STATUS OF SEA TURTLES IN THE ARAFURA AND TIMOR SEAS

A Literature Review

This report was prepared by Nicolas J. Pilcher, Marine Research Foundation for The Arafura and Timor Seas Ecosystem Action Phase 2 (ATSEA-2) Project.
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EXECUTIVE SUMMARY

Sea turtles are important species with multiple values to natural ecosystem processes, and to customs and traditions of indigenous people of the Arafura and Timor Seas (ATS) Region. Sea turtles have been utilised for food, trade and have been part of ceremonial practices for thousands of years. Sea turtles also play important ecological roles, cropping seagrasses, foraging on sponges on coral reefs, and acting as top and middle predators in marine ecosystems. Sea turtles have also been subjected to pressures including bycatch in commercial and artisanal fisheries, and in discarded fishing gears, climate change, egg and turtle take (legal and illegal), light pollution, along with habitat loss and degradation.

The region is home to six species of sea turtle including green turtle (*Chelonia mydas*); hawksbill (*Eretmochelys imbricata*); loggerhead (*Caretta caretta*); leatherback (*Dermochelys coriacea*); olive ridley (*Lepidochelys olivacea*); and flatback turtle (*Natator depressus*). All species are listed as Vulnerable, Endangered or Critically Endangered, and are subject of protection via a number of National legislation instruments and via international conventions.

POPULATION DISTRIBUTION, CONNECTIVITY STATUS AND GENETIC STRUCTURE

Green turtles are distributed throughout the Arafura and Timor Seas, but generally remain in coastal waters where they inhabit shallow water development and foraging areas. Nesting has been documented throughout the northern shores of Australia and in the Torres Strait islands, and also in the Aru Islands in Indonesia, where the majority of nesting occurs on Enu Island. There is also green turtle nesting in Kaimana, in the northwest extend of Indonesia's West Papua coast, but little is known or quantified for the remainder of the Indonesian West Papua coast. Green turtle nesting has been also reported for East Nusa Tenggara; and on the Tanimbar and Kei Islands in the Moluccas. Green turtles disperse extensively within the ATS region and there is also emigration into and immigration from other areas, such as northward into the Sulu and Sulawesi Seas and Pacific Ocean, or westward into Indian Ocean. Green sea turtles in the Arafura and Timor seas belong to two Regional Management Units, but 17 genetically distinct breeding stocks have been identified among green turtles nesting throughout the Australasian region, four of which occur within the ATS. Green turtle aggregations at feeding grounds are often derived from multiple breeding stocks, and turtles can move great distances between foraging areas and nesting sites. There are few studies of long-term trends for rookeries in the ATS region, and there remains the need for implementation and ongoing monitoring at key green turtle rookeries to confirm the abundance and trends in numbers of nesters at each key rookery.

Hawksbills nest on multiple islands scattered across the ATS region, but few estimates of abundance are available. In Indonesia hawksbill nesting has been reported for Roti, Dana and Semau Islands in East Nusa Tenggara; and also at the Tanimbar Islands and the Aru Islands in the Moluccas. There is also nesting of hawksbills along the Kaimana coast and offshore islands in West Papua. In Papua New Guinea there are several sites where nesting occurs, but the scattered nature of the surveys and the survey durations do not permit an updated assessment of nesting at a national level. In Timor-Leste, hawksbill nesting occurs on Jaco Island and on the beaches at

Com, Tutuala and Lore in the east of the country. At least some of the post-nesting hawksbills migrate from Timor-Leste to the northwest coast of Australia.

In Australia, hawksbills nest around much of the Northern Territory coastline and on virtually all islands that have sandy beaches. The region may be home to over 5,000 nesters each year. Hawksbill turtles from northeast Australia have been recorded in Vanuatu, Solomon Islands, Papua New Guinea and elsewhere in the Great Barrier Reef. Hawksbill foraging aggregations are typically mixed stocks of individuals originating from multiple nesting areas, but there is also a trend of foraging turtles coming from nearby nesting beaches – that is, little dispersal from hatchling to adult. There are two recognised genetic stocks of hawksbill turtle breeding in Australia, and each of these stocks supports an annual nesting population of several thousand females. Data on hawksbill nesting trends are not available within the ATS region given the lack of long-term studies on these smaller rookeries. Hawksbill turtles are difficult to monitor for a number of reasons: (a) small numbers of hawksbills nest on a wide variety of beaches across a broad geographic area; (b) hawksbill beaches tend to be remote, inaccessible and sometimes so narrow that the turtle leaves no crawl trace; and (c) hawksbill turtles exhibit large year-to-year fluctuations in nesting numbers so that single year counts cannot be used to determine trends.

Loggerheads are widespread throughout ATS waters but there is no breeding by loggerhead turtles in northern Australia, Indonesia, Papua New Guinea or Timor-Leste. Substantial movement has been documented of post-nesting loggerhead turtles into foraging areas in the Arafura and Timor Seas, with turtles following coastal routes along the Western and Northern Australia coast and assumed foraging that lay predominantly in waters of North Australia. Loggerheads of the southeast Indian Ocean are treated as a single Regional Management Unit (RMU), and this includes nesting turtles from Western Australia and foraging turtles throughout the ATS region. Little trend data exists to point to overall population trends.

Leatherback sea turtles migrate through ATS waters, and a handful of nesters use beaches in Northern Australia. The leatherback turtle does not nest elsewhere in the ATS region. Leatherbacks from Papua New Guinea or Indonesia generally do not move into the ATS region, but a small proportion of leatherbacks do move down into the Arafura Sea. The west Pacific leatherback turtle is considered a single RMU, and nesting in Australia has been in a continuous decline, similar to that in Papua New Guinea.

Olive ridley turtles are moderately abundant in the ATS region, and nest on beaches in Australia, Indonesia and Timor-Leste. However, nesting is dispersed and of low volume, and sometimes confounded with hawksbill turtle nesting. There are records of olive ridley turtles from West Papua moving into the Arafura Sea, and others that show how Australian olive ridley turtles may remain in Australian waters. The genetic structure and population connectivity is highly structured, and Australian and east Indonesian olive ridleys share many of the same haplotypes, but also displayed substantial differences. There are no long-term studies on the olive ridley in the ATS region and no indication of population trends.

The flatback turtle is unique in that is nests only in Australia, with some northward distribution of foraging grounds. Foraging flatbacks have been encountered in neighbouring Papua New Guinea and Indonesia but no nesting records for this species exist in those countries. Due to their non-oceanic nature, whereby flatback turtles are restricted to Australian waters and those of southern Papua New Guinea and Indonesia, the migration and habitat connectivity data for this species is limited mostly to the Australian continental shelf and the Timor Sea. Genetic structure of flatback turtles comprises seven genetic stocks, with geographic boundaries of rookeries varying from 160km to 1,300km. Population sizes appear to be stable at present.

THREATS

Key threats include bycatch in commercial and artisanal fisheries, entanglement in and ingestion of discarded fishing gears, predation, traditional turtle take, poaching and illegal egg harvests, climate change and light pollution. Given the lack of a complete understanding of the magnitude of impacts on sea turtle populations, it is not possible to accurately identify the highest and lowest priority threats. For instance, while climate change may impact sea turtle populations, it is currently unknown to what extent this occurs.

LEGAL INFRASTRUCTURE

Several provisions exist that provide protection measures to sea turtles including National legal instruments, international conventions, fisheries management plans – including enforcement measures, and indigenous community management plans. Each of these has certain limitations, but there are also a number of strengths, such as the mandatory use of Turtle Excluder Devices in Australia, and the collective community management in the Fly River region in Papua New Guinea.

RINGKASAN EKSEKUTIF

Penyu adalah spesies bernilai penting bagi berbagai proses ekosistem alami, serta adat dan tradisi masyarakat adat di Wilayah Laut Arafura dan Laut Timor (ATS). Penyu dimanfaatkan untuk makanan, perdagangan dan sebagai bagian dari praktik upacara selama ribuan tahun. Tak hanya itu, penyu juga memainkan peranan ekologis yang penting, diantaranya mengkonsumsi lamun, mencari spons sebagai sumber pakan di terumbu karang, dan berperan sebagai predator puncak dan menengah pada ekosistem laut. Meski demikian, penyu terus mengalami ancaman termasuk sebagai tangkapan sampingan (*bycatch*) dalam perikanan komersial dan artisanal, dan terhadap alat tangkap yang dibuang, perubahan iklim, pengambilan telur dan penyu (legal dan ilegal), polusi cahaya, serta degradasi dan hilangnya habitat.

Wilayah ATS merupakan rumah bagi enam spesies penyu termasuk penyu hijau *Chelonia mydas*; sisik *Eretmochelys imbricata*; tempayan *Caretta caretta*; belimbing *Dermochelys coriacea*; lekang *Lepidochelys olivacea*; dan pipih *Natator depressus*. Semua terdaftar sebagai spesies Rentan, Terancam Punah atau Sangat Terancam Punah, dan berada dalam perlindungan sejumlah instrumen perundang-undangan Nasional dan melalui konvensi internasional.

DISTRIBUSI POPULASI, STATUS KONEKTIVITAS DAN STRUKTUR GENETIK

Penyu hijau tersebar di seluruh Wilayah ATS, tetapi umumnya berada di perairan pesisir yang merupakan wilayah berkembang dan mencari pakan. Lokasi peneluran penyu telah didokumentasikan di seluruh pantai utara Australia dan di kepulauan Selat Torres, dan juga di Kepulauan Aru di Indonesia, khususnya di Pulau Enu. Penyu hijau juga ditemukan bertelur di Kaimana, di sepanjang barat laut pantai Papua Barat Indonesia, tetapi hanya sedikit informasi terkait penyu di pantai Papua Barat Indonesia lainnya. Habitat peneluran penyu hijau juga dilaporkan di Nusa Tenggara Timur; Kepulauan Tanimbar, dan Kepulauan Kei di Maluku. Penyu hijau tersebar secara luas di dalam wilayah ATS, serta penyu hijau tersebut juga melakukan imigrasi dari dan emigrasi ke daerah lain, seperti ke utara menuju Laut Sulu dan Sulawesi dan Samudera Pasifik atau ke barat menuju Samudera Hindia . Penyu hijau di laut Arafura dan laut Timor termasuk dalam satu Unit Pengelolaan Regional, namun teridentifikasi 17 stok penangkaran penyu hijau yang berbeda secara genetik yang bertelur di wilayah Australasia, yang 4 berada di wilayah ATS. Penyu hijau yang beragregasi di tempat mencari pakan sering kali berasal dari beberapa stok penangkaran, dan penyu dapat berpindah dengan jarak yang sangat jauh antara area mencari pakan dan lokasi peneluran. Selanjutnya, terdapat beberapa studi tentang tren jangka panjang untuk habitat peneluran di wilayah ATSEA, serta masih membutuhkan implementasi dan pemantauan berkelanjutan di habitat peneluran utama penyu hijau untuk mengkonfirmasi kelimpahan dan tren jumlah sarang di setiap penangkaran utama.

Lokasi peneluran penyu sisik tersebar di beberapa pulau di wilayah ATSEA, tetapi estimasi kelimpahan yang tersedia masih terbatas. Lokasi peneluran penyu sisik dilaporkan di Pulau Rote, Ndana dan Semau di Nusa Tenggara Timur; dan juga di Kepulauan Tanimbar dan Kepulauan Aru di Maluku. Penyu sisik juga ditemukan bertelur di sepanjang pantai Kaimana dan pulau-pulau lepas pantai di Papua Barat. Papua Nugini juga memiliki beberapa lokasi bertelur, tetapi kondisi yang tersebar dan durasi yang tidak menentu menyebabkan sulit untuk melakukan penilaian terkini terkait lokasi peneluran pada tingkat nasional. Di Timor-Leste, penyu sisik bertelur di Pulau Jaco, beberapa pantai di Com, pantai Tutuala, dan Lore di bagian Timur. Beberapa penyu sisik yang telah bertelur di Timor-Leste bermigrasi ke pesisir barat laut Australia, Lokasi peneluran penyu sisik di Australia ditemukan di sebagian besar garis pantai wilayah federal Australia bagian utara, *Northern Territory* (NT) dan di hampir seluruh pulau dengan pantai berpasir. Wilayah ini diprediksi menjadi rumah bagi lebih dari 5.000 penyu petelur setiap tahun. Penyu sisik dari timur laut Australia ditemukan di Vanuatu, Kepulauan Solomon, Papua Nugini dan tempat lain di *Great Barrier Reef*. Agregasi penyu sisik di tempat mencari pakan biasanya terdiri atas campuran individu yang berasal dari beberapa lokasi peneluran. Namun, ditemukan juga kecenderungan penyu yang berasal dari lokasi pantai peneluran terdekat untuk mencari pakan – dikenal sebagai penyebaran yang sempit dari tukik hingga dewasa. Penyu sisik yang bertelur di Australia teridentifikasi memiliki dua stok genetik, dan masing-masing stok ini mendukung populasi peneluran tahunan beberapa ribu betina. Data tentang tren peneluran penyu sisik tidak tersedia di wilayah ATSEA karena kurangnya studi jangka panjang tentang habitat peneluran yang lebih kecil ini. Pemantauan penyu sisik sulit dilakukan karena beberapa alasan: (a) jumlah sarang penyu sisik yang kecil di berbagai pantai peneluran di wilayah geografis yang luas; (b) pantai penyu sisik cenderung terpencil, tidak dapat diakses dan terkadang sangat sempit sehingga penyu tidak meninggalkan jejak; dan (c) penyu sisik menunjukkan fluktuasi jumlah sarang yang besar dari tahun ke tahun sehingga hitungan satu tahun tidak dapat digunakan untuk menentukan tren peneluran.

Penyu tempayan tersebar luas di seluruh perairan ATSEA tetapi tidak ada habitat peneluran penyu tempayan di Australia utara, Indonesia, Papua Nugini atau Timor-Leste. Pergerakan substansial penyu tempayan telah didokumentasikan dari pascabertelur menuju ke daerah mencari pakan di Laut Arafura dan Laut Timor, mengikuti rute pesisir di sepanjang pesisir Australia Barat dan Utara, dan diasumsikan lokasi mencari pakan yang sebagian besar terletak di perairan Australia Utara. Penyu tempayan di tenggara Samudra Hindia diperlakukan sebagai unit pengelolaan regional (UPR – *Regional Management Unit*/RMU) tunggal, termasuk penyu yang bertelur di Australia Barat dan penyu yang mencari pakan di seluruh wilayah ATSEA. Adapun, data tren yang menunjukkan kecenderungan populasi secara keseluruhan masih sangat terbatas.

Penyu belimbing bermigrasi melalui perairan ATSEA, dan sejumlah kecil penyu yang bersarang di pantai Australia Utara. Tidak ditemukan habitat peneluran penyu belimbing lainnya di kawasan ATSEA. Penyu belimbing dari Papua Nugini atau Indonesia umumnya tidak melalui kawasan ATSEA, tetapi sebagian kecil penyu belimbing bergerak turun ke Laut Arafura. Penyu belimbing Pasifik barat dianggap sebagai UPR tunggal, dan peneluran di Australia terus mengalami penurunan, hal serupa juga terjadi di Papua Nugini.

Penyu lekang cukup melimpah di wilayah ATSEA, dan mereka memanfaatkan pantai-pantai di Australia, Indonesia dan Timor-Leste untuk bertelur. Namun, sarang mereka bersifat tersebar dan dengan volume yang rendah, sehingga kerap terkecoh dengan sarang penyu sisik. Penyu lekang tercatat bergerak dari Papua Barat ke Laut Arafura dan informasi lainnya menunjukkan bagaimana penyu lekang Australia menetap di perairan Australia. Struktur genetik dan konektivitas populasi penyu lekang sangat terstruktur, dan memiliki banyak kesamaan *haplotype* antara penyu lekang Australia dan Indonesia timur, namun disaat yang bersamaan juga menunjukkan perbedaan yang substansial. Tidak tersedia studi jangka panjang tentang penyu lekang dan indikasi tren populasi di wilayah ATSEA.

Penyu pipih memiliki keunikan karena hanya bersarang di Australia, dengan tempat mencari pakan yang tersebar ke arah utara. Akan tetapi, penyu pipih tercatat mencari pakan di negara tetangga Papua Nugini dan Indonesia, namun tidak ditemukan habitat peneluran di kedua negara tersebut. Karena sifatnya yang *non-oceanic*, di mana penyu pipih terbatas di perairan Australia, Papua Nugini bagian selatan dan Indonesia, data migrasi dan konektivitas habitat untuk spesies ini sebagian besar terbatas pada landas kontinen Australia dan Laut Timor. Struktur genetik penyu pipih terdiri dari tujuh stok genetik, dengan batas geografis habitat peneluran bervariasi dari 160 km hingga 1.300 km. Untuk saat ini, jumlah populasi terlihat stabil.

ANCAMAN

Ancaman utama penyu antara lain, tangkapan sampingan (*bycatch*) dalam perikanan komersial dan artisanal, terjerat dan menelan alat tangkap yang dibuang, predasi, pengambilan penyu oleh masyarakat untuk keperluan tradisi, perburuan dan pengambilan telur illegal, perubahan iklim dan polusi cahaya. Mengingat kurangnya pemahaman yang lengkap tentang besarnya dampak kegiatan manusia terhadap populasi penyu, tidak memungkinkan untuk secara akurat mengidentifikasi ancaman prioritas tertinggi dan terendah. Misalnya, saat ini tidak diketahui sejauh mana perubahan iklim dapat berdampak pada populasi penyu.

INFRASTRUKTUR HUKUM

Adapun, beberapa ketentuan hukum telah memberikan perlindungan terhadap penyu termasuk diantaranya instrumen hukum nasional, konvensi internasional, rencana pengelolaan perikanan – termasuk upaya penegakan, dan rencana pengelolaan masyarakat adat. Masing-masing ketentuan tersebut memiliki keterbatasan, tetapi juga memiliki sejumlah kelebihan, seperti mewajibkan penggunaan *Turtle Excluder Devices* di Australia, dan pengelolaan komunitas kolektif di wilayah Fly River di Papua Nugini.

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CHAPTER 1. INTRODUCTION AND BACKGROUND

Sea turtles are important species with multiple values to natural ecosystem processes, and to customs and traditions of indigenous peoples of the Arafura and Timor Seas (ATS) Region; Figure 1). Sea turtles have been utilised for food, trade and have been part of ceremonial practices for thousands of years. Sea turtles also play important ecological roles, cropping seagrasses, foraging on sponges on coral reefs, and acting as top and middle predators in marine ecosystems. However, sea turtles have been subjected to increasing pressure as threats such as bycatch in commercial and artisanal fisheries have increased, and climate change threatens important nesting and feeding areas and sea turtle reproductive biology. The ATS Region is home to six species of sea turtles:

- The green turtle (*Chelonia mydas*)
- The hawksbill turtle (*Eretmochelys imbricata*)
- The loggerhead turtle (*Caretta caretta*)
- The leatherback turtle (*Dermochelys coriacea*)
- The olive ridley turtle (*Lepidochelys olivacea*); and
- The flatback turtle (*Natator depressus*)

Figure 1. Map of the Arafura and Timor Seas region

A clear understanding of population status is necessary for the development of targeted and prioritised management and conservation action. This status review will serve as the basis for the development of a focussed management strategy for the four countries that border the Arafura and Timor Seas (Australia, Indonesia, Papua New Guinea and Timor-Leste). Cognisant of the existence of National management and/ or recovery plans, management initiatives embedded in international agreements, and traditional indigenous management plans, the outcomes of this present initiative are to focus on the immediate timeframe (3-5 years) and pressing conservation intervention needs that will safeguard sea turtle populations in the ATS region.

CHAPTER 2. IUCN STATUS

Among the most recognised assessments population status are the status assessments conducted for the IUCN Red List. This assessment process objectively evaluates the trend in numbers of a species, the available habitat, limitations to habitat use, whether the population is fragmented, whether the population is genetically distinct, and a suite of other factors to produce a risk of extinction assessment that is comparable across species. That is, the risk of extinction to an orchid uses the same assessment process as that for a marine turtle, and the resulting risk extinction assessments are directly comparable. For sea turtles the most common criterion on which to determine risk extinction assessments is the trend in numbers of nesting turtles over time. This is because counts of nesting females and clutches of eggs on beaches is the most common type of data collected for sea turtles. These assessments are undertaken by members of the IUCN Species Survival Commission (SSC) Marine Turtle Specialist Group (MTSG). The 2020 IUCN Red List of Threatened Species lists the six marine turtle species found in the ATS region as follows:

- **Leatherback** (*Dermochelys coriacea*): Vulnerable (global)
  Critically endangered (West Pacific subpopulation)
- **Hawksbill** (*Eretmochelys imbricata*): Critically endangered (global)
- **Loggerhead** *Caretta caretta*): Vulnerable (global)
  Near threatened (Southeast Indian Ocean subpopulation)
- **Green** (*Chelonia mydas*): Endangered (global)
- **Olive Ridley** (*Lepidochelys olivacea*): Vulnerable (global)
- **Flatback** (*Natator depressus*): Data deficient (this does not mean that there is no data available, but merely that the data have not yet been compiled and assessed using IUCN criteria). However, under Australia’s Environment Protection and Biodiversity Conservation Act 1999), where flatbacks are endemic, they are listed as vulnerable.

CHAPTER 3. REGIONAL MANAGEMENT UNITS

The MTSG recognised long ago that it was unrealistic to assess sea turtles at a global scale due to the vast differences in trends at different locations, and in recent years has conducted assessments at a level commensurate with their movements and genetic linkages. This more regionally-restricted assessment of extinction risk is conducted at a level of Regional Management Units, or RMUs (Wallace et al. 2010). The RMU framework is a solution to the challenge of how to organize marine turtles into units of protection above the level of nesting populations, but below the level of species, within regional entities that might be on independent evolutionary trajectories. As new assessments are conducted by the MTSG, they now address sea turtle extinction risk at the RMU level. The Leatherback subpopulation assessment listed in Section 2.0 is an example of a recent assessments conducted using the RMU framework. The current recognised Regional Management Units of sea turtles in the ATS region are as follows:

Green (2) : Southwest Pacific, Southeast Indian Ocean

Hawksbill (3) : Southeast Indian Ocean, West Pacific / Southeast Asia, Southwest Pacific

Loggerhead (1) : Southeast Indian Ocean

Leatherback (1) : West Pacific

Flatback (2) **:** Southeast Indian Ocean, Southwest Pacific

Olive Ridley (1) **:** West Pacific

CHAPTER 4. MANAGEMENT UNITS

A more regionally restricted method of grouping turtle populations based primarily on genetic stocks is described by Management Units, or MUs (while the RMUs described above also address movements and connectivity and dispersal to foraging habitats). Green turtle Management Units were described by Dethmers et al. (2006); hawksbill Management Units were described by Broderick et al. (1994) and Vargas et al. (2016); and Management Units for olive ridley turtles were described by Jensen et al. (2013). More recently, Management Units for flatback turtles were described by FitzSimmons et al. (2020). Leatherback genetics structure by Dutton et al. (1999) that defined only one stock for the western Pacific and Southeast Asia. Australia does not further break the western Pacific stock down into Management Units. Loggerhead genetic structure was described by Bowen et al. (1994, 1995) and there are two distinct breeding stocks in the west Pacific region. Australia considers the western Australian stock ranging up into the Northern Territory to be a single Management Unit. This extensive genetic analysis work based out of Australia considered the linkages between Australian nesting and foraging turtle species with those of neighbouring countries, and the Management Units for turtles in the Arafura and Timor Seas, where these are described, are worthy of recognition, as follows:

Green : (1) Gulf of Carpentaria, (2) Ashmore Reef, (3) Scott Reef / Browse Island, (4) Aru Islands.

Hawksbill **:** (1) Great Barrier Reef (GBR), Torres Strait and Arnhem Land; (2) northwest shelf of Western Australia.

Loggerhead: One single Management Unit.

Leatherback: One single Management Unit.

Olive Ridley: (1) Cape York, (2) Northern Territory.

Flatback **:** (1) Joseph Bonaparte Gulf, (2) Arafura Sea.

CHAPTER 5. GREEN SEA TURTLES

5.1 DISTRIBUTION & MIGRATIONS

Green turtles are distributed throughout the Arafura and Timor Seas, but generally remain in coastal waters where, presumably, they inhabit shallow water development and foraging areas (Figure 2). Nesting has been documented in Australia and in the Torres Strait islands (Limpus 2007a), and also in the Aru Islands in Indonesia (Enu, Jeh and Karang Islands; Dethmers 2010), where the majority of nesting occurs on Enu Island. There is also green turtle nesting in Kaimana, in the northwest extend of Indonesia's West Papua coast (Tapilatu et al. 2017), but little is known or quantified for the remainder of the Indonesian West Papua coast. Green turtle nesting has been also reported for Roti and Dana Islands (East Nusa Tengggara); and on the Tanimbar Islands in the Moluccas, and on the Kei Islands (Schulz 1989). Schulz suggested estimates of annual nesting by green turtles as follows: 3,600 to 5,400 in the Aru and Tanimbar Island groups in the Moluccas; and 40-50 at the Kei islands. Dethmers (2010) estimates appear to support these estimates, at least for the Aru Islands. There are no ongoing monitoring programmes at other sites to provide estimates of annual nester abundance.

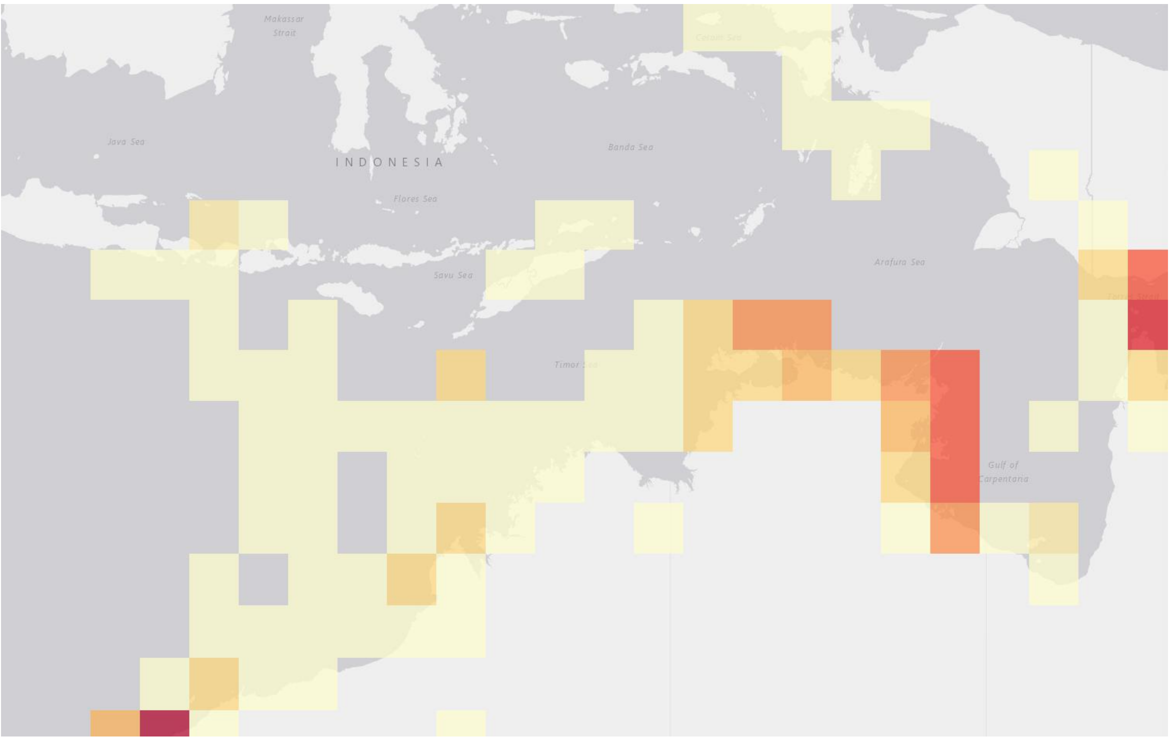


Figure 2. Green turtle distribution in the ATS region. Darker colours indicate greater number of records per 10 cells. Data includes nesting locations and at-sea data from satellite tracking. Image source: OBIS-Seamap 2021

In the Torres Strait, straddling Australia and Papua New Guinea, the majority of green turtle nesting occurs on the eastern islands and these turtles are more likely to be associated with the northern Great Barrier Reef (nGBR) region. Green turtle nesting in the Torres Strait occurs on Bramble Anchor, Don Dower, Maclennan and Underdown Cays. There are no reports of green turtle nesting on the Papua New Guinea mainland fronting the ATS region. In Timor-Leste, Jaco

Island and the beaches of Com, Tutuala and Lore have been identified as turtle nesting sites (Nunes 2001, Amaral pers. comm.), with green turtles nesting between February and August. Other breeding sites may exist on the south coast of Timor-Leste. Green turtles are also reported to nest in low numbers in Tibar bay, west of Dili, and at Ulmera (Eisemberg et al. 2014). Nest protection programs meare run by local communities in the east of the country and coordinated by CI. Since 2018, these programmes are providing the first species specific nesting data for Timor- Leste.

