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An evaluation of nest predator impacts and the efficacy of plastic meshing on marine turtle nests on the western Cape York Peninsula, Australia

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ABSTRACT

Nest predation is considered to be one of the most significant biotic threats to marine turtle populations globally. The introduction of feral predators to nesting beaches has dramatically increased nest predation, reaching near total egg loss in some regions. We monitored a 48 km stretch of beach along western Cape York Peninsula, Australia, from June – November 2018. We recorded a total of 360 nests comprising 117 flatback and 243 olive ridley nests. We installed plastic meshing (90 cm × 100 cm) on 110 olive ridley nests (45.2% of total olive ridley clutches laid) within the study area. We classified all nest predation attempts into three categories: complete, partial, or failed predation events. In total, 109 (30.2%) of all marine turtle nests were depredated by a variety of predators, including feral pigs, dingoes, goannas, and humans. The addition of plastic meshing reduced the likelihood of dingoes gaining access to eggs, but not goannas or feral pigs. Further, we found no difference in the proportion of hatchling emergence between meshed and un-meshed nests. Additionally, while hatchling emergence was reduced in nests that had been partially depredated, these nests still produced live hatchlings and contributed to recruitment. The success of particular predator control methods is often predator, and/or regionally, specific. Our findings highlight a thorough understanding of predator guilds and their relative impacts is required to deploy targeted and predator-specific strategies to maximize conservation results. We present a strong case for data-driven adaptive management that has implications for designing optimal predator management plans.

1. Introduction

Marine turtles face a series of threats globally. Their populations suffer from pressures both on land and at sea, ranging from pollution (Eckert, 1995; Lazar and Gračan, 2011), fisheries bycatch (Hall et al., 2000; Pandav et al., 1997), and challenges from rising sea water, including nest inundation, and lack of suitable nesting beaches due to erosion or beach habitat loss (Fish et al., 2005; Fuentes et al., 2010; Whytlaw et al., 2013). In addition, excessive predation rates from native and feral animals (Davis and Whiting, 1977; Whytlaw et al., 2013) lead to reduced recruitment and population declines (Engeman et al., 2003; Hamann et al., 2010; Stancyke, 1982). Marine turtle populations have declined globally in concert with many anthropogenic practices, in part because marine turtles face threats at all life-stages, from nestlings to adults.

Eggs may represent the most vulnerable life-stage for turtles, given their survival is dependent on various external environmental factors. While marine turtles are extremely fecund, and can lay multiple clutches of eggs per year (Miller, 2017), entire nests, or even nesting beaches, can be destroyed by predators (Engeman et al., 2005; Garmestani and Percival, 2005). A variety of native and feral animals, including invertebrates, reptiles, birds, and mammals are known to prey upon marine turtle eggs (Garmestani and Percival, 2005; Kurz et al., 2012; Lei and Booth, 2017; Stancyke, 1982). While native animals have sustainably harvested turtle nests for thousands of years, turtle populations are now at risk due to a lack of recruitment as a result of additional predation pressure from feral animals among other threats (Limpus, 2008; Whytlaw et al., 2013). For example, in Australia, feral red foxes

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(*Vulpes vulpes*) and feral pigs (*Sus scrufa*) are responsible for high levels of predation, exceeding 90% nest predation in some regions prior to predator control or nest protection programs (Limpus et al., 1993; Limpus, 2008; Whytlaw et al., 2013).

Conservation and management efforts vary regionally depending on what the primary threats are to hatchling recruitment. Marine turtle conservation efforts to reduce nest predation and increase hatching success have generally taken two pathways: 1) applying predator deterrents to nests such as physical barriers like wire cages or plastic meshing (Lei and Booth, 2017; O'Connor et al., 2017; Yerli et al., 1997); and/or 2) via predator control programs including trapping, baiting, or culling known nest predators (Engeman et al., 2005; O'Connor et al., 2017; Ratnaswamy et al., 1997). Conservation tools, such as installing nest cages or predator meshing, have been shown to drastically reduce predation rates in some regions (Engeman et al., 2006; Ratnaswamy et al., 1997; Smith et al., 2013). Despite concerns about cages and meshing trapping nestlings (Adamany et al., 1997) or interfering with the magnetic fields around the nest (Irwin et al., 2004), nest protection via these (or modified) practices have become standard protocol since the 1960s (Breckenridge, 1960; Engeman et al., 2016).

In Australia, a substantial portion of turtle conservation efforts are focused around the eastern and western coastlines that are well-known nesting grounds for loggerhead (*Caretta caretta*) and flatback (*Natator depressus)* turtles (Limpus, 2009). Most of northern Australia is extremely remote and difficult to access, especially during the wet season. This has led to a shortage of monitoring and therefore a lack of information on population dynamics in these remote areas. Queensland's remote Cape York Peninsula and the Torres Strait also support large populations of nesting marine turtles, predominantly flatback and olive ridley (*Lepidochelys olivacea*) turtles, but also green (*Chelonia mydas*) and hawksbill (*Eretmochelys imbricata*) turtles in lower densities (Limpus, 2009). Few quantitative studies have been conducted on the remote beaches of western Cape York Peninsula, despite their supporting thousands of nesting marine turtles each year. Historically, feral animals, such as pigs, have been a major threat to turtle nest predation in this region since their introduction (Limpus et al., 1993). The primary aims of this study were to 1) identify the primary predator groups of sea turtle nests along a remote stretch of beach on western Cape York Peninsula, 2) quantify successful and unsuccessful predation rates by each predator group, and 3) test the efficacy of adding plastic meshing over nests to reduce the likelihood of a predator gaining access to the egg chamber. Our overall aim was to evaluate and quantify the return-on-investment for existing management strategies and to inform future adaptive management strategies.

