12-2021

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Citation

Jones, A. K. (2021). Habitat Assessment of Ozark Glades Using Selected Epigeal Predatory Arthropods. *Graduate Theses and Dissertations* Retrieved from https://scholarworks.uark.edu/etd/4278

Habitat Assessment of Ozark Glades Using Selected Epigeal Predatory Arthropods

A dissertation submitted in partial fulfillment
of the requirements for the degree of
Doctor of Philosophy in Entomology

by

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December 2021
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Abstract

Epigeal predatory arthropods were collected with pitfall-style traps in nine sandstone glades of varying ecological stages, and management histories, in the Ozarks of Northern Arkansas over three years. These sites were categorized into three Site Types; Intact, high quality glades; Degraded, forest-encroached glades, and Restored glades which had received woody vegetation removal and burning. Collections of ground spiders (Gnaphosidae), predatory ground beetles (Carabidae), and the sole species of scorpion present in the Ozarks, *Centruroides vittatus*, were reported on as well as the applicability of arthropod collection methods for glade habitat assessment. Habitat characteristics were recorded for each site and analyzed for differences between sites and correlation to the diversity of arthropods identified from two seasons of trapping. Intact and Degraded glade sites were found to have different habitat characteristics as well as different species of epigeal arthropods in collections. In general, the Restored Site Type had two sites that appeared to group with Intact sites, and one seemingly more like Degraded sites. This study showed comparing trap catches of the selected taxa among glade sites had value for assessment.

Acknowledgements

I would first like to acknowledge Susie Makke, who, during the timespan of work for this dissertation, took on a diversity of roles including loyal supporter, loving partner, therapist, cheerleader, field hand—and fiancée. Susie Makke Jones, added not only a surname (and a "unique" husband), but also roles of wife, and mother as well. Atlas Driver Jones, happy first birthday, Son, I got you this dissertation. You and your mother are truly the loves of my life.

To Mom and Dad, who have always been an inspiration. Whether homesteading in the Rockies, building a tiny home in the Ozarks, or working from home while homeschooling me, you have always been ahead of your time, and amazing at what you do. You taught me how to work hard, made sure I could critically think and talk to people, and showed me what true freedom is. I hope to set Atlas on a course for love and adventure—like you did for me.

We are all reflections of the inputs we receive. Teachers bring focus to that reflection. Principles of Biology with Dr. Sandy Tedder inspired me to pursue Biology as a field. Dr. Max Meisch hooked me on entomology and inspired me to teach, and Dr. Carolyn Lewis (a.k.a. The Lady of the Insects) made me believe I could do anything I set my mind to.

Mostly, I owe the quality of this dissertation, and completion of my graduate degrees, to my Advisor and Mentor, Dr. Robert Wiedenmann. Your door was always open, and your encouragement and advice always genuine. The impact you have had on student lives and careers is immeasurable. You punctuated your retirement with glade fieldwork and editing every other element of this dissertation with your signature fine-tooth comb; you are a true humanitarian, and I am both honored and humbled to have had the opportunity to finish under your direction, learn from your leadership, and grow from our friendship. Someday, I aspire to have the impact on students, colleagues, Arkansas, and the world, that you have shown possible.

Preface

Nestled among the hills and hollows of the Ozark Mountains is a diverse array of habitats that contain unique communities of flora and fauna. It was amidst one of these communities, upon a limestone blufftop aside Whitney Mountain in Northwest Arkansas, where I was fortunate enough to be raised. Being a home-taught, only child, raised in this rural setting, I had plenty of time for discovery, and I began to appreciate the natural beauty and diversity of my home at an early age. I spent many days on our severely sloped acreage climbing trees and flipping rocks, catching whatever I could find. Before I knew that different habitats existed, I knew there was something different about the blufftop opening my family called the "Top of the Rock." It wasn't the same as surrounding forests. It was always wet in spring, so much so that "slime" I now know as cyanobacteria would grow, and yet sometimes this was only a meter or two away from cactus, which were more indicative of the scorching summertime heat and drought that would impact this same forest opening. Scorpions, fence lizards, coachwhip snakes, velvet ants, tarantulas, tarantula hawk wasps and giant desert centipedes all inhabit memories of my youthful adventures atop that bluff. These species also helped spark my affinity for studying the natural world, and eventually entomology. It would not be until many years later however, after earning degrees in both biology and entomology, and a serendipitous contact from a colleague seeking knowledge of our native tarantulas, that I would learn that this habitat was specific enough in nature to have its own habitat designation—glade. Upon compiling sources for the present study, I have come to find a dissertation on glade flora from 1978 authored by Keeland. Among his descriptions was that of a glade site named Lost Bridge Village—my home community—which, based on site descriptions, may have actually been within sight of my home. Likewise, glade sites studied by both Grimsley (2009) and Booth (2020) were only a few kilometers from the Top of the Rock. Ironically, the family name given in my youth to that small patch of a disappearing habitat, has since also become the name of an iconic upscale golf and adventure “preserve” outside of Branson, Missouri.

I have become fascinated not only with the uniqueness of the habitat in which I was raised, but also with its historical relevance to my home state and ecoregion. I have also learned about the disappearance of Ozark glades, and regional loss of species associated with them, which have become vividly apparent to me. Many glades remembered in youth have all but disappeared to ever-encroaching forests marching towards successional climax—aided by suppression of once-frequent fire. It is for this reason that glades have become the focus of this work, as I hope that someday others will be inspired by these extreme habitats that are part prairie, part wetland, part desert, and distinctively interwoven into the tapestry of the forest mosaic, as well as the history of Arkansans.

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INTRODUCTION AND LITERATURE REVIEW

Ozark Glades

The Ozark Mountains are the most prominent landscape feature of the Central United States and reach the highest elevations found between the Appalachians to the east and the Rockies to the west. They contain the highest point in Missouri, Taum Sauk Mountain (540 m), and reach an apex at Buffalo Lookout, Arkansas (780 m). Topographically they range from rolling hills draining into slow-moving rivers, to rugged ridges and cliffs connected by deep lush valleys and rocky spring-fed streams. There is great diversity of species and habitat types in the Ozarks with more than 65 unique communities having been reported (Nelson, 2012). The habitat of focus for the present study is that of an Ozark sandstone glade.

Many people are unfamiliar with the term glade and, indeed, those who are can convey different meanings in different regions of the United States with its use. In the Ozarks, glades are rocky grasslands composed of a mix of prairie-associated flora with fauna having lineages rooted in the desert Southwest. Glades were once considered widespread throughout the Ozark landscape mosaic, primarily alongside open woodlands and savannas—all of which have largely converted to forest-dominance in the modern era. Species that remain in the Ozarks as either glade obligates or associates, face challenges related to their isolation in shrinking and fragmented habitats that require management to resist forest succession. Ozark glades have adapted to frequent fire over the past several thousand years, and many of the issues faced by glade-specialist species are complications from a lack of this rejuvenating biotic force. In the absence of fire, invasive and introduced species threaten glade integrity, but through proper management, glades can be restored and maintained so that the diversity of species found here may persist for future generations to experience and learn from.

Region and Geologic History

Many attempts have been made over the last century to delineate physiographic or ecologically unique areas within the United States. These have been referred to as Physiographic Provinces, Natural Regions (Zachry et al., 1979), Natural Divisions (Thom and Wilson, 1980), and Geographic Regions (Skvarla, 2015) but, for the purpose of work herein, are referred to as ecoregions. Referenced ecoregions are those mapped by the United States Environmental Protection Agency.

The first and most inclusive ecoregion level for the Ozarks is that of eastern temperate forest. The Interior Highlands is a level-II ecoregion containing some of the westernmost expanses of eastern temperate forest spanning from eastern Oklahoma across Arkansas, Missouri, and ever so slightly into both Kansas and Illinois (Figure 1). These highlands are bounded by the Gulf Coastal Plain to the south and east and the Central Lowlands to the north and west (Zachry et al., 1979).

The Interior Highlands contain four recognized level-III ecoregions: the Ouachita Mountains, Arkansas River Valley, Boston Mountains and Ozark Plateau. The southern one-third of the Interior Highlands consists of the Ouachita Mountains and Arkansas River Valley. The Boston Mountains bound the Arkansas River Valley to the north. Together the Boston Mountains and Ozark Plateau comprise what most people commonly refer to as “the Ozarks,” which cover some 13.5 million hectares (Nelson, 2012). The Ozark Plateau ecoregion is again divided geologically into the Springfield Plateau, Salem Plateau, and St. Francois Mountains—a range comprising the oldest rock layers in the region.

The geologic history of the Interior Highlands that has led to the ecoregions and topography of today, has periodically been a refuge for untold numbers of species escaping climatic extremes. The shallow seas of the Cretaceous were unable to inundate these highland elevations. The Wisconsin Glacial Episode, the most recent glacial maximum occurring 25,000 years ago in the Pleistocene, was unable to overtake the highlands either, stopping at the northernmost edges of the Ozark Plateau (Dowling, 1956; Allen, 1990). The longevity of the Ozarks as a terrestrial habitat unencumbered by glaciation or inundation for some 300 million years, combined with isolation from other eastern temperate forests and mountains by the Mississippi River Delta, have no doubt been influential factors for the recognition of species endemism in the region. Robison and Allen (1995) suggested there may be as many as 300 species endemic to the Interior Highlands, some of which show connections to related species from a diversity of localities from the Appalachians to the desert Southwest. These species relationships reveal the connection of the Ozarks to several starkly different biomes throughout history. Indeed, it does not take a background in ecology or geology to see the influence of these connections in the region still today. Species assemblages can be found indicative of eastern temperate forests, such as oak-hickory and oak-pine climax communities, Great Plains prairies, and even assemblages of desert-associated species, relics from drier periods of Interior Highlands history. The Xerothermic Interval occurring after the glacial retreat, roughly 4,000-6,000 years ago, allowed species more associated with the desert Southwest to expand their ranges into the Ozarks (Dowling, 1956; Trauth, 1989; Templeton et al., 2001). Hall (1955) suggested the Ozarks have received more moisture since that dry period, which allows for the temperate forests that now dominate. This climatic shift has effectively marooned species such as prickly pear cactus (*Opuntia sp.*), Arkansas yucca (*Yucca arkansana* Trelease), striped bark scorpion (*Centruroides vittatus* Say), giant desert centipede (*Scolopendra heros* Girard), Eastern collared lizard (*Crotaphytus collaris collaris* Say), Western diamondback rattlesnake (*Crotalus* *atrox* Baird & Girard) and Arkansas brown tarantula (*Aphonopelma hentzi* (Girard)) within islands of xeric habitat, such as glades.

Meaning of the Term

If one were to investigate dictionary definitions of glade, it would seem that describing it as an opening in a forest would be accurate, and perhaps it is somewhat so, at least today, in the Ozarks. However, what about elsewhere? Or 150 years ago? And how does this definition allow for differentiation of a glade from a meadow, field, clearing, barren, savanna or prairie? It should be noted that, whereas the term glade has seen limited used in description of habitats internationally (Mota et al., 2008; Klapyta and Kolaczek, 2009), the following discussion is limited to the use of the term in the United States.

Understanding of the term glade entails 1) familiarity with where one happens to be and 2) that the definition is somewhat dependent on vernacular. When speaking of glades, use of the term can be a bit sticky, and it is often unfamiliar to the layperson altogether. Upon first introduction to the term, the Everglades of Florida (known sometimes simply as “The Glades”), are often first imagined. These are indeed, in essence, glades, but vastly different from those found in the Ozarks. Exposure of the public to the term glade has also come from the ad campaign of the popular line of Glade air fresheners sold by SC Johnson since 1956. Scents marketed under this brand have often been adorned with picturesque images of meadows and wildflowers, which, in reality, are more accurate depictions of Ozark glades than are those of the vast sawgrass wetlands of the Everglades. Whereas glade may be a misunderstood or somewhat cryptic term in modern America, it was once popular enough for places such as Glade, Arkansas, and Rocky Glade, Missouri, to bear its moniker.

In the Southeast United States, including Florida, glades are known for being composed primarily of annual grasses and lacking the need for fire to prevent succession to forest (Baskin and Baskin, 2000). However, the glades of the Interior Highlands are quite opposite, being dominated by perennial grasses and almost entirely reliant on fire for existence (Baskin and Baskin, 2000). To compound terminology even more, some authors refer to the glades bearing perennial grasses as “cedar glades” (Meyer, 1937; Steyermark, 1940; Hall, 1955), whereas others see cedar glade as the description of a single sere of glade succession (Keeland, 1978). Still others claim that “xeric limestone prairie” is a better term for rocky glades like those in the Ozarks, and that cedar glade should refer to microhabitats within them (Baskin and Baskin, 2000). It should be noted, however, that not all glades in the region are limestone. The underlying bedrock type is a common adjective in glade discussion, and, thus, as descriptors compound, one can feel the term glade being sliced by the double-edged sword of ecology in a noble attempt to simplify uniqueness with diverse and often inconsistent terminology.

Defining Features

Nelson’s (2012) description of glades as “essentially treeless shallow bedrock openings in woodlands ranging in size from one-half acre to 1,500 acres,” does not stray far from the dictionary description, but becomes exponentially more useful in differentiating glades from other types of open grasslands because of the mention of shallow bedrock. It is this shallow bedrock, overlain by thin soils, that leads to floral assemblages vastly different from surrounding forests, and closer in composition to that of prairies. Ozark glades are seen as early stages of forest succession that have prolonged due to extreme conditions arising from these thin soils, as well as historically frequent fire. Species of vertebrates associated with glades are often paralleled to desert-dwelling relatives, and some charismatic xerophilic invertebrates are also known from these habitats. However, there seems to be a paucity of information pertaining to

Ozark glade-inhabiting arthropods, the study of which could lead to discovering species indicative of, or endemic to, this habitat.

Rock

Many studies point to shallow or thin soils and rock outcrops as common to glades (Jefferies, 1985; Bergmann and Chaplin, 1992; Baskin and Baskin, 2000; Ware, 2002; Van Zandt et al., 2005; Ostman et al., 2007; Nelson, 2012), thus explaining why bedrock type is worthy of adjective status when accompanying “glade” in discussions. The Arkansas River Valley excluded, the Interior Highlands are underlain by Paleozoic sedimentary rocks and Precambrian igneous rock 1.5 billion years in age (Zachry et al., 1979; Nelson, 2012). In the Ozarks, limestone and dolomite predominate, therefore glades on these substrates do as well (Nelson, 2012). However, glades can also be found atop granite, rhyolite, chert, shale or sandstone bedrock (Templeton et al., 2001; Ware, 2002). The exposure of bedrock outcroppings is due to erosion (Van Zandt et al., 2005), which can be the result of bedrock layers near the surface, intersecting a slope, or underlying a drainage. Erosional forces are a major factor in resistance of glades to forest succession, and some suggest these forces can result in an edaphic subclimax being maintained nearly indefinitely (Hall, 1955). The power of erosional forces is displayed particularly well on south- and west-facing slopes, where extended periods of insolation and exposure to prevailing winds prevent organic buildup and decomposition (Steyermark, 1940).

Soils

The soils that are able to persist against erosive forces in glades are often shallow (Keeland, 1978; Ostman et al., 2007), of poor quality (Van Zandt et al., 2005), and unevenly distributed, creating pockets or mosaics of vegetation (Ware, 2002). Ware (2002) thoroughly outlined seven physical properties inherent to glade soils that contrast them with surrounding forests: full insolation, bare soils absent of leaf litter, xeric summer conditions, periodic hydric conditions, extreme soil temperatures, frost heaving, and extreme chemistry.

Thin soils have little ability to hold moisture during the periods of drought that often occur in the Ozarks, which leads to seasonally xeric conditions that limit plant inhabitation of the glade (Meyer, 1937; Ware, 2002; Van Zandt et al., 2005). However, the nature of existing atop bedrock that is impermeable to water can lead to periods of extremely hydric conditions as well, particularly if topography dictates that the area receive drainage from higher elevations with permeable overlying substrates. In the Ozarks, this hydric condition can exist at any time there is sufficient rainfall. However, it is more common in the spring and fall, with drought occurring during the summer months (Keeland, 1978). Depressions in exposed bedrock can act as cisterns harboring aquatic insects during wet periods. These pools, when filled with sediment and vegetation, can even resemble wetlands or bogs until drier conditions prevail.

Glades can be generally classified into categories of calcareous and non-calcareous, based on substrate composition (Ware, 2002). Limestone and dolomite glades are calcareous and tend to have a basic pH in general. The primary difference between soils associated with these two rock types is a higher concentration of magnesium accompanying the calcium in dolomite (Booth, 2020). Booth (2020) also reported limestone glades to be less basic (6.8-7.5) than dolomite glades (7.8-8.1). Sandstone and igneous glade soils are, in general, considered more acidic and non-calcareous. Jeffries (1985) reported that sandstone glades near Calico Rock, Arkansas, were commonly found to be acidic, but noted wide variance in pH (4.1-8.8) within a single glade.

Succession and Vegetation

Booth (2020) claims that the first records of Ozark flora from the early 19th century were from the region's explorers and surveyors, such as William Dunbar and George Hunter, who described oak, hickory and pine forests, with large areas cleared and burned by natives and immigrants alike for the purposes of hunting and agriculture. Many accounts depict the Ozarks as containing more open land than there is currently (Steyermark, 1940; Hall, 1955; Baskin and Baskin, 2000; Ware, 2002; Ostman et al., 2007; Nelson, 2012; Booth, 2020), with savannas existing over much of the Ozarks around the year 1800 (Templeton et al., 2001). One of the more colorful accounts in the literature comes from Hall (1955):

> "Those who know the Ozark landscape consider these local interruptions of forest cover as the particular mark of beauty of the uplands, and the 'openness' was a major feature in selection for settlement in pioneer days. Visitors to the Ozarks from foreign countries usually are most impressed by the red cedar glades which are island playgrounds in the present-day dense oak woods. In the proper season these open areas serve as edge for much wildlife activity, and perhaps the most characteristic features in this wise are the summer call of the chuck-will's-widow and the 'booming' of the nighthawk that so often chooses a glade for his target… Today the Ozarks are well covered with close forests, and the once open areas show signs of rapid forest encroachment."

The encroachment described by Hall, due to continued forest succession in the absence of periodic fire, comes as a result of native woody species increasing canopy cover and, subsequently, soil depths. Forest succession within the Ozarks was first reported by Steyermark (1940), who described oak-hickory as the "common and prevailing association over most of the Ozark ridges, uplands, and upper slopes of hills." Along with the oak-hickory communities, he also described sugar maple-white oak and oak-pine as being alternate climax communities. For all of these climaxes he suggested glades to be "the original condition which existed all over the Ozarks," and that this gives way to the second stage— "cedar glades, red cedar glades or red cedar balds" (Steyermark, 1940).

Studies of Ozark glade flora are far more prevalent than those of other aspects of the habitat. Glades progress from bare rock and lichen, to grasses and forbs (Steyermark, 1940; Keeland, 1978), which differ from those associated with disturbed areas (Ware, 2002). In the absence of periodic fire, glades eventually succeed to forest climax communities gradually changing in species composition. Differences in floral composition inherent with successional progression of Ozark glades, when combined with an array of possible soil compositions and bedrock types, make blanket statements about their species assemblages difficult.

Glades are first and foremost open grasslands, having no more than 30% canopy cover (Kimmel and Probasco, 1980), and are most often compared to prairies in vegetative composition (Steyermark, 1940; Hall, 1955; Baskin and Baskin, 2000; Ware, 2002; Van Zandt et al., 2005; Ostman et al., 2007). Whereas grasses do predominate in the communities, they are certainly not alone. Glade-associated forbs distinguish these habitats from being simply “rocky prairies” (Ware, 2002), and hundreds of species of plants, including endemics, have been recorded from glades of the Ozarks (Nelson, 2012; Ostman et al., 2007). Some endemics include Missouri coneflower (*Rudbeckia missouriensis* Engelm. ex Boynton & Beadle), Ozark coneflower (*Echinacea paradoxa var. paradoxa* (Norton) Britt.) (Thom and Wilson, 1980; Van Zandt et al., 2005), and Mead’s milkweed (*Asclepias meadii* Torr. ex A. Gray), which is considered an indicator species of Ozark glade health as well as endemic (Bell, 2007). These three endemic species seem to also have relationships to tallgrass prairie species.

The diversity found in glades is driven in part by the variety of soil chemistries that exist due to differences in underlying bedrock. Baskin and Baskin (2000) published a comprehensive synthesis of studies of Ozark limestone/dolomite glade flora. Their synthesis included summaries of descriptions from Steyermark (1940), Hall (1955), Kucera and Martin (1957), Keeland (1978), and several others. Since publication of that synthesis, Booth (2020) inventoried and compared vegetation of limestone and dolomite glades in Northwest Arkansas, finding dolomite glades to have both greater richness and more unique taxa. However, the overall floral diversity was not significantly different between glade substrates. Ware (2002) considered all early-sere Ozark glades to be floristically similar, as "most genera of cryptograms and small dicots occur on both calcareous and non-calcareous glades." However, he also noted that some species within these genera did differ between rock types. Compared to calcareous glades, sandstone glades have received little attention. Floral community composition of sandstone glades in west-central Missouri has been reported by Flaspohler (1999), and has been documented from sites near Calico Rock, Arkansas, by Jeffries (1983). The plant species inhabiting Ozark glades mentioned in the above-cited studies have been compiled in Appendix A by bedrock type for the most common rock types.

With factors for floral variation considered, if one were forced to select representatives of the Ozark glade community as a whole, two plant species stand out and are mentioned by almost every author, no matter what other descriptors the glade may carry: little bluestem grass (*Schizachyrium scoparium* (Michx.) Nash), and Eastern red cedar (*Juniperus virginiana* L.). These two species are "conspicuous and dominant plants when the entire vegetational complex is considered" (Ware, 2002), and can also be seen as opposing each other ecologically, with *S. scoparium* playing the role of conscientious objector to forest encroachment at the hands of *J. virginiana*. Other common early-sere glade plants include grasses, such as big bluestem (*Andropogon gerardii* Vitman), Indian grass (*Sorghastrum nutans* (L.) Nash), and switchgrass (*Panicum virgatum* L.), and forbs such as flowering spurge (*Euphorbia carollata* L.), prairie tea (*Croton monanthogynous* Michx.), whorled milkweed (*Asclepias verticillata* L.), and hairy woodland sunflower (*Helianthus hirsutus* Raf.) (Appendix A). Late-sere woody species that invade along with cedars include black oak (*Quercus velutina* Lam.), fragrant sumac (*Rhus aromatica* Aiton), and winged elm (*Ulmus alata* Michx.) (Appendix A).

Marooned Fauna

Certain faunal groups indicative of glades within the Interior Highlands have also received attention, but not nearly to the same degree as have plants. Charismatic and economically important vertebrates often capture hearts, minds and funding dollars for both research and management efforts. Vertebrates in Ozark glades are no exception, with reptiles, birds, and game species garnering most attention among fauna. Whereas some studies have reported on glade arthropods as well, typically the reports have been peripheral components of vegetation or vertebrate-focused work.

Much as with vegetation studies, animal species of interest to glade-o-philes are those not typically associated with surrounding forests. In contrast to floral studies, prairies are mentioned less often for comparison of fauna than are deserts of the Southwest. The comparison with desert fauna comes from the extreme insolation and surface temperatures of glades that can be reached and sustained by the shallow rock in these bastions of severe periodic xericity. Many species of Ozark glade fauna reach northern extremes of their ranges here, are thought to be relics of the climatic history of the region (Skvarla, 2015), and may be present here in the Ozarks only because of the existence of glades. Climatic shifts that bring more moisture to the region, along with a decrease in frequency of fire, have set the stage for canopy cover and leaf litter to rob glade soils of the ability to reach extreme temperatures. In the absence of fire, species adapted to hot and dry conditions become isolated in glades that are no longer connected because of ever-encroaching forests. Fragmentation of the glade habitat by forest encroachment can lead to genetic isolation and the associated impacts on populations.

One of the most charismatic faunal members of the Ozark glade community is *Crotophytus collaris collaris* (henceforth *C. collaris*), the Eastern collared lizard. They are the "largest vertebrate predator confined to the glades" within the Interior Highlands, reaching lengths that exceed 40 cm (Ostman, 2007). These colorful and often-flamboyant entomophagous predators can cause hearts to race as they—often nearly underfoot—burst into sprint from complete crypsis among brightly colored, lichen-covered rock outcroppings. Most residents of the Ozarks today are unfamiliar with the existence of *C. collaris* altogether, but older generations of rural Ozarkians might recall memories of watching, catching, or even holding races with these lizards known colloquially as "Mountain Boomers" that can run quickly, solely on their hind legs. This species is a glade obligate within the Ozarks (Templeton et al., 2001), has been noted to be in decline (Trauth, 1989; Brisson et al., 2003; Brewster et al., 2018), and has been reintroduced to some glades where extirpated (Neuwald, 2008; Sites, 2013; Dr. Casey Brewster, personal communication, June 12, 2018). This is the only animal known to hold this distinction.

Other reptiles and amphibians that have been associated with Ozark glades include the Western pygmy rattlesnake (*Sistrurus miliarius streckeri* (L.) Gloyd) (Bell, 2007), flat-headed snake (*Tantilla gracilis* Baird & Girard) (Nelson, 2012), red milk snake (*Lampropeltis triangulum syspila* Cope) (Bell, 2007), variable groundsnake (*Sonora semiannulata* Baird and Girard) (Kimmel and Probasco, 1980), Eastern coachwhip (*Masticophis flagellum flagellum* (Shaw)) (personal observation), six-lined racerunner (*Aspidoscelis sexlineatus* (L.)) (Kimmel and Probasco, 1980; Bell, 2007) and the terrestrial microhylid frog known as the Eastern narrow-mouthed toad (*Gastrophryne carolinensis* Holbrook) (Bell, 2007). The Eastern fence lizard

(*Sceloporous undulatus* (Bosc & Daudin)), has also received attention in glades because, whereas it inhabits many Ozark habitat types, it is considered a top entomophagous predator in glades where *C. collaris* is absent (Van Zandt et al., 2005).

Ostman et al. (2007) noted that "birds are rarely seen foraging in these glades, most likely because forest birds are not adept at foraging in these grasslands, and the isolated patches are too small to maintain viable populations of open grassland bird species." However, that same year Bell (2007) published on songbirds utilizing glades, and suggested there to be "true glade avian species" such as the prairie warbler (*Setophaga discolor* (Vielliot)), Bachman's sparrow (*Peucaea aestivalis* (Lichtenstein)), painted bunting (*Passerina ciris* (L.)), field sparrow (*Spizella pusilla* (Wilson)), brown thrasher (*Toxostoma rufum* (L.)) and grasshopper sparrow (*Ammodramus savannartum* (Gmelin)). It should be noted, however, that these species can be found in a number of habitats, such as woodlands or forest understory, and thus should not be considered restricted to glades. Probasco (1978) also indicated Ozark glades to be favorable habitat for populations of *P. aestivalis,* as well as the greater roadrunner (*Geococcyx californianus* Lesson).

Several gamebird species can also benefit from presence of glades. Northern bobwhite quail (*Colinus virginianus* (L.)) are associated with open grassland and shrub habitats such as glades, and have been of interest for management agencies both public and private due to socioeconomic interest in the species as a game bird once widely hunted for sport and sustenance (Crosby et al., 2013). Due largely to habitat degradation this species has "declined an average of 3% per year since 1966" (Palmer et al., 2011). Programs such as the Arkansas Game and Fish Commission campaign to "Save the Quail," in conjunction with the National Bobwhite Quail Initiative, target habitat improvement of open grasslands via prescribed burning and cedar removal. During fieldwork for this study, *C. virginianus* were present at all glade sites. This was made evident not only by vigorous calling throughout the warmer months, but occasional flush from cover as well. On one memorable afternoon in 2018, at one of the sites involved in this study, another popular gamebird, the mourning dove (*Zenaida macroura* (L.)), was present as a single flock of more than 200 individuals, the flock being many times larger than any I had personally ever witnessed. Wild turkeys (*Melegris gallopavo* L.) are also known to make use of glades (Bell, 2007).

The Texas mouse (*Peromyscus attwateri* J. A. Allen) has a range that spans from Texas into the Interior Highlands and is the sole species of mammal found to be mentioned as being restricted to glades, at least within northern extremes of its range in Missouri (Bell, 2007). Kimmel and Probasco (1980) reported that the deer mouse (*Peromyscus maniculatus* (Wagner)) and the harvest mouse (*Reithrodontomys spp*.) "occurred almost exclusively in open cover conditions" along forest/glade transects of their study.

Arthropods in Glades

There is a staggering diversity of described arthropods and perhaps an equally impressive potential for species discovery. Far less has been published about arthropods in relation to glades than has been plants or vertebrates, and the existence of undiscovered species within these persistent, yet fragmented, relictual habitats is undoubtable. Allen (1990) noted 68 endemic species of insect alone within the Interior Highlands, and the same factors that have led to diversity and endemism among plants within glades, have most certainly acted upon arthropods as well.

The majority of arthropod studies within glades come not from arthropod taxonomists and systematists, but rather from those studying prey for dominant glade predators such as *C.*

collaris and *S. undulatus,* as well as those looking at herbivorous insects, primarily orthopterans and, to a far lesser extent, Lepidoptera and Hemiptera (suborder Sternorrhyncha).

Very little has been reported about arthropods present within glades in general, let alone those of the Ozarks. In 1937, Adelphia May Meyer reported on invertebrates collected from a cedar glade near Nashville, Tennessee. This appears to be the most taxonomically broad-ranging report of glade arthropods known. In that work, she recorded diversity among samples collected over a calendar year for Hymenoptera, Isoptera, Diptera, Coleoptera, Lepidoptera, Hemiptera, Orthoptera, Collembola, Araneae, Opiliones, Acari, Chilopoda and Myriapoda. She differentiated the glade categories of open grass, cedar, or shrub vegetation, and then recorded species collected on the soil surface, up to 2 inches above it, and from 2-6 inches above. Noteworthy findings from her work include open and shrub habitats outperforming cedar for epigeal diversity, and a decrease in catch during dry summer months. She also composed the only work that could be found containing broad-ranging species-level inventory of arthropods in glades. Even though these glades were in Tennessee, descriptions indicate them to be of similar composition to that of the Ozarks.

Several arthropods are mentioned by Bell (2007) as “having a significant relationship with glades,” including the Ozarks’ sole species of tarantula (Theraphosidae), the Arkansas brown tarantula, *A. hentzi* (which has previously been published under the common name of the Texas brown (Setton et al., 2019), Oklahoma brown (Wagler, 2015), or even Louisiana brown tarantula (McCarthy et al., 2020)), as well as the striped bark scorpion (*C. vittatus*), and the immodest grasshopper (*Melanoplus impudicus* Scudder). Whereas these species may not be entirely restricted to glades within the Ozarks, they are associated with xeric habitats and well drained soils (Gurney, 1941; Baerg, 1958; Jones et al., 2014).

As poikilothermic organisms, arthropods develop in relation to temperature as well as time, a concept packaged into the units of arthropod development known as accumulated degree days (ADD). Northern reaches of range for *A. hentzi* and *C. vittatus* into the Ozarks may only be facilitated by habitats that achieve high soil temperatures that allow these species to reach developmental thresholds associated with ADD. The intense insolation periods and shallow bedrock found in glades, lead to hot and dry soil conditions during summer droughts, and a more rapid accrual of ADD than can be found in soils of other Ozark habitats or later stages of forest succession allowing these species to complete reproductive cycles before the chill of winter sets in. It also seems that xeric soils of glades are more desirable for habitation by *A. hentzi* than are cooler, damper, soils of forests, because these spiders are prone to fungal infections (personal observation). Even *A. hentzi* established within well-drained glade soils have been observed to fall victim to fungi during wet years. Of course, there are a host of other species that, along with this fungus, utilize glade animals as a resource, and therefore become glade associates as well, such as the acrocerid fly (*Lasia purpurata* Bequaert), or tarantula hawk wasps (*Pepsis sp*.), both of which are obligate parasitoids of *A. hentzi*.

Ostman et al. (2007) reported on sweep-net captures from 12 rhyolite glades in the St. Francois Mountains of Missouri. Orthoptera, Hemiptera and predatory arthropods comprised the bulk of samples, and “total foliage arthropod species richness was positively correlated with glade area.” They also found that where *C. collaris* was present, diversity of the prey groups of Orthoptera and predatory arthropods decreased, but that this was offset by an increase in diversity of Sternorrhyncha. That offset was hypothesized to be due to reduced predation on species of Sternorrhyncha that are too small to be viewed as prey by the lizards. Van Zandt et al. (2005) positively correlated herbivory of glade endemic dicots and grasshopper (Orthoptera:

Acrididae) abundance, to increasing distance from structures used as cover by *S. undulatus* within glades, but they made no mention of grasshopper species present.

Bergmann and Chaplin (1992) studied the diversity of Acrididae in relation to glade size in the Caney Mountain Wildlife Area of south-central Missouri, and found positive correlations between glade size, overall diversity, and abundances for certain species, stating that "larger dolomitic glades support a fairly characteristic assemblage of grasshopper species… commonly found on tallgrass and mixedgrass prairies." Hill (2007) found many of the same species mentioned by Bergmann and Chaplin within a dolomite glade in Alabama. This relation to prairie species assemblages is not surprising, seeing that vegetational complexes are similar to prairies as well.

The lichen grasshopper (*Trimerotropis saxatilis* McNeill) has received attention, as it is reported to be limited to rock outcrops and glades in the Ozarks and similar habitats in the Southern Appalachians, but its closest kin are more-western ranging species in the genus. This species has been the focus of work on population energetics (Duke and Crosley, 1975) and population genetics (Gerber and Templeton, 1996).

As an accompaniment to floral diversity work that recorded differences between limestone glade flora and dolomite glade flora, Booth (2020) used photos to document lepidopterans encountered. Interestingly, only one of the 18 species photographed was present in glades of both rock types, the silvery checkerspot (*Chlosyne nycteis* (Doubleday)). However, that work seems to have attempted to document only charismatic species from the order, as all species listed in the study are colorful and easily identified; small, cryptic, or difficult-to-identify species were seemingly overlooked or omitted. It would also appear that only diurnal species were active during the documentation period; again, a sampling bias likely limited the taxa collected.

Issues Faced

The Anthropocene has seen great human influence on the health and sustainability of ecosystems. Development and management of land for personal, economical, or agricultural reasons drives habitat invasion, modification and fragmentation, which are all tangible and apparent threats common to habitats globally. Bulldozers invading wilderness, slash-and-burn land clearing, and construction of housing developments, shopping centers, and interstate highways, are all blunt examples of activities directly detrimental to habitats. Whereas these examples are pertinent threats to Ozark glades today, the existence of these habitats is being challenged to a greater extent by far more subtle side effects of human activity. Glades of the Ozarks, and elsewhere in the rocky regions of the Midwest and Midsouth, have adapted to the occurrence of fire over the past several thousand years. These fires have all but ceased to exist in the past century, and absence of fire allows for woody species to convert glades to later stages of succession and, eventually, climax temperate forest communities. Native and invasive woody species, such as *J. virginiana*, are pioneer players in the succession of glades. Other non-native, introduced species can alter glade community composition and have also become a threat to these habitats, even when woody species are kept at bay.

Fire & Forest Succession

Two things are seemingly universally agreed upon in the literature: 1) that fire is essential to the maintenance of Ozark glades, and 2) fire historically was far more frequent in the Interior Highlands than has occurred in the past century (Kimmel and Probasco, 1980; Baskin and Baskin, 2000; Templeton et al., 2001; Ware, 2002; Van Zandt et al., 2005; Jenkins and Jenkins, 2006; Ostman et al., 2007; Eltorai, 2011; Nelson, 2012; Booth, 2020). As stated by Steyermark (1940), “natural succession, under the present mesophytic climatic climax, is towards a forest which sooner or later obliterates the glade.” Fire and erosion are the abiotic forces that have historically allowed for Ozark glades to persist in the face of this “obliteration.” Lowell and Astroth (1989) hypothesized that Ozark glades only exist because fire or human management halt encroachment and conversion to later-stage communities. This hypothesis seems to hold true in all but the most intensely erosive and insolated glades that occur on steep, south- or west-facing, rocky slopes, where soils have little chance to develop— an edaphic subclimax that resists woody species even without fire (Steyermark, 1940).

For many, prescribed burns are seen as a way to bring natural cycles back to an environment that were lost due to suppression of fire; indeed, this is accurate in many ecoregions. However, persistence of the vast glades described in the Ozarks by Hall (1955) is likely to have required fires more frequently than nature provided. The influence that native peoples had on forest succession with the use of fire was effectively the first habitat management in the Ozarks. A review of tree fire-scar studies was compiled and mapped by Guyette et al. (2006) showing that average intervals of 8-15 years between fire scars were common over much of the Interior Highlands prior to European settlement, and Guyette and McGinnes (1982) found fire occurrence in certain areas as often as every 3.2 years. Nelson (2012) presented convincing evidence that use of fire by native peoples was instrumental in the persistence of glades, savannas, and woodlands of the Interior Highlands for thousands of years leading up to European settlement. He also criticized the later, and highly influential, work of Steyermark (1959), who suggested climax forests were the ideal endgame for the Ozark landscape, and that burning was counter to management goals. Whether Dr. Steyermark’s work was influential in management decisions or not, the U.S. Forest Service has suppressed fire in the Ozarks for the better part of the 60 years since publication of his views (Kimmel and Probasco, 1980; Guyette and McGinnes, 1982; Guyette et al., 2006; Eltorai, 2011; Nelson, 2012; Booth, 2020). However, widespread fire suppression began in the region in the 1930s (Soucy et al., 2005). Indeed, use of prescribed burns may actually be restoring an unnatural abiotic influence to the Ozarks, but it is also an influence that some Ozark glade-associated species have adapted to over the past several thousand years, and require for existence.

Cedars

In the absence of periodic fire, several woody floral species have been implicated in conversion of glade to forest. These include several species of oaks (*Quercus spp*.), fragrant sumac (*R. aromatica*), winged elm (*U. alata*), and even the now exceedingly rare Ozark chinquapin (*Castanea ozarkensis*) (Appendix A), but none of these species have received the notoriety in glade conversion as have cedars.

The existence of “cedar glade” as a described habitat shows how commonly these trees are associated with glades and glade succession. Keeland (1978) proposed a reduction from Steyermark’s six forest successional stages to four: grass and cedar, cedar, cedar hardwood and finally, hardwood. No other tree has been mentioned more than cedars in relation to Ozark glade flora, forest succession, prescribed burning, or glade management.

Juniperus virginiana, to a lesser extent Ashe’s juniper (*J. ashei* Buchholz), and hybrids of the two, are capable of withstanding drought conditions and taking hold in the thin rocky soils of glades where most woody species cannot, and therefore are primary catalysts for succession advancement (Templeton et al., 2001). Cedars can form extremely dense stands that choke out other glade flora (Van Zandt et al., 2005). In prairies, which again are akin to glades, the invasion of *J. virginiana* has been shown to reduce herbaceous species diversity and overall productivity (Briggs et al., 2002), reduce forage production (Engle and Kulbeth, 1992), and decrease diversity of grassland birds (Chapman et al., 2004) and small mammals (Horncastle et al., 2005). Epigeal arthropods have also been reported to have less diversity in samples taken from cedar glade habitat when compared to "open glade" habitat (Meyer, 1937). Abiotic alterations known to occur with cedars other than increasing canopy cover include decreasing soil pH (Hall, 1955), intercepting precipitation, and having higher transpiration rates than prairie flora (Starks et al., 2014). Cedars can overtake areas rapidly, particularly in the absence of fire, and prevent insolation required by herbaceous glade natives (Eltorai, 2011). Engel et al. (1994) reported rangeland invasion by cedars in Oklahoma occurred at a rate of approximately 113,000 hectares per year from 1985-1994. This same report also noted that, whereas these cedars do provide food for some generalist vertebrates, those habitat specialists are negatively impacted by presence of cedars, and that "biosimplification is a characteristic of a closed stand of juniper and results in ecosystem deterioration." Largely because of mounting evidence correlating cedar invasion to negative impacts on habitats, cedar removal has become a primary goal of many grassland managers.

Introduced Species

Beyond successional woes, two introduced species—one plant, one animal—have become destructively invasive within Ozark glades, as witnessed firsthand: the herbaceous vegetative invader, sericea (*Lespedeza cuneata* (Dum. Cours.) G. Don), and the feral hog (*Sus scrofa* L.), scourge of many terrestrial habitats in North America, and particularly so in the Southeast U.S.

Purposefully introduced from Asia in the latter half of the 19th century, *L. cuneata* has been planted for forage and to prevent erosion. Since introduction, it has also invaded many open habitats where it can form “dense monocultures” (Allred et al., 2010). A large number of small seeds are produced by *L. cuneata* that can persist in the seed bank for as long as 30 years, from which, “over time, scattered seedlings emerge and insidiously expand into multi-stemmed patches that coalesce as the dominant component” in some grasslands (Wang et al. 2008). Coykendall and Houseman (2014) reported that soils where *L. cuneata* had previously grown were more conducive to germination and growth of future generations of the species, and they suggested that this could be due to *L. cuneata* altering soil composition.

In the sandstone glades chosen for the present study, *L. cuneata* has been managed via burning but is also one of the few glade invaders targeted for management with herbicide sprays (Idun Guenther, Wildlife Biologist, Ozark-St. Francis National Forest, Sylamore Ranger District, personal communication, January, 2019). The introduction of *L. cuneata* can easily take place in areas of disturbance, by improperly cleaned equipment or vehicles spreading the fine seeds. In discussions with local residents living near the glades of the present study, one couple noted that *L. cuneata*, known to them as bush clover, did not exist in the area in noticeable quantities until the local road was paved. Their explanation was that it had been a hitchhiker aboard road construction equipment. The earliest report found of *L. cuneata* presence in Ozark glades was published in 1999 (Flaspohler), and appearance of this species after glade disturbance supports the claim by Ware (2002) that disturbed areas do not naturally become glades, even if abiotic factors conducive to glades are present.

Glades rooted by feral hogs appear as though a tiller has gone through the shallow soils disturbing soils crusts, moss layers, and overturning stones in search of anything palatable (personal observation). In glades where cover stones are not large enough to prevent overturn by hogs, there is little refuge for species to escape from consumption by these insatiable beasts. Whereas not much mention of disturbance by hogs occurs in relation to Ozark glades, Booth (2020) suggested wild hog activity to have been a possible confounding variable in vegetational analyses of limestone and dolomite glades.

Benefits of Glades

Glades in the Ozarks of today remain as storytellers. They are a product of glacial and oceanic retreat, desert and prairie expansion, mesophytic shifts and forest succession—and human management or lack thereof. Ozark glades are not only intrinsically beautiful windows in the forest canopy, but also windows to a not-so-distant past when these habitats were much larger players in the overall regional forest mosaic.

Glade endemic species, and isolated populations of iconic species that require glade habitats, are not only valuable as a reflection of the past, but also as subjects of study. Presumably, the potential to connect Ozark glade habitats, on a large enough scale to promote future widespread gene flow for obligate animals such as *C. collaris,* or glade-associated species such as *A. hentzi,* is extremely low. To regain large-scale re-connectivity of glade habitats would entail far too great an effort—one that would extend beyond cutting and burning woody vegetation, to include large-scale removal of accumulated soils. Thus, proper management of the glades that remain, and restoration of those in threat of loss to succession, will still face challenges brought about by habitat fragmentation. Continued monitoring and management of glade-associated species can reveal knowledge about proper conservation of fragmented grassland habitats, and about the isolated populations of flora and fauna they support.

Very little is known about the natural history of many Ozark glade inhabitants, utilization of glades by forest-dwelling species, or composition of glade invertebrate communities, including arthropods, and the potential for discovery of endemics among this group cannot be overlooked. If Ozark glades are allowed to continue to convert to forest, the restoration of these habitats to functional levels may be impossible, and unanswered questions about glade communities may remain so—indefinitely.

Mounting evidence suggests that not only does loss of this habitat to forest succession have a negative overall impact on forest mosaic diversity, but also that removal of woody vegetation, particularly cedars, can improve grassland diversity, productivity, and connectivity (Brisson et al., 2003). Along with species associated with glades, known and unknown, these habitats also see use by species not considered to have particular glade affinity. Glades offer pollinators and herbivores diversity in dietary options not found in other stages of forest succession in the Ozarks, and glades support wildlife of economic value as hunting game such as quail, turkey and deer, which is of paramount importance to many land managers.

Glade Management

Nelson (2012) stated that there are no examples of open habitat types in the Ozarks that have “recovered or succeeded to an equally species rich ecosystem” without management. Habitats such as glades, existing as transient successional stages, albeit often prolonged, are always under threat of loss to later stages. In all but the most severe instances of erosion, glades cannot exist without fire—so as long as fire is rationed by people, so too will be glade existence. Species adapted to, or marooned in, these rocky grasslands await the end of historically infrequent fire-drought and the associated removal of woody species—in essence just as an aquatic species awaits rain from a shrinking ephemeral pool.

Size Decreases, Interest Grows

Indeed, within glades there is a likelihood that "conservative species decrease or disappear" in absence of fire (Nelson, 2012), and that prescribed burning is often accompanied by restorative effects (Brisson et al. 2003). For the glade manager, restoration and maintenance revolve around the proper use of fire, monitoring and, when appropriate, reintroduction of associates, or removal of introduced and invasive species not suppressed by fire. Sound perspective on end goals for habitat maintenance comes from Omernik (1987) who states that:

> "it is unrealistic to expect an attainable quality of water and land resources at the level possible before major human settlement. What is realistically attainable is a quality possible given a set of economically, culturally, and politically acceptable protective measures that are compatible with regional patterns of natural and anthropogenic characteristics."

Whereas Native Americans, and some early settlers, may have understood the importance of fire in creating fertile habitat for hunting, interest in fire's importance and use as a rejuvenating abiotic force maintaining Ozark glades progressed slowly throughout most of the 20th century. As previously discussed, this could be due to the views of Steyermark, who supported the idea of fire suppression (Nelson, 2012). However, rarity increases value. In the last several decades, as glades have shrunk to fractions of their former extent, interest in them has seemingly gained ground (Van Zandt et al., 2005) with field research leading the charge, and federal and local agencies adopting prescribed burns as well as funding glade restoration proposals. Agencies that are known to have attempted Ozark glade restoration include the National Park Service (Jenkins and Jenkins, 2006), U. S. Forest Service (Bell, 2007; Idun Guenther, personal communication, January, 2019), Arkansas Game and Fish Commission, (Brewster, 2019; personal observation), Missouri Department of Conservation (Comer et al., 2011), Arkansas Natural Heritage Commission, The Nature Conservancy (Idun Guenther, personal communication), Missouri Botanical Garden Shaw Nature Center, and Washington University (Eltorai, 2011). It is also worthy of note that the Tyson Research Center at Washington University in St. Louis, Missouri, has experimental glades within the confines of the 800-hectare complex for study of restoration (Lutz, 2012). Washington University is also tied to several of the works referenced herein (Templeton et al., 2001; Brisson et al., 2003; Van Zandt et al., 2005; Ostman et al., 2007; Neuwald, 2008). Ozark glades were the focus of the virtual workshop held by the Oak Woodlands Fire Consortium in October, 2020, titled, *Quail, Glades and Fire,* cosponsored by several state, federal, non-profit, and private management agencies.

Restoration and Maintenance

Longcore (2003) stated that "modern conservation planning increasingly relies on the use of ecological restoration techniques to improve conditions for natural communities." In 2011, Eltorai stated there was "an urgent need for scientific advancement in the fields of conservation biology and restoration ecology," and that "recovery of the natural environment is the only hope for preserving biodiversity." Restoration and recovery in the context implied by these authors is a return to a previous state, which hypothetically, is an attainable endpoint. That endpoint in grasslands can be a complete restoration, where historic accounts of species assemblages are targeted for recreation; a functional restoration, where non-native species assemblages perform actions indicative of the habitat type in question; or, finally, an experiential restoration, which in grasslands means that "one or two grasses and a few forbs are used to represent the entire community" (Sluis, 2002). Experiential restorations attempt to recreate the "feel" of a habitat more than its function.

Once the chosen endpoint for glade restoration is achieved, management effort from that point on can be considered maintenance. Maintenance activities often require less cost or involvement than do restorative actions. The transition from restoration to maintenance is beneficial for both the habitat that has achieved "restored" status, and land managers responsible for them. Upon completion of restoration, the habitat targeted should be reflective of, or minimally closer to, the habitat desired, in composition and/or function. It should also be beneficial for the manager to make transition from restoration to maintenance, as maintenance requires less management effort. For example, if one were to restore a climax community like the regionally dominant oak-hickory forest, it could entail species or population removal, enhancement, or reintroduction, which are often time- and cost-sensitive endeavors. Once restoration goals have been satisfactorily achieved in a habitat, maintenance would typically entail protections from disturbance attributable directly to humans, like prevention of logging or introduced species. These protections can come either physically, such as fencing, gating or patrolling, or as regulatory actions, such as laws and limitations. As in this example of oak-hickory climax community, transition of glades from restoration to maintenance revolves around suppressing anthropogenic disturbance. However, in contrast to the oak-hickory example, the disturbance caused by humans in glades is indirect, via suppression of fire—the primary factor holding destruction and fragmentation of glades at bay.

The sub-climactic nature of glades in forest succession means they are continually in transition. When fire is suppressed, the transition is most often away from the most desirable state, making glades moving targets for the managerial marksman's aim and, as any non-static target, more of a challenge to hit. The continual transition also means lines between restorative and maintenance actions become blurred, each utilizing similar practices differentiated solely by the successionary status of the glade in question.

