MONASH University

Between two worlds:
The implications of existing between land and sea for the conservation of mangrove ecosystems

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Abstract

Mangrove forests occur in the intertidal zone spanning marine and terrestrial environments, with elements of both, yet are traditionally viewed as marine and valued for the marine ecosystem services they provide (e.g. nursery areas for fisheries). The terrestrial role mangroves play has been largely neglected, due in part to the challenges surveying them. In consequence, we know little about the terrestrial animals that occur there. In addition to ecological knowledge gaps, complex arrangements surrounding intertidal governance create challenges in managing and conserving these declining ecosystems.

My research explores mangroves as terrestrial ecosystems, evaluating existing governance structures, and assessing and filling critical knowledge gaps to support improved conservation outcomes. Specifically I aim to: 1) determine the current state of global knowledge on terrestrial vertebrates (mammals, reptiles and amphibians) in mangroves; 2) provide a field approach that overcomes challenging survey conditions related to inundation; 3) assess the value of ecological field data to inform mangrove management strategies in Australia; and 4) evaluate the adequacy of existing Australian governance structures for the protection and management of intertidal ecosystems.

I carried out a comprehensive global literature review of records of terrestrial vertebrates occurring in mangroves. I found 464 species, a fivefold increase from previous global reviews. Record origins were often unknown due to the lack of empirical data, especially about the ecology of species found in mangroves. Faunal richness is correlated with mangrove floral richness, suggesting global knowledge gaps remain. To fill these knowledge gaps, I designed a novel field approach to detect terrestrial vertebrates in mangroves. I evaluated the effectiveness and efficiency of the survey techniques at ten locations along the East coast of Australia ranging from temperate to tropical regions, making it the most extensive vertebrate fauna survey in mangroves in the world. The approach detected 30 native species never before recorded in Australian mangroves, which is a 10% increase in global knowledge, displaying similar patterns in richness, resource use

and feeding ecology as revealed in the global review. The prevalence of threatened and invasive species in mangroves support the need for specific management objectives.

To investigate the management of mangroves further I undertook a review of Australia's federal and state legislation. My research has identified that the existing inconsistent governance structures for the intertidal zone in Australia mean that mangroves would benefit most from the acknowledgement as terrestrial plants. A focus on the mitigation of terrestrial threats and keeping mangroves wholly within terrestrial protected areas rather than split across marine and terrestrial protected areas ensures their governance is more consistent. The recommendations I make for strengthening their governance have global relevance, and involve practical solutions to ensure these critical systems do not fall through the cracks in global ecosystem protection.

Given the rate of loss of mangroves, we need to urgently address these identified gaps to be able to halt their decline. This study outlines a new agenda for further research and highlights improvements to management and governance required to protect these vital forests.

Declaration

This thesis contains no material which has been accepted for the award of any other degree or diploma at any university or equivalent institution and that, to the best of my knowledge and belief, this thesis contains no material previously published or written by another person, except where due reference is made in the text of the thesis.

Signature: [redacted]

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Date: 27/09/2017

Publications during enrolment

Chapter 2 Rog S.M., Cook C.N (2017) Strengthening governance for intertidal ecosystems requires a consistent definition of boundaries between land and sea. *Journal of Environmental Management*, **196,** 694-705.

Chapter 5 Rog S.M., Cook C.N, Clarke R.H (2017) More than marine: the critical importance of mangrove forests for terrestrial vertebrates. *Diversity and Distributions* **23**, 221–230.

Thesis including published works declaration

I hereby declare that this thesis contains no material which has been accepted for the award of any other degree or diploma at any university or equivalent institution and that, to the best of my knowledge and belief, this thesis contains no material previously published or written by another person, except where due reference is made in the text of the thesis.

This thesis includes two original papers published in peer reviewed journals and no submitted publications. The core theme of the thesis is mangrove conservation. The ideas, development and writing up of all the papers in the thesis were the principal responsibility of myself, the student, working within the School of Biological Sciences under the supervision of Dr Carly Cook and co-supervision Rohan Clarke. The inclusion of co-authors reflects the fact that the work came from active collaboration between researchers and acknowledges input into team-based research.
In the case of chapter 2, 3, 4 & 5 my contribution to the work involved the following:

Thesis Chapter	Publication Title	Status	Nature and % of student contribution	Co-author name(s) Nature and % of Co-author's contribution*	Co-author(s), Monash student Y/N*
2	**More than marine: the critical importance of mangrove forests for terrestrial vertebrates.**	**Published**	Conceived and designed study, conducted review, analysed data, wrote manuscript (80%).	-Carly Cook: Helped design study, proof read and contributed to manuscript 10% -Rohan Clarke: Contributed to manuscript 10%.	No
3	A rapid survey approach for terrestrial vertebrates in flooded forests.	Not submitted	Conceived and designed study, conducted review, analysed data, wrote manuscript (77%).	-Carly Cook: Helped design study, proof read and contributed to chapter 10% -Rohan Clarke: Contributed to field design, proof read and contributed to chapter 8% -Ernest Minnema: Contributed to the field design and assisted with data collection 5%.	No
4	The value of ecological field data on terrestrial vertebrates to inform mangrove management strategies.	Not submitted	Conceived and designed study, conducted review, analysed data, wrote manuscript (85%).	-Carly Cook: Helped design study, proof read and contributed to chapter 10% -Rohan Clarke: Contributed to chapter 5%.	No
5	**Strengthening governance for intertidal ecosystems requires a consistent definition of boundaries between land and sea.**	**Published**	Conceived and designed study, conducted review, analysed data, wrote manuscript (85%).	-Carly Cook: Helped design study, proof read and contributed to manuscript 15%.	No

I have not renumbered sections of submitted or published papers in order to generate a consistent presentation within the thesis.

Student signature: [redacted] **Date:** 27/09/2017

The undersigned hereby certify that the above declaration correctly reflects the nature and extent of the student's and co-authors' contributions to this work. In instances where I am not the responsible author I have consulted with the responsible author to agree on the respective contributions of the authors.

Main Supervisor signature Carly Cook: [redacted]
Date: 27/09/2017

Secondary supervisor signature Rohan Clarke: [redacted]
Date: 27/09/2017

Signature co-author chapter 3 Ernest Minnema: [redacted]
Date: 27/09/2017

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"The soupy mud and tepid shallow waters of mangrove swamps are physically unpleasant for humans, with stifling humidity, rank smell, a constant whine of biting insects plus a very real risk of being eaten by a crocodile. Naturalists seldom venture into these areas, and no sane person would spend longer there than absolute necessary" – Wilson & Swan 2013.

…would it then make me insane, that times spent in the mangroves were among the most wonderful of my life?

It has been a great privilege to return to full-time study to pursue research of my passion that explored gaps in our knowledge of mangrove ecology and conservation. This opportunity was possible with the support of an Australian Postgraduate Award Scholarship and with financial assistance from Holsworth Research Endowment and Paddy Pallin Science Grant. Many people have contributed to this work during the last 3.5 years.

Firstly, I am indebted to my supervisor Carly Cook who believed in my ambitious field plan and who has started to grow fond of mangroves too. I am very grateful for her enthusiasm, patience and sense of humour at all stages of the project. I thank my secondary supervisor Rohan Clarke for his ecological knowledge and invaluable advice on the field design, even though this meant testing out pitfall taps in the surprisingly hard mangrove soil just a few times too many. My panel Joslin Moore, David Chapple and Bob Wong for their helpful insights during my candidature. Fiona, Barbara, Leesa and Jody for all their support with the administrative circus; without your help arranging all the permits, ethics and grants, this project would not have been impossible. Ricardo for his enthusiastic assistance to make sure my fieldwork was as safe as possible. Steward and John for their help with the field vehicles and interstate transport of the equipment. Ryan to fight off the cassowaries and being my witness for spotting the elusive tree kangaroo and Louis who kept me motivated during the very last field trip, even when knee deep in mud. All the rangers and traditional owners from the national parks allowing me to enter their valuable mangroves, Amanda for analysing the bat calls, Barbara for identifying all the animal hairs. Melbourne Museum for allowing me to look at their hundreds of stuffed rats. Faunatech for their assistance with the electronic equipment.

Julie who has been with me throughout this whole PhD rollercoaster, I will miss you. I thank Beck for proofreading my thesis and reading the word mangrove at least 810 times. Josh who emptied my brain by almost killing me with your gym trainings. Nic for snapping me out of my hermit crab mode during those critical times when I was swamped by it all. Franciscus; lunch without you will just not be the same. Lotte for your Dutch humour and unforgettable writing boot camp race. Kelsey I'll keep sending you marine job links! Marcelo for being my serious-face writing partner for chapter 2– I remember our strict morning routine with mixed fondness and terror! And all the others who made coming to university more enjoyable; Hayley, Emily, Qam, Susie, Melissa, Yussi, Helena, Marie, Ash, Rowan Cath, Nancy, Stan, Chris Johnston, Grant, Stephen, Guilliaume and Shane.

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Chapter one: Introduction

Mangroves are wetland ecosystems growing in marine and brackish environments restricted to the tidal zone between land and sea, estuaries and along rivers (Spalding *et al.*, 2010). These ecosystems can occur as far as 60km inland and reach canopy heights exceeding 30m (Spalding *et al.*, 2010). Mangrove plants are salt resistant or salt tolerant halophytic tree and shrub species uniquely adapted to harsh environments where they face regular tidal inundation of roots and sediments (Feller *et al.*, 2010). Some well-known characteristics include exposed breathing roots to grow in anaerobic sediments, salt-excreting foliage to remove excess salt from sap, and buoyant propagules to promote dispersal and establishment of new stands (Tomlinson, 2016). These specialized attributes are found amongst different species, but not all are necessarily found within one single species. Therefore, although mangroves are highly specialised and adapted to intertidal environments, these plants are not of a single family but multiple families sharing the same ecological niche (Tomlinson, 2016).

Mangroves provide a variety of ecosystem services, for example in the form of coastal protection as a supporting service; in the form of high carbon sequestration as a regulating service (Donato *et al.*, 2011; Alongi, 2014); fisheries as a provisioning service (Carrasquilla-Henao & Juanes, 2017), cultural services (Uddin *et al.*, 2013), as well as providing habitat for fauna (Nagelkerken *et al.*, 2008). Despite the values of these services that when lost would lead to the loss of US$ 2.16 billion annually by 2050 in Asia alone (Brander *et al.*, 2012), mangroves are highly threatened. The major threats include population growth and economic development (IUCN SSC Mangrove Specialist Group, 2014) leading to land conversion for aquaculture and agriculture (Primavera, 2006; Richards & Friess, 2016) and for coastal urbanisation (Macintosh, 2002). Mangroves are also threatened by generally less visible observable drivers of degradation, such as pollution (Agoramoorthy *et al.*, 2008) and by threats related to climate change due to an altered precipitation regime, increasing temperatures and sea level rise (Ward *et al.*, 2016). Though data quality is highly variable, it has been suggested that 35% of original mangrove area was lost by the end of the 20th century (Valiela et al., 2001). Mangrove loss in the early 21st century has declined from expected highs in the mid- to late 20th century (Spalding et al., 2010), with a recent global-scale remote sensing study showing that annual rates of mangrove deforestation averaged 0.2–0.7% between 2000 and 2012 (Hamilton & Casey, 2016). While it is estimated that 6.9% of global mangroves receive legislative protection (Giri *et al.*, 2011), and while protected areas may protect mangroves from habitat loss, external influences - such as pollution, sedimentation and fragmentation of the wider landscape - can still lead to degradation (UNEP, 2014, Feller et al., 2017). In addition mangrove plant species diversity is insufficiently represented by marine protected areas (Daru & le Roux, 2016). The high level of global loss and low level of effective protection urges to review current mangrove conservation measures.

Mangrove ecosystems span marine and terrestrial environments, yet they have been traditionally conceptualized as marine environments (e.g. providing nursery areas for important commercial fisheries (Mumby *et al.*, 2004)). Historically faunal surveys in mangroves have focused on the macro benthos (Lee, 2008) and biological relationships between fish and marine invertebrate diversity in mangroves and marine habitats (Bloomfield & Gillanders, 2005; Sheaves, 2005; Jelbart *et al.*, 2007; Seemann *et al.*, 2017). Despite this, mangroves have been described as being comprised equally of both marine and terrestrial habitat in the 1960s (MacNae, 1968). As such, mangroves may be expected to support terrestrial fauna, however our knowledge on terrestrial species in mangroves has progressed little since this description. While there are a number of studies that have investigated bird richness in mangroves in Australia, India and Central America (Lefebvre & Poulin, 2009; Mohd-Azlan *et al.*, 2012; Kumar & Kumara, 2014), fewer studies have focused on other vertebrate fauna, like mammals, in mangroves (Nagelkerken *et al.*, 2008; Hogarth, 2015). Much of what we do know about terrestrial vertebrates in mangroves is based on mostly anecdotal records, and their ecology in mangroves is poorly understood (Luther & Greenberg, 2009; Gardner, 2016). It is likely that significant challenges for traditional survey designs related to frequent inundation, generally complex vegetation and muddy sediments (Blench & Dendo, 2007) have impeded the expansion of knowledge on terrestrial fauna in mangroves.

For a holistic approach to the management of mangroves we need to have information on both marine and terrestrial biodiversity features. Besides knowing what species occupy mangrove forests, we need to understand the ecological relationships between terrestrial fauna and mangrove ecosystems, as there may be interactions critical to the health of mangrove forests. Bats for example potentially act as mobile, genetic information linkers via pollination and seed dispersal (Buelow & Sheaves, 2014) and bats and kangaroos in providing nutrients (Reef *et al.*, 2014). It is therefore essential to the identification of management priorities for ecosystems that data on species richness is coupled with information on species identity and resource use (Fleishman *et al.*, 2006). The lack of empirical knowledge about which threatened species are utilising mangroves and how strongly they depend on the resources that these forests provide (Gardner, 2016), and vice versa, the lack of information about the essential resources terrestrial vertebrates provide to mangrove forests, are critical barriers to their effective conservation and management.

Filling ecological knowledge gaps may not be sufficient if appropriate governance structures are not in place to support conservation and management. Current governance structures, which focus on a separation between the marine and terrestrial realms (Alvarez-Romero et al., 2011), have the potential to create risks for intertidal ecosystems like mangroves. These risks are shown for example in the inconsistent governance arrangements and conflicting responsibilities that surround the intertidal zone at the interface of land and sea where mangrove forests occur (Friess *et al.*, 2016). A large source of this complexity lies in the legislative land-sea boundaries that are defined inconsistently on different tidal lines (Tagliapietra *et al.*, 2009; Clemens *et al.*, 2014; Harris *et al.*, 2014). Governance structures need to be carefully assessed to ensure they can provide for effective conservation and management of these important ecosystems.

The main aim of this thesis is to evaluate the terrestrial components and governance structures of mangrove ecosystems. Figure 1 provides a conceptual overview, demonstrating how my chapters contribute to answering the overall thesis aim.

Specifically, my research sought to achieve the following objectives:

1) Determine the current state of global knowledge on terrestrial vertebrate fauna (mammals, reptiles and amphibians) in mangrove forests (**Chapter 2**);
2) Develop a novel survey approach for terrestrial vertebrates that overcomes the challenges associated with mangrove environments (**Chapter 3**);
3) Assess the value of ecological field data on terrestrial vertebrates present in mangroves to inform mangrove management strategies in Australia (**Chapter 4**);
4) Evaluate the adequacy of existing Australian governance structures for the protection and management of mangrove ecosystems (**Chapter 5**).

Chapter 2 provides an extensive global review of records of terrestrial vertebrates occurring in mangroves, documenting patterns of species richness around the world. Logistically challenging survey conditions, the result of frequent inundation, are likely to contribute to the limited knowledge base of terrestrial vertebrate species in mangroves. To overcome these survey challenges, **Chapter 3** details a novel survey approach to detect terrestrial vertebrates in mangroves, including both conventional and newly developed techniques. Survey sites were distributed along the eastern seaboard of continental Australia, where mangrove forests span a latitudinal gradient incorporating temperate, subtropical and tropical climatic regions. **Chapter 4** explores the ecological patterns and resource use of species detected in mangrove habitat in East coast Australia and correlations of patterns of fauna richness with available habitat. To evaluate the governance structures for mangrove ecosystems in Australia, **Chapter 5** provides a comprehensive review of federal and state legislation and a set of practical recommendations relating to definitions and classifications of the land-sea boundary, mangrove ecosystems and plant taxonomy. **Chapter 6**, the final chapter, provides a synthesis of the major findings of the preceding four chapters and provides recommendations for future research directions.

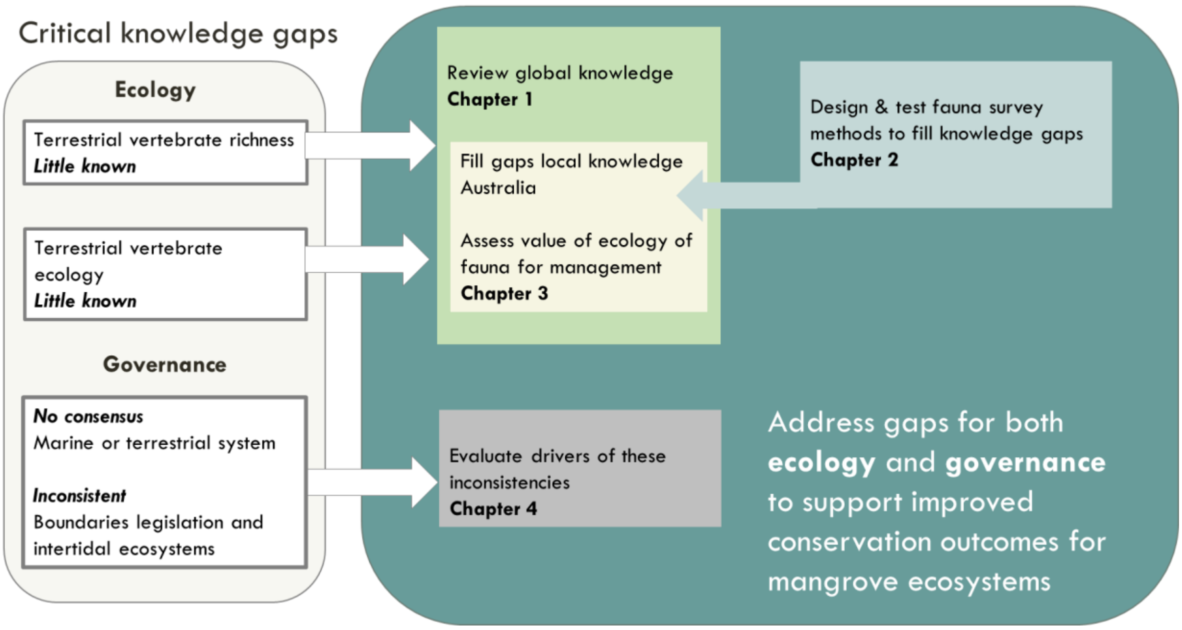

Figure 1. Conceptual overview of thesis "Between two worlds: The implications of existing between land and sea for the conservation of mangrove ecosystems".

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Chapter two: More than marine: revealing the critical importance of mangrove ecosystems for terrestrial vertebrates

Published in *Diversity and Distributions*

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Keywords: Coastal forest, ecosystem function, lagoon, swamp, transboundary, biodiversity patterns.

Abstract

Despite mangrove forests spanning marine, freshwater and terrestrial realms, their function as terrestrial ecosystems has been largely ignored. In light of the rapid global decline of mangroves it is critical to build a more holistic understanding to plan for effective management of the whole ecosystem. This study examines the importance of mangrove forests for terrestrial vertebrates.

An extensive review of records of the use of global mangrove forests by the most poorly studied terrestrial vertebrate groups in these ecosystems: mammals, reptiles and amphibians. I explored the species richness and distribution of these groups, along with their ecological characteristics. I also explored the relationship between animal and plant species richness across the distribution of mangrove forests.

Mangroves are globally used by a remarkable number of terrestrial mammal, reptile and amphibian species (n=464); five times more than previously reported. The diversity of species uncovered by this study, almost half of which are of conservation concern, underscores the value of mangroves as terrestrial ecosystems. Most species were facultative users of mangroves; however, there are critical knowledge gaps in how these species interact with these ecosystems. As hypothesised, animal richness was higher in regions of high mangrove plant richness.

This study highlights that mangrove forests are considerably more important for terrestrial animals than generally acknowledged. I present the most comprehensive review of the importance of mangrove forests for terrestrial vertebrates, but also reveal significant knowledge gaps in the ecology of these ecosystems. My study uncovers evidence that suggests these habitats may be increasingly important as refuges from anthropocentric disturbance. My findings emphasise the importance of moving beyond viewing mangroves as marine ecosystems, toward recognising their cross-realm

importance. Without such a shift, there will continue to be significant limitations in our ability to manage and conserve these ecosystems.

Introduction

Mangrove forests fringe intertidal zones spanning marine, freshwater and terrestrial realms in tropical, subtropical and temperate latitudes (between 30° N and 30° S; Giri *et al.*, 2011). They provide a wide range of ecosystem services, including coastal protection, carbon sequestration and opportunities for biodiversity (Macintosh, 2002). These services continue to be jeopardised due to high rates of mangrove decline, with major ecological and economic implications for both the wildlife and people that depend on them (Alongi, 2002). Globally, 35% of the total area of extent mangroves forests was lost over a recent 30 year period (Giri *et al.,* 2011) and currently, 40% of mangrove plant species are listed as threatened on the International Union for Conservation of Nature (IUCN) Red List (Polidoro *et al.*, 2010). A little-known fact is that mangrove forests are actually declining faster than inland tropical forests and coral reefs, both of which hold a prominent place in global ecosystem conservation concerns (Duke *et al.*, 2007). Despite the value of mangrove forests, aspects of their terrestrial ecology are poorly understood, limiting the capacity to effectively conserve and manage them (Nowak 2013).

Mangrove forests occur at the interface between land and sea, and as a consequence these forests span both aquatic and terrestrial realms, playing fundamental roles in both (Beger et al., 2010). Nevertheless, well justified concerns about the decline in mangrove systems have focused on their value as marine environments, albeit largely ignoring their value as terrestrial ecosystems (e.g. the provision of forest products for human benefit (Buelow & Sheaves, 2014) and habitat for terrestrial fauna (Meades *et al.*, 2002)). As mangrove ecosystems are traditionally conceptualized as marine environments (e.g. providing nursery areas for important commercial fisheries (Mumby *et al.*, 2004), there is a strong bias towards research on their marine fauna. This research focuses on species specific to their marine habitat, such as polychaetes (Metcalfe & Glasby, 2008), molluscs (Appadoo &

Roomaldawo, 2013), shrimp (Primavera, 1998), crabs (Schories *et al.*, 2003) and fish (Faunce & Serafy, 2006). This emphasis on mangroves as purely marine ecosystems is likely to have arisen from a historically commercial-use view of the ecosystem, and may explain as to why their value as terrestrial habitat for vertebrates remains poorly studied (Luther & Greenberg, 2009; Nowak, 2013).

Previous research on terrestrial vertebrates in mangroves has largely focused on birds or charismatic fauna, such as the Sumatran tiger, *Panthera tigris sumatra* (Noske, 1996; Barlow *et al.*, 2011). The assemblage of smaller mammals, reptiles and amphibians that occupy mangrove forests has, by contrast, received little attention (but see Nagelkerken *et al.*, 2008; Luther & Greenberg, 2009; Hogarth, 2015). The literature also over-represents species that depend on mangrove forests for all their critical resources (i.e. obligate mangrove users), resulting in a narrow focus that excludes many facultative mangrove users (Hansson & Åkesson, 2014). Yet facultative users continue to provide important ecosystem services, such as pollination of mangrove plants (McKenzie & Rolfe, 1986) and the transfer of nutrients from adjacent habitats (Reef *et al.*, 2014). In addition, the loss of primary habitats for these facultative users may mean mangrove forests are increasingly important habitat for these species in the future (Nowak, 2013, Rodrigues & Martinez, 2014). In order to better understand the functioning of mangrove ecosystems, it is important to understand their relationship with both the obligate and facultative fauna that uses them, in both the marine and terrestrial realms.

Here, I review existing knowledge of the terrestrial vertebrates known to use mangroves, and document the global distribution of species richness that can serve as a baseline for future studies. I explore what is known about the ecological relationships between terrestrial vertebrates and mangrove forests, identify critical knowledge gaps and provide recommendations for improved conservation management of these vital ecosystems.

Methods

Scope of the review

We sought records of terrestrial mammals, reptiles and amphibians that use the terrestrial component of mangrove ecosystems. I defined the terrestrial component of mangroves as those areas exposed to air at some time during the tidal cycle (e.g. branches, roots, adjacent mud flats). Semi-aquatic snakes, water rats and frogs were included as they routinely occupy terrestrial habitats (Fish & Baudinette, 1999; Gibbons, 2003; Willson *et al.*, 2006), whereas sea turtles were excluded as they exclusively use the aquatic aspects of mangrove systems (Lutz *et al.*, 2002). While birds are important inhabitants of mangroves, they were excluded from this study because they have previously received significant attention (e.g. Cawkell, 1964; Buelow & Sheaves, 2014; Kumar & Kumara, 2014). To ensure that as many records of terrestrial vertebrates in mangroves were captured as possible, I included species with only a single record in mangroves. It is possible that some vagrant species are included in the species list, so I have indicated how many times the species was recorded (see Supplementary Table 1). Both obligate and facultative users of mangrove forests were included to identify the full range of species that exploit mangrove resources and potentially play a role in ecosystem function. I defined the use of mangroves as when a species was reported to use any key resource provided by the mangrove forest (e.g. food, shelter, dispersal route – see "Characteristics of species reported in mangrove forests" below for more detail).

Search strategy and inclusion criteria

I conducted a comprehensive online search of the literature published up until the end of June 2016. The publication databases Web of Science, Trove Thesis Search and Google Scholar were searched using the following terms: (*fauna AND diversity AND mangrove**); (*mangrove* AND mammal*; mangrove* AND biodiversity*); (*mangrove AND reptile; mangrove AND amphibian*); (*mangrove AND monitoring OR techniques AND fauna OR reptile OR mammal OR amphibian*); (*mangrove AND survey*

AND fauna OR reptile OR mammal OR amphibian OR vertebrate); (*mangrove AND terrestrial AND vertebrates OR fauna OR mammal OR reptile OR amphibian*).I also searched synonyms for mangrove habitat (*coastal forests, swamp* and *coastal lagoon*) together with the above search terms. The reference list of each relevant publication was scrutinized to identify further relevant literature. All mammal, reptile and amphibian taxa reported as occurring in mangroves were documented. Taxonomy followed the IUCN Red List (IUCN, 2016).

Species records were also obtained from the peer reviewed literature (including PhD Theses) and three open source databases: World Wildlife Fund (WWF, 2016), ARKive (ARKive, 2016) and IUCN Red List (IUCN, 2016). These databases were included because they focus on global species records, allow for restriction of searches to mangrove habitat and have provision for peer-review by experts. Records from guidebooks, local management plans and additional online sources were excluded because it was not possible to determine the accuracy and consistency of the records in those sources.

Characteristics of species reported in mangrove forests

All literature that met the above criteria were read in full and records of terrestrial vertebrate species in mangroves were collated. For each species record I noted: i) countries in which the species was reported to use mangroves; ii) the total native range; iii) IUCN conservation status (including "not evaluated"); and iv) feeding guild (i.e. carnivore (including insectivore and piscivore), herbivore, omnivore). If these characteristics were not mentioned, I undertook targeted literature searches to ensure the dataset was comprehensive (e.g. consulting the IUCN Red List database for the total native range (when a species was not evaluated by IUCNI used Arkive and/or guidebooks to determine its range) and conservation status and Arkive for feeding guild).

A species' dependence on mangrove forests was classified as obligate, facultative or not reported based on information in the studies. Obligate users included those species described as found primarily in mangroves or with a life history tied to mangrove habitat, whereas facultative users were those species that occupied both mangrove and adjacent terrestrial habitats. Facultative users were further categorized by their resource use: i) feeding; ii) breeding; iii) dispersal route between primary habitats; iv) shelter from biotic (e.g. predators, competitors) and abiotic stressors (e.g. temperature extremes; desiccation); v) use and increased frequency of use as refugia from human disturbance; and vi) novel use of mangroves as a result of human disturbance.

Species distribution and richness

In many instances, species records were specifically focused on a subset of countries from within the species' broader native range (e.g. snakes in a mangrove patch in Singapore; Voris (2002)). To estimate the global species richness of terrestrial vertebrates, if a species was reported to use mangroves in part of its range I inferred that it may do so wherever its distribution overlapped with mangrove forest. For example, the Mexican Mouse Possum, *Marmosa mexicana*, occupies mangrove forest in Mexico (WWF, 2016), thus, based on the intersection of the species' range and extant mangrove forest, I inferred that it also uses mangroves in Belize, Costa Rica, El Salvador, Guatemala, Honduras, Nicaragua and Panama. With this approach, the alpha diversity of terrestrial vertebrate species that occupy mangroves was calculated for each country. As Brazil and Australia span >20 degrees of latitude, and are substantially larger than other countries that support mangrove habitat, outputs are presented at the state level for these two countries. Species ranges followed Hutchings & Recher (1981); Uetz (1995) and Wilson & Reeder (2005). Mangrove richness followed Spalding *et al.* (2010) and mangrove distribution followed Giri *et al.* (2011). A Pearson's correlation coefficient was conducted to determine whether there was a relationship between global mangrove plant and terrestrial animal richness. In an attempt to explain biogeographic patterns of species richness I also calculated the correlation coefficient for regional groupings of countries (Africa, Asia, Americas, Middle East and Oceania). While the world can be divided into regions in many different ways I followed the United Nation (2014) macro-geographical global regions as it is a commonly used source.

Results

We identified 464 terrestrial vertebrate species reported to occur in mangrove forests: 320 mammals, 118 reptiles and 26 amphibians (including 22 subspecies) (Supplementary material, Table S1). This is a five-fold increase in the number of terrestrial vertebrate species previously reported to use mangroves (excluding birds), including representatives of 14 additional families to those in previous reviews (Kathiresan & Bingham, 2001; Nagelkerken *et al.*, 2008; Luther & Greenberg, 2009; Hogarth, 2015). Of the species reported, 34 were reported fewer than five times or were stated to be rare in mangroves. Of the 391 vertebrate species whose conservation status has been assessed by the IUCN, 35% were classified as threatened. Fewer than 41% of the published species records I found (n=186 of 464) were derived from or could be directly linked to published field studies. As such, reports, many of which were reviews, of the overwhelming majority of species occurring in mangroves did not reference the field study or observation on which this species being present in mangrove habitat was based. Therefore, the origin of these records are uncertain and separate records may not be independent, suggesting more field studies on terrestrial vertebrates in mangroves are important to improve our understanding of these ecosystems.

Species distribution and alpha diversity

Mammals, reptiles and amphibians were reported to occur in mangroves in 73 of the 120 countries (60%) in which mangroves occur (Figure 1B & D), more than doubling previous estimates (Figure 1A). When accounting for the potential global distribution of these terrestrial vertebrates in mangrove habitat this estimate increases to 113 of 120 countries (93%) (Figure 1C), more than trebling previous reports (Figure 1A). My findings suggest that the highest alpha diversity of terrestrial vertebrates in mangroves occurs within Asia, northern Australia and the Central American land bridge (Figure 1). In contrast, alpha diversity is lowest on the east coast of Africa, southern Australia, New Zealand, the Middle East and small island nations (Figure 1). There were 24 countries that support extant mangroves where terrestrial vertebrates have yet to be documented as using this ecosystem as habitat; of these, 20 were island nations with a total land area of less than 2,535 km^2 (Figure 1B).

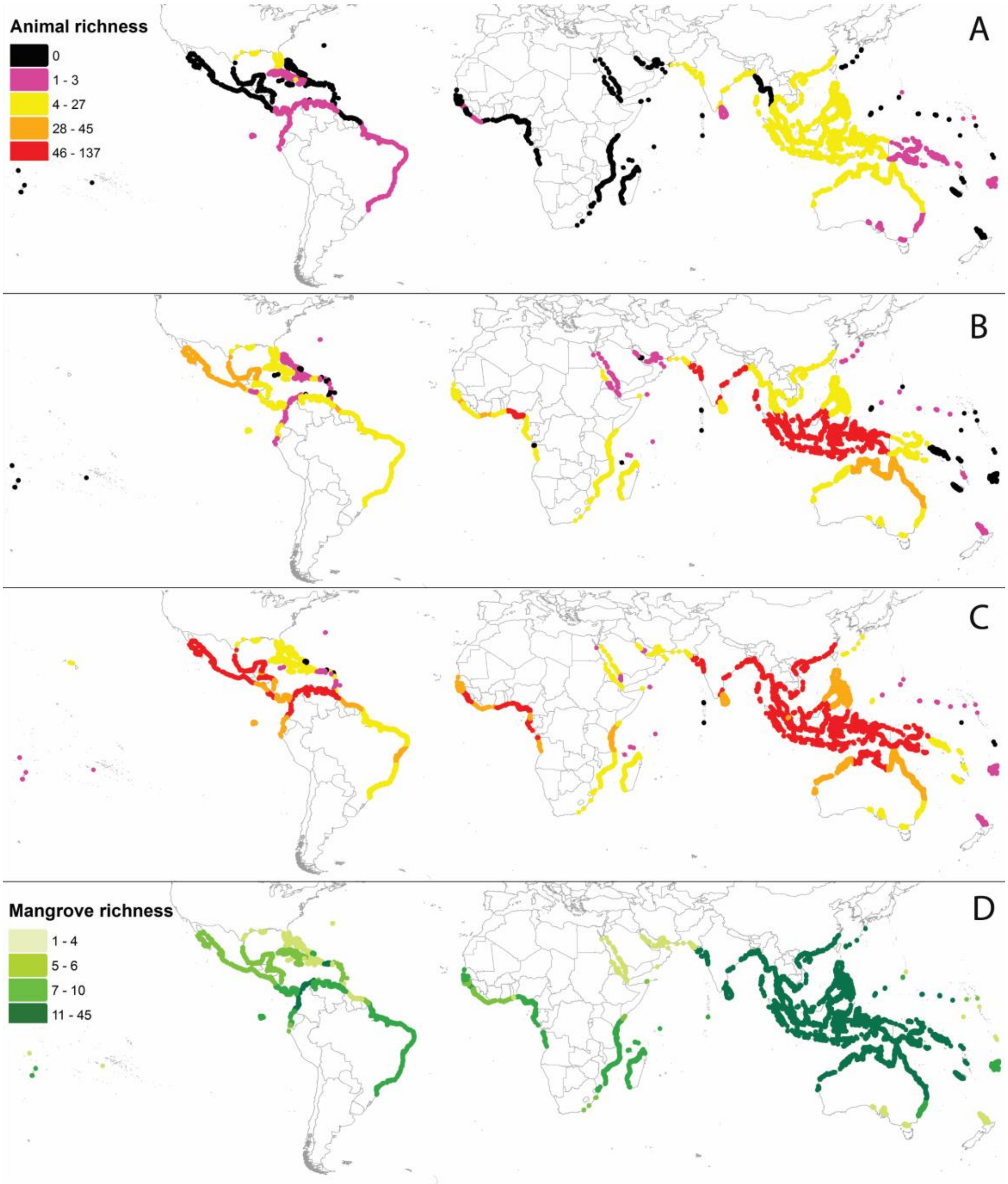

Figure 1. Known global number of terrestrial vertebrate species in mangroves per country: A) prior to this review, B) based on this review and C) after extrapolating data from this review to intersect with the global distribution of individual vertebrate species with that of D) the global distribution and richness of mangroves. As Brazil and Australia span >20 degrees of latitude, and are substantially larger than other countries that support mangrove habitat, outputs are presented at the state level for these two countries. Color categories are scaled to reflect the distribution centred on the median

number of species (C) to aid comparison between maps. Uncoloured countries do not support mangroves.

Relationship between mangrove plant and terrestrial vertebrate alpha diversity

Globally, there was a positive correlation between terrestrial vertebrate alpha diversity and mangrove plant richness for both the recorded and the potential distributions (Table 1; reported: $r=0.556$, $n=120$, $p<0.001$; extrapolated: $r=0.609$, $n=120$, $p=<0.001$). When separating the world into macro-geographical regions, the strength of the relationship varied, with the strongest correlations in Asia and Oceania for both the reported data (Asia: $r= 0.724$, $p=0.001$, $n=17$; Oceania: $r=0.581$, $p=0.005$, $n=22$) and extrapolated data (Asia: $r= 0.783$, $p=<0.001$, $n=17$; Oceania: $r=0.807$, $p=<0.001$, $n=22$). For the Americas the correlation between mangrove plant and animal richness was significant only when using the extrapolated data ($r=0.467$, $p=0.002$, $n=4$). No relationship was found in Africa (reported: $r=0.12$, $p=0.556$, $n=27$; extrapolated data $r=0.13$, $p=0.528$, $n=27$) and the Middle East (reported $r= -0.24$, $p=0.941$, $n=12$; extrapolated data $r=-0.04$, $p=0.902$, $n=12$).

Table 1. Correlation between mangrove plant richness and animal richness in mangrove ecosystems.

Region						Global
	Asia	Oceania	Americas	Middle East	Africa	Global
			Reported data			
Correlation coefficient	0.72	0.58	0.19	-0.24	0.12	0.55
Probability	0.001**	0.005**	0.232	0.941	0.556	0.001**
Number of countries	17	22	42	12	27	120
			Extrapolated data			
Correlation coefficient	0.78	0.81	0.47	-0.04	0.13	0.58
Probability	<0.001**	<0.001**	0.002**	0.902	0.528	0.001**
Number of countries	17	22	42	12	27	120

Use of mangrove forests by terrestrial vertebrates

For the 464 terrestrial vertebrate species reported in mangroves, 147 had information about the specific resources they were using. Species were most often reported to use mangrove forests as foraging grounds (Figure 2). Terrestrial vertebrates were also reported to use mangrove forests as refuges from human disturbance, shelter from stressors (both biotic and abiotic), to disperse between primary habitats and for breeding (Figure 2).

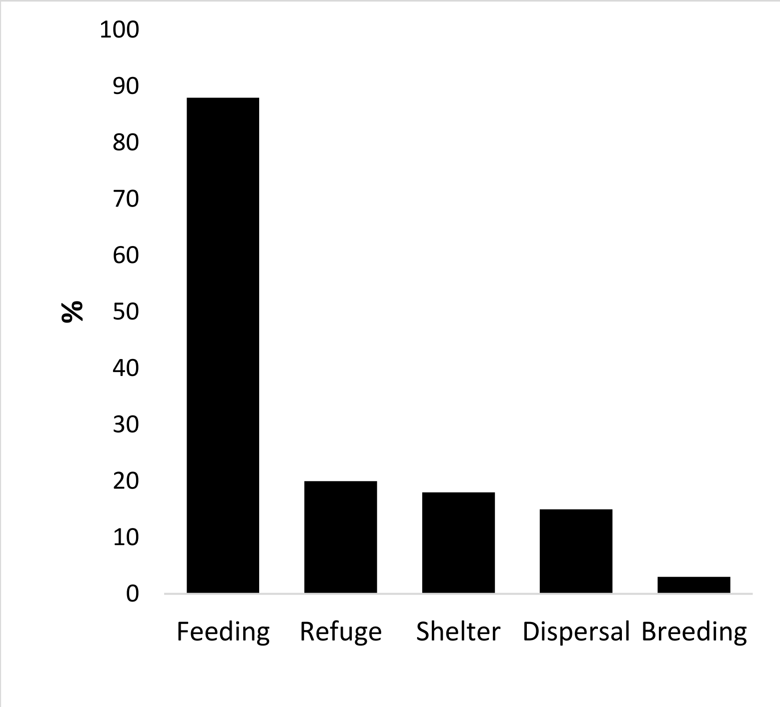

Figure 2. Mangrove use by facultative terrestrial vertebrates (n=147). Columns sum to more than 147 as reported uses are not mutually exclusive. Obligate users (n=24) are not shown in this figure as by definition these species use mangroves for all their resources.

If species were reported to use mangrove forests as refuges, most (19) were considered to be increasing their existing use of mangroves rather than using mangrove forests as a novel habitat (1). The majority of terrestrial vertebrate species using mangroves forests are categorized as carnivorous (52%; Figure 3).

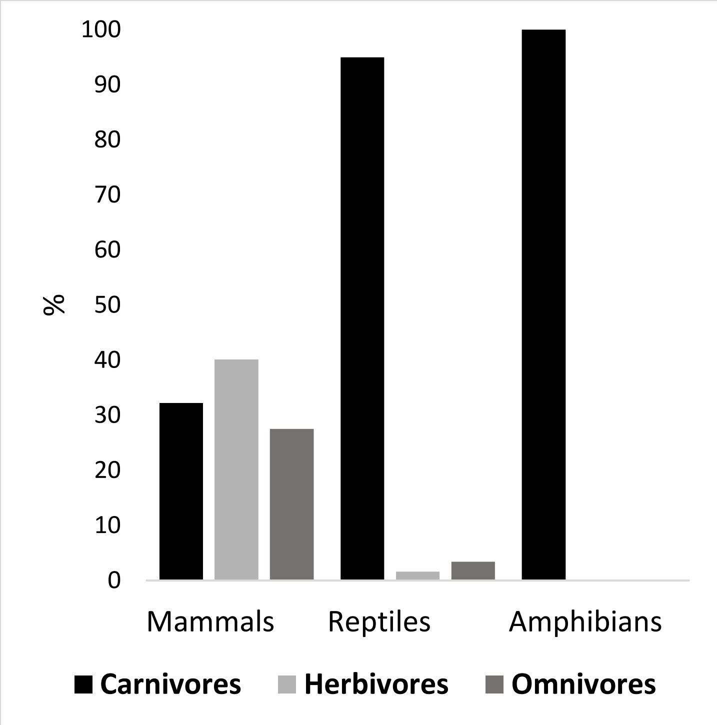

Figure 3. Feeding guild of terrestrial vertebrates in mangroves, including both facultative and obligate mangrove users (mammals, n=320; reptiles, n=118; amphibians, n=26). Insectivores and piscivores are included in the carnivore guild.

Discussion

Species distribution and richness

We found mangrove forests to be substantially more important for terrestrial vertebrates than previously reported, supporting a remarkable richness of terrestrial mammals, reptiles and amphibians globally. This finding underscores the view that the emphasis on mangroves as marine ecosystems has led to the importance of the terrestrial components of mangrove forests being undervalued (Luther & Greenberg, 2009; Nowak, 2013).

In addition to extending our knowledge of the global richness of terrestrial vertebrates in mangrove ecosystems, my results also reveal a wider range of countries in which these species use mangroves than previously reported (Figure 1B, C). There were few places in the world where terrestrial vertebrates have not been reported to use mangrove forests, mainly small islands (Figure 1B). The biogeographic patterns indicate a higher richness of species in the tropics (especially in Asia) and lower richness in temperate regions, which is one of the most well-established patterns in macro ecology and biogeography (Koch, 2000). Mammals and amphibians conform to the pattern of highest diversity in the tropics (Schreier *et al.,* 2009; Wiens *et al.,* 2006) and my findings demonstrate this pattern extends to mangrove forests. I can think of no reason why the key processes that drive biogeographic patters (speciation, extinction and dispersal (Wiens *et al.,* 2009)) should differ in mangrove ecosystems, but this could be a focus of further study.

