

The Impact of *Lantana camara* on Invertebrates and Plant Species of the Groenkloof Nature Reserve, South Africa

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Invasive plant species have negative ecological impacts such as displacing indigenous plants and invertebrates. These invasive plant species affect biodiversity by impacting indigenous vegetation and the food webs associated with this vegetation. We assessed how *Lantana camara* affects indigenous plant species richness and invertebrates and their feeding guilds in riparian habitats inside the Groenkloof Nature Reserve in South Africa. We showed: (1) A lower abundance and morphospecies richness of invertebrates as well as lower numbers of plant species in *lantana*-invaded habitat as compared to indigenous bush and grass-dominated habitats. (2) A Negative association between plant species richness and *L. camara* above ground mass and shoot density, but no association was found between plant species richness and the size of *L. camara* invaded areas. This finding suggests a link between the reduction in overall invertebrate abundance and morphospecies richness and the replacement of native plant species by invasive *lantana*. (3). This increased biomass in natural vegetation was even more evident for detritivores as compared to other feeding guilds. Extensive invasion by *L. camara* is affecting the quality of riparian ecosystems especially for invertebrates that rely on decaying plants and animals as food and this will affect overall biodiversity.

Key words: Bushland, Grassland, Riparian vegetation, Invasion, Feeding guilds.

BACKGROUND

Invasive plants can cause extensive impact on indigenous vegetation and their food webs as these plants tend to occupy large spaces (David et al. 2017). Also, several studies have shown that invasive plants negatively impact indigenous vegetation composition, diversity, richness and abundance (Dresseno et al. 2018). However, very little literature investigates the simultaneous impact of invasive plants on indigenous plants species richness, invertebrate abundances and their feeding guilds in riparian habitats, especially in South Africa. Such studies are needed in order

to understand how invasive plants alter processes and patterns of ecosystems that depend mostly on invertebrates and indigenous plants to thrive (Ramey and Richardson 2017). There is a suggestion that, in general, invasive plants foster smaller herbivore assemblages than indigenous plants (Foster et al. 2021; Štrobl et al. 2019; Stone et al. 2018), but research on the impacts of these invasive weeds on the biomass and species richness of different invertebrate feeding guilds at the habitat level are scarce (Barnes et al. 2017), most studies concentrate on vertebrates and how their food and shelter are affected by invasive plants species (Wei et al. 2020; Whisson et al. 2020).

Harris et al. (2004) reported abundance and species richness in one taxon associated with an invasive shrub to be higher than with the indigenous Kanuka tree (*Kunzea ericoides*) in New Zealand. Interestingly, however, the same Kanuka tree species supported higher numbers of invertebrates (Strong et al. 1984).

Herbivorous invertebrates, at the secondary producer trophic level, are accountable for most of the total flow of energy (Ullah et al. 2018). Therefore, reduction of herbivorous invertebrate biomass in areas disturbed by invaders, will also have consequences for mammals, birds, and predatory invertebrates (Pedersen et al. 2018). Assessing the effects of invasive weeds on the ecosystem could bring in new information that could help with the management, conservation, and protection of protected areas' ecosystems and indigenous biodiversity, in particularly invertebrates that are often neglected in the invasion literature (Gallardo et al. 2019).

The rapid spread of invasive species in South Africa poses a serious threat to biodiversity, with 29% of the Gauteng Province surface area affected by introduced exotic plant species (Versfeld et al. 1998). For example, exotic riparian plants alter the freshwater invertebrate communities by altering channel shape and reducing water flow through increased shading (Quinn et al. 1992). Versfeld et al. (1998) estimates that areas of South Africa riparian zones invaded by plant weeds have doubled from 1998 to 2018 because of the susceptibility of riparian areas to such invasions. In particular, the exotic invasive weed, *Lantana camara*, is reported to be problematic in the riparian zones of South Africa (Versfeld et al. 1998). However, we have found no study that has investigated the effects of these invaded riparian zones on invertebrates and indigenous vegetations richness in South Africa.

