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Effects of *Hedera helix* L. removal on the understory early regeneration in an oak temperate forest in Mexico City

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Abstract

Background *Hedera helix* L. is an invasive weed species introduced in several regions of the world, so it is relevant to study its effects on the plant community and the changes occurring after its eradication. We established invaded (with/without removal of *H. helix*) and non-invaded (with/without removal of the understory) plots to compare some community attributes in an oak forest in Mexico City and monitored them every 2 months for 1 year, taking soil samples to compare some edaphic variables between the plots. If invasive species negatively affect natural regeneration, seedlings of native species would be less abundant in plots invaded by *H. helix* than in the non-invaded ones. As invasive species modify the soil microbiota through exudates and often allelochemicals, soil chemical variables would differ between invaded and non-invaded plots.

Results The abundance of native species was lower in the invaded plots, but the richness and diversity of species were not statistically different. The composition of the two invaded plots was similar to each other and more alike to that of the non-invaded plots. The species composition of the invaded plots was associated with soil potassium and phosphorus concentrations, whereas that of the non-invaded plots was more associated with soil nitrogen concentration.

Conclusions Following the removal of *H. helix*, several native species were registered, leading to a similar species composition as in the non-invaded plots, suggesting that some vegetation recovery can be expected following *H. helix* removal in 1 year.

Keywords Ivy, Invasive plant, Mechanical control, Diversity, Species composition

Background

Invasive species are considered a global problem as one of the main threats to native species, together with habitat loss and the illegal extraction of species (Wilcove *et al.* 1998). Those species can modify the diversity of

communities (Woods 1997) by eliminating native species through competition for space, nutrients, light, pollinators, and dispersal agents (Daehler 2003; Morales and Traveset 2009; Thiébaud *et al.* 2019). Invasive plants may also substantially alter the composition and diversity of soil propagule banks (Vilà and Gimeno 2007; Gioria and Osborne 2009; Alharthi *et al.* 2021), can increase the susceptibility of a community to other disturbances such as fire and soil erosion, which at the same time can promote other species invasions (Simberloff and Von Holle 1999; Ma *et al.* 2020). Moreover, invasive plants can affect the soil chemistry through exudates or litter, altering the soil microbiome and nutrient cycling differently from native

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species (Allison and Vitousek 2004; Meiners et al. 2017; Thiébaud et al. 2019; Wang et al. 2016; Zhao et al. 2019).

Most research on biological invasions deals with biogeography and demography of invasive species, which usually derives from management plans to reduce or eliminate their populations (Woods 1997). In some cases, the removal of the invasive species may affect the plant community by increasing resource availability, allowing other invasive species or native weeds to colonize (Bergstrom et al. 2009), or by increasing herbivory on native species (D'Antonio and Vitousek 1992). Studies of the effects of invasive species removal are rare (Bergstrom et al. 2009), and it is therefore important to monitor the successional response of the community following the control or removal of invasive species (Woods 1997).

Hedera helix L. (English ivy) is a perennial weed native to Europe, Asia, and North Africa (Metcalf 2005); it has spread worldwide as an ornamental plant, escaping cultivation and causing diverse impacts on plant communities. For example, the understory of deciduous European temperate forests has been invaded increasingly by lianas, primarily English ivy (Perring et al. 2020). The invasion of lianas affects the species composition of the understory in part due to allelopathic effects (Ladwig et al. 2012; Madrigal-González et al. 2017). English ivy shows different growth forms: climber, liana, erect herb, and subshrub; it establishes in shady areas or in open and disturbed sites (Metcalf 2005). It reproduces vegetatively by resprouting from stem fragments and growing fast through adventitious roots and stolons (Waggy 2010). Besides, its seeds show a fast germination rate within the first 14 days after planting, reaching high germination percentages when the fruit pulp is cleaned naturally or artificially (Metcalf 2005). Also, some authors suggest that *H. helix* inhibits the germination and resprouting of some native species through allelopathy as that species is rich in phenolic and triterpenoid compounds (Biggerstaff and Beck 2007a; Marian et al. 2017; Shawky and El Sohafy 2020). Therefore, studies to predict or explain invasion by some plant species are relevant to measure propagules sources such as seed banks (Standish et al. 2001).

The *Quercus rugosa* and *Quercus laurina* temperate forest of the Magdalena River Basin (MRB) in Mexico City is subject to frequent anthropogenic disturbances such as urbanization and livestock raising (Santibáñez-Andrade et al. 2015). In this forest (Castillo-Argüero personal communication 2016), the abundance of *H. helix* has recently increased in areas where this species was not present a decade ago. It is possible that the increase in abundance of *H. helix* in the MRB forest is following the pattern of the increase in frequency of English ivy in temperate forests in Europe (Perring et al. 2020).

With the aim to assess the effect of *H. helix* on the plant community early regeneration, it is necessary to quantify the emergence of seedlings from the propagule bank in the soil. Thus, we asked: (1) Does the presence of *H. helix* hinder the germination or sprouting of other species? (2) Will the understory species composition differ between plots with *H. helix* and plots without it in a year time lapse? and (3) Due to the effects on soil chemistry of some invasive species, will some soil chemical variables differ between invaded plots with *H. helix* and non-invaded plots?

Methods

Study site

Our research site was an oak forest in the Magdalena River Basin (MRB), southwest of the Valley of Mexico (Fig. 1). This oak forest has two dominant oak tree species (*Q. rugosa* and *Q. laurina*), other tree species such as *Garrya laurifolia*, and a shrubby-herbaceous understory dominated by *Verbesina oncophora*, *Salvia microphylla*, among others. This forest is in the lowest altitudinal interval of this basin, 2700–3200 m (Santibáñez-Andrade et al. 2015). This area is constantly affected by urbanization, deforestation, and cattle grazing (Bonilla-Valencia et al. 2021). The climate is temperate sub-humid (García 2004). The dry season occurs from November to May, and the rainy one from June to October (Álvarez 2000), with a mean annual precipitation between 800 and 1000 mm (Jujnovsky 2006).