2. Methods

2.1. Study site

This study was conducted along a 48 km stretch of coastline between the Love and Kirk Rivers managed by the Aak Puul Ngantam (APN) Aboriginal ranger group, located approximately 30 km southwest of Aurukun, on the west coast of Cape York Peninsula, Queensland, Australia (Fig. 1). The site was chosen for its accessibility and a six-year history of marine turtle surveys and feral animal control.

2.2. Surveys for turtle nests

Nest monitoring took place between June and November 2018. Turtle nest monitoring frequency was dependent on the frequency of turtle nesting events. During the peak season (July–September) monitoring took place daily. This was important as challenging weather events (e.g., strong winds/tidal influences) would increase the difficulty of locating new nests over time. During the shoulder periods (June–July, October–November) nest monitoring was reduced to two–three surveys per week in concert with lower nesting activity.

Nest monitoring surveys were conducted in the early morning, late afternoon or evening, and overnight to avoid the hottest times of the day. We used quadbikes and side-by-side UTV vehicles to access the beach. A single drivable pathway north and south along the beach was chosen close to the edge of the dunes to minimize disturbance to the beach from multiple vehicle tracks. The same track was driven (where possible) each monitoring session to make it easier to locate known nests and identify new turtle tracks.

Upon locating a new turtle track, we identified the species by crawl characteristics and located the egg chamber where possible, marking it with an individually numbered heavy-duty sand peg. Each marked nest was checked every monitoring session thereafter, and new nests were added sequentially. Notable events such as signs of predation, tidal inundation, hatching, and general changes to nest characteristics were recorded. All data was recorded in a custom iPad application, called *Nestor*, which was developed to enable easy data entry in the field. *Nestor* standardized data collection and included mapping capability to reduce observer error.

To gain an understanding of nest success and hatchling emergence, we counted the total number of eggs deposited in nests when possible (e.g., a turtle was found while dropping eggs, or while searching for the egg chamber of a recently found nest.) Only a subset of nests were fully counted and reburied due to time constraints (n = 34 flatback nests; n = 150 olive ridley nests). We excavated all nests that had emerged or had reached 60 days of incubation. A partial nest excavation was conducted if a nest appeared to be depredated to identify if any eggs remained in the egg chamber. If some eggs remained within the egg chamber, the nest was reburied and a full excavation was conducted after the full 60-day incubation period. This information was important in identifying the proportion of eggs in each nest that successfully hatched and emerged from the nest.

2.3. Application of nest meshing

Upon locating nests, PVC-coated flower mesh (90 cm wide, with mesh size of 50 mm × 50 mm; Whites) was applied over turtle nests as a predator deterrent. Due to limited resources and differences in vulnerability listing status in Queensland (olive ridley turtles are listed as endangered, flatback turtles are listed as vulnerable; Nature Conservation Act 1992; September 2017 list), the application of plastic meshing was only implemented on olive ridley nests due to their shallow nest chambers (compared to flatback nests; Limpus, 1971; Santidrián Tomillo et al., 2017) and high susceptibility to predation (Perry, unpublished data). Mesh was cut into 90 cm × 100 cm pieces to fit over each nest and then dug down into the sand around the nest (10 cm deep), centering the mesh over the egg chamber. The mesh was secured around the perimeter with additional sand pegs (up to 9 pegs). The perimeter of the mesh was then covered with sand, leaving the centre of the meshing area unburied.

2.4. Nest predation assessment

We classified nest predation events into three types: failed, partial, or complete predation. Failed nest predation events involved signs of digging around the nest, but the predator was ultimately unable to access the egg chamber, leaving all eggs undamaged. Failed predation events occurred as a result of predator deterrents, such as the application of plastic meshing over the nest, or for unknown reasons, such as the predator being unable to locate the egg chamber. Partial predation events were recorded when digging occurred at the nest and the predator was able to access the egg chamber and consume or destroy some, but not all, of the eggs in the chamber. A complete predation event was recorded when a predator completely destroyed the nest, consuming or destroying all the eggs in the chamber, resulting in complete egg loss. Nest predators were identified via footprints, tracks, and characteristic digging and damage to nests.

