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Update to Assessing the Population-level Impacts of North Pacific Loggerhead and Western Pacific Leatherback Turtle Interactions: Inclusion of the Hawaii-based Deep-set and American Samoa-based Longline Fisheries

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List of Acronyms

ANE = adult nester equivalent

ASLL = American Samoa-based longline

DPS = Distinct Population Segment

DSLL = Hawaii-based deep-set longline

ESA = Endangered Species Act

ITS = Incidental Take Statement

NP = North Pacific (loggerhead turtles)

PIRO = Pacific Islands Regional Office

PIFSC = Pacific Islands Fisheries Science Center

PVA = population viability analysis

MVN = multivariate normal (distribution)

SSLL = Hawaii-based shallow-set longline

SCL = straight carapace length

WP = Western Pacific (leatherback turtles)

Executive Summary

This NOAA Technical Memorandum (TM) is intended to be read as part II of NOAA Technical Memorandum NMFS-PIFSC-95, February 2020, *Assessing the Population-level Impacts of North Pacific Loggerhead and Western Pacific Leatherback Turtle Interactions in the Hawaii-based Shallow-set Longline Fishery* (Martin et al. 2020). The overarching methods, marine turtle demographic data, and estimates on abundance and trends in this TM were used and developed for the Hawaii-based shallow-set longline fishery. Herein, we adapted the models to apply them to the Hawaii-based and American Samoa-based deep-set longline fisheries (DSLL and ASLL fisheries, respectively). This TM is largely an update to the methods to address partial observer coverage in the deep-set longline fisheries, as well as a presentation of new results. The two TMs should be read together and viewed in unison to understand the impacts of the shallow and deep-set longline fisheries in Hawaii and American Samoa.

As part of the fishery impact analysis, we estimated trends and abundance for the North Pacific (NP) loggerhead turtle (*Caretta caretta*) Distinct Population Segment (DPS) and the western Pacific (WP) leatherback turtle (*Dermochelys coriacea*) nesting populations. Both populations are listed as Endangered under the Endangered Species Act (ESA), NP loggerheads as a DPS and WP leatherbacks as a global species. The median population growth rate was positive for NP loggerheads (2.3% annually; 95% CI, −11.0% to 15.7%) and negative for WP leatherbacks (−6.1% annually; 95% CI, −24.0% to 12.3%). Current abundance for NP loggerheads was 4,538 total nesters (95% CI, 4,077 to 5,064 total nesters) for the three index beaches in Japan, which represent approximately 52% of all nesting individuals. Current abundance for WP leatherbacks was 787 total nesters (95% CI, 659 to 939 total nesters) for the two index beaches in Indonesia, which represent approximately 75% of all nesters. Importantly, for reasons discussed in Martin et al. (2020), these population growth rates better reflect long-term annual trends in the number of nesters rather than true population growth rates.

Impacts of the DSLL and ASLL fisheries were assessed in both historical and future contexts. The historical analysis involved removing the effect of historical take by an individual fishery (i.e., by adding back adult nester equivalents to the population); however, there was no difference between the trends for the “true” state and fishery-adjusted scenarios for either species. Population viability analysis (PVA) projections out 100 years in the future suggested a 100% chance of leatherbacks falling below 50% of their current abundance, with a mean of 13 years to reach that threshold, and a 32% chance of loggerheads falling below the 50% abundance threshold, with a mean of 24 years to reach that threshold. At 10 years in the future, the chance of falling below the 50% abundance threshold was 39% for leatherbacks and 9% for loggerheads. There were almost no changes to the probabilities of falling below abundance thresholds (50%, 25%, and 12.5% of current abundance) when comparing “no take” and “take” scenarios for the future for either the DSLL or ASLL fisheries. The one exception was that for leatherbacks, the difference between the “no take” and “take” scenarios became apparent after 2060 and the projection suggested the population would go extinct roughly 20 years sooner in the “take” scenario than in the “no take” scenario (around 2095 vs. 2115). It is important to note that this difference between the scenarios occurs when the population falls below 20 adult nesters. However, there is little difference between the two scenarios regarding when the population will reach 20 turtles. In the 10-year projection, which is perhaps more biologically

relevant to use for impact assessments, there was no discernible difference between the “no take” and “take” scenarios for either species.

Authors’ note

This NOAA Technical Memorandum (TM) is intended to be read as part II of NOAA Technical Memorandum NMFS-PIFSC-95, February 2020, Assessing the Population-level Impacts of North Pacific Loggerhead and Western Pacific Leatherback Turtle Interactions in the Hawaii-based Shallow-set Longline Fishery (Martin et al. 2020). The overarching methods, marine turtle demographic data, and estimates on abundance and trends used in this TM were used and developed for the Hawaii-based shallow-set longline fishery. Herein, we adapted the models to apply them to the Hawaii-based and American Samoa-based deep-set longline fisheries. This TM is largely an update to the methods to deal with partial observer coverage in the deep-set longline fisheries, as well as a presentation of new results. Much of the text is taken directly from TM NMFS-PIFSC-95 where no changes were made in the analyses. The two TMs should be read together and viewed in unison to understand the impacts of the shallow and deep-set longline fisheries in Hawaii and American Samoa.

R code for the original take model as applied to the Hawaii-based shallow-set longline fishery was provided as an appendix in NOAA-TM-NMFS-PIFSC-95. Modified R code specific to the analyses in this TM for the Hawaii-based deep-set longline fishery and the American Samoa-based longline fishery are publicly available on Open Science Framework: https://osf.io/bkmqv/.

Introduction

Background

Hawaii-based Deep-set Longline Fishery

The Hawaii-based deep-set longline (DSLL) fishery has historically interacted with five marine turtle species encompassing seven distinct population segments (DPSs), two species that are listed as global populations, and one species with a regionally recognized endangered population (Table 1). The DPSs include North Pacific loggerheads and East Pacific, Central North Pacific, East Indian-West Pacific, Southwest Pacific, Central West Pacific, and Central South Pacific green turtles. The global populations include leatherbacks and hawksbills. Olive ridleys are globally threatened but the Mexico population is endangered and grouped together with the remainder of the eastern Pacific population. The fishery has a rolling 3-year incidental take statement (ITS) covering marine turtle interactions as specified for each species and population in Table 1. If the amount of take specified in the ITS is exceeded, a reconsultation is required for the fishery under the Endangered Species Act (ESA). Since 2002, observer coverage and reporting of interactions have been managed separately for the DSLL and shallow-set longline (SSLL) fisheries. The DSLL fishery is observed at a minimum of 20% observer coverage (range since 2002: 20.1% to 26.1%). Observed interactions for loggerhead and leatherback turtles in the DSLL have ranged from 0 to 4 and 0 to 7 turtles per year, respectively. The interaction rates have ranged from 0.0 to 0.0006 loggerhead turtles per 1,000 hooks and from 0 to 0.0007 leatherback turtles per 1,000 hooks. As the DSLL fishery has partial observer coverage, the observed interactions are expanded to estimate total interactions for the year. Estimated total interactions with leatherbacks were highest in 2014 (38 turtles) and with loggerheads were highest in 2002 (17 turtles). These data can be found in the *Annual Stock Assessment and Fishery Evaluation Report for U.S. Pacific Island Pelagic Fisheries Ecosystem Plan* (WPRFMC 2019 and McCracken references therein).

American Samoa-based Deep-set Longline Fishery

The American Samoa-based deep-set longline (ASLL) fishery (note—there is no shallow-set fishery in American Samoa as there is in Hawaii) has historically interacted with five marine turtle species encompassing six DPSs, two species that are listed as global populations, and one species with a regionally recognized endangered population (Table 1). The DPSs include South Pacific loggerheads and East Pacific, East Indian-West Pacific, Southwest Pacific, Central West Pacific, and Central South Pacific green turtles. The global populations include leatherbacks and hawksbills. Olive ridleys are globally threatened but the Mexico population is endangered and grouped together with the remainder of the eastern Pacific population. The fishery has a rolling 3-year ITS covering marine turtle interactions as specified for each species and population in Table 1. If the amount of take specified in the ITS is exceeded, a reconsultation is required for the fishery under the ESA. The ASLL fishery has had observer coverage since 2006, with a coverage rate of close to 20% since 2010. The observer coverage ranged from 6.4% to 8.1% from 2006 through 2009 and from 15.7% to 33% since 2010. There have been no observed interactions of loggerheads in the ASLL. Observed interactions for leatherback turtles in the ASLL have ranged from 0 to 3 turtles per year since 2010. No leatherback interactions were observed from 2006 through 2009, during which the coverage was below 10%. The leatherback interaction rates since 2006 have ranged from 0 to 0.006 turtles per 1,000 hooks. As the ASLL

fishery has partial observer coverage, the yearly observed interactions are expanded to estimate total interactions for the year. For leatherbacks, estimated total interactions were highest in 2015 (22 turtles). These data can be found in WPRFMC (2019) and the McCracken references therein.

Table 1. Marine turtle populations and Incidental Take Statement (ITS) limits for the Hawaii-based deep-set (DSLL) longline and American Samoa-based longline (ASLL) fisheries. DPS = Distinct Population Segment. Endangered Species Act (ESA) status is threatened (T) or endangered (E). Each fishery has a rolling 3-year ITS, which authorizes a number of takes with corresponding mortalities in parentheses (note that fractional numbers are from modeled estimates). NLAA = not likely to adversely affect. ITS data from WPRFMC (2019).

Species	Population	ESA	DSLL ITS limits	ASLL ITS limits
Loggerhead (*Caretta caretta)*	North Pacific DPS	E	18(13)	NLAA
	South Pacific DPS	E	NLAA	6(3)
Leatherback (*Dermochelys coriacea)*	Global	E	72(27)	69(49)
Olive Ridley (*Lepidochelys olivacea)*	Mexico & Eastern Pacific Populations	E & T	144(134)	33(10)
	Western Pacific Population	T	42(40)	–
Green (*Chelonia mydas)*	East Pacific DPS	T	12(12)	7(6.48)
	Central North Pacific DPS	T	6(6)	–
	East Indian-West Pacific DPS	E	6(6)	1(1.08)
	Southwest Pacific DPS	T	6(6)	20(17.82)
	Central West Pacific DPS	E	3(3)	2(1.62)
	Central South Pacific DPS	E	3(3)	30(27)
Hawksbill (*Eretmochelys imbricata)*	Global	E	NLAA	6(3)

North Pacific Loggerhead Turtles

The following information is taken directly from the NOAA Technical Memorandum NMFS-PIFSC-95 (Martin et al. 2020). The North Pacific loggerhead population ("NP loggerheads" or "loggerheads" herein) is recognized as a DPS and listed as endangered under the ESA. NP loggerheads nest exclusively in Japan, in three regions or management units (MUs): mainland Japan, Yakushima, and Okinawa. After the turtles emerge as hatchlings on their natal beaches in Japan, they spend their developmental years (i.e., more than 20 years until sexual maturity (Tomaszewicz et al. 2015)) foraging in the North Pacific, with the oceanic central North Pacific and neritic zones off Baja California and California identified as two key developmental areas (Kobayashi et al. 2008; Polovina et al. 2006; Polovina et al. 2000). Once mature, they forage in oceanic or neritic waters closer to Japan in between breeding seasons (Hatase et al. 2002; Hatase et al. 2010), with adult females returning to nest, on average, every 3.3 years (mean "remigration interval") and laying 4.6 nests per season (mean "clutch frequency") (see Hatase et al. (2013)). Similar to most marine turtle populations worldwide, the only available monitoring data that are

suitable for assessing population status and trends are nest count data. The Sea Turtle Association of Japan (STAJ) provided annual nest count data for our use in this assessment, with the data coming exclusively from three beaches in Yakushima: Inakahama (1986–2015), Maehama (1989–2015), and Yotsusehama (1999–2015). The three beaches comprise an estimated 52% of annual nesting for the population (Matsuzawa, Sea Turtle Association of Japan, pers. comm.). We consider these data to be the best scientific information available and use them as an index of abundance for NP loggerheads.

Western Pacific Leatherback Turtles

The following information is taken directly from the the NOAA Technical Memorandum NMFS-PIFSC-95 (Martin et al. 2020). The western Pacific leatherback population ("WP leatherbacks" or "leatherbacks" herein) is currently listed as Endangered as part of a single global population of leatherback turtles. While it is not currently recognized as a distinct population segment under the ESA, it is genetically and ecologically different from other populations and is treated as such for management purposes related to the Hawaii-based and American Samoa-based longline fisheries. Nesting for this population occurs in the Indo-Pacific region, primarily in Indonesia, Papua New Guinea, and Solomon Islands, and to a lesser extent in Vanuatu. The WP leatherbacks are wide-ranging and undergo long migrations between nesting grounds and foraging areas. There are at least three important foraging regions for the WP leatherbacks, including pelagic areas of the Northeast Pacific (NEP; off the coasts of California, Oregon, and Washington), the North Pacific Transition Zone (NPTZ), and the South China Sea (Benson et al. 2011). The SSLL fishery is only known to interact with those foraging in the NEP and NPTZ, whereas we lack fisheries-independent data on the leatherbacks (mostly immature) that interact with the DSLL and ASLL fisheries. Females mature after 16.1 years (mean age at maturity) (Jones et al. 2011), and those from NEP and NPTZ foraging areas return to nest, on average, every 3.1 years (mean remigration interval, which was derived from Lontoh (2014)), laying 5.5 nests per season (mean clutch frequency) (Tapilatu et al. 2013). The only monitoring data that are suitable for trend analysis are nest counts from two beaches in the Bird's Head Peninsula of Papua Barat, Indonesia. Our Indonesian colleagues provided monthly nest count data for those beaches, Jamursba Medi (2001–2017) and Wermon (2006–2017), for our use in this assessment. However, due to the presence of gaps in the monthly data, we had to impute missing values to estimate annual nest counts prior to our trend analysis. While there is a single peak in nesting activity during the summer at Jamursba Medi, there are both summer and winter peaks in nesting at Wermon. The two beaches comprise approximately 75% of nesting activity for the population (NOAA Fisheries and USFWS Leatherback Turtle Status Review, in Review). We consider these data to be the best scientific information available and use them as an index of abundance for WP leatherbacks.

Methods

Data

The analyses in this report are based on the same nest count data that were provided by Martin et al. (2020). The nest count data originate from long-term monitoring programs in Japan (loggerheads) and Indonesia (leatherbacks) as stated previously. Raw nest count data were made available by international colleagues for this analysis (Figure 1). The loggerhead data from Japan were provided as annual nest counts from three index beaches (Maehama, Inakahama, and Yotsusehama) from 1985 to 2015. The leatherback data were monthly nest counts from two index beaches in Indonesia (Jamursba Medi and Wermon) from 2001 to 2017. The leatherback data set contained months with no monitoring effort; thus, we developed a model to impute the missing data in order to produce a time series of annual nest counts (Martin et al. 2020).

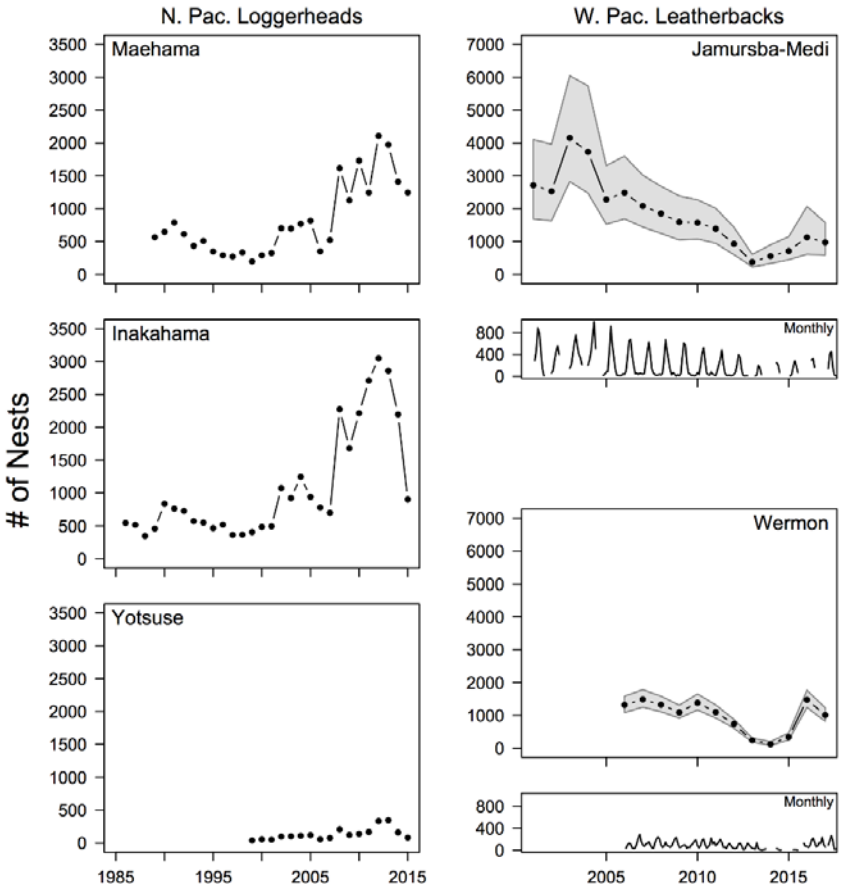

Figure 1. Raw nest count data available for this analysis (Figure 1 from Martin et al. (2020)). For North Pacific loggerhead turtles, annual nest count data came from three beaches in Japan (Maehama, Inakahama, and Yotsusehama) from 1985 to 2015. For western Pacific leatherback turtles, monthly nest count data came from two beaches in Indonesia (Jamursba Medi and Wermon) from 2001 to 2017. Annual counts for leatherbacks were imputed within this analysis; estimates are shown as median annual counts (points) with 95% credible intervals (gray shading).

Modeling Approach

Modeling methods followed those described for the SSLL fishery in Martin et al. (2020), with changes arising from differences in the observer data available. In contrast to the SSLL fishery, which employs 100% observer coverage, there is only partial observer coverage in both the DSLL and ASLL fisheries: approximately 20% (WPRFMC 2019). Here, we provide a description of the alterations to the methods and approach detailed in Martin et al. (2020). The two documents are intended to be used together to understand the complete approach to estimating impacts of the longline fisheries across the Pacific Islands region.

