

Setting Fire to the Field: The Effects of a Prescribed Burn on Arthropod Abundance and Diversity in a Contiguous Grassland Field, Edge, and Forest Area

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Makhali S. Voss
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Abstract

Edges (transitional zones) can be a combination of faunas of adjacent environments or can have their own unique subset of organisms. Edges are heavily traveled by organisms looking for food sources or refuge from the elements, and they have a greater diversity in vegetation. Arthropods in or adjacent to controlled burn areas are heavily understudied; holistic studies examining insect diversity and abundance in such systems are not common. By sampling an area with an edge, a basic understanding can be gained of its contribution to diversity and its value as a reservoir for repopulation of burned areas. It was hypothesized abundance and diversity would be higher in an edge (transitional) site than in the forest or burned sites, and that within the burn area abundance would decrease but diversity would increase over time. This study was composed of six contiguous sampling sites at Arnold Air Force Base, Tullahoma, TN: two in the forest, two on the edge, and two within a field that was subject to a prescribed burn in March 2023. Each of the sampling sites was equipped with one SLAM (Sea -Land- Air Malaise) trap and five pitfall traps. These traps operated for 10 -14 days, and all sites were sampled every other month for one year. Sweep-net sampling and organic litter collection for extraction in Tullgren funnels were performed. Data was analyzed using the Shannon-Wiener diversity index, Sørensen index, and a generalized linear mixed model was created to assess significance. Edge habitats had the highest arthropod abundance, but diversity was not statistically different. SLAM trap results and precipitation were correlated with arthropod abundances and diversity values. Repopulation of the burn area was most likely coming from the edge habitat rather than another source. The controlled burn did not significantly decrease arthropod numbers or diversity. In conclusion, a study such as this one provides invaluable information on the role of edges in arthropod biodiversity and effects of anthropogenic disturbance.

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Chapter 1: Introduction & Literature Review

Arnold Air Force Base (AAFB) has been an area of ecological interest for many years. A 1997–1998 study at AAFB assessed the terrestrial invertebrate community of part of the base (Lambdin & Grant 1999). About 25 years later, arthropod community structure was evaluated over a much larger region to further basic understanding of the arthropod biota and to evaluate the quantity and quality of food for insectivorous bats. One part of the recent study was to determine the effects of prescribed burns on arthropod numbers within the burn area and adjacent habitats. Controlled burns throughout AAFB are used to keep parts of the habitat, including indigenous animals, as they are believed to have existed before western colonization.

Declines in arthropod numbers and diversity have the potential to seriously impact the ecosystem and have been repeatedly noted for insect groups (Hallmann et al. 2017, Lhoumeau et al. 2022, Liu et al. 2021, Vaz et al. 2023). This is of concern because of how crucial insects are to ecological functions, such as pollination, nutrient breakdown, decomposition, and transfer of energy (Belitz et al. 2024, Filazzola et al. 2024, Hallmann et al. 2017). Not only are insects important ecologically, but they are also important economically as well. Many crops grown around the world are dependent upon insect pollination, and agriculture plays a large role in the American economy (Losey & Vaughan 2006). Decline and destabilization of arthropod communities can cause cascade effects elsewhere within the ecosystem. Biodiversity studies are crucial to understanding potential impacts within ecosystems and should be the first step in creating conservation plans. Biodiversity studies also allow researchers to know what organisms are in the area. Many new species or new ranges for species have been determined through biodiversity research (Lhoumeau et al. 2022). As global temperatures continue to rise, it is important to understand the current status of arthropod communities and how they will be affected. The reactions among insect taxa to rising temperatures and continued urbanization are expected to be species-specific, with some expected to react poorly, while others likely will thrive (Belitz et al. 2024, Filazzola et al. 2024).

Urbanization is one of the main drivers of patchy, fragmented habitats and is believed to be closely related to climate change and shifting rates of ecological processes (Belitz et al. 2024, Svenningsen et al. 2024). One of the most concerning issues with the process of urbanization is homogenization of the landscape, which is expected to have impacts upon the local insect community (Knop 2016). As urbanization proceeds, arthropods that fit into the feeding guild of generalists may fare better in the new habitat presented by these anthropogenic changes. Urban development can cause decreases in vegetative diversity of an area (homogenization), thus negatively impacting arthropods that are in a specialist feeding guild for that area (Belitz et al. 2024, Knop 2016, Magura et al. 2010). While urbanization can support some taxa, urban forests included in the new urban land coverings are not typically sufficient for maintaining pre-development biodiversity within insect taxa (Belitz et al. 2024). Some studies have found that these urban forests are able to support significant insect biodiversity, but most have only focused on the green spaces, and do not consider the surrounding gray spaces, which are unnatural places within the urban environment (Hu et al. 2024). Research on how impending homogenization due

to urbanization will affect the arthropod community lacks consensus, and differing reactions are more than likely because of life-history traits (growth rate, migration, reproductive patterns, longevity, etc.) (Knop 2016, Magura et al. 2010). The homogeneity of urban landscapes can become an issue when considering that this relatively consistent habitat type can be more vulnerable to invasive species. As most urban environments look similar (many buildings and human dwellings, less natural landscape), it is easier for invasive species to find common ground throughout the world. Urbanization is associated with the increasing disturbance hypothesis, which predicts a decrease in diversity due to urbanization (Magura et al. 2010). Urbanization is one example where there can be anthropogenically created edges, a factor that is contributing to arthropod decline.

Conversion of forest land to agricultural use is another anthropogenic change that modifies arthropod communities. An increase in farmland and agricultural processes results in a decrease in farmland insect biodiversity (Knop 2016). Agricultural systems can dramatically shift the arthropod community as many new factors are relevant to consider in a system such as pests and crop rotation. Rural/urban or forest/agricultural gradients create edges that become of ecological interest in the face of all these changes.

Any area can contain multiple types of habitats. Often, when two different types of habitats meet, an edge is created. This edge can potentially be home to a whole new subset of organisms or could be a blended mix from the two contiguous environments. The “edge effect” is generally considered to be a fundamental ecological concept (Ries et al. 2004). This concept is often found to be associated with other ecological theories or mechanisms, such as the intermediate disturbance hypothesis, resource mapping, and ecological energy flows (Ries et al. 2004). Edge effect theories have been studied and theories associated with edges have evolved throughout the years with a stronger understanding of the evidence associated with them.

Some of the more common edge types that occur in society today are agricultural edges, natural edges, and anthropogenic edges. Due to anthropogenic changes forest edges are becoming more abundant in different regions worldwide (Harper et al. 2005, Mangura & Lovei 2020). Logging, wildfires, and agriculture all contribute to the increase in forest fragmentation. When forests become more fragmented, or patchy, the number of edge habitats increases. Edge influence has been defined as “the effect of processes (both abiotic and biotic) at the edge that result in a detectable difference in composition, structure, or function near the edge, as compared with the ecosystem on either side of the edge” (Harper et al. 2005) and is of high importance as global warming and human influence affect environments. Essentially, an edge effect occurs when the landscape moves away from the “core” of a habitat, meaning that it moves further into another landscape or habitat type (Ries et al. 2004). Both biotic and abiotic factors can be vastly different from either of the habitats adjacent to the edge area. Some of these factors are differential light intensity, soil temperature, leaf-litter accretion, wind, and humidity, all allowing for microclimatic differences that are far-reaching in their impacts (Gonzales et al. 2016, Martello et

al. 2022). The differences experienced on the edge not only affect the organisms living there, but also ecological processes. Differences in seed dispersal, migration, and decomposition with edge environments can occur (Martello et al. 2022). In the past, edges have been associated with increased abundance and diversity, suggesting they positively affect the surrounding habitat (Leopold 1933), but the effect may actually be mixed (Sheridan et al. 2020). As more research is conducted the nature of edges and their effects on habitats evolve.

Edge effects tend to differ between taxonomic groups, including insect orders (Altamirano et al. 2016, Caitano et al. 2020, Gonzalez et al. 2016). Edge effects can differ even intra-specifically, which can be attributed to different life history traits, such as behavior, mating activity, or morphological variation (Ohwaki et al. 2015). For this reason, it is difficult to assess if edge effects are beneficial or detrimental to the environment, because it typically depends on the group of organisms being studied. It has been noted that many species within the orders Coleoptera, Diptera, and Hymenoptera are positively associated with the edge, with higher abundance and diversity there rather than in adjacent sites (Gonzalez et al. 2016). As studies work their way down to genus and species level, it can be seen how the edge affects community structure in regard to species-specific changes, as is the case with leaf cutter ants. Some leaf cutter ants are known to be associated with disturbance and do well on the edge, while others are “specialist species” that are negatively affected by disturbance, defined as an event that disrupts the current condition of a habitat (Barrera et al. 2015). When considering a species’ relationship with an edge it is important to consider life history traits, morphology, physiology, and behavior of that species. The different reactions and behaviors towards edge habitats from these organisms is one of the reasons ecological concepts, such as edge effect theories, have been studied so heavily.

An important point to consider when discussing insects and edge effects is that species specialization will play a role in the degree of response. In terms of forest fragmentation and edges, specialist species tend to exhibit a stronger edge response than the generalists, due to their differing niche amplitude (Peyras et al. 2013). Studies have shown that, depending on the surrounding habitats, edges harbor many generalist species (Magura & Lovei 2020, Ramalingam & Rajan 2021, Tscharntke et al. 2002). The habitats on either side of the edge provide generalist species from both environments and they tend to thrive in the edges as the altered conditions provide them with an advantage. However, their response to edges depends on their species-specific biology. In some cases, specialist natural enemies in a habitat can attack native species (Rand et al. 2006). The movement of specialists and generalists across these edges causes changes at the community level, which can cause disruptions within ecological processes (Frost et al. 2015). For example, a habitat generalist predator will typically spill over into the adjacent habitat causing disruption within the community, particularly with insect herbivores in an agricultural environment/natural environment edge landscape (Rand et al. 2006). Generalist predator movement from a plantation across the edge to natural forest was double that of specialist predators (Frost et al. 2015). Other studies may detect similar movements but in the

opposite direction. These movements can be attributed to resource allocation. Some of this movement may be attributed to the level of tolerance to environmental degradation as well. Reactions to many ecological processes within the arthropod groups are species-specific, influenced by that organism's biology, morphology, and physiology. For this reason, it is difficult to make blanket statements about the ecological perspective of edge effects, which is why it is a topic of interest in many ecology research groups.

The edge is important in terms of predator-prey relationships. Endangered bat species can be found in AAFB. Research is currently being conducted to identify insect prey species available throughout the year for these bats to eat, as well as different habitat types the bats may prefer. The diversity of the edge species, as well as abundance, could be an important factor to note as it helps add to the biological knowledge of bats in the area. Knowing what is available for nutrition within these edges can illuminate ecological interactions in the area. Studies that assess prey availability year-round become more important when considering the impact of white nose syndrome on these bat populations. Various other predator-prey relationships can be found on the base, but the endangered status of bats makes them of key interest.

Arthropods are the ideal study group for an edge-effect research project, as they are commonly used as ecological indicators (Ramalingam & Rajan 2021, Ruan et al. 2021, Scalercio et al. 2022, Vogt et al. 2022). Different guilds and groups of insects can be used for different types of ecological indicators. In edge effect studies many researchers will choose ground beetles (Carabidae) as their study organism, as ground beetles play a key role in nutrient cycling and decomposition (Magura et al. 2017, Magura & Lovei 2020, Ohwaki et al. 2015, Peyras et al. 2013, Wiggins et al. 2007). Many studies have also used ants, as they are not only important in ecological processes but can also be used to exhibit how invasive species fit into these theories (Ramalingam & Rajan 2022). Most insects experience sensitivity to environmental changes and will alter their behavior accordingly, especially if their resources are affected (Ruan et al 2021). As this study was examining not only edge effects but also the repopulation of an area after a prescribed burn, using arthropods (more specifically insects) as study organisms provided the best insight as to how the community was and could be affected. Focusing on arthropods for this research was also beneficial because the results can add to the small body of knowledge of the decreasing insect biomass and effects of rising temperatures on arthropod abundance and diversity.

Edge species tend to be important within the ecological community, thus numerous studies have assessed the species diversity and abundance of different organisms within these unique environments. Many of these studies, though, have been species-specific or focused only on mammals (Barrera et al. 2015). No consensus on how insects holistically behave within edge environments has been reached (Martello et al. 2022, Ohwaki et al. 2015). The edges of an ecosystem have the potential to host various species because of the conditions they exhibit and the different niches they provide. Species may group in edges for multiple reasons; it could be as

simple as an open niche or it could be that the temperature and light intensity are higher in forest edges than interior forest sites, making it more suitable for certain species to dwell within (Erds et al. 2013). The edge provides a “middle ground” for many species. The size of the edges will vary between habitats.

Edge species are important for a multitude of reasons, one of which is increasing the biodiversity of an environment (Gonzalez et al. 2016). The mixing of two different types of matrices potentially creates an area that allows for more species to be present since different resources can be available. Sometimes the differing abiotic factors allow for resources to be more abundant within the edge. Some studies have found that 40% of collected insects were found only in edge environments (Gonzalez et al. 2016). Useful genes within plant species could also be conserved within these edge habitats (Lee 2018). Knowing what species are abundant on these edges and how they have adapted to the edge habitat could lead to other important findings as global issues, such as climate change, become critical. Species range edges can be crucial in understanding how an organism is adapting to climate change as the genetic makeup of individuals that live at the edge of a species range may differ from individuals in the middle of said range (Angert et al. 2020). Some researchers are examining how forest fragmentation (more specifically edge effects) and climate change affect each other. For instance, a group in Germany studying the association between edges and climate change found that plant-insect herbivore networks (PIHNs) on the edge were more robust compared to the forest interiors, which allowed them to be more resilient with rising temperatures than the forest interior PIHNs (Bahner et al. 2017).

Ranges for insects are dynamic, often due to human activity such as urbanization and long-distance transport. Climate change is more than likely a driving factor as well (Belitz et al. 2024, Filazzola et al. 2024). Biodiversity research in itself can inadvertently provide a more complete record of species that may not have been documented in the area before (Lhoumeau et al. 2022). This instance can help data scientists create models that will predict when species might arrive in certain areas. This approach is valuable in tracking invasive pest species or when tracking natural predators of the invasive pest species. As exotic and invasive species can be detrimental to some of the native species, an *a priori* knowledge of an area’s biodiversity would be useful in a predictive model before the arrival of a potential pest. For example, the All-Taxa Biodiversity project in Great Smoky Mountains National Park sprang from concern that the pending arrival of the spongy moth (*Lymantria dispar dispar*) would decimate old-growth forests before the biota were well-known. Preventative measures become more feasible with more knowledge. Models like those mentioned here would be of the utmost importance in agricultural systems worldwide.

Controlled burns are a common method of habitat preservation and conservation efforts and are useful for many other ecological tasks. While a controlled burn may be positive for one group of organisms (Kortmann et al. 2024), it has the potential to be detrimental to others (Joly et al. 2007). One group of organisms that is not heavily studied in relation to controlled burns are arthropods. Similarly to edge effect research, various studies have assessed effects of a

controlled burn on different groups of organisms but have yet to come to a general agreement on how these groups are affected. Much of arthropod reaction to controlled burns is dependent on individual life history and physiology. While a prescribed burn does have the potential to directly kill an arthropod pest, it can also indirectly damage their population numbers by incinerating material they may use to overwinter in or feed on. Fire and insects both are integral parts to many ecosystems, and it is crucial to conservation plans that the associations between them be evaluated (McCollough et al. 1998).

Insect species will have different tolerances to both human interactions and fire interactions. Most arthropod tolerance to fire is best explained by life history traits and is variable between insect guilds, as are most ecological interactions (Javal et al. 2022). For some species, using a controlled burn is an effective way of controlling population numbers, which becomes especially relevant with pest species (McCollough et al. 1998). If the fire does not directly kill the arthropod, it can change aspects of their habitat, such as the soil or vegetation cover (McCollough et al. 1998). Insects and fire are interrelated, but over time fire suppression and control policies have altered forest composition, meaning that the relationship between insects and fire also may have been altered (McCollough et al. 1998). When using prescribed burns as a method of forest or arthropod community management, it is crucial to know how arthropod species will react to it, as well as how vegetation structure will be affected. It is important to understand that insects will have different tolerance levels to degradation of their environments as well (Ruan et al. 2021). For example, species that overwinter as adults are typically abundant in disturbed habitats, meaning they have a higher tolerance for disturbed or degraded environments (Magura & Lovei 2020).

Many of the studies done on edge effects are in relation to forest fragmentation and edges created by anthropogenic sources. Notable differences were found between an anthropogenic edge and a naturally maintained edge, which have led some to develop a history-based edge effect hypothesis (Magura et al. 2017, Magura & Lovei 2020). Edges that are consistently influenced by humans do not tend to have higher arthropod abundance, as is the case found within some ground beetle (Carabidae) studies (Magura et al. 2017). A naturally occurring and maintained edge typically has higher ground beetle abundance than an anthropogenic edge. Reasons include fluctuating microclimatic and other environmental conditions within a constantly disturbed edge (Magura et al. 2017). There is not enough time for the habitat to stabilize and provide specialist species with the necessary environmental requirements for a stable population. Another biotic factor that plays a role in insect reaction to edges is body size (Magura & Lovei 2020). A trait-based analysis that focuses on body size would help to not only understand how edge effects interact with insects in general but would reveal the insect reactions to edges with differing histories. Developing trait-based analyses is the first step to understand the interspecific and intraspecific differences among insect groups. In the ground beetle study, a meta-analysis of studies in relation to edge and forest interior was performed to assess how body size impacted beetle reaction to edge effects with differing histories (natural vs. anthropogenic: continued

human intervention). Naturally maintained edges were structural barriers to smaller species while the large and medium sized species could freely move through to the forest interior (Magura & Lovei 2020). Their roles within the environment could be another important factor to consider, such as their feeding behavior (generalist, specialist, predator). Anthropogenic edges were permeable for all beetle sizes (Magura & Lovei 2020). Having a completely permeable edge could negatively affect the ecological composition of the forest interior by shifting the community structure at the base level. Edge history is an important trait to recognize in not only edge effect research but also in making decisions on how to best manage a habitat.

Environmental disturbance is a key factor in abundance and diversity of species within an environment. The intermediate disturbance hypothesis (IDH) is a well-known ecological theory that relates disturbance patterns to species abundance and diversity. The thought is that when an environment reaches an intermediate level of disturbance it will have the highest abundance and diversity that area can experience. Too little disturbance will result in lowered abundance and diversity, as will too much disturbance. This hypothesis has been heavily studied throughout the years and there is much evidence that supports the hypothesis, as well as some that refutes the hypothesis (Lazarina et al. 2019, Mercado et al. 2008, Schwillk et al. 1997). Due to the controversy surrounding IDH, some have advocated abandonment of IDH (Fox 2013), while others say that it just needs to be better understood and then remodeled (Ries et al. 2004). Studies that add to this growing body of literature are relevant as they help to understand the complex ecological interactions going on within edge environments.

An environmental disturbance often associated with the IDH is fire. Prescribed burns and wildfires are common and therefore have been studied to assess their effects on the species occupying the area. Prescribed burns have often been used to help increase biodiversity, because if they are done correctly, they are thought to create an intermediate amount of disturbance; however, many factors must be considered when using prescribed burns. One of these factors is how frequently the burns are done. Different habitats may react in different ways to controlled burns, and a longer or shorter burn interval may need to be implemented depending on the intended outcome. Some habitats benefit from long-term burn schedules, while others do better with shorter burn schedules (Lazarina et al. 2019, McCollough et al. 1998). The shorter burn schedule could benefit an area that is having a consistent pest problem. Knowing the history of the areas to be burned is important as ecological effects of forest fires are highly variable and difficult to predict (McCollough et al. 1998). More research is needed to completely understand the relationships between certain arthropod species and effects of prescribed burns, as many of the results are inconclusive or contradictory (McCollough et al. 1998).

Resource mapping refers to the distribution of an organism in relation to its commonly used resources. Species interactions, especially in the case of predator-prey relationships, are important to consider when thinking about resource mapping. Resource mapping is useful in understanding arthropod distribution within the edge and adjacent habitats (Gonzales et al. 2016,

Ries et al. 2004). Positive, negative, or neutral effects of the edge on a species of interest should consider quality of habitat (e.g., food, shelter, water). As edges can be a place of increased energy flow, more opportunities may be available for a species to get the resources that enhance fitness. More of an organism's preferred resources may be available in the edge, depending on the two matrices that compose the transitional edge, thus making that area a more useful habitat than their original environment. Individuals will be more likely to move into the edge area or reproduce more rather than the adjacent habitats. Thus, this ecological tool is significant in understanding the shifting of arthropod communities within or near edge habitats.

Edge effects have been studied in an ecological setting for over a century. Clements first wrote about "ecotones" in his work, rather than edges (Clements 1907). He was explaining what the hard line is when two different habitats meet, what is well known today as an edge. Most notably, Leopold (1933) coined the term "edge effect"; he stated that edges should hold higher abundances of animals, an idea that came from his observation of increased game animals in patchy landscapes. Odum (1953) added to this theory when he proposed mechanisms that would support higher diversity in the edge. In the late 1970s, edges were beginning to be thought of as an ecological trap in the avian research community (Gates & Gysel 1978, Ries et al. 2003, Sheridan et al. 2020). The edge was attracting high numbers of reproductive birds, where their nests then experienced higher predation and parasitism. While the theory of ecological traps has often been applied only to birds, in the past 20 years more research has been applied to other groups, such as insects. Research done on mantids found that oothecae laid on the edge did have increased predation, but not parasitism, suggesting that these mantids could also be affected by edges as ecological traps (Ries et al. 2004).

While understanding how edge effects play a role in abundance and diversity is important, edge-mediated effects are also significant. The difference between edge effects and edge-mediated effects is that the latter focuses on how the edges influence the ecological processes in the area (Anastasio et al. 2021, Dauber & Wolter 2004). Those who focus on edge-mediated effects are able to better assess cross-edge spillover, which includes mobile natural enemies and competitors (Rand et al. 2006). This process is important because it has the ability to drastically affect the arthropod community structure, which can cause changes in the overall community structure over time.

With edge habitats it is inevitable that spillover of species happens across the habitats. While this may not be an issue for a habitat that is stable, it can be detrimental for others that may have been recently affected by an environmental change or ones that are sensitive to changes within their community structure (Rand et al. 2006). Spillover via edges can be understood by resource mapping, and affected species are likely to be more abundant where resources are more abundant. However, lack of spillover could be due to a behavioral response rather than an ecological response (Moron 2019).

Species-specific responses to edges may be driven in part by the plant community composition within the edge. Given that food webs are built upon plants and other primary producers, it is thought that species response to edges should be directed by plant community composition (Kung'u et al. 2023, Rand et al. 2006, Ulyshen et al. 2010, Wimp et al. 2021). Some attribute the richness, or uniqueness of an environment, to the richness of the surrounding vegetation of the sample area (Vlach et al. 2010). The plant community of the edge can be vastly different from that of the interior habitat, and this can again be attributed to differing climatic conditions, soil characteristics and human influences. Due to the differences in plant communities on the edge, herbivory can also be affected. Plants within these edges have a different composition in terms of water content, carbon, and nitrogen than those in the interior (Rosetti et al. 2014). One of the larger driving forces behind arthropod community structure on the edge is plant community composition and condition (Ramalingam & Rajan 2021). If the plant community within these edges is stunted or declining, then insect diversity will likely be lower (Anastasio et al. 2021). The level of degradation within the plant community and level of disturbance is an important factor to consider in terms of insect diversity and abundance (Ramalingam & Rajan 2021). This concept ties back to the concepts of resource mapping and the intermediate disturbance hypothesis; if the arthropod food source or habitat source is in a concentrated area, they will frequent that area. Disturbed forests tend to have more natural regeneration and increasing vascular plant diversity, which produce more habitat and potential food sources for arthropods in the area (Kortmann et al. 2024).

Seasonality is a large driver in insect behavior as they are ectothermic organisms. The majority of insects overwinter or die once the temperature becomes too cold for them to function, and edges can be important in this biological process. Many insect species that prefer open land tend to overwinter in field edges, and one of the reasons is believed to be the temperature change associated with seasonality (Ohwaki et al. 2015). For some arthropods, two different hosts are needed. The two common patterns are that the juvenile needs a different environment than the adult, as in the case of dragonflies; or the juvenile belongs to a different feeding guild than the adult, as is the case with most holometabolous insects. Disturbed areas can be important for some overwintering insects, as for some species of Carabidae (Mangura & Lovei 2020). Fortunately for some plant hoppers, but unfortunately for vineyard owners in Italy, the common vineyard pest *Erasmoneura vulnerata* Fitch is known to overwinter in the edge, then infest the vines when they begin growth again (Duso et al. 2020). Cotton mites in Australia are often found overwintering on weeds in the edge of the fields, rather than the interior, and this is attributed to nutritional value of the host plants (Wilson 1995). Many different arthropods from various taxonomic groups use edges as an overwintering site.

Arthropods are heavily influenced by abiotic factors within their habitats. One of the more important factors, other than temperature, is precipitation. As with other aspects surrounding edge effect research and arthropod research in general, many studies have contradictory results in regard to how arthropods are affected by precipitation (Belitz et al. 2024). Regardless, many have

found a relationship between precipitation and arthropod abundance (Chen et al. 2019, Dalton et al. 2023).

Edge areas are susceptible to invasive species. Depending on the type of edge, that area can be severely degraded, allowing for an opportunistic invasive species to colonize it. In India, restoration of thousands of hectares of fragmented lands is underway to limit an invasive ant species that actively colonizes degraded edges (Ramalingam & Rajan 2021). Both invasive and introduced species can change the community composition and structure (Anastasio et al. 2021, Cardenas & Buddle 2008, Vlach et al. 2010). In the previously mentioned example, the invasive ant species did not negatively impact the existing arthropod community structure, but the same species is known to decimate communities in other areas (Ramalingam & Rajan 2021). Another invasive ant, the Argentine ant (*Linepithema humile* Mayr), displaces native ants once they become established in the community (Holway 2004). The displacement of native ants increases the amount of citrus disease, as the Argentine ant protects the disease-carrying citrus psyllid for its honeydew, creating a mutualism between two problematic pests (Anastasio et al. 2021). In some cases, the introduction of a native species has been known to vertically shift an entire group's spatial distribution. In Hawaii, invasive rat species in an area were removed and subsequently the arthropod biomass shifted, in turn causing the avian community to shift (Rankin et al. 2018). One species can have major effects on an environment. It is important to consider introduced species when looking at species diversity as they may be dominant within their community structure (Cardenas & Buddle 2008). Introduced or invasive species may not only be important in terms of community structure, but also when considering the disturbance and anthropogenic changes as well (IDH, prescribed burns, etc.).