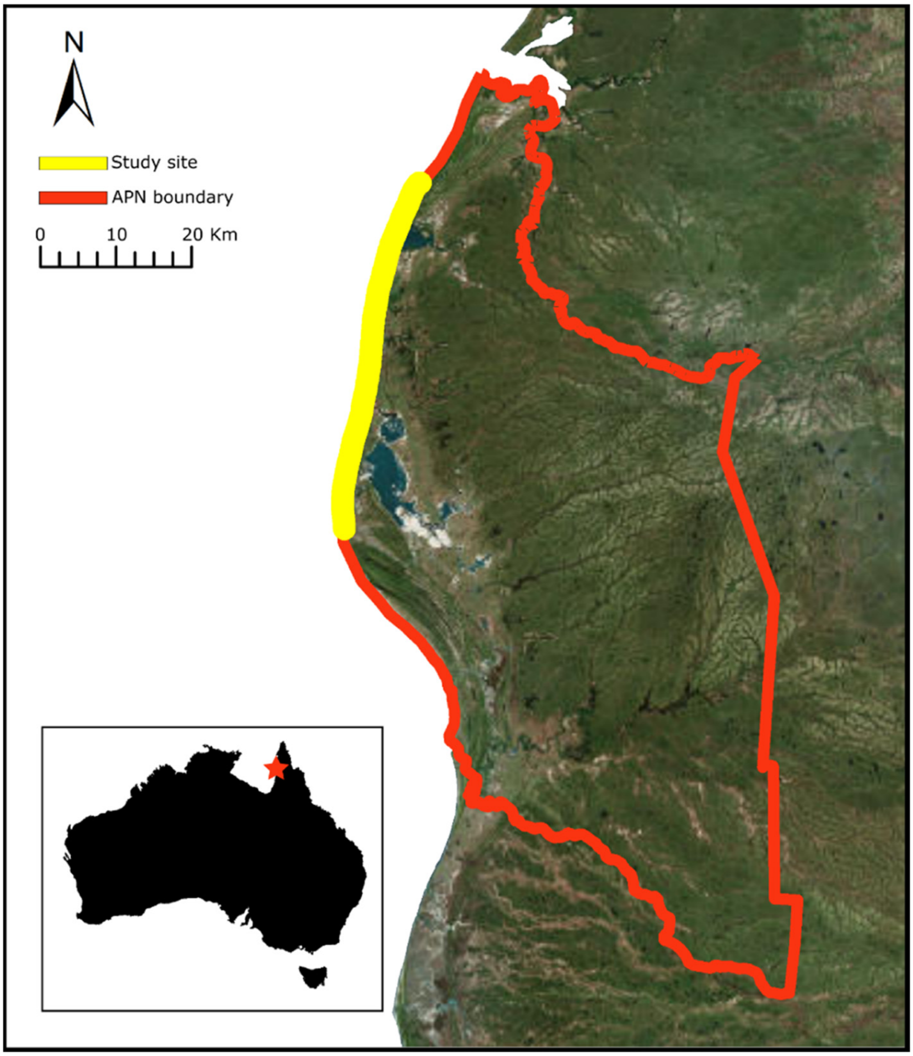

Fig. 1. Study site and Aak Puul Ngangtam (APN) property boundary along the western coast of Cape York Peninsula, Queensland, Australia.

2.5. *Statistical analyses*

We used Fisher's exact tests to compare the proportions of predation events between turtle species, and to compare the likelihood of failed predation events between meshed and un-meshed nests. We used a Wilcoxon test to compare the mean hatchling survival between meshed and un-meshed nests and a two-factor ANOVA to compare the mean hatchling emergence between meshing status and predator type. We inspected residual boxplots, conducted Shapiro-Wilk and Bartlett tests to determine if the assumptions of normality and homogeneity of variance were met, and used non-parametric tests where required. Means are presented as ± 1 SE. All statistical analyses were conducted in the R statistical programming environment (R Core Development Team, 2017).

3. Results

We conducted 107 beach surveys during the study period (Jun–Nov 2018). We found and monitored 360 nests from two species of marine turtle (243 olive ridley and 117 flatback). Turtle nests were found along the entire stretch of beach, with olive ridley nests being mostly evenly distributed within the study area (Fig. 2). In contrast, flatback nests showed a more clustered distribution with two higher density patches; one in the north and one in the south (Fig. 2).

Predation attempts were common, with 168 nests (46.6% of all nests) showing signs of digging at nest locations. We documented a total of 142 (58.4%) predation attempts on olive ridley nests compared to 26 (22.2%) predation attempts on flatback turtle nests (Fisher's exact test: P = 0.012). Overall, the most common predators of both olive ridley and flatback turtle nests were dingoes (*Canis lupus dingo*), goannas (*Varanus panoptes*), and feral pigs (*Sus scrofa*). Humans and unknown predators accounted for the remainder of predation events (Table 1). Note that “unknown” predators may still have been pigs/goannas/dingoes, but there was not enough evidence to confirm the identity. Predation attempts (disregarding meshing status) were generally evenly distributed across the study area, but more patchy with flatback nests, and bimodally distributed in olive ridley nests (Fig. 2). Predation events showed little pattern in their spatial distribution; goanna and dingo predation was evenly distributed along the beach, predation by humans was centered around an indigenous traditional owner camp in the middle of the beach, and predation by feral pigs was clustered in the northern stretches of the beach near a large freshwater wetland. Predation by feral pigs was both spatially and temporally constrained, as all predation by pigs occurred within a nine-day stretch in late September along 7 km of beach.

