Landscape issues for the macrofauna in temperate urban mangrove forests

Michelle N. Yerman[1] and Pauline M. Ross[2]

[1]Current address: Division of Invertebrate Zoology, Australian Museum, 6 College Street, Sydney, 2010.

[2]Centre for Landscape and Ecosystems Management, University of Western Sydney Locked Bag 1797, Penrith DC 1797.

Corresponding Author e-mail: michelleyerman@crustacea.net

ABSTRACT

Estuarine habitats along Australia's temperate shores generally comprise saltmarsh, mangrove forests and seagrass habitats. In urban areas these habitats have been progressively fragmented due to human population increase and industrial expansion. Saltmarshes are particularly vulnerable to urban expansion because of their close proximity to densely populated areas. However, there is limited understanding of what effect the reclamation of saltmarsh habitats has on the macrofauna in adjacent mangrove forests. We examined the importance of saltmarshes on adjacent mangrove forests at nine locations on the Parramatta River, Sydney, New South Wales. The habitats examined consisted of mangrove forests with and without an adjacent saltmarsh habitat. The diversity and abundance of macrofauna were sampled during spring 1999 and summer 2000. Overall, there was a trend for the diversity of macrofauna to be greater in mangrove forests with an adjacent saltmarsh compared to those with an adjacent park or bund wall. Macrofaunal diversity was 36% lower in mangrove forests without adjacent saltmarsh habitats. In addition, the diversity of macrofauna in mangrove forests adjacent to a saltmarsh showed the least variability, while those adjacent to a bund wall showed the greatest variability. This study has shown that the diversity and abundance of macrofauna in urban mangrove forests was correlated with the adjacent habitat, thus it is important to conserve remnant patches of saltmarsh in our urban environment.

Key words: Diversity, fragmentation, habitat linkage, invertebrates, macrofauna, mangrove forest, reclamation, saltmarsh, urbanisation

Introduction

The coastal estuaries of temperate southeast Australia are generally comprised of saltmarshes, mangrove forests and seagrass habitats (Clarke and Hannon 1967; McGuinness 1989; Adam 1990; Morrissey 1995). Of these habitats, temperate urban saltmarshes are disappearing rapidly. It has been estimated that there is only 1.5% of the national total of saltmarsh (354 km^2, Saintilan and Williams 2000) remaining in NSW. Those that remain in urban areas are rapidly deteriorating because of their location in the upper intertidal zone where development pressure from industry and housing is greatest. This is of concern because the saltmarshes are one of the rarest and least studied habitat types (Fairweather 1990; Adam 1996) and contain the greatest biodiversity and highest incidence of species endemic to Australia (Adam 1990, 1996; Bridgewater 1985; SOER 1997).

Compared to saltmarsh, the total area of mangrove forests is decreasing at a less rapid rate (Saintilan and Williams 1999) because they are closer to the water and require greater infilling (McGuinness 1989). Reclamation is not the only factor responsible for the decline in saltmarsh areas. It is exacerbated by the invasion of the grey mangrove *Avicennia marina*. Although the reasons for this are unknown, it could be part of the natural cycle of mangrove transgression or it may correspond to patterns of rising sea level in eastern Australia (Saintilan and Williams 2000). Whatever the cause, there is a need to assess the role of temperate urban saltmarshes in estuarine function.

Generally, the structure and role of estuarine habitats have been examined separately (saltmarshes: Clarke 1983; Adam 1990, 1996; mangroves: Skilleter 1996; Chapman and Underwood 1997; Kelaher *et al.* 1998a, b; Skilleter and Warren 2000; seagrasses: Posey 1988; Kemp 1989; Cambridge and Hocking 1997; Smith *et al.* 1997). As a consequence, there are limited data on how they interact and function together. However, investigations overseas of the consequences of saltmarsh loss on adjacent seagrass bed fauna found that when saltmarsh was adjacent to seagrass beds, pinfish *Lagodon rhomboides* moved between the two habitats, had greater growth and were more abundant than in seagrass beds without adjacent saltmarsh (Irlandi and Crawford 1997). Thus there is evidence to predict that the absence of saltmarsh in temperate urban NSW may decrease the biodiversity, abundance, growth rate and movements of organisms living in adjacent mangrove forests. The type of structure that replaces saltmarsh may also change biodiversity and abundance in adjacent mangrove forests.

The most common barriers used in urban areas are concrete retaining walls (bund walls) and parks. The interface between parks and mangroves is mainly grass (*Agropyron repens*, *Chloris truncata*, *Paspalum dilatatum*) and these areas may have different levels of nutrients and sediments, compared to one adjacent to a bund wall. However, the bund wall may alter the hydrology of the mangrove forest and impact on larval supply.

