

Dietary niche constriction when invaders meet natives: evidence from freshwater decapods

MICHELLE C. JACKSON^{1,2}, IAN DONOHUE^{3,4,*}, KATIE MILLER¹, J. ROBERT BRITTON⁵ AND JONATHAN GREY^{1,6}

1. School of Biological and Chemical Sciences, Queen Mary University of London, Mile End Road, London, England.
2. Centre for Invasion Biology, Department of Zoology and Entomology, University of Pretoria, Hatfield, South Africa.
3. School of Natural Sciences, Department of Zoology, Trinity College Dublin, Ireland.
4. Trinity Centre for Biodiversity Research, Trinity College Dublin, Ireland.
5. Centre for Conservation Ecology and Environmental Change, Bournemouth University, Bournemouth, England.
6. Lancaster Environment Centre, Lancaster University, Lancaster, England.

**Corresponding author:* Ian Donohue

Telephone: +35318961356

Email: ian.donohue@tcd.ie

Running head: Niche constriction and invasion success

Abstract

1. Invasive species are a key driver of global environmental change, with frequently strong negative consequences for native biodiversity and ecosystem processes. Understanding competitive interactions between invaders and functionally similar native species provides an important benchmark for predicting the consequences of invasion. However, even though having a broad dietary niche is widely considered a key factor determining invasion success, little is known about the effects of competition with functionally similar native competitors on the dietary niche breadths of invasive species.
2. We used a combination of field experiments and field surveys to examine the impacts of competition with a functionally similar native crab species on the population densities, growth rates and diet of the globally widespread invasive red swamp crayfish in an African river ecosystem.
3. The presence of native crabs triggered significant dietary niche constriction within the invasive crayfish population. Further, growth rates of both species were reduced significantly, and by a similar extent, in the presence of one another. In spite of this, crayfish maintained positive growth rates in the presence of crabs, whereas crabs lost mass in the presence of crayfish. Consequently, over the three year duration of the study, crab abundance declined at those sites invaded by the crayfish, becoming locally extinct at one.
4. The invasive crayfish had a dramatic effect on ecosystem structure and functioning, halving benthic invertebrate densities and increasing decomposition rates four-fold

compared to the crabs. This indicates that replacement of native crabs by invasive crayfish likely alters the structure and functioning of African river ecosystems significantly.

5. This study provides a novel example of the constriction of the dietary niche of a successful invasive population in the presence of competition from a functionally similar native species. This finding highlights the importance of considering both environmental and ecological contexts in order predict and manage the impacts of invasive species on ecosystems.

Key words: diet breadth, community structure, ecosystem functioning, interspecific competition, invasive species, *Procambarus clarkii*, stable isotopes.

Introduction

The increasingly rapid spread of biological invaders in recent decades (e.g. Ricciardi, 2006; Jackson & Grey, 2013) is a key driver of global environmental change with major implications for biodiversity and ecosystem functioning (Chapin *et al.*, 2000; Gurevitch & Padilla, 2004; Pejchar & Mooney, 2009). Invasive species can have dramatic and often irreversible effects on ecosystem services, frequently with considerable economic implications (Pejchar & Mooney, 2009), particularly in aquatic ecosystems which tend to be especially susceptible (Rahel & Olden, 2008). They can change fundamental ecological processes from the level of the individual to the whole ecosystem (Peltzer *et al.*, 2010; Penk *et al.* 2015), and frequently alter the distribution of native species via a range of competitive

interactions, predation pressure and/or habitat degradation (Mack *et al.*, 2000; Simon & Townsend, 2003; Hooper *et al.*, 2005). Classic invasion theory (Elton, 1958) predicts that a successful invader can establish itself in an unoccupied niche or displace similar native species by overlapping their niche and eventually out-competing them for limited resources. Having a generalist diet is a common trait of successful invaders, allowing them to opportunistically exploit diverse resources in novel environments and competitively exclude native species (Snyder & Evans, 2006). Hence, the decline of native species following an invasion is often a consequence of direct interspecific competition for resources (e.g. Porter & Savignano, 1990; Olsson *et al.*, 2009).

Classic niche theory predicts that species occupy a larger niche in the absence of interspecific competition (Hutchinson 1957; Van Valen 1965). Hence, competing invasive and native species would be expected to occupy smaller niches than their allopatric counterparts (Bolnick *et al.*, 2010). However, increased competition for resources might necessitate an increase in dietary niche breadth in order to maintain energy requirements (Svanbäck and Bolnick 2007). Empirical evidence indicates that interspecific competition can cause the niche widths of consumers to increase or decrease depending on the context but the reason for this variation is still unclear (Araújo *et al.*, 2011). Despite these apparently contradictory processes (Araújo *et al.*, 2011), the dietary niche breadth of invasive and native competitors has rarely been examined, even though it is likely a key factor in determining invasion success and predicting impacts of invaders on biodiversity (e.g. Jackson & Britton, 2013).

Some of the most destructive freshwater invaders are crayfish (Lodge *et al.*, 2012), which can alter ecosystem structure and processes by, for example, homogenising biotic assemblages and altering rates of both primary production and decomposition (Stenroth &

Nyström, 2003; Renai & Gherardi, 2004; Gherardi & Acquistapace, 2007; Jackson *et al.*, 2014). However, the full range of ecological mechanisms underpinning the impacts of invasive crayfish on ecosystem structure and functioning has yet to be examined fully (Lodge *et al.*, 2012). The red swamp crayfish (*Procambarus clarkii*) is a particularly important invader due to its global distribution (Capinha *et al.*, 2011) and the severe impacts it has on ecosystems (Klose and Cooper, 2012; Jackson *et al.*, 2014). In the 1960s, the species was introduced to multiple locations throughout East Africa for aquaculture (Harper *et al.*, 2002) where its effects remain largely unknown. This widespread introduction of crayfish in the region has led to encroachment on the range of many native freshwater crab species, several of which are endangered and endemic (Cumberlidge *et al.*, 2009). Indeed, crayfish have replaced crabs as the primary food source of African clawless otters in some Kenyan rivers (Ogada, 2006), providing indirect evidence that they are becoming more abundant and outcompeting crabs. Crabs and crayfish are both benthic omnivores with similar feeding mechanisms. They frequently occupy similar functional roles as shredders driving detrital processing (Nyström *et al.*, 1996; Dudgeon, 1999; Dobson *et al.*, 2002; Lancaster *et al.*, 2008) and consequently, the introduced crayfish might be expected to be functionally redundant (*sensu* Naeem, 1998). However, even morphologically similar species can vary significantly in their effects on ecosystems (O'Connor & Bruno, 2007; Matthews *et al.*, 2011; Jackson *et al.*, 2014; Penk *et al.*, 2015). Furthermore, highly invasive species tend to be more voracious than their native counterparts (Haddaway *et al.*, 2012; Alexander *et al.*, 2014), implying that red swamp crayfish could cause considerable alteration to ecosystem structure and processes were they to replace native crabs.

Using a combination of field experiments and surveys, we examined interactions between the globally widespread invasive red swamp crayfish and a native crab,

Potamonautes loveni, and compared their impacts on the structure and functioning of a Kenyan river ecosystem. We quantified dietary niche breadth in experimental and natural populations of both species in allopatry and sympatry by analysis of carbon and nitrogen stable isotopes from muscle tissue which provides information on dietary preferences integrated over time. We then obtained quantitative measures of total foraging niche space at the population level using methods that incorporate individual variation of stable isotope ratios (Layman *et al.*, 2007a; Newsome *et al.*, 2007; Jackson *et al.*, 2011). We hypothesised that (i) the invasive crayfish has stronger per-capita effects on ecosystem structure and functioning than the native crab and, (ii) the invasive crayfish and native crabs will have a larger niche breadth when present together.

Materials and Methods

Study system

The River Malewa is the primary tributary of Lake Naivasha (catchment area 1750 km²) in Kenya's Rift Valley in East Africa. Red swamp crayfish were introduced to Lake Naivasha in 1970 and subsequently spread throughout the catchment as a result of further introductions and natural dispersal, leading to their intrusion upon the range of a native river crab, *P. loveni* (Foster & Harper, 2007). The native range of *P. loveni* extends from Eastern Kenya into Western Uganda (Cumberlidge, 2008).

Experimental design

We conducted two field experiments in the River Malewa near Gilgil (0°31' 5" 5.42" S, 36° 24' 3.33" E) at a site at the crayfish invasion front where both species were present; the first in April 2009 and the second in March 2010. Both experiments were of 28 days duration and comprised the same two fixed factors in a fully-factorial design: the presence of the

native crab (two levels: present and absent) and the presence of red swamp crayfish (two levels: present and absent). The experiments differed, however, in both the level of replication (experiment 1: $n = 4$; experiment 2: $n = 6$) and the range of response variables that were quantified (Table 1).

Table 1. Response variables quantified during the two field experiments. The number of replicate experimental units is shown in parentheses.