Field sampling

We selected our study area since it has the highest density of *H. helix* within the forest. The vegetation is dominated by *Quercus rugosa*–*Quercus laurina* and *Garrya laurifolia*, with disturbed sites with an open canopy, between 2700 and 3100 m asl in an area of 102.51 ha on a north-west facing slope.

Then we determined two strata with different species composition; the first one was dominated by *H. helix* (19°17'00.3''N, 99°16'13.7''W), and the second one, where it was absent (19°16'59.3''N, 99°16'14.3''W). The area of each stratum was 500 m², and the distance between them was approximately 50 m. We randomly established ten plots (3 × 3 m) in each stratum. In total we had 20 plots (3 × 3 m, 180 m² total). Within each stratum, five plots were selected at random to clear the understory. We set four treatments: five plots invaded by *H. helix* (IH); five control plots without *H. helix* (CV); five plots where *H. helix* was manually removed (IR); and five control plots without *H. helix*, in which we manually removed the understory (CR). To detect changes in species composition and diversity in each sampling unit, we counted all seedlings (all species)

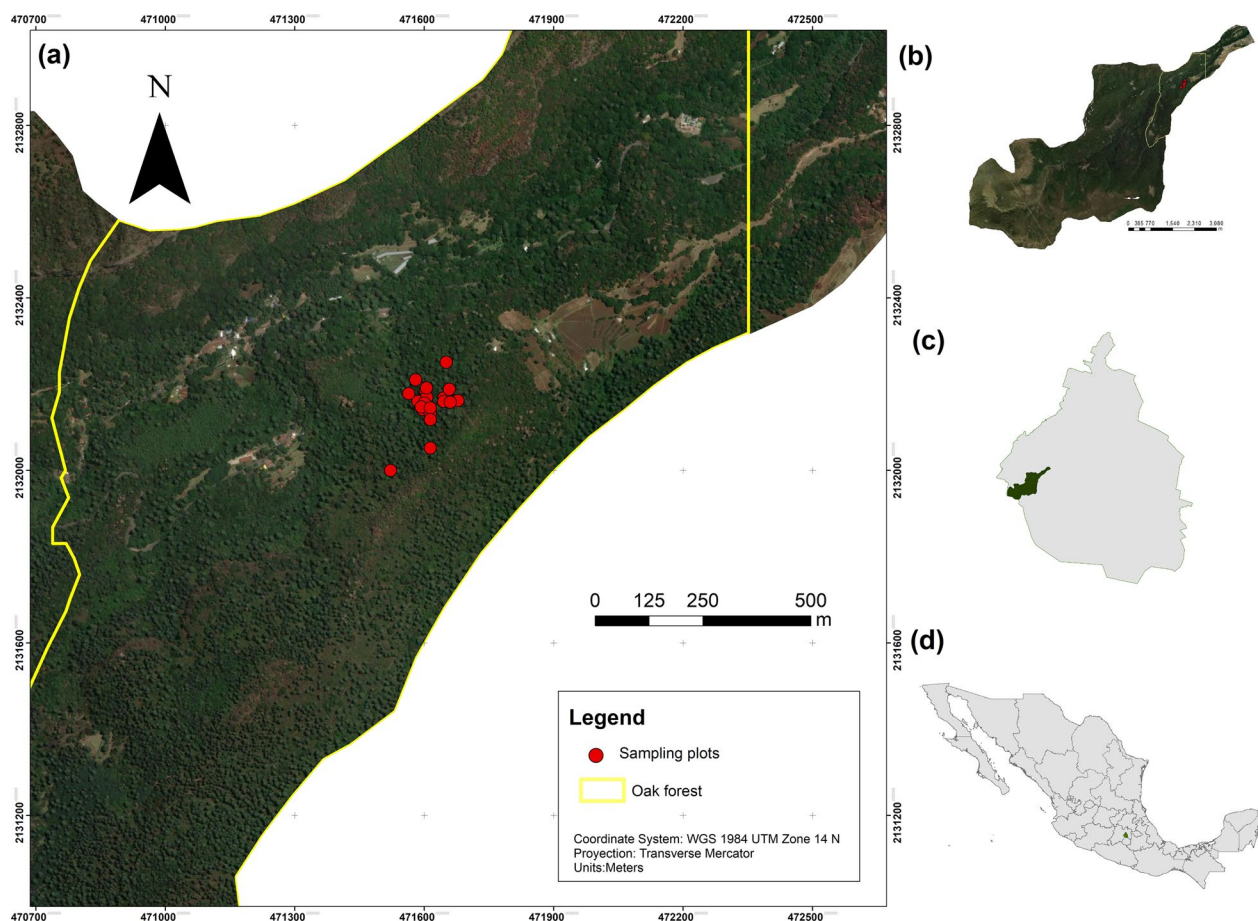


Fig. 1 Geographical location of the experimental plots (red circles) in the oak forest (a) of the Magdalena River Basin (b), Mexico City (c), Mexico (d). Map was produced with ArcGIS 10.3; Background map credits: ESRI, INEGI and Santibañez-Andrade (2015)

(<1 m) every 2 months for 1 year from August 2016 to September 2017; species identification followed the Flora of the Valley of Mexico (Calderón de Rzedowski and Rzedowski 2005). Species identity was confirmed by direct comparison with herbarium specimens deposited in the Herbaria of MEXU and FCME. 37% of species were not identified. We monitored the area during the study year to avoid other disturbances.

