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2021

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Nicolas J. Pilcher

TABLE OF CONTENTS

i. List of Acronyms and Abbreviations

IAC	Inter-American Convention for the Protection and Conservation of Sea Turtles
IATTC	Inter-American Tropical Tuna Commission
BCS	Baja California Sur
bp	Base Pairs
CBD	Convention on Biological Diversity
CI	Confidence Interval
CICI	Conflict Islands Conservation Initiative
CITES	Convention on International Trade in Endangered Species of Wild Fauna and Flora
CMI	Coastal, Marine and Island
CMM	Conservation and Management Measure
CMS	Convention on Migratory Species
CNMI	Commonwealth of the Northern Mariana Islands
ECTF	East Coast Trawl Fishery
ENSO	El Nińo Southern Oscillation
EP	Eastern Pacific
ETP	Eastern Tropical Pacific
FAO	Food and Agriculture Organization of the United Nations
FSM	Federated States of Micronesia
GBR	Great Barrier Reef
GEF	Global Environment Facility
GIS	Geographical Information System
HBF	Hook Between Float
IOSEA MoU	Memorandum of Understanding on the Conservation and Management of Sea turtles and their Habitats in the Indian Ocean and Southeast Asia
IUCN	International Union for the Conservation of Nature
MARPOL	International Convention for the Prevention of Pollution from Ships
mtDNA	Mitochondrial (mt) deoxyribonucleic acid (DNA)
MTSG	Marine Turtle Specialist Group
NMFS	National Marine Fisheries Service
MU	Management Unit (genetic differentiation)
nGBR	Northern Great Barrier Reef
NOAA	National Oceanic and Atmospheric Administration
NPTZ	North Pacific Transition Zone
NRC	National Research Council

NWHI	North West Hawaiian Islands
OCF	Observed Clutch Frequency
PEUMP	Pacific-European Union Marine Partnership
PNG	Papua New Guinea
QSCP	Queensland Shark Control Program
RMI	Republic of the Marshall Islands
RMU	Regional Management Unit
SD	Standard Deviation
sGBR	Southern Great Barrier Reef
SPC	Pacific Community
SPREP	Secretariat of the Pacific Regional Environment Programme
SSC	Species Survival Commission
STAMM	Sea Turtle Active Movement Model
SWOT	State of the World’s Sea Turtles
TED	Turtle Excluder Device
TSPF	Torres Straits Prawn Fishery
UNCLOS	United Nations Convention on the Law of the Sea
UNDP	United Nations Development Programme
UNEP	United Nations Environment Programme
USD	United States Dollar
USFWS	United States Fish and Wildlife Service
WCPFC	Western and Central Pacific Fisheries Commission
WCPFC-CA	Western and Central Pacific Fisheries Commission – Convention Area
WPRFMC	Western Pacific Regional Fisheries Management Council
WWF	World Wide Fund for Nature

ii. Scope of the Literature Review

This review presents a summary of the relevant literature and knowledge about sea turtle reproductive biology, movements, and connectivity, and presents these alongside relevant threats that, in combination, are pertinent to understanding the status of sea turtle populations in the Pacific Ocean. The review is not intended to be an exhaustive revision of all aspects of the biology and conservation of sea turtles – there is an extensive body of literature that has been widely published in peer-reviewed journals, internal documents, and a suite of other literature sources. Much of this is also aptly summarised in several key documents including, but not limited to:

1. The IUCN Red List Status Reports https://www.iucnredlist.org
2. The periodic status reviews prepared by the US National Oceanic and Atmospheric Administration's National Marine Fisheries Service https://www.fisheries.noaa.gov/resources/all-publications
3. The Biology of Sea Turtles (Volumes I, II and III); CRC Press http://crcpress.com

This review is a precursor to developing risk assessments for sea turtles in the Pacific Ocean. It sets out (where known and published) the relevant biological aspects of sea turtle reproductive output, survivorship, movements and genetic linkages and assesses threats to sea turtles such as by-catch, direct take and consumption, lighting, climate change and ocean health.

The review attempts to present the arguments in a manner that is easy to interpret and follow, chronologically relevant, and is non-exhaustive – the literature related to sea turtles grows voluminously each year, and while many additional publications could be presented, it is hoped that in its present form it sets the scene for the Risk Assessments to follow. Lastly, many excellent summaries have already been produced that detail reproductive biology and nesting demographics for all species of sea turtles, and these are drawn-upon heavily in this review. Readers are directed to those summaries for the original citations of some of the data presented herein. The author acknowledges the vast contributions from thousands of sea turtle biologists and researchers who contributed data to those summaries, and the wealth of information in their own original publications, and acknowledges that the ownership and intellectual property of that information resides entirely with all of the original data owners and providers.

It is hoped that this summary is a useful contribution and synthesis of all relevant information on sea turtles across the Pacific Ocean.

1.0 Introduction and Background

Sea turtles have played a significant role in the customs and traditions of Pacific island communities for thousands of years – and continue to do so to this day – featuring in many myths, legends, songs and traditions. However, sea turtles have been subjected to increasing pressure as customary practices have eroded and their popularity in commercial markets has increased, outside threats such as by-catch in commercial and artisanal fisheries has increased, and climate change threatens important nesting and feeding areas, along with sea turtle reproductive biology.

While some information exists with respect to the by-catch of sea turtles in the Pacific from industrial fisheries such as the tuna purse seines and to a lesser extent longline sectors, less is known about levels of use of sea turtles by coastal communities and impacts of small-scale fisheries across the Pacific. Similarly, little is known of the impacts of climate change on sea turtles and their important habitats across much of the Pacific, and of the status and trends of sea turtle populations at the local levels.

This literature review and risk assessment process is a part of the By-catch and Integrated Ecosystem Management (BIEM) Initiative being implemented by the Secretariat of the Pacific Regional Environment Programme (SPREP) through the Pacific-European Union Marine Partnership (PEUMP) programme. The BIEM Initiative, which is Key Result Area 5.4 of PEUMP, aims to understand turtle extinction risk and vulnerability across the Pacific. The project seeks to understand the extent and scope of harvest and trade of sea turtles across a range of Pacific island communities, building on work already undertaken recently in Papua New Guinea (PNG) and the Solomon Islands.

The first step in developing a risk assessment has been the compilation of the most recent literature and statistics on the status of sea turtles in the Pacific Ocean region, along with a clearer understanding of the risks to sea turtles and their habitats. This knowledge will be incorporated into a risk extinction model for sea turtles in the Pacific, to be used at national and regional levels to develop effective management and conservation programmes that ensure sea turtles continue to play their important ecological roles and continue to be part of local customs and traditions.

2.0 Extinction Risk Assessment

The risk assessment process will model the natural aspects of sea turtle biology and link these to existing threats such as by-catch, direct take, climate change and anthropogenic lighting. It is envisioned that the model will be used as a predictive tool to identify the most pressing threats and allow managers and policy-makers to address these as priorities.

2.1 The conservation equation

In its simplest form, the risk assessment process looks at two sides of an equation: the amount of turtles put into the system on one side, and the number of turtles taken out of the system on the other side. The number of turtles that are put in will depend on factors such as how frequently turtles lay eggs, how many eggs they lay, how many of these eggs survive, how many turtles survive in different age groups, etc. The number of turtles taken out of the system will depend on factors such as levels of traditional take and consumption, numbers of turtles caught in inshore artisanal fisheries, numbers of turtles caught in industrial fishing fleets, etc. Table 2-1 identifies the key parameters that will be addressed in the Risk Extinction model.

Table 2-1: Biological traits and threats that will be incorporated into a risk assessment model for sea turtles in the Pacific.

Biological traits	
Terrestrial	**Data sources**
Annual number of nesting female turtles	Reports, publications, data sets, models
Trend in number of nesting females	Reports, publications, data sets, models
Turtle dispersal range (tracking)	Reports, publications, models
Turtle dispersal range (genetics)	Reports, publications, models
Turtle dispersal range (isotopes, microchemistry)	Reports, publications, models
Remigration interval (distribution of number of years between effective nesting season)	Reports, publications, data sets, by proxy, models
Clutch frequency (distribution of number of clutches within a season)	Reports, publications, data sets, by proxy, models
Nesting success (ratio between clutches and tracks)	Reports, publications, data sets, by proxy, models
Hatching (probability that an egg ends incubation) and emergence success (probability that an egg produces a juvenile reaching the surface of the beach)	Reports, publications, data sets, by proxy, models
Natural sex ratio (inter-annual and intra-annual distribution of sex in embryos)	Reports, publications, data sets, by proxy, models
Natural hatchling survival probability	Publications, by proxy
Marine	**Data sources**
Natural adult survival probability	Publications, by proxy
Natural subadult survival probability	Publications, by proxy
Natural juvenile survival probability	Publications, by proxy
Threats	
Terrestrial	**Data sources**
Hatchling predation (on beaches)	Reports, publications, by proxy
Direct take of adult turtles (on beaches)	Reports, publications, current project
Direct take of eggs	Reports, publications, current project
Nesting habitat loss (complete loss of habitat due to erosion)	Reports, publications
Nesting habitat loss (sub-lethal condition for development)	Reports, publications
Nesting habitat alteration (temperature)	Reports, publications, current project, by proxy
Pollution (obstacles for adults)	Reports, publications, current project, by proxy
Pollution (obstacles for juveniles)	Reports, publications, current project, by proxy
Chemical pollution (alteration of development)	Reports, publications, current project, by proxy
Thermal pollution by objects on the beach	Reports, publications, current project, by proxy
Marine	**Data sources**
Commercial fisheries by-catch (juveniles)	Reports, publications, current project, SPC
Commercial fisheries by-catch (sub-adults)	Reports, publications, current project, SPC
Commercial fisheries by-catch (adults)	Reports, publications, current project, SPC
Artisanal fisheries by-catch (juveniles)	Reports, publications, current project
Artisanal fisheries by-catch (sub-adults)	Reports, publications, current project
Artisanal fisheries by-catch (adults)	Reports, publications, current project
Climate change (rising water temperatures)	Reports, publications, global data sets
Habitat alteration / loss (foraging grounds)	Reports, publications
Plastics / solid waste ingestion / entanglement	Reports, publications, local data

3.0 IUCN Status

Among the most recognised assessments of risk extinction are the 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 sea 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. These assessments are undertaken by members of the IUCN Species Survival Commission (SSC) Marine Turtle Specialist Group (MTSG). The assessments are then reviewed by an assessment committee, and then reviewed by the entire MTSG membership, before being submitted to the Red List for review and publication.

The 2020 IUCN Red List of Threatened Species lists the six sea turtle species found in the Pacific as follows:

- **Leatherback** (*Dermochelys coriacea*): Vulnerable (global)
  Critically endangered (West Pacific subpopulation)
  Critically endangered (East Pacific subpopulation)
- **Hawksbill** (*Eretmochelys imbricata*): Critically endangered (global)
- **Loggerhead** *Caretta caretta*): Vulnerable (global)
- **Green** (*Chelonia mydas*): Endangered (global)
  Least Concern (North Central Pacific subpopulation)
- **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)

4.0 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 organise sea 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 risk extinction at the RMU level. The leatherback and green turtle subpopulation assessments listed in Section 3.0 are examples of more recent assessments conducted using the RMU framework. The current recognised RMUs of sea turtles in the Pacific are as follows:

Green: Eastern Pacific, North Central Pacific, Northwest Pacific, South Central Pacific, West Pacific-East Indian Ocean
Hawksbill: North Central Pacific, South Central Pacific, West Pacific, West Central Pacific, East Pacific, Southwest Pacific
Loggerhead: North Pacific, South Pacific
Leatherback: East Pacific, West Pacific
Flatback: Southwest Pacific
Olive Ridley: West Pacific, East Pacific, East Pacific (arribadas)

5.0 Regional Conservation and Management Programmes

Several regional agreements and action plans address sea turtle conservation in the Pacific Ocean region, which are relevant to long-term management of these species and their habitats.

5.1 SPREP Regional Sea turtle Action Plan 2013–2017

The Marine Species Programme of the Secretariat of the Pacific Regional Environment Programme (SPREP) outlines a regional strategy for the cooperative conservation and management of dugongs, sea turtles, whales and dolphins. A new strategy has been developed and will come into effect in 2021. The strategy, which will be implemented during 2021–2025, will enable Pacific islanders to take a primary role in managing sea turtles, and meets the aspirations of Pacific island peoples and protects their natural and cultural heritage.

5.2 IAC

The Inter-American Convention for the Protection and Conservation of Sea Turtles (IAC) seeks to promote protection, conservation and recovery of sea turtle populations and habitats on which they depend, based on best available scientific evidence, taking into account environmental, socioeconomic and cultural characteristics of the Parties. The IAC came into force in 2001 and currently has 13 Parties. The IAC is the only binding treaty that focuses exclusively on sea turtles. The IAC prohibits intentional capture, retention or killing of, and domestic trade in, sea turtles, their eggs, parts or products. To date, the IAC parties have adopted one resolution on fisheries, which calls on Parties to incorporate the FAO Guidelines to Reduce Sea Turtle Mortality in Fishing Operations into their fisheries management programmes.

5.3 WCPFC

The Western and Central Pacific Fisheries Commission (WCPFC) seeks to ensure long-term conservation and sustainable use of highly migratory fish stocks in the western and central Pacific Ocean. The WCPFC requires shallow-set longline fisheries in the EEZs and on the high seas to use large circle hooks, whole finfish bait, or other mitigation measures proven to reduce interaction with or increase survivorship of sea turtles. These Commissions also require their members and cooperating non-members to safely handle and release sea turtles they encounter and provide sea turtle data to their respective secretariats.

5.4 IATTC

The Inter-American Tropical Tuna Commission (IATTC) seeks to maintain the populations of yellowfin and skipjack tuna and of other kinds of fish taken by tuna fishing vessels in the eastern Pacific Ocean. The IATTC parties have adopted several resolutions to address sea turtles. IATTC requires vessels fishing for tuna and tuna-like species operating on the high seas to take steps to reduce the frequency and severity of fishing gear interacting with sea turtles in accordance with the FAO guidelines.

5.5 WPRFMC

The Western Pacific Regional Fishery Management Council is one of eight regional fishery management councils established by the US Congress in 1976. Under the Magnuson-Stevens Fishery Conservation and Management Act, it has authority over fisheries seaward of state/territorial waters of Hawaii and the US Pacific islands. Management decisions are based on science and informed by traditional knowledge and practices of the local users for the benefit of the island communities and the nation. The WPRFMC addresses sea turtle conservation via conservation measures that include rules protecting sea turtles, gear restrictions and a cap on the number of sea turtles with which the fishery may interact. The fishery currently operates under an annual cap of 17 loggerhead and 16 leatherback turtle interactions and has 100% observer coverage. An interaction occurs whenever a sea turtle becomes hooked or entangled in longline gear, as recorded by the NOAA (National Oceanic and Atmospheric Administration) National Marine Fisheries Service (NMFS) observer. For many years the WPRFMC invested significant amounts of funding to support recovery of sea turtle populations in Japan, Papua New Guinea, Indonesia, Mexico and various other locations in support of Pacific sea turtle conservation.

5.6 NOAA's Threated Species Regional Programme

NOAA Fisheries Pacific islands region has a sea turtle recovery programme that supports sea turtle conservation and research projects across a large range of sites in the Pacific. The programme focuses particularly on sites in American Samoa, Guam, Hawaii, the Northern Mariana Islands and other US Pacific islands but also supports projects in other countries that may impact sea turtles that use US waters or nesting beaches. The programme addresses protected species and includes issues such as sea turtle injury and mortality as a result of commercial and non-commercial fishing, coastal development, military operations and other ocean/beach usage. The programme also works with partners and stakeholders to create effective messages and outreach materials and volunteer programmes.

6.0 International Conventions

There are five key international conventions that address sea turtles and their nesting, migratory and feeding habitats. These are the Convention on Biological Diversity (CBD) 1992; the Bonn or CMS Convention (Convention on the Conservation of Migratory Species of Wild Animals) 1979; the World Heritage Convention (Convention Concerning the Protection of the World Cultural and Natural Heritage) 1972; CITES (Convention on International Trade in Endangered Species of Wild Fauna and Flora) 1973; and the Ramsar Convention (Convention on Wetlands of International Importance) 1971. A list of countries that have signed, ratified or acceded to these top international conventions is presented in Table 6-1.

Table 6-1: Pacific Ocean signatories to the five key international biodiversity conventions (*except Tokelau).

	CBD	CMS	UNCLOS	CITES	Ramsar
Australia	✓	✓	✓	✓	✓
Chile	✓	✓	✓	✓	✓
Colombia	✓			✓	✓
Cook Islands	✓	✓	✓	✓	
Costa Rica	✓	✓	✓	✓	✓
Ecuador	✓	✓	✓	✓	✓
El Salvador	✓			✓	✓
Fiji	✓	✓	✓	✓	✓
France	✓	✓	✓	✓	✓
Guatemala	✓		✓	✓	✓
Japan	✓		✓	✓	✓
Kiribati	✓		✓		✓
Mexico	✓		✓	✓	✓
Micronesia	✓		✓		
Marshall Islands	✓		✓		✓
Nauru	✓		✓		
New Zealand	✓*	✓	✓	✓	✓
Niue	✓		✓	✓	
Nicaragua	✓		✓	✓	✓
Palau	✓	✓	✓	✓	✓
Panama	✓	✓	✓	✓	✓
Papua New Guinea	✓		✓	✓	✓
Peru	✓	✓		✓	✓
Philippines	✓	✓	✓	✓	✓
Samoa	✓	✓	✓	✓	✓
Solomon Islands	✓		✓	✓	
Tonga	✓		✓	✓	
Tuvalu	✓		✓		
United States of America				✓	✓
Vanuatu	✓		✓	✓	

The species covered by the CITES convention are listed in three Appendices, according to the degree of protection they need. Appendix I lists species that are the most endangered among CITES-listed animals. They are threatened with extinction and CITES prohibits international trade in specimens of these species except when the purpose of the import is not commercial, for instance for scientific research. All sea turtle species are listed on CITES Appendix I.

The text of the Convention on Migratory Species (CMS) defines the basic obligations of the Contracting Parties towards species listed on Appendix I and Appendix II. The leatherback, hawksbill, loggerhead, green and olive ridley turtles are all listed on Appendix I. The flatback is listed on Appendix II.

7.0 Biology and Ecology - Positive side of the Risk Assessment

7.1 Number of Nesting Turtles – Green Turtles

Maison et al. (2010) provide a very thorough review of green turtle nesting and numbers in the central and western extent of the Pacific region (Figure 7-1). Additional and updated information is provided by Seminoff et al. (2015) in a five year review of the status of green turtles, and also by Pilcher et al. (2011) for the north central Pacific RMU, and by Seminoff and Glass (2000) for the east Pacific RMU. In addition, the State of the World's Sea Turtles (SWOT) Report IV provides a summary map of global nesting sites for green turtles with estimates of abundance (Figure 7-2). Care should be taken when interpreting the SWOT graphic as this relies only on voluntarily contributed information and therefore is not exhaustive in its content.

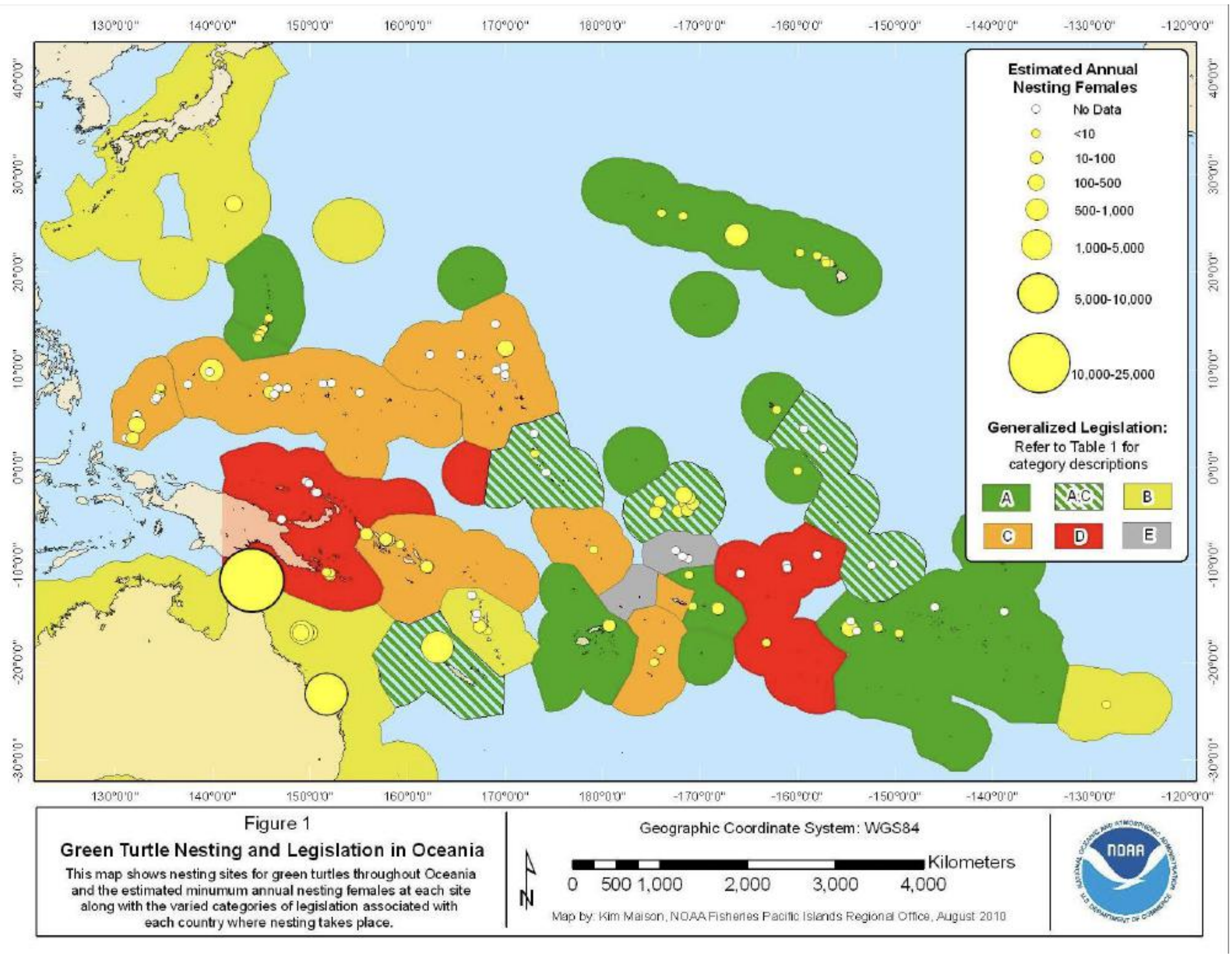


Figure 7-1. Green turtle nesting sites in the west and central Pacific. Source: Maison et al. 2010.

Figure 7-2. Worldwide green turtle nesting sites 2011. Source: SWOT Report Vol. VI.

Because the type of information available for nesting green turtle abundance in each country varies, estimates of annual nesting females may be binned into discreet categories, following a standardised approach adopted by Seminoff et al. (2015): 1 to 10, 11-50, 51-100, 101-500, 501-1000, 1001-5000, 5001-10000, and >10,000. Two additional bins were created for arribada olive ridley turtles: >100,000 and >500,000, given the large numbers present in distinct nesting events. It is envisioned that these bins would also assist in developing the model for overall nesting numbers and trends in the Pacific Ocean. A summary of annual nester abundances and the categorical bins for each is presented in Annex A.

Green turtles and hawksbill turtles are the most common species and most widely distributed across the Pacific, and these are presented first and second, followed by (decreasing abundance) loggerheads, leatherbacks, flatbacks and olive ridleys. Few sites in the Pacific have long-term data trends, but where these exist they are presented as examples that might typify trends for the varying RMUs. Graphics are also used where appropriate to clarify information summarised in the text.

7.1.1 Green turtle – North Central Pacific RMU

Hawaii: Extensive studies have been conducted on the green turtles nesting in Hawaii. Over 90% of these turtles nest on a few islets in the French Frigate Shoals and up to 2018 the trend in nesters was generally increasing (see Figure 7-3). In 2018 one of the main islands where green turtles nested was lost to hurricane Walaka and no systematic surveys have been conducted since. It is unknown what the future trajectory of this species will be in the Northern Hawaiian Islands. Seminoff et al. (2015) suggest the total nester abundance was ~ 3,800 females, while annual nester abundance was estimated to be ~450 (Balazs and Chaloupka 2004).

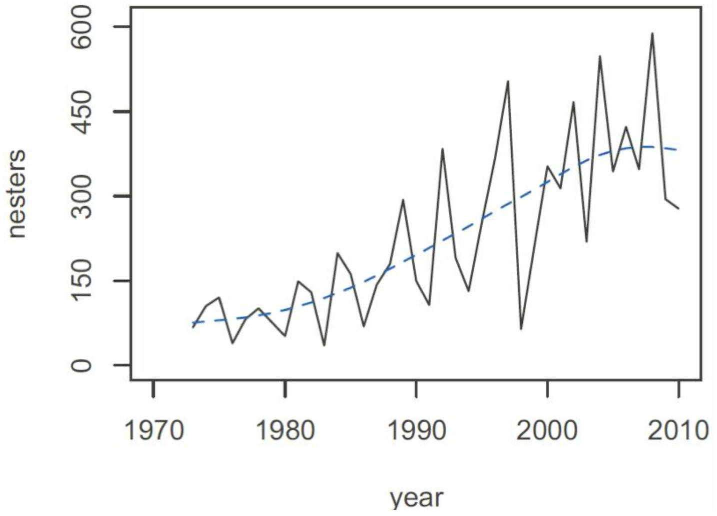


Figure 7-3. Trend in green turtle nests at French Frigate Shoals, Hawaii. Source: Pilcher et al. 2011

Palmyra: No evidence of nesting currently although there were records back in the 1930s of occasional nesting. Foraging turtles are recorded in the lagoon (Stirling et al. 2013).

Johnston Atoll: Green turtles do not nest at Johnston Atoll, but occur foraging within the atoll (Balazs and Forsyth 1986).

7.1.2 Green turtle – South Central Pacific RMU

American Samoa: The primary green turtle nesting location is at Rose Atoll with up to several dozen nests laid annually between October and March (Balazs 2009).

Cook Islands: The Cook Islands consist of fifteen islands divided into two distinct groups: the volcanic southern Cook Islands, and the northern Cook Islands. In the northern Cook Islands, green turtles nest at Penrhyn (Tongareva), Rakahanga, Pukapuka and Manihiki Atolls (Balazs 1995). In the southern Cook Islands, green turtles nest primarily at Palmerston Atoll, which hosts the majority of green turtle nesting. Foraging green turtles have also been observed in many parts of the Cook Islands (White 2012, 2013). The Cook Islands likely hosts <50 turtles per year (NMFS 2010).

Fiji: Fiji consists of an archipelago of more than 300 islands and more than 500 islets located in the South Pacific. There are no long-term studies in Fiji to provide information on sea turtle nesting trends but evidence suggests a decline in nesting green turtles due mainly to overharvest (Batibasaga et al. 2006). Based on Fisheries Surveys, Batibasaga et al. (2006) reported 4,000 to 6,000 resident foraging green turtles in Fiji, of which some 400−500 are killed each year. Between 50 and 75 green turtles are estimated to nest each year in Fiji (Batabisaga et al. 2006).

French Polynesia: French Polynesia consists of 130 islands and atolls spread over a large geographic area in the central south Pacific. In western and central French Polynesia, green turtles have historically been observed nesting at Tupai Atoll, Maupiti, Bellinghausen Atoll, Manihi Atoll, Tetiaroa Atoll, Bora Bora, Mopelia Atoll and Scilly Atoll (Maison et al. 2010). The green and hawksbill sea turtles are the most common species found in French Polynesia (Petit and Gaspar 2011). Currently only the green sea turtle is known to regularly lay eggs in French Polynesia. The major nesting sites are located in the Society Islands, such as Tetiaroa (Petit et al. 2013, Margaux et al. 2018), Scilly, Motu One and Mopelia (Balazs et al. 1995). Other less important sites have also been described, such as Tikehau (Tayalé 2007), Maupiti and Fakarava (association Te Honu Tea). Observations in the late 1970s, early 1980s, and early 1990s suggested 300 to 400 nesting females occurred there annually between November and March (Balazs et al. 1995, Lebeau 1985). While nesting females are counted the effort varies by year and location, and the number of tracks appears to have a more robust annual count. Tuoron et al. (2019) provide a time-series of nests for green turtles in Tetiaroa (Figure 7-4) that, notwithstanding the inter-annual fluctuation, suggest an average of 50 to 100 turtles nest there each year. Elsewhere in French Polynesia it is likely that <100 turtles nest each year.

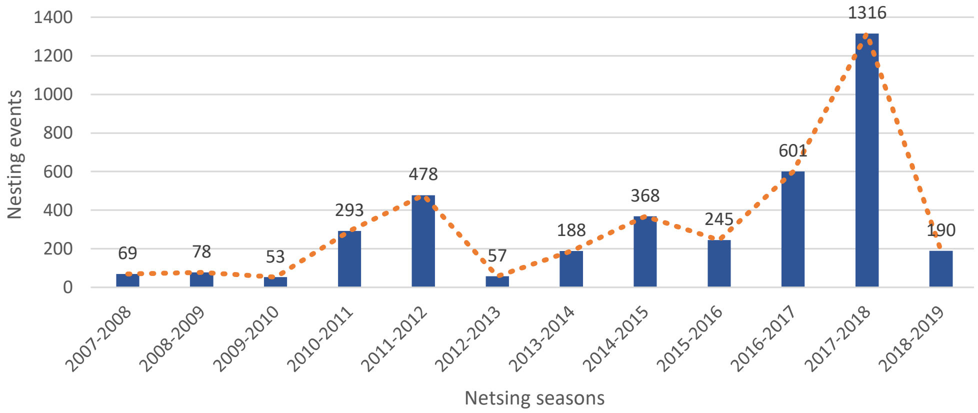


Figure 7-4. Trend in green turtle nests at Tetiaroa, French Polynesia. Source: Tuoron et al. 2019

Kiribati: Kiribati is an island nation that consists of 32 atolls and one raised coral island that are separated into three distinct chains, the Gilbert Islands, Phoenix Islands, and Line Islands (Maison et al. 2010). In the Gilbert Islands, turtle nesting occurs on most of the islands except Banaba, with the most important two areas being Katangatemau, a sandbank about 300 km northeast of Makin Island, and a sandbank by Nonouti Island. In the Phoenix group, nesting has been confirmed at Kanton, Nikumaroro (Gardner), McKean, Rawaki (Phoenix), Ederbury, Manra (Sydney), Orona (Hull) and Birnie. In the Line Group sparse nesting had been cited on Tabuaeran (Fanning) and Kiritimati (Christmas) Islands (Bell et al. 2010). Surveys have been of limited duration and coverage across much of the island chain, but it is likely that the Gilbert Islands host 10 to 50 green turtles per year, and the Phoenix Islands host 100 to 300 turtles per year (Maison et al. 2010). No data is available for the Line Islands.

Nauru: Nauru consists of one small island in the southwestern Pacific Ocean, and no green turtle nesting has been reported. However, the green turtle is listed as one of the country's species under the Convention on Migratory Species (www.cms.int), and sea turtles are also mentioned in the country's National Biodiversity Strategy and Action Plan (https://www.cbd.int/doc/world/nr/nr-nbsap-01-en.pdf). Historical records suggest green turtles in small numbers have nested on Nauru (Buden 2008), but there are no contemporary records. One satellite-tracked green turtle moved through Nauru waters during a circuitous 14,000 km migration from Tahiti (Moorea atoll) to an area near American Samoa, bypassing Fiji, Nauru, Kiribati, Tuvalu and the Marshall Islands.

Niue: Niue consists of a single island located in the centre of the triangle made up by Tonga, Samoa and the Cook Islands, and 2,400 kilometres northeast of New Zealand. Green turtles are found in Niue's waters (Friedlander et al. 2017, Bossarelle et al. 2018), but there are no reports of nesting.

Pitcairn Islands: Of the four islands within the group (Pitcairn Island, Henderson Island, Ducie Atoll and Oeno Atoll), only Henderson Island has records of turtle nesting (Brook 1995, Irving and Dawson 2012). Green turtles nest at Henderson Island with an estimated total of 10 turtles per year (Brooke 1995).

Samoa: Samoa consists of two main and seven small islands. No green turtle nesting sites have been reported for Samoa (Witzel 1982, Ward and Lemalu 2020).

Tokelau: Tokelau consists of three coral atolls, Atafu, Nukunonu and Fakaofo, all of which are known to have green turtle nesting (Maison et al. 2010). Balazs (1983) estimated 120 total nesting females annually in Tokelau.

Tonga: Tonga is composed of at least 170 islands. Green turtles nest in low levels on several islands in the Ha'apai Group as well as islands in the Vava'u Group, with an estimated 10 to 20 green turtles nesting annually based on anecdotal information from turtle hunters (Havea and MacKay 2009). These estimates are supported by field surveys in 2007 and 2008 (Bell et al. 2009).

Tuvalu: Tuvalu is made up of nine coral islands and atolls. Green turtles nest in the main island as well as on several outer islands (Pita 1980). The main nesting area is in the Funafuti Conservation Area on the western side of Tuvalu, which may host up to 10 nesting green turtles in a good year (Bowen 2020).

Wallis and Futuna: Wallis and Futuna consists of three main islands (Uvea, Futuna, Alofi) and 20 low coral or small volcanic islets. Green turtles are known to be present in the waters around Wallis and Futuna (Rudrud 2010) but no reports of green turtle nesting exist.

7.1.3 Green turtle – Central West Pacific RMU

Commonwealth of the Northern Mariana Islands: Summers et al. (2018) estimated a mean abundance of 11.9 nesters per year on Saipan, Tinian and Rota combined. It is estimated that possibly fewer than 10 individual turtles nest annually on the CNMI islands (NMFS and USFWS 1998).

Guam: Guam hosts only low level nesting by green turtles. It is estimated one to four turtles nest on Guam each year (Maison et al. 2010).

Federated States of Micronesia: FSM consists of 607 islands in the western Pacific divided into four states: Yap, Chuuk, Kosrae and Pohnpei. It is estimated that between 500 and 1000 green turtles nest annually in FSM (Maison et al. 2010). The largest rookeries appear to be Ulithi Atoll, ~ 185 km northeast of Yap, East Fayu Island in Chuuk, and Oruluk in Pohnpei. Little or no nesting is known on Kosrae. Since 2005, the Ulithi Sea turtle Program has worked with the local community of Falalop to tag and monitor nearly 3,000 green turtles nesting on the remote and uninhabited islands of Gielop and Loosiep (Cruce and Rulmal 2014). It is likely that up to 100 to 300 turtles per year nest on these islands annually.

Japan: Green turtle nesting in Japan occurs primarily in the Ryukyu and the Ogasawara Islands (Kamezaki et al. 1999). Green turtles nest mainly at Chichi-jima (Chaloupka et al. 2007). Records from the late 19th century show a rapid decline in the sea turtle population between 1880 and 1920. At present, sea turtle harvest in the Ogasawara Islands continues with a harvest limit of 135 mature turtles per year (Ishizaki 2007). The sea turtle population has steadily increased since the early 1980's and has exhibited an estimated annual population growth rate of 6.8% per year (Chaloupka et al. 2007). It is estimated a mean annual total of ~500 nesting females nest in Japan (Chaloupka et al. 2007).

Papua New Guinea: Green turtle nesting has been reported for Nago, Atmago, Ral, Limalam, Usen and Lemus islands, near Kavieng (Maison et al. 2010). Green turtle nesting also occurred in Manus Province, and Long Island in the Madang Province (Spring 1982). Pritchard (1978) identified green turtle nesting in East Sepik Province at Kwala Village, Wom Point, Musschu Island, Kairuru Island, Wuvulu Island and Kaniet Island; Manus Province at Tulu Village on the north coast, Ponam Island, Pak Island, Los Reyes Islands, Harengan Island, Bipi Island and the Ninigo Group of Islands; New Ireland Province in the Boloma Group of Islands; Emirau, Mussau and Emananusa Islands; Eloaue Island and the Tanga Islands; West New Britain Province on the islands off Provincial Capital of Hoskins; East New Britain at Nuguria; Madang Province on the north coast and at Long Island; and in Western Province along the whole coast. The only detailed nesting programme for green turtles has been in the West Calvados Chain and the Conflict Group in the Milne Bay Province (Kinch 2003, Wangunu et al. 2004). Generally there is a large paucity of data on green turtles in PNG given the lack of systemic and ongoing surveys (Kinch 2020), and an estimate of total nests per year is unavailable.

Philippines: Primary nesting of green turtles occurs in the Turtle Islands Wildlife Sanctuary in the Sulu Sea. An estimated 3,000 to 4,000 green turtles nest annually on the six islands (Bureau of Marine Resources, unpublished data), and the trend has been growing since the 1980s (Figure 7-5). It is unknown if these turtles interact with the greater Pacific Ocean populations, but turtles from Pacific countries have been recorded swimming to the Philippines (e.g. Kolinski et al. 2014), so potential exists for the hatchlings produced at this rookery to contribute to Pacific turtle populations.

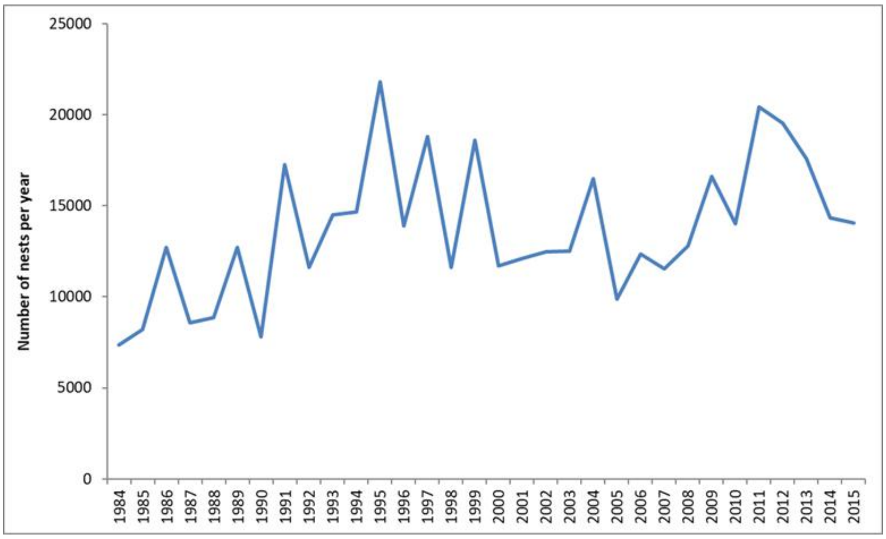


Figure 7-5. Trend in green turtle nests at the Turtle Islands Wildlife Sanctuary, Philippines. Source: Bureau of Marine Resources, unpublished data.

Republic of the Marshall Islands: RMI is made up of 29 atolls and five islands, and the most significant green turtle nesting areas include Bikar, Erikub and the island of Jemo (Maison et al. 2010). The level of exploitation of turtles is unknown, and there are no current data available on status of turtle stocks in the RMI (McCoy 2004). Anecdotal information suggests that the number of nesters has decreased over time, possibly by as much as 50% in the last 10 years (McCoy 2004). NMFS and FWS (1998) estimated a mean annual total of approximately 100 to 500 nesting females at Bikar Atoll.

Republic of Palau: Comprising four populated islands and several hundred smaller islands and atolls, Palau is a major green turtle nesting location, although the number of nesting turtles appears to be declining. Several hundred nests were laid in Sonsorol State in 2004−2005, and another 300 in Hatohobei State (PBMR 2008). This equates to some 100 to 150 individual turtles, and this estimate is supported by tagging data from Hatohobei in the late 2000s (Figure 7-6). Hatohobei State Government unpublished data also indicates that numbers of nesters are declining (Figure 7-6). The remaining states in Palau host one to ten turtles per year.

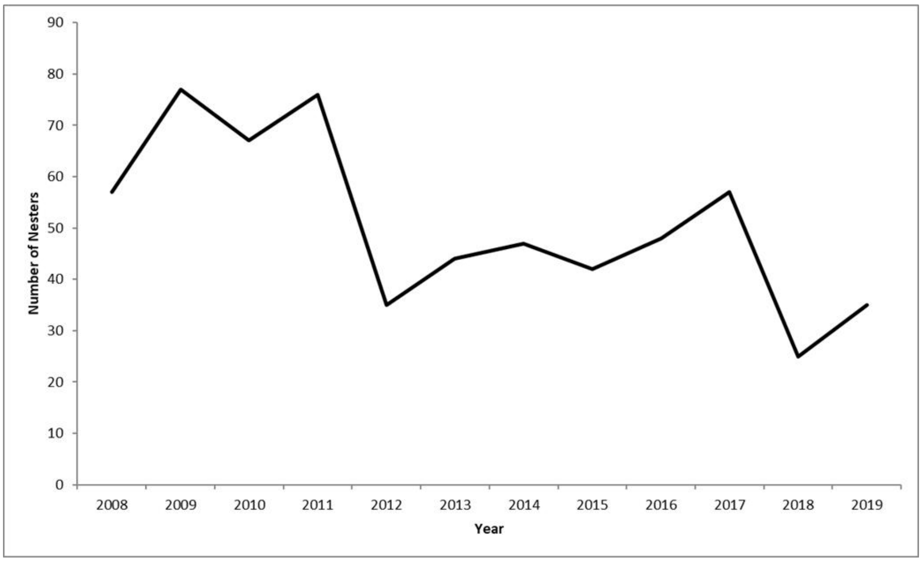


Figure 7-6. Trend in numbers of green turtles tagged at Hatohobei Island, Palau. Data source: Hatohobei State Government, unpublished data)

7.1.4 Green turtle – Southwest Pacific RMU

Australia: Green turtles nest at numerous sites along the east, north and west coasts of Australia as well as many islands and islets offshore and on the Great Barrier Reef and in the Coral Sea. According to Limpus (2007a) Raine Island, Moulter Cay, and Nos. 7 and 8 Sandbanks host over 90% of all green turtle nesting in the northern Great Barrier Reef (nGBR). Seminoff (2004) and NMFS and USFWS (2007) estimated 18,000 and 25,000 annual nesting females at Raine Island, respectively. Chaloupka et al. (2007) reported 4,000 to 12,000 annual nesting females at Raine Island. There is substantial inter-annual variation in the number of nesting females (Figure 7-7), and there was an upward trend in the size of the annual nesting population during 1976 to 1996 followed by a downward trend since 1996, but based on available information, a conservative estimate of the annual mean number of nesters in the nGBR of 10,000 to 25,000 females was suggested by Maison et al. (2010). In the southern Great Barrier Reef (sGBR), major green turtle breeding areas include the islands of the Capricorn Bunker Group: Northwest, Wreck, Hoskyn, Tryon, Heron, Lady Musgrave, Masthead, Erskine, Fairfax, North Reef and Wilson Islands. There is substantial inter-annual variation in the number of nesting females (Figure 7-8), but based on mid-season nightly track counts, the sGBR is estimated to support 5,000 to 10,000 nesting green turtles annually (Limpus 2007a). Lastly, the Coringa-Herald National Nature Reserve (CHNNR), located 440 km east of Queensland, also supports substantive green turtle nesting of between 1,000 and 5,000 turtles (Harvey et al. 2005).

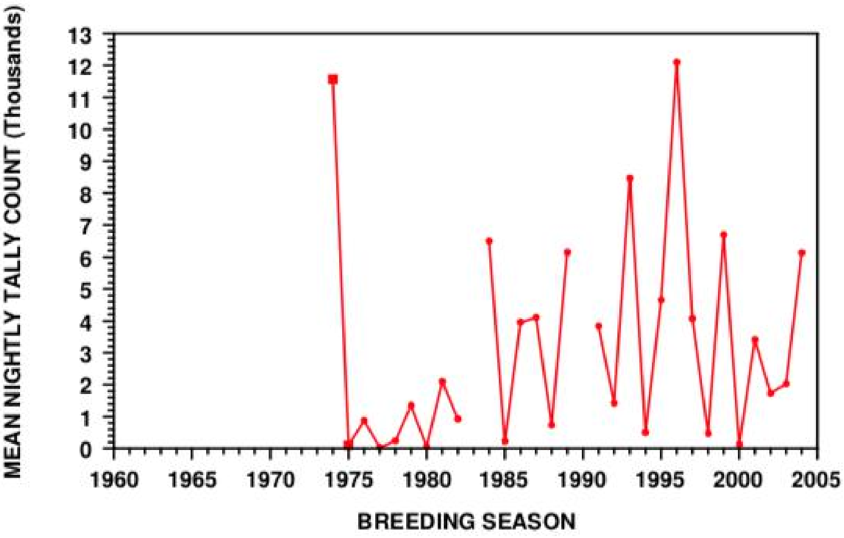


Figure 7-7. Trend in numbers of green turtles tagged at Raine Island, in the nGBR Australia. Data source: Limpus 2007a.

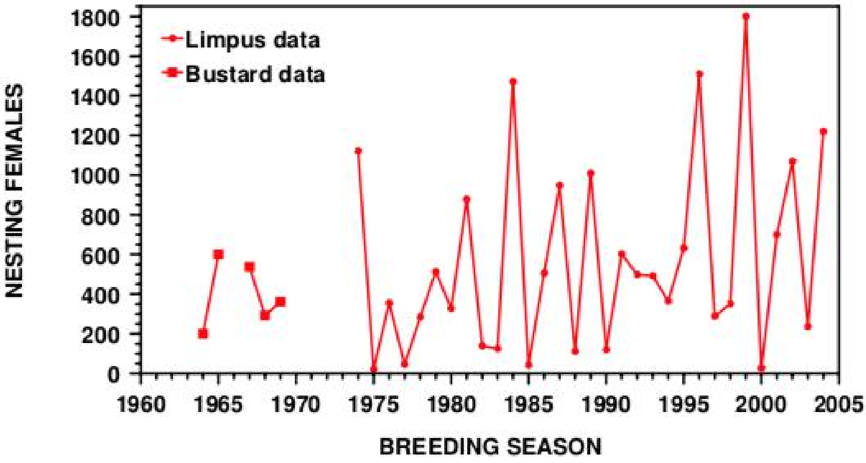


Figure 7-8. Trend in numbers of green turtles tagged at Heron Island in the sGBR, Australia. Data source: Limpus 2007a.

New Caledonia: The largest known nesting area for green turtles is the d'Entrecasteaux islands, located ~300 km north of the main island. Flipper tag recoveries of green turtles in New Caledonia indicate the turtles belonged to the sGBR genetic stock (Read 2015). Particular strong genetic and habitat linkages occur with the d'Entrecasteaux Islands and the sGBR in Australia (Read 2015). New Caledonia is the second largest breeding area for green turtles in the South Pacific (after Australia) with three large rookeries (d'Entrecasteaux: 5000−10,000 turtles; Chesterfields Atoll: 10,000−15,000; and Beautemps-Baupré: 100−500) along with a number of smaller nesting sites (Fonfreyde et al. 2012, Mounier 2007, Read 2012, Farman 2020). With a five to six year remigration interval recorded for turtles of the sGBR genetic stock (Limpus 2007a), this equates to approximately 800−2,000 turtles per year at d'Entrecasteaux, 100−200 turtles per year at the Chesterfields Atoll, and 20−100 turtles per year at Beautemps-Baupré.

Solomon Islands: The Solomon Islands consist of nearly one thousand islands. Three major nesting sites have been identified: Arnavon Islands, Hakelake Island and Kerihikapa Island (Maison et al. 2010). There is limited information available regarding current overall nesting of green turtles. Vaughan (1981) documented green turtle nesting activity within the provinces of Choiseul, Shortlands and Makira primarily on the islands of Wagina, Ausilala, Maifu, Balaka and Three Sisters (Malaulaul and Malaupaina), with approximately 50 to 100 green turtle nests per year at each island. Leary and Laumani (1989) estimated some 259 to 438 nests were deposited in Isabel Province. Vaughan (1981) estimated that the number of breeding individuals of all sea turtle species combined in the Solomons was about 1,500. Mortimer (2002) reported some 250 to 300 nests per year in the Shortlands and 70 to 100 nests in Santa Ysabel and Choiseul. Based on these widely differing estimates, it is likely that the total annual number of nesting females in the Solomon Islands is <300.

Vanuatu: The Republic of Vanuatu is comprised of 82 islands widely distributed in a northwest-southeast orientation. Green turtle nesting occurs on most islands, but systematic and ongoing surveys are lacking. Green turtle nesting has been recorded on the islands of Epi, Espiritu Santo, Malekula, Moso and Nguna, Pele, Motalava, Pentecost, Aniwa and Tegua, and Torres. The highest numbers were recorded on Malekula Island at Bamboo Bay. There is also green turtle nesting around the Maskelyne Islands of Vulai and Sakao (Hickey 2020). The Wan Smolbag Vanua-Tai Resource Monitor programme has a team of Turtle Monitors who collect information on turtles across the country (Hickey and Petro 2005). Recent surveys as part of a dugong and seagrass conservation programme also collected data on sea turtle abundance and trends, but this is not yet analysed. Maison et al. 2010 summarise known nesting as follows: Malekula island was identified in 1979 as an important nesting area with 40 to 120 turtles nesting annually (this likely refers to a combination of greens and hawksbills). Information collected at Wan Smolbag workshops in 2007 and 2008 by monitors of the Vanua-Tai network identified over 189 nesting sites on 33 islands of Vanuatu, with approximately 200 turtles (both green and hawksbill) nesting at Malekula island per year. Additionally, Santo island and Thion supported 50 or more nesting turtles per year, and approximately 30 turtles nest annually at Tegua and Hiu islands. Hickey (2020) reported a total of 353 nests per year, which could equate to a total of 60 to 100 turtles nesting each year. However, coverage of Vanuatu's beaches is not yet comprehensive so total nesting activity is likely to be substantially underestimated (Maison et al. 2010).

7.1.5 Green turtle –East Pacific RMU

Chile: Green turtles do not nest in Chile. Those turtles found in Chilean waters are likely juveniles or sub-adults (Sarmiento-Devia et al. 2015).

Colombia: Only infrequent green turtle nesting is recorded in Colombia, with <10 female turtles nesting each year on Isla Gorgona (Seminoff et al. 2015).

Costa Rica: Cost Rica records green turtles nesting at some 22 distinct nesting locations (Seminoff et al. 2015). Based on figures presented in Seminoff et al. (2015) and the remigration intervals suggested by Alvarado-Diaz and Figueroa (1990), the four largest of these host ~700 green turtles each year: Playa Nombre de Jesus-Zapotillal (~400 turtles per year); Playa San José (~150); Playa Cabuyal (~100), and Playa Colorada (~75). The balance rookeries contribute an additional 200 green turtles per year, although few long-term monitoring projects have been conducted at these sites (Seminoff et al. 2015). Conservatively, it is estimated some 1,500 green turtles nest in Costa Rica each year.

