UNIVERSITY of TASMANIA

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Institute for Marine and Antarctic Studies

Opportunities and impacts of range extending scalefish species

Understanding population dynamics, ecosystem impacts and management needs

Alexia Graba-Landry, Curtis Champion, James Haddy, Jeremy Lyle, David Mossop, Rod Pearn, Gretta Pecl, Heidi Pethybridge, Barrett Wolfe and Sean Tracey

May 2022

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Executive Summary

Project Summary

The biology and diet were quantified for three key range shifting species in Tasmania with both recreational and commercial fishery value—Yellowtail Kingfish (*Seriola lalandi*), Snapper (*Chrysophrys auratus*) and King George Whiting (*Sillaginodes punctatus*). The project was heavily reliant on engagement from the recreational fishing community and multiple citizen science initiatives. Our primary approach was a state-wide fish frame donation program which was implemented in 2019, with a network of 16 drop-off locations along most of coastal Tasmania. This data set was also supplemented with historical data from the IMAS archives, and with fishery-independent sampling for fish outside of the legal size limits. These data were used to quantify life-history parameters (age, population structure, growth, reproduction, mortality), and diet for the three species. This information was then used in both species distribution modelling and Atlantis ecosystem modelling to predict how suitable habitats for each species may shift under future climate change projections and indicate the potential changes to the ecosystem (i.e. food web) if and when they do shift. The species distribution modelling was in part reliant on citizen science data, specifically species occurrence records, reported to the Range Extending Database and Mapping Project (Redmap), survey data from the Reef Life Survey, and historical occurrence records from the Atlas of Living Australia. The results provide important baseline and predictive information on these species for effective management of these emerging fisheries in Tasmania.

Background

Ocean warming is already driving a shift in the distribution of marine species worldwide, with species from a wide range of taxonomic groups moving in a poleward direction. Due to disproportionate warming in Tasmania, driven by the extension of the Eastern Australian Current, this region is a hotspot for species extending their range, having more records of range shifting species than anywhere else in Australia. The potential implications of range shifting species for resource management is important to consider, as many stakeholder groups are already starting to adapt autonomously to these changes.

All three species in this study have occasional historical records of being present in Tasmanian waters ranging back to 1924 (Snapper and King George Whiting) and 1947 (Yellowtail Kingfish), but more recently, it is apparent that they are becoming increasingly abundant in Tasmanian waters. In their endemic ranges off southern mainland Australia, they are commercially and recreationally important. Thus, their increasing presence in Tasmanian waters is providing new fishing opportunities for recreational and commercial fishers. However, it is currently unclear as to whether these species have or are likely to become established as self-sustaining populations in Tasmanian waters, or simply persist as spill-over from their endemic range off mainland Australia. If the former is the case, it will be important to consider population attributes such as age structure, growth, mortality, and reproduction relevant to the Tasmanian populations when developing and refining management strategies to sustainably manage and maximize the opportunities these 'new' species bring. In addition, the broader ecosystem impacts of such range extending species, including competition with resident species at similar trophic levels, or increased predation to lower trophic levels are unknown but could have consequences for other recreationally and commercially important species or ecosystem function. Understanding these interactions will have benefits for the assessment and management of Tasmania’s natural marine resources more generally, ensuring Tasmania’s fisheries continue to thrive under climate change. As such, the overarching aim of this study was to understand the life-history, biology, distribution and future potential shifts of

Yellowtail Kingfish, Snapper and King George Whiting and the ecological implications of such shifts to inform management of these potential emerging fisheries in Tasmania.

Methodology

The Tassie Fish Frame Collection Program was launched by the Institute of Marine and Antarctic Studies (IMAS) from the University of Tasmania in December 2019 with the aim of creating a recognisable and ongoing fish frame collection program for Tasmania. The program has provided an opportunity for fishers to participate in citizen science through the donation of fish frames which complements the reporting of occurrence data to Redmap. A network of 16 drop-off points was established to provide strategic spatial coverage along most of coastal Tasmania. IMAS staff regularly liaised with the drop-off points to organise pick-up and transport of frozen fish frames to IMAS laboratories in Launceston and Hobart for processing. Additional fishery-independent sampling was conducted to account for individuals outside of the legal size limits, and historical data from the IMAS database was also used to supplement the data set.

From the samples and existing data, life-history characteristics were determined. Age was estimated using sectioned sagittal otoliths. Where data was sufficient, length and age frequency data were used to describe the regional and sex-specific stock structure and generate age-based catch curves to estimate total mortality. Natural mortality rates were estimated using a range of methods. To identify strong or weak recruitment years, interannual age frequency progressions were examined. Growth was estimated using von Bertalanffy growth models fit to size at age data. Gonads were also staged to determine maturity, and seasonality of spawning via examining monthly trends in the gonadosomatic index (GSI) which was calculated as the proportion of gonad weight versus length-weight converted or actual total weight. Diet was analysed by quantifying gut-contents to the lowest taxonomic level possible and mean proportional composition of prey taxa were summarised.

