Biological Conservation 202 (2016) 10–19

Contents lists available at ScienceDirect

Biological Conservation

journal homepage: www.elsevier.com/locate/bioc

The ecological response of insectivorous bats to coastal lagoon degradation

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ARTICLE INFO

Article history:
Received 10 May 2016
Received in revised form 9 August 2016
Accepted 15 August 2016
Available online 23 August 2016

Keywords:
Contamination
Greater Sydney region
Myotis macropus
Chiroptera
Toxic metals
Trawling bat
Urbanization

ABSTRACT

Coastal lagoons provide key habitat for a wide range of biota but are often degraded by intense urbanization pressures. Insectivorous bats use these highly productive ecosystems and are likely to be impacted by their decline in quality. We compared bat activity and richness and invertebrate biomass and richness across a gradient of lagoon quality (9 lagoons) in the Greater Sydney region, Australia to determine the extent to which bats and their prey were impacted by lagoon degradation. Bats were more diverse and 19 times more active at higher quality lagoons. The trawling bat, *Myotis macropus*, was absent from all low quality lagoons, but these lagoons were used by other species such as *Miniopterus schreibersii oceanensis*. Invertebrate richness and biomass did not differ significantly across lagoon quality. We examined potential mechanisms of insectivorous bat decline at degraded lagoons by measuring toxic metal concentrations in bat fur, invertebrates and sediment. Lead and zinc were detected at environmentally significant levels in the sediments of lower quality lagoons. Furthermore, lead concentrations were 6 times the lowest observable adverse effects level for small mammals in the hair of one individual *M. macropus*. The present study demonstrates that coastal lagoons support a rich bat community, but ongoing development and pollution of these habitats is likely to negatively impact on insectivorous bat species, especially trawling species.

1. Introduction

In 2012, the world's population reached 7 billion with the United Nations projecting an increase to 8 billion by the end of the decade (Department of Economic and Social Affairs/UN, 2014). Currently, 60% of the global population lives on the coast, which accounts for only 4% of the Earth's total landmass (UNEP/UN, 2006; Bellio and Kingsford, 2013). This proportion is much higher in Australia, where 85% of people live within the coastal zone, exposing adjacent ecosystems to high levels of disturbance (Creighton et al., 2015; McGuirk and Argent, 2011). Despite this, there are no formal, global estimates for the amount of degradation coastal systems have experienced (Koutsodendris et al., 2015; Lotze et al., 2006) and regional baseline data is often lacking (Pérez-Domínguez et al., 2012). It is well established, however that when coastal ecosystems are urbanized, habitat quality degrades as native vegetation is cleared (Gedan et al., 2010), waterways are eutrophied (Lapointe et al., 2015) and water quality declines (Newton et al., 2014). This degradation in habitat quality alters the community dynamics of a wide range of biota (Grimm et al., 2008).

Insectivorous bats are sensitive to urbanization at the species and community level, and are therefore considered indicators of wider ecosystem health (Jones et al., 2009). When mature trees are cleared for urban developments, bat species lose potential roosting habitat and communities shift towards disturbance-tolerant species (Basham et al., 2011; Threlfall et al., 2012). Contamination of coastal waterways is likely to be a key issue for many insectivorous bat species as aquatic invertebrates provide a vector for sediment-based contaminants to reach higher trophic positions (Walters et al., 2008; Mendoza-Carranza et al., 2016). The risk of bats consuming contaminated prey is relatively high in coastal lagoons where high population density, industrial activity and decreased water exchange with the ocean ("flushing") can result in high contaminant concentrations (Birch et al., 2015).

Prior to urbanization, coastal lagoons likely supported diverse and abundant food webs, including bat communities, due to the productivity of these systems (Lee et al., 2006). Urban coastal lagoons, however are commonly dredged, which activates otherwise dormant contaminants allowing them to interfere with biological systems (Mertens et al., 2001, 2004). Contamination at low trophic levels can have a profound ecological impact (Oberholster et al., 2008, 2012), where

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http://dx.doi.org/10.1016/j.biocon.2016.08.014
0006-3207/

contaminants restructure prey communities by creating conditions that exclude disturbance-sensitive species (Naidoo et al., 2013; Greig et al., 2012). Insectivorous bats are exposed to direct contamination from lower trophic levels as a result of their high metabolic and food consumption rate (Sánchez-Chardi and Nadal, 2007). These shifts in prey communities may also reduce foraging opportunities for urban- or pollution-sensitive bat species while increasing them for the urban-tolerant (Abbott et al., 2009; Vaughan et al., 1996). Despite these threats, coastal lagoons remain highly productive compared to the surrounding urban landscape, providing rich foraging grounds that may buffer insectivorous bat communities against effects of urbanization. Coastal lagoons are important habitats, yet the impacts of urbanization on their bat communities are poorly known.

It is likely that the impacts of lagoon degradation on insectivorous bats will be greatest on trawling bat species as they are adapted to prey upon surface-dwelling aquatic invertebrates and small fish of waterways (Brigham et al., 1992; Frick et al., 2007). They are particularly reliant on their associated riparian zones for roosting and terrestrial foraging (Campbell, 2009, 2011). Pesticide contamination, for example, significantly lowered survivorship of juvenile *Myotis yumanensis* (Frick et al., 2007), which forage on emerging aquatic invertebrates (Brigham et al., 1992; Frick et al., 2007). Sub-lethal symptoms, such as immune system suppression have been found in *M. daubentonii* populations, contaminated with higher concentrations of organic tin compounds (used in antifouling paints for marine vessels; Lilley et al., 2013). As a result, trawling species are intrinsically linked to the state of coastal lagoons, potentially acting as indicators of both lagoon degradation and restoration.

We investigated the relationship between coastal lagoon degradation and insectivorous bats and their prey by measuring bat species richness and activity and prey richness and biomass across a degradation gradient at nine coastal lagoons in the Greater Sydney region, Australia. We further investigated contamination of bat and prey tissues, and lagoon sediments by measuring concentrations of ten metallic contaminants. We hypothesised that insectivorous bat richness and activity would be greater at higher quality lagoons than at their degraded counterparts. Similarly, prey species richness would decline with lagoon quality, though biomass may increase due to eutrophic conditions, which has been shown to stimulate invertebrate biomass in aquatic ecosystems (Dunck et al., 2015). We expected that sediments sourced from degraded lagoons would have higher concentrations of metallic contaminants than those from high quality sites, and these metallic contaminants would be present in the insectivorous bats and invertebrates at these lagoons. Finally, we hypothesised that the activity of the trawling specialist, *M. macropus* (Campbell, 2011) would decrease with lagoon degradation and that the tissues of this species would have higher concentrations of metallic contaminants than the generalist species, *Miniopterus schreibersii oceanensis*.

2. Materials and method

2.1. Study area

The study was conducted along the Australian south-east coast surrounding Sydney (Fig. 1; For Sydney during sampling period: Max. air temperature (°C) = 21.7 ± 4.7, min. air temperature (°C) = 13.8 ± 4.4, annual rain fall (mm) = 1198 ± 350; Bureau of Meteorology, 2014). Four lagoons of low-moderate quality were located within metropolitan Sydney (16–23 km from Sydney's CBD, Supp. Table 1) and included those at Curl Curl, Dee Why, Manly and Narrabeen. Much of the native vegetation surrounding these moderate and low quality lagoons was cleared for suburban development (Pressey, 1996; Roper et al., 2010). For instance, Curl Curl Lagoon is located near a former landfill site where metal-loaded ground water enters the lagoon (Supp. Table 2; Healthy Rivers Commission/NSW, 2002). The two remaining moderate lagoons, Smiths Lake and Kioloa, were the most northerly and southerly sites in the study, respectively. The vegetation at these sites was identified as sensitive to human intervention due to population growth and local economies (Benson and Picone, 2009; Healthy Rivers Commission/NSW, 2002; Roper et al., 2010). Finally, high quality sites (Marley, Meroo and Termeil lagoons) experienced high levels of environmental protection within national parks (Royal and Meroo National Parks) and have few anthropogenic pressures (Supp. Tables 1, 2).

2.2. Study design

The nine coastal lagoons were selected with reference to a 'pressure index' based on indicators such as water quality, proportion of cleared land and human population densities associated with the lagoon (Roper et al., 2010; Supp. Table 1; Fig. 1). Lagoons were classified as either 'High', 'Moderate' or 'Low' quality, with three lagoons of each quality. Coastal lagoons are defined by a salinity gradient ranging from a non-permanent channel open to the ocean (the saltwater inlet) to the freshwater tributaries (the freshwater outlet; Tagliapietra et al., 2009). We sampled bats (acoustically) and terrestrial and aquatic invertebrates at three different localities across each lagoon: the saltwater inlet (within 150 m of the inlet; mean electrical conductivity (EC) = 29.72 ± 1.62), an intermediate position on the lagoon (within 75 m of the lateral midpoint; mean EC = 22.41 ± 0.83) and freshwater outlet (on the tributary within 150 m of lagoon; mean EC = 16.75 ± 2.25).

