

IMPRACTICABLE SAMPLING REQUIREMENTS TO MONITOR
BLACKFEET NATION CARBON RELATED SOIL HEALTH INDICATORS

by
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DEDICATION

I dedicate this master's thesis to my extraordinary family, Susan, Brad, and Ella Boyd. You have always been there for me and have had an indescribable impact on my life. Your dedication to me has been apparent, and I want to dedicate this thesis to you because with you because, I would not be where I am today without you three.

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ABSTRACT

Before the effects of grazing management on soil health can be assessed, a baseline analysis of soil conditions needs to be completed. This study, conducted across the Blackfeet Nation in Northwest Montana, explores patterns and variability of volumetric soil organic carbon (SOC_v), using catena-based sampling and co-located holes to account for local soil variability. The research also conducts power analyses, a statistical test to determine the needed sample size to have a high likelihood of detecting a real effect, by examining key soil health indicators variability to assess the feasibility of detecting changes in SOC_v. My findings indicate that detecting small shifts in SOC_v requires potentially hundreds of thousands of samples per site due to the high local soil heterogeneity of these semiarid rangeland soils. These findings are supported by others in the literature. SOC_v stocks were partly influenced by soil-forming factors such as vegetation and climate. However, other factors, such as rangeland management and hyperlocal conditions, may have a greater effect on explaining SOC_v stocks and hillslope variability than soil-forming factors, although a copious number of samples will be needed to confirm this. These findings highlight the importance of integrating current and historical land use into SOC_v models. Future research should explore SOC_v changes after rangeland management changes while factoring in sample size and developing methods to extrapolate better SOC_v and other soil health data across large rangeland landscapes.

INTRODUCTION

Bison (*Bison bison*) have been grazing on the Northern Great Plains for tens of thousands of years, including the land of the Blackfeet Nation. Bison began to disappear after Europeans moved rapidly into the region and extensively hunted bison, which increased forage for non-native cattle introduced by those same Europeans (Freese et al. 2007). Bison were quickly replaced with cattle (*Bos taurus*) that ranchers raised on landscapes shaped by Bison and Indigenous People of North America. Because of their selective grazing method, reluctance to naturally rotate, and increased maintenance requirements, these cattle began deteriorating the once native, highly productive prairies of the Great Plains. In partnership with humans, cattle overgrazed these lands and removed wildlife habitat, created monocultures, brought in invasive, non-native species, and decreased the overall ecosystem health of the Northern Great Plains (Kohl et al. 2013).

Ranchers have realized rangelands are in trouble and want to rejuvenate the prairie, including its vegetation, biodiversity, and soil health, by having their cattle imitate the native, once-abundant bison grazing patterns (Teague et al. 2013). To imitate bison grazing, ranchers use higher stocking rates for shorter periods to get cattle to graze pastures more evenly (Beukes and Cowling 2003). This can be referred to as high-intensity rotation grazing. The more general term for attempting to use cattle to restore degraded rangelands through altering grazing patterns is referred to as regenerative grazing (RG). RG is an adaptive land management approach that mimics natural grazing patterns, promoting ecosystem health by integrating practices such as rotational grazing, diverse plant species, and reduced chemical inputs (Newton et al. 2020).

Unlike conventional grazing systems, which can lead to soil degradation and biodiversity loss, regenerative grazing encourages conditions that enhance soil structure, water infiltration, and nutrient cycling (Teague et al., 2016). These practices can improve pasture productivity and contribute to broader environmental goals, including climate change mitigation.

A highly concerning result of climate change for ranchers is the possibility of decreased precipitation, higher temperatures, higher evapotranspiration, and more prolonged and intense droughts (Moore and McEvoy 2022). With rising temperatures, water retention could be the salvation of rangelands everywhere. One of the best ways to increase soil water holding capacity is to have more soil organic matter (SOM) and soil organic carbon (SOC) in particular (Bhadha et al. 2017). More SOC leads to a higher quantity and quality of forage, even into the late summer (Reddy 2016). Better and more forage means more grass for cattle to eat, increasing their body weight gain or even providing the potential for ranchers to increase their herd size, increasing profits. The problem with this idea is twofold: 1) There is mixed evidence that regenerative grazing significantly affects soil health and SOC, and 2) Soils are highly variable, so detecting changes in soil properties requires an incredibly high sampling density, complicating experimental design and increasing costs.

Regenerative grazing has shown potential in regions with greater moisture and higher plant productivity; for example, a study in Georgia reported a five-fold increase in soil organic carbon stocks after 20 years of regenerative grazing (Rowntree et al. 2020). However, many studies have found no SOC change due to high-intensity grazing events (Sanjari et al. 2008, Chan et al. 2010, Schon et al. 2011). And some studies have even found detrimental effects, such

as a decrease in daily weight gain for cattle in RG treatment groups (Barnes and Denny 1991, Augustine et al. 2023) and a reduction in SOC (Beukes and Cowling 2003, Sarula et al. 2014).

Soil spatial variability occurs on every scale, from across an entire landscape to a spatial scale of a few centimeters (Wachinger et al. 2000, Stanley et al. 2023). This poses a major problem when conducting research on large landscapes without an unlimited budget. Therefore, our project aims to assess baseline soil health across the Blackfeet Nation. A “power analysis” statistical test can calculate the number of samples needed to detect a specific change between two groups, helping researchers shape their experimental design. This study investigates how many samples are needed by calculating the local variability of ~100 hillslope positions across 28 ranches. And quantifying how hard it may be to find improvements in soil health due to the introduction of regenerative grazing, SOC in particular, based on baseline soil variability.

My findings indicate that although some locations need hundreds of thousands of samples to find even small changes in SOC, as the literature supports, others need only dozens. These low variability locations provide solid areas to implement RG for the best possibility of finding conclusive results with minimal samples taken and thus evaluating the efficacy of RG practices.

CHAPTER 2: LITERATURE REVIEW

Introduction

Rangelands, or open country for grazing or hunting animals, are vast ecosystems that cover approximately 659 million acres (267 million ha), or 28% of the United States (Winters-Michaud et al. 2024). These lands provide essential services, such as forage for livestock, wildlife habitat, and carbon storage. Yet these vital landscapes face growing threats from climate change, including prolonged droughts, erratic rainfall, and rising temperatures (Svejcar et al. 2014). Healthy rangeland soils offer a natural defense against disturbances, including drought, by enhancing water storage, reducing erosion, and supporting robust forage production (Blanchy et al. 2023).

Despite their extensive coverage and ecological importance, the historical bias in agriculture research has been to focus research on croplands. (Sayre 2023). This is concerning given that rangeland soils are often far more variable than cropland soils (Herrick and Whitford 1995, Stanley et al. 2023), making their management more complex.

Improved soil management is increasingly recognized as a strategy to mitigate greenhouse gas emissions (Bangroo et al. 2013). Soils store three times more carbon than the atmosphere and four times more than above-ground vegetation, making even small changes in soil carbon highly impactful (Minasny et al. 2017). Given the extensive area rangelands cover, their potential contribution to climate mitigation could be immense. However, this idea may be too optimistic because carbon sequestration outcomes depend on complex conditions, and

management alone may not always yield the desired results (Abule et al. 2005, Minasny et al. 2017, Lal et al. 2019).

Additionally, there is currently no widely accepted set of practices for improving rangeland soil health, nor a set of inexpensive, low-variability, and relevant indicators to assess the impacts of grazing practices on soil health (Spangler et al. 2012, Liptzin et al. 2022). This study seeks to address how to measure soil health by evaluating soil health indicators based on baseline data collected across the Blackfeet Nation in northwest Montana, USA.

Rangeland Management

Historical Management

For tens of thousands of years, indigenous people and bison co-evolved on the Northern Great Plains, compared to the short period (~250 years) that Europeans and cows have presided over these lands. Today's western U.S. rangelands are largely divided by barbed wire into designated plots of land that may be owned or leased. However, prior to that, livestock once roamed freely without man-made physical boundaries (DelCurto et al. 2023). Early ranchers managed these lands by relying on informal agreements, often leading to overgrazing and land conflicts (Sayre 2023). When overgrazing reached a breaking point, grazing leases were established to distribute the grazing pressure and ease land use conflicts (Sayre 2023). The removal of traditional rangeland management techniques, such as controlled burns, exacerbated mismanagement issues, leading to an increased dependence on grazing leases to distribute pressures. Once fences were established, ranchers began to manage their leases to increase forage. One way of doing this was to use a tractor or truck to pull out sagebrush (*Artemisia*

tridentata). This decreased habitat for wildlife, namely sage grouse (*Centrocercus urophasianus*), increased bare ground and invasive species abundance (Knick et al. 2011). Another historical rangeland management technique used to grow forage during drought was planting crested wheatgrass (*Agropyron cristatum*). Once recommended for drought resilience, the introduction of crested wheatgrass, while offering short-term forage solutions, has contributed to ecological degradation by promoting monocultures that suppress soil health and biodiversity (Rogler and Lorenz 1983). Crested wheatgrass is a highly aggressive species, quickly creating monocultures and decreasing the biodiversity of the plant and animal species in the pastures (Vaness and Wilson 2007). Additionally, crested wheatgrass has been shown to decrease soil stability, leading to problems with erosion and compaction (Broersma et al. 2000). Examples of poor range management like this allow ranchers to adapt their strategies by learning from their mistakes, and over time, can lead to improved stewardship practices such as rotational grazing, restoration of native plants, and integrated land management techniques.

Modern Management

There are still practices being used today that cause degradation of rangelands, such as sod-busting or the practice of plowing rangelands to convert them to croplands (Tucker 2008). Sod-busting removes native vegetation, impacting wildlife, soil health, and water quality due to nutrient runoff and increased sedimentation near water bodies (Silcock and Fensham 2019, Angerer et al. 2023). Science-based awareness of harmful rangeland management practices, shared through university extensions and agencies like the NRCS and USDA, helps ranchers move away from environmentally damaging methods such as sod-busting (Briske et al. 2017). So, even though rangeland management still has major problems, land managers have learned

that sustainable management strategies can prevent overgrazing and control invasive species (DiTomaso 2000, Abdelsalam 2021).

While detrimental practices like sod-busting persist in modern rangeland management, sustainable strategies have also emerged to mitigate these issues and promote healthier ecosystems. Most of these strategies revolve around grazing. One key aspect of sustainable management involves understanding how livestock grazing impacts plant growth dynamics. Livestock removes above-ground biomass from grasses and forbs, which leads to plants investing energy in growing different parts of the plant (Sollenberger et al. 2019). For example, after being heavily grazed, plants may invest more energy into growing leaves and less into growing and developing their root system (Tesk et al. 2018). This makes the plant more vulnerable to additional grazing pressure if it is not given enough time to recover; eventually, if a plant is grazed too often, it will be unable to recover, causing it to die. This is problematic, but cattle grazing can be managed to allow plants to adequately recover before experiencing an additional grazing event. When leaf biomass is low, the plant is in a growing stage to increase photosynthesis capacity. When cattle take leaf biomass they can trigger these growing points and allow the plant to grow and reproduce if grazing is done at the correct time and intensity (Wentao et al. 2023). Animals can even improve spring green-up by grazing at a high intensity by stimulating plant growth points (Geremia et al. 2019). Grazing intensity is often quantified in the units of Animal Unit Months (AUM). An AUM represents the dietary requirements for a 1,000 lb cow-calf pair with 1 AUM hectare⁻¹ being low and 3 AUM hectare⁻¹ being high (Reinhart et al. 2022). These AUM can be used to maximize forage abundance by allowing increased rest (Bartley et al. 2023). For example, a study examining grazing pressure and timing

effects in Montana found that grazing cattle at a light, 1 AUM hectare⁻¹ rate between late October and early November, increased vegetation aboveground biomass (Reinhart et al. 2024).

Because not all cows weigh 1,000 lbs, ranchers should adjust their AUMs based on the size of their cattle. If a cow-calf pair weighs 1,400 lbs, then the rancher should recognize that each cow-calf pair is 1.4 AUMs (Stam et al. 2018). Cows nutritional demands also change throughout the year. For example, if a cow is lactating, then she will consume 10-20% more forage than when dry (Sprinkle et al. 2021). Additionally, AUMs are difficult to work with because forage availability is often inconsistent on a single ranch, making it difficult to use AUM ha⁻¹ across different regions.

Grazing Effects on Soil

Just as grazing affects plants, grazing can affect soil (Teague et al. 2011). Cattle can physically impact soil properties through hoof action, which can cause compaction when soils are wet or when grazing pressure is too high (Drewry 2022). Cattle can also alter the nutrient cycle in soil by providing extra nutrients from dung or urine (Haynes and Williams 1993). Ideally, these changes can be observed by measuring soil properties called “soil health indicators” before and after implementing a management change. These indicators change in value as soil health changes by responding to management and can distinguish signal from noise, allowing one to analyze what effect management can have on soil health. Soil health is the optimum status of the soil's biological, physical and chemical functions (Al-Kaisi 2014).

Before examining specific soil health indicators, it is essential to understand different grazing practices and the reasons ranchers implement them. Traditional grazing systems are managed through continuous grazing, where livestock roam freely within fenced pastures,

resulting in prolonged but typically low grazing pressure on vegetation. When stocking rates are too high (e.g., 5 AUM ha⁻¹), continuous grazing can degrade soil by compacting it, breaking down soil structure, decreasing soil organic carbon (SOC, Tables 1.1 & 1.2), and reducing water infiltration, which limits plant-available water (Thurow 1991, Webber et al. 2010). Such degradation processes increase surface runoff and erosion, depositing nutrients in downhill water bodies and impacting freshwater ecosystems (Burkart and Stoner 2002, Babiker et al. 2004). Continuous grazing with a forage utilization rate of 50% can reduce soil organic carbon stocks by 30% compared to continuous grazing at a 10% utilization rate (Ingram et al. 2008).

Table 1.1. Advantages and disadvantages of continuous and rotational grazing.

	Continuous Grazing	Rotational Grazing
Advantages	Low complexity of management, lower production costs	Reduces erosion, increases plant biomass with more rest, maintains or improves SOC _m , decreases tick loads
Disadvantages	Reduced plant diversity, increases soil compaction, increase pest infestation	Higher production costs, more complex management

Rangeland Management Strategies

Regenerative grazing is a promising approach to managing these lands, with claims to increase soil organic matter (SOM; approximately double SOC), improve ecosystem resilience, and observe beneficial changes in vegetation (Sanderman et al. 2015, Rhodes 2017, Ahlering et al. 2021). Any increases in SOC following the implementation of regenerative grazing are most likely driven by shifts in the plant community resulting from grazing practices (Derner and Schuman 2025). Regenerative grazing's claimed benefits are the opposite of continuous grazing's claimed detriments (see Table 1.1). The harmful effects of continuous grazing

highlight the necessity for improved management practices; thus, regenerative grazing presents vital solutions for enhancing soil health (Ahlering et al. 2021). While total SOM content is a useful metric, focusing on specific SOM fractions — like analyzing the specific human blood components of glucose or oxygen — may provide deeper insights into soil health and function. This study focused on soil organic carbon (SOC) rather than total SOM because SOC is crucial in nutrient cycling, soil structure formation, and soil water availability, making it a key soil health metric, especially for resilient rangeland ecosystems (Lal et al. 2019, Rumpel et al. 2020). Although many so-called regenerative practices might boost SOC levels (Figure 1.1), soil aggregation, and soil stability, thus improving overall ecosystem function, there are just as many studies showing only modest or no measurable differences in these types of soil properties (Teague et al. 2011, Aguilera et al. 2013, Chaplot et al. 2016). Due to the lack of consensus on the definition of "regenerative grazing," this paper will refer to the grazing systems discussed as “range management strategies” unless they specifically address regenerative agriculture. Reported changes in SOC attributable to rotational grazing ranged between -2.6 and $+18\%$ per year, with a median shift of $+2.3\% \text{ y}^{-1}$. Our review of conventional grazing effects, for comparison, ranged between -32 and 7.4% , with a median shift of $0.0\% \text{ y}^{-1}$.

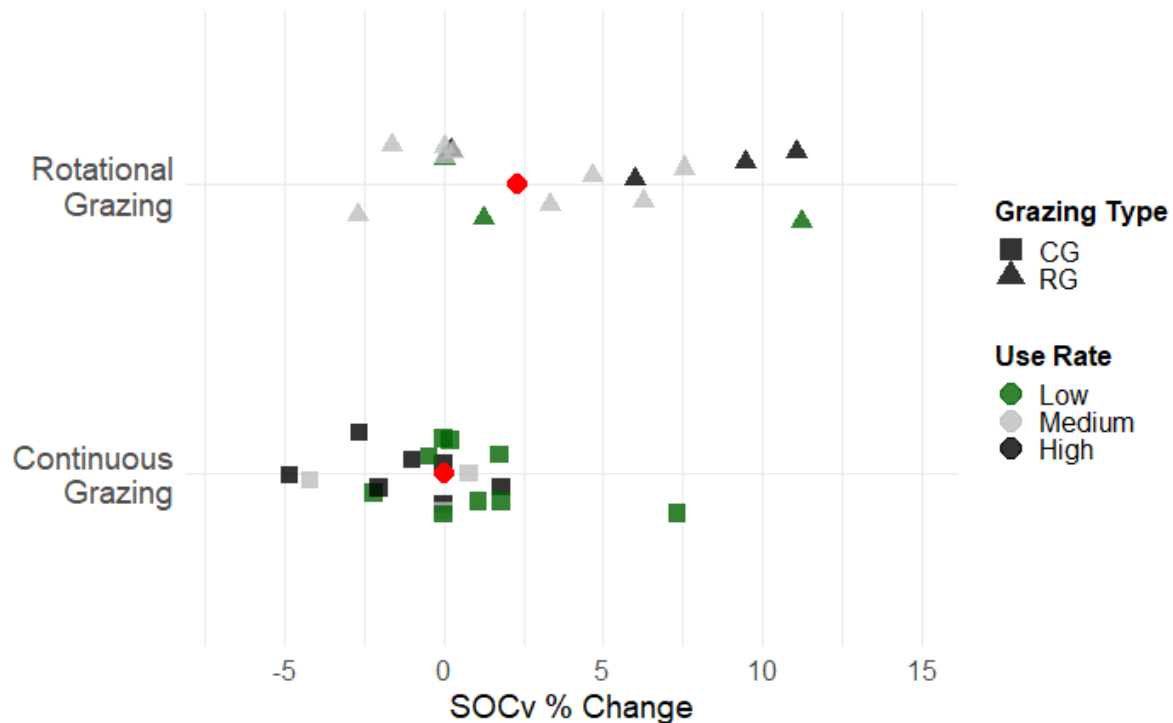


Figure 1.1. Average percentage volumetric soil organic carbon (SOCv) shifts year⁻¹. Data is from a collection of 45 studies (see references in Appendix B); 2 outliers (Rotational Grazing 18%, Continuous Grazing -32%) are excluded for clarity. Though different use categories are highlighted with different color symbols, these assignments often required considerable interpretation of reported methods. “Light use” rates are 0.5-2 animal unit months (AUM) hectare-year⁻¹, “medium use” rates are usually 2-3 (with most being 2.7) AUM hectare-year⁻¹, and “heavy use” rates are usually 3+ AUM hectare-year⁻¹. Red diamonds depict the grazing type medians: 0.0% for continuous grazing and 2.3% for rotational grazing.

Dozens of different rangeland management strategies exist. From the literature six of the most popular strategies include mob-grazing, adaptive multi-paddock, silvopasture, winter grazing, strip grazing, and targeted grazing (Hoekstra et al. 2019, Bachler 2019, Bailey et al. 2019, Karki et al. 2021, Augustine et al. 2023, Wagner et al. 2023). These strategies have no set intensity, and differ not in AUM’s but in the pattern and actual movement of the livestock.

Mob-grazing is a strategy that attempts to replicate natural ruminant grazing, such as bison, using high-density, short-duration grazing events with AUMs ha⁻¹ year⁻¹ of 3-10. Mob-

grazing has been suggested to increase desirable forage and SOM (e.g., SOC_m increased by 4.6% relative to conventionally grazed controls according to Segoli et al. (2015), but it can lead to soil compaction and decreased soil aggregation (Roberts and Johnson 2021). Soil aggregates are groups of soil particles (sands, silts, or clays) that group together through cohesion, creating clusters of soil that make up the soil structure in the soil profile (Nimmo 2013). Aggregates affect soil structure by influencing water retention, nutrient cycling, and soil stability (Yudina and Kuzyakov 2023). Specifically, Roberts and Johnson (2021) reported a 10% increase in soil compaction in mob-grazed paddocks relative to conventionally grazed paddocks, and a 40% decrease in aggregate stability in mob-grazed paddocks relative to ungrazed paddocks. As with all grazing strategies, and mob grazing in particular, the precise effect on SOC can further depend and interact with precipitation, soil texture, and existing forage composition (Piñeiro et al. 2010).

Another popular regenerative grazing technique is adaptive multi-paddock (AMP) management, where short-duration grazing periods (between a few hours and a few days) are followed by long pasture rest periods (>8 months) to allow plants to recover between grazing events (Augustine et al. 2023). AMP is like mob grazing but is more adaptive and can have varying grazing intensities (typically between 2-5 AUMs ha⁻¹ year⁻¹). Mob grazing is distinctive from AMP because it usually involves higher intensities and shorter grazing rotations to mimic native grazing herd movements (Wagner et al. 2023). Although mob grazing and AMP require multiple paddocks per herd to ensure the proper grazing pressure, AMP is more adaptive to weather patterns and pasture conditions (Apfelbaum et al. 2022). This tool does not have rigid movement dates, and each ranch grazing plan adapts to pasture conditions based on herd nutrient

requirements, weather, and other factors that can affect pasture conditions (Teague and Kreuter 2020). This tool allows ranchers to change herd sizes or split herds depending on behavior and economic needs, while continuous grazing is less flexible with all animals having unrestricted access to a large pasture (Johnson et al. 2022). For example, ranchers can sell cattle when prices are high, and weather conditions are poor, or buy heifers (female cows) when prices are low and weather conditions are good. Although many field trials do not reveal measurable pasture improvements using AMP grazing compared to continuous grazing, Teague et al. (2013) claims, “When applied rigidly at these larger scales, [AMP trials] have often led to different and unsatisfactory outcomes.” The same study states that by using AMP, “Many ranchers have achieved excellent animal production and soil and vegetation improvements using AMP and find that the flexibility and timeliness of feedback inherent in (AMP) facilitate improved management compared to continuous grazing.” However, the compounded variability of soil properties and grazing implementation introduces significant noise into monitoring efforts. While ranchers may observe improvements anecdotally, detecting a consistent management signal in the data remains a challenge.

Re-integrating tree crops, whether orchards or timber stands, and livestock could be a way to maximize land use more sustainably. An example is silvopasture, a system where cattle are grazed in forested or orchard settings. Thinning of trees allows cattle to quickly move between plants, grazing grasses while not causing any harm to trees (Karki et al. 2021). In a five-year silvopasture grazing system in Alabama, soil biological activity increased by 91% compared to ungrazed woodland plots; in addition, silvopasture grazing lead to a 73% percentage increase in mass-based carbon (SOC_m) compared to ungrazed woodland plots, leading to improved soil

health, which can benefit pasture and timber profits (Table 1.2; Karki et al. 2021). SOC_m is the percentage of the soil that is made from organic carbon. This measurement allows for estimation of how much carbon is in the soil profile, but neglects the portion of the soil profile that is taken up by rocks, and how dense the soil is. Although an increase in microbial activity can decrease SOC_m because the microbes are feeding off the SOC_m, this study had an increase in the SOC_m of the silvopasture plot due to an increase in plant biodiversity and increased grasses and forbs (Karki et al. 2021). In a nine-year study in a semi-arid region of India, a silvopasture system increased soil organic volumetric carbon (SOC_v) by 50% or 1.8 tons per acre-foot (t ac-ft^{-1} , $4.1 \text{ Mg ha-0.3m}^{-1}$; Ramakrishnan et al. 2021). Thinning trees and introducing livestock allow for more pasture utilization while simultaneously increasing ecosystem biodiversity and soil function. Although silvopasture is unlikely to be a popular strategy across the Northern Great Plains as the region generally lacks timber, it can be a potential regenerative grazing option for ranchers who live near forests, such as on the western, forested edge of the Blackfeet Nation.

Table 1.2. Comparison between silvopasture studies (Karki et al. 2021 and Ramakrishnan et al. 2021). Numbers are averages $\pm$ 1 standard deviation (SD)

Property	Karki et al. 2021	Ramakrishnan et al. 2021
Location	Alabama	India
Study Length	5 years	9 years
Practice	Pine forest vs pine silvopasture	Grass pasture vs guava silvopasture
Biological Silvopasture	β -glucosidase 61.7 $\pm$ 3.34 (nmol g ⁻¹ hr ⁻¹)	Microbial Biomass 56.7 $\pm$ 0.6 (mg kg ⁻¹)
Biological Control	β -glucosidase 32.3 $\pm$ 3.34 (nmol g ⁻¹ hr ⁻¹)	Microbial Biomass 47.2 $\pm$ 0.5 (mg kg ⁻¹)
Biological Change	91% increase	20% Increase
Soil Carbon Silvopasture	2.6 $\pm$ 0.12% SOCm	12.2 $\pm$ 1.5 Mg ha ⁻¹ 0.15m ⁻¹
Soil Carbon Control	1.5 $\pm$ 0.12% SOCm	8.1 $\pm$ 1.1 Mg ha ⁻¹ 0.15m ⁻¹
Soil Carbon Change	73% increase	50% increase
Precipitation ¹	52 in (132 cm)	33 in (85 cm)

¹Browning, Montana averages 15 in yr⁻¹ (38cm)

Another form of silvopasture that is common on the Blackfeet Nation is shelterbelts.

Shelterbelts are rows of shrubs and trees that are planted to protect the land from wind erosion (Stoeckeler 1970). These shelterbelts are often referred to as living snow-fences because of their ability to trap snow, increasing localized soil moisture, and growing more forage around and within the shelter belt (Bartling 2020). This form of silvopasture provides protection for livestock, access to forage, and increased soil health (Guernsey 1996, Smith et al. 2022).

Winter grazing is a general term, like regenerative grazing, meaning several strategies exist. Winters, especially in the Northern Great Plains, are long and cold, requiring feeding cattle for up to five months of the year. This can account for 41-62% of total costs per head, depending on market prices, management techniques, and location, but no matter the situation, winter feeding cuts into profits (DiLorenzo 2012). Finding a way to reduce production costs is helpful for ranchers, and several of these strategies can potentially improve pasture conditions, providing a benefit to the ecological and economic state of the operation. Standard cattle systems in the

winter put out bales of hay each day, which is not too different from a traditional feedlot. This can cause manure build-up, soil compaction, and considerable labor and feed costs (Flores and Tracy 2012, Bachler 2019).

Innovative ranchers developed several different winter grazing strategies to avoid some, or all, of these problems. The first is bale grazing, which involves strategically laying out bales in pastures where cattle gain access, feed on the hay, and then move on to the next bale once they are given access (Donohoe et al. 2021). Bale grazing can be done with large round or small square bales, but because round bales are typically larger (~1100 lbs vs 75 lbs [499 kg vs 34 kg]), they are more time-efficient and easier to roll out (Donohoe et al. 2021, Jones et al. 2024). Bale grazing eliminates labor costs in traditional feedlot systems, such as manure removal and daily feeding (Kelln et al. 2012).

Potential ecological benefits include increased pasture nutrients from manure and organic matter input. Over time, this can increase pasture forage quality and quantity by increasing organic inputs, removing bare-ground patches, and increasing microbial diversity (Kelln et al. 2012, USDA 2023). Bale grazing can decrease biomass for the first few years after grazing takes place because of the large amount of hay and manure residue at bale centers (Donohoe et al. 2021). However, bale grazing can create nutrient-loaded zones on a patch of bare ground, encouraging undesirable species, such as noxious weeds, to be established (Bachler 2019). Another project in the Midwest focused on bale grazing during drought, even if pastures are left to graze. This method gives cows bales as supplemental feed before moving to a new pasture to extend the time they can spend in one pasture before overgrazing the future pasture (Halich

2025). This maximizes the time a pasture rests before being grazed and allows for ranchers to not overstock pastures in times of low forage

Another cost-reducing, winter grazing method is winter swath grazing. In this management system, forage, either perennial grasses or crops, is cut before winter and consolidated in strips across the field (Baron et al. 2012). Snow falls, keeping the forage from being dispersed by wind. Cattle are then introduced into the pasture and can dig through up to 2 feet (0.6 m) of snow to gain access to this forage (Tanaka et al. 2005). This method can reduce invasive species' chances to establish themselves while cutting costs by 30% for winter feeding per head (Tanaka et al. 2005). This method is helpful because it reduces the need to bale hay, purchase other animal feed, and eliminate labor-feeding costs in the winter. However, its application is limited to wind-free zones, or swaths of forage will blow away. Referencing a late 1800s observation about the Blackfeet Nation, Foley (1974) stated "...wind sweeps with great power to and from the mountains about 9 days in the week"; thus swath grazing may not be an appropriate recommendation for Blackfeet producers.

Stockpiling forage for winter grazing is a popular strategy in the Northern Great Plains. One study found that 29% of ranchers stockpile forage to extend the grazing season and cut costs during the winter (Sheppard et al. 2015). This grazing strategy takes even less work than other winter grazing strategies because all the ranchers need to do is not graze a pasture and let cattle access the standing forage in the winter. This can even be done in snowy areas because of cattle's ability to deal with a couple of feet of snow to access food (Tanaka et al. 2005). This management technique requires excess rangeland that can go unutilized until the winter, when forage nutrients are lower than in summer (Sheppard et al. 2015). Another consideration is that

cattle need access to water, so stock tanks must be present in the grazed pasture and maintained during winter grazing of stockpiled forage. Additionally, stock tanks must be regularly checked for ice during winter to ensure a reliable water source for the cattle. Tanks should also be strategically placed close enough to cattle herds to minimize the energy cattle expend reaching water, which is especially critical in colder months when energy demands are higher (Wang et al. 2023).

Rotational grazing comes in multiple varieties, one of which is strip grazing. This management system relies on temporary electrical fences or, in some instances, virtual fencing to control which part of the pasture cattle can graze (Staahltoft et al. 2023). A temporary electrical fence can be set up across the whole pasture in a strip-grazing system. The fence is moved gradually back, giving cattle access to new forage (Figure 1.2). The rate of movement depends on the desired intensity, but it is common to give each cow an additional 500-3,000 ft² (46-279 m²) per day. This form of rotational grazing has been found in Europe to result in 13-36% greater forage biomass production than ungrazed control plots (Starz et al. 2013, Hoekstra et al. 2019). This is a promising form of regenerative grazing but requires extensive grazing planning as well as infrastructure investments (Hoekstra et al. 2019).

