Response of a Soil Invertebrate Community to a Brief Flood Event

Erica A. Corbett
Department of Biological Sciences, Southeastern Oklahoma State University, Durant OK 74701

Abstract: Transient flooding affects both above and below ground ecosystems. Soil invertebrates may be especially susceptible because of their small size and, in some cases, lack of a cuticle. A degraded grassland area on Lake Texoma flooded in early 2019; I examined soil invertebrate communities following flooding. Transects were established in the flooded area, and compared to an adjacent non-flooded area about 3.5 meters higher. I calculated Shannon diversity indexes for each sampling period and compared abundance and number of orders present. I also examined rainfall data for the sampling periods. Soil invertebrate communities varied widely across sampling times, with a general trend of the diversity being higher in unflooded area in 2019, but both areas converging in 2020. In general, invertebrate communities recovered rapidly following flooding, suggesting some taxa may have used behavioral mechanisms to avoid the flooded area, or else survivors were able to rapidly reproduce.

Introduction

Soil invertebrates recycle leaf and plant litter and return nutrients to the soil. Their biomass varies widely between ecosystems; one estimate suggests a range from 83 to 786 kg/ha in temperate deciduous forest (Landsberg and Gower 1997). However, they are not widely studied because they are small and inconspicuous (Corbett 2015, Coyle 2017) and because identification to species requires a high degree of specialization and is often based on examination of mouthparts (Seastedt 2000). To examine the community of soil invertebrates as a whole, and especially to track changes in it over time, identification to species may not be necessary and a more-general taxonomic level could be used, especially for community-scale monitoring studies. In this study I examined abundance of individuals by order, and diversity based on order. I used the same method as in Corbett (2015), of identifying soil invertebrates to order to monitor community changes over time in response to environmental conditions.

Environmental conditions can impact soil invertebrate abundance and diversity. Temperature fluctuations (Dowdy 1944), fire (Seastedt et al 1988), grazing (Seastedt and Reddy 1991), and drought (Corbett 2015) all have the ability to affect soil invertebrate communities. Barnett and Facey (2016) suggested that terrestrial arthropods are generally sensitive to moisture changes in their habitats, because they have a high surface-to-volume ratio and some soil arthropods lack the waxy, moisture-barrier cuticle that other arthropods have. However, soil invertebrates are also known to have population resilience (Wall et al 2008), and seem to rebound fairly quickly following disturbance.

Flooding has the potential to alter soil invertebrate communities. It can cause direct mortality (Vetz et al 1979), induce dormancy, may alter soil chemistry in ways that may be harmful (e.g., anaerobic decomposition) (Ausden et al 2001), or cause them to migrate out of the area (Plum 2005). The pattern of community recovery may vary, as different groups experience different effects, or escape flooding in different ways.

Plum (2005) catalogued a variety of negative physiological effects that could happen to soil invertebrates in a flooded area, ranging from physically being carried off by flowing water, to swelling of the invertebrate's body from extreme water uptake, to poisoning by pollutants in the water or by compounds produced during anaerobic decomposition. Additionally, flooding can affect the soil habitat (through compaction, loss of soil structure, or siltation filling soil macropores) in ways that will make it less habitable by invertebrates in the future.

Vetz et al (1979) noted that an increased frequency of flooding in an area that had experienced infrequent flooding in the past would reduce species diversity and abundance, especially as compared to areas with a regular flooding regime. Over time, changes in soil invertebrate diversity and abundance could affect soil chemistry and plant growth by altering the rate of litter breakdown, nutrient turnover, and nutrient availability.

Mites, springtails, earthworms, and other small invertebrates are important parts of the detritivore web in the soil, but many of these groups are little-studied and there is not much known about community patterns over time in response to natural disturbance cycles. Vetz et al (1979) noted that "little [was] known" about the effect of disturbance on detritivore food webs and Plum et al (2005) emphasized the lack of knowledge about the effect of flooding on smaller species such as mites. There is likely a difference in effects of regular, periodic flooding versus flooding as an infrequent disturbance event, even though Plum (2005) notes that "there are no typical 'wetland' soil megafauna" and only species more-tolerant of wet conditions. Ausden et all (2001) suggest some earthworm species are more flood-adapted than others. There is evidence that isopods, millipedes, and centipedes are particularly sensitive to the effects of flooding (Plum 2005).

