# Review of the Status of Sea Turtles in the Pacific Ocean
## 2021

EUROPEAN UNION

Sweden
Sverige

PEUMP
Pacific-European Union Marine Partnership Programme

SPREP
Secretariat of the Pacific Regional Environment Programme

**SPREP Library Cataloguing-in-Publication Data**

Pilcher, Nicolas J.

Review of the status of sea turtles in the Pacific
Ocean 2021. Apia, Samoa : SPREP, 2021

vi, 129 p. ; 29 cm.

ISBN: 978-982-04-1009-1 (print)
978-982-04-1008-4 (ecopy)

1 Conservation of marine species – Oceania
2. Sea turtles – Protection – Oceania.
I. Pacific Regional Environment Programme (SPREP).
II. Title.

597.92095

This document has been prepared and printed with the financial support of the Pacific-European Union Marine Partnership (PEUMP) programme, funded by the European Union and the Government of Sweden. Its contents do not necessarily reflect the views of the European Union or the Government of Sweden. This document has been compiled in good faith, exercising all due care and attention. SPREP does not accept responsibility for inaccurate or incomplete information.

This report should be cited as: Pilcher N.J. 2021. Review of the status of sea turtles in the Pacific Ocean 2021. Secretariat of the Pacific Regional Environment Programme, Apia, Samoa. 136 pp.

Cover image by: jakubgojda, Canva
Copyright all other images: Nicolas Pilcher, Marine Research Foundation, 2021.

**PO Box 240, Apia, Samoa**
**www.sprep.org**

**Our vision: The Pacific environment, sustaining our livelihoods and natural heritage in harmony with our cultures.**

# Review of the Status of Sea Turtles in the Pacific Ocean
# 2021

**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.

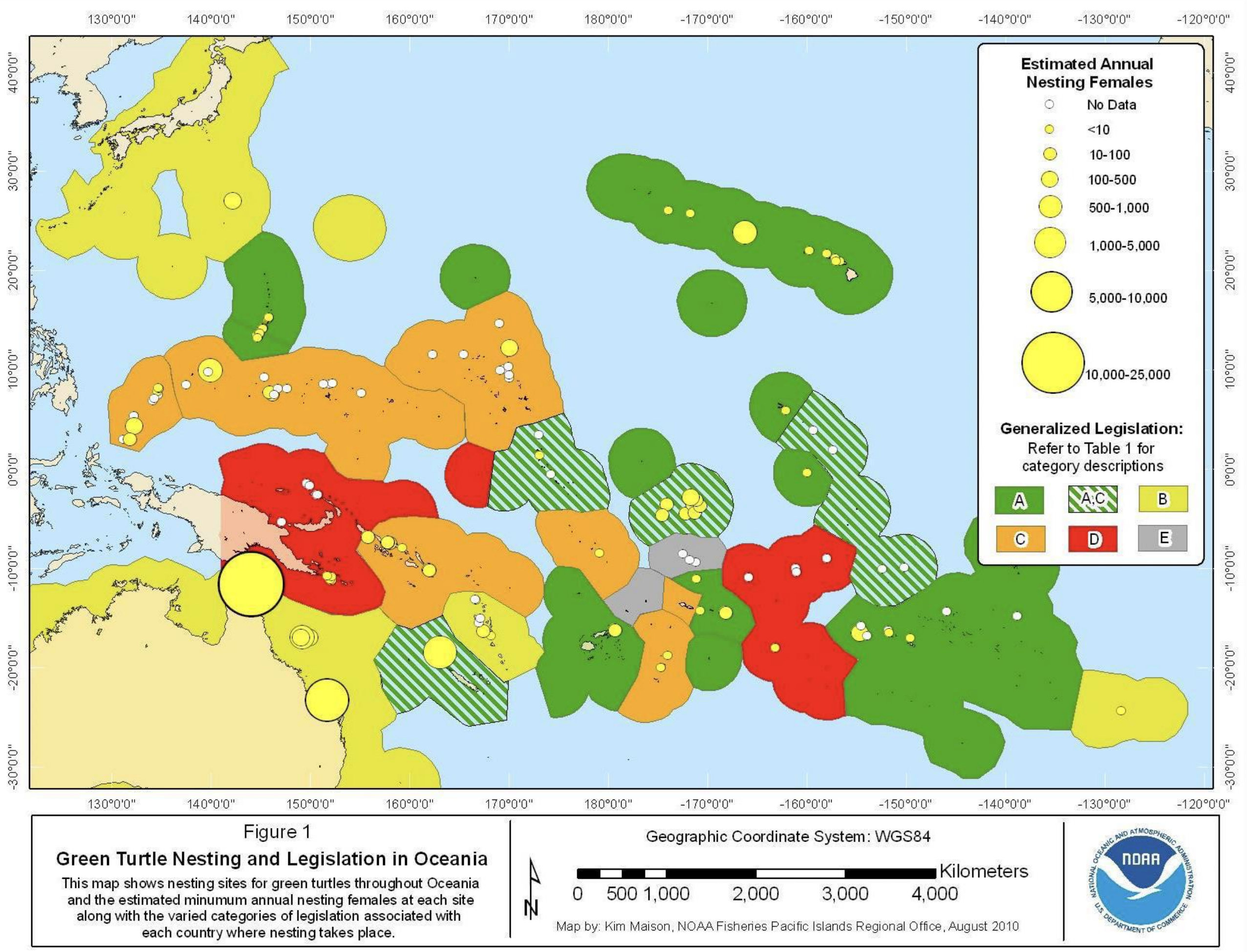


Figure 7-1. Green turtle nesting sites in the west and central Pacific. Source: Maison et al. 2010.

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).

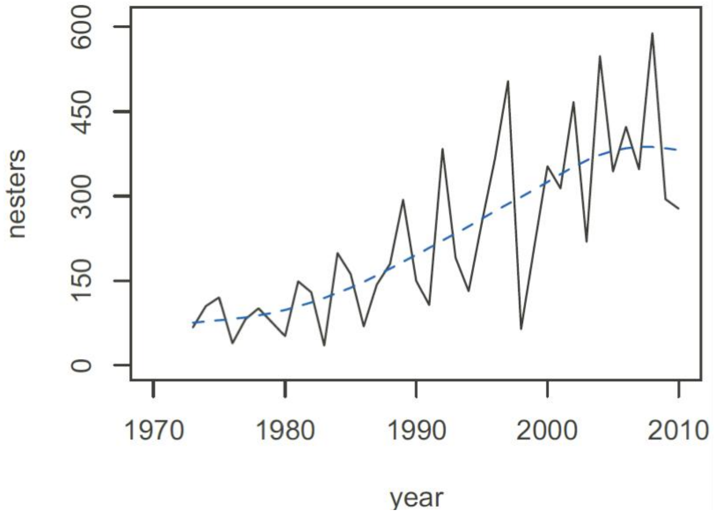


Figure 7-3. Trend in green turtle nests at French Frigate Shoals, Hawaii. Source: Pilcher et al. 2011

**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.

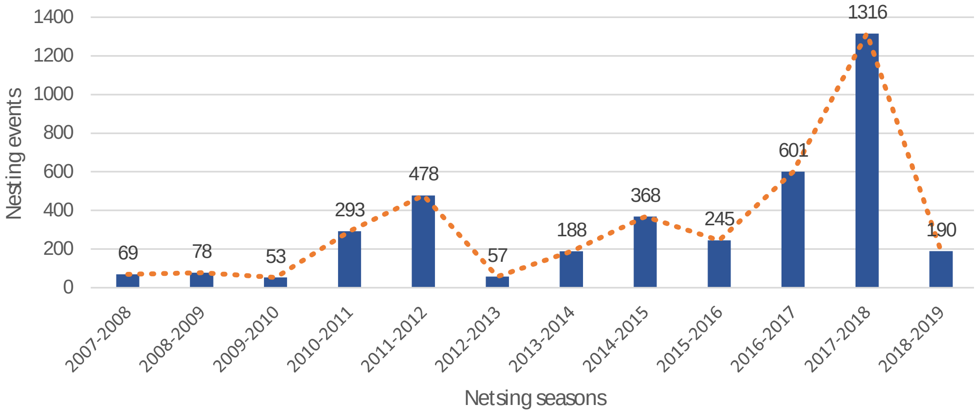


**Figure 7-4. Trend in green turtle nests at Tetiaroa, French Polynesia. Source: Tuoron et al. 2019**

**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.

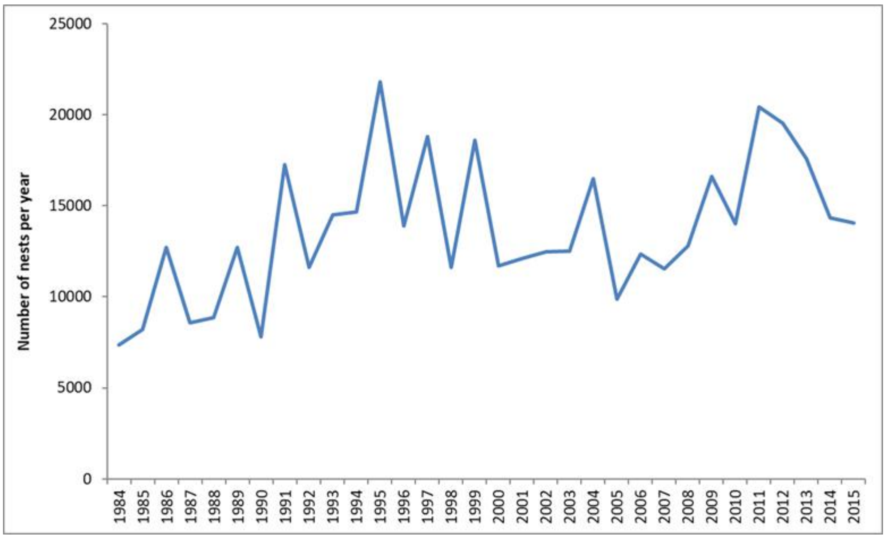


Figure 7-5. Trend in green turtle nests at the Turtle Islands Wildlife Sanctuary, Philippines. Source: Bureau of Marine Resources, unpublished data.

**Republic of the Marshall Island**s: 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.

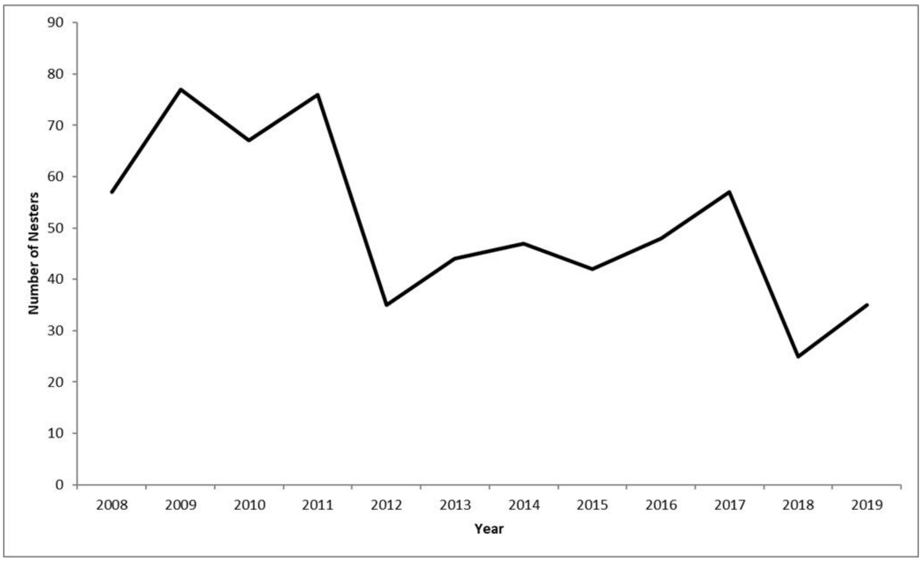


Figure 7-6. Trend in numbers of green turtles tagged at Hatohobei Island, Palau. Data source: Hatohobei State Government, unpublished data)

### 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).

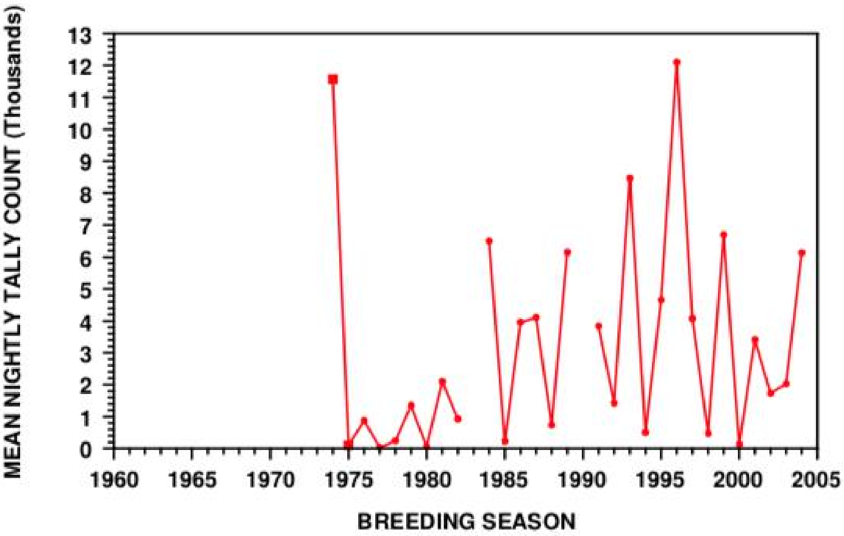


Figure 7-7. Trend in numbers of green turtles tagged at Raine Island, in the nGBR Australia. Data source: Limpus 2007a.

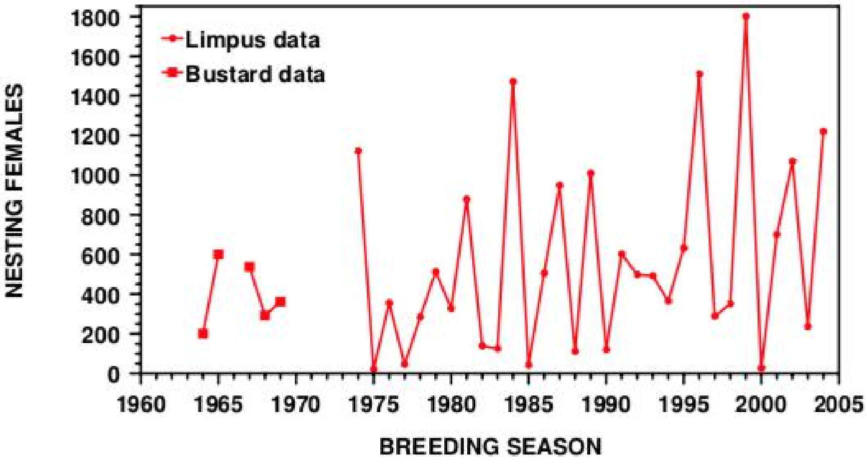


Figure 7-8. Trend in numbers of green turtles tagged at Heron Island in the sGBR, Australia. Data source: Limpus 2007a.

**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

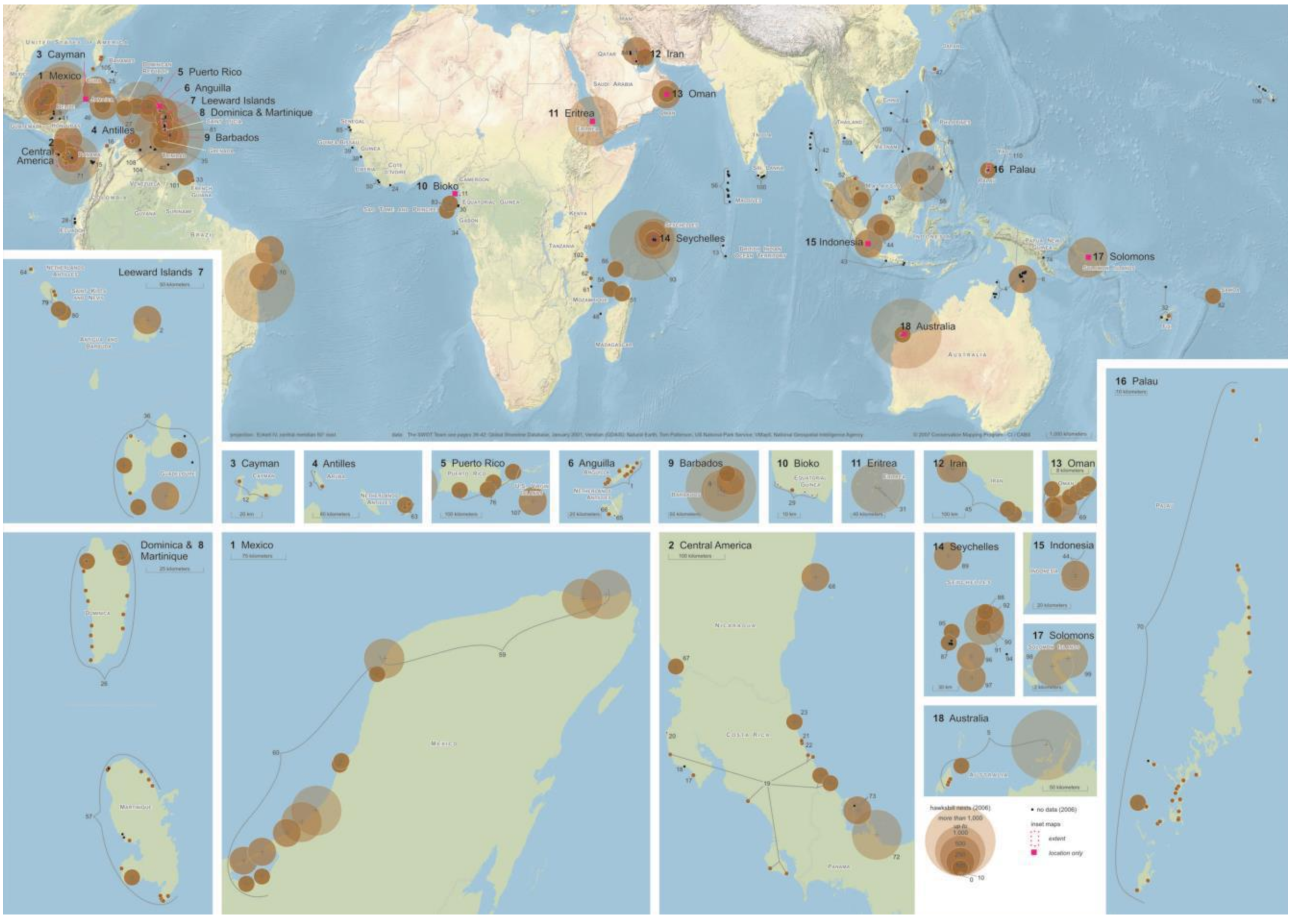


Figure 7-9. Global hawksbill nesting sites reported to SWOT in 2006. Image source: SWOT Report III.

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.

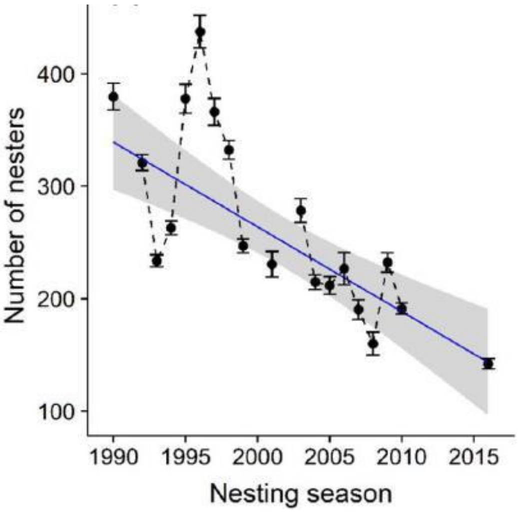


Figure 7-10. Modelled trend in numbers of hawksbill tagged nesting on Milman Island, Australia.

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.

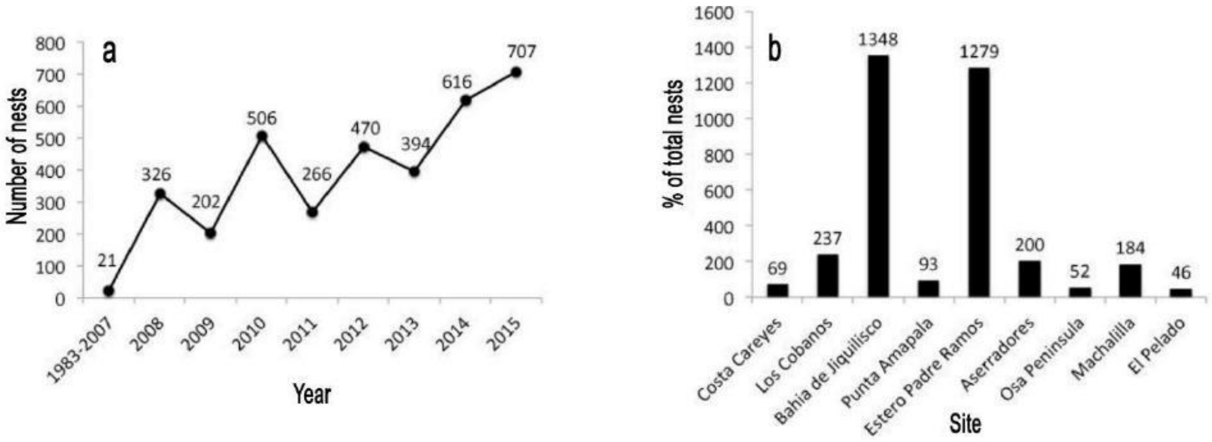


Figure 7-11. Trend in numbers of hawksbill nests (a) and proportional rookery contribution in the east Pacific hawksbill RMU. Data source: Gaos et al. 2017.

**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).

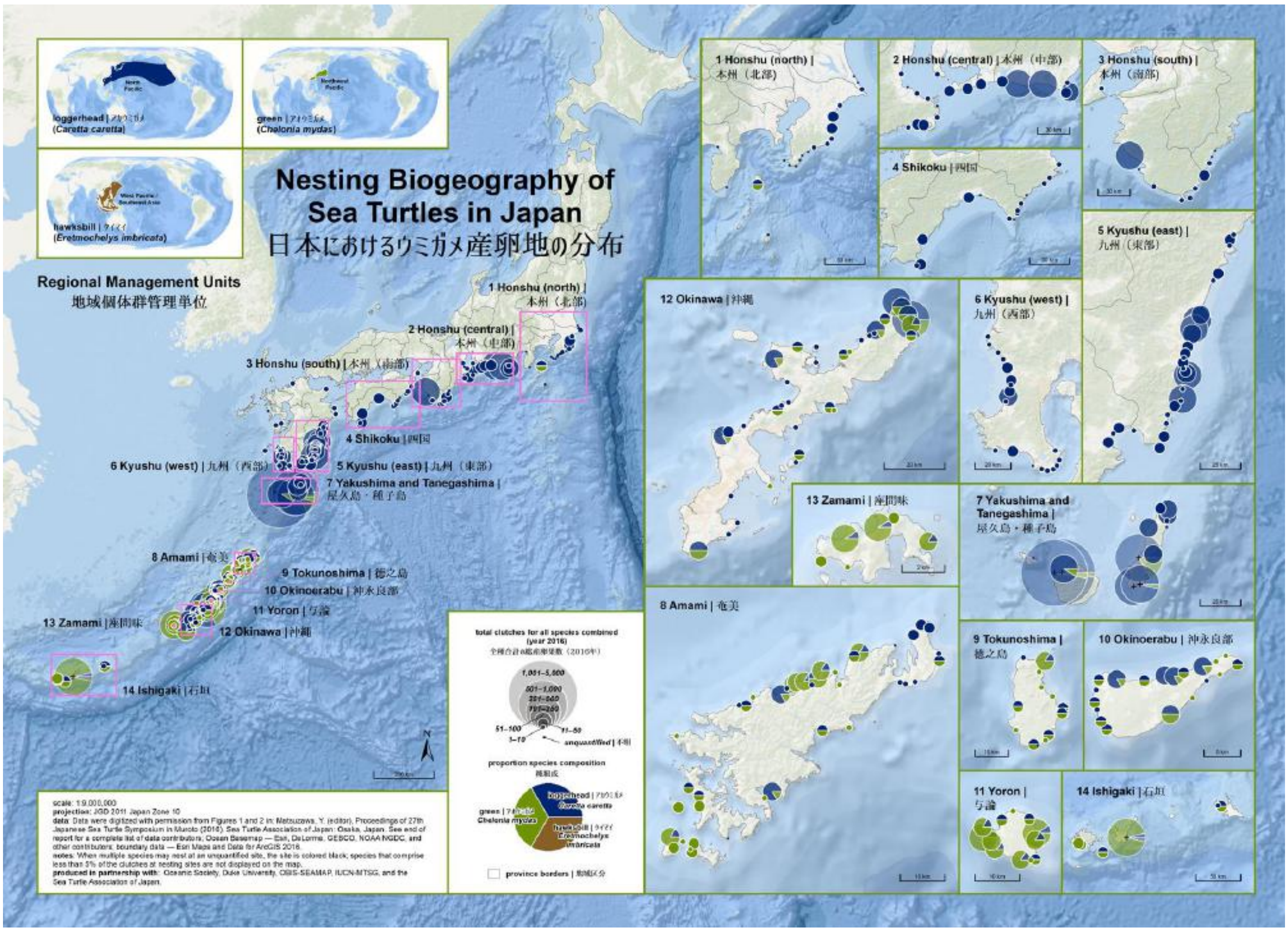


Figure 7-12. Nesting biogeography of sea turtles in Japan (blue circles denote loggerhead nesting). Image source: SWOT Report XIII.

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.

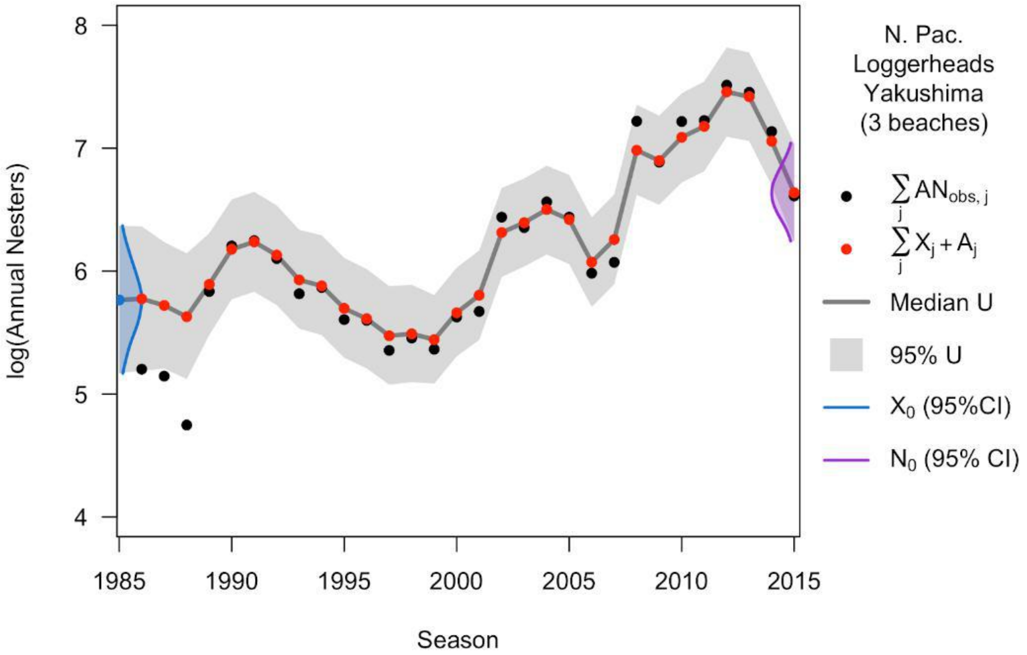


Figure 7-13. Model of nesting trend from 1985 to 2015 in Yakushima, Japan. The grey line depicts the median long-term trend. The grey shading depicts the 95% credible interval of the model fit. Observed data are shown in black and model-predicted data are shown in red. The purple line depicts the distribution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

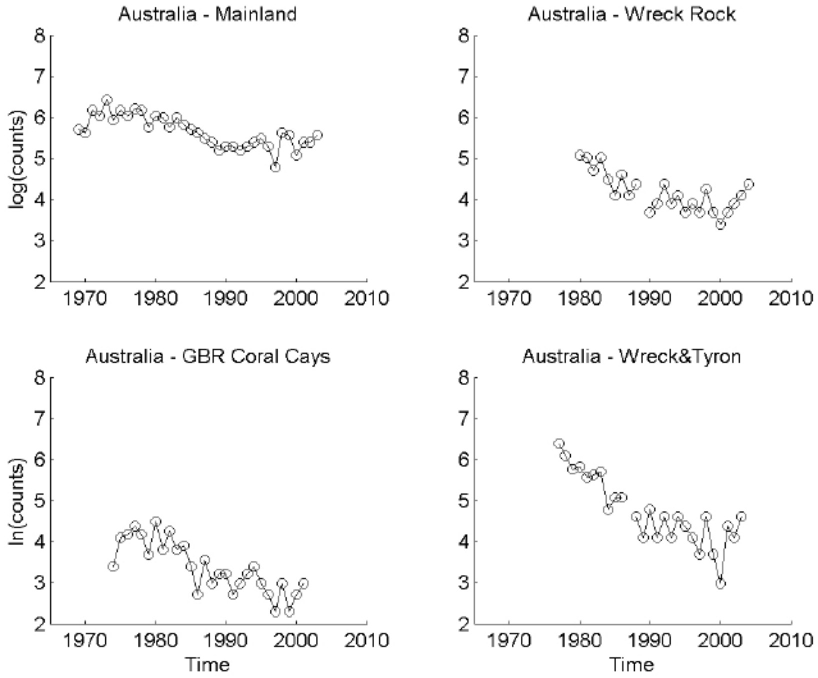


Figure 7-14. The change in the numbers of nesting females at four nesting beaches in the South Pacific Ocean RMU. Image source: Limpus 2007c.

**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.

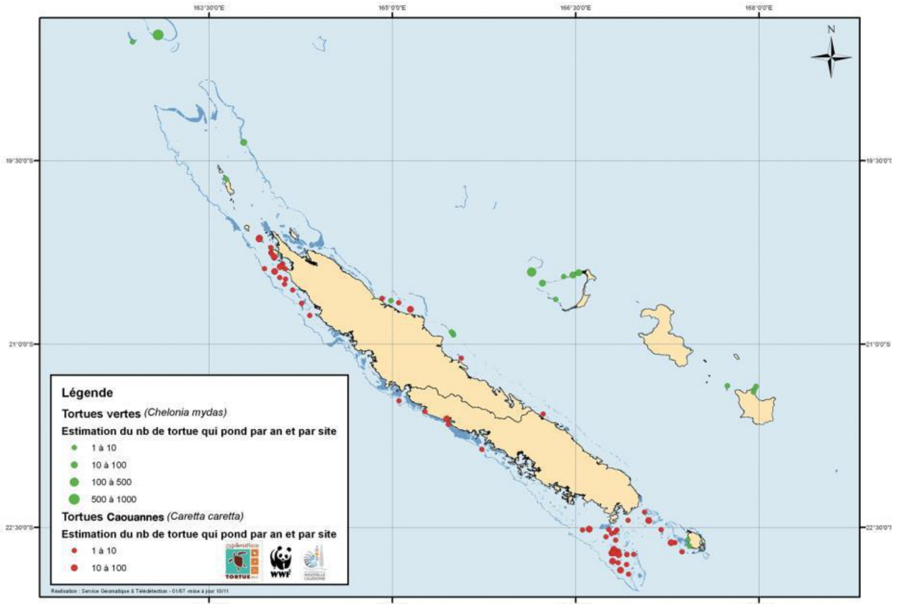
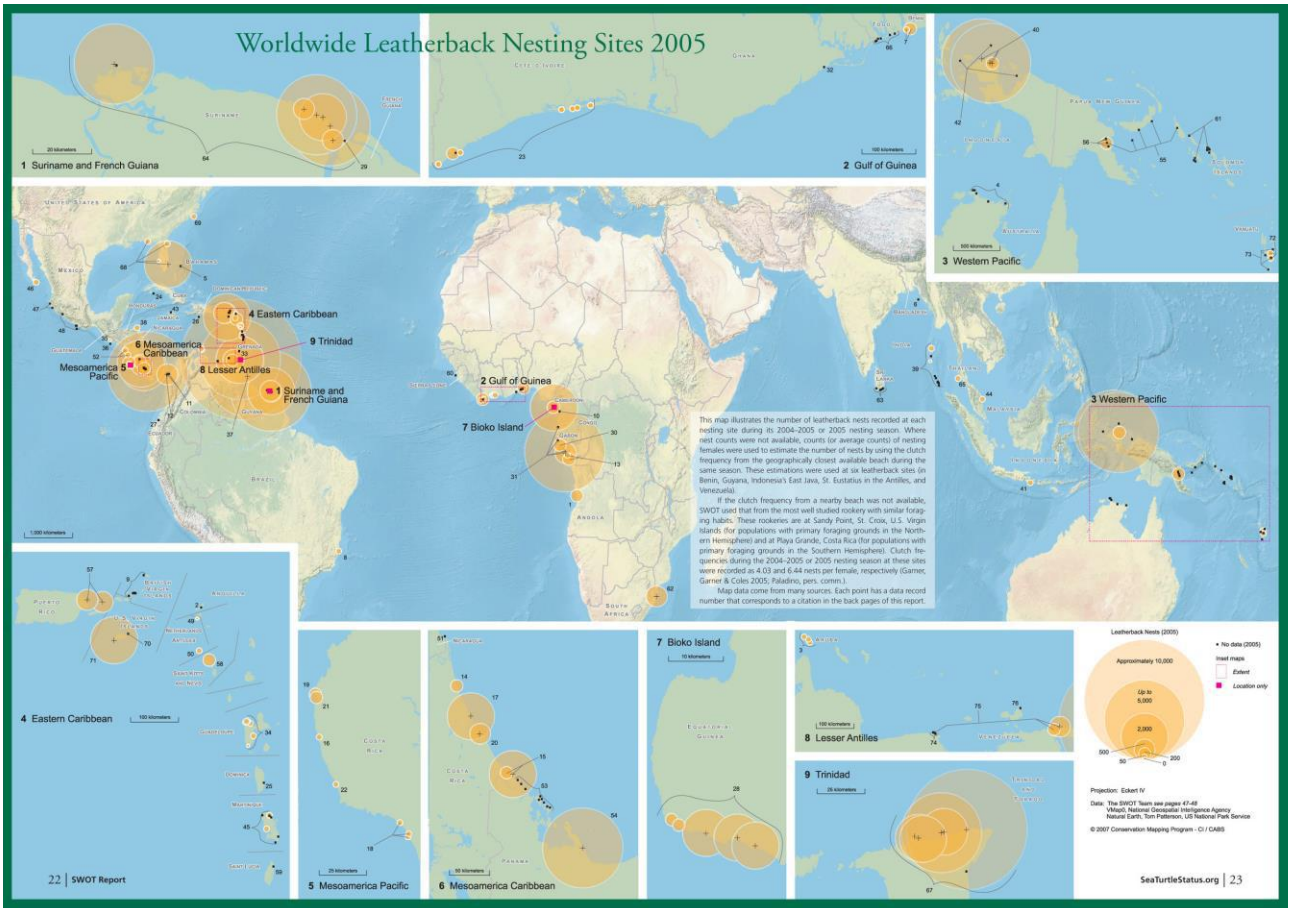


Figure 7-15. Location of green and loggerhead (in red) nesting sites in New Caledonia. Image source: Étaix-Bonnin et al. 2011.

## 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.

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.

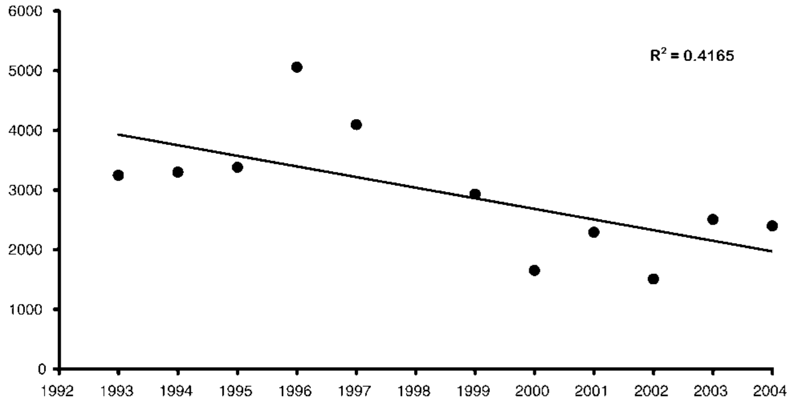


Figure 7-17. Decline of leatherback nesting at Jamursba Medi. Nest counts are for a portion of the nesting season from beginning of June through end of September and do not represent total nests for the year. Image source: Hitipeuw et al. 2007.

**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.

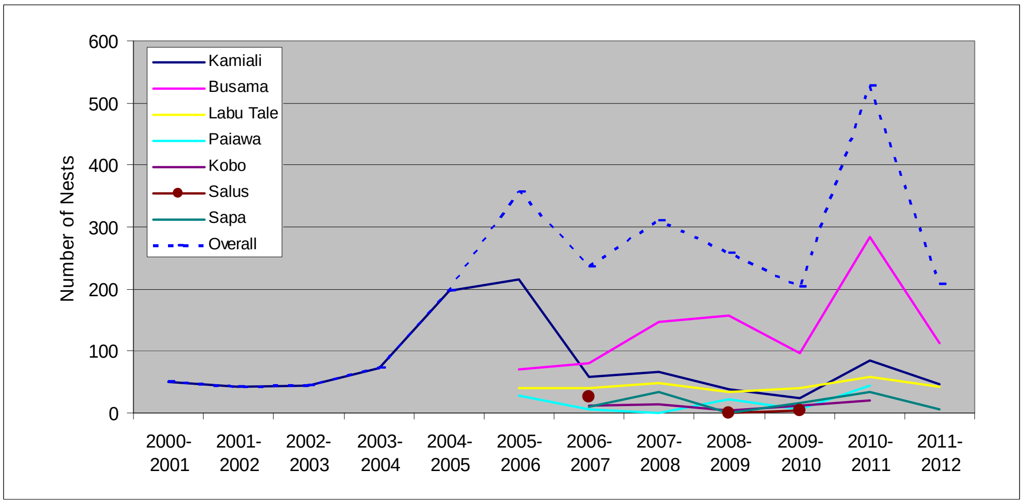


Figure 7-18. Nesting trends along the Huon coast of PNG. It is important to note that the increase at Kamiali in 2004 and 2005 was the result of an expansion of the monitored area, rather than an increase in nesting abundance. Image source: Pilcher 2013.

