Ground-dwelling Invertebrate Community Composition Changes between Coastal Sage Scrub Community of San Diego and Tijuana with Urbanization

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ABSTRACT

Invertebrates are some of the most diverse and abundant groups of terrestrial organisms. They provide important functions within an ecosystem and can rapidly respond to environmental changes, such as urbanization, making them excellent ecological indicators for ecosystem function and health. Urbanization, and the disturbance and introduction of exotic plants often associated with it, affects invertebrate taxa, changing community composition, structure, diversity, and abundance of invertebrates. Some invertebrate taxa can exploit these changes and increase in abundance; while others exhibit sensitivity and are at risk of extirpation. The San Diego and Tijuana region has been increasing in urbanizing pressure for decades. Tijuana, in particular, has experienced exponential population growth over the past 30 years, and urbanization threatens the integrity of much of the remaining coastal sage scrub, an endangered and rare ecosystem unique to the coasts of southern California and northwestern Mexico composed of many coastal deciduous shrubs and annual wildflowers. However, the invertebrates currently residing in these semi-disturbed landscapes have rarely been sampled. In this study, ground-dwelling invertebrates from a native plant nursery in an urbanizing area of Tijuana at Vivero Hormiguitas and a protected site in Border Field State Park in San Diego were sampled to compare the changes in coastal sage scrub ground-dwelling invertebrate community composition in response to urbanization. Beetle abundance and richness decreased in the urbanizing site in Tijuana, while spider abundance and richness increased in Tijuana compared to the protected site in San Diego. Particularly, we noted decreases in Tenebrionidae abundance and failed to detect Silphidae, suggesting slower decomposition rates in the urbanizing Tijuana site. Although spider abundance in general increased, mostly due to increased prevalence of one genus, *Xysticus*, Gnaphosidae abundance decreased in the Tijuana site. Lastly, non-native invertebrate taxa, especially isopods, showed dramatic increases in abundance at the urbanizing Tijuana site, contributing to overall increases in invertebrate abundance in Tijuana compared to the protected site in San Diego. These results indicate that urbanization has negatively affected ecosystem functions in coastal sage scrub communities and that it has drastically impacted the diversity and health of this rare ecosystem, much of which is already fragmented, and actions need to be taken to protect this rare habitat as much of its range continues to become more urbanized.

KEYWORDS

Beetle; Coastal Sage Scrub; Community Composition; Disturbance; Invasive Species; Invertebrates; San Diego; Tijuana; Spider; Urbanization

INTRODUCTION

Invertebrates are one of the most diverse and abundant groups of organisms in terrestrial ecosystems.[1] They are ubiquitous to almost every terrestrial habitat and contribute to many important ecological processes. Invertebrates are often the first facilitators of decomposition and can influence the rate at which nutrients are recycled in food webs.[2] Invertebrates also play important roles in the reproduction of other species through mechanisms such as pollination and seed dispersal.[3] Invertebrates often provide important food sources for higher trophic levels and can serve as biological control, increasing an ecosystem's resilience to change.[4] The health of an ecosystem and food web integrity can be reflected by the diversity and community structure of its invertebrate taxa.[5] However, these ecological processes, including those involving invertebrates, often fall out of balance when a habitat is altered due to human activities, resulting in changes in invertebrate community composition.[2, 5]

Using ground-dwelling invertebrates as bioindicators for ecosystem health is a well-established method within ecological studies for ecological impact assessment and restoration ecology.[6, 7] Recently, bioindicators have been categorized into three distinct categories: (i) environmental indicators, which indicate the abiotic and biotic states of an ecosystem, (ii) ecological indicators, which reveal changes in an ecosystem and evidence for impacts from changes caused by natural or human processes, and (iii) biodiversity indicators, which correspond to the diversity of taxa and community of an area.[8, 9] Changes in invertebrate assemblage and composition can reflect environmental change, especially when using ground-dwelling invertebrate taxa such as spiders (Araneae) (ex: Gnaphosidae), ground beetles (Carabidae), rove beetles (Staphylinidae), and ants (Formicidae) as bioindicators.[10–14]

Ground-dwelling invertebrates are good ecological and biodiversity indicators as they are particularly sensitive to changes in microhabitat and ecosystem structure and are more abundant and diverse than many vertebrate species.[10, 15, 16] Short generation times and high mobility of ground-dwelling invertebrates allow for rapid response to degraded or changing environments, enabling rapid detection of biodiversity change.[17] For example, urbanized spaces in southern California have decreased beetle abundance and diversity, especially larger beetles, contributing to slower decomposition rates of carrion.[5] Additionally, spider communities, in particular, are sensitive to a wide variety of environmental factors such as habitat structure, type, wind exposure, temperature, and moisture, potentially providing important information on the ecological health of a system or assessment of restored ecosystems.[12, 14, 18–20] Invertebrate bioindicators can be classified into three groups based on their response to environmental change: (i) detectors, which are sensitive to environmental stress and respond by decreasing, (ii) exploiters, which can increase in abundance by exploiting the environmental changes, and (iii) accumulators, which often accumulate toxins from the environment that can be measured to assess environmental toxin levels.[7]

Urban landscapes provide perhaps some of the best examples of how invertebrates reflect the health of an ecosystem.[7] One of the most deleterious human impacts is modern development and design of urban landscapes.[21, 22] Urbanization often involves significant disturbance of native soil and plant communities, which facilitates the spread of invasive species, in particular, plants that are disturbance-tolerant, ruderal, pioneer, and/or generalists.[23–27] Invasive plants can alter soil chemistry and nutrient cycling,[28–31] change community structure through trophic interactions,[32– 35] and facilitate further invasion of exotic species.[36, 37] Changes in the community structure can negatively affect invertebrate diversity, especially habitat specialists such as many carabid beetles.[38–41]

Marschalek and Deutschman found that for ground-dwelling taxa, abundance of beetles in the family Silphidae (carrion beetles) and Staphylinidae (rove beetles) exhibited a strong negative correlation with urbanized landscapes and showed a positive correlation with sage scrub habitat, a rare and endangered habitat in southern California and northwestern Mexico.[5] Spiders, on the other hand, have been found to increase in richness within urban landscapes.[42] Depending on the taxonomic group, diversity and abundance can be affected either negatively or positively.[5, 42] Such alterations create trophic effects down the food chain by affecting parasitoids and predators, which may also be directly affected by plant diversity,[32-35] changing the invertebrate community assemblage and structure of an ecosystem.[43–45] Urbanization can also alter insect-mediated ecological processes, including insect herbivory and tritrophic interactions.[46–49]

Tijuana, Mexico is a rapidly urbanizing municipality on the US-Mexico border. Tijuana's rapid development is due to ample employment opportunities, both within the city and across the border in San Diego, California.[50] Adverse possession laws in Tijuana allows the acquisition of land as private property upon showing proof of continuous and exclusive possession.[51] These laws encourage squatting and incentivize the poorest residents of the municipality to build homes in precarious areas that result in intensifying social risk and serious ecological consequences. The area surrounding the urban centers exhibits intense anthropogenic disturbance, including pollution due to industry, lack of waste management, and massive stripping of native plants and topsoil within canyons and on the crest of hillsides.[52, 53] These disturbances facilitate the establishment of many aggressive invasive plant species. Much of Tijuana and San Diego sits on coastal sage scrub habitat, much of which is fragmented in San Diego and the majority of which is under threat of urbanization in Tijuana.[52, 53] Coastal sage scrub is dominated by deciduous Shrubs; it is very similar to the chaparral habitat of southern California, except that the majority of the shrubs are deciduous and not evergreen. In these rapidly changing landscapes, evaluating the health of this rare and endangered ecosystem are of great importance not only for conservation, but for human health.[54] By using invertebrate sampling to obtain information on ecosystem function and structure, it will be possible to understand some of the effects of this rapid development on the system.

Although the Tijuana region shares similar plant communities and ecotypes to San Diego, such as coastal sage scrub and chaparral, the invertebrate communities have not been well sampled, and areas within the canyon have little to no records for their current invertebrate communities.[55–57] With urbanization and invasive species disturbances, the invertebrate community composition of Vivero Hormiguitas, Tijuana is expected to exhibit some key differences from similar, undisturbed habitats in San Diego within the Border Field State Park. For instance, habitat loss appears to be the primary driver of decreased invertebrate diversity and beetle abundance in coastal southern California.[5] Adams et al. found insect diversity and abundance was over 30% higher at sites with native or drought tolerant plants.[58] Exotic insects are also found in higher abundance at urbanized sites with low native plant diversity.[58] These results support the idea that invertebrate communities within disturbed Tijuana sites may be experiencing, or already have experienced, a rapid change in community structure and taxonomic diversity compared to protected areas within San Diego.

Sampling the sites within Tijuana and San Diego allows not only for a unique opportunity to understand current invertebrate composition in the rapidly developing Tijuana area but also for the study of how invertebrate communities may be expected to change with urbanization. This work in the southern California and northern Baja California region will provide information on the current ecological health and function of each area and will assist in creating roadmaps for site restoration in the future.

In this study, invertebrates sampled at the disturbed and rapidly urbanizing Tijuana site and the protected site in San Diego will be used as bioindicators for ecosystem function and health; comparison of invertebrate samples at these sites will enable assessment of how invertebrate communities are changing. Currently, some of the hypotheses that explain invertebrate community compositions across urban-rural gradients include: (i) the increasing disturbance hypothesis, which state that species richness monotonously decreases from rural to urban areas as disturbance increases;[42] (ii) the matrix species hypothesis, which claims that matrix (open habitat) species dominate disturbed open habitats and abundance increases with urbanization;[42] (iii) the opportunistic species hypothesis, which predicts that opportunistic and disturbance tolerant species should increase in urban areas;[36, 37, 42] and (iv) the habitat specialist hypothesis, which posits that the abundance and richness of specialist species decreases with disturbance and therefore also in urban areas while generalists are less affected.[42]

Based on these hypotheses proposed by past research, the diversity and abundance of ground-dwelling spiders are predicted to increase in the urbanizing Tijuana site. This follows the habitat specialist hypothesis and the matrix species hypothesis as ground-dwelling spiders, such as those in Lycosidae and Agelenidae are mostly generalists, which have been found to be affected less by disturbances and in some cases, increase in abundance and diversity due to the penetration of matrix spider species into disturbed sites.[59] Beetle abundance and diversity are hypothesized to be greater in less urbanized spaces. This follows the habitat specialist hypothesis as many beetle species are habitat and diet specialists and often require specific microhabitats, that are easily destroyed by urbanization.[60] Abundance of invasive or exotic invertebrate species are expected to be greater at the urbanizing Tijuana site, following the opportunistic species hypothesis as many successful invasive or exotic invertebrate species often thrive in disturbed and urban environments.[42] Finally, following the increasing disturbance hypothesis, there should be a higher invertebrate abundance but a lower Shannon diversity index (H') value in the disturbed site of Tijuana due to increases in disturbance tolerant and/or non-native species.[42]

METHODS AND PROCEDURES

Study sites

An area located at the coordinates 32.513363, -117.091483 of approximately 3000 m^2 in Tijuana, Mexico was sampled (**Figure 1a**). The site is situated on a slight slope near a canyon peak and much of the surrounding area has been disturbed by urbanization and unofficial settlements. The site is granted by the city of Tijuana and is managed as a native plant nursery known as Vivero Hormiguitas, Tijuana (TJ). The nursery is often disturbed by weedy, invasive plants, and the areas surrounding it are disturbed by illegal dumping, clearing of vegetation, and sediment transfers from construction. Vivero Hormiguitas contains some remnant plantings of native plants, as well as horticultural exotic species such as *Schinus terebinthifolia* (Brazilian peppertree) and *Selenicereus unatus* (white-fleshed pitahaya). Much of the landscape here now is sparse and open, dominated by weedy invasive grasses and forbs such as *Glebionis coronaria* (garland daisy), *Salsola tragus* (prickley Russian thistle*)*, *Silene gallica* (small-flowered catchfly), *Malva spp.* (mallows*)*, *Bromus spp.* (brome grasses*)*, and *Erodium spp.* (stork's bills) among other annual invasives. There are a few annual, native species that occur in low abundance, including *Lupinus succulentus* (arroyo lupine*)*, *Calochortus splendens* (splendid mariposa lily), *Primula clevelandii* (Padre's shooting star), and *Sisyrinchium bellum* (western blue-eye grass). Some perennial native plants also occur in low abundance, including: *Isocoma menziesii* (coastal goldenbush), *Cneoridium dumosum* (bushrue), *Solanum parishii* (Parish's nightshade), *Hazardia orcuttii* (Orcutt's goldenbush), *Rhus integrifolia* (lemonade berry), *Eriogonum fasciculatum* (California buckwheat*)*, *Baccharis sarothroides* (desert broom), *Acmispon glaber* (deerweed), *Artemisia californica* (California sagebrush), *Malacothamnus fasciculatum* (chaparral mallow), and *Malosma laurina* (laurel sumac). Much of this slope before disturbance would have been dominated by coastal deciduous and perennial shrub-like species such as *Salvia apiana* (white sage), *Salvia mellifera* (black sage), *Encelia californica* (California brittlebush), succulents like *Dudleya spp.* (Dudleyas), and coastal cactus species mixed with evergreen chapparel plants such as *Malosoma laurina, Rhus integrifolia, and Baccharis sarothroides*. Diversity of annual forbs would likely have been higher including *Eschscholzia californica* (California poppy) and many Boraginaceae and Asteraceae species.