Lake Texoma, found in Southeastern Oklahoma, periodically floods. The lake was originally constructed, in part, for river flood control, but in the past 20 years has experienced an increased rate of flooding (USACE History of Lake Texoma). This is not an intentional management tool but rather the result of an unusually high period of rainfall. Many of the published studies on the effects of flooding (e.g., Vetz et all 1979, Ausden et al 2001) examine flooding as a regular (annual or seasonal) event, rather than an uncommon disturbance, and as a result, the soil invertebrate communities there may respond differently to a community suffering a rare flood event.

In the location of the current study, Lake Texoma near the border between Bryan and Johnston Counties, Oklahoma, flooding is infrequent and not used as a management tool. However, flooding frequency seems to be increasing in the past 20 years – after ~30 years without a flood event, the lake flooded in 2007, 2015, 2017, and again briefly (and less severely) in 2019. It is possible that climate change and increasingly-unpredictable patterns of rainfall are contributing to an increased frequency of flooding. As a result, this may increasingly be a factor in soil chemistry, soil moisture levels, compaction, and other factors that could affect soil invertebrates.

Materials and Methods

The research was conducted on a plot of US Army Corps of Engineers-managed (hereafter: USACE) land adjacent to Lake Texoma (33.99 N, 96.58 W). This land is shared between SE Oklahoma State University and USACE and is used for research and class field trips. The land is mostly used for recreation and lake access. Baseline lake level (called the conservation pool) is 619 feet above sea level (USACE Lake Texoma Data). Often in the summer the lake level is below this level. This site has occasionally flooded; when the lake is high enough to crest the spillway most access to the site is cut off, which happened in 1957, 2007, and twice in 2015 (USACE History of Lake Texoma). Parts of the site also flooded in 2017, and, most recently, in 2019. The most prolonged and extreme flood event was the 2007 event, where elevations exceeding 620 feet lasted from mid-May to mid-September, and the water crested the emergency spillway. In 2015, the flood event lasted from mid-May to Mid-August and there were two points where it was over 640 feet and crested the emergency spillway (USACE History of Lake Texoma).

The 2019 flooding was more limited in scope; maximum lake elevation was 630' feet above sea level, which flooded part of the research area. The flood lasted from early May through early July 2019. A low-lying area just north of Highway 70 was flooded during that time; an adjacent area that was about 3.5 meters higher remained dry. It is also unclear how deep the flooding penetrated: whether it was merely standing water inundating the top dozen centimeters of soil, or if it went deeper. The depth of wetting was not measured.

Having previously examined changes in soil invertebrate communities over time (Corbett 2015) and noting that the drought of 2011 affected their abundance and diversity, I wondered what effect flooding would have and how rapidly communities would rebound. After flooding receded (July 2019), I established two transects at the site; one in the recently flooded area and the other in an adjacent upland area that had not flooded. The two transects were separated by about six meters, and the unflooded area was 3-4 meters higher than the flooded area. Because soil invertebrate communities may have initially differed between the two locations due to elevation and vegetation differences, the primary objective was to compare the changes in the two communities. The flooded area was dominated by black willow saplings (*Salix nigra*), gaura (*Gaura biennis*), and grasses including Scribner's panic grass (*Dichanthelium oligosanthes var. scribnerianum*). The upland area was dominated by sericea lespedeza (*Lespedeza cuneata*) and Scribner's panic grass (Corbett, unpublished data). Both transects appear (from the USDA soil map: USDA 1978) to be a mixture of soil series; the area was disturbed and the soil replaced after disturbance – the code in the soils manual describes it as "pits." I did not do a laboratory test on the soil texture, but a quick field test suggested it was closest to a sandy clay loam in texture.