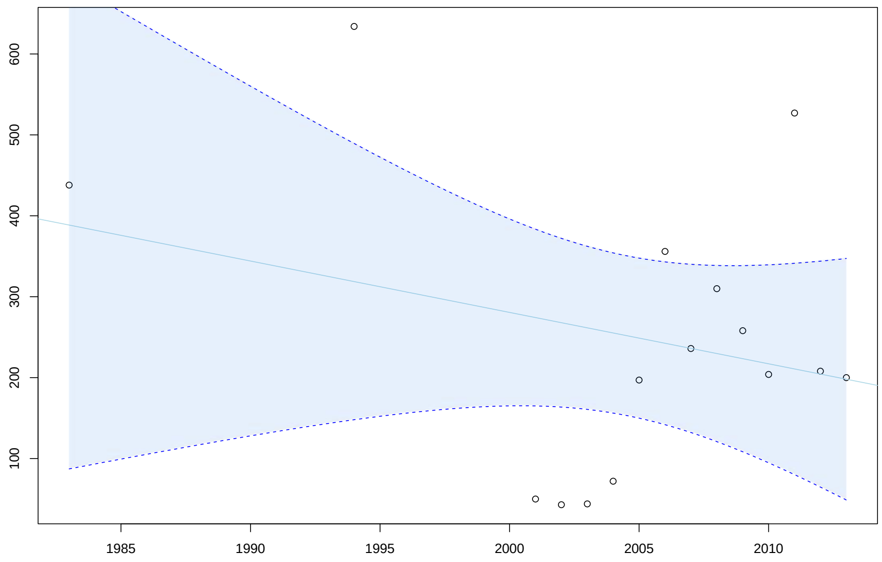


Figure 7-19. Declining trend in nest deposition along the Huon coast, Papua New Guinea. Shaded areas represent 95% confidence intervals. Image source: Pilcher et al. unpublished.

**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.

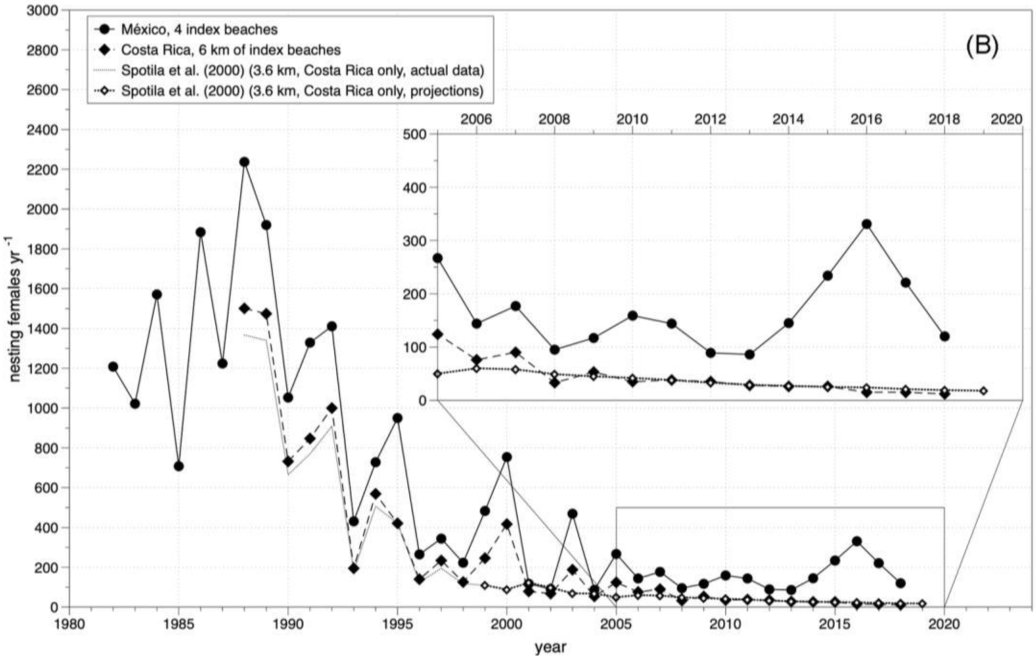


Figure 7-20. Population trends for eastern Pacific leatherbacks. Image source: Laúd OPO Network 2020.

## 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.

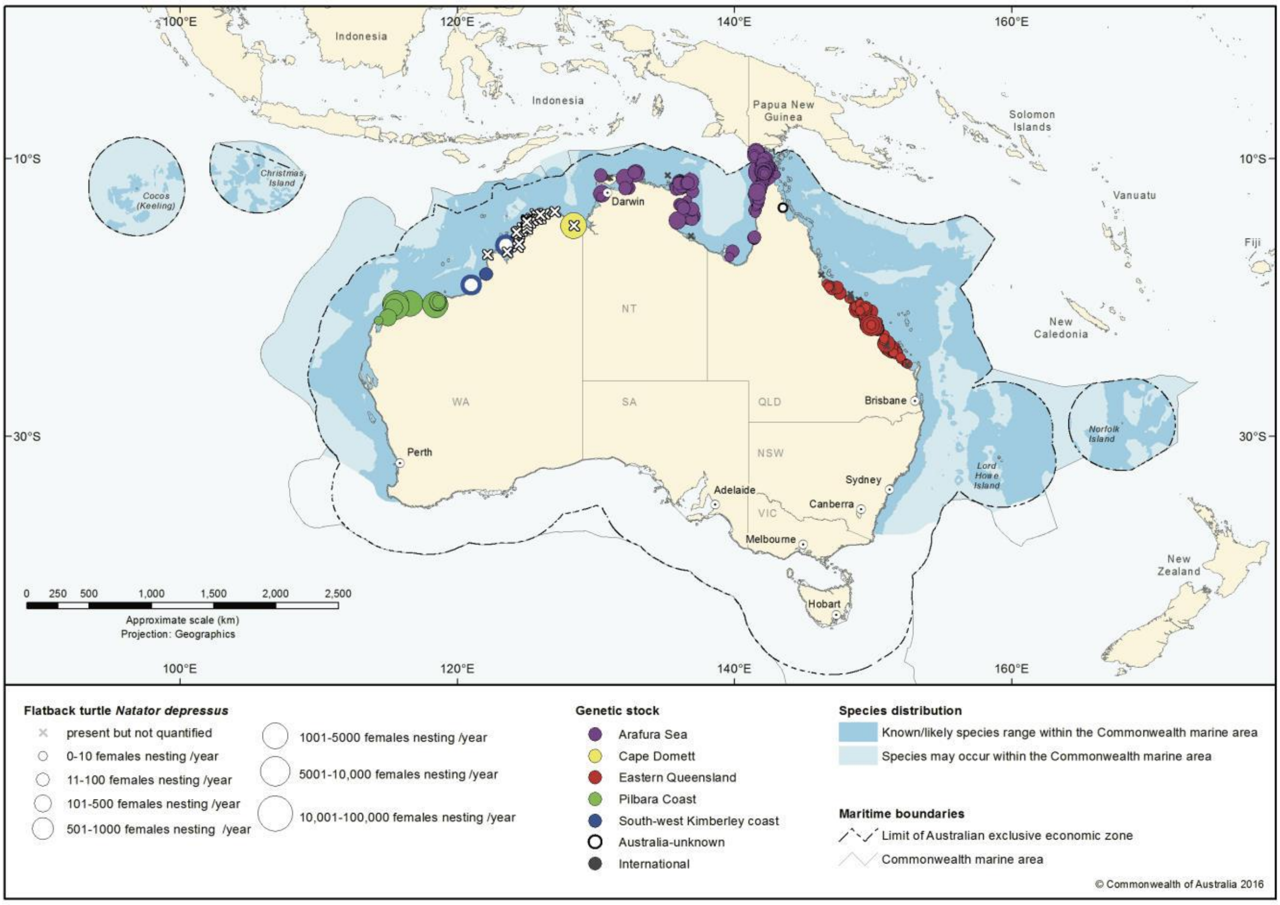


Figure 7-21. Flatback turtle (*Natator depressus*) nesting sites in Australia and surrounding regions. Image source: Commonwealth of Australia 2017.

## 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.

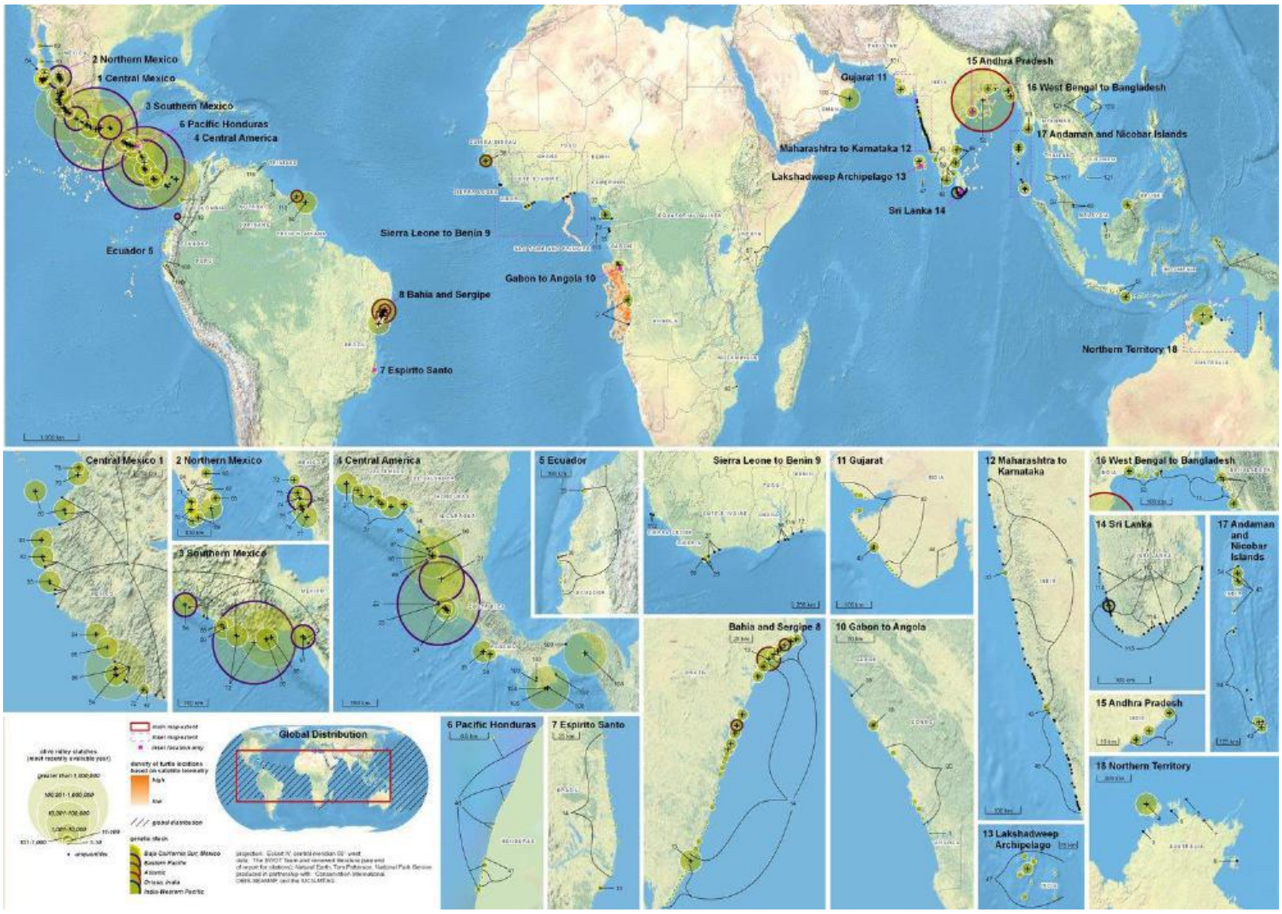


Figure 7-22. Global biogeography of olive ridley nesting sites. Image source: SWOT Report IV.

### 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).

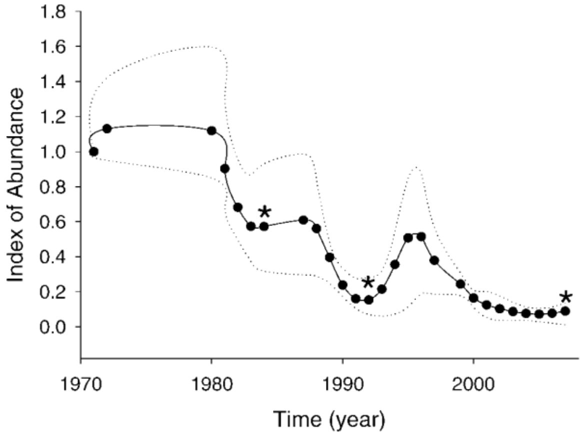


Figure 7-23. Modelled trend for arribada size based on the index of abundance for the Nancite rookery (1971–2007) using a GAM fitted to nesting female estimates. Image source: Fonseca et al. 2009.

**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.

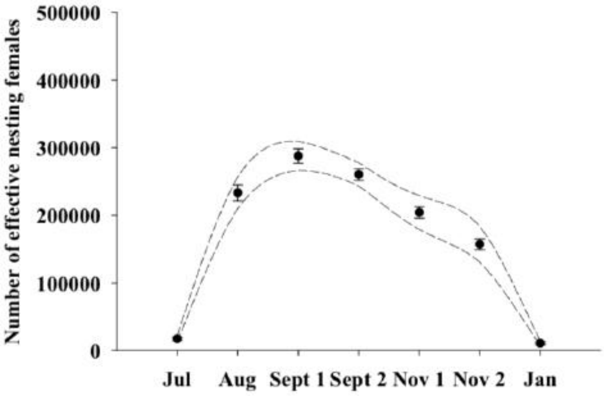


Figure 7-24. Arribada abundance at La Escobilla Beach, Mexico during the 2006 season. Data are plotted as number of effective nesting females, which is equivalent to number of nests laid. Image source: Valverde 2007.

**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).

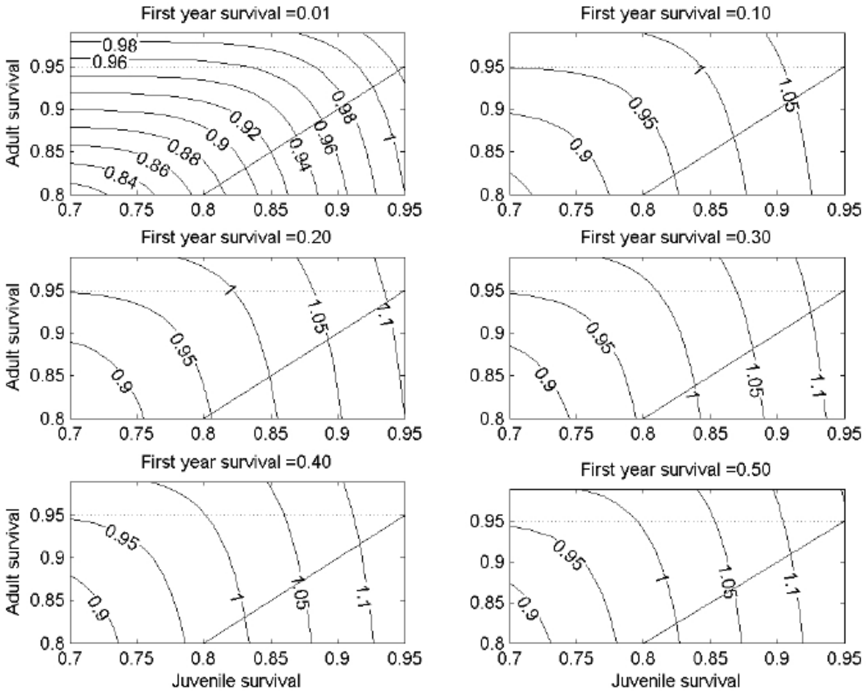


Figure 7-25. Relationships among adult survival rate, juvenile survival rate, first year survival rate, and the dominant eigenvalue for the Southwest Pacific RMU. Contour lines indicate the dominant eigenvalues, whereas the diagonal lines indicate equal annual survival rates between juveniles and adults. 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).

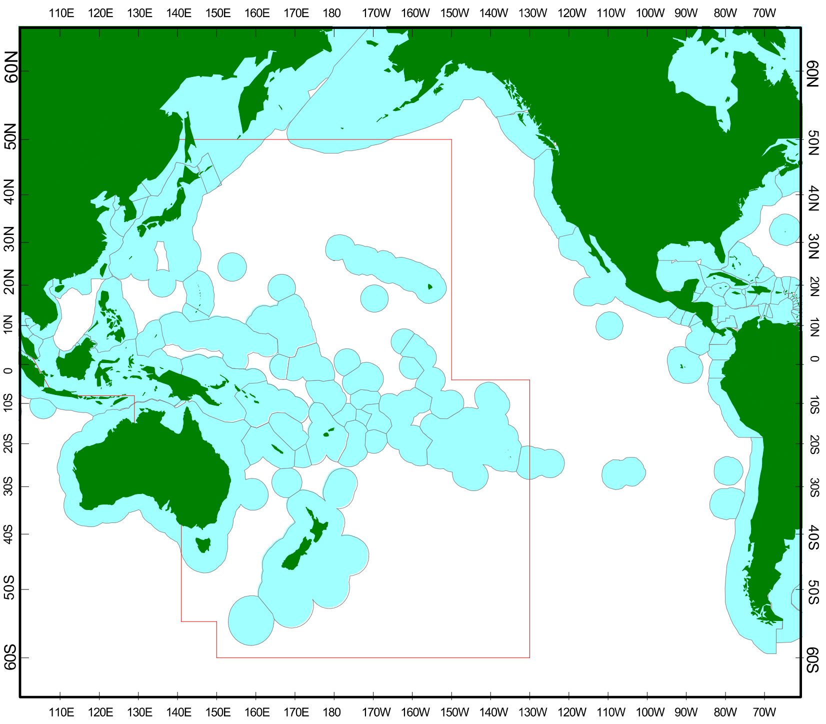


Figure 8-1. Pacific Ocean and the boundaries (red) of the WCPFC statistical area. Image source: Hampton 2010.

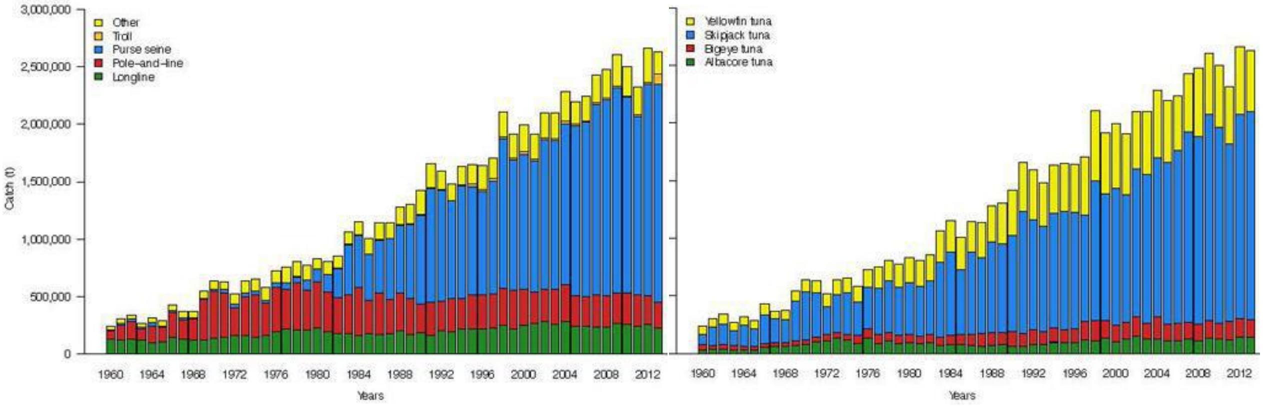


Figure 8-2. Catch (in tonnes) by gear (left) and species (right) for the Western and Central Pacific region, 1960-2013. Image source: Harley et al. 2015.

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).

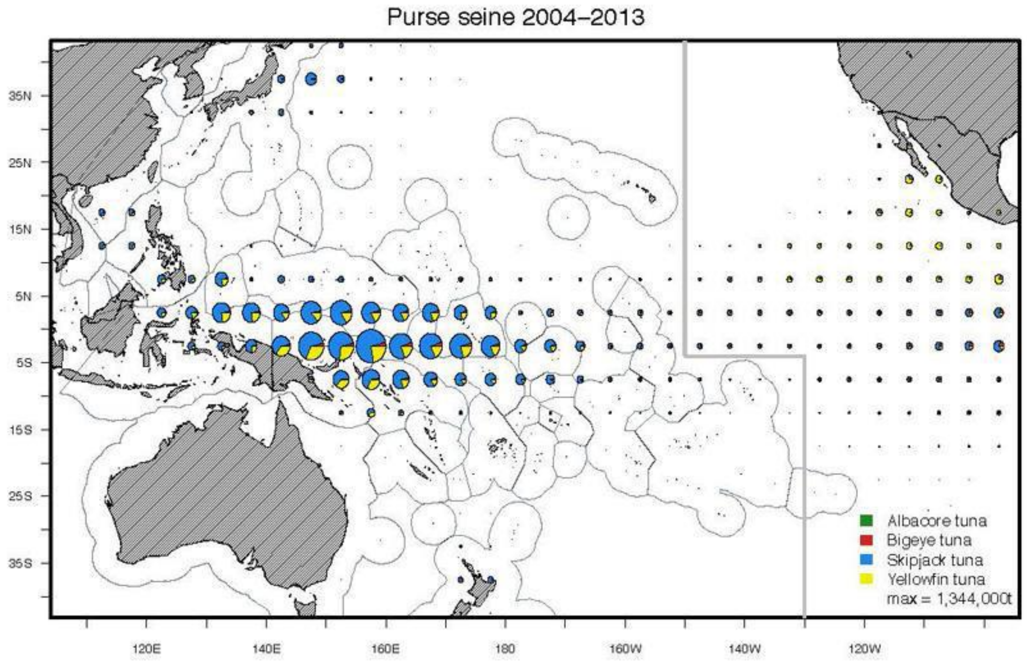


**Figure 8-3. Catch composition and spatial distribution from purse seine vessels operating in the WCPFC. The size of the largest pie corresponds to the maximum value of 1,344,000 t. Image source: Harley et al. 2015.**

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.

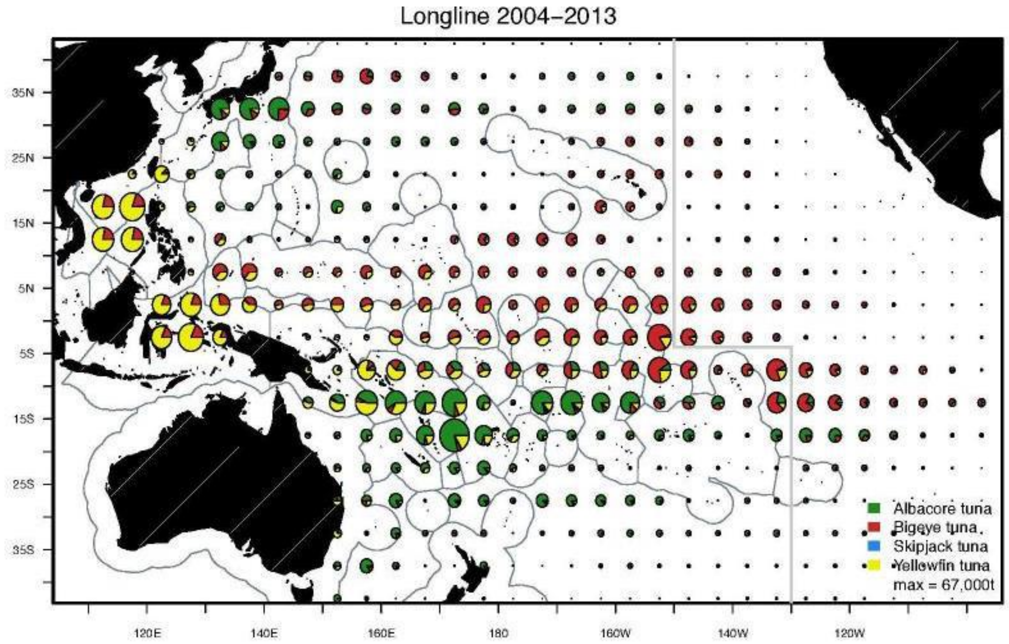


Figure 8-4. Catch composition and spatial distribution from longline vessels operating in the WCPFC. Image source: Harley et al. 2015.

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.

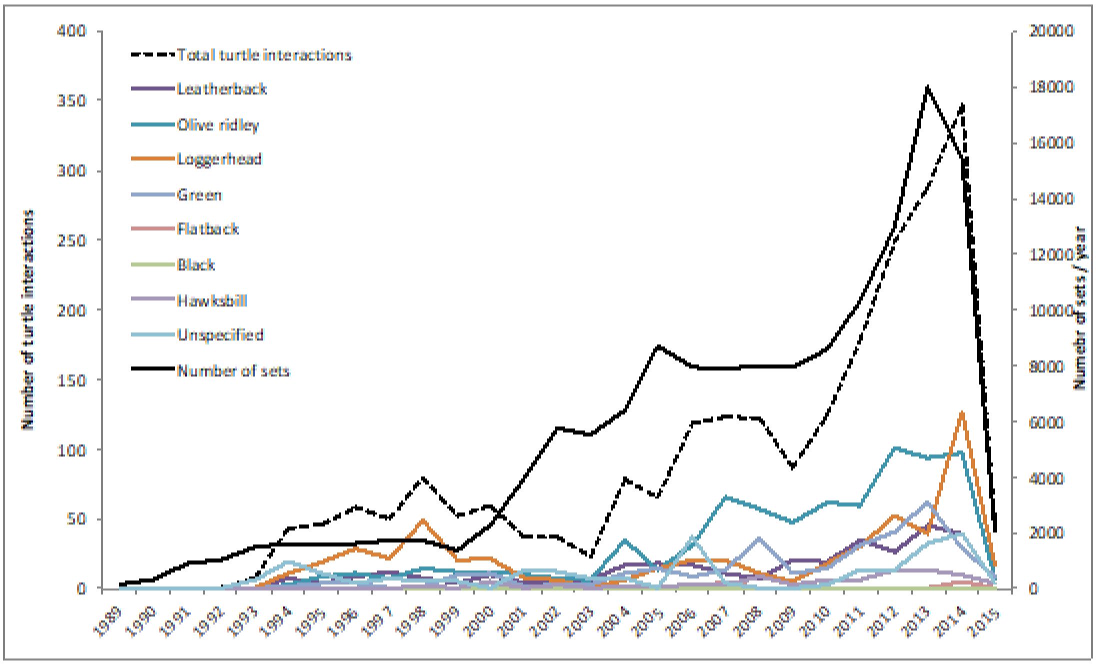


**Figure 8-5. Temporal trend in sea turtle interactions in the WCPO longline fishery compared to number of longline sets. Data extracted from WCPFC 2017.**

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).

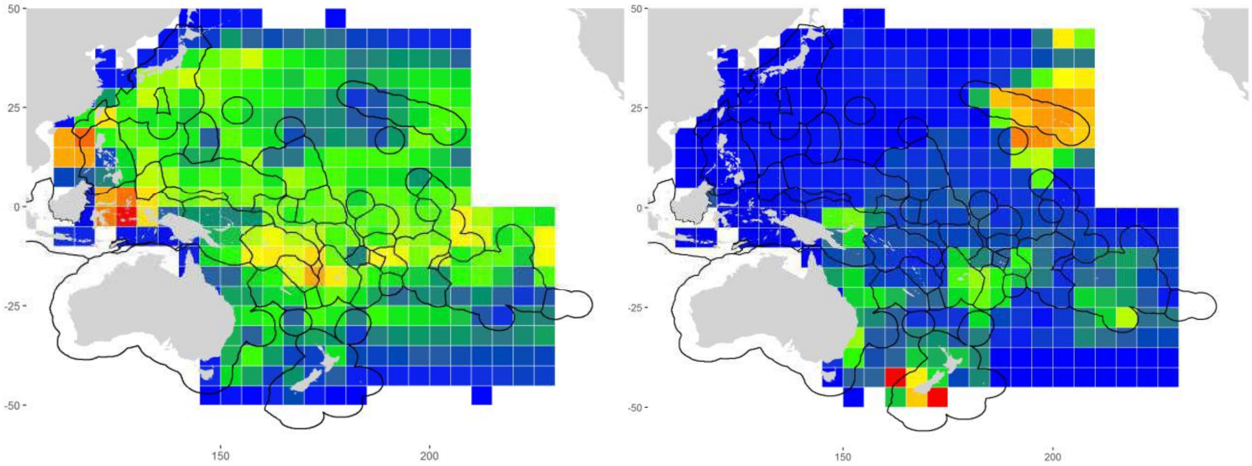


**Figure 8-6. Hotspot graphics of reported effort (left) and observer coverage (right; as a proportion of hooks) for longline fleets in the WCPFC-CA from 2003 to 2017. Image adapted from: Peatman et al. 2018b.**

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 |

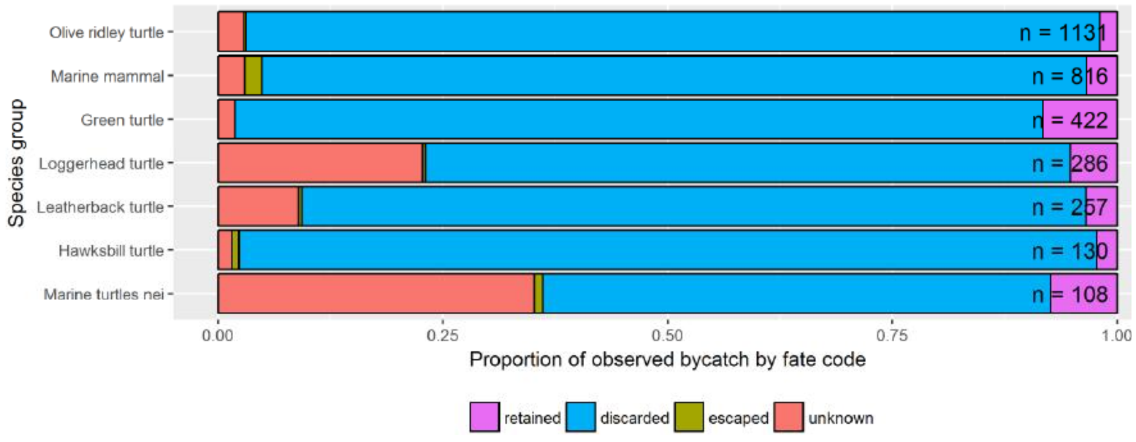


**Figure 8-7. Recorded fate of observed species of marine mammals and sea turtles catch by species/species group, as a proportion of total observed catch (number of specimens) for the species/species group in the longline fisheries. Image source: Peatman et al. 2018b.**

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).

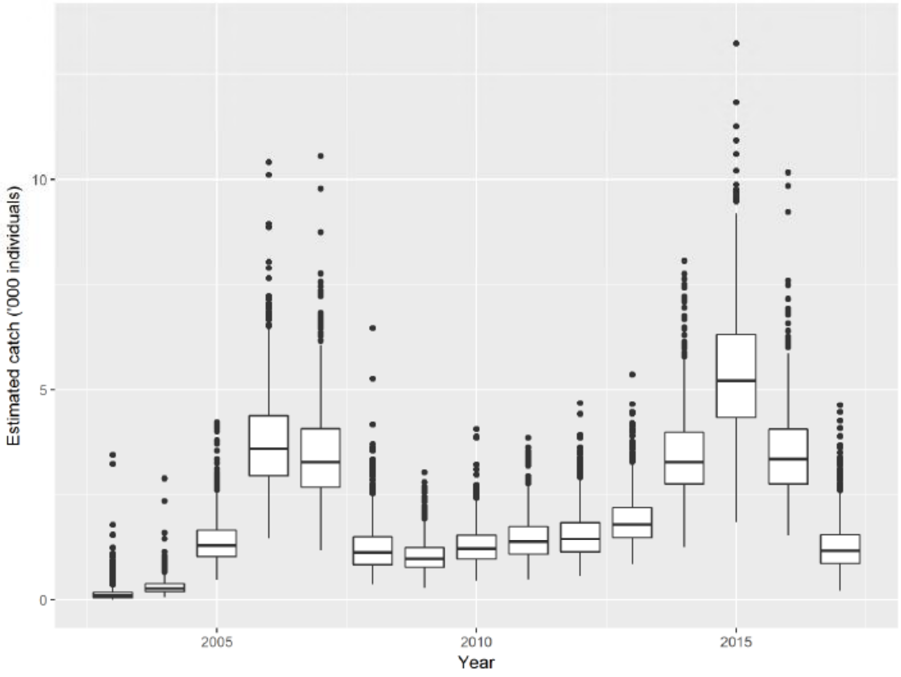


Figure 8-8. Total estimated loggerhead turtle by-catch per year. Image source: Peatman et al. 2018b.

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).

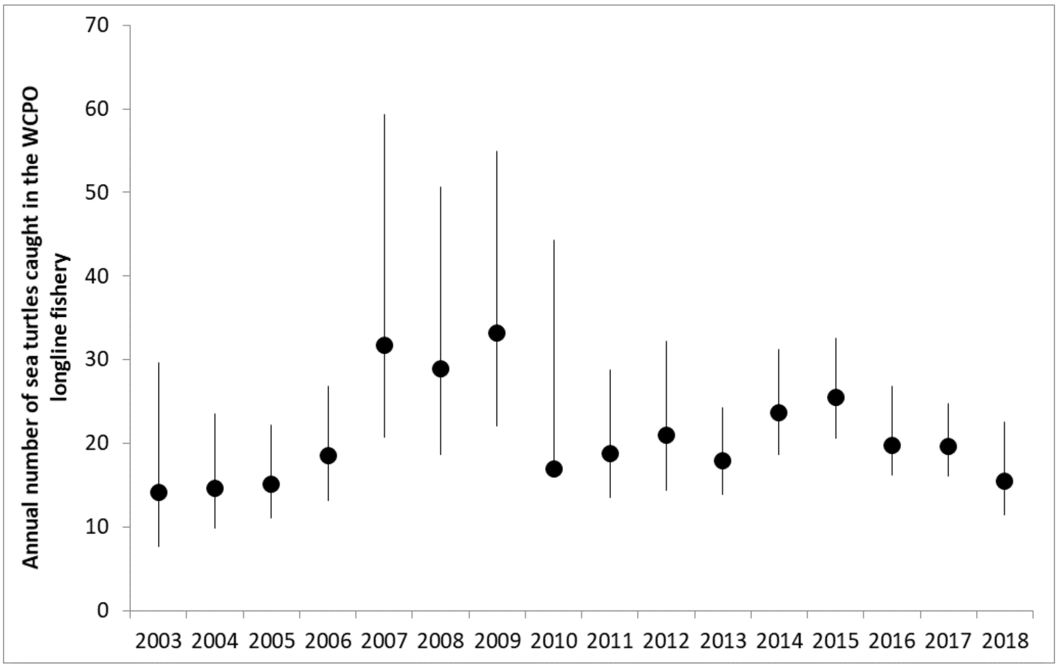


**Figure 8-9. Estimated annual longline catch (in '000s individuals). Median catch (black circles), along with upper and lower 95% confidence intervals. Data source: Peatman and Nicol 2020.**

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<br>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<br>(0.00003-0.0075) | 0.002<br>(0.0002-0.0054) | 0.002<br>(0.0005-0.0067) |
| PAC_TW | 0.003<br>(0.0014- 0.0071) | 0.011<br>(0.0080-0.0159) | 0.002<br>(0.0009-0.0040) | 0.006<br>(0.0032-0.0096) |
| PAC_SE | 0 | 0 | 0 | 0 |
| PAC_SW | 0.004<br>(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).