The hexapod class Collembola (springtails) occupies nearly every terrestrial niche. Most species are soil and leaf-litter dwellers, but a large fraction climb vegetation, live in trees or occupy the burrows and nests of other animals. Springtails are the most abundant hexapods and are essential in certain ecological processes, such as nutrient turnover and decomposition of leaf-litter (Bernard & Felderhoff 2007, Felderhoff et al. 2010). They can be used as indicators of overall soil quality (Milano et al. 2018). Collembola provide valuable insight to abundance and diversity changes because, along with mites (Acari), they can make up approximately 95% of the soil micro-arthropods (Milano et al. 2018). Collembola are ideal organisms to use when quantifying changes throughout the year or from an event that changes the landscape because they are so abundant.

With Collembola being crucial within ecological processes, it is important to understand how fire affects their abundance and diversity. Shifts in the collembolan community following fire are dependent on time since the fire disturbance, burn intensity and species-specific traits (Hubener et al. 2011). Fire intensity is a factor in using prescribed burns as a management technique, as most Collembola can be found within the leaf-litter; eliminating habitat and food sources can reduce their numbers (Hubener et al. 2011). Controlled burns also may result in the appearance

of different species in the burn area (Shaw 1997). Low-intensity burns can drastically decrease micro-arthropod density, number of taxa, and diversity, but they tend to rebound to their previous community structure within a year (Çakır et al. 2023)

The importance of soil-dwelling arthropods is recognized widely by researchers, and yet many of these groups are understudied. For Collembola, one reason is the comparative lack of molecular information for species diagnosis (Porco et al. 2013, Recuero et al. 2024). This problem creates multiple knowledge gaps, which leads to problems such as incomplete taxonomy, misidentification of a species, or misunderstanding an ecological process.

Soil-dwelling organisms are likely to be affected by rising temperatures (Xie et al. 2023, Zhang et al. 2024). For some species it is expected to increase fitness while being detrimental for others (Xie et al. 2023). By knowing what species are present in the three habitat types of this study, future changes can be better understood. Documentation of these species provides baseline data that can be used for comparisons in years to come.

AAFB is located about 113 kilometers away from Nashville and is in both Coffee and Franklin Counties. To ensure that biodiversity and natural resources are preserved, the Department of Defense (DoD) has supported many ecological studies. AAFB is the ideal setting for a study such as this, as it has high biodiversity as well as species that are not always found in other areas of Tennessee (Vlach et al. 2010, Wiggins et al. 2007). AAFB is also home to endangered and threatened insect-eating bat species; a study that catalogs a holistic view of the prey available could be beneficial to future bat protection or research (Samoray et al. 2019). Prey habitat can also be linked to bat foraging strategies.

This study assessed how edge effects are holistically affecting the arthropods in the area, and how an environmental disturbance, such as a controlled burn, would affect arthropod numbers. The site used at AAFB for this project was ideal to assess both of these ecological processes, as it had both a transitional edge zone and a recent controlled burn. It was hypothesized that abundance and diversity of arthropods would increase within the transitional edge zone. It was also hypothesized that arthropod diversity would increase after the controlled burn and that fire would destabilize the arthropod community long enough for new arthropod families to come to the area, thus increasing diversity. Arthropod abundance was hypothesized to decrease after the fire (May) as the fire would have a deleterious effect on arthropod community.

Chapter 2: Materials & Methods

Site Description

This study was conducted at Arnold Air Force Base (AAFB), which occupies 15,816 ha of varied habitats and vegetation types (Comprehensive Report 2021). The base is located in middle Tennessee and south of Nashville, adjacent to the towns of Tullahoma and Manchester. A large portion of base property is undeveloped and available for seasonal hunting. The site chosen for this study was one of many from a large overlying, intensive survey of arthropod diversity and abundance that was focused on a multitude of environment types

The study site for the research described in this thesis was ideal for this research as it had the successional gradient needed to test edge effects in the natural environment. The study site had six contiguous sites composed of three habitat types (two sites per habitat type). The habitat types were forest, edge, and field where the controlled burn took place in mid-March 2023 (Table 1). At the north end of the study area is the grassland field that experienced significant regrowth after the burn, dominated by herbaceous plants. The field blends into a transitional zone, or edge. The edge is composed mainly of thick new growth and has a high diversity of plant life that includes shrubs, grasses, and tree saplings. This edge leads into a secondary hardwood forest dominated by oak, maple and hickory. The understory of this forest was controlled burned in 2021 but has not had another prescribed burn since and is not on a consistent burn schedule. The field portion within this site has received prescribed burns every 2-4 years since 1997. The biologists employed by AAFB use the controlled burns as a method to return the habitat to what it was before European colonization to maintain unique biota native to the area.

Study Site Abiotic Conditions

AAFB received a total of 1.39 m of precipitation in 2023 (Fig. 1). The warmest month was July with a mean temperature of 25.38 C and the coldest month was December with a mean temperature of 7.77 C (NOAA Accessed 19 January 2024) (Fig. 2). The Koppen Climate System classifies the Tullahoma, TN climate as Cfa, which is humid subtropical (Koppen Climate System Accessed 19 January 2024). Cfa environments typically experience warm to hot summers with cold to mild winters and rainfall can be expected at any time during the year.

The soil within the study area is primarily composed of Dickson silt loam. Within the forested area a 2-ha area composed of Guthrie silt loam is present. Within the field, a small portion is composed of Greendale cherty silt loam (USDA Web Soil Survey 2024).

Treatments

The coordinates for the three treatments (forest, edge, and field) are given in Table 1 and a map of the area is provided in Figure 3. Each of the three habitats had two separate sites, each with a full complement of collection methods. An unimproved dirt access road ran approximately north to south between the three pairs of sites.

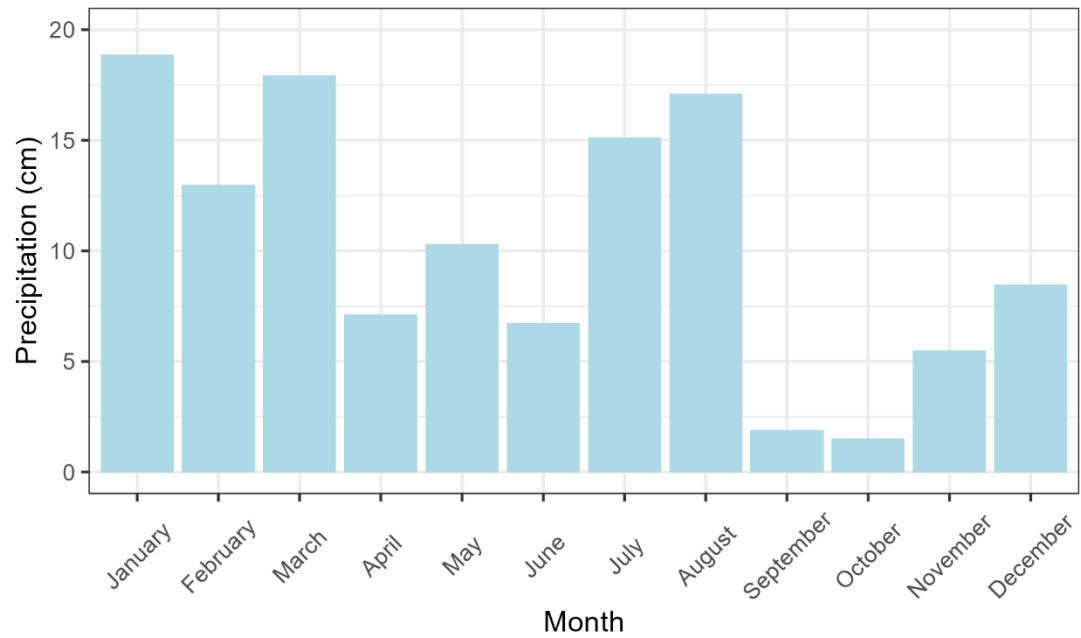


Figure 1: Total precipitation (cm) per month, Arnold Air Force Base, 2023.

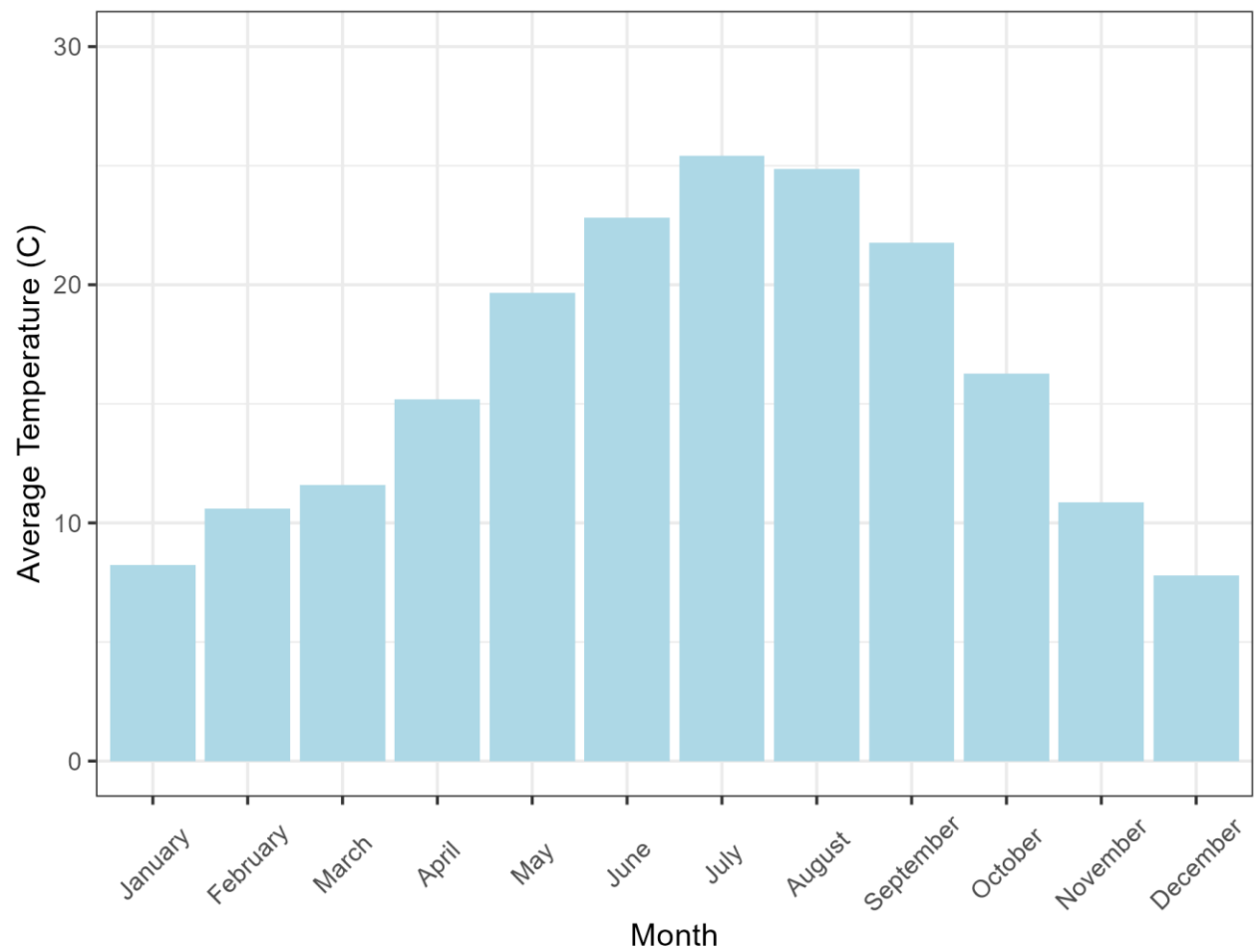


Figure 2: Average temperature (C) per month, Arnold Air Force Base, 2023.

Table 1: Habitat type and coordinates for the six contiguous sites of this study, Arnold Air Force Base, 2023.

Site Number	Habitat Type	Coordinates
1	Forest	35.34789, -86.13016
2	Forest	35.34841, -86.13078
3	Edge	35.34947, -86.12903
4	Edge	35.34924, -86.1298
5	Field/ Burn Zone	35.34981, -86.12894
6	Field/ Burn Zone	35.34985, -86.12971



Figure 3: Map of the study sites with distances, created via Google Earth, Arnold Air Force Base, 2023.

All collections were done on foot to avoid vehicle influences. Controlled burns are dependent on the weather conditions, so short notice was given as to when the burn would take place. For this reason, March was split into two samples; forest and edge habitat traps were placed before, during, and after the burn. SLAM (Sea-Land-Air-Malaise) and pitfall traps were placed in the newly burned field habitat about one week after the fire took place. All active sampling took place when the burned field habitat passive sampling was collected.

Sample Collection & Preservation

A variety of sampling techniques were used within this research project, as the aim was to assess biodiversity within the different habitat types. Samples were collected in 2023. Sampling began in January and occurred every other month, ending in November. Sampling dates for each month are given in Table 2. Both passive and active sampling were utilized for this study. Active sampling consisted of sweep-net samples and collection of leaf-litter to process in Tullgren funnels. Passive sampling consisted of SLAM traps and pitfall traps (Fig. 4). One SLAM trap was placed at each site and left deployed for 14 days (January traps were deployed for 10 days due to weather conditions). Insects entering SLAM traps fell into bottles charged with 95% ethyl alcohol.

SLAM traps are a modified version of a Malaise trap, in use since 1975 for their ease of use and high consistency in composition of higher taxa collected (Hallmann et al. 2017, Skvarla et al. 2020). They have been known to outcompete other traps in catching certain families, such as Cerambycidae (Miller & Crowe 2022). While they are able to catch larger specimens of orders such as Lepidoptera and Orthoptera, they are not the best for those orders specifically, and therefore in a biodiversity study they work best when paired with other trapping methods (Kortmann et al. 2024, Miller & Crowe 2022, Skvarla et al. 2020).

Pitfall traps were placed near each slam trap. Each site had five pitfall traps, placed in a transect with 3 m between traps. These traps were charged with a 1:1 mixture of 95% ethyl alcohol and propylene glycol and were deployed for the same length of time as SLAM traps. Pitfall samples were pooled to form one sample for each site. When pitfalls were not in use the cups were capped and plastic lids were flipped to ensure no organisms were caught during non-sampling months. This approach also ensured that the pitfall locations remained the same for all sampling periods.

Active sampling was conducted when the SLAM traps and pitfall traps were collected. Approximately 4 L of surface leaf-litter and other organic detritus were collected per site. Organisms were extracted with Tullgren funnels into 95% ethanol.

Sweep-net samples were collected for each sampling session. Each site had five sweep-net samples per sampling month. One sample consisted of 20 passes with the sweep-net, for a total of 100 sweep-net passes from each site per sampled month. Each sweep-net sample (20 sweep-

Table 2: Sampling dates, Arnold Air Force Base, 2023.

Month	Set Traps Out	Retrieved Traps & Active Sampling Dates*
January	January 19	January 30
March (Forest & Edge)	March 6	March 18
March (Field)	March 18	March 29
May	May 8	May 22
July	July 12	July 26
September	September 13	September 27
November	November 15	November 29

*Active sampling is sweep-net samples and leaf-litter samples; passive sampling are pitfall traps and SLAM traps.



Figure 4: Pitfall trap (left) placed in a transect near SLAM (Sea -Land- Air Malaise) trap (right), Arnold Air Force Base, 2023.

net passes) was emptied into its own Ziploc bag. These samples were transported back to the UTK campus, where they were frozen until sorting of the catch.

All identifications were done using appropriate dichotomous keys and manuals (Covell 1984, Evans 2014, Goulet & Huber 1993, Marshall 2013). Most specimens were identified to family level and counted. Some specimens were not identified family due to difficulty of identification, including most micro-lepidopterans, most mites (Acari), and juvenile and male Coccoidea. Larval forms and nymphs were not included in statistical analyses. Micro-lepidopterans (< 5 mm) and scale insects were included in the counts of abundance and these groups as a whole were included in calculating diversity. Macro-lepidopterans were identified to family.

Missing Data

Passive sampling methods were placed for 14 days for all sampling sessions, except January, when they were placed for 10 days due to weather conditions. Upon collection of sampling methods in January, it was discovered that the site 1 burn site SLAM trap had been blown over, but the jar still contained specimens. The other burn site SLAM trap was missing a key component, and the temporary solution was not adequate, resulting in only a single mite being collected. In March one of the burn site SLAM traps was blown loose by strong winds, resulting in no specimens being collected from that site. One of the edge site sweep-net samples from March did not collect any specimens.

Statistical Analyses

Diversity was measured using the Shannon-Wiener (H') equation for each habitat and each sampling session. The Shannon-Wiener diversity index formula is $H = -\sum p_i * \ln(p_i)$. As the sites within the same habitats acted as biological replicates, their data were analyzed together by habitat type (forest, edge, and field). The same process was followed for assessing abundance. To test for differences between habitats and trapping methods the Sørensen index was used to calculate a percentage of heterogeneity. The equation used was $C_s = 2a/(2a+b+c)$, where a is representative of the number of species common to both habitats being tested, b is representative of the number of species collected only in one habitat type, and c is representative of the number of species collected only in the other habitat type. Many times, Sørensen values are presented as a measure of dissimilarity; for this project it is presented in terms of similarity for ease of reading. The value calculated from the equation can be multiplied by 100 and presented as a percentage of similarity between the two chosen habitat types. For overall percentages all of the families and groups collected for the trapping methods were used. Each percentage was rounded to the nearest tenth. To assess species richness with relation to minimum sample size, rarefaction curves were generated for each sampling type. Any sample that did not have more than two values (two different families captured) was excluded as the vegan package in R requires at least two values to run the rarefaction process. Family accumulation curves were created in R, and

then plotted against the Michaelis-Menten model to estimate maximum family richness. Family richness numbers were rounded to the nearest whole number. While this model is typically used in association with enzymatic relationships, some groups are applying it within insect research (Eloi et al. 2020). The Michaelis-Menten equation will provide a K value as well, and this value can be used to assess sampling efforts (if more samples need to be taken to get close to the theoretical maximum family richness).

Venn diagrams were generated to help visualize repopulation of the burned area. The Venn diagrams were created using the shared groups of arthropods between the habitats as well as arthropod groups that were found only in one habitat type. Coupled with the test of heterogeneity, Venn diagrams provide valuable information on similarity between habitats. Venn diagrams were created within the website Canva and exported as PDFs.

Graphing and statistical analyses were done in both R studio version 4.3.2 and JMP Pro 17.0. Packages used in R were ggplot2, vegan, and plotly. To test for statistical significance between habitat type and month, a generalized linear mixed model (GLMM) was created within JMP Pro. Arthropod abundance or arthropod diversity was used as the dependent variable; habitat type and sampling month were used as the independent variables. An alpha value of 0.05 was used for all tests and models. Distribution was strongly skewed for all sampling methods, and so Poisson distribution was used when doing statistical analyses. H' diversity values, family richness, and the Michaelis-Menten model were calculated in R studio. Results were first assessed by habitat totals, increasing number of sampling events for statistical measures. An average of all three habitat types was taken for total mean arthropod abundance and mean H' diversity values. Then results were calculated by grouping month and habitat, and separating sampling method to assess sampling month influence on abundance or diversity.

Correlation of arthropod abundance and precipitation was tested. Information from the AAFB weather station on monthly precipitation for each sampling month was compared to arthropod abundance. This comparison was done for individual sampling methods. The null hypothesis was that no significant correlation occurred between the amount of precipitation in a sampling month and the number of captured arthropods. The alternative hypothesis was that precipitation and arthropod abundance would be significantly correlated. Correlation of precipitation with diversity values was measured the same as with abundances, keeping the same null and alternative hypotheses.

Collembola - May

Collembola were the second most abundant taxonomic group collected in this study. Due to high abundance and diversity, the specimens collected in May were looked at in-depth. Therefore, the May data were used as a case study for Collembola diversity and abundance. Specimens were identified to genus, and most to species (Bellinger et al. 2024, Christiansen & Bellinger 1998).

Ecological statistics were determined as given above. The May collections were made one month after the controlled burn.

Collembola from all trapping types were assessed for May. Numbers were much higher for pitfall traps and leaf-litter samples than for SLAM traps. Sweep-net samples did not include any Collembola. All graphics and analyses for this chapter were performed in R studio; packages used were *vegan* and *plotly*. Analyses included Shannon diversity measures (H'), abundance counts, and a test of heterogeneity using the Sørensen index.

Chapter 3: Results

Abundance & Diversity

A total of 62,503 specimens in 26 orders and 175 families were collected. Of that total, 322 specimens were larvae or nymphs that were not used in statistical analyses. Bycatch consisted of voles, shrews, and salamanders. Data analyzed by habitat are presented in Table 3 and Figure 5. The values were analyzed by habitat, rather than totals by traps and month as the rest of the analyses were. This data analysis allows for statistics to be calculated with a higher N value than the other analyses. None of the variables tested showed significance when analyzed by habitat type ($p \geq 0.05$; $df = 2$).

Other than Acari (mites) (27,015), the most abundant higher taxon was Collembola (17,940), followed by Diptera (5,864), of which 1,474 were Cecidomyiidae. Hymenoptera also were abundant (4,244); the most abundant family was Formicidae (2,871). A sunburst plot is provided to present diversity in Figure 6, and an interactive version is provided at the following link: https://github.com/Mvoss09/interactive_sunburst/blob/75d1f94f186972397c8cfa0466ddecfc9c5fa88c/sunburstoverall.html

A total of 8,283 arthropods were collected in SLAM traps (Fig. 7); 2,841 were caught in the forest habitat, 3,187 were caught in the edge habitat, and 2,255 were caught in the field habitat. Peak abundance was recorded in May with a total of 2,457 arthropods caught in all three habitats. Abundance was significantly different between habitat types (p -value $< .0001$, $df = 2$; $\chi^2 = 145.49$) and sampling dates (p -value $< .0001$, $df = 5$; $\chi^2 = 678.55$). Arthropod abundance for forest sites increased from January to July, then decreased. This general trend was similar for edge habitats as well. Field habitats differed by having two peaks; the first peak for field arthropod abundance in the SLAM traps occurred in May, and the second in September. SLAM traps had H' diversity values above 2.0 for almost the whole year, except they were below 2.0 in January for the field habitat and in March for the forest habitat (Fig. 8). The decrease in the forest habitats for March was likely due to extensive flooding of those sites that month. The forest and field habitat types had their highest diversity in the month of May, whereas the edge habitat experienced highest diversity in September. Neither sampling month (p -value = 0.9693; $df = 5$; $\chi^2 = 0.91$) nor habitat type (p -value = 0.9860; $df = 2$; $\chi^2 = 0.03$) was statistically significant for diversity in SLAM trap catches. The rarefaction curves for SLAM traps (Fig. 9) have low family richness when compared against the minimum sample size. Sampling provided the highest family richness for SLAM traps (Fig. 10). The family accumulation curve was plotted against the Michaelis-Menten model to see where the curve would theoretically level, thus providing maximum family richness (Appendix 1). For SLAM traps maximum family richness is 185 families with a calculated K value of 6.6.

Pitfall traps caught a total of 22,422 arthropods (Fig. 11), with 6,916 caught in the forest habitat, 9,105 in the edge habitat, and 6,401 in the field habitat. Highest abundance was recorded in May

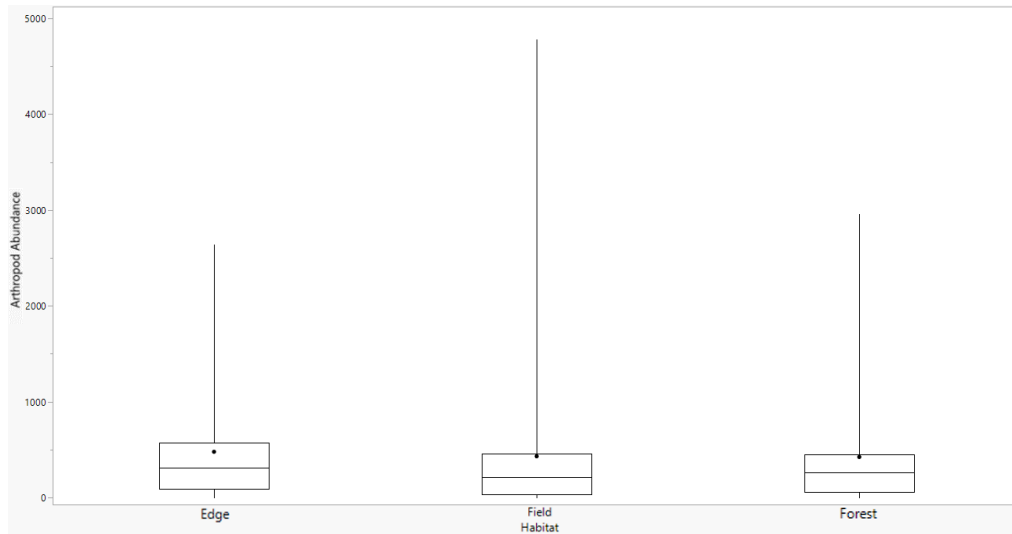
Table 3: Statistics calculated using number of sampling events and habitat totals, Arnold Air Force Base, 2023.

