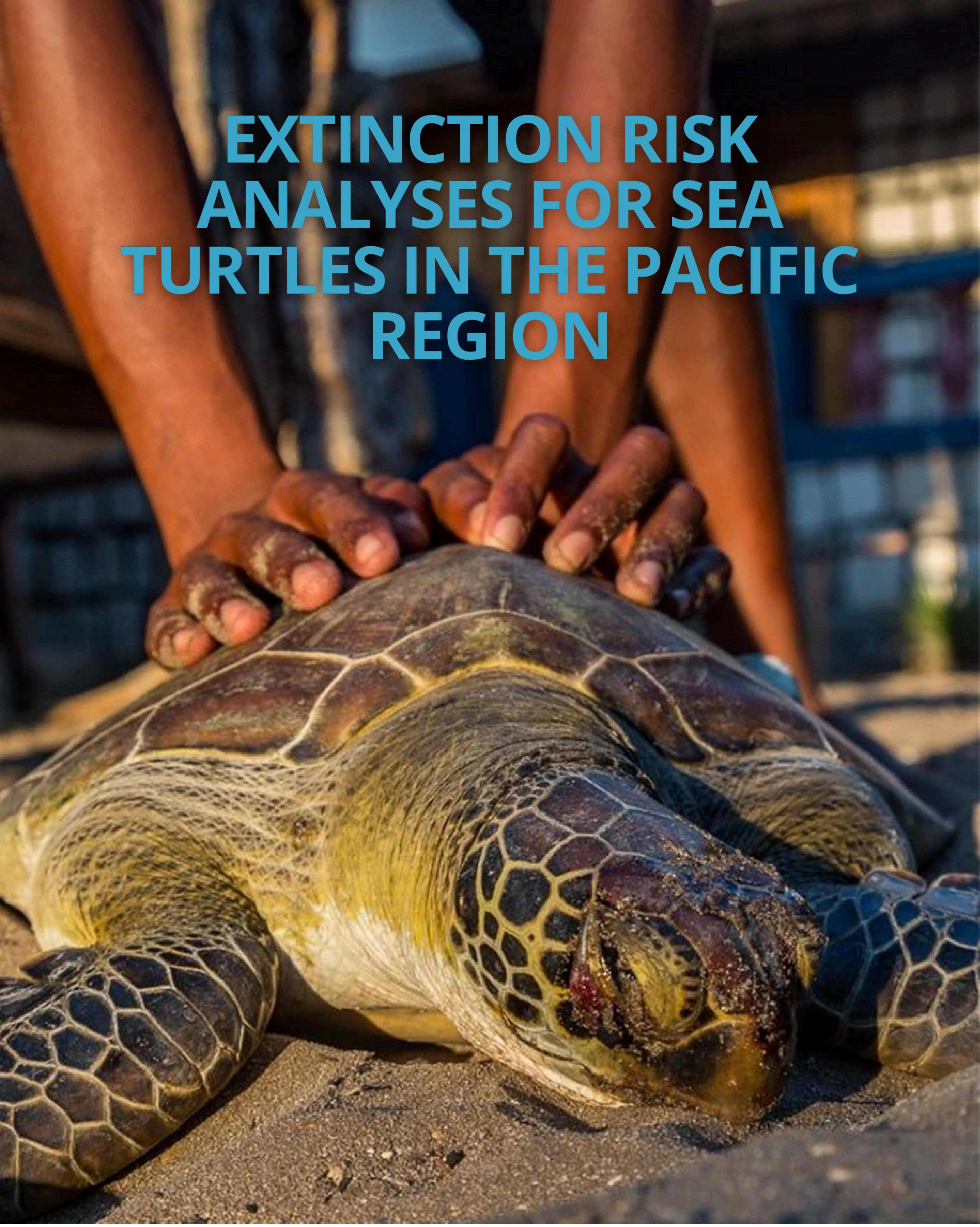
EXTINCTION RISK ANALYSES FOR SEA TURTLES IN THE PACIFIC REGION



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Extinction Risk Analyses for Sea Turtles in the Pacific Region

Nicolas J Pilcher

Turtle hatchery shaded naturally by overhanging trees

Executive Summary

This report addresses risks to extinction for sea turtle populations in the Pacific, specifically for countries and island territories within the SPREP region (herein SPREP region). The extinction risk analysis has been made based on modelled scenarios given much of the region suffers from data uncertainty in terms of biological attributes of sea turtle populations. For example, clutch frequency is unknown for most of the Pacific region, pivotal temperatures are unknown for many species and populations, and there are almost no long-term trends in numbers of turtles. A modelled approach that takes into account natural variability, climate, condition of foraging grounds, etc. is a more useful approach to determining risks of extinction than current IUCN risk categorisation given these data gaps. Sea turtles are facing a number of threats, including climate impacts, light pollution, and coastal development, but are primarily threatened by commercial and local community-based fisheries. As such, proportions of 'take' of sea turtles in both community-based, artisanal and commercial fisheries have been manipulated to simulate ongoing and potential threats to sea turtles in the region.

The extinction risk model was developed by Prof. Marc Girondot at the University of Paris-Saclay called vTurtle. The model makes a number of assumptions (see Section 1.3) and borrows biological attributes from neighbouring populations or stocks when these are not available for a specific species in this region. Mortality of turtles in the model is broken down into three categories: direct take by communities, bycatch in commercial fisheries, and take of nesting females. Other forms of mortality (for instance high early life stage mortality) can be embedded into these categories as necessary. It is envisioned that the model will be used as a predictive tool in the future to identify the most pressing threats and allow managers and policy-makers to address these as priorities.

The largest source of uncertainty comes from assumptions of the *proportion* of sea turtles taken in the various forms of mortality. While reports from empirical studies point to numbers of turtles being taken (e.g. for Fiji, Solomon Islands, Papua New Guinea, and some commercial fisheries) it is problematic to determine what proportion of all turtles in the population are represented by these numbers. A recent report from the Solomon Islands suggests some 10,000 turtles are harvested each year, of which some 7,700 might be green turtles. But out of *how many* total green turtles in the region? This uncertainty is discussed further in Section 1.1.

Following the modelling process, and taking on board what is known from IUCN Red List assessments and threats to turtles in the region, the following risk assessment ratings are proposed for turtles in the SPREP region, at varying geographical levels. For simplicity, the IUCN Red List Categories are used to describe the proposed risk of extinction, however it must be noted that these are not official IUCN assessments nor did the assessment follow the IUCN criteria. The assessments in this review were based on the vTurtle modelled outputs, given known population numbers and biological traits and change in their status under various modelled scenarios. They are used as purely indicative and for ease of understanding. Separate ratings are made for Australia for green turtles because of the substantially higher understanding of turtle population status in that country. These ratings should be considered of concern for the region, and management interventions should be considered a priority. Given estimated population sizes and trajectory, and modelled impacts, the leatherback and hawksbill sea turtles are considered most at-risk. Table EI provides a summary of the extinction risk status for turtles in the Pacific region, alongside the current IUCN listings.

Table E1: vTurtle modelled risk extinctions alongside current IUCN listings for turtles in the Pacific region.

vTurtle Modelled Risk Assessment Ranking for the SPREP Region			IUCN Assessments	
	Regional	**National**	**Global**	**Regional**
Green turtle *Chelonia mydas*	Least Concern	Australia: Least Concern Other countries and territories: Endangered	Endangered	Endangered
Hawksbill turtle *Eretmochelys imbricata*	Critically Endangered	All countries: Critically Endangered	Critically Endangered	n/a
Leatherback turtle *Dermochelys coriacea*	Critically Endangered	All countries: Critically Endangered	Critically Endangered	Critically Endangered
Loggerhead turtle *Caretta caretta*	Endangered	All countries: Endangered	Vulnerable	Critically Endangered
Olive Ridley turtle *Lepidochelys olivacea*	Critically Endangered	Australia: Critically Endangered	Vulnerable	n/a

The current small number of leatherback turtles, low survival of eggs and hatchlings due to substantive egg depredation, and bycatch of turtles (both immature and adult) have led to the species being listed as Critically Endangered. There is a need to increase egg and hatchling survival, and reduce bycatch of larger turtles in commercial fisheries, if the leatherback population is to have any chance of survival. Both an increased early-stage survival and a decrease in adult mortality via bycatch or take on nesting beaches are needed to recover this population.

Green turtle numbers in the western Pacific appear to be stable and slowly recovering. However, substantial take is occurring in many Pacific countries, and bycatch rates in commercial fisheries do not appear to be diminishing. In order for green turtles to survive in the Pacific islands the amount of take needs to be moderated and commercial bycatch needs to be addressed for the levels of turtle mortality to be sustainable.

Hawksbill turtles have been pushed to the brink of extinction in many parts of the Pacific. Ongoing trade, direct take and accidental bycatch have led to continued declines. There is an urgent need to address bycatch and accidental take, minimise direct take of hawksbills, eliminate the international trade in hawksbill products, and to protect the remaining nesting hawksbill turtles

Bycatch of loggerheads in the western Pacific appears to be the main driver behind population declines, and any reduction in bycatch in other commercial fisheries will have a significant impact on population recovery. Only increases in early-stage survivorship and removal of substantial bycatch pressures are projected to have an impact on the species recovery.

While the olive ridley turtle is rare in the Pacific region, depredation of eggs on beaches and bycatch in commercial fisheries are the two key threats. Only the removal of egg depredation and fisheries bycatch appear to be the options for recovery of this species.

There are a number of management response options that can help in halting and reversing the declines in sea turtles, which can be adopted at local, national and regional levels. Some of these are already underway, with various degrees of effectiveness. Others will require commitments from government agencies and buy-in from local communities and the fishing industry including regional fisheries bodies. Four key strategies are suggested for consideration: Firstly, addressing mortality of eggs and hatchlings on nesting beaches: increasing early life-stage survival (eggs, hatchlings, juveniles) for depleted populations is one of the more impactful ways of reversing declines and increasing population growth. This can be achieved by reducing as much as practical the collection of eggs for consumption; minimising or eliminating egg predation by feral or domestic animals (pigs, dogs, rats, monitor lizards, etc.); or as a last resort, such as when eggs are guaranteed to be lost to erosion, moving eggs to hatcheries to protect them; and implementing monitoring and enforcement programmes in areas where egg collection is illegal.

Another key management response involves addressing incidental capture of all age classes in artisanal and commercial fisheries. This is possibly the most complex aspect of sea turtle conservation. Some technological solutions already exist (e.g. Turtle Excluder Devices in trawl fisheries, LEDs in gillnets, circle hooks and fin fish bait in longline fisheries), but the problem is usually more sociological than technological. Although mandated mitigation in industrial fisheries is an important step to address bycatch, there is also a need for government will and capacity (skills, resources) to enforce the use of bycatch measures, and for local fishers and communities to want (and have the capacity and knowledge) to implement the measures with a long-term view to sustainable fishing. Addressing bycatch may be resource intensive, expensive, and takes time for results to manifest. Section 3.0 discusses ingredients for bycatch reduction in more detail.

Also key to recovering sea turtle populations is the protection of nesting females on nesting beaches. Female turtles of reproductive age are extremely valuable biologically, as they have survived several decades of threats and are in a position to lay eggs. They lay the eggs that constitute future generations. Acknowledging that traditional take is permissible in different locations under different provisions in the SPREP region, the collection of nesting adult turtles – particularly in species and populations that are extremely vulnerable – should be reconsidered amongst local communities with a view to determining suitable courses of action that preserve local traditions while also improving the conservation status of sea turtles.

Addressing unsustainable local use and consumption of sea turtles and their products is an important management response that can also reverse the decline in sea turtles and promote recoveries. Extensive collection of hawksbill sea turtles for the curio trade has occurred in the past, and persists to this day with substantial take that includes harvesting for all parts and products (e.g. meat, eggs, shell) in the Pacific, including within SPREP region. Green turtles are also taken for both consumption and trade. The social norms, cultures and traditions surrounding these harvests aside, the number of turtles that are removed from the population can have severe impacts on population trajectories, or prevent already-collapsed populations from recovering, even in the absence of other threats.

The vTurtle model simulated a number of 'take' scenarios across all five species in the SPREP region. In a status quo (do no more) scenario, several of these species are likely to become locally extinct. If management interventions are made with impact and at scale, the declining species' trajectories can be reversed.

Ultimately, the management solutions to promote the recovery of sea turtle populations will vary by country, and likely by different communities. Given the dire state of Pacific turtle populations in the SPREP region, many of which are already at reduced levels, there is a need to address these with urgency and for countries to work together to do this.

Loggerhead turtle hatchling entering the sea after emergence

Table of Contents

Tables

Figures

List of Acronyms and Abbreviations

CITES	Convention on International Trade in Endangered Species of Wild Fauna and Flora
CCL	Curved carapace length
CNMI	Commonwealth of the Northern Mariana Islands
ENSO-	El Niño Southern Oscillation
FAO	Food and Agriculture Organization of the United Nations
FFS	French Frigate Shoals
FSM	Federated States of Micronesia
GEF	Global Environment Facility
GIS	Geographical Information System
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
mtDNA	Mitochondrial DNA
MTSG	Marine turtle Specialist Group
NMFS	National Marine Fisheries Service
NOAA	National Oceanic and Atmospheric Administration
NRC	National Research Council
PEUMP	Pacific European Union Marine Partnership
PICTs	Pacific Island Countries and Territories
PNG	Papua New Guinea
RMI	Republic of the Marshall Islands
RMU	Regional Management Unit
SCL	Straight carapace length
SPC	The Pacific Community
SPREP	Secretariat of the Pacific Regional Environment Programme
SSC	Species Survival Commission
WCPFC	Western and Central Pacific Fisheries Commission
WWF	World Wide Fund for Nature

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 (Kinan & Dalzell 2005). Across much of the Pacific Ocean, sea turtles feature as key diet components in local households, augmenting protein sources from fish and other marine species. In recent decades sea turtles have been subjected to increasing pressure as customary practices have eroded, their popularity in commercial markets has increased, and/or a lack of alternatives for food security has become more apparent. Outside threats such as bycatch in artisanal and commercial fisheries have also increased (e.g. Bardach & Riding 1985, Lewison et al. 2004), and climate change threatens important nesting and feeding areas, along with sea turtle reproductive biology (e.g. Hamann et al. 2007, Hawkes et al. 2007, Witt et al. 2010, Fuentes et al. 2013, Pike 2013).

Understanding how sea turtles adapt to these pressures is challenging. On one hand they have survived on the planet for 120 million years, leading many to think they are extremely resilient and not impacted by human activities. On the other hand, turtles face pressures today that did not exist in the past: bycatch in commercial fisheries; unprecedented coastal development; a never-before-witnessed rate of increase in global temperatures; and expanding coastal communities and demands on ecosystem services for food security. A useful way to understand the magnitude of these impacts is that the human population (and its pressures) have grown nearly four-fold in the last 40 years (Clelan 2013), but sea turtle populations have not. Indeed, sea turtle populations in many areas have declined during that period. Today we are exerting greater pressures (via these threats) on populations that are less able to withstand them (due to their small size and lower reproductive output).

While a certain degree of human take has existed across the Pacific region for millennia, alongside certain levels of bycatch and trade, and some degree of coastal erosion or fluctuating temperatures have always been a feature on sea turtle nesting beaches, today the magnitude of these is likely to be outstripping local sea turtle populations' ability to maintain themselves, let alone recover.

A useful modern-day case study to exemplify this trend lies in the green sea turtles of the Ogasawara islands, lying to the east of Japan (Figure 1). Uninhabited until the 1800s, these islands hosted thousands of sea turtles each year. Following human settlement in the 1930s, annual harvests of turtles by the thousands quickly saw the population crash catastrophically. It was not until World War II, with the evacuation of people and a temporary US occupation, followed by careful management since the 1970s, that the population started to recover (Kondo et al. 2017).

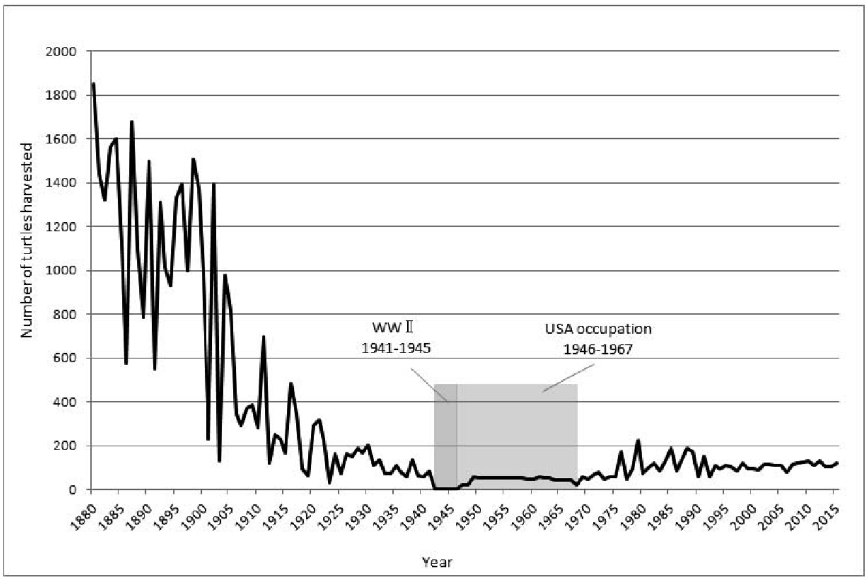


Figure 1. Number of harvested turtles from 1880 to 2015 in Ogasawara including male and female turtles. Source: Kondo et al. (2017).

Today the population is slowly recovering at a rate of about 6% per year (Chaloupka et al. 2007), with the Japan government still allowing a modest level of annual harvests. However, the population is far from recovery to pre-exploitation levels. The lesson to take from this, in short: take too many turtles and the population will likely collapse, or take only a few turtles and chances are the population will survive – so long as other external pressures are managed.

1.1 Approaches to determining extinction risk

More often than not extinction risk analyses are undertaken using the IUCN Red List numerical criteria approach. The IUCN Red List process was designed by quantitative statisticians and ecologists so that there would be a comparable and defensible way to assess risk across all taxa, from plants to turtles to insects, always using the same metrics to determine extinction risk. Other approaches to determining population risk include modelling and population viability analyses, which have been used in cetaceans and birds, and occasionally for sea turtles. However, these methods rely on a good understanding of population metrics, such as those related to reproduction, and absolute levels of threats, which are not available for many turtle species in the Pacific region.

In the past, Red List assessments used to look at all turtles of one species across their entire global range. In the last decade these assessments have been undertaken at a subpopulation level, so that they are more '*regionally*' realistic. Also, the assessments conducted for the IUCN Red List (currently) provide the only other assessment of extinction risk for sea turtles in the Pacific. While in some cases these assessments draw on outdated and incomplete datasets, there is significant overlap with the results of the current modelled risk assessment process, in spite of these limitations. The IUCN Red List 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. For sea turtles, the most common criterion on which to determine extinction risk assessments is the trend in numbers of nesting turtles over time which, as noted above, are usually lacking in the Pacific region. The main challenge in relying on the IUCN Red List for many Pacific turtle populations is the lack of long-term datasets that indicate population trajectories. As of 2025, the 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 (global)[1]
  Least Concern (North Central Pacific subpopulation)
  Near Threatened (Central West subpopulation)
- **Olive Ridley** (*Lepidochelys olivacea*): Vulnerable (global)
- **Flatback** (*Natator depressus*): Data deficient (assessment has yet to be submitted)

1.2 Challenges in quantifying Extinction Risk

Extinction risk is difficult to quantify in sea turtles. They are widespread and travel far across ocean basins making them difficult to count. Some biological traits that are needed to develop extinction risk predictions are unknown for many parts of the Pacific. In some places the numbers of turtles are growing while in others

[1] Scheduled for publication in 2025

the numbers are going down. And for many parts of the region, long-term data that would point to a trend are lacking. A key challenge is that documenting trends often requires multiple years of study, so that robust analytics can tell us about extinction risk. A good example of this is the population trend in Hawaii (Figure 2). In this example, turtle numbers fluctuate from year to year, but it has taken 25 years of monitoring for scientists to detect an upward trend with any confidence. Normal year to year fluctuations in nesting activity means that it is hard to understand population trends with short-term data sets.