3.1.1. *The effectiveness of meshing turtle nests as a predator deterrent*

A total of 110 olive ridley nests (45.2%) were meshed across the

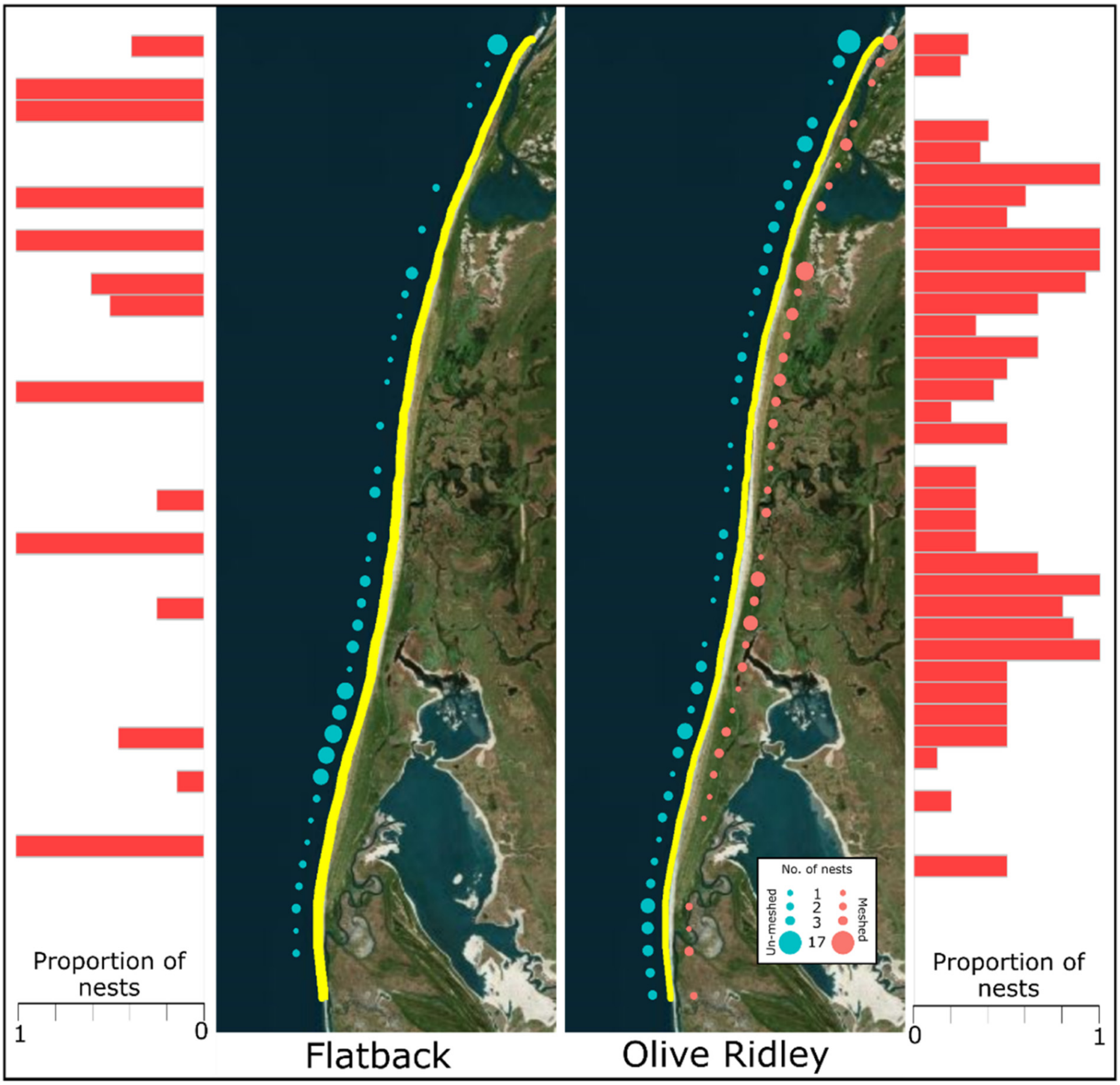

Fig. 2. Distribution of nests of two species of marine turtle along a 48 km stretch of beach. Circles show the number (size) and meshing status (colour) of nests in each 1 km subsection. The red bars represent the proportion of nests in each subsection with signs of predation attempts (includes all predation types: complete, partial, and failed attempts). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

study site (Flatback nests were not meshed). Meshing occurred approximately evenly along the entire study area; however, predation events were not evenly distributed (Fig. 2). Predation by feral pigs only occurred along a 7 km stretch of beach within the northern section of the study area and, when pig predation events occurred, it resulted in complete egg consumption (Fig. 3). In contrast, predation of both meshed and un-meshed nests by dingoes and goannas occurred along the majority of the study site with only the most southerly 6 km of beach not experiencing any predation (Fig. 3). This is possibly due to the southern stretch of beach being on a relatively isolated spit (see Fig. 2).

The addition of plastic meshing to olive ridley nests provided some predator-specific success in mitigating nest predation. Plastic meshing significantly increased the likelihood of a failed predation attempt by dingoes (Fisher's exact test: $P = 0.007$) but not by goannas or feral pigs ($P > 0.05$ for both groups; Fig. 4). Further, meshing nests shifted the majority of dingo predation events from complete predations in un-meshed nests to partial predation events when meshing was applied (Fig. 5). Even though both complete and partial predation events resulted in a loss of eggs, partially depredated nests still produced hatchling turtles (see below).

Hatchling emergence was not significantly different between meshed and un-meshed non-depredated nests (Wilcoxon test: $W = 2521.5$, $P = 0.546$). Hatchling emergence (calculated as the proportion of eggs in each nest that successfully hatched and made it to the surface) from non-depredated nests was 78.4% ($n = 124$) in olive ridley nests. Among partially depredated nests by dingoes and goannas, both meshing status and predator group did not have a significant effect on hatchling emergence (Two-factor ANOVA: mesh status: $F_{1,16} = 2.603$, $P = 0.126$; predator group: $F_{1,16} = 2.296$, $P = 0.149$; Fig. 6).