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The aim of this study was to test the prediction that the absence of saltmarsh and the type of structure replacing saltmarsh (park versus bund wall) will be correlated with a reduction in macrofauna diversity in adjacent mangrove forests.

Methods

This study was undertaken in the Parramatta River (33°51′ S, 151°16 ′ E), within Sydney. Mangrove forests that line the river edge are mostly comprised of the grey mangrove tree *A. marina* while adjoining saltmarsh is comprised of *Sarcocornia quinqueflora, Sueda australis, Sporobolus virginicus* var. *minor, Juncus kraussii, Cotula coronopifolia, Samulos repens and Selleria radicans* (Adam 1990, 1996). Moving landwards, the saltmarsh gives way to *Casuarina glauca* in some areas. Within the mangrove forest, aerial roots of *A. marina,* algae on the pneumatophores, *Bostrychia-Caloglossa* and algae on the sediment, *Catnella nipae* (King and Wheeler 1985; Hogarth 1999) are present. Along with the sediment, algae provide a habitat for crabs, *Sesarma erthrodactyla,* and gastropods *Assiminea buccinoides, Ophicardelus quoyi, O. sulcatus, Salintor solida, S. fragilis*, *Tatea rufilabris* and *T. huonensis*; amphipods, *Orchestiae* spp. and isopods, *Syncassidina aestuaria* (Hutchings and Recher 1982; Berents 1996).

To determine whether the diversity and abundance of macrofauna in a mangrove forest varied in relation to the adjoining habitat, three differing habitats were sampled. The first included mangrove forests where the adjacent saltmarsh habitat and, in some instances, the *C. glauca* forest, was intact (Yaralla Bay, Brays Bay and Glades Bay). The second habitat type included mangrove forests where the adjacent saltmarsh habitat had been reclaimed (approximately 50 years earlier, Lynch *et al.* 1976) and replaced with a grassy park (Settlers Park, Hughes Ave and Ermington). The third habitat type included mangrove forests where the adjacent saltmarsh habitat had been reclaimed (approximately 50 years earlier, Lynch *et al.* 1976) and replaced with a bund wall (George Kendall Reserve, Majors Bay and Eric Primrose Park). Nine mangrove forests were sampled (three of each habitat type), distributed along the Parramatta River. Locations were selected based on whether they were accessible, of adequate size and type of adjacent reclamation. It is known that the abundance and diversity of gastropods varies depending on where in the mangrove forest they are sampled (Kaly 1988; Underwood and Barrett 1990). To ensure that a similar tidal elevation was sampled at each location, tidal heights were measured when the absolute high tide was between 1.4 and 1.5 m. To do this, wooden stakes were placed every four metres from the landward to mid-zones, at two sites in the mangrove forest, at each of the nine locations. In the landward zone, the high tide level ranged from 5 to 18 cm (in absolute terms) above the mangrove forest floor, and in the mid-zone, the high tide level ranged from 30 to 45 cm above the mangrove forest floor.

Sampling was undertaken in the mid to landward zone at each of two sites in each location during spring 1999 and summer 2000. Six randomly placed quadrats (0.32 cm x 0.32 cm) were sampled in each site. The use of six quadrats ensured a low mean to standard error ratio (Kaly 1988; McGuinness 1990). In each quadrat, the numbers of gastropods (*A. bucccinoides*; *O. quoyi*; *O. sulcatus*, *S. fragilis*; *S. solida*; *T. rufilabris*) were counted on the surface of sediments, together with a count of the number of holes of the crab *Heloecius cordiformis* (crab burrow numbers are directly correlated to crab numbers, Warren and Underwood 1986; Warren 1990). The remaining macrofauna, including gastropods, amphipods, isopods and dipteran insect larvae living in the algae of the pneumatophores and sediment, were quantified by collecting the algae and returning it to the laboratory for sorting. A knife was then used to scrape off the upper 1 cm of sediment above the nutritive root layer. Leaf litter was also collected, by removing all leaves and twigs. This follows a modified, but similar procedure, previously used to estimate epifaunal macroinvertebrate densities (Skilleter 1996; Chapman 1998; Kelaher *et al*. 1998 a).

In the laboratory, each sample of leaf litter, algae and sediment was fixed in 7% formalin/seawater and dyed using Rose Bengal. The samples were washed through a 350µm sieve, the macrofauna removed with the aid of a magnifying lamp, and all macrofauna were counted and identified to the lowest taxonomical resolution feasible.