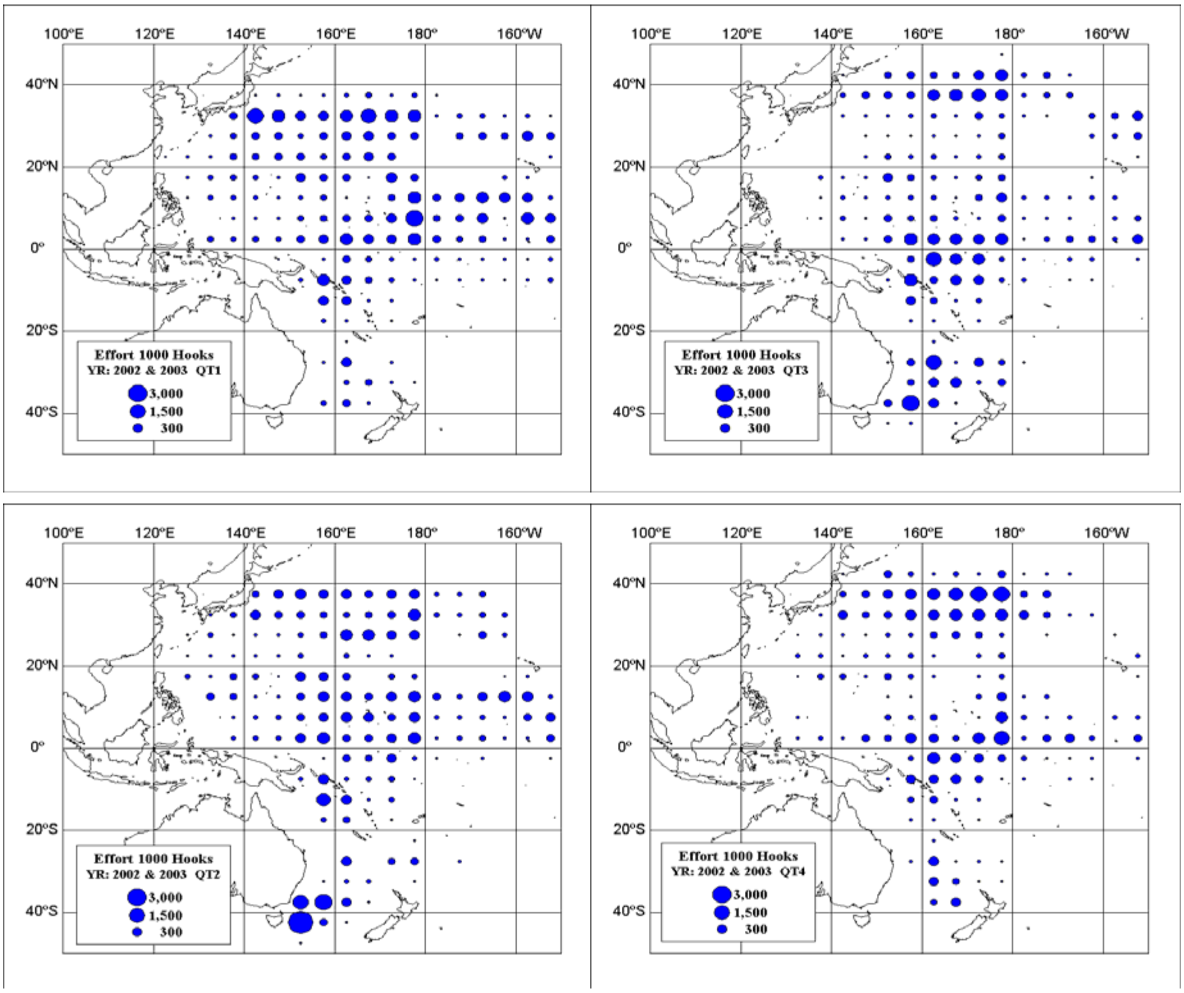


**Figure 8-10. Quarterly distribution of fishing effort for the Japanese offshore and distant water longline fisheries in the western and central Pacific Ocean in 2002–2003. Image Source: Uosaki et al. 2005.**

### 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.

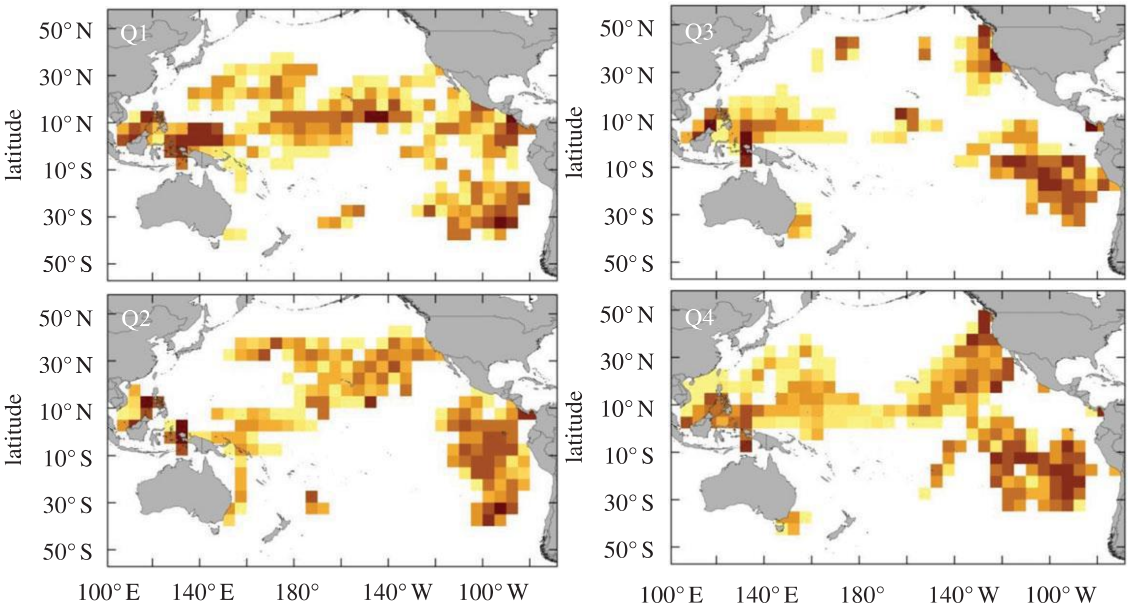


Figure 8-11. Relative use-intensity distributions for leatherback turtles in the Pacific Ocean within 58 x 58 grid cells. Values represent the proportion of all time-weighted and population-normalised positions by annual quarter. Image adapted from: Roe et al. 2014.

### 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.

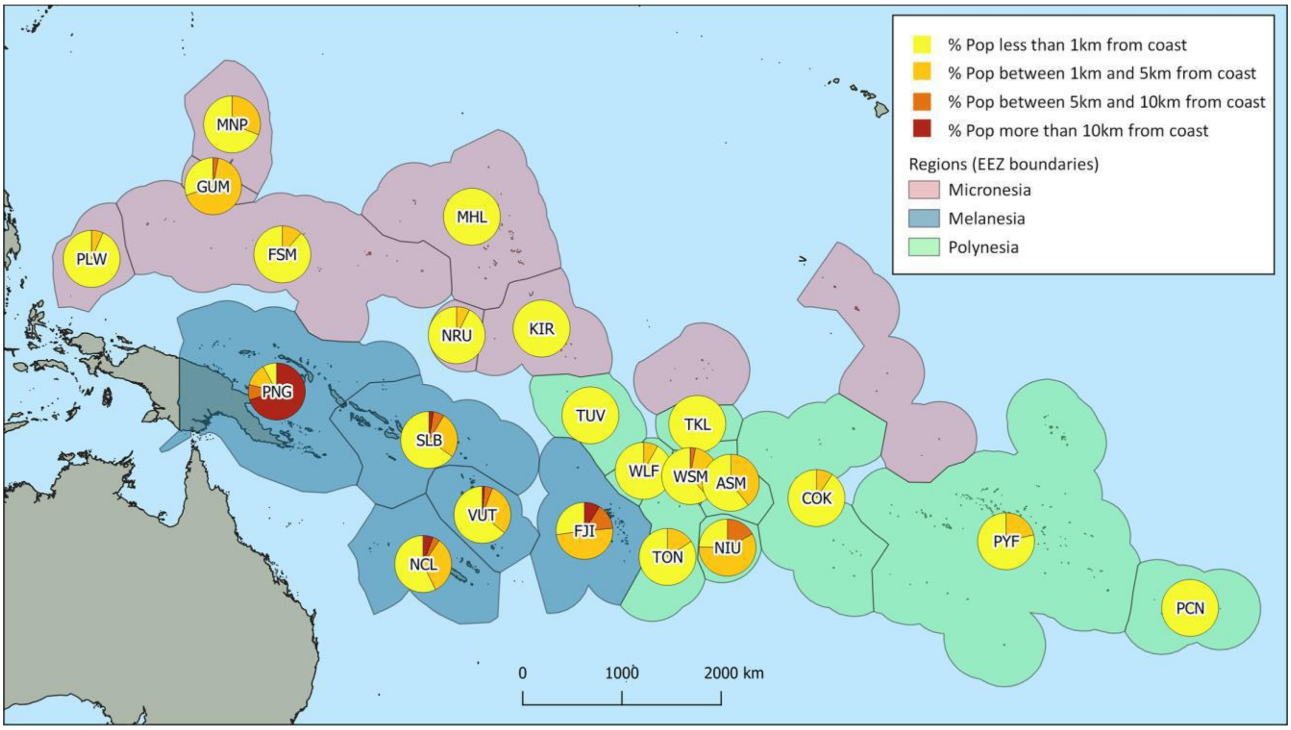


Figure 8-12: Proportions of households within 1, 5 and 10 km from the coast in the Pacific. Source: Andrew et al. 2019.

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).

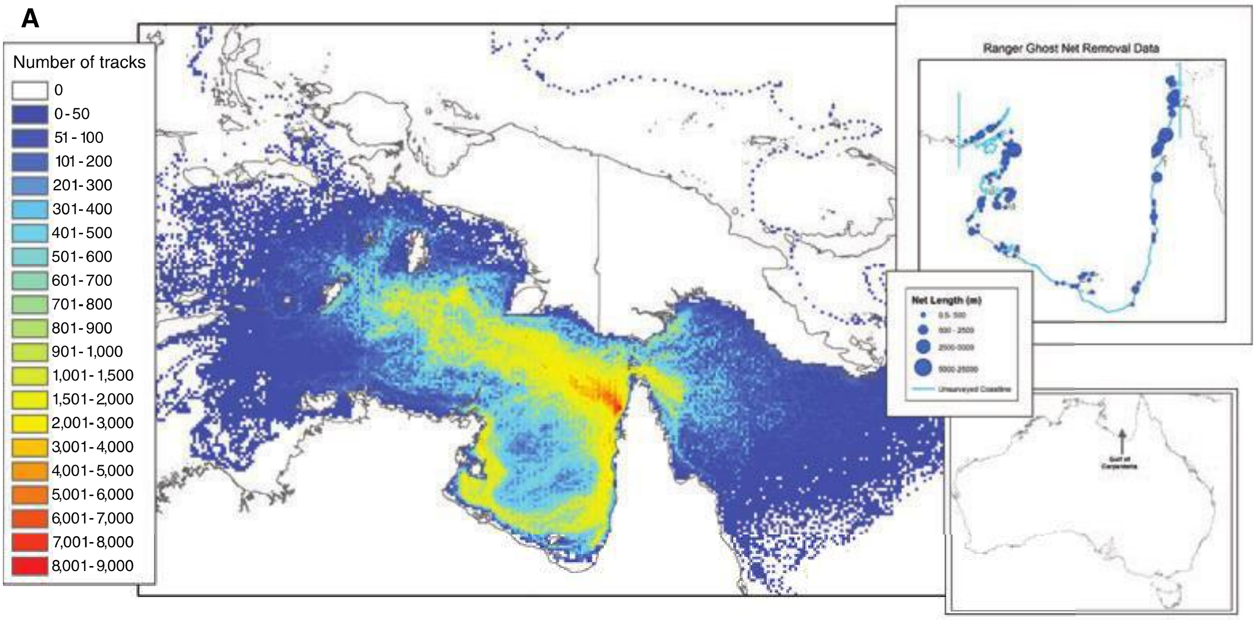


Figure 8-13: Components of a risk model for ghost net impacts to turtles. (A) Potential ghost net tracks based on daily particles releases (1996–2008) and (inset) total length of net found at sites along the Gulf of Carpentaria coastline. Source: Wilcox et al. 2012.

**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.

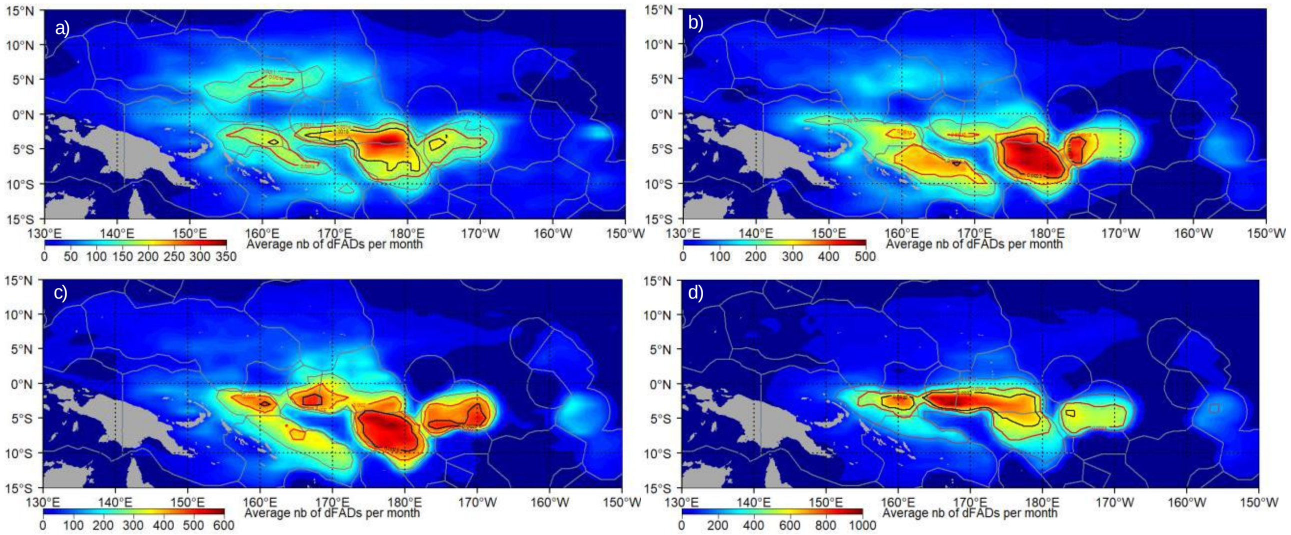


**Figure 8-14: Smoothed kernel density of the average number (nb) of FAD satellite buoys transmitting at least once per month and per 1° grid cell during a) 2016, b) 2017, c) 2018, and d) 2019. Red lines correspond to the 95th quantile. Colour scale corresponds to the average number of buoys transmitting 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).

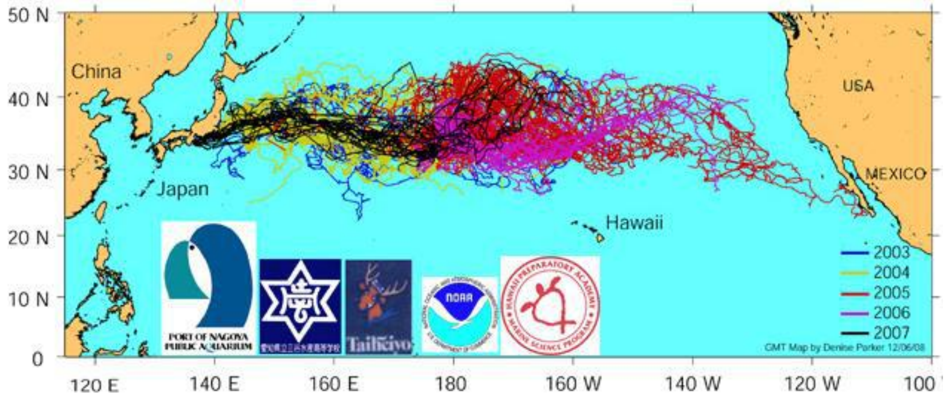


Figure 9-1: Migrations of aquarium-reared loggerhead turtles in the northern Pacific. Image source: NOAA NMFS and HPA.

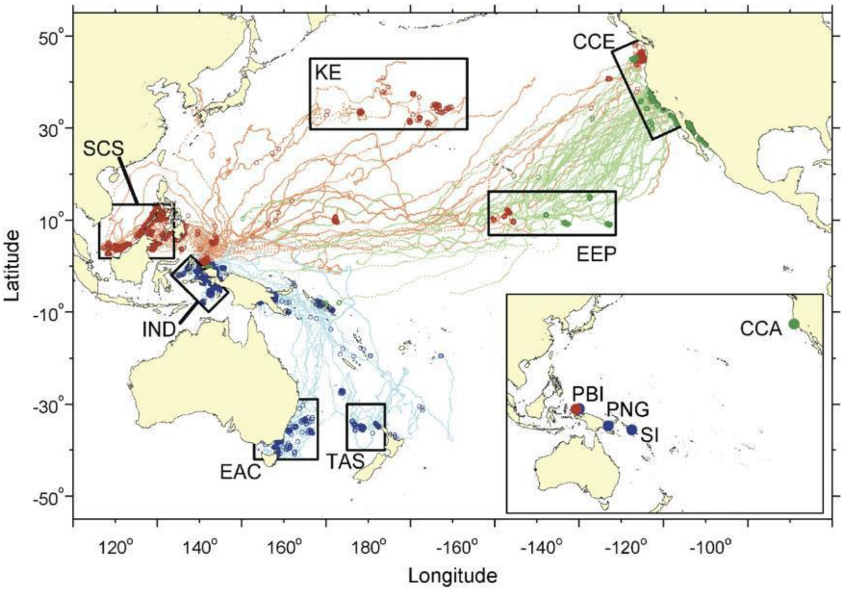


Figure 9-2: Migrations of leatherback turtles in the Pacific. Image source: Benson et al. 2011.

### 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.

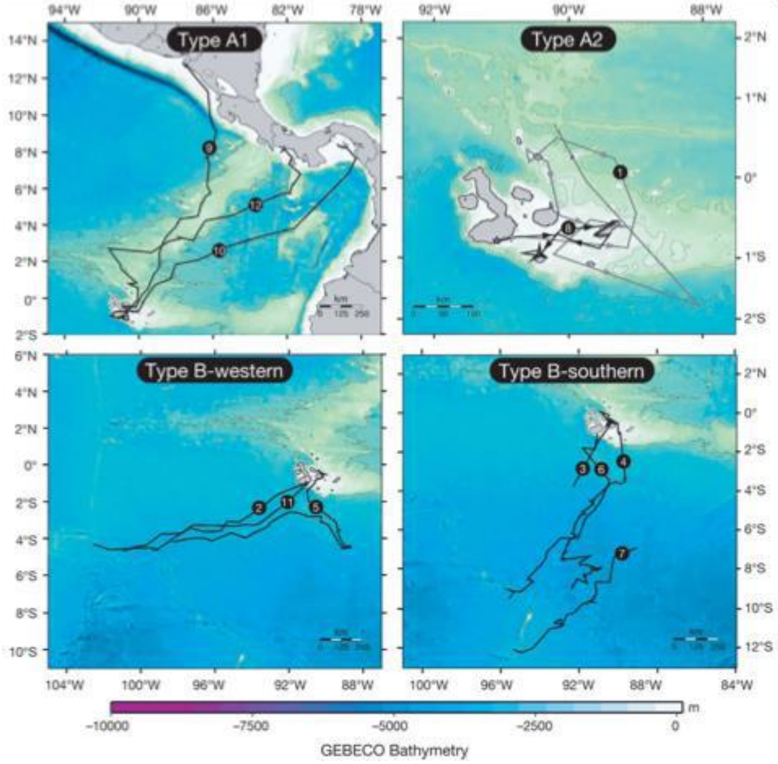


**Figure 9-3. Satellite-tracked post-nesting movements of green turtles nesting in the Galápagos. Image source: Seminoff et al. 2007.**

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).

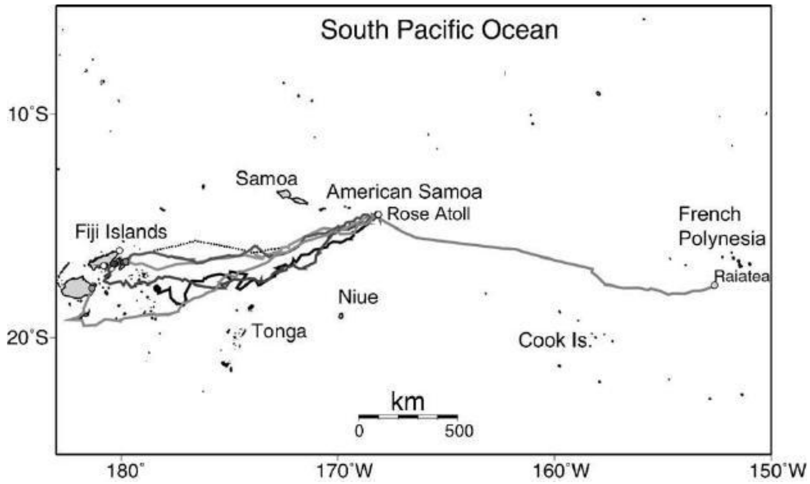


**Figure 9-4. Post-nesting movements of satellite-tracked of green turtles from Rose Atoll in American Samoa. Image source: Craig et al. 2003.**

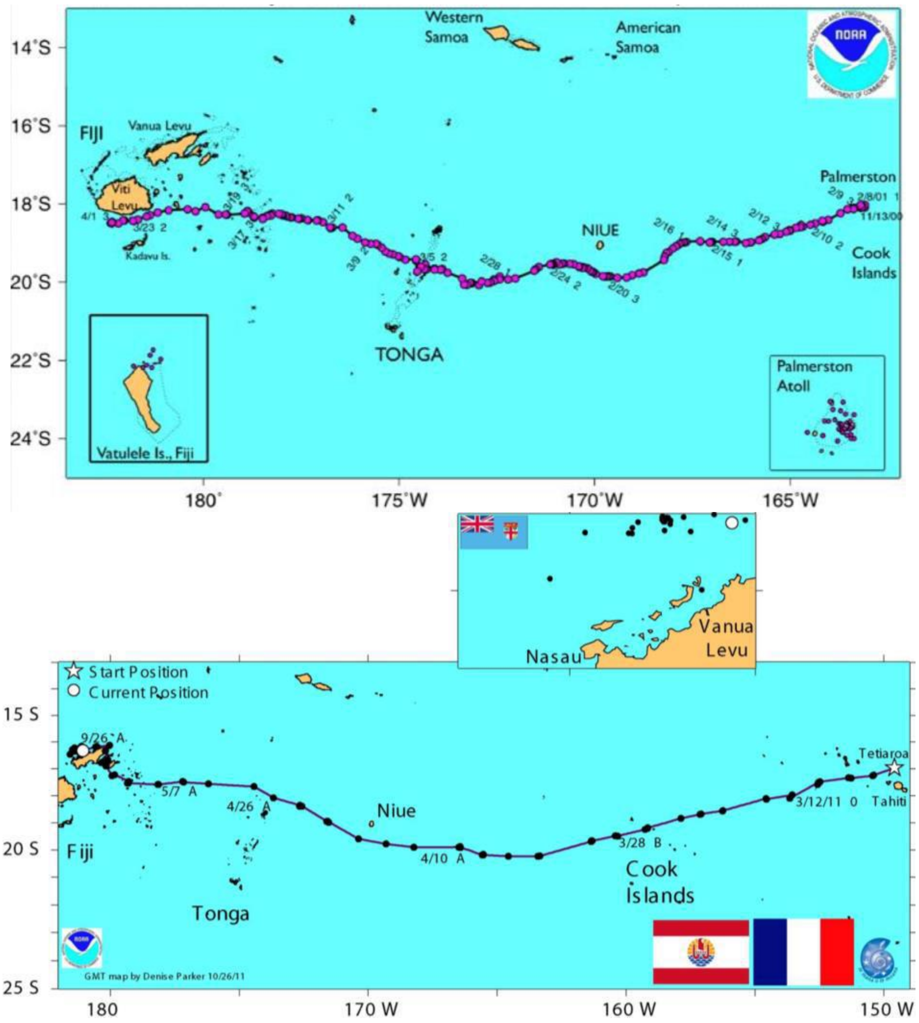


Figure 9-5. Post-nesting movements of a satellite-tracked green turtle from Palmerston Atoll (top) and another from Tetiaroa, French Polynesia (bottom). Image source: NOAA NMFS, GH Balazs, NOAA.

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.

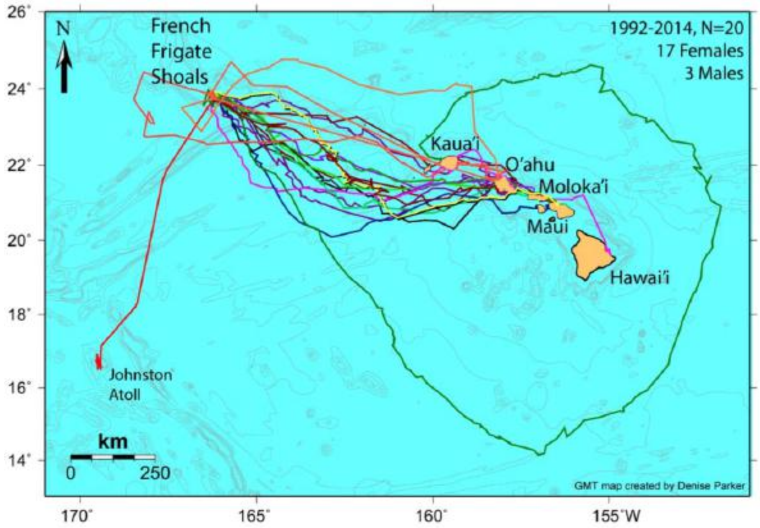


Figure 9-6. Composite view of the ocean pathways by 20 green turtles tracked during their breeding migrations linked to the colonial nesting site of French Frigate Shoals. Turtles were randomly selected for satellite tagging between 1992-2014. Image source: Balazs et al. 2017.

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).

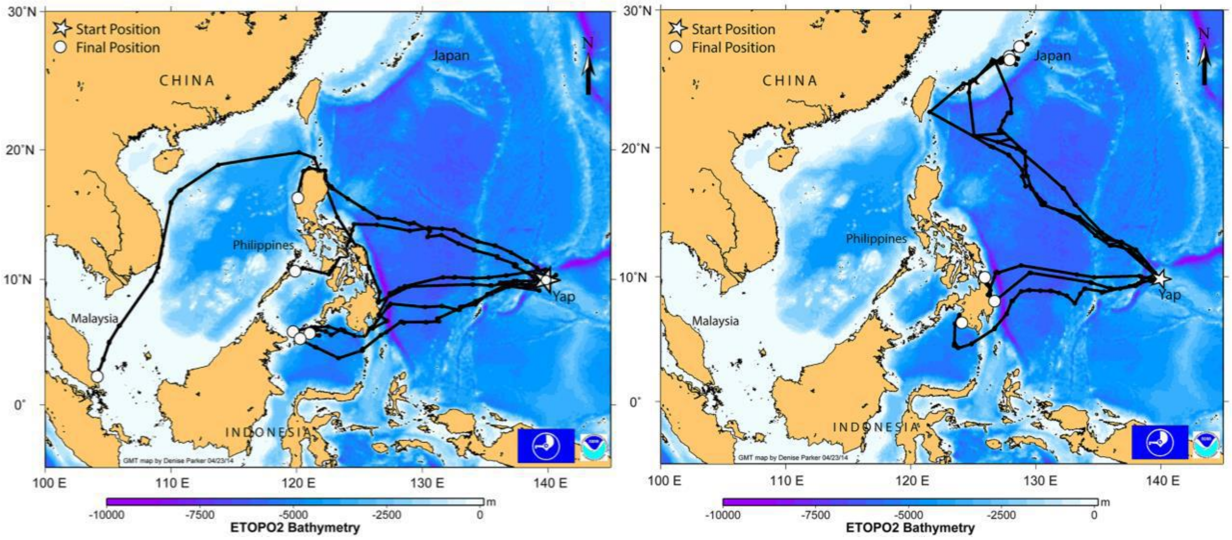


Figure 9-7. Post-nesting migrations of green turtles from Gielop Island, Federated States of Micronesia in 2005 and 2006 (left) and in 2007 (right). Image source: Kolinaksi et al. 2014.

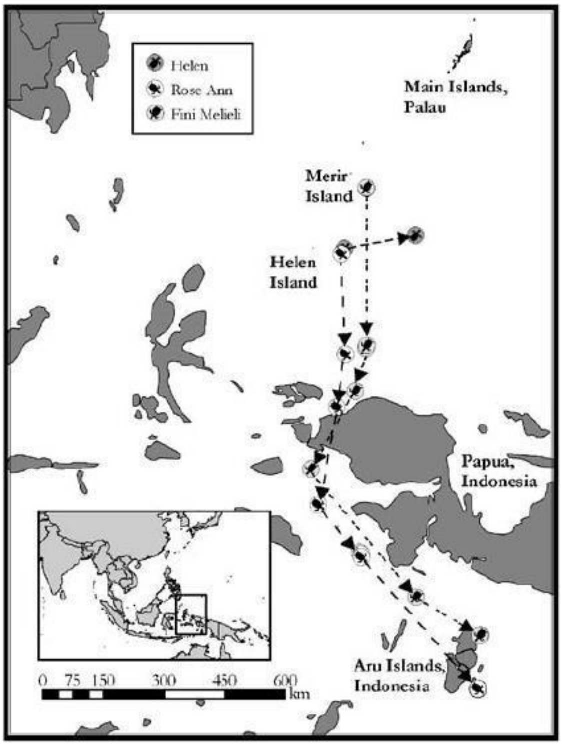


Figure 9-8. Post-nesting migrations of green turtles from Merir and Helen Islands, Palau. Image source: Klain et al. 2007

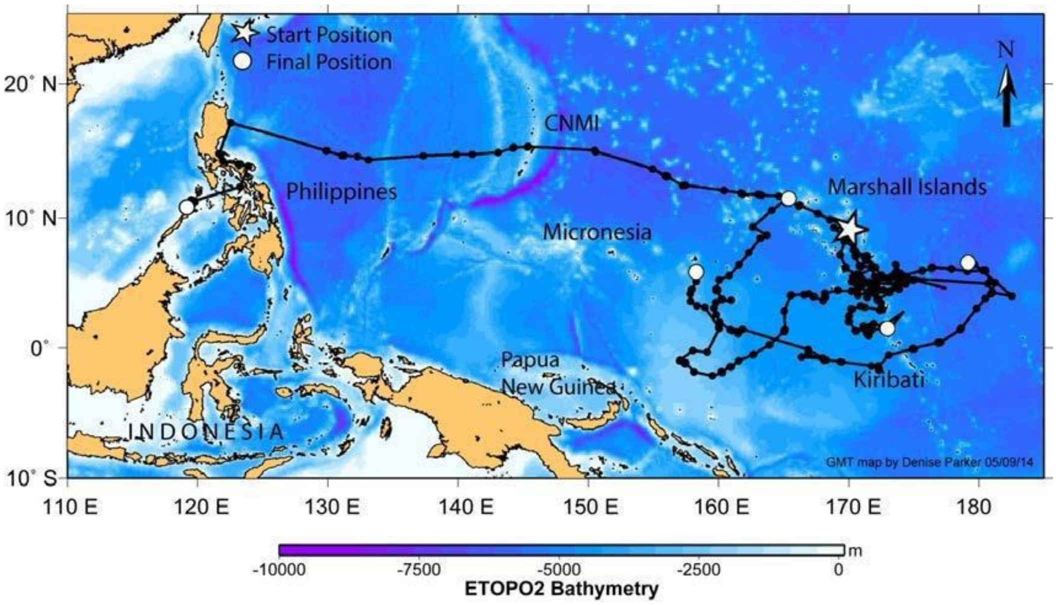


**Figure 9-9. Post-nesting movement of five green turtles from Erikub Atoll, Republic of the Marshall Islands in 2007-2008. Image source: Parker et al. 2015.**

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.

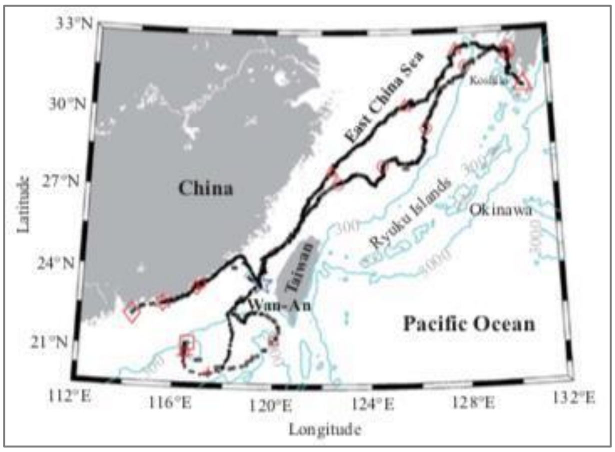


**Figure 9-10. Post-nesting movements of satellite-tracked of green turtles nesting in Taiwan. Image source: Cheng and Wang 2009.**

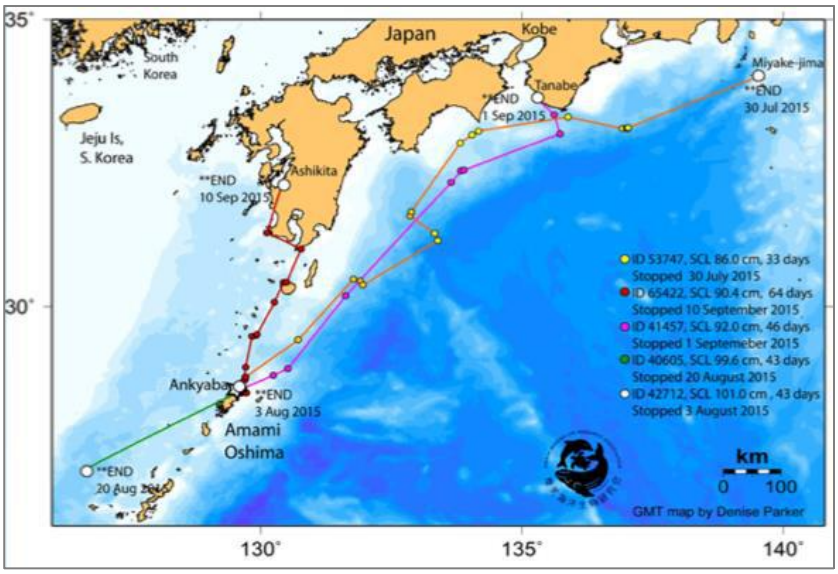


Figure 9-11. Post-nesting movements of satellite-tracked of green turtles from Amami, Japan. Image source: Parker 2006

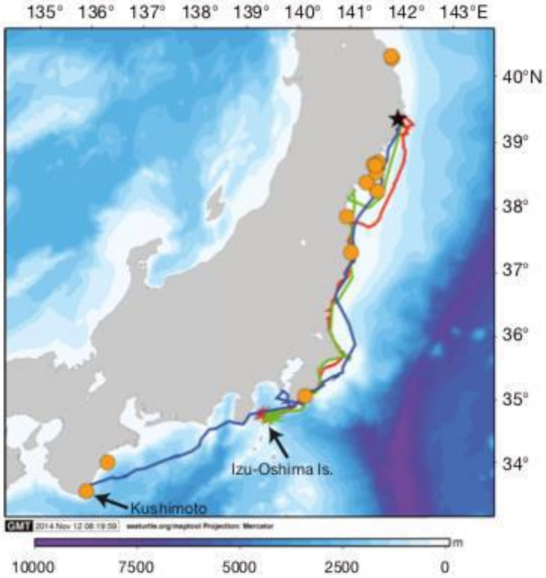


Figure 9-12. Migration paths of the green turtles equipped with satellite relay data loggers (SRDLs) and re-capture points of tagged turtles in the mark-recapture study. Star: release point; orange circles: recapture points. Image source: Fukuoka et al. 2015.

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).

