

Water hyacinth, *Eichhornia crassipes* (Pontederiaceae), reduces benthic macroinvertebrate diversity in a protected subtropical lake in South Africa

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Abstract The socio-economic impacts of the free-floating aquatic plant water hyacinth, *Eichhornia crassipes* (Pontederiaceae), on aquatic systems are well documented, yet the impacts on aquatic biodiversity, particularly invertebrate biodiversity, are less well understood. This study aimed to determine whether the presence of water hyacinth altered the diversity and assemblage structure of benthic macroinvertebrates in a conservation area. The benthic macroinvertebrate assemblage was sampled over 1 year at five sites under water hyacinth mats and at five sites without water hyacinth at Lake Nsezi—Nseleni River in the vicinity of Richards Bay, KwaZulu-Natal, South Africa. Artificial substrates were placed beneath water hyacinth mats or in the open water to allow for colonization by freshwater macroinvertebrates, and left for a period of 6 weeks, repeated on seven occasions. Twenty nine families comprising 18,797 individuals were collected, 817 (13 families) individuals were from under water hyacinth mat sites compared to 17,980 (27 families) individuals from open water sites. Ninety-eight percent of individuals collected were, however, the invasive snail, *Tarebia granifera*. Open water samples were separated from samples beneath the water hyacinth mat by non-metric Multidimensional Scaling, indicating reduced biodiversity associated with the presence of water hyacinth. Exclusion of the dominant Thiariidae from the analysis did not alter the groupings. Family richness (S) and abundance (N) were significantly higher in open water communities (S : $H_3 = 21.09$; $P = 0.0001$; N : $H_3 = 22.58$; $P = 0.00001$), while evenness (J') was higher under water hyacinth mats ($H_3 = 20.13$; $P = 0.0002$). The presence of water hyacinth had a significantly negative impact on aquatic macroinvertebrate biodiversity in a conservation

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area, and therefore the control of this invasive aquatic plant must play a major role in catchment management.

Keywords Aquatic invasions · Artificial substrates · Biodiversity · Conservation area

Introduction

Water hyacinth, *Eichhornia crassipes* (Pontederiaceae), a free-floating aquatic plant native to the Amazon Basin, is listed as one of the top 100 invaders in the world (Global Invasive Species Database (GISD) 2005). While the statement ‘the world’s worst aquatic weed’ is frequently cited, relatively few studies have actually quantified the socio-economic and environmental impacts associated with this weed. A primary focus of invasion biology is assessing the impacts of invaders, yet conservation biologists and managers spend an inordinate amount of time trying to control invasive species in order to mitigate their effects, and repair ecosystems, without having quantifiable goals against which to measure success.

Introduced into South Africa as an ornamental plant in 1908 (Hill and Cilliers 1999), water hyacinth remains the country’s most difficult aquatic weed to control despite efforts initiated in the 1960s (Coetzee et al. 2011). Because southern Africa lacks natural lake systems due to the geology of the region, it has a depauperate indigenous floating aquatic plant diversity (Cook 2004). The construction of manmade dams and impoundments has therefore created ideal habitats for invasive floating macrophytes, such as water hyacinth, to invade as a result of vacant niche availability (Davies and Day 1998). Since its introduction, water hyacinth has spread throughout South Africa, including into many protected areas, such as the Kruger National Park, South Africa’s flagship protected area.