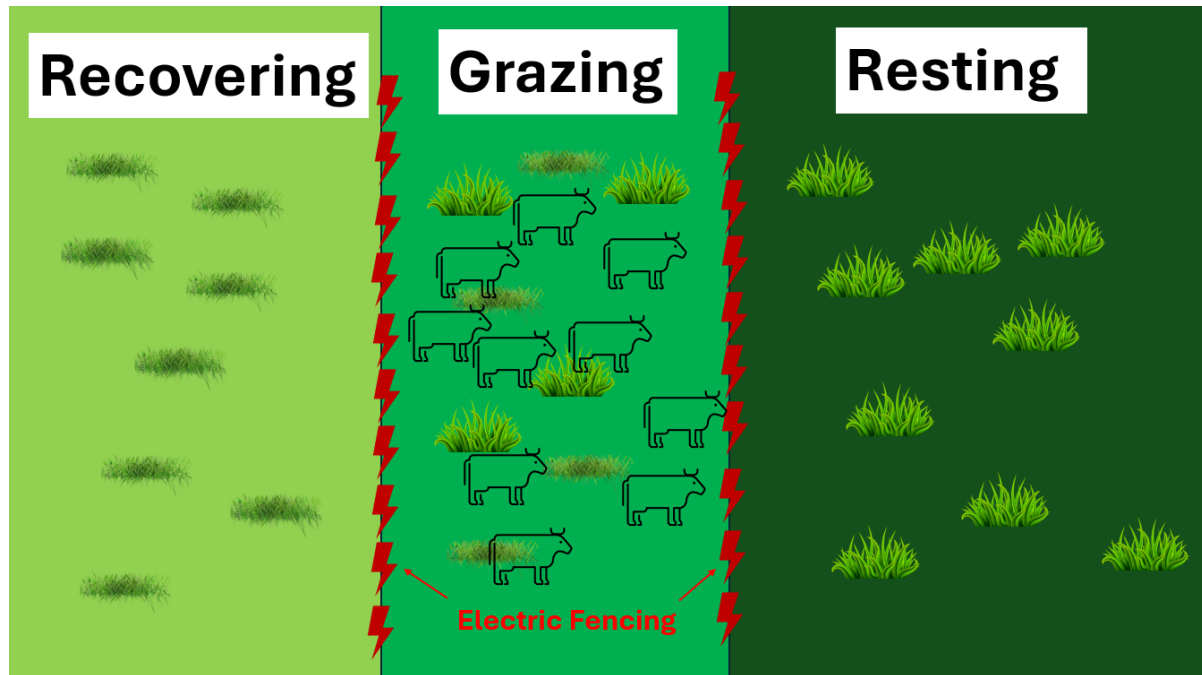


Figure 1.2. Diagram illustrating strip grazing. The dark green area on the right represents recently ungrazed forage, while the red lightning bolts represent electric fences. The middle pasture ($\sim 26,000 \text{ ft}^2$ [$2,415 \text{ m}^2$]) represents the section of the pasture currently accessed by cattle following their most recent movement. This diagram represents a grazing pressure of $1.5 \text{ animal unit months hectare}^{-1}$ per year and assumes one move per day.

Grazing is a tool, and although most rangeland grazing management strives to increase desirable species and forage biomass, one grazing tool, targeted grazing, is primarily utilized in specific scenarios to remove or reduce non-desirable above-ground biomass from rangelands. Targeted grazing removes fuels for fire prevention and targets undesirable species (Bailey et al. 2019). From a land management aspect, using targeted grazing is one of the few large-scale fuel reduction options considered financially viable (Svejcar et al. 2014); for example, $\$110 \text{ acre}^{-1}$ for brush cutting fuel reduction vs. $\$28 \text{ acre}^{-1}$ for grazing fuel reduction (values adjusted for inflation; Magadlela et al. 1995). Although cattle can be used in this method, removing woody fuels and woody invasive species is better done using goats (Bailey et al. 2019).

Targeted grazing differs from other forms of rangeland management because its goal is to remove as much above-ground biomass as possible to reduce species' abundance instead of utilizing the entire pasture sustainably. Focusing on specific parts of a pasture that are overwhelmed by weeds or that can cause potential fire hazards allows ranchers to have high-intensity grazes that can last for as long as the rancher deems necessary while not sacrificing high-quality forage (Marchetto et al. 2021). Beyond costs, the benefits of this tool include the absence of herbicides, its ability to be used in remote settings, and increased plant biodiversity (Marchetto et al. 2021). Targeted grazing on invasive species patches is often a multiyear process; however, weed patches need several intensive grazing events to remove or reduce the species of interest from a pasture (DiTomaso 2000). This strategy often uses goats instead of larger livestock to help eliminate these weeds. Goats are browsers, herbivores adapted to feed on woody vegetation such as leaves, twigs, and bark, so they are better equipped to eliminate large weeds, including knapweed and thistle (Banks 2007), while cattle are larger and better at reducing cheatgrass (*Bromus tectorum*) and other non-toxic grasses (Codron et al. 2006, Bailey et al. 2019).

Grazing Intensity and Introduction to Soil Health

For all of these rangeland management strategies, appropriate grazing intensity is essential for maintaining soil structure and preventing overgrazing, which can lead to soil degradation (Frank and Groffman 1998, Volesky et al. 2011). With increased grazing intensity, bulk density (BD) can increase, associated with potential soil compaction (Byrnes et al. 2018).

However, increased BD usually only occurs in the top horizon, commonly the A horizon, with deeper horizons' BD not correlating with grazing pressure (Tate et al. 2004).

There is no perfect grazing intensity for all pastures because the optimal intensity depends on what livestock is being reared, climate, topographical position, existing forage, and what rangeland management strategies are used. A wetter, warmer climate tends to yield more forage, meaning a higher grazing intensity can be implemented provided sufficient rest is part of that management (Sala et al. 1988). Different species of plants need different amounts of rest, so the ratio between grazing intensity and rest is location- and situation-dependent. Well-managed systems balance grazing pressure with rest periods, benefiting soil health and livestock productivity by preventing soil compaction and excessive defoliation (Teague and Barnes 2017).

Soil texture, a relatively stable property, influences how grazing impacts soil infiltration. Clay and silt soils have illustrated more soil health improvement under managed grazing than sandy soils, which can be less prone to compaction (Xu et al. 2018). So, although some evidence suggests soil properties such as aggregate stability can be changed through grazing management, any effects are sensitive to factors such as local climate, soil texture, and relief (Piñeiro et al. 2010, Bartley et al. 2023).

Variability

It is easier to assess carbon stocks at shallower depths (2-4 in [5-10 cm]) because the A horizon is where most carbon is stored (Kim et al. 2023). Since soil organic carbon (SOC) is concentrated near the surface of rangeland soils, sampling deeper than the topsoil horizon can dilute the SOC "signal" due to the presence of low-SOC mineral soils. While sampling to a depth

of one foot (30 cm) provides a more accurate estimate of total soil carbon stocks, it is more challenging to detect changes in SOCv compared to sampling at a shallower depth of 4 inches (10 cm). Just like sampling deeper, taking more samples gives a more accurate representation of true carbon stocks.

Understanding changes in soil organic carbon volume (SOCv) is crucial for ranchers and researchers alike, but drawing accurate conclusions can be challenging. Imagine trying to determine whether a new grazing practice is truly improving soil health—too few samples— independent of depth characterized—might fail to detect a genuine shift in SOCv. If an experiment finds a measurable increase in SOCv, for example, ranchers would like this increase to be real. Sometimes, however, simply due to probabilities and various sources of error, we might measure an increase in SOCv that is not real—we can think of this as a false positive (Type I error). A false negative (Type II) error can occur when a genuine increase in SOCv is missed, leading to a mistaken conclusion that no change occurred; this can be minimized by conducting a power analysis to determine the appropriate sample size (Figure 1.3). Insufficient samples may lead to Type II errors, while underpowered studies can result in false positives (Type I errors), such as mistakenly concluding that rotational grazing increases SOCv when, in reality, it does not (Figure 1.3). Both errors are costly and agents of accidental miscommunication. These errors cannot be eliminated, but they can be minimized by conducting a power analysis (Stanley et al. 2023).

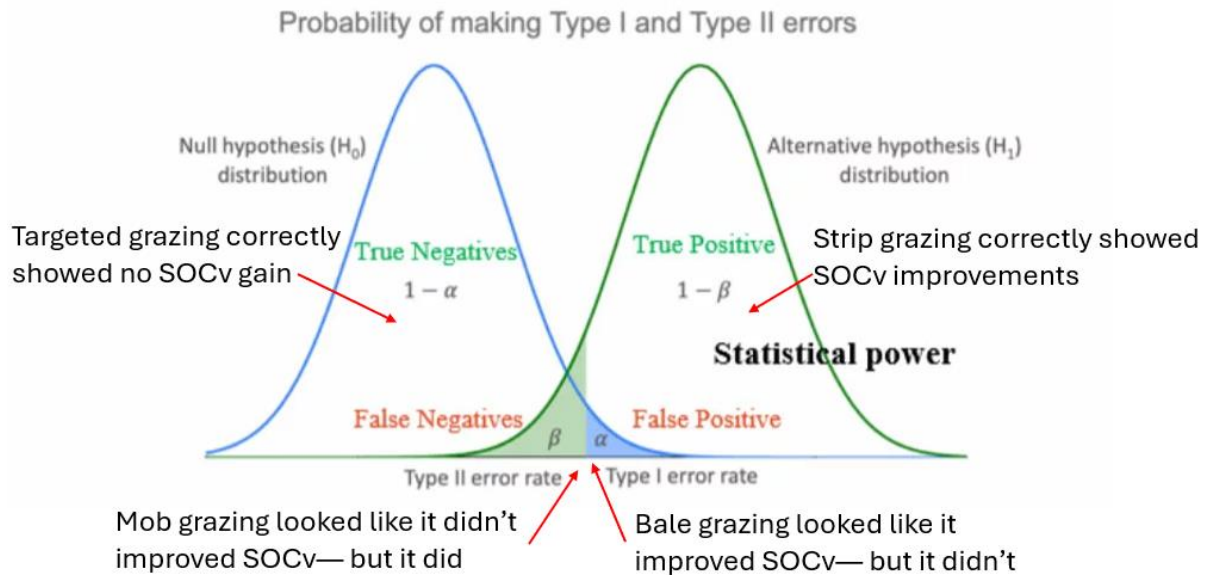


Figure 1.3. Diagram of Type I and Type II errors with rangeland management technique examples; adapted from The Center of Excellence at Orbeli Institute of Physiology (<https://era-chair.am/resource-information/>).

Instead of asking how much an “improved” versus “business-as-usual” grazing practice might affect SOCv, scientists need to shift mindsets to ask questions such as: given the variability of the baseline soil data, how many samples do we need to collect to detect a meaningful change between practices. Under-sampling diminishes the study's power, increasing the likelihood of correctly detecting a +10% increase in a soil health indicator, for example, from an ideal 80% to merely 30%. Therefore, carefully considering the appropriate number of samples during experimental design helps achieve reliable and statistically valid soil health results. If we were using a greater change in the soil property, say 50%, we would need fewer samples than if we were hoping to see a 10% increase, but increasing any soil properties by 50% over a short time is extremely unlikely, especially in rangeland soils (Figure 1.1; Henry et al. 2024).

Understanding Variability in Soil Properties

The importance of variability in soil properties can be difficult for researchers to convey to land managers, even though they are familiar with seasonal or annual fluctuations in other aspects of ranching such as rainfall and beef prices that affect their profits. Cattlemen, experienced in managing variability, adapt by shifting grazing rotations or adjusting selling times for steers based on drought and market fluctuations. However, western scientific practices often adopt a rigid framework that overlooks the unique adaptive strategies employed by ranchers in managing variability. Thus, using relatable agricultural examples can illustrate to ranchers the need for sufficient sampling to accurately assess soil properties instead of relying on small, potentially skewed subsets. Discussing the implications of variability can pave the way for enhanced collaboration between researchers and ranchers, leading to more cost-effective sampling and monitoring approaches for rangeland soil health assessments.

Types of Soil Variability

Vertical variability in physical and chemical properties is an intrinsic characteristic of most stable and relatively undisturbed soils; we designate soil horizons based on these vertical and visible morphologic differences. Horizontal variability, by contrast, is typically much less pronounced in most soils (Figure 1.4; Franz et al. 2013).

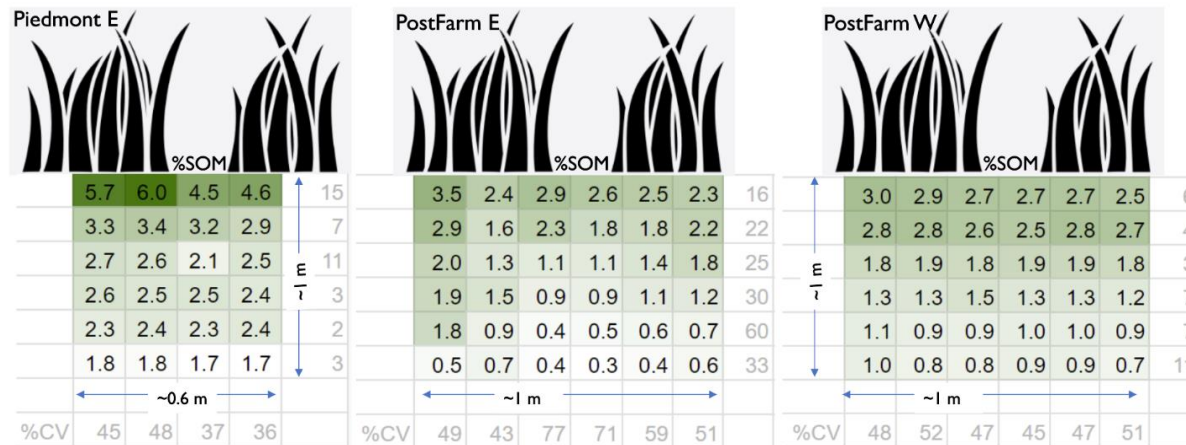


Figure 1.4. SOM variability for three soil profiles near Bozeman, MT. Vertical SOCm CVs ranged 36-77% and were typically 10 times larger than horizontal SOM CVs ranging from 2-60%.

For example, soil horizon depths often remain consistent, and biological characteristics such as microbial activity can fluctuate across space and time (Ishaq et al. 2020, Carey et al. 2020, Fierer et al. 2021). Buyer & Sasser (2012) found that different soil textures can have phospholipid fatty acid analysis (PLFA) concentrations that vary by ~50%. Ishaq et al. (2020) saw on average a 47% increase in bacterial richness over the course of a month. Soil aggregates a few centimeters apart, can exhibit varying biological (e.g., microbial activity), chemical (e.g., nutrient levels), and physical (e.g., water retention) properties. For example, with an uneven distribution of organic matter at the microscopic level, soil methane emissions measurements can differ by 180% at the centimeter scale (Wachinger et al. 2000). Although fine-scale variability often challenges researchers, acknowledging it during experimental design and sample collection enables more reliable findings at larger landscape scales (Welsch et al. 2019).

Addressing Variability in Experimental Design

To examine the effect of different rangeland management strategies on soil health, a statistically robust, adequately powered experimental design is needed (Stanley et al. 2023). Detecting changes in any one soil health indicator could require thousands of samples per site, emphasizing the challenge of high variability. Different soil health indicators such as aggregate stability, however, are less variable and may only require a few dozen samples per site to detect similar changes. Financially, it can cost hundreds of thousands of dollars less to get similar information about a ranch's soil health after a change in grazing management. To understand why, we need to incorporate a measure of variability into our experimental design.

Drawbacks of Composite Sampling

Most soil studies use a composite sampling method because laboratory analytic costs can be prohibitive. Compositing samples – combining samples from multiple locations into a single sample– is a common approach to examining soil properties at a coarse scale (Kang and Mills 2006). Compositing may reduce numerical variability in a spreadsheet because no explicit estimate of local spatial or intrinsic variability is possible, but this approach cannot account for true landscape or spatial variability because large differences can occur over small vertical and/or horizontal distances (Welsch et al. 2019). Stanley et al. (2023) stated “Spatial heterogeneity is likely to dominate measurement uncertainty in many scenarios. We recommend three ways ... to reduce uncertainty in SOC estimates ...: provide stratified sampling guidance, minimize compositing, and, most importantly, require larger sample sizes.” Therefore, we chose a different approach. We sampled and analyzed four, co-located holes on a catena, a transect consisting of a hillslope (Milne 1947); in other words, instead of four sets of composite samples, we tested each sample individually. This allowed us to quantify local variability by calculating the coefficient of

variation of each portion of a catena (e.g., crest vs toeslope). We also sampled by morphological soil horizon (e.g., A, Bw, Bt), as each has unique characteristics and variable thicknesses, instead of taking samples at predefined and fixed-depth intervals (e.g., 0-5 cm, 5-10 cm, 10-30 cm).

Inherent Uncertainty in Soil Testing

Even the same exact soil tests using the same exact procedures applied to the same exact soils can give slightly different results; this is termed analytical uncertainty. We can evaluate this by comparing values between tests run on the same sample from the same hole. For example, our soil respiration tests can vary around 33% on the same soil (Herbst et al. 2009). Therefore, we need more replicates to determine if a management practice increases the carbon mineralization rates. Of course, our study design is not perfect, nor are our laboratory methods; however, there is no way of analyzing soil carbon that addresses all the inherent variability seen in soil carbon (Wielopolski et al. 2010).

Two key challenges arose when addressing variability in the sampling process. First, the research was baseline, meaning a ranch soil's variability was unknown to our team. As a result, conducting a power analysis was not possible until after sampling had been completed. Second, as previously discussed, sampling is costly. Collecting four samples from each hillslope position proved expensive, and sampling at a much higher density may be unrealistic.

Overgrazing

While many range management practices display promise, incorrect management can still lead to overgrazing, causing soil structure deterioration (Byrnes et al. 2018), increased surface runoff, and erosion (Fierer and Gabet 2002). Severe defoliation before adequate regrowth

weakens the plant stand, which can lead to bare ground exposure and soil degradation (Briske and Richards 1995, Yisehak et al. 2013, Gastal and Lemaire 2015). Such issues emphasize the need for balanced grazing pressure and recovery times, which are fundamental to maintaining healthy, resilient pastures. Since improper grazing management can lead to soil degradation and bare ground exposure, evaluating soil health through indicators like SOC is essential to understanding and mitigating these impacts on rangeland ecosystems. As mentioned before, grazing management tries to distribute grazing pressures over time in space using different methods. If a rancher's calculations are wrong or there is miscommunication leading to a pasture with a high stocking rate, this can lead to overgrazing in certain circumstances. So, rotational grazing has some risks that come with it because of its complexity compared to continuous grazing (See Table 1.1).

Indigenous communities like the Blackfeet Nation, as highlighted in Carlisle's (2022) *Healing Grounds*, have worked with the land for millennia, gaining valuable insights into managing variability. By adopting thoughtful, well-designed sampling strategies, we can effectively address variability to produce more reliable findings. By embracing rather than ignoring the complexities of soil variability, we can better assess 1) which changes in grazing practices yield the greatest soil improvements and, conversely, 2) which changes in grazing practices do not seem to affect soil properties.

Soil Health Indicators

To determine whether changing grazing strategies affects soil health, soil properties must be measured before and after implementing the new practices. Assessing soil health requires an integrative approach, using indicators from multiple categories—physical, chemical, and biological properties—to capture the complexity of soil functions – the biogeophysical processes that occurs within a soil – and their responses to grazing (Doran and Zeiss 2000, Stott 2019). Soil health is best evaluated using indicators linked to soil functions (Table 1.3), which vary with land management and environmental conditions (Guo 2021). While total SOM remains a commonly used indicator, novel metrics focusing on ecological functions may provide a more accurate understanding of soil health under regenerative practices (Hurisso et al. 2016, Oldfield et al. 2024). Developing standardized soil health assessment protocols can help connect soil indicators to agroecological outcomes, such as productivity, carbon storage, and water quality (Janzen et al. 2021). A critical consideration in this soil health assessment is how much it will cost to find a change in soil health. Before this question can be answered, the variability of each soil health indicator must be assessed.

Table 1.3. Modified NRCS indicators of soil health, with more stars signaling a stronger relationship between an indicator of soil health and soil function (NRCS 2021). Indicators are ranked by total sum of stars (last column) within each of the three categories (first column): biological (B), chemical (C), and physical (P). According to this listing the top-ranked indicators for each category would be earthworms, particulate organic matter, and bulk density.

Category	Soil Quality Indicator	Sustain biological diversity, activity, and productivity	Regulate and partition water and solute flow	Filter, buffer, degrade, detoxify organic and inorganic materials	Store and cycle nutrients and carbon	Physical stability and support for plants and structures associated with human habitation	Total Stars (out of 30)
B	Earthworms	★★★★★	---	★★★★★	★★★★★	★★★★★	24
B	Bulk Density	★★★★★	★★★★★	---	★★	★★★★★	20
B	Aggregate Stability	★★★★	★★★★	---	★★★★	★★★★	17
B	Available Water Capacity	★★★★★	★★★★★	---	★★★★	---	15
C	Particulate Organic Matter	★★★★★	★★★★★	★★★★★	★★★★★	★★★★★	30
C	Reactive Carbon	★★★★	★★	★★★★★	★★★★	★★★★	20
C	Potentially Mineralizable Nitrogen	★★★★★	---	---	★★★★★	---	12
C	Slaking	★★	★★★★★	---	---	---	8
C	Infiltration	---	★★★★	★★	---	---	6
P	Soil pH	★★★★	★★★★★	★★★★★	★★★★★	---	22
P	Soil Respiration	★★★★★	---	★★	★★★★★	★★★★	18
P	Soil Structure and Macropores	★★★	★★★	★★	★★	★★★★	14
P	Soil Enzymes	★★★★★	---	---	★★★★★	---	12
P	Soil Electrical Conductivity	---	★★★★★	---	---	---	6
P	Soil Crusts	---	★★★★★	---	---	---	5
P	Soil Nitrate	★★	★★	---	---	---	4

Chemical Indicators

Soil organic carbon (SOC) is a fundamental indicator because of its critical role in nutrient-cycling; soil structure, including compaction resistance, soil porosity, water retention,

water infiltration, and hydraulic conductivity (Rumpel et al. 2020). It also provides organic binding agents that promote aggregation and reduce bulk density, further benefiting soil structure (Blanco-Canqui and Benjamin 2013). SOC has an inverse relationship with bare ground as reduced vegetative cover leads to more significant erosion and diminished soil functions (Wang et al. 2018).

SOC is also central to climate mitigation efforts, making it a key metric to measure for sustainable grazing operations. Plants uptake CO₂ for photosynthesis, so increasing the rate of photosynthesis means more CO₂ removal if done at the right time and with the correct intensity. When grazers, such as cattle, remove above-ground biomass from plants, the plant can put more energy into its root production. This encourages root exudates, carbon-packed compounds that plants excrete into soil, and root growth (Lal 2004). When roots decompose, they add carbon to the soil, so more root growth means more potential SOC (Lal 2004). Additionally, rangeland management strategies attempt to evenly distribute grazing pressure, which can increase biodiversity and increase organic inputs from plant litter and livestock manure (Six et al. 2002, Provenza 2008, Teague et al. 2013).

Before endorsing regenerative grazing as the correct approach to increase SOC, evaluating how higher soil organic volumetric carbon (SOCv) is crucial to improved soil health. Additionally, it is essential to determine whether grazing strategies, such as rotational grazing, reliably enhance SOCv. SOCv is a more reliable and fairer estimate of soil carbon than mass-based soil organic carbon (SOCm) because it considers the (bulk) density, depth, and rock content of the soil. Studies suggest that while rotational grazing may increase SOCv, these

effects are highly context-dependent, influenced by precipitation, hillslope position, and elevation (Zhang et al. 2018, McKenna et al. 2022).

To illustrate, a study in South Africa found that rotational grazing had 28.6% less SOC_v than continuous grazing (Sandhage-Hofmann et al. 2015). This study area has 10-18 (25-46 cm) inches of rain per year, comparable with the Blackfeet Nation's rainfall of 11 inches (28 cm), although this study site is noticeable hotter with an average temperature of 64°F (18°C), which is much warmer than the Blackfeet Nation's average temperature of 46°F (8°C; NOAA 2025). Comparatively, in a study taking place near Queensland, Australia, rotational grazing increased SOC_v by 63%, with the average rainfall being 29 inches (74 cm) and an average temperature of 73°F (23°C). In a study taking place near Brandon, Manitoba, rotational grazing had no effect on SOC or BD, resulting in no significant difference in SOC_v (Banerjee et al. 2000). Brandon, Manitoba averages 18 inches (46 cm) of rain a year and has an average temperature of 38°F (3°C), so it is wetter and colder than the Blackfeet Nation (Government of Canada 2024).

Some studies have found increases in SOC_v using rotational grazing. A study in Texas found 7.5% higher SOC_v at sites with rotational grazing (Kim et al. 2023). Although the raw data were not reported, the statistics indicated that heavy, continuous grazing had the least SOC_v. and rotational grazing has the highest SOC_v. Similarly, rotational grazing in degraded South African soils decreased SOC_m by ~8% (Beukes and Cowling 2003). It is important to note that this is done in an area with only eight inches (20 cm) of rain a year, less than the 11-16 inches (28-41 cm) of precipitation that most Blackfeet Nation sees (NOAA 2025).

Also, SOC_m decreased on the non-degraded site when rotationally grazed at a high intensity (>1.5 AUMs ha⁻¹ year⁻¹). This supports the idea that sequestering carbon can occur

when the soil is degraded and not in an arid/semi-arid region, however, because SOC_v was not measured, the total soil carbon could have followed a different pattern (McSherry and Ritchie 2013).

The Manitoba experiment indicates that places with low rainfall and cold temperatures have trouble sequestering SOC_v from rotational grazing. If extrapolating to the Blackfeet Nation, it appears that increases in SOC_v may also be difficult to detect due to the cold temperatures and low rainfall. It is important to note that higher rainfall can enhance plant productivity, which in turn boosts root exudates and plant litter, both of which contribute to SOC (Lal 2004).

Though several rangeland management strategies show the potential to increase SOC_v by removing atmospheric CO₂, studies have yielded mixed results. A 36-study meta-analysis by (Piñeiro et al. 2010) revealed mixed results: where increased grazing pressure had been implemented, 12 showed decreased SOC_m, 12 showed increased SOC_m, and 12 showed no change from paired ungrazed plots. These studies were not directly related to rangeland management techniques; they only compared grazing with a high AU to a low AU but did not mention what they classified as high vs low AUs. Another more recent 40-study meta-analysis by Conant et al. (2017) revealed that half of the studies found 10% greater SOC_v but one-third with lower SOC_v after implementing “improved” grazing practices. These types of meta-analyses suggest that grazing effects on carbon sequestration can vary widely, with factors like soil type, climate, management intensity, and photosynthetic pathways (C3 grasses vs C4 grasses) all affecting outcomes (Teague and Kreuter 2020).

When rotational grazing was compared to continuous grazing in southern Australia, there was no difference in soil and rangeland health between native grasslands and the two treatment

groups (Sanderman et al. 2015). In another study from Australia, regenerative grazing practices were implemented for ~15 years, and the study could not conclude whether regenerative grazing could accelerate land improvements (Bartley et al. 2023). In a third Australian study, intensive rotational grazing was compared to continuous grazing at a constant and variable stocking rate. The results found that cattle weight gain was less under rotational grazing, implying it would be unlikely for such an approach to be adopted. That study also found no SOCv increases over time with rotational grazing (Schatz et al. 2020).

A study on the Northern Great Plains found that severe grazing ($1.5 \text{ AUM ha}^{-1} \text{ year}^{-1}$) was the best way to increase SOCv (22% increase over five years), which is unsustainable from a management perspective because it will reduce vegetation and increase bare ground (Reinhart et al. 2022). Even if SOCv is increased in the short term, the high intensity will lead to overgrazing, increasing bare ground cover, and decreasing SOCv in the long run. Just because very intense grazing can increase SOCv, does not mean that this should be a recommended intensity. The objective of ranchers is to sustainably graze, and severely grazing pastures has only short-term rewards.

A study in Africa found that AMP ($1.8 \text{ AUM per hectare}$) did not increase SOC at the top of the soil profile (5 cm). Although plant litter increased with high-intensity rotation grazing, hoof action did not break up the plant litter into more decomposable pieces (Beukes and Cowling 2003). These studies support work by Teague et al. (2013), who theorized that “positive feedbacks of grazers on ecosystems” are limited by low precipitation or extreme temperatures. The Blackfeet Nation experiences very cold temperatures, supporting the C3 grass-dominated ecosystem but resulting in a short growing season that restricts plant growth. On the contrary,

some African and Australian studies have found no increase in SOC_v in locations where water availability is limited, and temperatures can get extremely hot.

Vegetation affects soil properties and development, such as C3 vs C4 grasses (Jenny 1941). The main differences between these grass categories are that they have different photosynthetic pathways, with C3 grasses being less efficient in warm climates due to their inefficiency in losing water when fixing CO₂ (Taylor et al. 2010). Most studies reporting increases in SOC_v were done in C4-dominated areas, which are warmer climates. Several grazing strategies, however, recommend grazing intensities that are high or equivalent to continuous grazing systems (Zeitsman 2014). Cool-season, C3 grasses dominate Montana, and it has been demonstrated that grazing these grass systems at high-moderate to heavy intensities has negatively impacted SOC_m by 6-7%, as reported by McSherry and Ritchie's study (2013). This approach could have detrimental effects in dry regions dominated by C3 grasses, such as the Blackfeet Nation (McSherry and Ritchie 2013).

Montana has a cold, semi-arid climate, resulting in lower root abundance (Salazar et al. 2020, Sierra Cornejo et al. 2020). This results in a shallow biological zone where the root and microbe interaction is concentrated (Heckman et al. 2023). Montana's climate exacerbates this small zone because the ground is frozen during the winter, so the biological zone is active only 200-250 days a year (Chen et al. 2022). This limited biological zone results in Montana soils being slower to sequester SOC because microbes cannot decompose organic matter (Rieke et al. 2022). Furthermore, SOC accumulation is constrained by saturation limits; soils eventually reach an equilibrium where carbon gains plateau (Lavallee et al. 2020). Stored carbon may also be

susceptible to mineralization (loss) if management practices change or drought occurs, which can quickly reverse any sequestration gains (Freibauer et al. 2004, Teague et al. 2013).