In Australia, nesting sites are extensive and occur along much of the northern region. In the eastern Gulf of Carpentaria nesting occurs at the Wellesly Group (Bountiful, Pisonia & Rock Islands). An order of magnitude estimate of the annual nesting population in the Wellesley Group is ~5,000 females (Limpus 2007a). In the western Gulf of Carpentaria nesting occurs along the Arnhem Land coast, Groote Eyland and Sir Edward Pellew Islands (SEP; Limpus 2007a and references therein). A preliminary estimate of the size of the annual green turtle nesting population for eastern Arnhem Land is thousands of females annually (at present there are no precise population size estimates, nor an understanding of population size trends). The principal nesting sites include: mainland beaches from Binanangoi Point (Port Bradshaw) south to Cape Shield, especially between Binanangoi Point and Wanyanmera Point; northern beaches of Woodah Island; eastern Groote Eylandt area, especially North East Island and south-eastern Groote Eylandt (south from Ilyungmadja Pt.; south from Ungwanba Point; Marangala Bay); and Sandy Islet. In the SEP the majority of sea turtle nesting activity occurred on two islands, West Island and Vanderlin Island. The low-density green turtle on the mainland and adjacent islands of northeast Arnhem Land that lie south from Cape Arnhem are within the Dhimurru Indigenous Protected Area. Low-density green turtle nesting also occurs on the Crocodile Islands, including Murrungga island and Gurriba island, northern Arnhemland. Gurriba was declared turtle sanctuary by the traditional owners of the land. The land and sea country are managed by the Crocodile Island rangers, based in Milingimbi. Declaration of Indigenous Protected Areas over Lhanapuy and Groote Eylandt means that the majority of the green turtle nesting habitat in western Gulf of Carpentaria is now on indigenous protected and managed lands.

Hamann et al. (2006) suggested the green turtle nesting population of the Sir Edward Pellew Islands was in the order of 100s of females per year. However, they indicate that this came from a single season survey and that there may be substantial changes from year to year.

Western Australia supports one of the largest green turtle populations in the world and may potentially be the largest in the Indian Ocean (Limpus 2007a). The principal rookeries include the Lacepede Islands, Monte Bello Islands, Barrow Island, North West Cape and Browse Island. Numerous small rookeries also occur in Western Australia. While most of these sites lie outside of the ATS region, there is significant migration of post-nesting turtles into the Timor and Arafura seas and large expanses of foraging areas (Ferreira et al. 2020). Limpus (2007a) indicated that in an average nesting season, tens of thousands of green turtles may breed on western Australian beaches. Recent data from key rookeries in Western Australia (IUCN MTSG, unpublished) suggests this number may be lower but still significant, ca. 5,000 annual nesters.

Post-breeding migration of turtles in the Gulf of Carpentaria has been derived from flipper tag recoveries from the Wellesley Group Rookeries and from satellite telemetry of females from eastern Arnhem Land Rookeries (Figure 3; Kennett et al. 2004). All foraging areas linked to this breeding assemblage by tag recovery and satellite tracking lie within the Gulf of Carpentaria, and this appears to be a very regionally-restricted foraging distribution (Limpus 2007a).

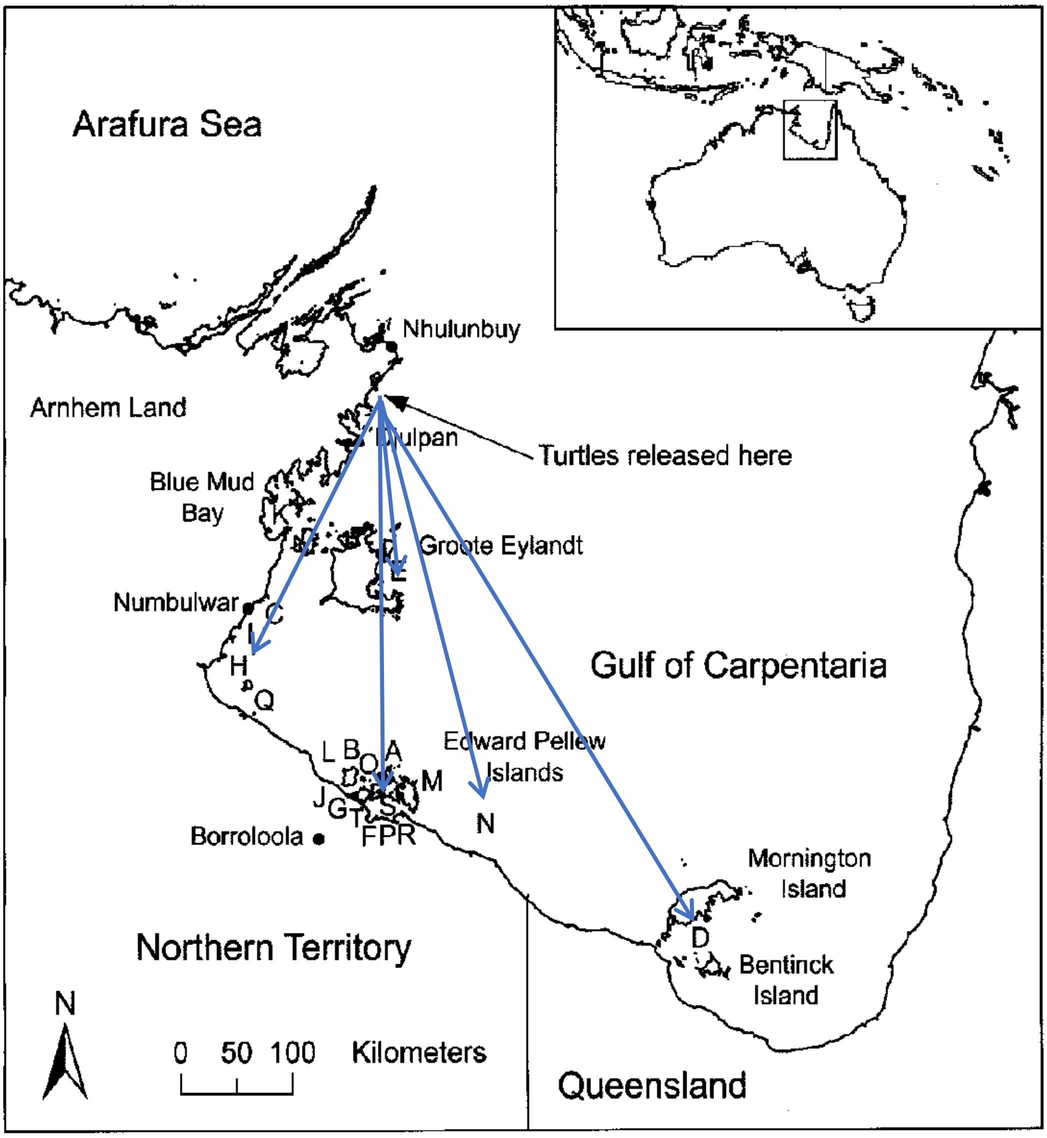


Figure 3. Green turtle movements in the Gulf of Carpentaria (adapted from Kennet et al. 2004)

Post-nesting migrations of 96 green turtles from Western Australia highlight the linkages of this turtle stock to the Arafura and Timor Seas, and indicate that turtles remain in coastal waters for most of the time and spend most of their time in Australian waters (Figure 4; Ferreira et al. 2020). However, the study also documented turtles moving to Sumba Island (Indonesia) and West Papua (Indonesia) and Torres Strait, passing through Papua New Guinea waters (Ferreira et al 2020). Key foraging areas for western Australian green turtles lie predominantly close to the Australian mainland (Figure 5; Ferreira et al 2020).

There is also green turtle nesting on the Australian islands in the Timor Sea. The National Nature Reserve (NNR) in the Indian Ocean encompasses three vegetated cays which support marine turtle nesting (West Island, Middle Islet, and East Islet) and one unvegetated cay (Cartier Reef) that also is a green turtle rookery (Whiting et al. 2000). Most nesting occurs on West Island

(Whiting et al. 2000). Early season nesting counts suggest that the total green turtle nesting population is of the order of hundreds of females annually (Guinea 1995, Whiting et al. 2000). There is also a small green turtle population nesting on Scott Reef.

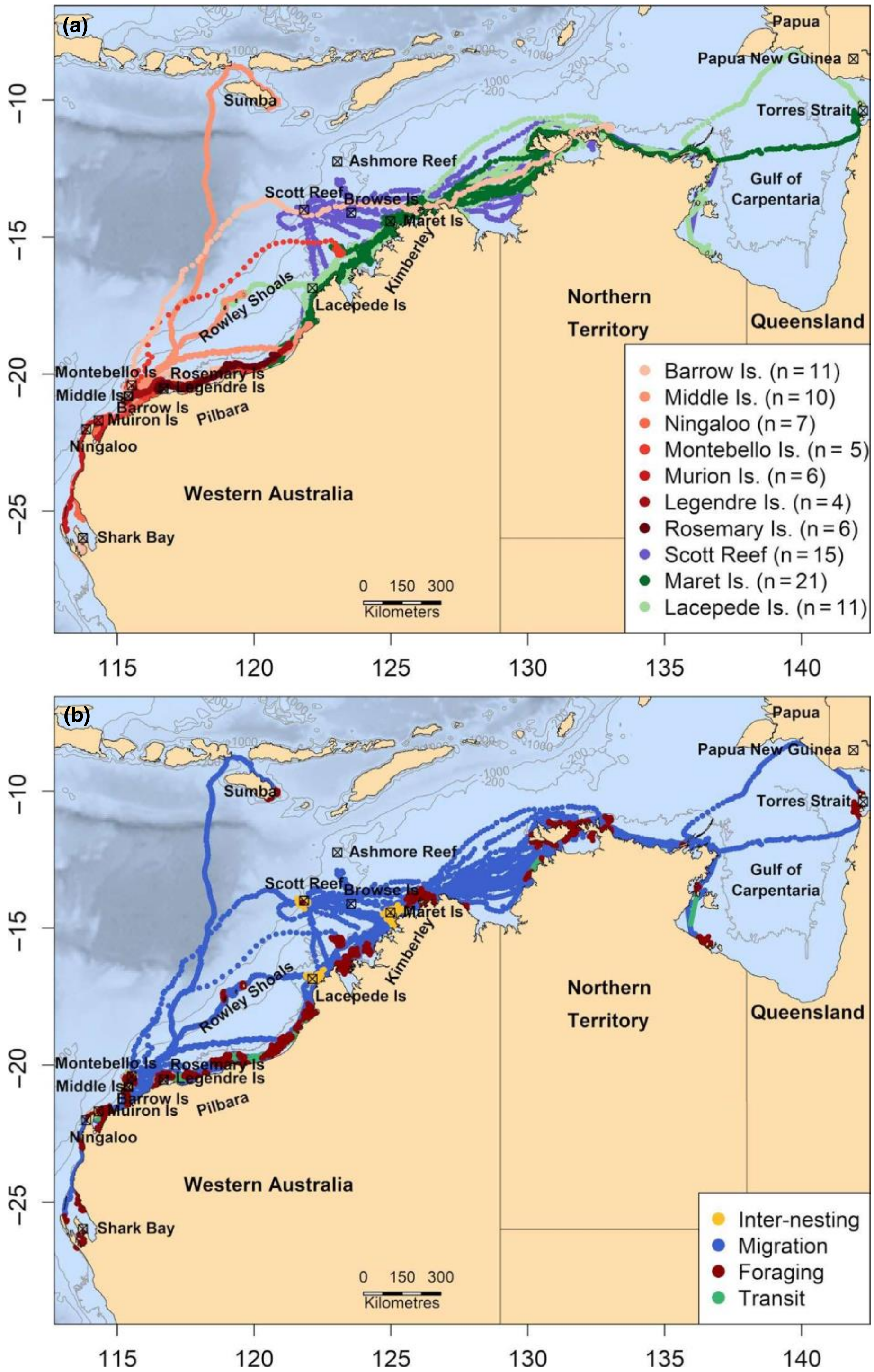


Figure 4. Top: Satellite tracks of turtles from the Norwest Shelf in Western Australia (red), the NWS-Kimberly stock (green) and the Scott-Browse stock (purple). Bottom: Movements colour-coded by activity. Image source: Ferreira et al. 2020

Limpus (2007a) documented migrations of nGBR and southern Great Barrier Reef (sGBR) turtles into Gulf of Carpentaria foraging grounds from flipper tag recoveries, and two satellite-tracked green turtles from Palau moved south into Indonesian waters (Figure 6a; Klain et al. 2007), indicating there is substantial immigration / emigration of green turtles in the ATS region to other areas. Post-nesting migrations of a green turtle from Jaco island - Nino Konis Santana NP, Timor LesteTimor-Leste to Cobourg Peninsula – Ggarik Gunak Barlu NP, Australia was also recorded (Figure 6b).

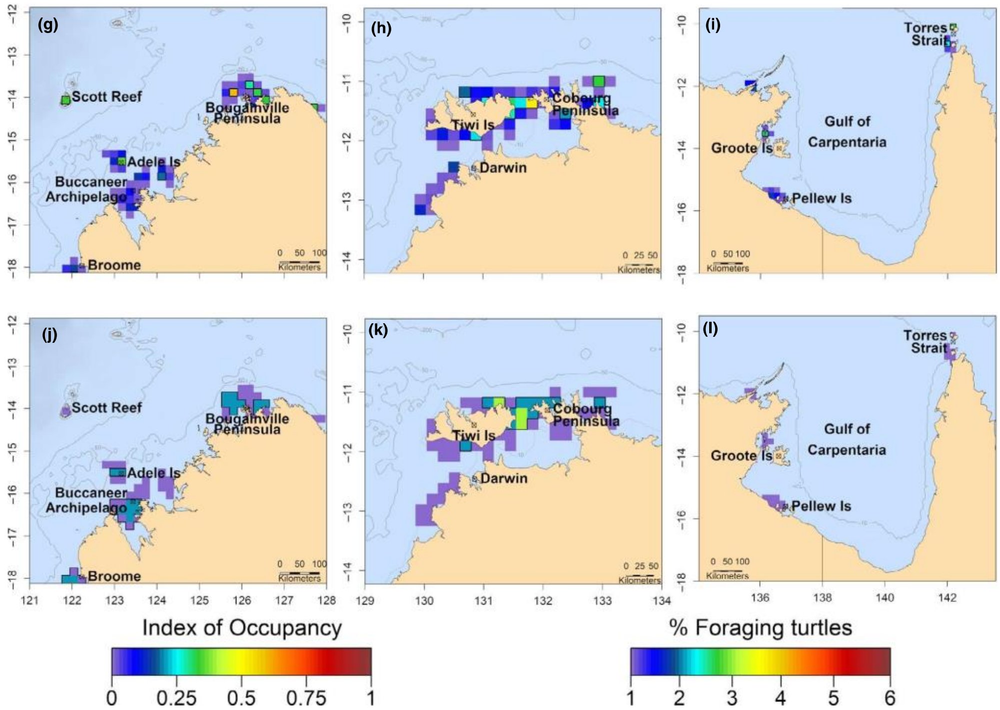


Figure 5. Top: Foraging distributions of green turtles using an occupancy index off Kimberly & Scott Reef (g); Coburg Peninsula & Tiwi Islands (h); and Gulf of Carpentaria (i). Plates j, k & l as above using percentage of foraging turtles. Image source: Ferreira et al. 2020

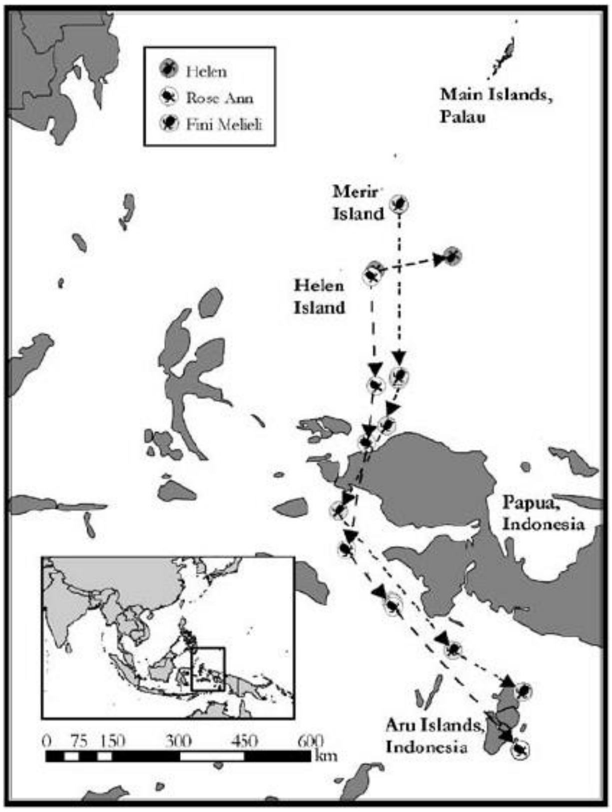


Figure 6a. Post-nesting migrations of green turtles from Merir and Helen Islands, Palau. Image source: Klain et al. 2007

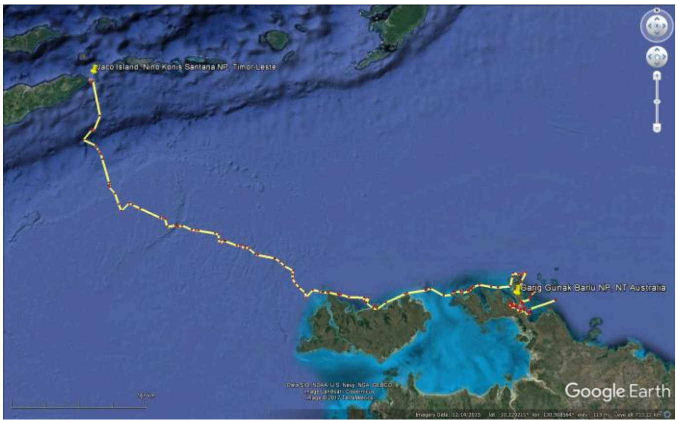


Figure 7b. Post-nesting migrations of a green turtle from Jaco island - Nino Konis Santana NP, Timor-Leste to Cobourg Peninsula – Ggarik Gunak Barlu NP, Australia. Image source: CI Timor-Leste

5.2 GENETIC STRUCTURE

Green sea turtles in the Arafura and Timor seas belong to a single RMU. However, Moritz et al. (2002) described smaller management units, or Ecologically Significant Units, that might be more applicable to understanding finer-scale differences in population structure. Subsequently, Dethmers et al. (2006) indicated there were 17 genetically distinct breeding stocks for turtles foraging in Australasian waters, and that these individual rookeries or groups of rookeries were generally separated by more than 500 km. Of note, this study demonstrated a significant discontinuity in genetic structure between Pacific Ocean stocks and those found further to the west (Figure 7). That is, the turtles in the Arafura and Timor Seas are genetically distinct from those green turtles that breed and forage in the Pacific. Dethmers et al. (2010) assessed linkages between nesting and foraging grounds via migration data and found that green turtle aggregations at each of the feeding grounds were derived from multiple breeding stocks. The geographic distance between breeding and feeding habitat strongly influenced whether a breeding population contributed to a feeding ground; however, neither distance nor size of a breeding population was a good predictor of the extent of their contribution (Dethmers et al. 2010).

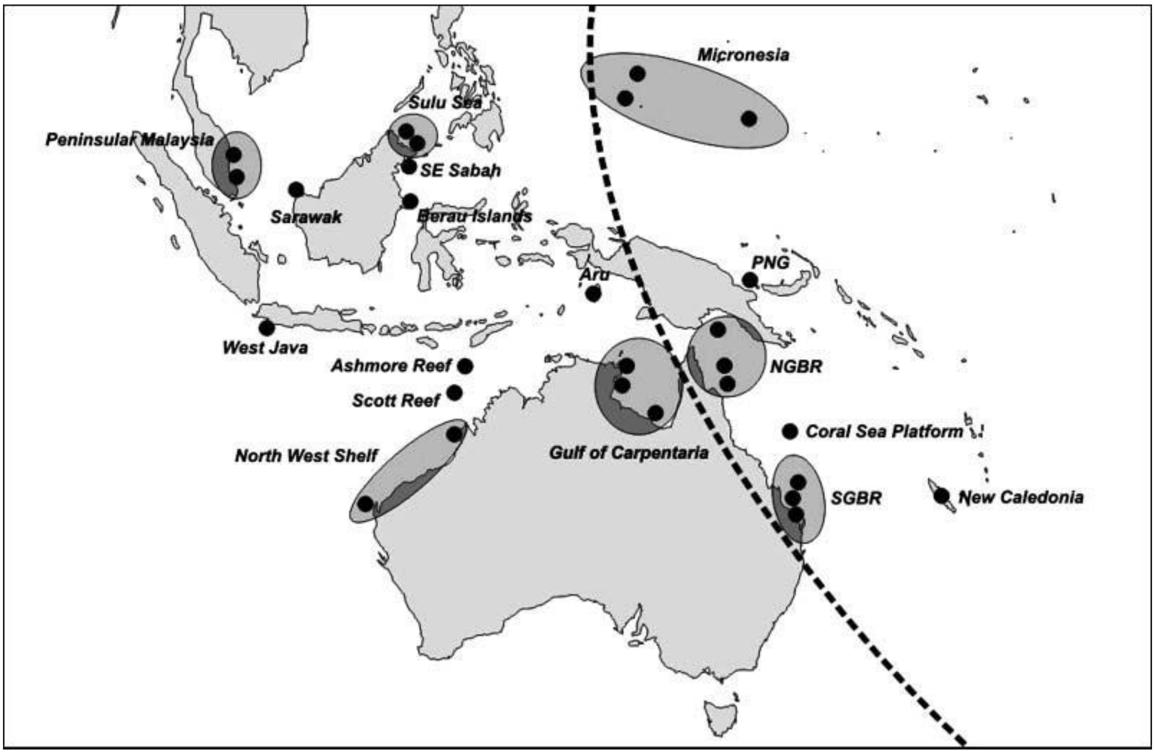


Figure 8. Location of 17 genetically distinct breeding stocks or management units as inferred from analysis of geographical structure of mtDNA variants and position of the genetic barrier (dashed line), indicating the major genetic discontinuity between the turtles west of Cape York and those of the Pacific Ocean. Image source: Dethmers et al. 2006

Mixed-stock estimates at four of the feeding grounds (Ashmore Reef, Field Islands, Aru islands and Sir Edward Pellew Islands) revealed a dominance of a single stock, with a mean contribution of 50% or more. This means that the Gulf of Carpentaria comprised one genetic stock, or Management Unit. For Ashmore Reef and Sir Edward Pellew Islands, this involved the geographically most proximate breeding stock at Aru and the Gulf of Carpentaria, respectively, both within a distance of 200 km. However, at the Ashmore Reef feeding ground, 75.4% of the contributions were assigned to the North-west Shelf stock, 960 km distant. Interestingly, the Ashmore Reef stock (at ~50km distance) had little representation at Ashmore Reef while, in contrast, 11.2% of turtles at the Cobourg Peninsula feeding grounds were estimated to have originated from the Ashmore Reef stock, 950km away. This study is a clear example of how some turtles move great distances between foraging areas and nesting sites, while there may be more suitable areas closer to home, and worthy of consideration in approaches to management and conservation of sea turtles in the Arafura and Timor Seas.

5.3 POPULATION TRENDS

There are few studies of long-term trends for rookeries in the ATS region. This is related to a lack of ongoing monitoring programmes at those key beaches in the ATS region – which more often than not are sampled opportunistically or as part of specific studies. As part of a recent IUCN Red List Assessment, numbers of green turtles at key Western Australian rookeries were used to indicate population trends (Figure 8), and these trends might reflect green turtle nesting trends elsewhere along the northern rookeries in Australia. However, there remains the need for implementation and ongoing monitoring at key green turtle rookeries in the Gulf of Carpentaria and along the beaches of the Northern Territory to confirm the abundance and trends in numbers of nesters at each key rookery.

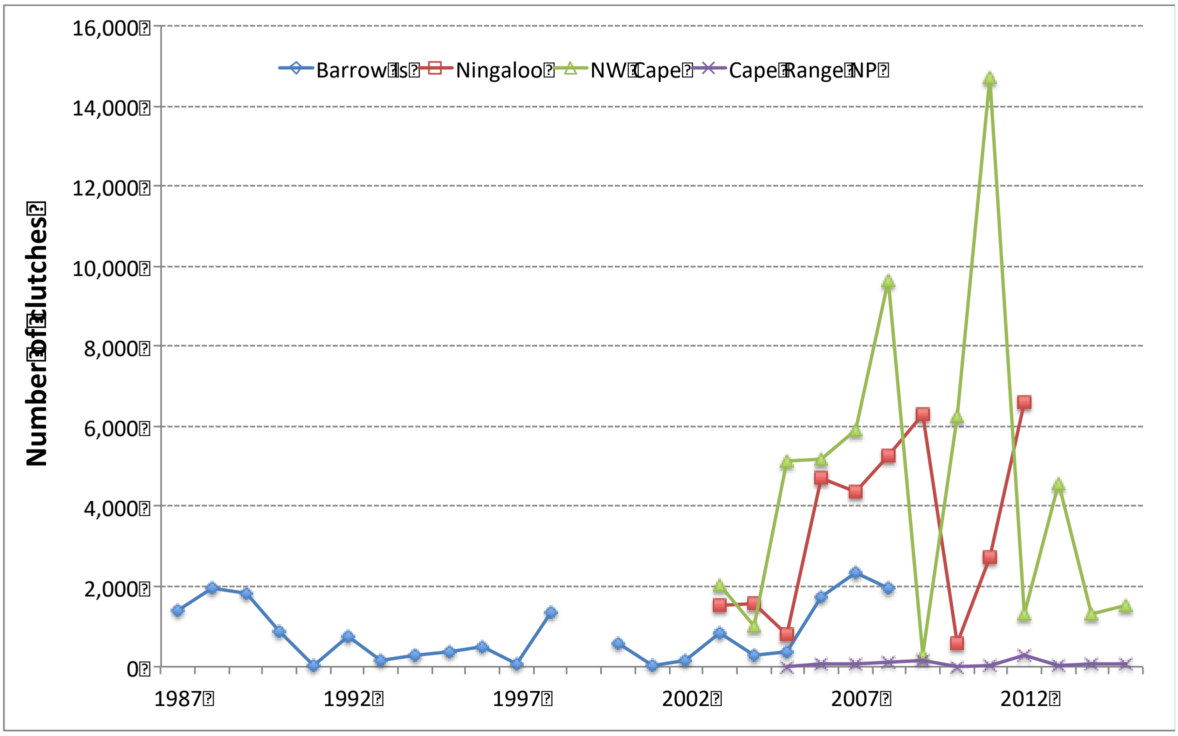


Figure 9. Trends in number of green turtle nests deposited each year on key Western Australian beaches. Image source: IUCN MTSG, unpublished

CHAPTER 6. HAWKSBILL SEA TURTLES

6.1 DISTRIBUTION & MIGRATIONS

Hawksbills nest on multiple islands scattered across the ATS region, but few estimates of abundance are available. In Indonesia hawksbill nesting has been reported for Roti, Dana and Semau Islands in East Nusa Tenggara; and also at the Tanimbar Islands and the Aru Islands in the Moluccas (Tomascik et al. 1997). However, they provide no estimates of rookery size. Tapilatu et al. (2017) indicate there is nesting of hawksbills along the Kaimana coast and offshore islands, although this site lies north of the ATS region, but similarly do not provide estimates of rookery size. Hawksbills were also reported for the Aru islands by Dethmers (2010), again with no rookery size estimates (although her study was aimed primarily at green turtles). While Asaad et al. (2018) indicate hawksbills were widespread throughout the Coral Triangle region, they did not indicate large assemblages or hawksbills in the Arafura and Timor seas. Similarly, Mortimer & Donnelly (2008) do not indicate any major nesting sites for hawksbills in the Indonesian ATS region. It is likely then that the hawksbill nesting sites in the Indonesian reaches of the ATS region are common, widespread, but of small size.