The observed pattern of high animal richness in tropical mangrove regions was consistent, regardless of whether data were derived from previous reviews, this review or the maximum possible richness of terrestrial vertebrates based on my extrapolation (the countries where the known distribution of a species intersects with the global distribution of mangroves) (Figure 1). Because the extrapolated data assume that species found in mangroves in one part of their range could also be found in mangroves across their entire range (Figure 1C), this may overestimate species occurrences. In particular, species may not occupy habitat at the edges of their predicted distribution, in marginal habitat or areas experiencing human disturbance or other pressures. The fact that the correlation between floral and faunal richness improved or remained unchanged with the extrapolation suggests that any overestimation of richness is unlikely. Nevertheless, the extrapolated data provide a worst case scenario about the knowledge gaps associated with the species richness and distribution of terrestrial vertebrates in mangrove forests, which can help target future research toward areas of potentially high richness but few records. To be able to make more detailed predictions and identify important mangrove areas for terrestrial vertebrates, it is necessary that future studies report characteristics (e.g. total mangrove area, source of disturbance, adjacent habitat etc.) about the mangrove region where a species was observed.

Relationship between mangrove plant and terrestrial vertebrate alpha diversity

Mangrove plant richness was not always a consistent predictor of animal richness, although correlations suggest that this relationship is stronger when the data were extrapolated to the possible extent of recorded species (Figure 1D, Table 1). Further fieldwork may reveal a higher richness of terrestrial vertebrates in mangrove forests in areas where there is a mismatch between floral and faunal diversity. Mangrove plant diversity is an order of magnitude higher in the Indo-West Pacific (IWP) than it is in the Atlantic, Caribbean and Eastern Pacific (ACEP) (Ellison *et al.*, 1999), and plant richness may directly influence richness of other taxa by determining the variety of food items or habitat structural elements that create niches for other organisms (Hawkins & Porter, 2003). A global concordance of plant and animal richness has been shown for the three vertebrate groups covered in this study (Qian & Ricklefs, 2008). However, I found some areas of low mangrove plant richness, such as Mexico and parts of South America, where vertebrate richness was high and similar to that of parts of Asia where mangrove plant richness is far higher (Figures 1B, C & D). This suggests that factors other than low mangrove plant richness, such as the influence of adjacent terrestrial habitat (e.g. deserts have lower animal diversity than tropical rainforests and could affect number of facultative species using adjacent mangroves) or the availability of prey species to support carnivores and omnivores, may also strongly influence animal richness in some key mangrove ecosystems. The large proportion of species that rely on animal protein instead of mangrove plant material (Figure 3) provides some support for the latter hypothesis, but a deeper understanding of the community ecology of these ecosystems is needed. Temperate regions in Oceania (e.g. New Zealand and southern Australia) showed both low floral and faunal richness in line with global biogeographic patterns (Wiens *et al.,* 2009). However, I note that there has been limited survey effort in temperate areas of these regions where diversity would be expected to be lower. The weak correlations between animal and mangrove plant richness in Africa and the Middle East may be a product of these regions having received little study (fewer than 10% of the studies I found specifically discussed these regions), possibly because of political unrest (Reddy & Dávalos, 2003), as opposed to a genuinely lower animal richness. The discrepancy between predicted and observed richness was highest in East Africa, a biodiversity hotspot (Myers *et al.*, 2000) and with a similar mangrove richness to that of West Africa where a higher animal richness was observed. One reason for lower than predicted animal richness in Africa may be the prevalence of threats, such as hunting pressure for the bush-meat trade, which can be high in East Africa (Cawthorn & Hoffman, 2015), depleting local populations to levels that make detection of some species less likely, especially when combined with lower survey effort in the region. The variability or lack of research effort across regions, lack of insight into regional differences and a somewhat inconsistent relationship between mangrove plant richness and animal richness suggest that additional field studies are needed to achieve a more complete understanding of the occurrence of terrestrial vertebrates in mangroves.

Although I present the most comprehensive review of terrestrial vertebrates in mangroves, and have identified significantly more terrestrial vertebrate species using mangrove forests than previously reported, my findings probably still underestimate species richness. There was limited direct evidence of species occurrence from published field studies and most of the records I found did not report the origin of the record, suggesting many come from unpublished data. The small number of field studies carried out in mangroves could be due to the combination of challenges associated with accessing these environments and the difficulty of sampling tidal environments due to regular inundation. New approaches may be needed to facilitate faunal surveys in mangroves. My results provide a valuable starting point from which to target survey effort. Similarly, my findings highlight the importance of clearly reporting the source of species records to help identify where research effort is genuinely low.

Use of mangrove forests by terrestrial vertebrates

Understanding habitat use might help explain why biogeographic patterns differ from predictions associated with floral richness. My results highlight that facultative users of mangroves substantially outnumber specialists, as is predicted by theory (Wilson & Yoshimura, 1994). This suggests that the tendency of previous studies to focus on obligate users of mangrove forests (Nagelkerken *et al.*, 2008; Luther & Greenberg, 2009) has distorted our understanding of the faunal species-types that use mangroves and contributed to the perception that vertebrate species richness is low in these environments. A striking finding from my study is how little is known about the interaction between terrestrial vertebrate species and mangroves. For example, the relationships between seasonality and timing in mangrove use is rarely studied. I only found 10 records in which seasonality or tidal phase was reported, including lemurs feeding on mangrove flowers in the dry season (Gardner, 2016) and varanids feeding in mangroves during low tide (Kutt, 1997). Understanding the resources consumed by facultative users of mangrove forests, and when these resources are available, is a particularly important area for future study because habitat destruction and environmental change is likely to lead to changes in resource availability (Nowak, 2013).

My results support the suggestion that at least some terrestrial vertebrates are using mangrove forests as a refuge from anthropocentric disturbance or loss of their primary habitat (Nowak, 2013; Figure 2). Although the number of species reported to be using mangrove forests as refuges is low, this has been reported for many different species groups (e.g. felids, Barlow *et al.* 2011; Nowak, 2013; snakes, Nagelkerken *et al.,* 2008; monkeys, Nowak, 2013) in much of the world (e.g. Asia, Nowak 2013; Africa, Nagelkerken *et al.,* 2008; South America, Rodrigues & Martinez, 2014), suggesting it is a relatively widespread and possibly underestimated phenomenon. Much of the evidence for the increasing use of mangroves is anecdotal, with just a few examples in which empirical data document a novel expansion into mangrove forests (e.g. Wied's marmosets, *Callithrix kuhlii* (Rodrigues & Martinez, 2014)). Some new records may also reflect a lack of historical data rather than evidence of a genuine expansion. It is also unclear whether mangrove forests are acting as population sinks or constitute important habitat for self-sustaining populations. Understanding this relationship may help predict the likely future impact of increased use on current mangrove inhabitants.

Knowledge gaps concerning vertebrate use of mangroves are particularly concerning for the many species that are of conservation significance. Given the high probability that species classified as "Data Deficient" or "Not Assessed" by the IUCN Red List are actually of conservation concern (Bland *et al.*, 2014; 72 species in this review), there is a pressing need to better understand the importance of mangroves as a resource for endangered species. Better information about the dependence of terrestrial vertebrates on mangrove forests could help identify the regions of the world in which mangrove conservation should be given highest priority.

The high proportion of carnivorous and omnivorous vertebrates in mangrove forests (Figure 3) is likely to be due to the marked seasonality of other food sources, such as fruit and nectar (Fernandes, 1999), and the low palatability of mangrove leaves given their high salt content (Kathiresan & Bingham, 2001). This finding also accords with the general prevalence of carnivores among reptiles and amphibians (Huey, 1982). Given that animal protein appears to be the most important source of nutrition for terrestrial vertebrates in mangrove forests, this may account for some of the variability I found in the relationship between mangrove plant and animal richness (see Species distribution and richness).

While vertebrate species were most often reported to use mangrove forests for feeding (Figure 2), for the majority of species there was no information about how they use mangroves. I found few primary records of field studies that documented habitat use by terrestrial vertebrates in mangroves. Given that most animals spend the majority of their time feeding, it is not surprising that opportunistic reports of species in mangroves often report individuals to be using mangroves as feeding grounds.

However, I did find evidence that facultative users of mangroves rely on these areas for critical stages in their life cycles as well, using these areas to breed (e.g. the Estuarine Crocodile, *Crocodylus porosus*, Hutchings & Recher, 1981; and the Sea Krait, *Laticauda colubrine*, Hogarth, 2015). There were also records of species using mangroves to shelter from heat stress (e.g. bat species and kangaroos in Australia; Reef *et al.*, 2014) and to disperse between primary habitats (e.g. the marsh rabbit, *Sylvilagus palustris*, in the United States; Kathiresan & Bingham, 2001) (Figure 3). A better understanding of the dependence of both obligate and facultative users on mangrove ecosystems will enable the implications of anthropogenic disturbances on ecosystem function of these forests to be better understood.

Importance of terrestrial vertebrates for mangrove ecosystems

The roles of terrestrial vertebrates in the health of mangrove ecosystem has been poorly studied, leaving another substantial gap in our understanding of mangrove ecosystem functioning. Terrestrial vertebrates can play an important role in the health of mangrove forests through the provision of essential ecosystem services, such as pollination of mangroves (Megachiroptera, Ashraf & Habjoka, 2013) and nutrient transfer (e.g. Reef et al 2014; Kristensen, 2008). The ecosystem services provided by terrestrial vertebrate fauna associated with the health of mangrove forests should therefore be explored as an important precursor to more effective conservation of these ecosystems.

Conclusions

My data demonstrate that mangrove forests support a considerably higher diversity of terrestrial animals than previously recognized. In spite of this, the terrestrial components of mangroves are generally ignored, overlooking a large part of this transboundary ecosystem. My findings highlight a wide range of knowledge gaps in relation to the diversity, distribution and ecology of species using the terrestrial components of these systems. Future research should focus on undertaking field assessments of terrestrial vertebrates, particularly in regions where my findings suggest animal richness is low relative to floral richness (e.g. small islands, temperate regions, the Middle East and East Africa). Greater attention to reporting the source of species records could help identify where research effort should be directed. There is also an urgent need to understand better the ecological relationships between mangroves and terrestrial vertebrates to plan for effective conservation of these forests. In particular, it is important to understand better the role mangrove forests are playing in providing a refuge for species suffering from the loss of their primary habitat. This knowledge will help identify regions where patterns of habitat loss and human disturbance may elevate remaining mangrove forests to an indispensable status, whilst also identifying areas where current users of mangroves may find resources under increasing pressure. In summary, I recommend a more holistic view of mangrove forests, as only when this is realized will it be possible to effectively conserve these vital ecosystems.

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Chapter two - Supplementary Details

Methods

1) Exclusion criteria were "benthos"; "macrofauna"; "invertebrate". 2) Results were limited to English literature, thereby potentially missing information published in Spanish, French and Bahasa on species occurring in South and Central America and Africa. 3) In total 37 papers were used.

Flora/fauna diversity

Mangrove plant diversity is an order of magnitude higher in the Indo-West Pacific (IWP) than it is in the Atlantic, Caribbean and Eastern Pacific (ACEP) (Ellison *et al.*, 1999), related to speciation patterns in the early Tertery period around the Theys sea. Ricklefs, R. E., & Latham, R. E. (1993). Global patterns of diversity in mangrove floras. *Species diversity in ecological communities: historical and geographical perspectives. University of Chicago Press, Chicago*, 215-229.

Addition to statement "Plant richness may directly influence richness of other taxa by determining the variety of food items or habitat structural elements that create niches for other organisms (Hawkins & Porter, 2003)": Apart from plant richness fauna richness is driven by a multitude of other environmental factors, such as land cover, vegetation, climate, soil, topography, and topographic heterogeneity and these complex relationships warrants further investigation. Stein, A., Gerstner, K., & Kreft, H. (2014). Environmental heterogeneity as a universal driver of species richness across taxa, biomes and spatial scales. Ecology letters, 17(7), 866-880.

Addition to Qian & Ricklefs 2008*:* however this study was based on multiple ecosystems incorporating more niches than mangroves (e.g., terrestrial shrub layer missing in mangrove due to tidal influences) which make a one-on-one comparison difficult. I found nonetheless that some of these richness patterns hold even in mangroves with low plant richness, such as Mexico and parts of South America, where vertebrate richness was high and similar to that of parts of Asia where mangrove plant richness is far higher (Figures 1B, C & D).

Giri et al., 2011 was used in error for the statement: Globally, 35% of the total area of extent mangroves forests was lost over a recent 30 year period. The correct reference is: Hamilton, S.E. & Casey, D. (2016) Creation of a high spatio-temporal resolution global database of continuous mangrove forest cover for the 21st century (CGMFC-21). Global Ecology and Biogeography, 25, 729-738.

<u>Supplement Table S1 : Terrestrial vertebrates reported to occur in mangroves</u>

Taxonomic group	Number of species reported	Taxonomic group	Number of species reported
Primates	79	Lemurs and loris	27
Chiroptera	57	Marsupials	13
Rodentia	30	Squirrels	5
Carnivora (incl. civet, mongoose, otters, genet, racoon, skunk)	30	Canidae	2
Felidae	26	Ursidae	1
Ungulata	31	Eulipotyphla (shrews)	4
Pilosa (Sloths)	2	Ferae (pangolins)	2
Xenarthria (anteaters, armadillos)	4	Lagomorpha	3
Serpentes	63	Squamata	37
Crocodylidae	11	Anura	26

Chapter three: Evaluating survey techniques to detecting terrestrial vertebrates in flooded forests

Not published as of yet

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Keywords: Biodiversity inventory, costs, methods, inundation, mangrove, monitoring, success, survey design, survey efficiency, trapping techniques

Abstract

Our understanding of the occurrence of terrestrial vertebrates in flooded forest ecosystems worldwide is exceptionally poor, with very little empirical data from field surveys available. This knowledge gap is linked to the challenging survey conditions associated with these environments. Here I develop and evaluate a rapid survey approach to assess the terrestrial mammals, reptiles and amphibians, the most difficult taxonomic groups to detect, in these ecosystems. I assessed eight commonly used for fauna detection techniques, namely incidental sightings, high frequency acoustic monitoring (bat detector), live traps, camera traps, night line transects, hair tubes, artificial terrestrial refuges and artificial arboreal refuges. Techniques were as evaluated according to their ability to address challenges in relation to tidal inundation, and their effectiveness and efficiency in detecting different taxa. The approach was employed at 10 sampling sites across temperate, subtropical and tropical mangrove forests along the eastern seaboard of Australia.

Patterns in the species richness detected are comparable to global patterns observed in mangrove forests for the same vertebrate groups, suggesting the approach was effective at detecting terrestrial fauna. Species accumulation curves indicate four consecutive nights were likely sufficient to detect most species in temperate regions, but longer surveys may be required in the tropics and subtropics due to the expected higher species richness in these regions. The approach is flexible and can be tailored to the most efficiency approach (cost per species detected) considering equipment, field time, processing time and taxonomic group.

This study provides the first evaluation of multiple detection techniques in flooded forests, such as mangroves, offering a flexible survey tool that can be adapted to different regions, target taxa and budgetary constraints. Insight into optimal survey practices in these ecosystems should facilitate a

better understanding of the species richness and ecology of flooded forests to inform their management and conservation.

Introduction

Biogeographic patterns of species occurrence have interested ecologists for centuries (Wiens & Donoghue, 2004), yet we know more about species richness in some ecosystems than others. For example, it is estimated that 99% of fauna occurring in temperate European heathlands have been documented (EEA, 2010), yet the biota of tropical areas is generally studied to a far lesser extent (Ferrier, 2002). Some of this variation may be explained by the ease (or degree of difficulty) with which certain ecosystems can be accessed and surveyed (Anderson, 2001; Kier *et al.*, 2005). With knowledge on species distribution being a key attribute for conservation reserve selection (Rondinini *et al.*, 2006), and an ecological understanding of ecosystems being essential for targeting conservation management actions (Zipkin *et al.*, 2010), bias in baseline biodiversity survey effort can potentially leave some ecosystems more vulnerable to threats. With knowledge on species distribution being a key attribute for conservation reserve selection (Rondinini et al., 2006), and an ecological understanding of ecosystems being essential for targeting conservation management actions (Zipkin et al., 2010), bias in baseline biodiversity survey effort can potentially leave some ecosystems more vulnerable to threats (e.g., poor coverage in protected areas or the presence of invasive species going unrecognised). Without knowledge of the species present and potential threats impacting these areas, developing effective conservation measures becomes a difficult task.

Flooded forests like tidal mangrove and estuarine forests, and seasonally flooded peat swamp and freshwater forests (e.g. Amazon floodplain forests and freshwater forests in the United States) are among the most poorly surveyed terrestrial ecosystems (Prentice & Parish, 1991; Goulding, 1993; Hawes *et al.*, 2012), with limited efforts to obtain species inventories (e.g. Haugaasen & Peres, 2005; Posa *et al.*, 2011b; Nowak, 2013; Rog *et al.*, 2017). These ecosystems provide a range of important ecosystem services, such as carbon sequestration (peat swamps; Jauhiainen *et al.*, 2005; mangroves;

McLeod et al., 2011), food provision (mangroves; Kathiresan & Rajendran, 2002), flood prevention (mangroves, Duarte et al., 2013, peat swamps; Parish, 2002, freshwater forests; Hey& Philippi 1995) and erosion mitigation (mangroves, Koch et al., 2009). Despite their importance they are rapidly declining, with estimates of up to 70% of their global extent lost in recent decades (Giri *et al.*, 2011; Posa *et al.*, 2011b). A wide range of threatening processes are driving this loss, including coastal development, climate change (MacDicken, 2002; Conner *et al.*, 2007; Craft, 2012; Rogers *et al.*, 2016), aquaculture (Primavera, 2006), logging, fire, land conversion (Posa *et al.*, 2011b), dredging and dams (Knutson & Klaas, 1998). In light of these threats, there has been increasing efforts to document the faunal diversity and ecology of these ecosystems, generally with an emphasis on aquatic fauna (e.g. fish or cetaceans (Ng *et al.*, 1994; Martin & Da Silva, 2004), crustaceans (Murugan & Anandhi, 2016) and benthic species (Nagelkerken *et al.*, 2008). Despite these ecosystems spanning both the aquatic and terrestrial realms, there has been little research into the terrestrial fauna utilising flooded forests (e.g. mammals, reptiles and amphibians; (Rog *et al.*, 2017), though birds are a notable exception (Borges & Carvalhaes, 2000; Mohd-Azlan, 2011; Posa, 2011a). Furthermore, the information that does exist for most terrestrial taxa is largely anecdotal (Rog *et al.*, 2017). While anecdotal data can provide important insight into species that are potentially using flooded forests (e.g., as a refuge when primary habitat is lost; (Nowak, 2013), these data provide little insight into species occurrence, density or habitat dependence. The lack of empirical data on the terrestrial fauna within flooded forests has led to calls for a greater emphasis on field studies to document the diversity and ecological role of fauna within these ecosystems (Gardner, 2016).

The paucity of field studies within flooded forests is perhaps not surprising given these ecosystems pose significant challenges for ecological surveys (Blench & Dendo, 2007; Hogarth, 2015; Luiselli *et al.*, 2015). The complex branch and root structures of flooded ecosystems, such as mangrove forests, combined with muddy or silty substrates, make accessing and manoeuvring in these environments difficult and slow. While challenging survey conditions are not unique to flooded forests and have impeded data collection in other habitats as well (e.g.; the deep sea, Haedrich et al., 2001; mountain cliffs, Larson et al., 2005; or cave communities, Weinstein & Slaney, 1995) the tidal inundation or seasonal flooding means that there are unique risks to traditional terrestrial survey techniques resulting in fauna drowning or equipment being damaged. In tropical flooded forests regions – as in other tropical forests, there are also threats to personal safety due to predators, such as crocodiles or large cats that may further limit research capacity. In combination, these challenges and risks create barriers to researchers working in these environments.

I contend that the difficulties of surveying tidal forests need not prevent surveys within these environments, but do require survey techniques, or their execution, to be modified to suit the conditions. While faunal survey techniques are often re-evaluated (e.g. aerially derived imagery, (Verner, 1984; Hodgson *et al.*, 2016)) or tested in new environments, few have been evaluated under the challenging conditions of flooded forests. To further facilitate surveys in flooded forests, techniques need to be adaptable to local circumstances and easily replicated so that data can be compared within and between regions (Larsen, 2016).

My aim was to identify the optimal survey approach for a broad census of multiple terrestrial taxa; mammals, reptiles and amphibians within flooded forests. I targeted these groups because they are the most understudied components of the terrestrial vertebrate community (Nowak, 2013; Rog *et al.*, 2017) and their traditional detection methods are more likely to be impacted by inundation than those that target birds (e.g., live traps versus observing species along transects). My approach involved a selection of detection techniques that are commonly used in terrestrial habitats. As there are issues in flooded forests that pose unique risks to species captured with these standard approaches – i.e., drowning – it is necessary to modifications the standard design to mitigate these risks. For example, identifying alternative techniques to detect reptiles and amphibians given pitfall traps cannot be used even above the high tide line due to hydraulic pressure. As such, we need to know if the approach is still successful in habitat that is under the influence of inundation. I therefore evaluated the effectiveness and the efficiency of these standard approaches, using mangrove forests as an exemplar of flooded forests. My goal was to determine the ability of techniques to detect the three taxonomic groups along a latitudinal gradient that encompasses temperate, subtropical and tropical regions. There are three reasons why it is important to test the effectiveness of the approach along this gradient: 1) tidal ranges vary significantly across the regions (0.1-3.5 m), 2) there is a latitudinal cline in vertebrate species richness that should be able to be reflected if the approach is effective and 3) the broad differences that occur in community composition of vertebrates along this gradient must be able to be detected by the sampling regime. My results provide insight into the value of rapid assessments in flooded forests and how surveys may be tailored to target specific taxonomic groups and deal with resource limitations (e.g., time and budget). In doing so, I seek to facilitate further biodiversity surveys that will improve knowledge of vertebrate fauna within these important yet threatened ecosystems.

Methods

Study system

Mangroves are woody plants that usually grow in tropical and subtropical latitudes along the land–sea interface of bays, estuaries, lagoons, and backwaters (Mukherjee et al., 2014). The survey area consisted of areas of mangrove that are regularly inundated, as well as those areas that are inundated during tidal extremes (e.g. king tides) where mangrove plants may occur sympatrically with adjoining terrestrial vegetation. Mangrove forests represent the extremes of challenges experienced when surveying flooded forests, with daily to seasonal cycles of saltwater inundation, physical challenges of tangled aerial root systems, soft and unstable substrates like muddy quagmires, and large predators. .

They therefore provide an ideal study system in which to trial and assess a survey design for flooded forests.

Study region

Survey sites were distributed along the eastern seaboard of continental Australia, where mangrove forests span a latitudinal gradient incorporating temperate, subtropical and tropical climatic regions. Data were collected at 10 sites in national parks (NP) that were >75 km apart (Figure 1). National parks were specifically selected on the assumption that these typically large reserves would harbour relatively intact and diverse communities of terrestrial fauna in each region. Survey sites were selected within 2 km of the nearest vehicle access in order to be easily accessible from the landward side on foot, and to include contiguous mangrove extent that spanned at least 500 m of shoreline.

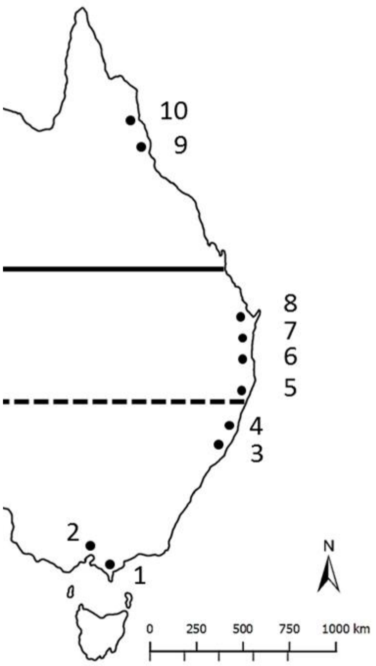


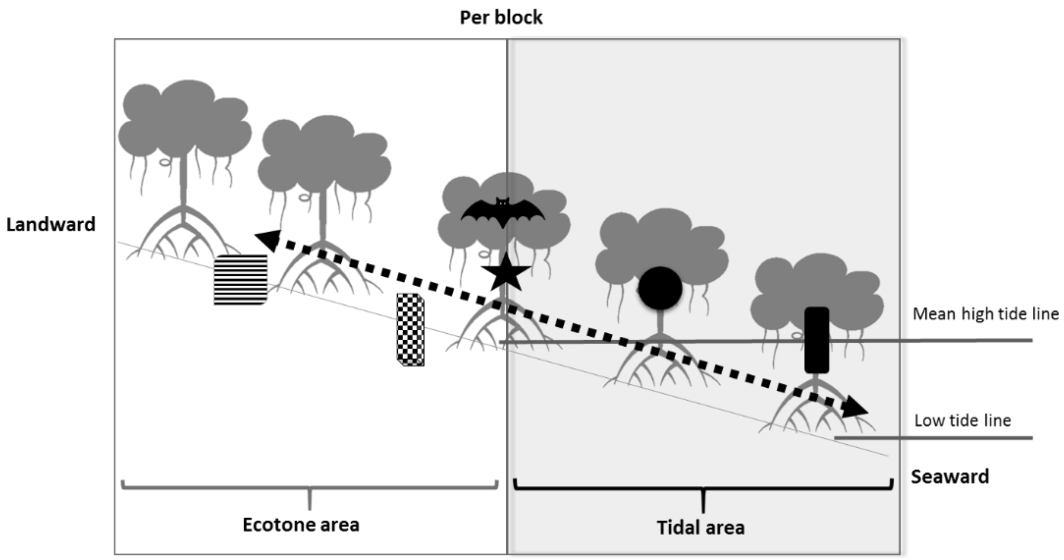
Figure 1. Survey sites in mangrove forests of eastern Australia where a rapid approach to detect terrestrial vertebrates in flooded forests was evaluated. The sites span a latitudinal gradient (15.56^{O} S to 39.13^{O} S) with the dashed line indicating the boundary between temperate and sub-tropical regions (-30^{O} S) and the solid line indicating the boundary between sub-tropical and tropical regions (-23^{O} S). Numbers correspond with the following sites: (1) Wilsons Promontory National Park (NP), (2) French Island NP, (3) Royal NP, and (4) Limeburners Creek NP, (5) Bundjalung NP, (6) North Stradbroke Island NP, (7) Bribie Island NP, (8) Poona NP, (9) Daintree NP, and (10) Annan River NP. Temperate to tropical zone boundaries follow (Corlett, 2013).

Techniques and execution rapid survey

I compiled a list of commonly used faunal survey techniques (Corn & Bury 1990; Garden et al., 2007) that collectively target the complete assemblage of the three target groups of terrestrial mammals, reptiles and amphibians (e.g. ground dwelling and arboreal, small to large; Table 1), and scored these techniques on an 11 point scale related to effort required related to set up, transport and check traps (see Supplementary table S1) A key criteria was that the techniques could be implemented within a maximum of 7 days. Targeted literature searches were used to identify less commonly used or newly emerging techniques that could potentially replace unsuitable techniques, which were scored in a similar manner as the other techniques (Supplementary material S1). I consulted with experts with knowledge in each taxonomic group about the suitability of potential techniques and possible adaptations. Pitfall traps (embedded 20 L buckets with removable lids plus drift fence) were initially deemed suitable for survey above the high tide line, but were abandoned because hydraulic pressure from ground water ejected the buckets from the substrate, even when three 50 cm long stakes were inserted diagonally into the substrate to hold the buckets in position. The number of survey nights and techniques and traps per site were chosen based on the ability to conduct the entire survey with two people in 7 days, seeking a compromise between detection probability and survey effort. Detection techniques and traps were deployed for four days and four nights. Incidental sightings were also gathered during the set up and removal of traps, such that observations were made for a total of six days. The surveys were carried out during the austral spring and summer of 2015/16 to coincide with the most active seasonal windows for the three taxonomic groups and the lowest tides, during which the largest area was accessible due to low tides reducing the area inundated. Surveys ran over multiple weeks so it was not possible to avoid rainfall events; however, surveys were timed to avoid any significant forecasted storm events.

Survey approach

The configuration of techniques was replicated using a block design (Table 1; Figure 2). Within each site, the design (Table 1) was placed within four 100 m wide and 50 m deep blocks spanning the high tide mark (25 m frequently inundated and 25 m mangrove ecotone area) (Figure 2). Four blocks were used for efficient set up and to estimate the total number of detection techniques and traps required to detect the maximum species richness per region. Blocks were separated by 20 m (Figure 2) and oriented parallel to the shoreline. To minimise the risk of fauna drowning during trap inundation, reconnaissance trips through the potential trapping area were made during two high tides to establish the tidal range before the survey commenced. Ground locations, and suitable elevations for the arboreal techniques, were marked during these trips and any incidental sightings (species observed directly or indirectly, through calls, scats or tracks) were recorded. To avoid interactions with estuarine crocodiles, techniques were placed away from deep channels and mudslides, and night transects were walked with a third person keeping watch in the tropical region.



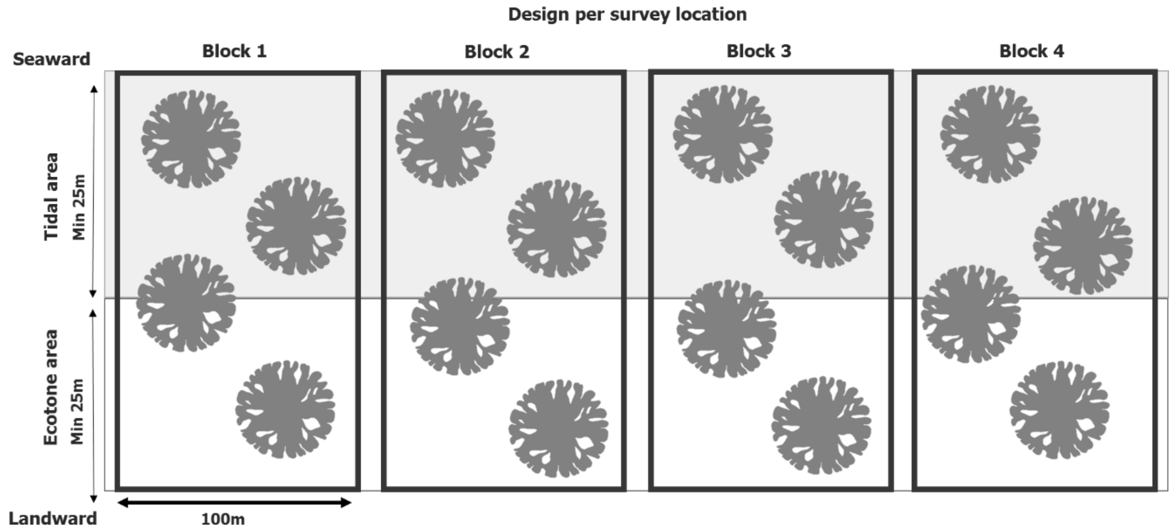


Figure 2. Blocked survey design employed in mangrove forests. The upper panel shows the placement of techniques in a cross-section view. The lower panel illustrates the block designs at each site from an aerial view. Trapping techniques within the ecotone were spaced 10 m apart and included 10 terrestrial refuges (striped square) and 10 live traps (blocked rectangle). A single camera trap (star) was placed at the boundary between the tidal and ecotone areas. One bat detector (bat) was placed per survey site, in the centre of the four blocks. Within the tidal area: 6 hair tubes (black circle) and 5 arboreal refuges (black rectangle), were placed on suitable branches. A single 100 m night transect (dashed arrow) was walked diagonally across each block to span both the tidal range and into the ecotone area. Incidental sightings were recorded across all four blocks.

Species identification for live captures and camera trap images followed Cogger (2014) and Menkhorst and Knight (2001). High frequency sonograms derived from micro bat recordings were viewed in Kaleidoscope Pro 4 (Wildlife Acoustics, 2015) and species identifications were verified by experts with access to reference libraries. Recordings unable to be identified to species level, owing to recording quality, poor separation of call structure or frequency between species pairs, were omitted. Hairs collected with hair tubes were identified to species by an expert based on identification of hair cross-sections.

Analysis

Effectiveness of survey approach

To assess the effectiveness of the survey approach, results were pooled by region (tropical, subtropical and temperate) (Figure 1). The effectiveness of the approach was assessed based on: (1) whether techniques and traps had overcome inundation events, (2) the number of species detected within each region and (3) the taxonomic composition of species relative to a global review of vertebrate taxa in mangrove forests (Rog *et al.*, 2017); (4) whether the mean number of species detected reached an asymptote by the fourth trap night; and (5) the number of blocks where species were detected.

Effectiveness per technique - Species richness, unique species and trap success

Effectiveness for each technique was judged relative to the detection success for each technique, pooling results from across all ten sites. Detection success was assessed as: 1) the mean number of species detected by each technique, (2) the mean proportion of unique species detected by the technique (i.e. those species that were not detected by any other technique) (calculated from the regional averages because species distributions may not cover the whole survey area but did overlap within regions), and (3) trap success (the mean number of individuals detected by a technique during four trap nights, across all sites). The bat detector and the incidental sightings were not included in the trap success calculations as the number of individuals detected could not be determined, nor standardized according to effort.

Efficiency of survey approach – return on investment

Return on investment was defined as the mean of number of species (pooled across the 10 sites) per AU$1,000 spent. The total cost of each technique were calculated as: (1) initial investment (the average costs involved in purchasing equipment based on the most expensive and cheapest model available within Australia; plus the (2) ongoing costs - field time (preparation and setup of detection techniques and traps at a site, setting and checking each survey day and removal of the traps at the end of survey), and post-survey time (data processing to complete species identifications). The cost of field and post-survey time were calculated based on a hypothetical wage of AU$20 per hour. Differences in costs among techniques were assessed using a Poisson General Linear Model.

The return on investment for each survey method was used to determine the most effective combination of techniques to: (1) maximize species richness and (2) minimize costs. This assessment was calculated for ongoing costs only, as equipment costs can vary significantly between countries and may or may not be available. These data were also used to assess the optimal design for all taxa, as well as mammals, reptiles and amphibians separately. The differences in return on investment per taxon for the different techniques were assessed using a Poisson GLM with a log linear function. Incidental sightings were excluded from the efficiency calculations as they occurred during other activities related to the survey approach.

Results

Effectiveness of survey approach

All techniques were found to avoid inundation in flooded forests when their field placement was selected based on the 1-2 day tidal range scouting and flagging, with 0 detection techniques or traps inundated over 5,160 trap nights. No encounters with estuarine crocodiles were experienced during the surveys.

Species richness detection and assemblage composition

As expected, the mean number of species detected varied across regions, with more species detected in the tropical region ($\bar{x}$ = 20.5 ± 1.93 SE) when compared with the sub-tropical ($\bar{x}$ = 11.5 ± 1.09 SE) and temperate ($\bar{x}$ = 8.5 ± 1.28 SE) regions (Figure 3A). Across all regions, mammals were the most frequently detected taxa (Figure 3A), accounting for 64% of all species detected, followed by reptiles (29%) and amphibians (6%). See Supplementary material S2 for details of the families detected by each technique and Chapter 4 for the specific species detected at each site).

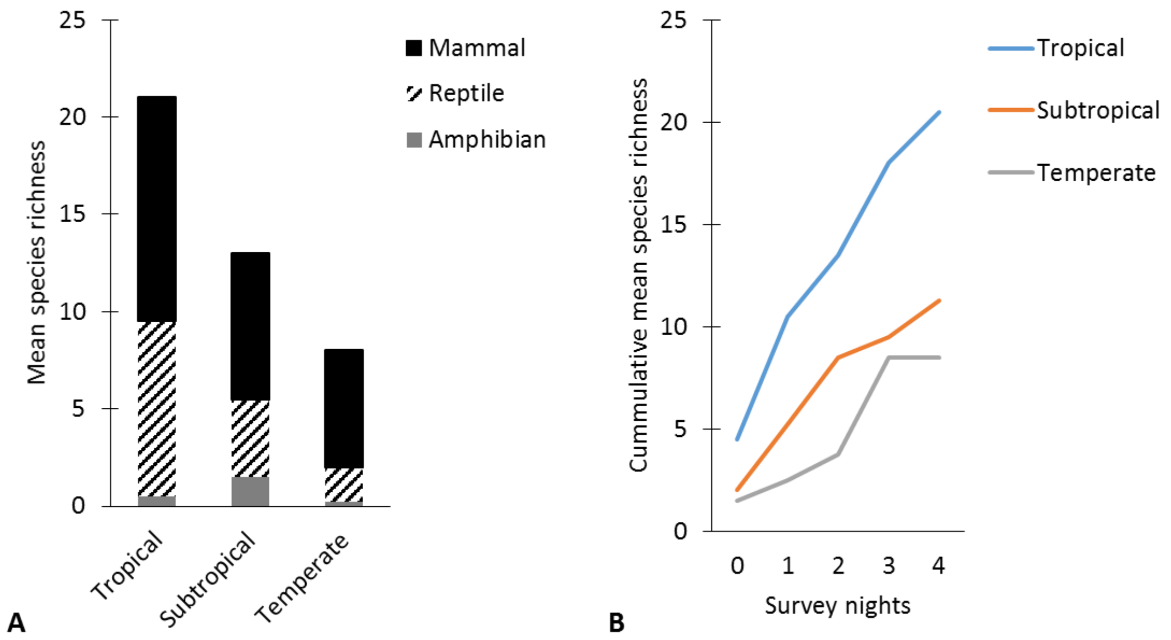


Figure 3. The mean species richness detected by the full survey approach within each region during the survey period for: (A) mammals, reptiles and amphibians within eastern Australia mangrove forests, and (B) cumulative across each survey night. Survey nights commence at 0 to account for incidental sightings taking place 1-2 days prior to the trapping events. Species detected by the hair tubes and bat detector were excluded from B because hair tubes were only checked at the end of the four nights and calls were of inconsistent quality across the four nights.

Survey duration

No new species were detected on the fourth night of trapping in the temperate region, although there were in the sub-tropical and tropical regions (Figure 3B), suggesting further survey effort may yield greater species richness in these regions. However, some techniques in the subtropical and tropical regions failed to detect new species on the fourth night of survey (e.g., arboreal refuges and live traps, Supplementary material S3).

Number of blocks

In all regions, every block detected at least one species using at least one technique. However, for most techniques, it was rare for a species to be detected in all blocks (Figure 4). Most blocks detected at least one unique species in the topical region (Figure 4) an outcome that was consistent with more species being detected in the tropics generally (Figure 3A).

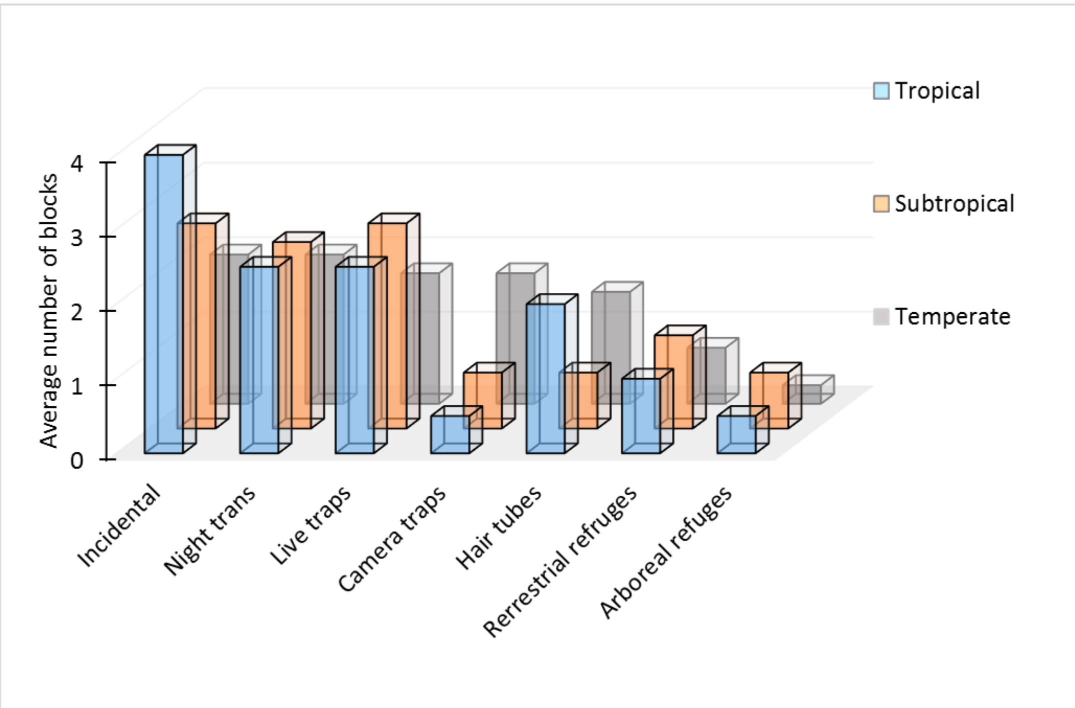
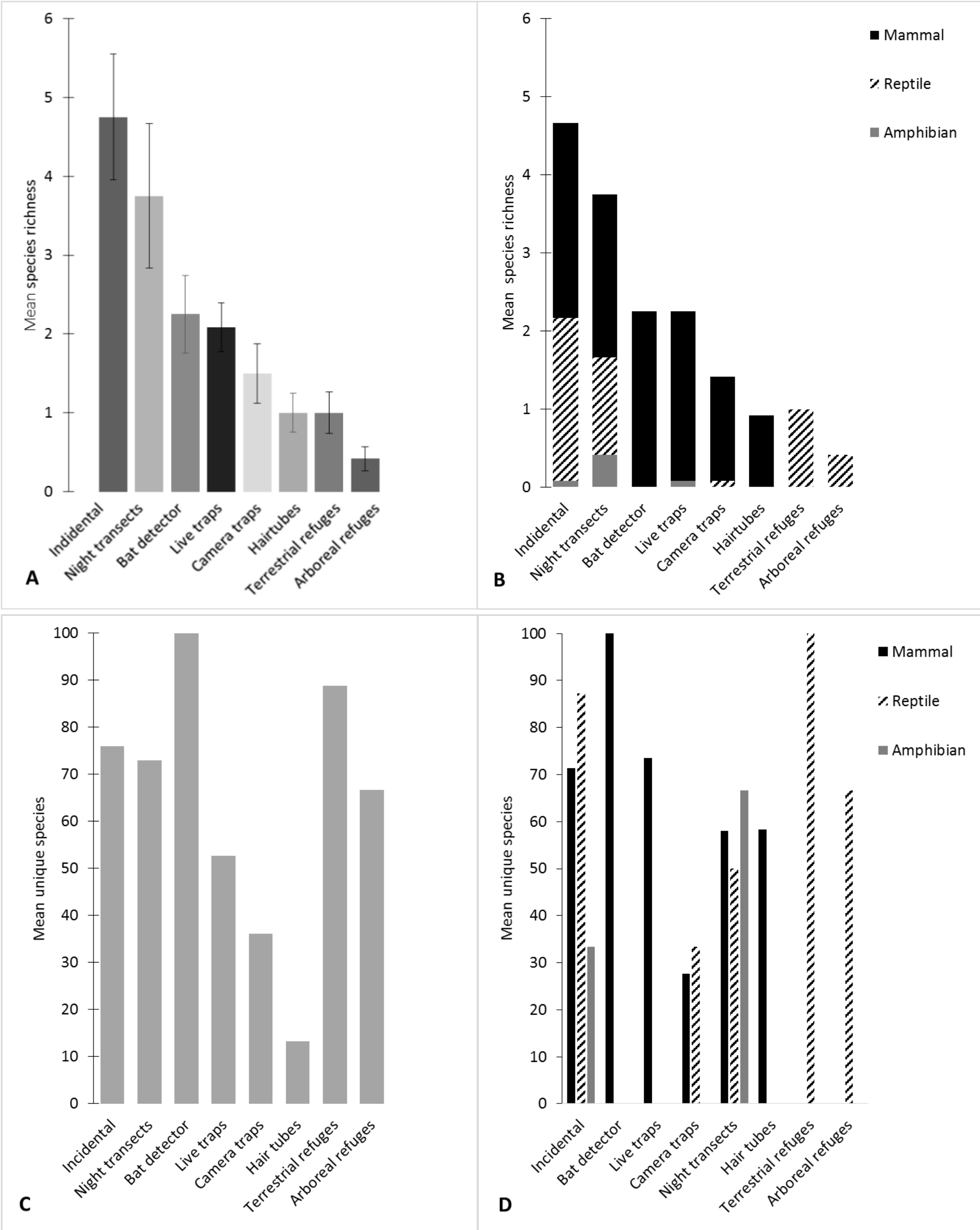


Figure 4. Average number of blocks per technique that detected at least one mammal, reptile or amphibian species per region. Number of traps per block are detailed in Table 1.