2.3. Bat and invertebrate surveys

We surveyed insectivorous bats and invertebrates in February (late-Summer) to March (early-Autumn) 2013. One ultrasonic bat detector (Anabat, Titley Electronics) was paired with one light trap (for sampling nocturnal invertebrates) and deployed at each lagoon's saltwater inlet, freshwater outlet and intermediate locality, both sampling for an entire night. Anabats and microphones were positioned 1 m above the ground on the lagoon shore, parallel to the water's surface. Light traps were positioned at least 100 m from the Anabat to avoid influencing bat activity (Adams et al., 2005). Aquatic invertebrates were sampled at the water's edge adjacent to each lagoon locality within five separate quadrants (1 × 1 m), by agitating the lagoon bed and skimming the surface using a sweep-net. Samples from these five quadrants were then pooled. Invertebrates captured in the light traps and sweep-nets were frozen as soon as possible after collection. Invertebrates were identified to order or family, and then assigned a 'morphospecies' classification. Anabats were deployed for three consecutive nights, while light traps were deployed for two. Within each lagoon, each locality was sampled concurrently to reduce variability of weather on bat and insect activity.

Bat calls were identified using an automated key specific to the Sydney region (B. Law unpubl. data), within the bat call identification software program 'Anascheme' (Adams et al., 2010). Calls with fewer than 3 valid pulses (pulse = minimum of 6 data points and model quality of ≥0.8) were not analysed by Anascheme. Since multiple bat species may call simultaneously, calls were only assigned to a species if >50% of pulses within the sequence were attributed to that species and only calls with a minimum of three pulses classified to the same species were identified. The key grouped all steep linear calls of *M. macropus* and *Nyctophilus* spp. together and these calls were checked manually to confirm identifications. Species that have been infrequently recorded in the study area (e.g. *Chalinolobus dwyeri*) were also confirmed manually.

2.4. Sample collection and preparation for toxic metal analysis

We collected sediment, invertebrate and bat hair samples to investigate the concentration of metals within lagoon food webs. One benthic sediment sample (0 to 3 cm depth) was collected from the freshwater outlet of each lagoon. Only the freshwater outlet was sampled as

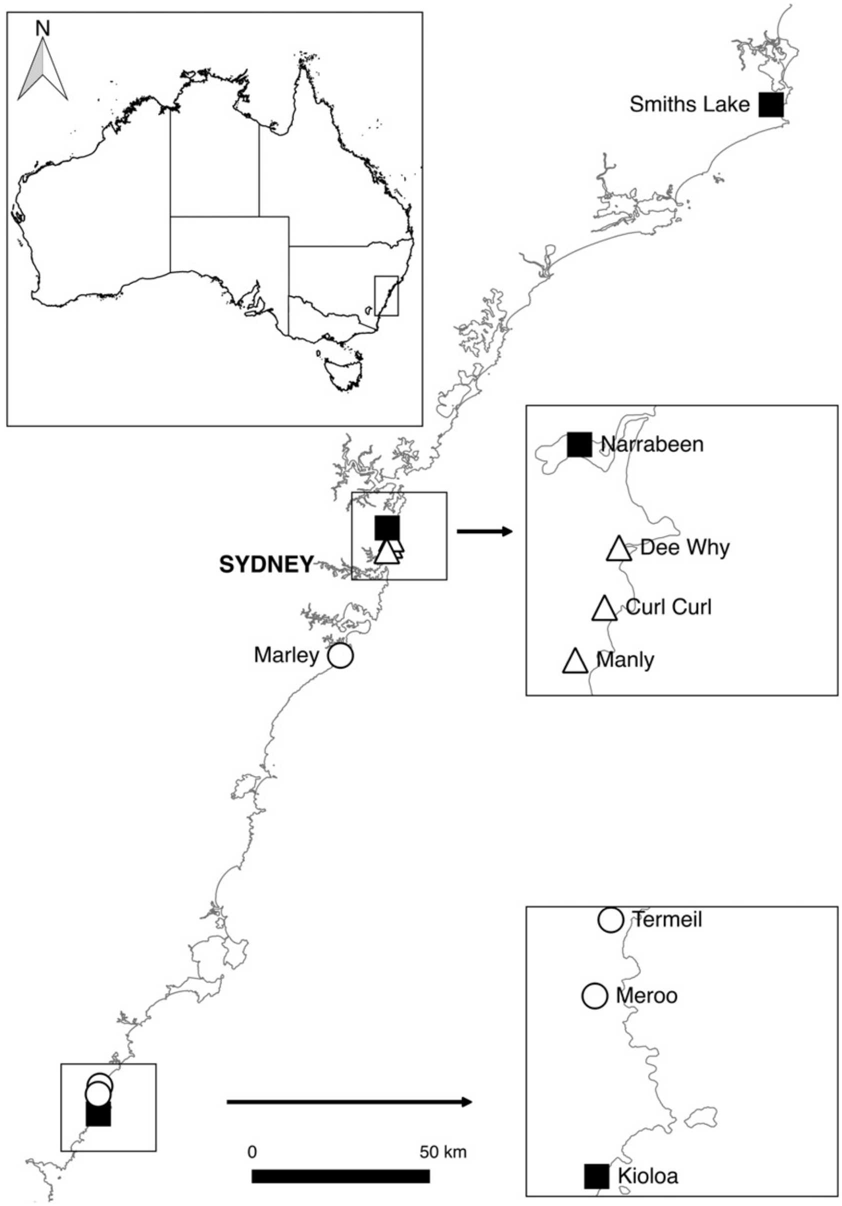

Fig. 1. Map of lagoons (triangles) sampled on the south-east coast of Australia. Lagoon quality is shown by white circles (high), black squares (medium) and white triangles (low).

contaminants are able to bond to fine clay particles and therefore this location was expected to show the highest levels of contamination (Fernandes et al., 2007; Peng et al., 2009). Sediment samples (n = 9; 3 per lagoon quality) were frozen as soon as possible after collection (Supp. mat. 3).

Invertebrate specimens were sub-sampled from freshwater outlet aquatic invertebrate samples and frozen at −30 °C soon after collection. Only one representative from each of the key aquatic invertebrate families sampled (i.e. Culicidae, Corixidae and Notonectidae) was selected for analysis, due to budget constraints. Aquatic invertebrate results were then pooled across these families (n = 3 per site). Invertebrate, sediment and hair samples from bats (see below) were dried at 55 °C for 48 h. Where necessary, invertebrates were homogenized whole, while shell and wings of larger specimens were removed, and only their soft, internal tissue processed.

M. macropus (aquatic specialists) and *M. s. oceanensis* (generalists) were selected as the 'focal' bat species for toxic metal analysis. *M. macropus* were absent from poor quality lagoons and, *M. s. oceanensis* were not captured at high quality lagoons (see Results). Therefore, *M. macropus* was sampled from lagoons of High (Termeil and Meroo; n = 6 bats) and Moderate quality (Narrabeen, Smiths Lake and Kioloa; n = 6 bats), and *M. s. oceanensis* (n = 6 per lagoon quality) was sampled from sites of Moderate (Narrabeen, Smiths Lake and Kioloa) and Low qualities (Curl Curl and Dee Why). Insectivorous bats were captured in harp traps or directly from known roosts using a hand-net in February (late-Summer) to March (early-Autumn). Upon capture, individuals had 5–10 mm of hair from their back-hind region removed with clippers. The hair was stored in polypropylene vials and frozen at −30 °C.

2.5. Toxic metal analysis

Bat hair, invertebrate and sediment samples were analysed for metal concentration using inductively coupled plasma mass spectroscopy and methods described in Supp. mat. 3. Metal concentrations were reported in mg/kg. Ten metallic contaminants were chosen for detection and included Al, As, Cd, Co, Cr, Cu, Fe, Ni, Pb and Zn. Concentrations that were below this method's detection limit were recorded as zero (Supp. Table 3). Sediment concentrations were compared to the Australian and New Zealand Guidelines for Fresh and Marine Water Quality (ANZECC and ARMCANZ, 2000), which identifies a low (trigger value) Interim Sediment Quality Guideline (ISQG) and a high ISQG, corresponding to concentrations which were occasionally and frequently associated with adverse biological effects, respectively. Where metals were not listed in the ANZECC and ARMCANZ (Al, Co & Fe), we compared metal concentrations in sediments to background values reported for the region (Supp. mat. 2). Similarly, as toxicity reference values are not widely available for wildlife tissues (Mayfield and Fairbrother, 2013), we used available risk levels and drew remaining comparisons from the literature (Supp. mat. 2).