Experiment 1 (April 2009)	Experiment 2 (March 2010)
Litter breakdown rates ($n = 4$)	Litter breakdown rates ($n = 6$)
Algal biomass ($n = 4$)	Benthic invertebrate assemblage structure ($n = 5$)
	Stable isotope analysis (crayfish [crabs absent]: $n = 6$ [24 individuals]; crabs [crayfish absent]: $n = 5$ [20 individuals]; crayfish [crabs present]: $n = 5$ [10 individuals]; crabs [crayfish present]: $n = 5$ [10 individuals])

We manipulated the presence of our target organisms using steel-framed cages (Figure S1). The cages had a 0.315 m^2 solid base surrounded by stainless steel mesh (5 mm aperture) to regulate the presence of crayfish and crabs and allow the free movement of other smaller invertebrates, algae and organic matter, and were 30 cm in height. The entire base of each cage was covered with homogenised pebble and gravel substratum from the adjacent river bed and a native macrophyte (Family Scrophulariaceae, ~5 g fresh weight) was added to simulate natural habitat. A 400 cm^2 tray was also filled with river substratum and inoculated with benthic invertebrates from a uniform kick sample taken from the experimental site.

Crayfish and crabs used in the experiment were collected from allopatric populations in the River Malewa. All treatments comprised four individuals of crabs and/or crayfish, with a combined biomass of ~30 g fresh weight. This was within the range of natural densities of both species at each of our survey locations (12 individuals m^{-2} ; Harper et al. 2002). The weight (g) and size (carapace length [cephalic groove to the end of the rostrum] in crayfish and carapace width [at the widest point] in crabs; (mm)) of all individuals was recorded before addition to the enclosures and did not vary between species in each enclosure (RANGE?). Where both crabs and crayfish were present together, treatments contained two individuals of both species, all of similar biomass. Results from substitutive experiments that manipulate density and biomass, such as we use here, can be sensitive to the selected total mixture density (Inouye, 2001). However, they can effectively quantify the extent to which the measured outcome is a consequence of the mixture components (in our case, presence or absence of crayfish and crabs at natural densities). Comparison of the treatment with crayfish present and crabs absent with the treatments without crayfish allowed us to quantify the effect of the invader on the ecosystem, while comparison of the single species treatments with the two-species treatment allowed us to quantify the effect of interspecific competition on crayfish and crab growth and diet. Our four treatment combinations (crayfish only; crabs only; crayfish and crabs; neither crayfish nor crabs) were assigned to cages placed along a 40 m stretch of the river following a randomised block design. The cage mesh was cleaned every four days to remove debris, and at the same time we checked for survival. Survival rates were 100% throughout most the experiment, however there was low mortality of crabs in the last week of the first experiment and therefore it was only used for measures of ecosystem functioning (Table 1). Unfortunately, two enclosures were damaged during the second experiment and were omitted from analyses (Table 1).

Experiment sampling protocol

We quantified leaf litter breakdown using plastic mesh bags (aperture 5 mm) containing a known mass (~ 2.5 g) of air-dried leaf litter (from a native riparian shrub; *Dovyalis abyssinica*) fastened to the side of each cage at the start of the experiments. The leaf litter remaining in each mesh bag after 72 hours (to avoid total decomposition of material) was washed and then dried at 60°C to constant weight. The exponential decay rate coefficient (k) was calculated for each treatment as a measure of the rate of leaf litter breakdown as:

$$k = \frac{\log\left(\frac{M_t}{M_0}\right)}{t}$$

where t is the duration of exposure (in days), M_0 is the initial dry mass (in grams) and M_t is the dry mass at time t (Hieber & Gessner, 2002).

We measured epilithic algal standing stock by placing a terracotta tile (10 x 10 cm) in each cage at the start of the experiment. Tiles were removed on the final day and all biofilm was washed off and filtered through GF/C filters (Whatman[®], Maidstone, UK) before adding 15 ml of 90% acetone. Chlorophyll a concentration of the filtrate, quantified following Jeffery and Humphrey (1975), was used as a measure of periphyton biomass. We quantified the benthic invertebrates in the cages at the end of the experiment by removing, counting and identifying all organisms in 20 cm² trays that were filled with substrate from the experimental site at the commencement of the experiment. All invertebrates were stored in 70% ethanol prior to identification (to genus) and enumeration.

On the final day of each experiment, crayfish and crabs were removed from the enclosures and their total mass recorded. Measurements of total fresh mass were used to

calculate per capita growth rates over the duration of the experiment to compare growth in allopatry and sympatry. All individuals were then frozen overnight and thawed before dissecting a portion of muscle from the tail of crayfish and the abdomen of crabs for the analysis of stable isotopes. The duration of the experiment (28 days) allowed sufficient time for crayfish and crab muscle tissue to equilibrate with their diet in experimental conditions at warm river temperatures (15 to 20 °C; e.g. Fantle *et al.*, 1999; Fry *et al.*, 2003; McIntyre & Flecker, 2006). We also collected epilithic algae, macrophytes, detritus and dominant benthic invertebrate species from the experimental location for stable isotope analysis. All tissue samples were oven-dried at 60°C, ground using an agate pestle and mortar and 0.6 to 1.0 mg weighed into 6 x 4 mm tin cups using a microbalance. Stable carbon and nitrogen isotope analyses were carried out using an elemental analyser (Flash EA 1112 series, Thermo-Finnigan, Waltham, Massachusetts, USA) coupled to a continuous flow isotope ratio mass spectrometer (Finnigan MAT Delta^{Plus}, Thermo-Finnigan, Waltham, Massachusetts, USA). Secondary standards (sucrose for carbon; ammonium sulphate for nitrogen) with known relation to international standards (Pee Dee Belemnite for carbon; nitrogen in air for nitrogen) were used as reference materials. Cyclohexanone-2,4-Dinitrophenylhydrazone was used as an internal standard and resulted in typical precision of < 0.1‰ for carbon and < 0.3% for nitrogen.

Finally, in the first experiment only, the guts were dissected from 5 randomly selected individual crayfish from each treatment and stored in 50% ethanol. A Novex Holland Microscope at x 10 magnification was then used to separate vegetation and invertebrate contents before drying to constant weight to calculate the relative proportion of plant to animal ingested by each individual. *Field surveys*

We examined the distribution of both crabs and crayfish at four sites in the River Malewa, located approximately 10, 30, 35 and 40 km upstream from Lake Naivasha, in 2008, 2009 and 2010. We quantified the abundance of adult crayfish and crabs on 3 to 8 occasions at each site in each year over a 6 week period, using a series of crayfish traps of 5 mm mesh, baited with fish pellets, and left for between 1 - 2 hours. All crayfish and/or crabs were removed and counted and we used the catch per unit effort (CPUE; number of individuals per trap per hour) to quantify relative abundance among sites. Samples ($n = 6$ to 18) of a selected common size range of crabs (26 - 35 mm carapace width) and crayfish (31 - 45 mm carapace length) from all sites on all sampling occasions were prepared for stable isotope analysis as described previously. Epilithic algae, macrophytes, detritus and dominant benthic invertebrate species were also collected from all the sites for subsequent stable isotope analysis.

Data analyses

We used analysis of variance (ANOVA) to test for independent and combined effects of crayfish and crabs on algal biomass, rates of leaf litter breakdown, trophic niche widths (see below), total invertebrate abundance and invertebrate taxon richness. We measured leaf litter breakdown in both field experiments (Table 1) and tested for differences between experiments by incorporating experiment number as an additional random factor. There was no difference between the two experiments and this random effect was then removed to maximise the power of the subsequent analysis. The Student-Newman-Keuls procedure was used to make post-hoc comparisons among levels of significant terms. We used permutational multivariate analysis of variance (PERMANOVA; Anderson et al., 2008) to test for effects of our experimental manipulations on the structure of invertebrate assemblages using the PERMANOVA+ add-in to PRIMER[®] version 6.1 (PRIMER-E Ltd, Plymouth, UK).

These analyses were performed with 9999 permutations of the residuals under a reduced model (McArdle & Anderson, 2001; Anderson *et al.*, 2008) and were based on Bray-Curtis similarity matrices calculated from $\log(X + 1)$ -transformed abundance data. Similarity percentages (SIMPER; Clark & Warwick, 2001) analysis was used to determine the contribution of each benthic invertebrate taxon to the pairwise Bray-Curtis dissimilarities between experimental treatments. All analyses were balanced and variables were transformed where necessary prior to analysis to homogenise variances.

The stable isotope-derived population metrics of carbon range (CR_b) and nitrogen range (NR_b) were used as measures of the trophic niche width of crabs and crayfish (Layman *et al.*, 2007b; Jackson *et al.*, 2012) in both the field experiments and field surveys. CR_b and NR_b were calculated as the Euclidean distance between the individuals with the lowest and highest $\delta^{13}C$ and $\delta^{15}N$ respectively and therefore reflect the total range of isotopic space used by a group of organisms. To enable comparison among variables and to account for small sample sizes, the metrics were bootstrapped ($n = 10000$; indicated with a subscript 'b') based on the minimum sample sizes (2 individuals in the field experiment [in the enclosures containing 2 crabs and 2 crayfish – each measurement was therefore replicated independently five times (Table 1)] and 6 individuals in the field surveys [at field sites with declining crab numbers]). We then used ANOVA to test for differences in the trophic niche width of crayfish and crabs when together and apart in the field experiment. However, given that one of our experimental treatments contained no crayfish or crabs, and could not therefore provide measurements of trophic niche widths of either species, our statistical model testing for differences in trophic niche width comprised two fixed factors: species (two levels: crabs and crayfish) and the presence of interspecific competition with a functionally similar species (two levels: competition and no competition). For the field survey data, we tested for correlations between

our metrics of trophic niche width of each species and the corresponding CPUE of its functionally similar competitor. To ensure comparability in the isotopic data among survey sites and years, we used PERMANOVA to test for any effect of site or year on the stable isotopic signature of putative resources (biofilm, macrophytes, detritus and dominant invertebrate taxa). We found similar stable isotope values of putative resources both among survey sites and years (Table S1), indicating that any differences in trophic niche width between crayfish and crabs were attributable to actual dietary variation.