Effectiveness of techniques

Species richness, unique species, trap success

Incidental sightings detected the highest species richness of all techniques (Figure 5A), followed by night transects and the bat detector. Assessing species richness across all sites masks some regional variability between techniques (e.g., hair tubes detected more species in the temperate region than the bat detector; Supplementary material Figure S4.1A). Some techniques failed to detect some of the target taxa (e.g., terrestrial and arboreal refuges did not detect amphibians (Figure 5B, Table 1)). While there was a degree of overlap in the species detected by each survey method, all methods detected at least one species not detected by any other method, which varied per region (i.e., unique species; Figure 5C and Supplementary material Figure 4.2A). Some techniques detected solely unique species e.g. terrestrial refuges (Figure 5D). Of the detection techniques night transects yielded the highest trap success (Figure 5E).



A
Mean species richness
0
1
2
3
4
5
6
Incidental
Night transects
Bat detector
Live traps
Camera traps
Hairtubes
Terrestrial refuges
Arboreal refuges
B
Mammal
Reptile
Amphibian
Mean species richness
C
Mean unique species
0
10
20
30
40
50
60
70
80
90
100
Hair tubes
D

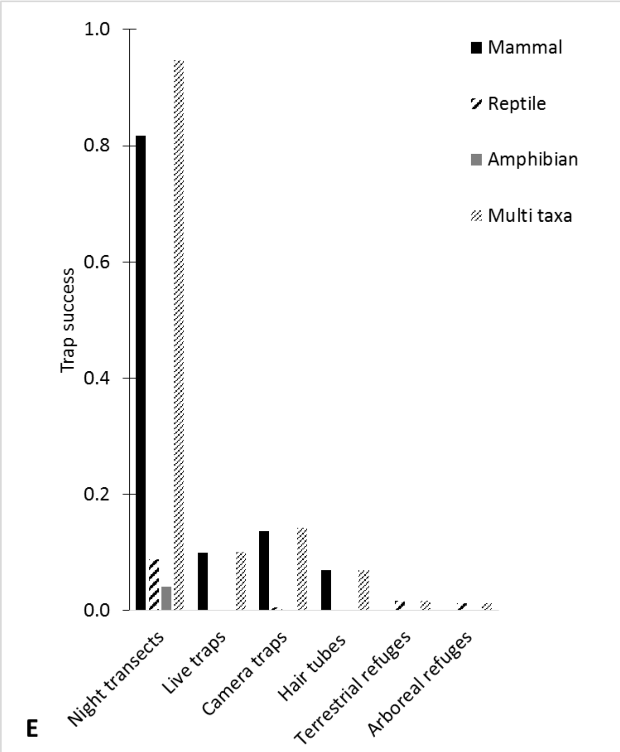


Figure 5. The mean number of species detected across all 10 sites for: (A) all taxa; (B) mammal, reptiles and amphibian; (C) those species not detected by any other technique (unique species); and (D) unique species in each taxonomic group, and (E) Trap success (i.e., the mean number of individuals detected by a technique during four trap nights across all sites).

Effectiveness by taxonomic group

Four techniques detected all the target taxonomic groups predicted, and three of these technqiues also detected groups that were not anticipated (Table 2). Nevertheless, all target vertebrate groups were detected by at least one technique. While there was often some redundancy in the techniques that detected small and medium sized species, larger bodied species were often detected by only a single technique (Table 2).

Mammals

Six techniques out of eight detected mammals across all regions; incidental sightings, live traps, bat detector, night transects, camera traps and hair tubes (Figure 5B). The highest richness and amount of assemblages for mammals was detected by incidental sightings (Figure 5B), only failing to detect medium sized arboreal mammals. Small and medium sized mammals were detected by a wide range of techniques, althought incidental sightings were the only way large arboreal mammals were detected (Table 2). The night transects had the highest detection success (Figure 5E), although no large mammals were detected with this technique (Table 2). Each of these six techniques detected unique mammal species (Figure 5D). Live traps, hair tubes and camera traps were effective at detecting mammals from both strata, which was not anticipated (Table 2).

Reptiles

Five techniques detected reptiles; incidental sightings, camera traps, night transects, terrestrial and arboreal refuges (Figure 5B). Terrestrial refuges were added to capture ground strata reptiles in the absence of pitfall traps, and while detection success was low (Figure 5E), they captured unique species not detected by other techniques (Figure 5D). However, they were not able to detect medium-sized ground dwelling reptiles (Table 2). The greatest richness and amount of assemblages of reptile species was detected by incidental sightings, which only failed to detect large ground dwelling reptiles. Night transects were also highly effective, detecting reptiles almost five times as often as other techniques (Figure 5E), but only from two of the targeted body sizes and strata (Table 2). Multiple techniques detected most small and medium sized reptiles, but only camera traps detected medium sized arboreal reptiles or large ground dwelling reptiles (Table 2).

Amphibians

Amphibian richness was low, and only three techniques detected amphibians; night transects, incidental sightings and live traps (Figure 5B), with a noticable non-detection of frogs by terrestrial and arboreal refuges (Table 2). Two of the techniques detected unique species (Figure 5D), and live traps unexpectedly detected ground dwelling amphibians (Table 2). The highest species richness was detected using night transects (Figure 5E), and both night transects and incidental sightings were effective for amphibians in both strata.

Efficiency of survey approach/costs

The equipment costs and ongoing costs varied significantly among the survey techniques (χ^2 = 7299.229, *df* = 1 and *P* = *<0.001*; Figure 6). The costs related to equipment purchase contribute to the largest differences (Figure 6). Relatively lower ongoing costs related to camera traps and the bat detector (Figure 6) are explained by less time for set up and checking in the field, even though these techniques have higher costs associated with post-survey processing time (Figure 6). Equipment, preparation, and field costs for incidental sightings are by definition negligible, as they take place during other activities related to the survey approach (Figure 6), therefore only costs assosicated with post survey data processing for incidental sightings are displayed.

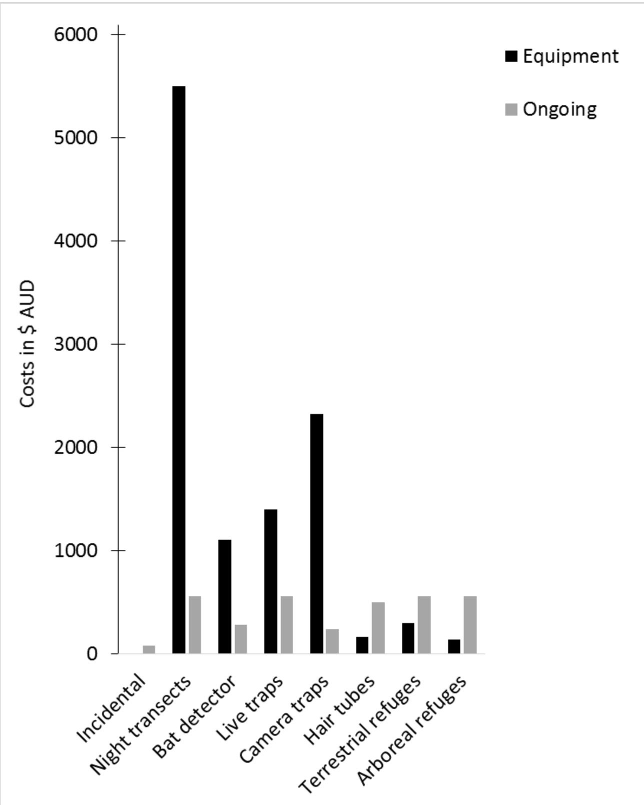


Figure 6. Cost in Australian dollars of each technique per survey site for 4 nights (numbers of traps shown in Figure 1). Shown separated by equipment costs and ongoing costs (field costs, post-survey data processing).

Efficiency of survey approach – return on investment

In the multi taxon approach the equipment costs (Figure 6) vary significantly per technique. The return on investment for ongoing costs per technique varied significantly by taxon (χ^2 = 975.423, *df* = 6 and *P* = <0.001). The bat detector had the highest return on investment, for both the multi taxa and mammal approach (Figure 7). This reflects the large number of species detected and the relatively low ongoing costs. The night transect technique is the most efficient technique for detecting reptiles, followed by terrestrial refuges, while night transects remain the most efficient way to detect amphibians (Figure 5B). The efficiency of techniques ongoing costs did vary slightly between regions although there were no consistent patterns observed (Supplementary material S6.1A).

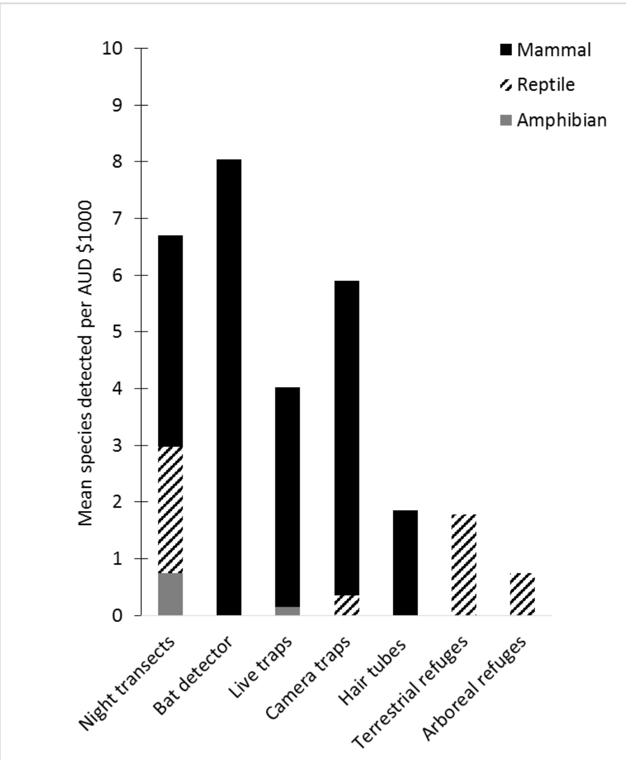


Figure 7. Mean return on investment (species detected per $1000) by taxon for each technique, measured using ongoing costs (field time (preparation and setup of traps at a site, setting and checking each survey day and removal of the traps at the end of survey)), and post-survey time (data processing to complete species identifications).

Discussion

While flooded forests are a challenging environments to survey due to their frequent inundation, I demonstrate that my survey approach including terrestrial survey techniques proved successful at detecting a wide range of terrestrial vertebrates. Scouting and marking the tidal range to inform the placement of detection techniques and traps successfully mitigated the risk of animals drowning and equipment being lost due to inundation. All of the techniques were effective at detecting a range of target taxa, except for pitfall traps, which failed due to the hydraulic pressure associated with saturated substrate and were removed from the design. Risks to fieldworkers from predators were successfully mitigated by adding an additional person when surveying the tropics, and placing traps away from deep channels. My approach shows that valuable empirical information about species richness in flooded forests can be collected through a rapid field approach to complement the largely anecdotal occurrence records that already exist (Rog *et al.*, 2017).

Effectiveness of survey approach

I was able to detect a range of mammal, reptile and amphibian species across the 10 sites. The richness patterns I detected are in line with global ecological patterns for mangrove forests, with significantly higher mammal species richness relative to other taxa, and richness increasing from temperate to tropical regions (Rog *et al.*, 2017). Some minor differences in the effectiveness of different techniques were found between regions. For example, amphibians were not detected in the tropics despite being present in these environments in other countries (Rog et al. 2017). However, the failure to detect amphibians may be explained by dry conditions during the survey period at tropical sites. Or may indicate lower detection with artificial refuges than might be expected with pitfall traps (Corn & Bury, 1990). However, the patterns in the effectiveness of techniques did not differ between regions (Supplementary material S6), suggesting this approach can be implemented successful across the range of climate regions where mangroves occur. The mammal richness I detected with live traps in the tropical region is comparable with results from Metcalfe (2007) who carried out mammals surveys with live traps in tropical mangroves in the Northern Territory of Australia. The species richness of bats I detected in the tropics was lower than expected from other bat surveys in tropical mangroves conducted in Western Australia (McKenzie, 2012) and Northern Australia (Melkalfe, 2007). This is potentially explained by the fact that surveys ran over multiple weeks so it was not possible to avoid rainy or windy nights that might have influenced bat activity. Additional surveys that target mammals, reptiles and amphibians in tropical as well as subtropical and temperate mangroves would allow for a better interpretation of the representativeness of my results.

My data show that less than 7 days and 4 trap nights is sufficient to detect a wide range of species from different taxonomic groups. While as little as two to three survey nights has been shown to add valuable data for conservation (Skalak *et al.*, 2012; Roberts & Daly, 2014), I found evidence that the number of trap nights may need to be adjusted to the anticipated species richness. Four trap nights were sufficient in temperate sites (Figure 3B), where species richness was lower (Rog et al. 2017). However, as species richness increased, new species were being detected on the fourth night (Figure 3B), suggesting that more trap nights could be of value in tropical and sub-tropical regions.

Importantly, there is likely to be little in the way of efficiency savings by reducing the survey length because field time is a minor component of the cost (Figure 6). I would also caution against reducing the number of blocks or detection techniques and traps used in the design, as all blocks detected species across all regions (Figure 4), meaning that any reduction will likely impact the effectiveness of richness estimates. I found that there was greatest value in implementing the full design, including the full suite of techniques, if the aim was to estimate species richness. All techniques were successful at detecting at least one species in their target groups (Table 2), and were highly complementary, identifying unique species not detected by other techniques (Figure 5D).

Efficiency of survey approach

To detect the highest richness and the full assemblage of target taxonomic groups in flooded forest, I found that all techniques in the design are required. The value of using multiple techniques to detect the broadest range of species is in line with studies in other habitats (e.g., mountain valley; Manley *et al.*, 2004). The complementary nature of the species detected by different techniques offers the potential to tailor the approach to target specific taxonomic groups, and to accommodate budget limitations.

Equipment, field time and processing time

The return on investment for different survey techniques was strongly driven by the cost of purchasing equipment (Figures 6 & 7), highlighting the importance of considering the different types of costs (equipment, field time and processing time) when selecting the most efficient survey design. For example, camera traps are costly to purchase but require less field time as they can be deployed for long periods of time before checking, as opposed to live traps. The trade-off between detecting more species by leaving camera traps in the field for longer (e.g., by detecting more elusive species; Hamel *et al.*, 2013) and incurring higher post-survey data processing costs, can provide important information when selecting the optimal survey period for a given budget.

In designing the most efficient survey approach for particular objectives and budgets, it is important to remember that there are some fixed costs related to the tidal range scouting time. These costs, as well as the time required to setup and check traps, may vary between regions due to differences in the complexity of the vegetation (e.g., both plant species richness and structural complexity is higher in tropical regions) influencing the ease of movement through the site. Likewise, costs will be higher in regions where the risk to fieldworkers from large predators (e.g., crocodiles, tigers) requires a third person to keep watch. The addition of a "control" carried out in adjacent, non-inundated habitat that is conducted in parallel with the surveys in flooded forests would enable comparison of the return on investment, which could then be used to calculate the impact of inundation on survey efficiency. Nevertheless, the relative cost of the techniques evaluated in this study will remain the same, which should enable researchers to identify the most efficient design for their context.

Incidental sightings

While any variation on the survey design includes the same fixed costs, I found that the time spent in setting up and implementing the design contributed valuable incidental sightings, covering the broadest range of target taxa of any technique (Table 2). Incidental sightings are known to increase species detected during faunal surveys, often adding species not detected by other survey techniques (Larsen, 2016), which was the case for my study (Figure 5, Supplementary material S2). Thus, even when there is no budget to purchase equipment, simply spending time in the field will contribute valuable information on the presence of several different taxonomic groups. Incidental sightings include capturing indirect signs of species, such as tracks and scats, and can therefore be less effective in the flooded season of seasonally flooded forests (i.e., flooded for prolonged periods of time) relative to forests that experience daily inundation (e.g., mangroves) because the forest floor is not be visible during inundation.

Redundancy

I found that multiple techniques were able to detect some of the target taxa (e.g., six techniques detected small mammals; Table 2). This may enable some efficiencies to be achieved when the aim is to determine whether specific taxa are present, rather than assessing the full assemblage of different sizes and different strata. Together night transects and incidental sightings were able to detect species from the full assemblage (Table 2). However, I found that all techniques detected unique species (Figure 5D, so the omission of any single technique would underestimate species richness. For example, live traps proved to be the most efficient technique, detecting the greatest richness, particularly of mammals (Figure 5B). While this technique also detected the largest number of unique species (Figure 5D), relying on live traps to detect mammals would mean that bats medium-sized and large mammals would not be detected. Several of these groups could still potentially be detected through incidental sightings but could not fully compensate if other techniques were omitted.

Considerations for single taxon group approaches

Few studies have quantified the return on investment of survey approaches. Even the few studies that have considered costs have assumed equal survey time and costs, and fail to breakdown result by taxonomic group (e.g., Rohr *et al.*, 2007). This breakdown is important for those wanting to adapt the techniques for different objectives and constraints, because I found considerable differences among techniques in how efficiently they detected different target groups (Fig 7). For mammals, efficiency is largely similar to the multi taxa approach because are the most common and drive the richness patterns (Figure 3A). For reptiles and amphibians however, less costly techniques can be used (e.g. night transects do not require a thermal imaging camera for these taxa). However, the low detection of these groups by the artificial refuges suggest that surveys targeted at reptiles and amphibians may need to consider adding additional techniques, such as day transects, or target surveys to warmer, wetter conditions (e.g., Spence-Bailey *et al.*, 2010), to compensate for the loss of pitfall traps from the design. More work is needed to determine the most effective and efficient ways to detect reptiles and amphibians in flooded forests.

Limitations

My data demonstrates that a rapid approach is capable of providing significant information on a broad range of taxonomic groups in flooded forests. Nevertheless, there are some limitations of the current design that need to be considered for future surveys. Little is known about detection and trapping techniques of amphibians and reptiles in flooded forests with which to compare the effectiveness of my approach for these taxa. Anecdotal records suggest that these taxa are much less common than mammals in mangroves (Rog et al. 2017) and the richness I detected followed this pattern (Fig 3A). However, it is not clear whether this genuinely reflects low richness for these groups or widespread limitations in my ability to detect these groups. Furthermore, there is almost no information about these groups in different types of flooded forests. The low richness of medium and large reptiles may be because of the depauperate understorey in flooded forests, providing little habitat for this assemblage (Luiselli & Akani, 2002). Terrestrial refuges detected small reptiles, including a high portion of species other techniques did not capture (Figure 5D). I was able to detect reptiles using this technique within the four days despite the fact they are generally placed in the field for a longer period (Supplementary material S3) (O'Donnell & Hoare, 2012). It was hoped that artificial refuges would function as a replacement for the pitfall traps because they are known to detect both reptiles and amphibians (Hampton, 2007), although only two studies have tested the effectivenes of aboreal refuges for these groups (Bell, 2009; Tomlinson, 2012). However, the limited success of this technique in my study suggests more research is required to understand their effectiveness in flooded forests, particularly in saline habitats.

The detection of arboreal and ground dwelling assemblages could have been effected by the tidal height during the night transects. Night transects were walked evenly during both low and high tides on the four temperate and the four sub-tropical sites, as the surveys were conducted in a consecutive order. They were walked during low tide at the two tropical sites these to avoid encounters with crocodiles. The effect of tides on present assemblages and thereby on their detection requires further investigation. The effect of inundation also needs to be considered for seasonally flooded forests were prolonged periods of inundation likely influences the ground dwelling assemblages of mammals, reptiles and amphibians. This situation dictates the elimination of ground based techniques (e.g., ground covers) while other techniques can be adjusted to target arboreal assemblages during these inundated periods (e.g., live traps and camera traps).

Conclusions

My results present the first evaluation of survey techniques to detect terrestrial vertebrates in flooded forests. Further studies are required to verify my results and whether they are a representation of fauna richness in flooded forests. Nonetheless the patterns in taxonomic assemblages detected mirrors that of global patterns (Rog *et al.*, 2017) implying the composition of taxonomic groups is likely representative. By detailing a rapid and flexible approach, which can be tailored to a range of different objectives and constraints, I hope this study will facilitate the expansion of knowledge about the poorly surveyed terrestrial vertebrate fauna that inhabit flooded forests. My findings highlight a need for further field studies to build knowledge on the effectiveness and efficiency of detection techniques in these and similar ecosystems. A critical next step is to adapt the technique to explore the diversity and ecological role of terrestrial fauna in these ecosystems, with particular emphasis on the resources they rely on and the services they provide to these ecosystems to aid deeper understanding of the importance of these taxa to the health of flooded forests. By building empirical data on the species richness of these ecosystems, I hope that effective conservation measures will be developed for these important, yet under-studied ecosystems.

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Table 1. Techniques in the survey approach for tidal forests, including survey nights (techniques and traps per site for four nights); target taxa; target body size (small circle: <0.5 kg, medium circle: 0.5-2 kg, large circle: >2 kg) and target strata (ground, over storey). Taxonomic key: kangaroo = mammals; lizard = reptiles; frog = amphibians.

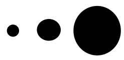
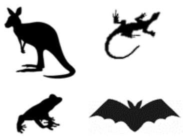
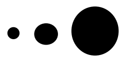
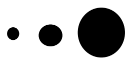
Technique	Method	Traps nights	Target taxa	Body size	Strata
Live traps	Trap lines of 10 traps, 10 m apart, baited with peanut butter and oats.	40			
Hair tubes	Attached to branches, baited with peanut butter ~1.8 m height.	24			
Camera traps Buckeye camera	Placed 30 cm above ground, baited with peanutbutter/oats.	4			
Terrestrial refuges	Traplines of 10 corrigated metal sheets, 10 m apart.	40			
Arboreal refuges	Flexible foam attached to tree truncks (~1.5 m height).	20			
Bat detector model Anabat Em3	Attached chest height on tree, run during night transects.	1			
Night transects	Two people, one spotlight one thermal imaging camera.	400 m			
Incidental	All opportunistic sightings of tracks, scats and animals.	n/a			

Table 2. Target taxa (small = <0.5 kg, medium sized = 0.5-2 kg, large = >2 kg; ground and over storey refer to target strata) detected by techniques in the survey approach for tidal forests. White cells indicate where technique was predicted to be effective, grey cells where it was not predicted to be effective (see Table 1). Tick indicates target group detected. Cross indicates target group not detected when predicted to be effective.

Target group	Incidental sightings	Bat detector	Live traps	Camera traps	Night transects	Hair tubes	Terrestrial refuges	Arboreal refuges
Mammals								
Small, ground	✓		✓	✓	✓	✓		
Small, over storey	✓	✓	✓		✓	✓		
Medium, ground	✓			✓	✓	✓		
Medium, over storey	✗			✓	✓	✓		
Large, ground	✓			✓	✗			
Large, over storey	✓				✗			
Reptiles								
Small, ground	✓				✗		✓	
Small, over storey	✓				✓			✓
Medium, ground	✓			✓	✗		✗	
Medium, over storey	✓				✗			
Large, ground	✗				✓			
Large over storey	✗				✗			
Amphibians								
Small, ground	✓		✓		✓		✗	
Small, over storey	✓				✓			✗

Chapter four: Rapid surveys of terrestrial vertebrates provide critical information for management strategies of mangrove ecosystems

Not published as of yet

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Key words: biodiversity assessment, ecological monitoring, intertidal, resource use, species richness, transboundary

Abstract

Mangroves span marine and terrestrial environments, and so support a wide range of fauna from both realms. Knowledge on the terrestrial species that occupy these declining forests is crucial for their conservation and management, yet has received little attention. I sought to assess the value of field surveys to address this knowledge gap, aiming to generate data that will inform the conservation and management of mangroves.

I implemented rapid field assessments in ten national parks distributed along a latitudinal gradient in eastern Australia. I documented the presence of terrestrial vertebrate species and their use of mangroves, and compared this to existing knowledge at these sites using interviews with site managers.

Significantly more species than were previously thought to occur in mangroves were detected, including both threatened and invasive species. Despite identifying species of conservation concern, that likely degrade mangroves, sites had few explicit management actions to protect mangroves.

I found a correlation between fauna richness and mangrove plant richness, with evidence that the richness of adjacent habitat potentially influence the vertebrates using mangroves based on the prevalence of non-mangrove specialists.

In order to facilitate holistic and effective management of these ecosystems, it is imperative that we improve our understanding of the terrestrial vertebrates utilizing mangrove habitats. I argue that rapid field assessments are an effective avenue to establish baseline knowledge about the species present, which can inform the management and conservation of mangrove forests.

Introduction

Mangrove forests span the intertidal zone in many tropical, subtropical and temperate regions around the world (Spalding et al., 2010). They provide a range of high value ecosystem services (Salem & Mercer, 2012), including coastal protection (Mazda et al., 2002), carbon sequestration (Alongi, 2014) and the provision of nursery grounds for local fisheries (Carrasquilla - Henao & Juanes, 2017). Yet globally, these forests are rapidly declining Hamilton, S.E. & Casey, D. (2016)). Remaining areas of mangrove forests are experiencing increasing degradation due to a range of threats associated with both marine and terrestrial pressures (Rog & Cook, 2017). These include direct threats, such as clearing to accommodate increasing urbanisation and agriculture (UNEP 2014, Thomas et al., 2017), as well as emerging threats such as those related to a changing climate (e.g., sea level rise and dieback associated with rising sea surface temperatures; Alongi, 2015). In order to effectively protect and manage remaining mangrove forests, this diverse range of threats requires explicit management objectives, set within robust management frameworks (Friess et al., 2016; Rogers et al., 2016). Currently, the required objectives and frameworks for effective management are demonstrably lacking (Rog & Cook, 2017).

The importance of mangroves as a habitat for terrestrial fauna has traditionally been overlooked due to ta focus on their marine fauna (Nagelkerken, 2008). This awareness is however changing with a recent study demonstrating that mangroves support over 460 species of terrestrial mammals, reptiles and amphibians globally - five times more species than previously reported (Rog et al., 2017). Over one third of the terrestrial fauna reported in mangroves is considered globally threatened (Rog et al., 2017), including species that are described to be using mangroves as a last refuge after the loss of their primary habitat (e.g., Zanzibar red colobus, Procolobus kirkii; Nowak, 2013). Patterns in floral and faunal species richness suggest that the importance of mangroves to terrestrial vertebrates continues to be underestimated, especially given a dearth of field studies (Rog et al. 2017). Faunal species richness in mangroves is also likely to be influenced by the richness of adjacent ecosystems, as most terrestrial vertebrates are facultative users (i.e., utilise only some resources) of mangrove ecosystems (Rog et al., 2017). Therefore, the availability of resources both within and adjacent to mangrove forests is potentially important in predicting faunal richness in these ecosystems. It is essential to effective conservation planning, and to the identification of management priorities, that data on species richness are coupled with information on species identity and ecology (Fleishman et al., 2006). A lack of knowledge concerning which threatened species occur in mangroves and how strongly they depend on the resources these forests provide, is a critical barrier to their effective conservation and management. Likewise, a lack of information concerning the essential resources terrestrial vertebrates provide to mangrove forests (e.g. nutrient transfer, Reef et al., 2014; pollination, Ashraf, 2013), leaves a gap in our understanding of the functional ecology of these ecosystems.

Mangroves span the boundary between land and sea, making these ecosystems vulnerable to poor governance structures due to confusion about whether their management falls under terrestrial or marine jurisdictions (Rog & Cook, 2017). The lack of a clear governance framework for mangrove ecosystems may also relate to the physical challenges of access due to complex and often obstructive vegetation structure, deep mud, frequent inundation and a general lack of infrastructure. Managers are likely to know more about the ecosystems they access regularly (Hockings, 2006) and so if site managers rarely visit mangrove forests, this reduces the likelihood that local ecological knowledge is obtained opportunistically (Cook et al., 2012). Likewise, a general lack of site visits potentially hampers responsive management by missing emerging issues, such as detecting new infestations of invasive species, or illegal activities (e.g. logging) (Hockings et al., 2004). A reduced capacity to detect emerging issues, the lack of a clear governance framework and poor ecological understanding, mean mangrove systems are particularly vulnerable to degradation. Here we sought to document the richness of terrestrial vertebrates in mangrove forests in Australia, through the development and implementation of rapid fauna surveys. We aimed to determine the extent to which the assemblage of mammals, reptiles and amphibians, and the resources they use, could be characterised. To understand how existing knowledge of these ecosystems compared with data gathered during rapid assessments we also conducted semi-structured interviews with site managers. Specifically, these interviews sought to assess managers' current knowledge regarding terrestrial vertebrate occupancy within mangrove forests that are managed for conservation purposes.

Methods

Study region and survey techniques

As detailed in Chapter 3, I undertook rapid field surveys across ten national parks along the eastern seaboard of Australia (Figure 1) targeted at detecting terrestrial mammals, reptiles and amphibians in mangrove forests. Birds were excluded from the design as this group is relatively well studied in these ecosystems (Mohd-Azlan *et al.*, 2012) see Chapter 2). These surveys spanned a latitudinal gradient from 15.56^{O} S to 39.13^{O} S, with sites representing the tropical, subtropical and temperate climatic regions in which mangroves forests occur. National parks were specifically selected on the assumption that these typically large reserves would harbour relatively intact and diverse communities of terrestrial fauna in each region, including relatively intact adjacent terrestrial vegetation communities. Survey sites were selected within 2 km of the nearest vehicle access to be accessible from the landward side on foot, and to include contiguous mangrove extent that spanned at least 500 m of shoreline.

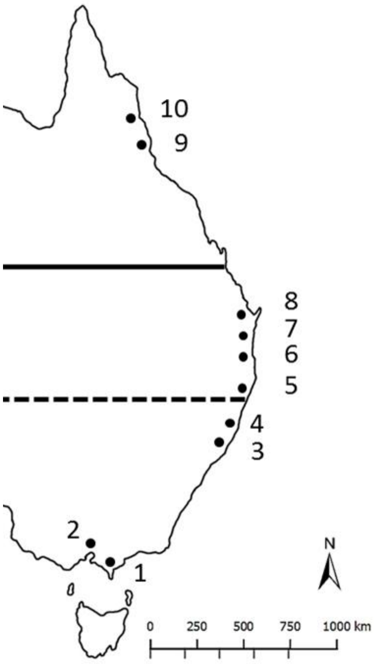

Figure 1. Survey sites in mangrove forests along eastern Australia. The sites span a latitudinal gradient of 15.56^{O} S to 39.13^{O} S. (1) Wilsons Promontory National Park (NP), (2) French Island NP, (3) Royal NP, (4) Limeburners Creek NP, (5) Bundjalung NP, (6) North Stradbroke Island NP, (7) Bribie Island NP, (8) Poona NP, (9) Daintree NP, and (10) Annan River NP.

The survey design utilised a range of techniques that would target all of the desired taxonomic groups on the condition the techniques could be adapted to avoid inundation, thereby reducing risk to fauna or equipment (see Chapter 3). The design involved a series of 4 blocks, each with a combination of 10 live traps, 10 artificial terrestrial refuges (45 x 50 cm^2 corrugated metal sheets), 6 hair tubes, 5 artificial arboreal refuges (50 x 70 cm^2 foam pads wrapped around tree trunks or limbs), 1 camera traps, and a 100 m night transects (two observers using one handheld spotlight and one thermal imaging camera) (Figure 2). A single bat detector was placed at the centre of the 4 blocks (Figure 2; and Chapter 3 Survey design). Two days were spent scouting the tidal range to identify appropriate trap placement. Traps were deployed continuously for four days and nights. I recorded any incidental sightings of species and any evidence of resource use during the 6 days required to set up, check and remove all the traps. Field work was carried out during the Austral spring and summer of 2015/16 to coincide with the most active seasonal windows for the three taxonomic groups. The survey periods spanned multiple weeks so it was not possible to avoid rainfall events; however, surveys were timed to avoid any significant forecasted storm events.

Species identification for live captures and camera trap images followed Cogger (2014) and Menkhorst and Knight (2001). High frequency sonograms derived from micro bat recordings were viewed in Kaleidoscope Pro (Wildlife Acoustics, 2015) and species identifications were verified by experts with access to reference libraries. Recordings unable to be identified to species level owing to recording quality or poor separation of call structure or frequency were omitted. Hairs collected with hair tubes were identified to species by an expert based on identification of hair cross-sections.

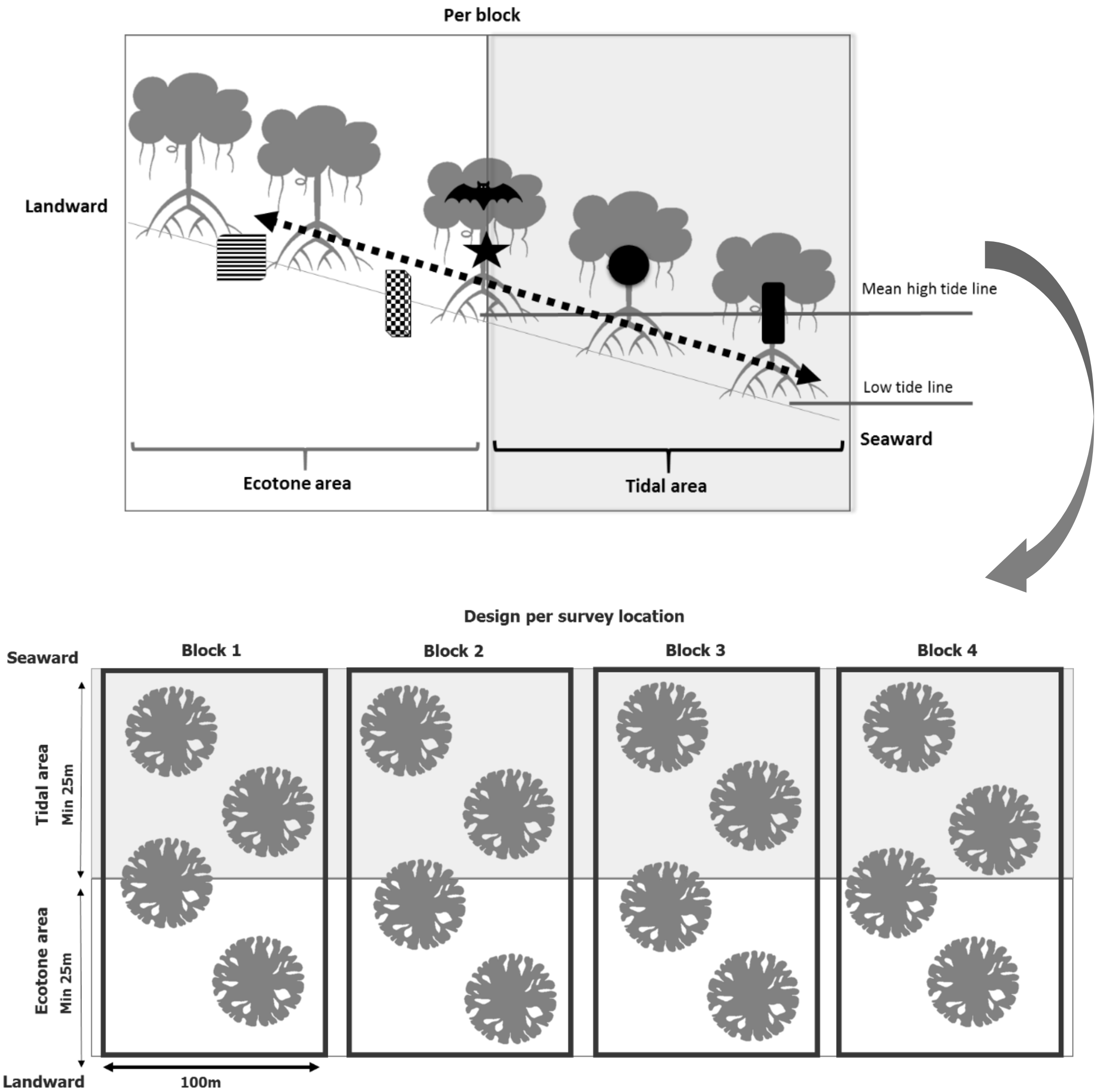

Figure 2. Blocked survey design employed in mangrove forests. The upper panel shows the placement of trapping techniques in a cross section view. The lower panel illustrates the block designs at each site from an aerial view. Trapping techniques: Terrestrial refuges (striped square), live traps (blocked rectangle), camera trap (star), bat detector (bat), hair tubes (black circle), arboreal refuges (black rectangle) and night transect (dashed arrow).

Characteristics of fauna detected in mangrove forests

I was interested in several ecological attributes of the vertebrate taxa observed in mangroves. I captured direct observations of resource use in the field. The evidence for resource use from field observations was supplemented by records from a targeted search of peer-reviewed literature. The specific resources used were categorized as: (1) feeding, (2) breeding, (3) dispersal route between primary habitats (4) shelter from biotic (e.g. predators, competitors) and abiotic stressors (e.g. temperature extreme; desiccation); and (5) use as refuge from human disturbance. Targeted literature searches were used to establish each species' dependence on mangrove forests, classifying species as either obligate or facultative users where known. Obligate users were those species described as being found primarily in mangroves or require the mangroves for an important part of their life history. Whereas facultative users were those species that occupied both mangrove and adjacent terrestrial habitats (reviewed in Rog *et al.* (2017), Chapter 2). Separate from the resource use detected or reported in mangroves, all species were classified according to their feeding guild: 1) carnivore (including insectivores and piscivores), 2) herbivore (folivores, frugivores), and 3) omnivore (any combination of animal protein and fruit, seeds or leaves). Feeding ecology was derived from (Menkhorst & Knight, 2001) and targeted searches of the peer-reviewed literature.

Species of conservation concern

For each species I noted their conservation status at both the international (i.e., IUCN conservation status following taxonomy from the IUCN red list (IUCN, 2017) and national (i.e., conservation status under relevant state legislation (*Nature Conservation Act,* 1992; DSE, 2013; *Biodiversity Conservation act*, 2016). I also noted whether a species was native or introduced to Australia based on "Mammals of Australia" Menkhorst and Knight (2001).

Relationships between terrestrial vertebrate richness and available habitat

To understand whether vertebrate richness correlated with attributes of the available habitat, I recorded the floral richness of the mangrove forests surveyed and of the vegetation community directly adjacent to the field sites. I also recorded the number of tree hollows found in the mangroves during the field survey, as an additional measure of habitat complexity. I determined mangrove richness following "Australia's mangroves" by Duke (2006) together with targeted searches of literature to identify the potential variations of mangrove plant species richness at each site (Supplementary table S1). I recorded the number of hollows observed per site while walking the four 100 m transects. I defined hollows as a depression of a minimum of 10 cm in diameter (Vesk *et al.*, 2008) and 10 cm in depth. I determined the adjacent vegetation community for each site by using state government vegetation maps (for references to these maps see Supplementary table S1). I used the species richness values for each vegetation community defined in the associated condition benchmark associated with each vegetation type (for references to these benchmarks see Supplementary table S1).

To examine any association between faunal richness and the different habitat attributes at the 10 sites, Pearson's correlation coefficients were calculated for faunal richness and: 1) mangrove plant species richness, 2) total number of mangrove hollows per site, and 3) floral richness of the adjacent vegetation.

National park management and attention to mangroves

To understand the existing knowledge about the mangrove forests at each of the survey sites, I interviewed the site manager for each of the national parks. To understand the existing knowledge about the mangrove forests at each of the survey sites, I interviewed the site manager for each of the national parks. The site managers were identified through direct communication with staff at each national park to identify the head ranger responsible for daily management of the park. I conducted ten face-to-face, semi structured interviews, ranging between 30 and 60 minutes. (Supplementary table S2) where managers were asked about: 1) the management objectives present at the site (for the whole national park), 2) any specific management objectives for mangroves within their national park, 3) species they were aware of occurring in mangroves at their site, and 4) any monitoring conducted focused on mangrove forests. They were also asked to provide some background information about themselves and their experience with the site (Supplementary table S2).

Results

I detected 65 terrestrial vertebrate species in mangrove forests across 10 sites distributed along the eastern seaboard of Australia, including 42 mammals, 19 reptiles, 4 amphibians (Table 1, Supplementary Table 3). Over 45% (n=30) of the native species I detected had not previously been recorded in mangroves (Rog *et al.*, 2017). While only sampling 10 sites on the east coast of Australia, this result increases the number of (native) terrestrial vertebrates reported in global mangroves (463 species; (Rog *et al.*, 2017) by 10%. In addition, 12 invasive species were also detected making the total of new species I detected 42. Interestingly, most of the new species in Australian mangroves (n=22 out of 42) I detected through incidental sightings, of which 9 that were only detected through incidental sightings rather than any of the formal survey techniques (Supplementary table S4 and Chapter 3 for the effectiveness and efficiency of each of the techniques).

Characteristics of fauna detected in mangrove forests

Only one of the species detected (*Xeromys myoides*) has been reported to be an obligate user of mangroves (2% of total species detected), with no information available for the other species with which to classify them as either obligate or facultative users. I characterised resource use for 42% of species (n=27) of which the resource use of 10 species was detected through direct observations in the field (Supplementary Table S3). The literature review yielded information on resource use for 18 of the species I detected in mangroves, including 1 species for which I made direct observations of feeding which was supported by the literature (Supplementary Table S3). For five of these species, the literature sources merely reported that a species used a resource in mangroves, without providing evidence from field observations (marked with * in Supplementary table S3).