Despite all the negatives surrounding invasive plants on ecosystems, there is a lack of literature on this issue in South Africa. Studies have shown that exotic invasive fish can change the composition, abundance, or behaviour of grazing invertebrates by exerting a top-down impact on a river ecosystem (Power 1992; McIntosh and Townsend 1995 1996), but no study has concentrated specifically on invasive plant species and how they can affect invertebrate assemblages in riparian habitats especially in South Africa. Therefore, in this study, we set out to (1) determine differences in indigenous plant species richness across areas invaded with *L. camara* and indigenous riparian vegetation threatened by *L. camara* invasion, (2) identify characteristics of *L. camara*-invaded plots that are correlated with plant species richness, and also with invertebrate abundance, biomass and richness, (3) compare the abundance, biomass and morphospecies

richness of invertebrates for different foraging guilds in riparian zones supporting indigenous vegetation with those in habitats invaded by *L. camara*.

MATERIALS AND METHODS

Study area

This study was conducted in riparian habitats inside the Groenkloof Nature Reserve in central Pretoria, South Africa. Twelve sites were sampled between 2019 and 2020 and all *L. camara* found in identified sites were sampled irrespective of their colour variety, as all types of *L. camara* are invasive in the riparian zones of South Africa (Spies 1984).

To assess how *L. camara* affected invertebrate assemblages and vegetation composition, we conducted a vegetation survey along both sides of the Apies River, found in the reserve. We distinguished three different vegetation types in the study site as follows: (1) areas invaded by *L. camara*, (2) areas with grass-dominated vegetation and (3) bush-dominated vegetation. For the analyses, we constructed six 2 × 2 m plots chosen randomly inside the three different vegetation types. The plots were marked by blue plastic stakes driven to a depth of 30cm at opposite corners of each quadrat, a protocol adapted from Woods and Schiel (1997). These plots will be referred to as *L. camara*, bush-dominated and grass-dominated plots. Between 1 and 5 October 2019, we measured the surface areas invaded by *L. camara* using a Global Positioning System (GPS). The size of the *L. camara*-invaded areas ranged from 40–400 m², the smallest one was 10 m × 4 m, and this range represents the level of infestation found inside the Reserve (GNR report 2004). Data were collected inside the 2 × 2 m plots, and the plots were strategically placed away from adjacent collection plots to avoid collection bias.

Vegetation sampling

We recorded plant species in each plot in the Spring (October–November 2019) and the Summer (January–February 2020) to include species with different phenology. The number of plant species was recorded within each 2 × 2 m plot to assess species richness for the understory and bush layers. We estimated the Above Ground Biomass (AGB) of *L. camara* by measuring the shoot-base of all shoots inside each plot using the shoot-base diameters (Verwijst and Telenius 1999; Gerber et al. 2008; Guendehou et al. 2012).

We recorded several indigenous plant species inside the *L. camara* patches: *Ehretia rigida* ($n = 8$),

Azolla filiculoides ($n = 3$), *Certophyllum demersum* ($n = 5$), and *Senna didymobotrya* ($n = 2$). Also, plants common inside the reserve such as *Paraserianthes lophantha* ($n = 2$); *Dipogon lignosus* ($n = 3$) and *Rhynchosia ferulifolia* ($n = 4$) were rarely recorded inside the *L. camara* patches, probably because they could not compete with the *L. camara* for resources.

Invertebrate sampling

To assess invertebrate biodiversity, we used two standardised collecting methods: pitfall traps that are more suited to ground dwelling invertebrates and window traps for flying insects (Gerber et al. 2008; Moretti et al. 2004).

We used 18 window traps (for flying insects; made with 23×20 cm polycarbonate sheets painted with a neutral colour of grey, placed 1.5 m above ground level) and 18 pitfall traps (for ground dwelling invertebrates; made from 9 cm diameter \times 12 cm depth plastic cups) to capture individuals that were moving at different heights, and also target invertebrates moving on leaves or grass that might fall inside the traps on the ground due to winds. This was to compare invertebrate assemblages in *L. camara* invaded vegetation, grassland and bushland vegetation. The traps were half filled with a solution of water, salt and soap (Karlsson et al. 2017) and were checked and emptied once a week throughout the study for four weeks.

