Stock assessment of saddletail snapper (*Lutjanus malabaricus*) in east coast, Queensland, Australia, with data to June 2025

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Queensland Government

This publication has been compiled by A.R. Fox and L.I. Sumpter of Fisheries Queensland, Department of Primary Industries

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Contents

Executive summary

Introduction

Saddletail snapper (*Lutjanus malabaricus*) are a key species targeted by the Reef Line Fishery, which operates along Queensland's east coast, predominantly within the Great Barrier Reef Marine Park. The fishery is generally considered to have commenced in 1961 and extends from the northern tip of Cape York to south of Brisbane. Saddletail snapper have separate sexes, a lifespan of up to 49 years and can grow to a maximum weight of 12 kg and a total length of 101 cm. They can reach maturity as early as three years of age, with the proportion of mature individuals increasing as they age and grow. Half of the population are expected to be mature by the time they grow to 57 cm fork length.

This stock assessment uses a two-sex age-structured population model to determine the spawning biomass at the beginning of the 2026 financial year (i.e. beginning 1 July 2025). In this assessment, the stock was assumed to be in an unfished state in 1961. The term 'biomass' refers to the total weight of a biological population or stock. 'Spawning biomass' refers to the total weight of mature or reproductively capable individuals within the stock. The 'spawning biomass ratio' is the key biomass estimate presented in this stock assessment. This ratio is simply the current spawning biomass as a percentage of the unfished spawning biomass (i.e. the estimated spawning biomass in 1961).

This stock assessment estimates that the biomass at the beginning of 2026 was between 19% and 42% of unfished levels. The most probable estimate within this range is 28% of unfished levels, which represents the median value.

Several key aspects of this assessment differ from the previous saddletail snapper assessment:

- The membership of the project team that guides the assessment was expanded, to include fishery stakeholders (one commercial, one recreational and one Traditional Owner fisher) as well as an independent scientist and a representative from the Great Barrier Reef Marine Park Authority.
- The catch reconstruction accounted for additional mortality as a result of depredation.
- Five additional years of catch, effort, length and age data were included (i.e. 2021–2025).
- Catch rate indices were adjusted for long term changes in fishing efficiency.
- Recreational catch rates were included as an additional index of abundance, spanning 2017–2025.
- Recreational harvest beyond 2020 was estimated using observed changes of indices of recreational fishing effort and recreational catch rates.
- The key outputs (including biomass estimates) were constructed from an ensemble across multiple plausible model scenarios, rather than selecting a particular preferred scenario. These models used a Markov chain Monte Carlo estimation framework rather than maximum likelihood estimation.

Methods

The assessment used Stock Synthesis to construct a two-sex age-structured model with an annual time step, fitted to standardised catch rates, length composition data and age-at-length composition data. The model incorporated data from Queensland east coast waters spanning the period 1961–2025, collected from the commercial, recreational, charter and Traditional Owners sectors. A full listing of all data inputs and sources is given in Table 2.1, with a description of each in Section 2.1.

This assessment modelled the east coast saddletail snapper population as a single genetic stock. However project team members flagged the regional differences in fishing behaviour along the Queensland coast. Separate modelling of age, length and sex data was conducted to investigate and identify any spatio-temporal patterns in these data. No evidence was found that would warrant a more spatially-complex stock assessment, given the data available (see Section 2.7). Still, regional variation was captured through the catch rate standardisation process and the relative weighting of length composition data. A full list of model assumptions can be found in Section 2.8.3.

All assessment inputs and outputs are referenced on a financial year basis, where ‘2025’ means ‘July 2024–June 2025’.

Unlike the previous assessment, this assessment uses an *ensemble* approach to combine the outcomes from a suite of 12 chosen models. The previous stock assessment promoted the outputs from a single ‘base case’ model chosen from the suite of models based on a consensus of the opinions of experts in the project team. The ensemble approach, used here, allocates equal weighting to each of the 12 models and combines their outputs. This reduces reliance on experts correctly choosing a single ‘most likely’ model among the suite of realistic models. This ensemble approach was also used in recent stock assessments of king threadfin and Spanish mackerel (Campbell et al. 2024; Sumpter et al. 2025).

Numerous scenarios were run to explore different assumptions, including variations of the natural mortality rate, steepness, data weighting, depredation level, historical commercial catch adjustment, fishing efficiency change and fishing selectivity approach. From these exploratory scenarios, an ensemble of 12 final scenarios was chosen for inclusion in summary reporting, covering two values of natural mortality (fixed at 0.11 and 0.14 yr^{-1}), three values of steepness (0.65, 0.75 and 0.85) and two different methods for the relative weighting of length composition data (described in Section 2.6.2). All results were produced from the final ensemble of 12 scenarios, where each scenario was run using a Markov chain Monte Carlo framework, for 250 000 iterations per scenario.

This report includes discussion on the influence of the exploratory scenarios which were not included in the ensemble. These scenario results are retained for comparison in Appendix B.

Results

Biomass

The results of this assessment (Table 1 and Figure 1) indicate that at the start of 2026, the biomass was between 19% and 42% of unfished levels, with 95% confidence that the value falls within this range (95% credible interval). The median biomass ratio estimate is 28%, meaning this is the most probable estimate of saddletail snapper biomass.

The median is preferred as a central tendency measure (over the mean or mode, for example) as it is less prone to the influence of outliers or extreme values, making it a more reliable measure for skewed distributions, such as the probability distribution of 2026 biomass.

We also report the probability of the estimated 2026 biomass ratio falling into four categories—below 20%, between 20% and 40%, between 40% and 60%, and above 60%. In addition, we report the estimated fishing mortality rate in 2025, relative to F_{60}—that is, the fishing mortality rate required to maintain biomass at 60% of unfished levels. This result represents the entire ensemble and is inclusive of fishing mortality from depredation. The fishing mortality estimate for 2025 exceeds that which would sustain the population at 60% biomass.

Table 1: Stock status indicators for Queensland east coast saddletail snapper

Indicator	Value
Biomass ratio (relative to unfished)	
Median	28%
Range (95% credible interval)	19–42%
Probability below 20%	5%
Probability between 20% and 40%	91%
Probability between 40% and 60%	4%
Probability above 60%	0%
Fishing pressure ratio (relative to F_{60})	
Median	3.85
Range (95% credible interval)	2.48–5.84

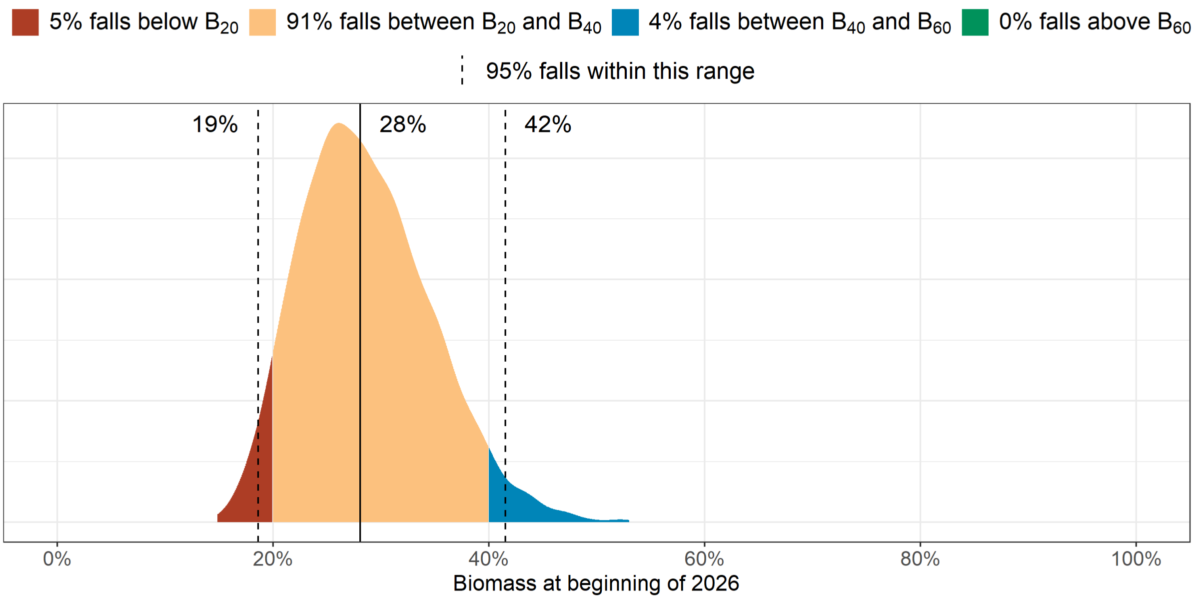

Figure 1: Probability distribution of the biomass ratio of Queensland east coast saddletail snapper at the beginning of 2026 across the full ensemble of scenarios with the credible interval and probability of biomass falling into the four categories indicated

The initial assessment of this stock in 2021 estimated the biomass ratio at the beginning of 2021 to be 23%, with a confidence range of 13–73% (Campbell et al. 2021). Now, the range of uncertainty around the final biomass estimate has reduced to 19–42% in this current assessment, including model outputs from the full ensemble. This is a significant improvement on estimate confidence.

The biomass trajectory (Figure 2) shows a decline from an assumed unfished level in 1961, until around 2016 after which the median value has slowly increased.

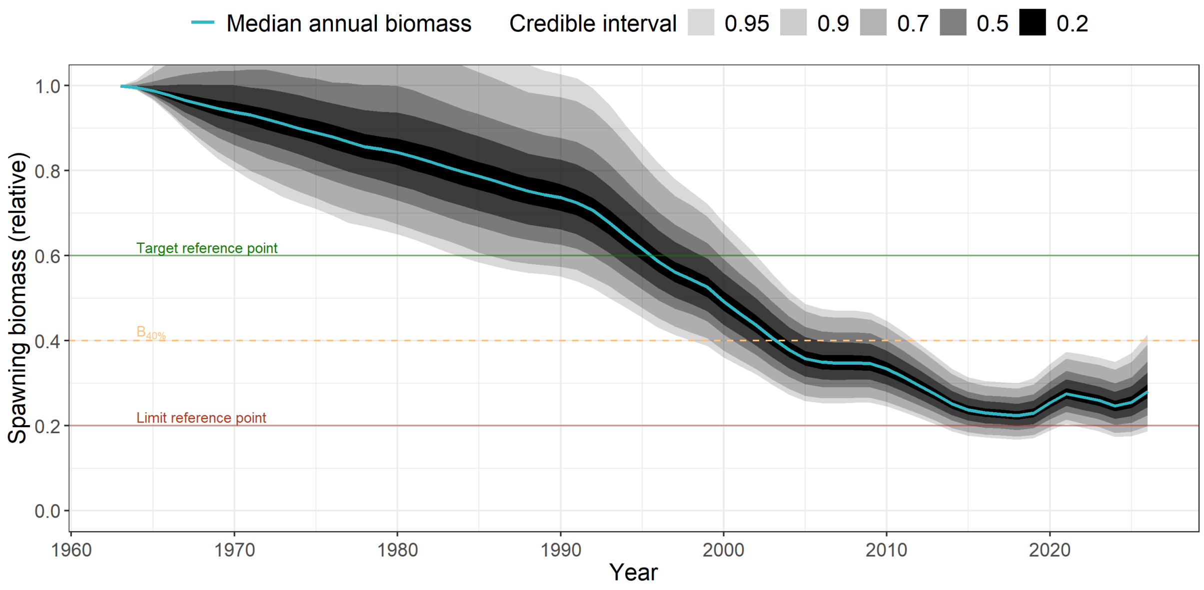

Figure 2: Predicted spawning biomass trajectory relative to unfished levels (1961) for Queensland east coast saddletail snapper, from Markov chain Monte Carlo ensemble scenarios

Catch

Over the last 5 years (2021 to 2025), total retained catch (from all sectors combined) averaged 376 tonnes per year from Queensland east coast waters (Figure 3).

Depredation data were used in the catch reconstruction to account for the additional mortality of fish lost to sharks and other predators in Queensland waters. The base rate used for depredation in 2025 was 58% for recreational, charter and Traditional Owner catch. For commercial boats, half of this rate was used (29%). Depredation rates were assumed to be a constant 1% between 1961 and 2009. Between 2009 and 2025, the depredation rates increased linearly. For full detail on these methods, see Section 2.2.5. Additional modelling was done to explore the impact of alternative depredation rates, and results indicated that depredation levels had a minimal impact on the 2026 biomass rate (Appendix B.1).

When including depredated catch, the total modelled 'retained catch' (retained by either fishers or underwater predators) in 2024–25 was 594 tonnes.

Before depredation was accounted for, approximately 25% of catch in 2024–25 was taken by the commercial sector, 68% by the recreational and Traditional Owners sectors combined, and 7% by the charter sector. When factoring in depredation rates, these values become 19% for the commercial sector, 73% for the recreational and Traditional Owners sectors combined, and 8% for the charter sector.

A full breakdown of catch shares—with and without depredated catch—can be found in Appendix E.1

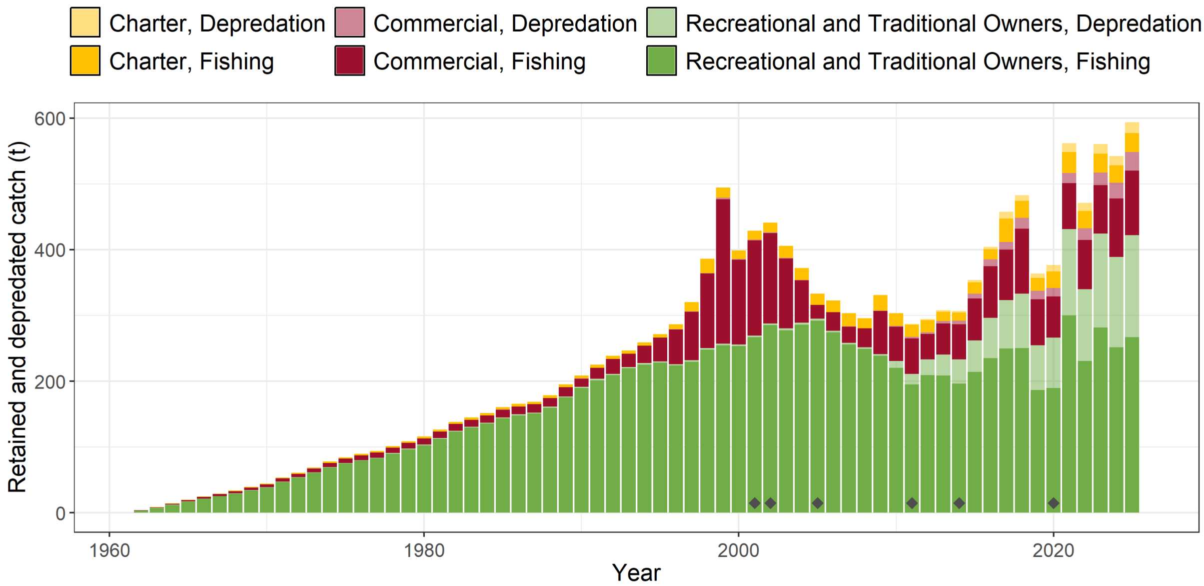

Figure 3: Estimated catch between 1961 and 2025 for Queensland east coast saddletail snapper–lighter sharded areas represent the estimated additional mortality due to depredation, diamonds represent years where statewide recreational fishing surveys were conducted

Catch rates

Catch rates from both the commercial and recreational sectors were standardised, using statistical methods, to estimate two annual indices of abundance of saddletail snapper through time (Figure 4). The response variable for the commercial catch rates model was based on the kilograms of saddletail snapper caught per daily fishing operation, which may comprise a primary vessel and multiple dories (collectively quantified as "number of boats"). The commercial catch rate model included terms for year, month, latitude, lunar phase, number of boats, wind speed, targeting behaviour and an identifier for the fishing operator. The response variable for the recreational catch rates model was based on the number of saddletail snapper caught per fishing trip. The recreational catch rate model included terms for year, region, month, number of fishers, length of trip, wind speed and whether fishing occurred on a weekday or weekend. The standardised catch rates for both sectors were adjusted to account for fishing efficiency gains. As a baseline, we assumed a 2% efficiency gain each year, except in recent years (2021–2025) where this was reduced to 1% as the prevalence of sharks has been reported to reduce fishing efficiency.

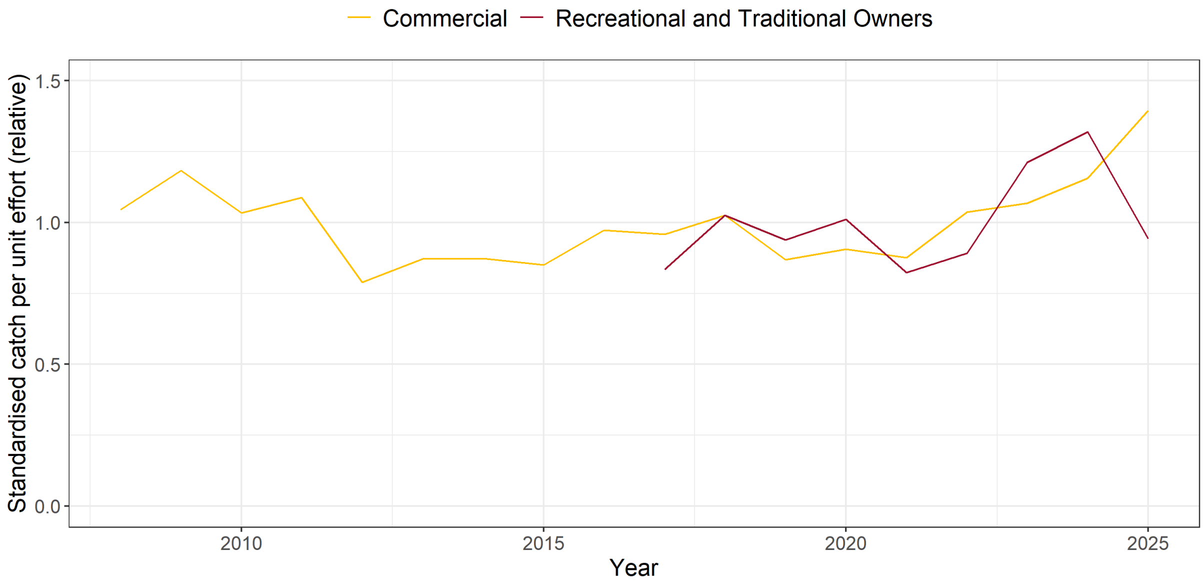

Figure 4: Annual standardised catch rates for Queensland east coast saddletail snapper after adjusting for fishing efficiency

Length and age data

The population model was fit to a combination of marginal length composition and conditional age-at-length composition data. The mean (average) age of fish shows no clear trend overall, however troughs in 2020 and 2024 may indicate a strong year class entering the fishery as a result of a positive preceding recruitment year, assuming consistent distribution of age samples (Figure 5).

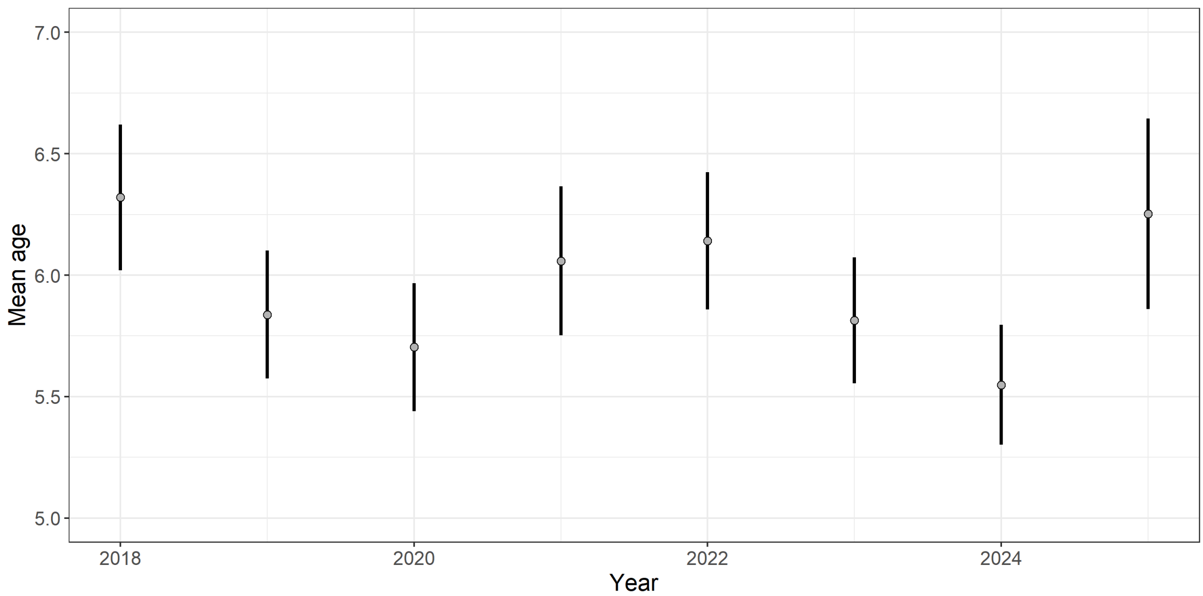

Figure 5: Mean age from conditional age-at-length data over time (aggregated across length bins) for commercial and recreational sectors combined for Queensland east coast saddletail snapper

Age data were input to the model as conditional age-at-length data for each sex, rather than as marginal age data. Figure 6 shows the extent of matching between observed marginal age compositions versus those predicted by the model. This is a useful diagnostic tool, recognising that the model did not explicitly fit to the age composition data. As evident in Figure 6, the model can predict very well the marginal age compositions, providing increased confidence in the model. Figure 6 shows both sexes combined—plots for each sex individually are shows in Figures 3.23 to 3.23.

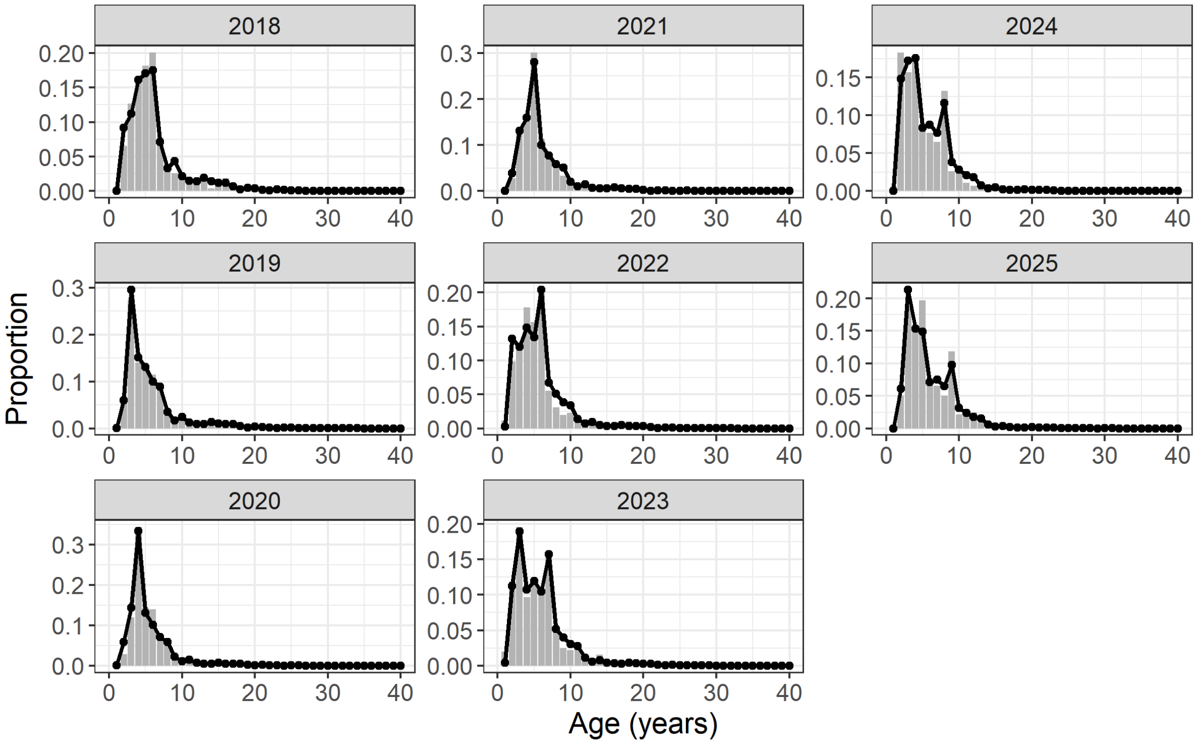

Figure 6: Comparison of modelled age compositions to observed annual age compositions of saddletail snapper for both females and males combined

Key model influences

Key assumptions in the model that affected assessment results are natural mortality rates, steepness, the weighting of length data inputs, and recent years of catch rates.

Natural mortality and steepness: Natural mortality and steepness values had an influence on biomass outcomes. Data were not available to support the estimation of either parameter, so both were fixed in every scenario. The suite of 12 scenarios explored six combinations of natural mortality and steepness, to understand the interaction between the parameters.

Length data weighting: The method used to weight individual fish samples in the length compositions was one of the most influential axes of uncertainty. This impacted the emphasis the model placed on the signal from the length compositions compared to other data inputs (such as catch rates), as well as the relative contribution that the commercial and recreational sectors had on that signal, respectively.

Catch rates: Saddletail snapper typically do not recruit to the fishery until they reach about 3 years of age. This means that the most recent few years of age and length data are not able to effectively inform the model on recruitment or relative abundance levels for the most recent few years, so for abundance the model instead has to rely on signal from catch rates indices alone, and for recruitment, on that predicted from the stock-recruitment relationship.

Analyses were conducted to understand the influence of other factors—such as spatial structure in the stock, and the impact of depredation—however these factors were not as influential as those listed above.

Recommendations to improve future assessments

A comprehensive list of recommendations for future work have been made in Section 4.5. At a glance, these are summarised below:

Stock assessment

- **Steepness and natural mortality:** Future assessments should continue to monitor current best practice for the selection of steepness values and natural mortality rates, as well as the suitability of estimating either parameter.
- **Time-varying selectivity:** Though plausible, and potentially beneficial to annual length composition fits, time-varying selectivity could not be effectively modelled here. We suggest revisiting this topic in future assessments, when additional years of length data become available, testing alternative approaches available in Stock Synthesis.
- **Depredation:** If updated data on depredation rates become available, they should be included in future assessments.

Research and monitoring

- **Updates to depredation data:** Given indications of recent rapid change in depredation rates, continued monitoring and research and periodic updates to these estimates are recommended.
- **Changes to fishing efficiency:** Future assessments would benefit from an up-to-date study on annual changes to the efficiency of catching saddletail snapper, or at least fish species targeted by the Reef Line Fishery, including the effects of recent technological advancements, changes to commercial fishing fleets, and interactions with sharks.
- **Biological data collection:** Continued collection of age, length, and sex data is critical. These data underpin the model's estimation of key biological processes, especially selectivity and natural mortality, which was influential in this assessment. These data are particularly important in cases like saddletail snapper, were selectivity varies between fleets and may vary over time within fleets. As time series of these data accumulate, combined with catch and abundance data, future models will likely become better equipped to estimate important parameters like natural mortality internally, reducing assessment uncertainty.

Conclusion

This stock assessment was commissioned to establish the status of the Queensland east coast saddletail snapper stock. This stock assessment benefited from incorporating recommendations from recent Queensland stock assessments and reviews, expanding the project team, and adding new data sources to address knowledge gaps. These factors all contributed to a more robust model-based assessment, with reduced uncertainty compared to the previous assessment (Campbell et al. 2021).

This assessment show that the most probably estimate of spawning biomass ratio at the beginning of financial year 2025–26 is 28%, with a 95% credible interval of 19–42%. This means that the spawning biomass of saddletail snapper is 72% smaller today than it was in 1961. The range of uncertainty in this assessment is an improvement compared to the previous assessment (13–73%, Campbell et al. 2021), so there is much greater confidence in the results from this current assessment. The current fishing mortality exceeds that which would sustain the population at 60% biomass.

Specific recommendations around any particular management proposals would require separate analysis, such as forward projection modelling using the current assessment model.

Acknowledgements

We acknowledge the Gubbi Gubbi and Kabi Kabi people who are the Traditional Owners of the land where this work was primarily conducted. We recognise their ongoing connection to Country, and their role in caring for and maintaining the land and sea for thousands of years.

This stock assessment was capably guided by a project team with an extensive knowledge base and a diverse range of perspectives on the Queensland east coast stock of saddletail snapper. In addition to the Departmental staff involved (including stock assessment scientists, fishery managers, fisheries monitoring staff and data specialists), the team included an independent scientist, three fishing industry representatives and a representative from the Great Barrier Reef Marine Park Authority. The project team operated under a terms of reference, designed to ensure a transparent and evidence-based approach. The project team members were, in alphabetical order:

- Chris Bolton—Fishery Stakeholder, commercial representative
- Jason Bradford—Fishery Stakeholder, recreational representative
- Alise Fox—Stock Assessment (Fisheries Queensland, DPI)
- Peter Green—Fishery Stakeholder, Traditional Owners representative
- Thomas Hatley—Great Barrier Reef Marine Park Association (since September 2025)
- Alex Hesp—Independent Scientist (Western Australia Department of Primary Industries and Regional Development)
- Kyle Hillcoat—Fisheries Scientist (Agri-Science Queensland, DPI)
- Jennifer Larkin—Fisheries Resource Officer (Data) (Fisheries Queensland, DPI)
- Chad Lunow—Fishery Manager (Fisheries Queensland, DPI)
- Amos Mapleston—Biological monitoring representative (Fisheries Queensland, DPI)
- Lucas Sumpter—Stock Assessment (Fisheries Queensland, DPI)
- James Webley—Recreational fishing data representative (Fisheries Queensland, DPI)
- Matthew Wilkie—Great Barrier Reef Marine Park Association (until September 2025)
- Michelle Winning—Chair/Fisheries Assessment Manager (Fisheries Queensland, DPI)

In addition, Ari Gilham contributed to interpretation and analysis of recreational fishing data, and James English assisted with project team meeting minute support.

The entire team is thanked for contributing their time and knowledge, and engaging constructively over the course of the project.

The authors acknowledge the significant time commitment of the fishery stakeholder representatives: Chris Bolton, Jason Bradford and Peter Green. Their involvement in this project has meant that all aspects of this assessment could be considered not only in a scientific context, but also in the context of their extensive on-water experience. Outside of project team meetings, they consulted with their own industry networks to ensure the perspective they brought the project was balanced. Chris, Jason and Peter's contributions have set an excellent standard for continued collaboration with industry members to further enhance stock assessments.

We also thank independent scientist Dr Alex Hesp for his ongoing collaboration throughout the project, and we acknowledge his substantial time commitment to the project. Alex's thorough and constructive feedback has greatly added to the robustness of the analyses and the comprehensiveness of the final report. In additional, we thank two more stock assessment scientists from Western Australia—Ainslie Denham and Rubie Evans-Powell—for contributing their expertise with catch rate standardisation and Stock Synthesis model configuration.

We thank both Dr Jaeden Vardon and Dr Stacy Bierwagen for generously taking the time to discuss their research on saddletail snapper and to answer our questions to ensure we were applying and interpreting their research correctly.

Our sincere thanks to the many DPI Fishery Monitoring staff that have participated in the collection and quality assurance of the length, age and sex data for Queensland east coast saddletail snapper, which were integral to this assessment. We also extend our gratitude to all fishers who have voluntarily assisted the collection of biological data. These data always play a crucial role in Queensland stock assessments. We thank Amos Mapleston for sharing his intimate knowledge of the data and advising on their usage.

We also wish to acknowledge and thank the many fishers, seafood processors and scientists who have contributed to past and current research on the Queensland east coast stock of saddletail snapper.

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Glossary

ACN	Authority chain number
age	Age within this report refers to age group unless otherwise stated
B	Biomass, total weight of a population or of a component of a population. This assessment refers to spawning biomass, measured by spawning egg production
B_{limit}, B_{20}	Biomass limit reference point, the point below which the risk to the population is regarded as unacceptable under the Sustainable Fisheries Strategy
B_{MSY}	Biomass at maximum sustainable yield
B_0	Mean equilibrium unfished biomass, average biomass level if fishing had not occurred. Unfished state corresponds to the first year assessed in 1961.
BRS	Boat ramp survey
Catch rate	Also referred to as 'CPUE'. Index of fish abundance, referred to as average (mean) catch rates standardised (adjusted) to a constant vessel and fishing power through time. All references to catch rates were standardised unless specified to be different.
Catchability, q	The ability to catch fish. More formally, it is defined as the probability of catching a fish with a single unit of standardised fishing effort. Catchability is the interaction of the fishing gear and a fish's behaviour, whereas fishing power is a property of the fishing effort, gear and practices.
CFISH	Commercial fisheries information system, which is the compulsory commercial logbook database managed by Fisheries Queensland
CPUE	See 'catch rate'
DPI	Queensland Department of Primary Industries
ESS	Effective Sample Size, a statistical term that relates to the sample size needed to achieve a specific level of precision, relevant to population model tuning for age and length composition data
EC	East coast
Fishery	This stock assessment evaluated Queensland east coast saddletail snapper in the Reef Line Fishery. The assessment was conducted on the whole (genetic) stock. The fishery covers all fishing sectors: commercial, charter, recreational and Traditional Owners.
Fishing year	1 July to 30 June. Also labelled as 'year' within. Fishing years were equal to financial years to group the seasonal and biological patterns of saddletail snapper. Labelling used the second year in the financial year string. For example the financial year July 2019 to June 2020 was labelled as 2020 fishing year.
FL	Fork length, measured from the tip of fish's nose to the fork in its tail (caudal fork)
fleet	A Stock Synthesis modelling term used to distinguish types of fishing activity. Typically a fishing fleet will have a unique selectivity curve that characterises the likelihood that fish of various sizes (or ages) will be caught by the fishing gear, or observed by the survey.
GAM	Generalised additive model
h	Also called 'steepness', a model parameter which indicates how recruitment responds to changes in biomass, influencing the stock's ability to recover from depletion.
ITQ	Individual transferable quota
JL	Jaw length
M	Natural mortality
MCMC	Markov chain Monte Carlo—a statistical simulation method for approximating the final ('posterior') distribution of a quantity
MLE	Maximum Likelihood Estimation—a statistical method of estimating the parameters of an assumed probability distribution

MLS	Minimum legal size
MSY	Maximum sustainable yield, the long-term average maximum level at which the species can be routinely exploited without long-term depletion
NRIFS	National Recreational and Indigenous Fishing Survey, funded by the Fisheries Research and Development Corporation (2000–01)
Overfished	A fish population with a biomass below the biomass limit reference point (B_{limit})
Overfishing	The condition where a population is experiencing too much fishing and the removal rate is unsustainable, that is, fishing mortality is higher than fishing mortality at maximum sustainable yield.
Qld	Queensland
R_0	Unfished recruitment
Reference point	An indicator of the level of a key performance indicator (e.g. spawning biomass or fishing mortality), used as a benchmark for interpreting the results of an assessment
RFish	Recreational fishing surveys conducted by Fisheries Queensland (1997, 1999, 2002, 2005)
SRFS	Statewide Recreational Fishing Surveys conducted by Fisheries Queensland (2010–11, 2013–14, 2019–20)
SS	Stock Synthesis software for fishery stock assessment
steepness	Also called h, a model parameter which indicates how recruitment responds to changes in biomass, influencing the stock's ability to recover from depletion.
t	Tonnes
TL	Total length, measured from the tip of fish's nose to the end of its tail lying freely in its normal position
Selectivity	The probability that a fish is exposed to fishing mortality. This probability varies by size and age and is determined by the spatio-temporal overlap between fish and fishing activity, encompassing both fish availability to the fishery and their susceptibility to capture by the fishing gear.
Wonky hole	a depression in the seabed typically occurring in shallower waters, where artesian freshwater is expelled seasonally, where this species is sometimes found

1 Introduction

Saddletail snapper (*Lutjanus malabaricus*) are a key species targeted in the Reef Line Fishery (formerly the Coral Reef Fin Fish Fishery). Alternative common names for the species include large mouth nannygai, scarlet sea-perch, Malabar blood snapper, red jew or saddletail. Saddletail snapper are one of three large species of lutjanids that have historically dominated the Other Species quota category within the Reef Line Fishery, alongside red emperor and crimson snapper (*L. sebae* and *L. erythropterus* respectively). Due to their similar appearance, saddletail snapper are sometimes misidentified as crimson snapper (Allen 1985). The Reef Line Fishery (RLF) operates largely within the Great Barrier Reef Marine Park (GBRMP), extending from the northern tip of Cape York to 24°30' S (south of Brisbane).

Distribution and movement

Saddletail snapper have a widespread distribution, from the Indo-West Pacific from Fiji to the Arabian Sea and Persian Gulf, and from Australia to southern Japan (Allen 1985). Around Australia, the species can be found from Shark Bay in Western Australia, across northern Australia to the east coast of Queensland (Newman 2002). The species is comprised of three biological stocks, located in the North Coast Bioregion (Western Australia), northern Australia (including the Timor Sea, Arafura Sea and the Gulf of Carpentaria) and the east coast of Queensland (Elliott 1996; Salini et al. 2006). The latter is the focus of this stock assessment. Elliott (1996) suggests little, if any, movement of genes through the Torres Strait since its opening about 8000 years ago. Salini et al. (2006) suggest a genetic boundary exists between Kupang and Sape (districts in Indonesia).

Tag and recapture data collected from the central Queensland coast indicate that saddletail snapper have a high recapture rate of 13.5% (Platten and Sawynok 2007). Of those recaptured individuals, the maximum distance moved was 5 km, while 97% indicated no movement from their initial location (Platten and Sawynok 2007). Given the limited number of recaptures in this study, most of which were below minimum legal size (MLS), this serves as only a preliminary indication of movement behaviour.

Habitat

Saddletail snapper are found on coastal and offshore reefs, shoal grounds, and areas of flat bottom with occasional epibenthos or vertical relief (Newman 2002). They have been observed at depths between 12 and 140 m and frequently form mixed shoals with crimson snapper (*Lutjanus erythropterus*) (Allen 1985; Newman 2002). They feed mainly on finfish and benthic crustaceans (Carpenter 2001). Saddletail snapper tend to exhibit an ontogenetic habitat shift, in which they spend their juvenile phase in nearshore habitat, then migrate to offshore inter-reef areas in their adulthood (Davis et al. 2025).

Saddletail snapper frequently suffer from barotrauma when caught, which is especially prevalent when they are caught in deeper waters. Brown et al. (2008) reported a post-capture released survival rate for saddletail snapper just above 50%, however this decreased to 10% survival if fish were in the lowest category for release condition in this study. It is important to note that this study was conducted on an inshore wreck near Townsville, Queensland at a depth of approximately 22 m with a high proportion of individuals under the MLS. Juvenile saddletail snapper (below the MLS) are more likely to be found in this habitat as they inhabit inshore shallow waters (< 25 m). Both saddletail snapper and crimson snapper (*Lutjanus erythropterus*) form part of a 'coastal' assemblage of species found only in the shallowest stations sampled (15–24 m) in the central Great Barrier Reef (Jones and Derbyshire 1988). Conversely, larger individuals (above the MLS) generally inhabit deeper waters with the exception of 'wonky holes'—depressions in the seabed typically occurring in shallower waters, where artesian freshwater is expelled seasonally (Nowak 2002; Stieglitz and Ridd 2000).

Biology

McPherson et al. (1992) reported that spawning activity occurs during the spring and summer months in the Great Barrier Reef waters, with a 5 month spawning period that peaks during November–January. Histology data suggest October as the middle of the spawning on the Great Barrier Reef (Hillcoat 2022). McPherson et al. (1992) reported there is no apparent relationship between spawning activity and lunar cycle.

Saddletail snapper are gonochoristic (seperate sexes) (O'Neill et al. 2011). They are relatively long-lived, and grow slowly after becoming reproductively mature (Newman et al. 2000; Fry and Milton 2009). Biological age data collected and used in this assessment show that, in the Queensland east coast stock, saddletail snapper can live for at least 49 years, determined using counts of validated annual increments in otoliths (Department of Primary Industries, unpublished data).

In Great Barrier Reef waters, the fork length of females at 50% maturity was estimated to be 57.3 cm (Hillcoat 2022). An earlier study estimated the same value to be 57.6 cm (McPherson et al. 1992). Saddletail snapper can reach an estimated maximum fork length of 101 cm and weight of 12 kg in Great Barrier Reef waters (McPherson et al. 1992). This is substantially larger than those found in northern and western Australia (McPherson et al. 1992).

Fishery

This report considers four sectors in the Reef Line Fishery: commercial, recreational, Traditional Owners and charter fishing operators.

The commercial sector of the fishery mainly targets coral trout for a live export market. Commercial fishers also catch redthroat emperor and 'other species' of coral reef fish, including saddletail snapper (see Appendix F.2 for a full list of other species). When commercial fishers target live coral trout, saddletail snapper are rarely bycatch of that activity. Commercial fishers who target and catch saddletail snapper typically operate in different areas and depth ranges to those targeting live coral trout. A number of key target species for commercial fishers in the Reef Line Fishery are also targeted by recreational anglers and charter operators (Fisheries Queensland 2020).

The popularity of saddletail as a recreational fishing target species has markedly expanded in recent years and is reflected in recent annual recreational catch totals. This is partially attributed to improvements in technology and resources allowing more fishers to find suitable saddletail ground, such as bigger more powerful boats, side-scan depth sounders, knowledge sharing and training, and high-resolution bathymetric mapping.

Saddletail snapper are also caught by Aboriginal peoples and Torres Strait Islander peoples. Limited data exist for catch and effort in the Traditional Owners sector, however it is expected that this sector has comparatively low levels of effort, with fishing activities aligning closely with the recreational fishing sector (Fisheries Queensland 2020).

The Reef Line Fishery earns Queensland's second highest gross value of production (GVP) behind the East Coast Otter Trawl Fishery, with an estimated GVP of $31 million (Fisheries Queensland 2020). The saddletail snapper component is estimated to be $0.6–0.9 million (BDO EconSearch 2020).

The fishing season is 1 July to 30 June annually, with two five-day spawning closures between October and November each year for coral reef fin fish species. These closures apply to all line fishers targeting coral reef fin fish (commercial, charter, recreational) between latitude 10° 41'S and 24° 50'S to the eastern boundary of the Great Barrier Reef Marine Park. In the commercial sector, vessel length is restricted to a maximum of 25 m, tenders are limited by number, size and proximity to the primary vessel, and gear is restricted to three fishing lines at a time with no more than six hooks total. Recreational fishers accessing the fishery can use hook and line, rods and reels, and spearfishing gear (excluding hookah/scuba). Species-specific individual transferable quotas are in place for coral trout and redthroat emperor, however saddletail snapper and other targeted species are managed using a combined/basket individual transferable quota.

The Reef Line Fishery is managed under the *Fisheries Act 1994* and its subordinate legislation. The Traditional Owners sector of the fishery is managed in consideration of the *Native Title Act 1993*, which allows fishing by Traditional Owners using prescribed traditional and non-commercial gear, and removes restrictions on size, possession limits and seasonal closures (Fisheries Queensland 2020).

Key management measures in the fishery that pertain to saddletail snapper include minimum size limits, compulsory log books, total allowable commercial catch limits, individual transferable quotas, gear restrictions, vessel and tender restrictions, spawning closures, and possession limits for recreational fishers (Fisheries Queensland 2020). The history of saddletail snapper management is summarised in Table 1. A more detailed history can be found in Table F.1 in Appendix F.1.

Table 1.1: Abridged history of saddletail snapper management in Queensland—more details can be found in Table F.1

Year	Management
1957	Minimum size of 14 inches (35.56 cm)
1975	Inclusion of no-fishing zones in the Great Barrier Reef
1982	Section 35 permit: recreational fishers allowed to sell surplus fish
1993	Recreational possession limits of a combined total of 30 coral reef fish covering 26 species. Minimum size limit of 40 cm. Charter vessel possession limit arrangements. Restructure of commercial line fishery into regional endorsements.
1994	Section 35 permits to sell recreationally caught fish repealed

Continued on next page

Table 1.1 – *Continued from previous page*

Year	Management
2003	Fisheries (Coral Reef Fin Fish) Management Plan implemented. Recreational possession limits of a combined total of 9 crimson and saddletail snapper. Commercial fishery symbols revised: 'L1', 'L2', 'L3' and 'L8' govern some gear and fishing area definitions, 'RQ' quota provides access to prescribed coral reef fish. Commercial licence holders require 'Other Species' (OS) units to take saddletail snapper, via individual transferable quotas. The total yearly catch of Other Species available for allocation is 902.2 t. New reporting requirements. Annual seasonal closures across the Great Barrier Reef. Restrictions implemented on gear, primary fishing boats, and fishing tender boats.
2004	New zoning arrangements for the Great Barrier Reef Marine Park, allocating approximately 33% of the marine park as no-take zones.
2019	Fisheries (Coral Reef Fin Fish) Management Plan repealed. New legislation enacted.
2019	The total yearly catch of Other Species available for allocation is 955.597 t

Stock assessment history

This stock was first formally assessed in 2021, using financial year data from 1961 to 2020, and reported on 16 modelling scenarios (Campbell et al. 2021). The assessment estimated the biomass at the beginning of 2021 by reporting the biomass result of a single 'base case' (or most plausible) scenario selected by the previous project team, using the range of estimates from the suite of modelling scenarios to define the uncertainty around the reported biomass estimate. The previous assessment estimated the biomass of east coast saddletail snapper at the beginning of 2021 to be 23%, with an uncertainty range of 13–73%.

2 Methods

2.1 Data sources

Data sources included in this assessment (Table 2.1) were used to produce key population model inputs such as catch rates, length compositions, conditional age-at-length compositions, discards and annual catches. The assessment period includes years from 1961 up until and including 2025 based on available information.

Table 2.1: Data inputs for the assessment

Years	Description
Commercial	
1989–2025	Compulsory commercial logbook database (CFISH) collected by Fisheries Queensland
Recreational	
2016–2025	Boat ramp survey, conducted by Fisheries Queensland, providing catch, discard, catch rate, depredation and length composition information for the recreational fishing sector
2002, 2005	Recreational fishing surveys (RFISH) conducted by Fisheries Queensland (Higgs et al. 2007; McInnes 2008)
2011, 2014, 2020	Statewide Recreational Fishing Survey (SRFS) conducted by Fisheries Queensland (Taylor et al. 2012; Webley et al. 2015; Teixeira et al. 2021)
2001	Recreational fishing survey (the National Recreational and Indigenous Fishing Survey, NRIFS) conducted by the Australian Department of Agriculture, Fisheries and Forestry (Henry and Lyle 2003)
Charter	
1997–2025	Logbook data collected by Fisheries Queensland
Wind	
1989–2025	Weather data collected by Bureau of Meteorology (BOM)
Lunar	
1989–2025	Continuous daily luminous scale of 0 (new moon) to 1 (full moon) (R package 'lunar')
Biological	
2017–2025	Collaborative collection of regional demographic data (age, length and sex) sourced from both commercial and recreational fisheries (biological monitoring as above undertaken by Fisheries Queensland)
2018–2025	Biological monitoring (sex, age and length from the commercial line fishery) undertaken by Fisheries Queensland (Fisheries Queensland)
1986–2008	Tag and release data, providing information on length composition of discarded fish (Brown et al. 2008)

2.1.1 Commercial

Estimated weights of commercial catches of saddletail snapper were recorded in the Queensland logbook system. The logbook system consists of daily retained catches (landed weight in kilograms) of all fish species from each individual fishing operator (license) since 1988. In addition to landed weight, logbooks also record the location of the catch (aggregated to 30 minute or 6 minute grid identifiers), the number of boats (dories) that were fishing, and the number of crew.

2.1.2 Recreational

2.1.2.1 Statewide surveys

The statewide recreational fishing surveys (RFISH and SRFS) provided estimates of the number of fish caught, retained and discarded. Using additional information on the lengths of fish caught, estimates of the annual weight of saddletail snapper caught be the recreational sector at state and regional scales could be made. See the references listed in Table 2.1 for more detail.

The statewide recreational fishing surveys used telephone surveys of random households to estimate recreational fishing participation, catch and effort. Logbook records were also maintained by a sample of fishing households for 12 months. Fishing householddata were demographically weighted and expanded to estimate total catches of fish (in numbers) and fishing effort by factors such as key species, seasons and regions.

2.1.2.2 Boat Ramp Survey

The Fisheries Queensland boat ramp survey program has collected recreational fishing data in 18 different regions, extending from Aurukun to the Gold Coast. Fifteen of these regions were on the Queensland east coast, with Cooktown being the northernmost region. Staff trained in fish identification and the survey protocol, and identifying fish, interview recreational fishers at boat ramps during survey shifts. The program has collected data from between 12,000 and 18,000 fishing boat trips annually across Queensland. The surveys recorded day and location fished, and for key species, catch (in numbers), discards (in numbers), depredation, and measured fork lengths (cm) of retained fish (Northrop et al. 2018; Fisheries Queensland 2017). As boat ramp surveys interview only a subset of recreational fishers, these data cannot be used to estimate total statewide catches. However, the data reveal spatial and temporal trends in recreational fishing effort, and patterns of discards and depredation.

The catch and effort data from boat ramp surveys were used to refine total catch estimates for the years 2018–2019 and 2020–2025, where total catch estimates from SRFS were not available (see Section 2.1.2 for more detail). Boat ramp surveys were also used to inform recreational catch rates, recreational discarding behaviour and depredation (see Section 2.3).

2.1.3 Charter sector

Catches of saddletail taken by Queensland charter vessels were recorded through the logbook system from 1997 to 2025. This provided the operator identifier, the date, the location fished, retained catch by species (recorded in kg) and the number of guests on the trip.

2.1.4 Traditional Owners

The National Recreational and Indigenous Fishing Survey in 2000 attempted to redress the lack of Indigenous fishing information on a national scale by involving Indigenous communities in the gathering of fisheries statistics. A point estimate for total catch and discard of saddletail snapper for Indigenous communities was provided in this study (Henry and Lyle 2003).

2.1.5 Age and length compositions

Age, length and sex data of saddletail snapper have been collected routinely by Fisheries Queensland since 2018 to serve as the primary data source for age and length compositions. Details of the age and length sampling are provided in Appendix D.2, with further information available in Fisheries Queensland (2021) and Sumpton and O'Neill (2004).

In addition, length data of saddletail snapper have been collected by Fisheries Queensland through the Boat Ramp Survey which were used to inform length compositions caught by the recreational sector (Fisheries Queensland 2017).

The Keen Angler Program collects fish skeletons from recreational fishers to help monitor Queensland's key fish stocks, from which length, age and sex information can be obtained (White et al. 2025). This data set served as an additional source of length-at-age data that were input to the model.

Data collected by Brown et al. 2008 provided information on the size distribution of undersized fish, which were used to inform the length composition of discarded fish.

2.2 Retained catch estimates

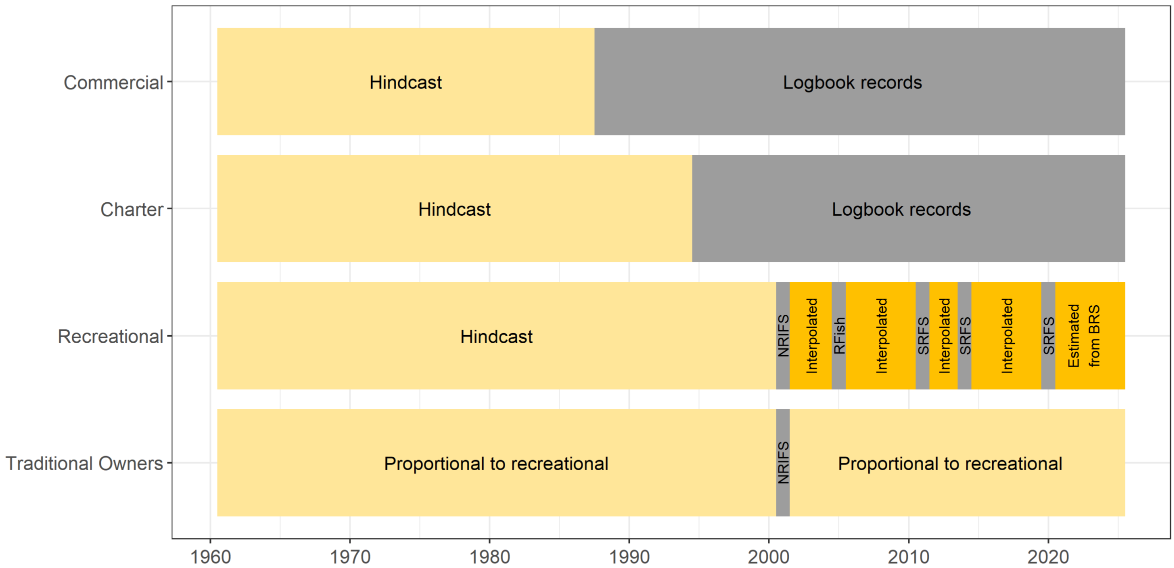

Figure 2.1: Overview of the methods used to reconstruct the history of Queensland east coast saddletail snapper retained catch

Catch data from all sectors were combined to reconstruct the history of retained catch from 1961 until 2025. Prior to 1961, saddletail snapper retained catch is assumed to be negligible based on low catches in the early years of the logbook system and project team feedback.

2.2.1 Commercial retained catch

Between 1988 and 2025, a baseline retained catch of saddletail snapper was set to the weight of records specified as ‘nannygai, large mouth’ in the CFISH logbooks within the east coast region, under all fishery codes except charter (CV) and reef line multi-hook (MH).

2.2.1.1 ‘Nannygai, Unspecified’ catch

From 1988 to 2009, CFISH logbooks also contained an ‘nannygai, unspecified’ label, which represented an ambiguous mixture of large mouth nannygai (saddletail snapper) and small mouth nannygai (crimson snapper).

The ratio of saddletail snapper retained catch to crimson snapper retained catch was calculated for each year, for two regions: north and south of 21° S, to account for regional variation while also maintaining a large sample size. The proportion of saddletail retained catch was applied to the unspecified nannygai retained catch and added to the baseline data.

