

# Declines in Woodland Salamander Abundance Associated with Non-Native Earthworm and Plant Invasions

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**Abstract:** Factors that negatively affect the quality of wildlife habitat are a major concern for conservation. Non-native species invasions, in particular, are perceived as a global threat to the quality of wildlife habitat. Recent evidence indicates that some changes to understory plant communities in northern temperate forests of North America, including invasions by 3 non-native plant species, are facilitated by non-native earthworm invasion. Furthermore, non-native earthworm invasions cause a reduction in leaf litter on the forest floor, and the loss of forest leaf litter is commonly associated with declines in forest fauna, including amphibians. We conducted a mark-recapture study of woodland salamander abundance across plant invasion fronts at 10 sites to determine whether earthworm or plant invasions were associated with reduced salamander abundance. Salamander abundance declined exponentially with decreasing leaf litter volume. There was no significant relationship between invasive plant cover and salamander abundance, independent of the effects of leaf litter loss due to earthworm invasion. An analysis of selected salamander prey abundance (excluding earthworms) at 4 sites showed that prey abundance declined with declining leaf litter. The loss of leaf litter layers due to non-native earthworm invasions appears to be negatively affecting woodland salamander abundance, in part, because of declines in the abundance of small arthropods that are a stable resource for salamanders. Our results demonstrate that earthworm invasions pose a significant threat to woodland amphibian fauna in the northeastern United States, and that plant invasions are symptomatic of degraded amphibian habitat but are not necessarily drivers of habitat degradation.

**Keywords:** *Alliaria petiolata*, *Berberis thunbergii*, earthworm invasion, forest leaf litter, invasive plants, salamander

Declinaciones en la Abundancia de Salamandras Asociadas con Invasiones de Lombrices y Plantas No Nativas

**Resumen:** Los factores que afectan negativamente a la calidad del hábitat de vida silvestre son una preocupación mayor para la conservación. Las invasiones de especies no nativas en particular, son percibidas como una amenaza global para la calidad del hábitat. Evidencias recientes indican que algunos cambios en las comunidades de plantas de sotobosque en bosques templados de Norte América, incluyendo invasiones por tres especies de plantas no nativas, son facilitados por la invasión de lombrices de tierra no nativas. Más aun, las invasiones de lombrices de tierra no nativas pueden causar una reducción en la capa de hojarasca en el suelo del bosque, y la pérdida de hojarasca comúnmente se asocia con declinaciones en la fauna forestal, incluyendo anfibios. Realizamos un estudio de marca-recaptura para determinar si las invasiones de lombrices o plantas se asociaban con la abundancia de salamandras. La abundancia de salamandras declinó exponencialmente con la disminución del volumen de hojarasca. No hubo relación significativa entre la cobertura de plantas invasoras y la abundancia de salamandras independientemente de los efectos de la pérdida de hojarasca debido a la invasión de lombrices de tierra. El análisis de la abundancia de presas

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*selectas de salamandras (excluyendo lombrices de tierra) en cuatro sitios mostró que la abundancia de presas declinó con la declinación de bojarasca. La pérdida de capas de bojarasca debido a las invasiones de lombrices no nativas parece estar afectando negativamente a la abundancia de salamandras, en parte porque las declinaciones en la abundancia de artrópodos pequeños que son un recurso estable para las salamandras. Nuestros resultados demuestran que las invasiones de lombrices representan una amenaza significativa para la fauna de anfibios en el noreste de Estados Unidos y que las invasiones de plantas son sintomáticas de un hábitat de anfibios degradado, pero no necesariamente son las causas de esa degradación.*

**Palabras Clave:** *Alliaria petiolata*, *Berberis thunbergii*, hojarasca de bosque, invasión de lombrices de tierra, plantas invasoras, salamandra

