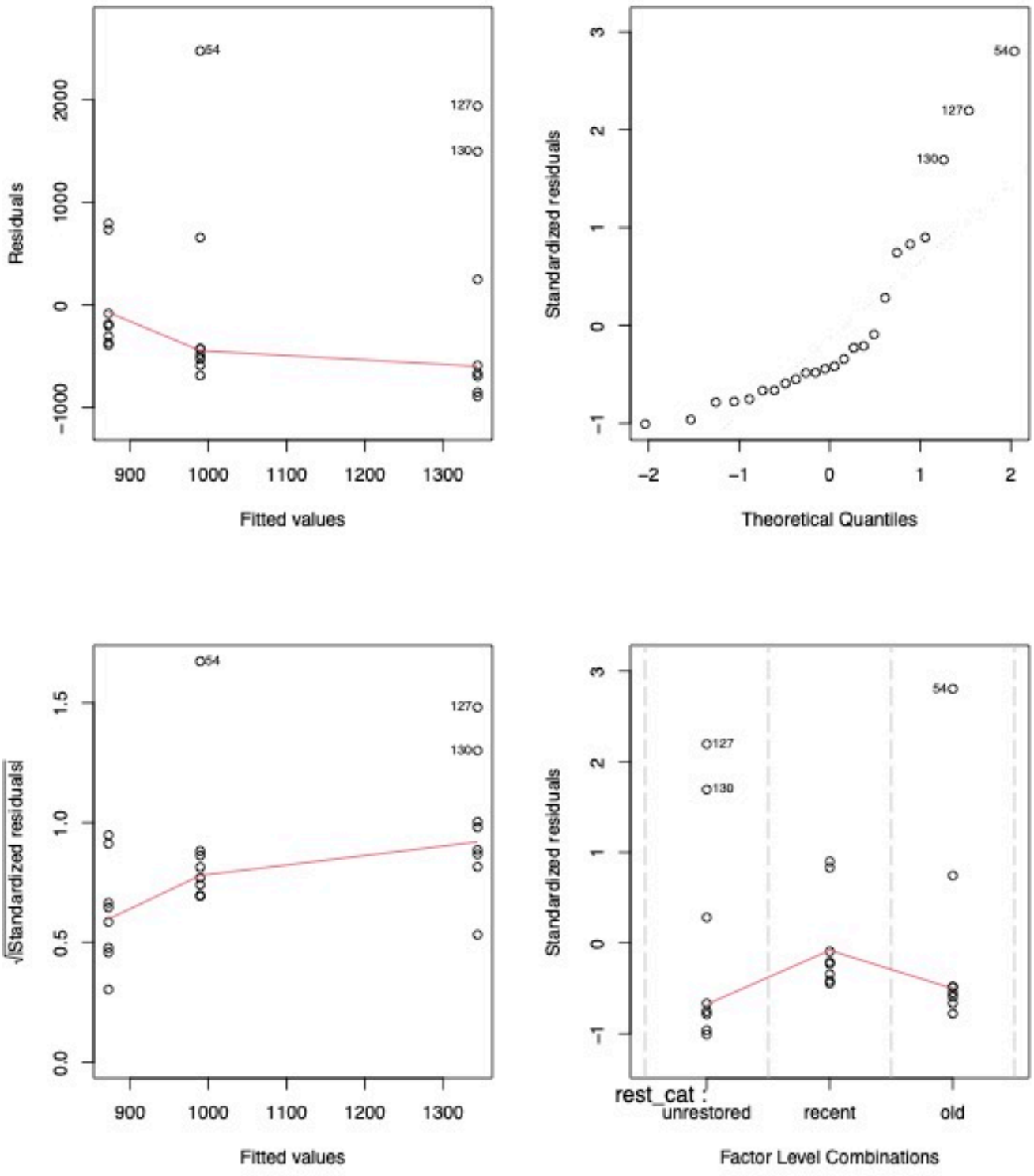
2024 | Top (0–10 cm) | Root Dry Mass



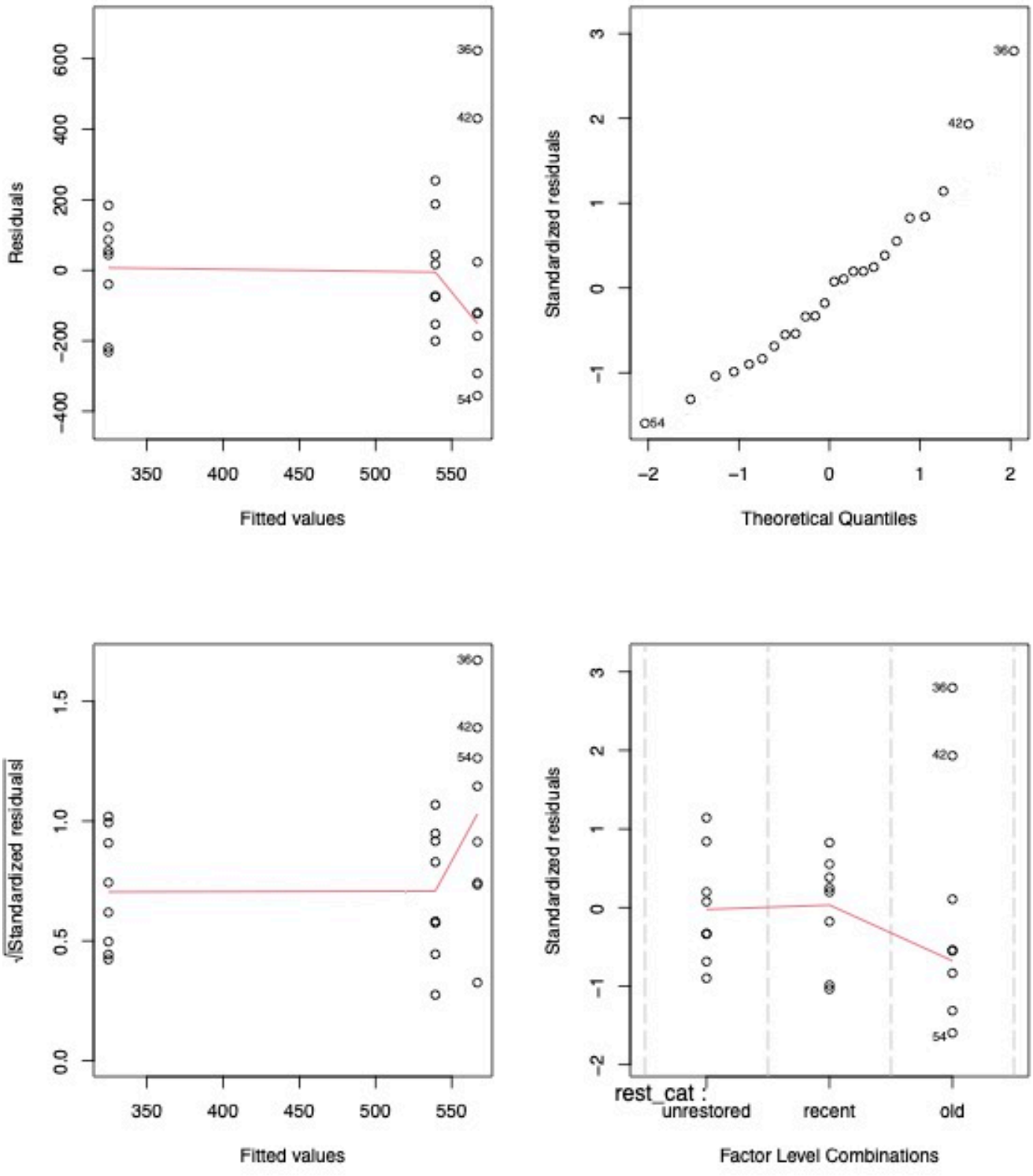
2025 | Top (0–10 cm) | SOM



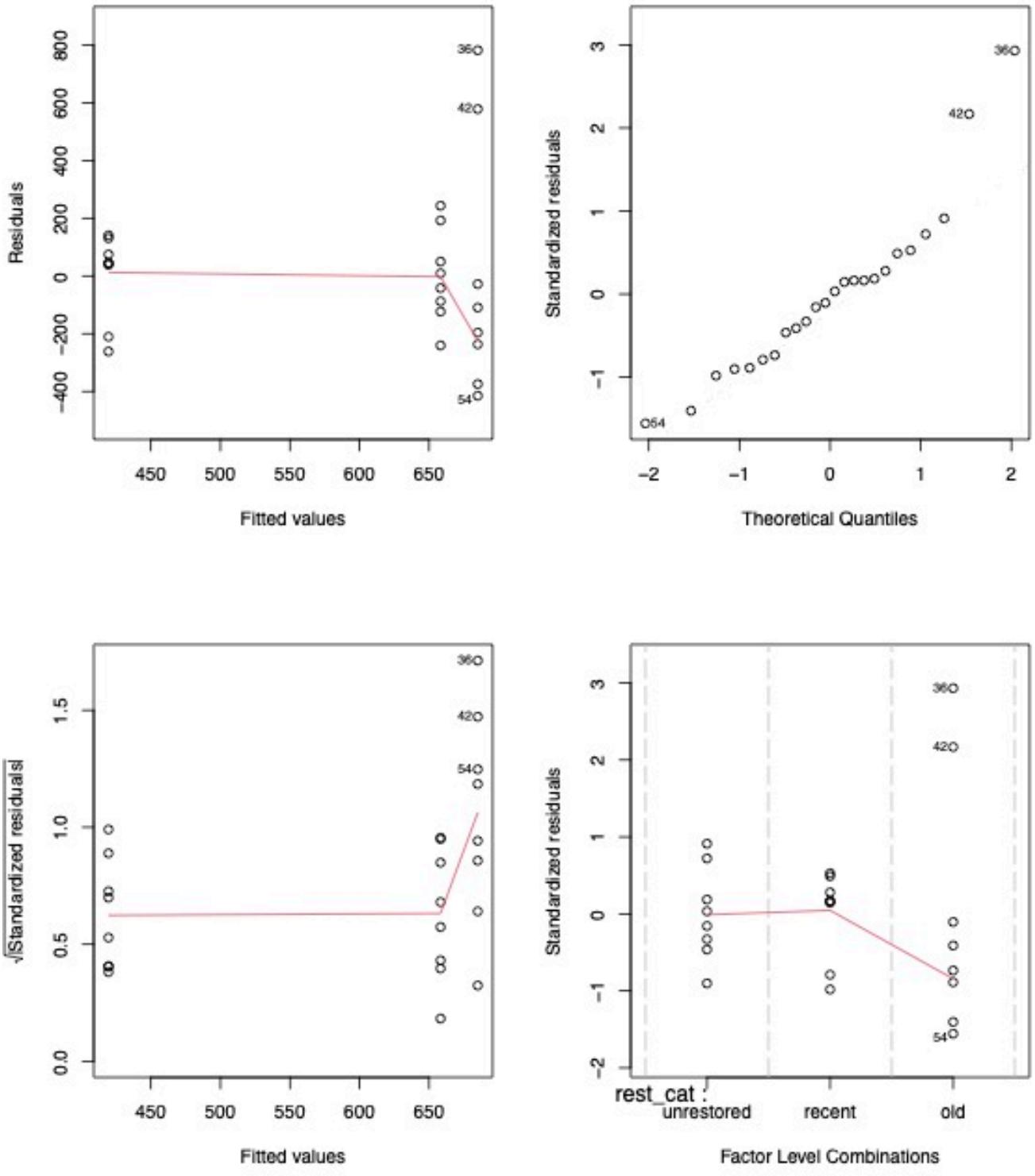