2.6. Statistical analysis

We modelled the response of bats (total activity, species richness, *M. macropus* activity and *M. s. oceanensis* activity) and invertebrates (terrestrial invertebrate biomass and richness, aquatic invertebrate biomass and richness) to lagoon quality and locality using linear mixed effects models constructed using the package *lme4* (v1.1-7; Bates et al., 2014). All response variables except bat richness and aquatic invertebrate richness were log transformed ($\log(x + 1)$) to satisfy the assumption of normality. Log-transformed response variables in linear models are considered robust for testing statistical significance of regression coefficients (Ives, 2015). Lagoon quality (high, moderate, low) and locality (inlet, intermediate, outlet) were included as fixed factors, and random factors were used to account for temporal correlation among multiple nights at each site and spatial correlation among multiple sites at each lagoon. Significance of fixed effects was tested using the Chi squared (X^2) statistic to compare a full model to a model without the variable to be tested. Significance of different levels of each fixed factor was tested using Satterthwaites approximation in the *lmerTest* (v2.0-6) package (Kuznetsova et al., 2012).

We modelled the relationship between bats and their prey separately, looking for changes in the relationship across different lagoon qualities using an interaction term. As terrestrial invertebrates were only sampled for two nights, the third night of bat sampling was omitted in this analysis. We did not investigate the relationship between bats and aquatic prey because of low sample sizes and catch yields. We used a base model including lagoon quality, locality and prey richness and biomass (separately) as fixed effects, with lagoon as a random effect. We then compared a null model (with lagoon quality only as a main effect) to an interaction model (with lagoon quality as a main effect and an interaction effect with prey richness and biomass (separately)) using a Chi- squared (X^2) test to assess how lagoon quality affects the relationship between bats and their prey. Concentrations of metals across different lagoon qualities were compared graphically for sediments, invertebrates and bat hair and assessed against relevant risk levels (more details in the Supp. mat. 2/3).

Bat community differences were first assessed using non-metric multidimensional scaling (nMDS) of a similarity matrix (Bray-Curtis similarity) of activity of 16 species for each sampling night ($n = 81$) using *Primer 6* (v.6.1.15; Clarke and Gorley, 2006). Confidence interval ellipses (95%) were added to the nMDS plot for each level of lagoon quality. Drivers behind community differences were further analysed by employing a model-based analysis of multivariate abundance data in the *mvabund* (v3.9.4) package (Wang et al., 2012). We resampled at the lagoon level to account for nestedness of sampling nights within lagoons, using 999 Bootstrap iterations. Significance of the fixed effects (lagoon quality and locality) was tested using an analysis of variance, comparing a full model to a model without the variable to be tested. We expressed the contribution of each species to the variability of a community effect by calculating the proportion of total variance explained by that species. Throughout the text, means are given with ± standard errors unless otherwise specified.

Table 1
Percentage of sites where bat species were recorded ($n = 9$) and mean ± standard error number of calls per night for bat species recorded using Anabat detectors at coastal lagoons along the south-east coast of Australia.

Species	Percentage of sites where species was recorded	Mean calls per night
Chalinolobus dwyeri	44.4	0.9 ± 0.3
Chalinolobus gouldii	100	12.9 ± 2.4
Chalinolobus morio	33.3	0.3 ± 0.1
Falsistrellus tasmaniensis	55.6	0.6 ± 0.2
Miniopterus australis	77.8	1.5 ± 0.4
Miniopterus schreibersii oceanensis	100	41.8 ± 15.9
Mormopterus norfolkensis	66.7	108.7 ± 43.8
Mormopterus ridei	100	12.7 ± 2.8
Myotis macropus	66.7	19.0 ± 3.4
Rhinolophus megaphyllus	44.4	0.4 ± 0.1
Saccolaimus flaviventris	11.1	0.02 ± 0.02
Scoteanax rueppellii	33.3	0.9 ± 0.3
Scotorepens orion	33.3	1.9 ± 0.5
Tadarida australis	66.7	1.7 ± 0.4
Vespadelus darlingtoni	44.4	57.0 ± 30.7
Vespadelus vulturnus	44.4	2.4 ± 1.0
Total		262.7 ± 14.3

3. Results

Anabat detectors recorded 21, 277 bat calls from 16 species over 81 nights (262.7 ± 14.3 calls per night; Table 1). *M. norfolkensis* was the most active species with 8803 recognised calls, occurring in 67% of sites, but rarely at low and moderate quality urban lagoons. The most widespread species were *M. s. oceanensis*, *C. gouldii* and *M. ridei*, which were recorded at all lagoons (3rd, 5th and 6th most active species, respectively; Table 2). Eight bat species were not recorded at all in low quality lagoons: *C. morio*, *F. tasmaniensis*, *M. macropus*, *R. megaphyllus*, *S. flaviventris*, *Sc. rueppellii*, *Sc. orion* and *V. darlingtoni*.

3.1. The impact of lagoon degradation on insectivorous bats

Bat activity and richness were significantly associated with lagoon quality (bat richness: $X^2 = 17.917$, $p < 0.001$; total activity: $X^2 = 12.314$, $p = 0.002$). Total activity was on average 19 times higher in high quality than low quality lagoons and seven times higher in high quality than moderate quality lagoons (Fig. 2), though the difference between high and moderate quality lagoons was not significant (Table 2).

Table 2
Summaries of models relating total bat activity (calls per night), bat richness (no. species recorded per night) to Lagoon Quality (high, moderate and low) and Locality (inlet, intermediate and outlet). Lagoon is included as a random effect.

Response	Predictors	Variance	Coefficient	Sd/se[a]	Test statistic	*p*
Total activity[b]	Lagoon[c]	0.650		0.807		
	Intercept (high quality & inlet)		5.935	0.544	10.906	<0.001
	Low quality		−3.029	0.727	−4.166	0.006
	Moderate quality		−1.224	0.727	−1.683	0.143
	Intermediate		−0.274	0.308	−0.888	0.377
	Outlet		−0.082	0.308	−0.264	0.792
Bat richness	Lagoon[c]	1.413		1.189		
	Intercept (high quality & inlet)		7.79	0.796	9.777	<0.001
	Low quality		−6.296	1.067	−5.901	0.001
	Moderate quality		−1.519	1.067	−1.423	0.205
	Intermediate		0.963	0.443	2.171	0.03
	Outlet		0.111	0.443	0.251	0.803

[a] Standard deviation (sd) is shown for random effects and standard error (se) for fixed effects.
[b] Response variables log-transformed.
[c] Random factors.

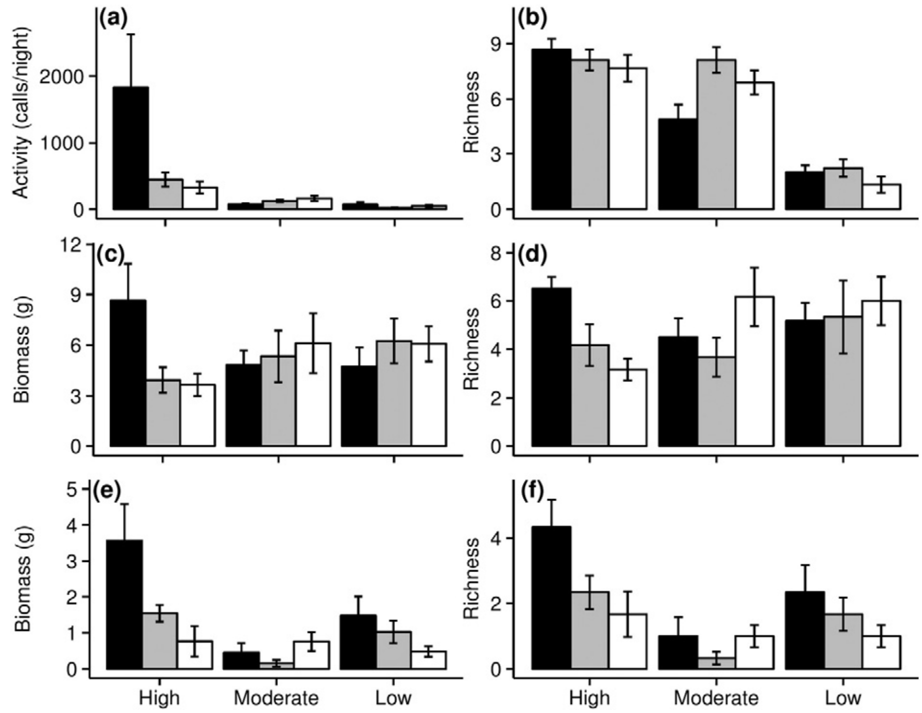

Fig. 2. Mean and standard errors of (a) totally bat activity, (b) bat richness, (c) terrestrial invertebrate biomass, (d) terrestrial invertebrate richness, (e) aquatic invertebrate biomass and (f) aquatic invertebrate richness with relation to lagoon quality (high, moderate, low) and locality (black: saltwater inlet, grey: intermediate, white: freshwater outlet).