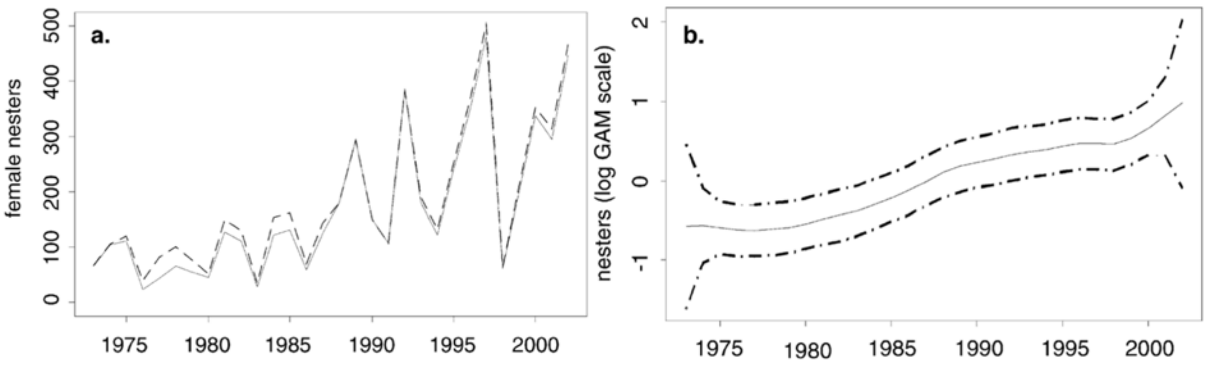


Figure 2. Trends in nester abundance. Left panel shows a time series plot of number of female green turtles nesting each year at East Island over a 30-year period from 1973 to 2002. Right panel shows the estimated long-term trend in nester abundance using a Bayesian nonparametric regression model. Source: Balazs & Chaloupka 2004.

The IUCN Red List process similarly requires multiple years of data, and in some cases requires information that we currently do not have. An excerpt of the criteria most commonly used in assessing risk of extinction in sea turtles is shown in Figure. 3.

A. Population size reduction. Population reduction (measured over the longer of 10 years or 3 generations) based on any of A1 to A4

	Critically Endangered	Endangered	Vulnerable
A1	≥ 90%	≥ 70%	≥ 50%
A2, A3 & A4	≥ 80%	≥ 50%	≥ 30%

A1 Population reduction observed, estimated, inferred, or suspected in the past where the causes of the reduction are clearly reversible AND understood AND have ceased.

A2 Population reduction observed, estimated, inferred, or suspected in the past where the causes of reduction may not have ceased OR may not be understood OR may not be reversible.

A3 Population reduction projected, inferred or suspected to be met in the future (up to a maximum of 100 years) *[(a) cannot be used for A3].*

A4 An observed, estimated, inferred, projected or suspected population reduction where the time period must include both the past and the future (up to a max. of 100 years in future), and where the causes of reduction may not have ceased OR may not be understood OR may not be reversible.

based on any of the following:

(a) direct observation *[except A3]*

(b) an index of abundance appropriate to the taxon

(c) a decline in area of occupancy (AOO), extent of occurrence (EOO) and/or habitat quality

(d) actual or potential levels of exploitation

(e) effects of introduced taxa, hybridization, pathogens, pollutants, competitors or parasites.

B. Geographic range in the form of either B1 (extent of occurrence) AND/OR B2 (area of occupancy)

	Critically Endangered	Endangered	Vulnerable
B1. Extent of occurrence (EOO)	< 100 km²	< 5,000 km²	< 20,000 km²
B2. Area of occupancy (AOO)	< 10 km²	< 500 km²	< 2,000 km²
AND at least 2 of the following 3 conditions:			
(a) Severely fragmented **OR** Number of locations	= 1	≤ 5	≤ 10

(b) Continuing decline observed, estimated, inferred or projected in any of: **(i)** extent of occurrence; **(ii)** area of occupancy; **(iii)** area, extent and/or quality of habitat; **(iv)** number of locations or subpopulations; **(v)** number of mature individuals

(c) Extreme fluctuations in any of: **(i)** extent of occurrence; **(ii)** area of occupancy; **(iii)** number of locations or subpopulations; **(iv)** number of mature individuals

Figure 3: Summary of the two most common criteria (A and B) used to evaluate if a sea turtle species belongs in a threatened category (Critically Endangered, Endangered or Vulnerable). Source: IUCN Red List Guidelines (2022).

In its most basic form, what IUCN categorisation depicted in Figure 3 means is: if a species has declined by more than 80% to 90% it qualifies as being Critically Endangered (red shaded cells). If the decline was 50% to 70% then it qualifies as Endangered (orange cells); and if it has decreased by 30% to 50% then it would be Vulnerable (yellow cells). Unfortunately, it gets a bit more complicated from there, and it is worth pointing out a few of these problematic challenges:

I. These all require some way of assessing a decline in the population measured over 10 years or three generations (dark grey cell up top);

II. Provisions such as "causes of reduction are reversible and understood and have ceased" or where these "may not have ceased" or "may not be reversible" need to be justified (see classifications A2, A3 and A4); and

III. Geographical area assessment (see criterion B) requires consideration of accompanying provisions such as 'continuing decline", "extreme fluctuations" and "fragmentation of the population".

Right away one thing should become obvious: If a turtle breeds when it is thirty years old then we would need over 100 years of data to meet the first point (a minimum of three generations of data). This is not available for any turtle species anywhere in the world. Second, there is a need to quantitatively understand threats, and more importantly, the *magnitude* of these threats. That is, we know some level of traditional take of sea turtles occurs. We also know that a certain number of turtles are taken in commercial and artisanal fisheries. What *proportion* of the sea turtle populations, however does that 'take' represent? This is largely unknown across vast extents of sea turtle ranges.

Here is an example: The modelled estimated total turtle bycatch in the Western and Central Pacific Fisheries Commission (WCPFC) longline fisheries increased from 13,500 to 42,921 individuals per year between 2006 and 2009, and then declined from 2009 to 2017 (Table I). Green turtles comprised 20% of these captures, Olive ridleys 59%, loggerheads 9%, leatherbacks 7% and hawksbill turtles 5% (Peatman et al. 2018a). The challenge lies in trying to understand what *proportion* of all green turtles in a population (or region) the bycatch take of green turtles represents. Was it 10% of all adults swimming in the Western and Central Pacific region? Or 27.3% of adults? Because this bycatch data is not readily reported by age classes and sex, we do not know what proportion of adults, sub-adults or juveniles were taken, what genetically distinct population they came from, and whether these were male or female. Our lack of knowledge on the numbers of turtles in each of these groupings leads to great uncertainty when trying to assess extinction risk.

Table I. Estimated annual bycatch and bycatch rates for WCPFC longline fleets in 1,000s of turtles. Data extracted from: Peatman et al. (2018a).

Year	Turtles ('000s)		
	Low	Med	High
2003	10.8	16.6	28.7
2004	12	17.5	25.3
2005	10.3	13.6	19.3
2006	10.3	14.3	21.5
2007	22	32.8	62.3
2008	23.7	36.1	70.1
2009	29.2	44.2	76.5
2010	19.7	29.2	53.6
2011	14.8	21.7	39.2
2012	15.7	24.2	43.1
2013	13.2	18.5	30.1
2014	15.5	21.6	32.1
2015	24.7	32.1	44.8
2016	18.3	24.1	35.4
2017	12.5	17.8	26.1

There are likely many, many sea turtles in the oceans for most species. The question is how many are being removed – or rather what *proportion* are being removed – by artisanal and commercial fisheries, and direct take for traditional use or consumption, and are these levels sustainable? As an exercise, let's agree that the average clutch size of green turtles in the Pacific region is around 105 eggs (this is the average of data sets reported from the Commonwealth of the Northern Mariana Islands (CNMI), Japan, and Papua New Guinea, which comprise the Central West Pacific green turtle population for which we have the bycatch data mentioned above). In this population (which excludes Australia) there is an average of about 3,000 annual nests (MTSG, unpublished data). That means there are some 300,000 eggs laid each year. With an average hatching success of 80% under natural conditions, that means about a quarter of a million hatchlings enter the sea each year. Of course, not all of these survive. Many become food for fish in the first few days of their lives, and many others succumb to a range of other pressures. Each year of their lives more and more are lost to predators, disease, and all kinds of other mortality – including fisheries. Let's not forget that the very next year however, there are another quarter of a million hatchlings flowing into the system. And the following year. And the next… We know that a large number of turtles enter the system each year, and that a large number are lost over the years to varying pressures. What we don't know is where the estimated thousands of mortalities from the longline industry fit into the equation.

Another challenge is that the number of turtles that nest each year changes. In some years the numbers are higher, and in others they are lower. This is linked to food availability, seasonality, and levels of external threats. Figure 2 above demonstrated how the Hawaii green turtle nesting population fluctuated up and down nearly every second year. If one were to go to the beach in, say, 1993, the nesting numbers would be high, but conversely in 1998 the numbers would be lower. For anyone to understand long-term trends in turtle abundance, there is a need for long-term monitoring.

In the case of the western central Pacific green turtles, we happen to know quite a lot about this particular population. We know how many turtles there are because of the amount of monitoring that has been undertaken, and in some places this monitoring has continued long enough for the datasets to be meaningful and able to be statistically assessed. There are long-term datasets for Ogasawara, and for CNMI, the Federated States of Micronesia (FMS), and Palau. We also have an idea of how many nests are laid each year, and how many eggs these contain, and what the average emergence rate in hatchlings is.

Unfortunately, however, for much of the rest of the Pacific region, these data are lacking for greens and also for other species. This is because it is really difficult to collect long-term datasets in remote locations. It is hard and expensive to get to remote islands; it is hard and expensive for communities to do this at the expense of making enough money to live and eat; it is hard and expensive to keep research teams there long enough to collect meaningful biological data. And it is hard and expensive to do this year after year for multiple years.

Similarly, we have a low level of information on threats and numbers of turtles (proportions of turtles actually) that are lost to the population each year. Note how the IUCN Red List in Figure 3 requires knowledge of 50% or 70% or 90% declines. These require an understanding of what *proportion* of a population is impacted. While some information exists with respect to the bycatch of sea turtles in the Pacific from industrial fisheries such as the tuna purse seine and long line sectors (such as the data presented in Table I), less is known about levels of use of sea turtles by coastal communities, and impacts of small-scale fisheries across the Pacific. Today there is a growing understanding of turtle use, with country-wide surveys being undertaken in some countries (Fiji – Batibasaga et al. unpublished data; Tonga – Stone et al. unpublished data; Papua New Guinea - Opu 2018, Haskin et al. unpublished data; Solomon Islands - Vuto et al. 2019, Hamilton et al. 2024). Similarly, little is known about 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 - although again, recent work is also making inroads into this subject (e.g. Staines et al., unpublished data; Staines et al. 2023).

The solution, when critical data points are missing, and long-term datasets are unavailable, is to use mathematical models to predict what could be happening with a population. Mathematical models can calculate, very quickly, a range of scenarios and average these for a hypothetical population, providing information on what that population is likely doing. This is what we have done for the SPREP region, particularly for those Pacific populations for which data are lacking, and the model outcomes inform us of the risk of extinction to sea turtles in the Pacific and SPREP region.

A simple example of how modelling can help fill gaps in research is shown in Figure 4. In this hypothetical example monitoring occurs at all sites during Year 1 (orange), but in Year 2 (green), two sites are not visited for some reason. With a model, the proportion of the total nesting occurring at all of the monitored sites can then be compared to what is known from the missing sites, and the amount of nesting that *might* have taken place at these sites can be modelled (blue) with varying levels of precision (grey bracket bars).

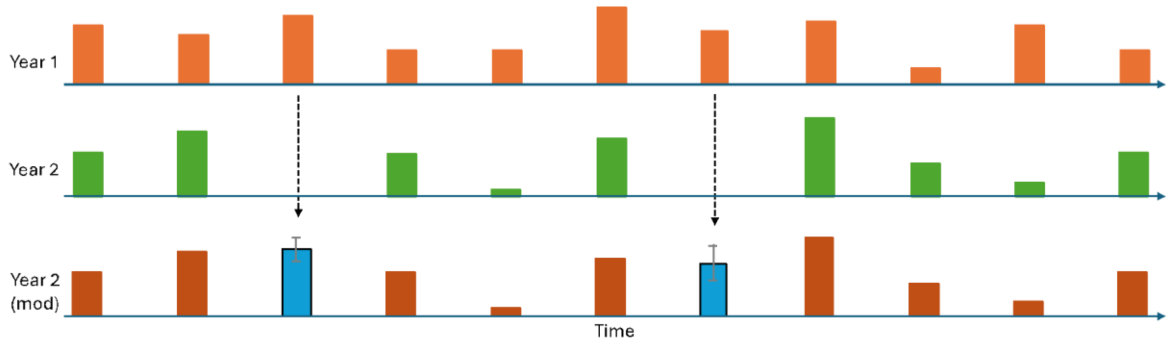


Figure 4: Example of how modelling can fill gaps in knowledge in sea turtle populations.

1.3 Risk Assessment: methodology, assumptions and datasets

1.3.1 Methodology

Given the absence of long-term nesting trend data for most Pacific sea turtle populations and the limited data on reproductive aspects of their biology in the region, this risk assessment process uses a modelling approach to project long-term population size trends, from which risk assessments can be derived. In collaboration with Prof. Marc Girondot at the University of Paris-Saclay, we developed a model called vTurtle to 'simulate' what the trend in these populations might look like in coming years. The model can take advantage of knowledge gaps, and use data from nearby and similar populations to make predictions of what might happen to a sea turtle population. We then used this model to simulate the impact of the two main threats to sea turtles in this region - the take of turtles in community harvests (or small-scale turtle fisheries) and commercial fisheries by-catch.

1.3.2 Assumptions

The model is based on a number of assumptions which are quite straightforward:

1. We assume that biological processes follow a normal gaussian (bell curve) distribution. That is, if we were considering habitat quality as an example, in some years turtle feeding habitat quality might be exceptional, in some years turtle feeding habitat quality might be really poor, and in most years it will be somewhere in between (Figure 5). The model is set to run multiple times (300,000 times) and ends up running scenarios where 'really bad quality' is assumed in 2.5% of cases (the bottom left end of the curve in Figure 5), and similarly extremely high quality is only assumed 2.5% of the time (the bottom right end of the curve in Figure 5), with the balance 95% of time somewhere in between.

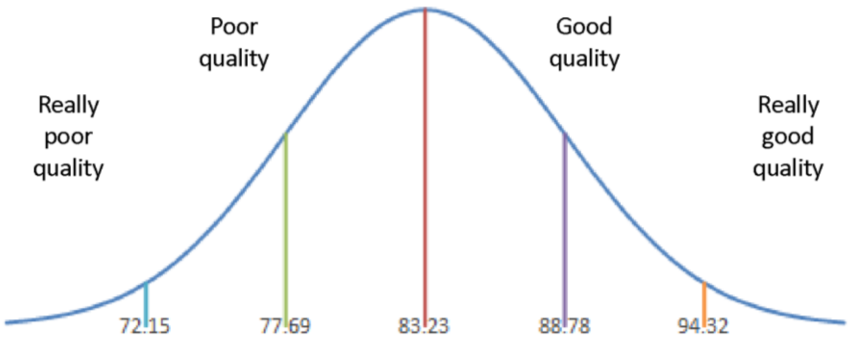


Figure 5: Hypothetical Gaussian distribution of habitat quality.

2. Where we do not know the clutch size of a particular population, we use values from the nearest known population of the same species. This works well because we already know that physiologically clutch sizes are about the same, and biologically they are mostly similar, so they likely can develop a similar number of eggs. Egg development is linked to habitat quality (the same as food availability) and this is taken into consideration by the model;
3. Similarly, where we do not know the emergence success, we can use a known value from a neighbouring area (until such a time as we have better data from each site);
4. Where we do not know the pivotal temperature (the pivotal point in which a male or female embryo is determined) for a particular species in the Pacific, we use the nearest known pivotal temperature as a proxy;
5. Survivorship at varying age classes is different, whereby small turtles have higher mortality (lower survivorship), due to their greater vulnerability to predators; and nesting turtles have higher vulnerability than non-nesting adult turtles due to increased risks on beaches and in nearshore waters;
6. Growth is a function that is informed by growth data and where not available utilises the nearest known dataset, adjusted for average adult sizes.

1.3.3 Data used in the risk assessments

The primary datasets and biological attributes used in the model were extracted from the Status Review for Sea Turtles in the Pacific region conducted by Pilcher (2021). These datasets include estimates of current adult female abundance (estimates were available for all Pacific Island nations and territories), clutch frequencies (available for CNMI, Japan, Australia, and New Caledonia); clutch size (available for Cook Islands, French Polynesia, CNMI, Japan, PNG, Australia, New Caledonia, and the Solomon Islands); hatching success (available for French Polynesia, CNMI, Australia and New Caledonia).

The assessments also make use of published bycatch estimates for the Western and Central Pacific Fisheries Commission region (Peatman et al. 2018a,b, 2020, 2023), New Zealand (Dunn et al. 2023), and harvest rate reports for several Pacific countries (Fiji – Batibasaga et al. unpublished data; Tonga – Stone et al. unpublished data; Papua New Guinea - Opu 2018, Haskin et al. unpublished data; Solomon Islands - Vuto et al. 2019, Hamilton et al. 2024). In addition, the risk assessment draws on the published scientific literature for data on turtle size in bycatch events, turtle growth rates, and population recovery toolkits (that comprise specialised actions for different sections of the species' population), such as that for the leatherback turtle (Jones et al. 2012). The assessment also considers more recent reviews of turtle status and conservation challenges by the IUCN Marine Turtle Specialist Group (Wallace et al. 2023); alongside past existing IUCN risk assessments for all species.

1.4 vTurtle

The vTurtle risk assessment process models the natural aspects of sea turtle biology and links these to existing threats such as bycatch and take by local communities (direct take). Other impacts such as from climate change, and anthropogenic lighting are expected to further impact sea turtle stocks but these remain unquantified at present. However, it is safe to say that across many parts of the Pacific region issues such as storm frequency, beach inundation and rising sand temperatures will be substantial management issues in coming years. It is envisioned that the model will be used as a predictive tool in the future to identify the most pressing threats and allow managers and policy-makers to address these as priorities. It is envisioned that in the near future the model will be published in the peer-reviewed literature, and will be available to modellers with an understanding of the R programming language (https://www.r-project.org/) coupled with a deep understanding of turtle ecology.

In its simplest form, the vTurtle risk assessment model looks at two sides of an equation: the number 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 depends on factors such as levels of direct take for traditional use and consumption, and numbers of turtles caught in offshore and inshore fisheries (Figure 6).

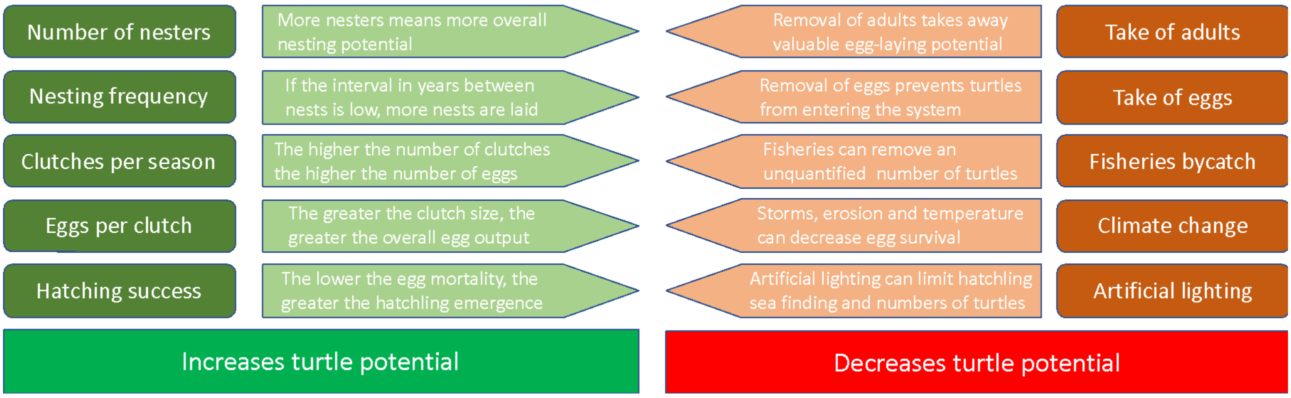


Figure 6: An imaginary balance scale investigated by vTurtle to determine extinction risk in sea turtles of the Pacific.

vTurtle was assembled from multiple published and peer-reviewed models. These include models that describe (among others) the phenology (timing) of marine turtle nesting seasons (Girondot et al. 2006), model the thermal environment of turtle nests (Girondot et al. 2024); the size and age relationships in turtle growth (Girondot et al. 2021); embryonic growth rates (Girondot & Kaska, 2014); clutch frequencies (Briane et al. 2007); turtle growth (Chevallier et al. 2020); and sea turtle age (Morales-Merida et al. 2024).