In Papua New Guinea, Kinch (2020) reports on several sites where nesting occurs, but the scattered nature of the surveys and the survey durations do not permit an updated assessment of nesting at a national level. It is suggested that the total annual nesters in PNG may be <500 turtles per year, but it is unknown how many of these are from the Torres Straits islands.

In Timor-Leste, hawksbill turtles nest on Jaco Island and Com, Tutuala and Lore beaches, between January and July. While there are no publications describing hawksbill nesting in Timor-Leste in the published literature, hawksbills nesting in the Nino Konis Santana National Park have been tracked with satellite transmitters moving through the Timor Sea and south to Western Australia (Figure 12). Nest protection programmes are run by local communities in the east of the country and coordinated by CI. Since 2018, these programmes are providing the first species specific nesting data for Timor- Leste.

In Australia, Hamann et al. (2006) indicate hawksbill nesting in the Sir Edward Pellew islands in the range of <100 turtles per year. There were also 220 nesting females in 2009 and 580 females in 2010 at the Groote Eylandt archipelago in the western Gulf of Carpentaria (Hoenner et al. 2016). Hawksbill turtles nest around much of the Northern Territory coastline and on virtually all islands that have sandy beaches (Chatto 1998). In general, most of this nesting occurs east of Darwin with the best areas found between Bathurst and North Goulburn Islands, from the east of Elcho Island, east and south to the southern end of Groote Eylandt, and the outer Sir Edward Pellew Islands. The size of the nesting population at each of the numerous hawksbill rookeries in the Gulf of Carpentaria remains incompletely surveyed (Limpus et al. 2000). Approximately 40 nesting sites were recorded for hawksbill in northeastern Arnhem Land during a spring aerial survey (Limpus et al. 2000). Additional low density nesting beaches probably occur in the region; however, their identification may be obscured by concurrent olive ridley nesting for those sites where positive distinction between these species could not be made for all tracks observed.

Limpus et al. (2000) found 12 sites with an estimated > 100 nesting female hawksbill annually as follows:

1. Outer islands of the English Company Islands area: Truant Island and Bromby Island.
2. Northeastern Groote Eylandt area: North East Island, Hawk Island, and Lane Island, which are the extreme northeastern beaches of Groote Eylandt. This area appeared to be the most significant area for hawksbill nesting in the Northern Territory.
3. Northwestern Groote Eylandt area: Hawknest Island, Bustard Island, and the small island southwest of Bustard Island.
4. Southeastern Groote Eylandt area: Two small islands of Cape Beatrice and the southeast coast of Groote Eylandt.

Some low-density nesting also occurs within the Gurig National Park on the Coburg Peninsula. For each site with high-density nesting there was a series of lower density nesting sites in the vicinity (Limpus 2007b). Most of the hawksbill rookeries of Arnhem Land lie outside National Park or other habitat managed for conservation purposes except for the low-density hawksbill on the mainland and adjacent islands of northeast Arnhem, and the land south from Cape Arnhem that are within the Dhimurru Indigenous Protected Area.

There have been no detailed monitoring studies of the size of the annual hawksbill breeding population at any of the Arnhem Land hawksbill rookeries (Limpus 2007b), and Limpus et al. (2000) suggested a preliminary estimate of the current size of the annual hawksbill nesting population for eastern Arnhem Land of ~2,500 females annually.

In Western Australia the Dampier Archipelago supports the largest hawksbill rookery in Australia (~1,000 females nesting annually; Limpus 2007b), but there are no long-term quantified census statistics to determine population trends or current abundance. Sporadic to low-density nesting occurs over a much wider area, including the Ashmore Reef National Nature Refuge (Guinea 1995). Outside of the ATS region, but with migratory links to the ATS region, hawksbill turtles nest in low density on multiple islands throughout the nGBR and Torres Strait areas (Limpus 1980, Limpus & Miller (2008), with Milman Island historically being one of the largest rookeries. However, current estimates suggest the annual number of nesters at Milman Island is down to ~200, and Bell et al. (2020) predict the species could be extirpated by 2036.

There is limited data on hawksbill movements within the ATS Region and neighbouring countries. Hawksbill turtles from northeast Australia have been recorded in Vanuatu, Solomon Islands, Papua New Guinea and elsewhere in the Great Barrier Reef (Figure 6-1; Miller et al. 1998). One flipper tag recovery from a hawksbill on Milman Island was recovered in Merauke (Figure 9; Miller et al. 1998), but numerous turtles tracked from the Solomon Islands (the closest large rookery outside of the ATS region) did not enter the Gulf of Carpentaria (Figure 10; Hamilton et al. 2015). It appears the movements in and out of the Torres Strait may be limited only to hawksbills from Australian rookeries and foraging areas. This is supported by flipper tag recovery data presented by Limpus (Figure 11; 2007b). While some hawksbills from Timor-Leste move south into the Timor Sea and Western Australia (Figure 12), there is likely substantial movement of this species northwest and northeast into other Indonesian sites.

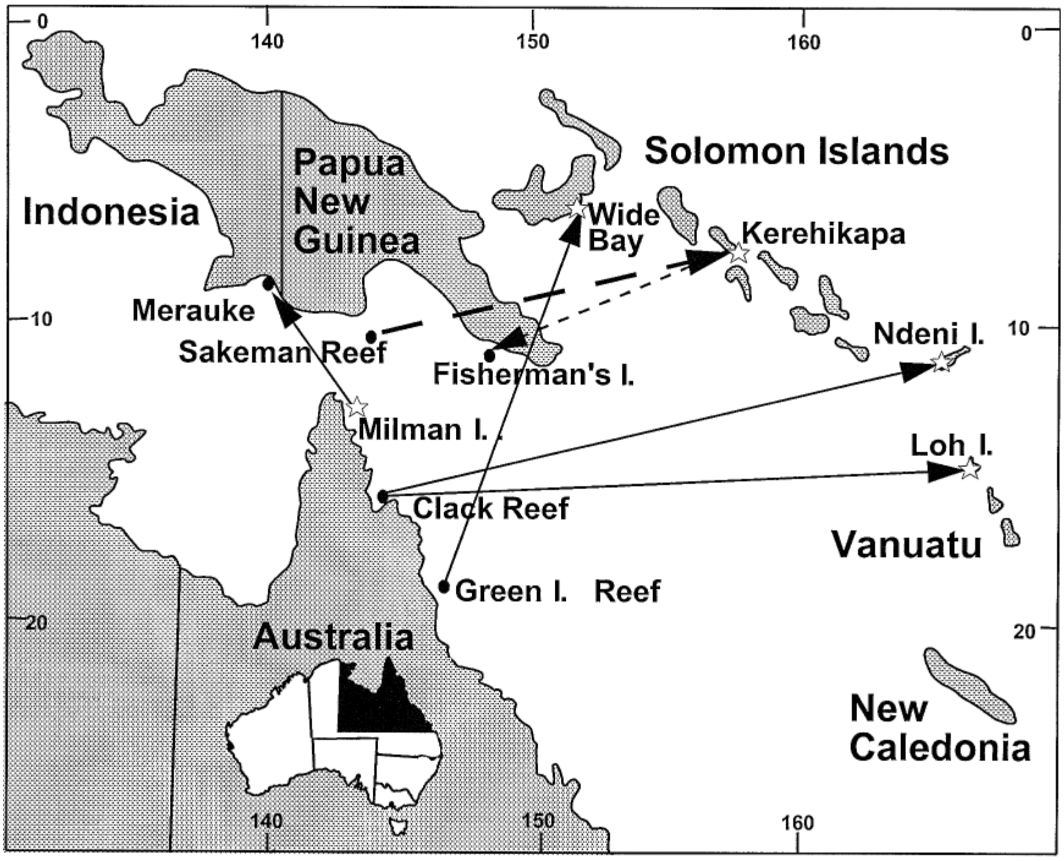


Figure 10. Hawksbill movements from the northern GBR into the Arafura Sea. Image source: Miller et al. 1998

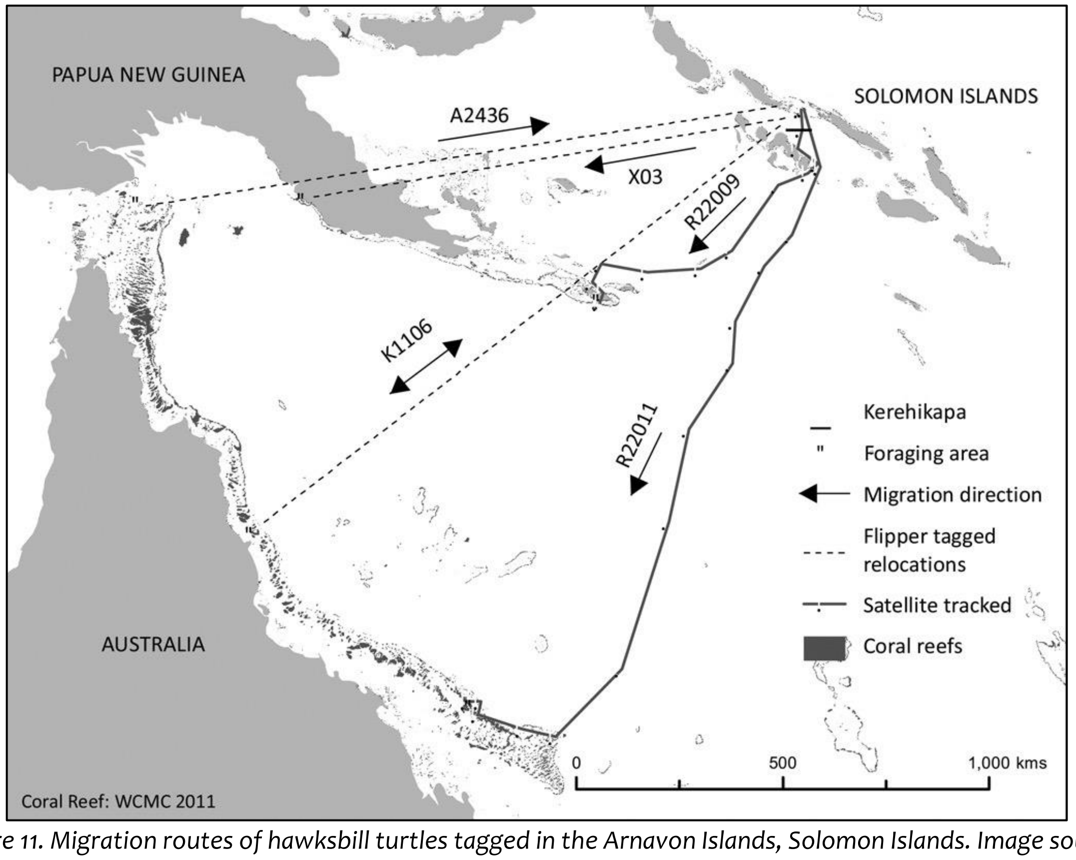


Figure 11. Migration routes of hawksbill turtles tagged in the Arnavon Islands, Solomon Islands. Image source: Hamilton et al. 2015

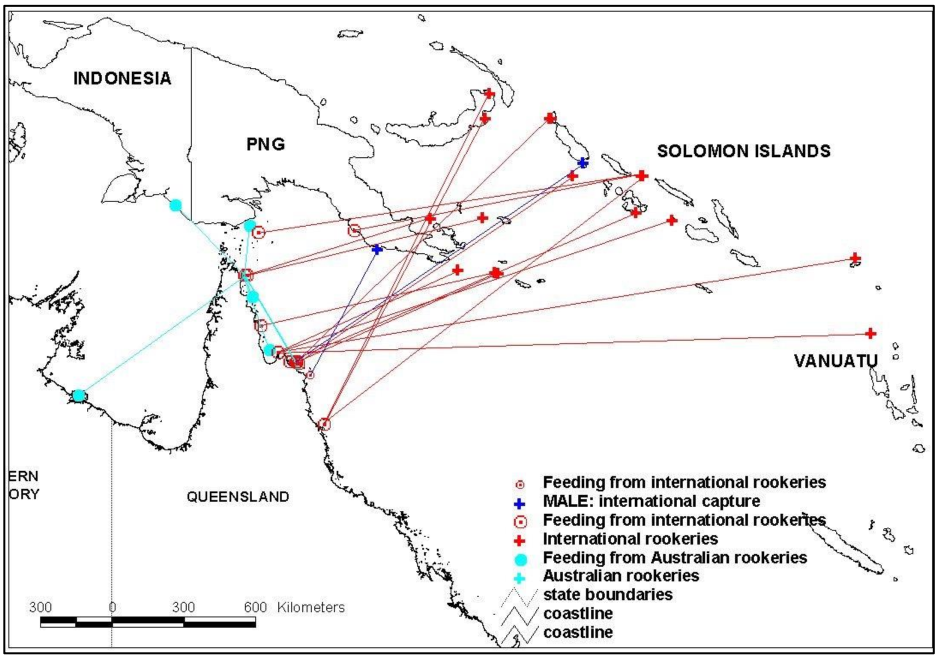


Figure 12. Migration routes of hawksbill turtles tagged in Australia and neighbouring countries. Image source: Limpus 2007b

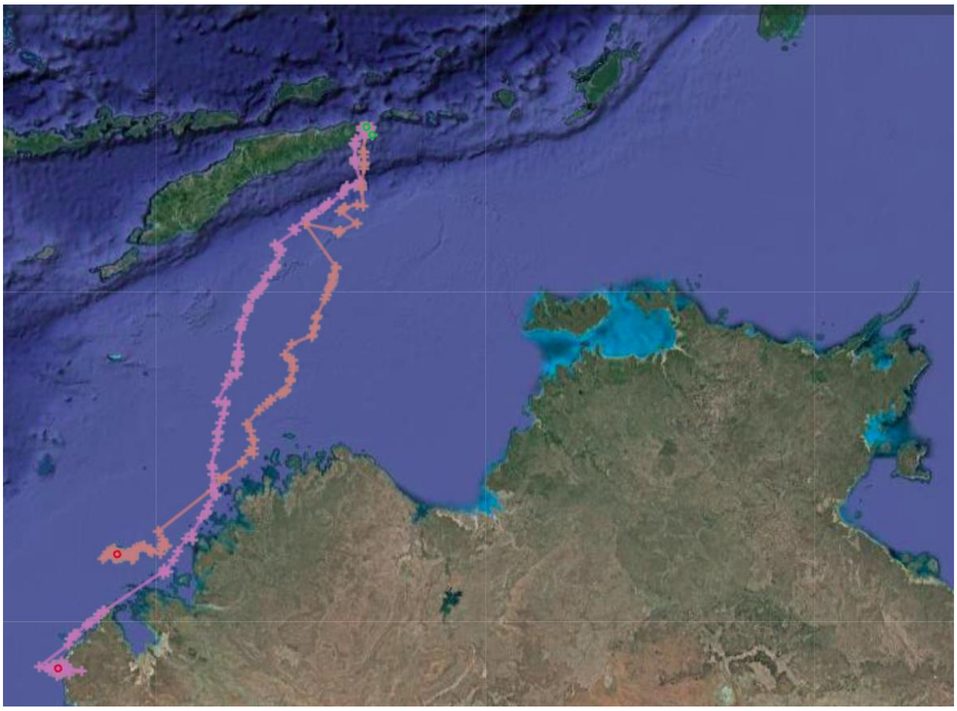


Figure 13. Migration routes of hawksbill turtles tagged in Timor-Leste. Image source: Conservation International, unpublished data; https://zoatrack.org/projects/560/analysis

Limited flipper tagging data also has demonstrated that northeast Australia nesting hawksbills have been found in Papua New Guinea, and nesting PNG hawksbills have been reported within their Australian foraging range. Hoenner et al. (2015) reported that post-nesting female turtles tagged within the Gulf of Carpentaria remained in the Gulf, suggesting minimal dispersal of adult females from these rookeries (Figure 13). This same study also noted that key rookeries likely seed areas in close proximity, and that post-hatchling turtles from these sites might seed areas in the Torres Straits and the northern Coral Sea (Figure 14). This study modelled the dispersal of hatchlings from two key rookeries in Australia and demonstrated how those from North East Island in the Western Gulf of Carpentaria were more likely to remain in the ATS region (Hoenner et al. 2015).

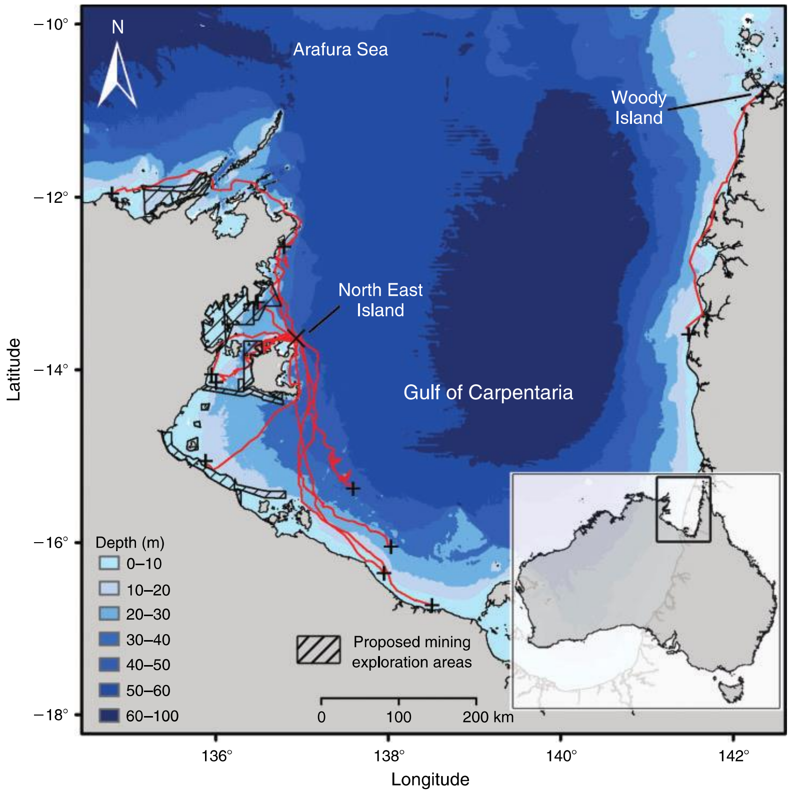
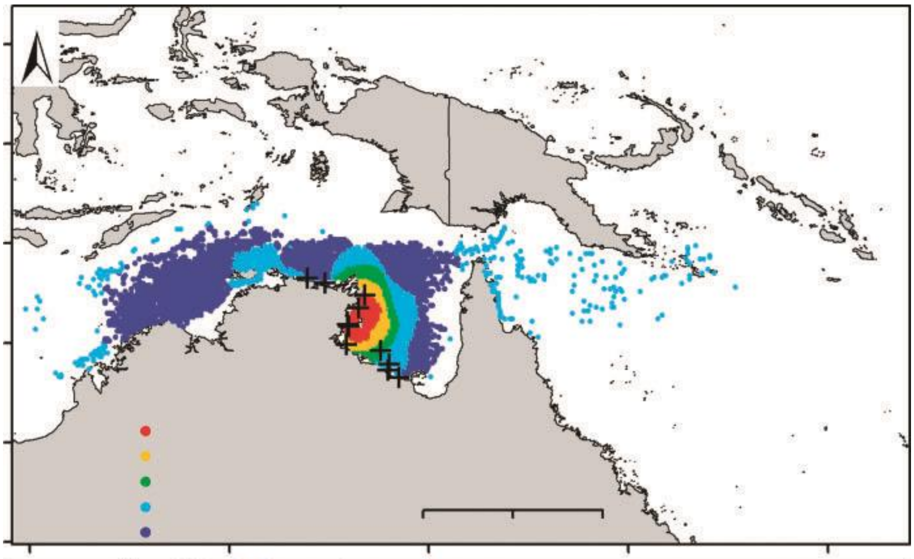


Figure 14. Hawksbill movements within the Gulf of Carpentaria. Image source: Hoenner et al. 2015

6.2 GENETIC STRUCTURE

A study on the global phylogeography of hawksbill turtles was recently undertaken by Arantes et al. (2020). They noted that hawksbill foraging aggregations are typically mixed stocks of individuals originating from multiple nesting areas, but there was also a trend of foraging turtles coming from nearby nesting beaches – that is, little dispersal from hatchling to adult. This study identified that Western Australia, Solomon Islands and Eastern Pacific hawksbills were related – and interestingly this group was also related to the Persian Gulf, while the east Pacific hawksbills formed another group, and a third group occurred in the Northern Territory and North Queensland, Australia (Figure 15). Vargas et al. (2016) noted that hawksbill turtles had a complex pattern of phylogeography, showing a weak isolation by distance and evidence of multiple colonization events. This explains the shared haplotypes across much of the Pacific region (pink colours, Figure 15).



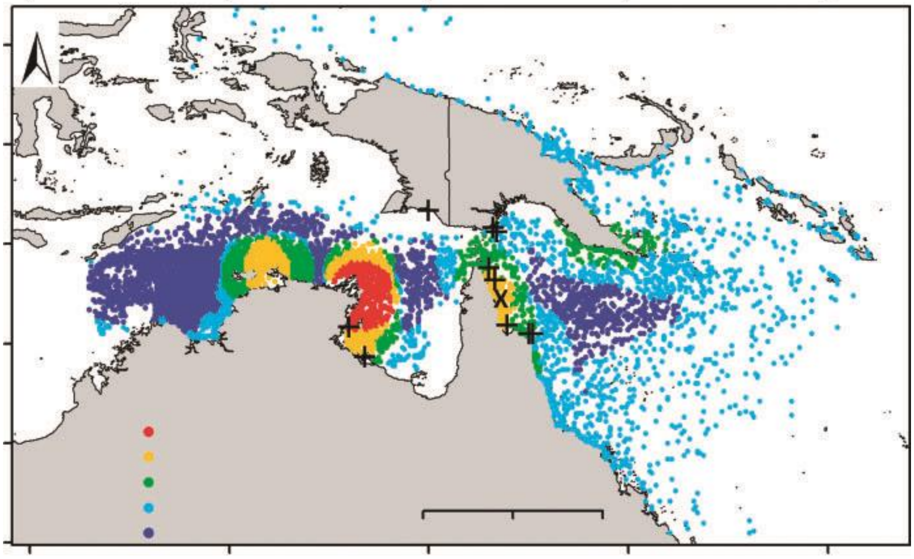


Figure 15. Modelled distribution and clustering probability of post-hatchling turtles from (a) North East Island and (b) Milman Island, whose locations are shown by crosses. Red markers indicate spatial clustering of high probabilities, whereas light blue markers indicate spatial clustering of low probabilities. Image source: Hoenner et al. 2015

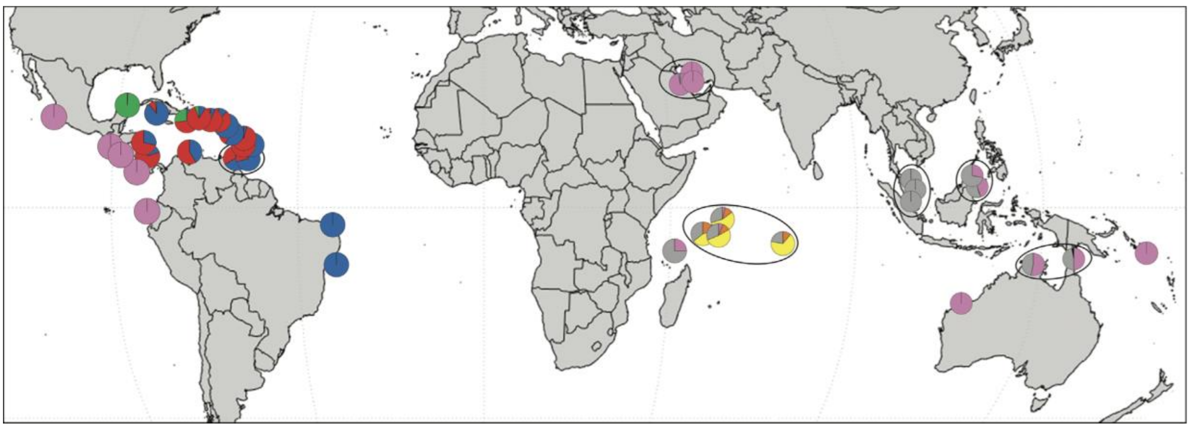


Figure 16. Frequencies of control region haplotypes (739 bp) from each of nine mtDNA lineages in the hawksbill turtle rookeries. Image source: Arantes et al. 2020

Findings by Arantes et al. (2020) were mirrored by findings using mtDNA studies within Australia (Broderick et al. 1994). There are two recognised genetic stocks of hawksbill turtle breeding in Australia (Moritz et al. 2002, Dutton et al. 2002), and each of these stocks supports an annual nesting population of several thousand females (Limpus & Miller 2008). Genetic analysis indicated that there was one stock that incorporated the hawksbill rookeries of the northern Great Barrier Reef (nGBR), Torres Strait and Arnhem Land that was independent of a second stock that breeds at rookeries on the northwest shelf of Western Australia (Broderick et al. 1994). Limpus (2007b) indicates that the GBR and Torres Strait turtles are unlikely to be interbreeding with Arnhem Land turtles given differences in breeding timing.

6.3 POPULATION TRENDS

Data on hawksbill nesting trends are not available for the rookeries within the ATS region given the lack of long-term studies on these smaller rookeries. Hawksbill turtles are difficult to monitor for a number of reasons: (a) small numbers of hawksbills nest on a wide variety of beaches across a broad geographic area; (b) hawksbill beaches tend to be remote, inaccessible and sometimes so narrow that the turtle leaves no crawl trace; and (c) hawksbill turtles exhibit large year-to-year fluctuations in nesting numbers so that single year counts cannot be used to determine trends.

Outside of the ATS region, data are available for three different locations in Indonesia (Alas Purwo National Park, East Java; Jamursba-Medi beach, West Papua; and Sukamade beach, Meru Betiri, East Java (Figure 16; Dermawan 2002), and for one site in Australia.

At the Indonesian sites there has been a general decline in nesting, predominantly due to the harvesting of hawksbill turtles for their shell (Dermawan 2002). In Australia, Milman Island was considered one of the most important hawksbill rookeries (Miller et al. 1995), but has witnessed severe declines in the last three decades (Figure 17; Bell et al. 2020).