While moderate and high quality lagoons recorded similar species richness (on average 7–8 species per night), significantly fewer species were recorded at low quality lagoons (on average 2 species per night; Fig. 2, Table 2). There was no effect of locality on bat activity, though bat richness was greatest at the locality with intermediate salinity (Table 2). Bat communities were distinct between high, moderate and low quality lagoons (Fig. 3; LR = 463.8, $p = 0.017$), but not across lagoon localities (LR = 71.288, $p = 0.795$). *M. macropus* (LR = 98.171, $p = 0.010$) was absent from low quality lagoons, and *C. gouldii* (LR = 66.229, $p = 0.015$), *M. norfolkensis* (LR = 52.524, $p = 0.048$) and *A. australis* (LR = 25.836, $p = 0.039$), were significantly less active at low quality lagoons (Fig. 4). Differences in activity of these species explained 21.2, 14.3, 11.3 and 5.6% (52.4% in total) of the community variation across lagoon qualities, respectively (Fig. 4). All four species were recorded significantly most often in high quality lagoons.

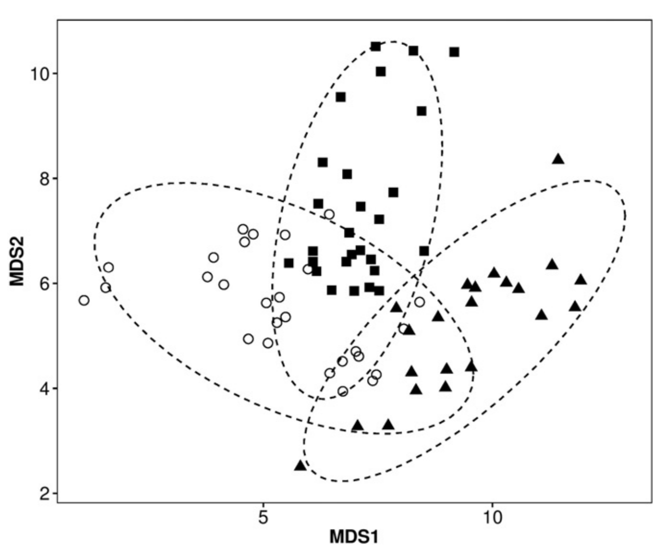

Fig. 3. nMDS plot of insectivorous bat community composition with respect to lagoon degradation (stress = 0.12) shown with 95% confidence ellipsoids for each level of lagoon quality (high: open circle, moderate: closed squares, low: closed triangles).

3.2. Relationship between bats and availability of terrestrial and aquatic prey

Contrary to our predictions, there was no effect of lagoon quality on terrestrial ($X^2 = 0.193$, $p = 0.908$) or aquatic ($X^2 = 3.587$, $p = 0.166$) invertebrate biomass, or richness (terrestrial: $X^2 = 0.671$, $p = 0.715$; aquatic: $X^2 = 3.162$, $p = 0.206$). Similarly, locality was not associated with terrestrial ($X^2 = 0.872$, $p = 0.647$) or aquatic ($X^2 = 3.349$, $p = 0.187$) invertebrate biomass or richness (terrestrial: $X^2 = 3.170$, $p = 0.205$; aquatic: $X^2 = 3.619$, $p = 0.164$). Bat richness was positively influenced by terrestrial invertebrate biomass at moderate lagoons (Fig. 5, $X^2 = 5.992$, $p = 0.050$), although this relationship was not observed at other lagoon qualities or with aquatic invertebrates. Bat activity did not interact significantly with aquatic or terrestrial biomass and richness (Supp. Table 4).

3.3. Toxic metal contamination

Though metal concentrations varied between lagoons, all metals detected were, on average, at their lowest concentrations in high quality lagoon sediments, while eight out of the nine metals detected had the highest average concentrations in low quality lagoon sediments (Fig. 6). Low quality lagoon sediments had metal concentrations ranging from four (Fe) to over twenty times higher (Cu, Pb, Zn; Fig. 6) than in high quality lagoons. Moderate and high quality lagoons did not differ significantly in metal concentrations. Arsenic, Nickel and Cobalt were not detected in any samples from high quality sites.

Sediment from two low quality lagoons (Curl Curl: 499.73 mg/kg and Manly: 609.65 mg/kg) and one moderate quality lagoon (Narrabeen: 337. 57 mg/kg) were contaminated with Zn above the ISQG-Low value (200 mg/kg). Similarly, sediment from three low quality lagoons (Curl Curl: 150.09 mg/kg, Dee Why: 146.1 mg/kg and Manly: 151.61 mg/kg) and one moderate quality lagoon (Narrabeen: 126.23 mg/kg) were concentrated with Pb above the ISQG-Low value

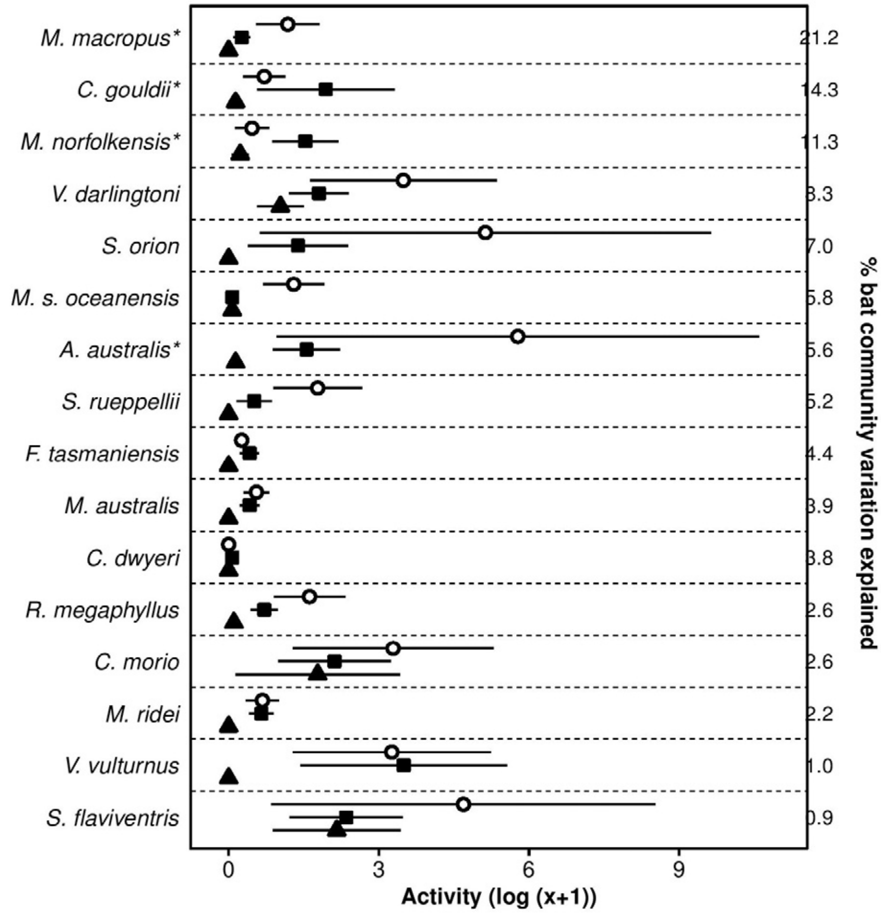

Fig. 4. Mean abundance (log(x + 1)) with standard error bars for activity of each bat species recorded in coastal lagoons of south eastern Australia for each level of lagoon quality (high: open circles, moderate: closed squares, low: closed triangles). Species whose activity differed significantly between lagoon qualities are shown with an asterisk. Numbers on the right y-axis show the percentage of the community variability between lagoons of different quality explained by each individual bat species.

(50 mg/kg). Other metallic contaminants were either not detected at environmentally significantly levels in the sediment or were below the method detection limit for ICP-MS (Cd).