More explicitly, an annual proportion ($p_{y,r}$) was calculated for each year (y) and region (r, north or south of 21° S), equal to the tonnage of saddletail snapper retained divided by the combined tonnage of reatined saddletail snapper and crimson snapper (the known nannygai). Then for each year and region, the tonnage of unspecified nannygai retained catch was multiplied to by the associated $p_{y,r}$ and added to the tonnage of saddletail snapper retained catch.

2.2.1.2 Adjustment for 1997–2004

Members of the project team noted that it was plausible that the investment warning ahead of individual transferable quota (ITQ) allocation in 2004 could habe led to over-reporting during the 1997–2004 period. If over-reporting had occurred, this would lead to artificially inflated annual catch totals during this period, followed by a sudden decline after 2004 as quota was implemented. This issue was appraised by the project team, which led to a clearer indication that over-reporting in this period was likely to have occurred. Project team discussions gave broad agreement that some degree of adjustment would be required for catches in this period, by comparing the 1997–2004 period to a reference period post-2004.

The project team also acknowledged that some of the reduction in recorded catch following 2004 was likely associated with a real effect, as the number of active fishing operations for saddletail dropped from 140 in 1997–2004 compared to 112 in 2007–2014. Since the number of active fishing operations dropped by 20%, a similar drop in reported catch may be expected. However, raw annual Queensland commercial catches dropped by 69%, across the same periods. This comparison corroborates that some of over-reporting of commercial catches likely occurred during the 1997–2004 period.

A scaling factor was calculated to apply to the catch in 1997–2004, such that the subsequent drop in total catch post-2004 was proportionally equivalent to the drop in the number of active licences.

The average commercial catch rate in 1997–2004 was 1056 kg/operator compared with in 2024 when it was 873 kg/operator. This is a decrease of 17%. By applying a scaling factor of 83% to the catch in 1997–2004 (i.e. multiplying the catch during that period by 0.83, or $1-0.17$), the catch rates in 1997–2004 and 2024 become the same (873 kg/operator).

A resulting scaling factor of 83% was used as a base assumption, with two alternative assumptions explored in Appendix B.2: 38% (using 2007–2014 as a reference period) and 100% (no adjustment).

2.2.1.3 Hindcast

Commercial catch was fixed to 0 kg in 1961, and linearly interpolated between 1961 and the beginning of CFISH logbook data in 1988.

2.2.2 Charter retained catch

Charter catch was calculated using a similar approach to commercial catch. Between 1995 and 2025, a baseline retained catch of saddletail snapper was set to the weight of records specified as 'saddletail snapper' in the CFISH logbooks within the east coast region, under the fishery code charter (CV). The method for unspecified nannygai estimates and hindcasting were the same as for commercial retained catch. The adjustment made to commercial retained catch data between 1997 and 2004 was not required for charter fishing operations.

2.2.3 Recreational retained catch

Recreational catch estimates for all years were input to the model as numbers of fish. These estimates were then converted to weight of retained catch by the population model internally, informed by selectivity patterns and length-weight relationships (Methot 2000).

2.2.3.1 Data and interpolation

Recreational catch estimates for 2000, 2011, 2014 and 2019 were set to equal the values reported in the NRIFS (2000) and SRFS (2011, 2014 and 2019) surveys respectively.

The recreational catch estimate for 2005 was set equal to a re-scaled RFish estimate. Estimates from the RFish surveys had higher participant drop out than SRFS surveys in later years. This may bias the mean catch rates and fishing effort upwards and result in an overestimate of recreational fish catches. To account for this bias, a simple ratio method from Leigh et al. (2017) was used. This method was applied to reduce RFish catch estimates to better align with the 2000, 2011, 2014 and 2019 surveys. The assumption in applying this scaling was that the RFish estimates were overstated by the same fraction in all survey years in which the RFish methodology was employed. This re-scaled estimate was calculated as the 2005 RFish estimate multiplied by a scaling factor (the 2002 estimate divided by the NRIFS estimate for the year 2000).

Estimates for recreational retained catch in non-survey years between 2001 and 2017 were linearly interpolated between data points.

2.2.3.2 Forecast

In the absence of recent Statewide Recreational Fishing Surveys from 2019 onwards, trends in catch and effort data from Boat Ramp Survey were used to estimate retained catch on a state wide scale, using the following method:

1. Calculate the average saddletail retained catch per interview, for each ramp in each year.
2. Calculate the average daily trailer count for each ramp in each year (keeping only the ramps present in 2020 boat ramp survey).
3. Multiply the above two trends together to calculate retained catch rate index (number of saddletail retained per ramp) for each ramp in each year.
4. Transform the retained catch rate into an index relative to 2020 (the year of the SRFS survey) by dividing the annual retained catch rate index by the 2020 retained catch rate index.

5. Summarise into the average retained catch index per year.
6. Multiply this index by the 2020 SRFS retained catch estimate to rescale to a statewide magnitude.

2.2.3.3 Hindcast

Recreational retained catch was assumed to be zero in 1961 and increased proportionally to Queensland population growth to reach the first data point in 2000.

2.2.4 Traditional Owners retained catch

The relative magnitude of retained catch by Traditional Owners was derived by comparing the NRIFS estimate to the SRFS estimate in 2000. This returned a scaling factor of 0.05, which was then applied to annual recreational catch totals to obtain annual Traditional Owner catches throughout the time series.

Recreational retained catch and Traditional Owners retained catch were combined into a single fleet for input to the population model.

2.2.5 Depredation

Existing literature on depredation in fisheries provides valuable insights, although specific information on saddletail and the Reef Line Fishery remains limited. Vardon (2025) explored depredation dynamics in a controlled experimental setting on charter vessels targeting reef fish, reporting that sharks successfully depredated just over 50% of the fish they followed to the surface. Vardon (2025) highlighted the significant impact of depredation on fishing success and the behavioural adaptations of fishers, such as only relocating after prolonged depredation events. Similarly, Mitchell (in prep) provided estimates of depredation rates for Spanish mackerel in Queensland's east coast waters, revealing substantial differences in depredation rates between commercial and recreational fishing sectors. However, these studies primarily focus on other species and fisheries, with limited direct application to saddletail snapper or the Reef Line Fishery. This underscores the need for further targeted research to better understand the extent and drivers of depredation of saddletail snapper, as well as its implications for stock assessments and sustainable management.

2.2.5.1 Boat Ramp Survey data

The estimation of depredation rates for saddletail snapper was done using data collected through Fisheries Queensland's BRS program. This method provides a quantitative approach to assess the impact of depredation on recreational fishing mortality.

The BRS program collects data from recreational fishers returning to surveyed boat ramps between 9:00 am and 4:00 pm. For the purposes of this analysis, only data from day trips (including overnight trips) were used, as multi-day trips are prone to greater recall bias and incomplete reporting. The first depredation data for saddletail were recorded on 6 July 2023.

The BRS program has been designed such that each BRS interview is likely to represent a randomly selected fishing trip and is representative of the diversity of boat-based recreational fishing activity in Queensland. There are coverage gaps in the sampling of the data (e.g. trips returning outside survey hours or those returning to non-surveyed ramps). However, these gaps are likely to have minimal impact on estimates due to the small proportion of missed trips outside survey hours, and the likely similarity in fishing characteristics between surveyed and non-surveyed ramps because the fishing activity from both sets of ramps occurs in similar water bodies (e.g. in Cairns, fishers leaving from surveyed and non-surveyed ramps both fish in the same offshore water bodies).

Usually, where fish are depredated and completely lost, fishers are unable to determine what species of fish was depredated. To estimate the number of depredated fish in the dataset that were saddletail snapper, for each interview we calculated the proportion of the total recorded catch (kept + released) that was saddletail snapper, then multiplied the number of fish recorded as depredated by that proportion. This provided an estimate of the number of saddletail snapper depredated for each fishing trip that caught saddletail snapper. These estimates were then summed to calculate the number of kept saddletail snapper (retained catch) and the sum of the saddletail snapper depredated for all those interviews.

Because the program was designed so that each interview is a representative, random sample of recreational fishing activity, this method should provide a representative estimate of the number of saddletail snapper depredated as a proportion of the saddletail snapper retained catch. This proportion was then applied to the total saddletail snapper recreational retained catch estimate for the respective years in the stock assessment to estimate the additional recreational fishing mortality due to depredation.

For the subset of trips where depredation occurred, the proportion of saddletail snapper depredated ranged from 1.0 (100%) to 0.03 (3%). For the boat ramp survey data available, the number of saddletail snapper retained was 1138. The estimated number depredated was 666, giving a depredation rate of 0.585. This was used to estimate the additional mortality due to depredation.

2.2.5.2 Application to catch estimates

While the above gives a sound static estimate of depredation rates in the recreational sector, empirical data are still lacking on the yearly rate of increase in depredation, as well as the year or period where rates began increasing. However looking forward, as the boat ramp survey depredation time series grows, future changes in depredation will become apparent.

There is widespread belief that depredation has increased in recent years. A theoretical basis for this was suggested by Tanimoto et al. (2021) however, where it was assumed that depredation increased from 2009 onwards, when fishery management introduced Queensland commercial quota for east coast shark and the requirement to hold a commercial shark fishing ‘S’ symbol. Queensland commercial east coast shark catch decreased following the management changes in 2009 (Queensland annual total east coast shark quota was 600 t per financial year; mean annual shark retained catch pre-quota, 2000–2009, was 1190 t; mean annual shark retained catch for 2010–2020 was 338 t). There was a belief among some fishers that these changes have directly resulted in higher numbers of sharks and higher depredation rates Tanimoto et al. (2021).

In addition, to support the notion of increased depredation since 2009, annual nominal levels of otter trawling have roughly halved since 2009, with a decline in bycatch discarding on which sharks may scavenge and feed (Wang et al. 2020; Hill and Wassenberg 2000). As mentioned byTanimoto et al. (2021), with less discarded trawl bycatch, one could speculate that sharks may alter their scavenging patterns as needed to consume more fish from offshore line fishing catches.

Appendix F of Tanimoto et al. (2021) explored a stock assessment model that incorporated these assumptions of depredation.

Sumpter et al. (2025) conducted a phone survey of fishers to gather feedback on the depredation in the east coast Queensland Spanish mackerel fishery. This survey provided broad support for the use of 2009 as a sound approximation of when depredation rates began to increase.

Project team discussions resulted in the following assumptions:

- The base rate of depredation in 2025 was 58% for recreational, charter and Traditional Owner sectors combined. Given the sectoral disparity in depredation rates found in Sumpter et al. (2025), it was approximated that the commercial sector experienced half this rate of depredation, resulting in an assumed rate of 29%.
- Depredation rates were assumed to be a constant 1% between 1961 and 2009
- Between 2009 and 2025, the depredation rates increased linearly.

Due to the uncertainty associated with calculating a base depredation rate for 2025, other options were explored in Appendix B.1, to investigate the model's sensitivity to alternate depredation scenarios.

This resulted in a time series for depredation rates shown in Figure 2.2. Retained catch estimates for each fleet were multiplied by $1 + d$ where d is the depredation rate shown in Figure 2.2.

Figure 2.2: Depredation rates used in catch reconstruction for Queensland east coast saddletail snapper

2.3 Discards

For many species, greater than half of the fish caught by recreational anglers are released (McLeay et al. 2002). Generally these released fish are under the MLS which, for saddletail snapper, is 40 cm total length for both the recreational and commercial fisheries. Following Jones and Derbyshire (1988) it was previously hypothesised that a large proportion of discarded fish are undersized and from inshore waters, the typical focus of smaller recreational boats. Larger recreational and commercial vessels typically fish further offshore in deeper waters where the chance of encountering individuals above the MLS is higher. Lower rates of survival for larger released fish in deeper water seems logical, however, these fish are not generally released when above the MLS suggesting limited discard mortality. Boat ramp survey data confirmed that a significant fraction of the recreational saddletail catch was released, and it was therefore important to model discarding explicitly for the recreational-Traditional Owners fleet. Commercial discarding, however, is uncommon due to the absence of a bag limit and the offshore focus of commercial fishers (assumed by the project team) and so for the commercial fleet discarding was assumed negligible.

Length compositions of discarded fish were informed by tagging data conducted by Brown et al. (2008).

Discard data were input to Stock Synthesis as an absolute index of discarded fish, as opposed to a proportion. Data from BRS and SRFS were combined to create discard data inputs as follows:

1. An annual index of BRS saddletail discard rates was calculated by summing the total number of released saddletail and dividing by the number of interviews.
2. The discard index was normalised relative to 2020 (i.e. annual index/2020 index).
3. This index was multiplied by the 2020 SRFS discard estimate to rescale to a statewide magnitude.

2.3.1 Post-release mortality

Discard mortality was set at 0.75 (i.e. 75% mortality, 25% survival), based on Brown et al. (2008) as per the previous assessment (Campbell et al. 2021). The rationale for using this value is as follows.

An average of the best (50%) and worst (10%) case of survival overall (30%) was taken for discards below the MLS based upon Brown et al. (2008) with few discards above the MLS for all sectors. This resulted in an average discard mortality rate of 70% ($100\% - 30\%$). However Brown et al. (2008) studied predominantly under-sized individuals from shallow waters, and therefore the average mortality of 70% was considered an underestimate. Hence a discard mortality rate of 75% was used.

2.4 Standardised indices of abundance

Commercial and recreational catch rate indices were used as indices of legal-sized fish abundance. Both indices were subject to data filtering and a statistical standardisation procedure to remove extraneous influences from factors unrelated to abundance. This section outlines the standardisation procedure.

2.4.1 Commercial catch rates

CFISH logbook data were used for the commercial catch rate analysis. Daily commercial catch records were filtered prior to analysis to include data where the following criteria were true, in order to obtain a data set that is typical of the fishery:

- line fishing only, from the “LF” logbook code
- daily logbook records (no ‘bulk’ reports, where daily quantities cannot be derived)
- latitude and longitude information exists
- latitude is less than -12 and greater than -27
- longitude is greater than 142
- authority chain number has at least two years of saddletail catch
- authority chain number is in the top 95% of operators catching saddletail
- authority chain number holds an L2 and/or L3 licence
- authority chain number holds an RQ symbol
- catch comes from the top 95% of grids from which saddletail are caught
- year is between 2008 and 2025 (for a longer time series see Appendix B.8.1)
- saddletail snapper catch is greater than 0 kg

Year range was restricted to 2008–2025 based on project team concerns regarding the accuracy and comparability of catch records prior to this point. Over a longer time series, the commercial catch rate index showed a significant drop around 2004 (shown in Figure B.22). This drop aligned with a series of changes to the management of the fishery and Great Barrier Reef Marine Park, including the introduction of individual transferable quota allocation and new zoning arrangements that limited fishing grounds. With the data available, we were unable to quantify how much of this drop in catch rates reflects a change in fish abundance, as opposed to extraneous influences (e.g. changes in fisher behaviour, availability of fishing ground, misleading signal due to misreporting). As such, the project team decided to limit the time span of commercial catch rates to 2008 to 2025 (inclusive) to reduce the effect of amplifying spurious signal from fishery changes and reporting patterns, that may incorrectly be attributed to change in abundance.

Each statistical observation in the commercial catch per unit effort (CPUE) analysis represented a single day of fishing by a single fishing operation, where a fishing operation may comprise a primary vessel and multiple dories. The response variable, 'logwt', was the log transformed weight of saddletail caught by that fishing operation. Each observation was constructed with separate fields for the following covariates:

- **year**: the year during which the fishing operation took place
- **month**: the month during which the fishing operation took place
- **latitude**: the latitude of the location in which the most saddletail were caught for that fishing operation
- **lunar**: the lunar phase at the time of the fishing operation
- **nboats**: the number of boats used
- **wind**: a binary variable, coded as '1' where the average wind speed on that day in that location exceeded 20 kilometres per hour (approx. 11 knots), otherwise '0'. More options were explored to define a threshold for wind speed (see Appendix B.8.3)
- **acn** ('authority chain number'): the identifier of the fishing operation
- **target**: a binary variable, coded as '1' where the fisher was deemed to be targeting saddletail during the fishing operation, otherwise '0'. Targeting was defined as a catch comprising of at least 70% saddletail. More options were explored to define targeting (see Appendix B.8.2).

The analysis was carried out using the software R (version 4.3.2, R Core Team (2023)). A Generalised Additive Model (GAM) approach was taken, using the *mgcv* package (Wood 2011).

The form of the model was:

$$\text{logwt} \sim \text{year} + \text{month} + \text{latitude} + \text{lunar} + \text{nboats} + \text{wind} + \text{acn} + \text{target} \tag{2.1}$$

GAM basis splines for the covariates were specified as follows:

- **latitude**: cubic regression spline of dimension 10
- **month**: cyclic cubic regression spline of dimension 10
- **lunar**: cyclic cubic regression spline of dimension 10
- **nboats**: thin plate regression spline of dimension 4
- **acn**: random effect

2.4.2 Recreational catch rates

Boat ramp survey data were used for the recreational catch rate analysis. The data were filtered prior to analysis according to the following criteria:

- interviews where East Coast fishers had the ‘opportunity’ to catch saddletail snapper, whereby they were:
  - Not fishing in rivers
  - Catching at least one “co-caught species” (saddletail, crimson, red emperor, common coral trout)
  - Using at least one suitable fishing method (line, spear)
  - Not charter fishing
- the number of released fish was less than 33 fish (which corresponds to the 97.5th percentile of discard quantities)
- the time spent fishing was less than 24 hours (i.e. day trips)

Each statistical observation in the recreational catch per unit effort (CPUE) analysis represented a fishing trip by the boat. The response variable, ‘saddletail’, was the number of saddletail caught by that fishing trip. Each observation was constructed with separate fields for the following covariates:

- **year**: the year during which the fishing trip took place
- **region**: the broad region where the fishing trip took place (above and including Cairns, between Cairns and Mackay, Mackay and below)
- **month**: the month during which the fishing trip took place
- **number of fishers**: the number of fishers on board the vessel (1, 2, 3, 4, 5+)
- **length of trip**: short (0–6 hr), medium (6–13 hr), long (13+ hr)
- **wind**: a binary variable, coded as ‘1’ where the average wind speed on that day in that location exceeded 15 kilometres per hour (approx. 8 knots), otherwise ‘0’.
- **day type**: weekend or weekday

The analysis was carried out using the software R (version 4.4.1, **r˙2025**). A Generalised Linear Model approach was taken, employing a negative binomial distribution and log-link function, using the *MASS* package (Venables and Ripley 2002). Other model forms were explored, including Tweedie, Poisson and quasi-Poisson families and GAMs which gave similar results.

The form of the model was:

$$\text{saddletail} \sim \text{year} * \text{region} + \text{month} + \text{number of fishers} + \text{length of trip} + \text{wind} + \text{day type} \tag{2.2}$$

2.4.3 Fishing efficiency

Accounting for fishing efficiency trends in catch rate indices is essential for ensuring accurate stock assessments and sustainable fisheries management. Fishing efficiency, often driven by advancements in technology, gear improvements, and increased fisher knowledge, can lead to higher catch rates even when fish abundance remains constant or declines. If these trends are not accounted for, catch per unit effort indices may overestimate stock abundance, resulting in overly optimistic assessments and potentially unsustainable retained catch levels. Adjusting catch rate indices for fishing efficiency means it is more likely that that changes in catch rates reflect true changes in fish populations rather than artefacts of improved fishing power. This is particularly important in longer catch rate time series, where cumulative increases in efficiency can significantly distort trends in abundance indices.

Many studies, both Australian and international, have explored the impact of technological advances (or “effort creep”) on a vessel’s fishing efficiency. Generally, it is estimated that fishing efficiency increases annual catch rates by 1–4% per year (Eigaard et al. 2014; Palomares and Pauly 2019; Sumpton et al. 2013; Lestang et al. 2018; O'Neill and Leigh 2007; Thurstan et al. 2016). Estimates are heavily influenced by the region, fishery and time period studied.

Project team members noted that the increased presence of sharks in recent years have influenced catch rates in the opposite direction: that is, depredation has contributed to fishing *inefficiency*. Specifically, project team members cited the impact on fisher behaviour due to the presence of sharks, whereby skippers are required to move spots more frequently to avoid losing fish to sharks. It was considered that fishers would have to drive further, spend more time driving or searching, and hook more fish to achieve the same daily retained catch in the presence of sharks.

As a baseline, we assumed a 2% efficiency gain each year, except in recent years (2021–2025) where sharks have become more prevalent, making the annual efficiency gain only 1% overall. Additional fishing efficiency schemes were explored in Appendix B.3.

2.4.4 Hyperstability

Hyperstability, or the "illusion of plenty", refers to a phenomenon in which stable catch rates mask a broader decline in the population (Hilborn and Walters 1992; Erisman et al. 2011). This occurs when fishing practices target predictable, dense aggregations of fish. Saddletail snapper are rarely solitary—they aggregate to spawn (Claydon 2005) and form large schools, often mixed with crimson snapper (*Lutjanus erythropterus*). Both of these behaviours make saddletail snapper catch rates prone to hyperstability.

To minimise the risk of hyperstability, the catch rate indices were given less emphasis in the model, by being input to the population model with broad standard errors (SE=0.2). This gave the population model greater flexibility to favour signal from marginal length and conditional age-at-length data inputs.

2.5 Biological relationships

2.5.1 Fork length and total length

All length measurements were provided in fork length (FL) and the population model was run using FL. The following conversion was applied where necessary (McPherson and Squire 1992):

$$TL_{mm} = 1.04 \times FL_{mm} - 0.45$$

where TL_{mm} is total length (mm) and FL_{mm} is fork length (mm).

2.5.2 Fecundity and maturity

Maturity values in the model were length-based, following a logistic function with coefficients supplied from Hillcoat (2022):

$$mat = \frac{1}{(1 + exp((57.3 - FL_{cm})/3.72))}$$

where mat is maturity and FL_{cm} is fork length in centimetres.

Following Equation 2.5.2, L_{50} (the length at which 50% of fish are reproductively mature) is equal to 57.3 cm. A previous study (McPherson and Squire 1992) reported L_{50} at 58 cm.

No information was available on the fecundity for saddletail snapper. For this assessment we assumed that weight and fecundity of saddletail snapper are parametric functions of their size. Per the Stock

Synthesis manual, by using Fecundity Option 1 and setting $a = 1$ and $b = 0$, eggs were equivalent to spawning biomass (Methot et al. 2024).

2.5.3 Weight and length

The weight-length relationship was taken from McPherson et al. (1992):

$$W_{kg} = \exp(-10.5 + (2.83 * (\log(FL_{cm}))))$$

where W_{kg} is weight of a saddletail in kilograms.

2.6 Length and age data

2.6.1 Input to the population model

Three composition data sets were input to the population model: commercial lengths, recreational lengths, and conditional age-at-length (in which both sectors were combined).

For conditional age-at-length compositions, only length and age data of specified sex (male or female) were included, with fish of unknown sex (sex not able to be determined from sample) excluded. Given the sex-specific growth patterns exhibited by saddletail snapper, it was important that the sex-specific growth curves were produced only from samples where the sex was known.

In the case of both commercial and recreational length compositions, length data of all sexes (male, female, unknown) were included, and input to the model as pooled (sex = 0 means combined male and female, see page 61 of Methot et al. (2024)). A large proportion of the available length samples are obtained from whole fish, where sex cannot be determined. Length data were input to the population model in 2 cm length bins.

2.6.2 Data weighting

Each population model used one of two methods to weight the length and age data. Both methods for weighting data (Method 1 and Method 2) were applied in separate population models, and the results from both models were included in the ensemble, contributing equally to the final stock assessment outcomes.

For each of the three composition data sets, there were three stages of data weighting:

- Stage 1: Pre-adjustment of sampling weights
- Stage 2: Annual sample size adjustment
- Stage 3: Statistical weighting in the model

Stage 1 (pre-adjustment of sampling weights) serves two purposes: to address irregularities and potential bias in sampling of data (across space and time), and to ensure the data is fit-for-purpose when entered into the model.

Length compositions inform several aspects within a stock assessment model: both fishery dynamics (e.g. selectivity, fishing mortality) and population dynamics (e.g. recruitment). Inspired by Maunder et al. (2020), two methods have been used to weight length composition data; one to represent the removals from the population by fishing, and one to represent the population distribution.

- Method 1, which was used in Campbell et al. (2021), weighted individual samples (i.e. individual fish) such that the composition data were more representative of the fish removed from the fishery.

This was done by adjusting the contributions of individual commercial fishing samples from each region and year proportional to the relative catch of saddletail snapper from commercial logbooks. Regional total catch data are not available for the recreational sector so this process was not applied to recreational length compositions collected by the boat ramp survey program. Therefore, each individual fish from the boat ramp survey program was given the same weighting.

- Method 2 weighted individual samples such that the composition data were more representative of the population. This was done by applying a statistical algorithm that calculated the effective weight of each individual such that there was equal weighting across each strata (i.e. year and region) while preserving the variance of the original weighting. The algorithm accounts for bias by reducing the effective sample size proportionally to the degree of adjustment applied to the weights. This approach preserves the variance of the original weighting while improving statistical robustness.

Both methods incorporated a subsequent adjustment to sample sizes to account for the percentage of catch that was measured by fishery staff.

This stage was not applied to conditional age-at-length data; each individual in that dataset was given the same weighting.

Stage 2 (annual sample size adjustment) (referred to as 'Nsamp' in the Stock Synthesis manual) was dependent on the method used in stage 1.

For Method 1, initial sample sizes of length compositions were based on the sampling unit of a 'catch' (defined in Appendix D.2).

For Method 2, the initial sample sizes of length compositions were the sum of the effective weights of each individual.

As is accepted practice, initial sample sizes of conditional age-at-length compositions were based on the sampling unit of individual fish.

Stage 3 (statistical weighting in the model) was implemented via variance adjustment factors in the Stock Synthesis control file, using the Francis weighting values suggested by an initial run of the model.

2.7 Spatiotemporal patterns in age and length

Exploratory analysis was conducted for the age and length data using a suite of GAMs (Generalised Additive Models) prior to population model optimisation. The purpose of this analysis was to identify any spatial or seasonal structure in the data that may warrant stratification in the population model structure (i.e. areas as fleets, or growth morphs (see Stock Synthesis manual, Methot et al. (2024)). This provided insight into structure in the biological data, and hence informed the structure of the population model.

In order to model the spatial structure of the composition data effectively, the catch location of each sample was required as a continuous latitude. Since the spatial resolution of the catch location field varies as described in Appendix D.2, a consistent method of extracting latitude information from these catch locations was required. The following process was undertaken to obtain continuous measures of latitude for age and length samples:

- For each individual catch location reported in the age and length data, a list was created of 30x30 minute grid cells that comprise that location. Depending on the inherent resolution of the original catch location, this may be just one grid, or it may be multiple.
- From the resultant groupings of grids, a continuous latitude and longitude point was sampled at random. This informed the latitude and longitude fields associated with each sample.
- The subsequent models were constructed and run multiple times, to investigate whether spatial and seasonal trends were sensitive to the random sampling process. Bias due to the random sampling of spatial data from within catch locations was not observed.

2.7.1 Age response model

Two GAMs were constructed to explore spatial and seasonal patterns in the age of saddletail snapper. The form of the first age-response model was:

$$Age \sim Sex + te_{Month} + te_{Latitude} + ti_{Month,Latitude} + Year + Sector \quad (2.3)$$

and the second was:

$$Age \sim Sex + te_{Month} + te_{Latitude.Sex} + ti_{Month,Latitude} + Year + Sector \quad (2.4)$$

where:

- **Sex** was the sex of fish (male or female)
- **Month** was the calendar month
- **Latitude** was continuous latitude, sampled as per methods in the above section
- **Year** was the fishing year
- **Sector** was the fishing sector (either commercial or recreational)

2.7.2 Length response model

A GAM was constructed to explore spatial and seasonal patterns in the length of saddletail snapper. The form of the length-response model was:

$$FL \sim Sex + te_{Month} + te_{Latitude} + ti_{Month,Latitude} + Year + Sector \quad (2.5)$$

where:

- **FL** was the fork length of fish (in cm)
- **Sex** was the sex of fish (male or female)
- **Month** was the calendar month
- **Latitude** was continuous latitude, sampled as per methods in the above section
- **Year** was the fishing year
- **Sector** was the fishing sector (either commercial or recreational)
- **te** represents a tensor product
- **ti** represents a tensor interaction

2.7.3 Age-at-length model

A GAM was constructed to explore spatial and seasonal patterns in the age-at-length of saddletail snapper. In this case, age was still the response variable, but length was included as a model term. The form of the age-at-length model was:

$$Age \sim Sex + te_{FL} + te_{Month} + te_{Latitude} + ti_{FL,Latitude} + Year + Sector \quad (2.6)$$

where:

- **FL** was the fork length of fish (in cm)
- **Sex** was the sex of fish (male or female)
- **Month** was the calendar month
- **Latitude** was continuous latitude, sampled as per methods in the above section
- **Year** was the fishing year
- **Sector** was the fishing sector (either commercial or recreational)
- **te** represents a tensor product
- **ti** represents a tensor interaction

2.8 Population model

2.8.1 Description

The software Stock Synthesis (SS) (Methot and Wetzel 2013; Methot et al. 2024), version 3.30.23.2, was used for the population model. A full technical description of Stock Synthesis is given in Methot et al. (2024).

2.8.2 Model structure

Biological monitoring data indicated a growth difference between the sexes with males growing larger than females. The population model was therefore set up as a two-sex model.

The stock assessment model had three fleets, representing different components of the fishery: 1) commercial, 2) recreational and Traditional Owners combined, and 3) charter. The selectivity pattern for the charter fleet was configured to mirror that of recreational and Traditional Owners, in lieu of charter specific length composition data.

2.8.2.1 Spatial structure

The spatio-temporal modelling of Queensland east coast age and length data (detailed in Section 2.7, results in Appendix D.4.3) indicated that the data did not support any hypotheses of potential spatial or seasonal variation in the size availability of saddletail snapper. This result did not provide evidence against the assumption that selectivity is comparable throughout Queensland east coast waters and as such, a single area stock assessment model was used.

Some trend is visible in the partial effect of latitude upon fork length in Figure D.8, however this is largely attributable to data sparsity in the far north of the region, with very few samples and some large individuals. A cyclical trend is observed in the partial effect of month upon lengths, though this is expected and reflects the seasonal timing of new recruits entering the fishery (nominal birth date of 1 October). When the full interaction between latitude and month is observed (Figure D.8), it can be seen that the data is somewhat noisy and there is no clear latitudinal or seasonal trend in lengths.

While some potential evidence of differing age-at-length by latitude was apparent for particularly large individuals (see Figure D.11), this is equally likely to be driven by data sparsity in the largest size classes at the extremities of latitude. No such latitudinal trend in age-at-length was evident for fish of more common size classes (see Figure D.11). This result was deemed as insufficient evidence to implement the ‘growth morphs’ functionality of Stock Synthesis (Methot et al. (2024), page 88) and model separate growth patterns by latitudinal strata. Therefore, a single growth curve per sex was estimated throughout Queensland east coast waters and across seasons. As such, the model fleets were not further split by area or season.

2.8.3 Model assumptions

Models rely on a set of assumptions in order to simplify real-world processes into mathematical equations. When data is scarce, modellers may need to make additional assumptions to fill gaps or simplify processes that cannot be directly observed or measured. These assumptions help the model function but may not fully capture the complexity of the real world. These assumptions are often necessary to make the model workable, especially when data is limited. Attempts to use limited data to capture too many fishery and biological processes can lead to an *over-parameterised* model, in which the data is spread too thinly and statistical confidence in the results is misleadingly high. While assumptions are typically a necessity for a workable model, they can introduce biases or caveats, which should be acknowledged and considered when interpreting the results.

The main assumptions underlying the model included:

- Fish swim freely and mix instantaneously throughout Queensland east coast waters. In reality, the time-scale of mixing is unknown, but slower.
- The Queensland east coast can be considered a single stock with no immigration or emigration.
- At finer spatial scales, demographic structure can be handled through weighting of input data.
- The stock was in an unfished state in 1961.
- The fraction of fish that are female at birth is 50% and fish do not change sex during their life.
- Growth occurs according to the von Bertalanffy growth curve.
- Length at age follows a single growth curve (by sex).
- The weight and fecundity of saddletail snapper are parametric functions of their size (Figures D.3 and D.7).
- Fish can reach maturity as early as three years of age, after which the proportion of mature fish increases with age and size (Figure D.4).
- Fishing selectivity of saddletail snapper is a function of length, and not age.
- Expected mean annual recruitment for a given stock size follows a Beverton-Holt function.
- The instantaneous natural mortality rate M is constant and does not depend on size, age, sex, or year.
- Standardised catch rates, following standardisation and after accounting for fishing efficiency changes, are proportional to whole-Queensland east coast abundance, and inform proportionally on the annual change in abundance of legal-sized saddletail snapper.
- Discards of commercially caught saddletail snapper are negligible.
- Depredation was assumed to be constant at 1% for all fleets until 2009, after which it was assumed to increase linearly to a maximum of 29% (commercial fleet) or 58% (other fleets) by 2025.

2.8.4 Model parameters

Model parameters are listed in Table 2.2. Recruitment deviations were estimated for all years of the stock assessment model. Care was taken to ensure that no significant trend in recruitment deviations was present in years exceeding the reach of influence from cohorts of fish contained in the available years of age data. Efforts were also made to ensure that the time series of recruitment deviations did not begin or end suddenly with a large value, as this may cause results to depend on the choice of recruitment deviation years to an unacceptable degree.

Table 2.2: Parameters included in the population model

Symbol	Description	Value
M	Natural mortality (yr^{-1})	Fixed at 0.11 or 0.14
L_{minF} L_{minM}	Mean fork length of fish at minimum age (male & female, in cm)	Estimated
L_{maxF} L_{maxM}	Mean fork length of fish at maximum age (male & female, in cm)	Estimated
A_{min}, A_{max}	Reference minimum and maximum age for estimation of growth curve (yr)	2, 20
K_{F} K_{M}	von Bertalanffy growth coefficient (yr^{-1}) (male & female)	Estimated
$\mathrm{SD}_{\mathrm{youngF}}$ $\mathrm{SD}_{\mathrm{youngM}}$	Standard deviation of fork length at minimum age (male & female, in cm)	Estimated
$\mathrm{SD}_{\mathrm{oldF}}$ $\mathrm{SD}_{\mathrm{oldM}}$	Standard deviation of fork length at maximum age (male & female, in cm)	Estimated
h	Beverton-Holt stock recruitment steepness	Fixed at 0.65, 0.75 or 0.85
$\ln R_0$	Log of number of recruits when unfished (1961)	Estimated
σ_R	Standard deviation of natural log recruitment	0.6
$\mathrm{Sel}L_{50,com}$	Length at 50% commercial selectivity (FL cm)	Estimated
$\mathrm{Sel}L_{\mathrm{width,c}}$	Difference in lengths at 50% and 95% commercial selectivity (FL cm)	Estimated
$\mathrm{Sel}L_{p1,rec}$	Gradient at first node in recreational selectivity	Estimated
$\mathrm{Sel}L_{p2,rec}$	Gradient at last node in recreational selectivity	Estimated
$\mathrm{Sel}L_{p3,rec}$	Position of node 1 for recreational selectivity spline (FL cm)	37
$\mathrm{Sel}L_{p4,rec}$	Position of node 2 for recreational selectivity spline (FL cm)	54
$\mathrm{Sel}L_{p5,rec}$	Position of node 2 for recreational selectivity spline (FL cm)	80
$\mathrm{Sel}L_{p6,rec}$	Recreational selectivity value at node 1 (in ln(selectivity))	Estimated
$\mathrm{Sel}L_{p7,rec}$	Recreational selectivity value at node 2 (in ln(selectivity))	Fixed (others estimated relatively)
$\mathrm{Sel}L_{p8,rec}$	Recreational selectivity value at node 2 (in ln(selectivity))	Estimated
recdev	Recruitment deviations between 1961 and 2025	Estimated

Likelihood profile diagnostics indicated that the data inputs did not contain enough signal to appropriately estimate either natural mortality or steepness (see Appendix E.2.1). Instead, natural mortality was fixed at either at 0.11 or 0.14 yr^{-1}. These values were derived from the empirical formula $M = 5.4/A_{max}$ (Hamel and Cope 2022), where A_{max} is equal to 49 or 39 years—the oldest or second oldest fish in the dataset respectively. The second oldest fish was deemed appropriate to consider in this context, as the oldest fish could potentially represent a statistical outlier, be influenced by ageing error, or fail to reflect the typical longevity of the population. Additional higher values of natural mortality were explored (consistent with hypothetical lower maximum ages), with results shown in Appendix B.7.

Beverton-Holt stock recruitment steepness (h) is difficult to estimate reliably, and a common approach for h is to represent uncertainty by considering a range of values and report on them all. Accordingly, h was fixed at a range of different values in this assessment: 0.65, 0.75 and 0.85. The previous assessment estimated steepness at 0.76, using an informative prior with a median of 0.73 from Thorson (2020). The range of values used in this assessment centre around this estimate. Two further values of steepness were explored to test the model's sensitivity to h (0.55 and 0.95), with results shown in Appendix B.4.

Commercial length-based selectivity was defined by a logistic curve with two parameters (Stock Synthesis pattern 1, Methot et al. (2024), page 149), both of which were estimated. The recreational length-based selectivity curve was define as a cubic spline with three nodes (Stock Synthesis pattern 27, Methot et al. (2024), page 156). This pattern was used to provide extra flexibility to capture the bimodal nature in the recreational length distributions (see Results, Section 3.1.5 for more details). Stock Synthesis' auto-generation tool was used for the recreational parameter values (p1 code = 1), which allowed Stock Synthesis to determine suitable locations for each node of the spline. Knot positions were fixed, and the value of selectivity at each node was estimated except for one (which was held constant to allow others to be estimated relative to it). Gradients at the first and last node were also estimated. Alternative knot positions and values were explored in Appendix B.6. Selectivity for the charter fleet was mirrored from the recreational fleet.

Standard deviation of natural log recruitment (σ_R) was fixed at 0.6. This is an increase from the value of 0.3 in the previous assessment. Recruitment deviations between 1961 and 2025 improved fits to composition data and abundance indices. The previous assessment allowed recruitment deviations from 1981 until the end of the assessment period, for the base case.

2.8.5 Ensemble

The project team considered the likelihood of the 12 models and recognised that they all were realistic and reasonable explanations of the fishery. No model significantly stood out to justify being treated as a 'base case'. Therefore, the ensemble method was considered appropriate in this stock assessment. All twelve model scenarios were included in the final ensemble to determine the model's sensitivity to different parameter values and assumptions (Table 2.3).

Each model scenario was initially optimised using a maximum likelihood estimation with one iteration of re-weighting using Francis variance adjustment factors (Methot et al. (2024), page 171). Each scenario was then run using a Markov chain Monte Carlo (MCMC) framework, with 250 000 iterations per scenario, using no burn in and a thinning interval of 1000 (Methot et al. (2024), page 194). Convergence was assessed via potential scale reduction (R-hat) values, and visual assessment of posterior density and trace plots (Appendix E.3). An ensemble result was created by combining MCMC outputs from each scenario into a single dataframe, giving equal weighting to each scenario.

The ensemble explored three axes of uncertainty, testing two values of natural mortality (M), three values of steepness (h), and two methods for weighting the length composition data that were input to the model (as described in Section 2.6.2).

Table 2.3: Scenarios tested to determine sensitivity to parameter values and assumptions

Scenario	M (yr^{-1})	h	Length data weighting
1	Fixed at 0.14 yr^{-1}	0.75	Method 1
2	Fixed at 0.14 yr^{-1}	0.65	Method 1
3	Fixed at 0.14 yr^{-1}	0.85	Method 1
4	Fixed at 0.14 yr^{-1}	0.75	Method 2
5	Fixed at 0.14 yr^{-1}	0.65	Method 2
6	Fixed at 0.14 yr^{-1}	0.85	Method 2
7	Fixed at 0.11 yr^{-1}	0.75	Method 2
8	Fixed at 0.11 yr^{-1}	0.65	Method 2
9	Fixed at 0.11 yr^{-1}	0.85	Method 2
10	Fixed at 0.11 yr^{-1}	0.75	Method 1
11	Fixed at 0.11 yr^{-1}	0.65	Method 1
12	Fixed at 0.11 yr^{-1}	0.85	Method 1

2.8.6 Exploratory sensitivity scenarios

Additional stock assessment model runs were undertaken, to better understand the model's sensitivity to various modelling assumptions. Many of these scenarios did not substantially affect results, and were therefore excluded from the final ensemble. We report on the influence of these factors nonetheless, and results of these scenario runs can be seen in Appendix B. These additional scenarios explore the stock assessment model's sensitivities to the following factors:

- Depredation
- Historical commercial catch adjustment
- Fishing efficiency trends over time
- Additional values of steepness
- Time-varying commercial selectivity
- Adjusting length data sample sizes to improve fits to commercial length compositions
- Alternative specifications for recreational selectivity patterns, including a logistic curve
- The interaction between the start year of recruitment deviations, steepness, and natural mortality, and how this interacts with the removal of catch rate data

In addition to these exploratory stock assessment models, Appendix B.8 includes the results of some exploratory work to explore the sensitivity of the commercial catch rate analysis to various assumptions; namely the time span of data included, the way in which 'targeting' was defined and the impact of the threshold used for the wind variable.

3 Results

3.1 Model inputs

3.1.1 Data availability

Model inputs are described for Queensland east coast saddletail snapper. Model outputs in this section relate to Scenario 1 as a reference scenario (as defined in Table 2.3).

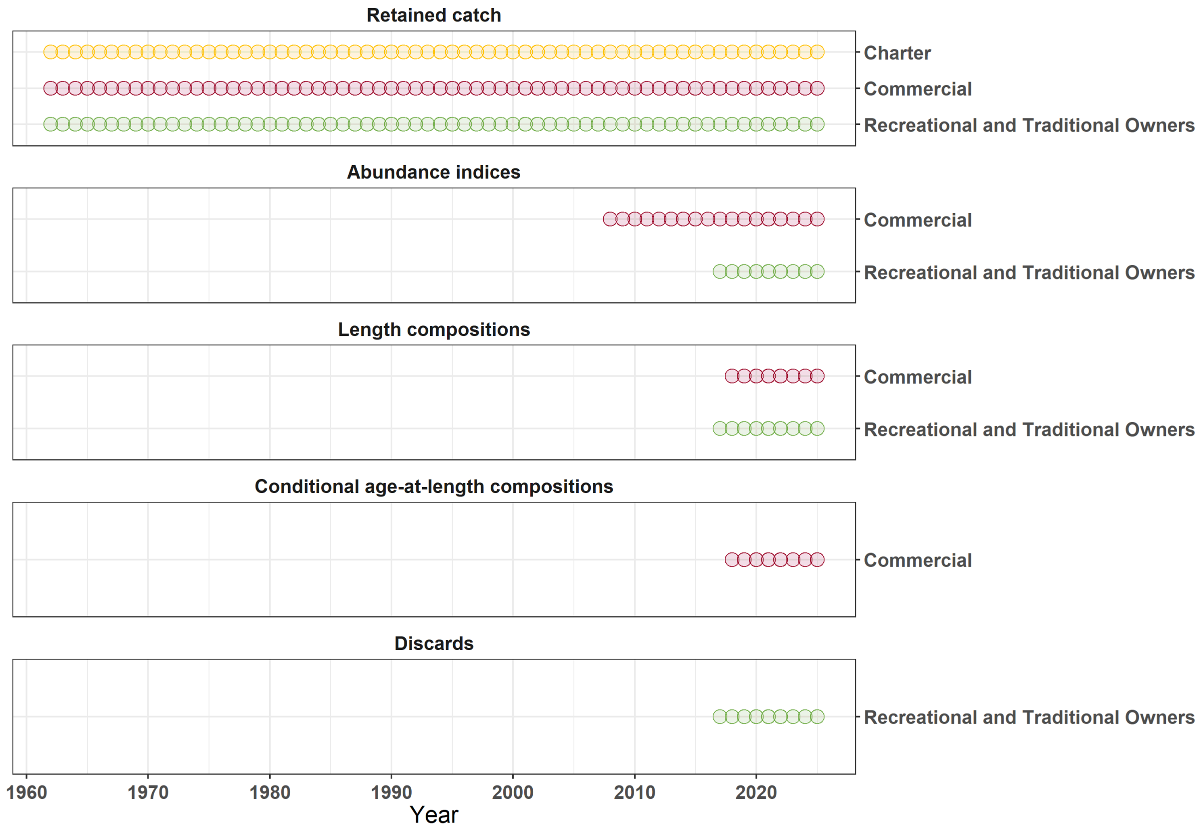

Figure 3.1: Data presence by year for each category of data type for Queensland east coast saddletail snapper

3.1.2 Retained catch estimates

Time series of saddletail snapper retained catch are shown in Figure 3.2, separated by fishing fleet. Annual catches were estimated using data sources and methods specific to each sector, as described in Section 2.2. The catch estimates shown in Figure 3.2 include the estimated additional mortality due to depredation adjustments outlined in Section 2.2.5, indicated by the lighter shading. These estimates do not include discard mortality, as discard estimates and discard mortality rates are input to the population model separately.

These estimates indicate that total annual catches (all sectors combined) on Queensland's east coast increased consistently since 1961, reaching an initial peak in 1999 of 668 t, decreasing around 2007 to approximately 478 t and remaining relatively stable until 2014, then increasing to an all time high in

2021 of 851 t. The majority of catch was contributed by the recreational sector. The peak in recent years would not be as prevalent if depredation was not accounted for in the catch reconstruction. Over the last five years, on average depredation accounted for 31% of total catch across all sectors. Total catches with or without the influence of depredation can be seen in Appendix B.1.

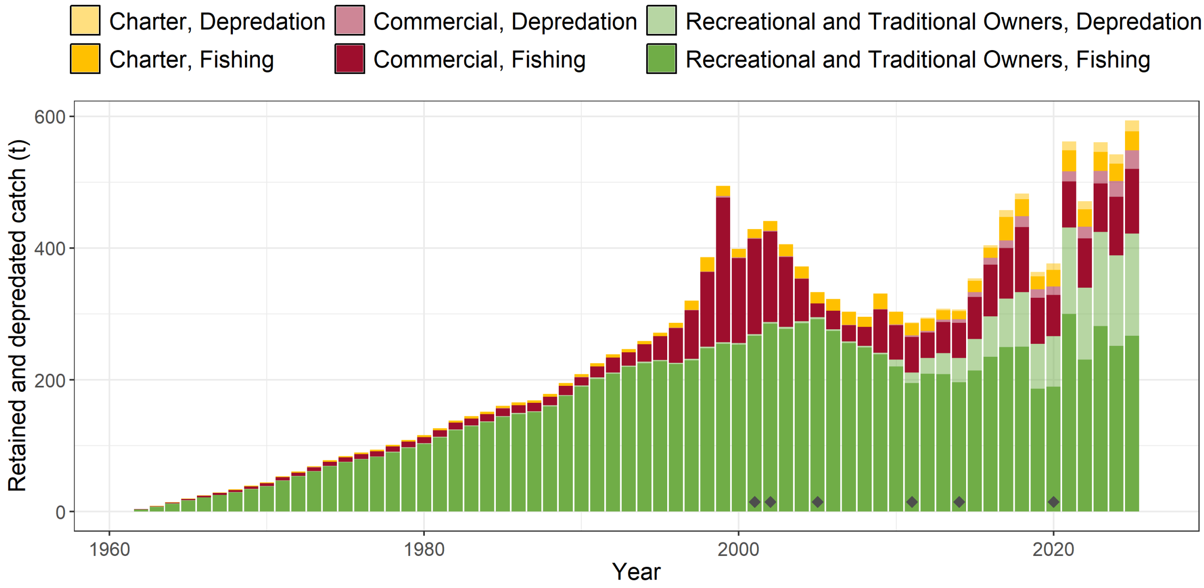

Figure 3.2: Estimated retained catch by sector between 1961 and 2025 for Queensland east coast saddletail snapper—lighter shaded areas represent the estimated additional mortality due to depredation, diamonds represent years where statewide recreational fishing surveys were conducted

Figure 3.3 shows the retained catch from the commercial sector (inclusive of depredation) in grey, with three alternative adjustments to account for potential overreporting between 1997 and 2004 as described in Section 2.2.1.2. The green line (an adjustment of 83%) was used as a base case scenario, with the other options explored in Appendix B.2

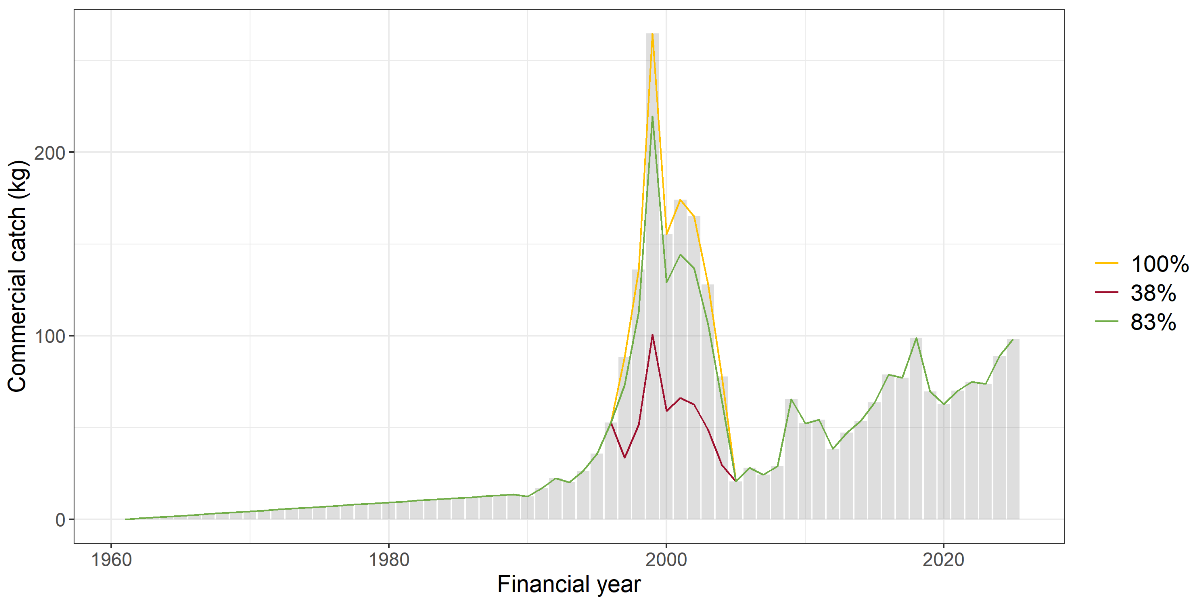

Figure 3.3: Estimated retained catch by sector between 1961 and 2025 for Queensland east coast saddletail snapper

Spatial patterns of retained catch from the commercial sector for periods 1989–2004 and 2005–2025 are shown in Figure 3.4. This shows some expansion into the northern part of the fishery in 2005–2025.

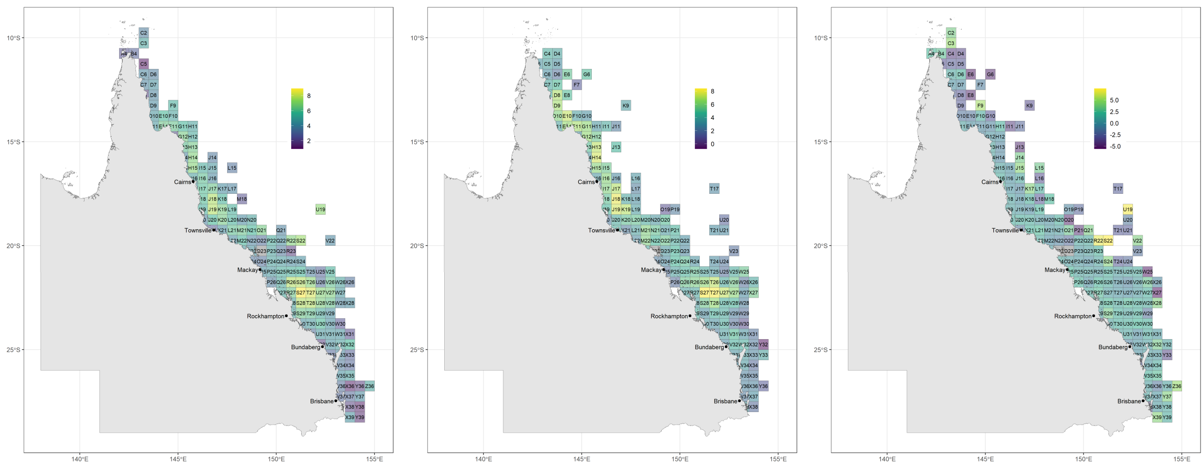

Figure 3.4: Comparison of spatial distribution of commercial catch (excluding depredation, on a logarithmic scale) for Queensland east coast saddletail snapper—the left panel shows the mean annual commercial catch between 1989 and 2004, the middle panel shows the same between 2005 and 2025, and the right panel shows the difference between the two

3.1.3 Standardised indices of abundance

Overall, the index shows little contrast over the time series (Figures 3.5 and 3.6, solid lines) after accounting for gains in fishing efficiency. This stable pattern could be due to several reasons, including a stable population or one that exhibits hyperstablity (due to the tendency of saddletail snapper to school in numbers, leading to potential hyperstability in catch rates).

When input into Stock Synthesis, the catch rate index was given a standard error of 0.2 to allow the population model freedom to fit more closely to biological data inputs. Model fits to the index shown in Section 3.2.2 (Figures 3.14 and 3.15). Model diagnostics were satisfactory and can be found in Appendix .