I located seven sampling points along each transect, separated by approximately 12 meters. On the first sample date (19 July 2019), surveyor's flags were placed so sample points could be relocated in the future. In total, there were five sampling events: July 2019, September 2019, October 2019, and July 2020 and October 2020. The author would have liked to have collected more samples in 2020, but campus closure in early 2020 due to the pandemic prevented samples being collected before July 2020.

Soil samples were collected using the same technique as in Corbett (2015): a 6.5 cm by 5 cm deep bulb planter was used to collect five haphazardly-spaced cores from a 2 m radius around the sampling point. Each set of cores was placed in a labeled zip-top bag and transported back to the biology department at Southeastern Oklahoma State University for extraction.

Samples were extracted by being placed in a large "funnel" with a plastic grid on the bottom (see Corbett 2015 for details) and were set over beakers containing 70% isopropyl alcohol. An incandescent light (40 watts, to avoid overheating during times when the room was unoccupied) shone on the soil for 48 hours to drive as many invertebrates as possible into the preservative. After the 48 hours, the preservative and any invertebrates captured was stored in a 100 mL plastic specimen cup with a lid. After that, a "float method" was applied (small subsamples of the soil mixed heavily with water and explored with a dissecting needle to find remaining invertebrates). Any additional invertebrates found were added to the appropriate specimen cup.

Following extraction, each sample was evaluated. The liquid and sediment in each specimen cup was dispensed into petri dish halves and examined under a dissecting microscope at 20x magnification. Organisms found were identified to order with the assistance of the "Kwik-Key to Soil Invertebrates" (Meyer, 1994). Each sample, thus, yielded both data on what orders were present in a sample as well as how many individuals of each order were present. This allowed for calculations of diversity and abundance.

To analyze the data, I first prepared tables showing the abundance of each order at each sampling date. The sample size was small (five dates with two transects on each date) so statistical comparisons were complicated by that fact. I used nonparametric testing (the Mann-Whitney U test: IBM SPSS 20, 2011) to compare total abundance and number of orders represented for the flooded vs. unflooded areas.

I also calculated Shannon indexes (H') for each transect for each sampling time. I used a base-10 logarithm with this calculation and also calculated evenness (J) as (H'/H' max)*100, where H' max was the base-10 logarithm of the number of orders represented in the sample (Magurran, 1988). Because of the nature of how Shannon indexes are calculated, the standard error is calculated differently from a typical t test. I followed the method given in Zar (2010) to perform a two-sample t test on Shannon index data, which is somewhat similar to the Behrens-Fisher t test. The standard error is calculated from a variance that is calculated based on a modification of the calculations used for the Shannon index, and the degrees of freedom is calculated similarly to that for the Behrens-Fisher test.

Five comparisons were made, flooded vs. unflooded at each sampling time.

Results and Discussion:

Table 1 lists the abundances by order for each of the sampling events; Table 2 lists the Shannon diversity and evenness values for each sampling event. Figure 1 shows a graph of the Shannon index values across the five sampling times. Notably, the flooded area starts out lower than the unflooded area, but the values converge in the second year of the study, when the unflooded area's diversity declines and the flooded area's diversity increases slightly. In fact, the main pattern seems to be one of higher but decreasing values in the unflooded area and low but increasing values in the flooded area. It is not clear why the unflooded area's diversity would decline in 2020.

The t-test comparisons of the Shannon indexes failed to achieve significance at the 0.05 value for all sampling periods. However, for the September 2019 period, the comparison was close to there being a significant difference: $t = 1.93$, critical value = 1.97 d.f. = 322. In this case the flooded area had a nearly-significantly lower value of diversity (table 2). None of the other pairs approached significance. This may be a result of small sample size (seven soil samples per transect) or that flooding genuinely does not affect species diversity of soil organisms. Because of the nature of how "standard error" is computed for statistical analysis, these values are not shown on the graph – they are typically only used for statistical analysis and as a result, they are not shown on Figure 1. Also, as Shannon index values are typically reported to three or four significant figures, I retained four decimal places in the Y axis of the graph.

There was no significant difference between flooded and unflooded areas over the course of the entire study for number of organisms ($p=.690$; SPSS does not post U values for Mann-Whitney tests). However, order number differed, with a p value of .032. The number of orders in the unflooded area was significantly higher over the span of the study (Table 1). This suggests that the main effect of flooding on the sites was in breadth of the community, rather than overall diversity.