Developing a length-mortality distribution from observed interactions

Observers measured turtles and assessed condition, to later estimate post-interaction mortality rates, for individuals encountered during observed trips. These observations are only a sample of the total potential interactions due to partial observer coverage. Observer length estimates and assigned mortality rates were provided by the Pacific Islands Regional Office (PIRO) for this analysis. In the SSLL fishery (2004–2018), 149 of 188 (79%) loggerhead interactions had associated straight carapace length (SCL) measurements, and 81 of 114 leatherback interactions (71%) had associated length estimates or measurements (leatherbacks were typically too large to board the vessel, often requiring visual estimates). These lengths and assigned mortality rates were used in the SSLL model to parameterize a multivariate normal (MVN) distribution (technically bivariate with the two variables being length and mortality) that characterized historical length-mortality interactions for the fishery and was used to generate the length and mortality rates for potential future interactions. Historical turtle interactions without lengths were imputed with the median length by species for the SSLL model. The quantity of observed interactions with associated length and mortality data was much lower for the DSLL fishery (2004–2018), with 10 of 12 loggerhead interactions (83%) having length measurements, and 24 of 31 leatherback interactions (77%) having length estimates or measurements, and with those numbers representing approximately 20% of fishing trips. There were no loggerheads observed in the ASLL fishery (2006–2018), but all 12 leatherback interactions had length estimates; however, roughly 80% of fishing trips were unobserved. Due to the limited number of known lengths and assigned mortality rates for the DSLL and ASLL fisheries, the length-mortality distributions used in the model are much less informed (i.e., less data-rich), thereby leading to greater uncertainty about historical and future population impacts.

Incorporation of Unobserved Past and Future Interactions

Unike in the SSLL model, lengths and mortality of historical as well as future unobserved turtle interactions were drawn from the MVN distribution developed for each species, informed by the observed historical length and post-interaction mortality rates. As a result, stochasticity was added to both historical and future fishery interactions with adult nester equivalents (ANEs). An ANE is the conversion of a specified number of turtle takes, which is composed of various lengths, ages, stages (i.e., juvenile or adult), and sexes, to adult nesters, which are the only index of population abundance available (Martin et al. 2020). This discrepancy from the SSLL model compensated for the lower observer coverage in the DSLL and ASLL fisheries. For comparison, the SSLL fishery had 15 times more length-mortality data for loggerheads, 3 times more data for leatherbacks than the DSLL fishery, and 7 times more data than the ASLL for leatherbacks. To account for unobserved historical interactions in the individual fisheries in the historical take

analysis, estimates of total fishery and species specific interactions were taken from the *Annual Stock Assessment and Fishery Evaluation Report for U.S. Pacific Island Pelagic Fisheries Ecosystem Plan 2018* (WPRFMC 2019 and McCracken references therein). The estimated unobserved interactions (historical) were the difference between the estimated total interactions and the known observed interactions.

Long-term Trend and Current Abundance

Martin et al. (2020) provide estimates of the population growth rate and current nester abundance for the portions of the populations represented by the index nesting beach data (see Background). Current nester abundance was taken to be the estimated abundance as of 2015 for loggerheads and 2017 for leatherbacks from Annual Nester estimates derived from the trend analysis for the final 4 years of data per Equation 13 in Martin et al. (2020). To evaluate whether the SSLL fishery affected the historical trends and, if it had, to eliminate the possibility of double-counting the fishery's impacts in future projections, population growth rates were provided for two scenarios: "take" (i.e., estimating the trend using the unaltered nesting time series) and "no take" (i.e., removing the historical impact of the SSLL fishery before estimating the trend). Abundance estimates, however, were only provided for the "no take" scenario (i.e., removing the SSLL fishery historical impact). The abundance estimates included in Martin et al. (2020) therefore reflect the state of the population had the SSLL fishery never existed (i.e., the "no take" scenario). In the present analysis, we clarify the "true" state of trends and abundance for the populations as compared to the "no take" scenarios conducted for each fishery (SSLL previously; DSLL and ASLL in this analysis) by presenting results for all four scenarios together (three fishery-adjusted scenarios and one "true" state scenario). We also revised our description of the historical scenarios as follows: the "take" scenario became the "true" state (i.e., incorporates all threats, fishery-dependent and independent) and the "no take" scenario became the "fishery-adjusted" scenario.

As noted in Martin et al. (2020), the purpose of removing the historical fishery impacts prior to conducting the trend analysis within a fishery-specific analysis is to avoid double-counting the future impact on the population. This is because the true historical trend includes all threats acting on the population. The model-based Annual Nester estimates (and derived current nester abundance estimates) change when running the analysis for the different fisheries (i.e., SSLL, DSLL, ASLL) because they are part of the trend analysis, which only removes the historical impacts of the specific fishery being analyzed. The fishery-adjusted abundance estimates are important within the analysis because they serve as reference points for the future projections and allow assessment of the populations falling below abundance thresholds. Importantly, the fishery-adjusted estimates do not reflect the "true" state of the population outside of the fishery take model analysis.

Modeling Assumptions

Due to the high proportion of unobserved trips in the DSLL and the ASLL, we had to assume that (i) the observed lengths and post-interaction mortality rates were representative samples of the total interactions, and (ii) the estimated total interactions were acceptably accurate and unbiased with respect to the actual number of interactions. The former assumption affects the mean length and mean post-interaction mortality rate as well as the variance and covariance in both characteristics. Changes in the variance and covariance are more likely to affect the

calculation of ANEs (see Incorporation of unobserved past and future interactions) of the historical and future interactions if the observed lengths and post-interaction mortality rates are more overdispersed or underdispersed relative to the total interactions. Either over- or underdispersion could result in ANEs that skew more conservative or more liberal for the species. The latter assumption affects the total ANE estimated for the historical scenarios and inaccuracies between the true total take and the estimated total take would solely affect the fishery-adjusted trend analysis. However, as shown in Results, small changes in the historical take of ANEs (e.g., 1–10 ANEs) have a minimal effect on the estimation of the population growth rate.

Conservation Implications

We have carried over the conservation implications from the SSLL fishery analysis (Martin et al. 2020) as they remain relevant to the analyses for the DSLL and ASLL fisheries. Martin et al. (2020) described modeling decisions for the SSLL fishery in detail and highlighted the ways in which they were inherently conservative for the turtle species in question. In summary, those decisions included (1) applying the full take from each fishery to the index nesting populations (i.e., not the full nesting population), which errs on the side of over-estimating the fishery impacts; (2) reporting and incorporating the uncertainty in the estimated population growth rate (r), which we believe reflects our understanding of the population trend; (3) acknowledging that the anticipated take level has a potentially high bias due to the estimation methods currently in use; and (4) assuming typical survival rates for adults in our historical analysis of fishery impacts which may have over-estimated fishery impacts (i.e., we added nesters back to the population when they potentially would not have survived due to other existing threats). We assumed that the full take from the fishery was independent of the population trend and fishery effort in the future, resulting in the full take in terms of ANEs being taken from the population at all time points in the projection. This is inherently conservative for declining populations because taking a constant number of turtles would be unlikely as a population declines (note—the opposite would be true for an increasing population such that applying a constant level of take into the future would mean applying a diminishing *rate* of take as the population increases). There are no additional considerations specific to the DSLL and ASLL fisheries.

Results

Incorporating Fishery Interactions into the Model

Assigned Lengths and Mortality for Historical and Future Interactions

As described by Martin et al. (2020), we used previously observed turtle lengths and assigned post-interaction mortality rates to characterize MVN distributions for each species in each fishery. The puropose of the MVN distributions was to allow us to generate lengths and mortality rates for unobserved interactions (both historical and future). A key parameter of the MVN distribution was mean length, $\theta_{L,j}$, a linear function of the annual number of fishery takes, Fj, with intercept, α_0 (i.e., the expected length of turtles in log space), and slope, α_1 ($\theta_{L,j}=\alpha_0 + \alpha_1 F_j$; see Equations 16–18 in Martin et al. (2020)). The slope, α_1, quantifies the relationship between the number of fishery takes and the expected (mean) length. For the DSLL fishery, this slope was positive (i.e., as the number of takes increases, the mean length of turtles increases), and for the ASLL fishery, it was negative (i.e., as the number of takes increases, the mean length of turtles decreases); however, neither slope was statistically significant. For the DSLL fishery, $\exp(\alpha_0)$ was 75 cm SCL and α_1 was 0.041 (90% CI, –0.044 to 0.129) for loggerheads, and for leatherbacks, $\exp(\alpha_0)$ was 115 cm SCL and α_1 was 0.015 (90% CI, –0.025 to 0.057) (see Martin et al. (2020) for details on those parameters). For the ASLL fishery, was 141 cm SCL and α_1 was –0.16 (90% CI, –0.33 to 0.04) for leatherbacks. For the DSLL fishery, the median correlation between length and mortality parameters was –0.15 for loggerheads (weakly negative) and –0.55 for leatherbacks (moderately negative). For leatherbacks in the ASLL fishery, the median correlation between parameters was –0.45 (moderately negative).

The mean assigned post-interaction probability of mortality, μ_D, was 0.95 for loggerheads in the DSLL fishery, 0.67 for leatherbacks in the DSLL fishery, and 0.88 for leatherbacks in the ASLL fishery. The posterior distributions of all bivariate normal parameters for the take demographics are included in Figure 2 for the DSLL fishery and Figure 3 for the ASLL fishery. To impute the lengths and mortalities of the historical unobserved interactions as well as draw lengths and mortalities of future interactions, the MVN distribution of log lengths and logit mortalties was used. The parameter values used to characterize this MVN distribution were the medians of each parameter's posterior distribution, which was estimated from the observed historical lengths and mortalities (i.e., median values from Figures 2 and 3).

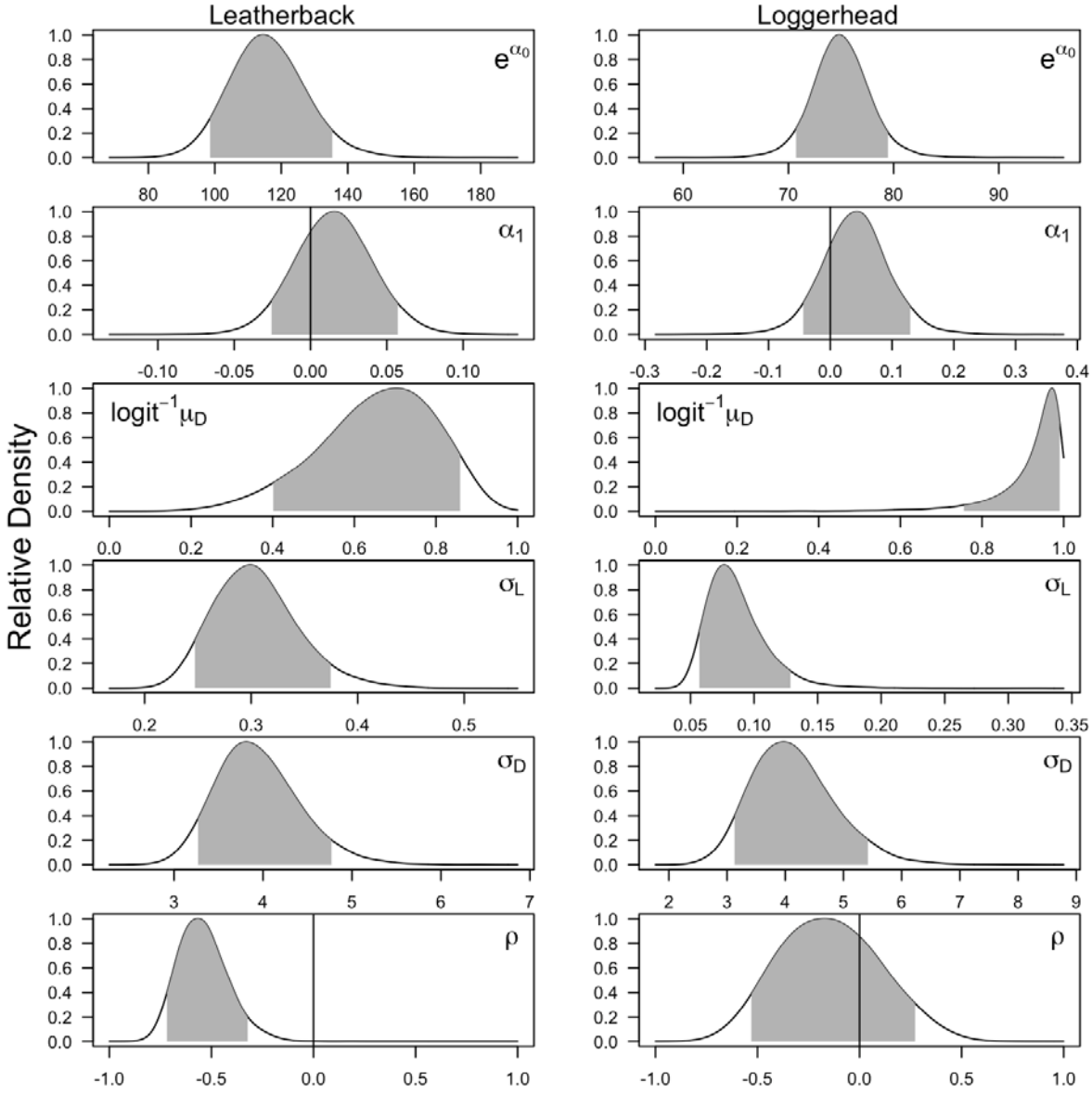

Figure 2. Hawaii-based deep-set longline (DSLL) fishery—posterior distributions of parameters (*y*-axes are relative densities) used in the multivariate normal distribution to generate lengths and assigned fishery mortality for each individual in the annual anticipated take and historical unobserved take. The intercept (α_0) and slope (α_1) were used to model the mean log lengths as a function of the take. For visualization, the intercept has been exponentiated to the anti-log scale in cm of SCL. The remaining parameters are the mean logit assigned fishery mortality (using an inverse logit transform; $\text{logit}^{-1}\ \mu_D$), the standard deviations of log lengths (σ_L) and logit fishery mortality (σ_D), and the correlation between the log lengths and the logit fishery mortality (ρ).

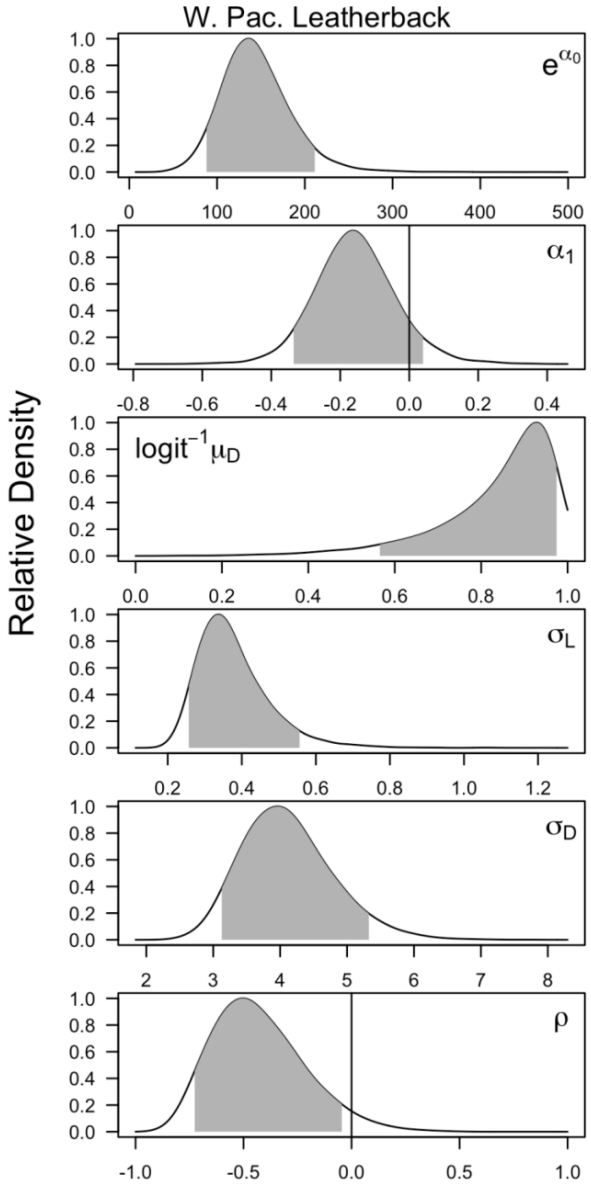

Figure 3. American Samoa-based longline (ASLL) fishery—posterior distributions of parameters (*y*-axes are relative densities) used in the multivariate normal distribution to generate lengths and assigned fishery mortality for each individual in the annual anticipated take and historical unobserved take. The intercept (α_0) and slope (α_1) were used to model the mean log lengths as a function of the take. For visualization, the intercept has been exponentiated to the anti-log scale in cm of SCL. The remaining parameters are the mean logit assigned fishery mortality (using an inverse logit transform; $\text{logit}^{-1}\ \mu_D$), the standard deviations of log lengths (σ_L) and logit fishery mortality (σ_D), and the correlation between the log lengths and the logit fishery mortality(ρ).

Historical Take

DSLL fishery interactions with loggerheads have been exlusively with immature turtles (mean SCL 69.0 cm; n = 10; 100% immature using 86.9 cm SCL size at maturity from the Appendix), similar to the SSLL fishery (mean SCL 58.0 cm; n = 147; 98.6% immature). With respect to leatherbacks, the SSLL fishery interacted with a higher proportion of adults (mean SCL 143.4 cm; n = 89; 51.7% adults using 142.7 cm SCL size at maturity from the Appendix) than both the DSLL (mean SCL 119.0 cm; n = 24; 25% adults) and ASLL (mean SCL 100.0 cm; n = 12; 25% adults) fisheries. Size-at-maturity of 142.7 cm SCL is conservative for the species, as it is on the low end of the spectrum for this population; a longer SCL would be associated with a longer period of growth at a lower survival rate and thus would decrease the ANE.

