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Short communication

Earthworm densities correlate with aboveground plant biomass and vegetation type across residential properties in Madison, Wisconsin, USA

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ABSTRACT

Non-native earthworms have been increasing in density within cities in the USA and around the world, but how earthworms respond to variation in plant cover within these systems remains unclear. We collected earthworms and standing plant biomass from five residences in Madison, Wisconsin, USA, each possessing two plant cover types—a turfgrass lawn and a prairie garden containing flora native to the Midwestern USA—on two occasions (once each in 2007 and 2008). Soil moisture and temperature were also quantified. On average, volumetric soil moisture was 2.8% lower and standing plant biomass 384 g m^{-2} greater in the prairie gardens than in the turfgrass lawns. Earthworm densities increased by 0.1 and 2.7 individuals m^{-2} on average with each increase of 1 g m^{-2} in standing plant biomass and of 1% in soil moisture, respectively. Further, when the effects of plant biomass and soil moisture were factored out, earthworm densities were estimated to be 55 individuals m^{-2} lower in prairie gardens than in adjacent lawns. These findings suggest native-flora prairie gardens may provide less favorable habitat for non-native earthworms, all else being equal.

Many cities in the USA and elsewhere are now home to large assemblages of non-native earthworms (Steinberg et al., 1997). Urban systems may be especially hospitable for such earthworms because of their often favorable soil climatic conditions (Scharenbroch and Johnston, 2011), high organic matter content (Smetak et al., 2007), and elevated nutrient influx (Szlavec et al., 2006). Earthworms have been shown to increase the activity rate of soil microbes (Scharenbroch and Johnston, 2011), foster soil aggregation (Marashi and Scullion, 2003), alter soil structure (Smetak et al., 2007), and accelerate nitrogen cycling (Steinberg et al., 1997) in urban systems, which suggests they may be changing soil function in these systems substantially (Chang et al., 2016). However, questions persist about how invasive earthworms interact with, alter, and are dispersed across different urban environments (Byrne et al., 2008; Xie et al., 2018; Moore et al., 2017).

Another change taking place in urban areas is a transition in plant cover types (Lepczyk et al., 2017). In the USA, an increasingly common practice is the replacement of residential turfgrass lawns with patches of plants native to the region (Helfand et al., 2006), in part because native-flora gardens can have lower maintenance costs than lawns while providing higher aesthetic value (Pataki et al., 2011). Native-flora gardens are also thought to contribute more than turfgrass lawns to important ecosystem services such as stormwater runoff mitigation (Roy-Poirier et al., 2010). For example, the city of Madison in south-central Wisconsin, USA, encourages the planting of residential prairie

gardens as a low-cost practice for reducing runoff (City of Madison, 2018). Previous work (e.g. Johnston et al., 2016) suggests prairie gardens can change soil properties over time relative to those below adjacent turfgrass lawns, yet the use of prairie gardens as habitats by urban earthworm assemblages has not yet been explored. We identified residences at which prairie gardens had been maintained adjacent to turfgrass lawns for at least three years (Johnston, 2006) and used these as a “natural experiment” to address the following questions: Does earthworm density vary significantly between prairie gardens and turfgrass lawns as well as with differences in aboveground plant biomass and soil climate? We hypothesized that differences in earthworm density would indeed exist as a function of these three factors.

This study was conducted in Madison, Wisconsin, USA ($43^{\circ} 08' \text{ N}$, $89^{\circ} 21' \text{ W}$), which has a temperate climate with a mean summer temperature of 21° C and an average annual precipitation of 800 mm (Wisconsin State Climatology Office, 2015). Soils in the study area are Alfisols mapped to the Dodge (fine-silty, mixed, mesic Typic Hapludalfs), McHenry (fine-loamy, mixed, mesic Typic Hapludalfs), and St. Charles (fine-silty, mixed, mesic Typic Hapludalfs) series (Soil Survey Staff, 2014). Madison is an ideal city in which to investigate the effects of residential land use changes on soil biota because landscaping utilizing native flora has become an increasingly common practice there over the past 20 years (Wisconsin Department of Natural Resources, 2016). Our study used five residences where the homeowners had

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converted a portion of their turfgrass lawn to a prairie garden of between 10 and 12 m² each in size. These residences were chosen from a larger pool studied previously (Johnston et al., 2016); we selected those that were large enough to accommodate our sampling scheme and whose landowners would cooperate. Previous work has indicated that each prairie garden and turfgrass lawn at each residence had a similar management history and soil properties (i.e., slope, aspect, landscape position, pH, texture, etc.), so it is unlikely prairie gardens were sited with respect to growing conditions for prairie flora.