Number of total organisms is extremely variable and can be influenced by small-scale or transient-in-time site factors. For example, if an active anthill is near where a sample was collected at a particular sampling time, large numbers of foraging ants may be collected in the sample, but nowhere else along the transect. Collembolans (springtails) also seemed to vary widely in population size between sampling periods. Russell et al (1992) suggest that collembola "react very flexibly to disturbance" and that they tend to have rapid population rebound after a disturbance. It seems likely different taxonomic groups will be affected to

Table 1. Abundance of different invertebrate orders (or higher taxonomic group, in some cases) by sampling date and site condition. Note that larvae are included in the count for their respective orders. Only orders with at least one individual present are noted for a time period.

July 2019	Unflooded area	Flooded area
Acarina	41	15
Aranae	2	0
Aschelminthes	2	0
Chilopoda	1	0
Coleoptera	31	21
Collembola	62	15
Diplura	4	4
Diptera	3	1
Gastropoda	1	0
Homoptera	2	0
Hymenoptera	13	0
Isopoda	1	0
Pauropoda	2	0
Thysanura	2	1
Total number	167	57
Orders represented	14	6
September 2019		
Acarina	34	24
Annelida	6	0
Aranae	1	3
Aschelminthes	1	0
Coleoptera	30	42
Collembola	41	118
Diplura	4	0
Diptera	4	3
Homoptera	2	1
Hymenoptera	5	3
Isoptera	0	1
Total number	94	195
Orders represented	10	8

Table 1. Continued

October 2019		
Acarina	23	79
Annelida	2	0
Aranae	1	2
Aschelminthes	1	0
Chilopoda	1	1
Coleoptera	13	20
Collembola	9	10
Diptera	1	1
Homoptera	4	4
Hymenoptera	8	8
Isopoda	3	3
Total number	145	128
Orders represented	10	9
July 2020		
Acarina	88	83
Annelida	5	2
Aranae	0	1
Coleoptera	38	53
Collembola	61	17
Diplura	10	5
Hymenoptera	11	69
Isopoda	0	1
Isoptera	0	3
Thysanura	1	0
Thysanoptera	1	0
Total number	215	238
Orders represented	9	10
October 2020		
Acarina	130	57
Annelida	3	5
Aranae	1	0

Table 1. Continued

Aschelminthes	2	1
Coleoptera	28	28
Collembola	53	15
Diplura	2	0
Homoptera	1	0
Hymenoptera	6	0
Isopoda	0	2
Pauropoda	4	0
Thysanoptera	3	0
Total number	233	108
Orders represented	11	6

different degrees by flooding; that difference might or might not show up in changes in overall community diversity. Some species of invertebrates (some Coleopteran larvae) have adaptations that allow them to withstand being in flooded areas; for example, water-repellant hair that will maintain a "bubble" of air around the larva (Barnett and Facey 2016). Ausden et al (2001) noted that recolonization following flooding could reconstitute soil-invertebrate communities (though possibly different taxonomic groups differ in their recolonization rate). In previous studies of grassland soil invertebrates (Corbett 2015), collembolans and mites (especially oribatid mites) were among the highest-abundance groups; this was also true in the current study. Beetles were the third-most-abundant group in the current study; beetles tend to be more mobile than many of the more hypogeic groups like proturans, and could recolonize the area following flooding or other disturbance.

There is a trend (Table 1) that the total numbers were higher in 2020 than in 2019; this could be a result of differences in rainfall, or recovery after a wetter year. Additionally, there may be some patterns within orders: Collembolans showed a large increase in the September 2019 sample, and there is some evidence (Coyle et al 2017) that their populations can rebound rapidly after flooding. It is also possible some organisms had migrated either laterally or to deeper regions of the soil (I did not test how deep the standing water penetrated) and migrated back as the site recovered. Presumably, different taxa will have different dispersal abilities, and that could affect community recovery over time, just as differences in reproduction rate between taxa could affect community recovery.

