TRANSFORMING LANDSCAPES: QUANTIFYING SOIL HEALTH, CROP YIELD AND CARBON DYNAMICS FOLLOWING FOREST AND GRASSLAND CONVERSION

by

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A thesis submitted in partial fulfillment of the requirements for the degree of:

MASTER OF SCIENCE IN ENVIRONMENTAL SCIENCE

Kamloops, British Columbia

July 2025

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ABSTRACT

Climate change is projected to expand agriculture into regions previously unsuitable for crop production. However, the effects on soil health, crop performance, and carbon dynamics remain uncertain across diverse local contexts. To address this knowledge gap, I conducted a short-term study of a semi-arid rangeland near Kamloops, British Columbia, where contiguous forested and grassland areas were converted to corn for cattle grazing with inputs of compost, chemical fertilizer and herbicide. I quantified total ecosystem carbon (TEC), 13 biochemical and physical soil variables, and corn yield and quality metrics. Soil variables were synthesized into a Soil Quality Index (SQI) and a Soil Multifunctionality Index (SMF), based on enzyme activities representing carbon, nitrogen, and phosphorus cycling capacities.

Forest conversion reduced total ecosystem carbon (TEC) by 41%, whereas grassland conversion had no significant effect on TEC. Although forest-converted soils showed an 86% increase in soil organic carbon (SOC) stocks, and grassland-converted soils showed no significant SOC stock change despite identical compost and corn residue amendments. These gains masked active microbial utilization and rapid carbon turnover: soil respiration rose by 80% in forest plots and enzymatic activities increased under both conversions. Compared to the forest conversion, the grassland conversion achieved higher Soil Quality Index (SQI) and Soil Multifunctionality Index (SMF) scores and produced 21% more corn (14.95 versus 12.39 Mg ha^{-1}) with greater crude protein in stalks (11.16% versus 7.59%) and cobs (4.57% versus 3.74%). However, the accelerated SOC mineralization in both conversion scenarios undermines the potential for long-term carbon sequestration under these agricultural practices.

These results demonstrate that, despite short-term SOC gains in forest converted soils and enhanced soil function in the grassland conversion, annual corn with grazing at the studied site was a suboptimal management strategy for maintaining stable soil carbon pools. Future work should assess long-term, post-conversion carbon dynamics and alternative cropping systems to inform sustainable, context-specific land-use decisions.

Keywords: land conversion, carbon loss, ecosystem impacts, soil health, crop yield

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ACKNOWLEDGEMENTS

I'm deeply grateful to everyone who helped me pull off this "group" project.

First, my heartfelt thanks to my supervisor, Dr. Lauchlan Fraser, for taking a chance on me and making this long-dreamed achievement possible. I'm forever grateful. I also extend my appreciation to my supervisory committee and especially would like to acknowledge Dr. Michael Preston for carving out time from his busy schedule to guide me and challenge my results.

Huge thanks to the Fraser Lab crew for braving every field condition (rain, thunder, snow, and ankle-deep manure) with good humor and a real appreciation of the benefits of type 2 fun. Kaitlyn, Sam, Keenan, Ashley, Ghassen, Matt F., Adam, Rabeya, Behnaz, and John – you're the best crew a person could ask for. A special shout-out to Dr. Jay Prakash Singh for his patience and exceptional instruction in statistics and soil microbial ecology, and to Matthew Coghill, the R whisperer, for generously sharing his expertise. I wouldn't have completed this thesis without all of you.

Thank you to Agriculture and Agri-Food Canada for the extensive soil and crop analyses. I'm especially indebted to Dr. Kirsten Hannam and Dr. Shabtai Bittman whose patience, insights, and probing questions enriched this work.

I'm also deeply thankful to the Devick family for letting me dig holes on their ranch for two years. Farming is hard enough. Thank you for tolerating the extra boots in your fields and for your generous support of this project.

Finally, to my family, the ultimate cheer squad. Thank you for every "so proud of you," "you got this," and "well, you asked for it" pep talk that carried me across the finish line. Graduating alongside my boys has been a true gift (and perhaps fodder for a future book. Ben and Pat, you'll be main characters). To my husband, Brad: I am endlessly grateful for your encouragement, for tolerating my frequent absences and marathon desk sessions, and for mastering corn harvesting in support of this project. Your unwavering support has made all the difference. I couldn't have asked for a better partner in crime.

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CHAPTER 1: INTRODUCTION

Climate-driven Agricultural Frontiers and Future Land Conversion

Climate change is anticipated to increase Earth's arable land area in northern latitudes, as warmer temperatures make regions previously unsuitable for agriculture more conducive to crop production. Hannah et al. (2020) combined projections from 17 global climate models for temperature and precipitation with agricultural models to predict the suitability for growing key food crops. They defined these emerging areas as "agricultural frontiers," estimating that they encompass between 10.3 and 24.1 million km² globally (Figure 1.0). Similarly, Zabel et al. (2014) projected an expansion of suitable cropping areas by approximately 5.6 million km², with both studies identifying northern high-latitude regions in Canada, China, and Russia as the primary zones of change. These regions are expected to increasingly accommodate the cultivation of crops such as potatoes, wheat, maize, and soy (Hannah et al., 2020; Rosenzweig et al., 2013).

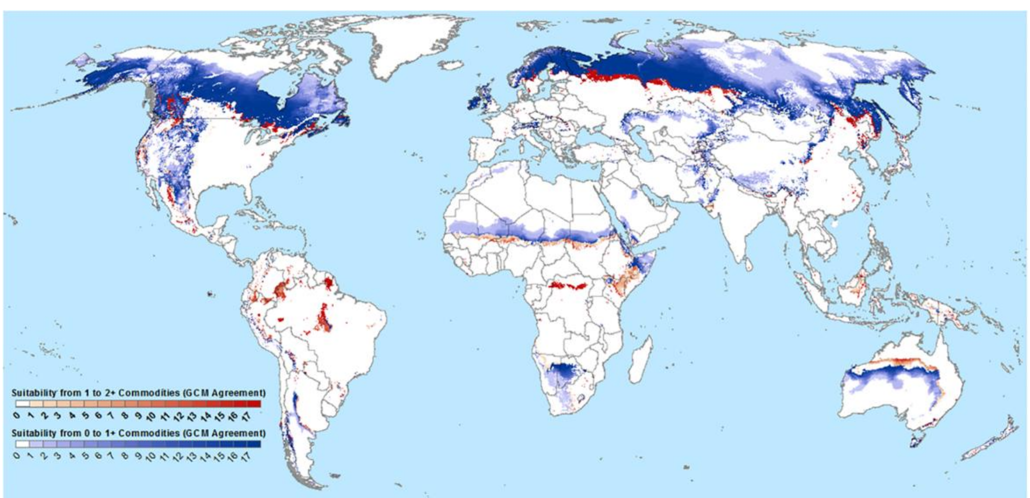

Figure 1.0. Map of global climate-driven agricultural frontiers. Areas that transition from no current suitability for major commodity crops to suitability for one or more crops are depicted in blue, while currently uncultivated areas that transition to suitability for multiple major commodity crops are shown in red (Hannah et al., 2020).

Evidence suggests that populations are already exploring food production opportunities in these areas; for instance, the government of the Northwest Territories recently implemented an agricultural strategy to promote the development of northern lands (Government of NWT, 2017). While this expansion presents opportunities to enhance food security for northern communities and address global food demands, it also necessitates careful consideration of the balance between agricultural production and the preservation of biodiversity and ecosystem services.

Challenges in Quantifying the Impacts of Land Conversion

Maintaining the balance between agricultural expansion and ecosystem preservation is critical in the future, as researchers predict global biodiversity losses of 26% in regions affected by unbridled agricultural expansion (Zabel et al., 2017). In addition, declines in downstream water quality, and consequent risks to human, ecosystem, and fisheries health, are anticipated from fertilizer and biocide runoff (Bennett et al., 2001; Hannah et al., 2020). Perhaps the most significant anticipated impact, however, concerns climate services, in particular soil organic carbon storage (Foley et al., 2005; Wei et al., 2014). Hannah et al. (2020) predicted that the total carbon potentially released from the cultivation of these climate-driven agricultural frontiers could reach up to 177 GtC, which is equivalent to 119 years of current U.S. CO_2 emissions (Boden et al., 2016). This loss is expected to result primarily from accelerated decomposition of soil organic matter due to altered microclimatic conditions, changes in the quality and quantity of carbon inputs, the breakdown of soil aggregates, and enhanced surface erosion (Martens et al., 2004; Wei et al., 2014; Jiang et al., 2023). To mitigate these impacts, agricultural expansion must be strategically managed to preserve soil health, given that the capacity of soils to sequester carbon fundamentally depends on the vitality of their complex ecological processes (Lal, 2016; Liptzin et al., 2022).

Despite the critical importance of preserving soil health to maintain carbon sequestration and support ecosystem services, a significant challenge remains: the lack of consensus regarding both the definition of soil health and the indicators used to assess it. Historically, soil health was defined primarily as a measure of crop production. However, most researchers and policymakers today have broadened this definition to include the soil's capacity to support food and fiber production while delivering essential ecosystem services that maintain human quality of life and conserve biodiversity (Kibblewhite et al., 2008; Lehman, 2020).

Defining the biological, chemical, and physical parameters of a healthy soil remains contentious. For example, the Soil Health Institute conducted a three-year, $6.5-million study comparing 30 soil health indicators across 124 conventional and regenerative farming sites and recommended three minimum metrics—soil organic carbon concentration, carbon mineralization potential, and aggregate stability—ideally measured across the 0–1.0 m soil profile (Soil Health Institute, 2021). In contrast, other researchers contend that these indicators do not fully capture the complexity of soil biochemical and physical processes, advocating for the inclusion of additional measurements such as pH, bulk density, electrical conductivity, various indices of soil microbial activity and crop yield (Alkorta et al., 2003; Cardoso et al., 2013; Martinez et al., 2018; Goodwin et al., 2023). Moreover, there is ongoing debate regarding which indicators are most effective in capturing changes in soil health following land conversion (Graham et al., 2021; Benalcazar et al., 2022).

Scientists have addressed the challenges of selecting individual soil health indicators by developing comprehensive soil quality and multifunctionality indices that account for the complex interactions among biological, chemical, and physical processes (Rayesi, 2017; Lenka, 2022). These indices eliminate redundant variables and retain only those most strongly correlated, thereby creating minimum data sets that more accurately represent soil function. For example, Yu et al. (2018) initially evaluated 11 soil indicators when studying the effects of different cropping systems in the alpine grasslands of China. After applying statistical analyses to identify the most informative variables, only 4 indicators were retained in the final minimum data set. Similarly, Raisi (2017) began with 15 soil health indicators to assess the impact of rangeland conversion to annual cropland in Iran, reducing the list to just 3 key indicators to determine soil quality.

Collectively, these studies suggest that soil quality indices (SQIs) are effective tools for evaluating the impacts of land conversion. By integrating multiple soil indicators and capturing their interactions, SQIs overcome the limitations of analyses based solely on individual metrics. This comprehensive approach provides valuable insights into soil function, thereby offering additional insights into the complex processes underlying soil health and the consequences of land use changes on carbon cycling.

Land Conversion Impacts: Research Limitations and Response Variability

Regardless of whether researchers rely on individual soil indicators or comprehensive indices, studies examining soil health and quality following land conversion have yielded mixed results, posing an additional challenge to mitigating soil carbon loss during conversion. The limited body of literature suggests that these outcomes largely depend on factors such as the type of ecosystem converted, post-conversion management practices, crop and livestock system selection, and variations in soil properties and climate at regional and micro-regional scales (Liptzin et al., 2022).

For example, a meta-analysis of global land conversion reported that, between 2010 and 2018, deforestation resulted in a loss of 12.3 Gt of carbon from both below- and aboveground biomass, with most forests converted for agricultural purposes (Hu, 2021). However, local rates of soil carbon mineralization following deforestation are more heterogeneous and largely climate-dependent, with the greatest decreases observed in temperate (52%), tropical (41%), and boreal (31%) regions (Wei et al., 2014). Even within these regions, soil organic carbon (SOC) losses are variable and site-specific, depending on the dominant soil type. For instance, forested podzols and vertisols have the potential to release more soil carbon than forested luvisols and ultisols (Tarnocai, 2005; Bruun et al., 2013). Similarly, Liang et al. (2023) found that, in eastern Canada, SOC losses after forest conversion to agriculture were 18 t C ha^{-1} for coarse-textured soils, 43 t C ha^{-1} for medium-textured soils, and 65 t C ha^{-1} for fine-textured soils. By comparison, on the Canadian prairies, forestland conversion to cropland resulted in a loss of 27 t C ha^{-1}, or 25%, for medium-textured soils, with no significant change observed for coarse-textured soils.

Similar variability is evident in grassland systems. Globally, temperate grasslands are estimated to store an average of 236 tonnes of carbon per hectare; when tilled for row crop production, nearly two-thirds of this carbon is lost to the atmosphere as CO_2 (IPCC, 2000). As with forests, however, the magnitude of carbon loss in grasslands varies considerably among sites due to factors such as climate, topography, and soil type. For example, a meta-analysis of SOC losses in Canadian Prairie grasslands converted to agriculture reported average reductions of 15, 26, and 5 t C ha^{-1} in coarse-, medium-, and fine-textured soils, respectively (Liang et al., 2023). Collectively, these findings underscore the need to consider local soil and climate conditions when evaluating the impacts of land conversion on carbon dynamics.

Less understood, however, is which management systems best mitigate the negative impacts on soil health and carbon cycling following land conversion. Numerous studies have examined practices that improve soil carbon storage in established agricultural soils, such as no-tillage, low-tillage, no-tillage with crop residue mulch, incorporating forages into rotation cycles, diverse annual cover crops, and the application of manure and other biosolids (Lal, 2005; Bai et al., 2019; Abbas et al., 2020; Valkama et al., 2020). Although these practices are expected to enhance soil fertility, nutrient cycling, and reduce carbon emissions in converted soils, few studies have directly evaluated optimal management systems post conversion, particularly in the short term.

In one of the limited investigations on this topic, Jiang et al. (2023) employed a predictive model to assess long-term changes in soil organic carbon (SOC) across six farms in northern Ontario employing various cropping systems following conversion. Their results indicated that predicted SOC stocks were highest under continuous pasture, followed by a rotation from barley–oats to pasture, then from pasture to barley–oats, next legume hay to barley–oats, with continuous barley–oats cropping yielding the lowest SOC levels. In another long-term study in southern Brazil, Balota et al. (2015) examined microbial activity and soil carbon sequestration after conversion from forest to either a perennial coffee crop or annual maize and soybeans managed under conventional tillage or no-tillage. The study determined that the perennial cropping system exhibited the highest SOC content, followed by the no-tillage annual system and then the conventional tillage system. Similarly, Rasouli-Sadaghiani et al. (2018) demonstrated a progressive deterioration in soil quality following land-use changes, with conditions declining from intact forest to grassland to orchard land to annual cropped arable land. While these studies provide valuable insights into long-term trajectories, they do not address the immediate consequences of different cropping practices on recently converted soils.

Research on incorporating livestock grazing into recently converted lands in the northern hemisphere is similarly limited. Studies from subtropical environments offer some insights, where forest conversion to cattle pasture has been extensively studied and is generally associated with soil carbon losses. For example, Fearnaside et al. (1998) reported that clearing 1.38×10^6 ha of Amazonian forest for cattle pasture in 1990 resulted in a net release of 11.7×10^6 tonnes of carbon. However, dos Santos et al. (2019) found that converting Brazilian Atlantic forest to a *Brachiaria brizantha* pasture for controlled cattle grazing initially resulted in a loss of 12.6 Mg C ha^{-1} but subsequently led to a net gain of 43.2 Mg C ha^{-1} to a depth of 100 cm over 16 years. This recovery was attributed to the slow decomposition of forest-derived carbon and the accumulation of carbon from *Brachiaria*. These contrasting outcomes underscore the complexity of soil carbon dynamics under different grazing regimes, although subtropical responses may not directly translate to the northern hemisphere due to significant differences in climate and soil type.

Additional insights could be gained from studies examining the impact of various perennial grazing practices on soil organic carbon in established agricultural operations in the northern hemisphere, though these results also exhibit substantial variability due to climate differences. Mosier et al. (2021) demonstrated that adaptive multi-paddock grazing on southeastern U.S. rangelands resulted in 13 tonnes more soil carbon compared to conventionally grazed sites. Similarly, Bork et al. (2020) reported increased soil carbon in U.S. grasslands under higher stocking rates, attributed to an increased abundance of non-native species. In contrast, McSherry et al. (2013) found that increasing grazing intensity led to SOC increases of 6–7% in C4-dominated and C4–C3 mixed grasslands, while SOC decreased by 18% in C3-dominated grasslands.

The integration of grazing cattle into annual cropping systems represents another management approach with limited soil carbon research. Tracy et al. (2008) demonstrated in Illinois that a rotational system incorporating winter grazing of corn resulted in higher total soil carbon concentration and microbial biomass compared to continuous corn production, suggesting that grazing may promote the accumulation of labile carbon. While this finding has potential implications for including grazed corn as part of a post-conversion rotation with perennial crops, the outcomes likely depend on soil type and climate conditions and may not fully translate to converted soils.

When assessing the impacts of land conversion, it is essential to also consider the effects on crop yields. It is generally accepted that converted lands result in lower yields over varying durations, depending on initial climate and soil type as well as the quantity and quality of external inputs applied (De Laporte et al., 2022). Yield reductions are attributed not only to the negative effects on soil health and quality following conversion but also to high carbon-to-nitrogen ratios that limit the availability of nitrogen essential for crop productivity (NDSU, 2021). In a U.S. meta-analysis of crop yields, 69.5% of newly converted croplands produced yields below the national average, resulting in a mean yield deficit of 6.5%—a phenomenon often termed "yield-drag" (Lark et al., 2020). Understanding the mechanisms driving these yield reductions and adjusting management practices accordingly is critical to ensuring that the crop potential justifies the costs associated with converting lands.

From Carbon Sources to Sinks: Adaptive Strategies in Ecosystem Conversion

Assessing the condition of intact or managed ecosystems is critical in land conversion studies. Research demonstrates that the original state of a grassland or forest, in terms of plant species diversity, drought conditions, and soil biochemistry can exert a strong legacy effect on subsequent crops (Crotty et al., 2016; Grange et al., 2022). For example, variations in plant diversity and biomass production in grasslands can either enhance or impede subsequent crop yield, depending on how shifts in soil biochemistry affect nutrient availability (Eisenhower et al., 2016; Eisenhower et al., 2017). In forest conversions, changes in soil pH, carbon, and nitrogen alter the composition of soil microbial communities thereby influencing nutrient cycling processes and future crop performance (Liu et al., 2020; Zong et al., 2022; Liu et al., 2023).