SOC is a valuable property to track overall shifts in soil health, but increasing SOC this in cold, arid areas presents significant challenges. Even if SOC gains are achieved, a drought could quickly reverse those gains in just a few years, regardless of management practices (Soares et al. 2023). During drought conditions, microbial communities become less efficient at utilizing carbon. One field study revealed that for every decrease in the Aridity Index, carbon use efficiency was reduced by half (Pei et al. 2024). As microbial efficiency declines in droughts, the same microbes use significantly more carbon, leading to notable losses in SOC, especially where SOC inputs might already have been low. Given that climate change can lead to longer and more severe droughts, ranchers must focus not only on sequestering carbon but also on enhancing their soil's resilience to drought conditions (Moore and McEvoy 2022, Williams et al. 2024).

Unpublished long-term data from this project suggest that Blackfeet soils may respond slowly to management changes (see Appendix D), indicating that grazing strategies intended to increase SOC may not yield quick or significant results. Thus, these results suggest that we are unlikely to observe measurable differences in SOCv or other soil health indicators due to practices of similar intensity over shorter intervals or practices of lesser intensity over the same or longer intervals. Therefore, considering these findings, exploring alternative soil health indicators, especially for shorter grazing experiments, is crucial to better inform ranchers' adaptive strategies in a changing climate.

Although SOC has been discussed and is a very popular soil health indicator, a variety of soil health indicators need to change for a rangeland management strategy to benefit soil health.

It is challenging to find changes in SOC_v for the above reasons; there is new interest in a subset of that SOC. SOC is not all the same, and that fact has captured the attention of many scientists, such as when humics, fulvics, and humins were considered the drivers of soil fertility in the 1960s (Schnitzer 1969, Pettit 2020). Currently, there is considerable interest in soil density and/or soil size fractions including mineral-associated organic matter (MAOM) and particulate organic matter (POM), which may reflect carbon pools with different stabilities or residence times (Cotrufo et al. 2019, Lorenz et al. 2019, Wade et al. 2022). The foundational concepts of MAOM and POM, which date back to the 1980s (Christensen 1987), continue to influence modern research and soil health understandings.

MAOM is typically defined as being associated with particles smaller than 53 μm (silts and clays); MAOM is smaller than POM ($>53 \mu\text{m}$) and binds to clays ($<2 \mu\text{m}$) and fine silts ($2\text{--}5 \mu\text{m}$) in soils. MAOM is considered a longer-lasting form of SOC due to its microbial origins and resistance to decomposition (Shirazi et al. 2001, Cotrufo et al. 2019, Heckman et al. 2023). Over time, organic matter loses molecular diversity and complexity during long-term decomposition and transforms into a more stable form, MAOM (Kleber et al. 2015). Unlike more fibrous particulate organic matter (POM), which has faster turnover and is more vulnerable to disturbance, MAOM is thought to provide stable carbon storage that can persist even under changing soil conditions (Figure 1.5, Poeplau et al. 2018, Lavalley et al. 2020).

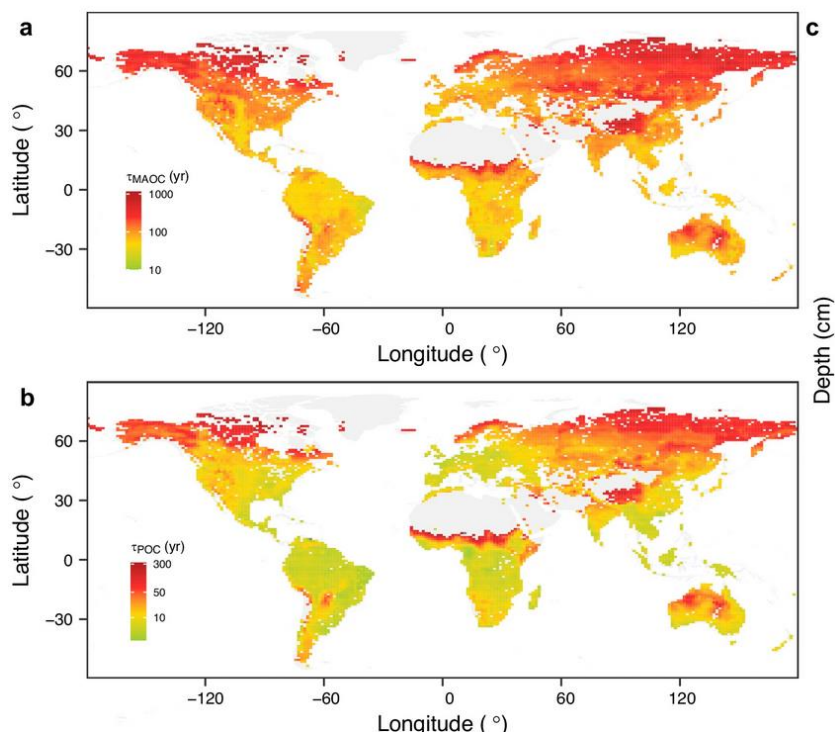


Figure 1.5. **a** mineral-associated organic carbon (MAOC) turnover time (0–100 cm). **b** particulate organic carbon (POC) turnover time (0–100 cm, from Zhou et al. 2024).

Quantifying MAOM, however, is complex and not yet fully standardized across soil health assessments. Studies indicate that MAOM can range from 15% to 98% of total SOM in mineral soils, with discrepancies often stemming from soil type and local environmental conditions (Gregorich et al. 2006, Haddix et al. 2020). While clay-rich soils might have a higher total surface area to sorb and protect MAOM from decomposition, studies have also suggested that increased clay does not guarantee increased MAOM levels (e.g., Midwood et al. 2021). Thus, soil management strategies, including enhancing water-holding capacity, applying organic amendments, and managing grazing practices, may help enhance MAOM without reliance on soil texture alone (Midwood et al. 2021, Khatri-Chhetri et al. 2024).

Despite its stability, MAOM, like SOC, has a “carrying capacity” in soils, where further increases become challenging, particularly in systems where MAOM content has already saturated soil binding sites (Stewart et al. 2007). Attempts to increase MAOM have been successful with rangeland management strategies that increase straw in soil (bale grazing) and adaptive multi-paddock grazing systems (Christensen 1987, Khatri-Chhetri et al. 2024). The potential for MAOM as a soil health metric, especially in regions like the Blackfeet Nation’s drylands, remains promising but needs refinement before broad implementation (Bartley et al. 2023, Augustine et al. 2023).

While MAOM’s variability and high measurement costs currently limit its practical applications, alternatives like permanganate oxidizable carbon (POXC) are emerging as promising soil health indicators (Romero et al. 2021). POXC can provide timely insights into biological function and nutrient mineralization (Bongiorno et al. 2019). POXC indexes active soil carbon, or carbon that can be easily oxidized and serves as an immediate, easy energy source for microbes (Weil et al. 2003). The benefit of this indicator is its consistency; it has the same values when the same soil is used (CVs<6%), irrespective of the test method or sample size used (Pulleman et al. 2021). This advantage is similar to that of SOC_m, with CVs<5% being common (see Appendix F, Table F.1); water-stable soil aggregation results, by contrast, can depend sensitively on sample weight and aggregate size (Flynn et al. 2020). Another benefit of POXC is its relationship with management regarding different farming practices. POXC can be more closely linked with management than SOC_m, microbial biomass, and particulate organic carbon (similar to POM; Culman et al. 2012). POXC is a reliable biological soil health indicator that can pick up carbon stock shifts faster than well-known indicators (Pulleman et al. 2021).

The Autoclaved Citrate Extractable (ACE) Protein Index (Soil Protein) indicates protein-like substances in soil organic matter. Soil Protein represents the largest pool of organically bound nitrogen (N) in soil organic matter, which microbes can mineralize and make available for plant uptake.

Soil Protein not only relates to biomass but also exhibits less fluctuation throughout the growing season compared to total and inorganic soil nitrogen, indicating its reliability as a soil health indicator for sustainable agricultural practices (Naasko et al. 2024). This suggests that the time of sampling has a limited impact on soil properties, which is advantageous for producers who may struggle to sample during specific times of the year, particularly during busy periods like calving, weaning, and branding. Soil Protein positively correlates with less intensive farming practices, such as no-till methods and increasing protein levels (Naasko et al. 2024). In rangelands, applying AMP grazing with a multi-species livestock system involving pigs, sheep, chickens, and cattle grazing in the same pasture resulted in a remarkable 500% increase in Soil Protein levels (Rowntree et al. 2020). This emphasizes diversifying livestock practices as a viable method for improving soil health. Since rangeland soils are less disturbed and not tilled, they should exhibit higher ACE protein levels compared to the typically more disturbed soil tested by Cornell University (Sever 2024). It is worth remembering that soil protein is a very small fraction of the overall soil organic matter. For example, if soil contains 2% SOC, then at a C:N ratio of 10, there might only be 0.2% SON, and perhaps 1/6 of that total pool of soil nitrogen may be Soil Protein.

Biological Indicators

Biological indicators play an essential role in understanding soil health, albeit with certain limitations (Fierer et al. 2021). In working with ranchers, a common goal is to sequester SOC_v, which means having greater carbon inputs than carbon losses/ mineralization rates. While greater soil biological activity generally promotes soil health, it also leads to increased microbial consumption of soil organic matter (SOM), which releases carbon back into the atmosphere as CO₂ (Trivedi et al. 2016, Rieke et al. 2022). Soil microbial biomass (SMB) and DNA or RNA tests are also popular methods for indexing the soil microbial community composition, but their high within-sample variability can complicate soil health assessments.

Mitchell et al. (2023) reported results for four microbial analyses (active and total fungi, active and total bacteria) and 8 treatments (32 combinations): the average coefficient of variation (CV) was 70%, with 7 of the 32 combinations having a CV >100%. These high CV's indicate high variability between the tests. A CV > 100% means that the standard deviation is greater than the mean. Seventeen combinations had CVs between 50% and 100%; of the remaining 8 combinations, only 2 had CVs <25%. This variability makes it extremely difficult and expensive to find significant differences between treatments.

Intra-sample variability is a common shortcoming of Soil microbial biomass (SMB) indicators. While high SMB can enhance SOM and aggregation due to microbial activity and polysaccharide production (Bhaduri et al. 2022), these tests may have limited utility in producer-focused soil health assessments due to high spatial and temporal variability. Microbial soil tests can be more influenced by seasonal changes than by management practices, raising concerns over their reliability as rangeland soil health indicators; for example, Ishaq et al. 2020 found that soil microbial diversity was significantly higher at the beginning June than at the end, yet there

was no evidence that organically managed farms had a different microbial diversity compared to conventional farms. The relationships between microbial abundance and ecological function, particularly for ranching or pasture management, remain uncertain and costly to interpret (Fierer et al. 2021). One study (Vries et al. 2012) examined biological indicators across fertilized, lightly grazed, and traditionally managed sites, revealing significant variability in SMB (Figure 1.6). At the fertilized and grazed sites, the SMB average ($\pm 1SD$) was $803 \pm 780 \mu\text{g g}^{-1}$ (CV 97%), and at the traditionally managed sites, the SMB was $1175 \pm 991 \mu\text{g g}^{-1}$ (CV 84%). Such large CVs make it challenging to draw definitive conclusions about how different management strategies might affect SMB (Vries et al. 2012). Using ClinCalc's sample size calculator (<https://clincalc.com/Stats/SampleSize.aspx>) and inputting the average and standard deviation of each treatment groups, detecting a 10% increase in SMB at a power of 0.80 and $\alpha=0.05$, would require nearly 3,000 samples for the fertilized and grazed treatment. Using the same power and alpha, approximately 2,300 samples would be needed to see a 10% increase for the traditionally managed group, raising concerns about feasibility. At \$30 per sample, detecting such a change in SMB could cost up to \$157,000, highlighting one of the financial burdens soil health assessments could pose for producers and researchers alike (Cornell University 2024, UC Davis 2024).

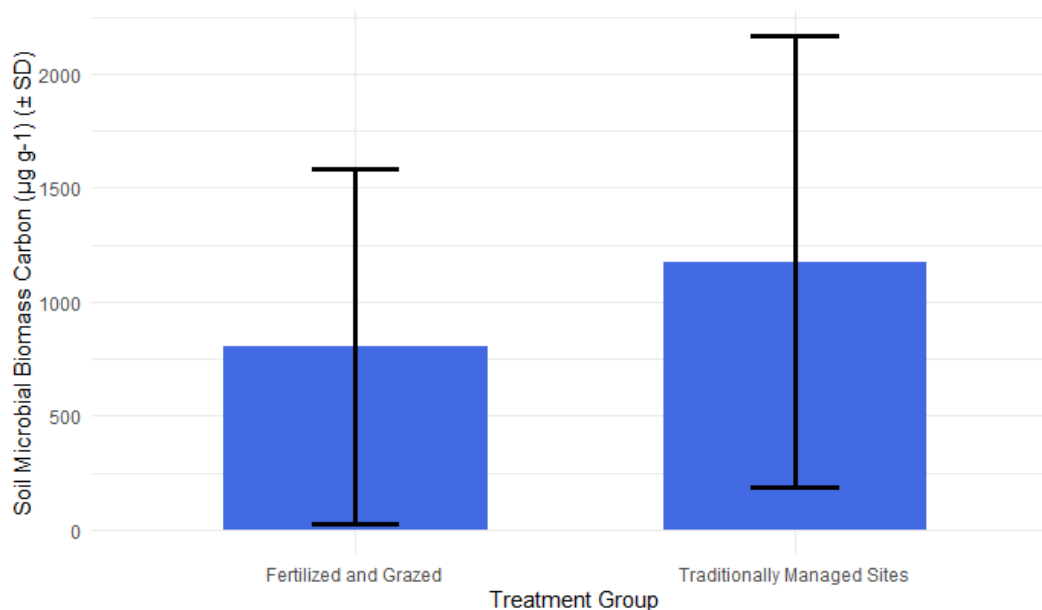


Figure 1.6. Soil microbial biomass C (MBC) data reported by Vries et al. (2012) allow for calculations of CVs (fertilized and grazed = 97%; traditionally managed = 84%) as well as power analyses to explore the number of samples required to detect different magnitudes of MBC improvement. Error bars represent one standard deviation (fertilized and grazed = 780; traditionally managed = 991).

Potential carbon mineralization (PCM) is a useful biological indicator of soil health. It's measured by rewetting soil, adding a substrate like diluted yeast, and tracking CO₂ release to estimate microbial activity (Zibilske 1994). Often called a 24-hour “burst” test, this method—commonly done with the Solvita CO₂ Burst—reflects how easily microbes can metabolize dissolved organic carbon (Haney et al. 2008). Substrate additions trigger the breakdown of more complex carbon forms (Birch 1958). The test is now standard for assessing soil biology's role in carbon stability (Schädel et al. 2020), with higher PCM values seen in undisturbed soils due to stronger microbial activity and organic matter retention (Nunes et al. 2021).

However, PCM assessments face interpretation challenges; higher values don't always equate to improved soil function. Like indicators of carbon storage or plant nutrition, they may simply reflect a temporary spike in SOM turnover. This activity can boost CO₂ output without necessarily leading to stable carbon retention or long-term soil structure and fertility improvements (Nunes et al. 2021, Fierer et al. 2021).

PCM is complex to contextualize because although higher PCM indicates higher biological activity, it also means that SOC is being mineralized at a higher rate, signifying a less stable SOC_v pool. So, sustained higher PCM rates could lead to lower SOC_v. Because increasing SOC_v is a major goal of ranchers on this project, it is important to note that if the PCM is high, sequestering carbon will be tough because carbon inputs are greater than carbon losses. Implementing carbon inputs is particularly challenging in semi-arid environments due to limited moisture for plant growth, frequent droughts, and low organic matter turnover, making it more difficult for ranchers to sequester carbon effectively (Bartley et al. 2023).

Typical rangeland PCM rates are between 8 and 16 mg kg⁻¹d⁻¹, indicating that with a constant rate of ~10 mg kg⁻¹ d⁻¹ it could take ~10,000 days (~30 years) to mineralize 100 megagrams of carbon ha⁻¹0.3m⁻¹, about the average found on the Blackfeet Nation from a previous study (Haigh et al. 1979, Cheng et al. 2024). Higher PCM values may reflect improved residue quality, which in rangeland ecosystems often includes plant litter originating from forage that has been subjected to varying levels of grazing intensity (O'Dea et al. 2015). Studies are working to clarify which microbial communities drive these measurements under minimal disturbance to better link PCM values to actual soil functions, offering more practical insights for agricultural management (Malik et al. 2018, Liu et al. 2022, Barnett et al. 2022). This biological

soil health indicator represents how well carbon is cycled. High rates indicate an active microbial community that can be quite productive in the presence of additions of organic matter. However, suppose a disturbance or land management change decreases organic matter input. In that case, excess carbon can be volatilized rapidly, resulting in significant SOC_v loss on the landscape.

Physical Indicators

Good soil health indicators should exhibit low variability while representing multiple soil functions. Physical soil properties are often linked to several soil functions, so it is important to include physical indicators in a soil health assessment (Cardoso et al. 2013, NRCS 2021). Physical indicators include porosity, bulk density, soil texture, soil structure, aggregate stability, color, temperature, and moisture content. Not all physical properties of soil are effective indicators; for instance, soil temperature and moisture content change too quickly, whereas soil structure and color evolve too slowly (Pyke et al. 2002, Cardoso et al. 2013, Guo 2021).

Bulk density (BD) is an indicator influenced by grazing practices, yet it tends to remain stable throughout the year. In various studies, overgrazing has been shown to significantly affect the soil's bulk density, leading to compaction, while certain grazing strategies can actually reduce BD and enhance the soil's capacity to retain water (Drewry et al. 2008, Lal et al. 2019, Augustine et al. 2023). BD represents how tightly soil particles are packed into the soil profile, which can correspond to infiltration and compaction but is also texture-dependent (Heckman et al. 2023). No grazing or very light stocking rates can decrease BD, while continuous or rotational, intensive grazing can increase BD (Li et al. 2012, Lal et al. 2019). Therefore, reducing BD through intensive grazing poses challenges unless the soil is already compacted or if significant accrual of SOC is anticipated.

BD measurements are crucial in SOCv estimation. Yet, BD is difficult to collect accurately, as measurement tools and methods influence results. Standard soil push probes, for instance, express four times the error rate of cylinder methods (Walter et al. 2016; Figure 1.7). When collecting soil samples, coring can lead to compaction, particularly in wet soils, which could yield deceptively high BD measurements.



Figure 1.7. A standard push probe used to collect soil cores mentioned in Walter et al. (2016).

The Cornell Soil Health lab employs a rainfall simulator, which can cost around \$1,200 (Schindelbeck et al. 2016), highlighting the financial burden associated with standard soil testing methods. Fortunately, the free Slakes App (<https://soilhealthinstitute.org/our-work/initiatives/slakes/>) developed by the Soil Health Institute provides a more affordable and efficient method for assessing this physical soil health indicator than traditional methods. This slaking method is more cost-effective and demonstrates greater sensitivity to changes in management practices (Flynn et al. 2020), making it highly relevant for soil health assessments. To conduct the slaking method, only a smartphone with the app, a petri dish (or any suitable

cup), and a stable surface are needed. A low-cost and easy-to-conduct test like the Slakes app represents an advantage for ranchers, providing essential soil health insights without any associated costs. A key advantage of the Slakes app is that ranchers can use it in the field to obtain immediate data, empowering timely decisions regarding their soil management.

Conclusion

In summary, regenerative grazing offers potential benefits for soil health through improved SOC, infiltration, and nutrient cycling. However, results can vary widely based on specific contextual factors, with some studies reporting only modest improvements. Future research should prioritize standardized methods, region-specific practices, and long-term monitoring to better understand and optimize regenerative grazing's impact on soil health (DeLuca and Zabinski 2011, Booker et al. 2013). In light of these complexities, scientists should focus on adaptive management strategies rather than simple, standardized treatments, acknowledging the multitude of factors influencing treatment outcomes (Teague et al. 2013). Variability in precipitation across regions leads to diverse vegetation types, influencing the management strategies employed in grazing. In the same pasture, there can be differences in hillslope and soil type, both of which affect soil health and add to the heterogeneity of the landscape (Aguilar et al. 1988, Heimsath and Burke 2013). Consequently, cattle management must be conducted using flexible grazing strategies that tailor plans according to local conditions and operational styles (McSherry and Ritchie 2013, Wang et al. 2021).

Thus, monitoring systems must be refined for clarity and standardized methods of measuring soil health indicators. The same indicators should be used to compare soils to find

changes or differences in soil health. Current studies assess a range of soil health indicators; however, inconsistencies in methodologies hinder effective comparisons. Therefore, adopting a comprehensive suite of cost-effective soil health indicators that exhibit low variability and align with management practices is essential for evaluating the effectiveness of rangeland management strategies.

CHAPTER THREE: BASELINE PATTERNS OF RANGELAND
SOIL HEALTH - INDICATORS ACROSS THE BLACKFEET
NATION

Contribution of Authors and Co-Authors

Manuscript in Chapter 2

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Manuscript Information

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3.1 Abstract

Soils are critical to ecosystem health, supporting biodiversity, water cycling, and carbon storage. However, baseline soil data for Indigenous lands, such as the Blackfeet Nation in northern Montana, remains limited. Here, we characterize soil health across 28 ranches and explore these soil properties using the CLORPT framework (climate, organisms, relief, parent material, time) and focusing on ecologically relevant and scalable indicators. Volumetric organic carbon (SOC_v) is a critical metric due to its mathematical incorporation of both varying soil bulk densities and rock contents, providing a more integrated representation of landscape patterns of carbon storage than mass-based measures of SOC. SOC_v to 1 foot depth (30 cm) ranged from 7 to 124 tons per acre-foot (16-279 Mg ha-0.3m⁻¹), averaging ($\pm$ 1SD) 42 $\pm$ 21 tons per acre-foot (94 $\pm$ 47 Mg ha-0.3m⁻¹). Not surprisingly, different ranches, and different hillslope positions within those ranches, showed different levels of SOC_v heterogeneity, even when sampled ~10 feet apart, which we quantified as coefficients of variation (CV) using a quadruplicate sampling approach. For example, across more than 100 hillslope positions sampled as part of this project, CVs ranged from 4 to 76% (averaging 21 $\pm$ 14%). These results suggest that most Blackfeet Nation sites will require unrealistic numbers of soil samples to detect the modest increases in SOC_v reported in the literature for improved grazing practices. For example, our power analysis calculations suggested that Ranch J alone would require 210,000 or 2,200 soil samples to detect a +1% or +10% annual increase in SOC_v, respectively, more samples than we collected across this entire project. Consistent and aligned with Blackfeet agricultural management priorities, we therefore recommend using our baseline soil findings to prioritize the lowest-variability Blackfeet soils for pilot-scale interventions while reviewing alternative measures of rangeland soil health such as topsoil erosion rates, forage quantity and quality, and soil moisture.

3.2 Introduction

Soils are foundational to ecosystem health, supporting biodiversity, water cycling, and carbon storage, yet baseline soil data for many Indigenous lands, including the Blackfeet Nation in northern Montana, remain limited. The Blackfeet Nation, located in North-western Montana, in the Northern Great Plains, with its diverse climates, glacial histories, and combinations of traditional and modern land management practices, provides an ideal and unique setting to explore the factors that shape soil properties. These state factors have frequently been conceptualized as climate, organisms, relief, parent material, and time (hereafter referred to as CIORPT (Jenny 1941, McKenna et al. 2022), which provides a systematic approach for examining soil landscape patterns. Some authors (e.g., Teague and Kreuter 2020, Kim et al. 2023) have suggested that soils can also be influenced by a sixth ‘state factor’—humans—so we consider management effects in our discussion.

Despite advances in soil science, significant knowledge gaps persist. For instance, the relative importance of individual CIORPT factors and their interactions varies across regions, making it difficult to predict soil properties in diverse settings like the Blackfeet Nation. CIORPT factors greatly influence soil properties; for example, soil organic carbon (SOC) can vary by threefold across different climates (McKenna et al. 2022); threefold across a tenfold increase in quartz presence in parent materials (McKenna et al. 2022); and sixfold across a 20-fold shift in duration of soil-forming and soil-transforming factors (Zheng et al. 2019). Additionally, the role of "humans" as a sixth state factor remains understudied, particularly in contexts where Indigenous land management practices have been occurring for tens of thousands of years.

This study addresses these gaps by documenting baseline soil properties as part of a multiyear, multiorganization research and education effort that examines how different management practices may shift those properties toward improved soil health. Soil health in this study was measured using indicators that are both ecologically relevant and thought to be feasible to assess at scale. Specifically, soil health indicators focus on physical, chemical, and biological properties such as soil texture, pH, organic matter content, and nutrient availability.

SOM is closely linked to soil functions, making it a crucial measure of ecosystem health (Lorenz et al. 2019, NRCS 2021, Wade et al. 2022). The importance of SOM has been recognized for centuries, as historical agricultural texts (Wallerius 1753) and (Thaer 1815) suggest early interest in its role in soil crop productivity. We chose to examine the SOC fraction of soil organic matter (SOM) specifically because although SOM is essential for soil function, it is a complex mixture of substances that vary greatly depending on soil moisture, microbial activity, and decomposition stages (Gregorich et al. 2006). Labile SOC, easily decomposable fractions, provide immediate benefits to soil microbes and nutrient cycling, while more stable SOC fractions contribute to long-term improvements in soil structure and water retention, ultimately reducing erosion (Six et al. 2002, Yudina and Kuzyakov 2023). These properties are fundamental to soil resilience and productivity, particularly under the changing environmental conditions facing ranchers.

Volumetric Carbon (SOC_v) offers a more accurate representation of total soil carbon storage than mass-based SOC (SOC_m) measures alone. In rangelands, where CIPORT and management practices can significantly influence carbon dynamics, understanding SOC_v is particularly important. For example, within the US, climate can drive SOC accumulation; for

example, subtropical grasslands, such as those in the Southeastern USA, typically have moderate SOC levels, which can increase further through management practices such as adaptive multi-paddock grazing (also referred to as regenerative grazing [RG]; Santos et al. 2024). This may be attributed to warmer temperatures, deep-rooted grasses, and clay-rich, older soils (Jackson et al. 1996, Jobbágy and Jackson 2000). In contrast, the Northern Great Plains, including the Blackfeet Nation, are much colder, receive far less precipitation during the growing season, support mostly shallow-rooted plants (except for sagebrush), and are generally underlain by younger soils—factors that contribute to lower average soil organic carbon levels, such as 39 tons per acre-foot (87 Mg SOC [ha-0.3m]⁻¹) compared to 47 tons (105 Mg SOC [ha-0.3m]⁻¹) in other regions (Haigh et al. 1979, Post et al. 1982, Jackson et al. 1996, Poggio et al. 2021).

The real challenge of measuring carbon stocks is how soil forming factors, including management and other disturbances, interact with the landscape, resulting in a heterogeneity of soil patterns. SOCv is especially variable, often having a low signal-to-noise ratio (Welsch et al. 2019). Most sampling techniques and experimental designs were created for croplands, so the fact that rangelands have a higher spatial heterogeneity than croplands makes accurately measuring SOCv difficult (Stanley et al. 2023). Because ranchers are interested in quantifying the effect of their management change on soil health, this variability needs to be assessed before regenerative grazing is implemented on these ranches. With the soil variability known, experiments can be designed for areas low in variability, where fewer samples would be needed to have a better chance of detecting small shifts in soil properties, including SOCv.

An increase in SOCv should result in a better water holding capacity, lower bulk density, easier rooting and germination for native plants, better soil structure, and less soil erosion (Ontl

and Schulte 2012, Blanco-Canqui and Benjamin 2013, Liptzin et al. 2022). Many factors influence SOCv, which is asymmetrically important for explaining the variances measured across the Blackfeet Nation. In other words, SOCv is likely greater where it is:

1. Wetter;
2. Dominated by native vegetation;
3. In toeslopes
4. Underlain by clayier substrates
5. Has not been glaciated relatively recently

Using these factors, we explain SOCv patterns and variability while informing rangeland management and monitoring. Because our baselining methods allow for a quantification of local variability at each hillslope position sampled, we used our data to calculate the number of samples that would be needed to see a change in SOCv. Sites with a lower coefficient of variation (CV) should be targeted for RG implementation changes in soil properties are easier to detect where the variability is low.

3.3 Materials and Methods

3.3.1 Site Description

The study area is the Blackfeet Nation in northern Montana, bordering Canada and Glacier National Park (Figure 2.1). Although the Blackfeet Nation was created by treaty in 1855, today it stretches over one and a half million acres (MBMG 2020, Indian Affairs 2024).

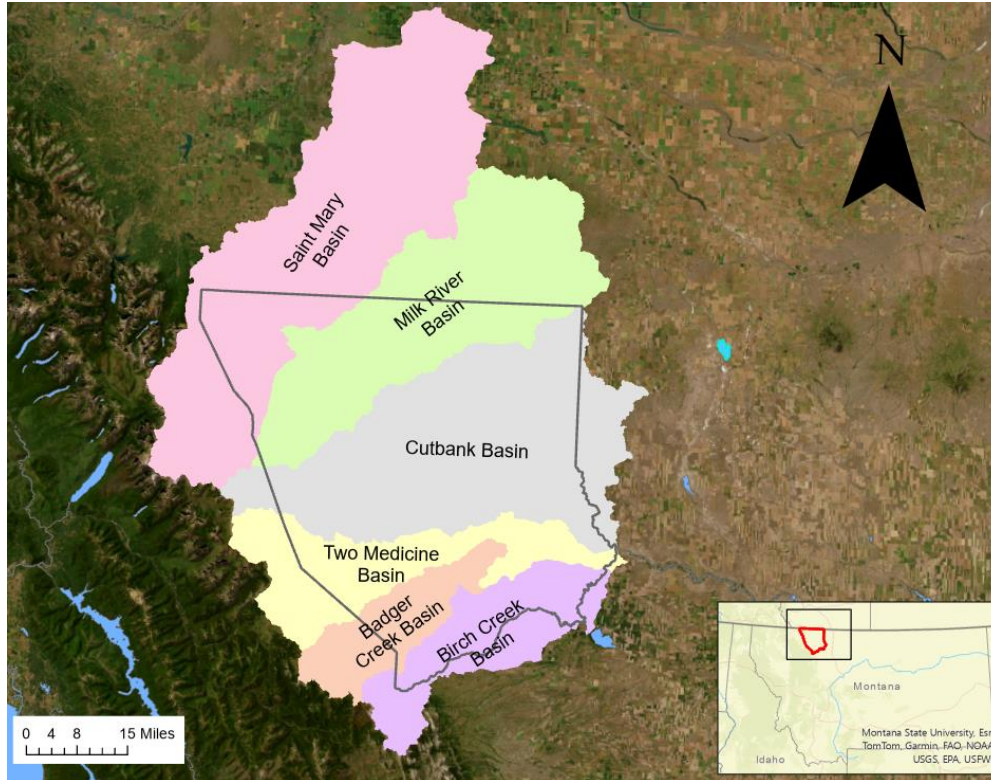


Figure 2.1. This map shows the Blackfoot Nation, bordered by Glacier National Park to the east and Canada to the north. Twenty-eight ranches were sampled across six major watersheds: St. Mary's Upper Basin ($n = 3$), Milk River Upper Basin ($n = 1$), Cutbank Upper Basin ($n = 5$), Cutbank Lower Basin ($n = 8$), Two Medicine Upper Basin ($n = 1$), Badger Creek Upper Basin ($n = 2$), Badger Creek Lower Basin ($n = 2$), Birch Creek Upper Basin ($n = 4$), and Birch Creek Lower Basin ($n = 2$).