The negative effects of water hyacinth on freshwater systems are undoubtedly numerous and varied, largely due to its rapid growth which results in expansive colonies of tall interwoven floating plants. As its doubling time is as little as 1 week under suitable conditions (Edwards and Musil 1975), infestations blanket large water bodies soon after it invades, creating impenetrable barriers which have significant negative socio-economic impacts (Coetzee et al. 2009). Whilst the socio-economic effects of water hyacinth have been fairly well documented, its impact on biodiversity and ecosystem functioning are less well understood [see Villamagna and Murphy (2010) for a full review]. Water hyacinth competes with other plants when drifting mats that scour vegetation destroy native plants and habitats (Center et al. 2002). Increased detrital production and siltation under dense water hyacinth mats results in higher sediment loading, which, in combination with a reduction in light availability to submerged plants, depletes oxygen for aquatic communities (Ultsch 1973; Mitchell 1985; Rommens et al. 2003). Phytoplankton diversity is thereby reduced (Rommens et al. 2003; Mangas-Ramirez and Elias-Gutierrez 2004) which alters the composition of invertebrate communities (Brendonck et al. 2003; Toft et al. 2003). For example, Masifwa et al. (2001) showed an increase in macroinvertebrate abundance at the edges of water hyacinth mats on Lake Victoria, Uganda, while Toft et al. (2003) demonstrated significant ecological alterations to invertebrate assemblages in the surrounding community as a result of water hyacinth invasion in the Sacramento-San Joaquin Delta, California, USA. However very few, if any studies, have investigated the

effect of water hyacinth invasion on benthic macroinvertebrate assemblages. Most studies concentrate on epiphytic invertebrate abundance which often show inflated assemblage diversity due to the abundant habitat provided by water hyacinth roots (e.g. Schramm et al. 1987; Brendonck et al. 2003; Rocha-Ramírez et al. 2007). Yet due to their sensitivity to environmental conditions, benthic macroinvertebrates are an ideal indicator group to monitor the impacts of an invasive species on community structure and biodiversity (Thirion 2000). The importance of benthic macroinvertebrates in freshwater species was reviewed by Covich et al. (1999) who highlighted their fundamental role in food webs through their acceleration of detrital composition, which results in the release of bound nutrients into the system, and their significant positions as both predators and prey in these diverse systems. Any alteration to the composition of these communities could have adverse effects on the integrity of the freshwater system which depends on how the benthic species contribute to complex food webs.

With the exception of a study by Midgley et al. (2006) who showed that water hyacinth mats significantly reduced the diversity and abundance of benthic invertebrates in an impoundment in the Eastern Cape Province of South Africa, there are no examples of quantified impacts of water hyacinth on the biodiversity of aquatic systems in South Africa, let alone within protected areas which need biodiversity benchmarks against which to measure control efforts. One such protected area is the Enseleni Nature Reserve, an Ezemvelo KwaZulu-Natal Wildlife protected area near Richards Bay in the KwaZulu-Natal Province of South Africa, where water hyacinth was first identified in the Nseleni and Mposa Rivers in 1982. By 1983, the river was 100 % covered. An integrated management plan combining biological and herbicidal control measures was implemented in 1995 and resulted in a reduction of the weed from 100 % cover to less than 20 % cover in 5 years (Jones 2001). Due to the limited amount of information available on the impact of water hyacinth on biodiversity in South Africa, particularly in conservation areas, this study aimed to determine the effect of water hyacinth mats on benthic aquatic macroinvertebrate diversity, under experimental conditions, in the Enseleni Nature Reserve. Understanding the effect that invaders such as water hyacinth have on biodiversity justifies the expectations of their control.

Methods

Study area

This study was conducted on the Nseleni River, in the proximity of Lake Nsezi (28°43'54.67''S and 31°58'47.77''E), to the west of Richards Bay on the north coast of KwaZulu-Natal, South Africa, between November 2007 and November 2008. The Enseleni Nature Reserve is the only formally protected area within the City of uMhlatuze. All of the surrounding area has been transformed to residential areas, sugar cane farms and eucalyptus tree plantations. The nature reserve has therefore become a refuge and nucleus for biodiversity, with the Nseleni River being a conduit for aquatic biodiversity between the protected area and outlying areas.

Experimental design

Five paired sites were randomly chosen, approximately 3 m from the water's edge, which had permanent mats of water hyacinth (extending approximately 8 m from the edge which were constant in size), and adjacent control sites, 5 m away, that had no water hyacinth

cover and had not had any water hyacinth cover for at least 3 years. The five sites were between 50 and 80 m apart, and approximately 1.5 m deep.