2025 | Top (0–10 cm) | SCC



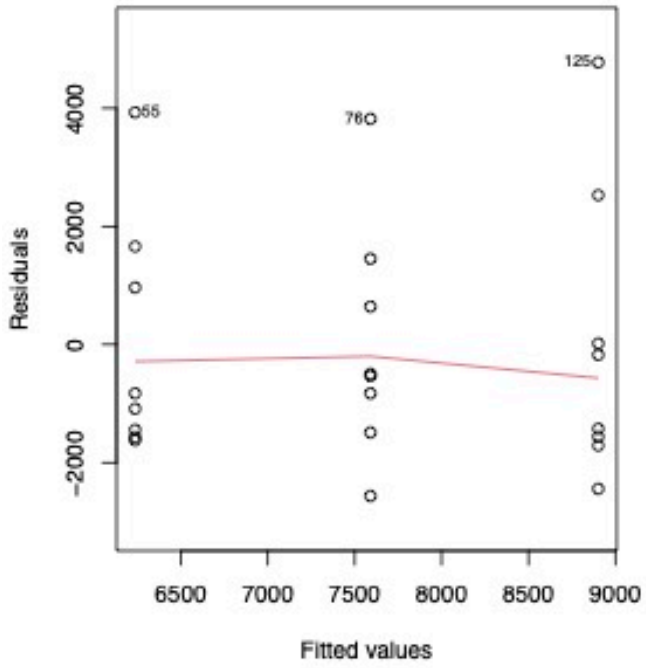
2025 | Top (0–10 cm) | ROM

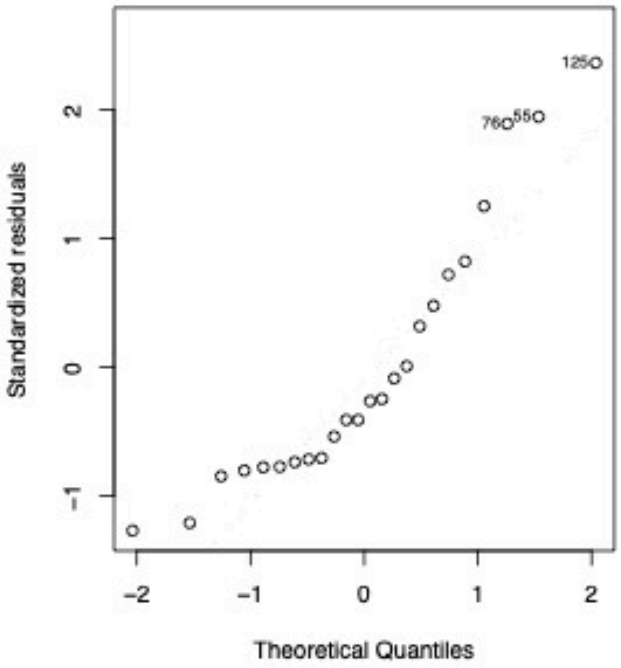


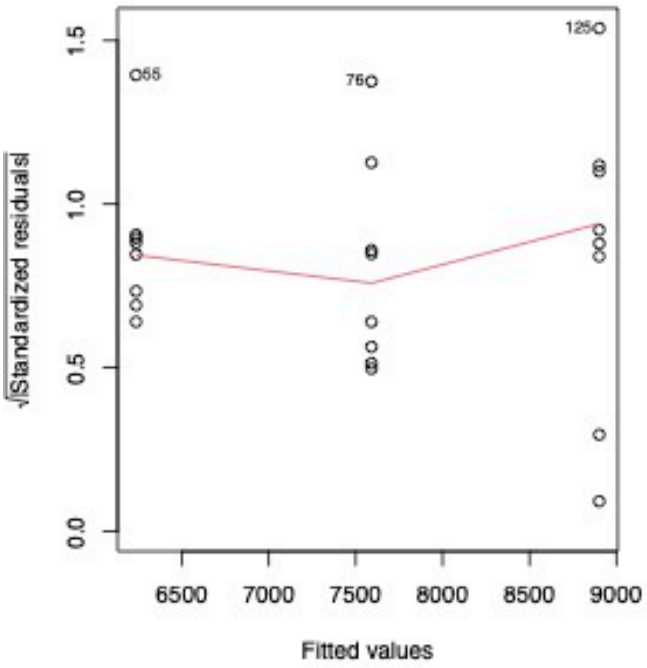
2025 | Top (0–10 cm) | Root Dry Mass