Ecuador: Green turtles nest on the Galapagos Islands and at the Machalilla National Park (MNP) on the Pacific coast of mainland Ecuador (Anhalcer et al. 2012). Seminoff and Glass (2020) indicate that current nester abundance for the Galapagos is in the order of 2,000 female turtles. MNP hosts only 10 to 20 turtles per year. Alvarado-Díaz and Figueroa (1990) indicate a remigration interval of three years for these turtles, which suggests an annual nester abundance for Ecuador of ~ 600−700.

El Salvador: There are sporadic nesting events at Barra de Santiago and Jiquilisco, but nesting abundance is undocumented. No green turtle nesting estimates are available for El Salvador, and no beaches are listed as nesting sites in Seminoff et al. (2015). It is likely <10 turtles per year nest in El Salvador.

Guatemala: There are sporadic nesting events, but nesting abundance is undocumented. No green turtle nesting estimates are available for Guatemala, and no beaches are listed as nesting sites in Seminoff et al. (2015). It is likely <10 turtles per year nest in Guatemala.

Mexico: Colola Beach, Michoacán, hosts the largest rookery for green turtles in the East Pacific, with approximately 8,000 total nesting females (Seminoff and Glass 2020). Accounting for remigration intervals, this equates to some 2,700 nesters per year. It is estimated that ~75% of the entire Michoacán population of adult females nests in Colola, with the balance rookeries located at Llorona, Bahia Maruata, Martin de Oro, Arenas Blancas (all in Michoacan) and at Care Region, Baja California Sur and on Revillagigedos archipelago (Seminoff et al. 2015). Given the proportion of nests from Colola, and based on figures presented in Seminoff et al. (2015) and the remigration interval presented by Alvarado-Diaz and Figueroa (1990), it is suggested Mexico has a total of ~3,500 green turtles nesting each year.

Nicaragua: There are sporadic nesting events, but nesting abundance is undocumented. No green turtle nesting estimates are available for the Pacific coast of Nicaragua, and no beaches are listed as nesting sites in Seminoff et al. (2015).

Panama: There are sporadic nesting events, but ongoing nesting assessments are lacking. Green turtles nest at Playa Rio Amarillo on Coiba Island (at least 200 nests) and at Playa Gato on mainland Panama (~300 nests) (E. Flores, unpublished data). This likely equates to some 400 to 500 annual nesting green turtles.

Peru: There are sporadic nesting events, but nesting abundance is undocumented. Forsberg et al. (2012) documented the first known instance, but ongoing monitoring is sporadic. It is likely <1 turtle per year nests in Peru.

7.2 Number of Nesting Turtles – Hawksbill Turtles

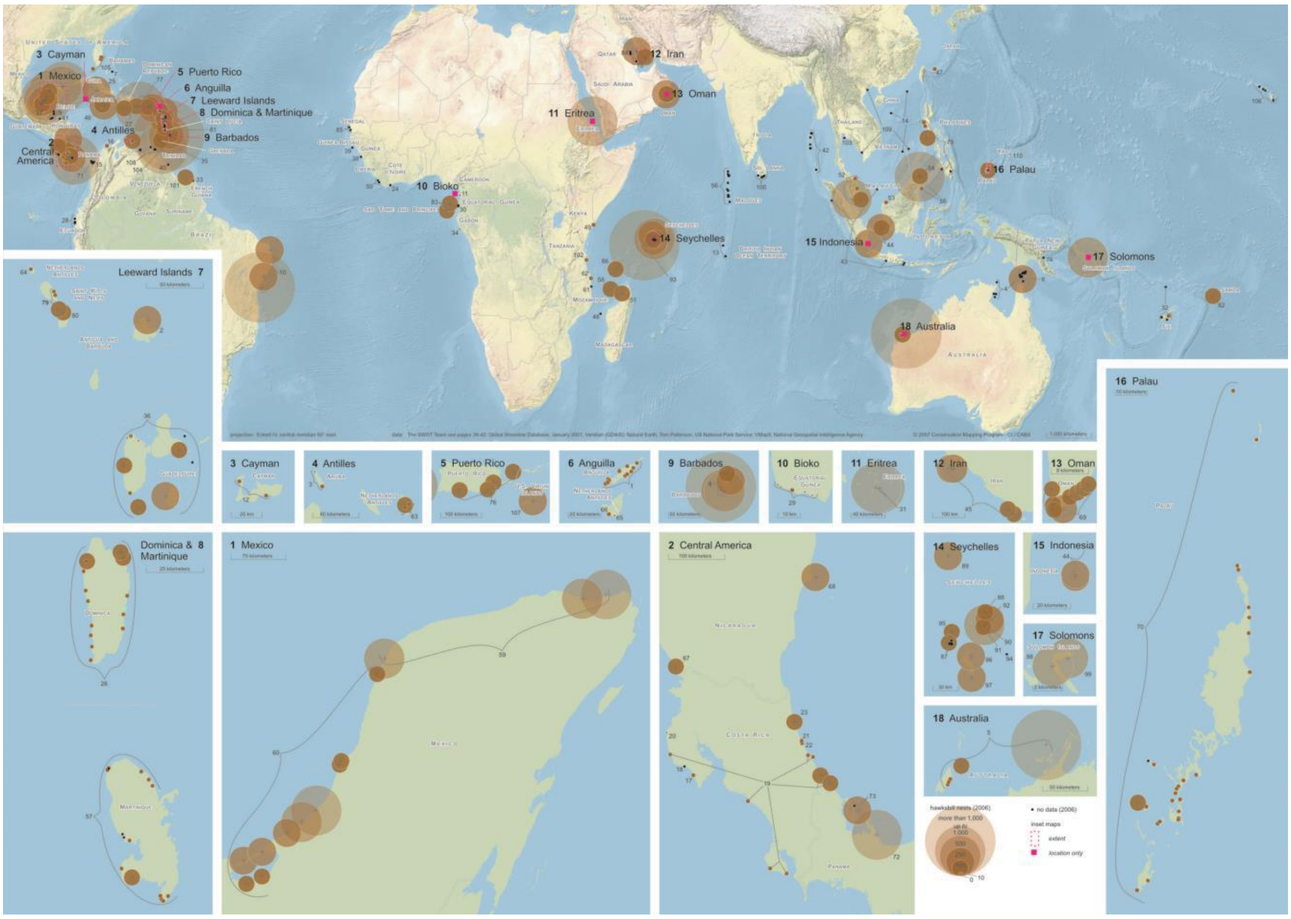


Figure 7-9. Global hawksbill nesting sites reported to SWOT in 2006. Image source: SWOT Report III.

Among the most comprehensive updates on global hawksbill nesting is the IUCN Assessment conducted by Mortimer and Donnelly (2008). This assessment summarised the extent of knowledge on nesting distribution and numbers, trends and threats for hawksbills at a global level. Since that time there has been a substantial increase in knowledge on the species in the east Pacific and a substantial amount of updated literature; but fewer updates for countries across the central and west Pacific except for Australia and the Solomon Islands, where research and conservation activities are ongoing and permanent. Where these are available they are presented, and in the absence of newer information the Mortimer and Donnelly (2008) assessment provides the most recent information.

7.2.1 Hawksbill turtle – North Central Pacific RMU

Hawaii: Hawksbill turtles nest (Parker et al. 2009) and foraging (e.g. Graham 2009, King 2013) in Hawaiian waters. Van Houtan et al. (2016) noted that small hawksbills were found in Hawaii without any evidence of long-distance oceanic phases, which suggests there is a possibility turtles that nest in Hawaii remain in Hawaii. Snover et al. (2012) indicate that the Hawaiian hawksbill population has fewer than 20 females nesting per year. This data is supported by tagging data presented by Steitz et al. (2012). Van Houtan et al. (2012) report on extremely sporadic hawksbill nesting in the Northwest Hawaiian Islands. It is likely only 10 to 20 hawksbills nest each year in Hawaii.

Palmyra: Sterling et al. (2013) report on foraging hawksbills in Palmyra but do not indicate hawksbills as frequent nesters. It is likely fewer than one turtle per year, if any, nests on Palmyra.

Johnston Atoll: No evidence of hawksbill nesting on Johnston Atoll was reported by Baker et al. (2006). Balazs and Forsyth 1986 reported that "no historic records are known of turtles nesting at Johnston Atoll" and similarly did not record any of their own during two field expeditions to the island.

7.2.2 Hawksbill turtle – South Central Pacific RMU

American Samoa: NMFS and USFWS (1998) indicated there may be up to 80 nesting females per year in Tutuila and the Manu's island group. However, Mortimer and Donnelly (2008) indicate only <10 to 30 female hawksbill turtles nest per year in American Samoa and Samoa combined. It is likely that <10-15 female hawksbill turtles nest in American Samoa annually.

Cook Islands: White (2012) reported that "Hawksbill turtles *E. imbricata* have not been found nesting in the Cook Islands recently", but hawksbills are found in Cook Islands waters (White 2012, 2013).

Fiji: Mortimer and Donnelly (2008) suggested 100 to 200 female hawksbill turtles nest per year in Fiji. However, the most recent nesting assessments are presented by Prakash et al. (2020), who report on nesting records from 2015 to 2019. There are multiple nesting sites in Fiji (27) with only 147 nests recorded over the four years. Given the distance and isolation of many of the nesting sites, it is likely that (a) there are a few nesting turtles per site in any given year, and (b) that this is an underestimate due to logistics of full-time monitoring. Yadu and Yadua Taba recorded 35% of all nesting in Fiji over this period, followed by Katawaqa and Nukuvadra (29%). Given the lack of long-term mark-recapture surveys and clutch estimates per female per season, it is not possible to ascertain how many female turtles nest in a given year, but it is likely Fiji has an annual nester abundance of only 20 to 30 turtles.

French Polynesia: Mortimer and Donnelly (2008) do not provide any reports on hawksbill nesting in French Polynesia. Similarly the US Recovery Plan from 1998 simply states that "nesting [in the Central Pacific] is widely distributed and scattered and in very low numbers" (NMFS and USFWS 1998). At present it is not possible to determine a number of females per year.

Kiribati: NMFS and USFWS (1998) and Mortimer and Donnelly (2008) do not provide any reports on hawksbill nesting in Kiribati. At present it is not possible to determine a number of females per year.

Nauru: NMFS and USFWS (1998) and Mortimer and Donnelly (2008) do not provide any reports on hawksbill nesting in Nauru. At present it is not possible to determine a number of females per year.

Niue: NMFS and USFWS (1998) and Mortimer and Donnelly (2008) do not provide any reports on hawksbill nesting in Niue. At present it is not possible to determine a number of females per year.

Pitcairn Islands: NMFS and USFWS (1998) and Mortimer and Donnelly (2008) do not provide any reports on hawksbill nesting in the Pitcairn Islands. At present it is not possible to determine a number of females per year.

Samoa: Mortimer and Donnelly (2008) indicate <10 to 30 female hawksbill turtles nest per year in Samoa and American Samoa combined. It is likely that <5 to 15 female hawksbill turtles nest in Samoa annually.

Tokelau: NMFS and USFWS (1998) and Mortimer and Donnelly (2008) do not provide any reports on hawksbill nesting in Tokelau. At present it is not possible to determine a number of females per year.

Tonga: NMFS and USFWS (1998) and Mortimer and Donnelly (2008) do not provide any reports on hawksbill nesting in Tonga. At present it is not possible to determine a number of females per year.

Tuvalu: NMFS and USFWS (1998) and Mortimer and Donnelly (2008) do not provide any reports on hawksbill nesting in Tuvalu. At present it is not possible to determine a number of females per year.

Vanuatu: Vanuatu may well fall under the southwest Pacific RMU following upcoming revisions by the MTSG, particularly with the linkages between Vanuatu turtles and Australia/New Caledonia recently presented by Rice et al. (2018), but at present it is listed as being in the south central Pacific – albeit at the extreme western extent. Mortimer and Donnelly (2008) report > 300 female hawksbill turtles per year in Vanuatu, nesting on Banks/Torres, Malekula, Epi, Green and Aneitum. Recent research on dugongs in Vanuatu also acquired information on sea turtles, but these data have yet to be analysed. Rice et al. (2018) report ~7 nests/year at Wiawi and 58 nests per year at Bamboo Bay, on Malekula Island. Mortimer and Donnelly (2008) also suggest the numbers may be declining, but the available information suggests ~300 nesting female hawksbills per year.

Wallis and Futuna: NMFS and USFWS (1998) and Mortimer and Donnelly (2008) do not provide any reports on hawksbill nesting in Wallis and Futuna.

7.2.3 Hawksbill turtle – West Central Pacific RMU

Mortimer and Donnelly (2008) combine nesting for Micronesian islands and suggest ~300 nesting females use the islands each year. These include the Commonwealth of the Northern Mariana Islands, Guam, the Republic of Palau, the Federated States of Micronesia, the Marshall Islands and Kiribati. However, they do point out those places where some data is available.

Commonwealth of the Northern Mariana Islands (CNMI): NMFS and USFWS (1998) and Mortimer and Donnelly (2008) do not report any nesting on CNMI. Summers et al. (2013) and Summers et al. (2017) documented predominantly juvenile and sub-adult sized turtles in the waters of CNMI, and they referred to nesting hawksbills, but no additional data was provided.

Guam: NMFS and USFWS (1998) indicated there was no hawksbill nesting on Guam. More updated information in Mortimer and Donnelly (2008) indicate 5 to 10 female hawksbill turtles nest per year on Guam.

Federated States of Micronesia (FSM): There is little information on hawksbills in FSM. Buden and Edward (2001) indicate that nesting was infrequent in Pohnpei, but offer no suggestion on annual nester volume. Buden (2000) did not report any hawksbill nesting in Yap. It is likely few hawksbills nest on these islands, possibly less than 10 to 20 per year.

Japan: The hawksbill turtle occasionally nests in the Ryukyu Archipelago (Hirate and Shimoike 1995) and on Okinawajima Island (Teruya 1995), and nesting has historically been reported for Kuroshima Island (Hirate 1988). However, these instances are rare and scattered, and it is likely less than 10 to 20 turtles per year nest in Japan annually.

Philippines: Hawksbills nest in low densities throughout the Philippines, but no major nesting aggregations have been identified (Palma 1994, 1997). Likely only 20 to 30 nesters per year (DENR, unpublished data).

Republic of the Marshall Islands (RMI)s: NMFS and USFWS (1998) indicated that infrequent nesting may occur in RMI, but no current data are available.

Republic of Palau: Both the NMFS and USFWS (1998) and Mortimer and Donnelly (2008) indicate 20 to 50 female hawksbill turtles nest per year in the Republic of Palau.

7.2.4 Hawksbill turtle – Southwest Pacific RMU

Australia: Hawksbill turtles nest in low density on multiple islands throughout the nGBR and Torres Strait areas of eastern Australia (Limpus 1980, Limpus and Miller (2008) and Limpus (2009b) lists the following breeder abundances: an estimated >500 nesting females per year on Long (Sassie) Island, Hawkesbury Island and Dayman Island in the Torres Strait; an estimated 100–500 nesting females per year on the following Great Barrier Reef islands: Milman Island, Boydong Island, Mt Adolphus Island, Albany Island. Torres Strait: Zuizin Island, Mimi Island, Bourke Island, Aukane Island, Layoak Island, Bet Island, Saddle Island, Dadalai Island, Albany Island and Mt Adolphus Island; an estimated 10–100 nesting females per year on other islands throughout the Great Barrier Reef and Torres Strait and the mainland coast of Western Cape York Peninsula north of Cotterell River; and an estimated 1–10 nesting females per year on other islands. Milman Island was considered one of Australia's most important hawksbill rookeries (Miller et al. 1995), but has witnessed severe declines in the last three decades (Figure 7-10; Bell et al. 2012). Current estimates suggest the annual number of nesters at Milman Island is down to ~200. In addition, there were 220 nesting females in 2009 and 580 females in 2010 at the Groote Eylandt archipelago in the Gulf of Carpentaria, northern Australia (Hoenner et al. 2016). This suggests the total Torres Strait and eastern Australia annual hawksbill nester abundance is some 500 to 1,500 females each year.

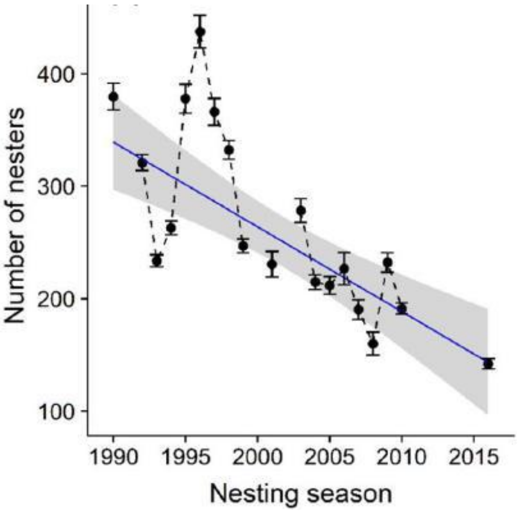


Figure 7-10. Modelled trend in numbers of hawksbill tagged nesting on Milman Island, Australia.

Data source: Bell et al. 2020.

New Caledonia: Meylan and Donnelly (1999) indicated that few hawksbills were reported to nest in New Caledonia. D'Auzon (2007) reported that the main population (about 200 individuals) is located on the northeast coast, but it was unclear if this referred to in-water turtles, and no key nesting site was reported. Recent surveys suggest there is no nesting in New Caledonia (T. Read, pers. comm.).

Papua New Guinea: Mortimer and Donnelly (2008) suggested 500 to 1,000 females may nest annually in PNG. The Conflict Island Conservation Initiative (2008) encountered seven turtles in two weeks of surveys in the Conflict Islands in 2018−2019. 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 may be < 500 turtles per year.

Solomon Islands: The largest rookery for hawksbill turtles in the oceanic South Pacific is the Arnavon Islands, in the western Solomon Islands. Following 150 years of commercial exploitation, the population appears to be recovering (Hamilton et al. 2015). Mortimer and Donnelly (2008) suggested an annual breeder abundance of 200 to 300 females per year. However, the average number of turtles per year at Kerehikapa between 2000 and 2012 was only 50 turtles (Hamilton et al. 2012), and Kerhikapa represents some 35%−45% of all nests in the Arnavons (Mortimer 2002), which suggests the total number of females using the islands each year might more realistically be 110 to 140. These numbers are confirmed also by counts in recent years, during which an estimated 600 egg clutches were laid annually in the Arnavons, representing a minimum of ~125 to 150 females nesting annually (Mortimer 2002).

7.2.5 Hawksbill turtle – East Pacific RMU

Records of nesting for the east Pacific (EP) hawksbill RMU have grown steadily since 2007 (Figure 7-11; Gaos et al. 2017) following the 'rediscovery' of the species on EP beaches. This increase likely reflects an increase in documentation rather than zero nesting between 1983 and 2007.

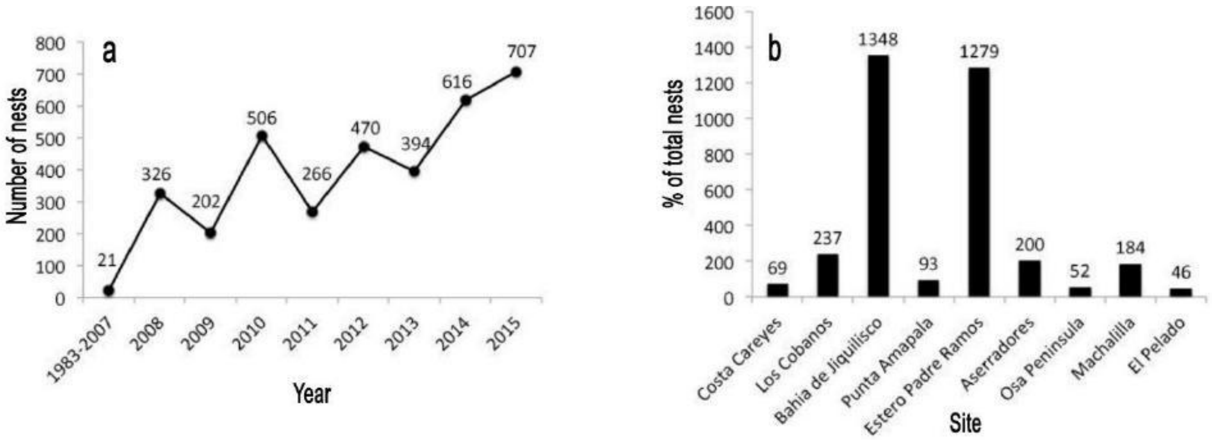


Figure 7-11. Trend in numbers of hawksbill nests (a) and proportional rookery contribution in the east Pacific hawksbill RMU. Data source: Gaos et al. 2017.

Chile: No records of hawksbill turtles exist for mainland Chile. However, hawksbill turtles have been identified in Easter Island (Rapa Nui) waters (Álvarez-Varas et al. 2015a,b).

Colombia: No records of nesting hawksbills were identified by Gaos et al. 2010.

Costa Rica: Gaos et al. (2010) reported 48 records of nesting hawksbills in Costa Rica over a 25 year period. An additional season of data in 2010 recorded 52 nests, and given the latter figure, it is possible that Costa Rica hosts one to five nesting female turtles per year.

Ecuador: Gaos et al. (2010) reported 31 records of hawksbills in Mexico nesting over a 25 year period. Gaos et al. (2017) reported an average of ~20 nests per year at Machalilla since 2007, and one season of 46 nests at El Pelado. Given this, it is possible Ecuador hosts 10 to 15 turtles per year.

El Salvador: Gaos et al. (2010) reported 538 records of hawksbills in El Salvador nesting over a 25 year period (~22 nests per year). However, annual monitoring since 2008 has documented ~60 nests per year at Los Cobanos, ~170 nests per year at Bahia de Jiquilisco, and ~23 nests per year at Punta Amapala (Liles et al. 2011, Gaos et al. 2017). Clutch frequency for these sites was 2.1 nests / year (Gaos et al. 2017), which equates to ~120 turtles per year.

Guatemala: Gaos et al. (2010) reported only 2 records of hawksbills in Guatemala nesting over a 25 year period, which equates to <1 nesting female turtle per year.

Mexico: Foraging juvenile turtles have been recorded off Baja California, but no nesters were reported historically (Seminoff et al. 2003a). Gaos et al. (2010) reported 73 records of hawksbills in Mexico nesting over a 25 year period. Using an annual average of ~7 nests per year at Costa Careyes, this equates to two or three nesting female turtles per year.

Nicaragua: Gaos et al. (2010) reported 31 records of hawksbills in Nicaragua nesting over a 25 year period. However, surveys since 2010 at Estero Padre Ramos and 2014 at Aserradores have documented a total annual average of approximately 215 nests per year, which equates to some 95 female turtles per year.

Panama: There is no current published data on hawksbill turtle nesting in Panama.

Peru: There is no current published data on hawksbill turtle nesting in Peru.

7.3 Number of Nesting Turtles – Loggerhead Turtles

7.3.1 Loggerhead turtle – North Pacific RMU

Japan: Nesting for this RMU only occurs in Japan (Kamezaki et al. 2003, Casale and Matsuzawa 2015, Martin et al. 2020). Annual nesting counts In Japan occur at 35 index beaches, while non-contiguous data are available for an additional 27 smaller beaches (Figure 7-12). Foraging occurs at different locations in waters of the north Pacific Ocean, dependent on life stage and foraging strategy (Peckham et al. 2011). Based on stable isotope analyses and satellite telemetry, Hatase et al. (2002) also demonstrated that some adult female loggerheads nesting in Japan inhabit oceanic habitats rather than neritic habitats. Small juveniles are passively transported via the Kuroshio Current, Kuroshio Extension, and other parts of the north Pacific Gyre (Okuyama et al. 2011). They also actively swim against currents to remain in the Kuroshio Extension Current Bifurcation Region and the Transition Zone Chlorophyll Front (Polovina et al. 2004, Briscoe et al. 2016) or to reach foraging areas in the eastern Pacific (Okuyama et al. 2011).

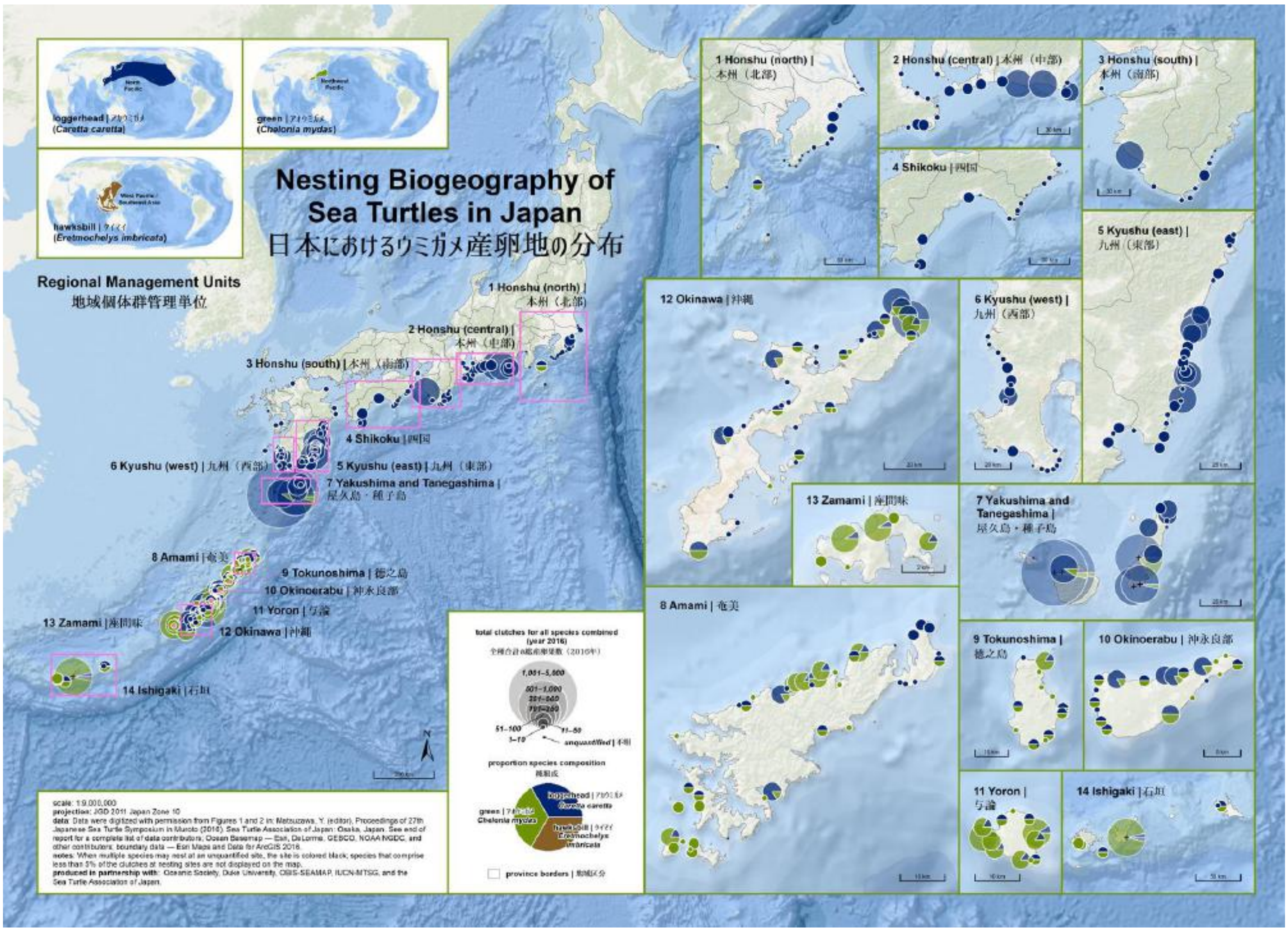


Figure 7-12. Nesting biogeography of sea turtles in Japan (blue circles denote loggerhead nesting). Image source: SWOT Report XIII.

Casale and Matsuzawa (2015) assessed time series datasets of 10−41 yrs from 35 nesting beaches in Japan. They estimated the total number of nests in the subpopulation, including beaches with <10 years of monitoring (62 nesting beaches in total), was about 9,050 nests/year (average between 2009 and 2013). NMFS/USFWS (2020) estimated that the extrapolated 2015 total nester abundance for the Japanese beaches was approximately 8,733 nesting females (95% credible limit of 7,834 to 9,736 nesting females). The report noted that the narrow credible interval reflects high confidence in the available data but indicated that data were unavailable for 48% of the nesting population. Martin et al. (2020) reported an increase in nesting at the three key beaches (Inakahama, Maehama and Yotsusehama on Yakushima) since the 1980s (Figure 7-13), in agreement with the estimates developed by Casale and Matsuzawa (2015), but this is problematic to reconcile when other large-scale beaches, such as Miyazaki, where over 1,200 nests are recorded per year

(Takeshita 2006), are included in the assessment. Given an average three-year remigration period, it is likely then that the annual breeder abundance in Japan is ~2,800 to 3,500 loggerheads/year. This estimate is further supported by the combined data of 9,050 nests/year from Casale and Matsuzawa (2015) and Sato (2016) that suggests loggerheads deposit an average of 3 to 5.5 nests per season, which would equate to an annual nester abundance of ~1,600 to 3,000, particularly given that Martin et al. (2020) indicate the numbers of nests per year have increased slightly since the 2015 assessment, and considering the Martin et al. assessment did not consider all Japan's nesting beaches.

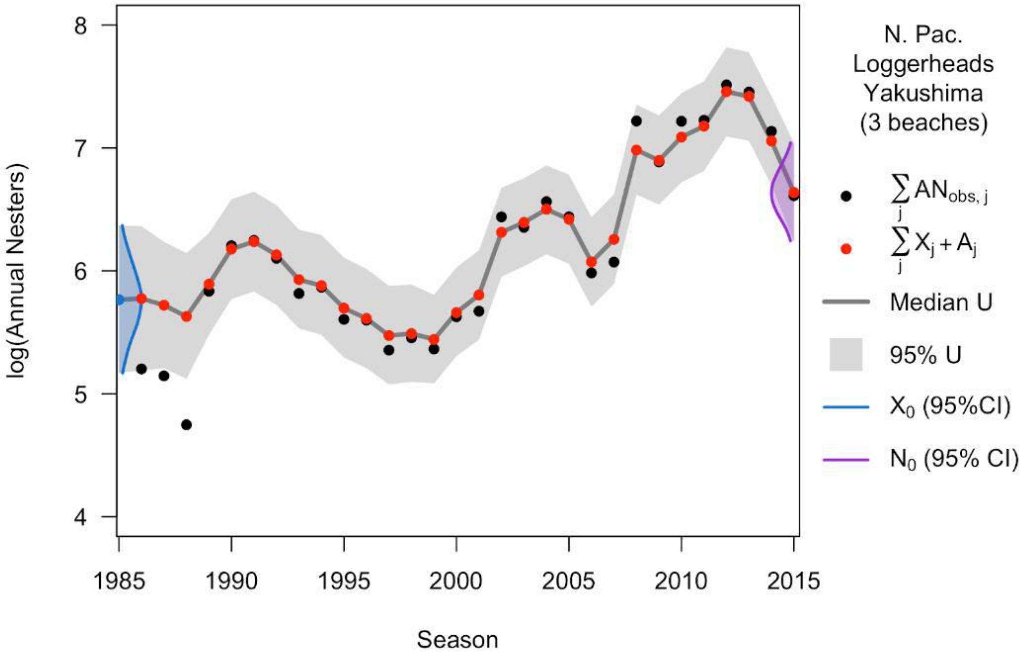


Figure 7-13. Model of nesting trend from 1985 to 2015 in Yakushima, Japan. The grey line depicts the median long-term trend. The grey shading depicts the 95% credible interval of the model fit. Observed data are shown in black and model-predicted data are shown in red. The purple line depicts the distribution around 2015 model-predicted data. Image source: Martin et al. (2020).

7.3.14 Loggerhead turtle – South Pacific RMU

Australia: The major eastern Australian breeding aggregations occur in three main areas: on the mainland coast of southeast Queensland (especially Mon Repos and adjacent beaches of the Woongarra Coast and Wreck Rock Beach; minor breeding aggregations on the mainland coast south from Bustard Head to the Sunshine Coast and on northern ends of Fraser, Moreton and North Stradbroke Islands; and on the 13 islands of the Capricorn-Bunker Groups of the southern Great Barrier Reef and the Swain Reefs. The latter sites are mostly small aggregations.

The eastern Australian nesting population declined from approximately 3,500 females per year in the mid-1970s to approximately 500 in 2000 (Limpus and Limpus 2003, Limpus 2007c; Figure 7-14). The decline in breeding numbers was attributed primarily to by-catch mortality in otter trawl fisheries of northern and eastern Australia (Robins et al. 2002). Of note however, is that this fishery now uses Turtle Excluder Devices and mortality is now negligible.

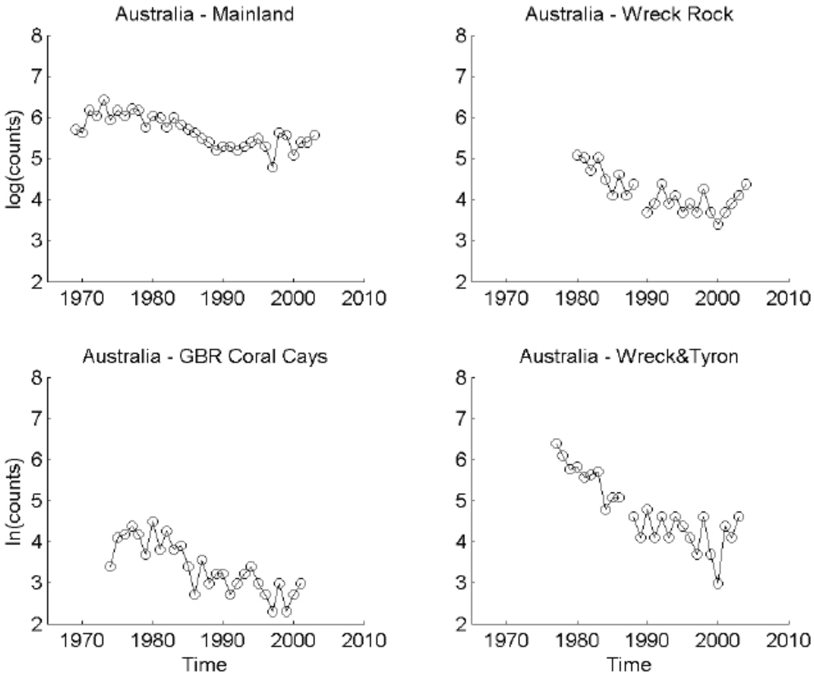


Figure 7-14. The change in the numbers of nesting females at four nesting beaches in the South Pacific Ocean RMU. Image source: Limpus 2007c.

New Caledonia: Recent genetic analysis has confirmed that loggerhead turtles nesting on the beach of Roche Percée belong to the same population as the turtles at Mon Repos in Queensland and therefore the Australian ("southwestern Pacific stock") genetic group (Boyle et al. 2009). The major nesting site for loggerhead turtles is on the Grande Terre in the heart of the Roche Percée Nature Reserve, with ~300 crawls per year (resulting in 182 nests; Fournière et al. 2015), and Farman (2020) reports an additional ~100 to 120 nests/year at multiple non-index sites (Figure 7-15). d'Auzon (2007) reported some 200 nests at Roche Percée and suggested the total number of mature females was ~250. Étaix-Bonnin et al. (2011) reported the number of nesting females was ~200.

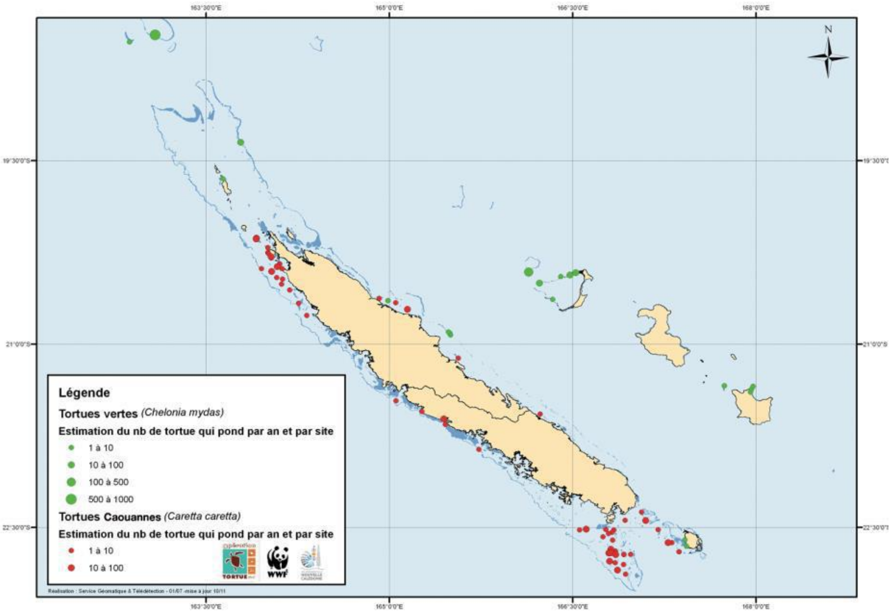
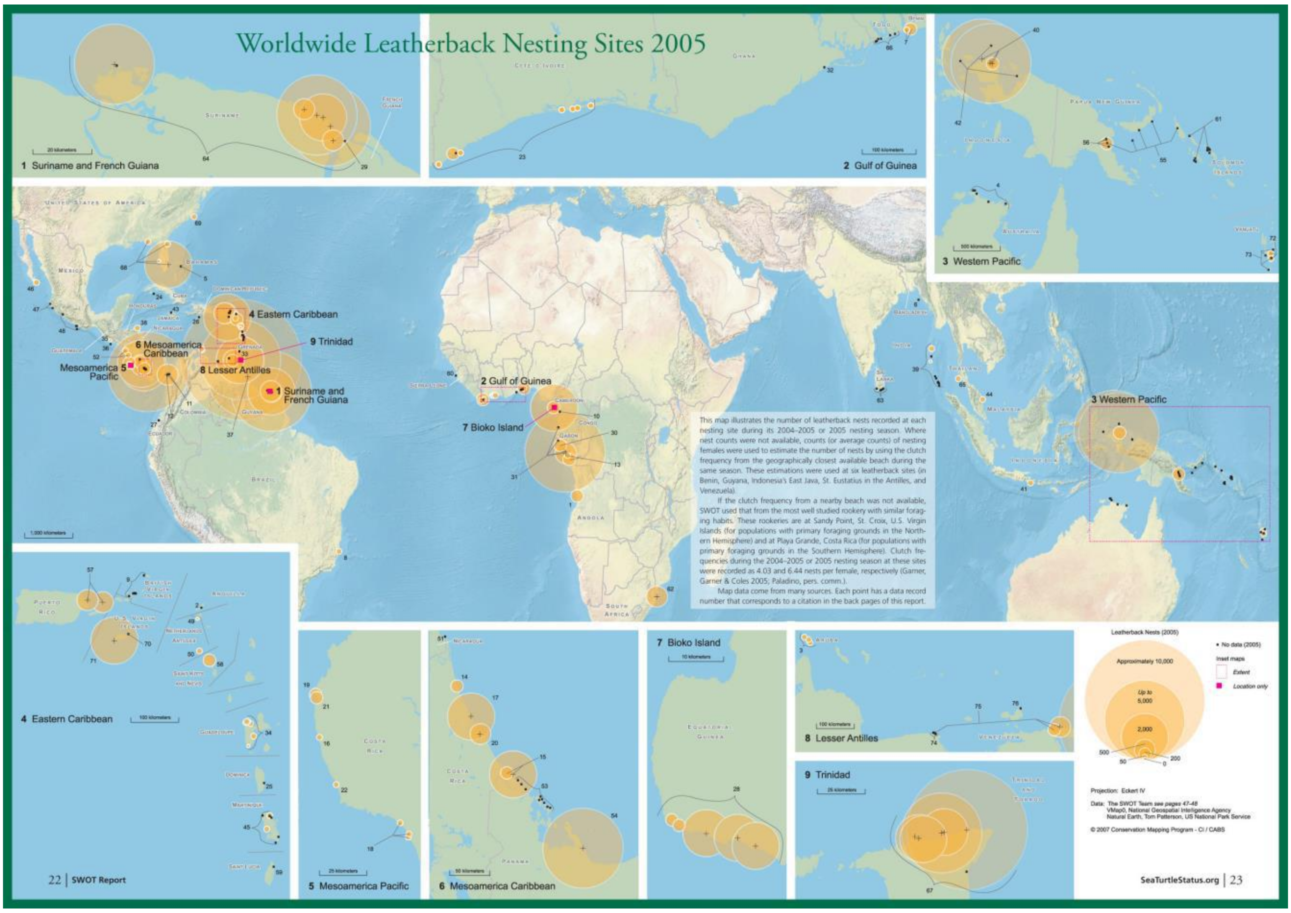


Figure 7-15. Location of green and loggerhead (in red) nesting sites in New Caledonia. Image source: Étaix-Bonnin et al. 2011.

7.4 Number of Nesting Turtles – Leatherback Turtles

Leatherback turtles in the Pacific Ocean are divided into two RMUs. The western Pacific encompasses nesting sites in West Papua, Indonesia, Papua New Guinea and the Solomon Islands. Occasional leatherback nesting also occurs on the northern Australian coast. In the eastern Pacific, nesting occurs in Mexico and Costa Rica (Figure 7-16).



This map illustrates the number of leatherback nests recorded at each nesting site during its 2004–2005 or 2005 nesting season. Where nest counts were not available, counts (or average counts) of nesting females were used to estimate the number of nests by using the clutch frequency from the geographically closest available beach during the same season. These estimations were used at six leatherback sites (in Benin, Guyana, Indonesia's East Java, St. Eustatius in the Antilles, and Venezuela).

If the clutch frequency from a nearby beach was not available, SWOT used that from the most well studied rookery with similar foraging habits. These rookeries are at Sandy Point, St. Croix, U.S. Virgin Islands (for populations with primary foraging grounds in the Northern Hemisphere) and at Playa Grande, Costa Rica (for populations with primary foraging grounds in the Southern Hemisphere). Clutch frequencies during the 2004–2005 or 2005 nesting season at these sites were recorded as 4.03 and 6.44 nests per female, respectively (Garner, Garner & Coles 2005; Paladino, pers. comm.).

Map data come from many sources. Each point has a data record number that corresponds to a citation in the back pages of this report.

Projection: Eckert IV

Data: The SWOT Team see pages 47–48
VMap0, National Geospatial Intelligence Agency
Natural Earth, Tom Patterson, US National Park Service

© 2007 Conservation Mapping Program - CI / CABS

Figure 7-16. Worldwide nesting sites in 2005. Image source: SWOT Report II.

7.4.1 Leatherback turtle – West Pacific RMU

Western Pacific leatherbacks have declined by more than 80% since the mid-20th century, from greater than 12,000 nests per year (which corresponds to 2,600 females per year) to less than 2,200 nests per year (Tapilatu et al. 2013). Key nesting sites are in Indonesia, Papua New Guinea, the Solomon Islands, occasionally in Vanuatu, and with negligible, remnant nesting in Australia.

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 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 and McLachlan, 1994; Limpus et al. 1984). A single nesting was recorded at Mackay in central Queensland. 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.

Indonesia: The key nesting beaches in Indonesia are at Jamursba Medi and at Warmon, West Papua province. 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 at Warmon in 2003 and 2004 respectively. No data were available on clutch frequency, but the authors used proxy data from the East Pacific (4.4 to 5.8 nests per female) to calculate the number of nesters. They surmised annual nesters at Jamursba Medi at 501 to 660 in 2003; and 667 to 879 in 2004 after adjusting for season length (Hitipeuw et al. 2007). However, this population has continued to decline since 1982 (Figure 7-17). The number of females nesting annually as of 2011 was estimated to be 382 during the boreal summer and 131 during the austral summer, based on estimated clutch frequency and clutch interval (Tapilatu et al. 2013). Indonesia likely has a total annual nester abundance of ~500 turtles.

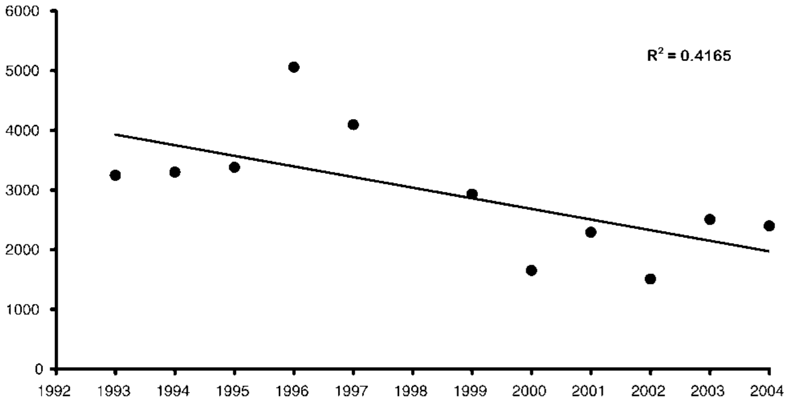


Figure 7-17. Decline of leatherback nesting at Jamursba Medi. Nest counts are for a portion of the nesting season from beginning of June through end of September and do not represent total nests for the year. Image source: Hitipeuw et al. 2007.

Papua New Guinea: Spring (1982) reported regular, but low-density leatherback nesting activity along the north coast of PNG and on several islands including Manus, Long, New Britain, New Ireland and Normanby. However, leatherback nesting in Papua New Guinea takes place nearly entirely on the northern Huon coast in the Morobe province (Pilcher 2013). Long-term assessments of turtles along the Huon coast documented nesting at Busama, Labu Tale and Labu Miti, Kamiali, Sapa, Kobo and Paiawa. Nesting trends from 2000 to 2012 suggest no noticeable trend across years (Figure 7-18), however an analysis of longer term trends suggest this population has also declined, much as the one in Indonesia (Figure 7-19). Given known clutch frequencies (Kisokau 2005, Pilcher 2006), it is estimated that the total annual breeder abundance along this coast is 50 to 200 female leatherbacks per year.

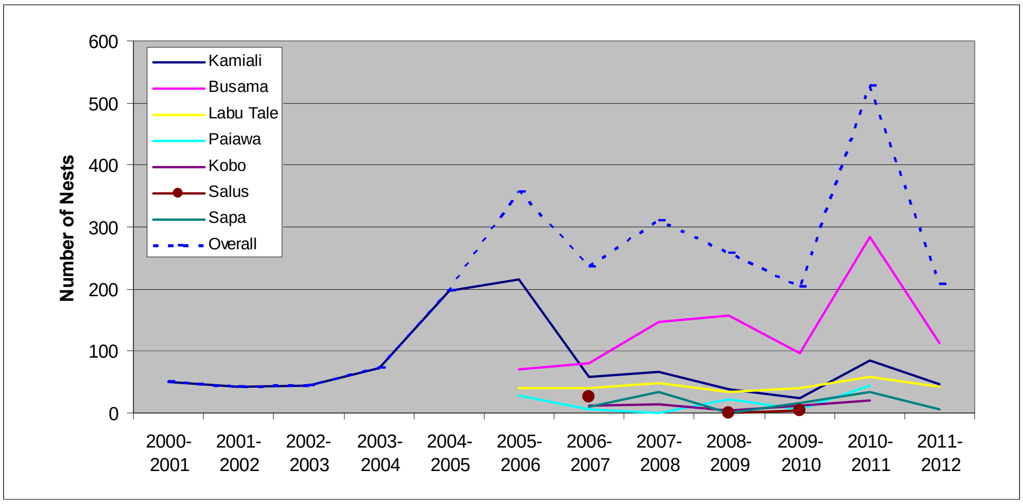


Figure 7-18. Nesting trends along the Huon coast of PNG. It is important to note that the increase at Kamiali in 2004 and 2005 was the result of an expansion of the monitored area, rather than an increase in nesting abundance. Image source: Pilcher 2013.

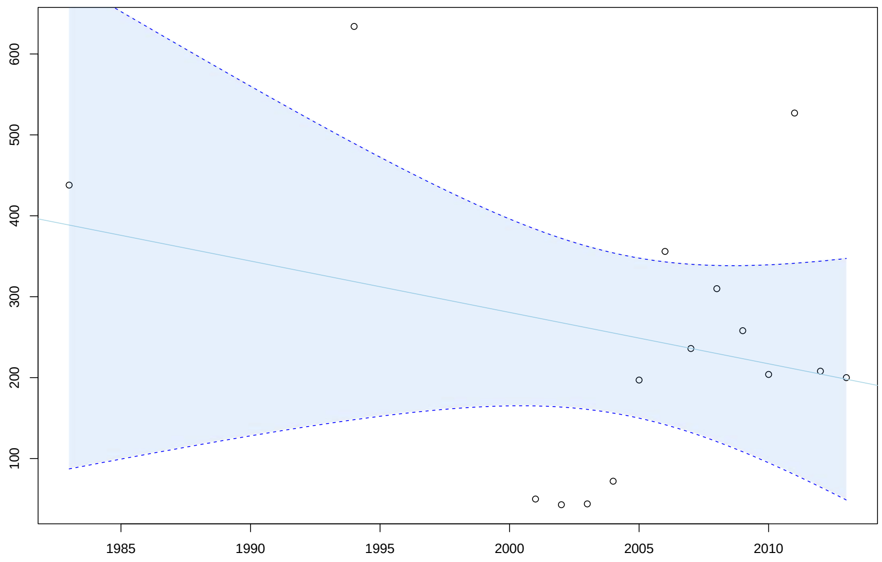


Figure 7-19. Declining trend in nest deposition along the Huon coast, Papua New Guinea. Shaded areas represent 95% confidence intervals. Image source: Pilcher et al. unpublished.

Solomon Islands: Leatherback turtle nesting in the Solomon Islands was documented on Santa Isabel, Choiseul, Rendova and Tetepare, but sporadic nesting was also reported at most other major islands (Vaughan 1981). Activity across all beaches varied from a few nests per season to over 20 nests in a single night, totalling a minimum of several hundred nests throughout the entire Solomon Island Archipelago per season (Vaughn 1981). Pita (1994) indicates an increase in nests at Litoghahira and Sasakolo from 50−150 turtles in the 1980s to nearly 1,000 in 1993; however, these estimates were derived by extrapolating directly from a two-week survey across the entire year, when nesting is primarily restricted to November through January. It is also likely that during this season there is a central peak during which the highest nesting is recorded and thus these estimates are unlikely to be realistic. This is supported by follow-up surveys in 1995 at Sasakolo, during which only 83 nests were recorded over the six-week peak period. It is likely several hundred nests are deposited throughout the entire Solomon Island Archipelago, potentially representing 30 to 50 annual nesters.

7.4.2 Leatherback turtle – East Pacific RMU

Eastern Pacific leatherbacks have declined by more than 97% since the 1980's (Santidrián-Tomillo et al. 2007, Laúd OPO Network 2020; Figure 7-20). According to the Laúd OPO Network (2020), three-quarters (75.4%) of total region-wide nesting activities between 2004 and 2015 occurred in México, while the remainder occurred in Costa Rica (22.2%) and Nicaragua (2.4%).