Of the 27 species for which resource use was reported food was the most commonly used resource in mangroves, with 76% of species recorded as feeding. Terrestrial vertebrates in Australian mangroves were also reported to use mangroves for shelter (13%) and breeding (13%) (Figure 3, Supplementary Table S3). Species were most often only recorded using a single resource, but 13% of species were recorded using 2 or 3 resources. There were fewer records of species providing ecosystem services to mangrove ecosystems. I observed nutrient provision (i.e., the decomposing carcass of *Macropus giganticus* and recorded one species (*Pteropus palecto*) known to be a pollinator of mangroves (Hogarth, 2015) (Supplementary table S3).

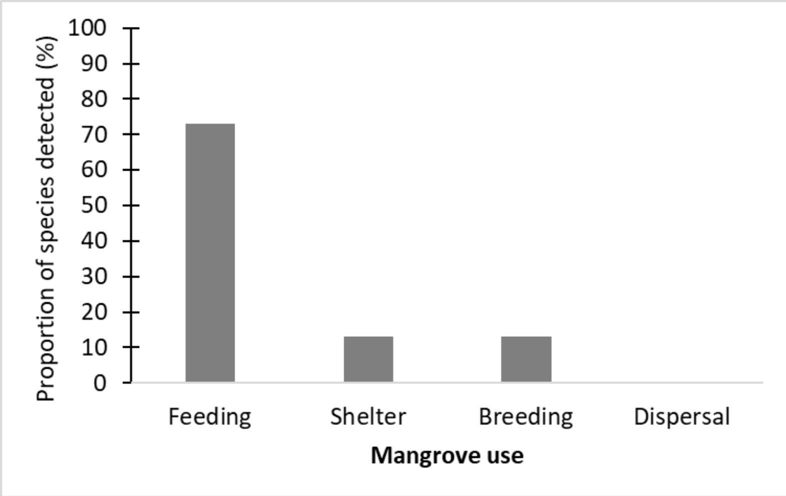

Figure 3. Mangrove use reported for terrestrial vertebrates (n = 27) in Australian mangroves. Reported uses are not mutually exclusive. Obligate users (n = 1) are not shown in this figure as by definition these species use mangroves for all their resources.

Of the 65 species detected in mangroves 66% are carnivorous (Figure 4, n=43; Supplementary Table 3). Invertebrates are the most common food source, with 85% of carnivores including this group in their diet.

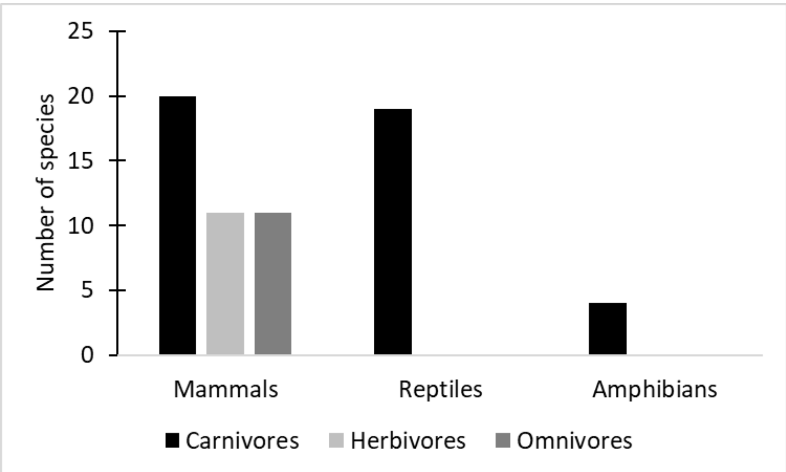

Figure 4. Feeding ecology of terrestrial vertebrates including both facultative and obligate mangrove users (mammals, reptiles and amphibians) in mangrove forests at 10 sites on the east coast of Australia. Insectivores are included in the carnivore guild.

Species of management concern

Thirty five of the 65 species I detected had had their conservation status assessed by the IUCN, of which three were classified as globally threatened Bennet's tree-kangaroo (Dendrolagus bennettianus), bush-tailed phascogale (Phascogale tapoatafa), and (Xeromys myoides). A further 5 species were considered threatened under the relevant state threatened species legislation (large-footed Myotis (Myotis macropus), Lumsden's mormopterus (Mormopterus lumsdenae), eastern bentwing bat (Miniopterus orianae oceanensis), eastern falsistrellus (Falsistrellus tasmaniensis), and glossy grass skink (Pseudemoia rawlinsoni). In total, 8 species (12%) were of conservation concern. Invasive species made up 18% (n=12) of the 65 species detected (Table 2, Supplementary Table S3). Overall, species of management concern, both threatened and invasive, constituted almost a third (30.7%) of the species detected.

Relationships between terrestrial vertebrate richness and available habitat

There was a strong positive correlation between terrestrial vertebrate richness and mangrove plant richness, although the relationship was not significant (r=0.55; n=10; p= 0.101; R^2 =0.300). There was a weak correlation between terrestrial vertebrate richness and adjacent habitat richness, although again the relationship was not significant (r=0.26; n=10; p=0.255; R^2=0.158). No relationship was found between terrestrial vertebrate richness and the number of hollows in mangroves (r=0.12; n=10; p=0.737; R^2=0.015) (Supplementary Table S5).

Knowledge of site managers about management relevant to mangroves

The managers I interviewed had an average of 9.85 years' (±2.27 SE; range: .5-25) experience managing the sites. They reported that on average, they visited the total site 76 (± 31.53SE; range: 12-365) times a year. No planned visits specifically to mangroves were reported for half of the sites. At the other 5 sites, the average number of times a year the site manager visited the mangroves was 10 (±1.26 SE; range: 2-12). Site managers identified 6 main management objectives for their sites, of which invasive species and fire management were common to all sites (Table 3). Across the 10 national parks, specific management objectives were related to protecting mangroves from illegal fishing and boating related activities (Table 3). Four sites had management objectives specific to mangroves. When compared with the number of sites at which these management issues were directly observed in mangroves, there is a clear mismatch between current knowledge of pressures related to invasive and threatened species present in mangroves and the on-ground reality (Table 3). Managers were aware of management issues, such as invasive species in the whole park, but only knew of a quarter of the species I detected within mangroves (3 of 12 species). They were also largely unaware of threatened species using mangroves forests at their sites (Table 3), only mentioning of 2 of 8 threatened species I detected as using mangroves. No correlation was found between the numbers of years of experience or site visits and the managers' knowledge on issues in mangroves (years' experience r=0.073; n=10; p= 0.841; site visits r=0.045; n=10; p=0.902). Monitoring was carried out in 3 of the 10 sites, where extent was monitored rather than the health of the mangrove vegetation. Two site managers reported that they believed mangroves did not require active management (i.e., they were reported as "self-caring" or "self-regulating").

Discussion

This study has significantly expanded the number of terrestrial vertebrates known to occur in Australian mangroves, detecting 30 native species never before recorded in mangrove forests (Rog *et al.*, 2017) as well as adding 12 invasive species not previously documented. Given this increase is based on rapid biodiversity assessments at 10 sites on the east coast of Australia, my study demonstrates the value of field studies in expanding knowledge about the terrestrial vertebrates that use mangrove forests. Indeed, the 65 species I detected is likely an underestimate, and longer surveys, at more sites, across multiple seasons may further increase the knowledge about the species present in the mangrove forests of Australia. Nevertheless, the patterns in species richness I observed in my study on a national level match well with those of a global review of terrestrial vertebrates in mangroves, which found species richness to increase from temperate to tropical regions and mammals to be the dominant taxonomic group of the three taxonomic groups studied (Rog *et al.*, 2017).

Characteristics fauna in mangroves

There is limited knowledge about whether the species I detected were obligate or facultative users of mangroves, with only one of the species I detected being known to be a mangrove specialist (*Xeromys miodes*; (Hogarth, 2015) although there are sources that do not see this species as obligate (Luther & Greenberg, 2009). All of the other species I detected in mangroves are known to occur in other habitats without evidence to suggest they are obligate (Menkhorst & Knight, 2001), implying the vast majority of species are facultative mangrove users. While there was only a weak correlation between the richness of adjacent habitat and faunal richness at a site, the prevalence of facultative users of mangroves in Australia (98%), and globally (86%; (Rog *et al.*, 2017), suggests that adjacent habitat may be an important contributor to the faunal richness of mangroves. The divide between obligate and facultative users of mangroves may not be fixed, with species reported more broadly as facultative users of mangroves utilising these areas as their primary habitat in some locations (e.g., *Alouatta pigra,* Mexico (Bridgeman, 2012)) potentially explained by disturbance or resource availability of adjacent habitat.

The majority of species from which resource use was reported were using the forest for food resources (Figure 3). The overall patterns in feeding ecology I observed in this study (Figure 4) match global patterns, with carnivores being the most common feeding guild within mangroves (Rog *et al.*, 2017). Terrestrial invertebrates make up a large part of the diet of the species I detected (e.g. bats, reptiles and amphibians), yet apart from the impact of some invertebrate groups in on vegetation and functioning of mangrove plants (e.g. ants and herbivorous insects (Cannicci *et al.*, 2008)), there is little known about their relationship with terrestrial species richness in mangroves. Some of the species I detected make use of marine food resources (e.g. feeding on crabs by the water rat (*Hydromys chrysogaster)*, water mouse (*Xeromys myoides*) and monitor lizard (*Varanus indicus*) meaning that the marine habitat should be included when further investigating resource use of terrestrial vertebrates in mangroves.

While making up a smaller fraction of the assemblage, I also detected a range of herbivores and omnivores feeding in mangroves (Figure 3), suggesting mangrove plants could provide a food resource, but it is not known whether all herbivores are in mangroves to feed or for other resources. While mangroves are often considered suboptimal environments for herbivores, supposedly because of the low-quality vegetation (Kathiresan & Bingham, 2001), I observed deer (*Rusa timorensis*), giant rats (*Uromys caudimaculatus*) and rabbits (*Oryctolagus cuniculus*) browsing on mangrove leaves. Mangrove leaves can be unpalatable because of high salt content (Kathiresan & Bingham, 2001), explaining a low portion of herbivores detected in the field survey and globally (Rog *et al.*, 2017). However, not all Australian mangrove species secrete salt through their leaves (Scholander *et al.*, 1962) and the water content in mangrove leaves can be higher than in surrounding eucalypt tree species (Winter, 2004). Likewise, mangrove leaves from some South American species have been shown to have a similar nutritional value to other trees in the area (Bridgeman, 2012). Because it is suggested that because some species might be adapted to feeding on plants rich in unpalatable components (*Hapalemur spp., Prolemur simus,* Madagascar), more species that are specialised to feeding on mangrove leaves are expected (Gardner, 2016). These examples support a rethinking of the vegetation of mangrove forests as a potential resource for herbivores.

If foliage is an important resource for terrestrial vertebrates in mangroves (e.g., cows in New Zealand are fed on foliage of certain mangrove species because they are high in nitrogen (Waisel, 1986), it might be expected that the proportion of herbivores would increase with mangrove floral richness, with more species providing greater variety of resources. Nevertheless, I found the number of herbivores was actually double in the temperate region (8 herbivore species) compared to subtropics and tropics (4 herbivore species) despite a significant increase in floral richness towards the tropics, potentially explained by some mangrove plant species that occur in these regions being less palatable for vertebrates (but not necessarily for insects; Feller, 2002), while the only temperate species Avicennia are high in nitrogen. Three herbivore species in the temperate region were however invasive (e.g. cows, deer and rabbits) as Australia lacks groups of herbivores (e.g. bovines, camelids, equids, deer and iguanas) these patterns might be different in regions were these do occur. While mangrove seeds can be an abundant resource at some times of year, few species were observed eating mangrove seeds (e.g., *Wallabia bicolor*, field observation). Given mangroves do not rely on vertebrates for seed dispersal (Bridgeman, 2012), it is possible that seeds do not provide an important source of nutrients for terrestrial vertebrates.

There remain significant knowledge gaps about how terrestrial vertebrates use mangrove forests in Australia (51.5% of species without information available on resource use, Supplementary table S3) and globally (31.6% of species without information available on resource use, (Rog *et al.*, 2017). This highlights the need to move beyond documenting species richness, to studies that detail the resources provided by mangroves to vertebrates, and also to expand our knowledge about the services vertebrates provide to support mangrove ecosystems. While food was the most common resource provided to terrestrial vertebrates, I found that mangroves were providing other resources like shelter in the form of tree hollows (through field observations of the lizards *Leptodactylus lugubris* and *Hemidactylus frenatus*, and breeding (reported in the literature of the salt water crocodile, *Crocodylus porosus,* flying fox *Pteropus Alecto*; Hogarth, 2015 and deer *Axis porcinus*; Kathiresan & Bingham, 2001). The role of mangroves as a corridor or (temporal) refuge between optimal habitats that are fragmented is poorly recognised (Nowak 2103). This role may become more important as habitat fragmentation increasingly isolates species' preferred habitat. The other mangrove resource reported by Rog *et al.* (2017; e.g. dispersal and refuge from human disturbance) were not detected and ethology studies are required to determine resource use of terrestrial vertebrates in mangroves. Without a deeper understanding of the ecology of the terrestrial components of mangroves, it will be difficult to plan for effective management and achieving positive conservation outcomes for these ecosystems.

Relationships between terrestrial vertebrate richness and available habitat

While I observed the same positive correlation between mangrove plant richness and faunal richness as that observed at a global scale (r=0.55; Rog et al. 2017, were the highest faunal richness and mangrove richness were detected in the tropics) the small number of sites may have contributed to the non-significant results within my study. The richness of the adjacent habitat and the structure of the vegetation (e.g., number of hollows) may also help contribute to the faunal richness at a site. Up to 11 times more hollows have been reported in mangroves than adjacent forest (McConville *et al.*, 2013), possibly related to the fact that mangroves are reported to be the last remaining old growth forest in several countries (Nowak, 2013). Hollows may also be particularly apparent in mangroves because of the high abundance of wood boring insects (Feller, 2002). Additionally *Avecinnea* has unique wood structure (successive cambia) that may be related to the formation of hollows in this species (Schmitz et al., 2007) providing roosts for bats (McKenzie & Rolfe, 1986). I found lizard species *Eulamprus tenuis, Gehyra dubia* and *Leptodactylus lugubris* using hollows, the latter with eggs (Supplementary table S3). As the use of hollows is predominant in Australian vertebrate fauna (Gibbons, 2002), it is vital to further investigate the role of hollows in mangroves as a critical resource. Other factors like the proportion of invasive species and the influence of seasonality or abiotic disturbances (e.g. illegal extractions (Table 3)) could have affected the faunal richness at time of surveying. The findings of my study suggest further research is needed to better understand the predictors of faunal richness in mangrove habitats.

Implications for the management of mangroves

I detected five threatened species using mangrove forests for shelter, and one that is obligate, of which most were unknown to site managers (Table 3). All threatened species are endangered due to habitat loss (*Dendrolagus bennettianus, Phascogale tapoatafa, Myotis macropus, Mormopterus lumsdenae, Miniopterus orianae oceanensis, Falsistrellus tasmaniensis,* and *Pseudemoia rawlinsoni)* and some are also threatened by predation by introduced predators (*Phascogale tapoatafa, Xeromys myoides*). This finding of threatened species emphasises the potential role for mangroves as refuges for species that have lost their primary habitat.

My results demonstrate that invasive species make up a significant proportion of the terrestrial vertebrates within mangroves in Australia. While managers tended to be aware that invasive species are a management issue within the rest of the national park, they were often ignorant of the presence of these species within mangroves (Table 3), which is potentially related to the fact that half of the managers visited mangrove ecosystems less than once per year. Potentially, varying resource availability between parks contributed to differences in management intensity and therefore knowledge of the mangroves at the site. There is little information about how these invasive species impact mangrove ecosystems, or whether mangroves may be providing a refuge for some invasive species that are managed elsewhere in the park. Nine of the 12 invasive species I detected have been listed as key threatening processes under Australian legislation, because of their impacts on native vegetation (e.g., rabbits, Eldridge & Simpson, 2002; hog deer – known to prevent regeneration of mangroves by browsing on young mangrove shoots; Parks Victoria, 2003), or on a wide range of native fauna (e.g., cats, Dickman, 1996; foxes Saunders *et al.*, 2010, black rats, Stokes *et al.*, 2009). Five of the 12 invasive species I detected have hard hoofs such as deer, pigs and cows, which can damage native vegetation and alter hydrology of coastal vegetated systems (e.g., water buffalo, *Bubalus bubalis;* Finlayson *et al.* (1997)). Due to various factors influencing the regeneration success of mangroves (Bosire *et al.*, 2008) and the slow growth of mangroves at certain latitudes (Lovelock *et* *al.*, 2007), structural damage to these ecosystems from hard hooved species could have significant and long lasting impacts. Habitat complexity of mangrove forests can create significant barriers for invasive species control, e.g., rat poison application (Harper & Bunbury, 2015), which may mean strategies need to be adapted to suit conditions in these ecosystems. Mangroves can in some areas make up relatively small parts of a national park (e.g., they occur on linear strips of less than 200m wide on the surveyed sites in the temperate region), making the low rate of visitation not solely a function of accessibility, but available time being allocated relative to activities in other areas of the park. The capacity to recognise these threats, or emerging management issues, however will be limited by the lack of monitoring I documented within mangrove forests and reinforces the need to build our knowledge base on the ecological impact of invasive species on the mangrove community. The perception that mangroves are inconsistently perceived as marine or terrestrial in Australia (Rog & Cook, 2017), and the view of some managers that mangroves self-regulate and do not require active management (reported by 2 interviewees), may explain the lack of knowledge on species of conservation concern. Interestingly, incidental sightings made during the field surveys detected the highest species richness including threatened and invasive species. This suggests that site visits are a promising low-cost activity that can yield crucial information on emerging issues for mangrove management and should therefore be considered the minimum management activities for mangroves.

Conclusions

My study demonstrates the value of rapid fauna assessments in mangrove forests to document the terrestrial vertebrates that use these ecosystems. This is particularly important, because relying on the expert knowledge of site managers alone appears to be insufficient. In order to improve the effective management of these systems, I provide three recommendations: 1) Mangrove forest require periodic monitoring using rapid assessments, such as the methods outlined here, to formally identify the species present, determine any conservation measures they require, and track any threats they pose to the mangrove ecosystem. 2) Survey data should be publicly available (i.e., open access databases) to build baseline knowledge of terrestrial vertebrates in mangroves and develop an evidence base that can inform best practice management for this vulnerable ecosystem. 3) Where resources are limited, managers can gain valuable information from incidental sightings generated during regular site visits to mangroves, enabling them to build knowledge of the species present and identify on-ground management issues. Without a deeper understanding of the terrestrial components of mangrove forests, we cannot hope to ensure their effective conservation and management into the future.

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Table 1. Species per family detected of each taxonomic group.

	AMPHIBIA	REPTILIA			MAMMALIA						
FAMILY	*Anura*	*Crocodylinae*	*Lacertilia*	*Serpentes*	*Chiroptera*	*Canis*	*Felis*	*Lagomorpha*	*Marsupialia*	*Rodentia*	*Ungulata*
NUMBER OF SPECIES	4	1	15	3	14	3	1	1	10	9	4

Table 2. Taxonomic groups detected in mangrove forests and the proportion of species of conservation concern.

TAXONOMIC GROUP	TOTAL	GLOBALLY OR LOCALLY THREATENED	INVASIVE
MAMMALS	42	n=7	n=10
REPTILES	19	n=1	n=1
AMPHIBIANS	4	n=0	n=1

Table 3. The management objectives relevant to the 10 national parks surveyed based on interviews with site managers and direct observations in the field.

Management objective	Number of sites where managers reported issue as relevant	Number of sites where managers reported issue as relevant for mangroves	Number of sites where issue was identified during field work
Invasive species			
Mammals	10	3	10
Reptiles	0	0	1
Amphibians	4	0	2
Threatened species			
Mammals	10	4	6
Reptiles	9	0	1
Amphibians	7	0	0
Other issues			
Fire management	9	0	-
Invasive weed	10	1	0
Illegal access (cars, boats) damaging native vegetation	9	4	1
legal extraction of natural resources (hunting, fishing, crabbing)	7	5	6

Chapter five: Strengthening governance for intertidal ecosystems requires a consistent definition of boundaries between land and sea

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Abstract

The protection of intertidal ecosystems is complex because they straddle both marine and terrestrial realms. This leads to inconsistent characterisation as marine and/or terrestrial systems, or neither. Vegetated intertidal ecosystems are especially complex to classify because they can have an unclear border with terrestrial vegetation, causing confusion around taxonomy (e.g., mangrove-like plants). This confusion and inconsistency in classification can impact these systems through poor governance and incomplete protection. Using Australian mangrove ecosystems as a case study, I explore the complexity of how land and sea boundaries are defined among jurisdictions and different types of legislation, and how these correspond to ecosystem boundaries. I demonstrate that capturing vegetated intertidal ecosystems under native vegetation laws and prioritizing the mitigation of threats with a terrestrial origin offers the greatest protection to these systems. I also show the impact of inconsistent boundaries on the inclusion of intertidal ecosystems within protected areas. The evidence presented here highlights problems within the Australian context, but most of these issues are also challenges for the management of intertidal ecosystems around the world. My study demonstrates the urgent need for a global review of legislation governing the boundaries of land and sea to determine whether the suggestions I offer may provide global solutions, to ensuring these critical systems do not fall through the cracks in ecosystem protection and management.

Introduction

Intertidal ecosystems occur at the interface of land and sea, encompassing environments such as sandy beaches and rock platforms through to vegetation communities like mangrove and saltmarsh. Intertidal ecosystems provide important ecosystem services such as coastal protection, carbon sequestration and critical habitat for a wide range of both marine (Nagelkerken et al., 2008; Yates et al., 2014) and terrestrial biodiversity (Rog et al., 2017). Despite their ecological importance, globally, intertidal ecosystems are in decline due to increasing anthropogenic pressure on coastal areas, including development, climate change and sea level rise (Giri et al., 2011; UNEP, 2013). However, the ability to effectively conserve these ecosystems is currently hampered by the complexity of managing intertidal ecosystems, due to uncertainty around land-sea boundary definitions (Clemens et al., 2014; Harris et al., 2014; Tagliapietra et al., 2009).

A large source of complexity in defining the boundaries of intertidal ecosystems lies in the multitude of legislative land sea boundaries based on tidal lines (e.g. seaward between land and sea generally the Low Tide, and between land and intertidal generally the Astronomical High Tide or Mean High Water Mark), which are fuzzy and dynamic (Friess et al., 2016) and difficult to accurately locate. Unambiguous boundaries of ecosystems are vital to enforcing legislation, as demonstrated in Indonesian rainforests, where poorly defined protected forest area boundaries have enabled illegal logging to slip through cracks in the legal system (Sahide and Giessen, 2015). Uncertainty around the boundaries between land and sea has also led to inconsistency in how these boundaries are applied both within and between countries (Abdullah et al., 2013; Day et al., 2012; Liu et al., 2014). This can have serious implications in the many cases where the national and international legislation that overlaps in the intertidal zone has inconsistent laws and regulations (Cao and Wong, 2007) and competing and unclear objectives (Friess et al., 2016) leading to ineffective protection of this zone.

The inconsistent definition of the land-sea boundary creates challenges for broad-scale analyses and global assessments of biodiversity of intertidal ecosystems, generating potentially large mapping inconsistencies (Friess et al., 2012). This inconsistency has been specifically cited as the reason why intertidal mangrove ecosystems have been excluded from global assessments of threatened ecosystems (Chape et al., 2005; Hoekstra et al., 2005) or grouped with tidal marsh ecosystems (Costanza et al., 2014). Likewise, because there is no consistent definition of the bounds of intertidal ecosystems their original global extent is not possible to estimate (Friess et al., 2012). As a result, there is great uncertainty surrounding estimates of the rate of global decline and the adequacy of protection measures currently in place, making it difficult to anticipate future trends on which management actions can be built.

Another major point of uncertainty complicating the management of intertidal ecosystems is whether the ecosystems themselves are characterized as marine or terrestrial environments. Marine and terrestrial ecosystems have been separated historically, which is apparent across agencies, NGO's, scientific institutions (Álvarez-Romero et al., 2011) and national policies (Friess et al., 2016). The uncertainty to which of the two environments intertidal systems belong is exemplified by the variability on how studies on threats to intertidal ecosystems classify them: marine (e.g. Halpern et al., 2008); terrestrial (e.g. Olson et al., 2001); or both (e.g. Joppa et al., 2016). While it is important to take a comprehensive and cross system approach to studying threats to these ecosystems (Álvarez-Romero et al., 2011) as threats to intertidal in many coastal systems can be diverse in origin (Friess et al., 2015), without a cohesive approach there is a risk that some threats are being missed, while others over-emphasized. One practical implication of whether intertidal ecosystems are characterized as marine or terrestrial is whether threat mitigation is the responsibility of marine or terrestrial protected area managers. This distinction is vital for the effective protection and management because protection for native (terrestrial) vegetation versus the marine environment differs in emphasis, and often in management practices (Adams et al., 2014; Boon and Beger, 2016) and conservation values (Álvarez-Romero et al., 2015). For example, the most significant threat to the marine environment, over-fishing (Halpern et al., 2008), is not the greatest threat to intertidal ecosystems, such as saltmarsh and mangroves, which are most vulnerable to clearing for coastal development (Giri et al., 2011). In recent years increased attention has been given to integrated coastal zone management (Álvarez-Romero et al., 2011; Beger et al., 2010), however as long as separate marine and terrestrial protected area boundaries exist, their respective threats need to be considered when aiming to protect intertidal ecosystems.

For vegetated intertidal ecosystems, this marine-terrestrial distinction is even more complex on a finer scale as vegetated intertidal systems occur along an environmental gradient, where a transition zone can make it difficult to define the boundary of the intertidal ecosystem with adjacent vegetated terrestrial ecosystems (Boon et al., 2014; Duke, 2006a). Vegetated intertidal ecosystems also potentially fall under legislation related to native vegetation management (where native vegetation is generally defined as aquatic or terrestrial plant or plants indigenous to the region of interest under Australian legislation; Table S1), adding a further layer of complexity. The vegetated intertidal ecosystems mangroves and saltmarsh have species within them that can be classified as both marine and terrestrial (Boon et al., 2011), most likely related to their physiological adaptations to exposure to both marine and terrestrial conditions (Tomlinson, 2016). While this taxonomic classification may seem trivial, it can have important implications for how species are managed and conserved (Fraser et al., 2015). Variation in the taxonomic classification of the species within these ecosystems as marine or terrestrial can also lead to them being divided between the types of protection, complicating management responsibility, or missing protection altogether (Boon et al., 2011). Indeed, there is concern that intertidal ecosystems are underrepresented in protected areas (Banks et al., 2005), possibly due to this difficulty in determining whether they should be included within marine or terrestrial protected areas. Without a consistent classification of intertidal plant species related to a consistent characterisation as marine or terrestrial, intertidal ecosystems are at risk of a lack of specific management objectives necessary for effective protection (Harris et al., 2014).

The aforementioned inconsistent definition of boundaries, marine or terrestrial characterization, and confusion around taxonomic classification has set intertidal systems up for poor governance. Recent studies have highlighted the complexity in intertidal ecosystem management and the urgent need to improve their protection (Banks et al., 2005; Friess et al., 2016; Rogers et al., 2016). My study is the first to consider the drivers of this complexity from an ecosystem boundary perspective. I explore the complexity in how the land and sea boundaries are defined among jurisdictions and types of legislation, the characterization of vegetated intertidal ecosystems as marine or terrestrial and the taxonomic classification of intertidal plant species, using Australian mangroves ecosystems as a case study. I use these data to evaluate how this complexity affect the protection of intertidal ecosystems, with the goal of identifying how governance structures for these complex ecosystems can be strengthened.

Methods

Study region

I focus on intertidal ecosystem governance within Australia. Australia is a federation of six states and two territories united under a national government, creating nine jurisdictional boundaries. These boundaries mirror the complexity associated with international boundaries that have created significant international transboundary governance issues discussed elsewhere (Liquete et al., 2011; Bartier and Sloan, 2007; Rahibulsadri et al., 2014). More than 85% of Australia's population live within 50 km of the coastline creating increasing pressure on intertidal ecosystems from encroaching coastal development; the most significant threat to intertidal ecosystems globally (Giri et al., 2011; Foster et al., 2013).

Study system

Mangroves occur along the coastline of five out of six Australian jurisdictions. Mangrove ecosystems make an ideal case study for assessing inconsistent legal boundaries because they can occur across the full intertidal zone from the lowest tide line to the highest (Astronomical) tide line (Figure 1), thereby crossing all tidal lines which are potential boundaries used to define land and sea (see Knight et al. (2008) for detail about the more complex relationships between micro-topography and tidal influences). The other two vegetated intertidal ecosystems (saltmarsh and seagrass) generally occur at the extremes of the tidal range. Due to their occurrence across two realms, mangroves also play important ecological roles in both marine and terrestrial communities (e.g. their roots can provide refuge for fish (Nagelkerken et al., 2010); and coral (Yates et al., 2014) and their branches and canopy provide habitat for terrestrial vertebrates (Rog et al., 2017).

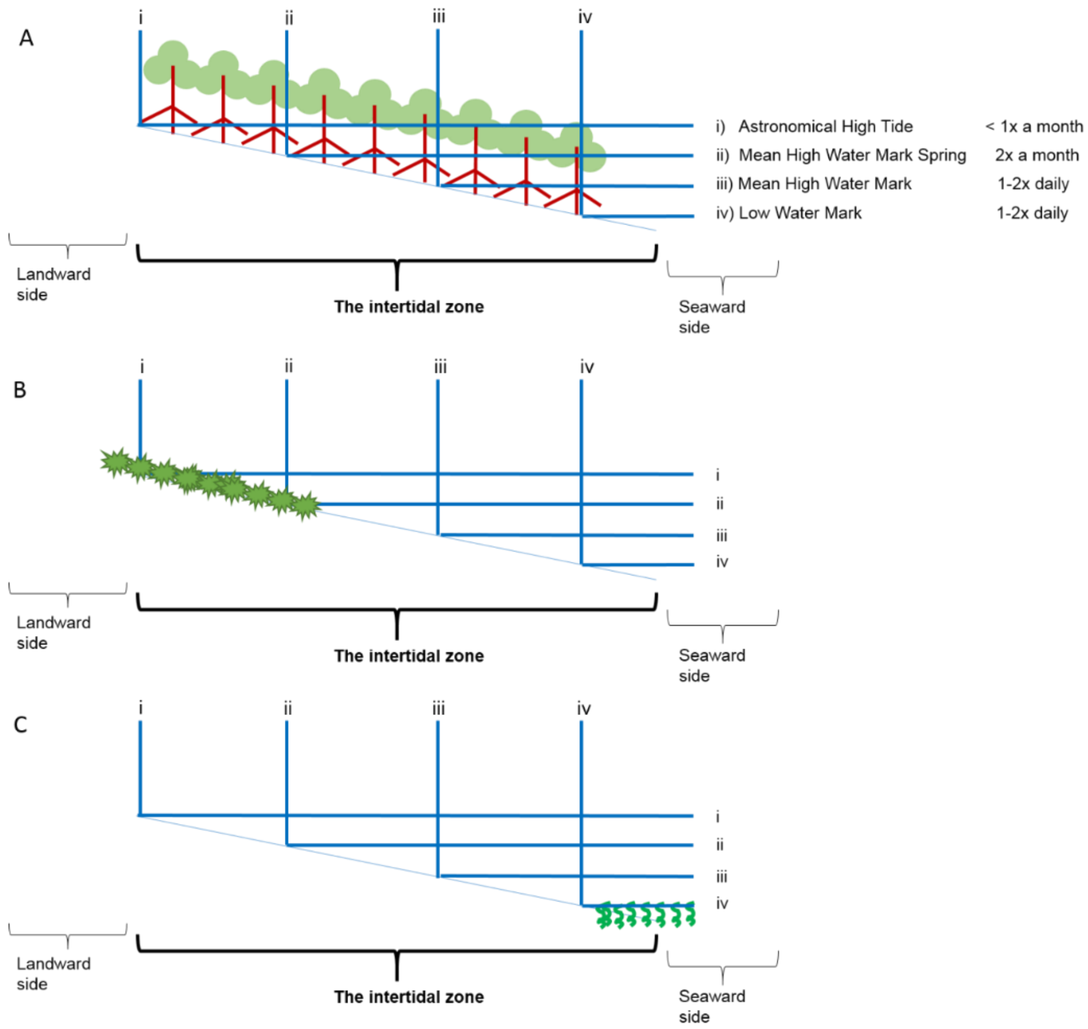

Figure 1. A diagram of the intertidal zone, spanning the full tidal range from Highest Astronomical Tide to the Low Water Mark, showing where vegetation fall across the tidal range for: A) the mangrove community (Hogarth, 2015), B) the saltmarsh community (Saintilan and Rogers, 2013) and C) the seagrass community (Hogarth, 2015). I-IV represent different tidal lines that are used to define the boundaries between land, sea and the intertidal zone, and shows where they fall within the different vegetation communities.

Data collection

To assess the governance structures for intertidal ecosystems, specifically for mangrove ecosystems in Australia, I focused on five aspects; 1) The definition of the legislative boundaries between land and sea; 2) Characterisation as marine or terrestrial of the vegetated intertidal ecosystem (mangrove) by the legislation; 3) Classification of what plant species are seen as mangroves (taxonomy); 4) The legislative mechanisms for the protection of vegetated intertidal ecosystems (e.g., fisheries protection, protected areas) and 5) The ability of protection mechanisms to mitigate threats to vegetated intertidal ecosystems.

To evaluate the variability in these aspects of intertidal governance for mangrove systems I conducted a comparative review of Australia's federal and state legislation. This review was based on sources identified by Rogers et al. (2016) in their assessment of mangrove policy and legislation in Australia, along with additional searches on land sea boundaries, management literature and relevant government websites. The sources used were placed into the following five categories;

i) Legislation on intertidal boundaries

ii) Legislation on native vegetation

iii) Legislation on fisheries management

iv) Legislation on protected areas

v) Legislation on threatened species and communities

See Supplementary table S1 for an overview of documents used for analysis. In all cases these documents were read and relevant information and definitions related to intertidal boundaries, mangroves ecosystems and plants and intertidal vegetation in general were extracted.

Legislation defining boundaries between land, sea and the intertidal zone

Multiple boundaries exist between land and sea related to the intertidal zone. Australian law decrees that the intertidal zone in Australia cannot be privately owned, with specific legislation that defines where privately owned land defaults back to state ownership (LexisNexis, 2010) thereby creating the need for a landward boundary between private land and the intertidal zone. A range of other legislation provided reference to both a landward boundary and a seaward boundary (e.g. legislation on the management of native vegetation, fisheries management and protected areas). All of the documents were sourced and coded according to how the boundaries between land and sea, and land and the intertidal zone were defined in relation to tidal lines (e.g. Low Water Mark (LWM), Mean High Water Mark (MHWM), Mean High Water at Spring Tides (MHWS) and Highest Astronomical Tide (HAT)). The specific description of the practical location of those boundaries was also recorded when present. Where the definition of the boundaries was not linked to a specific tidal line (e.g., tidal zone, sea) the boundary was classified as "unclear".

The marine or terrestrial characterization of intertidal ecosystems by legislation

To determine how consistently legislation throughout Australia characterized mangroves as marine or terrestrial environments, documents in all five categories of legislation (see above) were read for the following pieces of information: a) discussion of marine plants or marine vegetation and whether mangroves were explicitly mentioned, and b) specific reference to mangroves as terrestrial vegetation. Where no specific reference was made between the marine or terrestrial systems, documents were coded as unclear.

Taxonomic classification of mangrove species and ecosystems

What a "true mangrove" plant species is has a long history of debate and I found this is far from over (Mukherjee et al. 2014). The variation in how true mangroves are defined has led to different estimates of the number of mangrove species, with estimates as high as 71 and as low as 40 species (Spalding, 2010; Sandilyan and Kathiresan, 2012; Polidoro et al., 2010; Tomlinson, 2016). As such, legislation would not be expected to provide specific details on what classifies as a mangrove plant. Therefore, to determine which plant species are classified as mangroves by different jurisdictions I included an additional search of a wide range of relevant government documents, including mangrove factsheets, mangrove status reports and state wide coastal plans using a digital search via Google Scholar, Web of Science and Google (specific protected area management plans were not included). Coastal managers in all jurisdictions were also contacted to ensure that all the relevant documents were captured in the analysis and the most current versions were obtained. These documents were read for any reference to the number of mangrove plant species acknowledged to occur within a jurisdiction, a description of individual mangrove species and reference to “true” or associated mangrove plant species.

The protection for vegetated intertidal ecosystems

Protection by native vegetation legislation

As mangroves are vegetated intertidal systems I was interested in determining the proportion of Australian jurisdictions where mangroves were acknowledged as terrestrial vegetation and thereby included under native vegetation laws (Supplementary Table S1s, or if they were included under fisheries acts and thereby seen as a marine feature.

Protection within protected areas

To quantify the proportion of Australian mangrove ecosystems within protected areas I conducted a spatial analysis to determine the extent of mangrove ecosystems within marine protected areas, terrestrial protected areas and with no protection. These analyses were conducted in ArcGIS using a global mangrove distribution layer (Hamilton and Casey, 2016) and the extent of protected areas based on the Collaborative Australian Protected Area Database (Department of the Environment (DotE, 2014) providing estimates accurate as at 2014. All layers were projected in ArcGIS, using the Australian Albers GDA 1994 projection. All protection was taken as one category without distinction into IUCN protected area levels (Dudley, 2008).

The ability of marine or terrestrial protected areas to mitigate threats

To evaluate the effectiveness of marine or terrestrial protection offered to mangrove ecosystems I assessed the capacity of current protected areas to mitigate key global threats to mangrove ecosystems (Duke, 2006b; Sandilyan and Kathiresan, 2012; Mukherjee et al., 2014). Based on information about the types of impacts the threats have on mangroves and the types of controls provided by marine or terrestrial protected areas, I ranked the potential each threat could be mitigated by either protected area based on three criteria: i) area protected (e.g. whole area protected from clearing or only partly protected), ii) flow-on effects of protection to other realm (e.g. prevention of clearing upland also stops sediment run-off into sea and iii) regulating an activity. A low rank meant none of these criteria were met, medium the regulation and/or flow- on effects were met and high that all criteria were met. I also classified threats as those that could be directly managed by protected area management agencies (e.g. illegal fishing, pest control) and threats beyond the control of protected area legislation (e.g. climate change, oil spills).

Results

Legislation defining boundaries between land, sea and the intertidal zone

Federal definition of the intertidal zone

For the purposes of defining land tenure, the formal boundary of the intertidal zone under federal legislation in Australia is the “High Water Mark”, which determines where private land defaults back to public ownership to the “Low Water Mark” (Figure 1). An important implication of this legislation is that it determines where development can take place on the landward size of the boundary, potentially affecting a portion of the vegetated intertidal systems that occur here (Figure 1). While there is a description of how the high or low water mark boundaries are defined under federal law linked to specific tidal lines, these definitions are open for interpretation in the legislation for each jurisdiction.

Jurisdictional definition of the intertidal zone

Within their legislation, each jurisdiction provides their interpretation of the High Water Mark and the Low Water Mark set out under federal legislation. In Queensland and Western Australia the definition is the Mean High Water Mark at Spring tides (Line ii in Figure 1), which is reached twice a month. In South Australia, Victoria and New South Wales the Mean High Water Mark is used, defined as the average of all high tides across a year (Line iii in Figure 1) including more of the landward area of the intertidal ecosystems in private land and thereby more area at risk to development than in Queensland and Western Australia. In the Northern Territory, the legislation does not provide a description on how the High Water Mark is defined. The Low Water Mark does not vary per jurisdiction and is defined as the height of the lowest Low Water Mark at spring tide. Variable interpretation of tidal lines between jurisdictions can have significant consequences, because the tidal range can vary between 0.5 and 13 m around Australia and therefore the intertidal area at risk of development when applying different definitions can vary in magnitude of several kilometres (e.g. depending on slope and tide, the tidal zone can include up to 60 km inland in parts of Northern Australia) (Figure 2).

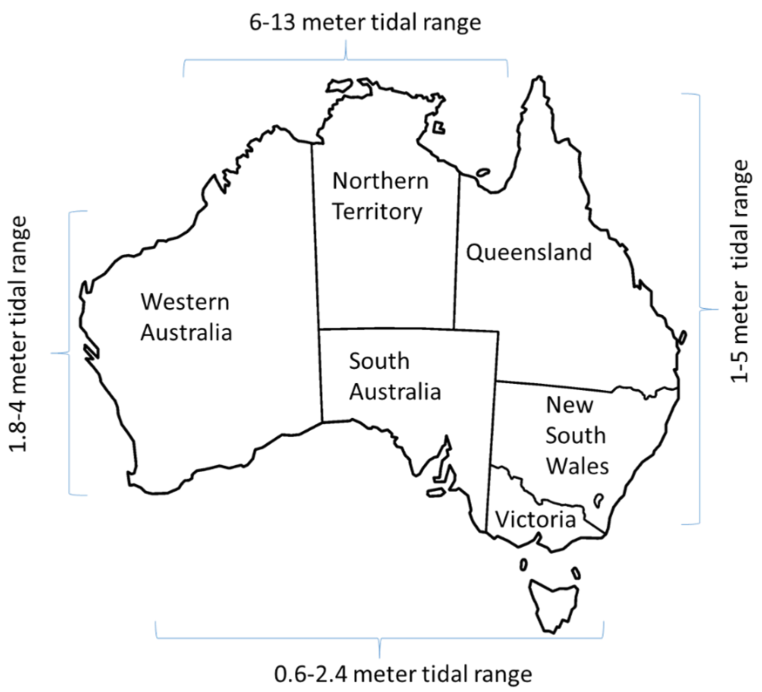

Figure 2. The tidal ranges along the coast of Australia displayed as the minimum and maximum range across the Eastern, Western, Northern and Southern gradients (adapted from Haigh et al., 2014). The intertidal area is generally larger with a greater tidal range, depending on the slope of the surface of land.

I found 29 different pieces of legislation that provide a definition of boundaries between land and sea and land and intertidal area in Australia to allocate management responsibilities. I found a defined boundary (e.g. “High Water Mark is high water mark at average of annual spring tide”) in 20% of the legislation under 4 jurisdictions (Figure 3A). Within the legislation however, a wide range of definitions exist, sometimes in direct conflict with the definitions provided in other relevant legislation for that jurisdiction. For example, in one jurisdiction, the boundary of one marine protected area is defined as “excepting land from High Water Mark 1000 m seaward” and for a terrestrial protected area “including land 150 m seaward from the Mean High Water Mark”. For the largest part of the legislation, the land sea boundary definitions are ambiguous, including definitions such as “waters include tidal waters”, or “includes land covered by water”.

Even in the 20% of cases where a clear description of the High Water Mark boundary was provided, it is not assured that the description would be sufficient to locate the boundary in the field, leaving the location of the High Water Mark boundary open to interpretation by different individuals and creating ambiguity for ecosystem management. For example, in South Australia the advice is to observe the water's edge in calm conditions, at a point with a sharp gradient and that "great care is necessary" in flat graded areas, such as where mangroves occur (Supplementary table S1).