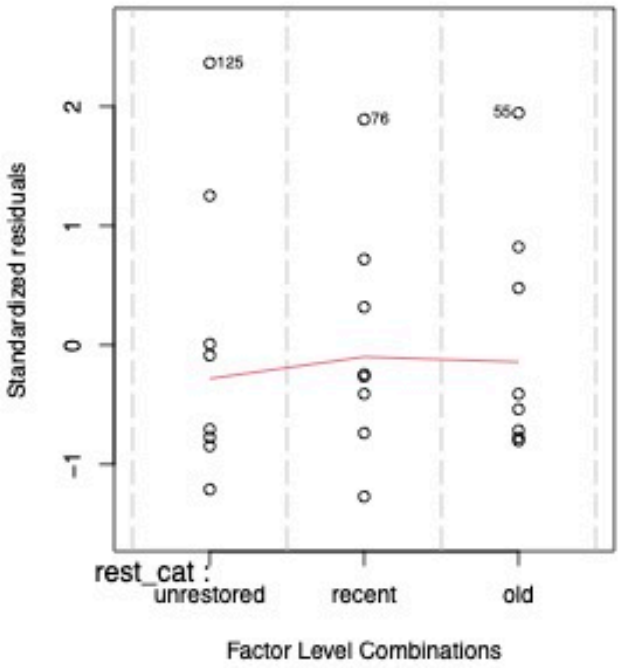


2025 | Mid (10–20 cm) | SOM

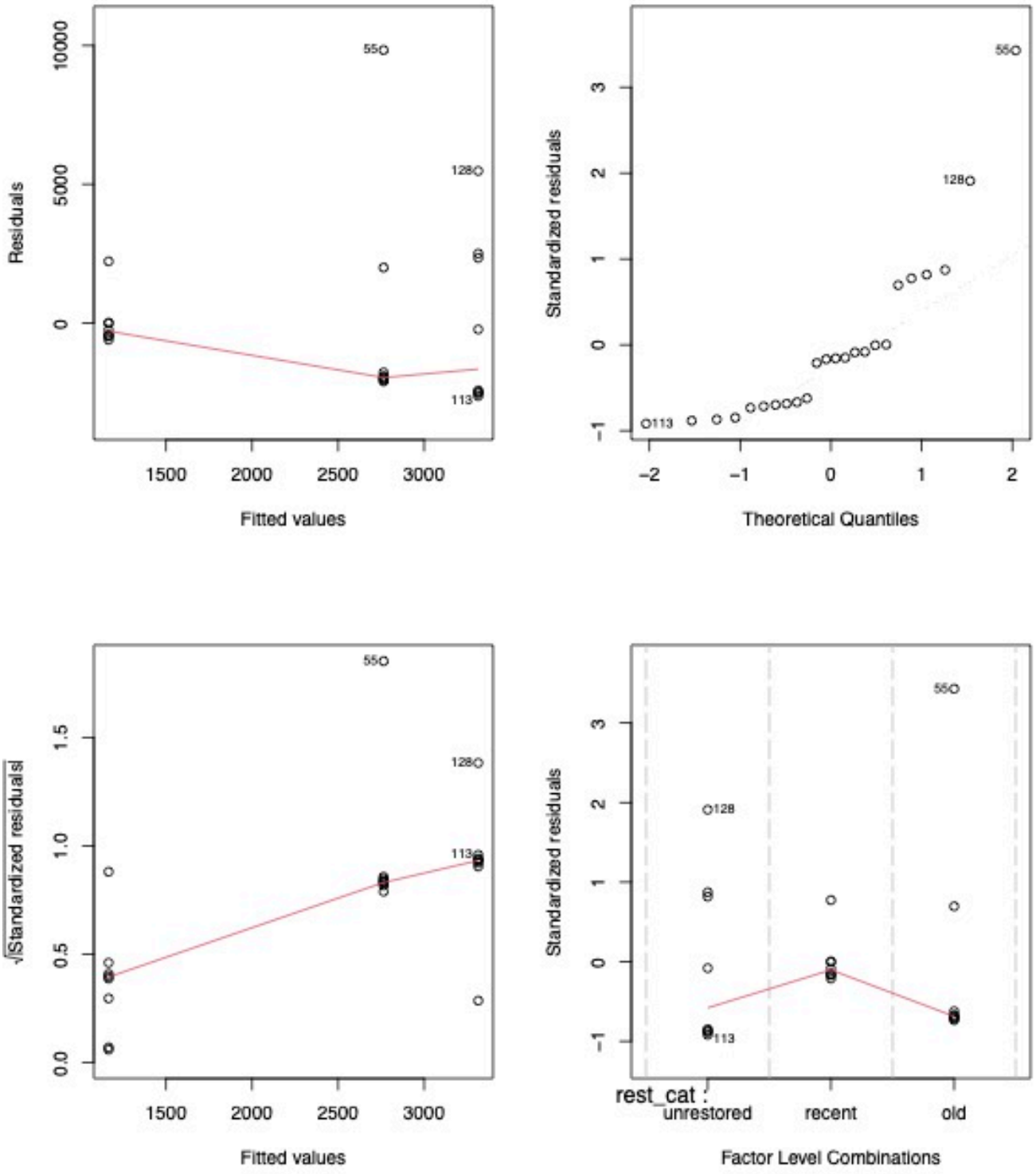




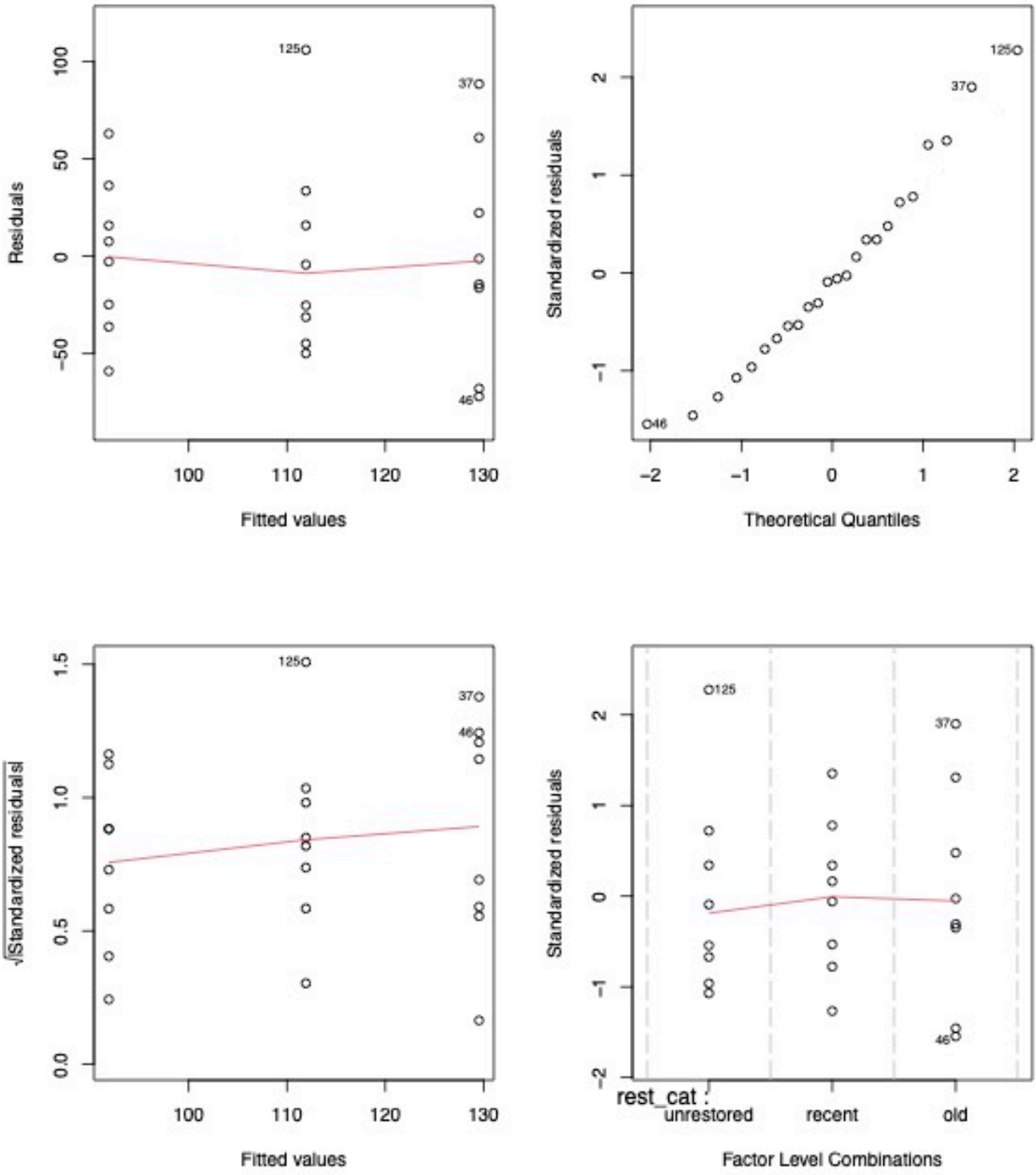




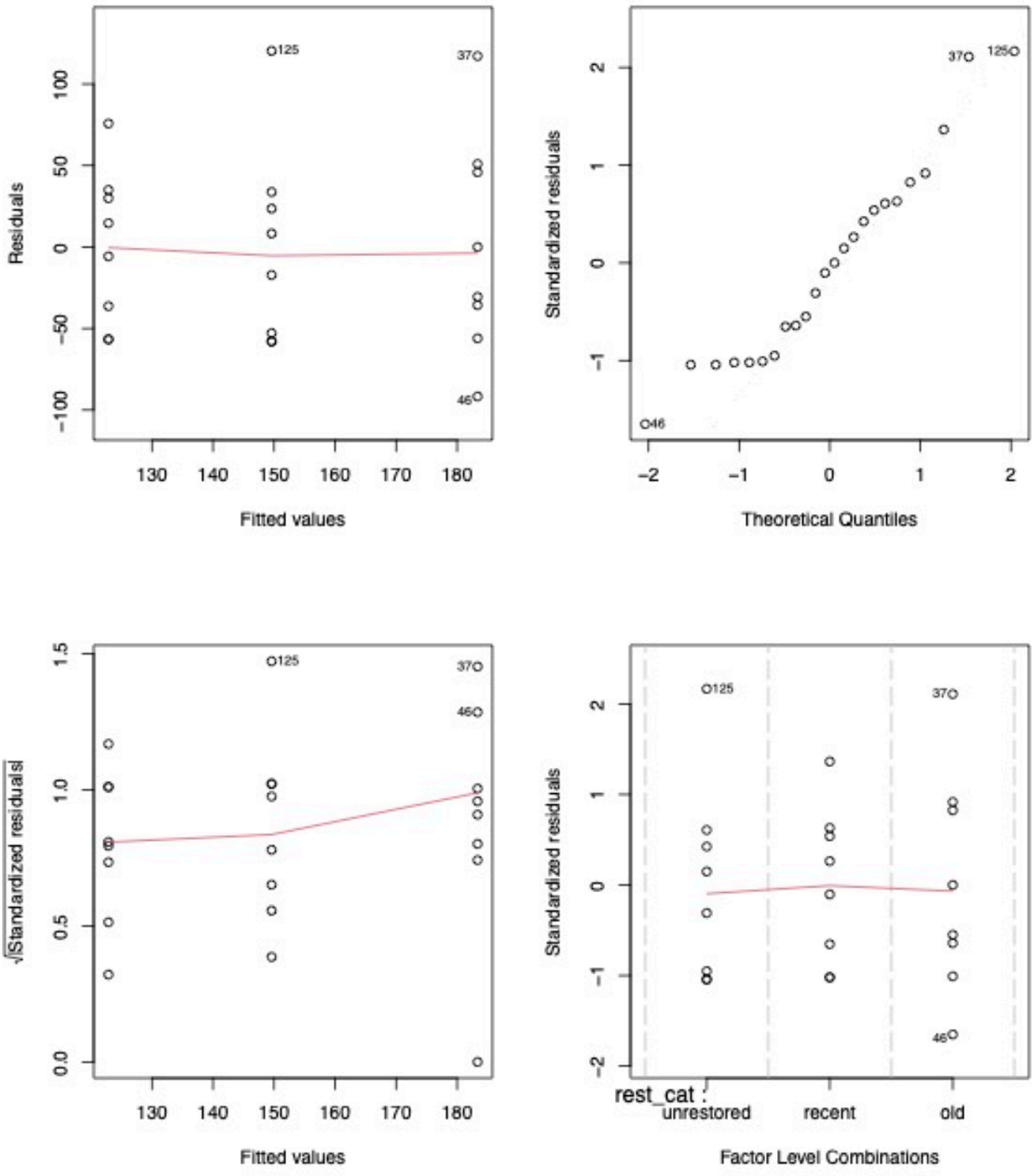