Costa Rica: The most recent assessment of nesting abundance in Costa Rica suggests nesting has declined significantly (Wallace et al. 2013), with a total of 184 ± 44.1 nests at Parque Nacional Marino Las Baulas (Santandrián-Tomillo et al. 2017). Additional beaches in Costa Rica with lower levels of nesting activity include Ostional National Wildlife Refuge; Caletas; and San Miguel, Guanacaste with a total of 62.8 ±7.0 nests between 2012 and 2017 (Santandrián-Tomillo et al. 2017). Following the continued declines in nesting, between 2011 and 2015, an estimated ~1,050 clutches per year were laid across all east Pacific rookeries (Laúd OPO Network 2020), of which ~230 were deposited in Costa Rica. This equates to approximately ~35 turtles per year.

Mexico: Annual nester abundances in 2010 were estimated to be 130 turtles at Mexiquillo, 103 at Tierra Colorada, 130 at Cahuitán, 24 at Cacahua and 130 at Barra de la Cruz, for a total nester abundance of 517 turtles (Wallace et al. 2013). As above, the numbers of nesters have continued to decline, and only ~800 were deposited in Mexico. This equates to approximately 125 turtles per year (Laúd OPO Network).

Nicaragua: Annual nester abundances in 2010 were estimated to be 34 turtles at Veracruz, 24 at Salamina, and 28 at Juan Venado, for a total nester abundance of 76 turtles. However, there have been substantial declines since that time, and based on data presented by Laúd OPO Network (2020) Nicaragua hosted only ~25 nests or some four turtles per year.

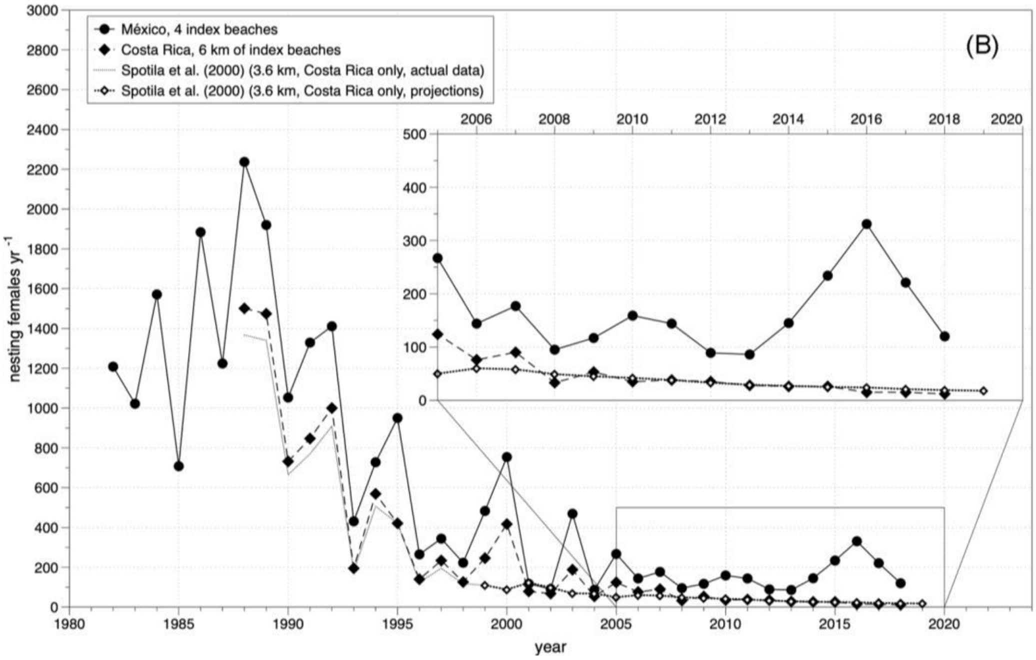


Figure 7-20. Population trends for eastern Pacific leatherbacks. Image source: Laúd OPO Network 2020.

7.5 Number of Nesting Turtles – Flatback Turtles

Flatback turtle nesting is exclusive to Australia. Foraging flatbacks have been encountered in neighbouring Papua New Guinea and Indonesia, but no nesting records for this species exist in those countries (Figure 7-21). A comprehensive summary of breeding biology of this species in Australia is presented by Limpus (2007e).

7.5.1 Flatback turtle – West Pacific RMU

Australia: Australia distinguishes turtles by management units (MUs) and of relevance to the flatback populations in the Pacific Ocean are the Eastern MU, with major rookeries at Peak and Wild Duck Islands, and minor rookeries at Mon Repos, a number of islands in central Queensland, and at Cape Cleveland and Cape Bowling Green. The second MU of interest is the Gulf of Carpentaria/Torres Strait MU, where the largest nesting sites for flatbacks include Crab Island, Deliverance Island and Kerr Island. Peak and Wild Duck Islands support the two largest nesting populations ineastern Australia (Limpus 2007e). At Peak Island two complete season census surveys resulted in ~250 nesting females per year, and Wild Duck Island has supported some 20 to 55 clutches (or seven to 30 females) per year (Limpus et al. 2002). At Curtis Island the annual number of nesters has fluctuated around 55 turtles per year, and along the Wonngarra coast the annual number of nesters is <10 per year (Limpus 2007e). At Avoid Island the number of annual nesters declined from ~75 per year to ~40 per year between 2013 and 2018. Total tagging census studies at Peak Island indicate that numbers of nesting females appear to have increased slightly, while Curtis Island and the Woongarra Coast have demonstrated no obvious trend in the size of the annual nesting population over three decades (Limpus 2007e).

In the Gulf of Carpentaria and Torres Strait, the largest flatback rookery 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 (Environment Australia 2003). 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). Limpus et al. (1989) concluded that Deliverance Island was at least as significant as the Peak Island rookery in central Queensland, the second largest nesting aggregation recorded for the species. This suggests that ~250 flatbacks may also nest on this island each year. Although slightly west and likely of little consequence to the Pacific Ocean population, Groom et al. (2017) calculated the number of flatbacks using Kakadu National Park to range between 97 and 183 turtles per year with no significant trend over 12 years of monitoring.

Bustard (2016) reported a rough estimate for the nesting population of 20,000 adult female flatbacks across all of Australia. Given this, and the summary data presented by Limpus (2009e), it is likely that the north and east beaches in Australia host 3,000 to 4,500 nesting flatback turtles each year.

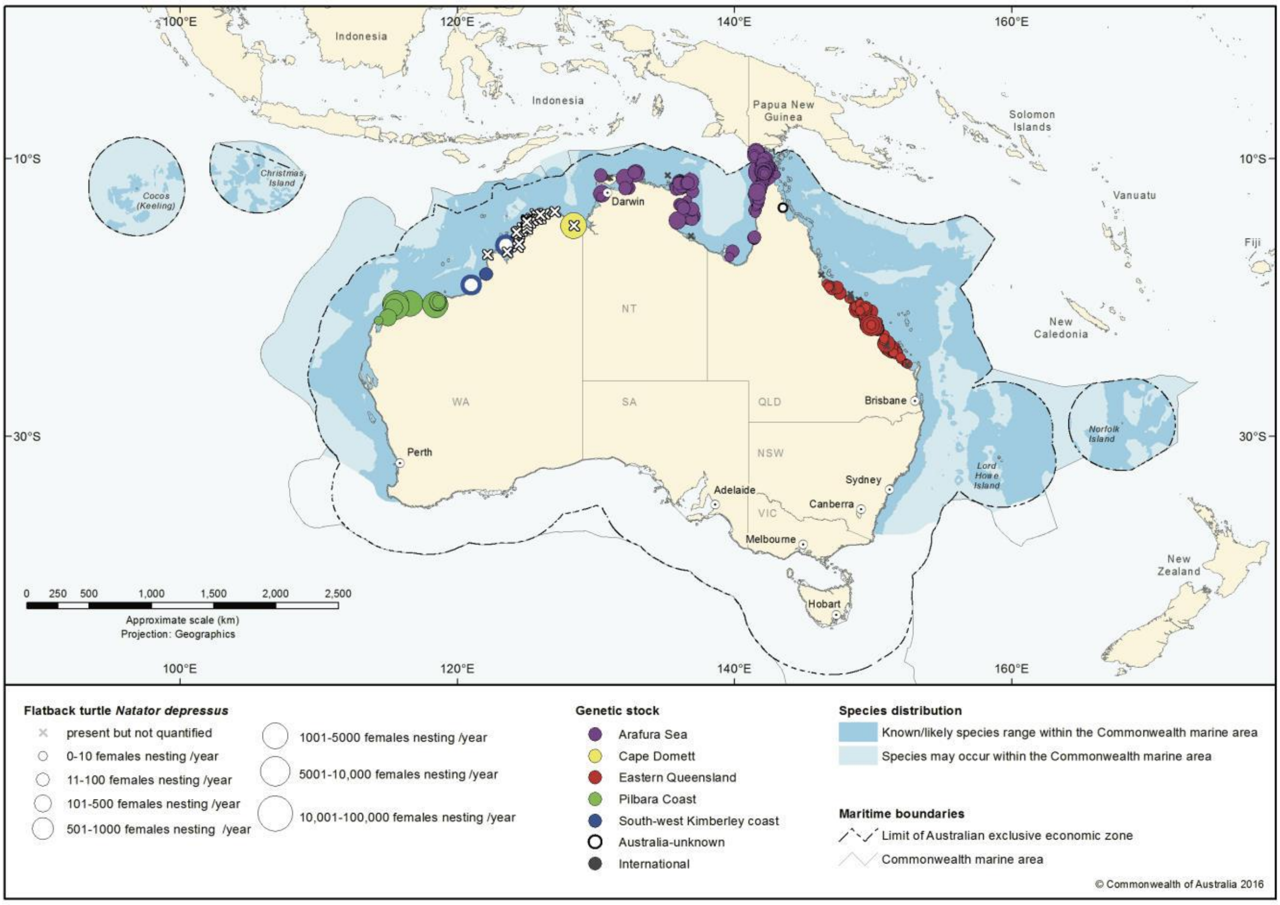


Figure 7-21. Flatback turtle (*Natator depressus*) nesting sites in Australia and surrounding regions. Image source: Commonwealth of Australia 2017.

7.6 Number of Nesting Turtles – Olive Ridley Turtles

Olive ridley turtle nesting is limited to a few countries in the western Pacific and there are few assessments for this species in the western Pacific. The largest nesting aggregations occur in the eastern Pacific, in Chile, Colombia, Costa Rica, Ecuador, El Salvador, Guatemala, Honduras, Mexico, Nicaragua and Panama (Figure 7-22). The most recent and comprehensive summaries of breeding biology of this species at a global level are presented by Plotkin (2007), the 2008 IUCN Red List assessment (Abreu-Grobois and Plotkin 2008), and the NOAA USFWS 5-Year review (Conant et al. 2014). The olive ridley (and its conspecific Kemp's ridley *Lepidochelys kempii*) have two nesting modes: solitary nesting and mass (or *arribada*) nesting, during which tens to hundreds of thousands of turtles nest over spans of just a few days. The beaches on the eastern Pacific coast include solitary and *arribada* nesting, with key *arribada* nesting beaches at Mismaloya, Ixtapilla and La Escobilla. Only solitary nesting occurs in the western Pacific.

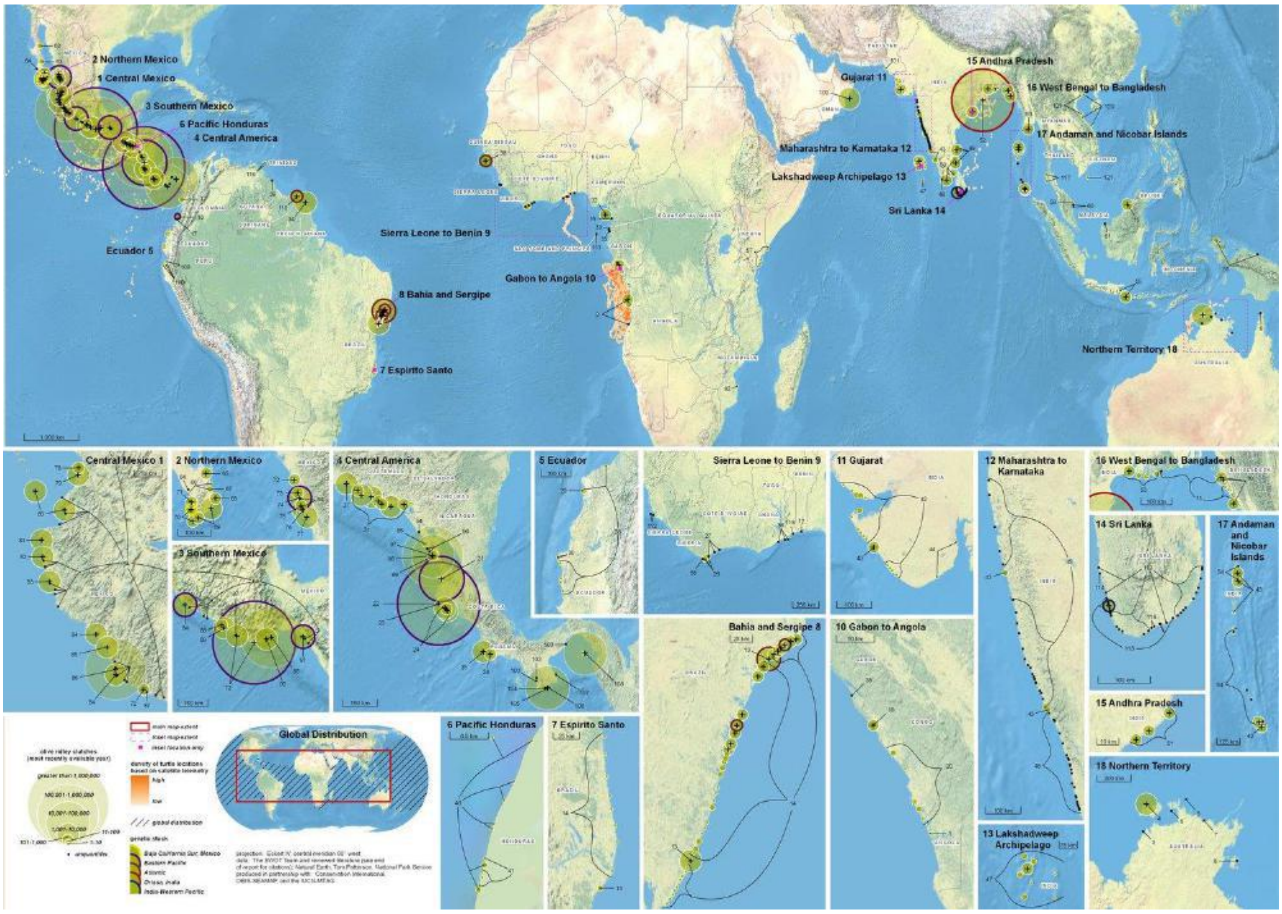


Figure 7-22. Global biogeography of olive ridley nesting sites. Image source: SWOT Report IV.

7.6.1 Olive ridley turtle – West Pacific RMU

Australia: There are no records of *L. olivacea* nesting from the east coast of Australia, but low-density nesting occurs along the northwestern coast of Cape York Peninsula between Weipa and Bamaga (Limpus and Roper 1977; Limpus et al. 1983b, Limpus 2007f). The balance of nesting in Australia occurs in the Northern Territory with minor nesting in Western Australia (Limpus 2007f). 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.

Federated States of Micronesia: Falanruw et al. 1975 report on the presence of olive ridleys, but no details are available for current status. It is likely, given the ongoing surveys that have documented green turtles in FSM, that the olive ridley is an extremely infrequent, if at all, nester in FSM.

Papua New Guinea: Spring (1982) reported olive ridley nesting in Turubu, East Sepik; Garu in New West Britain; and at Atailikilum in East New Britain. Due to a lack of systematic surveys, there is no current estimate of the number of solitary olive ridleys nesting annually in Papua New Guinea

Republic Of The Marshall Islands: A single olive ridley was found opportunistically on Arno Island in the Marshall Islands. It is unknown if the turtle intended to nest, and the authors suggested it may have been washed onto the beach by swells. There are no other records of olive ridleys in the Marshall Islands.

Philippines: Dispersed and infrequent nesting by olive ridleys occurs throughout much of the Philippines islands but currently remains unquantified. 2020 records indicate ~15 turtles (R. Araceli Salinas, pers. comm.).

Solomon Islands: McKeown (1977) reported that olive ridley turtles had been found in the Solomon Islands, but no accounts of nesting exist. Interestingly, two of the turtles sighted were a mating pair found near the capital Honiara. However, recent fieldwork conducted by a community monitor in Makira revealed olive ridley turtles nesting at Waihaoru beach, and one juvenile was caught at Wagina, Choiseul (Vuto et al. 2019). There is no current estimate of the number of olive ridleys nesting in the Solomon Islands.

7.6.2 Olive ridley turtle – East Pacific RMU

Colombia: Low-density nesting occurs, principally in the Playon de El Valle and Parque Snaguianga in the south (Amorocho et al. 1992). During 2003−2007, 25 olive ridley nests were documented on Parque Gorona, a small 1.2 km island in the south. Amorocho (1994) reported olive ridley nesting on Playa Larga but did not provide the numbers of turtles or nests. Martinez and Paez (2000) reported 112 olive ridley nests in 1998 on another beach, La Cuevita. There is no current estimate of the number of solitary olive ridleys nesting in Colombia, but based on the numbers of nests and localities identified, it is likely there are <200 annual nesters in Colombia each year.

Costa Rica: Although there is widespread, low-density olive ridley nesting, there is no current total estimate of the number of olive ridleys nesting in Costa Rica. There are a few non-arribada beaches where data have been collected (Conant et a. 2014): an average of 180 nests were documented in San Miguel from 1998 through 2004. At Playa Caletas, 71 olive ridley nests were documented during the 2002–2003 nesting season; there were an additional 226 unconfirmed events, most of which were believed to be olive ridleys. During the 1993−1994 nesting season on the Osa Peninsula, 3,155 olive ridley nests were recorded (Drake 1996). There is no current estimate of the number of solitary olive ridleys nesting in Costa Rica.

Ecuador: Olive ridleys have not been recorded to nest on Ecuadorian beaches. In 2004, a single nest was identified as an olive ridley nest based on an examination of a late-stage embryo (Alava et al. 2007).

El Salvador: There is low-density olive ridley nesting, but there is no current estimate available of the number of solitary olive ridleys nesting along the coast of El Salvador.

Guatemala: There is widespread, low-density olive ridley nesting in Guatemala. The most current estimate available indicates there were over two million olive ridley eggs laid on the coast of Guatemala in the late 1990s (Muccio 2000). Assuming the average clutch size is 100 eggs, then this represents approximately 20,000 nests. Current estimates of annual nesters are ~1,000 females (Conant et al. 2014).

Honduras: There is widespread, low-density olive ridley nesting on the shores of the Gulf of Fonseca. In Punta Raton, Lagueux (1989) reported 742 nests from July through December 1987. There is no current estimate of the number of solitary olive ridleys nesting along the coast of Honduras.

Mexico: An annual estimate of ~7,000 nesters is reported for the key solitary nesting beaches in Mexico (Abreu-Grobois and Plotkin 2008, Conant et al. 2014, Hart et al. 2014). These are distributed as follows: El Verde (~600); El naranjo (~50); Platanitor (~650), Cayutlán (~600); Maruata-Cololoa (~2,000), Puerto Arista (~350), Nuevo Vallarta (~2,500), San Crostóbal (~50) and El Suspiro (~100).

Nicaragua: In Nicaragua there is widespread, low-density olive ridley nesting. However, there is no current estimate of the number of solitary olive ridleys nesting along the coast of Nicaragua.

Panama: Widespread, low-density olive ridley nesting still occurs in Panama. There is no current reliable estimate of the number of olive ridleys nesting on beaches along the coast of Panama, and population trend data are unavailable. NMFS and USFWS 1998 reported that 10,000 solitary olive ridleys may nest annually throughout Panama.

Peru: In Peru, nesting is rare and only one or two nests have been recorded (Kelez et al. 2009).

7.6.3 Olive ridley turtle – East Pacific (arribadas) RMU

Costa Rica: Arribada nesting occurs at Ostional beach, where a community-based egg-harvest programme operates legally to generate important income for the community while promoting the protection of the ridley nesting assemblage. Arribadas ranged between ~3,500 and 475,000 egg-laying females between 2006 and 2010, with an average of ~ 120,000 nesters per year (Valverde et al. 2012). Arribada nesting also occurs at Nancite, where sizes of arribadas have declined by ~95% since the 1970s, attributed in part to low embryonic survival (Figure 7-23; Fonseca et al. 2009). Current arribadas host ~3,000 to ~10,000 turtles (Fonseca et al. 2009).

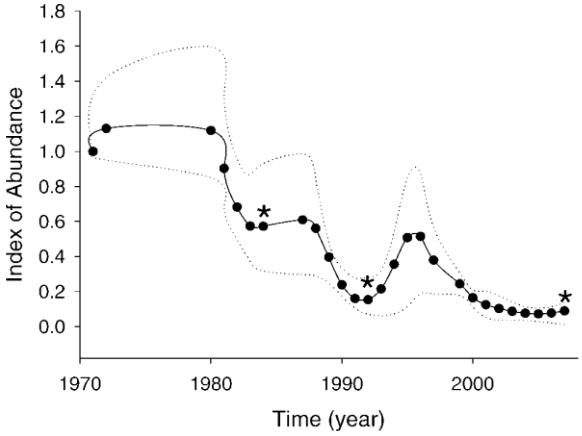


Figure 7-23. Modelled trend for arribada size based on the index of abundance for the Nancite rookery (1971–2007) using a GAM fitted to nesting female estimates. Image source: Fonseca et al. 2009.

Mexico: The current abundance of olive ridleys compared with former abundance at each of the large arribada beaches indicates the populations experienced steep declines due to over-exploitation (Abreu-Grobois and Plotkin 2008). Annual nesters at these beaches are as follows (calculated as number of nests divided by two – the average clutch frequency): Mismaloya: ~1,200 nesters; Tlacoyunque: 300 nesters; Ixtapilla: ~1,500-5,000 nesters; Chacahua: ~1,000 nesters; La Escobilla: ~500,000 nesters; and Moro Ayuta: ~5,000 to 50,000 nesters (Abreu-Grobois and Plotkin 2008). In the past there were much larger arribadas at Mismaloya, Tlacoyunquye, Chacahua and Moro Ayuta but these no longer occur. An example of the scale and magnitude of nesting during an arribada is presented in Figure 7-24.

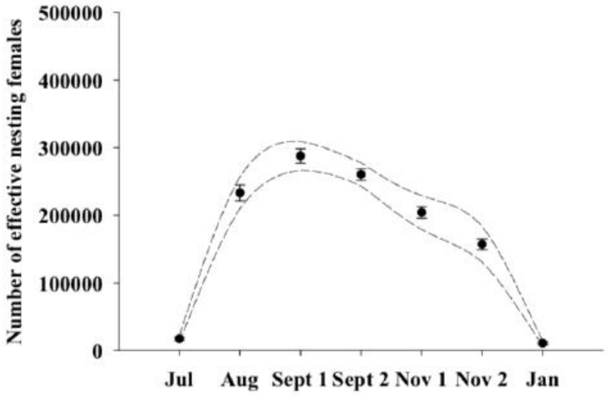


Figure 7-24. Arribada abundance at La Escobilla Beach, Mexico during the 2006 season. Data are plotted as number of effective nesting females, which is equivalent to number of nests laid. Image source: Valverde 2007.

Nicaragua: Two arribada beaches exist in Nicaragua: Playa Chacocente and Playa La Flor. Combined data from these two sites suggested a mean arribada size of 66,885 turtles from 1993 to 1999 (Hope 2002). More recently, during the 2008 and 2009 nesting seasons, 27,947 females were estimated to nest in nine arribadas at Playa Chacocente and 521,440 females in eight arribadas at Playa La Flor (Gago et al. 2012).

Panama: One arribada site is located at Isla Cańas and it is estimated that ~8500 to 9,000 turtles nest at this site each year (Abreu-Grobois and Plotkin 2008).

7.7 Clutch frequency

Clutch frequency refers to the number of clutches a female turtle deposits in a nesting season. This value is important in calculating the total annual reproductive output for a sea turtle species that is calculated by multiplying the total number of nesters × the average number of clutches (clutch frequency, this Section) × the average clutch size × the emergence success of hatchlings from a nest. Measuring clutch frequency requires season-long monitoring and mark-recapture of nesting females (so that one can know how many times each female lays eggs). Unfortunately, due to the logistical challenges and widely dispersed nature of many rookeries there are few programmes across a large range of the Pacific Ocean that have assessed clutch frequency. Additionally, estimating clutch frequency for olive ridley turtles is problematic because of the very nature of their nesting habits: during arribadas it is impractical to tag all turtles given the numbers, and therefore detecting renesting events is not possible. At many solitary nesting sites nesting is infrequent and sporadic, making detection of renesting events by individual turtles a challenge.

7.7.1 Green turtles

In the north central Pacific, Balazs (1980) reported a smaller clutch frequency (1.8 nests per season) than that recorded elsewhere for green turtles at French Frigate Shoals. However, a subsequent review indicated clutch frequencies of 4.96 (Balazs et al. 2015, Piacenza et al. 2016).

In the central Pacific, Summers et al. (2018) determined that clutch frequency in green turtles was 7.0 ± 1.3 nests per female in the Northern Mariana Islands.

At the Ogasawara Islands in the western Pacific, clutch frequency was reported as 4.1 clutches per season (Seminoff et a. 2015). Hirth (1997) indicated slightly lower clutch frequency of 3.9 at the Ogasawara Islands in Japan.

In the southwest Pacific, in Australia, Limpus (2007a) reported clutch frequencies of 5.06 at Heron Island. Clutch frequency in New Caledonia was 4.1 (Read et al. 2014).

In the eastern Pacific, Alvarado and Figueroa (1991) reported clutch frequency of 3.5 for Colola and Maruata in Mexico; and Marquez et al. (1981,1990) reported clutch frequencies of 4 at Michoacan in Mexico. Green (1994) reported clutch frequencies of ~ 3 nests per season on the Galapagos Islands. At Colola beach in Mexico, Alvarado-Diez et al. 2003) reported a mean observed frequency of 2.5 and a mean estimated clutch frequency of 3.1. Blanco et al. calculated mean estimated clutch frequency (ECF) to be 3.7 ± 1.8, but adjusted this to 5.1 ± 1.3 to include clutches recorded in the last ultrasound. This frequency is greater than previously described for east Pacific green turtles. Santandrián-Tomillo et al. (2014) indicate estimated clutch frequencies of 4.3 ± 2.3 at Playa Cabuyal in Costa Rica.

7.7.2 Hawksbill turtles

Clutch frequency in the Solomon Islands ranges from 3 to 5 clutches per season (Mortimer 2002). Clutch frequency in Australia was 2.4 nests on Milman Island and 3 nests in Campbell Island (Limpus 2007b). Clutch frequency in El Salvador was 2.1 nests / year (Gaos et al. 2017).

7.7.3 Loggerhead turtles

Clutch frequency at Mon Repos, Australia, was 3.41 nests/season (Limpus 2007c). In Japan the clutch frequency was 4.6 nests per year (Hatase et al. 2013). In Japan, Hatase and Omuta (2020) identified differing clutch frequencies based on foraging pattern: Annual mean (±SD) clutch frequencies for oceanic foragers ranged from 2.1 ± 1.3 (n=10) to 3.7 ± 1.1 (n=18), whereas those for neritic foragers ranged from 2.9±1.3 (n = 10) to 4.5 ± 1.1. Iwamoto et al. (1985) recorded clutch frequencies of 1.10 at Miyazaki, Japan, while Nishimura (1994) reported clutch frequencies of 2.06 at Yakushima.

7.7.4 Leatherback turtles

In the western Pacific, clutch frequency was estimated to reach up to four clutches per season in Australia (Limpus 2007d); and 2.2 nests per season in PNG (Kisokau 2005). Pilcher (2006) reported a similar but slightly higher 2.9 clutches per season in PNG. In Indonesia observed clutch frequencies ranged from 3−10 per season with a mean of 5.5 ± 1.6 (Tapilatu et al. 2013).

In the east Pacific, clutch frequency was 3.2−5.6 at Playa Grande (Reina et al. 2002) and 2.9−4.6 at Playa Langosta (Piedra et al. 2007), Costa Rica. Santandrián-Tomillo et al. (2006) also recorded clutch frequencies of 9.45 ± 1.63 for leatherbacks at Playa Grande. Price et al. (2006) reported an estimated clutch frequency of 7.87 ± 0.12) at Las Baulas in Costa Rica. In Mexico, the estimated clutch frequency was 5.5 (Sarti et al. 2007). At Las Baulas, Costa Rica, Reina et al. (2002) estimated clutch frequency (ECF) ranged from 4.3 ± 0.2 to 7.9 ± 0.3 clutches per female per nesting season. Sarti Martinez et al. (2007) reported an average estimated clutch frequency of 5.5 ± 1.9 for all Mexican Pacific beaches.

7.7.5 Flatback turtles

Along the Bundaberg coast in eastern Australia flatback clutch frequency was 2.84 nests per year (Limpus et al. 1983). Limpus et al. (1984) also describe a range for clutch frequency of one to four nests per female per season.

7.7.6 Olive ridley turtles

In general, individual olive ridleys may nest one, two, or three times per season but on average two clutches are produced annually (Pritchard and Plotkin 1995). Clutch frequency was reported to average 2.2 nests by Spotila (2004). Observed Clutch Frequency (OCF) at Playa Grande, Costa Rica, was lower at 1.13 clutches per female (Dornfelt et al. 2015).

Clutch frequency has not been determined in Australia due to lack of season-long studies on this species.

A summary of clutch frequencies by species is presented in Annex B.

7.8 Clutch size

Clutch size refers to the number of eggs (on average) deposited in each clutch per species per season. This value is important in calculating the total annual reproductive output for a sea turtle species that is calculated by multiplying the total number of nesters × the average number of clutches × the average clutch size (number of eggs, this Section) × the emergence success of hatchlings from a nest.

7.8.1 Green turtles

In the north central Pacific, Balazs (1980) reported clutch sizes of 104 eggs in the French Frigate Shoals. Niethammer et al (1997) determined that clutch frequency in green turtles in the French Frigate Shoals was slightly lower at 92.4 eggs.

In Tetiaroa (French Polynesia), clutch size was 76.5 ± 18.9 eggs (Touron et al. 2019), and White (2012) reported clutch sizes of 81−171 eggs in the Cook Islands.

In the western Pacific, Summers et al. (2018) determined that clutch size in green turtles was 93.5 ± 21.4 eggs in the Northern Mariana Islands, and Seminoff et al. (2015) reported clutch sizes of 102 eggs on the Ogasawara islands.

In Australia, Limpus (2009a) reported a clutch size of 112 ± 21.56 to 115.2 ± 27.88 eggs on Heron Island and 103.9 eggs on Raine Island. At Bramble Cay the clutch size was 111.1 in 1978 and 102.1 in 1979 (Limpus 2007a). Clutch size in the McCluer Islands in the Gulf of Carpentaria was 109 eggs. In the Solomon Islands clutch sizes averaged 84.6 to 97.3 (Vaughn 1981); in Papua New Guinea, Spring (1983) reported clutch sizes of 107.3 eggs; in New Caledonia, clutch size was 112 (Read et al. 2014).

In the eastern Pacific, clutch sizes ranged from 66 at Michoacan, Mexico (Marquez et al. 1981,1990) to 69.2 at Cololoa and Maruata (Alvarado and Figueroa 1991). Mean clutch size in the Galapagos was 71.2 ± 18.3 eggs (Zarate 2013). At Las Baulas, Costa Rica, Reina et al. (2002) reported mean clutch sizes of 64.7 ± 1.4 yolked and 38.5 ± 1.0 yolkless eggs. Santandrián-Tomillo et al. (2014) indicate estimated clutch sizes of 76.9 ± 18.2 eggs at Playa Cabuyal in Costa Rica.

7.8.2 Hawksbill turtles

In the south Pacific, Vaughn (1908) reported clutch sizes of 151 eggs in the Solomon Islands. Clutch size was 120.1 at the Groote Eylandt archipelago in Northern Australia in 2009−2010 (Hoenner et al. 2016); and was 121.7 on Milman Island, 131.8 on Campbell Island, and 139.3 to 142.2 on Crab Island (Limpus 2007b).

East Pacific RMU clutch sizes were 132.4 to 167.8 in El Salvador, 150.9 to 154.8 in Nicaragua, and 159.1 in Machalilla (Gaos et al. 2017).

7.8.3 Loggerhead turtles

Clutch size in Australia was reported as 127 eggs at Mon Repos and 124.4 eggs at Heron Island (Limpus 2007c). In Japan the average clutch size was 122 eggs per nest (Hatase et al. 2013).

7.8.4 Leatherback turtles

In the western Pacific, limited numbers of samples indicated a clutch size of 86.1 eggs in Queensland and 97.7 eggs in New South Wales, Australia (Limpus 2007d). In Indonesia, clutch size was 79.6 at Jamursba Medi and 76.2 at Warmon (Tapilatu and Tiwari 2007). In PNG, clutch size was 94.7 in 2006 (Pilcher 2006) and 79.3 in 2013 (Pilcher 2013); Hamman et al. (2006) report a clutch size of 88.2 at multiple PNG sites. Pita and Rovally (1996) reported an average of 88.2 eggs per clutch at Sasakolo, Solomon Islands, in 1995.

In the eastern Pacific, clutch size at Playa Grande, Costa Rica was 61.8 (Wallace et al. 2007); and 64.7 (Reina et al. (2002). At Playa Langosta clutch size was 64.5 (Piedra et al. 2007). Santandrián-Tomillo et al. (2006) also recorded clutch sizes of 62 ± 10 for leatherbacks at Playa Grande. In Mexico, clutch size was 66.0 at Jalisco (Castellanos-Michel et al. 2006) and 62.0 at Michoacan (Sarti et al. 2007). Price et al. (2006) reported a clutch size of 62.6 ± 1.28) eggs at Las Baulas in Costa Rica.

7.8.5 Flatback turtles

Along the Bundaberg coast in eastern Australia, flatback clutch size was 50.2 eggs, while at Peak Islands it was 53.4 and at Wild Duck Island it was 53.8 eggs (Limpus 2007e). At Crab Island, clutch size was 55.9 eggs, and at Deliverance Island, clutch size was 52 eggs (Limpus 2007e). On Deliverance Island Limpus et al. (1989) reported clutch sizes of 62, 55 and 43 eggs. Southerland and Southerland (2003) also assessed clutch size on Crab Island, at 56 ± 6.91 eggs.

7.8.6 Olive ridley turtles

In general, individual olive ridleys have clutch sizes of 100−110 eggs per clutch (Pritchard and Plotkin 1995). Mean clutch sizes at Nancite, in Costa Rica, were 99.5 in 1984 and 107.0 in 2007 (Fonseca et al. 2009). At Baja California, The number of eggs varied from 17 to 143, a mode of 102 (López-Castro et al. 2004). At Playa Grande, Costa Rica, mean clutch size was 87.5 ± 33.6 eggs (Dornfelt et al. 2015).

Clutch sizes in Australia were 109 eggs per clutch on Crab Island (Limpus et al. 1983).

7.9 Hatching/Emergence success

Hatching success refers to the number of hatchlings (on average) that successfully hatch and emerge from a nest on a beach. This number can vary depending on factors such as temperature, inundation, predation, insect infestation, etc. This value is important in calculating the total annual reproductive output for a sea turtle species that is (as shown above) calculated by multiplying the total number of nesters × the average number of clutches × the average clutch size × the emergence success of hatchlings from a nest (this Section). Emergence success is a more realistic figure than hatching success (those eggs that actually hatch down under the sand) as this accounts for the turtles that actually make it to the sand surface and contribute to reproductive output, but these figures are not always differentiated in the literature.

7.9.1 Green turtles

In the French Frigate Shoals, Niethammer et al. (1997) determined that emergence success in green turtles was 71.1%, with an overall hatching success of 81.1%. In the Northern Mariana Islands, Summers et al. (2018) reported a hatching success of 77.9 ± 27.0%, and an emergence success of 69.6 ± 30.3%. In Tetiaroa (French Polynesia), hatching success was ~94% (Touron et al. 2019), and on Moepelia hatching success rates ranged from 50% to 98.1% (Goutenegre et al. 2011).

In Australia, Limpus (2009a) reported a hatching success of 84.1% on Heron Island. At Bramble Cay the emergence success was 41% in 1977, and at Rainse Island it ranged from 73.9% in 1979 to 83.9% in 1983 (Limpus 2007a). In New Caledonia, emergence success was 90% (Read et al. 2014).

In the Galapagos islands, overall mean hatching and emergence success for 1039 nests was 46.0% ± 33.4 and 45.6% ± 33.4, respectively (Zarate 2013). Santandrián-Tomillo et al. (2014) indicate estimated hatching success of 89% ± 1.7 for clutches in the shade of trees and 75% ± 3.3 for clutches laid in the open at Playa Cabuyal in Costa Rica.

7.9.2 Hawksbill turtles

In Australia hatch success in Saunders islands was 90.9%, and on Milman Island it was 80.0%. However, due to predators on the mainland rookeries egg loss is ~90% (Limpus 2007b). Hatch success at the Groote Eylandt archipelago in Northern Australia was 85.2% but emergence success was only 69.3% (Hoenner et al. 2016).

Hawksbill hatch success in the east Pacific was 52% in Mexico, 52.5% to 72.3% in El Salvador, 59.8% to 60% in Nicaragua, and 59.7% in Ecuador (Gaos et al. 2017).

7.9.3 Loggerhead turtles

Hatching success was 81.9% in 1978 and 80.4% in 1979 at Mon Repos in the absence of predators (Limpus 2007c). However, predation has been varied across years and in the past was a major impact to overall hatchling production (Limpus 2007c). Wood et al. (2014) found loggerhead hatching success to be 80.2 ± 1.4 and emergence success of 78.2 ± 1.4 at Mon Repos. On Heron Island hatching success was 60% to 66.7% between 1987 and 1989 (Limpus 2007c).

Hatching success at Minabe beach in Japan varied from 40% to 96% (mean=74.9%, SD=15.8, n=18; Matsuzawa et al. 2002). At the Minabe-Senri beach were 24% (1996), 50% (1997), 53% (1998), 48% (1999), 62% (2000), 41% (2001), 34% (2002) (Matsuzawa unpublished data). Moriya et al. 2012 reported a hatching success of 62.4%.

7.9.4 Leatherback turtles

In west Papua, Indonesia, hatching success of leatherback turtle nests ranged from 25.5% at Jamursba Medi and 47.1% at Warmon in undisturbed nests (Tapilau and Tiwari 2007), but noted that hatching success could be as low as 9.3% in Wembrak to a high of 44.7% at Batu Rumah, indicating that nesting success is varied and dependent on factors such as low sand temperature, pig predation and nest inundation (Tapilatu and Tiwari 2007). In Papua New Guinea, Pilcher (2013) reported hatching success of 37% to 87% and emergence success rates of 31% to 82% across Labu Tale, Busama and Kamiali. Pilcher (2013) calculated an overall emergence success of protected nests for all sites at 49%.

In the eastern Pacific, emergence success was 41.0% at Playa Grande (Wallace et a. 2007) to 44.0% (Santandrián-Tomillo et al. 2009); and hatching success ranged from 47.9% to 51.4% at Playa Langosta (Piedra et al. 2007), both in Costa Rica. Santandrián-Tomillo et al. (2006) also reported an emergence success of 38% ± 27 for leatherbacks at Playa Grande. In Mexico, Sarti et al. have reported an emergence success of 66.4%.

7.9.5 Flatback turtles

Limited sampling in Australia suggests hatching success in Queensland is low (15.3%) and in New South Wales it was higher at 60.5% (Limpus 2007d). The low success in Queensland was attributed to unnaturally low ambient temperatures during embryonic development (Limpus 2007d). The seasonal hatching success for flatback turtles on Bare Sand Island ranged from 48% to 84% (Guinea 2015).

Hatching success of flatback turtles from Australian rookeries were as follows: Mon Repos: 70.3%−81.8%; Curtis Island: 83.4%; Peak Island: 74.6%, and Wild Duck Island: 88.7% (Limpus 2007e). In the Gulf of Carpentaria/Torres Strait, hatching success was 77.9%−92.4% on Crab Island, and 83.4% on Deliverance Island (Limpus 2007e). On Deliverance Island Limpus et al. (1989) reported overall emergence success ranging from 72% to 100%. Southerland and Southerland (2003) also assessed hatching and emergence success on Crab Island, at 95% ± 5.05 and 93.4% ± 8.21% respectively.

7.9.6 Olive ridley turtles

Hatching success during arribadas is typically low, as successive waves of turtles disinter previously-deposited clutches. At Nancite, in Costa Rica, emergence success was 17.72% and 26.75% during 2007 (Fonseca et al. 2009). At Playa Grande, Costa Rica, mean hatching success (78.5% ± 23.4% SD) was higher than at nearby arribada beaches (Dornfelt et al. 2015). At Las Barracas, Baja California Sur, nest success was 73.73% (López-Castro et al. 2004). In Honduras, mean clutch hatching success differed significantly between beach site and hatchery nests - beach: 83.22% ± 4.04 SE; hatchery: 24.08% ± 6.00 SE, $t_{10} = 6.818$ (Royo 2015).

Hatching success of olive ridley turtles in Australia has only been quantified in the Northern Territory, but it is likely that similar survivorship rates are experienced in Queensland: 84.4% at the McCluer Islands (Limpus and Preece 1992) and 79.6% at Bare Sand Island (Whiting 1997).

7.10 Survival rates

7.10.1 Green turtles

Limpus (2009a) reported hatchling survival in the nest at 84.1% on Heron Island, but hatchling survivorship was reported to be as low as 40% in nearshore waters (Gyuris 1994).

Seminoff et al. (2015) report juvenile survival rates of 58% and adult survival rates of 85%−97% at Michoacán, Mexico.

7.10.2 Hawksbill turtles

Bell et al. (2012) modelled the decline of hawksbills on Milman Island and determined that annual survival rate was high (0.972, 95% CI = 0.965 to 0.977).

7.10.3 Loggerhead turtles

Limpus (2009c) reported adult survivorship at 87.5% and 85.9% for all age classes, corrected to 91.8% to correct for possible transients (Chaloupka and Limpus, 2002). Conant et al. (2009) also provide high estimates of survival for first years and juveniles in the southwest Pacific RMU (Figure 7.25).

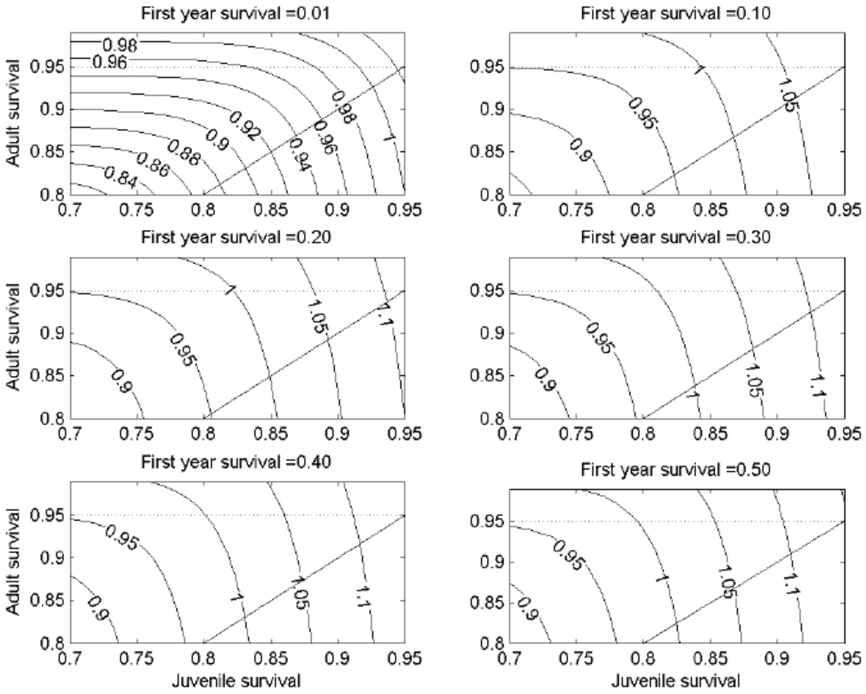


Figure 7-25. Relationships among adult survival rate, juvenile survival rate, first year survival rate, and the dominant eigenvalue for the Southwest Pacific RMU. Contour lines indicate the dominant eigenvalues, whereas the diagonal lines indicate equal annual survival rates between juveniles and adults. Dashed horizontal lines indicate the assumed adult survival rate for the base matrix model. Mean age at first reproduction was assumed at 30 and its standard deviation 5. Image source: Conant et al. 2009.

In Japan, NMFS and USFWS (2020) indicate survival rates of loggerheads as follows: hatchlings 0.400; oceanic juveniles: 0.858; neritic juveniles: 0.928; and adults: 0.950. Sasso and Epperly (2007) calculated an annual survival rate of 0.814, 95% CI = 0.557 to 0.939 for small immature loggerheads that had been lightly hooked (10) or dip netted (7) and found no difference in survival rates amongst groups.

7.10.4 Leatherback turtles

Laúd OPO provide robust estimates of survival for different age classes of east Pacific leatherbacks (Table 7-1).

Table 7-1. Mean survival estimates for E Pacific leatherback. First year and juvenile survival and fecundity, survival to the water and sex ratio were based on published estimates, while estimates for transient probability, survival probability and breeding propensity probabilities come from models with constant values to derive the variance-covariance matrix. Table source: Laúd OPO 2020.

Parameter	Costa Rica	México	Source
First year survival, S1	0.063	0.063	ref. [32]
Juvenile survival, S2	0.500	0.500	This study
Subadult and adult survival, S3	0.788	0.705	ref. [57]; This study
Breeding probability the year after breeding, B1	0.005	0.003	This study
Breeding probability 2 years after breeding, B2	0.272	0.169	This study
Breeding probability 3 years after breeding, B3	0.540	0.379	This study
Breeding probability ≥4 years after breeding, B4	0.681	0.550	This study
Probability of being transient	0.131	0.279	This study
Fecundity (eggs/female/year)	403	390	This study
Current egg harvest level	1%	4.2%	This study
Survival to water (from egg)	0.31	0.47	ref. [37]; This study
Sex ratio	0.840	0.840	ref. [19], based on the TSD-curved developed by ref. [36]

7.10.5 Flatback turtles

Survivorship from hatchling emergence to maturity has been estimated to be <0.0026 (Parmenter and Limpus 1995). Annual survivorship of adult females on the nesting beaches is >0.99 (unpublished data, EPA Queensland Turtle Conservation Project; Limpus 2007e). Groom et al. (2017) calculated adult annual survival as 0.97.

7.10.6 Olive ridley turtles

Data are lacking on post-hatchling and other life stage survival rates (Abreu-Grobois and Plotkin 2008). Presumably, similar to other sea turtles, olive ridley turtles experience high mortality in their early life stages (Abreu-Grobois and Plotkin 2008).

8.0 Threats - negative side of the risk assessment

Sea turtles suffer mortality through a wide range of anthropogenic and natural impacts. These include by-catch in fisheries, loss or alteration of nesting habitat and incubating clutches of eggs through erosion and climate change, boat strikes, predation, direct capture and consumption by coastal communities, and more. Different life stages are impacted at different rates, and the impacts to each life stage vary (Crouse et al. 1987). For instance, the loss of a few eggs is negligible compared to the loss of a reproductive adult female turtle. Because of this, it is necessary to understand not just the number of turtles taken out of the conservation equation, but also the life stage. Knowledge like this can help in the design of mitigation measures. For instance, Santandrián-Tomillo et al. (2008) demonstrated via a modelling exercise that different levels of egg collection and different levels of adult mortality in fisheries led to different population scenarios, and concluded that it would actually take both protection of eggs and adults at sea to recover the east Pacific leatherback population. Where possible in the coming sections, these levels of mortality are broken down by life stage, however in many cases, reports do not indicate turtle sizes from which an assumption of life stage could be made.

8.1 By-catch

By-catch refers to the unintended catch of sea turtles during fishing operations, be it hooked in the mouth or on some part of the body, or entrained in gillnets or trawl nets. Given sea turtles are reptiles and need to breathe air, and that fishing sets are typically of long duration, a large proportion of by-caught turtles drown. Not all interactions result in death however, and in some responsible commercial fisheries, there are handling protocols for disentangling turtles and de-hooking turtles so that they may survive. However, in many artisanal fisheries this is rarely practiced, and indeed, turtles caught unintentionally in nets are often retained as food. Fisheries are divided into the larger commercial (large boat) long-line, trawl and purse seine fisheries, while the smaller (in scale) coastal, local community fisheries are deemed artisanal fisheries. Small-scale fisheries have the potential to severely deplete sea turtle stocks in coastal waters (Lewison et al. 2004, Peckham et al. 2007), while the sheer scale of large industrial fishing, with millions of hooks deployed annually, can also impact sea turtles on the high seas. Estimating total mortality from fisheries by catch is problematic as 1) there is generally insufficient observer coverage and by-catch reports can not be extrapolated across entire fisheries; 2) not all turtles die after being caught in fishing operations; 3) the widely dispersed fishing operations with differing reporting requirements means getting the 'big picture' view of total impacts of all fisheries is problematic (although the Pacific Community – SPC, the Western and Central Pacific Fisheries Commission, the Western Pacific Regional Fisheries Management Council and NOAA Fisheries do a remarkable job with these limited data sets); and 5) illegal and unreported fishing effort is just that – unreported, and by-catch in these fisheries is unquantified.

Several proxy methods have been trialled for determining at sea mortality. For example, Hays et al. (2003) used satellite telemetry data to arrive at a (acknowledgedly tentative) mortality rate of 0.31, after recording six instances where the satellite telemetry data suggested a turtle had been captured, following a review of 5,923 tracking days (n = 50 turtles). Direct observations in fishing villages indicated that in 3 of the 6 cases, the turtles were dead. Similarly, Sasso and Epperly (2007) estimated post-release annual survival rate of loggerheads hooked in longlines using satellite telemetry as 0.814, 95% CI = 0.557 to 0.939. However, Chaloupka et al. (2004) noted that satellite data were not completely reliable for this sort of process given the need to determine end results inferentially. Given this, the best estimates at present come from observer programmes.

8.1.2 Commercial

North Pacific

Loggerhead turtles in the north Pacific overlap with the longline fisheries based out of Hawaii (Polovina et al. 2000) and American Samoa. The Hawaii fishery, with 145 active vessels, operates two distinct fisheries: a shallow-set fishery for swordfish, and a deep-set fishery to maximise catches of bigeye tuna. Catches by the Hawaii fleet also include yellowfin tuna, mahimahi, wahoo, blue and striped marlins. In 2017 bigeye tuna accounted for ~68% of the tunas and ~46% of all pelagic catch. Swordfish comprised ~51% of the billfish landings and 9% of the total catch (WPRFMC 2017). The Hawaii fishery does not freeze its catch, which is sold to the fresh fish and sashimi markets in Hawaii, Japan and the US mainland. The American Samoa longline fleet of ~ 40 vessels fishes almost exclusively for albacore, which is landed at the cannery in American Samoa. Pelagic landings consisted primarily of four tuna species: albacore, yellowfin, bigeye and skipjack. The small vessels average 350 hooks per set, while vessels over 50 ft. set 1700 to 2000 hooks per set (WPRFMC 2017).