To determine the biomass of the captured individuals we first sorted the invertebrates into taxonomic groups and dried them for 16 hours at 60°C (Butler et al. 2018). Invertebrates were pooled within taxonomic groups based on their predominant feeding modes as herbivores, predators and detritivores for analysis purposes and finally recorded as taxa to accommodate the morphospecies. As with most invertebrate trapping studies, in our study the taxonomic groups were used to class the invertebrates after feeding guilds data was recorded (Bersosa et al. 2019; Park and Huh 2018; Saunders et al. 2021).

The study obtained permission to capture invertebrates from the Tshwane Metropolitan Municipality (Permit number: GNR 2019/20).

Data Analysis

The data set was highly skewed (Shapiro-Wilk test) and did not meet the assumption of normality, therefore we used nonparametric statistical tests for the hypothesis testing. We first assessed how vegetation type affects plant species richness using a generalised linear model (GLM) with a Poisson distribution. We further ran a series of Spearman's rank correlation tests

to ascertain the relationships between plant species richness and each of *L. camara* above ground mass, shoot density, and the size of a *L. camara* invaded area. The same tests considered the relationship between *L. camara* biomass and both *L. camara* density and *L. camara*-invaded area size.

Abundance, morphospecies richness and biomass of invertebrates were analysed separately as response variables; vegetation type, feeding guilds (and their interactions) were set as main effect predictor variables. For all models, we included sites as a covariate to account for the potential differences in trapping sites. The most appropriate model was selected based on the plot of the residuals against the fitted values (Crawley 2007).

For all models, significance was determined using Wald (χ^2) statistics and *P*-values were generated using Analysis of variance (ANOVA) (Matuschek et al. 2017). We also reported the estimates, standard error and *Z*-values for the levels of the fixed predictors. We used Fisher's LSD post hoc tests to identify the influence of predictor variables on the dependent variables.

All statistical tests were performed using R Statistical Software (www.r-project.org, R version 4.1.0 (2021-05-18), and the level of statistical significance was set at $P \leq 0.05$. Data visualisation graphs were produced using the GGplot2 package in the R software.

RESULTS

Invertebrates Assemblage

We collected a total of 1,086 individual invertebrates during the sampling period across the study. Assemblages of invertebrate taxa recorded in the pitfall and window traps in the different vegetation types are given in table 1.

From table 1, each of Annelida, Mollusca and Nematoda are more abundant than Diptera. This could be explained by several facts. Annelids, especially earthworms, were previously reported as abundant (Edwards 1997). In a study on Annelids by Nxele et al. (2015), earthworms were reported to be most endemic in South African grassland. Freshwater snails, the only mollusks captured in our study, have been reported to be dispersed by birds' feet and floods between bodies of water (Coughlan et al. 2017). Therefore, their abundance could be attributed to the abundance of birds in our study area (Parker 2014) and the fact that freshwater snails tend to colonize freshwater environments (Strong et al. 1984). Nematodes are reported to be abundant in freshwater ecosystems. In particular, free-living Nematodes were abundant in a similar environmental

setup to ours in the Eastern Cape of South Africa (Marais and Swart 2014). Lastly, similar patterns of some invertebrates abundance were found in several historical studies conducted in South Africa (Kotze and Samways 1999; Nxele et al. 2015). However, these studies focused on specific invertebrates.

The window traps had less diversity of invertebrate taxa particularly in the *L. camara* invaded plots as compared to the two indigenous vegetation plots (Table 1). Bush-dominated vegetation had the highest abundance of individuals totalling 634 followed by grass dominated vegetation with 252 individuals and the *L. camara*-dominated regions with 200 individuals. More individuals were captured in the pitfall traps as than the window traps and Annelida was the most trapped taxa across all traps (Table 1). Insect larvae, Blattodea, and Diplura were not captured in any window traps in the *Lantana* plots. Moreover, Mollusca and Diptera were not found in the pitfall traps of the *Lantana* plots (Table 1).

Across the study detritivores were the most frequent captures (55–70%) and were dominant in the pitfall traps. Predators were the most frequent captures in the window traps (50–65%) but were generally less abundant in the pitfall traps. Herbivores were the least

abundant captures in the pitfall traps (40–69%).