3.3.2 Soil Sampling Collection

We collected soil from ranches on the Blackfoot Nation (see Figure 2.1). Soils were sampled by landscape location, which are categorized from uplands and are the equivalent of "crests" through "shoulders," to "footslopes" and "toeslopes" (Milne 1947;). When arranged linearly, these positions create a soil catena- a series of soils arrayed down a slope (Milne 1947). We sampled 118 hillslope positions on 28 catenas or rhachnes. To protect rancher privacy, each ranch was assigned a unique letter.

At each hillslope position, four soil pits were dug to about one foot deep and wide. One pit face was analyzed for vertical differences in soil properties (e.g., rocks, colors, textures), defining horizons (Schoeneberger et al. 2012). Horizon depths and coarse fragment percentages were recorded, and samples were collected for lab analysis.

3.3.3 Laboratory Methods

After sampling, we processed air-dried soil in-house at Montana State University for SOM, SOC on a mass-basis (SOC_m), BD (bulk density), porosity, and aggregate stability. To quantify SOM, we used the loss-on-ignition (LOI) method (muffle furnace at 680°F [360°C] for 2 hours). SOC was estimated as half of SOM on Equation 2.11 that used a Costech ECS4010 analyzer to establish a statistically significant empirical relationship between SOC and SOM using 27 triplicates ($R^2=0.96$, $p<0.05$):

$$\text{SOC}_m (\%) = 0.493 \times \text{SOM} (\%) + 0.99 \quad (\text{Equation 2.1})$$

Volumetric carbon accounts for bulk density, coarse fragment content, and SOC_m (Chow et al. 2007). Our lab employed the water displacement method to measure bulk density (Hamakareem 2021).

We calculated SOC_v using Equation 2.2

$$\text{SOC}_v = \sum_i \left(\frac{\text{SOC}_m}{100} \right) * (\text{BD}_i) * (\text{HT}_i) * \left(1 - \frac{R_i}{100} \right) * 44.609 \quad (\text{Equation 2.2})$$

Where SOC_v is measured in English tons (2000 lb) per acre-foot, SOC_{m_i} is the %SOC_m for the *i*th horizon, BD_{*i*} is bulk density (g cm⁻³) for the *i*th horizon, HT_{*i*} represents the horizon thickness (cm) of the *i*th horizon, and R is the rock content for the *i*th horizon. A conversion factor of 44.609 is required to convert the horizon *i*-specific g cm⁻² results to horizon *i*-specific SOC_v units (t ac⁻¹) before summing all horizon-specific SOC_v to 30cm (1 ft) to get the final units of SOC_v in units of tons per acre-feet (t ac-ft⁻¹), which can be converted to Mg SOC ha⁻¹ by multiplying by 2.2417.

BD was also used to calculate porosity (Equation 2.3).

$$Porosity = \left(1 - \left(\frac{BD}{Particle\ Density} \right) \right) * 100 \quad (Equation\ 2.3)$$

Where porosity is the percent of airspace in the soil by volume, BD is the bulk density of the soil, and particle density is the density of the individual soil particles in g cm⁻³. We calculated particle density using the boiling method (GLOBE 2014). In this method, air-dry soil is placed in a volumetric flask; water is added and boiled for 10 minutes to remove all air from the soil pores. The original weight of the soil is divided by the volume of the soil, giving us a particle density (GLOBE 2014).

Soil aggregate stability was measured with a *Slakes* smartphone app developed by the Soil Health Institute (<https://soilhealthinstitute.org/our-work/initiatives/slakes/>) for rapid assessment (Flynn et al. 2020). The app quantifies soil aggregate disintegration in water and calculates a slaking index (SI) score, with lower scores indicating reduced stability. SI values range from 0 to 1, and replicate sample CVs averaged 8%.

3.3.4 Statistical Analysis

We performed simple linear regressions and multiple linear regression analyses in R version 3.4.1 (R Core Team 2021) to assess how the CLORPT factors independently and collectively influence SOC_v on 118 hillslope locations across 28 ranches. First, we ran individual linear regressions to evaluate the effect of each independent CLORPT factor separately on SOC_v (Appendix B). Recognizing that these factors are inherently interrelated, we used the coefficient of determination (R^2) to assess how much variability in SOC_v each factor could explain. Statistical significance was determined using a standardized threshold of $\alpha=0.05$.

Next, we performed a multiple linear regression to evaluate the combined effect of all CLORPT factors on SOC_v. After evaluating the predictive power of each of the five soil-forming factors (CLORPT) individually with SOC_v (Appendix A), a multilinear regression with the variables climate index (climate), vegetation index (organisms), hillslope position (relief), parent materials (parent materials), and glacial formations (time), was run to predict SOC_v. We employed a stepwise selection procedure using the *step()* function to refine the model, considering both forward and backward selection. While stepwise selection is computationally efficient, we acknowledge its potential limitations, such as overfitting or instability in coefficient estimates, and mitigate these risks by applying stringent criteria for model selection (Rose and McGuire 2019). Specifically, the final model was selected based on three criteria: the lowest p-value, the highest adjusted R^2 (which accounts for added complexity while adjusting for variance), and the lowest Akaike information criterion (AIC; Akaike 2014), which penalizes model complexity to prevent overfitting. The model selection process also explored all possible

two-way and three-way interactions among predictors, prioritizing interactions informed by best professional judgement.

To ensure model robustness, we assessed model fit using residual diagnostics. All regressions were conducted using the *lm()* function for continuous variables and *aov()* for categorical variables. Diagnostic plots were used to verify that each model met the assumptions of linearity, normality, and homoscedasticity. Multicollinearity was evaluated using variance inflation factors (VIF), and a threshold of five identified problematic predictors. We identified potentially influential data points using Cook's distance, with a threshold of $4/n$, where n is the number of observations. Observations exceeding this threshold were removed from the final model to minimize their impact on parameter estimates. The results were compared using their corresponding R^2 and AIC values.

We also used a Huber regression model to minimize the influence of outliers on the estimated coefficients; it combines the properties of the ordinary least squares regression and robust regression (LeRoy and Rousseeuw 1987). This model was used because of the relatively high variability of the dataset, as indexed by our SOCv CV of 51%. Instead of eliminating influential points, it reduces their impact on the model by applying a weight, or inequality of <0.001 . The Huber regression model was fitted using the *lmrob()* function from the *robustbase* package in R (Maechler et al. 2024).

3.4 Results

3.4.1 Blackfeet Nation-Wide Trends in SOCv and SOCm

Across 118 hillslope positions on 28 ranches, baseline soil organic carbon on a volume basis (SOCv) and soil organic carbon on a mass basis (SOCm) exhibited 18X and 47X

differences between our lowest and highest SOCv and SOCm, respectively. SOCv, for example, averaged (± 1 SD) 42 ± 21 t ac-ft⁻¹ (94 ± 47 Mg ha-0.3m⁻¹; CV 49%), with a range from 7 to 124 t ac-ft⁻¹ (16-279 Mg ha-0.3m⁻¹). SOCm averaged $3.3 \pm 1.9\%$ (CV 57%) and ranged from 0.3 to 14%. The hillslope position averages were right-skewed due to a subset of hillslope positions (n=19) with lower carbon (SOCm < 2% or SOCv < 20 t ac-ft⁻¹ [48 Mg ha-0.3m⁻¹], Figure 2.2). The highest SOCv and SOCm values (124 t ac-ft⁻¹ and 13.6%, respectively) were associated with a shoulder position of Ranch J in the lower Cutbank basin (Figure 2.5). Conversely, the lowest SOCv and SOCm (7 t ac-ft⁻¹ and 0.7%, respectively) were associated with the footslope of Ranch M.

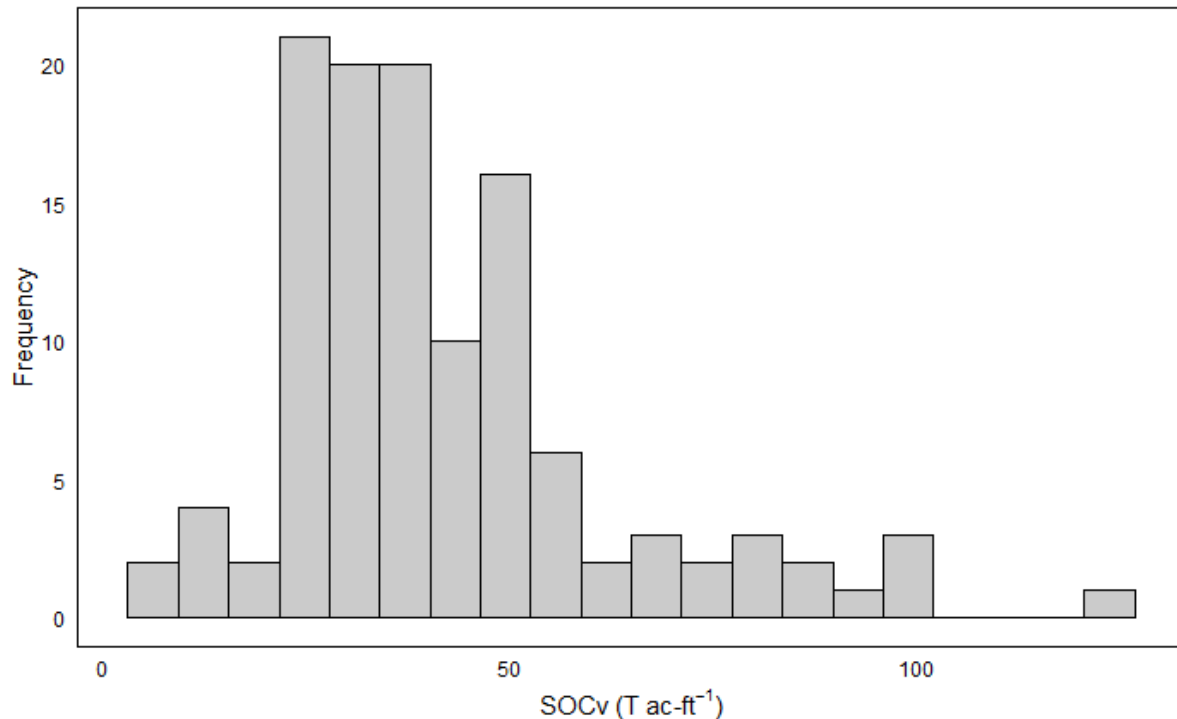


Figure 2.2. Histogram of all individual hillslope position SOCv values with an average of (± 1 SD) 42 ± 21 t ac-ft⁻¹ (94 Mg ha-0.3m⁻¹).

Because topography is part of the “relief” factor of CIORPT, and often influences soil properties, all hillslope positions were binned into crests, shoulders, or toeslopes to see the catena’s position effect on SOCv. Somewhat surprisingly, SOCv did not differ statistically among the hillslope positions ($p=0.08$). Crests averaged 38 ± 17 t ac-ft⁻¹ (94 Mg ha-0.3m⁻¹; CV=45%), shoulders averaged 40 ± 19 t ac-ft⁻¹ (90 Mg ha-0.3m⁻¹; CV=48%), and footslopes averaged 48 ± 24 t ac-ft⁻¹ (107 Mg ha-0.3m⁻¹; CV=50%; Figure 2.3).

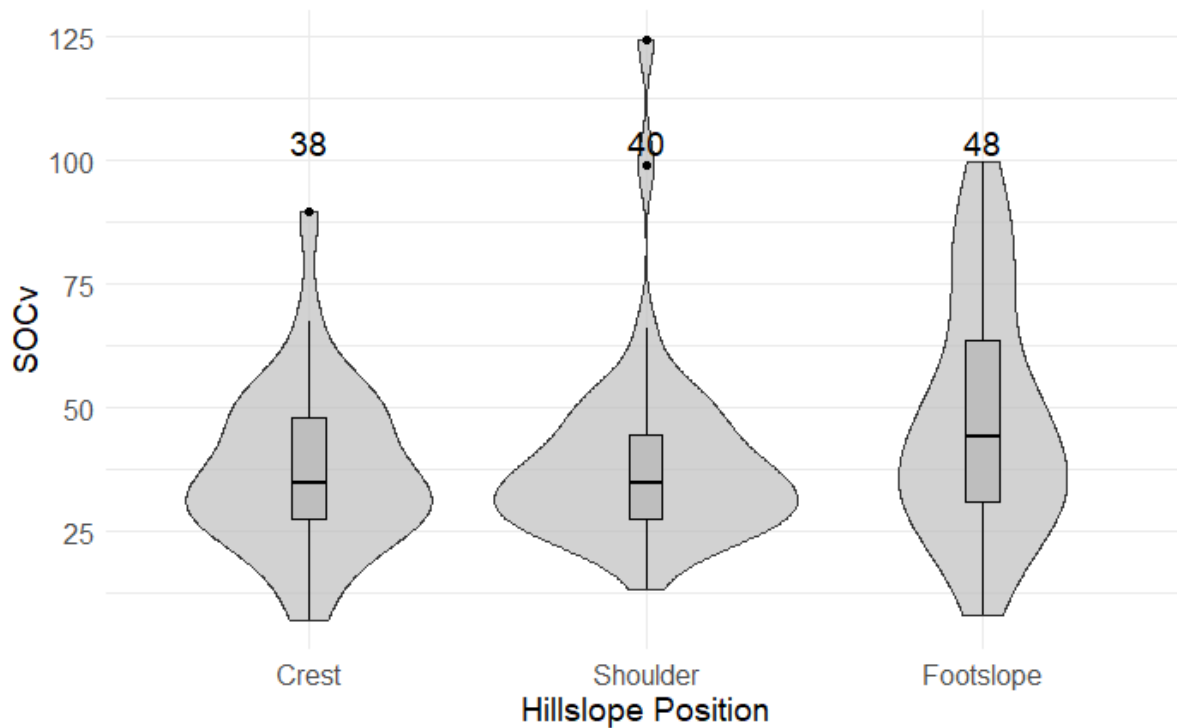


Figure 2.3 Violin plots of SOCv (t ac-ft⁻¹) by hillslope position; rectangle lengths correspond to the interquartile range (IQR), running from the 25th percentile (lower box edge) to the 75th percentile (upper box edge). The bolded horizontal line represents the median (50th percentile) while the vertical lines extend to 1.5 times the IQR. Width of each plot correlates to frequency of observations

We analyzed the influence of four climate zones by dividing the Nation into a northwestern, highest-elevation, wettest quadrant (average of 26.1 ± 6.1 in of rain, $39.3 \pm 1.1^\circ\text{F}$); a

northeastern, lower-elevation, driest quadrant (average of 12.5 ± 0.2 in of rain, $41.8 \pm 0.1^\circ\text{F}$); a southeastern, lowest-elevation, dryer quadrant (average of 14.6 ± 1.0 in of rain, $41.7 \pm 0.4^\circ\text{F}$); and a southwestern, higher-elevation, wetter quadrant (average of 20.5 ± 6.4 in of rain, $40.7 \pm 0.6^\circ\text{F}$). Average SOCv did reveal the expected trend, with decreasing SOCv with increasing aridity, but because of the inter quadrant variability, there was not a significant relationship ($p=0.12$, Figure 2.4).

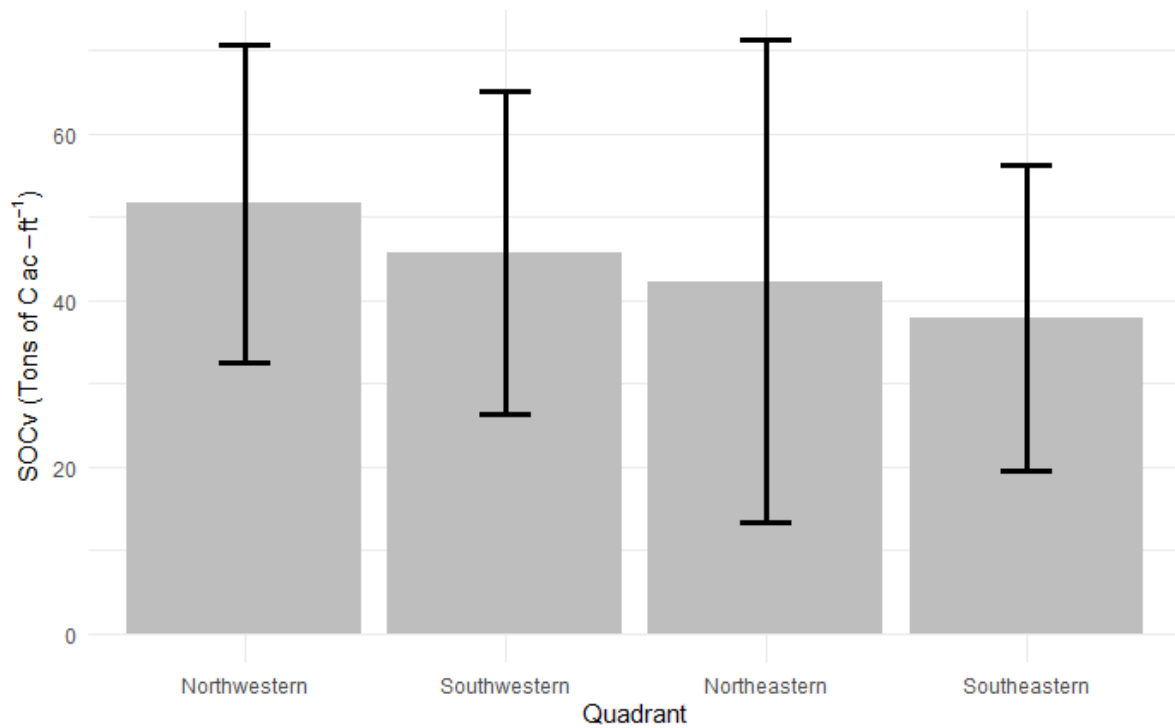


Figure 2.4 Average hillslope SOCv (t ac-ft^{-1}) by the quadrant within the Blackfoot nation that the ranch is located in. Quadrants are arranged from least to most arid. Error bars show 1SD.

3.4.2 Ranch-Specific Trends in SOCv and SOCm

At the individual ranch scale, spatially integrating across hillslope positions, SOCv was more variable than SOCm. For example, ranch SOCv averaged $41 \pm 16 \text{ t ac-ft}^{-1}$ ($90 \text{ Mg ha-0.3m}^{-1}$

[CV 39%]) and ranged from 20 to 63 t ac-ft⁻¹ (44 to 130 Mg ha-0.3m⁻¹). Depth-averaged SOC_m, by contrast, averaged 3.3±1.1 (CV 33%) and ranged from 1.6 to 6.3%.

These averages minimize substantial between-ranch differences in SOC_v variability. For example, ranch H was the most consistent ranch, averaging 32 t ac-ft⁻¹ (72 Mg ha-0.3m⁻¹; CV 20% [Figure 2.5]), whereas ranch J, the most variable ranch, averaged 59 t ac-ft⁻¹ (131 Mg ha-0.3m⁻¹; CV 94% [Figure 2.6]), nearly a 5-fold difference.

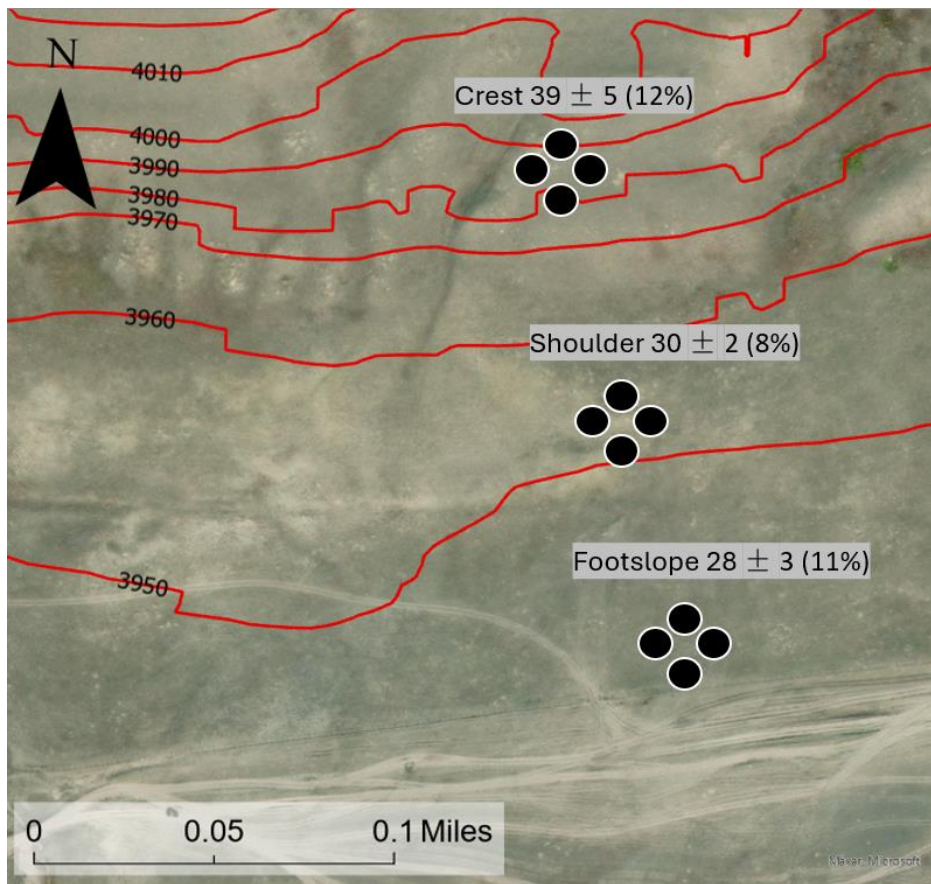


Figure 2.5. Map of low-variability ranch H showing average ±1SD (CV%) hillslope SOC_v in t ac-ft⁻¹. Black circles are stylized locations of quadruplicate soil pits. Note that the catena generally runs north to south-southeast.

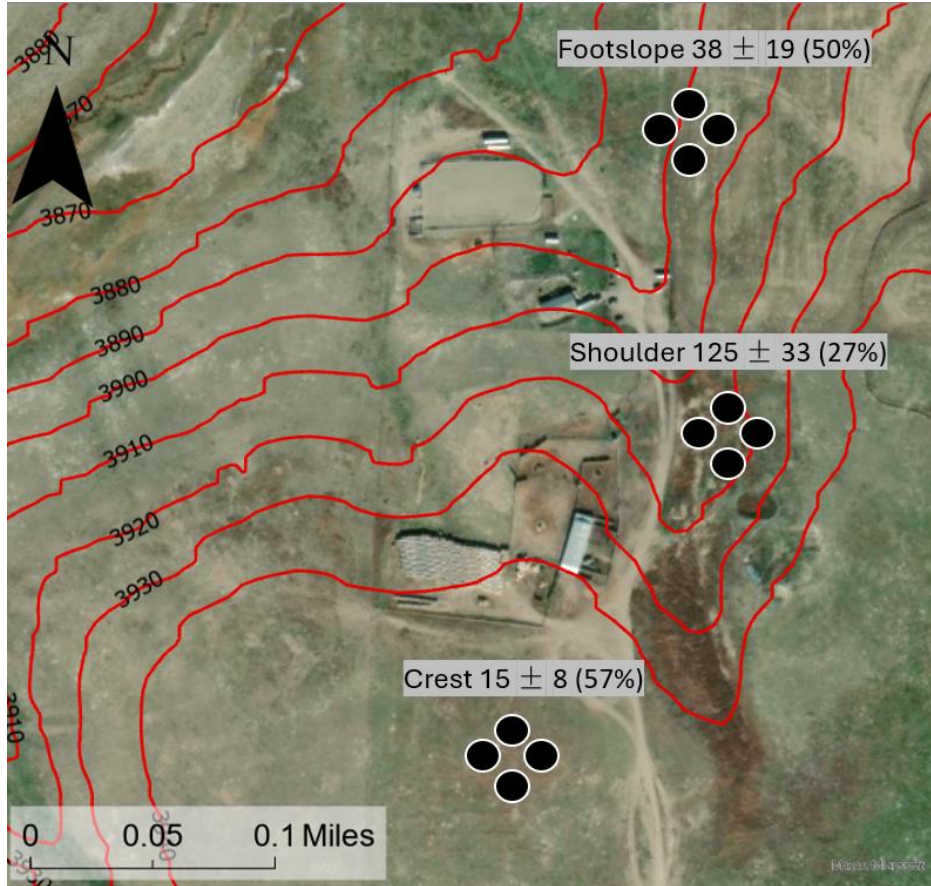


Figure 2.6. Map of high-variability ranch J showing average $\pm$ 1SD (CV%) hillslope SOCv in t ac-ft^{-1} . Black circles represent stylized locations of quadruplicate soil pits. Note the catena runs generally south to north-northwest.

We also summarized hillslope SOCv in 2.47-acre (1 ha) boxes to a depth of one foot (30 cm). This “box” method allows us to extrapolate from our point-based measurements to a more integrated pasture-level perspective. This method revealed which ranches had the most/least SOCv. Six of the 28 ranches we sampled showed relatively high hillslope-integrated SOCv ($>129 \text{ t } 2.47\text{-ac-ft}^{-1}$; $>117 \text{ Mg ha-0.3m}^{-1}$; Figure 2.7). By contrast, six ranches exhibited relatively low SOCv ($<77 \text{ t } 2.47\text{-ac-ft}^{-1}$; $<70 \text{ Mg ha-0.3m}^{-1}$; Figure 2.8).

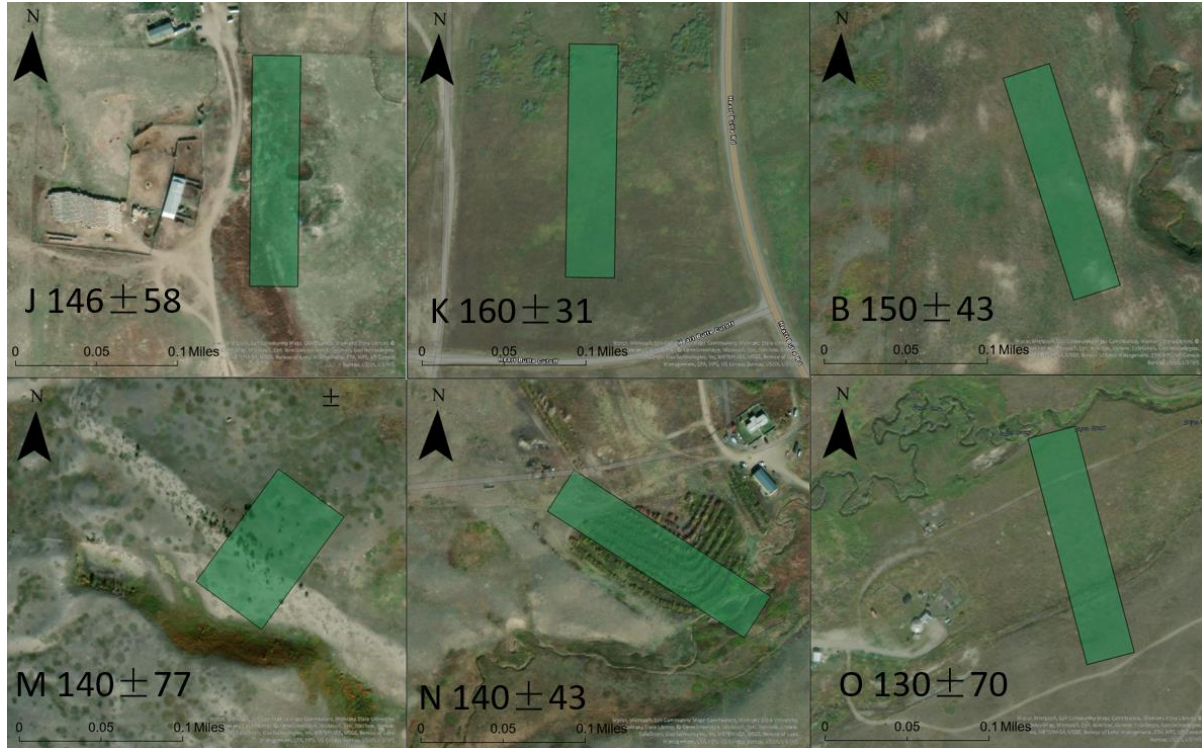


Figure 2.7. Six ranch summary of hillslope-integrated SOCv (average $\pm 1SD$) tons per 2.47-acre [1 ha] box); selected ranches showed the highest average SOCv ($>129 \text{ t } 2.47\text{-ac-ft}^{-1}$; $>117 \text{ Mg ha-0.3m}^{-1}$) across 28 ranches. Alphabetically (B, J, K, M, N, O), ranch SOCv variability at this scale was 29%, 40%, 19%, 55%, 31%, 54%.