An area located at the coordinates 32.538950, -117.105543 of approximately 3000 m^2 in San Diego Border State Field Park, San Diego (SD) was also sampled (**Figure 1b**). The site is situated on the foothill rise of the northeast side of Bunker Hill, CA. Vegetation here is more intact and less disturbed, as areas around this site are protected due to the cultural and historical significance to the Kumeyaay. indigenous people. Native plants, and especially shrubs dominate the landscape, although invasive species such as *Erodium spp*., *Bromus spp*., *Urtica urens* (dwarf nettle) and other invasive grasses and weeds are still present in small quantities on the hillside. *Malosma laurina*, *Rhus integrifolia*, *Claytonia perfoliata* (miner's lettuce), *Nuttallanthus texanus* (Texas toadflax), *Ferocactus viridescens* (San Diego barrel cactus), *Dudleya pulverulenta* (chalk dudleya), *Salvia apiana, Salvia mellifera*, *Lupinus succulentus*, *Pseudognaphalium biolettii* (bi-colored rabbit tobacco), *Heteromeles arbutifolia* (toyon), *Sairocarpus sp*. (snapdragons), *Encelia californica*, *Camissoniopsis sp.* (suncups), *Solanum sp.* (nightshades), and various other native shrubs and annuals characteristic of coastal sage scrub exist in the sampling site. The invasive annual forbs at this site was generally spread evenly throughout, using any little bit of open space between the large shrubs to grow.

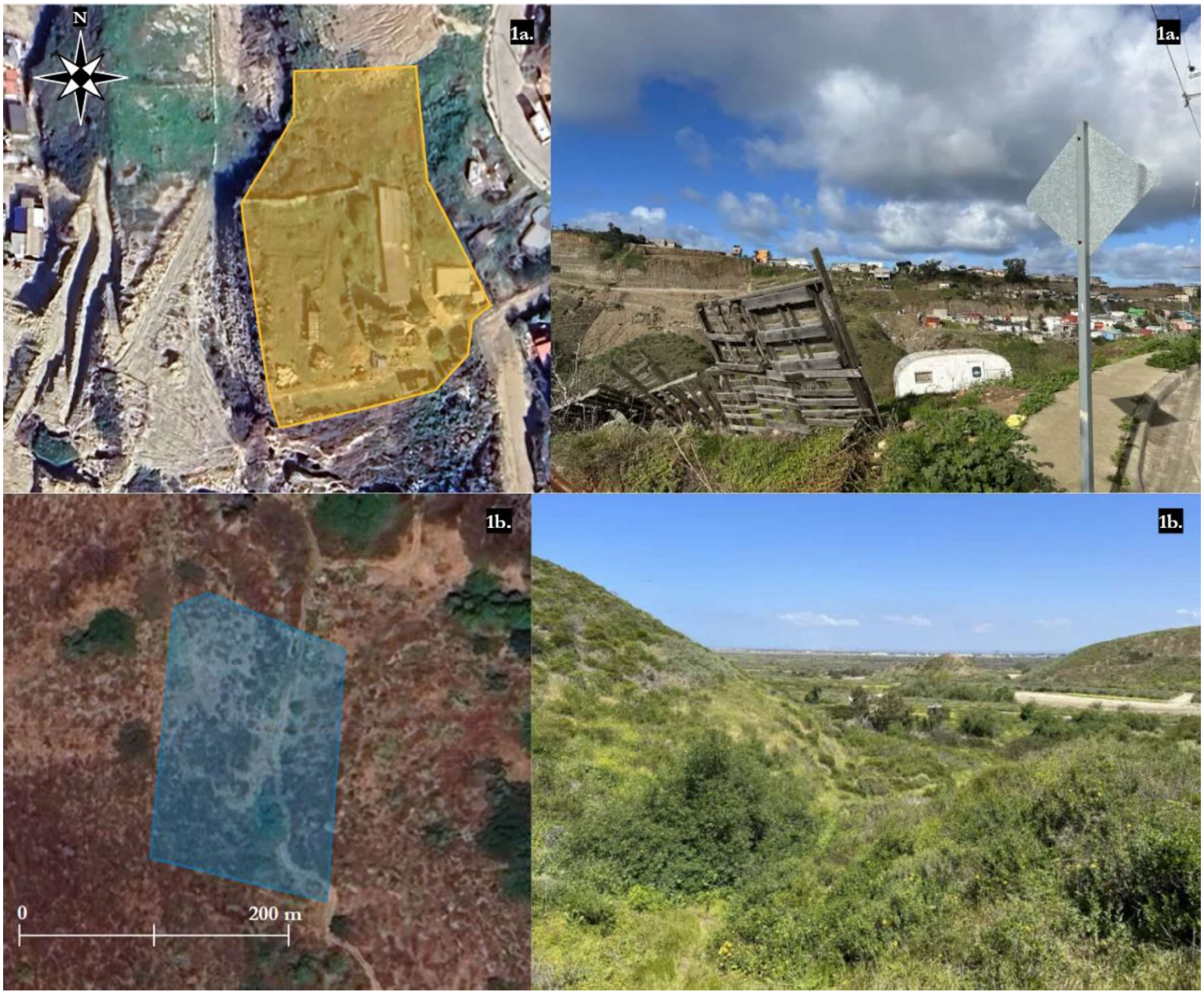

Figure 1. Shows the sampling sites from Tijuana in Vivero Hormiguitas (**1a.**) and from San Diego in Bunker Hill of Border State Field Park (**1b.**) in top view form from satellite image alongside general landscape view captured through camera on the ground.

Sampling method

Pitfall traps were placed flush to the ground using 532 ml plastic cups. The traps were filled with solution composed of 10% salt water and 5% bleach. An orange flag was placed adjacent to the trap to mark the traps (**Figure 2**). Twenty traps were randomly placed at each site by randomly generating coordinates with a computer within the select site. As invasive plants were generally homogenous throughout the sites, this method allowed for the most straightforward randomization. The traps were collected two weeks after placement and invertebrate identification followed. The site in Tijuana was sampled from February 29 to March 14, 2024, and the San Diego site was sampled from April 8 to April 22, 2024. Invertebrates were identified to the most specific taxa possible and by morphospecies. Spiders (Araneae) and beetles (Coleoptera) were heavily focused on due to their abundance, diversity, and functional group representation within terrestrial ecosystems, making them excellent ecosystem indicators.

Morphospecies were used for many spiders and insect larvae. Spiders were mostly identified to family level based on eye positioning, spinnerets, and pedipalps. Beetles, ants, isopods, dermapterans, orthopterans, hemipterans, mantids, scorpions, and cockroaches were mostly able to be identified to species or genus level. Each beetle was also categorized into their respective families for analysis. For individuals in the same family that appeared to be different, they were counted as different morphospecies. Some individuals were not able to be identified and were not the same as the other species morphologically, those individuals were counted as one morphospecies without being assigned to a family and labeled "unidentified". These "unidentified" individuals did not contribute to an additional family level richness count. Hard to identify groups such as Lepidoptera larva, silverfish, centipedes, millipedes and harvestmen that occurred in low abundance used morphospecies.

For the purpose of this study, invasive invertebrate species and other exotic species are all grouped under the label of "non-native", as it can be hard to identify what may be considered invasive, as there is often a lack of research on impacts of some of these exotic invertebrates. However, some of these invertebrates are already considered invasive, such as *Coccinella septempunctata* (seven-spotted lady beetle), *Cornu aspersum* (garden snail), and *Forficula mediterranea* (European earwig). Furthermore, all non-native invertebrates are identified to the species level as it would be almost impossible to know if an invertebrate is non-native without doing so. Due to rain causing mud to slide into and burying some traps on the slope, a total of only 17 traps were collected from the site in Tijuana and a total of 16 traps were collected from the site in San Diego. Permit to sample invertebrates in the Border Field State Park was provided by the California State Parks Department of Parks and Recreation. The permit granted is Scientific Collecting Permit 24-669-03, and is valid from March 19, 2024 to March 19, 2025.

Figure 2. Pitfall trap set up with a plastic cup flush to the ground filled with 10% salt solution + bleach (5%) with an orange flag beside it.

Pooled data analysis

The samples from each site were pooled together so that two samples are obtained, the total SD and total TJ samples. Using these pooled data, a table was created to summarize the raw data findings of the sites. The table reports raw data in both the abundance (how many counts) and richness (how many types) of each category. The categories include the total beetle abundance and richness (Order Coleoptera), Coleoptera families at each site (number of families), including the abundance and richness of species each of those families, the total spider abundance and richness (Order Araneae), Araneae families (number of families at each site), and the abundance and richness of species within each of those families. The table also reports the total abundance and richness of all other invertebrate orders collected from each site, and the total abundance and species richness of non-native invertebrate species, native invertebrate species, and total invertebrates. Centipede and millipede classes were included and counted as an order each because there was no more than one individual of each at each site and identification to order was more difficult for these groups.

From this pooled data, a couple important taxonomic groups of interest are noted, these are: (i) Carabidae, as they are abundant in both sites, and often used as biological indicators for restoration and for biodiversity assessment; they are also a group of predaceous beetles that can be used as one functional group;[60–63] (ii) Tenebrionidae, as they are also abundant within both of the sites and serve as a good alternative functional group to Carabidae, as they are mostly scavengers and detritivores;[64] (iii) Gnaphosidae, as a highly cursorial ground-dwelling spider that is often used as an environmental indicator due to their particularly sensitive nature to disturbance;[11] and (iv) Isopods, as they show a particularly large change in abundance between sites. Most of the analysis will be focused on spiders (Araneae) and beetles (Coleoptera) as these two groups are very abundant within the samples, can serve as large functional groups, and are used extensively as bioindicators.[7–11]

Statistical analysis

The Shannon diversity index (H') was calculated as a measure for diversity in morphospecies for each individual pitfall trap to conduct t-test analysis—this is termed all-taxa invertebrate H'. It is important to note that "native invertebrates" here are invertebrates excluding the non-native invertebrates identified, as some individuals not identified as non-native due to difficulty of species identification can still be exotic.

Abundance here is measured by the number of individuals sampled within the family, order, or in all-taxa totals. To carry out a t-test for abundance of a specific family or order, the number of individuals in that family or order was counted for each individual pitfall trap. To carry out a t-test for an all-taxa invertebrate abundance, the total count of invertebrates from all taxa for each individual pitfall trap was used. Total abundance for each site was also calculated by combining all pitfall traps within a site together. To carry out a t-test for non-native invertebrate all-taxa abundance, abundance was calculated for each individual pitfall trap. Total non-native invertebrate abundance is the abundance of non-native invertebrates calculated by counting the combined non-native invertebrate taxa for each site. By excluding non-native invertebrates, the all-taxa native invertebrate abundance for each pitfall trap was calculated to perform a t-test and the total native abundance was calculated.