We measured light incidence by calculating the global site factor (GSF) using hemispherical photographs and analyzing them with the Hemiview ver. 2.1 (Delta-T 1998) software. We took one photo per plot, placing the camera towards the magnetic north, using a NIKON camera D80 (Nikon, New York, USA) with a fisheye lens EX SIGMA (Sigma, New York, USA) 4.5, 1.28 DC DSM. We collected one soil sample per plot (200 g) to analyze the concentration of available nitrogen (N), potassium (K), phosphorus (P), organic matter (OM), pH, and electric conductivity (EC); we also assessed soil moisture content by the gravimetric method (Reynolds 1970). Light and soil variables were recorded twice, at the beginning of the study in

August 2016 and 6 months later in March 2017 (wet and dry seasons, respectively).

Data analysis

To know if there were differences between treatments and sampling months, we performed a repeated measures ANOVA using the number of seedlings (abundance) and species richness data for each plot with the R software (R Development Core Team 2018). We calculated the species diversity of seedlings through the Shannon–Wiener (H') index (Magurran 1988) and the Hutcheson t -test (Hutcheson 1970) to compare between treatments.

We constructed a matrix for a two-dimensional solution with 999 random iterations based on species abundance and the Bray–Curtis similarity index. We used a similarity analysis (ANOSIM) to estimate the species composition changes among the four treatments, considering bi-monthly records in each plot ($n=24$), with the statistical package vegan (Oksanen et al. 2013) of R software (R Development Core Team 2018). The ANOSIM calculated the R statistic from the Bray–Curtis

dissimilarity matrix. For this, we calculated the statistic R based on the difference in mean ranks among and within groups. The R-value varies between zero and one, where zero indicates completely random or no differences among treatments, and values close to one indicate dissimilarity among treatments (Oksanen et al. 2013). We evaluated composition changes graphically by considering species similarity between treatments using a non-metric scaling ordination analysis (nMDS) ($n=24$). We used the square root transformation on the abundance values to avoid biases of the highly abundant species.

We evaluated the effect of the treatment, the time (sampling months), and the interaction between treatment and time on the composition changes through a permutational multivariate analysis of variance (PERMANOVA) (Anderson 2001) with distances of Bray–Curtis obtained through 999 random iterations. We performed a multivariate homogeneity test of the distribution of groups based on permutations to determine whether the trajectories of each treatment were constant over the months. We calculated the average distance to the centroid of each graph in multivariate space (generated by a Bray–Curtis distance matrix) by using the betadisper function in the vegan package of the statistical program R. Through Permutest, we tested the null hypothesis that there was no difference in the distribution between treatments (999 permutations) (Anderson 2006). Also, to evaluate the dispersion pattern of the samples within treatments, we performed an analysis of multivariate homogeneity of group dispersions (composition dispersion) (Anderson 2006). We conducted a canonical correspondence analysis (CCA) to examine the relation between light, soil variables, and species composition for each plot, based on the matrix of species abundance transformed with $\log(x+1)$ with the statistical package vegan (Oksanen et al. 2013) of R software (R Development Core Team 2018). The CCA analysis was carried out through 999 permutations to determine the variables with the most significant ($P<0.05$) influence on species composition (R Development Core Team 2018).

Results

We found 47 native species in 22 families: the best represented were the Asteraceae (five species), the Solanaceae, and the Rosaceae (three species each).

The repeated measures ANOVA showed significant differences for the abundance of each species between treatments ($F(3,16)=6.92$, $P<0.05$) and months ($F(5,80)=9.35$, $P<0.05$). The interaction of both variables also showed significant differences ($F(15,80)=2.28$, $P<0.05$).

The plots invaded by *H. helix* (IH) exhibited the lowest abundance values, whereas the non-invaded control

plots with vegetation (CV) showed the highest abundance values in almost all the months (except January and July) (Fig. 2). For the non-invaded control plots (CV), invaded removal plots (IR), and control removal plots (CR), we observed an increase in the abundance in November, reaching a maximum value in January; later, a decrease in March and May (the end of the dry season), and finally a new increase in July and September (the beginning of the rainy season). A similar trend occurred between months for the IH treatment. For species richness, there were no significant differences between treatments ($F(3,16)=3.33$, $P\geq 0.05$), months ($F(5,80)=1.99$, $P\geq 0.05$), and neither in the interaction of both variables ($F(15,80)=1.04$, $P\geq 0.05$).

The Shannon–Wiener diversity index was higher in the invaded plots (IH) than in the other treatments in almost all the months (Table 1). In the first 3 months, both removal treatments (IR and CR) showed low diversity values that increased throughout the year; therefore, for the last month there were no significant differences with the control plots (CV).

The ANOSIM test showed a composition dissimilarity between treatments ($R_{\text{global}}=0.893$, $P=0.001$). The stress value (0.149) showed a good fit for our data. In the nMDS, we can observe that the first group corresponds to the invaded plots (IH) represented by the species *H. helix*, *Cirsium* sp., *Verbesina oncophora* B.L. Rob. and Seaton, and *Salvia microphylla* Kunth (Fig. 3). The second group corresponds to the non-invaded control plots with vegetation (CV) where the representative species were *Iresine diffusa* Humb. and Bonpl. ex Willd., *Salvia mexicana* L., *Cupressus lusitanica* Mill., and *Philadelphus mexicanus* Schltdl. (Fig. 3). The third group joined both removal treatments (IR and CR) represented by species such as *Prunus serotina* Ehrh., *Didymaea floribunda* Rzed., *Trisetum virletii* E.Fourn. ex Hemsl, and *Solanum pubigerum* Dunal. Also, the control plots with vegetation (CV) were closer to the removal plots (IR and CR) than to the invaded ones (Fig. 3).

The PERMANOVA analysis showed a significant effect of treatment on species composition changes but no significant effect of the interaction (treatment \times month) (Table 2). The Permutest did not show significant differences between treatments ($F=1.71$; $P>0.05$), which indicates that the distances to the centroids in each treatment are similar. For the four treatments, the distances to the centroid are small. In each treatment, the rates of change were low between the months (mean distance to the median IH=0.20, CV=0.18, IR=0.27, CR=0.25) (Fig. 4).