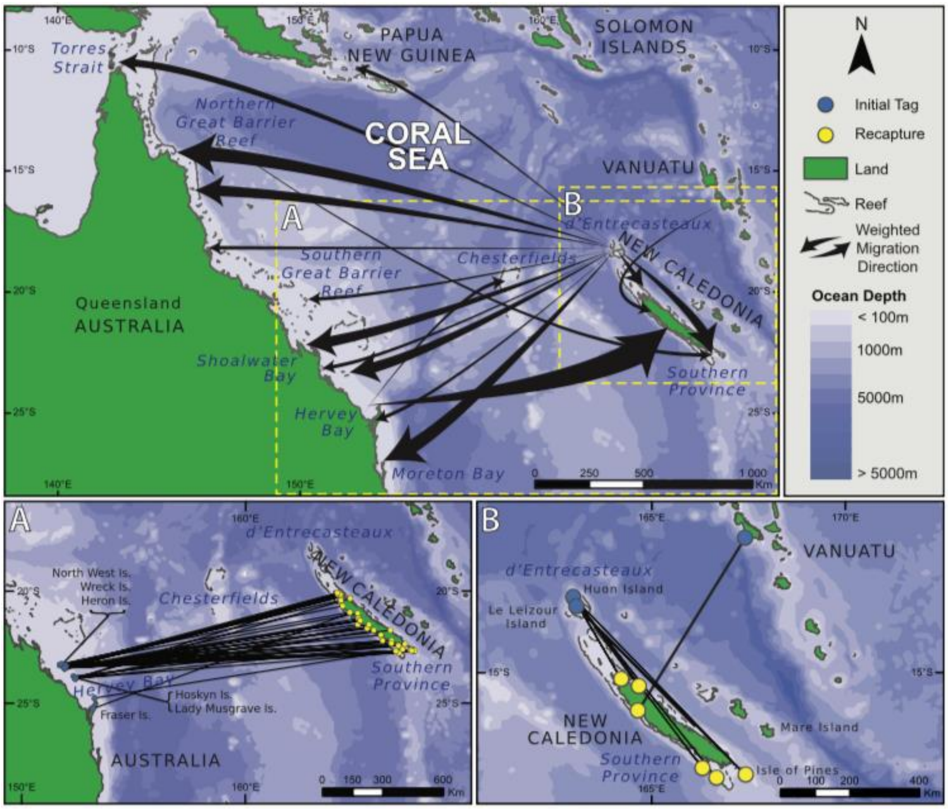


**Figure 9-13. Trajectory maps obtained by the tag recoveries (n = 93) and satellite tracking of *C. mydas* in the Coral Sea (n = 1). Image source: Read et al. 2014.**

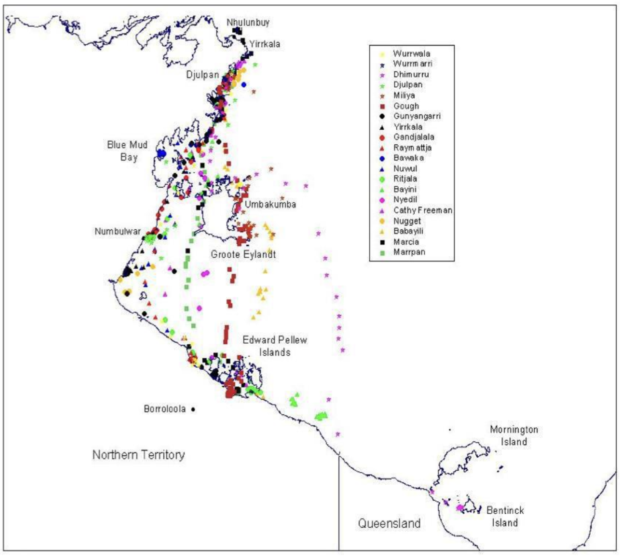


**Figure 9-14. Post-nesting migratory tracks of adult females from north-eastern Arnhem Land nesting beaches identified by satellite telemetry. Image supplied by R. Kennett. Image source: Limpus 2007a.**

## 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).

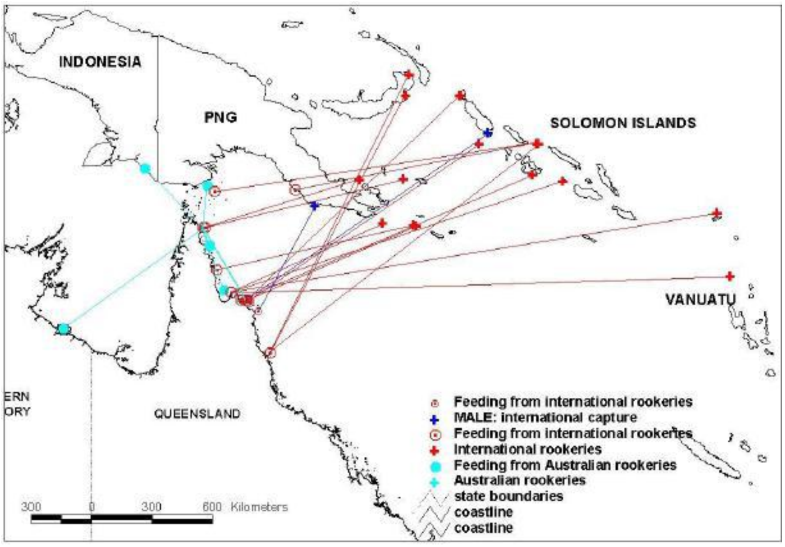
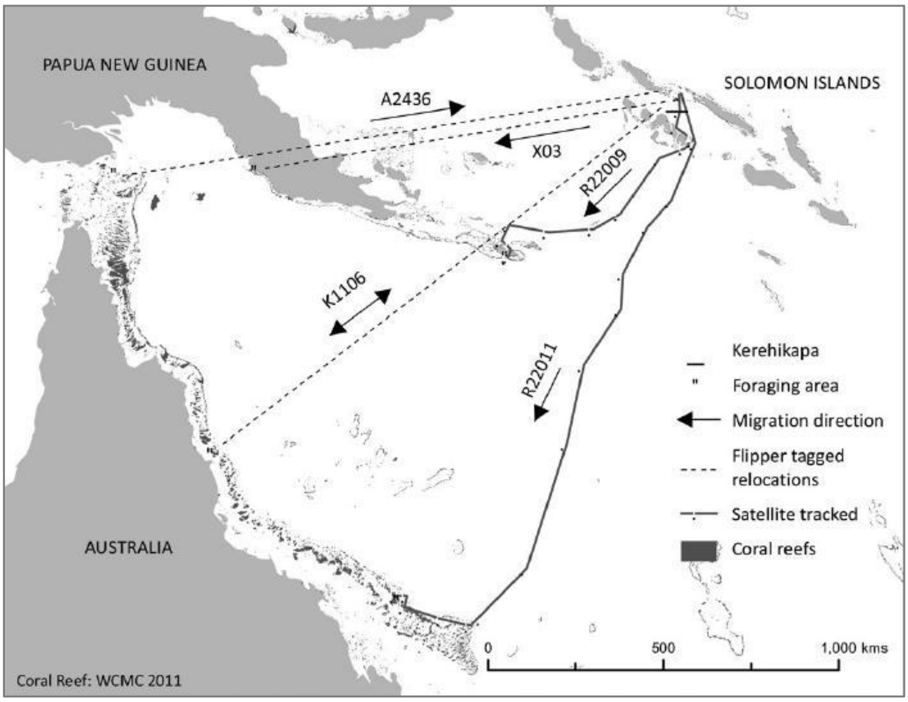


**Figure 9-15. Migration by adult female and male hawksbill turtles, *Eretmochelys imbricata*, in the Coral Sea Region of the southwest Pacific Ocean. Image source: Limpus 2008.**



Coral Reef: WCMC 2011

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

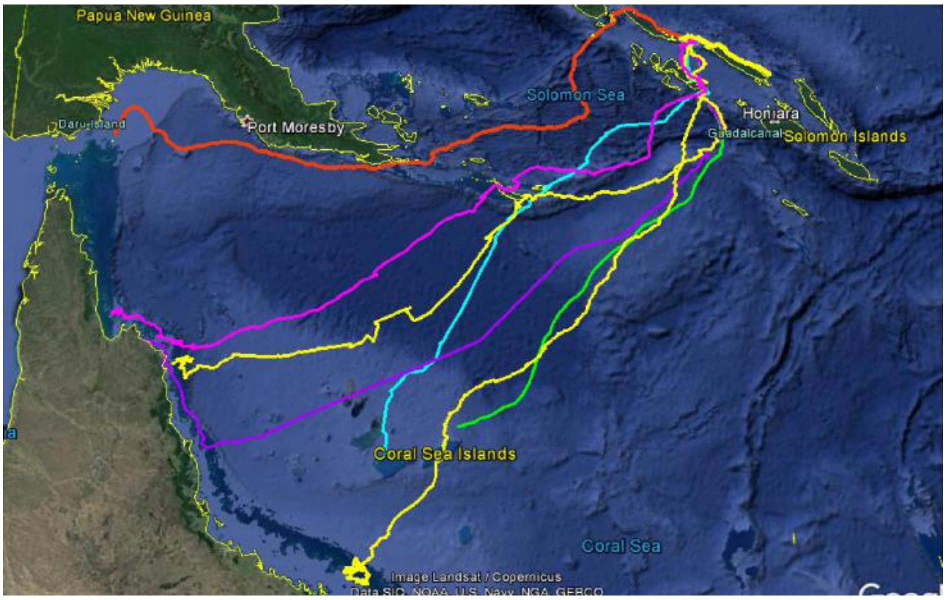


**Figure 9-17. Migration routes of eight hawksbill turtles tagged in the Arnavon Islands, Solomon Islands. Image source: Richard Hamilton/The Nature Conservancy.**

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.**

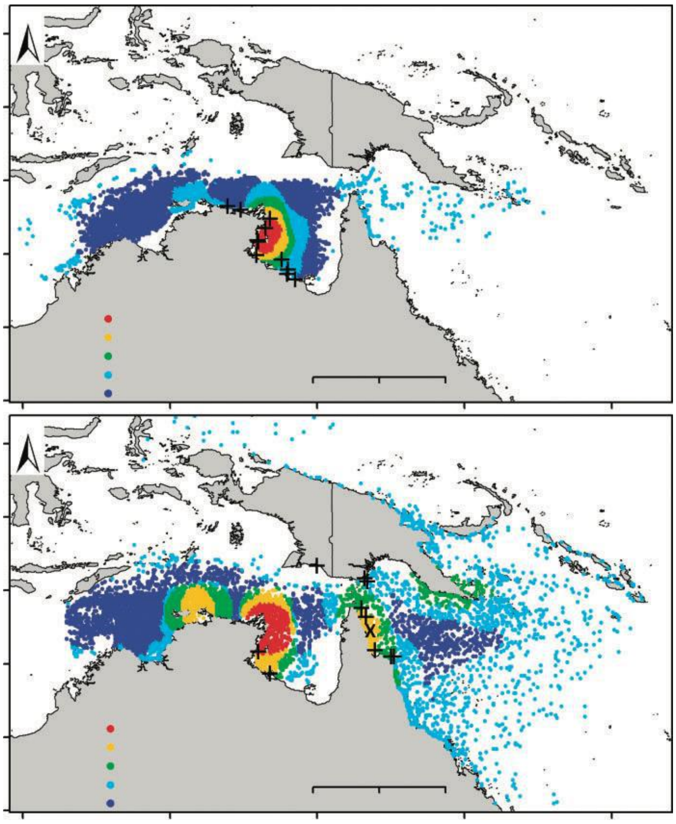


**Figure 9-19. Modelled distribution and clustering probability of post-hatchling turtles from (a) North East Island and (b) Milman Island, whose locations are shown by crosses. Red markers indicate spatial clustering of high probabilities, whereas light blue markers indicate spatial clustering of low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

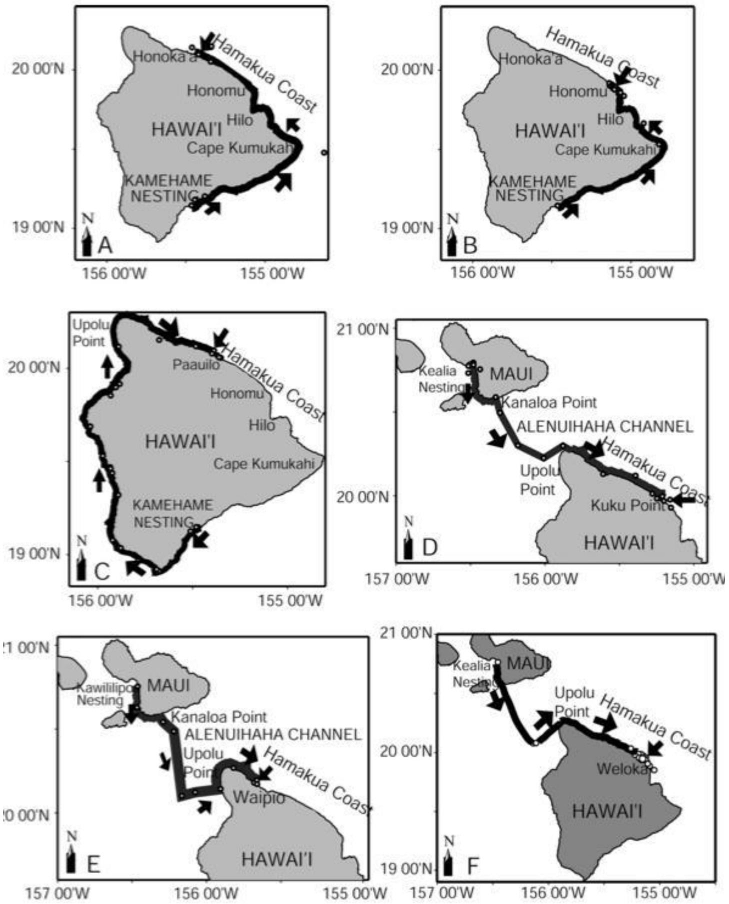


Figure 9-20. Post-nesting movement of six hawksbill turtles from nesting beaches to foraging areas on the Hāmākua Coast of the island of Hawaii. Image source: Parker et al. 2009.

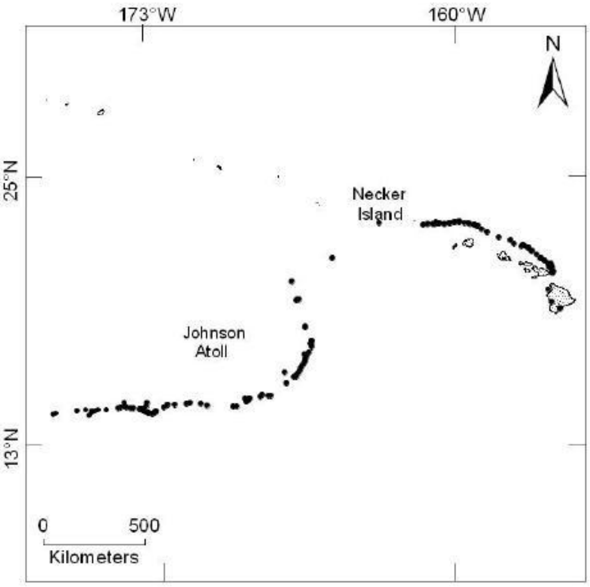


Figure 9-21. Post nesting migration of a hawksbill turtle from Hawaii into the Pacific Ocean. Image source: Graham 2009.

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).

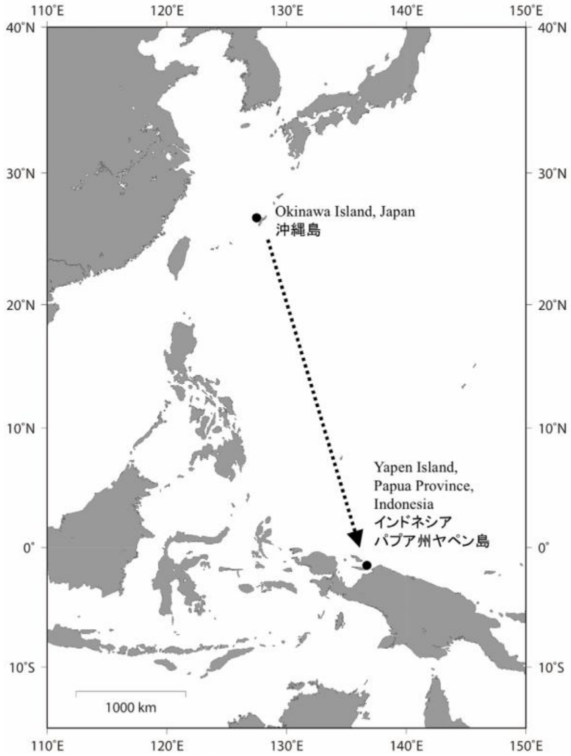


Figure 9-22. Long-distance movement of a male hawksbill turtle from Okinawa Island, Japan, to Yapen Island, Papua Province, Indonesia, based on tag recovery. Image source: Kawazu et al. 2018

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.

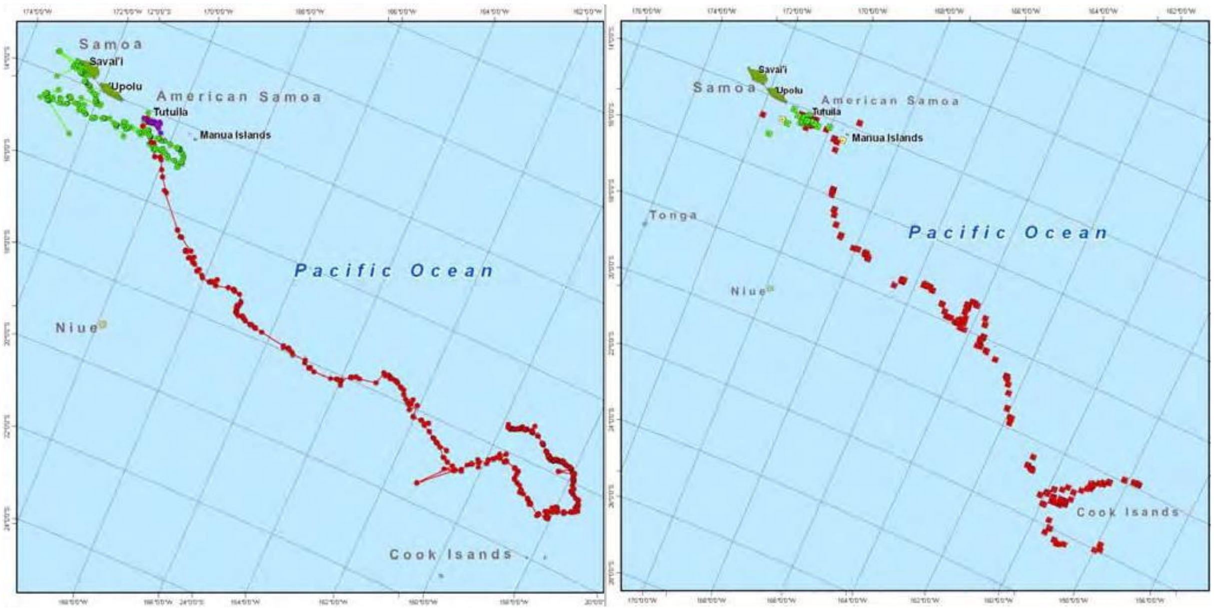


Figure 9-23. Movements of post-nesting hawksbill turtles from Tutuila, American Samoa, to the Cook Islands. Image source: Tagarino et al. 2008.

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)

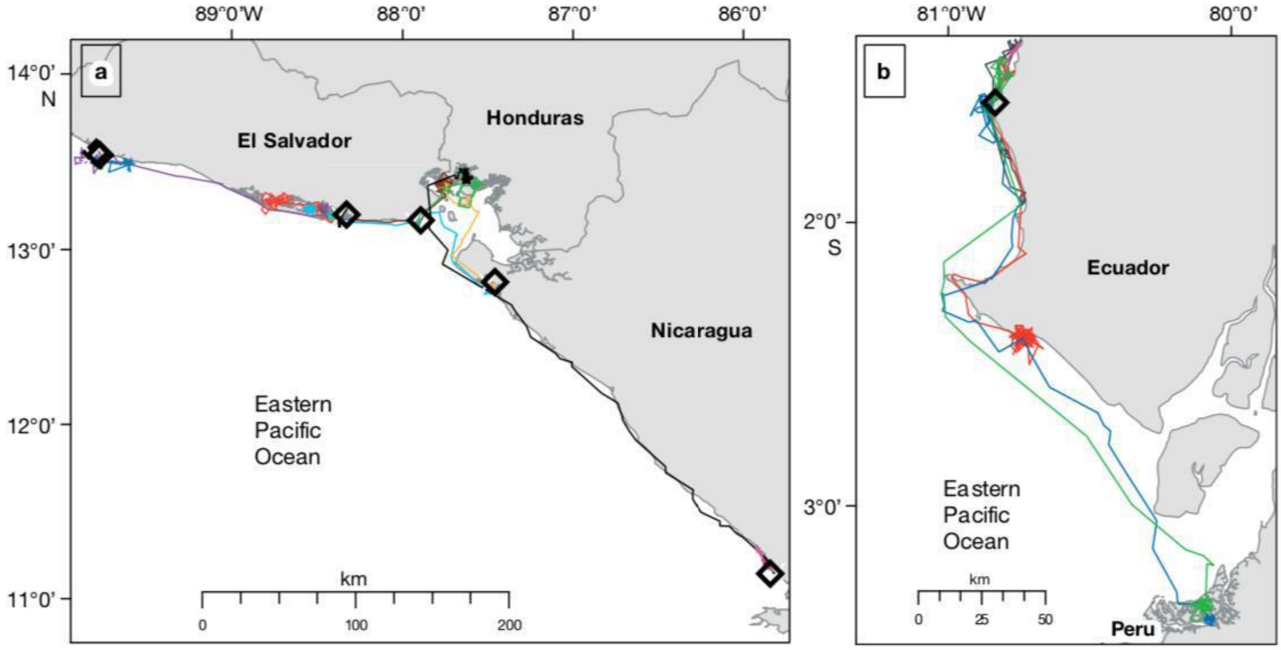
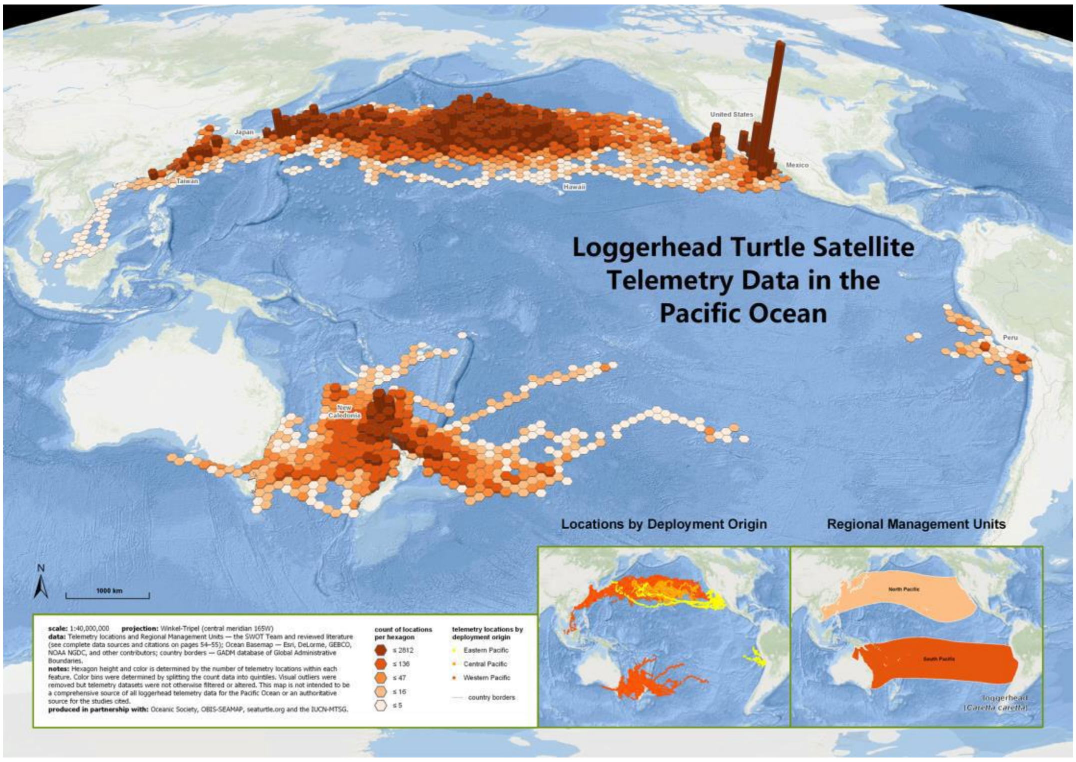


**Figure 9-24. Tracks of 15 hawksbills equipped with satellite tags in (a) central America and (b) Ecuador in the present study between 2008 and 2010. Image source: Gaos et al. 2012**

## 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).



**scale:** 1:40,000,000 **projection:** Winkel-Tripel (central meridian 165W)
**data:** Telemetry locations and Regional Management Units — the SWOT Team and reviewed literature (see complete data sources and citations on pages 54–55); Ocean Basemap — Esri, DeLorme, GEBCO, NOAA NGDC, and other contributors; country borders — GADM database of Global Administrative Boundaries.
**notes:** Hexagon height and color is determined by the number of telemetry locations within each feature. Color bins were determined by splitting the count data into quintiles. Visual outliers were removed but telemetry datasets were not otherwise filtered or altered. This map is not intended to be a comprehensive source of all loggerhead telemetry data for the Pacific Ocean or an authoritative source for the studies cited.
**produced in partnership with:** Oceanic Society, OBIS-SEAMAP, seaturtle.org and the IUCN-MTSG.

**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.

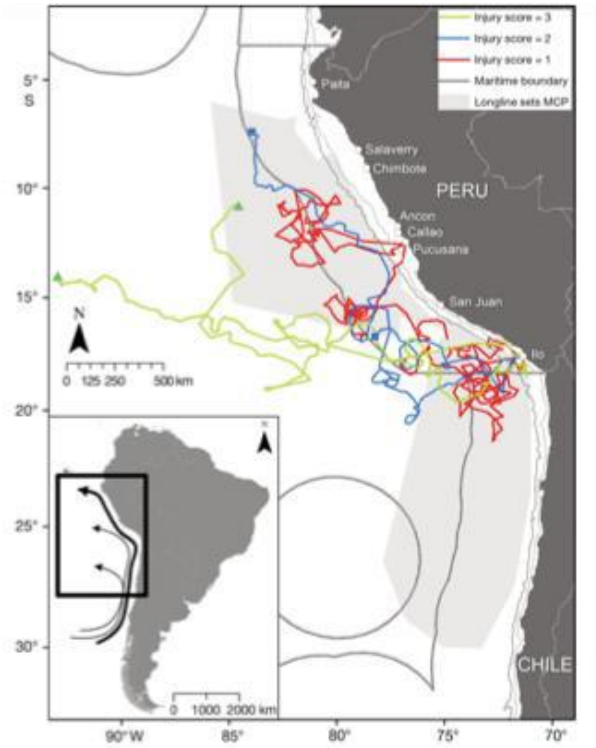


**Figure 9-26. Loggerhead movements after entanglement in Peruvian shark fisheries. Tracks are colour coded by injury level, and shaded polygon depicts the extent of the fishery. Image source: Mangel et al. 2011.**

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).

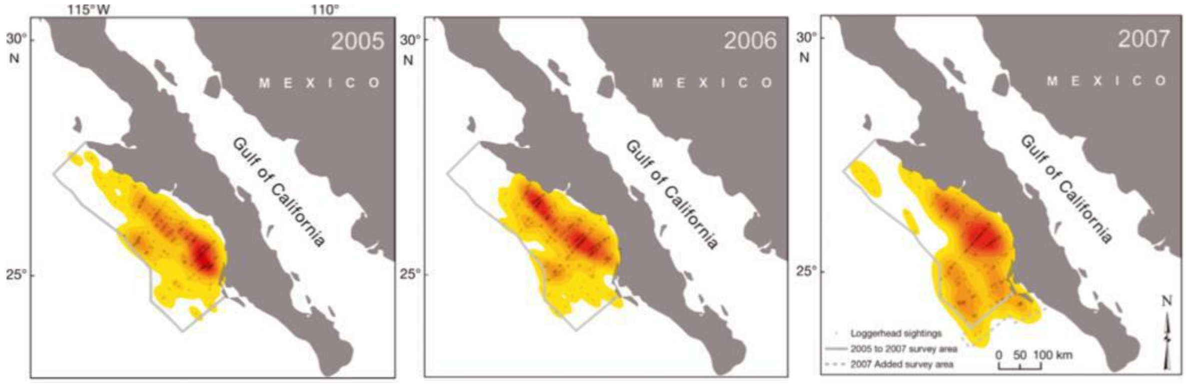


**Figure 9-27. Fixed kernel density plot for loggerhead sea turtle sightings during the 2005 to 2007 aerial surveys along the Pacific Coast of Baja California. Image source: Seminoff et al. 2014.**

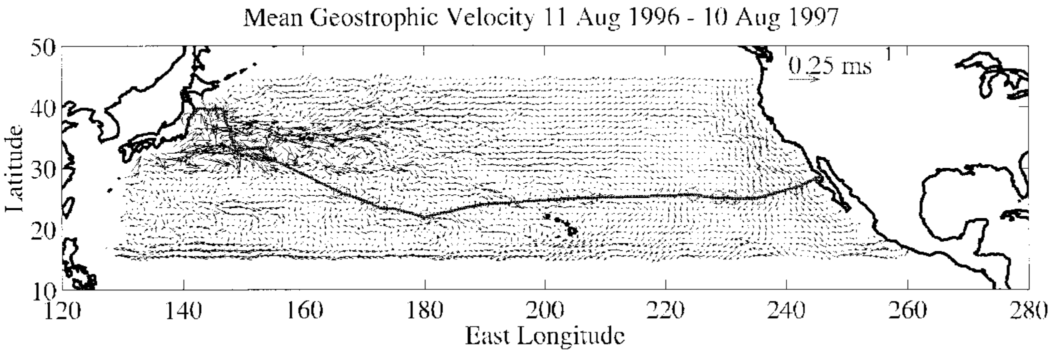


**Figure 9-28. Track of transpacific movement of loggerhead turtle (07667) from Mexico to Japan, during 1996–1997, monitored using satellite telemetry. Image source: Nichols et al. 2000.**

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).

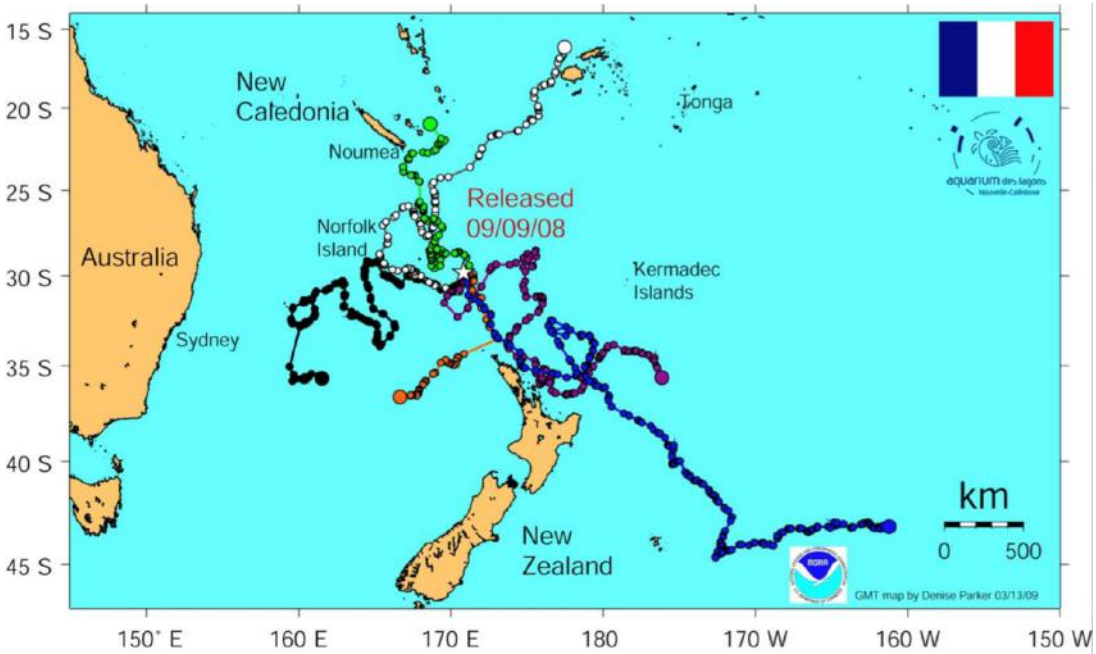


**Figure 9-29. Post-release movements of juvenile loggerheads following several years of being raised in captivity. Image source: Parker 2006.**

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).

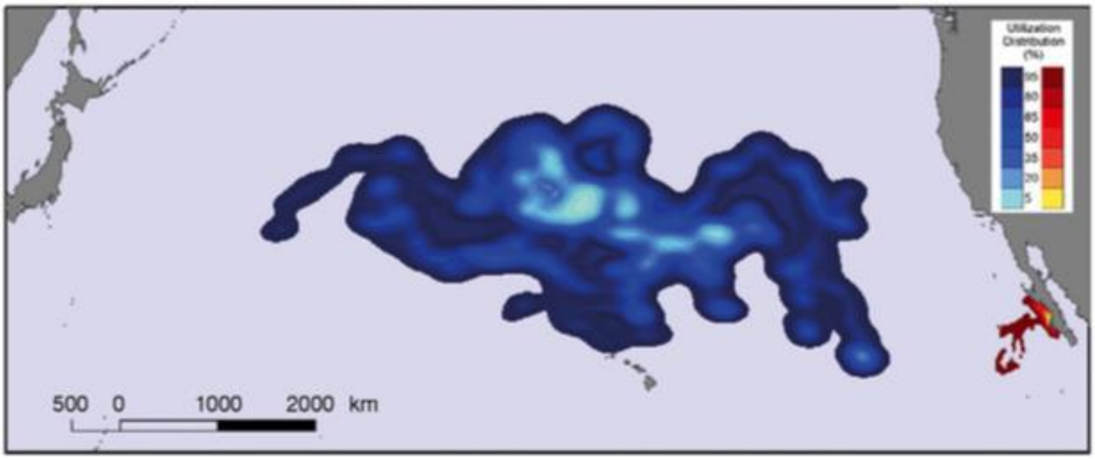


**Figure 9-30. Utilisation distributions of juvenile turtles in the central north Pacific (blue gradient) and at the Baja California peninsula (red gradient). Image source: Peckham et al. 2011.**

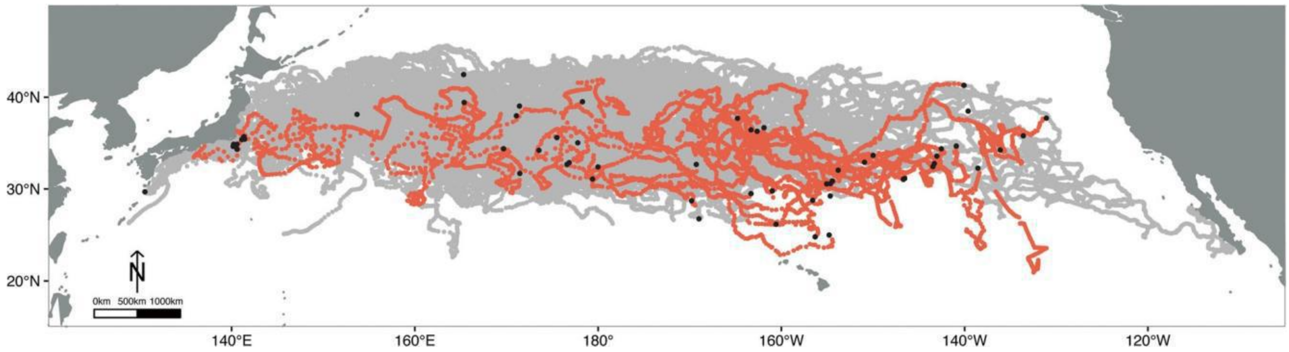


**Figure 9-31. Tracks of 231 satellite tagged juvenile loggerhead sea turtle locations from 1997 to 2014. Grey tracks represent 197 captive-reared juveniles; red tracks represent wild-caught juveniles. Black dots indicate the deploy locations for individual tracks. Image source: Briscoe et al. 2016.**

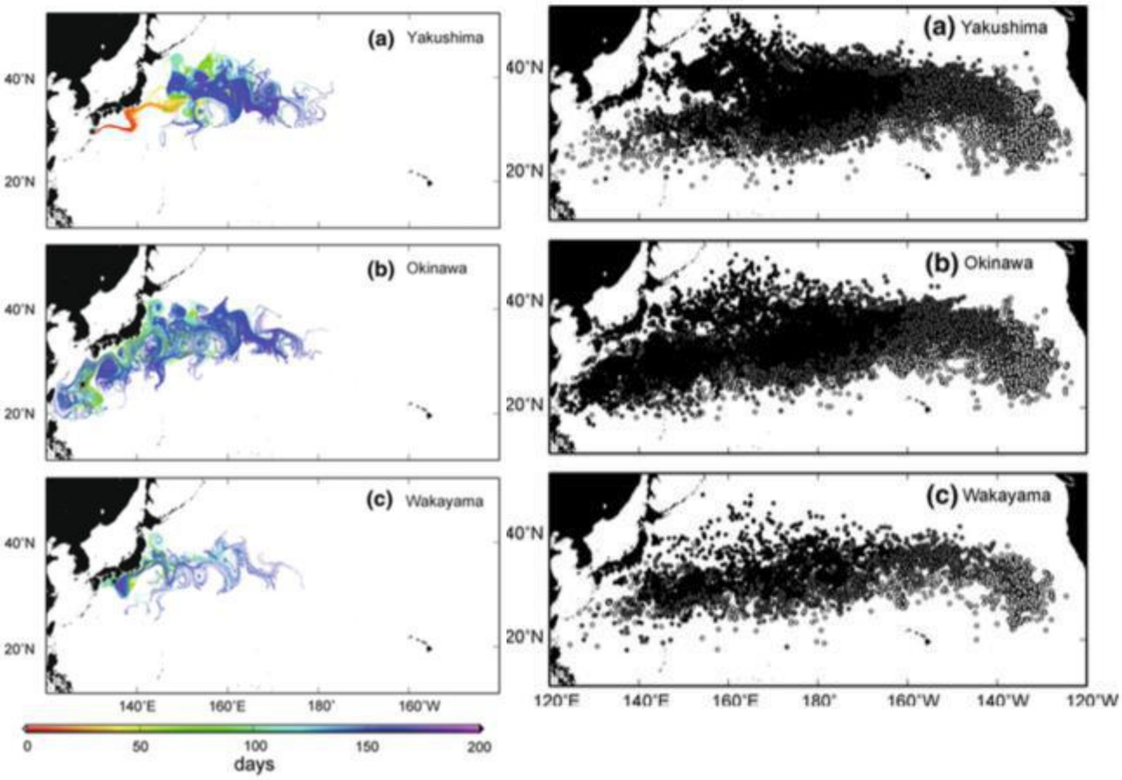


Figure 9-32. Modelled particle distribution from three key rookeries in Japan in the first 200 days (left), and after one (black dots) and five (white dots) years (right). Image source: Okuyama et al. 2011.

## 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.

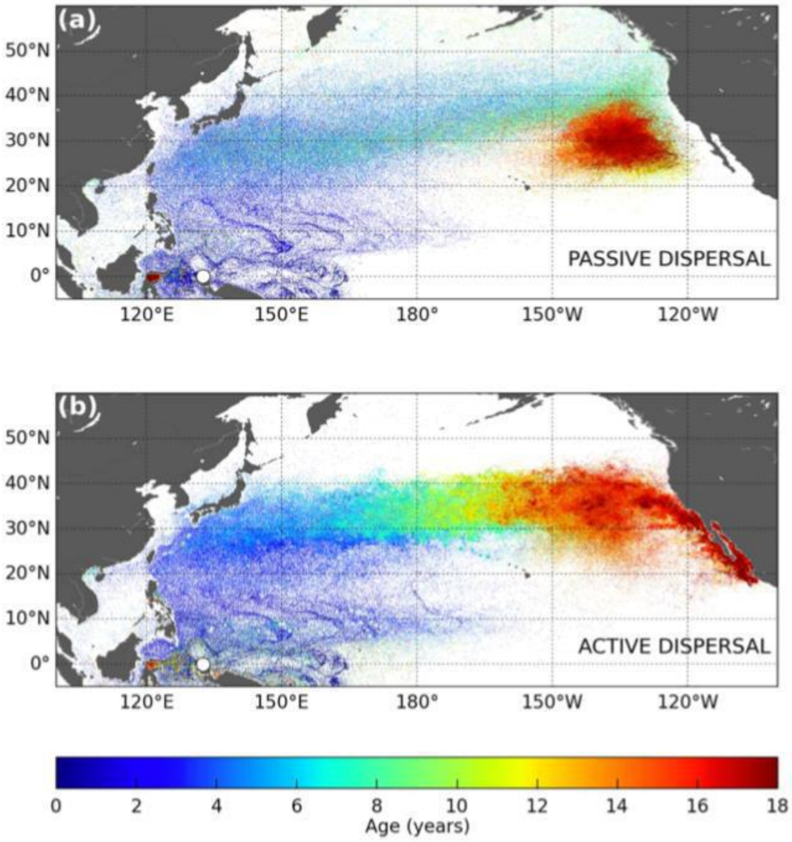


**Figure 9-33. Comparison of active and passive dispersal models for hatchling and juvenile leatherback turtles. Image source: Gaspar and Lalire 2017.**

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).