## Introduction

The spread of non-native species is a significant component of anthropogenic environmental change (Vitousek et al. 1997). Whether invasions by non-native species are a primary cause of the decline or extirpation of populations of native species is an area of active debate (Gurevitch & Padilla 2004; Clavero & García-Berthou 2005). One reason for the debate is that invasions by a particular species often coincide with other environmental changes, such as habitat fragmentation, habitat alteration, and invasions by other species, that may have their own direct or indirect contribution to declines (Gurevitch & Padilla 2004). For example, declines in coastal horned lizard populations (*Phrynosoma coronatum*) are linked to habitat loss and fragmentation and to the invasion of Argentine ants (*Linepithema humile*), which tend to occur in heavily fragmented habitats (Suarez et al. 1998; Suarez et al. 2000; Suarez & Case 2002). Similarly, studies show that increased predation rates on bird nests in heavily fragmented habitats may be linked to the greater abundance of invasive shrubs in those habitats (Schmidt & Whelan 1999; Borgmann & Rodewald 2004). In both cases the roles of either fragmentation or invasion in population declines cannot be fully understood without considering the other factors and their potential interactions.

Such is the case with plant invasions of northern temperate forests of North America. A suite of European and Asian plant species are invading mature forests, raising concerns about their impacts on wildlife habitat. For example, we received anecdotal reports of apparent declines in woodland salamander abundance coincident with invasions by garlic mustard (*Alliaria petiolata*). Because non-native plant invasions can negatively affect amphibian habitat (e.g., purple loosestrife [*Lythrum salicaria*] can reduce wetland quality for larval American toads [*Bufo americanus*] (Brown et al. 2006), and Japanese knotweed [*Fallopia japonica*] in old fields is linked to reduced amphibian foraging success) (Maerz et al. 2005a), it is possible that perceived associations are accurate, and that non-native plant invasions are contributing to declines in woodland salamander populations. However, it is also possible that a decline in sala-

mander abundance is coincidental with non-native plant invasion, and that separate processes explain the concurrent patterns.

Nuzzo et al. (2009), report that garlic mustard and 2 other non-native plant invasions are positively associated with non-native earthworm invasions and concurrent reductions of forest leaf litter layers (see also Bohlen et al. 2004a, 2004b; Hale et al. 2005, 2006). Other processes that contribute to litter loss, such as deforestation, lead to declines in plethodontid salamander abundance (Pough et al. 1987; Petranka et al. 1993; Ash 1997). The loss of leaf litter may alter forest floor microclimates, which have a significant impact on plethodontid salamander abundance (Feder 1983). Among the known impacts of leaf litter loss driven by earthworms is the decline in invertebrate soil fauna (reviewed by Bohlen et al. 2004b; Migge-Kleian et al. 2006), including small invertebrates such as springtails (Collembola), mites (Acarina), ants (Hymenoptera: Formicidae), beetles (Coleoptera), and Lepidopteran and Dipteran larvae that compose the more stable constituents of adult and juvenile salamander diets (Maglia 1996; Maerz et al. 2005c). Therefore, the loss of leaf litter due to earthworm invasions may be the actual cause of declines in salamander abundance and the simultaneous cause of invasions by non-native plants.

We used comparative mark-recapture studies at *A. petiolata* and *Berberis thunbergii* (*B. thunbergii*; Japanese barberry) sites described in Nuzzo et al. (2009) to test the hypothesis that non-native plant cover and reduced leaf litter volume are associated with reduced abundance of woodland salamanders. We also hypothesized that salamander prey abundance, excluding earthworms, declines with a reduced litter layer and an increased non-native plant cover.

## Methods

### Study Sites

We used 10 of the 15 mature hardwood forests described previously in Nuzzo et al. (2009) for our study: 5 in central New York with established invasions of *A. petiolata*,