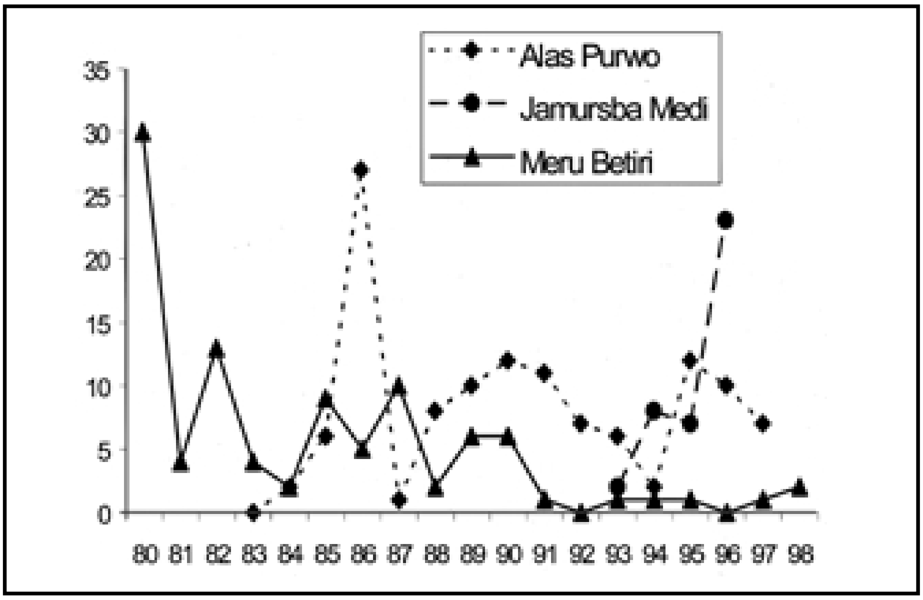


Figure 17. Annual hawksbill nesting at three Indonesia rookeries. Data source: Dermawan 2002

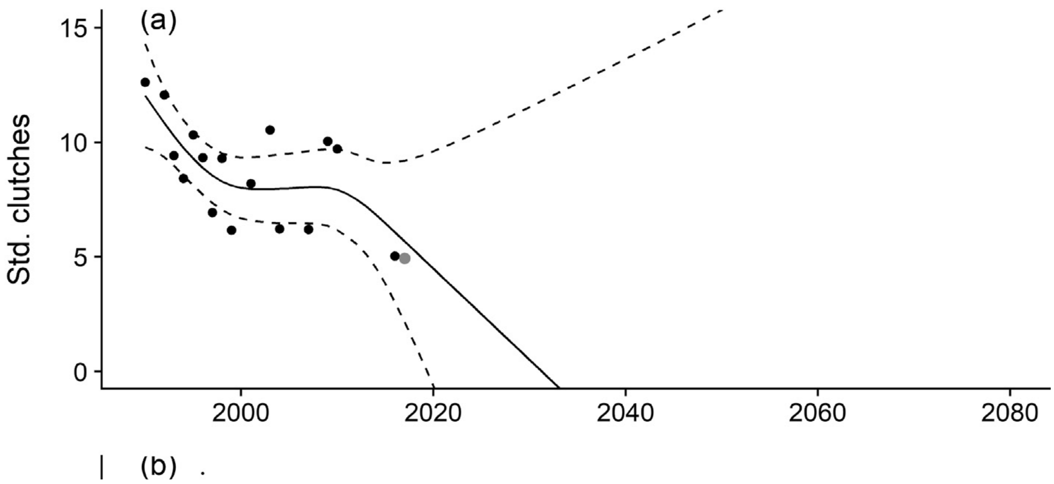


Figure 18. Projected trend in numbers of hawksbills nesting on Milman Island, Australia. Data source: Bell et al. 2020

CHAPTER 7. LOGGERHEAD SEA TURTLES

7.1 DISTRIBUTION & MIGRATIONS

Loggerheads are widespread throughout ATS waters (Figure 18, and Figure 19) but there is no breeding by loggerhead turtles in northern Australia, Indonesia, Papua New Guinea or Timor-Leste. The nearest nesting to the ATS region occurs in central Western Australia, from Shark Bay to the southern Northwest Shelf (Limpus & Limpus 2003). Nesting loggerheads have been flipper-tagged on Dirk Hartog Island nearly every year since 1993-1994 as part of a mark-recapture program started by the Western Australian Marine Turtle Project (Prince 2000, Reinhold & Whiting 2014). Dirk Hartog Island hosts approximately 70% of all loggerhead turtle nesting in WA, with an estimated 1,000 – 3,000 females nesting at this site annually (Baldwin et al. 2003 Limpus, 2007c). It was believed the annual nesting population for the entire stock was of the order of several thousand females (Baldwin et al. 2003), however, during 1998, 1999, 2000 and 2008, over 1,400 turtles were tagged at Dirk Hartog during several two-week peak periods (WAMTP unpublished data, Reinhold & Whiting 2014), indicating annual nesting numbers are likely substantially greater than previously estimated.

Substantial movement has been documented of post-nesting loggerhead turtles into foraging areas in the Arafura and Timor Seas (e.g. Figure 20; Tucker et al. 2020). These turtles followed coastal routes along the Western and Northern Australia coast, and assumed foraging that lay predominantly in waters of North Australia (Figure 21)

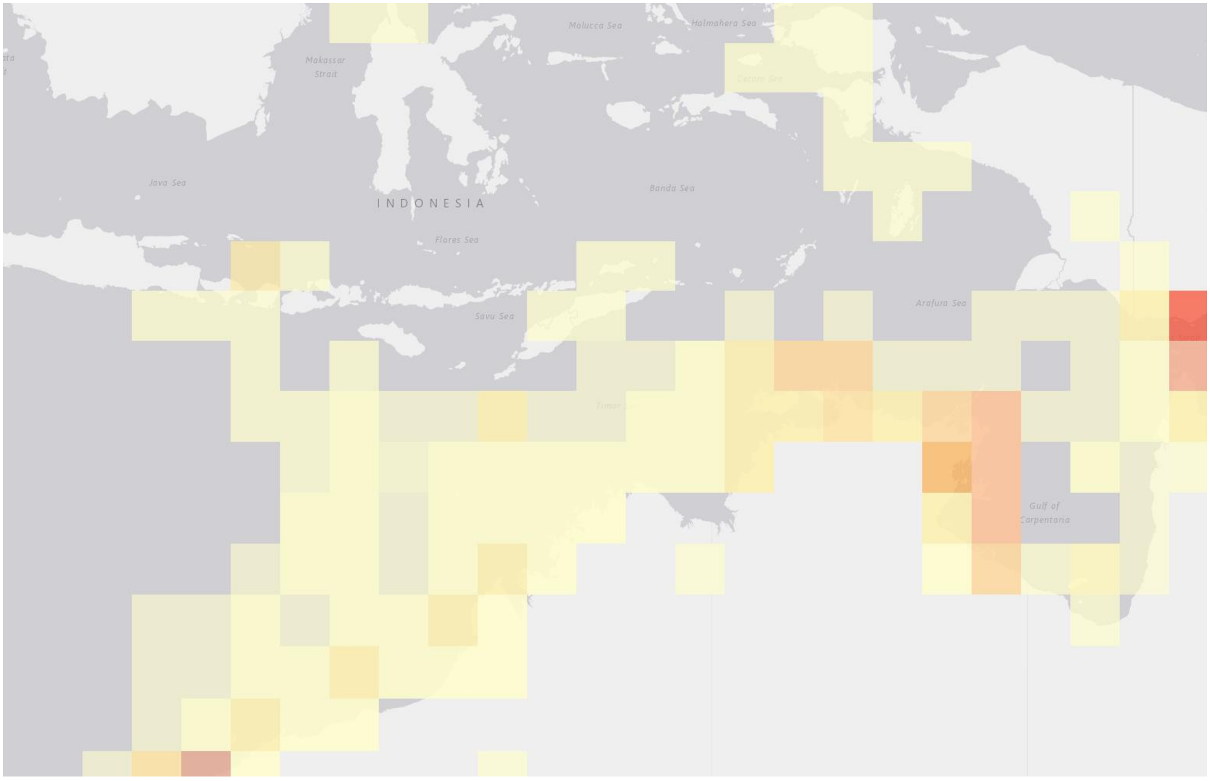


Figure 19. Loggerhead turtle distribution in the ATS region. Darker colours indicate greater number of records per 10 cells. Image source: OBIS-Seamap 2021

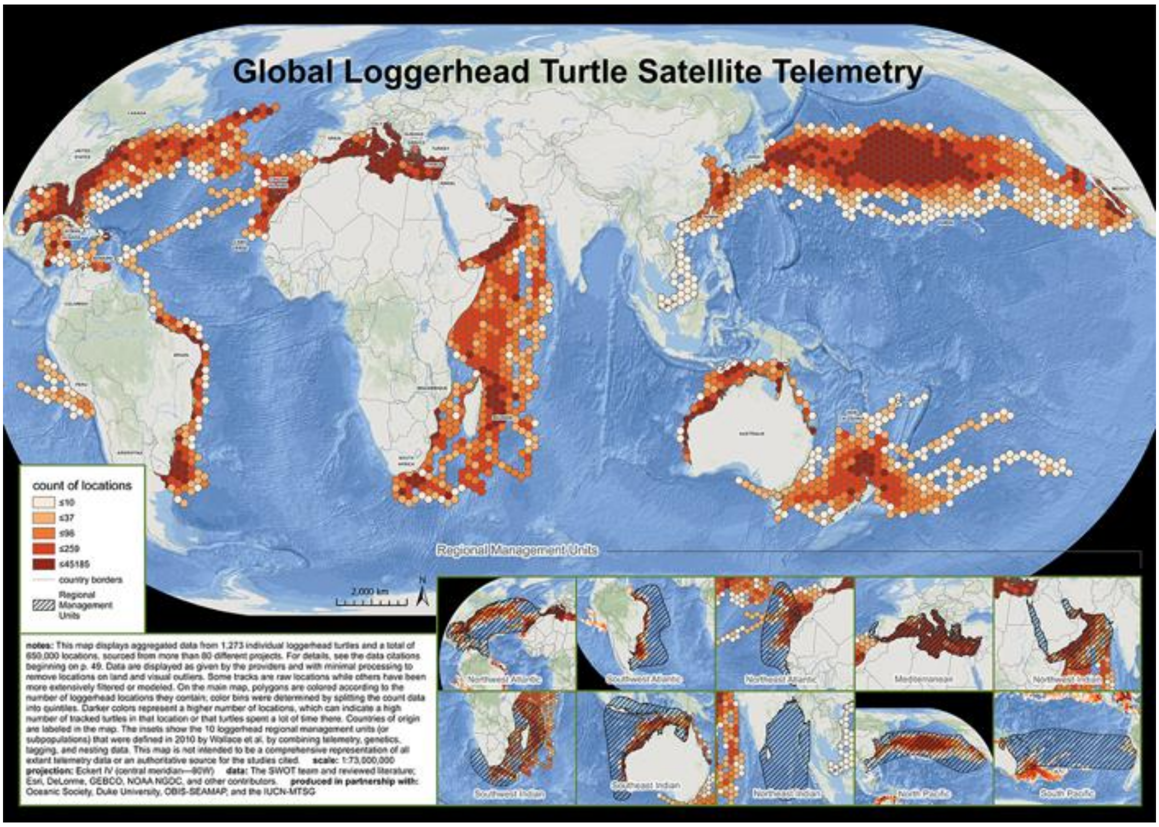


Figure 20. Global satellite telemetry data for loggerhead turtles. Image source: SWOT Report No. XV

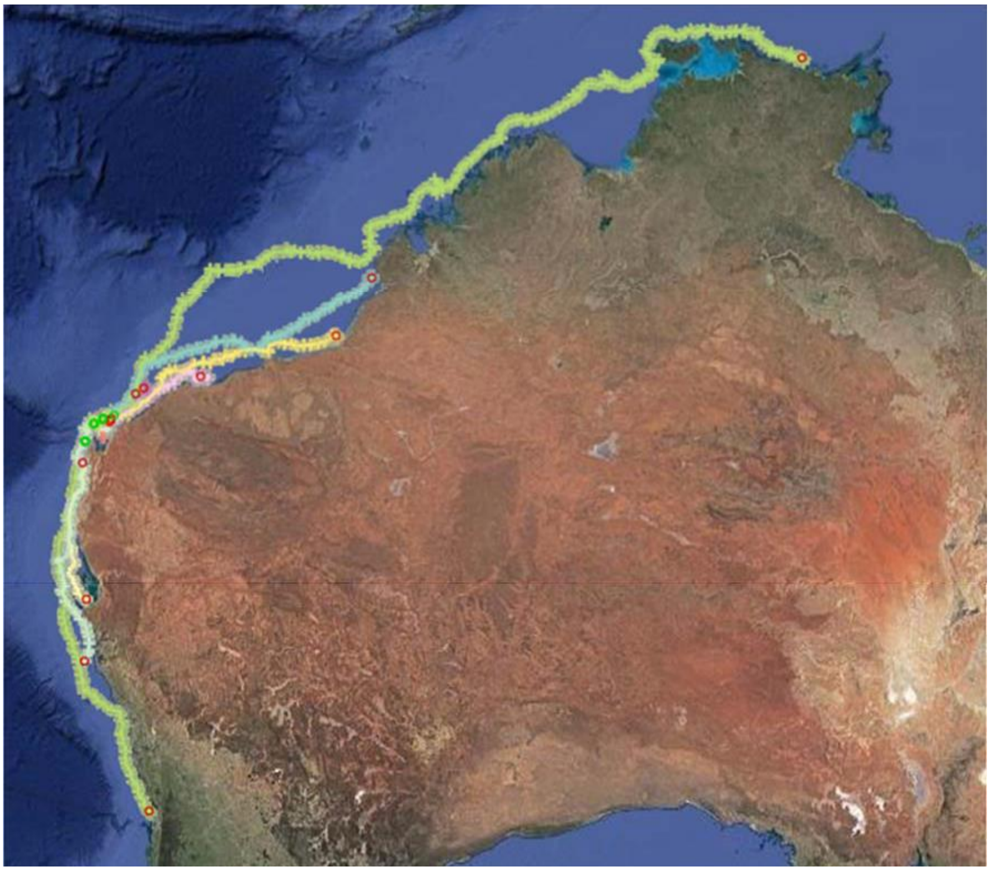


Figure 21. Movements of loggerhead turtles from Western Australia. Image source: Tucker et al. 2020

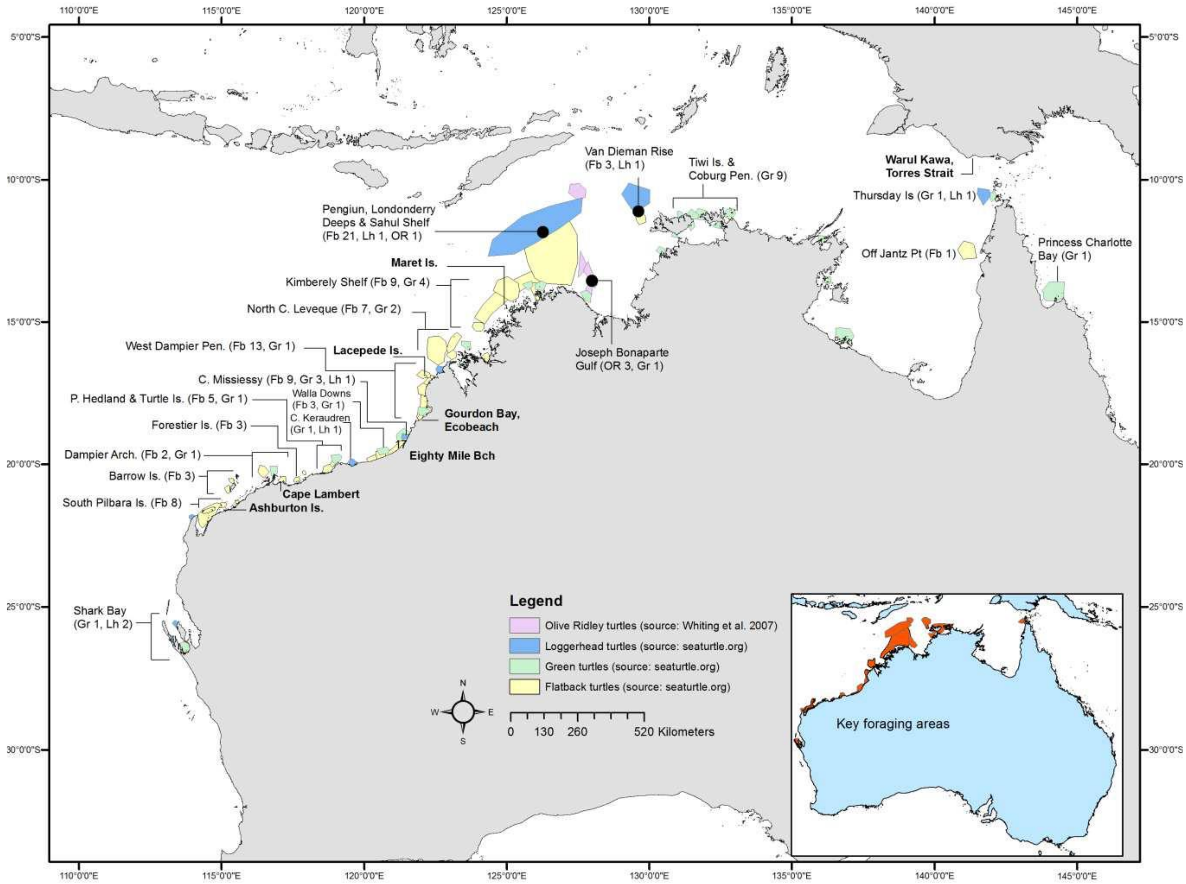


Figure 22. Foraging grounds of turtles tracked from Western Australia. Image source: Waayers et al. 2015

7.2 GENETIC STRUCTURE

Loggerheads of the southeast Indian Ocean are treated as a single RMU (Wallace et al. 2010), and this includes nesting turtles from Western Australia and foraging turtles throughout the ATS region. The nesting populations of the various loggerhead rookeries in Western Australia, from Shark Bay to the southern Northwest Shelf, are treated as a single interbreeding stock and independent of the other stocks that breed in eastern Australia and elsewhere in the east Indian Ocean (Bowen et al. 1994; Dutton et al. 2002, FitzSimmons et. al. 1996).

7.3 POPULATION TRENDS

While the western Australia nesting population is reported to number about 1,000 to 2,000 turtles annually (Baldwin et al. 2003), long-term census data from any index beach from which population trends can be assessed come from only a few sites. At the Northwest Cape (Figure 22; Prince 2000) the trend appears to show a fluctuating but steady trend, at least until 2000. In contrast, Thomson et al. (2016) documented a steady decline from ~840 to ~450 nets at Gnarloo on the mainland coast, and the trend at a State level likely warrants investigation.

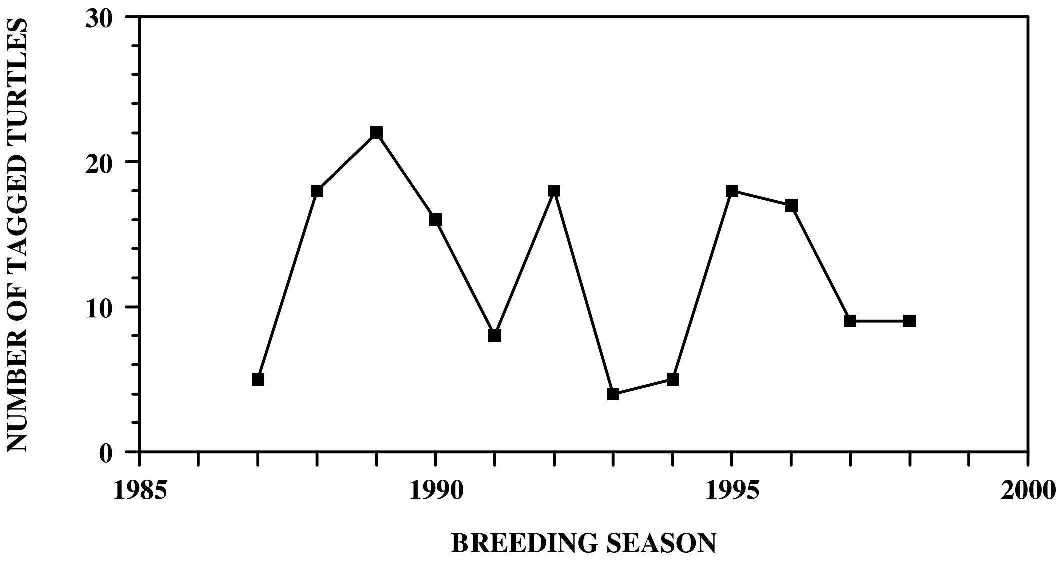


Figure 23. Trend in nesting loggerheads at Northwest Cape, WA. Image source: Prince 2000

CHAPTER 8. LEATHERBACK SEA TURTLES

8.1 DISTRIBUTION & MIGRATIONS

Leatherback sea turtles migrate through ATS waters (Figure 23), and a handful of nesters use beaches in Northern Australia. Scattered nesting may also occur on the beaches along the south coast of Timor-Leste. The leatherback turtle does not nest elsewhere in the ATS region, but the nearest rookery is on Buru Island, located just to the north in the Ceram Sea. The largest western Pacific rookery lies further north in West Papua, Indonesia.

In Australia, low-density nesting has been recorded at Wreck Rock Beaches and Rules Beach, southern Queensland and at the Coburg Peninsula and Arnhem Land, Northern Territory. Sporadic nesting by 0–3 females per year were also recorded on the southern Queensland coast between northern Hervey Bay (Bundaberg) and Roundhill Head (28 nesting attempts recorded from 1968 to 1990) in the late 1970s and early 1980s (Limpus & McLachlan, 1994; Limpus et al. 1984). Nesting appears to have declined since that time (Limpus 2007d). Based on these figures and trends it is estimated <3 turtles nest each year in Australia.

Although outside the ATS region, but with important migratory links (Figure 24), the key nesting beaches in Indonesia are at Jamursba Medi and at Warmon, West Papua province.

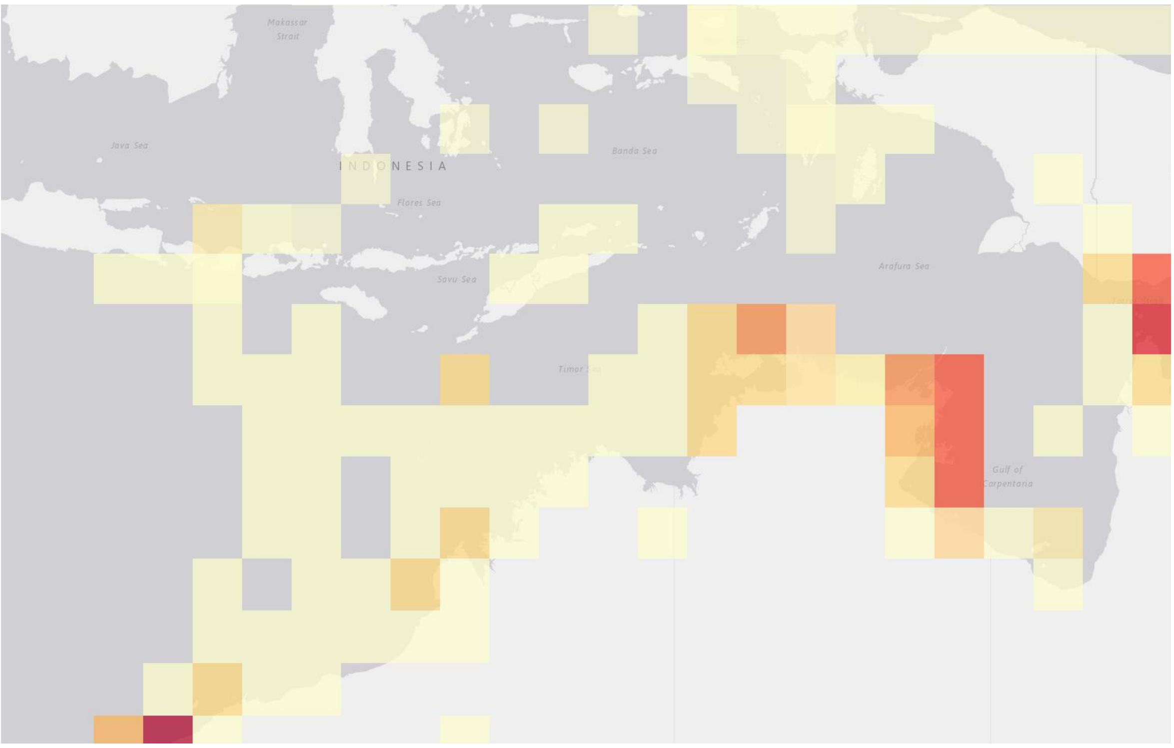


Figure 24. Leatherback turtle distribution in the ATS region. Darker colours indicate greater number of records per 10 cells. Image source: OBIS-Seamap 2021

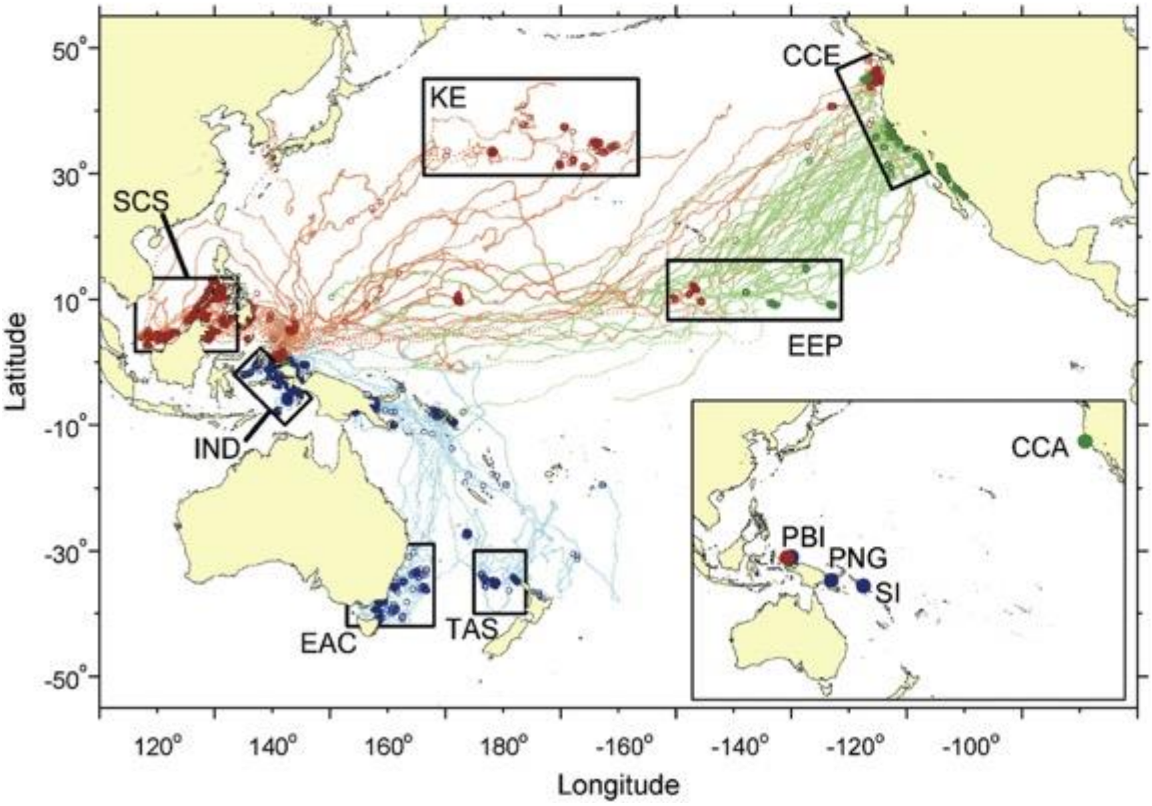


Figure 25. Regional leatherback turtle movements showing migrations of West Papua turtles into the Arafura Sea. Image source: Benson et al. 2011

Hitipeuw et al. (2007) recorded 1,865 and 3,601 nests at Jamursba Medi in 2003 and 2004, respectively; and 1,788 and 2,881 nests are Warmon in 2003 and 2004. Hitipeuw et al. (2007) surmised the number of annual nesters at Jamursba Medi was 501 to 660 in 2003; and 667 to 879 in 2004 after adjusting for season length. This number has continued to decline, and the number of females nesting annually as of 2011 was estimated to be <400 during the boreal summer and 131 during the austral summer, based on estimated clutch frequency and clutch interval (Tapilatu et al. 2013). However, most recently in 2021, 2,500 nests were recorded in Jamursba Medi and Wermon (Lontoh pers. comm.). In 2017 WWF-Indonesia started annual surveys of the 12.4 km beach of Fenaleisela, Buru Island and recorded an average of 250 leatherback nests per year (Suprapti pers. comm.).