Rainfall amounts varied during the period of sampling although the two years did not differ greatly in total rainfall (Oklahoma Mesonet, last accessed 4/22/21). Total rainfall for 2019 was 127 cm and for 2020, it was 125.5 cm. However, in 2019, April, May, and June were high-rainfall months (49.9 cm combined) and for 2020, the April, May, and June combined rainfall were 35.1 cm). The months sampled in 2019 (July,

Table 2: Shannon diversity index (H', calculated using $\log_{10}$) and evenness (j) by sampling date and site condition.

Date	Unflooded		Flooded	
	H'	J	H'	J
July 2019	0.7568	29.18	0.6075	33.92
September 2019	0.8135	35.34	0.4948	23.80
October 2019	0.5698	24.75	0.5634	25.64
July 2020	0.6348	28.89	0.6690	29.05
October 2020	0.5740	23.94	0.5302	29.59

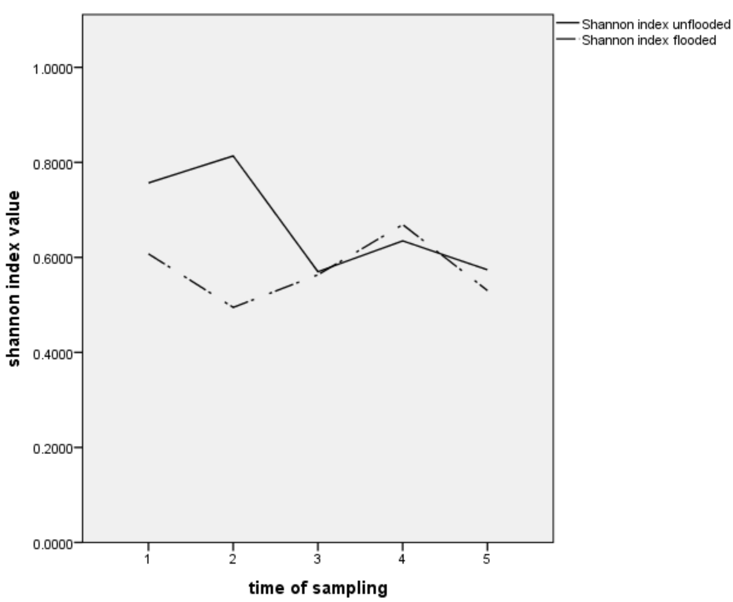

Figure 1: Shannon index values across the five sampling times. Sampling times 1, 2, and 3 were in 2019; sampling time 1 was shortly after flooding receded. Sampling times 4 and 5 were in 2020. The Shannon index values for the unflooded portion of the site are shown as a solid line, and for the flooded portion of the site, a dashed line. The Y axis is shown with four significant figures because typically the Shannon index is reported to three or four significant figures.

September, and October) had 7.8, 6.5, and 15.6 cm of rainfall, respectively, whereas July 2020 had 5.4 cm and October 2020 had 5.1 cm. Thus, the sampling times in 2020 were generally during dryer months which could affect the diversity levels of the sample; when it is drier, many soil invertebrates move to deeper levels of the soil (Barnett and Facey 2016, Dowdy 1944). The samples taken in this study were only about 5-7 cm deep in the soil.

Following the initial low level of invertebrates in the flooded area, populations seem to have rebounded quickly and there were few clear long-term affects on abundance and diversity. However, this was a short-term study (two years). Vetz et al. (1979) suggest that over longer term, with repeated flooding, there may be changes in nutrient cycling resulting from changes in the composition of the community. It is possible that climate change will cause long-term effects on both the invertebrate and plant communities throughout the temperate grassland that will alter nutrient cycling, community dynamics, and interspecific interactions (Barnett and Facey 2016), because of increased variability in rainfall regime and (possibly) increased frequency of flooding. Although these communities recovered quickly over the short term, it is possible increasing instability of rainfall regime with climate change could have long term effects for nutrient cycling and other soil alterations provided by soil invertebrates. Possible future studies could involve controlled flooding of areas and assessment of soil invertebrate communities before and after that process, focusing on individual taxonomic groups and their different responses.

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