Results

Field experiments

There was a significant interaction between the presence of crayfish and crab on rates of leaf litter breakdown (ANOVA; $F_{1,36} = 12.66$, $P = 0.001$). Rates of breakdown were significantly higher in the cages containing only crayfish compared with those containing none of the manipulated consumers (control treatment). In contrast, cages containing crabs were similar to those in the control treatment, while cages containing both crayfish and crabs had intermediate effect on breakdown rates (Fig. 1). The latter result indicates that the presence of crabs did not alter the per capita effects of crayfish on leaf litter breakdown rates. Neither crabs nor crayfish affected the standing stock of epilithic algae. \

Crayfish reduced the density of benthic invertebrates (ANOVA, $F_{1,16} = 9.39$, $P = 0.007$, Fig. 2) and altered their multivariate structure significantly (PERMANOVA; Pseudo- $F_{1,17} = 6.05$, $P = 0.004$). The latter result was driven primarily by reductions in the density of hydropsychid caddis flies (Trichoptera) together with an increase in the density of

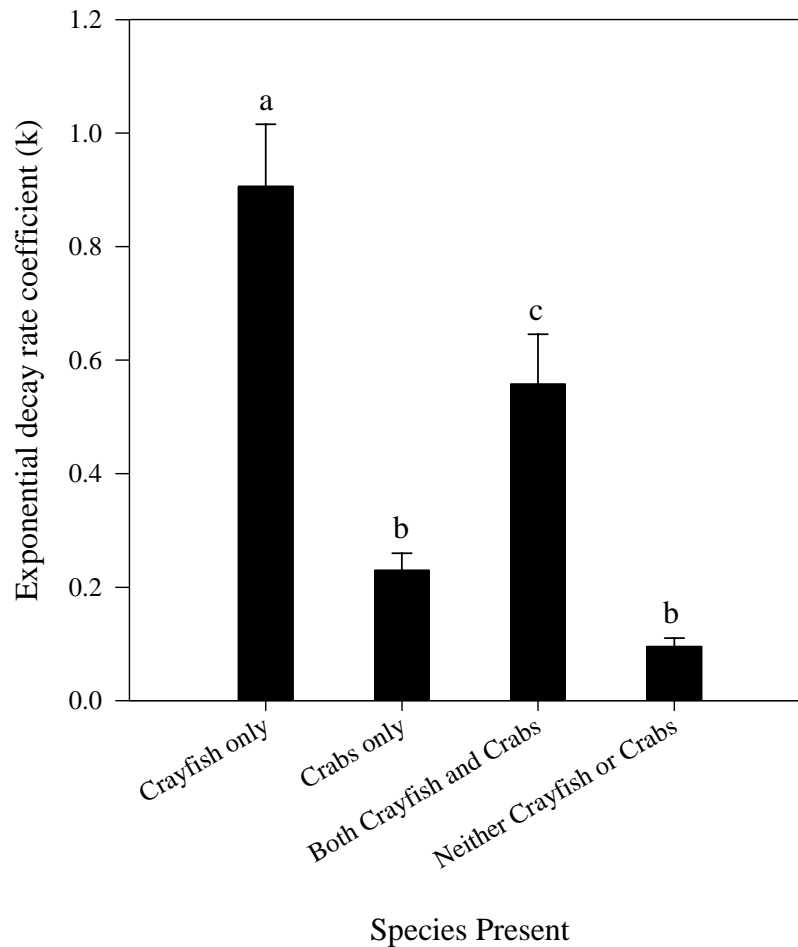


Fig. 1. Leaf litter breakdown rates (mean + s.e.) in the experimental cages, measured as exponential decay rate coefficients (k). Letters (a, b, c) indicate groups of treatments that are statistically indistinguishable from each other ($P > 0.05$).

oligochaete worms (SIMPER analysis; Table 2). In contrast, crabs had no effect on the density or multivariate structure of benthic invertebrate assemblages. However, both crabs and crayfish reduced the taxon richness of benthic invertebrates significantly, irrespective of the presence or absence of the other consumer species (ANOVA; crayfish: $F_{1,16} = 6.76$, $P = 0.019$; crabs: $F_{1,16} = 4.84$, $P = 0.043$; Fig. 2).

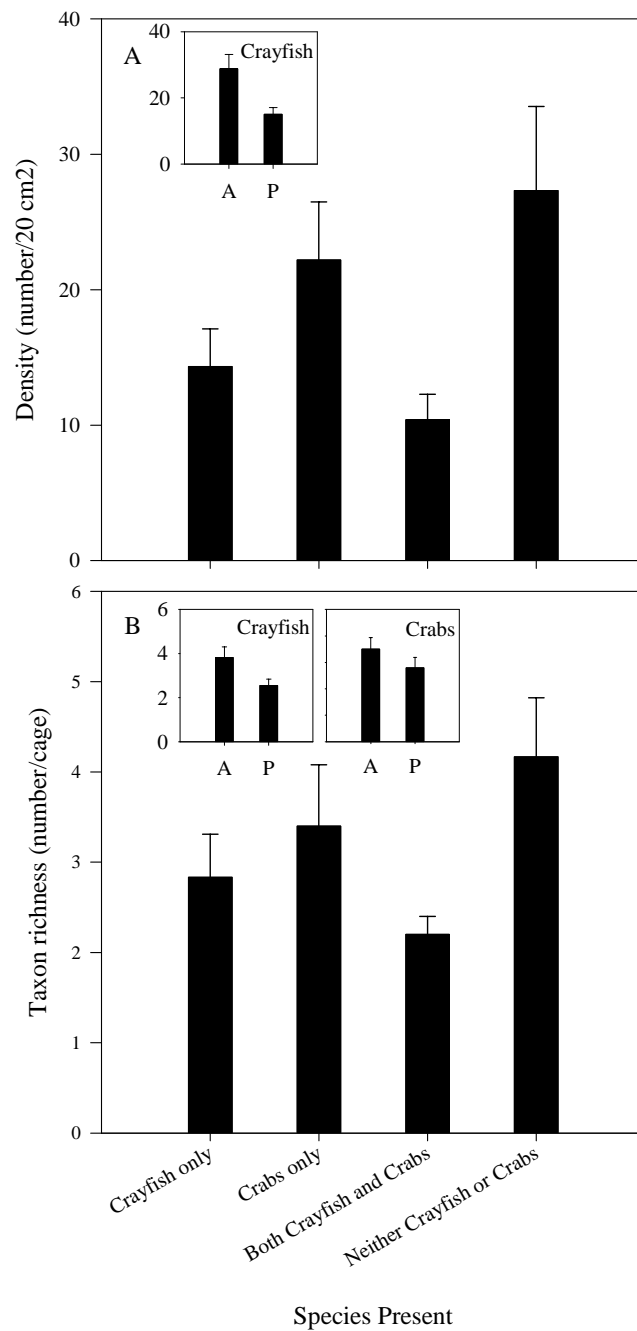


Fig. 2. Total density (A) and taxon richness (B) of benthic invertebrate assemblages in the experimental cages (mean + s.e.). Figure insets show effects of crayfish and/or crab presence (A: absent, P: present) where either had significant main effects ($P < 0.05$).

Table 2. Results of SIMPER analysis indicating the five benthic invertebrate taxa affected most strongly by the presence of crayfish.

Taxon	Mean density in crayfish absence (density m⁻²)	Mean density in crayfish presence (density m⁻²)	Contribution to dissimilarity (%)	Cumulative contribution (%)
Hydropsychidae	96.4	49.1	63.6	63.6
Oligochaeta	5.9	8.2	10.8	74.4
Heptageniidae	9.1	2.3	8.2	82.6
Baetidae	5.5	0.5	6.4	89.0
Heptageniidae	4.6	0.9	5.7	94.7

The nitrogen range (NR_b) of both crabs and crayfish was reduced significantly in the presence of the other functionally similar species (ANOVA; $F_{1,17} = 4.64$, $P = 0.046$; Fig. 3A). There was no effect of the presence of crab or crayfish on the carbon range (CR_b; Fig. 3B) of the other species. Gut content analysis revealed that the relative contribution animal and plant material to the diet of crayfish did not change in the presence of crabs (45.5 ± 17.7 % and 46.8 ± 9.8 % animal material in sympatric and allopatric populations, respectively).