Of the nine metals detected in lagoon sediments, six were detected in aquatic invertebrates (Al, Cu, Fe, Ni, Pb & Zn), of which all (excluding Ni) were also recorded in bat hair. Cr, however, was detected in an

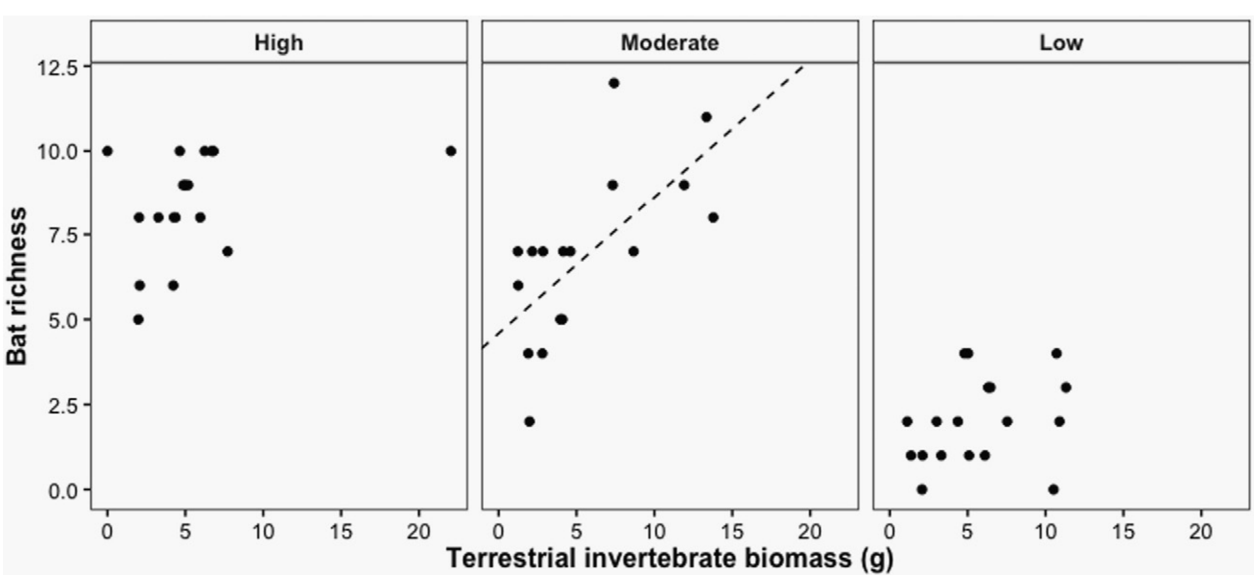

Fig. 5. The relationships between bat richness and terrestrial invertebrate biomass in coastal lagoons, showing interactions with lagoon quality, with the only significant interaction at moderate lagoons.

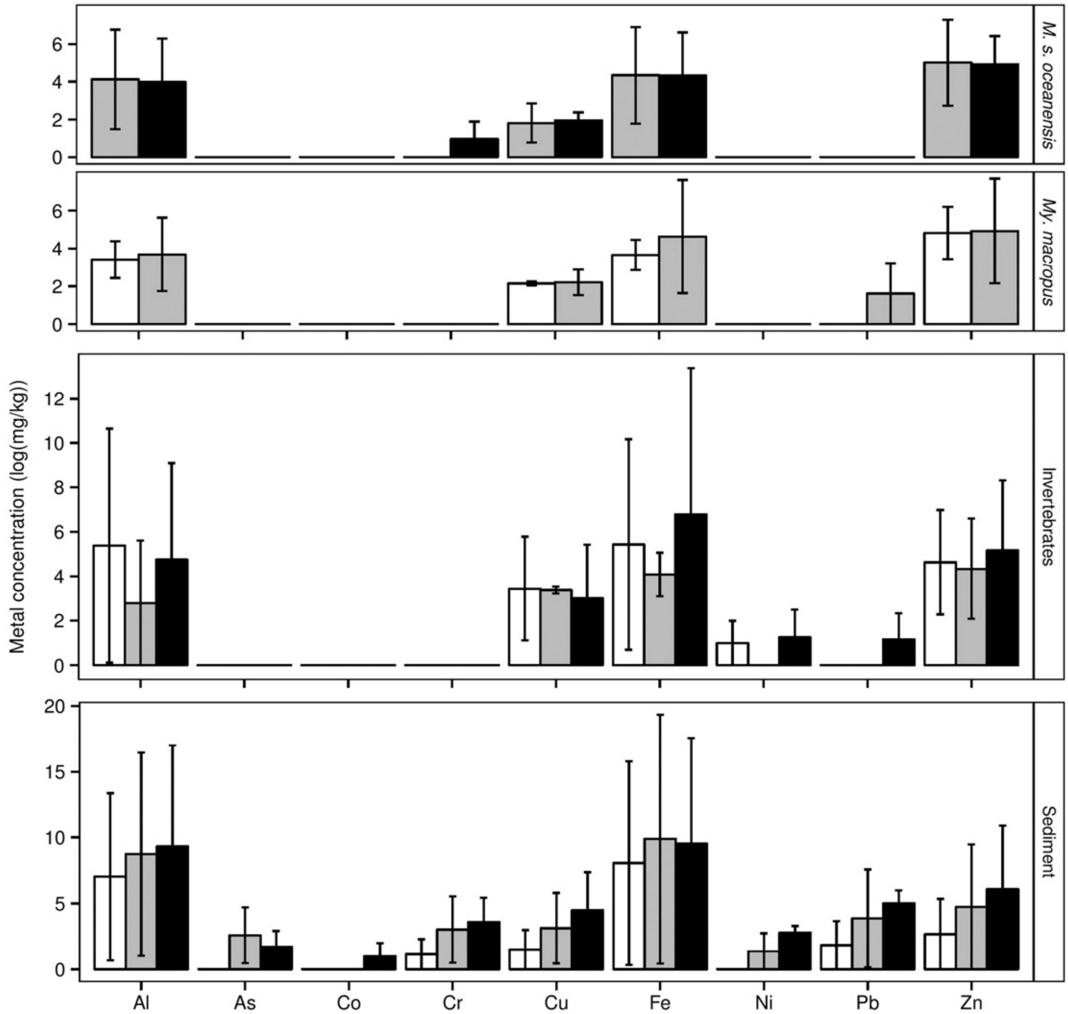

Fig. 6. Mean concentrations of metals (mg/kg) in benthic sediments, invertebrates and in bat hair of a generalist (*M. s. oceanensis*) and trawling specialist (*M. macropus*) across different site qualities (white: high, grey: moderate and low: black) within coastal lagoons of south eastern Australia. Mean concentrations and standard errors are log-transformed.

individual bat without being detected in sampled invertebrates (Fig. 6). Hair samples showed similar concentrations of metals between moderate and low quality lagoons for the generalist species (*M. s. oceanensis*). Likewise, the trawling specialist showed similar concentrations of metals between moderate and high quality lagoons (Fig. 6). However, hair from one individual *M. macropus* from a moderate quality lagoon had Pb concentrations 5 times above the lowest-adverse-observable-effects-levels (LOAEL) of Pb in small mammals (5.9 mg/kg).

4. Discussion

4.1. The impact of coastal lagoon degradation on insectivorous bat assemblages

We found that degradation of coastal lagoons had a major impact on insectivorous bat communities. Insectivorous bats were 19 times more active at undisturbed lagoons than their degraded counterparts. Coupled with this, degraded lagoons did not support the full assemblage of species found at the higher quality lagoons. Nevertheless, average nightly bat activity at low quality lagoon sites (52 $\pm$ 10 calls per night) was approximately three times that of other nearby urban sites, while species richness was four times greater (Threlfall et al., 2011, 2012). This suggests that while intense degradation and urbanization of coastal lagoons is having a negative impact on insectivorous bat assemblages, these ecosystems remain highly productive and are likely to have significant conservation value. Yet their productivity needs to be considered within the context of contamination from potentially active pollutants.

Bat community composition changed along a degradation gradient with significantly different bat assemblages detected at coastal lagoons of varying quality (Fig. 3). *A. australis*, *M. ridei*, *C. gouldii*, *M. norfolkensis* and *M. s. oceanensis* accounted for approximately 97% of all calls recorded at degraded lagoons, which is consistent with other urban studies that suggests species with traits suited to open-edge habitats with low-medium frequency calls are most active in urbanized habitats (Basham et al., 2011; Hourigan et al., 2006; Threlfall et al., 2011). As suggested by Jung and Threlfall (2016), urban-adapted species may be flexible in both foraging and roosting behaviour, allowing them to take advantage of a greater range of lagoon qualities and their open space environment. Species identified by Threlfall et al. (2012) as very sensitive to urbanization (*M. macropus*, *F. tasmaniensis* and *R. megaphyllus*) and some identified as moderately sensitive (*C. morio*, *S. rueppellii*, *S. orion*) were completely absent from degraded coastal lagoons in this study. Their absence at low quality lagoons is likely because of changes to habitat structure (e.g. lower tree cover) and reduced water quality (Bader et al., 2015; Roper et al., 2010). While some species possessed traits that were better suited to urbanized conditions (Bader et al., 2015; Threlfall et al., 2012), a marked decline in activity was still observed in 'urban-adapters'. *C. gouldii* and *A. australis* were negatively associated with lagoon degradation with higher activity recorded at pristine lagoons compared to degraded ones (Fig. 4). Similarly, *M. norfolkensis* exhibited significantly lower activity at degraded lagoons, consistent with previous findings that found urban areas were marginal habitat for this species (McConville et al., 2013; Threlfall et al., 2011), despite being ecomorphologically suited to open habitats (Bader et al., 2015).