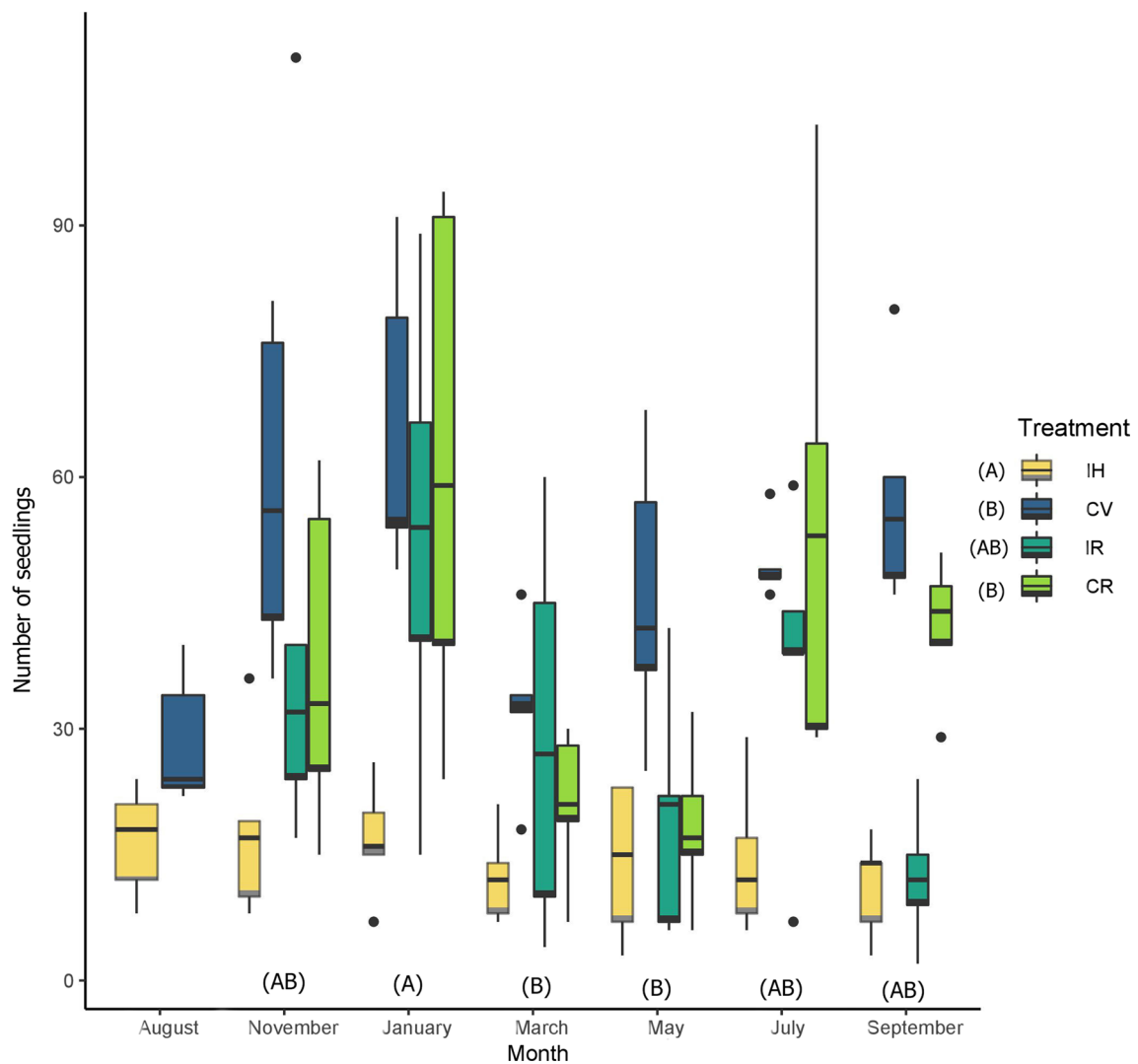


Fig. 2 Boxplot of the number of seedlings in each treatment every 2 months: plots invaded by *H. helix* (IH); control plots without *H. helix* (CV); invaded plots where *H. helix* was removed (IR); and control plots without *H. helix* where we manually removed the understory vegetation (CR). Letters represent statistically different groups

Table 1 Shannon–Wiener diversity index values for each treatment and sampling month

Treatments	August	November	January	March	May	July	September
IH	2.26 ^a	2.33 ^a	2.67 ^a	2.97 ^a	2.69 ^a	2.58 ^a	2.66 ^a
CV	1.83 ^b	2.14 ^a	2.34 ^b	2.05 ^b	2.07 ^b	2.11 ^b	2.18 ^b
IR		1.05 ^b	0.87 ^c	1.38 ^c	1.95 ^b	1.64 ^c	2.42 ^{ab}
CR		1.83 ^c	1.63 ^d	2.44 ^d	2.35 ^c	1.47 ^c	2.25 ^b

Letters represent significantly different groups according to Hutcheson t-test. Plots invaded by *H. helix* (IH); control plots without *H. helix* (CV); invaded plots where *H. helix* was removed (IR); and control plots without *H. helix* where we manually removed the understory (CR)