Restoration efforts in glades come in the form of prescribed burns, herbicide use, physical removal of mature trees (Idun Guenther, personal communication, January 29, 2019) and reintroduction of species (Trauth, 1989; Van Zandt et al., 2005; Neuwald, 2008; Eltorai, 2011; Brewster, personal communication, June 12, 2018). Glades seen as later-sere, containing woody species that threaten alteration of thin, well-insolated soils, often require the greatest restoration effort. Mature trees inhabiting glades are a product of a historically long interval between fire occurrences. Because the trees have been allowed to mature, the sparse ground cover often associated with Ozark glades, made even less so at this stage by canopy cover, does not contain enough fuel load to remove the trees or their glade-killing canopy. Restoration efforts effectively reset the sere of Ozark glades back to an earlier one. However, once reset, glades will invariably return to needing restorative effort if maintenance with periodic prescribed burns is not continued. Therefore, let it be said that anyone interested in successful management of glades would be unwise to undertake restoration without a plan for continued maintenance.

Inventory and Monitoring

Properly managing a habitat requires understanding both biotic and abiotic factors influencing communities. Knowledge of these factors can be gained either through descriptions in literature, or by firsthand experience and study. One could say those published descriptions of glade communities both early on (Meyer, 1937; Steyermark, 1940; Hall, 1955) and more recently (Baskin and Baskin, 2000; Ware, 2002; Nelson, 2012; Booth, 2020), can be seen as akin to stenographers dictating the living history of what it means to be a glade, and ultimately how these habitats have been and will be judged and managed in the future. Rohr et al. (2007) stressed the essential nature of “a baseline characterization of biodiversity” for monitoring habitat changes. In the context of management, use of these data to assess outcomes and guide implementation is critical.

Habitat monitoring is key for ecological maintenance (Rohr et al., 2007), particularly in glades, because these habitats can go from functional to uninhabitable by obligate species in as little as a few seasons (Brewster et al., 2018). Some ecosystems rely on periodic flood or inundation by water to maintain community structure, and inhabiting species are often highly adapted to, or entirely reliant on, this abiotic cycle. Ozark glade communities face much the same cycle of constraint, only with a different abiotic factor in play.

Reference to glade monitoring primarily manifests as accounts of glade area having been, or in the process of being, lost to succession (Steyermark, 1940; Hall, 1955; Soucy et al., 2005), in glades where management has not been implemented. Aerial photographs that confirm these accounts were taken of Hercules Glades Wilderness Area in southern Missouri between 1938 and 1986 (Kimmel and Probasco, 1980; Lowell and Astroth, 1989).

Monitoring also has occurred during and after management. Jenkins and Jenkins (2006) stressed that, particularly when managing glades with fire, a portion of the area should be maintained as a reference for monitoring changes attributable to its implementation. Eltorai (2011) stated:

> “It would be interesting to compile a comprehensive list of all dolomite glade wildflower species. The fact that such a list does not exist, serves as an indication that basic census surveys are fundamental and practical for further investigations. If such as list of expected Ozark dolomite glade wildflower species existed, future research could include a comparison of the wildflowers present in particular nature reserves to the expected list. This could serve as another means of evaluating glade health and restoration effort efficacy.”

Plant species reported from seven previous studies of dolomite, limestone, and sandstone glades have been compiled in Appendix A.

Applied Techniques

Fire is the most notable tool used in management for both restoration and maintenance of Ozark glades (Templeton et al., 2001; Ostman et al., 2007; Eltorai, 2011). Historically, fire occurred as frequently as every 3-4 years (Guyette and McGinnes, 1982), and it effectively prevented succession of glades to later stages. Succession is halted due to the intolerance to fire of woody species catalytic to the transformation of glade, such as immature *J. virginiana,* which are susceptible to fire, particularly if the growing tips can be burned. Simply re-enacting historic fire events seems to have the greatest restorative power among management techniques practiced. Jenkins and Jenkins (2006) showed a post-burn increase in floral diversity in communities sampled from limestone/dolomite/sandstone glade complexes within Buffalo River National Park, Arkansas. Comer et al. (2011) found similar effects on diversity after prescribed burns of rhyolite glades in Iron County, Missouri. The diversity of substrates and locations suggests fire could be effective in improving other glade situations as well. After all, fire is the primary abiotic force, limiting the primary threat to glade persistence. However, Jenkins and Jenkins (2006) pointed out that fire does not dictate alone, with “poorly understood factors including precipitation cycles, light regimes, topography, edaphic characteristics, resource heterogeneity, and disturbance” playing roles. Comer et al. (2011) also pointed out that glade communities are not recreated solely by fire.

Other forms of invasive species removal have also been practiced periodically. For cedars and other woody species, removal is performed mechanically, such as by felling and transporting offsite (Eltorai, 2011), felling and leaving them, or by chipping/mulching woody invaders onsite (personal observation; Guenther personal comm.). Herbicides can be used effectively for mitigation of weedy and introduced herbaceous species, such as *L. cunea* (Guenther personal comm.), that often can establish after disturbances, including mechanical removal of cedars.

Reintroduction of flora via transplant (Van Zandt et al., 2005) and seeding (Eltorai, 2011), have occurred in Ozark glades, and reintroductions of *C. collaris* (Neuwald, 2008; Sites, 2013; Brewster, personal communication, June 12, 2018) are also a restorative tactic.

Management Needs

Ozark glades will require management for the foreseeable future so long as fire is infrequent. Unlike other regional climax communities, the threat of glade succession to later stages looms as the primary opponent to their persistence, therefore, resisting succession is the primary concern for those wishing to maintain their existence. Sadly, forest succession cannot be mitigated with fencing, legislation, or law enforcement, removing some of the management tools that are often deployed for conservation and management of climax communities, whether restorative, maintaining, or somewhere in between.

Habitat conservation and management benefit from improved planning and implementation, and the feedback loop responsible for these improvements comes via monitoring and assessment. Bell (2007) stated that success of habitat restoration via fire has been poorly documented in the Ozarks. How can a glade be considered maintained, improved, or restored, if no assessment protocol is in place? Arguably, monitoring and assessment are the applied management techniques in greatest need of improvement in relation to Ozark glades. Development of a robust array of techniques helps overcome obstacles faced with an often-complex task of ecological assessment. Streamlined and more effective management approaches come from the ability to accurately: 1) implement appropriate restoration or management approaches, 2) determine effective glade management practices, 3) identify glade sites with high restoration potential, 4) triage glade sites with high risk of forest conversion, or 5) detect disturbances, species introductions, or population declines within glades. Evidence of successful management from assessments could help keep this shrinking habitat from also facing shrinking interest from decisionmakers keen to see a positive return on management investments.

Plant communities, and several species of charismatic and/or economically important bird, reptile, mammal and arthropod species, have now been discussed as being endemic to, occurring within, or benefitting from, Ozark glade habitat. Comprehensive species accounts exist for glade-inhabiting flora, in all seres of glade succession, making the plants easy to work with when assessing glade management. In fact, a protocol for use of wildflower sampling and diversity analysis has been suggested by Eltorai (2011) for use in assessment of glade health.

Inventories of animals associated with glades, beyond birds and reptiles, are severely lacking. Using presence and abundance of glade-endemic predators, such as populations of *C. collaris,* or nesting pairs of omnivorous and economically important *Colinus virginianus,* could also be an easy place to turn for data collection aimed to catch the attention of funding agencies and committees. However, between the vegetation and the vertebrates lies a vast sea of arthropods that could become the easiest and most accurate group to use in comparison and assessment of glades. Arthropods could also hold the key to tying together plant and vertebrate knowledge, especially for vertebrates that are entomophagous to some degree. Simply looking at plants or vertebrates in systems unnecessarily compartmentalizes interactions and pixelates our view of vital functions. It is important "to consider the entire community" holistically for effective management (Van Zandt et al., 2005). This holistic notion also extends to glade restoration efforts that seldom consider arthropods, but commonly work with plants (Van Zandt et al., 2005; Eltorai, 2011) and reptiles (Neuwald, 2008; Sites, 2013; Brewster, personal communication, June 12, 2018).

Creating an inventory is a required and vital step that must be completed before management efforts can properly be assessed. The absence of information on arthropods in Ozark glades, and glades in general, is stark—there is the lone diversity study occurring in a Tennessee cedar glade more than 80 years ago (Meyer, 1937). A lack of baseline information about what arthropods inhabit or utilize glades makes comparison impossible and, thus, not a current option for glade assessment.

Arthropods and Habitat Assessment

Comparing species assemblages within habitats to historical accounts, or to current examples of "pristine" or archetypal habitats, is useful for detecting environmental change (Kremen et al., 1993), and can generate valuable information for glade management (Eltorai, 2011). However, complete censusing of species for such comparisons is neither desirable, nor possible. Therefore, being able to identify "valid surrogates for biodiversity," and finding effective ways to monitor them, is desirable for reducing sampling and identification effort (Rohr et al., 2007; Eltorai, 2011). The first suggestions of indicator species being used as representatives of habitat diversity have been said to come from C. H. Merriam, in 1898 (Landres et al., 1988), as well as H. Hall and J. Grinnell, in 1919 (Landres et al., 1988; Niemi et al., 1997). The use of indicators in ecological analysis has been debated ever since these early publications; a thorough review of the topic can be found in the work of Landres et al. (1988). Indeed, indicator species may not accurately represent other "constituent" species in a community and, as stated by Landres et al. (1988), "may bear no direct or simple cause and effect relationship to the factor or factors of interest." However, confounding factors aside, sampling of ecological indicator species can generate data indicative of relative changes in a habitat, or differences between habitats, that can be used to hone management protocols (Holling, 1978).

Arthropoda is the most diverse phylum, containing between one and two million unique taxa. The order Coleoptera contains more described species than all plants combined. This breadth of diversity means that arthropods occupy more niches than any other phylum, and therefore, offer more options for habitat assessment and comparison as well (Kremen et al., 1993; Longcore, 2003). Whereas traditional terrestrial indicator species are large vertebrates (Landres et al., 1988), in aquatic environments the EPT test (a diversity analysis of the insect orders Ephemeroptera, Plecoptera, and Trichoptera) has been extensively used to assess water quality (Lenat and Crawford, 1994; Hodkinson and Jackson, 2005; Hamid and Rawi, 2017). However, despite the popularity in aquatic study, use of arthropod diversity sampling as assessment of terrestrial habitats is seemingly underrepresented in comparison to that of plants, birds and other vertebrates. Kremen et al. (1993) synthesized a review of studies that monitored terrestrial arthropod assemblages for conservation planning and reported a paucity of studies utilizing this seemingly ideal group; this seems particularly true in Ozark glades.

Arthropods, particularly those with low dispersal ability, can be very sensitive to localized environmental changes, such as forest conversion and habitat fragmentation (Kremen et al., 1993; Maleque et al., 2009), which are the main issues threatening glades. Longcore (2003) condensed the work of Kremen et al. (1993) by stating benefits of using arthropods as indicators to be “large population sizes, reproductive potential, and short generation times [allowing] the collection of statistically significant sample sizes using relatively passive methods with little potential for depleting populations.” Kremen et al. (1993) also stated the largest drawback to using arthropod monitoring for habitat change is relating variances in populations sampled to variables of interest. However, they also stated this can be somewhat mitigated with relative comparisons to control plots in quality habitats, or by long-term monitoring.

Pitfalls of Assessment

A popular way to sample terrestrial arthropod diversity is the pitfall trap, which, in essence, could be any implement made into a “pit” (collection container) that arthropods enter and cannot escape. Pitfall trapping has been said to be the most common way to collect epigeal (soil surface) arthropods (Leather, 2005). Simplicity and versatility of trap design, low cost, ease of implementation, and capability for specimen preservation during continuous sampling, top the list of positive attributes of pitfall traps. Pitfall trapping is also one of a very limited number of methods available for sampling “highly active, mostly polyphagous, invertebrate predators” (Leather, 2005), such as the epigeal predatory arthropods targeted for the current study.

Ironically, there are many pitfalls to avoid when designing experiments with, and analyzing data from, pitfall traps. Nearly every way a pitfall trap design can vary in construction, such as size of catchment basin, entrance opening size, color, type of preservative, presence of a lid, etc., seems to impact composition of arthropod species collected by a pitfall trap (Leather, 2005; Skvarla, 2015). The selectivity of certain pitfall trap designs for certain taxa limits the ability to make inferences about sample sites, or habitats, from sample data, as an unknown number of taxa may have failed to enter the trap. Instead, analysis of sample diversity should be limited to the samples themselves, or comparisons between samples (Leather, 2005). Analysis of pitfall-sampled diversity can also be improved by removing species rarely captured to lessen error, as these species may have been rarely sampled because they avoided a trap design element, or because they are “tourists (unrepresentative of the site in question), or very sedentary species”

(Dennis et al., 1997). Estimation of species densities at a site, or within a habitat, based on pitfall trap data should also be cautioned, as these traps produce activity-abundance information, and isolation of abundance from activity as a confounding variable is nearly impossible (Thiele, 1977; Leather, 2005).

Despite the drawbacks and limitations of pitfall trapping, Leather (2005) stated that the use of pitfall trapping "is probably no more questionable than most other sampling techniques used for invertebrates." Sherley and Stinger (2016) even considered it "ideal for open habitats such as grasslands and arable land." Use of an appropriate number of pitfall traps to generate sample sizes large enough for target taxa to be statistically analyzable, while minimizing sampling effort, is optimal. Preliminary testing can help determine what is appropriate based on the trap design, target taxa, and habitat in question. However, deployment of 10-12 traps for each sampling location is common (Obrtel, 1971; Dennis et al., 1997; Leather, 2005; Sherley and Stringer, 2016). Transects and grids are often used for selection of trapping locations within a site; transects are particularly desirable for sampling environmental gradients, and grids are said to provide "good even coverage of the sampling area" as well as the ability to manipulate statistical independence of traps based on grid spacing (Leather, 2005). Duration of sampling periods for diversity analysis, at least upon initial sampling, should encompass the seasonal activity period of the target taxa in entirety (Baars, 1979; Den Boer, 1979; Dr. Neelendra Joshi, University of Arkansas, personal communication, March, 2018). In the case of epigeal predatory arthropods in the Ozarks, summer, or spring-to-fall sampling may meet this criterion, but activity periods for certain species also peak in the winter months (Jones unpublished), thus, in general, a full calendar year is thought to be the minimum for an initial sampling period. Sherley and Stringer (2016) state that pitfalls "may be used to obtain an approximate index of relative abundance if used in the same way, in the same location(s), at the same time of year and if set over a long period of time." These authors also suggest that pitfall traps are capable of generating data on epigeal arthropod species richness that is worthy of consideration in habitat assessment.

In areas where substrates or regulations do not allow for collection containers to be dug into the soil, ramps can be used to direct target arthropods up and into a collection container sitting atop the ground surface: this is a style that has been referred to as a ramp pitfall trap, or ramp trap (Bouchard et al., 2000; Skvarla, 2015). As with other pitfall trap styles, design elements are sure to impact the composition of species captured when using ramp traps. For instance, it has been reported that ramp traps have outperformed standard, dug-in, pitfall traps in capturing wandering spiders (Patrick and Hansen, 2013). Another advantage of ramp traps is a lessened "digging in effect" versus standard pitfalls; which is where species are attracted, or repelled, by the disturbed earth for several days at the trap site (Leather, 2005).

Taxa Selection

Beyond the three insect groups used in EPT testing of aquatic environments, other arthropod groups commonly studied for "bioindicator potential" include ants, lepidopterans, carabids, cerambycids, dung beetles, spiders, syrphid flies, and parasitic wasps, according to a review by Maleque et al. (2009). Specific arthropod taxa are often selected for analysis due to limitations in manpower and/or identification abilities.

> "Often expertise is not available for determining all of the vast number of organisms that can be caught throughout a pitfall trap season. Limiting both trap number and the number of taxa that need to be assessed may greatly reduce the expense and duration of a study making this method more practical for determining a wider range of questions relating to conservation and ecology" (Sherley and Stinger, 2016).

Taxa selection for the current study was based on consideration of several factors including the work of Longcore (2003), preliminary studies passively trapping in shallow glade soils, and the breadth of available works relating to targeted taxa in relation to the Ozarks or glades.

The exploration of Travis Longcore (2003) into using terrestrial arthropods as indicators for coastal sage scrub habitat restoration was highly influential in the current study. His comparisons of terrestrial arthropod community composition between "disturbed, undisturbed and restored coastal sage scrub" habitats suggested that when restored sites had reached a floral community similar enough to undisturbed sites to be considered a restoration success, terrestrial arthropod communities had not, even up to 15 years post restoration. Indeed, he found restoration sites to have lower terrestrial arthropod diversity than both undisturbed or disturbed sites, and pointed to exotic and invasive species in the restored sites as the cause. Along with this finding, several families of epigeal predators, as well as certain individual species, were found to have significant differences in abundance between undisturbed and disturbed sites. This supported the author's introductory statement that "a good indicator of a successful restoration should be rare, predatory arthropods." Disturbed and restored sites were reportedly lacking abundances of scorpions (*Parurocotonus sylvestrii* (Borelli)) and trap door spiders (*Aposticus sp*.). He also stated that two of the arthropod groups sampled, spiders (Araneae) and beetles (Coleoptera), "could have served as indicators for the whole."

Many of the commonly used arthropod sampling techniques such as light trapping, sweep-netting, and malaise trapping are ineffective at sampling epigeal arthropods. Collection of leaf litter or soil samples for processing via Berlese funnel is often a technique used to sample epigeal arthropods. However, one of the key characteristics of Ozark glades is a lack of these substrates. Actively searching for specimens can work for creation of species inventories, but is undesirable when attempting to compare sites; achieving equal sampling effort at each location can be difficult or impossible due to differences in habitat size, available cover, etc. Actively searching along randomized transects of equal length to equalize sampling effort at each location can be difficult due to terrain, time involvement, or other confounding variables, such as weather, time of day or person performing the sampling. Pitfall trapping not only circumvents issues with epigeal sampling faced by other techniques, it also allows for continuous sampling, which reduces the influence of variables associated with shorter periods of collection. Passive, continuous trapping was seen as a requirement for this study due to limitations of manpower and distance of travel required to reach study sites.

Preliminary field testing of traps capable of sampling epigeal arthropods in shallow glade soils produced trap catches of a wide range of species including non-epigeal insects such as grasshoppers (Orthoptera), butterflies (Lepidoptera), leafhoppers (Hemiptera), small bees (Hymneoptera), and many species of flies (Diptera) and beetles (Coleoptera). However, epigeal predators were captured in numbers great enough to be considered statistically viable for analysis and primarily belonged to the Araneae. Active hunters are those most effectively sampled by pitfall-style trapping (Leather, 2005; Sherley and Stringer, 2016).

From the preliminary trap collections, two groups were chosen as targets for diversity study—ground spiders from the family Gnaphosidae and predatory ground beetles from the family Carabidae. This decision was made not only because these taxa existed in appreciable numbers, but also because they are diverse and well-documented families in Arkansas (Heiss and Allen, 1986; Carlton and Robison, 1998; Hamilton, 2015), and recent work (Hamilton et al., 2018) also provided data on pitfall capture of gnaphosids and carabids from oak-hickory forests of the Ozarks for contrast with the current study.

Why Gnaphosids?

Although diversity of spiders (Araneae, ~45,000 species) pales in comparison to that of insects, they are still among the most ubiquitous and diverse groups of organisms known. A lesser diversity of spiders can be seen as an advantage over working with insects because samples collected from passive traps are often overwhelming due to the amount of diversity, and identification, which is necessary to interpret sample composition. Another benefit of working with spiders is they are all thought to be obligate predators (with the exception of a single known phytophagous species, *Bageera kiplingi* Peckham and Peckham, and certain groups supplementing their diet with nectar and/or pollen (Lundgren, 2009)) and they tend to have population fluctuations more influenced by prey abundance than weather conditions (Uetz, 1975; Hamilton, 2015).

Many spiders are not considered epigeal, and those that are epigeal can broadly be considered sedentary hunters (web-builders, trapdoor spiders, etc.) and more active (hunting or wandering spiders). For obvious reasons, pitfall trapping cannot capture inactive epigeal spiders. Even though species of spiders that are generally more sedentary can become active seasonally for periods, typically it is the active species that are targeted with this trapping method. Ramp traps, such as those deployed for the current study, have been shown to catch more actively wandering spiders than pitfalls with soil-level entrances (Patrick and Hansen, 2013).

Members of the family Gnaphosidae are known by the common name of ground spiders. There are 118 genera and over 2,100 species worldwide with 255 species reported from North America (Bradley, 2013). They are active wandering spiders that are not known to disperse via ballooning (Dr. Kefyn Catley, Western Carolina University, personal communication, July 24, 2019), and are primarily nocturnal or crepuscular with the exception of the ant-mimicking genus, *Micaria* (Bradley, 2013).

Preliminary Ozark glade trapping trials produced more wandering spiders from the families Lycosidae, Salticidae, and Gnaphosidae than any other families (personal observation). Gnaphosids were selected over the other families that were abundant in samples for several reasons: 1) gnaphosids have distinct spinnerets and eye arrangements, making them relatively easy to identify to family when compared to other wandering spiders such as lycosids (Ubick et al., 2005; Hamilton, 2015), 2) a lack of ballooning may decrease gnaphosid dispersal ability to fragmented, isolated, or restored glade sites when compared to ballooning taxa, 3) the common name of ground spider alludes to the belief that these spiders are primarily epigeal, although it should be noted that life histories of gnaphosids are poorly known (Kamura, 1993), and 4) gnaphosids are believed to be appropriately sampled with pitfalls (Uetz and Unzicker, 1976; Gillespie et al., 2019) unlike salticids, which would require sweep-netting, pan trapping, or other techniques to appropriately sample family-wide diversity, as some species are seldom found on the ground.

Why Carabids?

Coleoptera is the largest order of known life, containing more than 150 families. The family Carabidae, commonly referred to as ground beetles or predaceous ground beetles, contains at least 40,000 described species worldwide (Lovei and Sunderland, 1996), which is roughly equivalent to the total number of all spider species. Approximately 2,000 species of carabids are known to inhabit North America (Lovei and Sunderland, 1996), and the amazing diversity, combined with ease of sampling with pitfalls, have made them often studied as ecological indicators (Dennis et al., 1997; Rainio and Niemela, 2003; Pawson et al., 2008).

Carabids have been reported to indicate intensity of habitat disturbance (Pearce and Venier, 2006), and fragmentation (Pawson et al., 2008). Large-bodied species have also been reported to disappear in some instances, even when species richness has increased (Vanbergen et al., 2005).

One of the disadvantages of using carabids for feeding guild study, is the diversity of guilds exhibited by this group. Even though carabids have received much attention from researchers, the vast majority of species have not received enough life history study to determine their true feeding habits. However, many of the prominent groups, such as Cicindelinae and Carabini, are believed to be composed of primarily, if not entirely, predatory species.

Species that were not believed to be predatory were omitted from the current study as they are beyond the focus, but it should be noted that omnivorous, granivorous or phytophagous carabid species diversity may indeed be just as relevant to assessment of Ozark glade habitats as are predatory taxa.

Purpose and Objectives

Habitat assessment can generate information vital to the creation of effective management strategies. Glades in the Ozark Mountains have had various means used to assess them, to varying degrees of success, that often rely on information from floral analysis or presence of glade associated vertebrates. The purpose of this dissertation work is to understand more about glades by studying sites in varying states of both forest succession and management history (Intact, Degraded and Restored Site Types), and to determine if monitoring selected taxa of epigeal predatory arthropods by land managers can be a practical and informative option for Ozark glade assessment. This research has several primary objectives, each with both general and specific hypotheses to be tested.

Objective 1: To determine if there are measurable differences among Intact, Degraded and Restored Site Type habitat characteristics.

Hypothesis: Measurements of selected groundcover and other habitat characteristics, will differ among Site Types.

Objective 2: To determine if there are measurable differences in selected taxa of epigeal predatory arthropods among Site Types, especially arthropods associated with glades, and whether there are correlations between the selected taxa and recorded habitat characteristics.

Hypothesis 1: Measurements of the abundance and diversity of selected taxa of epigeal predatory arthropods will differ among Site Types.

Hypothesis 2: Correlations exist between collection numbers of selected taxa and habitat characteristics recorded from glade sites.

Objective 3: To determine whether collection of selected taxa of epigeal arthropods (including individual species) can be useful for monitoring and assessing Ozark glade habitats.

Hypothesis: Collection and comparison of selected taxa can be useful in assessing the status of Ozark glade habitats.

Objective 4: To determine the utility and applicability of methods employed for habitat assessment and epigeal arthropod trapping by non-specialists.

Hypothesis: None applicable to this objective.

Note: One outcome of this work will be a report of collections for selected taxa, but creating a complete inventory of epigeal predatory arthropods of Ozark glades was not a primary objective.

MATERIALS AND METHODS

Site Descriptions

Sites were initially chosen based on accessibility as well as the presence of sandstone as the primary bedrock type. Floral community composition is influenced by bedrock type, and so, presumably, different bedrock types also differ in herbivorous arthropod community composition, which could in turn impact predator diversity. Therefore, sites were chosen with similar bedrock to eliminate bedrock type as a possible confounding variable. In total, nine sandstone glade sites were selected for the current study. All sites were in Stone County, Arkansas, and located within the Sylamore Ranger District of the Ozark-St. Francis National Forest. The exact locations of these sites have been intentionally withheld to help deter exploitation of flora and fauna, particularly *A. hentzi* and *C. collaris,* two species popular in the pet trade. This omission also aligns with the precedent set forth in the work by Brewster (2019), which included several of the same sites utilized for this study. Should exact locations be of interest, please contact the Sylamore Ranger District Office for further information. The estimates of glade perimeter and area, as well as the primary direction of slope for each site are recorded in Table 1. Individual site descriptions, including abiotic and biotic composition, as well as management history of each site, are detailed in Appendix B.

Site Types

Study sites were assigned to one of three site types based on the apparent stage of succession exhibited and/or recent known management history. Early-sere glade sites (hereafter, Intact) were chosen based on a lack of apparent encroachment by woody species, floral resemblance to early-sere glade descriptions of Steyermark (1940), and personal communication with Jim McCoy (District Ranger, Ozark-St. Francis National Forest, Sylamore Ranger District, June, 2018), who indicated that these sites were among the best examples of sandstone glades in the area. These sites have received maintenance efforts such as prescribed burns and herbicide treatment of introduced and invasive herbaceous plant species in recent years, but they have not received restorative efforts such as cedar removal or species reintroduction. Late-sere (hereafter, Degraded) glade sites were chosen based on apparent encroachment by woody species, particularly cedars (*J. virginiana* and *J. ashei*), as well as a lack of recent restorative effort. The third site type, Restored, consisted of glades that have received recent restorative management efforts, including mechanical removal of woody vegetation, prescribed burning, and, in the case of two sites, reintroduction of *C. collaris*.

Initially, seven sites were selected. In September, 2018, one site was dropped from the study due to re-evaluation of glade characteristics, and three other sites were added in October, 2018, to give a total of nine sites, with three sites assigned to each of the three site types.

Habitat Characteristics

For each glade site, estimated lines of glade perimeter and subsequent area calculations were generated using satellite images from Google Earth (Google, n.d.) and the imaging program ImageJ (Schindelin et al., 2012). The primary cardinal direction of glade slope and presence of *C. collaris* were recorded onsite and can be found along with perimeter and area estimates in Table 1. Groundcover was sampled onsite using a modified Step-Point method and photos were taken of quadrats surrounding each trap for estimation of groundcover. Specific sampling locations for both the modified Step-Point and photographic groundcover estimation methods were oriented to the randomized location of arthropod trap grids, which differed at each site, in each trapping season. Sampling occurred at each site on May 25-26, 2019, and again on June 20-21, 2020.

Modified Step-Point Method

One of the two methods used for groundcover analysis of each site, each trapping season, was modified from the Step-Point method described by Evans and Love (1957) for groundcover estimation of rangelands. The modified Step-Point method consisted of placing a rod (galvanized electrical conduit 3 m in length and 1.6 cm in diameter), with the tip on the ground and the rod perpendicular to the glade surface, and recording presence of groundcover categories touching it. This occurred every meter along four transects, each 50 m in length, for a total of 200 sample points per site (each trapping season). Transects were aligned with the four columns of traps, and equidistant in both directions from the center trap of a column (Figure 5). Groundcover categories were Grasslike (including grasses [Poaceae], sedges [Cyperaceae] and rushes [Juncaceae]), Forbs, Cedar, Woody (non-cedar), Moss, and Exposed Rock, and the presence of each category was recorded at each sample point. The number of times a groundcover category was detected at each site was analyzed for differences using one-way ANOVA and Tukey HSD with the programs R (R Core Team, 2020) and RStudio (RStudio Team, 2015). Variables analyzed for significance included the total number of groundcover category detections for each category from both seasons combined, the number of detections for each category in each individual season, and differences between Sites and Site Types for each category of groundcover.

Photographic Groundcover Estimation

Photographs of 1 m² quadrats around each individual trap location were taken for use in later classifications of groundcover. Photographs were taken of each trap location (12) in each sampling year (2019 & 2020), and thus yielded a total of 24 quadrats photographed for each of the nine sites. A square quadrat frame (1 m x 1 m) was constructed from PVC pipe, 2.1 cm in external diameter (Figure 6A-D). The quadrat frame was placed horizontally on the ground, with the trap approximately centered, before photographs were taken. The placement of the frame around each trap was random in orientation to cardinal direction. Photographs of framed quadrats were taken with an iPhone 7+ held directly above each trap from a height of approximately 2 m in full natural light. Midday photography was avoided to lessen the effect of shadows cast by the photographer.

The same six groundcover categories were classified for each photo used in the modified Step-Point method (Grasslike, Forbs, Cedar, Woody, Moss and Exposed Rock), as well as three more (Bare Soil, Dead Wood and Leaf Litter). Classification of ground cover categories followed the cover-class scale presented by Daubenmire (1959) which consists of six intervals: 1 = Present-5%, 2 = 6-25%, 3 = 26-50%, 4 = 51-75%, 5 = 76-94%, 6 = 95-100%. If a groundcover category was missing from a quadrat 0 was recorded. Examples of quadrats dominated by one groundcover type (those rated 4 - 6 on the Daubenmire scale) can be seen in Figure 6.

I, as well as another individual, classified photos using the Daubenmire scale. A random number generator was used to select the order in which photos were classified and all photo-grading periods were limited to a maximum of 45 minutes to reduce fatigue. The purpose of two individual graders was to determine the similarity of grading between myself and the second individual, who had no familiarity with the sites or quadrats photographed, but did have previous experience with plant identification. Both photo graders watched a 20 min informational video on how to apply Daubenmire cover classes to photos (Abbott, 2013).

Although both another grader and I evaluated the photographs, analysis of groundcover from photographs was performed only with the groundcover estimates I made. Estimates of cover class figures for each of the 12 photographed trap locations within a glade site were averaged in each of the years sampled (2019 & 2020). Differences in means of groundcover class categories between sites and Site Types were explored using one-way ANOVA and Tukey HSD with the programs R (R Core Team, 2020) and RStudio (RStudio Team, 2015).

Arthropod Sampling

Each site was sampled for epigeal arthropods with two different styles of modified pitfall traps designed to be workable in a variety of microhabitat conditions, including the shallow soils, or exposed rock, that define glades. The initial trapping season (2018-2019) began with deployment of traps on August 11, 2018. Traps were collected and emptied every four weeks. Because of the removal of one site, samples for September and October, 2018 were collected from only six sites. Addition of three sites gave the final, balanced design of three sites of each type, beginning with the collections in November, 2018. The first trapping season included 14 collection events and ended September 15, 2019. Based on the results of the first trapping season, traps for the second trapping season were deployed on March 28, 2020, and collection events occurred every four weeks until a final collection date of August 15, 2020.

Trap Designs

Creating ramp traps that intercepted arthropods and allowed a means of ascension, enabled collection containers to sit atop the ground surface, yet still create a basin deep enough for effective capture and preservation of a wide range of arthropod species and sizes. The ramp trap style allowed for sampling with pitfall trap collection containers that would have been infeasible in glades.

The first of two designs adopted for this study, termed “lunchbox-style” traps (LST), was very similar in design to traps used by Patrick and Hansen (2013). The collection containers consisted of clear polypropylene plastic containers (StarPlast, Haifa, Israel), 1.2 L in capacity, with squared sides and teal-blue, press-lock lids. These containers measured approximately 10 cm in height, and tapered in width on each side from approximately 12 cm at the lid to 10 cm at the base. Openings were created in the collection container for specimen entry on two opposite sides. A soldering iron was used to create these openings because the melting action made the somewhat brittle plastic less likely to crack. Each opening was 6 cm wide, 1.5 cm high, and 6 cm above the bottom of the collection container. Ramps were created from .25 mm thick aluminum flashing using a template to standardize ramp surface area. The flat flashing “slugs” (Figure 2A) were folded into ramps (Figure 2B) that included an edge designed to fit snugly into the side openings (Figure 2D). Ramps were coated with a textured paint (Rust-oleum Multicolor Textured, Desert color, Vernon Hills, Illinois, USA) to decrease daytime ramp thermal conductivity, as well as generate a surface for target arthropods to gain enough traction for ramp ascension.

The second trap design seems to be novel, and was designed around repurposed field-marker cones used in sports such as soccer (Figure 3A), termed “cone-style” traps (CST). Cones were 30.5 cm diameter and made of low-density polyethylene. Four slots, each 4 cm long. were cut into the cone surface, and the cone height was trimmed to 9 cm which increased the top opening diameter to 14 cm (Figure 3B). The key piece in the design for this trap is shown in Figure 3C. It is a component allowing for attachment of a lid, as well as an internal collection container. With the narrow end of this piece oriented toward the top of the cone body, the 6.3 cm tab on the bottom of this piece was inserted into the side of the cone through the 4 cm slot (Figure 3D). Each trap required four of these pieces. They were made from cone material using a template and attached to the cone by insertion into slots created in the cone surface. The exterior surface of the cone, which was coated with the same textured spray paint as the LST ramps, acted as a ramp with 360-degree coverage. Painting on CST was for the same purposes as was with the LST ramps, and effectively changed the color of the CST cones from yellow (Figures 3A-E) to a mottled tan/gray (Figures 4A-B). Lids for CSTs were made of red foam serving bowls, somewhat octagonal in shape, and 18 cm in diameter (Figure 4C). These were held in place with metal binder clips. Collection containers were plastic serving bowls (Merrick Engineering, Corona, California, USA) 6.3 cm in height made of red polypropylene plastic with an opening diameter of 15.2 cm, slightly larger than that of the top opening of the cone (Figures 3E and 4D).

Trap Grid Design

At each of the nine glade sites, 12 traps (6 LST, 6 CST), were deployed in a rectangular, 3 x 4 trap grid with traps spaced 13 m apart. Trap styles alternated within each column and row of the grid (Figure 5). Grids were chosen over transects due to the limited size of some sites, and the need to maintain a standardized distance between traps for possible future intra-site trapping analysis. For intra-site analysis of trap catch, it is also desirable for traps to be spaced such that each trap has an equal chance of catching each specimen sampled from an individual site (Leather, 2005). Due to the limited ability of some small arthropods to traverse long distances, it was presumed that a grid design would keep the distances between farthest-apart traps closer than would a transect. Initial trap deployment in both 2018 and 2020 consisted of randomizing the exact location of the first trap at each site with a series of dice rolls that dictated the location and the orientation of the trap grid to this initial trap location. The randomizing procedure is described in Appendix C.

Trap Deployment and Specimen Collection

When traps were deployed, care was taken to alter the trapping site as little as possible. However, it was necessary to initially level some sites during deployment each season for proper trap function. It was also necessary for proper deployment of traps to nestle the edges of ramps into substrates (if possible) each time they were set or reset. This was done to increase interception of target species that might otherwise go under a ramp if a gap was present between the ramp and the ground. Ramps of LSTs were always oriented with entrances perpendicular to slopes if present. This prevented water draining downhill from being funneled into the trap's collection container. Traps of both styles, when placed in areas of high wind exposure, were weighted with rock gathered from the trapping site (Figure 2C). At each collection, traps that were damaged, disappeared or lost to animal disturbance were replaced. During the 2018-2019 trapping season, sun exposure required maintenance of traps, consisting of replacing sun-damaged lids and/or repainting flaking ramp surfaces. The 2020 season did not expose traps long enough for sun damage to require maintenance.

The preservative used in collection containers was 100% propylene glycol (PG). In preliminary trials PG was found to evaporate slowly, be suitable for preservation of specimens during four-week collection intervals, and it is non-toxic to vertebrates. Upon initial trap deployment each season, 250 ml of fresh PG was poured into each collection container. Each time specimen collection occurred, PG was recovered from collection containers as specimens were separated. The recovered PG from all collection containers at a site was then pooled and, if necessary, fresh PG was added to achieve the volume required to redistribute 250 ml into each collection container. This reclamation procedure was implemented to reduce the amount of PG required, but also mixed any chemical signatures left behind from collected samples between all collection containers upon redistribution. Upon resetting of traps, collection containers were returned to locations among the grid that were within a trap type (*i.e*., LST collection containers were distributed among LST trap locations).

Trap-collection containers were each emptied over a 250-micron sieve and rinsed of excess PG with water from a squirt bottle. Specimens collected were then washed from the sieve into jars using 100% EtOH from a squirt bottle. Jars were returned to the laboratory and trap catch sorted using a stereo dissecting scope (Nikon SMZ745).

All spiders and scorpions were removed from trap catch and kept in 90% EtOH. All adult gnaphosid spiders were separated by sex (required for species-level identification), then identified to genus using *Spiders of North America* (Ubick et al., 2005), and to species using appropriate keys (Platnick, 1975; Platnick and Shadab, 1975a,b, 1976a,b, 1977, 1980a,b, 1981, 1982, 1983, 1988; Heiss and Allen, 1986).

Adult carabid beetles were pinned and identified to tribe and genus using *American Beetles Vol. 1* (Ball and Bousquet, 2001), as well as *Ground Beetles and Wrinkled Bark Beetles of South Carolina* (Ciegler, 2000). Because the taxonomic focus was on predatory arthropods, carabids collected belonging to the tribes Harpalini and Zabrini, as well as the genera *Agonum* and *Bembidion,* were omitted from the study, as these groups have been reported to be primarily composed of non-predatory species (Lundgren, 2009; Kulkarni et al., 2015). The genus *Brachinus* was also omitted because adults of some species are considered omnivorous, and because they are a “complex and taxonomically difficult group” (Erwin, 1970). Various keys were utilized for species identifications (Benschoter and Cook, 1956; Gidaspow, 1959; Bell, 1960; Choate, 2001; Purrington and Drake, 2005; Bousquet, 2012). Several small (< 3 mm length) carabid specimens were identified only to genus.

Richness and abundance for both gnaphosids and predatory carabids were recorded. Abundance was also recorded for the sole scorpion species found in the Ozarks, *C. vittatus*, and for immatures of the sole theraphosid spider, *A. hentzi* (traps were not capable of capturing adult *A. hentzi*).

Diversity Analysis

Species richness (Spellerberg and Fedor, 2003) was considered the number of species from a defined unit (Site, Site Type, sample, etc.), and species diversity referred to richness combined with collection total. All analyses utilized the programs R (R Core Team, 2020) and RStudio (RStudio Team, 2015). Collection totals were analyzed for each trapping season independently as well as combining seasons. Certain analyses were also performed on data trimmed to include only species present with sampled abundance ≥ 10 individuals, or by combining April-August trap collection dates.

The Shannon-Wiener Index of diversity was calculated for each site utilizing the formula $H' = -\Sigma (p_i * \ln(p_i))$, where p_i = the proportion of each individual species to the total numbers for each other species sampled (Shannon, 1948). The Simpson's Index was also calculated for each site using $D = 1 - ((\Sigma n (n-1))/(N(N-1))$, where n is the abundance of an individual species and N is the abundance of all species sampled (Simpson, 1949).

Identification of statistically separated groups of sites or Site Types based on Shannon-Wiener and Simpson's diversity indices required utilizing the Kruskal-Wallis test. The Kruskal-Wallis does not adhere to assumptions such as homoscedasticity or normality (Kruskal and Wallis, 1952), which may not apply to small sample sizes, as generated in the current study. Samples generated via pitfall-style trapping do not reflect abundances of species, as activity and abundance cannot be separated as variables. Therefore, the information lost in conversion from diversity index figures to the rank order utilized by the Kruskal-Wallis test is believed to be of little importance. When necessary, to determine what groups were statistically differentiated, the Dunn’s test was performed (Dunn, 1961).

The Morisita index of dissimilarity (Morisita, 1959) is a pairwise comparison used in this study to directly compare differences in species composition of sites. This index typically ranges between 0-1 in most cases, but can exceed one, with lower numbers meaning lower dissimilarity and zero meaning a lack of dissimilarity (complete similarity). Morisita index figures were calculated for each possible pair of sites with the formula:

$$C_M = \frac{\sum_{i=1}^{S_{1,2}} p_i \pi_i}{1/2(\sum_{i=1}^{S_1} p_i^2 + \sum_{i=1}^{S_2} \pi_i^2)}$$

as presented by Chao et al. (2006) where p_i = collection total of a species from Site X, π_i = collection total of Site Y, and S = number of unique species. Morisita index figures for each site were averaged and compared within, and between, Site Types as well.

Rarefaction curves were created from sampled abundance data for each site following methodology set forth by Sanders (1968). The function used to generate rarefaction curves was:

$$f_n = K - \binom{N}{n}^{-1} \sum_{i=1}^{K} \binom{N - Ni}{n}$$

Where n is the subsample taken, N is the total sampled abundance, K is the total number of collected species, and N_i is the collection total of an individual species.

Non-metric Multidimensional Scaling (NMDS) was performed on species abundance data using the *metaMDS* function within Package ‘vegan’ for R (Oksanen et al., 2020). The NMDS utilized Bray-Curtis dissimilarity index figures (Bray and Curtis, 1957) generated by the formula $BC_{ij} = 1 - (2C_{ij} / (S_i + S_j))$ where C_{ij} is the sum of the lowest collection total figures for each species shared between sites, and S_i and S_j are the collection total figures from each site comparison. Environmental variables were fitted to NMDS data using the vegan function *envfit* (Oksanen et al., 2020). Again, the non-parametric nature of this method does not rely on assumptions about homoscedasticity or normality as ranks are assigned to continuous variables.

RESULTS

Habitat Characteristics

Modified Step-Point Method

Frequencies of detection for groundcover categories did not differ between sampling years (one-way ANOVA, all $p > 0.05$). All groundcover categories were detected at each of the nine sites, except for a lack of moss at Site 6 and no cedar at Site 7 (Table 2). The frequency of detection differed among groundcover categories (F = 56.66; df = 5,12; $p < 0.0001$) when all sites and both years of sampling were combined. The groundcover category Grasslike was detected most frequently, and was detected significantly more often than the category Forbs, which was found more frequently than all other groundcover categories ($p < 0.05$, Tukey HSD, Figure 7).

Four of the groundcover categories showed significant differences in frequency of detection among Site Types (Table 2) based on one-way ANOVA analyses (Grasslike: F = 11.19; df = 2,15; $p = 0.001$; Figure 8A; Cedar: F = 14.57; df = 2,15; $p = 0.0003$; Figure 8B; Woody: F = 5.36; df = 2,15; $p = 0.018$; Figure 8C; Exposed Rock: F = 11.35; df = 2,15; $p = 0.001$; Figure 8D). Degraded sites had a significantly greater ($p < 0.05$, Tukey HSD) frequency of cedars than both Restored and Intact sites, and significantly greater frequencies of Grasslike and Woody categories than Intact sites ($p < 0.05$, Tukey HSD). Intact sites had significantly more exposed rock than Degraded sites ($p < 0.05$, Tukey HSD).

Photographic Groundcover Estimation

Average estimations of Daubenmire cover classes of groundcover from 12 quadrats at each site, along with averages for Site Types can be found in Table 3. All nine groundcover categories estimated were present at all Degraded sites, whereas only five groundcover categories were estimated present at all Intact and Restored sites (Table 3). One-way ANOVA showed significance among sites for the groundcover category of Cedar (F = 21.08; df = 8,9; $p < 0.0001$) with Degraded Sites 2 and 4 having significantly more cedar (Tukey HSD, $p < 0.05$) than all other sites. One-way ANOVA of the Moss groundcover category showed a significant difference among sites (F = 4.13; df = 8,9; $p = 0.025$). Degraded Site 2 had more moss (Tukey HSD, $p < 0.05$) than Restored Site 6. One-way ANOVA of the Exposed Rock category was significantly different among sites (F = 5.62; df = 8,9; $p = 0.009$) with more rock (Tukey HSD, $p < 0.05$) present at Site 6 than any of the Degraded sites (2, 4 and 9). Leaf Litter was significantly also different among sites based on one-way ANOVA (F = 5.64; df = 8,9; $p = 0.009$) with Degraded Site 9 having more leaf litter (Tukey HSD, $p < 0.05$*)* than Restored Sites 6 and 7.

Four of the groundcover categories showed significant differences in average groundcover class among Site Types (Figure 9) based on one-way ANOVA analyses (Grasslike: F = 8.09; df = 2,15; $p = 0.009$; Figure 9A; Cedar: F = 7.47; df = 2,15; $p = 0.006$; Figure 9B; Exposed Rock: F = 8.33; df = 2,15; $p = 0.003$; Figure 9C: Leaf Litter: F = 12.15; df = 2,15; $p = 0.0007$; Figure 8D). Degraded sites had significantly greater ($p < 0.05$, Tukey HSD) groundcover class averages for Cedar, Exposed Rock and Leaf Litter than both Restored and Intact sites, and Degraded sites also had a significantly greater average for the Grasslike groundcover category than Intact sites ($p < 0.05$, Tukey HSD). There were no significant differences found between Restored and Intact Site Types.

Agreement in Daubenmire groundcover class assignment between photo graders ranged between 38% for the Forbs category to 94% for Cedar. Further information regarding the inter-rater reliability of Daubenmire groundcover class assignment via photo grading can be found in Appendix D.

Gnaphosids

Two trapping seasons yielded 1,410 adult gnaphosids, 736 from 14 collection events in 2018-2019, and 674 from five collection events in 2020. A total of 1,412 immature gnaphosids was collected, but identifications were not made and were not included in this study. The average numbers of adult gnaphosids collected per trap, by each month of collection, are shown in Figure 10. The only collection event that failed to produce any gnaphosids occurred in February of 2019. The overall average catch for 2018-2019 was 0.64 per trap. From September 2018-March 2019, as well as from August and September of 2019, a maximum of 0.56 gnaphosids per trap was recorded (Figure 10). April-August collection dates in 2019 produced an average of 1.27 gnaphosids per trap, with a peak in June of 2.04 per trap. Sampling from April-August 2020 yielded an average of 1.40 per trap, with a low of 0.65 gnaphosids per trap in August and a peak of 2.57 per trap in June (Figure 10).

The gnaphosids collected in 2018-2019 consisted of 25 species from 10 genera; the 2020 trapping season produced gnaphosids from 27 species and 12 genera. In total, 32 species from 14 genera were collected. The genus *Drassyllus* was the most speciose collected, consisting of nine species, followed by the genera *Micaria* and *Zelotes*, each with five species collected. These three genera each contained one of the three most collected species of gnaphosids overall, *Drassyllus lepidus* (243), *Micaria punctata* (241), and *Zelotes aiken* (202). These three species were also the top three in collection totals for each trapping season, though their relative rank differed—in 2018-2019 *M. punctata* was the most numerous (131), whereas *D. lepidus* was the most numerous (135) in 2020.

The numbers of each species from each trapping season are shown by collection date in Table 4. Five species were captured only in 2018-2019, and seven species were present only in

2020. Two of the species collected in 2019 (*Drassyllus rufulus* and *Micaria longipes*) were present only in months not sampled in 2020. Ten species were represented by five or fewer individuals collected, five of which were present as single specimens—*Cesonia bilineata* and *Micaria seminola* in 2018-19 and *Litopylus temporarius, Micaria longispina* and *Synaphosis paludis* in 2020 (Table 4).

Analyzing the abundance of gnaphosids by sex revealed 487 males and 249 females in the 2018-2019 season, versus 420 males and 254 females in 2020 (Table 4). In 2018-2019 only males were found for four species, *C. bilineata, Drassodes gosiutus, Drassyllus frigidus* and *M. seminola*, whereas only females were found for *Drassyllus creolus* and *D. nannellus*. In 2020, *D. nannellus, M. longispina*, and *Sergiolus tennessensis* were present only as males, whereas *D. gosiutus, D. frigidus, Litopylus temporarius, M. laticeps*, *Sergiolus capulatus*, and *S. paludis* were present only as females.

The number of gnaphosids caught in Lunchbox Style Traps (LST) was 894 versus 516 in Cone Style Traps (CST). Of the 32 species captured, LST accounted for ≥ 50% of collection totals for 22, and seven species were caught only by LST. Three species were unique to CST catch, each of which were represented by a single individual. A listing of species collected by each trap type is in Appendix E.

The total numbers of gnaphosids collected from each site, both trapping seasons combined, are shown in Table 5. All sites produced seven or eight gnaphosid genera, but numbers of species collected ranged between 14 at Intact Site 8 and 22 at Degraded Site 2. Total numbers of individuals from both seasons range from 78 for Degraded Site 4, to 272 for Restored Site 7. All of the species unique to individual sites existed as single specimens. Degraded Site 2 samples showed the most unique species with three (*C. bilineata, L.*

temporarius and *M. seminola*). Restored Sites 6 and 7 each produced one unique species, *S. paludis* and *M. longispina*, respectively (Table 5).