2025 | Mid (10–20 cm) | SCC



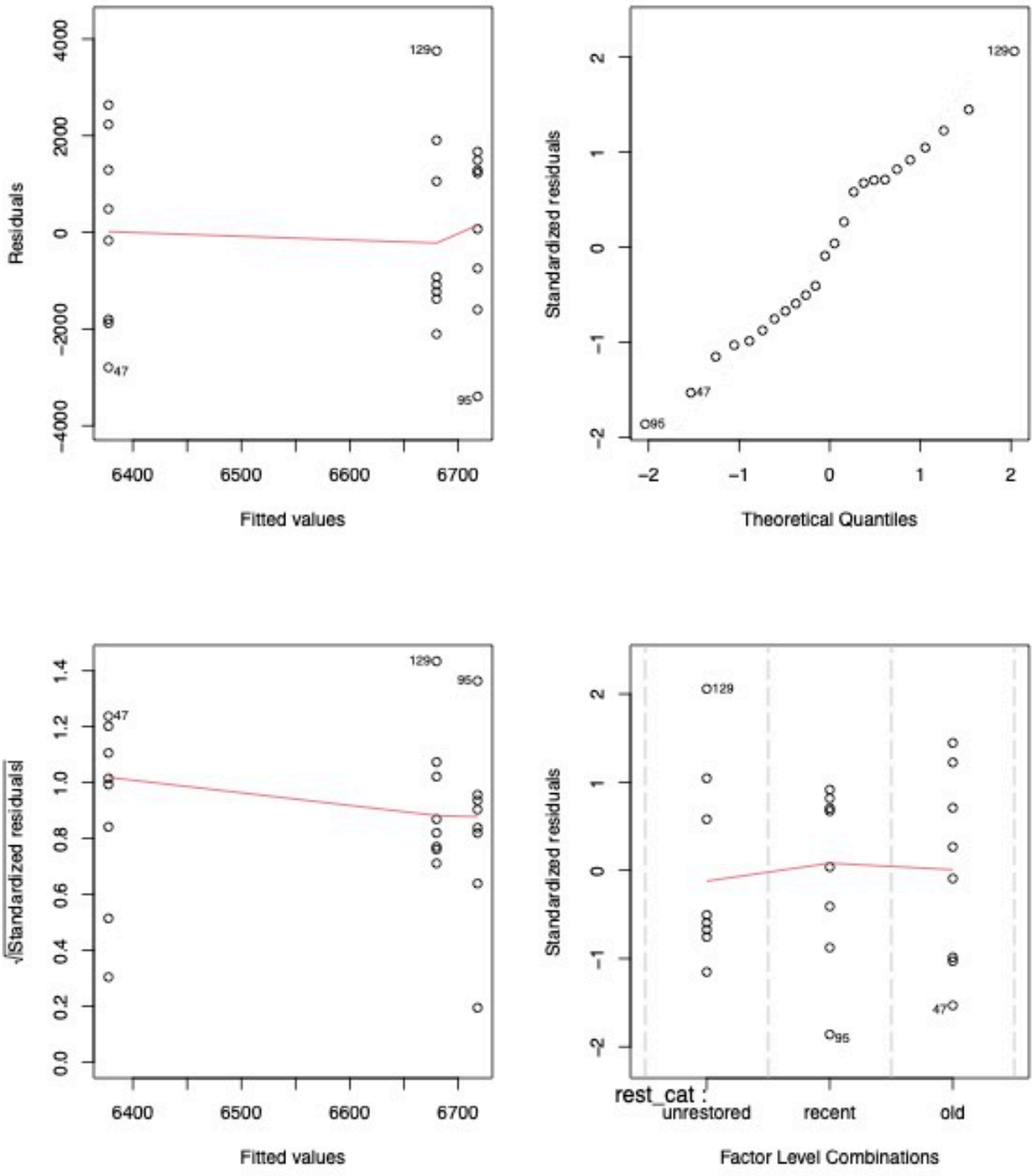
2025 | Mid (10–20 cm) | ROM



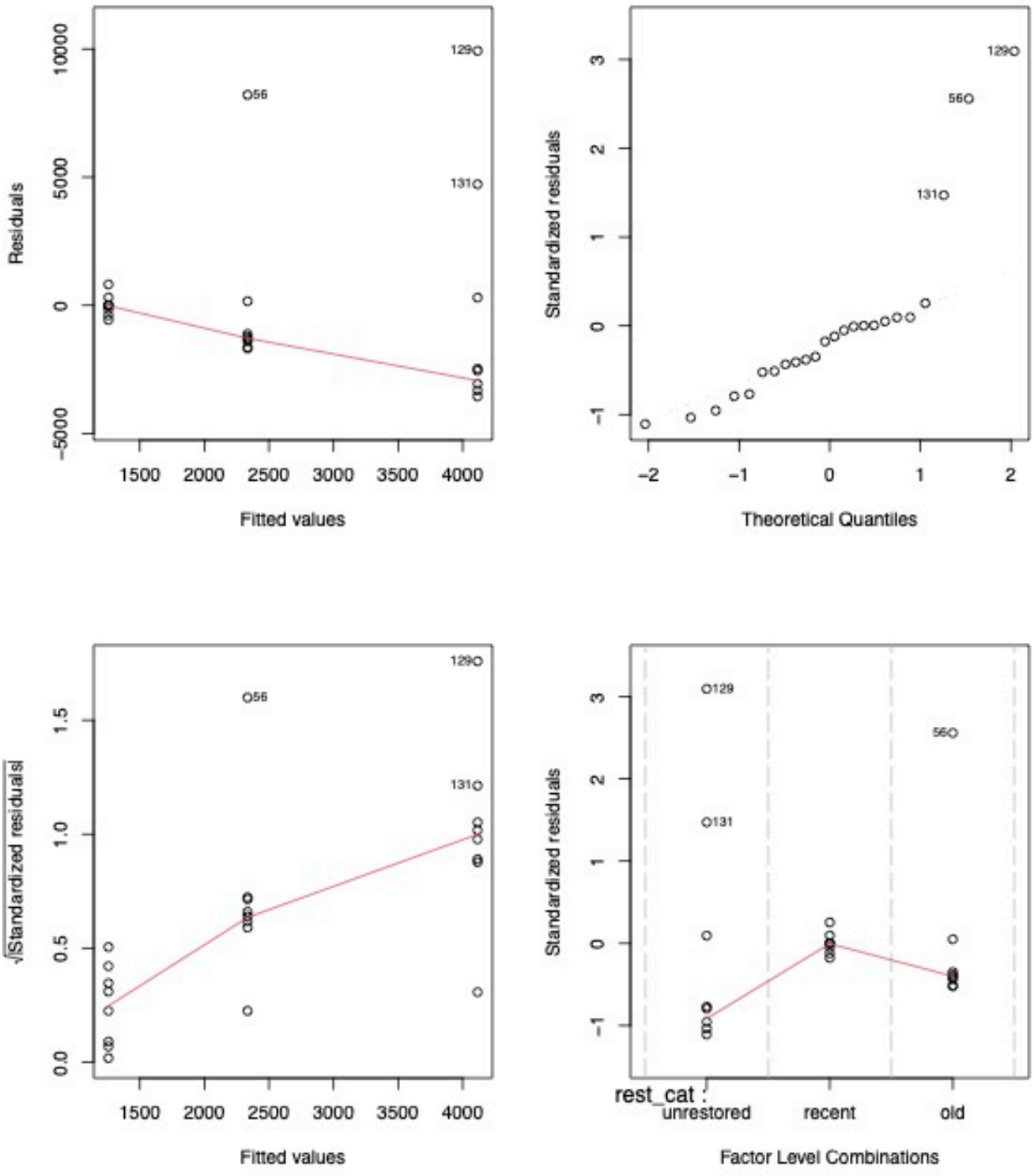
2025 | Mid (10–20 cm) | Root Dry Mass