To analyze the data obtained, multiple unequal variance t-tests were conducted after confirming unequal variance using F-test on the following response variables: beetle (Coleoptera) abundance (F-test p-value = 0.00179), spider (Araneae) abundance (F-test p-value = 7.355e-05), all-taxa invertebrate abundance (F-test p-value = 1.45e-12), non-native invertebrate abundance (F-test p-value < 2.2e-16), Carabidae abundance (F-test p-value = 1.172e-05), Isopod (Isopoda) abundance (F-test p-value < 2.2e-16), and Gnaphosidae abundance (F-test p-value = 5.923e-06). Student's t-tests were performed on variables that did not show significant unequal variances, these include: beetle (Coleoptera) H' at the species level (F-test p-value = 0.8822), beetle (Coleoptera) species richness (F-test p-value = 0.7021), spider (Araneae) H' at the species level (F-test p-value =0.4959), spider (Araneae) species richness (F-test p-value = 0.0746), Carabidae H' at the species level (F-test p-value = 0.4762), Tenebrionidae abundance (F-test p-value = 0.2195), all-taxa invertebrate species richness (F-test p-value = 0.8555), non-native invertebrate H' at the species level (F test p-value = 0.52), native invertebrate H' at the species level (F-test p-value = 0.6929), native abundance (F-test p-value = 0.2601), and all taxa H' at the species level (F-test p-value = 0.7072).

Species richness t-tests for Tenebrionidae, Carabidae, Isopoda, and Gnaphosidae were not performed because there were not enough morphospecies and individuals of each morphospecies to result in meaningful robust analysis. A t-test on non-native invertebrate species richness was not performed as the information provided by this test can be easily interpreted through the raw data as most of the same non-native species were present in both sites. A t-test on isopod species H' was not calculated or compared because there were only two non-native species of isopods present. A t-test on Gnaphosidae species H' was not calculated as only two morphospecies of Gnaphosidae were collected. All statistical analysis and data organization was performed using RStudio 2023.09.1+494 with R package ggplot2 (stacked bar graph made using function barplot())and Microsoft Excel.[65, 66]

RESULTS

In total, 923 invertebrates were collected from TJ. These individuals were made up of 766 non-native invertebrates from 10 species, 619 of which were isopods composing of two species, 189 *Armadillidium vulgare* (common pill woodlouse) and 430 *Porcellio laevis* (swift louse). 77 of the non-native individuals were dermapterans, all of which were *Forficula mediterranea* (European earwig), 33 were hymenopterans, all of which were *Linepithema humile* (Argentine ant), 23 of which were Stylommatophora (land snails and slugs), made up of 19 *Rumina decollata (*decollate snail) and four *Cornu aspersum* (garden snail), seven non-native beetles, six of which were *Coccinella septempunctata* (seven-spotted lady beetle) and one which was *Amara aenea* (common sun beetle), four non-native spiders, all of which were *Dysdera crocata* (woodlouse spider), and three non-native hemipterans, all of which were *Bactericera lavaterae* (island mallow psyllid). A total of 47 beetles of 11 morphospecies were collected from four families (also included one "unidentified" beetle larva): Carabiade, Tenebrionidae, Coccinellidae, Histeridae, and Curculionidae. 99 spiders from 10 morphospecies were collected comprising of eight families (also included one "unidentified" spider): Lycosidae, Agelenidae, Salticidae, Thomisidae, Anyphaenidae, Dysderidae, Gnaphosidae, and Theridiidae. The rest of the invertebrates collected were composed from the class of millipedes (Diplopoda) and centipedes (Chilopoda), and other invertebrate orders, including: Orthoptera, Hemiptera, Lepidoptera (larval form), and Opiliones. All individuals from Isopoda, Stylommatophora, Hymenoptera, and Dermaptera were non-native. (**Table 1**).

In total, 207 invertebrates were collected from SD. These individuals were made up of 51 non-native invertebrates from seven species, 30 of which were isopods composed of two species, three *Armadillidium vulgare* (common pill woodlouse) and 27 *Porcellio laevis* (swift louse), one of the non-native individuals was a dermapteran, which is a *Forficula mediterranea* (European earwig), five were hymenopterans, made of four *Linepithema humile* (Argentine ant) and one *Prenolepis imparis* (American winter ant), three were Stylommatophora, all of which were *Cornu aspersum* (garden snail), 12 non-native beetles, all which were *Amara aenea* (common sun beetle). A total of 112 beetles of 18 morphospecies were collected from eight families (also included one “unidentified” beetle): Silphidae, Tenebrionidae, Limnichidae, Carabidae, Scarabaeidae (subfamily Aphodiinae), Latridiidae, Hydraenidae, and Lampyridae. 32 spiders from nine morphospecies were collected from six families (also included one “unidentified” spider): Lycosidae, Salticidae, Thomisidae, Gnaphosidae, Trachelidae, and Theridiidae. The rest of the invertebrates collected were composed from various other invertebrate orders including: Orthroptera, Hemiptera, Lepidoptera (larval form), Hymenoptera, Mantodea, Zygentoma, Blattodea, Lithobiomorpha (stone centipede), Scorpiones, and Opiliones. All invertebrates from Isopoda, Stylommatophora, and Dermaptera were non-native (**Table 1**). Slugs were likely sampled at both sites, but their bodies melted in the solution and could not be identified.

TJ			**SD**		
Taxa	Total Abundance	Total Richness	Taxa	Total Abundance	Total Richness
Order Coleoptera	46	10	Order Coleoptera	112	18
Coleoptera Familes	46	4	Coleoptera Familes	112	8
Family Carabidae	16	5	Family Carabidae	39	5
Family Tenebrionidae	18	3	Family Tenebrionidae	46	7
Family Coccinellidae	8	2	Family Silphidae	13	1
Family Curculionidae	3	1	Family Limnichidae	1	1
Family Histeridae	1	1	Family Scarabaeidae	1	1
			Family Latridiidae	1	1
			Family Hydraenidae	9	1
			Family Lampyridae	1	1
Order Araneae	99	10	Order Araneae	32	9
Araneae Families	99	8	Araneae Families	32	6
Family Lycosidae	17	2	Family Lycosidae	6	2
Family Salticidae	4	1	Family Salticidae	2	1
Family Gnaphosidae	2	1	Family Gnaphosidae	20	2
Family Theridiidae	1	1	Family Theridiidae	1	1
Family Thomisidae	64	1	Family Thomisidae	1	1
Family Dysderidae	4	1	Family Trachelidae	1	1
Family Agelenidae	1	1			
Family Anyphaenidae	5	1			
Order Hemiptera	6	3	Order Hemiptera	2	2
Order Orthoptera	8	2	Order Orthoptera	3	1
Order Lepidoptera	4	4	Order Lepidoptera	3	1

Order Hymenoptera	33	1	Order Hymenoptera	11	3
Order Blattodea	0	0	Order Blattodea	3	1
Order Mantodea	0	0	Order Mantodea	2	1
Order Dermaptera	77	1	Order Dermaptera	1	1
Order Zygentoma	0	0	Order Zygentoma	1	1
Order Opiliones	5	1	Order Opiliones	1	1
Order Scropiones	0	0	Order Scropiones	1	1
Order Isopoda	619	2	Order Isopoda	30	2
Order Stylommatophora	23	2	Order Stylommatophora	3	1
Class Chilopoda	1	1	Class Chilopoda	1	1
Class Diplopoda	1	1	Class Diplopoda	0	0
Non-native Invertebrates	766	10	Non-native Invertebrates	51	7
Invertebrate Orders	917	13	Invertebrate Orders	207	14
Total Invertebrates	917	38	Total Invertebrates	207	45

Table 1. Raw data in number of individuals (abundance) and richness broken down by taxa. Total abundance and richness here is the combined count of each taxa from all pitfall traps within a site. Trends in total abundance and total richness of Order Coleoptera (total beetle species richness and abundance), Coleoptera families (total beetle abundance and number of beetle families), Order Araneae (total spider species richness and abundance), Araneae families (total spider abundance and number of spider families), Invertebrate Orders (number of invertebrates in total and number of invertebrate orders, the class Chilopoda and Diplopoda each count as one order as there is no more than one individual of these per site collected), Total Invertebrates (by species), Non-native invertebrates (by species), and all other Orders of invertebrates, millipedes (Class Diplopoda) and centipedes (Class Chilopoda). SD = Border Field State Park, San Diego and TJ = Vivero Hormiguitas, Tijuana.

Abundance

There was higher all-taxa invertebrate abundance in TJ than in SD (t = -2.2634, df = 16.286, p-value = 0.0376) (**Figure 3a**). This result was mainly due to an increase in non-native taxa, which was significantly higher at TJ than SD (t = -2.2804, df = 16.082, p-value = 0.03656) (**Figure 3b**), with the majority of the increase coming from non-native isopods, *Armadillidium vulgare* (common pill woodlouse) and *Porcellio laevis* (swift louse) (**Figure 3b** and **Figure 5c**). Abundance for the two common invasive isopod species was found to be marginally higher at TJ (t = -1.9693, df = 16.028, p-value = 0.06646) (**Figure 4c**). Native invertebrate abundance did not differ significantly between SD and TJ (t = 0.29028, df = 31, p-value = 0.7735) (**Figure 5d**).

Beetle abundance was found to be significantly higher at SD (t = 2.4618, df = 20.132, p-value = 0.02297) (**Figure 3c**), and spider abundance was found to be significantly higher at TJ (t = -3.0426, df = 19.506, p-value = 0.006557) (**Figure 3d**). The higher beetle abundance in SD comes from the increased prevalence of Tenebrionidae, a slight increase in Carabidae, and the presence of Silphidae beetles (**Figure 5a**). However, only Tenebrionidae showed a significantly higher abundance at SD (t = 2.2352, df = 31, p-value = 0.03275) (**Figure 4b**). There were more Carabidae individuals in the samples from SD, but the trend was not statistically significant (t = 1.1664, df = 17.375, p-value = 0.2592) (**Figure 4a**). The higher spider abundance in TJ is largely attributed to the prevalence of ground crab spiders (Family Thomisidae) in the genus *Xysticus* (**Figure 5b**). There was a significantly higher abundance of Gnaphosidae in SD than in TJ (t = 3.3697, df = 17.09, p-value = 0.003617) (**Figure 4d**).

Shannon Diversity Index (H')

There was no significant difference in the H' value between TJ and SD for beetles species level diversity (Order Coleoptera) (t = 1.5188, df = 31, p-value = 0.139), nor for spiders species level diversity (Order Araneae) (t = -1.7089, df = 31, p-value = 0.09747) (**Figure 6a** and **Figure 6c**). Similarly, Carabidae beetles species level H' values were not significantly different between SD and TJ (t = 0.19983, df = 31, p-value = 0.8429) (**Figure 6f**). While the non-native H' values at TJ was significantly higher than the H' value at SD (t = -5.7711, df = 31, p-value = 2.357e-06) (**Figure 6b**), there was no significant difference for the native H' values between SD and TJ (t = 0.95198, df = 31, p-value = 0.3485) (**Figure 6d**) and the all-taxa H' values between SD and TJ (t = -0.3608, df = 31, p-value = 0.7207) (**Figure 6e**).

Richness

There were significantly higher spider species richness in TJ compared to SD (t = -2.4164, df = 31, p-value = 0.02175) and higher beetle species richness in SD compared to TJ(t = 2.5504, df = 31, p-value = 0.01592) (**Figure 7b** and **Figure 7c**). The all-taxa invertebrate species richness was not found to be significantly different between SD and TJ (t = -5.1698, df = 31. p-value = 0.1266) (**Figure 7a**).

