Final Report

Improving the estimates of abundance of dugongs and large immature and adult-sized green turtles in Western and Central Torres Strait

Rie Hagihara, Christophe Cleguer, Shane Preston, Susan Sobtzick, Mark Hamann, Takahiro Shimada and Helene Marsh
In collaboration with the Mura Badulgal Registered Native Title Bodies Corporate

Tropical Water Quality Hub
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College of Marine and Environmental Sciences, James Cook University

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ACRONYMS

AFMA............ Australian Fisheries Management Authority

AIC................ Akaike Information Criterion

DOE .... Department of the Environment

GBR............. Great Barrier Reef

GLMM........... Generalized Linear Mixed Models

IUU................ Illegal, Unreported and Unregulated

JCU............... James Cook University

LC .... Location Classes

NESP .... National Environmental Science Programme

PBR .... Potential Biological Removal

PNG .... Papua New Guinea

PZJA............. Protected Zone Joint Authority

QFP............... Quick Fix Pseudoranging

RRRC............ Reef and Rainforest Research Centre Limited

TSRA .... Torres Strait Regional Authority

TWQ............. Tropical Water Quality

ZINB............. Zero-inflated negative binomial

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EXECUTIVE SUMMARY

- This research adds to other fisheries independent evidence that the Torres Strait dugong harvest is sustainable. The Torres Strait dugong population is substantially higher than previously estimated because most dugongs in Torres Strait occur in water 5-20m deep where they spend much more time out of the sight of aerial observers than previously assumed. The most credible estimates of the number of dugongs that can be sustainably removed each year from Central and Western Torres Strait from all human causes is similar to the (outdated) estimates of catch.

- The status of the foraging green turtle population in Torres Strait is less certain than that of the dugong. The fisheries independent evidence is limited, especially given the mounting evidence of recruitment failure at Raine Island, the major rookery for the Northern Great Barrier Reef (GBR) green turtle stock. The most credible estimates of the number of large immature and adult –sized green turtles that can be sustainably removed each year from Central and Western Torres Strait from all human causes is close to the (outdated) catch estimates when the likely Papua New Guinea (PNG) harvest is considered.

Recommendations

1. That the major priority for dugong and turtle management in Torres Strait be on-going support for the implementation of community-based management.

2. That the Protected Zone Joint Authority give high priority to:
    - continuing negotiations with Traditional owners and PNG about extending spatial closures as a culturally acceptable and logistically achievable method of controlling the levels of harvest;
    - assisting PNG to finalise and implement its Turtle and Dugong Management plan; and
    - facilitating complementary management of dugongs and turtles across and within justifications, especially the Northern Peninsula Area and along the PNG coast;

3. That the TSRA give high priority to:
    - implementing a rigorous program to record the current dugong and turtle harvest from all the major hunting communities in Torres Strait;
    - sharing learnings from the catch monitoring process with the agencies responsible for managing the dugong and turtle harvest in the Great Barrier Reef World Heritage Area and PNG;
    - investigating the impacts of illegal, unreported and unregulated (IUU) fishing and shipping on dugongs and turtles and their habitats in Torres Strait; and
    - implementing the humane methods of killing green turtles developed in consultation with a veterinarian.

1. INTRODUCTION

Torres Strait is the most important dugong habitat of the dugong, *Dugong dugon*, in the world (Marsh *et al.* 2011) and a globally significant green turtle (*Chelonia mydas*) habitat (Limpus and Parmenter 1986, Miller and Limpus 1991, Limpus *et al.* 2003, Fuentes *et al.* 2015). Green turtles have been harvested by the indigenous peoples of Torres Strait for at least 7000 years (Wright 2011). Similarly, dugongs have been harvested for at least 4000 years (Crouch *et al.* 2007), possibly 7,000 years (Wright 2011). Archaeological evidence indicates that the dugong harvest has been substantial for at least the last 400 – 500 years (McNiven and Bedingfield 2008).

Like all other Traditional Owners in northern Australia, Torres Strait Islanders have the right to hunt dugongs and green turtles in their sea country under the Australian Native Title Act (e.g. *Native Title Act* 1993). Environmental laws (e.g. the Australian *Environment Protection and Biodiversity Conservation Act* 1999 (*EPBC Act*); and the Queensland (State) *Nature Conservation Act* 1992) do not affect their Native Title rights. The harvests of dugongs and green turtles in Torres Strait are also classified as traditional fisheries guaranteed by the Torres Strait Treaty between Australia and Papua New Guinea (PNG) (Havemann and Smith 2007) and are regulated by Australian and State (Queensland) fisheries laws (the *Torres Strait Fisheries Act* 1984 Commonwealth and the *Torres Strait Fisheries Act* 1984 Qld). There are some input controls. Hunting from vessels longer than 6 m is illegal. Animals can only be hunted using a traditional spear ('wap') and by custom, only males can hunt. Dugong hunting is banned from the Dugong Sanctuary, which is a >13,000 km^2 region in Western Torres Strait (Figure 1). These regulations are supplemented by another restriction - it is illegal to sell the meat of either species in the Australian communities; the sale of dugong meat is also banned in the 13 PNG Treaty villages and Daru, the capital of the South Fly district of the Western Province of PNG. The sale of turtle meat is also banned in Daru (but not in the Treaty villages).

In recent years, community-based management for dugongs and turtles has been strengthened in the Australian communities of Torres Strait by the Australian Government investing millions of dollars in indigenous ranger programs. Fifteen communities have developed Turtle and Dugong Hunting Management Plans (Marsh *et al.* 2011). The Islanders see this reinforcement of community-based management of their dugong and marine turtle fisheries as an important means of maintaining and revitalising their culture (Marsh *et al.* 2011). The harvest of the residents of the 13 Treaty Villages along the PNG coast and Daru is effectively unregulated apart from the ineffective (PNG Department of Environment and Conservation, in prep) ban on the sale of meat mentioned above. A management plan setting out objectives and management arrangements for this region of PNG is currently under development in response to concerns about: (1) the large numbers of dugongs and green turtles caught by Treaty villagers using long mesh nets; (2) overharvest leading to the illegal sale of dugong meat in the Daru market; and (3) disturbance from large commercial vessels anchoring in channels adjacent to the feeding grounds in Bistow and Daru Islands (PNG Department of Environment and Conservation, in prep). Like the plans developed by the Australian communities, this draft plan aims to enable the sustainable use of dugong and turtle resources through reinforcement of traditional values, cultural protocols and ethics (PNG Department of Environment and Conservation, in prep).

Today, both dugongs and green turtles are harvested largely for their meat. Thus the fisheries of the Torres Strait have significant provisioning value, particularly for residents of the Outer Islands in the Australian jurisdiction, where residents suffer the double burden of low incomes and high food prices (Delisle *et al.* 2014) and the Western Province in PNG, where the Human Development Index is very low (Butler *et al.* 2015) making this region one of the poorest in the World.

The cultural values of dugong and turtle hunting are central to the cultural identity of Torres Strait Islanders (Beckett 1987, Johannes and MacFarlane 1991). Both dugongs and green turtles are considered to be cultural keystone species (Butler *et al.* 2012). Delisle (2013) quantified the benefits and costs associated with traditional dugong and turtle hunting in two Australian hunting communities in the western Torres Strait. Community members identified a range of cultural services associated with hunting and rated these services as significantly more important than the provisioning services associated with eating dugong and turtle meat. The gross benefits of the harvest (including associated provisioning and cultural services) exceeded 16% of the household income of Torres Strait Islanders living on the Outer Islands (approximately the proportion of income spent by the average Australian on mortgage repayments). Some of the cultural values of the harvest are expressed through the sharing of meat between communities in the Torres Strait, including the PNG Treaty villages, as well as between Torres Strait Islanders and their diaspora on mainland Australia. Sharing occurs primarily through visits of diaspora members to the Torres Strait, residents of the Torres Strait visiting family on the mainland, and Torres Strait Islander ceremonies and gift exchange. Sharing contributes to the wellbeing of the diaspora by strengthening social relationships and reinforcing cultural identity (Delisle *et al.* 2014, Watkin *et al.* 2016a).

Contemporary estimates of the size of the Torres Strait dugong and green turtle harvests are not available. Marsh *et al.* (2004) summarised the available estimates of the catch of dugongs in various communities in the Torres Strait Protected Zone between the 1970s and the 1990s. The most accurate records are those of Kwan (2002) who lived at Mabuyag Island and recorded carcasses as they were butchered at traditional sites by members of that community in 1997 and 1999. Kwan's data reinforced the conclusions based on the CSIRO/AFMA catch surveys that the annual harvest by the Australian communities was substantial (e.g., 805 s.e. 241 for the Protected Zone in 1994, Harris *et al.* 1997). Harris *et al.* (2000) estimated that the catch of turtles on the Australian side of Torres Strait had remained about 3000 ± s.e.1000 from the mid-1970s through the 1990s. The data of Harris *et al.* (1997) suggested that about half of these animals would have been caught in our survey area (Figure 1) and that about 90% would have been the large immature and adult-sized turtles visible from an aircraft i.e., 1350 ± s.e.450 animals. In addition, Kwan (1991) documented the annual sale of 658-871 turtles in the Daru markets in 1985-1987; 83.4% (i.e., 549-726) of which were large female green turtles taken in our survey area. Thus these combined figures indicate that the annual harvest of large immature and adult-sized green turtles in Central and Western Torres Strait must have been >2000 animals per year in the late 20th century.

Because dugongs and green turtles are simultaneously cultural keystones and threatened species (Butler *et al.* 2012), these fisheries generate substantive cultural values beyond the Torres Strait region. The important ecosystem services they provide not only support the wellbeing of Torres Strait Islanders, but are also of high international conservation interest. Consequently as with many other marine wildlife harvests (Robards and Reeves 2011), dugong and green turtle hunting, particularly dugong hunting, is controversial in Australia. The issue was featured in recent Australian and Queensland elections (Delisle *et al.* 2014, Watkin *et al.* 2016b). Concerns about the sustainably of dugong hunting have been fuelled by scientific modelling making it increasingly difficult for scientists to access catch data. Heinsohn *et al.* (2004) predicted severe and imminent reductions in dugong numbers and median times for quasi-extinction ranging from 42 -123 years using: (1) Population Viability Analysis and published estimates on dugong life history and population sizes from systematic aerial surveys conducted from 1987 to 2001 (Marsh *et al.* 1997, 2004), and (2) simulated hunting rates ranging from 250 to 1000 dugongs per year. Using the same life history and aerial survey data, and the Potential Biological Removal method (Wade 1998), Marsh *et al.* (2004) also concluded that the current harvest must be unsustainable by estimating the annual sustainable anthropogenic mortality from all causes, a value which was much smaller than the incomplete harvest estimates then available.

We now know that the dugong population estimates on which the analyses of Heinsohn *et al.* (2004) and Marsh *et al.* (2004) were based were too low. Torres Strait is estimated to contain between 13,425 km^2 (Coles *et al.* 2003) and 17,500 km^2 (Poiner and Peterkin 1996) of seagrass habitat, including the largest single continuous seagrass meadow in Australia (Taylor *et al.* 2010) incorporated within the Dugong Sanctuary (Figure 1). When the aerial survey area was extended in response to this finding, the extension was estimated to support more than 1000 dugongs (Marsh *et al.* 2011). In addition, Hagihara *et al.* (2014) reported that the availability of dugongs to aerial observers depends not only on environmental conditions (Pollock *et al.* 2006) but also bathymetry, a factor that has not been included in the aerial survey estimates for Torres Strait to date. Dugongs in waters 5-25 m deep (the depths where most dugongs are sighted in Torres Strait; Sobtzick *et al.* 2014) are less available to aerial observers than animals in shallower or deeper waters. Thus it is possible that the historical abundance estimates are negatively biased.

Marsh *et al.* (2015) used several lines of evidence to re-evaluate the sustainability of the Torres Strait dugong harvest. Their evidence suggested that the harvest is sustainable. Dugong relative density was significantly higher in 2013 than in any other survey year and their index of Area of Occupancy has trended slightly upward since 1987. The proportion of calves in 2013 was the highest recorded. Genetic diversity is high. Dugongs are caught in only 5.0% of the 5,268 km^2 of very high dugong density habitat as the result of the controls on the harvest and socio-economic factors. Nonetheless, this assessment was compromised by the absence of robust data on the absolute size of the dugong population or the harvest.

Fuentes *et al.* (2015) estimated the population of large immature and adult-sized turtles in Western and Central Torres Strait based on sightings from the 2013 dugong survey by:

(1) correcting for perception bias following the method of Pollock *et al.* (2006);

(2) developing correction factors to compensate for availability bias at the level of individual sighting by:

  (a) conducting experimental trials with a marine turtle Secchi Disk, to identify the depth of detection zones below the water surface where turtles are visible to aerial observers under different environmental conditions; and

  (b) estimating the proportion of time that turtles spend in these detection zones by analysing Time-Depth Recorder data from devices deployed on free-living turtles external to Torres Strait; and

(3) applying the resultant correction factors to aerial survey counts to improve abundance estimates. Their resultant estimate was 617,209 (± s.e. 83,717) large immature and adult-sized turtles of all species, most of which were presumably green turtles.

The objectives of our study were to inform the Indigenous management of dugongs and green turtles in Torres Strait, particularly the management of Mura Badulgal Sea Country by the Mura Badulgal Representative Native Title Body Corporate by:

- collecting movement and dive data from dugongs and green turtles caught with the assistance of members of the Badu Community and TSRA rangers and fitted with: (1) satellite tracking devices that recorded their two-dimensional space use, and (2) MiniPAT pop-up archival tags that recorded their depth use;
- using the resultant behavioural data to improve the estimates of the population size of both dugongs and green turtles in Torres Strait from aerial surveys conducted in 2006, 2011 and 2013; and
- Re-estimating the size of sustainable anthropogenic mortality of both dugongs and large immature and adult-sized female green turtles from all anthropogenic causes.

2. METHODS

2.1 Aerial survey data

The size of the populations of dugongs and large immature and adult-sized green turtles in Western and Central Torres Strait size was re-estimated using aerial survey data collected in 2006, 2011 and 2013 in the Torres Strait region (Figure 1). These surveys collected information on environmental conditions (e.g., water visibility) at the time of each animal sighting. This information was necessary for estimating animal abundance using the Hagihara methodology as described in Section 2.3. The data are available at https://dugongs.tropicaldatahub.org.

The 2006 and 2013 surveys were conducted in November. The 2011 survey was completed in March due to unsuitable weather conditions in the previous November. The design of all surveys was based on stratified random sampling (Figure 1). The survey region in 2006 encompassed eight blocks located between Cape York Peninsula and Papua New Guinea (blocks 0, 1A, 1B, 2A, 2B, 3, 4 and 5; Figure 1). The later surveys (2011 and 2013) covered additional area in Western Torres Strait (blocks 6, 7, 8 and 9). This extension included most of the Dugong Sanctuary (Figure 1). In 2006, the transects in blocks 0, 1A and 1B extended to the coastline of Papua New Guinea (PNG), but from the 2011 survey onwards the transects were truncated 5 nm from the coast as a result of additional restrictions on Australian light aircraft flying in PNG airspace. Our analyses adjusted for these differences in survey design.