The importance of assessing intact ecosystems is further underscored by the degradation observed in many remaining grasslands. Fewer than 10% of the world's grasslands remain intact, with most now utilized for livestock grazing at varying intensities (Scholtz et al., 2022). In these systems, increasing aridity, fire disturbance, rising livestock numbers, and overgrazing have disrupted the balance between soil carbon inputs and decomposition processes, causing many grasslands to shift from carbon sinks to sources of carbon emissions (Liu et al., 2022; Chang et al., 2021). Researchers estimate that due to human disturbance and a warming climate, grasslands now contribute 10–30% of total CO_2 emissions, with a positive feedback loop from global warming expected to further accelerate these emissions (Cao et al., 2025).

Forests, like grasslands, are significantly impacted by climate change and human activities. Warming temperatures are increasing the prevalence of forest pests and pathogens, which diminish stand health and reduce forests' capacity to provide essential ecosystem services. In recent decades, insect outbreaks have resulted in carbon losses of approximately 13.5 Tg C per year in Canada (Kurz et al., 2008). Additionally, wildfires are projected to increase in both frequency and severity; Phillips et al. (2022) predicted that wildfires in boreal North America could contribute nearly 12 gigatonnes of CO_2 by mid-century. These disturbances are converting forests from carbon sinks into carbon sources. Since 1990, Canada's forests have become net carbon sources due to increases in areas burned by wildland fires, insect outbreaks, and intensified logging in response to pest infestations (Natural Resources Canada, 2025). Similarly, in 2021, Finland's forests shifted from a carbon sink to a carbon source, contributing 1.12 Mt CO_2 to the atmosphere. Scientists suggest that this transition resulted from declining tree biomass growth, increased tree loss from logging and natural mortality, and rising soil emissions driven by altered soil carbon dynamics (LUKE, 2023).

Given that many ecosystems have shifted to become net carbon contributors, and agriculture continues to expand into new latitudes and elevations, future environments are expected to differ significantly from current conditions, necessitating a redefinition of conservation, restoration and sustainable land-use practices. Miller et al. (2007) and Herrick et al. (2012) advocated for adaptive strategies that transition ecosystems from their existing states to conditions that enhance carbon sequestration while reducing overall greenhouse gas emissions. Proven practices such as agroforestry, including alley cropping, silvopasture, and forest farming, offer potential solutions that support food production, enhance ecosystem services, and increase carbon storage, all while preserving natural or semi-natural habitats to support native biodiversity. Additionally, intensifying management in cropped areas through conservation tillage, residue incorporation, and diversified crop rotations with cover crops could boost carbon sequestration by 1–2 Pg C yr^{-1} globally and 0.1–0.3 Pg C yr^{-1} in North America (Lal, 2003), with Udawatta and Jose (2012) estimating that agroforestry practices alone could sequester approximately 530 Tg C yr^{-1} in the U.S.

However, agroforestry responses vary widely depending on site-specific biological, climatic, soil, and management conditions (Ramachandran et al., 2009). Moreover, a significant knowledge gap exists regarding integrated ecosystem and soil carbon responses following the partial conversion of intact areas through agroforestry. Addressing this gap is critical to fully harnessing agroforestry's potential for enhancing carbon sequestration and informing effective, adaptive land management practices.

Summary

The reviewed literature demonstrates that land conversion produces complex and variable effects on soil health, carbon dynamics, and crop yields. These effects depend on multiple, interacting factors from original ecosystem type and baseline soil metrics to post-conversion management practices and local climatic influences. Although comprehensive evaluation tools, such as soil quality indices, help to quantify conversion outcomes; and adaptive management strategies, including agroforestry, tailored grazing practices, and specific crop rotations, show promise in mitigating negative impacts, significant knowledge gaps remain. As climate change drives agricultural expansion into previously unsuitable northern regions, developing flexible, site-specific strategies that optimize productivity while enhancing carbon sequestration and preserving biodiversity become imperative. Future research should focus on local-scale investigations to inform broader conclusions. By refining adaptive approaches and understanding their impacts at the local level, researchers can ensure that land conversion practices remain economically viable and environmentally sustainable in a world with a rapidly changing climate.

Current Study

This study examines the impacts of land conversion on soil properties and crop performance in a high-elevation, privately owned rangeland near Kamloops, BC, where a grazed area encompassing contiguous forest and grassland was recently converted to corn production for cattle grazing. This "living laboratory" is uniquely suited for conversion analysis for several reasons:

1. **Altered baseline conditions:** Both grassland and forest have experienced intensive spring and fall grazing, shifting them from a potential natural community (PNC) climax stage to altered states dominated by agronomic species (Government of BC, n.d.).
2. **Simplified forest composition:** The forest is dominated by Interior Douglas-fir (*Pseudotsuga menziesii* var. *glauca)* after near-complete pine species removal, reducing its native diversity.
3. **Novel crop system:** Corn cultivation at this elevation is unprecedented in the region.
4. **Dual-ecosystem comparison:** Adjacent forest and grassland allow direct comparison of conversion responses between ecosystems.
5. **Unexplored carbon dynamics:** Although grazed grasslands in the studied region have been extensively mapped for carbon sequestration potential, carbon dynamics following conversion at these elevations remain unstudied (Harrower et al., 2012; Harrower, 2014).

The specific objectives of this study were to: (1) quantify changes in soil health indicators between forest and grassland conversions and their respective intact ecosystems; (2) assess soil quality and multifunctionality between conversion types; (3) evaluate changes in crop yield and quality metrics between conversion types; and (4) determine total ecosystem carbon losses or gains between intact and converted systems.

By addressing these objectives this study aims to contribute to the broader understanding of agricultural land conversion by providing detailed insights at the local landscape scale.

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CHAPTER 2: Evaluating the impacts of forest and grassland conversion on soil health, crop metrics and carbon dynamics

INTRODUCTION

"Each soil has had its own history. Like a river, a mountain, a forest, or any natural thing, its present condition is due to the influences of many things and events of the past."
— Charles Kellogg, *The Soils That Support Us*, 1956

Climate change, characterized by warming and altered precipitation patterns, alongside human activities such as overgrazing, represents a primary driver of ecosystem degradation in both forested and grassland rangelands (Hou et al., 2022, Liu et al., 2019). In grasslands, these stressors diminish primary productivity and fundamentally alter soil properties, ultimately leading to depletion of soil and ecosystem carbon pools. Given that grasslands account for approximately 30% of global carbon storage, their historical and continued degradation represents a significant threat to terrestrial carbon stocks (Booker et al., 2013, Liu et al., 2019).

Forest ecosystems face challenges analogous to those observed in grasslands. Climate change is projected to increase wildfire frequency and the prevalence of pests and pathogens, while overgrazing contributes to declines in forest floor biodiversity and alterations in forest soil properties (Chmura et al., 2011; Dukes et al., 2009). These combined stressors compromise forests' capacity for carbon sequestration, which currently offsets approximately one-third of anthropogenic carbon emissions (Natural Resources Canada, 2013). Consequently, the future role of degraded forests as carbon sinks remains uncertain, particularly considering evidence indicating that degraded forest systems can become net sources of carbon emissions (Nunes et al., 2020; Phillips et al., 2022).

Climate warming is also driving the transformation of these intact ecosystems as part of the emerging "northern agricultural frontier", with projections indicating that 10 to 20% of northern regions could be converted to agriculture by 2100 (Meyfroidt 2021, King et al., 2018). The Boreal region, Earth's largest forest biome, together with the Arctic's treeless tundra, maintains vast repositories of soil and vegetative carbon while supporting rich biodiversity. Boreal forests, comprising one third of global forest cover and storing 32% of the world's forest carbon stock, are increasingly targeted for agricultural expansion (Unc et al., 2021). While such conversion may enhance regional food security, it introduces significant uncertainties regarding crop performance on modified lands, and the subsequent impacts on soil biochemical processes, climate regulation services, and biodiversity conservation.

The human- and climate-driven transformation of forest and grassland ecosystems appears increasingly inevitable. In certain cases, converting a degraded system may present opportunities for ecosystem restoration and enhanced food security in remote regions. However, ecosystem responses to conversion can vary substantially, with some systems demonstrating greater resilience or adaptability than others. Numerous studies document that the legacy effects of the parent ecosystem play a critical role in shaping both the short- and long-term biochemical processes and functionality of the converted system (Abraha et al., 2018; Foster et al., 2003). Therefore, understanding how land conversion impacts soil health indicators, overall quality, and ecosystem carbon pools across different ecosystem types is essential for developing sustainable agricultural practices that enhance carbon sequestration and reduce greenhouse gas emissions.

A significant challenge in understanding these impacts is the considerable variability of soil health metric measurements reported in the literature on land conversion; outcomes can differ markedly across regions, and even within localized micro-regions, due to variations in soil parent material, climate, antecedent vegetation, agricultural management practices, and ecosystem legacy effects (Barbero et al., 2025, Cepeda et al., 2008, Dick, 1992, Khan, 1996, Rasck et al., 2000). An additional challenge lies in the limitations of singular soil health indicators to accurately capture conversion impacts, given the complex biochemical interactions within soil systems. Researchers have addressed this limitation by developing soil quality indices that comprehensively capture system complexity (Lenka et al., 2022, Raiesi 2017). Consequently, adopting a regional approach that accounts for local variability is critical, as is the development of comprehensive monitoring tools capable of capturing the intricate biochemical and physical processes governing soil dynamics during land-use transformation.

To address these challenges at a regional scale, we evaluated land conversion impacts on a privately owned property in the Kamloops region of British Columbia containing contiguous forested and grassland rangeland ecosystems. A portion of the property encompassing both ecosystem types was recently converted to corn production for managed cattle grazing. By comparing adjacent intact and converted areas within both forested and grassland ecosystems, we aimed to elucidate local impacts of conversion on soil quality, carbon dynamics and crop productivity. This site provided an ideal case study for examining climate- and human-driven transformation. The intact ecosystems were heavily grazed, and the newly cropped area represented a novel agricultural system because corn cultivation is uncommon at these elevations.

Building on this case study, we posed four research questions:

1. How does land conversion alter soil health indicators when comparing pre- and post-conversion conditions, and how do these indicators differ between forest and grassland conversion types?
2. How do soil quality and ecosystem function, as measured by a comprehensive Soil Quality Index and an enzyme activity-based Soil Multifunctionality Index, differ between forest and grassland conversion types?
3. How do corn crop yield and quality metrics differ between forest and grassland conversion types as indicators of conversion impacts on agricultural productivity?
4. How does Total Ecosystem Carbon (TEC) change between pre- and post-conversion conditions, as quantified through above- and below-ground biomass and soil organic carbon stocks.

By addressing these questions, this study aims to provide critical insights into ecosystem-specific responses to land conversion, thereby informing sustainable agricultural expansion and restoration strategies in regions undergoing transformation.

METHODS

Site Selection

In spring 2023, Devick Ranch, a cattle operation in Kamloops, British Columbia, volunteered land to the BC Living Labs initiative for research aimed at investigating best management practices for extending the cattle grazing season (Figure 2.0). The BC Living Labs, funded by Agriculture and Agri-Food Canada and other industry partners, integrates scientific research with farming operations to develop practical, climate-change friendly practices that farmers and ranchers are willing to adopt (BC Living Labs, 2024).

Until 2019, the site was a mix of grassland and forested rangeland utilized for shoulder season grazing of roughly 300 cow/calf pairs in spring and fall. In 2019, a 9-hectare section, with nearly equal parts forest and grassland, was cleared and converted to corn for cattle grazing using conventional practices. Compost, herbicide, and chemical fertilizer were applied at the rates listed in Tables 1 and 2 prior to planting. This setting offered an ideal environment for studying soil responses to land conversion, featuring both converted and intact ecosystems for comparison.

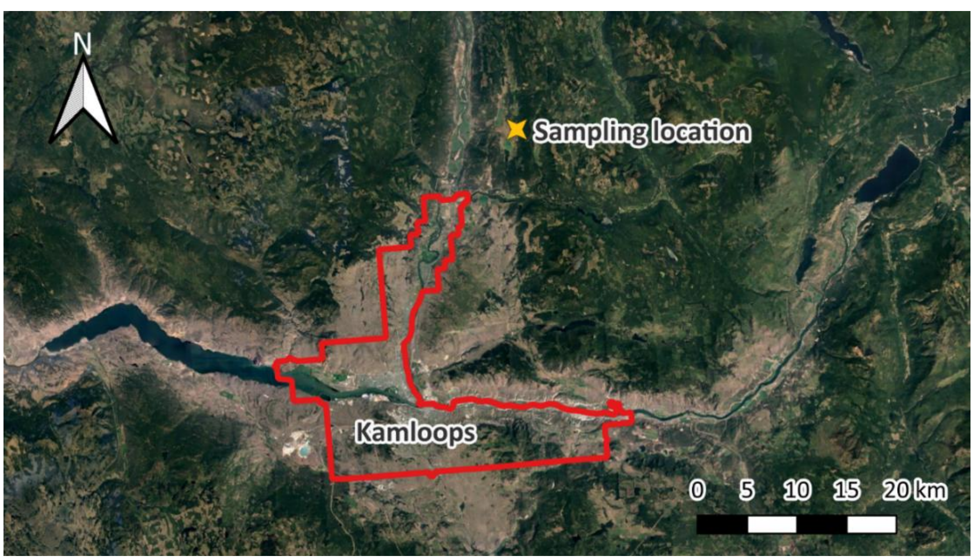

Figure 2.0. Regional context of the Devick Ranch within the Kamloops area. The map was created using QGIS version 3.42.1 (Hannover)

Study Design Layout

Using Google Earth historical imagery, the cleared area was divided into two 4.45-hectare sections corresponding to the former forest and grassland. These plots designated "Converted Forest" and "Converted Grassland," were each marked with 48 evenly spaced sampling points arranged in a grid pattern as described by Carter and Gregorich (2008). Adjacent intact forest and grassland study areas were similarly demarcated, with plots of matching area and sampling point density designated accordingly (Figure 2.1).

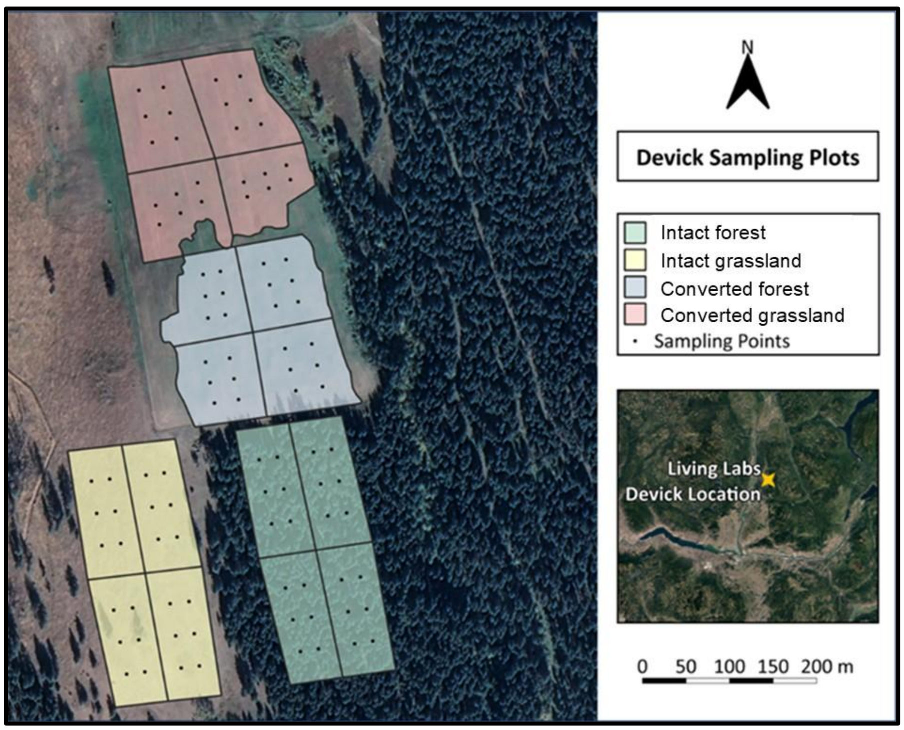

Figure 2.1. Devick ranch (50°55'19"N 120°10'15"W) – Plot layout and design. The map was created using QGIS version 3.42.1 (Hannover)

Site Characteristics

The study site was located within the Interior Douglas Fir Very Dry Hot BEC zone of the Thompson Nicola upper grasslands with an elevation range between 903 and 915 meters (Teucher et al., 2024).

Intact Forest and Grassland Ecosystems

Vegetation surveys revealed that the intact forested area was almost entirely dominated by Interior Douglas-fir (*Pseudotsuga menziesii* var. *glauca*), with a sparse understory and a thick litter layer. In contrast, the intact grassland was dominated by Kentucky bluegrass (*Poa pratensis L*). These findings suggest that the ecosystem has been altered, likely because of prolonged heavy grazing (Forest Renewal BC, 2001).

A visual inspection of soils during sampling determined that the intact forest soils exhibited characteristics typical of the Luvisolic order, while the dark-colored intact grassland soils appeared to be Chernozems. Hand texturing supported these findings, with the forest soils showing a higher sand content and the grassland soils containing finer materials. These conclusions coincide with the published literature of soils in the studied area (Soilx, n.d.).

Converted Areas

In 2023, the converted area was amended with beef compost and chemical fertilizer, treated with herbicide, disked, and seeded with a grazing corn variety.

Table 1 | 2023 agricultural management actions and inputs rates for the converted corn field

Date	**Input/Action**	**Rate**
May 3	Beef cattle compost	25 tonne wet mass/hectare
May 11	Glyphosate	4 L/hectare
May 15-18	Disked	3 passes
May 27	Fertilizer: 21.33-12.08-12.08-4.43-2.13Mg-0.352n-0.07B	594 kg/ha
May 28	Grazing corn: PS2142 RR	79,000 seeds/hectare
July 10	Glyphosate	3.3 L/hectare

During the growth period seven passes were made with the irrigation gun at approximately 3.8 centimeters of water per pass.

The converted corn field was subdivided using electric fencing into two 1.4-hectare sections and three 2.0-hectare sections. On November 20, 2023, 303 bred heifers were introduced to the first section and subsequently rotated to adjacent sections based on forage utilization assessments and behavioral indicators of depleted forage availability. Cattle maintained access to previously grazed sections throughout the rotation sequence. To mitigate the risk of ruminal acidosis, grass and alfalfa hay supplementation was provided on the first day following rotation to each new section. The grazing period concluded on January 6, 2024, totaling 47 days of corn residue utilization.

In spring of 2024, the corn field was disked and moldboard plowed to incorporate residual corn, forage, and manure, followed by herbicide treatment, chemical fertilizer application, and reseeding with a grazing variety of corn. During the growth period, the field received irrigation via seven passes with an irrigation gun, delivering 3.8 cm of water per pass.

