ABSTRACT

Title of Thesis: ECOLOGICAL INTENSIFICATION WITHIN FORAGE SYSTEMS BENEFITS SOIL ARTHROPODS AND SOIL BIOTA-MEDIATED ECOSYSTEM SERVICES

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Modern agricultural practices, such as intensive soil tillage, crop monocultures, and overfertilization pose sustainability challenges in forage and livestock farming, impacting soil quality and ecosystem stability. Throughout the world, studies have revealed that agricultural intensification negatively impacts aboveground arthropods, prompting interest in ecologically intensified forage systems for agricultural sustainability. Belowground, soil arthropods provide many essential ecosystem services on the farm, including decomposition, biological control, and bioturbation; however, these organisms and their services are generally overlooked. Understanding the response of soil arthropods and soil properties to changes in land use/agricultural management practices is vital for ecologically and economically balanced systems. To investigate how land use in forage systems influences soil arthropod communities, soil properties, and their ecosystem services, I sampled soil biotic and abiotic properties and measured the rate of feeding activity by soil biota. This study was conducted across a land use gradient ranging from intensively managed to semi-natural habitats, including established corn-soybean rotations, forage pastures, grass margins, and woodlots. I found significant differences in several soil properties including soil bulk density and soil moisture between land use types. Soil arthropod communities were more abundant and diverse in pastures compared to corn plots and the soil biological quality was significantly lower in corn plots compared to all other land uses. Soil taxa such as Acari, Collembola, Diplopoda, and Chilopoda were associated with soil properties investigated here and soil biota feeding activity was highest in ecologically intensified land use types (characterized by high plant diversity, plant perenniality, and system circularity). The results of this study suggest that ecological intensification, through the presence of plant diversity, perenniality, and system circularity, supports soil quality and soil arthropod communities within forage systems. This research informs decision-making in livestock systems as they continue to dominate land use throughout the United States.

ECOLOGICAL INTENSIFICATION WITHIN FORAGE SYSTEMS BENEFITS SOIL ARTHROPODS AND SOIL BIOTA–MEDIATED ECOSYSTEM SERVICES

by

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Abstract

Due to their large land-use footprint, livestock production systems throughout the United States are a target for sustainability improvements. These systems utilize 35% (656 million acres) of land in the US, including pastureland and feed fields used to produce corn and soybeans. Today, many conventional feed fields have undergone agricultural intensification which alters soil properties and introduces regular disturbance. Soil arthropods are especially sensitive to extreme environmental conditions and disturbance; therefore, it is important to understand their response to common land use types throughout forage systems. Leveraging plant diversity, perenniality, and system circularity has the potential to support soil arthropods and their natural ecosystem functions, increasing sustainability and resilience. This study investigated the response of soil properties and soil arthropod communities to four distinct land use types within livestock systems: corn-soybean rotation fields, perennial forage pastures, grass margins, and woodlots. I implemented subterranean pitfall traps to sample soil arthropods and sampled soil to measure several soil properties. Results from this research identified differences between land use types for several soil properties, including soil bulk density and soil moisture. Soil arthropod communities were more abundant and diverse in pastures compared to corn plots, and the soil biological quality – arthropod index was lower in corn plots compared to all other land uses. Soil taxa such as Acari, Collembola, Diplopoda, and Chilopoda showed strong associations with soil properties investigated here. The results of this study suggest that land use types influence both soil quality and arthropod communities, with more diverse and perennial systems supporting greater soil biological activity and potentially enhancing the ecological sustainability of livestock production.

Introduction

Generating an estimated 88.4 billion USD of revenue in 2023, the cattle industry depends on forages and is the largest agricultural industry in the United States (Knight, 2025). In 2017, approximately 656 million acres of land in the United States were utilized as pastureland, while another 127.4 million acres were used to produce feed for livestock consumption (Winters-Michaud, 2025). Most pasturelands throughout the United States are minimally managed, even though improvements in stocking rates, grazing methods, and management practices could lead to greater pasture health and resiliency within ruminant industries (USDA, 2025). Other, feed-based systems rely on linear, monoculture, and high-disturbance fields, intensely managed through agricultural intensification to maximize efficiency and yield. Management practices within the livestock industry give little credence to biodiversity and ecosystem services, although their benefits have been documented for supporting and regulating services throughout agroecosystems (Haan & Landis, 2023; Hendrix, et al., 1986; Bardgett & Chan, 1999).

In response to these land-use choices, ecological intensification across all lands utilized for ruminant production has been proposed (Picasso et al., 2023). Ecological intensification in agriculture is the intelligent utilization of the ecosystem's natural functions (support and regulation) to produce food, fiber, energy, and ecological services, and has become a popular ideology in recent decades (Tittinell, 2014). In practice, ecological intensification is characterized by maximizing plant diversity, perenniality, and system circularity throughout forage systems to leverage important ecosystem services. Plant diversity through time (crop rotations) and through space (local vs landscape) contribute to resource provisioning for soil arthropods. Plant perenniality serves as a mechanism for decreased disturbance in ecologically intensified systems, and system circularity recycles animal and plant residues back into the soil environment, supporting the next round of primary production.

Soil fertility and soil health are common concepts in forage/soil management and are important aspects of effective ecological intensification strategies. Soil fertility is the capacity of the soil to provide nutrients for plants. In pastures, the production of high-quality forages depends largely on the status of the soil and the release of plant-available mineral nutrients from organic returns (Moore et al., 2020). Soil nutrient availability to forages is influenced by the soil fauna and physiochemical characteristics such as soil pH, structure, organic matter, and water content (Moore et al., 2020). Nitrogen is an especially important element in forage systems because of its role in amino acid and protein synthesis in plants. Nonetheless, nitrogen frequently limits forage production because N-fixation in forages is energetically expensive; other limiting elements necessary for N-fixation are scarce; and N-fixing plants are targeted by many herbivores, negatively impacting their abundance (Moore et al., 2020). Given these constraints, maintaining optimal nutrient levels and physiochemical properties in the soil is critical for ensuring that feed and forages for ruminants reach their full potential in both yield and nutritional quality.

Soil health, which encompasses soil fertility, plays a fundamental role in sustaining agricultural productivity. It is defined as the soil's ability to provide essential ecosystem services, including serving as a reservoir for nutrients, hosting pollinators, controlling erosion, sustaining biodiversity, sequestering carbon, and regulating water availability (Moore et al., 2020). A key but often overlooked component of soil health is soil biodiversity, particularly the organisms that drive nutrient cycling. These organisms facilitate essential processes such as the fragmentation of organic residues, nutrient mineralization, and redistribution, ultimately influencing soil fertility and plant growth.

Among the most diverse ecosystems on Earth, soil harbors approximately 25% of all described species (Decaens et al., 2006). Soil arthropods play a wide range of ecological roles, including entomophagous, bacterivorous, coprophagous, and saprophagous functions, with the latter three contributing significantly to the detritus food chains. Acari, Collembola, Diplopoda, and Formicidae are common soil arthropod taxa involved in the decomposition of organic matter through fragmenting and chemically modifying parent material (Gobat, 2004). Several entomophagous, parasitic, and predatory arthropods are also common among epigeal and hypogeal environments, including Carabidae, Parasitoid Hymenoptera, and members of Chilopoda. Coprophagous arthropods such as members of the order Diptera and Coleoptera family Scarabaeidae are important in pasture environments because they distribute and recycle animal excrement back into these systems.

Complex relationships exist between soil biological activity, agricultural management practices, and soil properties (Fig. 1.1). Soil properties, including bulk density, moisture, and temperature, are directly influenced by management practices and interact with biological activity reciprocally (Ponge, 1993; Chikoski, Ferguson, & Meyer, 2006). Normally, environmental conditions within soils are more stable than aboveground environments where stochastic events are more frequent and their effects greater (Mammalo et al., 2019). The introduction of any management practice altering the physiochemical properties of the soil leads to levels of disturbance many soil arthropods have not evolved to cope with. Additionally, vegetation structure can further influence the soil structure and arthropod communities by providing structure, exudates, and organic matter. It is anticipated that soil properties affect not only the biological components within the soil but also other soil properties through both direct and indirect interactions.

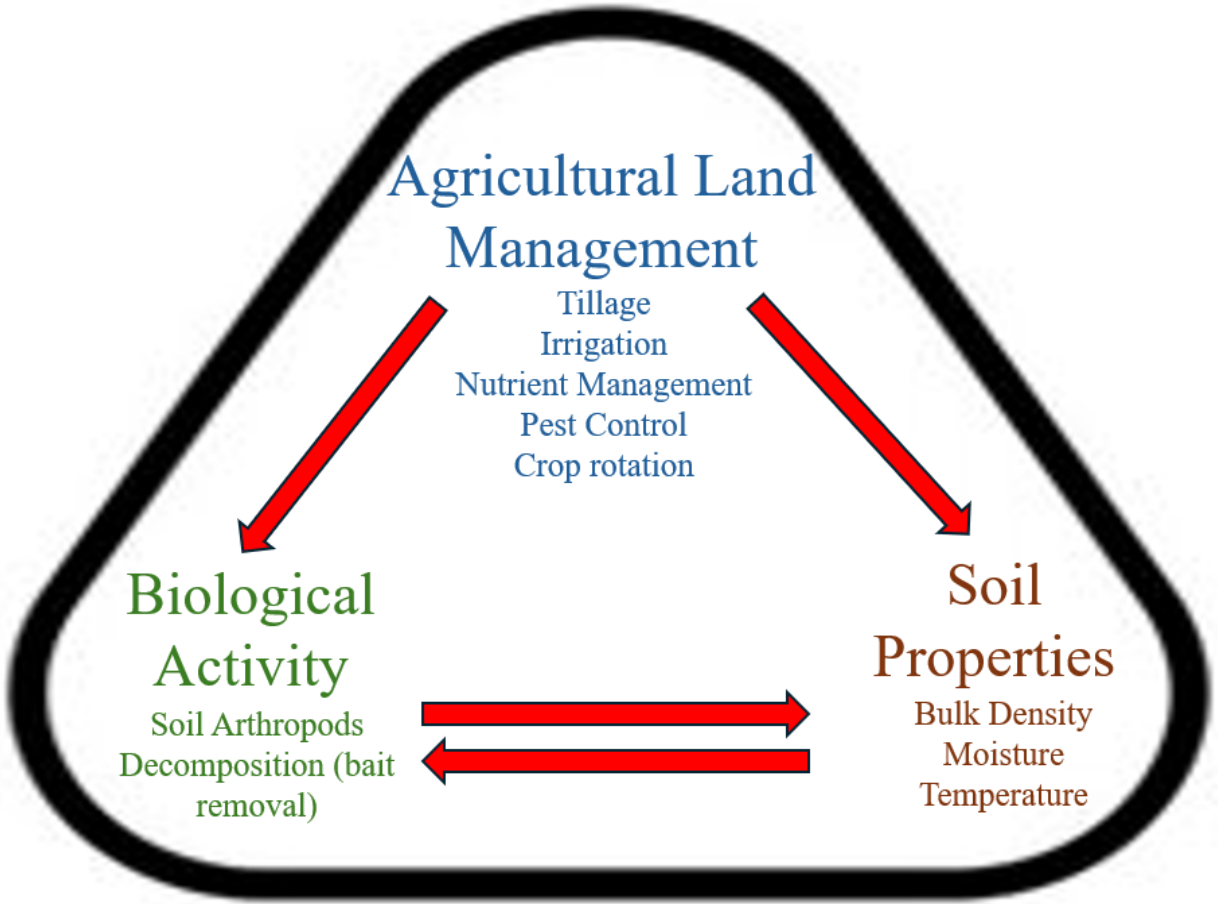


Figure 1.1 Conceptual diagram illustrating the interactions between agricultural land management practices, soil properties, and biological activity. Management practices influence both soil properties and biological activity, while soil properties and biological activity have a reciprocal relationship affecting soil health and ecosystem function.

Agricultural land management practices resulting in changes in soil properties like soil tillage, fertilization, pesticide application, irrigation, and low plant diversity in modern, conventional agriculture have directly resulted in adverse effects on various groups of soil biota (Mammalo et al., 2019; Bender et al., 2016; Tsiafouli et al., 2015). Across Europe, agricultural intensification has consistently led to negative impacts on the taxonomic and functional diversity of soil arthropods (Tsiafouli et al., 2015). In several areas, one or more functional groups were missing, which led to a long-term reduction in soil functioning. Increased soil temperatures resulting from the combined effects of decreased plant density and climate change cause shifts in soil biota community composition away from meso- and macro-fauna, towards microbiota (Briones et al., 2009). In some cases, the community composition of soil arthropods was driven by landscape context where adjacent donor habitats (woodlots) increased biodiversity in fields irrespective of management practices (Diekotter et al., 2010).

Losing soil biodiversity can impact the productivity and quality of crops and forages. Soils with abundant and diverse communities can provide ecosystem services benefiting humans, livestock, and soil environments. At high levels of soil arthropod biodiversity, the rate of litter decomposition is maintained (Wagg et al., 2014). Removing functional groups can result in a rapid decrease in the decomposition rate (Wagg et al., 2014) where areas possessing higher soil biodiversity increase plant productivity by enhancing plant nutrition through soil biota-mediated ecosystem services (Bender et al., 2016). Their role in decomposition and the nutrient cycle allows nutrients that were organically bound to plant residues and detritus to become available for plant uptake (Hendrix et al., 1986; Bardgett and Chan 1999), decreasing the need for fertilizer inputs.

The incorporation of ecologically intensified land use types into current forage systems is necessary to conserve soil arthropod diversity, bolster the ecosystem services they provide, and improve soil quality and resiliency. However, the impacts of integrating ecologically intensified systems characterized by high plant diversity, plant perenniality, and system circularity on soils and soil arthropods into current agricultural landscapes are unknown, and system-level studies like the one described here are required to begin to understand the importance of their inclusion.

To investigate how the implementation of ecologically intensified land use types within forage systems impacts the soil environment and soil arthropod communities, a study was conducted at the Clarksville, MD, Central Maryland Research and Education Center with two primary objectives. I aimed to (1) investigate the responses of soil arthropod communities to ecological intensification in forage systems and (2) evaluate the links between soil properties and soil arthropod communities resulting from management practices. Aside from sampling conventional feed fields and forage pastures (ecologically intensified), I included grass margin and woodlot plots, both semi-natural habitats common to farms across the Eastern United States. These areas served as a baseline for undisturbed areas, and including them will reveal their value in preserving soil arthropod diversity across the farm.

I hypothesized that pastures will have more abundant and diverse soil arthropod communities than conventional feed production systems. Conventional production systems will possess soil property metrics unfavorable to sensitive soil arthropods due to management practices and legacy effects from past practices. Soil bulk density, soil pH, and soil temperature range will be higher in the linear, monoculture, and high disturbance land use type (corn-soybean rotation plots) compared to perennial forage pastures. Conversely, volumetric water content, soil biological quality – arthropod index, and soil organic matter will be lower in corn plots compared to pasture plots. Due to similar management practices and vegetation structure, grass margins will not differ from pasture plots in soil arthropod community metrics and abiotic soil properties.

Methods

Site Description and Experimental Design:

This study was carried out at the University of Maryland's Central Maryland Research and Education Center (CMREC) in Clarksville, Maryland, USA, a 375-ha dairy farm that rotationally grazes a cowherd of 13-36 from June to October 2024. I implemented a fixed-effect block design for this project. Three sampling blocks (Fig. 1.2), each containing a replicate of the four land uses (corn-soybean rotation fields (conventional feed), forage pasture, grass margin, and woodlot) were identified based on location throughout the farm. All land use plots used in this study have retained their land use status for more than 30 years. The land use plots were 30m x 100m and located adjacent to other land use plots within the same block (Fig. 1.2).

The conventional corn-soybean rotation plots were planted in corn during this study. Corn varieties included Channel 212-40VT4 PIRB, DKC 65-95RIB, and H4390RC2P. All corn plots were planted directly into bare soil or triticale stubble. Conventional herbicide, pesticide, and fertilizer treatments were applied to plots throughout the sampling period (Table 1.1). No-till practices have been implemented for blocks B and C; however, block A corn field is tilled every 3 years.

Pastures were rotationally grazed throughout this study. Pasture A was planted with a Double Play Mix containing 45% gainer triticale, 33% oats, and 22% annual ryegrass along with

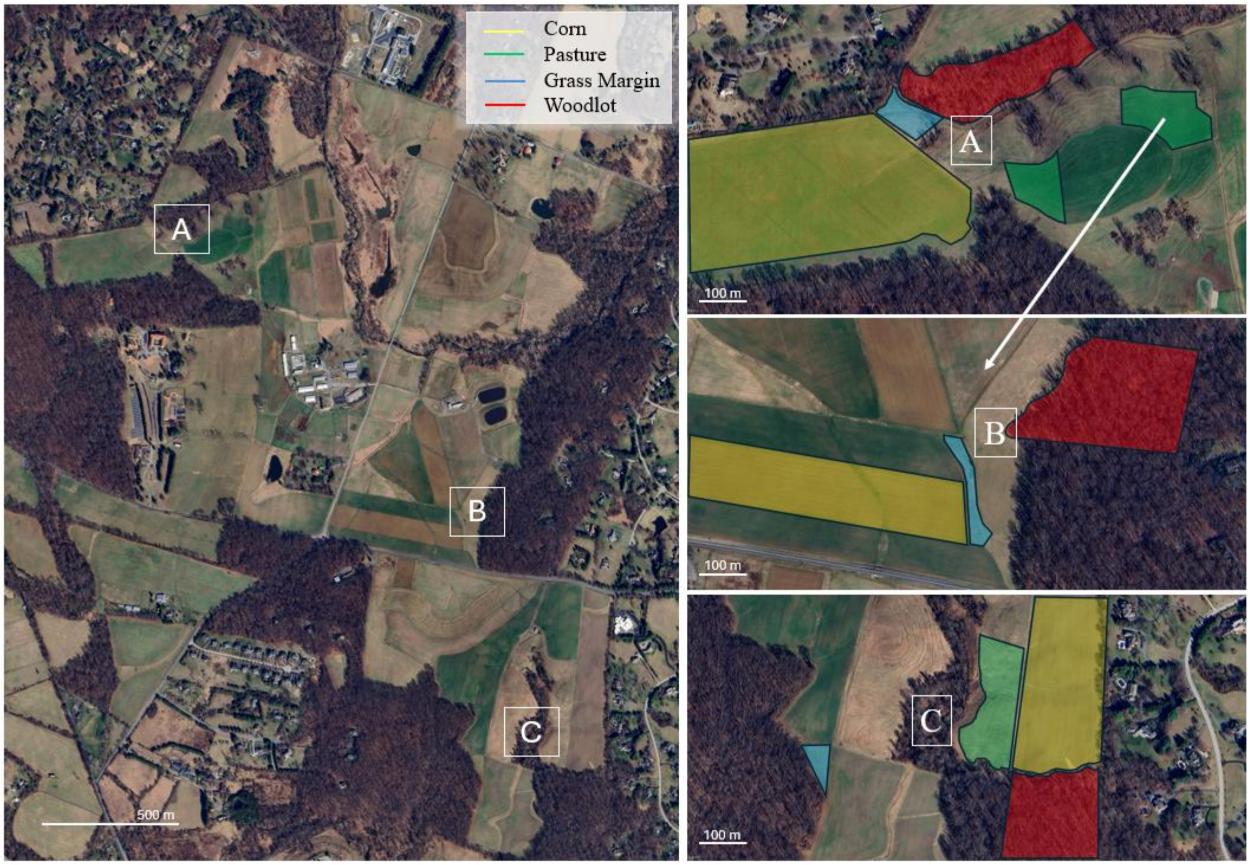


Figure 1.2 Map of the CMREC Clarksville farm in Clarksville, MD with each land use block represented by letters A-C and the distribution of land uses within the three land use blocks. Different colors represent different land uses. Pasture B was situated adjacent to block A due to pasture distribution throughout the farm. Google Earth Pro 7.3.6.10201.

crimson clover. Pastures B and C were overseeded with King's Beefmaster Mix containing 50% soft-leaf tall fescue, 20% perennial ryegrass, 20% leafy orchardgrass, and 10% Alice white clover. In both pastures, Kentucky 31 fescue was present during overseeding. Throughout this study, both annual and perennial forages were present in these systems including tall fescue (*Lolium arundinaceum*), perennial ryegrass (*Lolium perenne*), crimson clover (*Trifolium incarnatum*), and white clover (*Trifolium repens*). Cattle were left in the plot paddock(s) for no longer than five days and each pasture was grazed a minimum of three times throughout this study. Mowing occurred in pastures A and B following grazing and herbicides were applied in pasture C in spring 2024 (Table 1.1). No other management practices were present in forage pastures (Table 1.1).