Leatherbacks from Papua New Guinea or Indonesia generally do not move into the ATS region, but a small proportion of leatherbacks do move down into the Arafura Sea (Benson et al. 2011). While turtles nest year round on Buru island, there are distinct peaks in the cold-season (NW monsoon, November/December) and in the hot-season (end of the SE monsoon (June/July). The cold-season nesters migrate to the south, towards the Lesser Sunda islands and into the Timor Sea / Indian Ocean, while the hot-season nesters move northwards towards the Sulu-Sulawesi region (Suprapti pers.com).

There are several accounts of leatherback turtles caught as by-catch by Timorese fishing boats operating off the Timorese coast. There are also several reports of leatherback turtles coming ashore on the south coast of Timor-Leste (Amaral pers. comm.).

8.2 GENETIC STRUCTURE

The west Pacific leatherback turtle is considered a single RMU (Wallace et al. 2010). In the west Pacific, genetic analysis by using mitochondrial deoxyribonucleic acid sequences identified a total of six haplotypes among the 106 samples analysed for Solomon Islands, Papua, and Papua New Guinea, including a unique common haplotype that is only found in the western Pacific populations (Dutton et al. 2007). The genetic signature of the Buru Island leatherback turtles has yet to be determined.

8.3 POPULATION TRENDS

No data is available on trends of nesters in Australia, but Limpus (2007d) reported these to be declining. Outside of the ATS region, there has been a continuous decline in nesting of leatherbacks in West Papua (Figure 25), attributed in large part to terrestrial predators (Hitipeuw et al. 2007, Tapilatu et al. 2013).

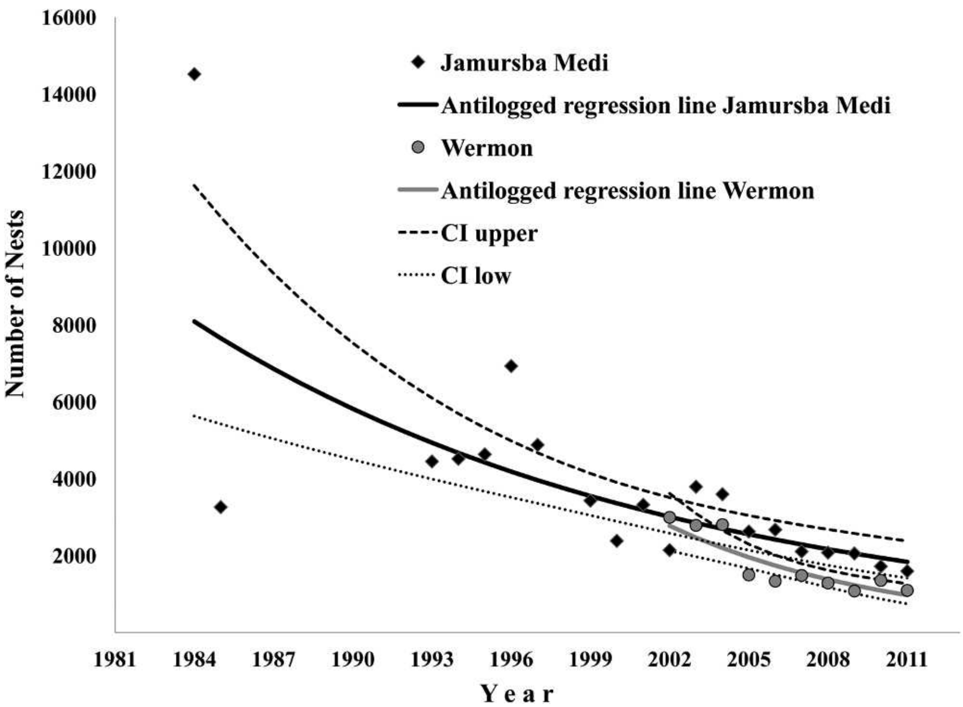


Figure 26. Decline of leatherback nesting at Jamirsba Medi. Leatherback nesting abundance (number of nests) trend at Jamursba Medi from 1984–2011 and Wermon from 2002–2011. Image source: Tapilatu et al. 2013

CHAPTER 9. OLIVE RIDLEY SEA TURTLES

9.1 DISTRIBUTION & MIGRATIONS

Olive ridley turtles are moderately abundant in the ATS region (Figure 26), and nest on beaches in Australia, Indonesia and Timor-Leste. However, nesting is dispersed and of low volume, and sometimes confounded with hawksbill turtle nesting.

In Australia, olive ridley nesting occurs from the western coastline of Cape York, in the east, westward to Fog Bay, NT (Whiting 1997). Olive ridleys have been recorded nesting on both mainland and on island beaches, but mainly on islands (Chatto & Baker 2008). Low-density nesting occurs along the northwestern coast of Cape York Peninsula between Weipa and Bamaga (Limpus & Roper, 1977; Limpus et al. 1983, Limpus 2007e). The balance of nesting occurs in the Northern Territory with minor nesting in Western Australia (Limpus 2007e). Over most of their range in the Northern Territory (which includes little of the western coast of the NT) they nest in low numbers. However, on some beaches (e.g. along the northern coast of the Tiwi Islands and some islands in north eastern Arnhem Land) they nest in nationally significant numbers in the order of several hundred nesters (Whiting et al. 2007a, Chatto & Baker 2008). The majority of nesting occurs in the Northern Territory, and it is likely that the total annual nester abundance in the north and east reaches of Australia is ~500 turtles per year. Olive ridleys also forage in north Australian waters (Prince et al. 2010; see also Figure 22) based on direct observations and captures in fisheries.

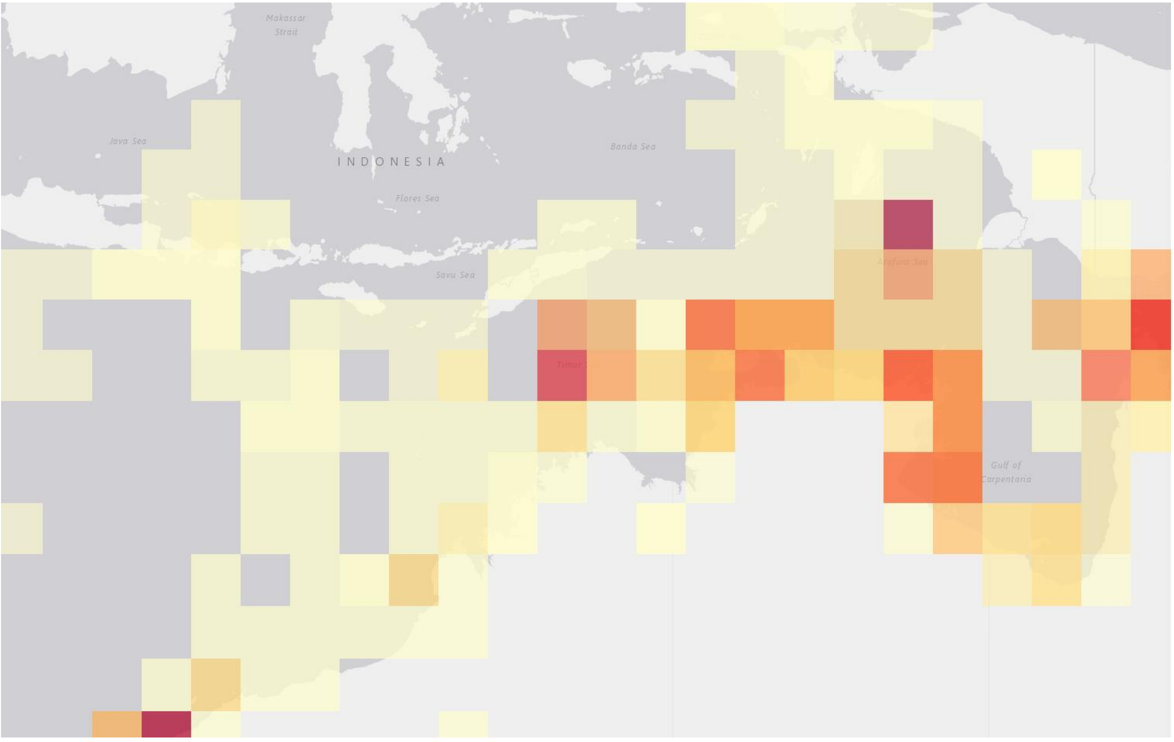


Figure 27. Olive ridley turtle distribution in the ATS region. Darker colours indicate greater number of records per 10 cells. Image source: OBIS-Seamap 2021

Spring (1982) recorded olive ridley nesting in Papua New Guinea. However, none is known for the islands and coastline fronting the ATS region.

In Timor-Leste, olive ridleys nest on Jaco Island and the beaches of Com, Tutuala and Lore (Amaral pers. comm.) between February and August. While there are no publications describing olive ridley nesting in Timor-Leste in the published literature, an olive ridley turtle nesting in the Nino Konis Santana National Park was tracked with a satellite transmitter moving through the Timor Sea and south to Western Australia (Figure 27a). Nest protection programmes are run by local communities in the east of the country and coordinated by CI. Since 2018, these programmes are providing the first species specific nesting data for Timor-Leste.

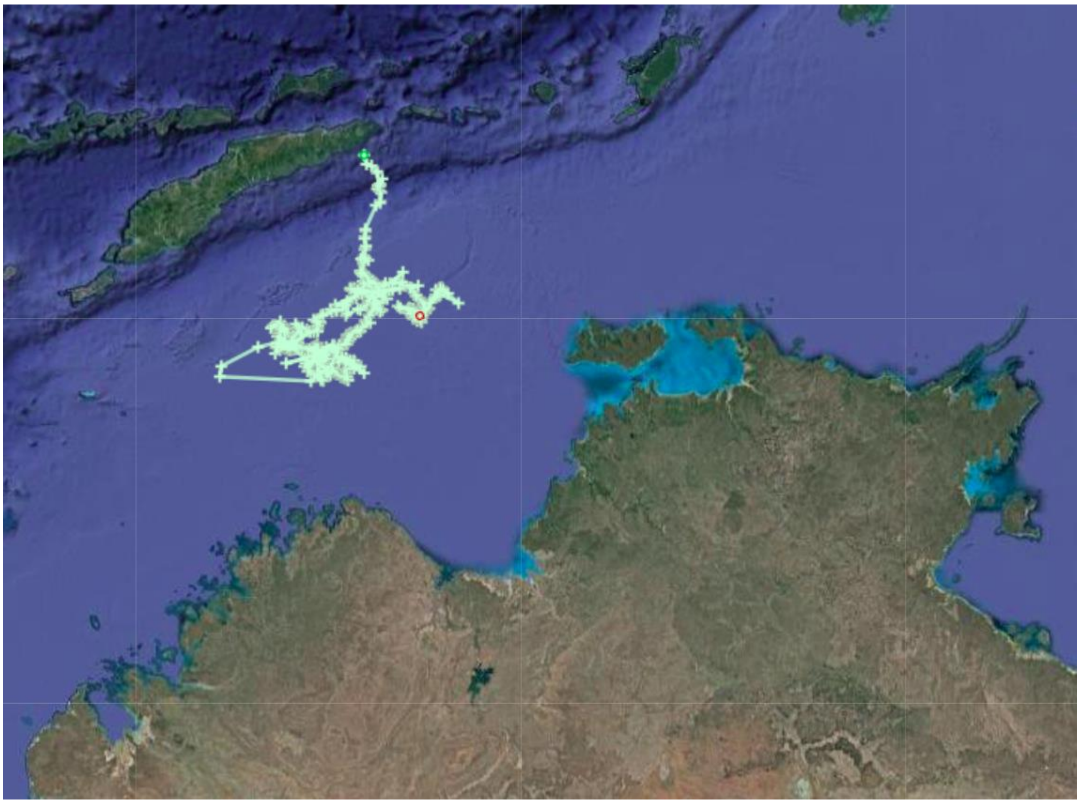


Figure 28. Migration routes of an olive ridley turtle tagged in Timor-Leste. Image source: Conservation International, unpublished data; https://zoatrack.org/projects/560/analysis

There are records of olive ridley turtles from West Papua moving into the Arafura Sea (Doi et al. 2019; Figure 28), and several studies that indicated Australian olive ridley turtles may remain in Australian waters (e.g. McMahon et al. 2007; Figure 29a, Whiting et al. 2007b; Figure 29b). Post-nesting olive ridleys from the Crocodile islands moved northward into the ATS (Dethmers *et al.* 2016). Analysis of combined tracking data of 27 olive ridleys released from various locations throughout Indonesia and north Australia (including those from the studies mentioned above), appear to indicate some high-density aggregation areas in the ATS (Figure 29c).

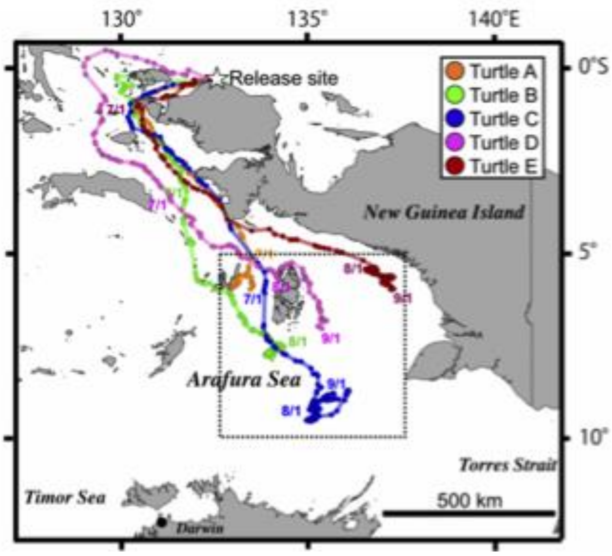


Figure 29. Olive ridley turtle movements from West Papua into the Arafura Sea. Image source: Doi et al. 2019

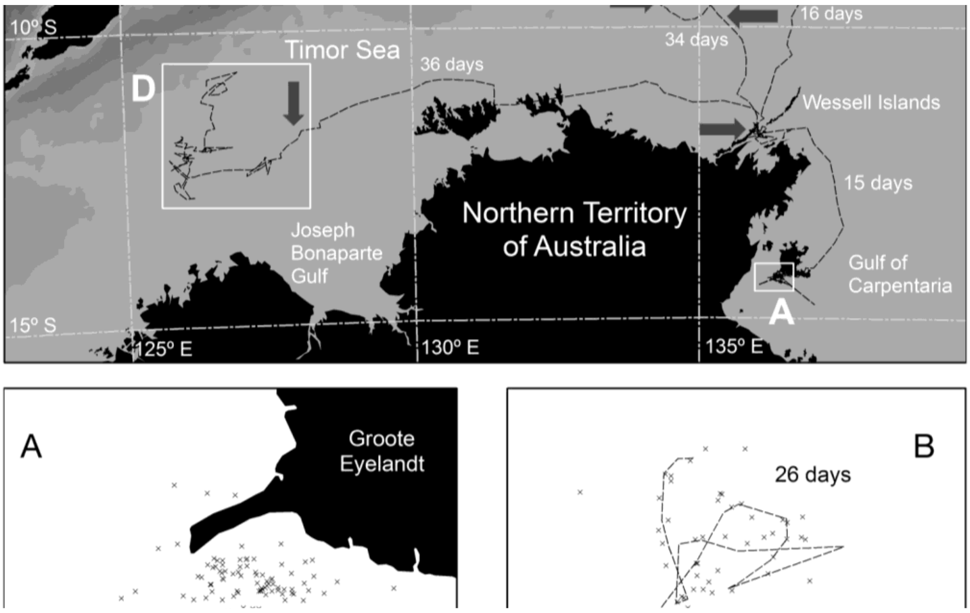


Figure 30a. Movement patterns during post-nesting migration and foraging of 4 olive ridley turtles tracked from the Wessell Islands in the Northern Territory of Australia. Image source: McMahon et al. 2007

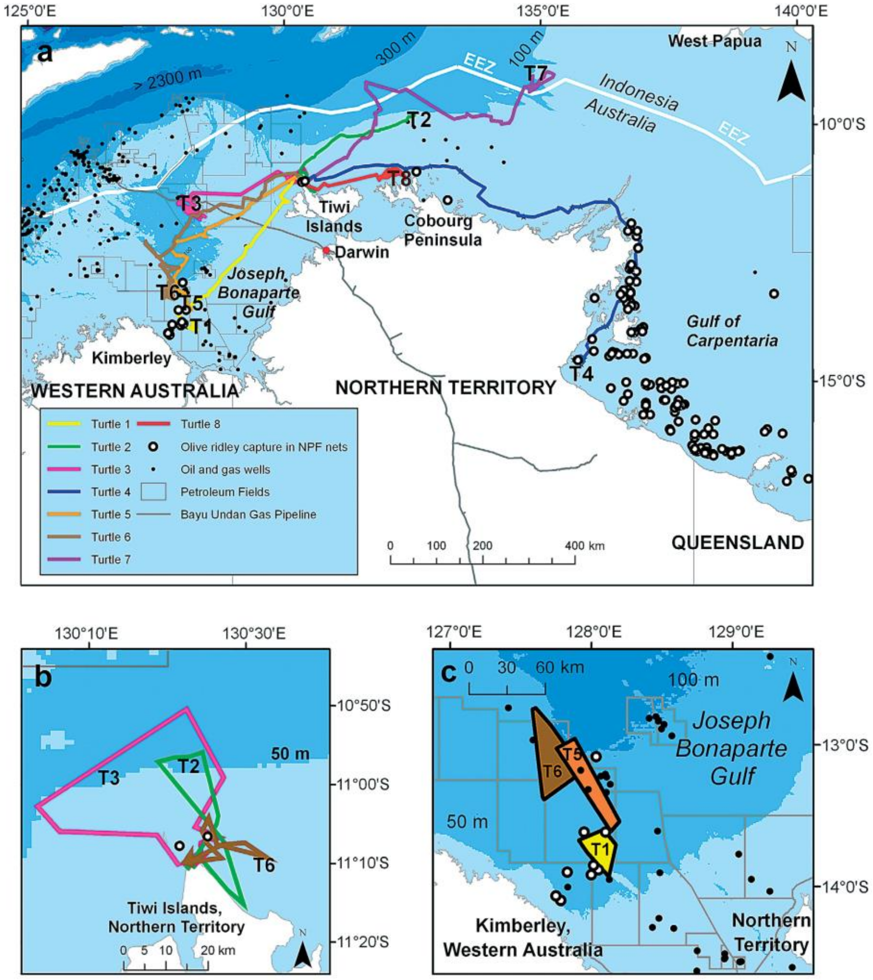


Figure 31b. Post-release movements of eight Olive ridley turtles from on Turtle Melville Island, northern Australia, in 2004 and 2005. Image source: Whiting et al. 2007b

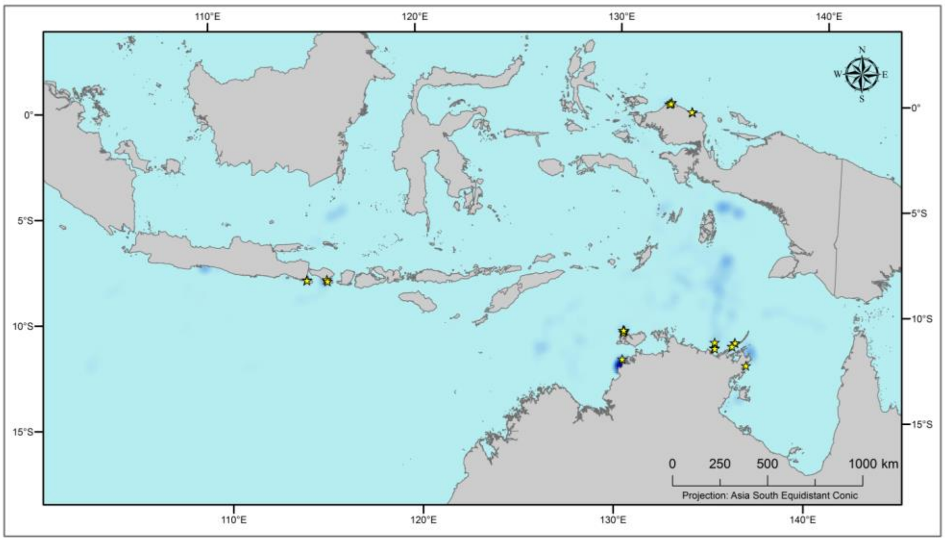


Figure 32c. Post-release movements of eight Olive ridley turtles from on Turtle Melville Island, northern c. High-density areas for olive ridleys in the ATS based on post-release movements of 27 olive ridley turtles (Dethmers et al. 2016)

In Indonesia, olive ridley turtles were not recorded in the ATS by Tomascik et al. (1997), but nesting has been documented in Tuafanu and in Kwatisore Cenderawasih Bay (based on samples collected by Madduppa et al. 2021). Dethmers (2010) indicated olive ridleys did not nest in the Aru islands. Olive ridley nesting is better known in Indonesia at sites outside of the ATS region (e.g. Alas Purwo National Park) and within the ATS the olive ridley likely nests at a handful of small and diffuse rookeries.

9.2 GENETIC STRUCTURE

Bowen et al. (1998) demonstrated strong geographic partitioning of mtDNA lineages between the Indo-West Pacific region and the East Pacific. Few studies have looked at genetics of olive ridleys in the west Pacific, primarily because of their diffuse nesting. A recent study was conducted to determine the genetic structure and population connectivity of olive ridley turtles across the Indonesian archipelago, that indicated the Indonesian olive ridley stocks were highly structured (Figure 30). While Australian and east Indonesian olive ridleys shared many of the same haplotypes, there appeared to be substantial differences between the two countries (Madduppa et al. 2021).

This was supported by genetic analyses of olive ridleys entangled in ghost nets in Northern Australia, which indicated the turtles came from nesting populations within the Northern Territory, but also that haplotypes not found in the Northern Territory were recorded, suggesting turtles may have come from Indonesia, Timor-Leste or Papua New Guinea (Jensen et al. 2013).

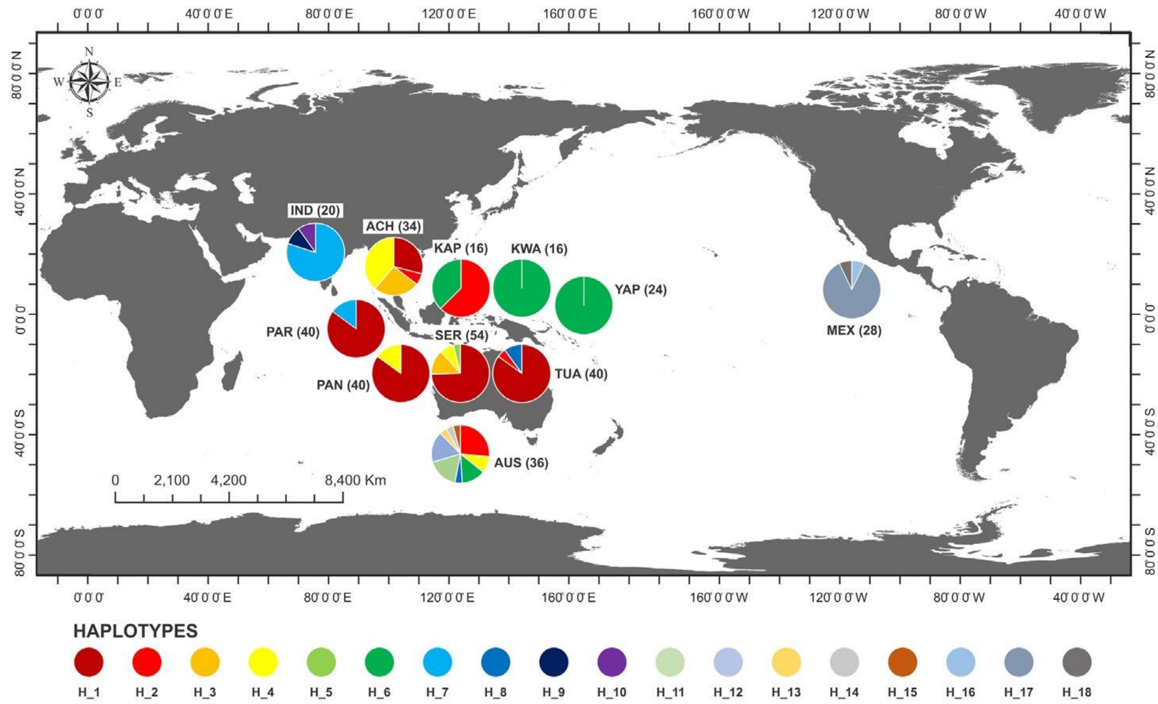


Figure 33. Haplotype distribution across the Indonesian olive ridley population. Pie charts represent the proportion of haplotypes defined in the network at each site. Image source: Madduppa et al. 2021

9.3 POPULATION TRENDS

There are no long-term studies on the olive ridley in the ATS region and no indication of population trends. Outside of the region, olive ridley numbers have increased in the Alas Purwo National Park, attributed primarily to conservation efforts, including nest relocation (Kurniawan & Gitayana 2020). There are no indications that olive ridley turtle numbers in the ATS region have increased or decreased, but accidental capture in fisheries and entanglement in ghost nets appears to be frequent and is cause for concern. In Australia, there was an estimated 90% loss of nests to pig predation on western Cape York (Limpus 2009) that is currently being addressed through predator control programmes.

CHAPTER 10. FLATBACK SEA TURTLES

10.1 DISTRIBUTION & MIGRATIONS

The flatback turtle is unique in that is nests only in Australia, with some northward distribution of foraging grounds. Foraging flatbacks have been encountered in neighbouring Papua New Guinea and Indonesia but no nesting records for this species exist in those countries. It is presumed that flatbacks also forage in waters of Timor-Leste (Figure 10-1, 10-2). Of relevance to the flatback populations in the ATS region are the Arafura Sea / Gulf of Carpentaria / Torres Strait flatback turtles, where the largest nesting sites for flatbacks include Crab Island, Deliverance Island and Kerr Island in the east; and the flatback turtles nesting at Cape Dommet in Western Australia and those nesting down to the Kimberly Islands. The largest flatback rookery in Queensland is on Crab Island just off the Northwest coast of Cape York Peninsula, Australia. Annual nesting numbers were reported as ~1,000 to 2,000 female turtles a year (Commonwealth of Australia 2017). However, this is likely a gross underestimation given recent studies by Leis (2008), who recorded 6,684 nesting events between August 27 and September 27, 2008. Deliverance Island (Warul Kara) hosts ~100-200 flatback turtles annually (Hamann et al. 2015). Between ~600 and ~1,000 nests are also laid in the Jardine River rookery, equating to some 200 to 500 annual breeders (Freeman et al. 2015). Limpus et al. (2016a, 2017a) recorded over 500 flatback clutches on Mapoon beaches in 2016, and over 600 clutches in 2017, highlighting the importance of these western Cape York peninsula beaches for flatback turtles.