The cumulative historical take ANE for each year (2005–2017) is shown in Figure 4 for the DSLL fishery and Figure 5 for the ASLL fishery. It is expressed as cumulative ANE ($\sum_{k=1}^{n} ANE_{realized,k,j}$), where *k* is an individual turtle, *n* is total number of takes for a given year, *j* is year, and *realized* indicates conversion of each turtle to an ANE to account for age, sex, and post-interaction mortality. In this analysis, the ANE for a particular year includes the take of adult females from that year plus any carryover ANE from previous years (i.e., immature turtles surviving and maturing or mature turtles returning to nest), and includes estimates for unobserved interactions. The loggerhead ANE for the DSLL fishery is 0–16 per year. The leatherback ANE for the DSLL fishery remained at 0–8 per year through 2013 then increased to 8–27 per year in 2014–2017. The leatherback ANE for the ASLL fishery is 0–0.3 per year in 2011–2017 (Figure 5).

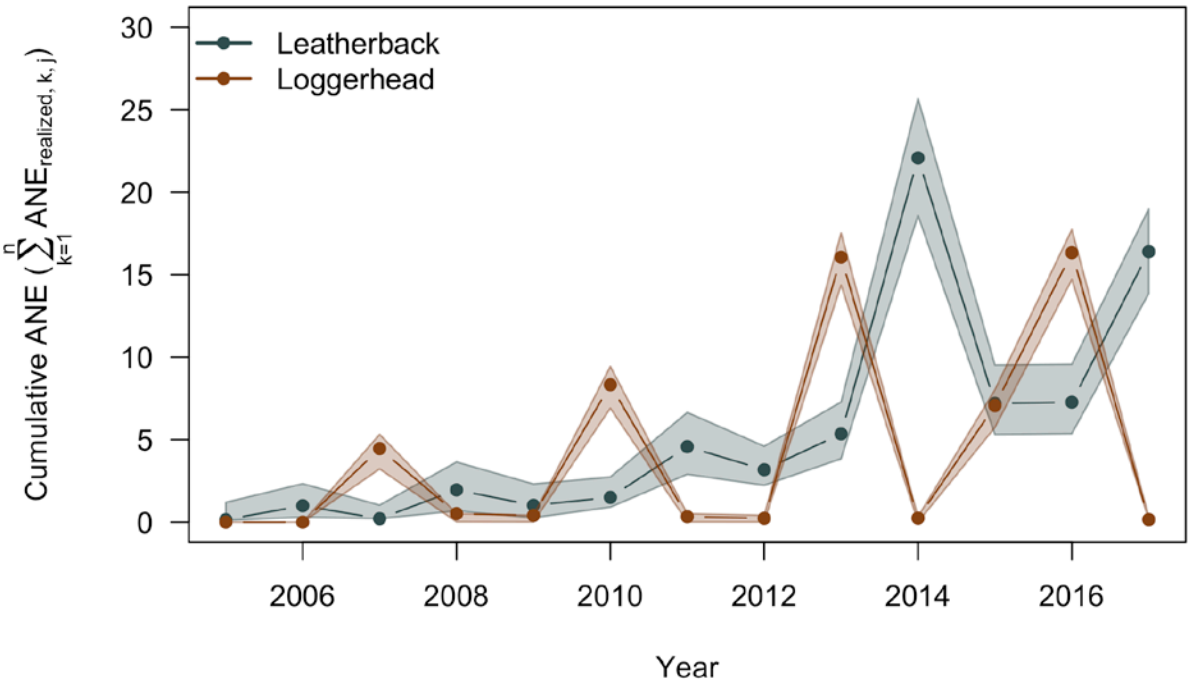

Figure 4. Hawaii-based deep-set longline (DSLL) fishery—the cumulative historical take as adult nester equivalents (ANE) for each species for each year (leatherbacks in dark blue; loggerheads in brown). The cumulative historical take ANE for each year is expressed as ($\sum_{k=1}^{n} ANE_{realized,k,j}$) where *k* is an individual turtle, *j* is year, *n* is total number of takes for a given year, and *realized* indicates conversion of the turtle to an ANE to account for age, sex, and post-interaction mortality. Note that the polygon surrounding each line is the 95% confidence interval generated from 10,000 permutations of the calculating the cumulative historical take.

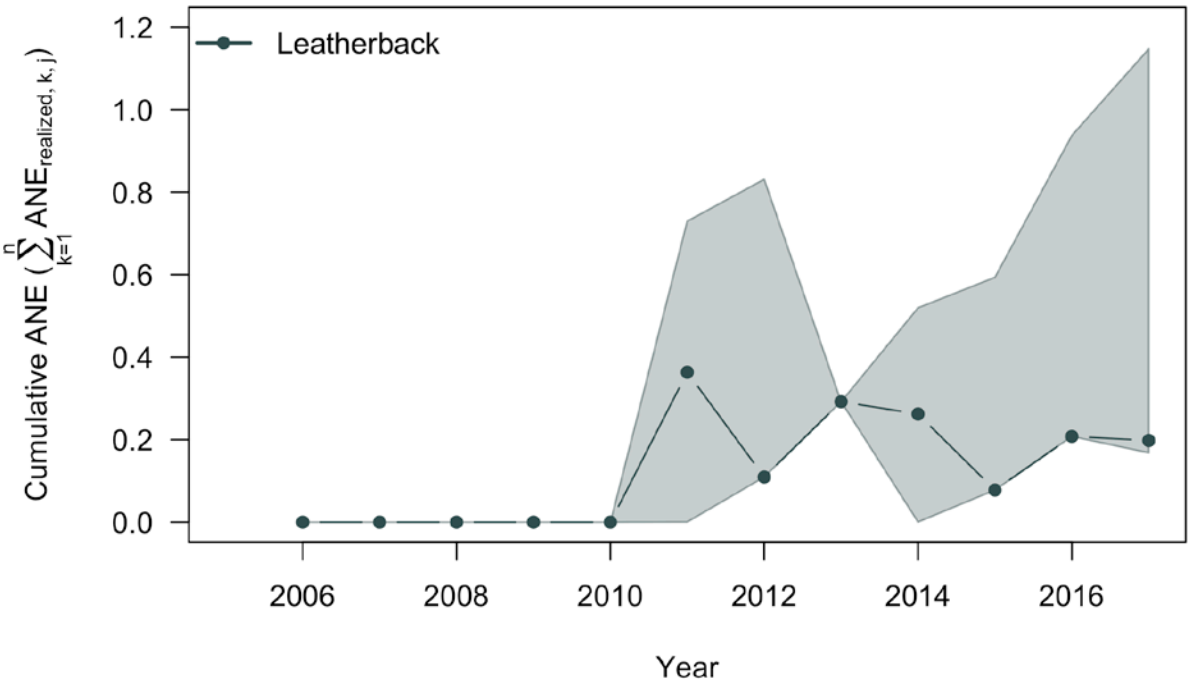

Figure 5. American Samoa-based longline (ASLL) fishery—the cumulative historical take as adult nester equivalents (ANE) for leatherback turtles for each year. There were no loggerhad turtle interactions. The cumulative historical take ANE for each year is expressed as ($\sum_{k=1}^{n} ANE_{realized,k,j}$) where *k* is an individual turtle, *j* is year, *n* is total number of takes for a given year, and *realized* indicates conversion of the turtle to an ANE to account for age, sex, and post-interaction mortality. Note that the polygon surrounding each line is the 95% confidence interval generated from 10,000 permutations of the calculating the cumulative historical take; the intervals are asymmetric as a result of a base amount of $ANE_{realized,k}$ occurring from interactions in a given year and the variation resulting from the assumed length of the unobserved interactions.

Long-term Trends and Current Abundance

Estimates of the population growth rate (i.e., long-term annual trend inferred from the number of nests) and current nester abundance are summarized for loggerheads in Table 2 and leatherbacks in Table 3. Estimates reflect the portion of each population represented by the index nesting beaches (i.e., 52% for loggerheads and 75% for leatherbacks). The tables include estimates corresponding to the "true" state (i.e., the trend and abundance inclusive of all threats acting on the populations in the past), as well as the removal of historical impacts of the SSLL fishery, DSLL fishery, and ASLL fishery (each case was treated separately). Results for the latter two fisheries are detailed in a later section (see Martin et al. (2020) for SSLL fishery results). While the estimates within the three fishery scenarios are useful as reference points within our population viability analysis (PVA) for assessing future impacts, they do not represent the "true" state of the populations based on the available nesting data. Only the estimates in the "true" state scenario should be referenced with respect to the status of the populations.

Theoretically, if the historical impact of a fishery on a turtle population was removed as if it never happened, then we would expect nester abundance to increase, a negative trend to become less negative (e.g., for leatherbacks), and a positive trend to become more positive (e.g., for loggerheads). If those changes are not observed upon removal of the fishery impact, then we would conclude that the fishery did not have an impact on the population in the past, or at least not an impact of discernible magnitude given the uncertainty in the estimated parameters.

For loggerheads, current abundance increases by 3 and 10 nesters when the SSLL and DSLL fishery impacts are removed, respectively (Table 2). For context, these differences mean that current abundance (the "true" state) is 99.9% of what it would have been without SSLL fishery impacts and 99.8% of what it would have been without DSLL fishery impacts over the period 2004–2015 (Table 2). The loggerhead trend remains the same for the "true" state (2.3% per year), the SSLL-adjusted scenario (2.3% per year), and the DSLL-adjusted scenario (2.4% per year) (Table 2 and Figure 6).

For leatherbacks, current abundance increases by 3, 22, and 0 nesters when the impacts are removed from the SSLL, DSLL, and ASLL fisheries, respectively (Table 3). For context, current abundance for the "true" state is 99.6%, 97.3%, and 100% of what it would have been without historical impacts from the SSLL, DSLL, and ASLL fisheries, respectively (Table 3). The leatherback trend remains the same for the "true state" (−6.1% per year), the SSLL-adjusted scenario (−6.1% per year), the DSLL-adjusted scenario (−6.0% per year), and the ASLL-adjusted scenario (−6.1% per year) (Table 3 and Figures 7 and 8).

Due to the indepedent trend estimation for each fishery, differences on the order of a tenth of a percentile of the population growth rate can result from stochasticity in the sampling of the posterior distribution of the population growth rate, especially under the vague priors assumed for the population growth rate, rather than from real differences between the "true" state and the fishery-adjusted trends.

Table 2. For North Pacific loggerheads, estimates of the long-term annual trend (i.e., population growth rate, *r*) and nesting female abundance under different scenarios pertaining to the removal of historical fisheries impacts. Fisheries include Hawaii-based shallow-set longline (SSLL), Hawaii-based deep-set longline (DSLL), and American Samoa-based longline (ASLL). Scenarios include a "true" state (i.e., all historical threats included), SSLL-adjusted, DSLL-adjusted, and ASLL-adjusted (not applicable for loggerheads, as no interactions occurred). Results include the median estimate with lower and upper 95% credible interval limits. Model-based abundance estimates for the four final years of data were used to calculate an estimate of current nester abundance. Note that these estimates are for the three nesting beaches used in this analysis, which represent 52% of total nesting activity. Minor differences in the trend are likely from posterior sampling stochasticity (as the SSLL-adjusted, DSLL-adjusted, and ASLL-adjusted trends were estimated separately) rather than real differences between the "true" state and the fishery-adjusted trend.

	True State			SSLL-adjusted			DSLL-adjusted			ASLL-adjusted		
	Med.	L95%	U95%	Med.	L95%	U95%	Med.	L95%	U95%	Med.	L95%	U95%
Trend (*r*)	0.023	−0.11	0.157	0.023	−0.11	0.156	0.024	−0.11	0.157	NA	NA	NA
2012	1853	1527	2235	1851	1529	2236	1853	1528	2235	NA	NA	NA
2013	1776	1462	2161	1778	1461	2153	1784	1469	2171	NA	NA	NA
2014	1203	986	1445	1202	977	1449	1204	987	1446	NA	NA	NA
2015	651	529	854	651	529	858	654	531	857	NA	NA	NA
Current Abund.	4538	4077	5064	4541	4074	5063	4548	4086	5074	NA	NA	NA

Table 3. For western Pacific leatherbacks, estimates of the long-term annual trend (i.e., population growth rate, *r*) and nesting female abundance under different scenarios pertaining to the removal of historical fisheries impacts. Fisheries include Hawaii-based shallow-set longline (SSLL), Hawaii-based deep-set longline (DSLL), and American Samoa-based longline (ASLL). Scenarios include a "true" state (i.e., all historical threats included), SSLL-adjusted, DSLL-adjusted and ASLL-adjusted (not applicable for loggerheads, as no interactions occurred). The trend and abundance are based on median monthly leatherback nest counts estimated by the imputation model in Martin et al. (2020). Results include the median estimate with lower and upper 95% credible interval limits. Model-based abundance estimates for the four final years of data were used to calculate an estimate of current nester abundance. Note that these estimates are for the two nesting beaches used in this analysis, which represent 75% of total nesting activity. Minor differences in the trend are likely from posterior sampling stochasticity (as the SSLL-adjusted, DSLL-adjusted, and ASLL-adjusted trends were estimated separately) rather than real differences between the "true" state and the fishery-adjusted trend.

	True State			SSLL-adjusted			DSLL-adjusted			ASLL-adjusted		
	Med.	L95%	U95%	Med.	L95%	U95%	Med.	L95%	U95%	Med.	L95%	U95%
Trend (*r*)	−0.061	−0.24	0.123	−0.061	−0.238	0.122	−0.060	−0.237	0.121	−0.061	−0.241	0.122
2014	160	116	208	162	118	211	171	123	222	160	116	208
2015	216	165	278	217	166	280	223	171	288	216	165	278
2016	340	253	446	340	256	445	344	255	453	340	253	446
2017	309	236	419	309	237	417	316	241	429	309	236	419
Current Abund.	787	659	939	790	666	942	809	678	966	787	659	939

North Pacific Loggerhead Turtles

As described in Martin et al. (2020), the population growth rate (r) estimation was conducted on the time series for either N_j, the number of annual nesters in year j, or $Nj + F$, annual nesters plus historical takes converted to ANEs (note—for this analysis, we have updated our notation from $Nj - F$ to $Nj + F$ when representing the historical fishery-adjusted scenario formerly identified in Martin et al. (2020) as a "no take" scenario). The DSLL-adjusted scenario removed the population impacts of historical interactions in the DSLL fishery. Similar to findings for the SSLL fishery (Martin et al. 2020), there was no notable difference between the posterior distributions of r estimated for the historical "true" state and DSLL-adjusted scenarios for NP loggerheads (see Figure 6 and Table 4). As shown in Figure 6, the distributions are closely overlapping, indicating the estimated population growth rate changed minimally when the turtles taken by the fishery were added back in to the population prior to estimating the trend. There have been no observed loggerhead interactions in the ASLL; therefore, we provide no figures, tables, or text.

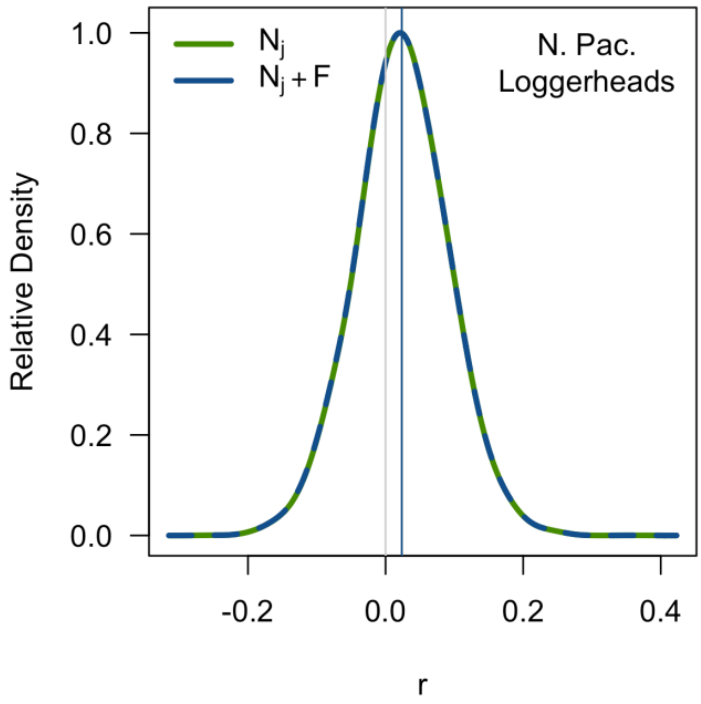

Figure 6. Hawaii-based deep-set longline (DSLL) fishery—posterior distribution of r (population growth rate) for North Pacific loggerheads under historical "true" state (N_j) and fishery-adjusted (N_j+F) scenarios. The two colored vertical lines are the median for the two scenarios. The gray vertical line indicates r = 0 (no growth).

The long-term annual trend (i.e., population growth rate) for North Pacific loggerheads over the period of available nest monitoring data (1985–2015) is summarized in Table 4, with the "true" state and fishery-adjusted scenarios presented for comparison. As was the case for the SSLL fishery, the estimates are based on the available data from three beaches in Japan (Inakahama, Maehama, and Yotsusehama) that comprise approximately 52% of total nesting (Martin et al. 2020). Results suggest an increasing trend in the number of females nesting annually. The

median growth rate estimate was 2.4% per year (95% CI, −11.0% to 15.7%) for the DSLL-adjusted scenario and 2.3% per year (95% CI, −11.0% to 15.7%) for the "true" state scenario (Table 4).

Table 4. Hawaii-based deep-set longline (DSLL) fishery—summary of the long-term annual trend (i.e., population growth rate) for North Pacific loggerheads under fishery-adjusted and "true" state scenarios over the period of available nest monitoring data (1985–2015). Results include log population trend (r) and λ with the mean ($\underline{x}$), median ($\tilde{x}$), variance (σ_x), lower 95% (L95%) and upper 95% (U95%) of the statistic.