To determine differences in the biomass of leaf litter and algae, the litter and algal samples were placed in aluminium trays, weighed and dried in an oven at 60°C for 48 hours.

To determine differences in soil moisture in mangrove forests and adjacent saltmarsh, park or bund wall, about 15 mL of sediment was removed, weighed and placed in an oven (heated to 60°C) in aluminium trays for 48 hours. The moisture content was then measured using the method of Rayment and Higgins (1992).

To determine if any differences existed in the concentration of chlorophyll between mangrove forests with adjacent saltmarsh, park or bund wall, the concentration of chlorophyll on the surface of the sediment was measured. Approximately 2 g of the top 1 mm layer of sediment was removed. The samples were placed into centrifuge tubes containing 5 ml of 100% acetone with a pinch of $MgCO_3$ and immediately stored on ice. Within 6 hours of collection, the chlorophyll was extracted in 90% acetone by grinding the sample in a mortar and then refrigerated overnight. The sediment samples were centrifuged at 90% for 10 minutes, the optical densities of the supernatant were read at 664, 647 and 630 nm, then converted to concentration of total chlorophyll (a, b and c), expressed as µg chlorophyll/g dry weight of sediment (*cf*. Jeffrey and Humphrey 1975).

Differences in the structure of assemblages of macrofauna among habitats were determined using ANOVA and non-metric multi-dimensional scaling ordination (nMDS, Clarke and Warwick 1994) on square root transformed data using the Bray-Curtis similarity measure (PRIMER v5, Clarke and Gorley 2001). The statistical significance among locations was analysed using analysis of similarity (ANOSIM). Matching of biotic to environmental patterns was achieved using log transformed data for number of pneumatophores and soil moisture with the BIO-ENV routine (Clarke and Ainsworth 1993). The relative dispersion of each sample was calculated, where greater values correspond to greater differences between samples (Clarke and Warwick 1994).

Figure 1. Saltmarsh replaced with a bund wall structure adjacent to the mangrove forest at Eric Primrose Park.

Figure 2. Saltmarsh replaced with a grassy park adjacent to the mangrove forest at Settlers Park.

Figure 3. Saltmarsh adjacent to the mangrove forest at Yaralla Bay.

Figure 4. *Salinator solida* on surface algae in the mangrove forest.

Figure 5. *Ophicardelus* spp. on the trunk of a mangrove tree.

Photos by M. Yerman.

Results

Twenty-five taxa were identified including gastropods (*A. tasmanica*, *O. ornatus*, *O. quoyi*, *T. huonensis, T. rufilabris, B. aurutum*, *S. solida*), amphipods: gen. nov. sp. nov., *Melita plumulosa*); bivalve (*Arthritica helmsi*), isopods (*Sphaeroma quoyana, Syncassidina aestauria*) polychaete (*Ceratonereis aequistis*), crab (*Heliocius cordiformis*), oligochates and dipteran insect larvae. Overall, there was a trend for the diversity of macrofauna to be greater in mangrove forests with an adjacent saltmarsh compared to those with an adjacent park or bund wall (Figures 2, 3). The analysis of variance showed a three-way interaction at the level of site (nested in treatment and location) in spring ($P < 0.01$; Table 1) and summer ($P < 0.05$; Table 2).

In spring and summer, the assemblages differed significantly among mangrove forests with an adjacent saltmarsh, park or bund wall (Figures 4, 5). The trends in the diversity of the macrofauna in mangrove forests with adjacent saltmarsh were significantly different to the diversity in mangrove forests with adjacent park or bund wall (spring, $R = 0.353$, $P<0.0001$;summer, $R = 0.229$, $P<0.0001$).

The amount of algae on the pneumatophores, total amount of algae (on pneumatophores and sediment), soil moisture, concentration of chlorophyll and the numbers of pneumatophores was correlated weakly with the macrofaunal assemblages (spring, BIOENV: $\rho_w = 0.29$; summer, BIOENV: $\rho_w = 0.79$). These results suggested that these factors were responsible for differences in macrofauna assemblages.

The greatest variability in faunal assemblages was in mangrove forests with an adjacent bund wall (1.34 relative dispersion). Mangrove forests that were adjacent to a park showed less variability (0.68-0.78 relative dispersion), and mangrove forests adjacent to a saltmarsh habitat showed the least variability (0.93-0.98 relative dispersion; Figures 3, 5).

