Research article

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

Stefanie M. Rog*, Carly N. Cook
School for Biological Sciences, Monash University, 25 Rainforest Walk, Melbourne 3800, Australia

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, we 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. We 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. We 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. Our study demonstrates the urgent need for a global review of legislation governing the boundaries of land and sea to determine whether the suggestions we offer may provide global solutions to ensuring these critical systems do not fall through the cracks in ecosystem protection and management.

© 2017 Elsevier Ltd. All rights reserved.

1. 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 (e.g., coastal protection and 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, 2014). 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 enforce legislation, as demonstrated for example in Indonesian rainforests where poorly defined protected forest area boundaries have enabled illegal logging (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 (Alvarez-Romero et al., 2011) and national policies (Friess et al., 2016). The uncertainty to which of the two 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 cross system approach to studying threats to these ecosystems (Alvarez-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 marine or terrestrial is whether threat mitigation is the responsibility of marine or terrestrial protected areas. 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 (Alvarez-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 (Alvarez-Romero et al., 2011; Beger et al., 2010), however as long as separate marine and terrestrial protected area boundaries exist the different focus points 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 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 characterisation, 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). Our study is the first to consider the drivers of this complexity from an ecosystem boundary perspective. We explore the complexity in how the land and sea boundaries are defined among jurisdictions and types of legislation, the characterisation of vegetated intertidal ecosystems as marine or terrestrial and the taxonomic classification of intertidal plant species, using Australian mangroves ecosystems as a case study. We use these data to evaluate how this complexity affects the protection of intertidal ecosystems, with the goal of identifying how governance structures for these complex ecosystems can be strengthened.

2. Methods

2.1. Study region

We 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; Barter 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 encroachment by coastal development; the most significant threat to intertidal ecosystems globally (Giri et al., 2011; Foster et al., 2013).

2.2. Study system

Mangroves occur along the coastline of five out of six Australian jurisdictions. Mangrove ecosystems make an ideal case study because they can occur across the full intertidal zone from the lowest tide line to the highest (Astronomical) tide line (Fig. 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)).

2.3. Data collection

To assess the governance structures for intertidal ecosystems, specifically for mangrove ecosystems in Australia, we 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 we 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 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.

2.4. 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 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 also provided reference to the landward boundary as well as the 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”.

2.5. The marine or terrestrial characterisation 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”.

2.6. Taxonomic classification of mangrove species and ecosystems

What a “true mangrove” plant species is has a long history of debate and we 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 (Spalding et al., 2010; Sandilyan and Kitharesan, 2012; Polidoro et al., 2010; Tomlinson, 2016). In many cases, 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 we 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.

2.7. The protection for vegetated intertidal ecosystems

2.7.1. Protection by native vegetation legislation

As mangroves are vegetated intertidal systems we were interested to determine the proportion of Australian jurisdictions where mangroves were acknowledged as vegetation and thereby included under native vegetation laws (see Table S1s), or if they were included under fisheries acts and thereby seen as a marine feature.

2.7.2. Protection within protected areas

To quantify the proportion of Australian mangrove ecosystems within protected areas we 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 Geocentric Datum of Australia (GDA 1994) projection. All protection was taken as one category without distinction into IUCN protected area levels (Dudley, 2013).

2.8. The ability of marine or terrestrial protected areas to mitigate threats

To evaluate the effectiveness of marine or terrestrial protection offered to mangrove ecosystems we assessed the capacity of current protected areas to mitigate key global threats to mangrove ecosystems (Duke, 2006b; Sandilyan and Kitharesan, 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, we 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. We 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).

3. Results

3.1. Legislation defining boundaries between land, sea and the intertidal zone

3.1.1. 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”. 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 (Fig. 1). While there is a definition 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.

3.1.2. 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 Fig. 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 Fig. 1) including more of the landward area of the intertidal ecosystems in private land and thereby more intertidal 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 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) (Fig. 2).

We 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. We 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 (Fig. 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 “excluding 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 (see Table S1).

3.2. The marine or terrestrial characterisation of intertidal ecosystems by legislation

We found 8 out of 24 legislative documents from the five legislation categories (see Section 2.3) that mention mangroves, aquatic plants or mention native vegetation (see Table S1). Of these, three acts from three jurisdictions describe mangroves as marine, while five from the other three jurisdictions provide no characterisation as marine or terrestrial (Fig. 3B).