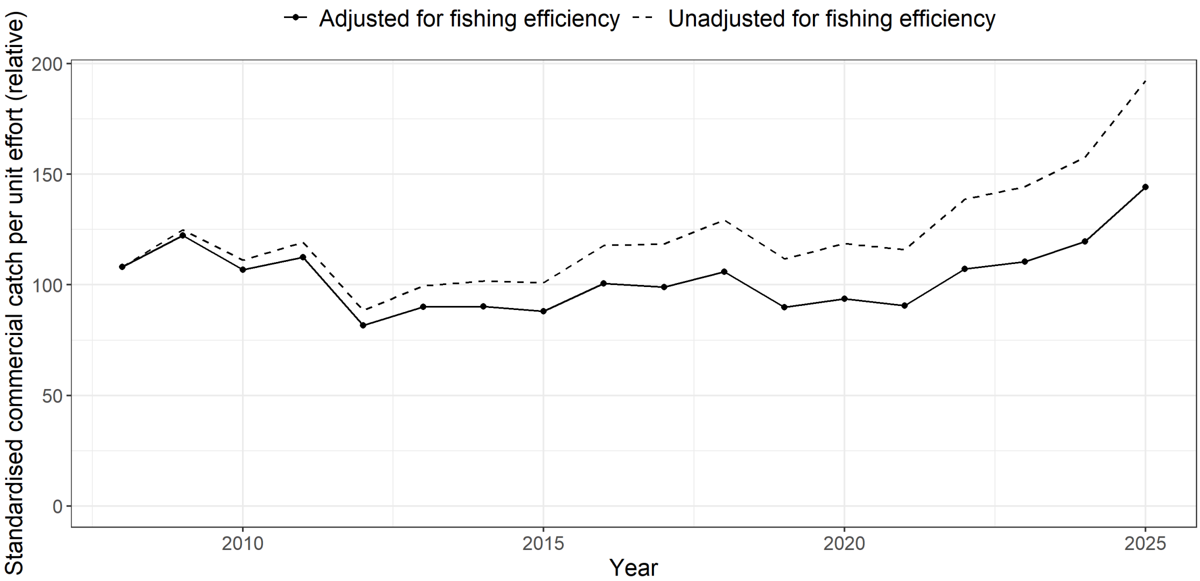

Figure 3.5: Annual standardised commercial catch rates for Queensland east coast saddletail snapper

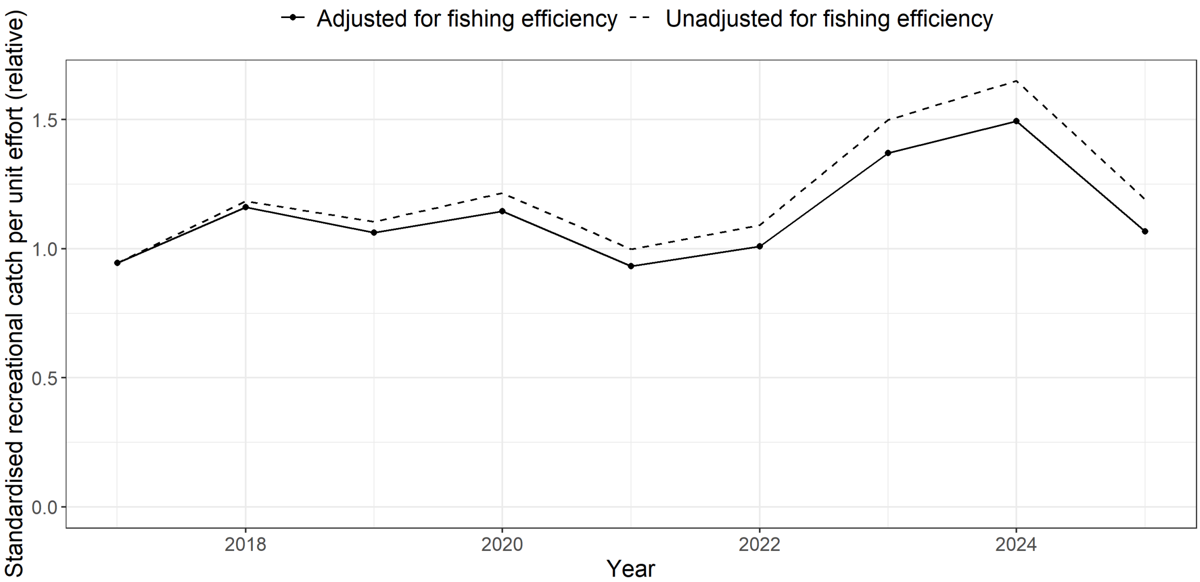

Figure 3.6: Annual standardised recreational catch rates for Queensland east coast saddletail snapper

3.1.4 Discards

Discard estimates (in tonnes) by the Queensland recreational fleets, as input into the model, are shown in Figure 3.7.

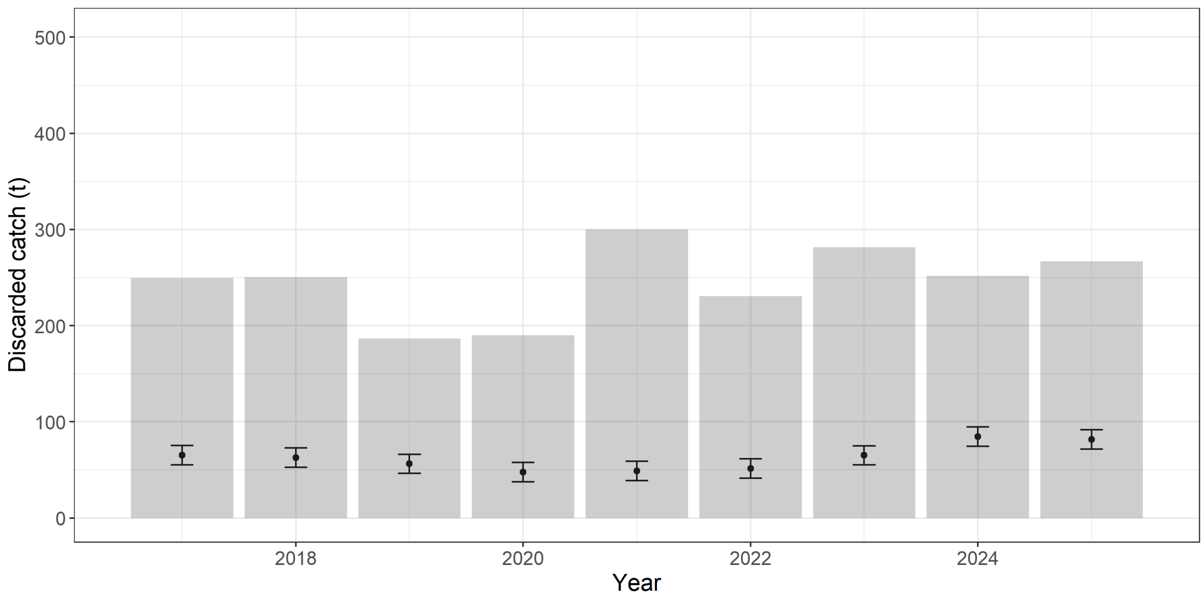

Figure 3.7: Annual discards of Queensland east coast saddletail snapper—points represent recreational discard estimates, whiskers represent uncertainty around the estimates, and grey bars represent recreational harvest between 2017 and 2025

3.1.5 Length composition

Length compositions of Queensland east coast saddletail snapper are shown by fleet, aggregated across time (Figures 3.8 and 3.9) and annually by sector (Figures 3.10 and 3.13).

The length compositions differ considerably by fleet (Figure 3.8), suggesting that selectivity patterns differ by sector. The commercial length data consist of only fish above MLS and the distribution of fish across length classes is relatively normal—this was compatible with a logistic length-based selectivity curve for the fleet. By contrast, the recreational length-based selectivity curve was defined as a cubic spline with three nodes (Stock Synthesis pattern 27, Methot et al. (2024), page 156). This pattern was used to provide the recreational selectivity additional flexibility needed to capture the bimodal nature of the recreational length frequencies (Figures 3.8). Bimodality of recreational length frequencies is attributable to the fact that recreational vessels fish both inshore and offshore, where the available size classes of saddletail differ due to ontogenetic shift (Davis et al. 2025). This is not observed in the commercial length distributions as the focus of this fleet is primarily offshore.

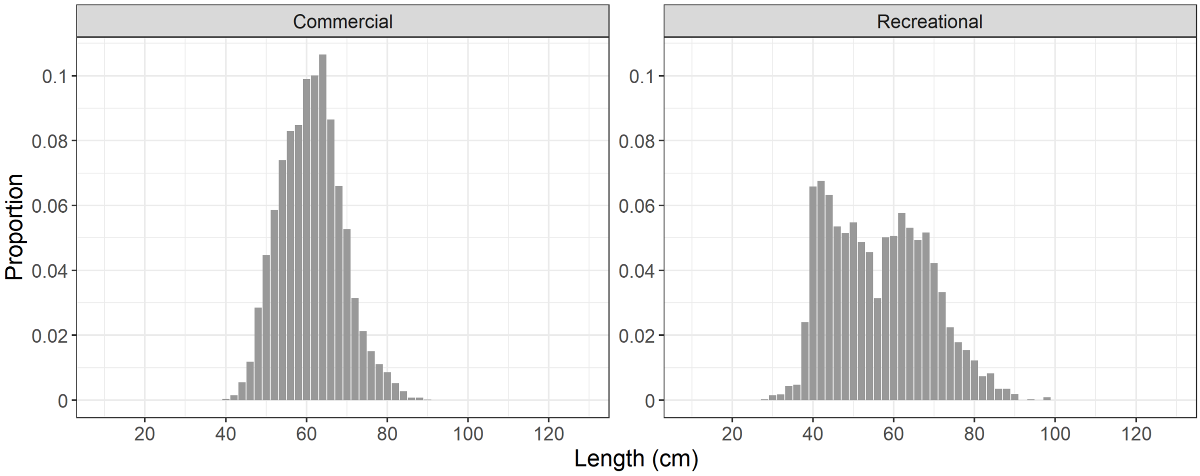

Figure 3.8: Length compositions of Queensland east coast saddletail snapper aggregated across time by fleet and by sex—scenarios 1–6, weighted using Method 1 described in Section 2.6.2

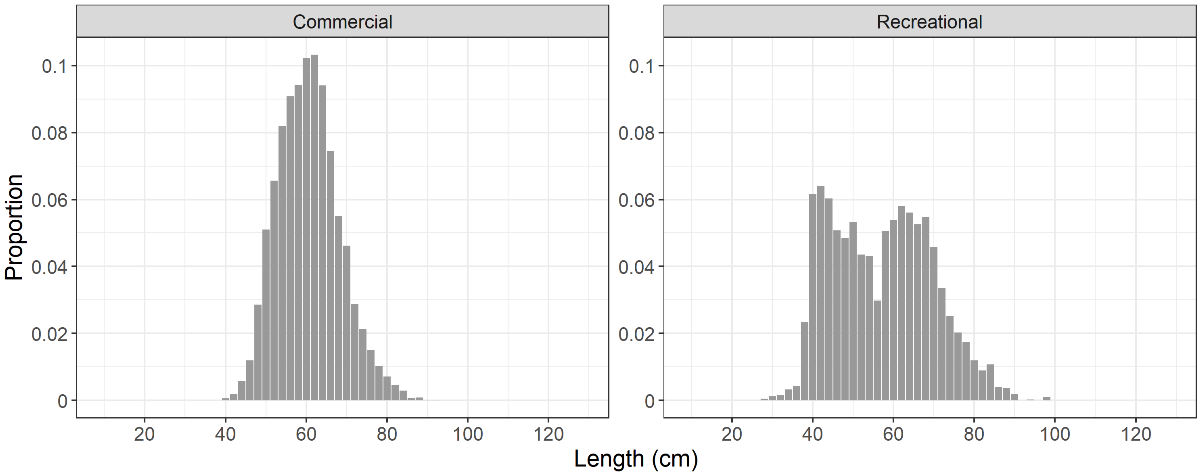

Figure 3.9: Length compositions of Queensland east coast saddletail snapper aggregated across time by fleet and by sex—scenarios 7–12, weighted using Method 2 described in Section 2.6.2

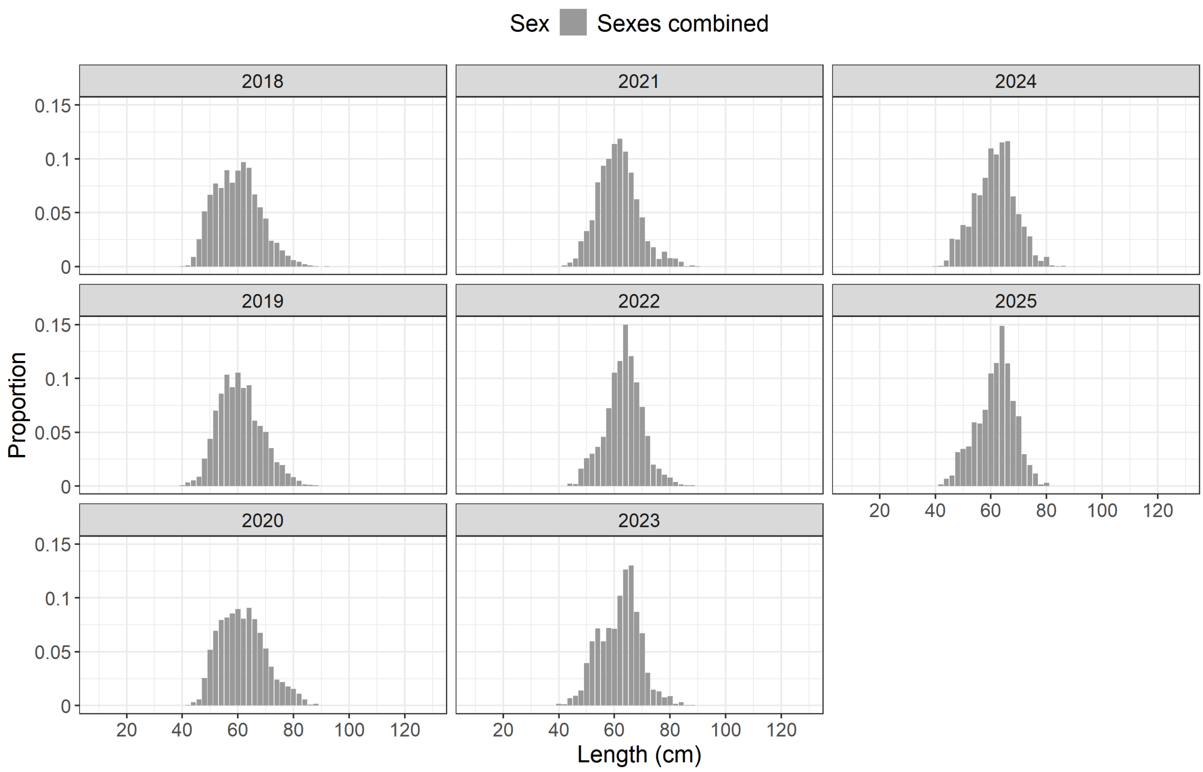

Figure 3.10: Annual commercial length compositions of Queensland east coast saddletail snapper—scenarios 1–6, weighted using Method 1 described in Section 2.6.2

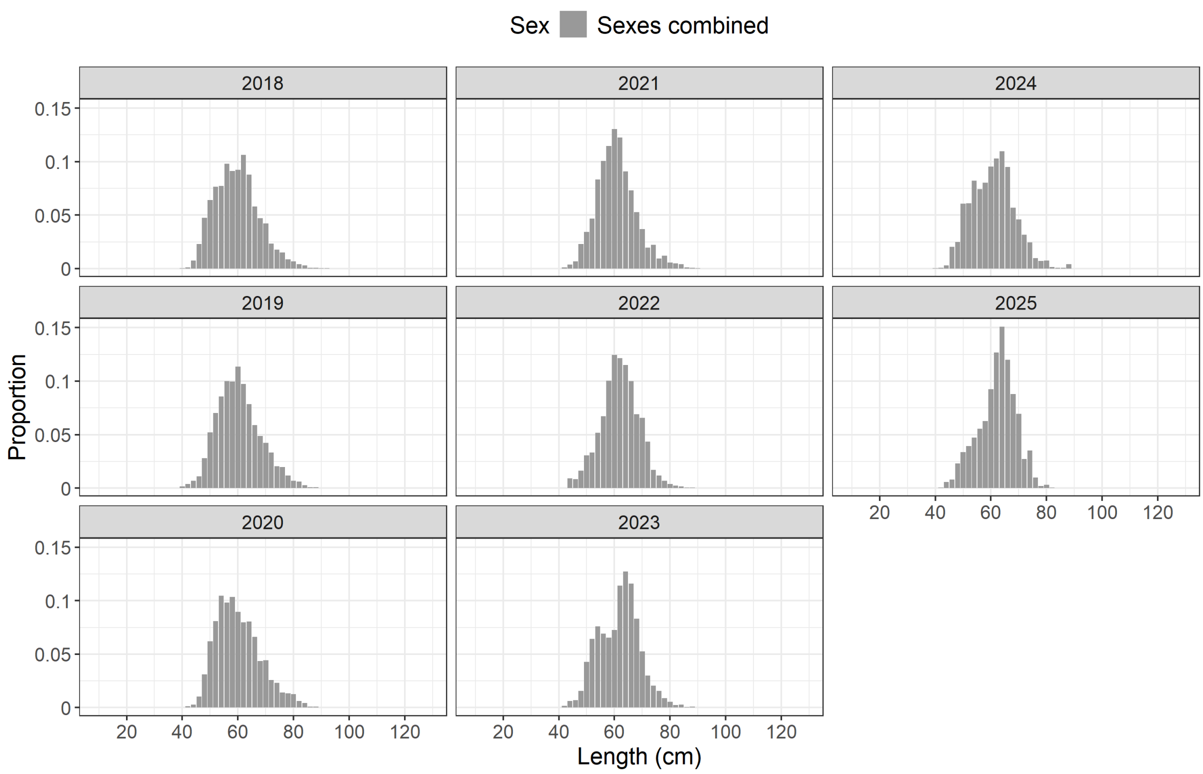

Figure 3.11: Annual commercial length compositions of Queensland east coast saddletail snapper—scenarios 7–12, weighted using Method 2 described in Section 2.6.2

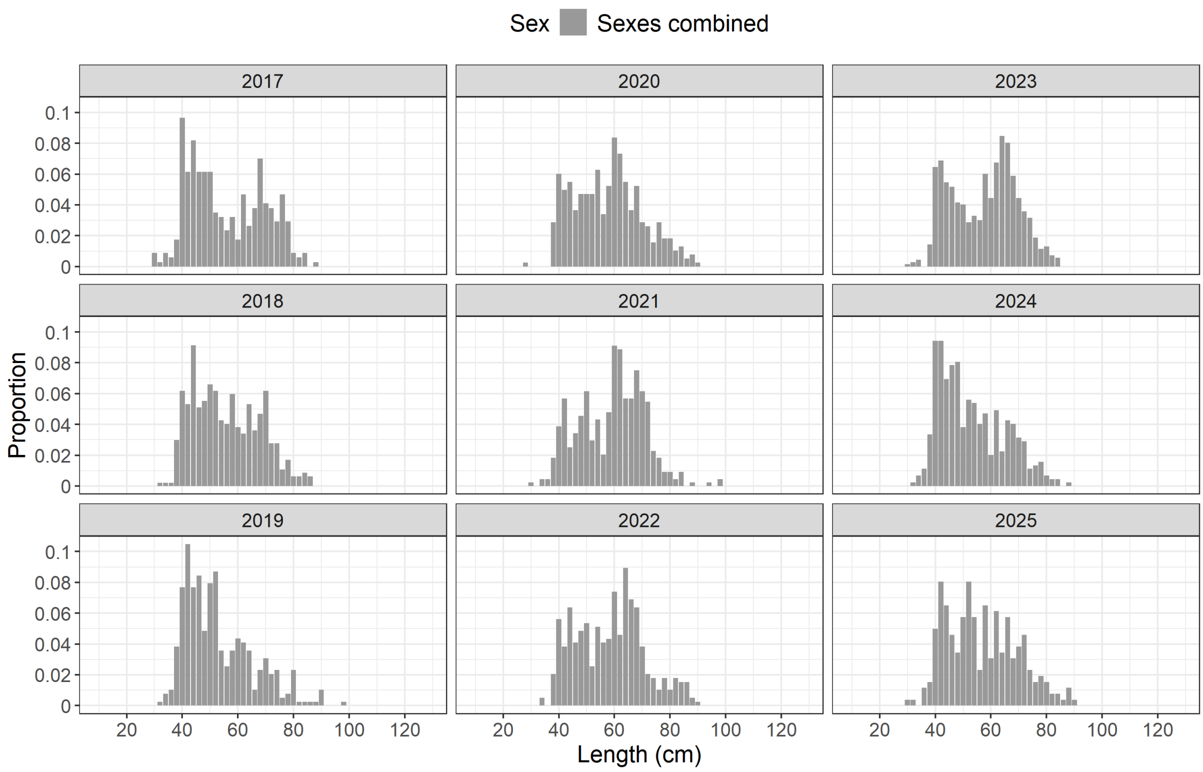

Figure 3.12: Annual recreational length compositions of Queensland east coast saddletail snapper—scenarios 1–6, weighted using Method 1 described in Section 2.6.2

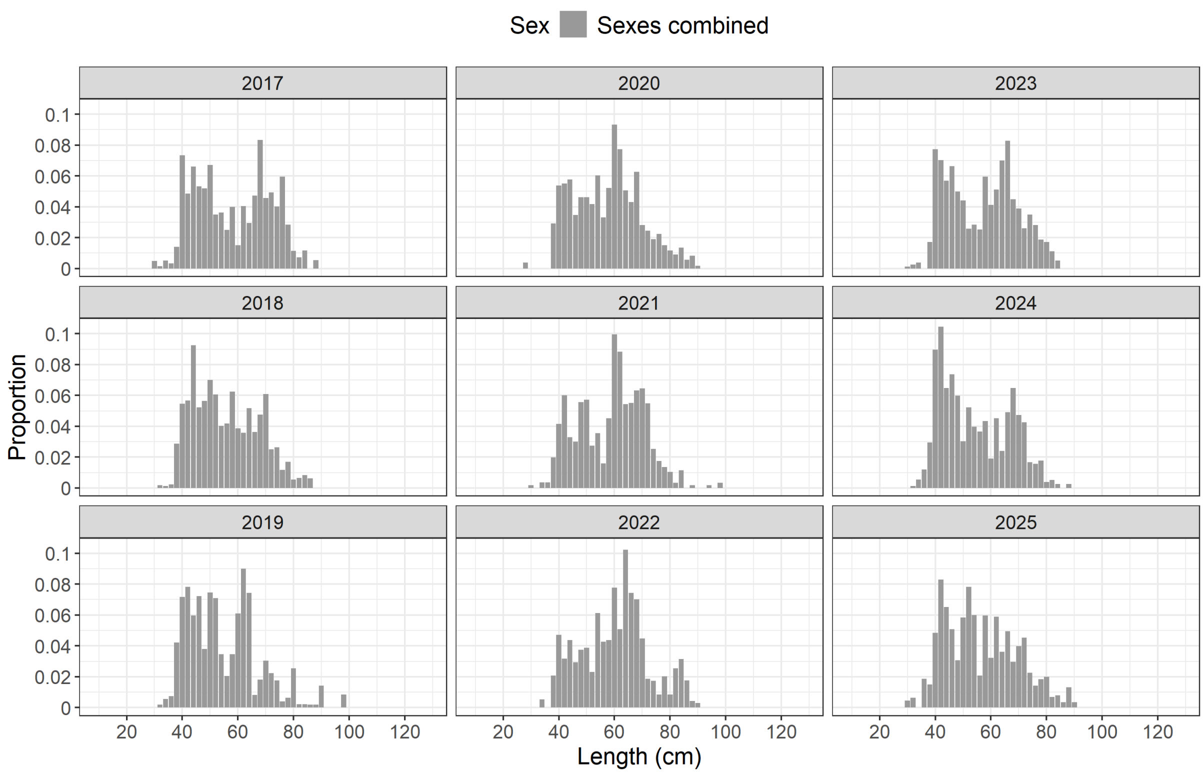

Figure 3.13: Annual recreational length compositions of Queensland east coast saddletail snapper—scenarios 7–12, weighted using Method 2 described in Section 2.6.2

3.1.6 Other model inputs

Fixed biological relationships are plotted in Appendix D.4, Figures D.3–D.7. These include the length–weight relationship, maturity at age, and individual spawning output (maturity multiplied by fecundity) by age and by length.

3.2 Model outputs

3.2.1 Model parameters

Parameter estimates across the ensemble are listed in Table 3.1.

Symbol	MCMC median	MCMC 2.5%	MCMC 97.5%
L_at_Amin_Fem_GP_1	39.4	38.4	40
L_at_Amax_Fem_GP_1	71.7	71.2	72.3
VonBert_K_Fem_GP_1	0.33	0.31	0.34
CV_young_Fem_GP_1	0.1	0.1	0.1
CV_old_Fem_GP_1	0	0	0
L_at_Amin_Mal_GP_1	40.9	39.66	42.03
L_at_Amax_Mal_GP_1	80.7	79.2	82.1
VonBert_K_Mal_GP_1	0.3	0.2	0.3
CV_young_Mal_GP_1	0.13	0.12	0.14
CV_old_Mal_GP_1	0.1	0.1	0.1
SR_LN(R0)	5.9	5.6	6.3
Size_inflection_Commercial(1)	50.41	49.21	51.95
Size_95SizeSpline_GradLo_Recreational(2)	0	0	0.1
SizeSpline_GradHi_Recreational(2)	-0.04	-0.1	0
SizeSpline_Val_1_Recreational(2)	-0.6	-0.9	-0.3
SizeSpline_Val_3_Recreational(2)	-0.6	-0.9	-0.3

Table 3.1: Summary of parameter estimates for Queensland east coast saddletail snapper from the ensembled scenarios—MCMC Median is median parameter value from robust MCMC scenarios, MCMC 2.5% and 97.5% indicates 95% credible interval

3.2.2 Model fits

Plots of model fit to standardised catch rates, mean age over time, length-composition, age-composition and discards by fleet are shown from Figure 3.14 to Figure 3.25.

By applying broad standard errors (SE=0.2) to the CPUE index, the model fits to the index were deliberately down-weighted. This was done to allow the model greater flexibility to fit closely to trends in the age-at-length data, and avoid fitting closely to a CPUE index that may be impacted by hyperstability. As a result, model fits to the CPUE index were intentionally less precise (Figures 3.14 and 3.15).

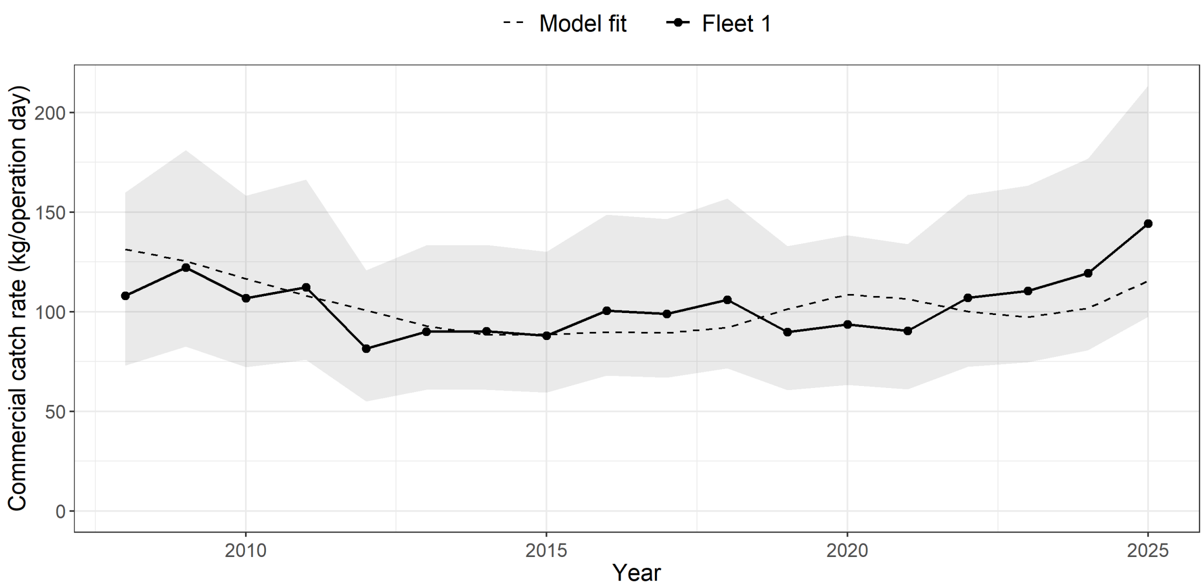

Figure 3.14: Model fits (dotted line) to standardised commercial catch rates for Queensland east coast saddletail snapper between the years of 2008 and 2025

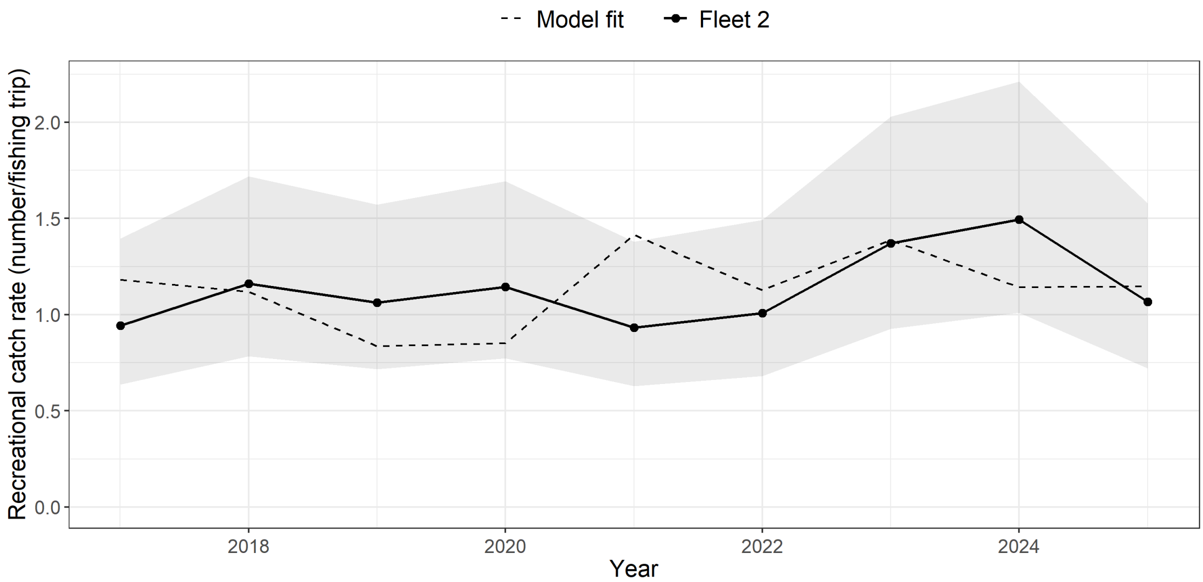

Figure 3.15: Model fits (dotted line) to standardised recreational catch rates for Queensland east coast saddletail snapper between the years of 2017 and 2025

Overall, the mean age plots (Figure 3.16) show reasonable fit to the age-at-length data. There is no clear trend of long term over- or under-prediction.

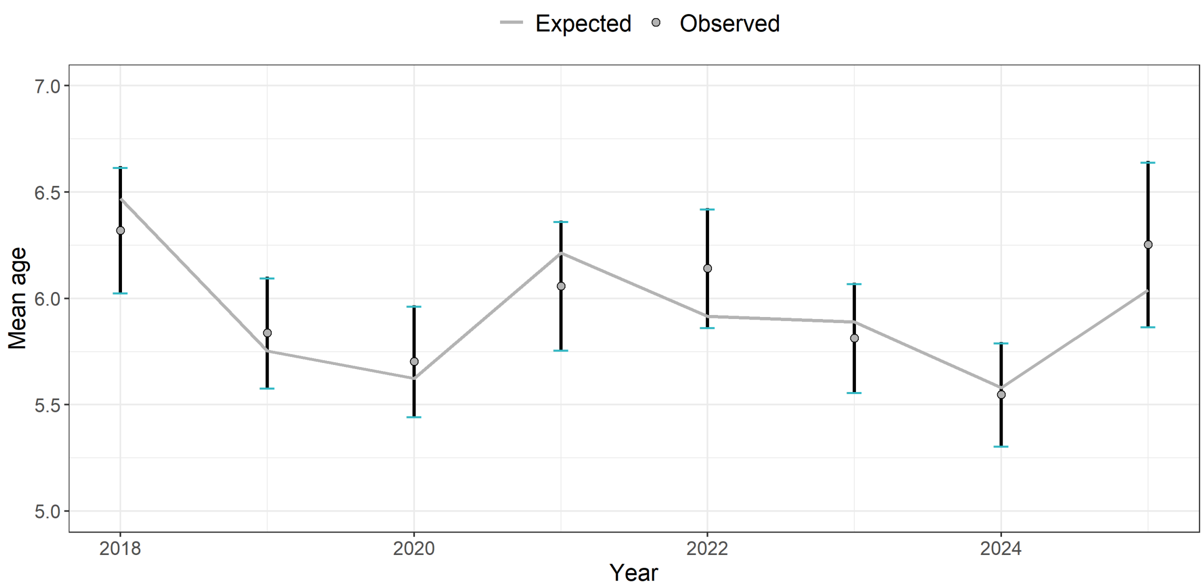

Figure 3.16: Model fits to mean age from conditional data (aggregated across length bins) with 95% confidence intervals based on current sample sizes: blue intervals (with capped ends) show result of further adjusting sample sizes based on suggested multiplier (with 95% interval) for conditional age-at-length data

Model fits to aggregated and annual length composition data by fleet are given in the plots below (Figures 3.17 to 3.22). In general, fits to length composition data were reasonable, especially aggregated across all years. In more recent years, the peak of the commercial length composition fit is misaligned with the peak of the input data. Attempts were made to improve this fit by adjusting data weighting and introducing time-varying commercial selectivity, however any practical implementations of this saw minimal change in the model fits or final biomass estimates (see Appendix B.5).

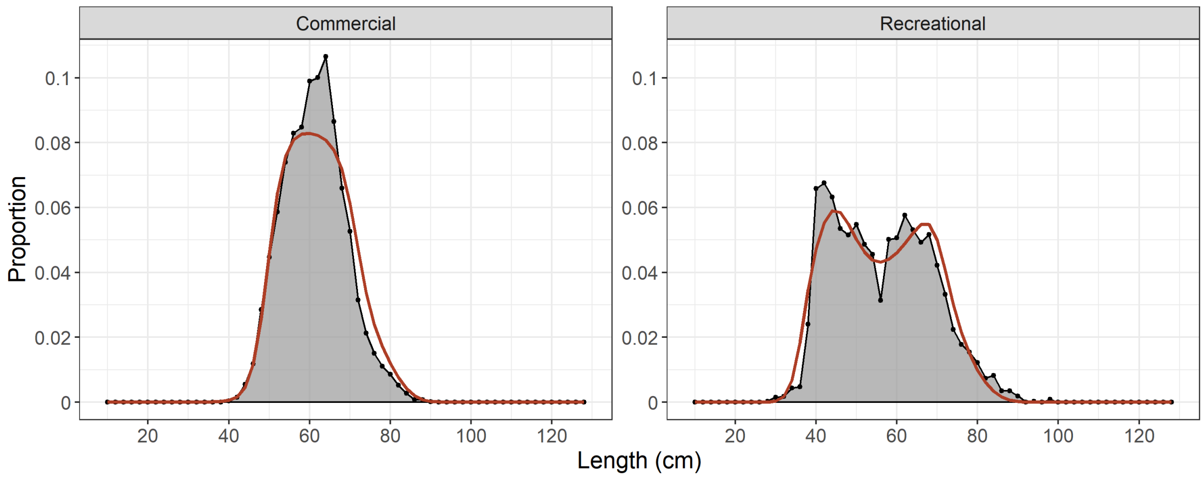

Figure 3.17: Length compositions of Queensland east coast saddletail snapper aggregated across time by fleet—scenarios 1–6, weighted using Method 1 described in Section 2.6.2

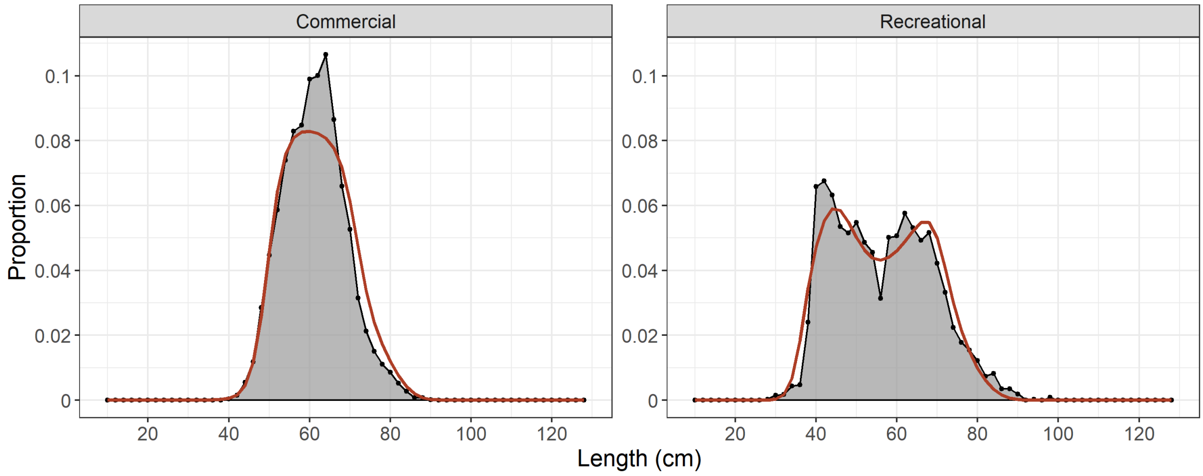

Figure 3.18: Length compositions of Queensland east coast saddletail snapper aggregated across time by fleet—scenarios 7–12, weighted using Method 2 described in Section 2.6.2

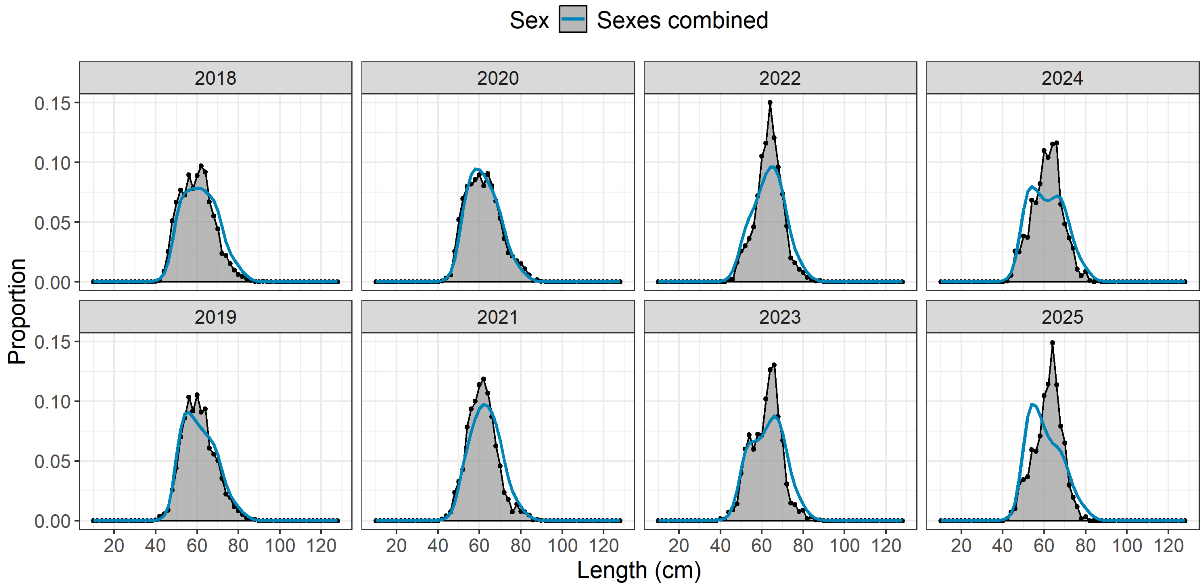

Figure 3.19: Model fits to commercial length structures, based on maximum likelihood estimation—the grey area and black line represent data inputs and the blue line represents the model fits—scenarios 1–6, weighted using Method 1 described in Section 2.6.2

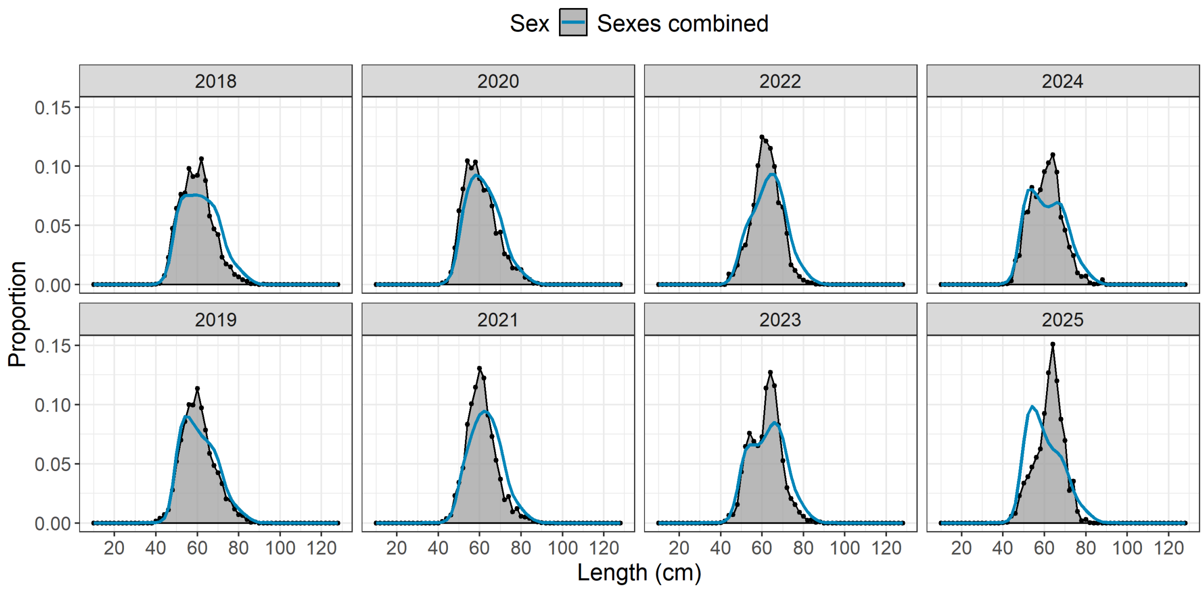

Figure 3.20: Model fits to commercial length structures, based on maximum likelihood estimation—the grey area and black line represent data inputs and the blue line represents the model fits—scenarios 7–12 weighted using Method 2 described in Section 2.6.2

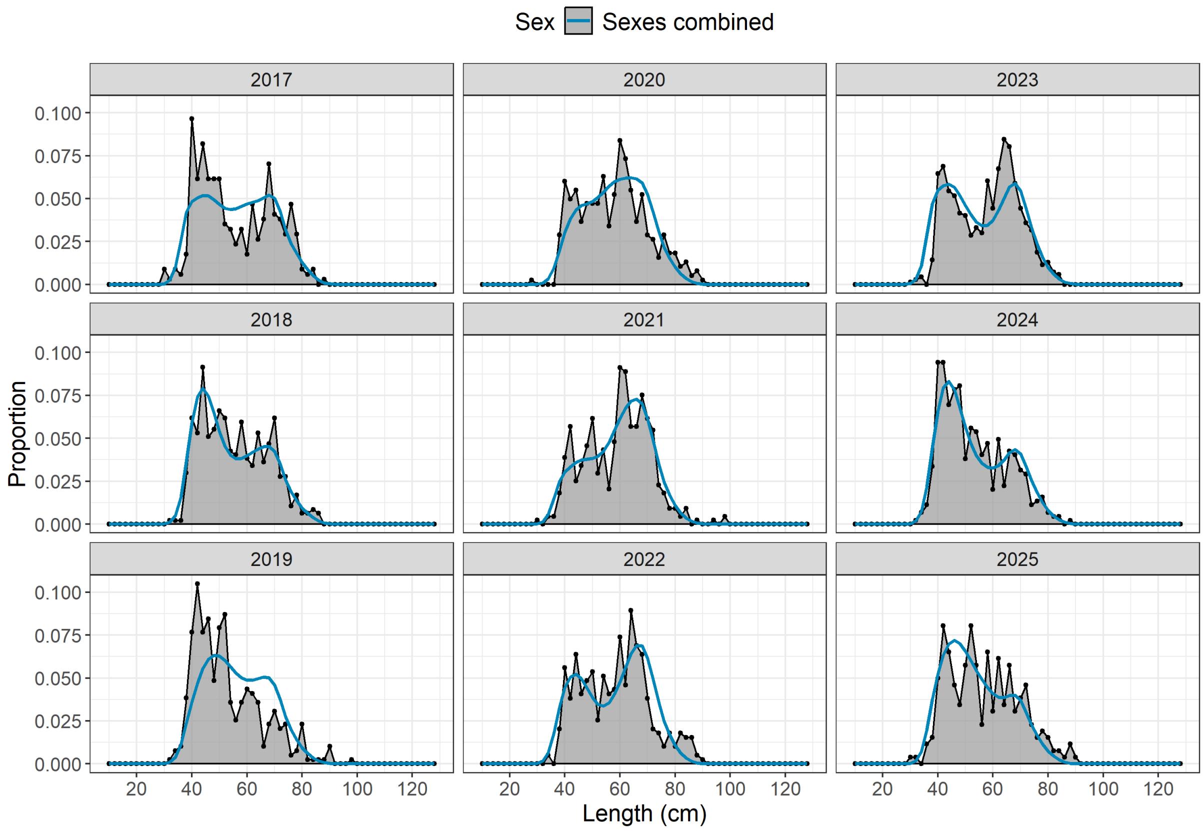

Figure 3.21: Model fits to recreational length structures, based on maximum likelihood estimation—the grey area and black line represent data inputs and the blue line represents the model fits—scenarios 1–6, weighted using Method 1 described in Section 2.6.2

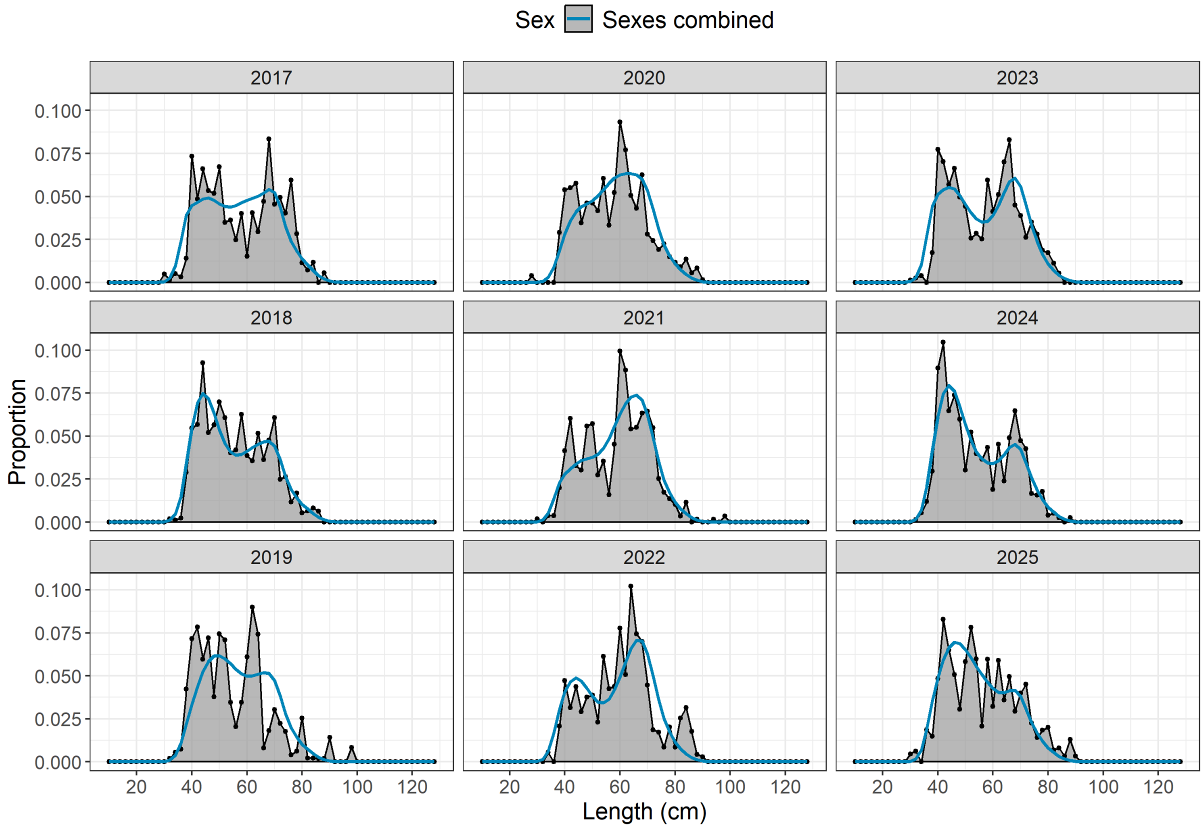

Figure 3.22: Model fits to recreational length structures, based on maximum likelihood estimation—the grey area and black line represent data inputs and the blue line represents the model fits—scenarios 7–12, weighted using Method 2 described in Section 2.6.2

Age data were input to the model as conditional age-at-length data, rather than age-only composition data derived outside the model. For model verification, plots showing the level of matching of model-predicted age compositions to observed age compositions were included for visualisation purposes (i.e. using the ‘ghost fleet’ approach in Stock Synthesis). Note that age-only composition data were not used as part of the model fitting process (Figures 3.23 to 3.24).

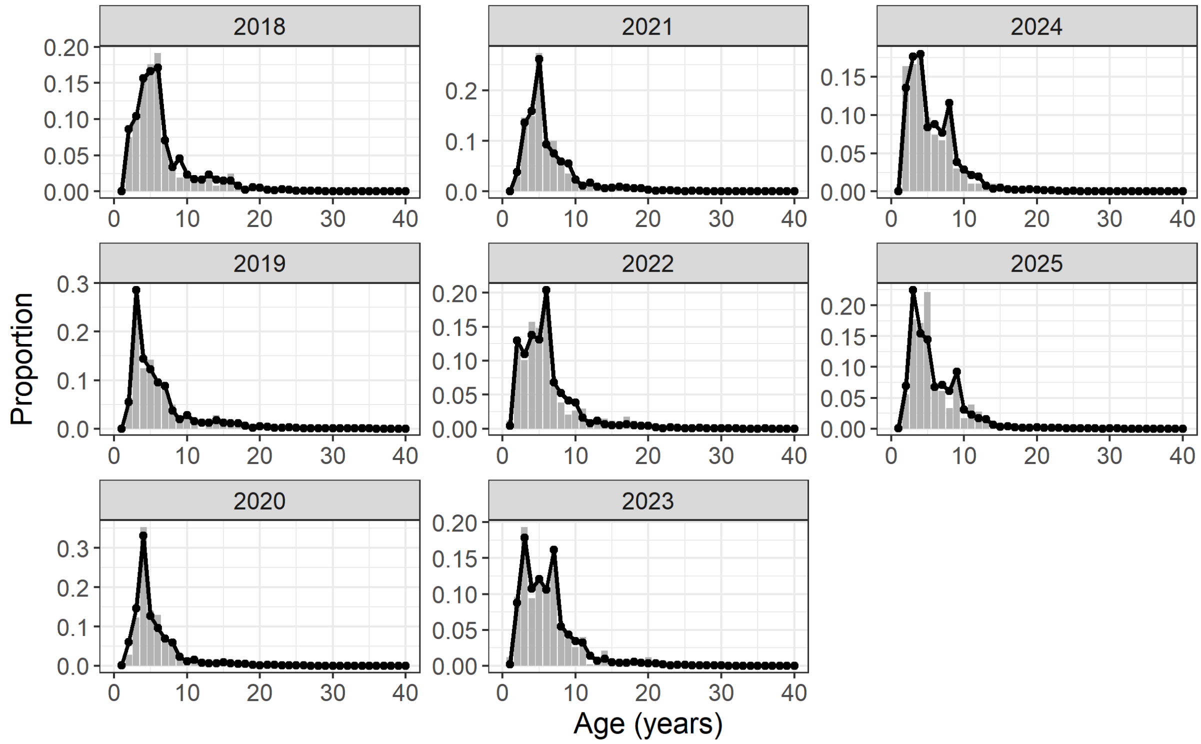

Figure 3.23: Matching of modelled age compositions to observed annual age compositions of female saddletail snapper (scenario 1)

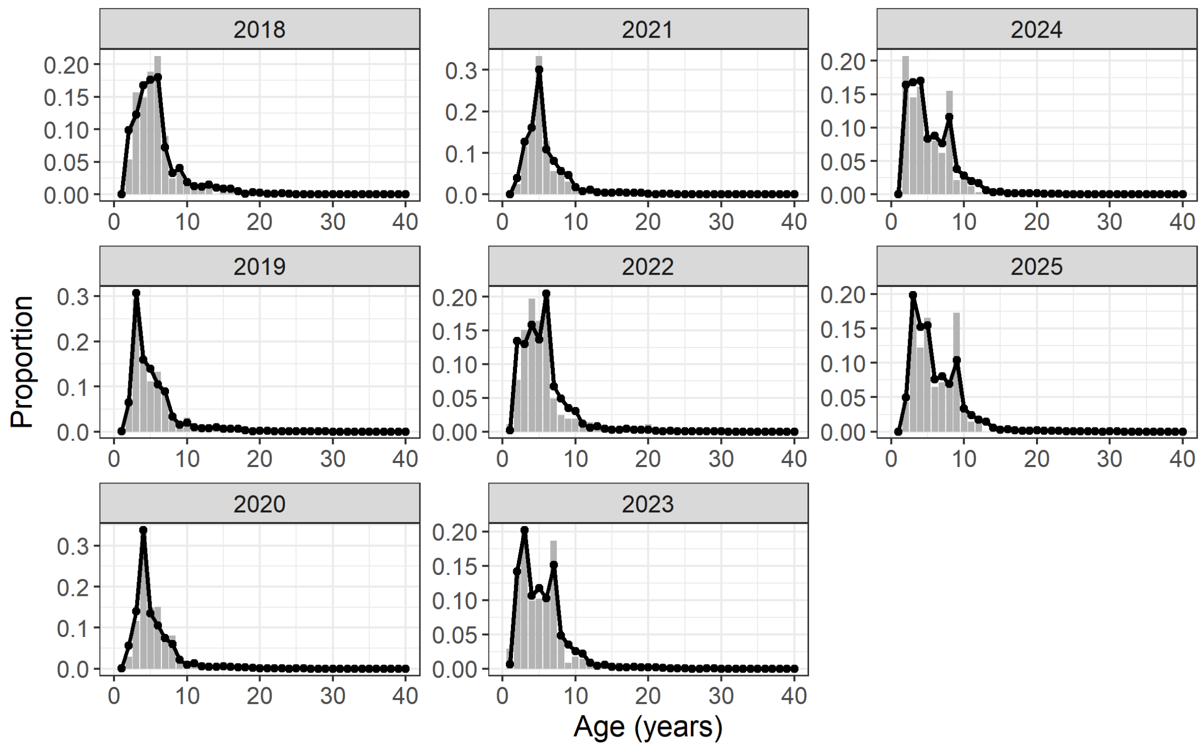

Figure 3.24: Matching of modelled age compositions to observed annual age compositions of male saddletail snapper (scenario 1)

Overall, there were reasonable visual fits to input discard information (Figure 3.25), considering the uncertainty in length distributions of released fish over time. The model underestimated the discards in 2018, 2019 and 2020, as it did in the previous assessment (Campbell et al. 2021).