	DSLL-adjusted	True State
$\underline{r}$	0.024	0.024
$\tilde{r}$	0.024	0.023
σ_r	0.005	0.005
$r_{L95\%}$	–0.11	–0.11
$r_{U95\%}$	0.157	0.157
$\underline{\lambda}$	1.026	1.026
$\tilde{\lambda}$	1.024	1.024
σ_λ	0.005	0.005
$\lambda_{L95\%}$	0.896	0.896
$\lambda_{U95\%}$	1.17	1.17

Western Pacific Leatherback Turtles

As described in Martin et al. (2020), the growth rate estimation was conducted on the time series for either N_j, the number of Annual Nesters in year *j*, or $N_j + F$, Annual Nesters plus historical takes converted to ANEs. The fishery-adjusted scenario removes the population impacts of historical interactions in either the DSLL fishery or the ASLL fishery (i.e., the analysis is conducted separately for each fishery). Similar to findings for the SSLL fishery (Martin et al. 2020), there was no notable difference between the posterior distributions of *r* (population growth rate) for western Pacific leatherbacks under historical "true" state and fishery-adjusted scenarios for the DSLL fishery (see Figure 7 and Table 5) or the ASLL fishery (see Figure 8 and Table 6). As shown in Figures 7 and 8, the distributions are closely overlapping, indicating the population growth rate had minimal or no change when adding the ANEs that were taken historically by either fishery back in to the historical population prior to estimating the trend.

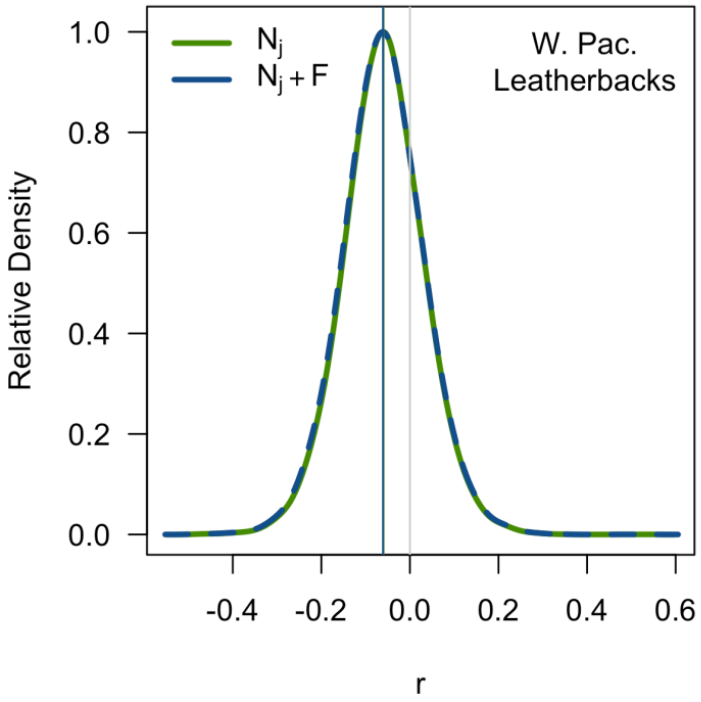

Figure 7. Hawaii-based deep-set longline (DSLL) fishery—posterior distribution of r (population growth rate) for western Pacific leatherbacks under historical "true" state (N_j) and fishery-adjusted (N_j+F) scenarios. The two colored vertical lines are the median for the two scenarios. The growth rate is based on median monthly leatherback nest counts estimated by the imputation model in Martin et al. (2020). The gray vertical line indicates r = 0 (no growth).

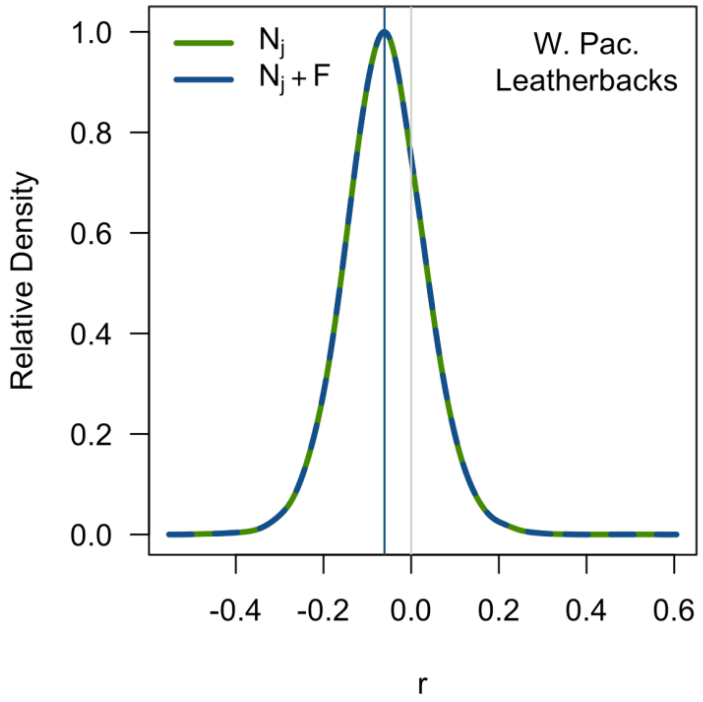

Figure 8. American Samoa-based longline (ASLL) fishery—posterior distribution of r (population growth rate) for western Pacific leatherbacks under historical "true" state (N_j) and fishery-adjusted (N_j+F) scenarios. The two colored vertical lines are the median for the two scenarios. The growth rate is based on median monthly leatherback nest counts estimated by the imputation model in Martin et al. (2020). The gray vertical line indicates r = 0 (no growth).

The long-term trend (i.e., population growth rate) for western Pacific leatherbacks over the period of available nest monitoring data (2001–2017) is summarized in Table 5 for the DSLL fishery and Table 6 for the ASLL fishery, with the fishery-adjusted and "true" state scenarios presented for comparison. As was the case for the SSLL fishery, the estimates are based on the available data from two beaches in Indonesia (Jamursba Medi and Wermon) that comprise approximately 75% of total nesting (Martin et al. 2020). Three estimates are provided (Median, Low, and High), corresponding with three different scenarios from the imputed monthly count estimates (median, lower 95%, and upper 95%) as described in Martin et al. (2020).

Results from the DSLL fishery analysis suggest a declining trend in the number of females nesting annually, with a median trend estimate of −6.1% per year (95% CI, −24.1% to 12.2%; median range of −5.5% to −6.4% across monthly nest count imputation scenarios) under the "true" state scenario and −6.0% (95% CI, −23.7% to 12.1%; median range of −5.4% to −6.2% across monthly nest count imputation scenarios) under the DSLL-adjusted scenario (Table 5).

Similarly, the ASLL fishery analysis suggested a median trend estimate of −6.1% per year (95% CI, −24.1% to 12.2%; median range of −5.5% to −6.4% across monthly nest count imputation scenarios) for both the "true" state and ASLL-adjusted scenarios (Table 6).

Table 5. Hawaii-based deep-set longline (DSLL) fishery—summary of the long-term annual trend (i.e., population growth rate) for western Pacific leatherbacks under fishery-adjusted and "true" state scenarios over the period of available nest monitoring data (2001–2017). Median, Low, and High scenarios correspond to the different outputs from the imputation model for monthly leatherback nest counts. Results include log population trend (r) and λ with the mean ($\underline{x}$), median ($\tilde{x}$), variance ($\sigma^2_{x_x}$), lower 95% (L95%), and upper 95% (U95%) of the statistic.

	Median		Low		High	
	DSLL-adjusted	True State	DSLL-adjusted	True State	DSLL-adjusted	True State
$\underline{r}$	−0.059	−0.06	−0.062	−0.063	−0.054	−0.055
$\tilde{r}$	−0.06	−0.061	−0.062	−0.064	−0.054	−0.055
σ_r^2	0.008	0.008	0.008	0.009	0.008	0.009
$r_{L95\%}$	−0.237	−0.241	−0.24	−0.246	−0.235	−0.238
$r_{U95\%}$	0.121	0.122	0.119	0.122	0.129	0.13
$\underline{\lambda}$	0.946	0.945	0.944	0.943	0.951	0.951
$\tilde{\lambda}$	0.942	0.941	0.94	0.938	0.947	0.946
σ_{λ}^2	0.007	0.008	0.007	0.008	0.008	0.008
$\lambda_{L95\%}$	0.789	0.786	0.787	0.782	0.79	0.788
$\lambda_{U95\%}$	1.128	1.13	1.126	1.13	1.138	1.139

Table 6. American Samoa-based longline (ASLL) fishery—summary of the long-term annual trend (i.e., population growth rate) for western Pacific leatherbacks under fishery-adjusted and "true" state scenarios over the period of available nest monitoring data (2001–2017). Median, Low, and High scenarios correspond to the different outputs from the imputation model for monthly leatherback nest counts. Results include log population trend (r) and λ with the mean ($\underline{x}$), median ($\tilde{x}$), variance ($\sigma^2{}_x$), lower 95% (L95%), and upper 95% (U95%) of the statistic.

	Median		Low		High	
	ASLL-adjusted	True State	ASLL-adjusted	True State	ASLL-adjusted	True State
$\underline{r}$	−0.06	−0.06	−0.063	−0.063	−0.055	−0.055
$\tilde{r}$	−0.061	−0.061	−0.064	−0.064	−0.055	−0.055
σ_r^2	0.008	0.008	0.009	0.009	0.009	0.009
$r_{L95\%}$	−0.241	−0.241	−0.246	−0.246	−0.238	−0.238
$r_{U95\%}$	0.122	0.122	0.122	0.122	0.13	0.13
$\underline{\lambda}$	0.945	0.945	0.943	0.943	0.951	0.951
$\tilde{\lambda}$	0.941	0.941	0.938	0.938	0.946	0.946
σ_λ^2	0.008	0.008	0.008	0.008	0.008	0.008
$\lambda_{L95\%}$	0.786	0.786	0.782	0.782	0.788	0.788
$\lambda_{U95\%}$	1.129	1.13	1.129	1.13	1.139	1.139

Population Viability Analysis (PVA) with Take and without Take

DSLL Fishery PVA Projections for North Pacific Loggerhead Turtles

Projections out 100 years for loggerheads reflect the 2.3% annual growth rate (Table 4) estimated from the 1985–2015 nesting data (Figure 9). Similar to results described in Martin et al. (2020) for the SSLL fishery, there is no discernible difference between the deterministic and stochastic models for the DSLL fishery (top vs. bottom panel in Figure 9). There is also no discernible difference between the "no take" and "take" scenarios for the future, shown by the completely overlapping medians and shaded credible envelopes in Figure 9 (note—natural log scale is used for the 100-year projections to facilitate visual detection of differences between the two scenarios) and the difference plot in Figure 10 centered on zero. It should be noted that the difference plot is made by taking the difference between paired projections of the "take" and "no take" scenario (i.e., both scenarios have the same trend parameters and the sole difference between them is applying the anticipated take).

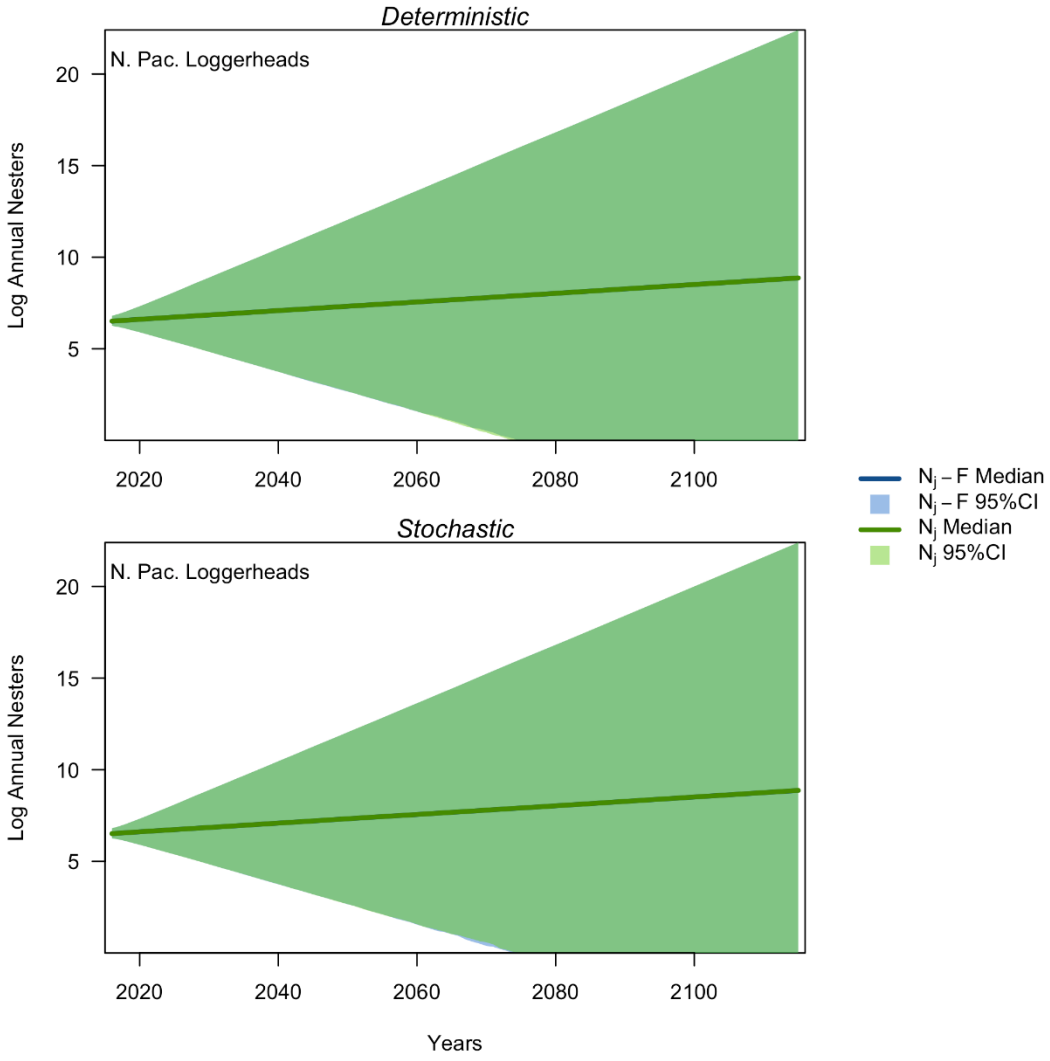

Figure 9. Hawaii-based deep-set longline (DSLL) fishery—for North Pacific loggerheads, 100-year projections of Annual Nesters (vertical axis is in natural log units) under future scenarios including take (N_j – F) and no take (N_j). Projections begin a year after the final year of available data (2015) and end 100 years later. Top panel shows results from a fully deterministic version of the take model in which the parameters are set at point estimates rather than pulled from distributions as they are in the stochastic version in the bottom panel. Note the almost exact overlay of the trend line and uncertainty envelope between the two scenarios.

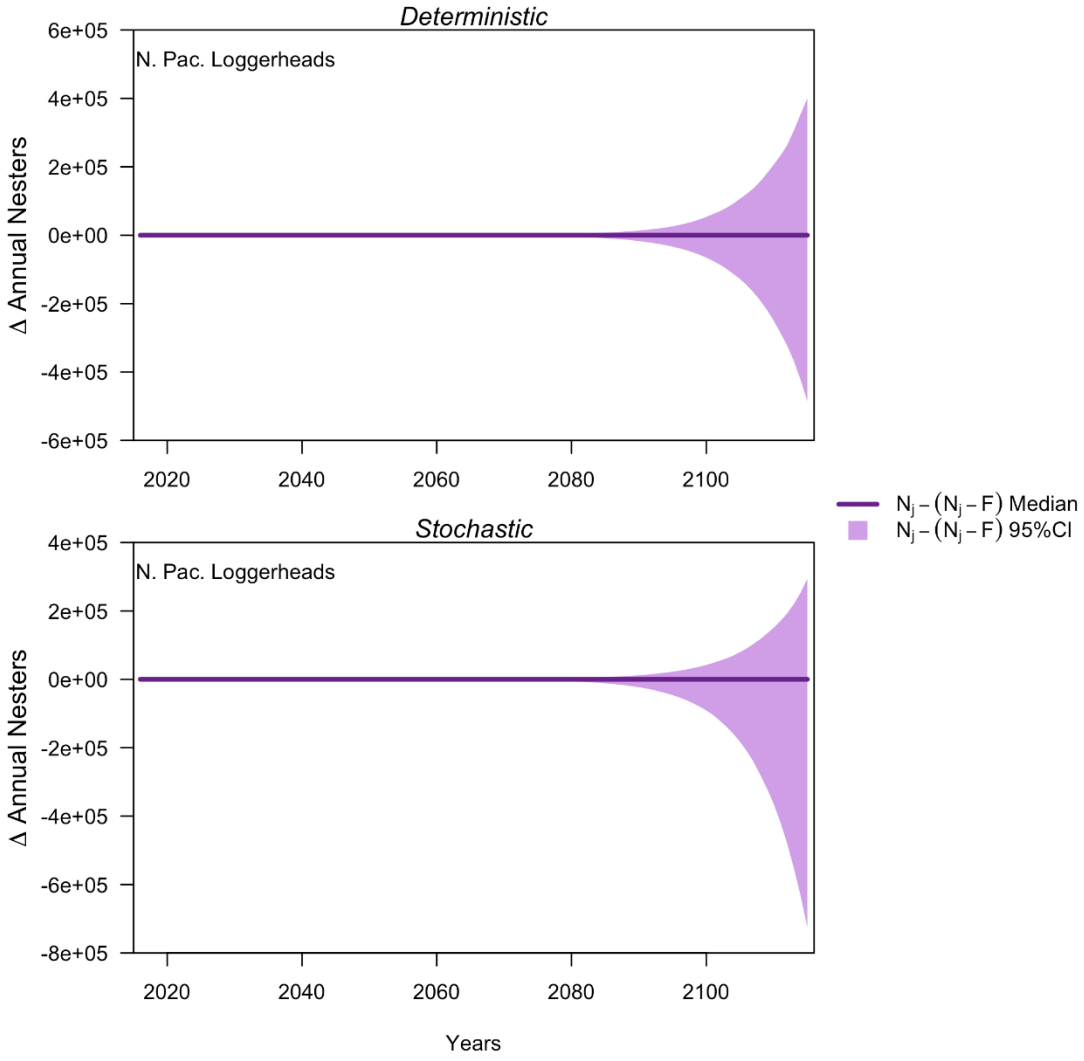

Figure 10. Hawaii-based deep-set longline (DSLL) fishery—for North Pacific loggerheads, difference plots of 100-year paired projections of Annual Nesters under future scenarios including take (N_j – F) and no take (N_j). Projections begin a year after the final year of available data (2015) and end 100 years later. Top panel shows results from a fully deterministic version of the take model in which the parameters are set at point estimates rather than pulled from distributions as they are in the stochastic version in the bottom panel.