Plant species richness

Species richness varied amongst sites ($\chi^2_5 = 1.84$, $P < 0.001$). In addition, the highest species richness was found in the grass-dominated plots followed by bush-dominated plots even though the difference was marginal. The *L. camara*-invaded plots had the lowest species richness (Fig. 1). Overall, within the *L. camara* plots the Spearman’s rank test revealed a negative relationship between plant species richness and *L. camara* above ground mass ($R_s = -0.740$; $P = 0.020$); between plant species richness and *L. camara* shoot density ($R_s = -0.748$, $P < 0.001$). However, there was no association between plant species richness and the size of the *L. camara* invaded areas ($P > 0.05$). There was a positive relationship between *L. camara* density and *L. camara* biomass ($R_s = 0.844$, $P = 0.003$), but no association was found between *L. camara* density and the size of *L. camara*-invaded areas ($P > 0.05$).

Morphospecies richness

Invertebrate morphospecies richness was

Table 1. Assemblage of invertebrate’s taxa (N = 1086) recorded in pitfall and window traps and vegetation types inside the Groenkloof Nature Reserve, South Africa

Taxa	Windows traps			Pitfall traps			Total
	IGP	IBP	LP	IGP	IBP	LP	
Mollusca	12	23	3	8	32	NC	78
Nematoda	8	14	4	9	12	12	59
Annelida	20	8	7	10	41	3	89
Arachnida	10	35	10	6	12	3	76
Isopoda	5	3	13	9	13	2	45
Hemiptera	3	6	1	4	13	4	31
Coleoptera	2	12	5	6	20	4	49
Cephalopoda	2	7	5	8	21	1	44
Isoptera	4	14	3	3	18	7	49
Ephemeroptera	7	12	7	2	12	6	46
Mecoptera	6	8	8	9	13	4	48
Orthoptera	8	11	10	7	10	5	51
Insect larvae	2	12	NC	4	9	5	32
Plecoptera	1	6	1	9	9	2	28
Dermaptera	2	4	1	9	9	8	33
Thysanoptera	2	12	1	8	6	1	30
Blattodea	6	5	NC	4	33	12	60
Amphipoda	6	5	7	5	28	13	64
Myriapoda	4	13	3	3	41	6	70
Diplura	3	6	NC	3	17	6	35
Diptera	3	12	1	3	12	NC	31
Psocoptera	4	2	1	3	23	5	38
Total	120	230	91	132	404	109	1086

NC = Not Captured, IBG = Indigenous Bush Plots, IGP = Indigenous Grass Plots, LP = *Lantana* Plots.

significantly influenced by vegetation type ($\chi^2_2 = 1.45$; $P < 0.001$), feeding guild ($\chi^2_2 = 4.2$; $P < 0.001$), site as a covariate ($\chi^2_5 = 1.07$; $P < 0.001$), as well as the interaction between vegetation type and feeding guild ($\chi^2_4 = 4.90$; $P < 0.001$). The highest morphospecies richness was recorded for predators in the grass-dominated vegetation. Interestingly, morphospecies richness of invertebrates was negligible for the *L. camara* plots across the study particularly for herbivores (Fig. 2A). Except for detritivores, all the feeding guilds had the highest morphospecies richness in grass-dominated vegetation, and the lowest richness in the

Lantana plots (Fig. 2A).

Significant differences were found for invertebrate morphospecies richness across all vegetation types and feeding guilds and site as a covariate. In addition, there were also significant differences found for the interactions between vegetation types and indigenous grass plots; *Lantana* plots and detritivores; also for indigenous grass plots and herbivores and predators. No significant differences were found for the interaction between vegetation type *Lantana* plots and feeding guilds herbivores and predators (Table 2).