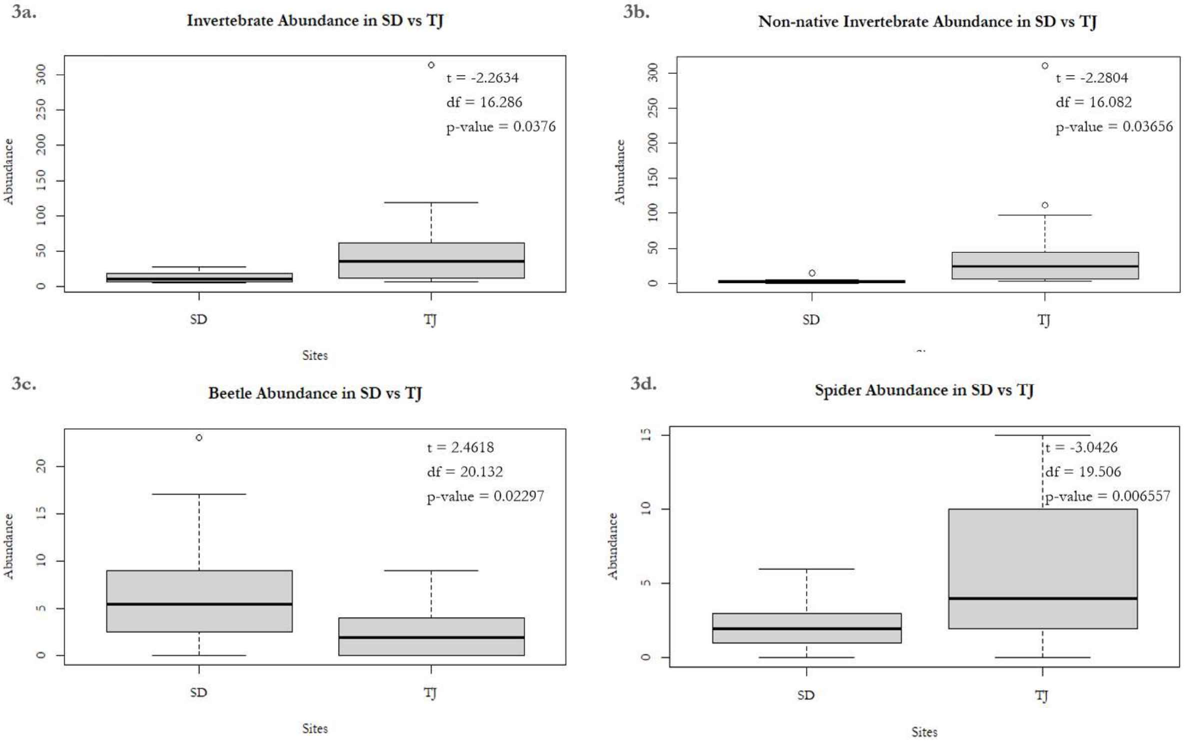

Figure 3. Boxplots depicts the all-taxa abundance of invertebrates (**3a.**), abundance of non-native invertebrates (**3b.**), beetle abundance (**3c.**), and spider abundance (**3d.**) across sites. Abundance is measured in counts of individuals. SD = Border Field State Park, San Diego and TJ = Vivero Hormiguitas, Tijuana. The degrees of freedom, t-statistic, and p-value are displayed in the legend located in the top-right corner. Outliers are represented with a white circle. The median is represented by the bolded line, first to third quartile represented by the box, and the whiskers represent the maximum and minimum values. There are more invertebrates at TJ than SD (**3a.**), more non-native invertebrates at TJ than SD (**3b.**), and more spiders at TJ than SD (**3c.**). There are more beetles in SD than TJ (**3d.**).

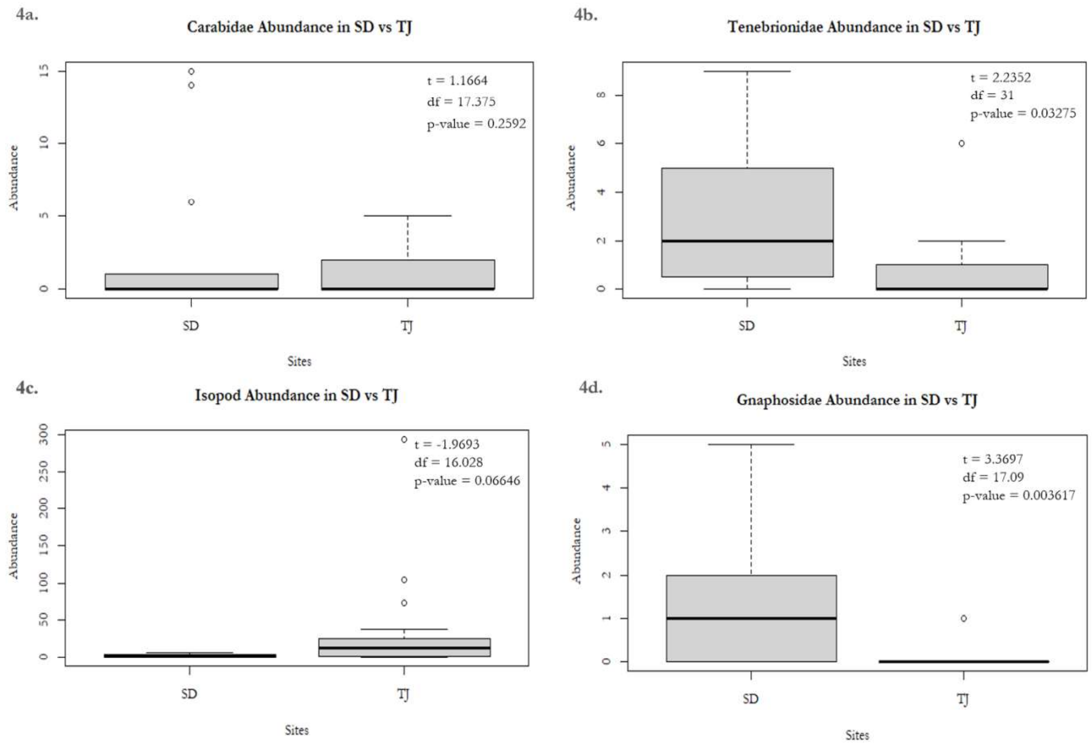

Figure 4. Boxplots of Carabidae (**4a.**), Tenebrionidae (**4b.**), Isopod (**4c.**), and Gnaphosidae (**4d.**) abundances across two sites. Abundance is measured in the count of individuals. SD = Border Field State Park, San Diego and TJ = Vivero Hormiguitas, Tijuana. The degrees of freedom, t-statistic, and p-value are displayed in the legend located in the top-right corner. Outliers are represented with a white circle. The median is represented by the bolded line, first to third quartile represented by the box, and the whiskers represent the maximum and minimum values. Carabidae abundance does not show significant difference between SD and TJ (**4a**.), while Tenebrionidae (**6b.**) and Gnaphosidae (**4d.**) abundance are higher in SD than TJ. Isopods are more abundant in TJ than SD (**4c.**).

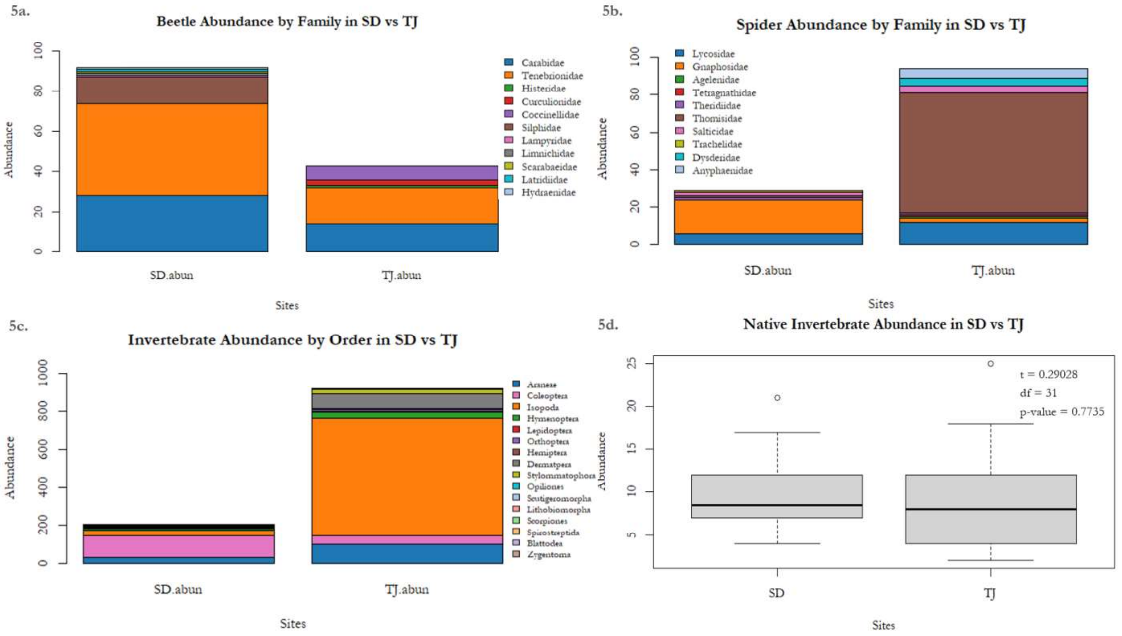

Figure 5. Stacked bar graphs of beetle abundance by family (**5a.**), stacked bar graph of spider abundance by family (**5b.**), a stacked bar graph of invertebrate abundance by order (**5c.**), and a boxplot of native invertebrate abundance (**5d.**). Abundance is measured in counts of individuals. SD = Border Field State Park, San Diego and TJ = Vivero Hormiguitas, Tijuana. The degrees of freedom, t-statistic, and p-value is displayed in the legend located in the top-right corner for **5d.** For **5d.**, Outliers are represented with a white circle. The median is represented by the bolded line, first to third quartile represented by the box, and the whiskers represent the maximum and minimum values. Tenebrionidae and Carabidae raw counts are noticeably more abundant in SD than TJ (**5a**.), Thomisidae is noticeably more abundant in TJ than SD (**5b.**), and Isopods are noticeably more abundant in TJ than SD (**5c.**). No significant difference in native invertebrate abundance between SD and TJ (**5d.**).