Due to highly variable biomass data and insufficient taxonomic resolution, we did not observe a clear relationship between lagoon quality, and aquatic and terrestrial invertebrate biomass and richness. While we sampled invertebrates in accordance with standard methods (Gonsalves et al., 2013; Kashian and Burton, 2000), a small yield in aquatic invertebrates could explain an undetected relationship. Aquatic invertebrate biomass and richness peaks in early summer and may have been depleted by the time our surveys were completed (McMeans et al., 2015). Other invertebrate diversity studies, however sampled aquatic communities at a greater intensity than ours (Naidoo et al., 2013; Kashian and Burton, 2000), which may also explain the lack of a detected relationship. Nevertheless while bat richness increased with terrestrial invertebrate biomass at moderate quality lagoons (Fig. 5), this was not uniform across lagoon qualities and was not reflected with aquatic invertebrates, significantly. A higher aquatic invertebrate sampling effort at our 9 sites as well as earlier sampling, may have better demonstrated how the contamination of coastal lagoons have wider implications for terrestrial food webs (Francis and Schindler, 2009; Fukui et al., 2006).

4.2. Toxic metal pollution and coastal lagoon biodiversity

Due to limited water transference out of coastal lagoons, contaminants become trapped and can accumulate to toxic levels (Birch et al., 2015). Coastal lagoons in metropolitan Sydney are highly urbanized and therefore are exposed to metal-loaded urban run-off, incurring significant ecological responses (Birch et al., 2015). Despite analysis costs resulting in a low sample size, we detected Zn and Pb at environmentally significant levels in the sediments of degraded lagoons. These concentrations were over twenty times those measured at high quality lagoons (Fig. 6). A larger comparison of metal concentrations with international studies is available in the supplementary material (Supp. Table 4), where the present degraded lagoons consistently have some of the highest metal concentrations reported. Despite this, aquatic invertebrates from degraded lagoons had similar toxic metal concentrations to those from higher quality lagoons. Zinc concentrations in aquatic invertebrates were comparable to concentrations derived from bioassays of arthropods fed a metal loaded diet, under laboratory settings (Santorufo et al., 2012). Santorufo et al. (2012) found that metal contamination in arthropods was related to sediment exposure concentrations and often resulted in increased mortality rates. As Zn and Cu are also considered essential metals, increased concentrations may be regulated by metabolic processes in biota, thereby limiting their lethal and sub-lethal effects (Cheruiyot et al., 2013; Norwood et al., 2003). However, given the high toxic metal concentrations observed in sediments, further research with a greater sample size and taxonomic resolution of invertebrates is warranted.

Overall, metal concentrations in the hair of *M. s. oceanensis* and *M. macropus* were similar at different lagoon qualities despite the presence of contaminants in sediment (noting that *M. macropus* was absent and so not sampled at low quality lagoons). This is inconsistent with a previous study that suggests bats that forage at polluted sites have an increased exposure risk, with toxic metals accumulating in their tissues (Naidoo et al., 2013). Moreover, adverse effects such as reproductive impediment and anaemia are known to impact bats resulting in changes to community compositions (Frick et al., 2007; Hernout et al., 2013), which was not investigated in this study. Metal contamination in insectivorous bats can be influenced by seasonal moults that shed contaminants from the body and potentially strengthen an individual's resilience against contamination (Fraser et al., 2013; Lilley et al., 2013). Although, one *M. macropus* individual from a moderate site in the present study exhibited Pb concentrations substantially above the LOAEL for small mammals (Shore and Douben, 1994), consistent with our hypothesis that the trawling *M. macropus* would be exposed to greater contaminant loads than its terrestrial foraging counterpart, *M. s. oceanensis*. Also, at one low quality lagoon, Cr concentration in *M. s. oceanensis* was recorded at greater concentrations than reported on average in studies of insectivorous bats and metal contaminants (Zukal et al., 2015). Concentrations of all other metals in bat hair were below mean levels reported from other studies investigating metals in bat tissues though these levels were pooled across bat species and tissue types and many of the studies included were conducted at contaminated sites (Zukal et al., 2015).

4.3. Myotis macropus and the degradation of coastal lagoons

Due to a highly specialized feeding ecology, *M. macropus* has a distribution restricted to waterways but across a range of water qualities (Anderson et al., 2006; Campbell, 2011). We identified a threshold of lagoon quality below which *M. macropus* appears to be excluded from coastal lagoons. Degraded lagoons below this threshold are characterized by high proportions of cleared land and high human population densities (Roper et al., 2010), but also significant chemical disturbances. As an example, Manly Lagoon is considered one of the most polluted in Australia and sediment analysis detected Pb between 23 and 270 mg/kg ppm (Manly Council/NSW, 2011), consistent with concentrations in the present study (Fig. 6). Recent sediment analysis at Manly Lagoon for Zn had the lowest concentrations at 7.5 mg/kg and the highest at 460 mg/kg (Manly Council/NSW, 2011), also consistent with this study (Fig. 6). This lagoon is also regularly partially dredged which disturbs sediment layers and dormant toxic metals. These concentrations generally exceed those identified as trigger values (ANZECC and ARMCANZ, 2000) or those detected in other studies that link toxic metal concentrations to bat health (Hernout et al., 2013; Pikula et al., 2010). It is unlikely that a scarcity of roosts resulted in the absence of *M. macropus* from these habitats, as storm water drains, culverts and bridges were present and used by *M. macropus* as roosts at other lagoons. Moreover, at least two local culverts were known to be occupied by *M. s. oceanensis*, which often roosts together with *M. macropus*.

Historical pollution events may have played a role in *M. macropus*'s *decline* at these lagoons as it has done for other insectivorous bat species in other aquatic systems (Campbell, 2011; Frick et al., 2007). As key *M. macropus* prey groups (Law and Urquhart, 2000) persist at degraded lagoons, we suggest direct contamination rather than a lack of available food has likely contributed to the exclusion of *M. macropus* from degraded lagoons. Due to a reliance on water, trawling species accumulate more metals, such as Zn, Cu and Pb than bat species that can source terrestrial prey (Méndez and Alvarez-Castañeda, 2000; Naidoo et al., 2013; Pikula et al., 2010). Though not investigated here, it is also possible that *M. macropus* forages to a greater extent on fish than invertebrates in saline waters and fish can have elevated levels of metals in polluted coastal lagoons (Roach et al., 2008). Organic compounds from pesticides can also have acute and lethal effects on bats (Bayat et al., 2014).

Despite this, *M. macropus*'s persistence at moderate-high quality lagoons suggests that restoring degraded lagoons would result in significant biodiversity gains. Remediation in the form of toxic sediment removal has demonstrated a marked decline in metal concentration in trawling bats (Flache et al., 2016), but this must be balanced by activation of dormant metals if sediments are not fully removed. Lagoons currently of intermediate quality are likely to be under continuing pressure of urban development. In such situations, protection of the remaining vegetated catchment should be a priority. For more degraded lagoons, the efficacy of trees such as *Eucalyptus*, in bonding contaminants to “dead” structural tissue suggests multiple benefits of habitat restoration (Luo et al., 2016). Although not a focus of this study, these approaches may offer a long term solution to activated toxic metals and habit loss associated urban lagoons, especially for those without a wide expanse. Further development and research into holistic management strategies such as this are needed if coastal lagoons are to be conserved as a refuge for insectivorous bats and wider biodiversity.

Acknowledgements

I would like to show my respect and acknowledge the traditional custodians of the land, of elders past and present, on which my fieldwork took place. I would like to acknowledge Pittwater Council, Shoalhaven City Council and Warringah Council for providing financial support that assisted with our analyses. Finally, I would like to acknowledge the camaraderie of my peers and colleagues at the Centre for Ecosystem Science.

Appendix A. Supplementary data

Supplementary materials are available here http://dx.doi.org/10.1016/j.biocon.2016.08.014.

References

Abbott, I.M., Sleeman, D.P., Harrison, S., 2009. Bat activity affected by sewage effluent in Irish rivers. Biol. Conserv. 142, 2904–2914.

Adams, M.D., Law, B.S., French, K.O., 2005. Effect of lights on activity levels of forest bats: increasing the efficiency of surveys and species identification. Wildl. Res. 32, 173.