Un-meshed nests suffered the highest rates of successful predation from dingoes (17.3% of available un-meshed nests), goannas (15.8%), and feral pigs (8.3%; Table 1). Dingoes were twice as likely to leave an un-meshed nest completely depredated than only partially depredated. Goannas were equally likely to partially depredate a nest as leave a nest completely destroyed. Although feral pig predation rates were relatively low, in contrast to predation by native predators, 100% of feral pig predation events led to a complete loss of eggs (Fig. 5). Failed predation events were rare among un-meshed nests, as predators were

Table 1

Predation attempts on olive ridley and flatback turtle nests. Predation types indicate successful predation events (‘Complete’ – all eggs eaten or destroyed, or ‘Partial’ – some eggs eaten or destroyed, but at least some eggs remain in the nest) or unsuccessful predation events (‘Failed’ – digging occurred at the nest, but the predator was unable to access the egg chamber). ‘% Nests’ indicates the proportion of the total number of nests each predator group came in contact with. ‘% Predator’ indicates the proportion of all predation events for which each predator group was responsible.

	Un-meshed olive ridley nests (n = 133)					Un-meshed flatback nests (n = 117)				
	Predation type					Predation type				
Predator	Complete	Partial	Failed	% Nests	% Predator	Complete	Partial	Failed	% Nests	% Predator
Dingo	14	7	2	17.3	37.1	4	2	5	9.4	37.9
Goanna	9	10	2	15.8	33.9	1	4	3	6.8	27.6
Pig	11	0	0	8.3	17.7	8	0	0	6.8	27.6
Human	3	0	0	2.3	4.8	0	0	0	0.0	0.0
Unknown	4	0	0	3.0	6.5	0	0	2	1.7	6.9

	Meshed olive ridley nests (n = 110)				
	Predation type				
Predator	Complete	Partial	Failed	% Nests	% Predator
Dingo	4	12	10	23.6	29.5
Goanna	14	30	14	52.7	66.0
Pig	3	0	0	2.7	3.4
Human	0	0	0	0.0	0.0
Unknown	1	0	0	< 1.0	1.1

generally able to locate the egg chamber (failed predation attempts by dingoes: 8.5% of predation attempts; goannas: 9.5% of predation attempts).

3.1.2. *Predation of flatback turtle nests*

Predators failed to locate the egg chambers of flatback turtles more often than olive ridley turtles (failed predation attempts by dingoes: 45.5% of predation attempts; goannas: 37.5% of predation attempts). Goannas, dingoes, and feral pigs depredated 6.8%, 9.4%, and 6.8% of available flatback nests respectively (Table 1). Similar to olive ridley nests, dingoes were more likely to completely depredate nests, goannas were more likely to partially depredate nests, and feral pigs always completely depredated nests (Appendix F1). Hatchling emergence from non-depredated nests was 87.5% (n = 28), and 94.4% (n = 2) from partially depredated nests.

4. Discussion

Predation of nests by both native and feral animals has become a significant threat to marine turtles around the world (Garmestani and Percival, 2005; Limpus, 2008; Stancyke, 1982). Feral pigs, foxes, raccoons, and wild dogs are among the top predators of sea turtle nests and continue to suppress recruitment of turtle populations through excessive predation (Garmestani and Percival, 2005; Norris et al., 2005; Stancyke, 1982; Whytlaw et al., 2013).

Western Cape York Peninsula (WCYP) represents an important nesting region for marine turtles, particularly olive ridley and flatback turtles (Limpus, 2008; Limpus et al., 1983; Norris et al., 2005; Whytlaw et al., 2013). Further, this region supports high densities of feral pigs, which pose a series of conservation concerns. Feral pigs represent a significant ecological threat along WCYP and Torres Strait (Limpus et al., 1993; Whytlaw et al., 2013), not only through predation of turtle nests, but additionally as predators of other native species (Fordham et al., 2006, 2008; Norris et al., 2005) and by causing habitat destruction around forested and wetland ecosystems (Campbell and Long, 2009; Engeman et al., 2007; Mayer et al., 2000; Norris et al., 2005). Historically, feral pigs have been the biggest threat to marine turtle nest survival in the study area, with nearly 100% nest predation by feral pigs in 2012 (Ross et al., 2017). Since then, a collaboration between APN and CSIRO has initiated an intensive aerial culling and baiting program to manage feral pig densities in the region (Ross et al., 2017). As a result, our study indicated a substantial reduction in turtle nest predation by feral pigs (from 100% nest predation by feral pigs in 2012 to 6% in 2018). Despite the reduction in feral pig predation since 2012, every existing feral pig predation event resulted in total egg loss, indicating that ongoing feral pig management is critical for the conservation of marine turtles in this region. As pig predation in our study was spatially and temporally constrained, we speculate that only a few individual pigs were responsible for the predation of turtle nests. This suggests that large scale aerial culling may no longer be the most effective way to manage the scattered remaining pigs. Rather, targeted hunts or bait stations may be more economically and biologically suitable. Especially in cases with feral pigs, few individuals can cause catastrophic damage to nests over a short period of time, and thus need to be closely monitored and controlled. Further, with continual changes in predator populations, funding allocation, etc., we highlight the importance of an adaptive management strategy that periodically assesses threats and solutions to maximize resource allocation.