1.5 Challenges in determining proportional turtle numbers - direct take or bycatch

As noted above, there is some information on the numbers of turtles caught as bycatch in some fisheries such as estimates of turtle bycatch for some commercial fisheries (e.g. Western Pacific Fisheries Commission (WCPFC), New Zealand), and some estimates of direct turtle take (and use) in some countries (e.g. PNG, Solomon Islands, Tonga and Fiji). What we do not know however, is the *proportion* of turtles out of the total population that are being taken by coastal communities or captured in commercial fisheries.

Bycatch and direct take data for leatherbacks exemplifies the challenges in assigning an annual proportional mortality rate to sea turtles. The most recent estimate of numbers of adult female leatherbacks in the western Pacific ranged from 2,700 to 4,500 turtles in 2013 (Dutton 2007, Wallace et al. 2013), and this number has been declining at an average annual rate of 5.9% (Tapilatu et al. 2013). This rate of decline would suggest that the number of adult females in the western Pacific in 2025 could range from 1,594 to 2,728 (x*(1-0.059)^12). Of course, adult females do not comprise the only turtles in any given population, with adult males and non-breeding turtles also needing consideration. Many population-advantageous primary sex ratios are typically represented by ~70% females and ~30% males (Santindrian-Tomillo 2022), suggesting that in addition to adult females, there may also be an additional 683 to 1,169 adult male leatherback turtles in the Western Pacific population (Table II).

Average age to maturity for leatherbacks is ~16 years, meaning there are also an additional 15 annual cohorts of younger turtles in the Western Pacific population. Each year a new batch of hatchlings emerges and joins the population, but not all of these survive. Annual survivorship rates for first-year leatherbacks have been estimated at 0.250 (meaning 25% of turtles survive to the next year), while for juveniles this is 0.810, and in the region of 0.800 for adult leatherback turtles (Jones et al. 2012). Working backwards from the 2025 estimated number of adults and assuming an equal probability of mortality among males and females, there could be an additional 196,393 to 336,110 smaller and non-breeding leatherbacks in the western Pacific (see Table II).

Table II: Possible leatherback turtle numbers in the western Pacific over multiple cohorts. Note: These numbers do not account for levels of egg harvest and reduction of hatchlings likely entering the system but they are useful in understanding the concept of total numbers of turtles.

	Lower estimate	Upper estimate
# Females 2013*	2700	4500
Decline‡	5.90%	
Possible # females 2025+	1,594	2,728
Possible male contribution	683	1,169
Males & females in 2025	2,277	3,897
Year 15	2,710	4,638
Year 14	3,225	5,519
Year 13	3,837	6,567
Year 12	4,566	7,815
Year 11	5,434	9,300
Year 10	6,467	11,067
Year 9	7,695	13,170
Year 8	9,157	15,672
Year 7	10,897	18,650
Year 6	12,968	22,193
Year 5	15,431	26,410
Year 4	18,363	31,428
Year 3	21,852	37,399
Year 2	26,004	44,504
Year 1	45,508	77,883
Total for 16 years	196,393	336,110
Annual mortality rate Year 10	1,229	2,103
Dutton et al. 2007, Wallace et al. 2013 ‡Tapilatu et al. 2013 +x(1-0.059)^12		

Size data for bycaught leatherbacks in the western Pacific are not usually reported, but the size ranges that have been reported further complicate assigning a *proportion* of bycatch from an entire population. Off the coast of Peru, 75% of leatherback turtles caught in the artisanal driftnet fishery were smaller than 120cm and considered juveniles (Mangel et al. 2024). de Paz et al. (2006) also report immature turtles being caught off Peru, with CCL ranging from 88-132cm. For New Zealand, Dunn et al. (2023) report that 'no data were available on size or sex of [271] captured leatherbacks'. In a study off the eastern US coast and the Gulf of Mexico, bycaught leatherback turtles had an average curved carapace length ± SD (CCL) of 146.6 ± 5.6 cm (n = 18; Stewart et al. 2016). The authors note in their study that these measurements may not be representative of all leatherback bycatch, as the measurements were biased toward smaller turtles, because larger turtles were more difficult to bring on board. A second study in the NW Atlantic indicated that only 20% of leatherback bycaught turtles were <140cm CCL (Dodge et al. 2022). This suggests that the smaller age classes of sea turtles are not taken as frequently in commercial fisheries as the larger size classes.

The next challenge lies in identifying how old those larger age classes might be. Leatherback turtles grow rapidly in contrast to other sea turtle species, with juvenile growth rates reported as 8.6–39.4 cm per year (Zug & Parham 1996), 18.3–50.2 cm per year (Avens et al. 2020), 18.3–50.2 cm per year (Wallace & Jones 2015) and 12.5–30.2 cm per year (Avens et al. 2020). This suggests, based on the von Bertalanffy growth curve proposed by Zug & Parham (1996), that the larger turtles more frequently encountered as bycatch in longline fisheries are likely 10 years of age and older.

If we were to assume this is the case for Western Pacific leatherbacks and consider only the larger size classes in the older 10 to 16–year age classes of sea turtles, this would suggest there could be some 28,516–48,803 larger size turtles 'available for capture' in the western Pacific (Table II, sum of years 10 to 16), that sustain losses of between 1,229 and 2,103 turtles per year due to natural mortality. If we were to consider younger immature age classes, there could be some 69,233–118,487 turtles 'available for capture' in the western Pacific (Table II, sum of years 6 to 16). If juvenile turtles were additionally considered, an even greater number of turtles would be available for capture.

Peatman & Nicol (2023) provide modelled estimates of leatherback bycatch in the WCPFC longline fisheries of between 11,076 and 34,811 turtles between 2003 and 2021. Recent work from New Zealand indicates annual bycatch rates averaging 15.5 turtles per year in surface longline fisheries, with reported captures increasing substantially to 50 in 2020–21 (Dunn et al. 2023). It is clear from the data presented by Peatman & Nicol (2023) modelled estimates of bycatch of leatherbacks are problematic: the upper estimate, if taken at face value, would have virtually extirpated leatherbacks from the western Pacific. The lower value would have removed nearly half of all adult-sized age classes, at a minimum. In addition, this report only addresses longline bycatch in the WCPFC region, and does not include other fisheries or areas outside of the WCPFC region. The WCPFC report on effectiveness of mitigation efforts suggest annual takes of 94 to 98 leatherback turtles per year (WCPFC-SC13-2017 EB-WP-10), two orders of magnitude smaller than the Peatman & Nicol (2023) estimates, further complicating predictions of total *proportional* loss.

As can be seen from the discrepancies in these bycatch estimates, interpreting proportional bycatch data is problematic. As Peatman & Nicol (2023) point out, there are difficulties in determining robust estimates of bycatch from observer data, particularly for rarely caught species such as leatherbacks, because of the low levels and imbalanced nature of observer coverage. They go on to suggest that the trends in estimated catch rates are more reliable than the magnitudes of the estimated catches. Given the uncertainty in absolute losses to the population, in the sizes and thus age classes impacted by the fisheries, and in the modelled estimates, it is challenging to determine what proportion of leatherbacks are being taken in a given year out of all leatherback turtles 'available for capture'.

Similarly, recent turtle use surveys in Fiji, Tonga and PNG suggest a combined total of around 51,404 sea turtles taken per year among communities living within 500m of the coast (data extracted from modelled median estimates in Batibasaga et al. unpublished data; Stone et al. unpublished data; Haskin et al. unpublished data). Given the number of turtles 'available for capture', these estimates seem unrealistic and thus challenging when assigning proportional levels of human take.

Using vTurtles, we therefore model different bycatch and take scenarios for each sea turtle species in the SPREP region. While Regional Management Units (RMUs) provide the latest division of turtle management units that are now used for the IUCN Red List processes (Wallace et al. 2023), in many cases there are insufficient data among the western Pacific RMUs to differentiate among these – and in these cases the extinction risks are presented simply for populations for each species of sea turtle overall in the SPREP region. These overall assessments make use of datasets from various neighbouring populations (e.g. where reproductive data are absent, or for which bycatch and direct take are reported) to develop biologically-appropriate models of population trends.

Leatherback turtle nesting in Kamiali, Papua New Guinea

2.0 Extinction Risk Analyses – Pacific Sea Turtles in the SPREP region

In the following sections, we present first the parameters that went into each model run, and then a modelled population trajectory, followed by a description of the model outcomes. We do not list *all* of the individual runs (in some cases we ran up to 40 per species) but focus on key examples from which to understand modelled population trajectories. In each case a variety of scenarios were tested that involved increasing and decreasing levels of accidental capture (bycatch) and targeted (direct) take of turtles, with an aim to determine the general conditions under which turtle populations would survive. These levels of take were based on estimates of turtles taken in fisheries and for human use, compared to estimated turtle population sizes.

The first model we present for each species is the one we believe best fits the current situation. Following this we present different models in which we alter the levels of take and bycatch to demonstrate the impacts the changes have on population trends. For clarity, the Figure 7 explains the data inputs and graphic outputs of the vTurtle model, which are then used consistently throughout the individual species risk analyses sections.

The *Initial population* (top left, Figure 7) comprises all age classes of male and female turtles and is modelled by vTurtle after running a 'burn in' period to stabilise the population. The 'burn in' process is a computer model simulation whereby the current known number of adult females in that population, along with all the biological traits that are known for the species in the region, but without any threats, are allowed to stabilise over 100,000 runs, based on the 'bell curve' distribution (whereby the simulation 'evens out' after a number of years). The model is then run for a 30-year period as this is a practical management lifespan.

In the future, the model can be run again and refined as and how additional data become available. The 'burn in' process is required to account for variability in habitat quality, environmental conditions, and turtle reproductive potential at any point in time. Natural survivorship (the proportion of turtles in a population that survive to the next year) and other biological traits, are based on best known scientific data for each turtle species (e.g. Pilcher & Chaloupka 2013, Jones et al. 2015 for leatherback turtles) and are fixed across all model runs for that species. The variables that are tested in the different model runs are 'take': direct human take for use and commercial take as bycatch, and these are expressed as proportions of turtles from that population that might be taken. For example, in the first scenario for leatherback turtles in Section 2.1, the commercial take of adults >140cm in carapace (shell) length was set at 0.001, which would be one in one thousand of the turtles available in that population after the model stabilised (in this case, 4,846, or roughly five adult turtles per year). This is proportional to the current reported rate of bycatch from WCPFC (Peatman et al. 2018a,b, 2023) and therefore is aligned with current known bycatch estimates.

Green turtle underwater

Parameter	Setting	Survivorship	SCL (cm)	s
Initial population size	14,807	Natural	0	0.250
Number of years for stabilization	30		85	0.800
Number of replicates	100		140	0.835
Average number of eggs per clutch	76.4	Human take	0	0.001
Standard deviation of number of eggs per clutch	16.1		85	0.001
Pivotal temperature model	WPac		140	0.001
Habitat mean	0.6	Commercial take	0	0.001
Habitat quality minimum limit	0.05		85	0.001
Habitat quality maximum limit	0.95		140	0.010

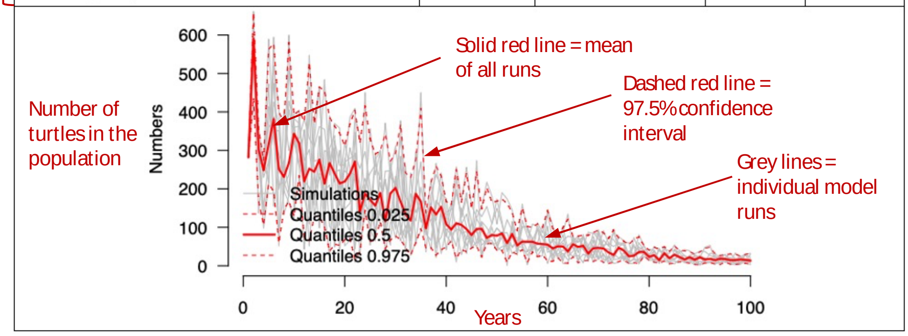


Figure 7: A model run showing data inputs and graphical output for the population trend, given varying levels of natural and fisheries mortality.

Juvenile green turtles in an experimental growout project

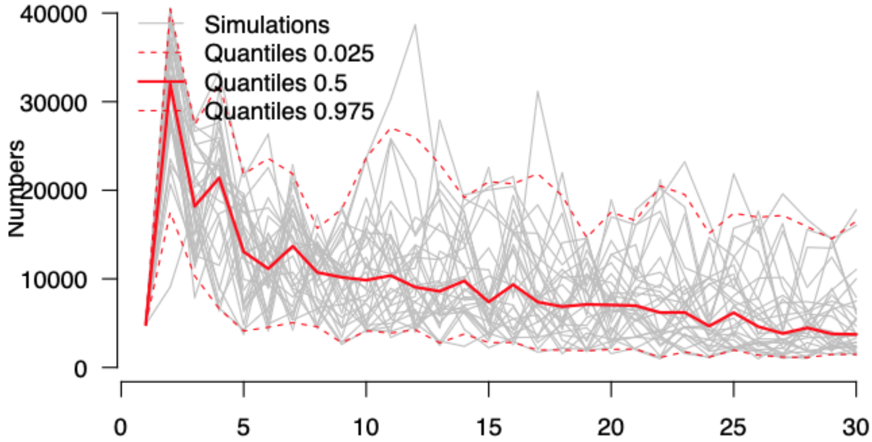
2.1 Leatherback turtle *Dermochelys coriacea*.

Scenario Dc1: Current situation

Basic Biological parameters used in the model scenario:

Parameter	Setting	Survivorship	SCL (cm)	s
Initial population size	4,846	Natural	0	0.200
Number of years for stabilization	30		85	0.800
Number of replicates	100		140	0.835
Average number of eggs per clutch	76.4	Human take	0	0.001
Standard deviation of number of eggs per clutch	16.1		85	0.001
Pivotal temperature model	WPac		140	0.001
Habitat mean	0.6	Commercial take	0	0.001
Habitat quality minimum limit	0.05		85	0.001
Habitat quality maximum limit	0.95		140	0.001

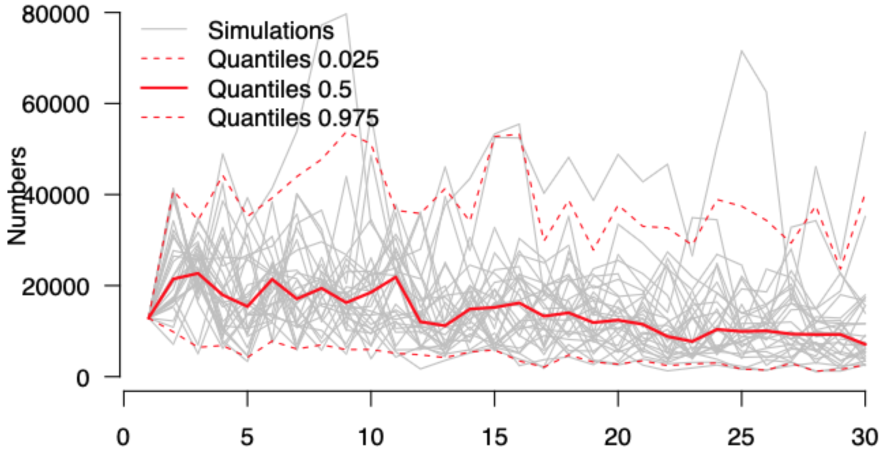
Predicted population trajectory (number of total individuals over years):



Descriptive summary:

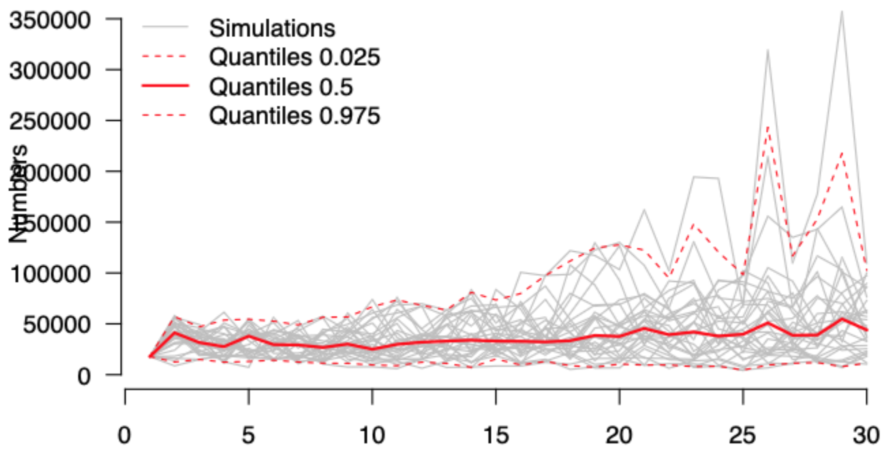
This is our best estimate of the current situation. Natural mortality is based on known findings from other locations (e.g. Papua New Guinea; Pilcher & Chaloupka 2013) and fisheries mortality is based on current estimated bycatch rate scenarios (e.g. Peatman et al. 2018a, 2018b, 2023). The starting number of turtles is based on current annual nesting stock (636 turtles), and then the Initial Population Size is modelled after the 'burn in' in to 'stabilise' turtle numbers, and includes all males and all other age classes following mathematical predictions based on habitat quality and reproductive output (this includes males and non-adult turtles). The model therefore determined there were 4,846 turtles at the start of the actual model runs. Harvest levels in this case are based on known moderate levels of bycatch in longline and purse seine industries (e.g. e.g. Peatman et al. 2018a, 2018b, 2023), alongside a low take of adults (e.g. in PNG; Pilcher 2013) and substantial take of eggs (e.g. PNG and West Papua, Indonesia; Tapilatu et al. 2013). Given the small initial population size of only 4,846 turtles, the impact of bycatch and take on long-term trends is significant. Given the low reproductive success of this species, and past and ongoing exploitation and bycatch, this population is shown to continue to decline under current levels of low survivorship of eggs and modest levels of bycatch in fisheries.

Scenario Dc2: Increased rate of bycatch in adults				
Parameter	**Setting**	**Survivorship**	**SCL (cm)**	**s**
Initial population size	12,869	Natural	0	0.200
Number of years for stabilization	30		85	0.800
Number of replicates	100		140	0.846
Average number of eggs per clutch	76.4	Human take	0	0.001
Standard deviation of number of eggs per clutch	16.1		85	0.001
Pivotal temperature model	WPac		140	0.001
Habitat mean	0.6	Commercial take	0	0.001
Habitat quality minimum limit	0.05		85	0.001
Habitat quality maximum limit	0.95		140	0.010



This model assumes the same basic parameters as Scenario Dc1 but with a higher mortality of adult nesters in commercial fisheries (0.010 instead of 0.001). Note that the burn in session ended with a starting population of 12,869 turtles, which was higher than that in Model Dc1. This is based on the model 'burn in' output following the same initial starting number of 636 adults, and the difference in number is simply variation in model outputs. What is more important to consider is the population trend rather than initial population size. This increase in initial population size also impacts the scale on the left hand, which is double the size of that in the earlier model. Under these conditions, and with natural mortality levels unchanged, the increased loss of adult turtles in commercial fisheries continues to impact the population and drives a decline in the population. Given the low reproductive success and past exploitation, and with the additional pressures of nester bycatch in fisheries and take on nesting beaches, these levels are detrimental and the population does not experience a reverse in the declining trend. While there are still substantial numbers of turtles that survive beyond 30 years under these conditions and some levels of take, the overall population trend is declining.