These fisheries have stringent management measures (WPRFMC 2017) including caps for take of both loggerhead and leatherback sea turtles. Protected species interactions in the Hawaii longline fishery have been monitored through mandatory observer coverage since 1994. Observer coverage was between 3% and 5% from 1994 to 1999 and increased to 10% in 2000. Since 2004, the shallow-set component of the Hawaii longline fishery has had 100% observer coverage. During 1994–2000, the Hawaiian longline fishery caused the loss of an estimated 64–106 loggerheads annually (from a catch of hundreds of loggerheads annually) but observer coverage was low. However, since 2004 there has been full observer coverage for this fishery, and all sea turtle interactions have been documented (Table 8-1).

Nearly all sea turtles observed in the Hawaii shallow-set longline fishery from 2004 to 2017 were released alive. One unidentified hard shell in 2013 was classified by NMFS as a loggerhead and was counted towards the annual shallow-set interaction limit for loggerheads. The highest interaction rates involved both leatherback and loggerhead turtles (average takes/1,000 hooks = 0.0061 and 0.0094, respectively), whereas interactions with greens, olive ridleys and unidentified hard shell turtles were much less frequent (0.0005, 0.0005, and 0.0003 respectively; WPRFMC 2017).

Management measures in the Hawaii shallow-set longline fishery have been effective in reducing the number of sea turtle interactions (WPRFMC 2017). The introduction of sea turtle by-catch-reduction measures in 2004, such as switching from J-hooks to circle hooks, and from squid bait to mackerel bait, resulted in an 89% decrease in sea turtle interactions in 2004−2006 compared to interactions observed in 1994−2002 (Gilman et al. 2007). The rate of deeply hooked sea turtles, which is thought to result in higher mortality levels, also declined after those measures were implemented (Gilman et al. 2007).

Table 8-1. Observed takes and takes per fishing effort (1,000 hooks) for sea turtles in the Hawaii shallow-set longline fishery, 2004–2017. Table Source: WPRFMC 2017.

Year	Observer Coverage (%)	Sets	Hooks	Green		Leatherback		Loggerhead		Olive ridley		Unidentified hard shell	
				Takes	Takes/ 1,000 hooks	Takes	Takes/ 1,000 hooks	Takes	Takes/ 1,000 hooks	Takes	Takes/ 1,000 hooks	Takes	Takes/ 1,000 hooks
2004	100	88	76,750	0	0.000	1	0.013	1	0.013	0	0.000	0	0.000
2005	100	1,604	1,328,806	0	0.000	8	0.006	10	0.008	0	0.000	0	0.000
2006	100	939	745,125	0	0.000	2	0.003	17[b]	0.023	0	0.000	2[c]	0.003
2007[d]	100	1,496	1,292,036	0	0.000	5	0.004	15	0.012	1	0.001	0	0.000
2008	100	1,487	1,350,127	1	0.001	2	0.001	0	0.000	2	0.001	0	0.000
2009	100	1,833	1,767,128	1	0.001	9	0.005	3	0.002	0	0.000	0	0.000
2010	100	1,879	1,828,529	0	0.000	7	0.004	5	0.003	0	0.000	0	0.000
2011	100	1,579	1,611,395	4	0.002	17	0.011	14	0.009	0	0.000	0	0.000
2012	100	1,307	1,418,843	0	0.000	7[e]	0.005	5	0.004	0	0.000	0	0.000
2013	100	912	1,000,084	0	0.000	7	0.007	5[f]	0.005	0	0.000	1[g]	0.001
2014	100	1,349	1,509,727	1	0.001	19	0.013	13	0.009	1	0.001	1	0.001
2015	100	1,178	1,286,628	0	0.000	6	0.005	15	0.012	1	0.001	0	0.000
2016	100	778	849,681	0	0.000	5	0.006	16	0.019	0	0.000	0	0.000
2017	100	973	1,051,426	2	0.002	4	0.004	16	0.015	4	0.004	0	0.000

In the Hawaii deep-set longline fishery, by-catch monitoring operates under a rolling 3-year period to track incidental take. In the 2014 3-year biological opinion the US National Marine Fisheries Service (NMFS) estimated total interactions with leatherbacks as 70 turtles with 27 mortalities (WPRFMC 2017). In the 2017 3-year biological opinion, by-catch rates were estimated as follows: green turtles: 54(49); loggerhead turtles: 18(13); and olive ridley turtles 183(174). Take, mortality and mortality/1000 hooks for the Hawaii deep-set longline fishery are presented in Table 8-2.

Small pelagic loggerheads are also incidentally captured in the large-mesh high-seas driftnet fisheries of Japan and Taiwan that operate in the north Pacific. In 1990 the mortality of loggerheads in this fishery was estimated at 1,380 (Weatherall et al. 1993), but there was substantial uncertainty around the estimate. Given the wide distribution range for these turtles, Polovina et al. (2003) suggested there was little that could be done to constrain fisheries in such a way as to not interact with these turtles. Current restrictions to the US swordfish component of this longline fishery mean the estimated mortality dropped to ~8 annually (Limpus (2007c). An estimated total of 30,000–75,000 loggerheads may have been captured by longline fisheries in the north and south Pacific Ocean in 2000 (Lewison et al. 2004), but these authors cautioned that data were insufficient to estimate the total loggerhead by-catch for the Indian Ocean when considering the fisheries of all flag countries, along with illegal and unreported fishing efforts.

In Japan, Ishihara (2009) reported 121 by-catch incidences reported by fishermen in pound nets (72%), gill nets (11%), trawl nets (8%), encircling nets (3%), and other fisheries (6%) between October 2006 and September 2007. It is presumed the vast majority of these turtles were loggerheads but this was not reported.

Table 8-2. Observed takes (#), mortalities (M), takes per fishing effort (1,000 hooks) for sea turtles in the Hawaii deep-set longline fishery, 2002–2017. Table adapted from: WPRFMC 2017.

Year	Obs. Cov. (%)	Sets	Hooks	Observed takes									
				Green		Leatherback		Loggerhead		Olive ridley		Unidentified	
				#(M)	#/1000 hooks	# (M)	#/1000 hooks	# (M)	#/1,000 hooks	# (M)	#/1,000 hooks	# (M)	#/1000 hooks
2002	24.6	3,523	6,786,303	1(1)	0.0001	2	0.0003	4(1)	0.0006	7(7)	0.001	0	0
2003	22.2	3,204	6,442,221	0	0	1(1)	0.0002	0	0	3(3)	0.0005	0	0
2004	24.6	3,958	7,900,681	1(1)	0.0001	3	0.0004	0	0	13(13)	0.0016	0	0
2005	26.1	4,602	9,360,671	0	0	1	0.0001	0	0	4(4)	0.0004	0	0
2006	21.2	3,605	7,540,286	2(2)	0.0003	12(2)	0.0003	0	0	11(10)	0.0015	0	0
2007	20.1	3,506	7,620,083	0	0	2	0.0003	1(1)	0.0001	7(7)	0.0009	0	0
2008	21.7	3,915	8,775,951	0	0	1	0.0001	0	0	3(3)	0.0003	0	0
2009	20.6	3,520	7,877,861	0	0	1(1)	0.0001	0	0	4(4)	0.0005	0	0
2010	21.1	3,580	8,184,127	1(1)	0.0001	1(1)	0.0001	1(1)	0.0001	4(3)	0.0005	0	0
2011	20.3	3,540	8,260,092	1(1)	0.0001	3	0.0004	0	0	7(6)	0.0008	0	0
2012	20.4	3,659	8,768,728	0	0	1(1)	0.0001	0	0	6(6)	0.0007	0	0
2013	20.4	3,830	9,278,133	1(1)	0.0001	3	0.0003	2(2)	0.0002	9(9)	0.001	0	0
2014	20.8	3,831	9,608,244	3(3)	0.0003	7(2)	0.0007	0	0	8(7)	0.0008	0	0
2015	20.6	3,728	9,393,234	1(1)	0.0001	4(2)	0.0004	2(2)	0.0002	13(12)	0.0014	0	0
2016	20.1	3,880	9,872,439	1(1)	0.0001	3(1)	0.0003	2(1)	0.0002	31(28)	0.0031	1(1)	0.0001
2017	20.4	3,832	10,148,195	3(1)	0.0003	0	0	3	0.0003	26(23)	0.0026	0	0

Western and Central Pacific – Tuna Fishery

The tuna fishery in the western and central Pacific Ocean includes large-scale, industrial purse seine, longline, and pole-and-line operations in both the exclusive economic zones of Pacific states and on the high seas. The main species targeted by these fisheries are skipjack tuna (*Katsuwonus pelamis*), yellowfin tuna (*Thunnus albacares*), bigeye tuna (*T. obesus*) and albacore tuna (*T. alalunga*).

The development and implementation of stock-wide management measures for WCPO tuna fisheries is the responsibility of the Western and Central Pacific Fisheries Commission (WCPFC). Commercial fishing in the western and central Pacific occurs within a Convention Area (WCPFC-CA; Figure 8-1) and catches and by-catch data within this area are reported by the Pacific Community (SPC). Annual catches of the four main tuna species in the WCPFC-CA have increased continuously since the beginning of commercial exploitation in the early 1950s (Figure 8-2), with the expansion in total catch over the past 30 years primarily due to the expansion of purse seine fishing. Today the catches landed in this region are estimated to be worth ~USD 4–5 billion (Allain et al. 2016, Harley et al. 2015).

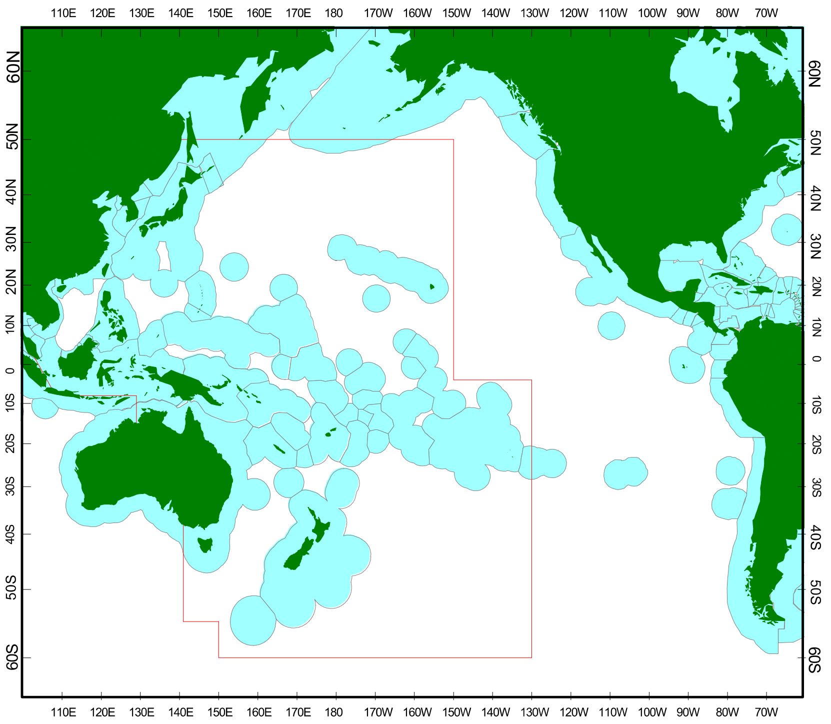


Figure 8-1. Pacific Ocean and the boundaries (red) of the WCPFC statistical area. Image source: Hampton 2010.

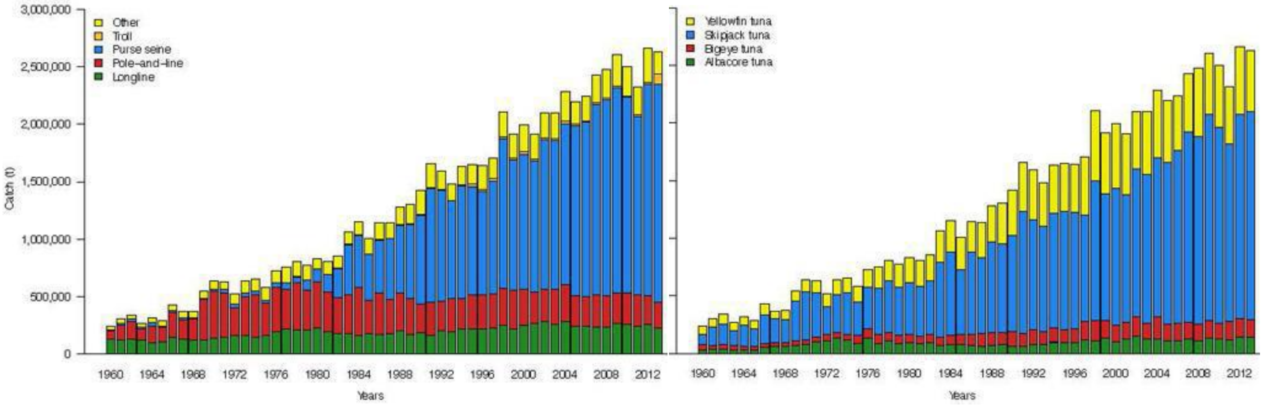


Figure 8-2. Catch (in tonnes) by gear (left) and species (right) for the Western and Central Pacific region, 1960-2013. Image source: Harley et al. 2015.

The purse seine fishing consists of encircling tuna schools with a large net (1,500−2,000 m long and 150−250 m depth), which is closed at the bottom before being hauled on board. The purse seine fishery is primarily a skipjack fishery, generally accounting for 65%−77% of the purse seine catch, and bigeye catches account for only a small proportion (20%−30%; Williams and Terawasi 2014). The majority of the historic WCPFC purse seine catch has come from the industrial fleets of the four main distant- water fishing nations: Japan (41), Korea (27), Chinese Taipei (34) and USA (40 vessels) in 2013 (Williams and Terawasi 2014). The geographical distribution of the purse seine fishery is concentrated in the equatorial band, with the highest catches between 5°N and 10°S (Figure 8-3).

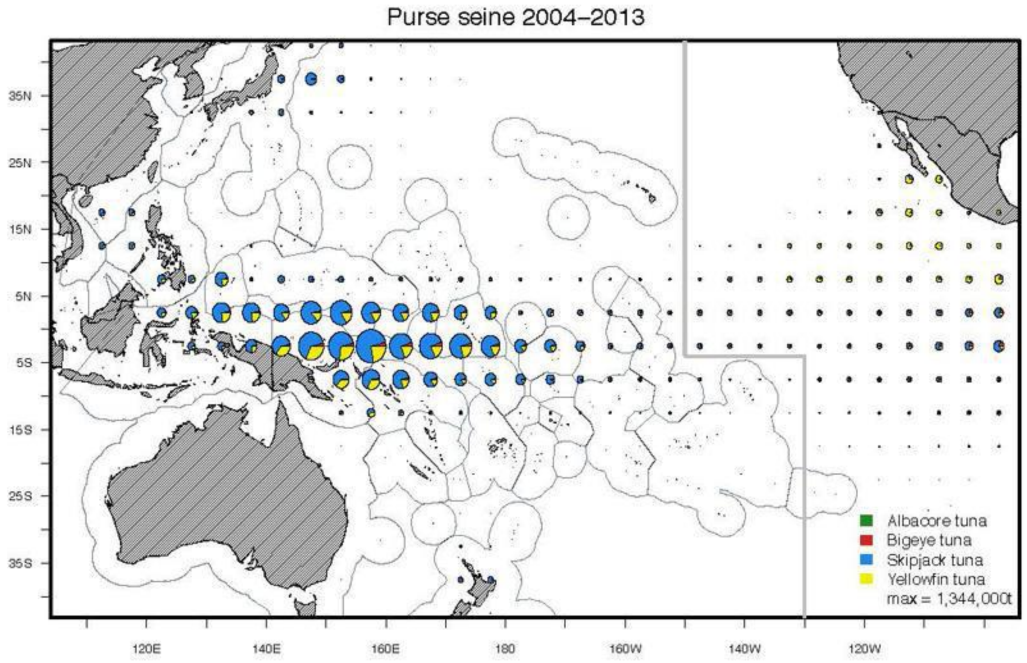


Figure 8-3. Catch composition and spatial distribution from purse seine vessels operating in the WCPFC. The size of the largest pie corresponds to the maximum value of 1,344,000 t. Image source: Harley et al. 2015.

Longline fishing consists of horizontally deploying a mainline, maintained on the surface with floats, and attaching to this mainline, vertical branchlines ending with baited hooks. The mainline can be as long as 100 km with as many as 3,000 hooks located between the subsurface and 100 m to 400 m depth. This technique typically catches larger tuna. In recent years albacore has been the main target, accounting for 44% of catch, while yellowfin and bigeye each accounted for 28% (Williams and Terawasi 2014). Catches of sea turtles (five different species) have been observed at a low encounter rate in the equatorial longline fisheries (Allain et al. 2016). The total number of vessels involved in the longline fishery has generally fluctuated between 3,000 and 6,000 over the past 30 years.

The longline fishery involves two main types of operations: large (typically > 250 gross register tonnage – GRT) distant-water freezer vessels that undertake long voyages (many months) and operate over large areas. These vessels may target yellowfin and bigeye tuna for the frozen sashimi market, albacore tuna for canneries, or recently, swordfish. The second class encompasses small (typically < 100 GRT) vessels which are usually domestically-based, undertaking trips of less than one month, with ice or chill capacity, serving fresh local or air-freight sashimi markets or albacore canneries (Williams and Terawasi 2014). Unlike the narrow range of operation in purse seine vessels, the geographical distribution of the longline industry is more widespread across the Pacific (Figure 8-4). Peatman et al. (2018b) split the WCPFC-CA into three regions to allow for more practical and spatially-relevant summaries of estimated catches: north temperate, ≥ 10°N; tropical ≥ 10°S and < 10°N; and south temperate < 10°S. Trends in effort (in thousands of hooks) in the deep and shallow-set fisheries across the three distinct zones of the WCPFC-CA are presented in Table 8-3. For each fish caught on a longline, the observers record the position of the hook (between two floats) where the fish was caught. A hook close to a float will be shallow while a hook far away from a float will be deep. This information provides insights in the interaction between the fish and the gear and can be used to set hooks at chosen depths to mitigate sea turtle by-catch.

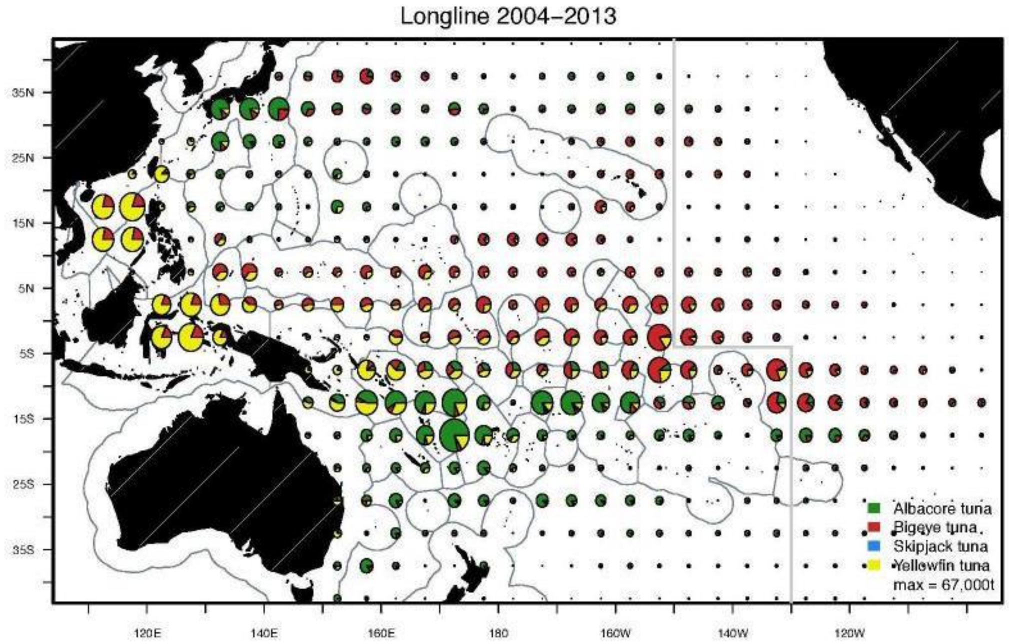


Figure 8-4. Catch composition and spatial distribution from longline vessels operating in the WCPFC. Image source: Harley et al. 2015.

The third key fishery is the pole-and-line industry, which consists of catching baited schools of tuna one by one, with long poles with a short line, with a feathered jig on a barbless hook. This technique catches small to medium-size fish, and skipjack tuna tends to account for the majority of the catch (~70%−83% in recent years, but typically more than 85% of the total catch in tropical areas), and albacore (8%−20% in recent years) is taken by Japanese coastal and oceanic fleets in the temperate waters of the north Pacific (Williams and Terawasi 2014). The remainder of the catch consists of yellowfin tuna (5%−16%) and a small component of bigeye tuna (1%−4%).

Table 8-3. Observed takes, mortalities (M), takes per fishing effort (1,000 hooks) for sea turtles in the Hawaii deep-set longline fishery, 2002-2017. Table adapted from: WPRFMC 2017.

	n.temp		trop		s.temp		WCPFC-CA		
Year	**shallow**	**deep**	**shallow**	**deep**	**shallow**	**deep**	**shallow**	**deep**	**Total**
2003	82.4	162.8	45.1	291.1	59.7	208.6	187.2	662.5	849.8
2004	100.4	174.8	24.9	321.4	58.1	188.8	183.4	685.0	868.4
2005	106.1	159.1	3.3	285.6	42.3	169.8	151.7	614.5	766.2
2006	104.9	163.7	6.7	270.4	28.8	183.5	140.4	617.6	758.0
2007	163.8	152.2	49.3	257.2	28.3	171.6	241.4	581.1	822.5
2008	178.5	164.0	33.5	247.7	19.0	197.1	231.0	608.8	839.9
2009	187.8	136.6	64.9	231.5	25.8	236.9	278.6	605.0	883.6
2010	138.9	134.2	48.3	247.4	35.1	284.3	222.3	665.8	888.1
2011	166.1	155.4	47.2	289.0	33.4	249.3	246.8	693.7	940.4
2012	127.9	172.0	72.8	311.9	18.0	306.3	218.7	790.2	1008.9
2013	72.0	163.5	40.1	260.2	14.5	285.9	126.5	709.7	836.2
2014	98.6	160.4	21.4	281.3	12.9	267.3	132.9	709.0	841.9
2015	101.1	167.5	12.1	351.5	11.6	240.1	124.8	759.1	883.9
2016	93.7	185.7	6.7	265.8	10.6	209.9	111.0	661.4	772.3
2017	68.8	187.4	12.2	222.4	11.2	273.7	92.3	683.6	775.8

Western and Central Pacific Ocean – Purse Seine Sea Turtle By-catch

The WCPFC also has a responsibility to assess the impact of fishing on non-target species. The Commission has adopted certain Conservation and Management Measures (CMMs) for non-target species such as sea turtles, to ensure the conservation of such species. One of these measures is the use of on-board observers, as this is the only reliable source of information on those species. The most updated assessment of by-catch of turtles in large scale purse seine fishery in the WCPFC-CA, spanning 2003 to 2017 and which accounted for >80% of reported catches, are provided by Peatman et al. (2018a). Observer coverage to 2009 was in the order of 15%−20%, but since that time observer coverage was 60% to 70% with the exception of 2017 (Peatman et al. 2018a).

Estimated total turtle by-catch in the WCPFC generally increased from 2004 to 2013, from 130 to 390 individuals per year, and then declined from 2014 to 2017 (Table 8-4). Green turtle (24%), olive ridley (23%), loggerhead (20%) and hawksbill turtles (16%) accounted for the majority of turtle by-catch from 2003 to 2017 (Peatman et al. 2018a; Table 8-5).

Uncertainty in these by-catch estimates was highest for 2003 to 2009, when observer coverage was comparatively low, but uncertainty decreased from 2010 to 2016 as a result of the increase in observer coverage. Estimates of by-catch for 2017 had relatively high uncertainty, due to the low levels of observer coverage when the data were extracted for analysis. The magnitude of uncertainty in by-catch estimates was primarily a function of how frequently the species were observed, with higher uncertainty for species that were more rarely caught. Sea turtle by-catch estimates had 95% confidence intervals of 35% for 2003 to 2009, but only 6% for 2010 to 2016 (Peatman et al. 2018a). That is, total by-catch from 2003 to 2009 may have been as high as 2,195 turtles or as low as 1,093 turtles, while from 2010 to 2016 this range was 1,664 to 1,889 turtles. By-catch estimates for 2017 should be considered preliminary.

Table 8-4. Estimated annual by-catch and by-catch rates for WCPFC large-scale purse seine fleets. Median by-catch (med), and lower (low) and upper (high) 95% confidence intervals. Data extracted from: Peatman et al. 2018a.

	Turtles (n)			By-catch rate per	
Year	Low	Med	High	set	'000 mt
2003	218	323	443	0.011	0.32
2004	77	129	199	0.004	0.12
2005	129	190	264	0.005	0.16
2006	119	171	234	0.005	0.14
2007	194	275	383	0.008	0.20
2008	160	227	305	0.006	0.16
2009	196	260	337	0.006	0.17
2010	198	214	230	0.004	0.14
2011	356	378	403	0.007	0.27
2012	264	282	301	0.005	0.17
2013	300	314	330	0.006	0.20
2014	195	209	224	0.004	0.12
2015	203	216	229	0.005	0.14
2016	148	160	172	0.003	0.10
2017	97	148	219	0.003	0.10

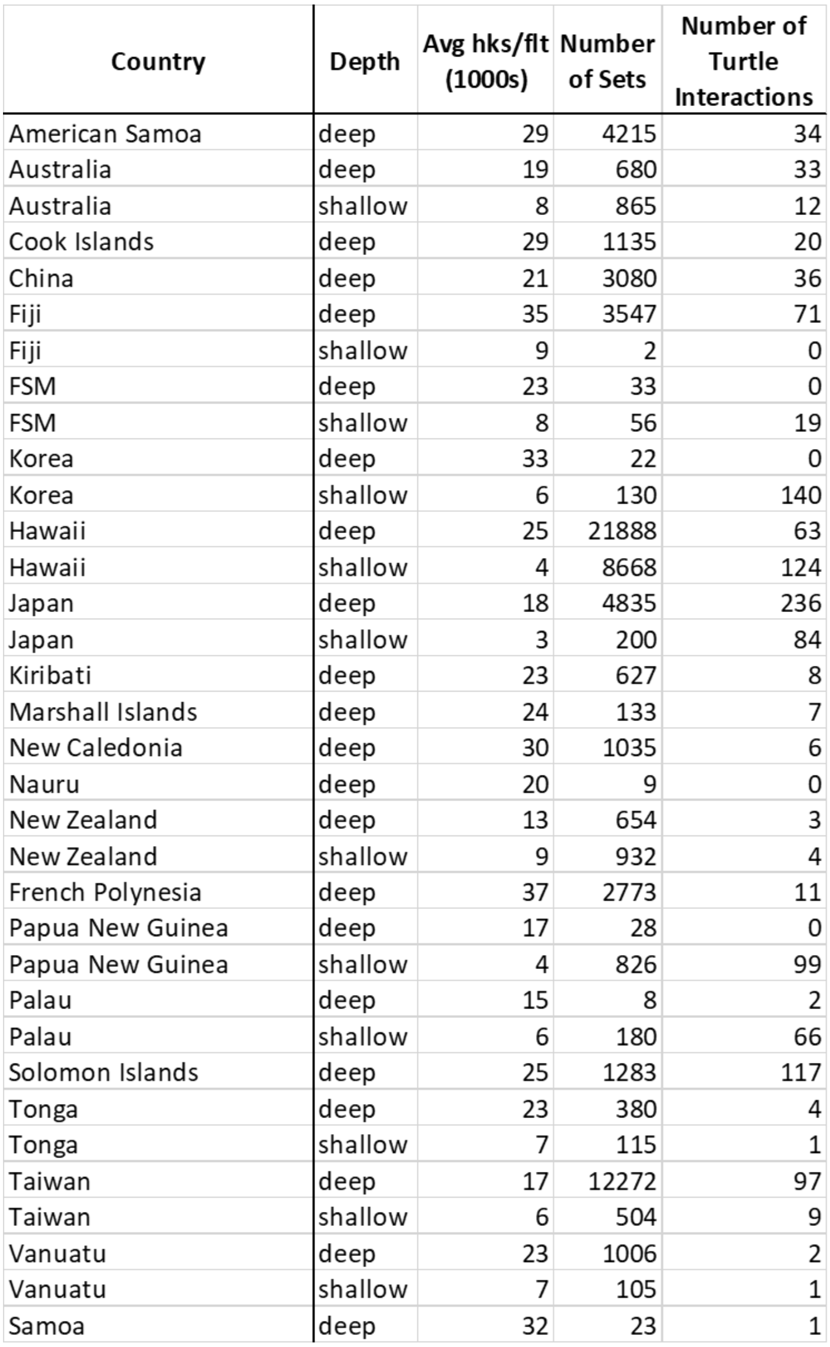
Table 8-5. Species composition of sea turtle by-catch in WCPFC large-scale purse seine fleets. Table Source: Peatman et al. 2018a.

Year	Green turtle	Olive ridley turtle	Loggerhead turtle	Hawksbill turtle	Leatherback turtle	Marine turtles nei	Total
2003	38	37	0	25	0	217	**323**
2004	0	16	0	15	12	84	**129**
2005	37	7	30	17	0	94	**190**
2006	20	63	26	30	13	14	**171**
2007	98	64	55	29	5	18	**275**
2008	41	36	100	32	8	7	**227**
2009	52	62	85	45	6	5	**260**
2010	58	40	56	43	7	9	**214**
2011	76	130	81	75	7	9	**378**
2012	73	78	57	50	6	17	**282**
2013	94	67	69	63	8	12	**314**
2014	63	51	29	44	9	13	**209**
2015	84	48	49	25	4	5	**216**
2016	41	44	35	19	14	5	**160**
2017	27	42	34	32	5	3	**148**
Species totals	**803**	**784**	**706**	**543**	**104**	**513**	**3,495**

Western and Central Pacific Ocean – Longline Sea Turtle By-catch

In 2017 the WCPFC, in a joint analysis of sea turtle mitigation effectiveness in the tuna fishery (WCPFC 2017), noted that there was generally insufficient information to quantify the severity of the threat posed by longline fisheries to sea turtle populations, but also that there was no information to suggest that such threats had appreciably diminished in recent years. The report noted that while approximately 20% of the WCPO longline effort was in shallow sets, analysis suggests that <1% of WCPO longline effort is currently subject to mitigation, and that while mitigating <1% of WCPO longline effort was marginally better than not mitigating at all, it did raise the question of effectiveness of the WCPFC's intent to mitigate sea turtle by-catch. Even with this level of coverage, the report recognised that there were some 2,300 turtle interactions in more than 148,000 observed sets representing 311 million observed hooks between 2009 and 2016 (WCPFC 2017). Japan provided additional observer data for the WCPFC (2017 report) for the eastern Pacific Ocean representing 31 trips with 1,927 sets and 79 sea turtle interactions between 2007 and 2015, from 150°W to 87°W and from 19°S to 28°N. Of note, this synthesis indicated a general positive correlation between the number of sets and the number of turtle interactions, which potentially might be used as a proxy in those cases where by-catch data are not reported (including in illegal, unregulated and unreported - IUU - fisheries). A summary of turtle interactions by fishery is presented in Table 8-6, while a temporal depiction of turtle interactions is shown in Figure 8-5.

Table 8-6. Summary of fishing effort, fishery type and turtle interactions in the WCPO fleets based on data provided since 2009 up to 2016 (noting that the most recent year varies by fleet). Table adapted from: WCPFC 2017.



Country	Depth	Avg hks/flt (1000s)	Number of Sets	Number of Turtle Interactions
American Samoa	deep	29	4215	34
Australia	deep	19	680	33
Australia	shallow	8	865	12
Cook Islands	deep	29	1135	20
China	deep	21	3080	36
Fiji	deep	35	3547	71
Fiji	shallow	9	2	0
FSM	deep	23	33	0
FSM	shallow	8	56	19
Korea	deep	33	22	0
Korea	shallow	6	130	140
Hawaii	deep	25	21888	63
Hawaii	shallow	4	8668	124
Japan	deep	18	4835	236
Japan	shallow	3	200	84
Kiribati	deep	23	627	8
Marshall Islands	deep	24	133	7
New Caledonia	deep	30	1035	6
Nauru	deep	20	9	0
New Zealand	deep	13	654	3
New Zealand	shallow	9	932	4
French Polynesia	deep	37	2773	11
Papua New Guinea	deep	17	28	0
Papua New Guinea	shallow	4	826	99
Palau	deep	15	8	2
Palau	shallow	6	180	66
Solomon Islands	deep	25	1283	117
Tonga	deep	23	380	4
Tonga	shallow	7	115	1
Taiwan	deep	17	12272	97
Taiwan	shallow	6	504	9
Vanuatu	deep	23	1006	2
Vanuatu	shallow	7	105	1
Samoa	deep	32	23	1

The report recommended expanding mitigation to deep-set longlines because sea turtles have a higher probability of asphyxiation in deep sets; and that expanding mitigation to deep-set longlines would also deliver stronger reductions in interaction rates due to the four-times greater effort in deep-set longline fisheries.

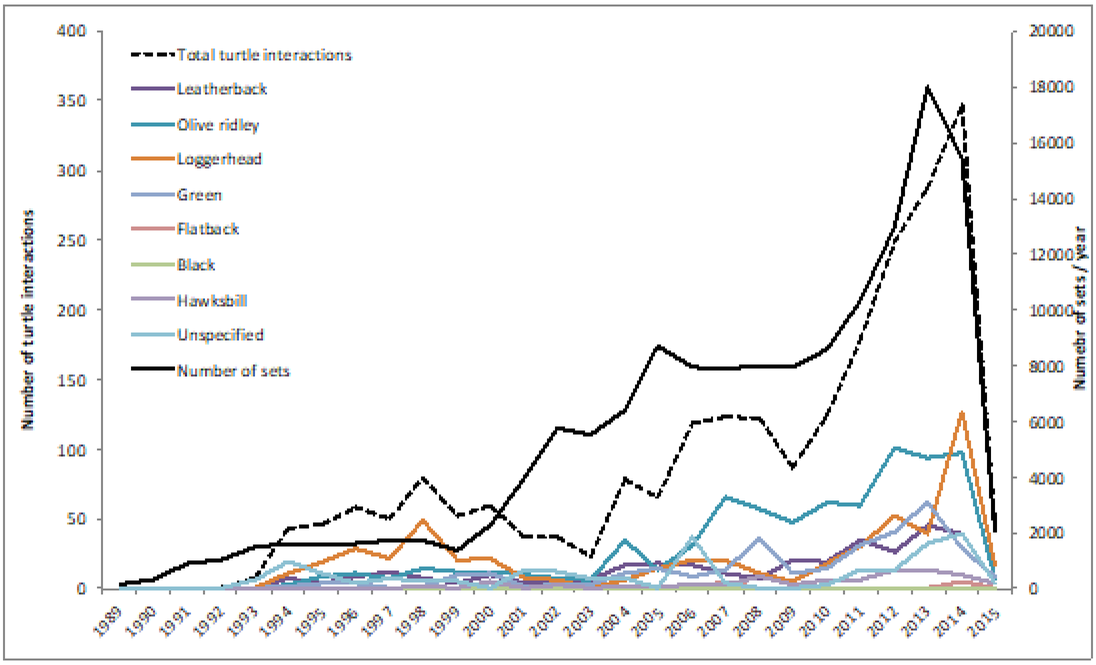


Figure 8-5. Temporal trend in sea turtle interactions in the WCPO longline fishery compared to number of longline sets. Data extracted from WCPFC 2017.

A summary of by-catch of longline fisheries from 2003 to 2017 in the WCPFC-CA is also provided by Peatman et al. (2018b), who summarise observer data and estimate catches for sea turtles, regardless of whether or not they were targeted or caught incidentally. Peatman et al. (2018b) used a simulation modelling framework to estimate longline catches by linking estimated catch-rates by estimated Hook Between Float (HBF) effort. Across-the-fishery observer coverage from 2003 to 2010 was in the range of ~20%, rising to a high of ~45% in 2013 but declining since then to about ~25%. A problem, however, was that observer coverage was not distributed evenly among the fisheries and in the WCPFC-CA – in some places coverage was 20% at the same time as it was 1% elsewhere. In addition, the level of observer coverage was not consistent with the level of effort – that is, in one low effort area (Hawaii) there was high observer coverage but in other higher effort areas, such as NE of New Caledonia, there was low observer coverage (Peatman et al. 2018b; Figure 8-6).

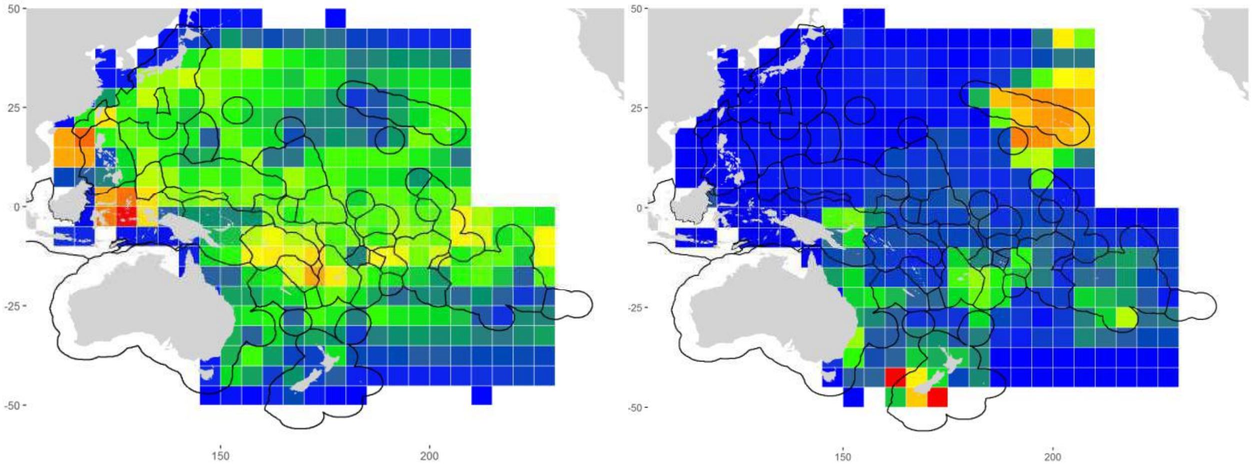


Figure 8-6. Hotspot graphics of reported effort (left) and observer coverage (right; as a proportion of hooks) for longline fleets in the WCPFC-CA from 2003 to 2017. Image adapted from: Peatman et al. 2018b.

Olive ridley turtles were the most frequently caught in both the deep and shallow set fisheries between 2003 and 2017 (Peatman et al. 2018b). A subset of 20% to 30% of the turtles were released alive (healthy or injured), and this was generally better for sea turtles caught by shallow sets (Peatman et al. 2018b). Catch estimates of the differing sea turtle species are presented in Table 8-7, while median, lower and upper 95%

confidence limits for loggerheads are presented in Table 8-8, and differences in take in shallow and deep longline sets are presented in Table 8-9. Sea turtles were mainly discarded but there are instances where individuals were retained (Figure 8-7).

Table 8-7. Species composition of sea turtle by-catch in WCPFC longline fleets. Table Source: Peatman et al. 2018b.

Year	Olive ridley	Green turtle	Loggerhead turtle	Leatherback turtle	Hawksbill turtle	Marine turtles nei	Annual total
2003	9,670	2,166	91	1,399	788	1,434	15,548
2004	6,495	4,186	266	1,847	850	2,695	16,337
2005	4,294	2,662	1,286	1,996	638	2,018	12,894
2006	5,065	1,509	3,591	1,496	534	1,359	13,555
2007	19,635	5,393	3,273	1,583	979	1,046	31,908
2008	20,296	9,645	1,120	1,000	1,149	540	33,750
2009	29,393	9,356	964	1,424	1,535	249	42,921
2010	18,900	4,219	1,209	1,856	1,586	177	27,947
2011	13,030	2,796	1,376	1,992	1,355	148	20,697
2012	14,711	3,211	1,443	2,153	1,428	271	23,217
2013	9,588	3,254	1,797	1,808	969	378	17,795
2014	10,184	3,555	3,272	1,840	996	835	20,681
2015	14,591	3,990	5,208	1,865	1,598	3,622	30,874
2016	12,364	3,586	3,345	1,132	1,597	1,161	23,184
2017	8,507	5,769	1,163	617	901	22	16,979
Species total	**196,722**	**65,296**	**29,405**	**24,006**	**16,902**	**15,956**	**348,286**

Table 8-8. Median (med) and lower (low) and upper (high) 95% confidence intervals for loggerhead turtle by-catch (individuals) by region in WCPFC longline fleets. Table Source: Peatman et al. 2018b.

	north temp			trop			south temp			Total		
Year	Low	Med	High	Low	Med	High	Low	Med	High	Low	Med	High
2003	2	34	501	1	15	236	1	11	148	16	91	714
2004	30	141	635	11	55	228	9	37	154	103	266	794
2005	286	753	1,976	103	259	701	81	200	505	673	1,286	2,670
2006	906	2,158	4,967	311	722	1,602	236	522	1,272	1,973	3,591	6,507
2007	937	2,117	5,139	252	707	1,769	144	334	730	1,855	3,273	6,351
2008	231	753	2,236	54	180	631	42	106	325	521	1,120	2,614
2009	204	553	1,592	83	237	725	40	102	263	479	964	2,010
2010	222	616	1,679	117	329	978	82	189	476	660	1,209	2,346
2011	288	787	2,018	127	332	990	71	180	427	726	1,376	2,667
2012	247	666	1,807	155	487	1,449	70	187	526	712	1,443	3,015
2013	322	804	1,987	236	587	1,699	129	288	650	1,018	1,797	3,407
2014	765	1,785	4,168	391	852	2,115	220	469	981	1,890	3,272	5,980
2015	1,220	2,636	5,899	798	1,642	3,897	348	699	1,502	3,254	5,208	8,776
2016	819	1,905	4,398	381	793	1,805	250	521	1,104	1,981	3,345	5,822
2017	140	555	2,178	70	257	923	59	220	822	522	1,163	2,927

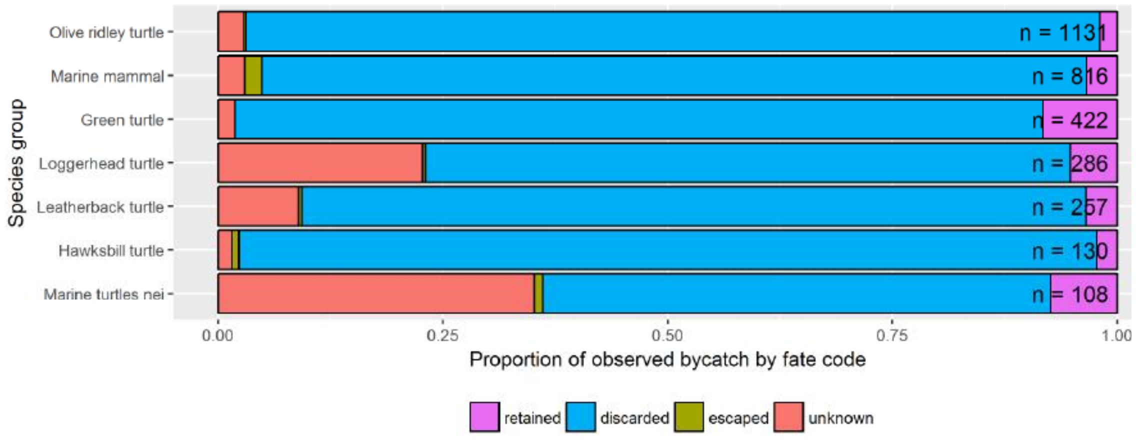


Figure 8-7. Recorded fate of observed species of marine mammals and sea turtles catch by species/species group, as a proportion of total observed catch (number of specimens) for the species/species group in the longline fisheries. Image source: Peatman et al. 2018b.

Table 8-9. Median (med) and lower (low) and upper (high) 95% confidence intervals for loggerhead catch (individuals) deep and shallow sets. Table Source: Peatman et al. 2018b.

	shallow			deep			Total		
Year	**Low**	**Med**	**High**	**Low**	**Med**	**High**	**Low**	**Med**	**High**
2003	6	40	377	8	46	296	16	91	714
2004	36	123	479	56	137	344	103	266	794
2005	274	617	1,526	351	646	1,190	673	1,286	2,670
2006	756	1,690	3,760	1,094	1,838	2,935	1,973	3,591	6,507
2007	1,123	2,182	4,756	631	1,079	1,772	1,855	3,273	6,351
2008	238	652	1,810	239	448	889	521	1,120	2,614
2009	255	586	1,429	197	367	667	479	964	2,010
2010	287	635	1,456	336	572	1,060	660	1,209	2,346
2011	331	771	1,719	348	600	1,053	726	1,376	2,667
2012	301	684	1,662	407	751	1,397	712	1,443	3,015
2013	367	737	1,608	630	1,043	1,877	1,018	1,797	3,407
2014	779	1,550	3,292	1,059	1,698	2,810	1,890	3,272	5,980
2015	946	1,867	3,992	2,095	3,291	5,525	3,254	5,208	8,776
2016	610	1,352	3,039	1,260	1,981	3,101	1,981	3,345	5,822
2017	150	408	1,396	347	740	1,630	522	1,163	2,927

Coefficients of variation (CVs) of the simulation model catch estimates were calculated for each species as a general measure of their uncertainty. In this scenario, strata with low catch rates generally have higher CVs and vice versa, in much the same way as the tendency for species with lower catch rates to have higher CVs compared to species with higher catch rates (Peatmen et al. 2018b). Coefficients of variation for sea turtles ranged from 60% (leatherback turtles) to 350% (hawksbill turtle), highlighting the difficulty in deriving precise estimates due to low catch rates coupled with low observer coverage. The uncertainty of the catch estimates only incorporates uncertainty in catch rates, and does not include uncertainty in the estimated proportions of effort. An example of the variance around median catch figures is presented in Figure 8-8. Given the consistency in reporting and coverage, it is likely that the *trend* in by-catch of sea turtles is accurate across years. What may be less reliable, however, is the *magnitude* of turtle by-catch. Peatman et al. (2018b) acknowledge that the turtle by-catch estimates are likely an overestimate, given an earlier assessment (Common Oceans 2017) that estimated roughly half these by-catch rates that accounted for estimates of sea turtle distributions, and benefited from additional observer data. Given this, the estimates of by-catch for sea turtles in the WCPFC-CA longline industry should be treated with caution but it would be reasonable to presume that by-catch rates are at least 50% to 100% of the results presented by Peatman et al. 2018b).

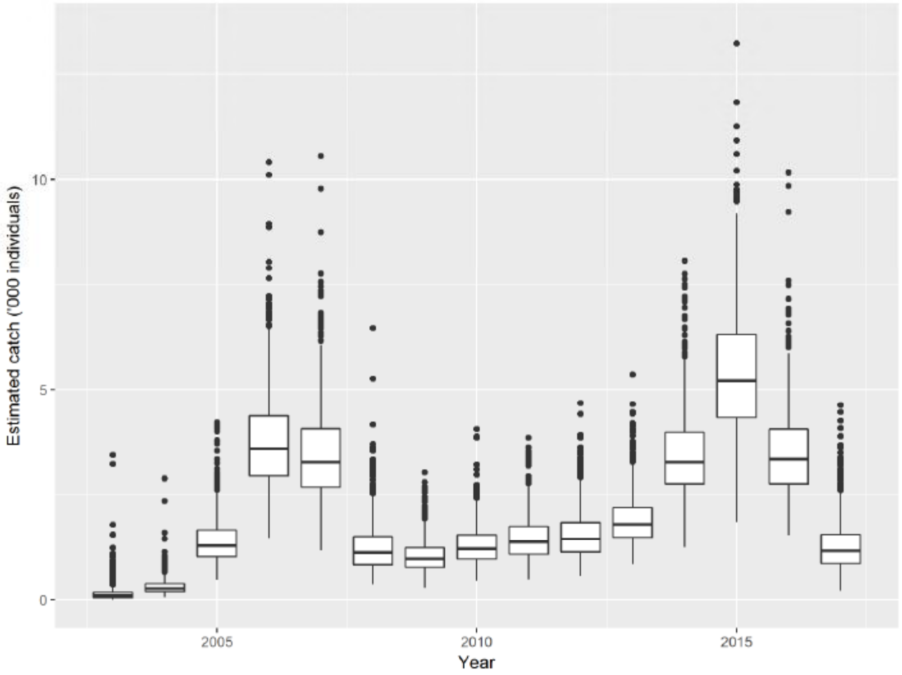


Figure 8-8. Total estimated loggerhead turtle by-catch per year. Image source: Peatman et al. 2018b.

There are limited interactions between the WCPFC purse seine fishery and protected species, and observed interaction rates between the purse seine fishery and sea turtles are low (< 1 interaction per 100 sets; Hare et al. 2019). Given the purse seine nets have averaged ~ 57,000 sets per year since 2008 (Hare et al. 2019), this equates to an annual by-catch of fewer than 570 turtles.

More recently in 2020, Peatman and Nicol (2020) indicate that it remained difficult to obtain reliable estimates of WCPO longline catches from observer data, given the low levels and imbalanced nature of observer coverage, and additionally the low coverage of available aggregate effort data disaggregated by hooks between float in the mid-2000s. They noted that observer coverage has been particularly low in the north west Pacific, and that the catch estimates for the region north of 10°N, and consequently the catch estimates for the WCPFC-CA as a whole, were unlikely to be reliable and should be viewed in that context. Through improved modelling methods, Peatmen and Nicol (2020) indicated that while the approach used to generate uncertainty in catch estimates was more statistically robust than that used in the 2018 study, it was still the case that the uncertainty in catch estimates did not include uncertainty in the estimated proportions of effort by hooks between float category. The number of hooks between floats, a proxy for the depth of fishing gear, has a large impact on the catch rates for a wide range of the species, and particularly sea turtles. As such, they noted that the uncertainty in catch estimates was likely underestimated between 2003 and 2009. Peatman and Nicol (2020) reported olive ridley turtle catch estimates had a peak of ~ 25,000 individuals in 2009, and represented 62% of total estimated catches of olive ridley, green, loggerhead and leatherback turtles. However, while Peatman and Nicol (2020) note that the sea turtle by-catch estimates are not entirely reliable, their findings are at least indicative of the magnitude of turtle interactions in this fishery over time (Figure 8-9).