Projections out 10 years into the future are more relevant biologically for management purposes than to 100 years given the estimated uncertainty in the population parameters. Specifically, the effects of the environmental or anthropogenic drivers on the population would be lagged; therefore, we think the first 10 years is largely based on the previously observed trend, but after that, we do not have sufficient information to account for uncertainty of the drivers that affect the populations. For loggerheads in the DSLL fishery, there was again no discernible difference between the deterministic and stochastic versions of the model or between the "no take" and "take" scenarios (Figures 11 and 12) (note—to provide a sense of the actual magnitude change in annual nesters over the shorter time frame, we do not use the natural log scale here).

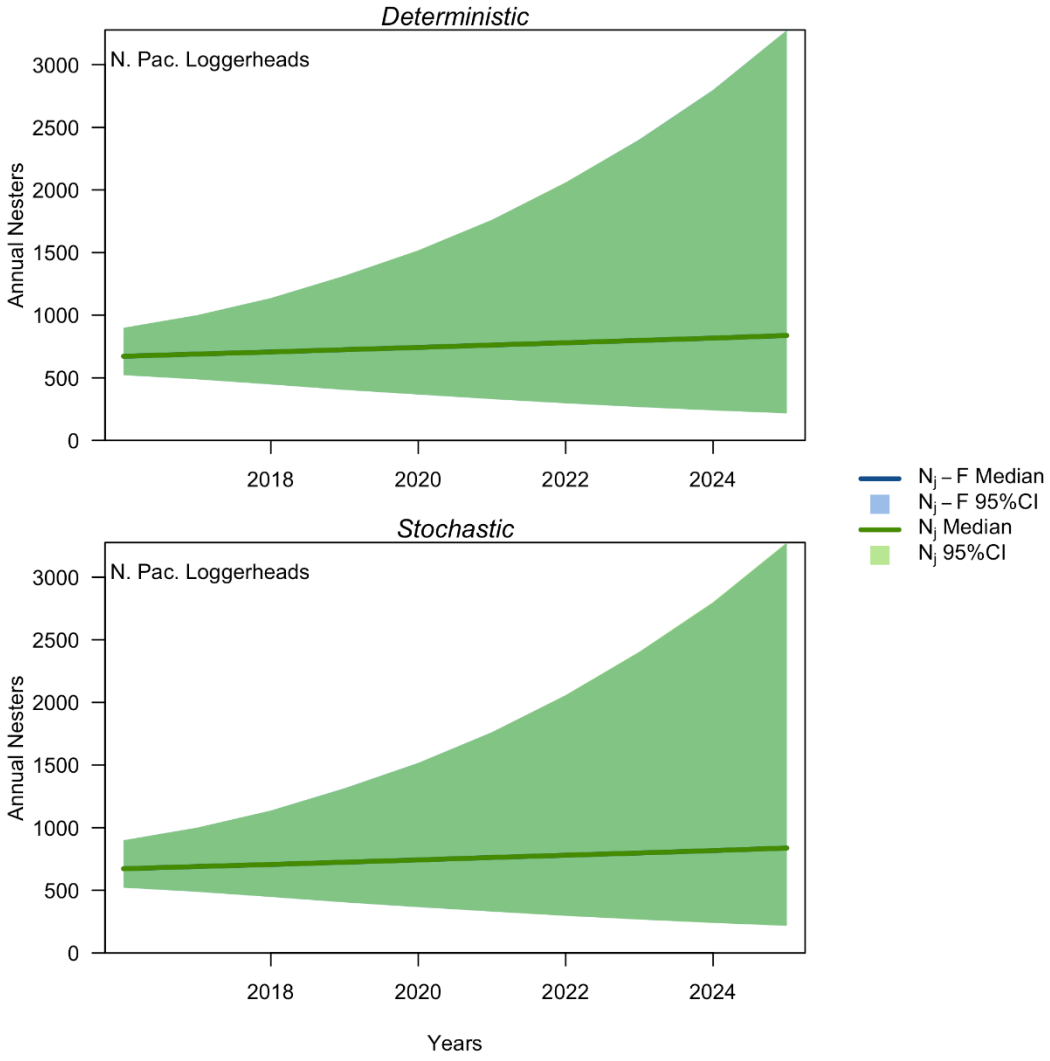

Figure 11. Hawaii-based deep-set longline (DSLL) fishery—for North Pacific loggerheads, 10-year projections of Annual Nesters under future scenarios including take (N_j – F) and no take (N_j). Projections begin a year after the final year of available data (2015) and end 10 years later. Top panel shows results from a fully deterministic version of the take model in which the parameters are set at point estimates rather than pulled from distributions as they are in the stochastic version in the bottom panel. Note the almost exact overlay of the trend line and uncertainty envelope between the two scenarios.

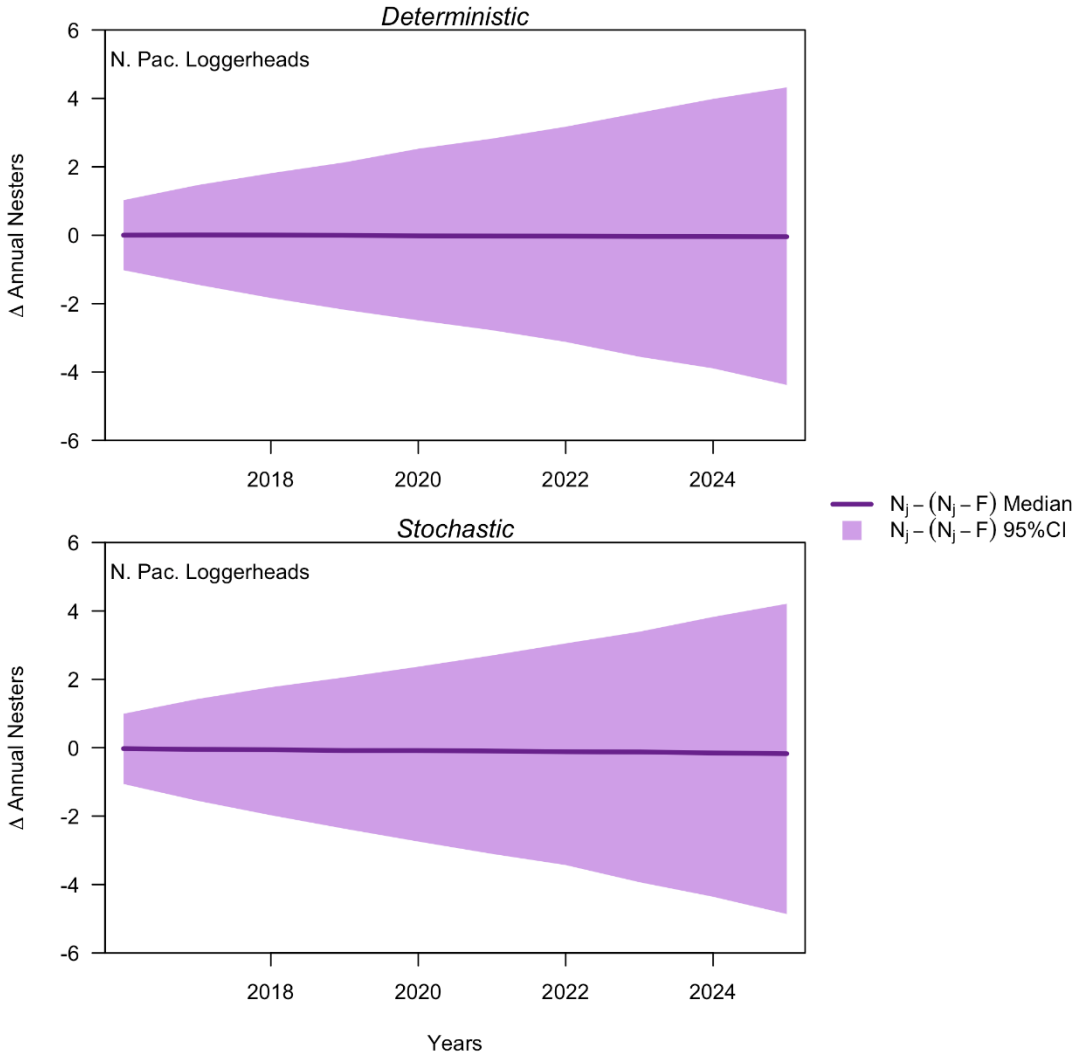

Figure 12. Hawaii-based deep-set longline (DSLL) fishery—for North Pacific loggerheads, difference plots of 10-year paired projections of Annual Nesters under future scenarios including take (N_j – F) and no take (N_j). Projections begin a year after the final year of available data (2015) and end 10 years later. Top panel shows results from a fully deterministic version of the take model in which the parameters are set at point estimates rather than pulled from distributions as they are in the stochastic version in the bottom panel. Note the almost exact overlay of trend and uncertainty envelope.

For loggerheads in the DSLL fishery, 32% of the simulation runs projected that Annual Nesters would fall below 50% of current annual nesters within 24 years (95% CI, 5 to 82 years), while 68% of runs ended with Annual Nesters above that threshold (Table 7). The chances of falling below the lower abundance thresholds (25% and 12.5% of current abundance) were lower (28% and 25%, respectively) and had longer associated time frames (37 and 45 years, respectively). There was no discernible difference in the probabilities of falling below any of the thresholds (50%, 25%, and 12.5% of current abundance) between the “no take” and “take” scenarios (Table 7).

Table 7. Hawaii-based deep-set longline (DSLL) fishery—for North Pacific loggerheads, the probability of the population being above or below ($p > \theta$ or $p < \theta$, respectively) abundance thresholds (θ = 50%, 25%, 12.5% of current Annual Nesters) within the 100-year simulation time frame, and the number of years (mean, median, & 95% credible interval [CI]) to reach each threshold for all runs that fall below them. Results are from the stochastic take model, both with and without take, and with historical ANEs added back into the population; results from the deterministic model were not notably different. Δ(NT – T) shows the difference between the take and no take projection scenarios.

Threshold	Scenario	$p > \theta$	$p < \theta$	Mean yr	Median yr	L95% yr	U95% yr
50%	No Take	0.68	0.32	24.3	17	5	82
	Take	0.68	0.32	24.3	17	5	82
	Δ(NT – T)	0	0	0.1	0	0	0
25%	No Take	0.72	0.28	36.6	30	10	90.5
	Take	0.72	0.28	36.6	30	10	90
	Δ(NT – T)	0	0	0.0	0	0	0.5
12.5%	No Take	0.75	0.25	45.17	40	14	96
	Take	0.75	0.25	45.14	40	14	96
	Δ(NT – T)	0	0	0.03	0	0	0

The probability of the loggerhead nesting population falling below the abundance thresholds within time frames shorter than 100 years ranged from 0 (for all thresholds at 5 years) to 0.29 (for the 50% threshold at 50 years) (Table 8). For each abundance threshold, the difference between the “no take” and “take” scenarios was non-existent to negligible for the median and 95% CI probability estimates (Table 8).

Table 8. Hawaii-based deep-set longline (DSLL) fishery—for North Pacific loggerheads, the probability (median with 95% credible intervals [CI]) of the population reaching abundance thresholds at 5, 10, 25, 50, and 100 years from final data year (2015). Results are from the stochastic version of the take model with historical takes accounted for by adding the ANEs back into the population. Scenarios with and without take are provided, with ∆(NT – T) showing the difference between the two scenarios. * indicates a difference attributable to sampling stochasticity.

Threshold	Scenario	5 yr	10 yr	25 yr	50 yr	100 yr
50%	No Take	0.01	0.09	0.21	0.28	0.32
	Take	0.01	0.09	0.21	0.28	0.32
	Δ(NT – T)	0	0	0	0	0
50%-L95	No Take	0.01	0.08	0.21	0.28	0.32
	Take	0.01	0.08	0.21	0.28	0.31
	Δ(NT – T)	0	0	0	0	0.01*
50%-U95	No Take	0.01	0.09	0.21	0.29	0.32
	Take	0.01	0.09	0.21	0.29	0.32
	Δ(NT – T)	0	0	0	0	0
25%	No Take	0	0.01	0.12	0.21	0.28
	Take	0	0.01	0.12	0.21	0.28
	Δ(NT – T)	0	0	0	0	0
25%-L95	No Take	0	0.01	0.11	0.21	0.28
	Take	0	0.01	0.11	0.21	0.28
	Δ(NT – T)	0	0	0	0	0
25%-U95	No Take	0	0.01	0.12	0.22	0.29
	Take	0	0.01	0.12	0.22	0.29
	Δ(NT – T)	0	0	0	0	0
12.5%	No Take	0	0	0.06	0.16	0.25
	Take	0	0	0.06	0.16	0.25
	Δ(NT – T)	0	0	0	0	0
12.5%-L95	No Take	0	0	0.06	0.16	0.24
	Take	0	0	0.06	0.16	0.24
	Δ(NT – T)	0	0	0	0	0
12.5%-U95	No Take	0	0	0.06	0.16	0.25
	Take	0	0	0.06	0.16	0.25
	Δ(NT – T)	0	0	0	0	0

DSLL Fishery PVA Projections for Western Pacific Leatherback Turtles

Projections out 100 years for leatherbacks reflect the −6.1% annual decline (Table 5) estimated from the 2001–2017 data (Figure 13). The difference between the deterministic and stochastic models is slight (top vs. bottom panel in Figure 13), with the population declining to zero nesters 5 years sooner in the stochastic version for the DSLL fishery "take" scenario (around 2095 vs. 2102). Note that this difference between models is roughly equivalent to 1–2 annual nesters surviving 5 years longer while the population continues to decline. There is a discernible difference between the "no take" and "take" scenarios, shown by the diverging medians and shaded credible envelopes in Figure 13 (note—natural log scale is used for the 100-year projections to emphasize the differences between the two scenarios) and suggests the population would go extinct roughly 20 years sooner than in the "no take" scenario (around 2095 vs. 2115). The 20-year difference in time to extinction between the medians of the "take" and "no take" scenarios is roughly equivalent to 3–4 annual nesters persisting for 20 years longer in the "no take" while the population continues to decline along its trajectory after the "take" scenario has gone extinct. The difference between the paired "take" and "no take" projections moves slowly toward a maximum of two annual nesters around 2070 (Figure 14, bottom panel). It again should be noted that the difference plot is made by taking the difference between paired projections of the "take" and "no take" scenario (i.e., both scenarios have the same trend parameters and the sole difference between them is applying the anticipated take). The U-shaped behavior in the median of the difference between the paired "take" and "no take" scenarios results from a reduction in the difference between scenarios as the proportion of individual projections going extinct increases along the time series.

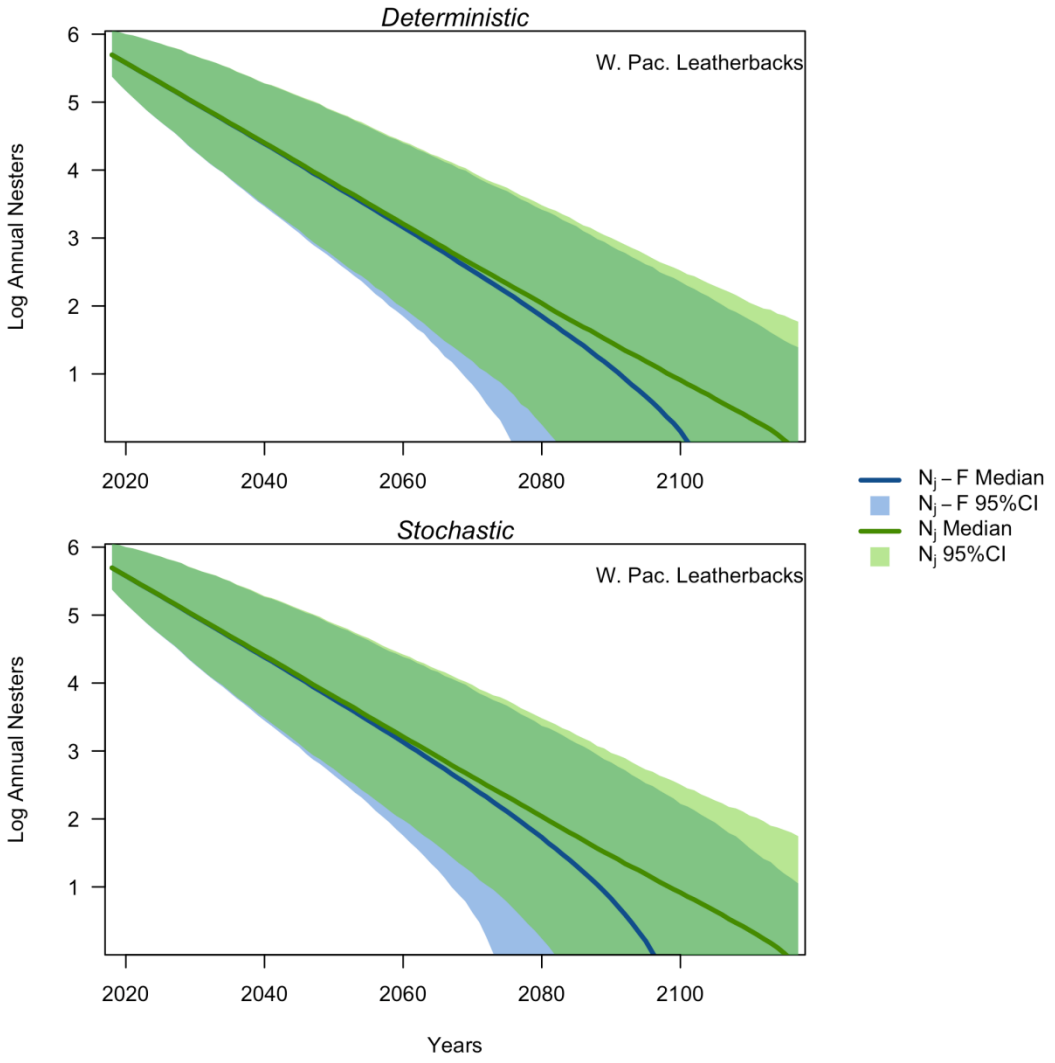

Figure 13. Hawaii-based deep-set longline (DSLL) fishery—for western Pacific leatherbacks, 100-year projections of Annual Nesters (vertical axis is in natural log units) under future scenarios including take (N_j – F) and no take (N_j). The projected trend is based on the median output scenario from the monthly nest count imputation model in Martin et al. (2020). Projections begin a year after the final year of available data (2017) and end 100 years later. Top panel shows results from a fully deterministic version of the take model in which the parameters are set at point estimates rather than pulled from distributions as they are in the stochastic version in the bottom panel.