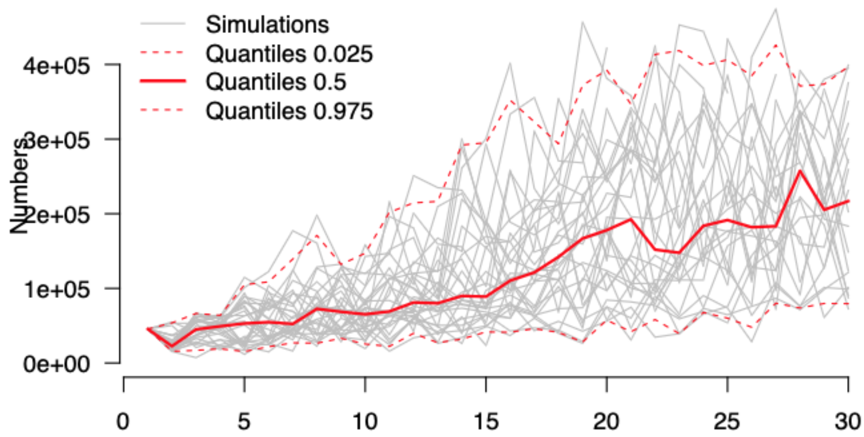
Scenario Dc3: Decreasing survivorship				
Parameter	**Setting**	**Survivorship**	**SCL (cm)**	**s**
Initial population size	17,880	Natural	0	0.250
Number of years for stabilization	30		85	0.800
Number of replicates	100		140	0.835
Average number of eggs per clutch	76.4	Human take	0	0.000
Standard deviation of number of eggs per clutch	16.1		85	0.000
Pivotal temperature model	WPac		140	0.000
Habitat mean	0.6	Commercial take	0	0.001
Habitat quality minimum limit	0.05		85	0.001
Habitat quality maximum limit	0.95		140	0.001



In this model (Dc3) we used the originally assumed current rate of fisheries and human take mortality, but decreased small turtle and adult natural survival (note 0.250 survivorship in small turtles, along with slightly lower adult survivorship: 0.835). This results in a stable population initially (again note the size of the overall population) although with substantial fluctuation in later years. If early years survival is increased this may offset the impacts of fisheries, but it is impractical given the wide dispersal of hatchlings as they depart nesting beaches.

Scenario Dc4: Increasing survivorship

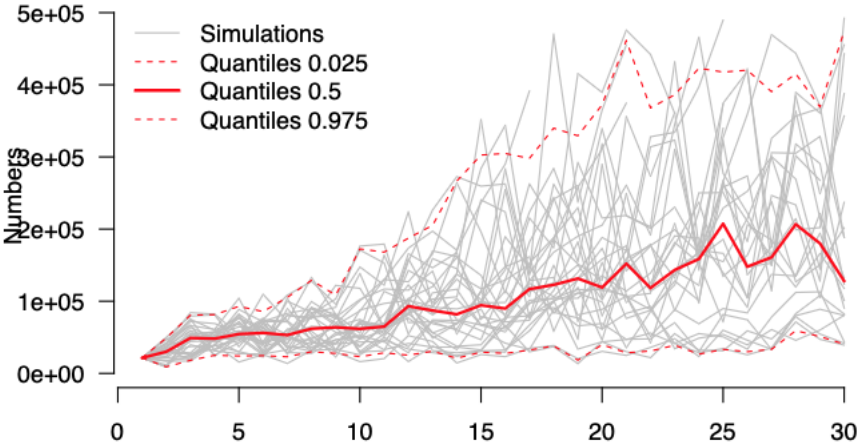
Parameter	Setting	Survivorship	SCL (cm)	s
Initial population size	45,591	Natural	0	0.300
Number of years for stabilization	30		85	0.800
Number of replicates	100		140	0.846
Average number of eggs per clutch	76.4	Human take	0	0.001
Standard deviation of number of eggs per clutch	16.1		85	0.001
Pivotal temperature model	WPac		140	0.001
Habitat mean	0.6	Commercial take	0	0.001
Habitat quality minimum limit	0.05		85	0.001
Habitat quality maximum limit	0.95		140	0.001



In this model Dc4 there is increased survivorship of eggs and hatchlings (note 0.300 survivorship vs. 0.200 survivorship in earlier Scenarios) and this results in an increasing overall population size due to increased survivorship of small life stages. While there is substantial fluctuation the model does not end in the extinction of the species; and it is likely that so long as mortality of adults and juveniles does not increase the population could recover slowly *if* we increase survivorship of eggs and hatchlings. This scenario demonstrates how any efforts to increase natural survivorship of hatchlings would be beneficial. In this model 'take' is estimated to be at current levels.

Scenario Dc5 Increasing Survivorship for small life stages

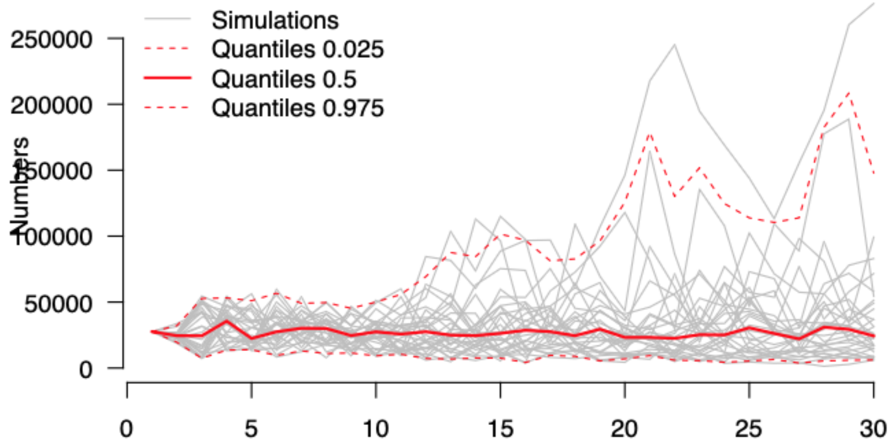
Parameter	Setting	Survivorship	SCL (cm)	s
Initial population size	21,508		0	0.300
Number of years for stabilization	30	Natural	85	0.800
Number of replicates	100		140	0.847
Average number of eggs per clutch	76.4		0	0.001
Standard deviation of number of eggs per clutch	16.1	Human take	85	0.001
Pivotal temperature model	WPac		140	0.001
Habitat mean	0.6		0	0.001
Habitat quality minimum limit	0.05	Commercial take	85	0.001
Habitat quality maximum limit	0.95		140	0.005



Again, this scenario (Model Dc5) also results in an increasing overall population size, due to higher survival of small life stages (note 0.300 survivorship vs. 0.200 survivorship in previous Scenarios). Even though in this scenario there is a slightly greater take of adults in fisheries and on nesting beaches (0.005 survivorship), the results project an overall increasing trend. Again, this scenario demonstrates that increasing the survivorship of eggs and hatchlings will result in an increasing population trend. However, it is important to note that the increased adult mortality may only represent ~10 adult turtles, so it is important to protect both adults and early years turtles.

Scenario Dc6 Higher juvenile / sub adult bycatch and take of nesters with a small increase in egg/hatchling survival

Parameter	Setting	Survivorship	SCL (cm)	s
Initial population size	27,568		0	0.250
Number of years for stabilization	30	Natural	85	0.800
Number of replicates	100		140	0.835
Average number of eggs per clutch	76.4		0	0.001
Standard deviation of number of eggs per clutch	16.1	Human take	85	0.010
Pivotal temperature model	WPac		140	0.001
Habitat mean	0.6		0	0.001
Habitat quality minimum limit	0.05	Commercial take	85	0.001
Habitat quality maximum limit	0.95		140	0.005



Here the model (Dc6) run has a higher juvenile / sub adult bycatch in fisheries and an elevated take in nesters. Again this model reveals substantial annual fluctuation after about 15 years, and projects a very gradual decline in numbers. A very small increase in egg and hatching survivorship is still insufficient to lead to population recovery, with minor impact from an elevated loss of subadults to fisheries and a few nesters on the beach.

2.2 Leatherback turtle conclusion and summary

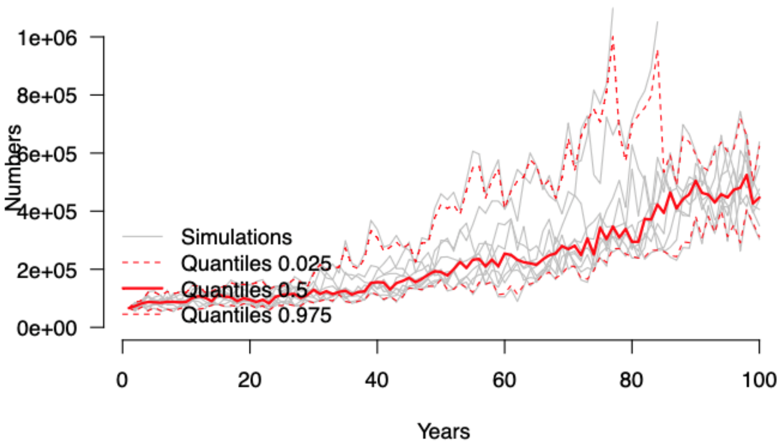
The vTurtle simulations presented above indicate that with the current small number of annual nesters (which average only 636 nesters per year in the western Pacific), the low survival of eggs and hatchlings due to substantive egg depredation on nesting beaches, and bycatch of a modest number of turtles (both immature and adult), management effort is needed. Of priority is a need to increase egg and hatchling survival, and reduce bycatch of larger turtles in commercial fisheries, if the Western Pacific leatherback population is to have any chance of survival. Caution is needed in simply increasing early year survival, as several trials with slightly increased mortality of adults and subadults and even nesters are insufficient to lead to substantial population recovery over the 30-year model run. Both an increased early-stage survival and a decrease in adult mortality via bycatch or take on nesting beaches are needed to recover this population.

Given the low numbers of leatherbacks remaining in the western Pacific the species is assessed as Critically Endangered.

Hatchling leatherback turtle making its way to the sea

2.3 Green turtle *Chelonia mydas*

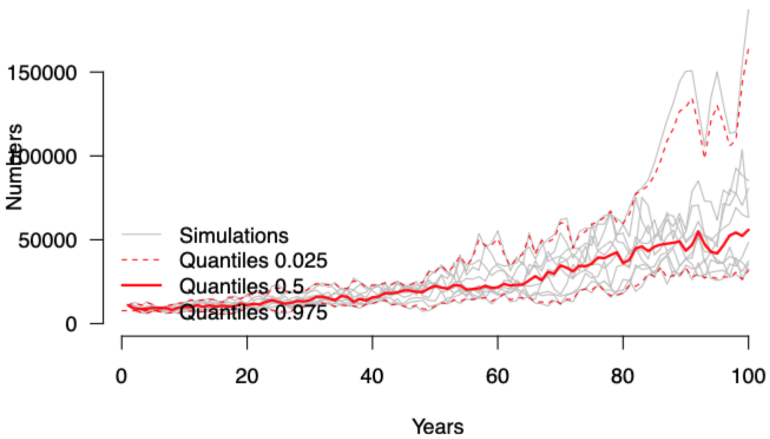
Scenario Cm1 Current situation – moderate direct take and bycatch of small turtles				
Parameter	**Setting**	**Survivorship**	**SCL (cm)**	**s**
Initial population size	97,854		0	0.200
Number of years for stabilization	100	Natural	40	0.800
Number of replicates	10		75	0.900
Average number of eggs per clutch	112		0	0.000
Standard deviation of number of eggs per clutch	21.56	Human take	40	0.001
Pivotal temperature model	WPac		75	0.001
Habitat mean	0.6		0	0.000
Habitat quality minimum limit	0.05	Commercial take	40	0.001
Habitat quality maximum limit	0.95		75	0.000



We believe models Cm1 (here) and Cm2 (below) to be representative of the current situation for green turtles in the Pacific region (the key difference between both models is the added take of adults in commercial fisheries in Cm2). Green turtles are modelled as being on a positive trajectory under moderate levels of human take (0.001 of both non-breeding turtles and adults) and similar bycatch of turtles in commercial fisheries. In this model this equates to a harvest of ~1,000 turtles in each category, but it is known that there are many more than just 97,854 green turtles in the Pacific region. Taking the same approach used above for leatherbacks (in Section 1.5), and understanding that the age to maturity is even longer for green turtles (30 years), it is likely this model represents only a small fraction of the turtles 'available to be taken' as described in Section 1.5. Indeed, rough calculations suggest that that the combined Central West and South Pacific nesting turtles could conservatively contribute some 2.3 to 4.7 million new hatchlings to the region each year. In model Cm1 we used a conservative estimate of 0.1% of the turtles in non-mature age classes were taken by commercial fisheries and both immature and adults were taken as human take. Recent findings from Solomon Islands: Hamilton et al. 2024; Fiji: Batibasaga et al. unpublished; Tonga: Stone et al. unpublished; PNG: Haskin et al. unpublished suggest this is commensurate with numbers of turtles taken in artisanal fisheries for consumption and trade. Similarly, Peatman et al. (2014) suggest several thousand green turtles are caught in longline and purse seine commercial fisheries each year, but the age class of these turtles is unknown.

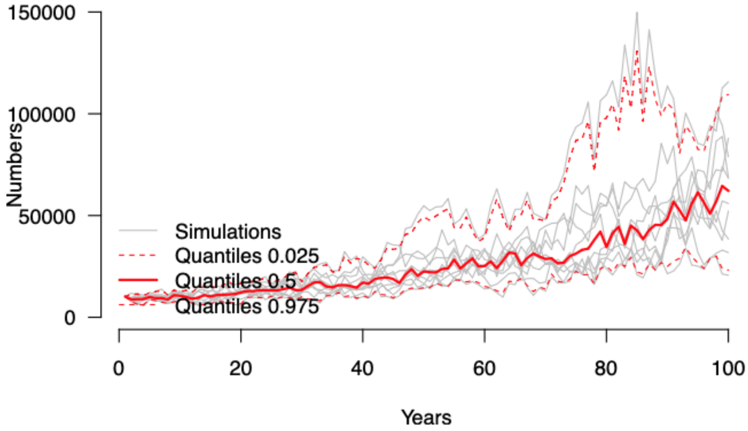
Hatchling green turtle migrating

Scenario Cm2 Current situation with bycatch of adults				
Parameter	**Setting**	**Survivorship**	**SCL (cm)**	**s**
Initial population size	10,998	Natural	0	0.200
Number of years for stabilization	100		40	0.800
Number of replicates	10		75	0.900
Average number of eggs per clutch	112	Human take	0	0.000
Standard deviation of number of eggs per clutch	21.56		40	0.001
Pivotal temperature model	WPac		75	0.001
Habitat mean	0.6	Commercial take	0	0.000
Habitat quality minimum limit	0.05		40	0.001
Habitat quality maximum limit	0.95		75	0.001



Model Cm2 is as presented in Cm1 above but with the added take of adults in commercial fisheries. After stabilisation this model run started with only 10,998 turtles, and therefore it is important to note the scale on numbers of turtles (recall that the starting number is simply the number of turtles after the run-in modelling phase, and not the actual number of turtles in the ocean). Despite the added commercial bycatch, the green turtles are modelled as being on a stable and positive trajectory under moderate levels of artisanal take (0.001) and to commercial fisheries.

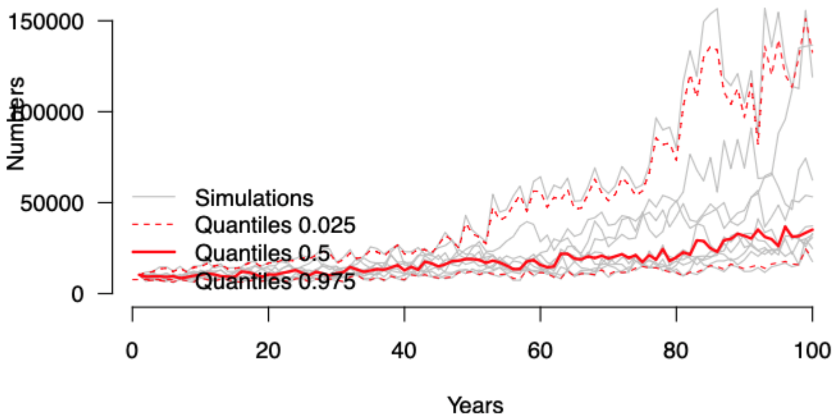
Scenario Cm3 Increased adult bycatch from commercial fisheries				
Parameter	**Setting**	**Survivorship**	**SCL (cm)**	**s**
Initial population size	10,175	Natural	0	0.200
Number of years for stabilization	100		40	0.800
Number of replicates	100		75	0.900
Average number of eggs per clutch	112	Human take	0	0.000
Standard deviation of number of eggs per clutch	21.56		40	0.001
Pivotal temperature model	WPac		75	0.001
Habitat mean	0.6	Commercial take	0	0.000
Habitat quality minimum limit	0.05		40	0.001
Habitat quality maximum limit	0.95		75	0.005



Increasing the level of commercial bycatch from 0.001 to 0.005 for adult turtles does not appear to have a major impact on green turtle population trends. Note that in this model (Cm3) the starting population size was similar to that in Cm2, and the levels of human take and bycatch still allow the population to grow. Here the total take and bycatch represents only 100 turtles, out of a starting population of 10,175. However, as noted above, green turtles are far more abundant in the western Pacific and data from turtle take interview questionnaires and bycatch estimates in commercial fisheries indicates they are taken in much larger quantities. Including Australia and New Caledonia, there are some 50,000 30-year adult female turtles, in addition to some 30% to 50% of this number as adult males. There are likely 15-20 additional years of adults in the ocean also given known length of breeding activity and multiple juvenile and subadult cohorts that are not yet reproductive, and these can number in the millions.

Scenario Cm4 No Bycatch and direct take only

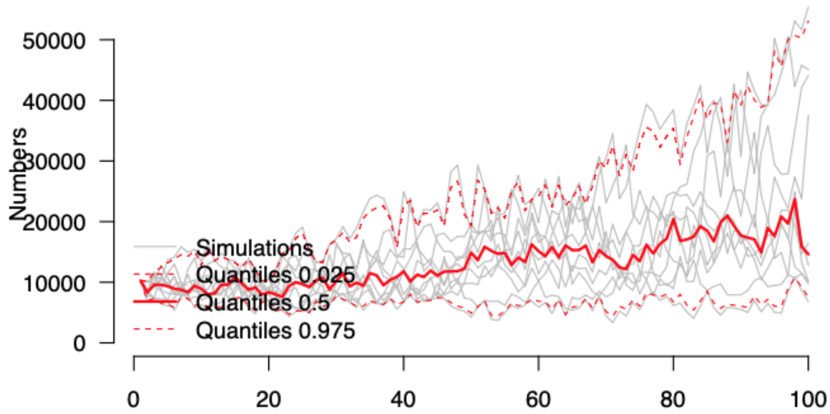
Parameter	Setting	Survivorship	SCL (cm)	s
Initial population size	10,318	Natural	0	0.200
Number of years for stabilization	100		40	0.800
Number of replicates	100		75	0.900
Average number of eggs per clutch	112	Human take	0	0.000
Standard deviation of number of eggs per clutch	21.56		40	0.001
Pivotal temperature model	WPac		75	0.005
Habitat mean	0.6	Commercial take	0	0.000
Habitat quality minimum limit	0.05		40	0.000
Habitat quality maximum limit	0.95		75	0.000



In this example the model (Cm4) the model ran without any bycatch in commercial fisheries, to demonstrate the impact of moderate levels of human take. In this scenario (Cm4) the model used a level of 5% take of non-adult turtles and 1% take of adults, as adults are far less frequently taken in artisanal fisheries (e.g. Hamilton et al. 2023). As can be seen from the graph, the population can withstand a certain level of human take but does not grow. Any additional pressures, such as the inclusion of bycatch in commercial fisheries (see next model scenarios, leads to population declines.