Table 2 | 2024 management actions and input rates for the converted corn field

Date	Input/Action	Rate
April 1 - 5	Disked	Two times
April 27	Moldboard plowed	Once
May 15	Disked	3 passes
May 27	Fertilizer: 21.33-12.08-12.08-4.43-2.13Mg-0.352n-0.07B	448 kg/ha
May 28	Grazing corn: PS2142 RR	79,000 seeds/hectare
June 25	Glyphosate	3.3 L/hectare

Soil Collection, Processing and Analysis

Determination of Soil Measurements

Before soil was collected, and based on the available literature, we selected a minimum suite of biological, chemical and physical soil health indicators to best determine the soil quality and functionality of the four treatments (Bagnall et al., 2023; Liptzin et al., 2023; Sainju et al., 2022).

Table 3 | Soil health indicators categorized by biological, chemical and physical attributes

Biological	**Chemical**	**Physical**
Carbon mineralization potential	pH	Bulk density
β-Glucosidase (BG)	EC	Soil aggregate stability
Cellobiohydrolase (CB)	Total soil carbon	
Phosphatase (PHO)	Soil organic carbon	
β-1,4-N-acetylglucosaminidase (NAG)	Total soil nitrogen	
Leucine aminopeptidase (LAP)	Ammonium	
	Nitrate	

Soil Collection

Initial soil samples were collected from October 10 to 12, 2023, to optimize collection and analysis protocols. Based on insights from these preliminary efforts, a second sampling was conducted from September 23 to 25, 2024, employing additional sample points and refined methods, with all subsequent analyses performed on the 2024 samples. It is important to note that soil for enzymatic activity analysis was exclusively collected during this second sampling. In both rounds, samples were taken from two depth intervals: 0 to 15 centimeters and 15 to 30 centimeters at the designated sample points.

Bulk density soils were collected using a core sampler attached to a slide hammer. The inner collar volume of core was 102.963 cm^3. In-field core collection was deposited into a medium Ziplock bag (Carter, 2007).

Soils for the analysis of carbon mineralization potential, and all chemical indicators, were collected using an Edelman auger, 5 cm type. Fifteen and thirty centimeters were marked on the auger with tape. Soils were collected from two, closely spaced holes from each depth to get sufficient soil for all indicator analyses. Soils were deposited into medium sized Ziplock bags (Wu et al., 2021).

Soil aggregates were collected from the sides of the auger-excavated holes at the two depths using a garden trowel. A chunk of undisturbed soil was removed from the wall and placed on a tarp, and the aggregates were then transferred carefully to a 50 mL flat-bottomed vial (Rieke et al., 2022).

Soils for enzyme activity analyses were collected from the hole sides at both depths using a dedicated garden trowel. The soil was placed in a 50 mL centrifuge vial, and the trowel was sanitized with alcohol wipes between samples. All vials were immediately placed in a cooler filled with ice packs. Soils were stored in the freezer until ready for analysis.

Soil Processing and Analysis

Bulk Density

Following the protocol of Ellert et al., 2006, bulk density samples were placed in 15.24-cm-diameter aluminum pie plates and oven-dried at 105°C for 24 hours. Empty plates were weighed before filling, and the filled plates were weighed both before and after drying to determine percent soil moisture. After drying, samples were sieved through a 2 mm mesh, and the weights of the <2mm fine fraction and the >2mm coarse material were measured. Additionally, the fine root mass within the coarse fraction was quantified. Whole soil bulk density (Eq. 1) and fine soil bulk density (Eq. 2) were then calculated as follows:

$$1)\ Whole\ soil\ bulk\ density = \frac{Dry\ weight\ of\ whole\ sample}{Collar\ inner\ volume}$$

$$2)\ Fine\ soil\ bulk\ density = \frac{Dry\ weight\ of\ whole\ sample \times Mass\ proportion\ of\ fine\ soil < 2mm}{Collar\ inner\ volume}$$

Soil Aggregates

Aggregates were oven dried at 30ºC until dry, with vial lids removed. Wet aggregate stability was assessed using the SLAKES smartphone application, which measures dissolution of aggregates in water over time. Three pea-sized soil peds were removed from the dried sample vial and placed into a petri dish of deionized water. A smartphone was positioned above the dish, and the app was run for 10 minutes until an on-screen measurement was obtained. This procedure was repeated three times for each soil sample (Flynn et al., 2020).

Air Drying of Soils

All soils for the following analyses of chemical indicators and carbon mineralization were prepared uniformly. Samples were placed in 22.86-centimeter-diameter aluminum pie plates and air-dried in a dedicated room within a temperature-controlled greenhouse until completely dry. Once dried, the samples were sieved through a 2 mm mesh and ground using a two-roll grinding mill (Karla et al., 1991).

pH and Electrical Conductivity

Soil pH and electrical conductivity (EC) were measured simultaneously using methods outlined in the *Canadian Society of Soil Science's Manual on Soil Sampling and Methods of Analysis* (McKeague, 1978). A 2:1 water-to-soil ratio was used, with 20 mL of water added to 10 g of soil. The mixture was shaken four times at regular intervals over 30 minutes, then allowed to settle for an additional 30 minutes. Measurements were recorded using a Fisherbrand Accumet AB200 dual-parameter meter, with a pH probe and conductivity probe inserted into the sample.

Total Soil Carbon (%) and Total Soil Nitrogen (%)

Air-dried, sieved, and ground soils were analyzed using a Leco CHN 628 Determinator (Leco Corp, MI, USA). Samples were combusted in a dual-stage furnace at 1000°C in a pure oxygen environment to ensure complete combustion. A three-point linear calibration procedure, employing LECO and Elemental Microanalysis Certified Soil Reference Materials spanning the expected C and N ranges, was used to standardize and monitor analytical performance. QA/QC protocols included duplicate samples as well as both in-house and certified reference standards (USDA NRCS, 2014; Wright et al., 2001).

Soil Organic Carbon (SOC)

Soil organic carbon was determined by subtracting soil inorganic carbon (SIC) measurements from total soil carbon measurements. The SIC protocol was adapted from Horvath et al. (2005) and Rochette et al. (2003). A known quantity of soil was placed in a sealed container and treated with 6N HCl to liberate inorganic carbon as CO_2; the resulting pressure was measured using a digital manometer. Pressure data from each sample were compared against a 9-point calibration curve constructed from pure, dry reagent calcium carbonate exposed to the same conditions. Quality control procedures included blanks, in-house soils with known SIC values, and sample duplicates. Samples with pressure readings near those of the blanks or the lowest calibration standard were considered below the method detection limit, and any SIC values calculated as negative were reported as 0.000% SIC.

Ammonium (NH_4^+) and Nitrate (NO_3^-)

Ammonium (NH_4^+) and nitrate (NO_3^-) were extracted from 5 g of air-dried, sieved soil using 50 mL of 2 M KCl. The suspension was shaken to facilitate extraction and then either filtered or allowed to settle before collecting the supernatant. Extracted NH_4^+-N and NO_3^--N concentrations were measured using a Lachat QuickChem 8500 Series 2 Flow Injection Analyzer, which reported nitrogen concentrations (ppm) on a solution basis. To express these values on a soil basis (ppm), raw instrument readings were multiplied by 10 to account for the 1:10 soil-to-solution ratio.

Soil Carbon Mineralization Potential

Soil carbon mineralization potential was assessed at the 0–15 cm depth, following the protocol of Franzluebbers et al. (2000) and the *Soil Health Institute's standard operating procedures for potential carbon mineralization* (2022). Thirty grams of air-dried soil were placed in an aluminum weigh boat (perforated for gas exchange) atop a filter paper-lined 250 mL mason jar equipped with a silicone-sealed rubber injection port. Twenty milliliters of distilled water were added to moisten the soil from below, and the jars were sealed and incubated in a 25°C darkened chamber for 24 hours.

After 24 hours, CO_2 production was measured using an EGM-4 portable CO_2 gas analyzer. A 25-gauge needle attached to a 30 mL syringe was inserted into the injection port, and 30 mL of headspace gas was withdrawn and injected into the analyzer. CO_2 concentrations were recorded, and results were expressed as carbon mineralization potential.

Calculations were made with the following assumptions: (1) data comparability was prioritized over absolute accuracy; (2) no corrections were made for temperature or pressure; (3) 30 g of sieved, air-dried soil was assumed to occupy 20 mL (based on a density of 1.5 g/cm^3), so that in a 250 mL jar the soil leaves 230 mL of air space; and (4) an EGM reading of 100 ppm CO_2 was assumed to correspond to approximately 100 mg CO_2/L, or 27 mg CO_2-C/L (given that carbon makes up about 27% of CO_2's mass).

CO_2 production was calculated as (Eq. 3):

$$3)\ \left(CO_2\left(\frac{mg}{L}\right)_{24-hr} - CO_2\left(\frac{mg}{L}\right)_{0-hr}\right) \times 0.23L \ x \ 0.27\ mg\ C/mg\, CO_2 \div 0.030\ kg \div 24hrs$$

Soil Enzymatic Activity

Soil enzyme activity was measured following the fluorescence enzyme assay protocol adapted from Steinweg et al. (2012). The enzymes assayed included β-d-cellobioside (CB), β-Glucosidase (BG), leucine aminopeptidase (LAP), N-acetyl-β-Glucosaminidase (NAG), and phosphatase (PHO), which were selected as indicators of soil health and key players in nutrient cycling (Marx et al., 2001; Fan et al., 2024).

For substrate preparation, 4-MUB-β-d-cellobioside, 4-MUB-β-d-glucopyranoside, L-leucine-7-amido-4-methylcoumarin hydrochloride, 4-MUB-N-acetyl-β-D-Glucosaminidase, and 4-MUB phosphate were each dissolved in 200 mL of deionized water to achieve a target concentration of 250 µM. Two synthetic fluorescent indicators—4-methylumbelliferone (MUB) and 7-amino-4-methylcoumarin (MUC)—were employed as the products; MUC was used exclusively for LAP analysis, while MUB was used for the remaining enzyme assays. MUC and MUB were initially prepared in methanol at target concentrations of 1 mM and 2 mM, respectively. Working stocks for calibration were then prepared by diluting these concentrated solutions in deionized water to a final concentration of 100 µM in a total volume of 50 mL, followed by serial dilutions to yield standard solutions with final concentrations of 50 µM, 25 µM, 12.5 µM, 6.25 µM, 3.125 µM, and a blank (0 µM).

Soil samples were prepared by adding 1 g of soil to 100 mL of 50 mM sodium acetate buffer and homogenizing for one minute. 150 µL of the soil slurry was dispensed into microplate wells, with each well subsequently receiving 150 µL of either substrate or standard solution, ensuring four technical replicates and a standard curve for all samples.

Filled plates were incubated at 25°C for three hours, after which each plate was transferred to the Agilent BioTek Synergy HTX Multi-Mode Microplate Reader for fluorescence measurements (excitation: 365 nm; emission: 450 nm). Calibration curves were constructed by converting the standard dilutions from µM to µmoles, plotting fluorescence intensity versus µmoles, and accepting only curves with an $R^2 \geq 0.99$.

Enzyme activity was calculated as follows:

1. Determine concentration from fluorescence (Eq. 4)**:**

$$4)\ y = mx + b \ \rightarrow \ x = \frac{y - b}{m}$$

where y is the measured fluorescence intensity, m is the slope, and b is the intercept of the calibration curve (with x in µmoles).

2. Calculate total µmoles in the slurry (Eq. 5)**:**

$$5)\ Total\ moles = x \times V_{buffer}$$

where V_{buffer} is the total volume of the buffer used in the soil slurry.

3. Normalize to obtain enzyme activity per gram of soil per hour (Eq. 6)**:**

$$6)\ Enzyme\ activity\ \left(\frac{\mu\text{mol}}{g \cdot hr}\right) = \frac{Total\ \mu\text{moles}}{t\ \times\ m_{soil}\ \times\ V_{well}}$$

where t is the incubation time (in hours), m_{soil} is the mass of dry soil (in grams) in the slurry, and V_{well} is the volume of slurry dispensed in each well.

4. Convert to obtain enzyme activity in nmol per gram of soil per hour (Eq. 7)**:**

$$7)\ Enzyme\ activity\ \left(\frac{nmol}{g\ \cdot hr}\right) = Enzyme\ activity\ \left(\frac{\mu mol}{g\ \cdot hr}\right)\ \times 1000$$

Crop Yield (Corn Biomass)

Corn sampling was conducted concurrent with soil sampling from September 23–25, 2024. Yield estimates followed protocols established by Iowa State University (2020). At three of the six plot points per block, a 5.31-m row length was measured northward from each designated sampling point. This length represents 1/1000th of an acre at 76.2-cm row spacing. Plant density was determined by counting all plants within this length. Five representative plants were harvested at 15 cm above ground level, processed into smaller segments (including cobs), and stored in paper bags for subsequent analysis.

In the laboratory, the wet biomass was first recorded, and samples were subsequently oven-dried at 60°C until a constant weight was achieved. The dried samples were then reweighed to determine total dry biomass, average dry biomass per plant, and percentage dry matter.

The estimated dry matter yield (kg/ha) was calculated for each plot point using the following equations (Eq. 8) :

$$8)\ Number\ of\ plants/hectare = \left(\frac{Number\ of\ plants\ in\ 5.31\ m\ row\ \ \times 1000 \times 2.47}{hectare}\right)$$

where 1000 is the conversion factor from 1/1000th of an acre to a full acre, and 2.47 is the conversion factor from acres to hectares (Eq. 9).

$$9)\ \text{Yield} = \text{Average dry weight/plant (kg)} \times \text{Number of plants/hectare}$$

These calculations provided an estimate of corn dry matter yield per hectare for each plot point.

Corn Quality Metrics

Crude Protein

The crude protein content of corn components (stalk, cob, and kernel) was determined using a LECO 828 C/N Analyzer using protocol determined by the manufacturer. The instrument was calibrated using a linear, forced-through-origin calibration with EDTA LCRM (502-896, Lot 1001) at fractional masses ranging from 0.1 g to 0.3 g, with a minimum of five replicates to ensure accuracy. Dried corn components were ground, and approximately 0.25 g of material was weighed into 502-186 tin foil cups for analysis. Total nitrogen values were multiplied by a conversion factor of 6.25 to calculate crude protein percentage, following standard protocols for plant tissue analysis (FAO, 2003).

Acid Detergent Fibre (ADF) and Neutral Detergent Fibre (NDF)

ADF content was determined using an ANKOM A200 Fiber Analyzer following the filter bag technique and using protocol determined by the manufacturer. Dried and ground corn components (stalk, cob, and kernel) were weighed (0.45–0.50 g) into F57 filter bags (ANKOM Technology), which were weighed both before and after filling. The bags were then heat-sealed and loaded into the analyzer vessel. Samples were extracted with an acid detergent solution (20 g/L cetyl trimethylammonium bromide in 1.00 N H_2SO_4) at 100°C for 60 minutes with constant agitation. Following extraction, the filter bags were rinsed with hot water, soaked in acetone for 3 – 5 minutes, and oven-dried at 102 ± 2°C until a constant weight was achieved. ADF content was calculated as the percentage of residual mass relative to the initial sample mass, corrected for the bag blank.

Neutral detergent fiber (NDF) content was determined using an ANKOM A200 Fiber Analyzer following the filter bag technique and using protocol determined by the manufacturer. Dried and ground corn components (stalk, cob, and kernel) were weighed (0.45–0.50 g) into F57 filter bags (ANKOM Technology), which were recorded both pre- and post-filling, then heat-sealed and loaded into the analyzer vessel. Samples were extracted with a neutral detergent solution consisting of 30 g sodium dodecyl sulfate (USP), 18.61 g ethylenediaminetetraacetic disodium salt (dihydrate), 6.81 g sodium borate, 4.56 g sodium phosphate dibasic (anhydrous), and 10 mL triethylene glycol in 1 L of distilled water, with the addition of sodium sulfite (0.5 g per 50 mL of solution) and 4.0 mL of alpha-amylase. The extraction was performed at 100°C with agitation for 75 minutes. Following extraction, samples were rinsed twice for 5 minutes with hot water containing alpha-amylase and once more for 5 minutes with hot water only. The bags were then soaked in acetone for 3–5 minutes and subsequently oven-dried at 102 ± 2°C until a constant weight was achieved. NDF content was calculated as the percentage of residual mass relative to the initial sample mass, corrected for the bag blank.

Total Ecosystem Carbon

Total ecosystem carbon (TEC) was quantified at each sample point across the four treatments by summing carbon pools from trees, shrubs, herbaceous vegetation (grasses and forbs), LFH layer, and soil organic carbon stocks. TEC was expressed as megagrams of carbon per hectare (Mg C ha^{-1}). In the converted forest and converted grassland treatments, TEC consisted solely of soil organic carbon stocks, as the corn biomass was removed through either grazing or silage harvest. Any residue left after harvest and grazing was incorporated in the spring and therefore captured within the soil organic carbon measurements rather than counted as a separate carbon pool.

Tree Carbon

Tree carbon was quantified using LiDAR data and validated through ground-truthing protocols. At each sample point, the point-quarter method (Cottam and Curtis, 1956) was used to estimate tree diameter at breast height (DBH) for the nearest tree in each quarter. DBH was measured at 1.3 m, with trees <3 cm DBH excluded. Tree species were recorded for each measured tree, and tree height was determined using a Suunto PM-5/360PC clinometer (Finland).

LiDAR data was acquired using a DJI Zenmuse L1 LiDAR sensor mounted on a DJI M300 remotely piloted aircraft system (RPAS). Terrain-following technology was utilized to maintain a constant ground sampling distance, minimizing image distortion and ensuring the safety of the RPAS. The RPAS was flown at a height of 30 m above vegetation. DJI Terra (version 3.5.5) was used to convert the proprietary file format to an open-sourced LAS file for use in other programs (i.e.: R statistical software).

Data processing of the LAS file included noise reduction using an isolated voxel filter (IVF) with a 4 m^3 voxel size and an isolation number of 5. Ground points were classified using a cloth simulation function (CSF). Tree heights were measured by normalizing the point cloud with a triangulated irregular network (TIN), and treetops were identified using a local maximum filter (LMF) with an automatically adjusting window size (Roussel, 2025). Ground-truthing data was used to develop a linear model for predicting tree DBH within blocks marked on the map. Using this model and the predict function in R, DBH was estimated for each individual tree in the forest treatment.

Tree biomass was calculated using the FAIBbase package in R, which applies the allometric equations of Lambert et al. (2005) (Province of British Columbia, 2019) (Eq. 10):

$$10)\ y_i = \exp\left(\beta_{l,i} + \beta_{2,i} lnD + \beta_{3,i} lnH\right)$$

where y_i is the dry mass of compartment i (Pseudotsuga menziesii) in kg, D is diameter at breast height (cm), and $\beta_{l,i}$, $\beta_{2,i}$, and $\beta_{3,i}$ are fitted parameters for compartment i.