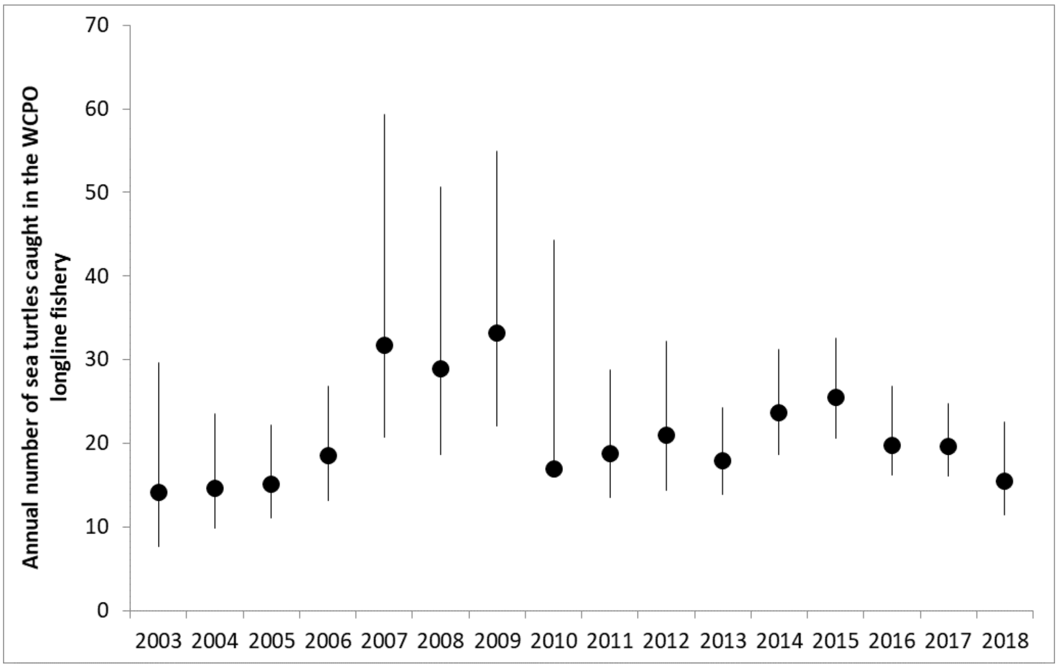


Figure 8-9. Estimated annual longline catch (in '000s individuals). Median catch (black circles), along with upper and lower 95% confidence intervals. Data source: Peatman and Nicol 2020.

Taiwanese and Japanese longline fleets also operate throughout the Pacific and by-catch data from these fisheries complements estimates of mortality from the WCPFC.

Taiwan operates both large-scale and small-scale longline vessels. In 2012, there were 87 large-scale vessels and 1,436 small-scale vessels. A total of 50 albacore large-scale vessel trips from Taiwan were observed from 2008 to 2013 and the majority of effort (88%) was in the southern hemisphere. Forty sea turtles were incidentally caught between 5°−15°S, 155°−170°W. Of these, 31 were olive ridley, 4 green, 3 loggerhead, 1 leatherback and 1 hawksbill turtle. A total of 33 (88%) were dead when landed. An additional 72 bigeye large-scale vessel trips were observed from 2008 to 2013 and most effort from these was between 5° N and 15° S in the western Pacific Ocean. A total of 33 sea turtles were incidentally caught between 10°N−10°S, 145°W−155°E. Of these, 12 were leatherbacks, nine olive ridley, seven green, three loggerhead and two hawksbill turtles; 27.3% were dead when landed. A total of 27 small-scale vessel trips were observed between 2012 and 2013, for which there were two major fishing grounds: between 20°−35° N and 135°−155°E, and the other between 5°N−15°S and 155°E−165°W. Fifty sea turtles were incidentally caught between 5°N−15°S, 155°−175°E. Of these, 38 were olive ridley, five green, two loggerhead, three hawksbill and two leatherback turtles; 73.5% were dead when landed (Huang 2014). Total estimated turtle by-catch between 2008 and 2013 in large-scale Taiwanese longline vessels was 1,147 turtles (Huang 2014). Based on data presented by Huang (2014) a possible additional 1,842 turtles were taken in small-scale longline fisheries during the same period. A summary of by-catch rates and 95% confidence intervals by ocean region for large-scale vessels is presented in Table 8-10. No comparable data is available for the small-scale longline vessels.

Table 8-10. Sea turtle by-catch rates and 95% confidence intervals in the Pacific Ocean between 2008 and 2013 for large-scale vessels. (NE-Northeast; NW-Northwest; TE-Tropical East; TW-Tropical West; SE Southeast; SW-Southwest). Table Source: Huang 2014

Season Area	Jan-Mar	Apr-Jun	July-Sep	Oct-Dec
PAC_NE	0	N.A.	0	0
PAC_NW	0	N.A.	N.A.	0
PAC_TE	0	0.001 (0.00003-0.0075)	0.002 (0.0002-0.0054)	0.002 (0.0005-0.0067)
PAC_TW	0.003 (0.0014- 0.0071)	0.011 (0.0080-0.0159)	0.002 (0.0009-0.0040)	0.006 (0.0032-0.0096)
PAC_SE	0	0	0	0
PAC_SW	0.004 (0.0001 -0.0225)	0	0	0

Japan deployed ~1400 vessels annually between 2000 and 2004. The Japanese fleets had a wide distribution across the Pacific: in the longline fishery, most vessels of 100−199 GRT operated inside the WCPFC-CA, while most of the boats >200 GRT operated outside the WCPFC-CA (Figure 8-10). Effort varied from 119 million to 128 million hooks per year and landings ranged from ~54,000 to ~66,000 metric tonnes (Uosaki et al. 2005). In comparison, there were ~47 pole and line vessels and ~36 purse seine vessels. In 2016 (Uosaki et al. 2017) reported ~350−400 vessels operated annually in the WCPFC-CA between 2011 and 2016, along with ~75−100 pole and line vessels and ~75 purse seine vessels. Between 2011 and 2016, there were 105 turtle by-catch events in 51 small-scale longline trips and an additional 80 by-catch incidences across 22 distant water trips. No data were provided that would assist in extrapolating these figures to annual total by-catch estimates (Uosaki et al. 2017).

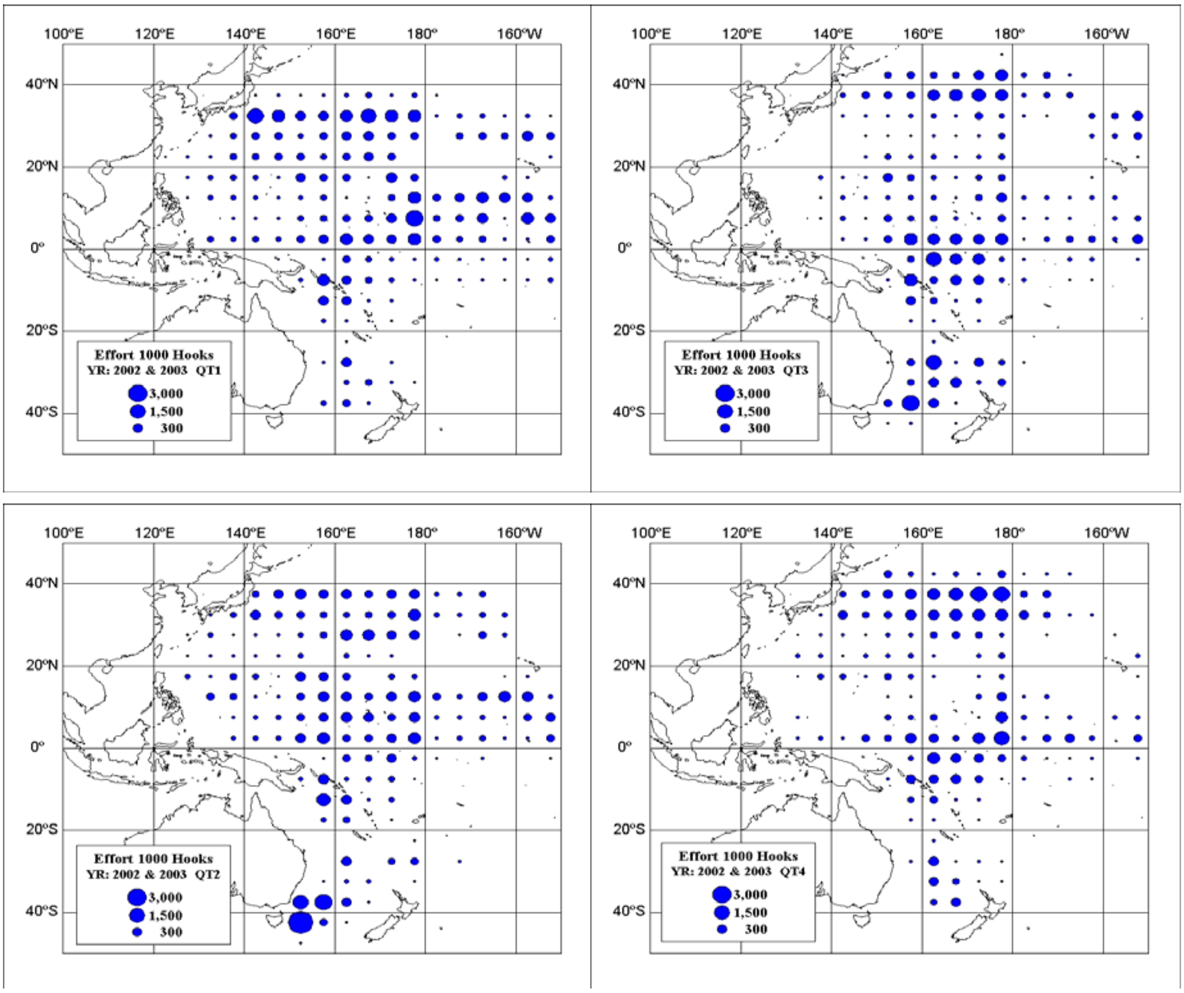


Figure 8-10. Quarterly distribution of fishing effort for the Japanese offshore and distant water longline fisheries in the western and central Pacific Ocean in 2002–2003. Image Source: Uosaki et al. 2005.

South Pacific

For the south Pacific, the decline in breeding numbers of loggerheads in eastern Australia, from approximately 3,500 females per year in the mid-1970s to approximately 500 by the year 2000 was attributed primarily to by-catch mortality in otter trawl fisheries of northern and eastern Australia (Robins et al. 2002). Threats from numerous fisheries have been reported that could impact this stock (Limpus and Reimer 1994, Poiner and Harris 1996, Robins et al. 1998, 2002, Limpus 2007c, Kelez et al. 2004, Alfaro Shigueto et al. 2008). Limpus (2007c) reported that the extrapolated mean annual catch of loggerheads was estimated at 2,938 in East Coast Trawl Fishery (ECTF) and 85 in Torres Straits Prawn Fishery (TSPF) between 1991 and 1996. The total annual direct mortality associated with these captures was estimated to be in the range of 1%–5% in the ECTF and 7%–14% in the TSPF. However, the introduction of Turtle Excluder Devices (TEDs) in 2000 to the fishery resulted in a two orders of magnitude reduction in turtle captures in the Northern Prawn Trawl Fishery (NPTF) trawls. In addition to this, there are now regulations that limit the number of days a trawler can be at sea, and a Marine Park zoning plan that has reduced the total area of coastal waters available for trawling. Recent data on by-catch rates in Australia indicate that the mitigation measures (e.g. TEDs) have been largely effective at preventing turtle by-catch. For instance, in the first three months of 2012 the Eastern Tuna and Billfish fishery caught two green turtles and one leatherback turtle, and the TSPF caught four green turtles and one unclassified turtle – all were released alive (AFMA 2012). The Australian Fisheries Management Authority has a mandatory reporting programme for by-catch events, and provides quarterly reports on endangered and protected species caught in various fisheries (https://www.afma.gov.au/sustainability-environment/protected-species-management/protected-species-interaction-reports). A summary of recent by-catch records in 2019 and 2020 is presented in Table 8-11.

Table 8-11. Sea turtle by-catch records for key fisheries interacting with Pacific sea turtles in Australia 2019-2020. Numbers in parenthesis indicate turtles released alive. Table Source: https://www.afma.gov.au/sustainability-environment/protected-species-management/protected-species-interaction-reports

Fishery	Bycatch species	2019				2020		
		Jan-Mar	Apr-Jun	Jul-Sep	Oct-Dec	Jan-Mar	Apr-Jun	Jul-Sep
Eastern Tuna & Billfish (longline)	Green	15(8)	23(21)	7(7)	7(6)	8(6)	12(11)	3(2)
	Hawksbill	2(1)	3(1)	1(1)	2(1)		3(1)	
	Leatherback	13(13)	22(21)	7(1)	10(10)	5(5)	7(7)	4(4)
	Loggerhead	4(1)	2(1)	1(1)	4(2)	19(1)	5(3)	3(3)
	Unclassified	6(3)	4(4)	4(3)	4(3)	5(3)	3(3)	
Northern Prawn (trawl)	Green		2(2)	12(12)	5(5)		5(5)	7(7)
	Hawksbill				1(1)			1(1)
	Leatherback							
	Loggerhead				3(3)			5(4)
	Unclassified		4(4)	16(16)	26(26)		7(7)	
								43(43)
Torres Strait Prawn (trawl)	Green		19(1)					

Flatback sea turtles constituted nearly 60% of sea turtles caught in the NPTF (Pointer and Harris 1996), and approximately 10% of sea turtles caught in the ECTF (Robins 1995, Slater et al. 1998). Hawksbills were also documented in the NPTF at a rate of 0.0018 ± 0.0007 turtles per trawl in 1989 and 0.0029 ± 0.0010 with a 19.2% probability of being landed dead in the sorting tray in 1989 and 33.3% probability in 1990. This study estimated that the NPTF killed approximately 68 and 64 hawksbills in 1989 and 1990, respectively (Poiner and Harris 1994, 1996). It is likely that TEDs are equally efficient at minimising flatback and hawksbill captures and that mortality in this fishery is now negligible. Leatherback turtles have been rarely captured in prawn trawls in eastern Queensland and Gulf of Carpentaria (Limpus 2007d). Limpus (2007d) indicates that mortality of leatherbacks in hook and line fishing is rare, but that entanglement in lobster traps is substantial.

The capture of leatherback turtles in gillnet fisheries in Australian waters has been widespread but the catch rate and numbers killed as by-catch within these fisheries has not been quantified (Limpus 2007d). Leatherbacks were captured regularly during the early 1990s and an undetermined, but probably low, number drowned in the barramundi gillnet fishery of the southeastern Gulf of Carpentaria (unpublished data, Queensland Turtle Conservation Project, Limpus 2007d). It is possible that this mortality however, led to only remnant nesting of leatherbacks along Australia's northern coast.

Loggerheads and leatherbacks are also recorded as by-catch in the Australian offshore longline fisheries of the eastern tuna and billfish fishery and the southern and western tuna and billfish fishery (Robins et al. 2002), and Limpus (2007c) indicated these fisheries may catch around 400 turtles (all species) per year, noting that the species composition of the catch is poorly reported and observer coverage of the effort is low (5%). Approximately 60% of the several hundred turtles captured annually in these fisheries are believed to be leatherbacks (Robins et al. 2002), but the mortality rate is probably low (Robins et al. 2002). In contrast to flatback and loggerhead captures, green turtles were caught less frequently in these fisheries (Limpus 2007a), with annual catches of greens representing only 5%−10% of all turtles caught. It is likely a small number of hawksbill turtles were also caught in these fisheries (Limpus 2007b).

Immature loggerheads comprised about 30% of the turtle by-catch by longline vessels off Peru in the eastern Pacific (Kelez et al. 2004). The catch per unit effort (CPUE) combined for all species of turtles caught was 0.296 turtles per 1000 hooks but the total size of the fleet and its total fishing effort was not described. A total of 323 loggerhead turtle captures was also recorded between 2000 and 2007 in Peruvian small-scale gillnet and long-line fisheries (Alfaro Shigueto et al. 2008). Curved carapace length ranged from 35.9 cm to 86.3 cm suggesting a predominance of juvenile turtles. Alfaro-Shigueto et al. (2009) indicated that by-catch composition in Peru consisted of 85.6% greens, 9.2% olive ridleys, 3% leatherbacks, 1% loggerheads and 1% unidentified. Some 87% of turtles were entangled alive and 58% were released without injury. Size classes included large juveniles for greens and olive ridleys, while both juvenile and adult leatherbacks were caught.

In the driftnet fishery, the observed by-catch rate was 1.6 ±2.7 turtles/trip; and in the bottomset fishery the observed by-catch rate was 1.8 ±2.7 turtles/trip. Given Alfaro Shigueto et al. (2008) reported ~63,083 gillnet and 11,316 longline trips annually from Peru, this suggested there are substantial impacts on sea turtles in the southeastern Pacific.

In international waters leatherback turtles are caught in longline fisheries, and across the entire Pacific Ocean it was estimated that 20,000–40,000 leatherbacks were captured as by-catch in longline fisheries in 2000 (Lewison et al. 2004).

For flatback turtles, entanglement in commercial fisheries gear, especially trawl and gill nets, is believed to have historically posed the greatest threat to adult and large immature flatback turtles resulting in the highest recorded mortalities for the species (Robins 1995, Robins et al. 2002, Limpus 2007e), but total mortality has not been quantified for the trawl fisheries of western and eastern Australia. The Northern Prawn Fishery historically caused an estimated annual flatback mortality of between 337 and 355 turtles (1990 and 1989 data respectively, Poiner and Harris 1994, 1996). Taking into account fishery effort between 1970 and 1999, this resulted in an average estimated mortality of 270 (SD = 111) turtles per year with an overall mortality of over 8,000 flatback turtles.

East Pacific

A major threat to east Pacific leatherback turtles is by-catch in small-scale gillnets and longlines within South American migration and foraging habitats (Ortiz-Alvarez et al. 2020). Leatherback by-catch was reported in Mexico, Nicaragua, Costa Rica, Panama and Colombia in a rapid assessment in 2016−2017 that estimated 345 ± 210 (mean ± SD) individual leatherbacks were caught annually across the 79 ports surveyed (Ortiz-Alvarez et al. 2020). The study also estimated that mortality after these interactions was low, at 1.0%. Leatherback interactions increased around index nesting beaches and ports with the highest by-catch were in Mexico: Paredón (118 ± 106), Barra Tecoanapa (72 ± 30), and Punta Maldonado (51 ± 20). The latter two are located close to index nesting beaches, Playa Tierra Colorada and Playa Cahuitán (Ortiz-Alvarez et al. 2020). In addition, the study also documented by-catch of olive ridley turtles (n=810), green turtles (n=124), hawksbills (n=57), one loggerhead and 150 turtles of unknown species. Leatherback turtle use of coastal waters by leatherbacks in Costa Rica at Las Baulas National Park during inter-nesting periods was found to be significantly concentrated, whereby by-catch potential was high, but also so was the opportunity for protecting important life stages for the species (Shillinger et al. 2010). Similarly, Roe et al. (2014) used satellite tracking data for leatherbacks to identify offshore hotspots where east Pacific leatherback turtles were particularly at risk (Figure 8-11). Based on existing by-catch assessments mentioned above, between 1,000 and 2,000 leatherbacks (adult males and females, as well as juveniles) are caught in nets and longlines annually, of which roughly 30%−50% die as a result of these interactions; therefore, estimated by-catch mortality is between 300 and 600 and as many as 1,000 leatherbacks annually.

In Baja California Sur, Senko et al. (2014) estimated that 30% (n=~233) of the green turtles they encountered on beaches and in villages were the result of by-catch, of which 95% were immature turtles (n=~217). Their study indicated that despite several decades of protection, by-catch and sea turtle mortality in the region continued to be high.

Arauz et al. (2000) describe a catch per unit effort of 0.85 green turtles per 100 hooks in Costa Rican longline fisheries operating east of the Galápagos. Green turtles are also the second most common sea turtle species bycaught in Peruvian artisanal shark longline fisheries (Kelez-Sara et al. 2006) and Colombian shrimp trawl fisheries (Amorocho et al. 2005). Alfaro Shuigueto et al. (2011) estimated an annual catch of ~5,900 turtles (3200 loggerhead, 2400 green, 240 olive ridleys and 70 leatherback turtles). The fishery is described as a small-scale fishery with >100 ports, >9,500 vessels, >37,000 fishers (Alfaro Shigueto et al. 2010). However, the observed effort between 2000 and 2007 constituted only ~1% of longline and net deployments, and Alfaro Shigueto et al. (2011) caution that the number of turtles captured per year is likely to be in the tens of thousands. Castro et al. (2012) also reported by-catch of turtles in the gillnet fisheries in Peru, indicating that 80.4% of captures were olive ridley turtles, 19.3 green and 0.2% hawksbill. Interestingly they did not report loggerhead captures, which contrasts substantially with findings by Alfaro Shigueto et al. 2011. Castro et al. (2012) suggest by-catch in these fisheries may increase during El Nińo Southern Oscillation (ENSO) years.

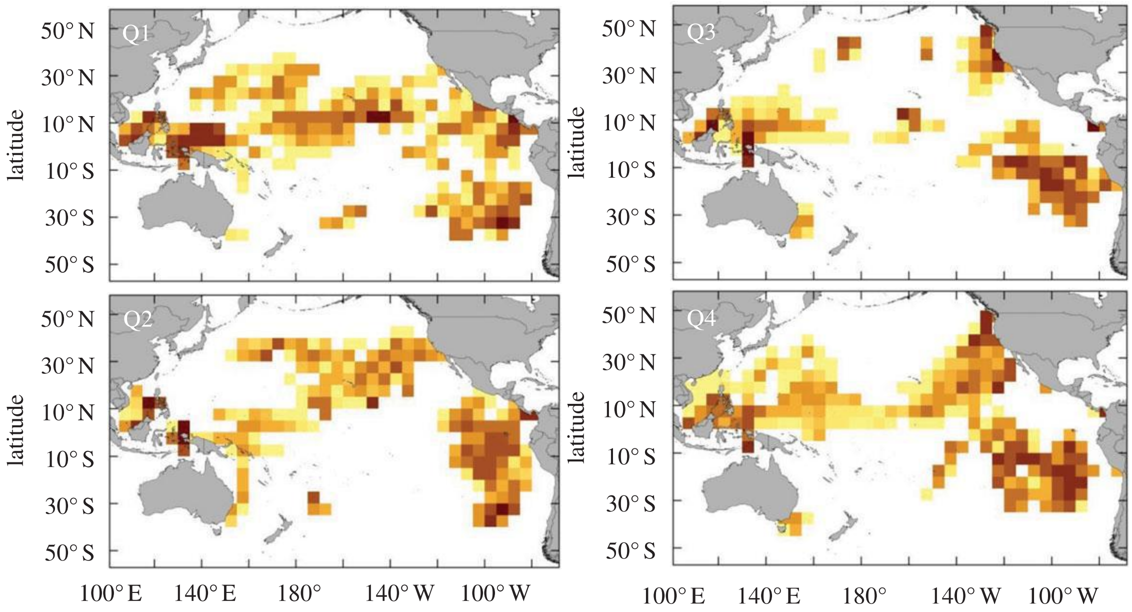


Figure 8-11. Relative use-intensity distributions for leatherback turtles in the Pacific Ocean within 58 x 58 grid cells. Values represent the proportion of all time-weighted and population-normalised positions by annual quarter. Image adapted from: Roe et al. 2014.

8.1.2 Artisanal

Artisanal fisheries, or small-scale fisheries, use a wide range of fishing methods including set and drift nets, pound nets, trawls and seines, surface, mid-water or demersal gear, longlines, and traps (Lewison 2013). Despite being described as small-scale, SSF or artisanal fleet sizes can be vast, with many thousands of vessels operating in a country or region (e.g. Alfaro-Shigueto et al. 2010). These fleets are often spread along long stretches of coastline, operating out of remote coastal communities. The fleets themselves are often dynamic, switching between gear types throughout the year to target seasonally abundant species. The fishing communities are often economically and politically marginalised, which typically means that few by-catch reduction measures and limited enforcement of existing by-catch mitigation measures exist (Lewison 2013). In addition, by-catch monitoring and management are often hard to assess due to the nature of the fisheries themselves, i.e. widespread effort, remote landing sites, political and economic marginalisation and customary practices (Chuenpagdee et al. 2006).

Research in recent years has shown that artisanal fisheries can have high and likely unsustainable levels of sea turtle by-catch (Lewison and Crowder 2007, Peckham et al. 2007, Gilman 2009, Alfaro-Shigueto et al. 2011). Estimates from some fisheries suggest that the amount of sea turtle by-catch may be comparable to by-catch levels in industrial fleets (Lewison and Crowder 2007). For example, Peckham et al. (2007) estimated annual mortality of loggerhead turtles off Baja California, and determined that minimum annual by-catch mortality in two fleets they investigated exceeded 1,000 loggerheads/year, rivalling that of ocean-wide industrial-scale fisheries and threatening the persistence of the Baja loggerhead population. Green turtle strandings along the Baja California Sur coastline were also deemed to have been the result of small-scale gillnet fisheries (Seminoff et al. 2003b), and green turtles were also taken in artisanal fisheries in Panama and Chile (Seminoff et al. 2007).

By-catch occurs in many of the different gear types employed by artisanal fisheries, including longlines, demersal gillnets, driftnets, pound nets and trawls (Arauz et al. 1998, Peckham et al. 2007, Gilman 2009, Alfaro-Shigueto et al. 2011). Hawksbill turtles have been reported in the coastal gillnet fisheries of Peru (Alfaro-Shigueto et al. 2010, Pingo et al. 2017). These were mostly juvenile sized turtles, although adult sized turtles have stranded on Peruvian shores.

For leatherbacks in the eastern Pacific, Laúd OPO (2020) assessed by-catch data from on-board observers (e.g. Instituto del Fomento Pesquero in Chile) as well as rapid by-catch assessments (e.g. ProDelphinus29,30, ACOREMA in Perú). They then used mortality rates of leatherbacks (i.e. the percentage of by-catch interactions that result in mortality) as 5% for longlines and 33% for nets. The by-catch data predominantly came from small-scale, national-level fisheries, specifically net gear. Laúd OPO (2020) indicated that very little information was available on leatherback by-catch interactions with industrial-scale fishing gear (e.g. longlines targeting tuna and other pelagic species). A minimum of 440 leatherback turtles (mostly adults, and some likely subadults) were caught by fisheries in the eastern Pacific since 2012 (~97 turtles per year), of which 132 turtles were caught in longlines, 176 in gillnets, 100 in driftnets and 32 in other fishing gear or in non-specified gear.

Unfortunately, for much of the Pacific Ocean range, by-catch from artisanal fisheries remains unquantified. However, it is likely that by-catch is greatest in those countries that utilise gillnets and longlines, while still being considered artisanal or small-scale. For instance, in the Philippines by-catch of turtles is common, but remains unquantified. In Hinatuan Bay, fronting the Pacific, Bryne and Hines (2005) reported 39 by-catch events between 1997 and 2002: 15 hawksbill, 10 olive ridley and 14 green; all juveniles sized turtles). Loggerheads have also been caught in Philippine artisanal fisheries, but only a handful of records exist (Sagun et al. 2005). In West Papua, Indonesia, although fisheries-related mortality of turtles has not been quantified, communities along the north coast and north islands of Papua reported dead leatherbacks entangled in fishing nets and marine debris (Hitipuew et al. 2007).

This lack of knowledge on the impacts of artisanal fisheries across the Pacific Ocean is a glaring gap in our understanding of total mortality in sea turtle populations.

8.2 Community consumption

Turtle populations have declined dramatically in the Pacific islands, and harvest by humans is a serious problem, particularly for green turtles that have historically been used for food (NMFS and USFWS 1998).

In the central Pacific, data is sparse on turtle and egg harvests due to a number of reasons including: a) some countries allow turtle harvests, but documentation of these is inconsistent or unrecorded; b) illegal harvests are unreported; and c) the distribution of small numbers of turtles over vast oceanic expanses and small, isolated atolls and islands precludes standardised and systematic counting. In the Northern Mariana Islands, Summers et al. (2018) reported that 32% of nesters were collected on Saipan, reducing the annual population growth rate from 11.4% to 7.4%. Maison et al. (2010) also document cases of nest collection on Tinian and Rota. In American Samoa the harvest of any turtles and eggs encountered is a major threat to turtle recovery (NMFS and USFWS 1998). Maison et al. (2010) also indicate that there have been uncontrolled, long-term harvests of eggs and females in the Federated States of Micronesia that is likely to have had an impact on current population numbers. In the Republic of the Marshall Islands, turtles have long been a food source and have played an important cultural role in the lives of inhabitants. Egg collecting and take of turtles while they are onshore is prohibited at all times, but current levels of exploitation are unknown (Maison et al. 2010). In Palau, green sea turtles are taken for food and hawksbill turtles are taken to support a tradition of gift exchanges of *toluk* (Pilcher, pers. obs.), despite traditional closures and moratoriums. Current levels of take are undocumented. At Tongareva, Rakahanga, Manihiki and Palmerston, and probably at other atolls, sea turtles are occasionally killed and eaten, but the true level of direct take remains unclear for the Cook Islands (White 2012). Batibasaga et al. (2006) reported that an estimated 400−500 green turtles were killed in Fiji each year, and that >90% of these were juvenile-sized. In Tonga it is estimated ~100 turtles are harvested annually, but eggs are not collected (K. Stone, pers. comm). In French Polynesia, despite legislation protecting turtles, adult green turtles have been harvested on Maupiti Island and Tupai Atoll (Leeward Islands) and also at Pukapuka Island (Maison et al. 2010). There are no estimates or reports of adult or egg harvests for Kiribati, Nauru, Niue, the Pitcairn Islands, Tokelau, Tuvalu and Wallis and Futuna.

The Hawaiian green turtle population was subjected to extensive human exploitation at foraging and nesting grounds from the mid-1800s until the early 1960s (Balazs and Chaloupka 2004). Since enactment of state and federal Endangered Species Act (ESA) protections in 1974 and 1978, respectively, the nesting population at French Frigate Shoals has exhibited a consistent upward trend over the past thirty years with an estimated annual growth rate of 5.7% (Chaloupka et al. 2008).

In Japan, green turtles at the Ogasawara Islands have been harvested locally for their meat since the 1830s (Ishizaki 2007). Today, sea turtle harvest in the Ogasawara Islands continues under strict governmental regulation with a harvest limit of 135 mature turtles per year (Ishizaki 2007).

In Australia, indigenous peoples with a recognised Native Title right can legitimately hunt marine turtles for communal, non-commercial purposes. Most indigenous turtle hunters appear to preferentially hunt green turtles. Examination of figures presented by Limpus (2007a) suggests there may be an annual take of some 5,000 to 10,000 green turtles harvested each year by indigenous people across Queensland and the Torres Strait. Records indicate that there is a bias towards collection of female turtles (Limpus 2007a), and that many of these originate from the sGBR genetic stock. Loggerheads are taken occasionally for food in Torres Strait and the far northern Great Barrier Reef (Limpus 2007c), and >10% of loggerhead tag recoveries had been captured for food. However, Limpus (2007c) also indicates that <1 tagged turtle per year is reported as eaten by indigenous people. Indigenous harvest of flatback turtle eggs is widespread along much of the northern Australian coast but the size of this harvest has been largely unquantified. This is similarly the case for loggerhead eggs (Limpus (2007c). There is no apparent preference for turtle eggs of a particular species (Kennett et al. 1998). Although the meat of the flatback turtle is less preferred than that of other turtle species especially the green turtle (Limpus 2007a); there is occasional take of flatback turtles for food from the Eastern Gulf of Carpentaria and Northern Territory stocks (Limpus 2007e). At nesting beaches, an average of 40 adult flatback females are taken annually in aboriginal harvest (Limpus 2007e). Being one of the main nesting turtle species in this region, olive ridley turtles are part of the traditional harvest (Kennett et al. 1998). Similarly, olive ridley eggs are collected by indigenous communities along the beaches in western Cape York Peninsula, Queensland (Limpus 2007f). On Crab Island, Southerland and Southerland (2003) documented egg harvests by traditional land owners and estimated these represented 0.44% of all clutches deposited during a six-week period (during which 4,234 nesting events were counted). Overall however, Limpus (2007f) concluded that based on available data, it was not possible to quantify the present magnitude of cumulative mortality from the wide array of anthropogenic sources impacting olive ridley turtles in Australia. Hawksbill turtles have also been hunted for centuries in Australia by indigenous people for tortoiseshell to manufacture items of everyday use and for ceremonial use. Its eggs and meat were eaten for food. The annual indigenous harvest for all of Queensland has not been quantified, but is probably of the order of 50–100 hawksbills plus probably hundreds of clutches of eggs (Limpus 2007b and references therein). In Queensland, reports suggest that less than one leatherback turtle per decade has been captured and killed by indigenous peoples (Limpus 2007d).

Limpus (2007a) also suggests ~500 to ~1,000 green turtles from the sGBR stock are also taken in neighbouring countries (Papua New Guinea, New Caledonia, Fiji, Vanuatu). Limpus (2007a) also indicates takes of ~1,000 turtles per year in northern Torres Strait/PNG waters of which 90% of turtles originate from Australian rookeries. Batibasaga et al. (2006) echo these suggestions, with several dozen loggerheads captured and sold in Fiji in the 1990s, believed to belong the Australian stocks.

In Papua New Guinea, Opu (2018) assessed domestic consumption of turtles via market surveys. A total of 37 reliable key stakeholders were interviewed and the results indicated that 4,760 turtles in 2016 and 5,320 turtles in 2017 were landed in markets for domestic consumption. Provinces that landed the most catches were Manus, Milne Bay and Western Provinces, and species composition showed a predominance of green, followed by hawksbill and lastly, leatherback turtles. No loggerheads or olive ridleys were recorded. These harvests included 1,200 green turtles harvested in Manus and another 1,350 in Daru alone. Opu (2018) suggested that these numbers were an underestimate given the limitations of the survey, and because many landed turtles likely never make it to the markets, were used for personal consumption or in the barter trade.

Votu et al. (2019) provide a recent update on turtle harvests in the Solomon Islands. They estimated that 9,473 turtles were harvested each year by spear fishers in Solomon Islands, with 95% confidence intervals of 5,063 to 22,423 turtles. Of these, 1,107 harvested turtles were actually observed by the community monitors. Most of these turtles were green (73.8%) and hawksbill (25.7%), with very few olive ridley turtles (0.5%). Juvenile turtles comprised 88.7% of green and 76.4% of hawksbill captures. Green and hawksbill turtles were most commonly used for subsistence purposes (88.2% and 81.6% respectively) and were most likely to be consumed by the family of the fisher that captured the turtle(s). Hawksbill turtle products were far more likely to be illegally sold (32.3%) than green turtle products (12.1%). Importantly to note in this summary is the high range between confidence intervals, which is understandable given the survey methods and the challenges in surveying communities spread across hundreds of islands. Given this reported annual harvest is probably an underestimate, it is quite likely that ~10,000 to ~20,000 turtles are taken each year in the Solomon Islands.

In Vanuatu there is a strong programme of local turtle monitors that aid in protecting turtles and convincing local communities to participate in turtle conservation efforts (Hickey and Petro 2006). They estimate that turtle take in the past may have been in the region of 1,500 turtles per year and they suggest that much of this harvest has since ceased. In a recent questionnaire survey for dugongs (Shaw, unpublished data), respondents were also asked about turtle captures; 21% of 435 respondents (91 people) indicated that they had caught turtles in the preceding 12 months. The average number of turtles per person was 1.4 ± 0.77, and was similar to catches in the previous five years and in their lifetimes. 59% of respondents indicated they would eat the turtles caught purposefully, 35% indicated they would use the meat in trade, and 12% indicated they would sell the meat. If the turtle was caught accidentally, 34% of respondents still indicated they would keep it for food, but 48% indicated they would release it alive. While this sample is not representative of the island chain as a whole, it does indicate that turtle captures continue to this day, and it is unknown what level of harvest of turtles and eggs may currently remain in Vanuatu.

In New Caledonia there is a legal, permitted take of green turtles that has averaged ~150−200 turtles per year (Farman 2020). In the northern province, the take of turtles is restricted to turtles <100 cm curved carapace length. No hawksbill turtles are targeted in this traditional take.

In the east Pacific, Ramirez-Gallego and Barrientos-Muńoz report on illegal hawksbill trafficking in Colombia, in which they estimated 1,800–2,800 items per year were offered for sale between 2008 and 2012, of which the majority of items (96.2%) were articles of jewellery. Sea turtle meat was only detected in two restaurants during the five-year study. Large-scale egg use historically occurred at arribada beaches in Mexico, concurrent with the use of adult turtles at these beaches (Cliffton et al. 1982). The high level of adult mortality is believed to be the reason why rapid and large nesting population declines occurred in Mexico (Abreu-Grobois and Plotkin 2008; Cornelius et al. 2007). Alvarado et al. (2001) indicate that nesting turtles are still taken at a rate of ~8−10 per year and up to 10% of clutches may also be collected at Michoacán, Mexico.

The collapse of the east Pacific leatherback stock has been attributed primarily to the illegal collection of >90% of eggs on nesting beaches for >20 years (Santandrián-Tomillo et al. 2008, 2017, Laúd OPO 2020), and while this has ceased at the protected areas, egg collection occasionally still occurs on secondary beaches.

Mancini and Kock (2009) found the carcasses of 1,014 sea turtles; the meat of 461 of these turtles (45.5%) had been consumed between March 2006 and February 2008. 77% of total consumed turtles were green sea turtles. They noted that while trade and consumption had decreased in recent years, there were still several places that supply sea turtle meat to local, regional, and sometimes even international markets. Seminoff et al. (2003b) also reported human consumption of green sea turtles in Baja California Sur and estimated 6.5% of the turtles they had tagged had been consumed. These turtles were 70 cm−75 cm in carapace length, suggesting they were sub-adult turtles. As recently as 2002, Nichols et al. (2002) described a black market that killed tens of thousands of green turtles each year in Baja California Sur. Historically, coastal regions in Peru and mainland Ecuador also had high levels of direct take (de Paz et al. 2002), although it is unclear to what degree this level of take continues.

8.3 Light impacts

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 and 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 mm−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 sea 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 unnecessary energy expenditure. In extreme cases hatchlings fail to reach the ocean.

Our understanding of impacts of anthropogenic light on sea turtles comes from only a handful of studies in the east and west Pacific. There are no empirical studies of lighting impacts in Pacific island nations, and the few studies that do exist are conducted mostly as academic exercises. Artificial light has been shown to disrupt natural night horizons in proximity to nesting beaches (Limpus and Kamrowski 2013). Lighting was found to impact flatback turtle orientation at Curtis Island, Australia, where multiple large industries are located. Hatchlings displayed reduced 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 on Curtis Island, 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". Similarly, at Playa Cabuyal, a remote, unprotected beach in the Guanacaste province of Costa Rica, beach illumination was not found to have any impact on sea turtle nesting success for green, olive ridley or leatherback turtles (Silowsky 2018). The presence of light at one private residence was found to have a beach-wide impact on the nesting success rate of black sea turtles ($p = 0.013$) irrespective of proximity to the sectors in which increased light intensity was detected, suggesting that visibility of the light fixture itself has a greater impact on sea turtle nesting success than light intensity at the site of the nest (Silowsky 2018).

This 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 night-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, presently at the population population level this does not appear to be a problem. Indeed, at all major global nesting sites where lighting has been 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 sea turtle nesting sites in Australia appeared minimally affected by light pollution exposure.

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 ± 5 min) 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).

A novel look at lighting impacts was conducted by Gless et al. (2008), who investigated impacts of lighting from longline vessels on leatherback turtles. Including previous results with loggerhead turtles, they concluded that there was no convincing evidence that sea turtles were attracted to the longlines by lights. However, sea turtles are often entangled in gillnets and Wang et al. (2010) investigated the use of lights on these nets to mitigate impacts to turtles. Their results indicated that nets illuminated by LED lights significantly reduced mean sea turtle catch rates by 40% while having negligible impacts on target catch and catch value. Similarly, nets illuminated by chemical lightsticks also significantly reduced mean sea turtle catch rates by 60% while having no significant impact on target catch and catch value. These results illustrate the potential for modifying fishing gear with visual deterrents to effectively reduce sea turtle catch rates (Wang et al. 2010).

At present (with the exception of Australia) there is a lack of knowledge on the overall impacts of light pollution on sea turtles in the Pacific Ocean region, and this warrants further investigation.

8.4 Predation

There are scarce records of at-sea predation and even fewer dedicated studies. This is because recording mortality presents severe challenges given the diffuse at-sea distribution of turtles and the potential for recording infrequent events. Most information on sea turtle predators comes from anecdotal accounts and stomach contents of predators, and a handful of studies related more to the predators themselves than to sea turtles. The majority of our understanding of predation on sea turtles relates to predation on eggs and hatchlings on beaches, where these are recorded by field teams.

At sea, the most common predators of adult sea turtles are sharks (Marquez 1990). Bull (*Carcharhinus leucas*), white (*Carcharodon carcharias*), and tiger (*Galeocerdo cuvier*) sharks are known predators of adult sea turtles (Witzell 1987, Cliff and Dudley 1991, Fergusson et al. 2000). For instance, Work et al. (2015) indicated tiger sharks accounted for ~20%–25% of all trauma cases examined in the Hawaiian islands. Summers et al. (2018) also recorded shark predation on turtles in Tinian and Saipan (CNMI) as did SPC (1979) in Kiribati. These accounts show that adults of all sea turtle species are susceptible to predators, although predation generally is rare and few predators take adult sea turtles frequently (Heithaus et al. 2008). Kauffman (1950) found hawksbill turtles in 11 of 22 tiger shark stomachs off the Philippines; green turtles were found in two stomachs. Similarly, turtles were found in 45% of the 84 sharks captured in Australia with food in their stomachs and in over 70% of the stomachs of sharks in the largest size classes (Simpfendorfer et al. 2001). Turtles were less common prey in Hawaii, where green turtle remains were found in 15% of 135 tiger sharks >300 cm total length and 7% of 118 sharks 200 cm–300 cm total length (Lowe et al. 1996). Engbring (1992) recorded instances of grey reef shark predation on juvenile leatherback turtles in Palau, while Adnyana (2006) recorded ghost crab predation of leatherback hatchlings in Indonesia. Neither of these provided any quantitative rates of predation. Balazs (1985) suggested the presence of large numbers of large turtles on Johnston Atoll could be explained by high rates of predation on smaller turtles. Salt-water crocodiles have been reported as predators of adult turtles at sea (e.g. Quinn et al. 1983, Quinn and Kojlis 1985, Kinch 2006, Southerland and Southerland 2003). Witzell (1981) also recorded predation on green turtles by a grouper (*Promicrops lanceolatus*) in the Kingdom of Tonga. Killer whales (*Orcinus orca*) are another predator of adult turtles. They have been observed killing leatherback turtles and may prey upon olive ridley and green turtles (Pitman and Dutton 2004, Fertl and Fulling 2007). There are even fewer records of juvenile-sized turtle mortality but it is believed this would be higher given their smaller size, as they are often ingested whole and mortality is unrecorded (Heithaus et al. 2008). Gyuris (1984) found mortality of hatchlings in nearshore waters to be ~20% within the first few hours after entering the sea. This work was reinforced by studies by Pilcher et al (2000), who found predation rates of ~20% in random (in space and time) emergences vs. ~60% predation in controlled releases from the same location at the same time daily. This highlights how hatchling release programmes need to align with natural biology of sea turtles to ensure adequate survival.

Predation is generally not quantified across Pacific island nations given the wide distribution and diffuse nesting, but nest predation has been documented on most of the islands. NMFS and USFWS (1998) indicate that the loss of eggs could be a severe problem in some areas with predators, including domestic animals such as cats, dogs and pigs; as well as wild species such as rats, mongoose, birds, monitor lizards, snakes, and crabs, ants and other invertebrates. A major threat to nesting turtles in the Republic of the Marshall Islands comes from rat predation on nests. The feral predators have been documented to consume large numbers of eggs on Erikub and Bikar Atolls (NMFS and USFWS 1998). Common ship rats are likely an additional problem on many of the smaller Pacific islands, such as was documented on Tetiaroa in French Polynesia (Touron et al. 2019). White (2012) indicated terrestrial predators were not of major concern at Palmerston Atoll, but indicated that across the Cook Islands potential terrestrial predators included rats (*Rattus exulans; R. rattus*), crabs, particularly the butcher landcrab or "tupa" (*Cardisoma carnifera*). Green (1978) reported predation by frigate birds, ghost crabs, hermit crabs and night herons on the Galapagos islands, and noted that feral cats and feral pigs, both introduced by humans, were causing extensive damage to eggs and hatchlings. In addition, he reported that the scarab beetle, *Trox suberosus*, was also a significant egg predator. This was subsequently recorded and quantified by Zarate (2013). Vaughn (1981) indicated that iguanas were a known predator of turtle eggs in the Solomon Islands, but reported that only 0.03% of the nests they inspected appeared to have been depredated by iguanas (likely *Varanus* spp.). He also recorded ghost crabs in 64 of 341 hawksbill nests, with 12% of nests completely destroyed. In addition, he indicated pigs and dogs were also of concern (Vaughn 1981). Summers et al. (2018) recorded monitor lizard crab and ant predation in nests on Rota Island (CNMI). In that survey, crabs depredated 5% of nests, destroying a mean of 3 eggs per nest; ants impacted 3% of nests, consuming a mean of 16 eggs per nest; and monitor lizards depredated 2% of nests, consuming a mean of 17 eggs per nest. Ghost crabs (*Ocypode* sp.) were also implicated in nest predation in Kiribati (SPC 1979, Balazs 1975) although rates were not quantified. In Tokelau, identified predators include hermit crabs, ghost crabs, Polynesian rats, frigate birds (*Fregata ariel, F. minor*), and reef herons (*Egretta sacra*; Balazs 1983).

In Yap State in the FSM, nest predation by ghost crabs was reported to be a substantial problem in the 1990s on Olimarao Island, as well as a potential threat on Falipi Island, both within the Olimarao Atoll (NMFS and USFWS, 1998). Ghost crabs were identified as nest predators on Gielop Island within the Ulithi Atoll (NMFS and USFWS, 1998). Monitor lizards depredated 23 of 28 (83%) marked green turtle nests on Loosiep Island (Cruce 2009). Wild pigs were observed to dig into nests that had been depredated by monitor lizards (Cruce 2009). Monitor lizards have also been documented as a predator of green turtle nests on Sorol Island (Sorol Atoll) in Yap State (Buden 2013).

Polynesian rat predation on nests was reported as a major threat to green turtles in the Republic of the Marshall Islands (NMFS and USFWS 1998), where Polynesian rat predation at Bikar was severe. Predators have also been documented to consume large numbers of eggs on Erikub Atoll (NMFS and USFWS 1998). On Guam, nest predation by monitor lizards, wild pigs, rats and crabs is a problem (Cummings 2002). Nest predation by wild pigs and monitor lizards is also a threat to green turtles in Palau however, the extent of the problem is unknown. In the Solomon Islands, nest predation by crabs, megapodes and iguanas is high in the Arnavon Marine Conservation Area, a major green turtle nesting beach in the Solomon Islands (Wilson et al. 2004). Predation of turtle nests and hatchlings by dogs and feral pigs has been identified as a problem on Warmandi beach in Papua, Indonesia (Maturbongs 2000). Within the Ogasawara Islands of Japan, ghost crabs (*Ocypode cordimana*) were documented to have completely depredated 14 to 23 percent of study nests each season from 1991 through 1994 (Suganuma et al. 1996). Ghost crabs were also a major predator at Ishigakihima (Okuyama et al. 2020).

At French Frigate Shoals in the Northwest Hawaiian Islands (NWHI), there is virtually no predation on eggs and, while numerous seabirds including frigate birds inhabit the area, no evidence of avian predation on hatchlings has been detected (NMFS and USFWS 1998). Predation on hatchlings at French Frigate Shoals when they pass through the nearshore waters en-route to the open ocean was also low (Balazs 1980).

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 while ashore for egg laying. Sutherland and 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 and 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 and 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. Lei and Booth (2017) identified the fox (*Vulpes vulpes*) and goanna (*Varanus* spp.) as key predators along the Queensland coast, and concluded that large male yellow-spotted goannas were the major predator of sea turtle nests at the Wreck Rock beach with nest visitation rates of 15% to 31% compared to 2% to 4% by foxes.

Introduced mammals are often opportunistic predators upon turtle eggs and include dogs (*Canus lupus*), feral pigs (*Sus scrofa*) and foxes (*Vulpes vulpes*). Introduced predators have caused almost total destruction of eggs at some rookeries (eg. areas in Western Cape York are thought to have had predation levels of ~90% over the last 30 years; Limpus 2007e). 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 2007f).

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 Islands, Hopley (2008) reported that predation of the hatchlings is high, especially by rufous herons, and only 6.7% of hatchlings may reach the sea. Nocturnal avian predation was also recorded on Bare Sand island (Giuliano et al. 2015). In contrast, Niethammer et al. (1992) determined that frigate birds were not a factor in hatchling predation in French Frigate Shoals.

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 1 m to >6 m, with numerous medium to large crocodiles (>3.5 m) observed. The amount of predation witnessed indicates that crocodiles are one of the major predators of hatchlings on the island (Leis 2008). Sotherland and Southerland (2003) also reported crocodile predation at a minimum rate of one adult flatback per week.

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 and Tiwari (2007) found pig predation rates of 29.3% in Jamursba Medi along with a lower predation rate by dogs. 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%.

In the east Pacific, dogs have been implicated in nest mortality in Panama (Meylan et al. 2013), and in 2003 54.3% of leatherback nests were disturbed by dogs. Another 28% of hawksbill nests were also disturbed by dogs. At four nesting beaches on the Osa Peninsula, Costa Rica, nest predation rates varied from 28% to 54% (Drake 1996). Dogs were responsible for 49% of the depredated nests (16% of the total nests recorded), humans for 42% (12% of the total) and coatis (*Nasua narica*) for 2% (less than 1% of total) on all four beaches (Drake 1996). Eggs of Pacific green turtles in Galápagos were preyed on by the native scarabeid beetle (*Omorgus suberosus*), the feral pig (*Sus scrofa*), and native ghost crabs (*Ocypode gaudichaudii*). Beetles affected 66.2% of nests and feral pigs destroyed 7.4% of the nests (Zarate et al. 2006).

8.5 Climate change impacts

Climate impacts can have multiple effects on sea turtles (e.g. Witt et al. 2010, Fuentes et al. 2013, Santandrián-Tomillo et al. 2012). 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).

In 2018 Hurricane Walaka virtually erased East Island in the French Frigate Shoals (Northern Hawaii Islands) and since that time the island has barely recovered. It is unclear what the long-term impacts will be on the Hawaiian green turtle stock, of which over 50% used to nest on East Island (Chaloupka and Balazs 2007). Baker et al. (2006) predicted that sea level rise would severely impact the French Frigate Shoals, however they suggested sea level rise would be implicated rather than severe storms. Ongoing studies at French Frigate Shoals in the coming two to three years will be crucial in our understanding of how turtle populations cope with the loss of major portions of their nesting habitat. Contrastingly, storm frequency along the Australian coast was projected to decrease (Fuentes and Abbs 2010) adding resilience to those 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 and be lost to turtles.