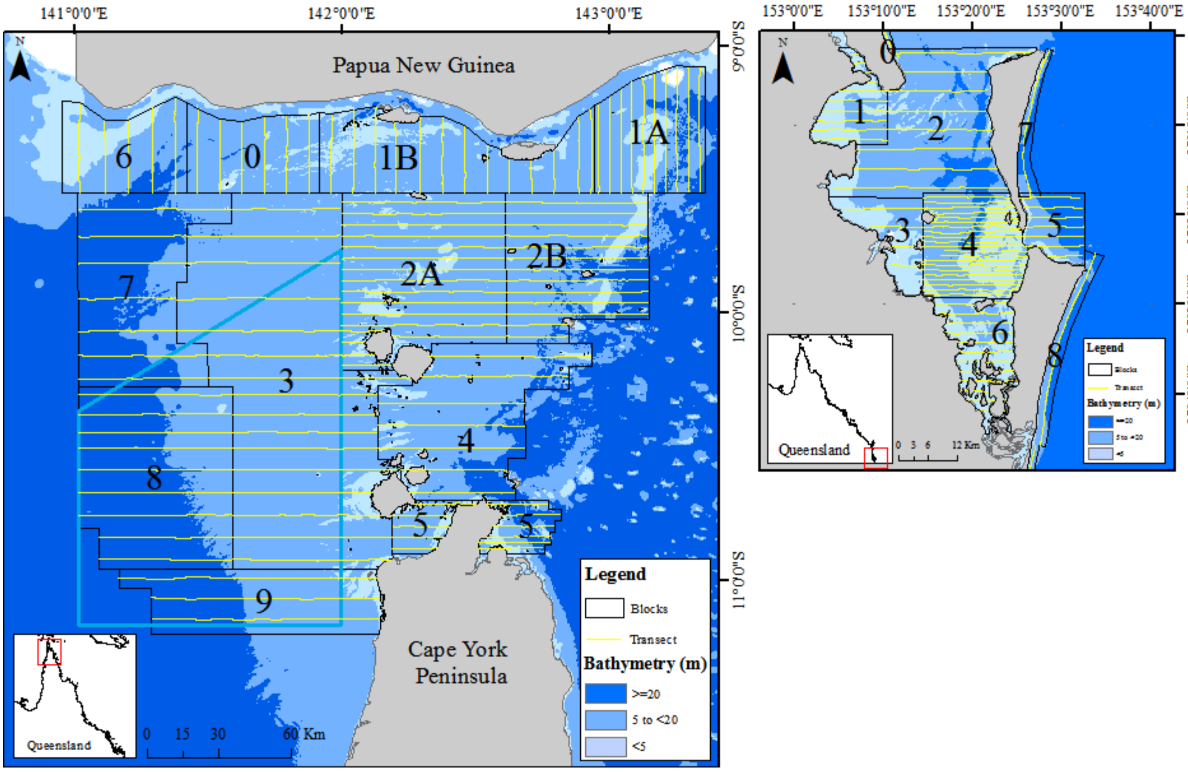

Figure 1: Left map: the Torres Strait aerial survey region showing the survey blocks, the transect lines flown within each block (yellow) including the truncated transects near the Papua New Guinea coast and the Dugong Protection Area (light blue). The bathymetry of Torres Strait is also shown and contrasted with the map on the right, which shows the bathymetry of the Moreton Bay aerial survey area in southeast Queensland, an area more typical of coastal habitats where a high proportion of dugongs and turtles are sighted in water <5m deep.

2.2 Aerial survey

The aerial survey methodology was based on Marsh and Sinclair (1989a,b), as improved by Pollock *et al.* (2006). All surveys were conducted from a 6-seater Partenavia 68B. Flight height in 2006 was 137 m (450 ft); 152 m ≈ 500 ft in 2011 and 2013. The experimental work of Marsh and Sinclair (1989b) indicates that this small difference in survey height should not make a substantive difference to the capacity of observers detecting animals. The transect widths were the same for all surveys. The aircraft was always flown as close as possible to a ground speed of 100 knots.

Trained tandem teams of two observers; one team on each side of the aircraft, scanned strip transects 200 m wide on the water surface. Each transect was demarcated using fiberglass rods attached to artificial aircraft wing struts. Distance categories (low=50 m, medium=100 m, high=150 m, and very high 150-200 m) within the strip were marked by colour bands on each artificial wing strut. The members of each tandem team of two observers sitting in the middle and rear seats on each side of the aircraft were visually and acoustically isolated and reported their sightings into separate tracks of an audio recorder. The distance categories of each sighting within the strip enabled the survey team to decide if simultaneous sightings by tandem team members were of the same group of animals when reviewing the recordings after each day's survey. This information was used to estimate perception bias (see Section 2.5). As explained by Pollock *et al.* (2006), although we found no decline in detection with distance across the strip, there was a large amount of measurement error in the assignment of sightings to distance classes within the transect strip. This problem was particularly challenging for dugongs which surface cryptically and for only 1–2 seconds (Anderson and Birtles 1978, Chilvers *et al.* 2004). The cryptic nature of dugong surfacing and the often high sighting rate of both species meant that observers could not afford to take their eyes off the water to read an inclinometer. Thus following Pollock *et al.* (2006), we decided not to use distance category as a co-variate in the analyses.

The surveys were conducted in passing mode. For each sighting, the observers recorded the total number of animals seen, number at the surface of the water, position in the transect sub-strip (e.g*.,* low or medium). The number of dugong calves (animals less than 2/3 of the size of the adult dugong and swimming in close proximity), was also recorded for dugong sightings. On three occasions, once in 2006 and twice in 2013, a group of dugongs was sighted that were too large to accurately count in passing mode ($\geq$10 animals). The aircraft discontinued flying the transect and went into circling mode in an effort to obtain a total count of the group before resuming the transect.

The survey leader seated next to the pilot collected data on environmental conditions at the beginning of each flight (cloud cover, cloud height, wind speed and direction, and air visibility) and each transect (cloud cover). There was a strict ceiling on weather: no precipitation and sea state $\leq$3. Every few minutes during each transect, and whenever conditions changed, the survey leader recorded sea state, water transparency, and glare (none; 0 to <25% of field of view affected; 25 to <50% affected, >50% affected) on each side of the aircraft (the last was assessed by the mid-seat observers).

2.3 Estimating abundance

The sizes of the populations of dugongs and all species of large immature and adult-sized turtles were estimated by correcting the sightings for availability bias (animals that are not visible to observers), perception bias (animals that are visible but missed by observers) and proportion of the area surveyed. The size of the population of green turtles was subsequently estimated by correcting the estimates based on turtle species sightings from helicopter surveys as outlined in Section 2.7 below.

We estimated the abundance of dugongs in Central and Western Torres Strait using the methodology of Hagihara *et al.* (2014) (hereafter the Hagihara method). This method accounts for the effects of water visibility and the changes in the diving patterns of the target species with water depth. This methodology is an improvement of the methodology developed by Pollock *et al.* (2006) (hereafter Pollock method) that accounts for the effects of water visibility and sea state but assumes that the time dugongs spend at or near the surface is homogeneous across water depths, an assumption that the Hagihara method has shown to be incorrect. Fuentes *et al.* (2015) (hereafter Fuentes method) adapted the Pollock method to estimate the population abundance of large immature and adult turtles in Central and Western Torres Strait by assuming that turtle diving behaviour was independent of water depth and we have also used the data collected in this study to improve their estimates of the abundance of large immature and adult green turtles in that region.

2.4 Availability bias

Availability detection probability requires: 1) estimates of the Detection Zone, the depth of water column in which animals are available for detection under defined environmental conditions; and 2) estimates of the proportion of time animals spend in that Detection Zone under such conditions.

We used the Detection Zones estimated by Sobtzick *et al.* (2015), who used finer resolution depth recorders than Pollock *et al.* (2006) (Table 1) on dugong and turtle secchi disks, 2-dimensional models that mimicked the shape of dugongs and turtles as seen by aerial observers under a range of environmental conditions. Environmental Conditions Index (ECI) is a function of water turbidity and sea state, and water depth as defined by Sobtzick *et al.* (2015) and summarised in Table 1.

Table 1: Estimates of Detection Zones for each Environmental Conditions Index (ECI) *sensu* Sobtzick *et al.* (2015).

Environmental Conditions Index (ECI)	In-water visibility	Depth range	Dugong		Green turtle	
			Average Secchi Disk depths ± SE	Detection Zone (m)	Average Secchi Disk depths ± SE	Detection Zone (m)
1	Clear	Shallow	n/a	all	n/a	all
2	Variable	Variable	2.07 ± 0.50	0 to 2.0	1.13 ± 0.63	0 to 1.0
3	Clear	Deep	3.45 ± 0.59	0 to 3.5	2.29 ± 0.73	0 to 2.5
4	Turbid	Variable	1.59 ± 0.70	0 to 1.5	0.67 ± 0.53	0 to 1.0

n/a: Sobtzick *et al.* (2015) did not conduct their Secchi Disk experiment for Environmental Conditions Index 1 because all animals were assumed to be available for detection under that condition.

The estimates of the proportion of time animals spend in each Detection Zone at various water depths were calculated from wild dugongs and green turtles captured in Torres Strait and fitted with the following devices as described in Section 2.4.1 below:

(1) Dugongs: a GPS (Global Positioning System)/Argos Systems unit (Telonics, Inpala, USA) and a pop-up archival tag MiniPAT (Wildlife Computers, Redmond, USA).;

(2) Green turtles: Argos-linked either a: (1) GPS SPLASH10-F-296A System unit (Wildlife Computers, Redmond, USA) or (2) a FastlockTM System unit F4G 676A (Sirtrack, Havelock North, New Zealand) and a pop-up archival tag MiniPAT (Wildlife Computers, Redmond, USA).

2.4.1 Animal tracking

We captured 10 dugongs and nine turtles near Badu Island in October 2015 as detailed in Cleguer *et al.* (2016). All animals were handled in strict accordance with local, state, national and international regulations. The field work was conducted under JCU Animal Ethics Approvals from JCU (A2072), Commonwealth Scientific Purpose Permit E2014/0091 and

Queensland Scientific Purpose Permit *WISP15058214* and the *Permit for Scientific Purposes* obtained from Torres Strait Regional Authority under the *Torres Strait Fisheries Act 1984*.

2.4.1.1 Dugongs

We captured the 10 dugongs using the Fuentes *et al.* (2013) technique. Each animal was fitted with a GPS/Argos Systems unit (Telonics, Inpala, USA) which generates GPS, QFP (Quick Fix Pseudoranging) and Argos location uplinks (Cleguer 2015). The transmitter was set to acquire GPS locations hourly. Table 2 provides details of the individual animals and tracking periods.

GPS and satellite data were retrieved from the Argos web site and decoded using software supplied by the manufacturers. We selected location data with higher quality indicators GPS (± 2 to <10 m), resolved QFP (± <75 m) and three Argos Location Classes (LC): LC3 (± <250 m), LC2 (± 250 to <500 m) and LC1 (± 500 to <1500 m). These data were then filtered using *SDLfilter* (Shimada *et al.* 2012, 2016) in *R 3.1.3* (R Development Core Team 2015). This process removed location points that are spatially or temporally duplicated, or that are highly unlikely given the individual's travel speed and turning angle (Shimada *et al.* 2012, Gredzens *et al.* 2014, Cleguer 2015). The lower quality fixes (LC2 and LC1) were removed after the filtration.

Dive records archived by each MiniPAT pop-up tag were transmitted to Argos satellites upon the tag's release from the tagged animal. Each tag was programmed to release after 60 days of deployment. The collected dive records were retrieved via the manufacture's portal to which the data were sent from Argos and stored upon tag release and subsequent transmission. Decoding of the depth records was performed in the portal. Eight tags stayed on dugongs till the end of intended deployment period; two tags were released prematurely about one week and one month after the animals were captured (Table 2). Although dive records were collected from all 10 dugongs, location data were not collected from four dugongs due to failure of the GPS/Argos Systems unit or breakage of the weak-link in the attachment tether (Cleguer *et al.* 2016). We examined the data collected from the dugongs that provided a complete set of location fixes and dive records.

To estimate the time dugongs spent in various Detection Zones, location data and dive records for each dugong were combined using information on record time and date and the software *DepthMatcher* (R. Jones 2013). Dive records were extracted within 10 min of each satellite fix to estimate the availability correction factors (Hagihara *et al.* 2014; Section 2.4.2 in this report). This process assumed that the water depth was constant for that 10 min period. By extracting data from six dugongs and across a range of sea states, we assumed that the resultant estimates of the time dugong spent in Detection Zones of various depths were representative of the conditions encountered during the aerial surveys.

2.4.1.2 Green turtles

All nine turtles were captured using the rodeo technique (Limpus 1978) and equipped with tracking units following Shimada *et al.* (2012). All captured turtles were kept in cool conditions at the Mura Badulgal Ranger Station on Badu Island during the process of attaching the GPS-satellite transmitters. The captured turtles were brought to land, and satellite and pop-up tags were attached to carapace using Sika (®Anchor Fix 3) two-part epoxy and fibreglass. The satellite unit was attached high on the carapace (approximately first vertebral scute) to increase the likelihood of satellite fixes. Seven of the nine green turtles each carried a pop-up tag which was attached to the lateral side of the carapace, approximately 50 cm apart from a FGPS tag to avoid potential interference in transmission and data recording. Each turtle was released the morning after it was captured when the epoxy had set. The GPS-satellite unit was programmed to acquire GPS locations every 30 min. Table 2 provides details of individual turtles and tracking periods.

The GPS-satellite data were retrieved from the Argos web site and decoded using software supplied by manufacturers. Six out of the seven MiniPATs fitted to the turtles were prematurely released (Table 2). While one MiniPAT stayed on a turtle for the entire deployment period, the satellite transmitter attached to this turtle failed and no location data were recovered. Sets of location and dive records collected from six turtles were examined further. The time turtles spent in various Detection Zones was estimated using the techniques similar to those outlined for dugongs above.

Table 2: Details of dugongs and green turtles caught in Torres Strait in October 2015 and fitted with satellite transmitters and MiniPAT pop-up archival tags.

Individual ID	**Sex**	**Length (cm)**	**Capture date**	**Depth at capture location (m)**	**MiniPAT tracking time (days)**	**Argos/GPS transmitter tracking time (days)**
Dugongs						
D1	M	260	7/10/2015	13.2	61*	No data
D2	M	210	8/10/2015	7.1	61	57
D3	M	230	9/10/2015	4.8	53*	No data
D4	M	240	10/10/2015	2.8	61	20
D5	M	270	12/10/2015	4.9	4	57
D6	M	240	12/10/2015	8.8	61	46
D7	M	210	12/10/2015	12.9	61	77
D8	M	280	14/10/2015	7.9	60**	No data
D9	M	270	14/10/2015	4.9	61	20
D10	M	250	15/10/2015	4.7	30*	No data
Green turtles						
T1	F	90.7	10/10/2015	3.9	24	112
T2	F	109.1	13/10/2015	3.1	8	133***
T3	M	99.6	15/10/2015	3.1	not deployed	109
T4	F	107.5	15/10/2015	2.8	40	130
T5	F	106.7	15/10/2015	2.0	not deployed	121***
T6	F	110.5	15/10/2015	2.0	26	123
T7	F	95.5	15/10/2015	2.4	29	104
T8	F	107.0	15/10/2015	2.9	5	130***
T9	F	98.9	15/10/2015	3.0	48*	No data

*satellite transmitter failed to transmit signals, dive records recovered but not used in analyses; **satellite transmitter was released from the dugong, dive records were recovered but not used ;*** satellite transmitter still attached to turtle and tracking at time of analysis on February 22 2016.

2.4.2 Availability detection probability

Availability detection probabilities for each Environmental Conditions Index (ECI) were estimated using separate Generalized Linear Mixed Models (GLMMs) for dugongs and green turtles assuming a binomial distribution (Hagihara *et al.* 2014). The response variable was the presence/absence of dugongs or green turtles in each Detection Zone. Water depth was the single explanatory variable with three depth categories: 1) water <5 m deep; 2) water 5 to <20 m deep; and 3) water ≥20m (see Figure 1). Individual animal was treated as a random factor in the models.