Belowground biomass was estimated as 22.2% of aboveground biomass (Addo-Danso et al., 2016), and carbon concentration in tree tissues was assumed to be 50% of total biomass (Paré et al., 2013).

Understory and Litter, Fibric, Humic Layer (LFH) Carbon

At each sample point, a 1 m^2 quadrat was placed, and the percent cover of all plant species was estimated. Woody shrubs <2 m tall were clipped and collected in paper bags. Within the center of this quadrat, a 0.5 m^2 quadrat was placed, and all grasses and forbs were clipped and collected in paper bags. Finally, a 0.25 m^2 quadrat was positioned at the center, where litter was removed down to the mineral soil layer and placed in a paper bag.

Shrub, grass/forb, and LFH layer materials were oven-dried at 60°C in a Quincy Lab Model 31-350ER bench oven until a constant weight was achieved. Dried samples were weighed to the nearest 0.01 g using a Fisher Scientific Accu-4102 scale. Total shrub and grass/forb biomass (above- and belowground) was estimated as 2.5 and 3 times the aboveground biomass, respectively (Johnston et al., 1996). Carbon concentration was assumed to be 48% of total biomass for shrubs and 45% for grasses/forbs (Vogel & Gower, 1998).

After drying, LFH materials were ground using a processor and passed through a 2 mm sieve. Organic carbon content was determined using a ThermoFisher CHNS FlashSMART elemental analyzer, following guidelines laid out by ThermoFisher Scientific, ISO 10694, and the Official Italian Method (Gazzetta Ufficiale, 1999). Approximately 10 mg of the prepared LFH sample was placed in a silver sample container and loaded into the total organic carbon (TOC) block alongside all other samples. To remove inorganic carbon, 50 µL of a 10% HCl solution was added to each silver container in a 1:1 ratio (volume) with the sample and left to dissolve for 4 hours. Samples were then placed in a dry bath at 65°C for 16 hours to ensure complete drying. Once dried, the capsules were folded, loaded into the appropriate positions in the autosampler wheel, and analyzed.

Quality control (QC) was maintained through blank samples to subtract the elemental composition of the samples, calibration standards to verify instrument accuracy, and QC samples analyzed at regular intervals to monitor instrument stability. Organic carbon content was reported as a percentage and extrapolated to calculate total LFH carbon stocks in megagrams (Mg) per hectare. The standard curves showed an $R^2 > 0.999$ for carbon, nitrogen and hydrogen, demonstrating optimal performance of the instrument.

Soil Organic Carbon Stock (SOC stock)

Soil organic carbon stocks were quantified following Ellert et al. (2007), using the fixed-mass method to correct for treatment-driven bulk density differences that can otherwise under- or overinflate SOC stock estimates. SOC stocks were calculated for each land-use pairing (forest with converted forest and grassland with converted grassland) to isolate the effects of each land-use change.

First, determine SOC stock (fixed depth) (Eq. 11):

$$11)\ \mathrm{SOC_{FD}} = \sum_{1}^{\mathrm{n}} \mathrm{D_{CS}}\,\mathrm{C_{CS}}\mathrm{L_{CS}} \times 0.1$$

where SOC_{FD} is the SOC stock to a fixed depth (Mg C ha^{-1} to the specified depth), Dcs is the density of core segment (g cm^{-3}), Ccs is the organic C concentration of core segment (mg C g^{-1} dry soil), and Lcs is the length of core segment (cm).

Next, determine SOC stock (fixed mass).

1. For all samples, calculate the mass of soil to the designated depth (Eq. 12):

$$12)\ \mathrm{M_{Soil}} = \sum_{1}^{n} D_{CS}\,L_{CS} \times 100$$

where M_{soil} is the mass of soil to a fixed depth (Mg ha^{-1}).

2. Select, as the reference, the lowest soil mass to the prescribed depth from the intact site (M_{ref}).
3. Calculate the mass of soil to remove from each fixed-depth core measurement so that soil mass is equivalent across all sampling sites (Eq. 13):

$$13)\ M_{ex} = M_{soil} - M_{ref}$$

where M_{ex} is the excess mass of soil, to be subtracted from fixed depth core measurement.

4. For each sampling site, calculate SOC stock to fixed mass (Eq. 14):

$$14)\ SOC_{FM} = SOC_{FD} - M_{ex} \times C_{SN}/1000$$

where SOC_{FM} is the SOC stock fore a fixed mass of M_{ref} and C_{SN} is SOC concentration in the deepest soil core segment (mg C g^{-1} dry soil).

Statistical Analysis

Measuring Changes in Soil Health Indicators.

Statistical analyses were conducted using R statistical software. Ordinary least squares (OLS) regression models were fitted to assess the effects of treatment and sample depth, including their interaction, on all soil variables measured in 2024. Sample depth was excluded from analyses of carbon mineralization potential across treatments. Model assumptions were evaluated using diagnostic checks, including residual simulations (DHARMa package). Data were log-transformed when assumptions of normality and homoscedasticity were not met. Two-way analysis of variance (ANOVA) was performed on the fitted models to determine the significance of treatment, sample depth, and their interaction, while one-way ANOVA was used for carbon mineralization potential. Estimated marginal means were computed for treatment and treatment-sample depth combinations, with post-hoc pairwise comparisons conducted using Tukey's adjustment. Significant differences between treatments ($\alpha = 0.05$) were identified using compact letter display grouping. Differences between treatments were expressed as percent changes—calculated as $(\exp(\Delta) - 1) \times 100\%$, where Δ is the difference between estimated marginal means on the log scale.

Soil Quality Index (SQI)

Statistical analyses of the SQI were performed in R. The equations used to calculate the SQI are listed below under their respective section headings.

The soil quality index was assessed for 2024 at the 0–15 cm depth, as this dataset was the most complete and contained all relevant variables. Only the converted forest and converted grassland were assessed for differences in soil quality, as comparing converted land types was considered more informative than comparing them to their parent soils, where obvious changes were expected. To ensure model robustness, both linear and nonlinear indices were modeled.

Selection of Minimum Data Set (MDS)

Principal component analysis (PCA) was conducted to identify key factors, retaining only components with eigenvalues greater than 1 and explaining more than 5% of the variance (Andrews et al., 2002a,b; Brejda et al., 2000; Shukla et al., 2004; Wander and Bollero, 1999). Subsequently, a varimax-rotated factor analysis was performed to obtain factor loadings, and soil attributes with absolute loadings within 10% of the maximum for each factor were retained. When multiple attributes were retained under the same factor, a Spearman correlation matrix was computed; for attributes exhibiting high correlation ($|r| \geq 0.70$), the one with the lower average loading was considered redundant and removed (Andrews et al., 2002b).

Transformation of MDS Indicators

After identifying the minimum data set (MDS) indicators, all observations were transformed using linear (L) and non-linear (NL) scoring functions (Liebig et al., 2001; Andrews et al.,2002b, 2004). Indicators were classified into three categories based on their assumed relationship with soil quality: “more is better,” “less is better,” and “optimal range.”

The corresponding linear scoring functions applied were: "more is better" (Eq. 15), "less is better" (Eq. 16), and "optimal range" (Eq. 17).

$$15)\ L(Y) = \frac{x}{x_{max}}$$

$$16)\ L(Y) = \frac{x_{min}}{x}$$

$$17)\ L(Y) = \begin{cases} 1, & \dfrac{if\ x_{low} \leq x \leq x_{high}}{otherwise} \\ 0, & \end{cases}$$

where L(Y) is the linear score ranging from 0 and 1, x is the observed soil indicator value, and x_{max} and x_{min} are the maximum and minimum value of each soil indicator, respectively. For Eq. 3, if x falls within the predefined optimal range [x_{low}, x_{high}], it gets a score of 1; otherwise, it is assigned a score of 0.

The non-linear (sigmoidal) scoring functions were similarly applied based on the assumed relationship between each indicator and soil quality: "more is better," "less is better" (Eq. 18), and "optimal range" (Bastida et al., 2006; Masto et al., 2008) .

$$18)\ \mathrm{NL(Y)} = \frac{\mathrm{a}}{1 + \left(\frac{\mathrm{x}}{\mathrm{x_0}}\right)^{\mathrm{b}}}$$

where NL(Y) is the non-linear score ranging from 0 to 1, a is the maximum value of the function defined as a=1, x is the observed indicator value, x_0 is the mean value of each indicator for the respective land conversion type, and b controls the slope of the function. The slope parameter was set to -2.5 for "more is better" indicators and +2.5 for "less is better" indicators.

For "optimal range" indicators, the following function was applied (Eq. 19 and Eq. 20):

If x ≤ m, then:

$$19)\ 1/(1 + (x/m)^{-2.5})$$

If x > m, then:

$$20)\ 1/(1 + (x/m)^{2.5})$$

where m is the midpoint of the optimal range. If x is near m, the score approaches 1 (indicating optimal soil conditions), while deviations above or below m result in a lower score, approaching 0.

Weighting of MDS Indicators

Once transformed, weights were assigned to each MDS indicator based on factor loadings and variance explained using the following equation (Eq. 21) (Andrews et al.,2002a,b) :

$$21)\ w_{ij} = \left(\frac{|loading_{ij}|}{\sum |loadings_i|}\right) \times \left(\frac{variance\ explained_i}{\sum variance\ explained}\right)$$

where W_{ij} represents the weight assigned to indicator *j* within factor *i*, $loading_{ij}$ is the absolute factor loading of indicator *j*, and variance $explained_i$ is the proportion of total variance accounted for by factor *i*.

Calculation of SQI

The SQI was computed as a weighted additive function, incorporating the selected indicators using the following equation (Eq. 22) (Andrews et al.,2002a,b) :

$$22)\ \mathrm{SQI} = \sum_{i=1}^{n} W_i S_i$$

where W is the weighting factor for the soil indicator derived from the factor analysis, and S is the corresponding soil indicator score, derived using either the linear (L-SQI) or non-linear (NL-SQI) transformation. Higher SQI values were interpreted as indicating better soil function.

Nutrient Cycling Based Soil Multifunctionality (SMF)

Soil multifunctionality (SMF) was assessed for 2024 at the 0–15 cm depth and only in the two converted sites. To quantify SMF and evaluate the magnitude of soil function, Z-scores were calculated for each of the five tested soil enzymes (Fan et al., 2024). Multifunctionality for individual nutrient cycles was determined using the following equations (Eq. 23, Eq. 24, Eq. 25):

$$23)\ CCM = \frac{Z_{BG} + Z_{CB}}{2}$$

$$24)\ NCM = \frac{Z_{LAP} + Z_{NAG}}{2}$$

$$25)\ PCM = \frac{Z_{LAP} + Z_{PHO}}{2}$$

where CCM represents carbon cycle multifunctionality, NCM represents nitrogen cycle multifunctionality, and PCM represents phosphorus cycle multifunctionality.

The overall SMF score was then computed as the mean of these three sub-indices (Eq. 26):

$$26)\ SMF = \frac{CCM + NCM + PCM}{3}$$

A linear model (ANOVA) was fitted to test for significant differences in SMF across treatments. Model assumptions were evaluated using DHARMa residual simulations and diagnostic checks. Estimated marginal means (EMMeans) were computed for each treatment, and pairwise comparisons were conducted with Tukey's adjustment for multiple comparisons. Treatment differences were further visualized using compact letter display (CLD), where treatments sharing the same letter were not significantly different at $\alpha = 0.05$. All statistical analyses were conducted in R software.

Total Ecosystem Carbon (TEC)

Total ecosystem carbon (TEC) for 2024 was calculated by summing the mean stocks of five component pools, soil organic carbon (SOC), litter C, shrub C, grass/forb C, and tree C, for each land-use treatment. Only SOC stocks were formally tested for significance using paired t-tests; all other pools were represented by their mean values. I then computed percentage changes in both TEC and SOC stock for converted versus intact treatments to quantify the impact of land conversion. Because the aim was to document and compare total carbon stocks across treatments rather than to test every pool for statistical differences, descriptive summaries provided a transparent and sufficient basis for assessing total carbon distribution.

Corn Yield and Quality Metrics

Corn yield and corn quality metric differences between conversion types were analyzed using a one-way analysis of variance (ANOVA), with treatment as the independent variable. Model assumptions were evaluated in R using DHARMa residual simulations and diagnostic checks form the performance package. Estimated marginal means (EMMeans) were computed for each treatment, and pairwise comparisons were conducted with Tukey's adjustment for multiple comparisons. Treatment differences were visualized using compact letter display (CLD), where treatments sharing the same letter were not significantly different at $\alpha = 0.05$.

RESULTS

Biological Soil Health Indicator Changes

Extracellular Enzymatic Activity

Enzymatic activities were consistently greater in the 0–15 cm layer than at 15–30 cm, higher in converted grassland than in converted forest, and elevated in both conversions relative to their intact counterparts (Figure 2.2). The sole exception was N-acetyl-glucosaminidase (NAG) activity in forest conversions, which showed no significant change. A significant treatment × depth interaction for all enzymes, except NAG, indicates that conversion effects on enzyme activity depend on soil depth (Table 4).

For β-glucosidase (BG) activity in the shallow layer, the converted grassland exhibited the highest levels, being 235% greater than those in the forest, 113% greater than in the grassland, and 65% greater than in the converted forest (Figure 2.2). In the deeper layer, although forest and grassland did not differ significantly, converted fest and converted grassland displayed 222% and 263% higher activity than forest and grassland, respectively (Figure 2.2).

Celliobioside (CB) activity followed a similar pattern. In the shallow layer, converted grassland was 93% higher than grassland, 166% higher than converted forest, and 200% higher than forest, while in the deeper layer it remained 46%, 116%, and 164% higher than converted forest, grassland, and forest, respectively (Figure 2.2).

Phosphatase (PHO) activity in the shallow layer was highest in converted grassland, which showed 74% higher activity than grassland, 41% higher than converted forest, and 148% higher than forest. In the deeper layer, converted forest and converted grassland exhibited 103% and 146% higher activity than forest and grassland, respectively (Figure 2.2).

For leucine aminopeptidase (LAP) activity, the converted grassland in the shallow layer exhibited 110% higher activity than forest, 45% higher than grassland, and 38% higher than converted forest. In the deeper layer, although forest and grassland were not statistically different, converted forest and converted grassland demonstrated 55% and 90% higher activity than their intact counterparts, respectively (Figure 2.2).

N-acetyl-glucosaminidase (NAG) activity was consistently lowest in the grassland. In the shallow layer, forest, converted forest, and converted grassland had 58%, 41%, and 77% higher activity than the grassland, respectively, with even more pronounced differences in the deeper layer (88%, 112%, and 114% higher, respectively) (Figure 2.2).

Table 4 | Results from 2-way ANOVA test, treatment (Converted forest, Converted grassland, Forest, Grassland) x sample depth (0 - 15 cm, 15 - 30 cm) for β-glucosidase (BG), celliobioside, phosphatase, leucine aminopeptidase and N-acetyl-glucosaminidase activity.

Enzyme		Df	Sum squares	Mean squares	F ratio	p value
BG						
	Treatment	3	52.56	17.52	81.12	< 0.001
	Sample depth	1	9.27	9.267	42.909	< 0.001
	Treatment*Sample depth	3	3.54	1.18	5.463	< 0.01
	Error	184	39.74	0.216		
CB						
	Treatment	3	34.79	11.598	34.359	< 0.001
	Sample depth	1	5.58	5.582	16.536	< 0.001
	Treatment*Sample depth	3	3.17	1.056	3.128	< 0.05
	Error	184	62.11	0.338		
PHO						
	Treatment	3	24.93	8.31	72.737	< 0.001
	Sample depth	1	5.21	5.21	45.604	< 0.001
	Treatment*Sample depth	3	1.52	0.507	4.434	< 0.01
	Error	184	21.02	0.114		
LAP						
	Treatment	3	13.179	4.393	36.327	< 0.001
	Sample depth	1	6.146	6.146	50.822	< 0.001
	Treatment*Sample depth	3	1.122	0.374	3.092	< 0.05
	Error	184	22.251	0.121		
NAG						
	Treatment	3	12.64	4.213	21.487	< 0.001
	Sample depth	1	6.24	6.237	31.811	< 0.001
	Treatment*Sample depth	3	1.02	0.34	1.733	0.612
	Error	184	36.07	0.196		

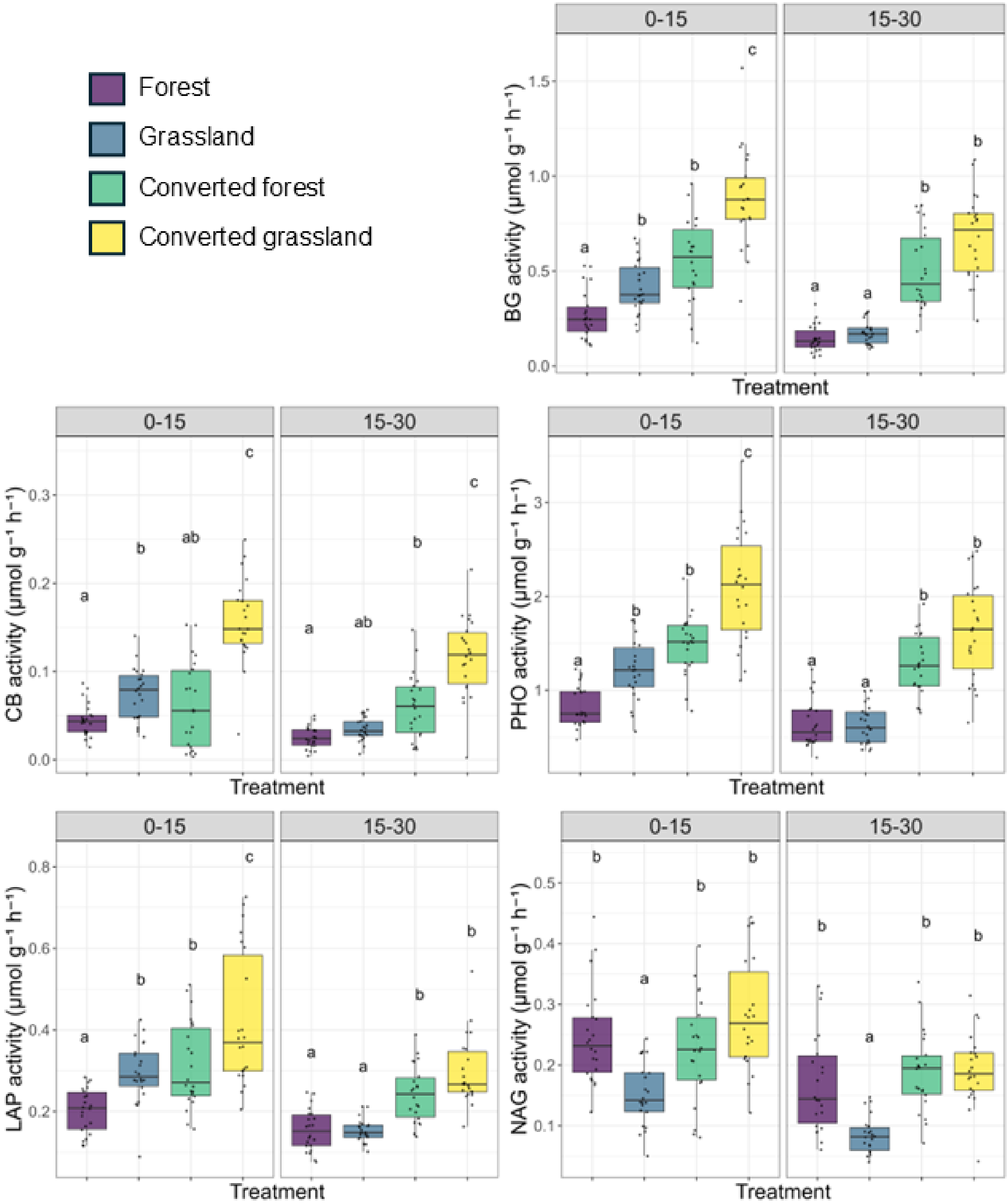

Figure 2.2. Log transformed enzymatic activity by treatment (Converted forest, Converted grassland, Forest, Grassland) x sample depth (0 - 15 cm, 15 - 30 cm) for β-glucosidase (BG), celliobioside (CB), phosphatase (PHO), leucine aminopeptidase (LAP) and N-acetyl-glucosaminidase (NAG) activity. Significance differences ($p<0.05$) are indicated by different letters according to Tukey's post-hoc test. Box plot whiskers represent data points that lie within 1.5 times the interquartile range (IQR) from the lower or upper quartile.