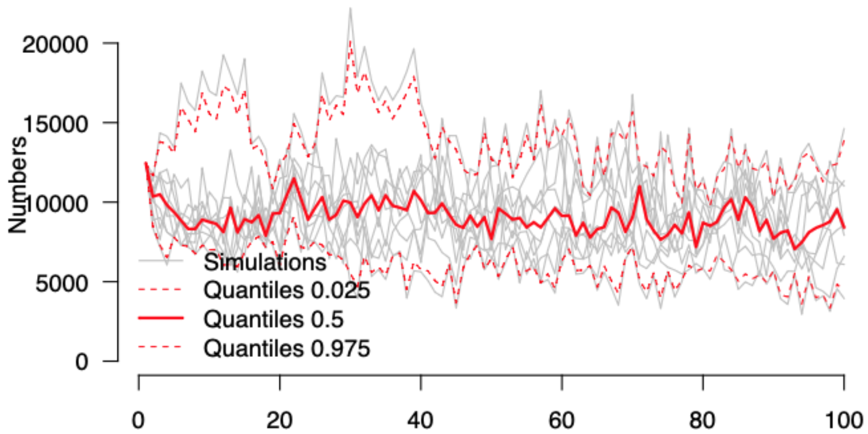
Scenario Cm5 Increased artisanal take of small turtles

Parameter	**Setting**	**Survivorship**	**SCL (cm)**	**s**
Initial population size	97,854	Natural	0	0.200
Number of years for stabilization	100		40	0.800
Number of replicates	100		75	0.900
Average number of eggs per clutch	112	Human take	0	0.000
Standard deviation of number of eggs per clutch	21.56		40	0.050
Pivotal temperature model	WPac		75	0.000
Habitat mean	0.6	Commercial take	0	0.000
Habitat quality minimum limit	0.05		40	0.000
Habitat quality maximum limit	0.95		75	0.000



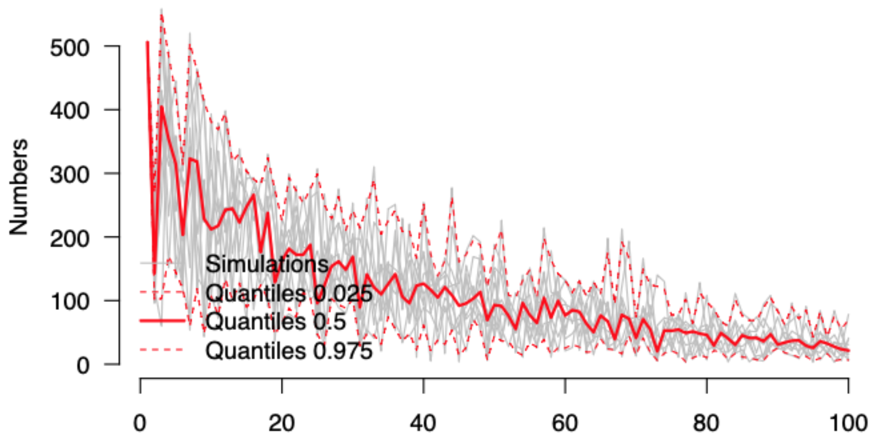
In this example the model (Cm5) the model ran with a larger amount of artisanal take of smaller turtles (0.050) and the numbers of turtles over time overall is an order of magnitude lower than in past scenarios, and the population is not growing at any substantive rate. It is likely that the long-term resilience of the population is lowered, as the modelled number of nesting females drops slightly in the first years and then grows only modestly. In this case the human take level represents some 4,900 turtles (starting population × human take), which is known to be lower than current take levels across the western Pacific. However, as noted above, it is likely that the numbers of juvenile turtles are an order of magnitude higher than this modelled starting population (but the model could not run with such large numbers as it models the individual trajectory of each turtle in the model).

Scenario Cm6 Increased direct take of juvenile turtles only				
Parameter	**Setting**	**Survivorship**	**SCL (cm)**	**s**
Initial population size	12,442	Natural	0	0.200
Number of years for stabilization	100		40	0.800
Number of replicates	100		75	0.900
Average number of eggs per clutch	112	Human take	0	0.000
Standard deviation of number of eggs per clutch	21.56		40	0.100
Pivotal temperature model	WPac		75	0.000
Habitat mean	0.6	Commercial take	0	0.000
Habitat quality minimum limit	0.05		40	0.000
Habitat quality maximum limit	0.95		75	0.000



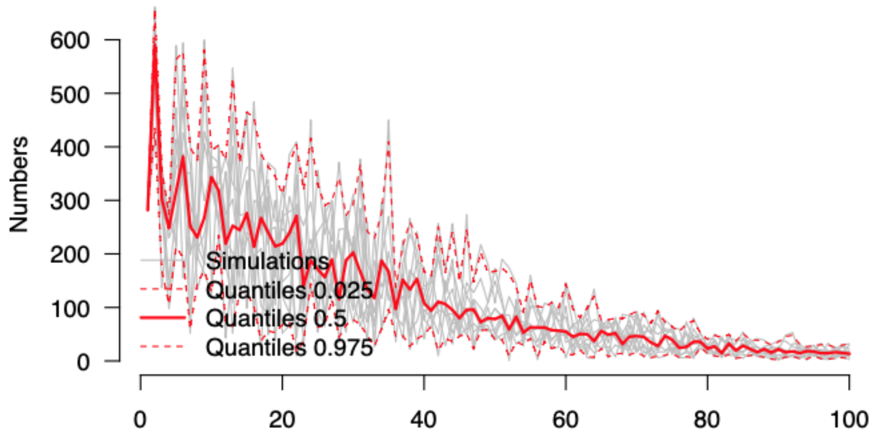
However, if the artisanal take of juveniles is increased to 0.100 (10% of the population) – again with no take of adults, the model suggests the population would gradually decline. There would be a large number of adults initially but the excessive take of smaller turtles would limit the number of females in the population which would then not be able to recover. That is, an excessively high take of juveniles can have significant negative impacts on the population. A rough calculation (along the lines of that for leatherbacks described in Section 1.5 above) suggests there might be 3.3 to 4.5 million juvenile green turtles in the western Pacific, therefore this 10% take would equate to 330,000 to 450,000 immature green turtles. Recent estimates for Solomon Islands, Fiji, and Tonga alone indicate take rates could exceed 20,000 to 120,000 green turtles each year (confidence intervals around PNG human take data preclude including them in this analysis).

Scenario Cm7 High artisanal take and high commercial bycatch				
Parameter	**Setting**	**Survivorship**	**SCL (cm)**	**s**
Initial population size	97,854	Natural	0	0.200
Number of years for stabilization	100		40	0.800
Number of replicates	100		75	0.900
Average number of eggs per clutch	112	Human take	0	0.000
Standard deviation of number of eggs per clutch	21.56		40	0.050
Pivotal temperature model	WPac		75	0.010
Habitat mean	0.6	Commercial take	0	0.000
Habitat quality minimum limit	0.05		40	0.100
Habitat quality maximum limit	0.95		75	0.010



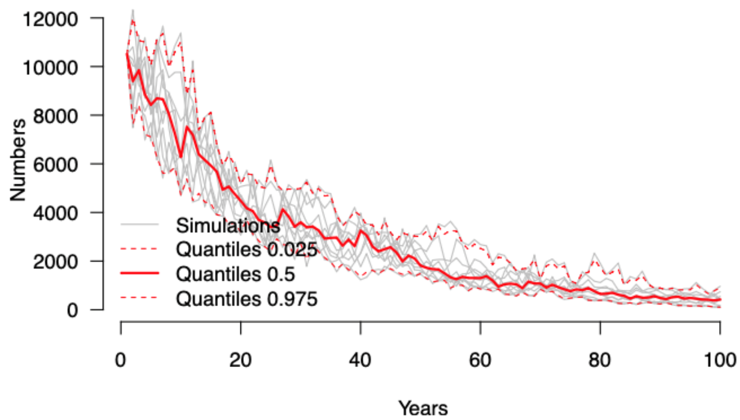
In this scenario (model Cm7) a combination of high artisanal take (0.050 and 0.010) and high commercial fishing bycatch (0.100 & 0.010) indicates the population would decline rapidly and not recover. Under these conditions, human take equates to 4,892 non-adult and 978 adult turtles, and bycatch accounts for 9,700 small and 978 adult turtles, or a total of 16,635 turtles of the starting population of 97,854. As noted above, the total number of green turtles in the Pacific region is far larger, but the model indicates that elevated bycatch combined with elevated human take is unsustainable. While the commercial fisheries in the Pacific generally do not impact green turtles unless the turtles are migrating from one area to another (commercial fisheries tend to operate in deeper offshore waters and not where green turtles feed), it is known that some level of take does occur, alongside substantial artisanal take, and therefore this scenario is realistic if high levels of bycatch were sustained, and is indicative of the need to reduce bycatch in commercial fisheries and lower the levels of human take.

Scenario Cm8 Increased direct take of immature turtles and high levels of bycatch				
Basic Biological parameters used in the model scenario:				
Parameter	**Setting**	**Survivorship**	**SCL (cm)**	**s**
Initial population size	10,058	Natural	0	0.200
Number of years for stabilization	100		40	0.800
Number of replicates	100		75	0.900
Average number of eggs per clutch	112	Human take	0	0.000
Standard deviation of number of eggs per clutch	21.56		40	0.025
Pivotal temperature model	WPac		75	0.000
Habitat mean	0.6	Commercial take	0	0.000
Habitat quality minimum limit	0.05		40	0.050
Habitat quality maximum limit	0.95		75	0.010



In this example the model (Cm8) the model had a substantial level of human take of immature turtles (much as is occurring in the Solomon Islands, for example) coupled with a high level of bycatch in commercial fisheries of both adults and non-adult turtles. In this case the combined pressures of take and bycatch would lead to a population decline to zero after 100 years. This model scenario demonstrates that elevated bycatch and significant levels of human are unsustainable and under those conditions both a reduction in bycatch and a reduction in human take are required for green turtle populations to survive.

Scenario Cm9 Increased direct take of turtles and bycatch – all sizes				
Parameter	**Setting**	**Survivorship**	**SCL (cm)**	**s**
Initial population size	10,507	Natural	0	0.200
Number of years for stabilization	100		40	0.800
Number of replicates	100		75	0.900
Average number of eggs per clutch	112	Human take	0	0.000
Standard deviation of number of eggs per clutch	21.56		40	0.025
Pivotal temperature model	WPac		75	0.050
Habitat mean	0.6	Commercial take	0	0.000
Habitat quality minimum limit	0.05		40	0.050
Habitat quality maximum limit	0.95		75	0.010



In this example (Cm9) the model was run with both elevated artisanal take of both adults and non-adult turtles, and commercial bycatch. Much as in Model Cm8 above, the combined pressures of high levels of take of small and large turtles, combined with high levels of bycatch of both age classes, leads to a population collapse after 100 years.

2.4 Green turtle conclusion and summary

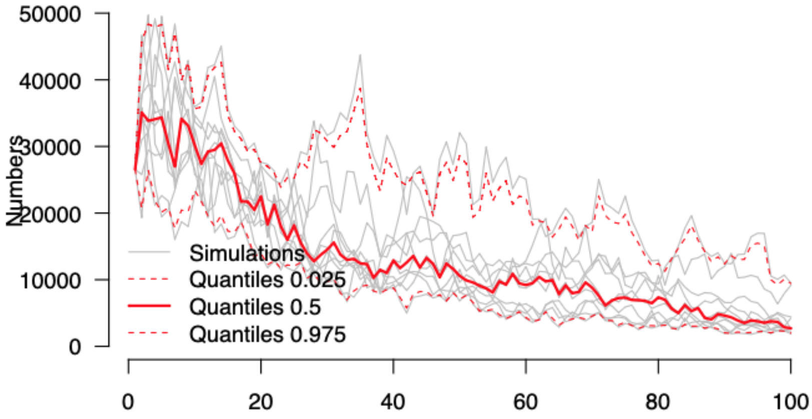
Based on recent trend estimates for multiple green turtle nesting sites in the Pacific region (MTSG, unpublished data) green turtle numbers in the western Pacific appear to be stable and slowly recovering. The southern Pacific stock numbers in the tens of thousands of nesters, providing added population resilience. However, substantial take is occurring in many Pacific countries, and bycatch rates in commercial fisheries do not appear to be diminishing. In the absence of substantive human take and fisheries bycatch the population demonstrates the potential to grow, but when modest levels of take are introduced at both sub-adult and adult stages, populations remain stable and do not grow. When levels of take become excessive, for example the recent reports of take in the Solomon Islands, Tonga, PNG and Fiji (Hamilton et al. 2024, Batibasaga et al. unpublished; Stone et al. unpublished; Haskin et al. unpublished), and when commercial fisheries bycatch levels are maintained, the populations generally decline over time. It is likely the decline would be more impactful and notable when the harvest of eggs is considered. In order for green turtles to survive in the Pacific islands the amount of take needs to be moderated and commercial bycatch needs to be addressed for the levels of turtle mortality to be sustainable.

Given the large number of nesters in Australia, this species is assessed as Least Concern for Australia. However, given that the majority of other nesting locations in the Pacific host only tens of green sea turtles per year as nesters, they could be considered Endangered in these locations due to the small numbers of adults and the continued pressures from human take.

Adult green turtle returning to sea after nesting

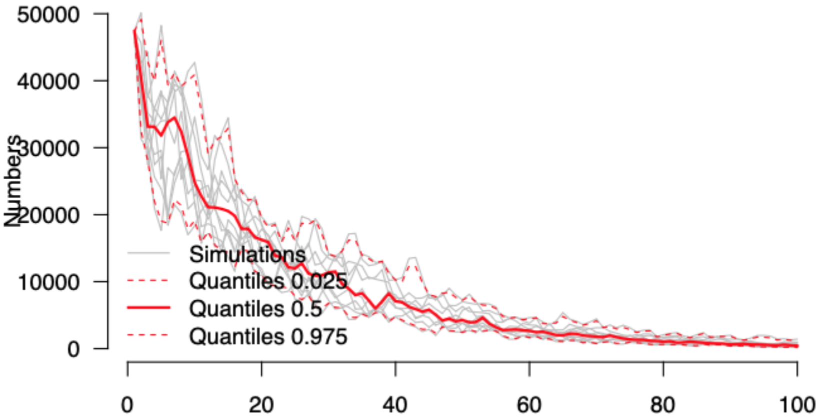
2.5 Hawksbill turtle *Eretmochelys imbricata*

Scenario Ei1 Current situation				
Parameter	**Setting**	**Survivorship**	**SCL (cm)**	**s**
Initial population size	26,560	Natural	0	0.200
Number of years for stabilization	100		45	0.800
Number of replicates	100		75	0.845
Average number of eggs per clutch	121.7	Human take	0	0.200
Standard deviation of number of eggs per clutch	23.4		45	0.025
Pivotal temperature model	SWPac		75	0.001
Habitat mean	0.6	Commercial take	0	0.000
Habitat quality minimum limit	0.05		45	0.001
Habitat quality maximum limit	0.95		75	0.005



This is a realistic current scenario (Ei1) for Hawksbill turtles in the western Pacific. Hawksbills are under substantial pressure across the region, and are taken as part of traditional or consumptive use and others as accidental bycatch. There is known take in both commercial and artisanal, from small plate-sized turtles to fully-grown adults. The model suggests Hawksbill turtles are on an extinction trajectory so long as there is a level of take of eggs on beaches, and turtles of all size classes taken as bycatch and/or human take at sea. In this example the mortality rates represent 6,133 turtles from the starting population of 26,560. However, as in the case of green and leatherback turtles, there are far greater numbers of hawksbills at sea in the western Pacific than those run in this model. The total annual hawksbill turtle take estimates in Tonga, Fiji and Solomon Islands amount to ~28,400 turtles, and it is likely that this exceeds the number of turtles available for take in the region.

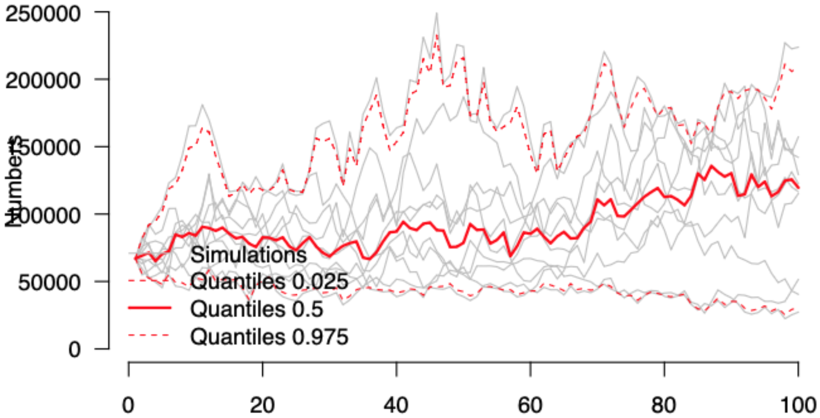
Scenario Ei2 Increased commercial bycatch				
Parameter	**Setting**	**Survivorship**	**SCL (cm)**	**s**
Initial population size	47,420	Natural	0	0.200
Number of years for stabilization	100		45	0.800
Number of replicates	100		75	0.845
Average number of eggs per clutch	121.7	Human take	0	0.200
Standard deviation of number of eggs per clutch	23.4		45	0.025
Pivotal temperature model	SWPac		75	0.000
Habitat mean	0.6	Commercial take	0	0.000
Habitat quality minimum limit	0.05		45	0.025
Habitat quality maximum limit	0.95		75	0.050



In scenario Ei2, the level of bycatch in commercial fisheries was increased leading to a steeper decline in turtle numbers. Specifically, when the bycatch of adults increases (note the adult bycatch was increased ten-fold from 0.005 in model Ei1 to 0.05 in model Ei2), the Hawksbill turtles are projected to decline on a more-rapid extinction trajectory so long as there is a continued level of take of eggs on beaches, and turtles are taken as bycatch and/or human take on beaches.

Scenario Ei3 Increased early stage survival and reduced take of eggs

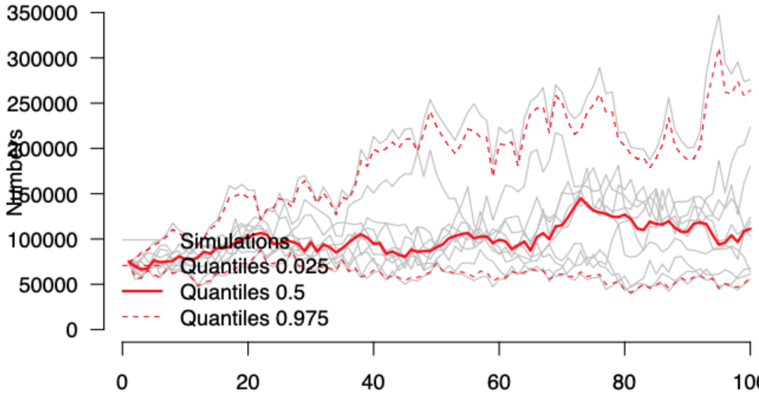
Parameter	Setting	Survivorship	SCL (cm)	s
Initial population size	67,010	Natural	0	0.300
Number of years for stabilization	100		45	0.800
Number of replicates	100		75	0.847
Average number of eggs per clutch	121.7	Human take	0	0.001
Standard deviation of number of eggs per clutch	23.4		45	0.001
Pivotal temperature model	SWPac		75	0.005
Habitat mean	0.6	Commercial take	0	0.000
Habitat quality minimum limit	0.05		45	0.015
Habitat quality maximum limit	0.95		75	0.050



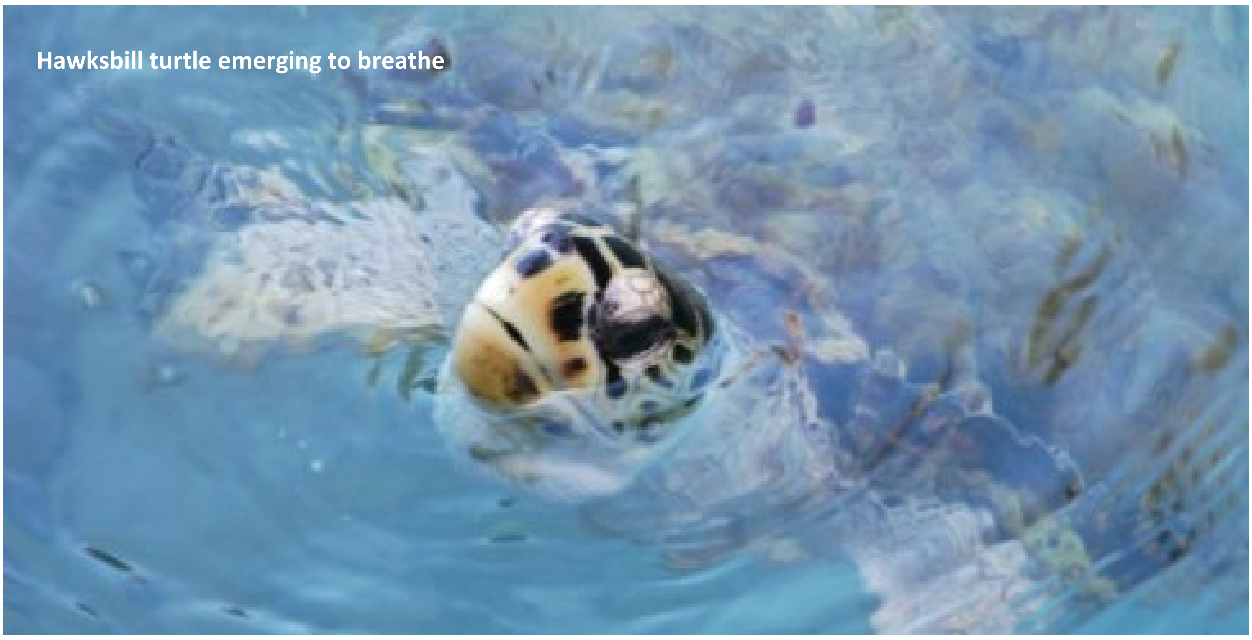
However, if early life stage survival is increased (note the 0-45 cm size class natural survival in model Ei3 has been increased from 0.200 to 0.300) and pressures on eggs decreased (note only 0.001 survivorship), the hawksbill population demonstrates the ability to recover slightly even when, interestingly, artisanal take is decreased to only occasional adult turtles, and with added pressure from commercial fisheries. This suggests that increasing early survivorship on beaches and nearshore waters, and decreasing the take of turtles in coastal fisheries has the potential to lead to population recovery. With increased early life stage survival (e.g. better protection of eggs on beaches and higher survival of offshore younger life stages), the population has the ability to maintain itself in a relatively stable trend even with low mortality through artisanal fisheries in all life stages and some level of commercial harvests.