Standard errors of the availability detection probabilities were estimated using the delta method, which approximates the variance on the probability scale, as in Hagihara *et al.* (2014). GLMMs were performed using the *lme4* package (lme4_1.1-7, Bates *et al.* 2012) in *R*

3.1.3 (R Development Core Team 2015). The standard error estimates were incorporated into the estimates of the standard errors of population abundance estimates using the simulation technique described in Section 2.6.

2.5 Perception bias

Perception bias was estimated in Mark-Recapture framework following Pollock *et al.* (2006) using program *Mark* (version 8.0, White 2014). We used the information from the following three sighting categories separately for: (1) dugongs and turtles (all species), and (2) both the port and starboard observer teams for each survey team within each survey: a) sighted by a mid-seat observer only; b) sighted by a rear-seat observer only; and c) sighted by both observers. We chose a Lincoln-Petersen method for a closed population and two visits (sightings by two observers) to the study area. Four scenarios (models) were examined: (1) all observers have the same perception detection probability; (2) mid-seat observers have the same detection probability and rear-seat observers have the same detection probability; (3) observers on the same side have the same detection probability; and (4) all observers have different detection probabilities. The best model was selected based on the Akaike Information Criterion (AIC). The perception detection probability for port or starboard side was estimated as:

$$\hat{p}_d = 1 - (1 - \hat{p}_1)(1 - \hat{p}_2)$$

where $\hat{p}_d$ is the combined perception probability for either port or starboard side, and $\hat{p}_1$ is the perception probability for the mid-seat observer, and $\hat{p}_2$ for the rear-seat observer.

2.6 Abundance estimation

Following Pollock *et al.* (2006), population abundance was estimated using the following detection probability model:

$$\hat{p}_j = p_b \hat{p}_{aj} \hat{p}_{dj}$$

where $\hat{p}_j$ is the probability of detection for animal j , $\hat{p}_{aj}$ is the probability that animal j was available for detection, and $\hat{p}_{dj}$ is the probability that the individual was detected given it is available for detection. p_b is the probability of sampling a transect strip in block b which is the proportion of the area sampled in that block. This probability ($\hat{p}_j$) was used in the Horvitz-Thompson estimator (Horvitz and Thompson 1952) to estimate the dugong and turtle population sizes as follows:

$$\hat{N} = \sum^{n} [1/\hat{p}_j]$$

Where $\hat{N}$ is the estimated population size for the whole survey region, and n is the number of distinct dugongs and turtles spotted in the whole survey region.

All dugongs sighted in the three herds of ≥10 animals were assumed to be counted and bias corrections were not applied to these sightings. As these groups appeared small (10-15 animals) failure to count all the dugongs in these groups should make only a trivial difference to the results.

Population size was estimated for each block in which ≥5 dugongs or turtles were recorded. Standard errors of the estimated population abundance were generated in Python (version 2.7.6) using a Monte Carlo simulation method with 1,000 iterations (based on Pollock *et al.* 2006).

2.7 Green turtle species and sex composition

To estimate the number of green turtles in Torres Strait, we determined the marine turtle species composition using helicopter surveys. Three species of marine turtles nest and forage in Torres Strait: green turtles (*Chelonia mydas*), hawksbill turtles (*Eretmochelys imbricata*) and flatback (*Natator depressus*) (Miller and Limpus 1991). Three other species (loggerhead turtles, *Caretta caretta*; olive ridley turtle, *Lepidochelys olivacea*; leatherback turtle, *Dermochelys coriacea*) also occur in the region (Fuentes *et al.* 2015).

Two helicopter-based flights were taken over Central Torres Strait on February 12 2016. Both flights were conducted with a single observer (Mark Hamann), who has considerable expertise in the identification of sea turtle species. Flights departed Horn Island airport and flew at 90 m (300 feet) and 80 knots. Flight 1 travelled along the western edge of the Orman Reefs as north as Turn-Again Cay and then south to Mabuyag and Badu and Dollar Reef. Flight 2 travelled north along the eastern side of the Orman Reefs until the northern extent of the reefs, then south to Mabuyag and along the western coast of Mabuyag and Badu. Only turtles of approximate adult size were counted. Flight 1 was conducted between 0845 and 1030; Flight 2 between 1430 and 1615. Whenever possible, each turtle sighted was identified to species.

We estimated the number of turtles sighted during the large-scale aerial surveys that were female green turtles based on the proportion of sightings identified to species that were classified as green turtles during the helicopter flights. The sex ratio of large immature and adult-sized green turtles in Torres Strait has not been published. However, in two capture-mark-recapture trips in 2008 and 2009, the ratio was calculated as 6 (female): 1 (male) (pers.comm, Mark Hamann). Published data from southern Queensland indicate the ratio is 3 (female): 1 (male) in Shoalwater Bay and Moreton Bay, and variable across age classes and years at Heron Island – juvenile and sub-adult (>65 cm) were significantly female biased and adults were slightly male biased (Chaloupka and Limpus 2001, Limpus *et al.* 2005). We therefore applied this 3:1 ratio to the population estimates of green turtles to calculate the number of large female juvenile and adult green turtles encountered during aerial surveys in Central and Western Torres Strait. This estimate of large female juvenile and adult green turtles was used to calculate the size of the sustainable female green turtle harvest in Central and Western Torres Strait using the Potential Biological Removal (PBR) method (refer Section 2.9).

2.8 Population trends

2.8.1. Dugongs

To determine the significance of the temporal and spatial variation in the number of dugongs sighted in the three surveys, we used a zero-inflated negative binomial (ZINB) model because of the large number of transects with no dugong sightings (47%). Exploratory analysis based on the saturated model (year, block and the interaction of year and block) based on AIC showed that a ZINB model was the best fit among the four models examined (Poisson, negative binomial, zero-inflated Poisson and ZINB). Explanatory variables in both count and zero components were year (2006, 2011 and 2013), block and the interaction of year and block. The response variable was the number of dugongs per transect corrected for the availability and perception biases. The log transformed transect length (km) was used as an offset in the count component.

The 2011 and 2013 surveys covered all blocks while the 2006 survey covered blocks 0 to 5 only (Figure 1). Consequently, two separate ZINB models were examined: (1) all three years with blocks 6, 7, 8 and 9 removed for 2011 and 2013; (2) two years (2011 and 2013) and all blocks (excluding blocks 6 and 7 where no dugongs were sighted in both years). The saturated model was reduced using the model selection based on AIC. The statistical analysis was performed in *pscl* (ver.1.4.8, Jackman 2015) in *R* (R Core Team 2015).

2.8.2. Turtles

As for dugongs, temporal and spatial variations in the number of turtle sightings were examined using a zero-inflated model. The exploratory analysis showed that the percentage of transects that contained no turtle sightings was smaller (11%) than for dugongs (47%). Nonetheless, exploratory analysis based on the saturated model (year, block and the interaction of year and block) showed that a ZINB model was the best fit (based on AIC) among the four models examined (Poisson, negative binomial, zero-inflated Poisson and ZINB). Explanatory variables in both count and zero components were year (2006, 2011 and 2013), block and the interaction of year and block. The response variable was the number of turtles (all species and sexes) per transect corrected for the availability and perception biases. The log transformed transect length (km) was used as an offset in the count component.

2.9 Potential Biological Removal (PBR)

Although the Potential Biological Removal (PBR) method was developed to estimate sustainable human-caused mortality limits for marine mammals (Wade 1998), this approach has also been used by Casale and Heppell (2016) to evaluate the anthropogenic mortality of green and loggerhead turtles from fishing bycatch using information on time-series nesting abundance (Chaloupka *et al.* 2008). Thus we calculated PBR values for both dugongs and harvestable green turtles in Central and Western Torres Strait as outlined below.

This conservative technique estimates the anthropogenic mortality that should enable the population to reach or exceed maximum net productivity without depletion. PBR is calculated as:

$$PBR = N_{min} \times \frac{1}{2} R_{max} \times F_R$$

where N_{min} is the minimum population estimate of the population, ½ R_{max} is one half the maximum theoretical or estimated net productivity rate of the population and F_R is a safety factor to account for additional uncertainties other than the precision of the abundance estimate (e.g., R_{max}) (Wade 1998) and ranges between 0.1 and 1.

N_{min} accounts for uncertainties in the precision of abundance estimate and is calculated as the 20th percentile log-normal distribution as below:

$$N_{min} = \frac{\hat{N}}{\exp(z\sqrt{\ln(1 + CV(N)^2)})}$$

where $\hat{N}$ is the abundance estimate, z is the standard normal variate and is replaced by 0.842 for the 20th percentile under the log-normal distribution (Wade 1998). *CV(N)* is the coefficient of variation of the abundance estimate.

2.9.1 Dugongs

R_{max} requires information on life history parameters (e.g., age of first calving, mean calving interval, adult survivorship). Kwan (2002) estimated the first two of these parameters for Torres Strait dugongs from carcass analysis conducted in 1997, 1998 and 1999. However, there is no contemporary information on any of the dugong life history parameters for Torres Strait. Accordingly, we adjusted for this uncertainty using R_{max} of 0.03 (calving intervals of ca. 3 years) and 0.05 (calving intervals of ca. 2.5 years) following Marsh *et al.* (2004). The estimates of growth rate used by Marsh *et al.* (2004) were based on the survivorship of the Florida manatee (*Trichechus manatus latirostris*) (dependent calves = 0.822 per annum (p.a.); independent young = 0.965 p.a.; reproductive adult = 0.965 p.a.; Boyd *et al.* 1999). Marsh *et al.* (2015) used several lines of evidence to re-evaluate the sustainability of the Torres Strait dugong harvest in the absence of robust data on the absolute size of this dugong population or the harvest. Their evidence suggests that the harvest is sustainable. In calculating the PBR, the US National Marine Fisheries Service has used values for F_R of 0.1 for endangered species, 0.5 for threatened stocks or stocks of unknown status, and 1.0 for secure stocks. The dugong is listed as Vulnerable by the IUCN (2016) and in Queensland but is not listed as threatened at the scale of Australia. Thus we used values of F_R of 0.5 and 1.0.

2.9.2 Turtles

Female green turtles that use Torres Strait as a feeding ground predominantly nest in Torres Strait and the northern Great Barrier Reef, in particular Raine Island (Limpus *et al.* 2003, Jensen *et al.* 2016). The maximum population rate of change reported from Torres Strait and northern Great Barrier Reef nesting population is not known. The green turtle growth rate estimated from >25 years of nesting surveys in the neighbouring green turtle population in the southern Great Barrier Reef was 3.8% (Chaloupka *et al.* 2008). As explained above the appropriate recovery factor (R_{max}) differs according to the status of the population. Although green turtles are listed as endangered at a global scale (IUCN 2016); they are listed as Vulnerable in Australia and in Queensland. We calculated the PBR for three values of R_{max} = 0.1, 0.5 and 1 (Wade 1998) and three values of F_R = 0.1, 0.5 (default value) and 1.

2.10 Repatriation of results

Helene Marsh visited Badu in April 2016 to repatriate the results of the project to the Mura Badulgal Representative Native Title Body Corporate and to obtain permission to release this report..

3. RESULTS

3.1 Dugongs

3.1.1 Availability detection probabilities

In Torres Strait, availability detection probabilities for dugongs were lowest when the animals were in water 5 to <20 m deep for all the environmental conditions under which the surveys were conducted. (Figure 2 and Appendix I). The detection probability in water <5 m deep was slightly lower than that calculated using the Pollock *et al.* (2006) for ECI 2 and ECI 4 and substantially higher for ECI 3. The availability detection probability for water exceeding 20 m deep was not estimated and was conservatively assumed to be 1, because no satellite fixes were collected from tracked dugongs in this category. Given the low number of dugong sighted during the aerial surveys in this depth stratum (Figure 3), our failure to correct for availability bias for sightings in this stratum must make only a trivial difference to the results.

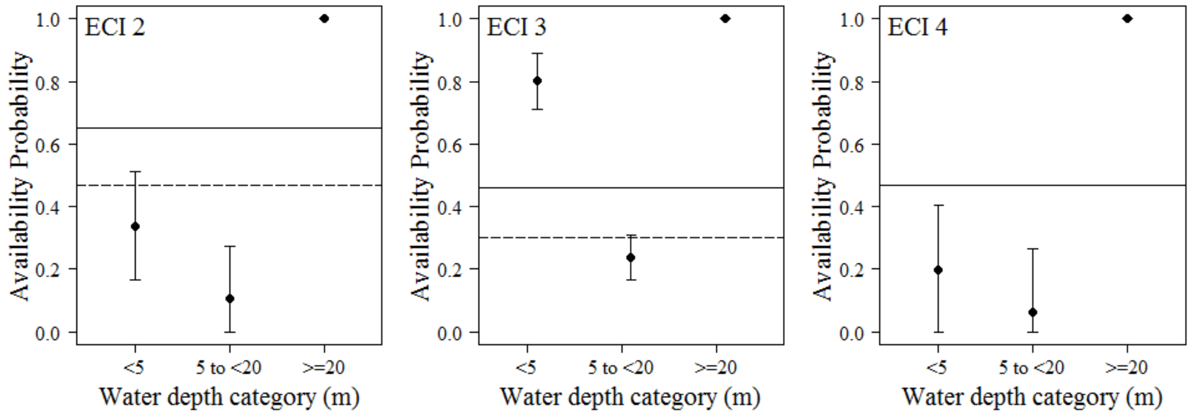

Figure 2: Availability detection probabilities and standard errors (vertical lines) estimated from dugongs tracked in Torres Strait under various levels of the Environmental Conditions Index (ECI). Horizontal lines represent availability estimates from Pollock *et al.* (2006) for optimal sea state (solid lines) and marginal sea state (dotted lines). The value for water $\geq$20 m was assumed to be one as no data were obtained from tracked dugongs in this deep water. Note the solid and dotted lines on the figure on the right (ECI4) overlapped and the dotted line is not visible.

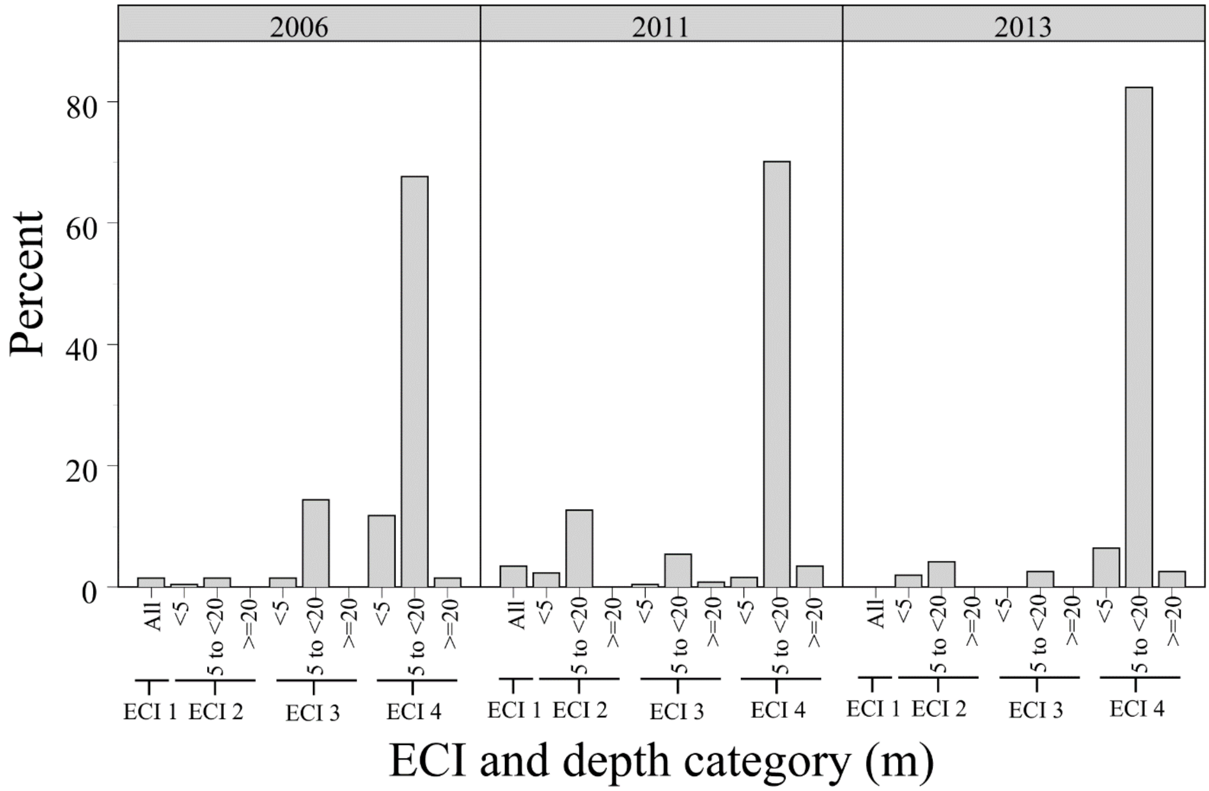

Figure 3: Percentage of dugong sightings recorded in each Environmental Conditions Index (ECI) and depth categories.