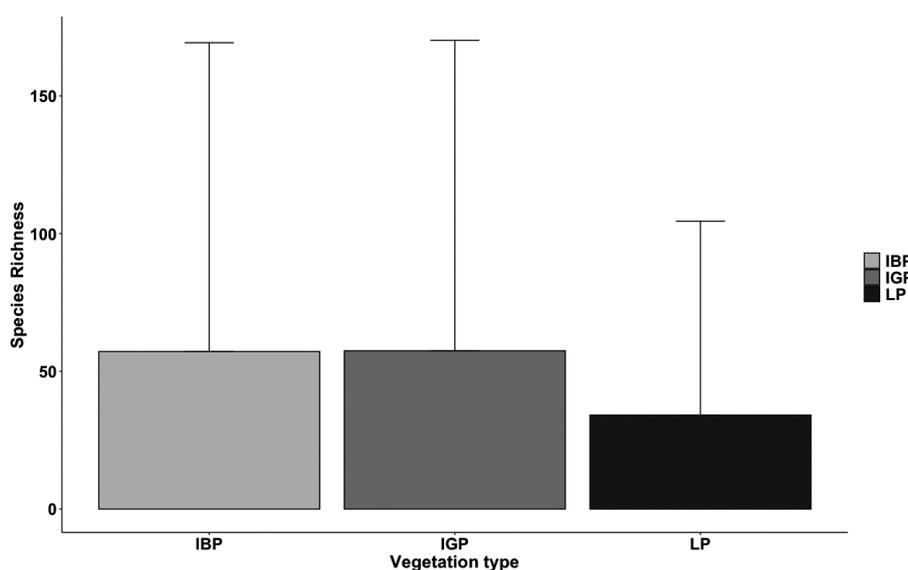


Fig. 1. Mean ± SE of species richness in the understory and the shrub layer of three different vegetation types. LP = *L. camara* plots, IBP = Indigenous bush-dominated plots and IGP = Indigenous grass-dominated plots inside the Groenkloof Nature Reserve, South Africa.

Table 2. Output of a GLM model showing the influence of vegetation type, feeding guilds and site as a covariate on morphospecies richness of invertebrates (N = 1086) inside the Groenkloof Nature Reserve, South Africa

Variables	Estimate	SE	Z value	P value
Vegetation type: IBP	5.569	0.013	423.187	$P < 0.001^{***}$
Vegetation type: IGP	5.797	0.011	490.681	$P < 0.001^{***}$
Vegetation type: LP	5.290	0.015	351.816	$P < 0.001^{***}$
Feeding guild: detritivores	-2.494	0.122	-20.371	$P < 0.001^{***}$
Feeding guild: herbivores	-3.396	0.190	-17.848	$P < 0.001^{***}$
Feeding guild: predators	-1.853	0.090	-20.547	$P < 0.001^{***}$
Site	-1.835	0.021	-85.581	$P < 0.001^{***}$
Vegetation type: IGP_ Feeding guild: detritivores	-0.242	0.171	-1.415	$P = 0.015^{**}$
Vegetation type: LP_ Feeding guild: detritivores	0.187	0.175	1.072	$P = 0.028^*$
Vegetation type: IGP_ Feeding guild: herbivores	-0.437	0.153	-2.845	$P = 0.004^{**}$
Vegetation type: LP_ Feeding guild: herbivores	-0.280	0.314	-0.894	$P = 0.371$
Vegetation type: IGP_ Feeding guild: predators	0.377	0.109	3.428	$P = 0.000^{**}$
Vegetation type: LP_ Feeding guild: predators	0.223	0.242	0.923	$P = 0.356$

LP = *L. camara* plots; IBP = Indigenous bush-dominated plots; IGP = Indigenous grass-dominated plots.

Abundance

Across the study, invertebrate abundance differed between the sampled sites ($\chi^2_5 = 4.37$; $P < 0.001$). Furthermore, invertebrate abundance was influenced by vegetation type ($\chi^2_2 = 1.48$; $P < 0.001$); feeding guild ($\chi^2_2 = 1.96$; $P < 0.001$); site ($\chi^2_5 = 5.22$; $P < 0.001$) and the interaction between vegetation type and feeding guilds ($\chi^2_4 = 4.36$; $P < 0.001$). The highest invertebrate abundances were in bush-dominated plots, followed by grass-dominated plots and fewer invertebrates were found in *L. camara*-invaded plots. Moreover, the most abundant feeding guild across the study were detritivores, even though they were less abundant in the *L. camara*-dominated plots than in the grass and bush-dominated plots (Fig. 2B).