The total numbers of each species of gnaphosid collected from each Site Type, with both trapping seasons combined, are presented in Table 6. Total numbers collected by Site Type ranged from 287 for the three Degraded sites, to 569 for the Intact sites. Whereas the total number of genera collected was 14, only 11 genera were found from any Site Type. Likewise, numbers of species present at a Site Type ranged from 23-27, versus the study-wide total of 32 species. Species unique to Degraded sites were *Drassyllus covensis, L. temporarius* and *M. seminola*. Species unique to Restored sites were *M. longispina* and *S. paludis*. No species were unique to Intact sites. Three species were shared only between Degraded and Restored sites, *D. frigidus, D. rufulus* and *Zelotes laccus* (Table 6). Four species were shared only between Degraded and Intact sites, *D. nannellus, M. laticeps, Nodocion floridanus* and *S. capulatus;* and Restored and Intact sites shared *D. gosiutus, Herpyllus ecclesiasticus,* and *S. tennessensis*. The most abundant species from Intact sites were the same, and in the same rank order, as the three most abundant species overall (*D. lepidus, M. punctata* and *Z. aiken*). The three most-abundant species from Restored sites were *M. punctata, Z. aiken* and *Gnaphosa sericata;* the three most-abundant species from Degraded sites were *Drassyllus dixinus, Zelotes hentzi* and *Z. lymnophilus* (Table 6).

Diversity Measures and Indices

Kruskal-Wallis testing of diversity indices for both trapping seasons combined showed no significant differences between individual sites for Shannon-Wiener ($\chi^2 = 15.05$, df = 8, $p = 0.058$), or Simpson's diversity indices ($\chi^2 = 13.33$, df = 8, $p = 0.101$) (Figure 11).

Kruskal-Wallis testing for Site Types did not show significance for either trapping season individually (Table 7), but when data from both trapping seasons were combined, the figures were found to be significantly different for Shannon-Wiener ($\chi^2 = 8.92$, df = 2, $p = 0.012$), and Simpson's ($\chi^2 = 6.18$, df = 2, $p = 0.039$) diversity indices. These differences occurred between Degraded and Intact Site Types according to the Dunn's test ($p < 0.05$; Table 7; Figure 12).

Morisita dissimilarity index figures for the 2018-2019 trapping season are shown in Table 8. Restored Site 6 and Intact Site 10 were the only two sites shown to be completely similar (dissimilarity figure = 0). Degraded Site 9 showed the only dissimilarity figures > 0.75 for this trapping season, and >0.90 dissimilarity was present between Site 9 and Restored Site 7, as well as between Site 9 and Intact Site 3 (Table 8). Average dissimilarity between Site Types was 0.646 for Degraded and Intact sites, 0.610 for Degraded and Restored sites, and 0.369 for Restored and Intact sites. Average dissimilarity figures within Site Types were 0.363 for Degraded, 0.524 for Restored, and 0.354 for Intact. The Morisita dissimilarity index figures are shown in Table 9 for the 2020 trapping season. Two pairs of sites, Degraded sites 2 and 4, as well as Restored Site 6 and Intact Site 3, were shown to be completely similar. Intact Site 10 presented the only index figures above 0.82, which were seen between it and sites 4, 5 and 9 (Table 9). Average dissimilarity between Site Types were 0.711 for Degraded and Intact sites, 0.531 for Degraded and Restored sites, and 0.438 for Restored and Intact sites. Average dissimilarity figures within Site Types were 0.160 for Degraded, 0.522 for Restored, and 0.371 for Intact sites.

Trimmed Data

A total of 15 species of the 32 species collected were represented by fewer than 10 individuals. Trimming the data to include only species with ≥ 10 individuals collected, the 2018-

2019 trapping season was reduced from 25 species and ten genera to 15 species from six genera, and the 2020 season was reduced from 27 species and 12 genera to 11 species from five genera (Table 4). Five species were found in the trimmed 2018-2019 data but not in trimmed 2020 data, and only one of the 11 species remaining in the trimmed 2020 dataset was unique to 2020 (Table 4). With both trapping seasons combined, 17 species from 6 genera remained in the trimmed dataset. Although *Z. laccus* had fewer than 10 individuals in either season, combining seasons yielded more than 10 individuals.

The genus *Drassyllus* was the most speciose of the genera remaining in the trimmed dataset, with six species (a reduction of three from unaltered data), followed by *Zelotes* with four species, and *Gnaphosa* and *Micaria* each with two species.

Total trimmed numbers collected by Site Type for both seasons were 273 for Degraded, 537 for Restored, and 554 for Intact sites. The percent of individuals in the trimmed dataset compared to the unaltered dataset for all three Site Types exceeded 95%: 95.1% (273 trimmed/287 unaltered) for the Degraded Site Type; 96.9% (537 trimmed/554 unaltered) for the Restored Site Type, and 97.4% (554 trimmed/569 unaltered) for the Intact Site Type.

Kruskal-Wallis testing of diversity indices for data trimmed to include only species with a total of ≥ 10 individuals collected in both trapping seasons combined found no significant differences between sites for Shannon-Wiener ($\chi^2 = 11.20$, df = 8, $p = 0.191$), or Simpson's diversity ($\chi^2 = 10.46$, df = 8, $p = 0.235$) indices (Figure 13).

For Site Types, neither diversity index was found to be significantly different for either trapping season individually (Kruskal-Wallis, $p < 0.05$; Table 7). However, when data from both trapping seasons were combined, figures were found to be significantly different between Site Types for Shannon-Wiener ($\chi^2 = 6.87$, df = 2, $p = 0.032$) and Simpson's ($\chi^2 = 6.22$, df = 2, $p =$

0.044) indices (Table 7 & Figure 14). The differences occurred between Degraded and Intact Site Types according to the Dunn's test ($p < 0.05$; Table 7; Figure 12).

The Morisita dissimilarity index figures for the 2018-2019 trapping season trimmed to include only species with ≥ 10 collected are shown in Table 10. A single pair of sites were shown to be completely similar, Restored Site 6 and Intact Site 10. Degraded Site 9 had two figures above 0.90, which were shared with Restored Site 7 and Intact Site 3 (Table 10). Average dissimilarity between Site Types was 0.642 for Degraded and Intact sites, 0.609 for Degraded and Restored sites, and 0.367 for Restored and Intact sites. Average dissimilarity figures within Site Types were 0.359 for Degraded, 0.519 for Restored, and 0.359 for Intact. The Morisita dissimilarity index figures for the 2020 trapping season trimmed to include only species with ≥ 10 collected are shown in Table 11. Two pairs of sites were shown to be completely similar, Degraded sites 2 and 4, as well as Restored site 6 and Intact site 3. Intact Site 10 had the only index figures above 0.82, which were seen between it and sites 4, 5 and 9 (Table 11). Average dissimilarity between Site Types was found to be 0.705 for Degraded and Intact sites, 0.538 for Degraded and Restored sites, and 0.434 for Restored and Intact sites. Average dissimilarity figures within Site Types were 0.151 for Degraded, 0.522 for Restored, and 0.378 for Intact.

Carabids

The two trapping seasons yielded 592 predatory carabids, 404 from 14 collection events in 2018-2019, and 188 from five collection events in 2020. The average numbers of predatory carabids collected per trap, by each month collected, are shown in Figure 15, and they ranged from a low of 0.01 per trap in February 2019 to a high of 0.95 per trap in August 2019. Not unexpected, the three-month collection period of January-March 2019 yielded a peak of 0.03 per trap, whereas the spring and summer months (April-August 2019) produced an average of 0.44

per trap and the autumn months (September-November) produced 0.56 per trap (Figure 15). The second trapping season of April-August of 2020 produced 0.43 per trap average with, again, August having the maximum 0.75 per trap (Figure 15).

The 404 predatory carabids collected in the 2018-2019 season consisted of 33 unique taxa from 21 genera, and 188 predatory carabids from the 2020 season yielded 25 taxa belonging to 19 genera (Table 12). In total, 34 species and three morphospecies from 23 genera were collected. The genus *Chlaenius* was the most speciose collected, with six species and one morphospecies. The second-most abundant genus was *Dicaelus* with three species. The tiger beetle, *Tetracha virginica*, was the most-captured predatory carabid species both seasons. The second-most captured each season was *Pasimachus depressus.* In 2018-2019 *Calosoma sayi* was the third-most captured species but was absent from 2020 samples (Table 12). There were 11 species and one morphospecies captured only in 2018-2019. Of these, three (*Chlaenius vafer, Cincindela sexguttata*, and *Pterostichus sculptus*) were present only in months not sampled in 2020, and another six were present as single specimens only. A total of three species and one morphospecies were present only in 2020 samples, of which three (*Chlaenius impunctifrons, Helluomorphoides texanus,* and *Semiardistomis viridis*) were singlets.

The numbers of predatory carabids caught in LST were 369 versus 223 in CST. Of the 37 taxa captured, ten taxa were only caught by LST and 28 taxa had ≥ 50% captured by LST (Appendix E). Three taxa were unique to CST, two of which were represented by a single individual (*Lebia analis* and *Tachys* sp.).

The total numbers of predatory carabids collected from both trapping seasons combined are shown by individual site in Table 13. Genera collected range in number from 9-15 and numbers of species range from 9 (Restored Site 7), to 18 (Restored Site 6). Numbers of individual carabids collected from a site ranged from 27 (Restored Site 7), to 111 (Restored Site 6). Of the 14 taxa unique to individual sites, ten existed as single specimens, and Site 2 had the greatest number of unique taxa, with five (Table 13).

The total numbers captured for each species of predatory carabid at each Site Type are presented in Table 14. Total numbers collected for both seasons ranged from 181 in the Intact sites to 217 in the Restored sites. The total number of genera collected over both seasons was 23, but the numbers of genera ranged from 16 (Intact Site Type) to 19 (Degraded Site Type). Although the total number of species was 37 (34 identified species plus 3 morphospecies), numbers of species at each Site Type ranged from 21 (Intact Site Type) to 27 at the Degraded Site Type. Five species were unique to Degraded sites, four were unique to Intact sites and three to Restored sites (Table 14). Six species were shared only between Degraded and Restored sites, whereas only one species was shared only between Degraded and Intact sites, and one species shared only between Restored and Intact sites (Table 14). It is worthy of note that *Calosoma sayi* had a single individual collected from Degraded sites, but had 20 each in collections from Restored and Intact sites. The most abundant species from Degraded sites was *P. depressus,* the second-most abundant overall. The most abundant species in samples from both Restored and Intact sites was the most abundant overall, *T. virginica*.

Diversity Measures and Indices

Kruskal-Wallis testing of diversity indices for predatory carabids from both trapping seasons combined found no significant differences among sites for Shannon-Wiener ($\chi^2 = 5.96$, df = 8, $p = 0.65$), or Simpson's ($\chi^2 = 7.37$, df = 8, $p = 0.498$) indices (Figure 16).

There were also no significant differences found among Site Types for either trapping season individually, or combined (Kruskal-Wallis, $p < 0.05$; Table 15).

Morisita dissimilarity index figures for the 2018-2019 trapping season are shown in Table 16. The most similar sites were Site 7 and Site 8 (dissimilarity = 0.01). The greatest dissimilarity figure (0.899) existed between Degraded Site 4 and Intact Site 3. Average dissimilarity between Site Types was found to be 0.563 for Degraded and Intact sites, 0.533 for Degraded and Restored sites, and 0.335 for Restored and Intact sites. Average dissimilarity figures within Site Types were 0.616 for Degraded, 0.476 for Restored, and 0.443 for Intact. The Morisita dissimilarity index figures are shown in Table 17 for the 2020 trapping season. Two pairs of sites were shown to be completely similar (dissimilarity = 0), Degraded Site 4 and Restored Site 5, as well as Restored site 7 and Intact site 10. The highest index figure of 0.934 existed between Degraded Site 2 and Restored Site 7. In general, sites 2, 4 and 5 were dissimilar from Intact and other Restored sites. Average dissimilarity between Site Types was found to be 0.714 for Degraded and Intact sites, 0.531 for Degraded and Restored sites, and 0.439 for Restored and Intact sites. Average dissimilarity figures within Site Types were 0.247 for Degraded, 0.727 for Restored, and 0.326 for Intact.

Trimmed Data

A total of 19 species of the 37 species collected were represented by fewer than 10 individuals. Trimming the predatory carabid abundance data to include only the remaining 18 species with ≥ 10 individuals sampled, the 2018-2019 trapping season consists of 13 species from ten genera, and the 2020 season consists of five species each from a different genus (Table 12). Seven species found in the 2018-2019 trimmed data were not found in the trimmed 2020 data, but only one of the five species remaining in the trimmed 2020 dataset failed to produce more than ten individuals in 2018-2019 samples (Table 12). With both trapping seasons combined, 554 individuals of 18 species from 13 genera remained. Although *Cicindelidia* *punctulata, Dicaelus elongatus,* and *Paraclivina bipistulata* did not yield ≥ 10 in either trapping season individually, all three had ≥10 individuals when the seasons were combined.

Total trimmed numbers collected by Site Type for both seasons were 181 for Degraded, 196 for Restored, and 167 for Intact sites. The percent of individuals in the trimmed dataset compared to the unaltered dataset for all three Site Types exceeded 90%: 93.2% (181 trimmed/194 unaltered) for the Degraded Site Type; 90.3% (196 trimmed/217 unaltered) for the Restored Site Type, and 92.2% (167 trimmed/181 unaltered) for the Intact Site Type.

Kruskal-Wallis testing of data trimmed to include only predatory carabid species with ≥ 10 individuals, from both trapping seasons combined, found no significant differences between sites at $\alpha = 0.05$ for Shannon-Wiener ($\chi^2 = 2.04$, df = 8, $p = .980$), or Simpson's diversity indices ($\chi^2 = 1.30$, df = 8, $p = .996$) (Figure 17).

There were also no significant differences found between Site Types for either trapping season individually, or combined (Kruskal-Wallis, $p < 0.05$; Table 15).

The Morisita dissimilarity index figures for the 2018-2019 trapping season trimmed to include only species with ≥ 10 collected are shown in Table 18. The two lowest dissimilarity figures were for Intact Site 8 and Restored Site 7 (0.025), as well as Intact Site 8 and Degraded Site 2 (0.039; Table 18). Intact Site 3 and Restored Site 6, as well as Degraded Site 4 and Restored Site 5, had dissimilarities < 0.10 (Table 18). Degraded Site 4 had the highest dissimilarity figure (0.897), which it shared with Intact Site 3. Average dissimilarity between Site Types was found to be 0.496 for Degraded and Intact sites, 0.503 for Degraded and Restored sites, and 0.334 for Restored and Intact sites. Average dissimilarity figures within Site Types were 0.599 for Degraded, 0.479 for Restored, and 0.446 for Intact. The Morisita dissimilarity index figures are shown in Table 19 for the 2020 trapping season. Three pairs of sites were shown to be completely similar (dissimilarity of 0), Degraded sites 2 and 9, Degraded Site 4 and Restored Site 5, and Restored Site 7 and Intact Site 10. All of the dissimilarity figures above 0.83 existed for Degraded Sites 2 and 4, and were between them and Restored and Intact sites (Table 19). A complete dissimilarity (dissimilarity of 1.0) was shown between Sites 3 and 9. Average dissimilarity between Site Types was found to be 0.790 for Degraded and Intact sites, 0.578 for Degraded and Restored sites, and 0.483 for Restored and Intact sites. Average dissimilarity figures within Site Types were 0.248 for Degraded, 0.656 for Restored, and 0.278 for Intact.

Scorpions

The two trapping seasons yielded 442 *Centruroides vittatus*—279 from 14 collection events in 2018-2019, and 163 from five collection events in 2020, making this the most abundant predatory species recorded overall. The average numbers of *C. vittatus* collected per trap each month are shown in Figure 18. In the 2018-2019 trapping season, the January-March collection dates produced zero *C. vittatus* and fewer than 0.16 per trap were recorded from November 2018-June 2019, or from April-June 2020 (Figure 18). Collections from July-October of both trapping seasons produced more than 0.38 scorpions/trap with a peak of 1.13 per trap in August 2020 (Figure 18). The number of *C. vittatus* caught in LST was 119 versus 323 in CST. The total numbers of scorpions collected at each site from both trapping seasons combined are shown in Figure 19. Degraded Site 4 and Restored Site 5 produced no *C. vittatus*, and Restored sites 6 and 7 produced ≥ 100 individuals each. Totals by Site Type were 62 for Degraded sites (zero from Site 4), 209 for Restored sites (zero from Site 5), and 171 for the three Intact sites.

Rarefaction Curves

Gnaphosids

Species rarefaction curves for gnaphosid spiders captured in the 2018-2019 trapping season are shown in Figure 20. Analysis of all curves at the smallest sample size (41; Site 9) showed the greatest species richness belonged to Degraded Site 2 followed by Degraded Site 4. The three Intact sites provided the least-speciose curves at the point of the smallest sample size, as well as the remainder of their durations from this point (Figure 20). Species rarefaction curves for gnaphosid spiders captured in the 2020 trapping season are shown in Figure 21. In this trapping season the smallest sample size (36) belonged to Degraded Site 4 which was followed closely by Restored Site 5. At the point of smallest sample size Degraded Sites 2 and 4 were shown to be the most speciose. The least speciose site at the point of smallest sample size belonged to Intact Site 10. However, before a sample size of 50 was reached, Restored Site 7 became the least speciose and remained so for the remaining of its duration (Figure 21).

When both trapping seasons were combined, again Site 4 was the limiting sample size for direct site comparisons, as it had the smallest sample size (78), and again Sites 2 and 4 were the most speciose at this point (Figure 22). The smallest number of species at the point of site comparison were for Site 8 and three of the four smallest species figures were Intact sites; the fourth was Restored Site 7 (Figure 22).

Species rarefaction curves for gnaphosid spiders from both trapping seasons combined and then trimmed to include only species with ≥ 10 individuals are shown in Figure 23. Site 4 was again the smallest sample size (76) and point of comparison for all sites. Site 4 was also the most speciose site at the point of comparison, followed by Degraded Sites 2 and 9, and then Restored Sites 5 and 6, all of which had more than 10 species at the point of comparison (Figure

23). Intact Sites 3, 8 and 10, as well as Restored Site 7, had between 8-10 species when compared at the point of smallest sample size, and these were also the least speciose curves for a duration prior to, and after this point of comparison.

When 2018-2019 data were reduced to the sampling period from April-August, and combined with the 2020 season, Site 2 was the most speciose at the point of comparison (67). Site 4 had the second most species and was followed by Degraded Site 9 (Table 24). Restored Site 7 had the smallest number of species at the point of comparison, but at a sample size of approximately 160, Site 8 became the least speciose.

Rarefaction curves for the combined April-August dataset discussed above trimmed to include only species with ≥ 10 individuals collected are shown in Figure 25. All three Degraded sites, as well as Restored Sites 5 and 6, had more than 10 species at the point of comparison (66), whereas Restored Site 7 and all three Intact sites had fewer than 10 species at the point of comparison. Site 7 also had the least speciose curve for the remainder of its duration past the point of comparison.

Carabids

Species rarefaction curves for predatory carabid beetles captured in the 2018-2019 trapping season are shown in Figure 26. The smallest sample size, and point of comparison (18) for all sites, was shown to be Restored Site 7. Degraded Sites 2 and 9, as well as Site 6 were more speciose than Site 7 at the point of comparison; the remaining sites were less speciose than Site 7 at this point. Beyond a sample size of approximately 30, Degraded Site 4 was the least-speciose site. Predatory carabid rarefaction curves for the 2020 trapping season will only be discussed for sites with sample size greater than 10 (Sites 3, 7, 9 and 10 omitted). Restored Site 5 showed the most speciose curve throughout its duration and was followed by Degraded Site 4. The smallest species richness was seen from Intact Site 8 and Restored Site 6.

Species rarefaction curves for carabid beetles from both trapping seasons combined are shown in Figure 28. Restored Site 7 had the smallest sample size (27), and along with Intact Sites 8 and 10 had the lowest richness (Figure 28). Degraded Site 9 had the most speciose curve at the point of comparison, and throughout much of its duration (Figure 28).

When predatory carabid data combined from both seasons were trimmed to include species that had ≥ 10 individuals in samples, Site 7 was again the limiting sample size (23) for the point of comparison (Figure 29). At this point, the Site 9 curve remained the most speciose with approximately 10 species. Site 10 had the least richness, but the range of species figures at the point of comparison between sites (excluding Site 9) was fewer than two species (Figure 29).

The 2018-2019 data were reduced to the sampling period from April-August, and combined with the 2020 data, and are shown in Figure 30. Degraded Site 9 had the smallest sample size (22), but the most species at that point (Figure 30). Restored Site 7 had the least species richness, and Restored Site 6 along with Intact Sites 8 and 10 all had between five and seven species at the point of comparison (Figure 30).

When April-August data were combined from both seasons and trimmed to include only the species that had ≥ 10 individuals captured, Site 9 again had the smallest sample size (19) and most species at the smallest sample size (Figure 31). All sites had between 4-8 species at the point of comparison and Intact sites along with Restored Site 7 had the lowest species richness in general throughout their duration.

Non-Metric Multidimensional Scaling

Non-metric multidimensional scaling (NMDS) of gnaphosid spider data from the 2018-2019 trapping season found five environmental variables significant ($p < 0.05$). These were Grasslike, Exposed Rock, Bare Soil and Leaf Litter groundcover categories, as well as slope (Table 20). Site ordinations, along with significant environmental variables are shown in Figure 32. Analysis of the 2020 trapping season found only the Leaf Litter groundcover category to be significant (Table 21; Figure 33). When both trapping seasons were combined, the same environmental variables were found significant as the 2018-2019 trapping season NMDS ordination (Table 22). Site ordinations, along with significant variables, for gnaphosid data combined from both seasons are shown in Figure 34.

Gnaphosid data from the 2018-2019 trapping season trimmed to include only species with ≥ 10 individuals collected showed eight NMDS environmental variables to be significant (Table 23; Figure 35); species ordinations for the 2018-2019 trimmed dataset are plotted in NMDS ordination Figure 36. Gnaphosid data from the 2020 trapping season, trimmed to include only species with ≥ 10 individuals collected, showed five environmental variables to be significant in NMDS ordination (Table 24; Figure 37); species ordinations for the 2020 trimmed dataset are plotted in NMDS ordination Figure 38.

Predatory carabid data from the 2018-2019 trapping season found three environmental variables significant at $\alpha = 0.05$. These were the Woody groundcover category, slope and presence of *C. collaris* (Table 25). Site ordinations, along with significant environmental variables are shown in Figure 39. The 2020 trapping season found only the Woody groundcover category to be significant at $\alpha = 0.05$ (Table 26; Figure 40). When both trapping seasons were combined, Woody and Leaf Litter groundcover categories, as well as slope, were significant

(Table 27). Site ordinations, along with significant variables, for predatory carabid data combined from both seasons are shown in Figure 41.

Predatory carabid data from the 2018-2019 and trapping seasons trimmed to include only species with ≥ 10 individuals collected showed woody groundcover and slope were significant (2018-2019: Table 28; Figure 42; 2020: Table 29; Figure 44); species included in this dataset are plotted in NMDS ordinations in Figures 43 (2018-2019) and 45 (2020).

DISCUSSION

The objectives of this dissertation were to assess Ozark glades of three Site Types, early-sere Intact glades, late-sere Degraded glades, and glades that have received restorative efforts (Restored), to determine if these Site Types differed in habitat characteristics and/or selected taxa of epigeal predatory arthropods collected; if differences in abundance and/or diversity of gnaphosid spiders, predatory carabid beetles and the single scorpion species collected existed among the Site Types; whether correlates exist between habitat characteristics and the arthropods collected; if any taxa of the selected arthropods could be useful in assessing and monitoring Ozark glade habitats; and whether the methods used are desirable for adoption into glade assessments performed by non-specialists associated with site management.

Habitat Assessment

Degraded Sites

Very few of the analyses of habitat characteristics strayed from showing the Degraded Site Type, consisting of late-sere glades, to be anything other than expected: more wooded (both cedar and non), and with more leaf litter and less exposed rock than other Site Types. Less expected was the overall consensus between analyses that Degraded sites held greater diversity of both Gnaphosidae and Carabidae. The correlation of habitat characteristics to sites via NMDS supported that leaf litter and woody groundcover were a possible factor in the Degraded Site Type being the most speciose.

Degraded Site 2 was one of the most interesting involved in the study. It was the only site where cedar encroachment included *Juniperus ashei*. It was also the only Degraded site where *C. collaris* was observed but that had virtually no history of management. This site presented itself as a mosaic of dense stands of cedars adjoining large patches of exposed sandstone bedrock, interwoven with sparse, relatively early-sere, glade vegetation. This list of characteristics may help explain why the site appeared to be a driver for the higher diversity of the Degraded Site Type, and also why this site shared some species only with Intact and/or Restored Site Types, including three species of *Micaria* (Table 5). Three species of gnaphosid and five species of carabid were also unique to Degraded Site 2 trap collections, more unique species than were collected from any other site. This site also accounted for the vast majority of scorpions collected from the three Degraded sites.

Based on Morisita dissimilarity index figures, the Degraded sites were relatively dissimilar to each other when compared to Intact sites. Interestingly, each Degraded site had a palpable difference in habitat "feel" when visited in person. The Morisita index figures showed Degraded Sites 2 and 4, in general, were very similar in gnaphosid diversity, but very different in carabid diversity. Degraded Site 4 also was similar and had nearly identical Morisita index figures when compared to the closely located Restored Site 5.

Intact Sites

No gnaphosid species were unique to the Intact Site Type, but the most-collected species from Intact sites were the same, and in the same rank order, as the three most abundant species overall (*D. lepidus*, *M. punctata* and *Z. aiken*). The Intact Site Type had small Morisita dissimilarity figures between its sites, indicating relative congruency when compared to other Site Types. Also, more similarities existed between Intact and Restored Site Types than other pairings. Overall, Intact sites had the lowest species richness of Site Types, but also typically the greatest abundance of species catch totals. Intact sites also had the greatest catch totals for species associated with Ozark glades—such as *C. vittatus*, *C. o. vulturina* and juveniles of the tarantula *Aphonopelma hentzi*, the latter two of which were caught nearly exclusively from Intact Site 3, which was by far the largest intact glade.

Restored Sites

It is not surprising that detectable differences between Intact and Degraded Site Types occurred for the taxa selected in this study, as well as the habitat characteristics recorded. This probably could have been surmised without trapping. The real story of this work is the apparent ability of taxa collected to show a division of sites within the Restored Site Type. Restored Site 5 was shown to be closer to Degraded than Intact in multiple analyses; conversely Restored Sites 6 & 7 were closer to Intact sites in most analyses. Clues from natural history observations also can be discussed as supporting evidence of restoration success, or lack thereof, such as the collection of glade-associated *C. o. vulturina* at Site 6, or the lack of *M. punctata* and *C. vittatus* at Site 5, which were collected frequently from other Restored and in Intact sites. The discussions below elaborate more on Restored sites and their characteristics.

Diversity Analyses

Shannon-Wiener and Simpson's Diversity Indices

Two key statements can be made about diversity analysis of gnaphosids and predatory carabids via Shannon-Wiener and Simpson's diversity indices. The first is that both indices were largely in agreement, with the Shannon-Wiener index showing slightly lower *p*-values overall. The second is that, in general, Degraded sites produced the highest diversity index figures, Intact sites the least diversity, and Restored sites were typically somewhere in between.

No significant differences were found among the nine glade sites for gnaphosid or carabid diversity index figures when both trapping seasons were combined. However, it is worthy of note that the Shannon-Wiener index had a *p*-value of 0.058.

The story for Site Types differs slightly. Carabids failed to show any significant difference between average indices of Site Types. Gnaphosids, in contrast, had significantly greater average diversity indices in Degraded sites than in Intact sites only when data from both trapping seasons were combined, but the Shannon-Wiener index figures for the 2020 trapping season again came very close to significance (p = 0.051).

Morisita Dissimilarity Index

The Morisita dissimilarity index figures for gnaphosids as well as those for predatory carabids were overall in agreement that the most dissimilar Site Types were Degraded and Intact. The highest average dissimilarity index figures among the sites of a single Site Type were found for Restored sites in all analyses of gnaphosids and carabids, with the exception of 2018-2019 for which carabid data were most dissimilar among Degraded sites. The high dissimilarity among Restored sites was largely due to Site 5, and among Degraded sites largely due to Site 2. Trimming of the predatory carabid data to include only species with ≥ 10 individuals collected shifted the greatest dissimilarity to between Degraded and Restored Site Types, but this did not occur with trimming the gnaphosid data.

Rarefaction Curves

The rarefaction curves generated from diversity data for each site were compared to each other at the sample size of the site with the smallest trap capture. This sample size will be referred to henceforth as the sample size threshold.

Gnaphosids

When rarefaction curves for gnaphosids were compared at their respective sample size thresholds, stratification became evident. Degraded sites, in general, had greater species richness and less abundance than other Site Types—particularly when compared to Intact sites. Intact sites had three of the four lowest and least speciose curves when crossing the sample size threshold for all rarefaction figures, whereas Degraded sites had at least two of the three steepest and most speciose curves when reaching the sample size threshold. Degraded Site 2 had the steepest trajectory in all untrimmed rarefaction figures, and Degraded Site 4 was the steepest when trimmed to include species with ≥ 10 collected. Degraded Site 9 had the smallest, and therefore limiting, sample size for direct comparison in the 2018-2019 season, but in all other datasets the smallest sample size belonged to Degraded Site 4.

Perhaps the most interesting finding was the separation of trajectories among Restored sites. Restored site curves primarily existed between extremes in species richness exhibited by other Site Types. In all rarefaction figures (Figures 20-31), Restored Site 7 was closer in trajectory and sample size to Intact sites than to Degraded, or even to other Restored sites. The Restored Site 6 trajectory was more like those for Degraded sites, but had a sample size more indicative of Intact sites. Restored Site 5 had a curve closer in appearance to those for Degraded sites and Restored Site 6, but had a sample size closer to those of Degraded sites. However, for the 2018-2019 trapping season, Site 5 had the median sample size of all nine sites.

When data were trimmed to include only species with ≥ 10 collected, differentiation of curves into two groups was seen at the sample size threshold, as well as among sample sizes (Figures 23 & 25). The first group (Group 1) consisted of all Degraded sites plus Restored Site 5. Members of Group 1 all reached the sample size threshold with more speciose curves than did members of the second group (Group 2), which consisted of Restored Site 7 and all Intact sites. Group 1 may have had higher richness than Group 2, but also had distinctly lower abundance. The Site 6 curve was not placed in either group, as its trajectory more closely matched those of Group 1 members, but it also had a sample size that more closely matched Group 2.

When data collected from April-August and combining seasons were considered (Figure 24), very little difference was observed from the untrimmed data from combined seasons. This was somewhat expected, as relatively few gnaphosids were collected outside of April to August.

It could be said that, based on gnaphosid rarefaction curves generated from this study, regardless of how data were examined, all rarefaction figures supported differentiation between Degraded and Intact Site Types. It also could be argued, based on groupings observed in gnaphosid curves, that Restored Site 5 was more like the Degraded sites; that Restored Site 7 was more like the Intact sites, and that Restored Site 6 fell somewhere between Sites 5 and 7.

The apparent division of Restored sites into those that were closer to Intact or Degraded Site Types could be due to a number of factors. Restored sites may be showing a gradient of restoration success, at least when compared relatively to other Site Types, with Site 7 achieving the most similarity to Intact sites, and Site 5 the least. The gradient of success may be due to a number of factors including time since restoration effort, intensity of restoration effort, or the state of the glade before restoration was initiated. Another possibility is that designation of the Restored Site Type based on management history and apparent successional attributes was flawed. Regardless of Site Type designation, Restored Site 5 having a northerly slope, and being burned earlier in the same year as initial trapping began—a timespan far closer to initial trapping than other Restored sites—may have impacted the results. The separation of Restored Sites 6 and 7 from each other is likely explained by differences in habitat characteristics (Table 1) as nearly all the characteristics contrasted between sites, including direction of slope and presence of *C. collaris*.

Carabids

Predatory carabid rarefaction curves lacked clear stratification by Site Type among trajectories. However, a trend toward lower species richness could be seen across figures for Intact site curves when compared to other Site Types. Also, in agreement with gnaphosid rarefaction figures, was a close grouping of Restored Site 7 to Intact sites across all rarefaction figures.

Unlike gnaphosid rarefaction curves, trimming the predatory carabid data to include only species with ≥ 10 collected did not differentiate curves into distinct groups of trajectories. Grouping of sites based on sample size can be seen in (Figure 31), but this grouping was a product of the small sample sizes generated for Sites 3, 7, 9 & 10 in April-August 2020 (Figure 27) rather than from trimming the species.

Several things can be said about rarefaction curves for predatory carabid beetles in relation to those discussed for gnaphosids. Sample sizes used to generate carabid curves were approximately half of those for gnaphosids and, in the 2020 trapping season (Figure 27), which included April-August only, were several times lower than for gnaphosids over the same period. In fact, only four sites generated sample sizes of predatory carabids great enough to create clearly differentiated curves in 2020. Degraded Site 9 was the most speciose site at the sample size threshold for all discernable rarefaction figures (the limiting sample size for site comparison was too small to interpret from April-August 2020), and had the smallest sample sizes in April-August collections. The smallest sample size was Restored Site 7 in all other analyses.

Non-metric Multidimensional Scaling

In relation to Site Type differences, results of Non-metric Multidimensional Scaling (NMDS) were generally in agreement with diversity indices and rarefaction results. All NMDS ordinations showed clear distinction between the plotted position of Site Type centroids. They also showed the greatest distance between centroids of Degraded and Intact Site Types; indicating that these Site Types had the most differences between each other in diversity of taxa collected. There were rare instances where individual Degraded sites were plotted closer to Intact sites than the centroid for their own Site Type, but only in predatory carabid data from 2020 (Site

9; Figures 40 & 44), which had a small sample size, and 2018-2019 data trimmed to species with ≥ 10 individuals collected (Site 2; Figure 42).

For all ordinations, Restored Site Type centroids were plotted between those of the other two Site Types on the NMDS1 axis, which is the axis that includes the compressed dimensions of the most influential diversity variables analyzed. However, the Restored sites were clearly split between Intact sites (Sites 6 & 7), and Restored Site 5 was more like Degraded sites—particularly Degraded Site 4, to which it was in close proximity. Had Site 5 been classified as Degraded instead of Restored, the mean (equivalent of centroid for two sites) of the Remaining Restored sites would have been closer to Intact sites than was the centroid.

The separation of the Restored sites supports the idea that Sites 6 & 7 have been more successful (closer to Intact sites) glade restorations than has Site 5, or possibly that Site Type classification based on management history was not appropriate in this instance, and that Site 5 should have still been considered Degraded despite cedar removal and burning.

Overlay of the significantly correlated habitat characteristics (including Daubenmire cover class ratings) showed that trimming gnaphosid data increased the number of habitat characteristics with significant correlation to ordination points. It also showed that gnaphosids were influenced by the Leaf Litter groundcover category no matter the dataset, and that beyond this category, slope and Exposed Rock were of significance more consistently than other habitat characteristics. Interestingly enough, Leaf Litter and Exposed Rock were also found significantly different between Site Types in Kruskal-Wallis analysis of Daubenmire cover classes between Site Types.

The results of gnaphosid NMDS suggested that the more leaf litter present as groundcover at a site, the more like a Degraded site it was, and the more Exposed Rock a site had, the more like an Intact site it was. Because gnaphosid diversity has been positively correlated to increasing depth, complexity and composition of leaf litter (Uetz, 1991; Hamilton, 2015), a lack of leaf litter may generally explain why fewer gnaphosid species were collected from Intact sites, as well as at Restored Sites 6 and 7.

The Woody groundcover category was significant throughout all NMDS analyses for predatory carabids, and slope nearly was, with the 2020 trapping season being an exception. Leaf Litter was significant for combined trapping seasons of predatory carabids (as it was with gnaphosids), and presence of *C. collaris* appeared as significant in 2018-2019. This was the only analysis to show these lizards as having a significant impact at $\alpha = 0.05$; but if thresholds were relaxed to 0.10, other 2018-2019 datasets (carabid and gnaphosid) would also be seen as significant (Tables 20, 23, 27 & 28). This may indicate an influence of these lizards on trap catch, which would not at all be surprising as they are considered one of the top entomophagous glade predators (Ostman, 2007).

The number of significant habitat characteristics did not increase when predatory carabid species were trimmed from the ordination, as had been seen with gnaphosids. It is also worthy of note that the 2020 trapping season, which did not produce large enough sample sizes to create meaningful rarefaction curves, did produce NMDS site plots that were, in general, the same pattern as others made from predatory carabid catches.

It seems clear, based on NMDS analyses, that woody vegetation, the leaf litter it produces, and the direction of glade slope are all influential habitat characteristics for species assemblages of predatory carabids and gnaphosid spiders.

Natural History Observations

Gnaphosids

The display of gnaphosids by species, sex and month collected (Table 4) provides valuable information on this spider family that expands the information reported from Arkansas by Heiss and Allen (1986) some 35 years prior to this study.

The species *Drassyllus nannellus* had one female collected in 2018-2019, and a single male in 2020. This species was not known to be reported previously from Arkansas. However, Platnick and Shadab (1982) reported on specimens of *D. nannellus* collected from "brushy prairie" in Northern Missouri during months similar to collections during this study (summer), where it was collected from Degraded Site 2 and Intact Site 10—two sites that had very different habitat characteristics recorded, but that also each had portions that could be considered "brushy" grasslands.

The species *Zelotes lymnophilus* also appears to be a new record for Arkansas, with the closest recorded specimens being from the Hill Country of Texas and Northern Georgia (Platnick and Shadab, 1983). This species was captured between May and August of both trapping seasons and was one of five gnaphosids that were collected from every site.

Only males of *Drassodes gosiutus* were captured in the 2018-2019 season, and only females were captured in the 2020 trapping season. Females were caught in months congruent with reports in the literature (Heiss and Allen, 1986); however, records of male *D. gosiutus,* or their activity period, were not found to be reported previously for the state.

Additionally, collection for three gnaphosid species, *Gnaphosa sericata, Drassyllus creolus* and *Zelotes aiken,* occurred outside of the temporal range of collection reported by Heiss and Allen (1986) for Arkansas.

Of the 19 species for which both sexes were collected in the 2018-2019 season, 17 of them had at least one month where only one sex was present; this was also true for 11 of 17 species in 2020. Months where both sexes were collected were often preceded by one or more months of a single sex (more typically male), and they were followed by a month or more of the opposite sex. These results suggest an offset in activity period for mature males and females for the majority of gnaphosid species captured (Table 4).

Carabids

None of the carabids collected were new to Arkansas or expand on known species ranges. Study of carabids in Arkansas has long been a pursuit of entomologists from both the state and region. One of the more notable of these was W. H. Whitcomb, an Arkansas entomologist about whom many stories have been told (Ruberson, 2019), and for whom the carabid species *Cyclotrachelus whitcombi* was named by Freitag. Interestingly enough, *Cyclotrachelus whitcombi* was collected during this study, but only from Degraded Site 2, and another species from the genus (*C. seximpressus*) was not collected at Site 2, whereas it was found at nearly all other sites. Notes on *Cicindelidia obsoleta vulturina*, and the two species of *Pasimachus* collected are discussed with glade associations.

Scorpions

The sole species of scorpion, *Centruroides vittatus*, was present in trap collections from April-December, which was a range of dates that persisted later into the fall/winter than expected. This extended activity period might be possible as a product of thermal storage provided by the sparsely covered bedrock found within glades.

The other curious outcome from trapping was a lack of scorpions at Degraded Site 4 and Restored Site 5, which were sites in close proximity to each other, and presumably had suitable habitat for scorpions.

Ecological Equivalencies

In several instances, some taxa appeared to have ecological equivalents and reciprocal appearance at sites. Restored Site 7 had the most gnaphosids collected, primarily *M. punctata*, as well as the second-most *C. vittatus* when compared to other sites. Site 7 also had the fewest predatory carabids. Restored Site 6 had the most predatory carabids and *C. vittatus*, and one of the smallest collections (11) of *M. punctata*. The abundance of this ant-mimicking species (*M. punctata*) at Site 7 and rarity at Site 6, and the reverse for predatory carabids, may have been due to the relative numbers of ants at these site as well, influencing the numbers of gnaphosids and predatory carabids present.

Another possible equivalency occurred between two members of the genus *Zelotes*, *Z. hentzi* and *Z. aiken*. Collection totals ≥ 9 were recorded for *Z. hentzi* from Restored Site 5, and from each of the Degraded sites, whereas < 9 were collected from both other Restored sites, and each of the Intact sites. The potentially equivalent species, *Z. aiken*, had ≥ 20 individuals collected from Restored Sites 6 & 7 as well as each Intact site, and ≤ 10 collected from Restored Site 5 and each of the Degraded Sites.

Another point to highlight when discussing possible equivalencies in activity-abundance for these groups, is that gnaphosid numbers sharply dropped in average catch per trap in August (Figure 10), which was the time when carabids were peaking (Figure 15). However, no clear patterns emerged for equivalencies, and such explanations would require explicit testing.

Glade Associations

Gnaphosids

In considering the glade associations among gnaphosids, the first consideration is their abundance in trap collections. Five species of gnaphosids had at least twice as many individuals collected from Intact sites as were from Degraded sites. Those were *Drassyllus creolus, D. lepidus, Gnaphosa sericata, Micaria punctata* and *Zelotes aiken*. Excluding *D. creolus*, the other four were also the four most collected species during this study. It is also worthy of note that *G. sericata* is the only species found to have been explicitly reported from a glade (Meyer, 1937). However, Heiss and Allen (1986) presented *G. sericata* and *D. lepidus* as being common from many open habitats in Arkansas, including open fields and monoculture crops. That finding indicates they may be numerous in trap catch from Intact sites more because of open canopy than because of a particular glade affinity. Also lacking in evidence of glade association, *D. creolus* has been reported from leaf litter, moss and under rocks from hardwood forests in Arkansas (Heiss and Allen, 1986).

Two of the most collected species remain relevant in the discussion of glade association. *Zelotes aiken* was collected two to four times more from both Restored and Intact Site Types when compared to Degraded sites (Table 5). When combined with reports from Heiss and Allen (1986) that this species was collected from prairies, *Z. aiken* may be of interest as a glade associate, given that these habitats share many qualities. It should be noted that both male and female *Z. aiken* are difficult to discern from *Z. hentzi*, with females requiring dissection of the epigynum to positively distinguish between the two species. This fact makes *Z. aiken* undesirable for quick species identification. Finally, *M. punctata* is worthy of discussion as it was overwhelmingly captured from Intact and Restored sites, and also has been reported from "bluestem prairies" (Platnick and Shadab, 1988), although it has also been captured in other open habitat types such as beaches and old fields. Unfortunately, *M. punctata* is also an identification challenge, as most specimens are between 2-3 mm in length.

Although no previous reports of *Zelotes lymnophilus* were found to exist for Arkansas, this species was captured at all sites, which could indicate an affinity for glades of all stages of succession, or this species could be equally present in surrounding forests as well. More would need to be known about presence of this species in surrounding forests to make statements on glade affinity. It is also possible that *D. nannellus*, an apparent new record for Arkansas, could be associated with glades, as specimens collected in Missouri were from glade-like habitat.

Of the remaining gnaphosid species collected, nearly all were found to be previously documented in Arkansas, and notes on their collection locations compiled by Heiss and Allen (1986) eliminate most of them from the glade discussion, with reports of collection locations such as “from the Delta Delta Delta sorority at the University of Arkansas.” Comparisons between this study and gnaphosids collected from forested areas of the Arkansas Ozarks by Hamilton (2015), showed 11 of the 15 species overlapped between studies. However, none of the top three most collected gnaphosids in this study, or any of the genus *Micaria*, were reported by from her work.

The five species of *Micaria,* a genus of ant-mimicking gnaphosid, were of particular interest in this study. Arkansas species of *Micaria* were not discussed by Heiss and Allen (1986) because the taxon (previously in the family Clubionidae) was mentioned as under revision, presumably by Platnick and Shadab, who published species descriptions and keys to them in 1988. However, all of the *Micaria* species collected in this study had previously been reported from the state (Dorris, 1985; Platnick and Shadab, 1988). One species, *M. longipes,* was only captured in fall and winter months during this study and thus only captured in 2018-2019. It has been reportedly collected from an array of habitat types including prairies and oak-pine barrens, which in general are habitats similar to glades. Catch totals for *M. longipes* were greater from Intact and Restored sites, with Restored Site 5 producing 22 specimens, 50% of the total catch (Table 5). It is curious that this species was collected in such high numbers at Site 5 (relative to other sites). Degraded Site 4 was in close proximity to Site 5, yet produced only two *M. longipes* in collections. It is possible that the high catch totals at Site 5 could have had something to do with more recent burning than other sites when trapping began. It would have been interesting to see if collection totals remained high at Site 5 in preceding years.

The other four species of *Micaria* collected have all been reported primarily from open habitats, including pine-dunes (*M. longispina* & *M. seminola*); sandy shores and beaches (*M. punctata* & *M. seminola*); abandoned fields, pastures and oak-savannas (*M. punctata*); and, perhaps most relevant to this study, from prairies in Arkansas (*M. laticeps* & *M. longispina*; Platnick and Shadab, 1988).

Much like *M. longipes*, the greatest numbers of *M. punctata* were also captured from a Restored site (Site 7), but in contrast *M. punctata* were mainly caught from April-June. This was one of the most captured species of the study, but only two individuals were captured from Degraded sites; both from Degraded Site 2, which had areas of early-sere glade vegetation.

Carabids

Five species of predatory carabids were at least twice as numerous in samples from Intact sites than those from Degraded sites (*Calosoma sayi*, *Chlaenius tomentosus*, *Cicindelidia obsoleta vulturina*, *Scarites subterraneus* and *Tetracha virginica*). Four of these species have wide ranges and seem unlikely to be true glade associates, including *T. virginica*, which was the most captured of all predatory carabids. The fifth species, *C. o. vulturina* is more commonly found in grasslands of Central Texas and Southern Oklahoma, but disjunct populations in

Arkansas and Missouri are strongly, if not entirely, associated with glades (Mawdsley, 2009), including in areas surrounding Calico Rock, Arkansas (Mawdsley, 2009; MacRae and Brown, 2011). This species was almost exclusively found at the largest Intact site trapped, but one individual was also collected from Restored Site 6, which produced the richest, as well as most abundant collections of carabid diversity when both trapping seasons were considered (Table 13).

Worthy of mention were differences in collection totals between two of the largest and most charismatic species of beetles collected in this study. The two species of *Pasimachus* collected were nearly indistinguishable from each other, as they have quite subtle differences in morphology. The first one of these large black and blue beetles encountered by me was in a woodland near the study sites, when I mistakenly identified it from a distance by its pronounced mandibles as a lucanid beetle. That particular *Pasimachus* speeding through the underbrush eluded me and for a time was “the one that got away.” I was very excited when the first one showed up in my traps as I was unsure at the time if these beetles would be prone to capture with unbaited ramp-style pitfalls; it ended up becoming the second most captured predatory carabid of the study.

Among the *Pasimachus* species collected, *P. depressus* far outnumbered *P. elongatus* (72 vs. 4), particularly in collection totals for Degraded sites and Restored Site 5. This is another example of Site 5 seeming out of place with other Restored sites. The less-collected *P. elongatus* was only caught in Intact and Restored sites. Natural history notes on *P. elongatus* from Cress and Lawson (1971) were recorded from Wyoming grasslands, which could arguably be a habitat physically similar to areas within a glade. However, they also reported that larvae overwintered more than 30 cm deep in the soil, which, if correct, would be incompatible with the thin soil layers of most Ozark glades. It is possible the larvae overwinter in habitats near the glades with deeper soils. Alternatively, cold winters in Wyoming may require deeper overwintering sites than in the milder Ozarks.

A final note about carabids comes from the genus *Chlaenius*. Six species of *Chlaenius* were identifiable to species level, but three specimens of one taxon (presumably) could not be identified to species level. The three specimens of this taxon were given morphospecies designation due to an inability to find a key with an appropriate set of characters, whereas other taxa were reported as morphospecies because of a complexity of identification due to size or species similarities. The three specimens of *Chlaenius* sp. deserve further investigation into proper identification, as these specimens were large enough to identify with a dissecting scope, and are classified in a genus with low diversity, compared to other carabids—and they were all collected from Intact Site 3.

Scorpions

Scorpions (*C. vittatus*), in this study, were far more prevalent in trap catches from Intact and Restored sites than Degraded, and indeed could be associated with glades as Bell (2007) suggested. This could be particularly true in colder months when glades receive more insolation, and surface bedrock remains warm longer than substrates in surrounding forest. In December of 2018, *C. vittatus* were still being caught, and seen actively hunting at night along bedrock. Also, due to the nature of pitfall trapping, it could be that activity of scorpions is increased instead of abundance in the more open Site Types because of this heating effect.

The increased woody vegetation of Degraded sites may also have allowed for more habitat structure that was not sampled effectively for *C. vittatus* via pitfalls; limiting their catch in these Site Types. On one instance, an individual was spotted at eye level among the bark of a cedar at Site 2. This was the only time I have personally seen this species of “bark” scorpion anywhere above knee-high off of the ground on vegetation, although, during preliminary glade arthropod trapping via malaise trap (typically targeting flying insects) one *C. vittatus* was found in a collection container roughly 2 m off of the ground after only a few days of trapping, indicating scorpions reaching this level was probably not a rare occurrence. Degraded Site 2 also had numbers of scorpions more equally comparable to other Site Types than its own. Again, Site 2 could be considered a Degraded site with qualities, in some areas, similar to that of an Intact site.