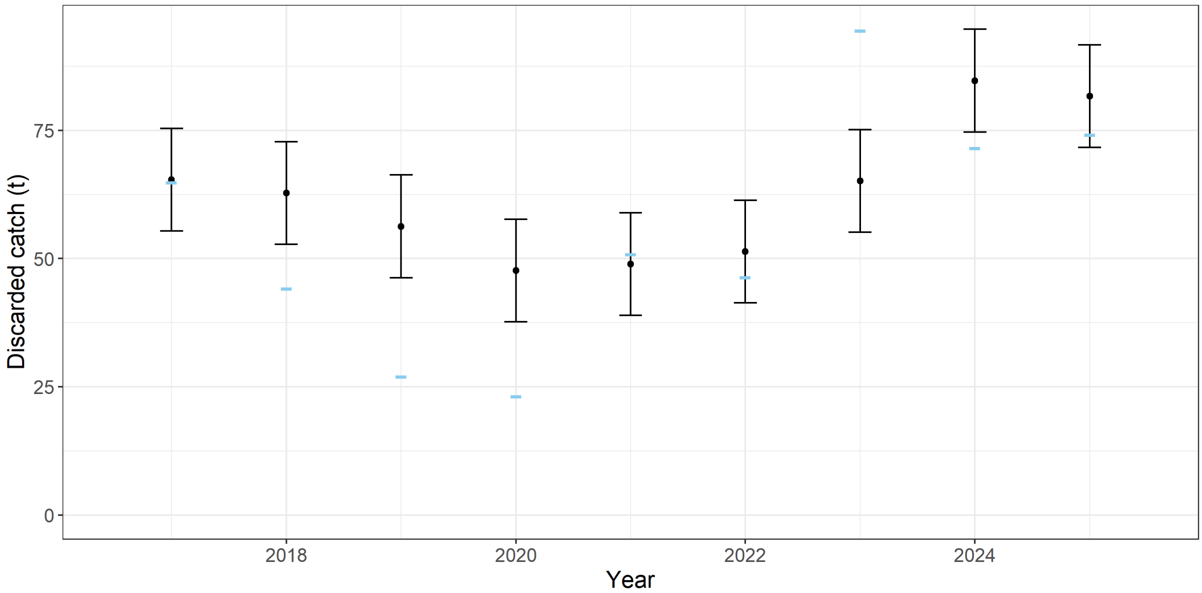

Figure 3.25: Fits to annual discards of Queensland east coast saddletail snapper (scenario 1)—only the recreational fleet has discards

Figures E.5, E.6 and E.7 (Section E.3) show diagnostics that indicate good model convergence for all scenarios.

3.2.3 Selectivity

Parameters for length-based selectivity to fishing were estimated within the model (Table 3.1). The resulting selectivity function (Figure 3.26) represents the relative proportion of saddletail snapper of a given length that can be caught by the fishing gear deployed by a fleet (ranging from 0% to 100%).

The model was given flexibility to estimate the recreational selectivity curve as a three-node cubic spline. This resulted in a bimodal shaped selectivity curve, which reflected the bimodal shape in the length distribution data inputs (compare Figure 3.26 and Figure 3.17). The author team noted the area under the recreational selectivity curve below MLS, and explored various models to assess its impact (see Section B.6).

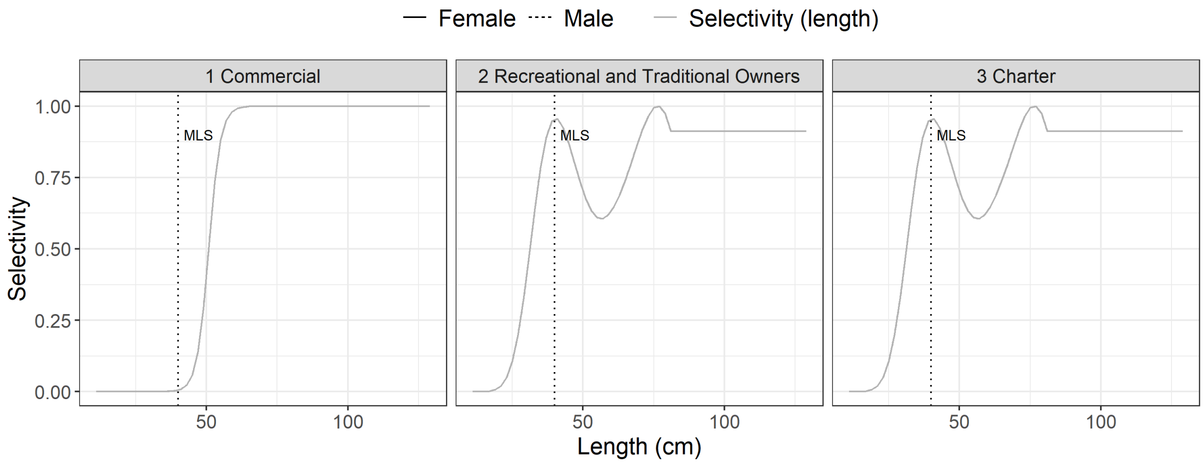

Figure 3.26: Model estimated length-based selectivity in 2026 for the representative MLE model for Queensland east coast saddletail snapper—the dashed line shows the current minimum legal size

3.2.4 Growth curves

Parameters for the von Bertalanffy growth curve, including standard deviations about the mean length for both old and young fish, were estimated within the model (Table 3.1).

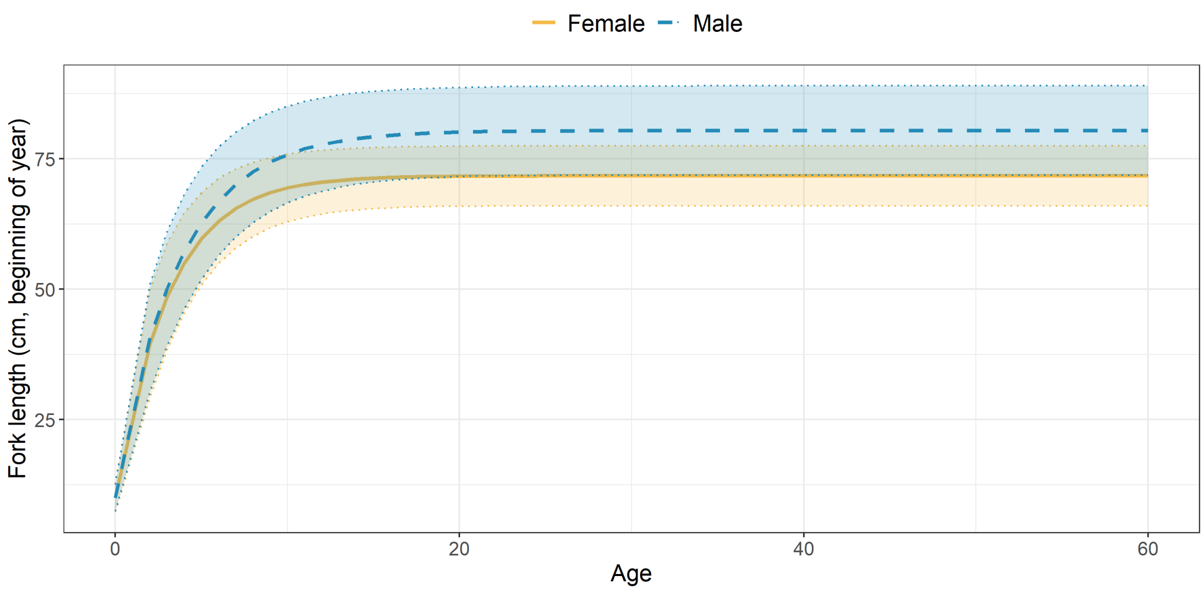

Figure 3.27: Estimated growth of Queensland east coast saddletail snapper (95% confidence intervals) for the representative MLE model (scenario 1)

3.2.5 Biomass

The time series of estimated spawning biomass ratio from all the scenarios listed in Table 2.3, relative to the unfished state, are shown in Figures 3.28 (Markov chain Monte Carlo results) and 3.30 (maximum likelihood estimates and MCMC results) and Table 3.2.

The median (or most probable) estimate of 2026 spawning biomass ratio was 28% (Figure 3.29). The coloured, shaded sections of the probability distribution plot (Figure 3.29) indicate the respective probabilities that the 2026 spawning biomass ratio outcome falls below, between, or above reference points (20%, 40% and 60% biomass). Figure 3.29 shows that 90% of the 250,000 MCMC model runs per scenario produced a result between 20 and 40% of unfished spawning biomass, meaning these models predict there is a 90% chance (high confidence) that the biomass falls between 20 and 40% of unfished spawning biomass.

Figure 3.29 differs from the way final outputs were presented in Campbell et al. (2021) and instead follows the presentation style of more recent Fisheries Queensland stock assessments (e.g. Campbell et al. 2024; Sumpter et al. 2025). Campbell et al. (2021) presented a 'base case' with an uncertainty range. A 'base case' means that the project team was asked for a preferred (or most likely) scenario from the larger suite of scenarios, and headline biomass results were reported from this chosen scenario alone. Instead, an ensemble approach (used here) directly represents all uncertainty considered in the final suite of scenarios, by bringing them into a single output distribution. For this assessment, all of the model scenarios included in the final ensemble were assumed equally likely.

While the 'base case' approach used in Campbell et al. (2021) presents biomass results from a 'most likely' scenario as chosen by project team members, the ensemble approach can show the most probable biomass outcome from the suite of scenarios assumed to be equally likely by the project team. Figure 3.29 shows this, displaying the relative probability of different final biomass ratio outcomes. Rather than the 'most likely' biomass outcome that would be given by a 'base case' approach, the median biomass is given as a measure of central tendency for 2026 spawning biomass outcomes across all plausible scenarios. The median is preferred as a central tendency measure (over the mean or mode, for example) as it is less prone to the influence of outliers or extreme values, making it a more robust measure for skewed distributions, such as the probability distribution of 2026 biomass.

A tabular summary of stock status indicators relating to biomass outputs is given in Table 3.2. A full time series of annual fishing mortality can be found in Figure E.11.

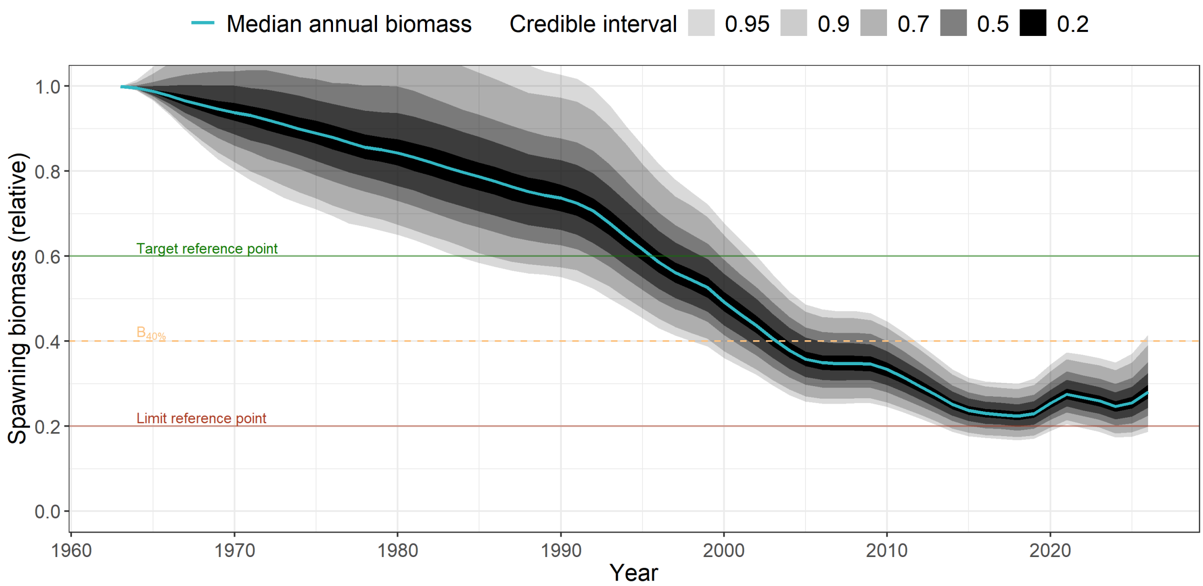

Figure 3.28: Predicted spawning biomass trajectory relative to unfished for Queensland east coast saddletail snapper, from MCMC ensemble scenarios

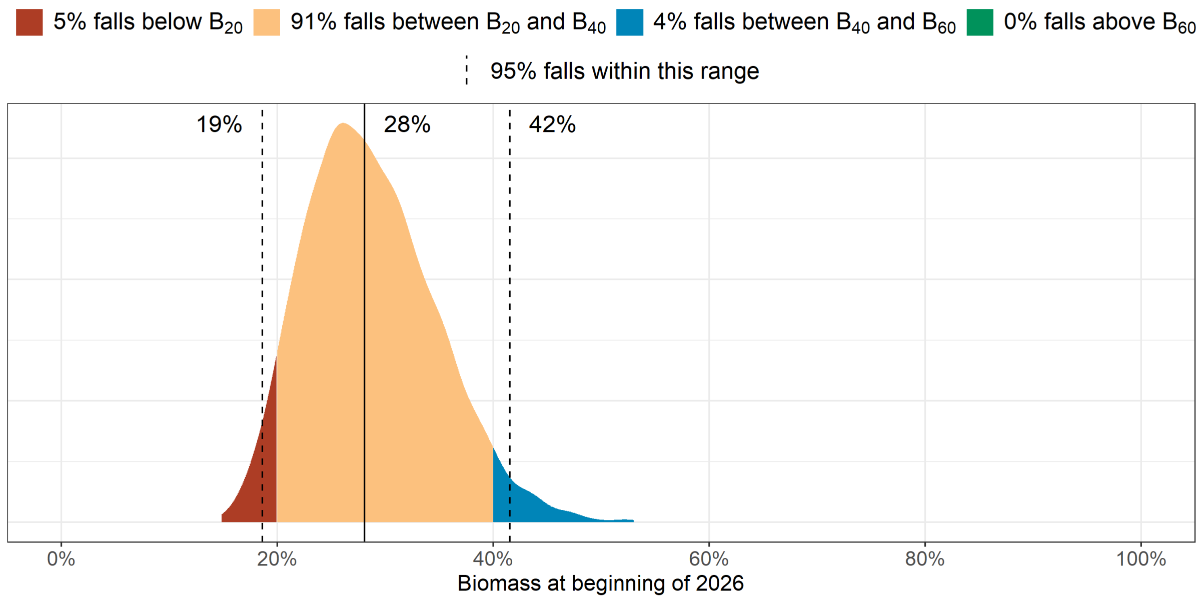

Figure 3.29: Probability distribution of the biomass ratio for Queensland east coast saddletail snapper at the beginning of 2026 across the full ensemble of scenarios with the credible interval and probability of biomass falling into the four categories indicated

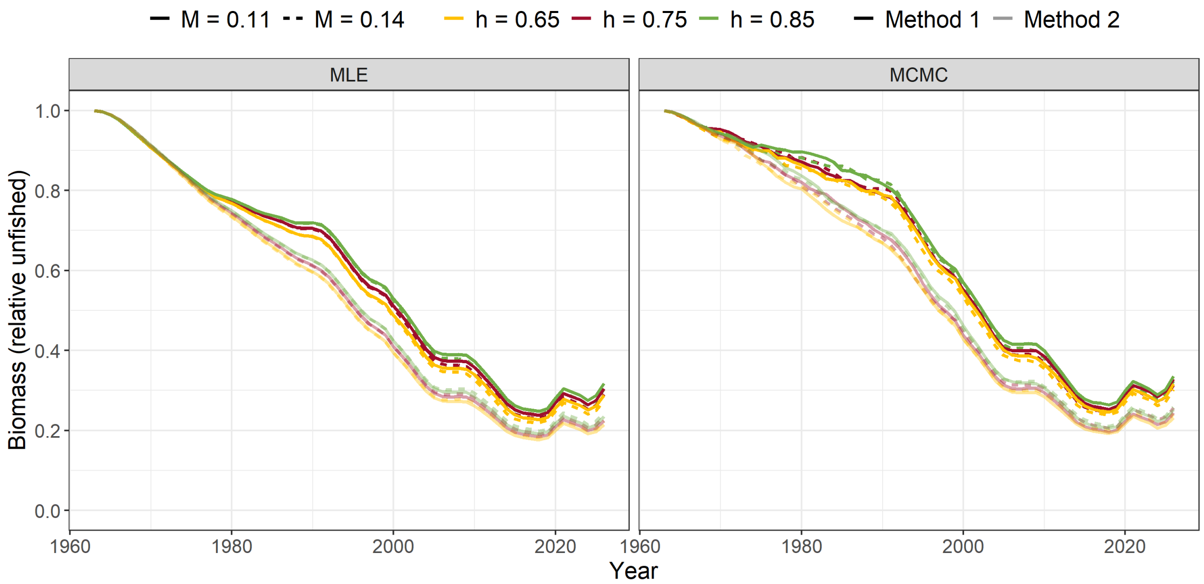

Figure 3.30: Estimated spawning biomass trajectory relative to unfished levels for Queensland east coast saddletail snapper from 1961 to 2026 for all scenarios. The left panel is the optimised maximum likelihood estimate and the right panel displays the “MCMC median” (i.e. median annual biomass value across all iterations within a scenario). Line type (solid or dashed) represents natural mortality parameter setting, colour represents steepness parameter setting, and the opacity represents the method used to weight length data.

Table 3.2: Stock status indicators for Queensland east coast saddletail snapper

Indicator	Value
Biomass ratio (relative to unfished)	
Median	28%
Range (95% credible interval)	19–42%
Probability below 20%	5%
Probability between 20% and 40%	91%
Probability between 40% and 60%	4%
Probability above 60%	0%
Fishing pressure ratio (relative to F_{60})	
Median	3.85
Range (95% credible interval)	2.48–5.84

In the following table, biomass ratio outcomes at the beginning of 2026 are given by scenario, for both maximum likelihood estimation (MLE) and Markov chain Monte Carlo (MCMC) model runs (Table 3.3). For MCMC runs, the biomass outputs shown here are relevant to the scenario providing the median 2026 biomass from each run's MCMC distribution. For a list of the twelve scenarios and their given parameter settings, refer to Table 2.3.

Table 3.3: Summary of model outcomes for all scenarios.$B_{2026}\%$ is the most likely biomass in 2026 relative to unfished in 1962 with the 95% confidence interval for maximum likelihoods estimations and 95% credible interval for MCMC estimations.

Scenario	MLE			MCMC		
	$B_{2026}\%$	$B_{2026,lower}\%$	$B_{2026,upper}\%$	$B_{2026}\%$	$B_{2026,lower}\%$	$B_{2026,upper}\%$
1	0.29	0.19	0.39	0.31	0.23	0.42
2	0.28	0.18	0.37	0.30	0.22	0.42
3	0.30	0.21	0.40	0.32	0.25	0.43
4	0.30	0.20	0.40	0.33	0.23	0.45
5	0.29	0.19	0.39	0.32	0.22	0.44
6	0.32	0.22	0.42	0.34	0.25	0.45
7	0.23	0.16	0.31	0.26	0.19	0.35
8	0.22	0.15	0.30	0.24	0.18	0.33
9	0.24	0.17	0.32	0.26	0.18	0.35
10	0.23	0.15	0.30	0.24	0.17	0.34
11	0.22	0.14	0.29	0.23	0.17	0.32
12	0.23	0.16	0.31	0.25	0.18	0.35

4 Discussion

4.1 Stock status

At the beginning of 2026, the stock was estimated to be between 19% and 42% of unfished spawning biomass (95% credible interval over the MCMC ensemble), with a median (most probable) estimate of 28%. It is unlikely that the biomass was below B_{lim} (i.e. 20% of unfished) at this time, with only a 5% probability of this outcome. The biomass trajectory indicates a decline from an assumed unfished state in 1962 to potentially as low as 12% in 2018.

4.2 Model influences

4.2.1 Natural mortality and steepness

When comparing the biomass trajectories across all twelve scenarios in Figure 3.30, the impact of natural mortality and steepness seem negligible. However, insight is gained from close inspection of the additional modelling reported in Appendix B.7, as follows.

The two fixed values of natural mortality used in the ensemble model (0.11 and 0.14 yr^{-1}) come from empirical estimations derived from Hamel and Cope 2022 ($5.4/A_{max}$), using the first and second oldest fish available in the data set. There is a sizeable gap between the oldest fish (aged 49 years) and the second oldest fish (aged 39 years) observed in the data. Instead, the signal that the age data are providing to the model is that of a heavily fished stock with few relatively older fish, which is in conflict with both the fixed value of natural mortality and the index of abundance.

Figures B.20 and B.21 illustrate the interaction between steepness and natural mortality. The relative impact of steepness is negligible if natural mortality increases past a biological plausible range. Within the plausible range of natural mortality and steepness values, the models responded intuitively to variation in steepness, in which a higher value of steepness produced a more optimistic biomass estimate, however overall variation from this effect was small (Figure 3.30). This predictable behaviour is likely a result of both steepness and natural mortality parameters being fixed, though this was necessary as likelihood profiles indicated insufficient information for estimation of these parameters. As Maunder et al. (2020) highlights, the success of estimating natural mortality within a stock assessment model depends on the amount and type of data available combined with model assumptions (including those around modelling natural mortality itself). Similar limitations apply to estimating steepness within a stock assessment model.

4.2.2 Length data weighting

The method used to weight individual fish samples in the length compositions (described in Section 2.6.2) was one of the most influential axes of uncertainty that was explored, in terms of the resulting biomass estimates (Figure 3.30). There was no obvious difference in the annual length composition inputs or resulting model fits from either weighting method, as seen in Figures 3.17 to 3.22.

Table D.1 shows the length composition effective sample sizes that were input to the model for both methods. This table highlights two key differences between the methods that are likely influencing the model results:

- The sum of the sample sizes for each method is different (Stage 2, Section 2.6.2). This, in combination with Francis weighting (Stage 3, Section 2.6.2), affects the overall contribution of the length compositions to the optimised likelihood, compared to other data inputs (e.g. conditional age-at-length and CPUE indices).
- Weighting Method 1 emphasised commercial length compositions over recreational, and Method 2 did the opposite. Given the commercial fleet fishes primarily offshore grounds, while the recreational fleet fishes both inshore and offshore, the distribution of their respective length compositions differ notably (Figure 3.8), with the recreational length compositions containing more fish of small size classes. As a result, placing more emphasis on one fleet versus the other will have a direct impact on the application of fishing mortality by size class, and the model's interpretation of patterns in recruitment.

Maunder et al. (2020) highlighted the importance of weighting length composition data so that it was fit-for-purpose for its use in a stock assessment model. Sumpter et al. (2025) explored how this work might be applied to Fisheries Queensland monitoring data. This current assessment extends that work, by explicitly defining two weighting methods (Section 2.6.2); one that focuses on the removals from the population by fishing, and one that focuses on the population distribution. By incorporating both methods, a more comprehensive range of uncertainty is explored. We recommend this work is continued to ensure any sampling bias (including spatio-temporal distribution of data) is correctly accounted for, and that length composition inputs are correctly informing the appropriate mechanisms within a stock assessment model.

4.2.3 Catch rates

Saddletail snapper do not recruit to the fishery until they reach about 3 years of age. This means that the most recent few years of age and length data are not able to effectively inform the model about recent recruitment levels, as pre-recruits are not part of the sample. Instead, the model has to rely more heavily on the estimated stock-recruitment relationship and CPUE indices. This is one reason why consistent biological monitoring, over a long time span, is crucial for informing stock dynamics.

For this reason, the most recent few years of CPUE are having a greater influence upon recent changes to biomass and recruitment deviations. The commercial catch rate data saw an increase in abundance since 2022, and this upwards trend has been reflected in the biomass trajectory (Figures 3.14 and 3.28). Both the commercial and recreational catch rate time series were adjusted, prior to input into the model, to account for assumed annual changes to fishing efficiency (Section 2.4.3). The calculated effect of fishing efficiency compounds over time, so the impact of any assumptions relating to fishing efficiency is most evident when comparing later years to earlier years. Exploration in Appendix B.3 demonstrated that this had only a small effect on final biomass estimates, due to the relatively limited number of years in the time series.

Appendix B.7 explored the influence of catch rates on the model in more detail, demonstrating that the influence of CPUE inputs was more pronounced with smaller values of natural mortality. Further, at these levels, when the annual catch rate input data were removed from the model, the stock predicted a sharp decline in biomass to the final year. This occurs because, the catch rates provide essential information to 'anchor' recent abundance, preventing the model from assuming unrealistically low recruitment for the most recent years.

4.2.4 Depredation

The impact of depredation on saddletail snapper population dynamics has been both a topic of public interest and a major area of exploration for this assessment. Multiple sources of information were scrutinised by the project team, to ensure the best available estimates of depredation rates were applied. Further research needs are described in Section 4.5.2.

Several scenarios were run to explore the impact of extreme estimates of depredation rates, ranging from 0% (i.e. no fish depredated) to 100% (i.e. depredation equal to the amount of fish being caught by humans). This was explored in Appendix B.1. Across these range of scenarios, final biomass estimates in 2026 only varied by approximately 6%. This is because the model compensated for the change in fishing mortality—with unchanged catch rates—by adjusting the *ln(R0)* parameter (i.e. the number of recruits in the unfished population). In other words, since all of these models had the same, relatively stable catch rate data inputs, the model reconciled a large volume of catch by rescaling the size of the entire population. Given the scale of the unfished population is not something we can observe or easily estimate, it is difficult to discern whether the model's rescaling of *ln(R0)* is a realistic adjustment or solely an artefact of model optimisation. Further investigation into the impact of depredation on catch rates may provide the model with additional flexibility, as discussed in Section 4.5.2.1.

4.3 Performance of the population model

Parameter estimation occurred using two methodologies across twelve scenarios: maximum likelihood estimation (MLE) and Markov chain Monte Carlo (MCMC) estimation. For MCMC estimation, each scenario was run with 250,000 iterations, using the standard Stock Synthesis algorithm. Final gradient diagnostics for all twelve MLE scenarios were satisfactory, indicating suitable convergence. Since final ensemble results were derived from MCMC outputs, model performance was also evaluated in the context of MCMC diagnostics. The trace plots, posterior density plots and R-hat plots in Appendix E.3 indicate convergence was reasonable.

Final biomass estimates produced by the MLE and MCMC scenarios were comparable. Generally, the 2026 median biomass estimates produced by the MCMC estimate were approximately 2–4% higher than the MLE equivalent for each scenario. This offset is also reflected in the 95% confidence/credible intervals estimated by each method.

The visual fits to input data (Section 3.2.2) were reasonable, with fits to the age and length data notably improved compared with the previous assessment (Campbell et al. 2021). The model fit closely to the mean age signal from conditional age-at-length data. The fits to the aggregated length compositions were good, especially with the spline shape of recreational selectivity allowing for a bimodal distribution as seen in the recreational length compositions. There is room for improvement in the annual fits to the later years of commercial length composition data. We attempted to improve these fits by implementing time-varying selectivity (Appendix B.5), as well as cubic-spline shaped commercial selectivity curves, however we did not see much improvement and increased the risk of overfitting the model by introducing more parameters. It is worth noting that the model's tendency to underpredict peak selectivity in these years (particularly 2024 and 2025) may be due to conflict in the data—the model may expect an influx of small to average-sized individuals, as abundance indices (both sectors) increase and mean age decreases over the same time frame. Future assessments, with additional years of length data, may have more success in fitting to these years of length data. The model underestimated the discards in 2018, 2019 and 2020, as it did in the previous assessment (Campbell et al. 2021), which may warrant further investigation, however it was deemed acceptable in this assessment given the uncertainty associated with the observed discard data

It is crucial to appropriately define fleet structure, as it determines how fishing effort and catch are partitioned across time and space and can avoid over-generalisation of selectivity patterns, influencing model fit to data and the accuracy of stock dynamics estimates. Misspecification of fleet structure (i.e. aggregating fleets in the presence of spatiotemporal structure in the data, or erroneously splitting fleets) can lead to biased estimates of fishing mortality and stock status. Such examples of structure in the input data include seasonal and spatial variations in the availability of fish of certain size classes or ages (see Sumpter et al. (2025)), which can be driven by natural phenomena such as movement or migratory behaviour. Generally speaking, analysis using GAMs (see Section 2.7) found little evidence of spatial or seasonal trends in length and age data for this stock. However, results from the length-response model (Figure D.10) did indicate a potential spatial difference in the lengths of fish in latitudes 21–23° S, which corresponds to the Swains Region.

Given the Swains Reefs are further offshore than most other saddletail grounds on Queensland's east coast, vessels that fish these grounds are typically larger, stay for longer trips and have greater capacity for holding large catches. These patterns may be a result of the differences in fishing practices/patterns (i.e. selectivity) in the Swains compared to other regions, which could potentially be captured through splitting the commercial fleet spatially into two fleets (i.e. the 'areas-as-fleets' approach, Punt (2019)). However, adequate data is needed for the Swains to stand alone as its own fleet, and the stock assessment model is relatively data-poor. Despite this, we explored additional models in which one or more fleets were split into regions, employing the 'fleets-as-areas' technique. In one such model, the commercial fleet was split into two regions (the Swains region, and everywhere else). This structural change to the model did not have a significant impact on the final biomass, however it resulted in poor fits to model inputs (particularly length data), and the estimated selectivity curves for both commercial fleets were nearly identical. This indicated that the more complex fleet structure (with the available data) was unnecessary and detrimental to model performance. This served as reinforcing evidence to retain the simpler population model structure of just three fleets (commercial, recreational & Traditional Owners, and charter), with no additional spatial or temporal fleet splits.

Additional models were run to explore alternative implementation of weighted length compositions. Attempts were made to implement a dual-weighting approach described in Maunder et al. (2020), in which the model was provided two versions of length composition data—one weighted by CPUE to represent the index of abundance, and one weighted by catch to represent the fish removed from the stock. This was implemented through the use of an extra (survey) fleet. This approach was experimental, especially given the limited availability of data, and resulted in poor fits to model inputs. This approach could be explored further in future assessments.

4.4 Unmodelled influences

4.4.1 Spatial structure

As part of this assessment, we included an investigation into spatial structure of the saddletail snapper population and fishery (Section 2.7). Additional analyses, including GAM modelling of length and age data, revealed no significant spatial or temporal patterns that would justify a more complex model structure than what has been presented in this report. More information on this is provided in Section 2.8.2. It is possible that spatial or temporal patterns exist in this stock but are not identifiable with the current sample sizes of biological data, or the available spatial metadata associated with each sampled fish.

Additional years of biological data may aid in confirming or dispelling the general spatio-temporal homogeneity in length and age data seen here. Spatial stock assessment models (even those that do not model movement, i.e. 'fleets-as-areas') require a large quantity of appropriately informative data, which was not available for this body of work.

As such, the model assumed a single homogenous population across the east coast of Queensland. Despite this, regional variation was incorporated into the model inputs and dynamics via numerous mechanisms. Regional variation in catch and effort was accounted for through standardised CPUE inputs, which included regional covariates in the standardisation models. Length composition inputs accounted for regional variation through careful data weighting. Recreational selectivity estimates used a cubic spline to account for the bimodal distribution of recreational length compositions, which likely arose from spatial differences in fishing activity inshore versus offshore. This bimodality might also be a result of a potential ontogenetic shift where saddletail snapper move offshore as they age (Davis et al. 2025).

While the assessment was not data-rich, a fleets-as-area model was trialled by splitting the commercial fleet into two: the Swains region, and elsewhere. However, this exploratory model had poor diagnostics and resulted in negligible differences in selectivity curves between the two fleets, indicating that this approach was unnecessarily complex at this time.

4.4.2 Fishery changes around 2004

Attempts have been made to capture the impact of changes in the fishery in the period leading up to 2004. After 2004, fishery management arrangements changed substantially. These changes included the introduction of Total Allowable Commercial Catch, changes in size limits and recreational bag limits, seasonal closures, and the rezoning of the Great Barrier Reef Marine Park increases in the area of no-take zones.

The flow-on impacts of these changes on fisheries and fish populations within the Marine Park have not been well quantified for deepwater and pelagic species. In particular, there has been limited assessment of how rezoning affected fishery operations, the availability of fishing grounds, and fishers' responses to changes.

For deepwater and pelagic species, there has been no explicit modelling of spatial variation in population dynamics that may have arisen as a result of rezoning, nor of the potential resilience benefits associated with spatial protection. However, available evidence suggests that rezoning has had a positive impact on populations of reef-associated species, such as common coral trout, within no-take areas.

This was explored in the previous assessment (Campbell et al. 2021) and is a topic common to most stock assessments for species on the east coast of Queensland. Most recently, it was considered in the assessment of Spanish mackerel, following an independent review of the previous Spanish mackerel stock assessment (Sumpter et al. 2025; Hoyle and Dunn 2023; Tanimoto et al. 2021).

Members of this project team noted that it was plausible that the investment warning ahead of individual transferable quota (ITQ) allocation in 2004 could have led to over-reporting during the 1997–2001 period when catch history counted towards quota allocation. If over-reporting had occurred, this would lead to artificially inflated annual catch totals during this period, followed by a sudden decline following 2001 at the end of years counting to allocation of quota, then another period from 2001–2004 as quota was implemented. This issue was appraised by the project team, which led to a clearer indication that over-reporting in this period may have occurred. These discussions were actioned in two ways: adjustments were made to the commercial catch estimates (see Section 2.2.1.2 and Appendix B.2), and catch rate data before 2004 were omitted.

The catch rate time series from 1989 to 2025 shows a sharp drop of ∼43% around 2002–2004, when these management changes were implemented (Figure B.22). Given CPUE indices are meant to indicate temporal changes in fish abundance, the model would interpret this drop as a marked reduction in abundance. Following project team concerns about catch reporting in this period, it is unclear whether this drop is a result of changes in abundance, changes in fisher behaviour as they responded to management changes or emerging markets, availability of fishing ground, or a misleading signal in the data due to incorrect reporting. It is possible that all of these factors (and more) may be responsible, however data is not available to inform this with any certainty. If the drop was a result of anything other than fish abundance, that signal should be removed from the catch rate time series, since its purpose is to act as an index of abundance in the stock assessment model.

As CPUE indices are to be interpreted by the model to reflect changes in abundance, it is paramount that CPUE values are directly comparabile through time. Ultimately, the project team deemed that the structural changes to the fishery in 2004 would render catch rates incomparable prior to, and post 2004. As a result, the standardised catch rate model was run over data from 2008 to 2025, to achieve the most temporally consistent index of abundance.

Alternative approaches were considered:

- The commercial catch rate time series could have been split into two separate data inputs (i.e. pre- and post-management changes). This would have enabled the stock assessment model to retain trends from both eras, and estimate a catchability coefficient for each. This is analogous to allowing the model to vertically offset the catch rate time series of one era, relative to the other. This was the approach used in many scenarios in the previous assessment, as well as in the stock assessment of crimson snapper (Campbell et al. 2021; Fox et al. 2021). Since then, best practice literature advised against this approach (Hoyle et al. 2024).
- A manual adjustment to catchability for both eras, based on best judgement, could have been applied to the catch rates prior to input to the stock assessment model, however no information is available to quantify this adjustment objectively.
- The catch rate standardisation could have included a coefficient for relative fishable area available to fishers in each year, following the 2004 rezonings of the Great Barrier Reef by the Great Barrier Reef Marine Park Authority, delivered through the Representative Areas Program that increased no-take zones. However, when implemented, the effect of this term confounded with the 'year' term.

The truncation of the commercial catch rate time series highlighted the trade-off between reducing data available to inform the assessment, and the risk of introducing bias. Removing these years of data reduces the signal for abundance in the model, however the reliability of that signal itself is unknown.

4.4.3 Climate change and environmental drivers

As discussed in Campbell et al. (2021), saddletail snapper are not solely reef-associated, nor known to be dependent on live coral cover. They have a complex life history that includes cross-shelf movement for inshore habitats to offshore deepwater habitats as they grow and mature. They are thus thought to be less vulnerable to coral bleaching and other forms of climate-induced coral reef degradation than other reef-associated or reef-dependent species. However, climate-change-related habitat degradation of deep inter-reef areas has not been investigated (Rogers et al. 2017).

As more becomes known of saddletail habitat in recent years, such as their tendency to also inhabit isolated inshore features such as ‘wonky holes’ and wrecks, it is possible that trends in rainfall and runoff into inshore areas could have impacts upon both habitat availability (‘wonky holes’ discharge artesian water after rain events, keeping them open) and food availability (primary productivity in nearshore waters following rain). These factors may impact the occurrence of favourable conditions for juvenile survivability and growth, with young saddletail making use of inshore habitats.

It is also possible that increases in sea surface temperature will affect growth, recruitment, reproduction and mortality rates, particularly for juvenile fish that are more common inshore where environmental variation and perturbation impacts are more likely. While the precise mechanisms by which climate change and environmental drivers may impact saddletail snapper remain unclear, and any impacts to date remain unquantified, this is an additional source of uncertainty that needs to be taken into account. Despite the lack of specific environmental data, the environmental impacts on recruitment, and overall abundance/age structure is accounted for in the model implicitly, by modelling recruitment deviations.

4.5 Recommendations

4.5.1 Stock assessment

4.5.1.1 Natural mortality and steepness

The selection of appropriate values of steepness and natural mortality were guided by current best practice from the literature on these topics (Hamel and Cope 2022; Punt 2023). A detailed explanation of the basis for the selected values used in this assessment is given in Section 2.8.4. We note that currently accepted practices in stock assessment regarding steepness and natural mortality are subject to change with future research, and authors of future assessments should ensure ongoing appraisal of the most recent literature regarding the selection of appropriate values.

4.5.1.2 Revisit time-varying selectivity

In this assessment, incorporation of time-varying selectivity was attempted using methods described in Appendix B.5, though attempts were largely unsuccessful. We recommend that future assessments revisit the topic of time-varying selectivity when additional years of length composition data become available, trialing alternative time-varying mechanisms if the same lack of fit to recent annual length compositions is encountered. If this is considered, care should be taken to ensure there are sufficient data to estimate time-varying selectivity, and that the data are not being over-fitted by the model.

4.5.1.3 Depredation data

Project team discussions about depredation consistently highlighted that depredation rates are changing rapidly over time, highlighting the need for regularly updated estimates. The boat ramp survey program currently collects depredation data from the recreational fishing sector which will be available for future assessments. Methods to estimate depredation rates for the commercial fishing sector in Australian east coast waters should be pursued in the future so that the next assessment can incorporate them. This may include data from Bierwagen et al. (2026) which is currently in preparation.

4.5.1.4 Targeting

Campbell et al. (2021) recommended further investigation into the quantification of targeting saddletail snapper, for the purposes of standardising catch rates. This was explored in more depth in the current assessment (Section B.8.2). Future assessments could benefit from further exploration, if more informative data sets become available.

Targeting behaviour was defined in this assessment using the species composition reported per fishing trip. Project team members suggested that the spatial distribution of co-caught species is spatially dependent (i.e. species found in by-catch varies regionally). Further, it was suggested that depredation has significantly impacted targeting behaviour, resulting in greater diversity in fishing locations and by-catch composition. These hypotheses could be explored in future assessments when appropriate data become available.

4.5.2 Research

4.5.2.1 Depredation

There is a need for targeted research on depredation to improve stock assessment model inputs. Existing research, such as that conducted on Spanish mackerel (Mitchell in prep), cannot be directly applied to this assessment due to differences in fishing gear, practices, and locations.

Depredation rates may be influenced by a wide range of variables. Possible influences include distance offshore, day versus night activity, differing fishing practices between sectors, angler experience, location and depth, target species, proximity to human population centres, proximity to other fishers, proximity to green zones, and seasonality. These complexities highlight the need for a nuanced understanding of depredation at a fishery, or even stock-specific level. This could inform multiple parts of future assessments, including catch reconstruction, catch rate analysis and selectivity assumptions.

Further research is required to understand long-term trends in depredation, including when and how depredation rates began to increase. Recent studies, such as those from Mitchell (in prep) and Vardon (2025), have provided point estimates for depredation rates. Phone survey conducted by Fisheries Queensland as part of Sumpter et al. (2025) suggest these rates have been increasing over time. While anecdotal and social media posts can help to inform hypotheses about depredation patterns, for robust incorporation into a stock assessment model, proper survey design and empirical data are required.

Future research should focus on updating depredation rate estimates and addressing these gaps, ideally for all affected species (or at least fisheries) that would require a stock assessment in the future. Anecdotal evidence consistently identifies saddletail snapper as particularly prone to the issue of depredation, due to being caught in deeper water (longer time spent in the water column while hooked) on isolated structure where sharks may be concentrated. This may highlight the need for compilation of a priority species list for any future depredation work. Researchers should consider the capacity for the study to be regularly updated, to create a time series of depredation rates.

4.5.2.2 Fishing efficiency

Catch rates were adjusted prior to the stock assessment model to account for changes to fishing efficiency over time (Section 2.4.3). Many studies have analysed the impact of technological advances on the ability to catch fish.

Project team members cited many technologies and other factors that affect a fisher's ability to catch saddletail snapper during the period represented by catch rate data, such as side-scan sonar, electric motors with 'spot lock' capability, radar, electric reels, bathymetric mapping, location sharing, information sharing (i.e. forums, social media, training courses), years of experience in the fishery and identification of wonky holes. It was suggested that data from retailers and manufacturers may contribute to this research. Information is missing on the specific timing, prevalence and quantified, sector-specific efficiency gains associated with each of these advancements. Similarly to the topic of depredation, anecdotal evidence is plentiful and has helped form hypotheses, but empirical data is required in order to model the impact of these technologies more directly.

Further, it was flagged that an increased presence of sharks in the fishery has had a negative impact on fishing efficiency. While this hypothesis was explored based on project team consensus (Section 2.4.3), it is worth noting that this effect was quantified based on expert judgement rather than data or literature.

No specific study exists on fishing efficiency trends pertaining to saddletail snapper up to 2025. Since the model relies on catch rate signal to inform the last few years of biomass trends, this gap in the literature is particularly relevant. In this assessment, we attempted to fill this gap based on experiences within the project team, but a more detailed study is required. Although the alternative schemes for fishing efficiency trialled here produced little variation in final biomass estimates—primarily due to the short time series over which fishing efficiency trends were applied—any research to fill this gap in the literature would benefit future stock assessments, as technology continues to advance and the impact of sharks evolves.

4.5.2.3 Environmental drivers

Recent research from Leahy et al. (n.d.) used long-term age composition data to reconstruct historic year class strength of Spanish mackerel and explore a broad range of potential environmental influences on recruitment variability in the fishery. Although the time series of age composition data for saddletail snapper is shorter compared to that available for Spanish mackerel, a similar study would be beneficial to understand the key environmental drivers for the saddletail snapper stock.

4.5.2.4 Fecundity

In this assessment, fecundity was assumed to scale linearly with age however there is evidence to suggest that larger, older female fish (known as 'big old fat fecund female fish' or BOFFFFs) contribute disproportionately more to the reproductive potential of a stock than smaller, younger female fish (Evans-Powell et al. 2024). Future stock assessment of saddletail snapper would benefit from further research into fecundity as a function of length and/or age, and to test the assumed linear relationship between fecundity and age.

4.5.3 Monitoring

4.5.3.1 Continued collection of length and age data

The age, length and sex information collected by Fisheries Queensland's biological monitoring program has been fundamental to this stock assessment. Age and length data are used to inform biological processes like selectivity, as well as fishery-independent processes like growth, mortality and recruitment. For stocks that are prone to hyperstable catch rate series, there is greater dependence on age and length data to inform upon fishery-independent processes. As highlighted in Section 4.2.3, in years where length or age data are missing or lacking signal (i.e. fish are too young), this stock assessment mode relies on catch rate data to inform annual recruitment. If influential parameters like natural mortality are to be estimated in the future, it is important they are guided closely by robust data.

Natural mortality, M, was shown to be an influential parameter in this assessment. As per the methods outlined in Hamel and Cope (2022), appropriate values of M can be obtained by using the equation M = 5.4/A_{max}. This means that appropriate values of natural mortality are highly influenced by the maximum observed age of fish in the biological monitoring data. Likelihood profiling suggested the available data are not sufficient for estimating M. In future, as data time series (particularly ageing data) accumulate, and as modeling approaches for spatial structure and data weighting improve, assessments will be better able to estimate natural mortality internally (Maunder et al. 2020). Hence, continued age sampling is crucial to informing appropriate estimates of M, an influential parameter for this stock.

We recommend the continued collection of age, length and sex data, as the quality of future stock assessments of saddletail snapper is contingent upon these data. Detailed spatial information on the catch location of each biological sample, even to a commercial grid level, would increase the quality of future spatial modelling.

4.5.3.2 Depredation

In addition to the research described in Section 4.5.2.1, further depredation information could be routinely gathered through existing Fisheries Queensland surveys. Commercial fishers could record their experiences via a dedicated field in CFISH logbooks. Boat Ramp Survey design could be improved by gathering better spatial information, for example using commercial fishing grids, and depredation per species (where identifiable), rather than per trip. Recreational fishers could be further incentivised to report depredation events through the Qld Fishing 2.0 app.

It should be noted that these suggestions may risk the continuity of existing, long-term datasets, as well as adding extra reporting requirements (voluntary or otherwise) of fishers, so any changes should be carefully considered.

4.5.4 Management

This assessment highlights that the stock is unlikely to be at target levels, with a most probable estimate of 28% of unfished spawning biomass, and a high probability (91%) of falling between 20 and 40% of unfished spawning biomass. The fishing mortality estimate for 2025 exceeds that which would sustain the population at 60% biomass.

Since the previous assessment, there have been many advancements in the stock assessment process. Many of the recommendations made in Campbell et al. (2021) have been implemented, resulting in a more robust estimate of biomass. The range of uncertainty around the estimate has reduced from 13–73% in Campbell et al. (2021) to 19–42% in this current assessment. As such, less caution is required when considering the results from this current assessment.

Given that L_{50} (length at which 50% of fish are reproductively mature) for saddletail is around 57 cm FL (Hillcoat 2022; McPherson and Squire 1992) and MLS is 40 cm total length, a large proportion of retained saddletail have not contributed to spawning. Hence, an increase in the MLS may be warranted for this species, akin to what was done for red emperor following evidence from McPherson and Squire (1992). Saddletail are prone to barotrauma related discard mortality (Brown et al. 2008), with their risk of barotrauma upon capture increasing in deeper water. This could be perceived to offset some benefits of a size-limit increase. However, captures of smaller saddletail are uncommon on deeper, offshore grounds, with fish of smaller size classes predominantly caught in much shallower, inshore areas, where their chance of successful release is higher. A size-limit increase therefore has potential to reduce pressure on smaller saddletail and increase the number of fish reaching spawning size.

Newman et al. (2000) conducted yield-per-recruit modelling on otoliths from tropical red snappers (including saddletail snapper) from the Queensland east coast stock. They concluded that the optimal yield would be obtained if saddletail snapper first entered the fishery at 6 years of age. Using the age-length relationship determined in the same study, this corresponded to a fork length of 59 cm. Using the growth curve output by the current stock assessment, 6 years of age corresponds more closely to around 65 cm (Figure 4.1). These findings might further support an increase in MLS for saddletail snapper.

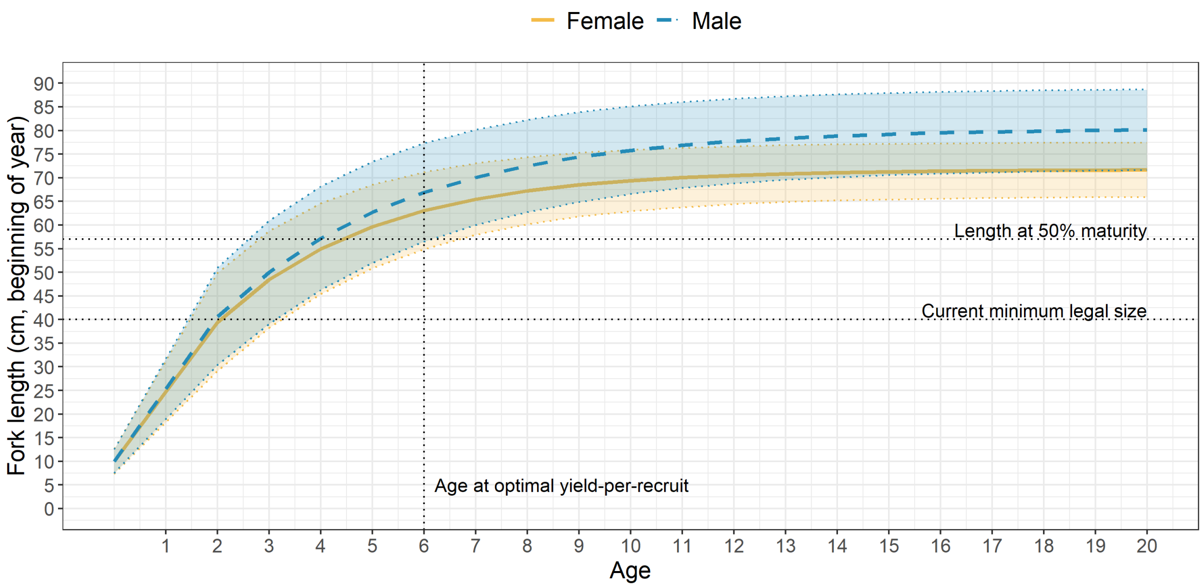

Figure 4.1: Estimated growth of Queensland east coast saddletail snapper (95% confidence intervals) for the representative MLE model (scenario 1)—truncated along the x-axis, with informative management indicators

Since 2004, there has been a combined possession limit of 9 saddletail and crimson snapper for recreational fishers. The species have been managed together because of their similar appearance and tendency to be caught together. Since the management arrangements were made, stock assessments have been conducted on both species (Campbell et al. (2021), Fox et al. (2021) and this report). Fishers tend to consider saddletail snapper a more favourable fish compared to crimson snapper, some even releasing crimson snapper in order to retain more saddletail (Fisheries Queensland 2024). In 2024, following consultation with the Reef Line Fishery Working Group, Fisheries Queensland sought public consultation on a proposal to introduce species-specific possession limits (specifically 4 crimson snapper and 4 saddletail snapper) (Fisheries Queensland 2024). This stock assessment, providing greater confidence around the biomass estimate, should be considered in subsequent management discussions.

Saddletail snapper are a relatively long-lived species, meaning natural mortality rates are low, and stock turnover is slow. This can mean that any positive recruitment events may take some time to propagate through the population and result in any spawning biomass increase. The tendency for saddletail snapper to inhabit small, isolated habitat features that have historically been hard to find, render them particularly prone to recent rapid advances in fishing technology that allow more fishers to locate these grounds. Appraisal of these species-specific factors by fishery managers is important when considering any future management actions for the species.

Specific recommendations around any particular management proposal would require a separate analysis that takes the current assessment and models the proposal by projecting forward, taking care to capture the estimated uncertainty in that projection. More complex management proposals may warrant a simulation-based management strategy evaluation. Management proposals should consider the uncertainty associated with the depredation rates used in this assessment—and their subsequent effect on biomass results (see Appendix B.1). In addition, management proposals should consider how engagement with the fishery varies regionally.

For convenience, a breakdown of annual catch by sector, with depredation separated out, can be found in Table E.1.

4.6 Conclusions

This stock assessment was commissioned to establish the status of the east coast saddletail snapper stock and inform the management of the Reef Line Fishery. Biomass was estimated to be between 19% and 42% at the beginning of 2026, relative to an assumed unfished state in 1961. This interval was generated over an ensemble of 12 scenarios. Some recommendations for future assessment and monitoring have been made.

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A Project Team decisions and recommendations

Project teams form an important part of the stock assessment process by providing guidance from experts from various disciplines relevant to the stock assessment. This approach ensures scientific validation and increases transparency.

The following sections of this appendix briefly describe decisions and recommendations made by the project team for this assessment, both in and out of session of project team meetings.

A.1 Catch reconstruction

A.1.1 Historical commercial catch adjustment

The project team recommended an adjustment to the commercial catch history from 1997 to 2004 as a result of potential over-reporting, using proportional decline in active licence holders to calculate the adjustment factor. The project team recommended adjustment factors be explored, to assess the impact on the final model results. Upon confirming the adjustment factor did not have a major impact on the final results, the project team confirmed an adjustment factor of 83% and cautioned against excessive scaling.

A.1.2 Recreational catch extrapolation

The project team were in favour of using boat ramp survey data to estimate trends in recreational catch since the more recent state wide fishing survey in 2019.

A.1.3 Depredation

The project team encouraged thorough consideration of the available data, literature and precedents pertaining to depredation rates for saddletail snapper. They noted a significant increase in depredation over time, particularly since 2009.

A project team member recommended seeking out any insights from an active study, conducted by the Australian Institute of Marine Science, which uses underwater cameras. The lead author met with the principal investigator, but no suitable data would be available for use in time for the current stock assessment.