Scenario Ei4 Reduced early-stage survival with no bycatch

Parameter	Setting	Survivorship	SCL (cm)	s
Initial population size	75,230	Natural	0	0.250
Number of years for stabilization	100		45	0.800
Number of replicates	100		75	0.845
Average number of eggs per clutch	121.7	Human take	0	0.001
Standard deviation of number of eggs per clutch	23.4		45	0.050
Pivotal temperature model	SWPac		75	0.050
Habitat mean	0.6	Commercial take	0	0.000
Habitat quality minimum limit	0.05		45	0.000
Habitat quality maximum limit	0.95		75	0.000



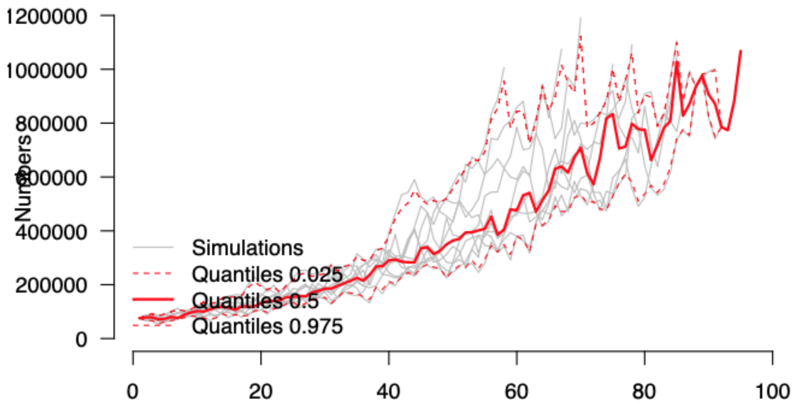
In model Ei4, the early-stage survival was lowered slightly to 0.250, and there was no bycatch in commercial fisheries (it is expected that hawksbills are less frequently caught in commercial fisheries as they are normally found on coral reefs). With a slightly higher early-stage survival, and in the absence of commercial fisheries but with substantial artisanal take (in this case ~7,500 turtles) the population remains stable through time but is not able to recover, and remains vulnerable to external pressures. With the addition of commercial fishery bycatch, this population would not survive, and this suggests that artisanal take of juveniles and adults is a key driver behind population declines and needs to be curtailed across the Pacific range for hawksbill turtles.



Hawksbill turtle emerging to breathe

Scenario Ei5 Modest direct take and no bycatch

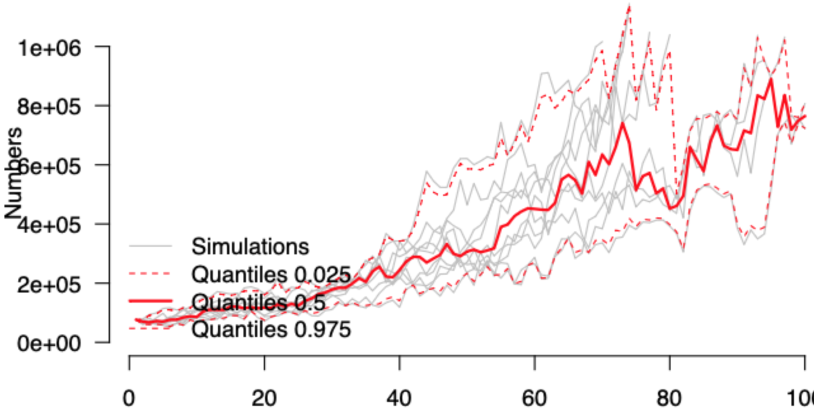
Parameter	Setting	Survivorship	SCL (cm)	s
Initial population size	75,640	Natural	0	0.300
Number of years for stabilization	100		45	0.800
Number of replicates	100		75	0.846
Average number of eggs per clutch	121.7	Human take	0	0.000
Standard deviation of number of eggs per clutch	23.4		45	0.025
Pivotal temperature model	SWPac		75	0.000
Habitat mean	0.6	Commercial take	0	0.000
Habitat quality minimum limit	0.05		45	0.000
Habitat quality maximum limit	0.95		75	0.000



However, when commercial bycatch is eliminated and human take is modest (in this Ei5 scenario it represents only 1,891 turtles from a starting population of 75,640) and only impacts mid-sized turtles, the population of hawksbills has the potential to recover. It is clear from this example that the declines in numbers of hawksbills in the Pacific have been due to human take and commercial captures in the past, which needs to be addressed and reversed if the population is to recover. If human take and commercial pressures are reduced there is hope for the hawksbills in the Pacific. However, the current estimate of 28,400 turtles from just Fiji, Solomon Islands and Tonga in an underestimate because it does not include other countries (e.g. PNG) or commercial fisheries (which could number in the 1,000s), and it is likely that a reduction in the range of 10 or more thousand hawksbills taken per year is required to facilitate recovery at the regional level.

Hawksbill turtle looking for a site to nest

Scenario Ei6 Increased early life stage survival				
Parameter	**Setting**	**Survivorship**	**SCL (cm)**	**s**
Initial population size	76,270	Natural	0	0.300
Number of years for stabilization	100		45	0.800
Number of replicates	100		75	0.846
Average number of eggs per clutch	121.7	Human take	0	0.000
Standard deviation of number of eggs per clutch	23.4		45	0.000
Pivotal temperature model	SWPac		75	0.001
Habitat mean	0.6	Commercial take	0	0.000
Habitat quality minimum limit	0.05		45	0.001
Habitat quality maximum limit	0.95		75	0.025



If early life stage survival is increased (note natural survival has been increased on 0.300), the population demonstrates the ability to recover even when, interestingly, artisanal take is decreased to only occasional adult turtles, and with added pressure from commercial fisheries. This suggests that increasing early survivorship on beaches and nearshore waters, and decreasing the bycatch of turtles in coastal fisheries has the potential to lead to population recovery.

2.6 Hawksbill turtle conclusion and summary

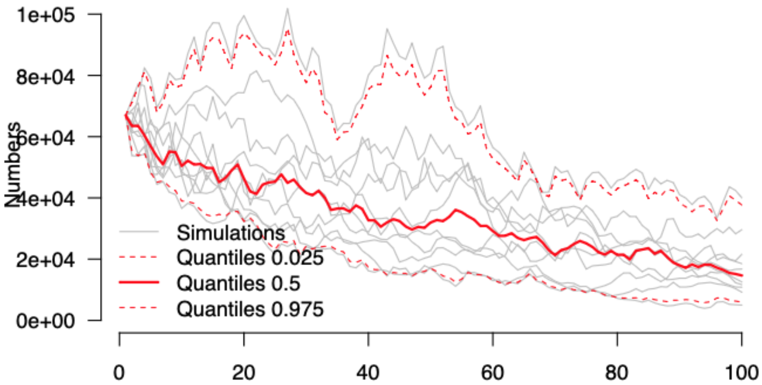
Hawksbill turtles have been taken as part of traditions in the Pacific region for many years (e.g. as *Toluk* in Palau), and in addition have been the focus of concerted commercial harvests to supply the *bekko* trade and more recently the curio trade in China. This has pushed populations to the brink of extinction in many parts of the world. Except in isolated cases (e.g. Arnavons, Solomon Islands; Hamilton et al. 2015), long-term protection of hawksbill nesting beaches has not been possible in many parts of the Pacific due to the diffuse nesting across multiple islands and the logistical challenges and costs in mounting long-term monitoring programmes.

The multiple vTurtle model outcomes all suggest that with ongoing trade, direct take and accidental bycatch the populations will still continue to decline. In order to reverse the decline, there is an urgent need to address bycatch and accidental take, minimise direct take of hawksbills, eliminate the international trade in hawksbill products, and to protect the few hawksbill turtles that are nesting. It may be preferable also to temporarily cease direct take (and all its uses e.g. tradition and consumption) of this species until it shows signs of recovery.

<u>Given the low numbers of hawksbill turtles remaining in the western Pacific the species is assessed as Critically Endangered.</u>

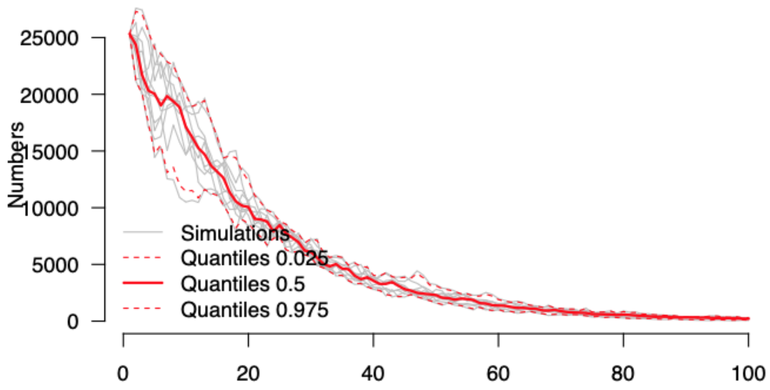
2.7 Loggerhead turtle *Caretta caretta*

Scenario Cc1 Current situation				
Parameter	**Setting**	**Survivorship**	**SCL (cm)**	**s**
Initial population size	66,954	Natural	0	0.200
Number of years for stabilization	100		45	0.800
Number of replicates	100		75	0.845
Average number of eggs per clutch	22.15	Human take	0	0.000
Standard deviation of number of eggs per clutch	3.82		45	0.0005
Pivotal temperature model	WPac		75	0.0001
Habitat mean	0.6	Commercial take	0	0.000
Habitat quality minimum limit	0.05		45	0.000
Habitat quality maximum limit	0.95		75	0.001



It is known that loggerhead turtles are not a favoured meat among local communities, and that accidental take of loggerheads in small-scale artisanal fisheries in the Pacific region is uncommon. However, commercial longline and purse-seine fisheries are known to impact sea turtles on a large scale due to their oceanic nature. Loggerheads nest primarily in Australia (~80% of the breeding population) and New Caledonia ~15-20% of the breeding population), and move across the southern Pacific to feed off the coast of South America. During these migrations the turtles are vulnerable to commercial fishing pressure. The total nesting population is projected to be in the region of 500-1,000 adult females. Given the small nesting population, even when a small number of adults is caught by commercial fisheries this can lead to population declines, as demonstrated in this model run Cc1. Interestingly this small number of mortalities was also modelled to result in the loss of Pacific loggerheads as far back as the 1990s (Heppell et al. 1996) and later by Chaloupka et al. (2004).

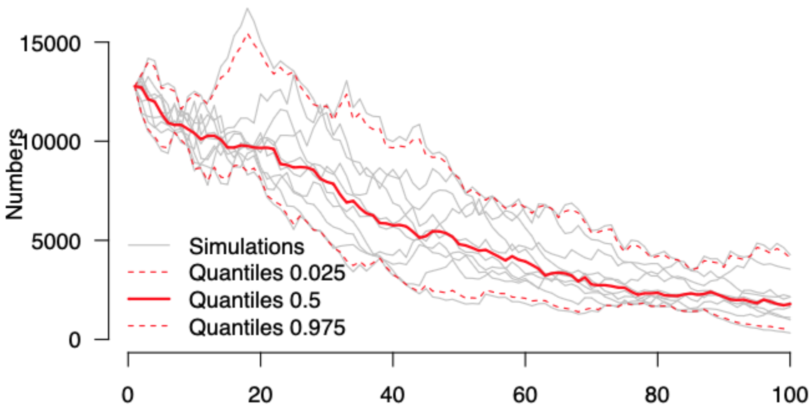
Scenario Cc2 Increased bycatch from commercial fisheries				
Parameter	**Setting**	**Survivorship**	**SCL (cm)**	**s**
Initial population size	25,319	Natural	0	0.200
Number of years for stabilization	100		45	0.800
Number of replicates	100		75	0.845
Average number of eggs per clutch	22.15	Human take	0	0.300
Standard deviation of number of eggs per clutch	3.82		45	0.000
Pivotal temperature model	WPac		75	0.001
Habitat mean	0.6	Commercial take	0	0.000
Habitat quality minimum limit	0.05		45	0.001
Habitat quality maximum limit	0.95		75	0.100



In this model Cc2, the bycatch rates were increased significantly, to depict what might occur in the absence of bycatch mitigation measures such as turtle excluder devices (TEDs), in prawn trawl fisheries. The decline in Pacific loggerheads is far more dramatic when fisheries bycatch of larger individuals is increased (note the 0.100 survivorship in adult turtles as Commercial take) and would result in the total loss of the population over a 100-year period. For this reason, the bycatch of loggerhead turtles needs to be mitigated to ensure there is no complete population loss. While TEDs are in use in Australian prawn fisheries and are extremely effective, there is continued bycatch of loggerheads in longline and purse seine industries, and there is a substantive loss of juvenile turtles to coastal fisheries of the coasts of Chile and Peru (e.g. Donoso & Dutton 2010, Velez-Zuazo & Kelez 2010).

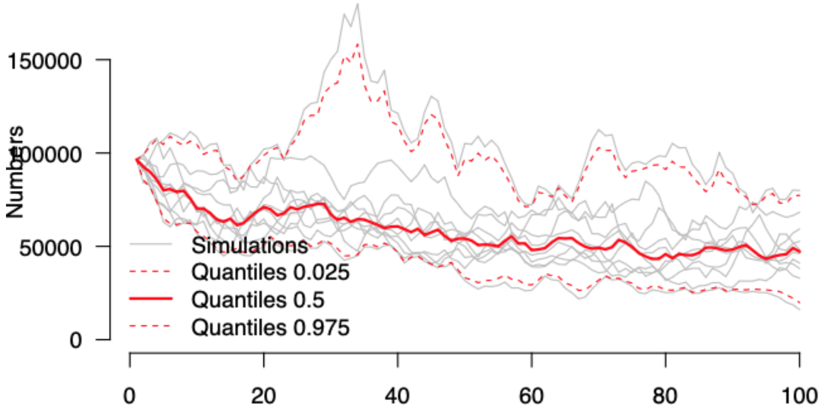
Scenario Cc3 Decreased survival of early stage turtles

Parameter	**Setting**	**Survivorship**	**SCL (cm)**	**s**
Initial population size	12,769	Natural	0	0.300
Number of years for stabilization	100		45	0.800
Number of replicates	100		75	0.847
Average number of eggs per clutch	22.15	Human take	0	0.000
Standard deviation of number of eggs per clutch	3.82		45	0.000
Pivotal temperature model	WPac		75	0.050
Habitat mean	0.6	Commercial take	0	0.000
Habitat quality minimum limit	0.05		45	0.000
Habitat quality maximum limit	0.95		75	0.010



In this scenario (Model Cc3), with an increased lower years survival (raised to 0.300; e.g. better protection of eggs on beaches and higher survival of offshore younger life stages) but with continued take of adults in commercial fisheries and occasional bycatch in artisanal fisheries loggerhead turtles continue to decline, although not reaching extinction within the 100-year time frame. This suggests that if incubation and early life stage survival is increased in Australia (the largest nesting site), the loggerhead turtle population would still be at risk due to the ongoing loss of turtles as bycatch.

Scenario Cc4 Increased survival of early stage turtles and no direct take				
Parameter	**Setting**	**Survivorship**	**SCL (cm)**	**s**
Initial population size	96,493	Natural	0	0.300
Number of years for stabilization	100		45	0.800
Number of replicates	100		75	0.847
Average number of eggs per clutch	22.15	Human take	0	0.000
Standard deviation of number of eggs per clutch	3.82		45	0.000
Pivotal temperature model	WPac		75	0.000
Habitat mean	0.6	Commercial take	0	0.000
Habitat quality minimum limit	0.05		45	0.050
Habitat quality maximum limit	0.95		75	0.010

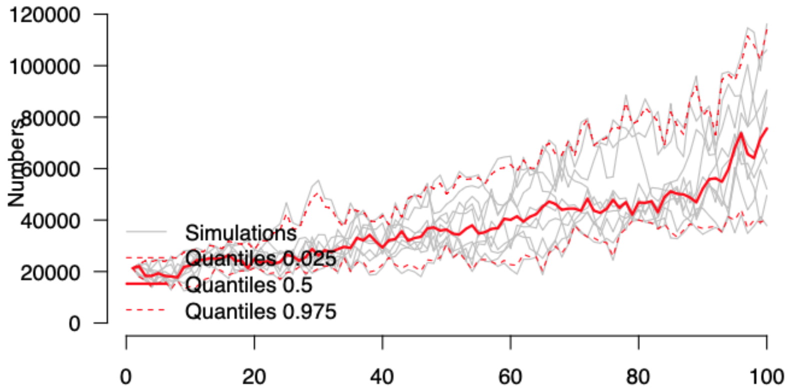


In model Cc4, the early life-stage survival was increased, and all human take of turtles was eliminated. When younger age class survival increases (note the change to 0.300 under natural survivorship, which would equate to increasing hatchling survival and reducing loss of small juveniles in eastern Pacific small-scale fisheries), and in the absence of artisanal take but with continued commercial fishing, the population of loggerheads in the Pacific would continue to decline although it would not reach extinction after 100 years. This demonstrates the value of increasing early-stage survival but does not eliminate the need to address bycatch in commercial fisheries, such as longline and purse seine fisheries in the central and western Pacific, and small-scale fisheries in the eastern Pacific.

Adult loggerhead turtle with a satellite transmitter

Scenario Cc5 Increased early stage survival and reduced bycatch

Parameter	Setting	Survivorship	SCL (cm)	s
Initial population size	21,293	Natural	0	0.300
Number of years for stabilization	100		45	0.800
Number of replicates	100		75	0.845
Average number of eggs per clutch	22.15	Human take	0	0.300
Standard deviation of number of eggs per clutch	3.82		45	0.000
Pivotal temperature model	WPac		75	0.001
Habitat mean	0.6	Commercial take	0	0.000
Habitat quality minimum limit	0.05		45	0.000
Habitat quality maximum limit	0.95		75	0.005



With natural survival rates in early years survival set slightly higher at 0.300, and with lower commercial and artisanal take (for example, if bycatch challenges had been addressed) the population of loggerhead turtles in the Pacific could increase over a 100-year period. This demonstrates the value of reducing impacts from commercial fisheries while also working to increase survivorship of clutches on beaches where there may be predation of eggs in the wild.

2.8 Loggerhead turtle conclusion and summary (Southwest Pacific RMU).

There is little demand for loggerhead turtles for consumption and use for traditional purposes, particularly as meat from other species is preferred. At some locations there is natural depredation of nests, which – if addressed – could positively impact recovery of the species. Bycatch of loggerheads in the western Pacific appears to be the main driver behind population declines. Commercial longline vessel bycatch peaked in 2006 and declined to roughly ¼ of the previous magnitude of take, and has remained relatively stable since that time (impacting an average of 2,600 loggerheads per year in the WPFC region). In the purse seine industry, bycatch peaked in 2011 but has declined substantially since, averaging a modelled 190 turtles per year. The loss of loggerheads declined dramatically following the mandatory use of Turtle Excluder Devices in the Northern Prawn Fishery in 2000 (Brewer et al. 2006). In the early 1990s, Australia's Commonwealth Scientific Industrial Research Organisation estimated that the country's Northern Prawn Fishery (NPF) incidentally caught 5,000–6,000 sea turtles each year, of which 39% likely died (Pilcher & Robins 2010), but this bycatch dropped by 99% following the use of TEDs (Brewer et a. 2006). Thus, any reduction in bycatch in other commercial fisheries will have a significant impact on population recovery. Loggerhead turtles are being taken as bycatch in distant fleets and some eggs are lost to predators on beaches, along with very occasional human take. Only increases in early-stage survivorship and removal of substantial bycatch pressures are projected to have an impact on the species recovery.