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3.1.2 Population size estimates

The estimated dugong population sizes using the Hagihara method were substantially higher than those using Pollock method (Table 3, Figure 4 and Appendix II) because most of the dugong sightings (84% in 2006, 88% in 2011 and 89% in 2013; Figure 3) were in water 5 to <20 m deep for which the estimates of availability bias estimates were much lower than those used by Pollock *et al.* (2006) (Figure 2). The coefficients of variation (CV) of the population sizes were very similar to those estimated using the Pollock method (Table 3).

Table 3: Comparison of the Coefficients of Variation (CV) of the dugong population abundance estimates for Central and Western Torres Strait based on the aerial surveys conducted in 2006, 2011, and 2013. Ratio of the abundance estimates using the Hagihara method (numerator) to the corresponding estimate using the Pollock method (denominator).

Year	CV		Ratio
	Pollock method	Hagihara method	
2006	0.16	0.16	5.72
2011	0.17	0.18	6.62
2013	0.19	0.20	6.52

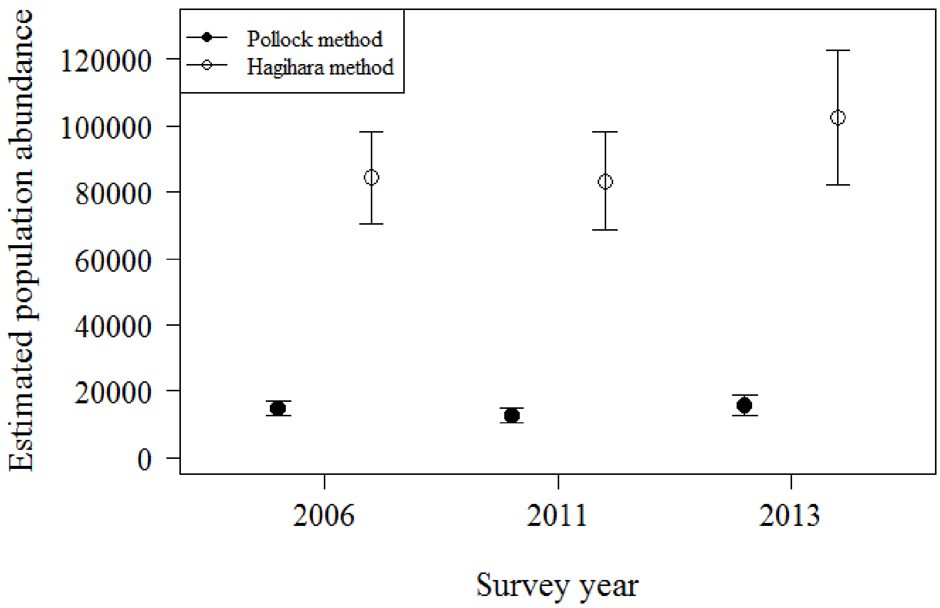

Figure 4: Estimates of dugong abundance and their standard errors for the aerial surveys conducted in 2006, 20011 and 2013 using the Pollock (closed circle) and Hagihara (open circle) methodologies. Note the aerial survey in 2006 covered a smaller areas than the 2011 and 2013 surveys.

3.1.3 Population trends

The corrected number of dugongs did not differ among years for the blocks (0-5 inclusive) that were surveyed in 2006, 2011 and 2013 (Figure 5). We also found no difference in the corrected number of dugongs between 2011 and 2013 (Figure 6). However, there was a significant difference in the number of dugongs among blocks. Block 2A always had the highest corrected number of dugongs, followed by block 1B and block 3 (Appendix IV and V). No yearly effect was found in the zero component. Block 2A had the lowest number of transects in which no dugongs were sighted; block 5 had the largest number of transects with zero dugongs (Appendix IV and V). No dugongs were sighted in blocks 6 and 7 in either 2011 or 2013.

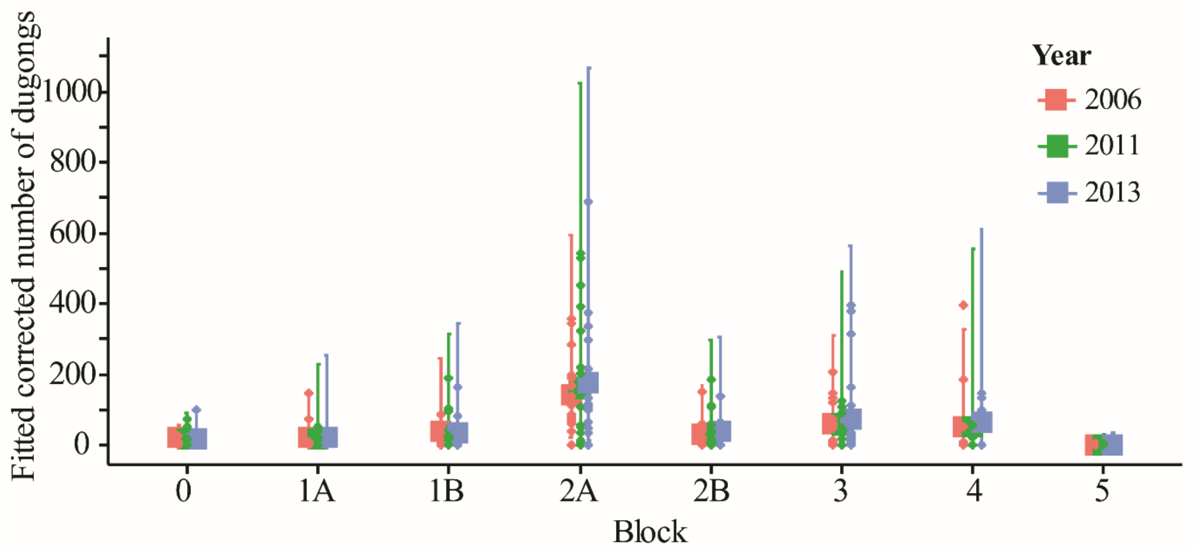

Figure 5: Fitted corrected number of dugongs per transect from a zero-inflated negative binomial model for the data collected in the aerial surveys conducted in 2006, 2011 and 2013 from blocks 0, 1A, 1B, 2A, 2B, 3, 4 and 5. Squares represent mean fitted values, and lines represent 95% confidence intervals. Dots are the observed corrected number of dugongs. The confidence intervals were estimated from the saturated model. The mean transect length of each block was used to calculate the fitted values for each block.

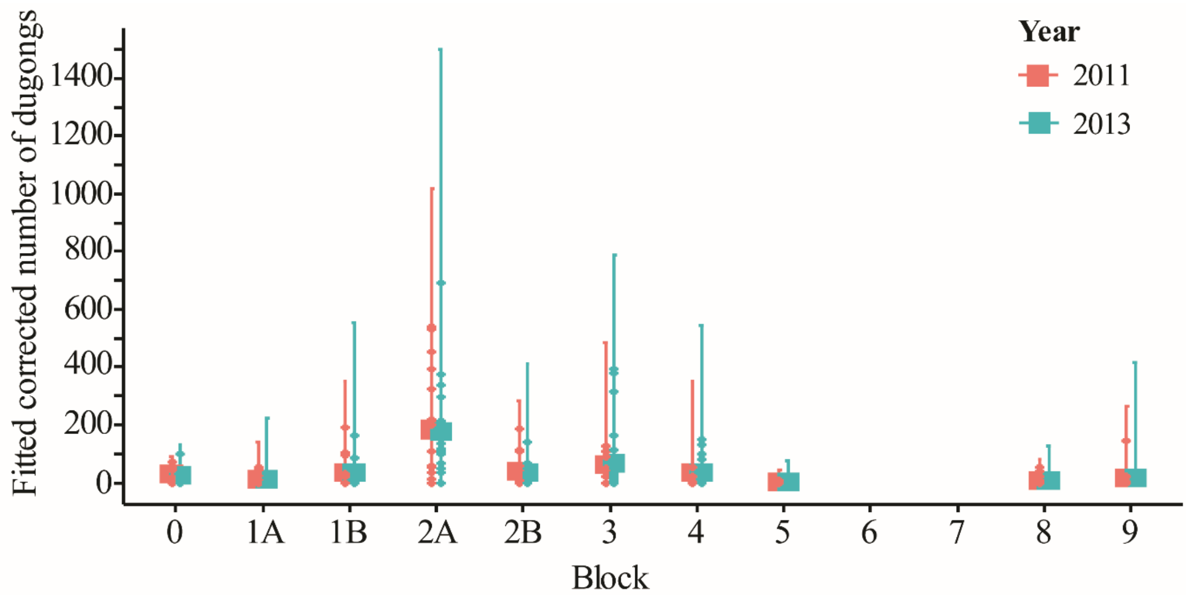

Figure 6: Fitted corrected number of dugongs per transect from a zero-inflated negative binomial model for the data collected in the aerial surveys conducted in years 2011 and 2013 from blocks 0, 1A, 1B, 2A, 2B, 3, 4, 5, 8 and 9. Blocks 6 and 7 were not included in the analysis as no dugong sightings were recorded in these blocks. Squares represent mean fitted values, and lines represent 95% confidence intervals. Dots are the observed corrected number of dugongs. The confidence intervals were estimated from the saturated model. The mean transect length of each block was used to calculate the fitted values for each block.

3.1.4 Potential Biological Removal (PBR)

Estimates of the number of dugongs that can be sustainably removed each year from Central and Western Torres Strait via all anthropogenic causes combined ranged from about 180 to 2200. If we assume the population to be stable as demonstrated by Marsh *et al.* (2015) then F_R = 1. Further assuming a medium productively level of R_{max} = 0.03, the estimated sustainable mortality from all causes using all three abundance estimates is 1100 to 1300 (Table 4).

Table 4: Potential biological removal (PBR) mortality limits calculated for dugong population estimates derived from the 2006, 2011 and 2013 aerial surveys using a range of net productivity rates (Rmax = 0.03 and 0.05) and recovery factor (F_R = 0.5 or 1). $\hat{N}$ is the estimated dugong abundance and N_{min} is the 20th percentile abundance using the log-normal distribution. Note the 2006 survey did not cover blocks 6-9.

				PBR	
Year	$\hat{N}$	N_{min}	R_{max}	F_R = 0.5	F_R = 1
2006	84,389	73,603	0.03	552	**1104**
			0.05	920	1840
2011	83,372	71,956	0.03	540	**1079**
			0.05	899	1799
2013	102,519	87,021	0.03	653	**1305**
			0.05	1088	2176

3.2 Turtles

3.2.1 Availability detection probabilities

Availability detection probabilities were lowest for Torres Strait turtles when they were in water 5 to <20 m deep (Figure 7 and Appendix VI). The detection probabilities in water <5 m deep were slightly higher for ECI2 and ECI4 and much higher for ECI3. As for dugongs, the availability detection probability for water exceeding 20 m deep was not estimated. We conservatively assumed the turtle availability in this water depth category to be 1, because no satellite fixes were collected from the six tracked turtles in this category. Given the relatively low number of turtles sighted during the aerial surveys in this depth stratum (Figure 8), our failure to correct for availability bias for sightings in this stratum makes only a trivial difference to the results.

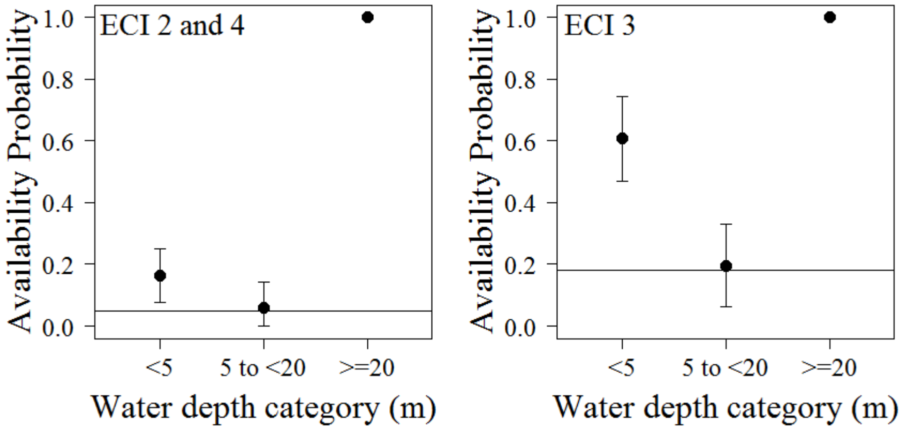

Figure 7: Availability detection probabilities estimated from adult green turtles tracked in Torres Strait for various levels of the Environmental Conditions Index (ECI). Horizontal lines represent availability estimates from Fuentes *et al.* (2015). The value for EC4 was assumed to be one as no data were obtained from tracked turtles in water >20m deep.