<u>Sweep Netting</u>				
Statistic	Forest (n = 12 events)	Edge (n = 11 events)	Field (n = 12 events)	Total (n = 35 events)
No. Orders	9	8	10	11
No. Unique Orders	1	0	0	1
No. Families	40	48	52	81
No. Unique Families	13	14	16	43
Mean No. Specimens	17.25	30.18	26.42	24.46
Mean Shannon-Weiner Diversity Index	1.82	1.46	1.80	1.69
<u>SLAM traps</u>				
Statistic	Forest (n = 12 events)	Edge (n = 12 events)	Field (n = 10 events)	Total (n = 34 events)
No. Orders	16	13	15	18
No. Unique Orders	3	0	1	4
No. Families	112	121	116	160
No. Unique Families	19	12	17	48
Mean No. Specimens	236.83	265.58	225.5	243.61
Mean Shannon-Weiner Diversity Index	2.50	2.61	2.64	2.58
<u>Leaf-litter Samples</u>				
Statistic	Forest (n = 12 events)	Edge (n = 12 events)	Field (n = 12 events)	Total (n = 36 events)
No. Orders	13	14	12	16
No. Unique Orders	1	2	1	4
No. Families	23	28	23	39
No. Unique Families	4	9	4	17
Mean No. Specimens	829.67	778.50	940.17	849.44
Mean Shannon-Weiner Diversity Index	0.68	0.73	0.82	0.74
<u>Pitfall Traps</u>				
Statistic	Forest (n = 12 events)	Edge (n = 12 events)	Field (n = 12 events)	Total (n = 36 events)
No. Orders	18	19	15	21
No. Unique Orders	2	3	0	5
No. Families	77	91	61	110
No. Unique Families	10	22	9	41
Mean No. Specimens	576.67	761.08	576.67	624.06

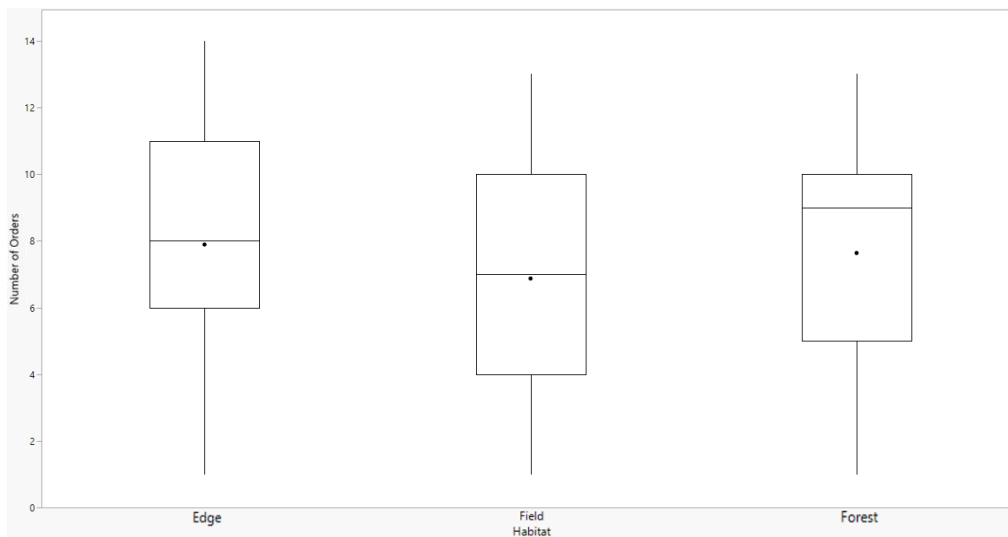
Table 3: Continued

Mean Shannon-Weiner Diversity Index	2.13	2.05	1.86	2.01
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<u>All Collection Methods</u>				
Statistic	Forest (n = 48 events)	Edge (n = 47 events)	Field (n = 46 events)	Total (n = 141 events)
No. Orders	22	23	20	26
No. Unique Orders	3	4	1	8
No. Families	140	159	145	197
No. Unique Families	19	19	16	54
Mean No. Specimens	413.54	465.85	439.33	439.39
Mean Shannon-Weiner Diversity Index	1.78	1.71	1.78	1.76

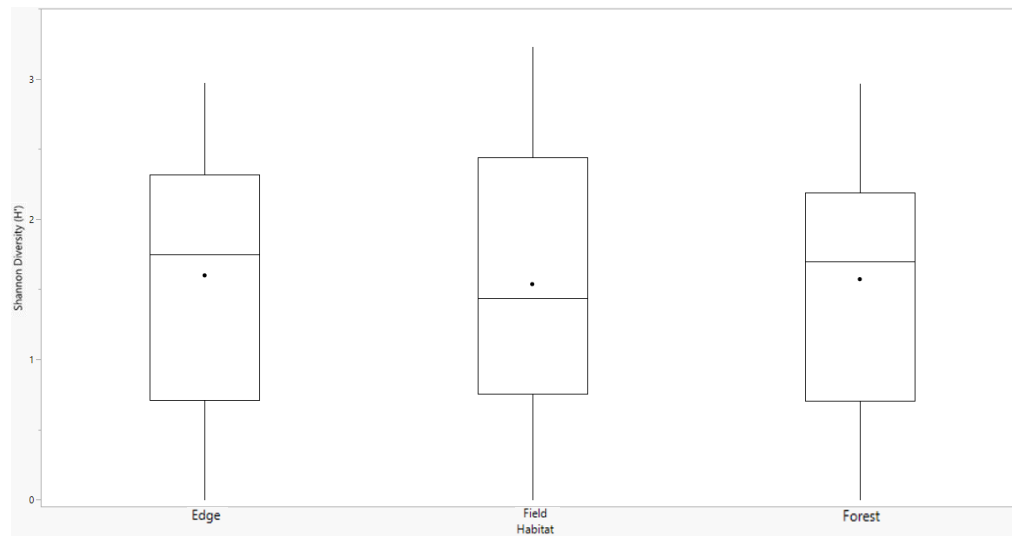


a.) Box-and-whisker plot of total arthropod abundance by habitat, Arnold Air Force Base, 2023.

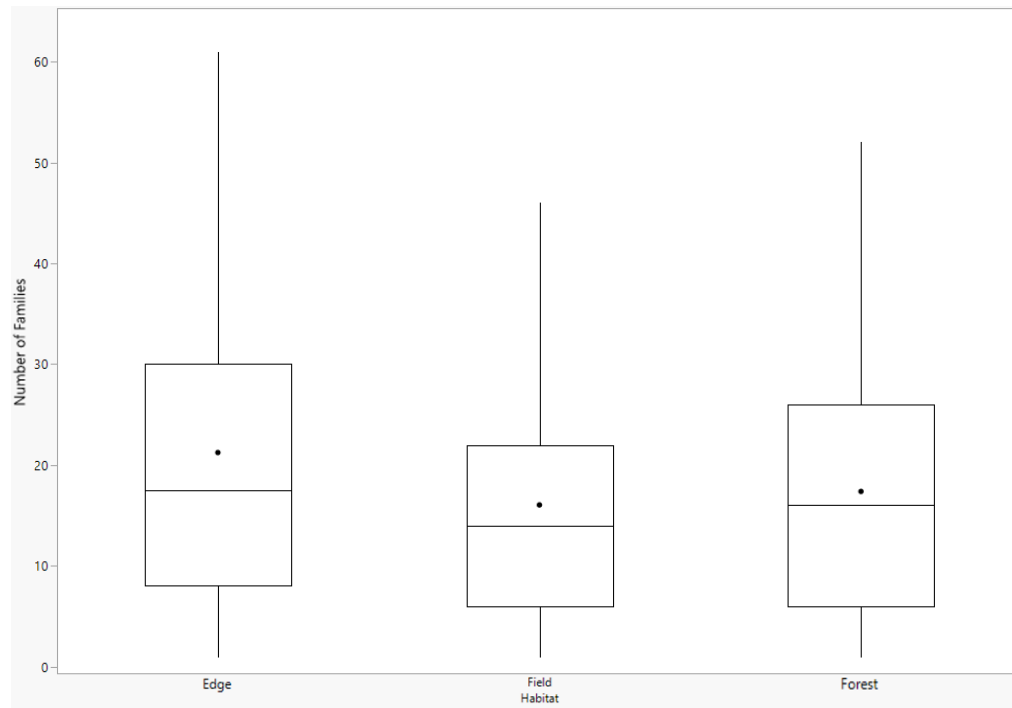


b.) Box-and-whisker plot of total number of orders by habitat, Arnold Air Force Base, 2023.

Figure 5: Box-and-whisker plots with data analyzed by habitats, Arnold Air Force Base, 2023. Mean denoted by ●.

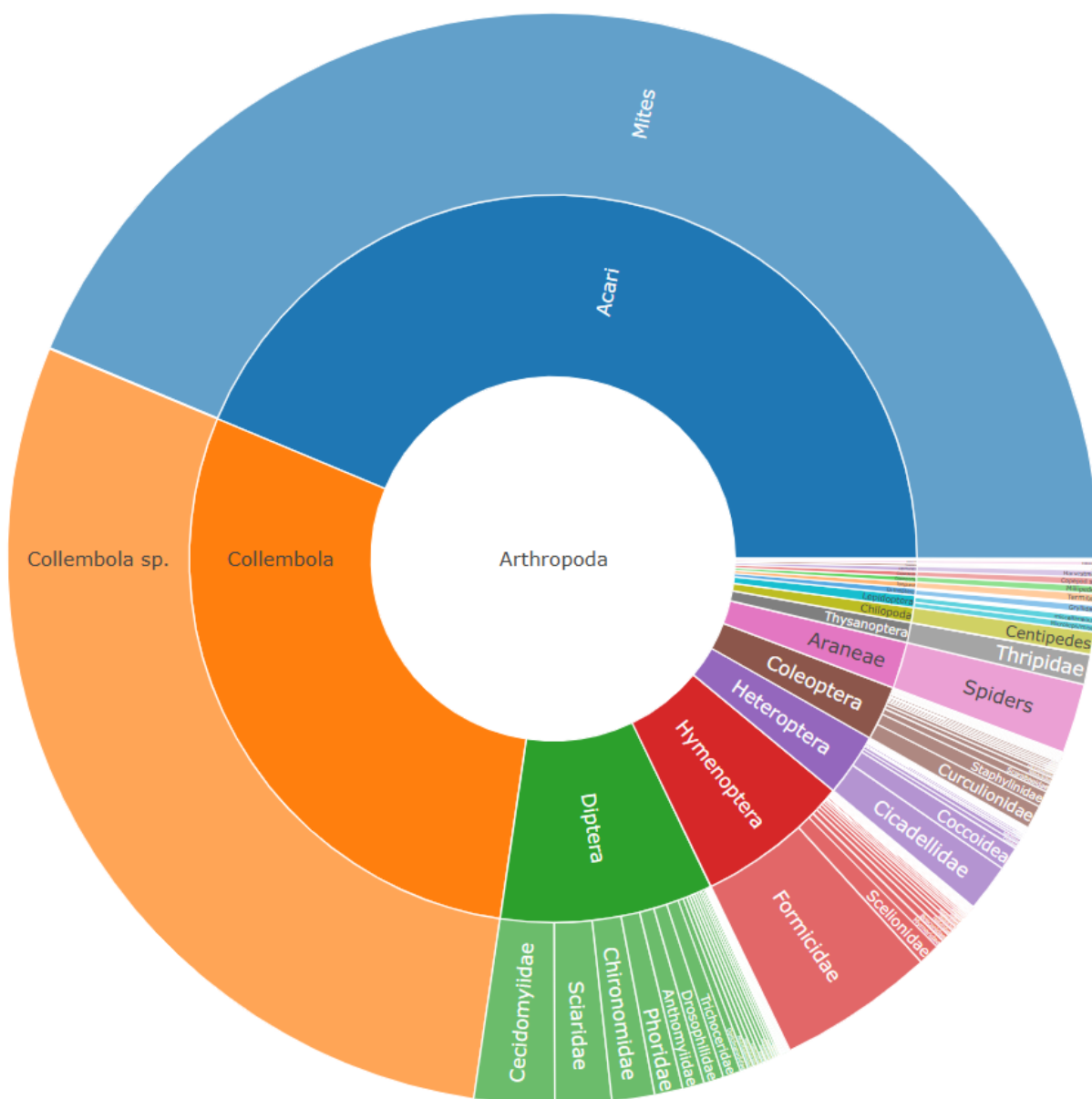


c.) Box-and-whisker plot of Shannon diversity value by habitat, Arnold Air Force Base, 2023.



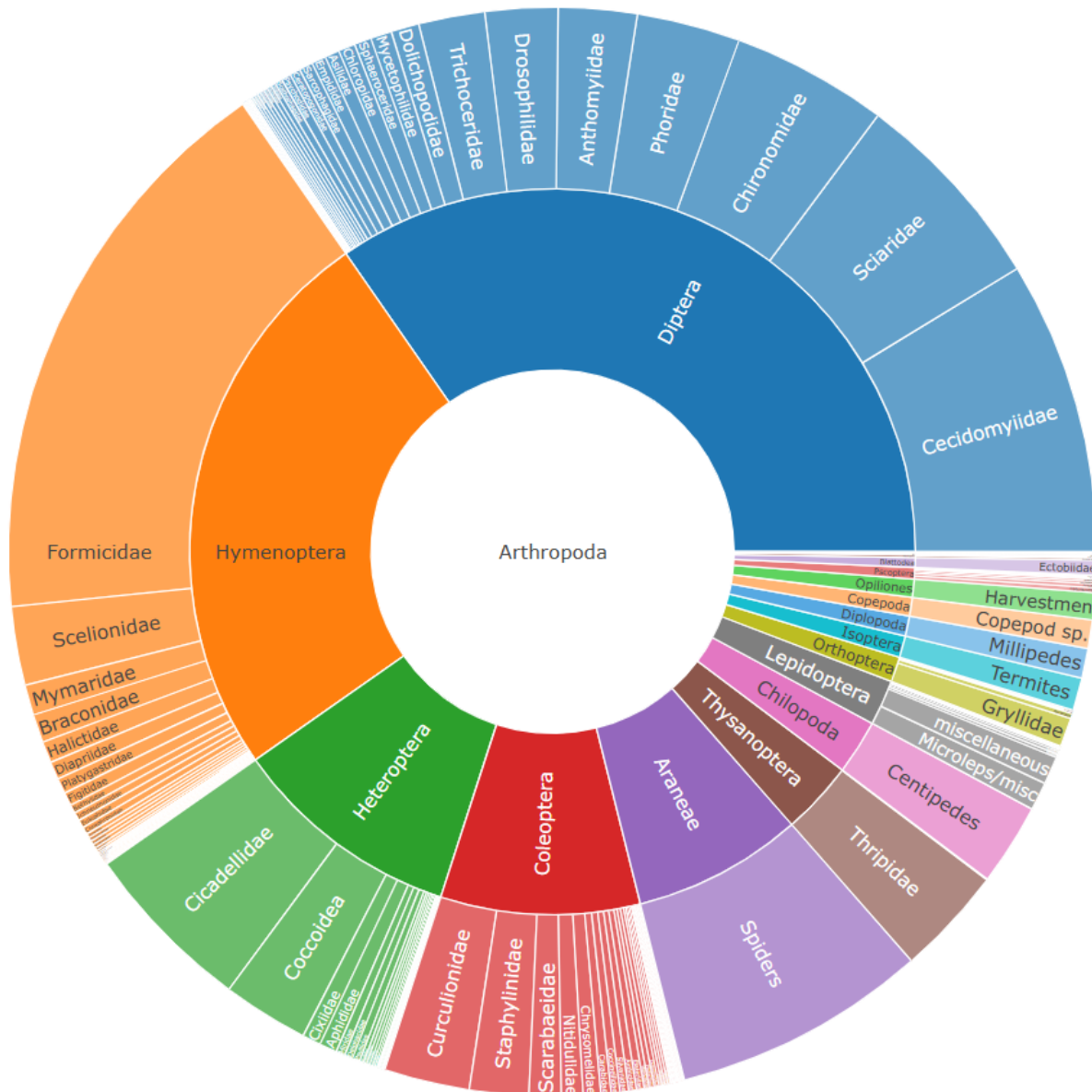
d.) Box-and-whisker plot of number of families by habitat, Arnold Air Force Base, 2023.

Figure 5 (continued)



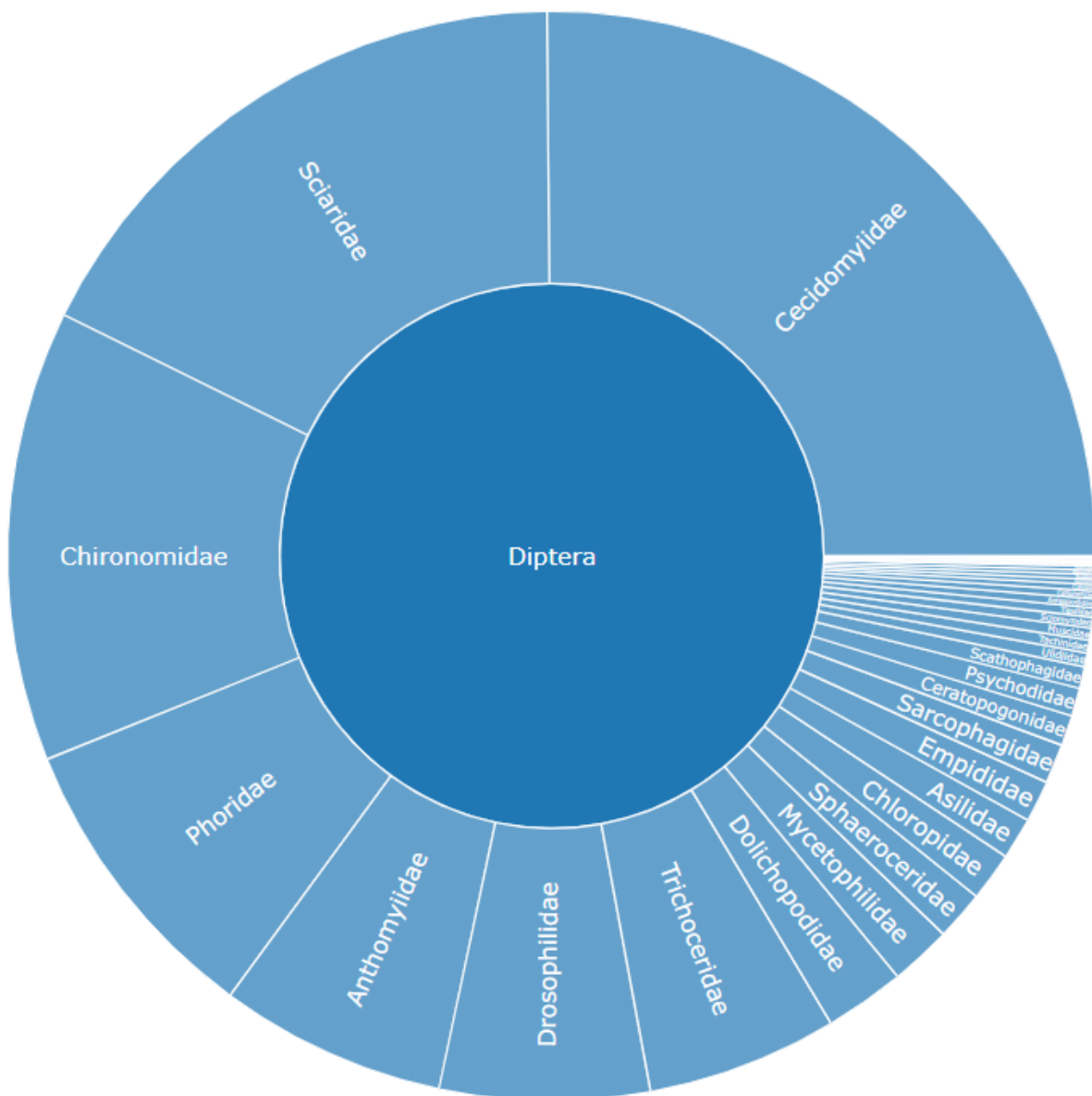
a.) Sunburst plot of arthropod orders and families.

Figure 6: Sunburst plots of diversity from collected specimens, Arnold Air Force Base, 2023.



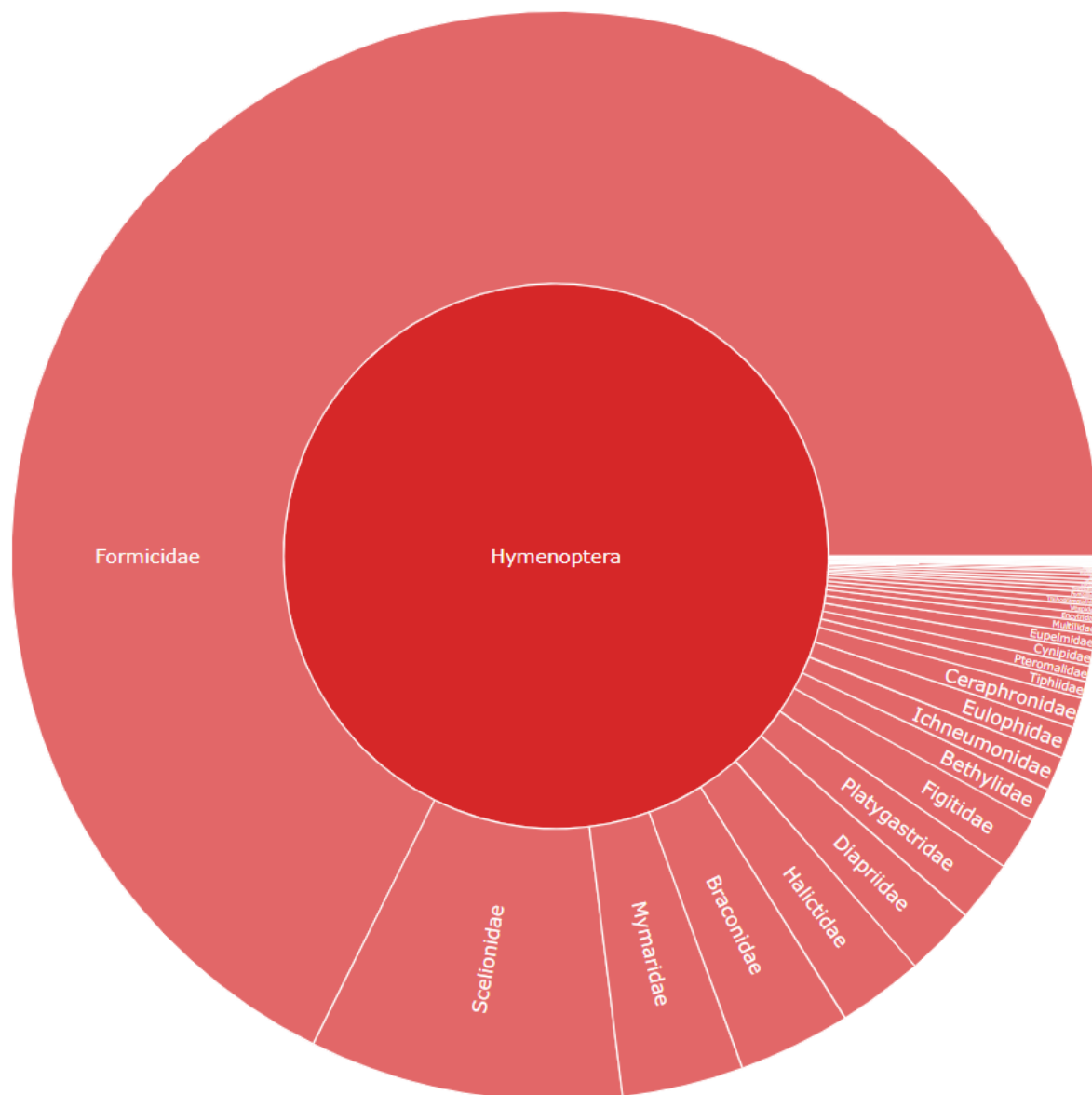
b.) Sunburst without Collembola or mite data.

Figure 6 (continued)



c.) Sunburst of the most abundant insect order, Diptera.

Figure 6 (continued)



d.) Sunburst of the second most abundant insect order, Hymenoptera.

Figure 6 (continued)

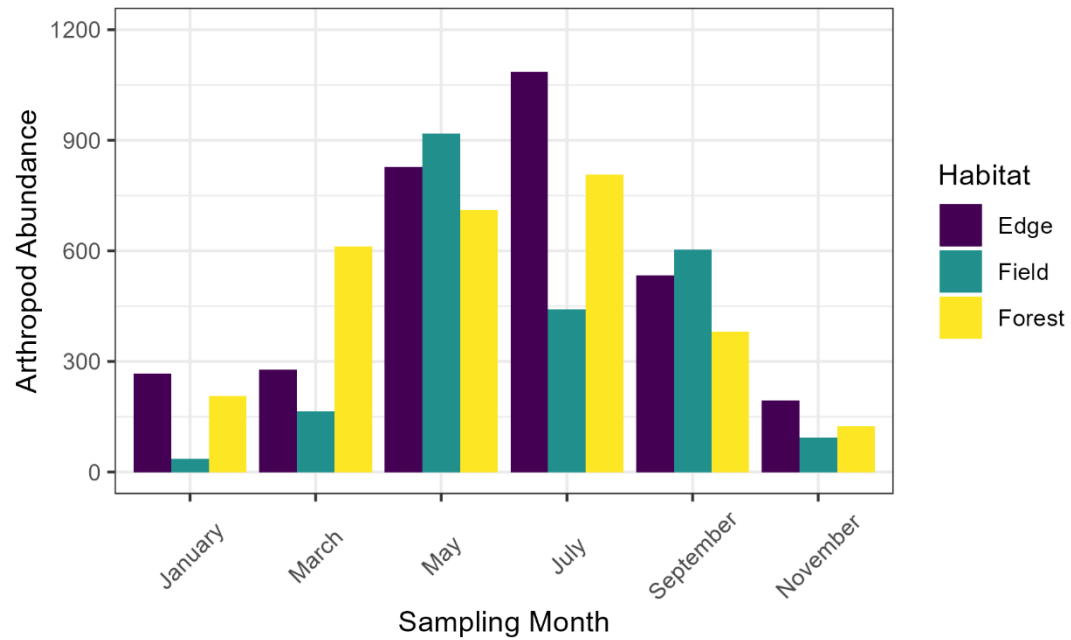


Figure 7: Arthropod abundance in SLAM traps for each month by habitat type, Arnold Air Force Base, 2023.

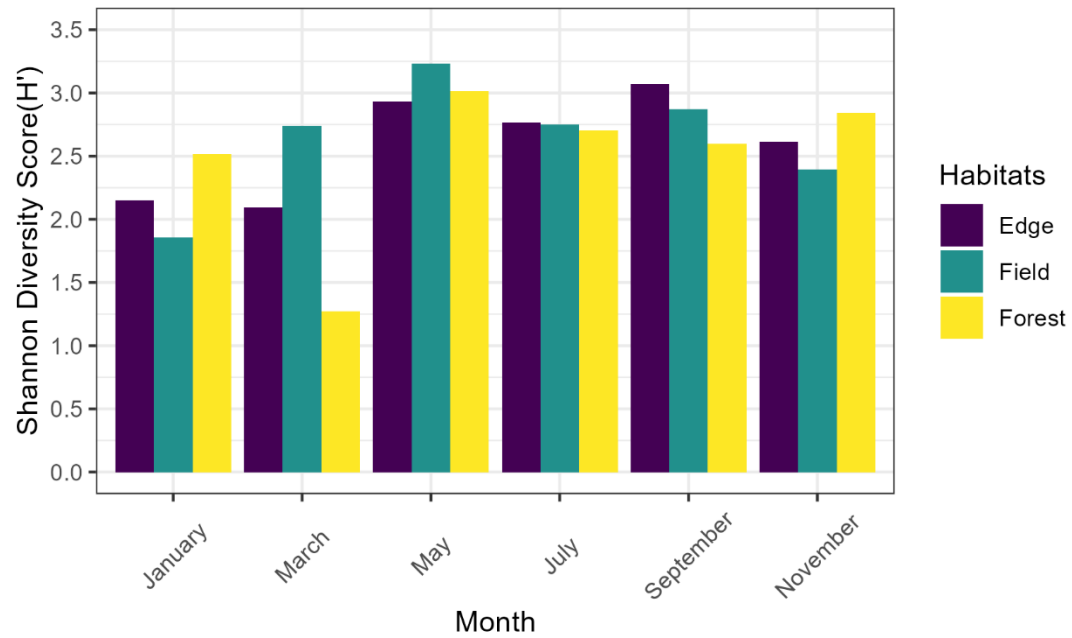


Figure 8: H' diversity values from SLAM traps for each month by habitat type, Arnold Air Force Base, 2023.