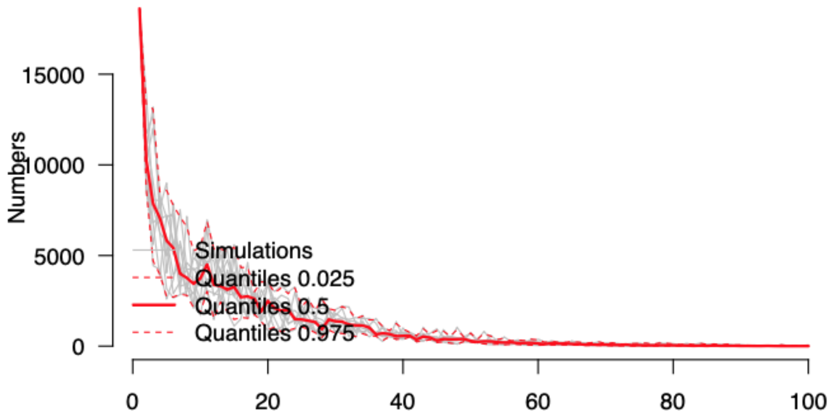
<u>The species is assessed as Endangered across its range in the western Pacific.</u>

2.9 Olive ridley turtle *Lepidochelys olivacea* (West Pacific RMU).

Olive ridleys are not as common in the Pacific region as other species. Small nesting populations exist in Australia, New Caledonia, Indonesia and Papua New Guinea. The population of Olive ridley turtles across the western Pacific is moderate in size, supported primarily by the nesting population in Australia. An estimate of the nesting population for Australia is 1,000-5,000 females annually, with most nesting in north west Arnhem Land (DCCEEW 2025). On the beaches there are a number of predators, and up to 50% of unguarded nests can be depredated (either fully or partially; Nordberg et al. 2019). Recent predator eradication programmes have been successful in some places (see Australia's Nest to Ocean Programme[2]) but require ongoing funding and are extremely resource-demanding.

Scenario Lo1 Current situation

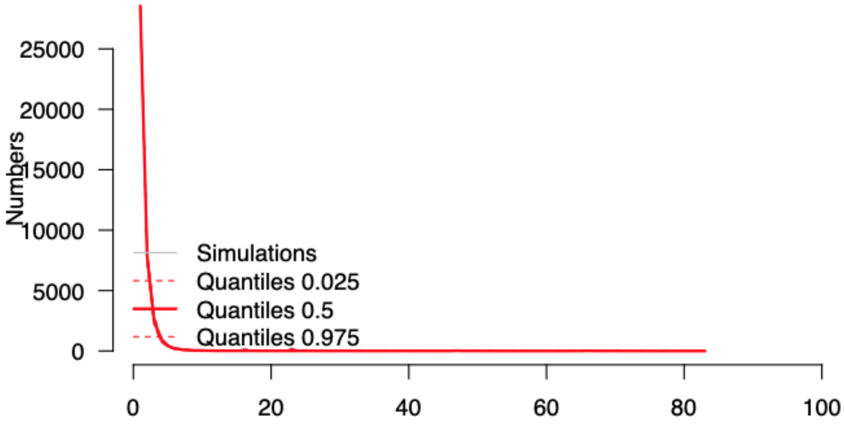
Parameter	Setting	Survivorship	SCL (cm)	s
Initial population size	18,612	Natural	0	0.200
Number of years for stabilization	100		35	0.800
Number of replicates	100		45	0.835
Average number of eggs per clutch	121.7	Human take	0	0.000
Standard deviation of number of eggs per clutch	23.4		35	0.000
Pivotal temperature model	WPac		45	0.001
Habitat mean	0.587	Commercial take	0	0.000
Habitat quality minimum limit	0.05		35	0.000
Habitat quality maximum limit	0.95		45	0.050



Under current conditions, Olive ridley turtles are not able to withstand small rates of take as bycatch, likely given the low initial population size and low survival probability of the younger life stages. This model scenario (Lo1) includes a low level of adult mortality as suggested by bycatch data from longline and purse seine industries, and outputs suggest that this level of take is unsustainable for Olive ridleys in the Pacific. However, the challenge lies in understanding whether the proportion of adults is reflective of the number actually taken in fisheries. In this model the number of turtles being removed from the population is ~950. The WPFC purse seine bycatch rate has averaged ~50 turtles since 2003 (Peatman et al. 2018b), while the longline bycatch rate has averaged ~990 turtles (Peatman et al. 2018a). While these values are not absolute, and are rather more indicative of changes in magnitude of catch, there is substantial similarity between this model and current estimated bycatch rates, and thus this scenario is likely realistic for Olive ridley turtles in the western Pacific.

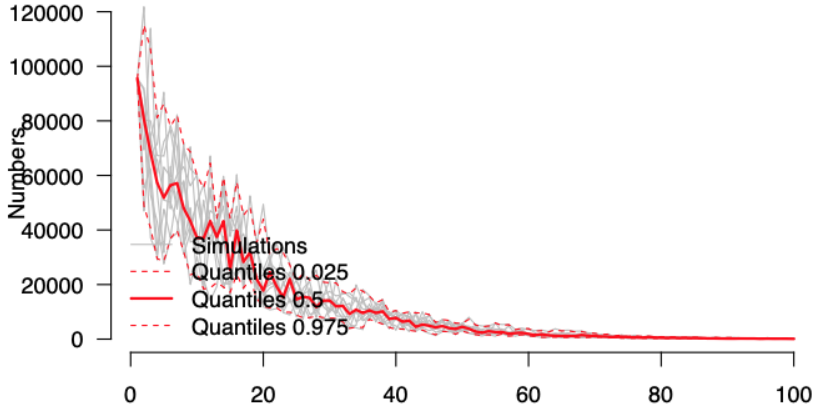
[2] https://www.stateoftheenvironment.des.qld.gov.au/biodiversity/management-responses/policy-and-programs/nest-to-ocean-program

Scenario Lo2 Increased bycatch in commercial fisheries				
Parameter	**Setting**	**Survivorship**	**SCL (cm)**	**s**
Initial population size	28,505		0	0.200
Number of years for stabilization	100	Natural	35	0.800
Number of replicates	100		45	0.835
Average number of eggs per clutch	121.7		0	0.000
Standard deviation of number of eggs per clutch	23.4	Human take	35	0.000
Pivotal temperature model	WPac		45	0.003
Habitat mean	0.587		0	0.001
Habitat quality minimum limit	0.05	Commercial take	35	0.003
Habitat quality maximum limit	0.95		45	0.005



In this model Lo2, bycatch rates were increased to demonstrate the impact of unregulated commercial fisheries. With higher levels of immature and adult bycatch, the decline of Olive ridley turtles is accelerated, with negligible fluctuation in model outputs. This species is not able to withstand a modest amount of egg mortality on the beaches, and is not in a position to withstand substantive losses in the form of bycatch.

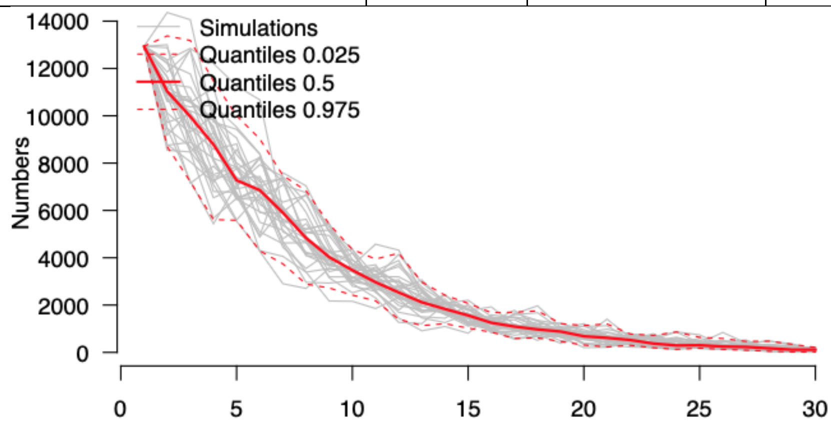
Scenario Lo3 Low level direct take and bycatch with increased early stage natural survivorship				
Parameter	**Setting**	**Survivorship**	**SCL (cm)**	**s**
Initial population size	94,490		0	0.300
Number of years for stabilization	100	Natural	35	0.800
Number of replicates	100		45	0.835
Average number of eggs per clutch	121.7		0	0.000
Standard deviation of number of eggs per clutch	23.4	Human take	35	0.000
Pivotal temperature model	WPac		45	0.005
Habitat mean	0.587		0	0.000
Habitat quality minimum limit	0.05	Commercial take	35	0.000
Habitat quality maximum limit	0.95		45	0.005



Model Lo3 used a low level of human take of adults and a low level of bycatch of adults in commercial fisheries, but with an increase in the natural early survivorship (note the increase from 0.200 to 0.300). This combination of a slightly higher natural younger years survival with a slightly higher human take of adults also leads to an overall decline in the species, demonstrating that Olive ridley turtles are not able to withstand small levels of take as bycatch in commercial fisheries even when nest survival is increased slightly.

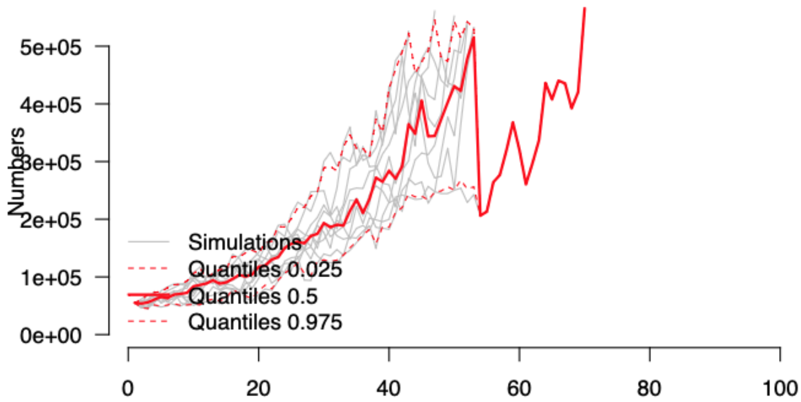
Scenario Lo4 Increased natural early stage survivorship with increased bycatch of adults

Parameter	**Setting**	**Survivorship**	**SCL (cm)**	**s**
Initial population size	12,937	Natural	0	0.300
Number of years for stabilization	100		35	0.800
Number of replicates	100		45	0.848
Average number of eggs per clutch	121.7	Human take	0	0.001
Standard deviation of number of eggs per clutch	23.4		35	0.001
Pivotal temperature model	WPac		45	0.001
Habitat mean	0.587	Commercial take	0	0.001
Habitat quality minimum limit	0.05		35	0.001
Habitat quality maximum limit	0.95		45	0.010



Even with a slightly higher natural younger years survival, when combined with higher adult take (e.g. bycatch) Olive ridley turtles are not able to survive, likely given the low initial population size and low survival probability of the youngest life stages.

Scenario Lo5 Reduction in all human induced mortality to zero				
Parameter	**Setting**	**Survivorship**	**SCL (cm)**	**s**
Initial population size	54,970	Natural	0	0.250
Number of years for stabilization	100		35	0.800
Number of replicates	100		45	0.835
Average number of eggs per clutch	121.7	Human take	0	0.000
Standard deviation of number of eggs per clutch	23.4		35	0.000
Pivotal temperature model	WPac		45	0.000
Habitat mean	0.587	Commercial take	0	0.000
Habitat quality minimum limit	0.05		35	0.000
Habitat quality maximum limit	0.95		45	0.000



Only if all instances of human mortality are reduced to zero the population is able to recover and grow (note all levels or mortality are set to 0.000). Given the unlikely scenario whereby all bycatch and beach loss of eggs are totally eliminated, this population is at risk from even low to modest bycatch in commercial fisheries, and of loss of eggs to predators on beaches.

2.10 Olive ridley turtle conclusion & summary (West Pacific RMU).

There are no realistic scenarios under the current simulation designs that lead to Olive ridley survival, unless with the removal of egg depredation and fisheries bycatch. Even increasing early-years survival, which had a positive impact on leatherback turtles, did not impact the overall survival of the population under any of the scenarios, likely due to lower population size and reproductive output.

Given the low numbers of Olive ridley turtles in the western Pacific, the loss of eggs to predators and the proportional impact of bycatch in commercial fisheries, the species is assessed as Critically Endangered.

Olive ridley turtle after nesting

3.0 Management Input Options – Pacific Sea Turtles

The following sections outline varying opportunities for management that will have high likelihoods of reversing declines and maintaining sea turtle populations in the Pacific for the SPREP region that fulfil their evolutionary ecological roles. While non-exhaustive in their application or scope, these immediate remedies are likely to positively impact sea turtle extinction risk outlooks across Pacific Island Countries and Territories. To recap, leatherback turtles are assessed under this process as being Critically Endangered in the western Pacific Region, along with hawksbill turtles, Olive ridley turtles. Loggerhead turtles are assessed as Endangered. Only the green turtle appears robust, being assessed as Least Concern for Australia, but elsewhere these are assessed as Endangered.

vTurtle Modelled Risk Assessment Ranking for the SPREP Region			IUCN Assessments	
	Regional	National	Global	Regional
Green turtle *Chelonia mydas*	Least Concern	Australia: Least Concern Other countries and territories: Endangered	Endangered	Endangered
Hawksbill turtle *Eretmochelys imbricata*	Critically Endangered	All countries: Critically Endangered	Critically Endangered	n/a
Leatherback turtle *Dermochelys coriacea*	Critically Endangered	All countries: Critically Endangered	Critically Endangered	Critically Endangered
Loggerhead turtle *Caretta caretta*	Endangered	All countries: Endangered	Vulnerable	Critically Endangered
Olive Ridley turtle *Lepidochelys olivacea*	Critically Endangered	Australia: Critically Endangered	Vulnerable	n/a

3.1 Addressing mortality of eggs and hatchlings on nesting beaches

As has been demonstrated in several of the model runs, increasing early-stage survival for depleted populations is one of the more impactful ways of reversing declines and increasing population growth. In practical terms on Pacific Island and Territory beaches this means increasing the survival of eggs and hatchlings. While it is clear little can be done for the hatchlings once they leave the beach, the following measures can be taken to increase survival of eggs and hatchlings on beaches:

a. Minimise and where possible eliminate egg predation by feral or domestic animals (pigs, dogs, rats, monitor lizards, etc.). Predator control measures can be challenging to implement, but recent successes demonstrate how effective they can be (e.g. removal of monitor lizards on Loosiep Island, FSM; Smith 2024; control of feral pig nest depredation; Karam 2014);

a. Reduce the collection of eggs for consumption. While adult and large subadult turtles have high biological values because they can reproduce (Crouse et al. 1987), the protection of eggs on beaches is also of paramount importance in population recovery (Heppell 1997, Santindrian-Tomillo et al. 2008);

b. Only where necessary or as a last resort, and with adequate training, move eggs to hatcheries to protect them from illegal poaching, predators, erosion or storm damage, and follow up with natural biology-mimicking hatchling release programmes. For example, temperature regimes need to be kept similar, so that if eggs are moved from a shaded location, they should be moved to a hatchery under shading, but if eggs are transferred from an exposed site they should be transferred to an exposed hatchery site. Hatcheries should not be used as a catch-all excuse for moving turtle eggs (Mortimer 1999) – they should be used prudently as and how varying situations dictate;

c. As culturally appropriate, work with local communities to minimise collection of eggs on nesting beaches, or establish levels of egg harvest that are sustainable. Given the critical status of hawksbill turtles in the region, temporary elimination of egg harvests should be considered; and

d. Implement monitoring and enforcement programmes in areas where egg collection is illegal.

3.2 Addressing incidental capture of all age classes in commercial and artisanal fisheries

Addressing impacts of fisheries bycatch is possibly the most complex aspect of sea turtle conservation across the world. There are some technological solutions (e.g. Turtle Excluder Devices in trawl fisheries, NRC 1990; LEDs in gillnets, Ortiz et al. 2016), but in truth, the problem is more sociological than technological (see, for example, Senko & Nalovic 2021). The challenge lies in getting fishers to *want* to implement bycatch reduction measures. Compounding this is that in many parts of the Pacific, local communities target sea turtles deliberately (this is then not really considered as bycatch), and others may not target turtles but may keep them if they catch them accidentally.

Reducing targeted community catch of turtles may require governance under traditional / customary practices, and at the practical level, requires changes to how fishing nets are deployed and tended to, or where one fishes - both of which have been optimised over hundreds of generations because this is the easiest way to catch fish, or because these are the best places to catch fish. Commercial fisheries do not target sea turtles. They target fish, squid, and other species, but sea turtles are accidentally caught in commercial operations. Changing commercial operations to be more turtle friendly invariably costs more, or takes more time, or is more inconvenient. Ultimately there is a need to fish more sustainably, recognising that the entire ecosystem and species are interconnected. Improved governance and enforcement is also required – whether at the traditional or government levels, via the adoption of improved monitoring technologies, use of mitigation technology and enhanced traditional governance and monitoring.

Another aspect of addressing bycatch is that the interventions are expensive, and take a long time for implementation. While Turtle Excluder Devices are effective at mitigating sea turtle bycatch in trawl fisheries, in Australia it took over 15 years before there was a move to make them compulsory. Similar technological solutions do not readily exist for artisanal gillnet fisheries. In most cases there is a need for continued training, reminders, refreshers, enforcement, guidance, financial assistance, and a suite of other support needs. Stronger and more effective management of bycatch in fisheries remains a key requirement however, and the adoption of modern technological solutions may provide advances where things might have stalled in the past. The use of electronic monitoring, vessel tracking devices, and enhanced satellite surveillance of vessel movements may all contribute to more effective management.

Regional Fishery Management Organisations promote bycatch reduction in commercial fisheries but the low levels of observer coverage mean that estimates of bycatch are often unreliable in terms of magnitude, but are useful to detect trends in bycatch rates. Artisanal fisheries are far harder to manage. Catches are rarely documented, the number of fishing boats can number in the thousands, and given the way they disperse to fish it is hard to monitor bycatch or enforce management measures (Lewison et al. 2011). In the PICT region, SPC developed with members and partners a regional strategy for implementing community-based fisheries management in coastal areas of Pacific Island countries and territories (SPC 2021). This strategy details approaches related to protected areas, fishery management areas, capacity, legal frameworks across the region, but there is still a need to better document and track community-based fishery management interventions, and clarify the synergies between biodiversity conservation, fisheries management and livelihood aims (Govan & Lalavanua 2022).

Below are some key ingredients to bycatch reduction programmes that are essential if sea turtles are to benefit. There needs to be:

- Long-term commitment from industry/community;
- A financing plan to keep the efforts running year after year;
- Local community / fisher buy-in from the very start, and livelihood considerations need to be a key aspect of the bycatch reduction efforts;
- Government buy-in from the very beginning, with a view to enforcement of eventual solutions; This includes willingness to agree to improved Conservation and Management Measures for turtles in RFMOs

- Acceptance and willingness to choose alternate bycatch reduction efforts, and to experiment until finding the compromise that works for industry, local communities and for sea turtles;
- Ample consultation / dialogue so that fisher voices are part of the solution;
- Bycatch reduction efforts that balance fisher livelihoods and sea turtle conservation; and
- Consideration of the cost of technological solutions to ensure it is not a detriment to the long-term financial viability of the fishery.

3.3 Addressing the loss of nesting females on nesting beaches

As noted above, female turtles of reproductive age are extremely valuable biologically, as they have survived several decades of threats and are in a position to lay eggs (Crouse et al. 1987). While it can be argued that all life stages are of value, adult female turtles are the proverbial *golden goose*. They lay the eggs that constitute future generations. Acknowledging that traditional take is permissible under varying conditions in some locations in the Pacific region, the collection of adult turtles – particularly in species and populations that are extremely vulnerable – should be discussed and considered amongst local communities with a view to determining suitable courses of action that preserve local traditions while also improving the conservation status of sea turtles. For example, in Palau, seasons and temporary closures (known locally as a *bul*) for sea turtles have been implemented (Putney 2008). In Fiji, a 10 year moratorium was put in place. Moratoriums on turtle collection are often in perfect alignment with traditional management measures in the Pacific region, so long as these are declared and enforced via customary processes. Given the critical status of hawksbill turtles in the region, PICTs might want to consider implementing temporary traditional or legal bans on hawksbill turtle harvests until such a time as populations recover.

As seen in a number of the model scenarios presented above, collection of even a few adult turtles in small populations and from the most endangered species (e.g. leatherback and hawksbill turtles) can lead to population extirpation, at least at the local level. The biology and ecology of sea turtles means that turtles migrate large distances between nesting and feeding areas, and thus impacts to sea turtles in one 'country' could mean losses of sea turtles in their 'home country'. Regional management actions – or at least actions that are commensurate with those of other PICTs, implemented at national and community levels, are required to adequately address losses of sea turtles and population declines.