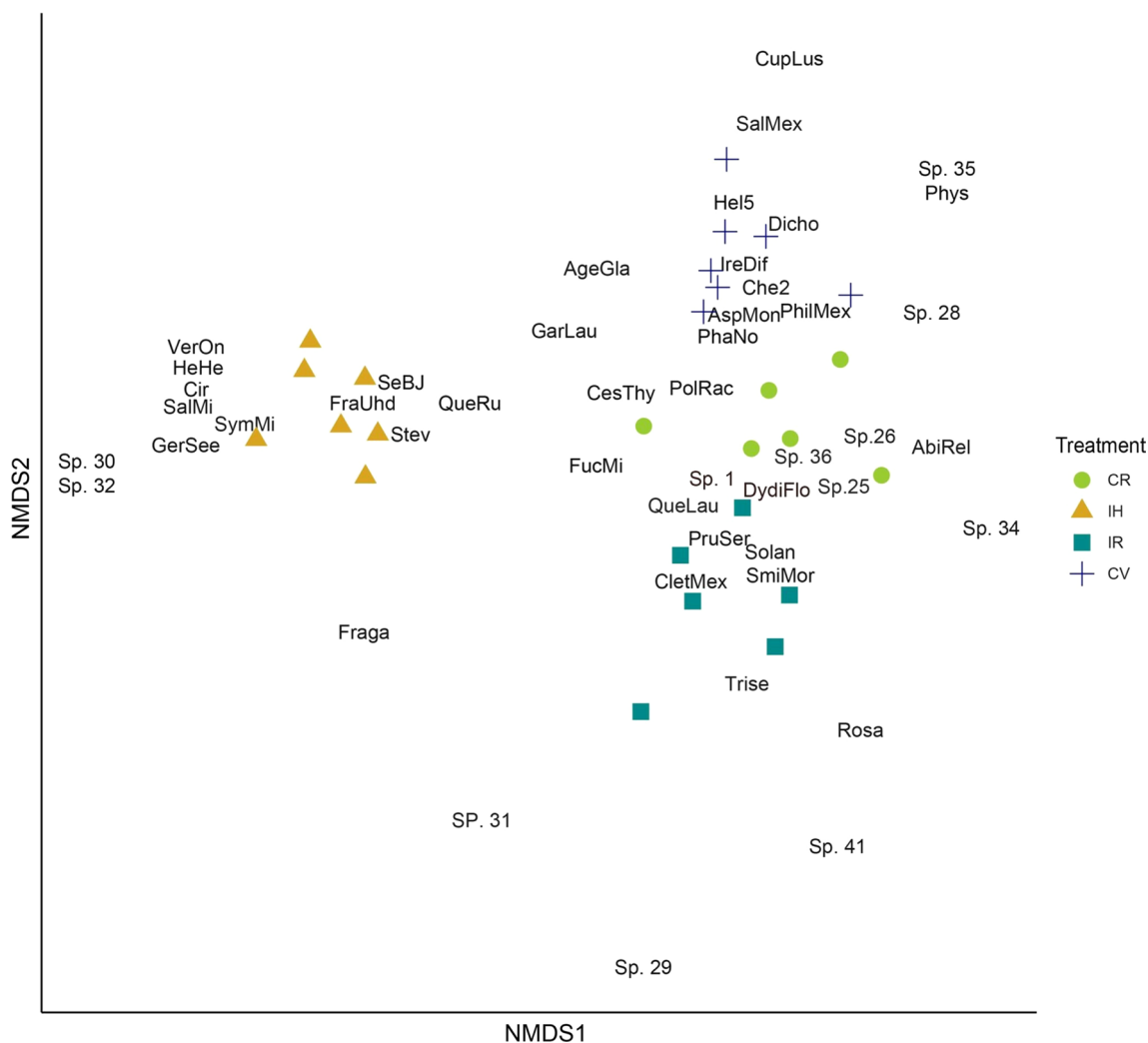


Fig. 3 Non-metric multidimensional scaling analysis (NMDS) of the species in each treatment and sampling month. Stress = 0.1358. Plots invaded by *H. helix* (IH); control plots without *H. helix* (CV); invaded plots where *H. helix* was removed (IR); and control plots without *H. helix* where we manually removed the understory (CR) (see Additional file 1: Appendix)

Table 2 Permutational multivariate analysis of variance (PERMANOVA) with Bray–Curtis similarity indices to evaluated the treatment and month effect

	df	Sum of squares	Pseudo-F	P
Treatment	1	1.45	9.18	<0.001*
Month	1	0.12	0.76	0.50
Treatment × month	1	0.06	0.43	0.79
Residual	22	3.48		
Total	25	5.13		

df degrees of freedom, SS sum of squares, Pseudo-F pseudo-F Fisher, P value P

*Significant effect

The canonical correspondence analysis (CCA) indicated that the invaded plots (IH and IR) had higher concentrations of potassium (K) and phosphorus (F) (axis 1 eigenvalue = 0.17; axis 2 eigenvalue = 0.12). The non-invaded site (CV and CR) had higher concentrations of nitrogen (N) and organic matter (O.M.). There was no direct relationship between treatments and electrical conductivity (E.C.) (Fig. 5).