Reducing the predation rates on marine turtle nests by feral animals should increase recruitment of hatchlings. In order to sustain turtle populations, the IUCN recommends that at least 70% of total eggs laid should remain protected (i.e., no > 30% egg loss; Eckert et al., 1999). Our data from this study falls right on the threshold, with 30.2% of our nests having been successfully depredated to some level, albeit only 21.1% (n = 76) of our nests were completely depredated (total egg loss). We highlight that predation of nests should not simply be classified in a binary construct (i.e., depredated or non-depredated) because even partially depredated nests can produce hatchlings. This is especially true for native predators, given that native predators such as goannas, would have historically fed on marine turtle nests for thousands of years without causing declines in turtle populations. Goannas make up the largest proportion of predation attempts on turtle nests at APN, but a majority of these attempts resulted in only partial predation events. Hatchling emergence remained relatively high even after partial predation events by dingoes (63.5 ± 11.7%) and goannas (34.3 ± 11.5%). We suspect that in other regions where predation is only classified as ‘depredated’ or ‘non-depredated’ managers may be overestimating the loss of clutches due to predators. However, in our

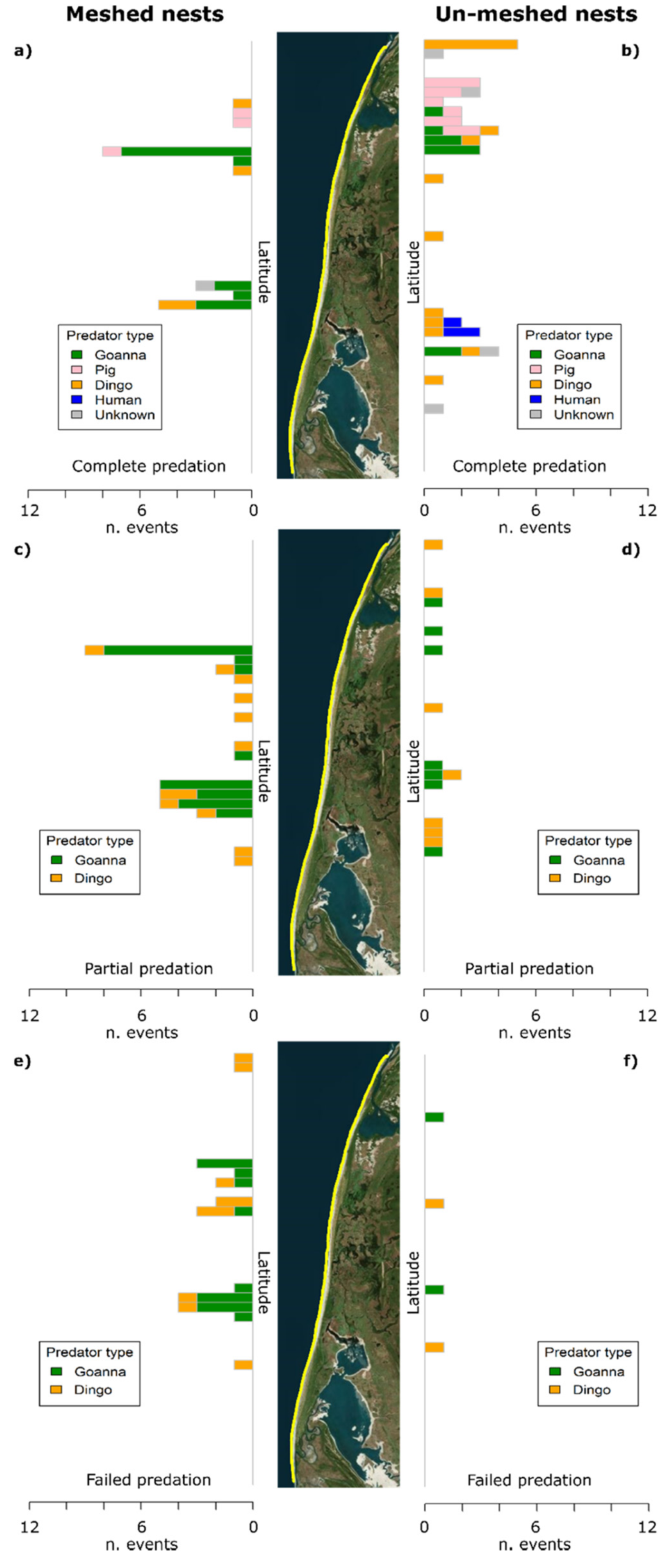

Fig. 3. The spatial distribution and frequency of predation events of olive ridley nests in 1 km subsection along 48 km of beach within the study area. The spatial and frequency distribution of complete predation events are shown for meshed (a) and un-meshed (b) nests; partial predation events on meshed (c) and un-meshed (d) nests; and failed predation events on meshed (e) and un-meshed (f) nests.