Utility of Study Components

One of the aims of this study was to look at different ways to generate and analyze data on arthropods in glades. The idea behind including material in this dissertation on different groundcover estimation methods, trap designs, analysis methods, included taxa, datasets used, or even different individuals grading photos was to present results from an array of methods that could be of interest to specific glade management situations.

Groundcover Estimation

Several takeaways from groundcover estimation worthy of mention were generated during this study. The first was an overall agreement between the modified Step-Point and Photographic Groundcover Estimation methods in describing compositions of glade sites with groundcover categories. Both methods showed Degraded sites had significantly more Grasslike and Cedar groundcover than Intact sites, that Cedar was also significantly more prevalent at Degraded than Restored sites, and that Exposed Rock was more prevalent at Intact sites than Degraded. However, the methods did not produce completely similar descriptions. The modified Step-Point method showed the Woody category to be significantly different between Degraded and Intact sites. The photo estimations showed the Moss category to be significantly less for Restored sites than the other Site Types when limits were relaxed to $\alpha = 0.10$ ($p < 0.09$).

The photo estimation involved the use of three more categories of groundcover (Bare Soil, Dead Wood, Leaf Litter) than did the Step-Point method, and of those, the Leaf Litter category was found to be significantly lower in both Restored and Intact Site Types than in Degraded sites. Leaf Litter was also an important category in NMDS analysis.

Cedars are not typically seen as groundcover, but the vast majority of cedars encountered on glades during this study were either young, or bushing with limbs near the ground, and thus able to be detected by each groundcover estimation method used. However, this was not the case at Site 9, where tall straight cedars were lacking bushy lower limbs and therefore not detected by either method effectively. It should be noted that, even without cedars being effectively detected as groundcover at Degraded Site 9, the Degraded Site Type still had more cedar than either of the other Site Types—meaning differences noted in cedar presence between Degraded sites and other Site Types were probably more pronounced in the field than were reflected in groundcover estimations.

The numbers of detections made with the modified Step-Point method for each groundcover category were not different between years. This indicates that somewhat similar groundcover compositions existed between the two trapping seasons—each of which had randomly chosen trap grid deployment locations within a glade site. It was also shown that the categories of Grasslike and Forbs were statistically more frequent overall when compared to other groundcover categories. This supports descriptions of glades as grasslands with mixed forbs.

The inter-rater reliability of photographic groundcover estimation between the two photo graders showed mixed results. Graders often agreed on the absence of a category, but total agreement in class assignment occurred at somewhat moderate levels for most of the groundcover categories. Total agreement was particularly low for the frequently graded groundcover categories of Grasslike (41%) and Forbs (38%; Appendix D). However, class assignments were congruent 88-100% of the time when a difference in one cover class rating was considered an agreement. This may be a more appropriate way to measure inter-rater reliability when using Daubenmire cover classes, because the classes are numerically discrete, whereas the placement into one or another class was differentiated by estimations of as little as one percentage point.

LST vs CST Construction

The two trap styles used in this study (LST and CST) each had advantages and disadvantages associated with their use and maintenance, as well as differences in species collected. Lunchbox Style Traps (LST) required less site disturbance for leveling (when necessary) and impacted less vegetation during deployment due to a smaller footprint than that of Cone Style Traps (CST). Maintenance requirements were also less for LST as they required no repainting of ramp surfaces, which was required both during and between trapping seasons for many CST, particularly those exposed to direct sunlight. Collection container lids of LST were also more resistant to deterioration from sunlight than those used on CST.

The reviews are not entirely in favor of the LST, however, as collection containers of LST were more prone to deterioration than CST, and many LST collection containers and lids became brittle to the point of requiring replacement during the second trapping season. Should LST be used, collection containers and lids of those in direct sunlight should be considered to have reached the end of their duty-cycle after one trapping season. The other noted advantage of

CST was a seemingly higher resistance to toppling from wind; however, no data were recorded to affirm this.

If only one trap style were chosen for this study based on ease of construction and deployment, as well as observed performance in field conditions, the LST would be the trap style of choice.

LST vs CST Capture

Both trap styles were capable of recording a satisfactory range of gnaphosid diversity for analysis. Adult gnaphosids were captured in greater overall numbers, and with greater overall species richness in LST. Based on these results, it could be argued that the LST outperformed the CST in gnaphosid capture. However, 24 of the 32 species were captured by both trap styles, and those captured by only a single trap style were species with only one or two individuals collected. It should also be noted that three species of *Drassyllus*, including the most collected gnaphosid overall, *D. lepidus*, had higher catch totals in CST.

Predatory carabids were also captured in greater numbers and with greater richness using LST and, again, 24 species were shared between trap types, but this figure represented a smaller percentage of species shared between trap types than with gnaphosids. Again, only species with one or two collected individuals were captured by a single trap style. The only species that was overwhelmingly captured by CST (18 vs. 5) was *Galerita janus*, a species common in many habitats, including urban areas, and is believed to be of little relevance to glades. In contrast, the one species of predatory carabid captured that has been mentioned as being associated with glades, *C. o. vulturina*, was overwhelmingly collected by LST (13 vs. 1). Based on these findings, it could be suggested that, unless *G. janus* is the targeted species, LST would be preferred over CST for predatory carabid sampling in Ozark glades.

As opposed to results generated from gnaphosid and predatory carabid trapping, *Centruroides vittatus*, the most commonly collected epigeal arthropod predator recorded for this study, was captured 119 times in LST versus 323 in CST. This would support CST being the preferred trap style if this, the only scorpion species found in the Ozarks, is targeted. This style of trap may also prove effective for scorpions in other habitats, or species inhabiting other regions.

Both vertebrate (salamanders, shrews, toads and lizards) and invertebrate bycatch was somewhat similar between trap styles (data not presented), although the only note made on this topic was that CST were more prone to capture wasps from the genus *Polistes* during the spring, presumably as the wasps were searching for nesting sites.

Trapping Procedure

Several trapping procedure notes were worthy of reflection. The 250ml of 100% PG preservative used in each trap preserved specimens well enough for identification, even after four weeks in the field, in often extremely hot conditions. Collection containers for each trap style had enough capacity for 4-week collection periods, although more frequent collection may have prevented some trap loss to animals due to the attractiveness of carrion (including vertebrate bycatch). Alternation of trap styles within the grid allowed for easier orientation to the grid during collection and deployment. Finally, 108 traps deployed among nine sites proved to be right at the limit of what could be managed by myself within a 24-h period, when both collecting samples and resetting traps.

Analytical Procedure—Trimming Data

The main purpose of trimming data was to see how little effort could be expended, in both trapping effort and specimen identification, while still retaining results robust enough to be meaningful for habitat assessment.

Habitat managers would seldom have two full years to invest in habitat assessment; therefore, data were examined for one full year (2018-2019 trapping season), a single summer (2020 trapping season), summers of both trapping seasons combined, and the entirety of both trapping seasons combined. Datasets for April-August collections were explored to determine whether only trapping during the summer, when land managers typically have a larger pool of seasonal workers at their disposal, would yield results comparable to trapping year-round.

Decidedly, trimming data to only April-August collections created a snapshot of gnaphosid diversity reflective of trapping throughout the year, as these were months of peak trap capture and diversity for this family. The outcome of temporally trimming data was not nearly as clear for predatory carabids, with the exception of analyses by NMDS, and April-August datasets for carabids were considered, in general, less meaningful than trapping throughout the calendar year, for multiple seasons.

Arthropod identification and the use of dichotomous keys may not be skills held by the average seasonal worker. Even fewer have experience identifying spiders to species. This means that training would likely be needed for anyone tasked with identification of collected specimens. By trimming the species counts to include only taxa with ≥ 10 individuals collected, much of the identification burden disappears. This is because the most time-consuming part of identification is often figuring out the proper identification for the first specimen of a taxon. From that point on, other specimens belonging to the previously identified taxon are much easier to identify. However, this argument does not acknowledge the difficulty of identifying small specimens (e.g., many *Micaria*), so it might not be possible with limited training to recognize whether a number collected represented one species or small numbers of similar taxa.

Trimming gnaphosid data to include only species with ≥ 10 individuals retained more than 95% of gnaphosids and 90% of carabids. Species trimming not only seemed to maintain a representation of analysis results when all species of gnaphosids were included, but also seemed to increase the stratification of sites (by Site Type) in rarefaction curve figures. Results of NMDS analysis revealed that, trimming species to those with ≥ 10 individuals collected, showed generally the same trends in site depictions as untrimmed data. This was even true for the small sample sizes of the 2020 predatory carabid data (which, when trimmed, retained only five species). Trimming of species also had little impact on rank of site or Site Type diversity index figures, but obviously decreased all figures related to diversity indices, as species richness was removed from analyses.

Scorpions, Spiders or Beetles?

The simplest to monitor of the three taxa discussed would be *C. vittatus,* as there is no identification burden, and large collection numbers (the most collected of any species analyzed), correlated to sites with attributes similar to Intact sites in this study. Peak numbers of collection occurred between July and October, which are typically the driest months in the glades (personal observation). Drawbacks to using scorpions would include that *C. vittatus* is known to inhabit a wide range of habitat types as well as glades; or that they could be completely absent from site collections—as happened for Degraded Site 4 and Restored Site 5 in this study. Of course, analyses would also be limited to comparisons of trap catch numbers alone and not diversity, which limits assessment techniques.

Gnaphosids are believed to have been more indicative of differences between Site Types than other taxa analyzed from this study. Gnaphosid richness was less among collections from Intact sites than from sites with attributes more like Degraded sites—such as leaf litter. One drawback to analyzing gnaphosid diversity figures for glade assessment is that low diversity may be an indicator that a glade is dissimilar to encroaching forest, but it also may be that diversity is low for other reasons. For example, a glade invaded by introduced plant species, such as sericea (*Lespedeza cuneata*), or one converted into a parking lot, would no doubt both show less diversity than surrounding forests, but those are certainly not glades. Evidence from this study reveals that appearance of the gnaphosid genus *Micaria*, may actually be counter to the trend towards lower gnaphosid diversity seen in early-sere glades, and holds particular promise for Ozark glade assessment. Large numbers of *M. punctata* were captured, particularly from Restored Site 7, and Intact Sites 3 and 8, and they were not collected from Degraded Sites 4 or 9. The only Degraded site where *Micaria* species other than *M. longipes* were found, Site 2, was a unique mosaic of habitat characters that included areas of habitat characteristics similar to Intact sites. It should also be mentioned that *Drassyllus* (the most diverse gnaphosid genus collected), as well as the genus *Zelotes*, may also be of interest as described with glade affinity.

Predatory carabids as a whole were considered the least effective indicators of differences between Site Types in this research, particularly when trimming already small numbers, such as the 2020 (April-August) trapping season. However, large numbers of *C. o. vulturina* collected at the largest Intact site (Site 3), strengthen hypotheses of this species being a glade obligate in the Ozarks. Detection of this species also occurred at third-largest Restored Site 6, which, in reality was larger in overall glade area than was recorded for Site Area in Table 1. This is because higher elevations of the glade were calcareous instead of sandstone, and therefore excluded from the Site Area (Table 1; Appendix B). This is mentioned here to preface that Arkansas and Missouri populations of *C. o. vulturina* may require large glades for existence, particularly those proximal to habitats with soils deep enough in areas for larvae to overwinter. Soils of this depth are seemingly uncharacteristic of Ozark glades. It could also be argued that the individual specimen captured at Restored Site 6 may be indicative of a success for restoration efforts at this location.

This study examined only carabid beetles considered predatory. It must be stated that, had all carabids collected been analyzed, regardless of feeding habit, this family may have had more to offer from analyses. The number of carabids collected would have increased more than two-fold, which may have strengthened diversity figures and possibly improved the accuracy of habitat representation in collections.

CONCLUSIONS

One could argue that the more taxa included in a diversity analysis, the more representative of the community it is; however, there is a necessary tradeoff between resolution and effort that must be addressed by those studying vast arthropod diversity. Groups of arthropod taxa must be selected for use in habitat assessment, particularly when time is limited. Differences between Degraded and Intact Site Types in this research were shown for collections of all three selected taxa, Gnaphosidae, predatory Carabidae and *C. vittatus*. Therefore, all showed potential for use in habitat assessment—at least for the sites studied. Inferences concerning other glades, particularly outside of sandstone glades of the Ozarks, should be cautioned, as analyses of species collected from sites in this study are only truly applicable to the mentioned sites. Information reported herein about methods employed for glade assessment, such as groundcover estimation, epigeal arthropod trapping, and diversity analyses, can be applied to comparative glade study elsewhere.

Across multiple analyses, Degraded and Intact Site Types significantly differed in habitat characteristics, particularly woody vegetation, leaf litter and primary direction of glade slope. For the most part, the designation of Site Types and assignment of sites to the Types appears accurate. However, some discrepancies were seen. Multiple analyses of habitat characteristics and diversity of selected taxa have hinted to the restoration status of Site 5, at the time of this study, as dubious at best. This site will require more time and/or management effort to achieve restoration success. Restored Sites 6 and 7 seemed to live up to their Site Type designation and had a far greater resemblance to Intact sites studied than to Degraded. Site 7 was probably the most like an Intact site in this study, as it had groundcover, gnaphosid, predatory carabid, and scorpion results that all were closer to Intact sites than any other Restored site. Interestingly enough, it was also the only site studied where a reintroduction of *C. collaris* was successful.

Degraded Site 2 was also considered to have some qualities of an Intact site, both in habitat characteristics and species diversity.

Overall, it can be determined based on the results of this dissertation work that measurable differences existed in habitat characteristics and selected taxa of epigeal predatory arthropods between Intact and Degraded sites. Restored sites were split between Sites 6 and 7 seemingly closer to restoration, and Site 5 farther from it. Correlations between selected taxa and recorded habitat characteristics were seen among NMDS analyses. It could be said that pitfall trap collections of gnaphosids, predatory carabids, and scorpions all show potential for use in monitoring glade habitats; the greatest utility in this capacity being seen among gnaphosids, with particular interest paid to the genus *Micaria*. It can also be said that many, if not all, of the methods employed in this study have applicability for habitat assessment by non-specialists with training and practice in their use.

If this study were to be expanded, identification and analysis of the remaining carabids collected (that were omitted due to perceived feeding habits), or other families of spiders, or predatory mites, would be valuable directions to proceed. A relative comparison of collections from a glade, to a prairie or open field, could also be of value; especially in further determining glade affinity of taxa.

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TABLES

Table 1. Characteristics of nine study sites in degraded, restored and intact glade types, and presence of the Eastern collared lizard, *Crotophytus collaris*.

Site Type	Site	Site Perimeter (m)	Site Area (m²)	Primary Slope	*Crotophytus collaris* Observed
Degraded	2	1,010	27,071	North	Yes
	4	1,751	20,766	North	No
	9	719	22,566	North	No
Restored	5	758	19,882	North	No
	6	2,152	57,628	West	No
	7	1,306	34,856	South	Yes
Intact	3	4,013	196,599	West	Yes
	8	1,134	35,852	South	Yes
	10	2,810	88,344	South	Yes

Table 2. Occurrence of groundcover categories from sampling via modified Step-Point method. Figures presented represent the number of occurrences out of 200 sample points, and averages of the three sites per site type. The Woody category includes all woody species of flora except for cedars (*Juniperus virginiana* & *J. ashei*).

Site Type	Site	Year	Grasslike	Forbs	Cedar	Woody	Moss	Exposed Rock
Degraded	Site 2	2019	147	31	89	18	102	15
		2020	138	24	68	61	76	16
	Site 4	2019	168	88	67	31	7	0
		2020	169	30	44	67	24	10
	Site 9	2019	188	59	24	37	19	0
		2020	157	36	14	21	14	24
Degraded Site Averages			161	45	51	39	40	11
Restored	Site 5	2019	115	112	9	37	8	2
		2020	139	72	3	47	6	17
	Site 6	2019	133	87	1	3	0	29
		2020	146	99	4	4	0	30
	Site 7	2019	139	73	0	11	35	18
		2020	151	33	0	15	6	34
Restored Site Averages			137	79	3	20	9	22
Intact	Site 3	2019	97	95	1	12	20	45
		2020	72	29	0	4	64	64
	Site 8	2019	143	67	11	13	33	25
		2020	125	76	19	15	8	65
	Site 10	2019	113	65	2	10	37	39
		2020	109	65	4	0	61	34
Intact Site Averages			110	66	6	9	37	45

Table 3. Average Daubenmire cover class figures for groundcover categories in three sites per three site types, and averages of each category per site type. Figures were recorded from digital photographs of $1 m^2$ quadrats (12 per site per year). Woody category includes all woody species except cedars (*Juniperus virginiana* & *J. ashei*).

Site Type	Site	Year	Grasslike	Forbs	Cedar	Woody	Moss	Exposed Rock	Bare Soil	Leaf Litter	Dead Wood
Degraded	2	2019	2.5	1.3	0.3	0.1	2.5	1.0	1.3	0.8	0.9
		2020	2.3	1.3	0.4	0.8	1.8	0.6	0.4	0.6	0.4
	4	2019	3.3	1.9	0.2	0.4	0.7	0.3	0.6	0.9	0.5
		2020	3.1	1.7	0.3	1.3	0.9	0.8	0.3	0.8	0.5
	9	2019	3.3	2.0	0	0.6	0.3	0.1	0.2	0.9	1.3
		2020	3.1	1.7	0	0.8	0.6	0.1	0.3	1.1	0.8
Degraded Site Averages			2.9	1.7	0.2	0.7	1.1	0.5	0.5	0.9	0.7
Restored	5	2019	2.4	2.4	0	0.8	0.2	1.4	0.9	0.8	0.6
		2020	2.8	1.8	0	1.3	0.3	1.0	0.6	0.3	0.8
	6	2019	1.3	1.9	0	0	0	2.7	1.4	0	1.3
		2020	2.4	2.4	0	0	0	2.5	1.1	0	0.7
	7	2019	2.4	2.0	0	0	1.2	1.1	1.3	0	0.4
		2020	2.8	1.6	0	0.8	0.5	1.1	1.4	0	0.3
Restored Site Averages			2.4	2.0	0	0.1	0.4	1.6	1.1	0.2	0.7
Intact	3	2019	2.1	2.1	0.1	0.2	0.9	1.6	1.8	0.4	0.2
		2020	1.6	0.8	0	0	2.3	2.1	0.3	0	0.2
	8	2019	1.9	1.8	0	0	0.9	0.8	1.4	0.7	0.7
		2020	2.2	1.5	0	0.1	0.3	2.4	0.8	0.1	0
	10	2019	2.2	1.9	0	0	1.1	1.5	0.6	0.2	0.8
		2020	1.5	1.7	0	0.2	2.1	2.0	0.4	0.1	0.4
Intact Site Averages			1.9	1.6	0	0.1	1.3	1.7	0.9	0.3	0.4

Table 4. Numbers of male and female gnaphosid species captured in the 2018-2019 and 2020 trapping seasons.

Species	2018-2019 Trapping Season														2020 Trapping Season				
	Sept. 9th	Oct. 6th	Nov. 3rd	Dec. 1st	Jan. 5th	Feb. 2nd	Mar. 1st	Mar. 30th	Apr. 28th	May 25th	June 24th	July 20th	Aug. 17th	Sept. 15th	Apr. 25th	May 23rd	June 20th	July 18th	Aug. 15th
***Calilepis imbecilla* (Keyserling)**											2♂	1♂		1♀				1♂ 1♀	
***Cesonia bilineata* (Hentz)**										1♂									
***Drassodes gosiutus* Chamberlin**			2♂	2♂											2♀				
***Drassyllus aprilinus* (Banks)**							1♂		3♂ 1♀	2♂ 5♀					10♂ 1♀	6♂	3♀	1♀	
***Drassyllus covensis* Exline**																3♂	1♀		
***Drassyllus creolus* Chamberlin & Gertsch**										1♀					1♂	4♂ 1♀	2♂ 6♀	4♀	
***Drassyllus dixinus* Chamberlin**										5♂	22♂ 9♀	3♂ 9♀	1♂ 3♀				31♂ 8♀	9♂ 10♀	1♀
***Drassyllus dromeus* Chamberlin**									1♂	4♂ 2♀	3♂ 3♀					1♂ 1♀	2♀	4♀	
***Drassyllus frigidus* (Banks)**							2♂	3♂								1♀			
***Drassyllus lepidus* (Banks)**										3♂	43♂ 22♀	16♂ 23♀				2♂	35♂ 15♀	25♂ 42♀	1♂ 15♀
***Drassyllus nannellus* Chamberlin & Gertsch**													1♀				1♂		
***Drassyllus rufulus* (Banks)**		8♂ 1♀	3♂ 2♀		2♀														
***Gnaphosa frontinalis* Keyserling**	1♀									2♂ 1♀	5♂ 1♀	2♀					10♂ 1♀	2♂ 3♀	3♂ 4♀
***Gnaphosa sericata* (L. Koch)**	3♂ 3♀										10♂ 2♀	43♂ 10♀	14♂ 8♀	1♂ 6♀			4♂ 1♀	31♂ 7♀	18♂ 11♀

Table 4 (cont.). Numbers of male and female gnaphosid species captured in the 2018-2019 and 2020 trapping seasons.

Species	**2018-2019 Trapping Season**														**2020 Trapping Season**				
	Sept. 9th	Oct. 6th	Nov. 3rd	Dec. 1st	Jan. 5th	Feb. 2nd	Mar. 1st	Mar. 30th	Apr. 28th	May 25th	June 24th	July 20th	Aug. 17th	Sept. 15th	Apr. 25th	May 23rd	June 20th	July 18th	Aug. 15th
***Haplodrassus signifer* (C. L. Koch)**								1♂	3♂	6♂ 1♀	1♀				4♂	6♂	3♂ 3♀		
***Herpyllus ecclesiasticus* Hentz**										1♂	1♂ 1♀	1♂							
***Litopylus temporarius* Chamberlin**																	1♀		
***Micaria laticeps* Emerton**										1♀		1♂ 1♀						2♀	2♀
***Micaria longipes* Emerton**		18♂ 1♀	1♂ 4♀	7♀	1♀									13♂					
***Micaria longispina* Emerton**																	1♂		
***Micaria punctata* Banks**	1♂ 2♀	2♂		1♀	1♂				54♂ 4♀	25♂ 16♀	4♂ 9♀	1♂ 3♀	4♂ 2♀	1♂ 1♀	44♂ 5♀	36♂ 2♀	7♂ 8♀	8♂	
***Micaria seminola* Gertsch**													1♂						
***Nodocion floridanus* (Banks)**																	1♂ 1♀		
***Sergiolus capulatus* (Walckenaer)**																		2♀	
***Sergiolus tennessensis* Chamberlin**																	2♂		
***Synaphosus paludis* Chamberlin & Gertsch**																	1♀		
***Talanites exlineae* (Platnick & Shadab)**					1♀						6♂ 2♀		1♀			1♀	2♂ 1♀		
***Zelotes aiken* (Platnick & Shadab)**	1♂	2♂	6♂				3♂		16♂ 2♀	44♂ 16♀	5♂ 12♀	1♂ 2♀		1♂	12♂	20♂ 7♀	24♂ 26♀	2♀	

Table 4 (cont.). Numbers of male and female gnaphosid species captured in the 2018-2019 and 2020 trapping seasons.

Species	**2018-2019 Trapping Season**														**2020 Trapping Season**				
	Sept. 9th	Oct. 6th	Nov. 3rd	Dec. 1st	Jan. 5th	Feb. 2nd	Mar. 1st	Mar. 30th	Apr. 28th	May 25th	June 24th	July 20th	Aug. 17th	Sept. 15th	Apr. 25th	May 23rd	June 20th	July 18th	Aug. 15th
***Zelotes duplex* Chamberlin**										4♂ 2♀	5♂ 2♀						3♂		1♀
***Zelotes hentzi* Barrows**		1♀	3♂ 1♀					7♂ 5♀	8♂ 8♀	3♂ 4♀	5♂ 3♀		2♀		2♂ 4♀	7♂ 5♀	4♂ 6♀	1♂ 3♀	
***Zelotes laccus* (Barrows)**											3♂ 4♀	1♂					4♂ 1♀	1♂	
***Zelotes lymnophilus* Chamberlin**										3♂	12♂ 3♀	3♂ 4♀				1♀	21♂ 7♀	6♂ 12♀	4♀
TOTAL	11	33	22	10	5	0	6	16	100	150	198	124	37	24	86	104	247	177	60

Table 5. Numbers of each species of gnaphosid, with total numbers of species, genera and individuals collected at each site in 2018-2019 and 2020 trapping seasons.

Species	**Degraded Sites**			**Restored Sites**			**Intact Sites**		
	2	4	9	5	6	7	3	8	10
Calilepis imbecilla **(Keyserling)**		1		4					1
Cesonia bilineata **(Hentz)**	1								
Drassodes gosiutus **Chamberlin**					2	2	2		
Drassyllus aprilinus **(Banks)**	15	7	5	3				3	
Drassyllus covensis **Exline**	3		1						
Drassyllus creolus **Chamberlin & Gertsch**	1	1	2		1	4	3	6	1
Drassyllus dixinus **Chamberlin**	17	12	17	22	9	14	4	8	4
Drassyllus dromeus **Chamberlin**	5		1	2	7	1	1		4
Drassyllus frigidus **(Banks)**		1		2		3			
Drassyllus lepidus **(Banks)**	14	1			25	37	39	39	88
Drassyllus nannellus **Chamberlin & Gertsch**	1								1
Drassyllus rufulus **(Banks)**	1	2		7	1	5			
Gnaphosa frontinalis **Keyserling**	7	7	8	4	1		3		5
Gnaphosa sericata **(L. Koch)**	7	3	3	13	41	31	18	27	29
Haplodrassus signifer **(C. L. Koch)**		7	5	4	3	1	3	2	3
Herpyllus ecclesiasticus **Hentz**					1		3		
Litopylus temporarius **Chamberlin**	1								
Micaria laticeps **Emerton**	2						1		4
Micaria longipes **Emerton**	1	2	1	22	6	4		2	6
Micaria longispina **Emerton**						1			
Micaria punctata **Banks**	2				11	106	63	57	2
Micaria seminola **Gertsch**	1								
Nodocion floridanus **(Banks)**			1				1		
Sergiolus capulatus **(Walckenaer)**	1							1	
Sergiolus tennessensis **Chamberlin**						1			1
Synaphosus paludis **Chamberlin & Gertsch**					1				
Talanites exlineae **(Platnick & Shadab)**	4	2	5	1				1	1
Zelotes aiken **(Platnick & Shadab)**	8	6	4	10	40	40	21	42	31
Zelotes duplex **Chamberlin**	1	1	8	2	2	1		2	
Zelotes hentzi **Barrows**	11	17	14	9	4	8	6	9	4
Zelotes laccus **(Barrows)**		2	1	2	8	1			
Zelotes lymnophilus **Chamberlin**	13	2	20	6	13	5	10	3	4
Total Genera	8	7	7	7	8	7	8	7	8
Total Species	22	17	16	16	18	18	15	14	17
Total Individuals	120	78	105	120	182	272	181	211	195

Table 6. Numbers of each species of gnaphosid collected at each site type, and totals collected, in 2018-2019 and 2020 trapping seasons combined.

Species	**Degraded**	**Restored**	**Intact**	**Total Collected**
***Calilepis imbecilla* (Keyserling)**	1	4	1	6
***Cesonia bilineata* (Hentz)**	1			1
***Drassodes gosiutus* Chamberlin**		4	2	6
***Drassyllus aprilinus* (Banks)**	27	3	3	33
***Drassyllus covensis* Exline**	4			4
***Drassyllus creolus* Chamberlin & Gertsch**	4	5	10	19
***Drassyllus dixinus* Chamberlin**	46	45	16	107
***Drassyllus dromeus* Chamberlin**	6	10	5	21
***Drassyllus frigidus* (Banks)**	1	5		6
***Drassyllus lepidus* (Banks)**	15	62	166	243
***Drassyllus nannellus* Chamberlin & Gertsch**	1		1	2
***Drassyllus rufulus* (Banks)**	3	13		16
***Gnaphosa frontinalis* Keyserling**	22	5	8	35
***Gnaphosa sericata* (L. Koch)**	13	85	74	172
***Haplodrassus signifer* (C. L. Koch)**	12	8	8	28
***Herpyllus ecclesiasticus* Hentz**		1	3	4
***Litopylus temporarius* Chamberlin**	1			1
***Micaria laticeps* Emerton**	2		5	7
***Micaria longipes* Emerton**	4	32	8	44
***Micaria longispina* Emerton**		1		1
***Micaria punctata* Banks**	2	117	122	241
***Micaria seminola* Gertsch**	1			1
***Nodocion floridanus* (Banks)**	1		1	2
***Sergiolus capulatus* (Walckenaer)**	1		1	2
***Sergiolus tennessensis* Chamberlin**		1	1	2
***Synaphosus paludis* Chamberlin & Gertsch**		1		1
***Talanites exlineae* (Platnick & Shadab)**	11	1	2	14
***Zelotes aiken* (Platnick & Shadab)**	18	90	94	202
***Zelotes duplex* Chamberlin**	10	5	2	17
***Zelotes hentzi* Barrows**	42	21	19	82
***Zelotes laccus* (Barrows)**	3	11		14
***Zelotes lymnophilus* Chamberlin**	35	24	17	76
Total Genera	11	11	11	14
Total Species	27	24	23	32
Total Individuals	287	554	569	1410

Table 7. Results of Kruskal-Wallis analysis of gnaphosid species diversity indices among Site Types. Figures followed by a single asterisk (*) are significant at $p < 0.10$, whereas those followed by double asterisk (**) are significant at $p < 0.05$.

Taxa	Index	Trapping Season	χ^2	*p*-value
Gnaphosids	Shannon-Wiener	2018-2019	3.20	.202
		2020	5.96	.051*
		Combined	8.92	.012**
	Simpson's	2018-2019	2.76	.252
		2020	3.82	.148
		Combined	6.18	.039**
Gnaphosids Trimmed Species ≥ 10	Shannon-Wiener	2018-2019	4.36	.113
		2020	4.36	.113
		Combined	6.87	.032**
	Simpson's	2018-2019	4.36	.113
		2020	3.29	.193
		Combined	6.22	.044**

Table 8. Morisita index of dissimilarity for pairwise site comparisons for gnaphosid spider species, 2018-2019 trapping season.

Site Type		Degraded			Restored			Intact	
		Site 2	Site 4	Site 9	Site 5	Site 6	Site 7	Site 3	Site 8
Degraded	Site 4	0.408							**Legend: 0**
	Site 9	0.469	0.212						**>0 - .250**
Restored	Site 5	0.491	0.448	0.577					**.251 - .500**
	Site 6	0.305	0.670	0.808	0.430				**.501 - .750**
	Site 7	0.449	0.827	0.914	0.728	0.408			**.751 +**
Intact	Site 3	0.584	0.830	0.916	0.886	0.622	0.025		
	Site 8	0.276	0.703	0.831	0.678	0.237	0.044	0.147	
	Site 10	0.370	0.524	0.781	0.360	0.000	0.466	0.663	0.252

Table 9. Morisita index of dissimilarity for pairwise site comparisons for gnaphosid spider species, 2020 trapping season.

Site Type		Degraded			Restored			Intact	
		Site 2	Site 4	Site 9	Site 5	Site 6	Site 7	Site 3	Site 8
Degraded	Site 4	0.000							**Legend: 0**
	Site 9	0.271	0.211						**>0 - .250**
Restored	Site 5	0.277	0.018	0.437					**.251 - .500**
	Site 6	0.549	0.543	0.596	0.492				**.501 - .750**
	Site 7	0.791	0.754	0.819	0.754	0.326			**.751 +**
Intact	Site 3	0.535	0.613	0.452	0.711	0.000	0.317		
	Site 8	0.742	0.729	0.782	0.770	0.196	0.002	0.193	
	Site 10	0.741	0.904	0.901	0.897	0.338	0.710	0.302	0.620

Table 10. Morisita index of dissimilarity for pairwise site comparisons for gnaphosid spider species with ≥ 10 collected (trimmed dataset) during the 2018-2019 trapping season.

Site Type		Degraded			Restored			Intact	
		Site 2	Site 4	Site 9	Site 5	Site 6	Site 7	Site 3	Site 8
Degraded	Site 4	0.404							**Legend: 0**
	Site 9	0.462	0.213						**>0 - .250**
Restored	Site 5	0.486	0.456	0.577					**.251 - .500**
	Site 6	0.296	0.667	0.817	0.424				**.501 - .750**
	Site 7	0.439	0.829	0.916	0.729	0.403			**.751 +**
Intact	Site 3	0.579	0.831	0.917	0.886	0.618	0.027		
	Site 8	0.255	0.702	0.830	0.675	0.227	0.046	0.157	
	Site 10	0.356	0.525	0.781	0.359	0.000	0.467	0.666	0.253

Table 11. Morisita index of dissimilarity for pairwise site comparisons for gnaphosid spider species with ≥ 10 collected (trimmed dataset) during the 2020 trapping season.

Site Type		Degraded			Restored			Intact	
		Site 2	Site 4	Site 9	Site 5	Site 6	Site 7	Site 3	Site 8
Degraded	Site 4	0.000							**Legend: 0**
	Site 9	0.256	0.196						**>0 - .250**
Restored	Site 5	0.297	0.078	0.432					**.251 - .500**
	Site 6	0.546	0.552	0.592	0.518				**.501 - .750**
	Site 7	0.778	0.745	0.818	0.739	0.309			**.751 +**
Intact	Site 3	0.525	0.607	0.454	0.714	0.000	0.315		
	Site 8	0.733	0.721	0.782	0.766	0.182	0.003	0.192	
	Site 10	0.727	0.901	0.902	0.890	0.327	0.713	0.315	0.627

Table 12. Numbers of predatory carabid species captured for collection dates in the 2018-2019 and 2020 seasons.

Species	**2018-2019 Trapping Season**														**2020 Trapping Season**				
	Sept. 9th	Oct. 6th	Nov. 3rd	Dec. 1st	Jan. 5th	Feb. 2nd	Mar. 1st	Mar. 30th	Apr. 28th	May 25th	June 24th	July 20th	Aug. 17th	Sept. 15th	Apr. 25th	May 23rd	June 20th	July 18th	Aug. 15th
***Calosoma externum* (Say)**										3		4	2	3		1			
***Calosoma sayi* (DeJean)**		13	22						1			5							
***Carabus sylvosus* Say**				10			3				4				1	2	18		
***Chlaenius emarginatus* Say**		1							1	1	1	1	12	9			2		6
***Chlaenius erythropus* Germar**											1								
***Chlaenius impunctifrons* Say**																		1	
***Chlaenius laticollis* Say**		1								1									1
***Chlaenius tomentosus* (Say)**									2	4	1	1	8	2		1		1	3
***Chlaenius vafer* LeConte**														2					
***Chlaenius* sp.**													1	2					
***Cicindela sexguttata* F.**									1										
***Cicindela splendida* Hentz**								1											
***Cicindelidia obsoleta* Say *vulturina* LeConte**	4												3	7					
***Cicindelidia punctulata* Olivier**											2	5	5					2	2

Table 12 (cont.). Numbers of predatory carabid species captured for collection dates in the 2018-2019 and 2020 seasons.

Species	2018-2019 Trapping Season															2020 Trapping Season				
	Sept. 9th	Oct. 6th	Nov. 3rd	Dec. 1st	Jan. 5th	Feb. 2nd	Mar. 1st	Mar. 30th	Apr. 28th	May 25th	June 24th	July 20th	Aug. 17th	Sept. 15th		Apr. 25th	May 23rd	June 20th	July 18th	Aug. 15th
***Cyclotrachelus seximpressus* (LeConte)**	3	4	6	4	4	1	1		3	1			1	5		1			1	
***Cyclotrachelus whitcombi* (Freitag)**		2	2	1						1	1	1		1			1	1		2
***Dicaelus elongatus* Bonelli**											2	1	1					2	4	
***Dicaelus purpuratus* Bonelli**	1													1					1	
***Dicaelus sculptilis* Say**											1									
***Galerita janus* F.**									1	8	4					3	3	3		1
***Helluomorphoides praeustus* (Dejean)**									2	2	2									
***Helluomorphoides texanus* LeConte**																				1
***Lebia analis* Dejean**													1							
***Microlestes* sp.**																2				
***Notiophilus novemstriatus* LeConte**									1							1				
***Omophron nitidum* LeConte**				1					1										1	1
***Panagaeus fasciatus* Say**												1								
***Paraclivina bipustulata* (F.)**										2	2		3				2	2		1

Table 12 (cont.). Numbers of predatory carabid species captured for collection dates in the 2018-2019 and 2020 seasons.

Species	**2018-2019 Trapping Season**															**2020 Trapping Season**				
	Sept. 9th	Oct. 6th	Nov. 3rd	Dec. 1st	Jan. 5th	Feb. 2nd	Mar. 1st	Mar. 30th	Apr. 28th	May 25th	June 24th	July 20th	Aug. 17th	Sept. 15th		Apr. 25th	May 23rd	June 20th	July 18th	Aug. 15th
***Pasimachus depressus* (F.)**	7	1							4	4	2	3	12	15		2		4	4	14
***Pasimachus elongatus* LeConte**	1									1				1					1	
***Perigona nigriceps* (Dejean)**											1					3		3	2	
***Pterostichus sculptus* LeConte**		1	19																	
***Scarites subterraneus* (F.)**									1		2	4				1	2	5	3	3
***Semiardistomis viridis* (Say)**																		1		
***Stenocrepis mexicana* (Chevrolat)**	1	2	4				1		2	3		4	1			1	8	7	6	1
***Tachys* sp.**													1							
***Tetracha virginica* (L.)**	9											13	33	25					8	31
TOTAL	26	25	53	16	4	1	5	1	20	31	26	39	84	73		15	19	48	39	67

Table 13. Numbers of each species of predatory carabid, with total numbers of species, genera and individuals collected at each site in 2018-2019 and 2020 trapping seasons.

Species	Degraded Sites			Restored Sites			Intact Sites		
	2	4	9	5	6	7	3	8	10
Calosoma externum (Say)		4		7				1	
Calosoma sayi (DeJean)			1	2	16	2	18	2	
Carabus sylvosus Say	18	5	6	2		1	1	1	4
Chlaenius emarginatus Say	3	4	2	4	7		12	1	1
Chlaenius erythropus Germar	1								
Chlaenius impunctifrons Say	1								
Chlaenius laticollis Say		1		2					
Chlaenius tomentosus (Say)	2				5	5	1	4	6
Chlaenius vafer LeConte					2				
Chlaenius sp.							3		
Cicindela sexguttata F.	1								
Cicindela splendida Hentz					1				
Cicindelidia obsoleta Say *vulturina* LeConte					1		13		
Cicindelidia punctulata Olivier	1	3		7	1	1		3	
Cyclotrachelus seximpressus (LeConte)		14	2	5	8	1		2	3
Cyclotrachelus whitcombi (Freitag)	13								
Dicaelus elongatus Bonelli		6	1	2	1				
Dicaelus purpuratus Bonelli			1	1	1				
Dicaelus sculptilis Say	1								
Galerita janus F.	2	8	5	8					
Helluomorphoides praeustus (Dejean)		5		1					
Helluomorphoides texanus LeConte		1							
Lebia analis Dejean									1
Microlestes sp.							2		
Notiophilus novemstriatus LeConte	1			1					
Omophron nitidum LeConte	1						1	2	
Panagaeus fasciatus Say							1		
Paraclivina bipustulata (F.)			1		7	2	2		
Pasimachus depressus (F.)	9	28	3	23			1		8
Pasimachus elongatus LeConte					1				3
Perigona nigriceps (Dejean)		2	1		2	3	1		
Pterostichus sculptus LeConte	2	2	4		2		8	1	1
Scarites subterraneus (F.)			1	1	7	4	1	4	3
Semiardistomis viridis (Say)					1				
Stenocrepis mexicana (Chevrolat)	3		5		23		8	2	
Tachys sp.				1					
Tetracha virginica (L.)	15	5	1	10	25	8	15	30	10
Total Genera	13	12	13	13	15	9	14	11	9
Total Species	16	14	14	16	18	9	16	12	10
Total Individuals	74	88	34	77	111	27	88	53	40

Table 14. Numbers of predatory carabids collected at each Site Type, and totals collected.

Species	Degraded	Restored	Intact	Total Collected
Calosoma externum (Say)	4	7	1	12
Calosoma sayi (DeJean)	1	20	20	41
Carabus sylvosus Say	29	3	6	38
Chlaenius emarginatus Say	11	9	14	34
Chlaenius erythropus Germar	1			1
Chlaenius impunctifrons Say	1			1
Chlaenius laticollis Say	1	2		3
Chlaenius tomentosus (Say)	2	10	11	23
Chlaenius vafer LeConte		2		2
Chlaenius sp.			3	3
Cicindela sexguttata F.	1			1
Cicindela splendida Hentz		1		1
Cicindelidia obsoleta Say *vulturina* LeConte		1	13	14
Cicindelidia punctulata Olivier	4	9	3	16
Cyclotrachelus seximpressus (LeConte)	16	14	5	35
Cyclotrachelus whitcombi (Freitag)	13			13
Dicaelus elongatus Bonelli	7	3		10
Dicaelus purpuratus Bonelli	1	2		3
Dicaelus sculptilis Say	1			1
Galerita janus F.	15	8		23
Helluomorphoides praeustus (Dejean)	1	5		6
Helluomorphoides texanus LeConte	1			1
Lebia analis Dejean			1	1
Microlestes sp.			2	2
Notiophilus novemstriatus LeConte	1	1		2
Omophron nitidum LeConte	1		3	4
Panagaeus fasciatus Say			1	1
Paraclivina bipustulata (F.)	1	9	2	12
Pasimachus depressus (F.)	40	23	9	72
Pasimachus elongatus LeConte		1	3	4
Perigona nigriceps (Dejean)	3	5	1	9
Pterostichus sculptus LeConte	8	2	10	20
Scarites subterraneus (F.)	1	12	8	21
Semiardistomis viridis (Say)		1		1
Stenocrepis mexicana (Chevrolat)	8	23	10	41
Tachys sp.		1		1
Tetracha virginica (L.)	21	43	55	119
Total Genera	18	19	16	23
Total Species	27	26	21	37
Total Individuals	194	217	181	592

Table 15. Results of Kruskal-Wallis analysis of predatory carabid species diversity indices among Site Types. Figures followed by a single asterisk (*) are significant at *p* < 0.10.

Taxa	Index	Trapping Season	χ^2	*p*-value
Carabids	Shannon-Wiener	2018-2019	0.87	.646
		2020	2.78	.249
		Combined	2.16	.340
	Simpson’s	2018-2019	1.72	.423
		2020	1.16	.561
		Combined	0.27	.875
Carabids Trimmed Species ≥ 10	Shannon-Wiener	2018-2019	0.09	.957
		2020	0.36	.837
		Combined	2.04	.980
	Simpson’s	2018-2019	0.09	.957
		2020	5.96	.051*
		Combined	1.30	.996

Table 16. Morisita index of dissimilarity for pairwise site comparisons for predatory carabid species, 2018-2019 trapping season.

Site Type		Degraded			Restored			Intact	
		Site 2	Site 4	Site 9	Site 5	Site 6	Site 7	Site 3	Site 8
Degraded	Site 4	0.814							**Legend: 0**
	Site 9	0.624	0.411						**>0 - .250**
Restored	Site 5	0.592	0.077	0.477					**.251 - .500**
	Site 6	0.507	0.757	0.546	0.617				**.501 - .750**
	Site 7	0.255	0.803	0.784	0.560	0.250			**.751 +**
Intact	Site 3	0.529	0.899	0.496	0.712	0.098	0.450		
	Site 8	0.215	0.839	0.834	0.575	0.376	0.010	0.477	
	Site 10	0.272	0.288	0.455	0.190	0.491	0.112	0.603	0.249

Table 17. Morisita index of dissimilarity for pairwise site comparisons for predatory carabid species, 2020 trapping season.

Site Type		Degraded			Restored			Intact	
		Site 2	Site 4	Site 9	Site 5	Site 6	Site 7	Site 3	Site 8
Degraded	Site 4	0.234							**Legend: 0**
	Site 9	0.006	0.500						**>0 - .250**
Restored	Site 5	0.337	0.000	0.681					**.251 - .500**
	Site 6	0.812	0.878	0.387	0.868				**.501 - .750**
	Site 7	0.934	0.747	0.007	0.826	0.486			**.751 +**
Intact	Site 3	0.885	0.720	0.444	0.814	0.436	0.096		
	Site 8	0.805	0.804	0.815	0.805	0.291	0.429	0.200	
	Site 10	0.886	0.871	0.200	0.741	0.335	0.000	0.429	0.348

Table 18. Morisita index of dissimilarity for pairwise site comparisons for predatory carabid species, 2018-2019 trapping season. Indices calculated for species that included those with ≥ 10 individuals.

Site Type		Degraded			Restored			Intact	
		Site 2	Site 4	Site 9	Site 5	Site 6	Site 7	Site 3	Site 8
Degraded	Site 4	0.777							**Legend: 0**
	Site 9	0.593	0.428						**>0 - .250**
Restored	Site 5	0.536	0.070	0.477					**.251 - .500**
	Site 6	0.415	0.750	0.560	0.606				**.501 - .750**
	Site 7	0.133	0.800	0.788	0.563	0.268			**.751 +**
Intact	Site 3	0.468	0.897	0.497	0.710	0.086	0.462		
	Site 8	0.039	0.833	0.838	0.579	0.345	0.025	0.485	
	Site 10	0.161	0.269	0.459	0.186	0.489	0.122	0.603	0.251

Table 19. Morisita index of dissimilarity for pairwise site comparisons for predatory carabid species, 2020 trapping season. Indices calculated for species that included those with ≥ 10 individuals.

Site Type		Degraded			Restored			Intact	
		Site 2	Site 4	Site 9	Site 5	Site 6	Site 7	Site 3	Site 8
Degraded	Site 4	0.145							**Legend: 0**
	Site 9	0.000	0.600						**>0 - .250**
Restored	Site 5	0.223	0.000	0.651					**.251 - .500**
	Site 6	0.814	0.860	0.418	0.827				**.501 - .750**
	Site 7	0.916	0.848	0.471	0.725	0.418			**.751 +**
Intact	Site 3	0.882	0.782	1.000	0.724	0.478	0.429		
	Site 8	0.796	0.739	0.794	0.651	0.292	0.252	0.040	
	Site 10	0.923	0.862	0.333	0.714	0.405	0.000	0.500	0.293

Table 20. Results of NMDS analysis of environmental variables and habitat characteristics for gnaphosids collected in the 2018-2019 trapping season, showing model fit (r^2) and significance (* $\alpha = 0.10$, ** $\alpha = 0.05$).

	Environmental Variable		r^2	*p*-value
Continuous Variables	Daubenmire Groundcover Class	Grasslike	0.722	.022**
		Forbs	0.146	.590
		Cedar	0.383	.236
		Woody	0.560	.095*
		Moss	0.151	.566
		Exposed Rock	0.712	.021**
		Bare Soil	0.696	.031**
		Leaf Litter	0.838	.010**
		Dead Wood	0.339	.260
	Site Perimeter		0.571	.068*
	Site Area		0.531	.057*
Categorical Variables	Slope		0.644	.013**
	C. collaris		0.263	.098*
	Site Category		0.501	.102

Table 21. Results of NMDS analysis of environmental variables and habitat characteristics for gnaphosids collected in the 2020 trapping season, showing model fit (r^2) and significance (* $\alpha = 0.10$, ** $\alpha = 0.05$).

	Environmental Variable		r^2	*p*-value
Continuous Variables	Daubenmire Groundcover Class	Grasslike	0.354	.261
		Forbs	0.288	.358
		Cedar	0.431	.176
		Woody	0.494	.127
		Moss	0.178	.561
		Exposed Rock	0.518	.112
		Bare Soil	0.207	.483
		Leaf Litter	0.760	.012**
		Dead Wood	0.156	.600
	Site Perimeter		0.342	.287
	Site Area		0.359	.246
Categorical Variables	Slope		0.518	.055*
	C. collaris		0.181	.265
	Site Category		0.491	.074*

Table 22. Results of NMDS analysis of environmental variables and habitat characteristics for gnaphosids collected in both the 2018-2019 and 2020 trapping seasons, showing model fit (r^2) and significance (* α = 0.10, ** α = 0.05).

	Environmental Variable		r^2	*p*-value
Continuous Variables	Daubenmire Groundcover Class	Grasslike	0.727	.022**
		Forbs	0.338	.274
		Cedar	0.518	.059*
		Woody	0.735	.022*
		Moss	0.516	.119
		Exposed Rock	0.679	.027**
		Bare Soil	0.520	.121
		Leaf Litter	0.893	.004**
		Dead Wood	0.256	.394
	Site Perimeter		0.552	.080*
	Site Area		0.479	.096*
Categorical Variables	Slope		0.645	.011**
	C. collaris		0.205	.193
	Site Category		0.466	.108

Table 23. Results of NMDS analysis of environmental variables and habitat characteristics for gnaphosids collected in the 2018-2019 trapping season, trimmed to include only species with ≥ 10 individuals collected, and showing model fit (r^2) and significance (* α = 0.10, ** α = 0.05).