and 5 in northeastern Pennsylvania with established invasions of *B. thunbergii*. All sites were north of the southern limit of the Wisconsin Glacial Boundary, which is within the region of northern hardwood forests that are believed to be historically earthworm-free (Hendrix & Bohlen 2002). The *A. petiolata* sites were generally low-elevation forests dominated by sugar maple (*Acer saccharum*), red oak (*Quercus rubra*), white ash (*Fraxinus americanus*), sweet birch (*Betula lenta*), and white oak (*Q. alba*). The *B. thunbergii* sites were high-elevation sites in the Pocono Mountains and were dominated by black oak (*Q. nigra*), chestnut oak (*Q. prinus*), sugar maple, white ash, and tulip poplar (*Liriodendron tulipifera*). Native understory plant communities were dominated by tree seedlings, perennial and annual herbs, and ferns at the *A. petiolata* sites, and by native shrubs (primarily Ericaceae) at the *B. thunbergii* sites (Nuzzo et al. 2009). Mature trees were generally shorter, with smaller crowns at the *B. thunbergii* sites compared with the *A. petiolata* sites, resulting in greater light penetration and consequently drier soils at the *B. thunbergii* sites. Some sites had a history of limited logging but not within the past 60 years, which is sufficient time (assuming no other disturbance) to restore forest organic layers and salamander populations to pre-cut levels (Petranka et al. 1993; Ash 1997). Each site had a well-developed invasion front (a clearly delineated transition from a habitat invaded by non-native plants to an area without non-native plants), with no other apparent features correlated with that transition. We designated the areas on each side of the invasion front as invaded or uninvaded, depending on the presence of the target non-native plant species.

### Salamander and Environmental Monitoring

At each site, we established 15, 1 × 1 m study plots on each side of the invasion front. We placed 2, 100 × 25 × 6 cm coverboards made from rough-cut, untreated sugar maple on opposing sides of each plot (one board per side). To install coverboards, we cleared away all leaf litter and duff, tamped the mineral soil, and placed boards directly on the mineral soil. We redistributed the leaf litter that we had removed around the perimeter of each board. All coverboards were installed at least 5 months prior to sampling. We checked coverboards on 12 occasions (every 2–4 weeks) from early April to late October. We collected all salamanders present under a coverboard, identified the species and sex, measured them (snout-vent length and wet mass), marked each individual uniquely using visible implant elastomers (Northwest Marine Technologies, Shaw Island, Washington, D.C.) (Davis & Ovaska 2001; Bailey et al. 2004b), and released them immediately at the edge of the coverboard.

The measures of native and non-native plant cover, leaf litter levels, and earthworm biomass are described in Nuzzo et al. (2009). In brief, we measured the per-

cent cover of all native and non-native plant species in each plot repeatedly for 2–4 years (depending on the site) to determine maximum cumulative cover. We measured leaf litter volume in early autumn when annual cumulative litter loss due to soil fauna was greatest but fresh litter had not yet fallen to the forest floor. The volume was determined by measuring litter depth at 4 locations within each plot, calculating the mean depth of all non-zero values, and multiplying that mean by the percent cover of organic matter within the plot. While sampling for salamanders under coverboards, we also counted the number of earthworms in each of 3 size classes (small, <5 cm; medium, 5–10 cm; large, >10 cm) for each of 3 guilds: epigeic, anecic, and endogeic (Hendrix 1995; Hendrix & Bohlen 2002; see descriptions by Hale et al. 2006). We multiplied count data by the mean fresh mass of earthworms for that guild and size class to determine the fresh biomass of worms for each plot during each visit. We then estimated the “mean active earthworm fresh biomass” for 1 year for each plot (from this point forward, the term earthworm biomass refers to active fresh weight).

We do not know whether salamanders or earthworms exhibit diel activity patterns, and the logistics of covering 600 coverboards over 10 sites across 2 states limited our ability to standardize sampling times or conditions. Multiple sites were seldom covered in a single day, so most sampling took place in the early morning. We reversed the order of visits when we were able to visit 2 sites in a single day. To minimize transport of invasive species, we always sampled uninvaded plots before sampling invaded plots at a site.

Once in spring and once in autumn of 2001, we measured salamander prey availability in the invaded and uninvaded habitats of 4 of the *A. petiolata* sites. In each season we placed a 25 × 25 cm frame at 4 random locations around the perimeter of each 1-m<sup>2</sup> plot and collected all leaf litter down to the mineral soil. The 4 samples were homogenized into a single sample for each plot and stored in a cotton pillowcase in a cooler for transport to the laboratory. At the laboratory we placed each sample in a Berlese funnel under a 25-watt incandescent bulb for 5 d (large samples were subdivided into multiple funnels). Invertebrates were collected in whirlpaks containing 15% EtOH. The invertebrates were identified to the highest taxonomic resolution possible. For each sample we counted the number of invertebrates in 2 select groups that are volumetrically important in salamander diets at similar sites in the region (Maerz et al. 2005c): ants (Hymenoptera: Formicidae) and beetles (Coleoptera: Carabidae, Curculionidae, and Staphylinidae).