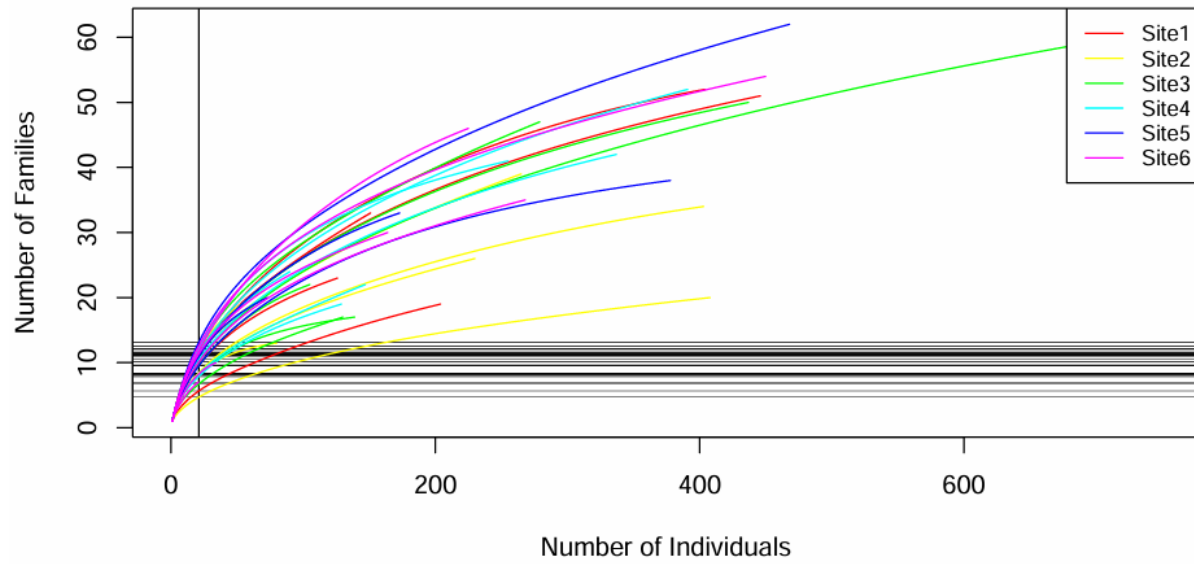


Figure 9: Rarefaction curves generated from SLAM trap data, Arnold Air Force Base, 2023.

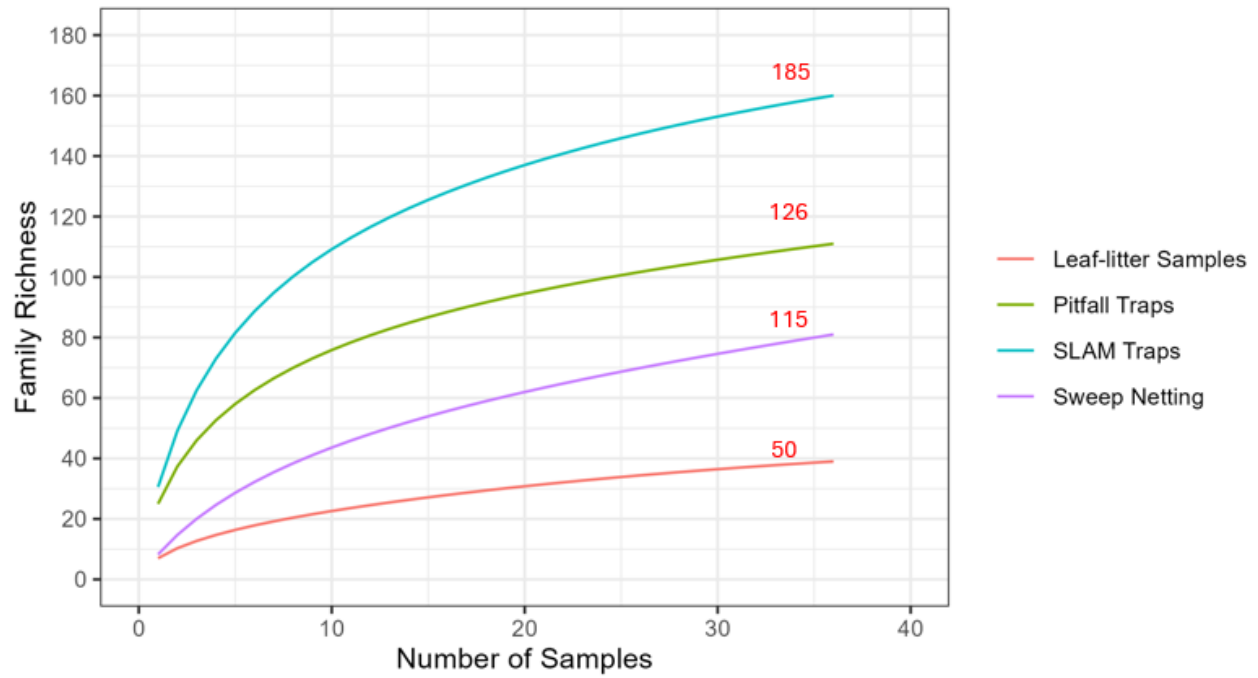


Figure 10: Family accumulation curves for all sampling types with estimated maximum family richness, Arnold Air Force Base, 2023.

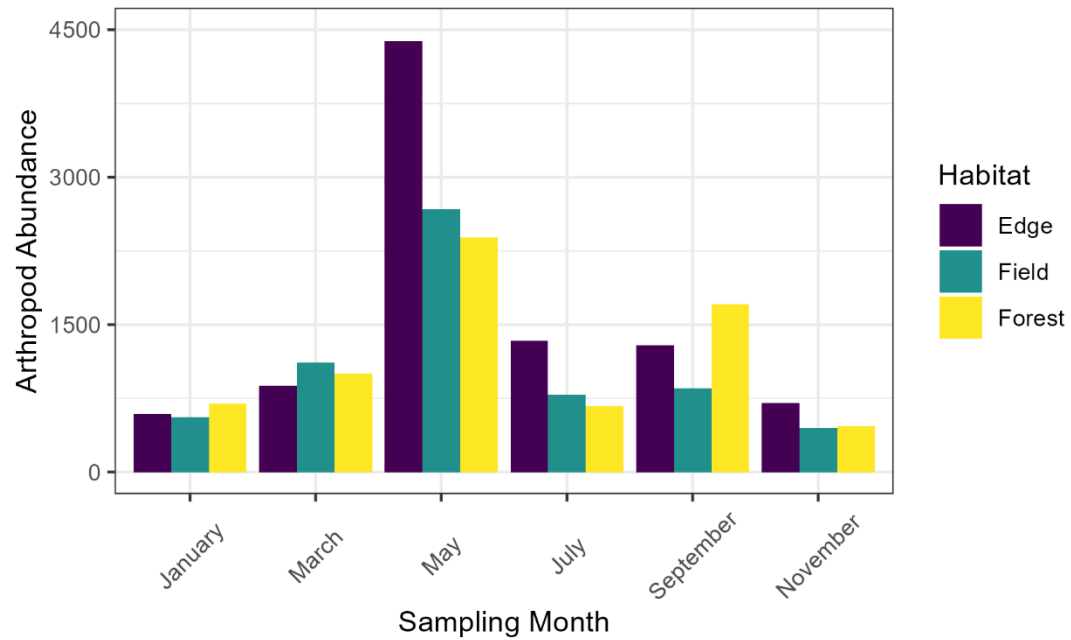


Figure 11: Arthropod abundance in pitfall traps for each month by habitat type, Arnold Air Force Base, 2023.

with a total of 9,396 arthropods caught between all three habitat types. Abundances were significantly different between habitats (p-value <.0001; df = 2; $\chi^2 = 494.06$) and sampling month (p-value <.0001; df = 5; $\chi^2 = 5,483.78$). In edge habitat pitfall traps abundance increased similarly to SLAM traps, peaking in May rather than July and then declining. For both the forest and field habitats abundance peaked in May, declined in July and increased in September. Pitfall traps had slightly lower H' diversity values than SLAM traps. In contrast to SLAM traps, H' diversity values were lower for this sampling method in May. The field habitat sites were typically the lowest habitat in H' diversity values and had the lowest diversity value overall in the month of November. Neither sampling month (p-value = 0.9804; df = 5; $\chi^2 = 0.75$) nor habitat type (p-value = 0.9297; df = 2; $\chi^2 = 0.15$) were statistically significant for pitfall trap H' diversity values. The highest H' diversity value occurred in September for edge habitats (Fig. 12). The minimum sample size for pitfall traps was much higher than SLAM traps and families collected when compared to minimum sample size is high (Fig 13). One sample caught just below 30 families when compared to the minimum sample size. Only one sample captured less than 10 families when compared against the minimum sample size. The family accumulation curve for pitfall traps did not grow as quickly as the SLAM trap curve did, but similarly to SLAM traps, multiple families were collected (Fig. 10). When plotted against the Michaelis-Menten model (Appendix 2), the equation provides a maximum family richness of 126 families, and a K value of 6.0.

A total of 861 arthropods were caught in sweep-net samples, with 212 caught in the forest habitat, 334 in the edge habitat, and 315 in the field habitat. Peak abundance was recorded in September with a total of 264 arthropods caught between all three habitat types (Fig. 14). Abundances were significantly different between habitat types (p-value <.0001; df = 2; $\chi^2 = 32.10$) and sampling month (p-value <.0001; df = 2; $\chi^2 = 161.45$). March and November were low in caught specimens for all three habitat types. For sweep-net samples in the edge habitat, two peaks occurred in July and September. For the forest habitat two peaks occurred in May and September. For the field habitat two peaks occurred in January and May. The general trends experienced for sweep-net samples are unlike those of the other sampling methods. Sweep-net samples had a wide range of H' diversity values, with the edge habitats having a H' diversity value of 0 in March all the way up to almost a 3.0 in the warmer months (e.g. May, July, and September) (Fig. 15). January, March, and November all had low H' values, and the only values that were above a 1.0 were the forest habitat sites for January and March. Habitat type (p-value = 0.8626; df = 2; $\chi^2 = 0.30$) and sampling month (p-value = 0.1462; df = 5; $\chi^2 = 8.19$) did not have statistical significance for the diversity values of sweep-net samples. The rarefaction curves for sweep-net samples do not have as many samples represented within them as so many of the samples from this trapping type had less than two families captured (Fig. 16). Most of the curves are still growing rather than exhibiting a plateau, suggesting that with more sweep-net samples a more representative diversity of the area would have been captured. When compared against the minimum sample size diversity would be low, as no samples collected above five families at that point. The family accumulation curve still appears to be increasing (Fig. 10). When plotted

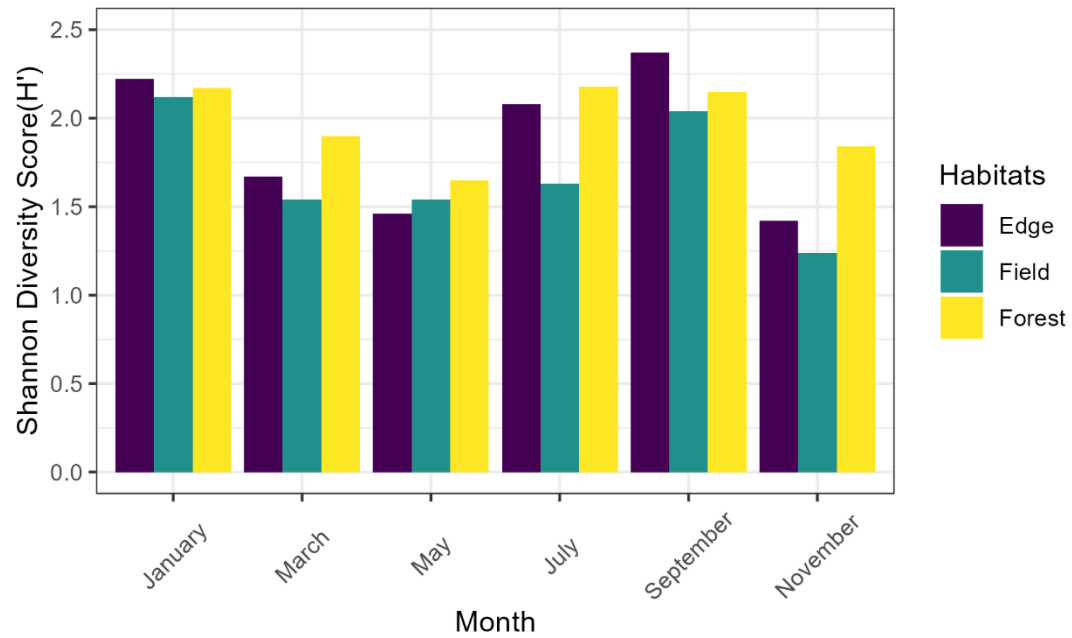


Figure 12: H' diversity values from pitfall traps for each month by habitat type, Arnold Air Force Base, 2023.

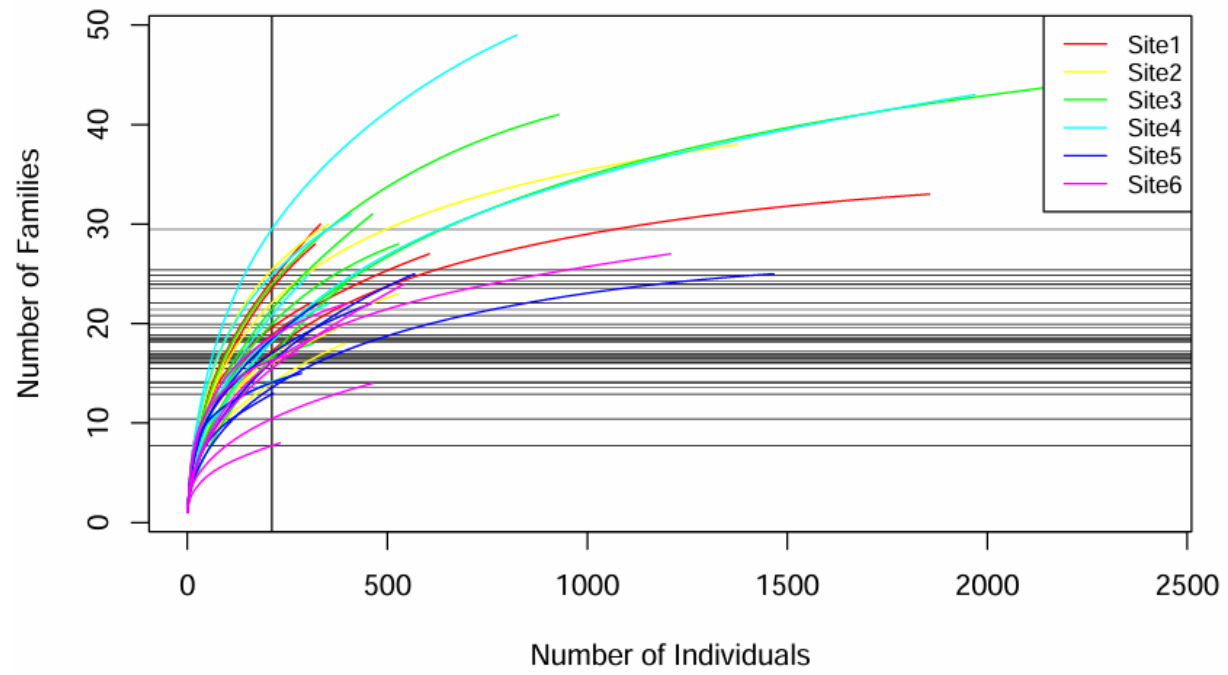


Figure 13: Rarefaction curves generated from pitfall trap data, Arnold Air Force Base, 2023.

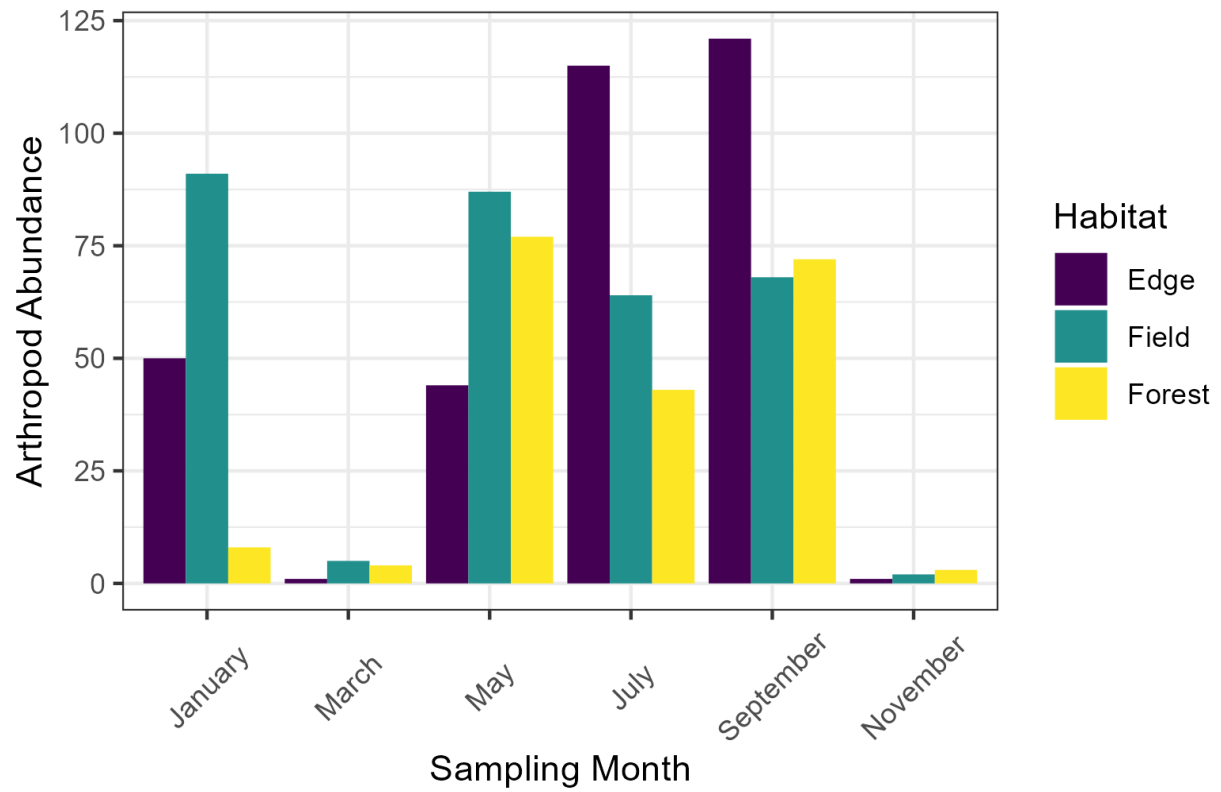


Figure 14: Arthropod abundance from sweep-net samples for each month by habitat type, Arnold Air Force Base, 2023.

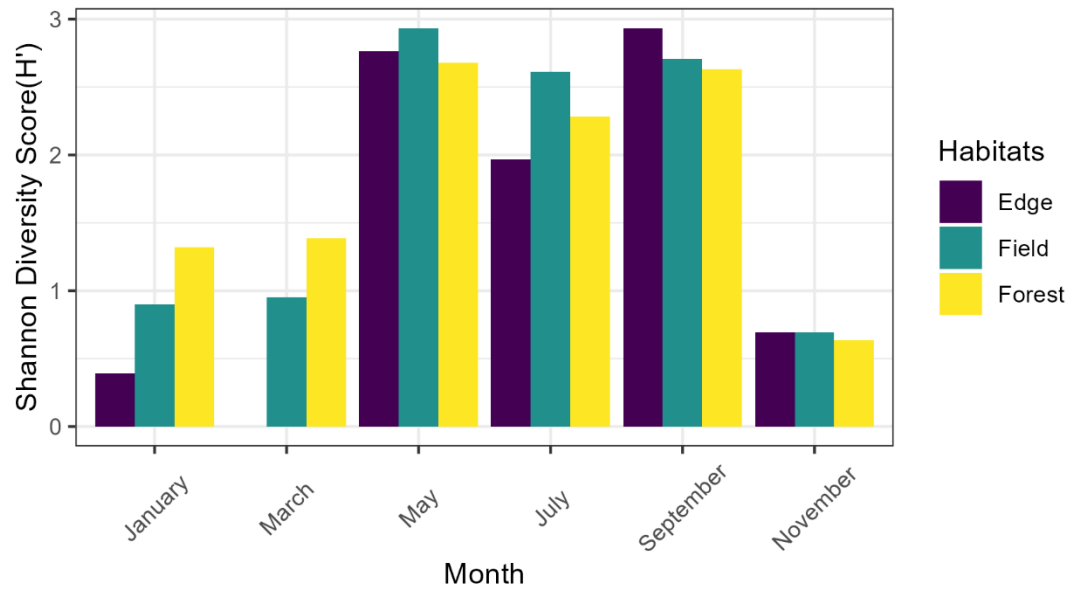


Figure 15: H' diversity values from sweep-net samples for each month by habitat type, Arnold Air Force Base, 2023.

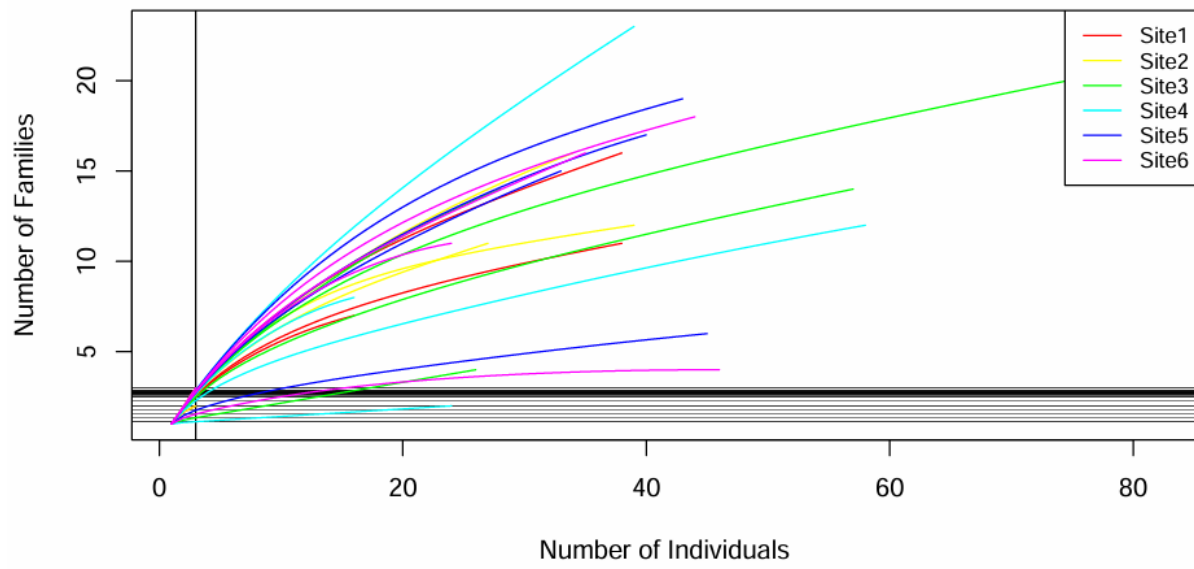


Figure 16: Rarefaction curves generated from sweep-net data, Arnold Air Force Base, 2023.

against the Michaelis-Menten model (Appendix 3), the maximum family richness would be 115 families, and a K value of 16.3.

Leaf-litter samples collected 30,615 arthropods. A total of 9,980 were collected from the forest habitat, 9,350 from the edge habitat, and 11,285 from the field habitat (Fig. 17). Peak abundance was recorded in May with a total of 18,438 arthropods collected in all three habitats. Abundances were significantly different between habitat types (p -value $< .0001$; $df = 2$; $\chi^2 = 192.46$) and sampling month (p -value $< .0001$; $df = 5$; $\chi^2 = 16730.15$). All three habitat types had low abundances for leaf-litter samples in September. Leaf-litter samples accounted for close to one-half of all specimens collected (~49%); of the 30,615 arthropods caught in leaf-litter samples 22,153 were mites (~72%). The H' diversity equation was run using the full data set including mites, as well as run without mites. January and March diversity values dropped, while May diversity rose by about 0.3 for the edge sites, with diversity of forest and field sites not changing much. H' diversity values were higher by about 0.5 in July for all three habitat types. A comparison of the mean diversity values by month is given in Figure 18. A side-by-side comparison of the two diversity graphs, one including mites and the other one excluding mites (Fig. 19). Ultimately, mites remained in data analyses. Leaf-litter samples had low H' diversity values throughout the entirety of 2023 (Fig. 20). The only sampling session that had a H' value over 1.0 was edge habitats in May. The edge habitats had highest diversity for the months of January, May, and July but there was no significance between sampling months (p -value = 0.9974; $df = 5$; $\chi^2 = 0.31$) or habitat types (p -value = 0.9627; $df = 2$; $\chi^2 = 0.08$). If family richness were to be compared against the smallest sample size, H' diversity values would have been much lower than what is already presented (Fig. 21). The family accumulation curve for leaf-litter samples differs from the other three trapping methods in the fact that it is flattening out (Fig. 10). Plotted against the Michaelis-Menten model (Appendix 4), maximum family richness would be 50 families, and a K value of 11.3.

Overall, no significance of arthropod diversity between the different habitat types and the different trapping methods used was seen ($p \geq 0.05$; $df = 2$). While the changes in abundance could be attributed to location and month, diversity measures could not be, suggesting that diversity within sites may be affected by other abiotic or biotic factors that were not examined.

Across the entire study arthropod abundance was highest within edge habitats throughout the year with 21,976 specimens, approximately 35% of the total number collected. Edge habitats also had the highest total abundance for all sampling methods, except that leaf-litter abundances were greatest in the field samples.