The project team endorsed the depredation rates that were used for the base case and recommended an investigation into alternative schedules of depredation rates (Appendix B.1).

A.2 Commercial catch rates

A.2.1 Truncation of CPUE time series

The project team acknowledged the potential bias that would arise from including CFISH logbook data from years leading up to 2005, due to potential spurious signal that arose from significant changes to the fishery management (resulting in both a change in fisher behaviour, availability of fishing ground and possible misreporting). The project team considered the loss of potentially valuable signal if the data were not spurious. Project team members considered splitting the commercial catch rate into two separate time series (represented by two commercial fleets in Stock Synthesis) or making other external adjustment for catchability changes.

Both alternatives of CPUE time periods (1989–2025 and 2008–2025) were input to the stock assessment model to assess the impact on final model results, which was relatively small.

The project team decided that removing data prior to these management changes was appropriate, citing the need for reliable data, and the precedence that has been set by adjusting the historical catch reconstruction due to the same concerns for misreporting. In addition, catch rate data from 2005 to 2007 was also excluded due to small sample sizes as the fishery adjusted to new management arrangements.

A.2.2 Targeting

The project team discussed several methods for identifying fishing trips in which saddletail snapper were targeted. They discussed the spatial variation in the composition of species that are commonly caught with saddletail snapper, indicating that co-caught species may not be a suitable indicator alone. It was also noted that depredation has significantly impacted targeting behaviour.

The project team were in favour of exploring multiple definitions of targeting behaviour as a preliminary investigation (Appendix B.8.2). One of these included the Stephens and MacCall (2004) method, which is a non-subjective statistical approach to account for targeting behaviour. As there was strong agreement between three of the tested methods, including the Stephens and MacCall (2004) statistical method, the project team endorsed a binary definition of targeting in which the catch from a targeted fishing event comprised at least 70% saddletail snapper.

A.2.3 Fishing efficiency

The project team suggested there have been significant increases in fishing efficiency in the past five years due to advancements like side-scan sonar, electric motors, radar, bathymetric mapping and online resources and education. They also suggested that the recreational sector experienced greater increase in fishing efficiency than commercial in the last five years.

The project team considered combining their industry experience to reconstruct a fishing efficiency index based on the perceived uptake and benefit of various advancements to fishing technology. When combined, this scale of this reconstructed index was a substantial underestimate compared to other sources from the literature, so it was not used for the base case scenario and instead only included as an additional modelling scenario, presented in Appendix B.3.

The project team recommended exploring multiple fishing efficiency trends in order to understand the scale of its impact on final model results. Ultimately the project team endorsed a single fishing efficiency index for the ensemble model.

A.3 Recreational catch rates

The project team discussed the regions used for the standardisation of recreational catch rates, noting the requirement to have sufficient data in each regional strata. The project team supported the stratification of three regions: north of Cairns (sparsely populated with less fishing pressure), Cairns to Mackay (higher population density, greater fishing pressure), south of Mackay (fishing generally further offshore). The project team noted concerns about the low number of interviews of fishing trips to high-abundance areas in boat ramp survey data.

A.4 Model specification

A.4.1 Initial fishing mortality

The project team discussed the possibility of starting the model at a non-virgin biomass (e.g. an unfished biomass less than 100%) to account for potential fishing prior to 1961, however the team agreed that fishing pressure prior to 1960 was negligible due to limited technology and human population size in the region.

A.4.2 Time-varying commercial selectivity

The project team considered the logic of introducing time-varying commercial selectivity as a means to improve model fits to length data. In future, when data permit, allowing for time-varying selectivity could be of benefit to better account for changes in spatial distribution and fishing behaviour.

The modellers explored time-varying selectivity scenarios for this assessment, and after reporting findings back to the group, the project team confirmed that commercial selectivity would not be time-varying in the ensemble of models (Appendix B.5).

A.4.3 Natural mortality and steepness

The project team endorsed the choice of fixed natural mortality and steepness parameters. Members noted that natural mortality may be time-varying, and potentially impacted by fluctuations in shark populations, however data were not available to support estimation of these variables and any variation over time.

B Additional scenarios

B.1 Depredation

Outside of the chosen ensemble scenarios, alternative rates for depredation were trialled to gauge the model's sensitivity to this mechanism. As described in Section 2.2.5, the scheme for depredation used in all ensemble scenarios was as follows:

- The rate used for depredation in 2025 was 58% for recreational, charter and Traditional Owner catch. For commercial boats, half of this rate was used (29%)
- Depredation rates were assumed to be a constant 1% between 1961 and 2009
- Between 2009 and 2025, the depredation rates increased linearly.

Table B.1: Retained catch per sector between 2009 and present, for take by sharks and humans, expressed in tonnes with annual percentages—percentages add to 100% for each row

Year	Commercial		Recreational & Traditional Owners		Charter	
	Fishing	Depredation	Fishing	Depredation	Fishing	Depredation
2009	65 t (20%)	1 t (0%)	239 t (72%)	2 t (1%)	23 t (7%)	0 t (0%)
2010	52 t (17%)	1 t (0%)	220 t (73%)	10 t (3%)	19 t (6%)	1 t (0%)
2011	54 t (19%)	2 t (1%)	195 t (68%)	16 t (6%)	18 t (6%)	1 t (1%)
2012	38 t (13%)	2 t (1%)	209 t (71%)	24 t (8%)	18 t (6%)	2 t (1%)
2013	47 t (15%)	4 t (1%)	208 t (68%)	32 t (10%)	14 t (5%)	2 t (1%)
2014	54 t (17%)	5 t (2%)	196 t (64%)	37 t (12%)	12 t (4%)	2 t (1%)
2015	64 t (18%)	7 t (2%)	214 t (60%)	48 t (14%)	17 t (5%)	4 t (1%)
2016	79 t (19%)	10 t (3%)	235 t (58%)	61 t (15%)	15 t (4%)	4 t (1%)
2017	77 t (17%)	12 t (3%)	249 t (55%)	74 t (16%)	35 t (8%)	10 t (2%)
2018	99 t (20%)	17 t (3%)	250 t (52%)	83 t (17%)	26 t (5%)	9 t (2%)
2019	70 t (19%)	13 t (4%)	186 t (51%)	68 t (19%)	19 t (5%)	7 t (2%)
2020	63 t (17%)	13 t (3%)	190 t (50%)	76 t (20%)	25 t (7%)	10 t (3%)
2021	70 t (12%)	15 t (3%)	300 t (53%)	131 t (23%)	32 t (6%)	14 t (2%)
2022	75 t (16%)	18 t (4%)	231 t (49%)	109 t (23%)	26 t (6%)	12 t (3%)
2023	74 t (13%)	19 t (3%)	281 t (50%)	143 t (26%)	29 t (5%)	15 t (3%)
2024	89 t (16%)	24 t (4%)	252 t (46%)	137 t (25%)	26 t (5%)	14 t (3%)
2025	98 t (17%)	28 t (5%)	267 t (45%)	155 t (26%)	29 t (5%)	17 t (3%)

Due to the uncertainty associated with depredation rate estimates, we explored alternative values for the base rate used for recreational, charter and Traditional Owner catch (i.e. 58%). These are defined in Table B.2 and covered a deliberately broad range of hypothetical depredation scenarios.

Independent of this stock assessment, two industry members used social media to gather feedback from peers with experience fishing this stock. While it was difficult for the authors to gauge the accuracy of many responses (or if they pertained to the stock in question), analysis of the data resulted in an approximate depredation rate of 50%. This value is in the middle of the range represented by the additional explorations.

These schedules for depredation rates were applied to the catch reconstruction (per Section 2.2.5) and run through the stock assessment model (maximum likelihood estimate) to estimate a final biomass for each scenario.

Table B.2: Scenarios tested to determine sensitivity to depredation assumptions

Scenario	Base depredation rate used for recreational, charter and Traditional Owner catch
Scenario A	0%
Scenario B	20%
Scenario C	40%
Scenario D	60%
Scenario E	80%
Scenario F	100%

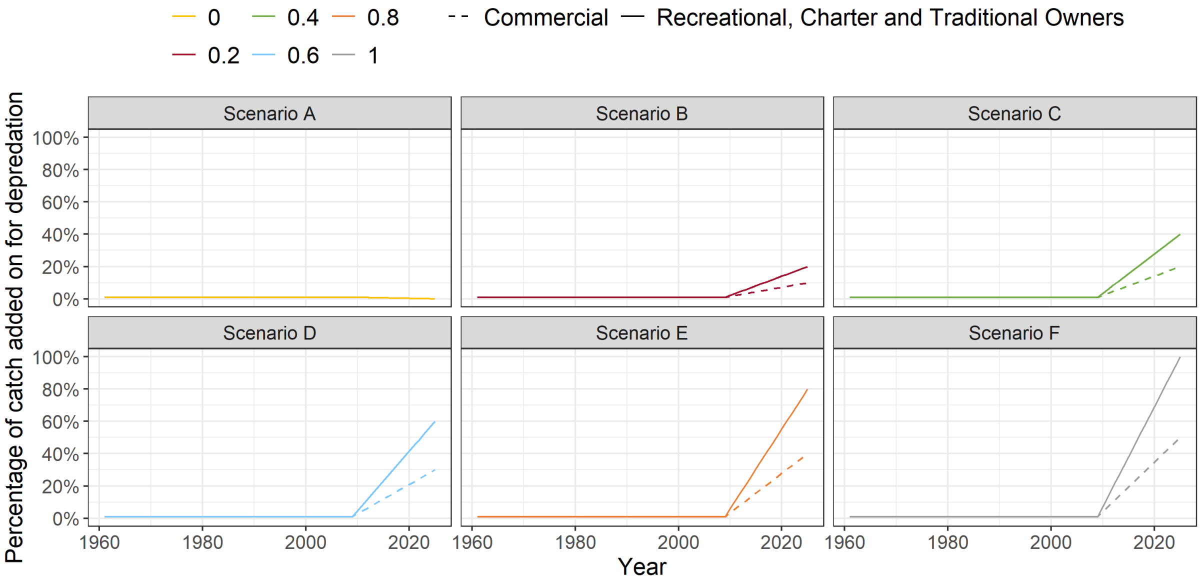

Figure B.1: Additional depredation rate schemes for Queensland east coast saddletail snapper

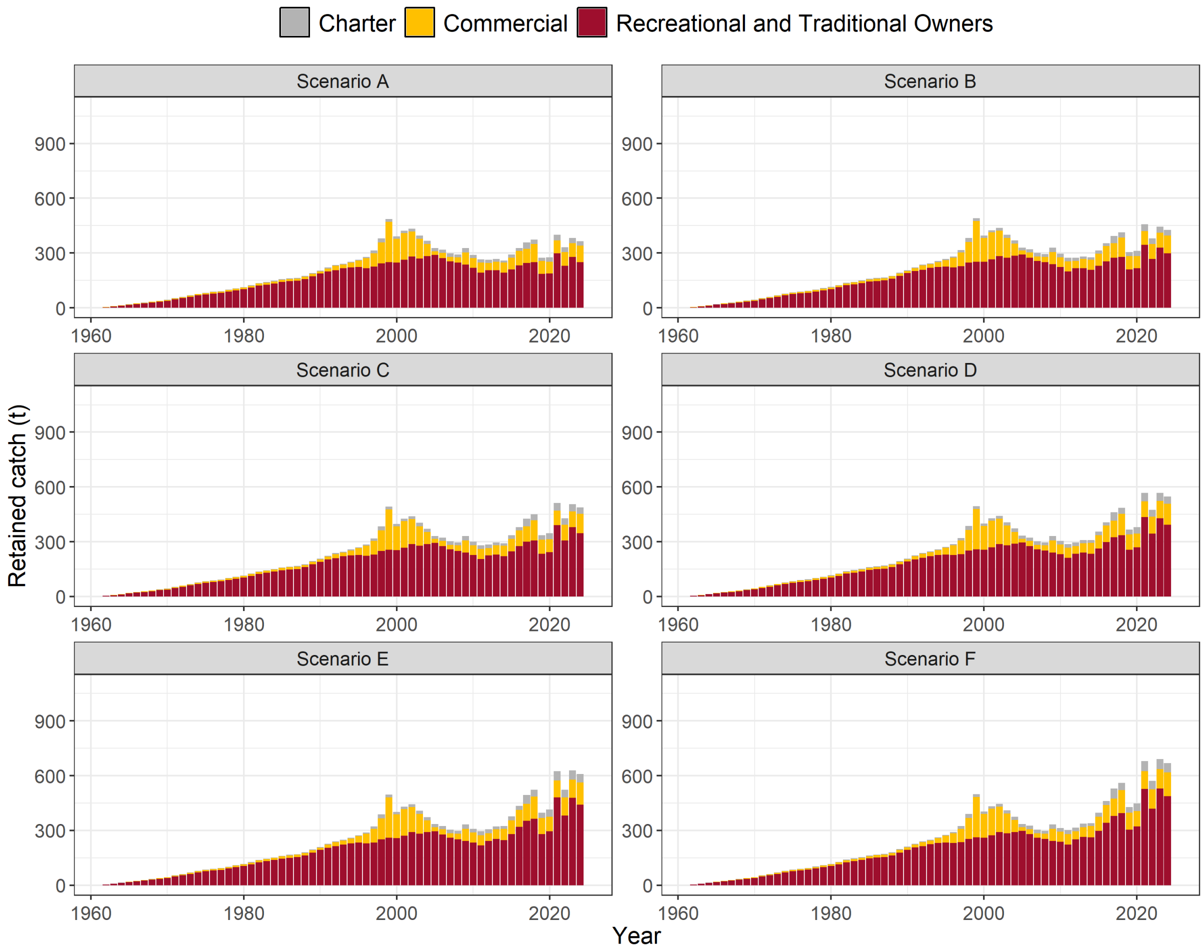

Figure B.2: Input catch data from additional depredation scenarios

The resulting biomass from these six scenarios are shown in Figure B.3.

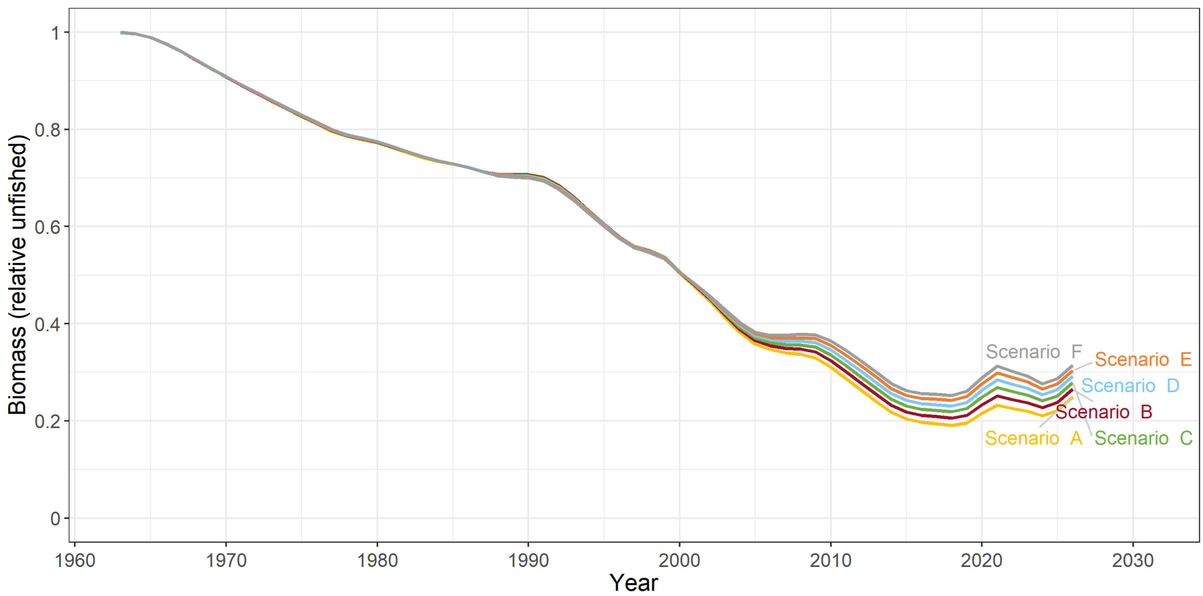

Figure B.3: Predicted spawning biomass trajectory relative to unfished for Queensland east coast saddletail snapper, from depredation scenarios

Notably, with fixed values for steepness and natural mortality, the model accounted for the increased fishing mortality by estimating a higher value of $ln(R0)$ (i.e. the number of recruits in the unfished population) (Figure B.4).

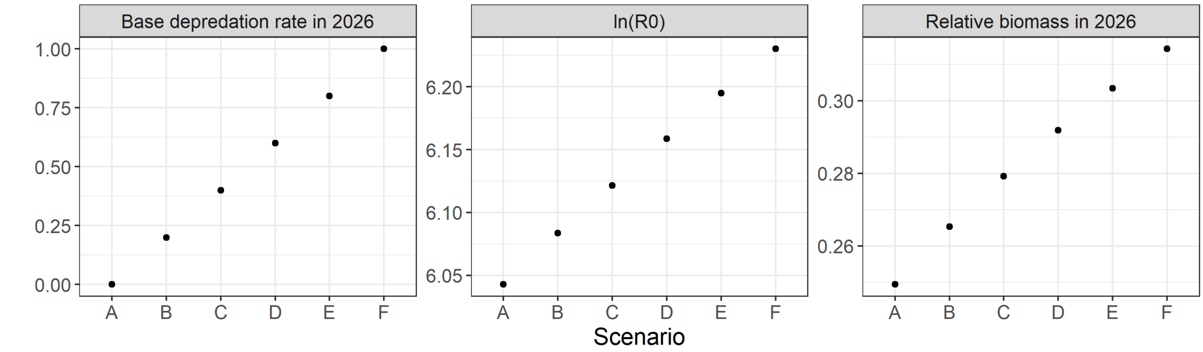

Figure B.4: Impact of increased depredation on biomass and ln(R0)

B.2 Historical commercial catch adjustment

Using the methods described in Section 2.2.1.2, three scaling factors (0.83, 1.00 and 0.38) were applied the commercial catch estimates from 1997 to 2004, to explore the impact of the uncertainty associated with potential over-reporting during this period (Figure B.5 and Table B.3).

This adjustment to the catch data input had negligible impact on the final biomass estimate when optimised in the stock assessment model (Figure B.6).

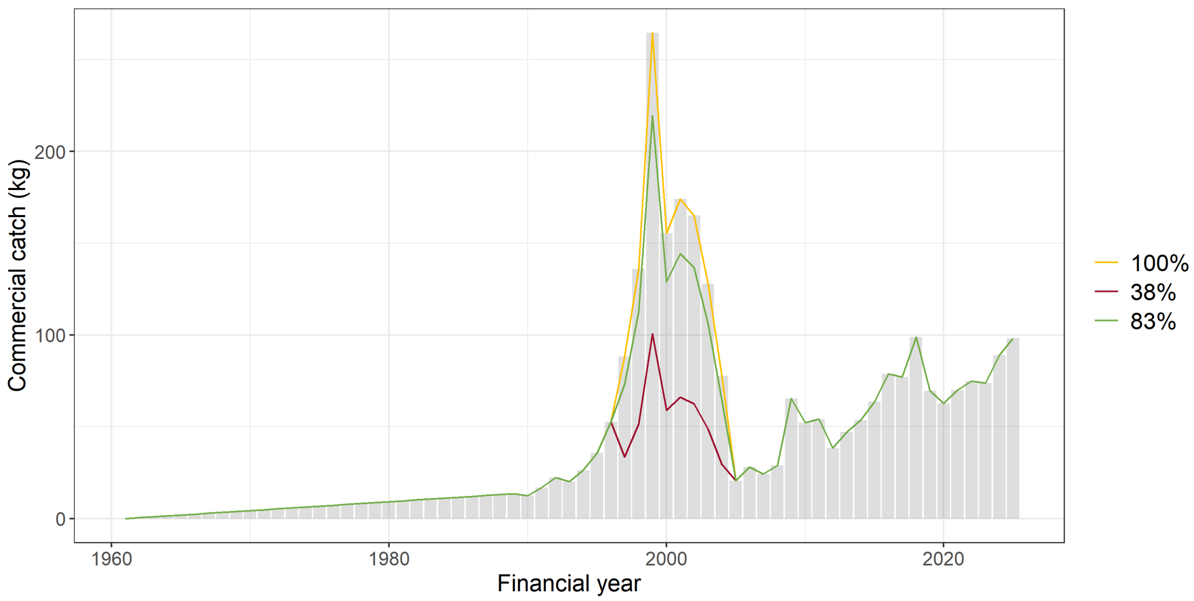

Figure B.5: Estimated retained catch by sector between 1961 and 2025 for Queensland east coast saddletail snapper

Table B.3: Scenarios tested to determine sensitivity to historical catch assumptions

Scenario	Scaling factor applied the commercial catch estimates
Scenario A	0.83
Scenario B	1.00
Scenario C	0.38

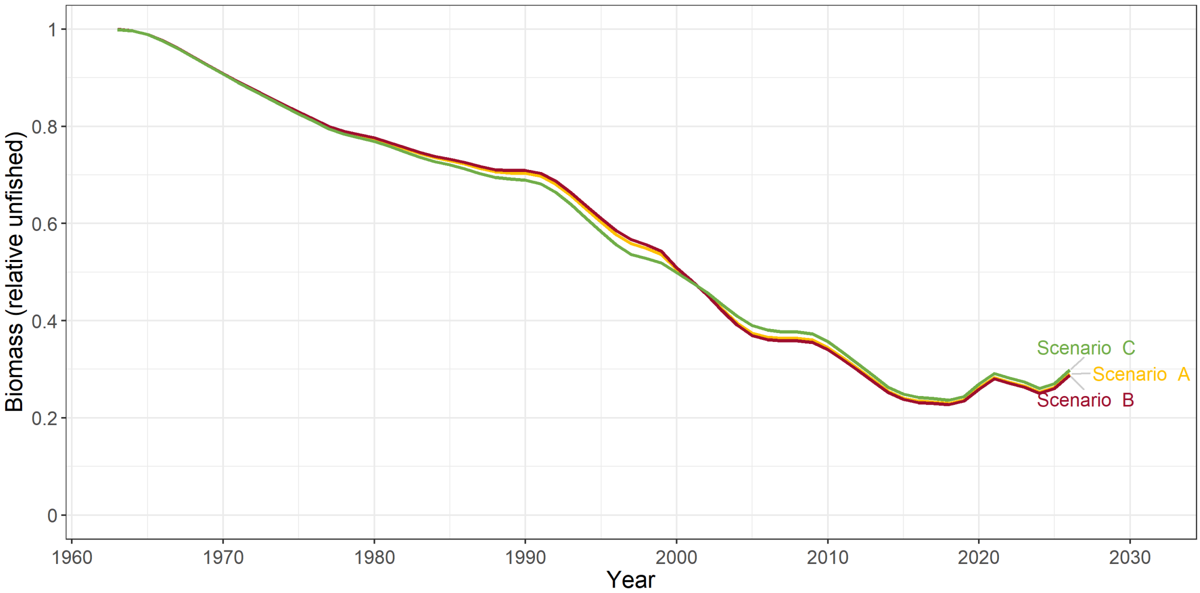

Figure B.6: Predicted spawning biomass trajectory relative to unfished for Queensland east coast saddletail snapper, from historical commercial catch scenarios

B.3 Fishing efficiency

As described in Section 2.4.3, adjustments were made to the catch rate time series—after the standardisation process and before input into the population model. This was done to account for changes in fishing efficiency which would otherwise impact on the reliability of the catch rate data as an index of abundance of saddletail snapper.

Several fishing efficiency schemes were trialled, as defined in Table B.4. Figure B.7 shows the annual increase to fishing efficiency for each additional scenario, and Figure B.8 shows how these annual changes translate into a fishing efficiency index, which was used to transform the catch rate indices that were input to the model (Figure B.9).

Figure B.10 shows that variations for fishing efficiency had minimal impact on the final biomass trajectories when optimised in the stock assessment model.

Table B.4: Scenarios tested to determine sensitivity to fishing efficiency assumptions

Scenario	Fishing efficiency pattern
Scenario A	2% increase each year, reduced to 1% since 2021 due to depredation
Scenario B	3% increase each year, reduced to 2% since 2021 due to depredation
Scenario C	1% increase each year, reduced to 0% since 2021 due to depredation
Scenario D	2% increase each year, offset by a linear increase in depredation rates (0 in 2009 to 0.016 in 2025)
Scenario E	2% increase each year
Scenario F	3% increase each year
Scenario G	1% increase each year
Scenario H	Custom project team pattern based on perceived uptake rate and impact of different technologies

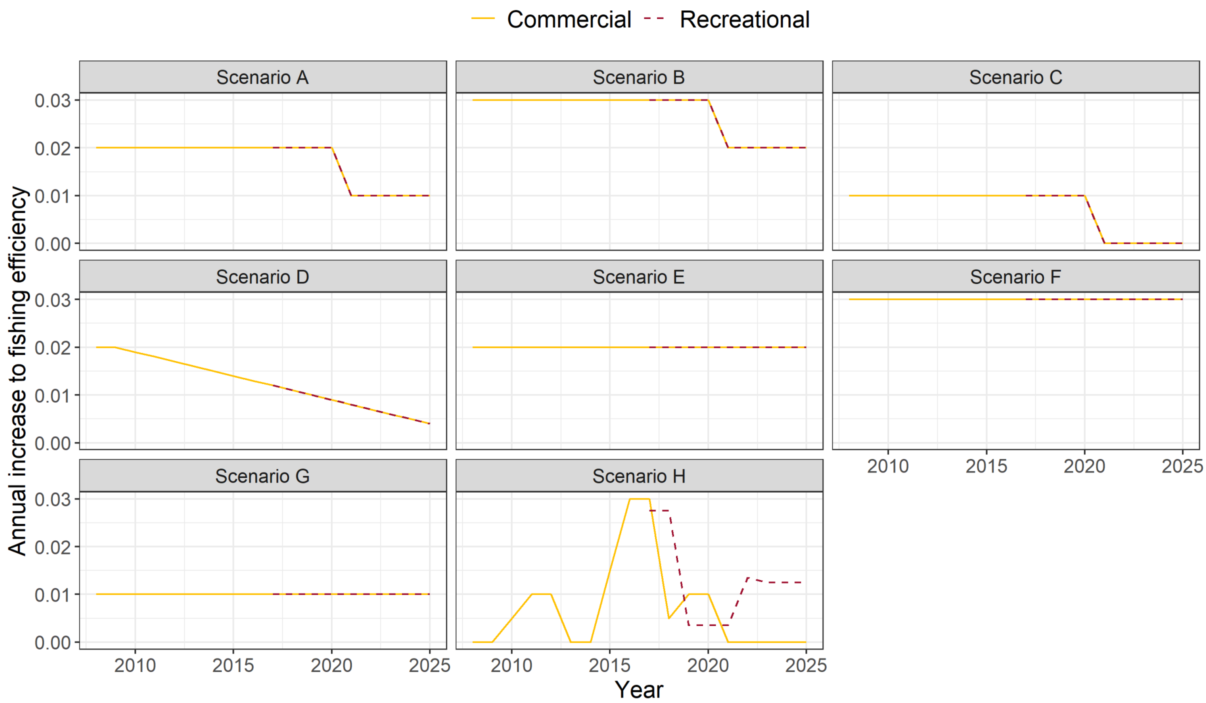

Figure B.7: Annual adjustments to fishing efficiency increase for each additional scenario

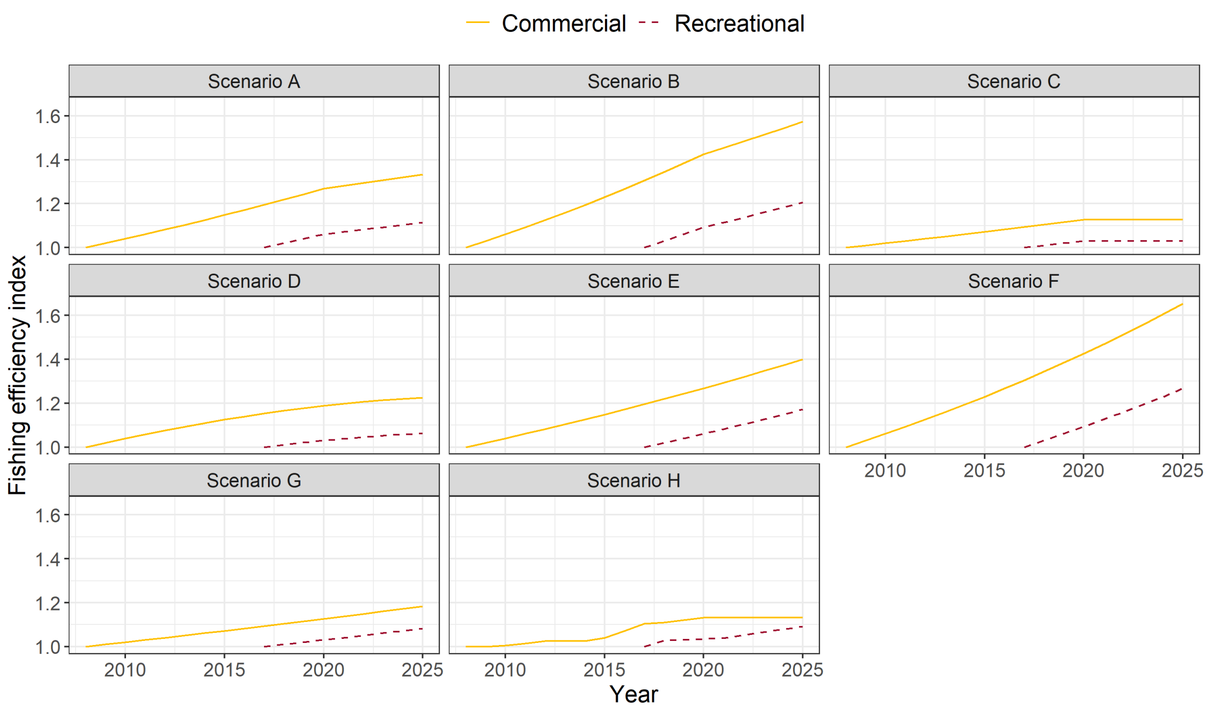

Figure B.8: Index of fishing efficiency for each additional scenario

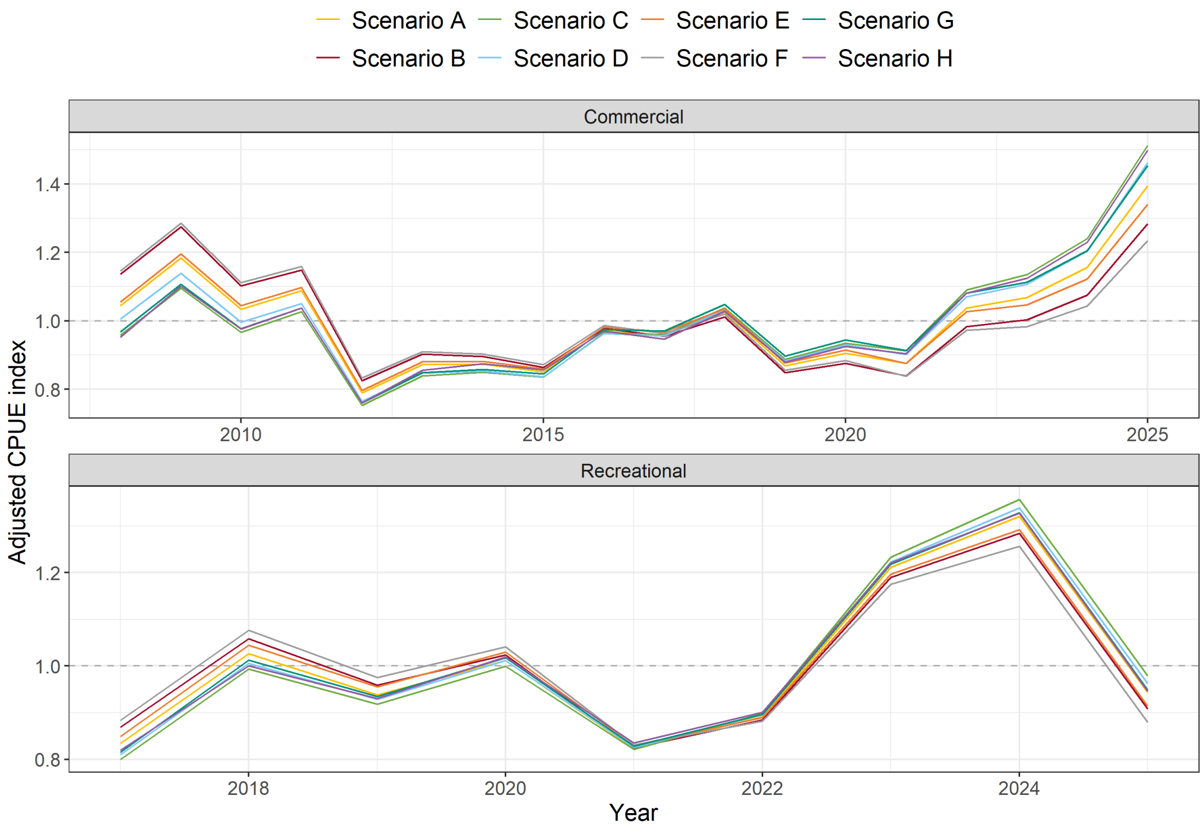

Figure B.9: Adjusted catch rate data entered into each stock assessment model after fishing efficiency transformation was applied (normalised to 1.0)

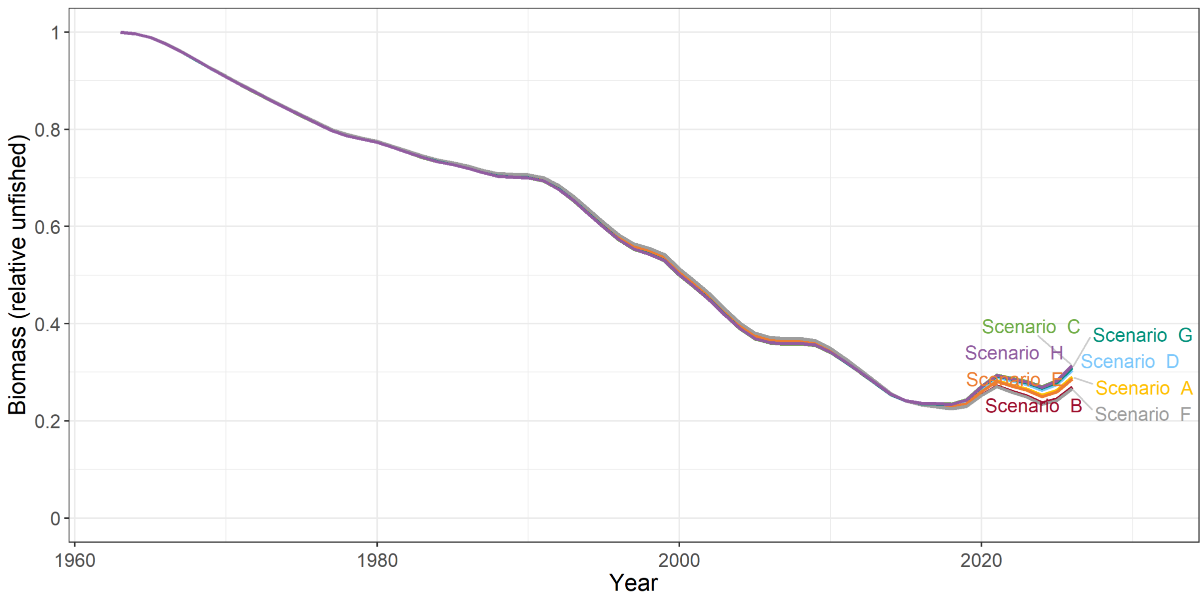

Figure B.10: Predicted spawning biomass trajectory relative to unfished for Queensland east coast saddletail snapper, from fishing efficiency scenarios

B.4 Steepness (h)

Scenarios were run to explore the effect of additional values for steepness (*h*). Both 0.55 and 0.95 were chosen as examples of the extremities of plausible steepness values, and explored along with the three values used in the ensemble (0.65, 0.75, 0.85). The effect on final biomass is shown in Figure B.11.

Table B.5: Scenarios tested to determine sensitivity to additional steepness values

Scenario	M **(yr^{-1})**	h	**Length data weighting**
1	Fixed at 0.14 yr^{-1}	0.75	Effort
2	Fixed at 0.14 yr^{-1}	0.65	Effort
3	Fixed at 0.14 yr^{-1}	0.85	Effort
4	Fixed at 0.11 yr^{-1}	0.75	Effort
5	Fixed at 0.11 yr^{-1}	0.65	Effort
6	Fixed at 0.11 yr^{-1}	0.85	Effort
7	Fixed at 0.14 yr^{-1}	0.75	Population
8	Fixed at 0.14 yr^{-1}	0.65	Population
9	Fixed at 0.14 yr^{-1}	0.85	Population
10	Fixed at 0.11 yr^{-1}	0.75	Population
11	Fixed at 0.11 yr^{-1}	0.65	Population
12	Fixed at 0.11 yr^{-1}	0.85	Population
A	Fixed at 0.14 yr^{-1}	0.55	Effort
B	Fixed at 0.14 yr^{-1}	0.95	Effort
C	Fixed at 0.14 yr^{-1}	0.55	Population
D	Fixed at 0.14 yr^{-1}	0.95	Population
E	Fixed at 0.11 yr^{-1}	0.55	Effort
F	Fixed at 0.11 yr^{-1}	0.95	Effort
G	Fixed at 0.11 yr^{-1}	0.55	Population
H	Fixed at 0.11 yr^{-1}	0.95	Population

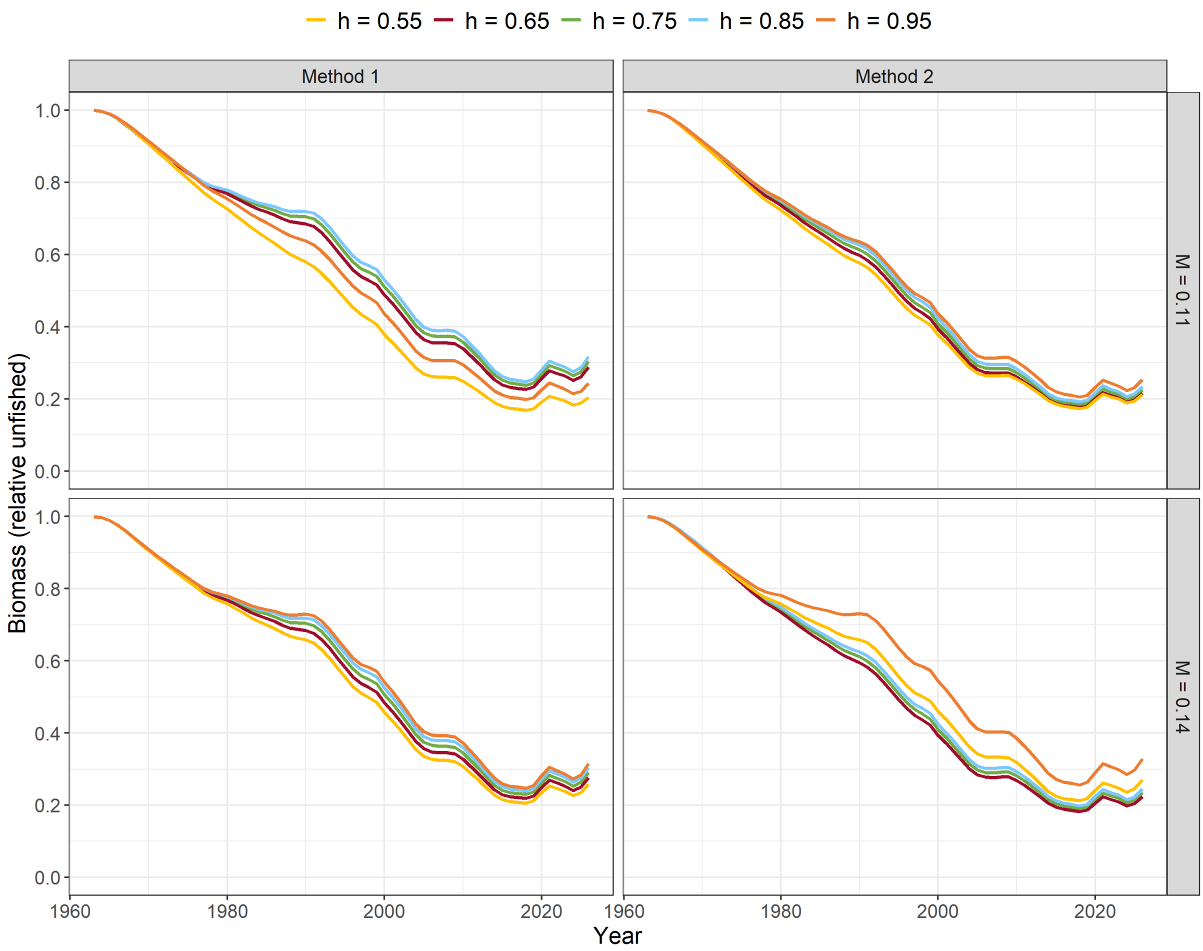

Figure B.11: Predicted spawning biomass trajectory relative to unfished for Queensland east coast saddletail snapper, from additional steepness scenarios

B.5 Commercial selectivity

In an effort to improve fits to the later years of commercial length data, time-varying selectivity mechanisms were explored (defined in Table B.6).

Figure B.12 shows that time-varying selectivity over the years 2018 to 2025 (for which there were length data to inform selectivity parameter estimates) had negligible impact on the final biomass estimate. Expanding the period for time-varying selectivity to 1961 to 2025 (i.e. the entire assessment period) has a much greater impact on the final biomass. However the estimates for selectivity during this extended period are not informed by input length composition data and, in the case of Scenarios F and G, do not result in plausible annual selectivity patterns in earlier years (Figure B.15). The implausible selectivity patterns estimated in these scenarios subsequently inferred unacceptable recruitment deviation trends (Figure B.14).

Figure B.13 shows that overall length data fits were not noticeably improved in Scenarios B to E (2018–2025) by the inclusion of time-varying selectivity. Scenarios H and I showed slight improvements to the aggregated commercial length fits, however this was achieved by applying time-varying selectivity for the full assessment period, including years where informative data were absent.

Table B.6: Scenarios tested to determine sensitivity to commercial selectivity assumptions

Scenario	Time-varying years	Time-varying parameters	Time-varying method (deviation link)
Scenario A	None	None	None
Scenario B	2018–2025	Inflection	Multiplicative
Scenario C	2018–2025	Inflection, 95% width	Multiplicative
Scenario D	2018–2025	Inflection	Additive
Scenario E	2018–2025	Inflection, 95% width	Additive
Scenario F	1961–2025	Inflection	Multiplicative
Scenario G	1961–2025	Inflection, 95% width	Multiplicative
Scenario H	1961–2025	Inflection	Additive
Scenario I	1961–2025	Inflection, 95% width	Additive

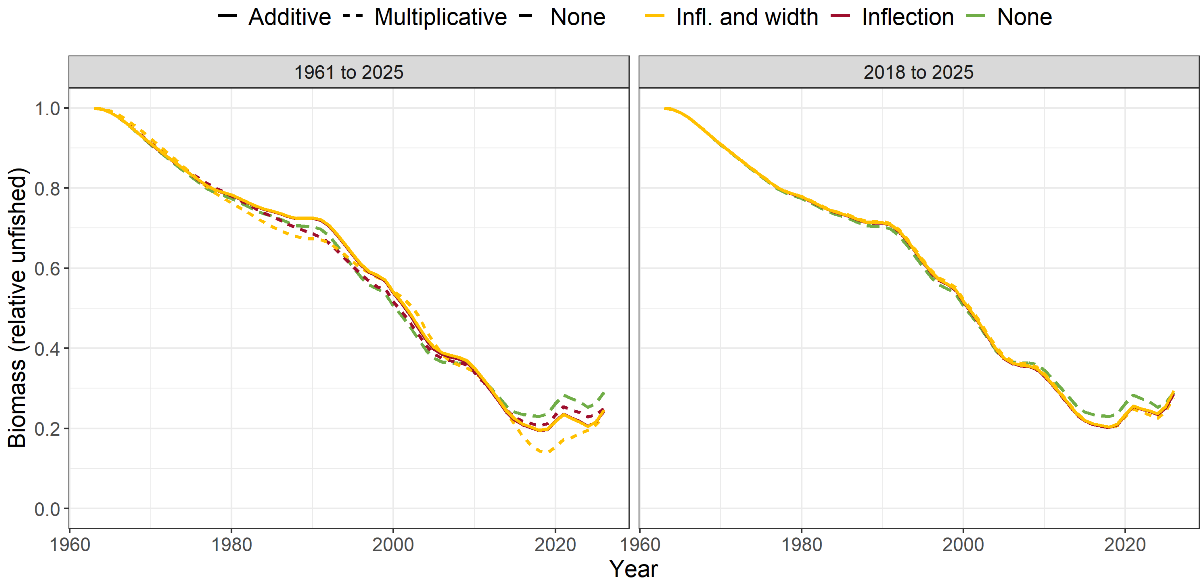

Figure B.12: Predicted spawning biomass trajectory relative to unfished for Queensland east coast saddletail snapper, from time-varying commercial selectivity scenarios

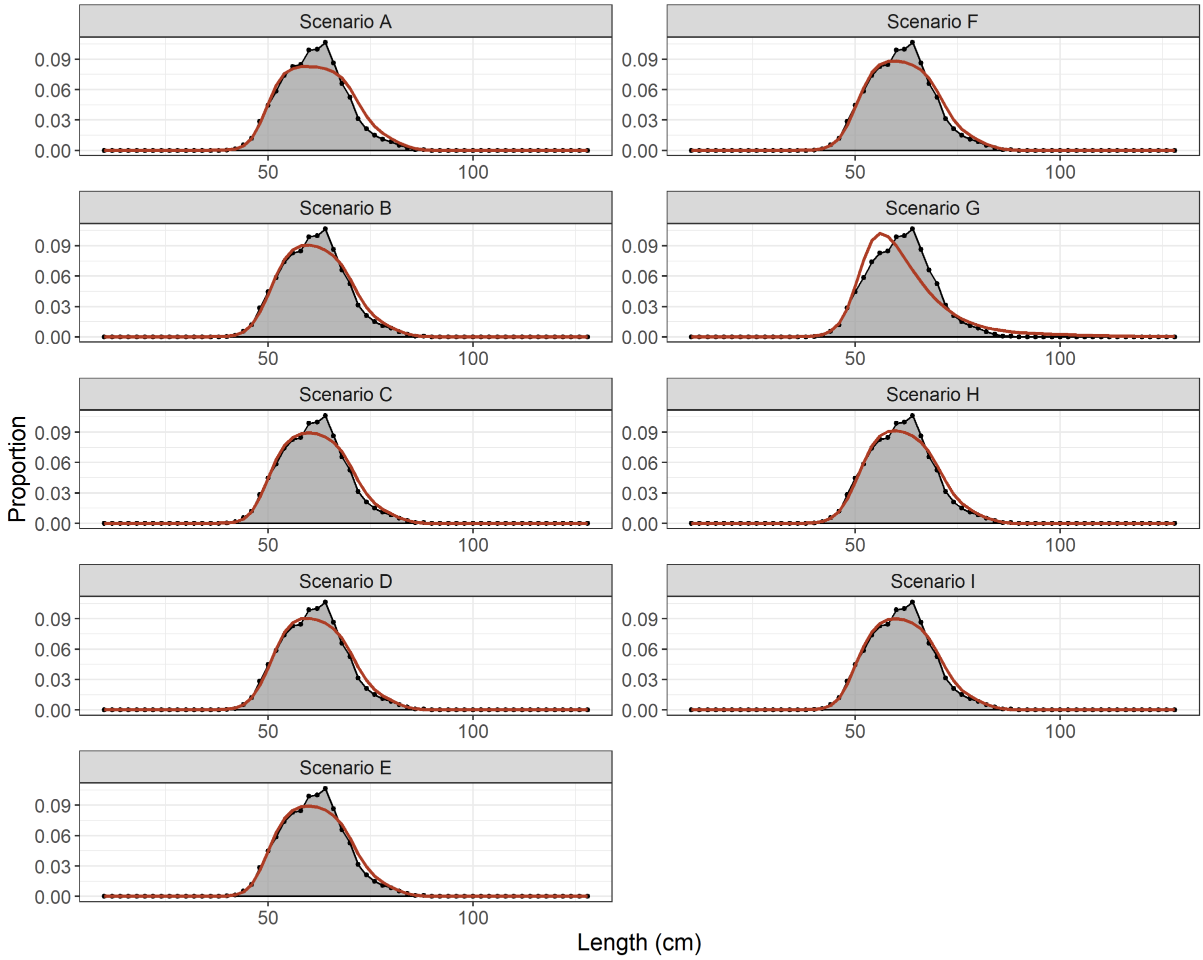

Figure B.13: Aggregated fits to commercial length data, from commercial selectivity scenarios

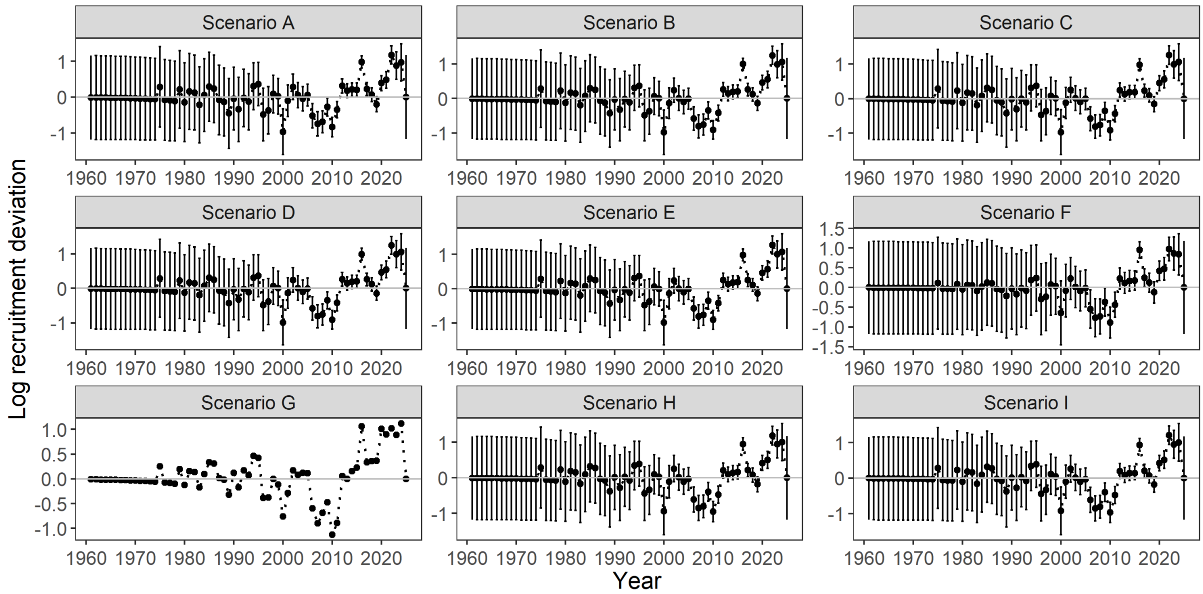

Figure B.14: Recruitment deviation estimates, from time-varying commercial selectivity scenarios

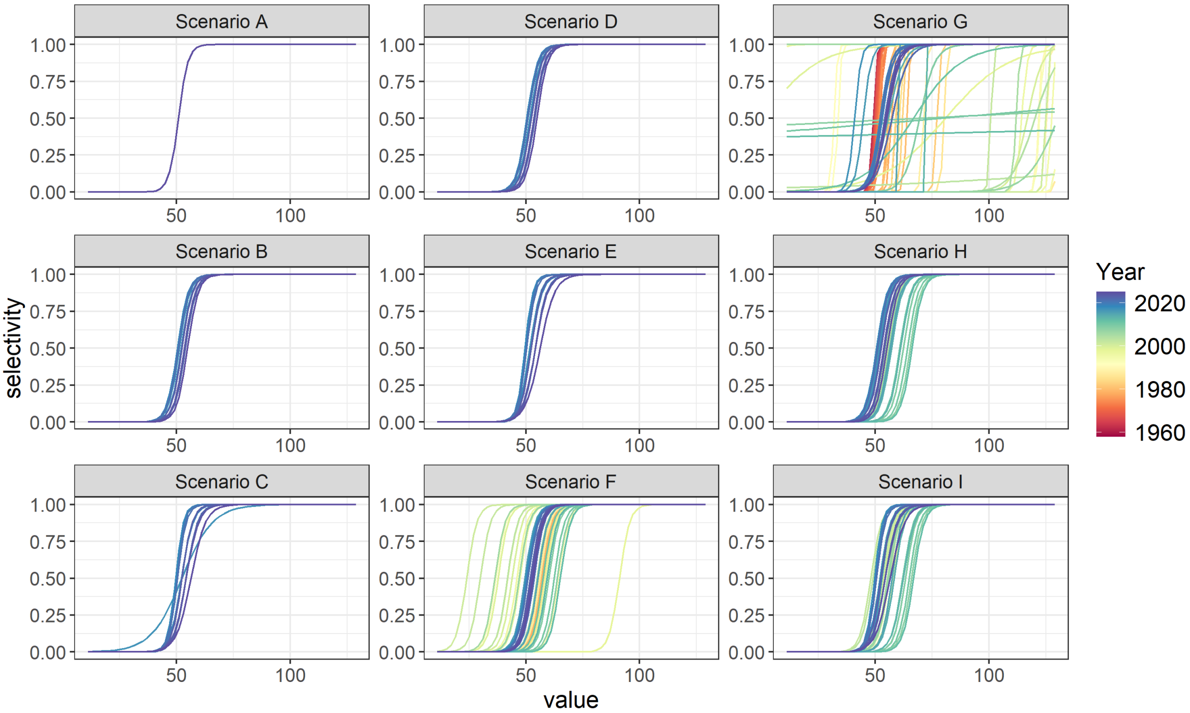

Figure B.15: Commercial selectivity estimates, from time-varying commercial selectivity scenarios

In further additional scenarios, the sample sizes of the last two years of length data were artificially inflated (Table B.7). This had negligible impact on the final biomass ratio (Figure B.16).