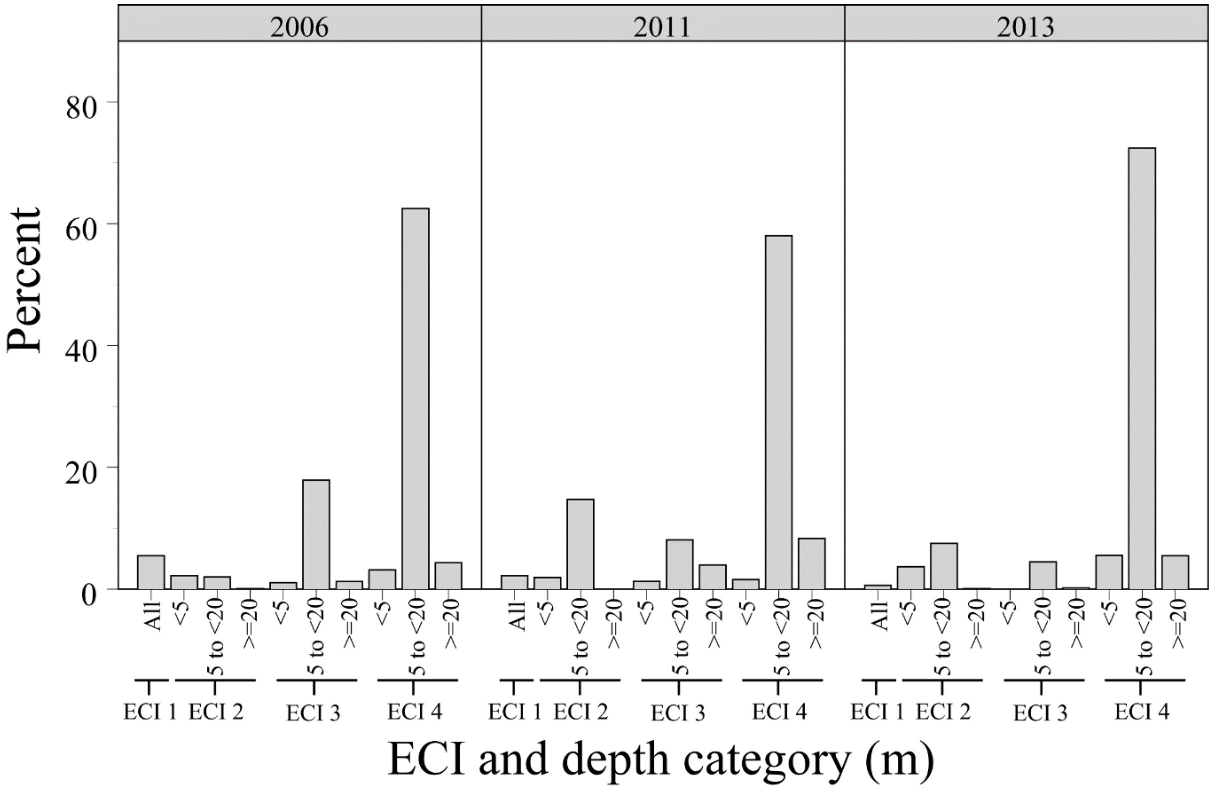

Figure 8: Percentage of all turtle sightings recorded in each Environmental Conditions Index (ECI) and depth categories.

3.2.2 Species composition

Three hundred and twenty adult-size turtles were spotted during the two helicopter flights on February 12, 2016: 107 green turtles, 1 flatback turtle, 2 loggerhead turtles (Dollar Reef) and 210 turtles that were not identified to species. The proportion of identified turtles that were green turtles was thus 0.973.

3.2.3 Green turtle population size estimates

The green turtle population size estimated using Hagihara method (adjusted assuming the proportion of green turtles was 0.973) was 72% of the estimated population size using the Fuentes methodology in 2013 (Figure 9, Table 5 and Appendix VII and VIII). This difference was largely due to: (1) 9% of turtle sightings being in water depths less than 5 m for which the availability detection probability was higher than those from Fuentes method, and (2) 6% of turtle sightings being from water exceeding 20 m for which availability correction was not applied. The other fixed wing aerial surveys had similar percentages of turtle sightings in these two depth categories (2006: <5 m – 6%, $\geq$20 m – 6%; 2011: <5 m – 5%, $\geq$20 m – 12%; Figure 8).

The estimated size of the population of large immature and adult-sized female green turtles (75% of total green turtles) in Central and Western Torres Strait is tabulated in Appendix IX. The coefficient of variation of the population size estimates in 2013 was very similar for both methodologies (Table 5).

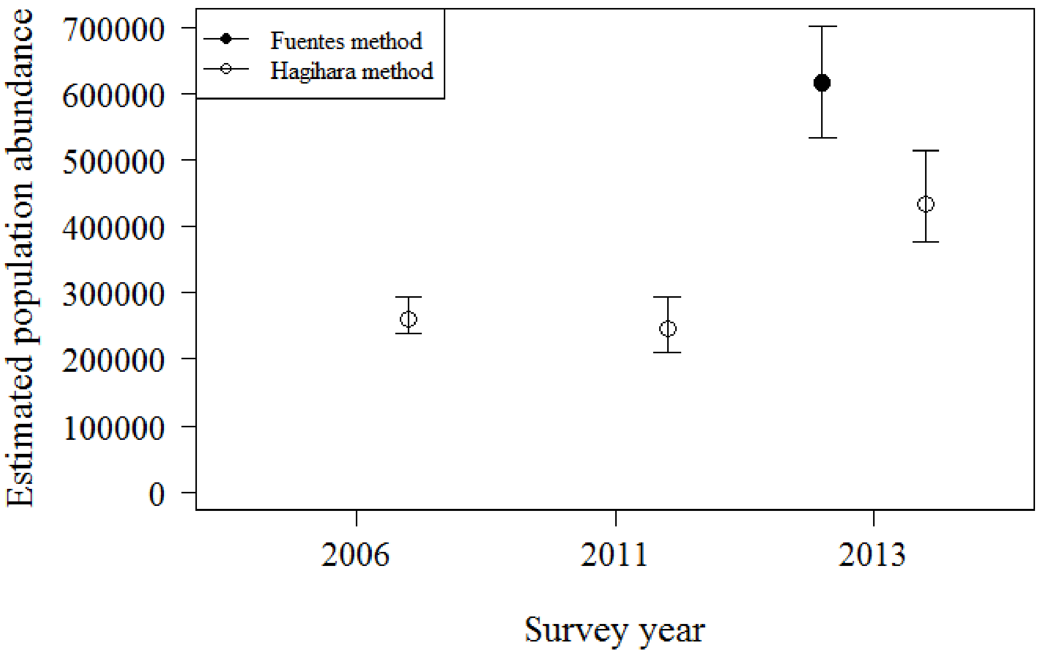

Figure 9: Estimates of green turtle abundance (both sexes) and their standard errors for the aerial surveys conducted in 2006, 20011 and 2013 using Hagihara (open circle) methodology all years and the Fuentes (closed circle) methodology 2013 only. Note the aerial survey in 2006 covered a smaller area than the 2011 and 2013 surveys.

Table 5: Comparison of the Coefficients of Variation (CV) of turtle (all species) population abundance estimates for Central and Western Torres Strait based on the aerial surveys conducted in 2006, 2011, and 2013. Ratio of the abundance estimates using the Hagihara method (numerator) to the corresponding estimate using the Fuentes method (denominator) for 2013 only.

Year	CV		Fraction*
	Fuentes method	Hagihara method	
2006	n/a	0.10	n/a
2011	n/a	0.16	n/a
2013	0.14	0.16	0.72

3.2.4 Temporal trends

The corrected number of all turtles (both sexes) per transect was significantly higher in 2013 than in 2006 or 2011, but there was no significant difference between 2006 and 2011 (Figure 10 and Appendix X). The analysis with the 2011 and 2013 survey data for all blocks also showed that the corrected number of turtles was significantly higher in 2013 than 2011 (Figure 11 and Appendix XI). In all three years, block 2A had the highest number of turtles, followed by block 4. The lowest corrected number of turtles per transect was found in block 0 in 2006 (Appendix X), and in 2011 and 2013 blocks 6-9 had lower numbers (Appendix XI). No yearly effect was found in the zero components; blocks 2A and 4 had the lowest number of transects on which no turtles were sighted. The block with the highest number of transects with no turtle sightings was block 1B in 2006; block 6 in 2011 and 2013.

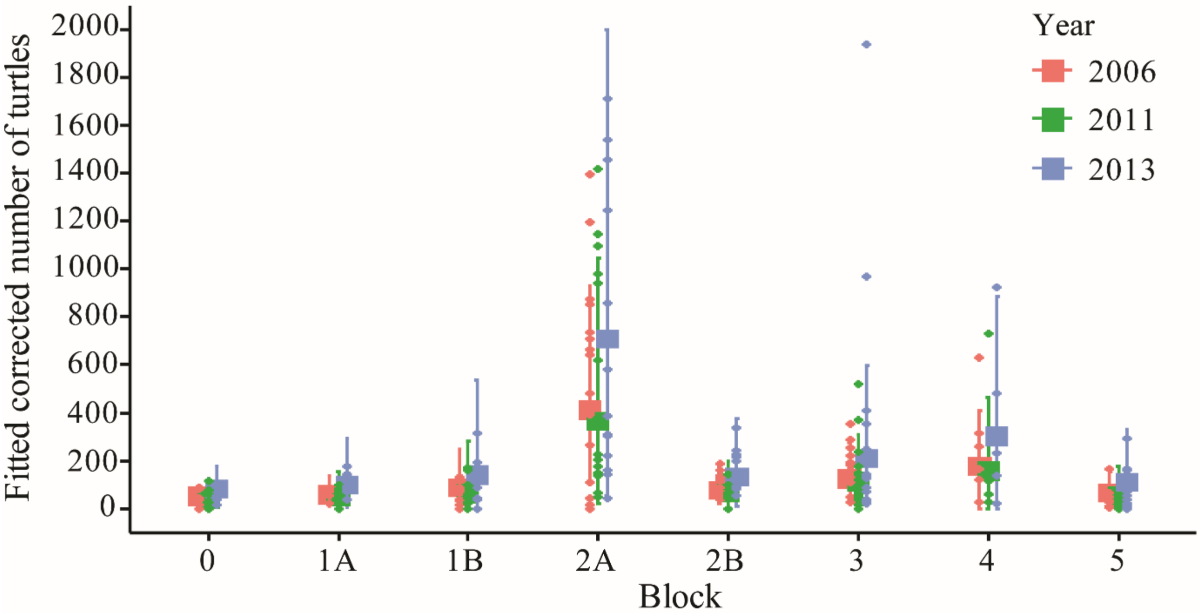

Figure 10: Fitted corrected number of all turtles (both sexes) per transect from a zero-inflated negative binomial model for years 2006, 2011 and 2013 and blocks 0, 1A, 1B, 2A, 2B, 3, 4 and 5. Squares represent mean fitted values, and lines represent 95% confidence intervals. Dots are the observed corrected number of turtles. The confidence intervals were estimated from the saturated model. The mean transect length of each block was used to calculate the fitted values for each block.

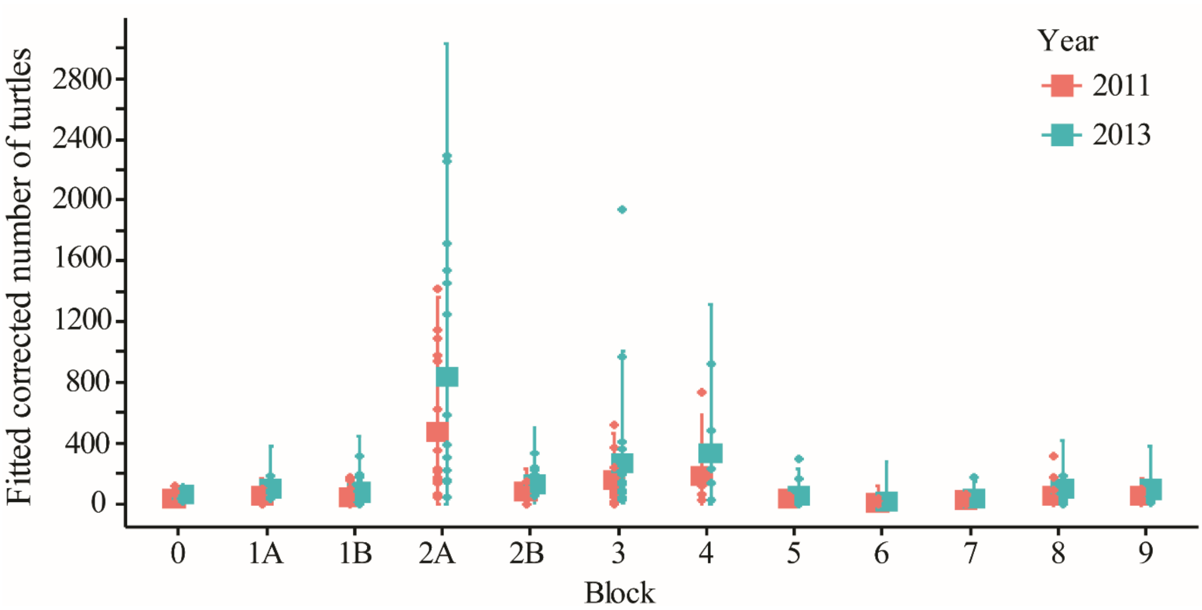

Figure 11: Fitted corrected number of all turtles (both sexes) per transect from a zero-inflated negative binomial model for years 2011 and 2013 and blocks 0, 1A, 1B, 2A, 2B, 3, 4, 5, 6, 7, 8 and 9. Squares represent mean fitted values, and lines represent 95% confidence intervals. Dots are the observed corrected number of turtles. The confidence intervals were estimated from the saturated model. The mean transect length of each block was used to calculate the fitted values for each block.

3.2.5 Potential Biological Removal (PBR)

Estimates of the number of female large immature or adult-sized green turtles that can be sustainably removed each year from Central and Western Torres Strait via all anthropogenic causes combined ranged from about 80 to 7,000 for various combinations of R_{max} and F_R. If we assume that R_{max} = 0.03 and F_R = 0.5 as befits their vulnerable conservation status, the estimates of the sustainable harvest ranges range from 1300 –2100 animals (Table 6).

Table 6: Potential biological removal (PBR) mortality limits calculated for large immature or adult-sized female green turtle population estimates derived from the 2006, 2011 and 2013 aerial surveys of Central and Western Torres Strait using a range of net productivity rate (R_{max} = 0.01, 0.03 and 0.05) and recovery factor (F_R = 0.1, 0.5 or 1). $\hat{N}$ is the estimated turtle abundance and N_{min} is the 20th percentile population abundance using the log-normal distribution. Note the 2006 survey did not cover blocks 6-9.

Year	Method	$\hat{N}$	N_{min}	R_{max}	PBR F_R = 0.1	PBR F_R = 0.5	PBR F_R = 1
2006	Hagihara	194,874	176,599	0.01	88	441	883
				0.03	265	**1324**	2649
				0.05	441	2207	4415
2011	Hagihara	184,193	157,258	0.01	79	393	786
				0.03	236	**1179**	2359
				0.05	393	1966	3931
2013	Hagihara	324,757	279,924	0.01	140	700	1400
				0.03	420	**2099**	4199
				0.05	700	3499	6998

4. DISCUSSION

4.1 Sustainability of the harvests

4.1.1 Dugong harvest

Our estimate of the size of the Central and Western Torres Strait dugong population is substantially higher than previously estimated because most dugongs in Torres Strait occur in water 5-20m deep where they spend much more time out of the sight of aerial observers than previously assumed. Given that this estimate does not correct for the dugongs in areas where no dugongs were sighted (see Martin *et al.* 2014) or the availability bias associated with animals in water >20m deep, it is likely that our revised population estimates are still underestimates by an unknown amount. In addition, there was no significant difference in dugong abundance between the (admittedly short) time series of surveys we reanalysed here.

Thus our results add to the other fisheries independent evidence of Marsh *et al.* (2015) that the Torres Strait dugong harvest in sustainable. The most credible estimates of the number of dugongs that can be sustainably removed each year from Central and Western Torres Strait via all anthropogenic causes is 1100 to 1300, a figure that is similar to the outdated estimates summarised by Marsh *et al.* (2004).

4.1.2 Green turtle harvest

As explained above, our estimate of the size of the large immature and adult-sized green turtle population in the Central and Western Torres Strait in March 2013 is substantially lower than Fuentes *et al.'s* (2015) estimate using the same data set, largely because Fuentes *et al.* (2015) did not compensate for the change in the availability detection probability with water depth. As for the dugong, our estimate does not correct for the turtles in areas where no animals were sighted (see Martin *et al.* 2014) or the availability bias associated with animals in water >20m deep. Thus it is likely that our revised population estimates are underestimates by an unknown amount. The significance of the differences we observed between years in the abundance of immature and adult-sized green turtles is impossible to interpret because the differences in actual populations size is confounded with the variable proportion of green turtles migrating from Torres Strait to breed at the time of the surveys (Limpus and Nicholls 2000). The 2006 and 2013 surveys were conducted in November coinciding with the start of the breeding season. Year 2006 was an above average breeding season (Mark Hamann pers comm.) so a higher than average number of adult females may have left on migrations. The 2011 survey was conducted in March about the time of the completion of the breeding season when females are returning.