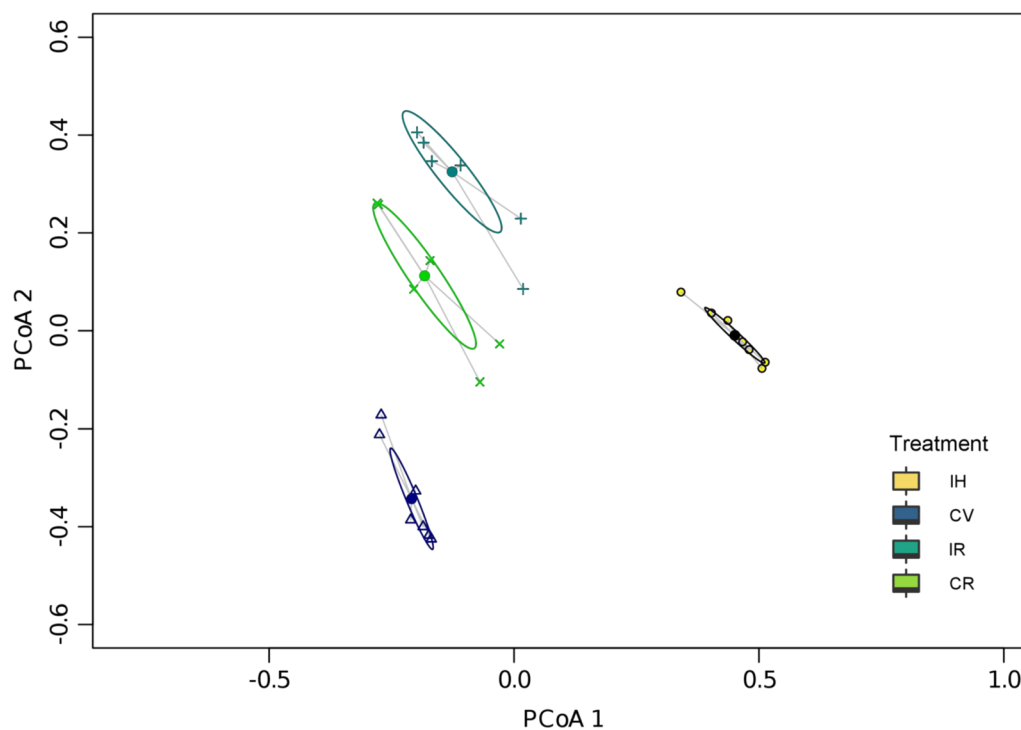


Fig. 4 Principal coordinate analysis (PCoA) plot with Bray–Curtis dissimilarity. Centroid distance IH=0.20, CV=0.18, IR=0.27, CR=0.25

Discussion

Hedera helix is a widely studied and well-known invader that grows in a wide range of ecological niches such as floodplains, urban forests, beech woods, and woodlands within its native and introduced distribution (Cortenbach and Madurapperuma 2018). As it is a physiologically plastic species, its growth form can also depend partially on climate, our study forest is spatially heterogeneous, and the existence of variable environmental conditions between microsites has already been reported (Santibáñez-Andrade et al. 2015). Therefore, we find it climbing tree trunks or forming dense ground covers as Strelau et al. (2018) report. In America it is found in humid areas of semi-open forests (as our study site) and in deeply shaded forests (as forests in west Canada).

Part of the success of *H. helix* in its introduced habitats is that it becomes widely distributed in urban parks and fragmented forests such as our study site, mainly because it reproduces sexually and spreads asexually as well (Okerman 2000). Other factors that may be favoring it are the high content of soil organic matter and anthropogenic disturbances in this site (Solís-Oberg 2015). This study showed that *Hedera helix* affects early forest regeneration because there was lower seedling recruitment in the invaded plots than in the other treatments; similarly, Biggerstaff and Beck (2007b) found lower abundances of

seedlings during 5 months in plots invaded by *H. helix*, suggesting that *H. helix* can inhibit the germination and or resprouting of some species. Also, Thomas (1980) showed that *H. helix* prevents the recruitment of native species in forests of Washington, USA. The inhibitory mechanism can be through allelopathic substances, as *H. helix* produces some secondary metabolites such as triterpene saponins, flavonoids, and phenolic compounds (Uddin et al. 2011; Shawky and El Sohafy 2020) that negatively affect the germination of some species seeds (Marian et al. 2017), which has been reported for other invasive species (Wang et al. 2016).

However, the species richness did not show significant differences between treatments, but the diversity index value was higher, in almost all months, in the invaded plots than in the other treatments. Similarly, Dlugosch (2005) did not find significant differences in richness and diversity between *H. helix*-invaded and non-invaded treatments.

Huston and DeAngelis (1994) suggest that many species can coexist, invasive and native, due to small-scale heterogeneity (the presence of microsites different in characteristics such as slope orientation, soil moisture, and light). As mentioned before, our study site is highly heterogeneous in space (due to its topography) (Santibáñez-Andrade et al. 2015). Therefore, invasion or coexistence depends on resource availability and