	Environmental Variable		r^2	*p*-value
Continuous Variables	Daubenmire Groundcover Class	Grasslike	0.703	.030**
		Forbs	0.160	.596
		Cedar	0.117	.717
		Woody	0.719	.020**
		Moss	0.180	.569
		Exposed Rock	0.583	.080*
		Bare Soil	0.801	.012**
		Leaf Litter	0.861	.005**
		Dead Wood	0.414	.206
	Site Perimeter		0.661	.039**
	Site Area		0.599	.038**
Categorical Variables	Slope		0.686	.013**
	C. collaris		0.378	.070*
	Site Category		0.631	.040**

Table 24. Results of NMDS analysis of environmental variables and habitat characteristics for gnaphosids collected in the 2020 trapping season, trimmed to include only species with ≥ 10 individuals collected, and showing model fit (r^2) and significance (* α = 0.10, ** α = 0.05).

	Environmental Variable		r^2	*p*-value
Continuous Variables	Daubenmire Groundcover Class	Grasslike	0.584	.079*
		Forbs	0.365	.265
		Cedar	0.479	.116
		Woody	0.833	.006
		Moss	0.411	.229
		Exposed Rock	0.591	.083*
		Bare Soil	0.640	.067*
		Leaf Litter	0.819	.013**
		Dead Wood	0.211	.481
	Site Perimeter		0.613	.045**
	Site Area		0.471	.100
Categorical Variables	Slope		0.574	.014**
	C. collaris		0.208	.195
	Site Category		0.593	.012**

Table 25. Results of NMDS analysis of environmental variables and habitat characteristics for predatory carabids collected in the 2018-2019 trapping season, showing model fit (r^2) and significance (* α = 0.10, ** α = 0.05).

	Environmental Variable		r^2	*p*-value
Continuous Variables	Daubenmire Groundcover Class	Grasslike	0.489	.160
		Forbs	0.459	.178
		Cedar	0.419	.165
		Woody	0.803	.014**
		Moss	0.412	.213
		Exposed Rock	0.394	.225
		Bare Soil	0.543	.078*
		Leaf Litter	0.517	.138
		Dead Wood	0.237	.466
	Site Perimeter		0.218	.477
	Site Area		0.281	.384
Categorical Variables	Slope		0.524	.048**
	C. collaris		0.466	.015**
	Site Category		0.175	.658

Table 26. Results of NMDS analysis of environmental variables and habitat characteristics for predatory carabids collected in the 2020 trapping season, showing model fit (r^2) and significance (* $\alpha = 0.10$, ** $\alpha = 0.05$).

	Environmental Variable		r^2	*p*-value
Continuous Variables	Daubenmire Groundcover Class	Grasslike	0.304	.308
		Forbs	0.012	.968
		Cedar	.0317	.291
		Woody	0.772	.013**
		Moss	0.015	.949
		Exposed Rock	0.235	.423
		Bare Soil	0.164	.524
		Leaf Litter	0.319	.281
		Dead Wood	0.107	.696
	Site Perimeter		0.127	.649
	Site Area		0.220	.500
Categorical Variables	Slope		0.474	.074*
	C. collaris		0.110	.445
	Site Category		0.197	.611

Table 27. Results of NMDS analysis of environmental variables and habitat characteristics for predatory carabids collected in the 2018-2019 and 2020 trapping seasons, showing model fit (r^2) and significance (* $\alpha = 0.10$, ** $\alpha = 0.05$).

	Environmental Variable		r^2	*p*-value
Continuous Variables	Daubenmire Groundcover Class	Grasslike	0.456	.175
		Forbs	0.143	.634
		Cedar	0.526	.094*
		Woody	0.753	.014**
		Moss	0.122	.667
		Exposed Rock	0.407	.216
		Bare Soil	0.323	.294
		Leaf Litter	0.728	.025**
		Dead Wood	0.086	.759
	Site Perimeter		0.393	.215
	Site Area		0.439	.136
Categorical Variables	Slope		0.690	.003**
	C. collaris		0.285	.096*
	Site Category		0.296	.356

Table 28. Results of NMDS analysis of environmental variables and habitat characteristics for predatory carabids collected in the 2018-2019 trapping season, trimmed to include only species with ≥ 10 individuals collected, and showing model fit (r^2) and significance (* α = 0.10, ** α = 0.05).

	Environmental Variable		r^2	*p*-value
Continuous Variables	Daubenmire Groundcover Class	Grasslike	0.534	.111
		Forbs	0.157	.579
		Cedar	0.002	.995
		Woody	0.800	.013**
		Moss	0.101	.722
		Exposed Rock	0.329	.323
		Bare Soil	0.505	.155
		Leaf Litter	0.487	.158
		Dead Wood	0.122	.669
	Site Perimeter		0.270	.386
	Site Area		0.380	.198
Categorical Variables	Slope		0.617	.007**
	C. collaris		0.331	.081*
	Site Category		0.165	.697

Table 29. Results of NMDS analysis of environmental variables and habitat characteristics for predatory carabids collected in the 2020 trapping season, trimmed to include only species with ≥ 10 individuals collected, and showing model fit (r^2) and significance (* α = 0.10, ** α = 0.05).

	Environmental Variable		r^2	*p*-value
Continuous Variables	Daubenmire Groundcover Class	Grasslike	0.513	.125
		Forbs	0.018	.951
		Cedar	0.509	.101
		Woody	0.723	.030**
		Moss	0.032	.911
		Exposed Rock	0.537	.099*
		Bare Soil	0.305	.320
		Leaf Litter	0.562	.079*
		Dead Wood	0.431	.172
	Site Perimeter		0.462	.158
	Site Area		0.559	.060*
Categorical Variables	Slope		0.587	.029**
	C. collaris		0.816	.502
	Site Category		0.368	.165

FIGURES

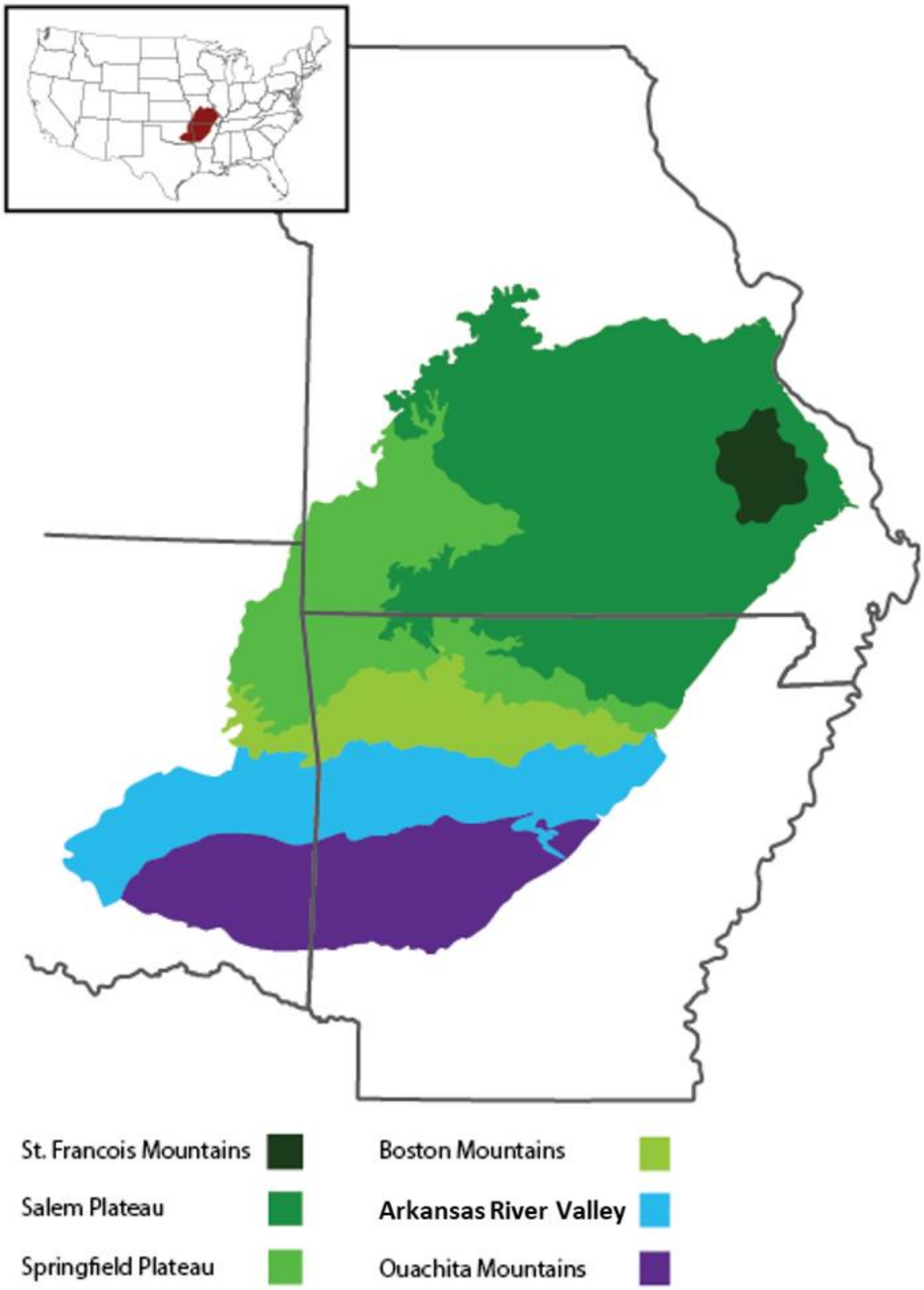

Figure 1. Ecoregion and geologic divisions of the Interior Highlands, reproduced from Skvarla et al. (2015) with permission. The Ozarks ecoregion encompasses the St. Francois Mountains, Salem Plateau, and Springfield plateau.

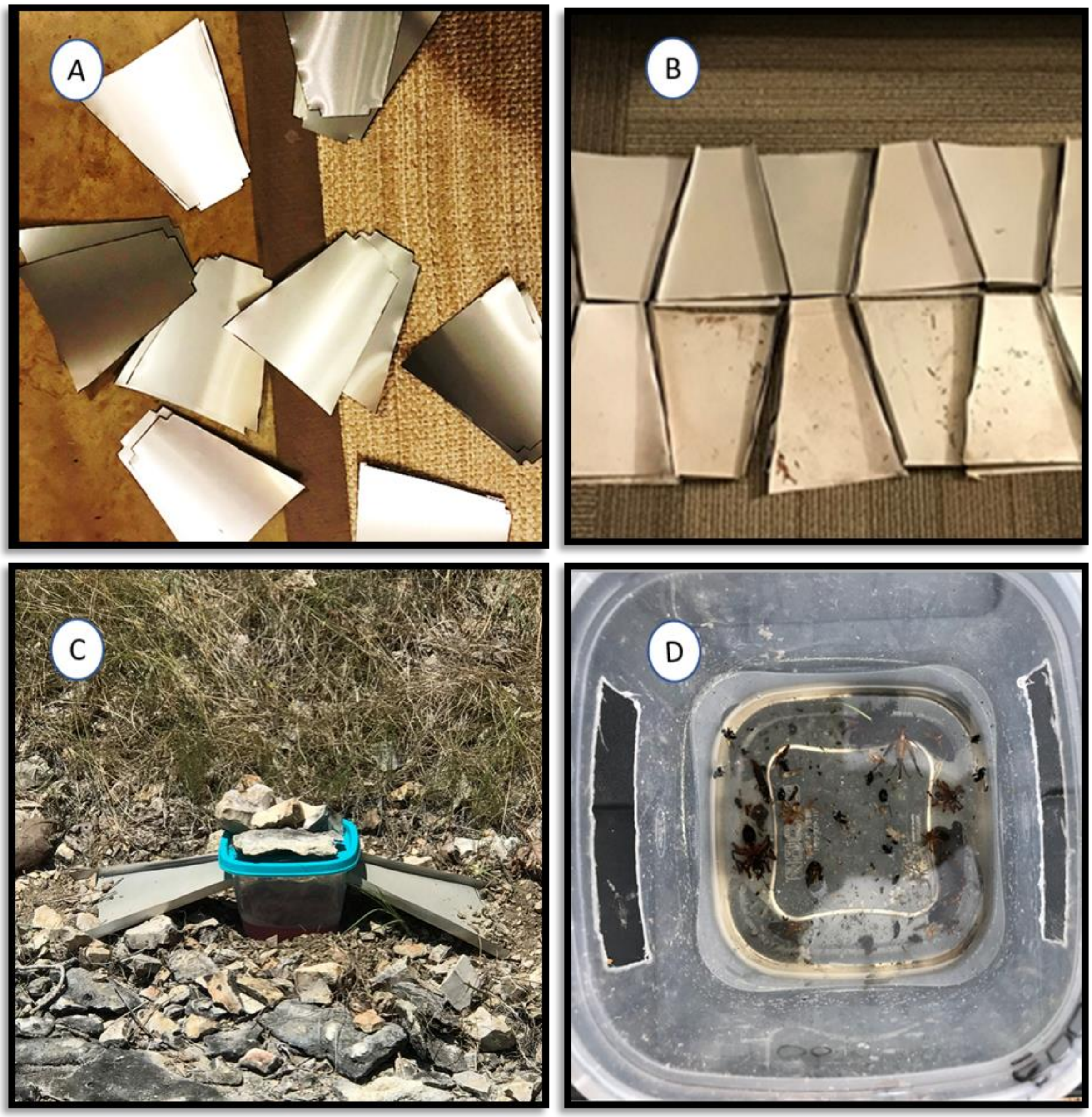

Figure 2. Lunchbox-style trap (LST) composite. Design is similar to Patrick and Hansen (2013). Figure 2A**.** Flat aluminum "slugs" (16 cm long X 12 cm in width on the wide end, tapering to 6 cm wide on the narrow end). Figure 2B**.** Stacked pairs of aluminum slugs folded into ramps and coated with textured spray paint (bottom row previously deployed). Figure 2C. LST deployed during preliminary testing with rocks added to resist toppling from wind. Figure 2D. Interior of LST collection container with lid removed revealing entrance openings and collected specimens. Photo credit: Austin Jones

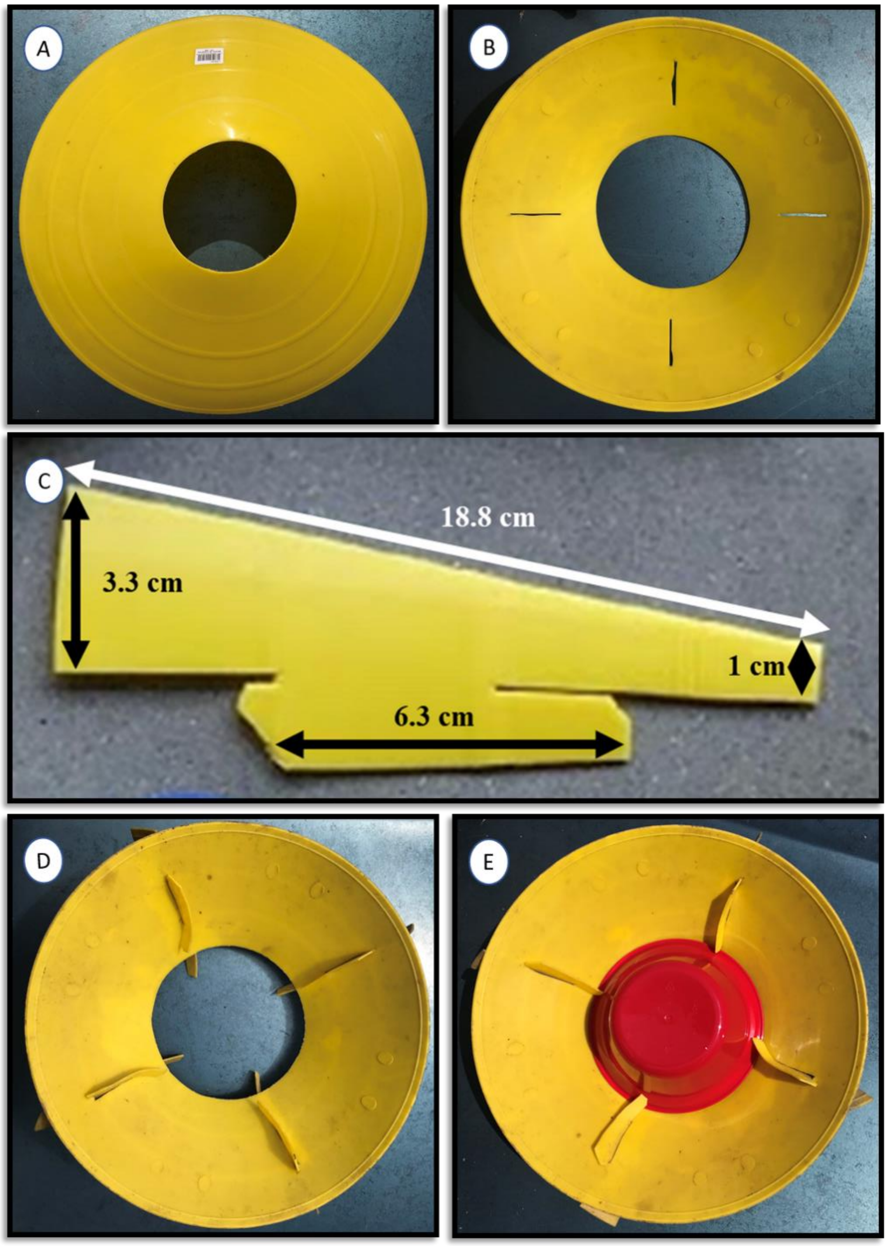

Figure 3. Stages of cone-style trap (CST) construction. Figure 3A. Top view of unmodified field-sports cone. Figure 3B. Bottom view of sports cone with modifications including increased top opening diameter and ramp slits. Figure 3C. Trap component that inserts into slits and allows for attachment of the collection container and trap lid. Figure 3D. Bottom view of CST assembled without collection container. Figure 3E. Bottom view of CST with collection container in place. Photo credit: Austin Jones

Figure 4. Cone-style trap (CST) composite. Figure 4A. Top view of painted CST without collection container or lid. Figure 4B**.** Top view of CST with collection container. Figure 4C. Testing of unpainted CST deployed on limestone shelf. Figure 4D**.** CST collection container with collected specimens. Photo credit: Austin Jones

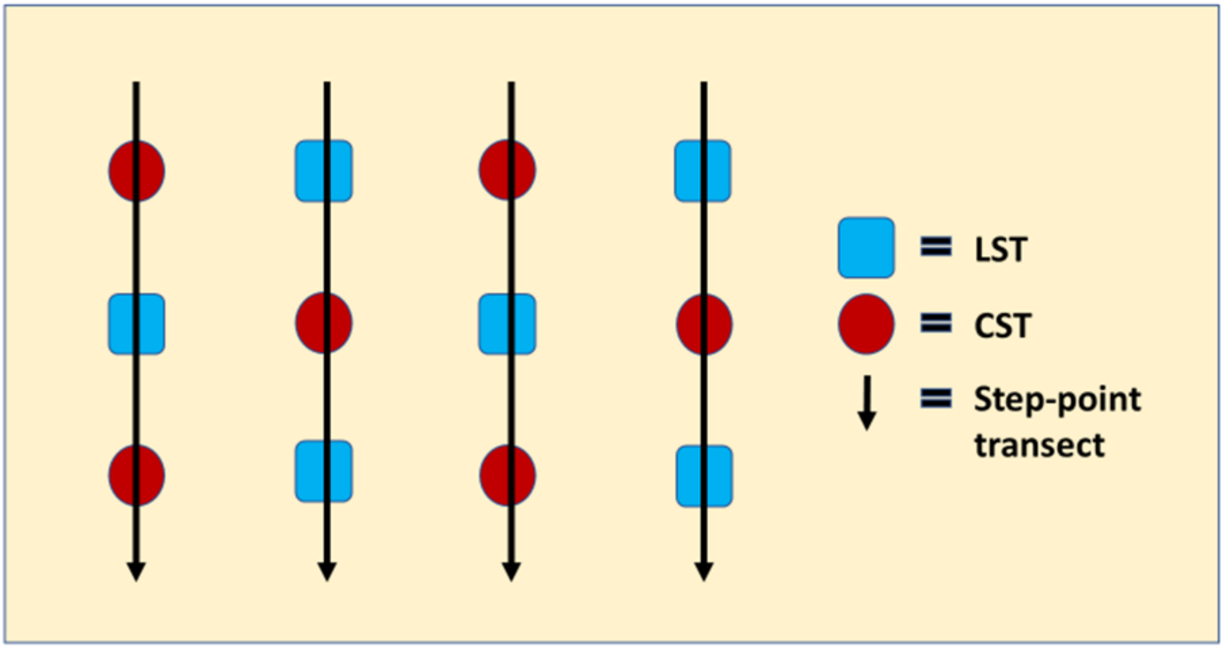

Figure 5. Trap grid design. Trap grids consisted of 6 Lunchbox-Style Traps (LST) and 6 Cone-Style Traps (CST) alternating in a 3 X 4 grid. Rows and columns of traps were spaced 13 m apart. Modified Step-Point groundcover sampling transects were 50 m in length, aligned with columns of traps, and equidistant in both directions from the center trap of a column. Image credit: Austin Jones

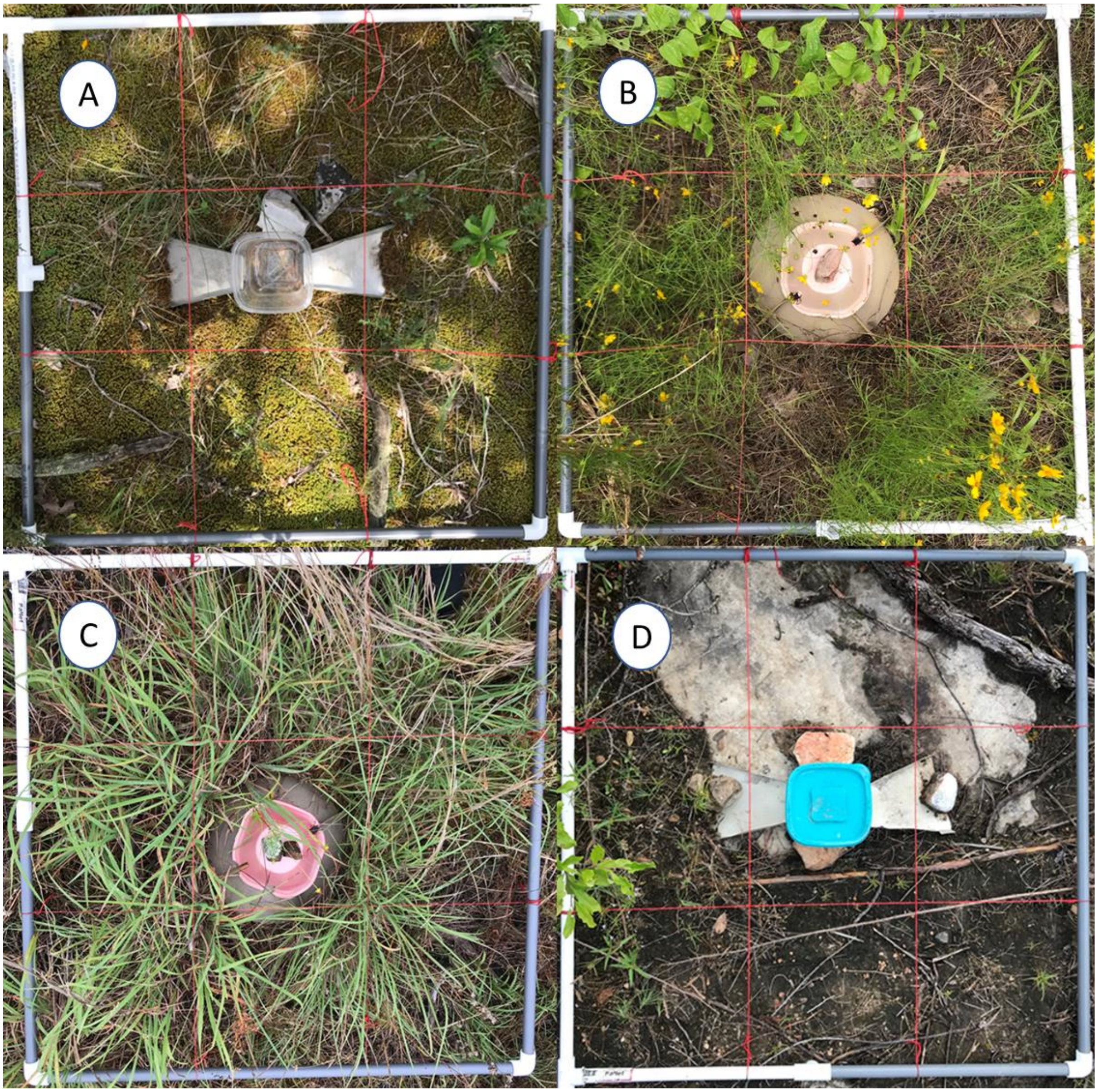

Figure 6. Composite of 1 m^2 quadrat photos centered around traps used in groundcover estimation. Figure 6A. LST quadrat dominated by the Moss groundcover category. Figure 6B. CST quadrat dominated by the Forbs groundcover category. Figure 6C. CST dominated by the Grasslike groundcover category. Figure 6D. LST deployed in predominantly Bare Soil and Exposed Rock groundcover categories. Photo credit: Austin Jones

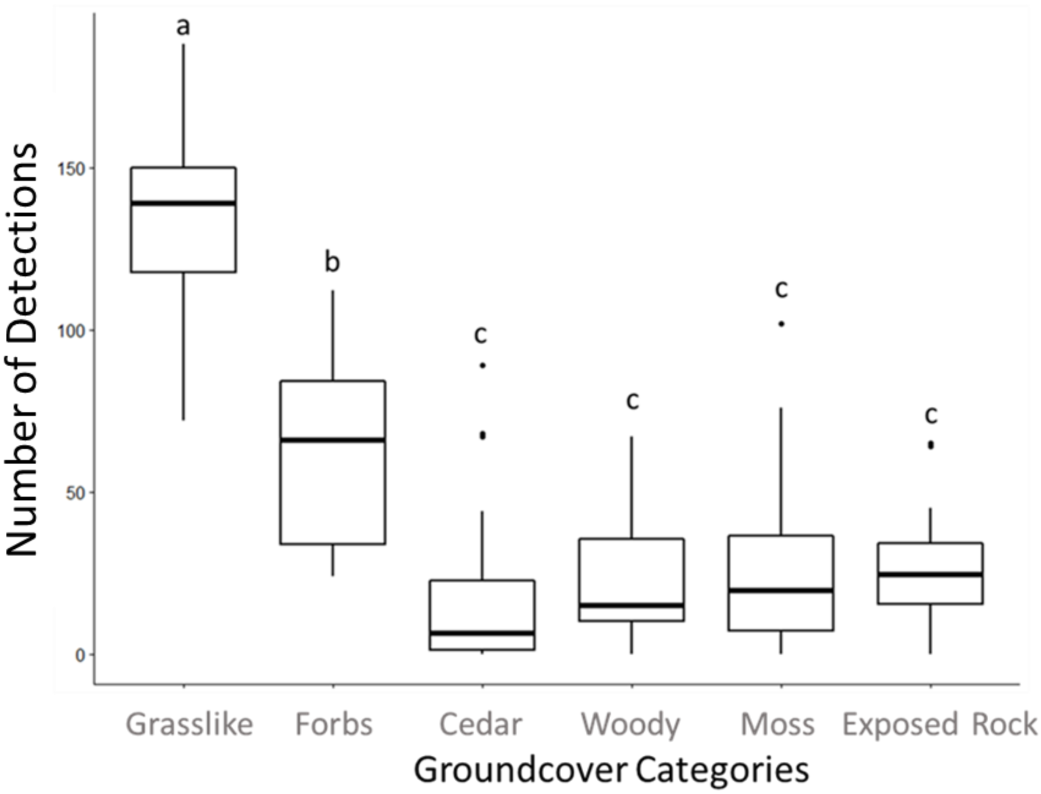

Figure 7. Total frequencies of occurrence for groundcover categories for all three Site Types (Degraded, Restored and Intact), sampled with the modified Step-Point method. Frequencies are based on 200 possible detections. Bars represent medians, boxes represent interquartile range (IQR) and whiskers represent 1.5(IQR). Different letters denote differences ($p < 0.05$) among groundcover category means based on Tukey HSD test.

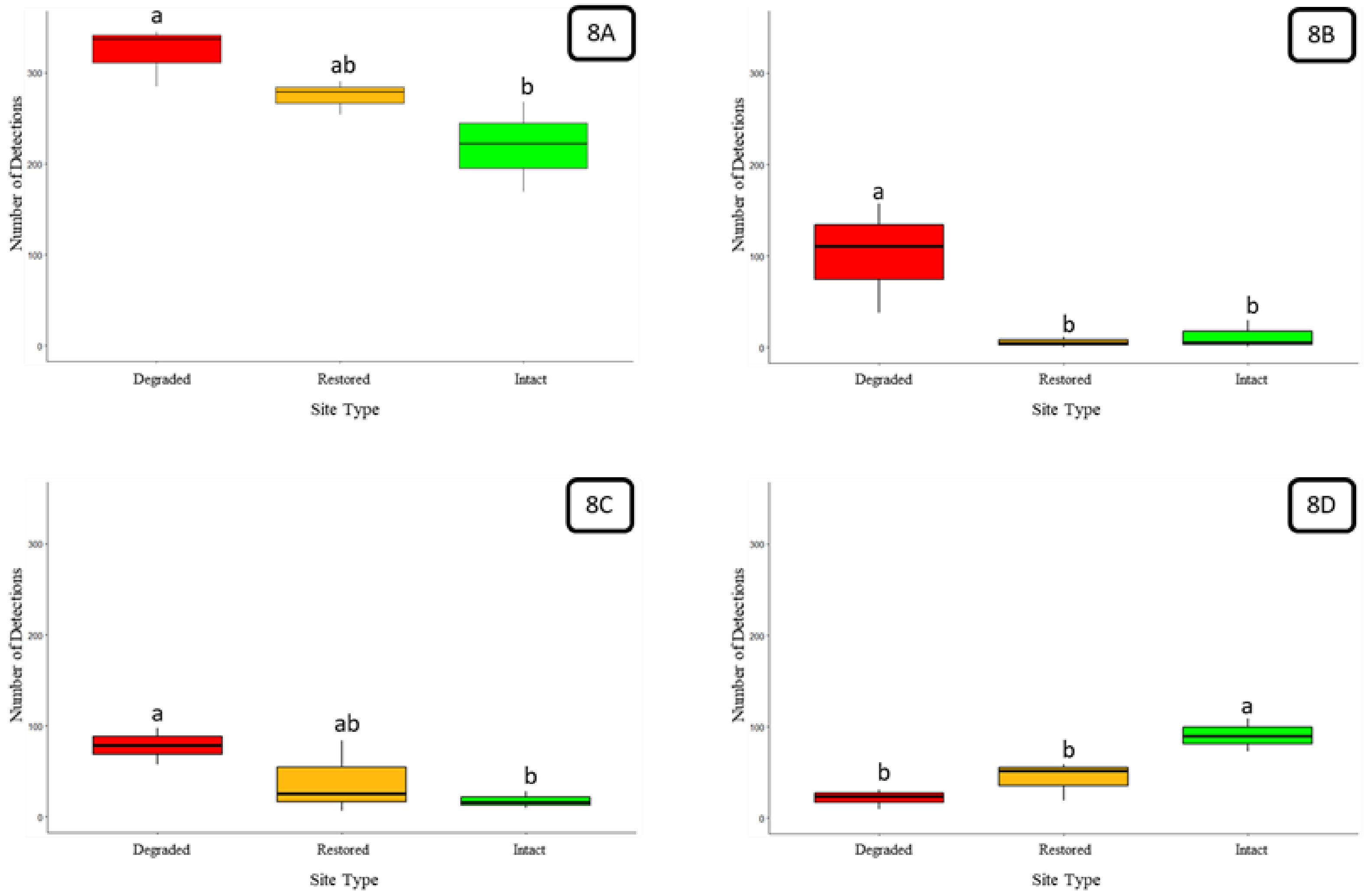

Figure 8. Number of detections at Degraded, Restored and Intact Site Types from modified Step-Point sampling, for Grasslike (Figure 8A), Cedar (Figure 8B), Woody (Figure 8C) and Exposed Rock (Figure 8D). Numbers of detections are based on 400 possible detections (200 each in 2018-2019 and 2020). Different letters denote differences ($p < 0.05$) among Site Type means based on Tukey HSD test.

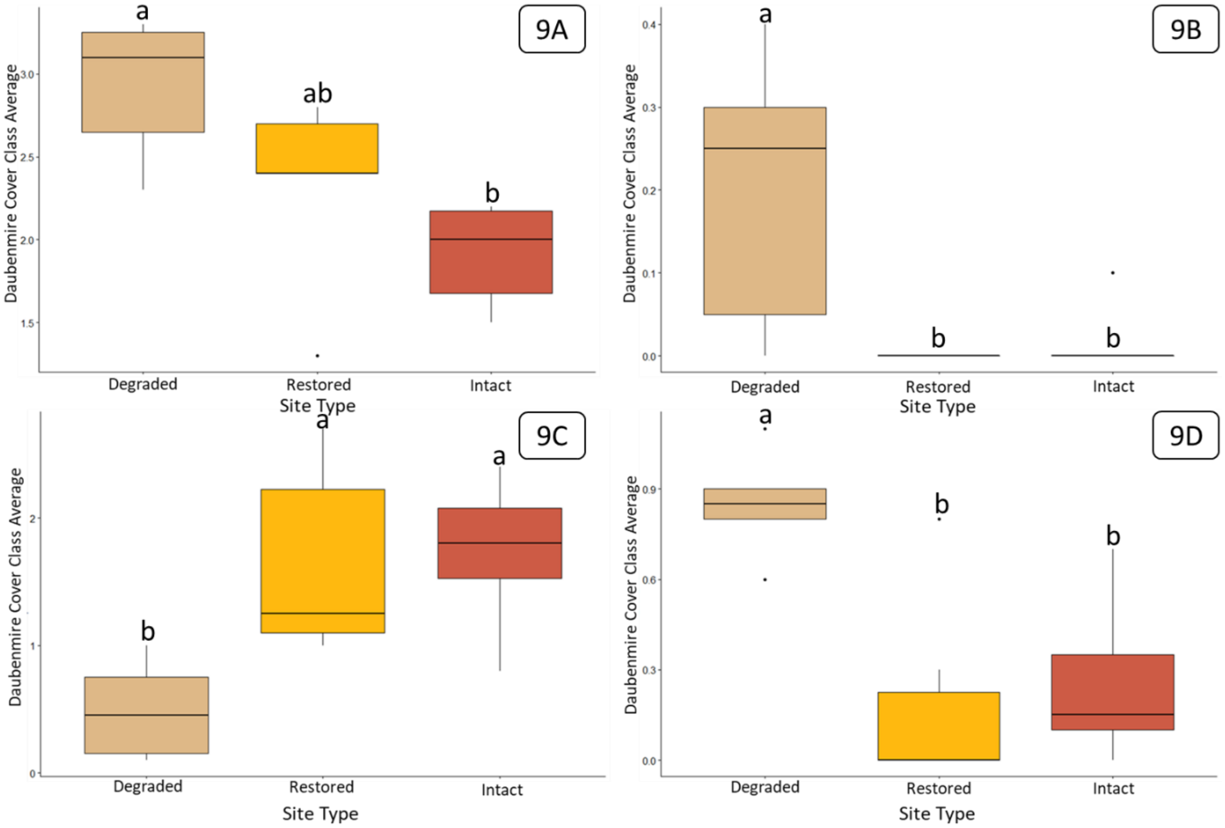

Figure 9. Daubenmire cover class averages for Degraded, Restored and Intact Site Types from photographic groundcover estimations of 12 quadrats (1 m^2) each year sampled (2019 & 2020). Shown are figures for Grasslike (Figure 9A), Cedar (Figure 9B), Exposed Rock (Figure 9C) and Leaf Litter (Figure 9D) groundcover categories. Bars represent the median of six figures for each site type as shown in Table 3. Different letters denote differences ($p < 0.05$) among Site Type means based on Tukey HSD test. Please note differences in scale of the y-axis.

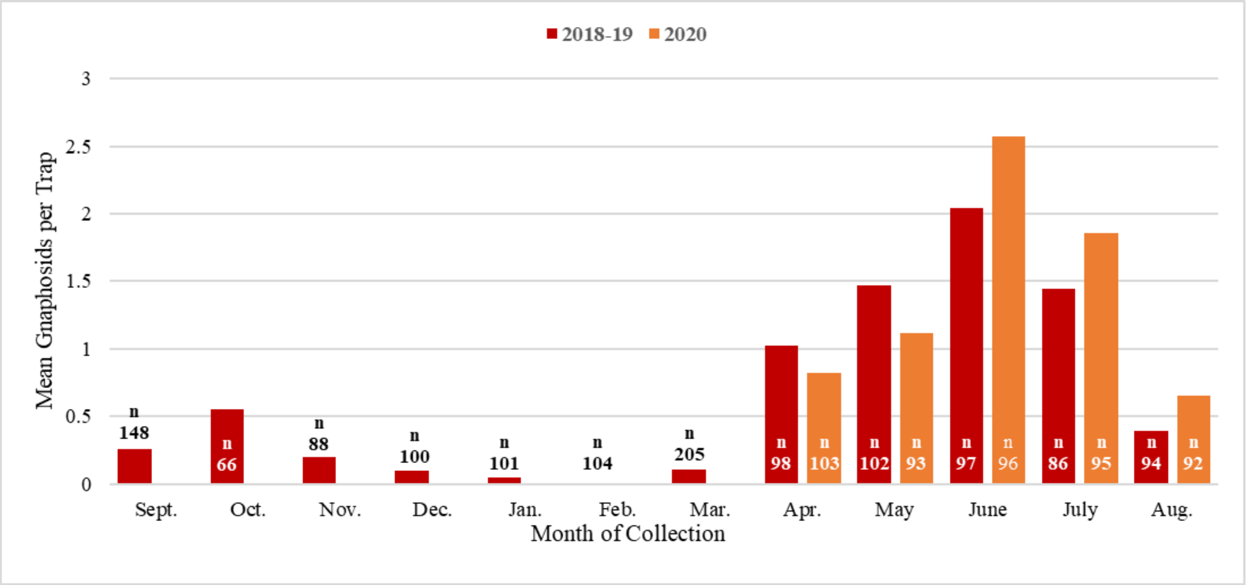

Figure 10. Mean number of adult gnaphosids captured per trap by month of collection. Numbers (n) indicate the number of traps recovered, reflecting trap loss as well as unequal sampling efforts for March (two collection dates in 2019), September (collections were combined from six sites in 2018 and nine sites in 2019) and October (only six sites were sampled in 2018).

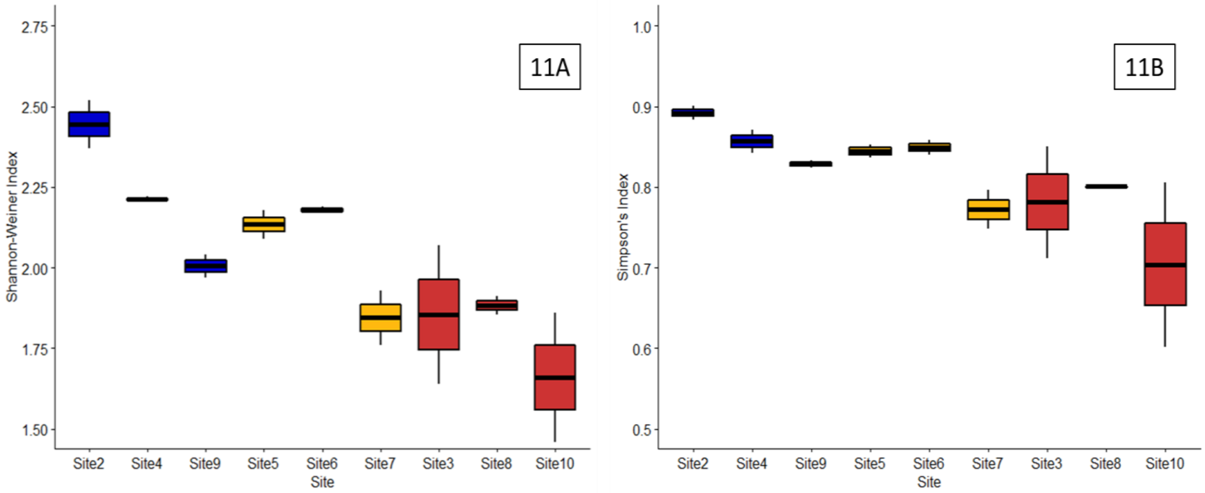

Figure 11. Diversity indices for gnaphosid spiders collected from nine glade sites in 2018-2019 and 2020, combined. Horizontal bars represent means of both trapping seasons. Figure 11A. Shannon-Wiener Diversity Index. Figure 11B. Simpson's Diversity Index.

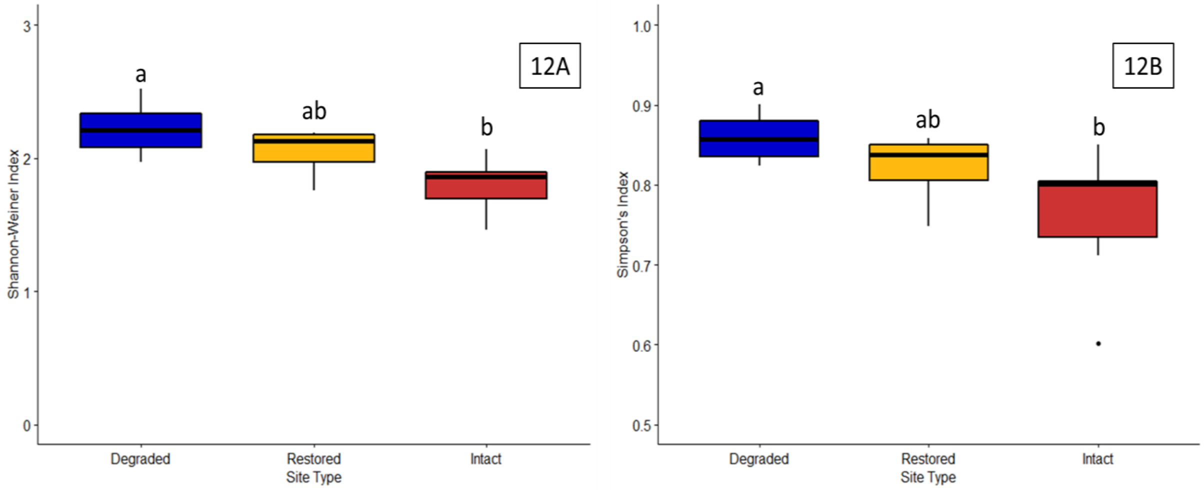

Figure 12. Diversity indices for gnaphosid spiders collected from three glade site types in 2018-2019 and 2020, combined. Horizontal bars represent means of both trapping seasons. Figure 12A. Shannon-Wiener Diversity Index. Figure 12B. Simpson's Diversity Index. Horizontal bars represent index medians and letters represent statistically different groups ($p < 0.05$) based on results of Dunn's test.

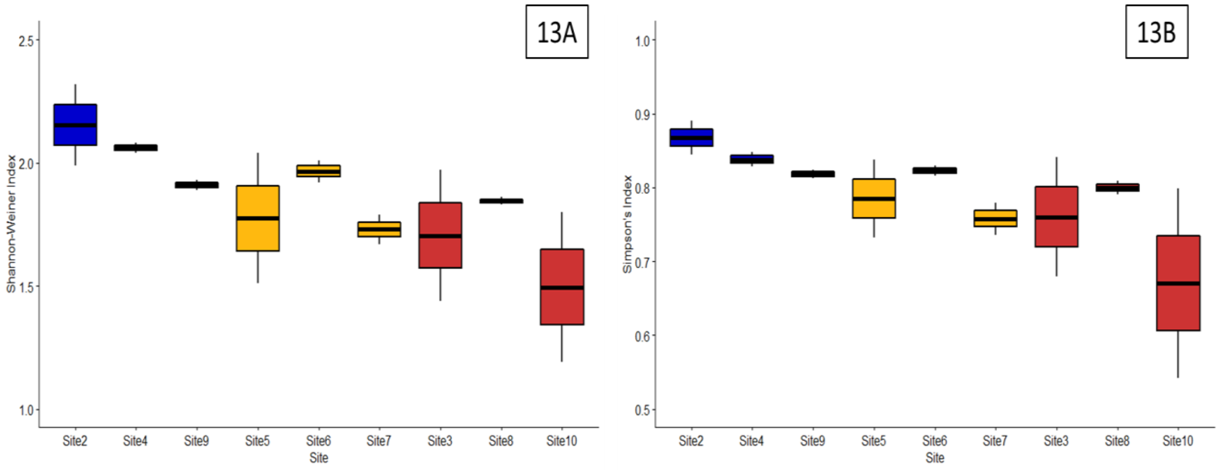

Figure 13. Diversity indices for gnaphosid spiders collected from nine glade sites for species with ≥ 10 collected in 2018-2019 and 2020, combined. Horizontal bars represent means of both trapping seasons. Figure 13A. Shannon-Wiener Diversity Index. Figure 13B. Simpson's Diversity Index.

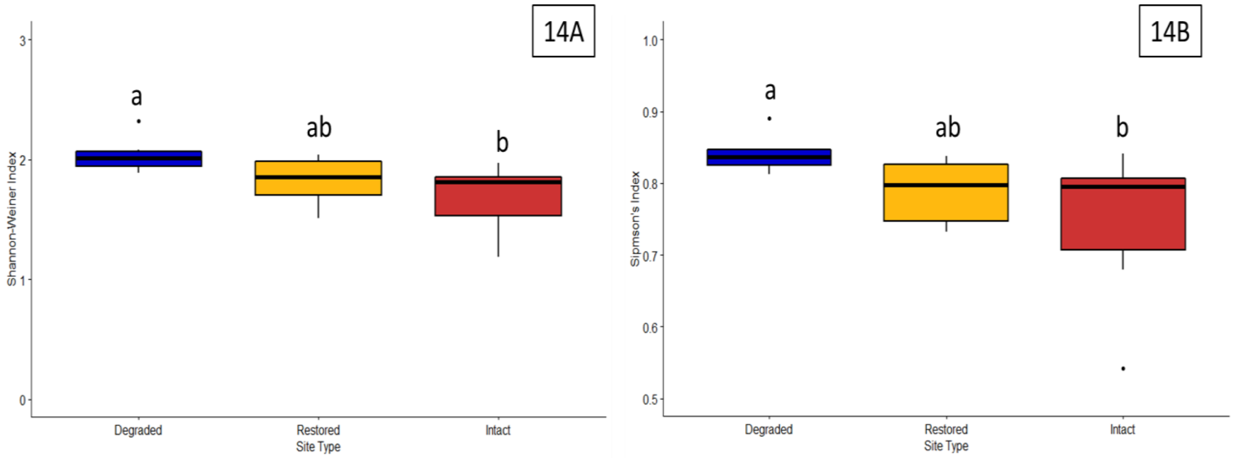

Figure 14. Diversity indices for gnaphosid spiders collected from three glade site types for species with ≥ 10 collected in 2018-2019 and 2020, combined. Horizontal bars represent index medians and letters represent statistically different groups (*p* < 0.05) based on results of Dunn's test. Figure 14A. Shannon-Wiener Diversity Index. Figure 14B. Simpson's Diversity Index.

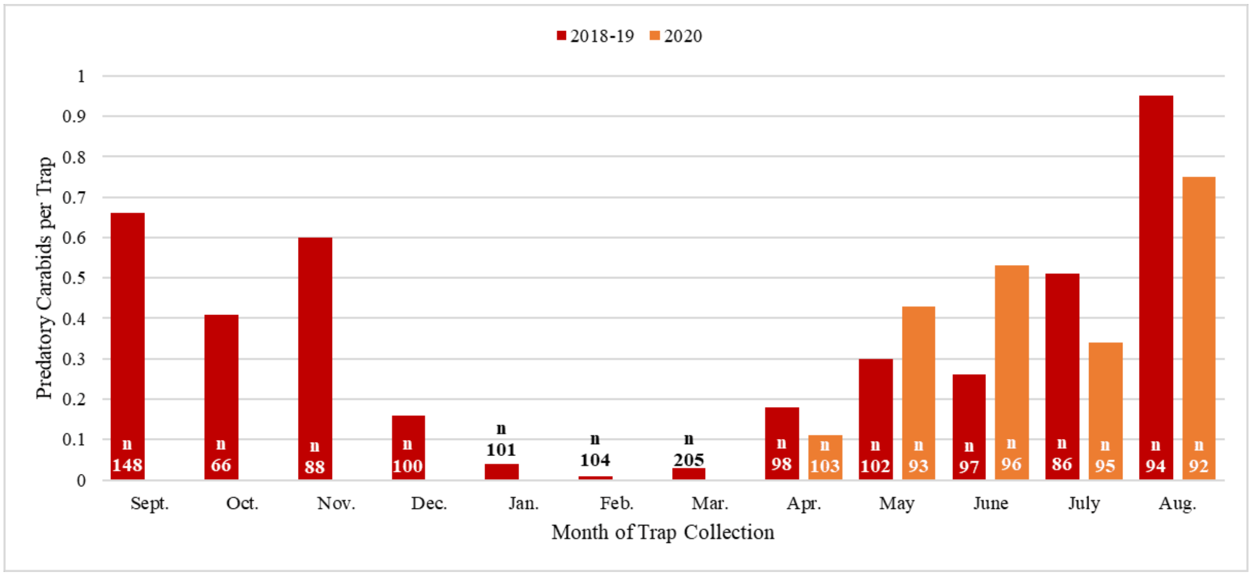

Figure 15. Mean number of predatory carabids captured per trap by month of collection. Numbers (n) indicate the number of traps recovered, reflecting trap loss as well as unequal sampling efforts for March (two collection dates in 2019), September (collections were combined from six sites in 2018 and nine sites in 2019) and October (only six sites were sampled in 2018).