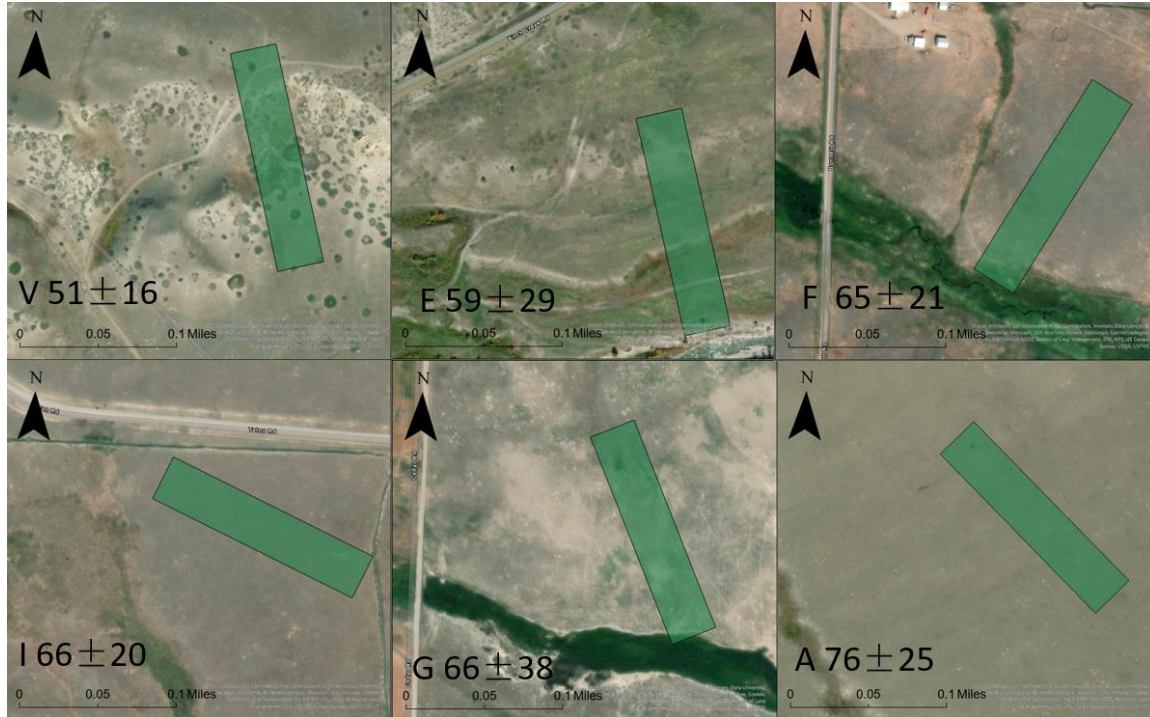


Figure 2.8. Six ranch summary of hillslope-integrated SOCv (average $[\pm 1SD]$ tons within the 2.47 acre [1ha] box); selected ranches showed the lowest average SOCv ($<77 \text{ t } 2.47\text{-ac-ft}^{-1}$; $<70 \text{ Mg ha-0.3m}^{-1}$) across 28 ranches. Alphabetically (A, E, F, G, I, V), CVs were 33%, 49%, 32%, 30%, 58%, 31%.

Higher SOCv ranches (see Figure 2.7) were distributed across the Blackfeet Nation with no clear geographic trend. In contrast, the lower SOCv ranches (see Figure 2.8) were mainly located in the relatively drier, southeastern part of the Blackfeet Nation. The 2.47-acre bounding box provides a standardized area to evaluate SOCv trends instead of looking at only the highest average catena SOCv.

Ranch W had the highest average ranch SOCv with $63 \pm 12 \text{ t ac-ft}^{-1}$ ($153 \text{ Mg ha-0.3m}^{-1}$). Ranch Z had the lowest ranch SOCv with $20 \pm 7 \text{ t ac-ft}^{-1}$ ($49 \text{ Mg ha-0.3m}^{-1}$). To increase Ranch Z's SOCv from 20 to 63 t ac-ft^{-1} , the management practice would need to achieve a 300% increase in SOCv. This means the current SOCv would need to triple. This could take anywhere

from 25-300 years depending on how fast management can change SOCv and assuming a linear growth with no SOCv losses (see figure 1.1).

We explored how sensitive SOCv appeared to be to small CIORPT differences by contrasting patterns at two ranches (A and B; Figure 2.9) located 42 miles (70 km) from one another. Despite having similar temperatures, vegetation biomass estimates, elevations, and soil ages, our analysis revealed that Ranch B's integrated SOCv across hillslope positions (150 t 2.47-ac-ft⁻¹) was significantly higher than ranch A's (76 t 2.47-ac-ft⁻¹; $p = 0.029$, Table 2.1). Ranch B experiences 8 more inches (20 cm) of precipitation a year based on Oregon State's PRISM model. Ranch SOCv CVs were comparable (27%, 31%). The soils on Ranch A are older than Ranch B's soils, yet SOCv is higher on Ranch B.

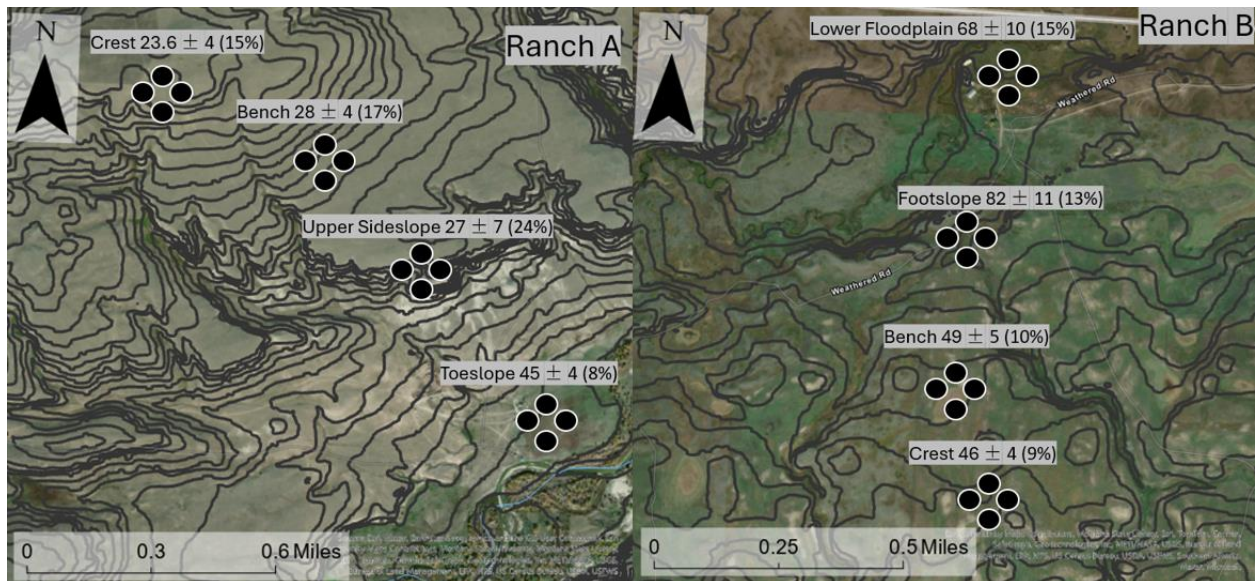


Figure 2.9. Site maps of Ranch A and B with individual hillslope position average SOCv values (t ac-ft⁻¹) $\pm$ 1SD (CV%). Black circles convey a stylized (not to scale) representation of the locations of quadruplicate soil pits with each pit 10 ft (~3m) from a center point.

Table 2.1. Comparisons between Ranch A and Ranch B variables to highlight the potential impacts of ~60% greater precipitation on SOCv. Columns convey ranch averages $\pm$ 1SD. Metric equivalents are bracketed in the same row.

Average Variable	Ranch A	Ranch B	Quotient (B/A)
Annual precipitation (in)	13.8 $\pm$ 0.1 (35.1 $\pm$ 0.3 cm)	21.8 $\pm$ 0.1 (55.4 $\pm$ 0.3 cm)	1.58
Annual air temperature (°F)	42.0 $\pm$ 0.2 (5.6 $\pm$ 0.1°C)	41.0 $\pm$ 0.0 (5.0 $\pm$ 0.1°C)	0.98
SAVI ¹	0.23 $\pm$ 0.07	0.34 $\pm$ 0.12	1.47
Elevation (ft)	3956 $\pm$ 98 (1206 $\pm$ 30 m)	4178 $\pm$ 16 (1274 $\pm$ 5 m)	1.06
Parent material	Glacial deposit, Alluvium	Glacial deposit	
Glacial formation	Pinedale Mountain glacial advance	Canadian ice	
Estimated surface age (y)	~25,000	~18,000	0.72
SOCv (t ac-ft ⁻¹)	31 $\pm$ 10 (69 $\pm$ 21 Mg ha-0.3m ⁻¹)	62 $\pm$ 17 (138 $\pm$ 38 Mg ha-0.3m ⁻¹)	2.00

¹ SAVI stands for soil adjusted vegetation index and is a measurement of greenness using remotely sensed spectral data. These values correspond to the 20th and 48th percentiles of all our hillslope-integrated SAVI estimates.

Though Ranch A's toeslope soils contained the most SOCv (45 t ac-ft⁻¹; 101 Mg ha-0.3m⁻¹), these values did not differ statistically from the other hillslope positions of ranch A (p=0.1; Figure 2.9). Ranch B displayed a stronger catena pattern, meaning lower, depositional hillslope positions contained more SOCv. For example, Ranch B's footslope has statistically significantly higher SOCv (82 t ac-ft⁻¹; 184 Mg ha-0.3m⁻¹) than Ranch B's midslope bench (49 t ac-ft⁻¹; 110 Mg ha-0.3m⁻¹; p=0.001) position. Although the lowest hillslope position, the lower floodplain, contained statistically significantly greater SOCv than the crest (68 vs 42 t ac-ft⁻¹; 152 vs 94 Mg ha-0.3m⁻¹, respectively; p=0.023), it was not statistically significantly greater than that of the bench soil (p=0.053). These hillslope positions are referred to in Figure 2.9.

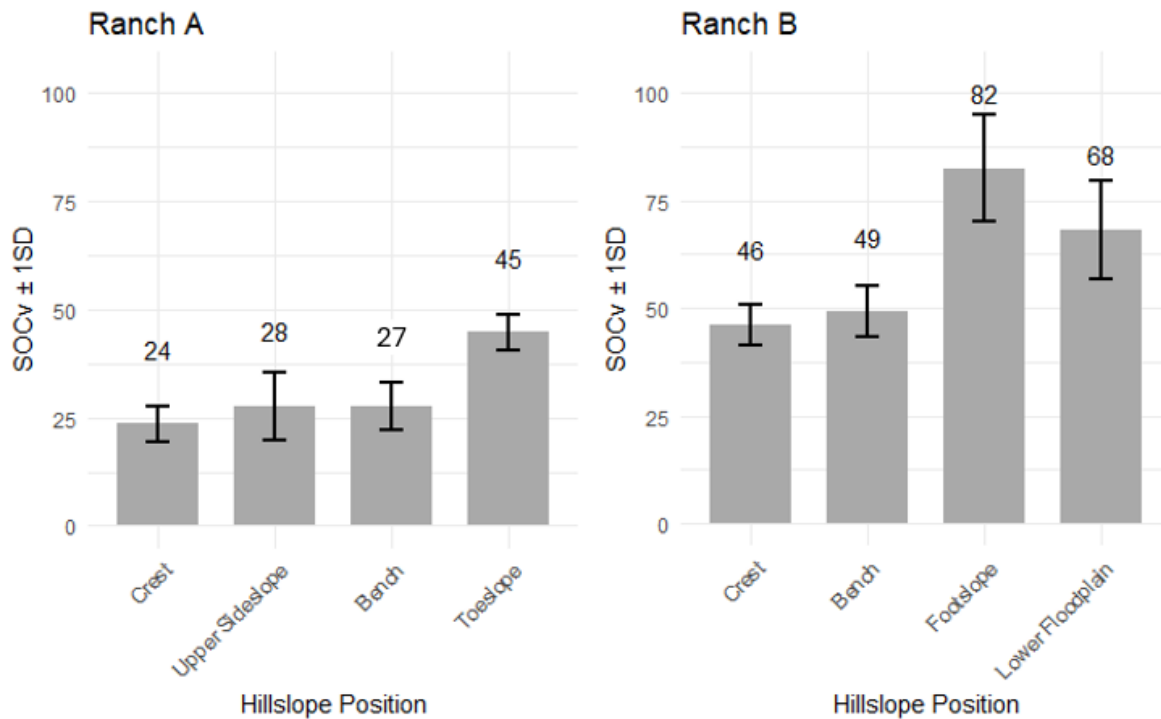


Figure 2.10. Average ($\pm 1SD$) SOCv (t ac-ft⁻¹; multiply by 2.24 to obtain Mg ha-0.3m⁻¹) for Ranch A and B, by hillslope position.

After running an ANOVA analysis on the top A horizons of each hillslope position, Table 2.2 demonstrates that the lowest hillslope position for both ranches had the most SOCm. The toeslope for Ranch A also had a thicker top horizon (7.6 cm) than the bench or the crest (4.8 and 5.7 cm, respectively). Although Table 2.2 indicates that there is more SOCm on the toe slope position at Ranch A, there is no significant difference when the other factors are accounted for (see Figure 2.10). While SOCm is an essential part of SOCv, SOCm comparisons can be a poor choice for assessing soil health differences across space (e.g., different parts of the same ranch or different ranches) or time (e.g., the same ranch location before and after a new grazing practice has been implemented). This is because SOCm does not directly reflect soil bulk density or rock content, potentially leading to incomplete or misleading interpretations of SOC differences. For

example, although Ranch B's footslope soil contained 5% more SOC_m to 30 cm than Ranch B's lower floodplain (11.4% compared to 10.8%), accounting for rock, horizon thickness, and BD differences showed Ranch B's footslope soil contained 14 t ac-ft⁻¹ (20%) more SOC_v than Ranch B's lower floodplain soil (Table 2.2). This comparison highlights two important takeaways: 1. Precipitation differences may drive increased SOC_v when other factors are similar, and 2. SOC_v provides a more comprehensive understanding of soil carbon compared to SOC_m.

Table 2.2. Comparison of the topmost soil horizon average (± 1 SD) SOC_m (%), thickness (cm), bulk density (g cm⁻³), and rock (%) between different hillslope positions. Different letters signify statistically significant differences between hillslope positions within a ranch (Tukey HSD; $p < 0.05$).

Variable/ Interaction	Thickness (cm)	SOC _m (%)	Bulk Density (g cm ⁻³)	Rock (%)
Ranch A				
Crest	5.1 $\pm$ 0.0 (b)	2.9 $\pm$ 0.2 (b)	1.2 $\pm$ 0.2 (b)	47.5 $\pm$ 23.6 (a)
Upper Sideslope	5.7 $\pm$ 1.3 (ab)	2.7 $\pm$ 0.2 (b)	1.5 $\pm$ 0.0 (ab)	30.0 $\pm$ 33.7 (a)
Bench	4.8 $\pm$ 1.6 (a)	3.0 $\pm$ 0.4 (b)	1.4 $\pm$ 0.0 (ab)	37.5 $\pm$ 22.2 (a)
Toeslope	7.6 $\pm$ 0.0 (a)	4.1 $\pm$ 0.6 (a)	1.5 $\pm$ 0.0 (a)	2.0 $\pm$ 2.0 (a)
Ranch B				
Crest	8.3 $\pm$ 3.2 (a)	2.7 $\pm$ 0.3 (b)	1.5 $\pm$ 0.2 (a)	8.8 $\pm$ 2.5 (a)
Bench	7.0 $\pm$ 1.3 (a)	3.3 $\pm$ 0.4 (b)	1.5 $\pm$ 0.2 (a)	5.0 $\pm$ 0.0 (b)
Footslope	11.4 $\pm$ 1.5 (a)	7.0 $\pm$ 0.6 (b)	1.0 $\pm$ 0.2 (b)	0.0 $\pm$ 0.0 (c)
Lower Floodplain	10.8 $\pm$ 3.8 (a)	20.6 $\pm$ 0.4 (a)	0.4 $\pm$ 0.0 (c)	0.0 $\pm$ 0.0 (c)

3.4.3 Number of Soil Samples Needed to Detect Soil Differences

Measuring any improvement in soil health properties represents a classic signal-to-noise issue. These measurements, specifically the numbers of soil samples required, are sensitive to both the magnitude of the “management signal” (e.g., +1% vs. +25%) as well as the magnitude of the “background ranch noise,” which we indexed here with SOC_v CVs, but which depend sensitively on the specific soil property of interest. For example, our literature review of

published CVs for soil properties showed a wide range between 3 and 53% (Table F2), though these within-sample CVs do not necessarily translate to ranch-scale CVs. Our within-hillslope SOCv CVs averaged 21%, whereas ranch-scale SOCv CVs averaged 50%. In other words, the number of samples any rancher will need will depend on 1) the magnitude of the improvement, and 2) the local baseline soil variability. Smaller improvements (e.g., +1%) require more samples than larger improvements (e.g., +100%) because they are relatively harder to detect; also more variable ranches (CVs ~120%) will require more samples than less variable ranches (CVs ~1.2%).

Our analyses showed it might take 170,000 soil samples to detect a +1% increase in SOCv (Table 2.3) for our most variable set of ranches. For more optimistic SOCv improvements (e.g., +10%), detecting such a change in the least variable soils would still require approximately 100 samples per ranch (Table 2.3). In contrast, the most variable soils would necessitate over 1,000 samples to detect the same +10% change, highlighting the substantial impact of baseline soil variability on sampling requirements.

Table 2.3. The recommended number of soil samples required to detect a specific difference in SOCv depends on the underlying variability in SOCv we quantified for each ranch as indexed by our estimated ranch-specific hillslope CVs ([standard deviation] average⁻¹; see Equation 2.4; 1st column). Here, we highlight sample numbers required to detect +1%, +10%, or +25% increases in SOCv (last three columns) for nine ranches (alpha 0.05, power 0.80).

SOCv category	Ranch	Samples to detect __ difference		
		+1%	+10%	+25%
Lower (CV: 10-14%)	C	6,400	66	10
	G	13,000	130	20
	H	11,000	110	18
	Average	10,000	100	16
Intermediate (CV: 16-18%)	K	62,000	620	98
	L	63,000	630	100
	P	64,000	640	100
	Average	63,000	630	100
Higher (CV: 25-45%)	J	210,000	2,200	350
	Q	190,000	1,800	290
	S	120,000	1,200	190
	Average	170,000	1,400	280
Overall average		24,000	240	37

Depending on the local variability at a ranch, the number of samples needed to find a change in SOCv can vary depending on where it was sampled. For example, Table E2 (in Appendix E) presents that the number of samples needed to see a +1% increase between Ranch O's west catena can take three-fold more samples than Ranch O's east catena ($76,000 / 24,000 = \sim 3$). Three of the four paired catenas' average SOCv are within 25% of one another, showing that our study design has captured local variability and soil characteristics well. This is comparable with Table 2.3, which conveys that soils with the highest local heterogeneity can take 33 times more samples to find a 10% difference in SOCv than the lowest variability soils.

3.4.4 Other Rangeland Soil Health Indicators

We also explored alternative rangeland soil health indicators across the Nation, including water aggregate stability (“slaking”) and soil porosity, as these indicators have been related to topsoil erosion and soil moisture, respectively (Bissonnais 1996, Ellies et al. 2000). Two additional soil properties, bulk density and rock, are required to convert SOC_m into SOC_v (Equation 2.2).

We also report correlations to estimate how much variance in SOC_m was explained by each indicator. Slaking was not correlated with SOC_m ($R^2 = 0.07$, $p < 0.001$; Figure 2.11A). As expected, we found a negative relationship between bulk density and SOC_m ($R^2 = 0.46$, $p < 0.001$; Figure 2.11B). Also as expected, we discovered a positive relationship between porosity and SOC_m ($R^2 = 0.40$; $p < 0.001$; Figure 2.11C). Finally, and unsurprisingly, rock content and SOC_m were uncorrelated ($R^2 = 0.03$; $p < 0.001$; Figure 2.11D).

Aggregate stability to 30 cm averaged 0.71 ± 0.12 and ranged from 0.39 to 0.91. Aggregate stability was significantly different between soil structure groups. Subangular blocky structure had significantly higher aggregate stability scores compared to single grained, or granular structures ($p < 0.01$). Porosity to 30 cm averaged $54 \pm 13\%$ and ranged from 21 to 86%. Bulk Density averaged to 30 cm was $1.2 \pm 0.3 \text{ g cm}^{-3}$ and ranged from 0.47 to 2.56 g cm^{-3} . Rock averaged to 30 cm was $21 \pm 20\%$ and ranged from 0-90%.

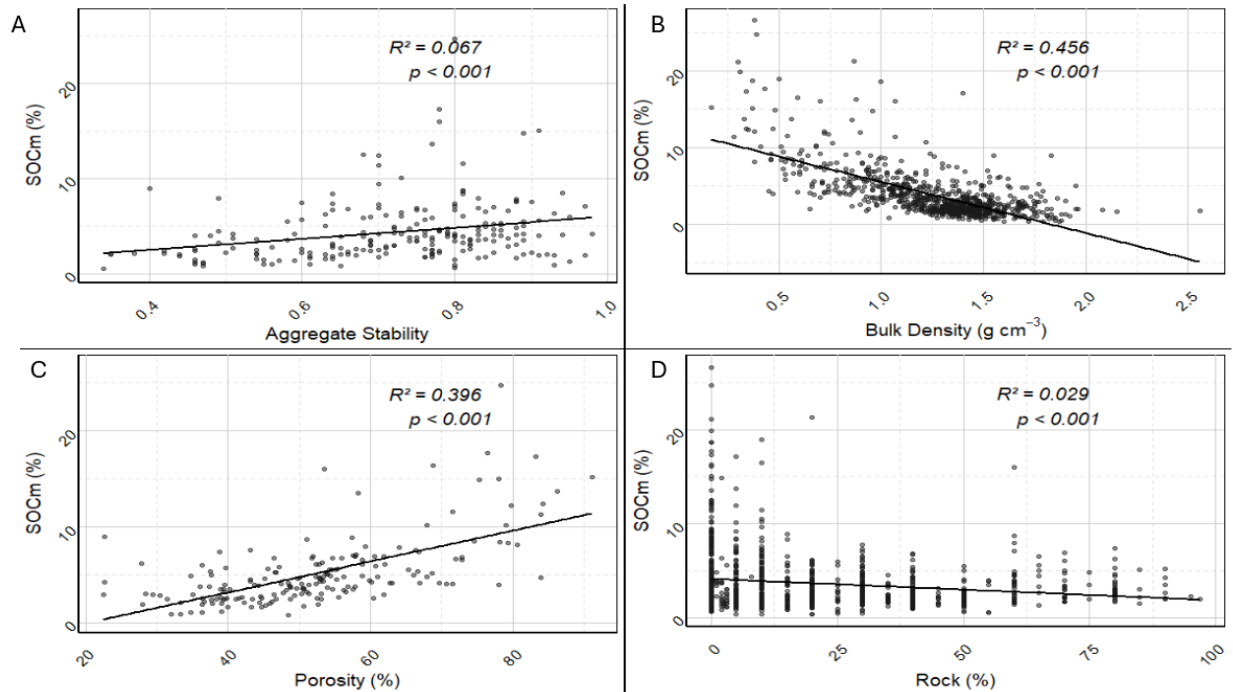


Figure 2.11. How four physical soil properties/indicators, aggregate stability (2.10A, Slaking), bulk density (2.11B, g cm^{-3}), porosity (2.11C, %), and rock (2.11D, %), relate to SOCm across the Blackfeet Nation.

We can re-examine the sample numbers different ranches might require to detect specific management improvements using one of these alternative rangeland soil health indicators. While Ranch J would require 210,000 samples to find a +1% increase in SOCv, “only” 24,000 samples would be required to find a +1% increase in aggregate stability, about a 9-fold difference (see Table 2. 3). Overall, aggregate stability is much less variable at the site-specific level having a 7% average CV compared to the 20% average CV of SOCv. Unfortunately, the low correlation between slaking results and SOCm ($R^2=0.07$) means we cannot reliably substitute slaking results for SOCm or SOCv results.

3.4.5 Multiple Regression Models

The model below was determined to be the best-fitting model after first running the *step()* function and then assessing for and removing influential data points with a second Huber regression model (Equation 2.5; Sun et al. 2020), which assigned a low weight (<0.001) to three overly influential data points. The Huber regression model had an AIC of 693 compared to the original model's AIC of 1017. Typically, lower AIC values are associated with more robust models. The relationship between SOC_v and CLORPT factors can be best represented as:

$$\text{SOC}_v \sim \text{RE} + \text{Glacier} + \text{SAVI} + \text{CI} + \text{PM} + \text{SAVI:CI} + \epsilon$$

(Equation 2.5)

Where Relative Elevation (RE) measures the change in elevation from the lowest to the highest point along a catena, capturing topographic variation. The Glacier variable represents the most recent glacial event and is used to approximate the surface age of soils, providing insights into their developmental history. Soil Adjusted Vegetation Index (SAVI) serves as a proxy for vegetation biomass, reflecting the biological activity and organic inputs to the soil. The Climate Index (CI) is calculated as the average temperature divided by precipitation, offering an estimation of regional climate conditions. Parent Material (PM) refers to the geological source material from which soils form, shaping their physical and chemical characteristics. Additionally, the interaction term SAVI:CI captures the combined influence of vegetation biomass and climate on SOC_v.

Table 2.4. Partial R^2 of each variable and interaction from Equation 2.5. These are the contribution of a specific predictor variable to the overall explanatory power of a regression model.

Variable/ Interaction	Partial R ²
Relative Elevation	0.04
Glacier	0.15
SAVI	0.74
CI	0.20
PM	0.04
SAVI:CI	<0.01
Full Model ¹	0.51

¹Full model's adjusted R²

Table 2.5. P values of variables and indicators from the model given in Equation 2.5. R² of the overall model is included in the last row. Reference variables are sandstone as the parent material, and the Bull Lake glacier.

Variable/ Interaction	P Value
Relative Elevation	<0.001
Early Pinedale Glacier	0.004
Pinedale Glacier	<0.001
Illinoisian Glacier	<0.001
Bull Lake Glacier	<0.001
Unglaciaded	0.083
SAVI	0.365
Climate Index	<0.001
SAVI : CI	<0.001

Table 2.5 Continued

Full Model P Value	<0.001
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Of note, this model still left 49% of the variance in SOC_v unexplained, possibly due to more localized factors like microclimates or soil texture. Also, current and historical management may explain a large part of this variance. Management changes may shift SOC_v differently for soils with different CIORPT factors. It is important to note that SAVI was the variable with the highest partial R² (0.72, see Table 2.4), yet it had the highest p-value (0.365, see

Table 2.5). SAVI has a strong effect on the model but the uncertainty in estimating its effect due to a large signal to noise ratio leads to a high p-value.

3.5 Discussion

3.5.1 Blackfeet Nationwide Trends

The practical implications of these findings are as follows: (1) Locations with more vegetation generally exhibited higher SOC_v, indicating that vegetation explained the most variation in SOC_v stocks; (2) An impractical number of samples would be required to detect realistic changes in SOC_v at a ranch-wide scale, with even low-variability ranches needing approximately 10,000 samples to identify a +1% change in SOC_v; and (3) Local variability can differ significantly between hillslope positions even within a single ranch (e.g., from 14% to 76% in only 500 ft (150 meters at Ranch Q). This suggests that some ranches may store more SOC_v due to their wetter conditions and lower carbon mineralization rates. However, detecting changes across an entire ranch is unrealistic, given the sample size requirements we determined for individual catenas. Alternative indicators should be considered for assessing soil health at a ranch-wide scale.

We found evidence supporting our hypothesis that Blackfeet rangeland soils have high variability, with SOC_v values ranging between 7 and 124 t ac-ft⁻¹ and averaging 42±21 t ac-ft⁻¹ (94±47 Mg ha-0.3m⁻¹), this is somewhat consistent with previously reported characterization efforts for both the Blackfeet Nation and the Northern Great Plains. For example, Haigh et al. (1979) reported 39±18 t ac-ft⁻¹ (87±40 Mg ha-0.3m⁻¹; Figure B1) with 16 profiles across the Blackfeet Nation; Aguilar (1984) reported 33±6 t ac-ft⁻¹ (73±14 Mg ha-0.3m⁻¹) in southcentral

North Dakota; and Karlstrom (2000) reported $25 \pm 11 \text{ t ac-ft}^{-1}$ ($56 \pm 25 \text{ Mg ha-0.3m}^{-1}$) for 17 profiles.

Although climate was only able to explain 36% of the variability within the data (Appendix A.1), there was a pattern between the three ranches in the northwest, and wetter corner of the Blackfeet Nation (Ranches B,C, and V) compared to the three ranches (Ranches E,G, and I) in the southeast and dryer corner of the Blackfeet Nation. Although not significant ($p=0.07$), the northwest corner averaged $50 \pm 12 \text{ t ac-ft}^{-1}$ ($112 \pm 56 \text{ Mg ha-0.3m}^{-1}$) and the southeast corner averaged $26 \pm 2 \text{ t ac-ft}^{-1}$ ($58 \pm 4 \text{ Mg ha-0.3m}^{-1}$). Higher carbon on the wetter corner implies that climate, and precipitation in particular, is an important factor for storing SOCv.

We also found things inconsistent with prior reports, such as Blackfeet Nation's variability. While the mean values are consistent, the CV differs from earlier findings. Our dataset's overall CV of 49% is comparable to Haigh et al. (1979) of 46% and Karlstrom (1981) of 45%, but much higher than Aguilar (1984) 19%, likely because Aguilar (1984) excluded rocks from his data collection, whereas both Haigh et al. (1979) and our study's final SOCv calculations included them. This highlights the importance of methodological consistency when comparing SOCv measurements across studies. Particularly in measuring rocks that are a driver of SOCv storage (Nave et al. 2021).

This discrepancy may stem from differences in sampling intensity, soil age, or climate. For example, Karlstrom (2000) excluded bulk density and rock estimates and sampled older soils, which may help explain how the Blackfeet Nation's younger, drier soils and historical grazing practices have reduced the influence of relief, or the topographical location of the soil, on SOCv.

3.5.2 Ranch Findings

While valuable for examining ranch-wide SOCv, replica catenas have the potential to complicate landscape representation. Mosier et al. (2021) divided their catenas into exactly three zones. However, if one catena has three positions and another has four, hillslope comparisons are complicated and can lead to biased interpretations of landscape-scale SOCv. Additionally, if a catena contains only one measurement point and lacks variability, it oversimplifies the spatial heterogeneity of soil properties, which may lead to biased interpretations of SOCv. Additional challenges emerge when SOCv is higher at the crest of one catena but reverses further downslope. This makes it more challenging to detect changes in SOCv because the increasing noise diminishes the clarity of the signal. Baseline data is critical for detecting changes, but inconsistent patterns along hillslopes make it unclear which values represent "typical" conditions. We can compare catenas to see how well a single catena captures ranch SOCv by looking for statistical differences between paired hillslope positions.

To see how climate might effect SOCv, we analyzed ranches with similar CLORPT factors but different climates. These ranches, A and B, provided valuable insights into how CLORPT factors might influence SOCv (see Table 2.1). At Ranch B, 60% higher precipitation and 50% greater vegetation cover appeared to contribute to significantly greater SOCv levels than at Ranch A (62 vs 31, respectively). These differences suggest that climate and vegetation play a larger role in SOCv storage than age of the soil. We should expect older soils to store more carbon; however, the opposite was true when looking at Ranches A and B. The presence of more carbon in cold and wetter locations aligns with Liptzin et al.'s (2022) findings.