The prairie gardens consisted mainly of *Andropogon gerardii* (Vitman), *Monarda fistulosa* (L.), *Schizachyrium scoparium* ([Michx.] Nash), *Solidago ohioensis* (Riddell), and *Symphytichum novae-angliae* ([L.] G.L. Nesom). The turfgrass lawns, meanwhile, were predominantly *Poa pratensis* (L.), *Trifolium repens* (L.), and *Glechoma hederacea* (L.). Prairie gardens were periodically weeded, and lawns were occasionally mowed with clippings mulched in place. Neither vegetation type was regularly fertilized, burned, or irrigated, so neither vegetation type received systematically more intensive maintenance than the other.

Property owners requested that we minimize disturbance to their properties. Therefore, all sampling was conducted on a single autumn day preceded by several days of dry, fair weather in each of two consecutive years (20 Oct. 2007 and 16 Sept. 2008). The following assumptions were implicit in our sampling approach: (1) Prairie gardens and turfgrass lawns would differ most in the fall with respect to earthworm density because differences in standing plant biomass would be greatest at that time of year; (2) an autumn sample would capture more adult earthworms than an earlier sample because earthworms would have had the entire season over which to mature; and (3) an autumn sample would be meaningfully representative of the earthworm assemblage at each site for comparison purposes.

Plant, soil, and earthworm data were collected from within 0.16-m² quadrats placed on the soil surface at four locations within each vegetation type at each residence. All aboveground plant biomass inside the quadrats was clipped at ground level, dried at 60 °C, and weighed. Soil moisture (m³ H₂O m⁻³ soil, hereafter in percent by volume) was measured 0.10 m outside each quadrat to a depth of 0.6 m (Model TH2O, Dynamax Inc., Houston, TX). Soil temperatures (± 1 °C) were measured with two analog soil thermometers placed at the center of each quadrat immediately following biomass harvesting at depths of 0.10 m and 0.30 m. These measurements were intended only to detect differences in soil climate between the two vegetation types on our sampling dates, not more broadly. Earthworms were collected by pouring 1 L of mustard solution (7.65 g L⁻¹) over the soil surface (Lawrence and Bowers, 2002); although the mustard extraction method may not capture every individual present, we assumed it would be a sufficient approach for comparing sites with similar soil conditions. Earthworms that emerged over the ensuing 30 min were collected, rinsed, and placed in jars containing moist paper towel for transport. We derived earthworm density estimates (individuals m⁻²) from the raw abundance data. We determined maturity and pigmentation from the presence of a clitellum and presence of dorsal color, respectively (Great Lakes Worm Watch, 2018). Our focus was on differences in earthworm densities among sites, so we did not quantify fresh weights. Mature specimens collected from each site were preserved to determine all taxa present. Taxonomic classifications were based on segment number, setae pairing, body size, and segment location of the clitellum (Hale, 2007).

We assessed whether aboveground plant biomass and soil climate (soil volumetric moisture content and temperature) varied by vegetation type (coded as: 1 = prairie gardens, 0 = turfgrass lawns) via a multiple mixed-effects regression model using R (package: *lme4*; function: *lmer*; Bates et al., 2015; R Core Team, 2018). Our sampling effort had to be minimally disruptive, so we were able to conduct just one data collection per year. As such, we obtained few fully independent data points. To utilize our data as powerfully as possible, quadrats (n = 80) were used as the units of replication in this model. Residence

(n = 5) and vegetation plot (a single garden or lawn at a single residence; n = 10) were included as hierarchical grouping factors to minimize the effects of pseudo-replication by penalizing our effective sample sizes and denominator degrees of freedom in proportion to the amount of pseudo-replication detected in the data (Zeger and Karim, 1991). Survey date (Fall 2007 and Fall 2008), vegetation type, and their interaction were included as fixed factors.