Artificial substrates were constructed using mesh bags ($20 \times 50 \text{ cm}^2$, 2 mm mesh size) filled with 2 kg of small pebbles ($\sim 3\text{--}5 \text{ cm}$ diameter) to allow for colonization by freshwater macroinvertebrates, the units of biodiversity assessed in this study. The bags were closed using a cable tie, and were fastened to a 2 m length of nylon string which was attached to two plastic toilet cistern floats and a 30 cm wooden dropper to allow for re-location at each sampling event. Five sample bags per site were placed in the centre of the sample sites and allowed to sink to the river bottom. They were left for a period of 6 weeks to allow for complete colonization by macroinvertebrates (Thirion 2000), and were then removed from the field for investigation. This was repeated on seven occasions, at six weekly intervals with the first set of bags placed in the river on 4 November 2007 and the last on 12 November 2008.

Sampling and data collection

At each sampling event, the artificial substrates were carefully removed from the water and placed into large plastic bags. The plastic bags were placed below the water surface with the openings slightly above the water level until the artificial substrate bag reached the surface, at which point the plastic bag was dropped below the water level to ensure no loss of specimens. Artificial substrate bags were then replaced at each site. All bags were returned to a field laboratory where the contents (pebbles) of each sample bag and the sample bag itself were emptied into a sorting tray, and were washed with clean water to separate any invertebrates from the bag and pebbles. Each pebble was cleaned with fresh water and a small paint brush. Both the sample bag and individual pebbles were examined for invertebrates before being removed from the sorting tray. The contents from the plastic bag were mixed with fresh water and were also emptied into the sorting tray. The contents of the sorting tray (water and substrate) were then strained through a sheet of mesh ($1 \times 1 \text{ mm}$) and macroinvertebrates and invertebrate body parts were removed with forceps and placed into glass vials, containing 75 % ethanol. Specimens were identified to family level, under a dissecting stereomicroscope, using a series of identification keys (Day and De Moor 2002; Gerber and Gabriel 2002; Day et al. 2003; De Moor et al. 2003a, b). In some instances, individuals were only identified down to order e.g. Hirudinea and Oligochaetae, as keys to families in these orders were not available. Numerical abundance of each taxon per sample bag, per site was quantified.

Data analysis

In order to determine if the study sampled sufficient invertebrate families, sample-based accumulation curves were compiled from the abundances of each sample bag, per site, using the analytically calculated S_{obs} (Mao Tao) (number of species expected), the Michaelis–Menten Mean (MMM_{Mean}) estimator (Toti et al. 2000) and the incidence-based coverage estimator (ICE) (Chazdon et al. 1998), for each site, using EstimateS V8.0.0 (Colwell 2005). The richness estimates are considered representative when the observed sample-based accumulation curves and the estimators converge closely at the highest observed richness (Longino et al. 2002).

In order to quantify the impacts of water hyacinth on benthic macroinvertebrate assemblages, multivariate community analyses of the family abundance data were conducted in Primer v6.1.11 (Clarke and Warwick 2001). Abundances from each sample bag

per site were summed and analysed as total numerical abundance. A non-metric Multi-Dimensional Scaling (nMDS) plot was constructed following resemblance analysis using the Bray-Curtis measure of similarity to examine relationships between benthic macro-invertebrate assemblages beneath water hyacinth mats and those in open water environments. In addition, due to the abundant presence of an invasive snail, *Tarebia granifera* (Thiaridae) in the open water samples, an additional nMDS plot was constructed excluding the family Thiaridae, to investigate its effect on the assemblages because the dominance by one group could mask the results.

A number of measures of diversity were calculated in EstimateS V8.0.0 to determine differences in invertebrate diversity beneath water hyacinth mats and in open water for each sample period using total abundance values, including total family richness (S), individual abundance (N), and Pielou's evenness (J') (Magurran 2004). Once again, analyses were conducted both including and excluding the dominant family, Thiaridae. Kruskal–Wallis ANOVAs, followed by multiple comparisons tests compared differences in diversity measures across the seven sample periods between beneath water hyacinth mats and open water sites, in the presence and absence of the Thiaridae.