Our data also indicates that one catena may be enough to represent ranch SOCv stocks accurately. Where we had co-located catenas ($n=4$), one instance showed no significant difference between paired hillslope positions. Another showed a significant difference between the crests and toeslopes. The other two ranches with co-located catenas, Ranches M and O, found an overall variability of 17% and 25%, respectively, between the catena with the highest SOCv and the catena with the lowest SOCv (Appendix E). Because these catenas only had single holes done on them, it is impossible to conduct an ANOVA between paired holes; however, the low CV indicates that they were good representations of their ranches.

3.5.3 Site Findings

Erosion processes, such as soil deposition, reducing erosion in lower landscape positions, and enhancing moisture retention, support microbial activity and organic matter decomposition (Rhanor 2011). These factors likely explain why Ranches A and B exhibited thicker top horizons (Oi/A/Ap), contributing to increased SOCv (see Table 2.1). Karlstrom 2000 also found that topsoil horizon thickness increases downslope. Although SOCv may not always be higher on footslopes compared to upper hillslope positions, it is often better protected, suggesting that intense grazing may deplete it less quickly.

Despite these patterns, hillslope position, or relief, showed no significant relationship with SOCv in this study (see Figure 2.3; $p=0.18$). While terrain attributes such as upslope accumulating area or topographic wetness index (Karlstrom 2000, Lindsay 2025) can certainly influence soil carbon variability at a landscape scale, but they do not appear to act as robust predictors in our context.

This finding contrasts with other research from the Northern Great Plains, where SOC_v tends to be lowest on crests and higher downslope, with parent material further shaping these patterns (Aguilar 1984, Zhu et al. 2024). Although the data reflects this trend on average (see Figure 2.3) the high variability within each hillslope position prevents the differences from achieving statistical significance (like ranches A and B). This outcome may result from the region's young and dry soils, which limit natural carbon sequestration processes.

Digging four holes per hillslope position—as opposed to the single-point sampling methods used by Haigh et al. (1979) and Karlstrom (1981)—allowed us to capture local variability and demonstrate the need for multiple samples to detect statistical differences over time and space. This approach aligns with recommendations from Karlstrom 2000, who emphasized the importance of intensive sampling based on catena position and management strategies.

Our relatively weakly weathered soils, paired with over 150 years of heavy grazing with minimal rest for vegetation, may continue to degrade pastures, increase bare ground, and limit SOC_v accumulations (Teague et al. 2011, Blanco-Canqui and Benjamin 2013). These findings highlight the importance of considering historical land use when interpreting landscape-scale variability in SOC_v.

3.5.3 Sample Numbers and Feasibility of Improvements

Variability is a leading reason why rangelands do not have a concrete suite of soil health indicators used in assessments. Our results are consistent with Karlstrom (2000), which found that rangeland carbon and bulk density exhibit high spatial variability, necessitating higher sample sizes than croplands. Furthermore, Bagnall et al. (2023) supports this protocol by

recommending that sampling for changes in soil properties after management interventions should occur as close as possible to previous sample locations, with non-composited, co-located holes.

Many soil health indicators are too variable to submit one soil sample before a grazing management change occurs, and then another a few years later to detect meaningful changes in soil health. Submitting more soil samples increases the cost of testing, which can be a financial challenge for ranchers in Montana. On average, Montana ranchers earn \$41,345 annually—about \$12,000 less than the state’s average salary (Montana Department of Labor & Industry 2024, Economic Research Institute 2024). We found that detecting a 10% increase in SOC_v for medium variability ranches would cost \$95,000 (at \$150 per sample), or over twice the average rancher’s salary (see Table 2.3). This cost rises even further when attempting to detect smaller incremental changes, such as shifts in carbon stocks of less than 1%, which many studies report (Kravchenko and Robertson 2011, Aguilera et al. 2013, Stanley et al. 2023). These challenges highlight the need for two key actions: first, becoming more comfortable with less statistically powerful datasets that require fewer samples, and second, exploring more affordable and innovative methods for assessing soil health improvements without placing an undue financial burden on ranchers.

Our power analysis results are consistent with Stanley et al. 2023 who found that rangeland carbon and BD had high spatial variability, resulting in a need for a higher number of samples than the tested croplands. Her study specifically found that SOC in the top horizon of a soil is more consistent compared to subsurface horizons, and her power analysis showed that compositing randomly selected samples resulted in the lowest power.

Because most rangeland management strategies generally only shift SOCv a small amount (conventional grazing: -10 to +7% per year; rotation grazing: -2 to +15%; see Figure 1.1), many samples will need to be taken to identify any management-related shift; in our case, an average of 240 samples must be taken to find a 10% change at each ranch (see Table 2.3). Stanley et al. (2023) found that it may take 200 samples to see a +20% increase in SOCv after a management change such as the addition of compost. This is higher than our averages but within our range, as the most variable catena needs 280 samples to find a +20% increase (see Table 2.3). A similar study found that it may take 50-180 SOCv samples (Welsch et al. 2019) to detect results +10% greater than the mean. Allen et al. (2010) also examined sampling design and found that paired sites are the best way to find changes in SOCv, of which we have four ranches with co-located catenas. Kravchenko and Robertson 2011 found that almost all of their rangeland studies showed very low power (<20%) and that finding a +50% increase in SOCv would require over 100 samples.

All the aforementioned studies emphasize the importance of conducting power analyses to ensure that the required number of samples is met or to verify the study's statistical power. Using the Blackfeet Nation's average and standard deviation for SOCv, we found that detecting a 10% increase in SOCv with our current sample size would yield a power of only 30%—far below the desired threshold of 80%. This low power is largely due to our limited sampling strategy: we are collecting only four samples per hillslope position rather than the hundreds or thousands typically needed to achieve robust statistical power (0.80). As a result, our ability to detect significant changes in SOCv is substantially reduced.

The literature review has suggested that there is potential to improve soil health, including SOC_v, (see Figure 1.1). The sheer number of samples that would be needed (see Table 2.3) means that any improvement will be hidden unless an immense amount of time and money can be spent assessing baseline and post-management soil conditions. This leads us to explore other soil health indicators to try and find inexpensive and reliable indicators that can be used to assess soil health without taking tens of thousands of samples at each ranch to try and find a +1, +10, or +25% improvement in any given soil health indicator.

3.5.4 Other Indicators

As seen in Figure 2.11B, BD is higher where there is an absence of SOC because minerals are heavier than SOM, implying that parent material may affect the BD of soil. Additionally, the negative relationship between BD and SOC% suggests that compacted soils are less effective at storing carbon, as higher BD inhibits the air and water flow necessary for the decomposition and accumulation of organic matter (see Figure 2.11B; Périé and Ouimet 2008). Aggregate stability can indicate improved soil structure stability, an important characteristic that enhances carbon storage potential (Teague and Kreuter 2020). Our aggregate stability data suggests that soils exhibiting this characteristic may have better structural stability, thus potentially enhancing carbon storage.

The positive correlation between porosity and SOC% also reflects the importance of relief and parent material in shaping soil health. Increased porosity promotes better water retention and microbial activity, vital for organic matter decomposition and carbon sequestration (Fukumasu et al. 2022). This is especially relevant when considering the parent material

component of CLORPT, as certain parent materials may contribute to soil properties that enhance or inhibit porosity (Raj et al. 2022).

The non-existent relationship between rock content and SOC_m contradicts the general assumption that rockier soils have more organic material (Schaetzl and Anderson 2005). This reinforces the idea that the influence of parent material on soil organic carbon can vary depending on the material's specific characteristics.

Although we could not determine that physical soil properties associated with relief and parent material are pivotal in determining soil organic carbon content (see Figure 2.11), BD and porosity emerged as influential soil health indicators in explaining SOC_m, while slaking index scores and rock content appeared to have a negligible impact. These findings highlight the need for a comprehensive understanding of a diverse range of soil properties, as described by the CLORPT model, to effectively manage soil health and optimize carbon storage understanding across diverse landscapes.

Aggregate stability appears to be a less variable soil health indicator compared to SOC_v. So, when a variable soil is identified, perhaps smartphone-enabled slaking index scores could be a useful property to measure instead of SOC_v. Even though SOC_v and slaking tests indicate different soil functions, both are still valuable indicators of soil health. For example, as we saw with our most variable ranch, slaking takes only 11% of the samples that SOC_v would need to find a +1% change. Alternate indicators are useful for ranchers who have soils that are highly variable for SOC_v, but may be much less variable for other indicators. Finally, while we found no relationship between SOC_v and rock content ($R^2=0.03$, $p<0.001$), future research should

explore how variability in rock associated with glacial till might influence SOC levels (Lai et al. 2022).

3.5.5 Regression Models

Understanding how SOCv influences soil functions is essential for improving land management practices in semi-arid regions, as higher SOCv can lead to increased soil health. The state factor model CIORPT provides a helpful way of thinking about this influence. By understanding soil formation factors, ranchers can identify which areas of land have the potential for higher productivity versus those that require careful management to maintain soil health. The state factor model CIORPT provides a helpful way of thinking about soil formation, including organic matter and SOCv. Areas predicted to have high SOCv based on the CIORPT model but exhibiting low SOCv may require targeted management changes, such as adjusting grazing patterns to improve soil health.

Local factors, such as vegetation cover, play a more substantial role in driving SOCv variability than landscape-scale factors like relief or parent material. The soil-adjusted vegetation index (SAVI), representing vegetation abundance and ground cover, expressed the strongest relationship with SOCv (see Table 2.4, Figure A2). Vegetation contributed to SOM through root exudates and plant litter, aligning with the NRCS's soil health management principles of maintaining living roots, reducing bare ground, and promoting biodiversity (NRCS 2021). Our CIORPT model supported this conclusion, as SAVI alone explained 72% of our model's variability, suggesting that vegetation may have been a key driver of SOCv variability in this region ($p < 0.001$). The final CIORPT model used SAVI in an interaction with soil age, suggesting that organisms were a key component of carbon stocks, despite Reinhart et al. (2022) finding no

relationship between biomass and species richness and increased SOCv in rangelands (Equation 2.6).

While four of the five CLORPT factors independently had weak relationships with SOCv, this is not necessarily a limitation (Appendix A). Most management interventions realistically target the organism factor—increasing vegetation cover, forage quality, and improving root systems (Jastrow 1996, Teague et al. 2013). For example, improving pasture biomass by reducing bare ground could enhance SOCv. Installing snow fences can also increase soil moisture, leading to more grass and, from there, potentially more SOCv (Guernsey 1996). Focusing on local factors that ranchers might be able to manage, such as SAVI, provides one opportunity to enhance soil health.

There was also limited evidence found that the sampled catenas represented the entire ranch (Appendix E.3). Additionally, rangeland management strategies and other factors seemed to affect SOCv more than factors supported in the literature, such as upslope accumulating area, climate, vegetation, or snowpack retention potential.

That said, because local variability varied across locations within a specific ranch, even ranches with high overall CVs may still have specific areas where sampling requirements might be more manageable. This is particularly relevant when implementing management practices on a trial basis, as small-scale evaluations require fewer samples and are less resource-intensive. Since most rangeland management changes are initially tested on smaller areas before broader application, focusing on these low-variability sites could provide a practical way to monitor soil health improvements at a fraction of the excessive sampling and monitoring costs we outline. Focusing even more on those low-variability sites with relatively low SOCv values may be one

way to maximize rancher-return-on-researcher-investment (RRORI), a soil health spin on the traditional return-on-investment framing.

3.6 Conclusions

Soil is a complex system shaped by numerous environmental and anthropogenic factors that vary widely across landscapes. Despite decades of inquiry reaching back to a National Academy of Sciences-commissioned study (National Research Council 1994), rangeland soil health monitoring remains a 'Wild West' endeavor characterized by fragmented datasets, inconsistent methodologies, and a lack of standardized indicators. Although such indicators should be responsive to management and show low intra-sample and spatial variability, enabling low-cost and straightforward interpretations (Liptzin et al. 2022), no agreed-upon set of indicators has emerged. Our baseline results reinforce this complexity, reflecting that in addition to CIORPT, physical soil properties such as bulk density (BD), porosity, and rock content play pivotal roles in determining SOC content. However, the weak correlations observed for all indicators tested underscore the challenges of identifying universally applicable indicators. While advances in remote sensing, such as machine learning and improved vegetation indices, offer opportunities for expensive, large-scale soil health assessments, these tools remain only as robust as the ground-truthed data upon which they are based. Because soils are highly variable, ground truthing remotely sensed-based models is extremely difficult, highlighting the need for on-the-ground approaches to assess soil health.

To bridge the gap between science and application, we propose prioritizing a suite of indicators that are not only scientifically rigorous but also practical and actionable for ranchers.

These indicators need to be relevant to everyone, from ranchers to private investors, who often have different goals for rangeland operations, with private investors more concerned with carbon storage than ranchers' interests in a sustainable increase in cattle carrying capacity. Additionally, quality soil health indicators should be useful for measurement, monitoring, reporting, and verification. This means test results should be replicable without spending potentially thousands of dollars per ranch, depending on the local soil heterogeneity. While progress has been made in identifying soil health indicators that represent soil condition, further investigation should be done to identify indicators that can be affordably ground-truthed while being extrapolated accurately using models to set benchmarks.

While our findings suggest that using SOC_v as a soil health indicator may be impractical in rangeland contexts like ours due to the prohibitively large number of samples required to detect meaningful changes, our research also points to alternative indicators that may be more responsive to improvements in management practices, even when SOC_v is not. Specifically, topsoil erosion rates (Olson et al. 1994, Fierer and Gabet 2002), carbon transit times (Anderson et al. 1947, Libby et al. 1949, Duborgel et al. 2023), mineral-associated organic matter (Kleber et al. 2015, Cotrufo et al. 2019), and soil moisture levels (Heckman et al. 2023, Lomax et al. 2024) warrant further investigation. These indicators hold promise for providing more sensitive and actionable insights into soil health dynamics under improved management practices and should be prioritized in future studies within this project.

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CHAPTER FOUR: CONCLUSION

Soil health is foundational to the resilience and productivity of rangelands, yet its heterogeneity makes it difficult to even baseline soil health. The literature provides evidence for rotational grazing even if studies tended to be in wetter locations, and sampling may have been underpowered. Additionally, our research showed that by characterizing the local variability of numerous ranches, locations can be identified as sites that can be measured for an increase in soil health properties such as SOCv. Ranchers should implement rotational grazing for reasons other than increasing soil carbon. Because of the variability, it is tough to find increases in SOCv, but that does not mean that 1) There was no increase in SOCv, or 2) Other soil health indicators, or even rangeland health, can be increased. So even though we recommended implementing better rangeland management practices on high variability sites, it is not recommended to try and evaluate these sites for increases in SOCv because without taking hundreds of thousands of samples, the results will be highly likely to fall victim to statical errors. Instead, ranchers should use their best judgment to look for increases in soil health, such as darker soil colors and vegetation differences, and use low-cost methods such as remote sensing or even slaking as estimates of improved soil and pasture health.

State factors were an interesting test of natural effects on SOCv. However, it seems that they cannot predict variability in soils, perhaps because rock content or management has a larger impact on SOCv heterogeneity. By focusing on low-variability locations and prioritizing indicators that balance affordability with sensitivity, ranchers can implement regenerative grazing practices with the best chance of yielding conclusive results.

Future research should continue exploring indicators that better explain management's effect on soil health, such as soil moisture levels, topsoil erosion rates, carbon transit times, and MAOM/POM dynamics. Soil moisture should help to explain both forage dynamics better as well as general soil health because increased soil holding capacity means there is an increase in water storage due to structure and increased SOC_v, and there is more water available to plants, increasing their biomass and growth rates late into the summer (Yang et al. 2024). Topsoil erosion is an exciting indicator that should be tested on the Blackfeet nation because it signals loss of structure and nutrients and because the topsoil is the most fertile layer of soil (Lal 1998, Thaler et al. 2022). Longer carbon transit times within a soil profile would indicate more protected carbon during drought or disturbances, allowing vegetation to grow during those times and helping ranchers through climate change (Sierra et al. 2018). Short transit times suggest a more rapid decomposition of labile organic matter (Sierra et al. 2018). MAOM and POM could be a more cost-effective option for carbon transit times as these analyses are generally done in university laboratories compared to a ¹³C analysis of \$450 (Poeplau et al. 2018, University of Georgia 2024). In summary, while the literature review reveals mixed results regarding the impact of rotational grazing on SOC_v, and our research highlights the challenges of detecting SOC_v changes due to management practices, it may be prudent for ranchers to adopt smarter grazing practices for benefits beyond carbon sequestration. These practices could instead focus on enhancing water storage capacity, boosting forage biomass, or achieving more easily measurable improvements in soil and rangeland health.

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APPENDICES

APPENDIX A

CHAPTER 2 SUPPLEMENTAL DATA

A.1. CIORPT Factors

A.1.1 Climate

Due to a lack of local weather data on the Blackfeet Nation, we utilized Montana's Relative Effective Annual Precipitation (REAP) data to estimate the role of effective precipitation as a climate factor. Effective precipitation is the portion of total precipitation contributing to vegetation growth after accounting for factors like evapotranspiration and soil water storage (Ali and Mubarak 2017). The REAP data set did not correlate with SOCv ($R^2=0.014$, $p=0.204$), necessitating exploring alternative methods. Precipitation and temperature are vital drivers in soil development, influencing moisture levels and nutrient availability, critical for vegetation sustainability (Jenny 1941, Haddix et al. 2020, McKenna et al. 2022). So another climate database, Oregon State's PRISM (Parameter-elevation Relationships on Independent Slopes Model), was used to obtain yearly precipitation and temperature data. Individually, temperature had a very weak but significant relationship with SOCv ($R^2=0.04$, $p=0.029$), but precipitation did not ($R^2=0.02$, $p=0.14$). Because temperature and precipitation had a strong negative linear relationship with one another ($R^2=0.82$, $p<0.001$), temperature was divided by precipitation to create a climate index (CI), which had also had no relationship to SOCv ($R^2=0.02$, $p=0.136$). Temperature and precipitation were combined instead of being included individually in the model because of their high multicollinearity (Generalized Variance Inflation factor >6). The CI also performed better in all predictive models.

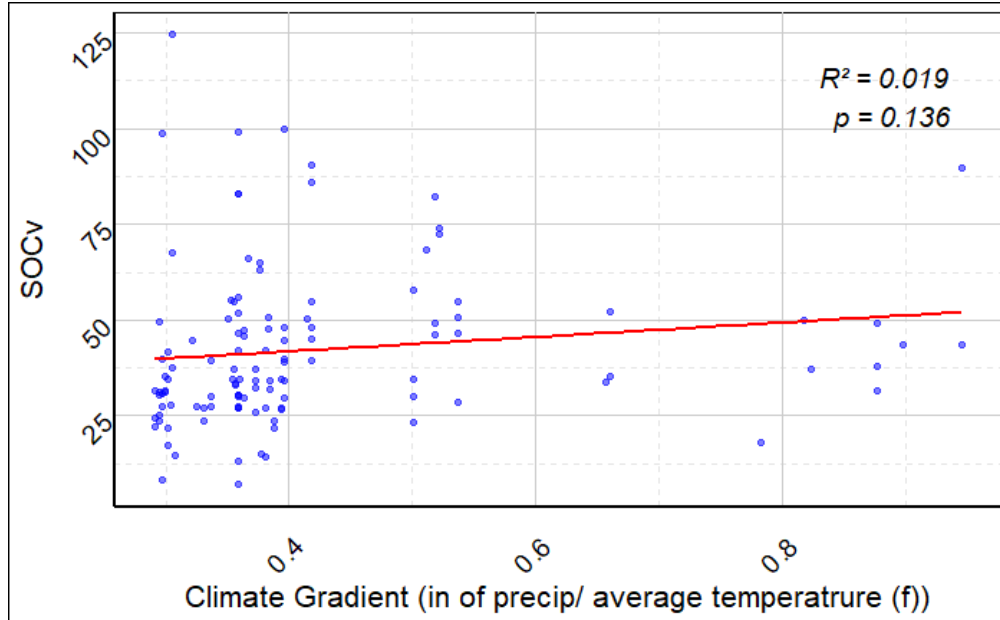


Figure A1. SOCv (t ac-ft⁻¹) shows a non-statistically relevant relationship to the climate index (inches of precipitation divided by temperature).

A.1.2 Organisms

Quantifying the impact of “organisms” is much more complex than climate and is scale-dependent. So, once again, we took a step-by-step approach to finding the best representative. SOCv increases with a vegetation index representing a higher forage quantity per pasture (Figure A2). Biomass is the weight of aboveground vegetation per unit area in a pasture. It was calculated through the Range Analysis Platform and had a very weak relationship with SOCv ($R^2=0.11$, $p<0.001$; Figure A3).

The SAVI and SOCv had a moderate relationship ($p<0.001$, $R^2 =0.31$; Figure 4B). We conducted a linear regression between biomass and the SAVI ($p<0.001$, $R^2 =0.37$). This is expected because the Range Analysis Platform uses a vegetation index to calculate an estimated biomass (Figure A3). Because Figure A2 has a stronger relationship with SOCv, we included it in the final multilinear regression model.

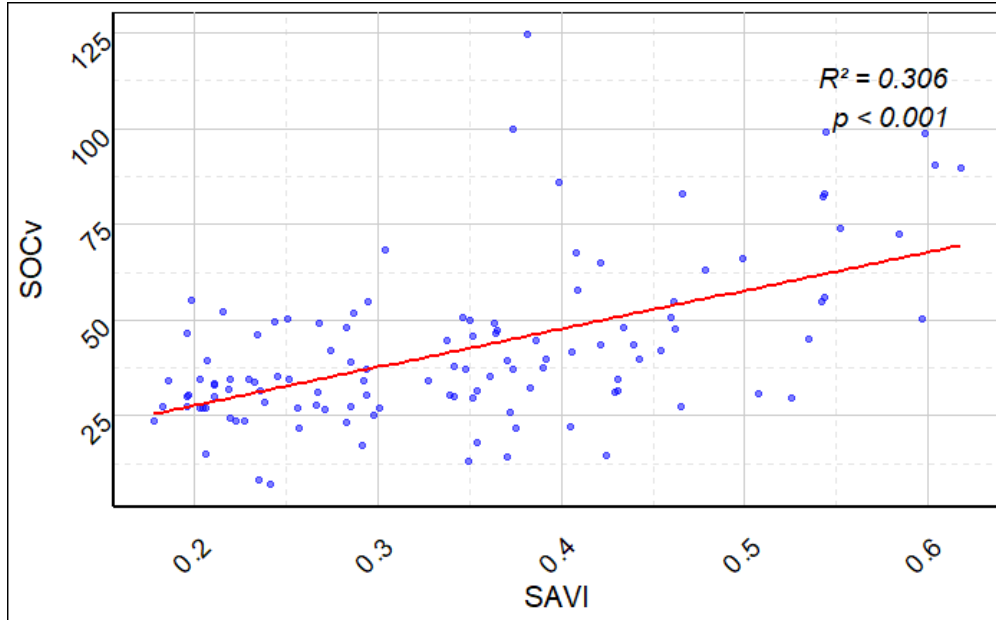


Figure A2. The relationship between SAVI and SOCv (t ac-ft⁻¹).

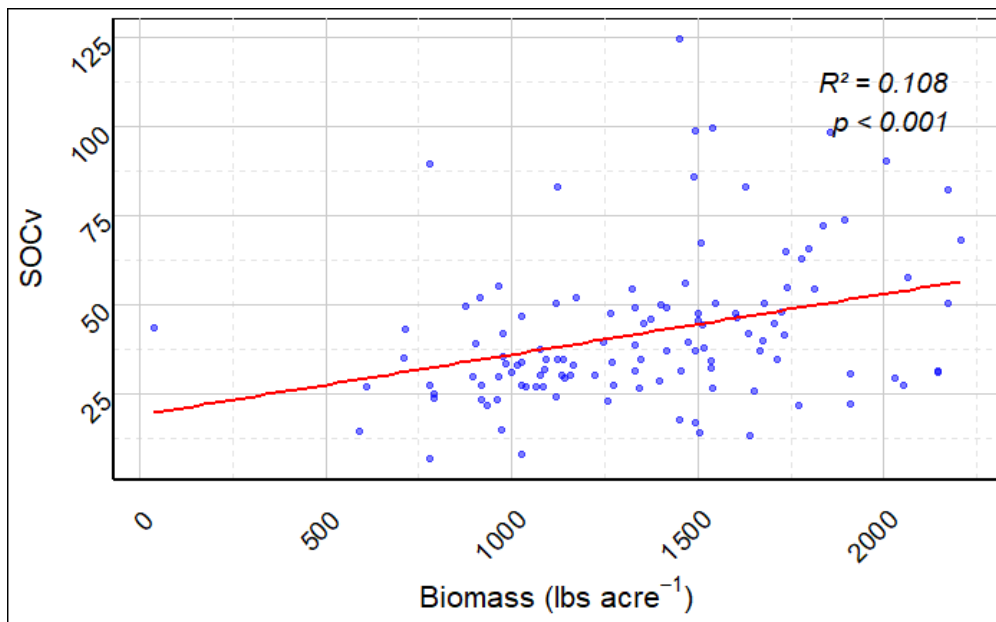


Figure A3. The relationship between Biomass and SOCv (t ac-ft⁻¹).

A.1.3 Relief

The first variable tested in a simple linear model to explain SOCv was hillslope position. SOCv did not differ statistically among the hillslope positions ($p=0.07$). Ranging from crests that averaged ($\pm 1SD$) 42 ± 22 t ac-ft⁻¹ (CV=52%) to footslopes averaging 46 ± 26 t ac-ft⁻¹ (CV=57%), the next variables tested were slope and aspect, both of which had a non-significant relationship with SOCv (Table A1). Elevation was derived from a digital elevation model in ArcGIS and was also non-significant by itself ($p=0.10$). Elevation was used to calculate relative elevation from the lowest sample's hillslope position (Figure A4). Figure A4 indicates that the lower a position is on a catena, the more SOCv. Because relative elevation had the highest R^2 and the lowest P value and AIC, it was used in the final multilinear model to represent Relief (Table A1).

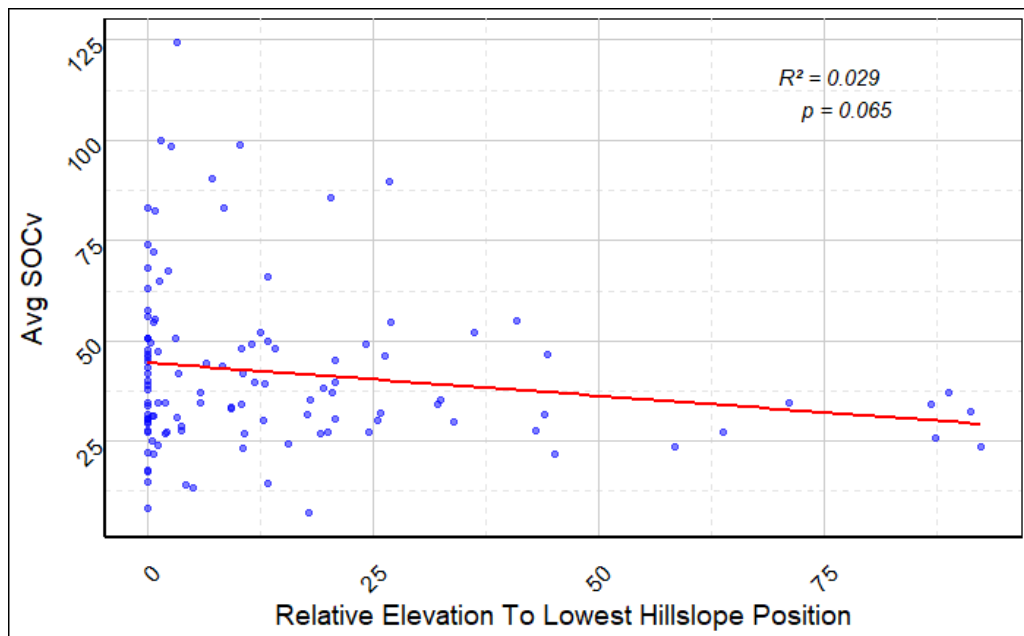


Figure A4. SOCv's (t ac-ft⁻¹) relationship to relative elevation (ft). Values that = 0 are the bottommost hillslope positions.

Table A1. Shows P-values R^2 s, and AICs for variables indicating relief. Where there is an NA in the R^2 column, that variable is categorical, and an ANOVA was run instead of a simple linear regression.

Variable	P Value	R^2	AIC
Hillslope Position	0.079	NA	1052.482
Aspect	0.149	NA	1056.594
Slope	0.434	0.01	1055.263
Elevation	0.099	0.02	1053.115
Relative Elevation	0.065	0.03	1052.406

A.1.4 Parent Material

Figure A5 compares SOC_v across three different parent material types: sandstone, alluvium, and glacial deposit. Parent material is a soil-forming factor in CIORPT (Jenny 1941). These different parent materials can weather into different soil profiles, affecting chemical and physical soil properties (Wilson 2019).

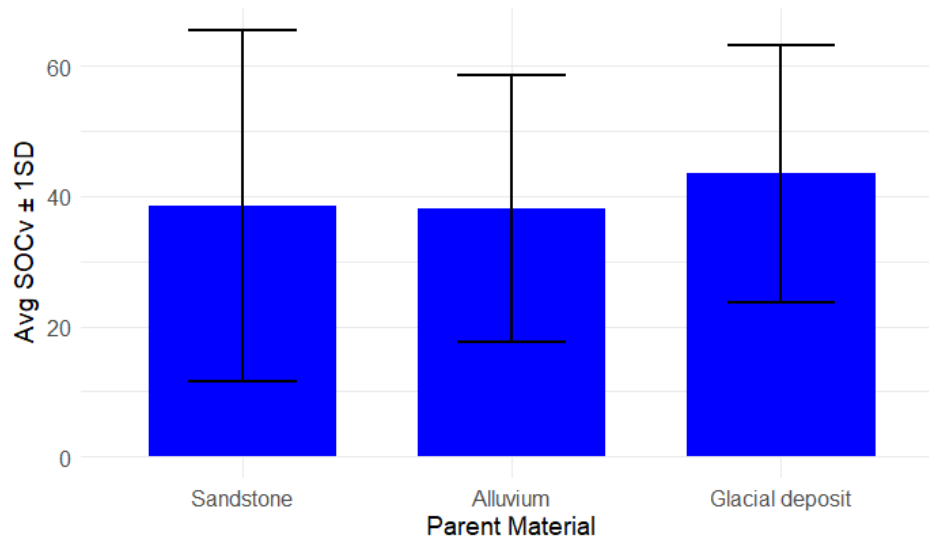


Figure A5. The average SOCv (t ac-ft⁻¹) was determined by different parent materials, and the ANOVA test highlighted no differences between groups ($p=0.502$).