Table B.7: Scenarios tested to determine sensitivity to commercial selectivity assumptions

Scenario	Nsamp multiplier (2024–2025)
Scenario J	5
Scenario K	10

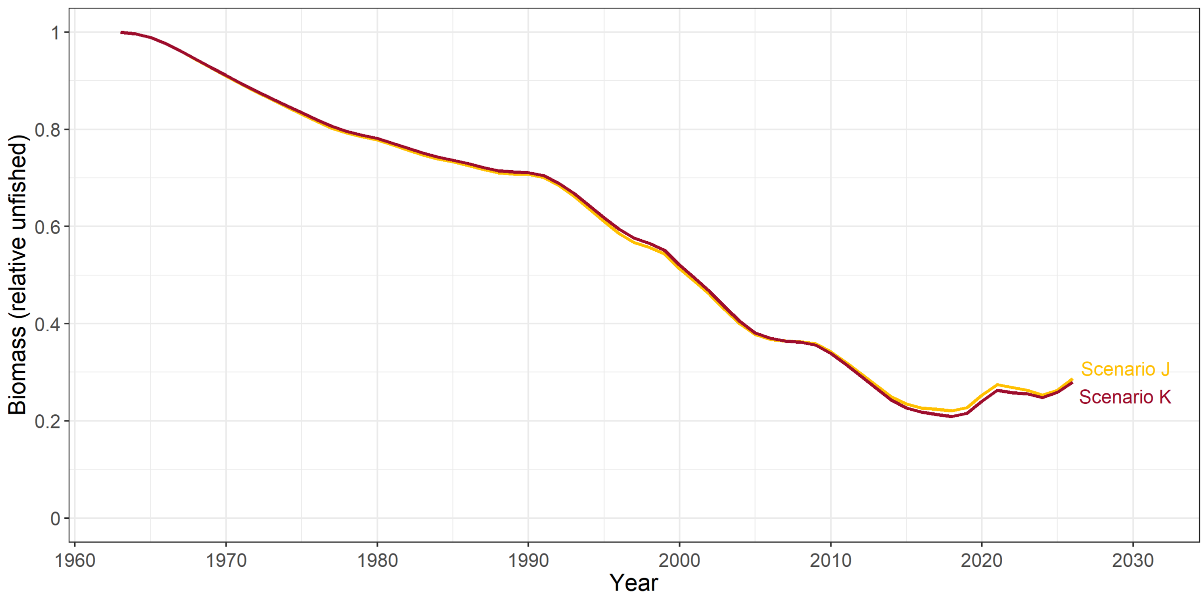

Figure B.16: Predicted spawning biomass trajectory relative to unfished for Queensland east coast saddletail snapper, from additional commercial selectivity scenarios with inflated Nsamps

B.6 Recreational selectivity

The selectivity curve that was estimated for the recreational fleet included a large proportion of area under the curve below MLS (40 cm), suggesting a considerable proportion of size classes below MLS are vulnerable to recreational fishing. Additional scenarios were run to explore the effect of reducing this area using different specifications for the recreational selectivity curve. These scenarios included adjusting the position and relative values of the nodes used to define spline parameter estimates, and changing the shape of the selectivity curve from spline to logistic (Table B.8).

Scenario A (Table B.8) represents the base case that was used in the ensemble. The initial positions and values for each node of the recreational selectivity curve were determined using the in-built auto-generation feature in Stock Synthesis.

Although these adjustments had varying effect on the fits to length composition data (Figure B.17), they had minimal effect on the final biomass estimates (Figure B.18). There was also varying difference in the recruitment deviation estimates between these additional scenarios (Figure B.19).

Table B.8: Scenarios tested to determine sensitivity to recreational selectivity assumptions

Scenario	Shape of curve	Position of first knot	Value of first knot
Scenario A	Spline	-0.54	37
Scenario B	Spline	-0.2	40
Scenario C	Spline	-0.1	40
Scenario D	Spline	-0.1	45
Scenario E	Logistic	NA	NA

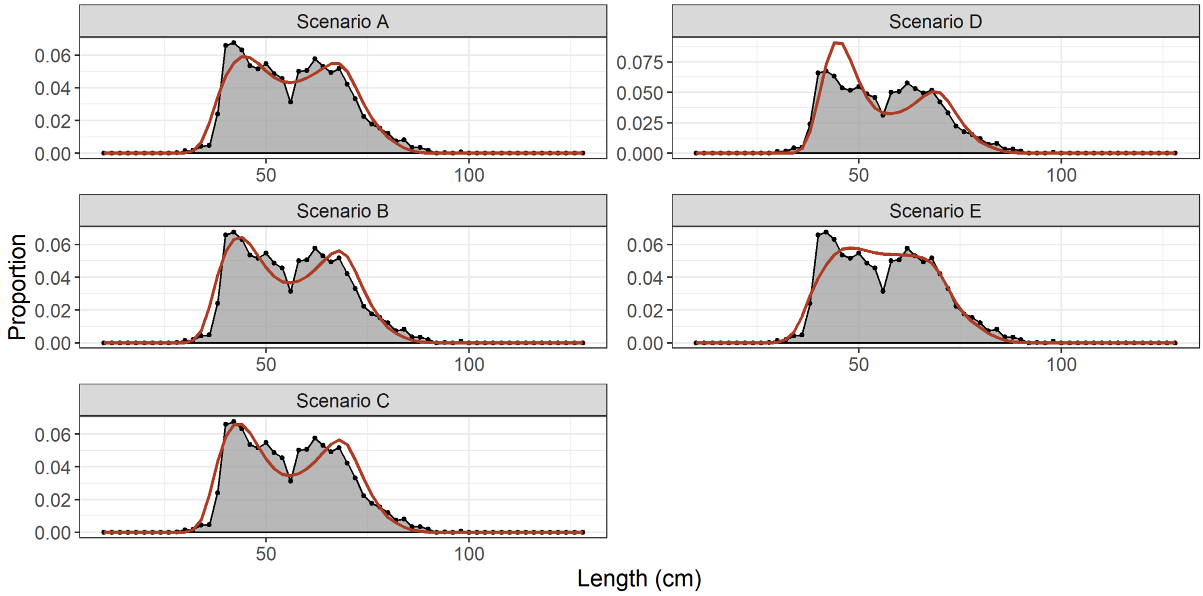

Figure B.17: Aggregated fits to recreational length data, from additional recreational selectivity scenarios

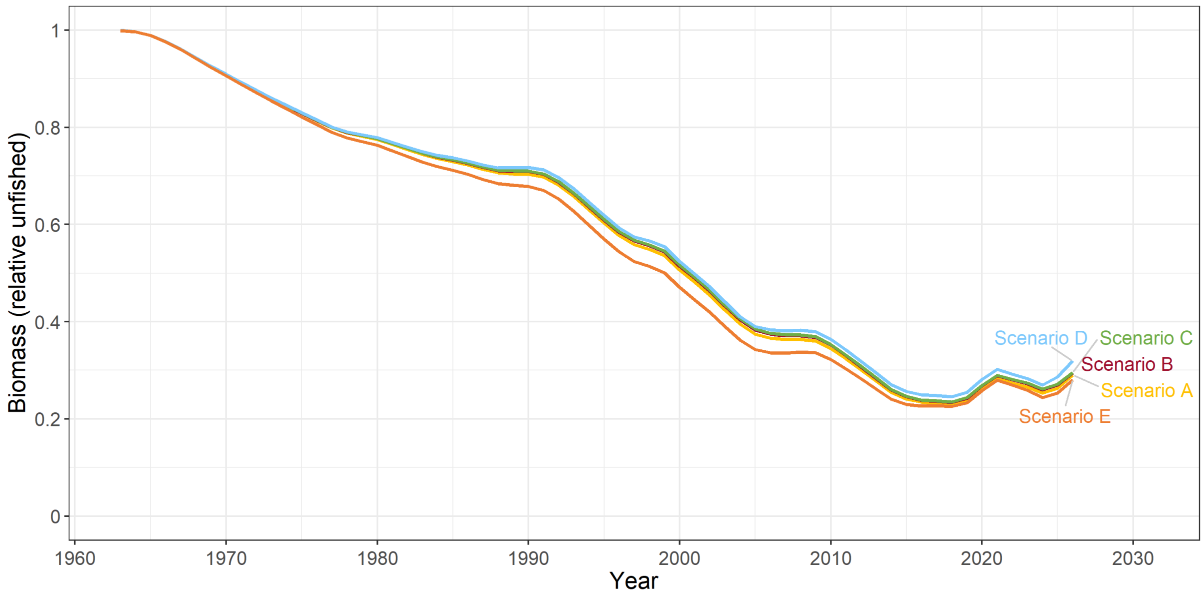

Figure B.18: Predicted spawning biomass trajectory relative to unfished for Queensland east coast saddletail snapper, from additional recreational selectivity scenarios

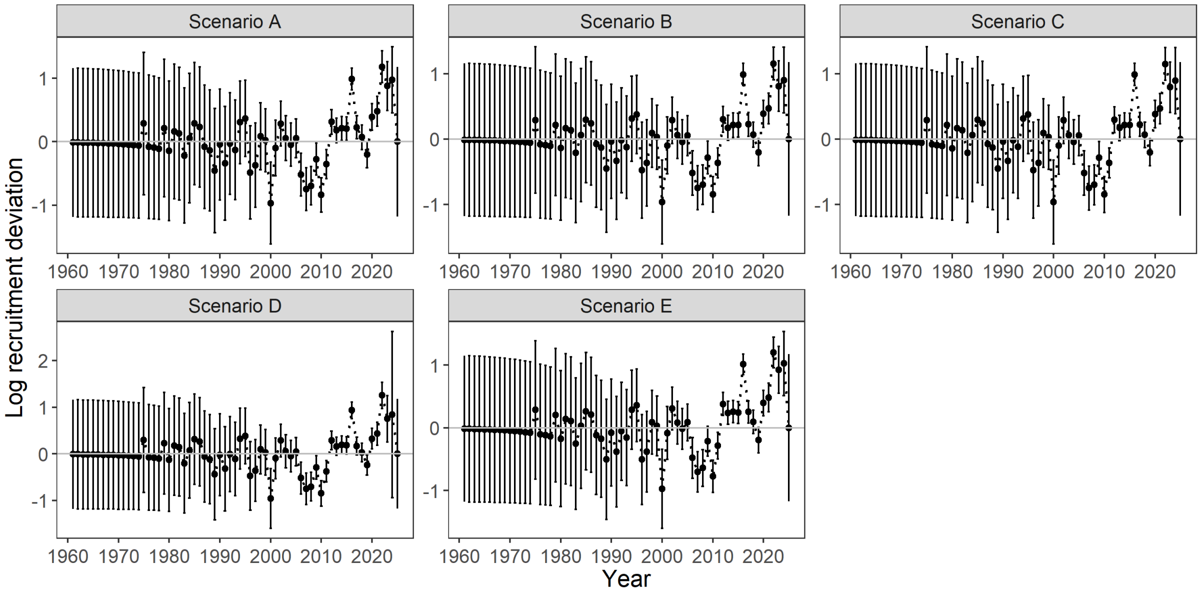

Figure B.19: Recruitment deviation estimates, from additional recreational selectivity scenarios

B.7 Interaction between catch rate data, recruitment deviations, h and M

Additional sensitivities were run to explore the effect of removing the CPUE data input, and fixing steepness and natural mortality parameters to extreme values.

Sensitivities were run across four dimensions:

- steepness (0.75, 0.95)
- natural mortality (0.11, 0.14, 0.18, 0.22, 0.26, 0.30 yr^{-1})
- CPUE (included or not)
- start year for recruitment deviation estimates (1961 or 1985 when the oldest fish was born).

Figure B.20 shows that the removal of CPUE didn't have an impact on biomass when natural mortality was unrealistically large (i.e. 0.26 or 0.3 yr^{-1}). These values are considered unrealistic as they correspond to maximum ages of 21 or 18 years respectively, whereas the observed data show saddletail snapper can live up to 49 years of age. The combination of low natural mortality and removal of CPUE causes the stock to crash in recent years without CPUE information. The interaction of CPUE removal combined with the steepness parameter value or start year of recruitment deviations, had little impact the final biomass estimates.

Figure B.21 shows that as natural mortality increases, the early decline in estimated recruitment deviations disappears. With very high natural mortality (higher than supported by observed data), the early period of informed recruitment deviations is above average, and the spike in later years is much less pronounced. The impact of steepness on recruitment deviations is only apparent when combined with low natural mortality.

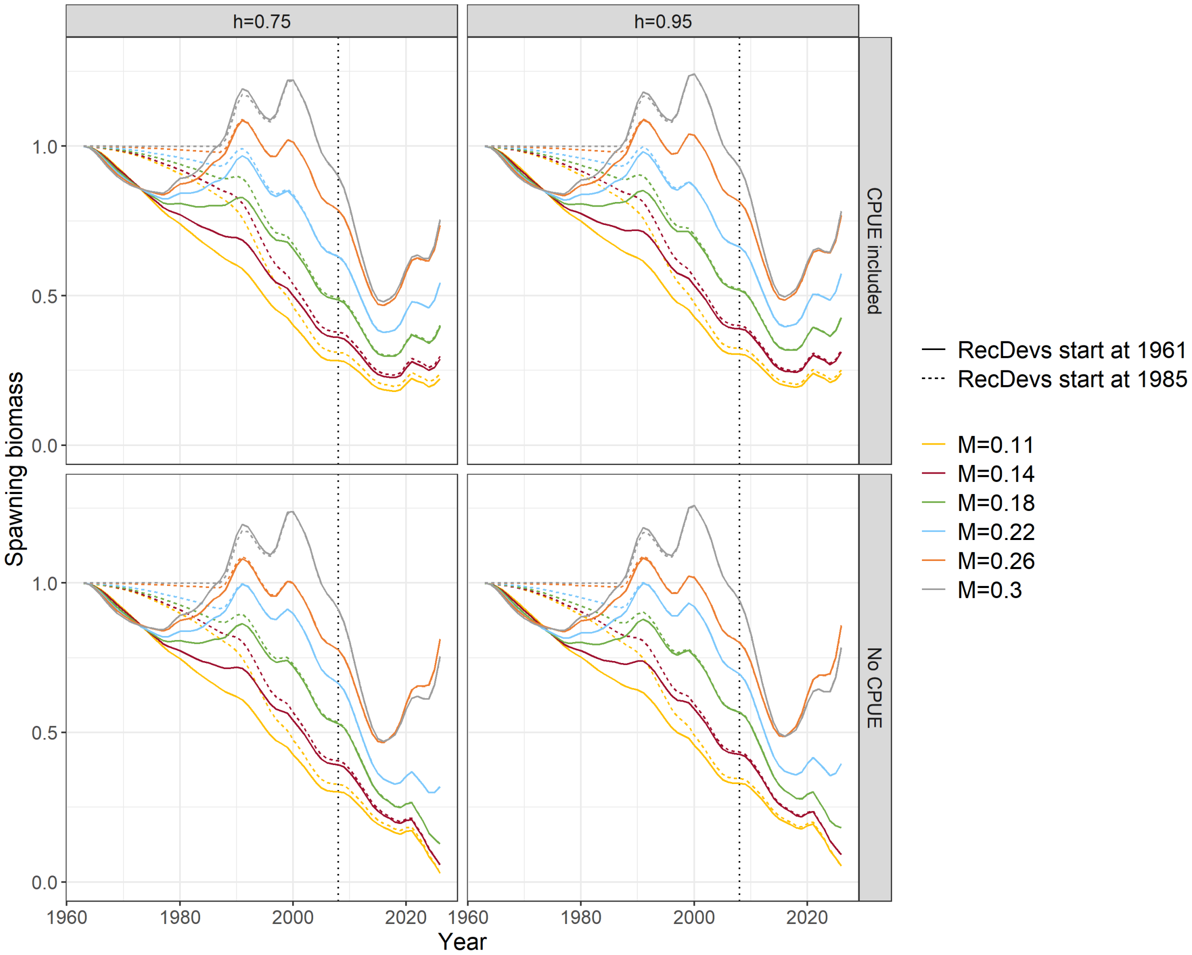

Figure B.20: Predicted spawning biomass trajectory relative to unfished for Queensland east coast saddletail snapper, from additional scenarios in which CPUE inputs were removed—the vertical line indicates where the commercial CPUE index begins

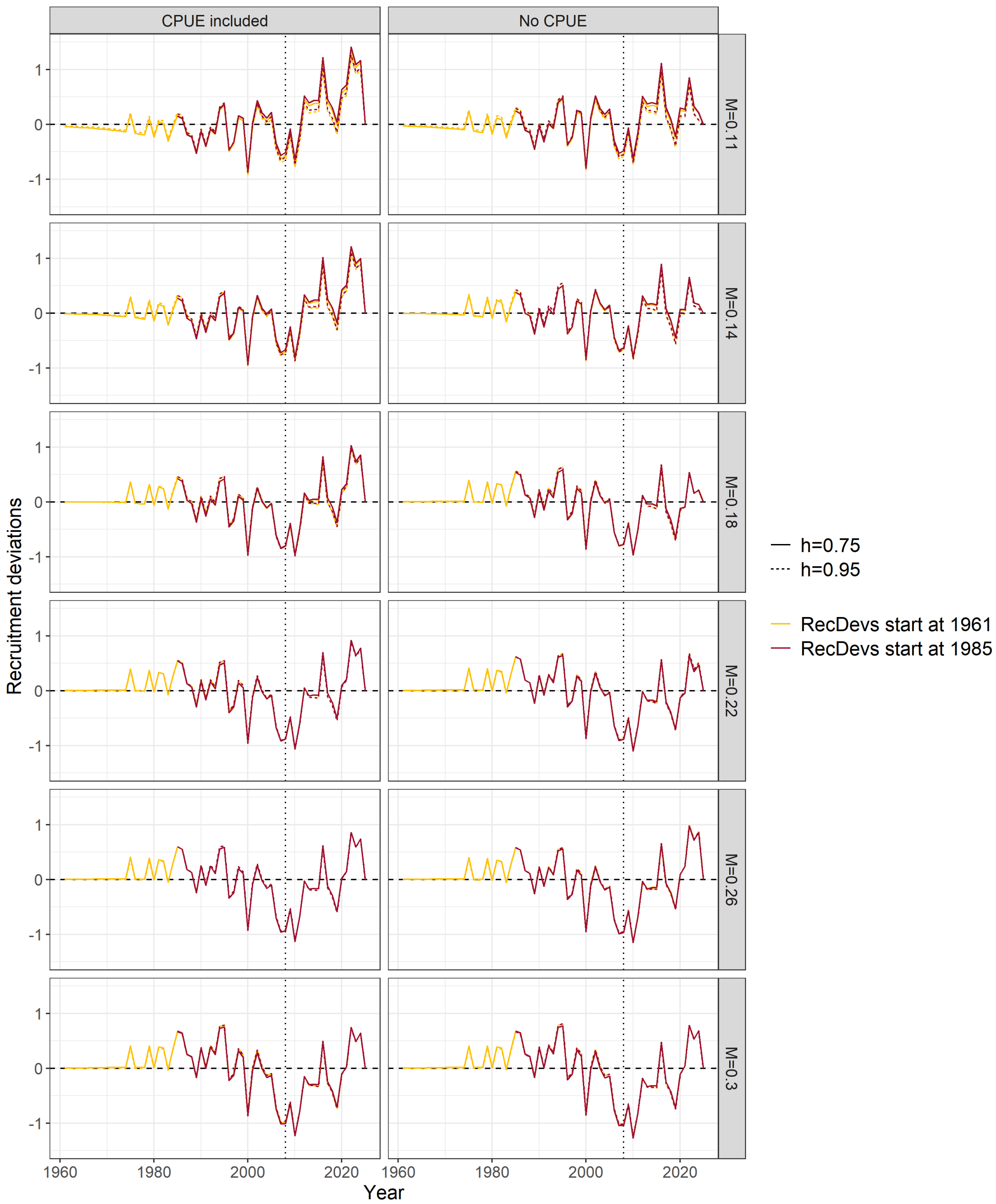

Figure B.21: Recruitment deviation estimates, from additional scenarios in which CPUE inputs were removed—the vertical line indicates where the commercial CPUE index begins

B.8 Commercial catch rates

B.8.1 Time span

Incorporating historical catch rates provides a longer-term perspective on stock trends, improving the model's ability to estimate historical abundance and fishing impacts. However, poor-quality historical data can introduce bias, and may cause the model to infer abundance trends erroneously, impacting biomass estimates.

The project team decided to limit the time span of commercial catch rates to 2008 to 2025 (inclusive) (see Section A). This was to reduce the effect of amplifying spurious signal in catch rates from 1997 to 2004 that was not abundance-related, due to fishery changes and reporting patterns. Leaving these data in the model risks them being incorrectly attributed to a shift in abundance. This decision was discussed further in Section 4.4.2.

Figure B.22 shows the two alternatives explored for the time scale of commercial catch rates: 1997–2025 and 2008–2025.

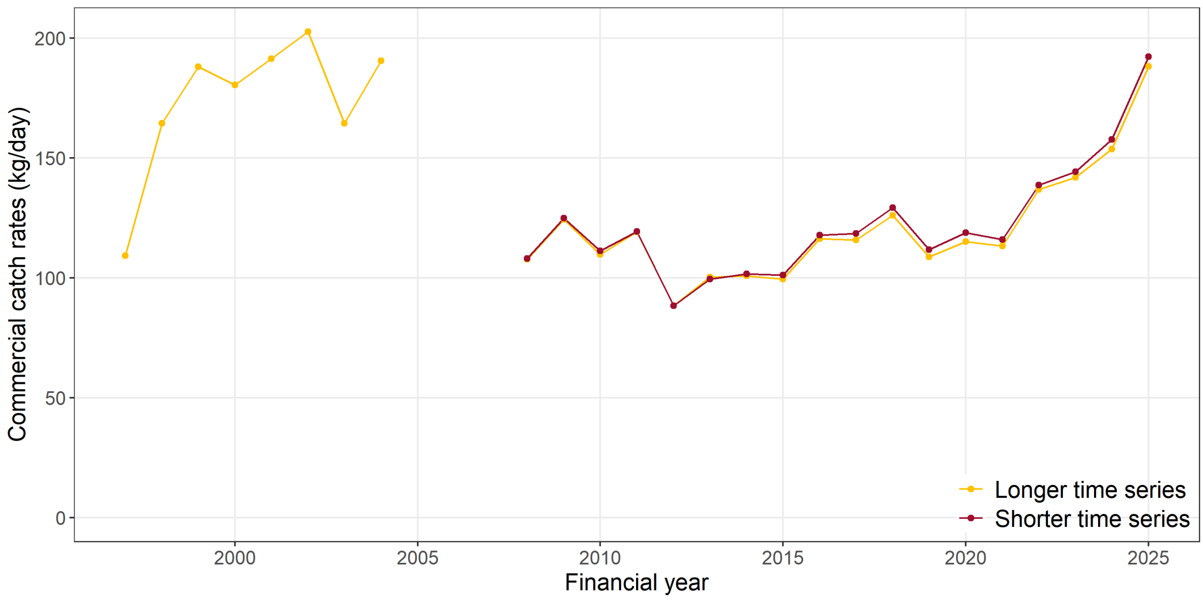

Figure B.22: Impact of time span on standardised commercial catch rates

B.8.2 Targeting

A definition for targeting was defined in Section 2.4. Alternative definitions were explored, as outlined here.

For the different targeting scenarios, no further filters were applied beyond those defined in Section 2.4.1. Instead, different ways of defining a targeting covariate in the model were explored:

- "Percent catch saddletail (above or below 70%)": Calculate the percentage of each individual catch that is saddletail and create a binary variable (i.e. '1' if >70%, '0' if <70%). This was chosen as the base case.
- "Percent catch saddletail (0-30%, 30-70%, 70-100%)": Calculate the percentage of each individual catch that is saddletail and create a ternary variable.
- "Percent catch saddletail (above or below 50%)": Calculate the percentage of each individual catch that is saddletail and create a binary variable.
- "Stephens and MacCall analysis": Using the analysis defined in Stephens and MacCall (2004) to determine if a trip targeting saddletail and create a binary variable.

- “Habitat and co-caught species”: If more kilograms of inter-reef fish (saddletail, crimson, red emperor or jobfish) caught than reef fish (coral trout or red throat emperor) on a trip, the fisher is considered to be targeting saddletail.

Figure B.23 shows that the method used to define targeting did not play a major role in the commercial catch rate trend.

Figure B.24 shows a breakdown of combinations of commonly co-caught species in the commercial logbook data. This plot was one of the tools used to develop the ‘habitat and co-caught species’ decision rule.

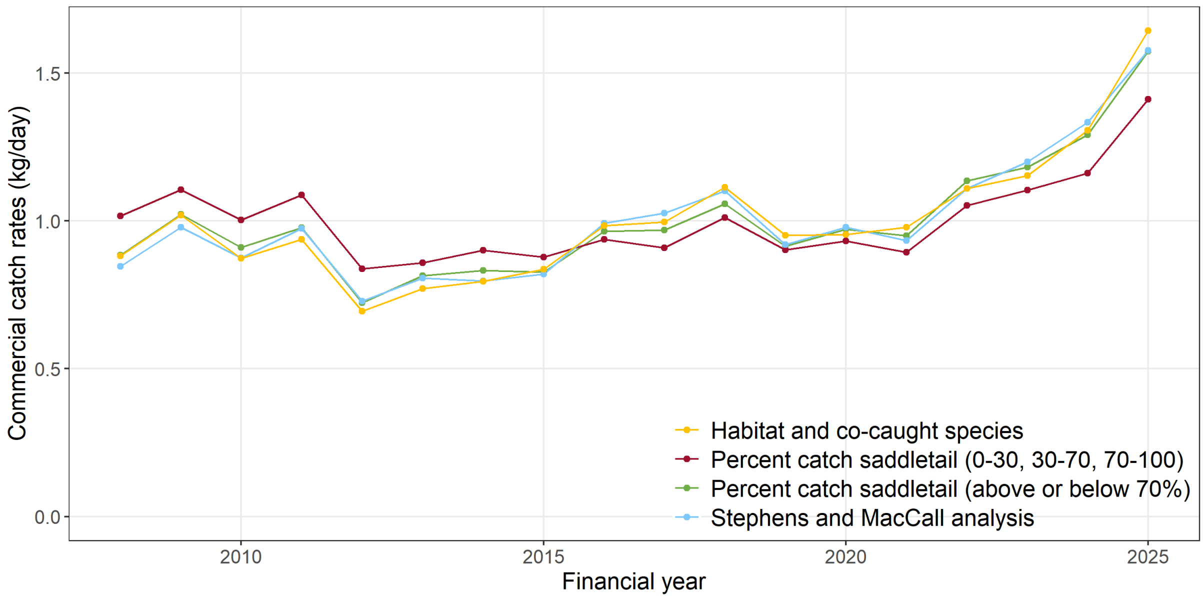

Figure B.23: Impact of targeting definition on standardised commercial catch rates

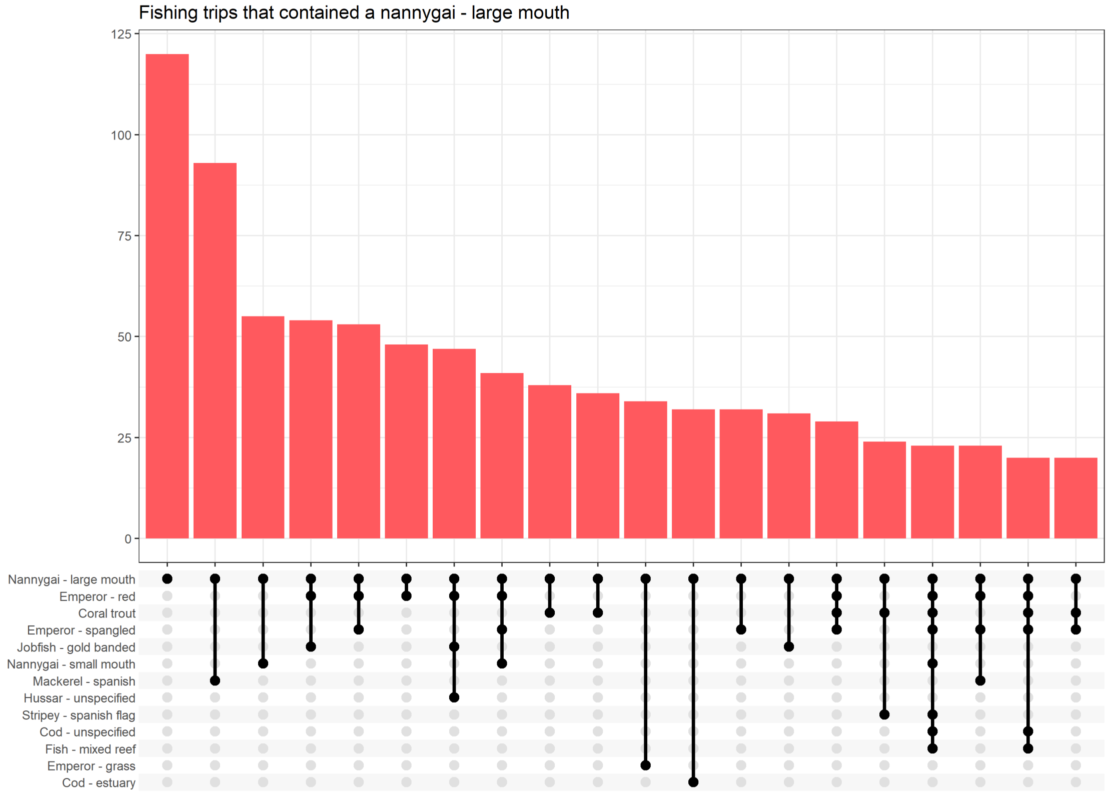

Figure B.24: Most common combinations of species caught with saddletail snapper, in the commercial logbook data

B.8.3 Wind speed

Figure B.25 shows the impact that different thresholds for 'wind' variable have on the standardiation of commercial catch rates.

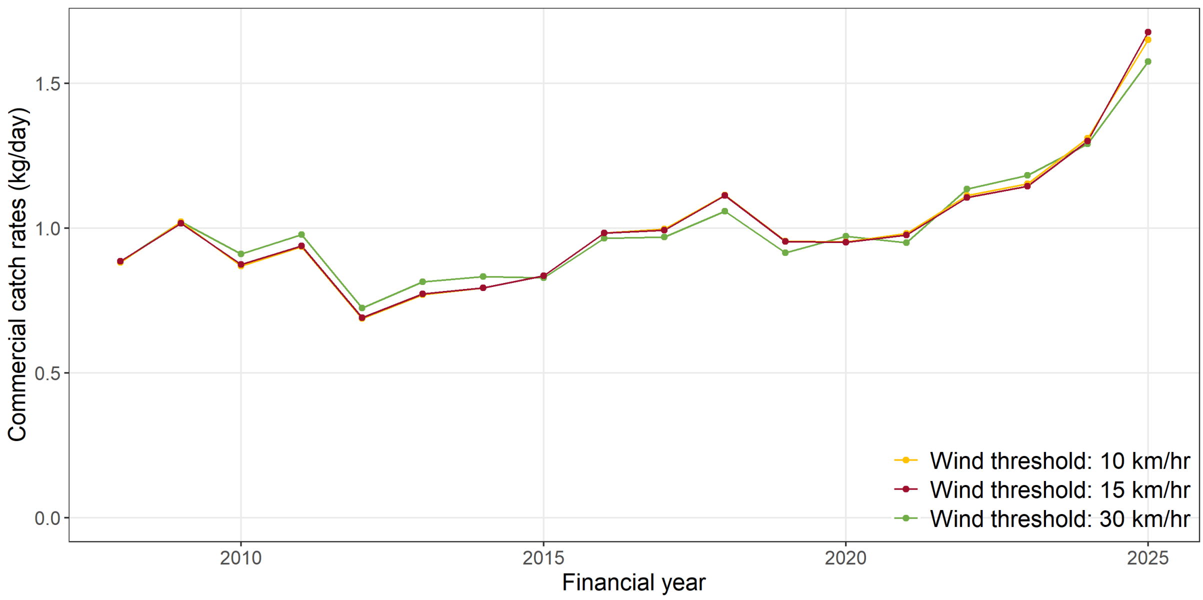

Figure B.25: Impact of wind speed threshold on standardised commercial catch rates

C Diagnostics for standardised indices of abundance

C.1 Commercial catch rates

C.1.1 Model performance and outputs

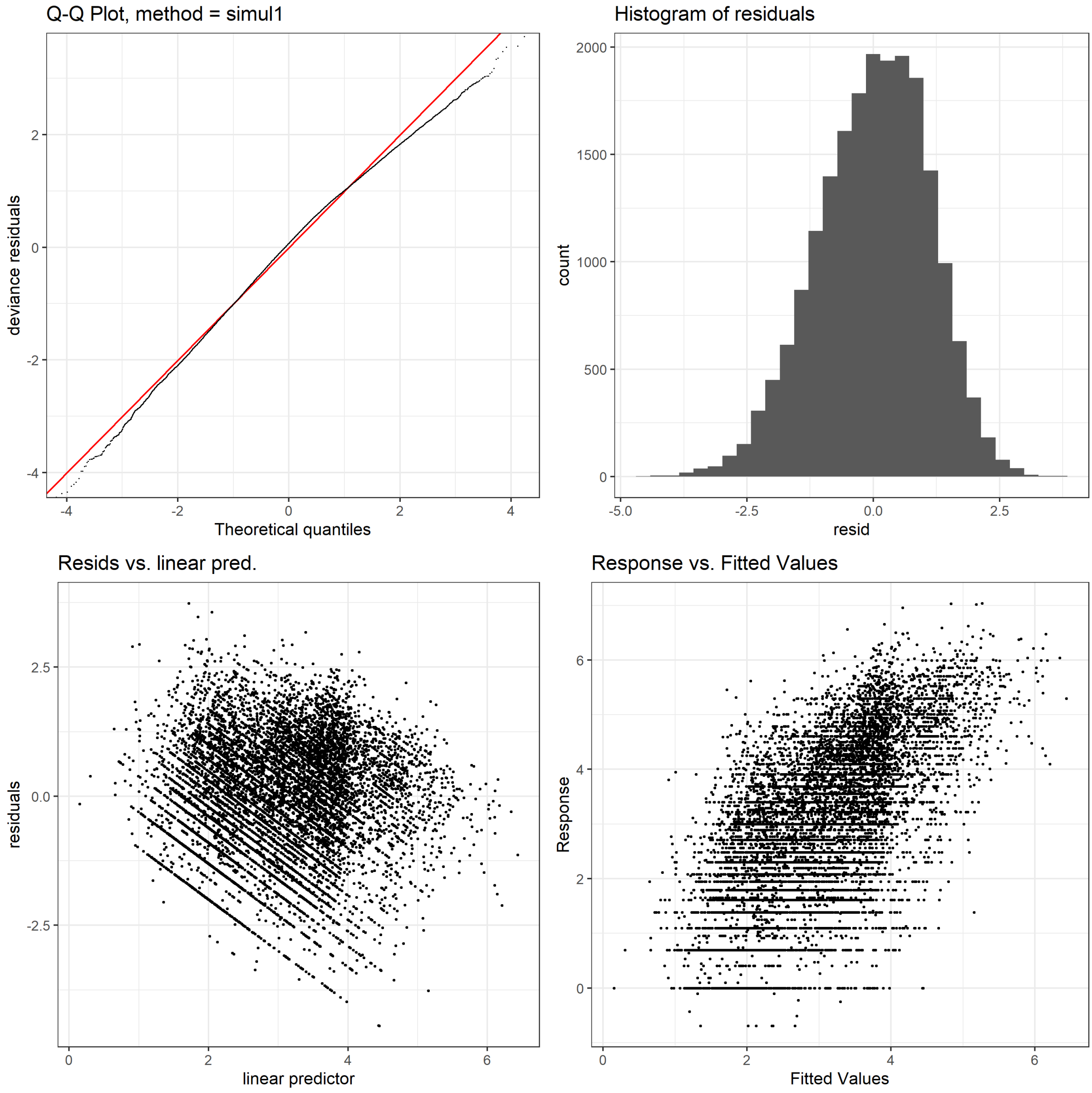

Figure C.1: Analysis of residuals for the commercial catch rate model

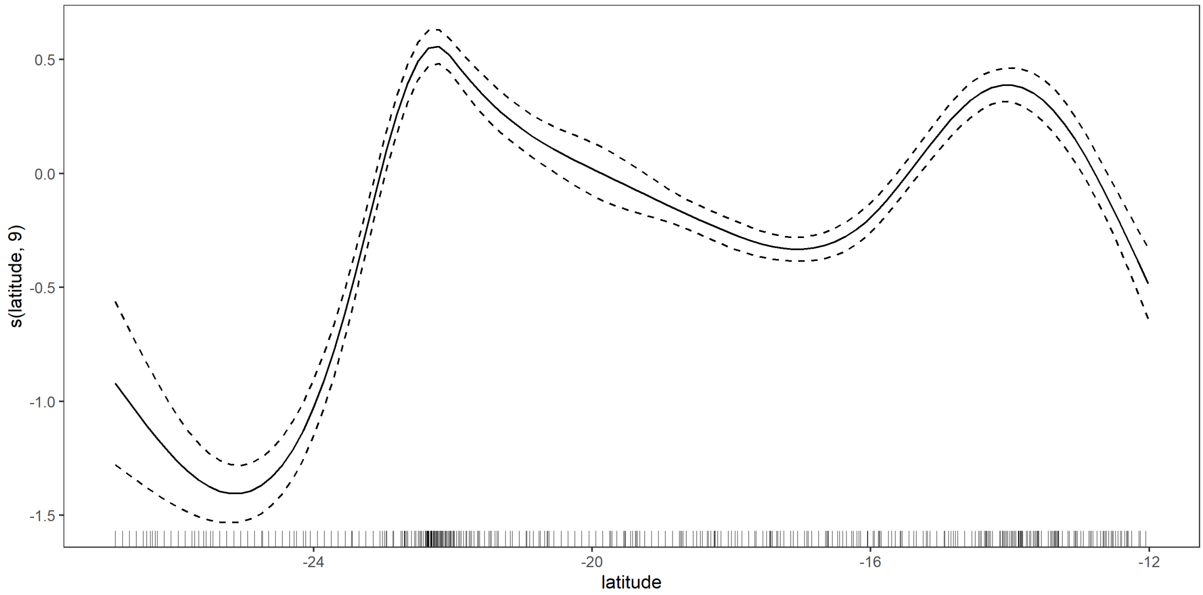

Figure C.2: Commercial catch rate model effect plot—latitude

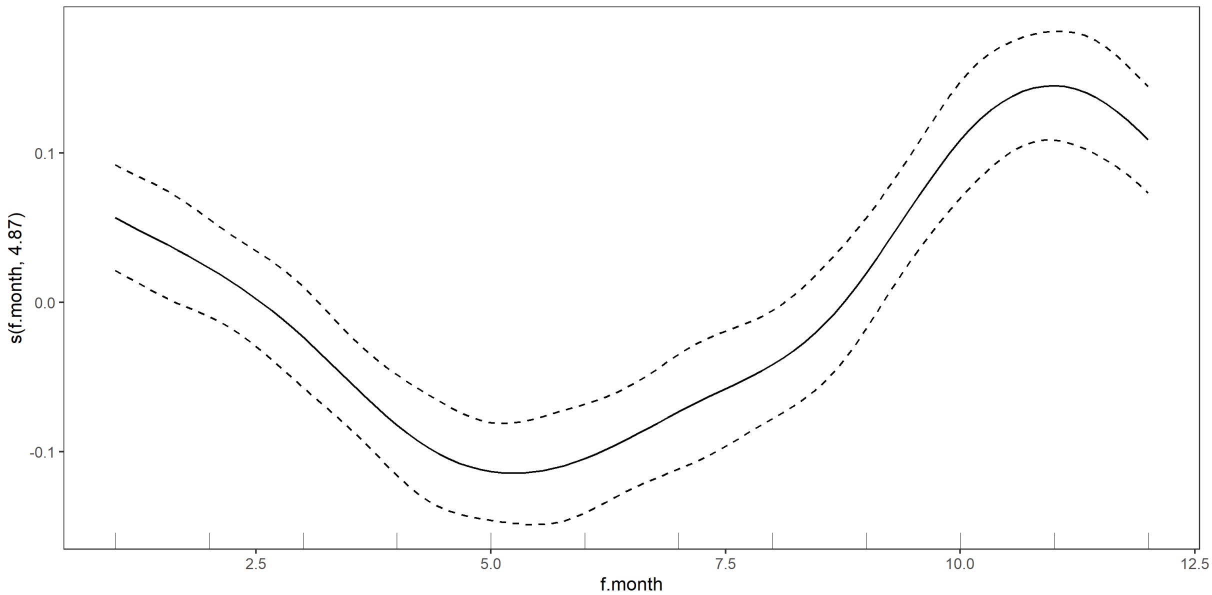

Figure C.3: Commercial catch rate model effect plot—month

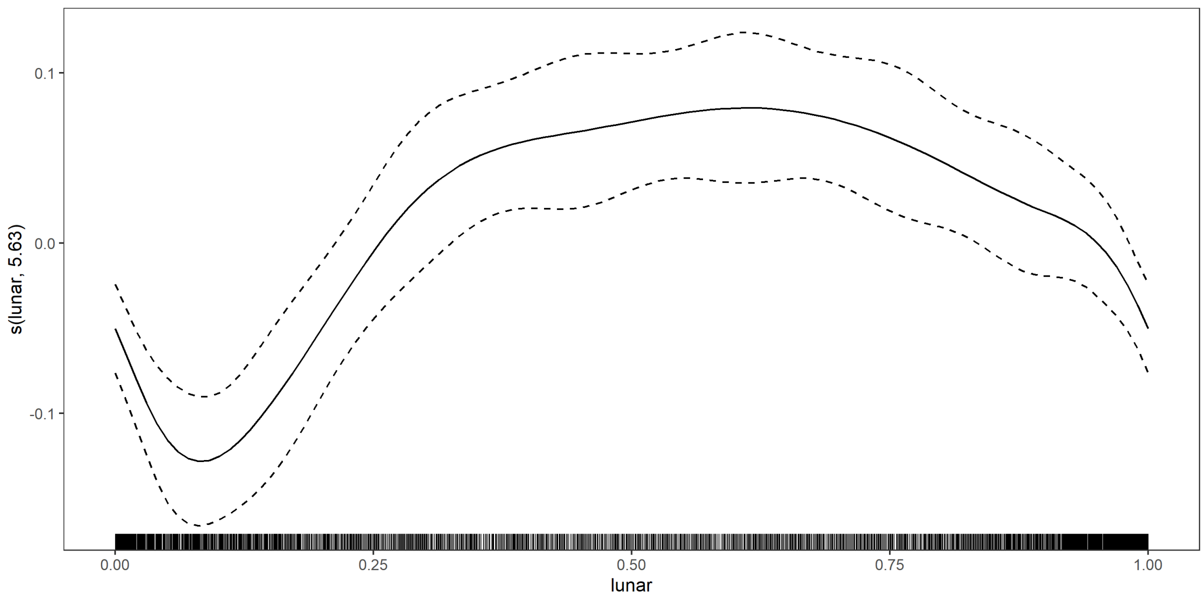

Figure C.4: Commercial catch rate model effect plot—lunar

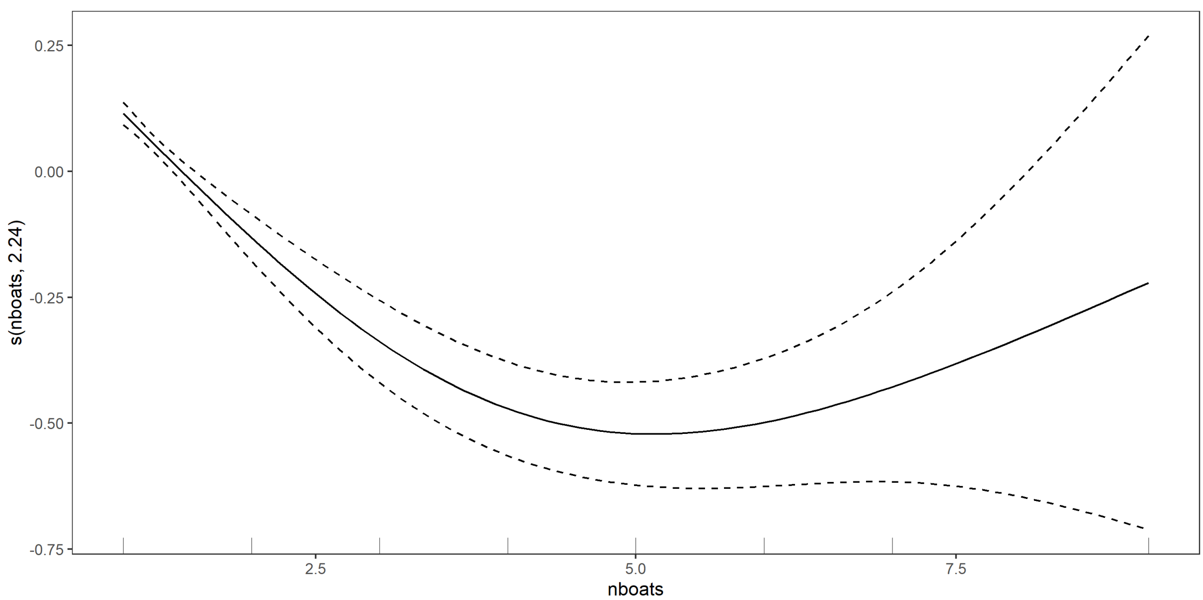

Figure C.5: Commercial catch rate model effect plot—number of boats

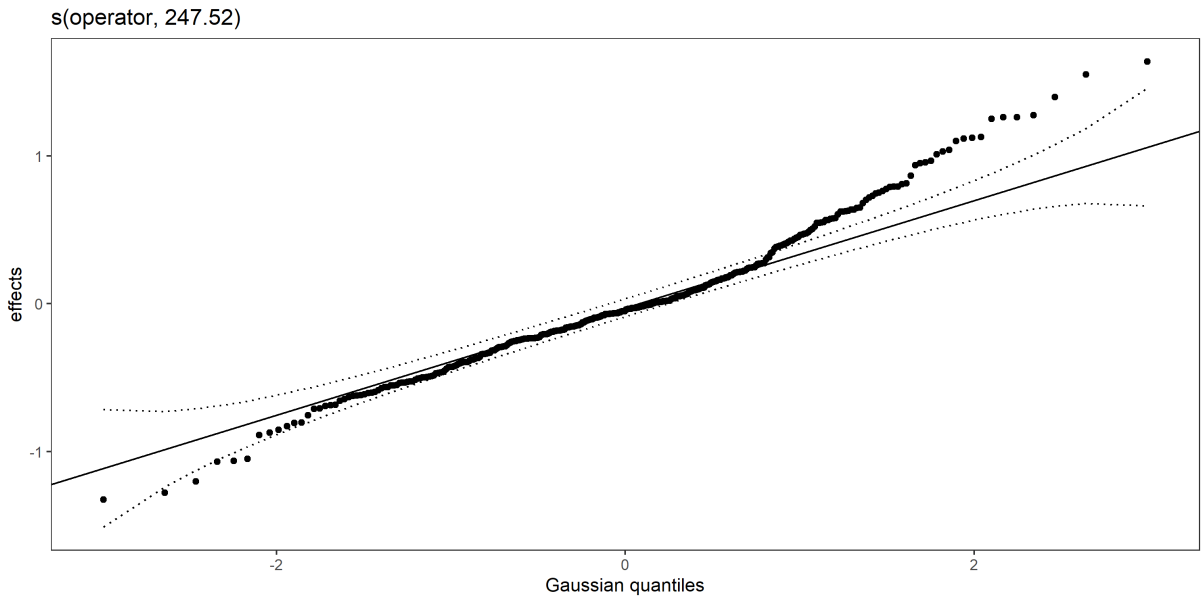

Figure C.6: Commercial catch rate model effect plot—fisher

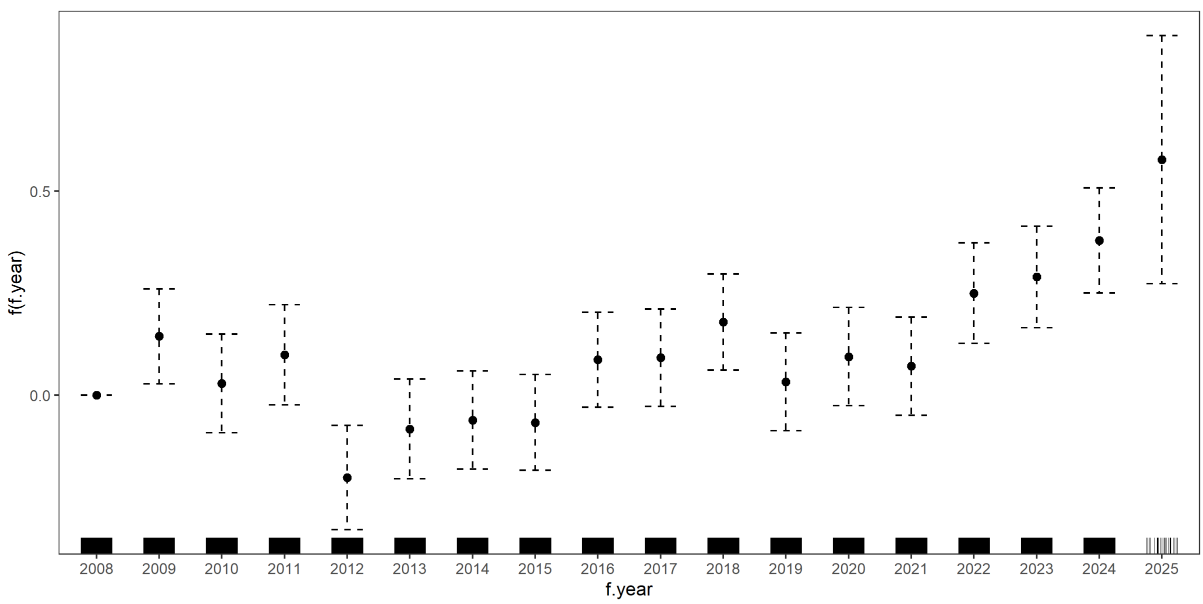

Figure C.7: Commercial catch rate model effect plot—year

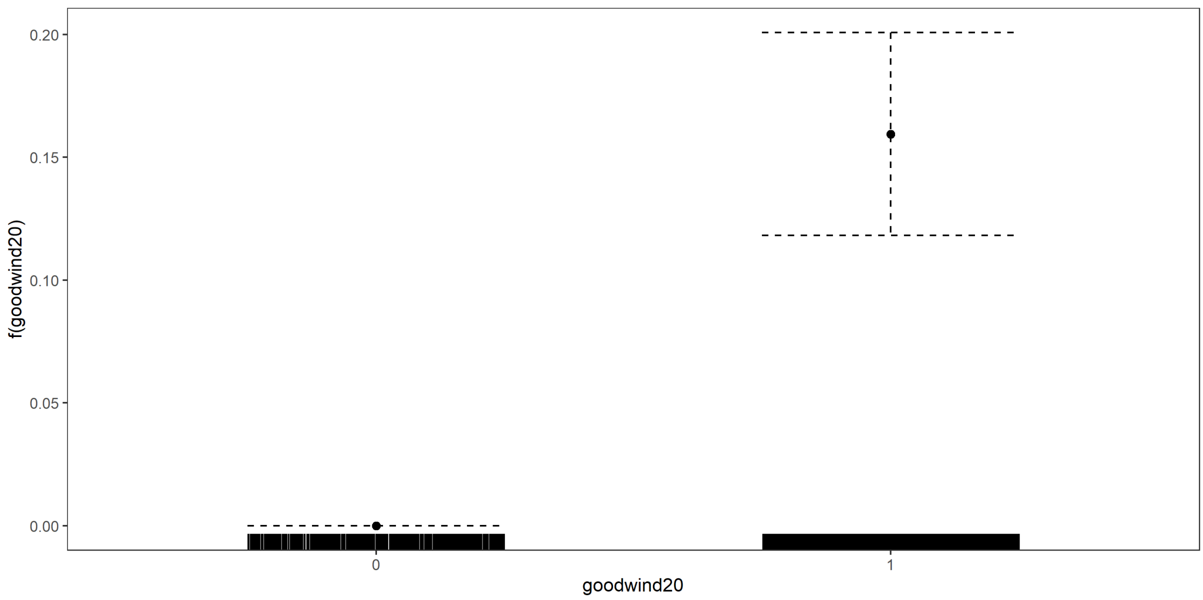

Figure C.8: Commercial catch rate model effect plot—wind

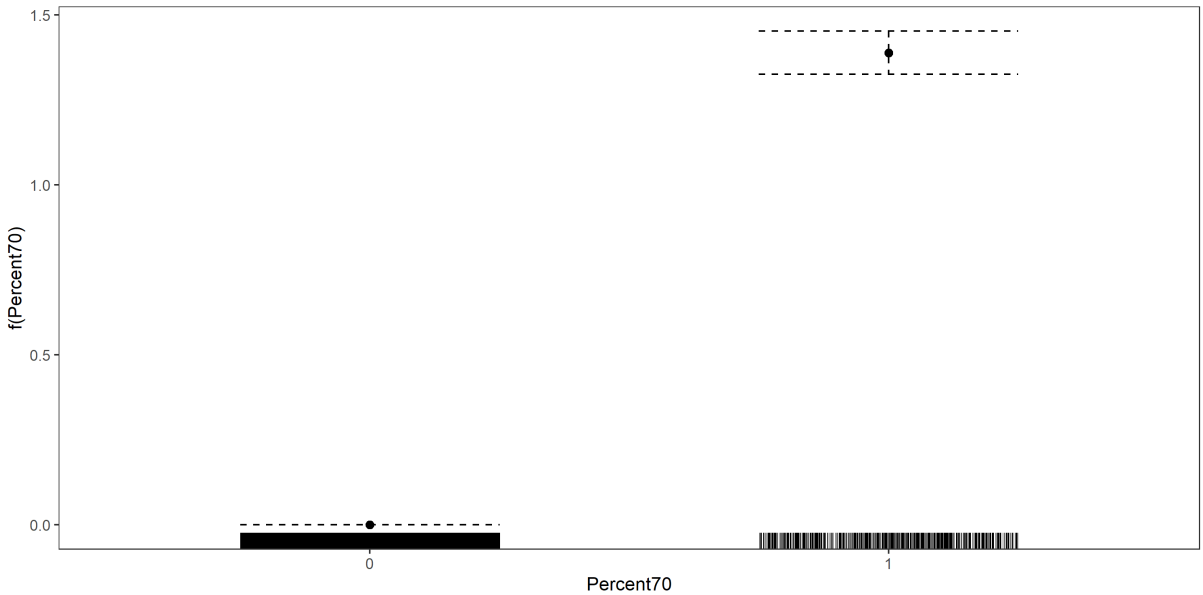

Figure C.9: Commercial catch rate model effect plot—targeting

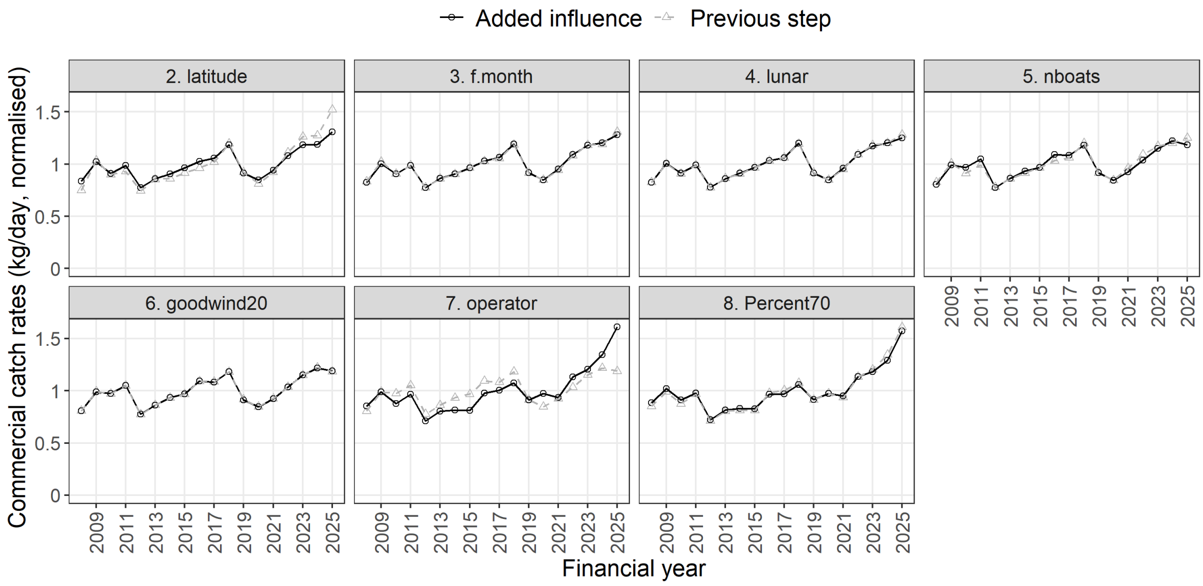

Figure C.10: Influence plot for terms added to standardisation model for commercial catch rates