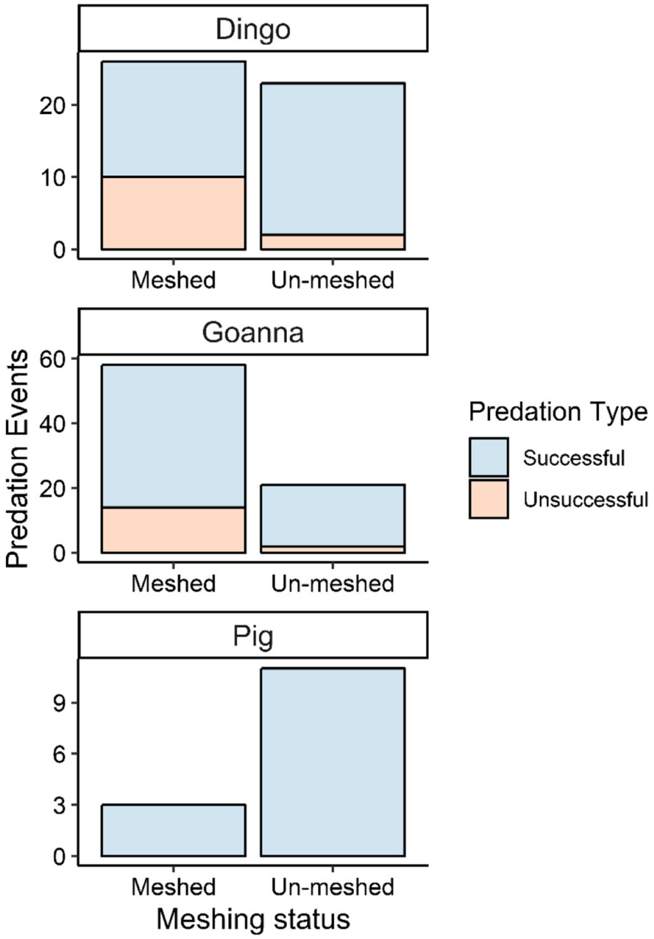

Fig. 4. Frequency of predation events on olive ridley turtle nests. Successful predation represents a combination of partial and complete predation (see text for details), where some or all eggs were consumed or damaged. Unsuccessful predation represents failed predation attempts, where predators dug into or around the nest, but were unable to reach the egg chamber. The addition of plastic meshing (meshed) significantly reduced the likelihood of successful predation events by dingoes (Fischer's exact test: $P = 0.007$), but not for goannas or feral pigs ($P > 0.05$ for both predator groups).

study, after discovering a partial predation event and examining the remaining contents, the nest was reburied. After a partial predation event, an un-manipulated nest (i.e., a nest not reburied/with exposed egg chamber) may face further challenges (e.g., increased nest temperatures or further predation events) that may further contribute to nest mortality. Further experiments examining the impacts of partial predation events and extended egg chamber exposure should be investigated.

The addition of plastic meshing to marine turtle nests showed a predator-specific response, reducing the predation success of dingoes, but not of goannas or feral pigs. In fact, predation attempt rates by goannas were 2.8 times higher on meshed compared to un-meshed nests. In a similar study, Lei and Booth (2017) found that goannas were generally unsuccessful at locating the edges of plastic meshing over turtle nests, resulting in low predation rates. One major difference in the mesh design between our study and that of Lei and Booth (2017), is the latter completely buried the mesh in sand, suggesting that goannas may be able to visually locate turtle nests with exposed mesh, increasing predation rates, as in our study. It is currently unknown what cues, specifically, goannas use to locate turtle nests (Lei and Booth, 2018), but it is possible that they may have used the visible plastic meshing or nest marking pegs as a visual or olfactory signal to identify the presence of turtle nests. This, however, is yet to be experimentally tested. Mammalian predators of turtle nests have been shown to either use (Mroziak et al., 2000) or not use (Burke et al., 2005; Riley and Litzgus, 2013; Strickland et al., 2010; Tuberville and Burke, 1994)

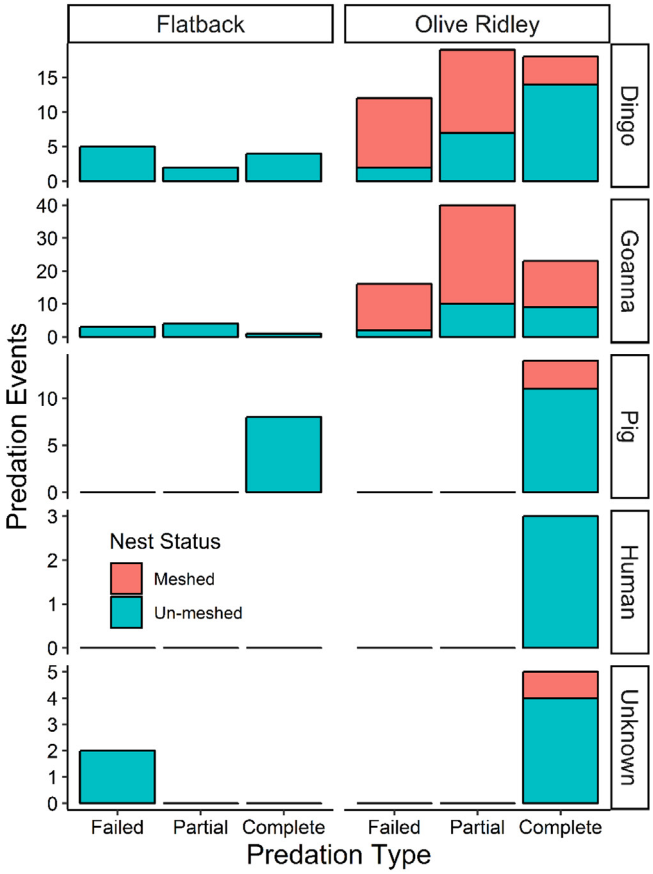

Fig. 5. Frequency of predation types by predator group for both flatback (n = 117) and olive ridley (n = 243) turtle nests. Failed predation ("Failed") represents events where the predator was unable to access the egg chamber, resulting in no egg loss. Partial predation ("Partial") represents predation events where some, but not all, eggs were consumed or damaged. Complete predation ("Complete") represents a total loss of the nest, where all eggs were consumed or damaged. Nests with plastic meshing applied over the egg chamber are represented in red, and un-meshed nests are represented in blue. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

visual markers including exclusion cages over nests to locate nests. For predators with a keen sense of smell, such as dogs, pigs, and even goannas, researchers may unintentionally identify nests for predators by leaving olfactory cues at the nest through digging and working around the nest. While untested to our knowledge, these types of experiments would be of value to managers and researchers trying to conserve sensitive species.