A.1.5 Time

Figure A6 compares the SOCv across six glacial events corresponding to different periods. This is to represent the “time” aspect of ClORPT. We did not radiocarbon date any of our soils, but we could estimate their age because we know when they last glaciated (Karlstrom 1999, Fullerton et al. 2004). Although no group is significantly different from another, the distribution of SOCv for the unglaciated soils, estimated to be around 1.8 million years old, is much less variable, with a CV of 20%. Glaciated soils have a higher CV, with Canadian ice at 28%, Early Pinedale at 34%, Pinedale at 46%, Illinoian at 65%, and Bull Lake at 45%. This implies that soils may become more variable as they age but become more homogenous after 128,000 years.

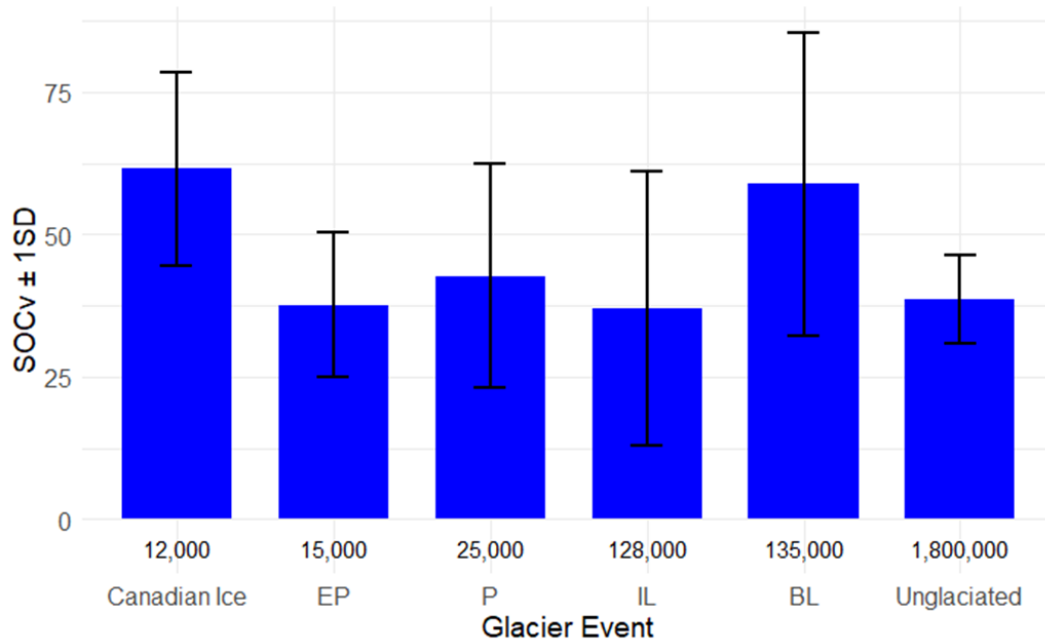


Figure A6. SOCv (t ac-ft⁻¹) by different glacial periods ($p=0.188$). Canadian ice is also referred to as a Laurentide sheet. EP stands for early Pinedale, and P stands for Pinedale. IL refers to an Illinoian, Bl refers to Bull Lake, a valley glacier, and unglaciaded means that the point was not glaciated for the last 1.8 million years. The number above the ice sheet's name is how long the ice sheet receded or how old the soil is.

Prior Research

A.2 Haigh et al. 1979 Data

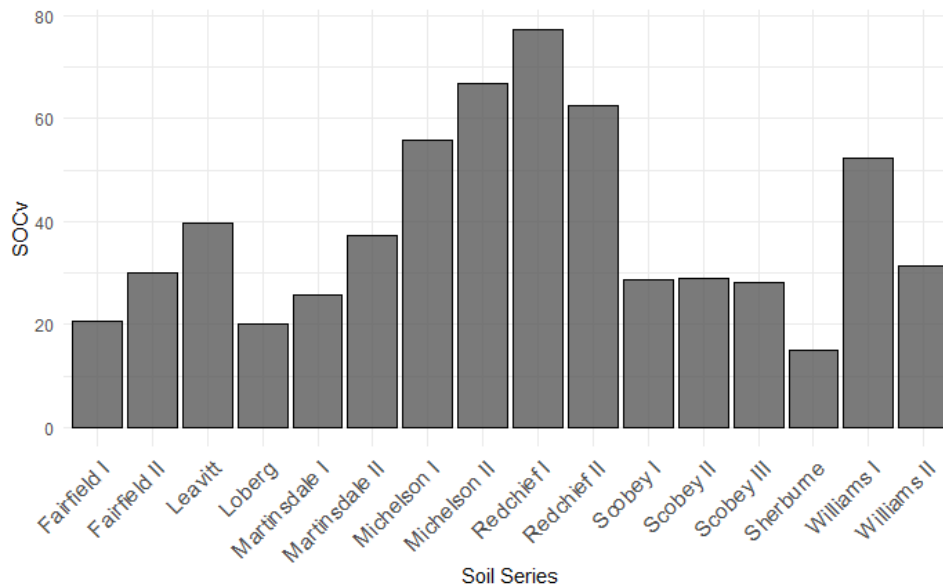


Figure A7. SOCv (t ac-ft⁻¹) of different soil pits from the Blackfeet Nation, average ± 1 SD 39 $\pm$ 18 (Haigh et al. 1979).

A.2 Karlstrom et al. 2000 Data

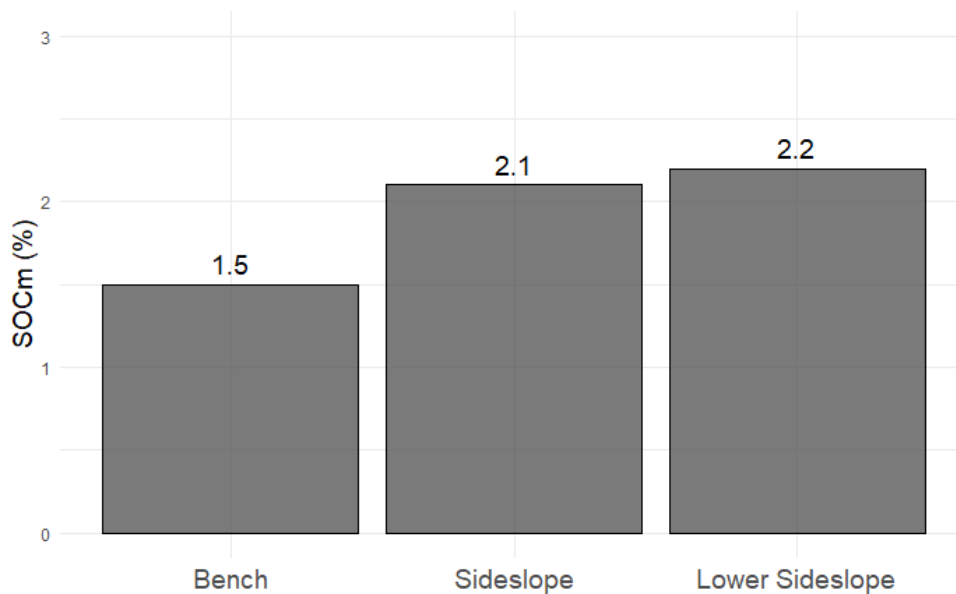


Figure A8. Depth-averaged (1-foot) SOCm% by catenal hillslope position (Karlstrom et al. 2000).

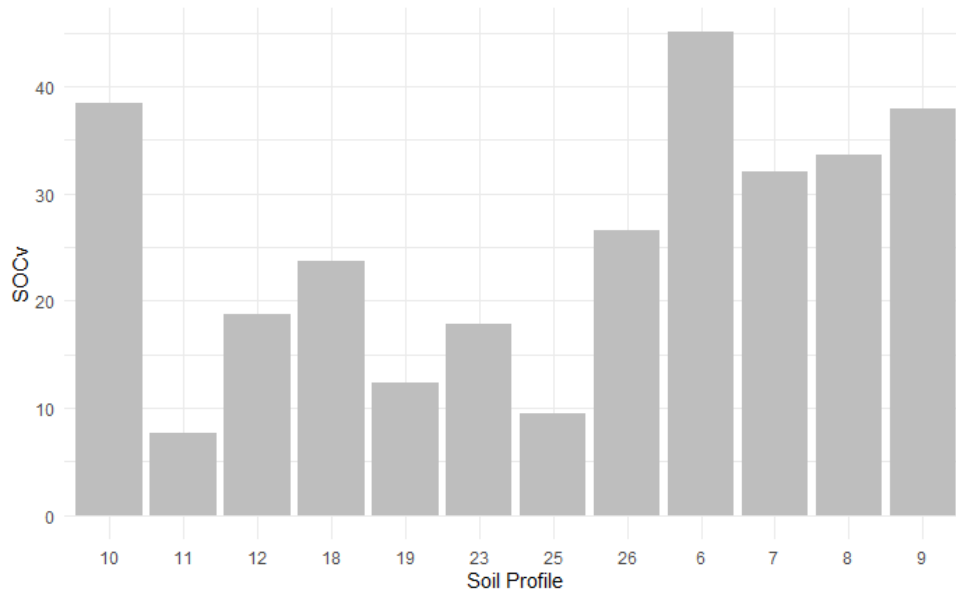
A.2 Karlstrom 1981 Data

Figure A9. SOCv (t ac-ft⁻¹) of different soil pits from the Blackfeet Nation, average ± 1 SD 25 $\pm$ 11 (Karlstrom 1981).

APPENDIX B

META-ANALYSIS REFERENCES

Table B1. Studies used in metanalysis to create Figure 1.1.

Title	Author(s)	Location	Year Published
Biodiversity and ecological long-term plots in Southern Patagonia to support sustainable land management: The case of PEBANPA network	Peri P.L., Lencinas M.V., Bousson J., Lasagno R., Soler R., Bahamonde H., Martínez Pastur G.	Patagonia	2016
Potential of grassland rehabilitation through high density-short duration grazing to sequester atmospheric carbon	Chaplot V., Dlamini P., Chivenge P.,	South Africa	2016
Changes in Soil Physical, Chemical, and Biological Characteristics of a Temperate Desert Steppe under Different Grazing Regimes in Northern China	Jiao T., Nei Z., Zhao G., Cao W.	North China Desert	2016
Grazing intensity and micro-topographical effects on some nitrogen and carbon pools and fluxes in sheep-grazed hill country in New Zealand	Hoogendoorn C.J., Newton P.C.D., Devantier B.P., Rolle B.A., Theobald P.W., Loyd-West C.M.,	New Zealand	2016
Do shrubs reduce the adverse effects of grazing on soil properties?	Eldridge D.J., Beecham G., Grace J.B.,	Eastern Australia	2015
Grazing intensity levels influence C reservoirs of wet and mesic meadows along a precipitation gradient in Northern Patagonia	Enriquez A.S., Chimner R.A., Cremona M.V., Diehl P., Bonvissuto G.L.	Northern Patagonia	2015
Managing Semi-Arid Rangelands for Carbon Storage: Grazing and Woody Encroachment Effects on Soil Carbon and Nitrogen	Yusuf H.M., Treydte A.C., Sauerborn J.	Southern Ethiopia	2015

Table B1. Continued

Rangeland management effects on soil properties in the savanna biome, South Africa: A case study along grazing gradients in communal and commercial farms	Sandhage-Hofmann A., Kotzé E., Van Delden L., Dominiak M., Fouché H.J., van der Westhuizen H.C., Oomen R.J., Du Preez C.C., Amelung W.	South Africa	2015
Improved grazing management may increase soil carbon sequestration in temperate steppe	Chen W., Huang D., Liu N., Zhang Y., Badgery W.B., Wang X., Shen Y.	Temperate steppe of China	2015
Managing cattle grazing intensity: effects on soil organic matter and soil nitrogen	Segoli M., Bray S., Allen D., Dalal R., Watson I., Ash A., O'Reagain P.	Northern Australia	2015
Effects of Grazing Intensity on Soil and Vegetation Properties in a Mediterranean Rangeland	Akhzari D., Pessarakli M., Eftekhary Ahandani S.,	Mediterranean	2015
Response of Vegetation and Soil Carbon and Nitrogen Storage to Grazing Intensity in Semi-Arid Grasslands in the Agro-Pastoral Zone of Northern China	Xu M.Y., Xie F., Wang K.,	Northern China	2014
The response of grassland productivity, soil carbon content and soil respiration rates to different grazing regimes in a desert steppe in northern China	Hou X., Wang Z., Schellenberg M.P., Ji L., Yun X.,	Desert steppe in northern China	2014
Carbon storage under different grazing management in the typical steppe	Chen H., Hou X., Ubugunov O., Vishnyakova O., Wu X., Ren W., Ding Y.	Steppe of China	2014
Impact of carbon farming practices on soil carbon in northern New South Wales	Cowie A. L., Londergan V.E., Rabbi S.M.F., Forasier F., Macdonald C., Harden, S., Kawasaki A., Singh B.K.	New South Wales	2014

Table B1. Continued

What determines soil organic carbon stocks in the grazing lands of north-eastern Australia?	Allen D.E., Pringle M.J., Bray S. Hall T.J., O'Reagain P.O., Phelps D., Cobon D.H., Dloesch P.M., Dalal R.C.,	North-eastern Australia	2013
Rangeland management impacts on the properties of clayey soils along grazing gradients in the semi-arid grassland biome of South Africa	Kotzé E., Sahndage-Hofmann A., Meinel J.A., Du Preez C., Amelung W.	South Africa	2013
Livestock grazing in intermountain depressional wetlands—Effects on plant strategies, soil characteristics and biomass	Teuber L.M., Hölzel N., Fraser L.H.,	British Columbia	2013
Response of soil properties to yak grazing intensity in a Kobresia parva-meadow on the Qinghai-Tibetan Plateau, China	Dong Q.M., Ahao X.Q., Wu G.L., Shi J.J., Wang Y.L., Sheng L.	Qinghai-Tibetan Plateau, China	2012
Impact of Grazing on Soil Carbon and Microbial Biomass in Typical Steppe and Desert Steppe of Inner Mongolia	Liu N., Zhang Y., Chang S., Kan H., Lin L.	Mongolia	2012
Effect of grazing on carbon stocks and assimilate partitioning in a Tibetan montane pasture revealed by $^{13}\text{CO}_2$ pulse labeling	Hafner S., Unteregelsbacher S., Seeber E., Lena B., Xu X., Li X., Guggenberger G., Miede G., Kuzyakov Y.	Tibetan montane pasture	2012
Warming and grazing affect soil labile carbon and nitrogen	Rui Y., Wang S., Xu Z., Wang Y., Chen C., Zhou X., Kang X., Lu S., Hu Y., Lin Q., Luo C.	Tibet	2011
Impact of different grazing systems for bovine cattle on the soil microbiological and chemical characteristics	Garcia M.R.L., Sampaio A.A., Nahas E.,	Brazil	2011
Grazing depresses soil carbon storage through changing plant biomass and composition in a Tibetan alpine meadow	Sun D.S., Wesche K., Chen D.D., Zhang S.H., Wu G.L., Du G.Z., Comerford N.B.,	Tibet	2011

Table B1. Continued

Grazing management impacts on vegetation, soil biota and soil chemical, physical and hydrological properties in tall grass prairie	Teague W.R., Dowhower S.L., Baker S.A., Haile N., DeLaune P.B., Conover D.M.,	Texas	2011
Assessing the relationship between fire and grazing on soil characteristics and mite communities in a semi-arid savanna of northern Australia	Beyer S., Kinner A., Hutley L.B., McGuinness K., Gibb, K.	Northern Austrillia	2011
Impacts of Livestock Grazing on Selected Soil Chemical Properties in Intensively Managed Pastures of Peninsular Malaysia	Ajorlo M., Abdullah R., Hanif A.H., Halim R., Yusoff M.,	Malaysia	2011
Changes in carbon and nitrogen in soil particle-size fractions along a grassland restoration chronosequence in northern China	He N., Wu L., Wang Y., Han X.	Northern China	2011
Soil fauna in sheep-grazed hill pastures under organic and conventional livestock management and in an adjacent ungrazed pasture	Schon N.L., Mackay A.D., Minor M.A.,	New Zealand	2011
Effects of livestock exclusion on vegetation and soil properties under two topographic habitats in an alpine meadow on the eastern Qinghai-Tibetan plateau	Huai Chen S., Wu Y., Wu N.	Tibet	2010
Changes in soil characteristics and grass layer condition in relation to land management systems in the semi-arid savannas of Swaziland	Tefera S., Dlamini B.J., Dlamini A.M.	Eswatini (Swaziland)	2010
Soil and herbaceous plant responses to summer patch burns under continuous and rotational grazing	Teague W.R., Dowhower S.L., Baker S.A., Ansley R.J., Kreuter U.P., Conover D.M., Waggoner J.A.	Texas	2010
Soil carbon stocks under different pastures and pasture management in the higher rainfall areas of south-eastern Australia	Chan K.Y., Oates A., Li G.D., Conyers M.K., Prangnell R.J., Poile G., Liu D.L., Barchia I.M.,	Eastern Australia	2010

Table B1. Continued

Field Trial Assessment of Biological, Chemical, and Physical Responses of Soil to Tillage Intensity, Fertilization, and Grazing	Vargas Gil S., Oddini C., Zuza M., Marinelli A., March G.	Argentina	2009
Comparing the effects of continuous and time-controlled grazing systems on soil characteristics in Southeast Queensland	Sanjari G., Ghadiri H., Ciesiolka C.A.A., Yu B.	Australia	2008
Effect of grazing intensity on carbon and nitrogen in soil and vegetation in a meadow steppe in Inner Mongolia	Han G., Hao X., Zhao M., Wang M., Ellert B.H., Willms W.	Mongolia	2008
Effects of grazing intensity and prescribed fire on soil physical and hydrological properties and pasture yield in the savanna woodlands of Burkina Faso	Savadoogo P., Savadoogo L., Tiveau D.	West Africa	2007
Influence of Goat Densities on Vegetation, Hydrology and Soil Properties in a Chihuahuan Desert Range	Mellado M., Duenez J., Foote R.H., Garcia J.E., Rodriguez A.	Northern Mexico	2005
Influences of continuous grazing and livestock exclusion on soil properties in a degraded sandy grassland, Inner Mongolia, northern China	Yong-Zhong S., Yu-Lin L., Jian-Yuan C., Wen-Zhi, Z.	Inner Mongolia, northern China	2005
The influence of woody plants and livestock grazing on grass species composition, yield and soil nutrients in the Middle Awash Valley of Ethiopia	Abule E., Smit G.N., Snyman H.A.	Ethiopia	2005
Response of Organic and Inorganic Carbon and Nitrogen to Long-Term Grazing of the Shortgrass Steppe	Reeder J.D., Schuman G.E., Morgan J.A., LeCain D.R.,	Colorado	2004
Non-selective grazing impacts on the soil-properties of the Nama Koo	Beukes P.C., Cowling R.M.	South Africa	2003
Effects of Grazing on Restoration of Southern Mixed Prairie Soils	Fuhlendorf S.D., Zhang H., Tunnel T.R., Engle D.M., Cross A.F.	Oklahoma	2002
Influence of pasture management on soil biological quality	Banerjee M.R., Burton D.L., McCaughey W.P., Grant C.A.	Canada	2000

Table B1. Continued

Impact of grazing management on the carbon and nitrogen balance of a mixed-grass rangeland	Schuman G.E., Reeder J.D., Manley J.T., hart R.H., Manley W.A.	Wyoming	1999
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Table B2. Studies used for cost analysis in Chapter Three.

Title	Author(s)	Year Published
Grazing effects on soil chemical and physical properties in a semiarid steppe of Inner Mongolia (P.R. China)	Steffens M., Kolbl A., Totsche K.U., Kogel-Knabner I.	2008
Extensive Management Promotes Plant and Microbial Nitrogen Retention in Temperate Grassland	Vries F.T., Bloem J., Stevens C.J., Bol R., Bardgett R.D.	2012
The measurement of soil fungal:bacterial biomass ratios as an indicator of ecosystem self-regulation in temperate meadow grasslands	Bardgett R.D., McAlister E.,	1999
Effects of Cultivation on Soils in Northern Great Plains Rangeland	Aguilar R., Kelly E.F., Heil D.,	1988
Soil organic carbon and black carbon storage and dynamics under different fire regimes in temperate mixed-grass savanna	Ansley R.J., Boutton T.W., Skjemstad J.O.,	2006
Impact of carbon farming practices on soil carbon in northern New South Wales	Cowie A. L., Londergan V.E., Rabbi S.M.F., Forasier F., Macdonald C., Harden, S., Kawasaki A., Singh B.K.	2013
Permanganate Oxidizable Carbon Reflects a Processed Soil Fraction that is Sensitive to Management	Culman S.W., Snapp S.S., Freeman M.A., Schipanski M.E., Beniston J., Lal R., Drinkwater L.E., Franzluebbers A.J., Glover J.D., Grandy A. S., Lee J., Six J., Maul J.E., Mirsky S.B., Spargo J.T., Wander M.M.	2012

Table B2. Continued

Grazing intensity levels influence C reservoirs of wet and mesic meadows along a precipitation gradient in Northern Patagonia	Enriquez A.S., Chimner R.A., Cremona M.V., Diehl P., Bonvissuto G.L.	2015
Changes in carbon and nitrogen in soil particle-size fractions along a grassland restoration chronosequence in northern China	He N., Wu L., Wang Y., Han X.	2009
Estimating organic carbon stocks of mineral soils in Denmark: Impact of bulk density and content of rock fragments	Harbo L.S., Oleson J.E., Liang Z., Christensen B.T., Elsgaard L.	2022
Comparison of Permanganate-Oxidizable Carbon and Mineralizable Carbon for Assessment of Organic Matter Stabilization and Mineralization	Hurisso T.T., Culman S.W., Horwath W.R., Wade J., Cass D., Beniston J.W., Bowles T.M., Grandy A.S., Franzluebbers A.J., Schipanski M.E., Lucas S.T., Ugarte C.M.	2016
Comparative assessment of soil health indicators in response to woodland and silvopasture land use systems	Karki, S., Shange R., Ankumah R., McElhenney W., Idehen O., Poudel S., Karki U.	2021
Evaluating the impacts of alternative grazing management practices on soil carbon sequestration and soil health indicators	Kim J., Ale S., Kreuter E.P., Teague R.W., DelGrosso S.J., Dowhower S.L.	2023
Rangeland management impacts on the properties of clayey soils along grazing gradients in the semi-arid grassland biome of South Africa	Kotzé E., Sahndage-Hofmann A., Meinel J.A., Du Preez C., Amelung W.	2013
Biodiversity and ecological long-term plots in Southern Patagonia to support sustainable land management: The case of PEBANPA network	Peri P.L., Lencinas M.V., Bousson J., Lasagno R., Soler R., Bahamonde H., Martínez Pastur G.	2016

Table B2. Continued

Silvopastoral system for resilience of key soil health indicators in semi-arid environment	Ramakrishnan S., Kumar S., Chaudhary M., Govindasamy P., Yadav M., Prasad M., Srivastava R., Kumari B., Prajapati K. 2021
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APPENDIX C

RANCH COORDINATES AND LETTER KEY

Table C.1. Ranch names with the letter used to refer to the ranch through this manuscript as well as decimal-degree coordinates (why not ranch-averaged SOCv as well?).

Ranch Name	Anonymized Letter	Latitude	Longitude
Redacted for rancher privacy	A	4	Redacted for rancher privacy
	B	4	
	C	4	
	D	4	
	E	4	
	F	4	
	G	4	
	H	4	
	I	4	
	J	4	
	K	4	
	L	4	
	M	4	
	N	4	
	O	4	
	P	4	
	Q	4	
	R	4	
	S	4	
	T	4	
	U	4	
	V	4	
	W	4	
	X	4	
	Y	4	
	Z	4	
	AA	4	
	AB	4	
	AC	4	

APPENDIX D

LONG-TERM EFFECTS OF CONSERVATION RESERVE
PROGRAMS

D.1 Results

One of our ranches we characterized provided an opportunity to investigate the long-term consequences of management as part of the Conservation Reserve Program (CRP). The CRP attempts to reduce water and wind erosion by establishing perennial grass on degraded crop and rangelands. These perennial grasses also help to increase carbon and turn carbon sources into sinks (Gebhart et al. 1999). We dug three pairs of holes roughly 100 feet apart—one on each side of the fence separating +CRP and -CRP areas (Figure D.1). This set of fence-line contrasts was designed to assess whether the sharp differences in above-ground biomass corresponded to measurable shifts in below-ground properties, such as SOC_v and SOC_m. Despite the +CRP soils being grazed at a lighter stocking rate for ~30 years, +CRP and -CRP average (± 1 SD) SOC_v (33 ± 4 and 33 ± 2 t ac-ft⁻¹ [74 ± 9 and 75 ± 5 Mg ha-0.3m⁻¹], respectively) were not statistically significantly different from one another ($p=0.941$). Contrary to our expectations, -CRP soils actually exhibited a slightly higher average SOC_v than adjacent +CRP soils.

D.2 Discussion

Consistent with our suggestion that SOC_m can provide a misleading picture of belowground differences, average SOC_m for +CRP soils were 220% greater than -CRP soils—for the uppermost inch (2.5cm). However, the relatively high CVs (+CRP: 43%, -CRP: 28%) meant we could not detect a statistically significant difference in these SOC_m results ($p=0.15$). Across our multiyear project, this specific finding has been one of the most powerful reminders that rangeland soil health presents a formidable signal-to-noise challenge. Because even this ~30-year +CRP management effect “signal” could not be detected against the background

“noise” of SOCv variability, the likelihood of a much shorter-duration management “signal” (e.g., 1-10 years) being detectable is extremely low. On a more positive rangeland soil health note, the “absence of evidence” as highlighted here is not “evidence of absence” (Rees 1971)--in other words, just because our team failed to detect a measurable difference in SOCv does not mean that future research will not be able to detect measurable (and meaningful) differences in other soil properties such as top-soil erosion rates, carbon transit times, or porosity.

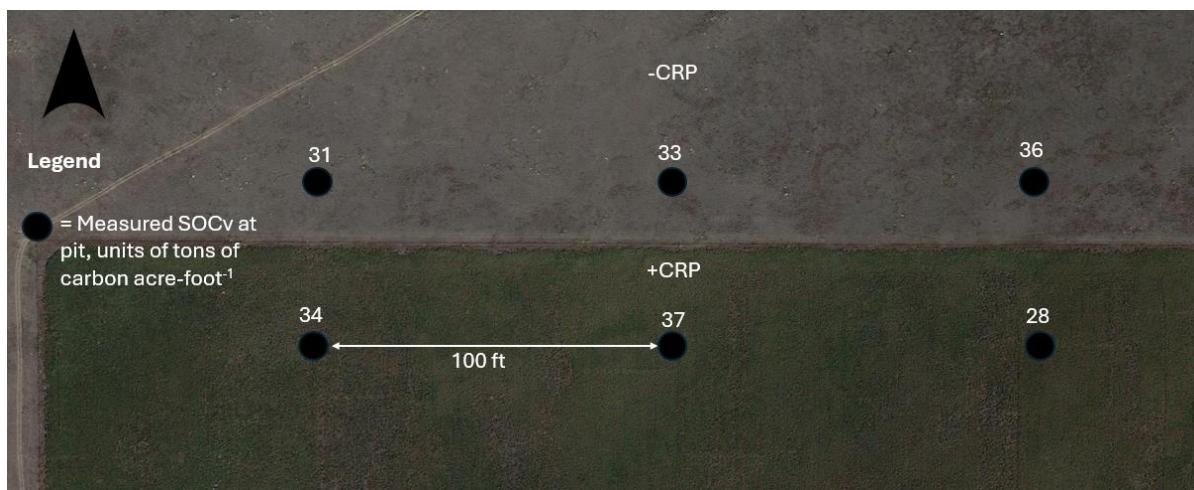


Figure D.1. Map of CRP experiment design with measured SOCv values (t ac-ft⁻¹; multiply by 2.24 to obtain Mg ha^{-0.3m}⁻¹). The distance between points is ~100 feet east to west and ~50 feet north to south.

APPENDIX E

CO-LOCATED CATENAS

E.1 Ranch S

A frequent criticism of catena studies is that any one selected catena may not adequately represent soil-landscape relationships (Malinowska and Szumacher 2013, Richter and Burras 2017). Our study sampled four pairs of catenas within 300 feet (100 m) of one another, although only two (ranches D and S) had multiple holes dug at each hillslope position.

Our ANOVA analysis of ranch S revealed no statistical differences between paired hillslope positions ($p>0.1$). However, significant differences in SOC_v were observed between specific positions, highlighting the influence of topographic location on soil carbon stocks (Figure E1).

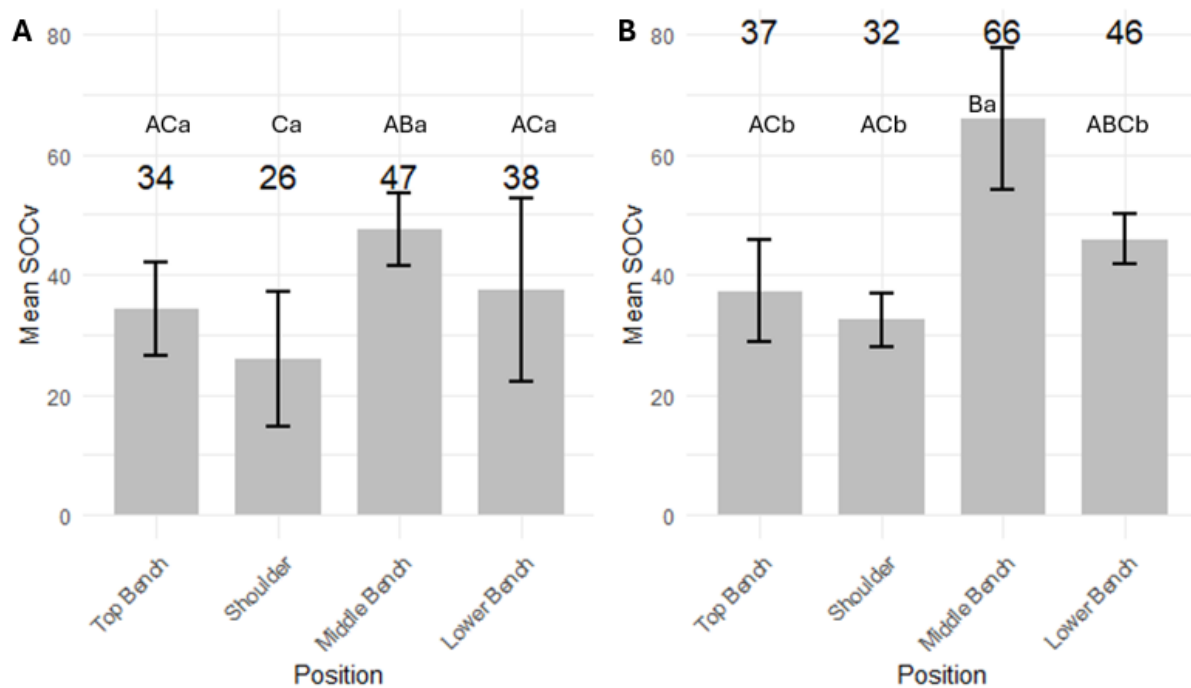


Figure E1. Average ($\pm 1SD$) SOC_v (t ac-ft⁻¹) at each hillslope position for ranch S's paired catenas: A- east, B, west. Numbers above each bar are averages; multiply by 2.24 to obtain Mg ha-0.3m⁻¹. Uppercase letters above data labels represent statistically significant ($p<0.05$) ANOVA differences between the paired catenas within hillslope position, and lowercase letters represent

statistically significant ($p < 0.05$) ANOVA differences between hillslope positions within an individual catena.