For abundance as a response variable, significant differences were found for all vegetation type, feeding guild, site as a covariate and the interactions between vegetation types IGP, LP and feeding guild for both detritivores and predators (Table 3), but no significant difference was found between vegetation types IGP, LP and feeding guild for the herbivores (Table 3).

Biomass

Vegetation type ($\chi^2_2 = 22.26$; $P < 0.001$) and feeding guild ($\chi^2_2 = 13.43$; $P < 0.001$) had a significant influence on the biomass of invertebrates. However, neither interaction between vegetation type and feeding guild ($\chi^2_4 = 8.07$; $P = 0.232$) nor site as a covariate ($\chi^2_5 = 2.62$; $P = 0.105$) predicted the biomass of invertebrates. The effect of vegetation type on biomass of invertebrates

was the lowest in the *Lantana* vegetation type across all feeding guilds. In addition predator biomass was negligible across all vegetation types with the least biomass in the *Lantana* vegetation type (Fig. 2C).

DISCUSSION

The Riparian zones invaded by *L. camara* inside the Groenkloof Nature Reserve support fewer plant species. These zones also support fewer invertebrates, which have less biomass and less morphospecies richness of invertebrates. Mungi et al. (2020) has associated *L. camara* with loss of large expanses of indigenous vegetation, and Sharma et al. (2007) reported a loss of 160,000 ha of indigenous vegetation in Hawaii because of this invader species. Therefore, the fewer plant species supported in the *L. camara* invaded areas found in our study were not surprising. Furthermore, a study by Vivrette and Muller (1997) reported *L. camara* to reduce the suitability of soil for indigenous plant species to grow and flourish by secreting salts. However, while this was not tested in our study, this could explain the lower plant species numbers found. The association found in this study between 1) indigenous plant species richness and *L. camara* shoot density and above ground mass, 2) *L. camara* biomass and *L. camara* density, and 3) both invertebrate abundance and morphospecies richness with indigenous plant species richness suggest that there is a link between the replacement of indigenous plant species by exotic invasive *L. camara* species and the negative impact of this on invertebrate assemblages

Table 3. Output of a GLM model showing the influence of vegetation type, feeding guilds on invertebrates Abundance (N = 1086) inside the Groenkloof Nature Reserve, South Africa

Variables	Estimate	SE	Z value	P value
Vegetation type: IBP	5.023	0.012	391.734	$P < 0.001^{***}$
Vegetation type: IGP	5.252	0.011	459.087	$P < 0.001^{***}$
Vegetation type: LP	4.744	0.014	321.818	$P < 0.001^{***}$
Feeding guild: detritivores	0.088	0.017	4.972	$P < 0.001^{***}$
Feeding guild: herbivores	-3.876	0.090	-43.069	$P < 0.001^{***}$
Feeding guild: predators	-1.990	0.036	-53.810	$P < 0.001^{***}$
Site	0.197	0.027	7.115	$P < 0.001^{***}$
Vegetation type: IGP_ Feeding guild: detritivores	-0.583	0.025	-22.686	$P < 0.001^{***}$
Vegetation type: LP_ Feeding guild: detritivores	-0.639	0.030	-21.189	$P < 0.001^{***}$
Vegetation type: IGP_ Feeding guild: herbivores	-0.094	0.123	-0.770	$P = 0.441$
Vegetation type: LP_ Feeding guild: herbivores	-0.175	0.144	-1.215	$P = 0.224$
Vegetation type: IGP_ Feeding guild: predators	0.203	0.047	4.264	$P < 0.001^{***}$
Vegetation type: LP_ Feeding guild: predators	0.294	0.052	5.595	$P < 0.001^{***}$

LP = *L. camara* plots; IBP = Indigenous bush-dominated plots; IGP = Indigenous grass-dominated plots.

(Gerber et al. 2008).