The marine or terrestrial characterisation of intertidal ecosystems by legislation

I found 8 out of 24 legislative documents from the five legislation categories (see "Data collection") that mention mangroves, aquatic plants or mention native vegetation (Supplementary table S1). Of these, three acts from three jurisdictions describe mangroves as marine, while five from the other three jurisdictions provide no characterization as marine or terrestrial (Figure 3B).

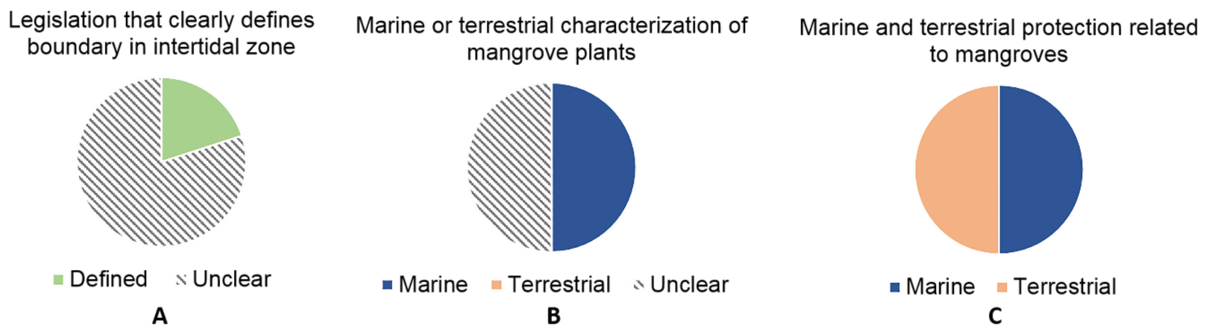

Figure 3. Intertidal governance in Australian based on the proportion where legislation: (A) clearly defines the boundary of the intertidal zone; B) characterizes mangrove plants as marine or terrestrial plants; and C) legislation related to mangrove plants or ecosystems. Legislation is drawn from the jurisdictions with mangroves present (five states and one territory).

Taxonomic classification of mangrove species and ecosystems

None of the legislative documents describe or define a mangrove plant species or mangrove ecosystem, leaving their classification open to interpretation in the Australian legislation. Government factsheets, status reports and state coastal plans also do not outline mangrove taxonomy, although all jurisdictions have at least one document that provides a general description of mangroves plant biology (e.g. "mangroves are salt tolerant plants", Supplementary table S2). Of these 15 descriptions, 10 classified mangroves as marine, while the remaining did not assign mangroves a classification. Where documents do discuss mangroves, generally the total number of mangrove species is given without detailing the species name or a species list (6 jurisdictions). However, one jurisdiction provided a complete species list for mangroves including identification plates, and two jurisdictions include information about associate mangrove plant species (Table S2).

The protection for vegetated intertidal ecosystems

Protection by native vegetation legislation

In Australia, native vegetation management acts set out the restrictions on clearing of native vegetation. While the specific definition of native vegetation under these laws varies among jurisdictions, they all relate to aquatic and terrestrial plant(s) indigenous to the region of interest, often qualified through specific inclusions or exclusion for the definition (Supplementary table S1). Mangroves are included with equal frequency under native vegetation management or fisheries management legislation (Figure 3C). In New South Wales and Queensland mangrove ecosystems are explicitly excluded from their native vegetation management acts and so are not offered protection from clearing (Supplementary table S1). From the 15 acts relating to the five legislation categories ("Data collection"), three acts specifically mention objectives for mangroves.

Protection within protected areas

I found that approximately 49% (9,391 km^2) of Australia's mangrove ecosystems are within some form of protected area. Across Australia, this protection is relatively equally distributed between marine (2,258 km^2 -24%) and terrestrial (1,989 km^2 -21%) protected areas; however the distribution varies significantly between jurisdictions (Figure 4). My analysis also identified 374 km^2 (4%) as being represented within both marine and terrestrial protected area boundaries, a number close to the 3% of intertidal habitat found to be falling in both protected areas by Dhanjal-Adams et al. (2016). However, due to the aforementioned problems with mapping mangrove extent (Hamilton and Casey, 2016), and potential issues mapping the coastal boundary of protected areas (DotE, 2014), this percentage may be the result of mapping errors.

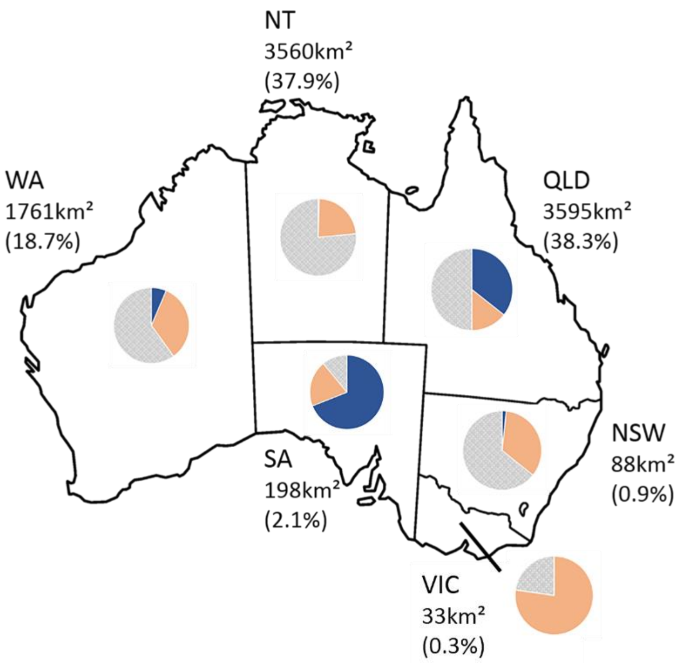

Figure 4. The distribution of protection for mangrove ecosystems across Australia. Pie charts indicate the portion of mangroves in each state that are protected within marine (blue) or terrestrial (orange) protected areas, and without protection (grey). Numbers represent the area of mangroves per jurisdiction, with the number in parentheses being the total percentage (%) of mangroves in Australia. NT is Northern Territory, QLD is Queensland, NSW is New South Wales, VIC is Victoria, SA is South Australia and WA is Western Australia.

The ability of marine or terrestrial protected areas to mitigate threats

Terrestrial protected areas ranked higher in their capacity to mitigate key global threats to mangroves than marine protected areas (Table 1). Terrestrial protected areas are potentially capable of high mitigation of four out of seven manageable threats, while marine protected areas are capable of high mitigation of one threat (Table 1). Three threats are outside direct control of protected areas (e.g. climate change, natural disasters and oil spills).

Discussion

Despite the importance of the ecosystem services provided by mangroves and their increasing global vulnerability to anthropogenic threats, I found that there are many different levels at which Australian governance structures fail to provide clear guidance for their management. Of great concern is that many of these failures represent problems for intertidal ecosystem governance more generally in other regions of the world.

Legislation defining boundaries between land, sea and the intertidal zone

Despite a general definition of the boundaries between land and sea under federal law setting the limit as the High Water Mark, I found that this definition is sufficiently vague that it is interpreted differently across Australia. There are many cases where legislation overlaps in the intertidal zone (Cao and Wong, 2007) requiring a definition of these boundaries to enable their jurisdictional delineation. Nevertheless, I found generally these acts fail to provide a clear definition for the interpretation of the boundary, with one of the most significant problems the level of encroachment from private land into intertidal ecosystems depending on different applications of the boundary between private and public (intertidal) lands. This means intertidal ecosystems are offered varying levels of protection across jurisdictions, which Australia demonstrates can be a significant a problem within a country, and are likely to be most important for areas with significant tidal ranges (Figure 2).

Coherent boundaries between land and sea will become an increasingly important global issue given the predictions for accelerated sea level rise and the potential for coastal development to impede the inland migration of intertidal ecosystems, such as mangroves, under climate change (Rogers et al., 2016). However, even with coherent boundaries the problem is not solved because sea levels rise could place coastal wetlands outside of current protected area boundaries (Rogers et al. 2013). The widespread use of static coastal and protected area boundaries (Rogers and Schofield 2016), will need to change for policy and law to be flexible enough to plan and adapt for the effect of sea level rise (Rogers and Saintilan, 2009). My findings provide clear support for global calls to strengthen governance to ensure these ecosystems are well managed and monitored (Friess et al., 2016; Harris et al., 2014; Rogers et al., 2016), and for a common definition of the intertidal zone to create a basis for adaptations to changing sea levels to be undertaken.

I also found that at present, no jurisdiction in Australia uses a definition of the land-sea boundary that encompasses the full extent of intertidal communities – the highest high tide mark (Highest Astronomical Tide) to the lowest water mark (Low Water Mark) (Figure 1), creating challenges for consistent protection. For intertidal communities on the seaward side of the land-sea boundary, such as seagrass, the impacts of a variable definition is less significant because they occur below the low tide mark, ensuring that no intertidal boundary intersects this ecosystem (Figure 1C). However, for ecosystems that can occur across the tidal range, such as mangroves and saltmarsh, the definition employed by some jurisdictions can place the landward boundary of the intertidal so that part of the ecosystem is considered mostly marine and part terrestrial (Figure 1A & B). The practical implication of such a division is that responsibility is divided between different jurisdictions, creating challenges for comprehensive management. This is clearly illustrated in the division of responsibility for mangrove ecosystems between fisheries and native vegetation management laws (Figure 3C), and likely contributes to the inconsistency with which mangroves are protected within marine versus terrestrial protected areas (see under "Protection within protected areas"). Where legal definitions divide intertidal ecosystems between marine and terrestrial realms, care must be taken to ensure consistent levels of protection are offered to the whole ecosystem. By creating a single, universal definition of the landward land-sea boundary at the highest Astronomical High Tide, it is ensured that vegetated intertidal ecosystems occurring across the tidal range are not divided.

Australia is not alone in having poor governance of intertidal ecosystems, with difficulties related to bounding the intertidal zone reported all over the world (e.g. the High Water Mark is not clearly defined in China (Liu et al., 2014), varying definitions for tidal boundaries are used in Malaysia (Abdullah et al., 2013) and the landward extent of coastal waters is not defined in Europe (Liquete et al., 2011)). This variability in how these boundaries are globally defined creates significant concerns, given the likelihood of poor management of intertidal ecosystems (Dhanjal-Adams et al., 2016) and supports the concerns about the ability to accurately map the extent of intertidal ecosystems necessary to identify global priority areas of decline (Friess and Webb, 2011). While the inconsistency in how the intertidal zone is defined needs to be resolved, I caution against the creation of an additional boundary by creating a separate protected area network for intertidal zones, as has been suggested by others (Banks et al., 2005). My study shows that the definition of boundaries is one of the main difficulties, and as such, creating a separate intertidal zone would double the number of boundaries and the potential for confusion and inconsistent governance.

The marine or terrestrial categorization of intertidal ecosystems by legislation

There is a long history of debate surrounding whether intertidal ecosystems should be considered marine or terrestrial ecosystems, with reference to mangroves in the debate stretching back to 1887 (Duke, 2006b). While I found evidence that the debate is far from over (Mukherjee et al., 2014), mangrove plants and ecosystems were more often characterised as marine ecosystems under Australian legislation and within relevant management documents. This marine portraying of mangroves is not limited to Australia (e.g. Mongabay, 2016) and may come from the emphasis placed on the marine ecosystem services they provide, like coastal protection, fish nurseries and carbon sequestration marketed as 'blue' carbon (McLeod et al., 2011). There has also been a strong historical focus of research into the marine components of mangrove ecosystems (Honda et al., 2013; Ellison, 2008; Faunce and Serafy, 2006). However, this marine focus has come at the expense of a better understanding of the terrestrial value of mangrove ecosystems to biodiversity, the terrestrial ecosystem services they provide (Rog et al., 2017; UNEP, 2014), and has implications for their protection from threats of a terrestrial origin (Table 1). With a terrestrial perspective of the vegetated intertidal ecosystems the most pragmatic approach to mitigate the largest threats to these ecosystems (Table 1).

Taxonomic classification of mangrove species and ecosystems

The widespread uncertainty around the taxonomy of mangrove species and which species are considered true mangroves versus mangrove associated species adds to the global vulnerability of these ecosystems. Firstly, without a clear definition of a mangrove plant, some species may not be included under legislation protecting mangroves, or enough uncertainty is created such that it would be difficult to prosecute individuals who destroy mangroves (Sahide and Giessen, 2015). Mangrove and saltmarsh species that occur in the transition zone with terrestrial vegetation are likely to be most vulnerable to uncertain taxonomy, because they do not always have the distinctive features associated with common families (e.g. air roots in mangroves, fleshy leaves in saltmarsh) and are therefore not always easily recognisable. Secondly, if the plant species included in mangrove and other intertidal ecosystems are not clear it becomes very difficult to determine the boundary between intertidal and adjacent vegetation communities, which are necessary to enable robust mapping and monitoring (e.g. in relation to global extent and change in distribution; Friess and Webb, 2011). Compared to legislation, management plans in Australia tended to provide more detail about which plant species are considered a mangrove (Supplementary table S2). Therefore, resolving the uncertainty around mangrove - and potentially other intertidal plant species - could be helped by creating closer links between legislation and management plans, such that plans support the legislation by providing details about which plant species are considered as mangrove/intertidal plants within a jurisdiction or country.

The protection of vegetated intertidal ecosystems

Protection by native vegetation legislation

The predominantly marine characterisation of mangrove ecosystems may be responsible for the fact that in several Australian jurisdictions mangroves are not protected by the native vegetation legislation that place restrictions on the clearing of indigenous plant species. This leaves them more vulnerable to destruction than other native vegetation communities. Mangrove ecosystems span the intertidal zone from the Highest Astronomical Tide line to the lowest, meaning that the majority of definitions of land-sea boundaries transect this ecosystem (Figure 1). In most cases, part of the intertidal zone is included under marine legislation, and as such, mangroves are only offered partial protection by this legislation (i.e., generally from the Low Water Mark seawards; Figure 1). Interestingly, the definition of tidal lines is not relevant to native vegetation laws in Australia, which include both aquatic and terrestrial vegetation wherever it occurs, thus offering unambiguous protection. The application of habitat protection that does not require accurate definition of dynamic tidal lines, in this case acknowledging mangroves under native vegetation, offers a promising avenue for the protection of vegetated intertidal ecosystems worth exploring globally.

Protection within protected areas

I found that mangrove ecosystems in Australia receive relatively high levels of representation within protected areas (49%) compared to global mangrove protection (6.9%; Giri et al., 2011) and most other threatened ecosystems (UNEP, 2014). Although variable across jurisdictions (Figure 4), the overall representation of mangroves in Australian protected areas is split almost equally across marine and terrestrial protected areas, showing similar patterns to intertidal habitat protection more broadly (Dhanjal-Adams et al., 2016). While this high level of representation within protected areas is generally a positive, it should be noted that the functional protection provided by marine versus terrestrial protected areas is different, offering protection from different threats and providing different approaches to management (Table 1). There is a clear potential for these differences to lead to inadequate management as terrestrial protected areas might not always prioritize their marine environments, and marine parks might underplay the importance of tidal habitats (Dhanjal-Adams et al., 2016).

Mangrove ecosystems, like other intertidal ecosystems, cross the boundary between land and sea, which means that their inclusion in one protected area (marine or terrestrial) leaves at least part of the ecosystem without protection. This approach also ignores the cross realm nature of many of the global threats to intertidal communities (Friess et al., 2016), which originate on land and impact the marine environment (e.g., sedimentation; Alana et al., 2012), or originate in the marine environment and impact land (e.g., sea level rise; Alvarez-Romeo, 2015a). To avoid a situation where protected areas only protect part of the intertidal there is a need to ensure that the way protected areas boundaries are drawn captures the cross realm nature of these ecosystems. Marine and terrestrial protected areas generally occur separately in a landscape, i.e. they do not always abut each other (Fuller et al., 2010). Therefore, to protect the whole intertidal zone, marine protected areas must extend to the Highest Astronomical Tide line, and preferably above that to allow for a buffer zone that could facilitate inland migration in response to sea level rise (Rogers et al., 2016). Likewise, the boundary of terrestrial protected areas should extend to the Low Water Mark, ensuring the whole intertidal zone is protected. These adaptions are in line with the IUCN Mangrove specialist group statement that recommends the expansion of global protected areas to include 30% of mangroves adjacent to terrestrial or marine protected areas by 2020 (IUCN, 2013).

While the proposed expansion of protected areas boundaries would provide consistent protection for intertidal ecosystems it creates the need for clear division of governance responsibilities where marine and terrestrial protected areas potentially overlap one another. It is outside the scope of this paper to discuss optimal management strategies for the intertidal zone, and other authors have provided recommendations for coastal zone management (Atkinson et al. 2016; Buelow and Sheaves 2014) and national and decentralized coastal management (Friess et al. 2016). However, in Australia at least, this situation is less of a concern because the same management agencies are generally responsible for the management of both marine and terrestrial protected areas.

The ability of marine or terrestrial protected areas to mitigate threats

While intertidal ecosystems experience threats originating from both marine and terrestrial environments, many of which are beyond the control of management agencies (e.g., sea level rise, oil spills), I found that for mangrove ecosystems, terrestrial protected areas are likely to be most effective at mitigating their key global threats (Table 1). This analysis contradicts suggestions that mangroves benefit most from increased protection under marine protected areas to reconcile (terrestrial) policy conflicts by protecting mangroves from illegal logging or land conversion threats (Friess et al., 2016). I believe protection for these ecosystems should be prioritized in terrestrial protected areas, because encroachment by coastal development is the most significant threat to intertidal ecosystems globally (Giri et al., 2011; Foster et al., 2013). Not only does terrestrial protection prevent direct habitat loss, but it can also accommodate the landward migration of saltmarsh and mangroves, which is predicted to be an increasing need under climate change induced sea level rise as suitable ecological conditions recede (Alana et al., 2012; Polidoro et al., 2010; Rogers et al., 2016). Marine systems, which are often more affected by threats from land than sea (Stoms et al., 2005), also benefit from a focus on terrestrial protection due to the indirect effect of reduced sedimentation, which is the largest threat to inshore coral reefs (Gilby et al., 2016; Klein et al., 2012) and projected to increase with climate change (Hoegh-Guldberg and Bruno, 2010). While terrestrial protected areas offer protection from the most significant threats, marine protected areas are required to protect the marine ecosystem services that intertidal vegetation provides (e.g. coastal protection and fish nurseries), and to mitigate threats from development within the marine environment (e.g., harbour development. aquaculture) (Table 1). From a marine protected area perspective, the inclusion of mangroves has also been shown to increase their resilience (McLeod and Salm, 2006). These arguments suggest that a combination of marine and terrestrial protected areas is still optimal to mitigate threats to intertidal ecosystems.

There is a lack of specific management objectives for intertidal ecosystems in Australia (Figure 3D) and globally (Harris et al., 2014, Clemens et al., 2014). Even when legal protection is offered it is likely that these ecosystems are not a management priority (Horigue et al., 2016). The failure to formulate specific objectives for these ecosystems may stem from the significant knowledge gaps about how these ecosystems function (Álvarez-Romero et al., 2011) and low resolution spatial data which impedes monitoring (Avery, 2003). There is an urgent need for these knowledge gaps on intertidal systems to be filled and to integrate into both marine and terrestrial protected area management. Without specific management objectives for mangroves and other intertidal ecosystems these environments will continue to be at risk to ineffective protection.

Conclusions

Numerous studies have addressed the complexity of intertidal ecosystem management but have failed to identify solutions to improve governance. The evidence presented here highlights problems within the Australian context, but most of these issues are also challenges for the management of intertidal ecosystems around the world. My finding suggests that there is an urgent need to harmonise the definitions of the intertidal zone boundaries and its vegetation across borders, with a focus on ensuring that the definitions used consider the implications these have for the effective protection and management of intertidal ecosystems and the ecosystem services they provide. At present the failure to ensure this consistency means that intertidal ecosystems receive partial protection, and often little protection from the key threats of terrestrial origin. My study demonstrates the urgent need for a global review of legislation governing the boundaries of land and sea to determine whether the suggestions I offer may provide global solutions to ensuring these critical systems do not fall through the cracks in ecosystem protection and management.

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Table 1. Key global threats to mangroves divided in manageable (1-7) and non- manageable threats (8-10). The ranking of a protected area (PA) on its capability to mitigate a threat (low - high) was based on three criteria: i) area protected (e.g. whole area protected from clearing or only partly protected), ii) flow- on effects of protection to other realm (e.g. prevention of clearing upland also stops sediment run off into sea) and iii) regulating an activity. S1a references.

		Threats	**How are mangroves affected**	**Rank marine PA in mitigating threat**	**Rank terrestrial PA in mitigating threat**	**Argumentation**
Manageable	1	Coastal development for housing, hotels, harbours, infrastructure and agriculture	Destruction by clearing and filling of the intertidal zone. Affecting total mangrove forest by: i) taking away the physical plants, i) negatively affecting the mangrove forest by altering tidal flows (Ammanulla et al., 2014), iii) reducing connectivity to adjacent vegetation possibly affecting dispersal of facultative mangrove users (Rog et al., 2017), iv) coastal squeeze where mangroves have no possibility to migrate inland when sea levels rise due to climate change (Rogers et al., 2016), and v) dieback of mangroves by pollutant run-off (Duke, 2005)	Low	**High**	Terrestrial PA's can protect the whole mangrove area from clearing when they extend to the low water mark. Terrestrial protection also has a flow-on effect on MPA's by decreasing the main threat of sedimentation and regulate terrestrial activities like development of infrastructure. MPA's can generally capture development up to the low water mark, but sometimes prohibit development directly adjacent to the MPA therby protecting larger parts of mangroves. Estuaries, river mounth and rivers (areas generally holding large parts of mangrove forests) often do not fall under MPA's and development along those banks can therefore only be mitigated by terrestrial PA's.
	2	Coastal aquaculture	Destruction by clearing of the intertidal zone to make space for shrimp farms or caged fish farms. Affecting total mangrove forest by: i) taking away the physical plants for ponds and infrastructure, ii) loss of habitats and nursery areas, iii) coastal erosion, iv) reduced biodiversity, v) acidification vi) and alteration of water drainage patterns (Paez -Osuna, 2001).	**High**	**High**	MPA's can mitigate aquaculture as this is generally not allowed within MPA boundaries. Aquaculture however is a relatively new industry and no clear guidelines are established throughout all jurisdictions. Terrestrial PA's can mitigate aquaculture by halting terrestrial infrastructure development and development of shrimp ponds above the low tide line. When mangroves upland are protected from clearing the runnoff of sediment and pollution to MPA's is prevented as well.
	3	Mining	Land mining for mineral products (iron,nickel, bauxite) affect mangroves through toxic waste and alteration of natural flows (Ohimain, 2003). Coral minging can affect mangroves by increased erosion rates caused by the loss of reef breakwaters (Dulvy, 1995).	Medium	**High**	Coral mining can be mitigated by MPA's. Land mining can be mitigated by terrestrial PA's with flow on effects to marine areas by preventing sediment and pollution runoff.
	4	Commercial exploitation fish shelfish	Mangroves are degraded by construction of boat moarings and infrastructure for land access.Fishing can alter foodwebs (e.g. less top predators), and can cause pollution by garbage and oilspills (Davenpoort, 2006)	Medium	Medium	Fisheries can be mitigated and regulated by MPA's by assigning certain periods of non-fishing, areas of no take, and regulations for waste disposal. Intertidal fisheries are generally included in MPA's, esturaries and river mouths (where large parts of mangroves occur) are sometimes regarded as internal water where it is not clear where the boundary lies for protection. Regulating fisheries might have a positive flow on effect on terrestrial parts of mangroves by keeping marine and links between marine-land foodwebs in tact. Terrestrial PA's can provide partial protection by mitigating or regulating development of boat moorings and infrastructure to transport catches and boats, thereby having a positive flow on effect on MPA's through decreased sedimentation runoff.
	5	Tourism	Tourism requires transport that links coastal road to hotels (Davenpoort, 2006). Sightseeing causes trampling of saplings (Ross, 2006), pollution and increase boat wash causes erosion (Davenport, 2006) and damage by anchoring (Burgin, 2011). Off road recreation vehicles affect mudflats (Bridgewater, 1999)	Medium	Medium	MPA's can regulate boating activities anchoring and waste disposal. For intertidal activity (kajakking in estuaries, walking along shore) regulation is not clear and depends on boundaries of MPA's. Terrestrial PA's can egulate tourism by guiding access to mangrove areas through boardwalks and assigning non-access areas for 4WD's.

		Threats	How are mangroves affected	Rank marine PA in mitigating threat	Rank terrestrial PA in mitigating threat	Argumentation
Manageable	6	Pests and diseases	Degrading of mangroves by invasive plant species, for example *Annona glabra* known as Pond Apple (Duke, 2006) are capable of replacing whole mangrove ecosystems and 23 invasive plant species are present in Sundarban mangroves (Biwas et al., 2007). Lionfish use mangroves for feeding ground habitat in Bahamas (Barbour et al., 2010) potentially affecting marine mangrove communities. Invasive black rats have been found to predate on birds nesting in mangroves (Harper et al., 2015), and invasive deer are known to trample and eat saplings.	Low	Medium	Being unable to see sub-tidal features poses particular problems in terms of management of marine species that may damage features within an MPA. Appropriate monitoring or surveillance can be undertaken but is expensive, requiring SCUBA diving. Regulations regarding the discharge of ballast water could help prevent marine invasive species spread. Terrestrial PA's can control terrestrial invasive plants and animals that invade mangroves from the landward side.
	7	Local exploitation	Exploitation for wood/charcoal, fish, crabs and other wildlife (Hoq, 2007). Grazing by domestic cattle (Hoppe-Speer et al., 2015). Degrading by accessing, trampling, clearing, burning, pollution and altering foodwebs.	Medium	**High**	MPA's can set up no take areas and regulations regarding fisheries. Terrestrial PA's can set up no take areas and regulations for wildlife use, (dead) wood harvesting and pollution. The prevention of the latter has a positive flow on effect on MPA's. These regulations however often do not apply to traditional owners which makes it difficult to assess the total use and the related pressures.
Not manageable	8	Climate change	Mangroves are affected by sea level rise and when this occurs faster than they can migrate inland, or when this migration is hampered by coastal squeeze from coastal development this can cause them to drown (Rogers et al., 2016). Climate change causes differences in weather patterns that is likely linked to recent dieback of 7000km mangroves in Nothern Australia (Duke, 2017).	na	Low	MPA's cannot mitigate this threat as MPA's can not set aside land for landwards migration of mangroves. Terrestrial PA's could potentially mitigate part of this threat by regulations related to buffer zones around development that allow mangroves to migrate inland. Terrestrial PA's could also stop development of artificial sea walls that causes erosion and lowers mangrove establishment.
	9	Natural disasters	Mangroves need years to recover from cyclone damage (Chotray et al., 2012)	Low	Low	Only indirect mitigation by both MPA's and terrestrial PA's by striving for healthy ecosystems. More resiliant mangroves can possibly withstand changes better due to genetic and phenotipic diversity, larger patches have also more possibility to re-establish due to higher amount of propagules.
	10	Oil spills from boats	Oil causes dieback of mangroves by smothering root systems (Duke, 2002)	Low	Low	Only indirect mitigation by both MPA's and terrestrial PA's by striving for healthy ecosystems. More resiliant mangroves can possibly withstand changes better due to genetic and phenotipic diversity, larger patches have also more possibility to re-establish due to higher amount of propagules.

Chapter five - Supplementary details

Additional reference to statement: This can have serious implications in the many cases where the national and international legislation that overlaps in the intertidal zone has inconsistent laws and regulations (Cao and Wong, 2007; Ache et al., 2015) and competing and unclear objectives (Friess et al., 2016) leading to ineffective protection of this zone.

Ache, B. W., Crossett, K. M., Pacheco, P. A., Adkins, J. E., & Wiley, P. C. (2015). “The coast” is complicated: a model to consistently describe the nation’s coastal population. Estuaries and coasts, 38(1), 151-155.

The reference Giri et al., 2011 was used in error for the statement: Globally, 35% of the total area of extent mangroves forests was lost over a recent 30 year period. The correct reference should be: Hamilton, S.E. & Casey, D. (2016) Creation of a high spatio-temporal resolution global database of continuous mangrove forest cover for the 21st century (CGMFC-21). Global Ecology and Biogeography, 25, 729-738.

Foster et al., has been used in error for the statement: ….”coastal development is the most significant threat to intertidal ecosystems globally”. The correct reference should be: UNEP (2014). The Importance of Mangroves to People: A Call to Action. van Bochove, J., Sullivan, E., Nakamura, T. (Eds). United Nations Environment Programme World Conservation Monitoring Centre, Cambridge. 128 pp.

Addition to section “The marine or terrestrial categorization of intertidal ecosystems by legislation”: In some cultures mangroves are conceptualised as terrestrial systems: In Southeast Asia for example, the governance of mangroves has been primarily terrestrial for at least 100 years (Jusoff, 2009), related to mangroves being managed by colonial Forestry Departments. Blue Carbon is could also be considered as a terrestrial and not a marine concept (despite the name), because carbon ecosystem services were first studied in terrestrial forests, and the use of carbon sequestration as an important impetus for conservation first came from tropical terrestrial systems (Lewis et al., 2009).

Jusoff, K. (2009). Managing sustainable mangrove forests in Peninsular Malaysia. Journal of Sustainable Development, 1(1), 88.

Lewis, S.L., Lopez-Gonzalez, G., Sonké, B., Affum-Baffoe, K., Baker, T.R., Ojo, L.O., Phillips, O.L., Reitsma, J.M., White, L., Comiskey, J.A. and Ewango, C.E., 2009. Increasing carbon storage in intact African tropical forests. Nature, 457(7232), pp.1003-1006.

Mangrove-like plants are plant species that are often grouped as “mangrove associates”.

To quantify the proportion of Australian mangrove ecosystems within protected areas I conducted a spatial analysis to determine the extent of mangrove ecosystems within marine protected areas, terrestrial protected areas and with no protection. These analyses were conducted in ArcGIS using the global mangrove distribution layer from Hamilton and Casey 2016. They however define mangroves as trees over 2 m tall, which omits most of the scrub forests of the world. Including the < 2 m mangroves occurring in Southern Victoria, Western Australia and South Australia could have altered the results. Future work could calculate how much area these shrublike mangroves encompass in protected areas and how much has been lost.

Chapter Six: Discussion

Biological diversity underpins ecosystem functioning and the provision of ecosystem services essential for human well-being (Krebs, 1972). Biological diversity is being rapidly lost, which threatens ecosystem functioning, requiring action to effectively conserve it. To identify management priorities to protect ecosystems, both data on species richness and information on species identity and ecology are required (Fleishman *et al.*, 2006). For some ecosystems however, we know less than others. Less well known ecosystems can result from human-made barriers, such as those in regions with long running conflict political conflict (e.g. The Darien rainforest in Colombia has only since 2016 been accessible to scientists (Negret *et al.*, 2017)), or through natural barriers where challenging survey conditions have impeded data collection (e.g.; the deep sea, Haedrich *et al.*, 2001; mountain cliffs, Larson *et al.*, 2005; or cave communities, Weinstein & Slaney, 1995). When we have no understanding of species richness and their ecological relationships, the risks are that we cannot effectively conserve and manage ecosystems and their biodiversity into the future.

Mangroves suffer from limited understanding of their ecology due to challenging survey conditions. Due to their position between land and sea they cross marine and terrestrial realms and therefore have both marine and terrestrial components. There is greater knowledge on biodiversity and ecosystem services from a marine perspective, for example their importance as nursery grounds for fisheries, their role in coastal protection and biological connectivity with other marine ecosystems (e.g. seagrass and coral reefs; (Olds *et al.*, 2012). However, far less is known about their terrestrial biodiversity and their ecological links with terrestrial systems. My review on the current state of global knowledge on terrestrial vertebrate fauna in mangrove forests (Objective 1) revealed that less than 20 published field surveys underpin our global understanding on the occurrence of terrestrial mammals, reptiles and amphibians in mangroves (Rog *et al.*, 2017). New studies are revealing the importance of biological connectivity between mangroves and surrounding terrestrial ecosystems

(Buelow & Sheaves, 2014; Reef *et al.*, 2014). Broad ecological patterns highlighted by my research however suggest that our understanding of terrestrial species in mangroves, and their ecological relationships with these ecosystems, is still far from sufficient.

The strong emphasis on the role of mangroves in marine systems seems to lead to a marine bias in their governance (Rog & Cook, 2017). My research evaluating the adequacy of existing Australian governance structures for the protection and management of mangrove ecosystems (Objective 4), has shown that the emphasis on their role in the marine realm leads to major gaps in their protection that risk the failure to identify and respond to key threats and management issues affecting mangroves that arise from the terrestrial realm (Rog & Cook, 2017). The fact that mangroves have a cross-realm nature, growing in the dynamic intertidal zone and integrate with adjacent terrestrial vegetation, has made it exceptionally difficult to define the boundaries of these forests. Tidal boundaries can be defined in a variety of ways, and tidal ranges are dynamic (Friess *et al.*, 2016). This creates a significant challenge for governance structures, such as legislation, which must provide clear definitions. This is a challenge for all intertidal ecosystems, but more so for mangroves which are divided by any of the currently used definitions of the boundary between marine and terrestrial systems (Rog & Cook, 2017). This blurring of the boundaries for mangroves leads to inconsistent classification of the taxonomy of mangrove plant species and the status of the ecosystem among marine and terrestrial realms, along with consistent jurisdiction responsible for their management (Rog & Cook, 2017). My research has identified that the existing governance structures for the intertidal zone in Australia mean that mangrove forests would benefit most from the acknowledgement of mangroves as terrestrial plants, a focus on mitigation of terrestrial threats, and keeping mangrove forests undivided when included within a protected area, to ensure their governance is more consistent and with that, more effective.

Improving the understanding of the terrestrial components of mangrove ecosystems means we need tools to fill ecological knowledge gaps to increase our ability to effectively manage the biodiversity of the whole ecosystem. There are risks to fauna and field researchers associated with traditional field survey approaches. These need to be addressed and overcome to facilitate a greater understanding of the terrestrial fauna in mangroves. My research developed a survey approach for terrestrial vertebrates that overcomes challenges associated with mangrove environments (Objective 2). Importantly, my research revealed the importance of using a mix of techniques to detect the range of fauna that may occur in mangrove forests, and that even rapid surveys can provide significant new knowledge on the species richness and important ecological patterns - including the presence of threatened and invasive species. The tested field approach could provide a valuable tool to increase the collection of empirical data not only in mangroves but also in other under surveyed ecosystems that face inundation challenges, like flooded forests (Haugaasen & Peres, 2005).

While my research has added significant empirical data on the occurrence of terrestrial fauna in mangroves, including some signs of resource use, there are still significant knowledge gaps about the ecological relationships and services provided by both terrestrial fauna and mangrove forests. With a better understanding of the ecology of these systems, we can identify biotic and abiotic relationships critical to the health of mangrove forests, and better predict the current and future importance of these ecosystems for the threatened species they contain. This would then allow us to paint a more complete picture of important areas of mangroves for biodiversity conservation that can be used to inform conservation planning (Atkinson *et al.*, 2016).

My thesis has provided important new knowledge to inform the more effective protection of mangrove forests by assessing the value of ecological field data on terrestrial vertebrates present in mangroves to inform mangrove management strategies in Australia (Objective 3). It has highlighted the significance of the knowledge gaps associated with these ecosystems, and the risks these knowledge gaps pose for the conservation of the critical ecosystem services they provide. A key research priority identified by my research is the exploration of the role of species that move in and out of mangrove forests, presumably only using them for some of the resources they require (i.e., facultative users). As the loss of mangroves and adjacent habitats continue, it will be important to understand whether mangroves can provide all the resources these species require, particularly if the role of mangroves as a refuge becomes more important. Likewise, how the services mangroves provide to these species might vary for degraded or restored mangroves requires further understanding of resource provision. The prevalence of invasive species within mangrove forests revealed by my research means that more research is needed to understand the impacts of these species both on the mangrove forests themselves and the native species that use them. The linear nature of mangroves, and the prevalence of species that seem to move in and out of the forests mean they could be important for connecting fragmented habitat. This could be used to inform protected area planning with the view to increasing connectivity. These examples all demonstrate that we are in need of more research on the interactions between mangroves and adjacent habitats in the dynamic zone between land and sea.

Conclusion

My thesis reveals that the terrestrial side of mangroves is more important than previously thought and that we are just starting to completely understand these ecosystems. This study outlines a new agenda for further research and highlights improvements to management and governance required for their protection. Given the rate of loss of mangroves, we need to urgently address the identified gaps to be able to halt the decline of these vital forests.