### Salamander Abundance Estimation

We used closed population models in program CAPTURE to estimate the mean salamander abundance for each

habitat (for a detailed description and justification for using these models in studies of plethodontid salamanders, see Bailey et al., 2004a). We used the “appropriate” function in CAPTURE to compare different closed population models, and then we estimated salamander abundance with each model. Although a single model was often determined “most appropriate,” other models were often relatively similar in their weight. Furthermore, the  $M_{tbb}$  model, which other studies indicate is the best-fit model for plethodontids, does not allow for population size estimation (Bailey et al. 2004a). Therefore, rather than using a single abundance estimate from the most appropriate model, we used an average of all abundance estimates, with each estimate weighted by its relative appropriateness.

### Statistical Analyses

We used estimated mean values for both habitats at each site in all our analyses. We used paired  $t$  tests to determine whether earthworm biomass and leaf litter volume differed between non-native plant-invaded and adjacent uninvaded habitats. We used a 2-factor analysis of variance (ANOVA) to test the hypothesis that non-native plant invasions are associated with reduced salamander abundance. We treated measures of salamander abundance in invaded and uninvaded habitats within sites as repeated measures, and because the study sites for our focal invasive plant species were distributed in 2 different regions, we included target invasive plant as a second fixed factor. Then we used a general linear model, again with target invasive plant as a fixed factor, to test the hypotheses that salamander abundance is positively correlated with native plant cover and litter volume and negatively correlated with invasive plant cover. We used this model to test only for main effects and treated habitats within sites as independent replicates because measures of plant cover and litter volume were generally different between habitats. We also used a general linear model to determine whether litter volume was negatively correlated with earthworm biomass in both focal regions. Finally, we used a general linear model with season as a repeated measure to determine whether the salamander prey abundance, excluding earthworms, declined with decreasing litter volume among habitats. When necessary, dependent variables were log-transformed for linear analyses (Zar 1999).

### Results

Independent of invasion, leaf litter volume was significantly greater among *Berberis* study sites compared with *Alliaria* study sites (mean squares [MS] = 0.0003,  $F_{1,16} = 9.233$ ,  $p = 0.008$ ; Fig. 1a), but declined similarly with increasing non-native earthworm biomass (MS = 0.0005,  $F_{1,16} = 15.712$ ,  $p = 0.001$ ; Fig. 1a) in both regions (fo-