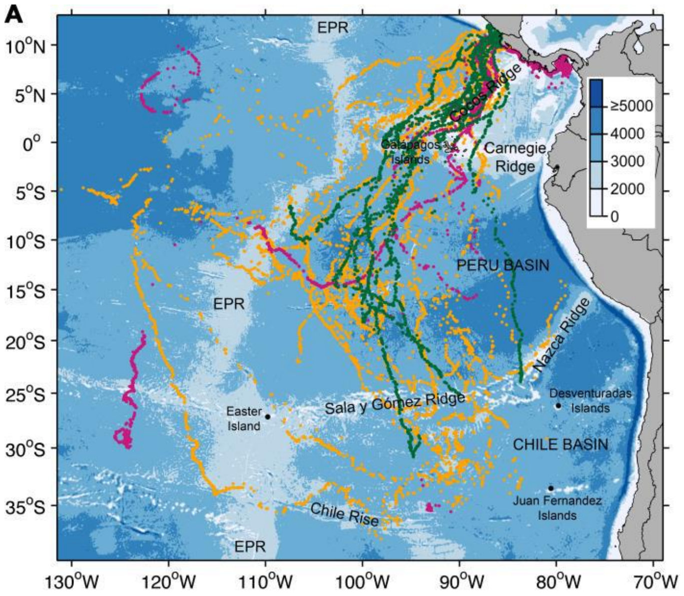


**Figure 9-34. Satellite transmission positions for 46 leatherback turtles from 2004 (n = 27, orange), 2005 (n = 8, purple), and 2007 (n = 11, green), tagged at Playa Grande, Costa Rica, overlaid on bathymetry (in m) Image source: Shillinger et al. 2008.**

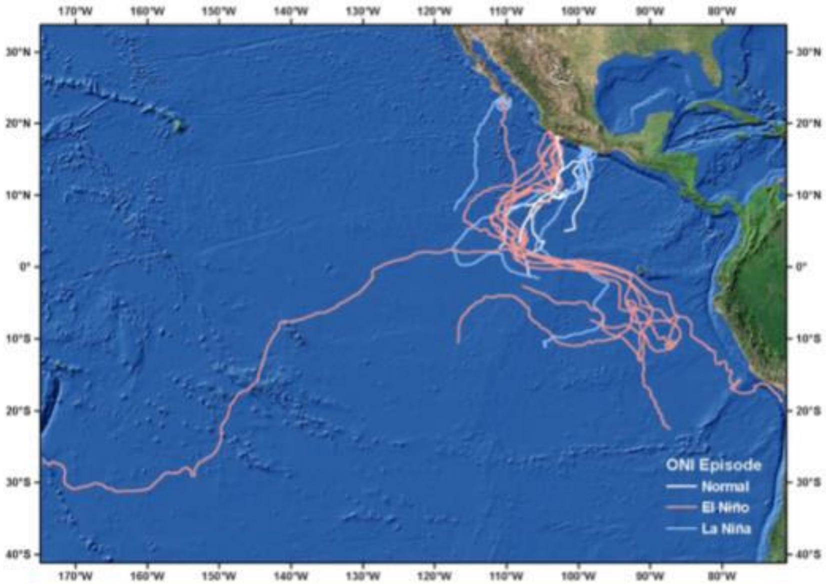


Figure 9-35. Overview map of the tracks from three different beaches in Mexico. Tracks are colour coded by Oceanic Niño Index (ONI) episode status. Image source: Shick et al. 2013.

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.

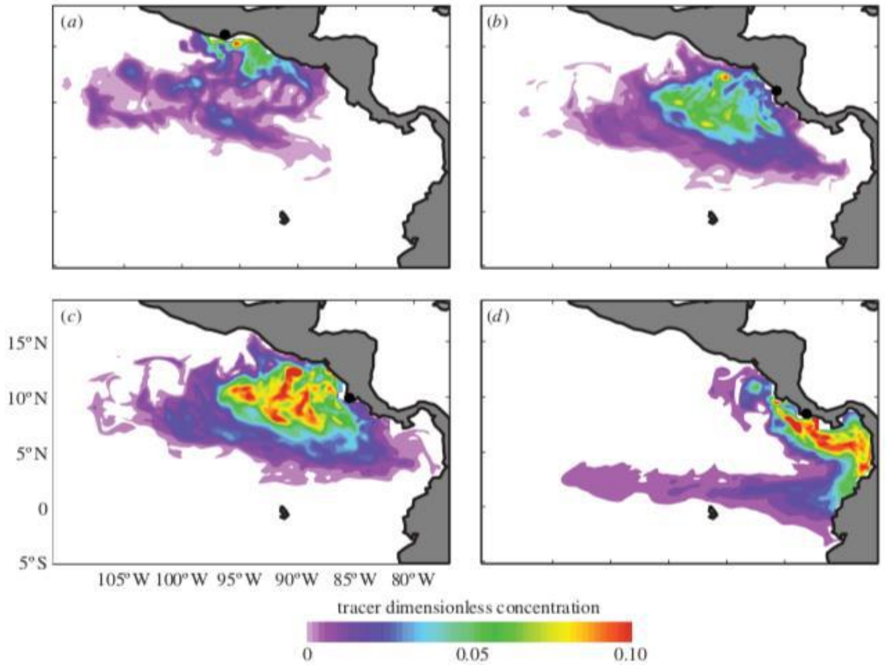


Figure 9-36. Long-term (2000–2008) mean tracer concentration based on continuous tracer releases from nesting beaches at (a) Barra de la Cruz, (b) Playa Chacocente, (c) Playa Grande and (d) Playa Carate. Image source: Shillinger et al. 2012.

## 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).

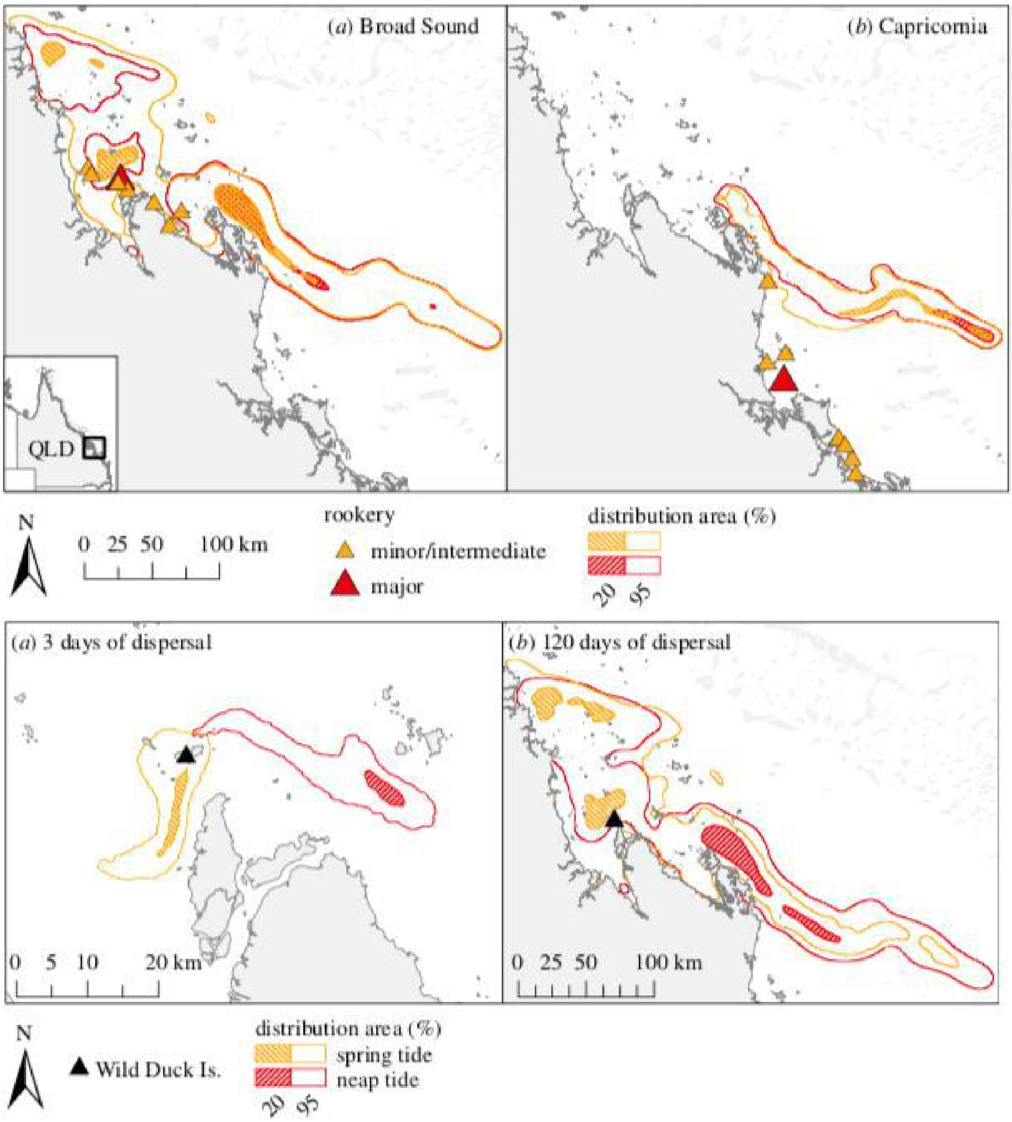


**Figure 9-37. (top) Flatback distribution probabilities after 120 days of passive dispersal from major and minor/intermediate rookeries in (a) Broad Sound and (b) Capricornia. Flatback distribution probabilities from Wild Duck Island after (a) 3 days and (b) 120 days of passive drift. Image source: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

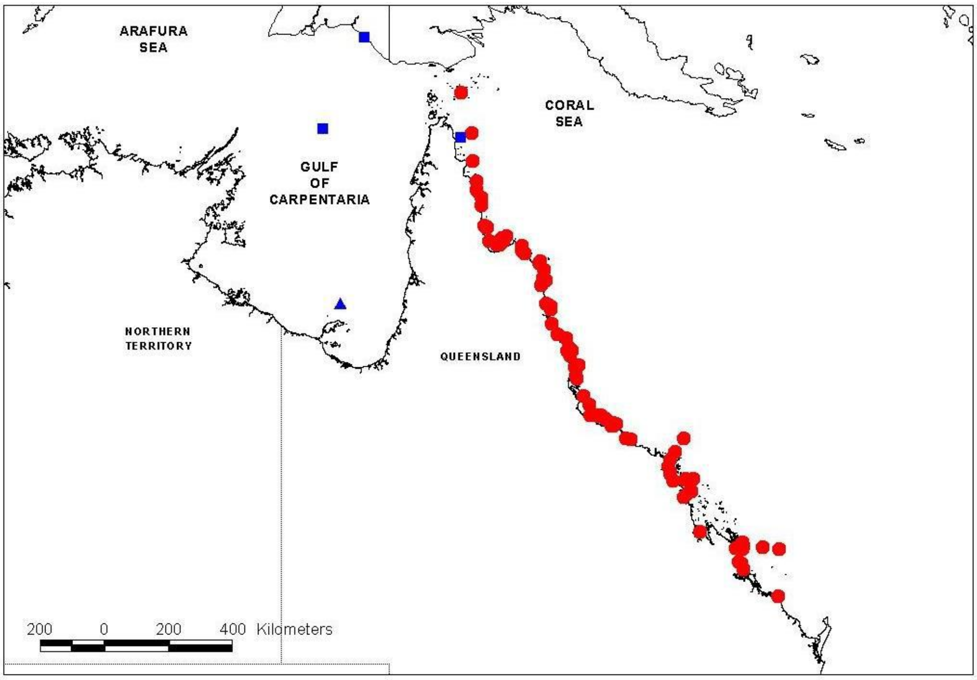


Figure 9-38. Post-nesting dispersal of *Natator depressus* from Queensland rookeries to their respective foraging areas. Symbols denote rookery of origin for the females: circles − eastern Australian rookeries of Mon Repos, Curtis Island, Peak Island and Wild Duck Island; squares – Crab Island in NE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

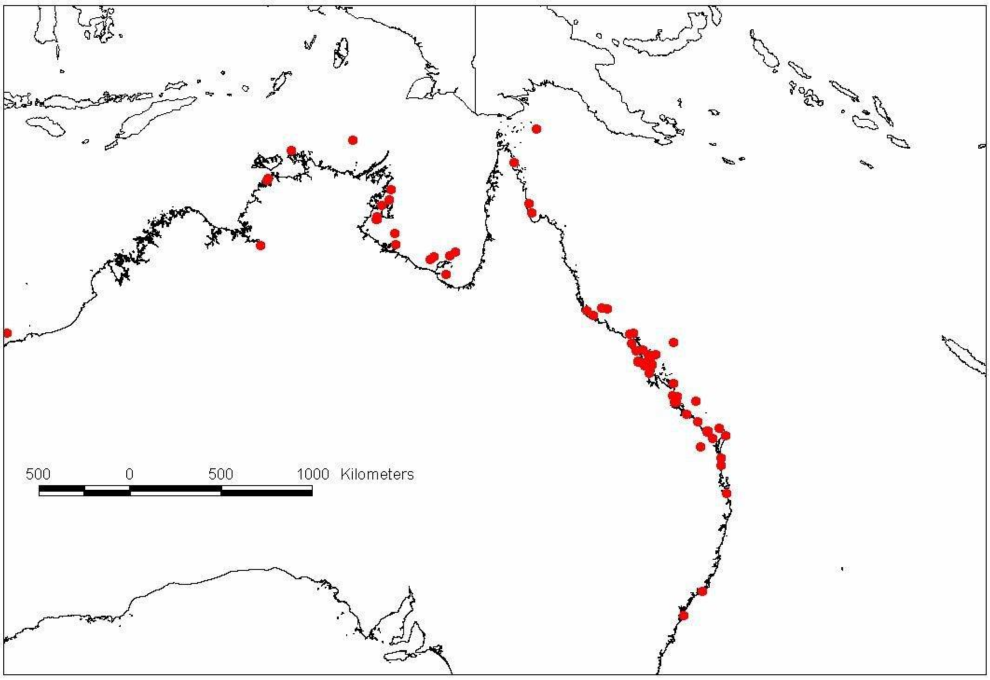


Figure 9-39. Distribution of post-hatchling records from northern and eastern Australia. Image source: Limpus 2007e.

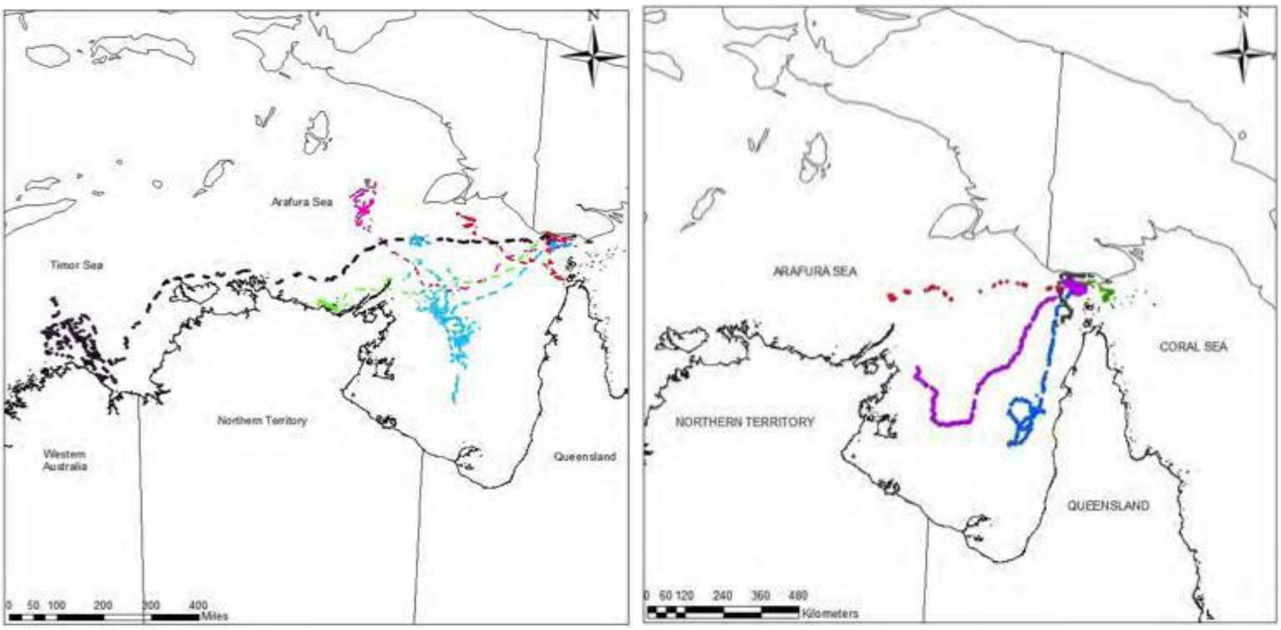


**Figure 9-40. Migration routes and foraging areas for five female flatback turtles after nesting at Warul Kawa in 2013 (left) and six female flatback turtles after nesting at Warul Kawa in 2014. Image source: Hamann 2015.**

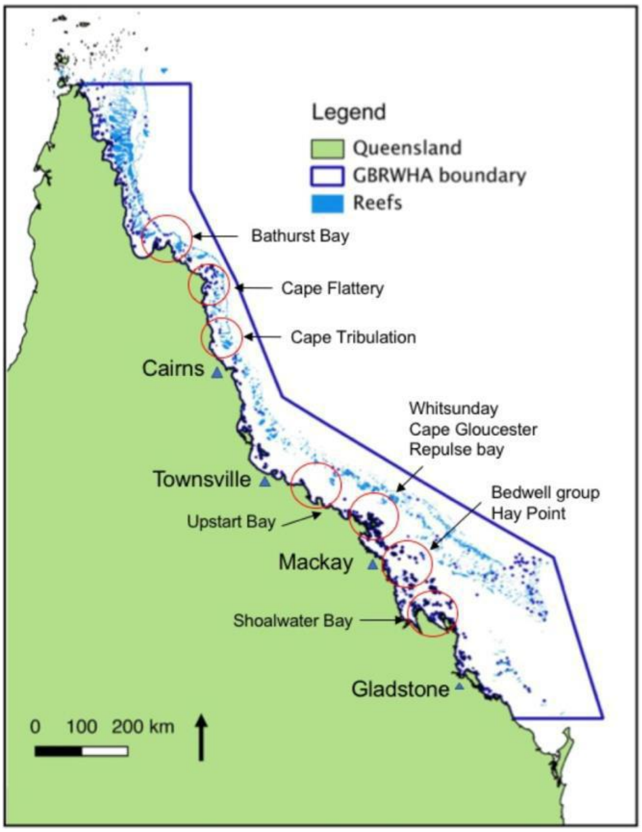


**Figure 9-41. Migration routes of flatback turtles after nesting at Curtis island 2013–2017. Image source: Hamann et al. 2017**

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).

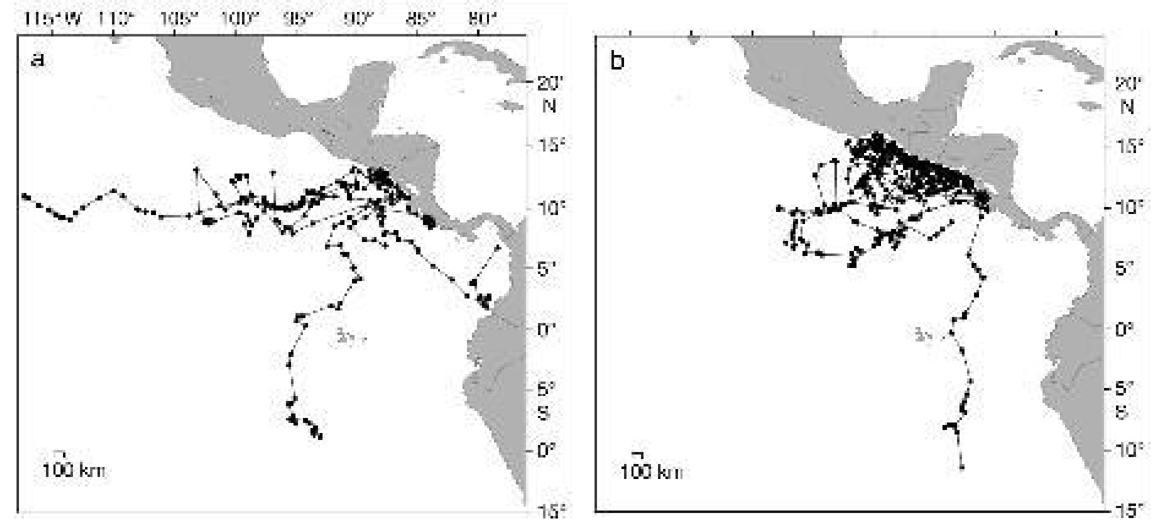


**Figure 9-42. Post-nesting migratory tracks of adult female olive ridley turtles from Costa Rica in (a) 1990–1991 and (b) 1992. Image source: Plotkin 2010.**

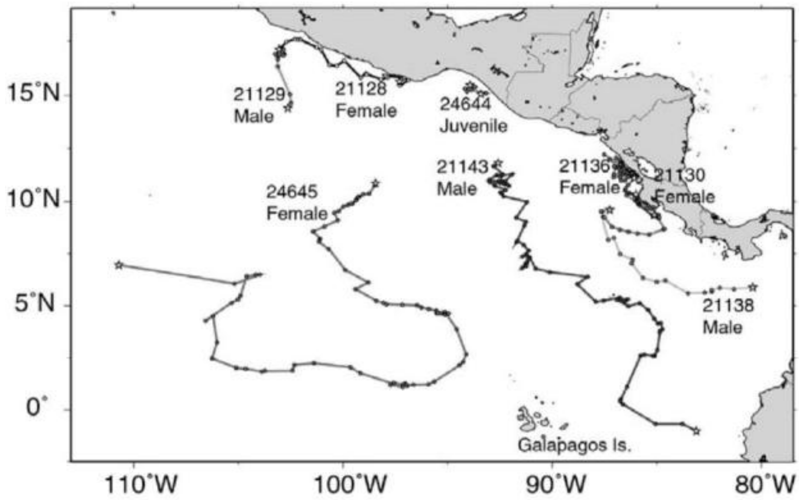


**Figure 9-43. Post-release movements of olive ridley turtles (four female, three male, and one juvenile) that were released during the 1999 Stenella Research Cruise in the ETP. Image source: Parker et al. 2003.**

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).

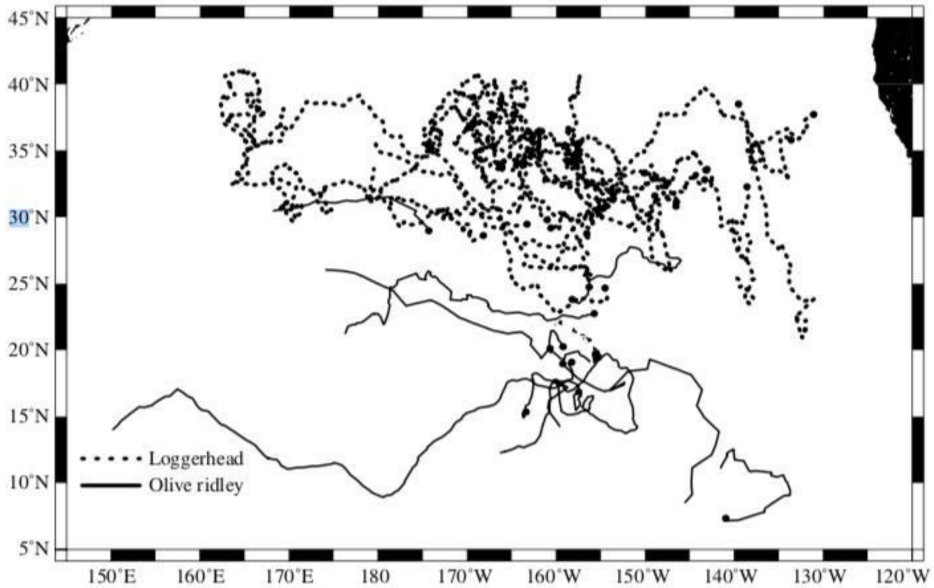


Figure 9-43. Post-release movements of 10 olive ridley turtles (solid lines) captured and released from longline fishing fleets. Image source: Polovina et al. 2004.

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).

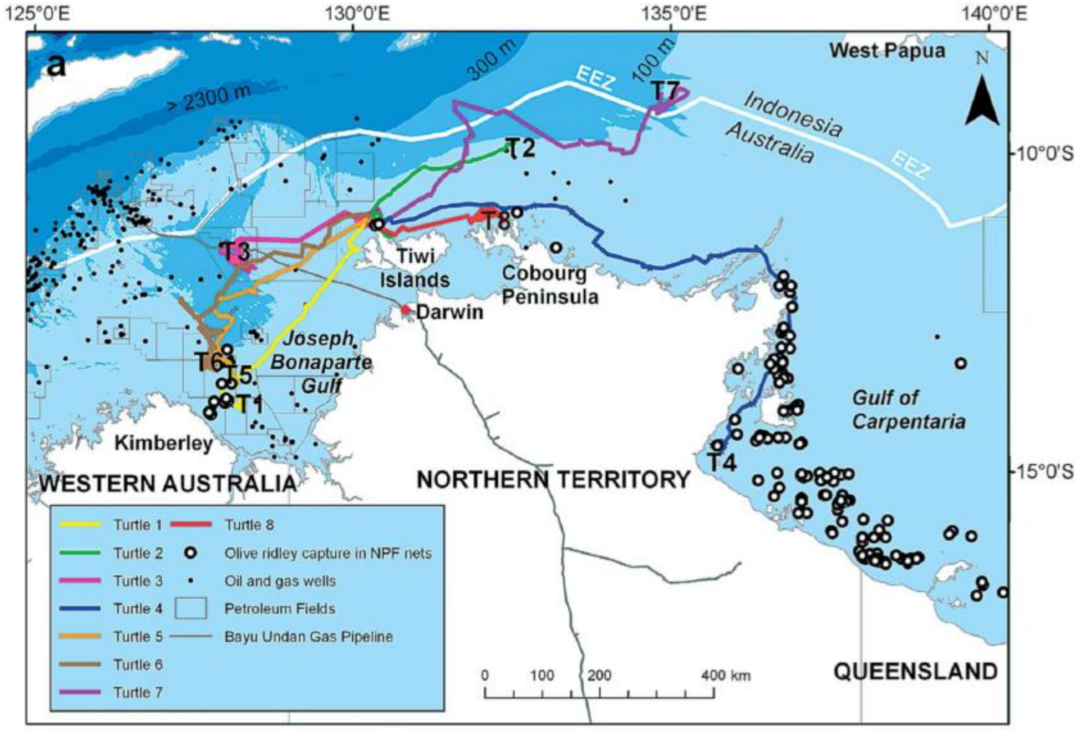


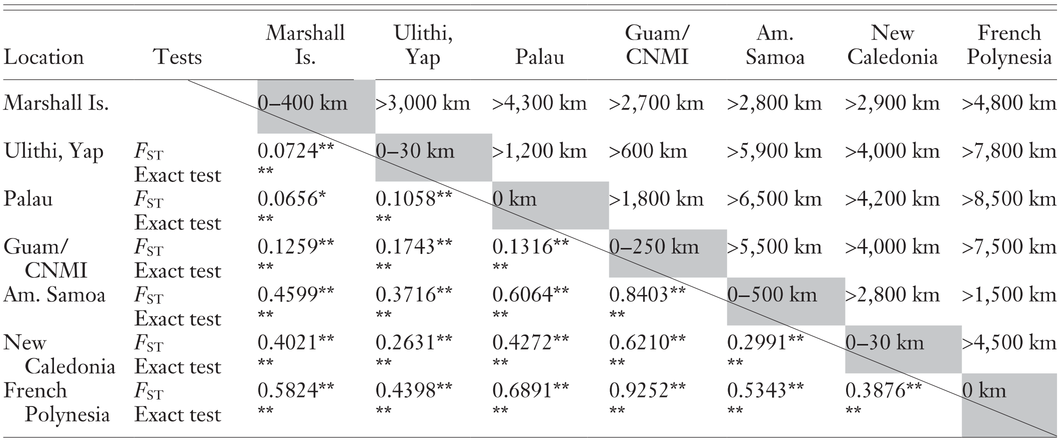
Figure 9-44. Post-release movements of eight olive ridley turtles from on Turtle Melville Island, northern Australia, in 2004 and 2005 demonstrating that none of the turtles reached the Pacific. Image source: Whiting et al. 2007.

# 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).

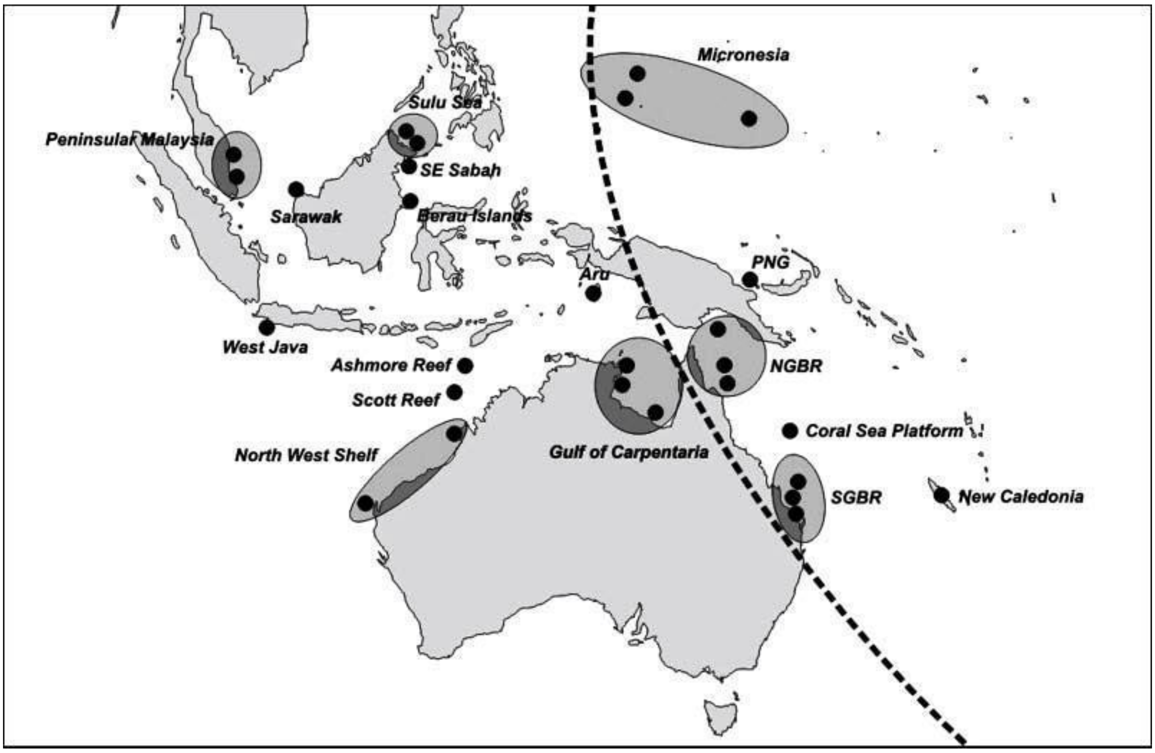


**Figure 10-1. Location of 17 genetically distinct breeding stocks or management units as inferred from analysis of geographical structure of mtDNA variants and position of the genetic barrier (dashed line), indicating the major genetic discontinuity between the Pacific Ocean rookeries from those to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

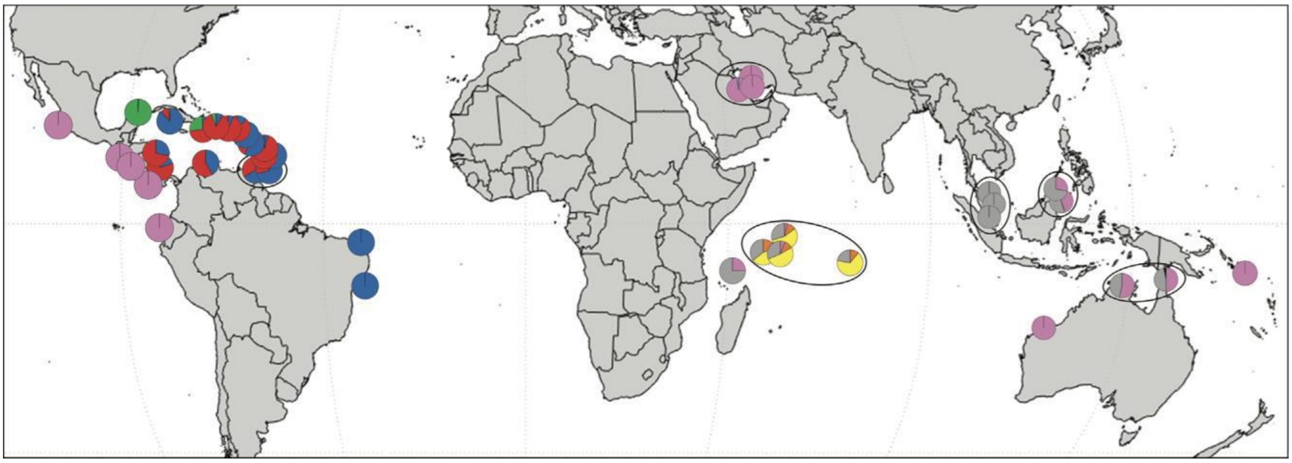


**Figure 10-2 Frequencies of control region haplotypes (739 bp) from each of nine mtDNA lineages in the hawksbill turtle rookeries. Image source: Arantes et al. 2020.**

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.

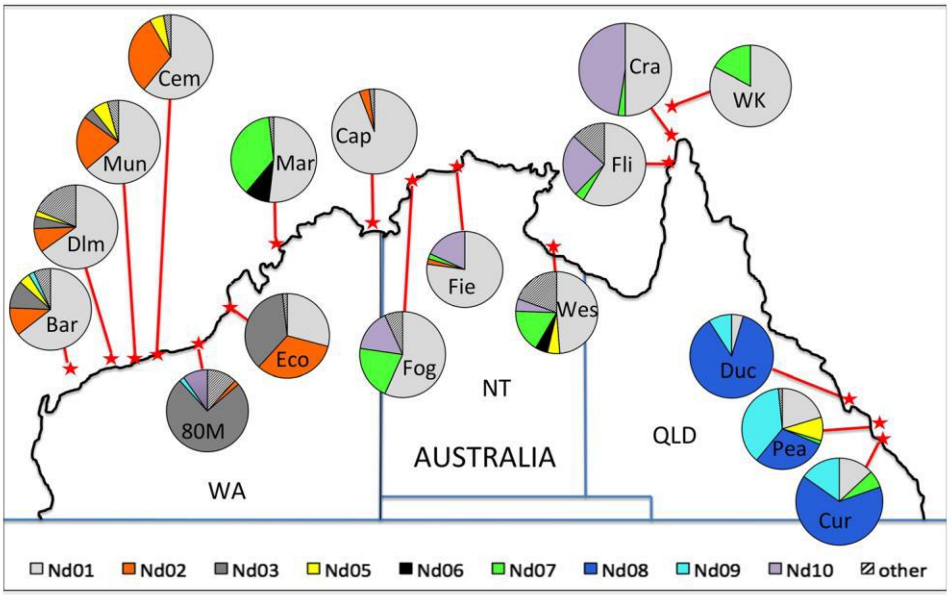


**Figure 10-3 Distribution of the nine most common mitochondrial DNA haplotypes and combined 'other' category, sampled from 17 flatback turtle (*Natator depressus*) rookeries. Image source: FitzSimmons et al. 2020.**

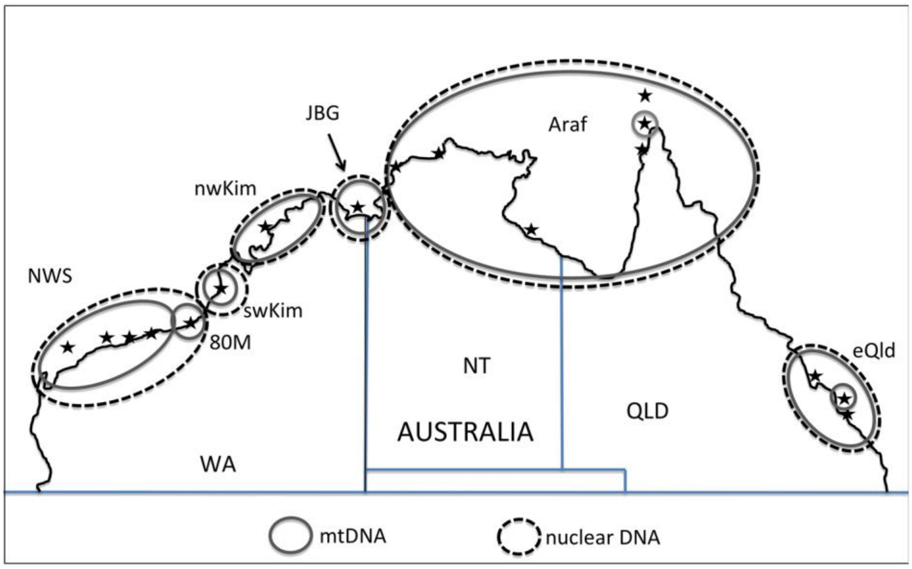


**Figure 10-4 Designated flatback turtle (*Natator depressus*) genetic stocks based on the analyses of 17 rookeries across their range. Image source: FitzSimmons et al. 2020.**

## 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.

# Literature Cited – Status of Sea Turtles in the Pacific

Abreu-Grobois A. and Plotkin P. 2008. *Lepidochelys olivacea*. The IUCN Red List of Threatened Species 2008: https://dx.doi.org/10.2305/IUCN.UK.2008.RLTS.T11534A3292503.en. Downloaded on 07 January 2021. IUCN SSC Marine Turtle Specialist Group.