As this study is a portion of a larger ongoing study, some results from 2022 to 2023 can be compared. In 2022 a SLAM trap within the forest area of the study site was placed for approximately six weeks in the summer (end of July and majority of August). The 2022 SLAM trap was placed in site 2 of this study, and so results from July for site 2 were compared. This

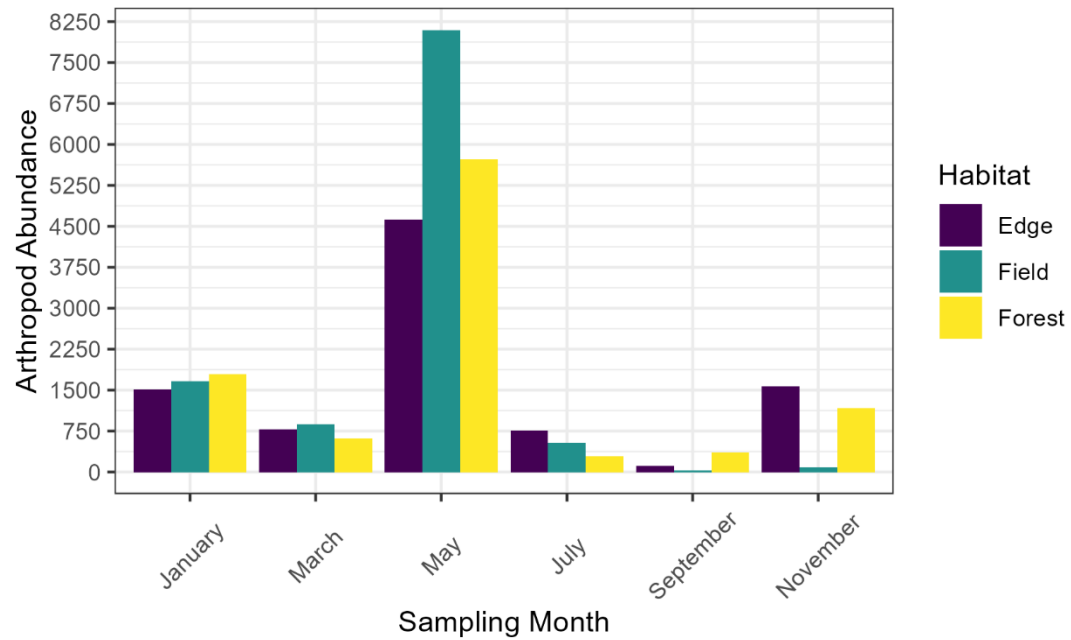


Figure 17: Arthropod abundance from leaf-litter samples for each month by habitat type, Arnold Air Force Base, 2023.

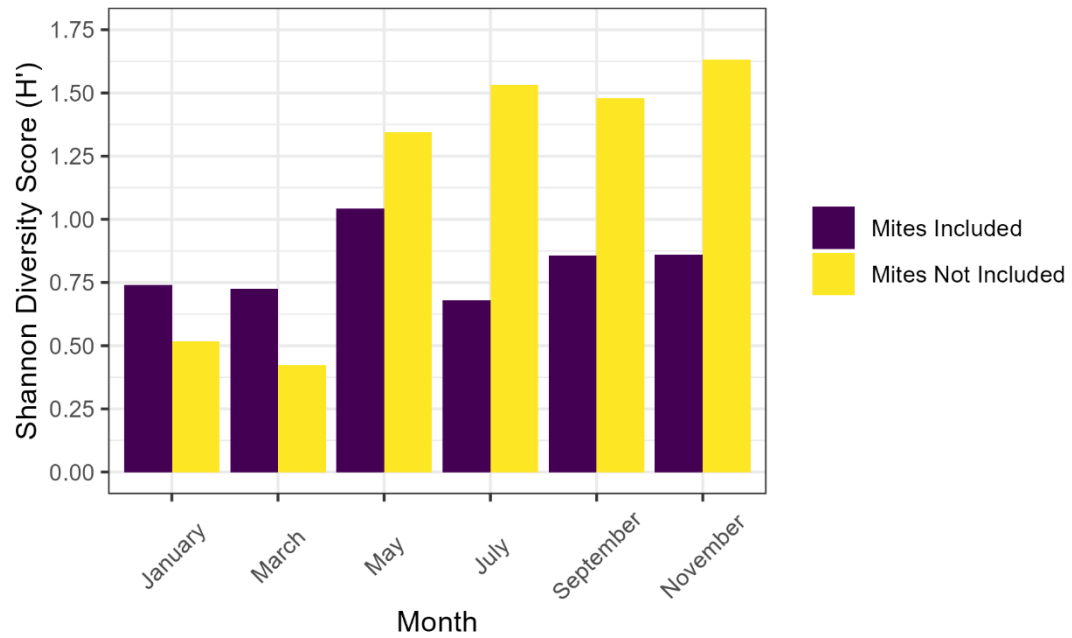


Figure 18: Comparison of average H' diversity values for leaf-litter samples with mites included and without mites included in the count for all habitat types, Arnold Air Force Base, 2023

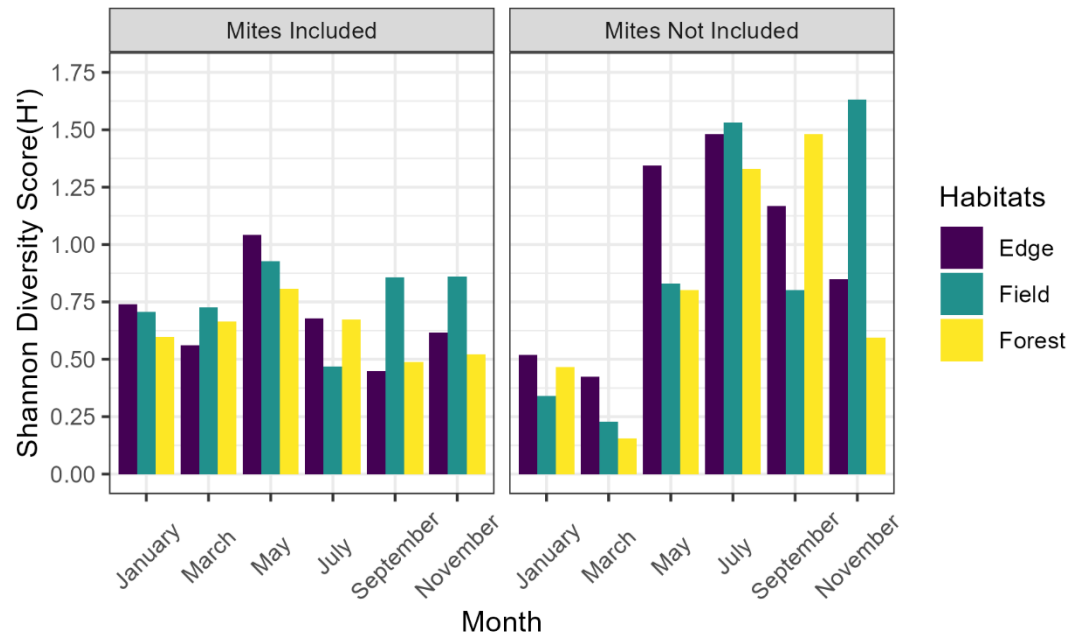


Figure 19: A side-by-side comparison of H' diversity values for leaf-litter samples with mites included and without mites included, separated by habitat type, Arnold Air Force Base, 2023.

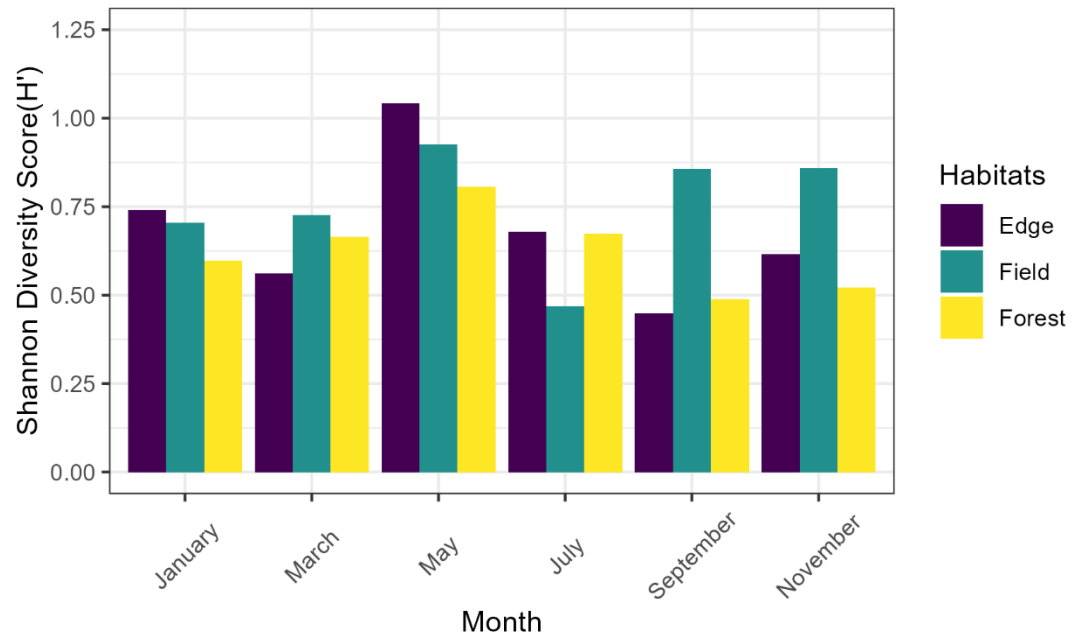


Figure 20: H' diversity values from leaf-litter samples for each month by habitat type, Arnold Air Force Base, 2023.

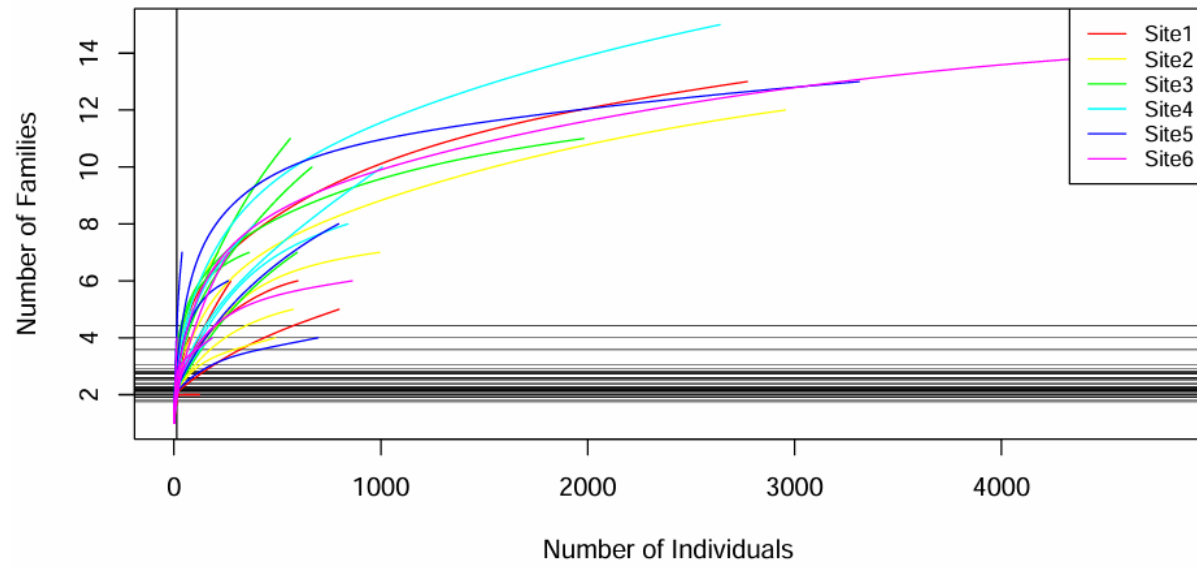


Figure 21: Rarefaction curves created from leaf-litter sample data, Arnold Air Force Base, 2023.

trap collected 1,069 specimens. July 2023 forest habitat SLAM traps collected a total of 814 specimens but were only left out for two weeks as opposed to the six for the 2022 SLAM trap. 402 of the 814 specimens came from site 2. A comparison of the summer 2022 SLAM trap and summer 2023 SLAM trap was made to assess differences of families captured (Fig. 22). In 2023 38 families were caught in site 2, and in 2022 87 families were captured in the SLAM trap. More families were captured in 2022 than in 2023, however, this trend was not the same for Collembola or Lepidoptera, as 2023 had more families within those orders

It was hypothesized that after the fire there would be a decrease in arthropod abundance due to the deleterious effects of fire on living organisms, followed by recovery of abundance. The controlled burn took place in mid-March 2023. Forest and edge habitat traps were deployed before, during, and after the burn. SLAM and pitfall traps were placed in the burned field sites about one week after the fire took place. Leaf-litter in the burn sites had few arthropods (2,280), as nearly all surface organic litter was reduced to ash. Abundances in the forest sites may have been low due to flooding that month. A major increase in abundance for all sampling types and all habitat types occurred in May. Thus, this hypothesis was not supported.

It was also hypothesized that abundance of arthropods would be higher in the edge habitats. In fact, abundance was highest in the edge habitats for all trapping methods except leaf-litter samples, likely due to the fact that there was relatively less leaf-litter within the edge. Overall, however, arthropod abundance was higher within the edge sites than in the other sites, supporting this hypothesis.

Similarly to abundance, it was hypothesized that the edge habitats would have the most diversity. In contrast to results for abundance, higher diversities for the different collection methods occurred in field or forest sites, not in the edge. Results were not statistically different for diversity within trap type or sampling month. Ultimately, this hypothesis was not supported.

Precipitation

Precipitation and arthropod abundance in SLAM traps were significantly correlated ($p = 0.0107$; $r = 0.05$). However, arthropod numbers in pitfall traps were not significantly correlated ($p = 0.6214$; $r = 0.01$), nor were leaf-litter samples ($p = 0.6751$; $r = 0.18$) or sweep-net samples ($p = 0.0563$; $r = 0.07$). Arthropods entering SLAM traps or on vegetation are likely more active when relative humidity is high, whereas litter and soil-dwelling arthropods are in naturally more humid environments and therefore less affected by precipitation. SLAM traps had a significant correlation of diversity and precipitation ($p = 0.0252$; $r = -0.53$), but neither pitfall H' diversity values were significantly correlated ($p = 0.7359$; $r = 0.09$), nor were leaf-litter samples ($p = 0.9301$; $r = 0.02$) or sweep-net sampling ($p = 0.0924$; $r = -0.41$).

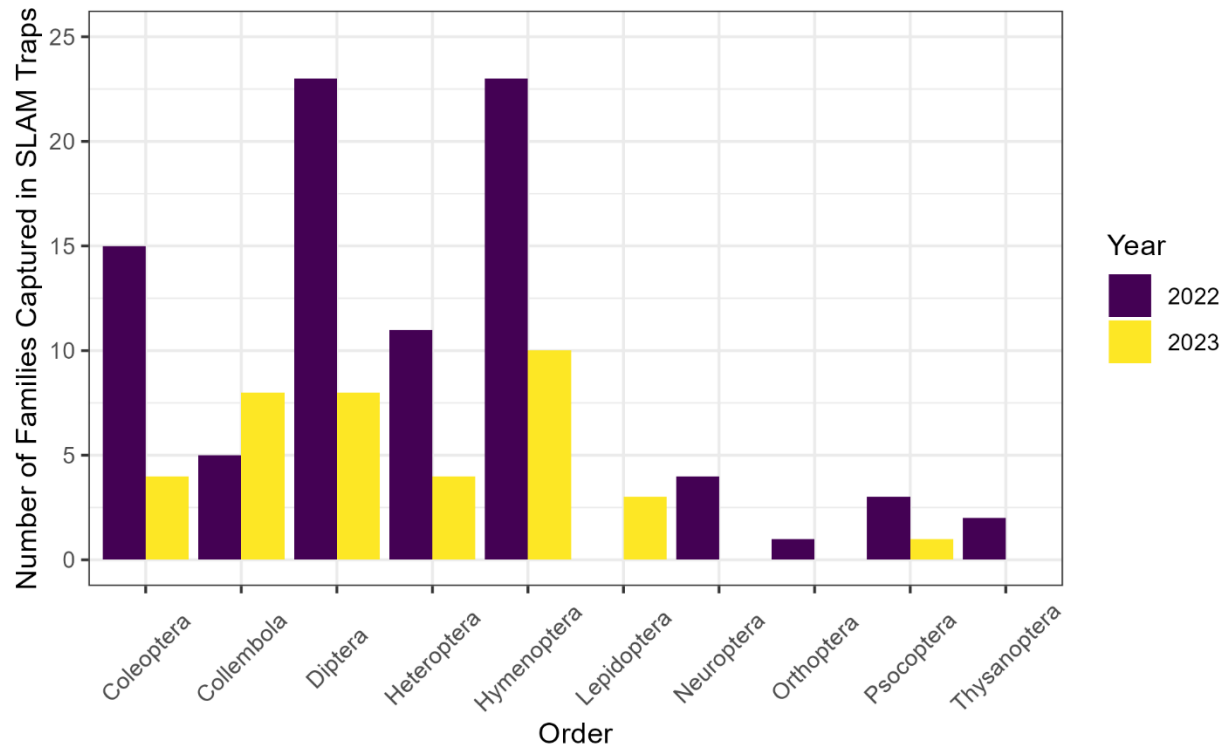


Figure 22: A comparison of the number of families collected in a SLAM trap in forest site, Arnold Air Force Base, 2022 and 2023.

Sørensen Index for Heterogeneity

Field and edge habitat sites had a Sørensen similarity of 81% overall, while the forest and the field were 72.5% similar (Table 4). These findings suggest that repopulation of the controlled burn field site was from the edge habitat more than the forest habitat. The field and the edge were most similar for SLAM traps and sweeps as well. For pitfall traps the highest similarity was with the forest and edge habitat sites; and for leaf-litter samples the highest similarity was between the forest habitat and field habitat sites.

To help better understand recolonization of the burned field area, Venn diagrams were developed. Examining the Venn diagrams by trapping method can be useful in determining individual method tendencies, but the overall Venn diagram can be used to determine ecological relationships. Many arthropod groups were shared between the three habitats; the highest number of shared groups for habitats was between the field and the edge environment (Fig. 23). Thus, recolonization of the burned field appeared to be mostly from the edge habitat. A Venn diagram of the different trapping methods was created with shared orders (Fig. 24). Sweep-net samples did not share as many collected orders as the other sampling methods did.

Results - Collembola

The May collections had by far the greatest number of collected Collembola (~8,700). An analysis of the data by habitat rather than totals by traps and month is presented in Table 6 and Figure 25. None of the variables tested showed significance when analyzed by habitat type ($p \geq 0.05$; $df = 2$). Collembola were most abundant in the burned field habitat (3,325 specimens collected), followed closely by the edge habitat (3,145) and forest (2,178) (Fig. 26). The most abundant Collembola family was Entomobryidae (2,174), followed by Tomoceridae (1,848), and then Isotomidae (1,644) (Fig. 27). All H' diversity values were above 1.5 (Fig. 28).

Leaf-litter samples of field habitats collected the most Collembola (2,108), followed by the forest (1,133) and edge (703) (Fig. 26). The most abundant family found within leaf-litter samples was Isotomidae (1,527) followed by the family Entomobryidae (1,114) (Fig. 29). Collembola diversity for leaf-litter samples was high, with the edge being the most diverse habitat (2.95), followed by the field (2.69) and then the forest (2.61) (Fig. 28).

For pitfall traps the edge had the highest number of Collembola (2,410), followed by the field (1,385) and the forest (999) (Fig. 26). The most abundant family found in pitfall traps was Tomoceridae (1,729) followed by Entomobryidae (972) (Fig. 30). Collembola H' diversity was lower for pitfall traps than for leaf-litter samples. The highest diversity occurred in the field (2.67), followed by the edge (2.35) and the forest (2.03) (Fig. 28).

Table 4: Sørensen index results for habitat comparisons by trapping type and overall, Arnold Air Force Base, 2023.

Habitats Being Compared	Overall*	SLAM Traps	Pitfall Traps	Leaf-litter Samples	Sweep-net Samples
Forest & Edge	79.6%	74.9%	81.8%	67.6%	53.5%
Edge & Field	82.0%	80.0%	71.1%	66.7%	66.7%
Forest & Field	73.6%	68.8%	74.5%	67.7%	56.8%

*Data for all trapping methods was compiled and assessed with the Sørensen equation to calculate overall heterogeneity.

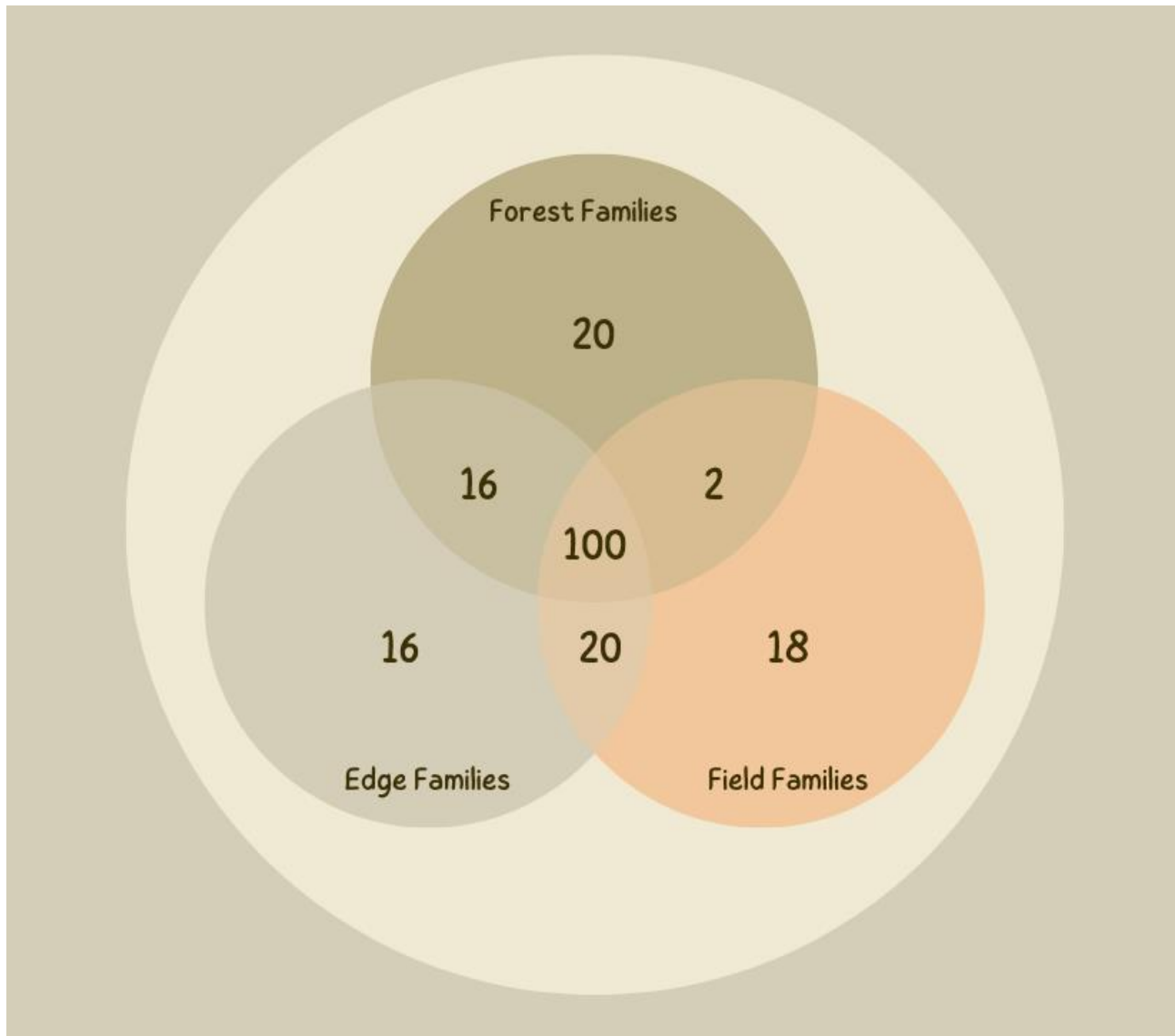


Figure 23: Venn diagram of arthropod family distributions within three different habitat types, Arnold Air Force Base, 2023.