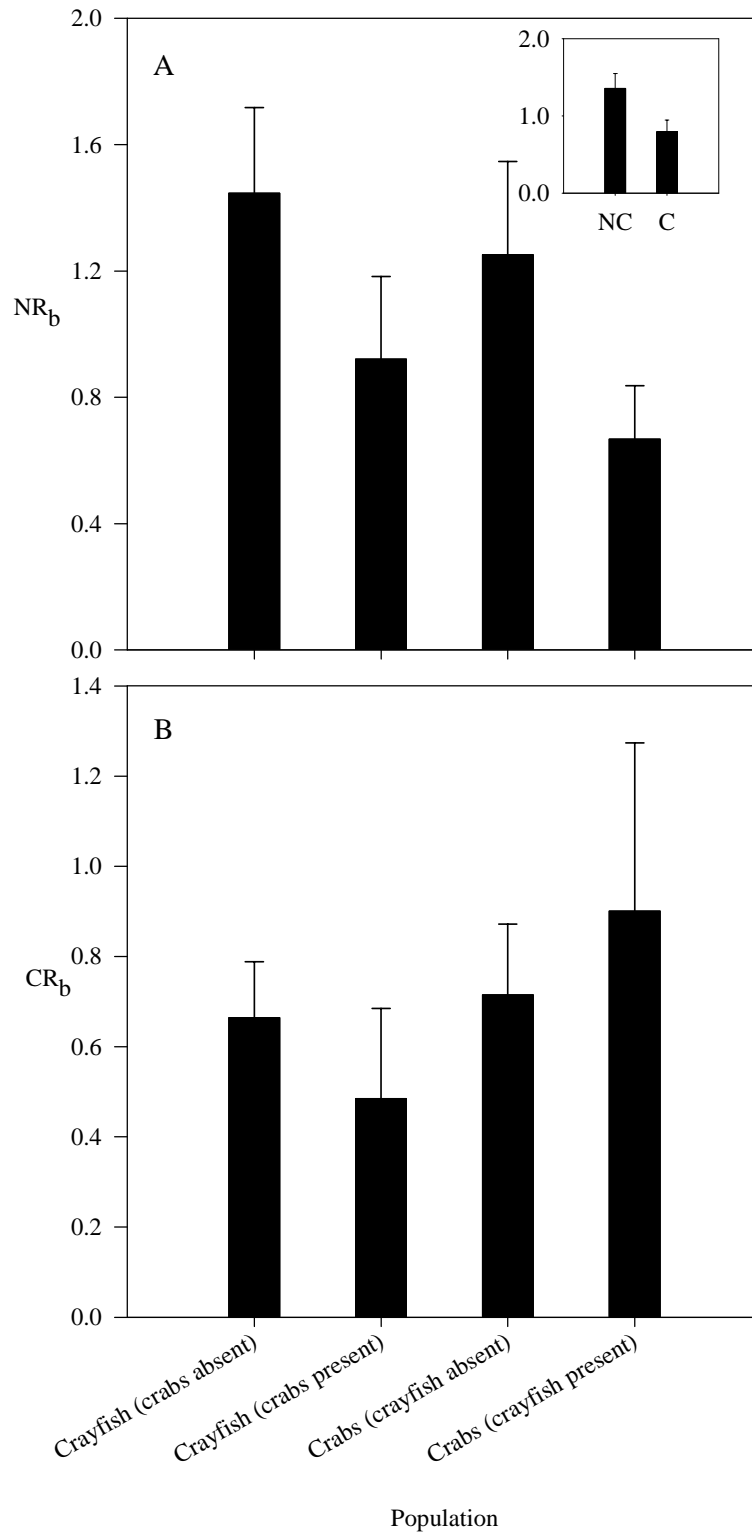


Fig. 3. The nitrogen (A) and carbon (B) range of crayfish and crabs in our experimental treatments (mean + s.e.). Figure inset shows the effect of the presence (C) and absence (NC) of interspecific competition on both species where significant main effects were detected ($P < 0.05$).

Per capita growth rates of both crabs and crayfish were reduced significantly by the presence of the other species compared with a similar biomass of conspecifics (ANOVA; $F_{1,48} = 6.64$, $P = 0.013$; Fig. 4). Both crabs and crayfish experienced similar reductions in both absolute (ANOVA; species*competition interaction: $F_{1,48} = 0.003$, $P = 0.96$) and relative ($F_{1,48} = 0.02$, $P = 0.9$) growth rates due to the presence of the other species. However, growth rates of crayfish were significantly higher than those of crabs ($F_{1,48} = 10.63$, $P = 0.002$). Thus, even though their growth rates were reduced significantly in the presence of crabs, crayfish still gained significant biomass over the course of the experiment. In contrast, however, crabs generally lost biomass when present together with crayfish (Fig. 4).

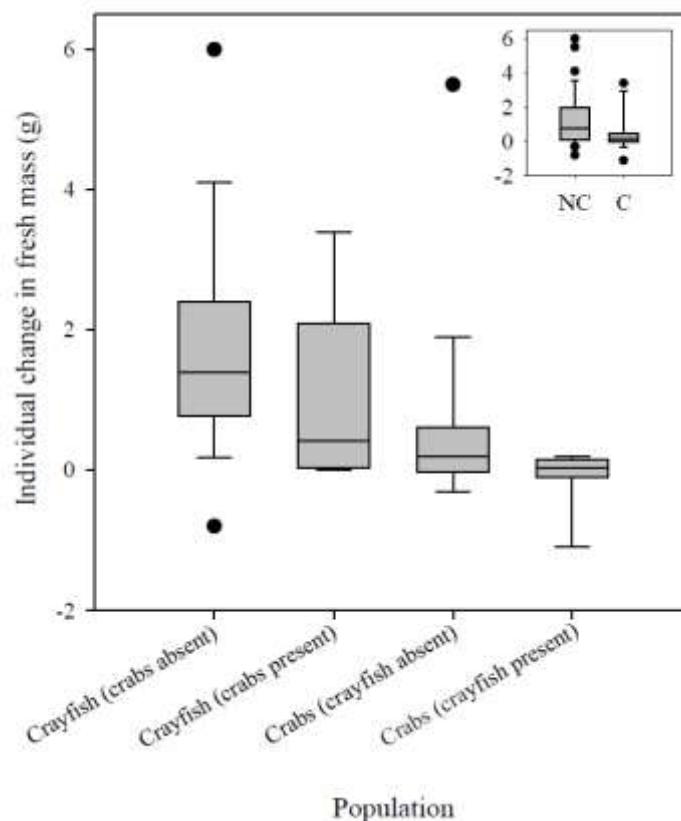


Fig. 4. Change in crayfish and crab individual body mass in our experimental treatments (median boxed by first and third quartiles). Figure inset shows the effect of the presence (C) and absence (NC) of interspecific competition on both species where significant main effects were detected ($P < 0.05$).