Implementing a predator-deterrent program, such as the application of predator-proof fencing or meshing, is a time consuming endeavor that often requires a lot of resources (e.g. time, manual labor, and funding). In our study system, the application of plastic meshing over marine turtle nests has shown predator-specific effectiveness. The biggest threat, historically, has been feral pig predation, for which the plastic meshing did not prevent access to the nests, as pigs were able to dig through and break the meshing. Even more robust predator-deterrents, such as aluminum cages, have proven ineffective in stopping feral pig predation of turtle nests, although they have been shown to increase the time it takes pigs to access the egg chamber, providing an extra few days or weeks for hatchling development and potential hatching (Engeman et al., 2016). As part of the larger project, additional control methods have been put in place to manage the feral pig numbers at the APN study site, which have dramatically reduced the predation attempts on turtle nests. In contrast, plastic meshing was an effective method in reducing successful predation events by dingoes. Further, meshing not only increased the number of failed predation attempts, but perhaps more importantly, also shifted the majority of predation events from complete predation to partial predation. While meshing was ineffective at blocking all access to the egg chamber by predators, it did reduce the number of eggs consumed by shifting the predation type.

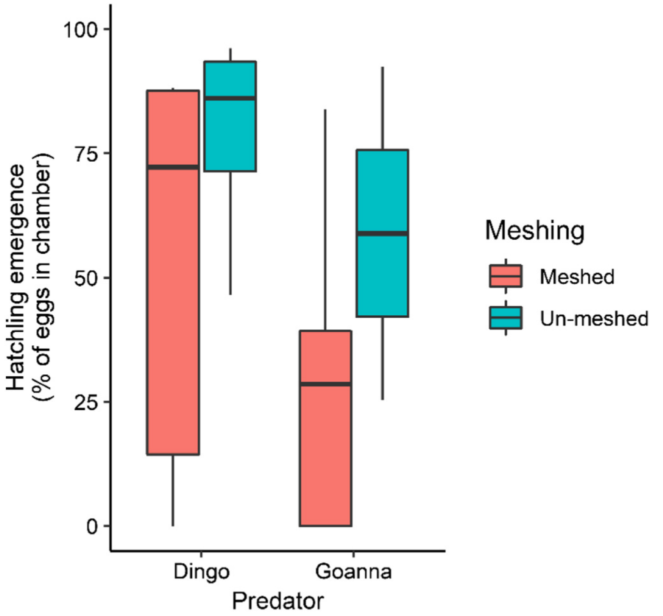

Fig. 6. Mean (± SE) hatchling emergence (percentage of laid eggs that emerged successfully) from olive ridley nests, after partial predation events by dingoes or goannas (n = 19 nests). Neither meshing status nor predator type had a significant effect on hatchling emergence (two-factor ANOVA: mesh status: $F_{1,16}$ = 2.603, P = 0.126; predator group: $F_{1,16}$ = 2.296, P = 0.149).

Alterations to meshing or cage designs, or other exclusion devices may or may not be worth additional resources, depending on the predators present in a system. It is worth noting that alterations to the meshing design used herein could further deter predators like goannas or dingoes. Both dingoes and goannas typically accessed the egg chamber on meshed nests by digging under or from the side of meshed area. Various meshing material and designs exist, and extending the flaps of mesh further away from the egg chamber may help increase the effectiveness. Additionally, minimizing any visual cues that a nest is present could also be worth trialing, such as making sure the mesh is fully buried or removing nest identification stakes. Given that some predators are notoriously successful at raiding nests despite predator exclusion devices, managers may want to consider the tradeoff in egg loss (e.g., are complete or partial predation events more likely) compared to costs and resources of installing predator excluder devices (given they may only be successful for some of the predators).

The success of conservation initiatives should be evaluated both on the basis of resource improvement as well as from an economic perspective to allow conservation managers to justify value for financial investments and to provide ongoing protection within annual budgets. Beach protection (Dutton et al., 2005), nest relocation (Garcia et al., 2003) and, more prominently, predator control programs (Engeman et al., 2003, 2005, 2016) have been shown to be successful strategies in reducing nest predation rates of marine turtles. However, these strategies are costly and funding is rarely available on an ongoing basis. Predator exclusion from individual nests on the other hand is relatively cheap, has been commonly used, and has been demonstrated to be effective (Engeman et al., 2006; Ratnaswamy et al., 1997; Smith et al., 2013). Our work demonstrates that inexpensive meshing is effective for some predators, but it may not exclude all predators and in this study did not deter feral pigs which had historically caused the most damage.

In addition, it is unclear (but likely) if predators will learn over time how to access even protected nests by digging under or around nest exclusion devices, further highlighting the need for continual and adaptive management plans. We also demonstrate that the level of protection may not affect the population viability given that unprotected nests can have similar hatchling success rates. This emphasises the need for a thorough understanding of predator guilds and ecosystem processes, as well as integrating predator management and nest protection strategies to make conservation practices more efficient and effective in the future.

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