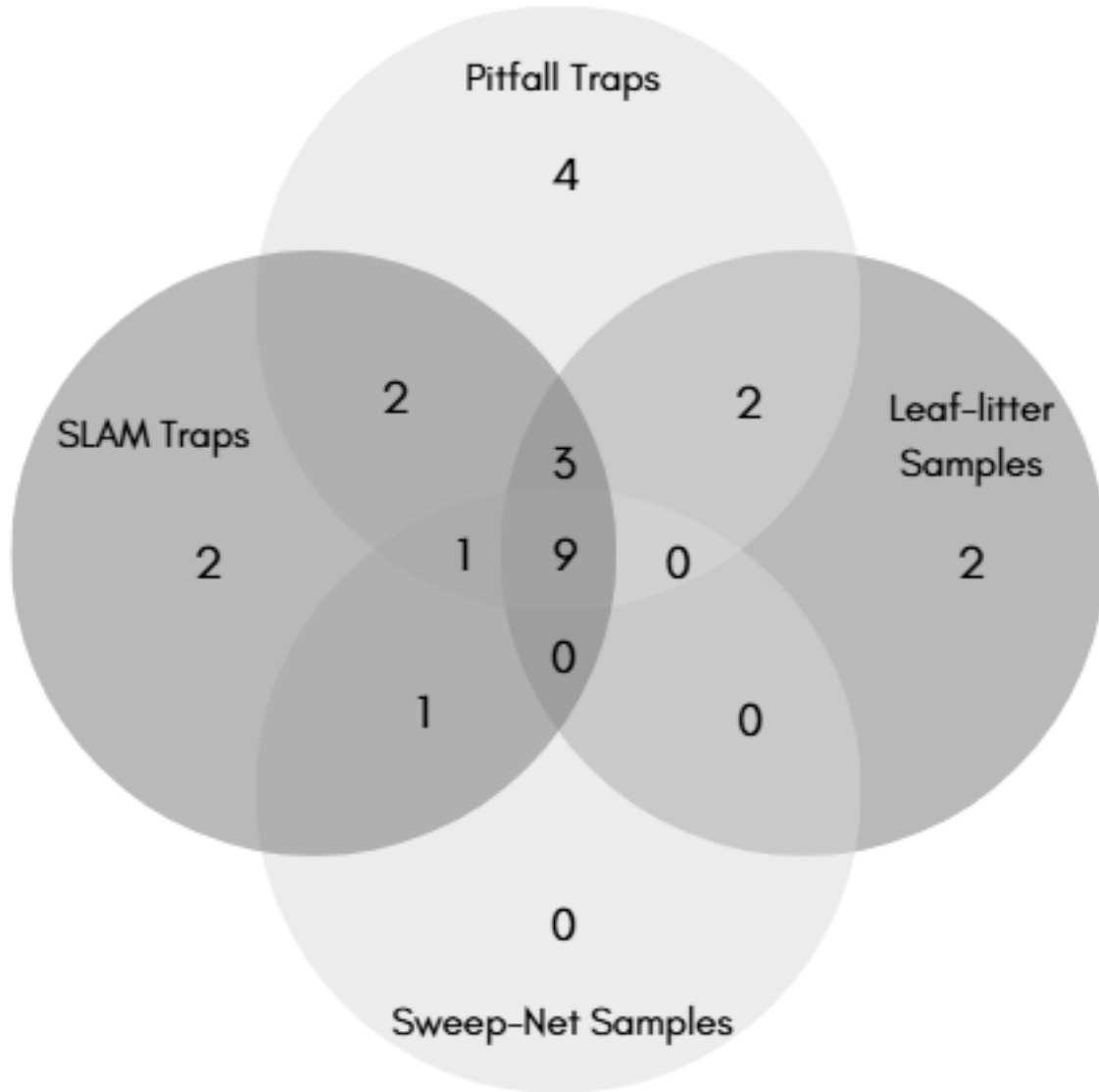
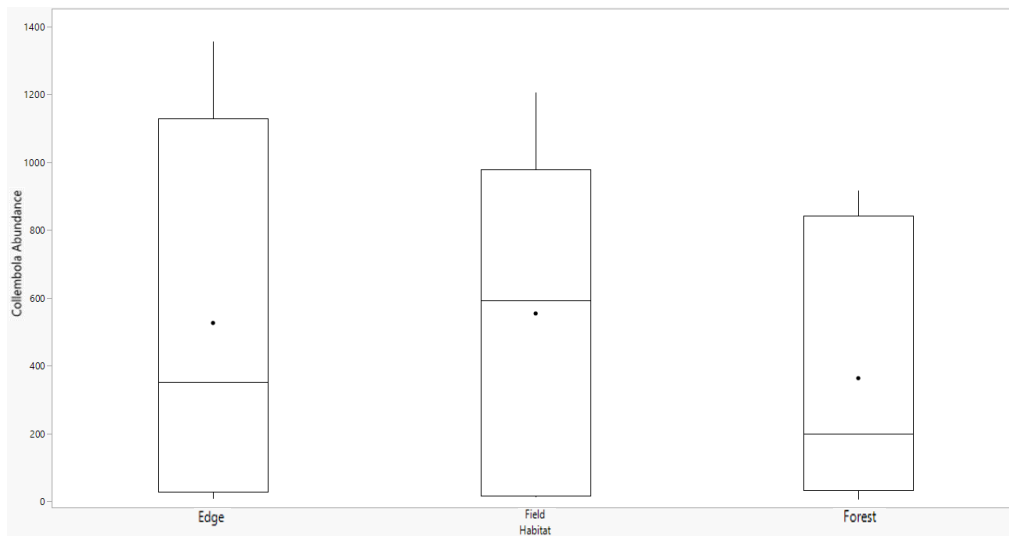


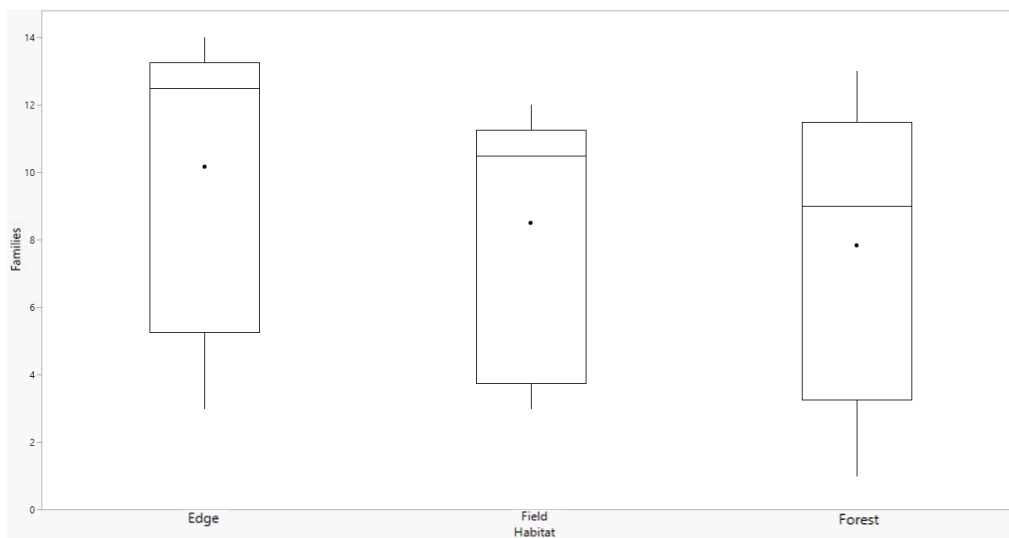
Figure 24: Venn diagram of Orders collected in different sampling methods, Arnold Air Force Base, 2023.

Table 5: Statistics calculated using number of sampling events and averages, Arnold Air Force Base, May 2023.

<u>SLAM traps</u>				
Statistic	Forest (n = 2 events)	Edge (n = 2 events)	Field (n = 2 events)	Total (n = 6 events)
No. Families	4	7	4	8
No. Unique Families	1	3	0	4
No. Species	7	10	5	38
No. Unique Species	2	4	1	7
Mean No. Specimens	24	22	16	62
Mean Shannon-Weiner Diversity Index	1.65	1.77	1.96	1.79
<u>Leaf-litter Samples</u>				
Statistic	Forest (n = 2 events)	Edge (n = 2 events)	Field (n = 2 events)	Total (n = 6 events)
No. Families	12	15	12	17
No. Unique Families	1	2	1	4
No. Species	27	32	29	51
No. Unique Species	3	9	7	19
Mean No. Specimens	566.5	351.5	1,054	657.33
Mean Shannon-Weiner Diversity Index	2.61	2.95	2.69	2.75
<u>Pitfall Traps</u>				
Statistic	Forest (n = 2 events)	Edge (n = 2 events)	Field (n = 2 events)	Total (n = 6 events)
No. Families	13	13	12	13
No. Unique Families	0	0	0	0
No. Species	19	22	19	47
No. Unique Species	0	3	1	4
Mean No. Specimens	499.5	1,205	592.5	765.67
Mean Shannon-Weiner Diversity Index	2.03	2.35	2.67	2.35
<u>All Collection Methods</u>				
Statistic	Forest (n = 6 events)	Edge (n = 6 events)	Field (n = 6 events)	Total (n = 18 events)
No. Families	16	15	15	17
No. Unique Families	1	0	1	2
No. Species	39	41	37	51
No. Unique Species	4	5	5	14
Mean No. Specimens	363.3	526.17	554.17	481.22
Mean Shannon-Weiner Diversity Index	2.10	2.36	2.44	2.3

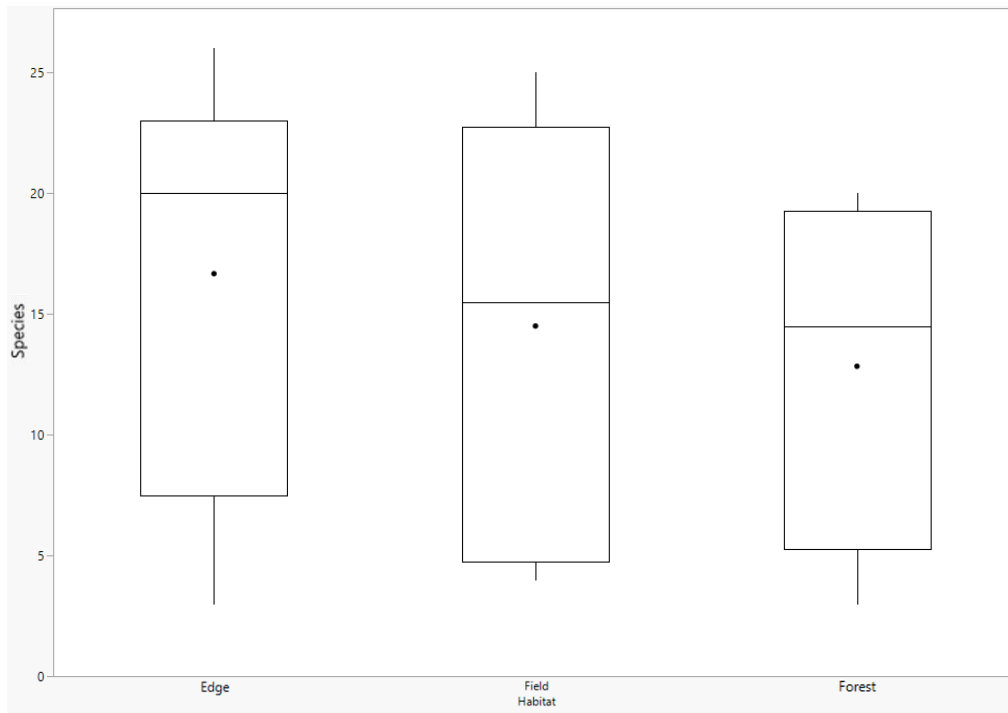


a.) Box-and-whisker plot of Collembola by habitat, Arnold Air Force Base, May 2023.

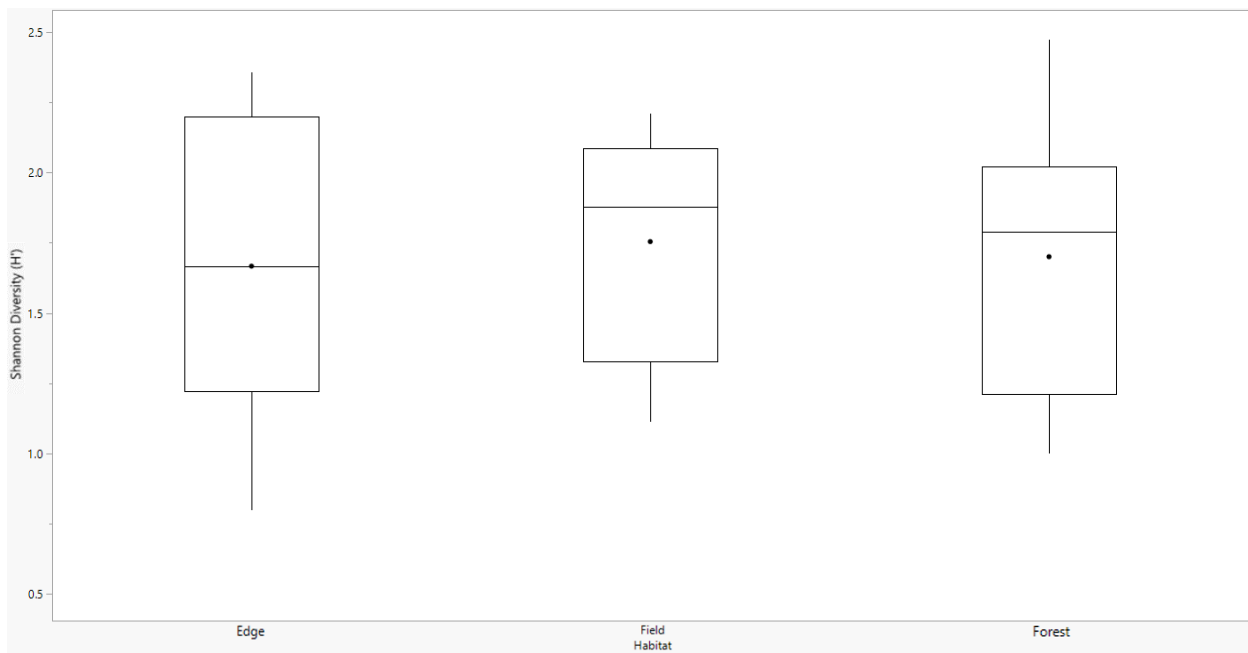


b.) Box-and-whisker plot of number of Collembola families by habitat, Arnold Air Force Base, May 2023.

Figure 25: Box-and-whisker plots of Collembola data analyzed by habitat, Arnold Air Force Base, May 2023. Mean denoted by ●.



c.) Box-and-whisker plot of number of Collembola species by habitat, Air Force Base, May 2023.



d.) Box-and-whisker plot of Shannon diversity values by habitat, Arnold Air Force Base, May 2023.

Figure 25 (continued)

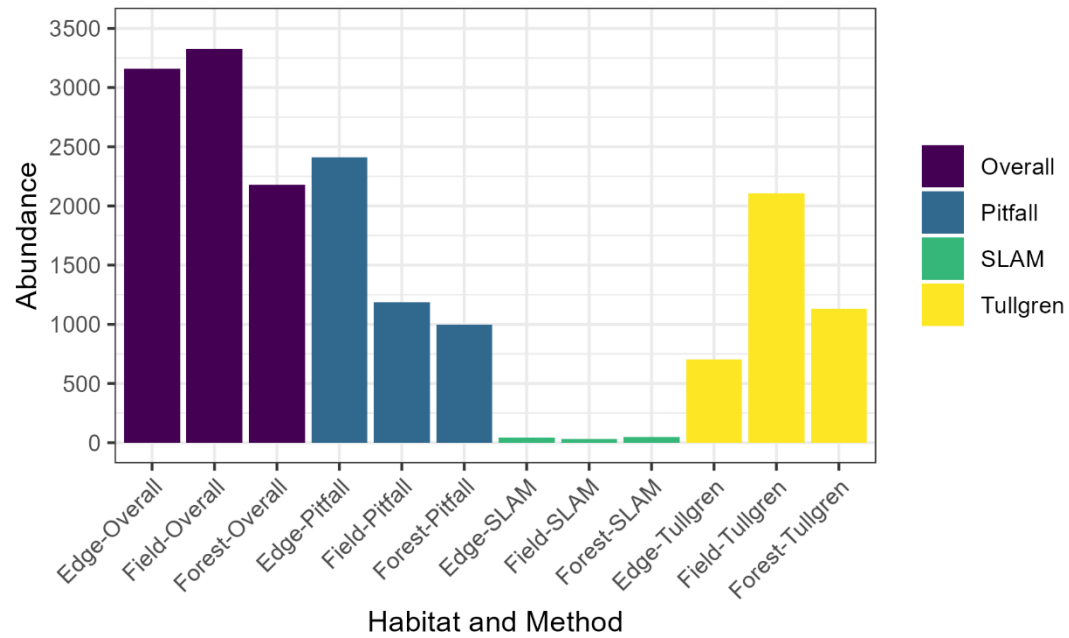


Figure 26: Collembola abundance for all trapping types and habitats, Arnold Air Force Base, May 2023.

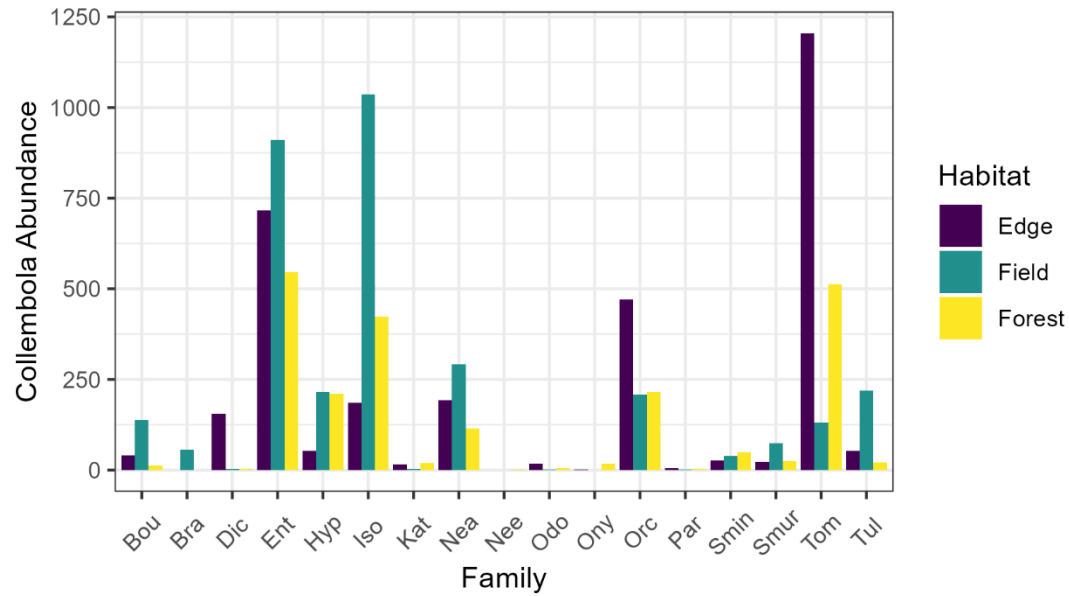


Figure 27: Collembola family abundances by habitat type, Arnold Air Force Base, May 2023. Abbreviations as follows: Bou = Bourletiellidae, Bra = Brachystomellidae, Dic = Dicyrtomidae, Ent = Entomobryidae, Hyp = Hypogastruridae, Iso = Isotomidae, Kat = Katiannidae, Nea = Neanuridae, Nee = Neelidae, Odo = Odontellidae, Ony = Onychiuridae, Orc = Orchesellidae, Par = Paronellidae, Smin = Sminthuridae, Smur = Sminthurididae, Tom = Tomoceridae, Tul = Tullbergiidae.

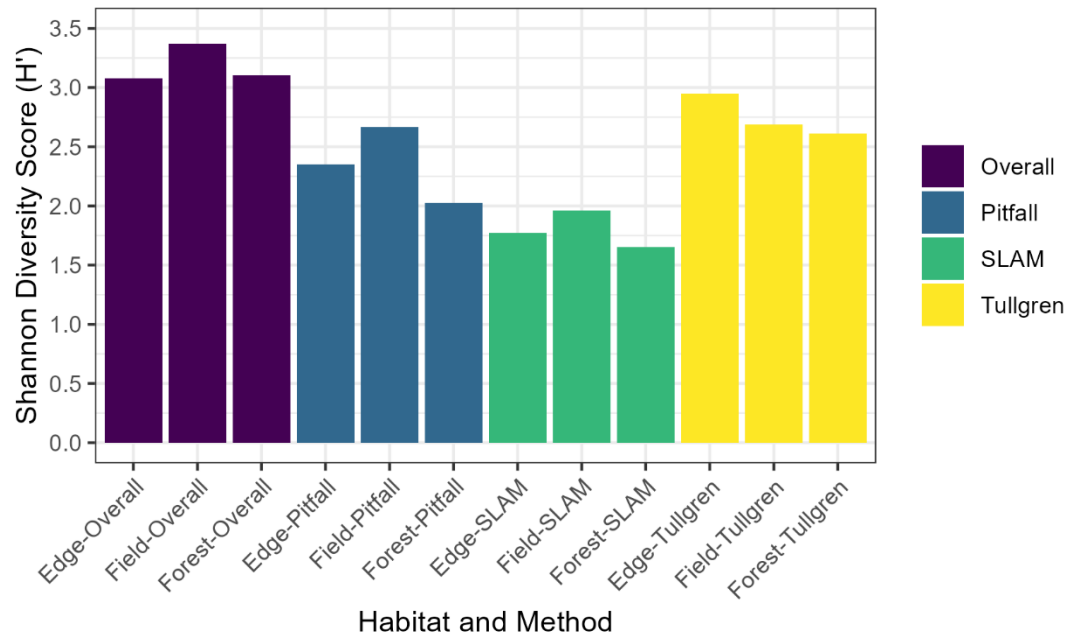


Figure 28: H' diversity values of Collembole for all trapping methods and overall, Arnold Air Force Base, May 2023.

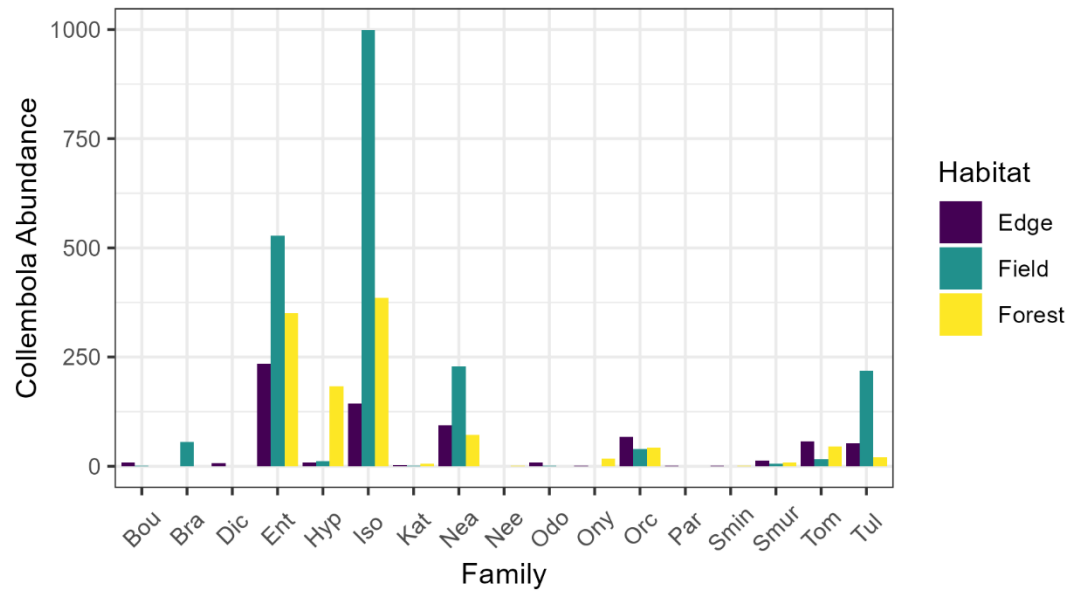


Figure 29: Collembola family abundance in leaf-litter samples by habitat type, Arnold Air Force Base, May 2023. Abbreviations as follows: Bou = Bourletiellidae, Bra = Brachystomellidae, Dic = Dicyrtomidae, Ent = Entomobryidae, Hyp = Hypogastruridae, Iso = Isotomidae, Kat = Katiannidae, Nea = Neanuridae, Nee = Neelidae, Odo = Odontellidae, Ony = Onychiuridae, Orc = Orchesellidae, Par = Paronellidae, Smin = Sminthuridae, Smur = Sminthurididae, Tom = Tomoceridae, Tul = Tullbergiidae.

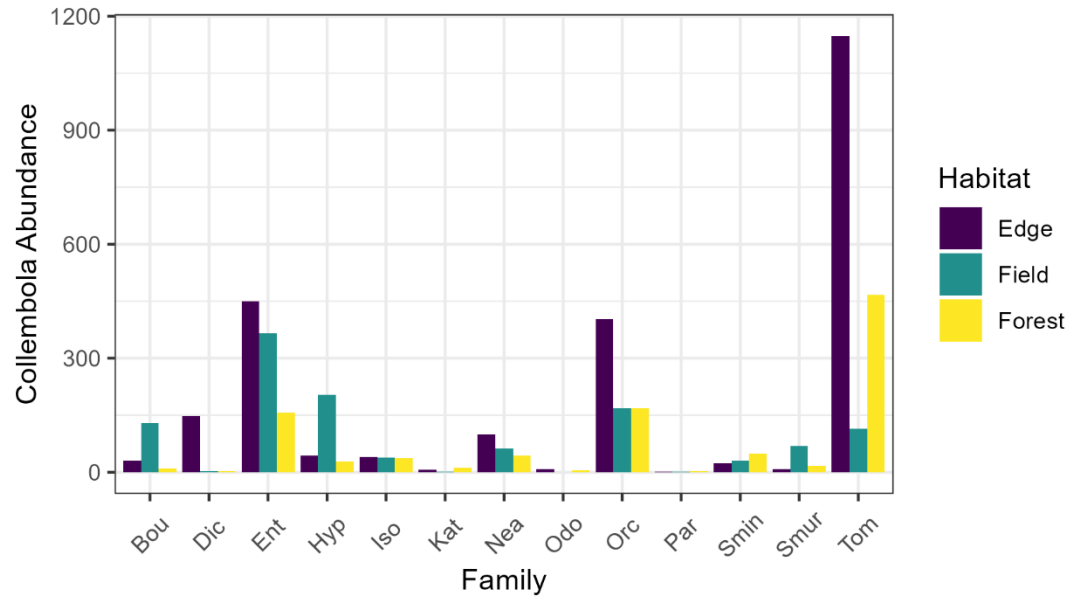


Figure 30: Collembola family abundances from pitfall traps by habitat type, Arnold Air Force Base, May 2023. Abbreviations as follows: Bou = Bourletiellidae, Dic = Dicyrtomidae, Ent = Entomobryidae, Hyp = Hypogastruridae, Iso = Isotomidae, Kat = Katiannidae, Nea = Neanuridae, Odo = Odontellidae, Orc = Orchesellidae, Par = Paronellidae, Smin = Sminthuridae, Smur = Sminthurididae, Tom = Tomoceridae.

SLAM traps had the highest number of Collembola in the forest (48), followed by the edge (44) and the field (32) (Fig. 26). The most abundant family found within SLAM traps in May was Entomobryidae (88) (Fig. 31). SLAM traps had the lowest Collembola diversity in May. All H' diversity values for SLAM traps were under 2.0, with the highest diversity in the field (1.96), followed by the edge (1.77) and the forest (1.65) (Fig. 28).

Isotomidae was the most abundant family in leaf-litter samples (1,527) and the third most abundant overall. Leaf-litter samples had 12 species of Isotomidae collected in May (Fig. 32). The most abundant species was *Folsomia prima* (735) with the majority of the specimens collected from the field habitat. The highest H' diversity for Isotomidae was in the edge, with a value of 1.7 (Fig. 33). Entomobryidae was the most abundant family; for both pitfall traps (920) and leaf-litter samples (1,034) *Lepidocyrtus* spp. had the highest abundance (Fig. 34, 35). H' diversity for Entomobryidae species was low for pitfall traps and leaf-litter samples (Fig. 33). All H' diversity values for Entomobryidae were below 1.0. The H' diversity values were also similar between the habitat types.

An interactive sunburst plot of the abundances for Collembola collected in May is provided in Figure 36 and at the following link:

https://github.com/Mvoss09/interactive_sunburst/blob/75d1f94f186972397c8cfa0466ddecfc9c5fa88c/sunburst_plot.html. From this link, choose “download raw file” to see the plot.

Overall, the forest and edge habitats were most similar at 96.7% followed by the edge and field habitat at 93.3%. In the overall project, edge and field were the most similar habitats. All Sørensen values are provided in Table 6. Similarity of Collembola communities was high for all trapping methods, and overall, SLAM traps had the lowest similarity while pitfall traps had the highest. A Venn diagram was developed for all of the species found for May; the forest and edge have the highest number of shared species (Fig. 37). Number of Collembola species collected differed by trapping type (Fig. 38).