Maintained grass margins were defined in this study as an area dominated by perennial grasses and mowed to 12-16 centimeters once or twice a month during the summer months (June- September) (Table 1.1). These areas border agricultural fields and woodlots throughout the farm (Figure 1.2). Dominant grasses include smooth brome (*Bromus inermis*), tall fescue (*Lolium arundinaceum*), and bluejoint grass (*Calamagrostis canadensis*).

Woodlots are scattered throughout the farm and are usually found in places that cannot/have not been cultivated in the past. Woodlot plots were consistent with a mesic mixed hardwood forest, dominated by various oak species (*Quercus* spp.), American beech (*Fagus grandifolia*), and tulip poplar (*Liriodendron tulipifera*), which are typical of the overstory. Dominant understory species include multiflora rose (*Rosa multiflora*), northern spicebush (*Lindera bezoiunm*), and basket grass (*Oplismenus hirtles*). Woodlots have remained unlogged for several decades and received no management throughout the past 30+ years (Table 1.1).

Table 1.1 Management practices within each land use at the CMREC Clarksville, MD farm in 2024.

	Corn	Grass Margin	Pasture	Woodlot
Rotation	Corn, Soybean, Triticale	-	-	-
Time Under Management (years)	30+	100+	30-100+	100+
Planting Date	5/22/2024 - 6/3/2024	-	3/16/2023 - 9/1/2023	-
Harvesting/Mowing Date	9/9/2024 - 10/2/2024	June - September (1/month)	June	-
Irrigation	Yes	No	No	-
Fertilizer	60-30-60-15-0.5 dry (5/13/24) 60b N with Herbicides (6/4/24) 60-0-30 side-dressed (6-25-2024 - 7-9-2024)	-	Animal Manure	-
Pesticide	Seed Land use: Acceleron Elite 1250-B-EDC-N or 500-B-EDC	-	-	-
Herbicide	Round-up or Gramoxone, Acuron, Atrazine, Surfactant (6/4/2024) Post sprayed with Round-up or Gramoxone, Status, Surfactant (6/26/2024 - 7/3/24)	-	2qts Crossbow/ac Spring 2024	-
Cover Crop	Triticale	-	-	-
Tillage	Yes	-	-	-
Head/Paddock	-	-	13-40	-
Grazing Frequency	-	-	3-4 per year	-
Grazing Timeline	-	-	21 - 100 days	-

Sampling:

All land use plots were set up on April 25th, 2024. Subterranean arthropod sampling was completed once a month during the cultivation of corn silage in June, July, August, and September, coinciding with a soil property sampling date. Soil property samples were taken every two weeks, ending on October 15th, 2024. Soil temperature was collected every hour throughout the project; however, interference with cattle in the pasture plots and wildlife in the woodlot plots produced generally unreliable data. Nonetheless, I chose to include it here because of its known importance in the status of many soil properties including shaping soil arthropod communities. On September 26th, 2024, hurricane Helene made landfall causing a prolonged rain event for approximately 2 weeks. This weather event was the only major disturbance throughout the project.

Subterranean Pitfall-Trap Sampling

To collect soil arthropods, I deployed a modified subterranean pitfall trap initially used for descriptive studies of soil arthropods and reported by Lopez and Omori (2010). Their initial design was a hollow cylinder, 75cm in length, with an 11cm diameter. Many small holes (5-7cm) were drilled along its surface to allow arthropods to enter the tube. They placed a collection cup in the bottom with a preservative to collect arthropods that entered the trap. To deploy the trap, the cylinder is buried until all holes are covered and a cap is placed on top to prevent water from entering the trap. The traps are baited with lindberger cheese to attract detritivore and saprophagous soil arthropods.

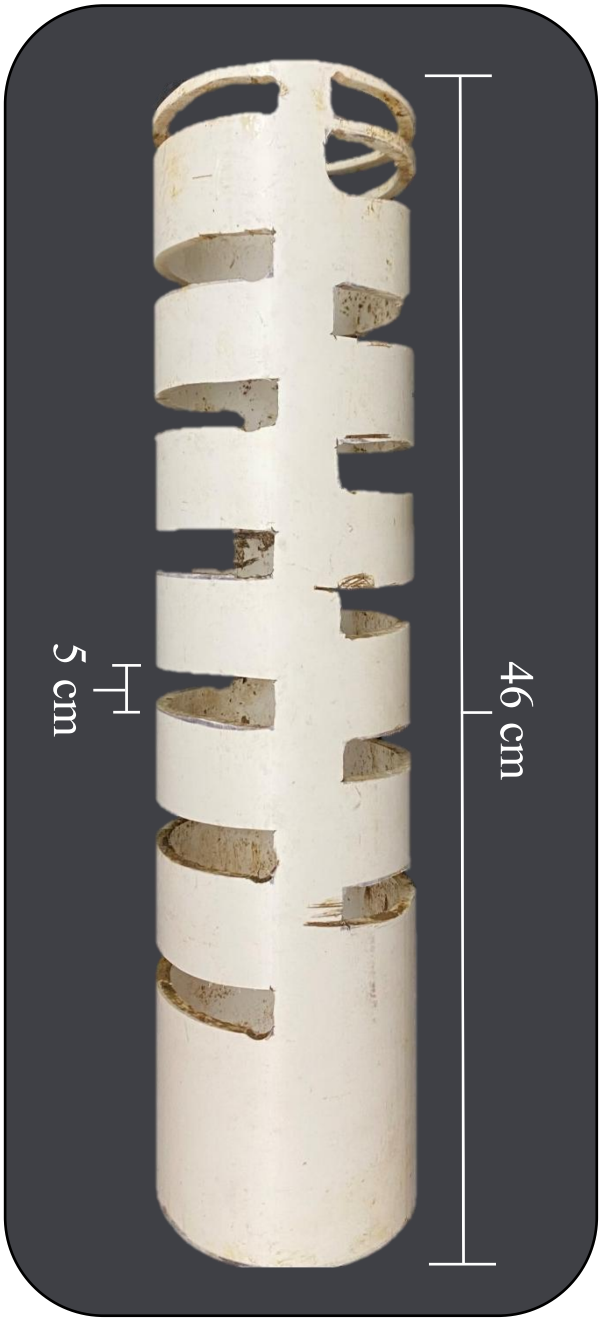


Figure 1.3 Subterranean pitfall trap design

The design utilized in this study was made from PVC and was slightly shorter at 45cm long while preserving the original 11cm diameter (Fig. 1.3). Instead of several small holes drilled around the outside of the cylinder, I cut horizontal slits approximately 5cm wide, half the circumference of the cylinder, and alternated those slits to the bottom of the cylinder, leaving approximately 10cm for the collection cup at the bottom. I also deployed a collection cup at the soil surface (similar to a normal pitfall trap). I began alternating the 5cm slits directly below a 1-2cm section used to hold the top sampling cup. To install the traps, I used a gas-powered soil auger to drill the initial holes. The auger diameter was approximately 15cm. I then placed the traps in the soil and covered them flush with the soil surface. These traps were baited with sardines (also commonly used as attractants in studies of soil arthropods). Packets of sardines in cheesecloth were suspended in the center of the trap using a metal hook pushed into the soil. Traps were set for 2 weeks at a time and collected four times throughout the growing season (June, July, August, and September). At the time of collection, top and bottom samples were recovered separately if possible, and placed in 240ml Corning Snap-Seal containers. All traps were installed in April to allow the surrounding soil to settle before commencing this study. At the time of collection, all contents of the sampling cup were placed in a collection container and transferred to the lab for immediate cleaning.

Due to high levels of soil entering the traps, all samples were cleaned before processing. To clean a sample, I first submerged the sample in approximately 1 liter of water along with 5-10 drops of mineral oil. Samples were then shaken vigorously until all soil was equally distributed throughout the water and the sample was allowed to rest for 1-2 minutes. Because of the non-polar hydrocarbons found in both mineral oil and arthropod exoskeletons, along with the difference in density of mineral oils and water, I was able to separate soil arthropods from the soil in these samples (Bouyoucos 1962). All arthropods that floated to the surface were strained off using a fine sieve and preserved in 80% ethanol until sample processing. The remaining soil-water-oil mixture was then spread out in a white tray to manually extract all arthropods missed during the separation. Usually, only large arthropods were left behind along with isopods which could be separated easily from the solution.

Following processing, arthropods were sorted into taxonomic groups using a Nikon SMZ800 dissecting microscope at 10x, counted, and identified to subclass, order, or family (Hennen and Brown 2021; Triplehorn & Johnson 2004). For analysis, any arthropod less than 2mm was considered a micro-arthropod (Acari, Collembola, Diplura, Ptillidae), while all other taxa were grouped as macro-arthropods.

Soil Biological Quality – arthropod index

The soil biological quality arthropod index (QBS-ar) is a method developed by Parisi et al., (2005) and recently revised by Menta et al., (2018). The ecological conditions of the soil can be investigated using soil arthropod communities because of their high abundance, role in soil formation, and life history (Parisi et al., 2005). The index provides a relatively quick assessment of the soil arthropod communities and can quantify soil health based on the soil arthropod presence. QBS-ar relies on the concept that soil arthropods with higher vulnerability will be present in soils with better conditions such as low degradation, pollution, and compaction. To develop this index, major taxa are assigned an ecomorphological index score (EMI) based upon the presence of characteristics such as short legs, short antennae, lack of pigmentation, and thinner cuticles. In all, these characteristics indicate a group's adaptation to general or specific soil environments. To calculate the QBS-ar for a single sample, arthropods are identified to the order/class level and an EMI value is assigned. Each taxon is only counted once (presence) and the sum of these scores is the QBS-ar score for that sample.

Soil Bulk Density and Volumetric Water Content

Soil bulk density and volumetric water content were measured using soil cores. All above-ground vegetation and organic matter were removed from the surface before sampling. A metal ring (with a volume of 98.17 cm^3) was hammered flush into the ground at eight locations within the plots. Locations were determined by splitting plots into 8 equal sections and haphazardly sampling within each of the 8 sections. Sampling was completed every two weeks from May to October 2024, totaling 13 sampling dates. Soil was removed from sampling rings and placed in an airtight 240ml Corning Snap-Seal container in the field. In the lab, samples were weighed, dried in an oven at 105°C for 24 hours, and weighed again to calculate soil bulk density and volumetric water content.

Bulk density (g/cm^3) was determined using the following equation:

$$\boldsymbol{Bulk\ Density} = \frac{\mathrm{Wd}}{\mathrm{V}}$$

W_d = Dry weight of the sample (g)
V = Volume of the sample (cm^3)

Volumetric water content (%) was determined using the following formula:

$$\boldsymbol{Volumetric\ Water\ Content} = \left(\frac{\mathrm{Vw}}{\mathrm{Vt}}\right) \text{ x } 100$$

V_w = Volume of soil water (cm^3)
V_t = Total sample volume (cm^3)

Soil Temperature

Variation in daily soil temperature (range) was measured throughout the entirety of the study. iButton Link devices (iButtonLink Technology, Model: DS1921G-F5# Thermochron, 4K) were used to measure temperature every hour from April to October in two locations within a land use plot. Each sampling location had two measurements, one at 15cm and one at 1-2cm below the soil surface, totaling 4 iButton Link devices per land use plot. Sampling locations were 30m and 90m along the length of the land use plots. Vegetation structure was left intact at the measurement sites. The daily range for soil temperature was calculated by subtracting the lowest temperature from the highest temperature during the 24-hour period. The mean temperature range for each land use type was calculated daily (176 days).

Soil Nutrient Analysis

Soil samples for nutrient profiles, soil type, soil organic matter, and soil pH were collected to a depth of 21 cm in December 2024 using a soil auger with a diameter of 5.1 cm. To ensure representative samples, four samples were taken at evenly spaced intervals along the length of each land use plot. The cores from each plot were combined into a single composite sample. Composite samples were air-dried and sent to A&L Great Lakes Laboratories Inc. (Fort Wayne, IN) for analysis, following the procedures outlined by Nathan and Gelderman (2015). Samples were dried overnight at 40 °C, ground with an Agvise flail-type grinder, and run through a 2 mm mesh sieve. Loss on ignition at 360°C was used to determine organic matter content. Soil pH was measured using a 1:1, soil:water slurry. Nutrient concentrations, including phosphorus (P), potassium (K), magnesium (Mg), and calcium (Ca) were assessed using Mehlich III extraction followed by ICP analysis.

Plant Coverage and Diversity

Plant coverage and diversity were collected two times, once in June and July using the modified Daubenmire method (Coulloudon et al., 1999). A 46cm x 46cm frame was randomly tossed ten times within a land use plot. All plant species were identified to species, and their density was quantified using a cover class scale (0-5%, 5-25%, 25-50%, 50-75%, 75-95%, 95-100%). Plots containing complex vegetation structures had the potential to have coverage scores over 100% due to multiple canopy layers. The canopy layer was estimated based on observations directly above the frame. Plant diversity was calculated using the Shannon diversity index. To calculate Shannon diversity for each sample, cover classes were converted into percentages based on the mean percentage for the cover class. For example, a cover class ranging from 0%-5% would receive a percentage score of 2.5%.

Data Analysis

Data analysis was conducted using R Statistical Software version (v4.4.1; R Core Team 2024) using the lme4, car, emmeans, vegan, and BiodiversityR packages (see R package citations in references). I tested for normality using the Shapiro-Wilk normality test. Arthropod counts were transformed using log(x + 1) transformation. For variables, including arthropod abundances, volumetric water content, and soil biological quality - arthropod index, I used linear mixed-effects models (LMER). These models were used for variables that met the assumptions of normality and homoscedasticity. I tested for the significance of fixed effects (land use, month, and their interaction) using Type III ANOVA tables derived from the fitted models. Following ANOVA, post hoc comparisons were performed through calculations of the estimated marginal means (least-squares means) for each factor level, adjusted for other terms in the model.

Variables that did not meet normality and homoscedasticity assumptions (soil bulk density, soil temperature ranges, plant diversity, and plant coverage) following model fitting were evaluated using a Kruskal-Wallis test comparing means between land use and date. For pairwise comparisons of non-normal data, the Wilcoxon rank-sum test was used with Bonferroni correction.

Differences in community composition were evaluated through both permutational multivariate analysis of variance (PERMANOVA) based on Bray-Curtis dissimilarity and pairwise PERMANOVA based on Bray-Curtis dissimilarity. I visualized similarities between the soil arthropod community samples using non-metric multidimensional scaling (NMDS) with BrayCurtis distance. Rényi diversity profiles were used to compare diversity between land uses, as they allow for direct comparison of several common diversity metrics. Rényi diversity profiles were obtained using 1000 permutations with the smallest sample size to determine margins of error and testing at $\alpha = 0, 0.25, 0.5, 1, 2, 4, 8$, and infinity. A redundancy analysis (RDA) was performed to examine the relationships between the most abundant taxa collected here and environmental variables.

Results

Soil Arthropods

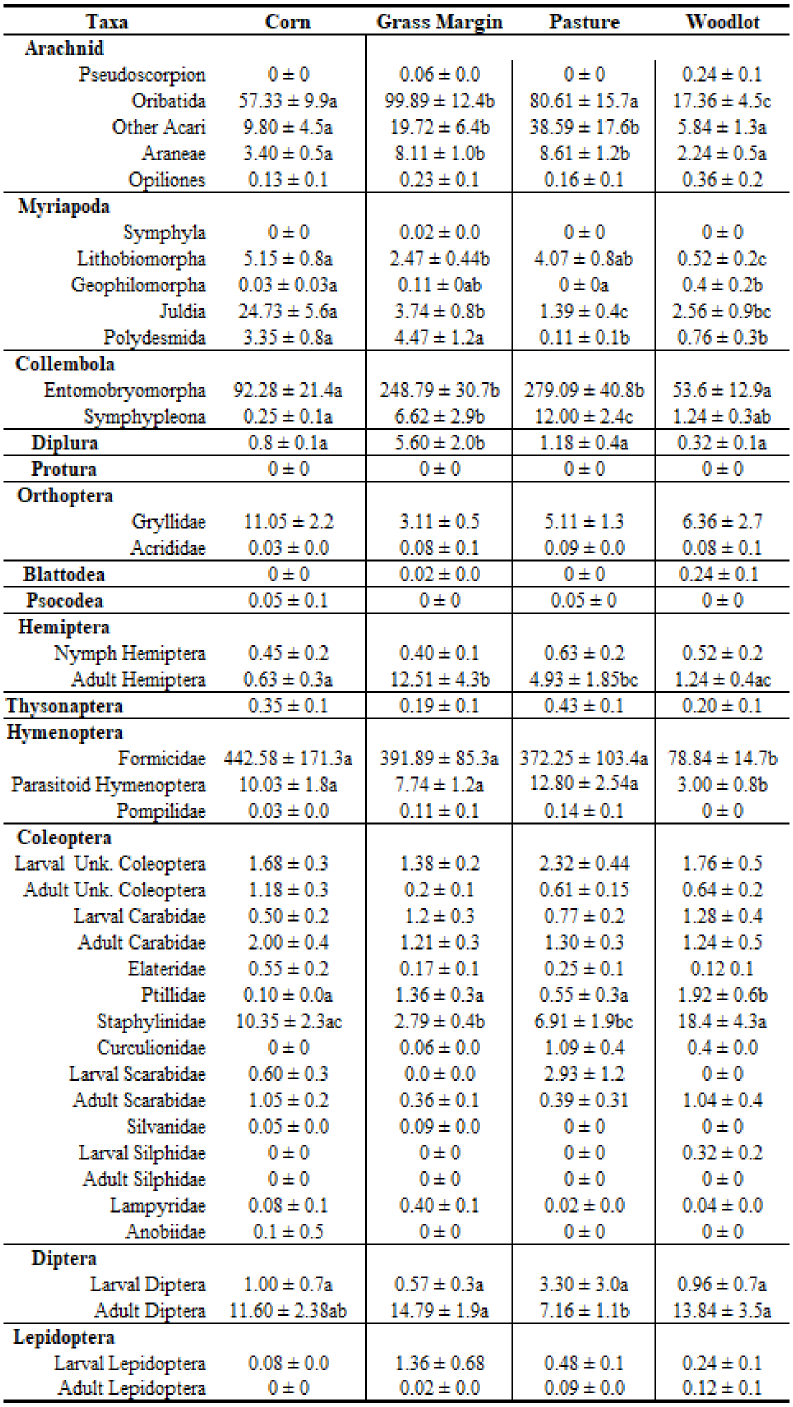
Table 1.2 Mean (±S.E.) values of biotic and abiotic soil properties in the four land use types at the CMREC Clarksville Farm collected in 2024. Within each row, values followed by different letters are significantly different from each other. Differences in parametric data were tested using estimated marginal means, while differences in nonparametric data were assessed using the Wilcoxon Rank Sum test with Bonferroni correction for multiple comparisons (n = 12).