The *L. camara* invasions in South Africa have been reported to decrease invertebrate diversity by Samways et al. (1996). Likewise, in our study some invertebrate taxa found in the indigenous vegetation were not found in the vegetation dominated by *Lantana* (See Table 1). In addition, Tanner et al. (2013) reported all functional feeding groups to be less abundant in plots threatened by the invasive plant species jewel weed (*Impatiens glandulifera*) in the United Kingdom. We also found the lowest number of invertebrates in the *L. camara* vegetation as compared to the bush and grass-dominated vegetation across the two trap types (See Table 1). This shows how invertebrates in our

study avoided the invaded plots (Table 1). Consistent with our findings, but in a laboratory setup, Reynolds et al. (1986) reported 100% mortality in subgroups of tobacco caterpillar (*Spodoptera litura*) fed with castor leaves treated with a 40% concentration of *L. camara* leave extract.

Morphospecies richness

Morphospecies richness of predators was highest in grass-dominated plots across the study, followed by the bush-dominated and *L. camara* invaded habitats. This is probably because spiders under the Arachnida taxa, the third most captured predator in our study, preferred the grass-dominated vegetation. In addition, according to a South African National Survey, Arachnida are endemic to the grassland biome (Haddad et al. 2013), as found in our study.

The negligible morphospecies richness of herbivores observed in our study, particularly in the *L. camara* vegetation, could be because this exotic plant is highly unpalatable to herbivorous invertebrates. Similar patterns were observed in Europe, where the exotic species *Fallopia* was never damaged by invertebrate herbivores (Gerber et al. 2008).

The insignificant differences found for the interaction between vegetation types LP and feeding guilds herbivores and predators for morphospecies richness could not be explained by our study since lots of factors contribute to interactions in most statistical models. However, since the highest morphospecies richness was for predators in the grassland vegetation, this insignificant result could be attributed to lower morphospecies richness figures for LP herbivores and predators (Fig. 2A). However, statistically insignificant does not mean biologically insignificant, especially when interactions are concerned.

Predators in our study differed in morphospecies richness among vegetation types but not in abundance or biomass.

Abundance

Lantana camara significantly influenced invertebrate abundance for the different vegetation types (Fig. 2B). That the *L. camara* invaded plots had the least abundance, especially with regard to detritivores, was not surprising. Invasive species typically cover an area extensively (Daniels and Larson 2020), and extensive cover by one plant species has been associated with unsuitable conditions for detritus invertebrate colonization (Palmer et al. 2004).

Indigenous bush can also form extensive cover. However, this was not the case with our study as it

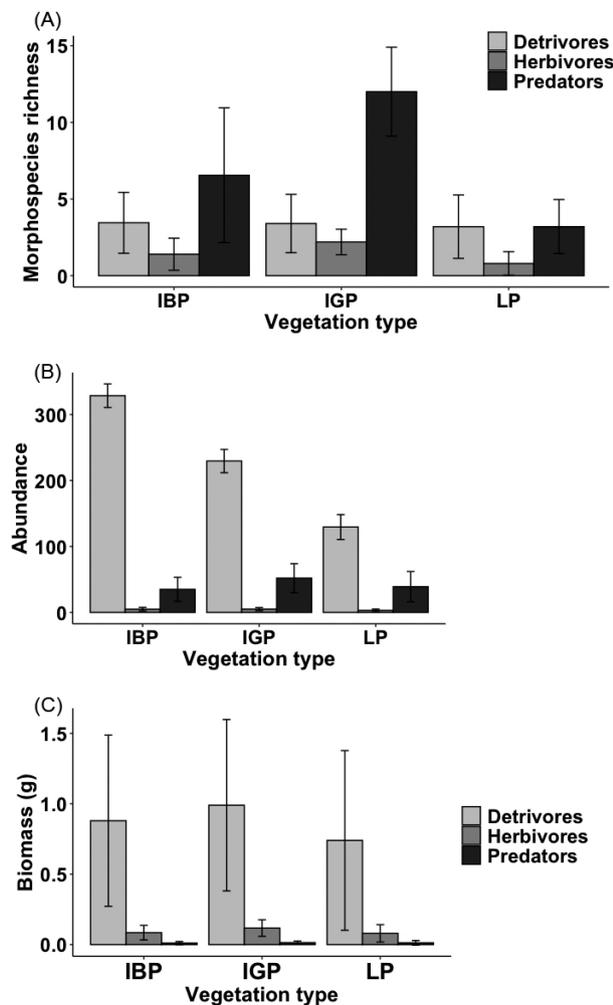


Fig. 2. Mean ± SE of morphospecies richness (top left), abundance (top right) and biomass (bottom left) of invertebrate recorded in pitfall and window traps in the three different vegetation types in the Groenkloof Nature Reserve. Means are shown for all invertebrates, herbivores, predators and detritivores. LP = *L. camara* plots; IBP = Indigenous bush-dominated plots; IGP = Indigenous grass-dominated plots.