There is considerable uncertainty about the status of the green turtle population in Central and Western Torres Strait Torres Strait, especially given the mounting evidence of recruitment failure at Raine Island, the major rookery and its possible impact on green turtles of Torres Strait (Limpus *et al.* 2003, Jensen *et al.* 2016). In contrast to the dugong, the fisheries independent evidence is limited. The most credible estimates of the number of green turtles that can be sustainably removed each year from Central and Western Torres Strait via all anthropogenic causes is 1300 –2100, a figure that is close to the outdated catch estimates summarised above, when the likely PNG harvest is considered.

4.2 Priorities for management

4.2.1 Ongoing support for community-based management

Given the mounting evidence that the dugong harvest in Torres Strait is sustainable and the large number of harvestable green turtles present in Central and Western Torres Strait, we consider that the major priority for the management of the Torres Strait dugong and green turtle fisheries should be the continued support of the culturally acceptable and scientifically robust community-based mechanisms to manage Indigenous hunting. Ongoing management is important given the escalating threats from illegal, unreported and unregulated (IUU) fishing, shipping and climate change outlined below. Alternative management approaches such as meat subsidies, a moratorium on the catch, or a ban on the transport of meat from Torres Strait to mainland Australia are almost certain to be expensive, unenforceable and have serious negative impact on the status of the dugong in the Great Barrier Reef World Heritage Area (Delisle 2013, Delisle *et al.* 2014).

The recent progress with community-based management of the harvest of dugongs and green turtles by the Australian communities in Torres Strait has been remarkable. With funding from the Australian Government, project officers employed by the Torres Strait Regional Authority (TSRA) have worked with 15 Indigenous communities to develop community-based Turtle and Dugong Management Plans. These plans have reinforced the statutory management arrangements imposed by the Commonwealth *Torres Strait Fisheries Act 1984* and its regulations by reinforcing cultural practices and protocols designed to control hunting (Marsh *et al.* 2011). This work needs to be appropriately supported with long-term program funding from government.

4.2.2 Extension of community based management to PNG

Parallel NESP funded research (Carter and Rasheed 2016) confirms that there are seagrass beds and dugong feeding trails along much of the Torres Strait coast of the Western Province. Large numbers of dugongs and green turtles are caught by villagers using long mesh nets, in this region PNG Department of Environment and Conservation, in prep). Thus the management plan setting out objectives and management arrangements for the management of the turtle and dugong fisheries in this region that is currently under development (PNG Department of Environment and Conservation, in prep) needs to be progressed with high priority. Stoeckl *et al.* (in press) concluded that regulation of the Torres Strait dugong and turtle fisheries is most vulnerable in the regions where PNG villagers take a greater proportion of catch, in areas with food shortages, and where regulatory effectiveness is hampered by weak governance. These factors are expected to be especially pertinent in regions experiencing limited capacity and the breakdown of cultural norms caused by migration. In the Fly Delta region of the Western Province, for example, migrants fleeing food insecurity have arrived in large numbers to engage in mining and logging (see Butler *et al.* 2015), resulting in significant habitat degradation.

4.2.3 Development of catch monitoring

Reliable estimates of the current dugong and green catches of each of the major hunting communities in Torres Strait including the PNG communities and the Northern Peninsula Area would enhance the trust of fisheries managers and the wider community in the fisheries independent evidence presented here and by Marsh *et al.* (2015) that dugong harvest in Central and Western Torres Strait is likely to be sustainable. Grayson (2011) offers important insights into how catch monitoring could be effectively implemented by transferring the reporting burden from the hunters to Indigenous survey agents, who could be trained to collect longitudinal data from each hunter at regular intervals. In work commissioned by TSRA, some work was done on customising this approach to Torres Strait using techniques used to survey recreational fishers (Pollock *et al.* 1994) and widely used in Australia for national and State surveys of recreational fishers (Lyle *et al.* 2002, Henry and Lyle 2003). We suggest that further work on developing a rigorous method of catch monitoring be given priority.

4.2.4 Co-ordination of management of dugongs and green turtle hunting across jurisdictions.

The current Turtle and Dugong Management Plans have been developed separately by each of the Australian communities. Gredzens *et al.* (2014) demonstrated using GPS satellite telemetry that the home ranges of dugongs in Torres Strait are generally much larger than those in the other areas where dugongs have been tracked (e.g., Hervey Bay, Shoalwater Bay and Cleveland Bay, Australia; Lease Islands, Indonesia, Cap Goulvain and Ouano regions in New Caledonia). Individual animals ranged widely across the sea countries of Torres Strait communities; one animal crossed the international boundary between Australian and Papuan New Guinean waters. In this study one animal moved twice between the waters of Badu and Boigu and all the animals used the Sea Country of the communities on Moa and possibly Mabuyag (Cleguer *et al.* 2016). In addition, the adult green turtles travel through the Sea Country of many communities during their nesting migrations (Limpus *et al.* 2003, Cleguer *et al.* 2016) further confirming the need for co-ordinated management of dugongs across jurisdictions.

Further consideration of spatial closures as a management tool will also require cross-jurisdictional collaboration, if this approach is supported by the Traditional Owners in the post-Native Title environment of Torres Strait. The modelling of Marsh *et al.* (2015) calculated two estimates of the spatial extent of take areas for dugongs in Central and Western Torres Strait with and without a depth limit on hunting. The areas are likely similar for green turtles in this region but do not include the take areas in Eastern Torres Strait, where there is a significant green turtle harvest (Harris *et al.* 1997, 2000, Kwan 1991). Assuming no depth restriction (which does not represent the actual situation), Marsh *et al.*'s (2015) model indicates that hunting mainly occurs in 38.5 % of the very high dugong density dugong habitats and 34.2 % of the high density dugong habitats. However, limiting take to waters ≤5 m deep (which they considered to be more realistic), indicates that dugong hunting occurs in only about 5.0 % of the very high and 7.9 % of the high density areas. The official Dugong Sanctuary in western Torres Strait comprises about a third of the unhunted area; the remainder is an unofficial sanctuary that results from: (1) cultural protocols that dictate where hunting should occur; (2) the *Torres Strait Fisheries Act* 1984 (Commonwealth) requirement that hunting must be carried out from vessels 6 m long or less, thereby limiting the amount of fuel that can be carried; and (3) the Torres Strait Islanders' double burden of low incomes and high commodity prices (Delisle 2013), especially the high cost of fuel in Torres Strait (up to $3 a litre).

The spatial model of Marsh *et al.* (2015 see their Figure 3) could be used by Traditional Owners to inform the design of future spatial management of hunting in Central and Western Torres Strait. We suggest that the TSRA continue to give high priority to further discussions with the Prescribed Bodies Corporate of the Top Western and Near Western Islands and the Protected Zone Joint Authority about the desirability of: (1) declaring some of the high density dugong areas as a no-hunting areas for an agreed period; and (2) determining how the Dugong Sanctuary might be extended. Dugongs and turtles are hunted together (Johannes and MacFarlane 1991) and there have been negotiations about making the Dugong Sanctuary a Dugong and Turtle Sanctuary.

Despite the jurisdictional and logistical differences between Torres Strait and the Northern Great Barrier Reef, there are several reasons why it is also important that the management of dugongs and green turtles is co-ordinated across these jurisdictions:

(1) Green turtles migrate from one region to the another to lay their eggs (Limpus *et al.* 2003, Cleguer *et al.* 2016);
(2) It is also likely that dugongs move from one area to another, especially in the Northern Peninsula Area;
(3) The Northern Peninsula Area straddles the two jurisdictions;
(4) There is considerable potential for mutual learning through a program of shared adaptive management; and
(5) Management practices in one area have the potential to impact on the status of stocks in the other area as a result of displaced effort.

Genetic, satellite tracking and aerial survey data all indicate that the appropriate ecological scale for management of dugongs and green turtles is large (Sheppard *et al.* 2006, Blair *et al.* 2014, Gredzen *et al.* 2014, Cleguer *et al.* 2016, Jensen *et al.* 2016) and both species are listed under the Convention on the Conservation of Migratory Species of Wild Animals http://www.cms.int/. Thus effective management requires initiatives to be co-ordinated across jurisdictions. Although we consider that it is sensible to continue to manage dugongs and green turtles in the Great Barrier Reef World Heritage Area separately from Torres Strait, we suggest that priority be given to joint policy for managing hunting by the Northern Peninsula Area communities. There would also be considerable advantages to encouraging mutual learning e.g., with respect to catch monitoring.

4.2.5 Management of illegal hunting

Delisle *et al.'s* (2014) study of the amount of dugong and green turtle meat consumed by the Torres Islander Diaspora and their information about the process of sharing dugong and turtle meat do not accord with allegations of an organised practice of 'illegal killing, poaching and transportation of turtle and dugong meat' (see http://www.greghunt.com.au/Media/MediaReleases/tabid/86/articleType/ArticleView/articleId/2638/Coalition-announces-Reef-2050-Plan.aspx). However, illegal, unreported and unregulated (IUU) vessels capture dugongs and green turtles in Torres Strait and Coastwatch sightings indicate the number of such vessels increased in recent years (Field *et al.* 2009). There is evidence of dugong artefacts (bones, teeth, tears and oil) being sold in Bali markets in 2013 (Nijman and Nekaris 2014), which accords with accounts of Indonesian traders travelling along the coast of Papua New Guinea to buy such artefacts along with other marine products (Sara Busilacchi CSIRO pers. comm. 2013). We suggest that it would be appropriate to investigate the capture of dugongs and green turtles by IUU vessels and the allegations of illegal trade on the Papua New Guinea coast.

4.2.6 Management of commercial fishing

In contrast to the situation on the north-eastern coast of Australia, we understand that the incidental catch of dugongs in large mesh nets set by commercial operators rarely occurs in Torres Strait except: (1) possibly in parts of the Northern Peninsula Area and (2) definitely in the Treaty villages along the PNG coast (where large green turtles are also caught and the development of alternative livelihoods will be a pre-requisite for effective change in practice). In the Australian waters of Torres Strait, the biggest indirect impact on dugongs and green turtles of changes to commercial fishing arrangements in Torres Strait would be to provide Indigenous crayfish fishers with excise relief on fuel. This action would probably have the unintended consequence of increasing hunting as most Indigenous crayfish fishers also hunt dugongs and turtles (Kwan *et al.* 2006). Since 2002 the Torres Strait Prawn Fishery has been required to use Turtle Excluder Devices in nets. As a consequence, bycatch of marine turtles in the fishery is considered to be negligible (Riskas *et al.* 2016).

4.2.7 Management of ports and shipping

Waterhouse *et al.* (2013) conducted a qualitative assessment of the key threats to the Torres Strait from water quality issues. They concluded that the threats from poor water quality to the environmental values of the area are currently relatively minor and that the largest threats in the future are most likely to be associated with the transit of many more large ships through the area. The volume of shipping transiting Torres Strait waters is projected to increase dramatically in the near future as a result of: (1) the port expansion along the urban Great Barrier Reef coast (Grech *et al.* 2013); (2) the development of a deep water sea port off the Island of Daru for the export of resources from the Ok Tedi Mine; and (3) expanded transhipment opportunities for other bulk commodities from PNG and northern Australia. Waterhouse *et al.* (2013) conclude that these increases will result in greatly increased risk of accidents such as oil spills in the Torres Strait. Currently there is very limited capacity to respond in any meaningful way to a large oil spill in Torres Strait. Because of the limited water exchange in and out of Torres Strait, there are concerns that if Torres Strait water became polluted, it would probably remain for some time, posing a risk to the seagrass communities on which dugongs and green turtles depend and to the animals themselves (Marsh *et al.* 2011). Islanders blame the extensive seagrass dieback event that occurred in Torres Strait in the 1970s on the oil spill from the *Oceanic Grandeur* in March 1970 (Johannes and MacFarlane 1991). However, this conclusion does not accord with the oceanographic evidence. The spatial model of dugong distribution and abundance resulting from the JCU aerial surveys (Marsh *et al.* 2015; Figure 3) could inform the development of oil spill response capability in Torres Strait.

4.2.8 Climate change

Katzfey and Rochester (2012) provide downscaled climate projections for the Torres Strait region for several climate scenarios; these results are considered further by Butler *et al.* (2015). Expected average and extreme changes in sea surface temperatures, rainfall, sea level, ocean chemistry and salinity and currents are likely to alter the biological productivity of the Torres Strait marine environment. Surface (air and sea) temperatures are expected to continue warming. Projected rainfall changes are more variable and uncertain, but are expected to amplify the seasonal cycle, with increases in wet season months relatively larger than for dry season months (e.g., June to August), and extreme rainfall events projected to occur more frequently. More extreme rainfall will most likely result in additional extreme flood events in the PNG coastal rivers that drain into the Torres Strait with consequential adverse impacts on agriculture and terrestrial runoff to the marine environment. The El Niño Southern Oscillation cycle will continue to be a source of interdecadal variability in the region (Lough and Hobday, 2011). The Torres Strait is north of the main cyclone belt (Green *et al.*, 2010). While there is high uncertainty about how tropical cyclones will change within a warmer climate, it is expected that the region will experience a similar or reduced number of cyclones, but of greater intensity. Increases in atmospheric CO_2 are projected to lead to substantial additional acidification of the ocean (reducing pH levels). Beyond the middle of the century, impacts are more uncertain as climate projections diverge. The situation may stabilise or become much worse (Butler *et al.* 2015).

Both green turtles and particularly dugongs are dependent on the extensive seagrass communities of the Torres Strait for food (André *et al.* 2005). Even though the seagrasses in the Torres Strait are currently in excellent condition (Carter *et al.* 2014a,b; Carter and Rasheed 2016), seagrass communities are expected to be vulnerable to increased sea surface temperature, decreased solar radiation, changing rainfall patterns and increases in cyclone intensity (Waycott *et al.* 2011, Carter *et al.* 2014c). The high sensitivity of seagrass to warmer temperatures means the effect of rising temperatures is likely to be greatest in shallow waters (Campbell *et al.* 2006, Collier and Waycott 2014). In addition to the effects of climate change, the extensive seagrass meadows in Torres Strait are known to disappear episodically over broad areas (Poiner and Peterkin 1996). The causes of such losses are unknown and it is uncertain how climate change will affect the scale and intensity of these events. Information is limited on the likely impact of climate change on dugongs (but see Marsh *et al.*, 2011 and Fuentes *et al.* 2016). There is, however, strong evidence of the dependence of dugongs on seagrass. Loss of available seagrass reduces dugong abundance through temporary migration, increased mortality, and negative effects on

dugong condition and female reproductive rates (Marsh and Kwan 2008, Meager and Limpus 2014).
Because the sex ratio of marine turtle populations is dependent on the sand temperature of nesting beaches, increasing surface temperatures are predicted to increase the female bias in the population (Fuentes *et al.*, 2009). Sea-level rise and ocean acidification have the potential to compromise the availability of nesting sites, particularly amidst coastal development or where beaches are narrow (Fuentes *et al.* 2010). Cyclones and storm surges can also impact these sites and the success of breeding (Fuentes and Abbs 2010). Inundation through storm surges has been shown to decrease the number of nests that develop to hatching stage and the number of hatchlings per clutch, though this may vary among species. Like dugongs, seagrass dieback also harms green turtle condition (Marsh and Kwan 2008).

In Stoeckl *et al.'s* (in press) opinion, the most effective means to ameliorate the social and economic impacts of climate change on the dugong and turtle fisheries of the Torres Strait will be to: (1) reinforce these the cultural services of these fisheries through the continued emphasis on community-based management in both Australia and PNG, and (2) invest in the development of alternative livelihoods, especially in PNG. Thus the likely impact of climate change reinforces the argument for strengthening community–based management of the dugong and green turtle harvests.