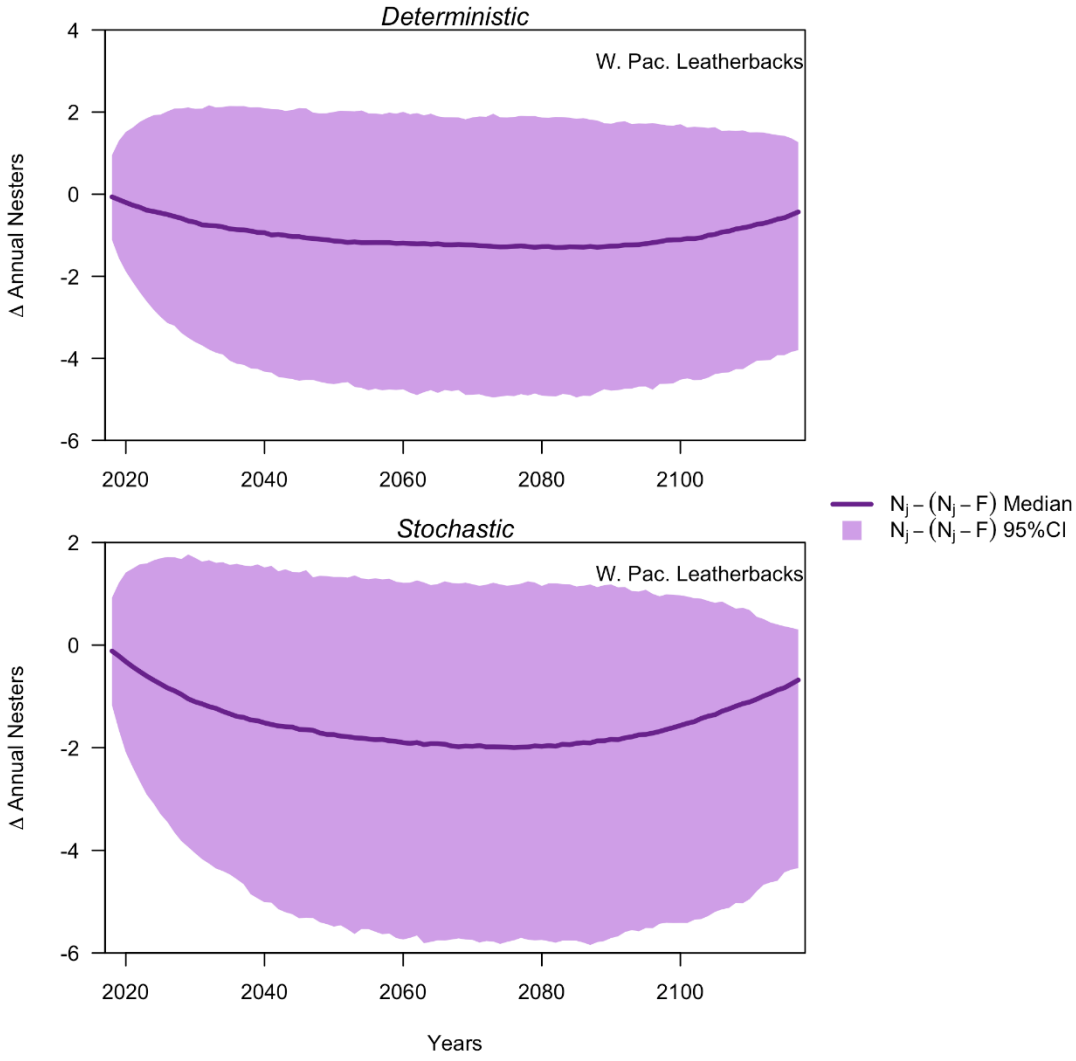

Figure 14. Hawaii-based deep-set longline (DSLL) fishery—for western Pacific leatherbacks, difference plots of 100-year paired projections of Annual Nesters under future scenarios including take (N_j – F) and no take (N_j). The projected trend is based on the median output scenario from the monthly nest count imputation model in Martin et al. (2020). Top panel shows results from a fully deterministic version of the take model in which the parameters are set at point estimates rather than pulled from distributions as they are in the stochastic version in the bottom panel.

Projections to 10 years into the future are more relevant biologically for management purposes than to 100 years given the estimated uncertainty in the population parameters. Specifically, the effects of the environmental or anthropogenic drivers on the population would be lagged; therefore, we think the first 10 years (Figure 15) is largely based on the previously observed trend, but after that, we do not have sufficient information to account for uncertainty of the drivers that affect the populations. For leatherbacks interacting with the DSLL fishery, there is a negligible difference (roughly 0.3 of an annual nester in 2027) between the deterministic and stochastic versions of the model when looking out only 10 years (Figures 15 and 16). Importantly, the difference we observed between the “no take” and “take” scenarios in the 100-

year projection is not seen in the 10-year projection (Figures 15 and 16) (note—to provide a sense of the actual magnitude change in annual nesters over the shorter time frame, we do not use the natural log scale here).

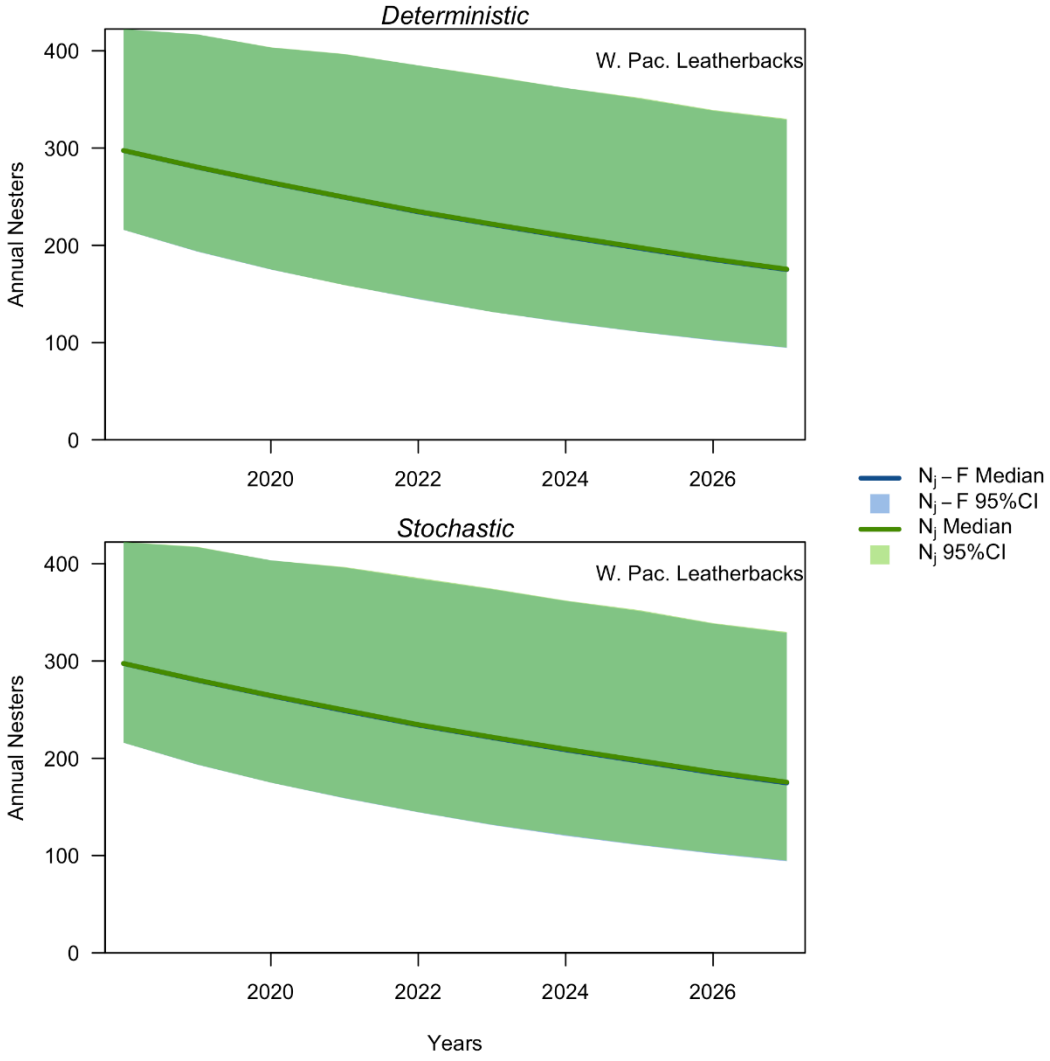

Figure 15. Hawaii-based deep-set longline (DSLL) fishery—for western Pacific leatherbacks, 10-year projections of Annual Nesters under future scenarios including take (N_j – F) and no take (N_j). The projected trend is based on the median output scenario from the monthly nest count imputation model in Martin et al. (2020). Projections begin a year after the final year of available data (2017) and end 10 years later. Top panel shows results from a fully deterministic version of the take model in which the parameters are set at point estimates rather than pulled from distributions as they are in the stochastic version in the bottom panel.

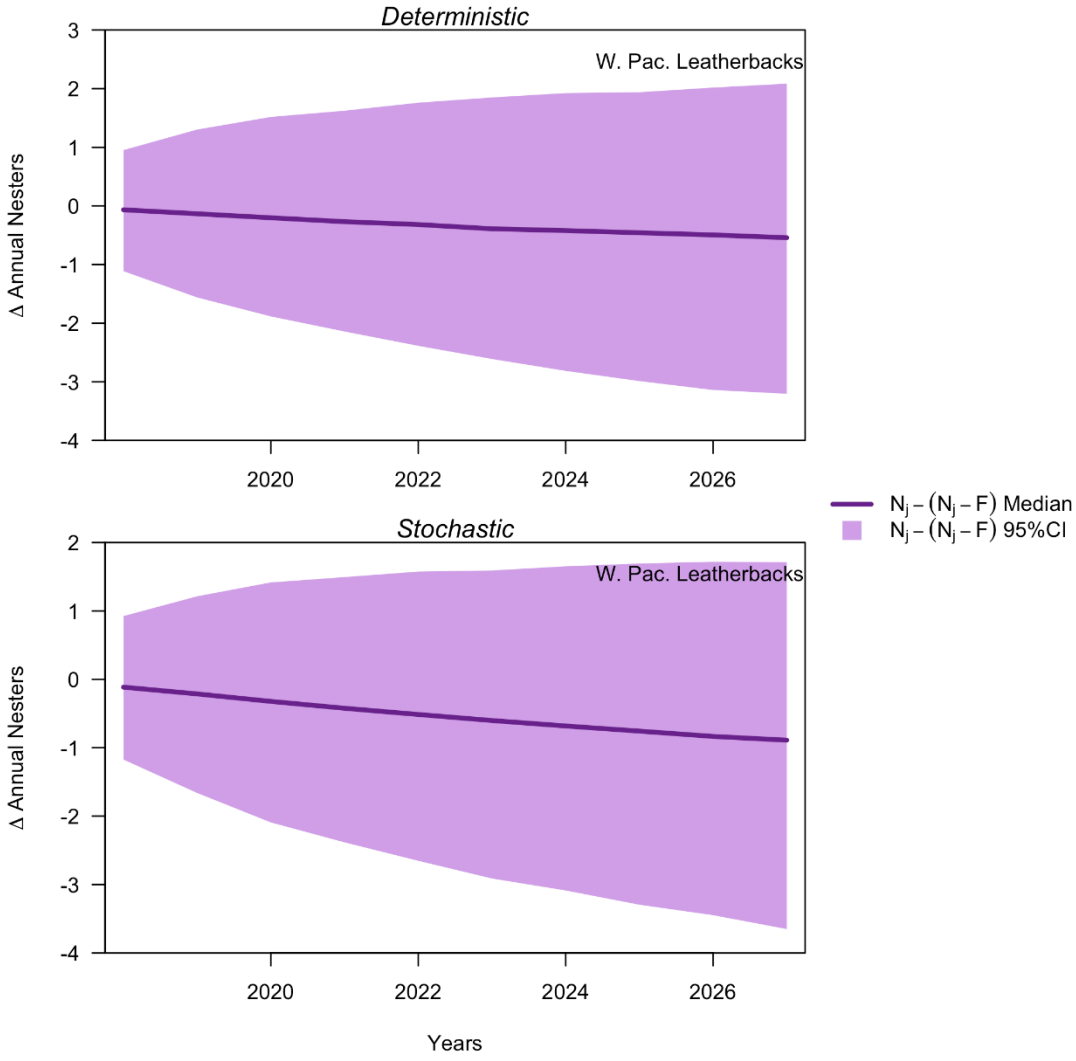

Figure 16. Hawaii-based deep-set longline (DSLL) fishery—for western Pacific leatherbacks, difference plots for 10-year paired projections of Annual Nesters under future scenarios including take (N_j – F) and no take (N_j). The projected trend is based on the median output scenario from the monthly nest count imputation model in Martin et al. (2020). Projections begin a year after the final year of available data (2017) and end 10 years later. Top panel shows results from a fully deterministic version of the take model in which the parameters are set at point estimates rather than pulled from distributions as they are in the stochastic version in the bottom panel.

For leatherbacks in the DSLL fishery, 100% of the simulation runs projected that Annual Nesters would fall below 50% of current Annual Nesters within 13 years (95% CI, 5 to 26 years) (Table 9). The chances of falling below the lower abundance thresholds (25% and 12.5% of current abundance) were also 100%, but it took longer to reach them (approximately 25 and 36 years, respectively). Comparing the “no take” and “take” scenarios, there was no discernible difference in the probability of falling below any of the abundance thresholds, but there were slight differences of less than 1 year in the three mean year estimates and 1–2 years for the 12.5%

threshold (Table 9). Those slight differences are not meaningful within the context of this analysis.

Table 9. Hawaii-based deep-set longline (DSLL) fishery—for western Pacific leatherbacks, the probability of the population being above or below ($p > \theta$ or $p < \theta$, respectively) abundance thresholds (θ = 50%, 25%, 12.5% of current Annual Nesters) within the 100-year simulation time frame, and the number of years (mean, median, & 95% credible interval [CI]) to reach each threshold for all runs that fall below them. The projected trend is based on the median output scenario from the monthly nest count imputation model in Martin et al. (2020); the low and high scenarios achieve similar results. Results are from the stochastic take model, both with and without take, and with historical ANEs added back into the population; results from the deterministic model were not notably different. Δ(NT – T) shows the difference between the take and no take projection scenarios. Results are based on the median imputed monthly count values.

Threshold	Scenario	$p > \theta$	$p < \theta$	Mean yr	Median yr	L95% yr	U95% yr
50%	No Take	0	1	12.9	12	5	26
	Take	0	1	12.8	12	5	26
	Δ(NT – T)	0	0	0.1	0	0	0
25%	No Take	0	1	24.6	23	13	42
	Take	0	1	24.3	23	13	42
	Δ(NT – T)	0	0	0.3	0	0	0
12.50%	No Take	0	1	36.3	35	22	58
	Take	0	1	35.6	34	21	56
	Δ(NT – T)	0	0	0.7	1	1	2

The probability of the leatherback nesting population falling below the abundance thresholds within time frames shorter than 100 years ranged from 0 (for 25% and 12.5% thresholds at 5–10 years) to 1 (for the 25% and 50% thresholds at 50 years) (Table 10). For each abundance threshold, the difference between the "no take" and "take" scenarios was non-existent to negligible for the median and 95% CI probability estimates (Table 10).

Table 10. Hawaii-based deep-set longline (DSLL) fishery—for western Pacific leatherbacks, the probability (median with 95% credible intervals [CI]) of the population reaching abundance thresholds at 5, 10, 25, 50, and 100 years from final data year (2017). The projected trend is based on the median output scenario from the monthly nest count imputation model in Martin et al. (2020); the low and high scenarios achieve similar results. Results are from the stochastic version of the take model with historical takes accounted for by adding the ANEs back into the population. Scenarios with and without take are provided, with ∆(NT – T) showing the difference between the two scenarios. * indicates a difference attributable to sampling stochasticity.