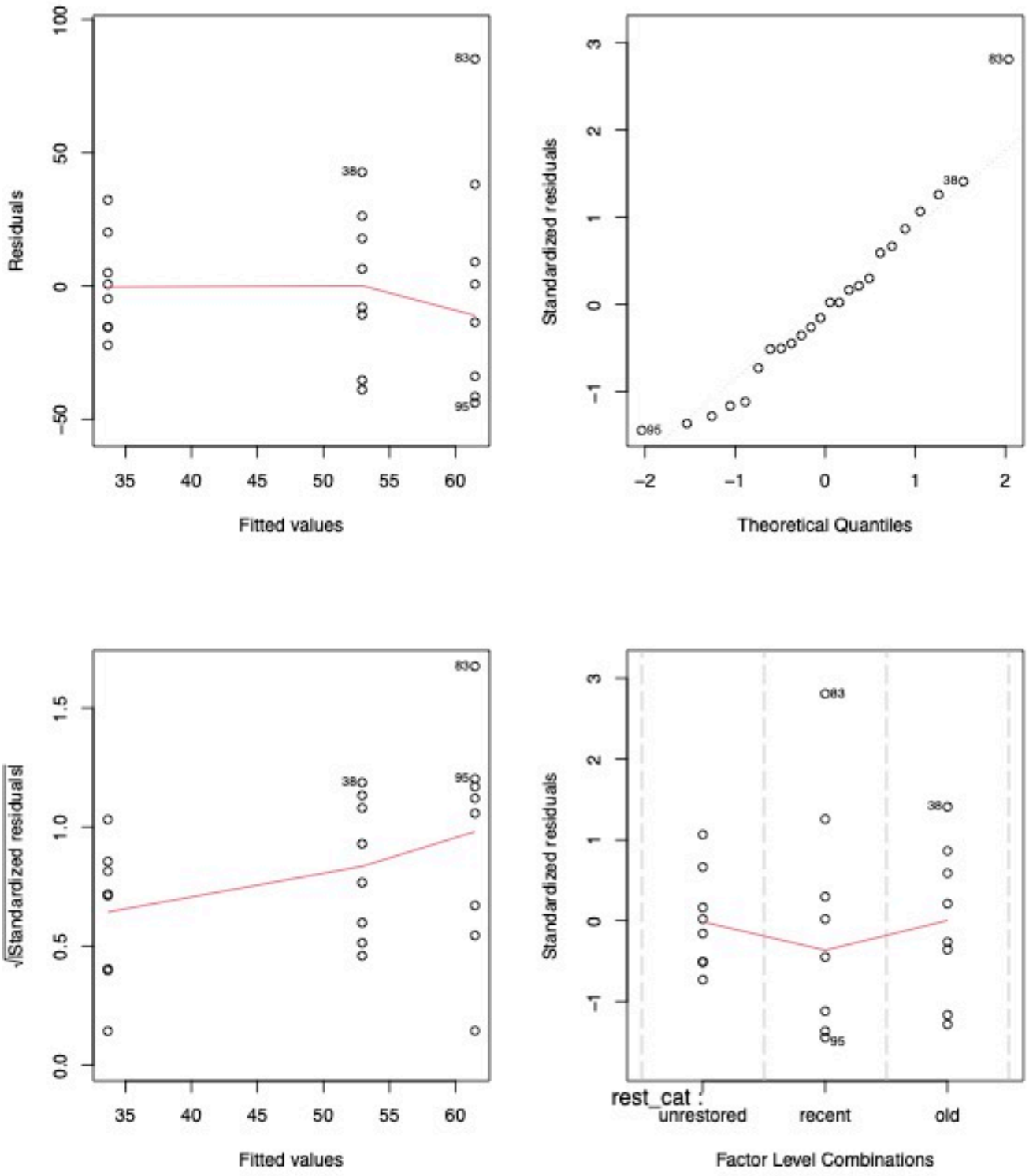
2025 | Bottom (20–30 cm) | SOM



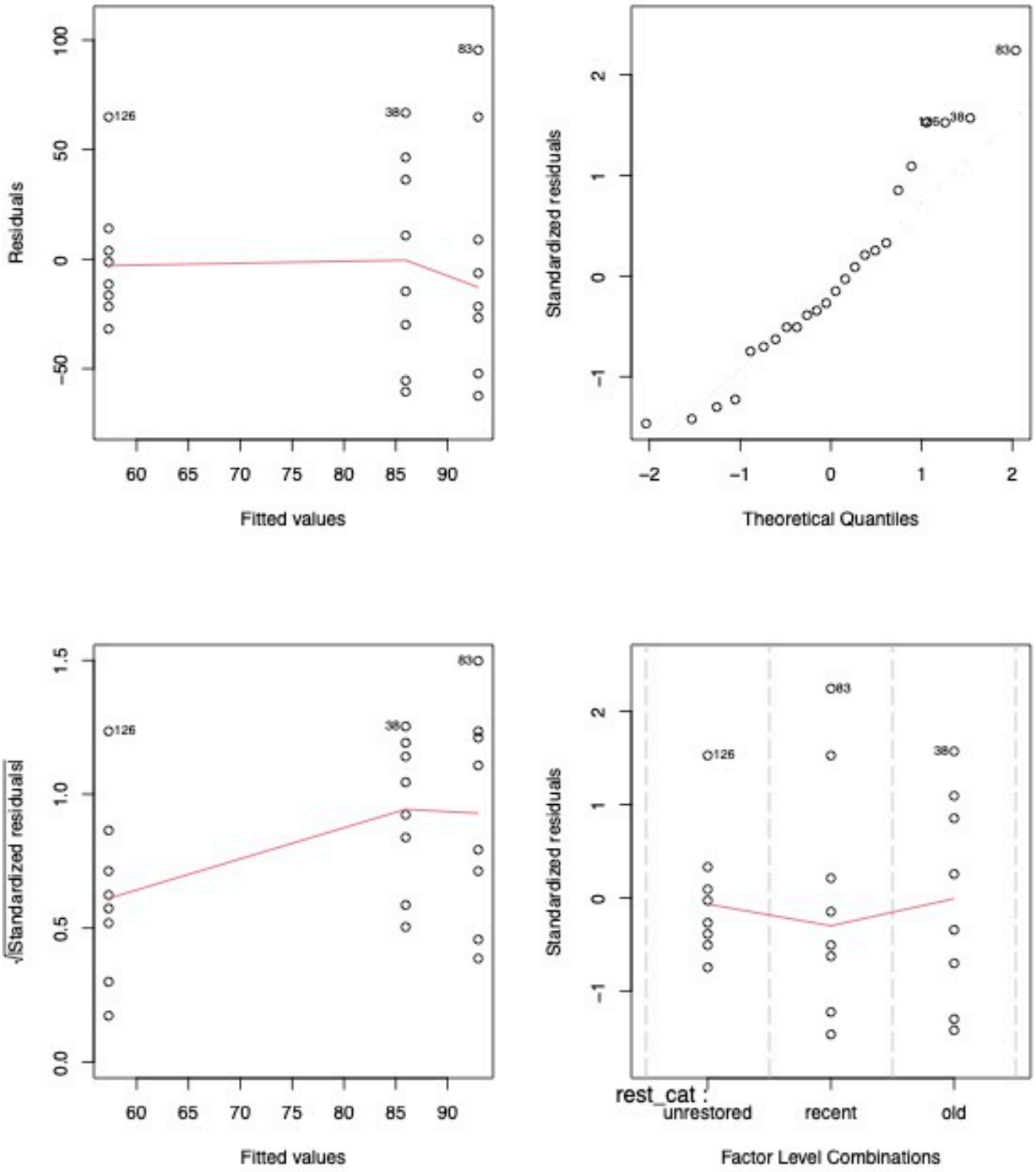
2025 | Bottom (20–30 cm) | SCC



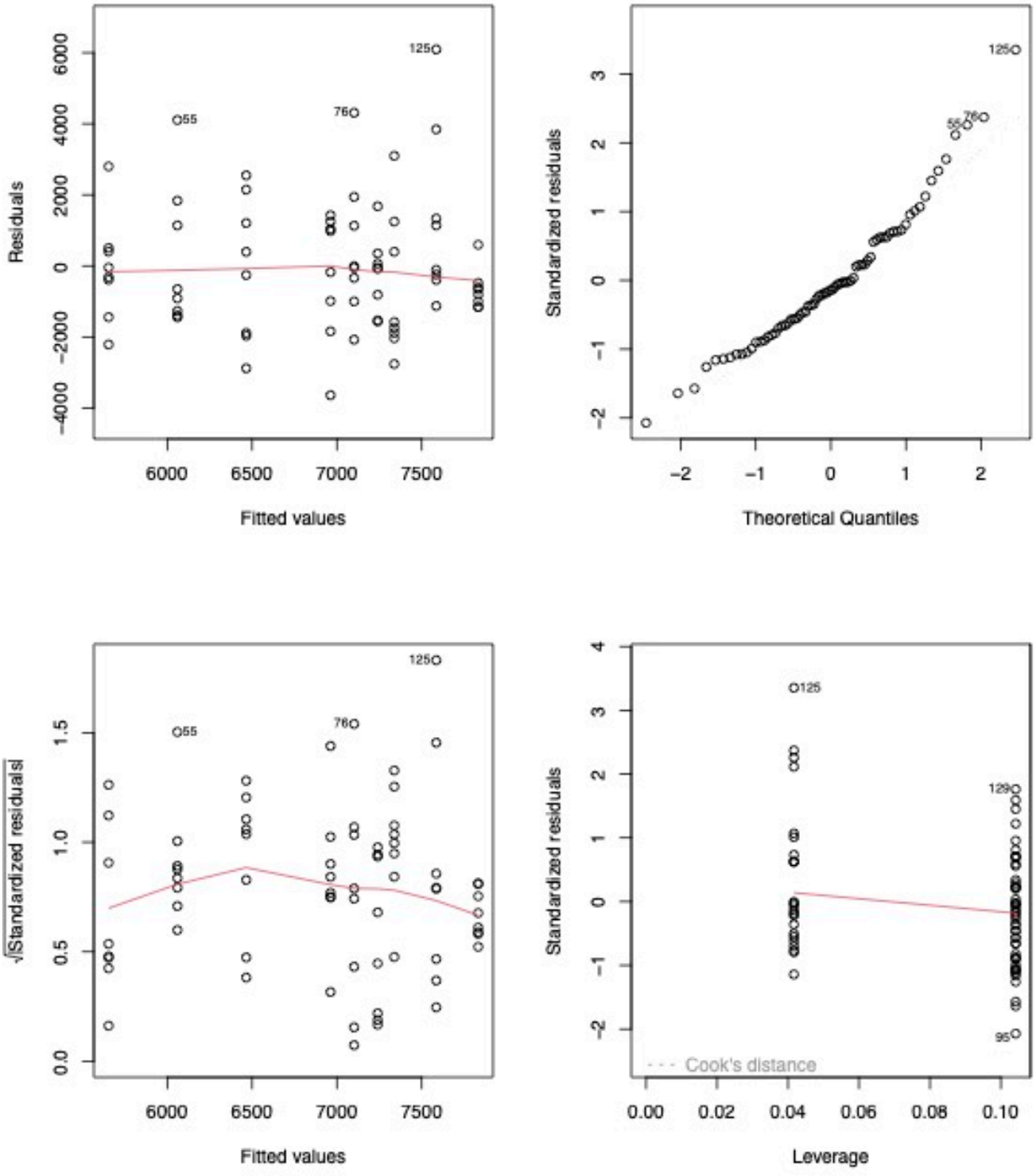
2025 | Bottom (20–30 cm) | ROM