3.3. 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", see Table S2). Of these 15 descriptions 10 classified mangroves as marine, while the others 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 (see Table S2).

3.4. The protection for vegetated intertidal ecosystems

3.4.1. 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 (see Table S1). Mangroves are included with equal frequency under native vegetation management or fisheries management legislation (Fig. 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 (see Table S1). From the 15 acts relating to the five legislation categories (2.3), three acts specifically mention objectives for mangroves.

3.4.2. Protection within protected areas

We found that approximately 49% (9391 km²) of Australia’s mangrove ecosystems are within some form of protected area. Across Australia, this protection is relatively equally distributed between marine (2258 km² -24%) and terrestrial (1989 km² -21%) protected areas; however the distribution varies significantly between jurisdictions (Fig. 4). Our analysis also identified 374 km² (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.

---

Fig. 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.

Fig. 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).
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. Three threats are outside direct control of protected areas (e.g., climate change, natural disasters and oil spills).

4. Discussion

Despite the importance of the ecosystem services provided by mangroves and their increasing global vulnerability to anthropogenic threats, we 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.

4.1. 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, we 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, we 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 problem within a country, and are likely to be most important for areas with significant tidal ranges (Fig. 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 Sainthilan, 2009). Our 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.

We 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) (Fig. 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 (Fig. 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 marine and part terrestrial (Fig. 1A and 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 (Fig. 3C), and likely contributes to the inconsistency with which mangroves are protected within marine versus terrestrial protected areas (see 3.1.2). 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 for 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, we caution against the creation of an additional boundary by creating a separate protected area network for intertidal zones, as has been...
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.

<table>
<thead>
<tr>
<th>Manageable</th>
<th>Threats</th>
<th>How are mangroves affected</th>
<th>Rank marine PA in mitigating threat</th>
<th>Rank terrestrial PA in mitigating threat</th>
<th>Argumentation</th>
</tr>
</thead>
<tbody>
<tr>
<td>1 Coastal development for housing, hotels, harbours, infrastructure and agriculture</td>
<td>Destruction by clearing and filling of the intertidal zone. Affecting total mangrove forest by: i) taking away the physical plants, ii) negatively affecting the mangrove forest by altering tidal flows (Amanullah 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 et al., 2005)</td>
<td>Low</td>
<td>High</td>
<td>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 thereby protecting larger parts of mangroves. Estuaries, river mouths 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.</td>
<td></td>
</tr>
<tr>
<td>2 Coastal aquaculture</td>
<td>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 v) and alteration of water drainage patterns (Páez-Osuna, 2001).</td>
<td>High</td>
<td>High</td>
<td>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. 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 runoff of sediment and pollution to MPA’s is prevented as well. 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.</td>
<td></td>
</tr>
<tr>
<td>3 Mining</td>
<td>Land mining for mineral products (iron, nickel, bauxite) affect mangroves through toxic waste and alteration of natural flows (Ohimain, 2003). Coral mining can affect mangroves by increased erosion rates caused by the loss of reef breakwaters (Dulvy, 1995).</td>
<td>Medium</td>
<td>High</td>
<td>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, estuaries 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.</td>
<td></td>
</tr>
<tr>
<td>4 Commercial exploitation fish shellfish</td>
<td>Mangroves are degraded by construction of boat moorings and infrastructure for land access. Fishing can alter foodwebs (e.g. less top predators), and can cause pollution by garbage and oils spills (Davenport and Davenport, 2006)</td>
<td>Medium</td>
<td>Medium</td>
<td>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 regulate tourism by guiding access to mangrove areas through boardwalks and assigning non-access areas for 4WD’s.</td>
<td></td>
</tr>
<tr>
<td>5 Tourism</td>
<td>Tourism requires transport that links coastal road to hotels (Davenport and Davenport, 2006). Sightseeing causes trampling of saplings (Ross, 2006), pollution and increase boat wash causes erosion (Davenport and Davenport, 2006) and damage by anchoring (Burgin and Hardiman, 2011). Off road recreation vehicles affect mudflats (Bridgewater and Cresswell, 1999)</td>
<td>Medium</td>
<td>Medium</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Manageable</td>
<td>6</td>
<td>Low</td>
<td>Medium</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
suggested by others (Banks et al., 2005). Our 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.