Threshold	Scenario	5 yr	10 yr	25 yr	50 yr	100 yr
50%	No Take	0.03	0.39	0.97	1	1
	Take	0.03	0.39	0.97	1	1
	∆(NT – T)	0	0	0	0	0
50%-L95	No Take	0.02	0.38	0.97	1	1
	Take	0.03	0.39	0.97	1	1
	∆(NT – T)	−0.01*	−0.01*	0	0	0
50%-U95	No Take	0.03	0.39	0.97	1	1
	Take	0.03	0.40	0.97	1	1
	∆(NT – T)	0	−0.01*	0	0	0
25%	No Take	0	0	0.61	0.99	1
	Take	0	0	0.62	1	1
	∆(NT – T)	0	0	−0.01*	−0.01*	0
25%-L95	No Take	0	0	0.60	0.99	1
	Take	0	0	0.62	0.99	1
	∆(NT – T)	0	0	−0.02*	0	0
25%-U95	No Take	0	0	0.61	1	1
	Take	0	0	0.63	1	1
	∆(NT – T)	0	0	−0.02	0	0
12.5%	No Take	0	0	0.10	0.93	1
	Take	0	0	0.11	0.94	1
	∆(NT – T)	0	0	−0.01*	−0.01*	0
12.5%-L95	No Take	0	0	0.10	0.92	1
	Take	0	0	0.11	0.94	1
	∆(NT – T)	0	0	−0.01*	−0.02*	0
12.5%-U95	No Take	0	0	0.10	0.93	1
	Take	0	0	0.11	0.94	1
	∆(NT – T)	0	0	−0.01*	−0.01*	0

ASLL PVA Projections for Western Pacific Leatherback Turtles

Results for the ASLL fishery are very similar to those for the DSLL fishery with some variation in the ending years (i.e., when the population falls to zero) due to the simulation nature of the analyses. Again, projections out to 100 years for leatherbacks reflect the −6.1% annual decline (Table 6) estimated from the 2001–2017 data (Figure 17). The difference between the deterministic and stochastic models is slight (top vs. bottom panel in Figure 17), with the population declining to zero nesters 5 years sooner in the stochastic version for the ASLL fishery "take" scenario (around 2092 vs. 2097). Note that this is roughly equivalent to 1–2 annual nesters surviving 5 years longer while the population continues to decline. There is a discernible difference between the "no take" and "take" scenarios, shown by the diverging medians and shaded credible envelopes in Figure 17 (note—natural log scale is used for the 100-year projections to emphasize the differences between the two scenarios) and suggests the population would go extinct roughly 20 years sooner than in the "no take" scenario (around 2092 vs. 2112). The 20-year difference in time to extinction between the medians of the "take" and "no take" scenarios is roughly equivalent to 3–4 annual nesters persisting for 20 years longer in the "no take" while the population continues to decline along its trajectory after the "take" scenario has gone extinct. The difference between the paired "take" and "no take" projections moves slowly toward a maximum of two annual nesters around 2070 (Figure 18, bottom panel) It again should be noted that the difference plot is made by taking the difference between paired projections of the "take" and "no take" scenario (i.e., both scenarios have the same trend parameters and the sole difference between them is applying the take). The U-shaped behavior in the median of the difference between the paired "take" and "no take" scenarios results from a reduction in the difference between scenarios as the proportion of individual projections going extinct increases along the time series.

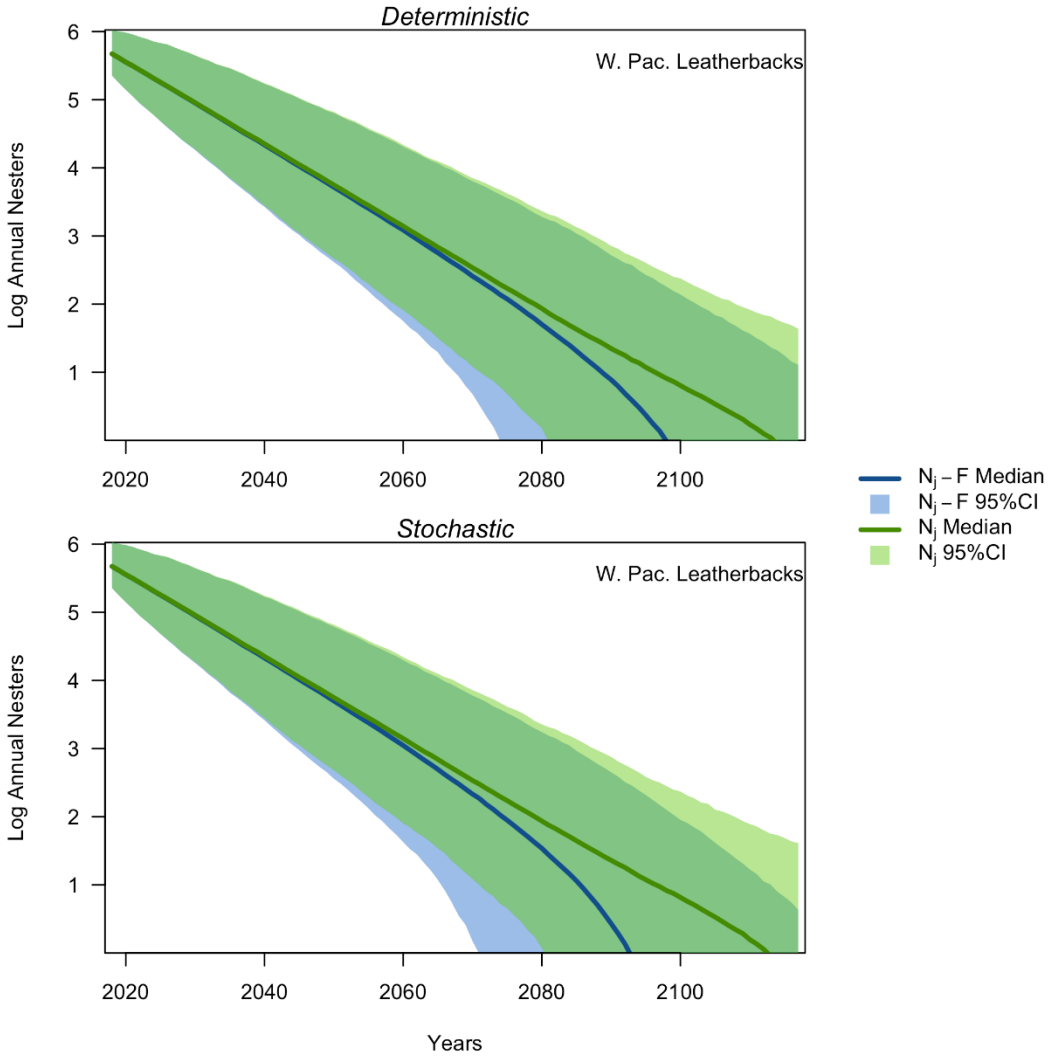

Figure 17. American Samoa-based longline (ASLL) fishery—for western Pacific leatherbacks, 100-year projections of Annual Nesters (vertical axis is in natural log units) under future scenarios including take (N_j – F) and no take (N_j). The projected trend is based on the median output scenario from the monthly nest count imputation model in Martin et al. (2020). Projections begin a year after the final year of available data (2017) and end 100 years later. Top panel shows results from a fully deterministic version of the take model in which the parameters are set at point estimates rather than pulled from distributions as they are in the stochastic version in the bottom panel.

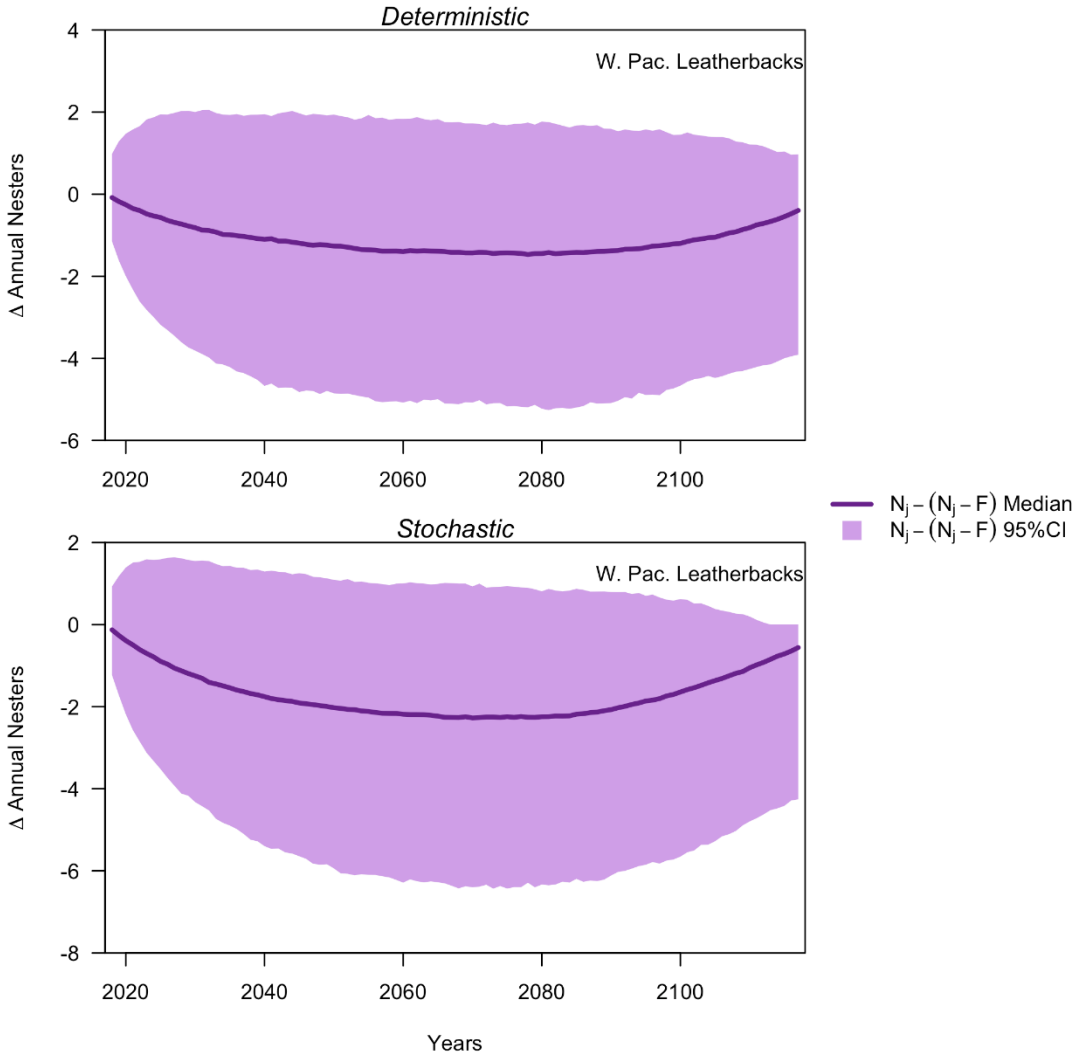

Figure 18. American Samoa-based longline (ASLL) fishery—for western Pacific leatherbacks, difference plots of 100-year paired projections of Annual Nesters under future scenarios including take (N_j – F) and no take (N_j). The projected trend is based on the median output scenario from the monthly nest count imputation model in Martin et al. (2020). Top panel shows results from a fully deterministic version of the take model in which the parameters are set at point estimates rather than pulled from distributions as they are in the stochastic version in the bottom panel.

Projections to 10 years into the future are more relevant biologically for management purposes than to 100 years given the estimated uncertainty in the population parameters. Specifically, the effects of the environmental or anthropogenic drivers on the population would be lagged; therefore, we think the first 10 years (Figure 19) is largely based on the previously observed trend, but after that, we do not have sufficient information to account for uncertainty of the drivers that affect the populations. For leatherbacks interacting with the ASLL fishery, there is a negligible difference (roughly 0.5 of an annual nester in 2027) between the deterministic and stochastic versions of the model when looking out only 10 years (Figure 20). Importantly, the difference we observed between the “no take” and “take” scenarios in the 100-year projection is

not seen in the 10-year projection (Figures 19 and 20) (note—to provide a sense of the actual magnitude change in Annual Nesters over the shorter time frame, we do not use the natural log scale here).

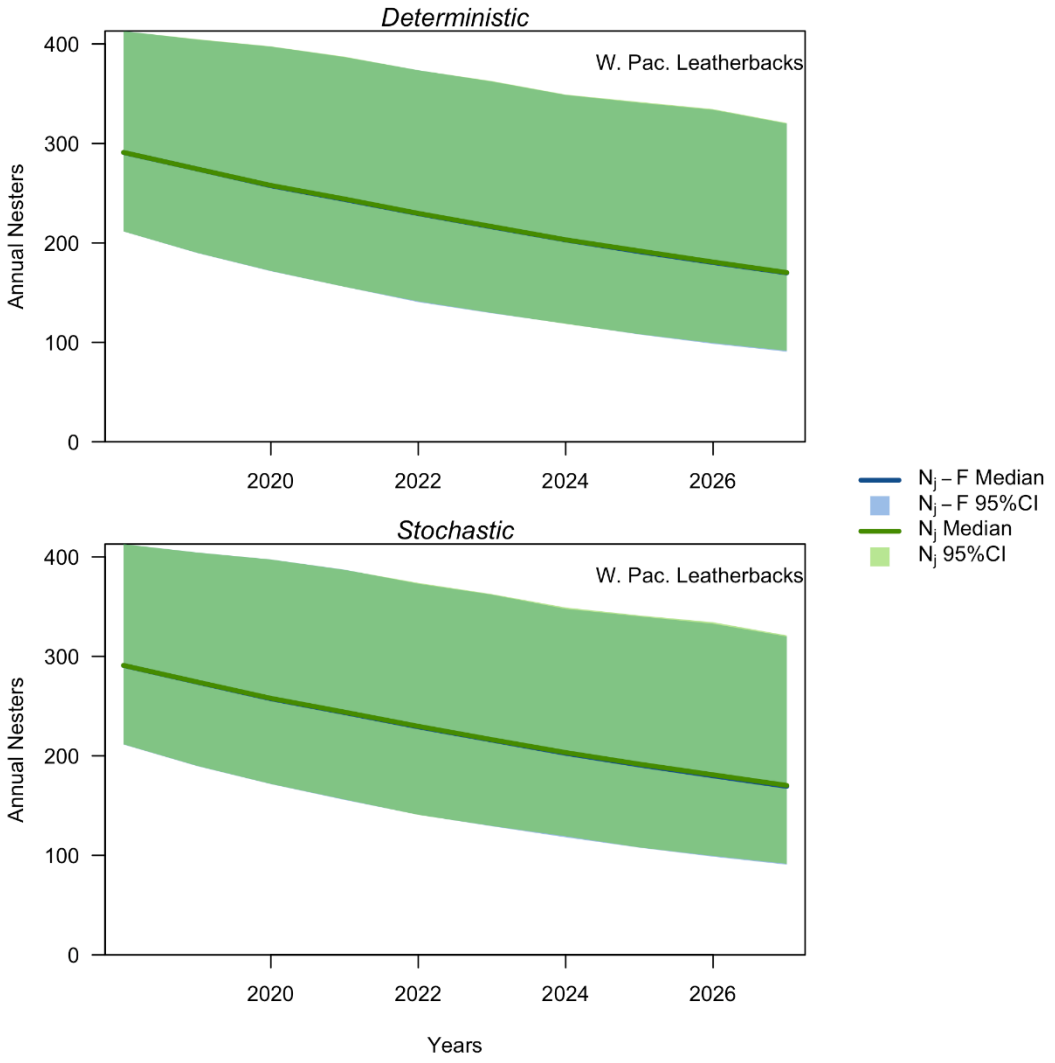

Figure 19. American Samoa-based longline (ASLL) fishery—for western Pacific leatherbacks, 10-year projections of Annual Nesters under future scenarios including take (N_j – F) and no take (N_j). The projected trend is based on the median output scenario from the monthly nest count imputation model in Martin et al. (2020). Projections begin a year after the final year of available data (2017) and end 10 years later. Top panel shows results from a fully deterministic version of the take model in which the parameters are set at point estimates rather than pulled from distributions as they are in the stochastic version in the bottom panel.

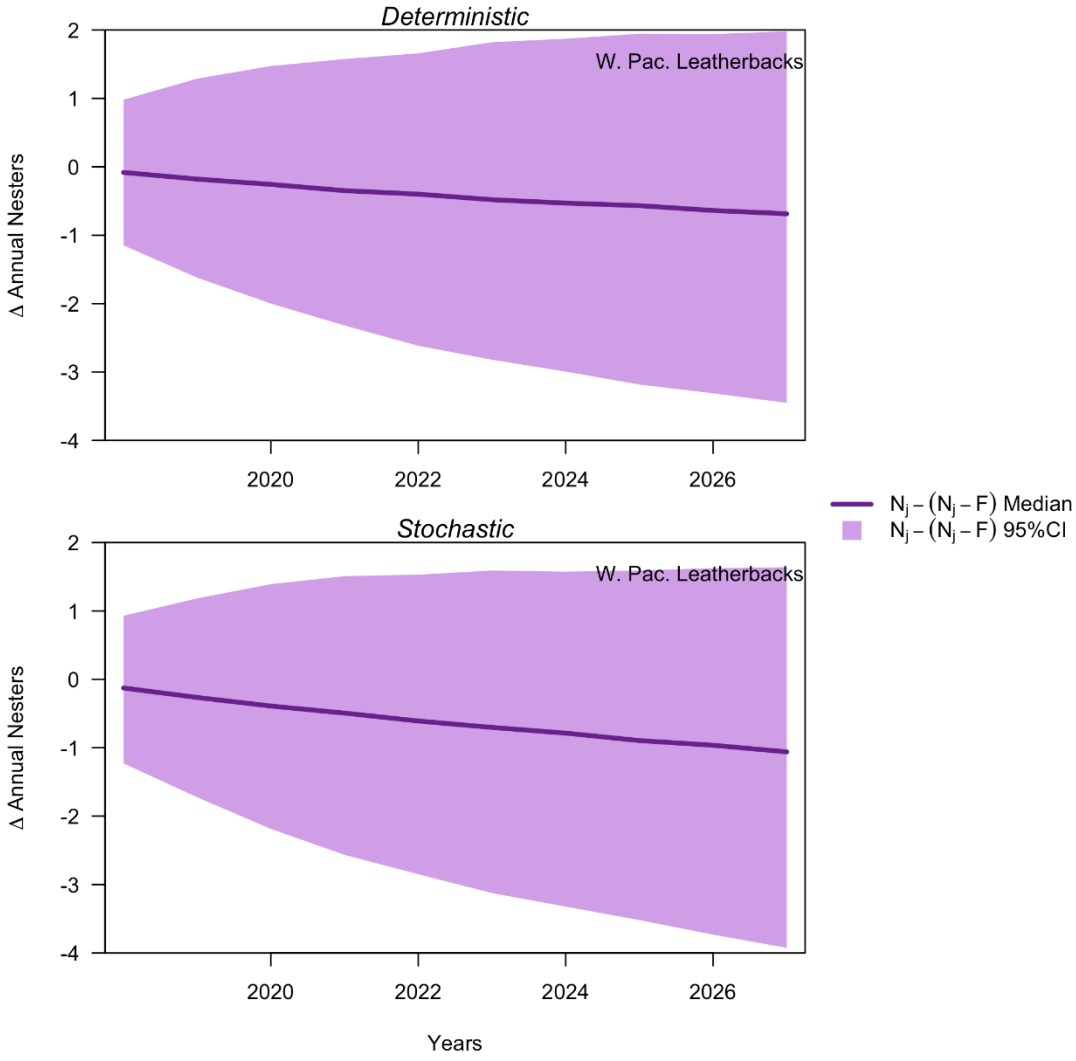

Figure 20. American Samoa-based longline (ASLL) fishery—for western Pacific leatherbacks, difference plots for 10-year paired projections of Annual Nesters under future scenarios including take (N_j – F) and no take (N_j). The projected trend is based on the median output scenario from the monthly nest count imputation model in Martin et al. (2020). Projections begin a year after the final year of available data (2017) and end 10 years later. Top panel shows results from a fully deterministic version of the take model in which the parameters are set at point estimates rather than pulled from distributions as they are in the stochastic version in the bottom panel.