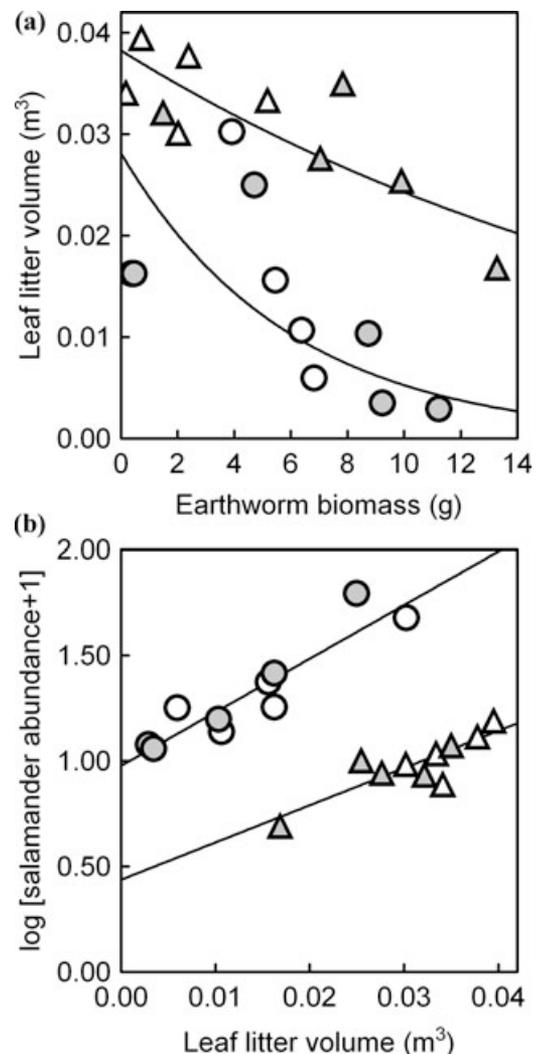


Figure 1. Correlations between (a) the mean “annual active earthworm biomass” and the mean autumn leaf litter volume and (b) the mean autumn leaf litter volume and *P. cinereus* abundance among forest habitats in central New York and northeastern Pennsylvania (U.S.A.) (circles, *Alliaria petiolata* sites; triangles, *Berberis thunbergii* sites; gray, habitats invaded by non-native plants; white, habitats without invasive plants).

cal plant/region  $\times$  earthworm biomass interaction: MS = 0.000012,  $F_{1,16} = 0.344$ ,  $p = 0.566$ ; Fig. 1a). Estimated earthworm biomass tended to be greater (paired  $t$  test:  $t = -3.900$ ,  $df = 9$ ,  $p = 0.004$ ) and, consequently, leaf litter volume tended to be lower (paired  $t$  test:  $t = 4.258$ ,  $df = 9$ ,  $p = 0.002$ ) in habitats invaded by non-native plants compared with adjacent uninvaded habitat.

Salamander communities were dominated by a single plethodontid species, the eastern red-backed salamander (*Plethodon cinereus*), which accounted for 80–99% of all individuals captured across sites. Other plethodontid

species encountered in low abundance were the northern slimy salamander (*P. glutinosus*), Allegheny mountain dusky salamander (*Desmognathus ochrophaeus*), northern two-lined salamander (*Eurycea bislineata*), four-toed salamander (*Hemidactylium scutatum*), northern spring salamander (*Gyrinophilus porphyriticus*), and northern red salamander (*Pseudotriton ruber*). Two salamander species that were not plethodontids were encountered in low abundance: yellow-spotted salamander (*Ambystoma maculatum*) and eft stages of the eastern red-spotted newt (*Notophthalmus viridescens*). The capture rates of species other than *P. cinereus* were too low to estimate whether species composition differed among sites or between invaded and uninvaded habitats. Limited captures and the fact that many of the subordinate species (except *P. glutinosus*) migrate seasonally made mark-recapture estimates of abundance untenable for these species, so we used data only on *P. cinereus* abundance in our subsequent analyses.

The paired analysis indicated that *P. cinereus* abundance among *Alliaria* sites was significantly higher than among *Berberis* sites (MS = 0.563,  $F_{1,8} = 7.682$ ,  $p = 0.024$ ; Fig. 1b), but within-site abundance did not differ significantly between invaded and uninvaded habitats (MS = 0.026,  $F_{1,8} = 2.178$ ,  $p = 0.178$ ) in either region (focal plant  $\times$  habitat: MS = 0.009,  $F_{1,8} = 0.709$ ,  $p = 0.424$ ). Within sites, *P. cinereus* abundance in invaded and uninvaded habitats was highly positively correlated ( $r = 0.872$ ,  $p = 0.001$ ). Within both regions, *P. cinereus* abundance declined exponentially with decreasing leaf litter volume (*Alliaria* sites:  $r = 0.918$ ,  $p < 0.001$ ; *Berberis* sites:  $r = 0.850$ ,  $p = 0.002$ ; Fig. 1b). General linear analysis in which transformed ( $\log_{10}[\text{abundance} + 1]$ ) abundance estimates were used also showed that *P. cinereus* abundance differed regionally (MS = 1.057,  $F_{1,15} = 116.878$ ,  $p < 0.001$ ; Fig. 1) as a positive function of leaf litter volume (MS = 0.581,  $F_{1,15} = 64.199$ ,  $p < 0.001$ ; Fig. 1) but was not correlated with percent cover of native (MS < 0.001,  $F_{1,15} = 0.006$ ,  $p = 0.939$ ) or non-native plants (MS = 0.010,  $F_{1,15} = 1.093$ ,  $p = 0.312$ ).