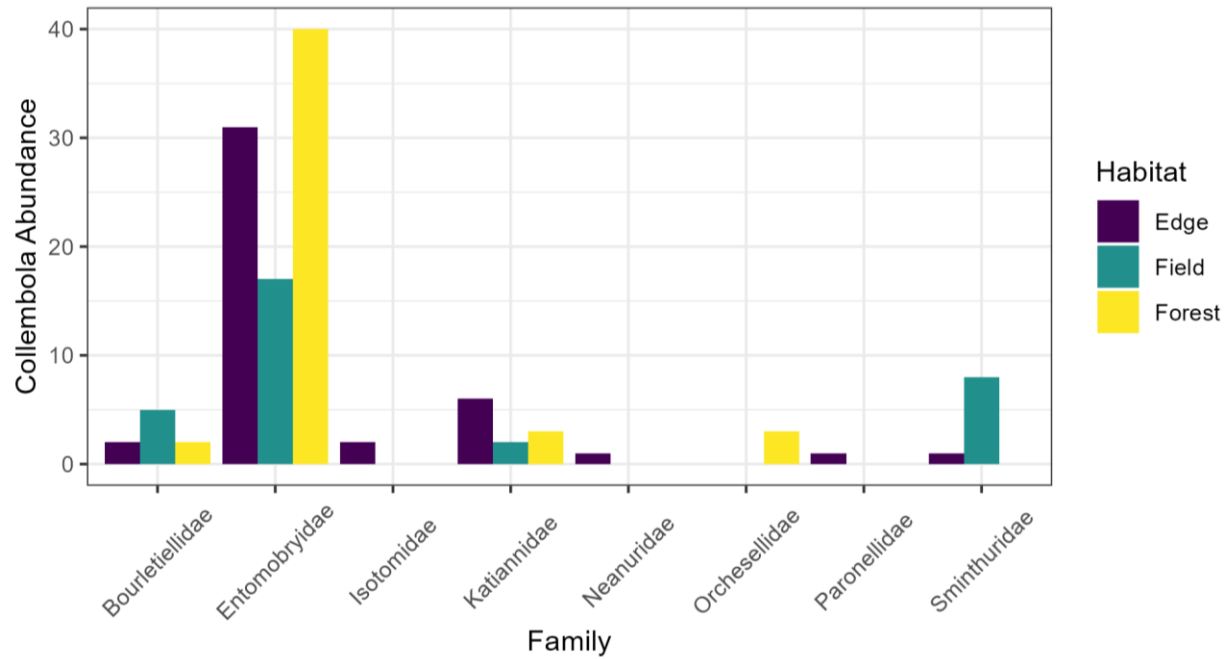


Figure 31: Collembola family abundances from SLAM traps by habitat, Arnold Air Force Base, May 2023.

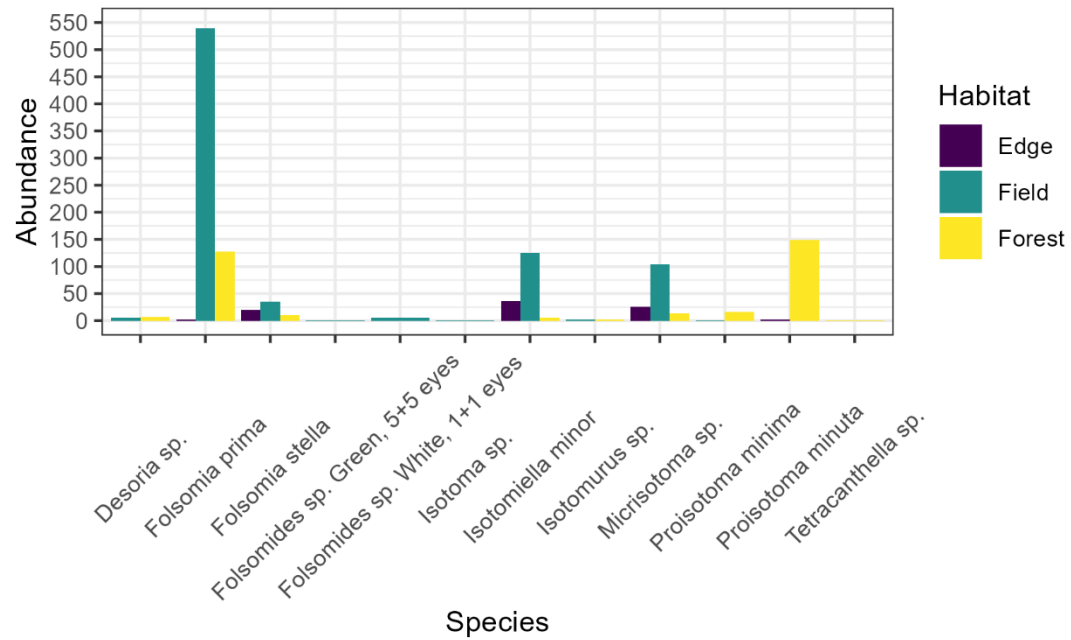


Figure 32: Species abundance from leaf-litter samples for Isotomidae species, Arnold Air Force Base, May 2023.

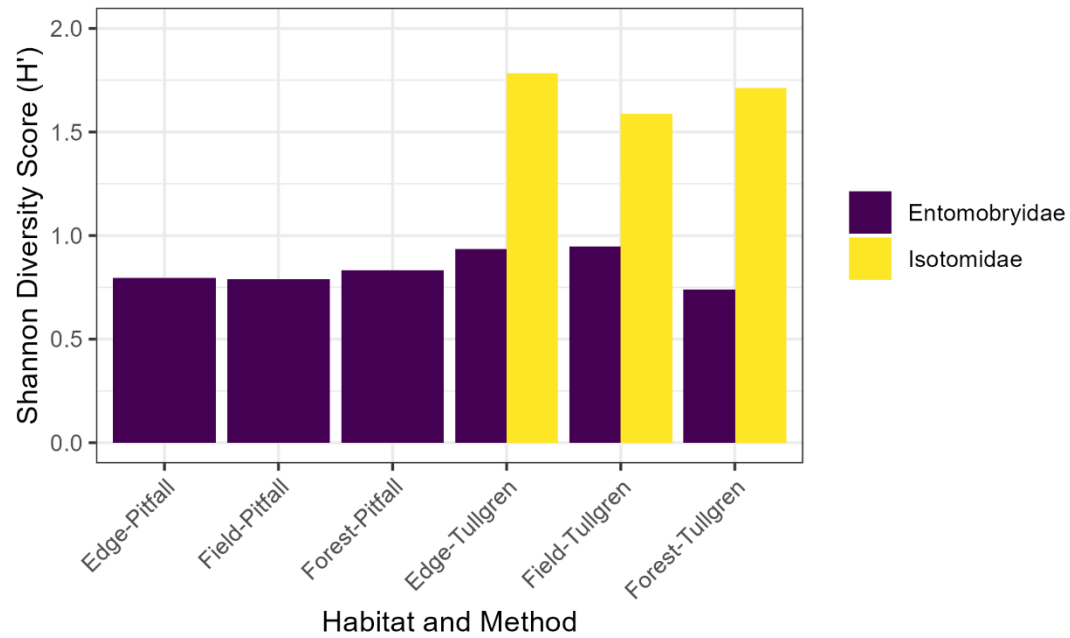


Figure 33: H' diversity values from leaf-litter samples and pitfall traps of Entomobryidae and Isotomidae, Arnold Air Force Base, May 2023.

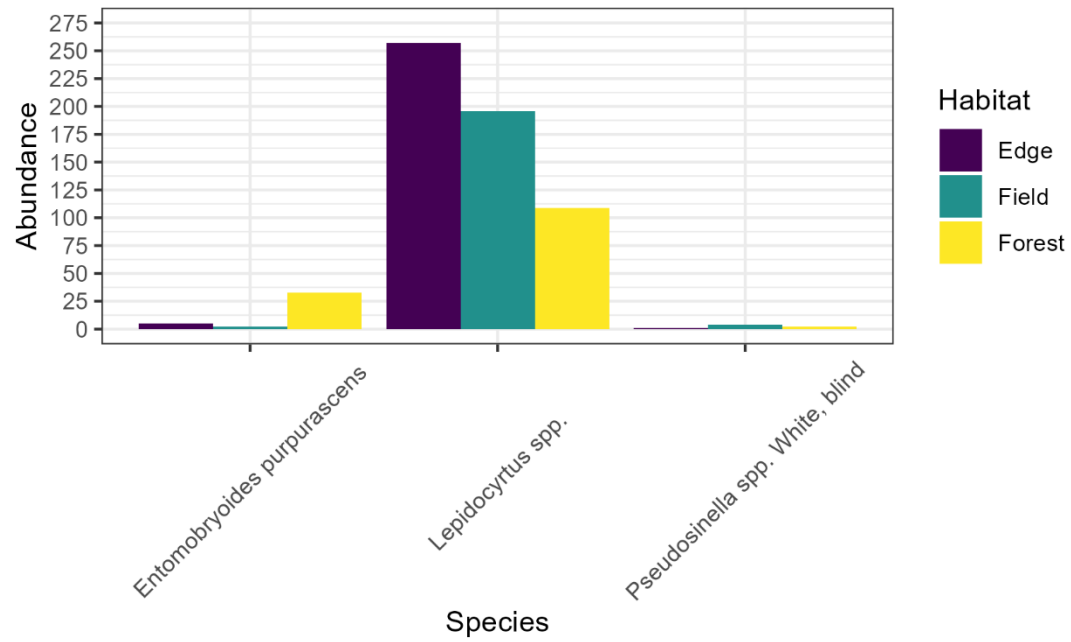


Figure 34: Entomobryidae species for pitfall traps, Arnold Air Force Base, May 2023.

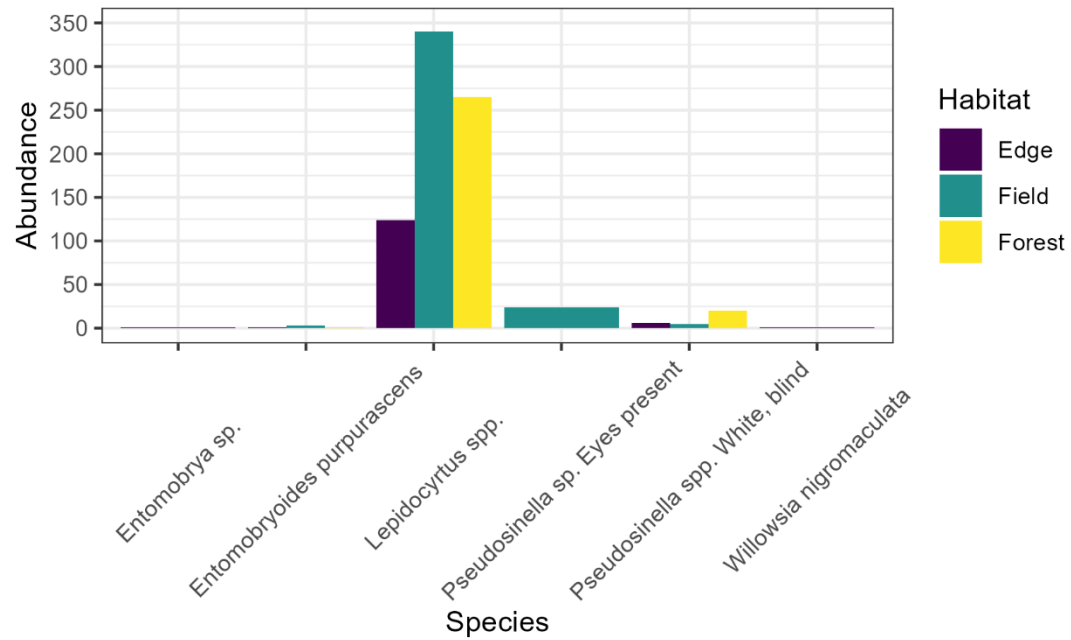


Figure 35: Entomobryidae species for leaf-litter samples, Arnold Air Force Base, May 2023.

Table 6: All Sørensen values calculated for the Collembola community, Arnold Air Force Base, May 2023.

Habitats Being Compared	Overall	SLAM Traps	Pitfall Traps	Leaf-litter Samples
Forest & Edge	96.8%	60.0%	100.0%	80.0%
Field & Edge	93.3%	72.7%	96.0%	80.0%
Forest & Field	90.3%	75.0%	96.0%	72.7%

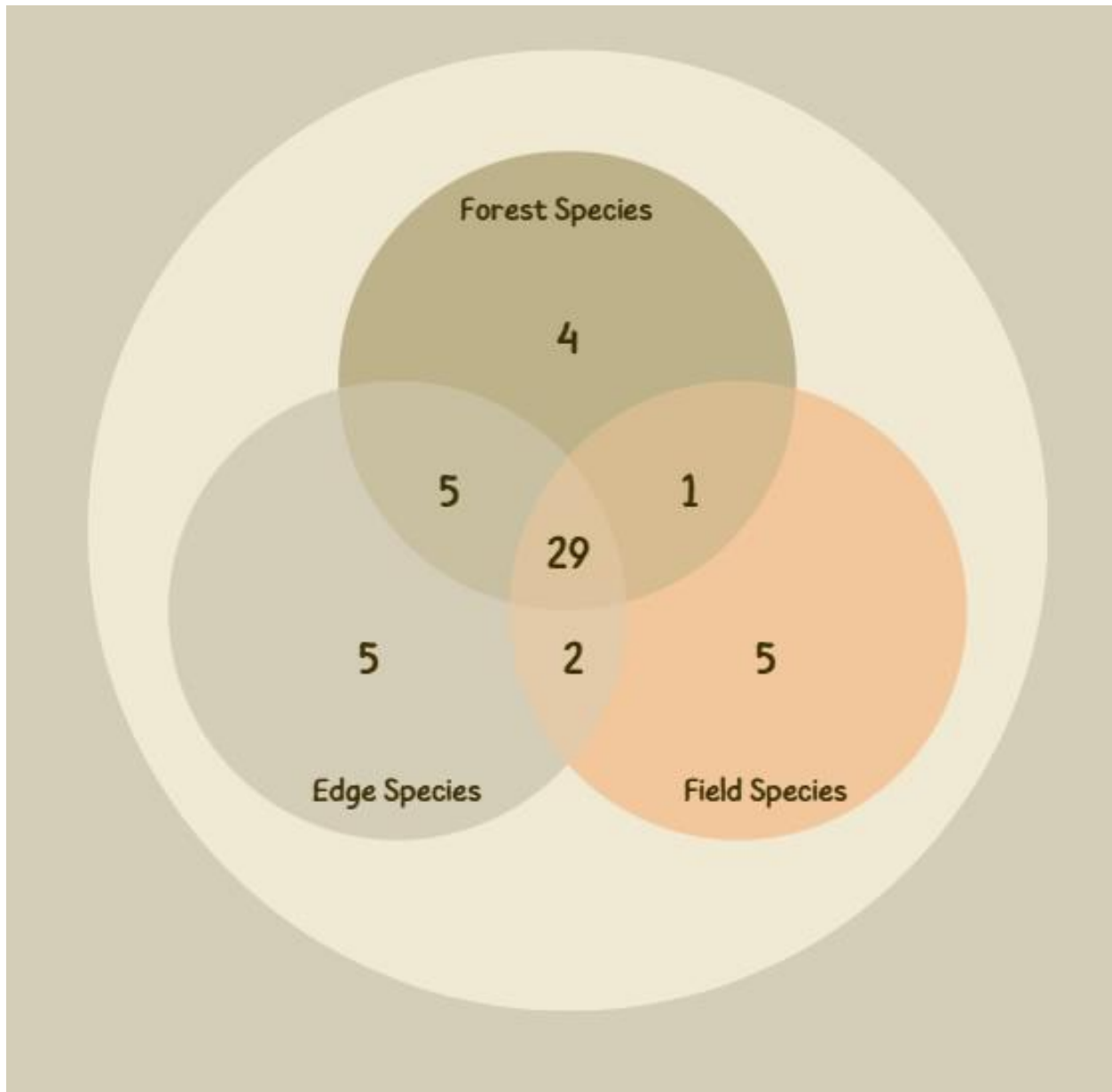


Figure 37: Venn diagram of Collembola taxa for all trapping methods, Arnold Air Force Base, May 2023.

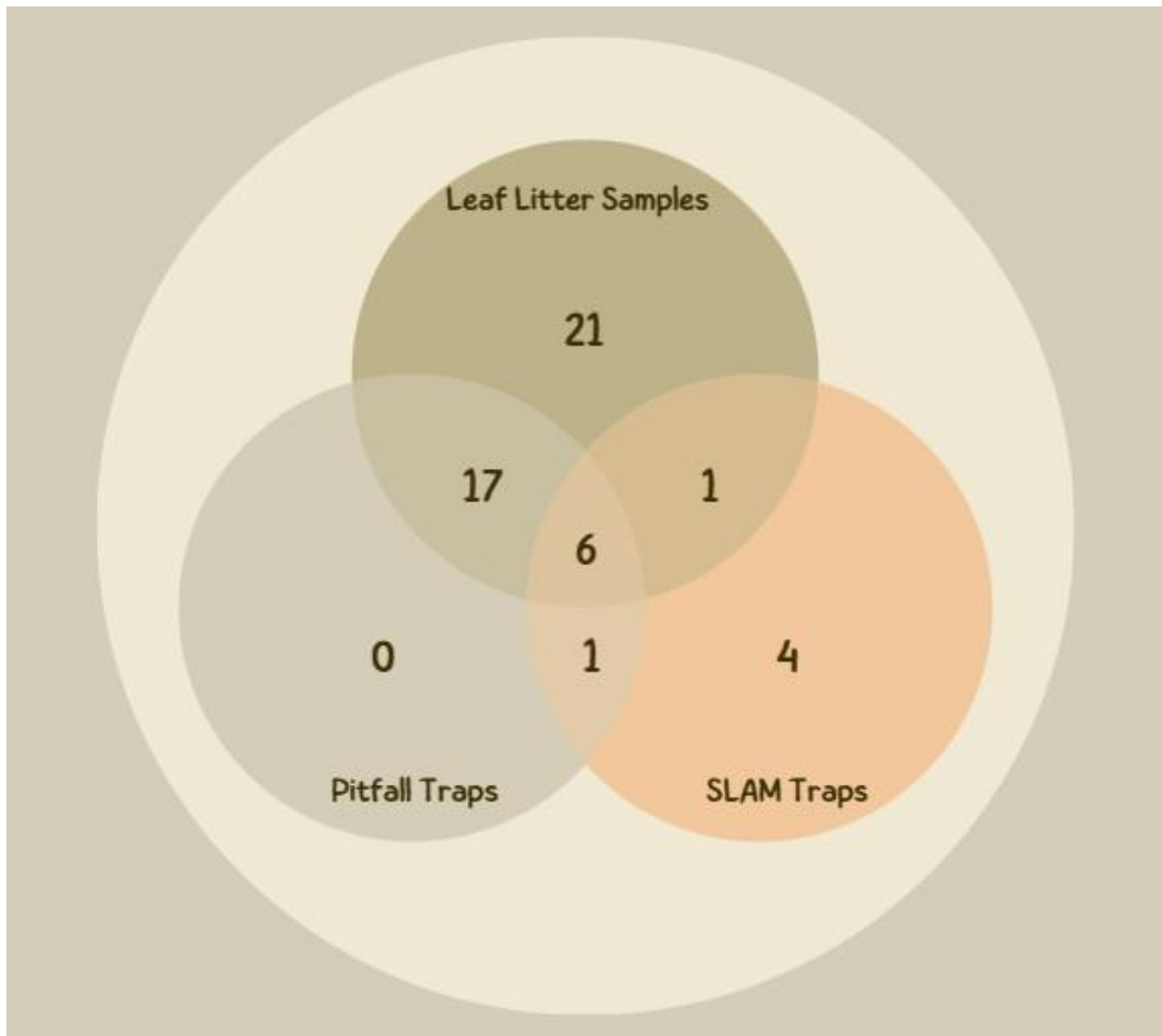


Figure 38: Venn diagram of Collembola taxa by collected method, Arnold Air Force Base, May 2023.

Chapter 4: Discussion, Future Directions & Conclusions

Discussion

Mites accounted for the majority of arthropods within leaf-litter samples (22,153) but were not differentiated into families due to both the lack of taxonomic expertise and the presence of most mites as six-legged, first-stage nymphs. The abundance of mites depressed most H' diversity values for leaf-litter samples; therefore, H' values were also calculated with mites excluded. Pitfall trap mite numbers were assessed, as pitfall traps also captured many mites. However, H' diversity varied little regardless of inclusion of mites in the calculations for pitfall traps.

Sweep-net samples captured the lowest number of arthropods, 861 (~1%). Sweep-netting was difficult given the brush and vegetation within the sites. For this reason, sweep-net samples do not accurately represent a good cross-section of arthropod fauna within the areas being sampled. It is also difficult to ensure sweeps are consistent as sweepers may vary from month to month and have different techniques. The forest groundcover consisted largely of tangled, thorny vines of *Smilax* spp., the field had woody brush remaining from the burn, and the edge habitat was thick with tree seedlings, small saplings and a variety of shrubs and herbaceous plants. Thus, while sweep-netting is a common collection method in monocultures, grasslands and meadows, it was not a useful choice for this project; insect numbers were low and largely a subset of SLAM trap catches. Sweep-net samples collected two fly families (Therevidae and Athericidae) that were unique to this sampling method.

Family accumulation curves demonstrated that for leaf-litter samples richness was beginning to plateau. Most trapping methods had low family richness when compared against the minimum sample sizes, but this was not the case for pitfall traps (Fig. 11). The family richness curve and rarefaction curves for these sampling methods suggest ground-dwelling and soil-dwelling organisms were adequately sampled. When considering family accumulation curves with the Michaelis-Menten model equation, pitfall traps, SLAM traps and leaf-litter samples performed well in capturing family richness. Their maximum family richness values were not far from the actual number of families collected. Sweep-net samples were not efficient and were more than 30 families short of the extrapolated maximum family richness. K values are relevant to family richness, as low K values typically mean higher efficiency or sensitivity to changes in sampling effort. Therefore, a low K value implies families would be collected quickly with moderate sampling effort. With sweep-net samples having the highest K value (16.3), sweep-netting would require more sampling effort than what was done in this project.

The differences in Sørensen heterogeneity between trapping methods suggest that ground-dwelling arthropods had different activity than flying arthropods. Pitfall traps in forest habitats and edge habitats were more similar than the other pairings, and for leaf-litter samples forest and field habitats were more similar. These similarities deviate from the overall results that edge habitats and field habitats were most similar. The heterogeneity of arthropods caught in forest and edge pitfall traps suggests that the specimens caught preferred a more shaded or sheltered

habitat. The heterogeneity of arthropods caught in forest and field leaf-litter samples is interesting because those habitat types were not directly adjacent to each other and have little apparent similarity in plant communities.

As expected, arthropod abundance was higher in warmer months. Active trapping methods had higher diversity values in the warmer months, but sampling month was not statistically significant with diversity values for any trapping methodologies. Abiotic or biotic factors that may have affected diversity values could include the degree of shading, relative amount of organic litter and the amount and diversity of vegetation cover and vegetative diversity within the habitat sites.

Correlation of precipitation with abundance and diversity was assessed. Accumulated precipitation within a sampling month was correlated with abundance and diversity for SLAM traps. SLAM traps and precipitation were positively correlated for measured abundance of specimens caught, suggesting that precipitation affects the capture of flying arthropods more than the that of ground-dwelling arthropods; measures of SLAM trap diversity were negatively correlated with precipitation.

An increase in arthropod abundance in forest sites occurred in March for passive trapping methods. These sites were partially flooded with ponding up to 40 cm deep. Most of the SLAM trap specimens collected in March were semi-aquatic dipterans such as Chironomidae; thus, abundance was high, but diversity was low. Maps of the study area show ephemeral ponds fed by a small watercourse, which from observation are dry in summer and fall. These forest sites likely serve as breeding areas for aquatic flies in late winter and spring.

Data was analyzed two ways for the results; totals by habitat type were used to calculate means of different statistical measures, and totals were split apart by trapping method, and grouped by habitat and month. Calculating statistics with totals by habitat type increases the number of sampling events (N) and allows for stronger statistical measures.

By-catch occasionally was found in pitfall traps. A forest pitfall trap collected in September contained 15 voles at the time of trap retrieval. These voles may have acted as an attractant for necrophilous beetles in the families Silphidae (carrion beetles) and Scarabaeidae (true dung beetles). Accidental baiting can potentially skew results by misrepresenting the densities of species attracted to the bait. On the other hand, specialized studies to collect decomposition and forensically relevant species are recognized as a valuable method (Leasure et al 2012). While this trap was not intentionally baited, it may be something to consider moving forward with other biodiversity and abundance research projects to obtain a more holistic view of the arthropod community in the area, including the decomposers such as those in this study.

In this project four different collection methodologies were used: SLAM and pitfall traps, leaf-litter samples and vegetation sweep-nets. However, it was recognized *a priori* that they would

not provide a complete inventory of the area's arthropods. Certain taxonomic groups were not efficiently collected for this project. Pheromone traps were not used and neither CO₂ traps nor light traps were deployed. Therefore, families such as Culicidae and many nocturnal lepidopteran families were not adequately sampled (Macgregor et al 2015). Few Heteroceridae or Trichoptera were caught, taxa that are typically attracted to light traps (Thein & Choi 2016). Other aquatic orders also were not collected. Arthropod communities can be vertically stratified, so this part of the fauna was not sampled. A suspended Lindgren funnel trap with pheromones at each site would have benefitted the analysis. No baited traps were used (except the vole-packed pitfall trap described earlier); such traps would have been useful in assessing the diversity of bark beetles as well as the necrophagous fauna of an area (Ozcan et al 2014). The chosen trapping methods seem to have provided a more realistic view of abundance rather than diversity. Additional collecting methodologies likely would have increased diversity.

As this project was a subset of a much larger project, data for comparison is available. In the year 2022 a SLAM trap was placed in the forest habitat (site 2) for over one month, compared to traps placed for two weeks in 2023. Number and composition of families caught and their abundances differed between the two years. It is difficult to evaluate these differences as the results are not directly comparable due to different time lengths of time they were deployed. The 2023 project had two SLAM traps placed per site rather than one. 2022 SLAM sampling was also done during a different month than the 2023 sampling periods; although varying weather and the field in a pre-burn condition would make alignment difficult, more data and consistent sampling methods need to be utilized to better understand the changes in arthropod abundance and diversity over a several-year span.

This project was structured in a way that makes it easily repeatable for meaningful estimates of the effects of climate change on the arthropod biota. Projects such as the AAFB survey can provide insight into the best ways to estimate and preserve arthropod biodiversity. General conclusions are difficult to draw between 2023 data and 2022 data, but if this project was repeated in 5, 10, or even 15 years the different data sets could be compared long term. This would add to an area of research that is of growing interest and importance. Studies that look at arthropod or insect biodiversity as a whole are scarce, and it is important to understand the changes within the community, as those changes affect ecological processes throughout the world. With this study having taken place at an Air Force Base, it is likely this habitat will be maintained for many years to come, making it an ideal location for a longitudinal arthropod abundance and diversity study.

Another limitation of this study is that it was only done for one year. While this study provides general trends throughout 2023, the possibility exists that 2023 had anomalies in arthropod abundance and diversity due to weather events. Repetition of this project, even with a focus on certain taxa rather than the entire community, would provide insight into weather events that might provide some predictability. Sampling for this project occurred during a controlled burn

year, but because the burn was conducted on very short notice there was not an opportunity for an immediate pre-burn sampling. Collecting data between the burn cycle years would show how the prescribed burn is affecting arthropod abundance, diversity and successional processes.