Extreme weather patterns might also more profoundly impact sea turtles during ENSO events. Recent investigations indicated that reproductive success declined for central eastern Pacific leatherback sea turtles, and suggested these events could become more frequent in the future (Santandrián-Tomillo et al. 2020). These investigations did not show such declines in green turtles, however abundance of green turtles did decline, likely related to decreased ocean productivity. Erosion from major storm events is also 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.

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. Recent studies point towards all Pacific island countries and territories being far more vulnerable to climate change via sea level rise than previously understood (Andrew et al. 2019). In this scenario, the impacts to sea turtles will likely be an afterthought when considering the human dimension, as this new study shows about 97% of the populations of the 21

Pacific island countries and territories (excluding Papua New Guinea) live within 10 km of the coast, and 90% live within 5 km of the ocean (Figure 8-12).

However, several studies suggest that some atolls and islands are actually growing (Hollingsworth 2020), and may be less sensitive to impacts of sea level rise. For instance, Jeh Island, in the Marshall islands, has increased in size by 13% since the 1940s (Ford et al. 2020). Thus caution should be taken in predicting a blanket loss of sea turtle nesting habitat across all Pacific islands.

Global warming patterns may also impact sea turtles in the Pacific. Feminisation of stocks is of concern, and a recent study pointed to 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 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 feminisation of this population was possible in the near future. Laloë et al. (2015) also detected female biased production of green and hawksbill turtles in the Caribbean 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 34°C in the Northern Mariana Islands (CNMI), and demonstrated that these impacts, in combination with egg poaching, could decrease nester abundance.

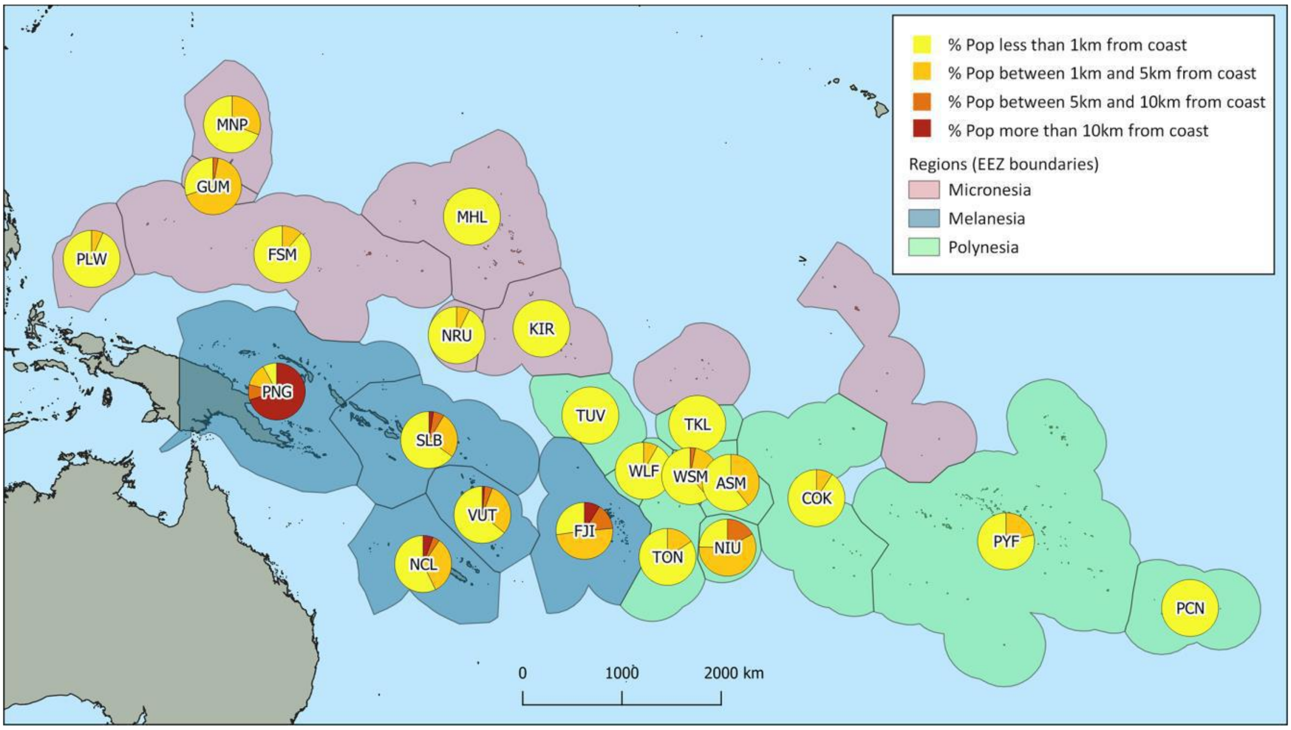


Figure 8-12: Proportions of households within 1, 5 and 10 km from the coast in the Pacific. Source: Andrew et al. 2019.

However, this may not be applicable to all species, as Howard et al. (2015) 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. (2015) also found an anomalous production of male flatback turtle hatchlings from Cape Domett (Western Australia).

At present, most research on impacts of temperature has focused on nesting turtles and developing embryos, given the ease of access. Chaloupka et al. (2008) 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 and 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 northern GBR and key to green turtle populations in the Pacific, 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 northern GBR by Jensen et al. (2018). 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 ENSO events would have important influences on the breeding behaviour of turtles, and research by Santandrián-Tomillo et al. 2020 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 then 5 m 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 Pacific.

8.6 Other Impacts

International tortoiseshell trade – Limpus (2007b) indicated that in the 17th and 18th centuries the trade in tortoise shell from Australia accounted for the loss of ~1,000 hawksbill turtles per year from the Northern Territory and another ~1,000 hawksbills from Queensland. However, that trade ceased during the 1930s and became illegal with the protection of *E. imbricata* in Queensland in 1968 (Limpus 2007b). While the legal trade in tortoise shell ended in 1991, there is continued illegal trade in hawksbill shells, and these are regularly found in shops in both source and destination countries. The continued trade in hawksbill shell warrants further investigation. Tortoiseshell trade continues to be an issue in the Solomon Islands, where Vuto et al. (2019) reported trade in hawksbill shell in 3 of the 10 communities surveyed, but which was only a common practice in Wagina. They report that in the Wagina community, the shells of 87.5% of hawksbill turtles harvested were sold to local buyers, who then on-sold to Asian buyers in Honiara. In the past the export of tortoiseshell from the Solomon Islands was among the ten highest in the world (Miller et al. 2019) In Papua New Guinea, Kinch and Burgess (2009) noted that the trade in hawksbill turtles was ongoing in coastal towns throughout Papua New Guinea. The trade was mainly in the form of tortoiseshell items supplying a domestic market, and potentially targeting international tourists as buyers. No estimate of total trade was provided, but they recorded ~1,400 tortoiseshell items at markets across the country in 2007. It is likely there is still a substantial illegal trade in hawksbill shells from various locations in the Pacific that warrants further study.

Ghost nets – In addition to commercial fisheries, lost fishing equipment from commercial fisheries poses a significant threat to sea turtles through entanglement and ingestion. Ghost nets continue to accumulate on Australia's northern shore due to both legal and illegal fishing; over 13,000 nets have been removed since 2005 (Wilcox et al. 2014). The problem is not as severe on the eastern coast of Australia but data for the Gulf of Carpentaria and the fact these nets have the potential to enter the Pacific Ocean (Wilcox et al. 2012) suggests ghost nets might be a substantive source of mortality. Quantified flatback mortality in ghost nets from 70 km of coastline (approx. 3.5% of the Gulf of Carpentaria coastline) averaged 2.3 turtles during a two-month period (Roeger et al. 2007). These limited data suggest that along the entire coastline, an estimated mid to high hundreds of flatback turtles die annually in these nets. Following Cyclone Abigail in February 2001, it is estimated that over 4,000 nets washed ashore containing in excess of 400 turtles along the eastern Gulf of Carpentaria (Limpus 2007a). Wilcox et al. (2014) estimated that the total number of turtles caught by the 8,690 ghost nets they sampled along Australia's northern coast was between 4,866 and 14,600, assuming nets drifted for 1 year. Wilcox et al. (2012) also demonstrated through particle modelling that a proportion of the nets could be entering the Pacific Ocean via the Torres Strait (Figure 8-13).

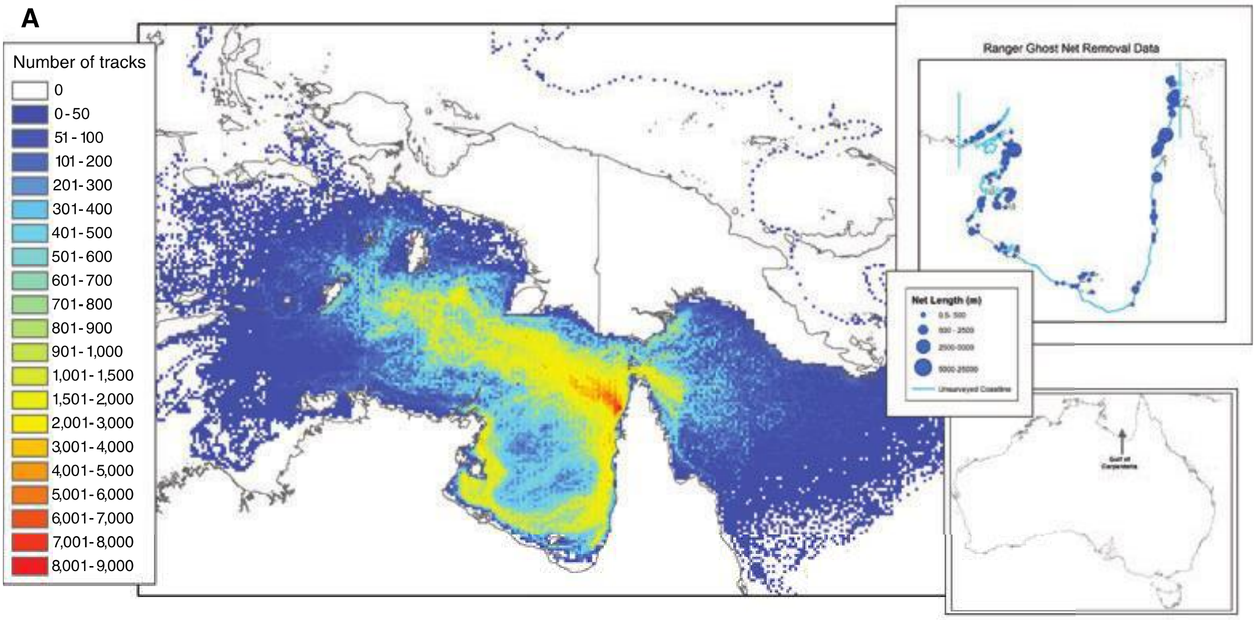


Figure 8-13: Components of a risk model for ghost net impacts to turtles. (A) Potential ghost net tracks based on daily particles releases (1996–2008) and (inset) total length of net found at sites along the Gulf of Carpentaria coastline. Source: Wilcox et al. 2012.

Shark Nets - Limpus (2007a,c) reported that green and loggerhead turtles were the most commonly captured turtle in the Queensland Shark Control Program (QSCP). Limpus (2007c) indicates some ~600 green turtles and 25 loggerhead turtles were likely killed between 1986 and 1992 in the QSCP. Following changes to the QSCP in 1992, Limpus (2007a,c) estimates annual mortality was closer to seven green and three loggerhead turtles per year. Hawksbills are rarely captured in the QSCP. The annual hawksbill mortality rate in the QSCP has been ~0.5 immature turtles per year (Limpus 2007b). Shark nets are not employed elsewhere in the Pacific and this is a localised issue restricted to Australia.

Fish Aggregating Devices (FADs) – Abandoned, Lost or Discarded Fishing Gears (ALDFGs), and particularly Fish Aggregating Devices used in large commercial fisheries, can impact sea turtles that may be attracted to floating objects, perhaps searching for food or shelter. As the FADs usually have webbing hanging below them, the turtle may become entangled in the FAD and if it is not released it may die. Blasi et al. (2016) found that FADs could impact all turtle life stages but that by-catch was more prominent in smaller sea turtles. They also recorded high (19.4%) levels of by-catch in FADs, which is worrying when considering the magnitude of their use in the Pacific. Escalle et al. (2020) describe the increase in use of FADs in the WCPO longline fishery and indicate the number of FADs in use has grown steadily, with an increase of ~30% per year in recent years (from 14,815 in 2016 to 21,080 in 2019). This study also estimated that 42.1% of FADs were lost, 9.4% were retrieved; 7.4% were beached; 20.0% were sunk, stolen or had a malfunctioning buoy; and 21.1% were deactivated by the fishing company and left drifting, unmonitored at sea. That is, in total over 60% of FADs (or ~48,000 FADs between 2016 and 2019) have been lost or abandoned at sea. Density plots of the use of FADs in the WPCO are shown in Figure 8-14, and these are suggested as areas where interactions with sea turtles may occur at similar density levels.

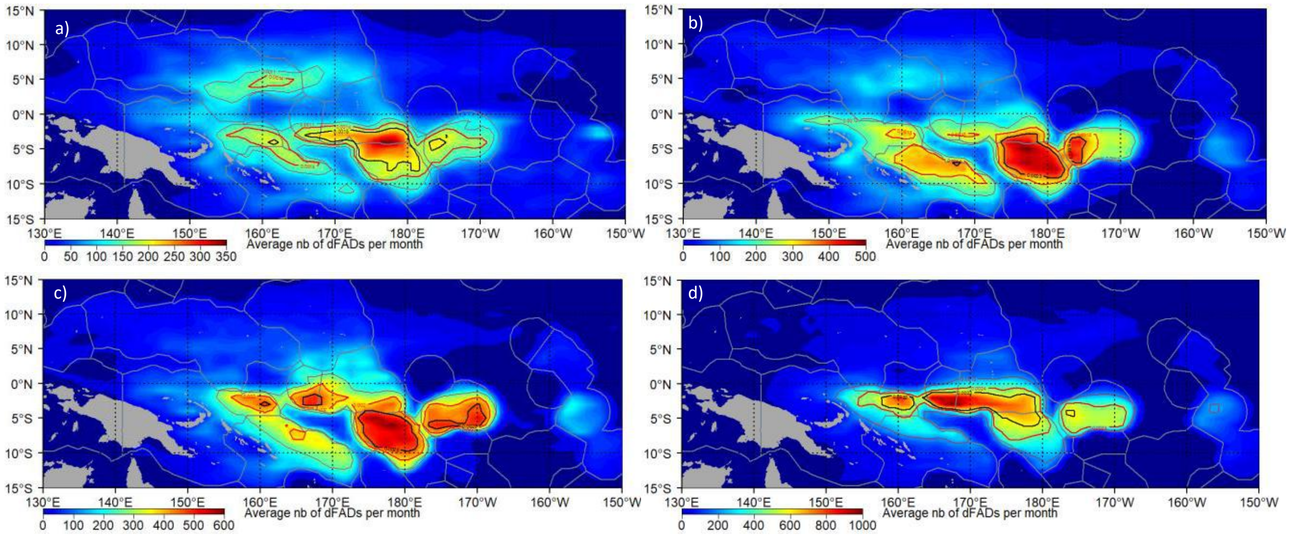


Figure 8-14: Smoothed kernel density of the average number (nb) of FAD satellite buoys transmitting at least once per month and per 1° grid cell during a) 2016, b) 2017, c) 2018, and d) 2019. Red lines correspond to the 95th quantile. Colour scale corresponds to the average number of buoys transmitting per 1° cell per month. Note that the scales are different on each plot. Source: Escalle et al. 2020.

Commercial exploitation – While there is no commercial exploitation of sea turtles across the Pacific today, the historical exploitation in both the east and west Pacific, and in Hawaii, led to significant declines and in some cases, the near extirpation of the species. There was appreciable commercial harvesting of green turtles in Australia until 1959 (Limpus 2007a). Tens of thousands of green sea turtles were collected for this trade and an estimated >18% of the sGBR nesting females were harvested in 1924–1930 alone. While the trade has ceased, there was likely local depletion of populations that are still recovering. Similarly, the olive ridley turtle fishery caused rapid, large declines at arribada beaches in Mexico (Cliffton et al. 1982) where an estimated 75,000 turtles were taken each year for over two decades until 1990 when the fishery closed (Aridjis 1990).

Boat strikes – Limpus (2007a) suggests several hundred sea turtles may be killed annually in Australia due to boat strikes. This is likely less of a problem in the Pacific island nations where occurrence of turtles coincident with vessels is likely a far less frequent event.

9.0 Migrations and Habitat Connectivity

Sea turtles are highly migratory, with foraging areas and nesting beaches that can be thousands of kilometres apart. In the Pacific Ocean, turtles have some of the longest known migratory routes, with loggerheads from Japan migrating across the entire northern Pacific (Figure 9-1), and nesting leatherbacks from West Papua, Indonesia, migrating to waters off Southern California to forage (Figure 9-2).

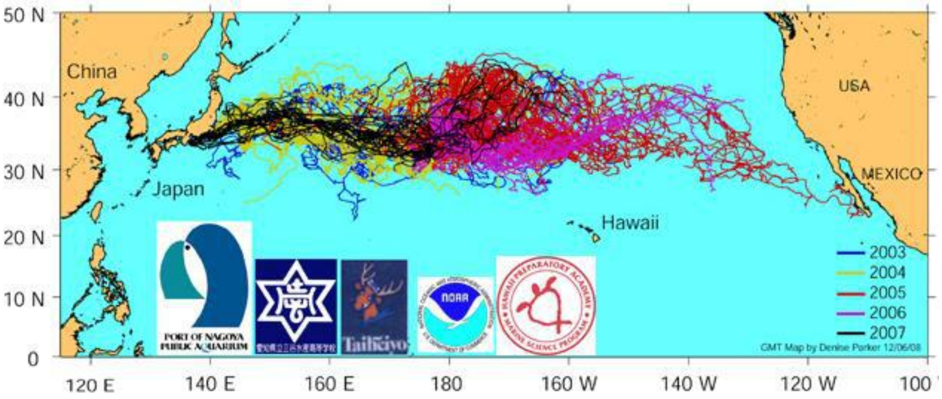


Figure 9-1: Migrations of aquarium-reared loggerhead turtles in the northern Pacific. Image source: NOAA NMFS and HPA.

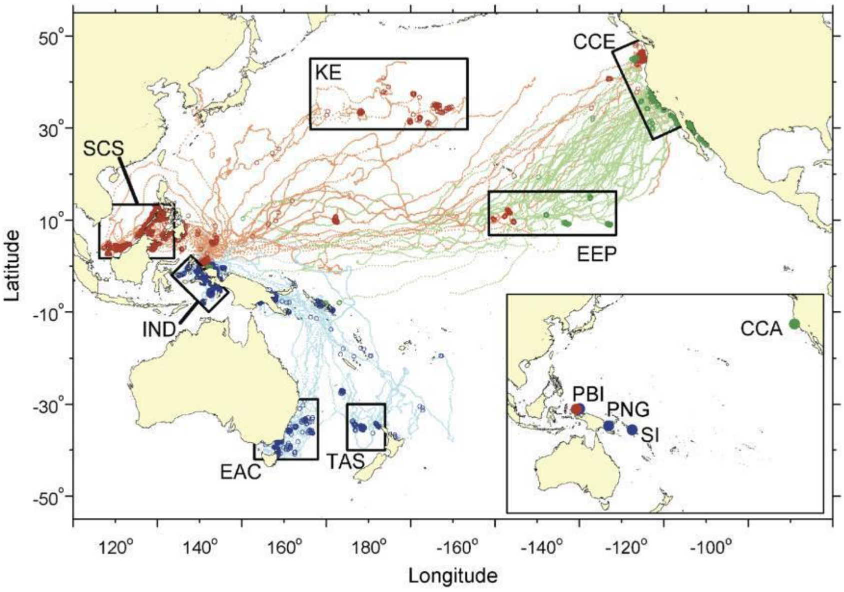


Figure 9-2: Migrations of leatherback turtles in the Pacific. Image source: Benson et al. 2011.

9.1 Green turtle migrations

Migrations of green turtles in the Pacific are substantial, with many turtles undertaking long distance migrations between island nations in the central and south Pacific, and turtles moving off the coasts of the Americas westward into the Pacific. In the eastern Pacific, Galapagos turtles migrated northeast to the American continent, south or west into the Pacific (Figure 9-3; Seminoff et al. 2007). These tracks are also supported by past flipper tag research, that documented 23 tag recoveries from green turtles tagged in the Galapagos Islands between 1970 and 1979 (Green 1984). Three recoveries were from Costa Rica, four from Panama, one from Colombia, five from mainland Ecuador and ten from Peru, with minimum displacement distances ranging from 1,233 km to 2,143 km. Other studies in the eastern Pacific have found less substantive movements (e.g. Blanco et al 2012) with turtles staying close to shore and with only limited north-south movement. Genetic studies also indicated most turtles from Gorgona Island in Colombia also originated from source rookeries located exclusively in the eastern Pacific (Amorocho et al. 2012). Similarly, Hart et al. (2015) documented north-south movements of green turtles along the American continent with little westward deployment, but highlighting the connectivity among green turtle nesting and foraging sites in the eastern Pacific.

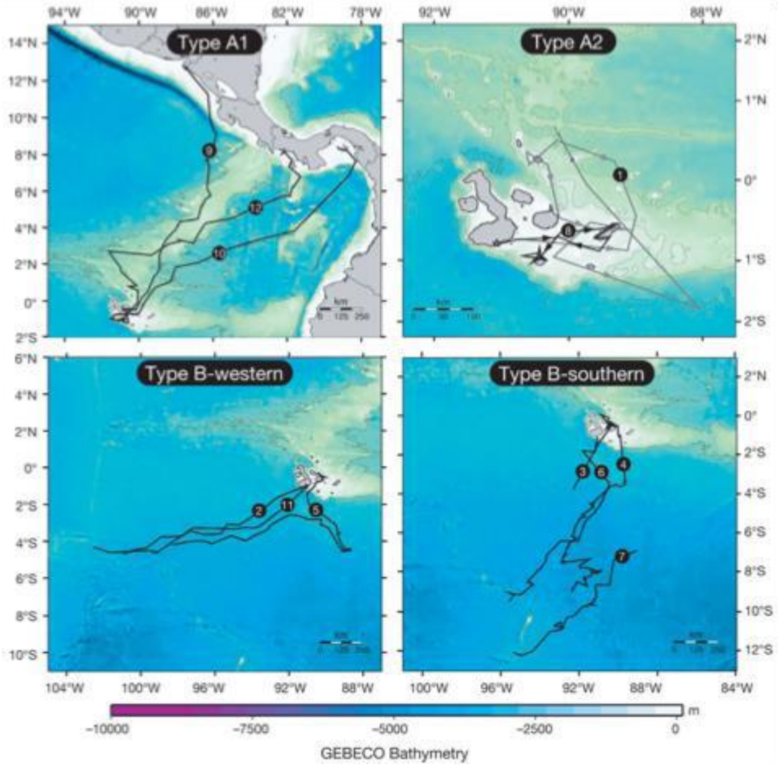


Figure 9-3. Satellite-tracked post-nesting movements of green turtles nesting in the Galápagos. Image source: Seminoff et al. 2007.

In the central south Pacific, migrations of seven post-nesting green turtles at Rose Atoll (American Samoa) between 1993 and 1995 indicated most turtles migrated 1,600 km to foraging areas in Fiji (Figure 9-4; Craig et al. 2004). Additional tag recaptures from other studies indicate a common pattern of turtle movement in the central south Pacific region. In total, the 26 recaptures of primarily post-nesting turtles from French Polynesia, American Samoa and Cook Islands showed a similar course of direction and destination: 96% migrated westward after nesting, with 58% going specifically to Fiji, likely towards Fiji's extensive seagrass and algae pastures that appear to be a significant resource for many green turtles in the region (Craig et al. 2004). In support of the importance of Fiji as a foraging area for green turtles (Piovano et al. 2020), a green turtle from Palmerston Atoll and another from Tetiaroa also travelled westwards towards Fiji (Figure 9-5; NOAA NMFS and GH Balazs).

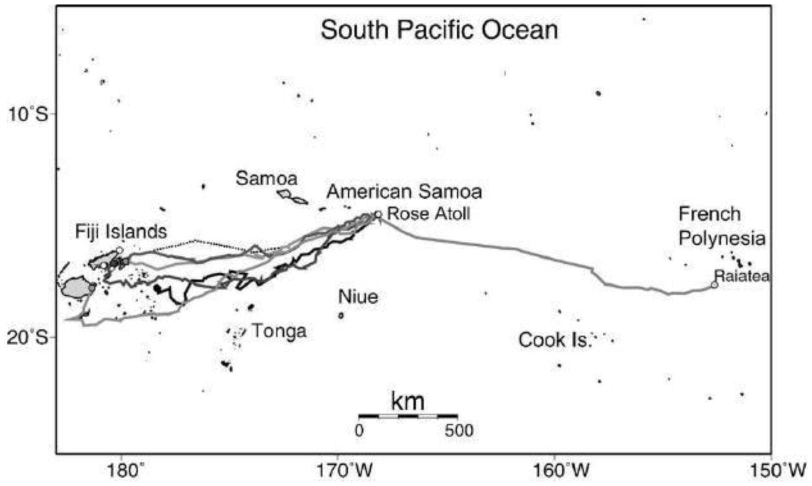


Figure 9-4. Post-nesting movements of satellite-tracked of green turtles from Rose Atoll in American Samoa. Image source: Craig et al. 2003.

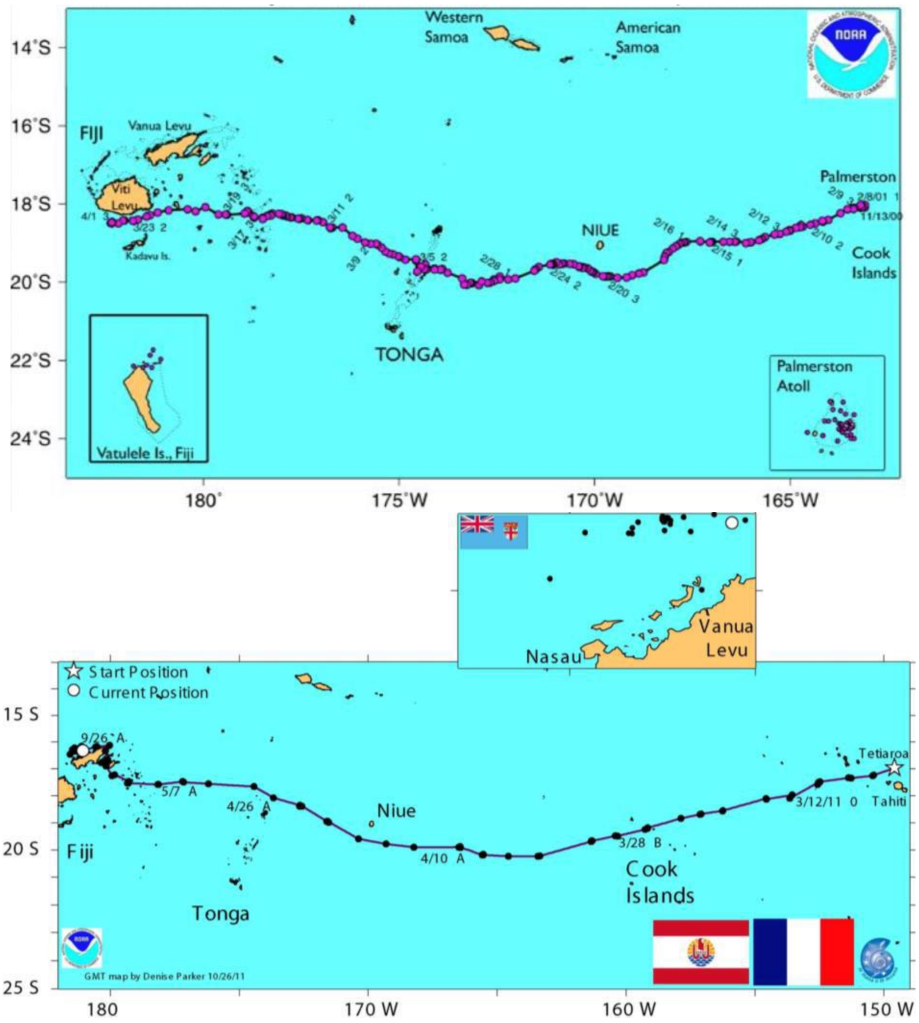


Figure 9-5. Post-nesting movements of a satellite-tracked green turtle from Palmerston Atoll (top) and another from Tetiaroa, French Polynesia (bottom). Image source: NOAA NMFS, GH Balazs, NOAA.

In the north central Pacific, turtles migrate between the main Hawaiian islands and the nesting sites in the French Frigate Shoals, but do not appear to move beyond these areas (Figure 9-6; Balazs et al. 2017). As with hawksbill turtles from Hawaii, there also has been a record of a green turtle moving towards Johnston Atoll, but this is the only event recorded over multiple years of research.

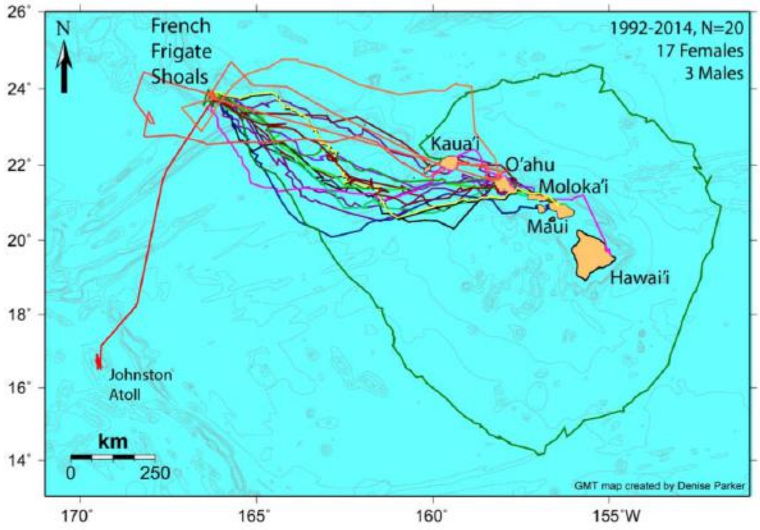


Figure 9-6. Composite view of the ocean pathways by 20 green turtles tracked during their breeding migrations linked to the colonial nesting site of French Frigate Shoals. Turtles were randomly selected for satellite tagging between 1992-2014. Image source: Balazs et al. 2017.

In the central west Pacific, green turtles have been found to move westward towards the Philippines, Taiwan and Japan, and even enter the South China Sea (Figure 9-7; Kolinski et al. 2014). Flipper tag returns also link Gielop Island in the FSM with foraging areas in Palau. Two satellite-tracked green turtles from Palau moved south into Indonesian waters (Figure 9-8; Klain et al. 2007), and flipper tag returns link Palau nesting sites to foraging areas in Sulawesi, Indonesia. Turtles from the Marshall Islands also moved generally west and south (Figure 9-9), but did also adopt oceanic 'wandering' strategies in several cases, and in one case the turtle moved east to pelagic waters east of Erikub Atoll, covering over 4,000 km (Parker et al. 2015). Green turtles foraging at the remote Palmyra Atoll National Wildlife Refuge in the central Pacific were assigned natal origins almost exclusively (approx. 97%) to the west central and south central Pacific combined Regional Management Units using genetic markers (Naro-Maciel et al. 2014) and mitochondrial deoxyribonucleic acid (mtDNA) analysis also indicates juvenile turtles disperse to the west (Nishizawa et al. 2014) even though this might be contrary to oceanic circulation (Nishizawa et al. 2013).

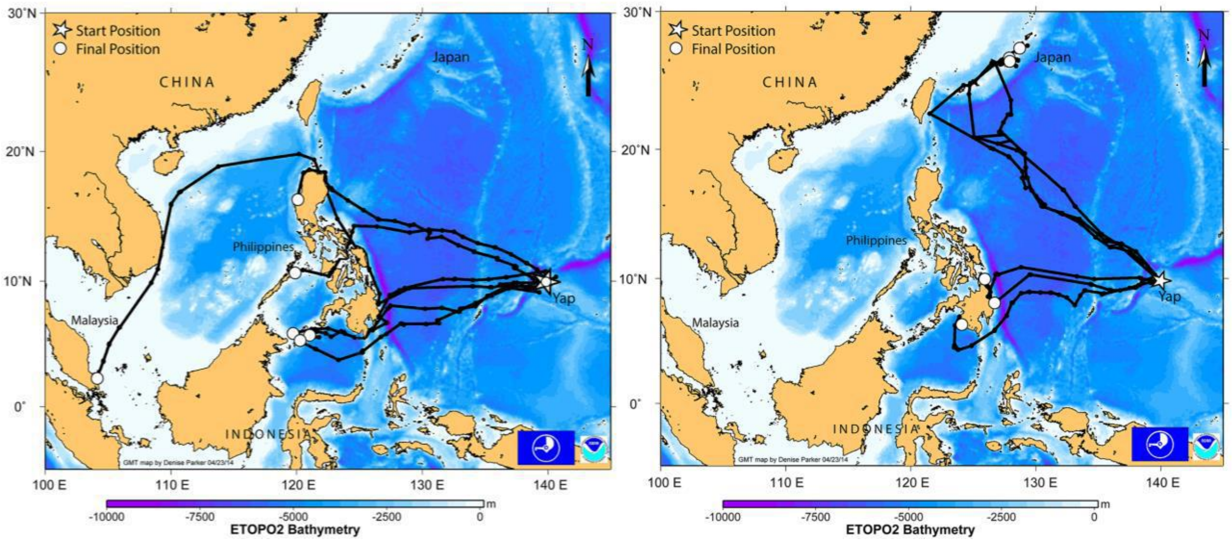


Figure 9-7. Post-nesting migrations of green turtles from Gielop Island, Federated States of Micronesia in 2005 and 2006 (left) and in 2007 (right). Image source: Kolinaksi et al. 2014.

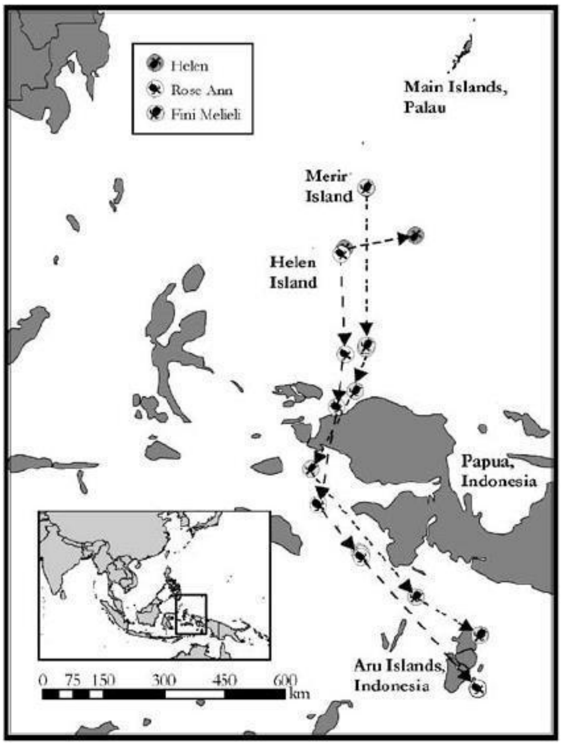


Figure 9-8. Post-nesting migrations of green turtles from Merir and Helen Islands, Palau. Image source: Klain et al. 2007

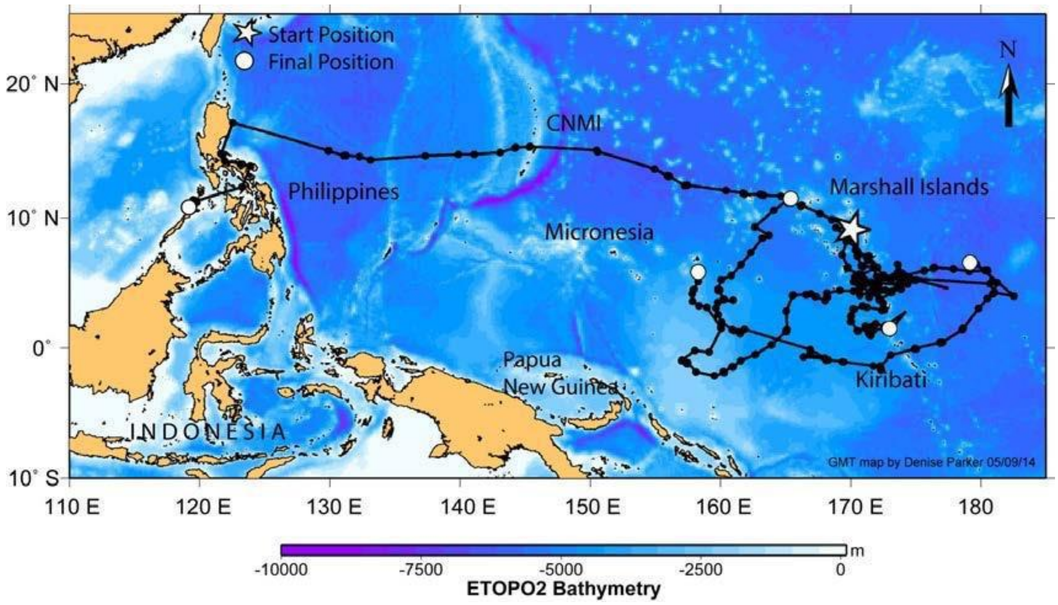


Figure 9-9. Post-nesting movement of five green turtles from Erikub Atoll, Republic of the Marshall Islands in 2007-2008. Image source: Parker et al. 2015.

As with central western green turtles, those in the northwest Pacific Ocean tend to remain in the western extent of the Pacific: green turtles from Taiwan migrated north towards Japan, and west into the South China Sea (Figure 9-10; Cheng and Wang 2009). Similar movements were recorded for green turtles departing from Amami, just north of Okinawa, which also moved northeast and remained close to the Japanese islands (Figure 9-11; Oki et al. 2019). Similarly, Fukuoka et al. (2015) found localised movements of green turtles during summer months in Japan (Figure 9-12) and Song et al. (2002) tracked turtles along the China coast with one reaching Japan.

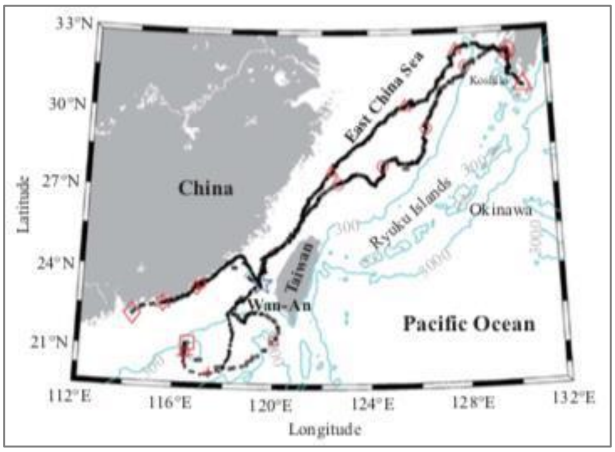


Figure 9-10. Post-nesting movements of satellite-tracked of green turtles nesting in Taiwan. Image source: Cheng and Wang 2009.

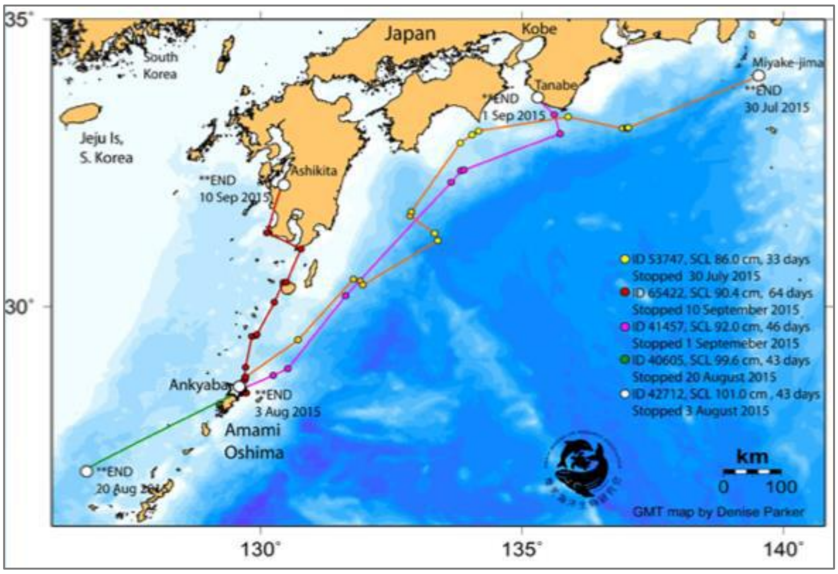


Figure 9-11. Post-nesting movements of satellite-tracked of green turtles from Amami, Japan. Image source: Parker 2006

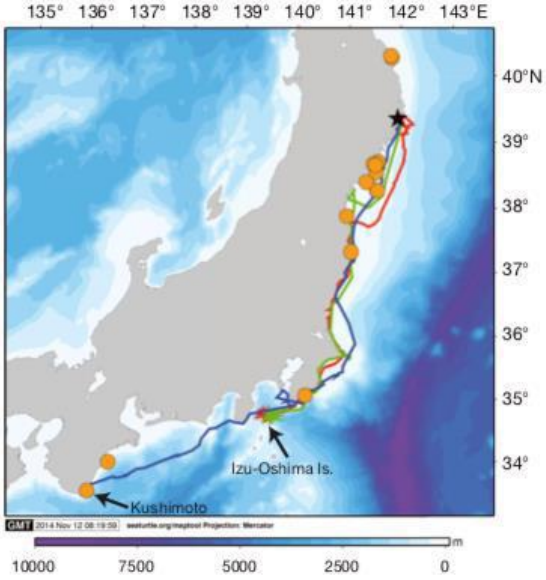


Figure 9-12. Migration paths of the green turtles equipped with satellite relay data loggers (SRDLs) and re-capture points of tagged turtles in the mark-recapture study. Star: release point; orange circles: recapture points. Image source: Fukuoka et al. 2015.

In the southwest Pacific, green turtles tend to remain between Australia and the outer island nations and territories (e.g. New Caledonia, Vanuatu, Solomon Islands), similar to the movements of hawksbill turtles. Read et al. (2014) compiled movements of satellite tracked green turtles and flipper tag recoveries and identified clear linkages between Australia and New Caledonia (Figure 9-13). Less substantive linkages were also identified between New Caledonia and Papua New Guinea, and between Vanuatu and New Caledonia. Tag recoveries recorded by Limpus (2007a) showed similar distributions, with tags deployed on foraging and nesting green turtles in Australia also recovered in Fiji, Vanuatu, the Solomon Islands and Papua New Guinea. At the same time however, Limpus (2007a) also highlighted the domestic movements of green turtles, with many turtles from Australian rookeries moving to foraging grounds in Australia (Figure 9-14).

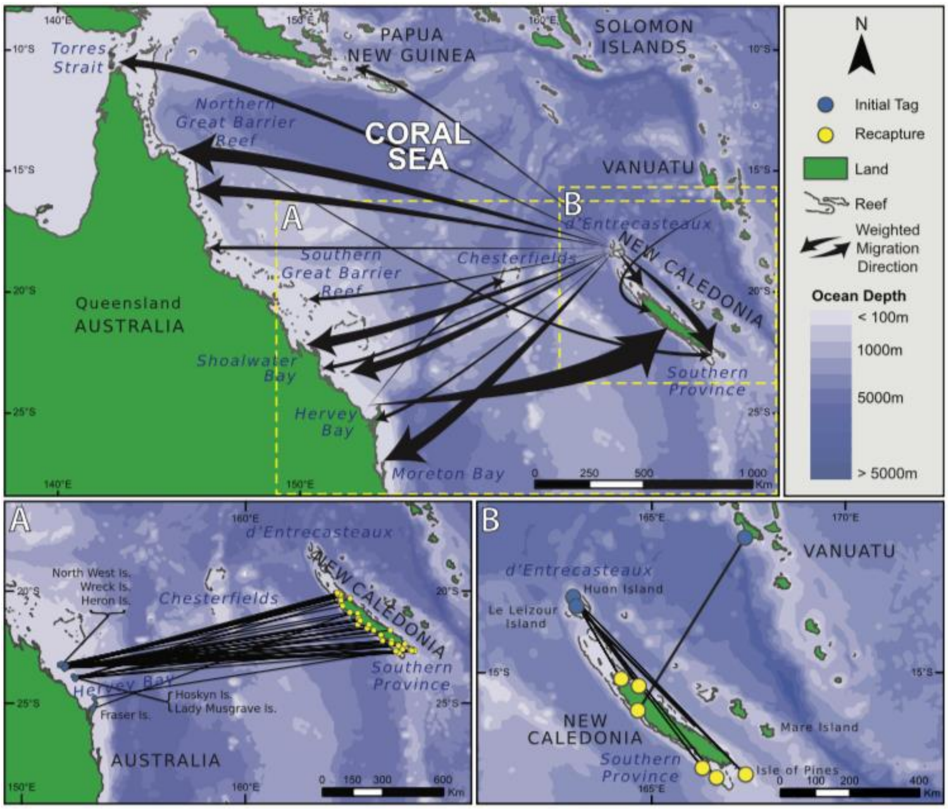


Figure 9-13. Trajectory maps obtained by the tag recoveries (n = 93) and satellite tracking of *C. mydas* in the Coral Sea (n = 1). Image source: Read et al. 2014.

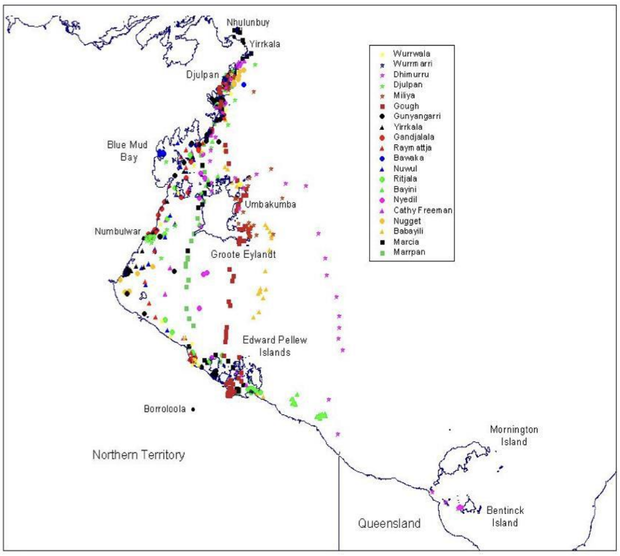


Figure 9-14. Post-nesting migratory tracks of adult females from north-eastern Arnhem Land nesting beaches identified by satellite telemetry. Image supplied by R. Kennett. Image source: Limpus 2007a.

9.2 Hawksbill turtle migrations

Migratory connectivity in hawksbill turtles is less well understood and typically of much shorter movement distances than those of leatherback turtles. Hawksbill turtles from north-eastern Australia have been recorded in Vanuatu, Solomon Islands, Papua New Guinea and sites elsewhere in the Great Barrier Reef (Figure 9-15; Miller et al. 1998). The Arnavon Islands are a particularly important destination for hawksbills residing in Australian foraging areas (Figure 9-16; Hamilton et al. 2015; Figure 9-17). Hawksbills that lay eggs on islands within the northern Great Barrier Reef have also been recorded migrating to foraging sites in Indonesia, Papua New Guinea and elsewhere in the northern Great Barrier Reef. These migration recaptures span time intervals between nesting and feeding recaptures of 0.5 yr–9.0 yr, and the individual minimum migration displacements from feeding to nesting have a range of 98 km–2,420 km (Limpus 2008).

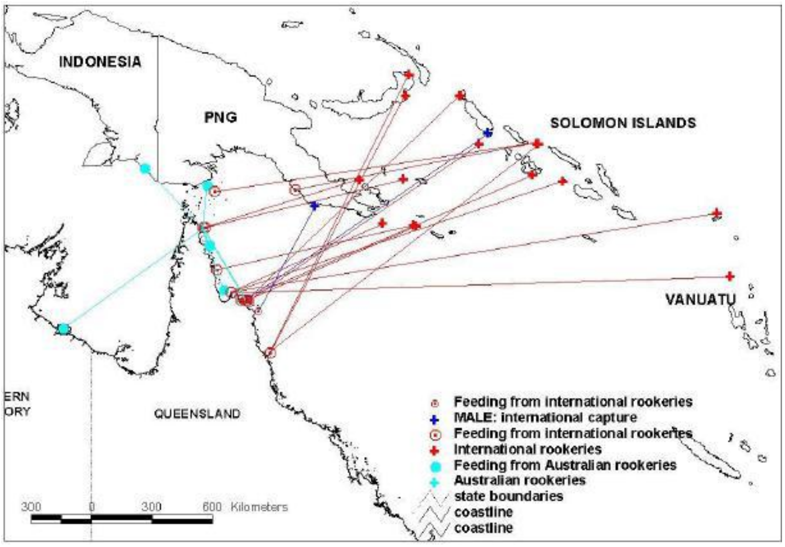


Figure 9-15. Migration by adult female and male hawksbill turtles, *Eretmochelys imbricata*, in the Coral Sea Region of the southwest Pacific Ocean. Image source: Limpus 2008.

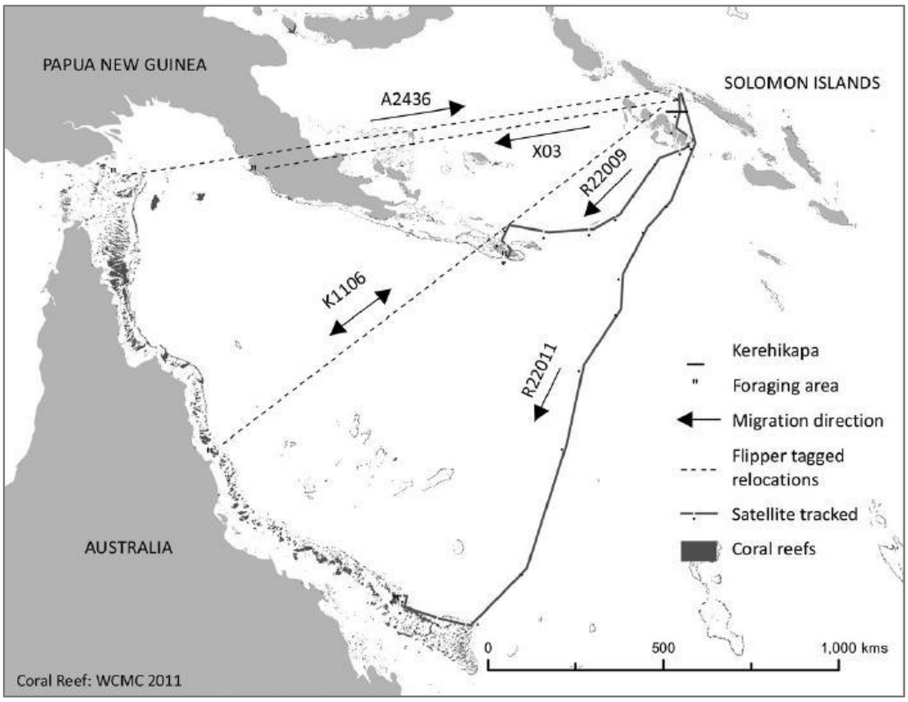


Figure 9-16. Migrations of adult female hawksbill turtles to and from Kerehikapa beach, Arnavon Islands. Image source: Hamilton et al. 2015.