Results

A total of 29 different families comprising 18,797 individuals was recorded (Table 1). Of the individuals collected, 817 were from under water hyacinth mat sites (WH) compared to 17,980 from open water sites (OW). More taxa were collected from the open water (27 families) in comparison to those from samples gathered from under water hyacinth mats (13 families). Snails in the family Thiaridae were by far the most abundant benthic invertebrate family sampled both under water hyacinth mats and in open water—15,694 individuals were collected (85 %) (Table 1). Ninety-eight percent of individuals in this family were identified as the invasive snail, *Tarebia granifera* (83 % of all specimens identified). Hirudinea, Polycentropidae, Oligochaetae and Leptophlebiidae were the next abundant taxa across both sites (Table 1). Rarefaction curves showed a significant difference in projected family richness between the open water and under water hyacinth samples, indicated by the non-overlapping confidence bands ($P < 0.05$) (Fig. 1). These results show that the presence of water hyacinth significantly reduced species richness of the benthic communities beneath the water hyacinth canopy.

Results of the nMDS indicated the presence of two distinct assemblages (Fig. 2a), comprising one group of open water sample events 1–7 and the other group of under the water hyacinth canopy sample events 1–7. Similarly, when the dominant family, Thiaridae, was excluded from the analysis, the same two distinct groupings are evident (Fig. 2b). Thus, despite the abundant presence of the invasive snail, the presence of water hyacinth significantly altered the benthic invertebrate assemblages.

A number of diversity measures were considered in this study to determine the effect of water hyacinth on benthic invertebrate assemblages. With the dominant family included in the analyses, total family richness (S) (Fig. 3a) and abundance (N) (Fig. 3b) were both significantly higher in open water communities (S: $H_3 = 21.09$; $P = 0.0001$; N: $H_3 = 22.58$; $P = 0.00001$). Pielou's evenness (J'), the ratio of observed diversity to maximum diversity, was used to measure evenness. Unlike the previous two measures, evenness was not significantly different between the two sites even though more individuals from fewer families were found underneath water hyacinth mats compared to open water ($H_3 = 20.13$; $P = 0.0002$) (Fig. 3c). When the Thiaridae were excluded from the

Table 1 Mean numerical abundance \pm standard error of benthic macroinvertebrate taxa, in order of dominance, from samples that inhabited artificial substrates placed under water hyacinth mats and from open water

Taxon (common name)	Under water hyacinth	Open water
Thiaridae (thiarid snails)	14.14 \pm 9.12	483.76 \pm 85.27
Hirudinea (leeches)	1.23 \pm 0.35	14.82 \pm 2.6
Polycentropidae (caseless caddisflies)	0.86 \pm 0.32	14.55 \pm 5.85
Oligochaeta (aquatic earthworms)	0.63 \pm 0.19	4.55 \pm 1.32
Leptophlebiidae (prong gilled mayflies)	0.06 \pm 0.04	2.55 \pm 1.13
Unionidae (pearly mussels)	0.71 \pm 0.36	2.12 \pm 0.63
Planorbidae (orb snails)	0.37 \pm 0.1	1.67 \pm 0.47
Polymitarcyidae (pale burrower mayflies)	0.03 \pm 0.03	0.52 \pm 0.28
Gomphidae (club tail dragonflies)	0.09 \pm 0.06	0.36 \pm 0.1
Atyidae (freshwater shrimp)		0.3 \pm 0.15
Chironomidae (midges)	0.14 \pm 0.08	0.27 \pm 0.16
Libellulidae (skimmer or percher dragonflies)		0.18 \pm 0.08
Coenagrionidae (narrow-winged damselflies)		0.18 \pm 0.11
Lestidae (spreadwing damselflies)	0.03 \pm 0.03	0.12 \pm 0.09
Chlorolestidae (damselflies)		0.09 \pm 0.09
Gyrinidae (whirligig beetles)		0.09 \pm 0.07
Hydropsychidae (caseless caddisflies)		0.09 \pm 0.07
Sphaeriidae (pill clams)		0.06 \pm 0.04
Corduliidae (emerald dragonflies)		0.06 \pm 0.06
Isopoda		0.06 \pm 0.06
Amphipoda (scuds, sideswimmers)		0.06 \pm 0.06
Baetidae (small minnow mayflies)		0.03 \pm 0.03
Aeshnidae (hawker dragonflies)		0.03 \pm 0.03
Potamonautidae—Crabs		0.03 \pm 0.03
Elmidae (riffle beetles)		0.03 \pm 0.03
Pleidae (pigmy backswimmers)		0.03 \pm 0.03
Dytiscidae (predacious diving beetles)	0.17 \pm 0.09	
Hydraenidae (minute moss beetles)	0.03 \pm 0.03	