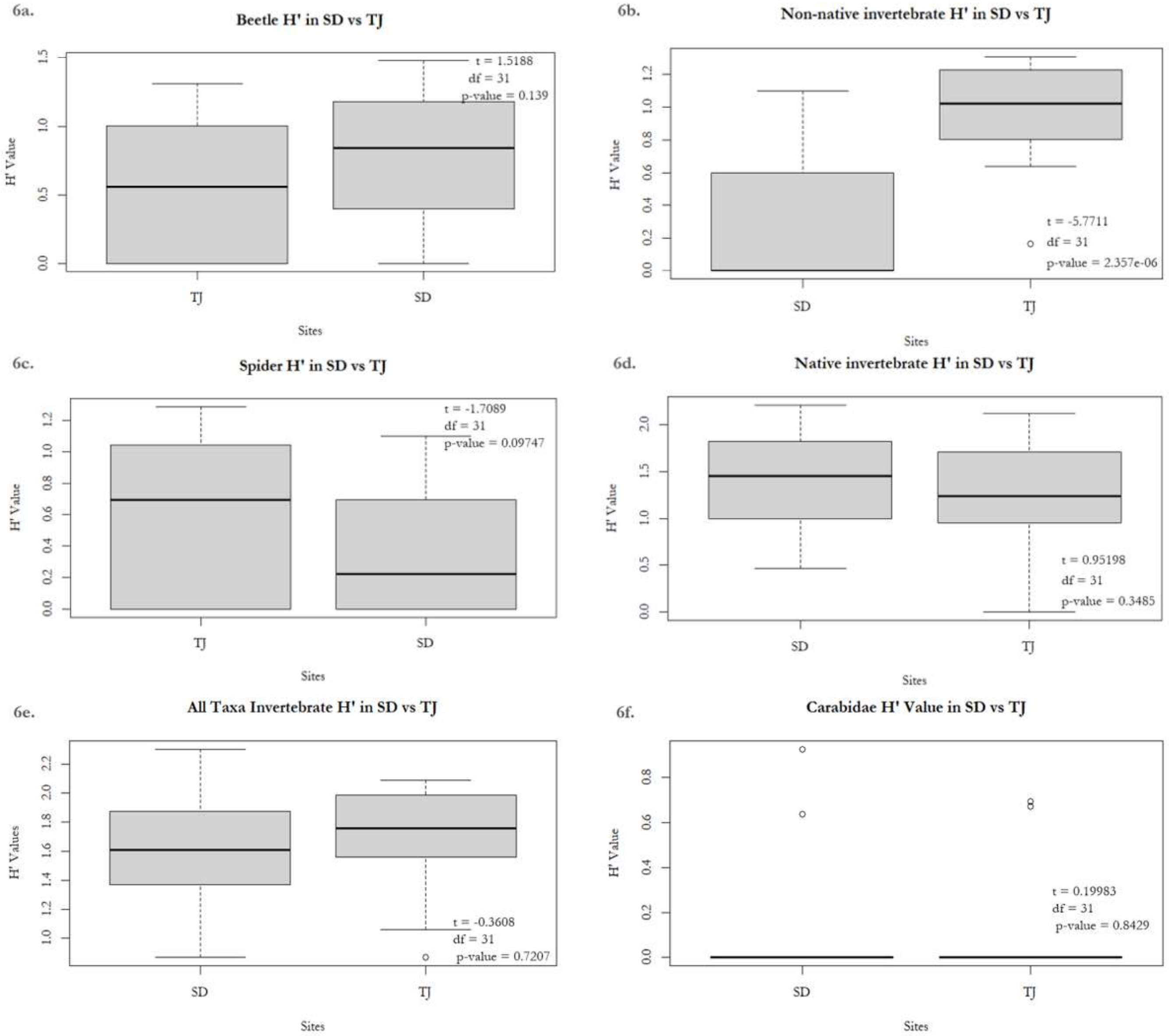

Figure 6. Boxplot of the Shannon diversity index (H') values of beetles from different samples (**6a.**), non-native invertebrate H' values (**6b.**), spider H' values (**6c.**), native invertebrate H' values (**6d.**), all taxa invertebrate H' values (**6e.**), and Carabidae H' values (**6f.**) across the two different sites. SD = Border Field State Park, San Diego and TJ = Vivero Hormiguitas, Tijuana. The degrees of freedom, t-statistic, and p-value are displayed in the legend located in the top-right or bottom-right corner. The median is represented by the bolded line, first to third quartile represented by the box, and the whiskers represents the maximum and minimum values. Beetle H' values (**6a.**), spider H' values (**6c.**), native invertebrate H' values (**6d.**), All taxa invertebrate H' values (**6e.**), and Carabidae H' values (where most samples only had one species) (**6f.**) did not differ significantly between SD and TJ . Non-native invertebrate H' is significantly higher at TJ than SD (**6b.**).

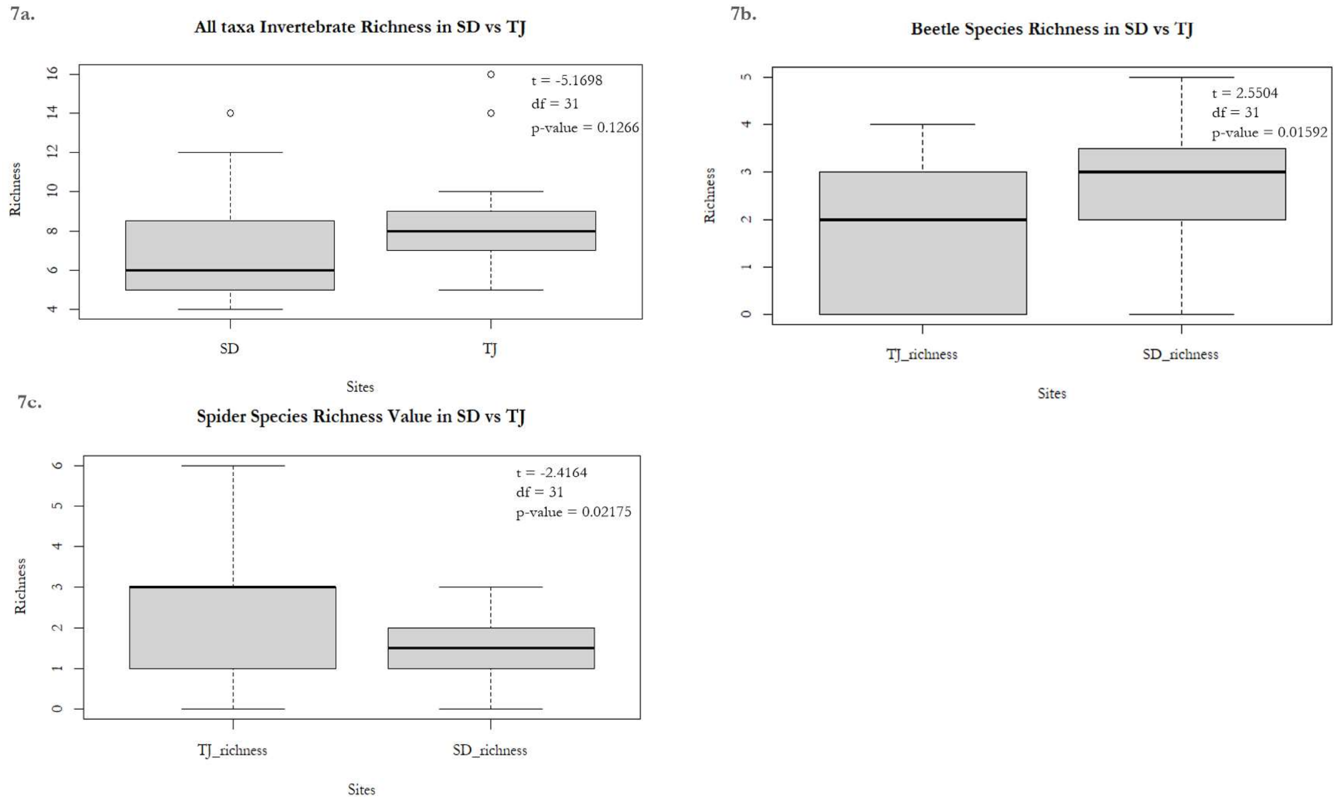

Figure 7. Boxplots of all-taxa invertebrate species richness (**7a.**), beetle species richness (**7b.**), and spider species richness (**7c.**), across two sites. SD = Border Field State Park, San Diego and TJ = Vivero Hormiguitas, Tijuana. The degrees of freedom, t- statistic, and p-value are displayed in the legend located in the top-right corners. Outliers are represented with a white circle. The median is represented by the bolded line, first to third quartile represented by the box, and the whiskers represent the maximum and minimum values. No significant difference in all taxa species richness (**7a.**) and spider species richness (**7c.**) between SD and TJ. There is significantly higher beetle species richness at the SD site compared to the TJ site (**7b.**).

DISCUSSION

The results of the study supported the hypotheses that spider abundance and non-native invertebrate abundance would increase, that all-taxa invertebrate abundance would increase, and that beetle abundance would decrease in urbanized spaces of Vivero Hormiguitas, Tijuana compared to less disturbed protected sites in San Diego Border Field State Park. The results of the study partially supported the hypothesis that beetle biodiversity would decrease in urbanized spaces of TJ compared to SD and spider diversity would increase as the species richness reflected this trend. However, the Shannon diversity index (H') did not show a significant difference between the two sites on beetle and spider diversity, nor did it show a significant difference in the all-taxa invertebrate H' value.

The all-taxa invertebrate abundance was found to be significantly higher in TJ than SD, and much of this increase in invertebrate abundance can be attributed to the non-native invertebrate species, which were significantly more abundant TJ than SD. This can be attributed to the lower abundance of invasive isopods *Armadillidium vulgare* and *Porcellio laevis* in SD. These results follow the opportunistic species hypothesis, which claims that generalist and opportunistic species, such as many non-native invertebrates in urban spaces, would increase with disturbance. These non-native invertebrates are therefore ecological exploiters, by exploiting the change caused by environmental disturbance, they become more successful in urbanized environments. One potential explanation for this finding is the relative differences in aridity of the study sites. Aridity in southern California plays an important role in resisting invasion in southern sage scrub.[67] There are also significantly fewer exotic invertebrates in less urbanized areas with less water subsidies (i.e., more arid regions).[67] One final contributing explanation could be that invasive plants are also facilitating the invasion of exotic invertebrate species.[36, 37] Many of these factors can be working in synthesis, contributing to the significant increase in non-native invertebrate taxa in urbanizing TJ compared to the protected state park in SD.

Interestingly, the native invertebrate abundance did not show a significant change between the two sites. This is potentially due to the variety of taxa and different functional groups included within this categorization. As such, some native invertebrate species that are generalists, and or ecological exploiters, may increase in abundance while others that are more sensitive to change, and/or are specialists, and are therefore detectors of change, decrease in abundance. Thus, this shifts the community composition but does not change the overall abundance.

The all-taxa invertebrate H' value did not show a significant difference between the sites, nor did the native invertebrate H' value; however, the non-native H' value was significantly higher at TJ than SD. This means that non-native species was not only more abundant in TJ than SD, but more consistently abundant across all samples in TJ compared to the SD samples. There are also a few non-native taxa completely absent in the SD samples. These included *Rumina decollata (*decollate snail), *Coccinella septempunctata* (Seven-spotted lady beetle)*, Dysdera crocata* (woodlouse spider), and *Bactericera lavaterae* (island mallow psyllid). These four species likely contributed to the difference in non-native species H' values between the two sites. It is important to note however, according to citizen-generated data on *iNaturalist*, all of these species except for *Rumina decollata (*decollate snail) have been recorded at the state park near the sample site in SD.[68] But failure to collect them within SD samples may suggest a decrease in abundance for these taxa within the protected site.

We also know from these results that the all-taxa invertebrate richness did not differ significantly between the two sites. From these results alone, it may seem like the only change when comparing the two sites is the non-native species H' and abundance. However, when we break down these broad categories into more specific taxa and categories such as native vs non-native, a different story emerges that can explain these results.

Spiders were found to be in higher abundance and richness in TJ than SD, indicative of an ecological exploiter; the change in habitat may be subsidizing or benefitting them in some way. Being generalists, these ground-dwelling spiders may be matrix species, exploiting the change from a dense shrub habitat to open grassland habitat and increases in invertebrate abundance. Shannon diversity index (H') for spiders showed only a marginally significantly higher value at TJ than SD. These results reflect the trend that generalist predators, such as many spiders, tend to benefit from urbanization. For example, generalist spiders have been found to increase in abundance with fragmentation and urbanization in Hungary.[59] Urban green spaces also tend to have increased spider abundance and richness compared to rural areas, consistent with the results of this study.[59, 69-72] Urbanization and human disturbance had little to no effect on abundance of many habitat generalist spiders.[73] However, some studies have found a reduction in spider diversity.[74] It is likely that taxonomic identity is critical when assessing the effects of human disturbance on the structure of spider communities, as although many spiders are generalist predators, they can differ drastically in hunting strategies and habitats needed to support their lifestyle.[73-75] For instance, invasive plants seem to exhibit effects on the structure of spider communities, notably by increasing preferred habitat for certain spider taxa, such as Thomisidae, Salticidae, and Theridiidae. Many invasive plant species promote spider abundance by providing ideal plant structure for webs and preferred hunting grounds, or directly, by subsidizing prey; for example, in the case of Lycosidae, where some invertebrates who are able to exploit the change in environment caused by invasive plants, such as by providing more shelter, increase in abundance, giving generalists Lycosidae spiders more prey to hunt and their abundance increases.[75-77] Still, others show a mix of effects due to the different urban environments that may occur, such as in Agelenidae.[78] The methods of sampling could influence which groups are more represented in the samples as well, an example of this can be seen from the results, wherein majority of spider taxa sampled are cursorial, such as Lycosidae, Gnaphosidae, Salticidae, and a ground-dwelling genus of Thomisidae. It should also be noted that due to the usage of morphospecies, some spider groups such as Lycosidae may be inflated in richness; however, this should have marginal effects on richness calculations as both sites compared use morphospecies, as most morphospecies are discerned because they are different families, and species richness within each spider family generally only consists of one or two morphospecies.