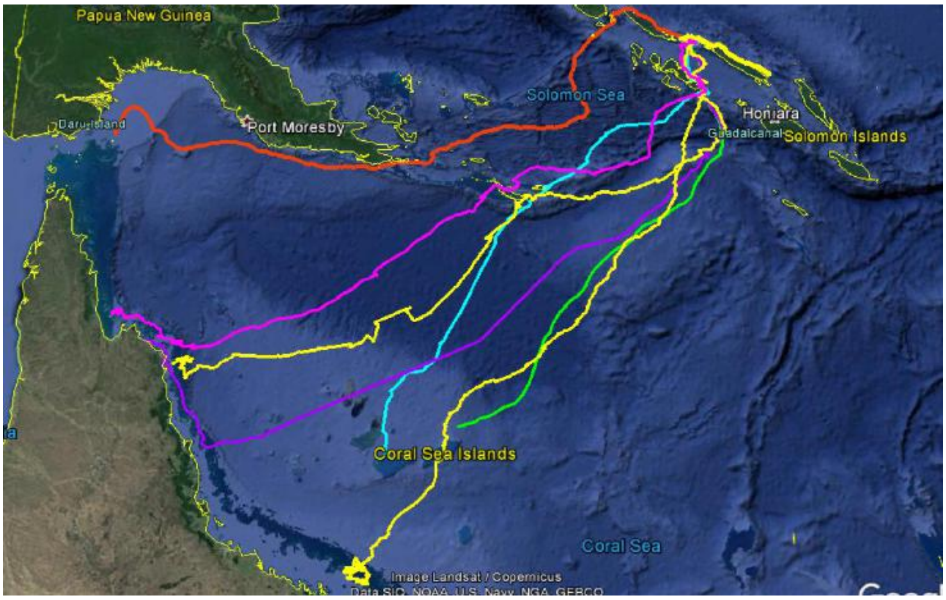


Figure 9-17. Migration routes of eight hawksbill turtles tagged in the Arnavon Islands, Solomon Islands. Image source: Richard Hamilton/The Nature Conservancy.

Similar movement patterns have been documented for hawksbills departing from the Conflict Islands in Papua New Guinea (CICI 2018). Hawksbills moved south and west into the Coral Sea to similar locations as turtles from the Solomon Islands (Figure 9-18). Limited flipper tagging data also has demonstrated that northeast Australia nesting hawksbills have been found in PNG, and nesting PNG hawksbills have been reported within their Australian foraging range. Interestingly, these movement patterns do not show movements between PNG and the Solomon Islands, or from the Solomon Islands to Vanuatu or Fiji, indicating the importance of the Coral Sea as a regional foraging area. Not all hawksbills from northern Australia migrate into the Coral Sea, as post-nesting females turtles tagged in the Gulf of Carpentaria remained in Australian waters (Hoenner et al. 2015). This same study, however, concluded that post-hatchling turtles from these sites might seed areas in the Torres Straits and the northern Coral Sea (Figure 9-19).

Figure 9-18. Post-nesting movements of 10 hawksbill turtles from the Conflict Islands, Papua New Guinea. Image source: CI 2018.

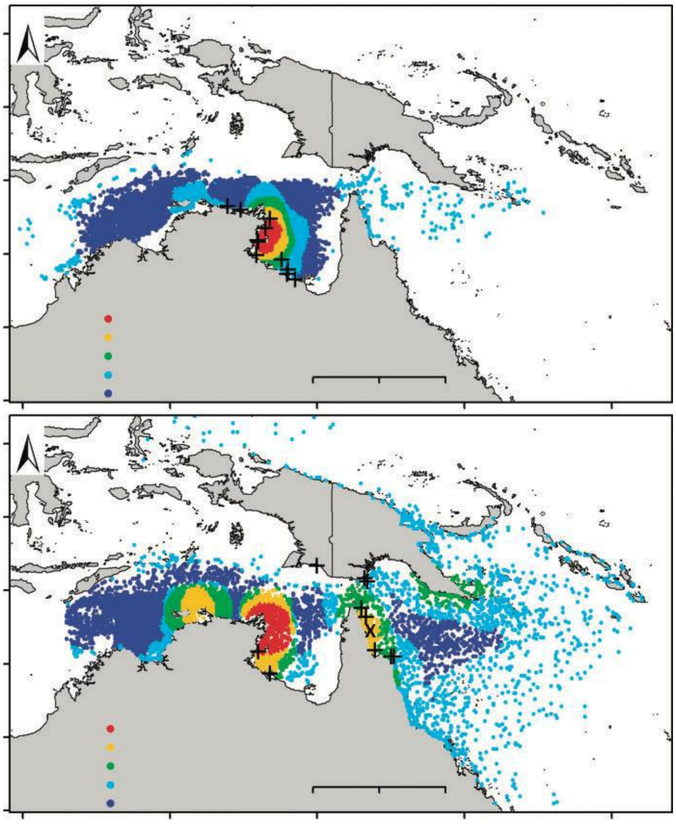


Figure 9-19. Modelled distribution and clustering probability of post-hatchling turtles from (a) North East Island and (b) Milman Island, whose locations are shown by crosses. Red markers indicate spatial clustering of high probabilities, whereas light blue markers indicate spatial clustering of low probabilities. Image source: Hoenner et al. 2015.

In the north central Pacific, hawksbills from the main Hawaiian islands typically remain within the islands (Figure 9-20; Parker et al. 2009), although a few long distance migrations have been recorded, including one track from Hawaii Island to a point some 650 km southwest of Johnston Atoll (Figure 9-21; Graham 2009). Passive drifters emulating hawksbill turtle hatchling dispersal suggest that hawksbills remain in the archipelago for several months or more, but that they may also be carried by currents in a westerly direction from the main Hawaiian islands (van Houtan et al. 2016).

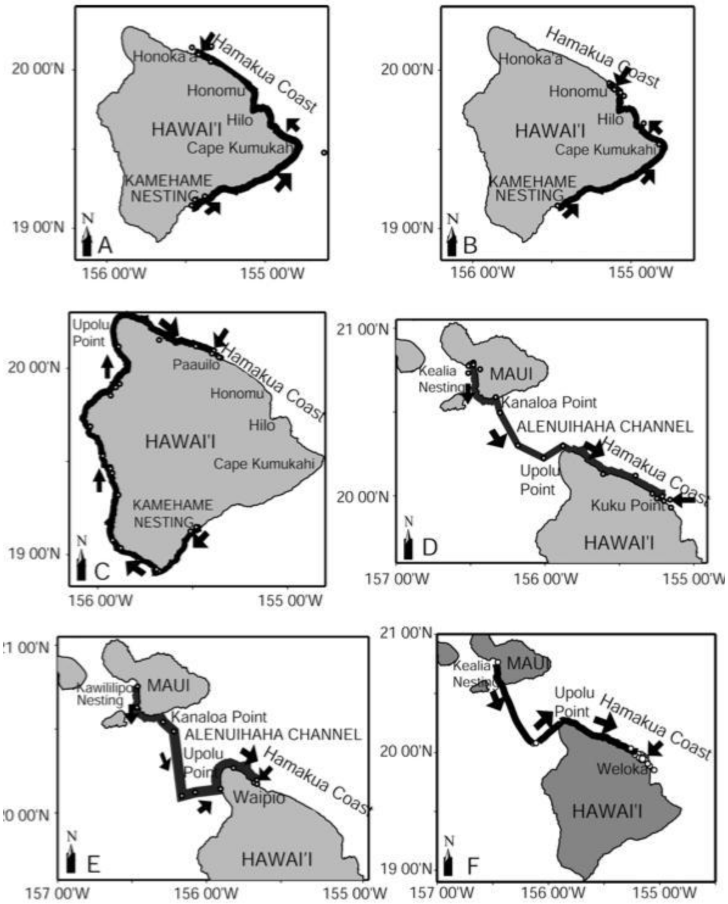


Figure 9-20. Post-nesting movement of six hawksbill turtles from nesting beaches to foraging areas on the Hāmākua Coast of the island of Hawaii. Image source: Parker et al. 2009.

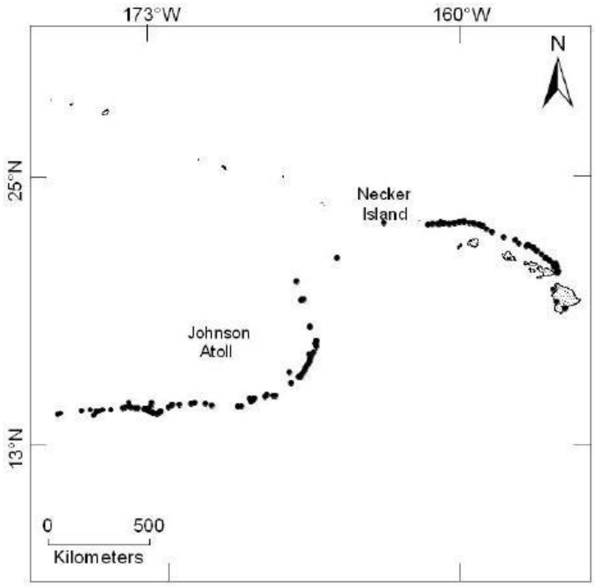


Figure 9-21. Post nesting migration of a hawksbill turtle from Hawaii into the Pacific Ocean. Image source: Graham 2009.

In the northwest Pacific there has been limited tracking of hawksbills. Six hawksbills were released in Kuroshima, Japan, of which five were recaptured in the waters adjacent to the Yaeyama Islands; and the remaining one was recaptured in the coastal waters of Okinawajima Island (Kamezaki and Hirate 1992). None of those turtles left Japan waters. One long-distance movement of a male hawksbill turtle was recorded from Okinawa, Japan to Yapen, Papua Province, Indonesia (Figure 9-22; Kawazu et al. 2018).

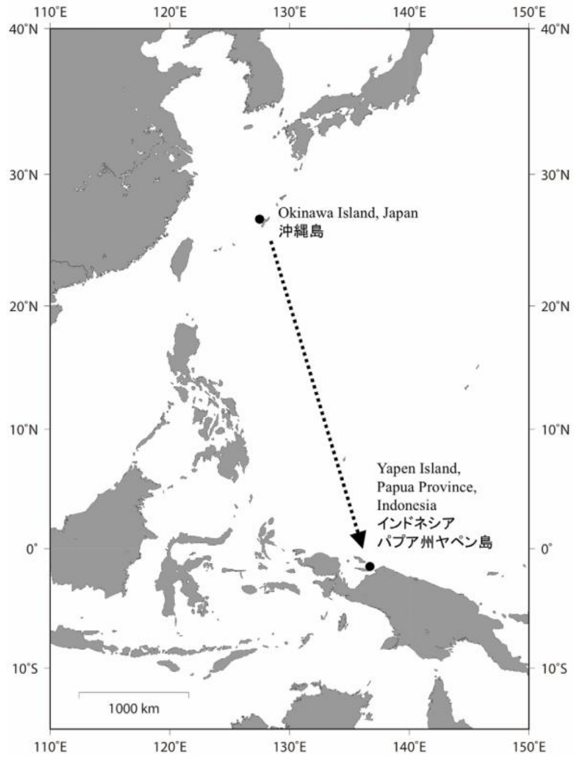


Figure 9-22. Long-distance movement of a male hawksbill turtle from Okinawa Island, Japan, to Yapen Island, Papua Province, Indonesia, based on tag recovery. Image source: Kawazu et al. 2018

In the central Pacific there has also been limited tracking of hawksbills, but those that have been tracked have demonstrated substantial movements. For instance, two turtles tracked from American Samoa travelled to the Cook Islands (Figure 9-23; Tagarino et al 2008), a straight-line distance of some 1,400 km.

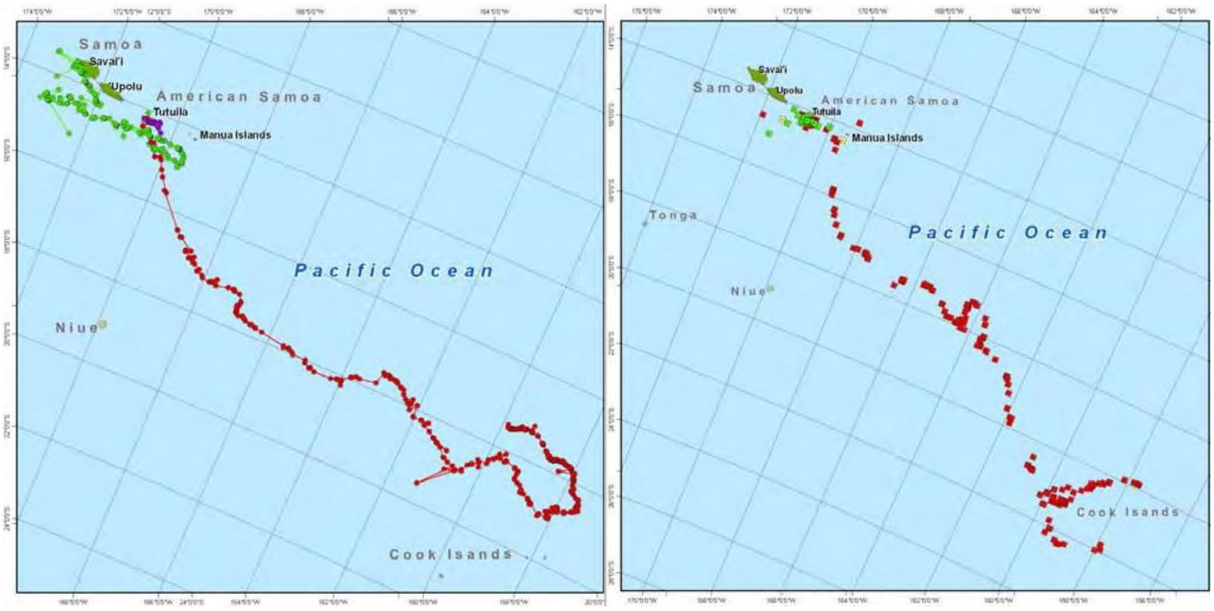


Figure 9-23. Movements of post-nesting hawksbill turtles from Tutuila, American Samoa, to the Cook Islands. Image source: Tagarino et al. 2008.

In the eastern Pacific, the movements of hawksbill turtles are still not well understood. Recent studies suggest that hawksbills in the eastern Pacific exhibit a higher degree of fidelity to their natal area than other species with restricted or non-existent migrations. Hawksbill turtles typically remain close to land, moving north and south along the American continents (Figure 9-24; Gaos et al. 2012)

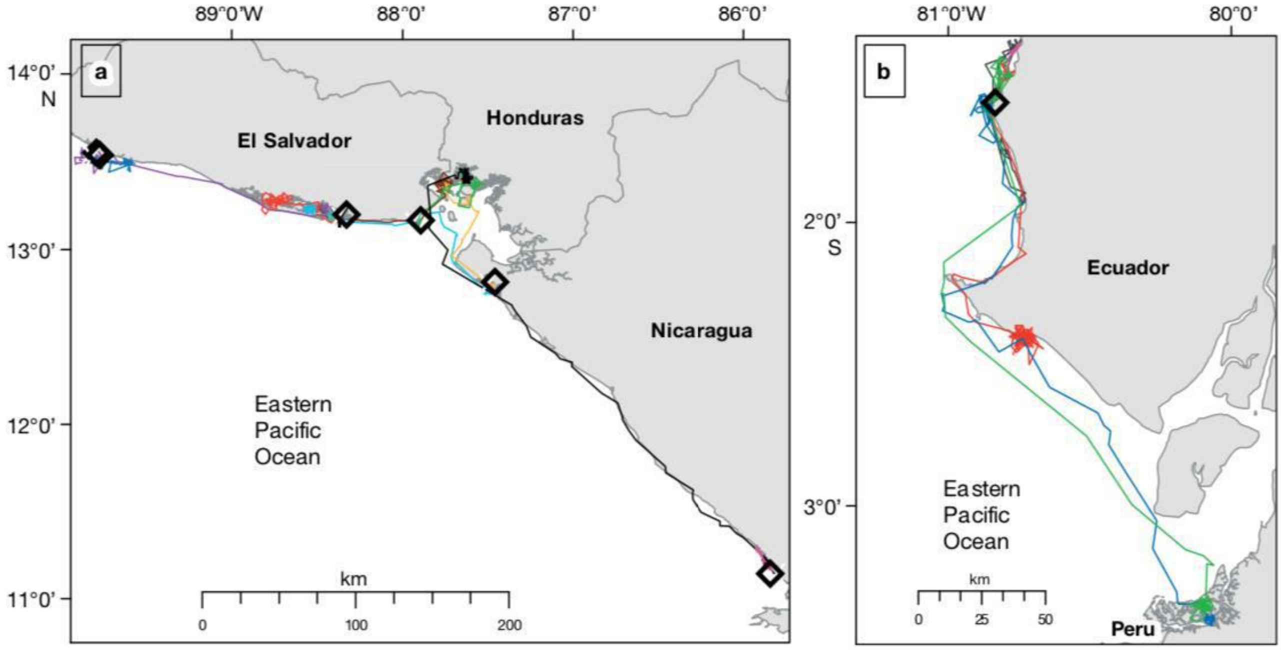


Figure 9-24. Tracks of 15 hawksbills equipped with satellite tags in (a) central America and (b) Ecuador in the present study between 2008 and 2010. Image source: Gaos et al. 2012

9.3 Loggerhead turtle migrations

Substantial tracking of juvenile, sub adult and adult loggerhead turtles has been conducted in the Pacific Ocean and shows that there is no mixing between north Pacific and south Pacific loggerhead stocks. Some of these turtles were aquarium-reared; others were caught on foraging grounds (e.g. Baja California), while others were released after entanglement in fishing gears (e.g. Peru). An overview of all satellite tracking data for the Pacific Ocean is presented in the Status of Sea Turtles Report No. XIII (Figure 9-25).

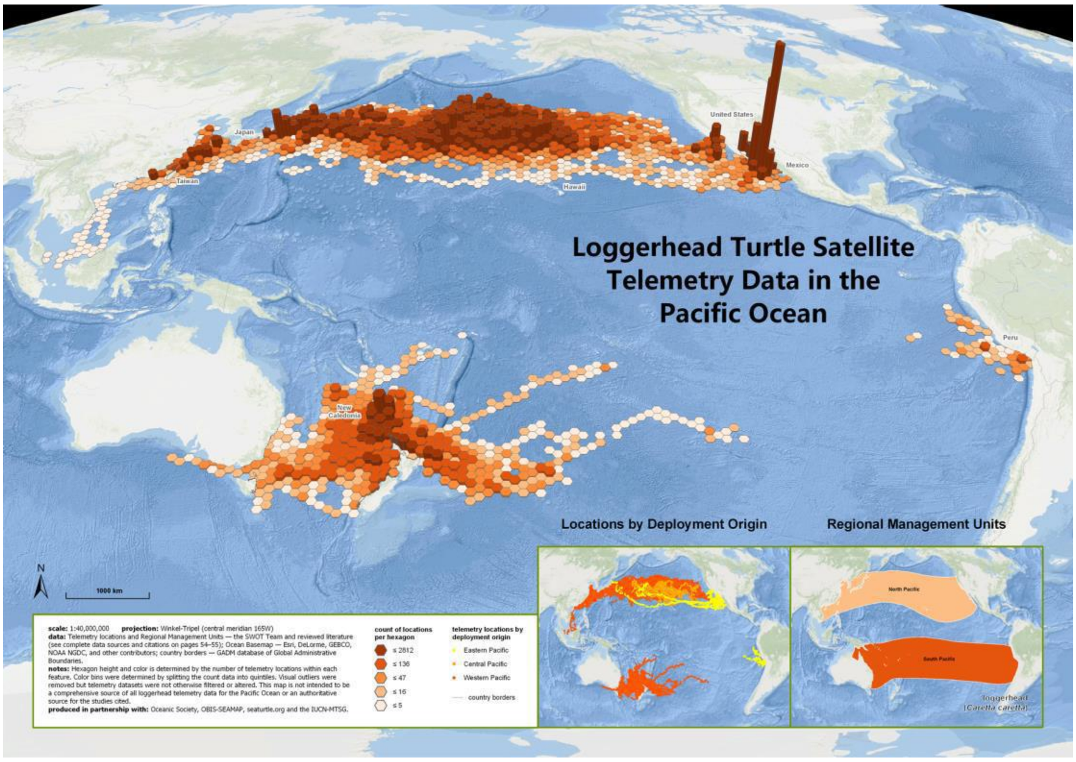


Figure 9-25. Loggerhead turtle satellite telemetry data in the Pacific Ocean. Image source: SWOT Report No XIII.

In the south Pacific loggerhead RMU, there is evidence of genetic connectivity between the Australian rookery and the foraging turtles of South America (Boyle et al. 2009). While protected at the nesting site, and via fishery management measures such as Turtle Excluder Devices in the Great Barrier Reef, these turtles are subject to pressures from artisanal fisheries off Peru and Chile. Post-capture movements of loggerheads from the Peruvian longline shark fisheries generally returned to oceanic waters (Mangel et al. 2011; Figure 9-26) and did not appear to return to coastal waters (< 200 m). This study indicated that many of these turtles were 'resident' in the waters off Peru and Chile, where they maintained a pelagic lifestyle for the duration of tracking. Turtles spent >97% of their time in waters in excess of 1,000 m depth (Mangel et al. 2011) Their results supported findings from other regions indicating that juvenile loggerhead turtles may be actively selecting key pelagic habitats and are not simply passively distributed by ocean currents.

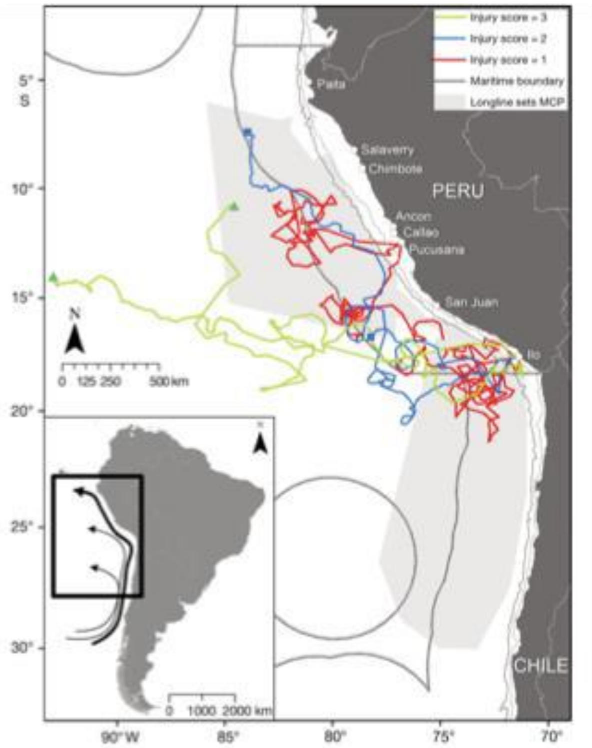


Figure 9-26. Loggerhead movements after entanglement in Peruvian shark fisheries. Tracks are colour coded by injury level, and shaded polygon depicts the extent of the fishery. Image source: Mangel et al. 2011.

In the north Pacific loggerhead RMU, loggerhead hatchlings entering the waters off their nesting beaches in Japan embark on a large-scale developmental migration that encompasses the entire north Pacific Ocean (Nichols et al. 2000). A large number of loggerhead turtles arrive along the coast of the Baja California peninsula (Ramirez Cruz et al. 1991), and this area is an important foraging ground for north Pacific loggerheads (Figure 9-27; Seminoff et al. 2014). It is unknown how long loggerhead turtles remain in Baja California waters, but studies in the region suggest that upon reaching maturity and reproductive condition, loggerheads are capable of migrating from Baja California to natal beaches in Japan and this is supported by satellite tracking evidence (Figure 9-28; Nichols et al. 2000).

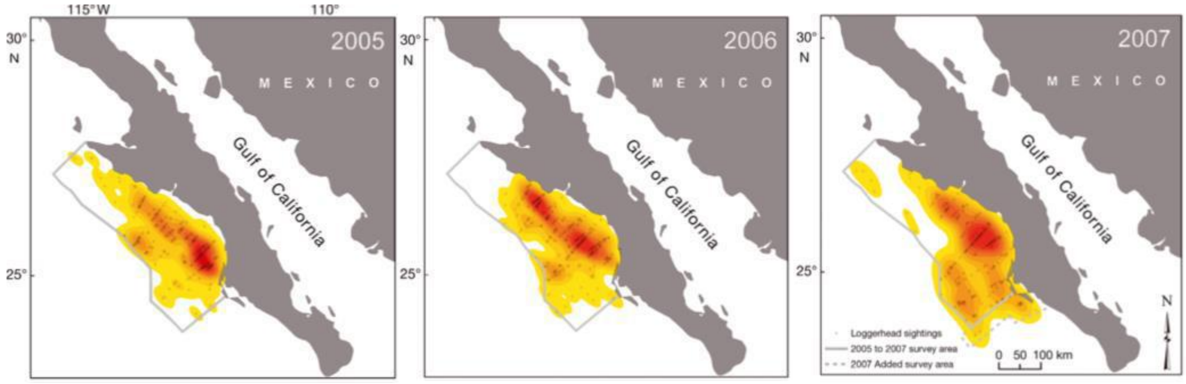


Figure 9-27. Fixed kernel density plot for loggerhead sea turtle sightings during the 2005 to 2007 aerial surveys along the Pacific Coast of Baja California. Image source: Seminoff et al. 2014.

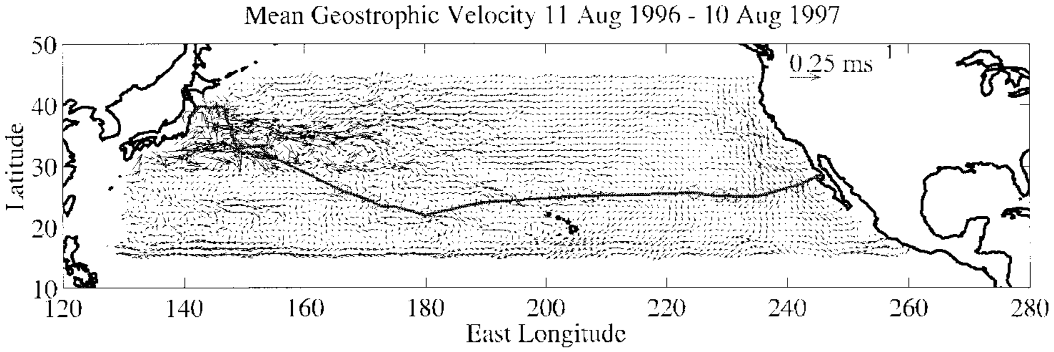


Figure 9-28. Track of transpacific movement of loggerhead turtle (07667) from Mexico to Japan, during 1996–1997, monitored using satellite telemetry. Image source: Nichols et al. 2000.

A release of captive reared south Pacific loggerheads from New Caledonia, indicated that these turtles moved in just about all directions, but it is unknown if this was simply an artefact of how they were reared, or related to natural dispersal patterns (Figure 9-29).

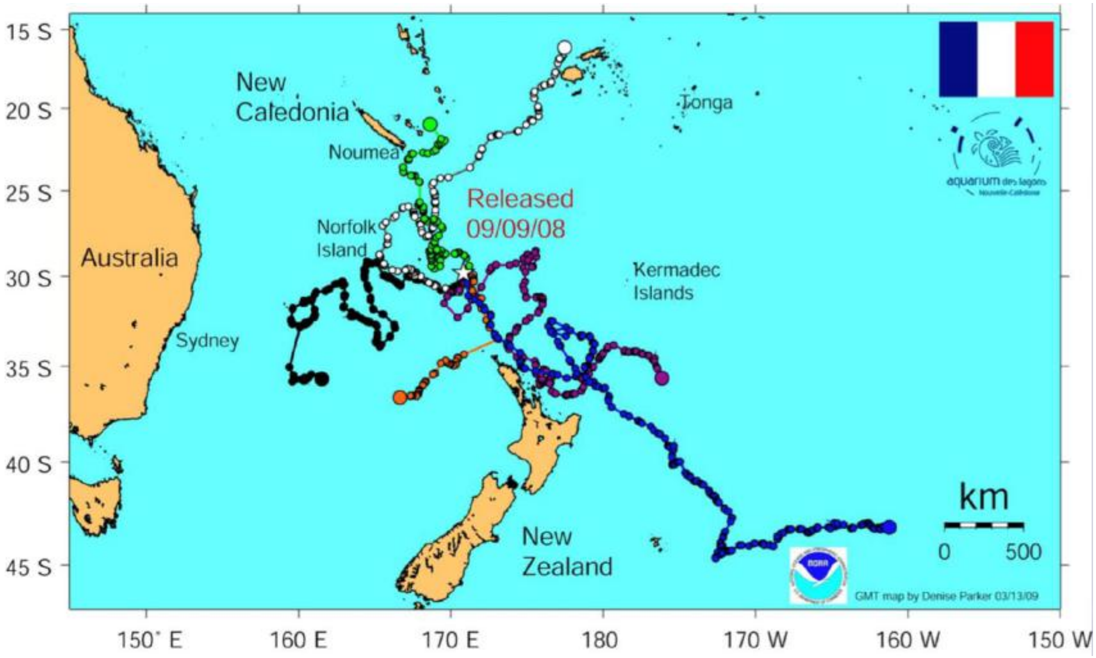


Figure 9-29. Post-release movements of juvenile loggerheads following several years of being raised in captivity. Image source: Parker 2006.

In the north Pacific loggerheads adopt two distinct foraging strategies, with some turtles residing in deeper oceanic waters of the north Pacific, while others migrate further to the waters off Baja California (Figure 9-30; Peckham et al. 2011). The oceanic waters are less productive, and resources undergo latitudinal shifts of ~1000 km (Polovina et al. 2001, 2004), forcing loggerhead turtles to undertake seasonal migrations in order to take advantage of oceanic productivity (Polovina et al. 2001, 2004). Results show high residency of juvenile loggerheads within the central north Pacific and a moderate influence of the Earth's magnetic field, but no real-time environmental driver to explain migratory behaviour. Multi-year tracking of juvenile loggerheads in the north Pacific has revealed that the pelagic phase of juvenile loggerheads is extensive, and that the central north Pacific acts as important developmental foraging grounds for young juvenile loggerhead sea turtles, rather than just a migratory corridor (Figure 9-31; Briscoe et al. 2016). These juvenile movement patterns are supported by modelled dispersal studies for hatchlings from key rookeries in Japan (Figure 9-32; Okuyama et al. 2011).

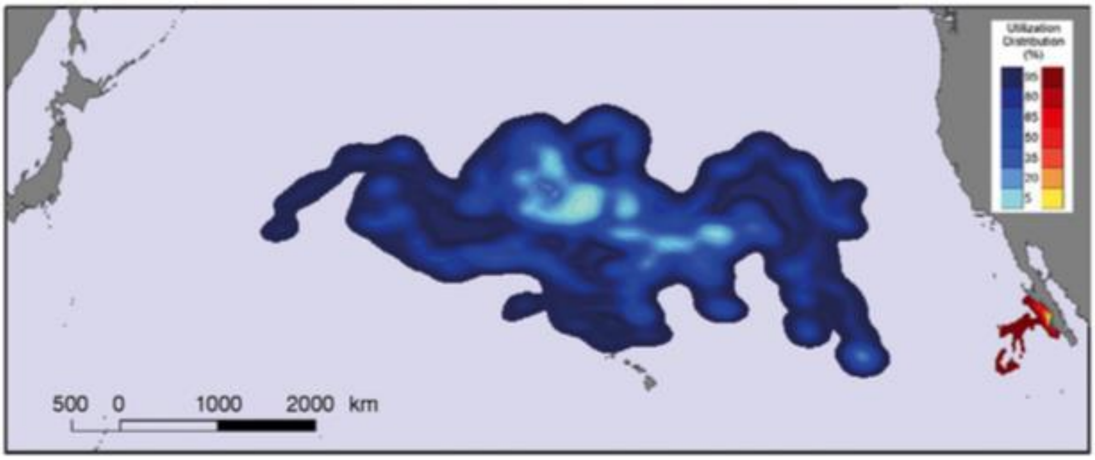


Figure 9-30. Utilisation distributions of juvenile turtles in the central north Pacific (blue gradient) and at the Baja California peninsula (red gradient). Image source: Peckham et al. 2011.

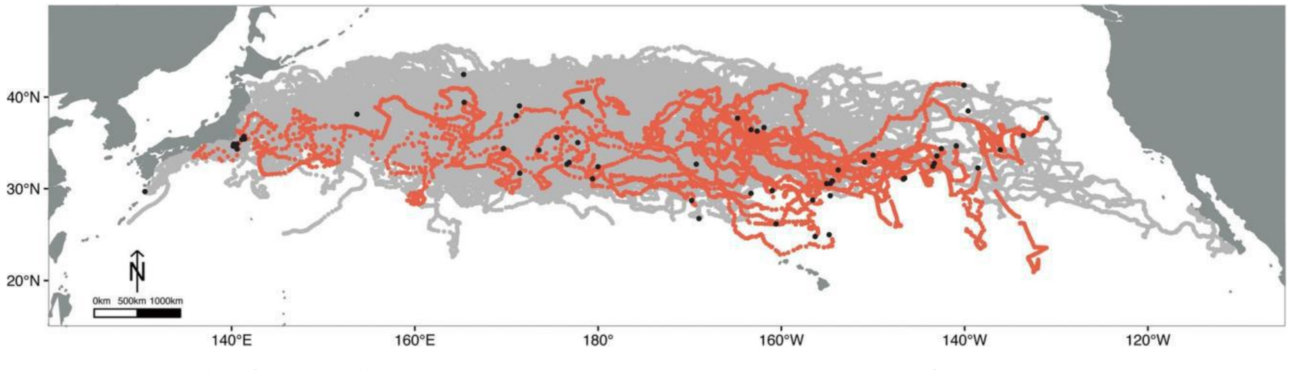


Figure 9-31. Tracks of 231 satellite tagged juvenile loggerhead sea turtle locations from 1997 to 2014. Grey tracks represent 197 captive-reared juveniles; red tracks represent wild-caught juveniles. Black dots indicate the deploy locations for individual tracks. Image source: Briscoe et al. 2016.

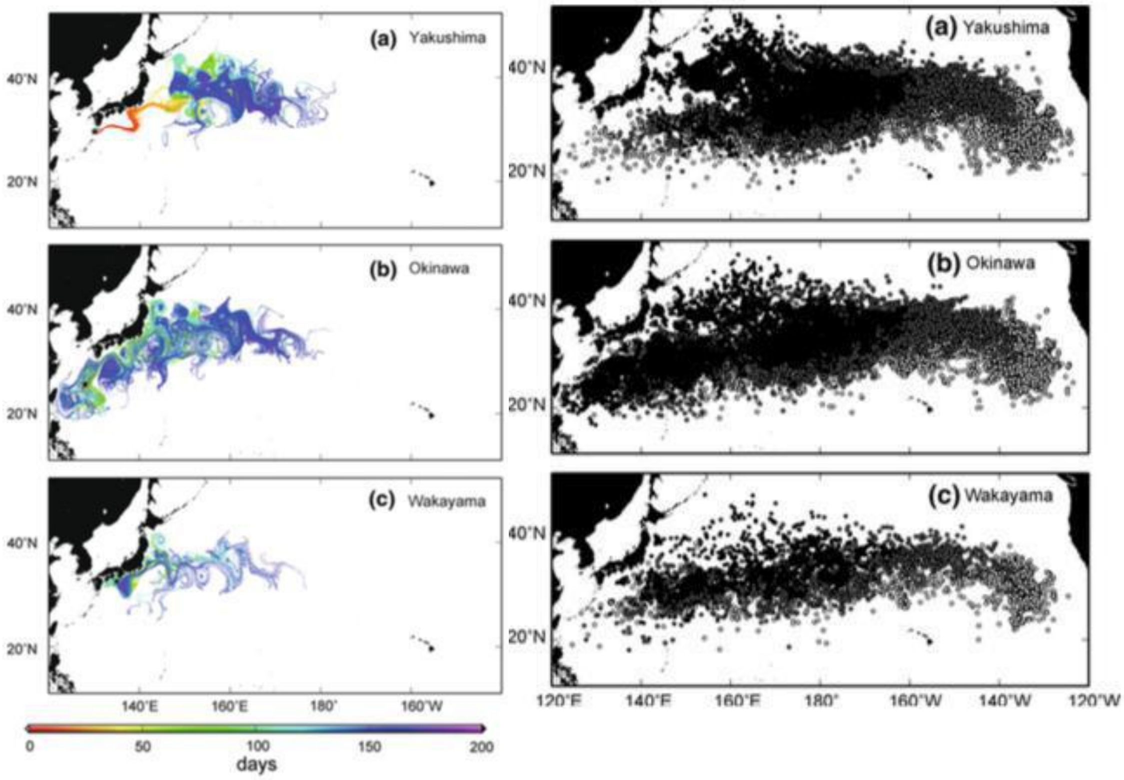


Figure 9-32. Modelled particle distribution from three key rookeries in Japan in the first 200 days (left), and after one (black dots) and five (white dots) years (right). Image source: Okuyama et al. 2011.

9.4 Leatherback turtle migrations

The primary nesting sites for leatherback sea turtles in the western Pacific are in West Papua, Indonesia, Papua New Guinea (PNG) and the Solomon Islands. Leatherbacks that nest during boreal summer typically move into the temperate north Pacific Ocean or into the South China Sea. Turtles that nest during boreal winter move into temperate and tropical waters of the southern hemisphere (Benson et al. 2007, 2011; Figure 9-1 above). Movements to the most distant and temperate foraging grounds off California requires a 10–12 month trans-Pacific migration and turtles then stay in the eastern Pacific for several seasons before returning to nest in the western Pacific (Benson et al. 2011). In the eastern north Pacific, sightings and incidental capture data indicate that the leatherback is found as far north as Alaska, but has been most frequently encountered off the coast of central California (Stinson 1984, Starbird et al. 1993). Genetic analyses of tissues from leatherback turtles stranded on California beaches or caught incidentally in the California-Oregon drift gillnet fishery indicate that these turtles originate from nesting beaches in the western Pacific (Dutton et al. 2007).

The southern tropical foraging destinations appear to support year-round foraging, potentially allowing a more rapid return to nesting beaches than the turtles that migrate to California (Benson et al. 2011). While there is a small amount of leatherback nesting in northern Australia, the leatherbacks that forage in Australian waters are presumed to migrate to Australian waters from Indonesia, Papua New Guinea and the Solomon Islands (Limpus 2008).

Because satellite tracking is not possible with hatchling sea turtles, a Sea Turtle Active Movement Model (STAMM) was developed by Gaspar and Lalire (2017) to simulate movements of hatchling and juvenile leatherbacks from West Papua, Indonesia under the combined effects of oceanic currents and habitat-driven movements. Simulation results demonstrated that juveniles gathered in the North Pacific Transition Zone (NPTZ) and undertook seasonal north-south migrations (Figure 9-33). Interestingly, these movements were rather similar to known adult tracks and used similar key foraging areas.

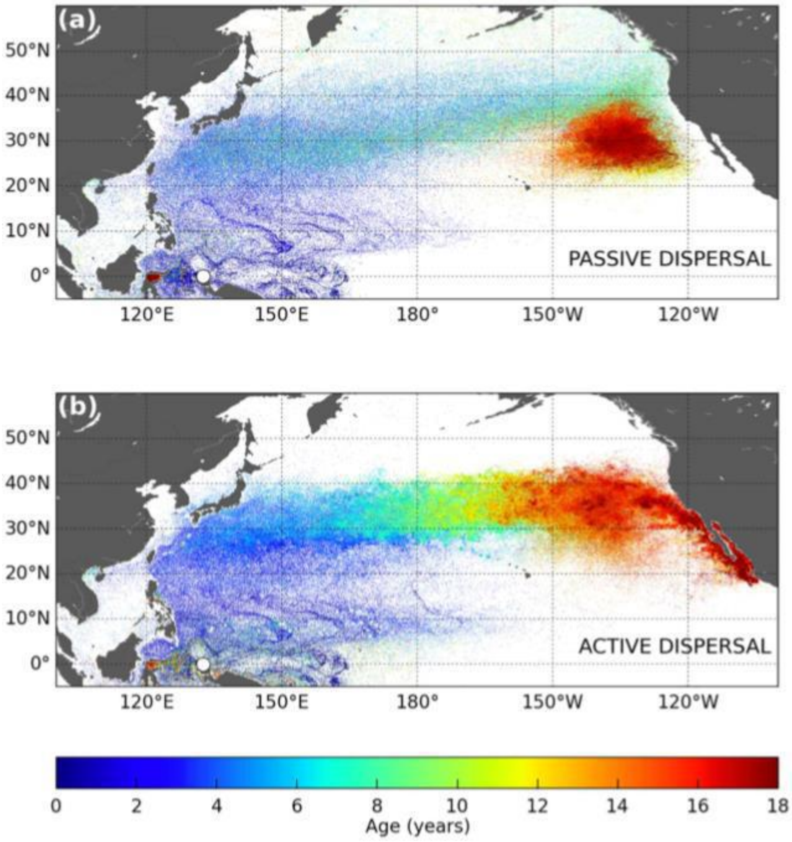


Figure 9-33. Comparison of active and passive dispersal models for hatchling and juvenile leatherback turtles. Image source: Gaspar and Lalire 2017.

One of the key nesting colonies in the eastern Pacific is at Playa Grande, in Costa Rica (Shillinger et al. 2008). Following nesting, the turtles generally head southward, within a persistent migration corridor from Costa Rica, past the equator, and into the South Pacific Gyre, a vast, low-energy, low-productivity region (Figure 9-34). Leatherbacks departing from nesting beaches in Mexico followed a similar pattern, moving south and southwest into the Pacific Ocean (Shick et al. 2013; Figure 9-35).

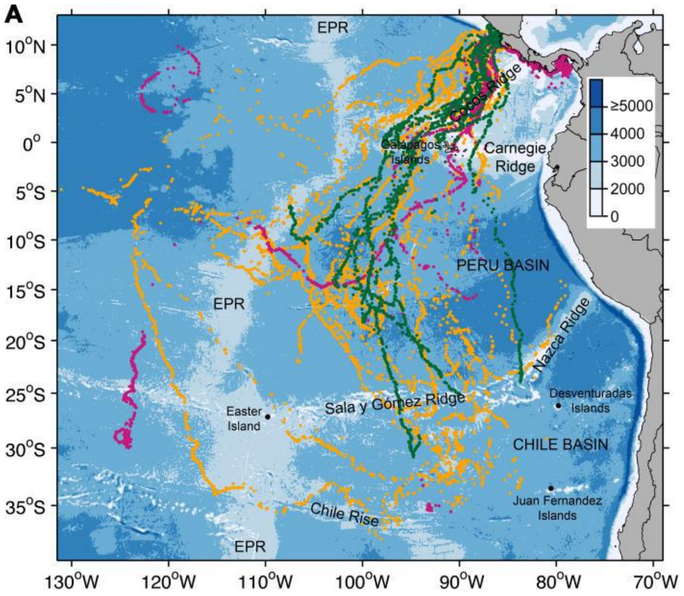


Figure 9-34. Satellite transmission positions for 46 leatherback turtles from 2004 (n = 27, orange), 2005 (n = 8, purple), and 2007 (n = 11, green), tagged at Playa Grande, Costa Rica, overlaid on bathymetry (in m) Image source: Shillinger et al. 2008.

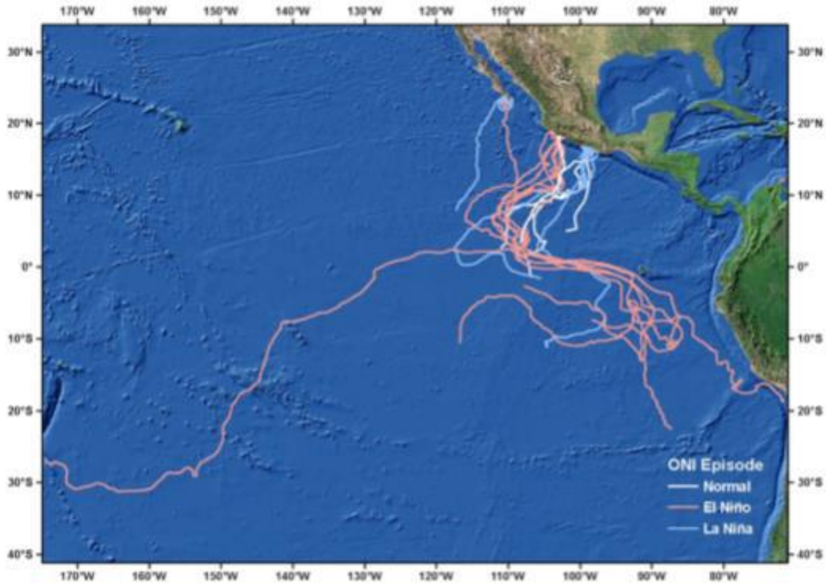


Figure 9-35. Overview map of the tracks from three different beaches in Mexico. Tracks are colour coded by Oceanic Niño Index (ONI) episode status. Image source: Shick et al. 2013.

Using modelled tracer experiments, Shillinger et al. (2012) demonstrated a strong influence of eddy transport and coastal currents, in which modelled 'hatchlings' from mesoamerican beaches were most likely to be entrained and transported offshore by large-scale eddies coincident with the peak leatherback nesting and hatchling emergence period (Figure 9-36). Shillinger et al. (2012) suggested that these eddies potentially serve as 'hatchling highways', providing a means of rapid offshore transport away from predation and a productive refuge within which newly hatched turtles can develop.

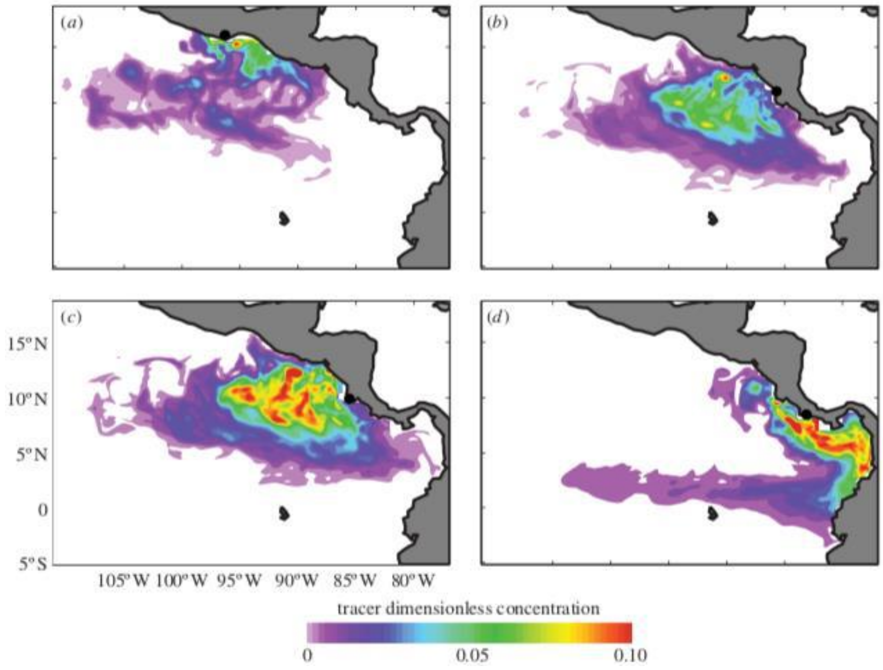


Figure 9-36. Long-term (2000–2008) mean tracer concentration based on continuous tracer releases from nesting beaches at (a) Barra de la Cruz, (b) Playa Chacocente, (c) Playa Grande and (d) Playa Carate. Image source: Shillinger et al. 2012.

9.5 Flatback turtle migrations

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. Flipper tag recoveries from female flatback turtles that were tagged while nesting on Queensland beaches indicated turtles moved some 200 km−1300 km north of their respective nesting beaches, in waters between the mainland and the Great Barrier Reef (Limpus et al. 1983). Modelling efforts of hatchling dispersal confirm the neritic lifestyle of flatbacks, with hatchlings moving only short 20 km−100 km distances from shore after 5 and 120 days (Figure 9-37; Wilderman et al. 2017). Post-nesting distribution of flatback turtles from Queensland rookeries (Figure 9-38; Limpus 2007e) demonstrate the restricted range of the species, as do locations of captures of post-hatchling flatbacks in eastern and northern Australia (Figure 9-39; Limpus 2007e). Movements of post-nesting female flatbacks from Torres Straits all oriented to the west into the Arafura and Timor Seas and not to the Pacific (Figure 9-40; Hamann 2015). Post-nesting female turtles tracked from Avoid Island (Figure 9-41) also remained within waters of the Great Barrier Reef (Hamann et al. 2018).

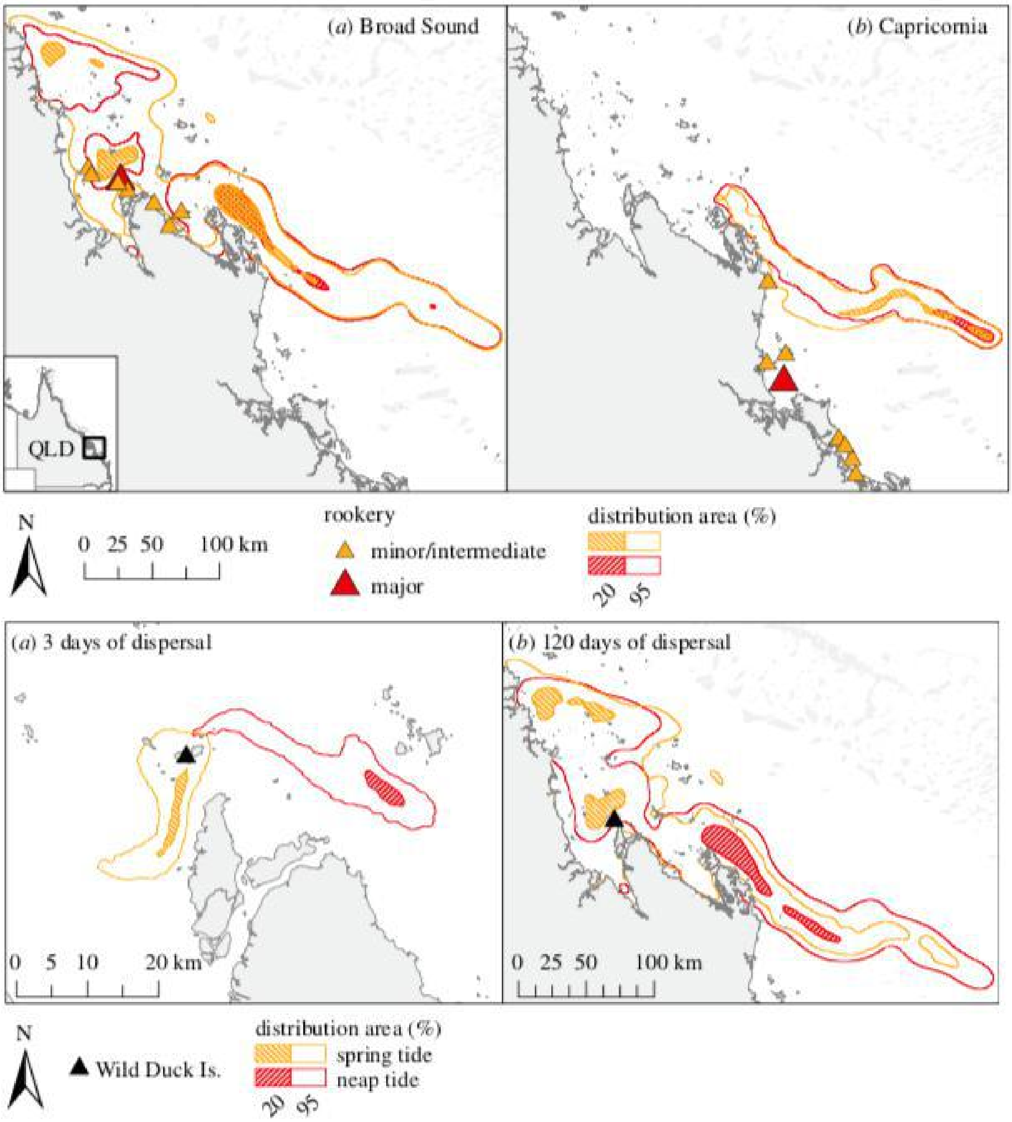


Figure 9-37. (top) Flatback distribution probabilities after 120 days of passive dispersal from major and minor/intermediate rookeries in (a) Broad Sound and (b) Capricornia. Flatback distribution probabilities from Wild Duck Island after (a) 3 days and (b) 120 days of passive drift. Image source: Wilderman et al. 2017.