Means derived from seven sampling events

analyses, there was no significant difference in family richness (S) within sites because one family was excluded from both treatments ($H_3 = 21.09$; $P = 0.0001$), but due to the sheer numbers of the Thiaridae in the open water sites, abundance (N) was significantly reduced in their absence ($H_3 = 22.58$; $P = 0.00001$). On the other hand, evenness was significantly higher in the open water sites with the Thiaridae excluded from the analysis, again due to their sheer numbers ($H_3 = 20.13$; $P = 0.0002$).

Discussion

The ultimate goal of tackling biological invasions ‘should be the conservation or restoration of ecosystems to preserve or re-establish native biodiversity and functions’ (Hulme

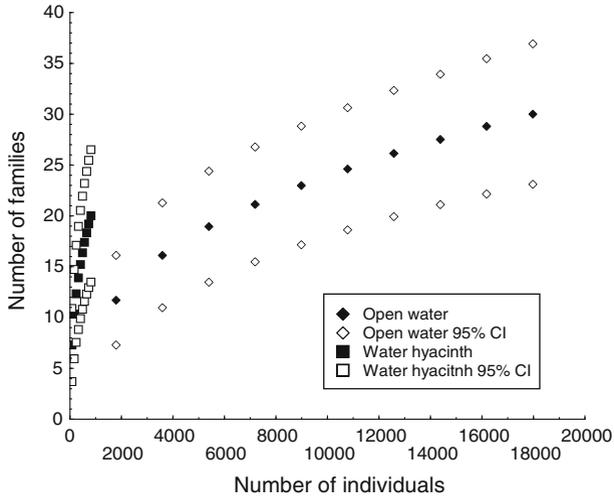


Fig. 1 Rarefaction curves, scaled by *number* of individuals, comparing observed family richness of benthic macroinvertebrates between open water samples and under water hyacinth canopy samples. Non-overlapping 95 % confidence interval *bands* indicate significant differences ($P < 0.05$) between the two curves