Gnaphosidae spiders were found to be almost entirely absent in samples from disturbed sites in TJ, while they were abundant in samples from SD. Gnaphosidae are a highly cursorial family, and evidence suggests they are more sensitive to disturbances than other groups.[79, 80] Urbanization has had a negative effect on Gnaphosidae species richness and abundance.[81] The results of this paper also show the high abundance of ground crab spiders in the genus *Xysticus* at TJ. While previous research has indicated negative effects of urbanization and human disturbance on the family to which this genus belongs (Thomisidae), studies on urbanized spaces demonstrate the continued presence of *Xysticus* in those areas.[82, 83] Results from this study and the current body of research confirms that crab spiders in the genus *Xysticus* show a tolerance to human urbanization. Taken together, these results demonstrating changes in spider abundance, richness, and community composition under anthropogenic disturbance suggest that certain taxa such as *Xysticus* may be taking advantage of reduced competition from other species. Further, it is likely that this group benefits from increased prey abundance in disturbed spaces, facilitated by increases in non-native ground-dwelling invertebrate taxa found in samples at TJ, while other taxa such as Gnaphosidae are being negatively affected by the change in habitat. These results indicate that a change in invertebrate predator community structure has occurred in the human-dominated landscape.

Beetles were found in lower abundance and richness in TJ than SD but Shannon diversity index scores (H') for these samples were not significantly different between the two sites. However, Shannon diversity index values (H') in beetles are often found to decrease across urbanized sites, implicating urbanization as a leading threat to beetle diversity.[84] This could be due to many beetle species being specialists within an ecosystem and occupying very specific niches and microhabitats, creating a mosaic of species distribution with high turnover rates at one site. Therefore, calculating H' for each individual site gives us H' values that are lower despite higher species richness, as one site may be a microhabitat that is dominated by a specialist beetle. In an urbanizing landscape such as that in TJ, this ecological complexity is destroyed and replaced with more generalist and monotonous habitat unsuitable for many specialist herbivores, such as many native beetles, so that beetle diversity within TJ is more homogenous and evenly distributed. In conclusion, while the richness and abundance of beetles may be higher in SD than TJ, the beetle species are more evenly distributed in TJ, resulting in finding little difference in H' between the two sites. Another explanation could also be that the sites in TJ were not completely urbanized as the slopes of the canyon still had intact sites with native vegetation, although it is disappearing rapidly.

There was no significant difference in Carabidae abundance between TJ and SD in this study. This can be explained as urbanization often has little effect on abundance of Carabid species as more generalist and tolerant species would increase in abundance in the absence of others.[60-62] However, this study suggests that there is no significant difference in Carabidae H' between the two sites either, which is not consistent with many other works that suggest there is a strong negative effect of urbanization on carabid beetle diversity. This is not entirely unheard of, as other studies have found that Carabidae diversity does not necessarily decrease or homogenize with urbanization, suggesting that composition is more influenced by location.[63] Most of the samples of this study also only had one species of Carabidae, suggesting that species turnover may also play a role in not finding significant differences in the H' values. This is not likely the case as both sites had the same total cumulative species richness for Carabidae. Another factor that can explain these results is the presence of invasive plant species. Invasive plants can also shift the microclimate of an environment to increase abundance of certain taxa of Carabidae while decreasing the abundance of rarer Carabidae species.[85] Human disturbance not only influences Carabidae fauna directly through alteration of landscape and native vegetation but also through the facilitation of invasive species.

There was a significant decrease in Tenebrionidae abundance within the disturbed site of TJ compared to that of SD. Although the total pooled sample count of Tenebrionidae species richness was lower in TJ than SD, there were not enough data points to run a t-test to confirm whether Tenebrionidae species richness was actually lower at TJ. Most species of Tenebrionidae within the samples are detritivores.[86] So a lower abundance in TJ could be explained by the decrease in native plant detritus due to the complete stripping of topsoil along with native plants, creating conditions that are far less hospitable for Tenebrionidae beetles. Presently, there exists little to no research on how urbanization and invasive plants could be influencing the composition of Tenebrionids. The results here suggest that there may be a negative effect on these detritivorous invertebrates, indicating a slower rate of decomposition at the human-dominated site. One other potential explanation for the difference in Tenebrionidae abundance between the two sites could be attributed to individuals within the genus *Eleodes,* which based on *iNaturalist* data suggests that they may be more abundant in May than April, as the SD dataset was sampled in May and the TJ dataset in April.[87]

Silphidae showed a presence at SD, concentrated within one sample due to the presence of a dead rodent within the sample. Dead rodents were also present in two samples in TJ, but no Silphidae beetles were observed. The Silphidae beetles collected from SD were likely attracted to recently deceased rodents that fell into the pitfall trap. Current published research indicates that Silphidae beetles are particularly sensitive to urbanization and have been shown to decrease in abundance and diversity with urbanization.[88, 89] Invasive plant species also have the potential to decrease the abundance of some Silphidae taxa.[90] Although no solid conclusions can be drawn from these samples, the observation suggests a decrease in Silphidae presence, potentially indicative of decreased decomposition rates by carrion beetles. A targeted study should be conducted to fully compare the difference in their presence between the two localities.

Limitations

Due to logistical and permitting constraints, the two locations were sampled at different times, about a month apart. However, temperature conditions were similar in March and April, as spring had already begun in coastal San Diego and Tijuana. Precipitation was slightly lower in the weeks sampling the San Diego site than the weeks sampling the Tijuana sites. However, just days before sampling began in the San Diego site, precipitation conditions were still similar in frequency and intensity. All identifiable beetles within this study's samples should be present in similar abundance in both March and April, with the possible exception of species in the genus *Eleodes*.[87] The method of invertebrate sampling using pitfall traps biases towards ground-dwelling invertebrate taxa. By focusing only on ground-dwelling taxa and only drawing conclusions based on ground-dwelling taxa, we eliminate some biases in comparing community composition. However, for groups that have large taxa in both ground-dwelling and arboreal or aerial lifestyles, such as spiders, it can only provide a limited view of the overall composition of the group. The pitfall traps are known for positive biases for ground-dwelling cursorial spiders that hunt actively, including spider families such as Salticidae, Gnaphosidae, and Lycosidae. However, differences in cursorial spider taxa among the two study sites can still yield insight into cursorial spider community composition.

The pitfall trap fluid chosen for this study could also influence the sample of ground-dwelling invertebrates.[91] However, while this may influence which invertebrates are more likely to be collected, it would not explain the difference in invertebrate abundance and composition from the different localities as both used the same solution.

CONCLUSIONS

This study is one of the few studies that has considered and characterized the invertebrate composition of a rapidly developing urban space in Tijuana, Mexico. Through the comparison of invertebrate community composition in similar, but far less disturbed habitat at the San Diego border, inferences regarding anticipated shifts in invertebrate community composition in response to the rapid and ongoing urbanization in Tijuana was made. The study found that the ground-dwelling invertebrate communities in urbanizing TJ differed from protected SD sites in a couple of key ways: (i) all-taxa invertebrate abundance was greater at TJ attributed largely to the increase in abundance of non-native invertebrate taxa; (ii) spider community composition changed with an increase in spider abundance in TJ compared to SD attributed to *Xysticus sp.* increasing in abundance, and more sensitive spider taxa such as Gnaphosidae decreased in TJ compared to SD; and (iii) Beetle abundance and richness decreased in TJ compared to SD, partially attributed to a decrease in Tenebrionidae abundance. More research on invertebrate community composition in Tijuana will provide valuable information about how rapid urbanization is changing the invertebrate composition of this area, in addition to providing important records of invertebrates in Tijuana. The Tijuana area is under-sampled, so little is known about the invertebrates there. Understanding the effects of rapid urbanization, coupled with the introduction of exotic plants in southern California coastal sage scrub is important to the continued preservation of sage scrub communities.

Most of the coastal sage scrub in San Diego is fragmented and destroyed, so the majority of intact habitat now lies in the rapidly developing areas of Tijuana, and as such is critically threatened. Future research should focus on long-term sampling of invertebrates in these areas to provide more robust datasets on invertebrate composition. Although outside the scope of this study, future research should also look at the effect of vegetation change and invasive plant species on invertebrate composition within coastal sage scrub in southern California and northwestern Baja California. This study did not sample for taxa that are not ground-dwelling, so future research should also study how aerial and arboreal taxa of this region are being affected by changes in vegetation and increasing human disturbance. The information on invertebrate community composition can be used to provide important ecological indicators to habitat restoration.

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REFERENCES

1. Eisenhauer, N., & Hines, J. (2021) Invertebrate biodiversity and conservation. *Current Biology*, 31(19), 1214–1218. *https://doi.org/10.1016/j.cub.2021.06.058*
2. Pechal, J. L., Benbow, M. E., Crippen, T. L., Tarone, A. M., & Tomberlin, J. K. (2014) Delayed insect access alters carrion decomposition and necrophagous insect community assembly. *Ecosphere*, 5(4), 1–21. *https://doi.org/10.1890/ES14-00022.1*
3. Martins, R., & Antonini, Y. (2016) Can pollination syndromes indicate ecological restoration success in tropical forests? *Restoration Ecology*, 24(3), 373–380. *https://doi.org/10.1111/rec.12324*
4. Altieri, M. (2018) Integrated Pest Management. In *Agroecology: The Science Of Sustainable Agriculture* (Altieri, M, Ed.), 2nd ed.,CRC Pres, Florida.
5. Marschalek, D. A., & Deutschman, D. H. (2022) Differing insect communities and reduced decomposition rates suggest compromised ecosystem functioning in urban preserves of southern California. *Global Ecology and Conservation*, 33, e01996. *https://doi.org/10.1016/j.gecco.2021.e01996*
6. Borges, F. L. G., Oliveira, M. da R., de Almeida, T. C., Majer, J. D., & Garcia, L. C. (2021) Terrestrial invertebrates as bioindicators in restoration ecology: A global bibliometric survey. *Ecological Indicators*, 125, 107458. *https://doi.org/10.1016/j.ecolind.2021.107458*
7. Gerlach, J., Samways, M., & Pryke, J. (2013) Terrestrial invertebrates as bioindicators: An overview of available taxonomic groups. *Journal of Insect Conservation*, 17(4), 831–850. *https://doi.org/10.1007/s10841-013-9565-9*
8. McGeoch, M. A. (1998) The selection, testing and application of terrestrial insects as bioindicators. *Biological Reviews*, 73(2), 181–201. *https://doi.org/10.1017/S000632319700515X*

9. Lawton, J. H., Gaston K. J. (2001) Indicator Species. Encyclopedia of Biodiversity, 437-450. *https://doi.org/10.1016/B0-12-226865-2/00156-5*

10. Perner, J., & Malt, S. (2003) Assessment of changing agricultural land use: Response of vegetation, ground-dwelling spiders and beetles to the conversion of arable land into grassland. *Agriculture, Ecosystems & Environment*, 98(1), 169–181. *https://doi.org/10.1016/S0167-8809(03)00079-3*

11. Majer, J. d., & Nichols, O. g. (1998) Long-term recolonization patterns of ants in Western Australian rehabilitated bauxite mines with reference to their use as indicators of restoration success. *Journal of Applied Ecology*, 35(1), 161–182. *https://doi.org/10.1046/j.1365-2664.1998.00286.x*

12. Wheater, C. P., Cullen, W. R., & Bell, J. R. (2000) Spider communities as tools in monitoring reclaimed limestone quarry landforms. *Landscape Ecology*, 15(5), 401–406. *https://doi.org/10.1023/A:1008171023039*

13. Eskchmitt, K., Weber, M., & Wolters, V. (1997) Spiders, Carabids, and Staphylinids: The Ecological Potential of Predatory Macroarthropods. In *Fauna in Soil Ecosystems* (Benckiser, G., Ed.), 1st ed., CRC Pres, New York.