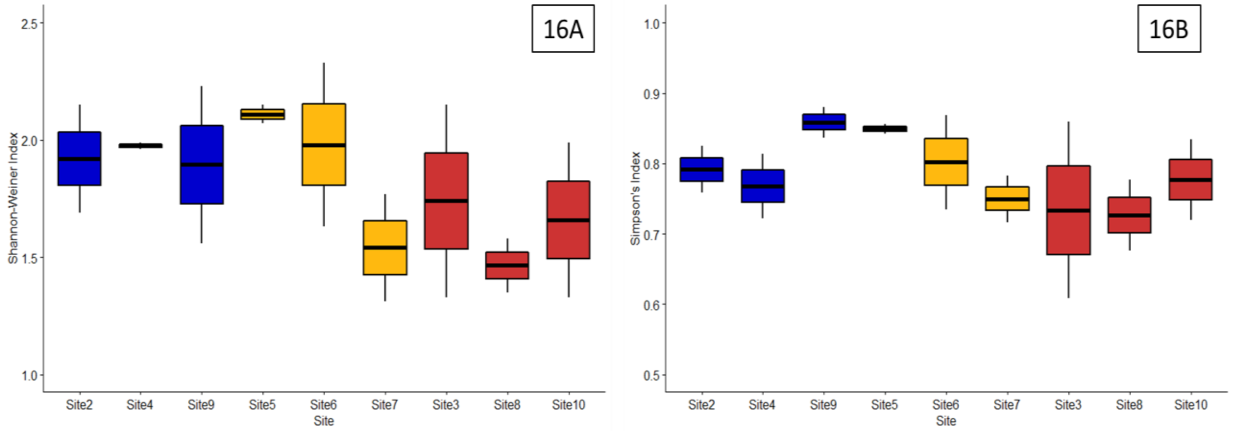

Figure 16. Diversity indices for predatory carabids collected from nine glade sites in 2018-2019 and 2020, combined. Horizontal bars represent index medians. Figure 16A. Shannon-Wiener Diversity Index. Figure 16B. Simpson's Diversity Index.

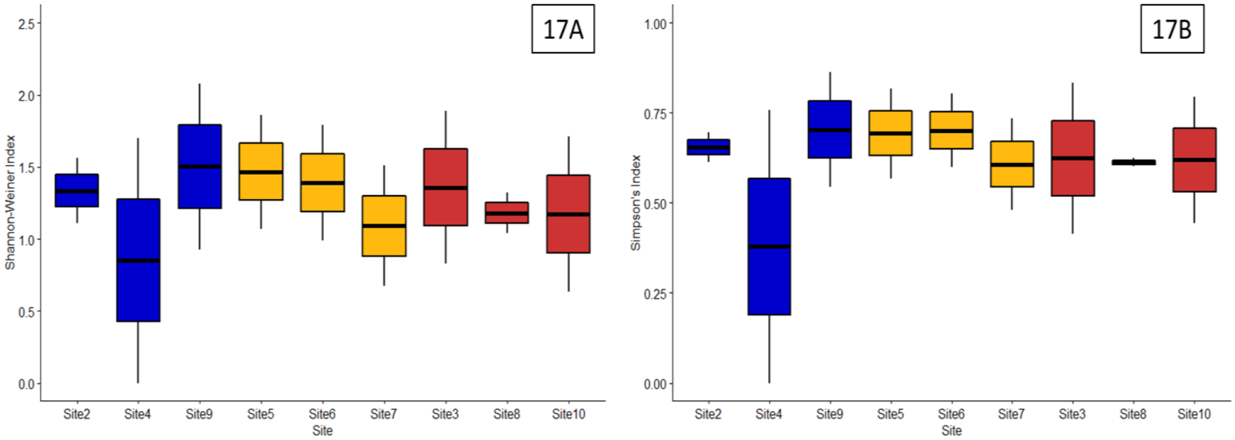

Figure 17. Diversity indices for predatory carabid species with ≥ 10 collected from nine glade sites in 2018-2019 and 2020, combined. Horizontal bars represent index medians. Figure 17A. Shannon-Wiener Diversity Index. Figure 17B. Simpson's Diversity Index.

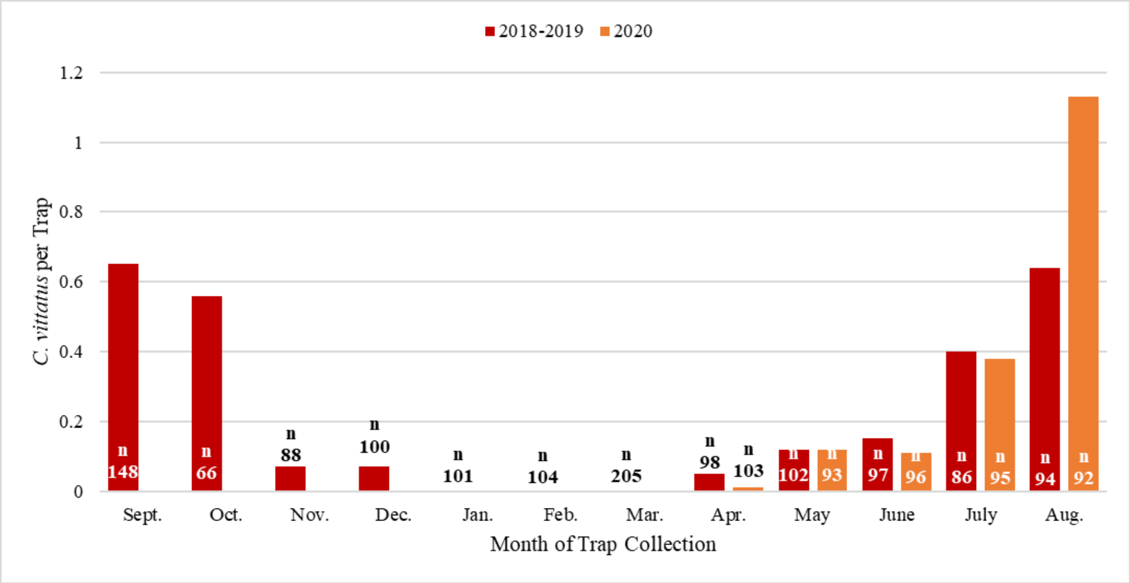

Figure 18. Mean number of *Centruroides vittatus* captured per trap by month of collection. Numbers (n) indicate the number of traps recovered, reflecting trap loss as well as unequal sampling efforts for March (two collection dates in 2019), September (collections were combined from six sites in 2018 and nine sites in 2019) and October (only six sites were sampled in 2018).

Figure 19. Number of *Centruroides vittatus* collected per site from both 2018-2019 and 2020 trapping seasons.

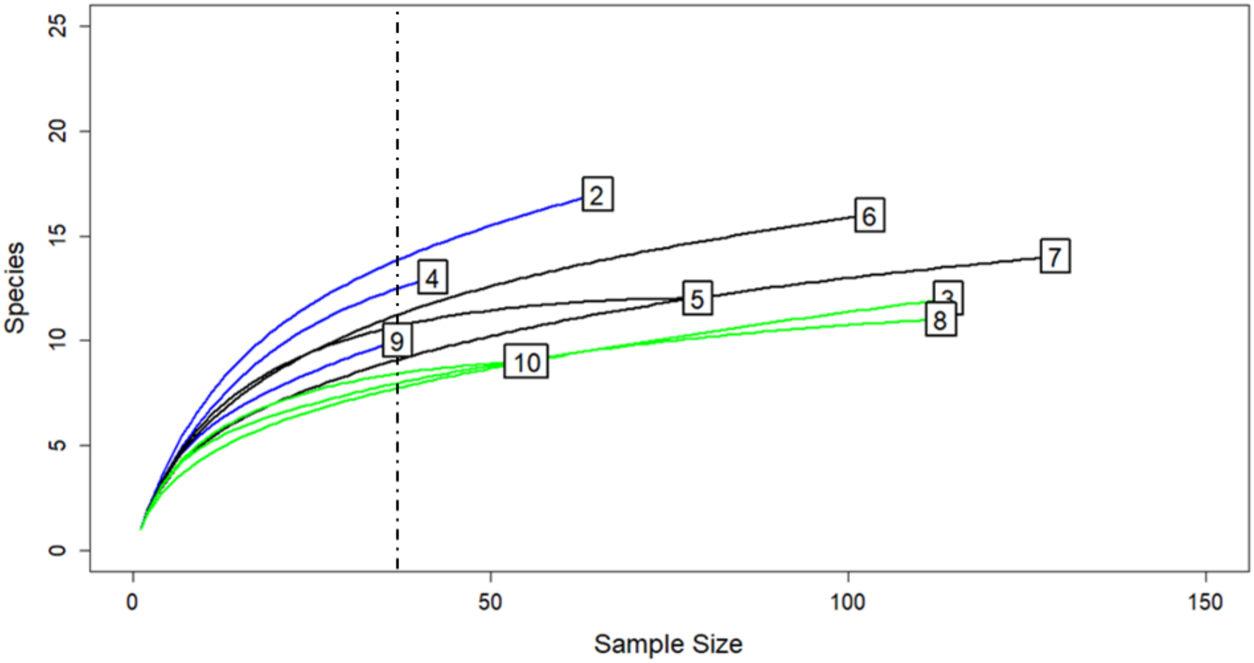

Figure 20. Species rarefaction curves for gnaphosid spiders captured in the 2018-2019 trapping season. Number labels on curves correspond to site numbers. Blue curves correspond to Degraded Site Types, black curves to Restored Site Types and green curves to Intact Site Types. Vertical line represents the smallest sample size, which was used for comparisons among sites.

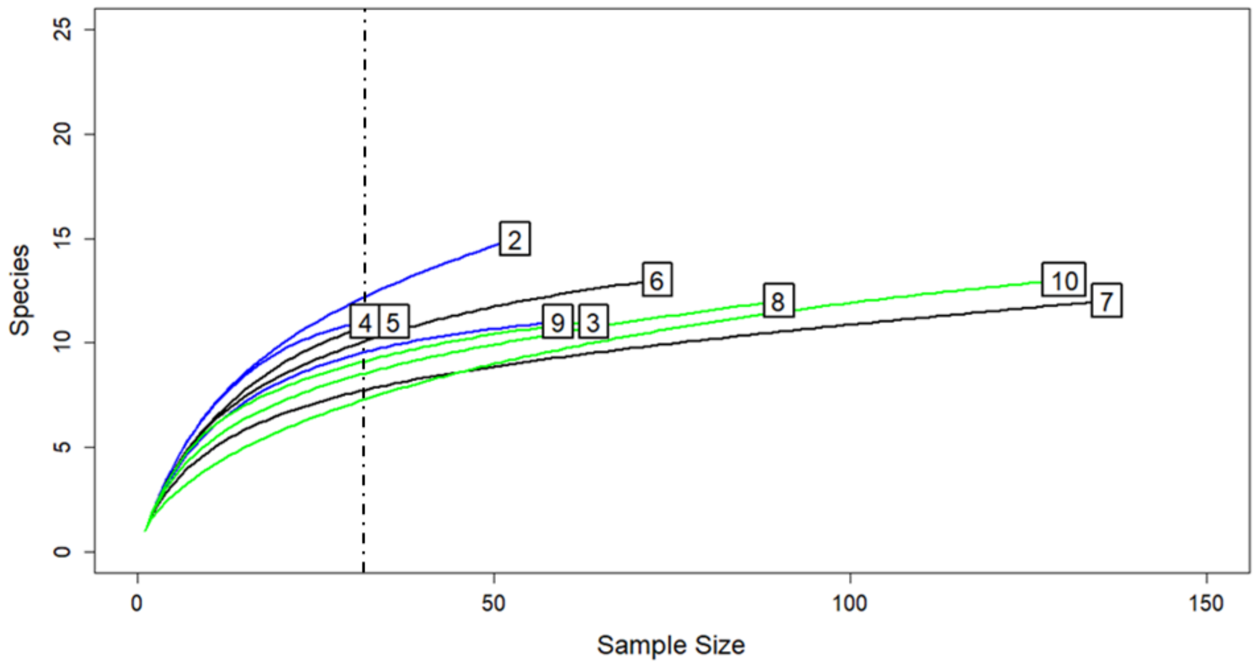

Figure 21. Species rarefaction curves for gnaphosid spiders captured in the 2020 trapping season. Number labels on curves correspond to site numbers. Blue curves correspond to Degraded Site Types, black curves to Restored Site Types and green curves to Intact Site Types. Vertical line represents the smallest sample size, which was used for comparisons among sites.

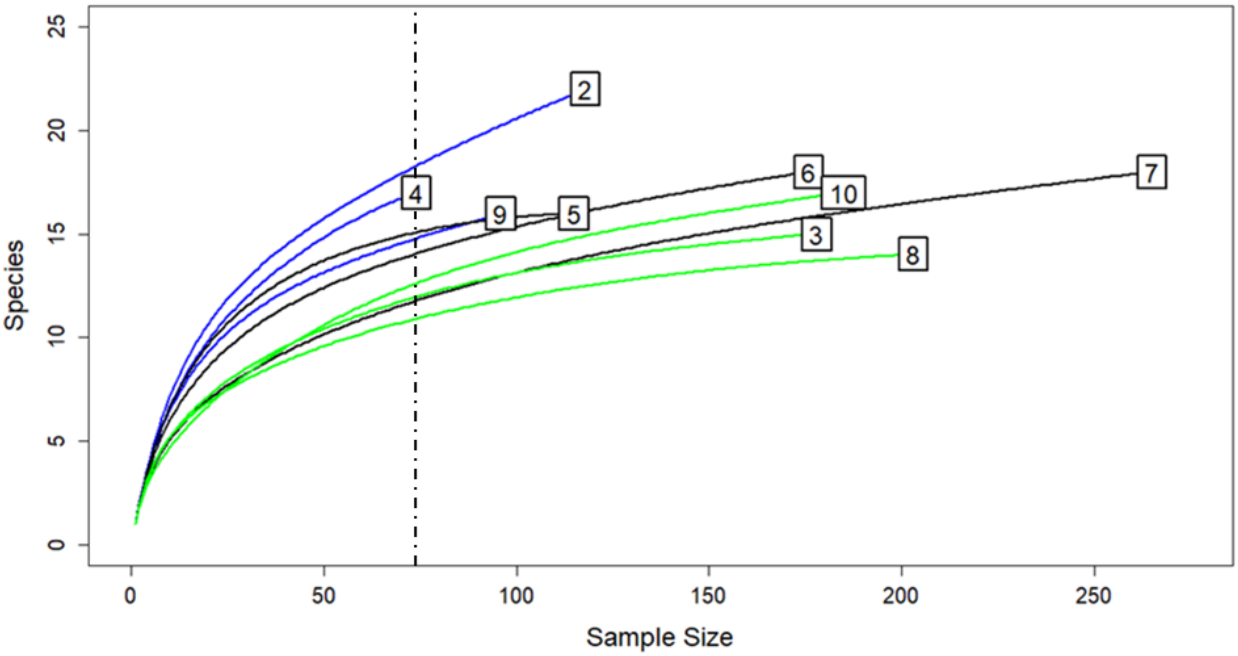

Figure 22. Species rarefaction curves for all adult gnaphosid spiders captured in both trapping seasons combined. Number labels on curves correspond to site numbers. Blue curves correspond to Degraded Site Types, black curves to Restored Site Types and green curves to Intact Site Types. Vertical line represents the smallest sample size, which was used for comparisons among sites.

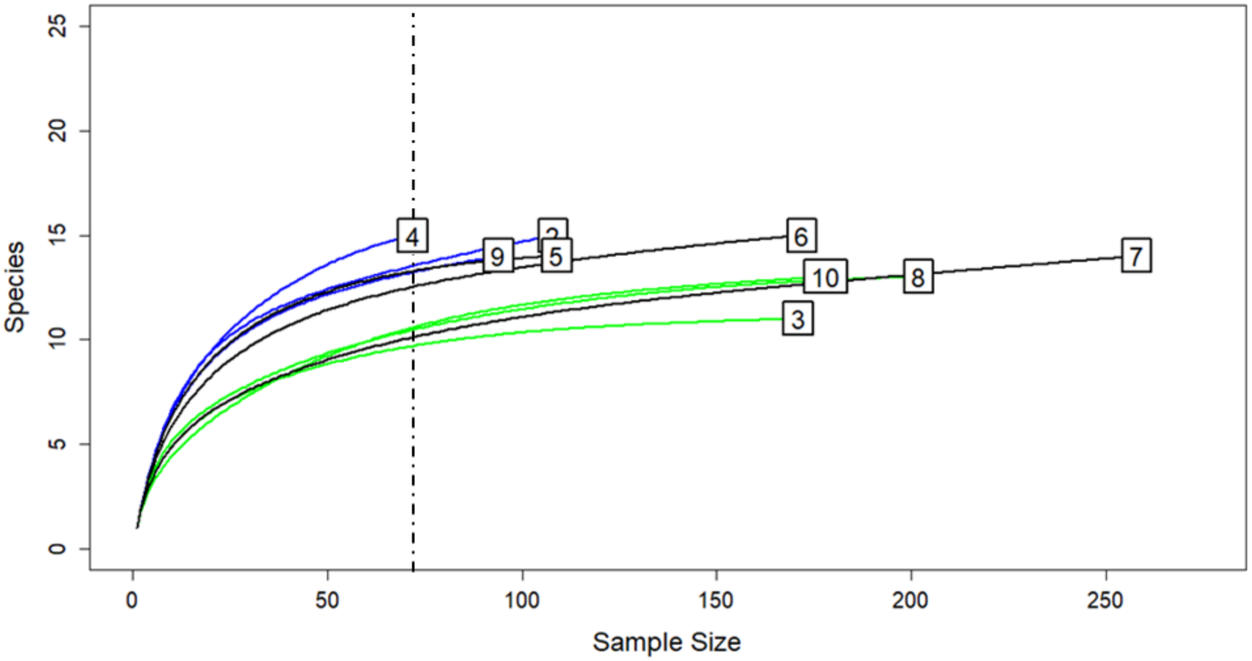

Figure 23. Species rarefaction curves for gnaphosid spiders with ≥ 10 individuals total collected in both trapping seasons combined. Number labels on curves correspond to site numbers. Blue curves correspond to Degraded Site Types, black curves to Restored Site Types and green curves to Intact Site Types. Vertical line represents the smallest sample size, which was used for comparisons among sites.

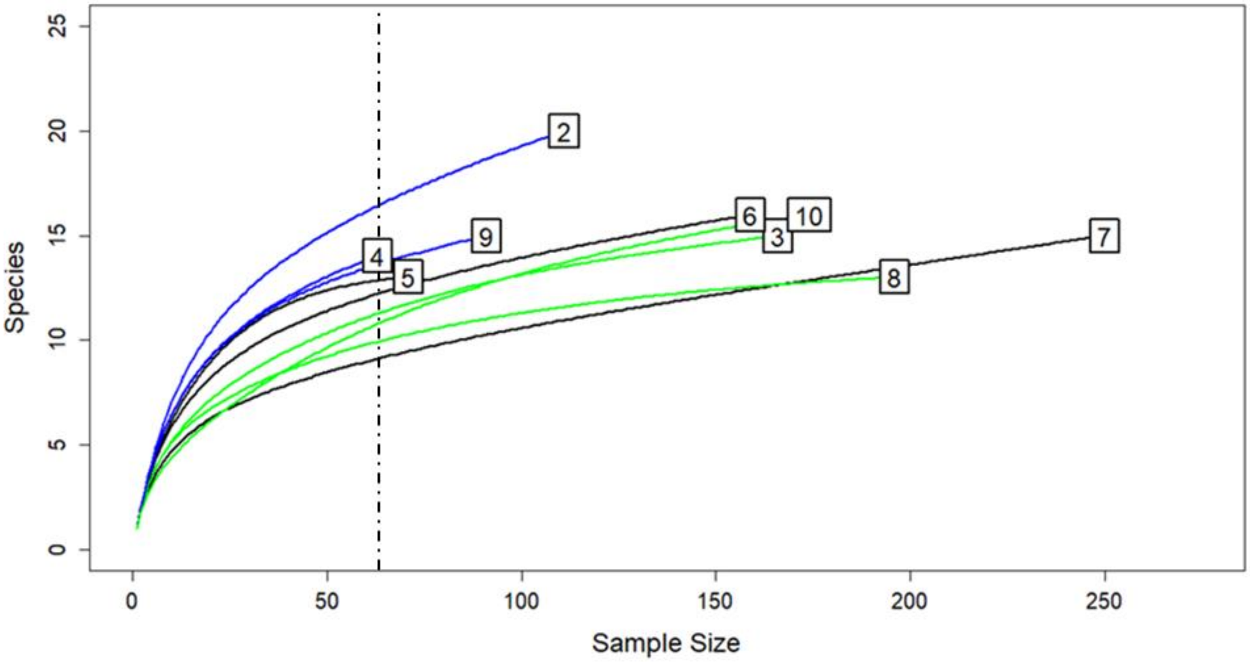

Figure 24. Species rarefaction curves for gnaphosid spiders captured in April-August of both trapping seasons combined. Number labels on curves correspond to site numbers. Blue curves correspond to Degraded Site Types, black curves to Restored Site Types and green curves to Intact Site Types. Vertical line represents the smallest sample size, which was used for comparisons among sites.

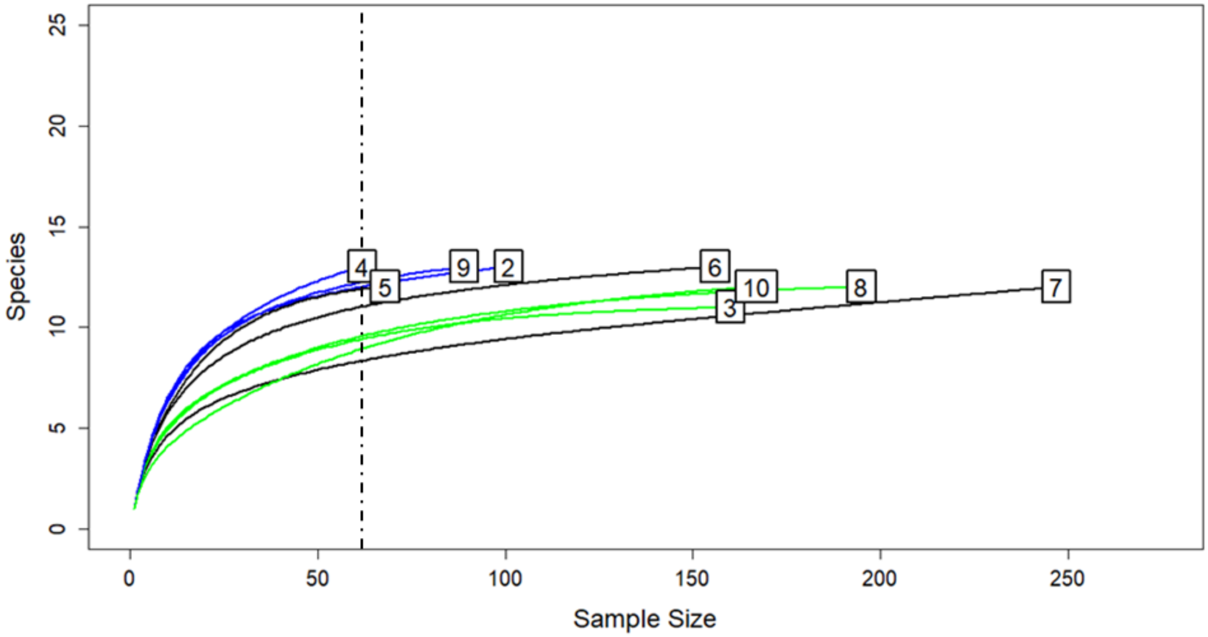

Figure 25. Species rarefaction curves for gnaphosid species with ≥ 10 individuals collected, captured in April-August of both trapping seasons combined. Number labels on curves correspond to site numbers. Blue curves correspond to Degraded Site Types, black curves to Restored Site Types and green curves to Intact Site Types. Vertical line represents the smallest sample size, which was used for comparisons among sites.

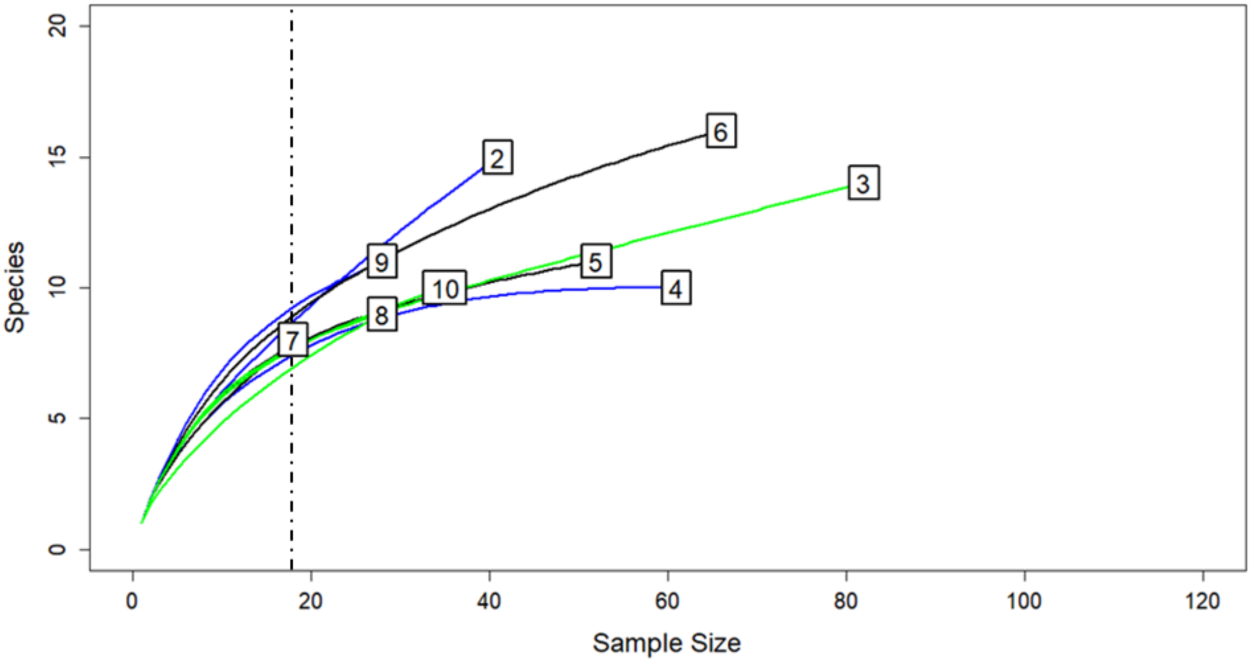

Figure 26. Species rarefaction curves for predatory carabid beetles captured in the 2018-2019 trapping season. Number labels on curves correspond to site numbers. Blue curves correspond to Degraded Site Types, black curves to Restored Site Types and green curves to Intact Site Types. Vertical line represents the smallest sample size, which was used for comparisons among sites.

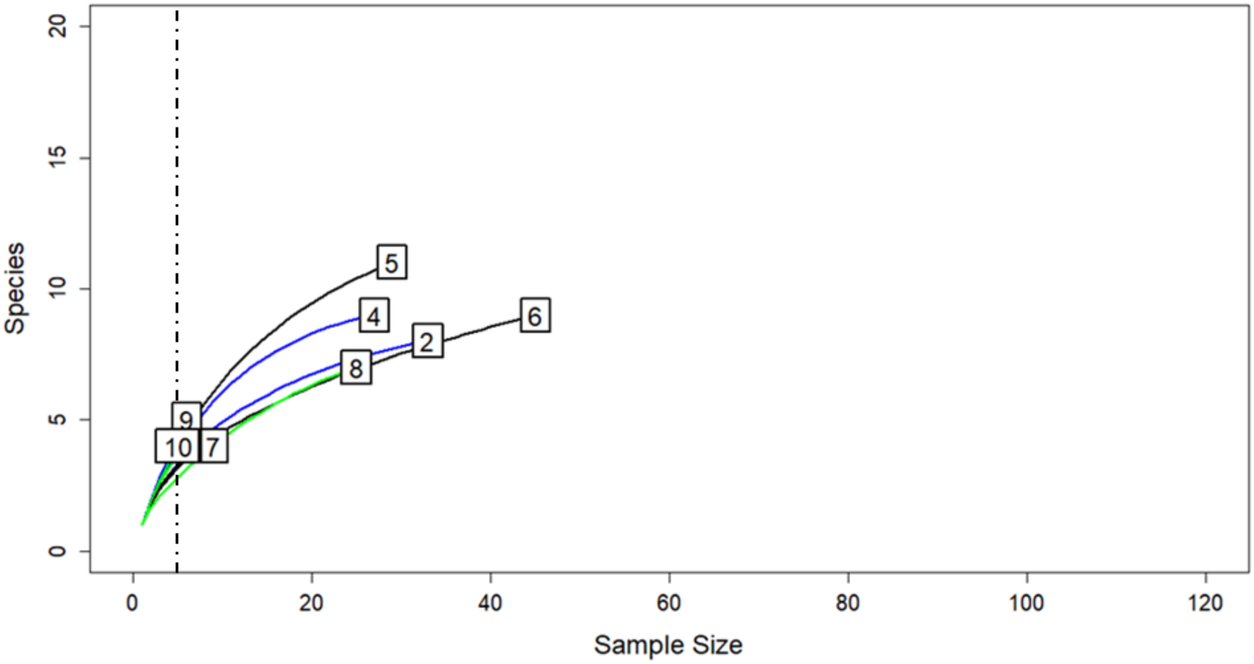

Figure 27. Species rarefaction curves for predatory carabid beetles captured in the 2020 trapping season. Number labels on curves correspond to site numbers. Blue curves correspond to Degraded Site Types, black curves to Restored Site Types and green curves to Intact Site Types. Vertical line represents the smallest sample size, which was used for comparisons among sites.

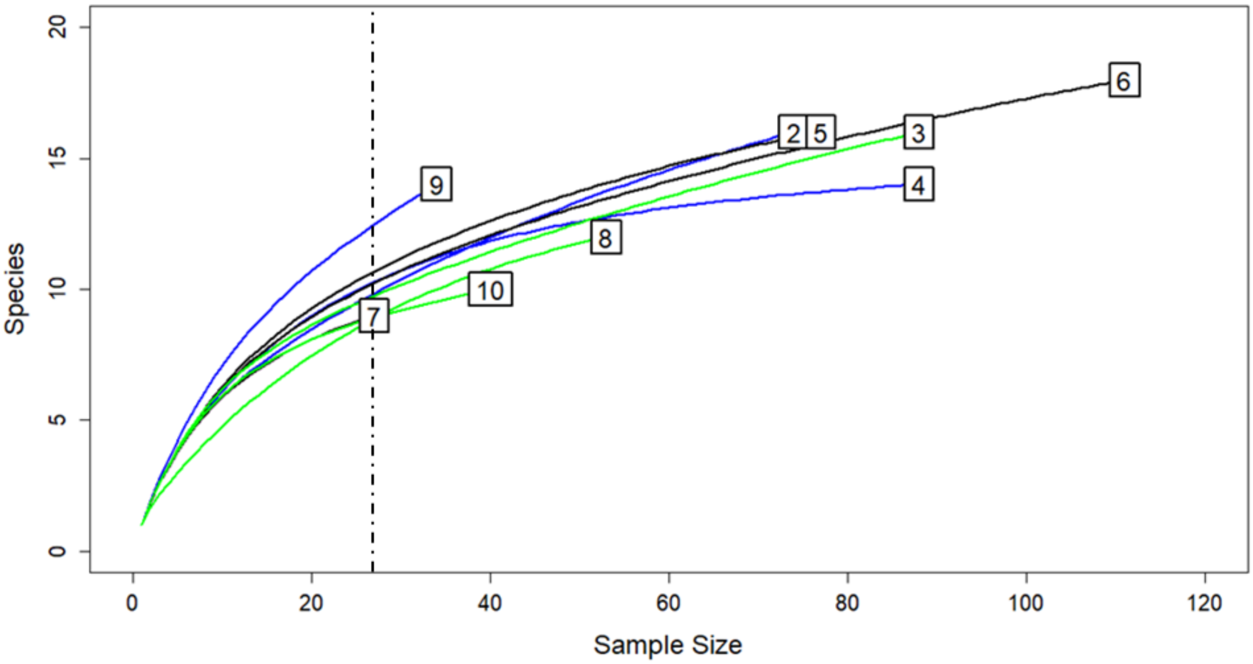

Figure 28. Species rarefaction curves for predatory carabids captured in both trapping seasons combined. Number labels on curves correspond to site numbers. Blue curves correspond to Degraded Site Types, black curves to Restored Site Types and green curves to Intact Site Types. Vertical line represents the smallest sample size, which was used for comparisons among sites.

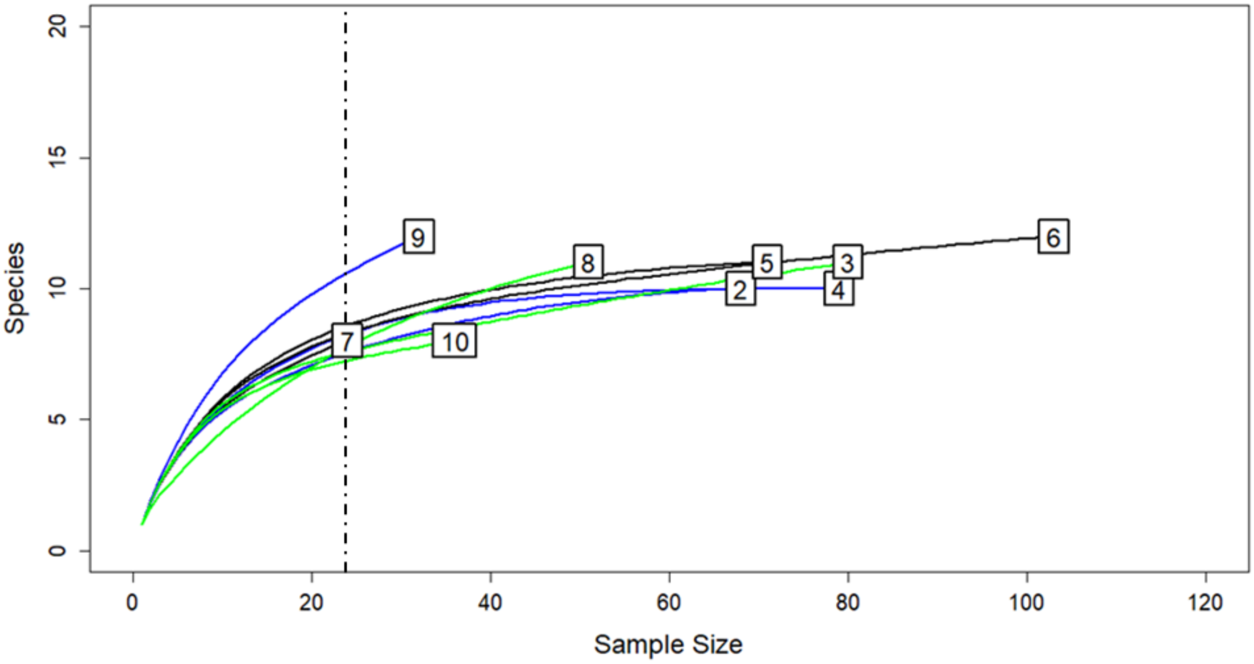

Figure 29. Species rarefaction curves for predatory carabid species with ≥ 10 individuals in both trapping seasons combined. Number labels on curves correspond to site numbers. Blue curves correspond to Degraded Site Types, black curves to Restored Site Types and green curves to Intact Site Types. Vertical line represents the smallest sample size, which was used for comparisons among sites.

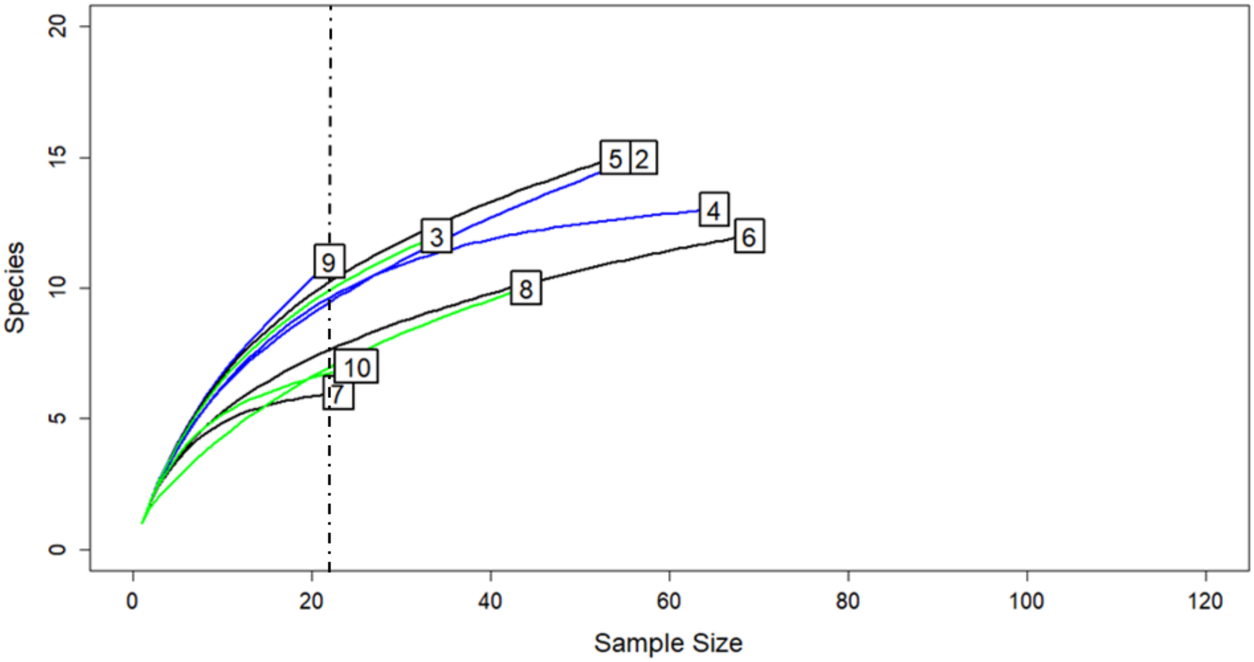

Figure 30. Species rarefaction curves for predatory carabids captured in April-August of both trapping seasons combined for each site. Number labels on curves correspond to site numbers. Blue curves correspond to Degraded Site Types, black curves to Restored Site Types and green curves to Intact Site Types. Vertical line represents the smallest sample size, which was used for comparisons among sites.

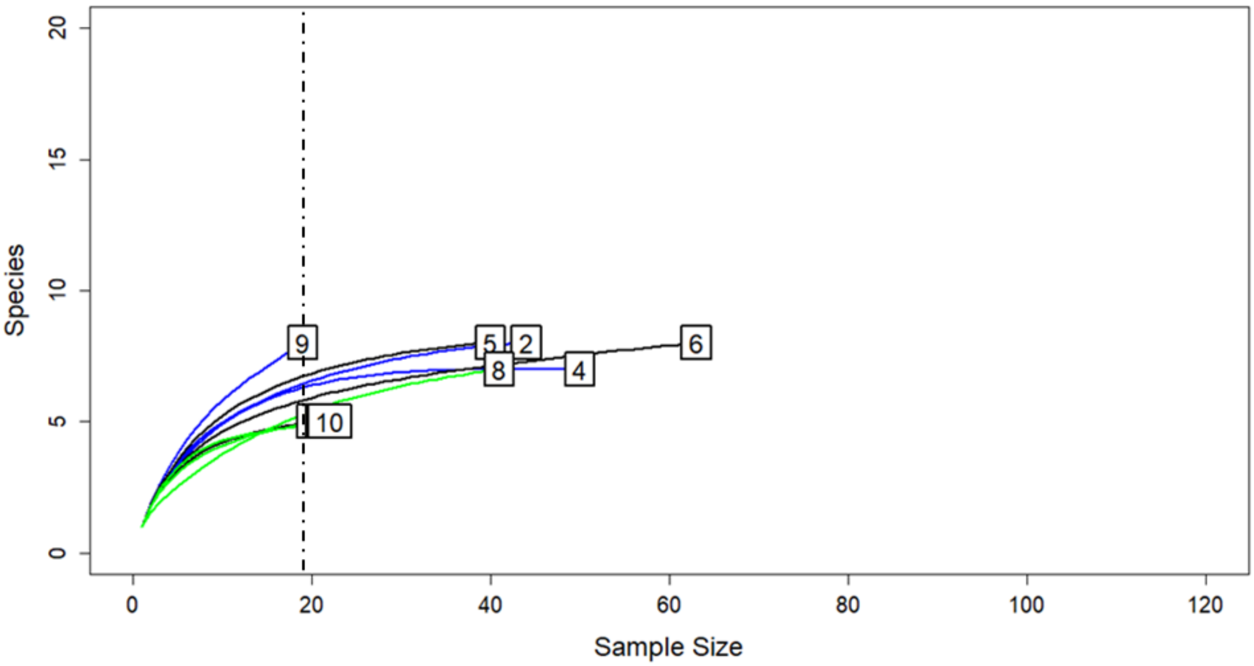

Figure 31. Species rarefaction curves for predatory carabid species captured at each site in April-August of both trapping seasons combined, trimmed to include only species with ≥ 10 individuals collected. Number labels on curves correspond to site numbers. Blue curves correspond to Degraded Site Types, black curves to Restored Site Types and green curves to Intact Site Types. Vertical line represents the smallest sample size, which was used for comparisons among sites.

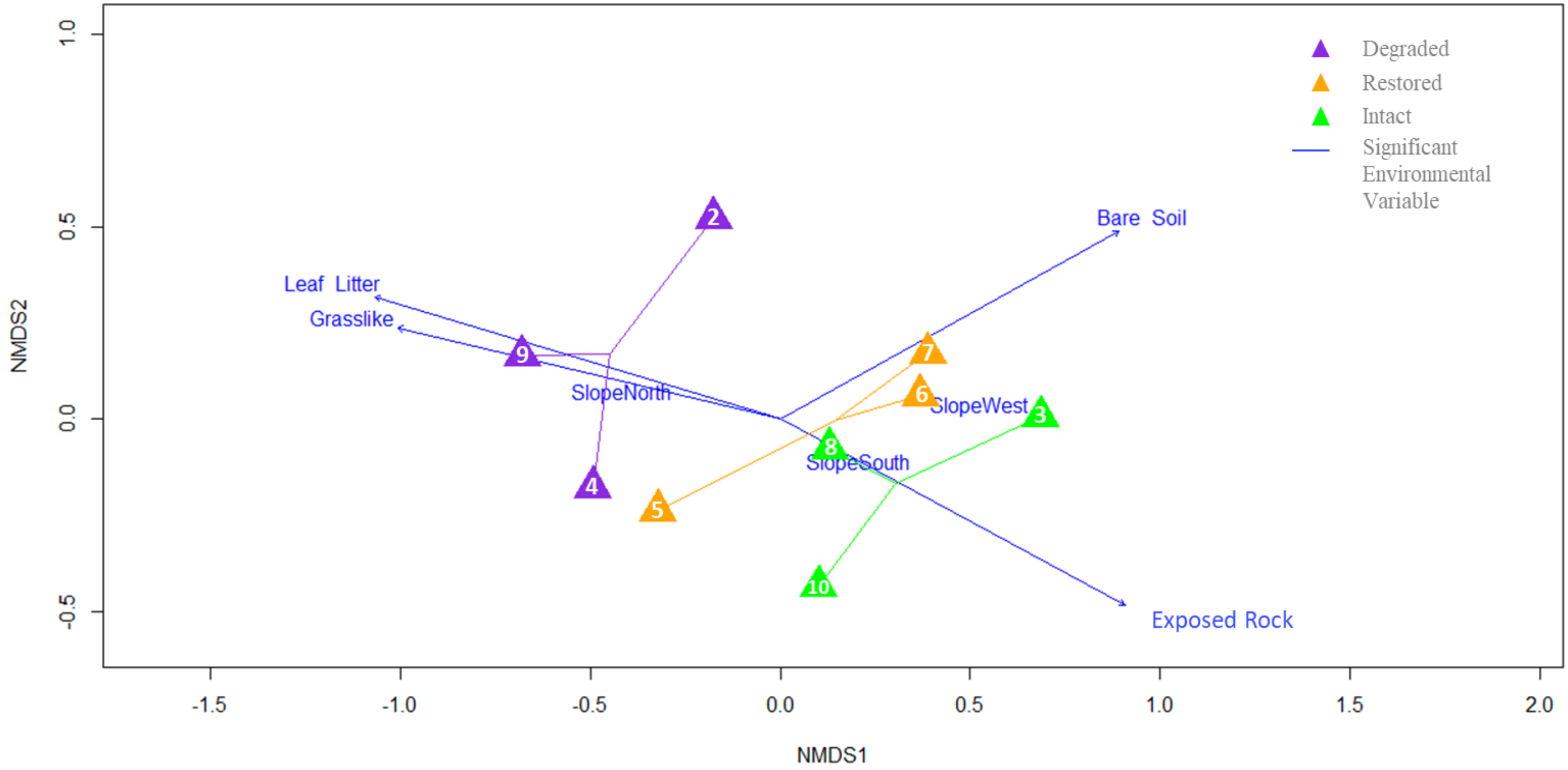

Figure 32. NMDS ordination of gnaphosid data from the 2018-2019 trapping season. Sites (numbered) and Site Types (colors) are shown with significant ($\alpha = 0.05$) environmental variables (Table 20). Lines connecting sites within a Site Type coincide at the centroid (midpoint of triangulated site medians).

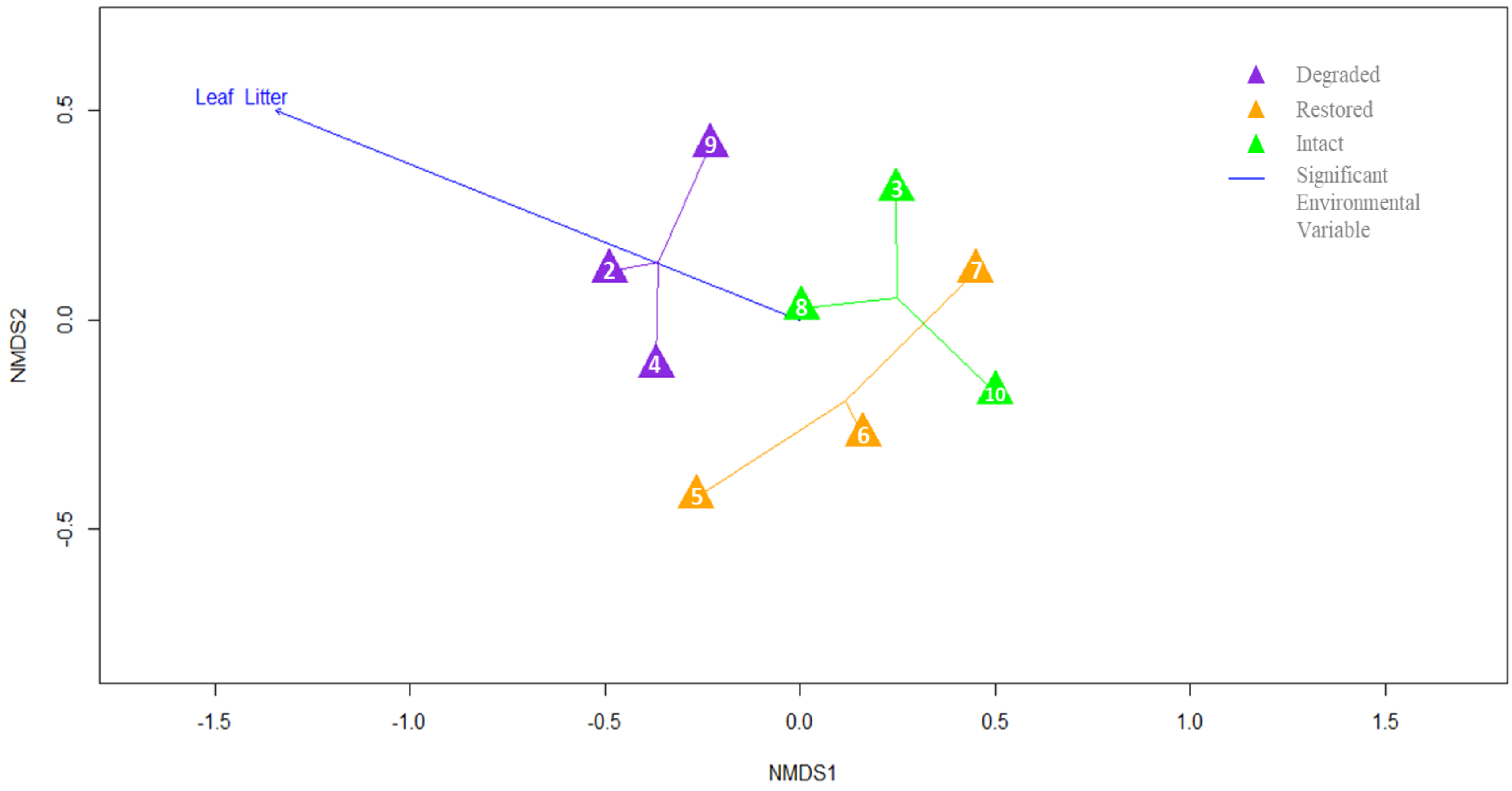

Figure 33. NMDS ordination of gnaphosid data from the 2020 trapping season. Sites (numbered) and Site Types (colors) are shown with significant (α = 0.05) environmental variables (Table 21). Lines connecting sites within a Site Type coincide at the centroid (midpoint of triangulated site medians).