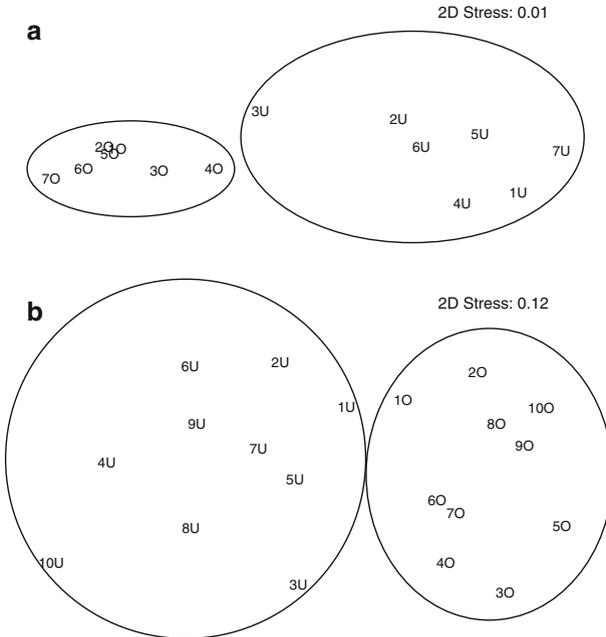
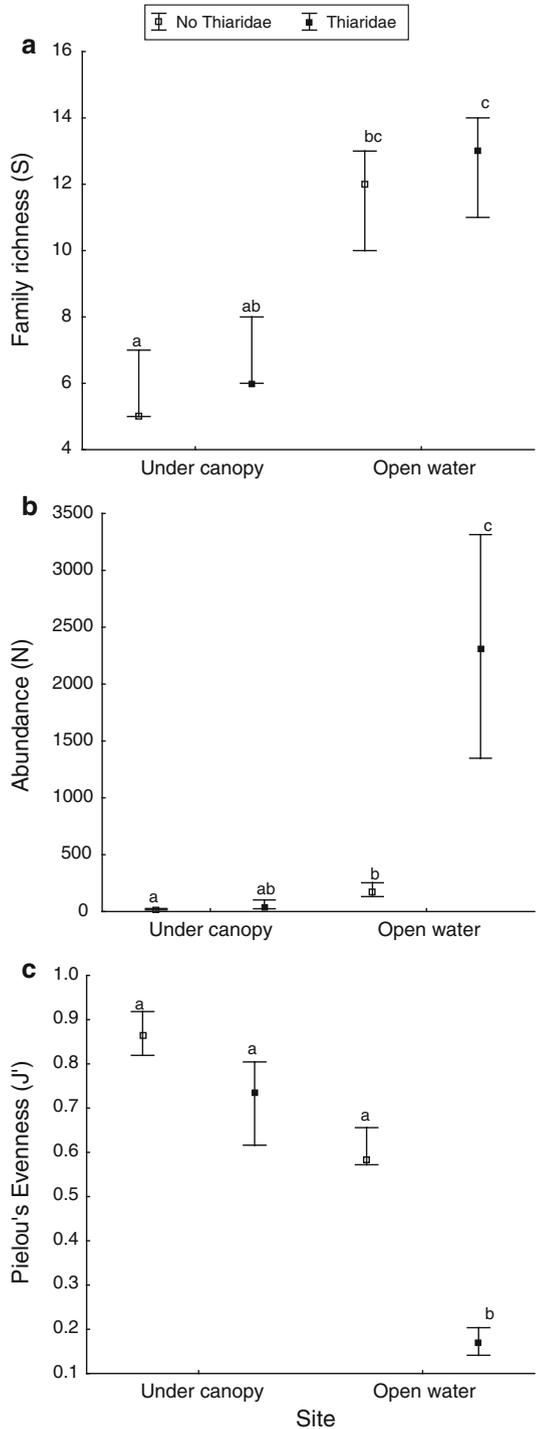


Fig. 2 Non-metric Multidimensional Scaling *plot* indicating a distinct difference between macroinvertebrate assemblages found beneath water hyacinth mats and those from open water, **a** in the presence of the dominant family, Thiariidae, and **b** in its absence

Fig. 3 Comparison of measures of diversity of aquatic macroinvertebrate communities between open water sites and from under water hyacinth mats, in the presence and absence of the dominant family, Thiaridae. **a**—family richness (S), **b**—abundance (N), **c**—Pielou’s evenness (J’). Medians compared by Kruskal–Wallis ANOVA tests, *error bars* represent 25 percentiles. *Different letters* indicate significant differences between medians ($P < 0.05$)



2006). Yet without knowing how invasive species impact indigenous community structure and functioning, there is no way of measuring a return to a functioning state. Conservationists and managers are under increasing pressure to mitigate the negative effects of invasive species on ecosystem dynamics, but in the absence of quantifiable results, it is difficult to understand these impacts (Richardson and Van Wilgen 2004), let alone manage them. This study quantified the effect of water hyacinth on assemblage structure and diversity of aquatic macroinvertebrates in a protected area, the Enseleni Nature Reserve, and demonstrated, categorically, that water hyacinth alters benthic invertebrate assemblages, and reduces macroinvertebrate diversity beneath its mats.