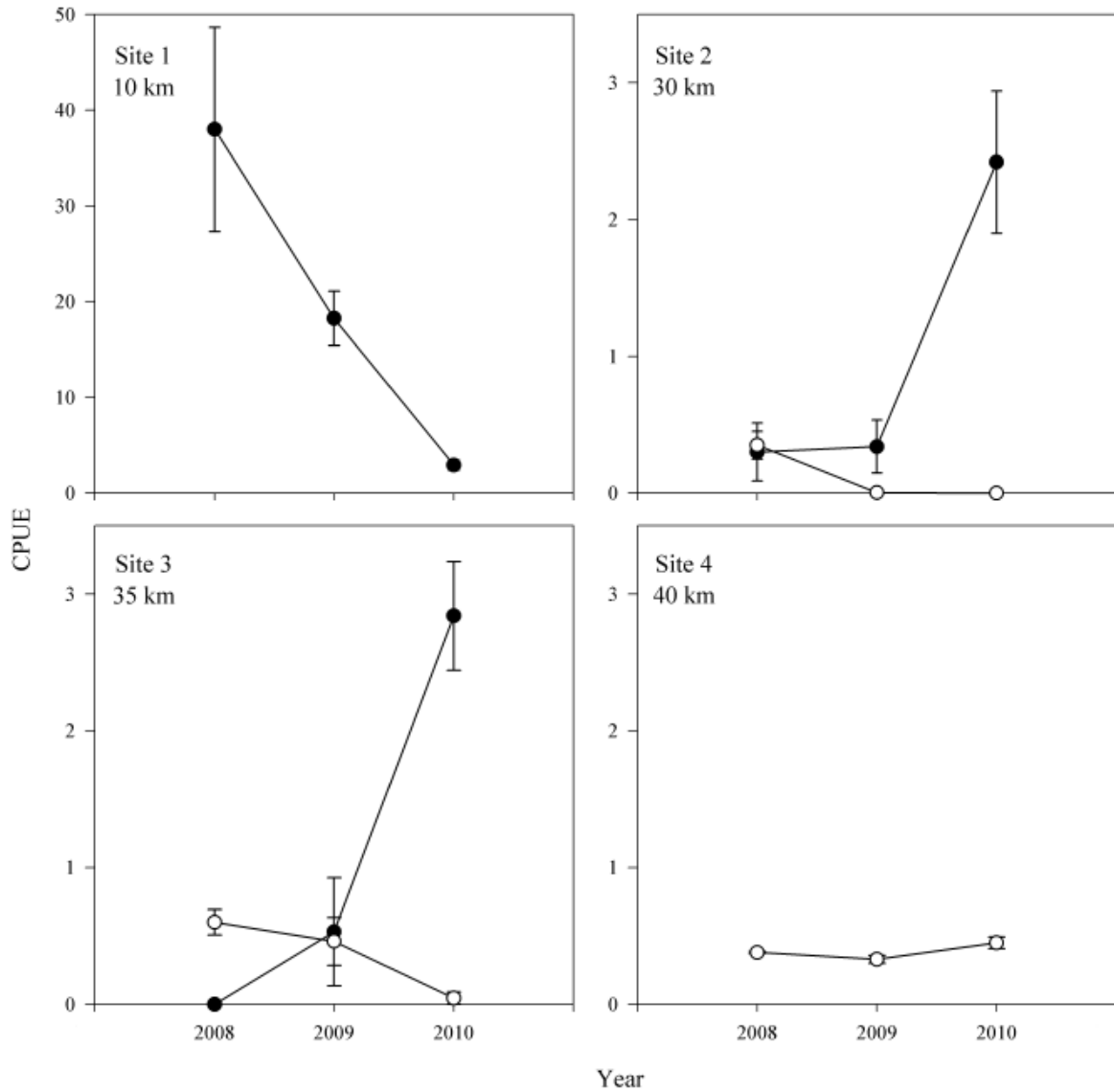


Fig. 5. Catch per unit effort (CPUE) of crayfish (black circles) and crabs (open circles) from 2008-2010 based on multiple trapping sessions at four sites (mean \pm s.e.). The distance indicated on the graphs refers to the total distance upstream from the lake. Note the different scale on the Y-axis for Site 1.

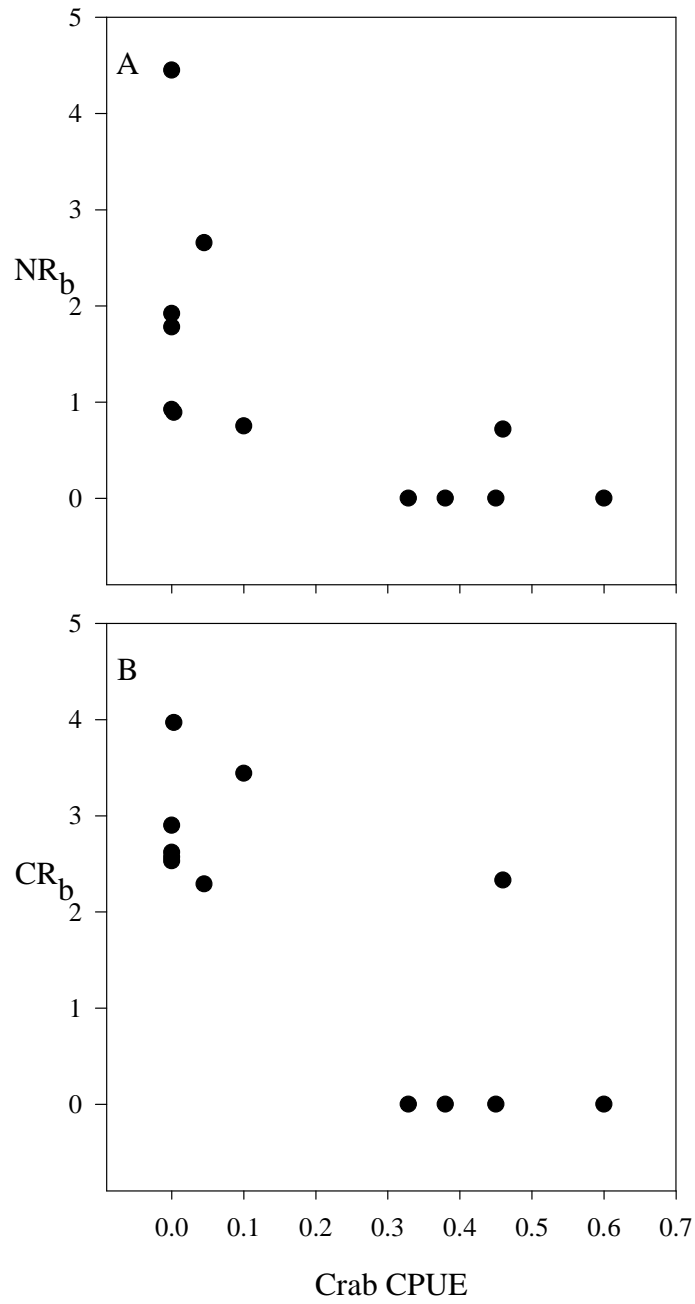


Fig. 6. Relationships between crab CPUE and the nitrogen range (A) and carbon range (B) of crayfish.

Field surveys

Crabs and crayfish were found coexisting at the two middle sampling sites (Sites 2 and 3) of the survey for at least two of the three study years (Fig. 5). Both sites showed a similar

trend from 2008 to 2010 of crayfish CPUE increasing simultaneously with a decline in crab CPUE. In fact, crabs appeared to be locally extinct at Site 2 by 2010, while the crayfish population was seven times larger compared with the previous year. In contrast, only crayfish were present at the most downstream of our survey sites, whereas only crabs were found at the most upstream location throughout the study period (Fig. 5).

We found significant inverse correlations between the abundance of crabs and both the nitrogen (NR_b; $r_{11} = -0.67$, $P = 0.018$; Fig. 6A) and carbon (CR_b; $r_{11} = -0.80$, $P = 0.002$; Fig. 6B) range of crayfish over the four sites. However, neither measure of crab trophic niche width correlated significantly with the abundance of crayfish (CR_b; $r_{11} = -0.54$, $P = 0.071$; NR_b; $r_{11} = -0.53$, $P = 0.074$).

Discussion

In 1958, Elton proposed that invasive species can out-compete similar native competitors by occupying a broader niche. This classic mechanism of invasion was exemplified recently by Olsson et al. (2009) and Ercoli et al. (2014), showing that trophic niche breadth of an invasive crayfish (*Pacifastacus leniusculus*) was double or treble that of a native crayfish (*Astacus astacus*). Indeed, it is typically reported that invasive species occupy a broad niche and suppress the niche of similar natives (e.g. Human & Gordon, 1996; Thomson, 2004). However, we found that competition between invasive crayfish and native crabs resulted in reduction in the diet breadth of both species. In our field experiments, the presence of interspecific competition between crayfish and crabs caused a decline in the nitrogen range of both species and our field surveys revealed that both the carbon and nitrogen ranges of the invasive crayfish correlated inversely with the abundance of crabs. While it has been shown that invaders can alter their feeding behaviour in the presence of native species (e.g.

Harrington *et al.*, 2009), our study provides a novel example of the constriction of the dietary niche of a successful invasive population in the presence of competition from a functionally similar native species.

Invasive species regularly outcompete and reduce the growth rate of native species (e.g. Smith, 2005; Riley, 2009; Britton *et al.*, 2011). However, we found strong and apparently symmetric interspecific competition, measured in terms of biomass change, between the invasive crayfish and native crabs. Growth rates of crayfish individuals were reduced in the presence of crabs compared to in the presence of a similar biomass of conspecifics, implying that interspecific competition was significantly greater than intraspecific competition. This is consistent with the presence of significant interspecific aggression, which we observed repeatedly during the study, causing reductions in foraging efficiency of both species when present together. Such aggressive behaviour between species is frequently greater than aggression with conspecifics (Polo-Cavia *et al.*, 2011). However, even though both absolute and relative reductions in growth rates were similar for both consumers, crayfish still exhibited positive per capita growth rates in the presence of crabs, whereas crabs generally lost mass in the presence of the invasive crayfish. This likely contributed to the decline in crab abundance at sites invaded by crayfish. Invasive crayfish have outcompeted native crayfish across North America and Europe by triggering similar declines in growth rates of natives (Hill & Lodge, 1999; Dunn *et al.*, 2009; Nakata & Goshima, 2006). However, as far as we are aware, this is the first study to test experimentally for interspecific competition between native crabs and introduced crayfish, despite numerous overlapping populations in both Europe and Africa (e.g. Barbaresi & Gherardi, 1997; Foster & Harper, 2007; du Preez & Smit, 2013).