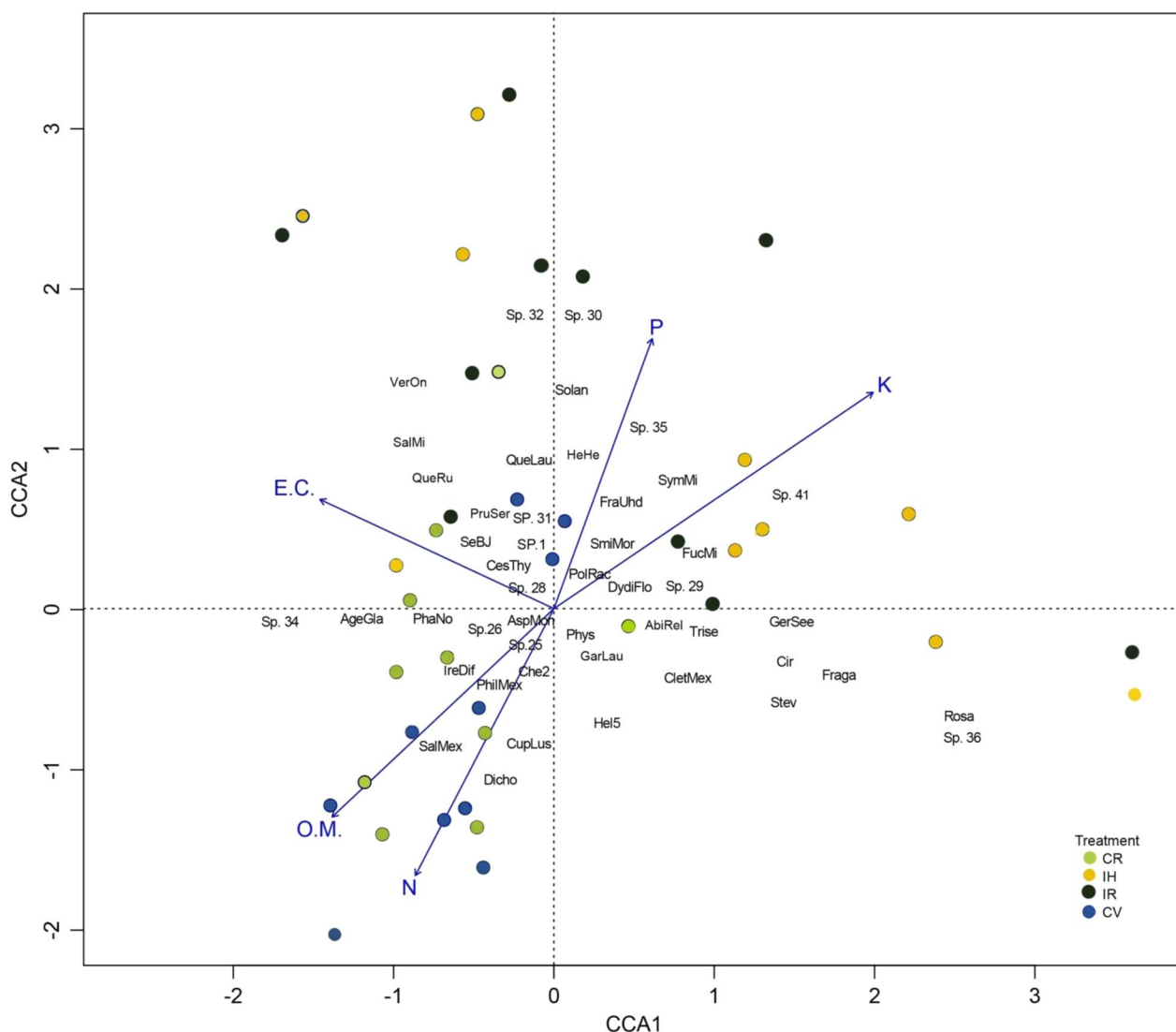


Fig. 5 Canonical correspondence analysis (CCA) for the soil environmental factors and the species found. Plots invaded by *H. helix* (IH); control plots without *H. helix* (CV); invaded plots where *H. helix* was removed (IR); and control plots without *H. helix* where we manually removed the understory (CR). Axis 1 eigenvalue=0.17; axis 2 eigenvalue=0.12. E.C.=soil electrical conductivity, P=soil phosphorus, K=soil potassium, O.M.=soil organic matter, N=soil nitrogen (see Additional file 1: Appendix)

microclimates; however, data on interactions such as predators or pollinators of ivy are missing for this area, as well as evidence of its potential effects on species diversity. Other reports for Mexico indicate that flowers are visited by a great variety of pollinators and that its invasibility can also be related with the fact that it is a vector of some plant species pathogens (SIVICOFF (2017)).

On the other hand, the role of the microbiota on ivy allelochemicals is unknown, but in other systems, microbes can degrade allelochemicals nullifying their effect and allowing the establishment of previously excluded plant species (Meiners et al. 2017). It is also

possible that native plants exposed to allelochemicals for a long time evolve resistance (Callaway et al. 2005).

Hejda et al. (2009) have also shown that the effects of invasive species on regeneration may be more associated with composition changes, as occurred in this study.

Other studies suggest that the impact of invasive species on community richness and diversity could depend on the species size. For example, Hejda et al. (2009) suggested that the difference between the cover and height of the invader and some dominant native species in the site determines the degree to which species diversity is reduced in the invaded community. Thus,

in our study plots, where most of the ivy was growing at the soil level (young individuals), seedlings were shorter than shrubs and trees already growing on the site, which did not seem to be affected by ivy in our experimental plots; this could explain the apparent little effect of *H. helix* on the species richness and diversity, in the short period of our study.

However, it is important to continue management programs because this species is found in our study plots as juvenile and adult, producing fruits (that are constantly dispersed by birds between the forest and the city). Therefore, continuous management efforts are necessary to mitigate the potential long-term impact of this species in the forest.

In addition, in the control vegetation plots (CV), there was a high abundance of *Iresine diffusa*, an annual weed that produces large amounts of seeds with high viability (Acosta et al. 2014). Contrastingly, in the invaded plots (IH), *I. diffusa* showed a lower abundance, probably because it depends on seed germination from the soil seed bank to maintain its population, and the shadow of ivy leaves hindered *I. diffusa* germination.

Other native species also showed high abundances in the plots where vegetation was removed (IR and CR), this was the case of *Prunus serotina*, which has a high seed production and, although its seedlings can remain on the ground for a long time, in this study we observed a high seedling mortality, which led to a monthly variation in its density.

A study in a riparian forest in North Carolina showed that *H. helix* grew associated with other introduced species (Vidra et al. 2006), however, in our study, all the species recorded are native to Mexico. Besides, five native weeds occurred in the invaded plots, a higher species number than in the other treatments. Nevertheless, we should consider that the colonization and spread of exotic species depends on time and it is influenced by anthropogenic disturbances (Vidra et al. 2006) which occur in our study site, mainly affected by livestock raising and urbanization (Bonilla-Valencia et al. 2021).

The differences resulting from the ANOSIM test, as well as the groups obtained from the nMDS, indicate that the invaded plots (IH) have a different composition than the other treatments, where we found established individuals of species like *Verbesina oncophora*, *Symphoricarpos microphyllus* Kunth, *Cirsium* sp. and *Salvia microphylla* Kunth. Meanwhile, the non-invaded plots (CV) had a more similar composition to the group conformed by both removal treatments (IR and CR), mainly due to the presence of secondary vegetation species like *Asplenium monanthes* L., *Phanerophlebia nobilis* (Schltdl. and Cham.) C. Presl and *Prunus serotina*, the native shrub *Philadelphus mexicanus*, and the

native weed *Iresine diffusa*. Several authors suggested that removals are an excellent technique to guide specific areas to a composition similar to that in non-invaded plots (Marchante et al. 2011; Eppinga et al. 2021). However, the close similarity in species composition between the IR and CR treatments suggests that, at small scales where native species propagules are available, removal of *H. helix* does not alter natural regeneration at early successional stages.