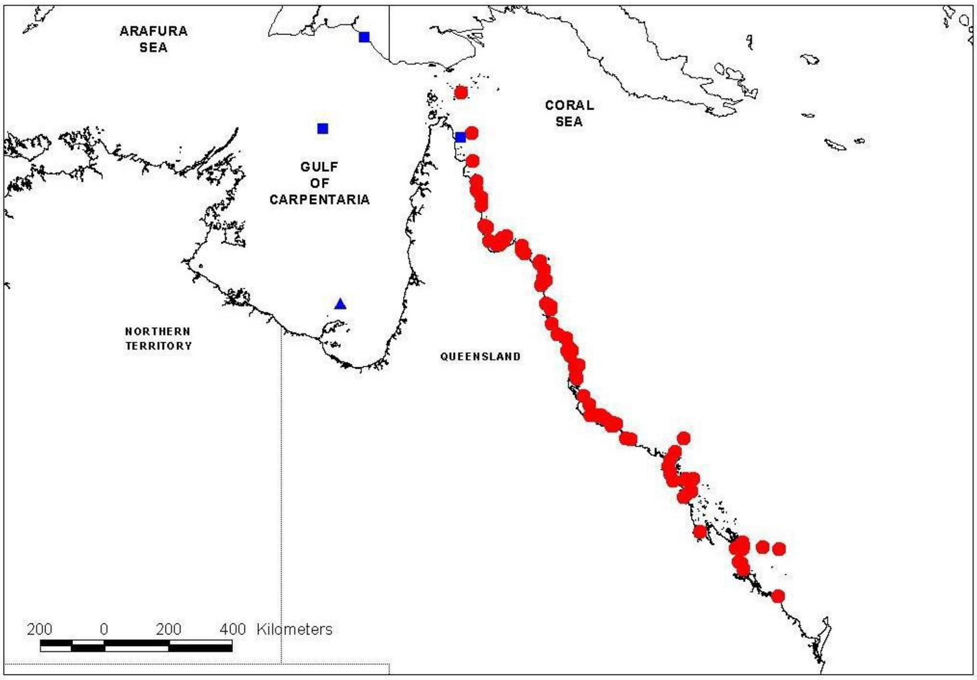


Figure 9-38. Post-nesting dispersal of *Natator depressus* from Queensland rookeries to their respective foraging areas. Symbols denote rookery of origin for the females: circles − eastern Australian rookeries of Mon Repos, Curtis Island, Peak Island and Wild Duck Island; squares – Crab Island in NE Gulf of Carpentaria; triangle – Bountiful Island, SE Gulf of Carpentaria. Image source: Limpus 2007e.

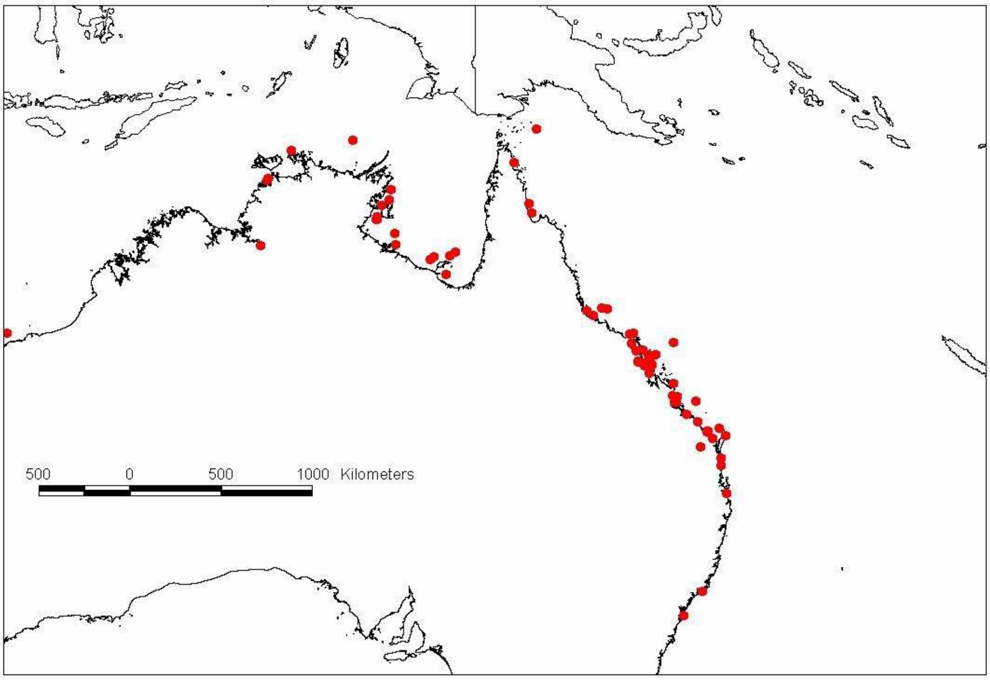


Figure 9-39. Distribution of post-hatchling records from northern and eastern Australia. Image source: Limpus 2007e.

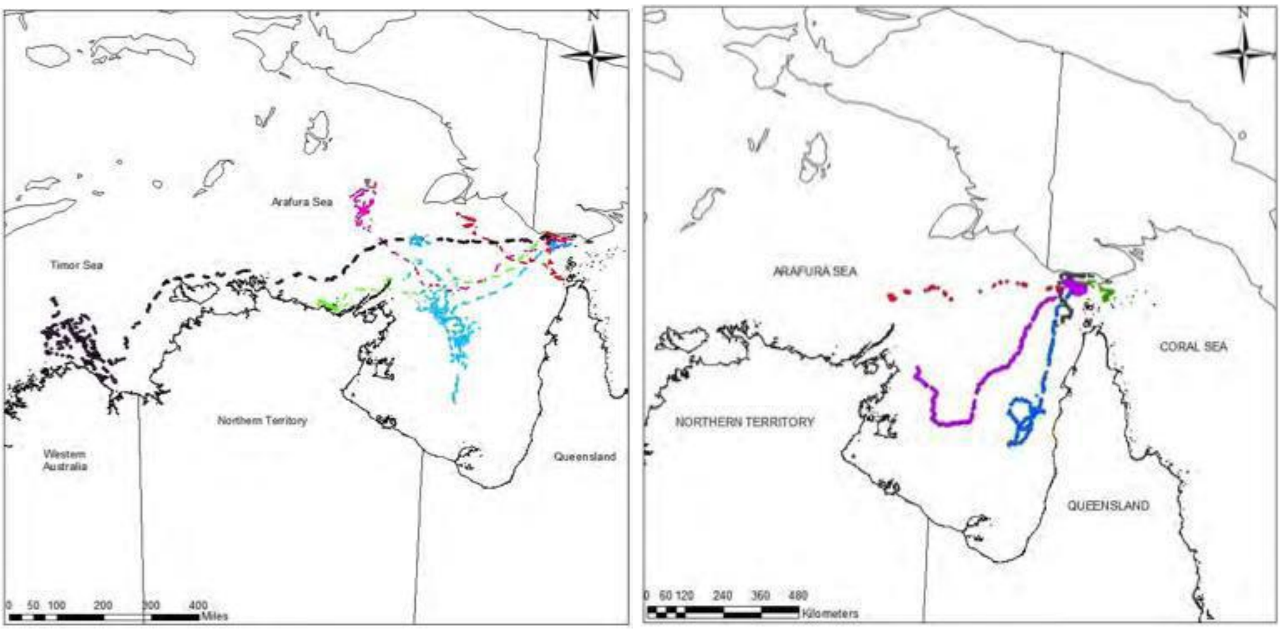


Figure 9-40. Migration routes and foraging areas for five female flatback turtles after nesting at Warul Kawa in 2013 (left) and six female flatback turtles after nesting at Warul Kawa in 2014. Image source: Hamann 2015.

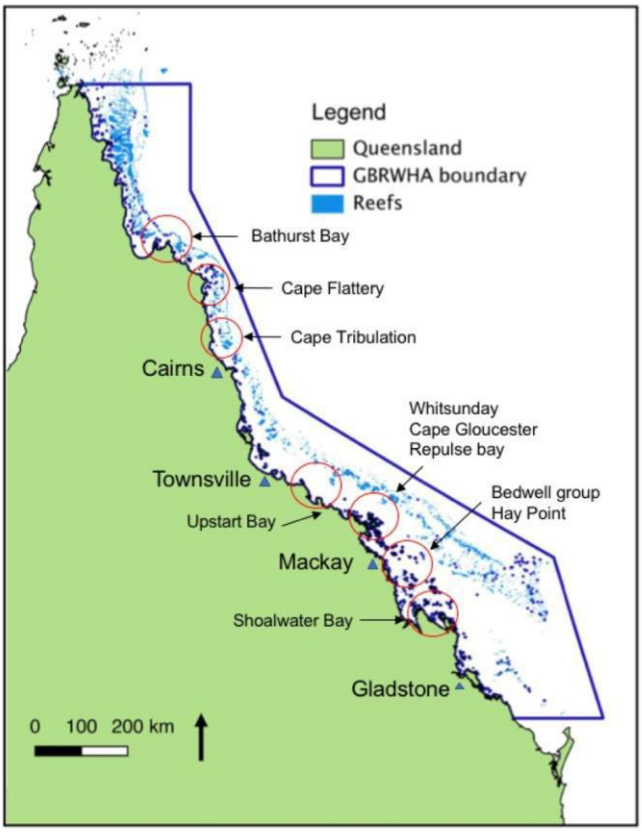


Figure 9-41. Migration routes of flatback turtles after nesting at Curtis island 2013–2017. Image source: Hamann et al. 2017

In short, dispersal of flatbacks has only a minor international implication, with the vast majority of movements of post-hatchlings and adults in and around Australian waters. Satellite tracking indicates flatbacks can move from the Torres Straits and the nGBR into the Arafura and Timor Seas, but there is limited movement into the Pacific from flatback turtles nesting in northern Australia.

9.6 Olive ridley turtle migrations

There is far less known of migratory behaviour of olive ridley turtles in the Pacific. Post-reproductive olive ridley sea turtles in the eastern tropical Pacific (ETP) were widely distributed in the pelagic zone from Mexico to Peru and lacked migratory corridors (Plotkin 2010). Plotkin (2010) also determined that olive ridleys migrated long distances, displayed no fidelity to specific feeding habitats, and were nomadic (Figure 9-42). Similar findings were found for rodeo-caught olive ridley turtles during the Stenella Abundance Research Survey (STAR) in the ETP (Figure 9-43; Parker et al. 2003) and tracking post-capture olive ridleys in the north central Pacific (Figure 9-43; Polovina et al. 2004). A biophysical model of connectivity and a seascape genetic analysis based on nuclear DNA variation of samples collected from 27 nesting areas identified two genetically distinct populations largely isolated during reproductive migrations and mating (roughly described as Mexico and Central America), each composed of multiple nesting sites linked by high connectivity (Rodriguez-Zarate et al. 2018). This was supported by Lagrangrian particle distribution modelling, which clearly identified the two populations (Figure 9-44; Rodriguez-Zarate et al. 2018). This work was further supported by an mtDNA study of ETP olive ridleys, which found that broadly dispersed family lineages had proliferated in the ETP, showing basin-wide connectivity and shallow population structure in the ETP (Silver-Gorges et al. 2019).

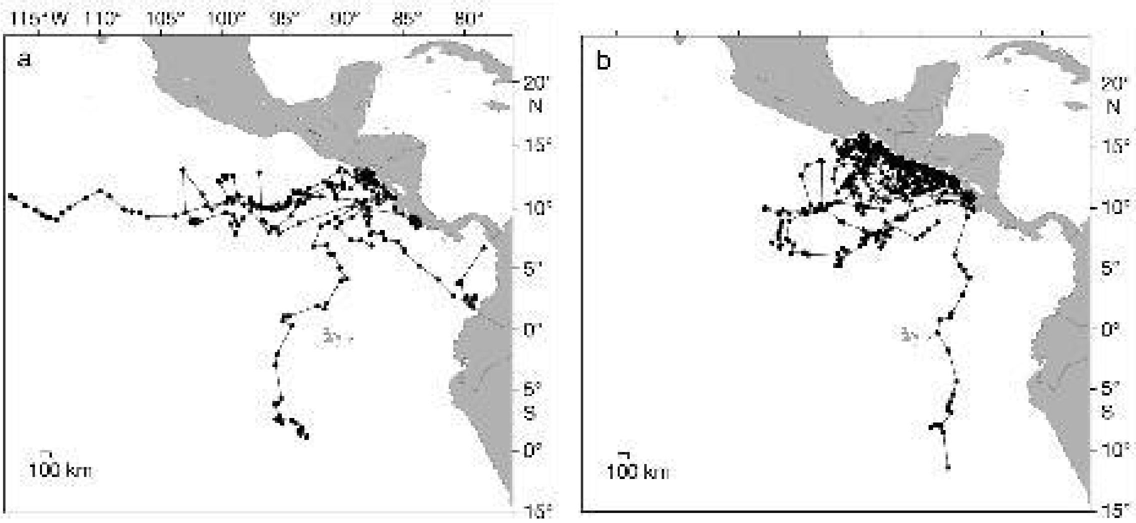


Figure 9-42. Post-nesting migratory tracks of adult female olive ridley turtles from Costa Rica in (a) 1990–1991 and (b) 1992. Image source: Plotkin 2010.

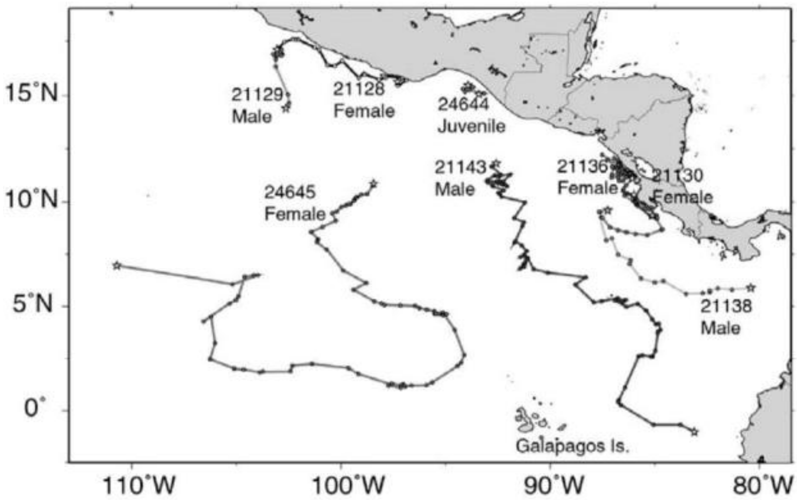


Figure 9-43. Post-release movements of olive ridley turtles (four female, three male, and one juvenile) that were released during the 1999 Stenella Research Cruise in the ETP. Image source: Parker et al. 2003.

In the north central Pacific olive ridleys similarly displayed no fidelity to specific feeding habitats, generally remaining south of 28°N in the subtropical gyre and the equatorial region (Figure 39), exhibiting both eastward and westward movements (Polovina et al. 2004).

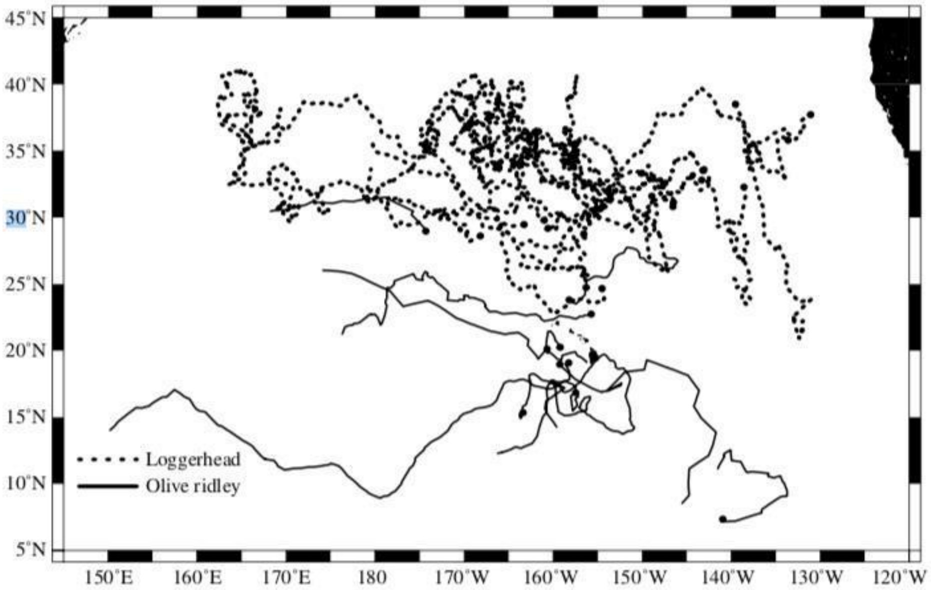


Figure 9-43. Post-release movements of 10 olive ridley turtles (solid lines) captured and released from longline fishing fleets. Image source: Polovina et al. 2004.

The Australian populations of olive ridley turtles are relatively small and their distribution is limited to remote areas in the northern part of the country. Olive ridley turtles from northern Australia tend to originate primarily from rookeries in SE Asia and do not appear to be highly connected to the Pacific Ocean (Jensen et al. 2013). Indeed, tracks from one study in northern Australia documented how the turtles were highly resident and noted that foraging behaviour differs from the meandering oceanic movements of turtles in the Pacific (Figure 39; Whiting et al. 2007). Similar lacks of long-distance movements in northern Australia sea turtles have been documented by McMahon et al. (2007) and Hamel et al. (2008).

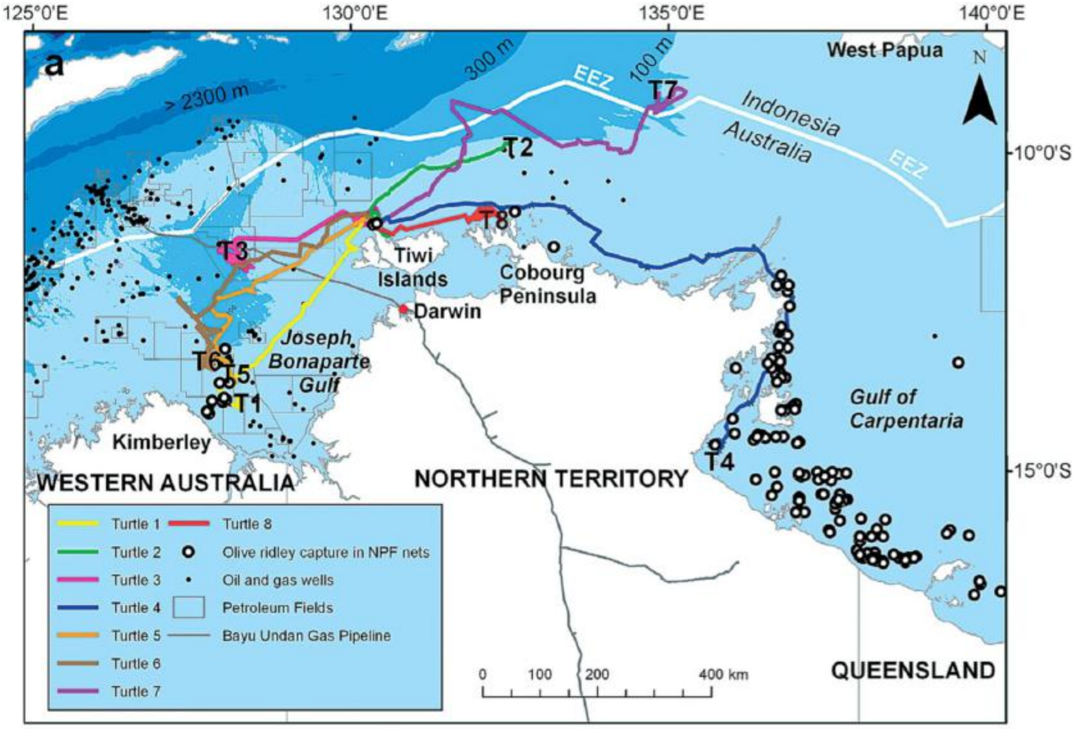


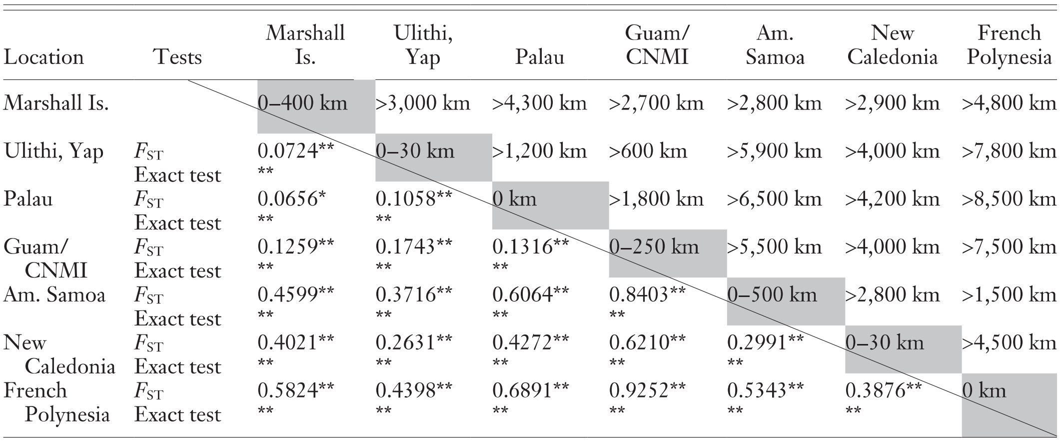
Figure 9-44. Post-release movements of eight olive ridley turtles from on Turtle Melville Island, northern Australia, in 2004 and 2005 demonstrating that none of the turtles reached the Pacific. Image source: Whiting et al. 2007.

10.0 Genetic Structure and Population Connectivity

10.1 Green turtles

Dutton et al. (2014) assessed the genetic structure of green turtles throughout the WCPO using mtDNA from 805 turtles sampled at 25 nesting locations. They examined whether sequencing longer fragments (770 bp) of the control region increased the resolution of stock structure and used genetic analysis to evaluate level of demographic connectivity among island nesting populations in the WCPO. They identified a total of 25 haplotypes, including five new variants that were indistinguishable from previous studies. Dutton et al. (2014a) concluded that there were at least seven independent stocks in the Pacific region and that rookeries separated by more than 1,000 km were significantly differentiated from each other, while neighbouring rookeries within 500 km showed no genetic differentiation (Table 10-1).

Table 10-1. Pairwise comparisons of the seven genetic stocks in the WCPO. Note: FST values (conventional haplotype frequencies) and results for the exact test are shown below the diagonal. P values are shown as <.05 (*) and <.005 (**). Approximate shortest distance between the stocks is shown above the diagonal. Grey areas show the overall distance between sampling locations within each stock. Table Source: Dutton et al. 2014



Location	Tests	Marshall Is.	Ulithi, Yap	Palau	Guam/ CNMI	Am. Samoa	New Caledonia	French Polynesia
Marshall Is.		0–400 km	>3,000 km	>4,300 km	>2,700 km	>2,800 km	>2,900 km	>4,800 km
Ulithi, Yap	F_{ST}	0.0724**	0–30 km	>1,200 km	>600 km	>5,900 km	>4,000 km	>7,800 km
	Exact test	**						
Palau	F_{ST}	0.0656*	0.1058**	0 km	>1,800 km	>6,500 km	>4,200 km	>8,500 km
	Exact test	**	**					
Guam/ CNMI	F_{ST}	0.1259**	0.1743**	0.1316**	0–250 km	>5,500 km	>4,000 km	>7,500 km
	Exact test	**	**	**				
Am. Samoa	F_{ST}	0.4599**	0.3716**	0.6064**	0.8403**	0–500 km	>2,800 km	>1,500 km
	Exact test	**	**	**	**			
New Caledonia	F_{ST}	0.4021**	0.2631**	0.4272**	0.6210**	0.2991**	0–30 km	>4,500 km
	Exact test	**	**	**	**	**		
French Polynesia	F_{ST}	0.5824**	0.4398**	0.6891**	0.9252**	0.5343**	0.3876**	0 km
	Exact test	**	**	**	**	**	**	

The Hawaiian green turtle was deemed to be genetically distinct from the green turtles in the WCPO (Dutton et al. 2008), after mtDNA control region sequences from immature and adult green turtles that forage around the Hawaiian Islands were compared to potential source nesting populations across the Pacific (Frey et al. 2013, Dutton et al. 2014a). Stock mixture analysis indicated that the Hawaiian foraging ground populations comprised one genetic stock derived from the nesting population at French Frigate. Evolutionarily, Hawaii green turtles were more closely related to the eastern Pacific than western Pacific populations (Dutton et al. 2014b).

Somewhat similar to Hawaii, the green turtles nesting in the Ogasawara Islands in Japan appear to be genetically isolated from the central and eastern Pacific stocks (Hamabata 2020). The uniqueness of the Ogasawara population in genome-wide single nucleotide polymorphisms identified that Ogasawara turtles were more likely to have been born on the Ogasawara Islands.

The east Pacific turtles are yet again genetically distinct from the WCPO (Dutton et al. 2014b). There is also limited gene flow outside of the east Pacific region. For example, Álvarez-Varas et al. (2017) demonstrated that the green turtles at Bahía Salgado (Chile), the southernmost foraging ground for east Pacific green turtles, had a strong genetic connectivity to the Galapagos green turtle rookeries.

The split between sampled central/ eastern and western Pacific haplotypes was estimated at around 0.34 million years ago, suggesting that the Pacific region west of Hawaii has been a more formidable barrier to gene flow in green turtles than the east Pacific Barrier. Dutton et al. (2014b) suggest that the eastern Pacific was colonised from the western Pacific via the central north Pacific and that the Revillagigedos Islands provided a stepping-stone for radiation of green turtles from the Hawaiian Archipelago to the eastern Pacific.

However, genetic composition is not as straightforward as having a country's turtles belonging to one or another stock. In Taiwan, for instance, Cheng et al. (2008) documented differences in genetic composition across two small Taiwanese islands that were in relatively close proximity (~200 km), and noted a lack of gene flow between the two rookeries. Similarly in Australia, Moritz et al. (2000) and Dethmers et al. (2006) demonstrated that haplotype frequency distributions indicated 17 genetically distinct breeding stocks for turtles foraging in Australian waters, and that individual rookeries or groups of rookeries were separated by more than 500 km. Of note however, this study demonstrated a significant discontinuity in genetic structure between Pacific Ocean stocks and those found further to the west (Figure 10-1).

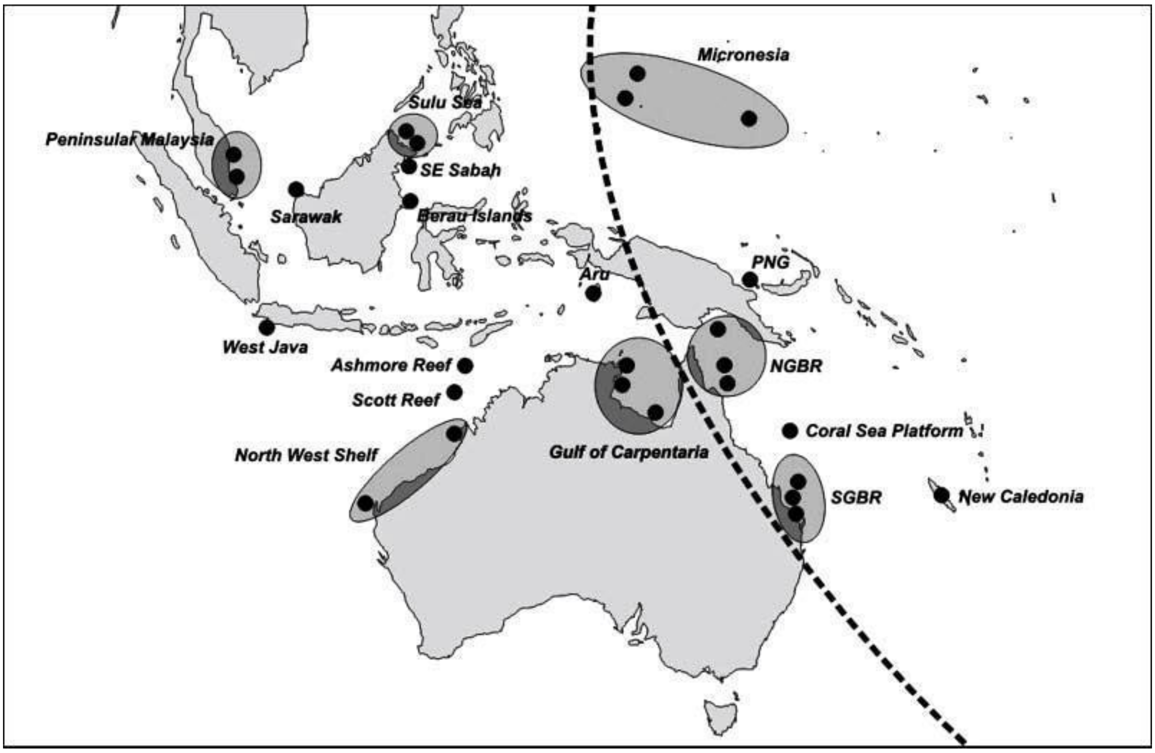


Figure 10-1. Location of 17 genetically distinct breeding stocks or management units as inferred from analysis of geographical structure of mtDNA variants and position of the genetic barrier (dashed line), indicating the major genetic discontinuity between the Pacific Ocean rookeries from those to the west. Image source: Dethmers et al. 2006.

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. 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. 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. 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 northwest 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 Pacific region.

10.2 Hawksbill turtles

A study on the global phylogeography of the critically endangered hawksbill turtle 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 is also a trend of foragers coming from nearby natal beaches. 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 Mexico, Nicaragua, El Salvador, Ecuador, Costa Rica and Panama hawksbills formed another group, and a third group in the Northern Territory and North Queensland, Australia (Figure 10-20). Vargas et al. (2016) noted that hawksbill turtles had a complex pattern of phylogeography, showing a weak isolation by distance and evidence of multiple colonisation events. This explains the shared haplotypes across much of the Pacific region (pink colours, Figure 10-2).

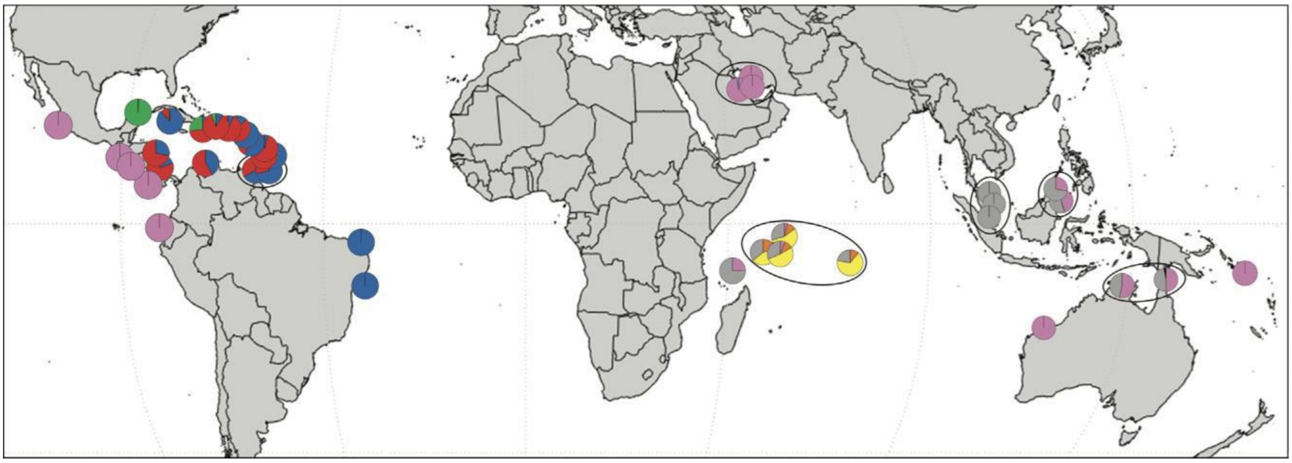


Figure 10-2 Frequencies of control region haplotypes (739 bp) from each of nine mtDNA lineages in the hawksbill turtle rookeries. Image source: Arantes et al. 2020.

Findings by Arantes et al. (2020) were mirrored by findings using mtDNA studies within Australia (Broderick et al. 1994). Genetic analysis indicated that there was one stock that incorporated the hawksbill rookeries of the northern Great Barrier Reef (GBR), Torres Strait and Arnhem Land that was independent of a second stock that breeds at rookeries on the northwestern 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.

In the east Pacific, Gaos et al. (2016) found that hawksbills demonstrated low genetic diversity, with only seven haplotypes across the region. Despite the low genetic diversity, they found strong stock structure between the four principal rookeries, suggesting the existence of multiple populations and warranting their recognition as distinct management units. Gaos et al. (2016) concluded that the low overall genetic diversity in the eastern Pacific is likely the combined result of few rookeries, extremely small reproductive populations and evolutionarily recent colonisation events. However, Zunige-Marroquin et al. 2017 identified haplotypes from the east Pacific turtle RMU, and concluded that some of the hawksbill hatchlings or juveniles from the eastern Pacific could be moving to the Indo-Pacific.

10.3 Loggerhead turtles

There are two distinct breeding stocks in the Pacific Ocean (Bowen et al. 1994, 1995) – extensive mtDNA studies show that the northern loggerhead populations are isolated from the southern Pacific populations. The north Pacific nesting occurs in Japan (Matsuzawa et al. 2016), while nesting in the south Pacific occurs primarily in Australia, with some loggerhead nesting in New Caledonia, Vanuatu and Tokelau. The divergence of haplotypes from both north and south Pacific loggerheads indicates that these two stocks have been isolated for approximately one million years (Bowen 2003). The loggerhead population that breeds at Mon Repos, Queensland is genetically different from the population that breeds in southern Japan (Bowen et al. 1994). Preliminary genetic analyses of the loggerhead populations of eastern and western Australia indicate that they constitute separate stocks (Dutton et al. 2002). Current scientific evidence, based on genetic analysis, flipper tag recoveries, and satellite telemetry, indicates that individuals originating from Japan remain in the north Pacific for their entire life cycle, never crossing the equator or mixing with individuals from the south Pacific (Hatase et al. 2002, LeRoux and Dutton 2006). Although studies indicate genetic distinctness between loggerheads nesting in Japan versus those nesting in Australia, Bowen et al. (1995) did identify individuals with the common Australian haplotype at foraging areas in the north Pacific, based on a few individuals sampled as by-catch in the north Pacific. More recently, Hatase et al. (2002) also detected this common haplotype at very low frequency at Japanese nesting beaches. However, the presence of the common Australian haplotype does not preclude the genetic distinctiveness of Japanese and Australian nesting populations, and is likely the result of rare gene flow events occurring over geologic time scales.

10.4 Leatherback turtles

There are two distinct breeding stocks of leatherbacks in the Pacific Ocean (Dutton et al. 1999) – those that nest in the east Pacific (Mexico, Costa Rica, Panama) and those that nest in the west (Indonesia, Papua New Guinea, Solomon Islands and Vanuatu). In the west Pacific, genetic analysis by using mtDNA sequences identified a total of 6 haplotypes among the 106 samples analysed for Solomon Islands, Papua, and Papua New Guinea, including a unique common haplotype that is only found in these western Pacific populations (Dutton et al. 2007). Barragan et al. (1998) demonstrated that leatherbacks in the Pacific had a high level of gene flow was found between rookeries but not with the western Pacific.

10.5 Flatback turtles

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 10-3; 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 1,300 km (Figure 10-4). Genetic divergence was consistently higher between the eastern Queensland rookeries and all other rookeries, highlighting the genetic distinction of the flatback turtles in the Pacific Ocean from other flatbacks in Australia.

FitzSimmons et al. (2020) noted that discontinuities in haplotype frequencies among rookeries may reflect historical patterns of low-frequency colonisation 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.

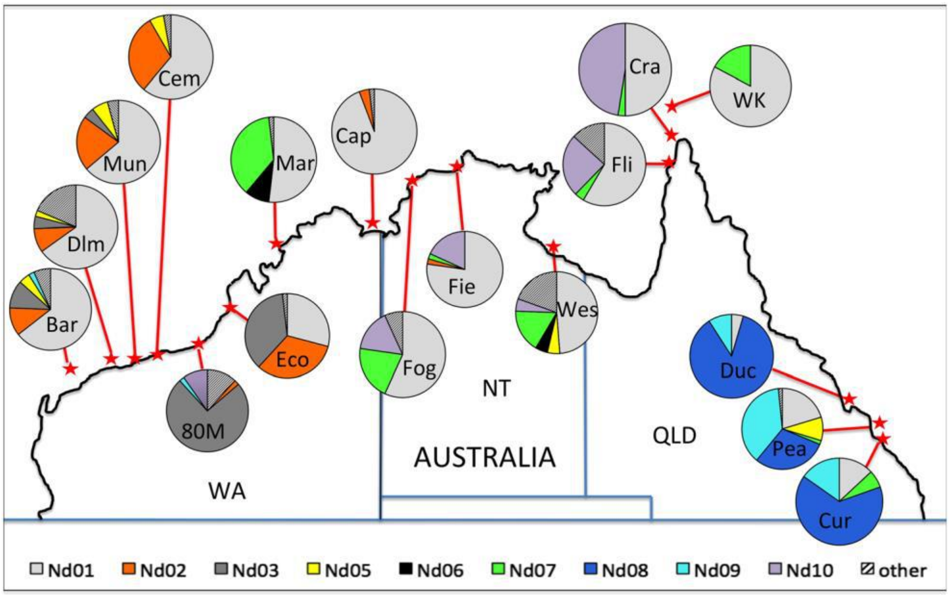


Figure 10-3 Distribution of the nine most common mitochondrial DNA haplotypes and combined 'other' category, sampled from 17 flatback turtle (*Natator depressus*) rookeries. Image source: FitzSimmons et al. 2020.

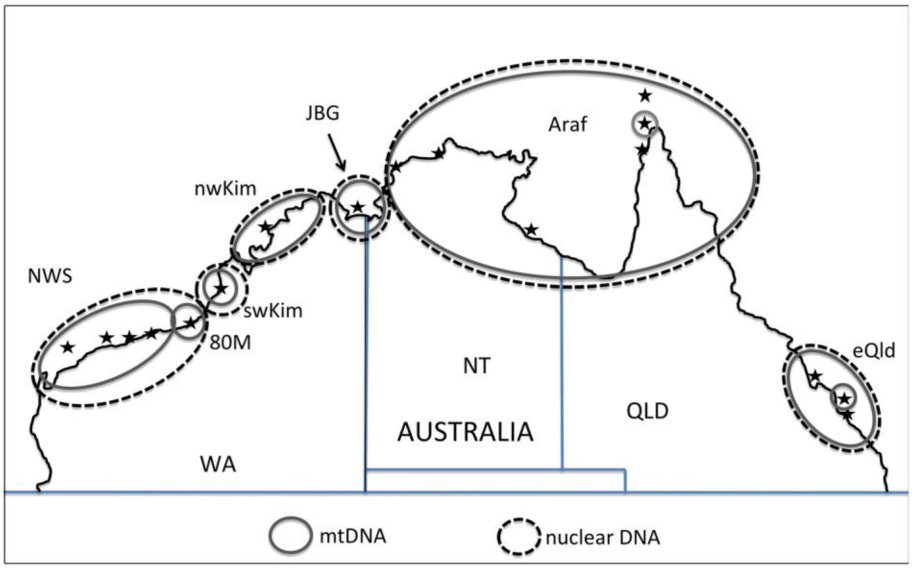


Figure 10-4 Designated flatback turtle (*Natator depressus*) genetic stocks based on the analyses of 17 rookeries across their range. Image source: FitzSimmons et al. 2020.

10.6 Olive ridley turtles

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. Genetic analyses of olive ridleys entangled in these ghost nets indicated these turtles came from nesting populations within the Northern Territory, but also haplotypes not found in the Northern Territory were recorded, suggesting turtles may have come from Indonesia or Papua New Guinea (Jensen et al. 2013). In the east Pacific Ocean, rookeries in Costa Rica and Mexico, separated by more than 500 km, were not genetically distinct (Bowen et al. 1998; Hahn et al. 2012). Fine-scale population structure also was not found across 13 solitary and arribada nesting beaches along the Baja Peninsula and the main coast of Mexico (Rodríguez-Zárate et al. 2013, 2018). Lopez-Castro and Rocha-Olivares (2005) found genetic diversity in solitary nesting assemblages from the Baja California Peninsula to be significantly lower than arribada nesting populations along the east Pacific coast of Mexico and Costa Rica, and concluded there was reproductive isolation and genetic differentiation.

Lopez-Castro and Rocha-Olivares (2005) demonstrated genetic differences between the solitary and arribada nesting populations, and Jensen et al. (2006) found a significant increase in multiple paternity (i.e. more than one male fertilising eggs in a clutch) in nests from arribada beaches and attributed population size and the associated increase in male encounter rates as the major factor. These studies demonstrate that solitary and arribada mating systems are distinct from each other, and explain the distinction in the two east Pacific RMUs.

Annex A: Summary of annual nester abundance by species and RMU for sea turtles in the Pacific.

	RMU	Green		RMU	Hawksbill		RMU	Loggerhead	
		Estimate	Bin		Estimate	Bin		Estimate	Bin
Hawaii	NC	450	101-500	NC	10-20	11-50			
Palmyra	NC	0	0	NC	1	1-10			
Johnston Atoll	NC	0	0	NC	0	0			
American Samoa	SC	25	11-50	SC	10-15	11-50			
Cook Islands	SC	<50	11-50	SC	0	0			
Fiji	SC	50-75	51-100	SC	20-30	11-50			
French Polynesia	SC	50-100	51-100	SC	na	na			
Kiribati	SC	110-350	101-500	SC	na	na			
Nauru	SC	0	0	SC	na	na			
Niue	SC	0	0	SC	na	na			
Pitcairn Islands	SC	10	1-10	SC	na	na			
Samoa	SC	0	0	SC	1-5	1-10			
Tokelau	SC	120	101-500	SC	na	na			
Tonga	SC	10-20	11-50	SC	na	na			
Tuvalu	SC	10	1-10	SC	na	na			
Wallis & Futuna	SC	0	0	SC	0	0			
CNMI	CW	<10	1-10	WC	na	na			
Guam	CW	1-4	1-10	WC	5-10	1-10			
FSM	CW	500-1000	501-1000	WC	10-20	11-50			
Japan	CW	500	101-500	WC	10-20	11-50	N	1600-3000	1001-5000
PNG	CW	na	na	SW	<500	101-500			
Philippines	CW	3000-4000	1001-5000	WC	20-30	11-50			
Marshall Islands	CW	100-500	101-500	WC	na	na			
Palau	CW	100-150	101-500	WC	20-50	11-50			
Australia	SW	15000-30000	>10000	SW	500-1500	1001-5000	S	500	501-1000
New Caledonia	SW	920-2300	1001-5000	SW	0	0	S	200	101-500
Solomon Islands	SW	<300	101-500	SW	125-150	101-500			
Vanuatu	SW	60-100	51-100	SC	300	101-500			
Indonesia	SW	na	na	SW	na	na			
Chile	E	0	0	E	0	0			
Colombia	E	10	1-10	E	0	0			
Costa Rica	E	1500	1001-5000	E	1-5	1-10			
Ecuador	E	600-700	501-1000	E	10-15	11-50			
El Salvador	E	<10	1-10	E	120	101-500			
Guatemala	E	<10	1-10	E	1	1-10			
Honduras	E	na	na	E	na	na			
Mexico	E	3500	1001-5000	E	1-2	1-10			
Nicaragua	E	0	0	E	95	51-100			
Panama	E	400-500	101-500	E	0	0			
Peru	E	1	0	E	0	0			

	RMU	Leatherback		RMU	Flatback		RMU	Olive Ridley (solitary)	
		Estimate	Bin		Estimate	Bin		Estimate	Bin
Hawaii									
Palmyra									
Johnston Atoll									
American Samoa									
Cook Islands									
Fiji									
French Polynesia									
Kiribati									
Nauru									
Niue									
Pitcairn Islands									
Samoa									
Tokelau									
Tonga									
Tuvalu									
Wallis & Futuna									
CNMI									
Guam									
FSM							W	<5	1-10
Japan									
PNG	W	50-200	101-500				W	na	na
Philippines							W	15	1-10
Marshall Islands							W	1	1-10
Palau									
Australia	W	<5	1-10	W	3000-4500	1001-5000	W	<500	101-500
New Caledonia									
Solomon Islands	W	30-50	11-50				W	na	na
Vanuatu									
Indonesia	W	500	101-500						
Chile									
Colombia							E	200	101-500
Costa Rica	E	35	11-50				E	na	na
Ecuador							E	1	1-10
El Salvador							E	na	na
Guatemala							E	1000	1001-5000
Honduras							E	na	na
Mexico	E	125	101-500				E	7000	5001-10000
Nicaragua	E	4	1-10				E	na	na
Panama							E	10000	>10000
Peru							E	0	0

	RMU	Olive Ridley (arribada)	
		Estimate	Bin
Costa Rica	E	3000-10000	5001-10000
Mexico	E	500000-600000	>500000
Nicaragua	E	50000-500000	>100000
Panama	E	8000-9000	5001-10000

Annex B: Summary of clutch frequencies by species and country in the Pacific.

	Green	Hawksbill	Loggerhead	Leatherback	Flatback	Olive Ridley
Hawaii	1.8, 4.96					
Palmyra						
Johnston Atoll						
American Samoa						
Cook Islands						
Fiji						
French Polynesia						
Kiribati						
Nauru						
Niue						
Pitcairn Islands						
Samoa						
Tokelau						
Tonga						
Tuvalu						
Wallis & Futuna						
CNMI	7.0					
Guam						
FSM						
Japan	4.1, 3.9		4.6, 2.1, 3.7			
PNG				2.2, 2.9		
Philippines						
Marshall Islands						
Palau						
Australia	5.6	2.4, 3.0	3.4		2.8	
New Caledonia	4.1					
Solomon Islands		3.0, 5.0				
Vanuatu						
Indonesia				5.5		
Chile						
Colombia						
Costa Rica	4.3			3.2, 5.6, 9.5		2.2, 1.1
Ecuador						
El Salvador		2.1				
Guatemala						
Honduras						
Mexico	3.5, 4.0, 3.1			5.5		
Nicaragua						
Panama						
Peru						

Note: Multiple values indicate results from different studies. Citations and additional values are provided in Section 7.7.

Annex C: Summary of clutch sizes by species and country in the Pacific.

	Green	Hawksbill	Loggerhead	Leatherback	Flatback	Olive Ridley
Hawaii	104, 92.4					
Palmyra						
Johnston Atoll						
American Samoa						
Cook Islands	81-181					
Fiji						
French Polynesia	76.5					
Kiribati						
Nauru						
Niue						
Pitcairn Islands						
Samoa						
Tokelau						
Tonga						
Tuvalu						
Wallis & Futuna						
CNMI	93.5					
Guam						
FSM						
Japan	102.0		122.0			
PNG	107.3			94.7, 79.3, 88.2		
Philippines						
Marshall Islands						
Palau						
Australia	112,111,102.1	120.1, 121.7	127.0, 124.4	97.7	53.8, 55.9, 52.0	
New Caledonia	112.0					
Solomon Islands	84.6	151.0		88.2		
Vanuatu						
Indonesia				79.6, 76.2		
Chile						
Colombia						
Costa Rica	64.7, 76.9			61.8, 64.7, 62.0		99.5, 107.0, 87.5
Ecuador	71.2					
El Salvador		132.4, 167.8				
Guatemala						
Honduras						
Mexico	69.2	159.1		66.0,62.0		102.0
Nicaragua		150.9, 154.8				
Panama						
Peru						

Note: Multiple values indicate results from different studies. Citations and additional values are provided in Section 7.8.

Annex D: Summary of hatching success rates by species and country in the Pacific.

	Green	Hawksbill	Loggerhead	Leatherback	Flatback	Olive Ridley
Hawaii	71.1, 81.1					
Palmyra						
Johnston Atoll						
American Samoa						
Cook Islands						
Fiji						
French Polynesia	94.4					
Kiribati						
Nauru						
Niue						
Pitcairn Islands						
Samoa						
Tokelau						
Tonga						
Tuvalu						
Wallis & Futuna						
CNMI	77.9					
Guam						
FSM						
Japan			74.9, 24.0, 53.0			
PNG				49.0		
Philippines						
Marshall Islands						
Palau						
Australia	84.1, 41.0, 73.9	90.9, 80.0, 85.2	81.9, 80.2, 78.2		60.5, 83.4, 74.6	
New Caledonia	90.0					
Solomon Islands						
Vanuatu						
Indonesia				25.5, 47.1		
Chile						
Colombia						
Costa Rica	89.0, 75.0			41.0, 47.9, 51.4		17.7, 26.8, 78.5
Ecuador	46.0	59.7				
El Salvador		52.5, 72.3				
Guatemala						
Honduras						83.2, 24.1
Mexico		52.0		66.4		73.7
Nicaragua		59.8				
Panama						
Peru						

Note: Multiple values indicate results from different studies. Citations and additional values are provided in Section 7.9.

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