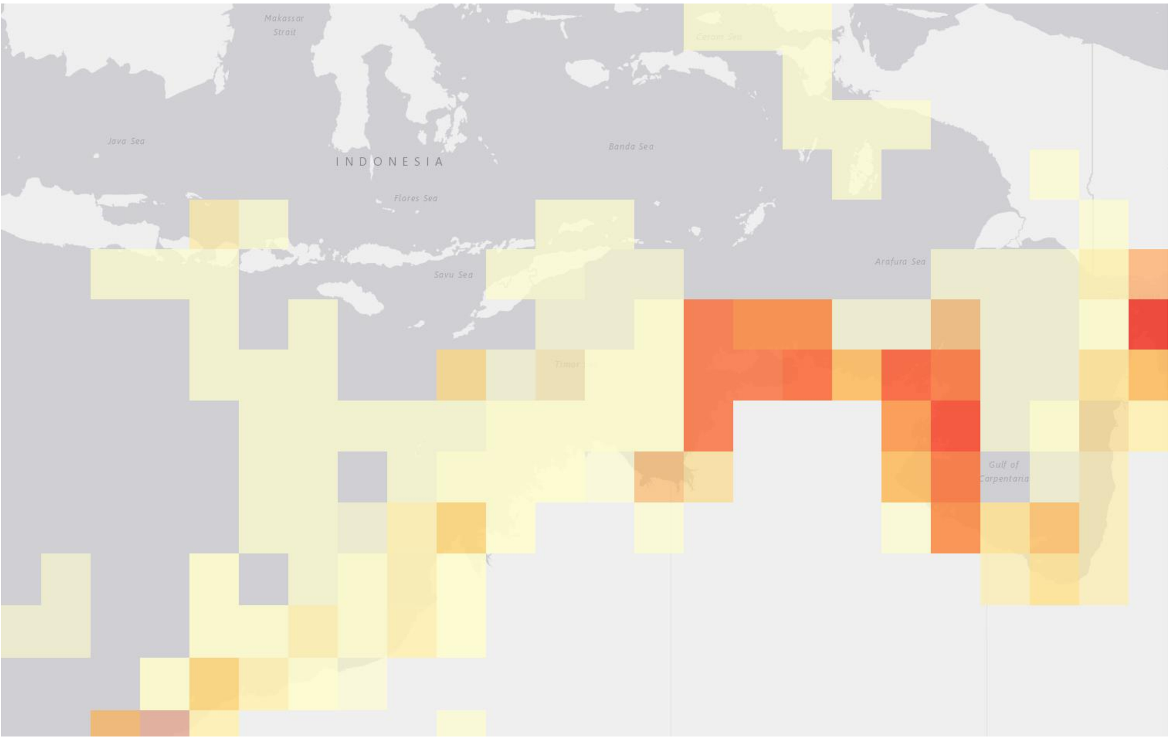


Figure 34. Flatback turtle distribution in the ATS region. Darker colours indicate greater number of records per 10 cells. Image source: OBIS-Seamap 2021

In the Northern Territory, flatback Turtles were recorded nesting all around the coast, on both mainland and on islands (Chatto & Baker 2008). Some 1,600 nests were recorded between 1991 and 2004 on islands, while mainland beaches recorded only ~200 (Chatto & Baker 2008). Flatbacks have been recorded nesting on nearly every Northern Territory beach where marine turtle nesting was confirmed, regardless of whether or not other species also nested at that location. Cape Domett supports one of the largest nesting flatback turtle populations with annual abundance in the order of several thousand individuals (estimated = 3,250, 95% CI = 1431–7757; Whiting et al. 2008). In the Kakadu National Park, Groom et al. (2017) calculated the number of flatbacks to be between 97 and 183 turtles per year with no significant trend over 12 years of monitoring. At Bare Sand Island, the estimated total number of turtles varied from 54 to 160 between 1996 and 2020, with no signs of population size change (Guinea 2020).

Western Australia also supports substantial flatback turtle nesting accounting for approximately one third of the total nesting flatbacks in Australia. There are two genetic stocks of flatback turtles, of which the northern stock, which breeds mainly at Cape Domett and presumably adjacent areas in western Arnhem Land (FitzSimmons et. al. 1996; Dutton et. al. 2002), is most pertinent to the ATS region. The southern stock, which nests throughout the Northwest Shelf from Exmouth to about the Lacepede Islands is linked to the ATS region via migratory data, with numerous Western Australian turtles foraging, and migrating through, in the Timor and Arafura Seas.

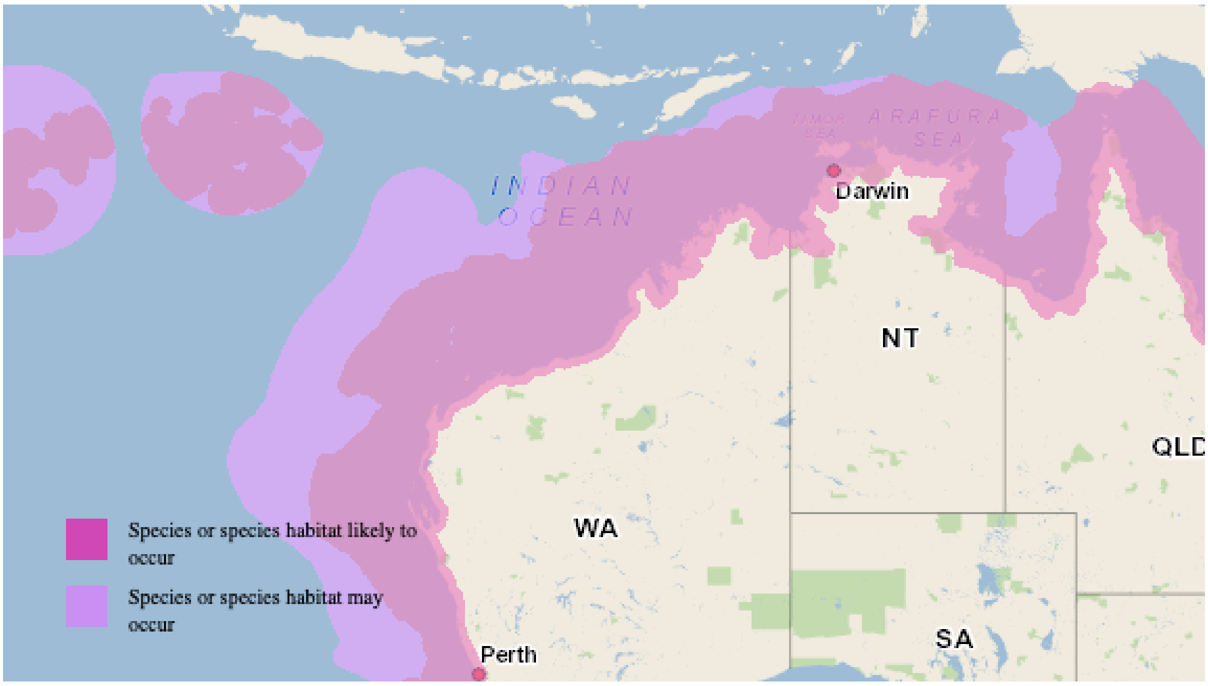


Figure 35. Flatback turtle distribution in the northern Australian region. Image source: Australia Species Profile and Threats Database, Accessed May 26, 2021

Due to their non-oceanic nature, whereby flatback turtles are restricted to Australian waters and those of southern Papua New Guinea and Indonesia, the migration and habitat connectivity data for this species is limited mostly to the Australian continental shelf and the Timor Sea. Flatbacks from the Lacepede Islands forage in the Timor Sea in average water depths of 74 ± 12 m, 135 ± 35 km from the Australian shore (Figure 33; Thumbs et al. 2017). Movements of post-nesting female flatbacks from Torres Straits all oriented to the west into the Arafura and Timor Seas and not to the east (Figure 10-4; Hamann 2015). Thums et al. (2018) also recorded movements of 35 flatback turtles from Bells Beach, ~38 km northeast of Karratha, and Delambre Island, ~18 km north of Bells Beach moving northeast into the Timor Sea (Figure 35).

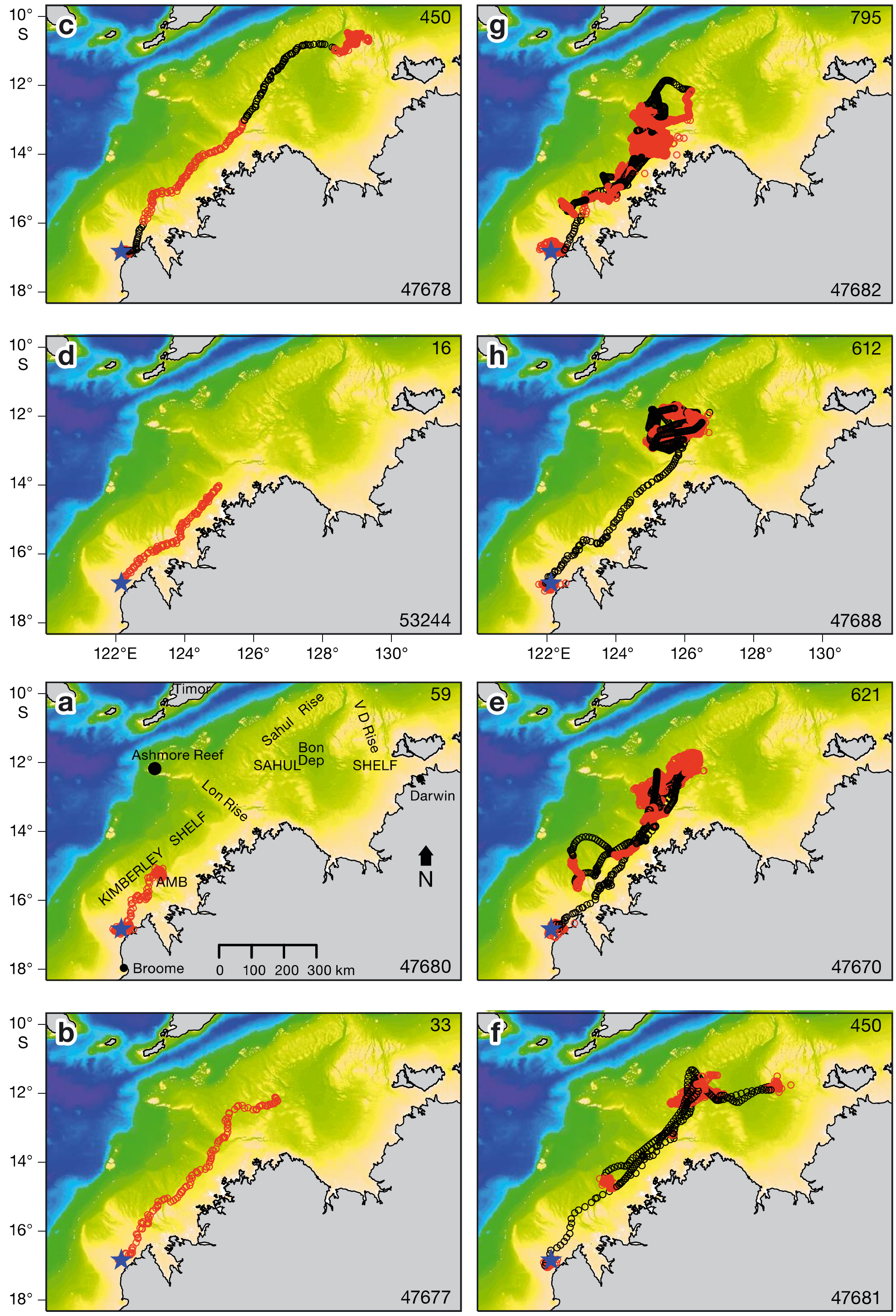


Figure 36. Flatback turtle dispersal from the Lacapede Islands in Western Australia. Image source: Adapted from Thums et al. 2017

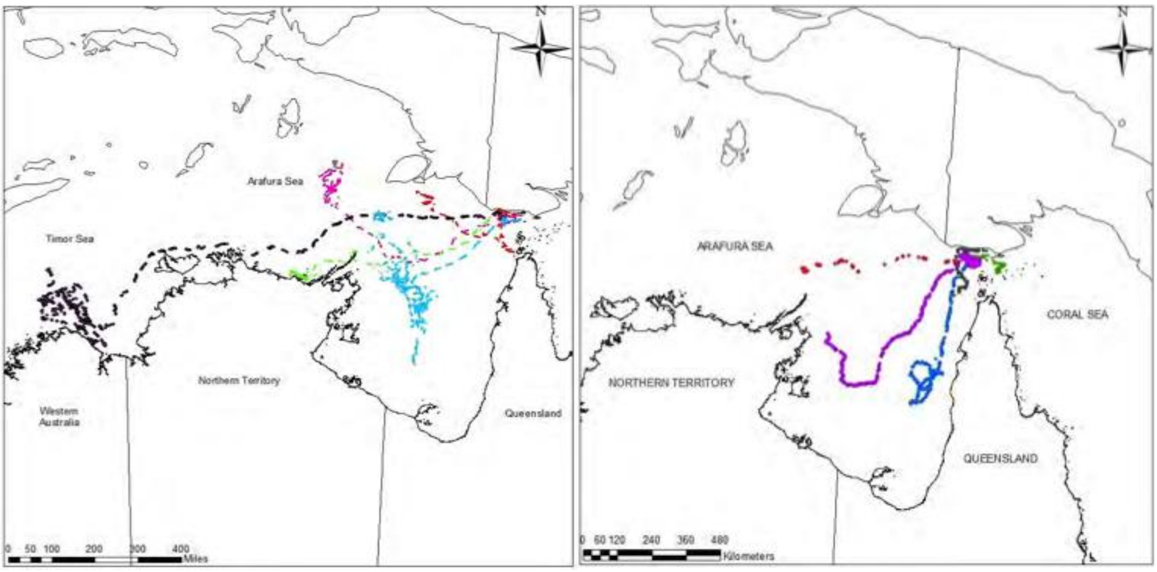


Figure 37. Migration routes and foraging areas for five female flatback turtles after nesting at Warul Kawa in 2013 (left) and six female flatback turtles after nesting at Warul Kawa in 2014. Image source: Hamann et al. 2015

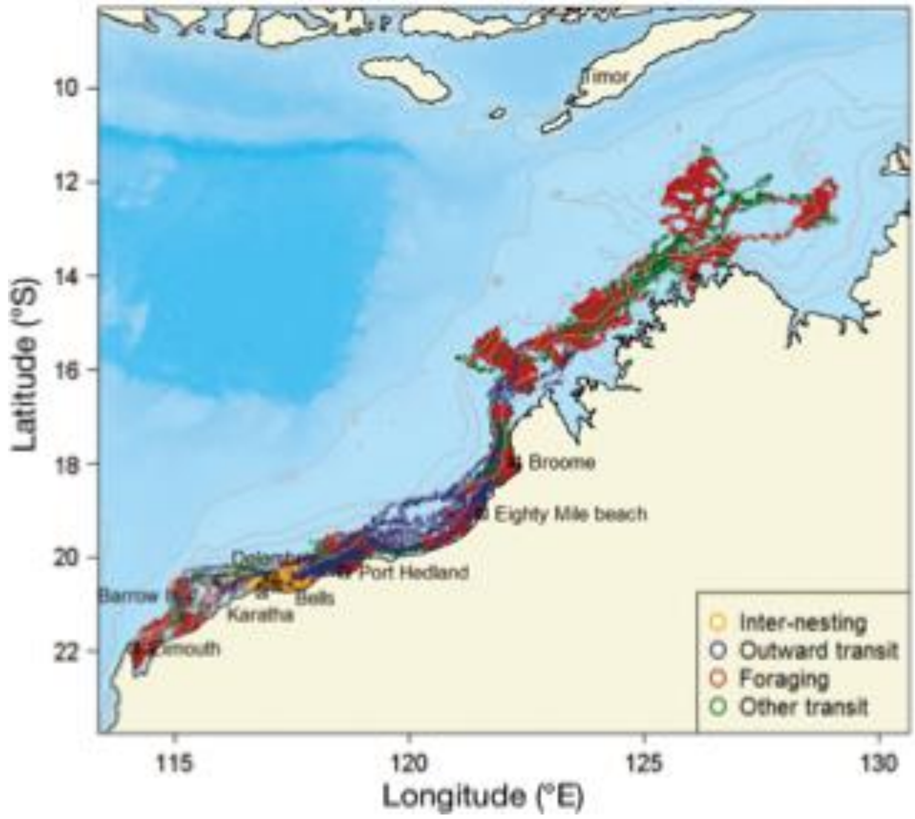
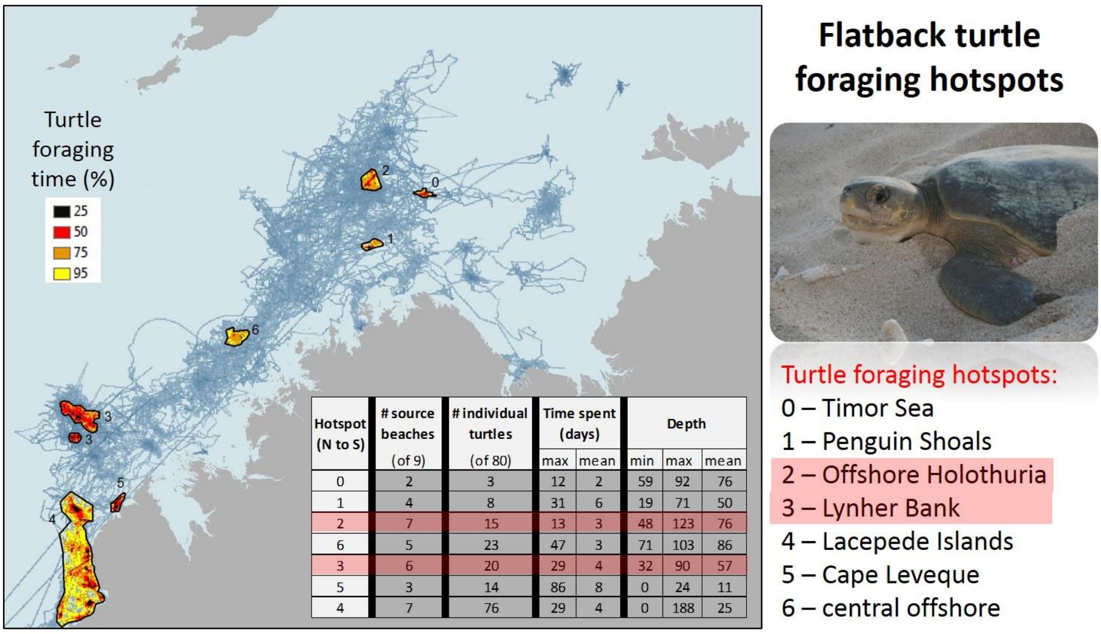


Figure 38. State-space model position estimates of flatback sea turtles from Western Australia. Tracks are coloured by behavioural mode: yellow: inter-nesting; blue: outward transit; red: foraging; green: other transit. Image source: Thumbs et al. 2018

A comprehensive analysis of flatback movements from Western Australia was compiled by Poutinen & Thums (2016), that identified seven key foraging areas for flatbacks in the Timor Sea, five onshore and two offshore (Figure 36). The study reported that turtles spent the most time in the inshore hotspot #5 (Cape Leveque; Figure 36), while the most individual turtles were recorded by far in inshore hotspot #4 (Lacepede Islands; Figure 36). While these two sites are outside of the ATS region, four other hotspot areas are within the Timor Sea and a large proportion of movements were recorded in the Timor Sea also (grey lines, Figure 36).



Hotspot (N to S)	# source beaches (of 9)	# individual turtles (of 80)	Time spent (days)		Depth		
			max	mean	min	max	mean
0	2	3	12	2	59	92	76
1	4	8	31	6	19	71	50
2	7	15	13	3	48	123	76
6	5	23	47	3	71	103	86
3	6	20	29	4	32	90	57
5	3	14	86	8	0	24	11
4	7	76	29	4	0	188	25

Figure 39. Post-nesting dispersal of flatback turtles showing the seven foraging area hotspots across NW Australia shown at a 2 km pixel scale. Image source: Poutinen & Thums 2016

Flatback turtles from the large rookery at Cape Domett also disperse widely across both the Arafura and Timor seas (Figure 37; Whiting et al. unpublished data). However, they appear to remain in shallow (<100m) waters on the Australian continental shelf (see also Figure 21).

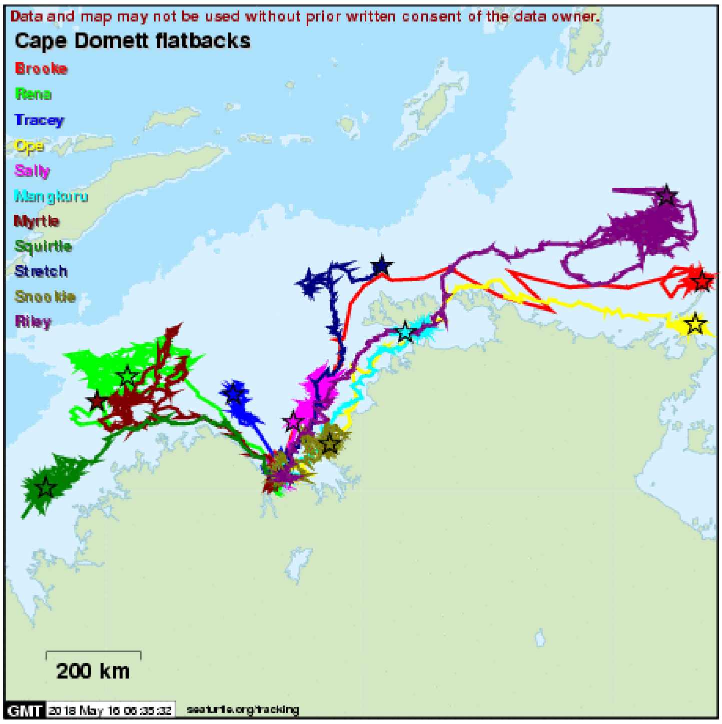


Figure 40. Dispersal of post-nesting flatback turtles from Cape Domett, NT: S. Whiting et al., unpublished

10.2 GENETIC STRUCTURE

The flatback turtle only breeds in Australia but has migrations that can include international waters. The most comprehensive assessment of genetic structure of flatback turtles in Australia is presented by FitzSimmons et al. (2020). One predominant haplotype was found across all rookeries, but other haplotype groups were regionally specific, across 17 main rookeries (Figure 10-6; FitzSimmons et al. 2020). This study led to the identification of seven genetic stocks, with geographic boundaries of rookeries used by genetic stocks varying from 160km to 1,300km (Figure 37). Genetic divergence was consistently higher between the eastern Queensland rookeries and all other rookeries, highlighting the genetic distinction of the flatback turtles in the east from other flatbacks across the north and west of Australia.

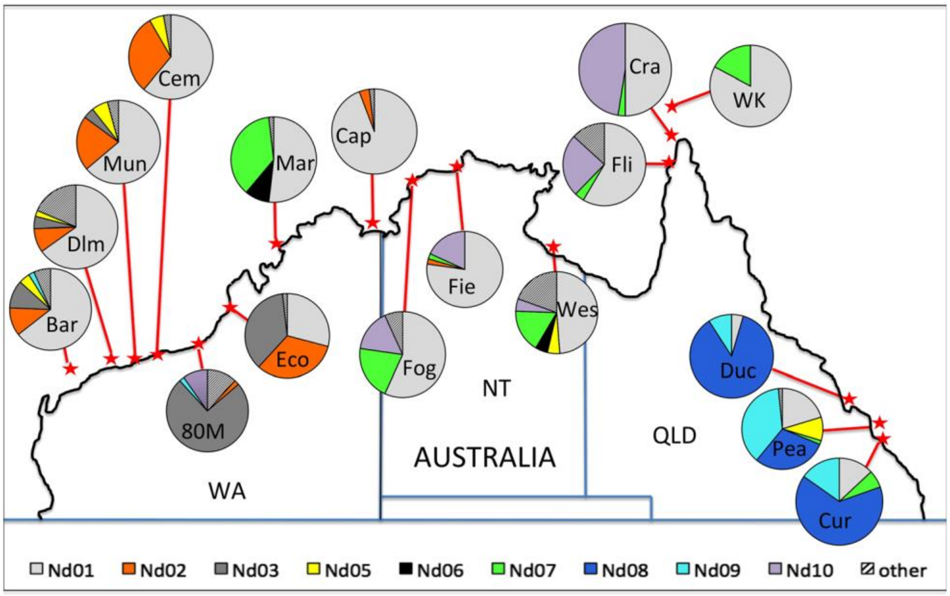


Figure 41. Distribution of the nine most common mitochondrial DNA haplotypes, and combined 'other' category, sampled from 17 flatback turtle (Natator depressus) rookeries. Image source: FitzSimmons et al. 2020

FitzSimmons et al. (2020) noted that discontinuities in haplotype frequencies among rookeries may reflect historical patterns of low-frequency colonization events by small numbers of turtles, followed by strong rookery fidelity of those turtles, and later fidelity of their offspring to natal regions for breeding. If so, observed patterns suggest that colonisation events do not necessarily involve turtles from nearby rookeries, as seen in the discontinuous distribution of some flatback haplotypes.

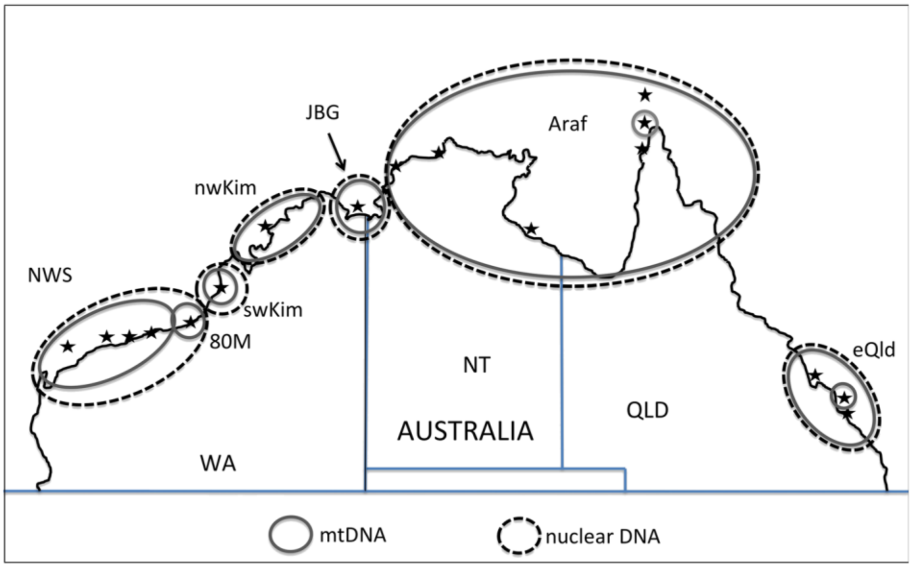


Figure 42. Designated flatback turtle (Natator depressus) genetic stocks based on the analyses of 17 rookeries across their range. Image source: FitzSimmons et al. 2020

10.3 POPULATION TRENDS

Long-term trends are available for only a handful of sites in Australia. Groom et al. (2017) conducted a long- term capture-mark-recapture program on nesting flatback turtles on Field Island in Kakadu National Park, a World Heritage Area that is jointly managed by Aboriginal landowners and the Australian Government, from 2002 to 2013, and determined there was a non-significant trend over 12 years of monitoring (Figure 40). This is mirrored by long-term data for Bare Sand Island from 1996 to 2020 (Figure 41, Guinea 2020).