Soil Carbon Mineralization Potential (0-15 cm)

Among the intact treatments, the grassland exhibited 80% higher respiration than forest. Forest conversion soils showed an 80% increase in respiration relative to intact forest, whereas converted grassland soils exhibited approximately a 57% reduction in CO_2-C respiration compared to intact grassland. When comparing conversions, converted forest soils demonstrated a 132% higher respiration rate than converted grassland (Figure 2.3).

Table 5 | Results from 1-way ANOVA test on treatment (Converted forest, Converted grassland, Forest, Grassland) for measurements of soil respiration.

	Df	Sum squares	Mean squares	F ratio	p value
Treatment	3	12.98	4.326	10.76	< 0.001
Error	**92**	**36.99**	**0.402**		

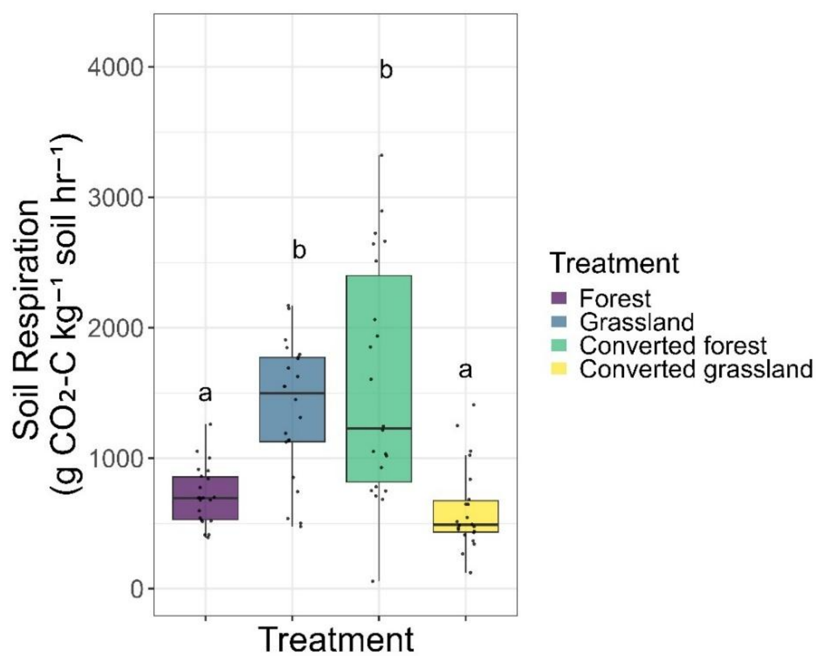

Figure 2.3. Treatment effects on log transformed soil CO_2-C respiration (g CO_2-C kg^{-1} soil hr^{-1}) in Forest, Grassland, Converted forest and Converted grassland. Significance differences ($p<0.05$) are indicated by different letters according to Tukey's post-hoc test. Box plot whiskers represent data points that lie within 1.5 times the interquartile range (IQR) from the lower or upper quartile.

Chemical Soil Health Indicator Changes

Two-way ANOVA results showed varying levels of significance for treatment × depth interactions across soil chemical indicators (Table 6).

The estimated marginal means of soil pH did not differ significantly between the shallow (0–15 cm; 6.26) and deeper (15–30 cm; 6.33) layers. However, within each depth, significant treatment effects were observed ($p < 0.001$). In the shallow layer, the forest treatment exhibited the lowest pH (5.90), while grassland, converted grassland, and converted forest showed significantly higher pH values, approximately 63%, 90%, and 36% greater than forest, respectively (Figure 2.4). In the deeper layer, pH in forest and grassland did not differ significantly from that in their corresponding conversion treatments (Figure 2.4).

Log transformed soil electrical conductivity (EC) data (dS m^{-1}) revealed significant effects of treatment ($p < 0.001$), sample depth ($p < 0.001$), and their interaction ($p < 0.05$). At both sampling depths, forest and grassland EC did not differ significantly; however, both conversion treatments exhibited significantly higher EC values than their respective parent ecosystems. In the shallow layer (0–15 cm), the converted forest showed 49% higher EC than forest, and converted grassland demonstrated approximately 62% higher EC than grassland, with no significant differences between the conversion treatments (Figure 2.4). In the deeper layer (15–30 cm), both conversions maintained significantly higher EC than their respective intact ecosystems, with converted grassland soils exhibiting 39% higher EC than converted forest soils (Figure 2.4).

Analysis of log transformed soil organic carbon concentration (SOC; g C kg^{-1} soil) revealed significant treatment effects at both depths ($p < 0.001$), while the treatment × depth interaction was not statistically significant. In the shallow layer (0–15 cm), the Forest treatment exhibited significantly lower SOC levels than grassland and both conversion treatments, with converted forest soils showing an approximate 98% increase in SOC relative to intact forest (Figure 2.4). Similarly, in the deeper layer (15–30 cm), converted forest soils demonstrated a 76% increase in SOC, while converted grassland soils did not differ significantly from intact Grassland. Both conversions at both depths did not differ significantly in SOC (Figure 2.4).

Log-transformed soil nitrogen data revealed a significant treatment × depth interaction ($p < 0.05$), with shallow soils (0–15 cm) exhibiting 83% higher nitrogen levels than deeper soils (15–30 cm) (Figure 2.4). At both depths, forest soils consistently had the lowest nitrogen levels compared to other treatments. In the shallow layer, grassland soils contained 280% more nitrogen than forest soils, while converted forest soils were 92% higher than intact forest (Figure 2.4). Although grassland and converted grassland soils did not differ significantly in the shallow layer, converted grassland soils exhibited approximately 92% more nitrogen than converted forest soils (Figure 2.4). At 15–30 cm, converted forest soils displayed a 61% increase in nitrogen over intact forest, and converted grassland soils had the highest nitrogen concentrations, being 48% and 92% higher than grassland and converted forest, respectively (Figure 2.4).

Log-transformed soil ammonium concentrations exhibited a similar treatment-by-depth effect ($p < 0.001$), with shallow soils having 52% higher ammonium levels than deeper soils. However, unlike nitrogen, converted forest and converted grassland treatments did not differ significantly from each other at either depth ($p > 0.05$). In the shallow soil layer, converted grassland showed 49% lower ammonium than intact grassland, while in the deeper layer, this pattern reversed with converted grassland showing 73% higher ammonium than its intact counterpart (Figure 2.4). Converted forest soils consistently showed higher ammonium levels than forest, with increases of 105% and 139% in the shallow and deeper layers, respectively (Figure 2.4).

Analysis of log-transformed nitrate concentrations revealed both significant depth ($p < 0.05$) and treatment ($p < 0.001$) effects, while the treatment × depth interaction was not significant. Averaging over treatments, shallow soils (0–15 cm) had nitrate concentrations 61% higher than those in deeper soils (15–30 cm). At 0–15 cm, all treatments differed significantly. Forest soils had the lowest nitrate levels, while intact grassland soils were approximately 15-fold higher than forest (Figure 2.4). Converted forest soils exhibited nitrate levels 52 times greater than those in forest, and converted grassland soils contained roughly 13-fold more nitrate than intact grassland (Figure 2.4). Notably, converted grassland soils showed about a 4-fold higher nitrate concentration than converted forest soils. In the 15–30 cm layer, forest again recorded the lowest nitrate levels; grassland soils were approximately 5-fold higher than forest soils (Figure 2.4). Converted forest soils showed a 32-fold increase over forest, while converted grassland soils were about 35 times higher than intact grassland (Figure 2.4). At this deeper depth, converted grassland soils had roughly 6.6 times the nitrate concentration of converted forest soils (Figure 2.4).

Analysis of log-transformed C:N ratios showed significant effects of treatment ($p < 0.001$) and depth ($p < 0.05$), without a significant treatment × depth interaction. Notably, C:N ratios were not significantly different between intact ecosystems and their converted counterparts. At 0–15 cm, forest and converted forest soils exhibited a C:N ratio approximately 85% higher than that of grassland and converted grassland soils, while at 15–30 cm the difference was even more pronounced, with forest and converted forest soils having a C:N ratio nearly 111% higher than Grassland and Converted grassland soils (Figure 2.4).

Table 6 | Results from 2-way ANOVA test, treatment (Converted forest, Converted grassland, Forest, Grassland) x sample depth (0 - 15 cm, 15 - 30 cm) soil chemical indicators, pH, EC, SOC concentration, total N, NH_4, NO_3 and C:N ratio. P values less than 0.05 are in bold.

Soil chemical indicator		Df	Sum squares	Mean squares	F ratio	p value
pH						
	Treatment	3	13.321	4.44	**37.497**	**< 0.001**
	Sample depth	1	0.239	0.239	2.022	0.157
	Treatment*Sample depth	3	0.363	0.121	1.022	0.384
	Error	184	21.79	0.118		
EC						
	Treatment	3	11.036	3.679	**77.568**	**< 0.001**
	Sample depth	1	0.935	0.935	**19.713**	**< 0.001**
	Treatment*Sample depth	3	0.634	0.211	**4.455**	**0.00483**
	Error	174	8.252	0.047		
SOC						
	Treatment	3	13.66	4.555	**22.313**	**< 0.001**

Soil chemical indicator	Df	Sum squares	Mean squares	F ratio	p value
Sample depth	1	12.2	12.201	**59.773**	**< 0.001**
Treatment*Sample depth	3	2.07	0.69	**3.381**	**0.0194**
Error	184	37.56	0.204		
Total N					
Treatment	3	55.03	18.343	**95.54**	**< 0.001**
Sample depth	1	17.38	17.377	**90.51**	**< 0.001**
Treatment*Sample depth	3	1.85	0.618	**3.22**	**< 0.05**
Error	184	35.33	0.192		
NH_4					
Treatment	3	35.99	11.998	**25.904**	**< 0.001**
Sample depth	1	8.29	8.29	**17.899**	**< 0.001**
Treatment*Sample depth	3	9.76	3.254	**7.026**	**< 0.001**
Error	184	85.22	0.463		
NO_3					
Treatment	3	650.1	216.72	**270.419**	**< 0.001**
Sample depth	1	11.1	11.09	**13.833**	**< 0.001**
Treatment*Sample depth	3	7.4	2.45	**3.058**	**0.02975**
Error	173	138.6	0.8		
C:N ratio					
Treatment	3	21.152	7.051	**191.974**	**< 0.001**
Sample depth	1	0.245	0.245	**6.681**	**0.0105**
Treatment*Sample depth	3	0.106	0.035	0.958	0.4138
Error	181	6.648	0.037		

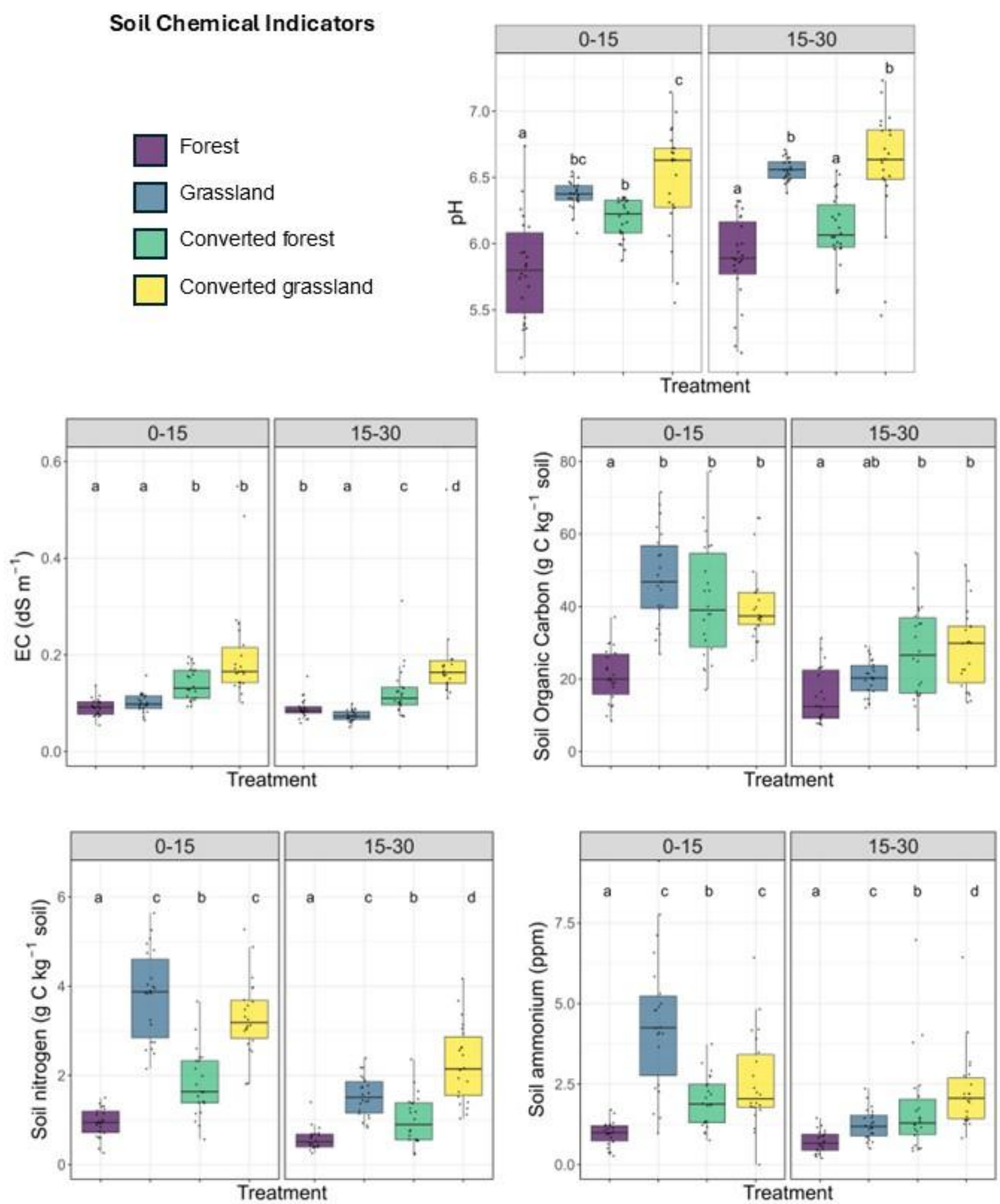

Figure 2.4. Log transformed soil chemical indicator measurements by treatment (Converted forest, Converted grassland, Forest, Grassland) x sample depth (0 - 15 cm, 15 - 30 cm) for pH, EC, SOC, total N, NH_4, NO_3 and C:N ratio . Significance differences ($p<0.05$) are indicated by different letters according to Tukey's post-hoc test. Box plot whiskers represent data points that lie within 1.5 times the interquartile range (IQR) from the lower or upper quartile.

Figure 2.4 continued.

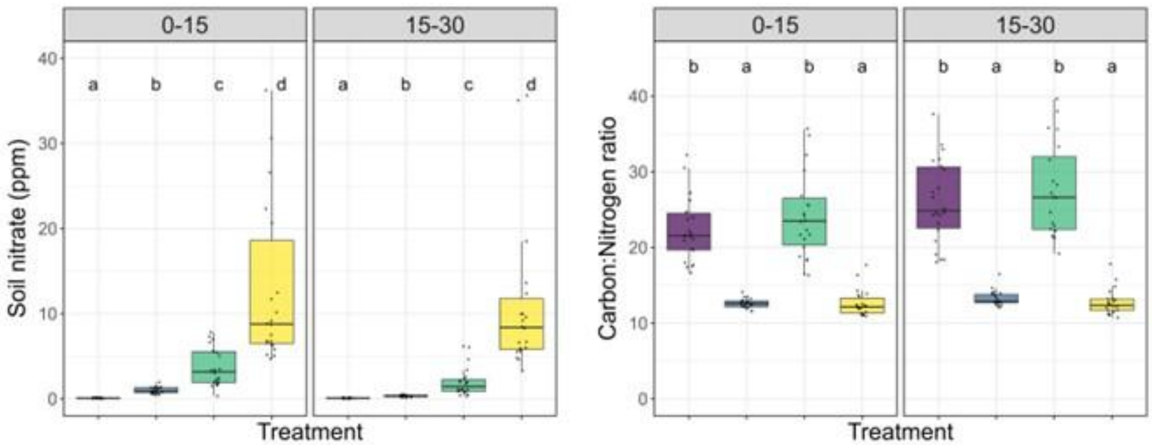

Figure 2.4. Log transformed soil chemical indicator measurements by treatment (Converted forest, Converted grassland, Forest, Grassland) x sample depth (0 - 15 cm, 15 - 30 cm) for pH, EC, SOC, total N, NH_4, NO_3 and C:N ratio . Significance differences ($p<0.05$) are indicated by different letters according to Tukey's post-hoc test. Box plot whiskers represent data points that lie within 1.5 times the interquartile range (IQR) from the lower or upper quartile.