C.2 Recreational catch rates

C.2.1 Model performance

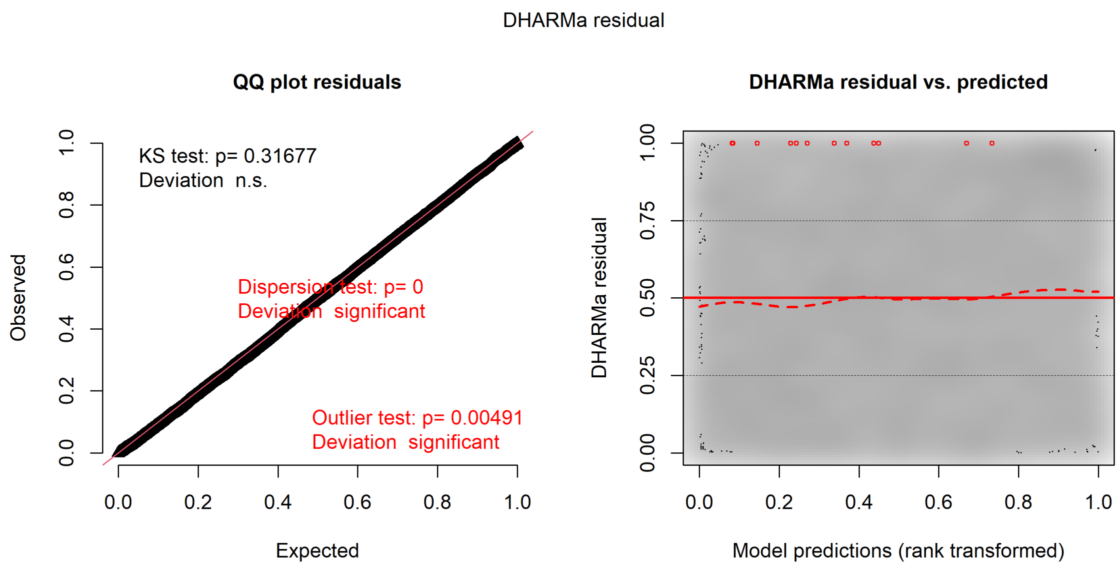

Figure C.11: Analysis of residuals for the recreational catch rate model

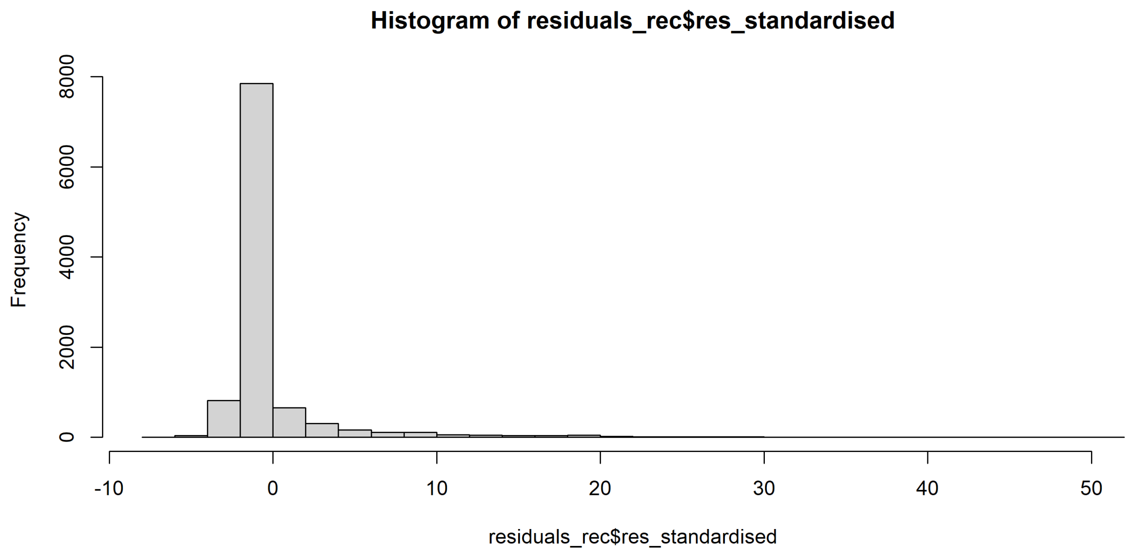

Figure C.12: Analysis of residuals for the recreational catch rate model

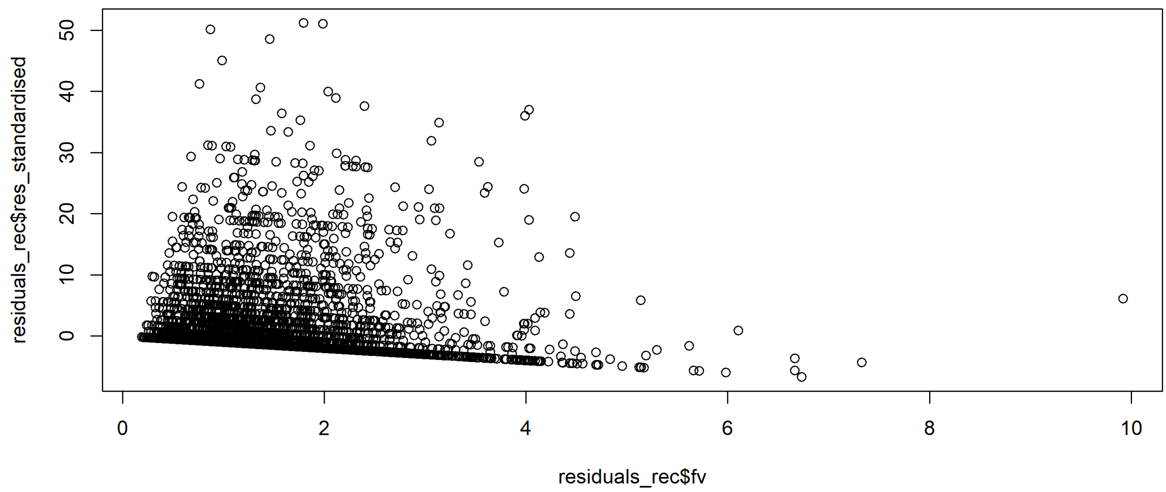

Figure C.13: Analysis of residuals for the recreational catch rate model

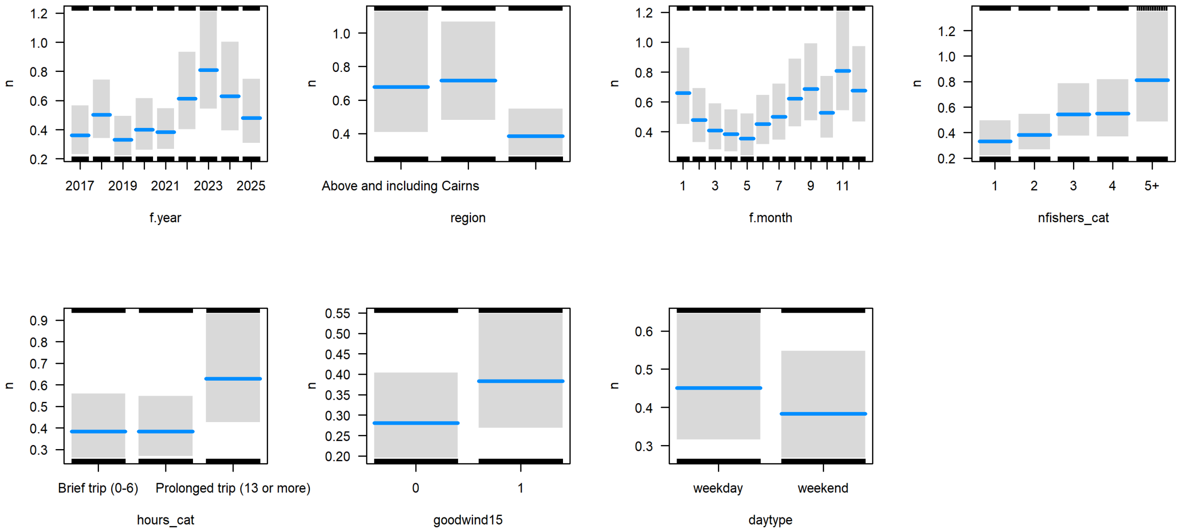

Figure C.14: Recreational catch rate model effect plot

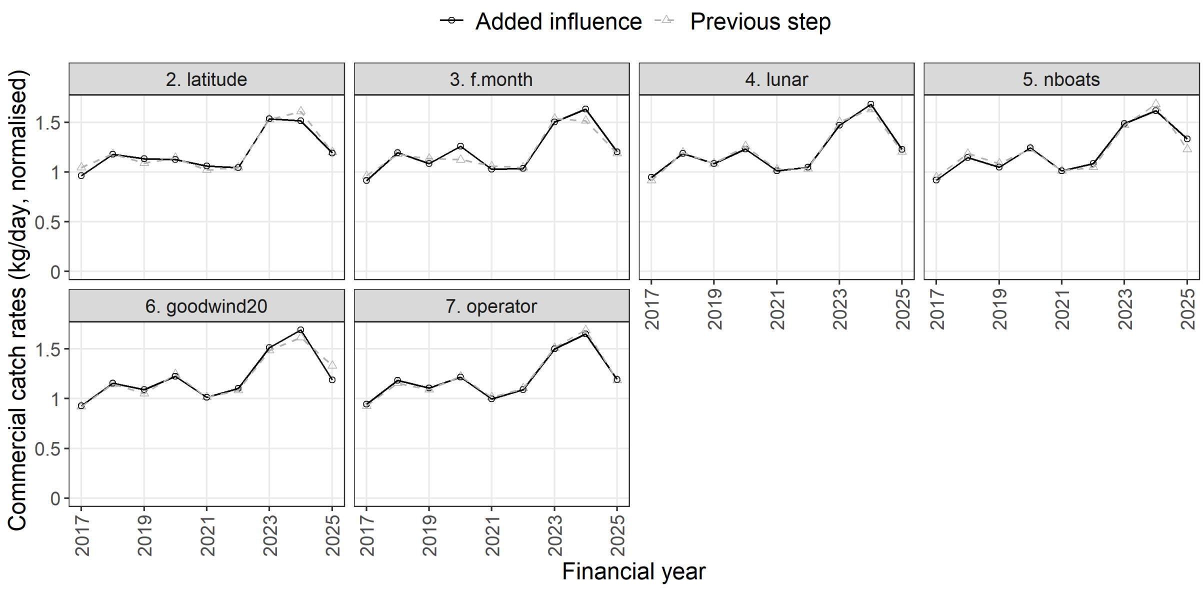

Figure C.15: Influence plot for terms added to standardisation model for recreational catch rates

D Model inputs

D.1 Initial weighting of length and age data

Weighting of length and age data, as input into the model, are shown in Table D.1.

Year	Age	Length: Unweighted		Length: Method 1		Length: Method 2	
		Commercial	Recreational	Commercial	Recreational	Commercial	Recreational
2018	917	4498	470	108	29	99	185
2019	928	5018	391	132	26	187	85
2020	775	2727	382	94	29	105	135
2021	687	3593	441	103	32	63	154
2022	702	2614	392	39	27	37	140
2023	768	2239	697	51	40	63	200
2024	726	2223	446	46	33	27	124
2025	320	2242	261	40	23	32	102

Table D.1: Annual weighting of length and age data sample sizes

D.2 Sampling of biological modelling data

Fisheries Queensland commenced routine, fishery-dependent biological data collection for saddletail snapper in 2018. The program's primary objective is to gather representative data about the length, sex and age of retained fish. For commercially caught fish, data are gathered through voluntary cooperation from commercial fishers and fish processors. These businesses facilitate data collection by providing access to fish within the supply chain and measuring or supplying samples from their own catches. Data from recreational and charter catches also involves voluntary cooperation, to allow staff to measure fish at boat ramps, or fishers to provide samples of their own catches that can be processed in the laboratory. The majority of saddletail snapper samples obtained on Queensland's east coast are contributed by the commercial fishery. Substantial sampling of the recreational sector also occurs. Some data are also available from the charter sector, though sample sizes are limited.

The primary sampling unit is the "catch", which comprises fish from an individual fishing session on a single day or spanning several days, by one fishing operation. The program is designed to collect data from the fishery that is representative of the full spatial extent of the fishery, by setting targets for the number of commercial catches to be sampled in each spatial stratum per year, and for the number of recreational fishing surveys to undertake at key boat ramps throughout a year. For commercial sampling, these targets are divided among the sampling regions, based on regional sample size analysis and recent catch reported in commercial fishing logbooks.

Fishing sector, catch date, location and fishing method are recorded for each catch. Location can be reported at various spatial resolutions, including monitoring region, 30x30 minute CFISH grid reference, or to the specific marine location (e.g. reef name). This recording preserves the confidentiality of the exact fishing location if the fisher wishes, whilst enabling the data collected to be aggregated to a suitable spatial scale.

Fish size is recorded as fork length (in cm, nose to caudal fork) whenever accurate measurement is feasible. In cases where the fish is damaged (e.g. frayed caudal fins) or incomplete (e.g. head only), alternate measures are taken. In these cases, total length (nose to end of tail) or jaw length (tip of the upper jaw to the end of the maxilla) measurements are taken as required. Where a catch has been identified as size biased, it is flagged for exclusion from analyses where representative length frequency data are needed.

When a catch is very large or access is limited by time constraints, a subsample of the catch is measured. The proportion of the catch measured is recorded.

Where possible, macroscopic examination of gonads is undertaken to determine sex, relying on colour, structure and texture of the gonad. Fish are classified as male, female, or unknown sex.

Otoliths (fish ear bones) are used to estimate the age of a fish. Each year, the program aims to collect otoliths from retained fish in every observed length class within every geographic area. To prevent oversampling of frequently retained length classes, or locations with higher availability, the number of otoliths collected per 1 cm length class in each area is capped at 20 fish. Otoliths are first dried and stored, before being embedded in resin and sectioned. The thin transverse sections are then mounted on slides, magnified images of section otoliths are captured using microscope and imaging software with the annual opaque zones counted. To estimate the age of each fish, a trained reader assigns an increment count and an edge type. Age is calculated based on capture date, increment count, edge type, the expected timing of new increments, and the assumed average birth date (time of year) for all fish in the stock. Each year, readers undergo training and testing on a reference collection of otoliths, before undertaking the current year's otolith reading this prevents bias and drift of increment counts over time. Annual reads must meet thresholds for precision for increment counts and edge assignment to be accepted.

Each year, both the number of catches sampled and the quantity of biological data collected vary. Sampling is heavily influenced by participation from fishers and processors, as well as by logistical considerations regarding access to the fish. The form in which fish are sold (e.g. whole, gilled and gutted, filleted) also significantly influences what data can be collected. All of these factors vary between years, seasons and regions.

D.3 Conditional age-at-length

Age data were input to the population model in the form of conditional age-at-length data (Figures D.1 and D.2).

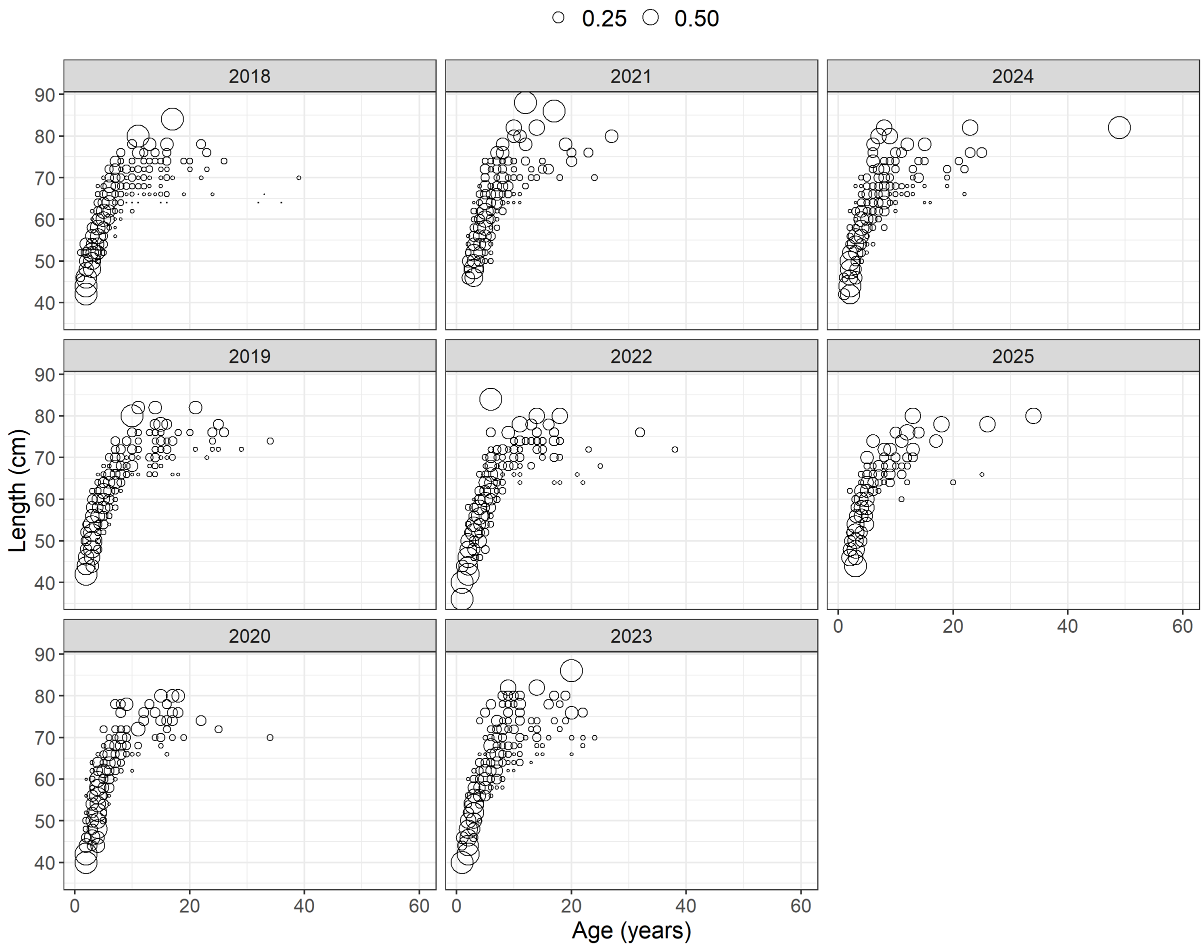

Figure D.1: Conditional age-at-length compositions for female saddletail snapper between 2018 and 2025—circle size is proportional to relative sample size in each bin across rows (i.e. for a given length bin)

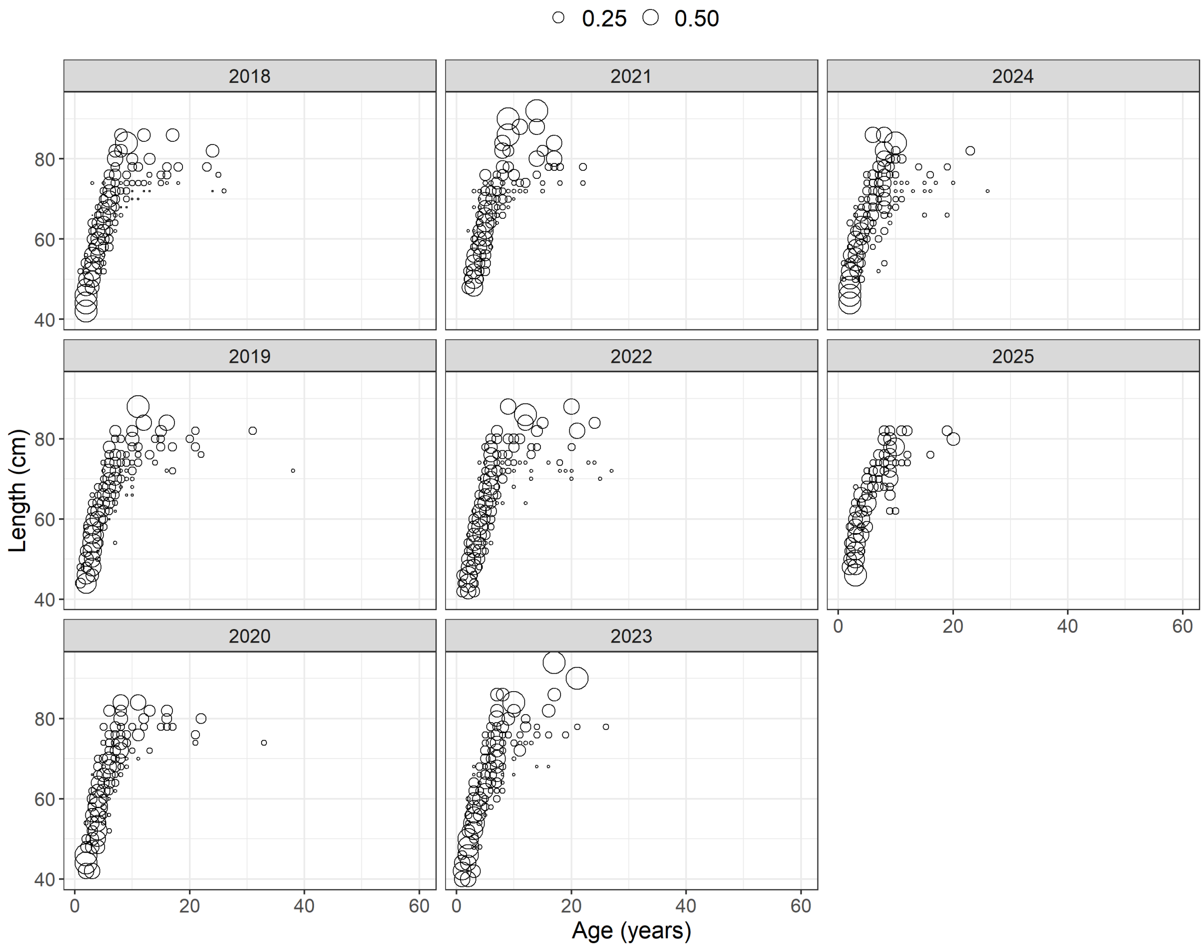

Figure D.2: Conditional age-at-length compositions for male saddletail snapper between 2018 and 2025—circle size is proportional to relative sample size in each bin across rows (i.e. for a given length bin)

D.4 Biological data

D.4.1 Weight and length

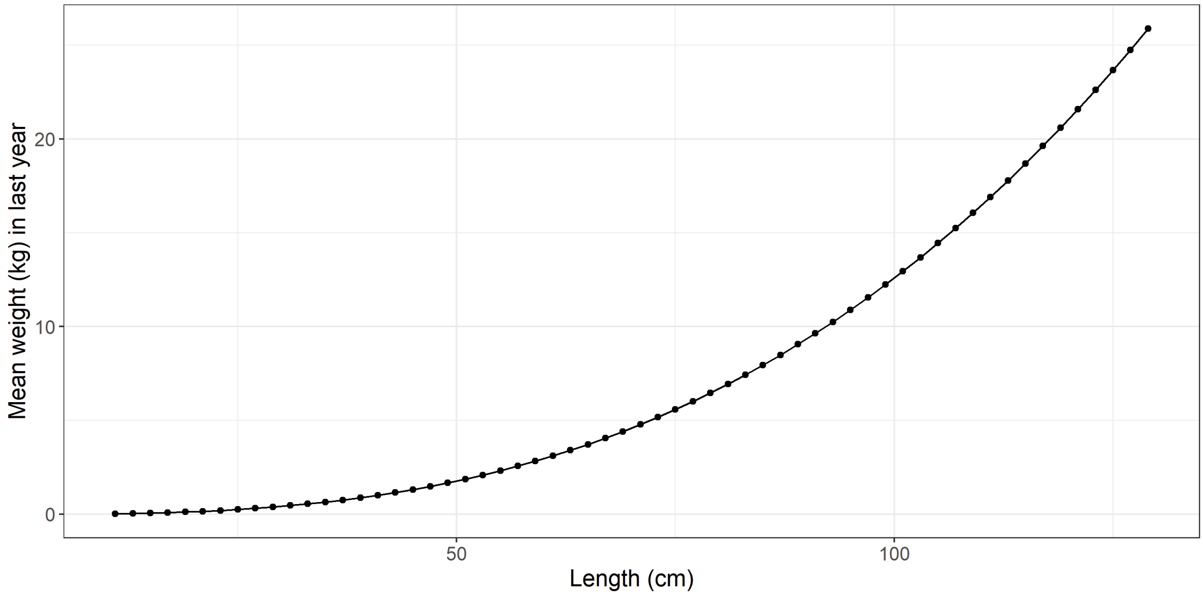

Figure D.3: Weight–length relationship for Queensland east coast saddletail snapper

D.4.2 Fecundity and maturity

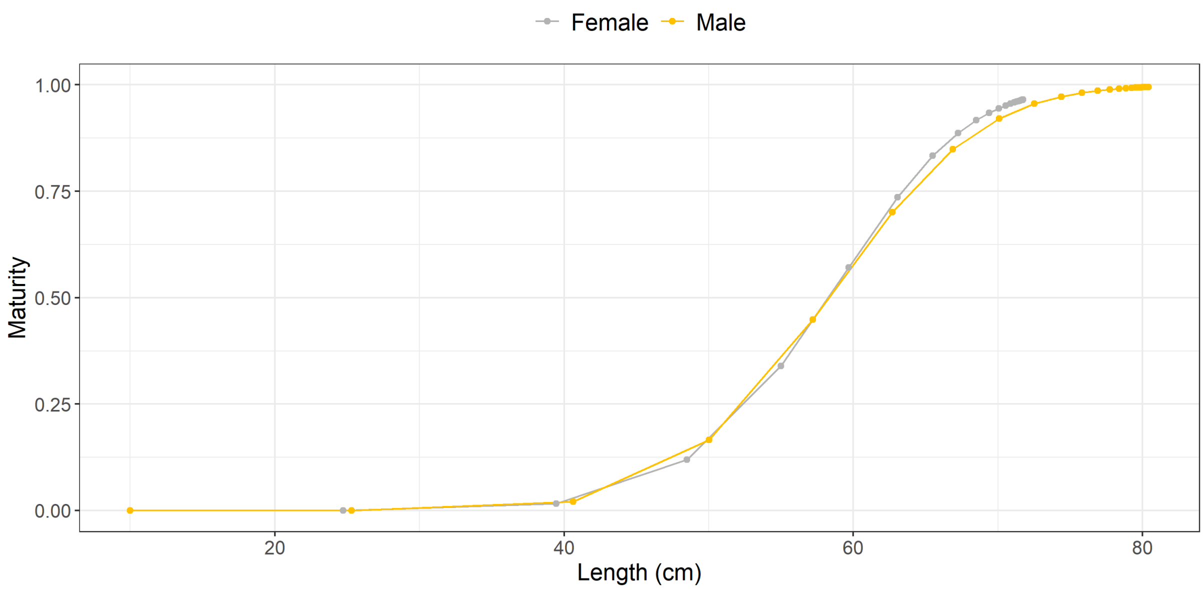

Figure D.4: Maturity at length for Queensland east coast saddletail snapper

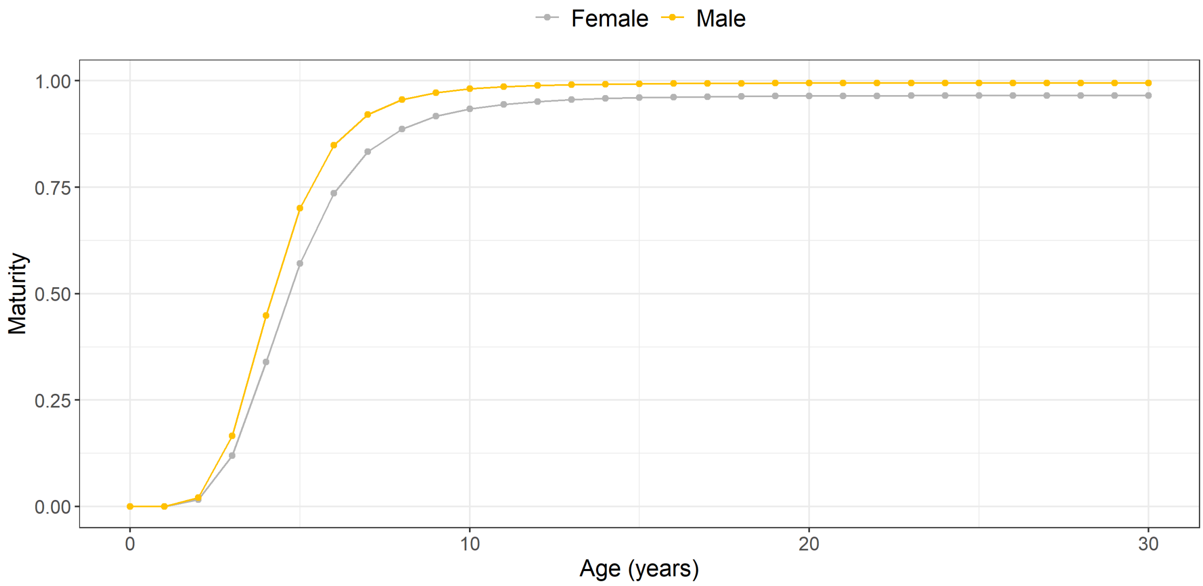

Figure D.5: Maturity at age for Queensland east coast saddletail snapper (transformed from maturity-at-length curve, using the estimated growth curve)

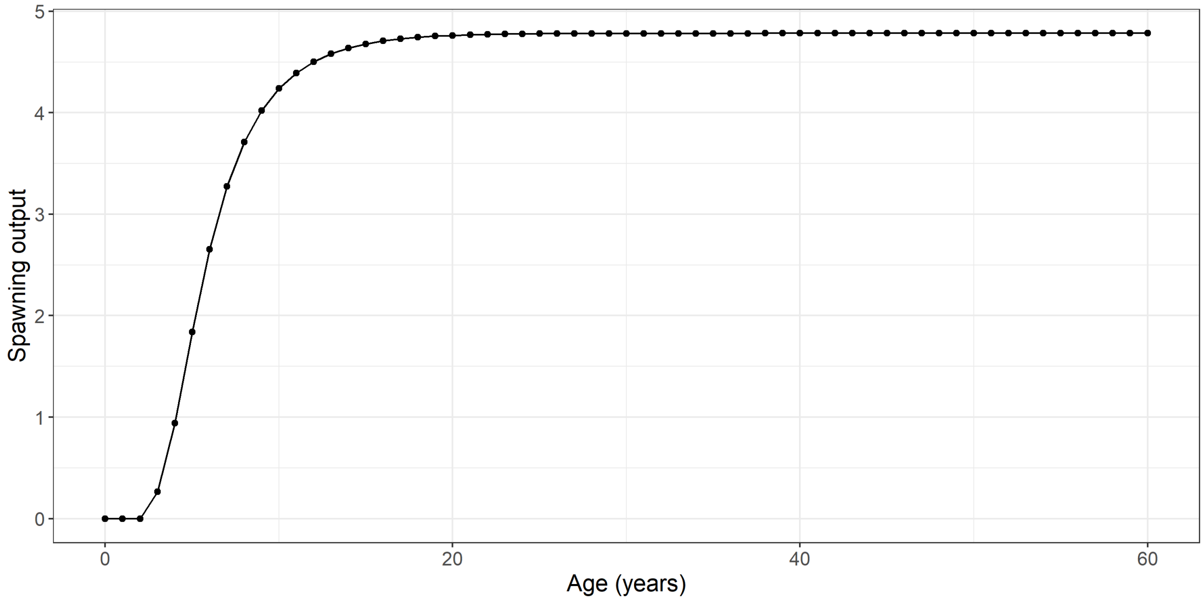

Figure D.6: Spawning output (maturity multiplied by fecundity) at age for Queensland east coast saddletail snapper

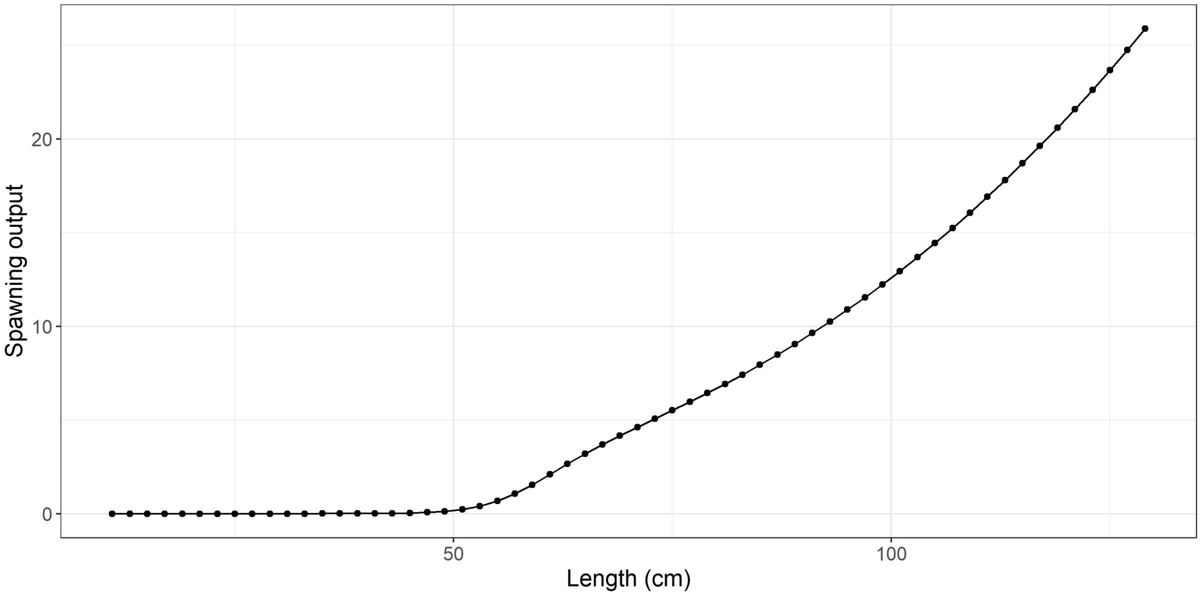

Figure D.7: Spawning output (maturity multiplied by fecundity) at length for Queensland east coast saddletail snapper

D.4.3 Spatiotemporal patterns in age and length

Figures D.8 to D.11 show the GAM model terms for Equations 2.3, 2.4, 2.5 and 2.6 respectively.

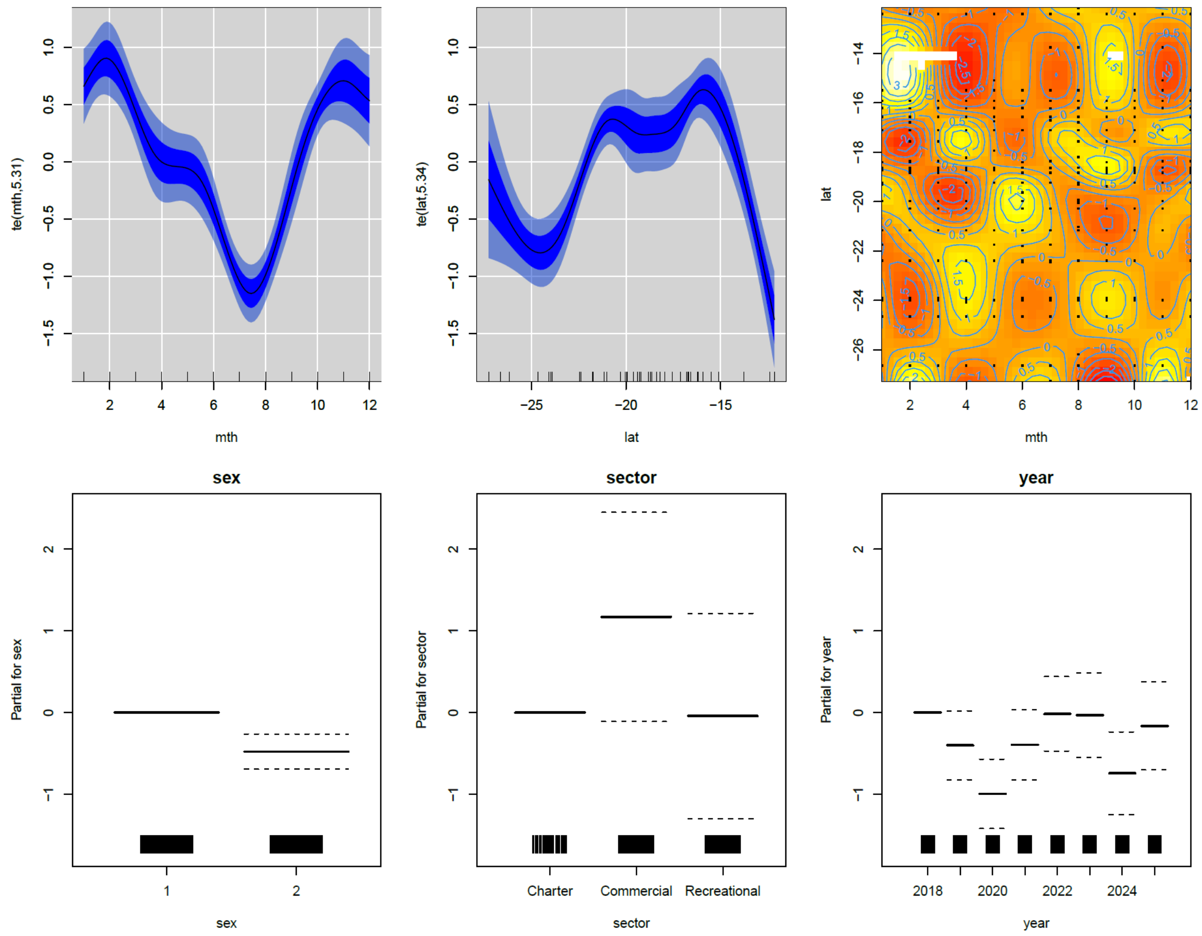

Figure D.8: GAM model terms in the first age-response model—for the latitude and month interaction subplot, white indicates a positive response (increased age) and red indicates a negative response (decreased age)

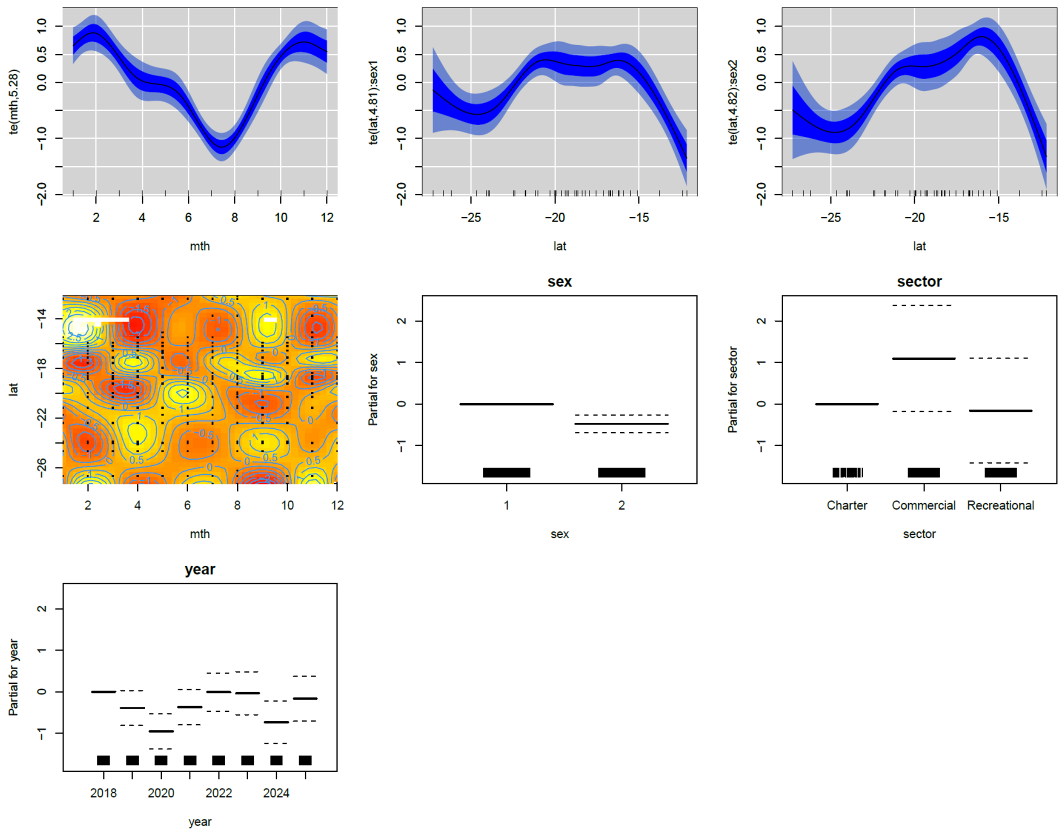

Figure D.9: GAM model terms in the second age-response model—for the latitude and month interaction subplot, white indicates a positive response (increased age) and red indicates a negative response (decreased age)

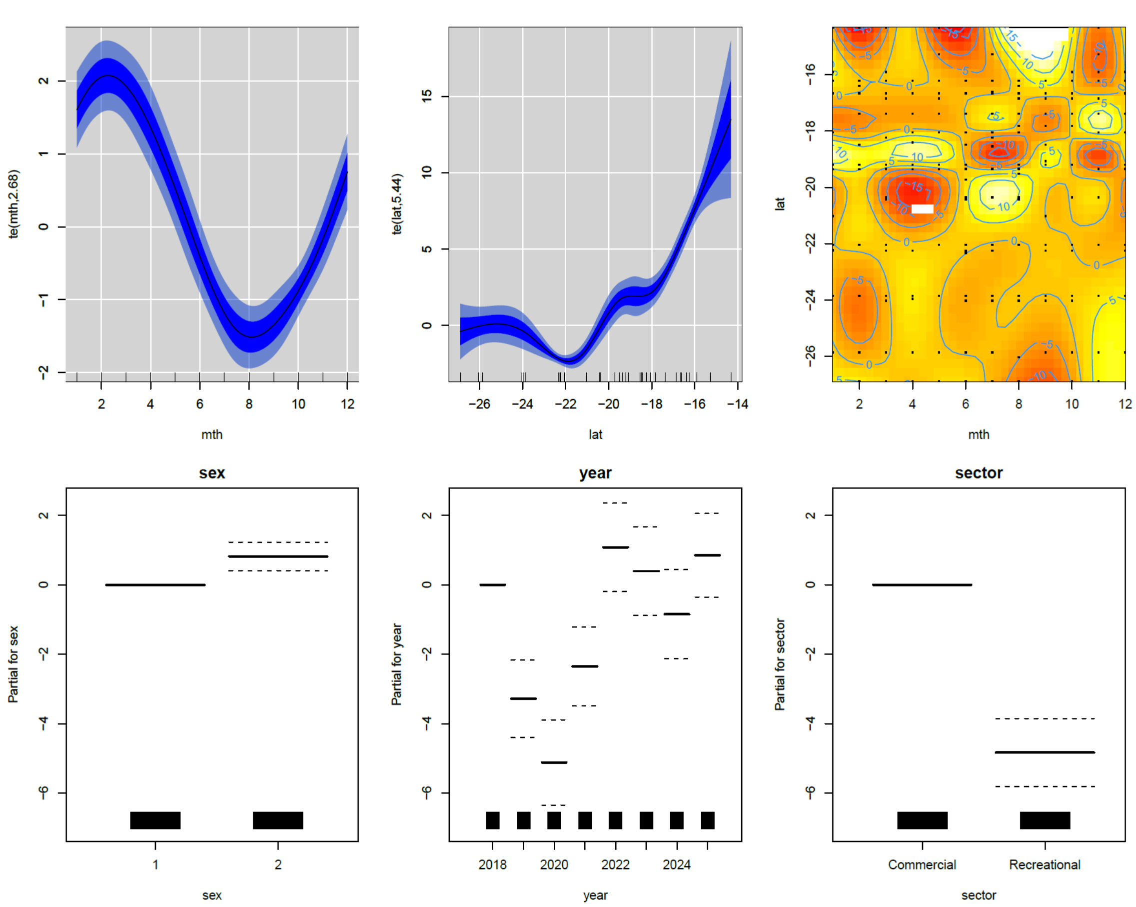

Figure D.10: GAM model terms in the length-response model—for the latitude and month interaction subplot, white indicates a positive response (increased length) and red indicates a negative response (decreased length)

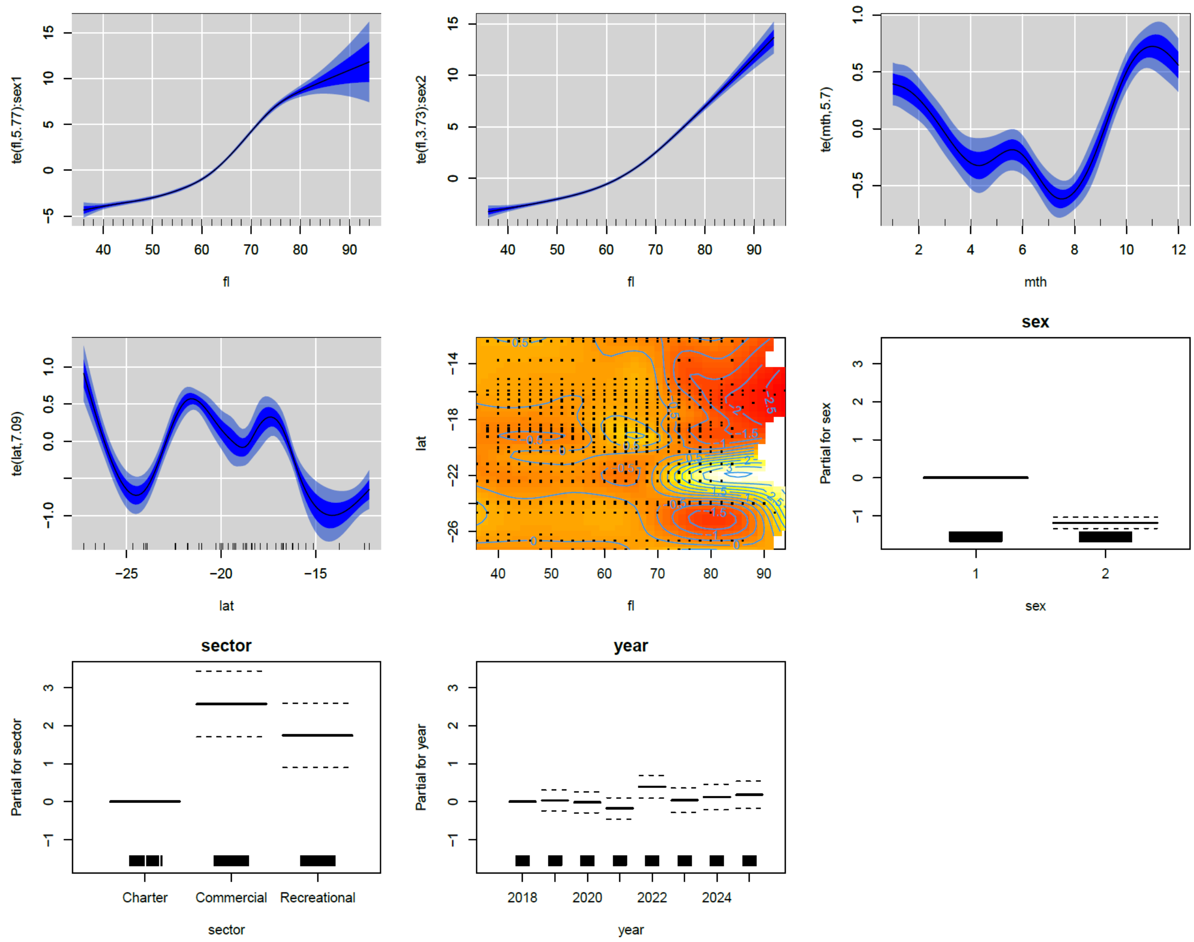

Figure D.11: GAM model terms in the age-at-length model—for the latitude and fork length interaction subplot, white indicates a positive response (increased age) and red indicates a negative response (decreased age)

E Model outputs

E.1 Catch shares

Table E.1 shows the distribution of catch across the three modelled fleets since 2009, excluding the additional fishing mortality that was added to account for depredation, and discards for the recreational and Traditional Owners fleet. In the table, the catch for the recreational and Traditional Owners fleet has been converted from numbers to tonnes using internal Stock Synthesis calculations.