The results from Collembola collections in May do not support the hypotheses made for the overall project. Highest abundance and diversity were in the field habitat. This result is surprising, as the controlled burn turned much of the leaf-litter to ash, and Collembola are associated with leaf-litter (Pollierer & Scheu 2021). The fires on base are typically low-intensity burns that appear to have little or no effect on organisms in moist surface organic matter and below the soil surface. Most Collembola deposit eggs in soil crevices or other moist, protected areas, and young juveniles tend to stay in those locations. The controlled burn could also have allowed for multiple generalist or opportunistic species to colonize within the newly-burned field areas, as other studies have shown (Urbanovičová et al. 2014). Surprisingly, the edge Collembola were more abundant than the forest, which had leaf-litter year-round, while the edge was thick and overgrown with shrubs, grasses, and lacking an abundance of leaf-litter. However, observations of leaf-litter abundance in this study were limited to the ease of obtaining it for Tullgren funnels, not on an analysis of total surface organic litter. Much litter deposit is rapidly transformed into smaller, more compact pieces difficult to recognize and collect. Furthermore, it is possible that the Collembola in these areas are better adapted to burns and perhaps selected for survivability since controlled burns are carried out every few years.

The numbers between trapping methods varied drastically. No Collembola were collected in sweep-net samples in May. SLAM traps caught 124 Collembola, but since SLAM traps are primarily used to collect flying arthropods, there may be a climbing cohort among these springtail taxa. The leaf-litter samples collected 3,944 Collembola and the pitfall traps collected 4,794. Leaf-litter numbers may have been lower than pitfalls because drier conditions could have driven springtails to more protected locations, reducing abundance in the drier litter.

According to the Sørensen test of heterogeneity all three habitat types are similar in Collembola composition. Overall Collembola diversity of these sites is also high, all sites having a H' value above 3.0. These findings suggest that the site chosen for this study (Site 26) has a widespread, diverse Collembola community. While diversity was high throughout the study area, overall diversity does not take into account potential behavioral differences and the trap type in which specimens were caught.

Two species of Entomobryidae, *Entomobrya assuta* and *Entomobrya clitellaria*, were collected only in SLAM traps, suggesting that these species are climbing species. Not much is known about the ecologies or species boundaries of these species (Katz et al. 2015). Most researchers sample Collembola with leaf-litter samples, but for some species the findings presented here suggest a SLAM trap may be more suitable. *Pogonognathellus* sp. (Tomoceridae) were caught in

high abundance for leaf-litter samples and pitfall traps, suggesting they are well adapted to strictly staying in leaf-litter.

Among Collembola families Isotomidae species diversity was above 1.5 in all habitats higher than the H' diversity of Entomobryidae species, which was below 1.0 for all habitats. Entomobryidae H' diversity was relevant for both pitfall traps and leaf-litter samples, but Isotomidae was not, as only one species of Isotomidae was collected in May pitfalls.

Looking forward, identification of the rest of the Collembola samples in 2023 will reduce the knowledge gaps of Collembola at AAFB. As with the other arthropod groups, a more detailed understanding of springtail communities over time will be of great value in recognizing effects of global climate change on detritus-based food webs.

Future Directions

Faculty at the University of Tennessee-Knoxville have done various studies in association with AAFB (Vlach et al 2010, Wiggins et al 2007). An established relationship with the base biologists and deep knowledge of this area provides an excellent opportunity for a long-term longitudinal arthropod abundance and diversity study. By doing this type of research in the same area, changes in the arthropod community due to abiotic and biotic factors can be assessed. By performing this study for multiple years, the burn interval schedule could be altered, if necessary, as controlled burn effects are habitat and environment-specific.

Projects such as the AAFB effort are important for multiple reasons. The results generated here contribute to a body of literature that becomes more relevant each year with anthropogenic and climate changes. As insects and other arthropods are ectothermic, changing temperatures will affect development rates and survival. For example, many fly species lay their eggs together in a phenomenon referred to as gregarious oviposition or aggregate oviposition (Brodie et al 2014, Brundage et al 2014, Ruiz-Dubreuil & Kohler 1994). This behavior is an evolutionary adaptation that typically allows for increased fitness of the fly species. Many eggs laid on the same carrion will become a mass of maggots in which the maggot mass secretes digestive enzymes in large quantities as a cohesive unit, allowing them to feed better. They also generate significant heat, which in turn changes their rate of development (Rivers et al. 2010, 2011). In some states where high temperatures are common, such as Texas, maggot masses may end up killing themselves due to extreme temperatures. Research like this can provide the baseline data needed to assess if certain arthropods, like blow flies, are declining or increasing in numbers throughout the years. Behavioral patterns could be assessed to see changes as temperatures fluctuate. Actions in the past that have increased fitness may not be as beneficial in the face of warmer temperatures.

The information from this project is an intensive examination of arthropods in certain habitats within AAFB. The overlying project is also aimed at assessing arthropod diversity and

abundance, but with the addition of other trapping methods (visual observation and beat sampling) and sampling periods. These projects need to be repeated periodically throughout time to develop a complete inventory of arthropods at AAFB, which can be used for extrapolation to other areas. For instance, the All-Taxa Biodiversity Inventory at Great Smoky Mountains National Park (1998-present) is a record of southern Appalachian species, while the AAFB work is a representative record of arthropods on the central plateau region of Tennessee. Continuing inventories serve conservation goals and provide enhanced knowledge of spatiotemporal distributions (Dawson et al. 2021). This is particularly important with regard to new invasive pest species. The oriental latrine fly (*Chrysomya megacephala*) recently was detected in Knoxville (Owings et al. 2021). This fly is a human health hazard as it is known to carry 11–12× the bacterial load of a house fly, including antibiotic-resistant strains of *E. coli*, is closely associated with myiasis and can fully develop within human feces (Chaiwong et al. 2014, Chomchat et al. 2020, Ferraz et al. 2010, Goff et al. 1991, Owings et al. 2018, Rofieq et al. 2020). The effects of weather and climate on the distribution dynamics of this fly species clearly have a human health component. Large-scale comprehensive studies increase the chances of detecting invasive pest species; this same reasoning also applies to predators, parasitoids, pollinators and arthropods of conservation interest.

Four trapping methods were used to gain an understanding of the arthropod fauna within the site, which consisted of mostly diurnal and ground-dwelling/organic litter taxa. Many nocturnal families and bark beetles were not collected in numbers that reflected their abundance, but the addition of additional types of traps as well as directed net collection (dragonflies, butterflies, etc.) could be added in the future. As with all trapping methods, limitations exist. If this project were to be repeated, incorporation of different trapping methods in addition to what was used here would be beneficial. Trapping should be done monthly, rather than bimonthly, if possible, to ensure the researcher is getting data from each month, rather than each season. However, the ability to sort and identify trap contents is a serious limitation, and increased levels of trapping will require more people to process and identify specimens or more time to count what was captured, as numbers would more than double with addition of other collection methods.

To further assess the influence of the edge effects on adjacent habitats, pitfall traps could be set up in a transect that increasingly moves away from the edge. This approach has been done before with specific groups of arthropods (Magura et al. 2017), but edge effect knowledge would benefit from a holistic approach as was done with this project. This approach would give a more precise estimate of how far the edge effect influence reaches for arthropod behavior.

Projects like the AAFB study could potentially be made easier with the use of DNA metabarcoding. The volume of specimens collected and sorted throughout the year limited finer taxonomic distinctions and made identifications of taxa to genus or species more difficult, especially for micro-lepidoptera, micro-hymenoptera, many acalyptate flies and several beetle families. Implementation of metabarcoding could drastically reduce time spent on taxonomic

identifications and allow researchers to apply scientific names to some of these problematic taxa. A metabarcoding approach does not predict the number of specimens but can provide identifications if sequences are deposited in an accessible database. Alternatively, sequences of individual specimens could be sequenced to establish identity, especially if the taxon is a new potential pest or suddenly abundant. DNA analysis would be helpful regardless of project objectives. However, many species have not yet been uploaded to the national database. For example, micro-lepidoptera are abundant and diverse but difficult to identify, thus the search for solutions to this issue (Belitz et al 2024). In the current study, 56 species of Collembola in 17 families were found. Some of the species are poorly described and at least three possible undescribed species have been collected. These taxa will need to be sequenced at a later date to get a better understanding of these arthropods.

Conclusions

Arthropod abundance was highest within edge habitats, but the diversity was not statistically significant between habitat types. Abundance and diversity increased after the controlled burn, suggesting that the fire did not negatively affect arthropod numbers. When testing for correlation of abundances and diversity values with precipitation, only SLAM trap results were significantly correlated for both. This correlation is likely due to SLAM traps catching flying or climbing arthropods that are likely more affected by rain, wind and temperature than ground-dwelling arthropods.

The SLAM traps were effective at catching arthropods within an area, especially winged species, but also collected wingless species that apparently climbed the netting and fell into the collection jar. Among these speculated climbers the most sharply defined groups were Collembola in Entomobryidae and Sminthuridae, species of which were rarely or never caught with any other method. However, SLAM traps do not necessarily indicate environmental preference, as they passively catch arthropods that may be flying through to their preferred habitat. In this project, sweep netting failed to provide much useful information. An expanded version of this project probably does not require sweep-netting. Instead, additional passive collection methods can be added, including Lindgren funnels and canopy traps. Active collection methods, such as black lighting, bait traps and CO₂ traps, also should be added for the most comprehensive results, with recognition that these traps may attract fauna from areas not in the survey plan.

Studies of arthropod ranges and expansions, ecological interactions, and ecological processes are important for developing predictive climate change-arthropod diversity models and changing ecological relationships in relation to other biota. As prescribed burns are used throughout the world for various purposes, their effects on invertebrate fauna are relevant understanding the effects of incorporation of a burn into a management environment. Edge environments, critical in conservation of the local fauna in fragmented environments, are cosmopolitan and can be found in a multitude of ecological states such as anthropogenically created edges or naturally created

edges. The interactions between not only arthropods, but all organisms, are important to better understand the basic relationships within that environment.

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Appendix

Orders, Groups & Families Collected for This Project

Orders/ Groups: Acari, Araneae, Archaeognatha, Blattodea, Chilipoda, Collembola, Coleoptera, Copepoda, Diptera, Diplopoda, Diplura, Heteroptera, Hymenoptera, Isopoda, Isoptera, Lepidoptera, Mecoptera, Neuroptera, Opiliones, Orthoptera, Phasmatodea, Protura, Psocoptera, Pseudoscorpiones, Thysanoptera, Trichoptera

Families:

Archaeognatha: Machilidae

Blattodea: Ectobiida

Collembola: Bourletiellidae, Dicyrtomidae, Entomobryidae, Hypogastruridae, Isotomidae, Katiannidae, Neanuridae, Neelidae, Odontellidae, Onychiuridae, Orchesellidae, Paronellidae, Sminthuridae, Sminthurididae, Tomoceridae, Tullbergiidae

Coleoptera: Anthicidae, Bostrichidae, Buprestidae, Carabidae, Cerambycidae, Coccinellidae, Cleridae, Chrysomelidae, Corylophidae, Curculionidae, Dytiscidae, Elateridae, Erotylidae, Heteroceridae, Histeridae, Latridiidae, Leiodidae, Lycidae, Melandryidae, Melodidae, Melyridae, Mordellidae, Nitulidae, Phalacridae, Ptiliidae, Ptilodactylidae, Scarabaeidae, Silvanidae, Staphylinidae, Tenebrionidae

Diptera: Agromyzidae, Anisopodidae, Anthomyiidae, Anthericidae, Asilidae, Bibionidae, Calliphoridae, Cecidomyiidae, Ceratopogonidae, Chironomidae, Chloropidae, Clusiidae, Conopidae, Culicidae, Drosophilidae, Dolichopodidae, Empididae, Ephydriidae, Helomyzidae, Micropezidae, Muscidae, Mycetophilidae, Pipunculidae, Phoridae, Psychodidae, Rhagionidae, Sarcophagidae, Scathophagidae, Sciaridae, Sciomyzidae, Sphaeroceridae, Syrphidae, Tabanidae, Tachinidae, Tephritidae, Thervidae, Tipulidae, Trichoceridae, Ulidiidae

Diplura: Japygidae

Heteroptera: Acanaloniidae, Aleyrodidae, Anthocoridae, Aphididae, Aradidae, Berytidae, Cercopidae, Cicadellidae, Cixiidae, Coccoidea, Corixidae, Delphacidae, Derbidae, Flatidae, Geocoridae, Lygaeidae, Membracidae, Miridae, Nabidae, Pentatomidae, Psyllidae, Reduviidae, Rhyparochromidae

Hymenoptera: Aphelinidae, Apidae, Bethyridae, Braconidae, Ceraphronidae, Chalcididae, Chrysididae, Cynipidae, Diapriidae, Dryinidae, Encytridae, Eulophidae, Eupelmidae, Eurytomidae, Evaniidae, Figitidae, Formicidae, Halictidae, Ichneumonidae,

Megaspilidae, Mymaridae, Multilidae, Pelecinidae, Platygasteridae, Pompilidae, Proctotrupidae, Pteromalidae, Rhophlosomatidae, Scelionidae, Tenthredinidae, Tiphidae, Trichogrammatidae, Torymidae, Vespidae

Lepidoptera: Arctidae, Crambidae, Lycaenidae, Noctuidae, Notodontidae, Nymphalidae, Pyralidae, Tineidae, Tortricidae

Mecoptera: Panorpidae, Chrysopidae, Coniopterygidae, Hemerobiidae

Orthoptera: Acrididae, Gryllidae, Rhaphidiphoridae, Tettigonidae, Trigonidae

Phasmatodea: Phasmidae,

Psocoptera: Amphipsocidae, Caeciliusidae, Dasydamelidae, Lachesillidae, Lepidopsocidae, Psocidae, Stenopsocidae

Thysanoptera: Aeolothripidae, Phlaeothripidae, Thripidae

Trichoptera: Molannidae

Collembola Species Collected in May

Bourletiellidae: Unknown species

Brachystomellidae: Brachystomella sp.

Dicyrtomidae: *Dicyrtoma hageni*, *Ptenothrix atra*, *Ptenothrix marmorata*,

Entomobryidae: *Entomobrya assuta*, *Entomobrya clitellaria*, *Entomobrya* sp., *Entomobryoides purpurascens*, *Lepidocyrtus* spp., *Pseudosinella* sp. (Eyes present), *Pseudosinella* spp. (White, blind), *Willowsia nigromaculata*

Hypogastruridae: *Ceratophysella* sp., *Xenylla* sp.

Isotomidae: *Desoria* sp, *Folsomia prima*, *Folsomia stella*, *Folsomides* sp. (Green, 5+5 eyes), *Folsomides* sp. (White, 1+1 eyes), *Isotoma* sp., *Isotomiella minor*, *Isotomurus* sp., *Micrisotoma* sp., *Proisotoma minima*, *Proisotoma minuta*, *Tetracanthella* sp.

Katiannidae: *Katiannina macgillivrayi*, *Polykatianna radicularia*, *Sminthurinus cf. quadrimaculatus*, *Sminthurinus* sp.

Neanuridae: Unknown species, *Friesea* sp., *Micranurida* sp. (1+1 eyes), *Micranurida* sp. (blind), *Morulina* sp., *Neanura* sp., *Pseudachorutes* sp., *Sensillanura* sp.

Neelidae: *Megalothorax* sp.

Odontellidae: *Superodontella* sp.

Onychiuridae: *Paronychiurus ramosus*

Orchesellidae: *Orchesella celsa*, *Orchesella* sp.

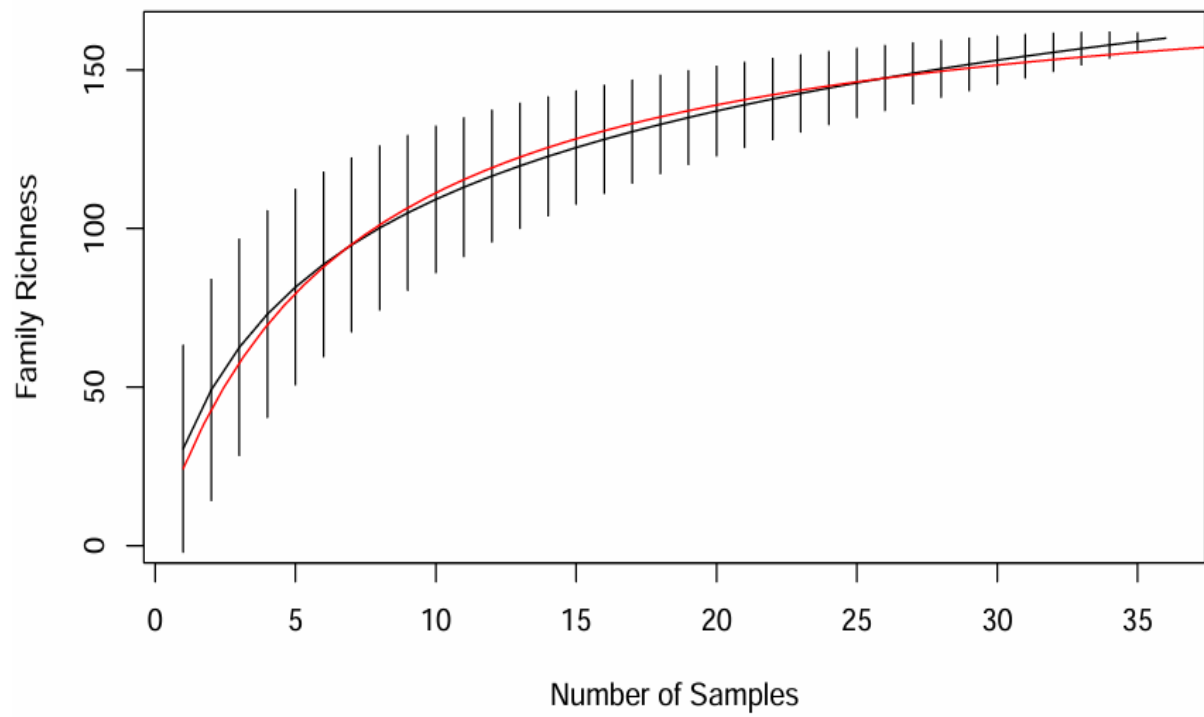
Paronellidae: *Salina* sp.

Sminthuridae: *Neosminthurus* sp., *Sminthurus* sp. (green)

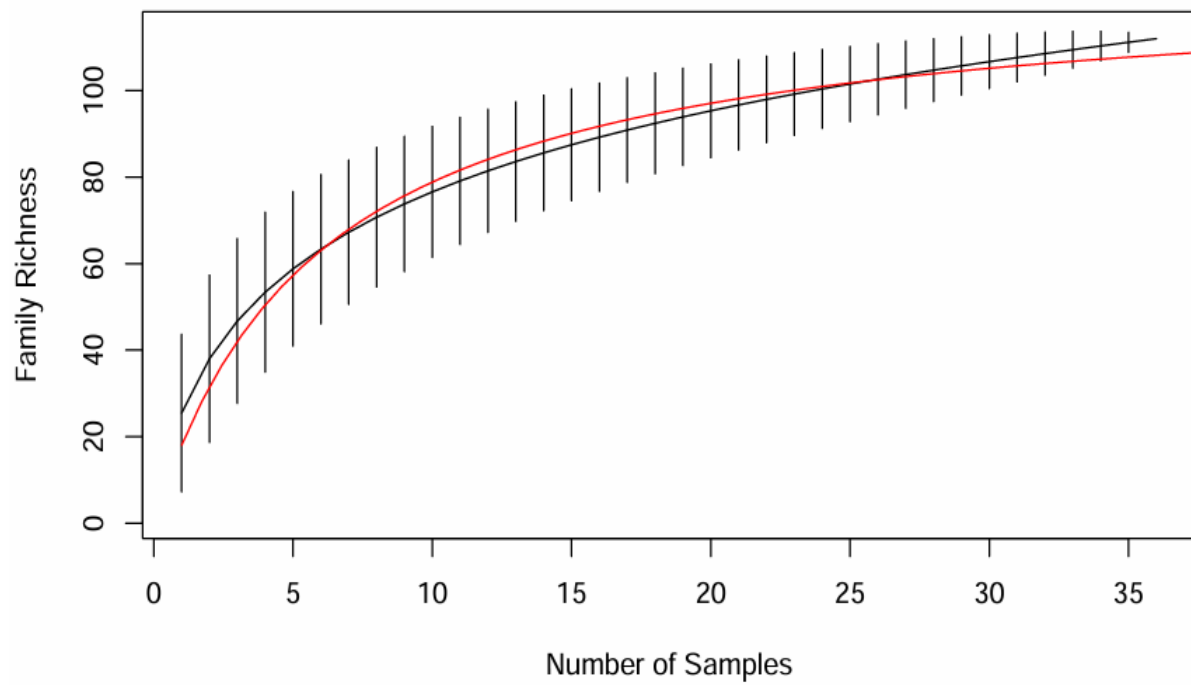
Sminthurididae: *Sminthurides* spp., *Sphaeridia pumilis*

Tomoceridae: *Pogonognathellus* sp.

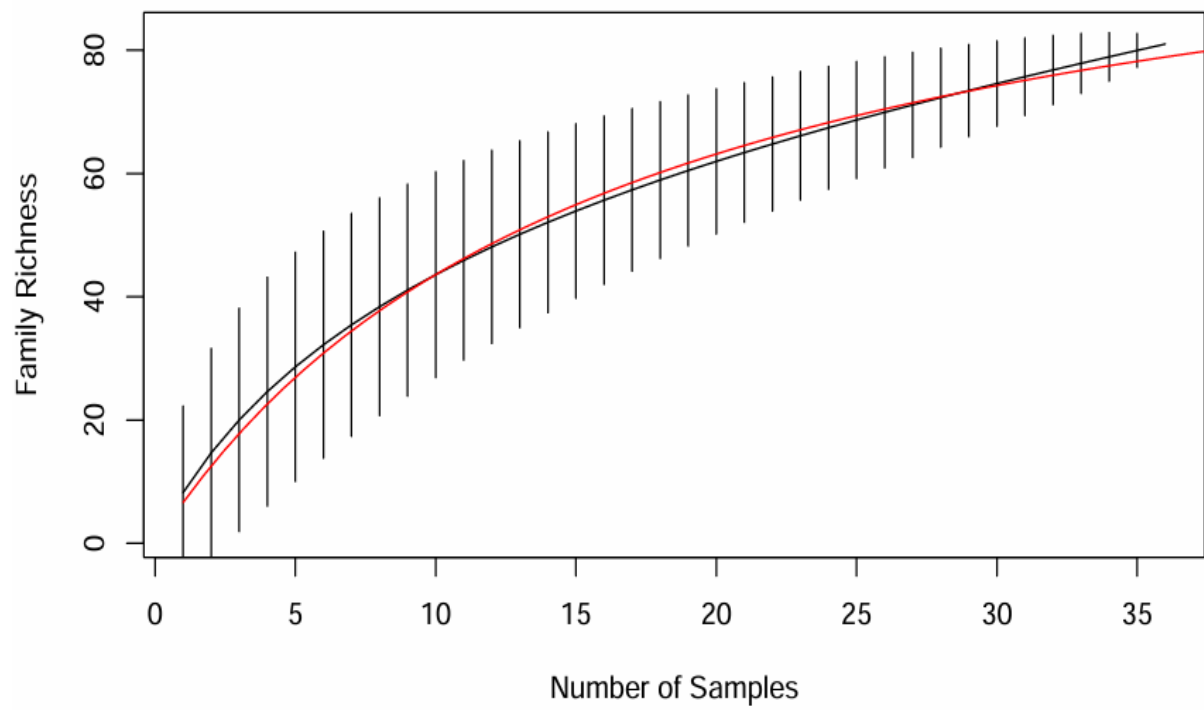
Tullbergiidae: *Mesaphorura* sp.



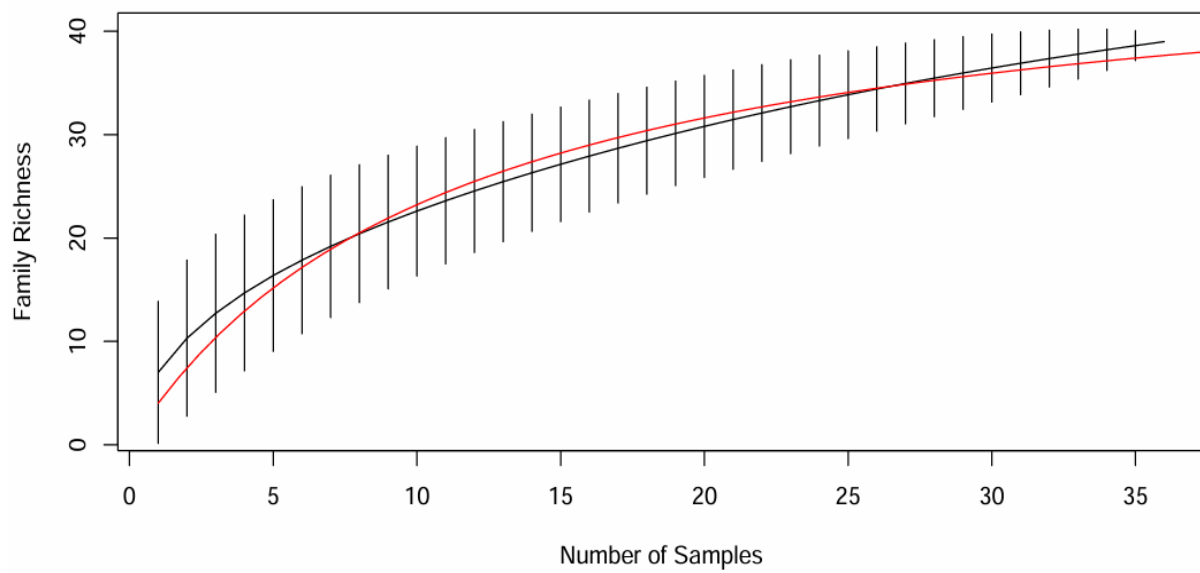
Appendix 1: SLAM trap family accumulation curve plotted against Michaelis-Menten Model.



Appendix 2: Pitfall trap family accumulation curve plotted against Michaelis-Menten Model.



Appendix 3: Sweep netting family accumulation curve plotted against Michaelis-Menten Model.



Appendix 4: Leaf-litter samples family accumulation curve plotted against Michaelis-Menten Model.

Vita

Originally from northern New York, Makhali Voss grew up in Oswego, NY. After high school, she attended the State University of New York College of Environmental Science and Forestry (SUNY ESF), where she received her Bachelor of Science degree in environmental biology, with a minor in business management. Before pursuing her graduate education, Makhali worked as an office manager at J&T Automotive in Syracuse, NY. After being in the workforce for a few years, she decided to go to graduate school, leading her to the University of Tennessee, Knoxville (UTK), where she obtained a Master of Science in entomology and plant pathology with a concentration in entomology. Her research interests include forensic entomology and disturbance ecology. After graduation, she will begin working towards a Doctor of Philosophy degree in entomology at UTK. She is incredibly thankful for her advisor, lab members, mother, and boyfriend, who have all provided the support needed while obtaining her degree.