Physical Soil Health Indicators Changes

Two-way ANOVA results showed varying levels of significance for treatment × depth interactions across physical soil indicators (Table 7).

In contrast to fine soil bulk density, which remained uniform, whole soil bulk density (g cm^{-3}) exhibited significant treatment ($p < 0.001$), depth ($p < 0.001$), and treatment × depth interaction ($p < 0.01$) effects. In the shallow layer (0–15 cm), conversion from forest to converted forest increased bulk density by 14.6%, while the converted grassland exhibited approximately 17.2% higher bulk density than the intact grassland; the two converted systems did not differ significantly at this depth (Figure 2.5). At 15–30 cm, the differences were more pronounced: the converted forest showed a 36.6% increase in bulk density relative to intact forest, whereas the converted grassland bulk density was 23.6% higher than in the intact grassland. Moreover, at this deeper depth, the converted grassland soils had 9.4% higher bulk density than in the converted forest soils (Figure 2.5).

In contrast to bulk density patterns, aggregate stability did not differ significantly between soil depths but showed marked differences between the forest and other treatments. Forest soils maintained significantly lower aggregate stability compared to all other treatments at both depths (Figure 2.5). Converted forest soils showed substantial increases in aggregate stability compared to forest soils: 40.3% higher at 0-15 cm and 43.5% higher at 15-30 cm (Figure 2.5). The remaining treatments (grassland and converted grassland) did not differ significantly from the converted forest at either depth (Figure 2.5).

Table 7 | Results from 2-way ANOVA test, treatment (Converted forest, Converted grassland, Forest, Grassland) x sample depth (0 - 15 cm, 15 - 30 cm) for whole soil bulk density (whole BD), fine soil bulk density (fine BD) and aggregate stability. P values less than 0.05 are in bold.

Soil physical indicator		Df	Sum squares	Mean squares	F ratio	p value
Whole BD						
	Treatment	3	2.221	0.7402	**18.248**	**< 0.001**
	Sample depth	1	1.605	1.6054	**39.576**	**< 0.001**
	Treatment*Sample depth	3	0.485	0.1618	**3.988**	**< 0.01**
	Error	184	7.464	0.0406		
Fine BD						
	Treatment	3	0.138	0.04599	1.915	0.129
	Sample depth	1	0.032	0.03213	1.338	0.249
	Treatment*Sample depth	3	0.069	0.02287	0.952	0.417
	Error	184	4.419	0.02402		
Aggregate stability						
	Treatment	3	2.076	0.6921	**14.609**	**< 0.001**
	Sample depth	1	0.006	0.006	0.127	0.722
	Treatment*Sample depth	3	0.038	0.0127	0.267	0.849
	Error	182	8.622	0.0474		

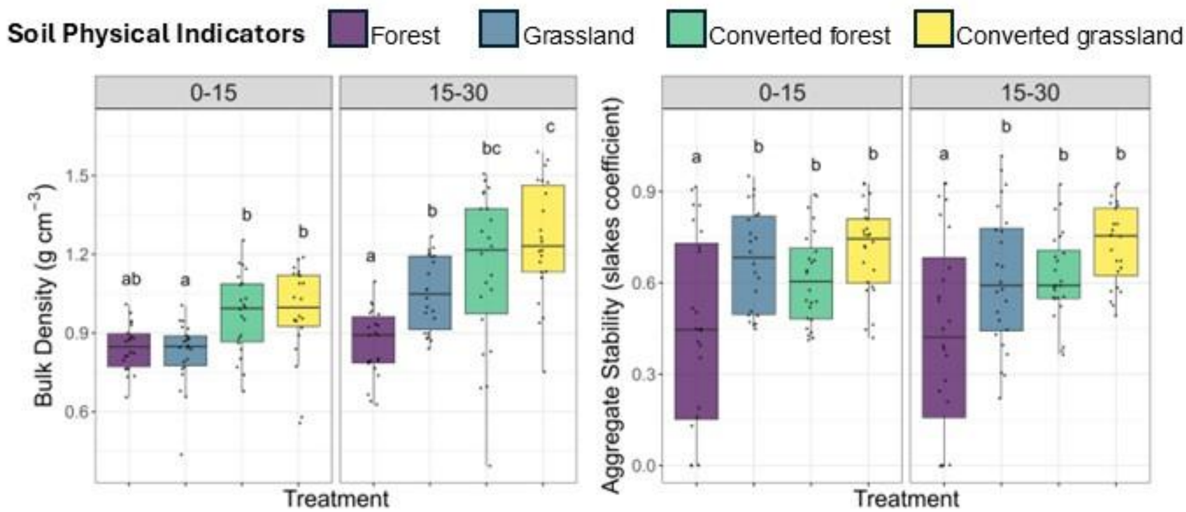

Figure 2.5. Soil physical indicator measurements by treatment (Converted forest, Converted grassland, Forest, Grassland) x sample depth (0 - 15 cm, 15 - 30 cm) for whole soil bulk density and aggregate stability . Significance differences ($p<0.05$) are indicated by different letters according to Tukey's post-hoc test. Box plot whiskers represent data points that lie within 1.5 times the interquartile range (IQR) from the lower or upper quartile.

Soil Quality Differences (SQI) Between Conversion Types

Factor analysis identified four main factors (RC1–RC4) that collectively explained 74.1% of the total variance in the soil data (Table 8). Variables with the highest loadings ($r > 0.7$) were then used to form a minimum data set (MDS) comprising whole soil bulk density, fine soil bulk density, electrical conductivity (EC), ammonium (NH_4^+), leucine aminopeptidase (LAP) activity, β-glucosidase (BG) activity, and soil respiration (Table 8). These MDS variables were subsequently weighted and categorized as "more is better" and "less is better" to reflect their roles in soil quality (Table 9).

Table 8 | Output of factor analysis (varimax rotation) showing explained variance, and factor loadings of soil physical, chemical and biological variables.

Variable	**RC1**	**RC3**	**RC4**	**RC2**
Whole soil bulk density	-	-	-	<u>**0.760**</u>
Fine soil bulk density	-	-	-	<u>**0.818**</u>
pH	-	-0.460	-0.562	-
EC	0.314	-	<u>**0.893**</u>	-
Aggregate stability	0.382	-	-	-
Soil N	**0.729**	-	0.323	-0.349
NH_4^+	<u>**0.761**</u>	-	-	-
NO_3^-/NO_2^-	-	-	**0.937**	-
Soil organic carbon (SOC)*	0.313	<u>**0.822**</u>	-	-0.305
LAP	<u>**0.814**</u>	-	-	-
CB	**0.622**	-	-	0.521
PHO	**0.734**	-	0.337	-
NAG	**0.802**	-	-	0.311
BG	<u>**0.814**</u>	-	-	0.351
Soil respiration	-	<u>**0.873**</u>	-	-
SOC stock	0.361	**0.830**	-	-
SS Loadings	**4.609**	**2.656**	**2.411**	**2.175**
Proportion Var (%)	**28.8**	**16.6**	**15.1**	**13.6**
Cumulative Var (%)	**28.8**	**45.4**	**60.5**	**74.1**

Note: Only loadings > |0.7| are shown. Dashes (-) indicate loadings < |0.3|. Underlined variables were retained in the minimum data set based on correlation analysis (r ≥ 0.70). Varimax rotation was applied to maximize the variance of squared loadings for each factor.

Table 9 | Soil quality indicator weights and scoring functions.

Indicator	**Weight**	**Scoring function curve used**	**Reference for use as SQ indicator**
NH_4^+	0.12388	More is better	Smith and Doran (1996), Wang et al., (2003)
LAP	0.13252	More is better	Fan et al. (2024)
BG	0.13255	More is better	Liptzin et al. (2022), Fan et al. (2024)
Soil SOC	0.10866	More is better	Andrews et al. (2002b; 2004)
Soil Respiration	0.11544	Less is better	Ouyang et al. (2015)
EC	0.20342	Less is better	Lenka et al. (2022)
Whole Soil Bulk Density	0.08843	Less is better	Lenka et al. (2022)
Fine Soil Bulk Density	0.09509	Less is better	Lenka et al. (2022)

The score values of the indicators were significantly different between linear and non-linear scoring methods ($p < 0.05$), except for β-glucosidase in converted forest (linear: 0.341, non-linear: 0.335, $p = 0.744$). The most pronounced differences were observed in soil respiration, where non-linear scores were substantially higher than linear scores in both converted forest (0.427 vs 0.0825) and converted grassland (0.792 vs 0.116) treatments ($p < 0.001$). Similarly, enzyme activities showed marked differences between scoring methods, with leucine aminopeptidase displaying significantly higher non-linear scores in both converted forest (0.352 vs 0.180) and converted grassland (0.504 vs 0.272) treatments ($p < 0.001$). The magnitude of these differences between scoring methods varied by treatment, suggesting that the choice of linear versus non-linear scoring substantially influences the final soil quality assessment.

A correlation analysis revealed several key relationships ($p < 0.05$) among the soil properties in the minimum data set (MDS) (Figure 2.6). Soil ammonium (NH_4^+) exhibited a strong positive correlation with soil organic carbon (SOC; r = 0.66), and it showed moderate positive associations with leucine aminopeptidase (r = 0.49) and β-glucosidase (r = 0.54). In contrast, soil respiration was moderately negatively correlated with both NH_4^+ (r = -0.46) and SOC (r = -0.56), suggesting that higher nutrient levels may be associated with reduced microbial activity. Similarly, electrical conductivity (EC) maintained a strong negative correlation with NH_4^+ (r = -0.66). Both whole and fine soil bulk density measures exhibited weak correlations with the other MDS variables; although the correlation between the two bulk density measurements was moderate, indicating partial redundancy.

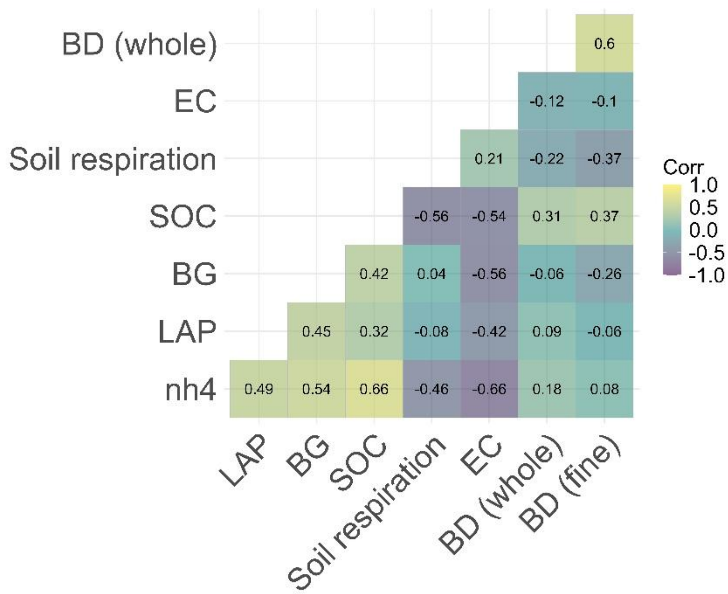

Figure 2.6**.** Correlation matrix of minimum data set (MDS) soil properties. NH_4^+ = ammonium, LAP = leucine aminopeptidase, BG = β-glucosidase, SOC = soil organic carbon, EC = electrical conductivity, BD = bulk density. Color intensity is proportional to correlation coefficients.

The results of the linear and non-linear Soil Quality Index (SQI) are shown in Figure 2.7. Both methods show the converted grassland treatment as having a significantly higher SQI (0.739 and 0.696 for non-linear and linear scoring, respectively) than the converted forest treatment (0.644 for both scoring methods) ($p < 0.05$). The magnitude of the difference between treatments is larger with non-linear scoring (0.095) compared to linear scoring (0.052).

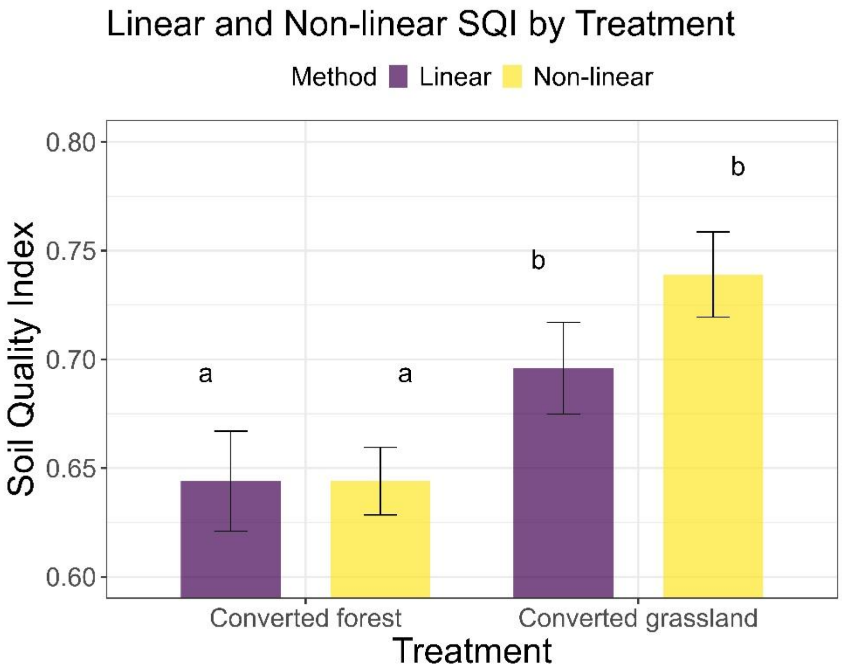

Figure 2.7. Comparison of Soil Quality Index (SQI) between Converted forest and Converted grassland treatments at the 0-15 cm depth based on linear and non-linear scoring methods. Significant differences ($p < 0.05$) are indicated by the different letters according to Tukey's post-hoc test. Error bars represent standard errors of the mean.

Nutrient-Cycling Based Soil Multifunctionality Differences Between Conversions

Nutrient cycling multifunctionality indices, derived from enzyme activity z-scores, were consistently and significantly higher in the converted grassland treatment compared to the converted forest treatment across all three nutrient cycles examined ($p < 0.05$ for all comparisons). For carbon cycling multifunctionality, z-scores were 0.367 and -0.367, nitrogen cycling z-scores were 0.277 versus -0.277, and phosphorus cycling z-scores were 0.412 versus -0.412 for converted grassland and converted forest treatments, respectively (Figure 2.8).

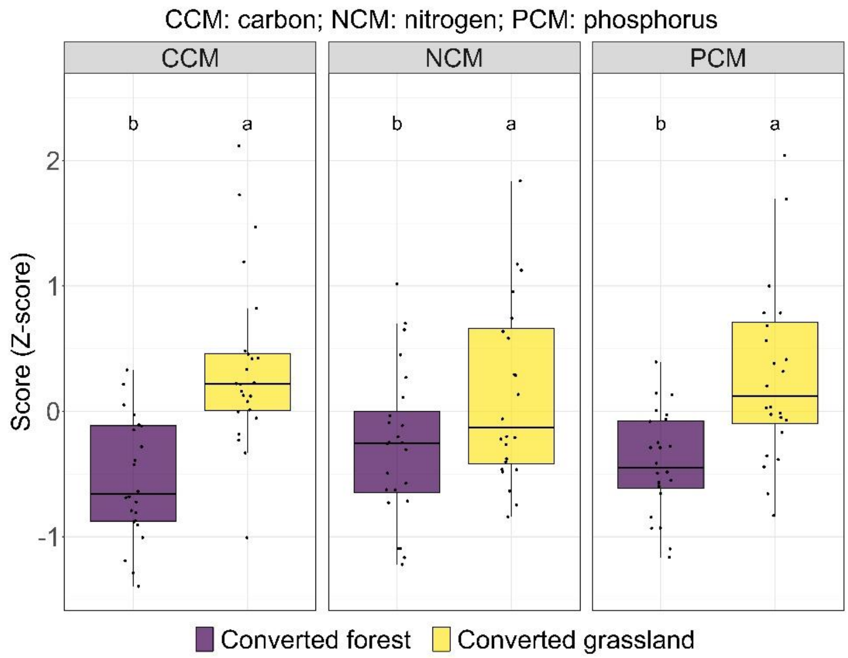

Figure 2.8. Comparison of soil multifunctionality (SMF) between Converted Forest and Converted Grassland treatments at the 0-15 cm depths based on three nutrient cycles: carbon, nitrogen and phosphorus. Significance differences ($p<0.05$) are indicated by different letters according to Tukey's post-hoc test. Box plot whiskers represent data points that lie within 1.5 times the interquartile range (IQR) from the lower or upper quartile.

When enzyme activities were integrated into a single soil multifunctionality index (calculated as the mean of carbon, nitrogen, and phosphorus cycling z-scores), the converted grassland treatment exhibited significantly higher nutrient cycling based multifunctionality (SMF = 0.352) compared to the converted forest treatment (SMF = -0.352; $p < 0.05$) (Figure 2.9).

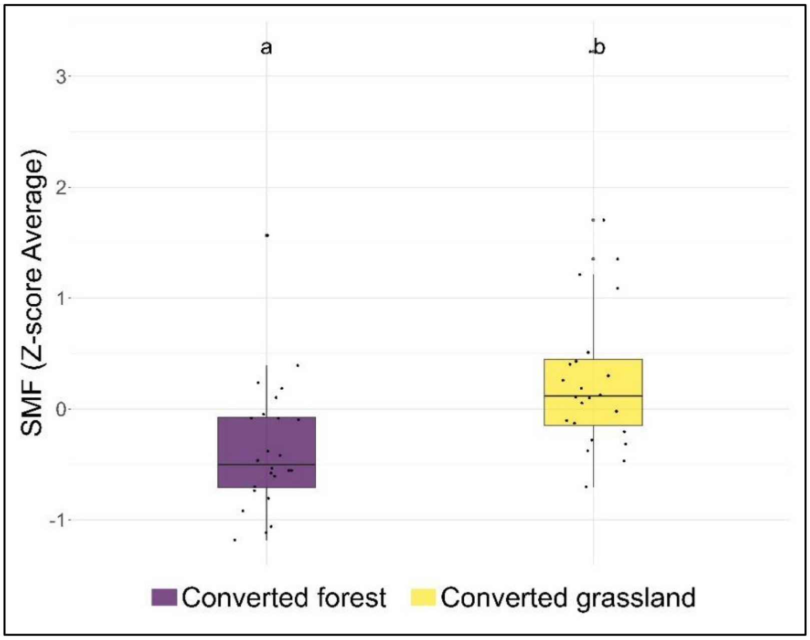

Figure 2.9. Comparison of total soil multifunctionality (SMF) between Converted forest and Converted grassland treatments at the 0-15 cm depths. Significance differences (p<0.05) are indicated by different letters according to Tukey's post-hoc test. Box plot whiskers represent data points that lie within 1.5 times the interquartile range (IQR) from the lower or upper quartile.