Soil Property	Corn	Grass Margin	Pasture	Woodlot	p - value
Bulk Density (g/cm^{-3})	1.14 ± 0.013^{A}	0.92 ± .008^{AB}	0.94 ± 0.015^{AB}	0.81 ± 0.017^{B}	< 0.05
Volumetric Water Content (%)	20.15 ± 0.98^{A}	34.96 ± 1.38^{B}	30.86 ± 1.78^{B}	35.94 ± 1.50^{B}	< 0.05
Soil 15cm Temperature Range	4.2 ± 0.13^{A}	2.63 ± 0.08^{AB}	1.72 ± 0.06^{B}	3.05 ± 0.11^{AB}	< 0.05
Plant Density	20.33 ± 5.82^{A}	96.58 ± 4.02^{B}	71.63 ± 8.54^{C}	215.23 ± 3.32^{D}	< 0.08
Plant Shannon Diversity	0.042 ± 0.021^{A}	0.684 0.138^{B}	0.446 0.081^{B}	1.458 0.048^{C}	< 0.03
QBS-ar	78.43 ± 3.26^{A}	118.17 ± 2.08^{B}	99.79 ± 4.04^{B}	107.28 ± 9.29^{B}	< 0.01
Meso-arthropods per trap	157.44 ± 36.91^{A}	378.26 ± 26.18^{B}	409.10 ± 60.66^{B}	93.74 ± 22.95^{A}	< 0.001
Macro-arthropods per trap	533.9 ± 174.3^{AB}	467.4 ± 86.1^{B}	441.2 ± 105.1^{B}	144.2 ± 24.3^{A}	< 0.05
Total arthropods per trap	683.15 ± 197.67^{A}	849.15 ± 87.46^{A}	853.96 ± 147.33^{A}	242.77 ± 40.44^{B}	< 0.03

Subterranean Pitfall Traps

Total arthropod abundance differed significantly between the four land uses (ANOVA, $F_{(3,39)} = 10.93$, $p < 0.001$, $n = 12$). I was able to identify temporal variation in total arthropod abundance between sampling dates (ANOVA, $F_{(3, 39)} = 2.99$, $p < 0.05$, $n = 48$). Average micro-arthropod abundance differed significantly between the four land uses (ANOVA, $F_{(3, 39)} = 20.27$, $p < 0.001$, $n = 12$). Acari and Collembola comprised most of the arthropods captured in this study (Table 1.3). Sample abundances for Acari and Collembola ranged from 0 to 1282 and 0 to 1437 respectively. The average macro-arthropod abundance also differed significantly between the four land uses (ANOVA, $F_{(3, 39)} = 20.27$, $p < 0.001$, $n = 12$) (Table 1.2). Macro-arthropod abundance was dominated by Formicidae, Coleoptera, and Diptera, with individual maximums per sample of 6639, 165, and 73, respectively. Chilopoda: Lithobiomorpha and Diplopoda: Juldia were most abundant in corn plots (Table 1.3).

Table 1.3 Mean (±S.E.) abundance of the major soil arthropod taxa collected using subterranean pitfall traps in four land use types at CMREC in 2024. Within each row, values followed by different letters are significantly different at the α = 0.05



Taxa	Corn	Grass Margin	Pasture	Woodlot
Arachnid				
Pseudoscorpion	0 ± 0	0.06 ± 0.0	0 ± 0	0.24 ± 0.1
Oribatida	57.33 ± 9.9a	99.89 ± 12.4b	80.61 ± 15.7a	17.36 ± 4.5c
Other Acari	9.80 ± 4.5a	19.72 ± 6.4b	38.59 ± 17.6b	5.84 ± 1.3a
Araneae	3.40 ± 0.5a	8.11 ± 1.0b	8.61 ± 1.2b	2.24 ± 0.5a
Opiliones	0.13 ± 0.1	0.23 ± 0.1	0.16 ± 0.1	0.36 ± 0.2
Myriapoda				
Symphyla	0 ± 0	0.02 ± 0.0	0 ± 0	0 ± 0
Lithobiomorpha	5.15 ± 0.8a	2.47 ± 0.44b	4.07 ± 0.8ab	0.52 ± 0.2c
Geophilomorpha	0.03 ± 0.03a	0.11 ± 0ab	0 ± 0a	0.4 ± 0.2b
Juldia	24.73 ± 5.6a	3.74 ± 0.8b	1.39 ± 0.4c	2.56 ± 0.9bc
Polydesmida	3.35 ± 0.8a	4.47 ± 1.2a	0.11 ± 0.1b	0.76 ± 0.3b
Collembola				
Entomobryomorpha	92.28 ± 21.4a	248.79 ± 30.7b	279.09 ± 40.8b	53.6 ± 12.9a
Symphypleona	0.25 ± 0.1a	6.62 ± 2.9b	12.00 ± 2.4c	1.24 ± 0.3ab
Diplura	0.8 ± 0.1a	5.60 ± 2.0b	1.18 ± 0.4a	0.32 ± 0.1a
Protura	0 ± 0	0 ± 0	0 ± 0	0 ± 0
Orthoptera				
Gryllidae	11.05 ± 2.2	3.11 ± 0.5	5.11 ± 1.3	6.36 ± 2.7
Acrididae	0.03 ± 0.0	0.08 ± 0.1	0.09 ± 0.0	0.08 ± 0.1
Blattodea	0 ± 0	0.02 ± 0.0	0 ± 0	0.24 ± 0.1
Psocodea	0.05 ± 0.1	0 ± 0	0.05 ± 0	0 ± 0
Hemiptera				
Nymph Hemiptera	0.45 ± 0.2	0.40 ± 0.1	0.63 ± 0.2	0.52 ± 0.2
Adult Hemiptera	0.63 ± 0.3a	12.51 ± 4.3b	4.93 ± 1.85bc	1.24 ± 0.4ac
Thysonaptera	0.35 ± 0.1	0.19 ± 0.1	0.43 ± 0.1	0.20 ± 0.1
Hymenoptera				
Formicidae	442.58 ± 171.3a	391.89 ± 85.3a	372.25 ± 103.4a	78.84 ± 14.7b
Parasitoid Hymenoptera	10.03 ± 1.8a	7.74 ± 1.2a	12.80 ± 2.54a	3.00 ± 0.8b
Pompilidae	0.03 ± 0.0	0.11 ± 0.1	0.14 ± 0.1	0 ± 0
Coleoptera				
Larval Unk. Coleoptera	1.68 ± 0.3	1.38 ± 0.2	2.32 ± 0.44	1.76 ± 0.5
Adult Unk. Coleoptera	1.18 ± 0.3	0.2 ± 0.1	0.61 ± 0.15	0.64 ± 0.2
Larval Carabidae	0.50 ± 0.2	1.2 ± 0.3	0.77 ± 0.2	1.28 ± 0.4
Adult Carabidae	2.00 ± 0.4	1.21 ± 0.3	1.30 ± 0.3	1.24 ± 0.5
Elateridae	0.55 ± 0.2	0.17 ± 0.1	0.25 ± 0.1	0.12 0.1
Ptillidae	0.10 ± 0.0a	1.36 ± 0.3a	0.55 ± 0.3a	1.92 ± 0.6b
Staphylinidae	10.35 ± 2.3ac	2.79 ± 0.4b	6.91 ± 1.9bc	18.4 ± 4.3a
Curculionidae	0 ± 0	0.06 ± 0.0	1.09 ± 0.4	0.4 ± 0.0
Larval Scarabidae	0.60 ± 0.3	0.0 ± 0.0	2.93 ± 1.2	0 ± 0
Adult Scarabidae	1.05 ± 0.2	0.36 ± 0.1	0.39 ± 0.31	1.04 ± 0.4
Silvanidae	0.05 ± 0.0	0.09 ± 0.0	0 ± 0	0 ± 0
Larval Silphidae	0 ± 0	0 ± 0	0 ± 0	0.32 ± 0.2
Adult Silphidae	0 ± 0	0 ± 0	0 ± 0	0 ± 0
Lampyridae	0.08 ± 0.1	0.40 ± 0.1	0.02 ± 0.0	0.04 ± 0.0
Anobiidae	0.1 ± 0.5	0 ± 0	0 ± 0	0 ± 0
Diptera				
Larval Diptera	1.00 ± 0.7a	0.57 ± 0.3a	3.30 ± 3.0a	0.96 ± 0.7a
Adult Diptera	11.60 ± 2.38ab	14.79 ± 1.9a	7.16 ± 1.1b	13.84 ± 3.5a
Lepidoptera				
Larval Lepidoptera	0.08 ± 0.0	1.36 ± 0.68	0.48 ± 0.1	0.24 ± 0.1
Adult Lepidoptera	0 ± 0	0.02 ± 0.0	0.09 ± 0.0	0.12 ± 0.1

Rényi Diversity Profile

The diversity profile indicates differences in diversity across the four land use types as more weight is given to dominant species (family richness at $\alpha = 0$, Shannon diversity at $\alpha = 1$, Simpson index at $\alpha = 2$, etc.) (Fig 1.4). Across all scales, the woodlot plots exhibit the highest diversity, with consistently greater values compared to the other land uses. Conversely, the lowest diversity was within corn plots, indicating lower taxa richness and greater dominance of a few species. Grass margin and pasture land uses display intermediate diversity values. Although there are differences in diversity between the four land use types as alpha increases, further statistical analysis using the Kruskal-Wallis rank sum test revealed no significant difference in diversity.

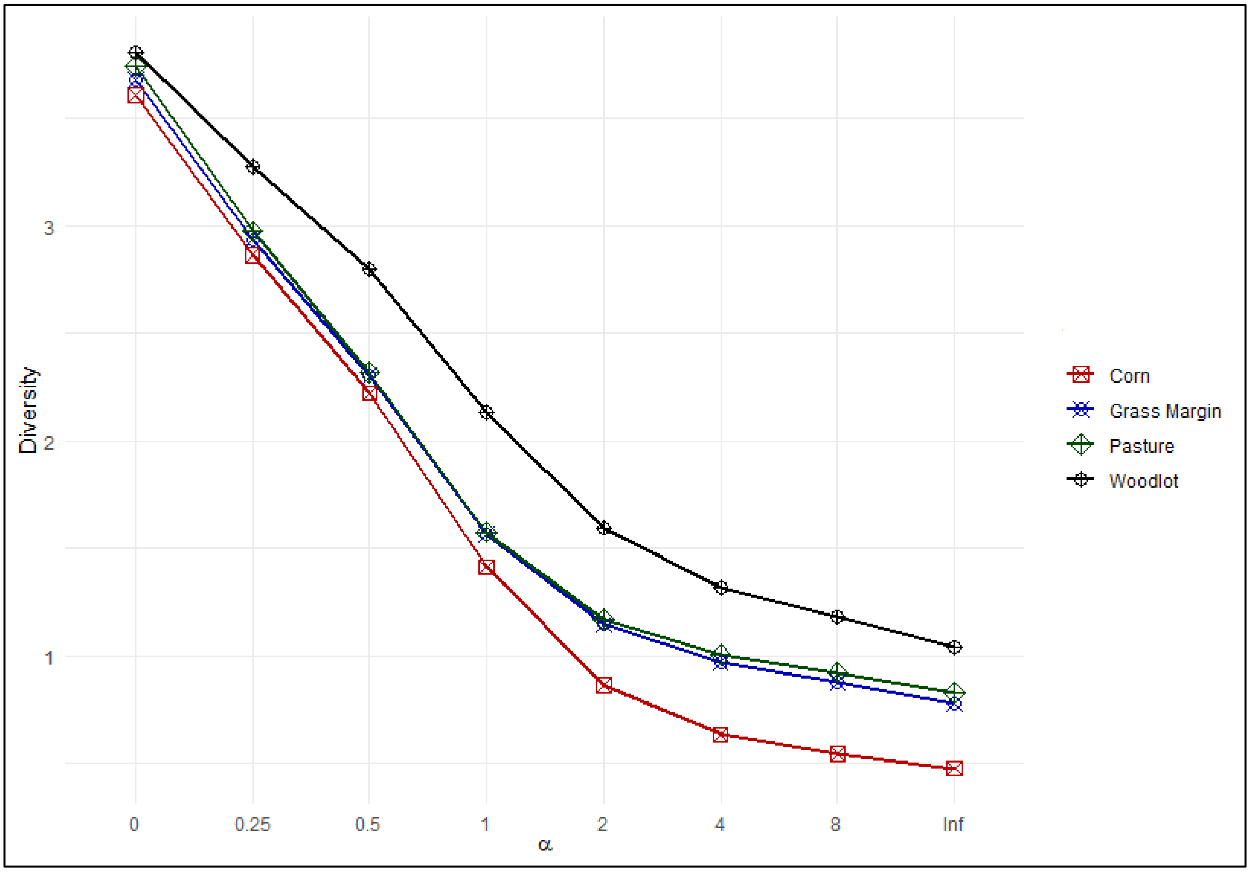


Figure 1.4 Rényi diversity profile comparing the four land use types from subterranean pitfall trap samples collected at CMREC in 2024. This graph compares Hill numbers to assess multiple diversity metrics at once (family richness at α = 0, Shannon diversity at α = 1, Simpson index at α = 2, etc.).

Non-Metric Multidimensional Scaling

The NMDS plot of soil arthropod communities within the four land uses showed a separation of woodlot and corn plots from pasture and grass margin plots (Fig. 1.5). The convex hulls outline the outermost points for each of the land use types in the ordination space. Using these hulls indicates the range in variation in community composition. The final stress value of the NMDS was 0.077, indicating a good fit of the model to the data.

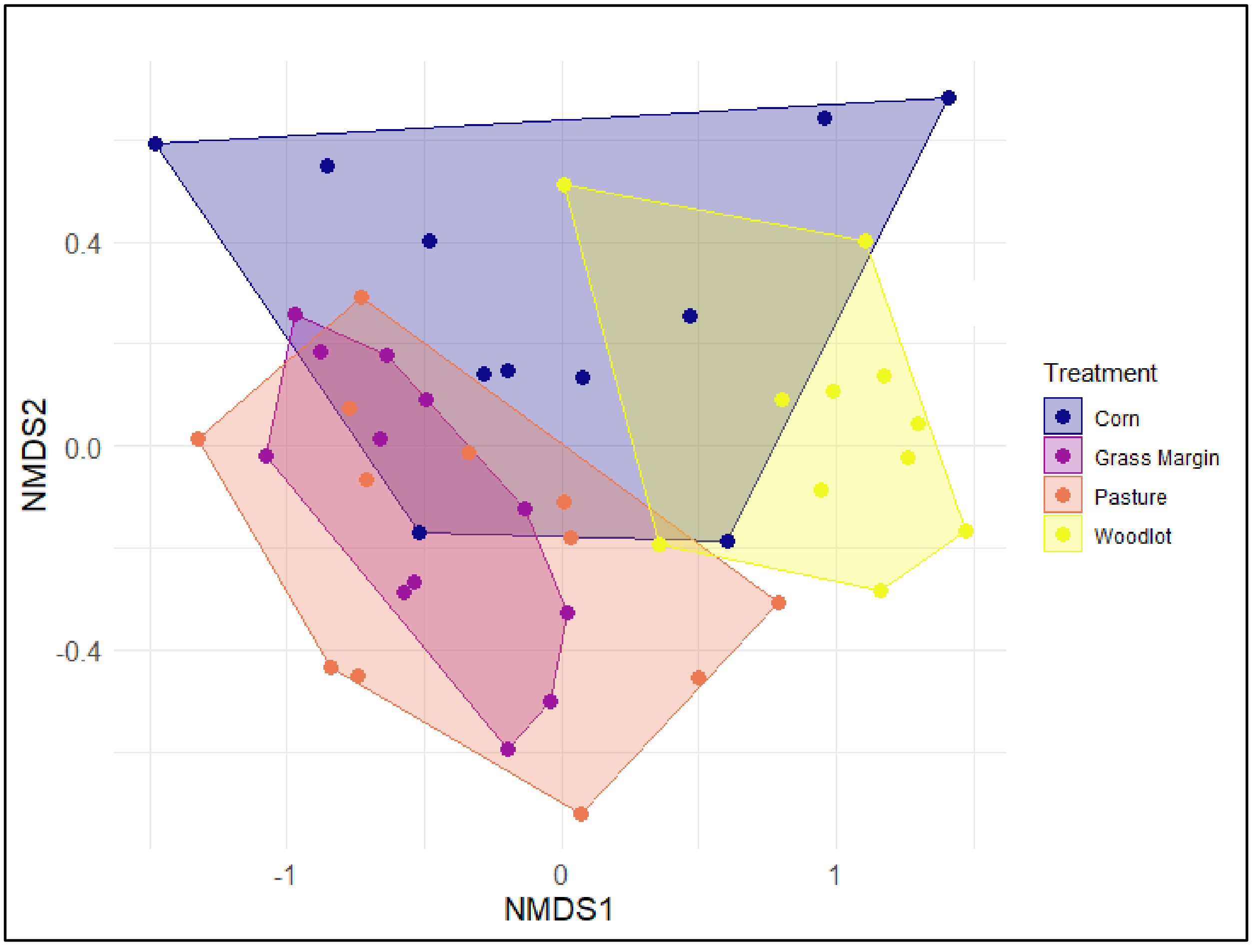


Figure 1.5 NMDS plot of subterranean trap samples taken from the CMREC farm in 2024 grouped by study land use type (stress = 0.077)

Soil Properties

Soil Biological Quality – arthropod index

Individual Soil Biological Quality – arthropod index values ranged from 45 to 174. Soil biological quality differed significantly between the four land uses (ANOVA, $F_{(3,8)} = 15.12$, $p < 0.01$, $n = 12$) (Fig. 1.6). Corn plots averaged the least while grass margins averaged the highest (Table 1.2). Standard error in the Soil Biological Quality - arthropod index was consistent between corn, grass margin, and pasture land uses; however, standard error in the woodlot land uses QBS-ar scores was almost 2 times as high. This indicates higher within-habitat heterogeneity in woodlots soil arthropod communities.