*P. cinereus* prey (ants and beetles) availability varied significantly between seasons and within and among sites as a function of leaf litter volume. Spring prey abundance was exponentially positively correlated with leaf litter volume, so we used transformed ( $\log_{10}[\text{abundance} + 1]$ ) prey abundance measures for the analysis. Prey abundance was significantly greater in spring samples compared with autumn samples (MS = 0.340,  $F_{1,4} = 7.980$ ,  $p = 0.048$ ; Fig. 2), primarily because of the high abundance of the non-native weevil, *Barypeithes pellucidus* (Coleoptera: Curculionidae) (Maerz et al. 2005c). Weevils were absent in autumn samples, and available preys were dominated by ants (Hymenoptera: Formicidae) and rove beetle (Coleoptera: Staphylinidae) species. Within both seasons, prey abundance declined significantly with decreasing leaf litter volume (leaf litter volume: MS = 1.047,

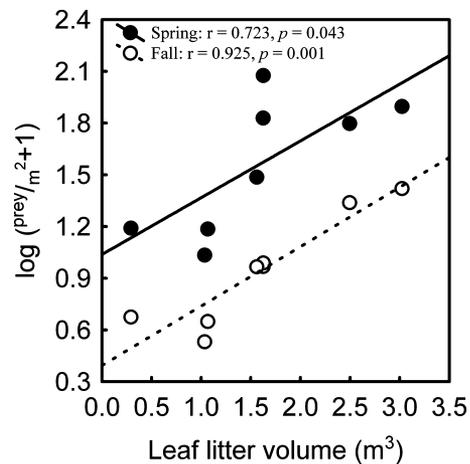


Figure 2. Correlations between the mean *P. cinereus* prey abundance, excluding non-native earthworms, and leaf litter volume among forest habitats in 4 central New York sites (black, spring samples; white, autumn samples).

$F_{1,4} = 9.766$ ,  $p = 0.035$ ; leaf litter volume  $\times$  season interaction term: MS = 0.003,  $F_{1,4} = 0.068$ ,  $p = 0.807$ ; Fig. 2).

## Discussion

The difficulty in determining whether invasion by a particular non-native species is affecting native species is that invasions are often correlated with other factors that may be the primary or a contributing cause of decline (Gurevitch & Padilla 2004). In some cases invasive species may increase the effects of other factors (e.g., habitat fragmentation) on wildlife (e.g., Schmidt & Whelan 1999; Suarez & Case 2002; Borgmann & Rodewald 2004). Our research and the research of others suggests that any apparent effects of non-native plant invasions on native plant communities (Bohlen et al. 2004b; Nuzzo et al. 2008) and *P. cinereus* abundance in north temperate North American forests are largely the result of invasions by non-native earthworms. Our results suggest that earthworm impacts on leaf litter have a strong negative effect on *P. cinereus* abundance, and that the apparent association between non-native plant invasions and *P. cinereus* declines is an artifact of the high likelihood that sites with high earthworm biomass are invaded by non-native plants. That is, plant invasions appear to be symptomatic rather than a cause of degraded forest habitats for salamanders.

Although earthworms were associated with leaf litter loss and declining *P. cinereus* abundance in both focal plant regions, there were consistent regional differences in these relationships that suggest that the magnitude of earthworm impacts can vary geographically. For example, the same biomass of non-native earthworms was associated with less leaf litter among the sites in

central New York compared with northeastern Pennsylvania sites, and *P. cinereus* abundance was greater for a given amount of leaf litter among the central New York sites. The higher amounts of leaf litter in northeastern Pennsylvania are not the result of greater leaf inputs. These sites are more stunted, with greater canopy gaps and smaller crowns. It is more likely that the differences in canopy composition and moisture account for these regional differences. The northeastern Pennsylvania sites were more xeric, had greater light penetration, and were dominated by oak species, whereas the central New York sites were more mesic and were dominated by highly dense canopies of sugar maple. Compared with maple litter, oak litter is less susceptible to breakdown by earthworms (Suárez et al. 2006). In addition, drier soil conditions among our northeastern Pennsylvania sites are likely to limit earthworm activity, further slowing the breakdown of leaf litter. Moisture conditions would also explain why salamander abundance is greater among the central New York sites despite less leaf litter. Cooler temperatures and increased moisture enable salamanders to forage more frequently (Feder 1983), remain active during the warmer summer months, and potentially be slightly less dependent on leaf litter to remain moist and cool.