E.2 Ranch D

Ranch D also had paired catenas ~300ft (100 meters) apart; however, their carbon distribution was much less similar (E2). Post-hoc Tukey HSD tests revealed statistically significant differences in SOC_v between corresponding hillslope positions of the two catenas. Ranch D's west crest, for example, had statistically significantly lower SOC_v compared to the east crest (30 vs 47 t ac-ft⁻¹, respectively; $p = 0.04$); conversely, ranch D's west toeslope showed statistically significantly higher SOC_v compared to the east toeslope (55 vs 35 t ac-ft⁻¹, respectively; $p = 0.008$). Figure E2 suggests that the west catena generally exhibits higher SOC_v, particularly in depositional areas like toeslopes, while the east catena has higher carbon storage at the crest position.

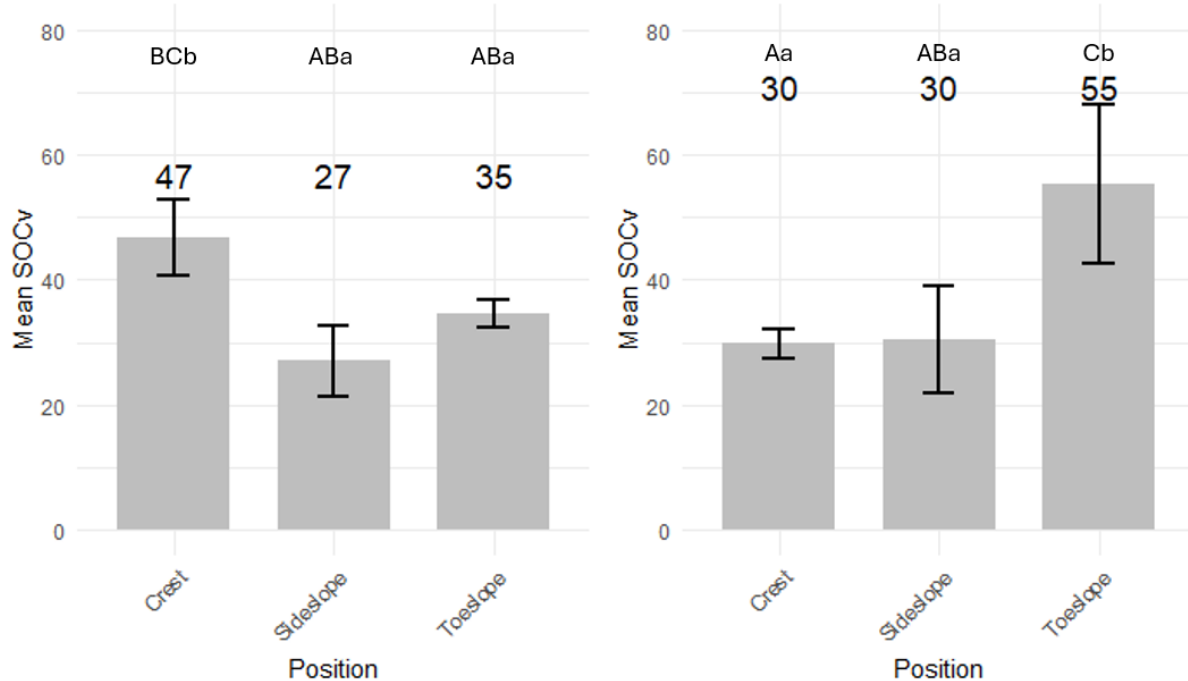


Figure E2. Average ($\pm 1SD$) SOCv ($t\ ac\text{-}ft^{-1}$) at each hillslope position for ranch D's paired catenas A- east, B, west. Numbers above each bar are averages; multiply by 2.24 to obtain $Mg\ ha^{-1}\ 0.3m^{-1}$. Uppercase letters above data labels represent statistically significant ($p < 0.05$) ANOVA differences between the paired catenas within hillslope position, and lowercase letters represent statistically significant ($p < 0.05$) ANOVA differences between hillslope positions within an individual catena.

Ranches S and D have different carbon storage patterns on a hillslope scale with ranch S and ranch D's west catena having more SOCv on lower, depositional positions (see Figures E1, and E2A), but ranch D's east catena expresses higher SOCv on the crest and lower SOCv on the sideslope and toeslope positions. Figure E1 shows that one catena was sufficient in calculating SOCv levels for ranch S and variability, while Figure E2 argues that two catenas are not enough to quantify ranch level SOCv because there was significant differences between the crests and footslopes of ranch D. Both Figures E1 and E2 show the need for sampling at different topographic positions because of the statistical differences between hillslope positions on the same catena.

E.3 Co-Located Catena Variability

Table E1. Variability in Volumetric Soil Organic Carbon (SOCv) Across Catenas. The ratio of the highest SOCv value to the lowest SOCv value is calculated for each catena to quantify variability. The farther the ratio is from one, the greater the variability in SOCv within the catena.

Ranch	Highest SOCv Catena	Crest	Shoulder	Footslope	Toeslope	Average Difference
D	West	1.56	0.89	0.63	N/A	1.08
M*	East	0.20	1.90	1.01	N/A	1.17
S*	East	1.09	1.25	1.39	1.54	1.25
O	West	1.78	1.28	1.29	0.76	1.50

* Signifies that one hole was dug per position on at least one catena instead of the normal four

Table E2. Average ($\pm 1SD$) hillslope SOCv ($t\ ac-ft^{-1}$), most variable position (SH indicates single holes per hillslope position), and the number of additional samples needed to detect an X% change in SOCv at alpha 0.05 and 80% power. The "Quotient" row depicts the highest to lowest catena average SOCv ratio.

Samples needed to detect __X% change				
Catena Sets	Hillslope SOCv	1%	10%	25%
D				
East	36±9	20,000	200	32
West	39±14	42,000	420	68
Quotient	1.08			
M				
East	54±20	44,000	430	70
SW	63±20	33,000	330	54
NE	63±36	104,000	1040	170
Quotient	1.17			
S				
East	45±10	16,000	160	24
West	36±7	12,000	120	20
Quotient	1.25			
O				
East	34±9	24,000	240	38
West	51± 25	76,000	750	120
Quotient	1.50			

APPENDIX F

TRADEOFFS FOR MEASURING
SOIL HEALTH INDICATOR CHANGES

F.1 Introduction

Grasslands are critical ecosystems that are essential in storing carbon, supporting biodiversity, and providing various ecosystem services, which are vital for ecological health. Sadly, overgrazing, land conversion, and climate change threaten these essential functions (Rumpel et al. 2020). While regenerative grazing offers ranchers the potential to increase forage production and raise more cattle on the same land area, its effectiveness remains to be fully proven. Additionally, although regenerative grazing is proposed to enhance soil functional capacity, support biodiversity, and promote plant growth, the evidence supporting these claims varies across different studies (Jastrow 1996, Teague et al. 2013). A comprehensive review by Newton et al. (2020) of 229 studies on “regenerative agriculture” and “regenerative grazing” revealed that only half the studies provided explicit definitions for these frequently utilized terms, highlighting the inconsistency in this field. We will use Rhodes's (2017) definition of regenerative grazing: “an agricultural practice that increases land productivity and improves soil health, biodiversity, ecosystem function, and water quality.” Of particular interest is whether regenerative grazing influences soil health, which is defined by the status of its biological, physical, and chemical functions (Al-Kaisi 2014, NRCS 2021).

The previous literature review examined nine rangeland management techniques considered tools for regenerative grazing, such as AMP and strip grazing (Bartley et al. 2023). These strategies rely on strategic herd movements that optimize forage availability and forage growth and generally aim to enhance soil health by increasing soil organic matter (SOM) inputs via increased plant productivity. Because SOM is half SOC (Pribyl 2010), these grazing strategies have been highlighted as a potential means of boosting soil carbon sequestration

(Teague et al. 2011, Reinhart et al. 2022, Stanley et al. 2024). It is pivotal to accurately identify low-variability soil health indicators to track changes in soil health over time (Jenny 1941, Birkeland 1999). Soil health directly influences soil function - vital benefits provided to humanity by the natural environment (NRCS 2021, Bagnall et al. 2023). Researchers can better monitor these essential functions using these indicators even as soil slowly evolves. Soil health indicators must closely link to specific soil functions outlined by the NRCS (see Table 1.3) to ensure effective and relevant soil testing. By identifying quality soil health indicators, this study will provide information essential for accurately assessing the impacts of regenerative grazing.

As soil health becomes a more familiar concept in ranching, it is essential to consider how to measure it and which metrics are most appropriate for assessing soil conditions before and after implementing specific management practices (O'Donoghue et al. 2022). This study examines the variability of soil health indicators and quantifies the sampling effort required to detect meaningful changes. We identified six relatively low-cost, relatively low-variability indicators responsive to management practices that could help ranchers implement effective soil health assessments. Ranchers are keenly interested in understanding how their land management practices impact soil properties and the associated costs of these changes. However, to make informed decisions regarding soil property changes, ranchers need to understand the quantifiable costs associated with these changes. This lack of information about financial costs associated with soil health assessment hinders ranchers' ability to make informed management decisions and is notably absent in soil science literature and agricultural guides.

F.2 Methods

F.2.1 Study Site and Experimental Design:

This chapter utilizes the same study design and experimental methods described in Chapter 2. Any chapter-specific adaptations or additional procedures are outlined below.

F.2.2 Workflow

After field sampling, 1004 soil samples were processed at Montana State University (MSU), and 37 soil samples were processed at Cornell University's Soil Health Laboratory (website). Together, 42 different soil health indicators were explored (Table F1, F2, and F3). The average cost per soil sample was estimated by averaging costs from Cornell University, UC Davis and Ward Laboratories. The number of soil samples identified through a power analysis, which estimated the total cost required for a rancher to detect a 10% improvement in specific soil properties. (Both smaller [+1%] and larger [+25%] estimates can be found in Chapter 2). Post power analysis, the number of samples was multiplied by the average cost to find the "extended cost" or the price a manager would have to pay to see a +10% increase in a soil health indicator. While less expensive and less variable tests are preferred, they must also correlate with soil function and be responsive to changes under different grazing management practices. We quantified management-responsiveness through a literature review (Chapter 1).

Experimental designs should begin with an initial variability assessment to inform the smallest sample size needed for meaningful results. Data were collected from four soil pits at

each sampling position to calculate the local variability of each soil property, so we conducted a power analysis using an online sample calculator (<https://clincalc.com/Stats/SampleSize.aspx>; Figure F1); an additional 19 programs and 10 online calculators are provided by Serdar et al. (2021). We used this calculator to estimate the number of samples required to detect a 10% change in each soil indicator at specified alpha (Type I [false positive] error 0.05) and beta (Type II [‘false negative’] error 0.8) levels (Figure F1). These levels represent a commonly accepted balance between Type I and Type II errors, with an 80% probability of correctly determining no difference while maintaining a 5% probability of falsely detecting no difference (Sommet et al. 2023; Figure F2). Thus, there remains a 5% chance of making a Type I error and a 20% chance of making a Type II error.

Statistical Parameters

Anticipated Means

Type I/II Error Rate

Group 1 ? ±

Group 2 ? %

Alpha ?

Power ?

Enrollment ratio ?

Figure F1. The power analysis calculator was used for all power/ sample size calculations for this study (<https://clincalc.com/Stats/SampleSize.aspx>).

	Reality: No Effect	Reality: There is an Effect
Conclusion: No Effect		Type I Error Regenerative grazing is determined to not increase SOCv, but it did
Conclusion: There is an effect	Type II Error Regenerative grazing is determined to increase SOCv, but it didn't	

Figure F2. Graphic showing Type I and Type II errors in testing whether regenerative grazing increases SOCv.

F.3 Results

F.3.1 Montana State University Results

F.3.1.1 Power Analysis Example

At one crest sampled, the average SOCv was $14.7 \text{ t ac-ft}^{-1}$ ($33.0 \text{ Mg ha-0.3m}^1$), with a standard deviation of 8.4 ($18.8 \text{ Mg ha-0.3m}^1$) across the four pits. A power analysis determined

that detecting a 10% increase in SOCv at 80% power with an alpha of 0.05 would require 1,026 samples—far more than the initial four.

F.3.1.2 Cost Analysis

Table F.1 summarizes extended costs for five different soil health indicators analyzed at MSU to detect a +10% improvement. Of these, the lowest-cost chemical soil health indicator is SOCm (\$30), while the lowest extended-cost for a physical indicator is soil porosity (\$60). In other words, if a ranch obtained baseline soil testing results showing average (± 1 SD) SOCm ($3.6 \pm 0.1\%$) and porosity ($54.0 \pm 1.6\%$) values, only \$90 would be sufficient to analyze the 10 samples (6 for SOCm, 4 for porosity) required to detect a +10% increase in both indicators. By contrast, testing for a 10% increase in SOCv could set a rancher back over \$20,000, 41 times the cost of all the other indicator tests combined. SOCv also has the highest cost per test and highest average CV, explaining the high cost of seeing a 10% change.

While some rangeland management techniques can increase SOCv, detecting such changes can be both difficult and costly. Our findings indicated that between four holes ~10 feet apart, SOCv had an average CV of 20% (Table F.1). A power analysis suggests that using the Blackfeet nation's average mean and average standard deviation $42 \pm 8 \text{ t ac-ft}^{-1}$ ($94 \text{ Mg ha-0.3m}^{-1}$) 138 samples are necessary for achieving 80% statistical power. So, on the Blackfeet Nation, the average price to find a 10% increase in SOCv for a single hillslope position at 80% power is \$22,000 (~\$150 per sample). In testing the power of this study using the same SOCv mean and standard deviation as before with four samples per hillslope position, a power of 16.9% was calculated. This indicates only a 16.9% chance of detecting a 10% increase in SOCv.

Table F1. Cost analysis of soil health indicators tested at MSU. The Category column shows whether an indicator is C= Chemical or P= Physical.

Category	Indicator	Average Cost (\$)	Average Indicator Value	Number of samples	Average CV (%)	Sample Size for +10% Increase	Extended Cost (\$)
C	SOC _m (%)	5	3.6	1,004	4	6	30
C	SOC _v (t ac-ft ⁻¹)	160	42.0	118	20	130	21,000
P	Porosity (%)	15	54.0	134	3	4	60
P	Bulk Density (g cm ³)	8	1.3	865	8	18	140
P	Aggregate Stability (slaking, %)	15	0.7	156	7	16	240

F.3.2 Literature Review Cost Analysis

Table F.2 presents 17 different soil health indicators taken from 17 different papers that explored how grazing affected soil health (see Appendix B). This table did not consider whether grazing affected soil health but served as an exploratory table investigating the variability of different indicators. Indicators were only included if an average for treatment was reported along with a metric of variability, which was then converted into a standard deviation required for the power analysis. Four soil health indicators—bacterial PLFA, fungal PLFA, particulate organic carbon, and mineralizable carbon—had an extended cost of over \$25,000 (Table F.2).

Interestingly, total PLFA had the lowest testing cost among biological indicators but still amounted to \$3,600. The lowest chemical soil health indicator cost was \$28 for pH. Bulk density had the lowest extended cost (\$540) of the two reported physical indicators. Although Table F.2 uses the prices from Cornell University, University of California (UC) Davis, and Ward Labs, those data did not come directly from those labs and instead originated from peer-reviewed articles.

Table F2. Meta-analysis power analysis results. Data was taken from 16 papers during a literature review (Appendix A). The Category column shows if an indicator is C=chemical or P=physical by B=biological. The average cost column was calculated using Cornell University's Soil Health Lab, UC Davis's Analytical Lab, and Ward Laboratories prices.

Category	Indicator	Average Cost (\$)	Average Indicator Value	n	Average CV (%)	Sample Size for +10% Increase	Extended Cost (\$)
B	Total PLFA (nmol g ⁻¹)	100	78.0	4	10	36	3,600
B	Basal Respiration (mg CO ₂)	30	0.8	12	28	250	7,400
B	Microbial Biomass (mg kg ⁻¹)	30	380.0	27	40	480	14,000
B	Bacterial PLFA (nmol g ⁻¹)	100	34.0	5	30	280	28,000
B	Fungal PLFA (nmol g ⁻¹)	100	0.7	5	63	1,300	130,000
C	pH	7	7.3	65	3	4	28
C	SOC _m (%)	7	2.8	94	17	92	640
C	Total Soil N (mg 100g ⁻¹)	11	1.7	38	15	70	770
C	SOC _v (kg m ²)	37	9.4	37	11	42	1,600
C	Electrical Conductivity (dS m ⁻¹)	34	0.5	40	15	68	2,300

Table F2 Continued

C	Extractable NO ₃ (mg 100g ⁻¹)	48	5.6	6	14	64	3,100
C	Extractable NH ₄ (mg 100g ⁻¹)	48	2.9	4	24	180	8,600
C	POXC (mg kg ⁻¹)	30	580.0	26	34	370	11,000
C	Mineralizable Carbon (mg kg ⁻¹)	30	250.0	16	53	880	26,000
C	Particulate Organic Carbon (Mg ha ⁻¹)	50	3.9	17	44	620	31,000
P	Bulk Density (g cm ⁻³)	30	1.3	111	8	18	540
P	Infiltration (cm hr ⁻¹)	40	5.1	13	24	180	7,200

F.3.3 Cornell University Soil Health – CASH Cost Analysis

Sixteen indicators were significantly correlated with the overall soil health score (Figure F1). This was determined using the correlation coefficient, which represents the strength and direction of the linear relationship between two variables (Asuero et al. 2006). This is commonly referred to as the Pearson (Product Moment) *r* correlation (Rodgers and Nicewander 1988). Additionally, latitude could be eliminated from the cost analysis because it is not a soil health indicator; its inclusion may relate to potential climate-predictive factors outside the study's core focus.

Because Autoclaved Citrate Extractable (ACE) protein tests process can be time-consuming and expensive, Cornell University's Soil Health lab created an ACE Protein

predictive model using a suite of parameters included in the standard soil health test, including % sand, % silt, % clay, Organic Matter, Active Carbon, Total Carbon, Total Nitrogen, and Tot C/Tot N. Compared with actual ACE Protein Index values, the model explains 76% of the variation (Cornell University 2024). Because the predicted protein and ACE Protein Index were closely related (Figure F3), the predicted protein values were used in the laboratory cost analysis. Despite our samples generally exceeding 10 g kg^{-1} in predicted ACE protein levels, we noted that predicted soil protein through linear regression indicates a strong correlation to overall soil health ($R^2 = 0.79$, $p < 0.001$). One study found that increasing grazing intensity increased ACE protein levels, but the max ACE protein value from that study was below 6 g kg^{-1} (Taylor et al. 2025). Achieving an ACE Protein level of 10 g kg^{-1} corresponds with a soil health score of 80/100 (Schindelbeck et al. 2016) suggesting a strong link between sufficient protein levels and a healthy soil ecosystem. This indicates a limited range in which patterns of soil health can be discerned through ACE protein levels. Predicted soil protein was also closely related to SOC and aggregate stability ($p = 0.005$).

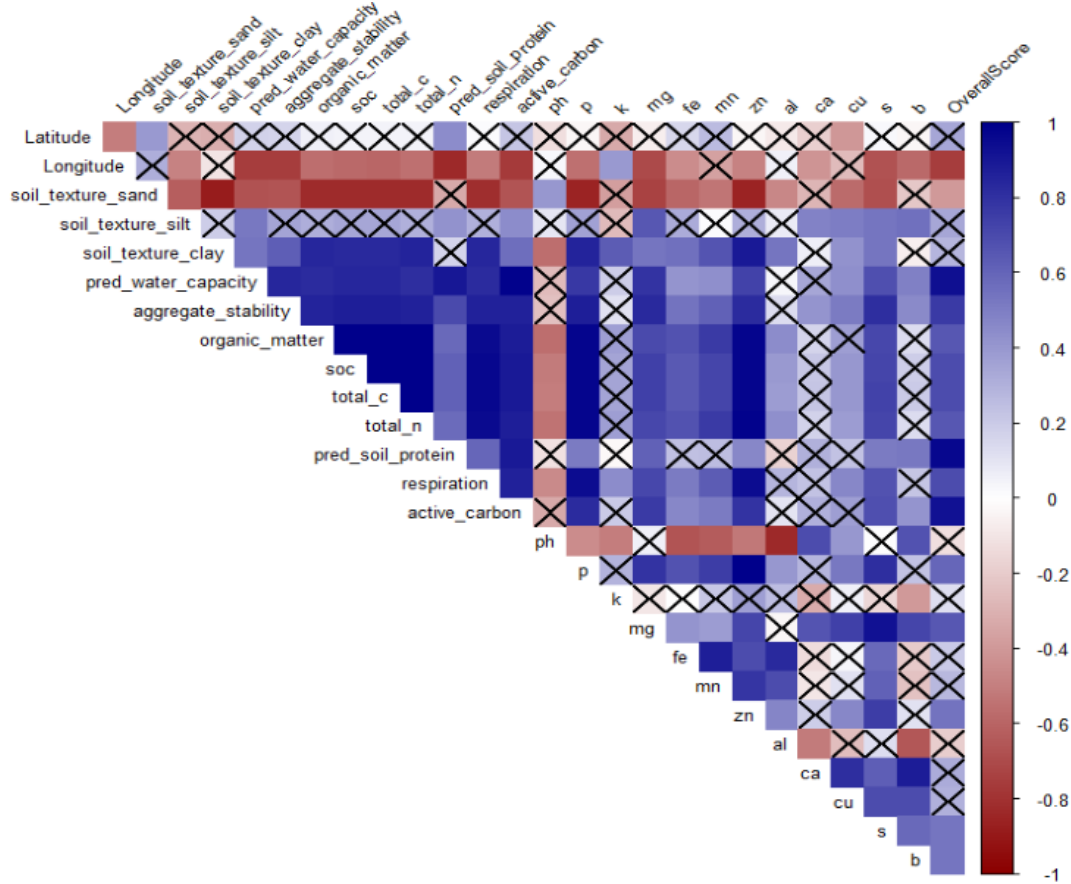


Figure F3. A Correlation matrix between different variables of the CASH data set. Boxes with an X in them have a non-statistically significant relationship ($p=0.05$). Blue signifies a positive correlation, while red indicates a negative correlation, as highlighted on the left x-axis. The overall score column is the overall soil health score as determined by Cornell University in the CASH ($n=37$).

The Cornell Soil Health Laboratory analyzed 37 soil samples, including four blind duplicates, to assess variability for the Comprehensive Assessment of Soil Health (CASH). These results (Table F.3) enabled a comparison of soil health indicators from MSU (see Table F.1) and Cornell University Laboratories, assessing variability and pinpointing the most consistent soil indicator values. According to the data, from the subset of samples sent to

Cornell's Soil Health Lab, the average CV of POXC was 6%, comparable to the well-supported indicator SOC with a CV of only 5%.

Identifying cost-effective soil health indicators was essential for land managers striving to enhance rangeland management sustainably and economically. By selecting the least expensive biological indicators correlated with soil health in each category in Table F3, it was determined that active carbon (\$360) provided significant value for effective management. Physical indicators, such as predicted water capacity (\$60) and aggregate stability (\$120), along with chemical indicators like SOM (\$60) and total nitrogen (\$30), reflected affordability while contributing to effective soil management.

Table F3. Cost analysis of soil health indicators from samples sent to Cornell University. The Category column shows if an indicator is B = Biological, C = Chemical, or P = Physical.

Category	Indicator	Average Cost (\$)	Average Indicator Value	Average CV (%)	Sample Size for +10% Increase	Extended Cost (\$)
B	Predicted Soil Protein (mg kg ⁻¹ soil)	30	8.8	60	2	60
B	Active Carbon (ppm)	30	700.0	6	12	360
B	Respiration mg (CO ₂ -C kg ⁻¹ soil 24 hr)	30	1.2	9	24	720
C	Total N (%)	15	0.3	3	2	30
C	SOM (%)	15	3.4	5	8	120
C	pH	15	6.8	1	2	60
C	SOCm (%)	15	3.4	5	8	120

Table F3 Continued

C	Potassium (ppm)	15	430.0	6	10	150
C	Magnesium (ppm)	15	360.0	6	10	150
C	Iron (ppm)	15	1.3	6	12	180
C	Total C (%)	15	3.4	6	12	180
C	Phosphorous (ppm)	15	2.7	12	42	630
C	Copper (ppm)	15	0.1	13	50	750
C	Sulfur (ppm)	15	4.8	14	64	960
C	Aluminum (ppm)	15	9.8	15	72	1,100
C	Boron (ppm)	15	0.4	16	84	1,300
C	Manganese (ppm)	15	3.8	17	94	1,400
C	Zinc (ppm)	15	0.3	20	128	1,900
C	Calcium (ppm)	15	3100.0	25	202	3,000
P	% Clay	10	28.0	3	4	40
P	Predicted Water Capacity (cm ³ water cm ⁻³ soil)	30	0.2	2	2	60
P	% Sand	10	45.0	5	8	80
P	% Silt	10	26.0	6	10	100
P	Aggregate Stability (%)	30	65.0	4	4	120
B,C,P	Overall Health Score	130	82.0	2	2	260

F.4 Discussion

F.4.1 Recommendations

Based on the results from Tables F1–F3 as well as the literature review, the following soil health indicators are recommended for land managers aiming to improve rangeland soil health through management changes:

- **Biological indicators**
 - Active carbon (\$360)
 - Respiration (\$720 see Table F3 \$26,460 see Table F2)
- **Chemical indicators**
 - Total soil carbon (\$180)
- **Physical indicators**
 - Predicted water capacity (\$60)
 - Aggregate stability (\$240 see Table F1, \$120 see Table F3)

The best soil health indicators: 1. Are management-sensitive 2. Have a low cost per sample, and 3. Have A low level of noisiness or low CV. Ranchers should pick indicators with a low price to see a +10% change, a function of the cost per sample and noisiness represented by the result of the power analysis or number of samples. Of course, these indicators are only valuable to ranchers if they can shift with management. The literature has provided evidence that all recommended soil health indicators can change with a change in rangeland management.

Although predicted soil protein has the lowest cost of any biological indicator, with a +10% change, there is no evidence that it can respond to a change in grazing management. Therefore, it is not a recommended soil health indicator. In addition, the Cornell Soil Health Assessment overall score has a low CV and low extended cost (see Table F3) but may not be a very robust indicator of soil health on its own. For example, the average soil health score $\pm 1SD$ was 90 ± 12 for a CV of 13%. In contrast, individual indicators like SOC exhibit much higher variability (CV = 70%), reflecting their responsiveness to environmental and management factors. The low CV of the data set for the overall health score indicates that it isn't sensitive enough to detect meaningful changes in soil health. Also, the overall soil health score was disproportionately high, even for soils exhibiting degradation, low SOC_m, aggregate stability, and high percent sand. This raises questions about the ability of the scoring system to reflect fine-scale variations in soil condition.

F.4.2 Comparison of Testing Costs

Although respiration is 37 times more expensive when comparing Cornell University's (see Table F3) results to the meta-analysis results (see Table F2), resulting in a huge discrepancy, we believe that this difference can be attributed to variations in methodology, experimental design, and analytical precision. For instance, Cornell's approach may involve more rigorous protocols or advanced equipment, which could yield more reliable results.

Table E1 depicts that the average SOC_v CV is 20% from MSU's dataset, compared to a meta-analysis average CV of 30% (see Table F.2). However, MSU's cost per sample is \$156,

which includes sampling (\$76), travel (\$7), and laboratory fees (\$73). In contrast, UC Davis and Cornell University charge \$15 per sample for SOC_m and an additional \$15 for bulk density (BD) for a total of \$30 per SOC_v cost. This is only for one soil horizon, and on average, each soil pit had ~3 soil horizons, costing \$90 to find the SOC_v for a single soil pit. This is \$66 more than MSU's but includes no sampling or traveling costs.

Variability greatly influences this cost analysis, as it dictates the number of samples required for statistically valid results. The various soil samples sent to Cornell University allowed me to calculate the variability for each soil health test, including aggregate stability, with a sample size of $n=4$. This analysis enabled me to compare variability and associated errors from different laboratories presented in Tables F1 and F3, guiding decisions on the most cost-effective lab for detecting soil health changes. Take aggregate stability, for example; while Cornell University charges \$30 per sample and MSU's charges half as much at \$15, the variability differs by 75% between the two institutions. Cornell University has a CV of only 4%, compared to MSU's 7%. Resulting in a cost of \$120 at Cornell University to achieve detect a +10% change in aggregate stability versus \$240 at MSU. This analysis indicates that the price per sample should not be the sole criterion for laboratory selection; instead, considering a subset of samples for variability assessment can provide significant long-term savings.

F.5 Conclusion

These findings presented in this study highlight the importance of balancing cost and variability when selecting soil health indicators to test. For ranchers aiming to monitor changes in their soil health they should primarily focus on key indicators. Not all soil health indicators are equally variable, cost the same to test, or even respond to management changes. Next, ranchers should conduct a preliminary variability assessment to estimate the number of samples that will be required. Finally, ranchers should select soil health indicators as well as laboratories by prioritizing precision over cost. Variability was a much bigger driver of extended cost than cost per sample (see Tables F1-F3), highlighting that the cheapest option may not always yield the most reliable or actionable data. By adopting these practices and using the recommended soil health indicators, ranchers can make informed decisions that balance affordability with scientific rigor, ensuring efficiency in soil health monitoring.