14. Uetz, G. W. (1991) Habitat structure and spider foraging. In *Habitat Structure: The physical arrangement of objects in space* S. (Bell, S., McCoy, E. D., & Mushinsky, H. R., Eds.), 1st ed., 325–348, Springer, Dordrecht.

15. Bisevac, L., & Majer, J. D. (1999) Comparative Study of Ant Communities of Rehabilitated Mineral Sand Mines and Heathland, Western Australia. *Restoration Ecology*, 7(2), 117–126.

16. Kremen, C., Colwell, R. K., Erwin, T. L., Murphy, D. D., Noss, R. F., & Sanjayan, M. A. (1993) Terrestrial Arthropod Assemblages: Their Use in Conservation Planning. *Conservation Biology*, 7(4), 796–808. *https://www.jstor.org/stable/2386811*

17. Samways, M. J., McGeoch, M. A., & New, T. R. (2010) *Insect Conservation: A Handbook of Approaches and Methods* (Sutherland, W. J., Ed.), 1st ed., Oxford University Press, New York.

18. Rushton, S. P., & Eyre, M. D. (1992) Grassland Spider Habitats in North-East England. *Journal of Biogeography*, 19(1), 99–108. *https://doi.org/10.2307/2845623*

19. Rushton, S. P., Topping, C. J., & Eyre, M. D. (1987) The habitat preferences of grassland spiders as identified using Detrended Correspondence Analysis (DECORANA). *Bulletin of the British Arachnological Society*, 7, 165– 170. *https://api.semanticscholar.org/CorpusID:132121890.*

20. Wise, D. H. (1995) *Spiders in Ecological Webs* (H. J. B., & J. A., Eds), 1st ed., Cambridge University Press, New York.

21. Bueno-Suárez, C., & Coq-Huelva, D. (2020) Sustaining What Is Unsustainable: A Review of Urban Sprawl and Urban Socio-Environmental Policies in North America and Western Europe. *Sustainability*, 12(11),4445. *https://doi.org/10.3390/su12114445*

22. Dupras, J., & Alam, M. (2015) Urban Sprawl and Ecosystem Services: A Half Century Perspective in the Montreal Area (Quebec, Canada). *Journal of Environmental Policy & Planning*, 17(2), 180–200. *https://doi.org/10.1080/1523908X.2014.927755*

23. Kowarik, I. (2008) On the Role of Alien Species in Urban Flora and Vegetation. In *Urban Ecology: An International Perspective on the Interaction Between Humans and Nature* (Marzluff, J. M., Shulenberger, E., Endlicher, W., Alberti, M., Bradley, G., Ryan, C., Simon, U., & ZumBrunnen, C., Eds.), 1st ed., 321–338, Springer, Boston.

24. Pickett, S. T. A., & Cadenasso, M. L. (2009) Altered resources, disturbance, and heterogeneity: A framework for comparing urban and non-urban soils. *Urban Ecosystems*, *12*(1), 23–44. *https://doi.org/10.1007/s11252-008-0047-x*

25. Ruas, R. de B., Costa, L. M. S., & Bered, F. (2022) Urbanization driving changes in plant species and communities – A global view. *Global Ecology and Conservation*, 38, e02243. *https://doi.org/10.1016/j.gecco.2022.e02243*

26. McKinney, M. L. (2001) Effects of human population, area, and time on non-native plant and fish diversity in the United States. *Biological Conservation*, 100(2), 243–252. *https://doi.org/10.1016/S0006-3207(01)00027-1*

27. McKinney, M. L. (2006) Urbanization as a major cause of biotic homogenization. *Biological Conservation*, 127(3), 247–260. *https://doi.org/10.1016/j.biocon.2005.09.005*

28. Cook, R. W., & Talley, T. S. (2014) The invertebrate communities associated with a Chrysanthemum coronarium-invaded coastal sage scrub area in Southern California. *Biological Invasions*, 16(2), 365–380. *https://doi.org/10.1007/s10530-013-0526-8*

29. Weidenhamer, J. D., & Callaway, R. M. (2010) Direct and Indirect Effects of Invasive Plants on Soil Chemistry and Ecosystem Function. *Journal of Chemical Ecology*, 36(1), 59–69. *https://doi.org/10.1007/s10886-009-9735-0*

30. Liao, C., Peng, R., Luo, Y., Zhou, X., Wu, X., Fang, C., Chen, J., & Li, B. (2008) Altered ecosystem carbon and nitrogen cycles by plant invasion: A meta-analysis. *New Phytologist*, 177(3), 706–714. *https://doi.org/10.1111/j.1469-8137.2007.02290.x*

31. Hawkes, C. V., Wren, I. F., Herman, D. J., & Firestone, M. K. (2005) Plant invasion alters nitrogen cycling by modifying the soil nitrifying community. *Ecology Letters*, 8(9), 976–985. *https://doi.org/10.1111/j.1461-0248.2005.00802.x*

32. Price, P. W., Bouton, C. E., Gross, P., McPheron, B. A., Thompson, J. N., & Weis, A. E. (1980) Interactions Among Three Trophic Levels: Influence of Plants on Interactions Between Insect Herbivores and Natural Enemies. *Annual Review of Ecology, Evolution, and Systematics*, 11(11), 41–65. *https://doi.org/10.1146/annurev.es.11.110180.000353*

33. Siemann, E., Tilman, D., Haarstad, J., & Ritchie, M. (1998) Experimental Tests of the Dependence of Arthropod Diversity on Plant Diversity. *The American Naturalist*, 152(5), 738–750. *https://doi.org/10.1086/286204*

34. Knops, J. m. h., Tilman, D., Haddad, N. m., Naeem, S., Mitchell, C. e., Haarstad, J., Ritchie, M. e., Howe, K. m., Reich, P. b., Siemann, E., & Groth, J. (1999) Effects of plant species richness on invasion dynamics, disease outbreaks, insect abundances and diversity. *Ecology Letters*, 2(5), 286–293. *https://doi.org/10.1046/j.1461-0248.1999.00083.x*

35. Hunter, M. D., & Price, P. W. (1992) Playing Chutes and Ladders: Heterogeneity and the Relative Roles of Bottom-Up and Top-Down Forces in Natural Communities. *Ecology*, 73(3), 724–732. *https://www.jstor.org/stable/1940152*

36. Jordan, N. R., Larson, D. L., & Huerd, S. C. (2008) Soil modification by invasive plants: Effects on native and invasive species of mixed-grass prairies. *Biological Invasions*, 10(2), 177–190. *https://doi.org/10.1007/s10530-007-9121-1*

37. Flory, S. L., & Bauer, J. T. (2014) Experimental evidence for indirect facilitation among invasive plants. *Journal of Ecology*, 102(1), 12–18. *https://doi.org/10.1111/1365-2745.12186*

38. Magura, T., Tóthmérész, B., & Molnár, T. (2004) Changes in carabid beetle assemblages along an urbanisation gradient in the city of Debrecen, Hungary. *Landscape Ecology*, 19(7), 747–759. *https://doi.org/10.1007/s10980-005-1128-4*

49. Magura, T., Tóthmérész, B., & Elek, Z. (2005) Impacts of Leaf-litter Addition on Carabids in a Conifer Plantation. *Biodiversity & Conservation*, 14(2), 475–491. *https://doi.org/10.1007/s10531-004-7307-8*

40. Sadler, J. P., Small, E. C., Fiszpan, H., Telfer, M. G., & Niemelä, J. (2006) Investigating environmental variation and landscape characteristics of an urban–rural gradient using woodland carabid assemblages. *Journal of Biogeography*, 33(6), 1126–1138. *https://doi.org/10.1111/j.1365-2699.2006.01476.x*

41. Elek, Z., & Lövei, G. L. (2007) Patterns in ground beetle (Coleoptera: Carabidae) assemblages along an urbanisation gradient in Denmark. *Acta Oecologica*, 32(1), 104–111. *https://doi.org/10.1016/j.actao.2007.03.008*

42. Magura, T., Tóthmérész, B., Elisabeth, H., & Horváth, R. (2008) Urbanisation and Ground-Dwelling Invertebrates, in *Urbanization: 21st century issues and challenges* (Wagner, L.N., Ed.), 1st ed., 213–225, Nova Science Publishers, New York.

43. Strong, D. R., J. H. Lawton, S. R. Southwood. (1984) *Insects on plants. Community patter and mechanisms* (Strong, D. R., J. H. Lawton, S. R. Southwood Eds), 1st ed, Blackwell Scientific Publication, Oxford.

44. Root, R. B. (1973) Organization of a Plant-Arthropod Association in Simple and Diverse Habitats: The Fauna of Collards (Brassica Oleracea). *Ecological Monographs*, 43(1), 95–124. *https://doi.org/10.2307/1942161*

45. Andow, D. (1991) Vegetational Diversity And Arthropod Population Response. *Annual Review of Entomology*, 36, 561–586. *https://doi.org/10.1146/annurev.ento.36.1.561*

46. Kozlov, M. V., Lanta, V., Zverev, V., Rainio, K., Kunavin, M. A., & Zvereva, E. L. (2017) Decreased losses of woody plant foliage to insects in large urban areas are explained by bird predation. *Global Change Biology*, 23(10), 4354–4364. *https://doi.org/10.1111/gcb.13692*

47. Moreira, X., Abdala-Roberts, L., Berny Mier y Teran, J. C., Covelo, F., de la Mata, R., Francisco, M., Hardwick, B., Pires, R. M., Roslin, T., Schigel, D. S., ten Hoopen, J. P. J. G., Timmermans, B. G. H., van Dijk, L. J. A., Castagneyrol, B., & Tack, A. J. M. (2019) Impacts of urbanization on insect herbivory and plant defences in oak trees. *Oikos*, 128(1), 113–123. *https://doi.org/10.1111/oik.05497*

48. Turrini, T., Sanders, D., & Knop, E. (2016) Effects of urbanization on direct and indirect interactions in a tri-trophic system. *Ecological Applications*, 26(3), 664–675. *https://doi.org/10.1890/14-1787*

49. Rocha, E. A., & Fellowes, M. D. E. (2020) Urbanisation alters ecological interactions: Ant mutualists increase and specialist insect predators decrease on an urban gradient. *Scientific Reports*, 10(1), 6406. *https://doi.org/10.1038/s41598-020-62422-z*

50. Anderson, J. B. (2003) The U.S.-Mexico border: A half century of change. *The Social Science Journal*, 40(4), 535–554. *https://doi.org/10.1016/S0362-3319(03)00067-3*

51. Zamora, S., CossÍO, J. R., Pereznieto, L., Roldá n-Xopa, J., & Lopez, D. (2005) Property Law and Inheritance Law, In *Mexican Law (*Zamora, S., Cossio, J. R., Pereznieto, L., Xopa, J. R., Instituto Tecnologico Automomo de Mexico, I. T. A. de Derece, M., & Lopez, D., Eds.), 1st ed., 483–503, Oxford University Press, New York.

52. Ojeda-Revah, L., & Ochoa, Y. (2016) Fragmentación conceptual de las áreas verdes urbanas en la administración pública de México. *Ciudades. Red de Investigación Urbana A.C.*, 2–10. *https://www.researchgate.net/publication/305993414_Fragmentacion_conceptual_de_las_areas_verdes_urbanas_en_la_administracion_publica_de_Mexico*

53. Eaton-Gonzalez, R., & Mellink, E. (2015) One Shared Region and Two Different Change Patterns: Land Use Change in the Binational Californian Mediterranean Region. *Land*, 4(4), 1138–1154. *https://doi.org/10.3390/land4041138*

54. Farley, J. (2012) Ecosystem services: The economics debate. *Ecosystem Services*, 1(1), 40–49. *https://doi.org/10.1016/j.ecoser.2012.07.002*

55. Alaníz Garcia, J. (2002) Fauna of the Municipality of Tecate. In *Tecate, Baja California: Realities and Challenges in a Mexican Border Community* (Ganster, P., Cuamea Velázquez, F., Luis Castro Ruiz, J., Villegas, A., Eds.), 1st ed., 147-154, San Diego State University Press, San Diego.