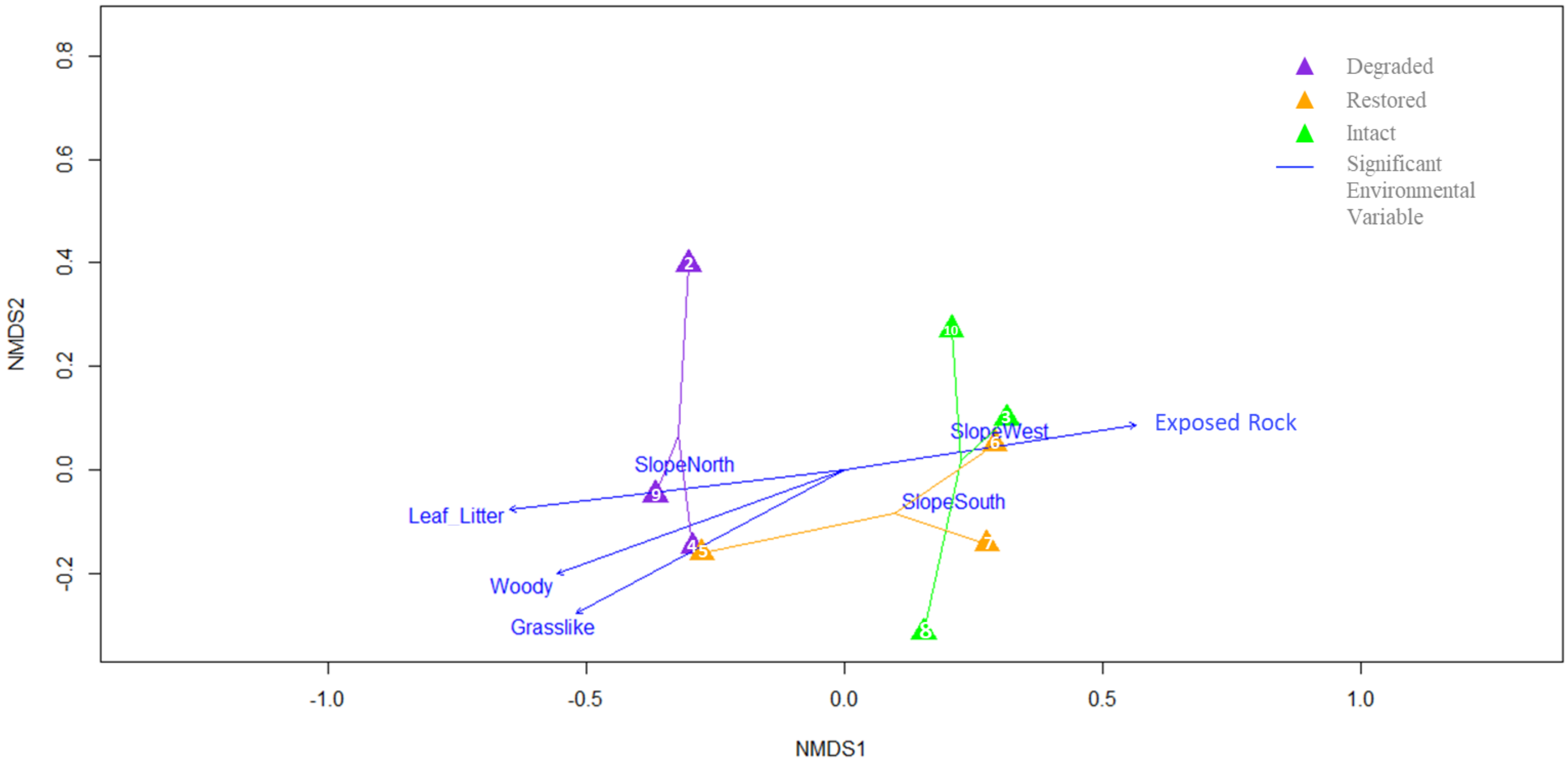

Figure 34. NMDS ordination of gnaphosid data from the 2018-2019 and 2020 trapping seasons. Sites (numbered) and Site Types (colors) are shown with significant (α = 0.05) environmental variables (Table 22). Lines connecting sites within a Site Type coincide at the centroid (midpoint of triangulated site medians).

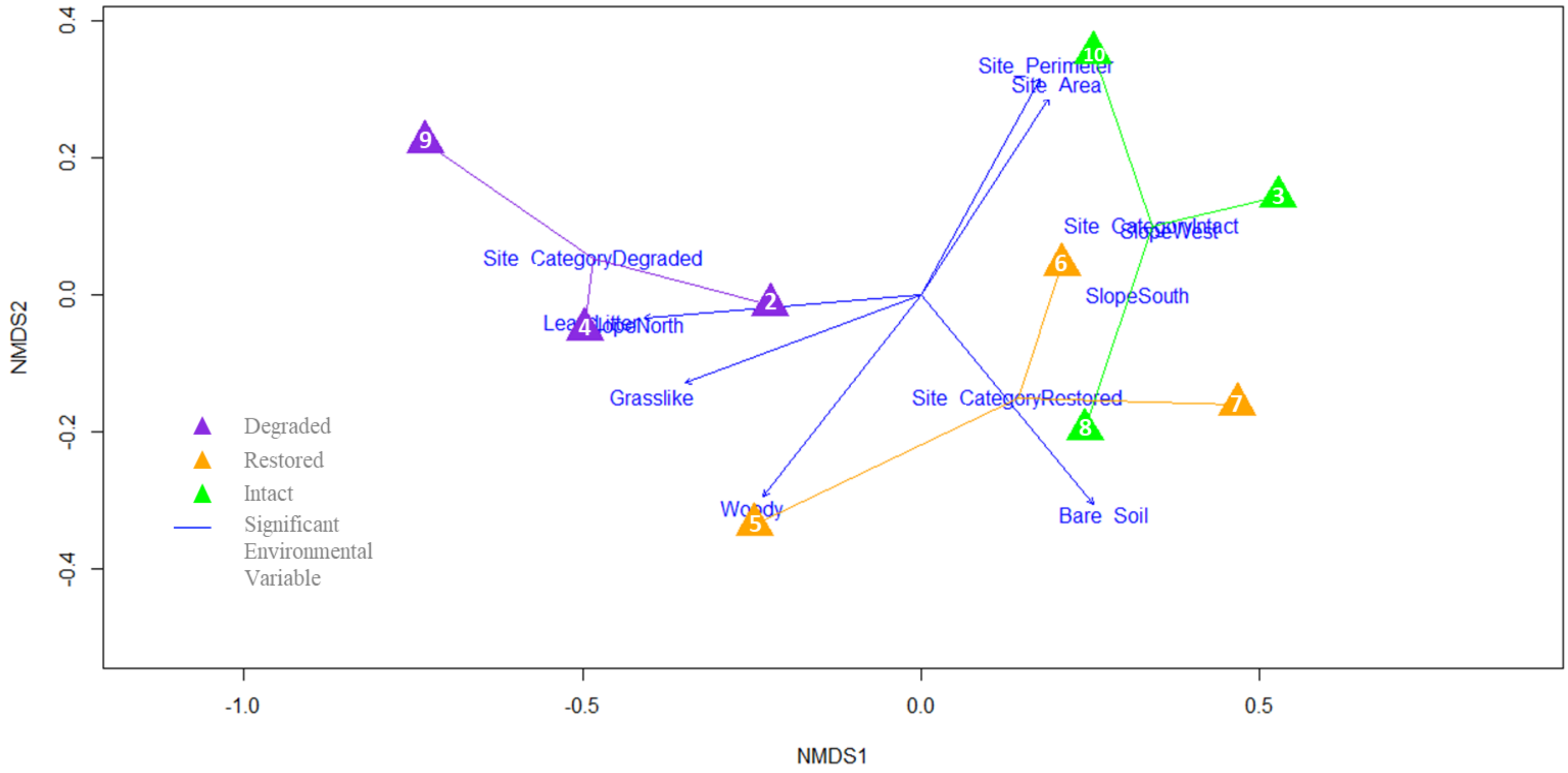

Figure 35. NMDS ordination of gnaphosid data from the 2018-2019 trapping season, trimmed to include only species with ≥ 10 individuals collected. Sites (numbered) and Site Types (colors) are shown with significant (α = 0.05) environmental variables (Table 23). Lines connecting sites within a Site Type coincide at the centroid (midpoint of triangulated site medians).

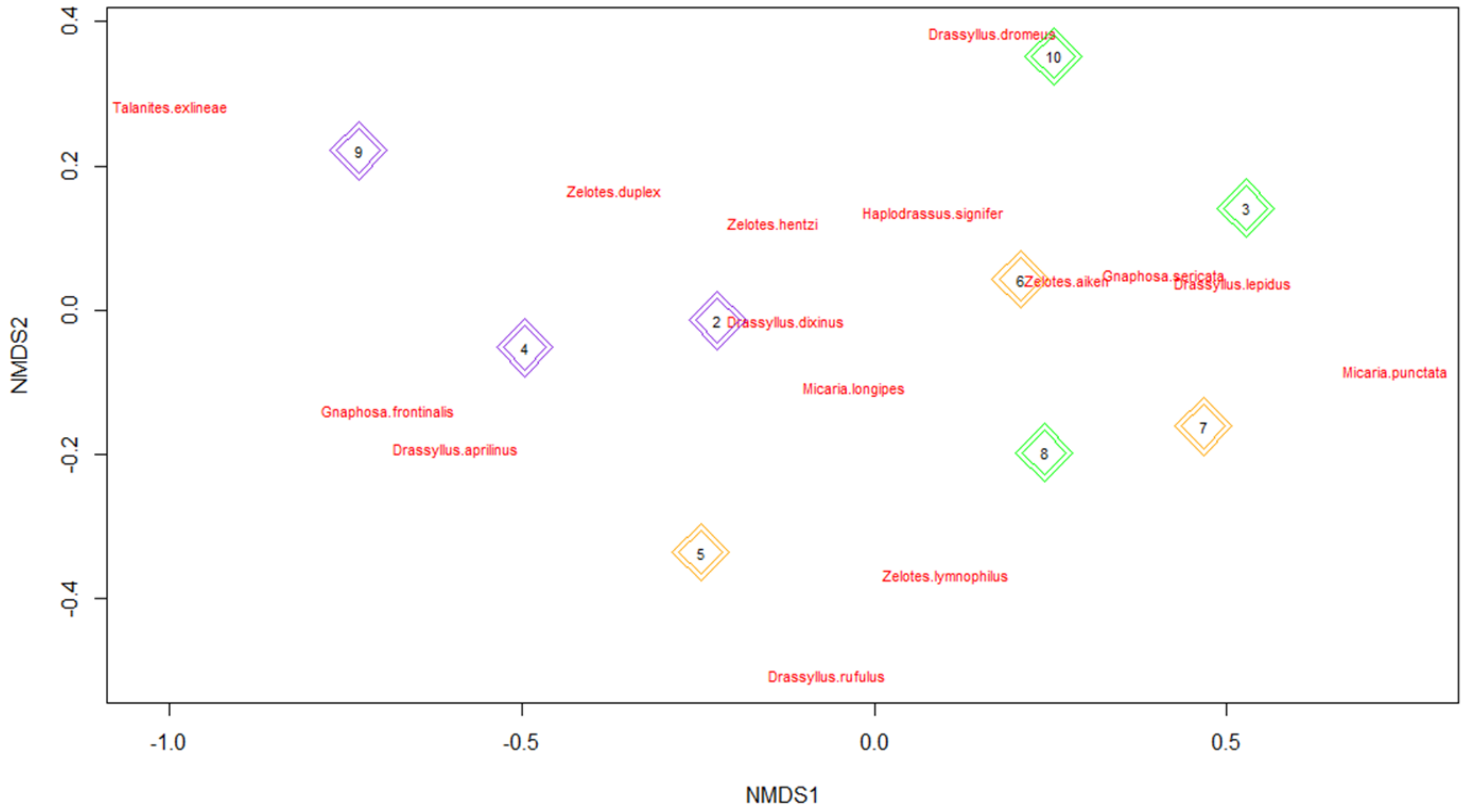

Figure 36. NMDS ordination of gnaphosid data from the 2018-2019 trapping season trimmed to species with ≥ 10 individuals collected, showing species and sites (purple = Degraded Site Type, yellow = Restored Site Type, green = Intact Site Type).

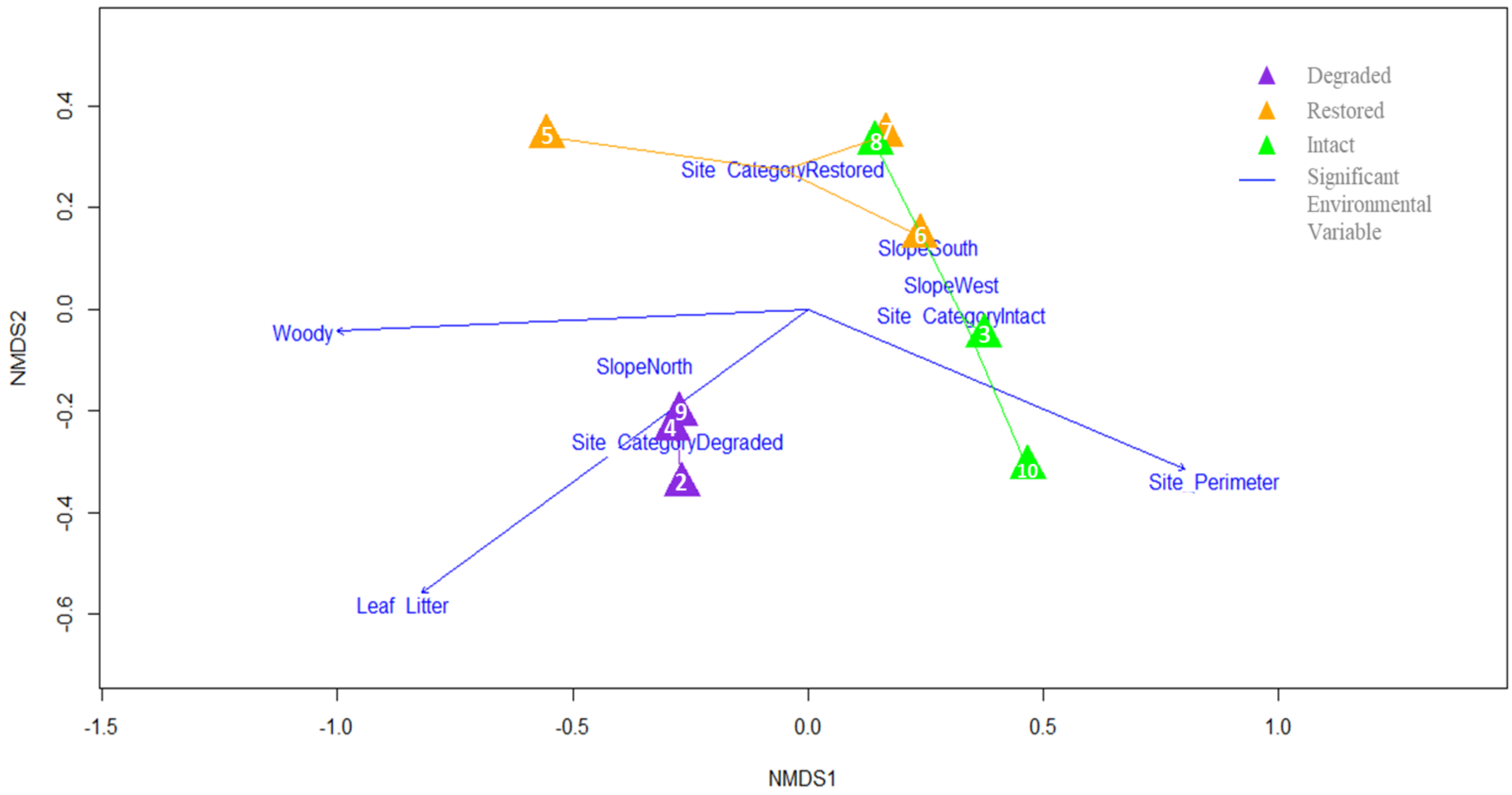

Figure 37. NMDS ordination of gnaphosid data from the 2020 trapping season, trimmed to include only species with ≥ 10 individuals collected. Sites (numbered) and Site Types (colors) are shown with significant (α = 0.05) environmental variables (Table 24). Lines connecting sites within a Site Type coincide at the centroid (midpoint of triangulated site medians).

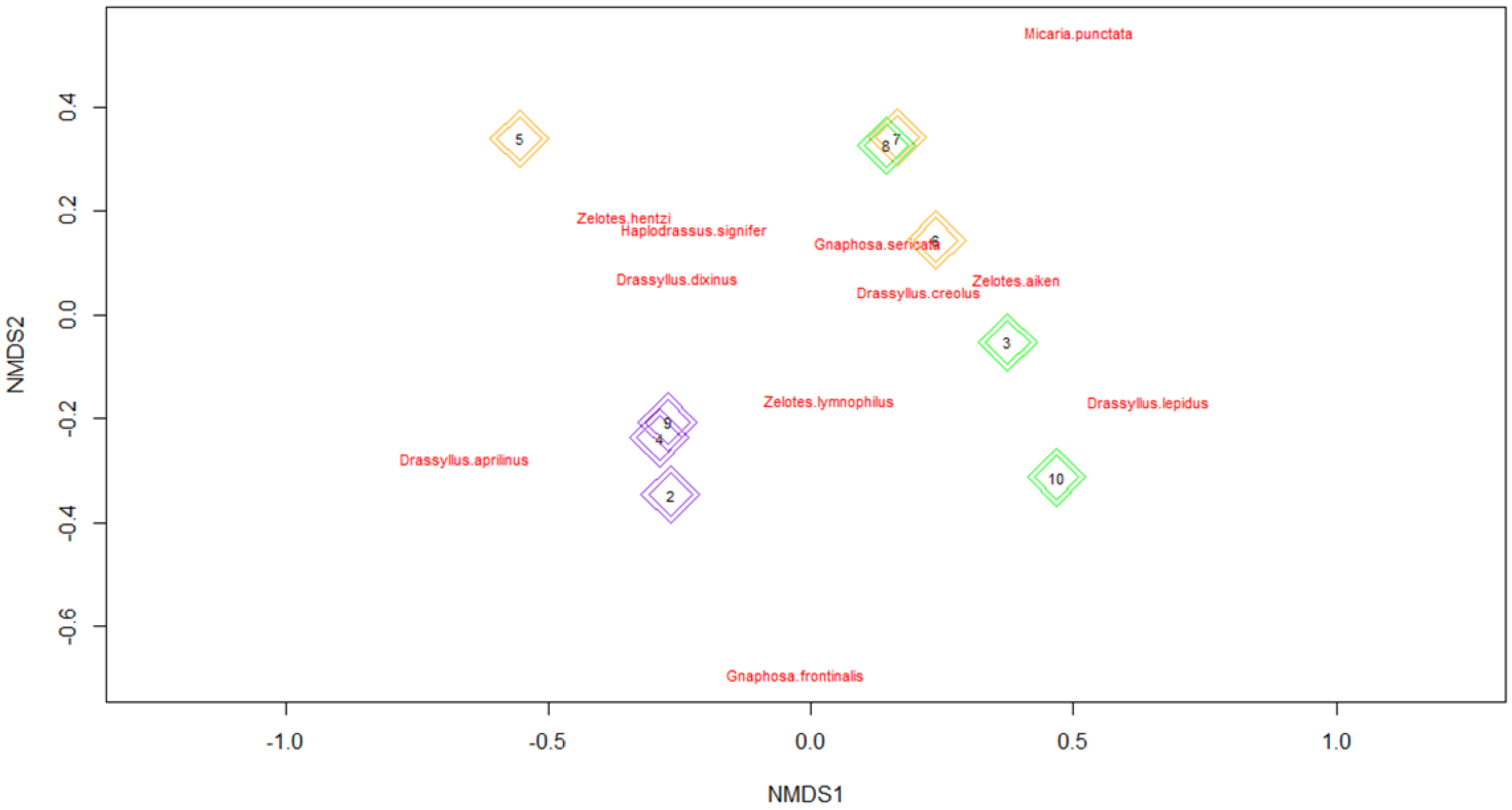

Figure 38. NMDS ordination of gnaphosid data from the 2020 trapping season trimmed to species with ≥ 10 individuals collected, showing species and sites (purple = Degraded Site Type, yellow = Restored Site Type, green = Intact Site Type).

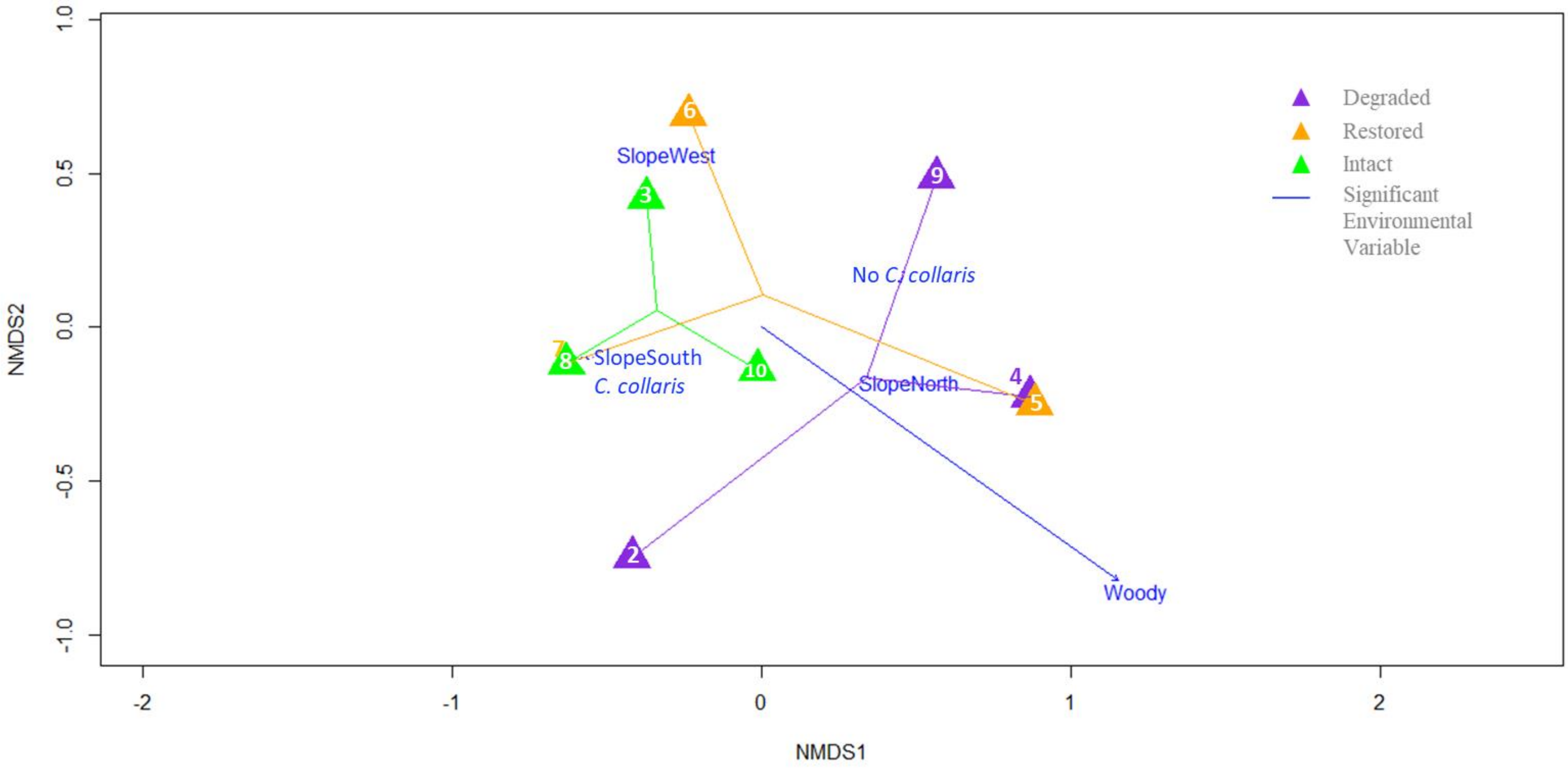

Figure 39. NMDS ordination of predatory carabid data from the 2018-2019 trapping season. Sites (numbered) and Site Types (colors) are shown with significant (α = 0.05) environmental variables (Table 25). Lines connecting sites within a Site Type coincide at the centroid (midpoint of triangulated site medians).

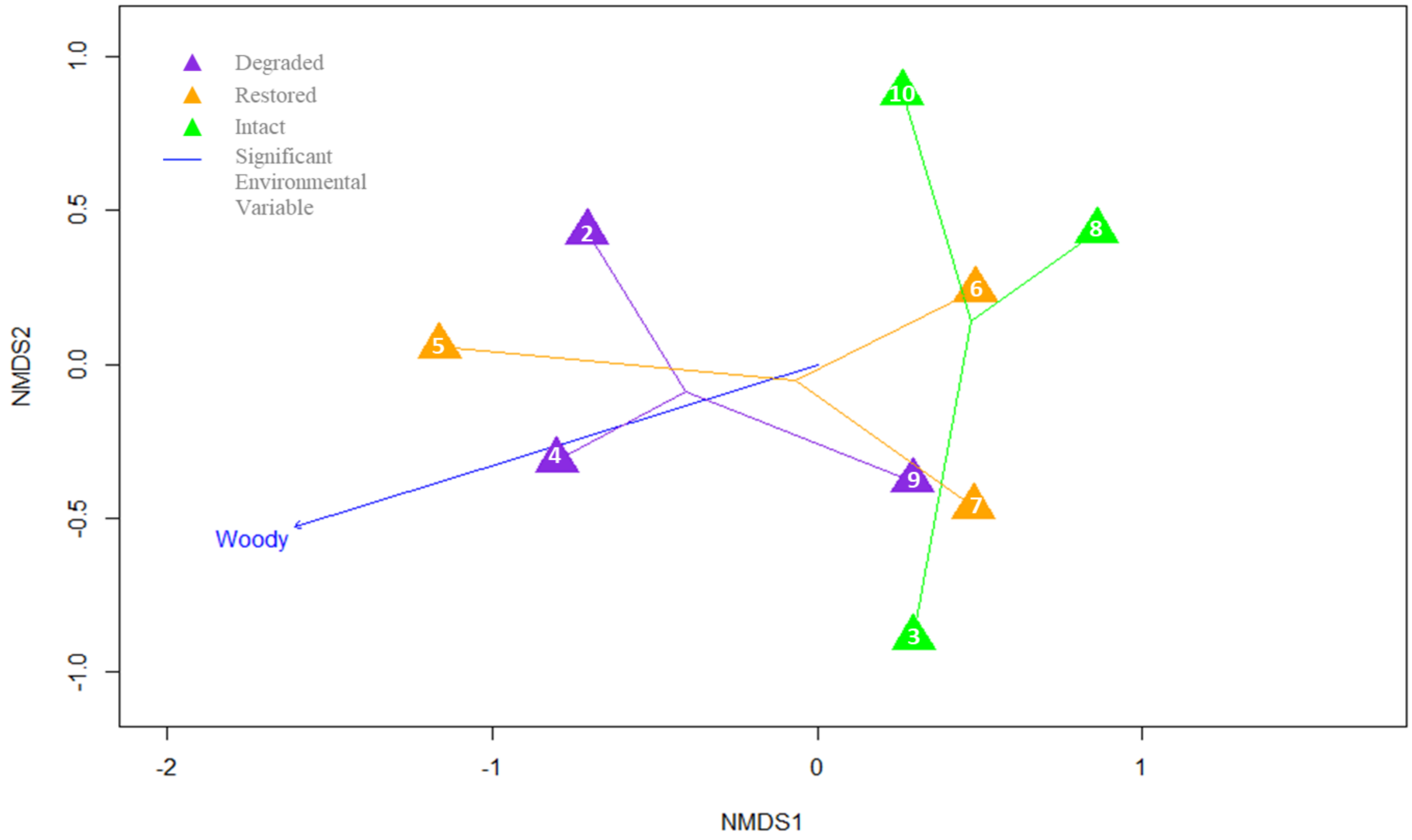

Figure 40. NMDS ordination of predatory carabid data from the 2020 trapping season. Sites (numbered) and Site Types (colors) are shown with significant (α = 0.05) environmental variables (Table 26). Lines connecting sites within a Site Type coincide at the centroid (midpoint of triangulated site medians).

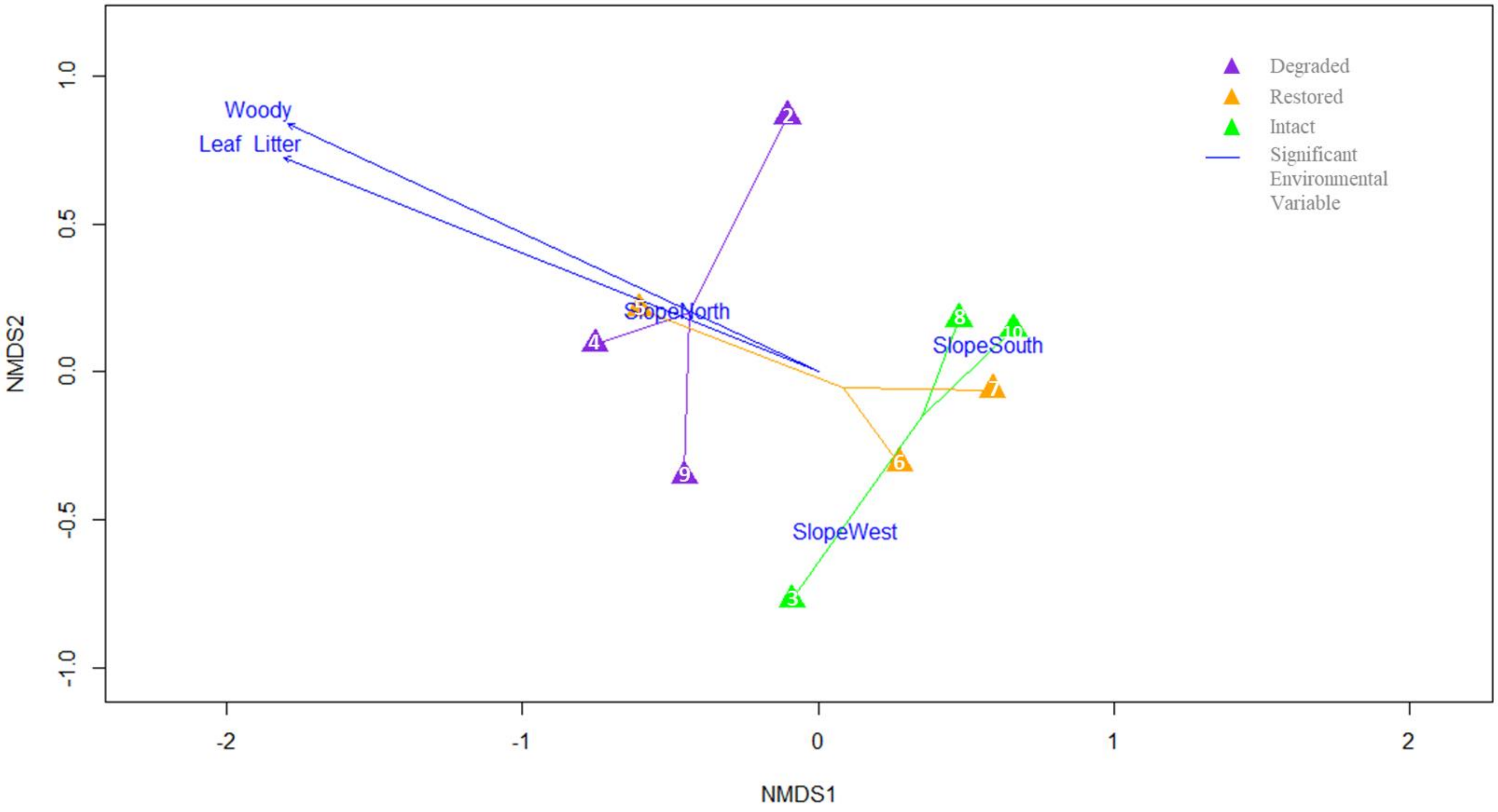

Figure 41. NMDS ordination of predatory carabid data from the 2018-2019 and 2020 trapping seasons combined. Sites (numbered) and Site Types (colors) are shown with significant (α = 0.05) environmental variables (Table 27). Lines connecting sites within a Site Type coincide at the centroid (midpoint of triangulated site medians).

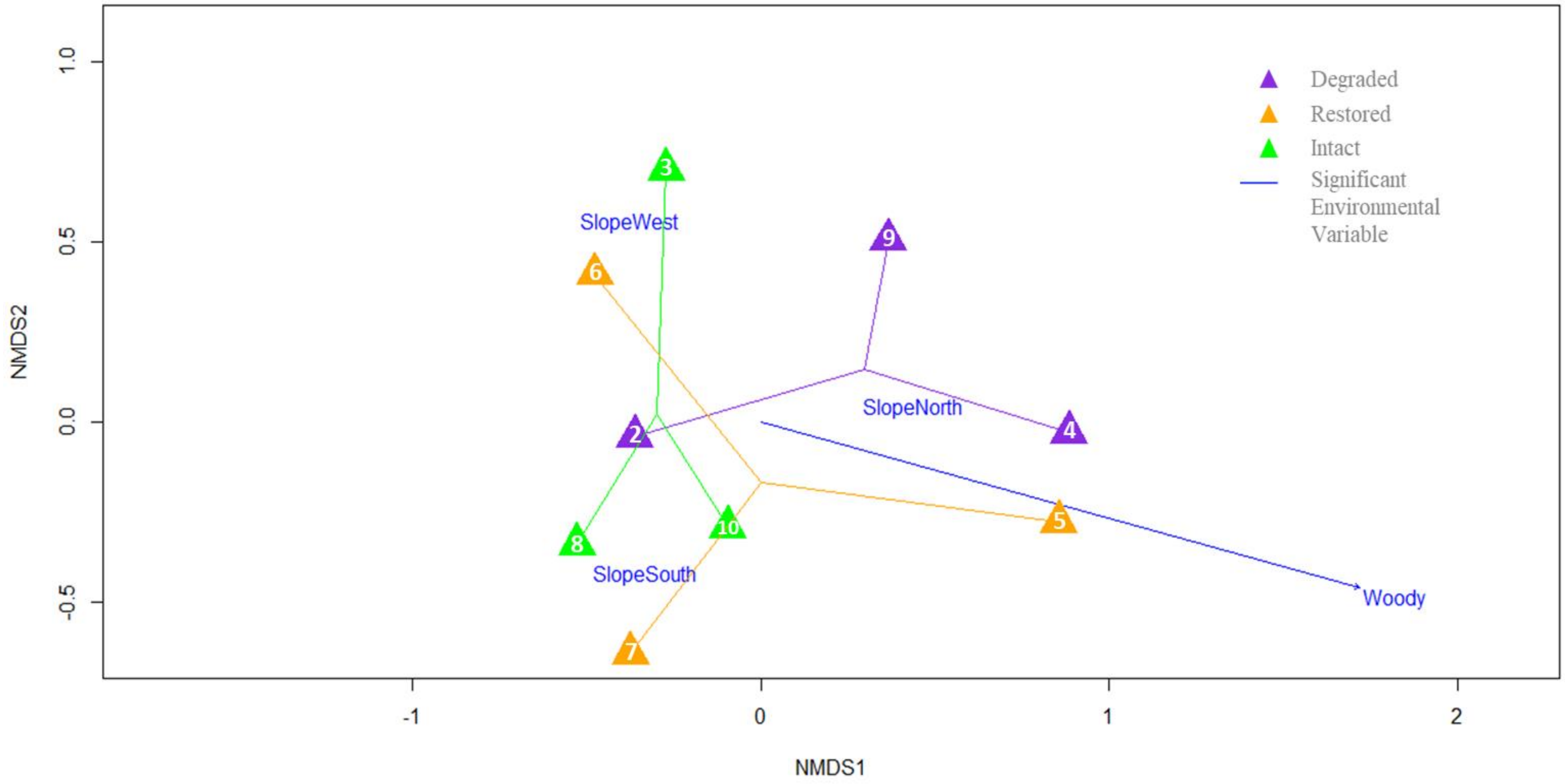

Figure 42. NMDS ordination of predatory carabid data from the 2018-2019 trapping season trimmed to include only species with ≥ 10 individuals collected. Sites (numbered) and Site Types (colors) are shown with significant (α = 0.05) environmental variables (Table 28). Lines connecting sites within a Site Type coincide at the centroid (midpoint of triangulated site medians).

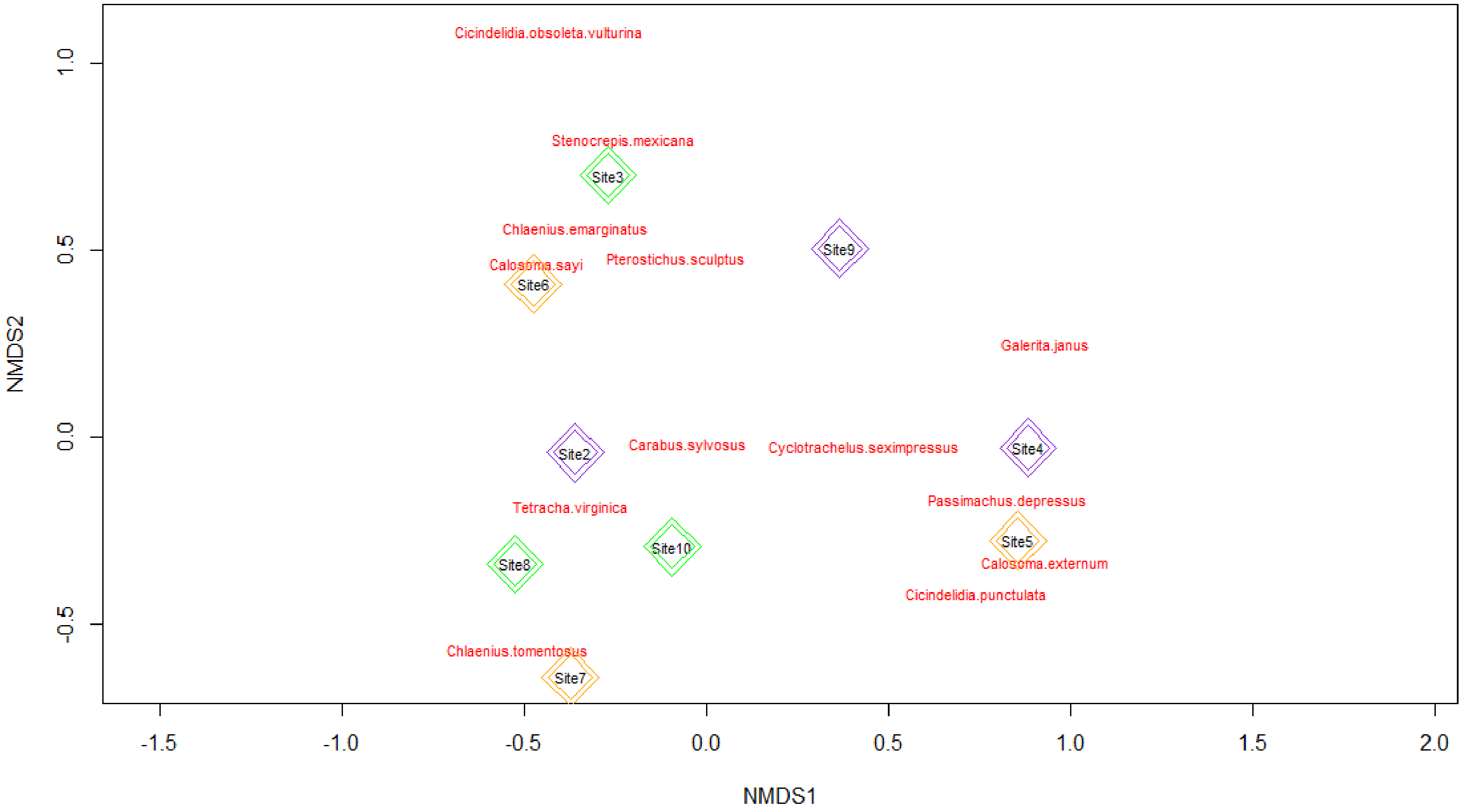

Figure 43. NMDS ordination of predatory carabid data from the 2018-2019 trapping season trimmed to species with ≥ 10 individuals collected, showing species and sites (purple = Degraded Site Type, yellow = Restored Site Type, green = Intact Site Type).

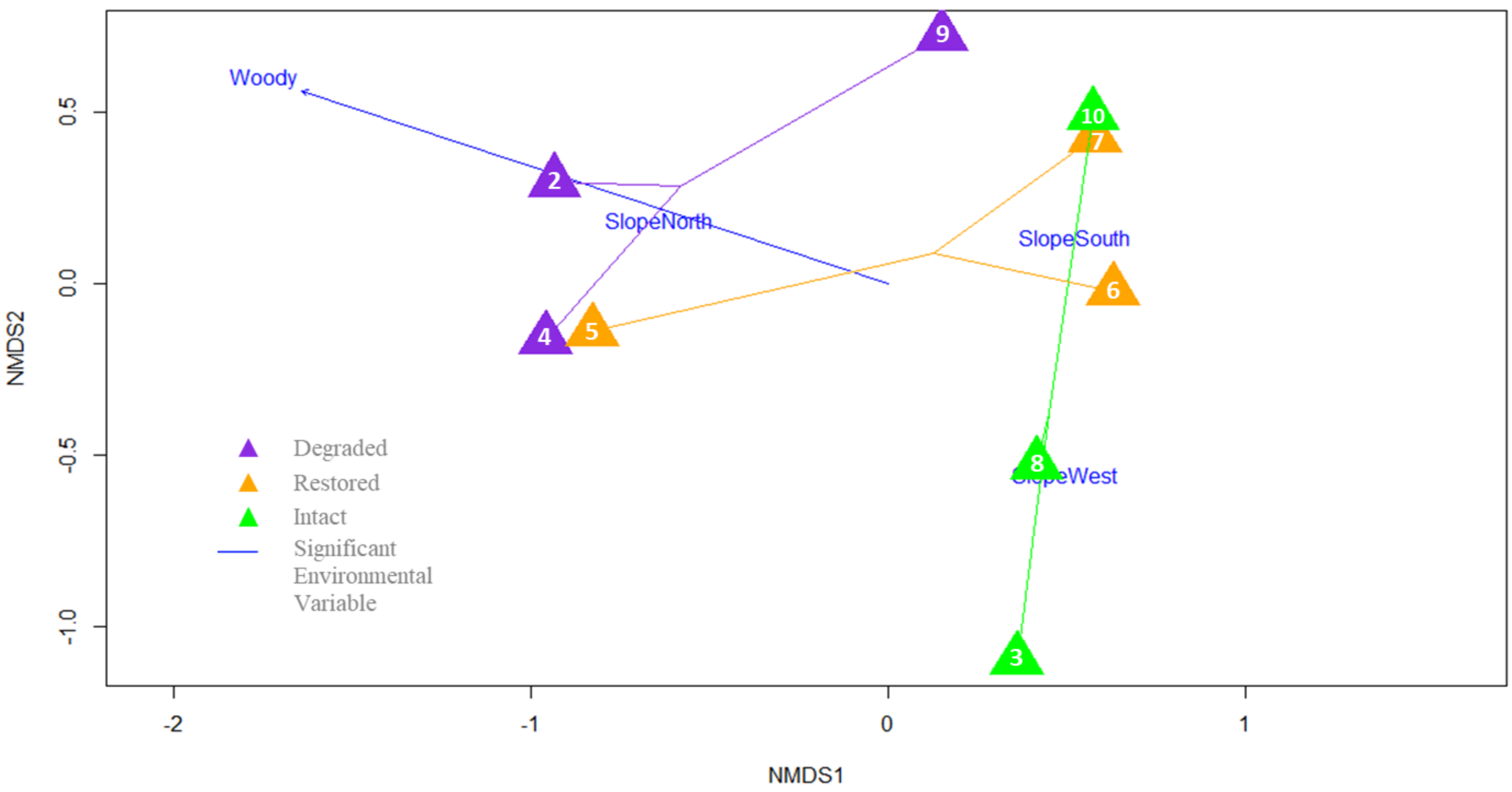

Figure 44. NMDS ordination of predatory carabid data from the 2020 trapping season, trimmed to species with ≥ 10 individuals collected. Sites (numbered) and Site Types (colors) are shown with significant (α = 0.05) environmental variables (Table 29). Lines connecting sites within a Site Type coincide at the centroid (midpoint of triangulated site medians).

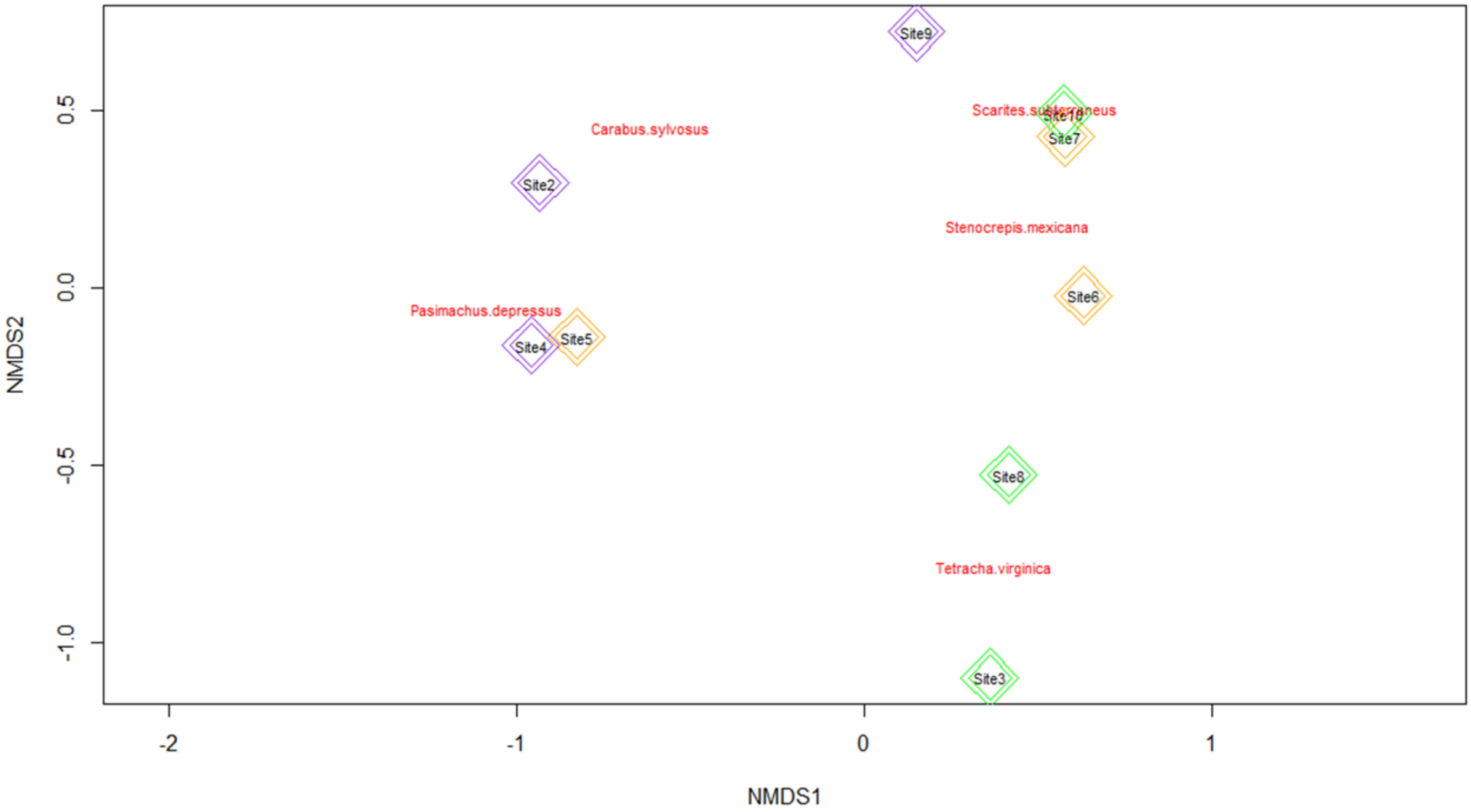

Figure 45. NMDS ordination of predatory carabid data from the 2020 trapping season trimmed to species with ≥ 10 individuals collected, showing species and sites (purple = Degraded Site Type, yellow = Restored Site Type, green = Intact Site Type).

APPENDICES

Appendix A. Synthesis of six studies of floral species reported from Ozark glades indicating the underlying bedrock type. Species reported from studies not distinguishing between limestone and dolomite bedrock are reported herein as calcareous.