Although invasive species are considered an important contributing factor in amphibian population declines, empirical evidence of such impacts is relatively limited (Kiesecker 2003). Introductions of predatory fishes or other amphibians that may compete with or prey upon native species are the primary examples of invader impacts on amphibians. There are no studies demonstrating bottom-up impacts of invasive species on amphibian abundance. Recent evidence suggests that some plant invasions may negatively affect larval amphibian performance (Maerz et al. 2005b; Brown et al. 2006; Leonard 2008) and terrestrial foraging success of frogs (Maerz et al. 2005a) but none of these studies has linked invaders to population declines. We suggest that the loss of leaf litter due to earthworm invasion negatively impacts *P. cinereus*, in part, because of the reduced abundance of small arthropods such as ants, beetles, and Collembola. Although we did not quantify Collembola abundance in this study, the results of many other studies show that these also decline with litter loss due to earthworm invasion (Migge-Kleian et al. 2006). Small arthropods are important stable resources for adult and juvenile *P. cinereus* (Maerz et al. 2005c). Earthworms are potentially important prey for adult *P. cinereus* and so may compensate somewhat for the loss of other taxa in salamander diets; however, *P. cinereus* predation on earthworms is generally restricted to larger adults and foraging on cool, rainy nights (Maerz et al. 2005c).

Earthworm impacts on *P. cinereus* may also be linked to effects of leaf litter loss on forest floor microclimates. Plethodontid salamanders are lung less and, like many

other organisms, depend on leaf litter as a moist substrate to buffer them from extreme temperatures and dehydration (Feder 1983). The loss of litter may promote water evaporation from forest soils, placing additional hydric constraints on salamanders.

Our results contribute to a growing body of research demonstrating the potentially large impacts of earthworm invasions on communities and ecosystem function in northern temperate forests of North America (reviewed by Bohlen et al. 2004b). By rapidly reducing forest leaf litter and mixing historically well-layered forest soils, earthworms affect nutrient cycling, microbial abundance and activity, and, consequently, soil invertebrate community structure. Earthworm invasions alter plant communities, causing declines or extinctions in some flora and increasing the relative importance of historically subordinate native and non-native plant species. Finally, earthworm invasions may alter resource dynamics (Maerz et al. 2005c) and be a source of soil pollution that affects vertebrate populations (Ireland 1977; Reinecke et al. 2000), both of which may contribute to population declines.

With the realization that earthworm invasions are major drivers of change in temperate forests comes the question of whether there are management options to control earthworms and their effects. The spread of earthworms to temperate forests that remain earthworm-free should be a high management priority. Because earthworm introductions are a legacy of agricultural landscapes, earthworm-free areas are likely to be more remote landscapes that are not utilized for agriculture. Earthworms are, however, also spread through their use as bait and increasingly as contaminants of ornamental plants (Cameron et al. 2008). Intervention to prevent the spread of earthworms through these pathways may be an important management objective (Hendrix & Bohlen 2002; Hale 2008). For areas currently invaded by earthworms, management options are far more limited. At present there are no published effective means of control for earthworms at the large scale. Within invaded landscapes, recognizing that earthworms are major drivers of forest change can better inform other management priorities. Specifically, it is important to realize that investment in the control of major herbivores, such as deer, or non-native plants may not result in the protection or restoration of native forest plants and animals affected by earthworm invasions. Finally, additional non-native earthworm species (e.g., *Amyntbas* spp.) from other parts of the globe such as Asia are arriving and spreading within the United States. These new species reflect new patterns of global trade, and how these new invaders are affecting recipient ecosystems is not well understood. With a growing and compelling body of evidence on the impacts of earthworm invasions, there is a need for greater vigilance about the deliberate and accidental introduction and spread of new earthworm fauna (Hendrix & Bohlen 2002).

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