Of the benthic invertebrates that were found under the water hyacinth mats, 49 % came from families that are considered highly tolerant of pollution (e.g. Hirudinea, Planorbidae, Oligochaeta and Chironomidae), 43 % were moderately tolerant of pollution (e.g. Leptophlebiidae and Lestidae) and 8 % have a very low tolerance to pollution (Polycentropidae) according to the South African Scoring System (SASS version 5) which was developed as a biotic monitoring tool for freshwater habitats (Dickens and Graham 2002). There was a far higher diversity of benthic invertebrates collected from open water sites without water hyacinth and represented families in the Ephemeroptera, Trichoptera and Odonata, all of which are indicative of less impacted aquatic ecosystems (Dickens and Grahams 2002).

The importance of benthic invertebrates in ecosystems generally goes unnoticed because they are cryptic, occurring below the water's surface, until unexpected changes occur in ecosystems (Covich et al. 1999). Any change or reduction in benthic invertebrate diversity has important consequences for ecosystem functions such as sediment mixing, nutrient cycling and energy flow through food webs (Lindegaard 1994; Wallace and Webster 1996). Indeed, Neira et al. (2005) demonstrated the multiple physical, chemical, biotic and trophic effects of the invasion of mudflats by a hybrid form of the eastern cordgrass *Spartina alterniflora* in San Francisco Bay, CA., USA, which have resulted in substantial changes in benthic communities that are likely to have important effects on the entire ecosystem. While the knock-on effects of altered benthic macroinvertebrate assemblages as a result of water hyacinth invasion were not investigated here, it is imperative to protect the diversity of these benthic communities to lower the risk of unexpected and unwanted consequences as demonstrated by studies such as Neira et al. (2005), particularly in conservation areas.

This study has also elucidated the impressive invasion of the Nseleni River by the snail *T. granifera*, which has been recorded before in high densities in this river by Appleton et al. (2009). Eighty-three percent of all specimens collected in this study were identified as *T. granifera*. This species is indigenous to south-east Asia but has been moved around the world in the aquarium trade and is now regarded as invasive in North and South America. It was first recorded in South Africa in 1999 from a reservoir south of Richards Bay/Empangeni, northern KwaZulu-Natal, and since then has become widespread in the eastern half of South Africa (Appleton and Nadasan 2002; Appleton et al. 2009). Despite its impressive invasive tendencies, even it is excluded by the presence of water hyacinth—in this study, there was a 90 % reduction in its abundance under water hyacinth mats. Furthermore, when excluded from the community and diversity analyses, there were no significant changes in the assemblage or diversity patterns, with the exception of abundance and evenness in the open water sites as a result of the sheer numbers of these snails. To date, the effect of *T. granifera* on the benthic fauna of invaded habitats in South Africa has not been demonstrated, but it has been associated with the disappearance of two indigenous benthic gastropod species from rivers in Puerto Rico (Samadi et al. 1997) suggesting that

there is potential for detrimental impacts in South Africa too, which need to be investigated.

In addition to the invasion of the Nseleni River by water hyacinth and *T. granifera*, an invasive loricariid catfish, *Pterygioplichthys disjunctivus* (Weber 1991), has recently established a self-perpetuating population in the system (Jones et al. 2013), suggesting that “invasional meltdown”, as postulated by Simberloff and Von Holle (1999), has occurred. This theory proposes that ecosystems become more easily invaded as the cumulative number of species introductions increases, and that facilitative interactions can exacerbate the impact of invaders. Future studies therefore need to prioritize investigating the combined effect of water hyacinth, the thiarid snail and the catfish on benthic biodiversity and ecosystem functioning of this protected area.

In conclusion, while studies have demonstrated the positive attributes of water hyacinth, such as nutrient control, phytoremediation and increased biodiversity in certain assemblages, the costs associated with its presence in the majority of systems around the world far outweigh its benefits. The results of this study and those of Midgley et al. (2006) have shown that dense water hyacinth cover significantly impacts the biodiversity of benthic invertebrate assemblages, which is of particular importance in protected areas, such as the Enseleni Nature Reserve, whose mission statement is “to ensure effective conservation and sustainable use of KwaZulu Natal’s biodiversity in collaboration with stakeholders for the benefit of present and future generations” (www.kznwildlife.com). Thus mitigating the negative impacts of water hyacinth invasion on biodiversity warrants its control.

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