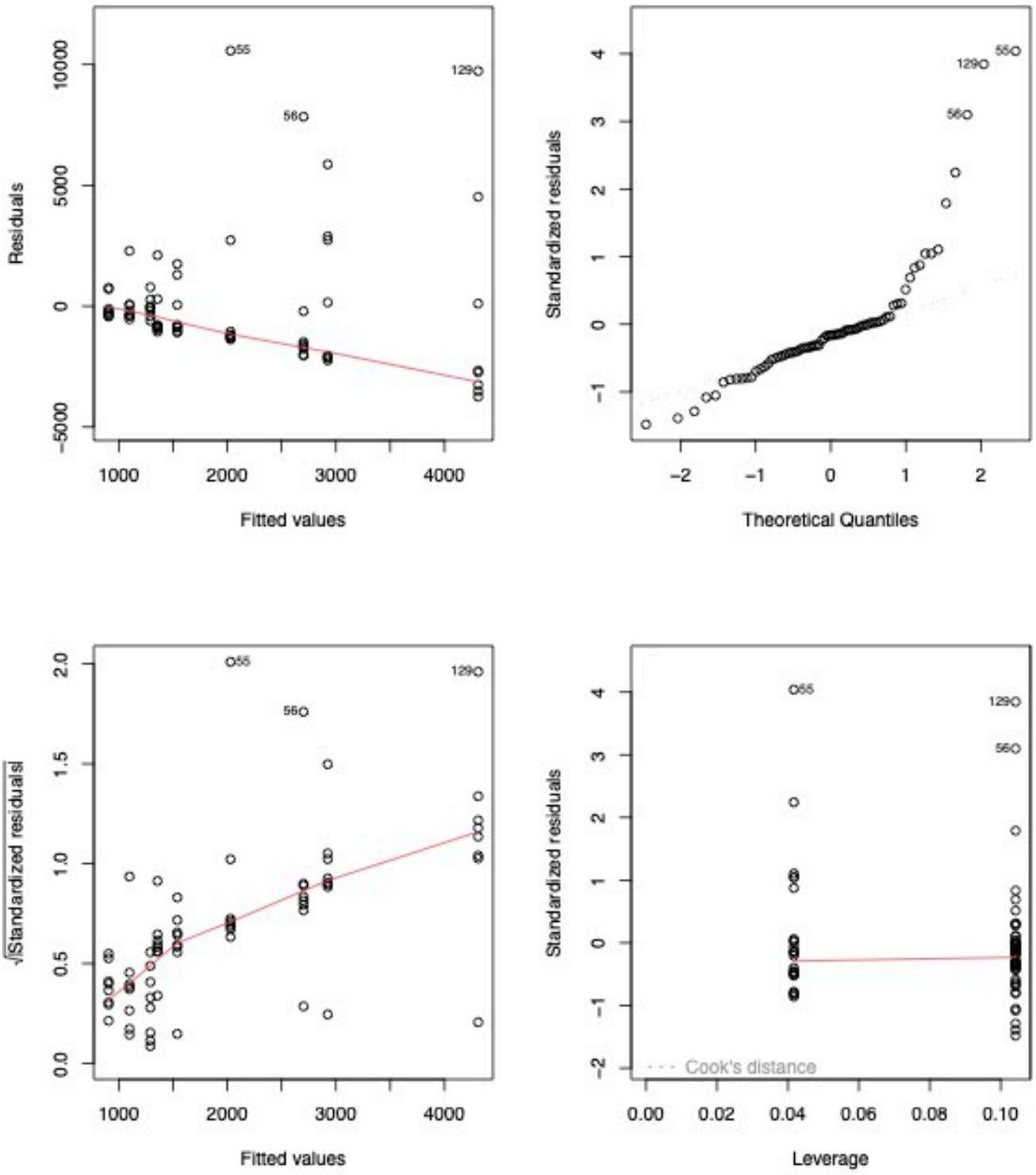


2025 | Bottom (20–30 cm) | Root Dry Mass

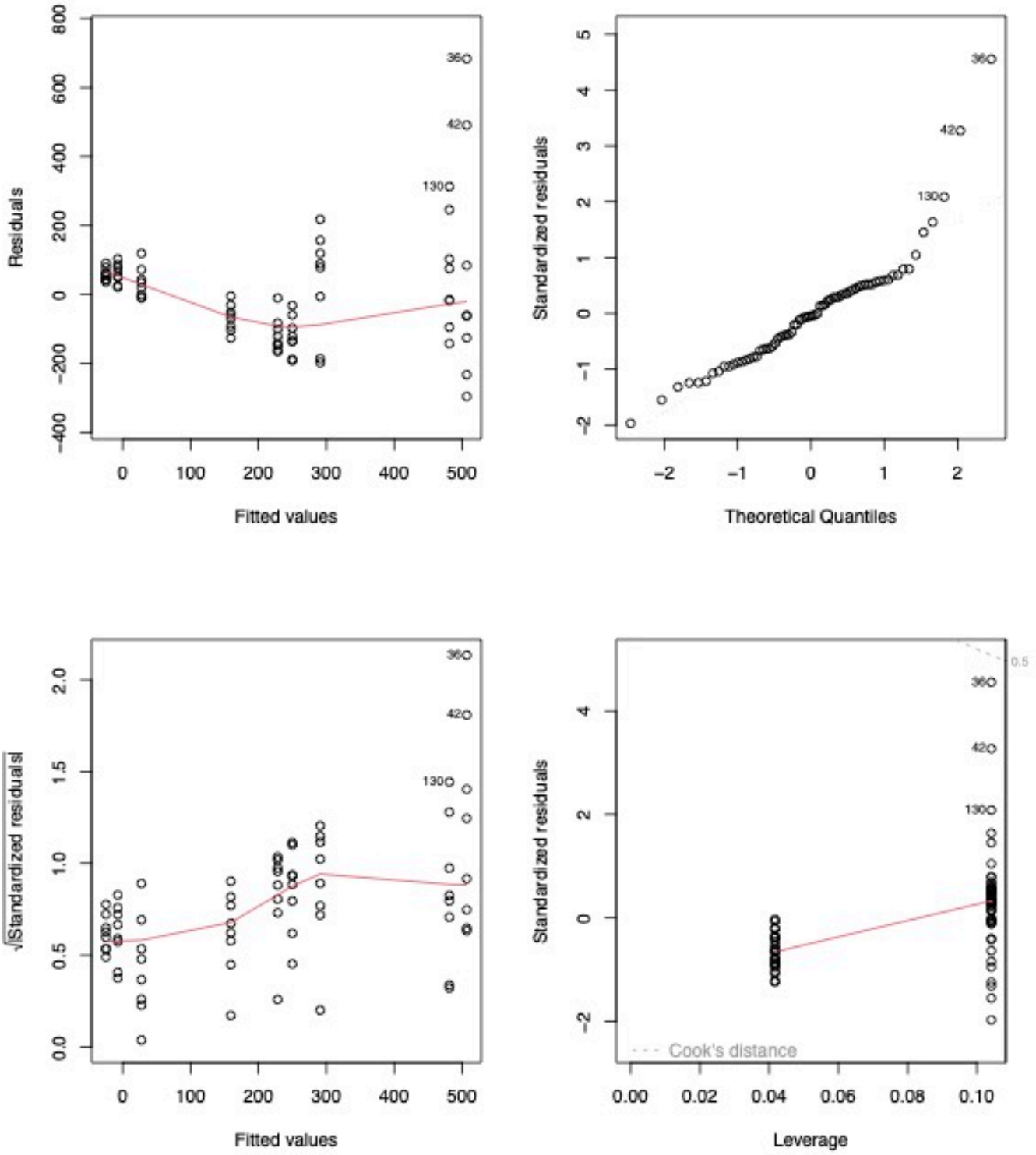


2025 | Two–Way (all depths) | SOM

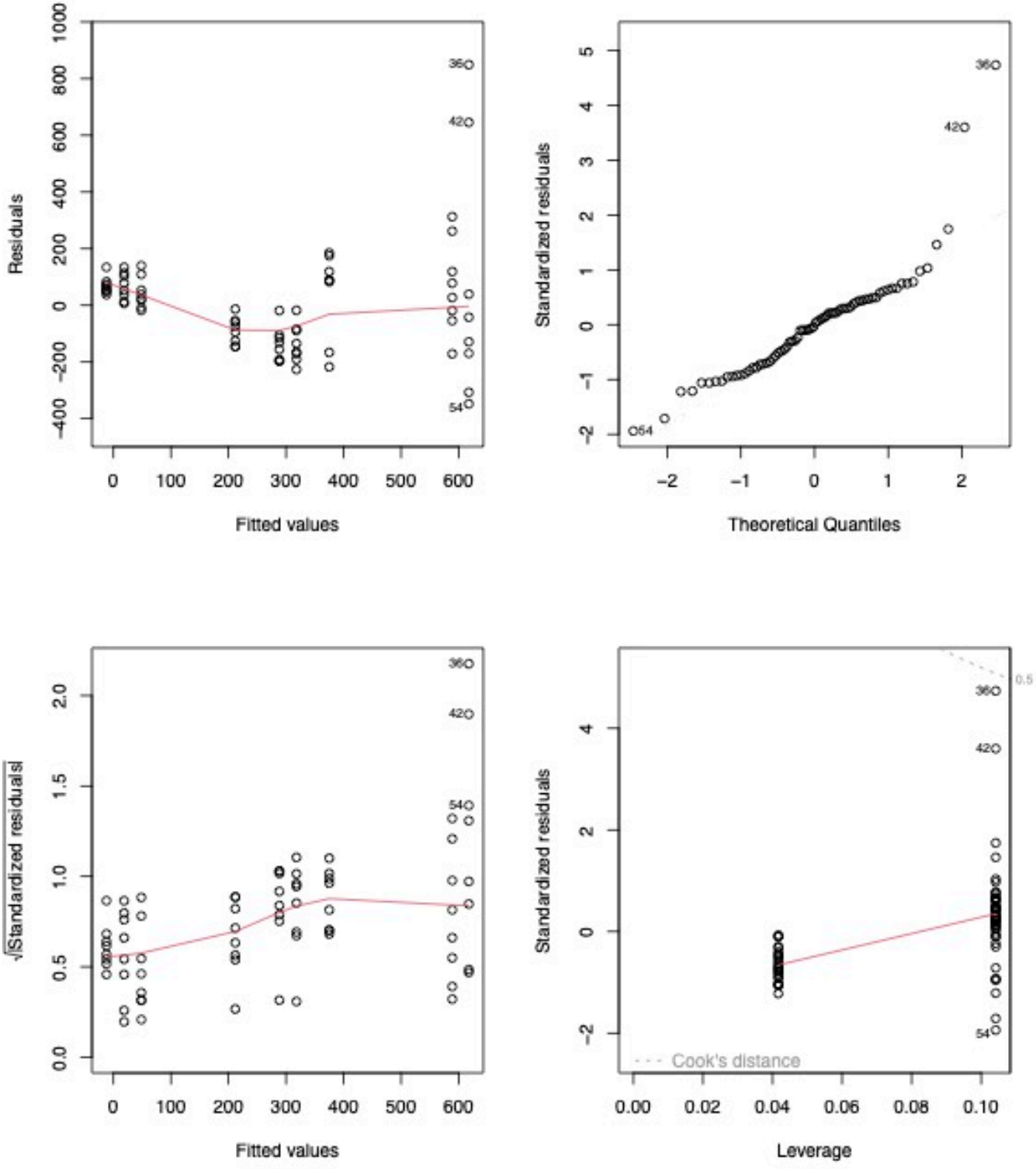



2025 | Two–Way (all depths) | SCC
Residuals
Fitted values
Standardized residuals
Theoretical Quantiles
√|Standardized residuals|
Fitted values
Standardized residuals
Cook's distance
Leverage
55
129
56