56. Lüderitz, V., Gerlach, F., Jüpner, R., Calleros, J., Pitt, J., & Gersberg, R. M. (2005) Biological Assessment of Tecate Creek (U.S.–Mexico) with Special Regard to Self-Purification. *Bulletin of the Southern California Academy of Sciences*, 104(1), 1–13. *https://doi.org/10.3160/0038-3872(2005)104[1:BAOTCU]2.0.CO;2*

57. iNaturalist community. Observations of Invertebrates from Tijuana, Baja California, Mexico. Exported from *https://www.inaturalist.org on July 5, 2024.*

58. Adams, B. J., Li, E., Bahlai, C. A., Meineke, E. K., McGlynn, T. P., & Brown, B. V. (2020) Local- and landscape-scale variables shape insect diversity in an urban biodiversity hot spot. *Ecological Applications*, 30(4), e02089. *https://doi.org/10.1002/eap.2089*

59. Magura, T., Horváth, R., & Tóthmérész, B. (2010) Effects of urbanization on ground-dwelling spiders in forest patches, in Hungary. *Landscape Ecology*, 25(4), 621–629. *https://doi.org/10.1007/s10980-009-9445-6*

60. Do, Y., & Choi, M. B. (2022) Effects of urbanization on carabid beetles in an urban riparian area. *Entomological Research*, 52(6), 281–294. *https://doi.org/10.1111/1748-5967.12601*

61. Martinson, H. M., & Raupp, M. J. (2013) A meta-analysis of the effects of urbanization on ground beetle communities. *Ecosphere*, 4(5), art60. *https://doi.org/10.1890/ES12-00262.1*

62. Hartley, J. D., Koivula, J. M., Spence, R. J., Pelletier, R., & Ball, E. G. (2007) Effects of urbanization on ground beetle assemblages (Coleoptera, Carabidae) of grassland habitats in western Canada. *Ecography*, 30(5), 673–684. *https://doi.org/10.1111/j.2007.0906-7590.05199.x*

63. Magura, T., Lövei, G. L., & Tóthmérész, B. (2010) Does urbanization decrease diversity in ground beetle (Carabidae) assemblages? *Global Ecology and Biogeography*, 19(1), 16–26. *https://doi.org/10.1111/j.1466-8238.2009.00499.x*

64. VanDyk, J. (2024) BugGuide.Net: Identification, Images, & Information For Insects, Spiders & Their Kin For the United States & Canada. Iowa State University. Accessed August 28, 2024. *https://bugguide.net/*.

65. R Core Team. (2024) R: A language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria. *https://www.R-project.org/*.

66. Wickham, H. (2016) ggplot2: Elegant Graphics for Data Analysis. Springer-Verlag New York. ISBN 978-3-319-24277-4, *https://ggplot2.tidyverse.org*.

67. Staubus, W. J., Bird, S., Meadors, S., & Meyer, W. M. (2019) Distributions of Invasive Arthropods across Heterogeneous Urban Landscapes in Southern California: Aridity as a Key Component of Ecological Resistance. *Insects*, 10(1), 29. *https://doi.org/10.3390/insects10010029*

68. iNaturalist community. Observations of Invertebrate from San Diego Border Field Staet Park, San Diego, California, USA. Exported from *https://www.inaturalist.org* on July 5, 2024.

69. Philpott, S. M., Cotton, J., Bichier, P., Friedrich, R. L., Moorhead, L. C., Uno, S., & Valdez, M. (2014) Local and landscape drivers of arthropod abundance, richness, and trophic composition in urban habitats. *Urban Ecosystems*, 17(2), 513–532. *https://doi.org/10.1007/s11252-013-0333-0*

70. Bolger, D. T., Suarez, A. V., Crooks, K. R., Morrison, S. A., & Case, T. J. (2000) Arthropods in Urban Habitat Fragments in Southern California: Area, Age, and Edge Effects. *Ecological Applications*, 10(4), 1230–1248. *https://doi.org/10.1890/1051-0761(2000)010[1230:AIUHFI]2.0.CO;2*

71. Alaruikka, D., Kotze, D. J., Matveinen, K., & Niemelä, J. (2002) Carabid Beetle and Spider Assemblages along a Forested Urban–Rural Gradient in Southern Finland. *Journal of Insect Conservation*, 6(4), 195–206. *https://doi.org/10.1023/A:1024432830064*

72. Moorhead, L. C., & Philpott, S. M. (2013) Richness and composition of spiders in urban green spaces in Toledo, Ohio. *The Journal of Arachnology*, 41(3), 356–363. *https://doi.org/10.1636/P12-44*

73. Niemelä, J., & Kotze, D. J. (2009) Carabid beetle assemblages along urban to rural gradients: A review. *Landscape and Urban Planning*, 92(2), 65–71. *https://doi.org/10.1016/j.landurbplan.2009.05.016*

74. Shochat, E., Stefanov, W. L., Whitehouse, M. E. A., & Faeth, S. H. (2008) Urbanization and Spider Diversity: Influences of Human Modification of Habitat Structure and Productivity. In J. M. Marzluff, E. Shulenberger, W. Endlicher, M. Alberti, G. Bradley, C. Ryan, U. Simon, & C. ZumBrunnen (Eds.), *Urban Ecology: An International Perspective on the Interaction Between Humans and Nature,* 455–472. Springer US. *https://doi.org/10.1007/978-0-387-73412-5_30*

75. Dudek, K., Michlewicz, M., Dudek, M., & Tryjanowski, P. (2016) Invasive Canadian goldenrod (Solidago canadensis L.) as a preferred foraging habitat for spiders. *Arthropod-Plant Interactions*, 10(5), 377–381. *https://doi.org/10.1007/s11829-016-9455-7*

76. Pearson, D. E. (2009) Invasive plant architecture alters trophic interactions by changing predator abundance and behavior. *Oecologia*, 159(3), 549–558. *https://doi.org/10.1007/s00442-008-1241-5*

77. Landsman, A. P., Burghardt, K. T., & Bowman, J. L. (2020) Invasive grass (Microstegium vimineum) indirectly benefits spider community by subsidizing available prey. *Ecology and Evolution*, 10(20), 11133–11143. *https://doi.org/10.1002/ece3.6752*

78. Pessman, B. J., Hays, M., Agpawa, E., & Hebets, E. A. (2023) Urbanization affects web abundance and aggregation of a funnel weaving spider, Agelenopsis pennsylvanica (Agelenidae). *Urban Ecosystems*, 26(5), 1275–1292. *https://doi.org/10.1007/s11252-023-01379-z*

79. Cole, L. J., Pollock, M. L., Robertson, D., Holland, J. P., McCracken, D. I., & Harrison, W. (2010) The influence of fine-scale habitat heterogeneity on invertebrate assemblage structure in upland semi-natural grassland. *Agriculture, Ecosystems & Environment*, 136(1), 69–80. *https://doi.org/10.1016/j.agee.2009.11.010*

80. Jennings, N., & Pocock, M. J. O. (2009) Relationships between Sensitivity to Agricultural Intensification and Ecological Traits of Insectivorous Mammals and Arthropods. *Conservation Biology*, 23(5), 1195–1203. *https://doi.org/10.1111/j.1523-1739.2009.01208.x*

81. Kaltsas, D., Panayiotou, E., Chatzaki, M., & Mylonas, M. (2014) Ground spider assemblages (Araneae: Gnaphosidae) along an urban-rural gradient in the city of Heraklion, Greece. *EJE*, 111(1), 59–67. *https://doi.org/10.14411/eje.2014.007*

82. Argañaraz, C. I., & Gleiser, R. M. (2017) Does urbanization have positive or negative effects on Crab spider (Araneae: Thomisidae) diversity? *Zoologia (Curitiba)*, 34, e19987. *https://doi.org/10.3897/zoologia.34.e19987*
83. Burkman, C. E., & Gardiner, M. M. (2014) Urban greenspace composition and landscape context influence natural enemy community composition and function. *Biological Control*, 75, 58–67. *https://doi.org/10.1016/j.biocontrol.2014.02.015*
84. El-Surtasi, E., Semida, F., Abdel-Dayem, M., & El-Bokl, M. (2012) The threat of urbanization on beetle diversity in New Damietta city, Egypt. *Nature and Science*, 10, 15–23. *https://www.sciencepub.net/nature/ns1001/003_7542ns1001_15_23.pdf*
85. Brigić, A., Vujčić-Karlo, S., Kepčija, R. M., Stančić, Z., Alegro, A., & Ternjej, I. (2014) Taxon specific response of carabids (Coleoptera, Carabidae) and other soil invertebrate taxa on invasive plant Amorpha fruticosa in wetlands. *Biological Invasions*, 16(7), 1497–1514. *https://doi.org/10.1007/s10530-013-0587-8*
86. Thomas, D. B. (1984) The Life History and Ecology of the Pinacate Beetle, Eleodes armatus Leconte (Tenebrionidae). *The Coleopterists Bulletin*, 38(2), 150–159. *https://www.jstor.org/stable/4008164*
87. iNaturalist community. Observations of Genus *Eleodes* from San Diego County, California, USA. Exported from *https://www.inaturalist.org* on July 5, 2024.
88. Konieczna, K., Czerniakowski, Z., & Wolański, P. (2019) The occurrence and species richnes of nicrophagous Silphidae (Coleoptera) in wooded areas in different degree of urbanization. *Baltic Journal of Coleopterology*, 19(2), 213–232. *https://bjc.sggw.edu.pl/article/view/6263*
89. Shizukuda, K., Saito, M. U.(2021) Effects of human-dominated landscape on the community structure of silphid and dung beetles collected by carrion pitfall trap. *Entomological Science*, 24(2), 157–168. *https://onlinelibrary.wiley.com/doi/full/10.1111/ens.12466*
90. Walker, T. L., Jr, & Hoback, W. W. (2007) Effects of Invasive Eastern Redcedar on Capture Rates of Nicrophorus americanus and Other Silphidae. *Environmental Entomology*, 36(2), 297–307. *https://doi.org/10.1093/ee/36.2.297*
91. Kwon, T.-S., Park, Y. K., Jung, J.-K., Lee, Y. G., Park, C. W., & Park, Y.-S. (2022) Effects of preservatives in pitfall traps for collecting arthropods: A comparison of ethylene glycol and five alternative preservatives. *Journal of Asia-Pacific Biodiversity*, 15(4), 541–546. *https://doi.org/10.1016/j.japb.2022.07.001*

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PRESS SUMMARY

Invertebrates such as spiders, beetles, and insects are some of the most diverse and abundant groups of terrestrial animals. This makes them excellent indicators for the health and well-being of an ecosystem. They provide important roles within an ecosystem and can quickly respond to environmental changes such as urbanization. Urbanization, human disturbance, and the introduction of invasive plants affect invertebrates and are changing community composition, structure, diversity, and abundance of them. Some invertebrates can exploit these changes and increase in numbers; others are sensitive and disappear. The San Diego and Tijuana regions have been increasing in urbanizing pressure for decades. Tijuana is now one of the fastest growing municipalities in Mexico and urbanization threatens the integrity of much of the remaining coastal sage scrub, an endangered ecosystem. The invertebrates currently residing in these semi-disturbed landscapes have rarely been sampled and investigated. In this study, one native plant nursery in an urbanizing area of Tijuana at Vivero Hormiguitas and a protected site in Border Field State Park in San Diego are sampled to compare the differences in the invertebrate communities. There were more beetles and beetle species in protected San Diego sites lacking these disturbances while spiders appear to increase in abundance and number of species in the human-disturbed site. Lastly, invasive species, especially pill bugs, showed dramatic increase in abundance in disturbed Tijuana sites, contributing to increased overall invertebrate abundance and decreased biodiversity in Tijuana compared to San Diego protected sites. The results show how the invertebrate communities of Tijuana are currently shifting due to human impacts.