Comparing Crop Yield and Quality Between Two Conversion Types

A one-way ANOVA revealed a significant treatment effect on crop yield and corn quality as measured by acid detergent fiber (ADF), neutral detergent fiber (NDF) and crude protein (CP) (Table 10). Corn crop yield in the converted forest treatment was 12,387.97 kg ha^{-1}, significantly lower ($p < 0.05$) than in the converted grassland treatment (14,953.54 kg ha^{-1}) (Figure 2.10). Similarly, corn stalk ADF and NDF measurements differed significantly between treatments ($p < 0.01$), with converted forest crop metric values of 35.6% ADF and 57.5% NDF versus 38.6% ADF and 61.2% NDF in the converted grassland (Table 11). In contrast, ADF and NDF values in corn cobs and kernels did not differ significantly between treatments. Crude protein (CP) content in corn components exhibited similar treatment effects, with significantly higher concentrations in the converted grassland treatment compared to the converted forest treatment for both stalk (11.16% vs. 7.59%, $p < 0.001$) and cob (4.57% vs. 3.74%, $p < 0.05$) (Table 12). Kernel CP percentages did not differ significantly between the two treatments.

Table 10 | Results from 2-way ANOVA test, treatment (Converted forest, Converted grassland) on corn crop yield and corn stalk, cob and kernel ADF, NDF, and crude protein. P values less than 0.05 are in bold.

		Df	Sum squares	Mean squares	F ratio	p value
Yield						
	Treatment	1	37775917	37775917	**4.68**	**0.0422**
	Error	21	169505470	8071689		
ADF stalk						
	Treatment	1	53.99	53.99	**14.6**	**< 0.001**
	Error	22	81.34	3.7		
NDF stalk						
	Treatment	1	79.68	79.68	**13.98**	**< 0.01**
	Error	22	125.38	5.7		
ADF cob						
	Treatment	1	0.01	0.01	0.001	0.972
	Error	22	181.29	8.24		
NDF cob						
	Treatment	1	0	0.001	0	0.996
	Error	22	373.4	16.975		
ADF kernel						
	Treatment	1	2.62	2.617	1.404	0.249
	Error	22	41.02	1.864		
NDF kernel						
	Treatment	1	18.06	18.063	3.584	0.0729
	Error	20	100.79	5.039		

Table 10 continued.

CP stalk						
	Treatment	1	76.44	76.44	**20.98**	**< 0.001**
	Error	22	80.17	3.64		
CP cob						
	Treatment	1	4.172	4.172	**4.723**	**0.0408**
	Error	22	19.431	0.883		
CP kernel						
	Treatment	1	6.85	6.852	4.141	0.0553
	Error	20	33.1	1.655		

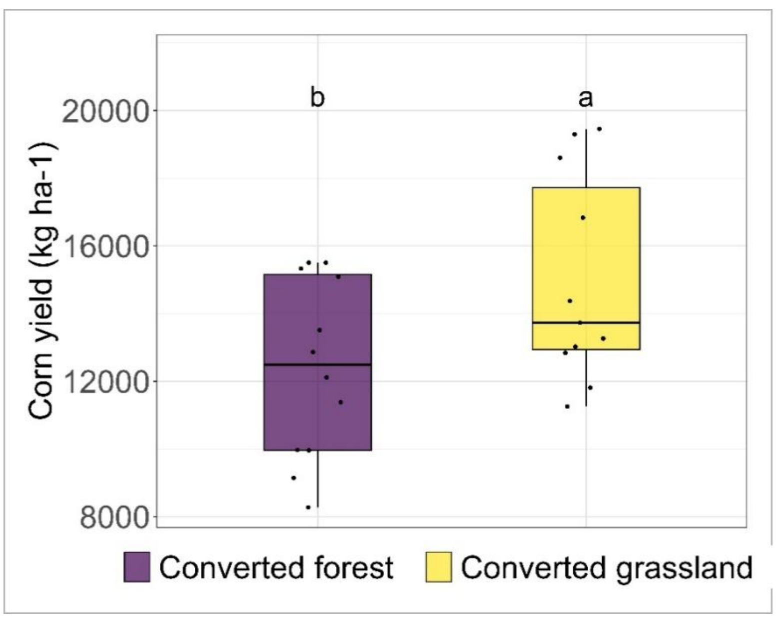

Figure 2.10. Comparison of crop yield between converted forest and converted grassland treatments. Significance differences (p<0.05) are indicated by different letters according to Tukey's post-hoc test. Box plot whiskers represent data points that lie within 1.5 times the interquartile range (IQR) from the lower or upper quartile.

Table 11 | Summary of fiber composition (digestibility) and statistical comparisons in corn crop components by treatment

Parameter (%)	Converted Forest (Mean ± SE)	Converted Grassland (Mean ± SE)	F-value	p-value	Significance
Stalk ADF	35.6 ± 0.56	38.6 ± 0.56	14.6	0.000931	***
Stalk NDF	57.5 ± 0.69	61.2 ± 0.69	13.98	0.00114	**
Cob ADF	42.6 ± 0.83	42.6 ± 0.83	0.001	0.972	ns
Cob NDF	77.6 ± 1.19	77.6 ± 1.19	0	0.996	ns
Kernel ADF	4.65 ± 0.39	5.31 ± 0.39	1.404	0.249	ns
Kernel NDF	16.2 ± 0.68	14.4 ± 0.68	3.584	0.0729	ns

*Note: Statistical analyses were conducted at a 95% confidence level using ANOVA. Significance codes: *** (p<0.001), * (p<0.01), ns (not significant).*

Table 12 | Summary of crude protein concentration and statistical comparisons in corn crop components by treatment

Crude protein (%)	Converted forest (Mean± SE)	Converted grassland (Mean± SE)	F-value	p-value	Significance
Stalk CP	7.59± 0.55	11.16± 0.55	20.98	0.000146	***
Cob CP	3.74± 0.27	4.57± 0.27	4.723	0.0408	*
Kernel CP	9.76± 0.37	10.88± 0.41	4.141	0.0553	ns

*Note: Statistical analyses were conducted at a 95% confidence level using ANOVA. Significance codes: *** (p<0.001), * (p<0.05), ns (not significant).*

Comparing Total Ecosystem Carbon (TEC) Between Intact Ecosystems and Their Conversions

Tree carbon was quantified using a distinct methodological approach from that applied to other carbon pools, therefore, tree carbon results are presented separately. Initial LiDAR point cloud data were used to validate tree top detection methodology (Figure 2.11). Following validation, a linear mixed model was developed to predict DBH from tree height, with results summarized in Table 13. Subsequent LiDAR analysis identified 888 trees within the sample area. The final tree carbon stock in the forested area was estimated at 75.38 Mg C ha^{-1}.

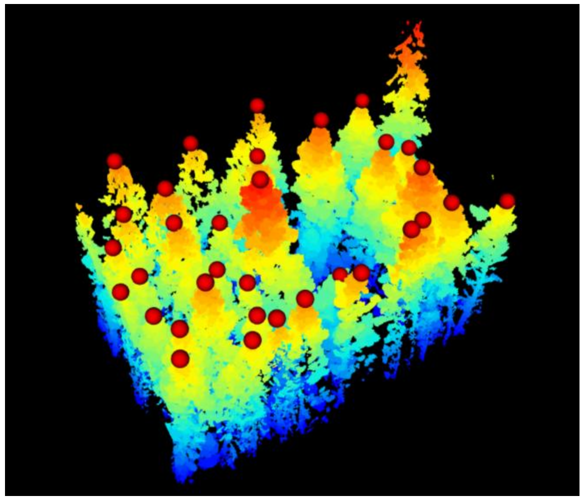

Figure 2.11. Point cloud data used to validate tree top detection methodology. Image created using R software.

Table 13 | Tree diameter at breast height (DBH) model summary

Parameter	Value
Fixed Effects:	
Intercept	-0.6697
Height (ht)	1.4379
Random Effects (Standard Deviations):	
Block (Intercept)	1.63
Residual	8.4
Model Statistics:	
Number of Observations	96
Number of Blocks	4
REML Criterion	682.18

Litter, shrub, grass/forb and soil organic carbon (SOC) stock for each treatment are summarized in Table 14.

Table 14 | Summary of carbon pools by source (Mg C ha^{-1}). SOC stock is measured from fixed mass calculations. Results are displayed as the mean ± SE.

Treatment	SOC stock	Litter	Shrub	Grass/Forb
Converted forest	77.59 ± 7.64	-	-	-
Converted grassland	62.11 ± 3.86	-	-	-
Forest	41.77 ± 4.07	13.33 ± 1.41	0.14 ± 0.04	0.27 ± 0.08
Grassland	65.90 ± 4.11	1.26 ± 0.19	0.54 ± 0.15	1.73 ± 0.16

A paired t-test on SOC stock measurements demonstrated a significant increase in SOC stock following forest conversion, with converted-forest soils showing an 85.8% higher SOC stock than their intact counterparts. In contrast, there was no significant difference in SOC stocks between grassland and converted grassland soils.

Table 15 | Paired t-test results for SOC stock changes by conversion type

Conversion type	n	SE (Mg C ha^{-1})	t-statistic	p-value
Forest	24	35.82	4.2062	**<0.001**
Grassland	23	3.80	-0.5727	0.572

Figure 2.12 presents mean total ecosystem carbon (TEC) stocks for each land-use treatment. Because above- and belowground pools were quantified using different methods, and directly comparable standard errors could not be derived, only mean TEC values are reported. Following conversion, TEC declined by 40.7 % in forest-derived systems, whereas grassland conversions showed no significant change in mean TEC.

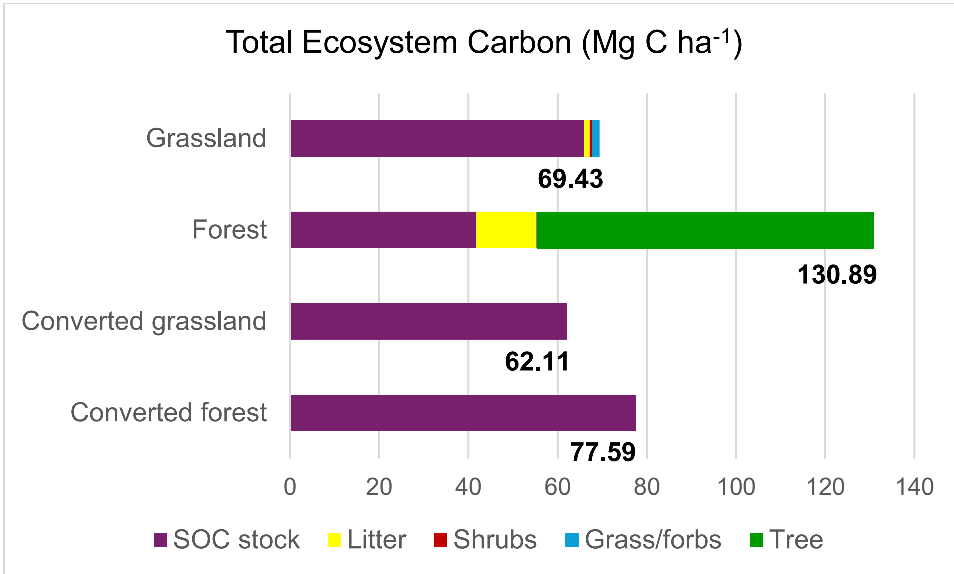

Figure 2.12. Summary of TEC by treatment (Mg C ha^{-1}).

DISCUSSION

Soil Health Indicator Changes During Conversions

Biological Indicator Changes

Biological soil health indicators revealed significant differences between intact and converted systems, as well as between the different conversion types. Except for N-acetylglucosaminidase (NAG) activity in forest conversions, which remained unchanged, the activities of all tested soil enzymes were significantly elevated in both forest- and grassland-converted sites across both soil depths, relative to their intact counterparts. Notably, the converted grassland exhibited the highest enzyme activities, surpassing those observed in the converted forest, a finding variably supported by the literature.

The literature on the effects of land conversion on soil biological activity is complex and often contradictory. Some studies report enhanced enzyme activity following disturbance, attributed to aggregate disruption, increased availability of labile carbon, improved microorganism–substrate contact via soil compaction, and even microbial stress (Khan, 1996, Latif et al., 1992, Resck et al., 2000, Cepeda et al., 2008). Conversely, other studies document decreased activity, citing reductions in soil organic matter from tillage (Barbero et al., 2025). Additionally, agricultural management practices yield variable results, for example, inorganic fertilizers and grazing have been linked to both increases and decreases in enzyme activity (Dick, 1992, Lynch and Panting, 1980), with grazing sometimes enhancing microbial activity via excreta inputs but also potentially suppressing it through soil structural degradation from trampling (Haynes and Williams, 1999, Cao et al., 2004).

In this study area, intact forests and grasslands experienced heavy grazing in both spring and fall, and recent drought and heat-dome events likely compounded ecosystem stress that can accelerate SOC loss (Dlamini et al., 2016). Moreover, Interior Douglas-fir stands are already characterized by relatively low carbon stocks compared to other forest types (Roach et al., 2021). Consequently, a conversion regime involving compost, fertilizers, animal excreta, and intensive irrigation may elevate soil enzyme activities by increasing substrate availability and microbial activity. However, it remains unclear whether these enzymatic surges indicate genuine improvements in soil health or simply reflect accelerated carbon mineralization and enhanced soil carbon losses.

Integrating soil respiration data with enzyme activity measurements provides a more comprehensive picture of soil function, consistent with previous findings (Ouyang et al., 2015; Mukumbuta et al., 2019). Notably, while CO_2-C respiration increased following forest conversion, it decreased from intact grassland to converted grassland. These contrasting patterns, coupled with higher enzyme activities in converted grassland compared to converted forest, suggests that different microbial communities may respond differently to conversion practices, likely influenced by pre-existing ecosystem properties. The results imply that microbial communities in the converted grassland may utilize carbon more efficiently, allocating a greater proportion of organic substrates to microbial biomass synthesis and nutrient transformation rather than immediate respiration which could have positive implications for carbon sequestration.

Further research is needed to elucidate these contrasting enzymatic and soil respiration trends. Detailed investigations into the particulate and mineral associated fractions of soil organic carbon within each conversion type, as well as comprehensive analyses of soil clay content, given preliminary indications of higher clay fractions in converted grassland, are warranted. Additionally, long-term temporal studies are essential to determine the stabilization period of enzymatic activities and soil respiration rates and to identify the transition point at which converted forest and converted grassland systems shift from carbon loss to carbon accumulation.

Chemical Indicator Changes

The distinct chemical responses between converted forest and converted grassland systems also suggests that pre-existing ecosystem properties continue to influence soil processes after conversion. The persistence of lower pH in converted forest soils likely reflects the long-term legacy of Interior Douglas fir (*Pseudotsuga menziesii*) vegetation through continued decomposition of residual organic matter and possibly remaining root systems. Coniferous forest soils typically accumulate organic acids from decomposing needles and maintain active fungal communities that can influence soil acidity, effects that may persist even after tree removal (Kageyama et al., 2008).

Nitrogen dynamics differed markedly between converted systems. The converted grassland treatment soils maintained ammonium concentrations similar to intact grassland and exhibited elevated nitrate levels, suggesting enhanced nitrification relative to the intact system. In contrast, converted forest soils consistently displayed lower concentrations of ammonium and nitrate, along with reduced electrical conductivity. These patterns align with known differences in microbial community composition between grassland and forest soils, where grasslands typically support more abundant bacterial communities, particularly nitrifying bacteria (Crowther et al., 2019). The reduced nitrogen cycling efficiency in converted forest suggests persistence of ecosystem legacy effects, where pre-existing microbial communities may be less adapted to processing agricultural nitrogen inputs.

The maintenance of C:N ratios across conversions, despite changes in absolute nutrient concentrations was unexpected and not generally supported by the literature (Li et al, 2020, Kim et al., 2023). The pattern observed in this study suggests carbon nitrogen balances are regulated during the decomposition processes and further implies that while management practices alter nutrient availability, fundamental patterns of organic matter processing remain tied to the original ecosystem type.

Long-term research is necessary to determine whether the observed soil chemical trends are transient or represent stable, reliable health indicators of a converted system.

Physical Indicator Changes

Land-use conversion substantially altered soil physical properties, with both forest and grassland conversions exhibiting increased bulk density compared to their intact counterparts, though the magnitude of change varied by depth. This outcome was expected (Gol et al., 2008, Tolimer et al., 2020). In shallow soil layers, the converted forest showed a modest increase in bulk density over intact forest, while both conversion types maintained similar compaction levels. However, at deeper depths, the converted grassland demonstrated substantially higher bulk density with potential implications of progressive compaction.

Several factors may have contributed to this pattern. The higher clay content observed in both intact and converted grassland soils may predispose these areas to greater compaction. Moldboard ploughing and other tillage practices have been shown to increase bulk density more substantially in clay soils compared to sandy soils, particularly at depth (Chen et al., 1993, Gameda et al., 1987). Additionally, field observations revealed that the cattle preferentially lingered in the converted grassland areas, likely due to their proximity to water sources, potentially intensifying compaction through concentrated animal traffic.

These findings have important management implications. Conversion, at this site, resulted in increased bulk density. Consequently, long-term monitoring of soil compaction is critical, as continued compaction may adversely affect key soil health indicators by constraining root penetration, water infiltration, and gas exchange, and ultimately impacting overall soil function.

Interestingly, while soil aggregate stability did not significantly change during the grassland conversion, it did increase following forest conversion. This was unexpected (Benalcazar et al., 2022) and may be attributed to the incorporation of organic amendments (compost, animal excreta and crop residues) during the conversion and previous year's cropping process. To determine the efficacy of this variable as a soil health indicator, long term monitoring would be necessary to determine whether this enhanced aggregate stability persists or represents a transitional phase in the conversion process.

Use of Soil Variables as Indicators of Health During Conversion

Although the tested variables have traditionally served as indicators of soil health, their utility in assessing land conversion outcomes may be limited. Conversion invariably alters these metrics, and many improved in this study, yet such changes may reflect transient legacy effects rather than true enhancements of soil function. For example, although converted grassland soils exhibited higher indicator values than converted forest soils, it remains unclear whether this difference signals a sustained functional improvement or simply residual pre-conversion conditions. Likewise, increases in enzyme activities and soil respiration could indicate enhanced microbial activity and soil health in both conversions, but they may also signify accelerated carbon mineralization and SOC loss, outcomes that are respectively beneficial or detrimental to long-term soil health and carbon stores.

This uncertainty underscores the inherent limitations of relying solely on individual soil health indicators, which do not fully capture the complex interactions within soil ecosystems. Consequently, these indicators are best employed as long-term monitoring tools for establishing baseline data and can also inform site-specific agricultural management strategies by accounting for the soil's ecological history when determining crop choice and amendment requirements.