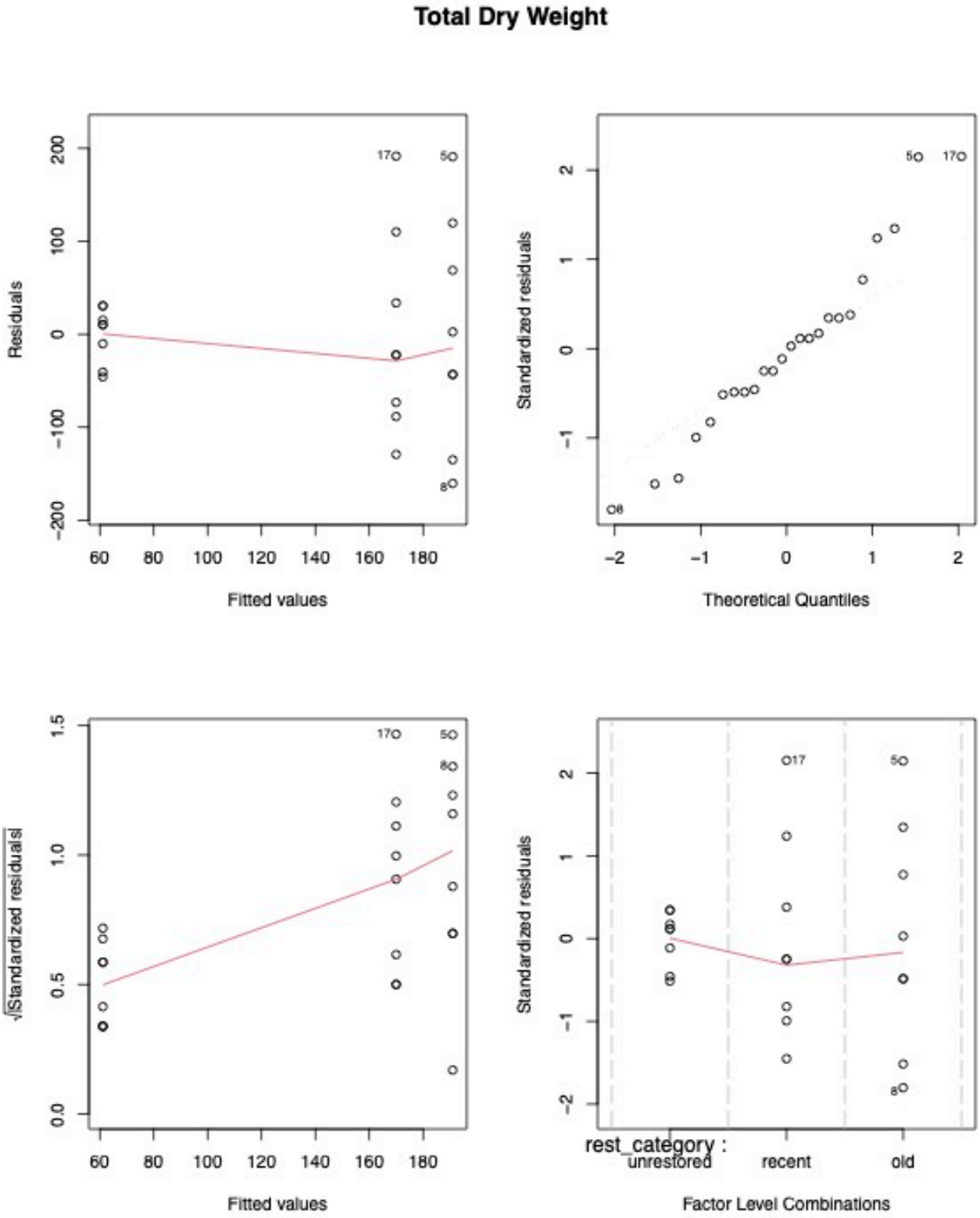
2025 | Two–Way (all depths) | ROM



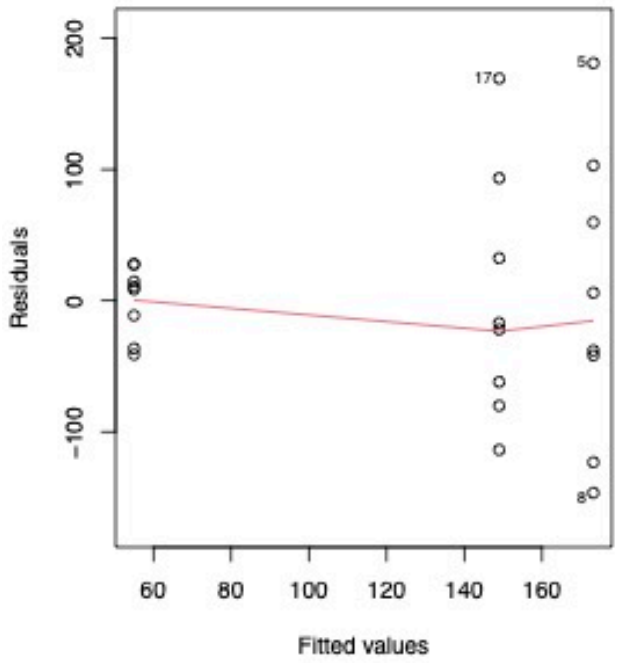
2025 | Two–Way (all depths) | Root Dry Mass

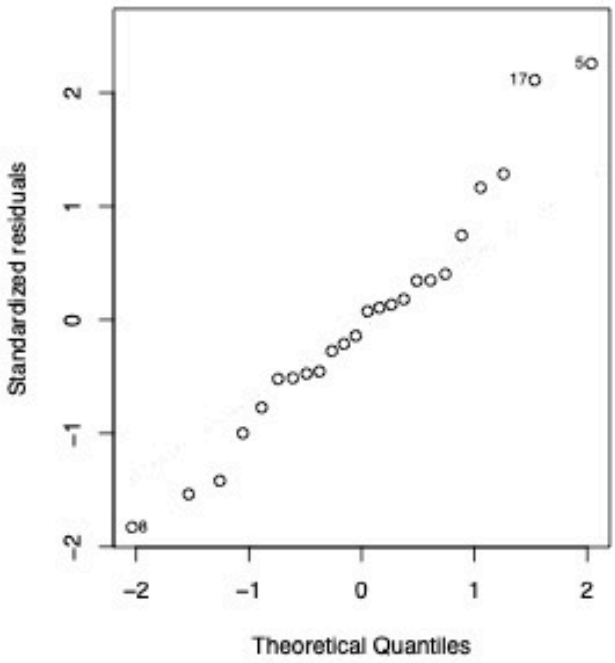


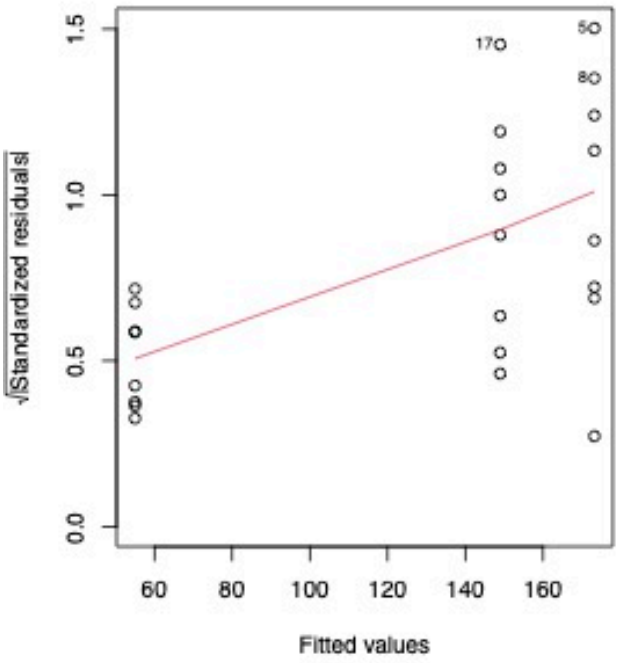
Appendix J – Diagnostic Plots for Ingrowth Cores

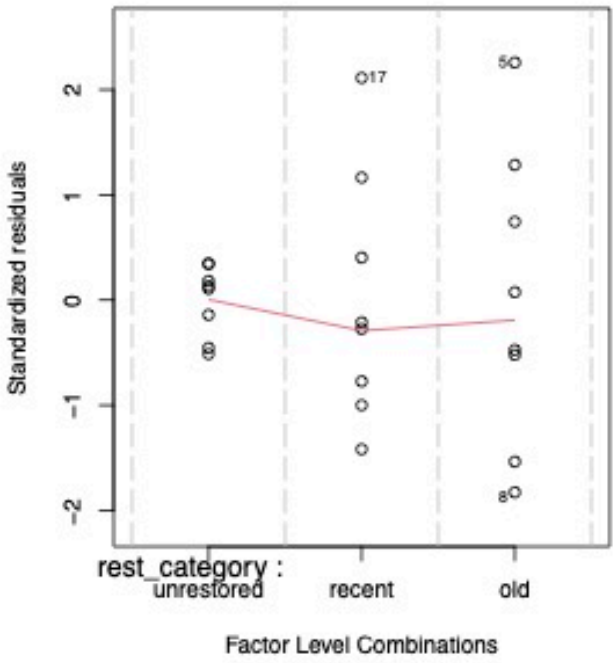


Organic Matter









Appendix K – Peak Aboveground Biomass Determination

To attempt to determine the timing of peak aboveground biomass, we returned weekly to five old meadow and five recent meadow sites between the weeks of June 24th and August 12th, 2024, clipping one 0.5 m × 0.5 m quadrat of live aboveground biomass at each site each week. We returned samples to the lab, dried them at 105ºC for 24 hours and weighed to the nearest 0.01 g (Sartorius QUINTIX2102-1S Toploading Balance). The figure below summarizes the results of this trial. It shows that while biomass increased overall over this period, there was not a distinct peak or major difference between the two meadow categories. Thus, in 2024 we collected aboveground biomass samples at all sites following the conclusion of these weekly samples between in late August/early September. In 2025, we collected in mid-August, coinciding with the collection of soil ingrowth cores. We did not repeat the weekly biomass collection in 2025, though the mild to moderate drought conditions could have affected the timing of peak aboveground biomass (Agriculture and Agri-Food Canada, National Agroclimate Information Service, 2025).

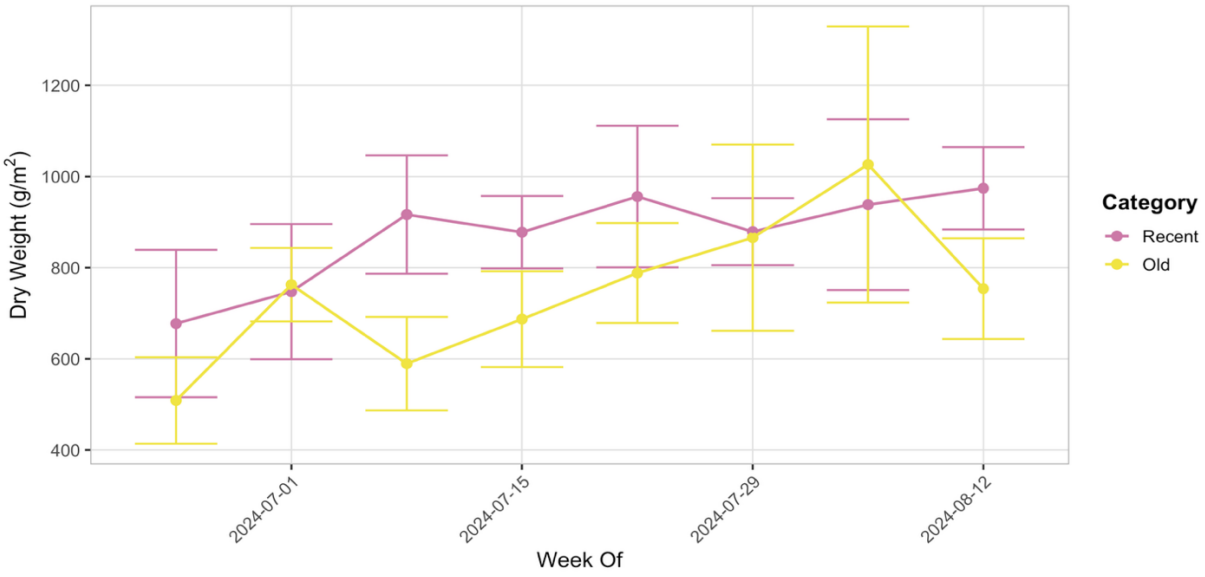


Figure K.1 Mean dry weight of aboveground biomass from June 24 to August 12, 2024 by meadow age category: recent meadow (n = 5) and old meadow (n = 5).

Appendix L – Raw Data

Raw data associated with the two data chapters can be found at:

https://doi.org/10.6084/m9.figshare.32288088.