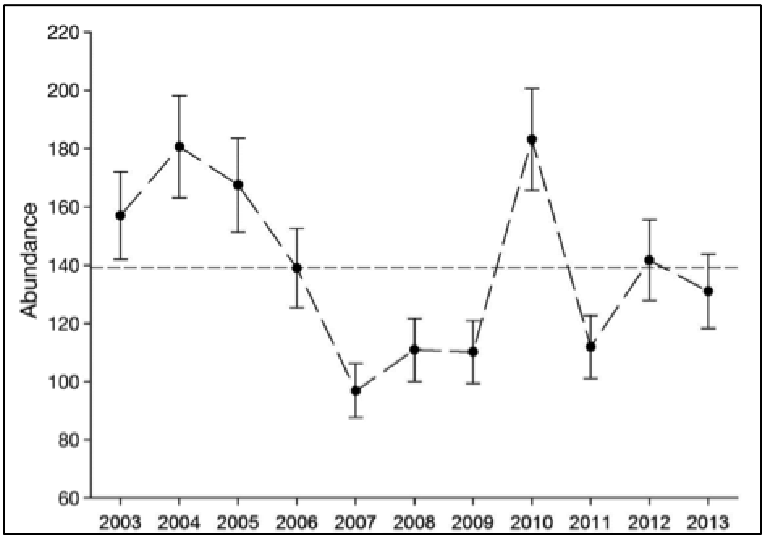


Figure 43. Nesting abundance of flatback turtles (Natator depressus) at Kakadu National Park, NT. Image source: Groom et al. 2017

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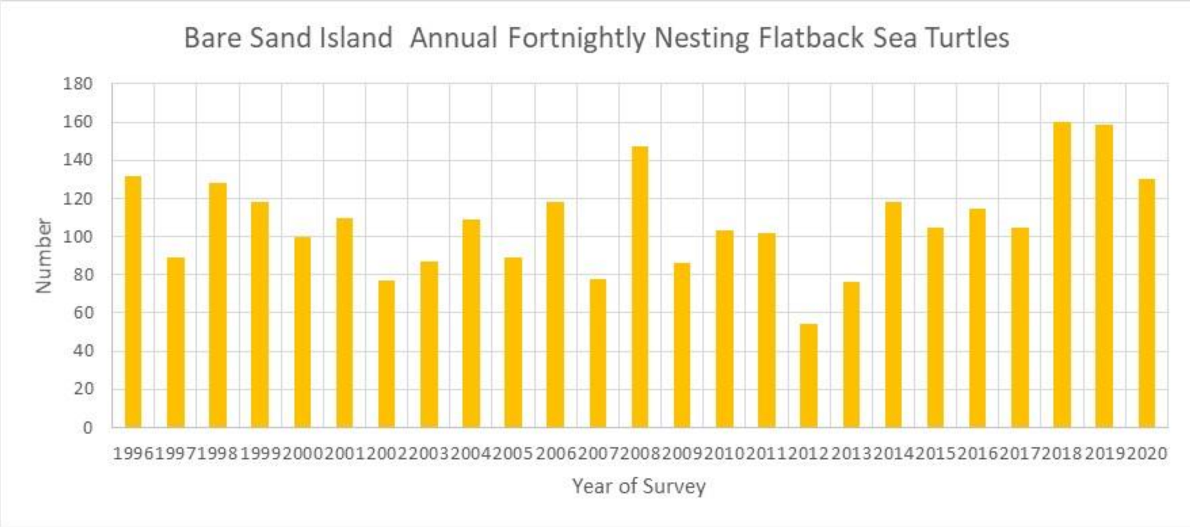

Figure 44. Nesting abundance of flatback turtles (Natator depressus) at Bare Sand Island, NT. Image source: M Guinea, 2020

CHAPTER 11. THREATS

Given the lack of a complete understanding of the magnitude of impacts on sea turtle populations, it is not possible to accurately identify the highest and lowest priority threats. For instance, while climate change may impact sea turtle populations, it is currently unknown to what extent this occurs. While fishery bycatch in Australia may be well managed, this is less so outside of Australian waters, and it is likely (as shown below) that thousands of sea turtles are lost to fisheries each year. Given these uncertainties, the threats listed below are not presented in any order of priority but are believed to be far higher priority than some other threats such as vessel strikes, oil pollution, and coastal development, which are not discussed herein.

11.1 BYCATCH IN FISHERIES

Fisheries in the ATS region include those managed by Australia, Timor-Leste, Indonesia and PNG as well as foreign vessels that may be operating under the flags of these and other countries (Wagey et al. 2009, Williams 2007). Indonesia is the highest contributor to the fisheries sector with ~250,000 fishers, followed by Timor-Leste with ~5,000 fishers and Australia with ~650 fishers (ATSEA 2011). Key fisheries include the Arafura Sea shrimp trawl fishery in Indonesia (the Arafura Sea shelf area between West Papua and Australia is shallow and hosts trawling for penaeid shrimps), deep water large-scale purse seines and artisanal pole-and-line, trolling gear and mini-seines that catch small pelagic fishes, tuna and skipjack, often using FADS (fishing aggregating devices). In Australia the Northern Prawn Trawl Fishery in Australia, the Torres Strait Prawn Fishery (TSPF), the Kimberley Prawn Fishery (KCPF) and the Northern Territory pelagic gillnet fishery are implicated in bycatch of ATS region turtles, albeit at low levels given the use of Turtle Excluder Devices.

In Australia significant steps have been taken to reduce fishery-turtle interactions. The introduction of turtle excluder devices (TEDs) in trawl fisheries has reduced turtle mortalities when used correctly, with fewer captures since 2001, and with the majority being released alive (Brewer et al. 2006). For example, in 1999, 780 turtles were caught and released by the Northern Prawn Fishery, with 96 turtle deaths. In 2006, following the introduction of turtle excluder devices, 31 sea turtles were caught and all were released alive (DEWA 2008). In addition, the use of de-hookers and line cutters in long-line fisheries has also improved marine turtle survival as they facilitate the live release of turtles captured on gear (Patterson et al. 2015).

However, there are still some areas where fishery-turtle interactions are of concern, as fisheries continue to interact with turtles (Figure 42). One area is the Gulf of Carpentaria and the Northern Territory near Darwin and throughout eastern Arnhem Land (Figure 43), where the highest rates of turtle/fishery interactions have been reported (Riskas et al. 2016). There is concern that the olive ridley turtle, which has seen large population reductions in western Cape York, may comprise a large portion of these bycaught turtles (Jensen et al. 2013). Riskas et al. (2016) also noted that while the bycatch near Darwin could be attributed to the Northern Territory pelagic gillnet fishery, the reports of turtle bycatch in the Gulf came almost exclusively from the Northern Prawn Trawl Fishery. They also noted that olive ridley turtles were reported in the Northern Prawn Trawl Fishery at an average rate of eight turtles per year. They suggested that these annual bycatch rates could place proportionally higher pressure on Australian olive ridleys, which are also threatened by egg depredation and mortality in ghost nets (Jensen et al. 2013, Limpus 2007e, Wilcox et al. 2013). Given the existence of several different fisheries, with different reporting avenues and log-book record programmes, Riskas et al. (2015) indicated that the cumulative impact of all fisheries on any given stock remained unquantified.

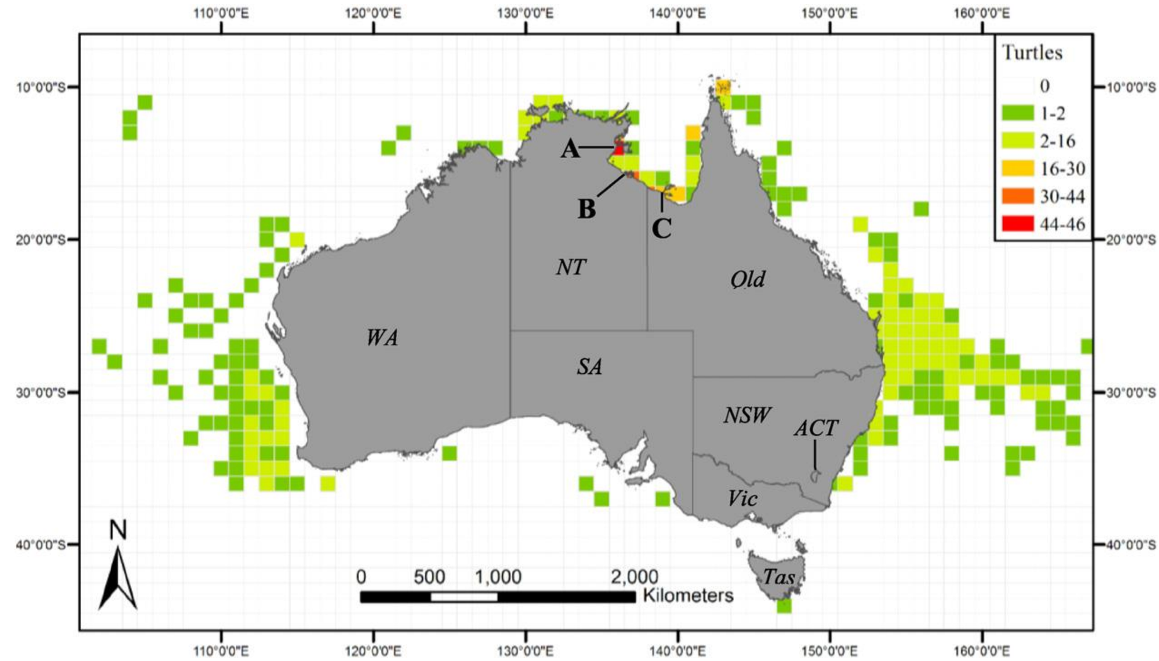

Figure 45. Spatial distribution of cumulative turtle interactions with Commonwealth-managed fisheries, 2000–2013. A: Groote Eyland; B: Sir Edward Pellew Islands; C: Wellesley Islands. Image source: Riskas et al. 2016

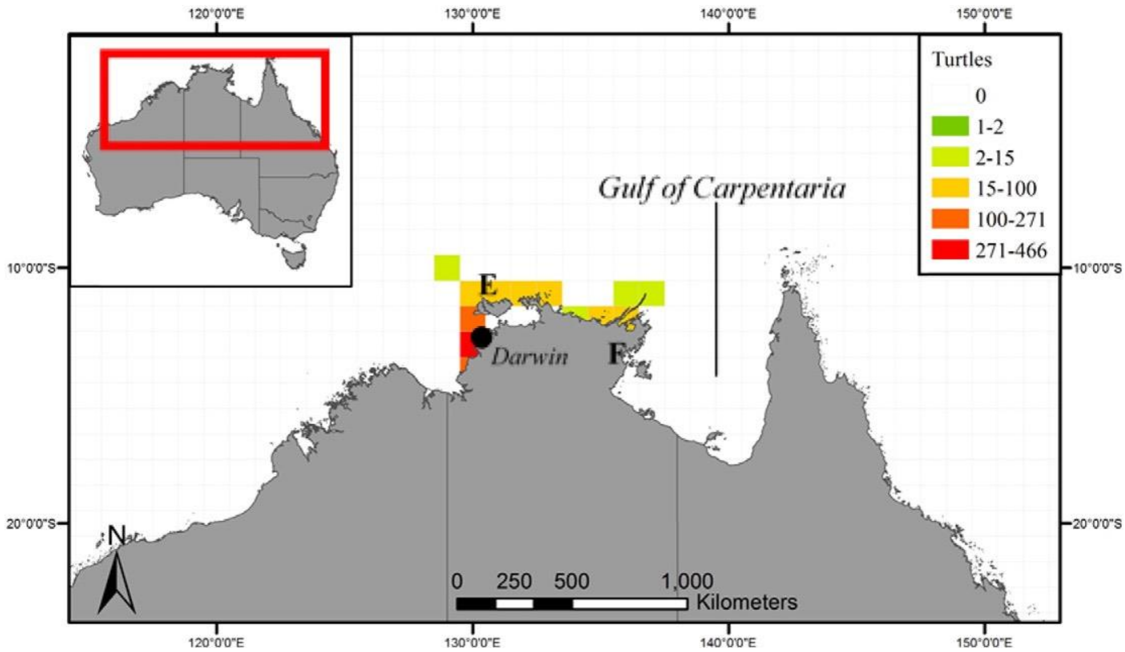

Figure 46. Spatial distribution patterns of cumulative turtle interactions with Northern Territory-managed fisheries, 2000–2013. E: Tiwi Islands; F: East Arnhem Land. Image source: Riskas et al. 2016

Turtles are also bycaught in Australian gillnet fisheries. As an example, 24 flatback turtles were estimated to have drowned in a 2,000m bottom set monofilament net shark net over a two-week period, approximately 4km off-shore in Fogg Bay, Northern Territory in 1991 (Guinea & Chatto, 1992). Similarly, an onboard-observer on a Taiwanese gill net boat off the Arnhem Land coast recorded seven flatbacks out of 16 turtles captured over approximately a four-month period, with 81 sets of a 10.5km monofilament net in 1985-86. (Limpus 2007f).

Immature flatbacks are also regularly captured in gill nets set along the coast of the south-eastern Gulf of Carpentaria and some of these turtles are drowned (unpublished data, EPA Queensland Turtle Conservation Project). The annual kill of turtles in the inshore gill net fisheries of the Gulf of Carpentaria and Arnhem Land remains unquantified.

In Indonesia there is less detailed information on bycatch rates, particularly in the ATS region. One account of the shrimp fishery in the Arafura Sea in the 1990s indicated that vessels did interact with turtles, and often did not use the mandated Turtle Excluder Devices. The Directorate General of Fisheries reported an interaction rate of 7 turtle in 450 hauls for one individual vessel in the 1990s. At slightly under 1,000,000 hauls per year across the entire fishery, this would equate to a bycatch of ~15,500 turtles per year in the 1990s. In 2004 there were 338 vessels operating in this fishery (Purbayanto et al. 2004), which constitutes a 20% to 25% decrease in the number of vessels in the late 1990s. It is possible then, that the bycatch of turtles has similarly decreased by this proportion, to ~11,500 to ~12,500 turtles per year. Onboard observations carried out by WWF in 2005 and 2006 in the Arafura Sea, Digul, Kalmana, and Timika fishing areas and reported 133 turtles in only 12 observed vessels in 2005, and in four vessels observed in 2006 an additional 26 turtles were recorded in just four months (DBC 2014). Interview data from 157 fishermen indicated that an average of one sea turtle was caught per individual vessel / trip. These estimates are comparable to the total bycatch estimates presented above and suggest losses of tens of thousands of turtles per year in the Arafura shrimp trawl fishery.

Indonesia's assessment of threatened species for the Coral Triangle Initiative (DMCB 2018) indicated that bycatch in longlines involved primarily olive ridley turtles (78.1% or 490 turtles) followed by green turtles (7.8% or 49 turtles). All other species were also hooked: hawksbills and loggerheads (5.3% each equivalent to 33 turtles), leatherbacks (1.9% or 12 turtles) and flatbacks (1.6%; 10 turtles). The bulk of the longline interactions occurred north of West Papua, north of Sulawesi and southwest of Java. However, while these interactions occurred outside of the ATS region it is likely that turtles from the ATS region are implicated in the catches given their migratory nature. Most turtles implicated in this fishery were reportedly juveniles (DMCB 2018).

WWF-Indonesia and the Directorate of Conservation and Marine Biodiversity (Ministry of Marine Affairs & Fisheries) indicated in 2015 that the tuna longline industry was unlikely to impact turtles in the ATS region given most interactions occurred much further west in the Indian Ocean (DKKLH 2015). Mustika et al. (2014) reported no instances of turtle bycatch in either coastal gillnets, long lines, and purse seines in Paloh and Adonara in 2013, and it is likely that the dispersed nature of turtles accounts in part for these findings. Purse seiners in Java indicated bycatch rates of at least one turtle per trip, especially where the fishing area was near a turtle nesting beach (DBC 2014).

Further information on origins of the bycatch, such as through genetic sampling, would be useful to clarify which turtle populations / stocks are implicated in the industrial fisheries bycatch, much as was done by Jensen et al. (2013) for turtles caught in ghost nets in northern Australia.

DBMC (2018) also indicate that substantial bycatch occurs in small-scale fisheries: In one WWF study in Sulawesi an estimated 20 to 30 turtles were caught per vessel per year. Given the vast numbers of boats operating in the ATS region it is likely that impacts on sea turtles are substantial – even alarming, should these interaction rates be similar to those in Sulawesi. Findings at other locations mirrored these high catch rates (Table 11-1). Impacts on turtle species from small-scale coastal fisheries differ from longlines, which operate in deep waters. In Kalimantan hawksbill turtles were the most common at ~42% followed by greens (~30%). Loggerheads, flatbacks and leatherbacks comprised <7% of all bycatch. Gill nets accounted for the vast proportion of bycatch in small-scale fisheries in Indonesia (Table 2).

Table 1. Estimates of turtle bycatch at a selection of locations in Indonesia. Image source: DCMB 2018

Location	Fishing Gear	Average turtle bycatch/year
Lunyuk (Sumbawa)	Gillnet	105
	Drifting Longline	1.67
	Bottom longline	-
Labuhan lalar (West Sumbawa)	Drifting Gillnet	49.5
Pototano (West Sumbawa)	Bottom longline	75
Hu'u (Dompu)	Fixed Gill Net	33.6
Rompo (bima)	Drifting Gill Net	180
Merpak(Central Lombok)	Drifting Gill Net	81.11
	Fixed Gill Net	539.84
	Bottom longline	118.33
	Drifting Longline	7.50

Table 2. Proportion of turtle bycatch by fishing gears in Indonesia. Image source: DCMB 2018

Fishing Gear	Green Turtle (%)	Hawksbill Turtle (%)	Olive Ridley Turtle (%)
Drifting Gill Net	51.11	38.00	100.00
Fixed Gill Net	28.89	36.00	-
Circle Gill Net	2.22	-	-
Purse Seine	6.67	-	-
Drifting Longline	4.44	16.00	-
Bottom Longline	6.67	10.00	-

It is likely that small-scale fisheries are a major source of bycatch, particularly those using gillnets. Additionally, illegal fishing in the ATS region is likely to be substantial, and bycatch from illegal and unregulated fisheries is likely to be higher than in regulated fisheries. Between 2000 and 2007 there was a two-fold increase in non-motorised vessels, and a five-fold increase in the number of motorised vessels, particularly in vessels less than 5GT in the ATS region (Edyvane & Penny 2017). The major increase in fishing activity in the Indonesian EEZ corresponded to a 3-fold increase in foreign fishing vessels (legal, illegal) sightings in northern Australian waters. Within the Australian EEZ, sightings of illegal foreign fishing vessels peaked and reached a maximum in 2005 (6,956 vessels). Numbers then sharply reduced (>80%) following major border control, surveillance and security operations in the northern Australia in 2005–2006. However, post-2007, illegal foreign fishing vessel sightings inside the Australian EEZ increased again (Edyvane & Penny 2017).

However, there is little in the way of current published statistics that might inform on the magnitude of turtle bycatch in Indonesian fisheries. This is even less so in Timor-Leste, and this information gap warrants further attention.

11.2 GHOST NETS

Ghost fishing is defined as the ability of fishing gear to continue to fish after all control of that gear is lost. This definition however, does not give specifics on how to identify mortality rates associated with ghost fishing. Ghost nets are of concern in the Arafura and Timor Seas given the high number of turtles entrained in these nets annually. Materials are transported into the gulf by southeast trade winds. These winds become northwesterly during the monsoon season (Wilcox et al. 2013). After this, a clockwise gyre current centred northwest of Groote Eylandt, exacerbates the problem of ghost nets in the region as it can prohibit ghost nets from escaping the region (Gunn et al. 2010). Thus, derelict nets in the Gulf become locked into an extended period of 'ghost fishing' until they are washed ashore (White 2003). Since the early 2000s, Australia's sparsely populated, remote northern shores have reported very high levels of foreign, fishing-related marine debris (Edyvane & Penny 2017). Northern Australia has some of the highest densities of ghost nets in the world, with up to three tons washing ashore per km of shoreline annually (Wilcox et al. 2015). The estimated total number of turtles caught from 2005 to 2012 by the ~9,000 ghost nets was between 4,866 and 14,600, assuming nets drifted for one year (Wilcox et al. 2015). Turtle species found in these nets included flatback (9.9%), green (13.8%), hawksbill (32.6%), loggerhead (1.1%), and olive ridley turtles (42.5%); approximately 24% of turtles were unidentified.

Nets with relatively larger mesh and smaller twine sizes (e.g., pelagic drift nets) had the highest probability of entanglement for marine turtles (Wilcox et al. 2015). During this study, net size was important, with larger nets having higher catch rates. These results point to issues with trawl and drift-net fisheries; the former due to the large number of nets and fragments found and the latter due to the very high catch rates resulting from the net design. However, other nets were also implicated: catch rates for fine-mesh gill nets could reach as high as four turtles / 100 m of net. Wilcox et al. (2015) concluded that ghost nets were an important and ongoing transboundary threat to biodiversity in the Arafura and Timor Seas.

Between 2003 and 2008, a total of 2,305 derelict fishing nets washed ashore in Northern Australia and of these, 89% were identified of foreign origin (i.e. manufacture), compared to 11% attributed to Australian fishing vessels or fisheries (Edyvane & Penny 2017). These authors concluded that industrial foreign and Indonesian-flagged fisheries - particularly, illegal, unreported and unregulated (IUU) trawling activity - and small-scale Indonesian IUU fisheries (primarily targeting shark) in the Arafura Sea were likely the major sources of these nets. The arrival and increase in derelict nets in northern Australia after 2000 coincided with sharp increases in both industrial foreign fishing (illegal and legal) and Indonesian small-scale fisheries within the Indonesian EEZ waters of the ATS region.

While the problem is one faced primarily on Australian beaches, recent genetic studies suggest that turtles entrained in these nets also originate from neighbouring countries, most likely from Indonesia, with a small number potentially also coming from Timor-Leste (Jensen et al. 2013). Solutions to the ghost net problem are complex and involve a wide range of stakeholders (Butler et al. 2013). These include net manufacturers, fishers, government regulatory agencies, local communities, conservation agencies and artists and art buyers. Some local communities along the Gulf of Carpentaria and the Northern Territory indicate the ghost net issue may have decreased slightly in recent years, but it is unlikely to go away and thus impacts to sea turtles in the ATS region warrants continued investigation.

11.3 PREDATION

There is an extensive understanding of predation in Australia, where multiple predators impact turtles and their eggs. Large crocodiles, *Crocodylus porosus*, are predators of nesting female flatback turtles and olive ridley turtles. Sutherland & Sutherland (2003) recorded a predation rate of 1.17 females/week by crocodiles during July 1997 at Crab Island. Predation of flatback clutches by feral mammals or varanid lizards did not occur at the major island rookeries such as Crab or Deliverance Islands (Limpus et al. 1989, 1993; Sutherland & Sutherland, 2003), but loss of clutches to feral pigs along the mainland coast south of the Jardine River was presumed to be ~90% (Limpus et al. 1993). Whytlaw at el. (2013) recorded an overall level of nest mortality of 40.2% with pigs being responsible for 93% of nest losses. Foxes also are predators of turtle hatchlings in Australia where the impact on overall hatchling production can be varied (King 2016). Butcher & Hattingh (2013) recorded 70% nest predation by introduced red foxes, along with additional predation by feral cats and wild dogs, and King (2016) recorded a nest predation rate of 26% by red foxes. Guiliano et al. (2015) also recorded predation by night herons (*Nycticorax caledonicus*), and reported that 100% of emerged hatchlings of 14 nests were predated by nocturnal avian predators within an opportunistic subsample of 35 nests. They point out that this was not total predation but that the issue of night heron predation required further investigation. Whiting et al. (2008) noted that feral dogs (*Canis lupus dingo*) were a predator on Cape Domett, taking at least one clutch of eggs per night. They also recorded several hundred Nankeen night herons each night but predation on hatchlings was unquantified. The study also documented large crocodiles attacking adult nesting turtles and also hatchlings (Whiting et al. 2008).

Introduced mammals are also opportunistic predators upon turtle eggs and include feral pigs (*Sus scrofa*) and foxes (*Vulpes vulpes*), and these predators have caused almost total destruction of eggs at some rookeries (e.g. areas in Western Cape York are thought to have had predation levels of ~90% over the last 30 years; Limpus 2007f). While the nesting in this region is primarily by flatback turtles, low density Olive ridley clutches are laid on the same beaches and both species are subjected to high rates of egg predation. Almost the entire Olive ridley nesting population for Queensland occurs in this area of intense egg predation (Limpus 2007e). However, recent pig removal programmes have resulted in the near-elimination of this threat at the Mapoon beaches (Limpus et al. 2017b). Rangers from Cobourg Marine Park suggest around 70-90% of nests are predated on by dogs and goannas. This site is an important site for olive ridleys and there is a Cobourg genetic stock of green turtles that could be impacted by the high predation. Surveys in the Tiwi Islands in 2005 indicated that dogs were still a primary predator of eggs (Whiting et al. 2007a). Limpus et al. (2016a) indicates that egg collection and predation by dogs and varanid lizards is a problem on Flinders, Back and Mapoon beaches, in the western Cape York peninsula, particularly following many years of pig depredation. Dogs and to a lesser extent goannas were the most significant predators of turtle eggs on Flinders Beach (Mapoon) during the 2016 and 2017 breeding seasons (Limpus et al. 2016a, 2017b). However, the 2017 turtle breeding season saw the lowest clutch loss to predators recorded in any one year since annual monitoring of Mapoon beaches began in 2004 (Limpus et al. 2017b).

On Crab Island, Rufous night herons, blacked-necked storks, beach stone curlews, silver gulls and pelicans were observed to either predate on hatchlings directly or were identified by their tracks around newly emerged clutches (Leis 2008). Similarly on Heron Island, Hopley (2008) reported that predation of the hatchlings was high, especially by Rufous herons, and that only 6.7% of hatchlings may have reached the sea. Nocturnal avian predation was also recorded on Bare Sand Island (Giuliano et al. 2015). Only silver gulls were observed to have predated hatchlings during the day. There was no evidence of predation by feral pigs, *Sus scofa*, or native varanids on the island during the study period. However, of concern, crocodiles were a major predator of hatchlings. Close to 30 crocodiles were consistently counted on each survey night in 2008 (Leis 2008). Crocodiles congregated in areas where the densest hatching occurred. Crocodiles size varied from 1m to >6m, with numerous medium to large crocodiles (>3.5m) observed. The amount of predation witnessed indicates that crocodiles are one of the major predators of hatchlings on the island (Leis 2008). Southerland & Southerland (2003) also reported crocodile predation at a minimum rate of one adult flatback per week.

Of concern to leatherbacks that migrate through the ATS region, predation of leatherback turtle eggs by pigs and feral dogs in West Papua is a grave concern, where clutch loss can reach 40% (Hitipeuw et al. 2007). Tapilatu & Tiwari (2007) found pig predation rates of 29.3% in Jamursba Medi along with a lower predation rate by dogs. However, recently improved management approaches appear to have an effect, nest predation is reducing (Lontoh pers. comm.). In PNG domestic dogs were the most common predator on eggs, and outside of protected and monitored areas nest loss could reach 100%. After the introduction of protective bamboo grids in 2006 (Pilcher 2006) the success of clutches was higher than 60%. However, this does not appear to work with the pig predation, given their size and strength.

Pig predation on nests has also been recorded in Timor-Leste (Eisemberg et al. 2014) and it is likely that feral dogs and varanid lizards are similarly a problem.