Table E.1: Breakdown of catch, depredation and discards per sector expressed in tonnes—'Kept' refers to retained catch, 'Dep' refers to catch lost due to depredation, 'Total' refers to the combination of retained and depredated catch, 'Disc' refers to discarded fish, 'Dead disc' refers to discarded fish who subsequently died, 'Surv Disc' refers to discarded fish who survived

Year	Commercial			Recreational & Traditional Owners							Charter		
	Kept	Dep	Total	Caught	Disc	Dead Disc	Surv Disc	Kept	Dep	Total	Kept	Dep	Total
1961	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
1962	0.48	0.01	0.49	3.71	0.40	0.30	0.10	3.31	0.04	3.35	0.15	0.00	0.15
1963	0.97	0.01	0.98	8.07	0.86	0.65	0.21	7.21	0.08	7.29	0.30	0.00	0.30
1964	1.45	0.02	1.47	13.45	1.43	1.07	0.36	12.02	0.12	12.14	0.45	0.00	0.45
1965	1.93	0.02	1.95	19.00	2.03	1.52	0.51	16.97	0.17	17.14	0.60	0.00	0.60
1966	2.42	0.02	2.44	23.88	2.55	1.91	0.64	21.33	0.22	21.55	0.75	0.00	0.75
1967	2.90	0.03	2.93	28.05	2.99	2.24	0.75	25.06	0.25	25.31	0.90	0.01	0.91
1968	3.39	0.03	3.42	32.73	3.49	2.62	0.87	29.24	0.29	29.53	1.05	0.01	1.06
1969	3.87	0.04	3.91	38.20	4.08	3.06	1.02	34.12	0.34	34.46	1.19	0.02	1.21
1970	4.35	0.05	4.40	42.91	4.58	3.43	1.15	38.33	0.38	38.71	1.34	0.02	1.36
1971	4.84	0.04	4.88	52.28	5.59	4.20	1.39	46.69	0.47	47.16	1.49	0.02	1.51
1972	5.32	0.05	5.37	59.72	6.39	4.79	1.60	53.33	0.53	53.86	1.64	0.02	1.66
1973	5.80	0.06	5.86	68.16	7.30	5.47	1.83	60.86	0.61	61.47	1.79	0.02	1.81
1974	6.29	0.06	6.35	77.03	8.25	6.18	2.07	68.78	0.68	69.46	1.94	0.02	1.96
1975	6.77	0.07	6.84	83.73	8.98	6.73	2.25	74.75	0.75	75.50	2.09	0.02	2.11
1976	7.25	0.08	7.33	89.02	9.78	7.34	2.44	79.24	0.79	80.03	2.24	0.02	2.26
1977	7.74	0.07	7.81	92.87	9.98	7.49	2.49	82.89	0.82	83.71	2.39	0.02	2.41
1978	8.22	0.08	8.30	100.60	10.78	8.08	2.70	89.82	0.90	90.72	2.54	0.02	2.56
1979	8.70	0.09	8.79	108.11	11.58	8.69	2.89	96.53	0.96	97.49	2.69	0.03	2.72
1980	9.19	0.09	9.28	115.44	12.65	9.49	3.16	102.79	1.02	103.81	2.84	0.03	2.87
1981	9.67	0.10	9.77	125.89	13.51	10.13	3.38	112.38	1.12	113.50	2.99	0.03	3.02
1982	10.16	0.10	10.26	138.77	15.17	11.38	3.79	123.60	1.24	124.84	3.14	0.03	3.17
1983	10.64	0.11	10.75	145.25	15.88	11.91	3.97	129.37	1.29	130.66	3.29	0.03	3.32
1984	11.12	0.11	11.23	151.79	16.25	12.19	4.06	135.54	1.36	136.90	3.44	0.03	3.47
1985	11.61	0.11	11.72	161.26	17.52	13.14	4.38	143.74	1.44	145.18	3.58	0.04	3.62
1986	12.09	0.12	12.21	166.49	18.48	13.86	4.62	148.01	1.49	149.50	3.73	0.04	3.77

Continued on next page

Table E.1 – *Continued from previous page*

Year	Commercial			Recreational & Traditional Owners							Charter		
	Kept	Dep	Total	Caught	Disc	Dead Disc	Surv Disc	Kept	Dep	Total	Kept	Dep	Total
1987	12.57	0.13	12.70	169.59	18.77	14.07	4.70	150.82	1.50	152.32	3.88	0.04	3.92
1988	13.06	0.13	13.19	179.04	19.36	14.52	4.84	159.68	1.60	161.28	4.03	0.04	4.07
1989	13.54	0.14	13.68	196.40	21.11	15.83	5.28	175.29	1.75	177.04	4.18	0.04	4.22
1990	12.38	0.12	12.50	212.19	22.45	16.84	5.61	189.74	1.90	191.64	4.33	0.04	4.37
1991	16.89	0.16	17.05	225.91	24.45	18.33	6.12	201.46	2.01	203.47	4.48	0.05	4.53
1992	22.44	0.23	22.67	234.30	25.03	18.77	6.26	209.27	2.10	211.37	4.63	0.05	4.68
1993	20.07	0.21	20.28	246.32	26.83	20.12	6.71	219.49	2.19	221.68	4.78	0.05	4.83
1994	26.29	0.26	26.55	252.68	27.46	20.59	6.87	225.22	2.25	227.47	4.93	0.05	4.98
1995	35.71	0.35	36.06	257.11	29.04	21.78	7.26	228.07	2.28	230.35	5.08	0.05	5.13
1996	52.69	0.52	53.21	252.77	28.89	21.67	7.22	223.88	2.24	226.12	6.94	0.07	7.01
1997	73.24	0.73	73.97	257.35	27.61	20.71	6.90	229.74	2.30	232.04	13.94	0.14	14.08
1998	112.90	1.13	114.03	277.90	29.81	22.35	7.46	248.09	2.48	250.57	21.41	0.21	21.62
1999	219.63	2.20	221.83	286.65	31.98	23.99	7.99	254.67	2.55	257.22	15.35	0.15	15.50
2000	129.00	1.29	130.29	285.06	31.89	23.92	7.97	253.17	2.53	255.70	12.89	0.13	13.02
2001	144.52	1.45	145.97	298.34	31.49	23.62	7.87	266.85	2.67	269.52	13.08	0.13	13.21
2002	136.94	1.37	138.31	320.65	35.56	26.67	8.89	285.09	2.85	287.94	14.57	0.15	14.72
2003	106.21	1.06	107.27	314.40	36.81	27.61	9.20	277.59	2.78	280.37	18.06	0.18	18.24
2004	64.63	0.65	65.28	328.07	42.11	31.58	10.53	285.96	2.86	288.82	17.88	0.18	18.06
2005	20.66	0.21	20.87	334.11	41.90	31.43	10.47	292.21	2.92	295.13	17.04	0.17	17.21
2006	27.97	0.28	28.25	313.87	39.83	29.87	9.96	274.04	2.74	276.78	17.71	0.17	17.88
2007	24.36	0.25	24.61	290.50	34.45	25.84	8.61	256.05	2.56	258.61	20.01	0.20	20.21
2008	28.91	0.28	29.19	280.91	32.02	24.01	8.01	248.89	2.48	251.37	15.20	0.15	15.35
2009	65.48	0.66	66.14	269.76	30.84	23.13	7.71	238.92	2.39	241.31	23.48	0.24	23.72
2010	52.35	1.44	53.79	256.54	36.44	25.68	10.76	220.10	10.39	230.49	18.84	0.86	19.70
2011	54.12	2.44	56.56	225.20	30.72	21.65	9.07	194.48	16.40	210.88	18.38	1.49	19.87
2012	38.49	2.40	40.89	242.31	34.20	24.20	10.00	208.11	25.29	233.40	18.10	2.11	20.21
2013	47.25	3.78	51.03	246.78	39.45	28.15	11.30	207.33	32.91	240.24	14.29	2.18	16.47
2014	53.68	5.23	58.91	231.87	36.97	26.42	10.55	194.90	38.17	233.07	12.44	2.34	14.78
2015	63.67	7.32	70.99	252.33	39.95	28.58	11.37	212.38	49.49	261.87	17.14	3.83	20.97
2016	78.76	10.43	89.19	276.59	43.53	31.18	12.35	233.06	62.98	296.04	15.33	3.97	19.30
2017	77.00	11.55	88.55	304.60	57.53	41.56	15.97	247.07	75.95	323.02	35.32	10.41	45.73
2018	98.81	16.55	115.36	295.55	47.80	34.36	13.44	247.75	85.37	333.12	25.85	8.55	34.40
2019	69.61	12.88	82.49	216.64	32.27	23.15	9.12	184.37	70.38	254.75	19.50	7.14	26.64
2020	62.85	12.73	75.58	218.28	30.79	22.06	8.73	187.49	78.55	266.04	25.17	10.12	35.29
2021	69.99	15.40	85.39	350.10	53.93	38.83	15.10	296.17	135.08	431.25	31.64	13.84	45.48
2022	74.86	17.78	92.64	271.60	43.98	31.76	12.22	227.62	112.27	339.89	25.97	12.29	38.26
2023	73.75	18.81	92.56	344.55	67.22	48.87	18.35	277.33	147.11	424.44	29.05	14.78	43.83
2024	89.01	24.26	113.27	302.43	54.46	39.53	14.93	247.97	140.75	388.72	25.90	14.10	40.00
2025	98.22	28.49	126.71	318.70	51.75	38.82	12.93	266.95	154.84	421.79	28.64	16.62	45.26

E.2 MLE diagnostics

E.2.1 Likelihood profile

Likelihood profiles on R0, steepness and natural mortality were conducted on scenario 1 as reference.

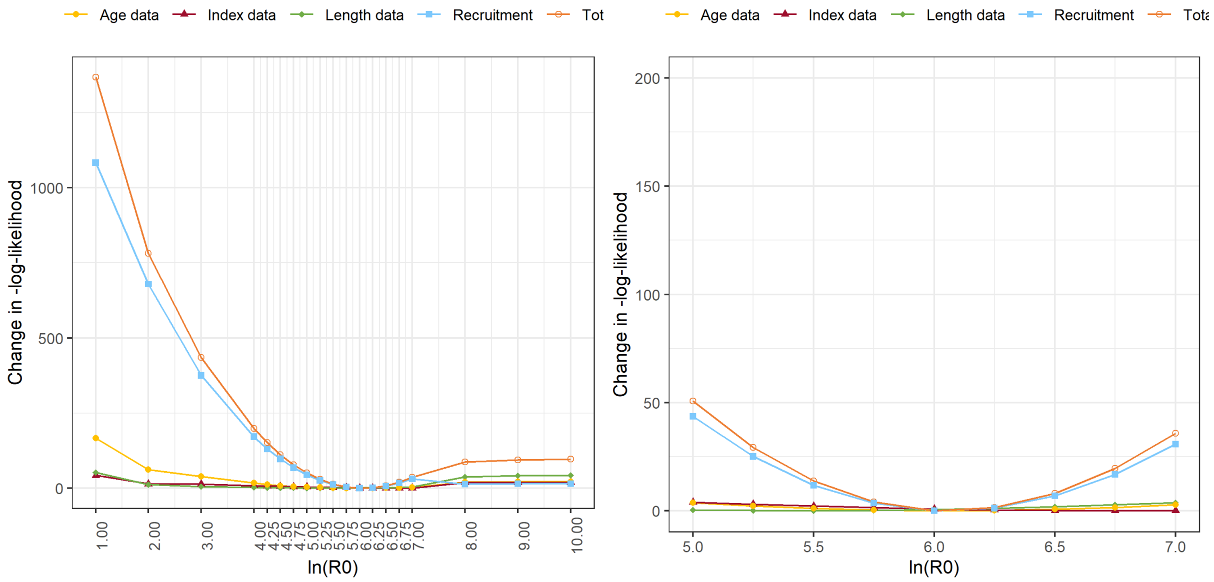

Figure E.1: Likelihood profile for ln(R0)—the left panel shows a wide span of tested values and the right panel zooms in on the area of interest

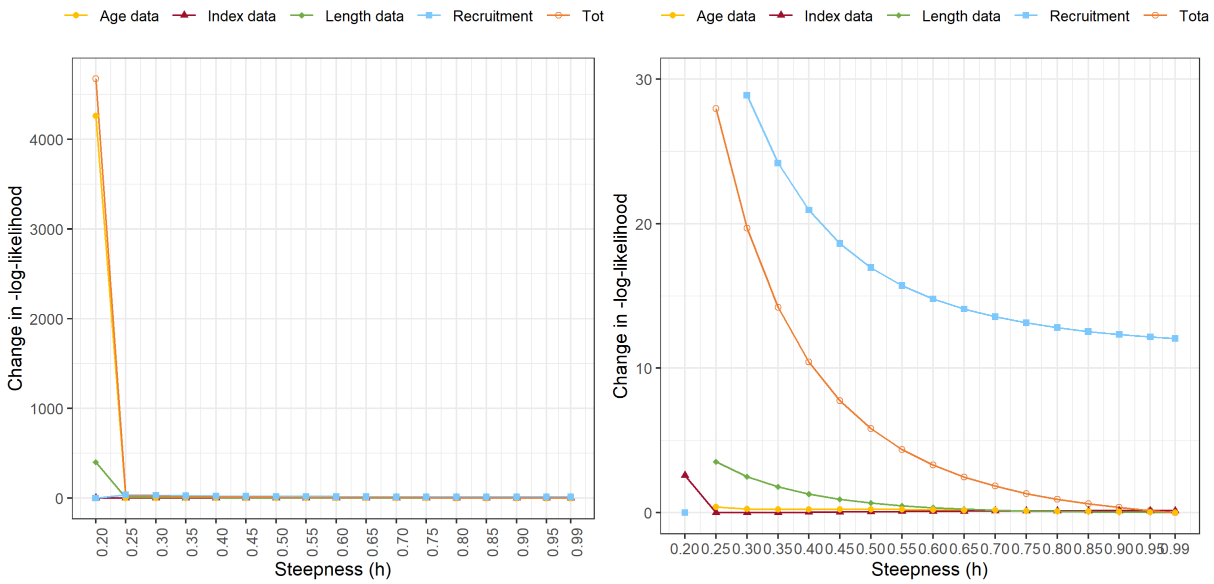

Figure E.2: Likelihood profile for steepness—the left panel shows a wide span of tested values and the right panel zooms in on the area of interest

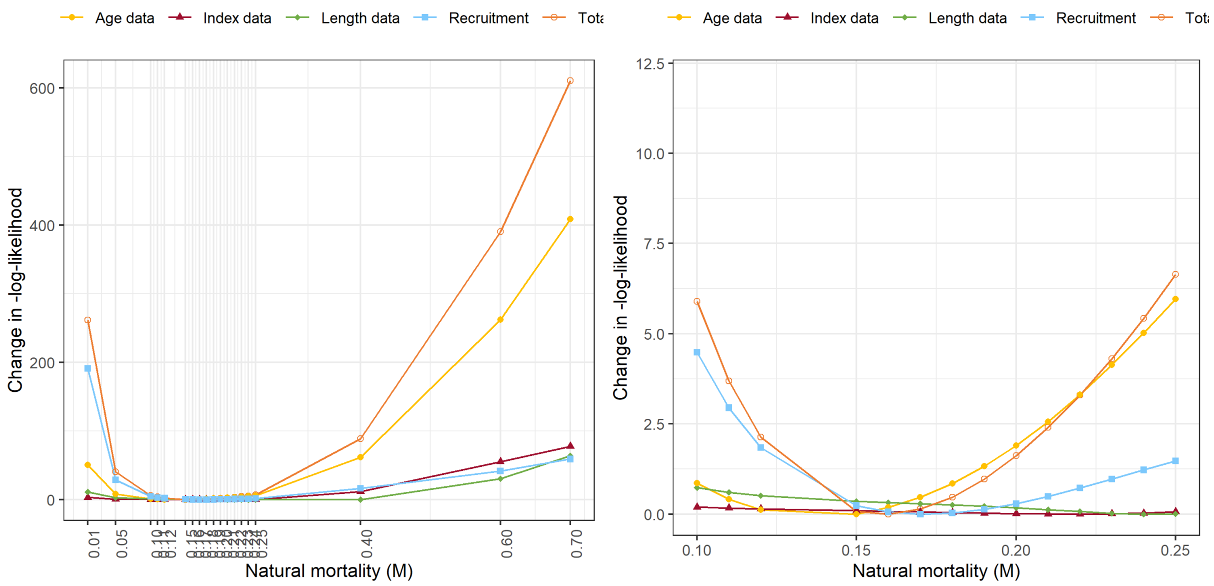

Figure E.3: Likelihood profile for natural mortality—the left panel shows a wide span of tested values and the right panel zooms in on the area of interest

E.2.2 Retrospective analysis

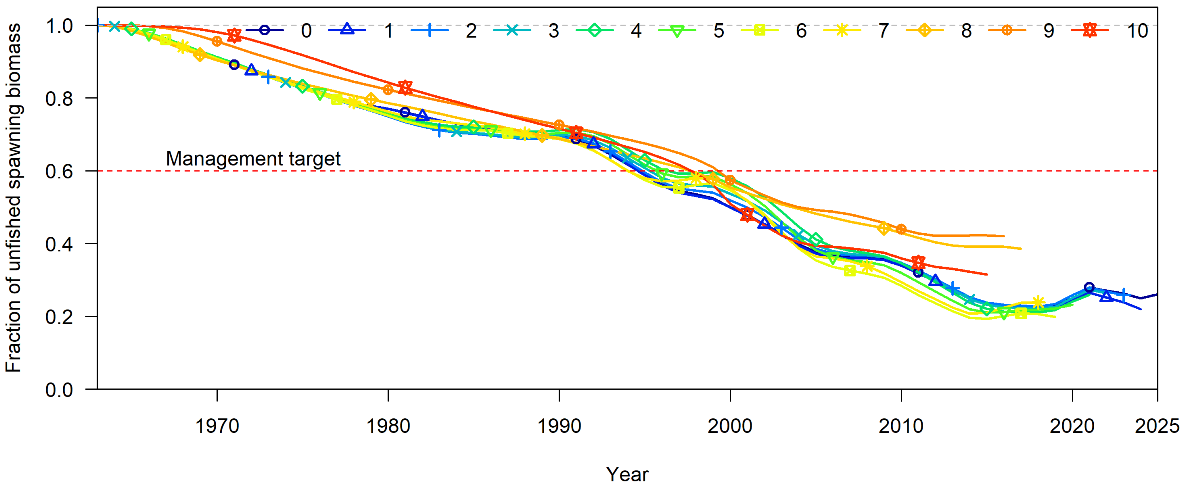

Figure E.4: Relative biomass trajectories from a retrospective analysis (scenario 1, MLE), in which the most recent years of input data were iteratively removed and the model re-optimised

E.3 MCMC diagnostics

Figures E.5, E.6 and E.7 show the potential scale reduction factor (R-hat), trace plots and posterior density function plots from the MCMC analysis for scenarios 1 to 12. These demonstrate good convergence of the MCMC analysis.

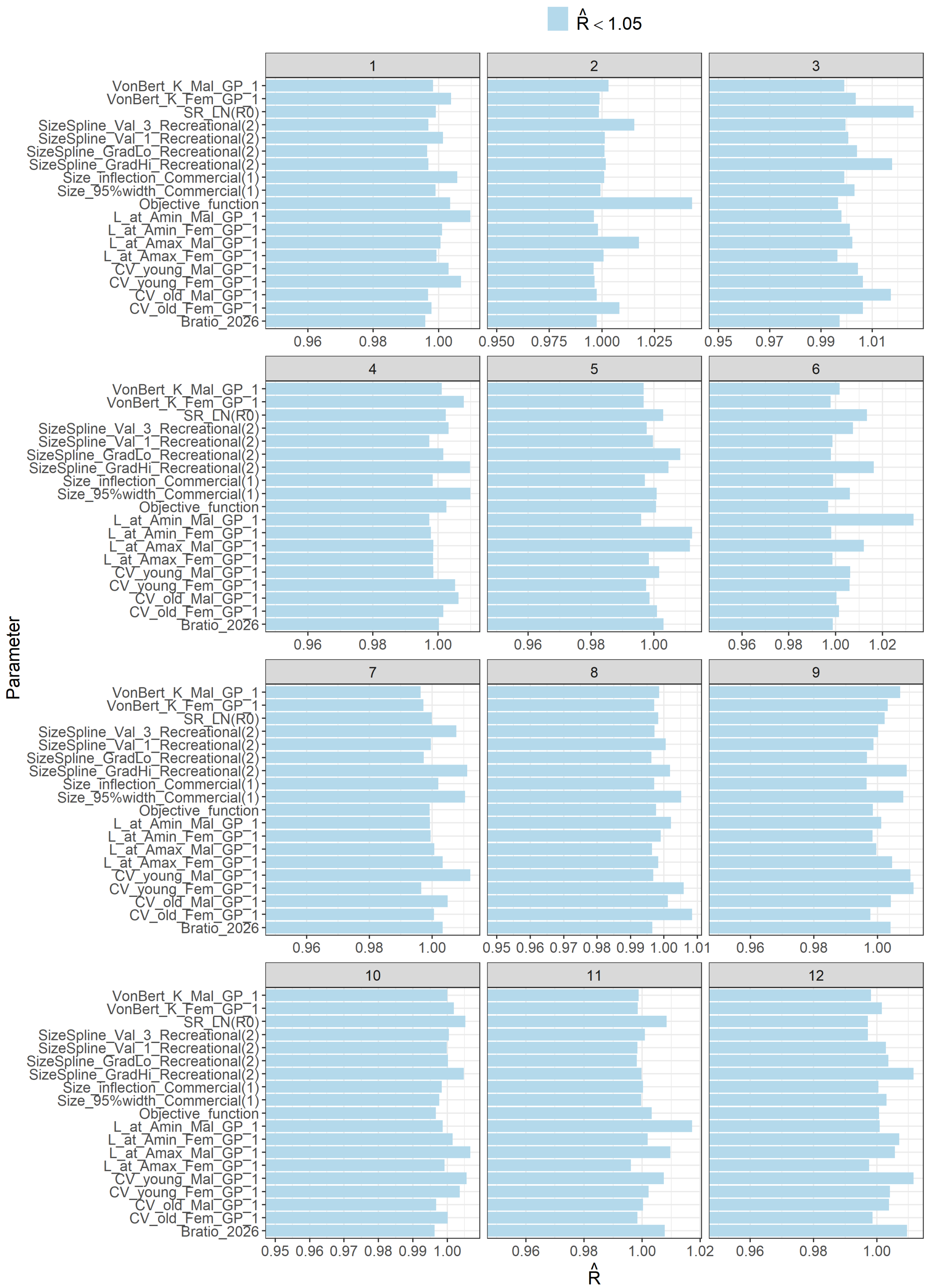

Figure E.5: Potential scale reduction factor, $\hat{R}$, values among the scenarios for Queensland east coast saddletail snapper—model is likely converged if $\hat{R} < 1.05$

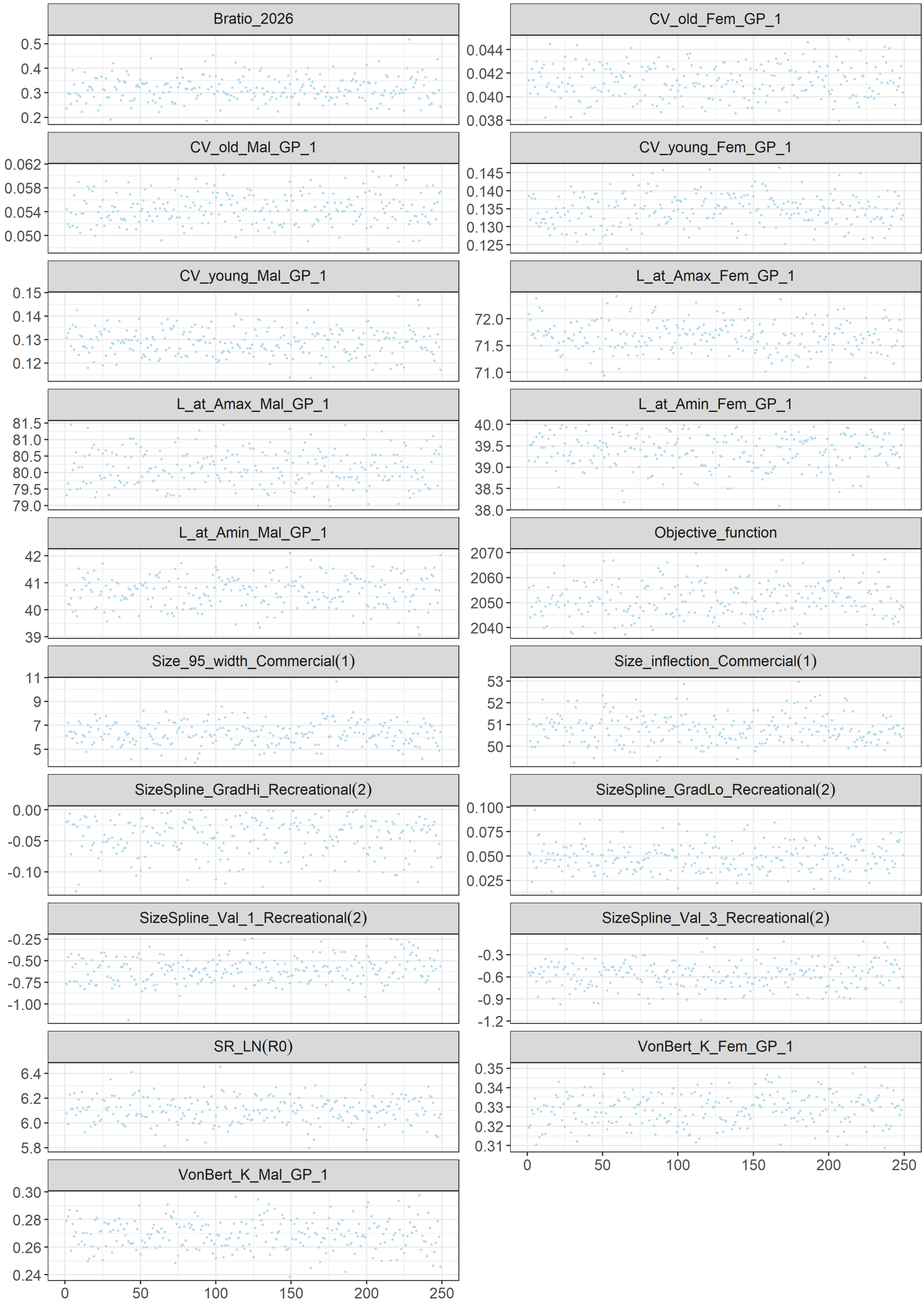

Figure E.6: Trace plots of MCMC iterations for scenario 1

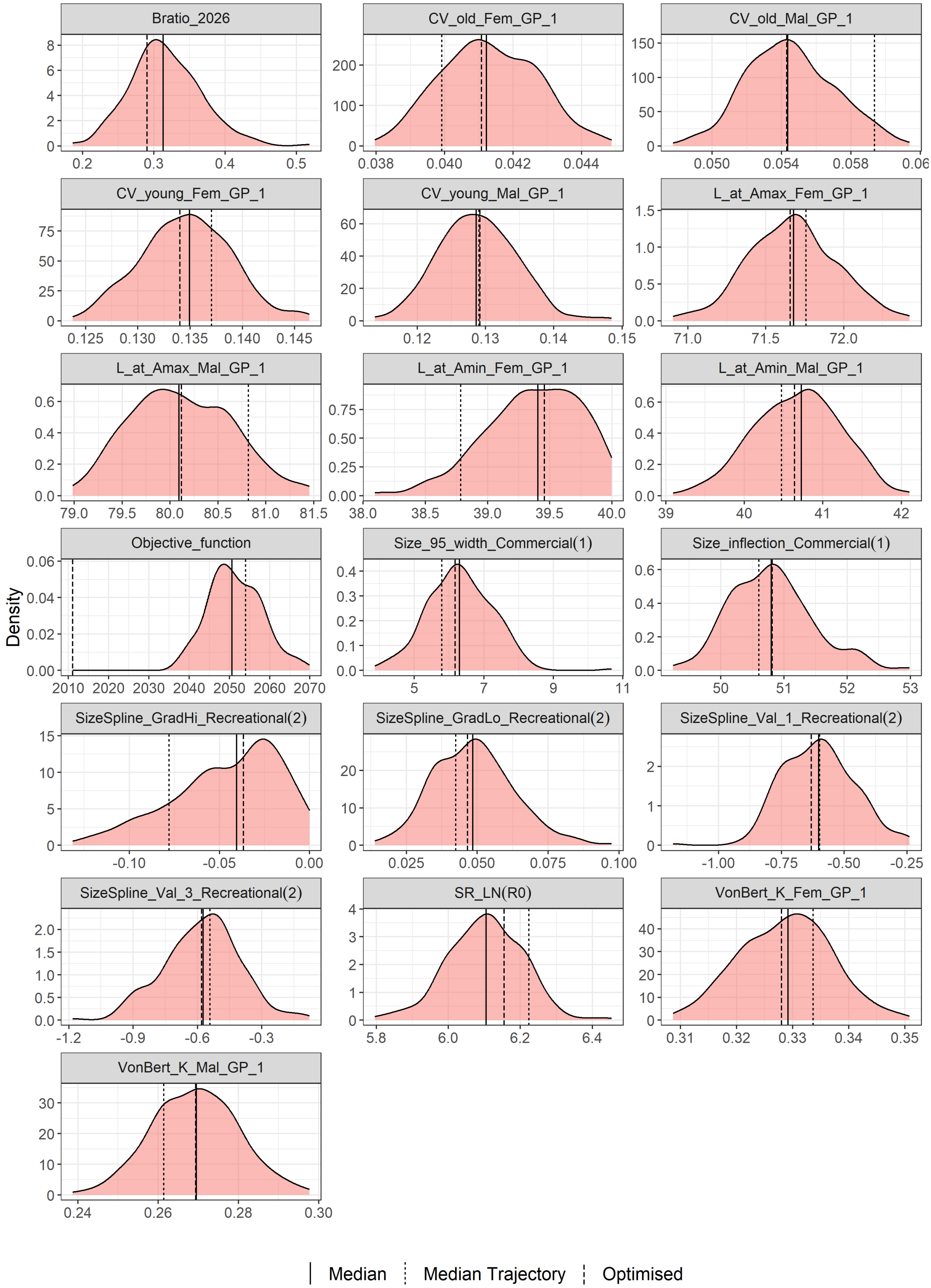

Figure E.7: Posterior density of MCMC iterations for scenario 1—“Median” line shows median parameter value for MCMC iterations “Optimised” shows the parameter value found from maximum likelihood estimates.

E.4 Other outputs

E.4.1 Andre plots

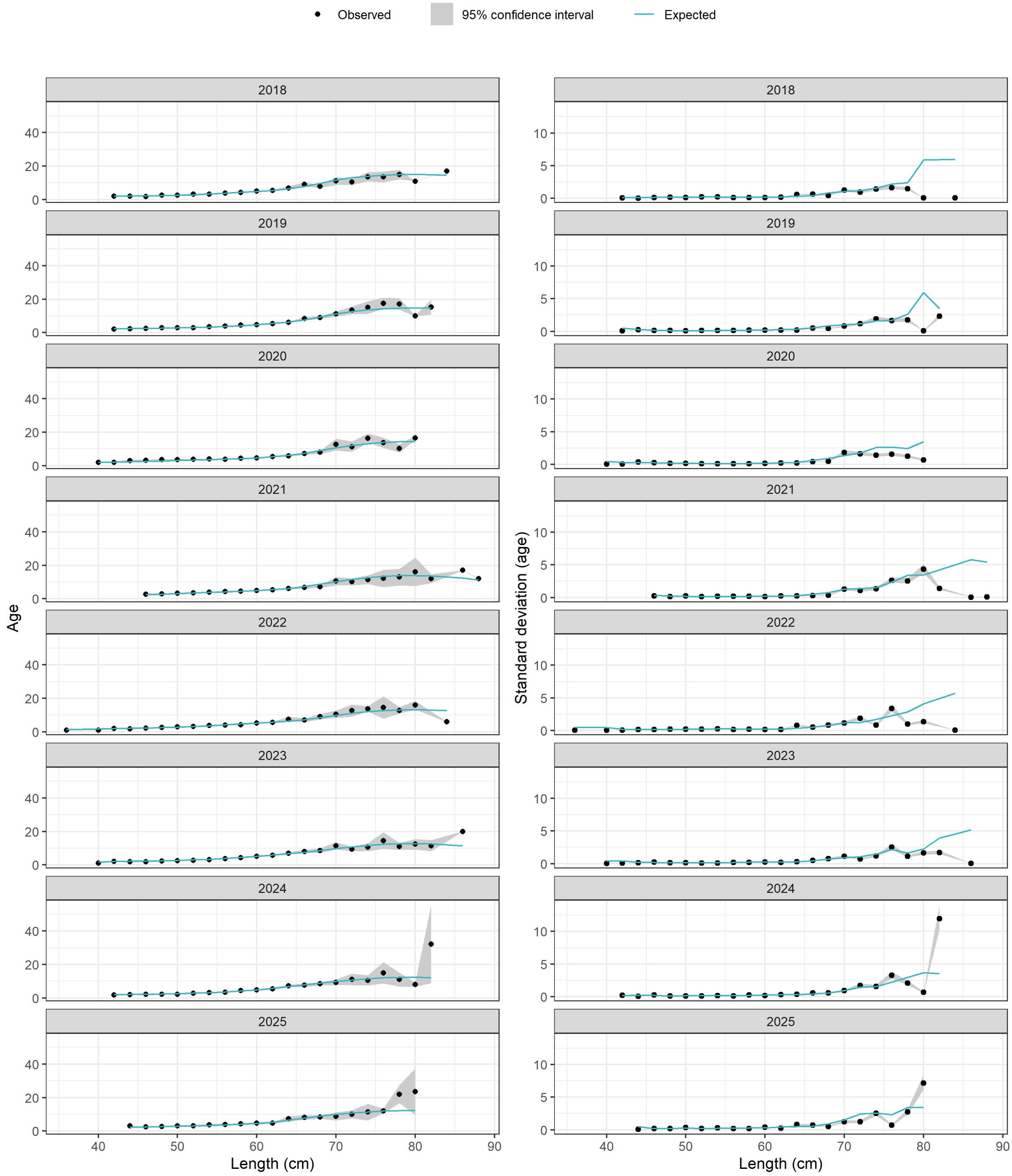

Figure E.8: Mean age and standard deviation in female conditional age-at-length data between 2018 and 2025—left plots are mean age at length by size-class (observed and expected) with confidence intervals obtained by adding the appropriate number of standard errors of mean to the data; right plots in each pair are SE of mean age-at-length with confidence intervals based on the chi-square distribution

NULL

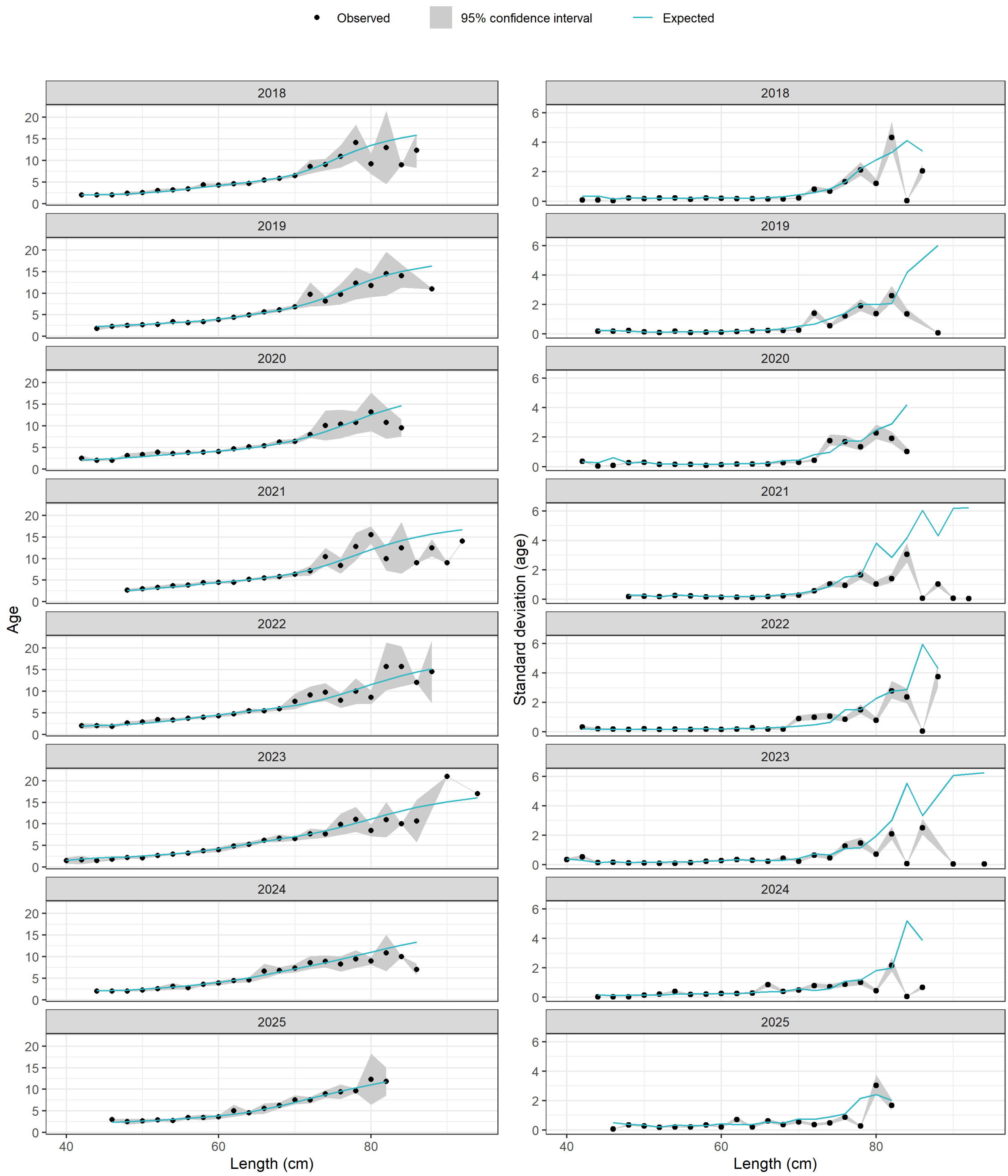

Figure E.9: Mean age and standard deviation in male conditional age-at-length data between 2018 and 2025—left plots are mean age at length by size-class (observed and expected) with confidence intervals obtained by adding the appropriate number of standard errors of mean to the data; right plots in each pair are SE of mean age-at-length with confidence intervals based on the chi-square distribution

NULL

E.4.2 Stock-recruit curve

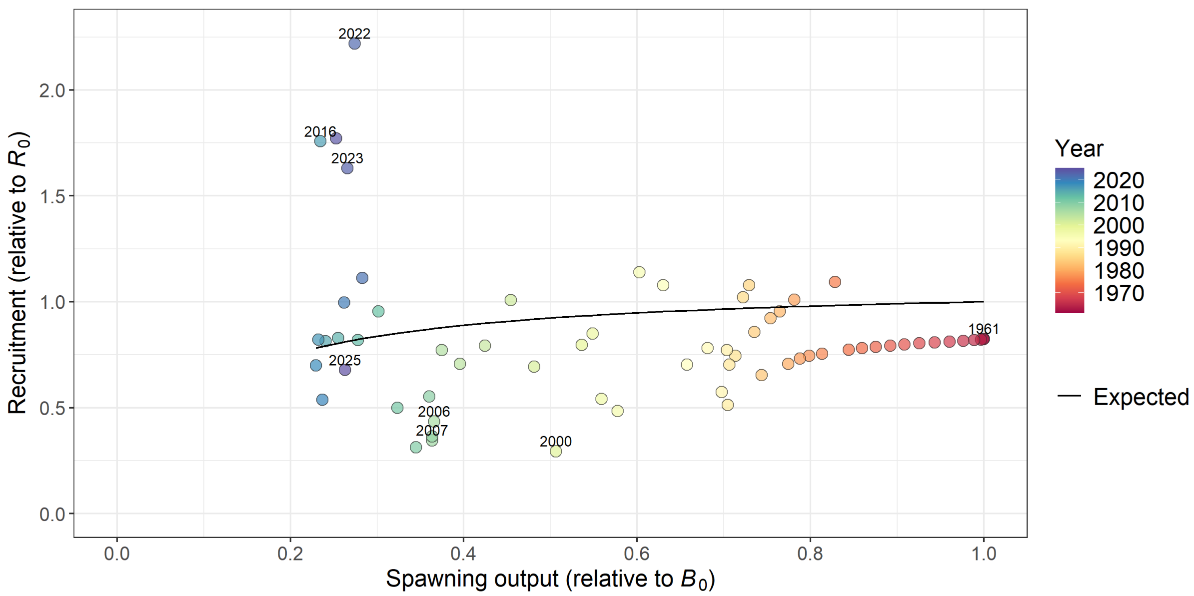

Figure E.10: Stock-recruit curve for Queensland east coast saddletail snapper based on the reference scenario 1—point colors indicate year, with warmer colors indicating earlier years and cooler colors in showing later years

E.4.3 Fishing mortality

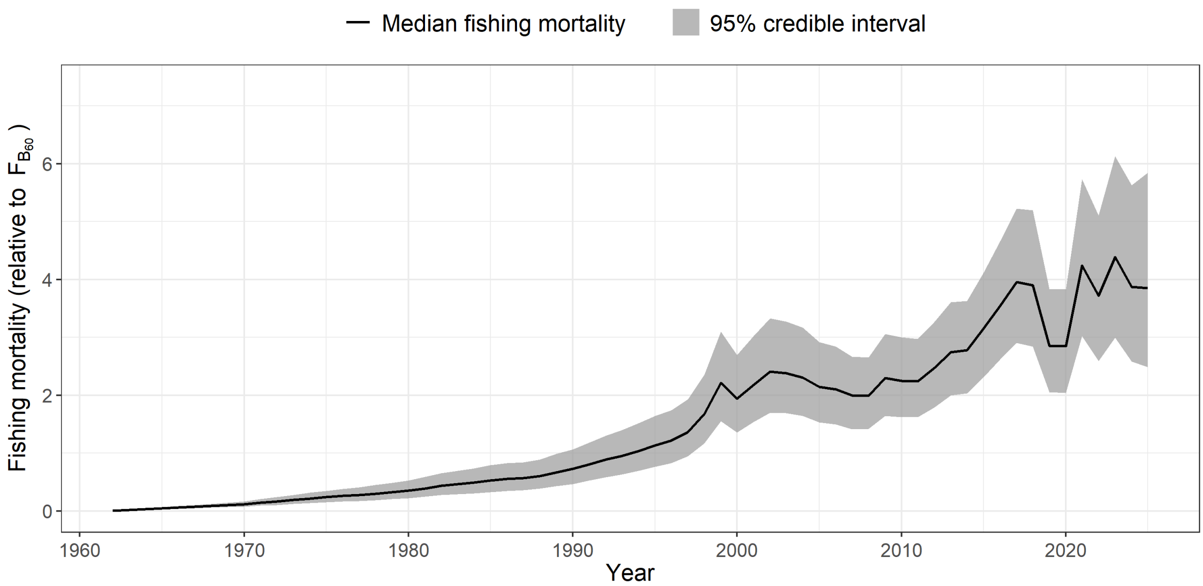

Figure E.11: Time series of fishing mortality ratio ($F/F_{B_{60}}$) from the ensemble model—whiskers represent 95% credible intervals, boxes represent 50% credible intervals, horizontal bars represent medians, and the points represent outliers

E.4.4 Recruitment deviations

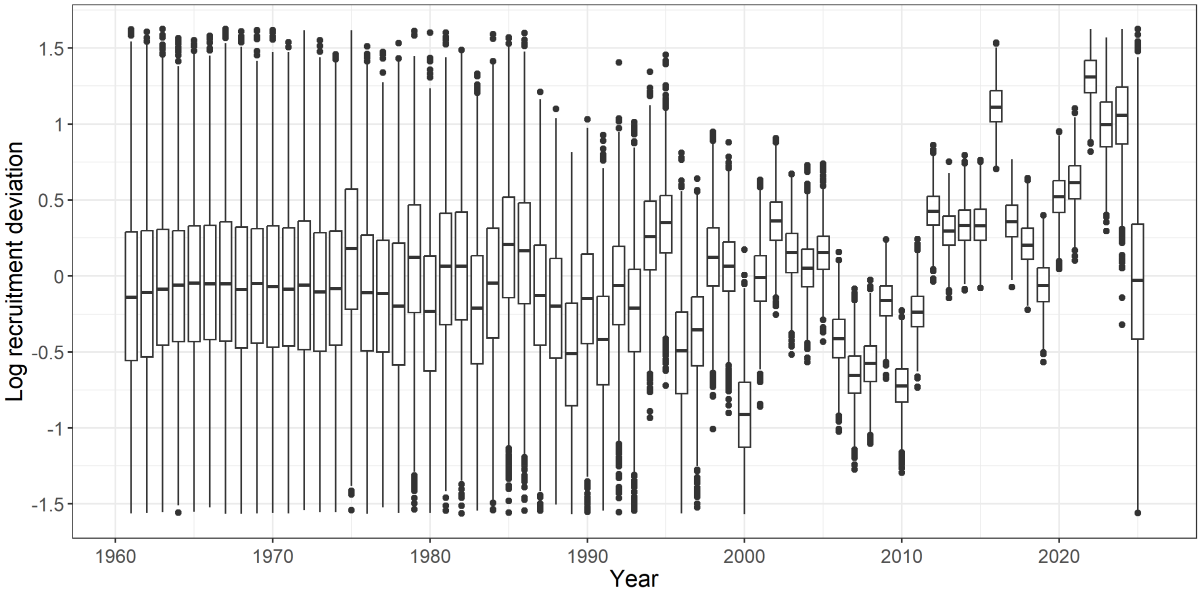

Figure E.12: Recruitment deviations from the ensemble model—whiskers represent 95% credible intervals, boxes represent 50% credible intervals, horizontal bars represent medians, and the points represent outliers

E.4.5 Sensitivity: parameter estimates and derived quantities

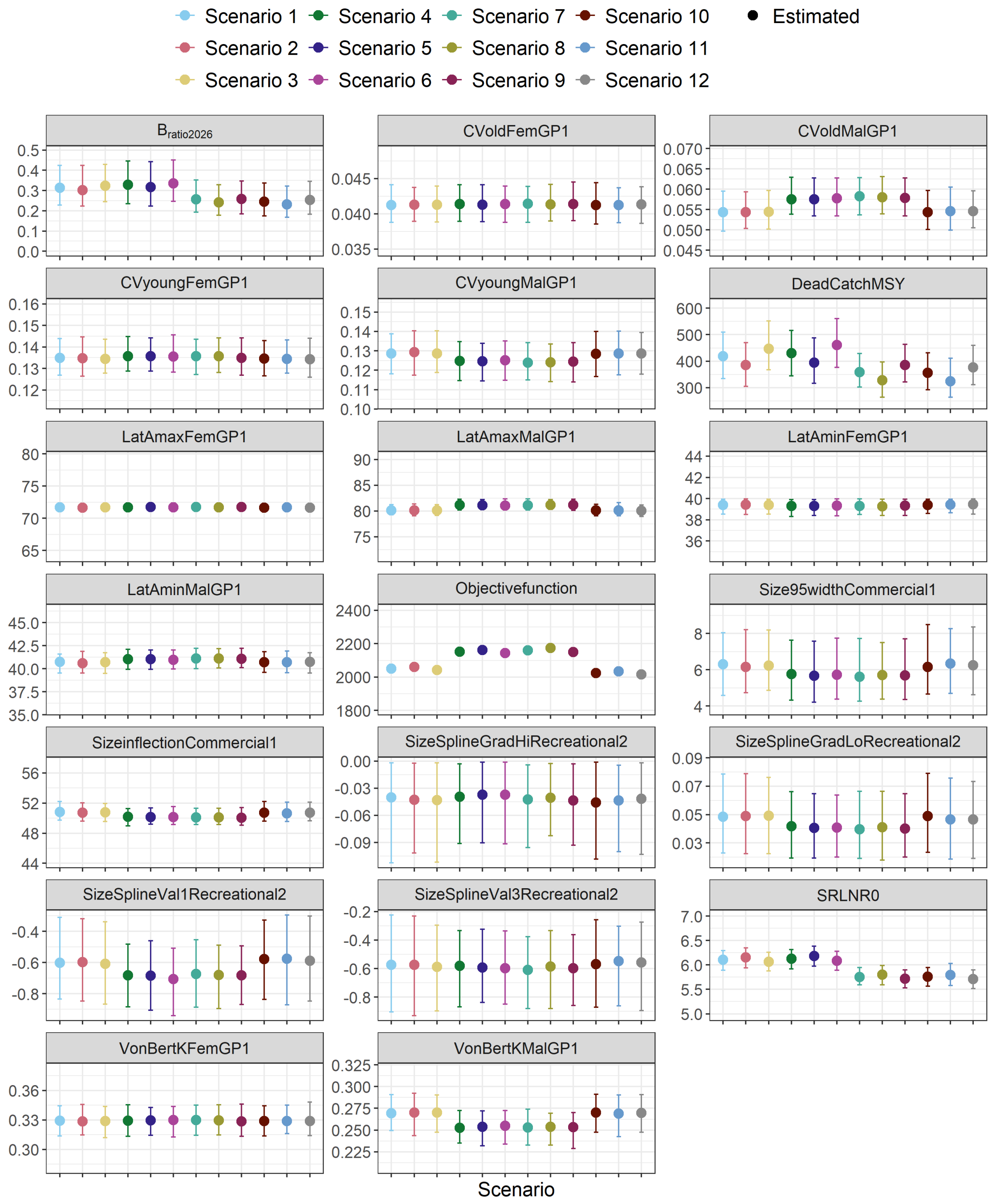

Figure E.13: Comparison of parameter estimates and derived quantities among the 12 scenarios included in the ensemble model

F Fishery management: detailed information

F.1 History of saddletail snapper management in Queensland

Table F.1: History of saddletail snapper management in Queensland

Year	Management	Legislation
1957	Minimum size of 14 inches (35.56 cm) (listed as Scarlet Sea-Perch (*Lutjanus malabaricus*))	*The Fisheries Acts, 1957 to 1962*
1975	Inclusion of no-fishing zones in the Great Barrier Reef	*Great Barrier Reef Marine Park Act 1975*
1982	Section 35 permit; issued to recreational fishers who had caught more fish than they could use, and therefore were allowed to sell that portion of their catch that was deemed surplus to their requirements.	*Fishing Industry Organization and Marketing Act 1982 (Qld) (FIOMA)*
1993	Recreational possession limits of a combined total of 30 coral reef fish covering 26 species. Skin not to be removed from fillets by recreational fishers, except in the case of charter vessels in excess of 48 hrs where the majority of the skin may be removed provided a minimum is left for identification. Minimum size limit of 40 cm. Charter vessel possession limit arrangements: extended charters in excess of 48 hrs allowed double the prescribed possession limit. Restructure of commercial line fishery into regional endorsements (L1,L2,L3). The existing L symbol was introduced into legislation with the numbers L1–L9 depicting different regions of operations. New format for landed fish, where a fish has been filleted there must be two fillets equal to one whole fish.	*Fishing Industry Organisation and Marketing Regulation 1993*
1994	Section 35 permits to sell recreationally caught fish repealed.	*Fisheries Act 1994*

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Table F.1 – *Continued from previous page*

Year	Management	Legislation
2003	Fisheries (Coral Reef Fin Fish) Management Plan implemented. Recreational in-possession limits reduced to a combined total of 9 crimson and saddletail snapper. The fishery symbols for the commercial fishery are ‘L1’, ‘L2’, ‘L3’ and ‘L8’ provide access to fishing areas in Queensland while RQ quota provides access to fish and both are required. RQ licence holders must hold appropriate line units (OS units) to take saddletail snapper, which take the form of individual transferable quotas. The total yearly catch of Other Species available for allocation is 955.5 t. New reporting requirements. Seasonal closures across the GBR for nine days around the new moon period in October, November and December each year. Fish may be taken only by using fishing lines. A commercial fisher must not use more than 3 fishing lines at the same time, and the total number of hooks or lures attached to the lines must not be more than 6. A primary boat longer than 20 m must not be used. The permitted distance for an assistant fisher to be under direction of a commercial fisher is 5 n miles. A tender boat must not be used more than 5 n miles from its primary boat. This does not apply if the tender boat and its primary boat are located on the same reef.	*Fisheries (Coral Reef Fin Fish) Management Plan 2003 (Queensland)*
2004	New Fisheries Management implemented (ITQ and RQ) and the Great Barrier Reef Marine Park Authority implemented new zoning arrangements for the Great Barrier Reef Marine Park. Under the rezoning approximately 33% of the marine park area is protected through closed green zones within which extractive uses are restricted.	*Great Barrier Reef Marine Park Zoning Plan 2003*
Sept. 1 2019	Fisheries (Coral Reef Fin FIsh) Management Plan repealed *Fisheries (General) Regulation 2019 (Queensland)*, *Fisheries (Commercial Fisheries) Regulation 2019 (Queensland)*, *Fisheries Declaration 2019 (Queensland)* and *Fisheries Quota Declaration 2019 (Queensland)* enacted.	
2019	The total quota entitlement for the commercial reef line fishery for each line year for OS line unit is 955.597 t	*Fisheries Quota Declaration 2019 (Queensland)*

F.2 List of 'other species' in fishery

- Cod - greasy
- Camouflage rockcod
- Cod - flowery
- Cod - bar
- Cod - white lined
- Radiant rockcod
- Cod - black-tipped rock
- Peacock cod
- Cod - black-finned
- Cod - tomato
- Cod - birdwire
- Cod - coral
- Cod - yellow spotted rock
- Cod - speckled fin
- Cod - blue maori
- Cod - hapuku
- Cod - red rock
- Cod - maori
- Cod - red flushed
- Cod - blue spot rock
- Cod - long finned
- Banded Rockcod
- Blacksaddle Rockcod
- Chinaman Rockcod
- Cod - brown banded
- Cod - leopard rock
- Cod - strawberry rock
- Cod - barramundi
- Cod - potato
- Cod - groper unspecified
- Cod - reef unspecified
- Cod - unspecified
- Speckled grouper
- Grouper - eight bar
- Grouper - comet
- Bass groper
- Whitespotted Grouper
- Emperor - spangled
- Emperor - Unspecified
- Lancer
- Emperor - long nose
- Emperor - pink-eared
- Emperor - red ear
- Emperor - yellow tailed
- Emperor - variegated
- Emperor - reticulated
- Emperor - orange striped
- Emperor - yellow lipped
- Bream - japanese large-eye
- Emperor - yellow spotted
- Smalltooth Emperor
- Ornate Emperor
- Longfin Emperor
- Bream - mozambique
- Bream - blubber lip
- Bream - sea
- Bream - japanese large-eye
- Bream - maori
- Seabream - Collar
- Sea bream - big eye
- Emperor - red
- Stripey - spanish flag
- Jobfish - gold banded
- Nannygai - small mouth
- Nannygai - large mouth
- Nannygai - unspecified
- Jobfish - rosy
- Jobfish - green
- Rusty jobfish
- Jobfish - small-toothed
- Jobfish - unspecified
- Hussar
- Hussar - unspecified
- Snapper - unspecified tropical
- Snapper - ruby
- Snapper - flame tail
- Snapper - onespot
- Snapper - pale
- Snapper - saddleback
- Olbique-banded snapper
- Midnight Snapper
- Ornate snapper
- Snapper - indonesian
- Goldeneye snapper
- Sharptooth snapper
- Lavender snapper
- Snapper - black and white
- Fiveline Snapper
- Snapper - black spot
- Cocoa snapper
- Tropical snapper
- Perch - moses
- Perch - dark tailed sea
- Perch - maori sea
- Bass - red
- Seaperch - swallowtail
- Paddle tail
- Chinaman
- Wrasse - unspecified
- Wrasse - sling-jaw
- Wrasse - humphead maori
- Foxfish
- Redbreast Maori Wrasse
- Reefcrest Parrotfish
- Pigfish - gold spot
- Eastern Pigfish
- Tusk fish - venus
- Tusk fish - unspecified
- Tusk fish - black spot
- Tusk fish - blue
- Tusk fish - purple
- Painted sweetlip
- Sweetlip - clown
- Oriental Sweetlips
- Sweetlip - striped
- Surgeon fish - convict
- Fusilier - yellow tail
- Fusilier - southern
- Fish - mixed reef b
- Fish - mixed reef a
- Fish - mixed reef