4.2. 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 we found evidence that the debate is far from over (Mukherjee et al., 2014), mangrove plants and ecosystems were more often characterized 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 components of mangrove ecosystems by legislation and within relevant management documents. 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). This marine portraying of mangroves is not limited to Australia (e.g. Mongabay, 2016) and may come from the emphasis placed on the marine components of mangrove ecosystems by legislation and within relevant management documents. 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

<table>
<thead>
<tr>
<th>Threats</th>
<th>How are mangroves affected</th>
<th>Rank marine PA in mitigating threat</th>
<th>Rank terrestrial PA in mitigating threat</th>
<th>Argumentation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Pests and diseases</td>
<td>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 (Biswas 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 and Bunbury, 2015), and invasive deer are known to trample and eat saplings. Exploitation for wood/charcoal, fish, crabs and other wildlife (Hoq, 2007). Grazing by domestic cattle (Hoppe-Speer and Adams, 2015). Degrading by accessing, trampling, clearing, burning, pollution and altering foodwebs.</td>
<td>Medium</td>
<td>High</td>
<td>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 adjacent terrestrial ecosystems. MPAs 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 potentially 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 in certain global regions.</td>
</tr>
<tr>
<td>Not manageable 8 Climate change</td>
<td>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 7000 km mangroves in Northern Australia (Duke, 2017).</td>
<td>na</td>
<td>Low</td>
<td>MPAs cannot mitigate this threat as MPAs 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 cause erosion that impedes mangrove establishment. Only indirect mitigation by both MPA's and terrestrial PA's by striving for healthy ecosystems. More resilient mangroves can possibly withstand changes better due to genetic and phenotypic diversity, larger patches have also more possibility to re-establish due to higher amount of propagules.</td>
</tr>
<tr>
<td>9 Natural disasters</td>
<td>Mangroves require years to recover from cyclone damage (Chhotray et al., 2012)</td>
<td>Low</td>
<td>Low</td>
<td>Only indirect mitigation by both MPA's and terrestrial PA's by striving for healthy ecosystems. More resilient mangroves can possibly withstand changes better due to genetic and phenotypic diversity, larger patches have also more possibility to re-establish due to higher amount of propagules.</td>
</tr>
<tr>
<td>10 Oil spills from boats</td>
<td>Oil causes dieback of mangroves by smothering root systems (Duke, 2002)</td>
<td>Low</td>
<td>Low</td>
<td>Only indirect mitigation by both MPA's and terrestrial PA's by striving for healthy ecosystems. More resilient mangroves can possibly withstand changes better due to genetic and phenotypic diversity, larger patches have also more possibility to re-establish due to higher amount of propagules.</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
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).

4.3. 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 (section 3.2) 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 destroyed 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 feature associated with common families (e.g. air roots in mangroves, fleshy leaves in saltmarsh) and so are 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. 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.

4.4. The protection of vegetated intertidal ecosystems

4.4.1. 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 (Fig. 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; Fig. 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.

4.4.2. Protection within protected areas

We found that mangrove ecosystems in Australia receive relatively high levels of representation within protected areas (49%) compared to global mangrove protection (6.8%; Giri et al., 2011) and most other threatened ecosystems (UNEP, 2014). Although variable across jurisdictions (Fig. 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; Álvarez-Romero, 2015). To avoid a situation where protected areas only protect part of the intertidal there is a need to ensure that the protected area boundaries are drawn to capture 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 adoptions 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 recommendation 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.

4.5. 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), we found that for mangrove ecosystems, terrestrial protected areas are likely to be most effective at mitigating their key global threats (see 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). We 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 (Fig. 3D, Section 3.1.4) 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 (Horigae et al., 2016). The failure to formulate specific objectives for these ecosystems may stem from the significant knowledge gaps about how these ecosystems function (Alvarez-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.

5. Conclusion

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. Our 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. Our study demonstrates the urgent need for a global review of legislation governing the boundaries of land and sea to determine whether the suggestions we offer may provide global solutions to ensuring these critical systems do not fall through the cracks in ecosystem protection and management.

Acknowledgements

This research did not receive any specific grant from funding agencies in the public, commercial, or not-for-profit sectors.

Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.jenvman.2017.04.052.

References