11.4 TRADITIONAL TURTLE TAKE

Sea turtles are protected by law in all four countries bordering the ATS region. However, in Australia, under Section 211 of the Native Title Act 1993, indigenous people with a native title right can legitimately take marine turtles and eggs in Australia for communal, non-commercial purposes, subject to limited exceptions. Little information is currently available on levels of Indigenous harvest of marine turtles in the Northern Territory and Queensland waters of the Gulf of Carpentaria but they are believed to be relatively low in some areas, and worryingly high in others. In the Torres Strait, a small number of nesting females and eggs used to be harvested annually from Bramble Cay, Dowar and other islands, which likely consisted of an annual nesting population of several hundred nesting females (Parmenter 1977, 1978). The current magnitude of take is unknown.

Historically, an estimated 2,410 (2,050–2,760) turtles (approximately 98% green turtles) were captured annually from the 14 inhabited islands of the Torres Strait Protected Zone, with the catch biased to females and the majority being adult and near adult turtles (Harris et al. 1992a,b). An estimated 4,000 might have been killed annually by islanders across the Queensland Torres Strait (Harris et al. 1992a,b, Limpus 2007a). While the majority of the turtles from this region originate from the nGBR breeding unit, there is known movement of turtles from the nGBR into the Gulf of Carpentaria and thus ghost nets are also likely to be of consequence to turtle stocks from outside the ATS region. Kennett et al. (1998) estimated that approximately 480 green turtles were collected annually on the northeast Arnhem coast but current levels of take are unknown. Tiwi Islanders in the Northern Territory continue to exercise their rights to customary harvest of sea turtles and anecdotal evidence suggest that green turtles are the main turtles harvested (Whiting et al. 2007a), however no estimates of annual take are available. On the Dampier peninsula of northern Western Australia, Morris & Lapwood (2001) recorded a harvest of 96 green turtles in 2002. Subsequently, Morris (pers. comm. In Limpus 2007a) suggested that the annual harvest for the Dampier Peninsula area could be about 500 green turtles annually. The total harvest in the Northern Territory is currently unknown, but is likely to be hundreds to several thousand, while Western Australia is estimated to be several thousand turtles annually (Kowarsky 1982, Henry & Lyle 2003).

In Papua New Guinea, within the north eastern area of the Torres Strait Protected Zone, there was a minimum harvest by the Kiwai people estimated at 953 to 1,363 turtles annually during 1985–1987, of which 94– 98% were green turtles (Kwan 1989, 1991). An independent study based in Tureture village during 1986 provided a larger estimate (by a factor of 2 or more) of the harvest by the Kiwai (Eley 1989). As noted above for the Queensland Torres Strait turtles, the majority of the turtles from this region originate from the nGBR breeding unit and are likely of little consequence to turtle stocks in the ATS region.

While not sanctioned at the national level, there has been a traditional take of leatherback turtles in the Kei Islands, Indonesia, for many years (Suarez & Starbird 1996, Suarez et al. 2000). Suarez & Starbird (1996) monitored the harvest between October and November 1994 and reported a catch of 23 leatherback turtles by Kei Islanders (six males and 17 females), and between October 1994 and February 1995 Suarez (2000) found 65 leatherback turtle captures (both sexes). More recently (Lawalata & Hitipeuw 2005) found that at least 29 leatherback turtles were hunted in the Kei Islands between November 2003 and October 2004 (18 females and 11 males). However, the number of turtles taken in this traditional practice has declined, and recently the number of turtles taken each year is down to only 5-10 (J. Wang, NOAA NMFS, pers. comm.). WWF-Indonesia, with the support of religious leaders, monitors the traditional harvesting and implemented an effective management strategy. In the 5 years since its implementation in 2017, the number of captures has decreased substantially (from 103 to 22 in a year, Suprapti pers. comm.).

Legal egg harvests are also significant: In Queensland, a large but unquantified annual egg harvest across the entire northern region. Much of this harvest occurs in eastern and central Torres Strait, particularly from Bramble Cay and the Murray Islands, and the small rookeries of the inner shelf of the nBGR (Limpus 2007a), but traditional egg collection occurs throughout the Northern Territory and in Western Australia. The majority of the turtles from the Torres Strait originate from the nGBR breeding unit and are likely of little consequence to turtle stocks in the ATS region. But eggs taken elsewhere directly impact populations in the ATS region. Flatback eggs have been gathered by indigenous peoples living adjacent to flatback rookeries across northern Queensland and the Northern Territory (Limpus et al. 1983, 1989, 2007f). Limpus et al. (2017b) indicated that collection of eggs on Back Beach (Mapoon was a significant issue in 2017.In the Groote Archipelago and along Arnhem Land, egg collection generally occurs wherever people can access the beach, and there are concerns in many areas about unsustainable take from hunting and collection. The size of the harvest is largely unquantified, but is of concern to many of the indigenous communities who live throughout the region. There are also many remote beaches that are inaccessible by road and are a long way by sea for community access where egg collection is not an issue. Tiwi islanders in the Northern Territory also take eggs of any species of turtle periodically (Whiting et al. 2007a). An emerging threat has been the use of 4X4 vehicles to cover large distances and collect eggs, but the extent of this practice also remains unquantified (Limpus 2007f).

There remains a need to explore the sustainability of legal turtle and egg harvests in the ATS region given that many communities target adult turtles and the overall number of turtles taken in the region remains unknown.

11.5 ILLEGAL TURTLE TAKE

The most glaring problem in assessing illegal turtle take is that it is illegal, and thus goes unreported and grossly unquantified. In Timor-Leste, illegal turtle harvesting has been reported as a major issue especially in the recently declared Nino Konis Santana National Park and Marine Park (Edyvane et al. 2009). Sealife Trust (2018) reported a brisk trade in turtle meat and ornaments made from tortoise shell in and around Dili. They reported that meat sales were common in local markets, and indicated that turtle shell parts came from Manatuto, Liquica, Same, Lospalos, Viqueque and Suai / Zumalai. The study also indicated that products were not always brought to market, but rather traded at the individual level, confounding any possibly quantification. In a personal communication to K. Edyvane in 2008, E. Vitorino reported on an extensive slaughter of turtles (most appeared to be olive ridley) on Jaco Island, where dozens of turtle carapaces and cooking / processing facilities were found in a cave. Olive ridley, hawksbill and green turtle shells were presented at homes along the road from Dili to the east of the country (Dethmers pers. comm., 2012). It is clear from these reports that illegal take of turtles across a large part of Timor-Leste is ongoing, possibly on a large scale, but currently unquantified.

In Indonesia all take of sea turtles is technically illegal, but this remains unquantified (with the exception of the traditional take in the Kei Islands and the Bali religious green turtle take). Illegal take of turtles reportedly declined on Rote Island following awareness programmes and implementation of local laws (Haning 2019). However, there are no estimates of annual take at this location. Febrianto et al. (2020) document trade of hawksbill shell in Kupang but similarly no estimates of annual take are available. Dethmers (2019) report that trade in green sea turtles in Bali continues, and historically Enu Island (Aru) has been a major source of turtles in this market. It is likely that illegal fishers operating in the Timor Sea also provide turtles for this trade. DBMC (2018) also indicate that the trade in meat, carapaces and eggs is still a major activity in traditional markets in Kei Kecil, Saumlaki, and Southeast Maluku, but no estimates of take are available. A total take across the Indonesian archipelago of 3,279 turtles per year was reported by Humber et al. (2014), although it is unclear how this figure was derived. But it is likely this is a gross underestimate, given that Dethmers (2000) reported an annual take of ~5,000 turtles in the Aru Islands alone. She estimated that, with the ongoing local exploitation pressure and turtles migrating to and from other regions, the Aru nesting population would go extinct within the next 50 years (Dethmers and Baxter 2010). Hilterman & Goverse (2005) and Nijman (2019) both document the ongoing illegal trade in turtle products in south Java, so it is evident that illegal harvest is ongoing but remains unquantified in the Indonesian ATS region. The lack of understanding of the magnitude of illegal take warrants further attention, and accurate assessment of the drivers and spatial distribution and impact level of this activity is needed.

While there was substantial harvest of green and hawksbill turtles in Australia in the past for commercial purposes, in recent years commercial harvest has not been permitted under any State or Federal legislation, and there is little documented illegal take of turtles in Australia. A few cases of illegal 'traditional' take have been recorded, although this is uncommon (Limpus, pers. comm.). As noted above, there is a traditional take of turtles in Australia, but today there is little or negligible other illegal turtle take in Australia.

11.6 EGG COLLECTION

Unquantified egg collection occurs in Indonesia and Timor-Leste. For instance, Dethmers (2010) reports egg collection in the Aru Islands, and Edyvane et al. (2009) indicate this happens in the Nino Konis Santana Marine Park in Timor-Leste, but no estimates of annual take are suggested. Sealife Trust (2018) reported the sale of turtle eggs in and around Dili and noted that the practice was common but again did not indicate how many clutches may be implicated on an annual basis. Eisemberg et al. (2014) reported egg collection west of Dili and indicated nesting in the areas was infrequent (<5 nests) but year-round. It is likely that egg collection occurs throughout the Indonesian islands to some extent, and on many - if not all - nesting beaches in Timor-Leste, and further investigation of this activity is warranted.

11.7 CLIMATE IMPACTS (STORMS, TEMPERATURE, EROSION)

Climate impacts can have multiple effects on sea turtles (e.g. Witt et al. 2010, Fuentes et al. 2013, Santandrián-Tomillo et al. 2009). Increased storm frequency can exacerbate erosion of nesting beaches. Sea level rise can lead to shallower beaches, or the loss of beaches altogether. Increased temperatures can lead to feminisation of stocks. Some studies suggest sea turtle ranges may be expanding due to climatic changes (e.g. Pike 2013), but caution is warranted in assuming this will be beneficial (e.g. through increased access to alternate habitats). As Pike (2013) points out, "some species may be able to disperse successfully to novel areas in an attempt to access critical resources eroded by climate change, which could allow persistence in changing environments"; "Other species will have difficulty shifting their ranges because of limitations imposed by dispersal behaviours (which could limit movements, and thus constrain the exploration and colonization of novel areas), life history (e.g., repeated use of fixed resources through time), or because the novel habitat does not contain sufficient resources necessary for survival or reproduction". In the case of sea turtles, it is likely that they have adapted evolutionarily to shifting habitats, but it is unknown if the current rate of change is one sea turtles can adapt to (e.g. Pilcher et al. 2015).

Extreme weather patterns might also profoundly impact sea turtles during El Nińo Southern Oscillation (ENSO) events. Recent investigations indicated that reproductive success declined in leatherback sea turtles, and suggested these events could become more frequent in the future (Santandrián-Tomillo et al. 2015). Contrastingly, storm frequency along the Australian coast was projected to decrease (Fuentes & Abbs 2010) adding resilience to turtle rookeries, and this suggests that impacts of storms will be localised and varied. Some places may experience violent storms and survive, while others may be exposed to less harmful storms but be lost to turtles. Erosion from major storm events is a concern, and Hitipeuw et al. (2007) describe conditions through which up to 45% of leatherback nests in West Papua, Indonesia, could be lost to erosion during the monsoon season. On the Tiwi Islands in Australia's Northern Territory, surveys after Cyclone Ingrid in 2005 showed that the beach was eroded substantially causing loss of nests but no large-scale change to nesting conditions for future nesters. Almost all nests laid eight weeks prior to Cyclone Ingrid were deemed to have been destroyed (Whiting et al. 2007a).

Rising sea levels is also of concern (e.g. Patino-Marquez et al 2014) as this raises the potential to significantly increase beach inundation and erosion (Pike et al. 2015). Nest site selection may also be impaired under less favourable conditions (e.g. Comer Santos et al. 2015), given turtles use a combination of cues to find nest sites, such as higher elevations and lower sand surface temperatures.

Global warming patterns may also impact sea turtles. Feminisation of stocks is of concern, and a recent study pointed to a 97% female bias in turtles from Australia's largest green turtle rookery (Jensen et al. 2018). In this study they determined that turtles originating from warmer northern Great Barrier Reef (nGBR) nesting beaches were extremely female-biased (99.1% of juvenile, 99.8% of subadult, and 86.8% of adult-sized turtles) and suggested that Australian green turtle rookeries had been producing primarily females for more than two decades and that the complete feminization of this population was possible in the near future. Sand temperature monitoring at Flinders Beach in Mapoon has shown that virtually all olive ridley and flatback offspring in 2015 were likely to be female based on the proportion of time the nests spend above pivotal temperatures (Limpus et al. 2016b). Laloë et al. (2015) also detected female biased production of Green and Hawksbill turtles and projected that this would increase with rising temperatures in the future. In the Central West Pacific, Summers et al. (2018) documented reduced hatching success and embryonic death above 34ºC in the Mariana Islands, and demonstrated that these impacts, in combination with egg poaching, could decrease nester abundance.

However, negative temperature effects may not be applicable to all species, as Howard et al. (2014) found that Flatback turtle embryos were resilient to the heat of climate change. They also recorded an unusually high pivotal sex-determining temperature in flatback turtles relative to other sea turtle populations, with an equal ratio of male and female hatchlings at 30.4°C. The authors suggested that this adaptation might allow some flatback turtle populations to continue producing large numbers of hatchlings of both sexes under the most extreme climate change scenarios. Alongside this, Stubbs et al. (2014) also found an anomalous production of male Flatback turtle hatchlings from Cape Domett (Western Australia).

At present most research on impacts of temperature have focused on nesting turtles and developing embryos given the ease of access. Chaloupka et al. (2007) demonstrated that loggerhead turtle nesting abundance in stocks from Australia and Japan decreased following warmer sea surface temperatures. They suggest the warmer waters may lead to reduced ocean productivity and that this could lead to long-term declines in loggerheads following protracted temperature increases. Rising temperatures may also impact hatchling fitness, as elevated water temperatures were found to decrease swimming performance in green turtles (Booth & Evans 2011). Little is known of impacts of temperature on other life stages, and this warrants further investigation.

Raine Island, the world's largest green turtle rookery, in the nGBR and a source of green turtles to the Gulf of Carpentaria / Arafura Sea, presents a good case study for predicted impacts of climate change: Back in 2008 increasing temperatures were projected to alter the sex ratios of turtle hatchlings and increase heat stress on turtles (Hopley 2008). This was later supported via research on sex ratios from the nGBR by Jensen et al. (2018) and Booth et al. (2020). It was predicted that sea level rise may not necessarily result in island erosion and that Raine Island may become even more unstable and respond to any changes in wind patterns. Erosion was later found to be a major problem in East Island, Hawaii, in 2018 when the entire island was lost to Hurricane Walaka. Similarly, Hopley (2008) predicted a sea level rise that would cause a rise in the water table increasing the risks of turtle nest flooding, and that sea level rise and temperature increase might change the ecology of the reef flat and delivery of sediment to the island. In the intervening years the Australian government has invested ~8 million AUD in trying to restore sand where it was lost, and to raise the sand level so that nests would not be inundated. Hopley (2008) also suggested El Niño Southern Oscillation (ENSO) events would have important influences on the breeding behaviour of turtles, and research by Santandrián-Tomillo et al. 2007 supports this prediction.

In short, climate has the potential to decrease reproductive output; to decrease nester abundance; to alter a species' distribution and nesting seasonality; to erode or cause the loss of entire nesting beaches; and to impact sex ratios of emerging turtles. On the other hand, sea turtles also possess evolutionary traits that have enabled them to adapt to these climatic changes over time: sea levels have gone up and down by more than 5m repeatedly in the last

100,000 years, and the planet has warmed and cooled repeatedly during the same period – sea turtles would surely have gone extinct had they not been able to adapt to these changes. Of concern, and worthy of recall, are two key issues: 1) turtles adapted to these changes in the absence of incremental human pressures; and 2) the rate of change today is roughly four times faster than anything experienced in the past. It is unknown what long-term impacts these two confounding factors will have on the viability and resilience of sea turtles in the ATS region.

11.8 LIGHT POLLUTION

Artificial light can be responsible for misorientation and disorientation in sea turtle hatchlings resulting in hatchlings moving away from the ocean and towards brighter light sources (Salmon et al. 1992, Witherington & Martin 1996). As hatchlings crawl to the ocean they have a primary tendency to orient away from a darker horizon (typically the darker rear beach dune silhouette, particularly when envisioned from hatchling eye height ~5-10 mm above the ground) and towards the brightest horizon, typically the ocean illuminated by the moon and/or stars. The presence of bright omnidirectional light, such as sky glow caused by anthropogenic light sources, or bright overhead moonlight coupled with low cloud cover, can disrupt hatchling sea-finding behaviour, causing disorientation (moving in random directions) and misorientation (orientation in the wrong direction), which can in turn affect hatchling survivorship. Sky glow (the incremental overhead brightness caused by urban centres and industrial facilities) has the potential to impact hatchling orientation, as do point-source lights directly visible from marine turtle nesting beaches. Point source lights typically attract hatchlings toward the brighter lights (misorientation), whereas sky glow typically causes general mass disorientation, where hatchlings roam in random patterns. Both of these effects cause hatchlings to remain on the beaches for unnaturally longer periods, increasing risks of predation and dehydration, and causing unnecessary energy expenditure. In extreme cases hatchlings may fail to reach the ocean, and even once at sea- may continue to be disoriented (Wilson et al. 2018).

Our understanding of regional impacts of anthropogenic light on sea turtles comes from only a handful of studies in Australia, and the few studies that do exist are conducted mostly as academic exercises or to detect impacts from major industries. There are no empirical studies of lighting impacts in the ATS region, but artificial light has been shown to disrupt natural night horizons in proximity to nesting beaches (Limpus & Kamrowski 2013). Lighting was found to impact flatback turtle orientation at Curtis Island, where multiple large industries are located. Hatchlings displayed disrupted sea-finding ability, with light horizons from the direction of nearby industry significantly brighter than from other directions. The sea-finding disruption observed at Curtis Island was less pronounced in the presence of moonlight (Kamrowski et al. 2014). However, Pendoley (2014) also investigated hatchling sea-finding in relation to light levels at the same location and determined that “flatback and green turtle hatchlings emerging from clutches located on the primary dune at both Curtis and Facing Islands orientated successfully toward the ocean without detectable disruption”.

This reported lack of impacts by anthropogenic lighting may be explained in part by the influence of cloud cover and lunar illumination, which have influenced hatchling orientation through history. Vandersteen et al. (2020) demonstrated that up to 80% of variation in nigh-time brightness was explained by the percentage of moon illuminated, moon altitude, and cloud cover. That is, anthropogenic lighting is not the only lighting that sea turtles are subjected to.

While individual turtles and hatchlings may be exposed to and impacted by light, at present at the population level this does not appear to be a problem. Indeed, at all major global nesting sites where lighting has been a cause of concern, populations all appear to be stable or on the rise (with the understanding that these turtle populations are also under considerable conservation and management). At the greater population level, Kamrowski et al. (2012) concluded that despite the broad geographic scale of impact, the majority of marine turtle nesting sites in Australia appeared minimally affected by light pollution exposure. However, it is worthy to note that our population level observations or today are of nesting adults and therefore the data we are considering here may actually be reflective of the hatchling light environment of 20-30 years ago.

Thums et al. (2016) investigated attraction of turtle hatchlings to stationary light sources (such as navigation beacons and jetty lights) and found that artificial lighting affected hatchling behaviour, with 88% of individual trajectories oriented towards light sources and spending, on average, 23% more time in a delineated area (19.5 ± 5min) than under ambient light conditions (15.8 ± 5 min). This study indicates that light can impact turtles even once they have entered the sea. On Heron Island turtle hatchlings were also disoriented, particularly on moonless nights, when 66.7% of tracking trials recorded hatchlings returning to shore, attracted by land-based light sources (Truscott et al. 2017).

Lighting associated with oil and gas facilities and coastal and island developments may have the potential to disturb the nesting regimes of sea turtles. On the North West Shelf in Western Australia, lighting from industrial complexes has been shown to affect flatback, green and hawksbill turtles (https://www.environment.gov.au/biodiversity/publications/national-light-pollution-guidelines-wildlife). In Western Australia, preliminary results of an investigation into the impact of flares and facility lighting suggest that impacts are determined by the phase of the moon, with disorientation greatest in the new moon nights. Another factor is the brightness and wavelength of the light sources. However, these reports should be interpreted with caution: ongoing studies at some of these locations do not find impacts from lighting at the population level – while a handful of hatchlings may be implicated in disorientation, the vast majority of hatchlings where light is managed all reach the sea, and there are examples of where light is not managed impacting significant numbers of hatchlings (Limpus 2020).

Given nesting beaches adjacent to the ATS region are predominantly located in isolated areas where lighting and flares associated with oil and gas facilities are virtually absent, impacts to turtles on land currently unlikely to be of concern (DAWE 2008). However, as demonstrated by Thums et al. (2016), offshore lighting can impact sea turtles and there is a potential impact from deep-water oil and gas exploration in the Timor Sea. However, the magnitude of this impact is hard to predict, given the Thums et al. (2016) study looked at hatchling orientation and it is unknown if hatchling sea turtles are concentrated in areas where rigs and offshore facilities are located. Further investigation into this potential impact is warranted.

CHAPTER 12. LEGAL INFRASTRUCTURE

12.1 NATIONAL LEGAL PROVISIONS

In Australia all species of sea turtles are protected via the Environment Protection and Biodiversity Conservation Act (1999) and via state/territory government legislation: The Northern Territory Parks and Wildlife Conservation Act (2014), the Queensland Nature Conservation Act (1992), and the Western Australian Wildlife Conservation Act (1950). Turtles may be legally hunted by Aboriginal and Torres Strait Islander people under section 211 of the Native Title Act 1993 for personal, domestic or non-commercial communal needs.

In Indonesia all species of sea turtles are protected via Government Regulation No. 7 (1999). In addition, there is the Bali Governor Decree No. 243 (1999) that revoked the green turtle take permit for religious festivals, and Act No. 5 /1990 concerning conservation of living resources and their ecosystems provides prohibition for and sanction of direct harvest of protected species.

In Papua New Guinea only the leatherback turtle is protected via the Fauna (Protection and Control) Act (1976).

In Timor-Leste all species of sea turtles are protected via the United Nations Transitional Administration in East Timor (UNTAET) Regulation No. 2000/19.

12.2 RELEVANT INTERNATIONAL CONVENTIONS

Australia, Indonesia, Papua New Guinea and Timor-Leste are all contracting parties to the Convention on Biological Diversity (CBD).

Australia, Indonesia, and Papua New Guinea are contracting parties to the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES).

Only Australia is a Party (since 1991) to the Convention on Migratory Species (CMS). However, Australia (2001), Indonesia (2005) and Papua New Guinea (2010) are Signatories to the CMS Memorandum of Understanding on the Conservation and Management of Marine Turtles and their Habitats of the Indian Ocean and South-East Asia (IOSEA MoU).

Australia, Indonesia, and Papua New Guinea are contracting parties to the Ramsar Convention on Wetlands of International Importance especially as Waterfowl Habitat (RAMSAR).

12.3 FISHERIES MANAGEMENT

In Australia, three key commercial fisheries that may interact with sea turtles include the Torres Strait Prawn Fishery managed under the Torres Strait Prawn Fishery Management Plan 2009, the Northern Prawn Trawl Fishery managed under the Northern Prawn Fishery Management Plan 1995 (amended 2012), and the North West Slope Trawl Fishery (NWSTF), managed under the North West Slope Trawl Fishery and Western Deepwater Trawl Fishery: statement of management arrangements (AFMA 2012). There is a turtle fishery in the Torres Strait managed under the Torres Strait Fisheries Act 1984 and Fisheries Management Notice No. 66. This is also a commercial fishery but is managed by States and Territories rather than the Commonwealth.

There are also multiple coastal fisheries using hook & line, gillnets, traps and other gears across all of the northern Australian region that can potentially interact with turtles. In Queensland these are governed under the Fisheries Act 1994, the Fisheries (General) Regulation 2019, the Fisheries (Commercial Fisheries) Regulation 2019, the Fisheries Declaration (2019) and the Fisheries (Quota) Declaration 2019. In the Northern Territory fisheries are governed under the Territory of Australia Fisheries Act 1988 and the Northern Territory of Australia Fisheries Regulations 1993. In Western Australia fisheries are governed under the Fish Resources Management Act 1994, the Pearling Act 1990, the Fisheries Adjustment Schemes Act 1987, the Fishing and Related Industries Compensation (Marine Reserves) Act 1997, and the Fishing Industry Promotion Training and Management Levy Act.

In Indonesia fisheries are managed by Fishery Management Areas, two of which are included in the Arafura and Timor Seas. Fisheries are regulated nationally via the Law 31 (2004), amended by Law 45 (2009) covering fisheries, license and vessel registration, management of IUU and destructive fishing, standardization of fish processing, tax, and conservation as well as estimation of the potency of fishery resources in the Fisheries Management Areas. There is also Law 27 (2007) as amended by Law 1 (2014) regarding management of coastal areas and small islands; Law 32 (2014) regarding maritime surveillance, management and harmonization among marine stakeholders; Law 7 (2016) regarding protection and empowerment of fishermen; Government Regulation 60 (2007) regarding conservation of fishery resources: Ministerial Decree 47 (2016) regarding total allowable catch and utilization rate of fishery resources; and Ministerial Decree 75-85 (2016) establishing fisheries management plans.

In Papua New Guinea the Fisheries Management Act (1998) and Fisheries Management Regulation (2000) regulate the set-up of the National Fisheries Authority, the supervision of pelagic fisheries, and local and species-specific fisheries management plans.

In Timor-Leste, laws and ministerial edicts governing fishery-related policy include Decree No. 5/2000 (General Regulation on Fishing); Decree-Law No 6/2004 (General basis of the legal regime for the management and regulation of fisheries and aquaculture); Ministerial Order 06/42/Gm/Ii/2005 (Sanctions for fisheries infringements); Ministerial Order 04/115/Gm/Iv/2005 (List of protected aquatic species); Ministerial Order 03/05/Gm/I/2005 (Percentages Of Bycatch); Ministerial Order 02/04/Gm/I/2005 (Main fisheries); Ministerial Order 01/03/Gm/I/2005 (Definition of fishing zones); Ministerial Order 05/116/Gm/Iv/2005 (Minimum size and weight of capture species); Ministerial Order 06/42/Gm/Ii/2005 (Fines for fishing infractions); and Decree-Law 21/2008 (Implementation of the satellite system for monitoring fishing vessels).

12.4 INDIGENOUS COMMUNITY MANAGEMENT

In Australia, under Section 211 of the Native Title Act 1993, indigenous people with a native title right can legitimately hunt marine turtles for communal, non-commercial purposes. In recent decades numerous indigenous communities across northern Australia have declared dedicated

Indigenous Protected Areas (IPAs) over their traditional land and sea Country and developed traditional land IPA management plans (sometimes known as Healthy Country Plans in the Northern Territory and Working on Country Plans in the Torres Strait). It is also worth noting that some Indigenous groups have agreed to put in place arrangements, e.g. in their IPA and Healthy Country Plans or in State/Territory/Regional plans, limiting or preventing the hunting of marine turtles or particular marine turtle species. These plans are built on customary practices and reflect the aspirations, customs, traditions and history of the traditional owners of the land. Sea turtles are sacred to all of these communities, and feature prominently in the Healthy Country Plans, where issues related to sustainability of turtle use are a key feature.

In Timor-Leste there exists a local resource management l ban called *Tara bandu*, which is a traditional community-based resource management mechanism. *Tara bandu* is a traditional Timorese custom that enforces peace and reconciliation through the power of public agreement. *Tara bandu* involves handing of culturally significant items from a wooden shaft to place a ban on certain agricultural or social activities within a certain area.

Sasi is a local traditional resource management system, used in Central Maluku and akin to Timor-Leste's *Tara bandu*. It implements spatial and temporal prohibitions on harvesting or gathering resources from the tidal zone or marine territory of a village.

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