4.2.9 Animal welfare issues

The Australian community campaigns against dugong and turtle hunting in Torres Strait includes concern about animal welfare issues (Delisle *et al.* 2014, Watkin *et al.* 2016a). Experience with whaling issues (e.g., Kalland 2012) suggest that the evidence presented here about the likely sustainability of the dugong harvest may refocus attention on these animal welfare issues n.

4.3 Use of Unmanned Aerial Vehicles

Aerial surveys conducted are expensive, especially the costs associated with keeping a crew on the ground in remote areas when the weather conditions are unsuitable for aerial surveys. In addition, the risks associated with flying light aircraft low over the sea in remote areas are not inconsequential. These problems could be reduced by using Unmanned Aerial Vehicles or drones for aerial surveys in Torres Strait (including Eastern Torres Strait, especially for sea turtles) when the technology matures (see Hodgson *et al.* 2013). Another benefit of using drones will be the archival of photographs of animals and potential for identifying turtles at the species level. The data on dugong and green turtles diving behaviour collected by this study could inform the development of availability correction factors for dugong surveys conducted by drones.

5. RECOMMENDATIONS

1. That the major priority for dugong and green turtle management in Torres Strait be on-going support for the implementation of community-based management.

2. That the Protected Zone Joint Authority give high priority to:
    - continuing negotiations with Traditional owners and PNG about extending spatial closures as a culturally acceptable and logistically achievable method of controlling the levels of harvest;
    - assisting PNG to finalise and implement its Turtle and Dugong Management plan; and
    - facilitating complementary management of dugongs and green turtles across and within justifications, especially the Northern Peninsula Area and along the PNG coast;

3. That the TSRA give high priority to:
    - implementing a rigorous program to record the current dugong and turtle harvest from all the major hunting communities in Torres Strait;
    - sharing learnings from the catch monitoring process with the agencies responsible for managing the dugong and green turtle harvest in the Great Barrier Reef World Heritage Area and PNG;
    - investigating the impacts of IUU fishing and shipping on dugongs and green turtles and their habitats in Torres Strait; and
    - implementing the humane methods of killing green turtles developed in consultation with a veterinarian.

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APPENDIX I

Table A1: Availability bias estimates (*Pa*) for dugongs and their Detection Zones in Central and Western Torres Strait and depth categories for each Environmental Conditions Index (ECI). The Pollock method does not use depth categories (Pollock *et al.* 2006).

Hagihara method					Pollock method						
						Optimal sea state			Marginal sea state		
ECI*	Detection zone	Depth category	*Pa*	SE	Turbidity	Detection zone	*Pa*	SE	Detection zone	*Pa*	SE
1	all	n/a	1.00	0.00	1	all	1.00	0.00	all	1.00	0.00
2	0-2.0	<5	0.34	0.17	2	0-2.5	0.65	0.05	0-1.5	0.47	0.05
		5 to <20	0.11	0.17							
		≥20	1.00	0.00							
3	0-3.5	<5	0.80	0.09	3	0-4.0	0.46	0.06	0-1.5	0.30	0.07
		5 to <20	0.24	0.07							
		≥20	1.00	0.00							
4	0-1.5	<5	0.20	0.21	4	0-1.5	0.47	0.05	0-1.5	0.47	0.05
		5 to <20	0.06	0.20							
		≥20	1.00	0.00							

*Environmental Conditions Index (ECI) is a function of water turbidity, sea state and water depth as defined by Sobtzick *et al.* (2015).

APPENDIX II

Table A2: Estimated dugong population abundance in Central and Western Torres Strait using the Hagihara and Pollock methods. The numbers in brackets represent standard errors.

	2006		2011*		2013	
Block	Pollock method	Hagihara method	Pollock method	Hagihara method	Pollock method	Hagihara method
0	tfe	tfe	578 (404)	3870 (3712)	401 (343)	2962 (2874)
1A	858 (516)	5323 (3478)	467 (206)	2008 (1191)	tfe	tfe
1B	1005 (435)	7405 (3182)	1573 (775)	9876 (4989)	1626 (593)	10840 (4419)
2A	4362 (919)	26824 (5050)	5214 (1514)	36228 (10026)	5879 (1727) ***	35380 (9412)***
2B	736 (318)	5166 (2238)	1117 (359)	6609 (3128)	792 (368)	4516 (1981)
3	5166 (1418) **	24496 (8495)**	2083 (862)	16843 (7365)	5542 (2159)	38417 (16185)
4	2640 (1356)	15175 (8091)	297 (222)	1839 (792)	1487 (638)	10404 (4859)
5	nds	nds			tfe	tfe
6	ns	ns	nds	nds	nds	nds
7	ns	ns	nds	nds	nds	nds
8	ns	ns	778 (386)	2636 (1795)	tfe	tfe
9	ns	ns	497 (396)	3463 (2719)	tfe	tfe
Total	14767 (2292)	84389 (13797)	12604 (2080)	83372 (14693)	15727 (2942)	102519 (20146)

*Due to unsuitable weather conditions in November, this Torres Strait survey was conducted in March 2011.

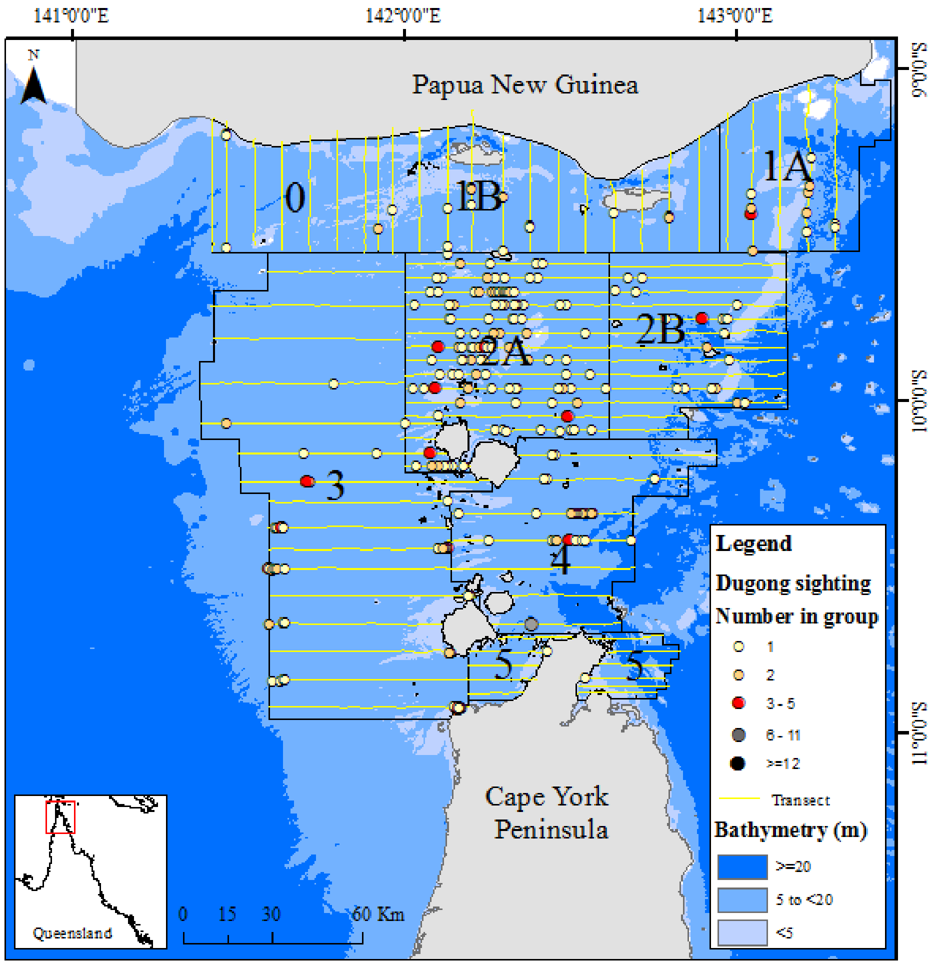
**a herd of 15 dugongs sighted

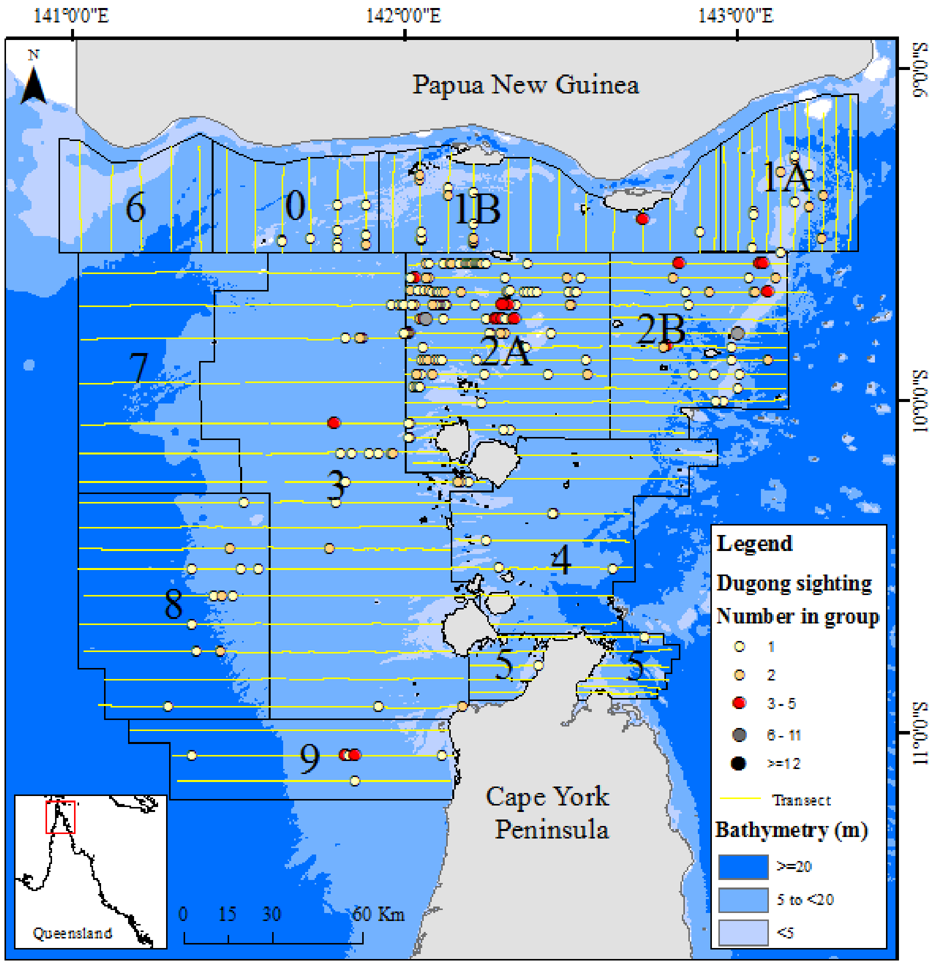
***herds of 10 and 15 dugongs sighted

tfe - too few sightings for population estimations; nds-no dugongs seen on this transect; ns-not surveyed.

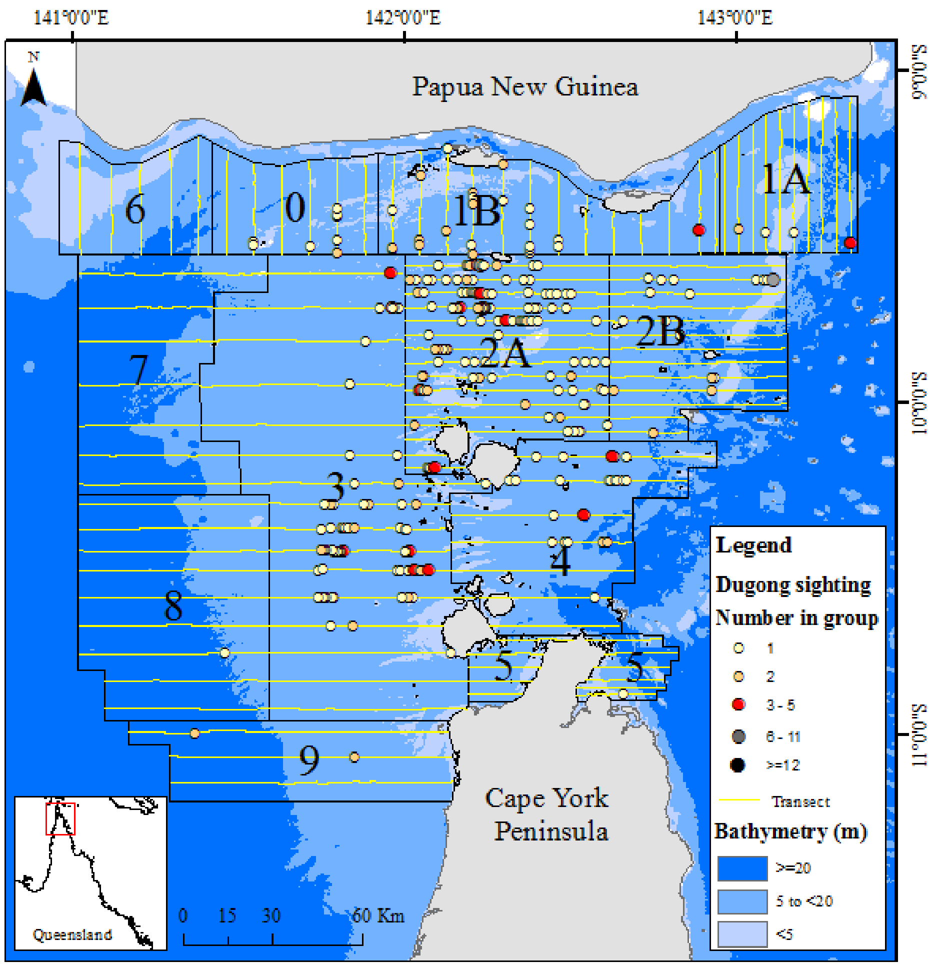
APPENDIX III

Figure A1: Dugong sightings from aerial surveys conducted in Central and Western Torres Strait in 2006 (left), 2011 (right) and 2013 (next page).





Dugong sightings from aerial surveys in Central and Western Torres Strait conducted in 2013.



APPENDIX IV

Table A3: Estimated coefficient of a zero-inflated negative binomial (ZINB) model using data from aerial surveys of Central and Western Torres Strait in 2006, 2011 and 2013 but excluding blocks 6-9. Response variable was the corrected number of dugongs per transect and explanatory variable was block. Year was not significant in both count and zero components. Transect length (km) was used as an offset in the model.

	Coefficient	Std. Error	Z value	Pr(>|z|)
Count				
Intercept	0.198	0.309	0.639	0.523
Block:1A	-0.433	0.409	-1.057	0.290
Block:1B	0.552	0.372	1.486	0.137
Block:2A	0.974	0.340	2.865	0.004
Block:2B	-0.256	0.358	-0.714	0.475
Block:3	0.271	0.347	0.780	0.436
Block:4	0.248	0.401	0.618	0.536
Block:5	-1.819	0.554	-3.282	0.001
Log(theta)	0.183	0.119	1.547	0.122
Zero				
Intercept	0.178	0.454	0.391	0.696
Block:1A	-0.129	0.610	-0.212	0.832
Block:1B	-0.307	0.561	-0.548	0.584
Block:2A	-2.222	0.658	-3.375	0.001
Block:2B	-0.917	0.574	-1.598	0.110
Block:3	-1.018	0.553	-1.842	0.066
Block:4	-0.681	0.642	-1.059	0.289
Block:5	1.356	0.683	1.986	0.047

APPENDIX V

Table A4: Estimated coefficient of a zero-inflated negative binomial (ZINB) model using data from aerial surveys of Central and Western Torres Strait in 2011 and 2013 and including all blocks except blocks 6 and 7 where no dugongs were sighted in both years. Response variable was the corrected number of dugongs per transect and explanatory variable was block. Year was not significant in both count and zero components. Transect length (km) was used as an offset in the model.