The number of turtles caught in commercial fisheries in some cases is already of grave concern, and these fisheries do not distinguish by size, gender or reproductive states. That is, any take of adult females in commercial fisheries is complex to address (see above section). However, capturing and retaining large female turtles in artisanal fisheries that are likely in reproductive condition, or the collection of female turtles nesting on beaches, needs to be extremely carefully managed. In those populations that are critically endangered (and/or genetically distinct from another and therefore more vulnerable to local extirpation), it is suggested that some form of traditional management will be required. This needs to be complemented by the introduction and enforcement of all applicable legal measures as rapidly as reasonably possible to prevent continued declines in turtle numbers.

3.4 Addressing local traditional and consumptive use of sea turtles and their products

Extensive collection of hawksbill sea turtles for the curio trade has occurred in the past (e.g. Canin 1991, Miller et al., 2019), and persists to this day with substantial take in the Pacific region (e.g. LaCasella et al. 2021, Madden Hof et al., 2022, Opu 2018, Vuto et al, 2019). Green turtles were taken for subsistence in the past in Hawaii (Van Houtan & Kittinger 2018), and continue to be taken traditionally in the Ogasawara Islands, Japan (Kondo et al. 2017), in Papua New Guinea (Opu 2018), in the Solomon Islands (Vuto et al. 2019, Hamilton et al 2024), Fiji (Piovano et al. 2019), and numerous other countries and island territories (Maison et al. 2010).

The social norms, cultures and traditions surrounding these harvests are not part of this discussion, but rather the *quantum* – or the number of turtles in question. It is impossible to state at this juncture whether reducing the number of turtles collected for traditional use or consumption should be ten, or seventy-one, or thirty-seven. Nobody has ever experimented with sustainable take and decided on a quantum. Rather, what is suggested is that traditional sea turtle use be managed – within the reaches of traditional roles, so that sea turtle populations have the potential to recover. As noted in the case of green sea turtles, they are not Endangered at a regional level, and populations in the Pacific region number in the tens of thousands. However, the numbers nesting in any one location in one of the PICTs might only be in the tens, and recognising this distinction is important. If we remove five turtles from a nesting population of 500 this is likely extremely manageable from a turtle survival point of view (0.01%). However, if we remove five turtles from a nesting population of 10 (50%) this will lead to population declines and eventual loss.

What is suggested here is as follows: In those populations where numbers of nesters (or turtles in general) are low, and traditional take is ongoing, it would be prudent to find a way to reduce the traditional take by a set proportion. If – over time – numbers of turtles increase, then the proportion of turtles taken traditionally may be increased. If the number of turtles continues to decline, then the proportion of turtles allowed as take likely needs to be lowered. For example, a community decides that 10 turtles per year is the limit. After 10 to 20 years the numbers of nesters continually increase. In this case the community may wish to increase the take to 12 or 15 turtles. But if the number decreases over time, then the traditional take limit is too high. How long is long enough to wait? At least 10 years, and preferably 15 to 20.

The Recovery Plan for Marine Turtles in Australia indicates that protecting a minimum of 70% of turtle nests and their habitats is essential for population maintenance – and possibly recovery. Where bycatch rates exceed the balance of 30%, countries need to effectively manage commercial fisheries and traditional take so that the *total* impacts fall below the 30% threshold. As noted above, it is problematic to determine what *proportion* of sea turtle populations are lost to commercial fisheries and human take, but in those populations that continue to decline (leatherbacks, hawksbills, loggerheads and Olive ridleys) it is suggested that all take rates need to drop - bycatch rates need to drop by at least 50%, and human take also needs to drop by 30% to 50% to reverse these declining trends.

3.5 Improved data collection

While not specifically actions that can be taken by local communities or individual PICTs, there is a need for better information on take - targeted and bycatch to be in a position to better understand these large-scale impacts. As noted for the commercial fisheries data, the low level of observer coverage means that there are large confidence intervals around the estimates or bycatch for each sea turtle species, and as noted by Peatman et al. (2018a) the bycatch rates are more reflective of the trend in bycatch rather than the quantum itself. To aid in the development of more robust estimates, the following improvements are suggested:

- Expand observer coverage on fishing vessels, either via human or electronic monitoring;
- Enhance voluntary reporting schemes and determine accuracy of voluntary reporting via secondary monitoring means;
- Gather finer-scale geographic resolution data to help with model accuracy;
- Collect and report carapace length data for as many bycaught turtles as possible;
- Improve post-release survival estimates by species caught in commercial fisheries;
- Develop community reporting schemes and develop accurate estimates of traditional take at national levels; and
- Take genetic samples to understand which nesting populations remain, and in foraging grounds (e.g. through community harvest, bycatch) how these populations are connected, and which populations are being impacted the most.


Green turtle returns to the sea after nesting

4.0 Literature Cited

Avens L, LR Goshe, GR Zug, GH Balazs, SR Benson & H Harris. 2020. Regional comparison of leatherback sea turtle maturation attributes and reproductive longevity. Marine Biology 167(1): 4.

Balazs GH & M Chaloupka. 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

Bardach J & P Ridings. 1985. Pacific Tuna: Biology, Economics, and Politics. In Ocean Yearbook 5. (E Mann Borgese & N Ginsburg, eds.). University of Chicago Press pp. 27-59.

Brewer D, D Heales, D Milton, Q Dell, G Fry, B Venables & P Jones. 2006. The impact of turtle excluder devices and bycatch reduction devices on diverse tropical marine communities in Australia's northern prawn fishery. Fisheries Research. 81. 176-188. 10.1016/j.fishres.2006.07.009.

Briane JP, P Rivalan, & M Girondot. 2007. The inverse problem applied to the observed clutch frequency of leatherback turtles from Yalimapo Beach, French Guiana. Chelonian Conservation & Biology 6(1): 63-69.

Canin J. 1991. International trade aspects of the Japanese Hawksbill shell ('Bekko') industry. Marine Turtle Newsletter 54: 17-21.

Chaloupka M, P Dutton & H Nakano. 2004. Status of sea turtle stocks in the Pacific. Papers Presented at the Expert Consultation on Interactions Between Sea Turtles and Fisheries Within an Ecosystem Context: Rome, 9-12 March 2004 738: 135.

Chaloupka M, KA Bjorndal, GH Balazs, AB Bolten, LM Ehrhart, CJ Limpus, H Suganuma, S Troëng, & M Yamaguchi. 2007. Encouraging outlook for recovery of a once severely exploited marine megaherbivore. Global Ecology & Biogeography 17: 297–304.

Chevallier D, B Mourrain & M Girondot. 2020. Modelling leatherback biphasic indeterminate growth using a modified Gompertz equation. Ecological Modelling 426: 109037.

Cleland J. 2013. World population growth; past, present and future. Environmental & Resource Economics, 55: 543-554.

Crouse DT, LB Crowder & H Caswell. 1987. A stage-based population model for loggerhead sea turtles and implications for conservation. Ecology 689(5): 1412-1423.

de Paz N, JC Reyes, M Ormeño, HA Anchante & AJ Altamirano. 2006. Immature leatherback mortality in coastal gillnet fisheries off San Andres, Southern Peru. In Book of Abstracts (p. 259).

Dodge KL, S Landry, B Lynch, CJ Innis, K Sampson, D Sandilands & B Sharp. 2022. Disentanglement network data to characterize leatherback sea turtle Dermochelys coriacea bycatch in fixed-gear fisheries. Endangered Species Research 47: 155-170.

Donoso M & PH Dutton. 2010. Sea turtle bycatch in the Chilean pelagic longline fishery in the southeastern Pacific: opportunities for conservation. Biological Conservation, 143(11): 2672-2684.

Dunn MR, B Finucci, MH Pinkerton, P Sutton & CAJ Duffy. 2023. Increased captures of the critically endangered leatherback turtle (*Dermochelys coriacea*) around New Zealand: the contribution of warming seas and fisher behaviour. Frontiers in Marine Science 10:1170632. doi: 10.3389/fmars.2023.1170632

Dutton PH, C Hitipeuw, M Zein, SR Benson, G Petro, J Pita, V Rei, L Ambio & J Bakarbessy. 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.

Fuentes MMPB, DA Pike, A DiMatteo & BP Wallace. 2013. Resilience of marine turtle regional management units to climate change. Global Change Biology 19: 1399–1406.

Girondot M, P Rivalan, R Wongsopawiro, JP Briane, V Hulin, S Caut, E Guirlet & MH Godfrey. 2006. Phenology of marine turtle nesting revealed by statistical model of the nesting season. BMC ecology 6: 1-12.

Girondot M & Y Kaska. 2014. A model to predict the thermal reaction norm for the embryo growth rate from field data. Journal of thermal biology, 45: 96-102.

Girondot M, B Mourrain, D Chevallier & MH Godfrey. 2021. Maturity of a giant: age and size reaction norm for sexual maturity for Atlantic leatherback turtles. Marine Ecology 42(5), e12631.

Girondot M, CJ Krueger, C Cléomène, Z Tran, D Chevallier & FJ Janzen. 2024. Developmental thermal reaction norms of leatherback marine turtles at nesting beaches. Animals, 14(21), p.3050.

Govan H & L Watisoni III. 2022. Status of community-based fisheries management in Pacific Islands countries and territories: survey report. Pacific Community, Noumea, New Caledonia. ISBN: 978-982-00-1477-0; 70 pp.

Hamann M, CJ Limpus & M Read. 2007. Vulnerability of marine reptiles in the Great Barrier Reef to climate change. In: Climate change and the Great Barrier Reef: a vulnerability assessment. J Johnson & P Marshall (eds.). Great Barrier Reef Marine Park Authority and Australian Greenhouse Office, Townsville, pp 465−496.

Hamilton RJ, T Bird, C Gereniu, J Pita, P Ramohia, RWalter, C Goerlich & CJ Limpus. 2015. Solomon Islands largest hawksbill turtle rookery shows signs of recovery after 150 years of excessive exploitation. PLoS One, 10(4), p.e0121435.

Hamilton R, S Vuto, C Brown, P Waldie, J Pita, R Babaua, R Masu, N Peterson, C Madden & CJ Limpus. 2024. Freedivers harvest thousands of sea turtles a year in the Solomon Islands. Aquatic Conservation: Marine and Freshwater Ecosystems, 34(1), p.e4050.

Hawkes LA, AC Broderick, MH Godfrey & BJ Godley. 2007. Investigating the potential impacts of climate change on a marine turtle population. Global Change Biology 13: 923−932.

Heppell, SS. 1997. On the importance of eggs. Marine Turtle Newsletter, 76:6-8

Heppell SS, CJ Limpus, DT Crouse, NB Frazer & LB Crowder. 1996. Population model analysis for the loggerhead sea turtle, *Caretta caretta*, in Queensland. Wildlife Research (Aust), 23, 143-159.

Jones TT, BL Bostrom, MD Hastings, KS Van Houtan, D. Pauly & DR Jones. 2012. Resource Requirements of the Pacific Leatherback Turtle Population. PLoS ONE 7(10): e45447. doi: 10.1371/journal.pone.0045447

Karam J. 2014. Western Cape Turtle Threat Abatement Alliance: Indigenous Rangers Driving Collaboration for Marine Turtle Protection in Cape York. Proceedings of the Second Australian and Second Western Australian Marine Turtle Symposia Perth 25-27 August 2014, pp 51.

Kinan I & P Dalzell.2005. Sea Turtles As a Flagship Species: Different perspectives create conflicts in the Pacific Islands. MAST 2005, 3(2) & 4(1): 195–212.

Kondo S, Y Morimoto, T Sato & H Suganuma. 2017 factors affecting the long-term population dynamics of green turtles (*Chelonia mydas*) in Ogasawara, Japan: influence of natural and artificial production of hatchlings and harvest pressure. Chelonian Conservation and Biology 16:83-92.

Lewison RL, LB Crowder, AJ Read & SA Freeman. 2004. Understanding impacts of fisheries bycatch on marine megafauna. Trends in ecology & evolution. 19(11): 598-604.

Lewison RL, CU Soykan, T Cox, H Peckham, N Pilcher, N LeBoeuf, S McDonald, J Moore, C Safina, & LB Crowder. 2011. Ingredients for addressing the challenges of fisheries bycatch. Bulletin of Marine Science, 87(2): 235-250.

LaCasella EL, MP Jensen, CA Madden Hof, IP Bell, A Frey & PH Dutton, 2021. Mitochondrial DNA profiling to combat the illegal trade in tortoiseshell products. Frontiers in Marine Science 7: 595853.

Limpus CJ, CJ Parmenter & M Chaloupka. 2013. Monitoring of Coastal Sea Turtles: Gap Analysis 1. Loggerhead turtles, *Caretta caretta*, in the Port Curtis and Port Alma Region. Report produced for the Ecosystem Research and Monitoring Program Advisory Panel as part of Gladstone Ports Corporation's Ecosystem Research and Monitoring Program.

Maison KA, IK Kelly & KP Frutchey. 2010. Green turtle nesting sites and sea turtle legislation throughout Oceania. NOAA Technical Memorandum NMFS-F/SPO-110. 52pp.

Mangel JC, S Pingo, A Jimenez, PD Doherty & J Alfaro-Shigueto. 2024. Post-release movements of leatherback turtles captured by the Peruvian small-scale driftnet fishery: insights from satellite telemetry. Endangered Species Research 54: 261-275.

Morales-Mérida BA, NJ Pilcher & M Girondot. 2024. How old Is a turtle? Challenges in interpreting age information in sea turtles. Ecologies 5(4): 502-511.

Mortimer JA. 1999. Reducing threats to eggs and hatchlings: hatcheries. I Research and Management Techniques for the Conservation of Sea Turtles (KL Eckert, KA Bjorndal, FA Abreu-Grobois & M Donnelly (eds). International Union for Conservation of Nature / Species Survival Commission Marine Turtle Specialist Group Publication No. 4. pp. 174–178

National Research Council. 1990. The Decline of the Sea Turtles. National Academy of Science Press, Washington, D.C. 259 p.

Nordberg EJ, S Macdonald, G Zimny, A Hoskins, A Zimny, B Somaweera, J Ferguson & J Perry. 2019. An evaluation of nest predator impacts and the efficacy of plastic meshing on marine turtle nests on the western Cape York Peninsula, Australia. Biological Conservation, 238, p.108201.

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 N, JC Mangel, J Wang, J Alfaro-Shigueto, S Pingo, A Jimenez, T Suarez, Y Swimmer, F Carvalho & BJ Godley. 2016. Reducing green turtle bycatch in small-scale fisheries using illuminated gillnets: the cost of saving a sea turtle. Marine Ecology Progress Series, 545: 251-259.

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

Peatman T, V Allain, S Caillot, T Park, P Williams, I Tuiloma, A Panizza, S Fukofuka & N Smith. 2018b. 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 & S Nicol. 2020. Updated longline bycatch estimates in the WCPO. Scientific Committee Sixteenth Regular Session WCPFC-SC16-2020/ST-IP-11. 37 pp.

Peatman T, V Allain, L Bell, B Muller, A Panizza, NB Phillip, G Pilling & S Nicol. 2023. Estimating trends and magnitudes of bycatch in the tuna fisheries of the Western and Central Pacific Ocean. Fish and Fisheries 24(5): 812-828.

Pike DA. 2013. Climate influences the global distribution of sea turtle nesting. Glob Ecology & Biogeography 22: 555−566.

Pilcher NJ. 2013. Community-based conservation of leatherback turtles along the Huon coast, Papua New Guinea 2012-2013. Project 12-turtle-003 Final Report. Western Pacific Regional Fishery Management Council, Honolulu, HI. 14pp.

Pilcher NJ & C Robins. 2010 Trials and tribulations of Turtle Excluder Devices. SWOT Report 5, pp. 18-21.

Pilcher NJ & M Chaloupka. 2013. Using community-based monitoring to estimate demographic parameters for a remote nesting population of the Critically Endangered leatherback turtle. Endangered Species Research 20: 49-57.

Piovano S, A Batibasaga, A Ciriyawa, E LaCasella & PH Dutton. 2019. Mixed stock analysis of juvenile green turtles aggregating at two foraging grounds in Fiji reveals major contribution from the American Samoa Management Unit. Scientific Reports 9(1): 3150.

Putney R. 2008. Customary marine tenure and traditional ecological knowledge in Palau. San Jose State University Master's Theses. 3493. doi: https://doi.org/10.31979/etd.32rm-qsxp

Santidrián Tomillo P. 2022. When population-advantageous primary sex ratios are female-biased: changing concepts to facilitate climate change management in sea turtles. Climatic Change 175(3): 15.

Santidrian-Tomillo P, VS Saba, R Piedra, FV Paladino & JR Spotila. 2008. Effects of illegal harvest of eggs on the population decline of leatherback turtles in Las Baulas Marine National Park, Costa Rica. Conservation biology 22(5) 1216-1224.

Senko JF & MA Nalovic. 2021. Addressing Sea Turtle Bycatch in Developing Countries: A Global Challenge That Requires Adaptive Solutions for the 21st Century. *In* Sea Turtle Research and Conservation, Academic Press, pp. 151-165.

Smith C. 2024. World-first monitor lizard eradication program aims to protect endangered turtles nesting on Micronesian island. ABC Science News; October 18, 2024. https://www.abc.net.au/news/science/2024-10-19/monitor-lizard-eradication-micronesia-turtle-island-loosiep/104375496

SPC 2021. Pacific Framework for Action on Scaling up Community-based Fisheries Management: 2021-2025. Noumea, New Caledonia: Pacific Community. 22 p. https://purl.org/spc/digilib/doc/yr5yv

Stewart KR, EL LaCasella, SE. Roden, MP. Jensen, LW Stokes, SP Epperly & PH. Dutton. 2016. Nesting population origins of leatherback turtles caught as bycatch in the US pelagic longline fishery. Ecosphere 7(3): e01272.

Tapilatu RF, PH Dutton, M Tiwari, T Wibbels, HV Ferdinandus, WG Iwanggin & BH Nugroho. 2013. Long-term decline of the western Pacific leatherback, *Dermochelys coriacea*: a globally important sea turtle population. Ecosphere, 4(2): 1-15.

Van Houtan KS & JN Kittinger. 2014. Historical commercial exploitation and the current status of Hawaiian green turtles. Biological Conservation 170: 20-27.

Velez-Zuazo X & S Kelez. 2010. Multiyear analysis of sea turtle bycatch by Peruvian longline fisheries: a genetic perspective. In Proceedings of the 30th Annual Symposium on Sea Turtle Biology and Conservation (2010): 24-30.

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

Wallace BP & TT Jones. 2015. Leatherback turtle physiological ecology: implications for bioenergetics and population dynamics. In: The leatherback turtle: biology and conservation. JR Spotila & PS Tomillo, eds. Johns Hopkins University Press, Baltimore, pp 149–161

Wallace BP, M Tiwari & M Girondot. 2013. *Dermochelys coriacea*. IUCN Red List of Threatened Species. Version 2013.2.

Wallace BP, ZA Posnik, BJ Hurley, AD DiMatteo AD, A Bandimere, I Rodriguez, SM Maxwell, L Meyer, H Brenner, MP Jensen, E LaCasella, BM Shamblin, FA Abreu-Grobois, KR Stewart, PH Dutton, H Barrios-Garrido, M Dalleau, F Dell'amico, KL Eckert, NN FitzSimmons, M Garcia-Cruz, GC Hays, S Kelez, CJ Lagueux, CA Madden Hof, A Marco, SLT Martins, A Mobaraki, JA Mortimer, R Nel, AD Phillott, NJ Pilcher, NF Putman, AF Rees, JM Rguez-Baron, JA Seminoff, A Swaminathan, O Turkozan, SM Vargas, PD Vernet, S Vilaça, SD Whiting, BJ Hutchinson, P Casale & RB Mast. 2023. Marine turtle regional management units 2.0: an updated framework for conservation and research of wide-ranging megafauna species. Endangered Species Research 52:209-223. https://doi.org/10.3354/esr01243.

Witt MJ, LA Hawkes, MH Godfrey, NJ Godley & AC, Broderick. 2010. Predicting the impacts of climate change on a globally distributed species: the case of the loggerhead turtle. Journal of Experimental Biology 213: 901−911.

Zug GR & JF Parham. 1996. Age and growth in leatherback turtles, *Dermochelys coriacea* (Testudines: Dermochelyidae): a skeletochronological analysis. Chelonian Conservation & Biology 2: 244-249.

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