We used a regression with the same grouping variables to assess whether vegetation type, soil moisture, and plant biomass (fixed factors) correlated with earthworm density. Initially, survey date (Fall 2007 or Fall 2008) was included as a covariate in this regression. However, the factor covaried strongly with soil moisture (Pearson's $r = -0.73$) and was excluded to prevent co-linearity. Similarly, while soil temperatures at both depths were initially included as well, these two factors were both strongly negatively correlated with soil moisture (both Pearson's r statistics < -0.64) and were also excluded. As a result, soil moisture can be thought of as a proxy for soil temperature. All models were checked via several diagnostic plots to verify they met the assumptions of linear regressions (i.e., normality of residuals, homoscedasticity, and no unduly influential points). We deemed p values less than 0.05 significant; these were derived using the Kenward-Roger approximation (package: *lmerTest*; Kuznetsova et al., 2016). Figures were made using the *ggplot2* package (Wickham, 2009). A conditional R² estimate for the earthworm density regression was returned using the *sem.model.fits* function in the *piecewiseSEM* package (Lefcheck, 2015). To facilitate comparisons of effect sizes, we additionally report the standardized regression coefficients ($\beta_{std.}$) from the earthworm density regression model.

Soil moisture values averaged 34.4% by volume across all quadrats (range: 16.2–52.6%) and varied significantly by sampling date ($\beta = -10.32$, $p < 0.001$) and vegetation type ($\beta = -2.826$, $p = 0.032$) such that they were lower in 2008 than in 2007 and higher in turfgrass lawns than in prairie gardens. Soil temperatures at 0.10 m depth ranged from 6 to 23 °C (mean: 14 °C) and from 6 to 19 °C (mean: 12 °C) at 0.30 m depth. Aboveground plant biomass values ranged from 89.00 g m⁻² to 3,096 g m⁻² across all quadrats with a mean of 529.3 g m⁻² (median 356.9 g m⁻²). There was a significant vegetation-type-by-sampling-date interaction for plant biomass ($p = 0.010$) such that values were higher in prairie gardens in 2008 ($\beta = 394.8$) than in any other vegetation-type-and-survey-date combination. Beyond this, biomass values were significantly higher in prairie gardens than in lawns overall ($\beta = 384.4$, $p = 0.021$). These data indicate there were appreciable differences in soil climate and plant biomass between the vegetation types across the two collection dates.

The four earthworm taxa present were from a few different ecological groups: the *Aporrectodea caliginosa* complex (endogeic), *Octolasion tyrtaeum* (endogeic), *Lumbricus rubellus* (epi-endogeic), and *L. terrestris* (anecic). We did not find any of the three Asian earthworm species of genera *Amyntas* and *Metaphire* that have recently been found in Wisconsin (Chang et al., 2017). Across all quadrats and survey dates, only 31% of earthworms collected were mature, and thus we used pigmentation (a characteristic generally associated with epigeic and anecic taxa) to further characterize earthworm assemblages (Brown, 1995). Overall, 29% of the earthworms collected were pigmented and a greater percentage of earthworms collected in 2007 was pigmented (39%) than in 2008 (19%). Also, lawns had, on average, a higher percentage of pigmented earthworms (32%) than prairie gardens (26%). These differences were larger in 2007, when 46 and 33% of earthworms were pigmented in lawns and prairie gardens, respectively, compared with 19% in both vegetation types in 2008.

The mean earthworm density across all quadrats was 165 individuals m⁻² (range: 0–544 individuals m⁻²). In raw terms, the average density in turfgrass lawns was higher than in prairie gardens in 2007 (192 versus 142 individuals m⁻²), whereas the opposite was true in 2008 (149 versus 176 individuals m⁻²). However, the results of our regression (Conditional R² = 0.641) indicated that earthworm density

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