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Supplementary material

Supplementary material Chapter two. More than marine: revealing the critical importance of mangrove ecosystems for terrestrial vertebrates

Supplementary table S1. Terrestrial vertebrates reported to occur in mangroves

Supplementary file S1 references with table S1

Supplementary table S1. Terrestrial vertebrates reported to occur in mangroves (mammals, reptiles and amphibians)

	Scientific name	Common name	Nr of individuals	Evidence for occurance in mangrove	Number of sources*	Sources that reported the species in mangroves
1	*Acrochordus granulatus*	Little file snake	–	Reports from multiple sources	6	Allison 2006; Kutt 1994; Luther and Greenberg 2009; Hutchlings 1982; Karangkutar 2013; Rajpar 2014; Sandilyan 2012
2	*Aipysurus duboisii*	Dubois' sea snake	Very few recorded	1 museum specimen	1	Kutt 1994
3	*Aipysurus eydouxii*	Beaded sea snake	–	Multiple sources	2	Hutchlings 1982; Kutt 1994
4	*Alligator mississippiensis*	American alligator	–	Mangroves listed as habitat	1	ARKive 2016
5	*Alouatta caraya*	Howler monkey	–	Generally reported to occur in mangroves	1	WWF 2016
6	*Alouatta palliata*	Mantled howler monkey	–	Generally reported to occur in mangroves	1	WWF 2016
7	*Alouatta pigra*	Guatemalan black howler	–	Generally reported to occur in mangroves	1	ARKive 2016
8	*Alouatta seniculus*	Red howler monkey	–	Generally reported to occur in mangroves	1	WWF 2016
9	*Anolis allisoni*	Allisons anolis	Multiple recorded	Multiple sightings +10 individuals	1	Charruau 2015
10	*Anolis grahami*	Graham's anole	–	Generic mangroves as habitat	1	Thomas and Logan 1992
11	*Antechinus bellus*	Fawn Antechinus	Very few recorded	Reported less than 5 records over 2 years	1	Metcalfe 2007
12	*Aonyx capensis*	African clawless otter	–	Multiple sources	2	Angelici 2005; Luiselli 2015
13	*Aonyx cinerea*	Small clawed otters	–	Multiple sources	2	Katherisan 2001; ARKive 2016
14	*Arctocebus calabarensis*	Angwantibo	–	Multiple sources	2	Nowak 2013; Luiselli 2015
15	*Aristelliger georgeensis*	Saint George Island Gecko	Multiple recorded	Multiple sightings +10 individuals	1	Charruau 2015
16	*Artibeus aztecus*	Aztec fruit-eating bat	–	Generally reported to occur in mangroves	1	WWF 2016
17	*Artibeus phaeotis*	Pygmy fruit eating bat	–	Generally reported to occur in mangroves	1	WWF 2016
18	*Artibeus toltecus*	Toltec fruit-eating bat	–	Generally reported to occur in mangroves	1	WWF 2016
19	*Astrotia stokesii*	Stokes' seasnake	Very few recorded	1 museum specimen	1	Kutt 1994
20	*Ateles geoffroyi*	Black-handed spider monkey	–	Multiple sources	2	ARKive 2016; WWF 2016
21	*Atilax paludinosus*	Marsh mongoose	–	Multiple sources	2	Blench 2007; Luiselli 2015
22	*Axis axis*	Axis deer, Chintal deel, Spotted deer	–	Multiple sources	3	Hogarth 2007; Katherisan 2001; Rajpar 2014
23	*Axis porcinus*	Hog deer	–	Multiple sources	2	Katherisan 2001; WWF 2016
24	*Baiomys musculus*	Southern pygmy mouse	–	Generally reported to occur in mangroves	1	WWF 2016
25	*Basiliscus basiliscus*	Basilisk lizard	–	Generally reported to occur in mangroves	1	WWF 2016
26	*Blarina carolinensis*	Southern short-tailed shrew	–	Generally reported to occur in mangroves	1	ARKive 2016
27	*Boa constrictor*	Boa	–	Generally reported to occur in mangroves	1	WWF 2016
28	*Boaedon lineatus*	Striped House Snake	Very few recorded	Reported 4 times over 355 hours	1	Luiselli 2002
29	*Boiga irregularis*	Brown tree snake	–	Generally reported to occur in mangroves	1	Hutchlings 1982
30	*Boiga wallachi*	Yellow ringed snake	–	Multiple sources, listed as obligate mangrove user	6	Hogarth 2007; Luther and Greenberg 2009; Nagelkerken 2008; Norma Rashid 2012; Rajpar 2014; Sandilyan 2012
31	*Bradypus pygmaeus*	Pygmy three-toed sloth	–	Multiple sources, listed as obligate mangrove user	4	Luther and Greenberg 2009; Sandilyan 2012; ARKive 2016; Voirin 2015
32	*Bradypus variegatus*	Brown-throated three-toed sloth	–	Generally reported to occur in mangroves	1	WWF 2016
33	*Bubalus arnee*	Water buffalo	–	Listed as common	1	Hogarth 2007
34	*Bubalus bubalis*	Water buffalo	–	Multiple sources	3	Dahdouh Guebas 2006; Hogarth 2007; Katherisan 2001

35	*Bufo marinus*	Canetoad, Giant toad	–	Multiple sources	2	Rajpar 2014; Kutt 1994
36	*Bufo melanostictus*	Asian common toad	–	Generally reported to occur in mangroves	1	Voris 2002
37	*Caiman crocodilus*	The spectacled caiman	–	Multiple sources	3	Hogarth 2007; Rajpar 2014; WWF 2016
38	*Caiman crocodilus chiapasius*	Caiman subspecies	–	Generally reported to occur in mangroves	1	WWF 2016
39	*Caiman latirostris*	Broad snouted caiman	–	Generally reported to occur in mangroves	1	ARKive 2016
40	*Callithrix kuhlii*	Wied marmoset	–	Reported to be common	1	Rodriques 2014
41	*Callosciurus notatus*	Plantain squirrel	–	Multiple sources	2	Norma Rashid 2012; Zakaria 2015
42	*Calotes versicolor*	Oriental garden lizard	–	Multiple sources	2	Karangkutar 2013; Voris 2002
43	*Camelus bactrianus*	Domestic camels	–	Generally reported to occur in mangroves	1	Hogarth 2007
44	*Canis latrans*	Coyote	–	Generally reported to occur in mangroves	1	WWF 2016
45	*Cantoria annulata*	Banded Mangrove Snake	–	Generally reported to occur in mangroves	1	Allison 2006
46	*Cantoria violacea*	Yellow-banded, Cantor water snake	–	Multiple sources, listed as obligate mangrove user	3	Luther and Greenberg 2009; Voris 2002; Sandilyan 2012
47	*Capromys pilorides*	Hutia	–	Generally reported to occur in mangroves	1	WWF 2016
48	*Caracal aurata*	African golden cat	–	Generally reported to occur in mangroves	1	Nowak 2013
49	*Cebus apella*	Capuchin monkey	–	Multiple sources,	2	Katherisan 2001; WWF 2016
50	*Cebus apella apella*	Capuchin monkey subspecies	–	Generally reported to occur in mangroves	1	WWF 2016
51	*Cebus capucinus*	White throated capucin	–	Multiple sources	3	Rajpar 2014; WWF 2016; Yaap 2015
52	*Cephalophus nigrifrons*	Black-fronted Duiker	Very few recorded	2 records	1	Blench 2007
53	*Cephalophus silvicultor*	Yellow-backed Duiker	Very few recorded	1 recorded	1	Blench 2007
54	*Cerberus australis*	Australian bockdam	–	Obligate mangrove user, listed as obligate	1	Luther and Greenberg 2009
55	*Cerberus rynchops*	Bockadam, dog faced water snake	–	Multiple sources, listed as obligate mangrove user	9	Allison 2006; Hutchlings 1982; Karangkutar 2013; Luther and Greenberg 2009; Norma Rashid 2012; Rajpar 2014; Rajpar 2014; Sandilyan 2012; Voris 2002
56	*Cercocebus atys*	Sooty mangabey	–	Generic mangroves as habitat	1	Nowak 2013
57	*Cercocebus torquatus*	Red capped mangabay	–	Multiple sources	3	Blench 2007; Nowak 2013; ARKive 2016
58	*Cercopithecus campbelli*	Campbell’s guenon	–	Generally reported to occur in mangroves	1	Nowak 2013
59	*Cercopithecus cephus*	Moustached guenon	Very few recorded	Reported as rarely sighted	1	Nowak 2013
60	*Cercopithecus erythrogaster*	White-throated guenon	–	Generally reported to occur in mangroves	1	Nowak 2013
61	*Cercopithecus mitis*	Sykes monkey	–	Generally reported to occur in mangroves	1	WWF 2016
62	*Cercopithecus mitis albogularis*	Sykes monkey	–	Generally reported to occur in mangroves	1	Nowak 2013
63	*Cercopithecus mitis albotorquatu*	Sykes monkey	–	Generally reported to occur in mangroves	1	Nowak 2013
64	*Cercopithecus mitis labiatus*	Samango	–	Generally reported to occur in mangroves	1	Nowak 2013
65	*Cercopithecus mona*	Mona monkey	–	Multiple sources	3	Blench 2007; Nowak 2013; Luiselli 2015
66	*Cercopithecus neglectus*	De Brazza’s monkey	–	Generally reported to occur in mangroves	1	Nowak 2013
67	*Cercopithecus petaurista*	Lesser white-nosed monkey	–	Generally reported to occur in mangroves	1	Nowak 2013
68	*Cercopithecus sclateri*	Sclater's monkey	–	Generally reported to occur in mangroves	1	WWF 2016
69	*Cerdocyon thous*	Crab-eating fox	–	Generally reported to occur in mangroves	1	WWF 2016
70	*Cervus duvaucelii*	Swamp deer	–	Listed as common	1	Hogarth 2007
71	*Chaerephon spp.*	Free tailed bat	Multiple recorded	44 times reported	1	Mc Kenzie 1986
72	*Chalinolobus gouldii*	Gould's wattled bat	–	Generic mangroves as habitat	1	Hogarth 2007
73	*Chalinolobus nigrogriseus*	Hoary wattled bat	Multiple recorded	50 times reported	1	Kutt 1994
74	*Cheirogaleus medius*	Fat tailed dwarf lemur	Very few recorded	Unconfirmed observation 1x	1	Gardner 2016

75	*Chiropotes satanas*	Bearded saki monkey	–	Generally reported to occur in mangroves	1	WWF 2016
76	*Chlorocebus pygerythrus*	Vervet monkey	–	Generally reported to occur in mangroves	1	Nowak 2013
77	*Chlorocebus pygerythrus hilgertis*	Vervet monkey	–	Generally reported to occur in mangroves	1	Nowak 2013
78	*Chlorocebus sabaeus*	Green monkey	–	Generally reported to occur in mangroves	1	Nowak 2013
79	*Choeropsis liberiensis heslopi*	Pygmy hippopotamus	–	Generally reported to occur in mangroves	1	WWF 2016
80	*Civettictis civetta*	Civet	–	Multiple sources	2	Juliana 2011; Luiselli 2015
81	*Cnemidophorus spp.*	Whiptail lizard	–	Generally reported to occur in mangroves	1	WWF 2016
82	*Colobus angolensis*	Angolan black-and-white colobus	–	Generally reported to occur in mangroves	1	Nowak 2013
83	*Conepatus leuconotus*	Hog-nosed skunk	–	Generally reported to occur in mangroves	1	WWF 2016
84	*Conilurus spp.*	Rabbit rat	–	Generally reported to occur in mangroves	1	Hutchlings 1982
85	*Corallus hortulanus*	Arboreal snake	–	Generally reported to occur in mangroves	1	WWF 2016
86	*Crayia smythii*	Smyth's Water Snake	Very few recorded	Reported 4 times over 355 hours	1	Luiselli 2002
87	*Cricetomys spp*	Giant pouched rat	–	By multiple sources in studies recorded (field, market, intervieuws)	1	Luiselli 2015
88	*Crocodilus acutus*	American crocodile	–	Multiple sources	4	Hogarth 2007; ARKive 2016; WWF 2016; Venegas 2015
89	*Crocodylus johnstoni*	Fresh water crocodile	–	Listed as rare	1	Hutchlings 1982
90	*Crocodylus moreletti*	Morelet’s crocodile	–	Generally reported to occur in mangroves	1	WWF 2016
91	*Crocodylus niloticus*	Nile crocodile	–	Multiple sources	3	Hogarth 2007; ARKive 2016; WWF 2016
92	*Crocodylus palustris*	Marsh Crocodile	–	Multiple sources	2	Rajpar 2014; WWF mangrove region
93	*Crocodylus porosus*	Estuarine, Saltwater crocodile	–	Multiple sources	10	Allison 2006; Gopal 2006; Hogarth 2007; Hutchlings 1982; Katherisan 2001; Kutt 1994; Nagelkerken 2008; Rajpar 2014; ARKive 2016; WWF 2016
94	*Crocuta crocuta*	Spotted hyena	–	Generally reported to occur in mangroves	1	Nowak 2013
95	*Crossarchus platycephalus*	Flat headed kusimanse	–	By multiple sources in studies recorded (field, market, intervieuws)	1	Luiselli 2015
96	*Crotalus durissus*	Brazilian rattlesnake	–	Generally reported to occur in mangroves	1	WWF 2016
97	*Crotaphopeltis hotamboeia*	Red-lipped snake	Very few recorded	Reported 4 times over 355 hours multiple years and field seasons	1	Luiselli 2002
98	*Cryptotis nigrescens*	Blackish small-eared shrew	–	Generally reported to occur in mangroves	1	WWF 2016
99	*Ctenosaura bakeri*	Utila spiny tailed iguana	–	Generally reported to occur in mangroves, listed as obligate	1	ARKive 2016
100	*Ctenosaura similis*	Black spiny-tailed iguana	–	Generally reported to occur in mangroves	1	WWF 2016
101	*Cuniculus paca*	Paca, spotted paca, augouti	–	Multiple sources	2	WWF 2016; Yaap 2015
102	*Cuon alpinus*	Wild dog	–	Generally reported to occur in mangroves	1	WWF 2016
103	*Cyclopes didactylus*	Pygmy anteater, Silky anteater	–	Generally reported to occur in mangroves	1	WWF 2016
104	*Cyclura cychlura*	Northern Bahamian rock iguana	–	Generally reported to occur in mangroves	1	ARKive 2016
105	*Cyclura nubila*	Ground iguana	–	Generally reported to occur in mangroves	1	ARKive 2016
106	*Cyclura rileyi*	San Salvador iguana	–	Generally reported to occur in mangroves	1	ARKive 2016
107	*Cynogale bennettii*	Otter civet	–	Generally reported to occur in mangroves	2	Norma Rashid 2012
108	*Cynopterus brachyotis*	The lesser short-nosed fruit bat	–	Generally reported to occur in mangroves	1	Rajpar 2014
109	*Dasypeltis fasciata*	Egg Eating Snake	Very few recorded	Reported 4 times over 355 hours multiple years and field seasons	1	Luiselli 2002
110	*Dasyprocta guamara*	Orinoco agouti	–	Generally reported to occur in mangroves	1	WWF 2016
111	*Dasypus novemcinctus*	Nine banded armadillo	–	Generally reported to occur in mangroves	1	WWF 2016
112	*Dasyurus hallucatus*	Northern quoll	–	Listed as common	1	Metcalfe 2007
113	*Daubentonia madagascariensis*	Aye-aye	–	Multiple sources	2	Nowak 2013; Gardner 2016
115	*Dendrelaphis punctulatus*	Green tree snake	–	Generally reported to occur in mangroves	1	Hutchlings 1982

116	*Dendroaspis jamesoni*	Jameson's mamba	Very few recorded	Reported 1 time over 355 hours multiple years and field seasons	1	Luiselli 2002
117	*Dendrobatidae spp.*	Poison dart frog	–	Generally reported to occur in mangroves	1	WWF 2016
118	*Dipodomys compactus*	Gulf Coast kangaroo rat	–	Generally reported to occur in mangroves	1	WWF 2016
119	*Duttaphrynus melanostictus*	Asian common toad	Multiple recorded	189 individuals reported	1	Jena et al 2013
120	*Duttaphrynus stomaticus*	Indian marbeled toad	Very few recorded	2 individuals reported	1	Jena et al 2013
121	*Eidolon helvum*	Straw-coloured fruit bat	–	Generally reported to occur in mangroves	1	ARKive 2016
122	*Eira barbara*	Tayra	–	Generally reported to occur in mangroves	1	WWF 2016
123	*Elephas maximus*	Wild elephant	–	Generally reported to occur in mangroves	1	WWF 2016
124	*Eleutherodactylus caribe*	Mangrove frog	–	Multiple sources, listed as obligate mangrove user	4	Hedges 1992; Luther and Greenberg 2009; Nagelkerken 2008; Sandilyan 2012
125	*Eleutherodactylus flavescens*	Tree frog	–	Generally reported to occur in mangroves	1	Sangermano 2015
126	*Emballonura furax*	New Guinea sheathtail bat	–	Generally reported to occur in mangroves	1	WWF 2016
127	*Emoia atrocostata*	Litorial, Mangrove skink	–	Multiple sources, listed as obligate mangrove user	4	Hutchlings 1982; Allison 2006; Rajpar 2014; Zakaria 2015
128	*Enhydrina schistosa*	Hook nosed sea snake	–	Listed as common	1	Karangkutar 2013
129	*Enhydris polylepis*	Macleay's water snake	–	Generally reported to occur in mangroves	1	Allison 2006
130	*Eonycteris spelaea*	Cave fruit bat , Cecadu bat	–	Multiple sources	2	Hogarth 2007; Norma Rashid 2012
131	*Ephalophis greyae*	North western mangrove snake	–	Multiple sources	3	Hutchlings 1982; Nagelkerken 2008; ARKive 2016
132	*Ephalophis mertoni*	Sea snake	–	Generally reported to occur in mangroves	1	Hutchlings 1982
133	*Epicrates striatus fosteri*	Bimini boa	–	Generally reported to occur in mangroves	1	Jennings 2012
134	*Epicrates subflavus*	Jamaican boa	–	Generally reported to occur in mangroves	1	ARKive 2016
135	*Eptesicus spp.*	House bat	–	Generally reported to occur in mangroves	1	Mc Kenzie 1986
136	*Erythrocebus patas patas*	Patas monkey	–	Generally reported to occur in mangroves	1	Nowak 2013
137	*Eulemur albifrons*	White-fronted brown lemur	Very few recorded	1 reported	1	Gardner 2016
138	*Eulemur coronatus*	Crowned lemur	–	Generally reported to occur in mangroves	1	Gardner 2016
139	*Eulemur flavifrons*	The blue-eyed black lemur	–	Generally reported to occur in mangroves	1	Gardner 2016
140	*Eulemur fulvus*	The common brown lemur	Multiple recorded	Groups observed several times	1	Gardner 2016
141	*Eulemur macaco*	Black lemur	Very few recorded	1x a large group reported	1	Gardner 2016
142	*Eulemur mongoz*	Mongoose lemur	Multiple recorded	Groups observed several times	1	Gardner 2016
143	*Eulemur rufus*	Red lemur	Multiple recorded	Groups observed several times	1	Gardner 2016
144	*Eulemur sanfordi*	Sanford's brown lemur	Very few recorded	1 recorded	1	Gardner 2016
145	*Eunectes murinus*	Anaconda	–	Generally reported to occur in mangroves	1	WWF 2016
146	*Euphlyctis cyanophlyctis*	Indian skipper frog	Multiple recorded	83 individuals reported	1	Jena et al 2013
147	*Euphlyctis hexadactylus*	Indian green frog	Multiple recorded	155 individuals reported	1	Jena et al 2013
148	*Fejervarya cancrivora*	Crab eating frog	–	Multiple sources	2	Rajpar 2014; Chandramouli 2015
149	*Fejervarya orissaensis*	Orissa frog	Multiple recorded	106 individuals reported	1	Jena et al 2013
150	*Fejervarya syhadrensis*	Bombai wart frog	Multiple recorded	29 individuals recported	1	Jena et al 2013
151	*Fordonia leucobalia*	White bellied mangrove snake	–	Multiple sources, listed as obligate mangrove user	5	Hutchlings 1982; Luther and Greenberg 2009; Sandilyan 2012; Voris 2002; Allison 2006
152	*Funisciurus sp.*	African striped squirrel	–	By multiple sources in studies recorded (field, market, intervieuws)	1	Luiselli 2015
153	*Galago senegalensis*	Northern lesser galago	–	Generally reported to occur in mangroves	1	Nowak 2013
154	*Galago zanzibaricus*	Zanzibar galago	–	Generally reported to occur in mangroves	1	Nowak 2013

155	*Galagoides demidovii*	Demidoff's dwarf galago	–	Generally reported to occur in mangroves	1	Nowak 2013
156	*Galictis vittata*	Greater grisón	–	Generally reported to occur in mangroves	1	WWF 2016
157	*Gastropyxis smaragdina*	Ermerald snake	Multiple recorded	Reported 15 times over 355 hours multiple years and field seasons	1	Luiselli 2002
158	*Gavialis gangeticus*	Gangetic gavial	–	Generally reported to occur in mangroves	1	WWF 2016
159	*Genetta cristata*	Crested Genet	Very few recorded	4 individuals reported	1	Blench 2007
160	*Genetta maculata*	Blotched genet	–	By multiple sources in studies recorded (field, market, intervieuws)	1	Luiselli 2015
161	*Geomys personatus*	Texas pocket gopher	–	Generally reported to occur in mangroves	1	WWF 2016
162	*Gerarda prevostiana*	Glossy marsh, Gerards water snake	–	Multiple sources, listed as obligate mangrove user	5	Luther and Greenberg 2009; Voris 2002; Karangkutar 2013; Katherisan 2001; Sandilyan 2012
163	*Gonyosoma oxycephalum*	Red-tailed green rat snake	–	Multiple sources, listed as obligate mangrove user	2	Sandilyan 2012; Luther and Greenberg 2009
164	*Gorilla gorilla gorilla*	Western lowland gorilla	–	Generally reported to occur in mangroves	1	Nowak 2013
165	*Hapsidophrys lineatus*	Black lined green tree snake	Very few recorded	Reported 4 times over 355 hours multiple years and field seasons	1	Luiselli 2002
166	*Helarctos malayanus*	Malayan sunbear	–	Generally reported to occur in mangroves	1	Norma Rashid 2012
167	*Heliosciurus rufobrachium*	Red legged sun squirrel	–	By multiple sources in studies recorded (field, market, intervieuws)	1	Luiselli 2015
168	*Heloderma horridum*	Mexican beaded lizards	–	Generally reported to occur in mangroves	1	WWF 2016
169	*Hemidactylus frenatus*	Common house gecko	–	Generally reported to occur in mangroves	1	Voris 2002
170	*Hemigalus derbyanus*	Banded palm civet	–	Generally reported to occur in mangroves	1	Norma Rashid 2012
171	*Herpailurus yagouaroundi*	Jaguarundi	–	Generally reported to occur in mangroves	1	WWF 2016
172	*Herpestes brachyurus*	Short tailed mongoose	–	Generally reported to occur in mangroves	1	Norma Rashid 2012
173	*Herpestes javanicus*	Small Asian mongoose	–	Generally reported to occur in mangroves	1	Cutter 2015
174	*Hippopotamus amphibius*	Hippopotamus	–	Multiple sources	2	Blench 2007; WWF 2016
175	*Hipposideros spp.*	Round leaf bat	Very few recorded	2 individuals reported	1	Mc Kenzie 1986
176	*Hoplobatrachus crassus*	Jerdon's bulfrog	Multiple recorded	8 individuals reported	1	Jena et al 2013
177	*Hoplobatrachus tigerinus*	Asian bulfrog	Multiple recorded	15 individuals reprorted	1	Jena et al 2013
178	*Hydrelaps darwiniensis*	Black-ringed Seasnake	–	Generally reported to occur in mangroves	1	Hutchlings 1982
179	*Hydrictis maculicollis*	Speckle troated otter	–	Generally reported to occur in mangroves	1	Blench 2007
180	*Hydrochaeris hydrochaeris*	Capybara	–	Generally reported to occur in mangroves	1	WWF 2016
181	*Hydromys chrysogaster*	Water rat	–	Multiple sources	4	Hutchlings 1982; Kutt 1994; Metcalfe 2007; ARKive 2016
182	*Hydrophis clarias*	Many Toothed Sea Snake	–	Listed as uncommon	1	Karangkutar 2013
183	*Hydrophis cyanocinctus*	Annulated Sea Snake	–	Listed as uncommon	1	Karangkutar 2013
184	*Hydrophis elegans*	Elegant sea snake	–	Generally reported to occur in mangroves	1	Hutchlings 1982
185	*Hydrophis mamillaris*	Bombay Sea Snake	–	Listed as rare	1	Karangkutar 2013
186	*Hydrosaurus pustulatus*	Sailfin lizard	–	Listed as common and obligate mangrove user	1	Siler et al 2014
187	*Hyemoschus aquaticus*	Water chevrotain mouse deer	–	Multiple sources	2	Blench 2007; Luiselli 2015
188	*Hylarana erythraea*	Common green frog	–	Generally reported to occur in mangroves	1	Juliana 2011
189	*Hylarana tytleri*	Green frog	Very few recorded	2 individuals reported	1	Jena et al 2013
190	*Hylobates agilis*	Agile gibbon	–	Generally reported to occur in mangroves	1	Nowak 2013
191	*Hylobates albibarbis*	Bornean white bearded gibbon	–	Generally reported to occur in mangroves	1	Nowak 2013
192	*Hylobates klossii*	Kloss's giboon	–	Generally reported to occur in mangroves	1	Nowak 2013
193	*Hylobates muelleri*	Muller's Bornean gibbon	–	Generally reported to occur in mangroves	1	Nowak 2013
194	*Iguana delicatissima*	Antillian iguana	–	Generally reported to occur in mangroves	1	ARKive 2016
195	*Iguana iguana*	Green iguana	–	Generally reported to occur in mangroves	1	WWF 2016

196	*Isoodon macrourus*	Northern brown bandicoot	–	Multiple sources	2	Kutt 1994; Metcalfe 2007
197	*Kaloula pulchra*	Banded bullfrog	–	Generally reported to occur in mangroves	1	Voris 2002
198	*Kaloula taprobanica*	Sri Lankan painted frog	Multiple recorded	15 individuals recorded	1	Jena et al 2013
199	*Laticauda colubrina*	Sea crait	–	Multiple sources	2	Hogarth 2007; ARKive 2016
200	*Lemur catta*	Ring tailed lemur	–	Listed as several times	1	Gardner 2016
201	*Leopardus pardalis*	Ocelot	–	Multiple sources	3	ARKive 2016; WWF 2016; Yaap 2015
202	*Leopardus wiedii*	Little spotted cat	–	Generally reported to occur in mangroves	1	WWF 2016
203	*Lepidodactylus lugubris*	Mourning gecko	–	Generally reported to occur in mangroves	1	Voris 2002
204	*Lepilemur edwarsi*	Milne-Edwards' sportive lemur	–	Generally reported to occur in mangroves	1	Gardner 2016
205	*Lepilemur tymerlachsoni*	Hawks' sportive lemur	Very few recorded	2 reported	1	Gardner 2016
206	*Liasis fuscus*	Brown water python	–	Multiple sources	2	Hutchlings 1982; Kutt 1994
207	*Liasis olivaceus*	Olive phyton	–	Listed as occasionally	1	Hutchlings 1982
208	*Limnodynastes convexiusculus*	Marbeled frog	–	Generally reported to occur in mangroves	1	Hutchlings 1982
209	*Litoria bicolor*	Northern dwarf tree frog	–	Generally reported to occur in mangroves	1	Hutchlings 1982
210	*Lontra longicaudis*	Central American otter	–	Generally reported to occur in mangroves	1	WWF 2016
211	*Lontra longicaudis annectens*	Central American otter	–	Generally reported to occur in mangroves	1	WWF 2016
212	*Lophocebus albigena*	White-cheeked mangabey	–	Listed as reare	1	Nowak 2013
213	*Lophognathus gilberti*	Gilbert's dragon	–	Generally reported to occur in mangroves	1	ARKive 2016
214	*Lophognathus temporalis (Gowic*	Northern water dragon	–	Generally reported to occur in mangroves	1	Hutchlings 1983
215	*Loris tardigradus*	Red slender loris	–	Generally reported to occur in mangroves	1	Nowak 2013
216	*Loxocemus bicolor*	Mexican python	–	Generally reported to occur in mangroves	1	WWF 2016
217	*Lutra maculicollis*	Otter, Spotted neck otter	–	Multiple sources	2	Angelici 2005; WWF 2016
218	*Lutra sumatrana*	Hairy-nosed otter	–	Generally reported to occur in mangroves	1	ARKive 2016
219	*Lutrogale perspicillata*	Smooth coated otter	–	Multiple sources	4	Katherisan 2001; Norma Rashid 2012; Rajpar 2014; ARKive 2016
220	*Macaca fascicularis*	Crab eating, long tailed Macaque	–	Multiple sources	6	Rajpar 2014; Juliana 2011; Norma Rashid 2012; Nowak 2013; Rajpar 2014; Zakaria 2015
221	*Macaca fascicularis umbrosus*	Nicobar long-tailed macaque	–	Generally reported to occur in mangroves	1	Katherisan 2001
222	*Macaca mulatta*	Rhesus macaque	–	Multiple sources	2	Katherisan 2001; Nowak 2013
223	*Macaca nemestrina*	Southern pig-tailed macaque	–	Generally reported to occur in mangroves	1	Nowak 2013
224	*Macaca pagensis*	Pagai Island macaque	–	Generally reported to occur in mangroves	1	Nowak 2013
225	*Macaca siberu*	Siberut macaque	–	Multiple sources	2	Nowak 2013; ARKive 2016
226	*Macaca sinica*	Toque macaque	–	Generally reported to occur in mangroves	1	Nowak 2013
227	*Macroderma gigas*	Ghost bat	–	Multiple sources	2	Mc Kenzie 1986 ; ARKive 2016
228	*Macroglossus minimus*	Long-tongued fruit or blossom bat	–	Multiple sources	6	Hogarth 2007; Kutt 1994; Mc Kenzie 1986 ; Norma Rashid 2012; Rajpar 2014; Metcalfe 2007
229	*Macroglossus minimus lagochilu*	Northern blossom bat	–	Generally reported to occur in mangroves	1	Hutchlings 1982
230	*Macropus agilis*	Agile Wallaby	–	Listed as uncommon	1	Kutt 1994
231	*Macropus robustus*	Euro kangaroo	–	Generally reported to occur in mangroves	1	Reef 2014
232	*Macropus rufus*	Red wallaby	–	Generally reported to occur in mangroves	1	Reef 2014
233	*Mandrillus sphinx*	Mandrill	–	Generally reported to occur in mangroves	1	Nowak 2013
234	*Manin pentadactyla*	Chinese pangolin	–	Generally reported to occur in mangroves	1	Gopal 2006
235	*Marmosa mexicana*	Mexican mouse opossum	–	Multiple sources	2	WWF 2016; Yaap 2015
236	*Mazama spp.*	Deer	–	Generally reported to occur in mangroves	1	WWF 2016

237	*Melomys burtoni*	Grassland Melomys Gras mouse	–	Multiple sources	2	Hutchlings 1982; Metcalfe 2007
238	*Melomys cervinipes*	Fawn footed melomys	–	Generally reported to occur in mangroves	1	Kutt 1994
239	*Mesembriomys gouldii*	Black footed tree rat	Very few recorded	Reported less than 5 times over 2 years	1	Metcalfe 2007
240	*Mesembriomys macrurus*	Golden backed tree rat	–	Generally reported to occur in mangroves	1	IUCN redlist 2014
241	*Mesocapromys angelcabrerai*	Cabrera's hutia	–	Multiple sources, listed as obligate mangrove user	4	Hogarth 2007; Luther and Greenberg 2009; WWF 2016; Sandilyan 2012
242	*Mesocapromys auritus*	Hutia, large eared	–	Generally reported to occur in mangroves	1	WWF 2016
243	*Mesocapromys sanfelipensis*	Hutia	–	Generally reported to occur in mangroves	1	WWF 2016
244	*Michrohyla heymonsi*	Frog	–	Generally reported to occur in mangroves	1	Voris 2002
245	*Micoureus demerarae*	Possum	–	Listed as common	1	Fernandes 2006
246	*Microcebus griseorufus*	Grey brown mouse lemur	–	Generally reported to occur in mangroves	1	Gardner 2016
247	*Microcebus mamiratra*	Claire´s mouse lemur	Very few recorded	2 times several individuals	1	Gardner 2016
248	*Microcebus myoxinus*	Peter’s mouse lemur	–	Generally reported to occur in mangroves	1	Nowak 2013
249	*Microhyla ornata*	Ornate narrow-mouthed frog	Multiple recorded	13 individuals reported	1	Jena et al 2013
250	*Miniopterus australis*	Little bent wing bat	–	Listed as common	1	Kutt 1994
251	*Miniopterus schreibersii*	Common Bent-wing Bat	–	Multiple sources	2	Kutt 1994; Metcalfe 2007
252	*Miopithecus ogouensis*	Gabon talapoin	–	Generally reported to occur in mangroves	1	Nowak 2013
253	*Miopithecus talapoin*	Talapoin monkey	–	Generally reported to occur in mangroves	1	WWF 2016
254	*Mirza zaza*	Northern giant mouse lemur	–	Generally reported to occur in mangroves	1	Gardner 2016
255	*Morelia amethistina*	Indonesian water phyton	–	Multiple sources	2	Hutchlings 1982; Kutt 1994
256	*Morelia spilota*	Carpet python	–	Multiple sources	2	Hutchlings 1982; Kutt 1994
257	*Mormopterus beccarii*	Beccari's Free-tailed bat	Multiple recorded	Reported 6 times over 2 years	1	Metcalfe 2007
258	*Mormopterus loriae*	Little northern freetail bat	–	Multiple sources	3	Hogarth 2007; Kutt 1994; Metcalfe 2007
259	*Mormopterus loriae cobourgene*	Little north-western freetail bat	–	Generally reported to occur in mangroves	1	Luther and Greenberg 2009
260	*Mormopterus norfolkensis*	East-coast free-tailed bat	Multiple recorded	21 individuals reported	1	McConville 2013
261	*Muntiacus muntjak*	Southern red muntjak, barking deer	–	Multiple sources	3	Gopal 2006; Norma Rashid 2012; WWF 2016
262	*Mus musculus*	House mouse	–	Generally reported to occur in mangroves	1	Hutchlings 1982
263	*Myotis adversus*	Large footed myotis	–	Listed as uncommon	1	Kutt 1994
264	*Myotis fortidens*	Cinnamon myotis	–	Generally reported to occur in mangroves	1	WWF 2016
265	*Myotis hasseltii*	Lesser large footed myotis	–	Generally reported to occur in mangroves	1	Norma Rashid 2012
266	*Myotis moluccarum adversus*	Arafura Large-footed myotis	Very few recorded	1 recorded	1	Metcalfe 2007
267	*Myotis volans*	Long legged myotis	–	Generally reported to occur in mangroves	1	WWF 2016
268	*Myrmecophaga tridactyla*	Giant anteater	–	Generally reported to occur in mangroves	1	WWF 2016
269	*Myron richardsonii*	Richardson’s mangrove snake	–	Multiple sources, listed as obligate mangrove user	5	Hutchlings 1982; Allison 2006; Luther and Greenberg 2009; Nagelkerken 2008; Sandilyan 2012
270	*Mysateles garridoi*	Garrido’s hutia	–	Multiple sources, listed as obligate mangrove user	3	Luther and Greenberg 2009; Sandilyan 2012; WWF 2016
271	*Naja melanoleuca*	Forest Cobra	Very few recorded	Reported 2 times over 355 hours multiple years and field seasons	1	Luiselli 2002
272	*Naja nigricollis*	Black-necked spitting cobra	Multiple recorded	reported 13 times over 355 hours multiple years and field seasons	1	Luiselli 2002
273	*Nandinia binotata*	African palm civet	–	By multiple sources in study recorded (field, market, intervieuws)	1	Luiselli 2015
274	*Nasalis larvatus*	Proboscis monkey	–	Multiple sources, listed as obligate mangrove user	7	Hogarth 2007; Luther and Greenberg 2009; Nowak 2013; ARKive 2016; Rajpar 2014; Sandilyan 2012; Bin Kombi 2015

275	*Nasua nasua*	Coati	–	Generally reported to occur in mangroves	2	WWF 2016: Yaap 2015
276	*Neofelis diardi*	Sunda clouded leopard	–	Generally reported to occur in mangroves	1	Nowak 2013
277	*Neofelis nebulosa*	Clouded leopard	–	Multiple sources	2	ARKive 2016; Nowak 2013
278	*Nerodia clarkii compressicauda*	Mangrove water snake	–	Listed as obligate mangrove user	1	Luther and Greenberg 2009
279	*Nerodia fasciata*	Mangrove snake	–	Generally reported to occur in mangroves	1	Hogarth 2007
280	*Noctilio albiventris*	Lesser bulldog bat	–	Generally reported to occur in mangroves	1	WWF 2016
281	*Noctilio leporinus*	Fishing bat	Multiple recorded	61 individuals reported	1	Bordignon 2006
282	*Nycticebus bengalensis*	Bengal slow loris	–	Generally reported to occur in mangroves	1	Nowak 2013
283	*Nycticebus coucang*	Sunda slow loris	–	Generally reported to occur in mangroves	1	Nowak 2013
284	*Nycticebus javanicus*	Javan slow loris	–	Generally reported to occur in mangroves	1	ARKive 2016
285	*Nycticebus menagensis*	Bornean slow loris	–	Generally reported to occur in mangroves	1	Nowak 2013
286	*Nycticeius spp.*	Evening bat	Multiple recorded	46 individuals reported	1	Mc Kenzie 1986
287	*Nyctophilus arnhemensis*	Northern long-eared bat	–	Generally reported to occur in mangroves	1	Hogarth 2007
288	*Odocoileus virginianus*	White-tailed Deer⍰	–	Multiple sources	2	Rajpar 2014; WWF 2016
289	*Odocoileus virginianus clavium*	Key Deer	–	Generally reported to occur in mangroves	1	Rajpar 2014
290	*Ophiophagus hannah*	King cobra	–	Multiple sources,	4	Hogarth 2007; Katherisan 2001; ARKive 2016; Nagelkerken 2008
291	*Oryzomys palustris*	Marsh Rat	–	Generally reported to occur in mangroves	1	Rajpar 2014
292	*Osteopilus pulchrilineatus*	Common tree frog	–	Listed as common	1	Hedges 1992
293	*Osteopilus septentrionalis*	Tree Frog	–	Multiple sources	2	Rajpar 2014; ARKive 2016
294	*Otolemur crassicaudatus*	Thick-tailed greater galago	–	Generally reported to occur in mangroves	1	Nowak 2013
295	*Otolemur garnettii*	Small-eared galago	–	Generally reported to occur in mangroves	1	Nowak 2013
296	*Paguma larvata*	Andaman masked civet	–	Generally reported to occur in mangroves	1	Nowak 2013
297	*Pan troglodytes troglodytes*	Central chimpanzee	–	Generally reported to occur in mangroves	1	Nowak 2013
298	*Pan troglodytes verus*	Western chimpanzee	–	Generally reported to occur in mangroves	1	Nowak 2014
299	*Panthera onca*	Jaguar	–	Generally reported to occur in mangroves	1	WWF 2016
300	*Panthera pardus*	Leopard	–	Multiple sources	3	Norma Rashid 2012; Rajpar 2014; WWF 2016
301	*Panthera pardus delacouri*	Indochinese leopard	–	Generally reported to occur in mangroves	1	Nowak 2013
302	*Panthera pardus kotiya*	Sri Lankan leopard	–	Generally reported to occur in mangroves	1	Nowak 2013
303	*Panthera pardus melas*	Javan leopard	–	Generally reported to occur in mangroves	1	Nowak 2013
304	*Panthera pardus pardus*	Leopard	–	Generally reported to occur in mangroves	1	Nowak 2013
305	*Panthera tigris*	Bengal tiger, Royal tiger	–	Multiple sources, listed as obligate mangrove user	9	Barlow 2011; Hogarth 2007; Katherisan 2001; Norma Rashid 2012; Rajpar 2014; ARKive 2016; WWF 2016; Sandilyan 2012; Naha 2015
306	*Panthera tigris corbetti*	Indochinese tiger	–	Generally reported to occur in mangroves	1	Nowak 2013
307	*Panthera tigris jacksoni*	Malayan tiger	–	Generally reported to occur in mangroves	1	Nowak 2013
308	*Panthera tigris sumatranus*	Sumatran tiger	–	Generally reported to occur in mangroves	1	Nowak 2013
309	*Panthera tigris tigris*	Bengal tiger, Royal Bengal tiger	–	Multiple sources	2	Gopal 2006; Nowak 2013
310	*Papio cynocephalus*	Yellow baboon	–	Generally reported to occur in mangroves	1	Nowak 2013
311	*Papio papio*	Guinea baboon	–	Generally reported to occur in mangroves	1	Nowak 2013
312	*Pardofelis badia*	Bay cat	–	Generally reported to occur in mangroves	1	Nowak 2013
313	*Pardofelis marmorata*	Marbled cat	–	Generally reported to occur in mangroves	1	Nowak 2013
314	*Pardofelis temminckii*	Asiatic golden cat	–	Generally reported to occur in mangroves	1	Nowak 2013
315	*Pecari tajacu*	Collared peccarie	–	Generally reported to occur in mangroves	1	WWF 2016

316	*Pelamus platurus*	Sea snake	Very few recorded	1 record museum specimen	1	Kutt 1994
317	*Perameles spp.*	Bandicoot	–	Multiple sources	2	Hogarth 2007; Hutchlings 1982
318	*Perodicticus potto*	Bosman's potto	–	Generally reported to occur in mangroves	1	Nowak 2013
319	*Peromyscus yucatanicus*	Yucatan deer mouse	–	Generally reported to occur in mangroves	1	WWF 2016
320	*Phacochoerus africanus*	Warthog	–	Generally reported to occur in mangroves	1	Nowak 2013
321	*Phataginus tetradactyla*	Black bellied pangolin	–	Generally reported to occur in mangroves	1	Blench 2007
322	*Philantomba walteri*	Walter´s duiker	–	By multiple sources in studies recorded (field, market, intervieuws)	1	Luiselli 2015
323	*Philothamnus nitidus*	Green bushadder	Very few recorded	Reported 2 times over 355 hours multiple years and field seasons	1	Luiselli 2002
324	*Pipa pipa*	Pipa frog	–	Generally reported to occur in mangroves	1	WWF 2016
325	*Pipistrellus tenuis*	Least pipistrelle	–	Generally reported to occur in mangroves	1	Hogarth 2007
326	*Pipistrellus vordermanni*	Vordermann's pipistrelle	–	Listed as obligate mangrove user	1	Luther and Greenberg 2009
327	*Pipistrellus westralis*	Northern Pipistrelle	–	Multiple sources, listed as obligate mangrove user	2	Luther and Greenberg 2009; Metcalfe 2007
328	*Pithecia pithecia*	Guianan saki	–	Generally reported to occur in mangroves	1	WWF 2016
329	*Planigale maculata*	Common planigale	–	Generally reported to occur in mangroves	1	ARKive 2016
330	*Polypedates leucomystax*	Common tree frog	–	Generally reported to occur in mangroves	1	Voris 2002
331	*Polypedates maculatus*	Indian tree frog	Multiple recorded	23 individuals reported	1	Jena et al 2013
332	*Pongo abelii*	Sumatran orangutan	–	Generally reported to occur in mangroves	1	Nowak 2013
333	*Pongo pygmaeus*	Bornean orangutan	–	Generally reported to occur in mangroves	1	Nowak 2013
334	*Potos flavus*	Kinkajou	–	Generally reported to occur in mangroves	1	WWF 2016
335	*Presbytis chrysomelas*	Red banded langur	–	Generally reported to occur in mangroves	1	Nowak 2013
336	*Presbytis femoralis*	Banded leaf monkey	–	Generally reported to occur in mangroves	1	Nowak 2013
337	*Presbytis hosei*	Hose's langur	–	Generally reported to occur in mangroves	1	Nowak 2013
338	*Presbytis melalophos*	Sumatran Surili	–	Generally reported to occur in mangroves	1	Nowak 2013
339	*Presbytis potenziani*	Mentawai langrur, joja	–	Generally reported to occur in mangroves	1	Nowak 2013
340	*Presbytis rubicunda*	Maroon leaf monkey	–	Generally reported to occur in mangroves	1	Nowak 2013
341	*Presbytis thomasi*	Thomas's langur	–	Generally reported to occur in mangroves	1	Nowak 2013
342	*Prionailurus bengalensis*	Leopard cat	–	Multiple sources	3	Gopal 2006; Nowak 2013; Norma Rashid 2012
343	*Prionailurus bengalensis iriomote*	Iriomote cat	–	Generally reported to occur in mangroves	1	Nowak 2013
344	*Prionailurus planiceps*	Flat-headed cat	–	Generally reported to occur in mangroves	1	Nowak 2013
345	*Prionailurus rubiginosus*	Rusty-spotted cat	–	Generally reported to occur in mangroves	1	Nowak 2013
346	*Prionailurus viverrinus*	Fishing cat	–	Multiple sources	5	Gopal 2006; Hogarth 2007; ARKive 2016; Nowak 2013; Cutter 2015
347	*Procolobus badius*	Senegal red colobus	–	Generally reported to occur in mangroves	1	Nowak 2013
348	*Procolobus kirkii*	Zanzibar red colobus	–	Generally reported to occur in mangroves	1	Nowak 2013
349	*Procolobus pennantii epieni*	Niger delta red colobus	–	Generally reported to occur in mangroves	1	Nowak 2013
350	*Procolobus verus*	Olive colobus	–	Generally reported to occur in mangroves	1	Nowak 2013
351	*Procyon cancrivorus*	Crab eating raccoon	–	Multiple sources	2	Hogarth 2007; WWF 2016
352	*Procyon lotor*	Northern raccoon	–	Multiple sources	3	Cuaron 2004; WWF 2016; Yaap 2015
353	*Procyon pygmaeus*	Pygmy raccoon	–	Generally reported to occur in mangroves	1	ARKive 2016
354	*Propithecus coquereli*	Coquerel's sifaka	–	Multiple sources	3	Nowak 2013; ARKive 2016; Gardner 2015
355	*Propithecus coronatus*	Crowned sifaka	–	Multiple sources	3	Nowak 2013; ARKive 2016; Gardner 2015
356	*Propithecus deckenii*	Von der Decken's sifaka	–	Generally reported to occur in mangroves	1	Gardner 2016
357	*Protoxerus stangeri*	African giant squirrel	–	By multiple sources in study recorded (field, market, intervieuws)	1	Luiselli 2015