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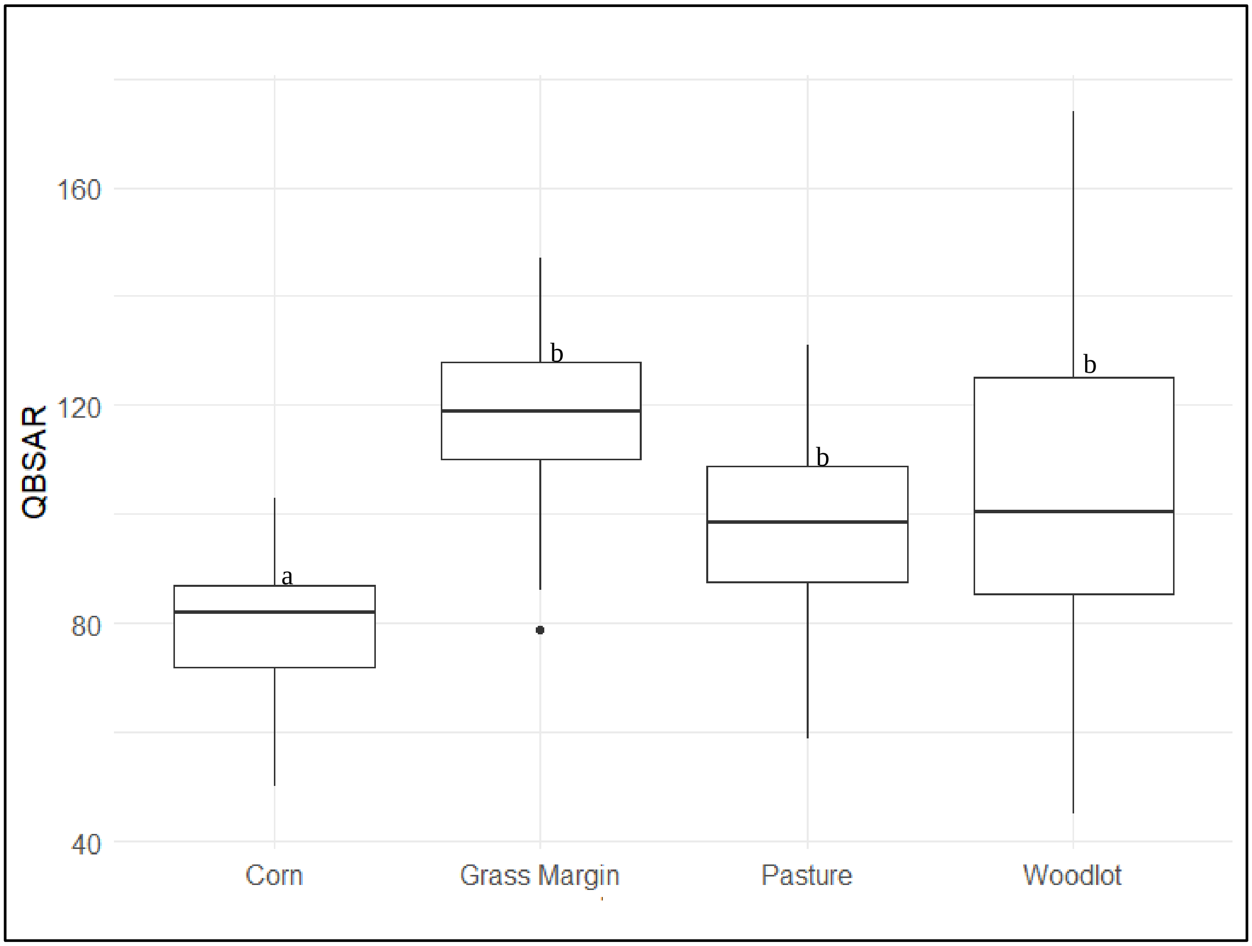

Figure 1.6 Box plot of the soil biological quality – arthropod index for all land use types throughout the four sampling dates at CMREC in 2024. Plots with different letters indicate that the mean soil biological quality – arthropod index scores are different from one another at the α = 0.05 significance level.

Soil Bulk Density

When comparing samples throughout the entire sampling period, the average soil bulk density differed between the four land uses (Kruskal-Wallis, $\chi^2_{(3)} = 8.69$, $p < 0.05$, $n = 12$) (Fig. 1.7). Dunn's post hoc test revealed that bulk density in the woodlots were different from that in Corn ($Z = 2.94$, $p < 0.05$) (Table 1.2). No other pairwise comparisons were statistically significant after Bonferroni correction. There were no significant differences in bulk density throughout the 13 sampling dates. (Kruskal-Wallis, $\chi^2_{(12)} = 8.07$, $p > 0.05$, $n = 156$) (Figure 1.8).

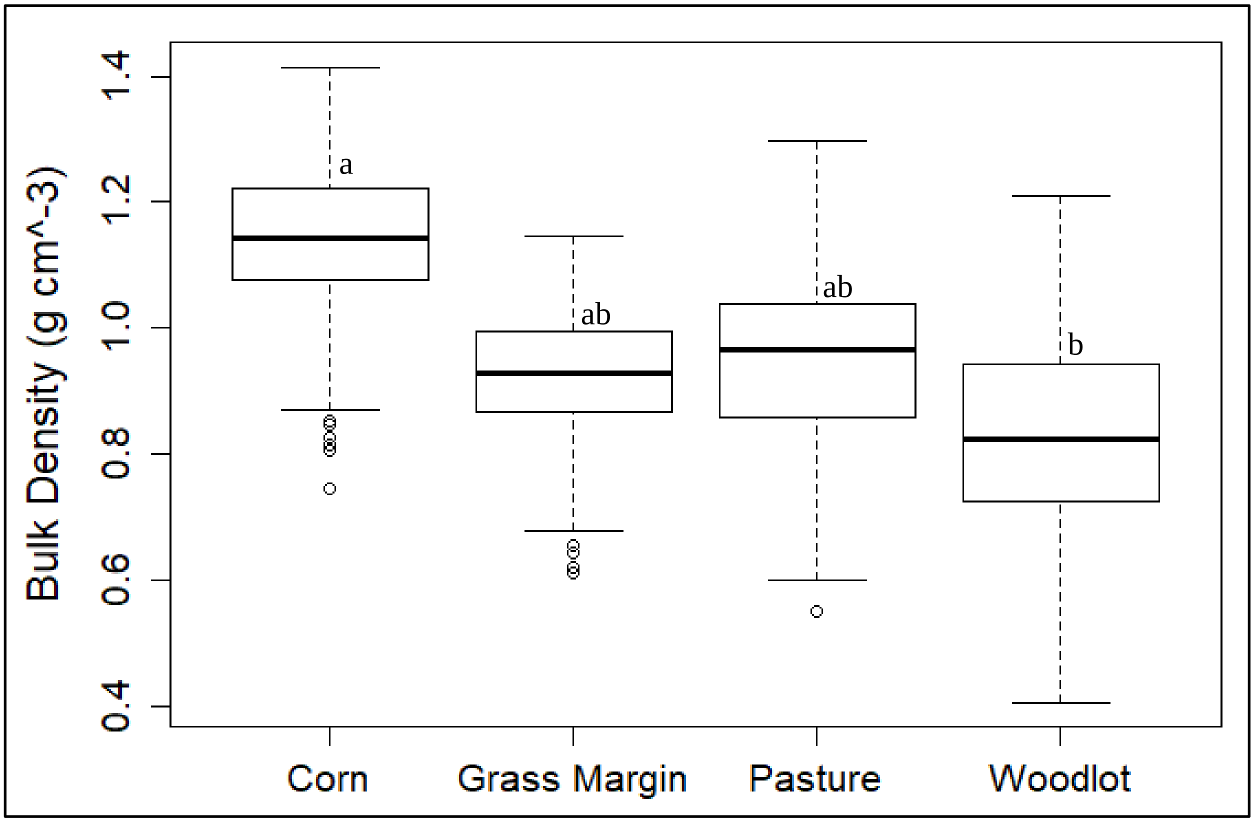

Figure 1.7 Box plot of soil bulk density for each land use type sampled at CMREC in 2024. Samples were taken using a 97.13 cm^3 sampling ring. Plots with different letters indicate differences in mean bulk densities at the $\alpha = 0.05$ significance level.

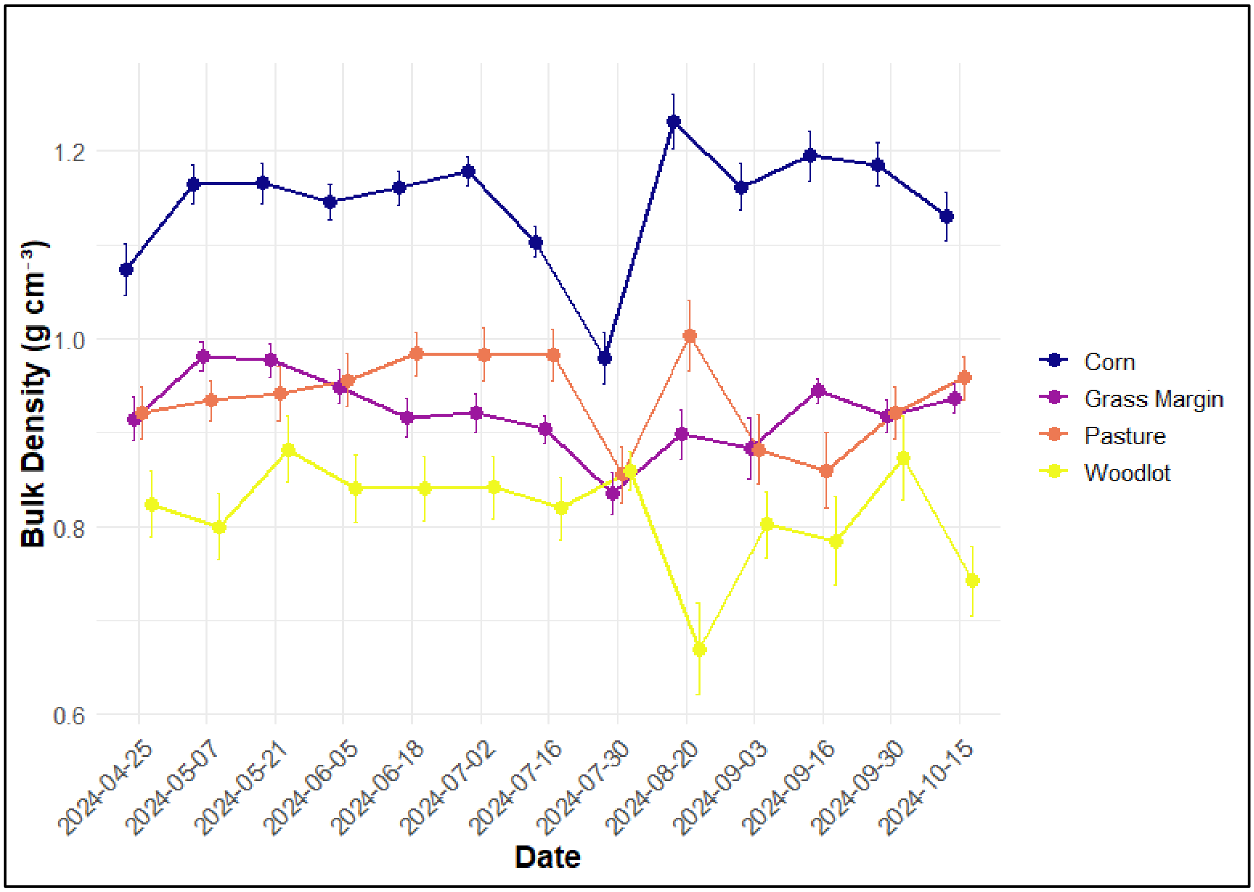

Figure 1.8 Line graph of average soil bulk density (± SE) for the four land use types across all 13 sampling dates from May to October 2024 at CMREC.

Volumetric Water Content

I found significant differences in volumetric water content among the four land uses (ANOVA, $F_{(3, 8)} = 8.69$, $p < 0.001$, $n = 12$) (Fig. 1.9). There was no significant difference between the average volumetric water content between grass margin, pasture, and woodlot land uses, which all possessed higher averages than corn plots (Table 1.2). Date had no significant main effect on volumetric water content (ANOVA, $F_{(3, 143)} = 16.63$, $p > 0.05$, $n = 156$) with distinct peaks in May, August, and October (Fig. 1.10). There was no significant interaction between land use and date for volumetric water content.

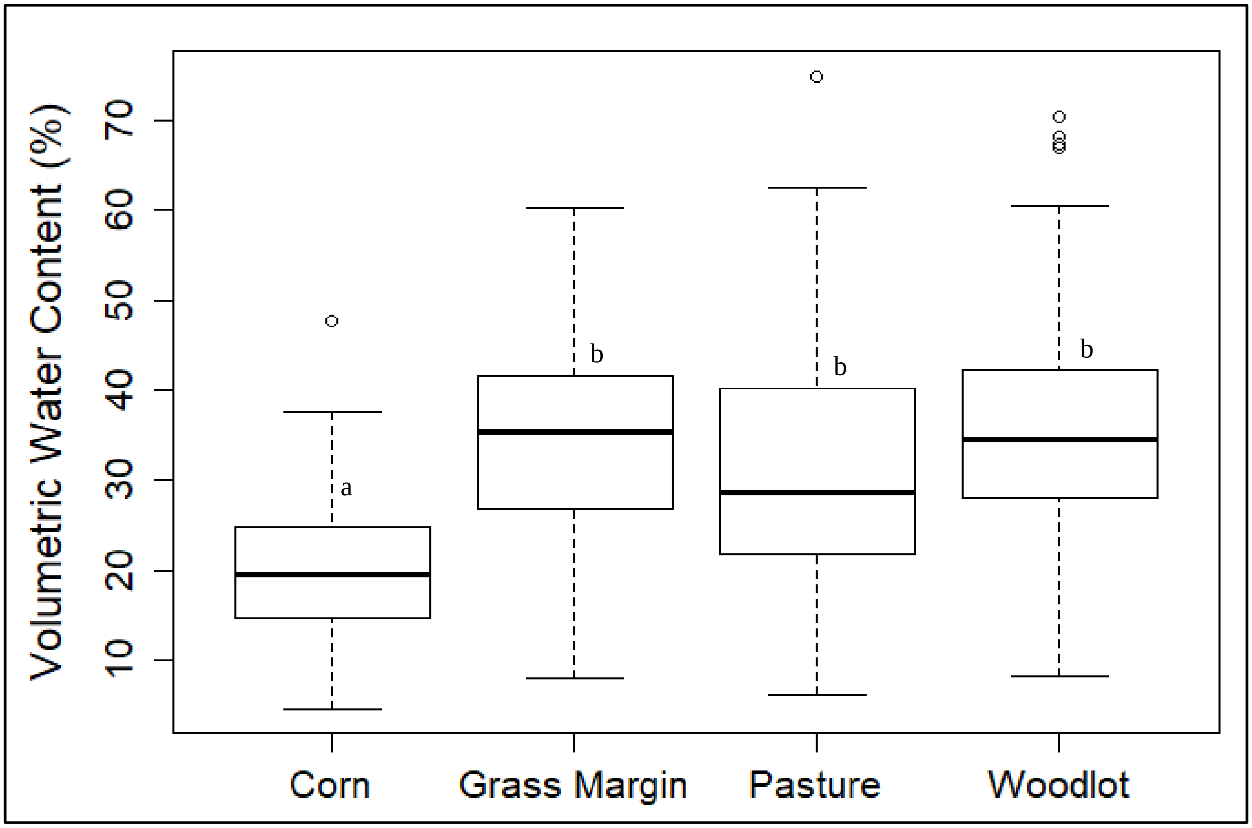

Figure 1.9 Box plot of volumetric water content for each land use type sampled at CMREC in 2024. Samples were taken using a 97.13 cm^3 sampling ring ($n = 156$). Plots with different letters indicate differences in mean volumetric water content at the $\alpha = 0.05$ significance level.

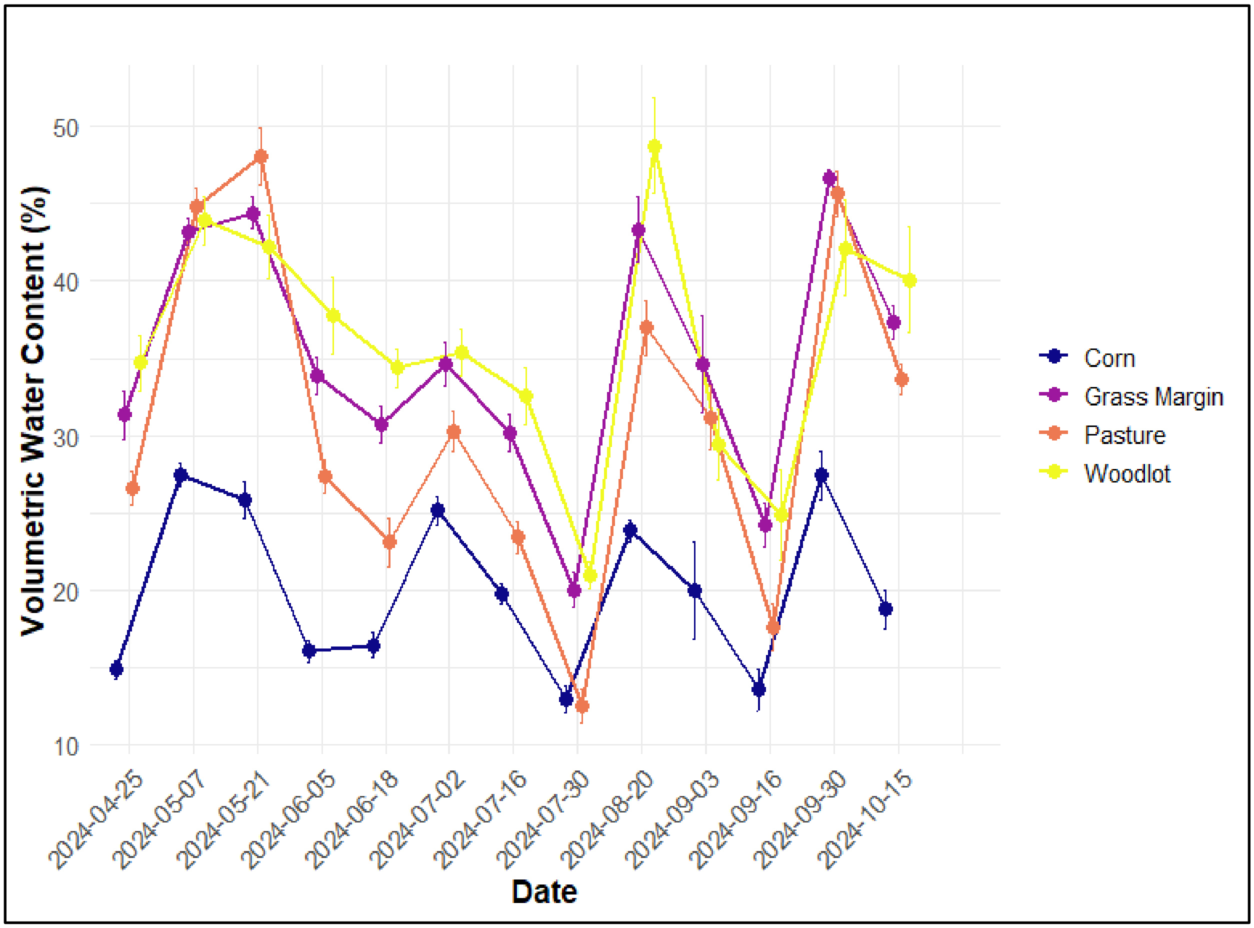

Figure 1.10 Line graph of average volumetric water content (± SE) for the four land use types across all 13 sampling dates from May to October 2024 at CMREC.

Soil Temperature

Soil temperature was measured at 15cm below the soil surface in all plots. The range in daily soil temperature is reported here. There were significant differences between the daily average range in soil temperature between land uses at 15cm depth (Kruskal-Wallis, $\chi^2_{(3)} = 8.13$, $p < 0.05$, $n = 12$) (Fig. 1.11). I did not investigate differences in the surface level measurements (2cm) because of high disturbances with sensors in the field. The average range at 15cm was highest in corn plots and lowest in pastures (Table 1.2).