Adnyana W. 2006. Status of leatherback turtles in Indonesia. In: Assessment of the conservation status of the leatherback turtle in the Indian Ocean and SouthEast Asia. Indian Ocean and SouthEast Asia (IOSEA) Species Assessment: Volume I (M Hamann, C Limpus, G Hughes, J Mortimer and N Pilcher, compilers). IOSEA Marine Turtle Memorandum of Understanding Secretariat, Bangkok, Thailand. pp. 51-63

Alava J.J., Pritchard P., Wyneken J. and Valverde H. 2007. First documented record of nesting by the olive ridley turtle. Chelonian Conservation and Biology 6(2):282-285.

Alfaro-Shigueto J., Mangel J. and Dutton P. 2006. Loggerhead turtle bycatch in Peru. In Proceedings of the Second Western Pacific Sea Turtle Cooperative Research and Management Workshop (I Kinan, compiler). Volume II: North Pacific Loggerhead Sea Turtles. Western Pacific Regional Fishery Management Council, Honolulu, Hawaii. pp. 43-44

Alfaro-Shigueto J., Mangel J., Diaz P., Seminoff J. and Dutton, P. 2008. Longlines and sea turtle bycatch in Peru. In: Proceedings of the Twenty-fifth Annual Symposium on Sea Turtle Biology and Conservation (H Kalb, A Rohde, K Gayheart, and K Shanker compilers). NOAA Technical Memorandum NMFS-SEFSC-582. p. 98

Alfaro-Shigueto J., Mangel J.C., Seminoff J.A. and Dutton P.H. 2008. Demography of loggerhead turtles *Caretta caretta* in the southeastern Pacific Ocean: fisheries-based observations and implications for management. Endangered Species Research, 5: 129-135. doi: 10.3354/esr00142.

Alfaro-Shigueto J., Mangel J., Donoso M. and Marquez J.C. 2009. Summary of Gillnet Fisheries and Sea Turtle Interactions in Peru and Chile. In: Procs. of the Technical Workshop on Mitigating Sea Turtle Bycatch in Coastal Net Fisheries (E Gilman, ed.). 20-22 January 2009, Honolulu. WPRFMC, IUCN, SEAFDEC, IOSEA MoU, NMFS SEFSC. pp. 18-20.

Alfaro-Shigueto J., Mangel J.C., Caceres C., Seminoff J.A., Gaos A. and Yanez I. 2010. Hawksbill turtles in Peruvian coastal fisheries. Marine Turtle Newsletter, 129: 19-21 pp.

Alfaro-Shigueto J., Mangel J.C., Pajuelo M., Dutton P.H, Seminoff J.A. and Godley B.J. 2010. Where small can have a large impact: Structure and characterization of small-scale fisheries in Peru. Fisheries Research, 106(1): 8-17.

Alfaro-Shigueto J., Mangel J.C., Bernedo F., Dutton P.H., Seminoff J.A and Godley B.J. 2011. Small-scale fisheries of Peru: a major sink for marine turtles in the Pacific. Journal of Applied Ecology, 48: 1432-1440.

Allain V., Pilling G.M., Williams P.G., Harley S., Nicol S. and Hampton J. 2016. Overview of tuna fisheries, stock status and management framework in the Western and Central Pacific Ocean. In: Fisheries in the Pacific: The challenges of governance and sustainability (S Pauwels and E Fache, eds.). Pacific-Credo Publications, Marseille pp. 19-48.

Alvarado-Díaz J. and Figueroa A. 1990. The ecological recovery of the sea turtles of Michoacan, Mexico. Special attention: Black turtle (*Chelonia agassizi*). Final Report to USFWS. 113 pp.

Alvarado J. and Figueroa A. 1991. Comportamiento Reproductivo de la tortuga negra *Chelonia agassizi.* Ciencia Y Desarrollo 17(98):43-49.

Alvarado-Díaz J., Delgado-Trejo C. and Suazo-Ortuño I. 2001.Evaluation of the Black Turtle Project in Michoacán, México. Marine Turtle Newsletter 92:4-7.

Alvarado-Díaz J., Arias-Coyotl E. and Delgado-Trejo C. 2003. Clutch Frequency of the Michoacán Green Seaturtle. Journal of Herpetology*,* 37(1):183–185**.**

Álvarez-Varas R., Contardo J., Heidemeyer M., Forero-Rozo L, Brito B., Cortés V., Brain M.J., Pereira S. and Vianna J.A. 2017. Ecology, health and genetic characterization of the southernmost green turtle (*Chelonia mydas*) aggregation in the Eastern Pacific: implications for local conservation strategies. Latin American Journal of Aquatic Research, 45(3): 540-554. doi: 10.3856/vol45-issue3-fulltext-4

Álvarez-Varas R., Flores M., Demangel D, García M. and Sallaberry-Pincheira N. 2015a. First confirmed report of Hawksbill Sea Turtle *Eretmochelys imbricata* in nearshore waters of Easter Island (Rapa Nui). Revista de biología marina y oceanografía, 50(3): 597-602. doi: 10.4067/S0718-19572015000400018

Álvarez-Varas R., Petitpas R., Stowhas P. and Fuentes-Hurtado M. 2015b. Conservation research needs of Easter Island (*Rapa Nui*) marine turtles. Chelonian Conservation and Biology, 14(2): 184-192.

Amorocho D. 1994. Marine turtles on Playa Larga (Chocó, Colombia): a conservation model. In: Proceedings of the Thirteenth Annual Symposium on Sea Turtle Biology and Conservation (BA Schroeder and BE Witherington, compilers). NOAA Technical Memorandum NMFS-SEFSC-341. pp. 4-5.

Amorocho D., Rubio H. and Diaz W. 1992. Oservaciones sobre el estado actual de las tortugas marinas en el Pacifico Colombiano. In: Contribucion al Conocimiento de las Tortugas Marinas de Colombia (R Machecha and H Sanchez, eds.). Serie de Publicaciones Especiales del INDERENA, Bogotá. pp. 155-179

Amorocho D., Barreto L. and Zapata L. 2005. Disminución del impacto por captura incidental en tortugas marinas con el uso de anzuelos curvos. Informe Técnico WWF-Colombia, Cali

Amorocho D.F., Abreu-Grobois F.A., Dutton P.H. and Reina R.D. 2012. Multiple Distant Origins for Green Sea Turtles Aggregating off Gorgona Island in the Colombian Eastern Pacific. PLoS ONE 7(2): e31486. doi: 10.1371/journal.pone.0031486

Andrew N.L., Bright P., de la Rua L., Teoh S.J. and Vickers M. 2019. Coastal proximity of populations in 22 Pacific Island Countries and Territories. PLoS ONE 14(9): e0223249. https://doi.org/10.1371/journal.pone.0223249

Anhalzer G., Ferrin J.L., Baquero A., Vallejo F., Muñoz J.P., Vera F. and Chalen X. 2012. Machalilla National Park, a critical nesting site for hawksbill (*Eretmochelys imbricata*) and green (*Chelonia mydas*) sea turtles in the eastern Pacific. In: Jones T.T. and Wallace B.P. (comps). Proc 31st Annual Symposium on Sea Turtle Biology and Conservation, San Diego, CA. NOAA Tech Memo NMFS-SEFSC-631. p 208–209.

Arantes L.S., Vargas S.M. and Santos F.R. 2020. Global phylogeography of the critically endangered hawksbill turtle (Eretmochelys imbricata). Genetics and Molecular Biology 43(2) doi.org/10.1590/1678-4685-GMB-2019-0264

Arauz R., Rodríguez O., Vargas R. and Segura A. 2000. Incidental capture of sea turtles by Costa Rica's longline fleet. In: Procs. of the 19th Annual Symposium on Sea Turtle Biology and Conservation (H. Kalb and T. Wibbels, compilers) NOAA Tech Memo NMFS-SEFSC-443, pp. 21–26

Aridjis H. 1990. Mexico proclaims total ban on harvest of turtles and eggs. Marine Turtle Newsletter 50:1-3.

Australian Fisheries Management Authority. 2012. Final Report: Protected Species interactions reported in Commonwealth Fishery logbooks for the period 1 January to 31 March 2012.

Baker J.D., Littnam C.L. and Johnston D.W. 2006. Potential effects of sea level rise on the terrestrial habitats of endangered and endemic megafauna in the Northwestern Hawaiian Islands. Endangered Species Research, 2:21-30.

Balazs G.H., 1975. Marine turtles in the Phoenix Islands. Atoll Research Bulletin 184, 1–7.

Balazs G.H. 1980. Synopsis of biological data on the green turtle in Hawaiian Islands. NOAA Tech. Memo. NMFS, Honolullu, HI 141 pp.

Balazs G.H. 1983. Sea turtles and their traditional usage in Tokelau. Atoll Research Bulletin No. 279. The Smithsonian Institution, Washington, DC. 30 pp.

Balazs G.H. 1985. Status and ecology of marine turtles at Johnston Atoll. Atoll Research Bulletin No. 285. The Smithsonian Institution, Washington, DC. 46 pp.

Balazs G.H. 1995. Status of sea turtles in the central Pacific Ocean. In: Biology and Conservation of Sea Turtles (Bjorndal K ed). Smithsonian Institution Press. Washington D.C., 615 pp.

Balazs G.H. 2009. Historical summary of sea turtle observations at Rose Atoll, American Samoa, 1839-1993. Unpublished internal report compiled by George Balazs, Marine Turtle Research Program, NOAA, National Marine Fisheries Service, Pacific Islands Fisheries Science Center.

Balazs G.H. and Forsyth R.G. 1986. Status and ecology of marine turtles at Johnston Atoll: 1985 Assessment. NMFS SWFC Honolulu Laboratory Administrative Report H-86-9.

Balazs G.H. and Chaloupka M., 2004. Spatial and temporal variability in somatic growth of green sea turtles (*Chelonia mydas*) resident in the Hawaiian Archipelago. Marine Biology 145: 1043–1059.

Balazs G.H. and Chaloupka M. 2004.Thirty-year recovery trend in the once depleted Hawaiian green sea turtle stock. Biological Conservation, 117(5):491-498. doi: 10.1016/j.biocon.2003.08.008

Balazs G.H., Siu P. and Landret J.P. 1992. Ecological aspects of green turtles nesting at Scilly Atoll in French Polynesia.

Balazs G.H., Houtan K.S.V., Hargrove S.A., Brunson S.M. and Murakawa S.K.K. 2015. A Review of the Demographic Features of Hawaiian Green Turtles (*Chelonia mydas*). Chelonian Conservation and Biology, 14(2): 119-129. doi: 10.2744/CCB-1172.1

Balazs G.H., Parker D.M. and Rice M.R. 2017. Ocean pathways and residential foraging locations for satellite tracked green turtles breeding at French Frigate Shoals in the Hawaiian Islands. Micronesica 2017-04: 1–19

Baldwin R., Hughes G.R. and Prince R.I.T. 2003. Loggerhead turtles in the Indian Ocean. In: Loggerhead Sea Turtles (AB Bolten and BE Witherington, eds.) Smithsonian Institution, Washington, DC pp. 218 – 232.

Barragan A.R., Dutton P.H. and Abreu-Grobois A. 1998. Population genetics of the leatherback turtle in the Mexican Pacific. In: Epperly S.P. and J Braun (compilers). Procs. Seventeenth Annual Sea Turtle Symposium. U.S. Dep. Commer. NOAA Tech. Memo. NMFS-SEFSC-415 pp 6-7.

Batibasaga A. Waqainabete S. and Qauqau A. 2006. Notes on Fijian sea turtles: estimates on population status. Fiji Fisheries Department, PO Box 3165, Lami, Fiji. Information provided for Sea Turtle Working Group Meeting – Nadave / CATD 31st May – 1st June 2006.

Bell I.P., Meager J.J., Eguchi T., Dobbs K.A., Miller J.D. and Madden Hof C.A. 2020. Twenty-eight years of decline: Nesting population demographics and trajectory of the north-east Queensland endangered hawksbill turtle (*Eretmochelys imbricata*). Biological Conservation, 241: 108376. Doi: 10.1016/j.biocon.2019.108376.

Bell L., Matoto L. and Fa`anunu U. 2009. Project Report: Marine Turtle Monitoring Programme in Tonga. Marine Turtle Conservation Act Project Report.

Bell L.A.J., Favae T.T., Ruatu N., Bebe R., Anderson P. and Siota C. 2010. Kiribati Marine Turtle Profile. Secretariat of the Pacific Regional Environment Programme. Report. 29 pp.

Benson S.R., Dutton P.H., Hitipeuw C., Samber B., Bakarbessy J. and Parker D. 2007. Post-Nesting Migrations of Leatherback Turtles *(Dermochelys coriacea)* from Jamursba-Medi, Bird's Head Peninsula, Indonesia. Chelonian Conservation and Biology, 6(1): 150-154. doi: 10.2744/1071-8443(2007)6[150:PMOLTD]2.0.CO;2

Benson S.R., Eguchi T., Foley D.G., Forney K.A., Bailey H., Hitipeuw C., Samber B.P., Tapilatu R.F., Rei V., Ramohia P., Pita J. and Dutton P.H. 2011. Large-scale movements and high-use areas of western Pacific leatherback turtles, *Dermochelys coriacea.* Ecosphere 2(7): Art 84. doi: 10.1890/ES11-00053.1

Blanco G.S., Morreale S.J., Velez E., Piedra R., Montes W.M., Paladino F.V. and Spotila J.R. 2011. Reproductive output and ultrasonography of endangered population of East Pacific Green Turtles. The Journal of Wildlife Management.76 (4), 841-846.

Blanco G.S., Morreale S.J., Bailey H., Seminoff J.A., Paladino F.V. and Spotila J.R. 2012. Post-nesting movements and feeding grounds of a resident East Pacific green turtle *Chelonia mydas* population from Costa Rica. Endang Species Res 18:233-245. doi: 10.3354/esr00451

Blasi M.F., Roscioni F. and Mattei D. 2016*.* Interaction of Loggerhead Turtles (*Caretta caretta*) with Traditional Fish Aggregating Devices (FADs) in the Mediterranean Sea Herpetological Conservation and Biology 11:386–401.

Bosserelle P., Singh N., Helagi Kavisi N., Gataua L., Rex F. and Halford A. 2018. Pacific Community, Noumea, New Caledonia. 84 pp.

Bowen B.W. 2003. What is a loggerhead turtle? The genetic perspective. In: Loggerhead Sea Turtles (AB Bolten and BE Witherington (eds.). Smithsonian Books, Washington D.C. pp 7-27

Bowen B.W. 2020. Tuvalu: Sea Turtle Distributions, Threats and Conservation Status. In: Sea Turtles in Oceania MTSG Annual Regional Report 2020 (TM Work, D Parker and GH Balazs, eds). pp. 564-576.

Bowen B.W., Kamezaki N., Limpus C.J., Hughes G.H., Meylan A.B. and Avise J.C. 1994. Global phylogeography of the loggerhead turtle (*Caretta caretta*) as indicated by mitochondrial DNA haplotypes. Evolution 48(6), 1820–1828.

Bowen B.W., Abreu-Grobois F.A., Balazs G.H., Kamezaki N., Limpus C.J. and Ferl R.J. 1995. Trans-Pacific migrations of the loggerhead turtles (*Caretta caretta*) demonstrated with mitochondrial DNA markers. Procs. of the National Academy of Sciences of the United States of America 92:3731-3734.

Bowen B.W., Clark A.M., Abreu-Grobois A.F., Chaves A., Reichart H.A. and Ferl R.J. 1998. Global phylogeography of the ridley sea turtles (*Lepidochelys* spp.) as inferred from mitochondrial DNA sequences. Genetica 101:179-189.

Boyle M.C., FitzSimmons N.N., Limpus C.J., Kelez S., Velez-Zuazo X. and Waycott M. 2009. Evidence for transoceanic migrations by loggerhead sea turtles in the southern Pacific Ocean. Proc. R. Soc. B 276:1993–1999.

Briscoe D.K., Parker D.M., Bograd S., Hazen E., Scales K., Balazs G.H., Kurita M., Saito T., Okamoto H., Rice M., Polovina J. J. and Crowder L.B. 2016. Multi-year tracking reveals extensive pelagic phase of juvenile loggerhead sea turtles in the North Pacific. Movement Ecology 4: 23. doi: 10.1186/s40462-016-0087-4

Broderick D., Moritz C., Miller J.D., Guinea M., Prince R.I.T. and Limpus C.J. 1994. Genetics studies of the hawksbill turtle (*Eretmochelys imbricata*): evidence for multiple stocks in Australian waters. Pacific Conservation Biology 1, 123–131.

Brooke M. de L. 1995. Seasonality and numbers of green turtles *Chelonia mydas* nesting on the Pitcairn Islands. Biological Journal of the Linnean Society 56 (1–2): 325–327.

Byrne R. and Hines E. 2005. The marinet of Hinatuan Bay, Surigao del Sur, Mindanao, Philippines. Marine Turtle Newsletter 107: p.8.

Buden D.B. 2008. The Reptiles of Nauru. Pacific Science 62(4), 499-507. doi: 10.2984/1534-6188(2008)62

Buden D.W. 2000. The reptiles of Sapwuahfik Atoll, Federated States of Micronesia. Micronesica, 32(2): 245-256.

Buden D.W. 2010. Reptiles of Ngulu Atoll, Yap State, Federated States of Micronesia. Pacific Science, 64(3): 473-480. doi: 10.2984/64.3.473

Buden D.W. and Edward A. 2001. Abundance and Utilization of Sea Turtles on Pohnpei, Federated States of Micronesia: Islanders' Perceptions. Micronesica 34(1):47–54.

Bustard H.R. 2016. The hawksbill turtle (*Eretmochelys imbricata*): conservation research. *Testudo* 8(3): 50-63.

Butcher M. and Hattingh K. 2013. Gnaraloo feral animal control program. Report for sea turtle nesting season 2012/13. Animal Pest Management Services and Gnaraloo Station Trust, 42 pp.

Casale P. and Matsuzawa Y. 2015 *Caretta caretta* North Pacific subpopulation. The IUCN Red List of Threatened Species 2015: e.T83652278A83652322 http://dx.doi.org/10.2305/IUCN.UK.2015-4.RLTS.T83652278A83652322.en.

Castellanos-Michel R., Martínez-Tovar C., EncisoPadilla I. and Jacobo-Pérez J. 2006. Low presence of the leatherback turtle *Dermochelys coriacea* and hawksbill turtle *Eretmochelys imbricata* in the Jalisco coast, Mexican Pacific (2000-2002). in Proceedings of the 23rd annual symposium on sea turtle biology and conservation (Pilcher N., compiler). U.S. Department of Commerce, National Oceanic and Atmospheric Administration Technical Memorandum NMFS- SEFSC-536. pp 31-34.

Castro J., de la Cruz J., Ramírez P. and Quiñones J. 2012. Captura incidental de tortugas marinas durante El Niño 1997-1998, en el norte del Perú. Latin American Journal of Aquatic Research, 40(4): 970-979.

Chaloupka M.Y. and Limpus C.J. 2002. Survival probability estimates for the endangered loggerhead sea turtle resident in southern Great Barrier Reef waters. Marine Biology 140:267-277.

Chaloupka M. and Limpus C. 2005. Estimates of sex- and age-class-specific survival probabilities for a southern Great Barrier Reef green sea turtle population. Marine Biology, 146: 1251-1261. doi: 10.1007/s00227-004-1512-6.

Chaloupka M., Parker D. and Balazs G. 2004. Modelling post-release mortality of loggerhead sea turtles exposed to the Hawaii-based pelagic longline fishery. Marine Ecology Progress Series, 280: 285-293.

Chaloupka M., Bjorndal K.A., Balazs G.H., Bolten A.B., Ehrhart L.M., Limpus C.J., Suganuma H., Troëng S. and Yamaguchi M. 2007. Encouraging outlook for recovery of a once severely exploited marine megaherbivore. Global Ecology and Biogeography. DOI: 10.1111/j.1466-8238.2007.00367.x

Chaloupka M., Work T.M., Balazs G.H., Murakawa S.K.K. and Morris R. 2008. Cause-specific temporal and spatial trends in green sea turtle strandings in the Hawaiian Archipelago (1982–2003). Marine Biology, 154(5): 887-898. doi:10.1007/s00227-008-0981-4

Cheng I.J. and Wang Y.H. 2009. Influence of surface currents on post-nesting migration of green sea turtles nesting on Wan-An Island, Penghu Archipelago, Taiwan. Journal of Marine Science and Technology, 17:306-311.

Cheng I.J., Dutton P.H., Chen C.L., Chen H.C., Chen Y.H. and Shea J.W. 2008. Comparison of the genetics and nesting ecology of two green turtle rookeries. Journal of Zoology, 276(4): 375-384.

Chuenpagdee R., Liguori L., Palomares M.L.D. and Pauly D. 2006. Bottom-up, global estimates of small-scale marine isheries catches. Fisheries Centre Research Reports, Vancouver, British Columbia, Canada.

Cliff G. and Dudley S.F.J. 1991. Sharks caught in the protective gill nets off Natal, South Africa. 4. The bull shark *Carcharhinus leucas* Valenciennes. South African Journal Marine Sciience 10, 253–270.

Cliffton K., Cornejo D.O. and Felger R.S. 1982. Sea turtles of the Pacific coast of Mexico. In: Biology and Conservation of Sea Turtles (Bjorndal K.A., ed.). Smithsonian Institution Press, Washington, DC pp. 199-209

Comer-Santos K., Livesey M., Fish M. and Lorences A.C. 2015. Climate change implications for the nest site selection process and subsequent hatching success of a green turtle population. Mitigation and Adaptation Strategies for Global Change, 22: 121-135.

Common Oceans. 2017. Joint Analysis of Sea Turtle Mitigation Effectiveness. WCPFC-SC13-2017/EB-WP-10.

Commonwealth of Australia. 2017. Recovery Plan for Marine Turtles in Australia, 2017-2027. Department of the Environment and Energy, Australian Government. 146 pp.

Conant T., Somma A., Lauritsen A.M., Bibb K. and Possardt E. 2014. Olive ridley sea turtle (*Lepidochelys olivacea*) 5-year review: summary and evaluation. National Marine Fisheries Service, Silver Spring, MD, and US Fish and Wildlife Service, Jacksonville, FL. 81 pp.

Conant T.A., Dutton P.H., Eguchi T., Epperly S.P., Fahy C.C., Godfrey M.H., MacPherson S.L., Possardt E.E., Schroeder B.A. Seminoff J.A., SnoverM.L., Upite C.M. and Witherington B.E. 2009. Loggerhead sea turtle (*Caretta caretta*) 2009 status review under the U.S. Endangered Species Act. Report of the Loggerhead Biological Review Team to the National Marine Fisheries Service, August 2009. 222 pp.

Conflict Island Conservation Inititative. 2018. PNG Field Research: Conflict Island nesting season 2017-2018. 8pp. Report prepared for the Conflict Island Conservation Initiative, Panasesa Island, Milne Bay Province, Papua New Guinea.

Conflict Island Conservation Inititative. 2018. PNG Field Research: Conflict Island nesting season 2018-2019. 8pp. Report prepared for the Conflict Island Conservation Initiative, Panasesa Island, Milne Bay Province, Papua New Guinea.

Cornelius S.E., Arauz R., Fretey J., Godfrey M.H., Márquez R. and Shanker K. 2007. Effect of land-based harvest of *Lepidochelys*. In: Biology and Conservation of Ridley Sea Turtles (Plotkin P.T. (ed.). Johns Hopkins University Press, Baltimore, Maryland. pp. 231-251.

Craig P., Parker D., Brainard R., Rice M. and Balazs G. 2004. Migrations of green turtles in the central South Pacific. Biological Conservation, 116: 433-438.

Crouse D.T., Crowder L.B. and Caswell H. 1987. A stage-based population model for Loggerhead sea turtles and implications for conservation. Ecology, 68(5): 1412-1423.

Cruce J.A. 2009. Monitoring of nesting green turtles (Chelonia mydas) in Ulithi Atoll, Yap, Federated Sates of Micronesia. Testudo 7.

Cruce J. and Rulmal Jr. J. 2014. Continuity in a Community Setting: The Ulithi Marine Turtle Program. SWOT Report No. 9. 14-16.

Cummings V. 2002. Sea turtle conservation in Guam. In: Proceedings of the Western Pacific Sea Turtle Cooperative Research and Management Workshop (I Kinan, ed.). February 5– 8, 2002. Honolulu, HI: Western Pacific Regional Fishery Management Council, Honolulu, Hawaii, USA, pp. 37–38.

d'Auzon J.M. 2007. Marine turtles of New Caledonia. In: Compendium of marine species from New Caledonia (CE Payri and R de Forges eds). TYPATOLL. Doe. Sei. Teeh.1l 3,125 p. pp 411-414

Dethmers K.E.M., Broderick D., Moritz C., FitzSimmons N.N., Limpus C.J., Lavery S., Whiting S., Guinea M., Prince R.L.T. and Kennett R. 2006. The genetic structure of Australasian green turtles *(Chelonia mydas)*: exploring the geographical scale of genetic exchange. Molecular Ecology, 15(13): 3931-3946.

Dethmers K.E.M., Jensen M.P., FitzSimmons N.N., Broderick D., Limpus C.J. and Moritz C. 2010. Migration of green turtles (*Chelonia mydas*) from Australasian feeding grounds inferred from genetic analyses. Marine and Freshwater Research, 61: 1376–1387.

Doi T., Storto A., Fukuoka T., Suganuma H. and Sato K. 2019. Impacts of temperature measurements from sea turtles on seasonal prediction around the Arafura Sea. Frontiers in Marine Science 6:719. doi: 10.3389/fmars.2019.00719

Donoso M. and Dutton P.H. 2010. Sea turtle bycatch in the Chilean pelagic longline fishery in the southeastern Pacific: Opportunities for conservation. Biological Conservation, 143: 2672–2684.

Dornfeld T.C., Robinson NJ, Santidrián Tomillo P. and Paladino F.V. 2015. Ecology of solitary nesting olive ridley sea turtles at Playa Grande, Costa Rica. Marine Biology 162:123–139.

Drake D.L. 1996. Marine turtle nesting, nest predation, hatch frequency, and nesting seasonality on the Osa Peninsula, Costa Rica. Chelonian Conservation and Biology 2(1):89- 92.

Dryden J., Grech A., Moloney J. and Hamann M. 2008. Rezoning of the Great Barrier Reef World Heritage Area: does it afford greater protection for marine turtles? Wildlife Research, 35: 477–485.

Dutton P.H., Bowen B.W., Owens D.W. and Bowagan A. 1999. Global phylogeography of the leatherback turtle *(Dermochelys coriacea)*. Journal of Zoology, 248, 397–409.

Dutton P., Broderick D. and FitzSimmons N. 2002. Defining management units: molecular genetics. In: Proceedings of the Western Pacific Sea Turtle Cooperative Research and Management Workshop (I. Kinan, ed.) Western Pacific Regional Fishery Management Council: Honolulu. pp. 93–101.

Dutton P.H., Hitipeuw C., Zein M., Benson S.R., Petro G., Pita J., Rei V., Ambio L. and Bakarbessy J. 2007. Status and genetic structure of nesting populations of leatherback turtles (*Dermochelys coriacea*) in the Western Pacific. Chelonian Conservation and Biology, 6(1): 47-53.

Dutton P.H., Balazs G.H., Leroux R.A., Murakawa S.K.K., Zarate P. and Martínez L.S. 2008. Composition of Hawaiian green turtle foraging aggregations: mtDNA evidence for a distinct regional population. Endangered Species Research, 5: 37-44.

Dutton P.H., Jensen M.P., Frutchey K., Frey A., LaCasella E., Balazs G.H., Cruce J., Tagarino A., Farman R. and Tatarata M. 2014a. Genetic stock structure of green turtle (*Chelonia mydas*) nesting populations across the Pacific Islands. Pacific Science, 68(4): 451-464. Doi: 10.2984/68.4.1.

Dutton P.H., Jensen M.P., Frey A., LaCasella E., Balazs G.H., Zárate P., Chassin-Noria O., Sarti-Martinez L. and Velez E. 2014b. Population structure and phylogeography reveal pathways of colonization by a migratory marine reptile (*Chelonia mydas*) in the central and eastern Pacific. Ecology and Evolution, 4(22): 4317-4331.

Engbring J., Idechong N., Cook C., Wiles G. and Bauer R. 1992. Observations on the defensive and aggressive behavior of the leatherback sea turtle (*Dermochelys coriacea*) at sea. Herpetological Review 23:70-71.

Environment Australia. 2003. Recovery Plan for Marine Turtles in Australia, Commonwealth of Australia, Canberra. 164 pp.

Escalle L., Muller B., Hare S., Hamer P., Pilling G. and the PNA Office. 2020. Report on analyses of the 2016/2020 PNA FAD tracking programme WCPFC-SC16-2020/MI-IP-14. Scientific Committee Sixteenth Regular Session Oceanic Fisheries Programme, The Pacific Community (SPC). 36 pp.

Étaix-Bonnin R., Farman R., Géraux H. and Faninoz S. 2011. Conservation et suivi des populations de tortues marines en Nouvelle-Calédonie. Bull. Soc. Herp. Fr. (2011) 139-140 : 151-165

Falanruw M.V.C., Mccoy M. and Namlug. 1975. Occurrence of Ridley sea turtles in the Western Caroline Islands. Micronesica 11(1): 151-152.

Farman R. 2020. New Caledonia: A Review of Sea Turtle Distributions, Threats and Conservation Status. In: Sea Turtles in Oceania MTSG Annual Regional Report 2020 (TM Work, D Parker and GH Balazs, eds). pp. 278-231.

Fergusson I.K., Conpagno L.J.V. and Marks M.A. 2000. Predation by white sharks *Carcharodon carcharias* (Chondrichtyyes: Lamnidae) upon chelonians, with new records from the Mediterranean Sea and a first record of the ocean sunfish Mola mola (Osteichthyes: Molidae) as stomach contents. Environment Biology and Fisheries 58, 447–453.

Fertl D. and Fulling G.L. 2007. Interactions between marine mammals and turtles. Marine Turtle Newsletter 115, 4–8.

FitzSimmons N.N., Pittard S.D., McIntyre N., Jensen M.P., Guinea M., Hamann M., Kennett R., Leis B., Limpus C.J., Limpus D.J., McCann M.J., MacDonald A.J., McFarlane G., Parmenter C.J., Pendoley K., Prince R.I.T., Scheltinga L., Theissinger K., Tucker A.D., Waayers D., Whiting A. and Whiting S. 2020. Phylogeography, genetic stocks, and conservation implications for an Australian endemic marine turtle. Aquatic Conservation: Marine and Freshwater Ecosystems doi: 10.1002/aqc.3270

Fonfreyde C., Jouffroy F., Bourguet E., Weiss G., Guillaumet B., Simoni P., Colombani N., Famoetau S., Desgrippes C. and Hnaje G. 2012. Chesterfield et Bellona : Mission de suivi terrestre Janvier 2012. SMMPM, Noumea.

Fonseca L.G., Murillo G.A., Guadamúz L., Spínola R.M. and Valverde R.A. 2009. Downward but stable trend in the abundance of arribada Olive ridley sea turtles (*Lepidochelys olivacea*) at Nancite beach, Costa Rica (1971–2007). Chelonian Conservation and Biology (1): 19–27.

Forsberg K.S., Casabonne F. and Torres J.C. 2012. First Evidence of Green Turtle Nesting in Peru. Marine Turtle Newsletter 133:9-11

Fournière K., Jacob T. and Lafage D. 2015. Bilan de huit années de suivi des tortues grosses têtes (*Caretta caretta*) par l'association Bwärä Tortues Marines et perspectives (sites de la Roche Percée et de la Baie des Tortues) pp. 58)

Frey A., Dutton P.H. and Balazs G.H. 2013. Insights on the demography of cryptic nesting by green turtles (*Chelonia mydas*) in the main Hawaiian Islands from genetic relatedness analysis. Journal of Experimental Marine Biology and Ecology, 442: 80–87.

Friedlander A.M., Arribas J., Ballesteros E., Betz J., Bosserelle P., Brown E., Caselle J.E., Cramp J.E., Gataua L., Helagi N., Mayorga J., McAloney D., Myers D., Pasisi B., Richmond'Rex A., Rose P., Salinas-de-León P., San Félix M., Thompson C., Turchik A. and Sala E. 2017. Exploring the marine ecosystems of Niue and Beveridge Reef. Report to the government of Niue. National Geographic Pristine Seas. 84 pp.

Fuentes M.M.P.B. and Abbs D. 2010. Effects of projected changes in tropical cyclone frequency on sea turtles. Marine Ecology Progress Series, 412: 283–292. doi:10.3354/meps08678.

Fuentes M.M.P.B., Pike D.A., Dimatteo A. and Wallace B.P. 2013. Resilience of marine turtle regional management units to climate change. Global Change Biology (2013) 19, 1399–1406, doi: 10.1111/gcb.12138

Fukuoka T., Narazaki T. and Sato K. 2015. Summer-restricted migration of green turtles *Chelonia mydas* to a temperate habitat of the northwest Pacific Ocean. Endangered Species Research, 28: 1–10. doi:10.3354/esr00671.

Gago P.T., Valverde R., Orrego C.M., Gutiérrez L., Salazar H., Spotila J., Romero L., García W., Ríos A., Toruño C., Espinoza A., Sánchez O., Orozco J., Gutiérrez W., Mairena D., Rodríguez M. and Urteaga J. 2012. Establishment of arribada censusing methodology at olive ridley (*Lepidochelys olivacea*) Nicaraguan rookeries. In: Procs. of the Thirty-First Annual Symposium on Sea Turtle Biology and Conservation (Jones T.T. and Wallace B.P., compilers). NOAA Technical Memorandum NMFS-SEFSC-631. pp. 2004-2005.

Gaos A.R., Abreu-Grobois F.A., Alfaro-Shigueto J., Amorocho D., Arauz R., Baquero A., Briseño R., Chacón D., Dueñas C., Hasbún C., Liles M., Mariona G., Muccio C., Muñoz J.P., Nichols W.J., Peña M., Seminoff J.A., Vásquez M., Urteaga J., Wallace B., Yañez I.L. and Zárate P. 2010. Signs of hope in the eastern Pacific: international collaboration reveals encouraging status for a severely depleted population of hawksbill turtles *Eretmochelys imbricata*. Oryx, 44(4): 595-601. doi:10.1017/S0030605310000773.

Gaos A.R., Liles M.J., Gadea V., de Niz A.P., Vallejo F., Miranda C., Darquea J.J., Henriquez A., Altamirano E., Rivera A., Chavarría S., Melero D., Urteaga J., Pacheco C.M., Chácon D., LeMarie C., Alfaro-Shigueto J., Mangel J.C., Yañez I.L. and Seminoff J.A. 2017. Living on the edge: Hawksbill turtle nesting and conservation along the Eastern Pacific Rim. Latin American Journal of Aquatic Research, 45(3): 572-584.

Gaos A.R., Lewison R.L., Wallace B.P., Yañez I.L., Liles M.J., Nichols W.J., Baquero A., Hasbún C.R., Vasquez M., Urteaga J. and Seminoff J.A. 2012. Spatial ecology of critically endangered hawksbill turtles *Eretmochelys imbricata*: implications for management and conservation. Marine Ecology Progress Series, 450: 181–194. doi:10.3354/meps09591.

Gaos A.R., Lewison R.L., Liles M.J., Gadea V., Altamirano E., Henríquez A.V., Torres P., Urteaga J., Vallejo F., Baquero A., LeMarie C., Muñoz J.P., Chaves J.A., Hart C.E., de Niz A.P., Chácon D., Fonseca L., Otterstrom S., Yañez I.L., LaCasella E.L., Frey A., Jensen M.P. and Dutton P.H. 2016. Hawksbill turtle terra incognita: conservation genetics of eastern Pacific rookeries. Ecology and Evolution, 6(4): 1251-1264.

Gaspar P. and Lalire M. 2017. A model for simulating the active dispersal of juvenile sea turtles with a case study on western Pacific leatherback turtles. PLoS ONE, 12(7): e0181595.

Gilman E. (Ed.), 2009. Proceedings of the Technical Workshop on Mitigating Sea Turtle Bycatch in Coastal Net Fisheries. 20-22 January 2009, Honolulu, U.S.A. Western Pacific Regional Fishery Management Council, IUCN, Southeast Asian Fisheries Development Center, Indian Ocean – South-East Asian Marine Turtle MoU, U.S. National Marine Fisheries Service, Southeast Fisheries Science Center: Honolulu; Gland, Switzerland; Bangkok; and Pascagoula, USA.

Gilman E., Kobayashi D., Swenarton T., Brothers N., Dalzell P., and Kinan-Kelly I. 2007. Reducing sea turtle interactions in the Hawai`i-based longline swordfish fishery. Biological Conservation, 139(1): 19-28.

Giuliano C., Guinea M., Wright D. and Raith A. 2015. Nocturnal avian predation of Flatback sea turtle hatchlings, *Natator depressus*, on Bare Sand Island, NT. In: Procs 2nd Australian Sea Turtle Symposium (Whiting S.D. and Tucker A., compilers) pp 26-29.

Gless J.M., Salmon M. and Wyneken J. 2008. Behavioral responses of juvenile leatherbacks *Dermochelys coriacea* to lights used in the longline fishery. Endangered Species Research, 5: 239–247. doi:10.3354/esr00082

Graham S.C. 2009. Analysis of the foraging ecology of turtles (*Eretmochelys imbricata*) on Hawaii Island: an investigation utilizing satellite tracking and stable isotopes, master’s thesis, University of Hawaii at Hilo, Hawaii, United States.

Green D. 1978. The east Pacific green sea turtle in Galapagos. Galapagos Green Sea Turtle Ecology Study. 4 pp.