As omnivores, crayfish and crabs can survive when a resource is limited by shifting their diet to alternative resources to compensate (Tillberg *et al.*, 2007; Grey & Jackson, 2012). This is advantageous when exploiting niches with novel resources, which may explain the successful widespread distribution of some invasive crayfish and crab species (Capinha *et al.*, 2011). Flexibility in diet choice does not necessarily imply a wide niche breadth because in some scenarios it may benefit the consumer to focus on one abundant but under-used resource (Popa-Lisseanu *et al.*, 2007). The observed reduction in crayfish and crab diet breadth in the presence of interspecific competition could be a result of increased dietary specialisation and hence a consequent decline in the variation in diet choice of individuals (Bolnick *et al.*, 2010). Alternatively, the isotopic variability could be reduced due to individuals becoming more generalist and therefore reducing the variation among the diets of each individual (Bolnick *et al.*, 2010). Examination of gut contents revealed no significant overall shift in the diets of crayfish in the presence of crabs. Therefore, that we found significant reductions in the trophic niche breadth of crayfish populations in the presence of crabs in both the field experiments and field surveys indicates that, although the overall dietary preferences of crayfish were not altered in the presence of crabs, the individuals within the crayfish population were tending to feed in a more similar manner. These results were consistent across both the field experiments and field surveys, strongly supporting the robustness and generality of this finding. The presence of the crabs appears to have triggered dietary homogenisation within the crayfish population. Crabs exhibited a significant decline in diet breadth only in the experiments and not in the field surveys. This inconsistency in results perhaps suggests less flexibility in diet choice of crabs compared to the crayfish.

Since the crayfish and crab species in our study ecosystem are both relatively large decapod Crustacea, individuals of each of the same biomass might be expected to have

similar effects on ecosystem structure and functioning. Both species are considered important shredders; however, invasive crayfish had a considerably greater influence on detrital processing in our field experiments, quadrupling rates of leaf litter breakdown compared to native crabs. Analysis of gut contents demonstrated that this was due to direct consumption.); indeed, leaf litter is considered to be an important part of crayfish diet when available (Bobeldyk & Lamberti, 2008). Some invasive predators consumer native shredders, causing a trophic cascade whereby leaf litter decomposition is reduced; crayfish, however, decouple this trophic cascade by feeding directly both on shredders and the leaf litter itself (Jackson et al. 2014).

Previous studies have found that functionally similar sympatric crab species can interact to moderate one another's impact due to consumer redundancy (Griffen & Byers, 2008). However, we found intermediate leaf litter breakdown rates in the combined species treatment in our field experiment suggesting there was no consumer interaction, as was also found for pairs of crayfish in the United Kingdom (Jackson *et al.*, 2014). We also found that the invasive crayfish significantly altered benthic community structure and approximately halved invertebrate densities compared with the native crabs. These latter findings are consistent with previous studies on invasive crayfish in Europe (Stenroth & Nyström, 2003; Haddaway *et al.*, 2012; Moorhouse *et al.*, 2013). In contrast, native crabs had no effect on the structure or density of benthic assemblages, which were similar to those in the experimental units containing no decapod consumers. In spite of the similarities between these decapod crustaceans, results from our field experiments suggest that replacement of the native crabs by the invasive crayfish will considerably alter both the structure and functioning of African river ecosystems. Moreover, these shifts in ecosystem structure and functioning may even have facilitated indirectly the replacement of the native crabs by the crayfish by reducing

suitable habitat for the crabs (Parker *et al.*, 1999) by, for example, removing leaf litter as refugia. Such changes also have the potential to alter key ecosystem services such as fishery production and water quality, both of which are extremely important in this semi-arid region of East Africa. Many river crabs in East Africa are already highly endangered (Cumberlidge *et al.*, 2009) and local biodiversity will only become increasingly impaired as crayfish spread or are introduced further in both rivers and lakes (Lodge *et al.*, 2005).

In contrast to classic invasion theory, here we have shown that competition with a native species can drive niche constriction in invader populations without impairing invasion success. Interactions between native crabs and invasive crayfish caused a decline in the dietary niche width of both species but this only resulted in a reduction in performance in the native crabs. We conclude that both environmental context and species interactions need to be considered in order to predict invasion potential and the impacts of invasive species on ecosystems.

Acknowledgements

We thank the National Council for Science and Technology, Kenya for permission (NCST 5/002/R/020-D), Dr David Harper and Velia Carn for logistical assistance and the Campbell-Clauses for hospitality. Reuben Ngeete, Phil Sanders, Jessica Hamill, George Gasston and Jessica Jackson were invaluable in the field. Prof Roger I. Jones and Dr Pavel Kratina provided useful feedback on the manuscript. MCJ was sponsored by the Percy Sladen Memorial fund.

References

- Alexander, M., Dick, J.T.A., Weyl, O.L.F., Robinson, T.B. & Richardson, D.M. (2014) Existing and emerging high impact invasive species are characterized by higher functional responses than natives. *Biology Letters*, 10, 20130946.
- Anderson, M.J., Gorley, R.N. & Clarke, K.R. (2008) *PERMANOVA+ for PRIMER: A guide to software and statistical methods*. PRIMER-E, Plymouth, U.K.
- Araújo, M.S., Bolnick, D I. & Layman, C.A. (2011) The ecological causes of individual specialisation. *Ecology Letters*, 14, 948–958.
- Barbaresi, S. & Gherardi, F. (1997) Italian freshwater decapods: exclusion between the crayfish *Austropotamobius pallipes* (Faxon) and the crab *Potamon fluviatile* (Herbst). *Bulletin français de la pêche et de la pisciculture*, 70, 731-747.
- Bobeldyk, A.M. & Lamberti, G.A. (2008) A Decade after Invasion: Evaluating the Continuing Effects of Rusty Crayfish on a Michigan River. *Journal of Great Lakes Research*, 34, 265-275.
- Bolnick, D.I., Ingram, T., Stutz, W.E., Snowberg, L.K., Lau, O.L. & Paull, J.S. (2010) Ecological release from interspecific competition leads to decoupled changes in population and individual niche width. *Proceedings of the Royal Society B-Biological Sciences*, 277, 1789-1797.
- Britton, J.R., Cucherousset, J., Grey, J. & Gozlan, R.E. (2011) Determining the strength of exploitative competition from an introduced fish: roles of density, biomass and body size. *Ecology of Freshwater Fish*, 20, 74-79.
- Capinha, C., Leung, B. & Anastacio, P. (2011) Predicting worldwide invasiveness for four major problematic decapods: an evaluation of using different calibration sets. *Ecography*, 34, 448-459.
- Chapin, F.S., Zavaleta, E.S., Eviner, V.T., Naylor, R.L., Vitousek, P.M., Reynolds, H.L., Hooper, D.U., Lavorel, S., Sala, O.E., Hobbie, S.E. & others (2000) Consequences of changing biodiversity. *Nature*, 405, 234–242.

Clark, K.R. & Warwick, R.M. (2001) *Changes in marine communities: an approach to statistical analysis and interpretation*. PRIMER-E, Plymouth, UK.

Cumberlidge, N. 2008. *Potamonautes loveni*. In: IUCN 2013. IUCN Red List of Threatened Species. Version 2013.2. <www.iucnredlist.org>. Downloaded on 17 March 2014.

Cumberlidge, N., Ng, P.K.L., Yeo, D.C.J., Magalhães, C., Campos, M.R., Alvarez, F., Naruse, T., Daniels, S.R. & Esser, L.J. (2009) Freshwater crabs and the biodiversity crisis: importance, threats, status, and conservation challenges. *Biological Conservation*, 142, 1665–1673.

Dobson, M., Magana, A., Mathooko, J.M. & Ndegwa, F.K. (2002) Detritivores in Kenyan highland streams: more evidence for the paucity of shredders in the tropics? *Freshwater Biology*, 47, 909–919.

du Preez, L. & Smit, N. (2013) Double blow: Alien crayfish infected with invasive temnocephalan in South African waters. *South African Journal of Science*, 109, 20130109.

Dudgeon, D. (1999) *Tropical Asian streams: Zoobenthos, ecology and conservation*. Hong Kong University Press, Hong Kong.

Dunn, J.C., McClymont, H.E., Christmas, M. & Dunn, A.M. (2009) Competition and parasitism in the native White Clawed Crayfish *Austropotamobius pallipes* and the invasive Signal Crayfish *Pacifastacus leniusculus* in the UK. *Biological Invasions*, 11, 315-324.

Elton, C.S. (1958) *The ecology of invasions by plants and animals*. Methuen, London.

Ercoli, F., Ruokonen, T.J., Hämäläinen, H. & Jones, R.I. (2014) Does the introduced signal crayfish occupy an equivalent trophic niche to the lost native noble crayfish in boreal lakes? *Biological Invasions*, Online first (DOI: 10.1007/s10530-014-0645-x).

Fantle, M.S., Dittel, A.I., Schwalm, S.M., Epifanio, C.E. & Fogel, M.L. (1999) A food web analysis of the juvenile blue crab, *Callinectes sapidus*, using stable isotopes in whole animals and individual amino acids. *Oecologia*, 120, 416-426.