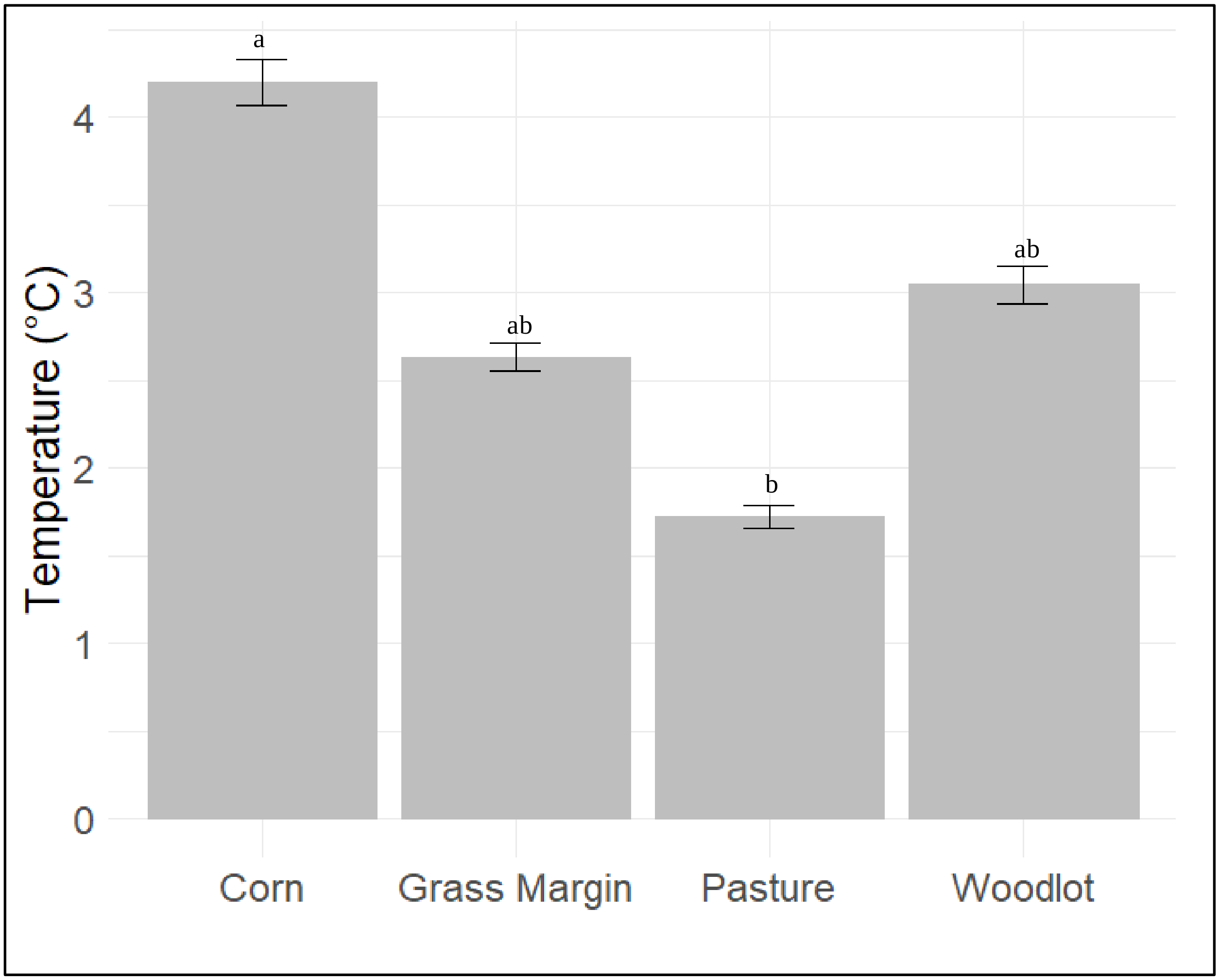

Figure 1.11 Average daily range in soil temperature at 15 cm for each land use type across all 13 sampling dates at CMREC in 2024. Bars with different letters have means that are significantly different from each other at the α = 0.05 significance level.

Soil Nutrient Analysis

Soil nutrient analysis revealed that pasture plots had the highest organic matter and sand percentage, along with the highest phosphorous, potassium, magnesium, and CEC levels (Table 1.4).

Table 1.4 Mean (±SE) values for nutrient soil properties (0–5 cm depth) in four land use types at CMREC in December 2024.

Soil Properties	**Corn**	**Grass Margin**	**Pasture**	**Woodlot**
Organic matter (%)	2.73 ± 0.09	3.5 ± 0.7	4.2 ± 0.3	3.3 ± 0.7
Phosphorus (mg/kg)	42 ± 18.9	45.3 ± 20.7	52 ± 9.8	8.0 ± 3.6
Potassium (mg/kg)	141.0 ± 16.2	255.7 ± 2.2	267.0 ± 65.4	113.3 ± 16.2
Magnesium (mg/kg)	113.3 ± 27.7	141.± 7 15.9	145.0 ± 5.8	76.7 ± 30.0
Calcium (mg/kg)	733.3 ± 83.3	766.7 ± 116.7	733.3 ± 83.3	216.7 ± 60.1
pH	6.4 ± 0.3	6.1 ± 0.2	5.9 ± 0.09	5.1 ± 0.3
CEC meq/100g	5.8 ± 0.9	6.9 ± 0.7	7.6 ± 0.7	5.6 ± 1.3
Sand (%)	50.0 ± 7.2	47.3 ± 8.7	61.3 ± 5.2	52.0 ± 5.0
Silt (%)	26.3 ± 3.5	29.0 ± 5.0	24.3 ± 2.9	28.3 ± 4.1
Clay (%)	23.7 ± 3.7	23.7 ± 3.7	14.3 ± 2.4	19.7 ± 2.4
Soil Type	Loam – Sandy Loam	Sandy Loam – Clay Loam	Loam – (Sandy) Clay Loam	Sandy Loam – Clay Loam

Plant Characteristics

Estimated plant coverage differed significantly between the four land use types (Kruskal-Wallis, $\chi^2_{(3)} = 8.13$, $p < 0.05$, $n = 12$) (Fig. 1.12). Estimated plant coverage in woodlot land uses averaged over 215%, whereas corn plots averaged 20.33% (Table 1.2). Plant species diversity was also significantly different between the four land uses (Kruskal-Wallis, $\chi^2_{(3)} = 9.67$, $p < 0.05$, $n = 12$) (Fig. 1.13). Plant diversity followed a similar trend to plant coverage, with woodlots having the highest average plant diversity at 1.46 and corn plots averaging the least at 0.042 (Table 1.2).

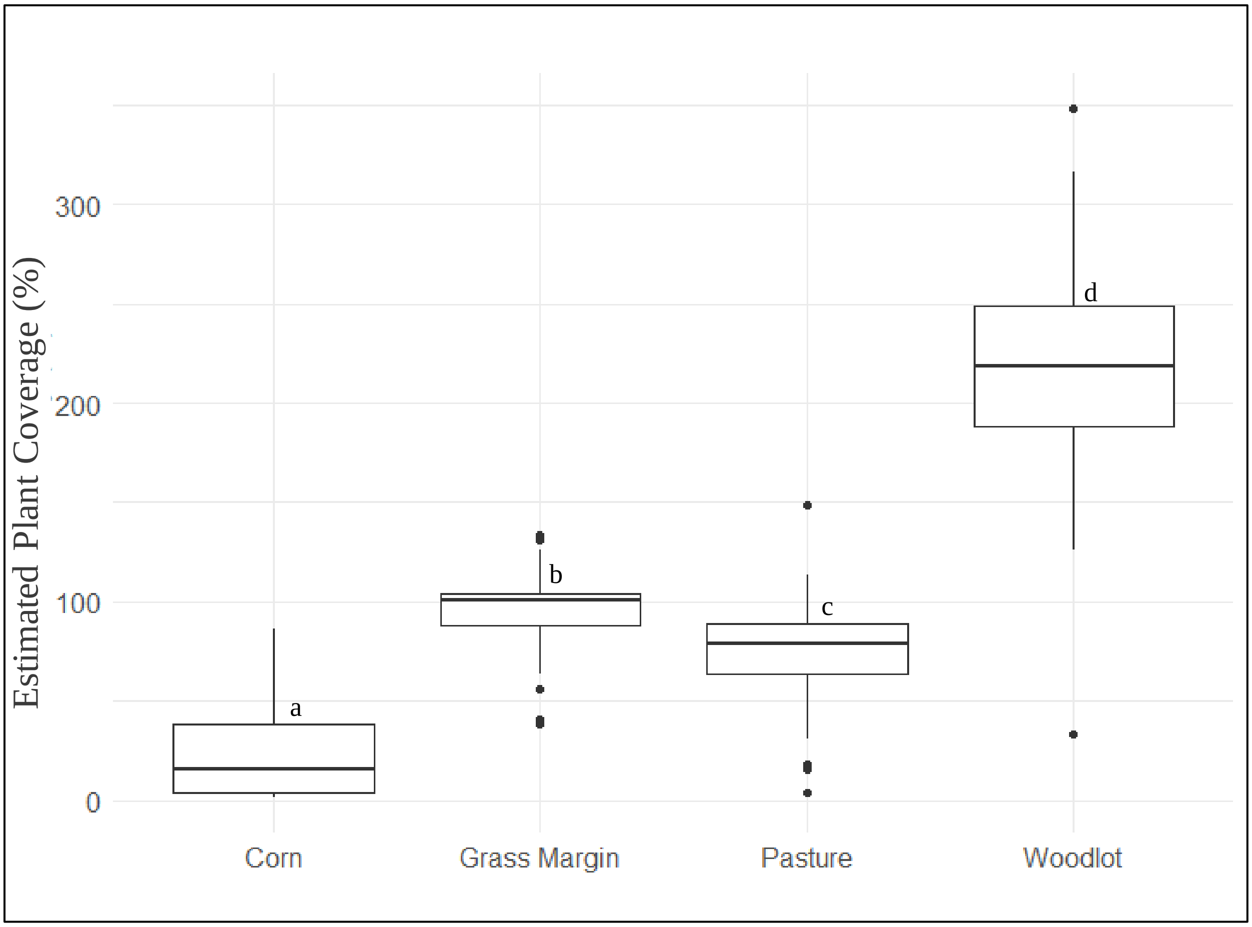

Figure 1.12 Box plot of estimated plant coverage measured with the modified Daubenmire frame method. Points were converted from a 6-point scale to the average percentage cover class to estimate plant coverage. Samples were collected twice during the early summer (June and July). Plots with different letters indicate that the estimated mean cover is different at the $\alpha = 0.05$ significance level.

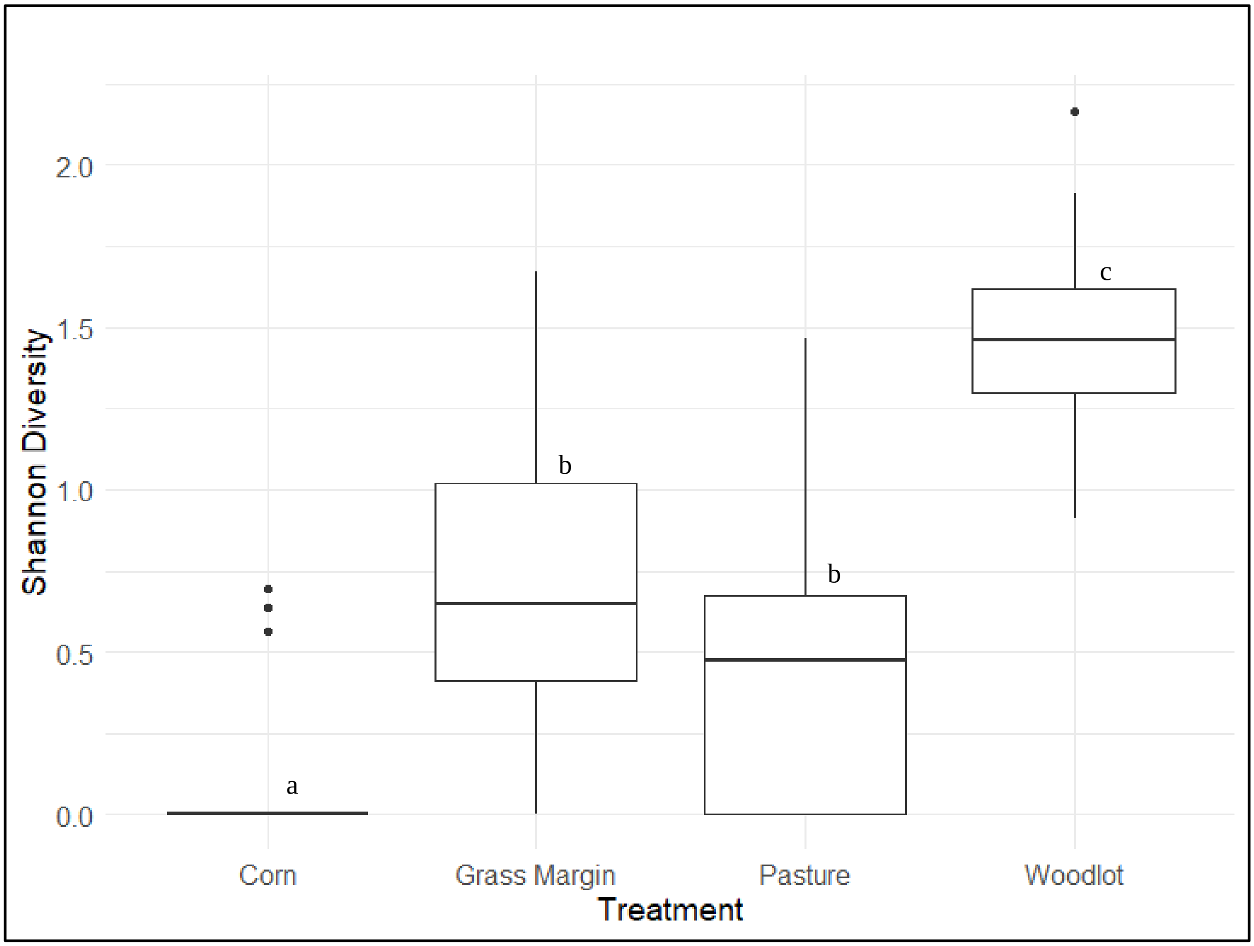

Figure 1.13 Box plot of plant species diversity (Shannon Diversity Index). Samples were collected twice (June and July). Diversity was calculated using the percent cover for each species. Plots with different letters indicate that the mean species diversity are different at the α = 0.05 significance level.

PERMANOVA

Differences between communities across study land uses were significant, with land use accounting for 26% of the total variance in communities for each sample (PERMANOVA, $p < 0.001$) (Table 1.5). Pairwise PERMANOVA revealed that all land uses have significantly different community compositions from one another ($p < 0.05$) aside from grass margins and pastures (Table 1.6).

Table 1.5 PERMANOVA results of soil arthropod communities in the four land use types relative to environmental variables sampled at CMREC in 2024.

	Df	Sum of Sqs	R^2	F	Pr(>F)
Land Use Type	3	2.09	0.264	5.13	0.001
Bulk Density	1	0.10	0.013	0.76	0.52
Volumetric Water Content	1	0.27	0.034	1.95	0.08
Temperature Range 1-2cm	1	0.14	0.018	1.06	0.30
Residual	39	5.29	0.67		
Total	45	7.89	1		

Table 1.6 Pairwise PERMANOVA results of soil arthropod communities between the four land use types from samples collected at CMREC in 2024

	Df	Sum of Sqs	R^2	F	Pr(>F)
Corn - Pasture	1	1.31	0.07	6.02	0.001
Corn - Grass Margin	1	1.72	0.09	8.92	0.001
Corn - Woodlot	1	1.46	0.1	6.89	0.001
Pasture - Grass Margin	1	0.37	0.02	2.15	0.061
Pasture - Woodlot	1	2.45	0.16	13.18	0.001
Woodlot - Grass Margin	1	3.27	0.23	20.68	0.001

Correlations

Volumetric water content was significantly correlated to the mean macro-arthropod (r = -0.26, p = 0.0076, n = 46), Formicidae, Parasitica, Chilopoda, and Diplopoda abundance (Table 1.7). Bulk density was significantly correlated with the soil biological quality - arthropod index (r = -0.44, p = 0.0022, n = 46). Diplopoda and Chilopoda had moderate positive correlations with bulk density (Table 1.7).

Table 1.7 Correlations between select soil properties and major soil arthropod taxa. Significant values are followed by astricts p < 0.10 (*), p < 0.01 (**), p < 0.001 (***).

Variable	Bulk Density	Volumetric Water Content	QBSAR
Microarthropod	0.08	0.09	0.23
Macroarthropod	0.14	-0.26*	0
Acari	-0.01	0.08	0.13
Collembola	-0.09	0.06	0.2
Formicidae	0.12	-0.25*	-0.04
Coleoptera	0.06	-0.2	0.41*
Diptera	-0.17	0.14	0.31*
Parasitica	0.22	-0.23*	-0.06
Diplopoda	0.53**	-0.25*	-0.14
Chilopoda	0.33*	-0.27*	-0.05
Carabidae	0.14	0.08	0.16
Staphylinidae	0.01	-0.18	0.44*

Redundancy analysis (RDA)

Diplopoda and Chilopoda had positive associations with bulk density (Fig. 1.14). In contrast, microarthropods including Collembola and Acari, along with Ptillidae are influenced by increased volumetric water content, lower soil temperatures, and lower soil bulk densities. The relationship between Formicidae, Staphylinidae, and Carabidae to environmental variables remains unclear from this work (Fig. 1.14).

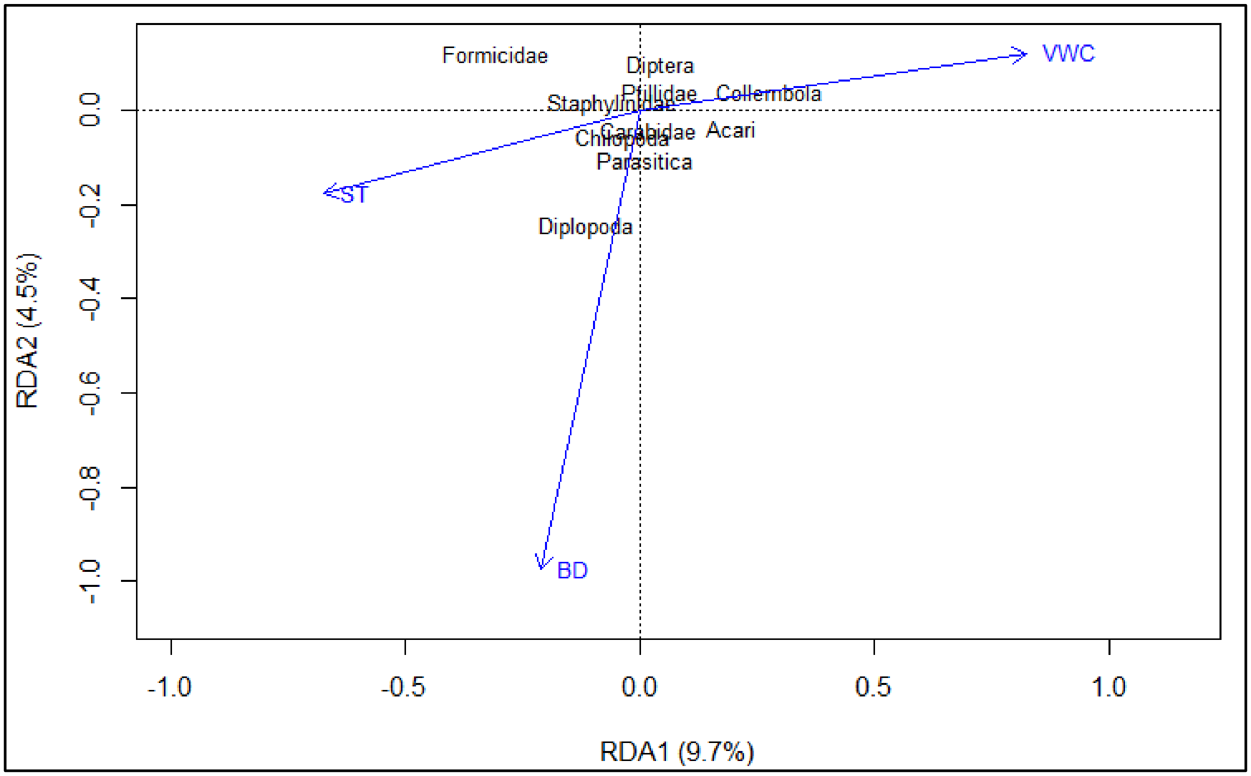

Figure 1.14 Redundancy analysis (RDA) of the major soil taxa groups in response to vectors of significant soil properties. ST represents soil temperature; BD represents soil bulk density; VWC represents volumetric water content (%).

Discussion

The goal of this research was to investigate the response of soil arthropod communities to land use type within a forage system and to relate differences in soil properties to soil arthropod communities. Here, I have shown that soil arthropod communities and soil properties differ significantly between the four land use types. Land use type explained over 26% of the variation in soil arthropod communities in this study, revealing that land use and management practices influence soil arthropod communities. These results agree with other studies investigating soil properties and arthropod communities across land use gradients (Menta et al., 2018; Creamer at al., 2015).