Green D. 1983. Galápagos sea turtles. Noticias de Galápagos 38: 22–25.

Green D. 1984. Long-Distance Movements of Galapagos Green Turtles. Journal of Herpetology 18(2):121-130

Groom R.A., Griffiths A.D. and Chaloupka M. 2017. Estimating long-term trends in abundance and survival for nesting flatback turtles in Kakadu National Park, Australia. Endangered Species Research 32: 203–211.

Guinea M. 2015. Assessing seasonal hatching success of Flatback (*Natator depressus*) sea turtles from hatchling track count data at Bare Sand Island, NT. In: Procs 2nd Australian Sea Turtle Symposium (SD Whiting and A Tucker, compilers) pp 33-36.

Gyuris E. 1994. The rate of predation by fish on hatchlings of the green turtle (Chelonia mydas). Coral Reefs 13,137–44.

Hahn A.T., Jensen M., Bowen B., Abreu-Grobois A., de Castilhos J.C., de Thoisy B., da Costa Gadegaste M., Limpus C., Whiting S., FitzSimmons N. and Bonatto S.L. 2012. Preliminary results on global genetic structure and phylogeography of olive ridleys (Lepidochelys olivacea). In: Proceedings of the Thirty-First Annual Symposium on Sea Turtle Biology and Conservation (Jones T.T. and Wallace B.P., compilers). NOAA Technical Memorandum NMFS-SEFSC-631. p. 207.

Hamabata T., Matsuo A., Sato M.P., Kondo S., Kameda K., Kawazu I., Fukuoka T., Sato K., Suyama Y. and Kawata M. 2020. Natal origin identification of green turtles in the north pacific by genome-wide population analysis with limited DNA samples. Frontiers in Marine Science, 7:658. doi:10.3389/fmars.2020.00658.

Hamann M., Limpus C., Hughes G., Mortimer J. and Pilcher N. 2006. Assessment of the conservation status of the leatherback turtle in the Indian Ocean and SouthEast Asia. Indian Ocean and SouthEast Asia (IOSEA) Species Assessment: Volume I. IOSEA Marine Turtle Memorandum of Understanding Secretariat, Bangkok, Thailand.

Hamann M., Schäuble C.S., Simon T., Johnson J., Evans S., Dorr T. and Kennett R. 2006. Sea turtles nesting in the Sir Edward Pellew Islands, Gulf of Carpentaria, Northern Territory. Memoirs of the Queensland Museum 52(1): 71–78.

Hamann M., Smith J. and Preston S. 2015. Flatback turtles of Torres Strait. Report to the National Environmental Research Program. Reef and Rainforest Research Centre Limited, Cairns. pp 9.

Hamann M., Wildermann N., Williams G., Daff K. and Limpus C.J. 2017. Final report on inter-nesting habitat use by flatback turtles off the Curtis Island coast − 2013 to 2015. Produced for the Ecosystem Research and Monitoring Program Advisory Panel as part of Gladstone Ports Corporation's Ecosystem Research and Monitoring Program. 27 pp.

Hamilton R.J., Bird T., Gereniu C., Pita J., Ramohia P.C., Walter R., Goerlich C. and Limpus C. 2015. Solomon Islands largest hawksbill turtle rookery shows signs of recovery after 150 years of excessive exploitation. PLos ONE, 10(4): e0121435. doi: 10.1371/journal.pone.0121435.

Hare SR, Williams PG, Ducharme-Barth ND, Hamer PA, WJ Hampton WJ, Scott R.D., Vincent M.T. and Pilling G.M. 2019. The western and central Pacific tuna fishery: 2019 overview and status of stocks. South Pacific Commission Oceanic Fisheries Programme Tuna Fisheries Assessment Report No. 20 56pp.

Harley S., Williams P.G., Nicol S. and Hampton J. 2015. The Western and Central Pacific Tuna Fishery: 2013 Overview and Status of Stocks (Tuna Fisheries Assessment Report N° 14). Noumea, New Caledonia: Secretariat of the Pacific Community, Oceanic Fisheries Programme.

Hart C.E., Ley-Quiñonez C., Maldonado-Gasca A., Zavala-Norzagaray A. and Abreu-Grobois F.A., 2014. Nesting characteristics of olive ridley turtles (*Lepidochelys olivacea*) on El Naranjo Beach, Nayarit, Mexico. Herpetological Conservation and Biology, 9(2): 524-534.

Harvey T., Townsend S., Kenyon N. and Redfern G. 2005. Monitoring of nesting sea turtles in the Coringa Herald National Nature Reserve: 1991/92 to 2003/04 nesting seasons. Indo-Pacific Sea Turtle Conservation Group, report for the Dept. of the Environment and Heritage. 67pp.

Hatase H. and Omuta K. 2020. Trophically polymorphic loggerhead sea turtles show similar interannual variability in clutch frequencies: implications for estimating population size of iteroparous animals. Journal of Zoology doi: doi:10.1111/jzo.12830

Hatase H., Kinoshita M., Bando T., Kamezaki N., Sato K., Matsuzawa Y., Goto K., Omita K., Nakashima Y., Takeshita H. and Sakamoto W. 2002. Population structure of loggerhead turtles, *Caretta caretta*, nesting in Japan: bottlenecks on the Pacific population. Marine Biology 141:299-305.

Hatase H., Omuta K. and Tsukamoto K. 2013. A mechanism that maintains alternative life histories in a loggerhead sea turtle population. Ecology 94: 2583-2594.

Havea S. and MacKay K.T. 2009. Marine turtle hunting in the Ha`apai Group, Tonga. Marine Turtle Newsletter 123: 15-17.

Hays G.C., Broderick A.C., Godley B.J., Luschi P. and Nichols W.J. 2003. Satellite telemetry suggests high levels of fishing-induced mortality in marine turtles. Marine Ecology Progress Series, 262: 305–309.

Hazel J., Lawler I.R., Marsh H. and Robson S. 2007. Vessel speed increases collision risk for the green turtle *Chelonia mydas*. Endangered Species Research, 3: 105-113.

Heithaus M.R., Wirsing A.J., Thomson J.A. and Burkholder D.A. 2008. A review of lethal and non-lethal effects of predators on adult marine turtles. Journal of Experimental Marine Biology and Ecology 356 (2008) 43–51.

Hickey F.R. and Petro G. 2005. Documentation of wan smolbag's vanua-tai resource monitor program in Vanuatu. Unpublished report. 58 pp.

Hickey F. 2020. Vanuatu: A Review of Sea Turtle Distributions, Threats and Conservation Status. In: Sea Turtles in Oceania MTSG Annual Regional Report 2020 (Work TM, Parker D and Balazs GH, eds). pp. 577-642.

Hirate K. 1988. On the nesting of the hawksbill turtle on Kuroshima Island. Marine Pavilion (Kushimoto Marine Park) 17: 32.

Hirate K. and Shimoike K. 1995. A new nesting record of the hawksbill turtle, *Eretmochelys imbricata* (Linnaeus), in Aka Island, Kerama Islands, Ryukyu Archipelago. Biological Magazine of Okinawa 33: 61-63.

Hirth H. 1997. Synopsis of the biological data on the green turtle *Chelonia mydas* (Linnaeus 1758). US Fish and Wildlife Service. Washington DC. 120 pp.

Hitipeuw C., Dutton P.H., Benson S., Thebu J. and Bakarbessy J. 2007. Population status and internesting movement of leatherback turtles, *Dermochelys coriacea*, nesting on the northwest coast of Papua, Indonesia. Chelonian Conservation and Biology, 6(1): 28–36.

Hoenner X. Whiting S.D., Enever G., Lambert K., Hindell M.A. and McMahon C.R., 2016. Nesting ecology of hawksbill turtles at a rookery of international significance in Australia's Northern Territory. Wildlife Research, 43(6): 461-473.

Hoenner X., Whiting S.D., Hamann M., Limpus C.J., Hindell M.A. and McMahon C.R. 2015. High-resolution movements of critically endangered hawksbill turtles help elucidate conservation requirements in northern Australia. Marine and Freshwater Research, 67(8): 1263-1278.

Hope R.A. 2002. Wildlife harvesting, conservation and poverty: the economics of olive ridley egg exploitation. Environmental Conservation 28(3):375-384.

Hopley D. 2008. Raine Island: its past and present status and future implications of climate change. Great Barrier Reef Marine Park Authority, Queensland. Unpublished report. 101 pp.

Howard R., Bell I. and Pike D.A. 2015. Tropical flatback turtle (*Natator depressus*) embryos are resilient to the heat of climate change. Journal of Experimental Biology, 218: 3330-3335. doi: 10.1242/jeb.118778.

Howell E.A., Kobayashi D.R., Parker D.M., Balazs G.H. and Polovina J.J. 2008. TurtleWatch: a tool to aid in the bycatch reduction of loggerhead turtles *Caretta caretta* in the Hawaii-based pelagic longline fishery. Endangered Species Research, 5: 267–278.

Huang, H.W. 2014. Seabirds and sea turtles bycatch of Taiwanese tuna longline fleets in the Pacific Ocean. Report to the Scientific Committee Tenth Regular Session. WCPFC-SC10-2014/ EB-WP-06 Rev 1, 17 pp.

Ishihara Y. 2009. Status of Japanese Coastal Sea Turtle Bycatch. In: Procs. of the Technical Workshop on Mitigating Sea Turtle Bycatch in Coastal Net Fisheries (E Gilman, ed.). 20-22 January 2009, Honolulu. WPRFMC, IUCN, SEAFDEC, IOSEA MoU, NMFS SEFSC. p. 15.

Ishizaki A. 2007. Towards reducing human-caused impacts on green turtle (*Chelonia mydas*) nesting activity in Ogasawara Islands, Japan. Thesis, Colorado State University.

Irving R. and Dawson T. 2012. The Marine Environment of the Pitcairn Islands. A report to Global Ocean Legacy, a project of the PEW Environment Group.

Iwamoto T., Ishii M., Nakashima Y., Takashita H. and Itoh A. 1985. Nesting cycles and migrations of the loggerhead sea turtle *Caretta caretta* in Maiyazaki, Japan. Japanese Journal of Ecology 35:505-512.

Jensen M.P., Abreu-Grobois A., Frydenberg J. and Loeschcke V. 2006. Microsatellites provide insight into contrasting mating patterns in arribada vs. non-arribada olive ridley sea turtle rookeries. Molecular Ecology 15:2567-2575.

Jensen M.P., Limpus C.J., Whiting S.D., Guinea M., Prince R.I.T., Dethmers K.E.M., Adnyana I.B.W., Kennett R. and FitzSimmons N.N. 2013. Defining olive ridley turtle *Lepidochelys olivacea* management units in Australia and assessing the potential impact of mortality in ghost nets. Endangered Species Research, 21(3): 241-253. doi: 10.3354/esr00521

Jensen M.P., Allen C.D., Eguchi T., Bell I.P., LaCasella E.L., Hilton W.A., Hof C.A.M. and Dutton P.H. 2018. Environmental warming and feminization of one of the largest sea turtle populations in the world. Current Biology, 28(1): 154-159. doi: 10.1016/j.cub.2017.11.057

Jensen M.P., FitzSimmons N.N., Bourjea J., Hamabata T., Reece J. and Dutton P.H. 2019. The evolutionary history and global phylogeography of the green turtle (*Chelonia mydas*). Journal of Biogeography, 46(5): 860-870. doi: 10.1111/jbi.13483

Kaiser H., Lewis M.M., Rickerl K.J., Hull M.M. and Zambada M.J. 2016. First verified observation of the Olive Ridley Sea Turtle (*Lepidochelys olivacea*) in the Republic of the Marshall Islands. Herpetology Notes 9: 311-314.

Kamezaki N. and Hirate K. 1992. Size Composition and Migratory Cases of Hawksbill Turtles, *Eretmochelys imbricata*, Inhabiting the Waters of the Yaeyama Islands, Ryukyu Archipelago. Japanese Journal of Herpetology, 14(4): 166-169.

Kamezaki N., Kikukawa A. and Ota H. 1999. Current status of the sea turtles nesting on Okinawajima and adjacent islands of the central Ryukus, Japan. Biological Conservation 87(1): 149-153.

Kamezaki N., Matsuzawa Y., Abe O., Asakawa H., Fujii T., Goto K., Hagino S., Hayami M., Ishii M., Iwamoto T., et al. 2003. Loggerhead Turtles Nesting in Japan. In: Loggerhead Sea Turtles (Bolten AB and Witherington BE, eds.) Smithsonian Institution. p. 210-217.

Kamrowski R.L., Limpus C., Maloney J. and Hamann M. 2012. Coastal light pollution and marine turtles: assessing the magnitude of the problem. Endangered Species Research 19: 85–98. doi: 10.3354/esr00462

Kamrowski R.L., Limpus C., Pendoley K. and Hamann M. 2014. Influence of industrial light pollution on the sea-finding behaviour of flatback turtle hatchlings. Wildlife Research 41: 421-434 pp.

Kawazu I., Inoguchi E., Maeda K., Fukada S. and Omata M. 2018. Long-distance movement of a tag-released hawksbill turtle from Japan to Yapen Island, Papua Province, Indonesia. Fauna Ryukyuana, 45: 1-3.

Kelez S., Velez-Zuazo X. and Manrique C. 2003. New evidence on the loggerhead sea turtle *Caretta caretta* (Linnaeus 1758) in Peru. Ecologia Aplicada 2:141-142.

Kelez S., Velez-Zuazo X. and Manrique C. 2004. Conservation of sea turtles along the coast of Peru. Partial project report. Unpublished report to UNEP Convention for Migratory Species.

Kelez S., Manrique-Bravo C. and Velez-Zuazo X. 2006. Shark longline fishey and sea turtles in Peruvian waters. In: Book of Abstracts; 26th Annual Symposium on Sea Turtle Biology and Conservation (Frick M., Panagopoulou A., Rees A.F. and Williams K., compilers). International Sea Turtle Society, Athens, p 262–263.

Kelez S., Velez-Zuazo X., Angulo F. and Manrique C. 2009. Olive ridley *Lepidochelys olivacea* nesting in Peru: the southernmost records in the eastern Pacific. Marine Turtle Newsletter 125:5-9.

Kennett R., Mununguritji N. and Yunupingu D. 1998. The Dhimurru Miyapunu Project. In 'Marine Turtle Conservation and Management in Northern Australia. In: Procs. of a workshop held at Northern Territory University, Darwin, 3-4 June 1997 (Kennett R., Webb A., Duff G., Guinea M. and Hill G., eds) Centre for Indigenous Natural and Cultural Resource Management, Northern Territory University, Darwin. pp. 69-75.

Kinch J. 2003. Sea Turtle Resources in the Milne Bay Province, Papua New Guinea: Results of a Nesting Survey (21-27/01/03) at Panayayapona and Panadaludalu Islands (Jomard Islands), with Additional Notes. Report prepared for Conservation International, Alotau, Milne Bay Province, Papua New Guinea.

Kinch J. 2006. Socio-economic assessment study for the Huon Coast. Final technical report to the Western Pacific Regional Fisheries Management Council. Honolulu, Hawaii.

Kinch J. and Burgess E. 2009. Assessment of the trade in Hawksbill Turtles in Papua New Guinea. TRAFFIC Bulletin. 22 (2): 62-72.

King C.S. 2013. An assessment of Maui hawksbill sea turtle (*Eretmochelys imbricata*) characteristics and habitat utilizations from in-water surveys and incidental observations (1998-2013). Hawksbill Recovery Project. Report. 52 pp.

King J. 2016. Flatbacks and foxes: Using cameras to capture sea turtle nest predation, Bachelor's thesis, Murdoch University, Perth, Australia.

Kisokao K. 2005. Community-based conservation and monitoring of leatherback turtles at Kamiali Wildlife Management Area performed by Kamiali Integrated Conservation Development Group. Final Report submitted to Western Pacific Regional Fishery Management Council—Contract No. 04-wpc-025. Honolulu, Hawaii.

Klain S, Eberdong J., Kitalong A., Yalap Y., Matthews E., Eledui A., Morris M., Andrew W., Albis D. and Kemesong P., 2007. Linking Micronesia and Southeast Asia: Palau sea turtle satellite tracking and flipper tag returns. Marine Turtle Newsletter, 118: 9-11.

Kobayashi D.R., Polovina J.J., Parker D.M., Kamezaki N., Cheng I.J., Uchida I., Dutton P.H. and Balazs G.H. 2008. Pelagic habitat characterization of loggerhead sea turtles, *Caretta caretta*, in the North Pacific Ocean (1997–2006): Insights from satellite tag tracking and remotely sensed data. Journal of Experimental Marine Biology and Ecology, 356 (2008): 96-114. : 10.1016/j.jembe.2007.12.019.

Kolinski S.P. 2001. Sea Turtles and their Marine Habitats at Tinian and Aguijan, with Projections on Resident Turtle Demographics in the Southern Arc of the Commonwealth of the Northern Mariana Islands. Southwest Fisheries Science Center Honolulu Laboratory, Honolulu, HI. Administrative Report H-01-06C. 88 pp.

Kolinski S.P. Parker D., Ilo L.I. and Ruak J.K. 1999. A Preliminary Assessment of the Sea Turtles and Sea Turtle Habitats of Saipan, Commonwealth of the Northern Mariana Islands. Report for the National Marine Fisheries Service, Southwest Fisheries Science Center Honolulu Laboratory, Honolulu, HI. 75 pp.

Kolinski S.P., Cruce J., Parker D.M., Balazs G.H. and Clarke R. 2014. Migrations and conservation implications of post-nesting green turtles from Gielop Island, Ulithi Atoll, Federated States of Micronesia. Micronesica, 04: 1-9.

Kolinski S.P., Cruce J., Parker D.M., Balazs G.H. and Clarke R. 2014. Migrations and Conservation Implications of Post-Nesting Green Turtles from Gielop Island, Ulithi Atoll, Federated States of Micronesia. Micronesica, 4: 1-9.

Laloë J.O., Esteban N., Berkel J. and Hays G.C. 2016. Sand temperatures for nesting sea turtles in the Caribbean: Implications for hatchling sex ratios in the face of climate change. Journal of Experimental Marine Biology and Ecology, 474: 92-99. doi: 10.1016/j.jembe.2015.09.015.

Laúd OPO Network. 2020. Enhanced, coordinated conservation efforts required to avoid extinction of critically endangered Eastern Pacific leatherback turtles. Scientific Reports 10, pp 4772. doi: 10.1038/s41598-020-60581-7

Leary T. and Laumani M. 1989. Marine turtles of Isabel Province: A report of a survey of nesting beaches (7-21 November 1989). Environment and Conservation Division and Fisheries Division MNR. 14pp.

Lebeau A. 1985. Breeding evaluation trials in the green turtle *Chelonia mydas* (Linne) on Scilly Atoll (Leeward Islands, French Polynesia) during the breeding season 1982-1983 and 1983-1984. Proceedings of the Fifth International Coral Reef Congress, Tahiti, 5:487-493.

Lei J. and Booth D.T. 2017. Who are the important predators of sea turtle nests at Wreck Rock beach? PeerJ, 5: e3515. doi: 10.7717/peerj.3515.

Leis B. 2008. 2008 Crab Island flatback sea turtle nesting study. Report prepared for the Cape York Peninsula Development Association. 17 pp.

LeRoux R.A. and Dutton P.H. 2006. Genetic stock determination of marine turtle by-catch from the California-based pelagic longline fishery and California/Oregon drift gillnet fishery. In: Book of Abstracts of the Twenty-sixth Annual Symposium on Sea Turtle Biology and Conservation (Frick M., Panagopoulou A., Rees A.F. and Williams K., compilers). International Sea Turtle Society, Athens, Greece. p. 253

Lewison R. 2013. Fisheries bycatch of marine turtles. In: Biology of Sea Turtles – Volume III (Musick J., Lohmann K. and Wyneken J., eds). CRC Press Boca Raton. pp. 329-251.

Lewison R.L. and Crowder L.B. 2007. Putting Longline Bycatch of Sea Turtles into Perspective. Conservation Biology, 21(1): 79-86. doi: 10.1111/j.1523-1739.2006.00592.x

Lewison R.L., Crowder L.B., Read A.J. and Freeman S.A. 2004. Understanding impacts of fisheries bycatch on marine megafauna. Trends in Ecology Evolution: 598–604.

Lewison R., Wallace B., Alfaro-Shigueto J., Mangel J.C., Maxwell S.M. and Hazen E.L. 2013. Fisheries Bycatch of Marine Turtles: Lessons Learned from Decades of Research and Conservation. In: The Biology of Sea Turtles, Volume III (Wyneken J., Lohmann K.J. and Musick J.A., eds.). CRC Press, Boca Raton. pp. 346-369.

Liles M.J., Jandres M.V., López W.A., Mariona G.I., Hasbún C.R. and Seminoff J.A. 2011. Hawksbill turtles *Eretmochelys imbricata* in El Salvador: nesting distribution and mortality at the largest remaining nesting aggregation in the eastern Pacific Ocean. Endangered Species Research, 14(1): 23-30. doi: 10.3354/esr00338

Limpus C.J. 1980. Observations on the hawksbill turtle *Eretmochelys imbricata* nesting along the Great Barrier Reef. Herpetologica 36, 265–271.

Limpus C.J. 2007a. A biological review of Australian marine turtles. 2. Green turtle, *Chelonia mydas* (Linnaeus). Report of the Queensland Environmental Protection Agency.

Limpus C.J. 2007b. A biological review of Australian marine turtles. 3. Hawksbill turtle, *Eretmochelys imbricata* (Linnaeus). Report of the Queensland Environmental Protection Agency.

Limpus C.J. 2007c. A biological review of Australian marine turtles. 1. Loggerhead turtle *Caretta caretta* (Linnaeus). Report of the Queensland Environmental Protection Agency.

Limpus C.J. 2007d. A biological review of Australian marine turtle species. 6. Leatherback turtle, *Dermochelys coriacea* (Vandelli). Queensland Environmental Protection Agency. 28 pp.

Limpus C.J. 2007e. A biological review of Australian marine turtles. 5. Flatback turtle, *Natator depressus* (Garman). Report of the Queensland Environmental Protection Agency.

Limpus C.J. 2007f. A biological review of Australian marine turtles. 4. Olive Ridley turtle *Lepidochelys olivacea (*Eschscholtz). Report of the Queensland Environmental Protection Agency.

Limpus C.J. and Roper P.A. 1977. A Pacific ridley turtle *Lepidochelys olivacea* from Victoria. Victorian Naturalist 94, 54–60.

Limpus C.J. and Preece N. 1992. One and All Expedition, 11–31 July 1992: Weipa to Darwin via Wellesley Group and the outer islands of Arnhem Land. Queensland Department of Environment and Heritage, Unpublished report. 41 pp.

Limpus C.J. and McLachlan N.C. 1994. The conservation status of the leatherback turtle, *Dermochelys coriacea*, in Australia. In James R. (compiler). Proceedings of the Marine Turtle Conservation Workshop ANCA, Canberra pp. 62–66.

Limpus C. and Reimer D. 1994. The loggerhead turtle, Caretta caretta, in Queensland: A population in decline. in James, R. (compiler). Proceedings of the Australian Marine Turtle Conservation Workshop. QDEH and ANCA, Canberra. Pp. 39−59.

Limpus C.J. and Limpus D.J. 2003. Loggerhead turtles in the Equatorial and Southern Pacific Ocean: A species in decline, 199 209.

Limpus C.J. and Miller J.D. 2008. Australian Hawksbill Turtle Population Dynamics Project. Report of the Queensland Environmental Protection Agency, 1-140 pp.

Limpus C. and Kamrowski R.L. 2013. Ocean-finding in marine turtles: the importance of low horizon elevation as an orientation cue. Behaviour, 150(8): 863-893. doi: 10.1163/1568539X-00003083

Limpus C.J., Parmenter C.J., Baker V. and Fleay A. 1983. The Flatback Turtle, *Chelonia depressa*, in Queensland: Post-Nesting Migration and Feeling Ground Distribution. Australian Wildlife Research 10(3) 557-561

Limpus C.J., Fleay A. and Baker V. 1984. The flatback turtle, *Chelonia depressa*, in Queensland: reproductive periodicity, philopatry and recruitment. Australian Wildlife Research 11, 579-587.

Limpus C.J., McLachlan N.C. and Miller J.D. 1984. Further observations on the breeding of Dermochelys coriacea in Queensland. Australian Wildlife Research 11, 567–571.

Limpus C.J, Zeller D., Kwan D. and MacFarlane W. 1989. Sea turtle rookeries in northwestern Torres Strait. Australian Wildlife Research 16: 517-525.

Limpus C.J., Couper P.J. and Couper L.D. 1993. Crab Island revisited: reassessment of the world's largest flatback turtle rookery after twelve years. Memoirs of the Queensland Museum, 33(1), 277-289.

Limpus C.J., Parmenter C.J., Baker V. and Fleay A. 1983. The Crab Island sea turtle rookery in the northeastern Gulf of Carpentaria. Australian Wildlife Research 10, 173–184.

Limpus C.J., Clifton D., Griffin K., Kemp L., Gallagher L., Gallagher L., Fisher S. and Parmenter C.J. 2002. Survey of marine turtle nesting distribution in Queensland, 2000 and 2001: Broad Sound to Repulse Bay, Central Queensland.

Lopez-Castro M.C. and Rocha-Olivares A. 2005. The panmixia paradigm of eastern Pacific olive ridley turtles revised: consequences for their conservation and evolutionary biology. Molecular Ecology 14(11):3325-3334.

López-Castro M.C., Carmona R. and Nichols W.J. 2004. Nesting characteristics of the olive ridley turtle (*Lepidochelys olivacea*) in Cabo Pulmo, southern Baja California. Marine Biology 145: 811–820

Lowe C.G., Wetherbee B.M., Crow G.L. and Tester A.L. 1996. Ontogenetic dietary shifts and feeding behavior of the tiger shark, *Galeocerdo cuvier*, in Hawaiian waters. Environment Biology and Fisheries 47, 203–211.

Maison K.A., Kelly I.K. and Frutchey K.P. 2010. Green turtle nesting sites and sea turtle legislation throughout Oceania. NOAA Technical Memorandum NMFS-F/SPO-110. 52pp.

Mancini A. and Koch V. 2009. Sea turtle consumption and black market trade in Baja California Sur, Mexico. Endangered Species Research, 7(1): 1-10. doi: 10.3354/esr00165

Mangel J.C., Alfaro-Shigueto J., Witt M.J., Dutton P.H., Seminoff J.A. and Godley B.J., 2011. Post-capture movements of loggerhead turtles in the southeastern Pacific Ocean assessed by satellite tracking. Marine Ecology Progress Series, 433: 261-272. doi: 10.3354/meps09152

Margaux T., Quentin G. and Cécile G. 2018. Final report on the green sea turtle egg-laying season of 2017-2018 (*Chelonia mydas*) on the atoll of Tetiaroa, French Polynesia. Te mana o te moana, French Polynesia. 54 pp.

Marquez R. 1990. FAO species catalogue. Volume 11: Sea turtles of the world. An annotated and illustrated catalogue of sea turtle species known to date. FAO Fisheries Synopsis 125, 1–81.

Márquez R., Villanueva A. and Peñaflores C. 1981. Anidacion de la tortuga laud (*Dermochelys coriacea* schlegelii) en el Pacifico Mexicano. Ciencia Pesquera, 1(1): 45-51.

Marquez R., Vasconcelos J. and Peñaflores C. 1990. XXV años de investigacion, conservacion y proteccion de la Tortuga marina. Instituto Nacional de Pesca. Secretaria de pesca, Mexico. 49 pp.

Martin S.L., Siders Z., Eguchi T., Langseth B., Yau A., Baker J., Ahrens R. and Jones T.T. 2020. Assessing the population-level impacts of North Pacific loggerhead and western Pacific leatherback turtle interactions in the Hawaii-based shallow-set longline fishery. U.S. Dept. of Commerce, NOAA Technical Memorandum NOAA-TM-NMFS-PIFSC-95, 183 pp. doi:10.25923/ydp1-f891

Martínez L.M. and Páez V.P. 2000. Nesting ecology of the olive ridley turtle (*Lepidochelys olivacea*) at La Cuevita, Chocoan Pacific coast, Colombia, in 1998. Actualidades Biologicas Medellin 22(73):131-143.

Matsuzawa Y., Sato K., Sakamoto W.and Bjorndal K.A. 2002. Seasonal fluctuations in sand temperature: effects on the incubation period and mortality of loggerhead sea turtle (*Caretta caretta*) pre-emergent hatchlings in Minabe, Japan. Marine Biology 140: 639–646

Matsuzawa Y., Kamezaki N., Ishihara T., Omuta K., Takeshita H., Goto K., Arata T., Honda H., Kameda K., Kashima Y., Kayo M., Kawazu I., Kodama J., Kumazawa Y., Kuroyanagi K., Mizobuchi K., Mizuno K., Oki K., Watanabe K.K., Yamamoto A., Yamashita Y., Yamato T., Hamabata T., Ishizaki A. and Dutton P.H. 2016. Fine-scale genetic population structure of loggerhead turtles in the Northwest Pacific, Endangered Species Research, 30: 83-93. doi: 10.3354/esr00724

Maturbongs J.A. 2000. Marine Turtles Nesting in Sorong, Irian Jaya, Indonesia. Marine Turtle Newsletter 97, 13.

McCoy M. 2004. Defining parameters for sea turtle research in the Marshall Islands. NOAA ADMIN REPORT AR-PIR-08-04.

McKeown A. 10977. Marine turtles of the Solomon Islands. Ministry of Natual Resoources, Honiara. 49 pp.

Meylan A.B. and Donnelly M. 1999. Status Justification for Listing the Hawksbill Turtle (*Eretmochelys imbricata*) as Critically Endangered on the 1996 IUCN Red List of Threatened Animals. Chelonian Conservation and Biology, 3(2):200–224

Meylan A.B., Meylan P.A. and Ordońez Espinosa C. 2013. Sea Turtles of Bocas del Toro Province and the Comarca Ngöbe-Buglé, Republic of Panama. Chelonian Conservation and Biology, 12(1): 17–33.

Miller J.D., Daly T., Card M. and Ludecke J. 1995. *Eretmochelys imbricata* Turtle Nesting in the Northern Great Barrier Reef and Torres Strait. Departmental report. Queensland Department of Environment and Heritage, Townsville, Qld.

Miller E.A., McClenachan L., Uni Y., Phocas G., Hagemann M.E. and Van Houtan K.S. 2019. The historical development of complex global trafficking networks for marine wildlife. Science Advances 5(3) eeaav5948

Moritz C., Broderick D., Dethmers K., FitzSimmons N. and Limpus C. 2002. Population genetics of Southeast Asian and Western Pacific green turtles, *Chelonia mydas*. Final report to UNEP/CM0053. 42 pp.

Moriya F., Moriya K. and Isumi C. 2012 Emergence of Hatchlings by Loggerhead Sea Turtle *Caretta caretta* at Isumi, Boso Peninsula, Central Japan, during 2007-2011. Unpublished Report. 7 pp.

Mortimer J.A. and Donnelly M. 2008. Eretmochelys imbricata. In: IUCN red list of threatened species, V. 2010.1. IUCN, Gland. Available at: www.iucnredlist.org (accessed 15 July 2019)

Mounier S. 2007. Bilan préliminaire des volets survol et mission terrain de l’opération tortue NC 2006/2007. WWF, Noumea

Muccio C. 2000. Guatemalan sea turtle conservation report. In: Kalb H.J. and Wibbels T. (compilers). Proceedings of the Nineteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-443. p. 206-207

Naro-Maciel E., Gaughran S.J., Putman N.F., Amato G., Arengo F., Dutton P.H., McFadden K.W., Vintinner E.C. and Sterling E.J. 2014. Predicting connectivity of green turtles at Palmyra Atoll, central Pacific: a focus on mtDNA and dispersal modelling. Journal of the Royal Society, 11: 20130888. doi: 10.1098/rsif.2013.0888.

National Marine Fisheries Service. 2010. Biological Opinion: Measures to reduce interactions between green sea turtles and the American Samoa-based longline fishery – Implementation of an

Amendment to the Fishery Ecosystem Plan for Pelagic Fisheries of the Western Pacific Region. National Marine Fisheries Service, Pacific Islands Region, Protected Resources Division

National Marine Fisheries Service and U.S. Fish and Wildlife Service. 1998. Recovery plan for U.S. pacific populations of the green turtle (*Chelonia mydas*). National Marine Fisheries Service, Silver Spring, Maryland. 84 pp.

National Marine Fisheries Service and U.S. Fish and Wildlife Service. 1998. Recovery Plan for U.S. Pacific Populations of the Hawksbill Turtle (*Eretmochelys imbricata*). Silver Spring: National Marine Fisheries Service, 82 pp.

National Marine Fisheries Service and U.S. Fish and Wildlife Service. 2007. Green Sea Turtle (*Chelonia mydas*). 5-Year Review: Summary and Evaluation. 105pp.

National Marine Fisheries Service and U.S. Fish and Wildlife Service. 2020. Loggerhead Sea Turtle (*Caretta caretta*) North Pacific Ocean DPS 5-Year Review: Summary and Evaluation. 84 pp.

Nichols W.J., Resendiz A., Seminoff J.A. and Resendiz B. 2000. Transpacific Migration of a Loggerhead Turtle Monitored by Satellite Telemetry. Bulletin of Marine Science, 67(3): 937-947.

Nichols W.J., Aridjis H., Hernandez A., Machovina B. and Villavicencios J. 2002. Black market sea turtle trade in the Californias. Unpublished Wildcoast report, Davenport, California.

Niethammer K.R., Balazs G.H., Nakai G.L. and McDermond D.K. 1992. Great Frigatebird (*Fregata minor*) Predation: Not a Factor in Hatchling Green Turtle (*Chelonia mydas*) Survival at French Frigate Shoals, Hawaii. Colonial Waterbirds, 15(1): 128-131.

Niethammer K.R., G.H. Balazs, J.S. Hatfield, Glynnis L., Nakai S. and Megyesi J.L. 1997. Reproductive Biology of the Green Thrtle *(Chelonia mydas)* at Tern Island, French Frigate Shoals, Hawai'i1 Pacific Science 51(1): 36-47

Nishimura W. 1994. Internesting interval and nest frequency for loggerhead turtles on Inakahama Beach, Yakushima Island, Japan. Marine Turtle Nerwsletter 67:21-22.

Nishizawa H., Narazaki T., Fukuoka T., Sato K., Hamabata T., Kinoshita M. and Arai N. 2014. Juvenile green turtles on the northern edge of their range: mtDNA evidence of long-distance westward dispersals in the northern Pacific Ocean. Endangered Species Research, 24(2): 171-179. doi: 10.3354/esr00592

Nishizawa H., Naito Y., Suganuma H., Abe O., Okuyama J., Hirate K., Tanaka S., Inoguchi E., Narushima K., Kobayashi K., Ishii H., Tanizaki S., Kobayashi M., Goto A. and Arai N. 2013. Composition of green turtle feeding aggregations along the Japanese archipelago: implications for changes in composition with current flow. Marine Biology, 160(10): 2671-2685. doi: 10.1007/s00227-013-2261-1

Okuyama J., Kitagawa T., Zenimoto K., Kimura S., Arai N., Sasai Y. and Sasakiet H. 2011. Trans-Pacific dispersal of loggerhead turtle hatchlings inferred from numerical simulation modeling. Marine Biology 158: 2055–2063. doi: 10.1007/s00227-011-1712-9

Opu J., 2018. An Assessment of Marine Turtle Exploitation in Papua New Guinea. Secretariat of the Pacific Regional Environment Programme, Apia Samoa. Final Report. 52 pp.

Ortiz-Alvarez C., Pajuelo M., Grados D., Abrego M.E., Rebeca Barragán-Rocha A., Barrantes M., Cotto Sánchez A., Fonseca L.G., Gadea Espinal V., Mangel J.C., Rodriguez-Baron J.M., Santidrian-Tomillo P., Sarti L., Santana-Hernández H., Shillinger G., Prado M., Wallace B., Williard A.S., Zavala-Norzagaray A.A. and Alfaro-Shigueto J. 2020. Rapid Assessments of Leatherback Small-Scale Fishery Bycatch in Internesting Areas in the Eastern Pacific Ocean. Front. Mar. Sci. 6:813. doi: 10.3389/fmars.2019.00813

Pacific Islands Regional Office. 2020. Federal Programs Office 2020 Annual Report. National Marine Fisheries Service, Honolulu, HI. 32 pp.

Palau Bureau of Marine Resources. 2008. Palau Marine Turtle Conservation and Monitoring Program Final Report. Unpublished grant report. 31pp.

Palma J.A.M. 1994. Marine turtle conservation in the Philippines. In: Nacu A., Trono R., Palma J., Torres D. and Agas F. (Eds.). Proceedings of the 1st ASEAN Symposium − Workshop on Marine Turtle Conservation, Manilla, Philippines, 1993. Philippines: World Wildlife Fund, pp. 105-120.

Palma J.A.M. 1997. Marine turtle conservation in the Philippines and initiatives towards a regional management and conservation program. In: Noor Y.R., Lubis I.R., Ounsted R., Troeng S. and Abdullah A. (Eds.). Proceedings of the Workshop on Marine Turtle Research and Management in Indonesia. Bogor, Indonesia: Wetlands International, PHPA/Environment Australia, pp.121-138.

Parker D.M., Balazs G.H., King C.S., Katahira L. and Gilmartin W. 2009. Short-Range Movements of Hawksbill Turtles (*Eretmochelys imbricata*) from Nesting to Foraging Areas within the Hawaiian Islands. Pacific Science, 63(3): 371-382.

Parker D.M., Balazs G.H., Frutchey K., Kabua E., Langridrik M. and Boktok K. 2015. Conservation considerations revealed by the movements of post-nesting green turtles from the Republic of the Marshall Islands. Micronesica 03: 9 pp.

Parmenter C.J. and Limpus C.J. 1995. Female recruitment, reproductive longevity and inferred hatchling survivorship for the flatback turtle (*Natator depressus*) at a major eastern Australian rookery. *Copeia* 1995: 474-477.