Discussion

Mangrove forest with adjacent saltmarsh had higher biodiversity than when adjacent to human structures (park and bund walls). This reduction in species diversity adjacent to such structures may have been induced by the lack of nutrients, in the absence of saltmarsh (Congdon and McComb 1980; Clarke 1983), which caused a decrease in total algal biomass (Yerman 2003). Larval supply and settlement of gastropods may also differ in these habitats without adjoining saltmarshes. The larval stages of aquatic organisms are important in determining adult distribution patterns and abundance in mangrove forests (Grosberg 1982; Bingham 1992; Ross 2001; Satumanatpan and Keough 2001). It has also been shown that when mangrove forests are absent, larval supply patterns become inconsistent (Satumanatpan and Keough 2001). Macrofauna larvae that enter mangroves without suitable habitat in which to settle suffer high mortality.

Macrofauna assemblages also varied among mangrove forests with different adjacent reclamation. When there were adjacent to saltmarsh, macrofauna variability was least. In addition, those with an adjacent park displayed greater macrofauna variability and those when there was an adjacent bund wall displayed greatest variability in spring and summer. Some studies have proposed that anthropogenic disturbances may alter temporal and spatial variability of macrofauna assemblages (eg. Underwood 1991, 1993) and it has been proposed that increased variability may be an important determinant of stressed macrofaunal assemblages, (Warwick and Clarke 1993). For example, in disturbed locations, Warwick and Clarke observed increased variability in the assemblages of meiobenthos, macrobenthos, corals and fish. They attributed this variability to anthropogenic habitat disturbance which resulted in an increase in the total number of individuals, the total number of species, or a change in the composition of taxa.

Differences in macrofauna densities were also apparent, with the greatest densities of macrofauna present in spring. This increase may have been due to the recruitment of the larvae. Summer decreases may have been due to desiccation (Ross 2001). Other factors, such as salinity and temperature (Hutchings and Recher 1982), may also influence diversity.

There were strong correlations between macrofauna assemblages and environmental variables (eg. pneumatophore number, macroalgae biomass on pneumatophores and the sediment surface, concentration of chlorophyll, leaf litter biomass) that probably contributed to macrofauna diversity by providing microhabitat (Dean and Connell 1987).

Previous studies on putative impacts in mangrove forests, saltmarshes and seagrasses have focused on one estuarine habitat (Irlandi and Crawford 1997). However, organisms use multiple habitat types. For example, estuarine-dependent transient juvenile fish regularly use flooded marshes (Hettler Jr. 1989; Irlandi and Crawford 1997; Thomas and Connolly 2001). These fish were in greater abundance (Hettler Jr. 1989; Irlandi and Crawford 1997; Thomas and Connolly 2001) and greater in weight (Irlandi and Crawford 1997) when they had access to an adjacent saltmarsh habitat.

Disturbances within mangrove forests have resulted in changes to macrofauna assemblages (Skilleter 1996; Kelaher *et al.* 1998a; Skilleter and Warren 2000). For example, Skilleter (1996) observed that anthropogenic damage resulted in lower macroinvertebrate diversity in the landward zones of mangrove forests. Similarly, Kelaher *et al.* (1998a) observed lower taxa diversity in association with boardwalks constructed through mangrove forests. There were also fewer pneumatophores adjacent to boardwalks, perhaps in partly due to their removal during the construction process (Kelaher *et al.* 1988a; Skilleter and Warren 2000). The reduction in pneumatophore number was correlated to, and perhaps caused, the changes to macrofauna (Skilleter and Warren 2000).

In recent years there has been an increasing appreciation of the importance of coastal wetlands (Fairweather 1990; Skilleter 1996), for example, as nursery sites for estuarine fish (Robertson and Duke 1987; Thomas and Connolly 2001), prawns (Robertson and Duke 1987) and shrimps (McIvor and Odum 1988), as a buffer zone between land and sea (Robertson and Phillips 1995), and as important sources of nutrients supplied to estuarine waters (Lee 1985; Riviera- Monroy *et al.*; Twilley 1985; Woodroffe 1985; Childers and Day 1988).

This study has shown that the diversity and abundance of macrofauna in urban mangrove forests was correlated with the adjacent habitat. Originally the adjacent habitat was saltmarsh, but since the 1950s these habitats have been reclaimed and replaced with a grassy park or bund wall. Although saltmarshes are protected by legislation in many parts of Australia, the ecology and role of these habitats in estuaries is largely unknown, while the impacts of the adjacent urban landscape pose a continual and unquantified threat to their integrity. Along with outright losses in saltmarsh habitats, there has been a reduction in size of individual saltmarshes. This study has shown, as has already been highlighted by others (Fairweather 1990; Adam 1996), that the role of temperate saltmarshes is critical for maintaining biodiversity, thus it is important to conserve remnant patches (Yerman 2003).

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