Through this work, I tested several hypotheses related to how land use affects soil arthropod communities and soil conditions. The hypothesis that pastures support more abundant and diverse soil arthropod communities than conventional feed systems was only partially supported, as only microarthropod abundance differed significantly. Nonetheless, through PERMANOVA analysis, I revealed that the community composition was different between the two land uses. Soil bulk density, soil pH, and soil temperature ranges were higher in the linear, monoculture, and high disturbance land use type (corn-soybean rotation plots) compared to perennial forage pastures. The hypothesis of lower volumetric water content and soil biological quality – arthropod index scores in corn plots compared to pasture plots was supported. I also found no difference between grass margins and pasture plots in soil arthropod community metrics and abiotic and biotic soil properties.

Soil arthropod communities differed significantly between the four land use types with grass margins and pastures possessing more similar arthropod communities. As in other studies, soil micro-arthropods such as Acari and Collembola dominated the samples collected here (Lui et al, 2016; Chikoski et al., 2006; O'Lear & Blair 1999); however, Formicidae represented the highest average abundance for all land uses. The abundance of Acari and Collembola, both containing important detritivore, saprophagous, and predatory species, were highest in grass margin and pasture plots.

In general, soil arthropod diversity is greatest in forested areas and grasslands, followed by cultivated land (Menta et al., 2018). Through network analysis, a method that maps co-occurrence patterns among organisms to reveal how taxa are interconnected within a community, Creamer et al. (2015) determined that the highest network density of soil organisms across Europe is found within forests, while the lowest is in arable soils dominated by a few taxonomic groups. These findings were confirmed here, as taxa evenness was lowest in the cultivated corn plots and highest in woodlot plots. All land use types contained similar average taxa richness; however, arthropod communities were less diverse at higher levels of alpha in corn plots compared to the other three. Conventional management practices, low plant diversity, and low habitat heterogeneity in corn plots are likely the cause of decreased abundance and diversity of soil arthropods (Tsiafouli et al., 2015).

Soil arthropods are a good indicator of soil quality because of their sensitivity to conditions within the soil environment. To measure and quantify soil quality, I used the soil biological quality – arthropod index, which considers arthropod biodiversity and individual taxa vulnerability (Menta et al., 2018). Through investigation of soil quality from subterranean trap samples, I observed that semi-natural habitats (grass margins and woodlots) possessed the highest soil quality averages. Nevertheless, the average soil biological quality score in pastures was higher than in corn plots. From a meta-analysis investigating the use of the QBS-ar index across Europe and Asia, Menta et al. (2018) determined that 93.7 was the threshold separating low-quality and high-quality soils. The average QBS-ar in corn plots fell below this threshold while all other land use type averages surpassed this threshold. The same study found that soils involved in human degradation, urban parks and forests, and agricultural lands possessed the lowest median QBS-ar scores, agreeing with my findings here (Menta et al., 2018). The index is not commonly used in the United States, resulting in data on QBS-ar metrics not being available; however, it seems that values derived from this research are similar to European countries such as Italy, Spain, Sweden, and the United Kingdom (Menta et al., 2018). Although this index has been used successfully to investigate soil health throughout different land use types, limitations exist, including low taxonomic resolution, presence-absence index vs abundance-biomass index, and its sensitivity to sampling effort and seasonality.

Throughout the agricultural landscape, biotic and abiotic properties within the soil are largely mediated by management practices. Several studies have identified tillage/plowing as one of the main factors influencing soil arthropod abundance and diversity (Jabbour et al., 2016; Lui et al., 2016). When frequently implemented, tillage can reduce the soil bulk density of conventionally managed fields and support relatively low diversity but abundant soil arthropod communities (Xin et al., 2018). Nevertheless, others have reported that physical disturbances such as tillage have no effect (Reilly et al., 2023). Compared to all others, soil bulk density was the highest in corn plots, with the grass margin and pasture plots measuring similarly to each other. Management practices, both past and present within corn fields—including tillage, irrigation, plant diversity/perenniality, and external inputs—alter the pedological structure of the soil, influencing pore spaces and water infiltration. Although grass margins and pastures undergo differing management practices such as grazing in forage pastures, soil bulk density did not differ between them in my study. Therefore, the presence of cattle within pastures over the past 30+ years had a neutral impact on soil bulk density. In this study, Chilopoda and Diplopoda had strong positive relationships with bulk density, whereas, in another study investigating the response of soil arthropods to soil properties, Coleoptera and Collembola had strong positive associations with bulk density (Liu et al., 2016). Differences between studies may be attributed to their scope and focus, as the assessment of select soil properties may lead to incomplete conclusions about their effect in relationship to one another.

Soil moisture has a strong relationship with soil bulk density because of bulk densities direct relationship with soil porosity (White 2006). Increasing soil porosity leads to increased water infiltration, resulting in increased water retention and decreased runoff due to surface pooling. Ideal porosity will allow water to drain with optimal retention, avoiding waterlogging roots and mediating their access to oxygen. Management practices and vegetation structure are most likely the main influences of soil moisture here and have been in other studies (Schmid et al., 2024; Mackay et al., 1986). Volumetric water content was highest in woodlots, which possessed the highest plant coverage as well as soil arthropod diversity. As expected, volumetric water content was lowest in corn plots with the lowest average plant coverage and soil arthropod diversity. Pastures had similar soil moisture to both woodlots and grass margins, indicating that management practices and soil properties can sustain moisture conditions similar to semi-natural habitats. Experimental results altering soil moisture have indicated that soil arthropod taxa such as Acari and Collembola respond differently to fluctuations in soil moisture; however, drought conditions generally reduce the diversity of soil arthropods (Tsiafouli et al., 2004).

Many of the arthropod taxa sampled here feed and burrow within the soil, improving porosity, water infiltration, and bulk density (Kishore et al., 2024). These taxa can influence soil conditions through burrowing, bioturbation, and redistribution of nutrients. Soil structure influences plant growth through nutrient availability to plant roots. Soil structures that provide plant roots access to appropriate nutrients, water, oxygen, and stability are important in forage systems to support the growth of high-quality forages. Millipedes burrow through tightly packed soils and open water channels, while ants have remarkable abilities to aggregate and redistribute soil with reports of ant species in grassland systems excavating 5kg to 13,000kg of soil per ha/year (Frouz & Jilkova et al., 2008). This has major implications as nutrients continue to become redistributed and brought to the soil surface by ants throughout their active season. Although soil arthropods can alter the soil environment in these study land uses, I do not expect soil arthropods to have a significant impact in conventionally managed corn plots due to chronic disturbances in these fields.

Temperature affects soil arthropods because of their evolution in underground environments with relatively stable moisture and temperature levels. When temperatures rise, evaporation increases, which can aid in pulling water from lower depths through cohesion (White 2006). High temperatures for extended periods can result in soil moisture levels dipping below arthropod thresholds causing migration to deeper soil depths or desiccation (Gobat et al., 2004). Vegetation structure and surface cover (such as litter and organic matter) play a role in temperature buffering because plants and surface cover intercept incoming solar radiation, and shade the soil surface. Although many soil arthropods are sensitive to desiccation because of their small body size and thin cuticles, others such as Acari:Oribatida and Diplopoda:Julida have evolved morphological or behavioral characteristics that allow them to persist in drought-prone environments (Taylor and Wolters 2005; Sjursen and Somme 2000; Cloudsley-Thompson 1959). These traits may partially explain the abundance of Acari and Diplopoda in corn plots, as both groups were numerically dominant.

The important role of plant diversity in arthropod communities is relatively well-known (Perez & Romero 2012; Harvey et al., 2008). Conventionally, corn fields are planted in monocultures whereas perennial pastures are lightly managed with the presence of introduced and native plant species increasing resource availability and microhabitat heterogeneity within pasture land uses. Increasing plant diversity in conventionally managed corn fields has the potential to increase the diversity and complexity of arthropod communities (Ebeling et al., 2018). Research at the USDA Agricultural Research Station in Beltsville, MD, found no difference in microarthropod communities between three agricultural management systems with differing management practices (crop rotation, tillage, weed control, fertility); however, differences in soil properties between the three were identified (Reilly et al., 2023). Herbicide applications are a common practice in forage systems to control weeds in both pastures and field crops. The response of soil arthropods to herbicide application has shown mixed results because of variables like handling time during the application period (Greenslade et al., 2010; Lins et al., 2007). On the other hand, effective herbicide application kills target weeds and accumulates plant residues, reducing arthropod exposure, increasing resource availability, and increasing soil arthropod abundance (Lui et al., 2016; Wardle et al., 1994).

Land use type resulted in the main differences in vegetation diversity and coverage in this study. Vegetation structure and community composition have been shown to influence arthropod communities, specifically herbivores, and their predators (Ebeling et al,. 2018). The main functional groups within soil samples in this study were detritivores and saprophagous arthropods while predatory arthropods were less abundant. In several studies, results have shown that altering species richness and diversity of the plant communities may not influence epigeal and decomposer arthropods (Ebeling et al., 2018; Seimann et al., 1998). Many decomposers are generalists and feed upon plant litter or animal excrement from several species in these systems. Generalist decomposer taxa such as Formicidae, Collembola, and Acari dominated our samples, and differences were observed in their abundance across land uses signifying that plant community composition may have some direct influence through resource provisioning or indirect influence through alteration of soil properties.

Using a redundancy analysis, I was able to identify relationships between environmental variables like soil bulk density, soil volumetric water content, and soil temperature, and major soil arthropod groups like Collembola, Acari, and Myriapoda. Other studies have also investigated the impact of similar soil properties showcasing differing results in the direction of impact for soil arthropod taxa and these soil properties (Manu et al., 2022; Jaksova et al., 2020; Kadamannaya et al., 2010; Tsiafouli et al., 2015; O'Lear & Blair 1999).

Conclusion:

This study highlights the significant influence of land use on soil arthropod communities and soil properties within forage systems. Management practices directly influence soil properties, which in turn interact with soil arthropods. My results here indicate that perennial pastures and semi-natural habitats, such as grass margins, support higher arthropod abundance, biodiversity, and soil quality compared to conventionally managed corn plots, an argument for their presence throughout the agricultural landscape. These findings align with previous research emphasizing the negative impacts of intensive agricultural practices on soil health and biodiversity (Menta et al., 2018; Creamer et al., 2015, Tsiafouli et al., 2015). The differences observed between land use types underscore the ecological benefits of promoting systems containing perennial forage pastures and semi-natural habitats such as grass margins and pastures.

The key factors that were important in shaping soil arthropod communities were soil bulk density, volumetric water content, and vegetation structure. Higher plant coverage, diversity, and perenniality, along with a decreased rate of disturbance in semi-natural habitats, likely enhanced soil porosity, water retention, and resource availability, providing a favorable environment for soil organisms, including sensitive and vulnerable soil arthropod taxa. Conventionally managed corn plots exhibited the lowest soil quality and arthropod diversity; and introducing management practices that benefit soil properties and reduce physical disturbances holds promise for supporting soil health and arthropod communities in conventionally managed fields (Perez & Romero, 2012; Harvey et al., 2008). For example, the inclusion of no-till and cover cropping management practices have been shown to support soil arthropod abundance and diversity

(Jabbour et al., 2015; House and Alzugaray 1989). Responses of soil arthropods to ecologically focused management practices take a considerable amount of time, and legacy effects of previous management practices could be present years after their introduction (Crotty et al., 2016; Jabbour et al., 2015).

Since this study lasted one summer, future research could explore the impacts of integrating diverse, perennial plant species and alternative management practices, such as reduced tillage or recycling ruminant excrement, on soil arthropod communities and ecosystem functions across a longer timeframe. Another expansion of this work could be the use of the soil biological quality- arthropod index across different regions to enhance our understanding of soil quality under varying land management practices. Critical studies further validating this index could provide useful information for its application too. By bridging gaps between agricultural production and ecological intensification, this work contributes to advancing sustainable farming practices and preserving soil ecosystems throughout the forage landscape.

Chapter 2: Comparison of Soil Biota Feeding Activity in Four Land Use Types in a Forage System in Central Maryland Using the Bait-Lamina Method

Abstract

Supporting ecosystem services that influence the nutrient cycle is imperative for sustainable agriculture. Soil invertebrates are important consumers of detritus and excrement, fragmenting organic matter into fine particles and pellets later used by microorganisms. In response, their activity influences the development and status of the soil, making soil biotic activity a good indicator of differences in soil quality. This study used the bait-lamina method to investigate differences in feeding activity of soil invertebrates between four land use types common to livestock systems: conventional corn-soybean rotation fields, perennial forage pastures, grass margins, and woodlots. I further investigated the relationships between several soil properties, including soil bulk density, soil moisture, and soil biological quality to feeding activity by soil biota. Results revealed lower levels of feeding activity in monoculture and high disturbance corn plots compared to all other land use types. Soil bulk density, soil moisture, and soil biological quality – arthropod index correlated significantly with feeding activity. The results presented here indicate that land uses dominated by perennial grass communities supported elevated levels of feeding activity, and soil properties can influence the ecosystem services that soil invertebrates provide. These findings support the use of sustainable land management practices that enhance soil quality and promote arthropod-driven decomposition processes. Ecologically

intensified systems, such as forage pastures, not only improve soil quality and resilience but also contribute to nutrient cycling and long-term agricultural sustainability within livestock systems.

Introduction

Approximately 90% of the biomass from primary production on land enters detritus food webs above or within the soil, where soil and soil biota recycle essential resources, supporting the next round of primary production (Gessner et al., 2010). Soil quality is the capacity of soil to perform ecological functions and ecosystem services, with healthy forage systems depending on high-quality soils to sustain the growth and development of ruminants (Karlen et al., 1997). Low-quality soils reduce forage yield, impair ruminant growth rates, and lead to nutrient deficiencies, poor-quality products, and instability within current forage systems. Soil animals, including mites (Acarina), springtails (Collembola), millipedes (Diplopoda), isopods (Isopoda), enchytraeids (Tubificida), earthworms (Haplotaxida), and many insects (Insecta) play a crucial role in forming high-quality, sustainable soils (Hamel et al., 2007). These organisms facilitate organic matter breakdown and prime microorganisms for mineralization through the displacement and fragmentation of litter, producing large amounts of feces readily available for microorganism colonization. In many cases, soil invertebrate activity influences the development of soil; therefore, soil biotic activity may be an indicator of differences in soil quality between land use types (LaRade et al., 2012).

The estimated global economic value of soil biodiversity in decomposition and soil formation exceeds 1.5 trillion US dollars annually (Pimentel et al., 1997). Although these services still provide immense value to our current farming systems, generally, they are not economically viable alone, prompting the need to supplement them with conventional management practices such as the application of synthetic fertilizers. The loss of ecosystem services provided by soil biota would have severe consequences for agricultural productivity and farmer resilience within today's systems.

Numerous studies have explored the effects of agricultural intensification on soil biota ecosystem services and diversity, consistently revealing negative impacts (Reilly et al., 2023; Creamer et al., 2015; Tsiafouli et al., 2015; Reinecke et al., 2008). Due to the overall stability of soil environments, soil arthropods have evolved a low tolerance to drastic abiotic fluctuations that are introduced by modern agricultural practices (Mammola et al., 2019; Bender et al., 2016; Tsiafouli et al., 2015). Management practices and vegetation types influence soil biota communities by altering critical microhabitat properties such as light penetration, soil pH, and nutrient/organic matter levels (Chikocki et al., 2006). Soil biota are concentrated within the litter layers and upper soil horizons resulting in their abundance and community composition largely mediated by individual temperature and moisture tolerance limits (Chikocki et al., 2006). Intensifying agricultural management practices to meet the expanding needs of the human population has resulted in the degradation of environmental conditions, particularly within soils (Foley et al., 2005).

Ecological intensification within forage systems through increased plant diversity, plant perenniality, and system circularity has been proposed as a strategy to enhance sustainability and resilience while avoiding the negative effects of conventional farming practices (Picasso et al., 2022). Ecological intensification leverages ecosystem services such as decomposition, pest control, soil aeration, and bioturbation in place of conventional management practices, reducing input and disturbance rates. Previous studies indicate that increasing forage and plant diversity enhances ecosystem service provisioning by arthropods (Haan et al., 2023; Wagg et al., 2014;

Nicholls and Altieri, 2013). Diverse soil biota communities contribute significantly to ecosystem functioning by accelerating litter decomposition rates and nutrient cycling, thereby reducing reliance on external inputs (Wagg et al., 2014; Bardgett & Chan, 1999; Hendrix et al., 1986).

Efforts to restore, maintain, and leverage the ecosystem services provided by soils and soil arthropods require deliberate landscape redesign, informed by ecological research on the connections between land use and ecosystem services (Haan et al., 2021). Hence, transitioning to more stable and sustainable forage systems that integrate ecological intensification is critical to supporting soil biota-mediated ecosystem services.

To address this knowledge gap, I conducted a study at the University of Maryland, Central Maryland Research and Education Center in Clarksville, MD with the primary objective, of evaluating differences in ecosystem service provisioning (feeding activity/decomposition) between conventional and ecologically intensified forage systems, linking bait removal to selected soil properties. In addition to sampling conventional and pasture forage systems, I included grass margin and woodlot plots as semi-natural habitats commonly found on farms in the Eastern United States to assess their role in supporting ecosystem service provisioning across the landscape.

I hypothesize that seasonal sampling of feeding activity between spring and fall will reveal temporal differences in ecosystem service provisioning, with higher levels of feeding activity in July. I anticipate that feeding activity will be highest in semi-natural habitat plots and lowest in conventionally managed feed plots. I further hypothesize that feeding activity will have a negative relationship with soil bulk density and a positive relationship with volumetric water content and the soil biological quality - arthropod index. Additionally, I expect that grass margins will not differ significantly from pasture plots in feeding activity or soil properties.

Methods

Site Description and Experimental Design:

This study was carried out at the University of Maryland's Central Maryland Research and Education Center (CMREC) in Clarksville, Maryland, USA, a 375-ha dairy farm that rotationally grazes a cowherd of 13-36 from June to October. I implemented a fixed-effect block design for this project. Three sampling blocks, each containing a replicate of the four land uses (corn-soybean rotation fields (conventional forage), forage pasture, grass margin, and woodlot) were identified based on location throughout the farm. All land use plots used in this study have retained their land use status for more than 30 years. Land use plots were 100m x 400m and located adjacent to other land use plots within the same block (Fig. 2.1). Four sampling locations equally spaced along the length of each sampling plot were chosen for bait-lamina testing.

The conventional corn-soybean rotation land use was planted in corn for all plots during this study. Corn varieties included Channel 212-40VT4 PIRB, DKC 65-95RIB, and H4390RC2P. All cornfield plots were planted directly into bare soil or triticale stubble. Conventional herbicide, pesticide, and fertilizer land uses were applied to these fields throughout the sampling period. No-till practices have been implemented for blocks B and C; however, block A corn field is tilled every 3 years (Table 1.1).

Pastures were rotationally grazed throughout this study. Pasture A was planted with a Double Play Mix containing 45% gainer triticale, 33% oats, and 22% annual ryegrass along with crimson clover. Pasture B and C were overseeded with King's Beefmaster Mix containing 50%

soft leaf tall fescue, 20% perennial ryegrass, 20% leafy orchardgrass, and 10% alice white clover. In both pastures, Kentucky 31 fescue was still present. During the duration of this study, both annual and perennial forages were present in these systems including tall fescue (*Lolium arundinaceum*), perennial ryegrass (*Lolium perenne*), crimson clover (*Trifolium incarnatum*), and white clover (*Trifolium repens*). Cattle were left in the plot paddock(s) for no longer than five days and each pasture was grazed a minimum of three times throughout this study. Mowing occurred in pastures A and B following grazing and an herbicide application was applied to pasture C in spring 2024 (Table 1.1). No other management practices were present in forage pastures.