358	*Psammophis phillipsi*	Sand snake	Multiple recorded	Reported 32 times over 355 hours multiple years and field seasons	1	Luiselli 2002
359	*Pseudis paradoxa*	Paradoxal frog	–	Generally reported to occur in mangroves	1	WWF 2016
360	*Pseudohaje goldii*	Gold's Tree Cobra	Multiple recorded	Reported 10 times over 355 hours multiple years and field seasons	1	Luiselli 2002
361	*Pseudomys fieldi*	Shark Bay mouse	–	Generally reported to occur in mangroves	1	ARKive 2016
362	*Pteronura brasiliensis*	Giant river otter	–	Generally reported to occur in mangroves	1	WWF 2016
363	*Pteropus aldabrensis*	Aldabra flying fox	–	Generally reported to occur in mangroves	1	ARKive 2016
364	*Pteropus alecto*	Flying fox , Black flying fox	–	Multiple sources	6	Hutchlings 1982; Katherisan 2001; Kutt 1994; Mc Kenzie 1986; Metcalfe 2007; ARKive 2016
365	*Pteropus anetianus*	Vanuatu flying fox	–	Generally reported to occur in mangroves	1	ARKive 2016
366	*Pteropus chrysoproctus*	Moluccan flying fox	Multiple recorded	300 individuals reported	1	Tsang 2015
367	*Pteropus conspicillatus*	Spectacled flying fox	–	Multiple sources	2	Hutchlings 1982; Katherisan 2001
368	*Pteropus faunulus*	Nicobar flying fox	–	Generally reported to occur in mangroves	1	ARKive 2016
369	*Pteropus melanopogon*	Black-bearded flying fox	Multiple recorded	200 individuals reported	1	Tsang 2015
370	*Pteropus melanotus*	Blyth's flying fox	–	Generally reported to occur in mangroves	1	ARKive 2016
371	*Pteropus ocularis*	Seram flying fox	Very few recorded	Reported 1 time	1	Tsang 2015
372	*Pteropus poliocephalus*	Grey headed flying fox	–	Multiple sources	3	Hutchlings 1982; Kutt 1994; ARKive 2016
373	*Pteropus rodricensis*	Rodrigues flying fox	–	Generally reported to occur in mangroves	1	ARKive 2016
374	*Pteropus rufus*	Madagascan flying fox	–	Generally reported to occur in mangroves	1	ARKive 2016
375	*Pteropus scapulatus*	Little red flying fox	–	Multiple sources,	3	Kutt 1994; Mc Kenzie 1986; ARKive 2016
376	*Pteropus temminckii*	Temminck's flying fox	Very few recorded	Reported 2 times unexpected	1	Tsang 2015
377	*Pteropus vampyrus*	Large flying fox	–	Generally reported to occur in mangroves	1	ARKive 2016
378	*Pteropus voeltzkowi*	Pemba flying fox	–	Generally reported to occur in mangroves	1	ARKive 2016
379	*Ptyas mucosa*	Oriental snake	–	Recported as uncommon	1	Karangkutar 2013
380	*Puma concolor*	Puma	–	Generally reported to occur in mangroves	1	WWF 2016
381	*Python bivittatus*	Indian Rock Python	–	Multiple sources	4	Gopal 2006; Katherisan 2001; Nagelkerken 2008; Karangkutar 2013
382	*Python regius*	Ball pythons	Multiple recorded	Reported 7 times over 355 hours multiple years and field seasons	1	Luiselli 2002
383	*Python sebae*	African Rock python	–	Multiple sources	3	Nagelkerken 2008; ARKive 2016, Luiselli 2002
384	*Rana cancrivora*	Crab eating frog	–	Multiple sources	3	Hogarth 2007; Norma Rashid 2012; Voris 2002
385	*Rattus colletti*	Dusky rat	–	Generally reported to occur in mangroves	1	Hutchlings 1982
386	*Rattus sordidus*	Canefield rat	–	Generally reported to occur in mangroves	1	Kutt 1994
387	*Rattus tunneyi*	Pale field rat	Very few recorded	Reported less than 5 over 2 years	1	Metcalfe 2007
388	*Rhacophorus leucomystax*	Flying foam nest frogs	–	Generally reported to occur in mangroves	1	Juliana 2011
389	*Rhamnophis aethiopissa*	Large eyed green tree snake	–	Reported 6 times over 355 hours multiple years and field seasons	1	Luiselli 2002
390	*Rhinoceros sondaicus*	Javan rhinoceros	–	Multiple sources	2	Hogarth 2007; Katherisan 2001
391	*Rhinolophus trifoliatus*	Trefoil horseshoe bat	–	Generally reported to occur in mangroves	1	ARKive 2016
392	*Rucervus duvaucelii*	Swamp deer	–	Multiple sources	2	Katherisan 2001; ARKive 2016
393	*Rusa timorensis*	Javan deer	Multiple recorded	15 individuals reported	1	Santosa 2015
394	*Rusa unicolor*	Sambar	–	Multiple sources	2	Norma Rashid 2012; WWF 2016
395	*Saccolaimus flaviventris*	Yellow-bellied Sheath-tailed bat	–	Generally reported to occur in mangroves	1	WWF 2016
396	*Saccopteryx bilineata*	Sac-winged bat	Very few recorded	Reported 1 time over 2 years	1	Metcalfe 2007
397	*Saimiri oerstedii*	Red-backed squirrel monkey	–	Generally reported to occur in mangroves	1	ARKive 2016
398	*Saimiri sciureus*	Squirrel monkeys	–	Generally reported to occur in mangroves	1	WWF 2016
399	*Scalopus aquaticus*	Nutria	–	Generally reported to occur in mangroves	1	WWF 2016

400	*Scincella spp.*	Skink	–	Generally reported to occur in mangroves	1	WWF 2016
401	*Sciuridae spp.*	Squirrel	–	Generally reported to occur in mangroves	1	Juliana 2011
402	*Scotorepens greyii*	Little broad nosed bat	–	Multiple sources	2	Hogarth 2007; Metcalfe 2007
403	*Sigmodon hispidus*	Cotton Rat	–	Generally reported to occur in mangroves	1	Rajpar 2014
404	*Simias concolor*	Pig tailed langur	–	Generally reported to occur in mangroves	1	Nowak 2013
405	*Sorex saussurei*	Saussure's shrew	–	Generally reported to occur in mangroves	1	WWF 2016
406	*Sphaerotheca rolandae*	Marble sandfrog	Very few recorded	Reported 4 times	1	Jena et al 2013
407	*Sus barbatus*	Bearded pig	–	Generally reported to occur in mangroves	1	ARKive 2016
408	*Sus scrofa*	Wild Pig	–	Multiple sources	5	Juliana 2011; Norma Rashid 2012; WWF 2016; Rajpar 2014; Zakaria 2015
409	*Syconycteris australis*	Common blossom bat	–	Generally reported to occur in mangroves	1	Kutt 1994
410	*Sylvilagus floridanus*	Cottontail rabbit	–	Generally reported to occur in mangroves	1	WWF 2016
411	*Sylvilagus palustris*	Marsh rabbit	–	Generally reported to occur in mangroves	1	Rajpar 2014
412	*Sylvilagus palustris hefneri*	Marsh rabiit	–	Generally reported to occur in mangroves	1	Katherisan 2001
413	*Symphalangus syndactylus*	Siamang	–	Generally reported to occur in mangroves	1	WWF 2016
414	*Tadarida jobensis (Chaerephon jobensis)*	Northern Free-tailed bat	–	Multiple sources	2	Hogarth 2007; Metcalfe 2007
415	*Tadarida planiceps (Mormopterus planiceps)*	Flat headed mastiff bat	–	Listed as ocassinally	1	Hutchlings 1982
416	*Tamandua mexicana*	Mexican anteater, Northern tamandua	–	Generally reported to occur in mangroves	1	WWF 2016
417	*Taphozous flaviventris*	Yellow-bellied sheath-tailed bat	–	Generally reported to occur in mangroves	1	Hogarth 2007
418	*Taphozous georgianus*	Common sheath-tailed bat	–	Generally reported to occur in mangroves	1	Hogarth 2007
419	*Tapirus bairdii*	Baird's Tapir	–	Generally reported to occur in mangroves	1	WWF 2016
420	*Tapirus indicus*	Malayan tapir	–	Multiple sources	2	Norma Rashid 2012; WWF 2016
421	*Tapirus terrestris*	South American tapir	–	Generally reported to occur in mangroves	1	WWF 2016
422	*Tardaria loriae (Mormopterus loriae)*	Little Northern mastiff bat	–	Listed as occasionally	1	Hutchlings 1982
423	*Tarsius bancanus*	Horsfield's tarsier	–	Generally reported to occur in mangroves	1	Nowak 2013
424	*Tarsius syrichta*	Philipine tarsier	–	Generally reported to occur in mangroves	1	ARKive 2016
425	*Tarsius tarsier*	Spectral tarsier	–	Generally reported to occur in mangroves	1	ARKive 2016
426	*Tayassu tajacu*	Collared peccarie	–	Multiple sources	2	WWF 2016; Yaap 2015
427	*Thelotornis kirtlandii*	Twig snake	Multiple recorded	Reported 30 times over 355 hours multiple years and field seasons	1	Luiselli 2002
428	*Thrasops flavigularis*	Yellow-throated bold-eyed ree snake	Multiple recorded	Reported 8 times over 355 hours multiple years and field seasons	1	Luiselli 2002
429	*Thryonomys swinderianus*	Greater cane rat	–	By multiple sources in studies recorded (field, market, intervieuws)	1	Luiselli 2015
430	*Tomistoma schlegelii*	False gavial	–	Generally reported to occur in mangroves	1	WWF 2016
431	*Toxicodryas blandingii*	Blandings tree snake	Multiple recorded	Reported 13 times over 355 hours multiple years and field seasons	1	Luiselli 2002
432	*Trachypithecus auratus*	Javan lutung	–	Generally reported to occur in mangroves	1	Nowak 2013
433	*Trachypithecus cristatus*	Silver leaf monkey	–	Multiple sources	3	Norma Rashid 2012; Nowak 2013; ARKive 2016
434	*Trachypithecus obscurus*	Dusky leaf monkey	–	Generally reported to occur in mangroves	1	Zakaria 2015
435	*Trachypithecus vetulus*	Purple faced langur	–	Generally reported to occur in mangroves	1	Nowak 2013
436	*Tragelaphus scriptus*	Bushbuck	–	Multiple sources	2	Nowak 2013; Luiselli 2015
437	*Tragelaphus spekii*	Sitatunga	–	Multiple sources	2	Blench 2007; Luiselli 2015
438	*Tragulus javanicus*	Javan chevrotain	–	Multiple sources	2	Norma Rashid 2012; WWF 2016
439	*Trichosurus arnhemensis*	Northern bursh tailed possum	–	Generally reported to occur in mangroves	1	Hutchlings 1982
440	*Trichosurus vulpecula*	Common brushtail possum	–	Multiple sources	2	Kutt 1994; Melcalfe 2007

441	*Trimeresurus purpureomaculatus*	Mangrove pit viper	–	Multiple sources, listed as obligate mangrove user	4	Luther and Greenberg 2010; Norma Rashid 2012; Voris 2002; Rajpar 2014
442	*Trimeresurus purpureomaculatus andersoni*	Mangrove pit viper	–	Listed as obligate mangrove user	1	Luther and Greenberg 2009
443	*Trimeresurus wegleri*	Pit viper	–	Generally reported to occur in mangroves	1	Norma Rashid 2012
444	*Tropidonophis mairii*	Keelback snake	Very few recorded	1 record museum specimen	1	Kutt 1994
445	*Uromys caudimaculatus*	White tailed rat	–	Listed as uncommon	1	Kutt 1994
446	*Vampyrum spectrum*	False vampire bat	–	Generally reported to occur in mangroves	1	WWF 2016
447	*Varanis rainerguentheri*	Monitor Lizard	–	Generally reported to occur in mangroves	1	Weijola 2010
448	*Varanus bengalensis*	Common indian monitor, monitor lizard	–	Multiple sources	2	Gopal 2006; Karangkutar 2013
449	*Varanus cumingi*	Water monitor	–	Listed as obligate mangrove user	1	Luther and Greenberg 2009
450	*Varanus flavescens*	Yellow monitor lizard	–	Generally reported to occur in mangroves	1	Katherisan 2001
451	*Varanus gouldii*	Sand monitor	–	Generally reported to occur in mangroves	1	Kutt 1994
452	*Varanus indicus*	Mangrove monitor	–	Multiple sources	7	Hogarth 2007; Hutchlings 1982; Luther and Greenberg 2009; Rajpar 2014; Sandilyan 2012; Allison 2006; Zakaria 2015
453	*Varanus marmoratus*	Water monitor	–	Listed as obligate mangrove user	1	Luther and Greenberg 2009
454	*Varanus niloticus*	Nile monitor	–	Generally reported to occur in mangroves	1	ARKive 2016
455	*Varanus panoptes*	Argus monitor lizard	–	Listed as occasionally	1	Blamires 2004
456	*Varanus prasinus*	Emerald tree monitor lizard	–	Listed as common	1	Hutchlings 1982
457	*Varanus salvator*	Asian water monitor lizard	–	Multiple sources	10	Gopal 2006; Katherisan 2001; Luther and Greenberg 2009; Norma Rashid 2012; Rajpar 2014; Voris 2002; Weijola 2010; WWF 2016; Sandilyan 2012; Zakaria 2015
458	*Varanus semiremex*	Rusty monitor	–	Multiple sources	2	Hutchlings 1982; Nagelkerken 2008
459	*Varanus timorensis*	Timor monitor lizard	–	Listed as common	1	Hutchlings 1982
460	*Varanus tristis*	Black headed monitor lizard	–	Listed as common	1	Hutchlings 1982
461	*Varanus varius*	Lace monitor	–	Generally reported to occur in mangroves	1	Kutt 1994
462	*Varanus yuwonoi*	Monitor Lizard	–	Generally reported to occur in mangroves	1	Weijola 2010
463	*Wallabia bicolor*	Swamp Wallaby	–	Generally reported to occur in mangroves	1	Kutt 1994
464	*Xeromys myoides*	Water rat, False water rat	–	Multiple sources, listed as obligate mangrove user	4	Hogarth 2007; Hutchlings 1982; Luther and Greenberg 2009; Kaluza 2015
*	Note: Many sources do not provide the origin of their report, which means that some records may not be independent					

Supplementary table S1a - References table 1

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Supplementary material Chapter three: A novel survey approach to detecting terrestrial vertebrates in flooded forests

Supplementary table S1. Assessment and scoring of techniques

I compiled a list of commonly used faunal survey techniques (including both established and emerging methods) that collectively target the complete terrestrial faunal assemblage, and could be executed within a maximum timeframe of 7 days. Targeted literature searches were used to identify less commonly used techniques that could replace commonly used but unsuitable approaches. All techniques were assessed for their ability to be adapted to avoid inundation based on literature reviews and discussion with experts; any that failed or were deemed not to be realistically modifiable within a rapid timeframe were excluded prior to scoring.

Technique	Criterea 1. Taxa groups detected	Criterea 2. Overcome inundation challenges		Scoring related to possible execution within a maximum timeframe of 7 days and feasability in relation to modification											Score	
	Species known to be detected with technique based on publications	Personal assesment of the possibility to mitigate inundation	Aleration required in areas with crocodiles based on Queensland crocodile expert advice	Preparation required	Effort required to set up in the field (attach to trees, set on ground, flag area)	Days before survey to determine tide	Daily checking	Checking of traps in tidal area	Additional checking effort in field	Species identification processing time	Species identification experts or reference library available	People needed for set up and checking	Weight and transport, per 10 traps	Bulkiness	Above 12 excluded	Additional argumentation behind eliminating or including techniques not based on score.
				0 =none, 1=bait 2= bait and tape 3= modifications to a conventional trap design	0 = just place 1= place above tidal area on tree 2=placement required more effort	0, 1, 2	0=no 1=yes	0=not in tidal area 1 = in tidal area	0= no cleaning 1= cleaning after each animal detected	1=direct species ID/pictures 2=send to lab for analyses	0=yes 1=no	1 = 1 or 2 =2	0= none 1=less than 1kg, 2=less than 10kg, 3 is less than 15kg 4 = less than 20kg 5= more than 30kg	Traps per 10 0=compact 1=not fit in 30l backpack		
Live traps in trees	Small ground dwelling mammals. De Bondi 2010, Tasker 2010	Placed above high tide line based on tidal predictions within survey period. Flag the high tide for 2 consequtive days before setting up and follow weather reports.	Make sure traps are placed at least 5 m above the highest tide line.	1	0	2	1	0	1	1	0	2	4	0	12	
Floating live traps	Small ground dwelling mammals, Desa 2012			3	2	0	1	1	1	1	0	2	4	0	15	
Live traps in trees	Small arboreal mammals, Malcolm 1991, Starr, 2012			1	2	2	1	1	1	1	0	2	4	0	15	
Cage traps floating	Small - medium ground dwelling mammals, Baker 1998			1	0	2	1	0	1	1	0	2	5	1	14	
Cage traps on ground	Small - medium ground dwelling mammals, De Bondi 2010			1	0	2	1	0	1	1	0	2	5	1	14	

Pitfall traps	Small ground dwelling reptiles and amphibians, Enge 2001	Flag the high tide for 2 consequtive days before setting up and follow weather reports. Placed above high tide line based on tidal predictions within survey period. Keep bucket in ground with 3* 50cm wooden stakes diagonally hammered into the ground (one edge cut out to fit over edge of bucket).	Make sure traps are placed at least 5 m above the highest tide line.	3	3	2	1	0	1	1	0	2	2	1	16	Initially included and tested because of reported high trap succes rate for ground dwelling reptiles
Pitfall traps	Small mammals, Umetsu 2006			3	3	2	1	0	1	1	0	2	2	1	16	
Hair traps	Small arboreal mammals, Pocock 2006	Placed if possible on chest height in trees. If no suitable branches at this height, place lower but flag high tide line on trees 1 day before placement.	Determine area within low tide window (the area that is dry 2h before and after low tide) to allow for set up and checking of traps.	2	2	1	0	1	0	2	0	1	2	1	12	
Print plates in trees	Arboreal mammals, Starr, 2012			3	2	1	1	1	1	2	1	1	2	0	15	
Print plates on ground	Mammals, Desa 2012			3	1	1	1	0	1	2	1	1	1	0	12	Excluded because of reported issues with mositure and identification by Desa 2012
Terrestrial artificial refuges	Small ground dwelling reptiles and amphibians, Michael 2012, Hampton 2007, Lettink 2007	Placed above high tide line based on tidal predictions within survey period. Flag the highest tide day before setting up and follow weather reports.	Make sure traps are placed at least 5 m above the highest tide line.	0	0	1	1	0	0	1	0	2	3	1	9	
Arboreal artificial refuges	Small arboreal reptiles and amphibians, Bell 2009, Thomlinson 2012	Placed if possible on chest height in trees. If no suitable branches at this height, place lower but flag high tide line on trees 1 day before placement.	Determine area within low tide window (the area that is dry 2h before and after low tide) to allow for set up and checking of traps.	0	2	1	1	1	0	1	0	2	1	0	9	
Arboreal funnel traps	Snakes, arboreal reptiles, Dorcas 2009			1	1	1	1	1	1	1	0	2	3	1	13	
Camera traps on ground	Medium to large ground dwelling mammals and reptiles, Espartosa 2011, Meek 2015, Paull 2012, Paull 2011, Diete 2016, Welbourne 2013.	Measure high tide on tree chosen to attach camera on day before placement.	Make sure traps are placed at least 5 m above the highest tide line.	1	1	1	0		0	1	0	1	2	0	7	

Camera traps in trees	Medium to largear boreal mammals and reptiles, Di Cerbo 2013, Gregory 2014.			1	2	1	0	1	0	1	0	1	2	0	9	Excluded because of budget
Bat detector	Bats, Milne 2004, Flaquer 2007	Place on shoulder height on tree.	Make sure traps are placed at least 5 m above the highest tide line.	0	1	0	0	0	0	2	0	1	1	0	5	
Harp traps	Bats, Duffy 2000	Need to be placed in open fly way.		1	2	1	1	1	1	2	0	2	3	1	15	
Night transects, night time visual encounter surveys, line transects with thermal imaging camera and spotlighting	All sizes ground dwelling and arboreal mammals, reptiles and amphibians, Focardi 2001, Goldinggay 2004	Explore and flag transect lines on highest tide the day before walking the night transect, to make sure transect does not traverse through areas with water higher than knee height. It can occur that the night transects are always walked at low tide- in this case the exploration the day before does not have to occur at the highest tide.	Determine area within low tide window (the area that is dry 2h before and after low tide to allow for walking of transect). This can mean that night transects commence at other hours than right after last light (e.g 2am)	0	1	1	1	1	0	1	0	2	0	0	7	
Day transects, visual encounter surveys, line transect	All sizes ground dwelling and arboreal mammals, reptiles and amphibians, Doan 2003, Rodel 2004, Silveira 2003	Same as night transects.	Same as night transects.	0	1	1	1	1	0	1	0	2	0	0	7	Excluded because of time required to check other traps in the approach interfering with sleep (night transects)

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Supplementary table S2. Families detected with each technique

○ Detected by 1 technique

● Detected by multiple techniques

Family	Number of species	Body size	Strata	Incidental (Arboreal and ground)	Bat detect (Flying)	Live traps (Ground)	Night transects (Arboreal and ground)	Camera trap (Ground)	Hair tubes (Arboreal)	Terestrial refuges (Ground)	Arboreal refuges (Arboreal)
Amphibians											
Frog	1	small	arboreal				○				
Frog	2	small	arboreal	○							
Frog	3	small	ground	●			●				
Frog	4	small to medium	ground	●		●	●				
Reptiles											
Crocodilians	1	large	ground				○				
Lizards	1	small	ground	●						●	
Lizards	2	small	grounf	●						●	
Lizards	3	small	ground	●						●	
Lizards	4	small	arboreal	●							●
Lizards	5	small	arboreal	○							
Lizards	6	small	arboreal and ground				●				●
Lizards	7	small	arboreal				●				●
Lizards	8	small	arboreal				○				
Lizards	9	small	ground	●						●	
Lizards	10	small	ground	●						●	
Lizards	11	small	arboreal	●			●				
Lizards	12	small	ground	●						●	
Lizards	13	small	ground	●						●	
Lizards	14	small	ground	○							
Lizards	15	medium	arboreal and ground	●				●			
Snakes	1	medium	arboreal and ground	○							
Snakes	2	medium	arboreal	○							
Snakes	3	small	ground	○							
Mammals											
Bats	1	small	flying		○						
Bats	2	small	flying		○						
Bats	3	small	flying		○						
Bats	4	small	flying		○						
Bats	5	small	flying		○						
Bats	6	small	flying		○						
Bats	7	small	flying		○						
Bats	8	small	flying		○						
Bats	9	small	flying		○						
Bats	10	medium	flying				○				
Bats	11	small	flying		○						
Bats	12	small	flying		○						
Bats	13	small	flying		○						
Canines	1	large	ground	●				●			
Canines	2	large	ground	○							
Canines	3	large	ground	●				●			
Felids	1	medium	arboreal and ground	○							
Lagomorphs	1	medium	ground	●				●			
Marsupials	1	small	ground			○					
Marsupials	2	large	arboreal	○							
Marsupials	3	large	ground	●				●			
Marsupials	4	large	ground	●				●			
Marsupials	5	medium	arboreal				○				
Marsupials	6	medium	arboreal				○				
Marsupials	7	small	ground						○		
Marsupials	8	medium	arboreal				○				
Marsupials	9	medium	arboreal					●	●		
Marsupials	10	large	ground	●				●			
Rodents	1	medium	ground				●		●		
Rodents	2	small	arboreal and ground	●		●	●		●		
Rodents	3	small	arboreal and ground	●		●	●		●		
Rodents	4	small	arboreal and ground			○					
Rodents	5	small	arboreal and ground				○				
Rodents	6	small	ground			●			●		
Rodents	7	small	ground			●		●	●		
Rodents	8	small	arboreal and ground			●			●		
Rodents	9	medium	arboreal and ground				●	●			
Rodents	10	small	ground				○				
Ungulates	1	large	ground	○							
Ungulates	2	large	ground	○							
Ungulates	3	large	ground	●				●			
Ungulates	4	large	ground	○							

Supplementary table S3. Species detected per survey night per technique per region

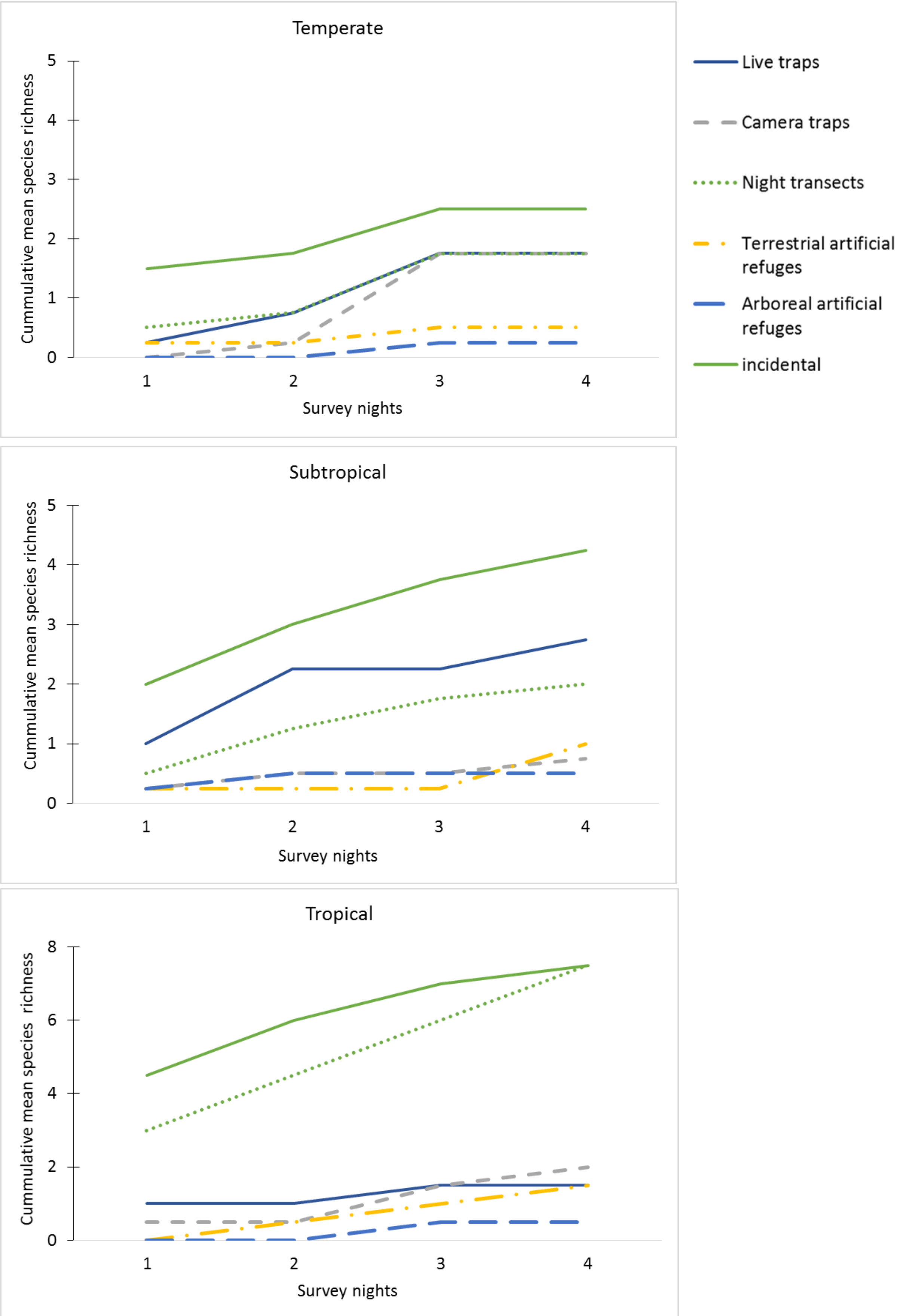
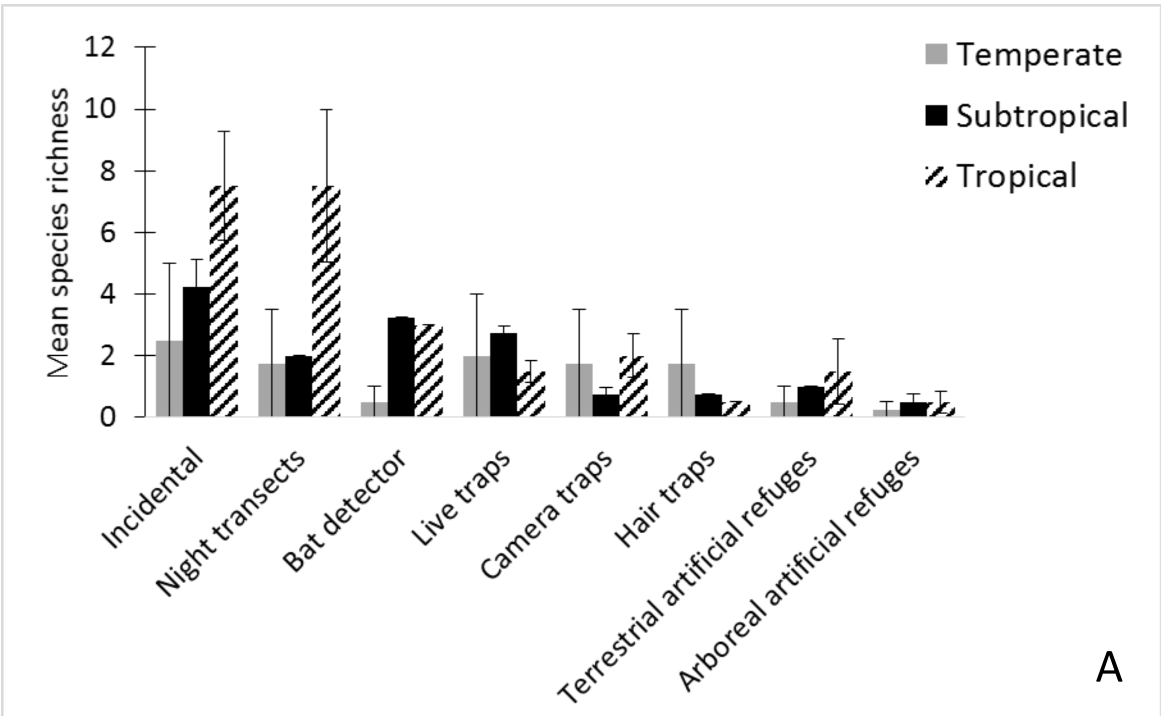


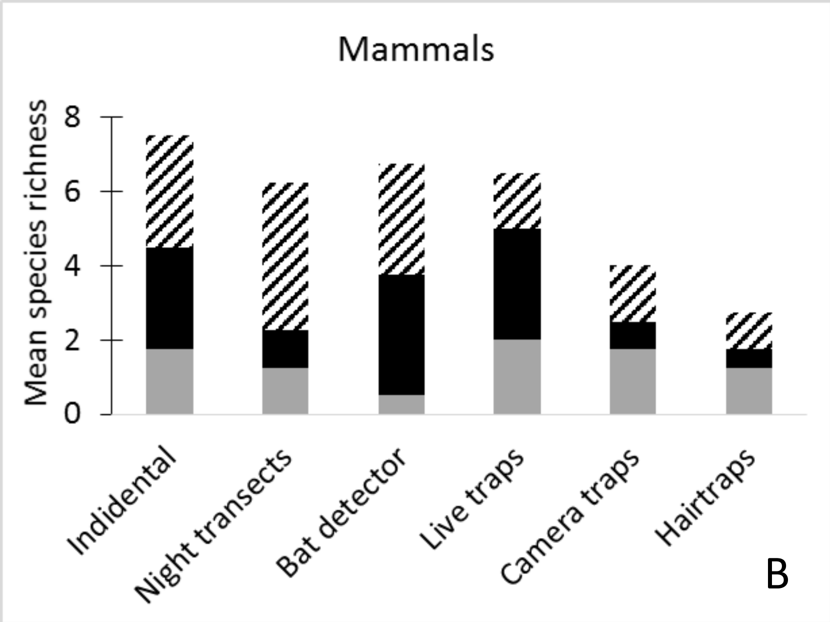
Figure S3.1. The mean species richness detected cumulative across each survey night per technique over the survey duration nights per region. Survey nights commence at 0 to account for incidental sightings taking place 1-2 days prior to the trapping events. Species detected by the hair tubes and bat detector were excluded from B because hair tubes were only checked at the end of the four nights and calls were of inconsistent quality across the four nights.

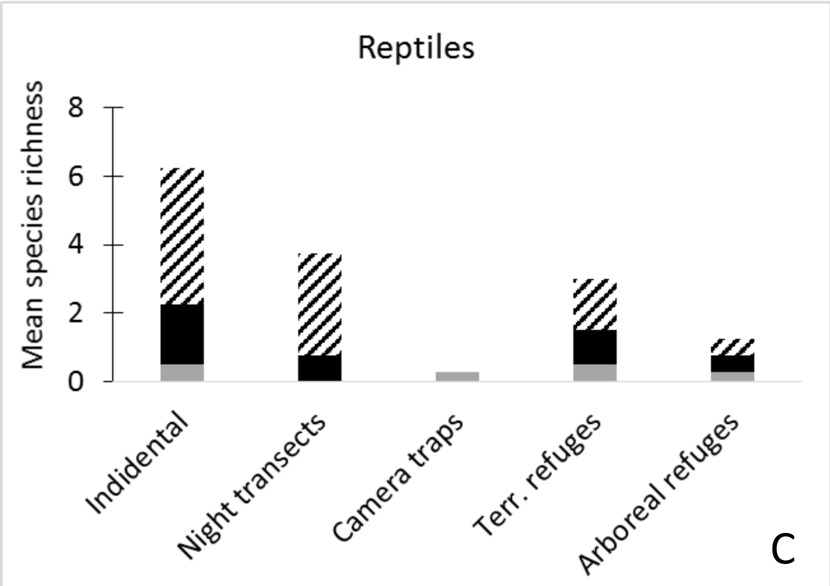
Supplementary table S4. Species richness detected per region

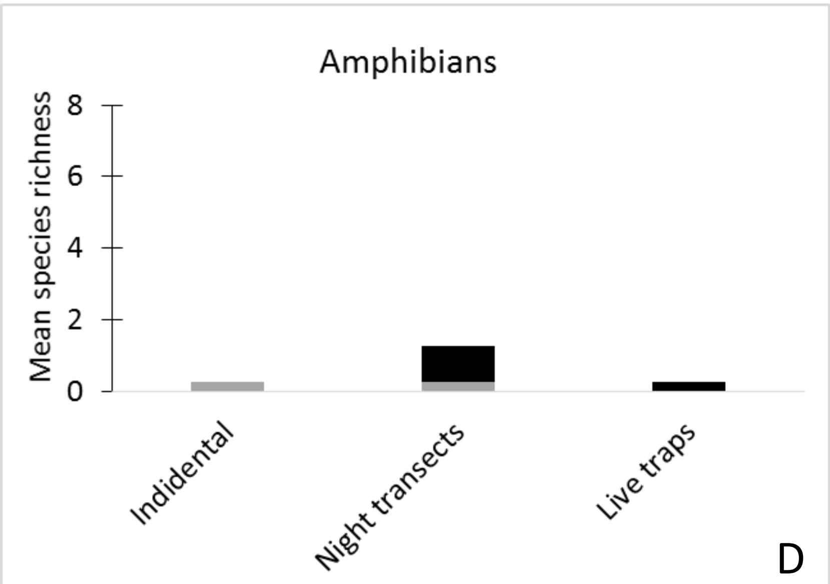
Methods

Effectiveness was judged relative to the return on investment for each technique, pooling results per region. The inputs were: 1) the average number of species detected by each technique across different regions (average over 4 sites for temperate and subtropical regions and over 2 sites for the tropical region); (2) the number of unique species detected by the technique (i.e. those species that were not detected by any other technique, hereafter referred to as unique species); and (3) trap success per region, calculated as the average number of detections by a technique over the survey period per region. The hair traps and bat detector were not included in trap success calculations, as the number of individuals detected could not be determined.




Mammals
Mean species richness
0
2
4
6
8
Indidental
Night transects
Bat detector
Live traps
Camera traps
Hairtraps
B


Reptiles
Mean species richness
0
2
4
6
8
Indidental
Night transects
Camera traps
Terr. refuges
Arboreal refuges
C


Amphibians
Mean species richness
0
2
4
6
8
Indidental
Night transects
Live traps
D

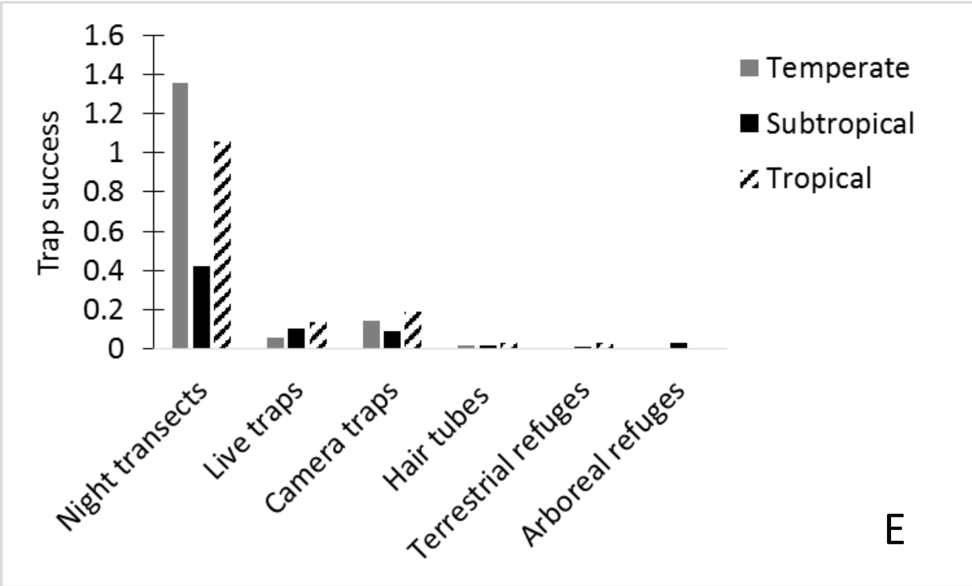
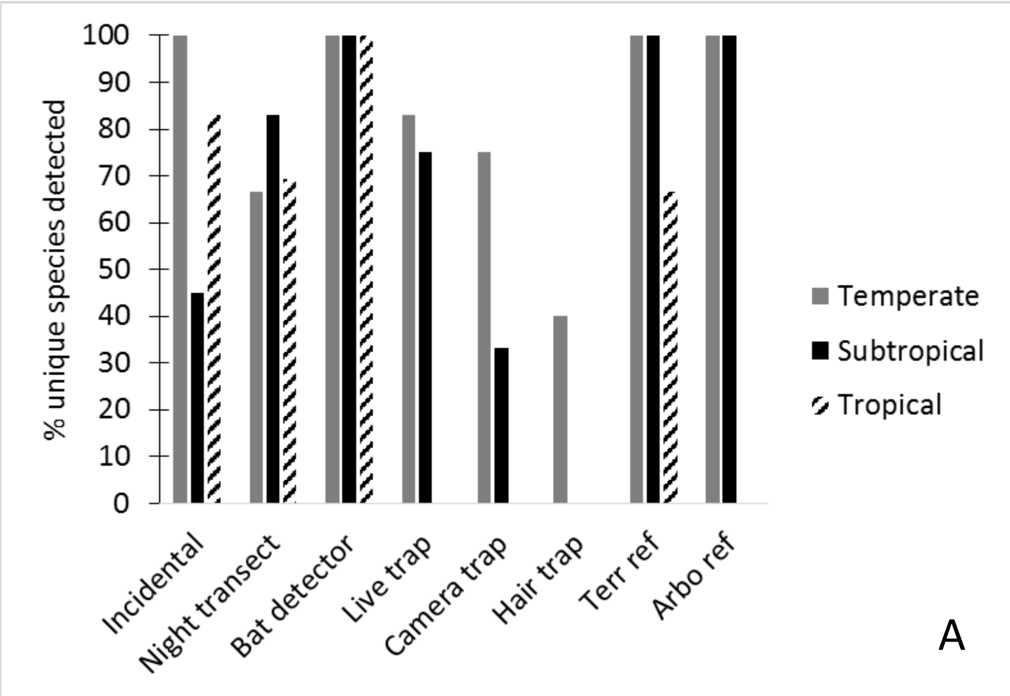


Figure S4 1 Mean species richness of mammals, reptiles and amphibians detected by survey techniques in temperate (grey bars), subtropical (black bars) and tropical (striped bars) mangrove regions over four nights. A) Mean species richness for all three taxa groups together. Error bars display standard deviation around the mean across the four sites in temperate and subtropical regions and two sites in the tropical region. B) Mean mammal species richness detected with each technique across temperate, subtropical and tropical regions. C) Mean reptile species richness detected with each technique across temperate, subtropical and tropical regions. D) Mean amphibian species richness detected with each technique across temperate, subtropical and tropical regions. E) Trap success of techniques in multi taxa approach per region.

Unique species



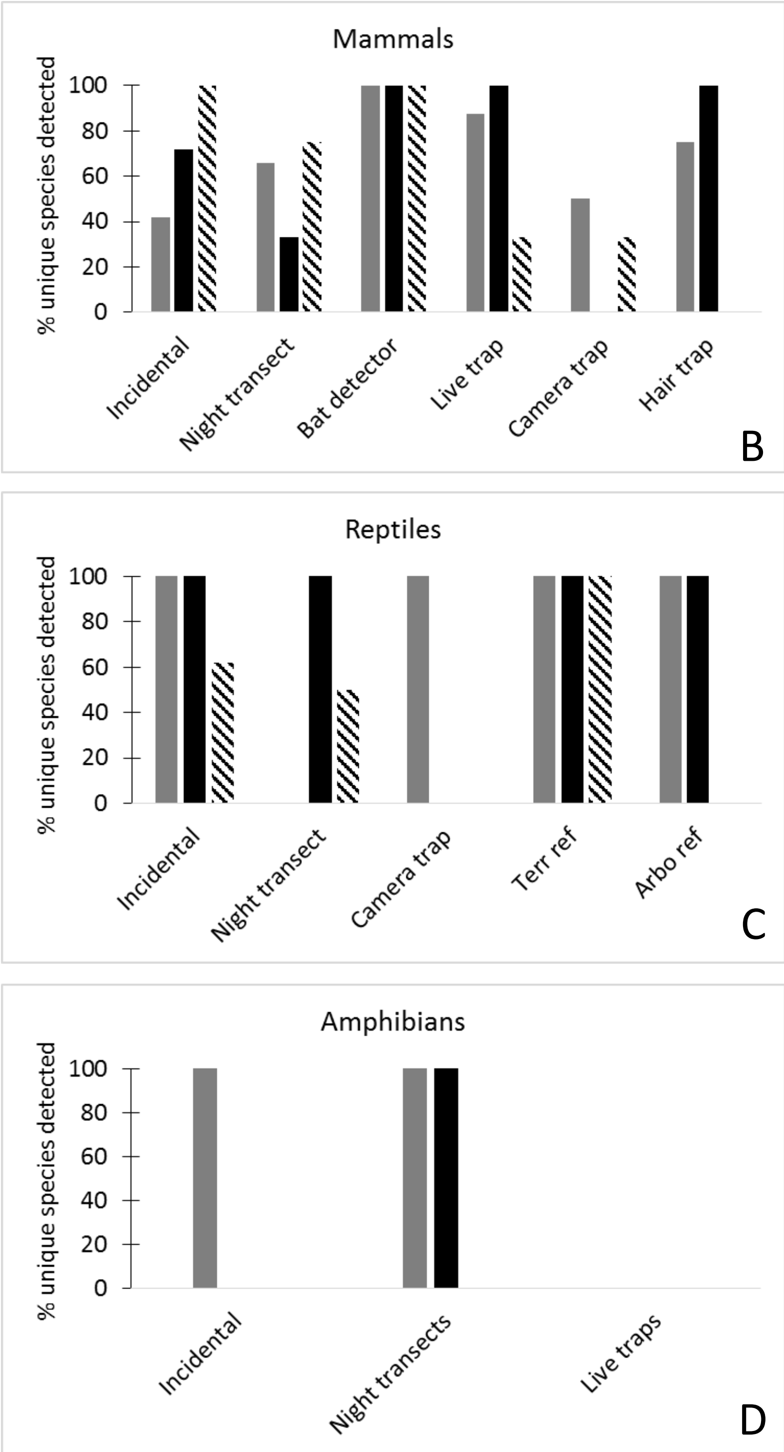


Figure S4.2 The portion of unique species detected per technique per region (i.e. those species that were not detected by any other technique, hereafter referred to as unique species). A) mean unique species per technique detected per region B) Mean percentage of unique mammal species per technique. C) Mean percentage of unique reptile species per technique D) Mean percentage of unique amphibian species per technique.