For leatherbacks in the ASLL fishery, 100% of the simulation runs projected that Annual Nesters would fall below 50% of current Annual Nesters within 13 years (95% CI, 5 to 26 years) (Table 11). The chances of falling below the lower abundance thresholds (25% and 12.5% of current abundance) were also 100%, but it took longer to reach them (approximately 24 and 36 years, respectively). Comparing the “no take” and “take” scenarios, there was no discernible difference in the probability of falling below any of the abundance thresholds, but there were slight differences of less than 1 year in the three mean year estimates and 1–2 years for the 12.5%

threshold (Table 11). Those slight differences are not meaningful within the context of this analysis.

Table 11. American Samoa-based longline (ASLL) fishery—for western Pacific leatherbacks, the probability of the population being above or below ($p > \theta$ or $p < \theta$, respectively) abundance thresholds (θ = 50%, 25%, 12.5% of current Annual Nesters) within the 100-year simulation time frame, and the number of years (mean, median, & 95% credible interval [CI]) to reach each threshold for all runs that fall below them. The projected trend is based on the median output scenario from the monthly nest count imputation model in Martin et al. (2020); the low and high scenarios achieve similar results. Results are from the stochastic take model, both with and without take, and with historical ANEs added back into the population; results from the deterministic model were not notably different. Δ(NT-T) shows the difference between the take and no take projection scenarios.

Threshold	Scenario	$p > \theta$	$p < \theta$	Mean yr	Median yr	L95% yr	U95% yr
50%	No Take	0	1	12.7	12	5	26
	Take	0	1	12.6	12	5	25
	Δ(NT – T)	0	0	0.1	0	0	1
25%	No Take	0	1	24.1	23	13	41
	Take	0	1	23.8	23	13	40
	Δ(NT – T)	0	0	0.3	0	0	1
12.50%	No Take	0	1	35.7	35	22	56
	Take	0	1	34.8	34	21	54
	Δ(NT – T)	0	0	0.9	1	1	2

The probability of the leatherback nesting population falling below the abundance thresholds within time frames shorter than 100 years ranged from 0 (for 25% and 12.5% thresholds at 5–10 years) to 1 (for the 25% and 50% thresholds at 50 years) (Table 12). For each abundance threshold, the difference between the “no take” and “take” scenarios was non-existent to negligible for the median and 95% CI probability estimates (Table 12).

Table 12. American Samoa-based longline (ASLL) fishery—for western Pacific leatherbacks, the probability (median with 95% credible intervals [CI]) of the population reaching abundance thresholds at 5, 10, 25, 50, and 100 years from final data year (2017). The projected trend is based on the median output scenario from the monthly nest count imputation model in Martin et al. (2020); the low and high scenarios achieve similar results. Results are from the stochastic version of the take model with historical takes accounted for by adding the ANEs back into the population. Scenarios with and without take are provided, with ∆(NT-T) showing the difference between the two scenarios. * indicates a difference attributable to sampling stochasticity.

Threshold	Scenario	5 yr	10 yr	25 yr	50 yr	100 yr
50%	No Take	0.03	0.39	0.97	1	1
	Take	0.03	0.39	0.97	1	1
	∆(NT – T)	0	0	0	0	0
50%-L95	No Take	0.02	0.38	0.97	1	1
	Take	0.03	0.39	0.97	1	1
	∆(NT – T)	−0.01*	−0.01*	0	0	0
50%-U95	No Take	0.03	0.39	0.97	1	1
	Take	0.03	0.40	0.97	1	1
	∆(NT – T)	0	−0.01*	0	0	0
25%	No Take	0	0	0.61	0.99	1
	Take	0	0	0.62	1	1
	∆(NT – T)	0	0	−0.01*	−0.01*	0
25%-L95	No Take	0	0	0.60	0.99	1
	Take	0	0	0.62	0.99	1
	∆(NT – T)	0	0	−0.02*	0	0
25%-U95	No Take	0	0	0.61	1	1
	Take	0	0	0.63	1	1
	∆(NT – T)	0	0	−0.02*	0	0
12.5%	No Take	0	0	0.10	0.93	1
	Take	0	0	0.11	0.94	1
	∆(NT – T)	0	0	−0.01*	−0.01*	0
12.5%-L95	No Take	0	0	0.10	0.92	1
	Take	0	0	0.11	0.94	1
	∆(NT – T)	0	0	−0.01*	−0.02*	0
12.5%-U95	No Take	0	0	0.10	0.93	1
	Take	0	0	0.11	0.94	1
	∆(NT – T)	0	0	−0.01*	−0.01*	0

Discussion

The purpose of this analysis was to apply the take model PVA approach developed by Martin et al. (2020) for the SSLL fishery to the DSLL and ASLL fisheries to assess the impacts of those fisheries on the western Pacific leatherback and North Pacific loggerhead populations. The approach included estimation of long-term annual trends (i.e., population growth rates) and current abundance (number of nesting females) for the populations using a Bayesian state-space population growth model. Those estimates were informed by time series of nest count data from index beaches in Indonesia (leatherbacks, 2001–2017) and Japan (loggerheads, 1985–2015), with missing monthly nest counts for leatherbacks imputed as described in Martin et al. (2020). For each population, we produced estimates corresponding to three scenarios: (1) the "true" state of the populations (i.e., inclusive of all past threats acting on the populations since the start of the nest count time series); (2) DSLL-adjusted to remove the impact of the DSLL fishery from 2005 to the end of the nest count time series; and (3) ASLL-adjusted to remove the impact of the ASLL fishery from 2006 to the end of the nest count time series. We also include the SSLL-adjusted scenario from Martin et al. (2020) in a comprehensive table with all four scenarios.

The purpose of removing the historical fishery impacts in the fishery-adjusted scenarios was to avoid double-counting the impact of a fishery in the future, as the true historical trend reflects all threats acting on the population. The estimates produced within the fishery-specific analyses were useful, with the trend driving the projections and the abundance serving as a reference point for assessment of the future population falling below specified thresholds. Outside of the fishery-specific analyses, however, those estimates do not reflect the "true" state of the population due to the removal of historical fishery impacts. Only the estimates in the "true" state scenario should be referenced with respect to the status of the populations.

The "true" state results indicated a declining trend for leatherbacks (−6.1% annually; 95% CI, −24.0% to 12.3%; from median imputed nest counts) and an increasing trend for loggerheads (2.3% annually; 95% CI, −11.0% to 15.7%). For loggerheads, current abundance was estimated at 4,538 total nesters (95% CI, 4,077 to 5,064 total nesters) for the three index beaches in Japan, which represent 52% of all nesting. For leatherbacks, the estimate of current abundance (from median imputed nest counts) for the two index beaches in Indonesia, which represent 75% of all nesting was 787 total nesters (95% CI, 659 to 939 total nesters). There were no notable differences between the trends for the "true" state and fishery-adjusted scenarios for either species.

Results of the PVAs, which included comparisons of future "no take" and "take" scenarios, were essentially the same for both the DSLL and ASLL fisheries for leatherbacks; only the DSLL fishery had past loggerhead interactions. PVAs projecting the trends 100 years into the future suggested a 100% chance of leatherbacks falling below 50% of their current abundance, with a mean of 13 years to reach that threshold. Loggerheads had a 32% chance of falling below the 50% abundance threshold, with a mean of 24 years to reach that threshold. At 10 years in the future, the chance of falling below the 50% abundance threshold was 39% for leatherbacks and 9% for loggerheads. There were no notable changes to the probabilities of falling below abundance thresholds (50%, 25%, and 12.5% of current abundance) when comparing the "no take" and "take" scenarios for the future.

The most noteworthy finding was that for leatherbacks, the difference between the “no take” and “take” scenarios in the 100-year projections became apparent after 2060 and the projections suggested the population would go extinct roughly 20 years sooner in the “take” scenario than in the “no take” scenario (around 2095 vs. 2115 for the DSLL fishery and 2092 vs. 2112 for the ASLL fishery). It is important to note that this difference between the scenarios occurs when the population falls below 20 adult nesters. However, there is little difference between the two scenarios regarding when the population will reach 20 adult nesters (i.e., 43 years from the starting point). Additionally, the 20-year difference in time to extinction between the medians of the “take” and “no take” scenarios is roughly equivalent to 3–4 annual nesters persisting for 20 years longer in the “no take” while the population continues to decline along its trajectory after the “take” scenario has gone extinct. However, it should be noted that for a particular assumption of the leatherback population growth rate, the maximum median difference between the “take” and “no take” scenarios was approximately 2 adult nesters for the DSLL and ASLL. In the 10-year future time frame, which is perhaps more biologically relevant to use for impact assessments, there was no notable difference between the “no take” and “take” projection scenarios for either species.

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Appendix: Parameter Estimates and Sources

Sp	Param	Description	Estimate	SE	Low	Upp	Pop	Source	Justification/Notes
Dc	Linf	VBGF: average size of nesting females (SCL in cm)	142.7	1.64			Mix	Jones et al. (2011) *JEMB*	best available growth model, both globally and for W. Pacific specifically
Dc	k	VBGF: Brody growth coefficient in VBGF	0.2262	0.021			Mix	Jones et al. (2011) *JEMB*	best available growth model, both globally and for W. Pacific specifically
Dc	t0	VBGF: hypothetical age animals would be length = 0	−0.17	0.07			Mix	Jones et al. (2011) *JEMB*	best available growth model, both globally and for W. Pacific specifically
Dc	Amat	VBGF: age at maturity (97.5% of Linf in VBGF)	16.1				Mix	Jones et al. (2011) *JEMB*	best available growth model, both globally and for W. Pacific specifically (Age at maturity confirmed recently with skeletochronology by L. Avens et al. (2020) in *Marine Biology*)
Dc	SCL_slo	slope: SCL conversion to CCL	1.04					Tucker & Frazer (1991) *Herpetologica*	best available but didn't end up needing, as no CCL measurements for Dc
Dc	SCL_int	intercept: SCL conversion to CCL	2.04					Tucker & Fraser (1991) *Herpetologica*	best available but didn't end up needing, as no CCL measurements for Dc
Dc	CF	clutch frequency = # of nests laid by a female in a season	5.5	SD 1.6	3	10	W. Pac	Tapilatu et al. (2013) *Ecosphere*	most recent estimate of clutch frequency
Dc	CS	clutch size = # of eggs laid in one nest	77.9	2.35			W. Pac	Tapilatu & Tiwari (2007) *Copeia*	most recent estimate of clutch size using direct count observations
Dc	RI	remigration interval = # of years between nesting years	3.06				W. Pac	derived from Lontoh (2014) master's thesis	most recent estimate; specific to CNP foraging area; estimated Poisson scale parameter was 2.36

Sp	Param	Description	Estimate	SE	Low	Upp	Pop	Source	Justification/Notes
Dc	Pj	survival of juveniles (probability from one age to next)	0.81	0.03			W. Pac	Jones et al. (2012) *PLoS One*	best available; based on population growth model calculating natural mortality based on known age at maturity
Dc	Pa	survival of adults (probability from one age to next)	0.893	0.013			CB	Dutton et al. (2005) *Biol Cons*	Jones et al. (2012) was too low due to conflating fisheries & natural mortality; thus, use Dutton et al. (2005) estimate as best available for a growing population at the time so probably a better estimate of what Pa actually is
Dc	PF	proportion female = sex ratio	0.73					Benson et al. (2011) *Ecosphere*	using Benson et al 2011 (Ecosphere) 27:10 female:male (27 females and 10 males captured/sat-tagged) as best (only) available estimate even though CA foragers different foraging stock and size ranges that are not necessarily representative of Dc taken in SSLL
Cc	CCL_slo	slope: CCL conversion to SCL	0.9084					SM model of Zug (1995)	best available SCL/CCL measurements data for CNP & specific sizes that interact w/ SSLL
Cc	CCL_int	intercept: CCL conversion to SCL	0.0303					SM model of Zug (1995)	best available SCL/CCL measurements data for CNP & specific sizes that interact w/SSLL

Sp	Param	Description	Estimate	SE	Low	Upp	Pop	Source	Justification/Notes
Cc	Pj	survival of juveniles (probability from one age to next)	0.8	0.031	0.74	0.86	Mix	Snover (2008) PIFSC IR-08-010; Conant et al. (2009)	these two papers are best available and generally accepted survival rates for the parameter; Conant et al. (2009) had mortality rates estimated from population model; Snover (2008) range includes Conant et al. (2009)
Cc	Pa	survival of adults (probability from one age to next)	0.895	0.028	0.84	0.95	Mix	Snover (2008) PIFSC IR-08-010; Conant et al. (2009)	best available and generally accepted survival rates for the parameter
Cc	PF	proportion female = sex ratio	0.65				Mix	Snover (2008) PIFSC IR-08-010; Conant et al. (2009)	best available and generally accepted sex ratio and consistent with expert consensus for this population given lack of empirical data; Conant et al. (2009) had mortality rates estimated from population model; Snover (2008) range includes Conant et al. (2009)
Cc	CF	clutch frequency = # of nests laid by a female in a season	4.6	SD 1.1			NP	Hatase et al. (2013) *Ecology*	most recent available; study based on Yakushima nesters which is the driver of NP pop (52% for 3 beaches in our analysis)
Cc	CS	clutch size = # of eggs laid in one nest	122	SD 18.4	83.5	148	NP	Hatase et al. (2013) *Ecology*	most recent available; study based on Yakushima nesters which is the driver of NP pop (52% for 3 beaches in our analysis)
Cc	RI	remigration interval = # of years in between nesting years	3.3	2.3	1	10	NP	Hatase et al. (2013) *Ecology*	most recent available; study based on Yakushima nesters which is the driver of NP pop (52% for three beaches in our analysis)

Sp	Param	Description	Estimate	SE	Low	Upp	Pop	Source	Justification/Notes
Cc	Linf	VBGF: average size of nesting females (SCL in cm)	86.9				NP	ZS model from data in Turner-Tomaszewicz (2015) *Biol Cons*	best available data used to construct VBGF model
Cc	k	VBGF: Brody growth coefficient in VBGF	0.09				NP	ZS model from data in Turner-Tomaszewicz (2015) *Biol Cons*	best available data used to construct VBGF model
Cc	t0	VBGF: hypothetical age animals would be length = 0	−2.467				NP	ZS model from data in Turner-Tomaszewicz (2015) *Biol Cons*	best available data used to construct VBGF model
Cc	Amat	Model generated. VBGF: age at maturity. # of ages as a juvenile (to get to Amat)	37.9				NP	ZS model of Loggerhead growth; Zug (1995) data, Turner-Tomascewiz (2015) *Biol Cons* data, & Hatase et al. (2002) *STAJ* nesters data	using 85 cm ± 4.5 cm (SD) as average nesting SCL length (Linf) from Hatase et al. (2002) *MEPS*
Cc	Linf_L0	VBGF: average size of nesting females (SCL in cm)	80.4474				NP	ZS model from data in Turner-Tomaszewicz (2015) *Biol Cons*	best available data used to construct VBGF model
Cc	k_L0	VBGF: Brody growth coefficient in VBGF	0.1396				NP	ZS model from data in Turner-Tomaszewicz (2015) *Biol Cons*	best available data used to construct VBGF model
Cc	L0	VBGF: length at birth (hatching)	4.7363				NP	ZS model from data in Turner-Tomaszewicz (2015) *Biol Cons*	best available data used to construct VBGF model
Cc	Amat	VBGF: age at maturity. # of ages as a juvenile (to reach Amat)	26.4951				NP	ZS model of Loggerhead growth; Zug (1995) data, Turner-Tomascewiz (2015) *Biol Cons* data, & Hatase et al. (2002) STAJ nesters data	using 85 cm ± 4.5 cm (SD) as average nesting SCL length (Linf) from Hatase et al. (2002) *MEPS*

Sp—species
Param—parameter
SE—standard error (except where SD is indicated for standard deviation)
Low—lower value if a range is known
Upp—upper value if a range is known
Pop—population of leatherback turtles or loggerhead turtles
Dc—Dermochelys coriacea (leatherback sea turtle)
Cc—Caretta caretta (loggerhead sea turtle)
Pop "Mix"—mixed population of leatherback turtles; not only western Pacific nesting population; or mix of loggerhead turtles
Pop "CB"—Caribbean nesting population of leatherback turtles
Pop "W. Pac"—western Pacific nesting population of leatherback turtles
Pop "N. Pac"—portion of western Pacific nesting population of leatherback turtles found foraging in the North Pacific
Pop "NP"—North Pacific DPS of loggerhead turtle
CNP—Central North Pacific foraging area used by western Pacific leatherback turtles
ZS—Zach Siders (coauthor on this report; produced loggerhead growth model used herein)
VBGF—von Bertalanffy growth function
CCL—curved carapace length
SCL—straight carapace length