Maintained grass margins were defined in this study as an area dominated by perennial grasses and mowed to 12-16 centimeters once or twice a month during the summer months (June- September). These areas border agricultural fields and woodlots throughout the farm. Dominant grasses include smooth brome (*Bromus inermis*), tall fescue (*Lolium arundinaceum*), and bluejoint grass (*Calamagrostis canadensis*).

Woodlots are scattered throughout the farm and are usually found in places that cannot/have not been cultivated in the past. These woodlots were consistent with a mesic mixed hardwood forest, dominated by various oak species (*Quercus* spp.), american beech (*Fagus grandifolia*), and tulip poplar (*Liriodendron tulipifera*), which are typical of the overstory. Dominant understory species include multiflora rose (*Rosa multiflora*), northern spicebush

(*Lindera bezoiunm*), and Basket Grass (*Oplismenus hirtles*). Woodlots have remained unlogged for several decades (Table 1.1).

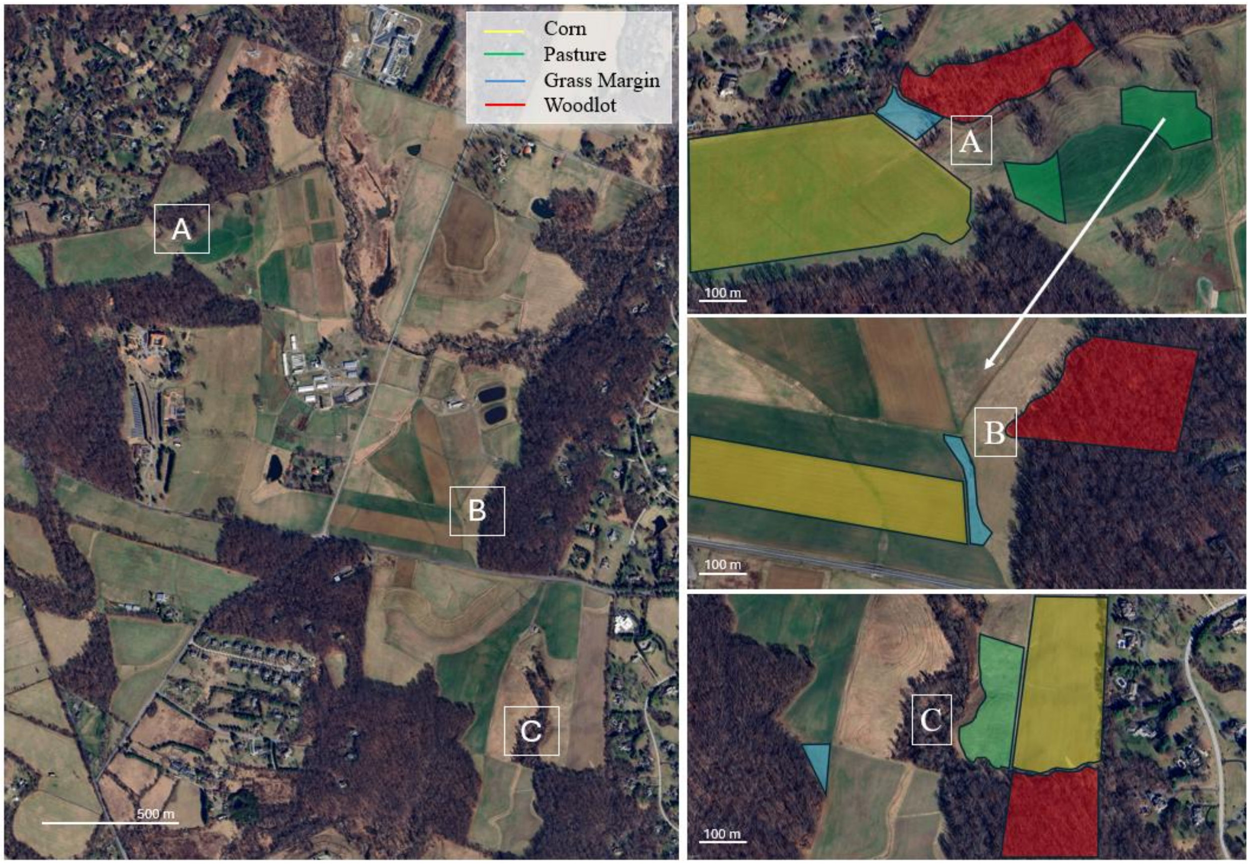

Figure 2.1 Map of the CMREC Clarksville farm in Clarksville, MD with each land use block represented by letters A-C and the distribution of land uses within the three land use blocks. Different colors represent different land uses. Pasture B was situated adjacent to block A due to pasture distribution throughout the farm. Google Earth Pro 7.3.6.10201.

Bait-Lamina Sampling:

To assess the activity of decomposer organisms in this study, I implemented the bait lamina method developed by Von Torne (1990). The bait lamina method has been repeatedly used in mesic environments, forest ecosystems (Rozen et al., 2010; Gongalsky et al., 2004), savannah/grassland ecosystems (Prozenrather et al., 2008; Hamel et al., 2007), and agricultural environments (Reikecke et al., 2008; Sturm et al., 2002) as a method to visualize soil biological activity and indirectly, the rate of decomposition (Reinecke et al., 2008). LaRade et al., (2012) utilized this method to compare feeding activities between native and seeded pastures, determining that bait lamina removal varied with vegetation type and soil depth. Generally, soil fauna feeding activity has been lower in conventional systems compared to ecologically intensified systems such as forage pastures or organic systems (Menta et al., 2018).

This method is based on the removal of bait substrate from a thin, perforated, laminated strip after exposure to decomposition processes in the soil. Although microbial breakdown of organic material is important for the nutrient cycle, Kratz (1998) concluded that microbial activity does not cause bait perforation within the typical exposure period of 7 to 20 days, which varies based on soil moisture (von Törne, 1990a; Kratz, 1998). Another study introducing soil arthropods following sterilization of soil concluded that perforation of bait-lamina strips was facilitated by soil-living animals (Gongalsky et al., 2008). Microbial effects may only result in thinning and subsequent destruction of the bait as an artifact, making meso- and macrofauna responsible for most bait removal here.

Bait-laminae were constructed following slight modifications from Reinecke et al., (2008). The plastic strips were 120 mm long, 5 mm broad, 1.5 mm thick, and 3D printed using yellow PLA (Fig. 2.2). The laminae were perforated at distances of 5 mm with 14 holes which were filled with a bait substance to attract soil organisms.

Bait-lamina strips were filled with a paste of agar, cellulose powder, and dried ground alfalfa. Strips were dried at 45 degrees Celsius for 24 hours. To insert strips into the soil, a modified butter knife was used to make the initial hole before inserting the strip into the ground.

Strips were placed in the ground so that the top of the uppermost hole settled flush with the top of the ground and the bottom hole rested at 65mm. For each sampling date in a singular sampling location, I placed 8 bait-lamina strips, with 4 plots per land use totaling 32 per land use block, 96 per total land use, and 384 in all, for each sampling date.

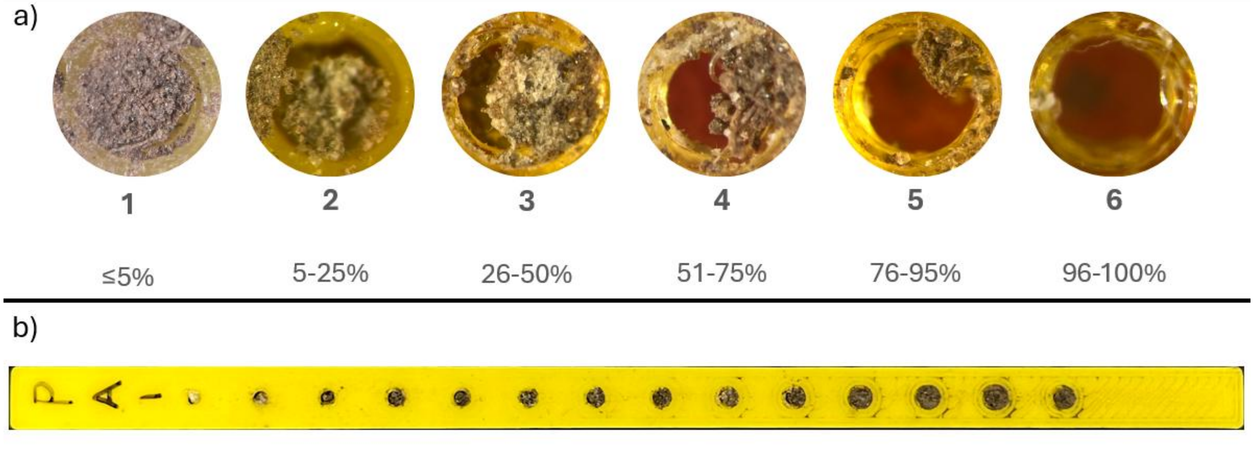

Fig 2.2 a) Reference 6-level scale for bait removal scoring, **b)** deployed bait-lamina from this study

Strips were left in the field for 14 days, collected, and placed on a metal sheet in the lab to air dry. All strips were analyzed within a week of collection. Spring (June) and Fall (September) sampling dates were used in this study. A six-level scale was used to classify the amount of bait removed from each hole, considering both the front and back sides of the strip (Fig. 2.2).

Soil Sampling:

Soil bulk density and volumetric water content were measured using soil cores. All above-ground vegetation and organic matter were removed from the surface before sampling. A metal ring (with a volume of 98.17 cm^3) was hammered flush into the ground at eight locations within the land use plots. Locations were determined by splitting land use plots into 8 equal sections and haphazardly sampling within each of the 8 sections. Sampling was completed two weeks before deployment, at deployment, and collection, totaling 3 soil sampling dates per bait lamina deployment. Soil was removed from sampling rings and placed in an airtight 240ml Corning Snap-Seal container in the field. In the lab, samples were weighed, dried in an oven at 105°C for 24 hours, and weighed again to calculate soil bulk density and volumetric water content.

Bulk density (g/cm^3) was determined using the following equation:

$$\boldsymbol{Bulk\ Density} = \frac{\mathrm{Wd}}{\mathrm{V}}$$

where:
W_d = Dry weight of the sample (g)
V = Volume of the sample (cm^3)

Volumetric water content (%) was determined using the following formula:

$$\boldsymbol{Volumetric\ Water\ Content} = \left(\frac{\mathrm{Vw}}{\mathrm{Vt}}\right) \text{ x } 100$$

V_w = Volume of soil water (cm^3)
V_t = Total sample volume (cm^3)

Soil Arthropod Sampling:

Soil cores were taken using a large metal ring (10.16 cm height x 10.16 cm diameter) at deployment and collection. Four subsamples were taken from each plot and homogenized in the field to create a composite sample. Two 820cm^3 samples were taken back to the lab for arthropod extraction within 4 hours of sample collection. Samples were placed in a Berlese-Tüllgren funnel for seven days for arthropod extraction. Following processing, arthropods were sorted into taxonomic groups using a Nikon SMZ800 dissecting microscope at 10x, counted, and identified to subclass, order, or family. Arthropods were able to be identified using the Borrer and Delong Study of Insects 7th edition dichotomous key to order and family (Tripelhorn & Johnson 2004).

To calculate an index of soil quality I used the soil biological quality - arthropod index (QBS-ar), a method developed by Parisi et al., (2005) and recently revised by Menta et al., (2018). The index provides a relatively quick assessment of the soil microarthropod communities and can quantify soil health based on the soil microarthropod presence. QBS-ar relies on the concept that soil microarthropods with higher vulnerability will be present in soils with better conditions such as low degradation, pollution, and compaction. To develop this index, major taxa were assigned an ecomorphological index score (EMI) based on the presence of characteristics such as short legs, short antennae, lack of pigmentation, and thinner cuticles. In all, these characteristics are an indicator of a group's adaptation to general or specific soil environments.

To calculate the QBS-ar for a single sample, arthropods are identified to the order/class level and an EMI value is assigned. Each taxon is only counted once (presence) and the sum of these scores is the QBS-ar score for that sample. A higher QBS-ar score could indicate that a higher taxa richness of soil arthropods is present or, that the arthropods present are more sensitive to disturbance. Both are indications of higher-quality soil.

Data Analysis:

Data analysis was conducted using R Statistical Software version (v4.4.2; R Core Team 2024) using the lme4, car, emmeans, and vegan packages. I tested for normality using the Shapiro-Wilk normality test. To assess differences in soil fauna feeding activity across land uses and sampling months, I used the Kruskal-Wallis rank sum test. This non-parametric test was selected because the data did not meet the assumptions of normality and homogeneity of variance required for parametric tests. Pairwise comparisons for non-parametric data were done using a Dunn test. For variables that met the assumptions of normality and homoscedasticity, I used linear mixed effect models (soil bulk density, volumetric water content, soil biological quality – arthropod index, and arthropod abundances). I tested for the significance of fixed effects (land use, month, and their interaction) using Type III ANOVA tables derived from the fitted models. Following ANOVA, post hoc comparisons were performed using estimated marginal means (least-squares means) for each factor level, adjusted for other terms in the model, and corrected for multiple comparisons using the Bonferroni method. A non-parametric Spearman's rank correlation was used to evaluate relationships between the proportion of bait consumed and other variables.

Results

Bait-Lamina Feeding Activity:

Overall, feeding activity from both months was significantly different between land uses (Kruskal-Wallis, $\chi^2_{(3)}$= 154.80, $p < 0.05$, $n = 24$, Figure 2.4). During both months, feeding activity was highest in grass margins. Although there was no significant difference between the feeding activity of both the sampling months, the pattern of bait removal throughout the soil column differed markedly. All land uses in July follow a pattern of elevated feeding activity in the 5mm to 25 mm range. September land use patterns are more variable, and while the 5mm – 10mm holes have higher feeding activities similar to July patterns, activity leveled off from 15mm on.

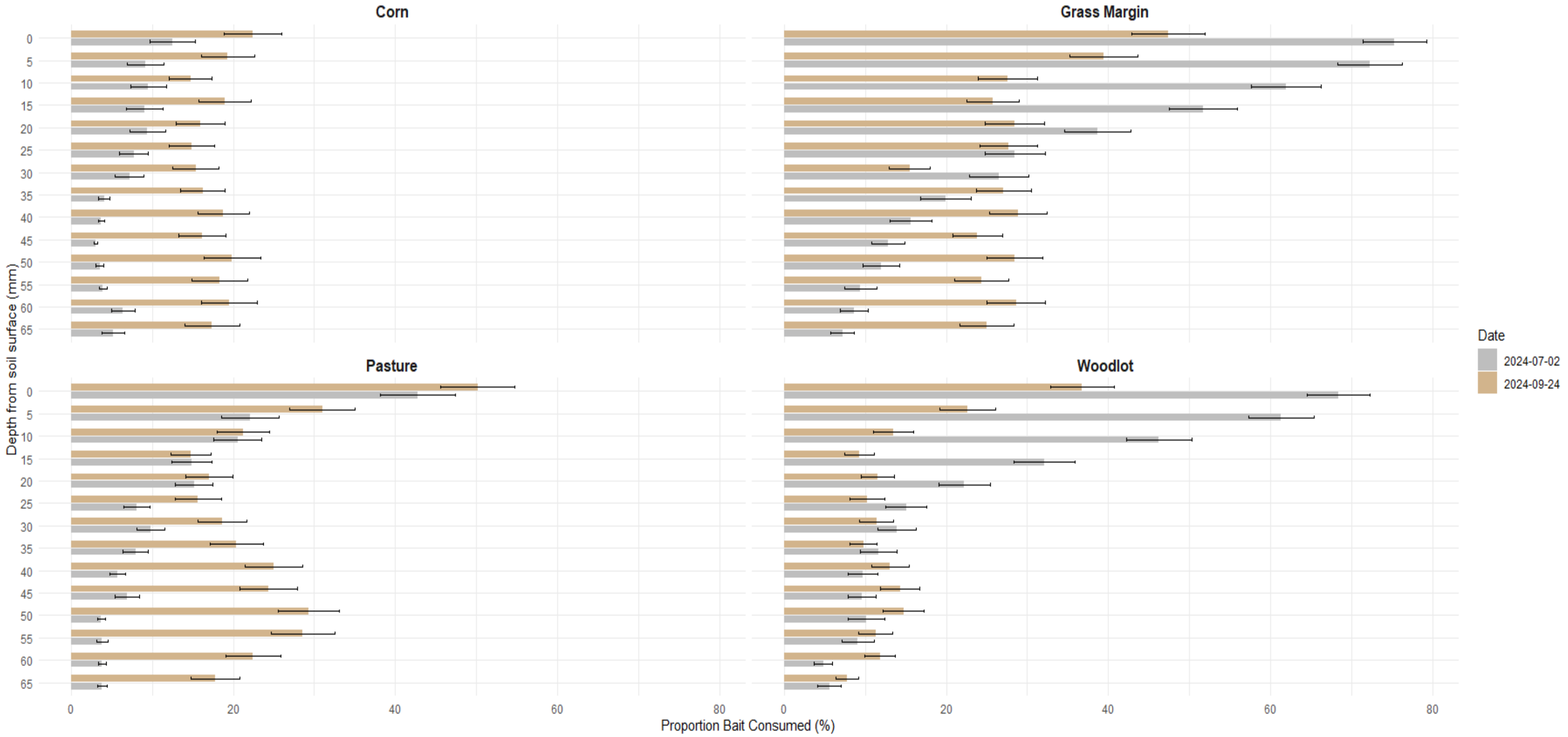

Fig 2.3 Vertical distribution of soil fauna bait consumption at CMREC in 2024 grouped by land use type.