Adams, M.D., Law, B.S., Gibson, M.S., 2010. Reliable automation of bat call identification for eastern New South Wales, Australia, using classification trees and AnaScheme software. Acta Chiropterologica 12, 231–245.

Anderson, J., Law, B.S., Tidemann, C., 2006. Stream used by the Large-footed Myotis (*Myotis macropus*) in relation to environmental variables in northern New South Wales. Aust. Mammal. 28, 15–26.

Australian and New Zealand Environment and Conservation Council, Agriculture and Resource Management Council of Australia and New Zealand (ANZECC, ARMCANZ), 2000n. Australian and New Zealand Guidelines for Fresh and Marine Water Quality, National Water Quality Management Strategy Paper 4.

Bader, E., Jung, K., Kalko, E.K.V., Page, R.A., Rodriguez, R., Sattler, T., 2015. Mobility explains the response of aerial insectivorous bats to anthropogenic habitat change in the Neotropics. Biol. Conserv. 186, 97–106.

Basham, R., Law, B.S., Banks, P.B., 2011. Microbats in a "leafy" urban landscape: are they persisting, and what factors influence their presence? Austral. Ecol. 36, 663–678.

Bates, D., Mächler, M., Bolker, B., Walker, S., 2014. Fitting linear mixed-effects models using lme4. J. Stat. Softw. 2 (67), 1–51.

Bayat, S., Geiser, F., Kristiansen, P., Wilson, S.C., 2014. Organic contaminants in bats: trends and new issues. Environ. Int. 63, 40–52.

Bellio, M., Kingsford, R.T., 2013. Alteration of wetland hydrology in coastal lagoons: implications for shorebird conservation and wetland restoration at a Ramsar site in Sri Lanka. Biol. Conserv. 167, 57–68.

Benson, D., Picone, D., 2009. Monitoring vegetation change over 30 years: lessons from an urban bushland reserve in Sydney. Cunninghamia 11 (2), 195–202.

Birch, G.F., Gunns, T.J., Olmos, M., 2015. Sediment-bound metals as indicators of anthropogenic change in estuarine environments. Mar. Pollut. Bull. 101, 243–257.

Brigham, R.M., Aldridge, H.D.J.N., Mackey, R.L., 1992. Variation in habitat use and prey selection by Yuma bats, *Myotis yumanensis*. J. Mammal. 73, 640–645.

Bureau of Meteorology, 2014. Air Temperature and Rain Fall, Climate Data. Sydney, Australia. URL http://www.bom.gov.au/climate/data/stations/ (accessed 1.7.16).

Campbell, S., 2009. So long as it's near water: variable roosting behaviour of the large-footed myotis (*Myotis macropus*). Aust. J. Zool. 57, 89.

Campbell, S., 2011. Ecological specialisation and conservation of Australia's Large-footed Myotis: a review of trawling bat behaviour. In: Law, B., Eby, P., Lunney, D., Lumdsen, L. (Eds.), The Biology and Conservation of Australiasian Bats. Royal Zoological Society of NSW, Mosman, NSW, pp. 72–85.

Cheruiyot, D.J., Boyd, R.S., Coudron, T.A., Cobine, P.A., 2013. Biotransfer, bioaccumulation and effects of herbivore dietary Co, Cu, Ni, and Zn on growth and development of the insect predator *Podisus maculiventris* (Say). J. Chem. Ecol. 39, 764–772.

Clarke, K.N., Gorley, R.N., 2006. PRIMER v6 (United Kingdom).

Creighton, C., Boon, P.I., Brookes, J.D., Sheaves, M., 2015. Repairing Australia's estuaries for improved fisheries production – what benefits, at what cost? Mar. Freshw. Res. 66, 493–507.

Department of Economic and Social Affairs/UN, 2014. Concise Report on the World Population Situation in 2014 (New York).

Dunck, B., et al., 2015. Responses of primary production, leaf litter decomposition and associated communities to stream eutrophication. Environ. Pollut. 202 (July), 32–40.

Fernandes, C., Fontaínhas-Fernandes, A., Peixoto, F., Salgado, M.A., 2007. Bioaccumulation of heavy metals in *Liza saliens* from the Esmoriz-Paramos coastal lagoon, Portugal. Ecotoxicol. Environ. Saf. 66, 426–431.

Flache, L., et al., 2016. Reduction of metal exposure of Daubenton's bats (*Myotis daubentonii*) following remediation of pond sediment as evidenced by metal concentrations in hair. Sci. Total Environ. 547, 182–189.

Francis, T.B., Schindler, D.E., 2009. Shoreline urbanization reduces terrestrial insect subsidies to fishes in North American lakes. Oikos 118, 1872–1882.

Fraser, E.E., Longstaffe, F.J., Fenton, M.B., 2013. Moulting matters: the importance of understanding moulting cycles in bats when using fur for endogenous marker analysis. Can. J. Zool. 91, 533–544.

Frick, W.F., Rainey, W.E., Pierson, E.D., 2007. Potential effects of environmental contamination on *Yuma myotis* demography and population growth. Ecol. Appl. 17, 1213–1222.

Fukui, D., Murakami, M., Nakano, S., Aoi, T., 2006. Effect of emergent aquatic insects on bat foraging in a riparian forest. J. Anim. Ecol. 75, 1252–1258.

Gedan, K.B., Kirwan, M.L., Wolanski, E., Barbier, E.B., Silliman, B.R., 2010. The present and future role of coastal wetland vegetation in protecting shorelines: answering recent challenges to the paradigm. Clim. Chang. 106, 7–29.

Gonsalves, L., Lamb, S., Webb, C., Law, B.S., Monamy, V., 2013. Do mosquitoes in fluence bat activity in coastal habitats? Wildl. Res. 40, 10–24.

Greig, H.S., Kratina, P., Thompson, P.L., Palen, W.J., Richardson, J.S., Shurin, J.B., 2012. Warming, eutrophication, and predator loss amplify subsidies between aquatic and terrestrial ecosystems. Glob. Chang. Biol. 18, 504–514.

Grimm, N.B., Faeth, S.H., Golubiewski, N.E., Redman, C.L., Wu, J., Bai, X., Briggs, J.M., 2008. Global change and the ecology of cities. Science 319, 756–760.

Healthy Rivers Commission/NSW, 2002. Coastal Lakes: Independent Inquiry Into Coastal Lakes. Healthy Rivers Commission of New South Wales, Sydney, Australia.

Hernout, B.V., Somerwill, K.E., Arnold, K.E., McClean, C.J., Boxall, A.B., 2013. A spatially-based modeling framework for assessing the risks of soil-associated metals to bats. Environ. Pollut. 173, 110–116.

Hourigan, C.L., Johnson, C., Robson, S.K.A., 2006. The structure of a micro-bat community in relation to gradients of environmental variation in a tropical urban area. Urban Ecosyst. 9, 67–82.

Ives, A.R., 2015. For testing the significance of regression coefficients, go ahead and log-transform count data. Methods Ecol. Evol. 6, 828–835.

Jones, G., Jacobs, D.S., Kunz, T.H., Willig, M.R., Racey, P.A., 2009. Carpe noctem: the importance of bats as bioindicators. Endanger. Species Res. 8, 93–115.

Jung, K., Threlfall, C.G., 2016. Urbanisation and its effects on bats - a global meta-analysis. In: Voigt, C.C., Kingston, T. (Eds.), Bats in the Anthropocene: Conservation of Bats in a Changing World. Springer Open, Cham, Switzerland, pp. 13–33.

Kashian, D.R., Burton, T.M., 2000. A comparison of macroinvertebrates of two Great Lakes coastal wetlands: testing potential metrics for an index of ecological integrity. J. Great Lakes Res. 26 (4), 460–481.

Koutsodendris, A., Brauer, A., Zacharias, I., 2015. Ecosystem response to human- and climate-induced environmental stress on an anoxic coastal lagoon (Etoliko, Greece) since 1930 AD. J. Paleolimnol. 53 (3), 255–270.

Kuznetsova, A., Brockhoff, P.B., Christensen, R.H.B., 2012. Package "lmerTest". [WWW Document]. URL https://cran.r-project.org/web/packages/lmerTest/index.html (accessed 5.6.16).

Lapointe, B.E., Herren, L.W., Debortoli, D.D., Vogel, M.A., 2015. Evidence of sewage-driven eutrophication and harmful algal blooms in Florida's Indian River Lagoon. Harmful Algae 43, 82–102.

Law, B.S., Urquhart, C.A., 2000. Diet of the Large-Footed Myotis (*Myotis macropus*) at a forest stream roost in northern New South Wales. Aust. Mammal. 22, 121–124.