Patino-Martinez J., Marco A., Quiñones L. and Hawkes L.A. 2014. The potential future influence of sea level rise on leatherback turtle nests. Journal of Experimental Marine Biology and Ecology, 461: 116-123. doi: 10.1016/j.jembe.2014.07.021

Peatman T. and Nicol S. 2020. Updated longline bycatch estimates in the WCPO. Scientific Committee Sixteenth Regular Session WCPFC-SC16-2020/ST-IP-11. 37 pp.

Peatman T., Allain V., Caillot S., Park T., Williams P., Tuiloma I., Panizza A., Fukofuka S. and Smith N. 2018a. Summary of purse seine fishery bycatch at a regional scale, 2003-2017. Report to the Scientific Committee Fourteenth Regular Session, 2018. WCPFC-SC14-2018/ST-IP-04 Rev 1. 13pp.

Peatman T., Bell L., Allain V., Caillot S., Williams P., Tuiloma I., Panizza A., Tremblay-Boyer L., Fukofuka S. and Smith N. 2018b. Summary of longline fishery bycatch at a regional scale, 2003-2017. Report to the Scientific Committee Fourteenth Regular Session, 2018. WCPFC-SC14-2018/ST-WP-02.

Peckham S.H., Diaz D.M., Walli A., Ruiz G., Crowder L.B. and Nichols W.J. 2007. Small-Scale Fisheries Bycatch Jeopardizes Endangered Pacific Loggerhead Turtles. PLoS ONE 2(10): e1041. doi: 10.1371/journal.pone.0001041

Peckham S.H., Maldonado-Diaz D., Koch V., Mancini A., Gaos A., Tinker M.T. and Nichols W.J. 2008. High mortality of loggerhead turtles due to bycatch, human consumption and strandings at Baja California Sur, Mexico, 2003 to 2007. Endangered Species Research 5, 171–183.

Peckham S.H., Diaz D.M., Tremblay Y., Ochoa R., Polovina J., Balazs G.H., Dutton P.H. and Nichols W.J. 2011. Demographic implications of alternative foraging strategies in juvenile loggerhead turtles *Caretta caretta* of the North Pacific Ocean. Marine Ecology Progress Series 425: 269-280. doi: 10.3354/meps08995

Pendoley Environmental. 2014. Gladstone Region 2013/2014 Marine Turtle Hatchling Orientation Monitoring Program for Gladstone Ports Corporation.

Petit M. and Gaspar C. 2011. Double programme de recherche sur les tortues marines de l'Archipel de la Société, Polynésie française. Te mana o te moana.

Petit M., Etienne S. and Gaspar C. 2013. Influence de la température du sable sur les nids de tortues vertes (Chelonia mydas) – Atoll de Tetiaroa, Polynésie française. Te mana o te moana.

Piacenza S.E., Balazs G.H., Hargrove S.K., Richards P.M. and Heppell S.S. 2016. Trends and variability in demographic indicators of a recovering population of green sea turtles *Chelonia mydas*. Endangered Species Research 31:103-117. doi: 10.3354/esr00753

Piedra R., Vélez E., Dutton P., Possardt E. and Padilla C. 2007. Nesting of the leatherback turtle (*Dermochelys coriacea*) from 1999−2000 through 2003−2004 at Playa Langosta, Parque Nacional Las Baulas de Guanacaste, Costa Rica. Chelonian Conservation and Biology 6:111-117.

Pike D.A. 2013. Forecasting range expansion into ecological traps: climate-mediated shifts in sea turtle nesting beaches and human development. Global Change Biology (2013) doi: 10.1111/gcb.12282

Pike D.A., Roznik E.A. and Bell I. 2015 Nest inundation from sea-level rise threatens sea turtle population viability. Royal Socociety Open Science 2: 150127. doi.org/10.1098/rsos.150127

Pilcher N. 2006. The 2005−2006 Leatherback nesting season, Huon Coast, Papua New Guinea. WPRFMC Final Report. 28 pp.

Pilcher N. 2013. Community-based conservation of leatherback turtles along the Huon coast, Papua New Guinea 2012-2013. WPRFMC Final Report Project 12-turtle-003. 13 pp.

Pilcher N.J., Enderby S., Stringell T. and Bateman L. 2000. Chapter 17 Nearshore Turtle Hatchling Distribution and Predation. In: Sea turtles of the Indo-Pacific: research management and conservation. 151-166 pp. University Malaysia Sarawak. Asean Academic Press.

Pilcher N., Al-Maslamani I., Williams J., Gasang R. and Chikhi A. 2015. Population structure of marine turtles in coastal waters of Qatar. Endangered Species Research 28(2) 163-174; DOI 10.3354/esr00688.

Pingo S., Jiménez A., Alfaro-Shigueto J. and Mangel J. 2017. Incidental capture of sea turtles in the artisanal gillnet fishery in Sechura Bay, northern Peru. Latin American Journal of Aquatic Research 45(3): 606-614. doi:10.3856/vol45-issue3-fulltext-10

Piovano S., Lemons G.E., Ciriyawa A., Batibasaga A., Seminoff J.A. 2020. Diet and recruitment of green turtles in Fiji, South Pacific, inferred from in-water capture and stable isotope analysis. Marine Ecology Progress Series, 640: 201-213. doi: 10.3354/meps13287

Pita E. 1980. The turtle status in Tuvalu. Joint SPC-NMFS Workshop on Marine Turtles in the Tropical Pacific Islands. SPC- NMFS/Turtle/WP.3 Noumea, New Caledonia.

Pita J. and Rovally A. 1996. Leatherback turtle survey October – November 1995, Sasakolo and Litoghahira-Isabel Province. Ministry of Forest, Environment and Conservation, Honiara, Solomon Islands.

Pitman R.L. and Dutton P.H. 2004. Killer whale predation on a leatherback turtle in the Northeast Pacific. Pacific Science 58, 497–498.

Plotkin P. 2007. Biology and conservation of Ridley sea turtles. John Hopkins University Press, Baltimore. 356 pp.

Plotkin P.T. 2010. Nomadic behaviour of the highly migratory olive ridley sea turtle *Lepidochelys olivacea* in the eastern tropical Pacific Ocean. Endangered Species Research 13:33-40. doi: 10.3354/esr00314

Poiner I.R. and Harris A.N.M. 1994. The incidental capture and mortality of sea turtles in Australia's northern prawn fishery. In: Procs. of the Marine Turtle Conservation Workshop (R James, compiler). Australian Nature Conservation Agency: Canberra. pp. 115-123.

Poiner I.R. and Harris A.N.M. 1996. Incidental capture, direct mortality and delayed mortality of sea turtles in Australia’s northern prawn fishery. Marine Biology 125:813-825.

Polovina J.J., Kobayashi D.R., Parker D.M., Sekiand M.P. and Balazs G.H. 2000. Turtles on the edge: movement of loggerhead turtles (Caretta caretta) along oceanic fronts, spanning longline fishing grounds in the central North Pacific, 1997-1998. Fisheries Oceanography 9:71-82.

Polovina J.J., Howell E., Kobayashi D.R. and Seki M.P. 2001. The transition zone chlorophyll front, a dynamic global feature defining migration and forage habitat for marine resources. Prog Oceanography 49:469–483.

Polovina J.J., Balazs G.H., Howell E.A., Parker D.M., Seki M.P. and Dutton P.H. 2004. Forage and migration habitat of loggerhead (*Caretta caretta*) and olive ridley (*Lepidochelys olivacea*) sea turtles in the central North Pacific Ocean. Fisheries and Oceanography 13:36–51.

Prakash S., Tuiono M., Clay S., Qarau P., Philip C., Miller K., Meo S., Tamata L., Sharma-Gounder S. and Piovano S. 2020. Temporal and geographic distribution of hawksbill turtle (*Eretmochelys imbricata*) nests in Fiji, South Pacific. Testudo 9(2): 12-23.

Price E.R., Wallace B.P., Reina R.D., Spotila J.R., Paladino F.V., Piedra R. and Vélez E. 2006. Size, growth, and reproductive output of adult female leatherback turtles *Dermochelys coriacea*. Endangered Species Research 1: 41–48.

Pritchard P. 1978. Marine turtles of Papua New Guinea. Report prepared for the Wildlife Division, Port Moresby, National Capital District, Papua New Guinea.

Pritchard P.C.H. and Plotkin P.T. 1995. Olive ridley sea turtle, *Lepidochelys olivacea*. In Plotkin P.T. (ed.). National Marine Fisheries Service and U.S. Fish and Wildlife Service Status Reviews of Sea Turtles Listed under the Endangered Species Act of 1973. National Marine Fisheries Service, Silver Springs, Maryland. pp. 123-139.

Quinn N. and Kojis B.L. 1985. Leatherback turtles under threat in Morobe Province Papua New Guinea. PLES 1: 79-99.

Quinn N.B., Anguru B., Chee K., Keon O. and Muller P. 1983. Preliminary surveys of leatherback rookeries in Morobe Province, with notes on their biology. Fisheries Research Report Series No. 83. University of Technology, Lae, Papua New Guinea.

Ramirez-Cruz J.C., Ramirez I.P. and Flores D.V. 1991. Distribución y abundancia de la tortuga perica, *Caretta caretta* Linnaeus (1758), en la costa occidental de Baja California Sur, Mexico. Archelon 1:1–4

Ramirez-Gallego C. and Barrientos-Muñoz K.G. 2020. Illegal hawksbill trafficking: five years of records of the handicrafts and meat trades of the hawksbill turtle (*Eretmochelys imbricata*) in Cartagena de Indias, Colombia. Journal of Aquaculture and Marine Biology, 9(3): 101-105.

Read T.C. 2012. Les récifs d'Entrecasteaux: Synthèse des missions de suivi terrestre des tortues ma- rines 2007-2011. Service de la Marine Marchande et des Pêches Maritimes, Noumea, New Caledonia.

Read T.C. 2015. Population Structure, Migration and Habitat Ecology of the Green Turtle (*Chelonia mydas*) in the Grand Lagon Sud of New Caledonia. Griffith University Gold Coast Campus. doi: 10.13140/RG.2.1.1491.6240.

Reina R.D., Mayor P.A., Spotila J.R., Piedra R. and Paladino F.V. 2002. Nesting ecology of the leatherback turtle, *Dermochelys coriacea*, at Parque Nacional Marino Las Baulas, Costa Rica: 1988-1989 to 1999-2000. Copeia 2002:653-664.

Rice M.R., Jim L.M.R., Hickey F.R. and Balazs G.H. 2018. Post Nesting Migrations of Hawksbill Turtles (*Eretmochtlys imbricata*) Nesting at Moso Island, Republic of Vanuatu. Reports to the Fisheries Department, Government of the Republic of Vanuatu. 14pp

Robins J.B. 1995. Estimated catch and mortality of sea turtles from the east coast otter trawl fishery of Queensland, Australia. Biological Conservation 4, 157-67.

Robins C.M., Goodspeed A.M., Poiner I.R. and Harch B.D. 2002. Monitoring the catch of turtles in the northern prawn fishery. Fisheries Research and Development Corporation Final Report, Canberra, Australia.

Rodríguez-Zárate C.J., Rocha-Olivares A. and Beheregaray L.B. 2013. Genetic signature of a recent metapopulation bottleneck in the olive ridley (Lepidochelys olivacea) after intensive commercial exploitation in Mexico. Biological Conservation 168:10-18.

Rodríguez-Zárate C.J., Sandoval-Castillo J., van Sebille E., Keane R.G., Rocha-Olivares A., Urteaga J. and Beheregaray L.B. 2018. Isolation by environment in the highly mobile olive ridley turtle (*Lepidochelys* *olivacea*) in the eastern Pacific. Proceedings of the Royal Society B: Biological Sciences, 285: 20180264.

Roe J.H., Morreale S.J., Paladino F.V., Shillinger G.L., Benson S.R., Eckert S.A., Bailey H., Tomillo P.S., Bograd S.J., Eguchi T., Dutton P.H., Seminoff J.A., Block B.A. and Spotila J.R. 2014. Predicting bycatch hotspots for endangered leatherback turtles on longlines in the Pacific Ocean. Proceedings of the Royal Society, 281: 20132559. doi: 10.1098/rspb.2013.2559.

Roeger S., Munuŋgurr M. and Wise P. 2007. Entanglement of Miyapunu (Marine Turtles) in ghost netting: Northeast Arnhem land, Northern Territory, Australia: Dhimurru Turtle Entanglement Report 2006. (Dhimurru Land Management Aboriginal Corporation: Nhulunbuy, Northern Territory)

Royo N.D. 2015. Reproductive ecology and hatchling behavior of Olive ridley sea turtles in Honduras. Loma Linda University Electronic Theses, Dissertations and Projects. 340 pp.

Rudrud R.W. 2010. Forbidden sea turtles: Traditional laws pertaining to sea turtle consumption in Polynesia (Including the Polynesian Outliers). Conservation and Society 8:84-97.

Salmon M., Wyneken J., Fritz E. and Lucas M. 1992. Sea finding by hatchling sea turtles: role of brightness, silhouette and beach slope orientation cues. Behaviour, 122, 56 – 77.

Santidrián-Tomillo P., Suss J.S., Wallace B.P., Magrini K.D., Blanco G., Paladino F.V. and Spotila J.R. 2006. Influence of emergence success on the annual reproductive output of leatherback turtles. Marine Biology doi: 10.1007/s00227-009-1234-x

Santandrián-Tomillo P., Fonseca L.G., Ward M., Tankersley N., Robinson N.J., Orrego C.M., Paladino F.V. and Saba V.S., 2007. The impacts of extreme El Niño events on sea turtle nesting populations. Climate Change, 159: 163-176. Doi: 10.1007/s10584-020-02658-w.

Santidrián-Tomillo P., Vélez E., Reina R.D., Piedra R., Paladino F.V. and Spotila J.R. 2007. Reassessment of the leatherback turtle (*Dermochelys coriacea*) nesting population at Parque Nacional Marino Las Baulas, Costa Rica: effects of conservation efforts. Chelonian Conservation and Biology 6(1):54-62.

Santandrián-Tomillo P., Saba V.S., Piedra R., Paladino F.V. and Spotila J.R., 2008. Effects of illegal harvest of eggs on the population decline of leatherback turtles in Las Baulas Marine National Park, Costa Rica, 2008. Conservation Biology, 22(5): 1216-1224. Doi: 10.1111/j.1523-1739.2008.00987.x.

Santidrián-Tomillo P., Suss J.S., Wallace B.P., Magrini K.D., Blanco G., Paladino F.V. and Spotila J.R. 2009. Influence of emergence success on the annual reproductive output of leatherback turtles. Marine Biology 156:2021-2031.

Santidrián-Tomillo P., Roberts S.A., Hernández R., Spotila J.R. and Paladino F.V. 2014. Nesting ecology of East Pacific green turtles at Playa Cabuyal, Gulf of Papagayo, Costa Rica. Marine Ecology doi.org/10.1111/maec.12159

Santandrián-Tomillo P., Saba V.S., Lombard C.D., Valiulis J.M., Robinson N.J., Paladino F.V., Spotila J.R., Fernández C., Rivas M.L., Tucek J., Nel R. and Oro D. 2015. Global analysis of the effect of local climate on the hatchling output of leatherback turtles. Scientific Reports, 5: 16789. Doi: 10.1038/srep16789.

Santandrián-Tomillo P., Robinson J.N., Fonseca L.G., Quirós-Pereira W., Arauz R., Beange M., Piedra R., Vélez E., Paladino F.V., Spotila J.R. and Wallace B.P. 2017. Secondary nesting beaches for leatherback turtles on the Pacific coast of Costa Rica. Latin American Journal of Aquatic Research, 45(3): 563-571. doi: 10.3856/vol45-issue3-fulltext-6.

Sarmiento-Devia R.A., Harrod C. and Pacheco A.S. 2015. Ecology and conservation of sea turtles in Chile. Chelonian Conservation and Biology, 14(1): 21-33.

Sarti Martinez L., Barragán A.R., García Muñoz D., García N., Huerta P. and Vargas F. 2007. Conservation and biology of the leatherback turtle in the Mexican Pacific. Chelonian Conservation and Biology 6:70-78.

Sasso C.R. and Epperly S.P. 2007. Survival of pelagic juvenile loggerhead turtles in the open ocean. The Journal of Wildlife Management, 71(6): 1830-1835. Doi: 10.2193/2006-448.

Sato C.L. 2016. Periodic status review for the Leatherback Sea Turtle in Washington. Washington Department of Fish and Wildlife, Olympia, Washington. 17 pp.

Schick R.S., Roberts J.J., Eckert S.A., Halpin P.N., Bailey H., Chai F., Shi L. and Clark J.S. 2013. Pelagic movements of pacific leatherback turtles (*Dermochelys coriacea*) highlight the role of prey and ocean currents. Movement Ecology, 1: 11. doi: 10.1186/2051-3933-1-11.

Seminoff J.A. 2004. Global Status Assessment: Green turtle (*Chelonia mydas*). IUCN Marine Turtle Specialist Group Review. 71pp.

Seminoff J.A., Nichols W.J., Resendiz A. and Brooks L. 2003a. Occurrence of hawksbill turtles, *Eretmochelys imbricata* (Reptilia: Cheloniidae), near the Baja California Peninsula, Mexico. Pacific Science, 57(1): 9-16. doi: 10.1353/psc.2003.0008.

Seminoff J.A., Jones T.T., Resendiz A., Nichols W.J. and Chaloupka M.Y. 2003b. Monitoring green turtles (*Chelonia mydas*) at a coastal foraging area in Baja California, Mexico: multiple indices describe population status. Journal of the Marine Biological Association of the United Kingdom, 83: 1355-1362. doi: 10.1017/S0025315403008816.

Seminoff J.A., Zárate P., Coyne M., Foley D.G., Parker D., Lyon B.N. and Dutton P.H. 2007. Post-nesting migrations of Galápagos green turtles *Chelonia mydas* in relation to oceanographic conditions: integrating satellite telemetry with remotely sensed ocean data. Endangered Species Research, 4: 57-72. doi: 10.3354/esr00066.

Seminoff J.A., Allen C.D., Balazs G.H., Dutton P.H., Eguchi T., Haas H., Hargrove S.A., Jensen M., Klemm D.L., Lauritsen A.M., MacPherson S.L., Opay P., Possardt E.E., Pultz S., Seney E., Houtan K.S.V. and Waples R.S. 2015. Status review of the green turtle (*Chelonia mydas*) under the Endangered Species Act. NOAA Technical Memorandum NMFS-SWFSC-539. pp 571.

Seminoff J.A., Eguchi T., Carretta J., Allen C.D., Prosperi D., Rangel R., Gilpatrick Jr. J.W., Forney K., Peckham S.H. 2014. Loggerhead sea turtle abundance at a foraging hotspot in the eastern Pacific Ocean: implications for at-sea conservation. Endangered Species Research, 24: 207-220. doi: 10.3354/esr00601.

Seminoff J.A. and Glass M. 2020. Red List Assessment. Green turtle, *Chelonia mydas*, Eastern Pacific Region. IUCN Marine Turtle Specialist Group. 24 pp.

Senko J., Mancini A., Seminoff J.A. and Koch V. 2014. Bycatch and directed harvest drive high green turtle mortality at Baja California Sur, Mexico. Biological Conservation 169 (2014) 24–30.

Shilinger G.L., Swithenbank A.M., Bograd S.J., Bailey H., Castelton M.R., Wallace B.P., Spotila J.R., Paladino F.V., Piedra R. and Block B.A. 2010. Identification of high-use internesting habitats for eastern Pacific leatherback turtles: role of the environment and implications for conservation. Endangered Species Research, 10: 215-232.

Shillinger G.L., Palacios D.M., Bailey H., Bograd S.J., Swithenbank A.M., Gaspar P., Wallace B.P., Spotila J.R., Paladino F.V., Piedra R., Eckert S.A. and Block B.A. Persistent leatherback turtle migrations present opportunities for conservation. PLoS Biol 6(7): e171. doi: 10.1371/journal.pbio.0060171.

Shillinger G.L., Lorenzo E.D., Luo H., Bograd S.J., Hazen E.L., Bailey H. and Spotila J.R. 2012. On the dispersal of leatherback turtle hatchlings from Mesoamerican nesting beaches. Proceedings of the Royal Society B, 279: 1737. Doi: 10.1098/rspb.2011.2348.

Silver-Gorges I., Koval J., Rodriguez-Zarate C.J., Paladino F.V. and Jordan M. 2020. Large-scale connectivity, cryptic population structure, and relatedness in Eastern Pacific Olive ridley sea turtles (*Lepidochelys olivacea*). Ecology and evolution, 10(16): 8688-8704. doi: 10.1002/ece3.6564.

Silowsky A. 2018. The Effects of Artificial Light Intensity on Sea Turtle Nesting Behavior at Playa Cabuyal, Guanacaste, Costa Rica. MSc Thesis, Purdue University 36 pp.

Simpfendorfer C.A., Goodreid A.B. and McAuley R.B. 2001. Size, sex and geographic variation in the diet of the tiger shark, *Galeocerdo cuvier*, from Western Australian waters. Environment Biology and Fisheries 61, 37–46.

Slater J., Limpus C.J., Robins J., Pantus F. and Chaloupka M. 1998. Risk assessment of sea turtle capture in the Queensland east coast otter trawl fishery. Report prepared for TRAWLMAC, Queensland Fish Management Authority on behalf of the Great Barrier Reef Marine Park Authority and the Queensland Departments of Environment and Primary Industries.

Snover M.L., Balazs G.H., Murakawa S.K.K., Hargrove S.K., Rice M.R. and Seitz W.A. 2012. Age and growth rates of Hawaiian hawksbill turtles (*Eretmochelys imbricata*) using skeletochronology. Marine Biology doi: 10.1007/s00227-012-2058-7

Song X., Wang H., Wang W., Gu H., Chan S. and Jiang H. 2002. Satellite tracking of post-nesting movements of green turtles *Chelonia mydas* from the Gangkou Sea Turtle National Nature Reserve, China, 2001. Marine Turtle Newsletter, 97: 8.

South Pacific Commission. 1979. Joint SPC-NMFS Workshop on Marine Turtles in the Tropical Pacific Islands. SPC-NMFS/Turtles/WP.7, Noumea, New Caledonia, 11 – 14 December 1979. 16 pp.

Spotila J.R. 2004. Sea turtles: A complete guide to their biology, behavior, and conservation. Johns Hopkins University Press. Baltimore.

Spring S. 1982. Status of Marine Turtles populations in Papua New Guinea. In: Biology and Conservation of Sea Turtles (K Bjorndal, ed), Smithsonian Institution Press, Washington, DC. pp: 281-289.

Starbird C.H., Baldridge A. and Harvey J.T. 1993. Seasonal occurrence of leatherback sea turtles (*Dermochelys coriacea*) in the Monterey Bay region, with notes on other sea turtles, 1986−1991. Calif. Fish and Game 79(2): 54–62.

Seitz W.A., Kagimoto K.M., Luehrs B. and Katahira L. 2012. Twenty years of conservation and research findings of the Hawai'i Island Hawksbill Turtle Recovery Project, 1989 to 2009. Technical Report No. 178. The Hawai'i-Pacific Islands Cooperative Ecosystem Studies Unit and Pacific Cooperative Studies Unit, University of Hawai'i, Honolulu, Hawai`i. 117 pp.

Sterling E.J., Mcfadden K.W., Holmes K.E., Vintinner E.C., Arengo F. and Naro-Maciel E. 2013. Ecology and Conservation of Marine Turtles in a Central Pacific Foraging Ground. Chelonian Conservation and Biology 12(1): 2–16.

Stinson M.L. 1984. Biology of sea turtles in San Diego Bay, California, and the north eastern Pacific Ocean. M.S. thesis, San Diego State Univ., San Diego, CA. 578 pp.

Suganuma H., Horikoshi K., Tachikawa H., Sato F. and Yamaguchi M. 1996. Reproductive characteristics of the Ogasawara green turtles, in: Keinath J.A., Barnard D.E., Musick J.A. and Bell B.A. (compilers). Fifteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-387, p. 318.

Summers T.M., Kelly I.K., Work T.M., Ruak J.K. and Hapdei J.R. 2018. Human induced trauma and directed take inhibits sea turtle recovery in the Commonwealth of the Northern Mariana Islands. Micronesica, 08: 1-19.

Summers T.M., Hapdei J.J., Ruak J.R. and Guererro J. 2013. Monitoring of green (*Chelonia mydas*) and hawksbill (*Eretmochelys imbricata*) sea turtles in the Commonwealth of the Northern Mariana Islands, Western Pacific. Mariana Archipelago Ecosystem Science Implementation Planning Workshop 2013. Pacific Islands Fisheries Science Center, NOAA Fisheries.

Summers T.M., Martin S.L., Hapdei J.R., Ruak J.K. and Jones T.T. 2018. Endangered green turtles (*Chelonia mydas*) of the Northern Mariana Islands: Nesting ecology, poaching, and climate concerns. Frontiers in Marine Science, 4: 428. doi: 10.3389/fmars.2017.00428.

Sutherland R.W. and Sutherland E.G. 2003. Status of the flatback turtle (*Natator depressus*) rookery on Crab Island, Australia with notes on predation by crocodiles. Chelonian Conservation & Biology 4: 612-619.

Tagarino A., Saili K.S. and Utzurrum R. 2008, Investigations into the status of marine turtles in American Samoa, with remediation of identified threats and impediments to conservation and recovery of species. NOAA/NMFS Unallied Management Grant: NA04NMF4540126. Final Report, 44pp.

Takeshita H. 2006. The current status of loggerhead sea turtle rookeries in Miyazaki, Japan. In: Kinan I. (ed.). Western Pacific Sea Turtle Cooperative Research and Management Workshop March 2-3, 2005; Honolulu, Hawaii. pp. 27-29.

Tapilatu R.F. and Tiwari, M. 2007. Leatherback turtle, *Dermochelys coriacea*, hatching success at Jamursba-Medi and Wermon beaches in Papua, Indonesia. Chelonian Conservation and Biology, 6(1): 154-158.

Tapilatu R.F., Dutton P.H., Tiwari M., Wibbels T., Ferdinandus H.V., Iwanggin W.G. and Nugroho B.H. 2013. Long-term decline of the western Pacific leatherback, *Dermochelys coriacea*: a globally important sea turtle population. Ecosphere, 4(2): 1-15.

Tayalé A. 2007. Etude de la saison de ponte des tortues marines 2007-2008 sur l'île de tikehau. Rapport de l'association Te honu tea.

Teruya H. 1994. First record of the nesting of the hawksbill turtle on Okinawajima Island with notes on culture of the hatchlings. In: Kamezaki N., Yabuta S. and Suganuma H. (eds.), Nesting Beaches of Sea Turtles in Japan. p. 17-20. Sea Turtle Assoc. Japan., Osaka.

Thums M, Whiting SD, Reisser J, Pendoley KL, Pattiaratchi CB, Proietti M, Hetzel Y, Fisher R and Meekan MG, 2016. Artificial light on water attracts turtle hatchlings during their near shore transit. Royal Society Open Science, 3(5): 160142.

Truscott Z., Booth D.T. and Limpus C.J. 2017. The effect of on-shore light pollution on sea-turtle hatchlings commencing their off-shore swim. Wildlife Research, 44, 127–134. doi: doi.org/10.1071/WR16143

Touron M., Gaspar C., Carpentier A., Van Der Helm M., Petit M. and Stabile V. 2017. Bilan de la saison de ponte 2016-2017 de la tortue verte (*Chelonia mydas*) sur l'atoll de Tetiaroa, en Polynésie française. Te mana o te moana. 55p.

Tuoron M., Genet Q. and Gaspar C. 2019. Final report on the green sea turtle egg-laying season of 2018-2019 (Chelonia mydas) on the atoll of Tetiaroa, French Polynesia. Te mana o te moana.

Uosaki K., Okamoto H., Sato S. and Miyabe N. 2005. National Tuna Fisheries Report of Japan as of 2005. Report to the 1st Meeting of the Scientific Committee of the Western and Central Pacific Fisheries Commission WCPFC–SC1. 22 pp.

Uosaki K., Kiyofuji H., Matsunaga H., Oshima K., Suzuki N., Satoh K., Semba Y. and Akatsuka Y. 2017. National Tuna Fisheries Report of Japan. Annual Report To The Scientific Committee Thirteenth Regular Session. WCPFC-SC13-AR/CCM-10. 42 pp.

Valverde R.A. 2007. Global Assessment of Arribada Olive Ridley Sea Turtles. Final Report for the USFWS MTCA award 6-G014. 32pp.

Valverde R.A., Orrego C.M., Tordoir M.T., Goómez R.M., Solís D.S., Hernández R.A., Gómez G.B., Brenes L.S., Baltodano J.P., Fonseca L.G. and Spotila J.R. 2012. Olive ridley mass nesting ecology and egg harvest at Ostional beach, Costa Rica. Chelonian Conservation and Biology, 11(1): 1–11.

Van Houtan K.S., Kittinger J.N., Lawrence A.L., Yoshinaga C., Ray Born V. and Fox A. 2012. Hawksbill Sea Turtles in the Northwestern Hawaiian Islands. Chelonian Conservation and Biology (2012) 11 (1): 117–121. doi.org/10.2744/CCB-0984.1

Van Houtan K.S., Francke D.L., Alessi S., Jones T.T., Martin S.L., Kurpita L., King C.S. and Baird R.W. 2016. The developmental biogeography of hawksbill sea turtles in the North Pacific. Ecology & Evolution 6:2378–2389. https://doi.org/10.1002/ece3.2034

Vandersteen J., Kark S., Sorrell K. and Levin N. 2020. Quantifying the impact of light pollution on sea turtle nesting using ground-based imagery. Remote Sensing 12, 1785; doi:10.3390/rs12111785

Vargas S.M., Jensen M.P., Ho S.Y.W., Mobaraki A., Broderick D., Mortimer J.A., Whiting S.D., Miller J., Prince R.I.T., Bell I. P., Hoenner X., Limpus C.J., Santos F.R. and FitzSimmons N.N. 2016. Phylogeography, genetic diversity, and management units of hawksbill turtles in the Indo-Pacific. Journal of Heredity, 2016, 199–213 doi:10.1093/jhered/esv091

Vaughn P.W. 1981. Marine turtles: A review of their status and management in the Solomon Islands. Fisheries Division, Ministry of Natural Resources, Honiara. 70 pp.

Virgilio G., Sagun V.G., Ramoso Jr, N.B. and Mejino B.H. 2005. New Records on the distribution of loggerhead turtles (*Caretta caretta*) in the Philippines. Marine Turtle Newsletter 107: p.12.

Vuto S., Hamilton R., Brown C., Waldie P., Pita J., Peterson N., Hof C. and Limpus C. 2019. A report on turtle harvest and trade in Solomon Islands. The Nature Conservancy, Solomon Islands. 34 pp.

Walker K., Oremus M., Lindsay R., Donoghue M., Constantine R. and Stone K. 2015. Cetaceans and Marine Turtles of Vava'u. PP. 193-209 In: Atherton, J., McKenna, S., Wheatley, A. Rapid Biodiversity Assessment of the Vava'u Archipelago, Kingdom of Tonga. Secretariat of the Pacific Regional Environment Programme, Apia, Samoa.

Wallace B.P., Sotherland P., Santidrián-Tomillo P., Reina R., Spotila J. and Paladino F. 2007. Maternal investment in reproduction and its consequences in leatherback turtles. Oecologia 152:37-47.

Wallace B.P., Tiwari M. and Girondot M. 2013. *Dermochelys coriacea*. IUCN Red List of Threatened Species. Version 2013.2.

Wang J.H., Fisler S. and Swimmer Y. 2010. Developing visual deterrents to reduce sea turtle by-catch in gill net fisheries. Marine Ecology Progress Series 408: 241–250. doi: 10.3354/meps08577

Wangunu N., Kwan D., Bell I. and Pita J. 2004. Turtle Tagging and Monitoring in Milne Bay Province, December 2003. Report prepared for the South Pacific Regional Environment Program, Apia, Samoa; the Department of Environment and Conservation, Papua New Guinea; and Conservation International, Alotau, Milne Bay Province, Papua New Guinea.

Ward J. and Lemaly F. 2020. The independent State of Samoa. In: Sea Turtles in Oceania MTSG Annual Regional Report 2020 (Work T.M., Parker D. and Balazs G.H., eds). pp. 486-505

Wetherall J.A., Balazs G.H., Tokunaga R.A. and Yong Y.Y. 1993. Bycatch of marine turtles in the North Pacific high-seas driftnet fisheries and impacts on the stocks. North Pacific Commission Bulletin 53, 519–538.

Western Central Pacific Fisheries Commission. 2017. Joint Analysis of Sea Turtle Mitigation Effectiveness WCPFC-SC13-2017/EB-WP-10. Report to the Scientific Committee 13th Regular Session. Rarotonga, Cook Islands. 143 pp.

Western Pacific Regional Fishery Management Council. 2017. Annual Stock Assessment and Fishery Evaluation Report for U.S. Pacific Island Pelagic Fisheries Ecosystem Plan. Honolulu, HI 531 pp.

White M. 2012. Sea Turtles in the Cook Islands: Volume One (2009-2012).

White M. 2013. The first study of sea turtles at Rarotonga, Southern Cook Islands. Testudo, 7(5): 12-29

Whiting S. 1997. Observations of a nesting olive ridley turtle in the Northern Territory. Herpetofauna 27(2), 39–42.

Whiting S.D., Long J.L. and Coyne M. 2007. Migrating routes and foraging behaviour of olive ridley turtles *Lepidochelys olivacea* in northern Australia. Endangered Species Research, 3: 1-9. doi: 10.3354/esr003001.

Whytlaw P.A., Edwards W. and Congdon B. 2013. Marine turtle nest depredation by feral pigs (*Sus scrofa*) on the Western Cape York Peninsula, Australia: implications for management. Wildlife Research 40(5):377-384.

Wilcox C., Hardesty B.D., Sharples R., Griffin D.A., Lawson T.J. and Gunn R. 2012. Ghostnet impacts on globally threatened turtles, a spatial risk analysis for northern Australia. Conservation Letters 1: 1–8

Wilcox C., Heathcote G., Goldberg J., Gunn R., Peel D. and Hardesty B.D. 2015. Understanding the Sources and Effects of Abandoned, Lost, and Discarded Fishing Gear on Marine Turtles in Northern Australia. Conservation Biology, 1–9 doi: 10.1111/cobi.12355

Wildermann N., Critchell K., Fuentes M.M.P.B., Limpus C.J., Wolanski E. and Hamann M. 2017. Does behaviour affect the dispersal of flatback post-hatchlings in the Great Barrier Reef? Royal Society Open Science, 4(5): 170164. doi: 10.1098/rsos.170164

Williams P.G. and Terawasi P. 2014. Overview of Tuna Fisheries in the Western and Central Pacific Ocean, Including Economic Conditions- 2013.10th Meeting of the Scientific Committee of the Western and Central Pacific Fisheries Commission, Majuro, Marshall Islands, 6-14 August 2014 (WCPFC-SC10-2014, GN-WP-01).

Wilson L.. MacKay K.T., Trevor A.P. and Solomona P. 2004. Melanesian Marine Turtles Conservation Forum workshop report. Western Pacific Regional Fishery Management Council (WPRFMC); the Secretariat of the Pacific Regional Environment Programme (SPREP); the Canada-South Pacific Ocean Development (C-SPOD) Program and WWF South Pacific, Gizo, Solomon Islands.

Witherington B.E. and Martin R.E. 1996. Understanding, assessing, and resolving light-pollution Problems on sea turtle nesting beaches. Technical Reports. F.M.R. Institute, Florida Department of Environmental Protection. 73pp.

Witt M.J. Hawkes L.A., Godfrey M.H., Godley B.J. and Broderick A.C. 2010. Predicting the impacts of climate change on a globally distributed species: the case of the loggerhead turtle. The Journal of Experimental Biology, 213(6): 901-911. doi: 10.1242/jeb.038133

Witzell W.N. 1981. Predation on juvenile green sea turtles, Chelonia mydas, by a grouper, *Promicrops lanceolatus* (Pisces; Serranidae) in the Kingdom of Tonga, South Pacific. Ibid. 31:935-936.

Witzell W.N. 1982. Observations on the green sea turtle (*Chelonia mydas*) in Western Samoa. Copeia, 1982(1): 183-185. doi:10.2307/1444286.

Witzell W.N. 1987. Selective predation on large cheloniid sea turtles by tiger sharks (*Galeocerdo cuvier*). J. Herpetolology 12, 22–29.

Wood A., Booth D.T. and Limpus C.J. 2014. Sun exposure, nest temperature and loggerhead turtle hatchlings: Implications for beach shading management strategies at sea turtle rookeries. Journal of Experimental Marine Biology and Ecology 451 (2014) 105–114.

Work T.M., Parker D. and Balazs G.H. (eds.) 2020. Sea Turtles in Oceania. MTSG Annual Regional Report 2020. Draft Report of the IUCN-SSC Marine Turtle Specialist Group, 2020. 675 pp.

Work T.M., Balazs G.H., Summers T.M., Hapdei J.R. and Tagarino A.P. 2015. Causes of mortality in green turtles from Hawaii and the insular Pacific exclusive of fibropapillomatosis. Diseases of Aquatic Organisms 115:103-110. doi: 10.3354/dao02890

Zarate P.M. 2013. Biology of the Green Turtle *Chelonia mydas* in the Galápagos Islands, Doctoral dissertation, University of Florida, Florida, United States.

Zárate P., Cahoon S.S., Contato M.C.D., Dutton P.H. and Seminoff J.A. 2006. Nesting beach monitoring of green turtles in the Galapagos Islands: a 4-year evaluation. Abstracts of the 26th Sea Turtle Symposium 26: 3–8.

Zuñiga-Marroquin T. and de los Monteros A.E. 2017. Genetic characterization of the Critically Endangered hawksbill turtle (*Eretmochelys imbricata*) from the Mexican Pacific region. Latin American Journal of Aquatic Research, 45(3): 555-562. doi: 10.3856/vol45-issue3-fulltext-5

SPREP
Secretariat of the Pacific Regional Environment Programme

2021