- Foster, J. & Harper, D. (2007) Status and ecosystem interactions of the invasive Louisianan red swamp crayfish *Procambarus clarkii* in East Africa. *Biological Invaders in Inland Waters: Profiles, Distributions and Threats*. (ed. by F. Gherardi), pp. 91-101. Springer, Netherlands.
- Fry, B., Baltz, D.M., Benfield, M.C., Fleeger, J.W., Gace, A., Haas, H.L. & Quinones-Rivera, Z.J. (2003) Stable isotope indicators of movement and residency for brown shrimp (*Farfantepenaeus aztecus*) in coastal Louisiana marshscapes. *Estuaries*, 26, 82-97.
- Gherardi, F. & Acquistapace, P. (2007) Invasive crayfish in Europe: the impact of *Procambarus clarkii* on the littoral community of a Mediterranean lake. *Freshwater Biology*, 52, 1249-1259.
- Grey, J. & Jackson, M.C. (2012) 'Leaves and eats shoots': direct terrestrial feeding can supplement invasive red swamp crayfish in times of need. *PloS one*, 7, e42575.
- Griffen, B.D. & Byers, J.E. (2008) Community impacts of two invasive crabs: the interactive roles of density, prey recruitment and indirect effects. *Biological Invasions*, 11, 927-940.
- Gurevitch, J. & Padilla, D.K. (2004) Are invasive species a major cause of extinctions? *Trends in Ecology & Evolution*, 19, 470-474.
- Haddaway, N.R., Wilcox, R.H., Heptonstall, R.E.A., Griffiths, H.M., Mortimer, R.J.G., Christmas, M. & Dunn, A.M. (2012) Predatory Functional Response and Prey Choice Identify Predation Differences between Native/Invasive and Parasitised/Unparasitised Crayfish. *Plos One*, 7, e32229.
- Harper, D.M., Smart, A.C., Coley, S., Schmitz, S., de Beaugard, A.C.G., North, R., Adams, C., Obade, P. & Kamau, M. (2002) Distribution and abundance of the Louisiana red swamp crayfish *Procambarus clarkii* Girard at Lake Naivasha, Kenya between 1987 and 1999. In *Lake Naivasha, Kenya* (pp. 143-151). Springer Netherlands.
- Harrington, L.A., Harrington, A.L., Yamaguchi, N., Thom, M.D., Ferreras, P., Windham, T.R. & Macdonald, D.W. (2009) The impact of native competitors on an alien invasive: temporal niche shifts to avoid interspecific aggression? *Ecology*, 90, 1207-1216.

Jackson et al.

Hieber, M. & Gessner, M.O. (2002) Contribution of stream detritivores, fungi, and bacteria to leaf breakdown based on biomass estimates. *Ecology*, 83, 1026-1038.

Hill, A.M. & Lodge, D.M. (1999) Replacement of resident crayfishes by an exotic crayfish: the roles of competition and predation. *Ecological Applications*, 9, 678-690.

Hooper, D.U., Chapin, F.S., Ewel, J.J., Hector, A., Inchausti, P., Lavorel, S., Lawton, J.H., Lodge, D.M., Loreau, M., Naeem, S., Schmid, B., Setälä, H., Symstad, A.J., Vandermeer, J. & Wardle, D.A. (2005) Effects of biodiversity on ecosystem functioning: a consensus of current knowledge. *Ecological Monographs*, 75, 3-35.

Human, K.G. & Gordon, D.M. (1996) Exploitation and interference competition between the invasive Argentine ant, *Linepithema humile*, and native ant species. *Oecologia*, 105, 405-412.

Hutchinson, G.E. (1957) The multivariate niche. In: Cold Spr. Harb. Symp. Quant. Biol, pp. 415-421.

Inouye, B.D. (2001) Response surface experimental designs for investigating interspecific competition. *Ecology*, 82, 2696-2706.

Jackson, A.L., Inger, R., Parnell, A.C. & Bearhop, S. (2011) Comparing isotopic niche widths among and within communities: Bayesian analysis of stable isotope data. *Journal of Animal Ecology*, 80, 595-602.

Jackson, M.C. & Grey, J. (2013) Accelerating rates of freshwater invasions in the catchment of the River Thames. *Biological Invasions*, 15, 945-951.

Jackson, M.C. & Britton, J.R. (2013) Stable isotope analyses indicate trophic niche overlap of invasive *Pseudorasbora parva* and sympatric cyprinid fishes. *Ecology of Freshwater Fish*, 22, 654-657.

Jackson, M.C., Jackson, A.L., Britton, J.R., Donohue, I., Harper, D. & Grey, J. (2012) Population-level metrics of trophic structure based on stable isotopes and their application to invasion ecology. *PLoS One*, 7, e31757.

Jackson, M.C., Jones, T., Milligan, M., Sheath, D., Taylor, J., Ellis, A., England, J. & Grey, J. (2014) Niche differentiation among invasive crayfish and their impacts on ecosystem structure and functioning. *Freshwater Biology*, Online first (DOI: 10.1111/fwb.12333).

Jeffrey, S.W. & Humphrey, G.F. (1975) New spectrophotometric equations for determining chlorophylls a, b, c 1 and c 2 in algae, phytoplankton and higher plants. *Biochemie und Physiologie der Pflanzen*, 167, 191-194.

Klose, K. & Cooper, S.D. (2012) Contrasting effects of an invasive crayfish (*Procambarus clarkii*) on two temperate stream communities. *Freshwater Biology*, 57: 526–540.

Lancaster, J., Dobson, M., Magana, A.M., Arnold, A. & Mathooko, J.M. (2008) An unusual trophic subsidy and species dominance in a tropical stream. *Ecology*, 89, 2325-2334.

Larson, E.R. & Olden, J.D. (2012) Using avatar species to model the potential distribution of emerging invaders. *Global Ecology and Biogeography*, 21, 1114-1125.

Layman, C.A., Quattrochi, J.P., Peyer, C.M. & Allgeier, J.E. (2007a) Niche width collapse in a resilient top predator following ecosystem fragmentation. *Ecology Letters*, 10, 937-944.

Layman, C.A., Arrington, D.A., Montaña, C.G. & Post, D.M. (2007b) Can stable isotope ratios provide for community-wide measures of trophic structure? *Ecology*, 88, 42-48.

Lodge, D.M., Rosenthal, S.K., Mavuti, K.M., Muohi, W., Ochieng, P., Stevens, S.S., Mungai, B.N. & Mkoji, G.M. (2005) Louisiana crayfish (*Procambarus clarkii*)(Crustacea: Cambaridae) in Kenyan ponds: non-target effects of a potential biological control agent for schistosomiasis. *African Journal of Aquatic Science*, 30, 119-124.

Lodge, D.M., Deines, A., Gherardi, F., Yeo, D.C.J., Arcella, T., Baldrige, A.K., Barnes, M.A., Chadderton, W.L., Feder, J.L., Gantz, C.A., Howard, G.W., Jerde, C.L., Peters, B.W., Peters, J.A., Sargent, L.W., Turner, C.R., Wittmann, M.E. & Zeng, Y. (2012) Global Introductions of Crayfishes: Evaluating the Impact of Species Invasions on Ecosystem Services. *Annual Review of Ecology, Evolution, and Systematics*, 43, 449-472.

Mack, R.N., Simberloff, D., Mark Lonsdale, W., Evans, H., Clout, M. & Bazzaz, F.A. (2000) Biotic invasions: causes, epidemiology, global consequences, and control. *Ecological Applications*, 10, 689–710.

Jackson et al.

- Matthews, B., Hausch, S., Winter, C., Suttle, C.A. & Shurin, J.B. (2011) Contrasting ecosystem-effects of morphologically similar copepods. *PloS one*, 6, e26700.
- McArdle, B.H. & Anderson, M.J. (2001) Fitting multivariate models to community data: A comment on distance-based redundancy analysis. *Ecology*, 82, 290-297.
- McCutchan Jr, J.H., Lewis Jr, W.M., Kendall, C. & McGrath, C.C. (2003) Variation in trophic shift for stable isotope ratios of carbon, nitrogen, and sulfur. *Oikos*, 102, 378-390.
- McIntyre, P.B. & Flecker, A.S. (2006) Rapid turnover of tissue nitrogen of primary consumers in tropical freshwaters. *Oecologia*, 148, 12-21.
- Mkoji, G.M., Hofkin, B.V., Kuris, A.M., Stewart-Oaten, A., Mungai, B.N., Kihara, J.H., Mungai, F., Yundu, J., Mbui, J., Rashid, J.R., Kariuki, C.H., Ouma, J.H., Koech, D.K. & Loker, E.S. (1999) Impact of the crayfish *Procambarus clarkii* on *Schistosoma haematobium* transmission in Kenya. *The American journal of tropical medicine and hygiene*, 61, 751-751.
- Moorhouse, T.P., Poole, A.E., Evans, L.C., Bradley, D.C. & Macdonald, D.W. (2013) Intensive removal of signal crayfish (*Pacifastacus leniusculus*) from rivers increases numbers and taxon richness of macroinvertebrate species. *Ecology and Evolution*, 4, 494-504.
- Naeem, S. (1998) Species Redundancy and Ecosystem Reliability. *Conservation Biology*, 12, 39-45.
- Nakata, K. & Goshima, S. (2006) Asymmetry in mutual predation between the endangered Japanese native crayfish *Cambaroides japonicus* and the North American invasive crayfish *Pacifastacus leniusculus*: a possible reason for species replacement. *Journal of Crustacean Biology*, 26, 134-140.
- Newsome, S.D., Martinez del Rio, C., Bearhop, S. & Phillips, D.L. (2007) A niche for isotopic ecology. *Frontiers in Ecology and the Environment*, 5, 429-436.
- Nyström, P.E.R., Bronmark, C. & Graneli, W. (1996) Patterns in benthic food webs: a role for omnivorous crayfish? *Freshwater Biology*, 36, 631-646.
- O'Connor, N.E. & Bruno, J.F. (2007) Predatory fish loss affects the structure and functioning of a model marine food web. *Oikos*, 116, 2027-2038.