Lee, S.Y., Dunn, R.J.K., Young, R.A., Connolly, R.M., Dale, P.E.R., Dehayr, R., Lemckert, C.J., McKinnon, S., Powell, B., Teasdale, P.R., Welsh, D.T., 2006. Impact of urbanization on coastal wetland structure and function. Austral. Ecol. 31, 149–163.

Lilley, T.M., Ruokolainen, L., Meierjohann, A., Kanerva, M., Stauffer, J., Laine, V.N., Atosuo, J., Lilius, E.M., Nikinmaa, M., 2013. Resistance to oxidative damage but not immunosuppression by organic tin compounds in natural populations of Daubenton's bats (*Myotis daubentonii*). Comp. Biochem. Physiol. - C Toxicol. Pharmacol. 157, 298–305.

Lotze, H.K., Lenihan, H.S., Bourque, B.J., Bradbury, R.H., Cooke, R.G., Kay, M.C., Kidwell, S.M., Kirby, M.X., Peterson, C.H., Jackson, J.B.C., 2006. Depletion, degradation, and recovery potential of estuaries and coastal seas. Science (80) 312, 1806–1809.

Luo, J., et al., 2016. Phytoremediation efficiency OF CD by *Eucalyptus globulus* transplanted from polluted and unpolluted sites. International Journal of Phytoremediation 18 (4), 308–314.

Manly Council/NSW, 2011. Manly Lagoon Dredging Project Application for a Resource Recovery Exemption. Manly Council, Sydney, Australia.

Mayfield, D.B., Fairbrother, A., 2013. Efforts to standardize wildlife toxicity values remain unrealized. Integr. Environ. Assess. Manag. 9, 114–123.

McConville, A., Law, B.S., Mahony, M.J., 2013. Are regional habitat models useful at a local-scale? A case study of threatened and common insectivorous bats in South-Eastern Australia. PLoS One 8, e72420.

McGuirk, P., Argent, N., 2011. Population growth and change: implications for Australia's cities and regions. Geogr. Res. 49, 317–335.

McMeans, B.C., McCann, K.S., Humphries, M., Rooney, N., Fisk, A.T., 2015. Food web structure in ecosystems. Trends Ecol. Evol. 30, 662–672.

Méndez, L., Alvarez-Castañeda, S.T., 2000. Comparative analysis of heavy metals in two species of ichthyophagous bats *Myotis vivesi* and *Noctilio leporinus*. Bull. Environ. Contam. Toxicol. 65, 51–54.

Mendoza-Carranza, M., Sepúlveda-Lozada, A., Dias-Ferreira, C., Geissen, V., 2016. Distribution and bioconcentration of heavy metals in a tropical aquatic food web: a case study of a tropical estuarine lagoon in SE Mexico. Environ. Pollut. 210, 155–165.

Mertens, J., Luyssaert, S., Verbeeren, S., Vervaeke, P., Lust, N., 2001. Cd and Zn concentrations in small mammals and willow leaves on disposal facilities for dredged material. Environ. Pollut. 115, 17–22.

Mertens, J., Vervaeke, P., De Schrijver, A., Luyssaert, S., 2004. Metal uptake by young trees from dredged brackish sediment: limitations and possibilities for phytoextraction and phytostabilisation. Sci. Total Environ. 326, 209–215.

Naidoo, S., Vosloo, D., Schoeman, M.C., 2013. Foraging at wastewater treatment works increases the potential for metal accumulation in an urban adapter, the banana bat (*Neoromicia nana*). Afr. Zool. 48, 39–55.

Newton, A., Icely, J., Cristina, S., Brito, A., Cardoso, A.C., Colijn, F., Riva, S.D., Gertz, F., Hansen, J.W., Holmer, M., Ivanova, K., Leppäkoski, E., Canu, D.M., Mocenni, C., Mudge, S., Murray, N., Pejrup, M., Razinkovas, A., Reizopoulou, S., Pérez-Ruzafa, A., Schernewski, G., Schubert, H., Carr, L., Solidoro, C., Pierluigi, V., Zaldívar, J.M., 2014. An overview of ecological status, vulnerability and future perspectives of European large shallow, semi-enclosed coastal systems, lagoons and transitional waters. Estuar. Coast. Shelf Sci. 140, 95–122.

Norwood, W.P., Borgmann, U., Dixon, D.G., Wallace, A., 2003. Effects of metal mixtures on aquatic biota: a review of observations and methods. Hum. Ecol. Risk. Assess. 9, 795–811.

Oberholster, P.J., Botha, A.M., Cloete, T.E., 2008. Biological and chemical evaluation of sewage water pollution in the Rietvlei nature reserve wetland area, South Africa. Environ. Pollut. 156, 184–192.

Oberholster, P.J., Myburgh, J.G., Ashton, P.J., Coetzee, J.J., Botha, A.M., 2012. Bioaccumulation of aluminium and iron in the food chain of Lake Loskop, South Africa. Ecotoxicol. Environ. Saf. 75, 134–141.

Peng, J.F., Song, Y.H., Yuan, P., Cui, X.Y., Qiu, G.L., 2009. The remediation of heavy metals contaminated sediment. J. Hazard. Mater. 161 (2), 633–640.

Pérez-Domínguez, R., Maci, S., Courrat, A., Lepage, M., Borja, A., Uriarte, A., Elliott, M., 2012. Current developments on fish-based indices to assess ecological-quality status of estuaries and lagoons. Ecol. Indic. 23, 34–45.

Pikula, J., Zukal, J., Adam, V., Bandouchova, H., Beklova, M., Hajkova, P., Horakova, J., Kizek, R., Valentikova, L., 2010. Heavy metals and metallothionein in vespertilionid bats foraging over aquatic habitats in the Czech Republic. Environ. Toxicol. Chem. 29, 501–506.

Pressey, R.L., 1996. Ad hoc reservations: forward or backward steps in developing representative reserve systems? Conserv. Biol. 8 (3), 662–668.

Roach, A.C., Maher, W., Krikowa, F., 2008. Assessment of metals in fish from lake Macquarie, new South Wales, Australia. Arch. Environ. Contam. Toxicol. 54.2, 292–308.

Roper, T., Creese, B., Scanes, P., Stephens, K., Robert, W., Dela Cruz, J., Coade, G., Coates, B., Fraser, M., 2010. Assessing the Condition of Estuaries and Coastal Lake Ecosystems in NSW. Estuaries and Coastal Lakes, Sydney.

Sánchez-Chardi, A., Nadal, J., 2007. Bioaccumulation of metals and effects of landfill pollution in small mammals. Part I. The greater white-toothed shrew, *Crocidura russula*. Chemosphere 68, 703–711.

Santorufo, L., Van Gestel, C.A.M., Maisto, G., 2012. Ecotoxicological assessment of metal-polluted urban soils using bioassays with three soil invertebrates. Chemosphere 88, 418–425.

Shore, R., Douben, P., 1994. Predicting ecotoxicological impacts of environmental contaminants on terrestrial small mammals. In: Ware, G.W. (Ed.), Reviews of Environmental Contamination and Toxicology. Springer New York, New York, pp. 49–89.

Tagliapietra, D., Sigovini, M., Ghirardini, A.V., 2009. A review of terms and definitions to categorise estuaries, lagoons and associated environments. Mar. Freshw. Res. 60, 497–509.

Threlfall, C., Law, B.S., Penman, T., Banks, P.B., 2011. Ecological processes in urban landscapes: mechanisms influencing the distribution and activity of insectivorous bats. Ecography 34, 814–826.

Threlfall, C., Law, B.S., Banks, P.B., 2012. Sensitivity of insectivorous bats to urbanization: Implications for suburban conservation planning. Biol. Conserv. 146, 41–52.

UNEP/UN, 2006. Marine and coastal ecosystems and human well-being. WWW Document]. URL http://www.unep.org/pdf/Completev6_LR.pdf (accessed 5.6.16).

Vaughan, N., Jones, G., Harris, S., 1996. Effects of sewage effluent on the activity of bats (Chiroptera: Vespertilionidae) foraging along rivers. Biol. Conserv. 78, 337–343.

Walters, D.M., Fritz, K.M., Otter, R.R., Luther, W.M., Drive, K., 2008. The dark side of subsidies: adult stream insects export organic contaminants to riparian predators. Ecol. Appl. 18, 1835–1841.

Wang, Y., Naumann, U., Wright, S.T., Warton, D.I., 2012. mvabund - an R package for model-based analysis of multivariate abundance data. Methods Ecol. Evol. 3, 471–474.

Zukal, J., Pikula, J., Bandouchova, H., 2015. Bats as bioindicators of heavy metal pollution: history and prospect. Mamm. Biol. 80, 220–227.