Species	**Common Name**	**Limestone** [1, 6]	**Dolomite** [1]	**Calcareous** [3, 5]	**Sandstone** [2, 4]
Acacia angustissima var. *hirta*		X			
Acalypha monocca (Engelm. ex A. Gray)	one-seed mercury	X			
Acer saccharum Marshall	sugar maple			X	
Achillea millefolium L.	common milfoil				X
Agalinus skinneriana (Alph. Wood) Britton	pale gerardia			X	
Agrostis alba L.	redtop				X
Agrostis hyemalis (Walt.) BSP. var. *hyemalis*	hair grass				X
Allium canadense L. var. *lavendulare* (Bates) Owenbey & Aase	tall pink glade onion	X	X		
Allium mutabile Michx.					X
Allium stellatum Nutt. ex Ker Gawl	glade onion		X	X	X
Ambrosia artemisiifolia L.	common ragweed	X		X	X
Ambrosia bidentata Michx.	ragweed				X
Amelanchier humilis Wieg. var. *humilis*	low bush serviceberry				X
Amelanchier arborea (Michx. F.) Fern.	common serviceberry			X	
Amorpha canescens Pursh	lead plant			X	X
Amphicarpa bracteata (L.) Fern.	hog peanut			X	
Andrachne phyllanthoides (Nutt.) Voronts. & Petra Hoffm.	buckbrush, maidenbush	X			
Andropogon gerardii Vitman	big bluestem grass	X	X	X	X
Andropogon ternarius Michx.	split bluestem				X
Antennaria sp.	pussytoes	X	X		
Antennaria plantaginifolia (L.) Hook	pussytoes				X
Apocynum cannabinum L.	dogbane, Indian hemp		X		
Arabis canadensis L.	sickle-pod			X	X
Arenaria patula Michx.					X
Aristida dichotoma Michx.	poverty grass	X			X
Aristida intermedia Scribn. & C. R. Ball				X	
Aristida longespica Poiret var. *geniculata* (Raf.) Fern.	slimspike three-awn			X	
Aristida oligantha Michx.	prairie three-awn	X			
Asclepias hirtella (Pennell) Woodson					X
Asclepias quadrifolia Jacq.	four-leaf milkweed	X	X	X	
Asclepias stenophylla A. Gray	narrow-leaf milkweed			X	
Asclepias tuberosa L.	butterflyweed			X	X

Appendix A (cont.). Synthesis of six studies of floral species reported from Ozark glades.

Species	**Common Name**	**Limestone** [1, 6]	**Dolomite** [1]	**Calcareous** [3, 5]	**Sandstone** [2, 4]
Asclepias verticillata L.	whorled milkweed	X	X	X	X
Asclepias viridiflora Ra. var. *lanceolata* (Ives) Torrey var. *linearis* (A. Gray Fern.)	green milkweed			X	
Asclepias viridis Walter	green milkweed	X	X		
Aster laevis L.	smooth aster			X	
Aster oolentangiensis Riddell	azure aster			X	
Aster patens Alt.	spreading aster			X	X
Aster pilosus Willd.	white heath aster			X	X
Aster praealtus Poir.	willow-leaved aster				X
*Aster ptarmicoides (*Nees) T. & G.			X		
Aster sericeus Vent. *f. sericeus*	silky aster			X	
Aster turbinellis Lindley	turbin-headed aster			X	
Astragalus crassicarpus Nutt. var. *trichocalyx* (Nutt.) Barneby	cream ground-plum	X			
Aureolaria grandiflora (Benth.) Pennell	yellow false foxglove	X			
Aureolaria grandiflora (Benth.) Pennell var. *serrata* (Torrey ex Benth) Pennell	big flowered gerardia			X	
Baptisia australis (L.) R. Br	blue wild indigo	X	X	X	
Baptisia bracteata Muhl. Ex Elliott var. *leucophaea* (Nutt.) Kartesz & Gandhi	cream wild indigo	X			
Baptisia bracteata Muhl. Ex Elliott var. *glabrescens* (Larisey) Isely	wild indigo			X	
Baptisia leucantha T. & G.			X		
Belamcanda chnensisis (L.) DC.	blackberry lily				X
Belphilia ciliata (L.) Benth.	downy wood mint		X		
Berchemia scandens (Hill) K. Koch	rattan-vine	X	X		
Bouteloua curtipendula (Michx.) Torr. var. *curtipendula*	side-oats grama grass	X	X	X	
Bromus commutatus Schrad.	hairy chess	X			
Bromus mollis L.	soft chess				X
Buchnera americana L.	blue hearts			X	
Bulbostylis capillaris (L.) C.B. Clarke	sedge				X
Bumelia lanuginosa Michx.	gum bumelia	X		X	
Cacalia plantaginea (Raf.) Shinn.	indian plantain			X	
Callirhoe digitata Nutt.	winecup	X	X		
Camassia scilloides (Raf.) Cory	wild hyacinth	X	X		
Carex annectens var. *xanthocarpa* (Bickn.) Wieg.	sedge				X

Appendix A (cont.). Synthesis of six studies of floral species reported from Ozark glades.

Species	**Common Name**	**Limestone [1, 6]**	**Dolomite [1]**	**Calcareous [3, 5]**	**Sandstone [2, 4]**
Carex Bushii Machenz.	sedge				X
Carex brevior (Dew.) Mackenz.	sedge				X
Carex complanata var. *hirsuta* (Bailey) Gl.	sedge				X
Carex crawei Dewey	Crawe's sedge	X	X	X	
Carex eburnea Hook.	sedge	X	X		
Carex Frankii Kunth	sedge				X
Carex hirsutella Mack.	hairy sedge	X			
Carex Leavenworthii Dew.	sedge				X
Carex meadii Dewey	Mead's sedge		X	X	
Carex molesta Mackenz.	sedge				X
Carex Muhlenbergii Schk.	sedge	X			X
Carya sp.	hickory	X			
Carya ovata (Mill.) K. Koch				X	X
Carya texana Buckl.	black hickory			X	X
Carya tomentosa Nutt.	mockernut hickory				X
Cassia fasciculata Michx.	partridge pea			X	X
Castilleja coccinea (L.) Spreng.	Indian-paintbrush		X	X	
Ceanothus americanus L.	wild snowball	X			
Celtis occidentalis L.	Northern hackberry			X	
Celtis tenuifolia (Nutt.)	dwarf hackberry			X	
Cercis canadensis L. var. *canadensis*	eastern redbud	X	X	X	
Chamaecrista fasciculata (Michx.) Greene var. *fasciculata*	showy partridge-pea	X	X		
Chamaecrista nictitans (L.) Moench var. *nictitans*	sensitive partridge pea			X	
Cheilanthes lanosa (Michx.) D.C. Eaton	hairy lip-fern				X
Chionanthus virginica L.		X			
Chrysopsis pilosus Nutt.	golden aster			X	X
Cirsium altissimum (L.) Spreng.	tall thistle				X
Clinopodium arkansanum (Nutt.) House	Arkansas calamint	X	X	X	
Clematis versicolor Small	pale leatherflower	X			
Comandra umbellata (L.) Nutt. subsp. *umbellata*	bastard-toadflax		X	X	
Conyza canadensis (L.) Cron. var. *canadensis*	horse weed				X
Coreopsis grandiflora var. saxicola Hogg ex Sweet	tickseed				X
Coreopsis lanceolata L.	lance-leaf tickseed	X	X		X

Appendix A (cont.). Synthesis of six studies of floral species reported from Ozark glades.

Species	Common Name	Limestone $^{1, 6}$	Dolomite ^{1}	Calcareous $^{3, 5}$	Sandstone $^{2, 4}$
Coreopsis palmata Nutt.	tickseed, finger coreopsis	X		X	X
Cornus asperifolia Michx.	tough-leaf dogwood	X			
Cornus florida L.	flowering dogwood			X	
Corylus americana Walter	hazelnut			X	
Cotinus obovatus Raf.	American smoketree	X			
Crotalaria sagittalis L. var. *sagittalis*	rattlebox				X
Croton capitatus Michx.	woolly croton		X		X
Croton monanthogynous Michx.	prairie-tea	X	X	X	X
Crotonopsis elliptica Willd.	rushfoil				X
Cunila origanoides (L.) Britton	dittany	X			
Cuphea viscosissima Jacq.	clammy cuphea			X	
Cyperus aristatus Rottb.	umbrella sedge				X
Dalea candida Michx. ex. Willd. var. *candida*	white prairie clover			X	
Dalea purpurea Vent. var. *purpurea*	purple prairie-clover	X	X	X	
Danthonia spicata (L.) P. Beauv. ex Roem. & Schult.	poverty oat grass	X	X		X
Daucus carota L.	wild carrot			X	
Delphinium carolinianum Walt.	Carolina larkspur				X
Delphinium treleasei Bush ex K.C.Davis	Trelease's larkspur	X	X		
Desmanthus illinoensis (Michx.) Macmillan ex Robinson & Fern.	prairie mimosa			X	
Desmodium ciliare (Muhl.) DC. var.	tick-trefoil				X
Desmodium perplexum B.G.Schub.	tick-trefoil	X			
Desmodium marilandicum (L.) DC.	tick-trefoil			X	X
Desmodium rotundifolium DC.	tick-trefoil				X
Dianthus armeria L.	deptford pink			X	X
Dicanthelium acuminatum (Sw.) Gould & C.A. Clark subsp. *fasciculatum* (Torr.) Freckmann & Lelong	hairy rosette grass	X	X	X	
Dicanthelium boscii (Pioret) Gould & C. A. Clark	panic grass			X	
Diodia teres Walt.	rough buttonweed				X
Diospyros virginiana L.	persimmon	X	X	X	
Dodecantheon meadia L. var. meadia f. *meadia*	shooting star			X	
Draba brachycarpa Nutt. ex. Torrey & A. Gray	whitlow grass			X	

Appendix A (cont.). Synthesis of six studies of floral species reported from Ozark glades.

Species	**Common Name**	**Limestone $^{1, 6}$**	**Dolomite ^{1}**	**Calcareous $^{3, 5}$**	**Sandstone $^{2, 4}$**
Echinacea pallida (Nutt.)	pale purple coneflower	X	X	X	
Echinacea paradoxa (Norton) var. *paradoxa Britton*	yellow coneflower			X	
Echinacea purpurea L.	purple coneflower				X
Eleocharis compressa Sull. var. *compressa*	Flat-stem spike-rush		X		
Eleocharis tenuis (Willd.) Schultes var. *verrucosa* Svenson	spike rush				X
Eleocharis verrucose (Svenson) L. Harms	spike rush			X	
Elymus virginicus L.	wild rye			X	X
Eragrostis spectabilis (Pursh) Steud.	purple lovegrass	X			
Erigeron annuus (L.) Pers.	annual fleabane			X	X
Erigeron strigosus Muhl. Ex Willd.	daisy fleabane		X	X	X
Eryngium yuccifolium Michx.	rattlesnake-master	X			
Erythronium sp.	trout-lily		X		
Eupatorium altissimum L.	tall thoroughwort		X		X
Euphorbia corollata L.	flowering spurge	X	X	X	X
Euphorbia dentata Michx.	toothed spurge	X	X		
Euphorbia missurica Raf.	Missouri spurge		X		
Festuca octoflora Walter		X			
Fimbristylis puberula (Michx.) Vahl var. *puberula*	fimbry		X	X	X
Fragaria virginiana Duchesne	wild strawberry			X	
Frangula caroliniana (Walter) A. Gray	Carolina buckthorn		X	X	
Fraxinus americana L. var. americana	white ash			X	X
Fraxinus pennsylvanica Marsh.					X
Fraxinus quadrangulata Michx.	blue ash		X		
Galacita volubilis (L.) Britton	downy milk-pea	X	X		
Galium arkansanum A.Gray var. *arkansanum*	Arkansas bedstraw	X			
Galium concinnum Torrey & A. Gray	shining bedstraw			X	
Galium circaezans Michx. var. *hypomalacum* Fern.	wild licorice			X	
Galium obtusum Bigel. var. *obtusum*	bedstraw				X
Galium virgatum Nutt. ex Tor. & A. Gray	southwestern bedstraw	X	X		
Geranium sp.	geranium	X	X		
Geranium carolinianum L.	geranium				X
Geum canadense Jacq.	white avens				X
Glandularia canadensis (L.) Nutt.	rose vervain	X			
Gleditsia triacanthos L.	Honey locust			X	

Appendix A (cont.). Synthesis of six studies of floral species reported from Ozark glades.

Species	**Common Name**	**Limestone** ^{1,6}	**Dolomite** ^{1}	**Calcareous** ^{3,5}	**Sandstone** ^{2,4}
Gnaphalium obtusifolium L.	sweet everlasting				X
Grindelia lanceolate Nutt.	gum-plant	X	X		
Hedeoma hispida Pursh	pennyroyal				X
Hedyotis nigricans (Lam.) Fosberg				X	X
Helianthus hirsutus Raf.	hairy woodland sunflower, bristly sunflower	X	X	X	X
Helianthus maximiliani Schrad.	Maximilian's sunflower	X	X		
Heliotropium tenellum (Nutt.) Torr.	heliotrope	X	X	X	
Hemerocallis fulva L.	orange day lily				X
Hieracium Gronovii L.	hawkweed				X
Houstonia longifolia Gaertn.	long-leaf bluet	X			
Hustonia nigricans (Lam.) Fernald var. *nigricans*	diamond-flower	X	X		
Hypericum drummondii (Grev. and Hook.) T. & G.					X
Hypericum gentianoides (L.) B.S.P.					X
Hypericum prolificum L.					X
Hypericum pseudomaculatum Bush ex Britton	false spotted St. John's-wort		X		
Hypericum punctatum Lam.	spotted St. John's wort			X	X
Hypericum sphaerocarpum Michx.	round-fruit St. John's-wort		X	X	
Hypoxis hirsuta (L.) Coville	yellow star-grass		X	X	
Ilex decidua Walter	deciduous holly	X	X		
Juncus Dudleyi Wieg.	rush				X
Juncus interior Wieg.	rush				X
Juncus marginatus Rostk.	rush				X
Juniperus virginiana L.	Eastern red-cedar	X	X	X	X
Krigia virginica (L.) Willd.	dward dandelion				X
Lactuca canadensis L.	wild lettuce				X
Lechea tenuifolia Michx. var. *tenuifolia*	pinweed				X
Lepidium virginicum L. var. *virginicum*	pepper grass				X
Lespedeza capitata Michx.	bush-clover				X
Lespedeza cuneata (Dumont) G. Don	sericea, Lespedeza				X
Lespedeza intermedia (S. Watson) Britton	bush clover			X	

Appendix A (cont.). Synthesis of six studies of floral species reported from Ozark glades.

Species	**Common Name**	**Limestone** [1, 6]	**Dolomite** [1]	**Calcareous** [3, 5]	**Sandstone** [2, 4]
Lespedeza procumbens Michx.				X	
Lespedeza stipulacea Maxim.	Korean clover				X
Lespedeza striata (Thunb.) H. & A.				X	
Lespedeza violacea (L.) Pers.	bush-clover	X			X
Lespedeza virginica (L.) Britton	slender bush-clover	X		X	
Liatris aspera Michx.	rough blazing-star		X	X	
Liatris hirsuta Rydb.	hairy blazing-star	X	X		
Liatris punctata Hook. var. *mucronate* (DC.) B.L.Turner	dotted gayfeather	X	X		
Liatris squarrosa (L.) Michx.	scaly blazing star			X	X
Linum sulcatum Riddell	grooved flax			X	
Ligusticum canadense (L.) Britton	lovage			X	
Liquidambar styraciflua L.					X
Lithospermum canescens (Michx.) Lehm.	hoary puccoon	X	X		
Lithospermum caroliniense (Walter) MacMillan	puccoon			X	
Lobelia spicata Lam.	pale-spike lobelia	X	X	X	X
Manfreda virginica (L.)	false aloe	X	X		
Melilotus officinalis (L.) Lam.	yellow sweet clover				X
Mimosa quadrivalvis L. var. *Nuttallii* (DC. ex Britton & Rose) Beard ex Barneby	sensitive-brier	X	X		
Minuartia patula (Michx.) Mattf.	Sandwort		X		
Mirabilis albida (Walter) Heimerl	white four-o'clock			X	
Monarda bradburiana Beck	beebalm			X	
Monarda fistulosa L.	wild bergamont			X	
Morus rubra L.	red mulberry			X	
Nothoscordum bivalve (L.)	crow-poison	X	X		X
Nuttallanthus canadensis (L.) Dummort.	blue toadflax				X
Oenothera fruticosa L.					X
Oenothera laciniata Hill	evening primrose				X
Oenothera macrocarpa Nutt. subsp. *macrocarpa*	Missouri-primrose		X	X	
Onosmodium bejariense A.DC. var. *subsetosum* (Mack. && Bush ex Small) B.L.Turner	marblleseed		X		
Ophioglossum engelmannii Prantl	limestone adder's-tongue fern		X	X	

Appendix A (cont.). Synthesis of six studies of floral species reported from Ozark glades.

Species	**Common Name**	**Limestone** [1, 6]	**Dolomite** [1]	**Calcareous** [3, 5]	**Sandstone** [2, 4]
Opuntia compressa (Salisb.) Macbr.	prickly pear				X
Opuntia humifusa (Raf.) Raf. Var. *humifusa*	Eastern prickly-pear	X			
Opuntia macrorhiza Engelm.					X
Ostrya virginiana (Mill.) K. Koch	hop-hornbeam	X	X	X	
Oxalis stricta L.					X
Oxalis florida Salisb.	yellow wood-sorrel			X	X
Oxalis violacea L.	violet wood-sorrel	X	X	X	
Palafoxia callosa (Nutt.) Torr. & A.Gray	small palafoxia	X	X		X
Panicum capillare L.	witch grass	X			
Panicum flexile (Gatt.) Scribn.	wiry witch grass	X	X		X
Panicum oligosanthes var. *Scribnerianum* (Nash) Fern.	panic grass				X
Panicum scoparium Lam.	panic grass				X
Panicum sphaerocarpon Ell.	panic grass				X
Panicum virgatum L.	switch grass	X	X	X	X
Parthenium hispidium Raf.	American feverfew			X	
Parthenium integrifolium L.	wild quinine	X			
Parthenocissus quinquefolia (L.) Planch.	Virginia-creeper	X	X	X	
Passiflora lutea L.	yellow passion-flower	X	X		
Pediomelum esculentum (Pursh) Rydb.	prairie turnip			X	
Pellaea atropurpurea (L.) Link	purple-stem cliff-brake		X	X	
Penstemon arkansanus Pennell	Arkansas beard-tongue	X			
Penstemon pallidus Small	pale beard-tongue			X	
Penstemon tubaeflorus Nutt.	beard-tongue				X
Phlox pilosa L. subsp. *ozarkana* (Wherry)	Ozark downy phlox		X		
Phyllanthus polygonoides Nutt.	knotweed leaf-flower		X		
Physalis pubescens L. var. *pubescens* (Dunal) Waterf.	ground cherry			X	
Physalis virginiana Mill.	ground cherry				X
Physocarpus opulifolius (L.) Maxim. var. *intermedius* (Rydb.) Robinson	ninebark			X	
Physostegia virginiana (L.) Benth. subsp. *praemorsa* (Shinners) P.D.Cantino	obedient-plant, false dragonhead	X		X	
Pinus echinata Mill.	shortleaf pine				X
Plantago aristata Michx.	bracted plantain				X

Appendix A (cont.). Synthesis of six studies of floral species reported from Ozark glades.

Species	**Common Name**	**Limestone** $^{1, 6}$	**Dolomite** ^{1}	**Calcareous** $^{3, 5}$	**Sandstone** $^{2, 4}$
Plantago pusilla Nutt.					X
Polygala sanguinea L.					X
Polygonella americana (Fisch. & Mey.) Small					X
Polygonum tenue Michx.					X
Potentilla recta L.	rough-fruited cinquefoil				X
Potentilla simplex Michx. var. *simplex*	common cinquefoil			X	
Primula meadia (L.) Mast & Reveal	shooting star		X		
Prunus serotina Ehrh.	black cherry			X	X
Psoralea psoralioides (Walt.)	snakeroot				X
Psoralidum tenuiflorum (Pursh) Rydb.	scruffy pea			X	
Ptelea trifoliata L. var. *trifoliata*	common hop tree			X	
Pycnanthemum pilosum Nutt.	hairy mountain mint			X	
Pycnanthemum tenuifolium Schrad.	slender mountain mint			X	X
Quercus alba L.	white oak			X	X
Quercus imbricaria Michx.	shingle oak				X
Quercus macrocarpa Michx.	bur oak			X	
Quercus marilandica Muenchh.	black jack oak			X	X
Quercus muehlenbergii Englem.	chinquapin oak	X	X	X	
Quercus prinoides Willd.	dwarf chestnut oak			X	
Quercus rubra L.	northern red oak		X	X	X
Quercus shumardii Buckley var. *shumardii*	Shumard's oak			X	
Quercus stellata Wangenh.	post oak	X		X	X
Quercus velutina Lam.	black oak	X	X	X	X
Ratibida pinnata (Vent.) Barnhart	grayhead prairie coneflower			X	
Rhamnus caroliniana Walter	Carolina buckthorn	X		X	
Rhus aromatica Aiton	fragrant sumac	X	X	X	X
Rhus copallinum L.	winged sumac	X			X
Rhus glabra L. var. *glabra*	smooth sumac				X
Robinia pseudoacacia L.	black locust			X	
Rosa carolina L.	Carolina rose	X		X	X
Rosa setigera Michx.	climbing rose		X		X
Rubus flagellaris Willd.	dewberry				X
Rudbeckia hirta L.	black-eyed susan	X	X		X
Rudbeckia missouriensis Englem. Ex E.L.Boynton & Beadle	Missouri coneflower	X	X	X	

Appendix A (cont.). Synthesis of six studies of floral species reported from Ozark glades.

Species	**Common Name**	**Limestone** [1, 6]	**Dolomite** [1]	**Calcareous** [3, 5]	**Sandstone** [2, 4]
Ruellia humilis Nutt.	hairy wild petunia	X	X	X	X
Rumex Acetosella L.	sheep sorrel				X
Rumex crispus L.	sour dock				X
Rumex hastatulus Balw.					X
Rhynchospor harveyi W.Boott					X
Sabatia angularis (L.) Pursh	rose-gentian	X	X	X	
Sabatia cammpestris Nutt.					X
Sanicula sp.	black-snakeroot	X	X		
Sassafras albidum (Nutt.) Nees	sassafras	X	X	X	
Satureja arkansana (Nutt.) Brig.					X
Schizachyrium scoparium (Michx.) Nash var. *scoparium*	little bluestem grass	X	X	X	X
Schrankia nuttallii (DC. Ex Britton & Rose) Standley	sensitive brier			X	
Schrankia uncinata Willd.	sensitive brier			X	X
Scleria oligantha Michx.	nut-rush		X		
Scleria pauciflora Muhl. ex Willd. var. *pauciflora*	nut-rush	X			X
Scirpus pendulus Muhlenb. ex Elliott	bulrush				
Scutellaria parvula Michx. var. *parvula*	small skullcap	X	X	X	
Selenia aurea Nutt.	golden selenia				X
Sedum nuttallianum Raf.					X
Sedum pulchellum (Michx.)	widow’s cross	X			X
Selaginella rupestris (L.) Spring	spikemoss				X
Setaria geniculata (Lam.) Beauv.	prairie foxtail				X
Setaria glauca (L.) Beauv.	yellow foxtail				X
Sida spinosa L.	prickly sida			X	
Sideroxylon lanuginosum Michx.	gum bumelia	X	X		
Silphium laciniatum L.	compass-plant		X	X	
Silphium terebinthinaceum Jaq. var. *terebinthinaceum*	prairie-dock	X	X	X	
Sisyrinchium campestre E.P.Bicknell	blue-eyed grass	X	X	X	
Smilax bona-nox L.	saw grenbrier	X	X	X	
Smilax hispida Muhlenb. ex Torrey	bristly greenbriar			X	
Solanum carolinense L.	horse nettle				X
Solidago arguta Aiton var. *arguta*	goldenrod			X	
Solidago gattingeri Chapm. ex A.Gray	Gattinger’s goldenrod		X	X	
Solidago sp.	goldenrod	X	X		X

Appendix A (cont.). Synthesis of six studies of floral species reported from Ozark glades.

Species	**Common Name**	**Limestone** [1, 6]	**Dolomite** [1]	**Calcareous** [3, 5]	**Sandstone** [2, 4]
Solidago missouriensis var. *fasciculata* Holzinger	goldenrod				X
Solidago nemoralis Alt.	old-field goldenrod			X	X
Solidago radula Nutt.	rough goldenrod				X
Sorghastrum nutans (L.) Nash	indian grass	X	X	X	X
Specularia lamprosperma (McVaugh) Fern.					X
Specularia leptocarpa (Nutt.) Gray	Venus' looking glass				X
Specularia perfoliata (L.) A.D.C.	Venus' looking glass				X
Spiranthes cernua (L.) Rich.	nodding ladies'-tresses		X		
Sporobolus clandestinus (Biehler) A. Hitchc.	dropseed			X	
Sporobolus compositus (Poir.) Merr.	tall dropseed	X			
Sporobolus heterolepis A. Gray	prairie dropseed	X			
Sporobolus neglectus Nash	bald grass, small dropseed	X			
Stellaria media (L.) Cyrillo var. *media*	common chickweed				X
Stylosanthes biflora (L.) B.S.P	pencil-flower	X		X	X
Symphotrichum anomalum (Englem. Ex Torr. & A. Gray) G.L. Nesom	aster	X			
Symphotrichum oblongifolium (Nutt.) G.L. Nesom	aromatic aster	X	X		
Symphoricarpos orbiculatus Moench *f. orbiculatus*	coral berry			X	
Symphotrichum urophyllum (Lindl. Ex DC.) G.L. Nesom	White arrow-leaf aster		X		
Symphyotrichum patens (Aiton) G.L. Nesom var. *patentissimum* (Lindl. Ex DC.) G.L. Nesom	late purple aster	X			
Taenidia integerrima (L.) Drude	yellow pimpernel	X			
Talinum calycinum Engelm.					X
Talinum parviflorum Nutt.	rock pink				X
Tephrosia virginiana (L.)	goat's rule			X	X
Thlaspi arvense L.	field penny cress			X	
Toxicdendron radicans (L.) Kuntze	poison ivy			X	
Tradescantia ohiensis Raf.	Ohio spiderwort	X		X	X
Tragia betonicifolia Nutt.	noseburn			X	
Tragia ramosa Torr.	noseburn	X	X		
Trifolium campestre Schreb.	large hop-clover				X
Trichostema brachiatum L.	false pennyroyal				X

Appendix A (cont.). Synthesis of six studies of floral species reported from Ozark glades.

Species	**Common Name**	**Limestone** [1, 6]	**Dolomite** [1]	**Calcareous** [3, 5]	**Sandstone** [2, 4]
Trichostema dichotomum L.					X
Tridens flavus (L.) Hitchc. var. *flavus*	purple-top tridens	X	X	X	X
Tripsacum dactyloides L.	eastern gama grass		X		
Ulmus alata Michx.	winged elm	X	X		X
Ulmus americana L.	American elm			X	X
Ulmus rubra Muhlenb.	slippery elm			X	
Uniola latifolia Michx.	spike grass				X
Vaccinium arboreum Marsh.	sparkleberry			X	X
Vaccinium vacillans Torr.	lowbush blueberry				X
Valerianella ozarkana Dyal	Ozark cornsalad	X			
Valerianella radiata (L.) Dufr.	corn salad				X
Veronia arkansana DC.	Arkansas ironweed		X		
Veronia baldwini Torr.	ironweed				X
Veronia crinita Raf.					X
Viburnum rufidulum Raf.	southern black haw			X	
Viola Kitaibeliana R. & S. var. *Rafinesquii* (Greene) Fern.	field pansy				X
Viola pedata L.	bird's-foot violet	X	X	X	
Vitis aestivalis Michx. var. *lincecumii* (Buckley) Munson	summer grape	X		X	
Vitis cinerea (Engelm.) Engelm. ex Millardet	grayback grape			X	
Woodsia obtusa (Spreng.) Torr.	blunt-lobed fern			X	X
Xanthium strumarium L. var. *canadense* (Miller) Torrey & Gray	common cocklebur			X	
Zizia aurea (L.) W.D.J.Koch	golden alexanders	X		X	

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4. Jeffries, D. L. (1983). *The vegetation and soils of sandstone glades of Northern Arkansas.* Master's thesis, University of Arkansas.
5. Keeland, B. D. (1978). *Vegetation and soils in calcareous glades of Northwest Arkansas.* Master's thesis, University of Arkansas.
6. Kucera, C. L., & Martin S. C. (1957). Vegetation and soil relationships in the glade region of the Southwestern Missouri Ozarks. *Ecology*, 38(2), 285-291.

Appendix B. Site Descriptions

Degraded Sites

Site management histories were provided by Idun Guenther, Wildlife Biologist, Ozark-St. Francis National Forest, Sylamore Ranger District, personal communication, January, 2019. Figures for each site perimeter and area, as well as primary cardinal direction of slope and presence of *Crotophytus collaris* can be found in Table 1.

Site 2

This was one of the more interesting sites chosen and had a very primitive feel and is shown in Figure B1. Many of the older cedars were draped in lichens and gnarled from several decades of life on sandstone bedrock, which had weathered to a dark, nearly black, coloration. Unlike other sites, here the encroaching cedars were primarily *Juniperus ashei* or hybrids of *J. ashei* and *J. virginiana*. Many of the rock outcrop margins, where soils were deep enough to hold vegetation, were encroached upon by cedars to the point of total canopy closure. Ground cover in these low-light areas consisted of mostly moss interspersed with sparse sedges and forbs. There were a few open expanses of glade not yet encroached upon by cedars containing mixed grasses and forbs, including *Schizachyrium scoparium*, *Croton willdenowii*, *C. capitatus* and *Coreopsis* sp. The open bedrock core of this site was the home to one of the study's more frequently sighted populations of *Crotophytus collaris*, which persists here despite forest encroachment. There was also a fairly extensive network of sandstone outcrops, pine (*Pinus echinata*) woodlands and unmaintained logging roads in the immediate area, which facilitated open-canopy connectivity between Site 2 and an unknown acreage of other woodland and glade habitat. The slope of Site 2 was negligible over much of the western expanses of the glade and sloped north over the majority of the remainder, with a typical slope being roughly 5-10 cm rise for every meter of run.

Site 2 had not received any recorded habitat management at the time of this study. However, there is a history of logging in the vicinity of this site.

Site 4

Site 4 can be seen in Figure B2. The sandstone bedrock of this site underlaid a limestone/dolomite layer, which created a small peak (~100 m higher in elevation) just to the south of Site 4. This topography gave Site 4 a primarily north-facing slope roughly equivalent to that described for Site 2. This site was positioned along the north edge of a firebreak/road, which was the dividing line for prescribed burns that occurred over much of the aforementioned peak and adjoining ridge to the south. Site 4 was also less than 150 m from Restored Site 5. Prescribed burns had reached some understory vegetation at Site 4, but failed to immediately kill older cedars (*J. virginiana*), which were the predominant encroaching species. However, in the two seasons of sampling that occurred after burning, many of the cedars died from damage to trunks during the fire, but remained standing. Very little bedrock was exposed at this site, but remnant grasses such as *S. scoparium* and *Sporobolus* sp. indicative of a more open canopy still persisted in pockets where sunlight reached the ground, as did the forbs *Clinopodium arkansanum* and *Tragia ramosa*. In areas of Site 4 where little light reached the ground, mosses prevailed. Notable woody species present included *Quercus* sp., *P. echinata,* and *Diospyros virginiana.* Management of Site 4 included a low-intensity prescribed burn of the understory in 1998 and incidental burning of the understory from prescribed fire crossing the firebreak in 2018. Again, logging has occurred near this site.

Site 9

Site 9 is shown in Figure B3 and was within a bastion of older *Juniperus virginiana* (most over 10 m tall) and *P. echinata*. In essence, this site was more woodland than glade, but was bordered on three sides by a complex of glades, savannas and woodlands that included a range of successional stages and management histories. It was located approximately 200 m north of a well-travelled gravel road and bordered to the west by private land managed for cattle. The tall and straight form of the cedar trunks indicated that perhaps soil depths at this site were greater than other sites, however, no soil depth measurements were taken. The community of grasses found at this site was not typical of sites involved in this study. No *S. scoparium* was noted in areas of trap deployment, which, instead, were dominated by *Chasmanthium latifolium* and *Dichanthelium* sp. Common forbs present included *Croton willdenowii* and *Pluchea camphorata*. Woody species present other than cedars and pines included *Carya* sp., *Baccharis halimifolia* and the vine *Smilax bona-nox*. This site received herbicide treatment of invasive plants in 2014, but no other management history was known. It is probable that *Crotophylus collaris* may exist within the adjoining open habitat complex, either the private land, or restored areas, but none were observed within the site and likely would not inhabit the areas of trap grid deployment due to dense canopy cover. This site again faced north with a typical slope being roughly 5-10 cm rise for every meter of run.

Intact Sites

Site 3

Site 3 is shown in figure B4 and was within a blufftop sandstone glade set atop a north-facing bluff. This was by far the largest glade in the study (Table 1). Portions of this glade sloped in all cardinal directions. However, slope was non-existent or very slightly west-facing (~2 cm per meter) where present at the location of Site 3 trap grids. This glade is crossed by a well-travelled gravel road and roadside waypoint which exposes the site to human disturbance more than other sites included in this study. This glade received a prescribed burn and herbicide treatment of invasive herbaceous species in 2014 with no other management history provided. Large expanses of lichen-covered bedrock occurred over much of this glade, which was the home of seemingly healthy populations of *C. collaris*. Notable species of groundcover included the grasses *S. scoparium* and *Sarghastrom nutans*, as well as the forbs *Coreopsis* sp., *Croton willdenowii* and *Clinopodium arkansanum.* The few woody invaders noted included *J. virginiana, P. echinata* and the vine *S. bona-nox*.

Site 8

Site 8 was the least accessible of glade sites chosen with no known roads reaching it. It was among the flattest of sites chosen, with an estimated typical rise of 5 cm per meter of rise creating a southern face to the slope. This site contained large expanses of exposed sandstone that had weathered to a dark color similar to that described for Site 2. Likewise, older cedars within the glade had often become covered by bearded lichens and gnarled from growing in small pockets within the bedrock layer. Several *C. collaris* were observed at this site, and flora noted here were *S. scoparium, Croton monanthogynous, Coreopsis* sp., *Opuntia* sp. *S. bona-nox, P. echinata* and *J. virginica*. This glade was the recipient of low-intensity understory burns in 1996, 2001 and 2004, and mechanical removal of invasives was documented in 2014. However, no evidence of cedar removal or other similar activity was detected in the vicinity of Site 8 trap grids.

Site 10

This site, shown in Figure B6, was located on a south-facing slope approximately 20 m elevation below an east-west ridge. At the core of this glade were several sandstone rock shelves that protruded through the most severe slopes of any sites involved in this study. These were often somewhere between 50-75 cm rise for every meter of run. Between this glade and the ridgeline showed evidence of clearing and burning at some point in its history. However, invasive species herbicide treatment in 2015 and 2016 is all that was recorded as management history for Site 10. There is an abrupt soil layer depth change that occurs along the margins of this glade, creating a home for brushier savanna-like vegetation, chest-high in places, and included species such as *Sorghastrum nutans, Callicarpa americana, Quercus* sp., *Passiflora incarnata, L. cunea,* and *Verbascum thapsus.* The bedrock outcroppings were home to populations of *C. collaris,* and notable groundcover within the glade included the grasses *S. scoparium* and *Aristida* sp. along with the forbs *Coreopsis* sp., *Croton monanthogynous*, and *Hexasepalum teres*.

Restored Sites

Site 5

This site was located to the south of the firebreak road mentioned in the Site 4 description above and can be seen in Figure B7. This site was selected not only because it had received woody species removal and a prescribed burn in February of 2018, but also because satellite images showed this site to have nearly been completely converted to forest prior to burning. A single sandstone bedrock opening roughly 15 m in diameter existed at the core of this site. Other areas were littered with sandstone chunk-rock mixed with limestone/dolomite chunk-rock, 50 cm-2 m in size, that had eroded downslope from the rise to the south. The slope of this site was to the north and was steeper than that of Site 4 (~ 20 cm rise per meter of run). Some cedars and other woody vegetation that had been cut and left in place near the bedrock opening prior to burning had not successfully incinerated. During the duration of this study, Site 5 succeeded from bare soils in most areas post-burn, to grasses and woody shrubs by completion of the final trapping season. Soil depth here appeared deeper than other sites, allowing for woody vegetation to regain a foothold in many areas prior to termination of the study. Grasses found here included *S. scoparium, Sporobolus* sp. and *Dicanthelium* sp. Notable forbs included *Croton capitatus, C. monanthogynous* and *Clinopodium arkansanum*. Woody invaders included *Cotinus obovatus*, *Quercus* sp., *Rhus copalinum*, *Diospyros virginiana*, and *Pinus echinata*.

Site 6

Site 6, shown in Figure B8, had undergone extensive cedar removal prior to this study via cutting, chipping/mulching and burning, that began in 2014 and was completed in 2016. This site, like Sites 4 and 5, sits along a sandstone layer that is topped by a limestone knoll. Site 6 was near a large sandstone outcrop. However, other areas of the glade existed with very little exposed rock. Slope was to the west and commonly around 10 cm of rise for each meter of run. This site was one of two involved in this study to have received a reintroduction of *C. collaris* in 2016. However, none were observed during this study, and reintroduced populations are believed to have become extirpated (Dr. Casey Brewster, personal communication, June 2021). Along margins of the bedrock outcrop, sedges (*Carex* sp.) predominated and, in slightly deeper soils, *S. scoparium* and *Sporobolus* sp. were dominant grasses. Notable forbs at this site included *Coreopsis* sp., *Rudbeckia* sp., *Croton monathogynous, C. capitatus,* and *Clinopodium arkansanum*. This site was also home to *Opuntia* cacti as well as the woody invaders *Baccharis halimifolia* and *Platanus occidentalis,* the latter not being present at other sites in this study. Site 6 had also received the same low-intensity understory burns as Site 8 in 1996, 2001 and 2004, and logging occurred along the southern edge of this glade during 2019.

Site 7

Site 7 is shown in Figure B9. Much as at Site 6, cedar removal began in 2014 and was completed, along with a prescribed burn and reintroduction of *C. collaris* in 2016. It also received a low intensity understory burn in 1997. An ephemeral drainage bordered the western edge of this site that held water throughout the wet season. A frequented primitive campsite exists within this glade and it is traversed by several logging roads. The *C. collaris* reintroduction at this site in 2016 was successful (Brewster, personal communication) and one adult female was sighted on the final visit of the 2020 trapping season. The slope of this site was 5-10 cm rise per meter of run over much of the upper elevations, but leveled off near the center of the glade to a negligible slope. Sandstone bedrock appeared in several outcrops, each approximately 5 m wide and located along a single contour of the more sloping region of the glade. Grasses were dominated by *S. scoparium* and *Dichanthelium* sp., with notable forbs including *Coreopsis* sp., *Croton willdenowii*, and *Hexasepalum teres*. Woody invaders included *J. virginiana*, *Quercus* sp. and the vine *S. bona-nox*.

Figure B1. Composite of Site 2 photographs. Photo credit: Austin Jones

Figure B2. Composite of Site 4 photographs. Photo credit: Austin Jones

Figure B3. Composite of Site 9 photographs. Photo credit: Austin Jones

Figure B4. Composite of Site 3 photographs. Photo credit: Austin Jones

Figure B5. Composite of Site 8 photographs. Photo credit: Austin Jones

Figure B6. Composite of Site 10 photographs. Photo credit: Austin Jones

Figure B7. Composite of Site 5 photographs. Photo credit: Austin Jones

Figure B8. Composite of Site 6 photographs. Photo credit: Austin Jones

Figure B9. Composite of Site 7 photographs. Photo credit: Austin Jones

Appendix C. Randomization protocol used for trap grid placement.

For each of the nine sites sampled, approximations of glade boundaries were traced on Google Earth (Google, n.d.) satellite images based on apparent forest/glade interface, as well as information from field notes (Figure C). In PowerPoint (Microsoft, Redmond, Washington, U.S.A.), six rectangular grids, composed of six square units of equal area, were placed within the traced boundaries (Figure C). One of the six grids was then chosen at random by rolling a standard six-sided die; then one of the squares within the selected grid was chosen at random with a second roll of the die. The randomly chosen square on the satellite image was translated to the physical location via GPS coordinates. Onsite, once the location was reached, randomization of the placement of the initial trap in a grid was furthered by a series of dice rolls performed while facing in the direction of travel upon arrival to the GPS coordinates. The series of dice rolls was as follows:

Roll 1: Even number, turn 180 degrees left; odd, turn 180 degrees right.
Roll 2: Take as many paces as the number rolled in the direction determined in Roll 1.
Roll 3: Even number, turn 180 degrees left; odd, turn 180 degrees right.
Roll 4: Take as many paces as the number rolled in the direction determined in Roll 3.
Roll 5: Even number, turn 180 degrees left; odd, turn 180 degrees right.
Roll 6: Even number, first trap is LST; odd, CST

From the initial trap, the remaining three traps of the middle row were placed in a straight line, 13 m apart, and the remaining two rows of the grid were spaced 13 m to either side of the middle row. When an obstacle, such as a tree or ledge, prevented trap placement in the required location, a dice roll again determined whether the trap would be placed to the left or right of the object in question.

Figure C. Site 6 satellite image (Google, n.d.) including traced sandstone glade boundary approximation, and numbered grids used to randomly select a location for trap grid deployment. In this case, the first dice roll selected grid 5, and the second dice roll selected square 3 of grid 5. It is worthy of note that at this site, the site boundary was influenced by the presence of overlying calcareous substrate as well as apparent forest-glade interface.

Appendix D. Inter-rater reliability of photographic groundcover estimation method.

Percent agreement between photo graders was calculated for all groundcover class assignments made via photographic groundcover estimation (Table D1). The lowest percentage of total agreement between photo graders occurred within the Grasslike and Forbs categories (Table D1). However, these were also the most frequently occurring categories and therefore subject to more estimation attempts than categories where absence (cover class 0) was recorded more frequently. For example, the groundcover category Cedar had the highest percent agreement between photo graders, but of the 203 photos agreed upon, 202 were scored by both graders as 0; only one of the 14 photos where cedar was present (according to at least one grader) was agreed upon by both graders. This example shows how the message conveyed by percent agreement can be skewed from accurately depicting congruence between graders for categories that are present in a photo. The example also shows that raters are likely to agree upon cedar absence.

Another representation of percent agreement is worthy of discussion in relation to this study. Since the groundcover classes being graded in this case (other than zero) are contiguous ranges of percentages, the difference between class assignments can come down to as little as 1% difference in estimation between graders. Therefore, figures have also been generated for the percent of groundcover class assignments that were only off by one groundcover class between graders (Table D1). These figures, when combined with true percent agreement, may be a better representation of congruence between graders if many of the estimations made for a category were “borderline” between two class ranges.

In an effort to more accurately interpret the inter-rater reliability between photo graders, Cohen’s Kappa figures were calculated for each groundcover category and are presented in

Table D2. Only data from photos where at least one grader indicated the presence of a groundcover category were considered for generation of these figures. According to McHugh (2012) Cohen's Kappa figures between 0.61-0.80 are considered to have substantial agreement. The categories of Grasslike and Dead Wood fall into the lower end of this range (Table D2). The categories of Rock, Leaf Litter and Bare Soil fall within the range for moderate agreement (0.41-0.60) presented by McHugh (2012). The Moss and Woody categories had fair agreement (0.21-0.40) and Forbs and Cedar categories fell below this range and were the least agreed upon of the study (Table D2).

Table D1. Percent agreement between photographic groundcover estimation of two individuals using Daubenmire cover classes

	Grasslike	Forbs	Cedar	Woody	Moss	Rock	Leaf Litter	Bare Soil	Dead Wood
% totally agree	41	38	94	81	62	63	75	63	79
% off by one category	48	50	4	12	28	30	21	27	21
Total	89	88	98	93	90	93	96	90	100

Table D2. Cohen's Kappa figures for photographic groundcover estimation using Daubenmire cover classes.

Grasslike	Forbs	Cedar	Woody	Moss	Rock	Leaf Litter	Bare Soil	Dead Wood
0.63	0.17	0.18	0.38	0.40	0.49	0.47	0.41	0.63

McHugh, M. L. (2012). Interrater reliability: the kappa statistic. *Biochemia Medica*, 22(3), 276-282.

Appendix E. Breakdown of trap catches by trap style.

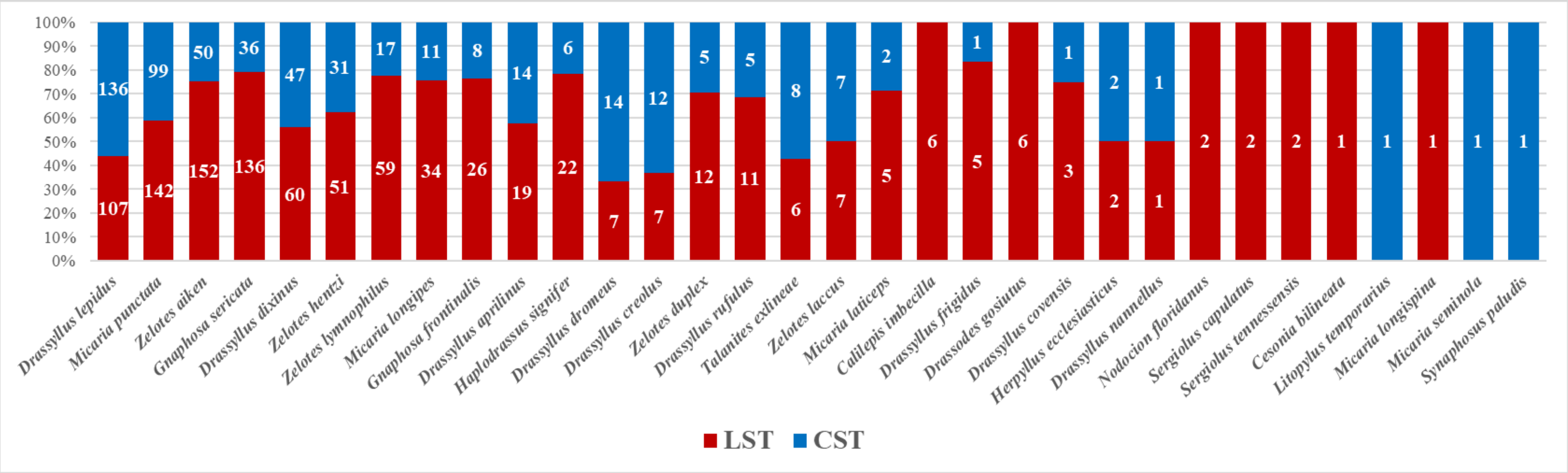

Figure D1. Gnaphosid spider catches by trap type for 2018-2019 and 2020 trapping seasons combined. LST = Lunchbox style traps, CST = Cone style traps.

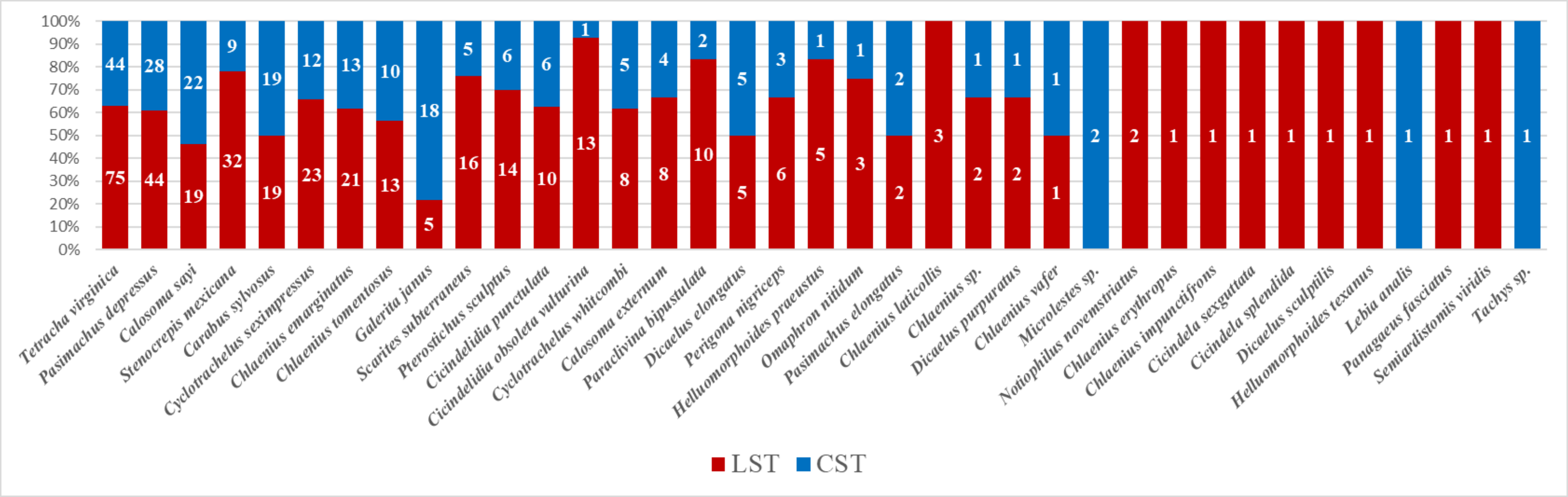

Figure D2. Predatory carabid catches by trap type for 2018-2019 and 2020 trapping seasons combined. LST = Lunchbox style traps, CST = Cone style traps.