- Ogada, M.O. (2006) *Effects of the Louisiana Crayfish Invasion and Other Human Impacts on the African Clawless otter in the Ewaso Ng'iro Ecosystem*. PhD Thesis, Kenyatta University, Kenya.
- Olsson, K., Stenroth, P., Nyström, P. & Granéli, W. (2009) Invasions and niche width: does niche width of an introduced crayfish differ from a native crayfish? *Freshwater Biology*, 54, 1731-1740.
- Parker, I.M., Simberloff, D., Lonsdale, W.M., Goodell, K., Wonham, M., Kareiva, P.M., Williamson, M.H., Von Holle, B., Moyle, P.B. & Byers, J.E. (1999) Impact: Toward a Framework for Understanding the Ecological Effects of Invaders. *Biological Invasions*, 1, 3-19.
- Parnell, A.C., Inger, R., Bearhop, S. & Jackson, A.L. (2010) Source partitioning using stable isotopes: Coping with too much variation. *PLoS One*, 5, e9672.
- Pejchar, L. & Mooney, H.A. (2009) Invasive species, ecosystem services and human well-being. *Trends in Ecology & Evolution*, 24, 497-504.
- Peltzer, D.A., Allen, R.B., Lovett, G.M., Whitehead, D. & Wardle, D.A. (2010) Effects of biological invasions on forest carbon sequestration. *Global Change Biology*, 16, 732-746.
- Penk, M., Irvine, K. & Donohue, I. (2015) Ecosystem-level effects of a globally-spreading invertebrate invader are not moderated by a functionally similar native. *Journal of Animal Ecology*. Early view. DOI: 10.1111/1365-2656.12402.
- Polo-Cavia, N., López, P. & Martín, J. (2011) Aggressive interactions during feeding between native and invasive freshwater turtles. *Biological Invasions*, 13, 1387-1396.
- Popa-Lisseanu, A.G., Delgado-Huertas, A., Forero, M.G., Rodriguez, A., Arlettaz, R. & Ibanez, C. (2007) Bats' Conquest of a Formidable Foraging Niche: The Myriads of Nocturnally Migrating Songbirds. *Plos One*, 2, e205.
- Porter, S.D. & Savignano, D.A. (1990) Invasion of Polygyne Fire Ants Decimates Native Ants and Disrupts Arthropod Community. *Ecology*, 71, 2095-2095.
- Rahel, F.J. & Olden, J.D. (2008) Assessing the Effects of Climate Change on Aquatic Invasive Species. *Conservation Biology*, 22, 521-533.

Jackson et al.

Renai, B. & Gherardi, F. (2004) Predatory Efficiency of Crayfish: Comparison Between Indigenous and Non-Indigenous Species. *Biological Invasions*, 6, 89-99.

Ricciardi, A. (2006) Patterns of invasion in the Laurentian Great Lakes in relation to changes in vector activity. *Diversity and Distributions*, 12, 425-433.

Riley, L.A. (2009) Invasive species impact: asymmetric interactions between invasive and endemic freshwater snails. *Journal of the North American Benthological Society*, 28, 746-746.

Simon, K.S. & Townsend, C.R. (2003) Impacts of freshwater invaders at different levels of ecological organisation, with emphasis on salmonids and ecosystem consequences. *Freshwater Biology*, 48, 982-994.

Smith, K.G. (2005) Effects of nonindigenous tadpoles on native tadpoles in Florida: evidence of competition. *Biological Conservation*, 123, 433-441.

Snyder, W.E. & Evans, E.W. (2006) Ecological effects of invasive arthropod generalist predators. *Annual review of ecology, evolution, and systematics*, 37, 95-95.

Stenroth, P. & Nyström, P. (2003) Exotic crayfish in a brown water stream: effects on juvenile trout, invertebrates and algae. *Freshwater Biology*, 48, 466-475.

Svanbäck, R. & Bolnick, D.I. (2007) Intraspecific competition drives increased resource use diversity within a natural population. *Proceedings of the Royal Society B: Biological Sciences*, 274, 839-844.

Thomson, D. (2004) Competitive interactions between the invasive European honey bee and native bumble bees. *Ecology*, 85, 458-470.

Tillberg, C.V., Holway, D.A., LeBrun, E.G. & Suarez, A.V. (2007) Trophic ecology of invasive Argentine ants in their native and introduced ranges. *Proceedings of the National Academy of Sciences of the United States of America*, 104, 20856-20861.

Van Valen, L. (1965) Morphological variation and width of ecological niche. *The American Naturalist*, 99, 377-377.

Supporting Information



Figure S1. Design of the cage enclosures used in the field experiments.

Table S1. Results of PERMANOVA testing for an effect of survey site and year on the stable isotope signature of putative resources of crabs and crayfish.

Source of variation	df.	SS	MS	Pseudo- <i>F</i>	<i>P</i>
Species	5	222.1	44.42	2.04	0.1
Site	3	55.98	18.66	1.44	0.3
Year	1	0.14	0.14	0.02	0.97
Species*Site	4	50.78	12.7	3.31	0.06
Species*Year	5	54.79	10.96	2.85	0.09
Site*Year	3	11.77	3.92	1.07	0.45
Species*Site*Year	4	15.35	3.84	1.82	0.1
Residual	42	88.63	2.11		

Table S2. Mean decomposition rates ($n = 10$), benthic invertebrate densities ($n = 5$), benthic invertebrate taxon richness ($n = 5$) and chlorophyll *a* concentrations on tiles ($n = 4$) in each treatment across both field experiments.

Treatment	Decomposition (<i>k</i>)	Invertebrate density (number/20cm²)	Invertebrate taxon richness	Chlorophyll <i>a</i> (mg/cm²)
Crayfish only	0.91 ± 0.11	14.33 ± 2.79	2.83 ± 0.48	0.39 ± 0.05
Crabs only	0.23 ± 0.03	22.20 ± 4.29	3.40 ± 0.68	0.50 ± 0.11
Both Crayfish and Crabs	0.56 ± 0.09	10.4 ± 1.89	2.20 ± 0.20	0.37 ± 0.05
Neither Crayfish or Crabs	0.10 ± 0.01	27.33 ± 6.21	4.17 ± 0.65	0.42 ± 0.10

Table S3. Carbon (CR_b) and nitrogen (NR_b) ranges of crayfish and crab populations in each experimental enclosure (Field Experiment 2).

Population	CR_b	NR_b
Crayfish (crabs absent)	0.70	2.24
Crayfish (crabs absent)	0.16	1.11
Crayfish (crabs absent)	0.83	1.65
Crayfish (crabs absent)	0.71	2.14
Crayfish (crabs absent)	1.06	0.71
Crayfish (crabs absent)	0.54	0.83
Crayfish (crabs present)	0.26	0.23
Crayfish (crabs present)	1.25	1.61
Crayfish (crabs present)	0.11	0.83
Crayfish (crabs present)	0.35	1.42
Crayfish (crabs present)	0.46	0.51
Crabs (crayfish absent)	0.46	1.91
Crabs (crayfish absent)	0.90	0.51
Crabs (crayfish absent)	0.44	1.96
Crabs (crayfish absent)	1.24	1.10
Crabs (crayfish absent)	0.53	0.78
Crabs (crayfish present)	0.63	1.15
Crabs (crayfish present)	0.41	0.96
Crabs (crayfish present)	0.63	0.36
Crabs (crayfish present)	2.38	0.28
Crabs (crayfish present)	0.45	0.59

Table S4. Change in crayfish and crab individual body mass in the experimental treatments (mean \pm s.e.).

Population	Change in mass (g)
Crayfish (crabs absent)	1.70 \pm 0.35
Crayfish (crabs present)	0.97 \pm 0.43
Crabs (crayfish absent)	0.55 \pm 0.31
Crabs (crayfish present)	0.08 \pm 0.15

Table S5. Crayfish and crab catch per unit effort (CPUE), carbon (CR_b) and nitrogen (NR_b) range in the field survey.

Site	Year	Crayfish CPUE	Crab CPUE	Crayfish CR_b	Crayfish NR_b	Crab CR_b	Crab NR_b
1	2008	38.0	0	2.57	1.92	-	-
1	2009	18.3	0	2.90	0.92	-	-
1	2010	2.9	0	2.62	1.78	-	-
2	2008	0.3	0.1	3.44	0.75	1.15	0.90
2	2009	0.3	0.003	3.97	0.89	1.79	0.89
2	2010	2.4	0	2.53	4.45	-	-
3	2008	0	0.6	-	-	2.15	1.25
3	2009	0.5	0.5	2.33	0.72	2.04	1.77
3	2010	2.8	0.05	2.29	2.66	1.11	1.48
4	2008	0	0.4	-	-	2.36	0.65
4	2009	0	0.3	-	-	2.02	0.81
4	2010	0	0.5	-	-	4.14	1.86