	Coefficient	Std. Error	Z value	Pr(>|z|)
Count				
Intercept	0.360	0.383	0.940	0.347
Block:1A	-0.946	0.513	-1.846	0.065
Block:1B	0.579	0.475	1.220	0.222
Block:2A	0.915	0.428	2.139	0.032
Block:2B	-0.363	0.455	-0.799	0.424
Block:3	0.198	0.442	0.449	0.654
Block:4	-0.285	0.524	-0.545	0.586
Block:5	-1.717	0.729	-2.354	0.019
Block:8	-1.806	0.530	-3.407	0.001
Block:9	-0.509	0.594	-0.857	0.391
Log(theta)	-0.005	0.147	-0.035	0.972
Zero				
Intercept	0.385	0.603	-0.639	0.523
Block:1A	0.513	0.762	0.674	0.500
Block:1B	0.348	0.723	0.481	0.631
Block:2A	-1.700	0.854	-1.990	0.047
Block:2B	-0.317	0.741	-0.428	0.670
Block:3	-0.207	0.707	-0.293	0.770
Block:4	0.054	0.819	0.066	0.948
Block:5	2.054	0.884	2.325	0.020
Block:8	0.481	0.787	0.611	0.541
Block:9	0.681	0.848	0.803	0.422

APPENDIX VI

Table A5: Green turtle availability bias estimates, Detection Zones and depth categories for each Environmental Conditions Index (ECI) for aerial surveys of Central and Western Torres Strait. Depth category is not applicable to Fuentes method (Fuentes *et al.* 2015).

			Hagihara method		Fuentes method	
ECI	Detection zone	Depth category	Pa	SE	Pa	SE
1	all	n/a	1.00	0.00	1.00	0.00
2	0-1.0	<5	0.16	0.09	0.05	0.01
		5 to <20	0.06	0.08		
		≥20	1.00	0.00		
3	0-2.5	<5	0.61	0.14	0.18	0.02
		5 to <20	0.20	0.13		
		≥20	1.00	0.00		
4	0-1.0	<5	0.16	0.09	0.05	0.01
		5 to <20	0.06	0.08		
		≥20	1.00	0.00		

APPENDIX VII

Table A6: Estimated abundance of large immature and adult-sized green turtles (both sexes) in Central and Western Torres Strait using data from aerial surveys and the Hagihara method (Hagihara *et al.* 2012, Sobtzick *et al.* 2015). Numbers in brackets represent standard errors. The estimates were corrected using the proportion of green turtles sighted during helicopter flights.

Block	2006	2011*	2013
0	9441 (4961)	7286 (6018)	4700 (1770)
1A	11661 (4233)	5997 (1848)	10575 (2664)
1B	18289 (5891)	13479 (6692)	22460 (10094)
2A	101487 (17799)	93032 (25211)	172110 (37360)
2B	19639 (3828)	9623 (2436)	21848 (4721)
3	60764 (10729)	58901 (19349)	114007 (49101)
4	35245 (13424)	29758 (19947)	53380 (25089)
5	3305 (1983)	1857 (892)	7306 (4085)
6	ns	tfe	tfe
7	ns	3358 (2111)	7330 (4076)
8	ns	15056 (9632)	9075 (5281)
9	ns	7241 (2882)	10217 (5497)
Total	259831 (26606)	245588 (40057)	433008 (68278)

*Due to unsuitable weather conditions in November, this Torres Strait survey was conducted in March 2011.
tfe - too few sightings for population estimations; ns-not surveyed.

APPENDIX VIII

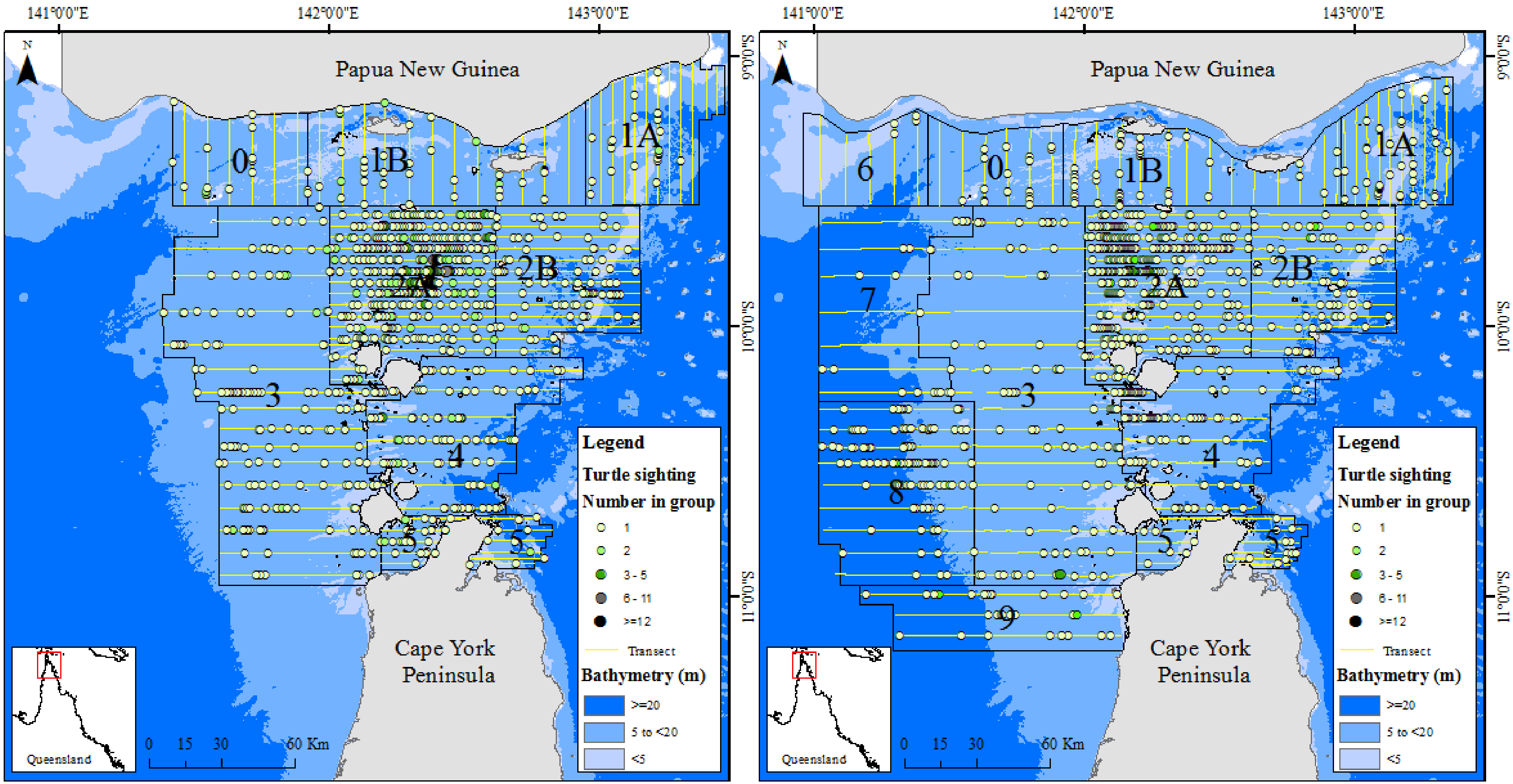
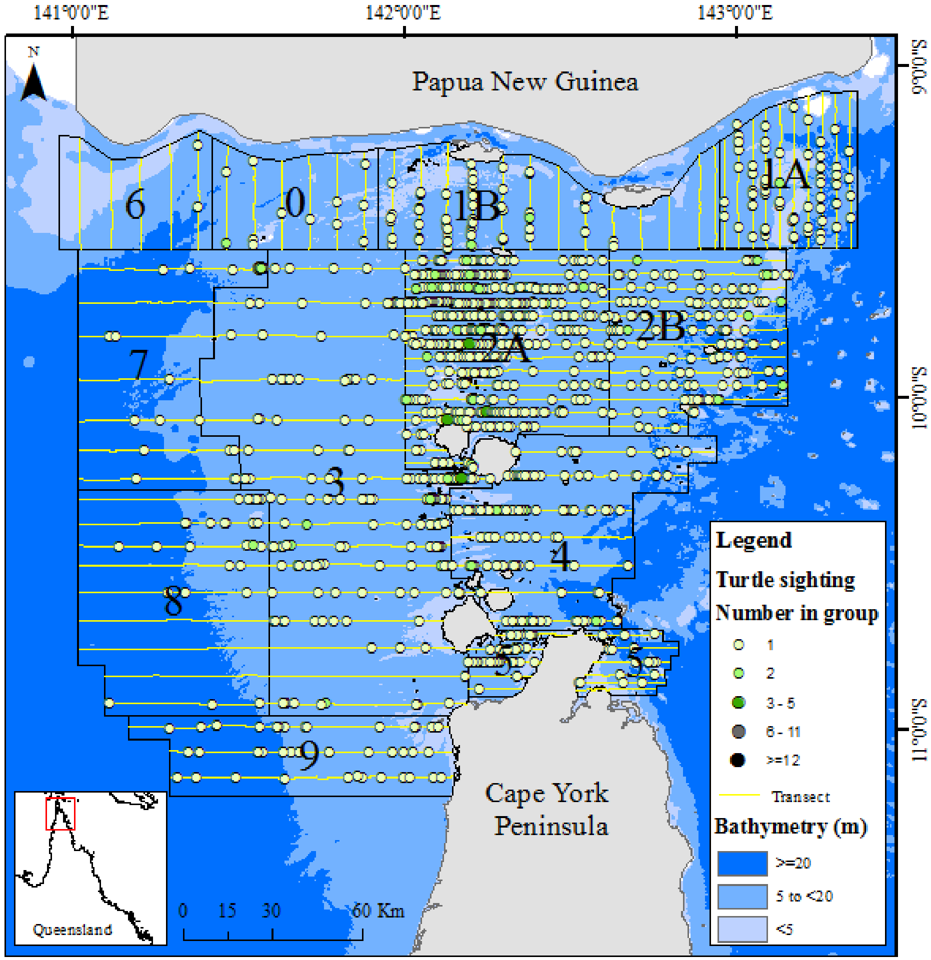


Figure A2: Turtle sightings (all species) from the aerial surveys of Central and Western Torres Strait conducted in 2006 (left), 2011 (right) and 2013 (next page).

Turtle sightings (all species) from aerial surveys of Central and Western Torres Strait conducted in 2013.



APPENDIX IX

Table A7: Estimated large immature and adult-sized female green turtle population abundance using data from aerial surveys of Central and Western Torres Strait and the Hagihara method. The numbers in brackets represent standard errors. The estimates were corrected from the proportion of green turtles sighted during helicopter flights. The number of female green turtles was calculated based on and assumed sex ratio of 3 (female):1(male) (Chaloupka and Limpus 2001, Limpus *et al.* 2005).

Block	2006	2011*	2013
0	7081 (4464)	5465 (5016)	3525 (1486)
1A	8746 (3691)	4498 (1570)	7931 (2430)
1B	13717 (5213)	10109 (5764)	16845 (8736)
2A	76115 (14945)	69774 (22608)	129083 (33268)
2B	14729 (3271)	7217 (2191)	16386 (4212)
3	45573 (9533)	44176 (16837)	85505 (40101)
4	26434 (11598)	22319 (16834)	40035 (21236)
5	2479 (1684)	1393 (814)	5480 (3434)
6	ns	tfe	tfe
7	ns	2519 (1726)	5498 (3467)
8	ns	11292 (7970)	6806 (4474)
9	ns	5431 (2574)	7663 (4785)
Total	194874 (22869)	184193 (34892)	324757 (57747)

*Due to unsuitable weather conditions in November, this Torres Strait survey was conducted in March 2011.
tfe - too few sightings for population estimations; ns-not surveyed.

APPENDIX X

Table A8: Estimated coefficient of a zero-inflated negative binomial (ZINB) model using aerial survey data from Central and Western Torres Strait in 2006, 2011 and 2013 but excluding blocks 6-9, which were not surveyed in 2006. The response variable was the corrected number of turtles (all species and both sexes) per transect. Explanatory variable in a count component was year and block, and in a zero component block was the single explanatory variable. Transect length (km) was used as an offset in the model.

	Coefficient	Std. Error	Z value	Pr(>|z|)
		Count		
Intercept	0.194	0.192	1.012	0.312
2011	-0.105	0.114	-0.919	0.358
2013	0.547	0.113	4.859	<0.001
Block:1A	0.027	0.236	0.113	0.910
Block:1B	0.648	0.228	2.839	0.005
Block:2A	1.995	0.211	9.474	<0.001
Block:2B	0.309	0.217	1.426	0.154
Block:3	0.787	0.209	3.776	<0.001
Block:4	1.121	0.241	4.650	<0.001
Block:5	0.167	0.226	0.739	0.460
Log(theta)	0.638	0.087	7.326	<0.001
		Zero		
Intercept	-1.746	0.633	-2.761	0.001
Block:1A	-1.492	1.253	-1.191	0.234
Block:1B	0.710	0.733	0.969	0.333
Block:2A	-2.105	1.212	-1.737	0.082
Block:2B	-1.933	1.230	-1.571	0.116
Block:3	-2.183	1.207	-1.809	0.070
Block:4	-16.832	2360.636	-0.007	0.994
Block:5	-1.037	0.988	-1.049	0.294

APPENDIX XI

Table A9: Estimated coefficient of a zero-inflated negative binomial (ZINB) model using aerial survey data from Central and Western Torres Strait in 2011 and 2013 and including all blocks. Response variable was the corrected number of turtles (all species and both sexes) per transect. Explanatory variable in a count component was year and block, and in a zero component block was the single explanatory variable. Transect length (km) was used as an offset in the model.

	Coefficient	Std. Error	Z value	Pr(>|z|)
		Count		
Intercept	0.111	0.252	0.442	0.659
2013	0.567	0.115	4.930	<0.001
Block:1A	0.070	0.314	0.224	0.823
Block:1B	0.758	0.320	2.371	0.018
Block:2A	2.018	0.290	6.956	<0.001
Block:2B	0.243	0.300	0.811	0.417
Block:3	0.826	0.288	2.869	0.004
Block:4	1.100	0.332	3.316	0.001
Block:5	0.198	0.314	0.632	0.527
Block:6	-0.389	0.542	-0.717	0.473
Block:7	-0.954	0.347	-2.746	0.006
Block:8	-0.119	0.323	-0.367	0.713
Block:9	-0.032	0.344	-0.093	0.926
Log(theta)	0.418	0.096	4.359	<0.001
		Zero		
Intercept	-2.437	1.083	-2.250	0.024
Block:1A	-0.614	1.570	-0.391	0.696
Block:1B	1.790	1.160	1.543	0.123
Block:2A	-17.168	3247.494	-0.005	0.996
Block:2B	-0.867	1.549	-0.560	0.576
Block:3	-1.097	1.510	-0.726	0.468
Block:4	-17.168	4832.423	-0.004	0.997
Block:5	0.071	1.338	0.053	0.958
Block:6	3.271	1.286	2.544	0.011
Block:7	0.561	1.360	0.413	0.680
Block:8	0.343	1.323	0.259	0.796
Block:9	-17.168	5219.613	-0.003	0.997


REWARD
STEEDEN