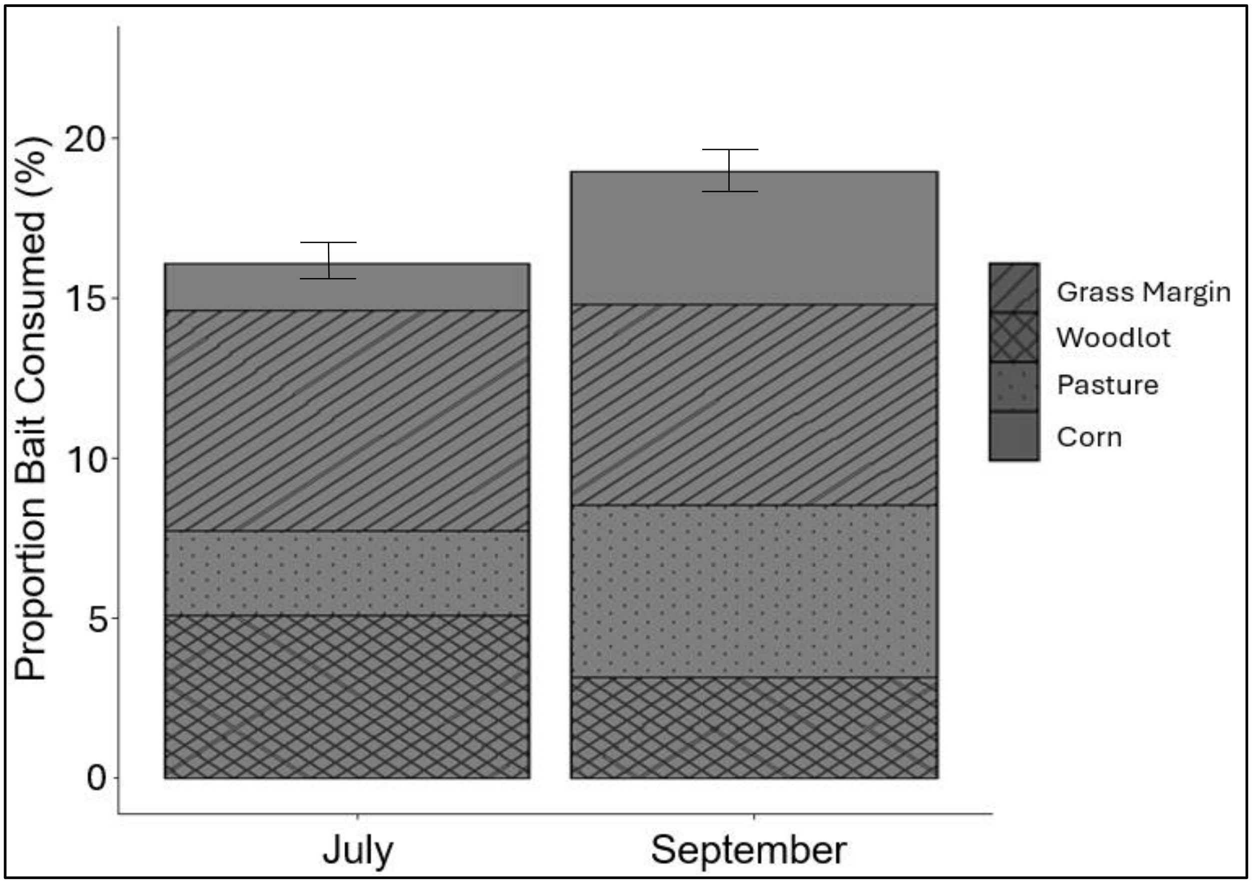

Fig. 2.4 Bait consumed, measured by bait lamina in July and September with the contribution of each land use to the overall mean coded

Table 2.1. Mean (±S.E.) values for the proportion of bait consumed, biotic, and abiotic soil properties in the four land use types at CMREC in 2024. Within each row, values followed by the same letter are not significantly different: a) July, b) September.

a)

Soil Property	Corn	Grass Margin	Pasture	Woodlot	p - value (n)
Proportion of Bait Consumed (%)	6.73 ± 0.872^{A}	31.2 ± 1.76^{B}	11.8 ± 0.95^{C}	23.0 ± 1.20^{B}	< 0.05 (12)
Bulk Density (g/cm^{-3})	1.07 ± 0.02^{A}	0.90 ± 0.01^{B}	0.95 ± 0.01^{BC}	0.85 ± 0.01^{D}	< 0.05 (12)
Volumetric Water Content (%)	17.8 ± 0.67^{A}	28.9 ± 0.83^{BC}	22.3 ± 0.91^{AC}	30.8 ± 0.92^{B}	< 0.01 (12)
QBS-ar	51.1 ± 7.1^{A}	101.9 ± 4.4^{B}	82.8 ± 2.9^{C}	97.2 ± 8.4B^{C}	< 0.05 (12)
Microarthropod Individuals (m^{-2})	4458.3 ± 1954.6^{A}	8765.6 ± 1521.1^{B}	11399.5 ± 3127.7^{BC}	3072.8 ± 303.6^{A}	< 0.01 (12)
Macroarthropods Individuals (m^{-2})	1022 ± 484.7^{A}	5391.1 ± 2628.5^{B}	2627 ± 1139.3^{AB}	2421.2 ± 663.6^{BC}	< 0.05 (12)

b)

Soil Property	Corn	Grass Margin	Pasture	Woodlot	p - value
Proportion of Bait Consumed (%)	18.7 ± 1.85^{AC}	28.4 ± 1.52^{B}	24.3 ± 1.8^{AB}	14.2 ± 1.10^{C}	< 0.05 (12)
Bulk Density (g/cm^{-3})	1.18 ± 0.01^{A}	0.91 ± 0.01^{B}	0.89 ± 0.02^{B}	0.82 ± 0.02^{B}	< 0.05 (12)
Volumetric Water Content (%)	20.3 ± 1.6^{A}	35.2 ± 1.9^{B}	31.5 ± 2.2^{B}	32.1 ± 2.0^{B}	< 0.10 (12)
QBS-ar	50.8 ± 8.4^{A}	97.7 ± 7.9^{B}	72.7 ± 4.2^{AB}	92.7 ± 8.4^{B}	< 0.01 (12)
Microarthropod Individuals (m^{-2})	1783.3 ± 336.9^{A}	10233.5 ± 2316.6^{B}	11262.3 ± 1601.9^{BC}	4074 ± 706.2^{A}	< 0.02 (12)
Macroarthropods Individuals (m^{-2})	391 ± 95.4^{A}	2030.2 ± 609^{B}	2599.5 ± 656.2^{BC}	2002.8 ± 1140.3^{BCD}	< 0.01 (12)

Abiotic Soil Properties:

I found significant differences in soil bulk density among the four land use types (ANOVA, $F_{(3,8)} = 16.5$, $p < 0.001$, $n = 12$), indicating that land use strongly affects soil bulk density (Fig. 2.5). Corn plots possessed significantly higher soil bulk densities compared to all land uses in both months (Table 2.1). There was no significant interaction between land use and date and no significant main effect of date alone, indicating that soil bulk density is not influenced by month here. I suspect the smaller sampling size in Chapter 2 has led to different conclusions about soil bulk density during the evaluation of statistical tests. Nonetheless, soil bulk density means does not significantly differ when comparing land use type between chapters.

I also found significant differences in gravimetric soil moisture among the four land uses (ANOVA, $F_{(3,6)} = 12.0$, $p < 0.01$, $n = 12$), indicating that land use had a strong effect on soil moisture content (Fig. 2.5). Corn had significantly lower gravimetric soil moisture levels than all other land uses for both July and September (Table 2.1). There was no significant main effect of date and no interaction between land use and date.

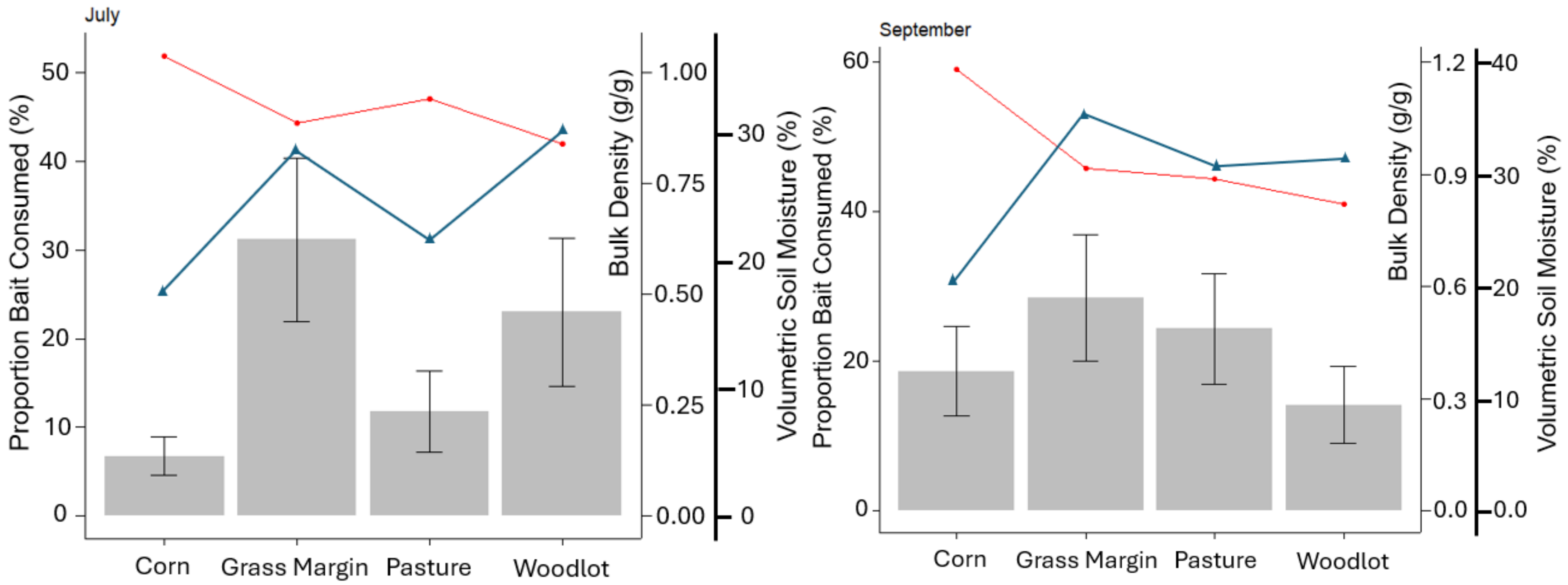

Fig. 2.5 The average proportion of bait consumed (July: n = 12; September: n = 12) and soil abiotic properties (circle: bulk density, triangle: volumetric water content) (July: n = 12; September: n = 12) in four study land uses at CMREC in 2024.

Biotic Soil Properties:

Soil Biological Quality - Arthropod Index

Soil biological quality scores ranged from 27 (Corn) to 150 (Grass Margin). My statistical model revealed that there are significant differences between land uses (ANOVA, $F_{(3,6)} = 18.6$, $p < 0.01$, $n = 12$, Fig. 2.6). Corn fields had significantly lower QBSAR scores than the other three land uses in both months aside from pastures in September (Table 2.1). There was no significant difference between the months, nor a significant interaction between month and land use.

Arthropod Communities

Micro-arthropod abundance ranged from 432 individuals/m^2 (corn) to 30,556 individuals/m^2 (pasture). Macro-arthropod abundance ranged from 0 individuals/m^2 (corn) to 25248 individuals/m^2 (grass margin). Micro-arthropod (ANOVA, $F_{(3,6)} = 7.80$, $p < 0.05$, $n = 12$) and not macro-arthropod abundances were significantly different between land use types (Table 2.1).

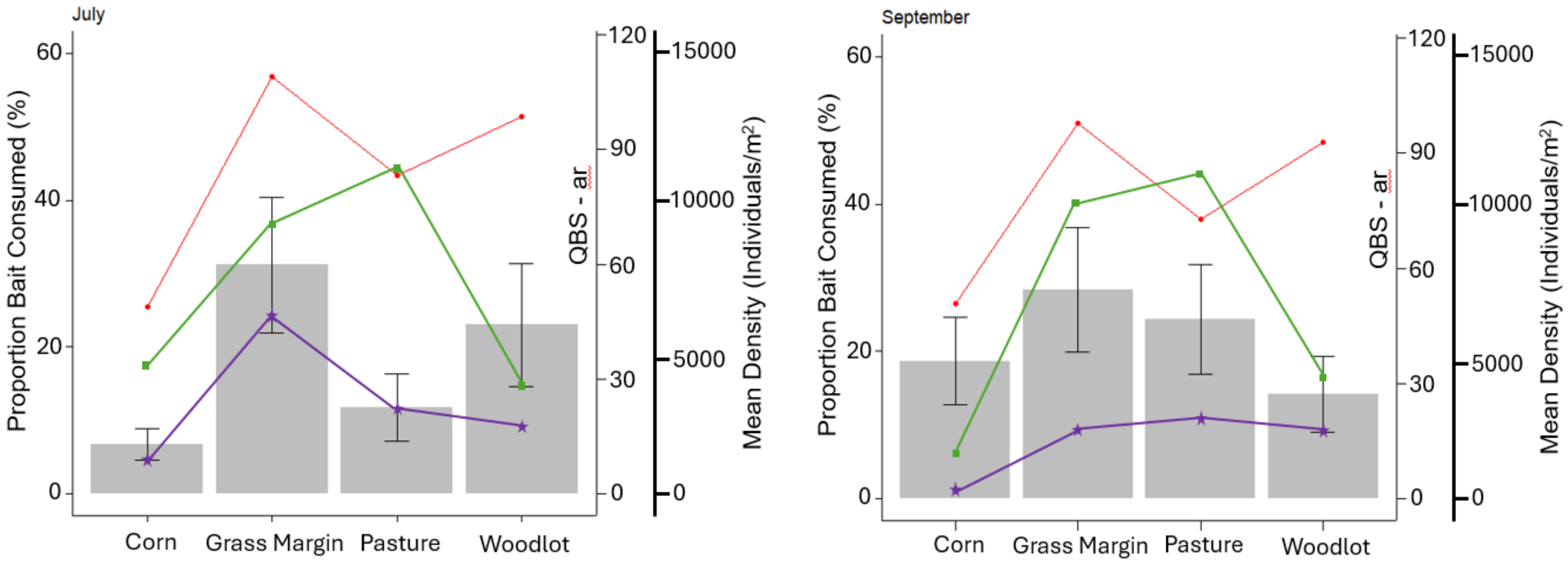

Figure 2.6. The average proportion of bait consumed (July: n = 12; September: n = 12), soil macro-arthropod (circle, n = 12), meso-arthropod abundance (square, n = 12), and soil biological quality arthropod index score (star, n = 12) in four study land uses at CMREC in 2024.

Correlations:

At the plot level, the proportion of bait consumed had a positive correlation with gravimetric soil moisture (r = 0.35, p = 0.003, n = 72) and the soil biological quality – arthropod index (r = 0.32, p = 6.8e-03, n = 72). Soil bulk density had a negative correlation with the proportion of bait consumed (r = -0.27, p = 0.02, n = 72). No significant correlation was found between meso- and macro-arthropod abundance and bait lamina feeding activity.

Discussion

I examined feeding activity and key soil properties in four land use types within common forage systems of the Northeast United States, including conventional corn silage fields, perennial forage pastures, and two semi-natural habitats. I hypothesized that feeding activity would be higher in July than in September; however, this was not supported, as overall feeding activity did not differ between months. In July, all land uses followed similar patterns of higher feeding activity from the 5mm to 25mm depths. feeding activity from lamia strips in corn plots was significantly lower than in pasture and grass margin plots for July. During the September sampling period, feeding activity leveled off after 10mm for all land uses, and bait removal was not different between corn and pasture land uses; however, both were significantly lower than grass margin plots. At the plot level, the proportion of bait consumed correlated significantly with volumetric water content, soil bulk density, and the soil biological quality – arthropod index, supporting my hypothesis. This study found that the feeding activity of soil biota occurs at its highest rates closer to the soil surface in all land use types, and soil moisture may be a limiting factor in mediating feeding activity by soil biota. These findings align with results from previous research investigating feeding activity using bait lamina strips (Reinecke et al., 2008; Vazquez et al., 2003).

Several factors, including soil texture, structure, and management practices, influence soil bulk density. Bulk density is a direct indicator of soil compaction, with high soil bulk density leading to decreased soil functioning. High bulk densities and compact soils have been shown to negatively impact decomposition mediated by soil meso-arthropods such as Collembola (Dittmer and Schrader 2000). Higher soil bulk densities have also been shown to negatively impact earthworms and enchyrtid densities (Smetak et al., 2007). Although I did not sample earthworms or enchytrids, both taxa were present in my samples. This relationship may explain the lower feeding activity in corn plots, as bulk density was the highest in these plots. Soil bulk density in pasture and grass margin plots did not differ due to similar vegetation structures and management practices. Although pastures experience rapid compaction from livestock pressure (opposite of grass margins), soil compaction is relieved through the decomposition of cattle dung (Herrick and Lal, 1995).

Differences in volumetric water content may have also resulted in differences in bait removal between land use types. Soil moisture plays a crucial role in decomposition, particularly when favorable conditions in other environmental factors such as temperature, vegetation structure, and soil type are present (Petraglia et al., 2018; Carter et al., 2010). Precipitation and soil moisture were highest during the September sampling period due to prolonged rain throughout the Eastern United States. From a weather station at the CMREC farm, approximately 10.16 cm of rain was received throughout the September sampling period on 12 days, whereas sampling plots received approximately 3.6 cm of rain in July over 2 days. My results did not indicate a significant difference in soil moisture between July and September, however, sampling soil three times surrounding deployment and collection of bait lamina may be too infrequent to capture this variation. Homogenization of moisture throughout all depths in September resulting from increased precipitation may have increased the soil fauna's ability to reach and feed upon bait at lower depths.

Soil bulk density and soil moisture alone highly influence the soil habitat, including water movement, compaction, aeration, and vegetation dynamics (White 2006). Here, volumetric water content has a significant negative correlation with soil bulk density (r = -0.56), supporting the idea that increasing bulk density limits the water's ability to infiltrate the ground and the soil's ability to hold onto water. Feeding activity correlated negatively with soil bulk density and positively with volumetric water content, as lower soil bulk density and higher soil moisture led to higher rates of feeding activity across all plots.

Several studies investigating the influence of vegetation type on feeding activity in both grassland and forest communities determined there was no significant effect (Klimek et al., 2015; Hamel et al., 2007). Another study, sampling 92 agricultural grasslands, revealed that feeding activity was positively related to plant species richness, specifically grass and legumes (Birkhofer et al., 2011). Both pasture and grass margin plots were dominated by grass species, which may explain the elevated proportion of bait consumed for those land uses due to increased plant diversity increasing feeding activity through alterations of the microclimate and resources. In Chapter 1, I found that plant diversity was highest in woodlots and lowest in corn plots. Increasing plant diversity has been shown to support biodiversity and the abundance of decomposer arthropods, along with increasing their rate of decomposition (Ebeling et al., 2018; Ebeling et al., 2014). Stark differences in vegetation type and management practices between land uses indirectly or directly alter soil properties and resource availability, resulting in differences in bait removal.

Because soil arthropod activity influences the development and quality of soil, soil biotic activity may be an indicator of differences in soil quality between land use types (LaRade et al., 2012). Here, I found that the soil quality differed significantly between the four land uses. The lowest scores were found in corn plots, along with the lowest average feeding activity. Although pasture QBS-ar scores were significantly higher than those of corn, I found that both fell below the 93.7 thresholds commonly used to separate lower and higher-quality soils. The soil biological quality – arthropod index had a significant positive correlation with the average feeding activity, indicating that arthropod communities in higher-quality soils mediate higher rates of bait removal. Generally, the QBS-ar scores were lower in all land uses compared to Chapter 1; however, the patterns observed between them were similar. This is most likely due to using subterranean pitfall traps in Chapter 1 and soil cores in Chapter 2 for arthropod sampling, biasing the sampling.

Soil meso-arthropod and macro-arthropod abundance showed no significant correlations with average bait consumption; however, abundances for both were higher in pasture plots compared to corn plots. These results indicate that the soil arthropods observed here do not significantly contribute to the overall feeding activity or that the sample size was not sufficient to capture the variation in soil arthropod abundance necessary to explain these differences. Other studies have noted that earthworms and enchytraids contribute to the majority of bait removal (Vorobeichik and Bergman 2023; Gongalsky et al., 2008), and these soil organisms were present in all land uses.

Conclusion:

This study demonstrated that land use type influences soil arthropod feeding activity through interconnected factors such as soil quality, soil bulk density, and soil moisture. The reduced feeding activity in corn plots, characterized by high bulk density and monoculture systems, highlights the impacts of intensive agricultural practices on soil biota ecosystem services. In contrast, pasture and grass margin plots, with their lower bulk density and more diverse vegetation dominated by grasses, supported higher feeding activity, emphasizing the benefits of these land use types throughout the agricultural landscape. Increased soil moisture during the wetter September period may have further facilitated deeper arthropod activity during the second sampling period.

These findings support the idea of conserving soil moisture and including higher plant density within forage systems to support ecosystem services mediated by soil biota. By supporting decomposition processes, ecologically intensified systems such as forage pastures not only improve soil quality and resilience but also contribute to nutrient cycling and long-term agricultural sustainability throughout forage systems (White et al., 2023; Basche et al., 2020). Due to climate limitations of the eastern United States, the livestock industry cannot rely on forage pastures throughout the year. Nonetheless, systems dominated by annual crop fields producing year-round feed for livestock should investigate methods to introduce forages through rotation and create more circular systems integrating agricultural residues across lands once used for annual, monoculture, and linear cropping systems.

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