S5 Time and costs of techniques in the design								
	Time represented as time for 1 person (two persons were out in the field at the same time for safety reasons)							
Time 1 block *4 nights	**Traps per block**	Preparation required (e.g. bait, hair tapes in tubes)	Set up in the field (make transect, attach to trees, set on ground, flag area)	Day before survey to determine tide	Daily checking	Total field time recalc to costs	Species identification processing time	
Incidental	**0**	0	0	0	0	0	60	ID with guidebook
Hairtraps	**6**	20	120	60	120	320	84	All 6 hair tubes full, 2 tapes per tube - this is already costs
Live traps	**10**	20	60	60	360	500	60	ID with guidebook
Night transect	**1*100m**	0	90	60	360	510	60	ID with guidebook
Camera traps	**1**	20	30	60	120	230	60	Analysing 200 pictures (based on maxium pictures taken over 4 nights per camera in this survey)
Arboreal refuges	**5**	0	120	60	360	540	60	ID with guidebook
terrestrial ref	**10**	0	60	60	360	480	60	ID with guidebook
Bat detector	**1**	0	20	60	120	200	720	3hours recordings a night *4 nights
TIME 4 block *4 nights	**Traps per location**	Preparation required (e.g. bait, hair tapes in tubes)	Set up in the field (make transect, attach to trees, set on ground, flag area).	Day before survey to determine tide.	Daily checking	Total field time recalc to costs	Species identification processing time	
Incidental	**0**	0	0	0	0	0	240	
Hairtraps	**24**	80	480	240	480	1280	336	
Live traps	**40**	80	240	240	1440	2000	240	
Night transect	**400m**	0	360	240	1440	2040	240	
Camera traps	**4**	80	120	240	480	920	240	
Arboreal refuges	**20**	0	480	240	1440	2160	240	
Terrestrial ref	**40**	0	240	240	1440	1920	240	
Bat detector	**1**	0	20	60	120	200	720	
COSTS 4 blocks *4 nights	**Traps per location**	Field costs time /60min*20dollar	Post survey costs	Ongoing costs	Equipment costs (see next sheet)	**Total costs**		
Incidental	**0**	0	80	80	0	**80**		
Hairtraps	**24**	160	336	496	160	**656**		
Live traps	**40**	480	80	560	1400	**1960**		
Night transect	**400m**	480	80	560	5500	**6060**		
Camera traps	**4**	160	80	240	2320	**2560**		
Arboreal refuges	**20**	480	80	560	140	**700**		
Terrestrial ref	**40**	480	80	560	300	**860**		
Bat detector	**1**	40	240	280	1100	**1380**		

Supplementary table S5a. Equipment costs

Type	# traps per site	Costs in AUD	Average costs in AUD	Description of equipment used in survey
Incidental sightings	0	0		Pocket camera for each investigator for pictures of detected and/or trapped animals. Used pocket camera (Lumix Tz-17 for all pictures as it has good macro and efficient zoom + takes good videos. Long trousers tucked into socks with light sneakers with laces. Waders are too clunky and too hot and difficult to manoeuvre through thick vegetation. 2L hydrapack each person.
Hairtraps	24	160	60	PVC tubes from Bunnings, 15cm long in 2 sizes 87mm and 40mm. Low cost, you can make 6 tubes from 1 m for $10. Specific double sided tape that has perfect stickyness K5300WH48/25 (25mm x 48mm) via www.primepaksupplies.com.au for $30 per roll. $6.80 for identification of one tube (2 tapes) to analyse (by Barbara Triggs). Other things you need: Zip ties of good quality/strong, black waterproof markers, thick plastic bags for transport, pincers to get tape out, ziplockbags, baking paper, nipper to cut zip ties on take out day.
Live traps	40	1400	30-50	From http://www.wapoultryequipment.net.au/products/alumium_folding_trap. Bait: peanutbutter and oats.
Night transect	400m	5500	1000 - 10000	Headlamps of 250 lumens and spotlight of 1000 lumens +8 spare batteries, cost depend on brand. Thermal imaging camera InfRec G100EX Thermo Gear Camera from Nec-Avio $8365 . Pink flagging tape $10 a roll, get 2 rolls for 400m transect. Trimmers for branches. We noticed that taking videos at night instead of pictures enhanced quality and increases possibility to identify or confirm ID species.
Camera traps	4	2320	170 - 1100	Depending on model camera. Buck eye cams 7X $1060 each+$85 metal casing, $30 phyton lock and SD cards. Large zip ties, plastic bait holders $5 each, peanutbutter.
Arboreal refuges	20	140	70	$10 per 100*100cm black closed cell foam (from Clarke Rubber) 6mm thick. Needed 20*50cm*70cm =7 meters needed. Nails, hammer, white marker, small bender to take nails out when packing up.
Terrestrial ref	40	300	250	$25 per 50*180cm to make 4 sheets of the size used 50*45cm.
Bat detector	1	1100	500 - 1600	Depending on model. Zipties to attach to tree, rechargable batteries + charger.

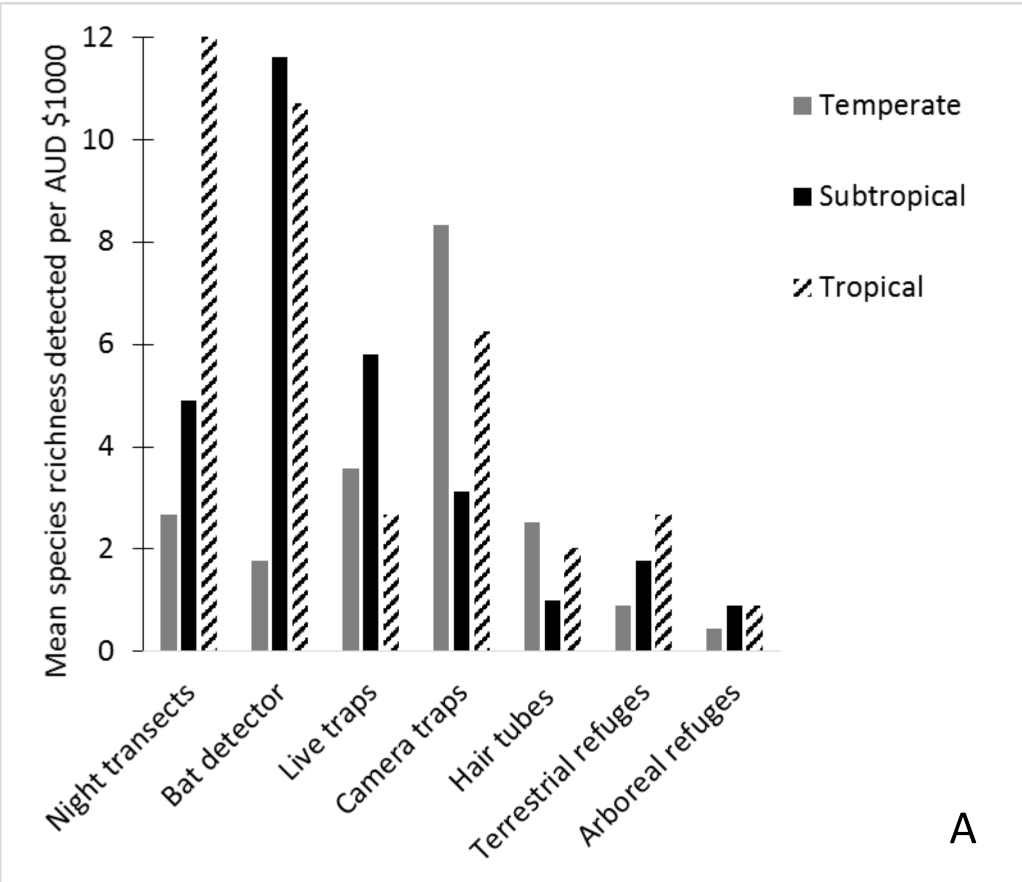
Supplementary table S6. Efficiency per region

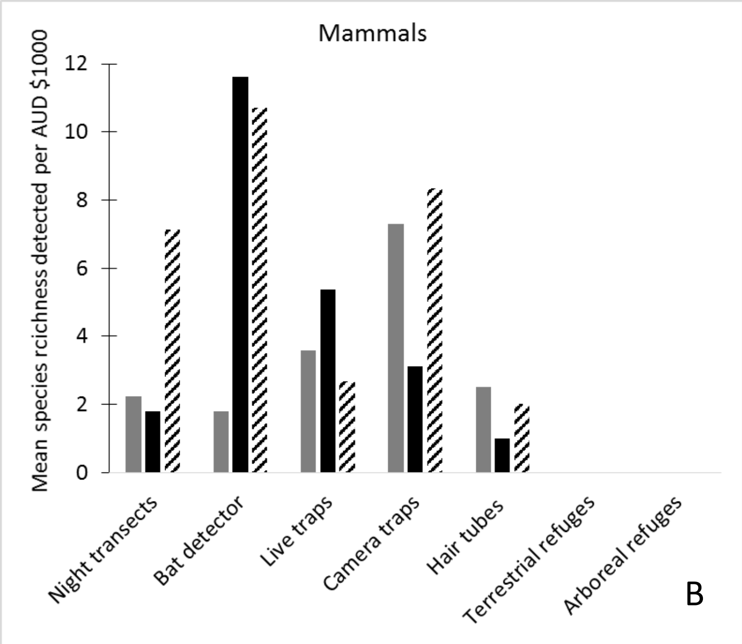
When initial equipment purchasing costs are omitted (i.e., only ongoing costs are considered), the most efficient techniques for multi taxa approach are variable between region (Figure S6.1 A).

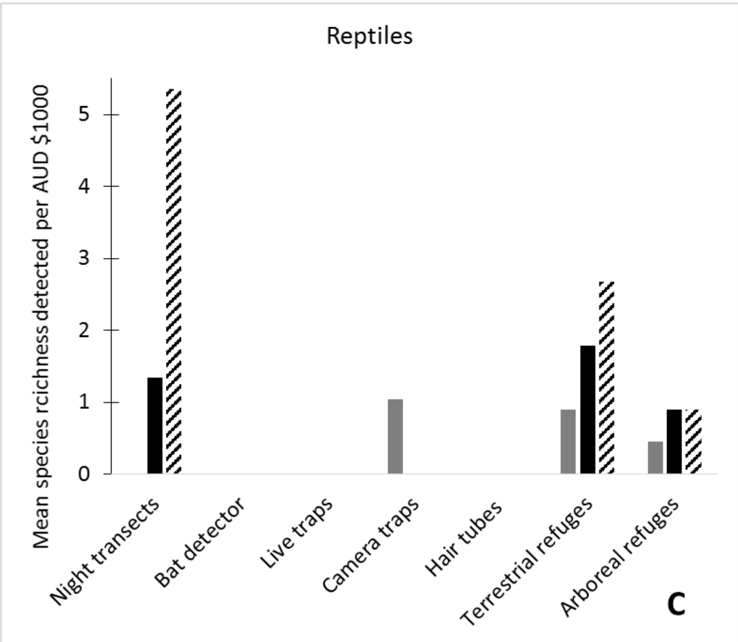
For mammals the bat detector is the most efficient technique in the tropics, while similar techniques remain most efficient in the subtropics and temperate region (Figure S6.1 B).

For reptiles the most efficient technique in the tropics is night transects, subtropics terrestrial refuges and x in the temperate region (Figure S6.1 C).

For amphibians night transects was the only technique that detected species in more than one region with night transects being the most efficient (Figure S6.1 B).




Mammals
Mean species rcichness detected per AUD $1000
0
2
4
6
8
10
12
Night transects
Bat detector
Live traps
Camera traps
Hair tubes
Terrestrial refuges
Arboreal refuges
B


Reptiles
Mean species rcichness detected per AUD $1000
0
1
2
3
4
5
Night transects
Bat detector
Live traps
Camera traps
Hair tubes
Terrestrial refuges
Arboreal refuges
C

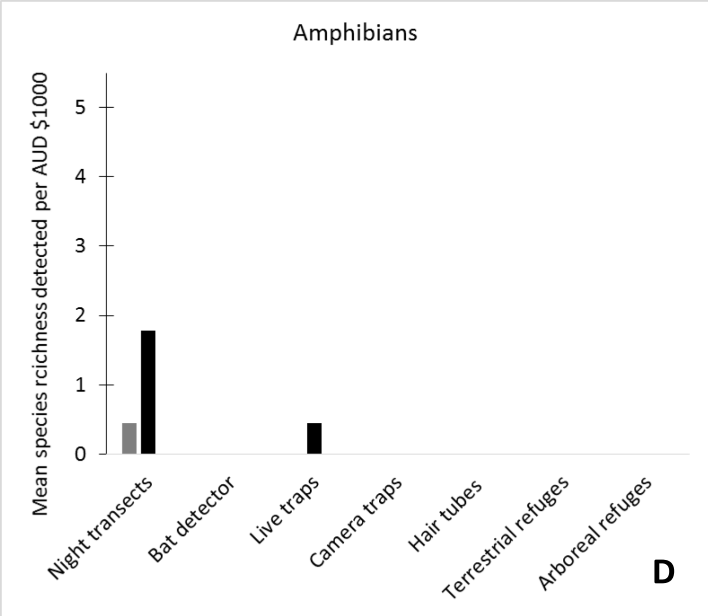


Figure S6. 1. Average success of each survey technique for each taxa group per region for ongoing costs, per $1000. Success is shown for: (A) total species detected per region per $1000 A) mammals detected per region per $1000, (B) with only ongoing costs, (C) reptiles detected per region, (D) amphibians detected per region per $1000. Note the difference in scale.

Supplementary material Chapter four: Rapid surveys of terrestrial vertebrates provide critical information for management strategies of mangrove ecosystems

Supplementary table S1. Mangrove richness, adjacent vegetation maps, benchmark report sources

Supplementary file S2. Interviews

Supplementary table S3. Species, threat categories, resource use, feeding ecology

Supplementary table S4. Species detected per technique

Supplementary table S5. Richness fauna and available habitat

S1 Sources mangrove plant richness, adajacent vegetation maps and benchmark reports

Site	Mangrove plant richness source	Map Source adjacent habitat vegetation community	Benchmark source for adajcent vegetation community	Adjacent habitat description
1	Duke, 2006	http://maps.biodiversity.vic.gov.au/viewer/?viewer=NatureKit	EVC/Bioregion Benchmark for Vegetation Quality Assessment Wilsons Promontory bioregion	Coastal saltmarsh
2	see 1	see 1	see 1	see 1
3	Duke, 2006* and Primefact 746 State of New South Wales 2008	Map Sutherland NSW Sutherland Shire Vegetation Communities Map, 2011. VIS_ID 4198	Targeted Vegetation Survey of Floodplains and Lower Slopes on the Far North Coast http://www.environment.nsw.gov.au/surveys/TargetedVegetationSurvey.htm Appendix 1 - Vegetation community profiles	Swamp oak floodplain forest (from map sutherland)
4	Duke, 2006 and Limeburners creek National Plan of management, NSW National Park and Wildlife Service 2013	Coastal Vegetation of North East NSW. VIS ID 3885	BenchmarksExport - Archive from VIS_ClassificationPublic 17August2017 http://www.environment.nsw.gov.au/projects/biometric-dataset.htm#vegtype	Wett Sclerophyll Forest (*Melaleuca quinquenervia*) & Subtropical Rainforest (*Archontophoenix cunninghamiana*)
5	Duke, 2006 and Broadwater and Bundjalung National Parks and Iluka Nature Reserve Management plan, NSW National Park and Wildlife Service 1997	Coastal Vegetation of North East NSW. VIS ID 3885	BenchmarksExport - Archive from VIS_ClassificationPublic 17August2017 http://www.environment.nsw.gov.au/projects/biometric-dataset.htm#vegtype	Dry Sclerophyll Forest & Woodland/*Corymbia intermedia-Eucalyptus tereticornis*
6	Duke, 2006 and Dowling, R. M. (1986). The mangrove vegetation of Moreton Bay. Queensland Botany Bulletin, (6).	Map in benchmarks report	BioCondition Benchmarks for Regional Ecosystem Condition Assessment. Department of Science, Information Technology and Innovation. QLD government. Last reviewed 5/10/2016	*Eucalyptus racemosa* open-forest on dunes and sand plains. Usually deeply leached soils
7	Duke, 2006 and Dowling, R. M. (1986). The mangrove vegetation of Moreton Bay. Queensland Botany Bulletin, (6).	Map in benchmarks report	BioCondition Benchmarks for Regional Ecosystem Condition Assessment. Department of Science, Information Technology and Innovation. QLD government. Last reviewed 5/10/2016	*Melaleuca quinquenervia* open-forest on coastal alluvium
8	Duke, 2006 and Great Sandy Strait A wetland of international s ignificance Fraiser Island defenders organisation, backgrounders 25 fido.org.au/moonbi	Map in benchmarks report	BioCondition Benchmarks for Regional Ecosystem Condition Assessment. Department of Science, Information Technology and Innovation. QLD government. Last reviewed 5/10/2016	*Melaleuca quinquenervia* open-forest on coastal alluvium
9	Duke, 2006 and Duke, N. C. (1997). Mangroves in the Great Barrier Reef World Heritage Area: current status, long-term trends, management implications and research. In State of the Great Barrier Reef World Heritage Area Workshop (pp. 288-299). Great Barrier Reef Marine Park Authority, Townsville.	Complex mesophyll vine forest found via maps via http://www.wettropics.gov.au/vegetation-maps.html and via Regional ecosystem mapping http://qldspatial.information.qld.gov.au/catalogue/custom/detail.page?fid={01972496-CD6D-4314-B0C0-DA0E0421FB0A} data request relevant adjacent ecosystems is 7.3.17	No benchmark documents for the Cape and Wet tropics. Used expert opinion for species richness from QLD herbarium	Complex mesophyll vine forest (on deep fertile soils). Rainforest. Very wet and wet lowlands and foothills mainly on basalts and alluvium.
10	Duke, 2006 and Duke, N. C. (1997). Mangroves in the Great Barrier Reef World Heritage Area: current status, long-term trends, management implications and research. In State of the Great Barrier Reef World Heritage Area Workshop (pp. 288-299). Great Barrier Reef Marine Park Authority, Townsville.	Medium Eucalyptus chlorophylla woodland and Corymbia clarksoniana and Eucalyptus platyphylla and Melaleuca viridiflora on shallow soil with impided drainage via http://www.wettropics.gov.au/vegetation-maps.html and via regional ecosystem mapping relevant adajcent ecosystems is 3.3.25b	No benchmark documents for the Cape and Wet tropics. Used expert opinion for species richness from QLD herbarium	Medium *Eucalyptus chlorophylla* woodland and *Corymbia clarksoniana* and *Eucalyptus platyphylla* and *Melaleuca viridiflora* on shallow soil with impided drainage. Moist foothills on alluvium

*Duke, N. C. (2006). *Australia's mangroves: the authoritative guide to Australia's mangrove plants* . MER.

Supplementary file S2. Interviews

Interview questions for managers/rangers of 10 National Parks in VIC, NSW and QLD in relation to the PhD study on mangrove conservation from Stefanie Rog, Monash University Melbourne.

Background on park management and monitoring

1) How would you describe your management role?
2) How long have you been managing this park?
3) For how many parks are you responsible?
4) When was the park declared?
5) What are the main management objectives for the park?
6) How often are you able to visit in the park during the year?
7) What sort of activities do you undertake within the park?
8) Which parts of the park are covered during these visits?
9) Is monitoring carried out in the park?
10) If yes, what sorts of things are you monitoring
11) How often are X, Y and Z monitored?
12) What would you say are the key knowledge gaps that influence your ability to carry out the management within the park?

Background on mangrove related management and monitoring

13) What are your management objectives for the mangroves within the park?
14) What information would you like to have to inform your management of the mangrove community?
15) Do you have any information about the condition of mangroves in the park?
16) Do you know whether terrestrial vertebrates like mammals, reptiles and amphibians are using the mangroves?
17) Where did you receive the information about mangroves from?
18) How recent is this information?

Knowledge sharing and communication

19) Do you talk to other park managers about how to manage mangroves?
20) If so, how do you share information (meetings, sharing reports/studies, sharing monitoring data)?

Supplementary table S3. Detected species threat category, resource use and feeding ecology

Identity					Resource use and feeding ecology			
Family group	**Common name**	**Species**	**Threat category**	**Resource use of mangroves detected in field**	**Resource use from in literature**	**Source**	**Feeding ecology**	**Not reported in Australian mangroves by previous reviews**
	Amphibians							
Frog	Cane toad	*Bufo marinus*	Invasive	Feeding	-	-	Carnivore	x
Frog	The Australian green tree frog	*Litoria caerulea*	NA	-	-	-	Carnivore	x
Frog	The eastern dwarf tree frog	*Litoria fallax*	NA	-	-	-	Carnivore	x
Frog	Rocket frog	*Litoria nasuta*	Na	-	-	-	Carnivore	
	Reptiles							
Crocodile	Satwater croc	*Crocodylus porosus*	LC	-	Feeding*, breeding*	Hogarth, 2015	Carnivore	
Lizard	closed-litter rainbow-skink	*Carlia longipes*	NA	-	-	-	Carnivore	x
Lizard	Shaded-litter rainbow-skink	*Carlia munda*	NA	-	-	-	Carnivore	x
Lizard	Brown bicarinate rainbow-skink	*Carlia storri*	NA	-	-	-	Carnivore	x
Lizard	Snake eyed skink p	*Cryptoblepharus pulcher*	NA	Feeding	-	-	Carnivore	x
Lizard	Snake eye skink v	*Cryptoblepharus virgatus*	NA	-	-	-	Carnivore	x
Lizard	Barr sided skink	*Eulamprus tenuis*	NA	Shelter (hollow)	-	-	Carnivore	x
Lizard	Dubious dtella	*Gehyra dubia*	LC	Shelter (hollow)	-	-	Carnivore	x
Lizard	House gecko	*Hemidactylus frenatus*	Invasive	Breeding	-	-	Carnivore	x
Lizard	Sunskink a	*lamphropholis amiculata*	NA	-	-	-	Carnivore	x
Lizard	Sunskink d	*Lampropholis delicata*	NA	-	-	-	Carnivore	x
Lizard	Mourning gecko	*Leptodactylus lugubris*	NA	Breeding	-	-	Carnivore	
Lizard	Glossy grasskink	*Pseudemoia rawlinsoni*	Jurisdiction vulnerable	-	-	-	Carnivore	x
Lizard	Pale lipped shade skink	*Saproscincus basiliscus*	NA	-	-	-	Carnivore	x
Lizard	Lace monitor	*Varanus varius*	NA	-	-	-	Carnivore	x
Lizard	Leggless lizard	*Lialis burtonis*	NA	-	-	-	Carnivore	
Snake	Spotted Python	*Anteresia maculosa*	NA	-	-	-	Carnivore	x
Snake	Tree snake	*Dendrelaphis punctulatus*	LC	-	-	-	Carnivore	
Snake	Marsh snake	*Hemiaspis signata*	NA	-	-	-	Carnivore	x
	Mammals							
Bat	White striped free tailed bat	*Austronomus australis*	LC	-	Feeding	McKenzie 1986	Carnivore	x
Bat	Gould's Wattled Bat	*Chalinolobus gouldii*	LC	-	Feeding	McKenzie 1986	Carnivore	
Bat	The chocolate wattled bat	*Chalinolobus morio*	LC	-	Feeding	McKenzie 1986	Carnivore	x
Bat	The eastern false pipistrelle	*Falsistrellus tasmaniensis*	Jurisdiction vulnerable	-	Feeding	McKenzie 1986	Carnivore	x
Bat	The little bent-wing bat	*Miniopterus australis*	NA	-	Feeding	McKenzie 1986	Carnivore	
Bat	Eastern Bentwing-bat	*Miniopterus orianae*	Jurisdiction vulnerable	-	Feeding	McKenzie 1986	Carnivore	x
Bat	Eastern Bentwing-bat	*Mormopterus lumsdenae*	Jurisdiction vulnerable	-	Feeding	McKenzie 1986	Carnivore	x
Bat	Eastern freetail bat	*Mormopterus ridei*	na	-	Feeding	McKenzie 1986	Carnivore	x
Bat	Southern Myotis	*Myotis macropus*	Jurisdiction vulnerable	-	Feeding	McKenzie 1986	Carnivore	x
Bat	Black flying fox	*Pteropus alecto*	LC	Feeding	Feeding*, shelter*, breed* pollinating*	Hogart 2015, Katherisan 2001	Herbivore	
Bat	The smaller horseshoe bat	*Rhinolophus megaphyllus*	LC	-	Feeding	McKenzie 1986	Carnivore	x

Bat	The little broad-nosed bat	*Scotorepens greyii*	LC	-	Feeding	McKenzie 1986	Carnivore		
Bat	The eastern forest bat	*Vespadelus pumilus*	LC	-	Feeding	McKenzie 1986	Carnivore	x	
Canid	Dingo	*Canis lupus dingo*	lc	-	-	-	Omnivore		
Canid	Domestic dog	*Canus lupus*	Invasive	-	-	-	Omnivore	x	
Canid	Fox	*Vulpus vulpus*	Invasive	-	-	-	Omnivore	x	
Felid	Domestic cat	*Felis catus*	Invasive	-	-	-	Carnivore		
Lagomorph	European rabbit	*Oryctolagus cuniculus*	Invasive	-	-	-	Herbivore	x	
Marsupial	Brown antichinus	*Antechinus stuartii*	LC	-	-	-	Carnivore	x	
Marsupial	Tree Kangaroo	*Dendrolagus bennettianus*	NT	-	-	-	Omnivore	x	
Marsupial	Agile wallaby	*Macropus agilis*	LC	-	-	-	Herbivore		
Marsupial	Grey Kangaroo	*Macropus giganteus*	LC	**Nutrient provision**	-	-	Herbivore	x	
Marsupial	Sugar glider	*Petaurus breviceps*	LC	-	-	-	Omnivore	x	
Marsupial	Brushtailed phascogale	*Phascogale tapoatafa*	VU	-	-	-	Carnivore	x	
Marsupial	Common planigale	*Planigale maculata*	LC	-	-	-	Carnivore		
Marsupial	Ringtail possum	*Pseudocheirus peregrinus*	LC	-	-	-	Herbivore	x	
Marsupial	Brushtail possum	*Trichosurus vulpecula*	LC	-	-	-	Herbivore		
Marsupial	Swamp wallaby	*Wallabia bicolor*	LC	Feeding, dispersal	-	-	Herbivore		
Rodent	Water rat	*Hydromys chrysogaster*	LC	Feeding, shelter	-	-	Carnivore		
Rodent	Grassland Melomys	*Melomys burtoni*	LC	-	-	-	Herbivore		
Rodent	Fawn footed melomys	*Melomys cervinipes*	LC	-	Feeding	McKenzie 1986	Herbivore		
Rodent	House mouse	*Mus musculus*	Invasive	-	-	-	Omnivore		
Rodent	Bush rat	*Rattus fuscipes*	LC	-	-	-	Omnivore		
Rodent	Swamp rat	*Rattus lutreolus*	LC	-	-	-	Herbivore	x	
Rodent	Black rat	*Rattus rattus*	Invasive	-	-	-	Omnivore	x	
Rodent	Water mouse	*Xeromys myoides*	VU	-	**Obligate***	Hogarth, 2015	Carnivore		
Rodent	Delicate mouse	*Pseudomys delicatulus*	LC	-	-	-	Omnivore		
Rodent	Giant white tailed rat	*Uromys caudimaculatus*	LC	Feeding	-	-	Omnivore		
Ungulate	Hog deer	*Axis porcinus*	Invasive	-	shelter*, breeding*	Kathiresan, 2001	Herbivore		
Ungulate	Cow domestic	*Bos taurus*	Invasive	-	Feeding*	Hogarth, 2015	Herbivore	x	
Ungulate	Rusa deer	*Rusa timorensis*	Invasive	-	-	-	Herbivore	x	
Ungulate	Wild pig	*Sus scrofa*	Invasive	-	-	-	Omnivore	x	
* without providing evidence from field observations									

Supplementary table S4. Species detected per technique									
○	Detected by 1 technique								
●	Detected by multiple techniques				**Survey technique**				
		Incidental	Bat detect	Live traps	Night transects	Camera trap	Hair tubes	Terestrial refuges	Arboreal refuges
Family	**Species**								
				Amphibians					
Frogs	*Litoria caerulea*				○				
Frogs	*Litoria fallax*	○							
Frogs	*Litoria nasuta*	●			●				
Frogs	*Bufo marinus*	●		●	●				
				Reptiles					
Crocodilians	*Crocodylus porosus*				○				
Lizards	*Carlia longipes*	●						●	
Lizards	*Carlia munda*	●						●	
Lizards	*Carlia storri*	●						●	
Lizards	*Cryptoblepharus pulcher*	●							●
Lizards	*Cryptoblepharus virgatus*	○							
Lizards	*Eulamprus tenuis*				●				●
Lizards	*Gehyra dubia*				●				●
Lizards	*Hemidactylus frenatus*				○				
Lizards	*Lamphropholis amiculata*	●						●	
Lizards	*Lampropholis delicata*	●						●	
Lizards	*Leptodactylus lugubris*	●			●				
Lizards	*Lialis burtonis*	●						●	
Lizards	*Pseudemoia rawlinsoni*	●						●	
Lizards	*Saproscincus basiliscus*	○							
Lizards	*Varanus varius*	●				●			
Snakes	*Anteresia maculosa*	○							
Snakes	*Dendrelaphis punctulatus*	○							
Snakes	*Hemiaspis signata*	○							
				Mammals					
Bats	*Austronomus australis*		○						
Bats	*Chalinolobus gouldii*		○						
Bats	*Chalinolobus morio*		○						
Bats	*Falsistrellus tasmaniensis*		○						
Bats	*Miniopterus australis*		○						
Bats	*Miniopterus orianae oceanensis*		○						
Bats	*Mormopterus lumsdenae*		○						
Bats	*Mormopterus ridei*		○						
Bats	*Myotis macropus*		○						
Bats	*Pteropus alecto*				○				
Bats	*Rhinolophus megaphyllus*		○						
Bats	*Scotorepens greyii*		○						
Bats	*Vespadelus pumilus*		○						
Canines	*Canis lupus dingo*	●				●			
Canines	*Canus lupus*	○							
Canines	*Vulpus vulpus*	●				●			
Felids	*Felis catus*	○							
Lagomorphs	*Oryctolagus cuniculus*	●				●			
Marsupials	*Antechinus stuartii*			○					
Marsupials	*Dendrolagus bennettianus*	○							
Marsupials	*Macropus agilis*	●				●			
Marsupials	*Macropus giganteus*	●				●			
Marsupials	*Petaurus breviceps*				○				
Marsupials	*Phascogale tapoatafa*				○				
Marsupials	*Planigale maculata*						○		
Marsupials	*Pseudocheirus peregrinus*				○				
Marsupials	*Trichosurus vulpecula*					●	●		
Marsupials	*Wallabia bicolor*	●				●			

Rodents	*Hydromys chrysogaster*				●		●		
Rodents	*Melomys burtoni*	●		●	●		●		
Rodents	*Melomys cervinipes*	●		●	●		●		
Rodents	*Mus musculus*			○					
Rodents	*Pseudomys delicatulus*				○				
Rodents	*Rattus fuscipes*			●		●	●		
Rodents	*Rattus lutreolus*			●			●		
Rodents	*Rattus rattus*			●			●		
Rodents	*Uromys caudimaculatus*				●	●			
Rodents	*Xeromys myoides*				○				
Ungulates	*Axis porcinus*	○							
Ungulates	*Bos taurus*	○							
Ungulates	*Rusa timorensis*	●				●			
Ungulates	*Sus scrofa*	○							

Supplementary table S5. Richness fauna and available habitat

Location	Fauna richness	Mangrove plant richness	Mangrove hollows	Adjacent habitat plant richness
1	8	1	1	11
2	5	1	11	12
3	8	2	17	23
4	13	2	3	51
5	8	1	12	43
6	12	5	20	40
7	19	5	22	22
8	14	10	25	22
9	13	26	4	105
10	22	22	6	45

Chapter five: Strengthening governance for intertidal ecosystems requires a consistent definition of boundaries between land and sea

Supplementary table S1. Legislation sources, boundaries and definitions

Supplementary table S1a. References for table S1

Supplementary table S2. Management sources and mangrove taxonomy

Supplementary table 1 Legislation sources, boundaries and definitions

Source documents	Land - Sea boundaries clearly described	High Water Mark definition	Refence to mangrove plant	Detail on Native vegetation definition
Jurisdictional boundaries				
Environment Protection and Biodiversity Conservation Act 1999 (Federal)	no	no	no	All Australian biodiversity (focus on nationally and internationally important flora).
Survey Regulations 2006 (NSW)	The normal baseline for measuring the breadth of the territorial sea is the low‑water line along the coast as marked on large‑scale charts officially recognized by	Line of mean high tide between the ordinary high water spring and ordinary high-water neap tides.	no	
Land Administration Act 1997 (WA)	Land means — (a) all land within the limits of the State; and (b) all marine and other waters within the limits of the State; and (c) all coastal waters of the State as defined by section 3 (1) of the Coastal Waters (State Powers) Act 1980 of the Commonwealth; and (d) the sea-bed and subsoil beneath, and all islands and structures within, the waters referred to in paragraphs (b) and (c).	Ordinary high water mark at spring tides.	no	
Survey and Mapping Infrastructure Act 2003 (QLD)	The high water mark along the coast.	Ordinary high water mark at spring tides.	no	
Seas and Submerged Land Act 1973 (Federal)	The normal baseline for measuring the breadth of the territorial sea is the low-water line along the coast as marked on large-scale charts officially recognized by the coastal State.	no	no	
Halbury's Laws of Australia (355 Real property) (Federal)	The boundary line for a Crown grant of land described as abutting the seashore is presumed, in the absence of a contrary intention in the transferring instrument, to be the mean high water mark.	The mean high water mark at common law is assessed by averaging out the tidal level reached by the springs (the highest tide of each lunar month) and the neaps (the lowest tide of each lunar month) over the period of a year.	na	
Native vegetation				
VIC - No specific Native Vegetation Act, but see Victoria Planning Provision – Definitions – Clause 72	no	no	no	Indigenous to Victoria, including trees, shrubs, herbs and grasses.
Native Vegetation Act 2003 (NSW)	no	na	"Native vegetation" does not include any mangroves, seagrasses or any other type of marine vegetation.	Native vegetation means any of the following types of indigenous vegetation: (a) trees (including any sapling or shrub, or any scrub), (b) understorey plants, (c) groundcover (being any type of herbaceous vegetation), (d) plants occurring in a wetland.

Native Vegetation Act 1991 (SA)	no	na	The Council may give its consent to the clearance of native vegetation — If an application for the Council's consent relates to mangroves (Avicennia marina) within the Adelaide Dolphin Sanctuary, the Council must, before giving its consent.	Native vegetation means a plant or plants of a species indigenous to South Australia including a plant or plants growing in or under waters of the sea but does not include—(a) a plant or part of a plant that is dead unless the plant, or part of the plant, is of a class declared by regulation to be included in this definition; or (b) a plant intentionally sown or planted by a person unless the plant was sown or planted.
Vegetation Mangement Act 1999 (QLD)	no	na	Vegetation is a native tree or plant other than the following: a mangrove	Vegetation is a native tree or plant other than the following— (a) grass or non-woody herbage; (b) a plant within a grassland regional ecosystem prescribed under a regulation; (c) a mangrove.
No specific Native Vegetation Act (NT)	na	na	na	
No specific Native Vegetation act (WA	na	na	na	
Fisheries management				
Fisheries Act 1995 (VIC)	To any waters of the sea not within the limits of the State that are on the landward side of waters adjacent to the State that, within the meaning of that Part, are within the Commonwealth proclaimed waters. Consistent with Seas and Submerged lands act 1973.	no	no	
Fishieries Management Act 1994 (NSW)	Foreshore means below high astronomical tide but do not identify whether this is the boundary applied.	no	"Marine vegetation" means any species of plant that at any time in its life must inhabit water (other than fresh water).	
Fisheries Management Act 2007 (SA)	Waters include intertidal area	no	Aquatic plant mangrove.	
Fishieries Act 1994 (QLD)	Land includes foreshores and tidal and nontidal land. foreshore means parts of the banks, bed, reefs, shoals, shore and other land between high water and low water.	High water at highest spring tide	Mangrove is marine plant.	
Fisheries Act 1988 (NT)	no	no	"Not defined aquatic life" means any species of plant or animal life (except species of birds) which, at any time of the life history of the species, must inhabit water, and includes the plant or animal at any stage of its life history, and also includes any part of such plant or animal, but does not include fish, or aquatic life declared by the Minister by notice in the Gazette to be aquatic life to	Plant, in relation to aquatic life, includes seaweeds, sea-grasses, and algae.
Fish Resources Management Act 1994 (WA)	no	no	Aquatic flora.	

Protected areas				
National Parks Act 1975 (VIC)	Includes Marine Parks. Inconsistent boundaries decribed for parts of these parks. Some examples: "MORNINGTON PENINSULA NATIONAL PARK17 "Excepted is any land between high water mark and low water mark", or FRENCH ISLAND NATIONAL PARK28: "excepted is any land between high water mark and 150 metres seawards of high water mark" and "Unregulated land" means that part of the land in the park that is not comprised of the land that is 200 metres seawards from high water mark.	Mean high water mark.	no	no
National Parks and Wildlife Act 1974 (NSW)	Tidal land needs to be discussed with fisheries.	no		Native plant means any tree, shrub, fern, creeper, vine, palm or plant that is native to Australia, and includes the flower and any other part thereof.
National Parks and Wildlife Act 1972 (SA)	no	no	no	
National Parks and Wildlife Act 1975 (QLD)	no	no	no	
Territory Parks and Wildlife Conservation Act 1976 (NT)	"Land" includes the sea above any part of the sea bed of the Territory. freshwater" means the water in a lake, lagoon or billabong whether or not it is at any time connected with the sea and water in any stream above the tidal limit.	no	no	
Reserves (National Parks And Conservation Parks) Act 2004 (WA)	no	no	no	no
Conservation and Land Management Act 1984 (WA)	Land includes tidal waters, intertidal zone means the land, or the land and waters, below the high water mark and above the low water mark.	no	no	no
Marine Parks Act 1997 (NSW)	Not clear, for 2 marine parks specified: marine park includes crown land and coastal waters up to mean high water mark.	no	no	no
Marine Parks Act 2007 (SA)	A marine park is to consist of a part of the sea that is within the limits of the State or the coastal waters of the State, and may include land or waters held by, or on behalf of, the Crown within or adjacent to the specified part of the sea.	no	no	no
Marine Parks Act 2004 (QLD)	Includes tidal waters.	no	no	

Threatened species and communities				
Flora and Fauna Guarantee Act 1988 (VIC)	Land includes land covered by water.	no	no	Flora means any plant-life which is indigenous to Victoria whether vascular or non-vascular and in any stage of biological development and includes any other living thing generally classified as flora.
Threathened Species Conservation Act 1995 (NSW)	land includes: (b) land covered with water, and (c) the sea or an arm of the sea, and (d) a bay, inlet, lagoon, lake or body of water, whether inland or not and whether tidal or not, and (e) a river, stream or watercourse, whether tidal or not.	no	Does not include marine vegetation within the meaning of Part 7A of the Fisheries Management Act 1994.	
National Parks and Wildlife Act 1972 (SA)	Land includes waters.	no		Native plant means any plant that is indigenous to Australia and includes any plant of a species declared by regulation to be a native plant.
Nature Conservation Act 1992 (QLD)	Land includes: (a) the airspace above land; and (b) land that is, or is at any time, covered by waters; and (c) waters.	no	no	
Territory Parks and Wildlife Conservation Act 1976 (NT)	"Land" includes the sea above any part of the sea bed of the Territory. "Freshwater" means the water in a lake, lagoon or billabong whether or not it is at any time connected with the sea and water in any stream above the tidal limit.	no	no	
Wildlife and Conservation Act 1950 (WA)	na	no	no	

Coastal management act				
Coastal Management Act 1995 (VIC)	Coastal Crown land includes any Crown land within 200 metres of high water mark.	none	no	no
Coastal Management Act 2016 (NSW)	Mean high water.	Mean high water.	Indirect: enhance coastal defence including vegetation and wetlands.	
Coastal Protection Act 1972 (SA)	Coast means all land that is (a)within the mean high water mark and the mean low water mark on the seashore at spring tides; or (b) above and within one hundred metres of that mean high water mark; or (c) below and within three nautical miles of that mean low water mark; or (d) within any estuary, inlet, river, creek, bay or lake and subject to the ebb and flow of the tide; or (e) declared by regulation to constitute part of the coast for the purposes of this Act.	Mean high Wwter at spring tide.	no	
Coastal Protection and Management Act 1995 (QLD	Coastal land is land above high water mark.	Ordinary high water at spring tides.	Mangroves mentioned as coastal wetland.	Vegetation includes trees.
No Specific Coastal Management Act (NT)	na	na	na	
No Specific Coastal Management Act (WA)	na	na	na	

Supplementary table S1a. References table 1

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Supplementary table S2. Management sources and mangrove taxonomy				
Jurisdiction	Sources	Mangrove plant species mentioned	Species list and species identification provided	Marine or terrestrial characterisation clarified
WA	WA department of Fisheries, Fisheries fact sheet: Mangroves Government of Western Australia, Department of Fisheries, Perth 2012.	Nr of species mentioned per region	No	Marine importance
WA	Western Australian Planning Commission (2001) Coastal zone management policy for Western Australia. Perth Australia.	No	No	Mentioned as important for fish
SA	Edyvane, K.S. (1999) Conserving Marine Biodiversity in South Australia - Part 2 - Identification of areas of high conservation value in South Australia. SARDI.	1 species, is total species number in SA	No	Marine plant
SA	Scientific Working Group (2011) The vulnerability of coastal and marine habitats in South Australia. Department of Environment, Water and Natural Resources. Adelaide, Australia.	1 species, is total species number in SA	No	Not clear
NSW	Stewart, M. and Fairfull, S. (2008). Mangroves. Primefact 746. NSW Department of Primary Industries.	All species mentioned	Species list and ID plates and discription	Marine plant
NSW	Fairful, S. (2013) Policy and guidelines for fish habitat conservation and management. NSW Department of Primary Industries. Wollongbar.	2 species mentioned	No	Marine plant
	Mangrove survey threathened species.	All species	Species list including assosiates	Not clear
Federal	Vegetation Profiles: Mangroves MVG23. Department of Environment and Energy. Commonwealth Australia.	Not all species provided, incomplete	No	Not clear
QLD	Goudkamp, K. and Chin, A. June 2006, 'Mangroves and Saltmarshes' in Chin. A, (ed) The State of the Great Barrier Reef On-line, Great Barrier Reef Marine Park Authority, Townsville.	Only the number of species given	No species list but hybrids mentioned	Marine plant but terrestrial values described
QLD	Mangroves, WetlandInfo, Department of Environment and Heritage Protection, Queensland	Only the number of species given	No species list but mention associates exist	Marine plant
QLD	Environment Planning (2012)Queensland Coastal Plan. State of Queensland (Department of Environment and Heritage Protection).	No		Not clear
VIC	Victorian Coastal Council (2014) The Victorian Coastal Strategy 2014. The State of Victoria Department of Environment and Primary Industries Melbourne.	No species given	No	Not clear
VIC	Harty, C. (2011) Mangroves of Victoria information kit. People & Parks Foundation Melbourne.	1 species, is total species number in VIC	List including assosiate species	Not clear
VIC	Victorian Environmental Assessment Council (2014) Marine Investigation Final Report. The State of Victoria.	No	No	Marine
NT	Lee, G.P. (2003), Mangroves in the Northern Territory, Department of Infrastructure, Planning and Environment, Darwin.	Total number species	Species list of 6 most common species, assosiates aknowledged but without species list	Marine and terrestrial values

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