took place during a very dry period and only invasive trees and weeds, such as *L. camara*, were flourishing during the study period because of their drought tolerant characteristics (Gunasekara and Ranwala 2018).

The significant differences found for abundance as a response variable across all predictor variables except between vegetation types IGP and LP and the feeding guild of herbivores (Table 3), show that vegetation type and feeding guild are significant factors in invertebrate assemblages and abundance in the ecosystem. Herrera and Dudley (2003) also reported vegetation type as a significant factor when it comes to reducing the abundance of and diversity of invertebrates in riparian ecosystems where the aggressive invasive *Arundo donax* occurs in Central California.

However, the lack of differences in abundance between vegetation types IGP and LP and the feeding guild of herbivores could be attributed to food availability. Naturally, there will be more food for herbivores in the *L. camara* and bush-dominated plots than in the grass-dominated plots. This could explain the lack of herbivores in these two vegetation types.

The higher abundance of detritivores found from our study (Fig. 2A), especially in bush-dominated plots, highlights the possible shade and food requirements by detritivores for survival. Bush-dominated vegetation provides more shade, and greater amounts of the foliage and dead material preferred by these detritus feeders, as compared to those dominated by grass vegetation. Surprisingly, detritivores were less abundant in the *L. camara* vegetation irrespective of the shady character of the understory. This could be because of the dense layers formed due to the low decomposition rate of *L. camara*, as is typical of invasive plant species.

Biomass

The exotic *L. camara* may be less palatable to many native herbivores, that will neglect these plots in search for food, as many non-insect invertebrates captured by our study were predators. This could reduce predator biomass in the *L. camara*-dominated vegetation.

Overall, our study revealed how *L. camara* affects invertebrates in the riparian zones inside the GNR. Even though there are several studies that investigate how invasive plants influence biodiversity, studies on how these invasive influence invertebrates' feeding guilds and assemblage and how they affect other animals that utilise arthropods as their food source are lacking in Africa (Tallamy 2004).

Our results showed that *L. camara* not only alters morphospecies richness and abundance of invertebrates but also has an influence on the productivity of riparian

habitats in terms of invertebrate biomass (Gerber et al. 2008). Invasions of riparian zones by exotic plants affects the ecosystems especially for invertebrates. This is a problem since invertebrate numbers and biomass have important implications for the earth's ecosystems. For example, invertebrates provide supporting services for primary production indirectly through nutrient cycling (Schindler et al. 2001).

Study Limitation

The study was limited by the short collection period of invertebrates. Further studies that will consider seasonal variations during invertebrates collections are recommended by this study to ascertain whether short collection period does represent an effective sampling of invertebrates in the study area.

CONCLUSIONS

In conclusion, the effects of *L. camara* on riparian zones inside Groenkloof Nature Reserve suggests more and different management strategies that include focussing on riparian zones to holistically conserve biodiversity, including invertebrates. Even though a complete removal of *L. camara* inside the reserve has been proven difficult because a complete removal of established *L. camara* is a costly task, the benefits of removal outweighs the costs as far as biodiversity is concerned, especially since the objective of protected areas is to conserve biodiversity.

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Availability of data and materials: The data set

used for this study is with the corresponding author and is available upon request.

Consent for publication: Both Authors have given a concern to publish this manuscript.

Ethical approval consent to participate: Data collection used non-invasive methods, and acceptable methods for trapping invertebrates. The traps were filled with a solution made with soap, water and salt instead of harmful chemicals such as alcohol and ethynyl glycol. Fieldwork was done with the permission of the Management of the Groenkloof Nature Reserve Permit Number: GNR 2019/20.

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