Evaluating Soil Quality in Forest and Grassland Conversions

To overcome the limitations of individual indicators, a comprehensive Soil Quality Index (SQI) was developed using both linear and non-linear approaches. This index was derived through a series of statistical analyses that integrated all measured variables and assessed their correlations, providing a robust assessment of soil health. The analysis focused solely on the two conversion types, based on the assumption that soil quality would be altered during conversion, rendering comparisons with intact conditions less meaningful.

Factor analysis identified a minimum data set of soil health indicators, including whole soil bulk density (WBD), fine soil bulk density (FBD), electrical conductivity (EC), ammonium (NH_4^+), leucine aminopeptidase (LAP) and β-glucosidase (BG) activities, and soil respiration. EC was assigned the highest weighting, followed by LAP and BG activities, which had similar contributions. In contrast, NH_4^+, soil respiration, and soil organic carbon (SOC) received moderate weightings, while both bulk density measurements were given the lowest weightings. These results suggest that chemical and biological properties were the most influential factors in distinguishing soil quality differences in the converted systems.

Correlation analysis revealed a strong positive association between NH_4^+ and SOC, aligning with previous research demonstrating that higher organic matter content enhances mineralization and ammonium release through microbial decomposition (Six et al., 2006). While total SOC did not differ significantly between conversion types, the study did not differentiate between labile and stable carbon fractions. The converted grassland may have contained a greater proportion of labile carbon, as suggested by the moderate positive correlations between NH_4^+ and enzyme activities (LAP and BG). These enzymatic responses typically indicate greater substrate availability from SOC (Allison et al., 2010) and were notably elevated in the converted grassland system.

Conversely, moderate negative correlations between soil respiration and both ammonium (NH_4^+) and soil organic carbon (SOC) suggest that higher nutrient levels may be associated with reduced microbial metabolic rates. One possible explanation is that, in nutrient-rich systems with enhanced substrate availability or a higher proportion of labile carbon, microbial communities may prioritize biomass synthesis and transformation processes over immediate respiration, thereby adopting a more efficient carbon use strategy (Kruse et al., 2013; Daunoras et al., 2024). This interpretation is reinforced by the observation that converted grassland soils exhibited higher enzyme activities yet lower respiration rates compared to converted forest soils. Although carbon fractionation was not assessed, both the correlation analysis and the higher NH_4^+ levels in converted grassland support these interpretations. Moreover, intact grassland contained significantly higher levels of NH_4^+ and total nitrogen than intact forest, suggesting that a legacy effect contributed to the nutrient profile observed in the converted grassland.

The strong negative correlation between electrical conductivity (EC) and ammonium (NH_4^+) suggests that soils with higher EC, potentially due to increased dissolved ion concentrations, might inhibit ammonium accumulation through mineralization processes (Rengasamy, 2010). However, my study does not support this relationship. At shallow depths, EC levels were similar between conversion types, whereas at deeper depths, EC was higher in converted grassland than in converted forest. In contrast to literature findings, NH_4^+ concentrations remained consistently higher in converted grassland across both depths. This apparent contradiction suggests that unmeasured ions—such as calcium (Ca^{2+}), magnesium (Mg^{2+}), sodium (Na^+), and potassium (K^+)—may be influencing ammonium retention through specific ion exchange processes. Future analyses that include these ions would help elucidate the mechanisms driving nitrogen dynamics between the two conversion types.

The moderate correlation between fine and whole bulk density indicates that while both indicators capture aspects of soil compaction, they are not entirely redundant and may reflect different elements of soil structural integrity. Additionally, the weak correlations between these bulk density measurements and other MDS soil variables suggest that soil compaction is likely more influenced by soil texture and management practices than by the biochemical processes driving nutrient cycling and microbial activity. Notably, whole soil bulk density showed only moderate differences between conversion types across both depths, whereas fine soil bulk density did not differ significantly between treatments at either depth. This is further supported by the lower weighting assigned to fine soil bulk density in the Soil Quality Index (SQI).

Collectively, the interactions among chemical, biological, and physical soil properties underscore the inherent complexity of soil systems and reinforce the value of a Soil Quality Index (SQI) that integrates multiple variables to provide a more comprehensive assessment of soil health than individual indicators can offer. My results demonstrate that, under both linear and non-linear scoring methods, converted grassland soils exhibit significantly higher SQI scores than converted forest soils, with the more pronounced differences observed using non-linear scoring suggesting that this approach is particularly sensitive in detecting treatment differences. These findings not only enhance our understanding of the legacy effects on converted soil quality but also highlight the benefits of employing a composite SQI as a valuable tool for long-term monitoring.

Evaluating the Nutrient Cycling Capacity of Forest and Grassland Conversions

The nutrient cycling multifunctionality index (SMF) provided a broader assessment of enzyme activity and nutrient processing capacity, complementing the Soil Quality Index (SQI), which factor analysis limited to two enzyme activities. SMF indices were consistently higher in converted grassland compared to converted forest across carbon, nitrogen, and phosphorus cycles, corroborating findings from both the SQI and individual indicator analyses. This pattern may be attributed to legacy effects resulting from differences in microbial community structure, diversity, and function between intact forest and grassland ecosystems (Kaiser et al., 2016). Coniferous forest soils, typically dominated by fungal communities specialized in decomposing acidic needle litter, may be less responsive to readily available agricultural nutrient inputs than grasslands (Trofymow et al., 2023).

As a response, the converted grassland exhibited enhanced nitrogen cycling capacity, suggesting a microbial community better adapted to nitrifying the higher levels of ammonium (Clark et al., 2020). Additionally, carbon cycling functionality was significantly higher in converted grassland, even though soil organic carbon (SOC) did not differ significantly between conversion types, potentially due to a greater proportion of labile carbon forms facilitating enzymatic activity.

The elevated phosphorus cycling multifunctionality score in converted grassland was variably supported by the literature. Forest soils generally have lower available phosphorus due to the predominance of organic phosphorus forms and aluminum binding; therefore, if legacy effects were the sole influence and phosphorus were the limiting nutrient, higher phosphatase activity would be expected in converted forests (Kunito et al., 2012; Margalef et al., 2017). Conversely, Margalef et al. (2017) did report that higher total nitrogen correlates with increased phosphatase activity, and this is consistent with my finding of significantly higher total nitrogen in the converted grassland than in converted forest. These results highlight the need for further investigation of phosphorus dynamics at this site. This study did not include direct phosphorus measurements, and the SMF did not account for potential interactions between phosphatase (PHO) and leucine aminopeptidase (LAP), both of which were considered in the phosphorus cycling assessment.

To establish the SMF as a standalone indicator, rather than merely a complement to the SQI, future work should integrate soil microbial community profiling with correlation analyses of soil texture, SOC fractionation, and phosphorus fractionation to clarify their complex dynamics. Moreover, future studies must determine whether the elevated nutrient-cycling signals in converted grasslands reflect genuine ecosystem function or simply accelerated mineralization and SOC loss, particularly as captured by carbon-cycling multifunctionality scores.

Comparing Soil Quality, Crop Yield and Crop Quality in Two Conversion Types

The significantly greater corn crop yield and quality observed in the converted grassland compared to the converted forest, as evidenced by differences in acid detergent fiber (ADF), neutral detergent fiber (NDF), and crude protein (CP) measurements, aligns with existing literature on soil quality-crop yield and quality relationships.

The positive correlation between the Soil Quality Index (SQI) and crop productivity is well-documented (Lenka et al., 2022; Mukherjee et al., 2014; Vasu et al., 2024; Huang et al., 2021), and the findings of this study support this relationship. Converted grassland exhibited both higher SQI values and superior yield outcomes. Moreover, soil enzyme activities, particularly β-glucosidase (BG) and N-acetyl-glucosaminidase (NAG) and phosphatase (PHO), have been positively associated with crop yield (Sainju et al., 2022; Yang et al., 2023, Quiao et al., 2022). These results align with study observations of elevated enzyme-driven carbon, nitrogen, and phosphorus cycling capacities in the higher-yielding converted grassland plots.

Regarding crop quality parameters, while Adhikari et al. (2022) found only weak correlations between soil health and corn grain quality, other studies have established links between soil nitrogen availability and corn protein content (Alijani et al., 2021; Oktem et al., 2010). In this study, most soil health indicators were higher in converted grassland than in converted forest, and the total soil nitrogen and the nitrogen cycling capacity of converted grassland was also superior, further supporting previous literature findings.

It is important to note that correlations between the SQI, SMF, or individual soil indicators and measured corn yield were not calculated due to a mismatch between the number of crop and soil sample points. Nevertheless, the results align with existing literature. Future research should increase the crop sample size to enable a robust correlation analysis and further substantiate these relationships. Additionally, long-term studies of the site are needed to determine the stabilization period for crop yields between the two conversion types.

Changes in TEC Between Intact Ecosystems and Converted Counterparts

The 40.1 % decline in total ecosystem carbon following forest conversion was expected, given the large biomass removal associated with tree harvesting (Pan et al., 2011). Likewise, the 85.8 % increase in SOC stocks under forest conversion aligns with the substantial inputs of compost, manure, and corn residues. In contrast, the grassland conversion, despite receiving identical input applications, exhibited no significant change in either TEC or SOC stocks.

Intact grasslands likely lost substantial carbon during conversion, mirroring the rapid SOC declines documented after grassland-to-cropland transitions (Liang et al., 2023; Tang et al., 2019; Wei et al., 2014; Qiu et al., 2012). In this study, the addition of exogenous carbon inputs (compost, manure, and corn residues) appears to have triggered a priming effect by accelerating microbial mineralization (Sun et al., 2019; Mazzilli et al., 2014) and thus destabilizing soil carbon pools. Mazzilli et al. (2014) specifically showed that corn residue inputs enhance mineral-associated carbon decomposition rates, a process likely exacerbated here by the observed clay content of converted grassland soils. This interpretation is supported by the markedly elevated extracellular enzyme activities in converted grassland plots. To confirm the instability of newly added carbon, future research should characterize SOC fractions (e.g., particulate vs. mineral-associated pools) and directly track carbon turnover rates.

Although forest-converted soils showed substantial SOC increases, these pools are likely labile and prone to loss, as evidenced by elevated extracellular enzyme activities and higher soil respiration rates following conversion (Nyberg et al., 2020). In the inherently soil carbon-poor Interior Douglas-fir ecosystem (Roach et al., 2021), large inputs of compost, manure, and crop residues, coupled with warm conditions and irrigation, likely induced a priming effect (Mgelwa et al., 2025). Furthermore, reduced crop yields in forest-converted plots may trigger a scenario whereby lower residue returns fail to compensate for ongoing SOC mineralization, ultimately leading to net soil carbon losses.

In this study, corn cultivation immediately following conversion generated labile carbon pools prone to rapid turnover. Therefore, long-term, high-resolution monitoring of carbon dynamics is essential to quantify turnover rates, determine when soils transition from net loss to net gain, and assess stabilization trajectories. This temporal data is critical for evaluating the viability of continuous corn in post-conversion systems and to guide the selection of alternative cropping strategies for enduring soil carbon sequestration.

LIMITATIONS

The primary limitations of this study include the absence of baseline testing, the extrapolation of small, single-time-point samples to a larger area, the lack of analysis of particulate and mineral associated carbon fractions, the omission of soil texture analysis, and the absence of true replicate sites.

The study site was identified and donated for research only after the land had been cleared for agriculture, making it impossible to conduct baseline tests before conversion. Testing the converted areas prior to clearing would have provided a more accurate measure of change than relying on adjacent intact ecosystems as proxies for the pre-conversion state. Additionally, because the land had been cleared five years before field testing, the period spent in an uncultivated state may have altered soil properties in ways not captured by study results.

The study's reliance on point-sampling at a single time interval represents an additional methodological limitation, especially given the well-documented spatial and temporal variation in soil biochemical properties (Chelabi et al., 2021; Gil-Sotres et al., 2005). Consequently, only an estimate of the site's overall characteristics can be derived using this approach.

An analysis of labile and stable carbon fractions would have enhanced the interpretation of biological indicators and may have clarified the apparent contradiction between enhanced enzyme activity but reduced soil respiration in the converted grassland versus the reverse pattern in the converted forest. Similarly, a more comprehensive soil texture analysis might have better explained the observed trends in soil biological indicators and nutrient cycling capacity, as preliminary observations revealed distinct soil texture patterns between conversion types.

Another key limitation was the absence of site replication, which confined this work to an observational study rather than a true experiment. The inclusion of additional converted sites under similar agricultural management practices, either within the same biogeoclimatic zone or at comparable elevations, would have strengthened the findings. Furthermore, testing converted sites across different geographical locations would have provided valuable comparative data. Multiple site replication would have validated or challenged the study's findings, thereby offering a more comprehensive contribution to the existing literature.

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CHAPTER 3: Research conclusions, implications and future directions

This study found that converting heavily grazed forests and grasslands to corn production with managed cattle grazing led to short-term improvements in multiple soil health indicators, including biological, chemical, and physical (aggregate stability) metrics. However, these gains may represent transient conversion effects rather than lasting enhancements of soil function. Elevated enzyme activities and soil respiration, commonly interpreted as signs of healthy microbial function, could instead indicate accelerated decomposition of labile carbon, pointing to increased mineralization and potential destabilization of soil carbon pools.

When comparing the performance of the two conversions, the converted grassland exhibited significantly higher scores than the converted forest on both the Soil Quality Index (SQI) and the nutrient cycling-based Soil Functionality Index (SMF). This was reflected in superior corn yield and forage quality metrics in the converted grassland. These findings suggest that legacy effects from the intact ecosystem may influence soil nutrient cycling and modulate the complex biochemical interactions within the soil environment following conversion. They also suggest that the conversion origin should be considered when developing an agricultural management strategy, particularly regarding input applications and crop selection.

Forest conversion caused a 41% decline in total ecosystem carbon (TEC), driven primarily by tree biomass removal, yet SOC stocks increased by 86% following compost and crop residue amendments. In contrast, the grassland converted soils showed no significant change in TEC or SOC stocks despite identical inputs, suggesting substantial initial soil carbon losses post conversion followed by rapid microbial turnover of added organic matter, as evidenced by elevated enzyme activities. In forest-converted soils, similarly high enzyme activities and increased soil respiration further indicate that most carbon inputs remain labile and are quickly mineralized rather than stabilized.

These findings suggest that integrating cattle grazing with corn on converted lands can yield short-term gains in soil health indicators, but long-term studies are essential to determine whether carbon inputs of corn residues and compost ultimately stabilize or continue to mineralize. Additionally, future research should evaluate whether annual corn represents the most effective post-conversion crop. Such insights will be critical for developing agricultural management strategies that sustain productivity, improve soil health and maximize long term carbon storage.

IMPLICATIONS

"We know more about the movement of celestial bodies than about the soil underfoot."

— Leonardo Da Vinci, circa 1500s

Research Implications

This study challenges prevailing paradigms by demonstrating that certain soil biochemical parameters can improve following conversion of an overgrazed rangeland to annual cropping systems, contrary to widespread reports of post-conversion soil degradation (Liang et al., 2023; Wei et al., 2014). However, these apparent gains warrant cautious interpretation. For instance, the elevated enzyme activities and soil respiration results in this study may reflect either enhanced microbial function or accelerated decomposition of added carbon inputs, leading to net destabilization of soil organic carbon. Thus, although managed grazing and organic amendments delivered short-term improvements in some soil health metrics, their longer-term effects on carbon stability and agroecosystem resilience remain uncertain.

A similar caveat applies to SOC stock changes. Although SOC stocks rose by 86% in forest-converted soils and remained unchanged in grassland-converted soils, these figures could obscure carbon losses. Large inputs of compost and crop residues, coupled with elevated enzyme activities in both conversions and heightened soil respiration in forest-converted plots, suggest that added carbon is largely labile, masking ongoing SOC mineralization and destabilization. Consequently, short term SOC stock gains may not equate to long term carbon sequestration, underscoring the need for metrics that distinguish transient inputs from stabilized soil carbon.

This study also demonstrates that different land conversion types yield distinct biochemical responses to post conversion agricultural practices. The Soil Quality Index and Soil Multifunctionality Index (SMF) showed that the converted grassland responded more favorably to conversion than the converted forest as evidenced by greater nutrient cycling function and enhanced corn yields and quality metrics. These findings highlight the strong legacy effects of the original ecosystems and suggest that corn may be a suboptimal crop choice, especially on forest-converted soils. Alternative cropping systems, particularly those incorporating perennials or annual cover crops, may enhance soil-building properties and promote carbon sequestration, making them more suitable for forest conversion scenarios.

This study additionally underscores the necessity of effective monitoring tools capable of discerning, in the long term, the complex chemical, biological, and physical interactions within soil systems. While the assessment of individual soil health indicators yielded interesting observations, the singular result often generated more questions than answers. In contrast, the use of composite indices, specifically the Soil Quality Index (SQI) and the Soil Multifunctionality Index (SMF), provided a more comprehensive evaluation. By integrating multiple variables, eliminating redundancies, and incorporating weighted scores supported by existing literature, these indices are well suited for monitoring the long-term evolution of altered soil systems. This integrated approach is markedly more effective in evaluating the entire soil system than the reliance on one or a few isolated indicators.

Building on these findings, future research should examine land conversion under different management systems including conventional, organic, and regenerative, to yield valuable insights into optimizing nutrient cycling and maximizing carbon sequestration. Such studies should investigate how these management practices affect soil biochemical processes and overall system resilience in converted lands. In addition, exploring the response of diverse crops and rotational systems to land conversion could further refine sustainable conversion strategies. For example, evaluating rotations that integrate grazed corn with perennial forage crops, or studying transitions from converted cropland back to managed rangeland systems, may illuminate practices that simultaneously optimize carbon storage and preserve ecosystem health. Collectively, these research directions are essential for identifying land conversion practices that enhance ecosystem functioning while maximizing carbon sequestration potential.

Policy Implications

In an era of accelerating climate change marked by more frequent wildfires, pest outbreaks, and emerging diseases, and with warming temperatures opening new regions to cultivation, intact ecosystems face mounting conversion pressures. It is therefore essential that research claiming gains in soil health or carbon sequestration rigorously distinguish between labile and stabilized carbon pools and demonstrate long-term soil function. Such evidence will enable policymakers and land managers to pursue agricultural conversions that genuinely mitigate climate change through long term carbon storage, rather than inadvertently degrading ecosystem services.

Embedding proven agroecological and agroforestry practices into conversion strategies can safeguard core ecosystem functions while enhancing productivity and strengthening soil-based climate mitigation. To drive widespread adoption, legislation must pair regulatory guidelines with financial incentives for private landowners who implement these practices. This integrated policy framework can foster future agricultural landscapes that ensure food security while sustaining ecosystem resilience in a changing climate.

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