A clear separation of the invaded and the non-invaded plots resulted from the canonical correspondence analysis (CCA), which relates to changes in species composition and edaphic variables. Higher concentrations of nitrogen and organic matter seem more related to the non-invaded plots than the invaded plots. In previous research in this forest, a similar value of organic matter was reported (17.6%) (Solis-Oberg 2015). This value is slightly lower than that in the non-invaded plots and higher than in invaded plots.

In temperate forests, the constant decomposition of organic matter also increases nitrogen concentration (Fassnacht and Gowerr 1999; Hasegawa et al. 2021). Our results showed a relation between *H. helix* and plots with low nitrogen concentration; however, our short-term study does not clarify if the low nitrogen concentrations promote the invasion of *H. helix* or if this is an invasion consequence, but according to SIVICOFF (2017), the increases in soil nitrogen due to *H. helix* invasion are registered in a period between 5 and 20 years. Probably, our results are related to the nitrogen fixing bacteria in the *H. helix* leaves and stems (Soares et al. 2016), so nitrogen is not a limiting factor for the invasive growth; also, there is a constant source of nitrogen from atmospheric deposition from the city. In other research in this forest, soil nitrogen concentrations were slightly lower (0.4%) (Solis-Oberg 2015) than those found in our study in both invaded and non-invaded plots.

Dolan (2013) observed that the increase in soil nitrogen did not stimulate the dominance of *H. helix*, and conversely, the native species showed a positive response to nitrogen increase in a temperate forest in Portland, USA. In addition, some studies have shown that some varieties of *H. helix* cultivated in subirrigation systems decrease their growth rate as the nitrogen concentration rises (Pennisi et al. 2005).

Potassium (K) and phosphorus (P) concentrations in the invaded plots soil were higher than in the non-invaded ones; this could be because the leaves of *H. helix* release potassium faster when they start to decompose, generating high accumulation of this element in the soil (Badre et al. 1998; Bonanoni et al. 2010). The introduction of invasive plants can drastically change the cycling of some nutrients in a system by modifying the quality

and quantity of the litter and changing the mineralization rates and their availability (Tateno et al. 2007; Follstad et al. 2010); also, they affect the soil microbiota through exudates and allelochemicals (Meiners et al. 2017; Zhao et al. 2019). Most studies report a higher reserve and availability of nutrients, such as soil P and K, under invasive plant species than in non-invaded ecosystems (Vanderhoeven et al. 2005). However, Solis-Oberg (2015) reported a similar K concentration, but the P concentrations in our invaded plots were threefold higher.

Other efforts to control English ivy through plant removal suggest that this method will begin to yield results soon. For example, Stanley and Taylor (2015) found similar results in terms of diversity and abundance within the four weeks after its removal; they also observed a recuperation of native plants 24 weeks after the applied treatments. However, they recommend the removal control during the growing season; thus, we suggest a frequent (at least once a year) removal of *H. helix* during the rainy season in our study site.

We prevented invasion or reinvasion by ivy in our experimental plots by removing all the ivy seedlings, runners, or stems every sampling month; thus, no new ivy individuals remained there during the study. Cutway (2017) tested the effectiveness of constant manual removal for 8 years of several invasive species, including *H. helix*, which only appeared in the 8th year in low densities. Therefore, given the current level of *H. helix* invasion in the Magdalena River Basin oak forest, the manual eradication of English ivy may be a feasible control method without negatively affecting soil fertility. However, this method may disturb the soil by promoting erosion or compaction (Biggerstaff and Beck 2007b), which may facilitate reinvasion by *H. helix* or the establishment of other invasive plants.

We acknowledge that our study was conducted over a short period of time. Such studies should be conducted over longer periods of time (more than 5 years) according to Kettenring and Adams (2011) so they can provide valuable ecological information. It may also be limiting that we do not know how long *H. helix* has been at this site and therefore do not have enough information on the edaphic conditions prior to its establishment. Nevertheless, our results indicate that the colonization of *H. helix* and its removal did not prevent the emergence of native species.

Conclusions

Despite our short-term study, we observed that mechanical control (removal) of *H. helix* allows the recovery of early successional vegetation because several native species resprouted or germinated. The dynamics of the forest (changes in richness and diversity), and the changes

in its soil, should be monitored to prevent the possibility of *H. helix* reestablishing itself since there is continuous dispersal of propagules by birds from the city.

Abbreviations

MRB	Magdalena River Basin
IH	Plots invaded by <i>H. helix</i>
CV	Non-invaded control plots with vegetation
IR	Invaded removal plots
CR	Control removal plots
N	Nitrogen
K	Potassium
P	Phosphorus
OM	Organic matter
EC	Electric conductivity
<i>H'</i>	Shannon–Wiener index
ANOVA	Analysis of variance
ANOSIM	Similarity analysis
PERMANOVA	Permutational multivariate analysis of variance
MEXU	National Herbarium Universidad Nacional Autónoma de México
FCME	Herbarium of the Faculty of Sciences Universidad Nacional Autónoma de México

Supplementary Information

The online version contains supplementary material available at <https://doi.org/10.1186/s13717-023-00443-y>.

Additional file 1: Appendix. Species registered in plots invaded by *H. helix*; control plots without *H. helix*; invaded plots where *H. helix* was removed; and control plots without *H. helix* where we manually removed the understory in the Magdalena river basin, Mexico City.

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Author contributions

HJA-H: conceptualization, investigation, formal analysis, writing original draft. LB-V: formal analysis, writing original draft. YM-O: investigation, writing original draft and editing. MJZ-A: investigation, writing original draft. FJE-G: supervision, conceptualization, writing original draft and editing. SC-A: supervision, conceptualization, investigation, formal analysis, writing original draft.

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