Species distribution models (SDM) were developed using occurrence records of each species from the Tassie Fish Frame Collection Program, Redmap and open access databases including the Atlas of Living Australia and Reef Life Survey and matching them to location-specific satellite-derived oceanographic data and mapped habitat data. Generalised additive models were used to spatially predict areas of suitable habitat. Climate projection data was then used to predict how habitat suitability, and temporal persistence of suitable habitat may change into the future (range: 2036-2065 centred around 2050).

Ecosystem modelling was also conducted to determine potential shifts to food-web dynamics under both climate change projections and range shift distribution scenarios of the three species. This was done by updating an existing Atlantis ecosystem model for the south-east Australian region, including coastal and open ocean regions, with new biological and ecological data derived from the other components of this project. To explore the effects of projected climate change on the Tasmanian marine ecosystem, we ran the model under the OFAM-v3 control (no greenhouse emissions) scenario out to 2060 and compared it to the *Status Quo* run under the RCP8.5 climate projections. To explore the effects of species redistribution, three scenarios were developed to reflect the trends of increasing population of each of the species of interest toward the south. These included: i) population density is even across portions of the modelled domain (boxes) around mainland Australia with median depth < 100 m, broader northern Tasmania was populated at 50% of mainland density, while the species are not present in the rest of Tasmania, ii) density is even across all boxes with a median depth < 100 m off the north of Tasmania, but only half the density off the east and south Tasmania, and iii) species present in every box at equal density off the north, east and southern parts off Tasmania with a median depth < 100 m.

Results/key findings

i) Ecosystem effects

This study used a biogeochemical and whole of ecosystem Atlantis model of south-eastern Australia to run a series of climate change and species distribution scenarios, representing incremental biomass changes down the eastern and southern parts of Tasmania. The Atlantis South East Australian model was very responsive to climate change with biomass projected to greatly increase for King George Whiting with more variable spatial dynamics observed for Yellowtail Kingfish and Snapper. This result compliments that of the SDMs, where habitat suitability was also predicted to increase across the state for King George Whiting and Snapper, and the temporal persistence of suitable habitat in south-east Tasmania is predicted to increase for Yellowtail Kingfish under the tested scenarios. Most of the temporal and spatial dynamics were determined by trends in the primary producers including picoplankton and macrophytes. In contrast, under the redistribution scenarios of the three focal species, the effects predicted by the ecosystem model were less pronounced. For example, single species redistribution scenarios typically only showed changes in relative abundance of less than ± 5% of trophic groups or species, with the species groups most negatively impacted including dominant prey items or potential competitors. Therefore, under both climate change and species redistribution scenarios, there was limited evidence of any ecosystem collapse driven by the range extending species examined.

ii) Yellowtail Kingfish

Yellowtail Kingfish are a pelagic species and are known to undertake long distance migrations. It is likely that juvenile fish are using the East Australian Current to migrate south-ward of their endemic range during the summer into Tasmanian waters, and then return to the waters adjacent to mainland Australia in the cooler seasons. There is also anecdotal evidence of larger Yellowtail Kingfish becoming seasonally present on the north-west coast of Tasmania, which are likely migrating from populations centred in the Great Australia Bight and migrating down during summer months with the Leeuwin current. However, neither of these theories have been verified via molecular analysis. Yellowtail Kingfish adjacent to the east coast of Tasmania were generally small (< 600 mm FL), immature fish (2–3 years) which were seasonally present from November to April. Due to the small sample size, and age truncation of the samples, we were unable to model growth rates. Also, given the size of the fish, it is highly unlikely that Kingfish spawn in Tasmanian waters as there was only one reproductively active female in our sample.

The results of the diet analysis suggest that Yellowtail Kingfish in Tasmania are generalist and opportunistic carnivores, 93% of stomachs examined were empty but for those with content the majority contained highly digested fish which could not be taxonomically resolved beyond the level of class. As such, our diet analysis suggests that Actinopterygii (fish) and Malacostraca (crabs) were the most dominant prey classes among Yellowtail Kingfish stomachs both in terms of frequency of occurrence (Actinopterygii: 95%, Malacostraca: 10%) and mean proportion (100%) of prey. The crabs that appeared in the stomachs of two fishes were the long pelagic megalopa larvae of the red rock crab *Plagusia chabrus*. Therefore, this generalist diet may be a facilitating factor for Yellowtail Kingfish at its range edge, as generalist diets are positively correlated with the capacity of a species to undergo a range shift.

Species distribution modelling determined that Yellowtail Kingfish have been moving southward over the past ~20 years, and that the temporal persistence (months per year) of suitable habitat is predicted to further increase in Tasmania as the climate continues to change. However, even under extreme warming (RCP8.5), it is unlikely that it will be warm enough for Yellowtail Kingfish to be present in Tasmania year-round. However, results of the SDM indicate that it is likely that

Yellowtail Kingfish may increase in seasonal abundance and in range extent under future climate change.

iii) Snapper

Snapper in our study reached a maximum size of 799 mm FL, and ranged in fork length from 241–700 mm FL for females, 233–633 mm FL for males, and 150–316 mm FL for juveniles of indeterminate sex, and 268–779 mm FL for individuals lacking gonadal material. There were no statistical differences in age/size structure between sexes, however we did find regional differences in Snapper age/size structure where individuals were larger and older in the north versus the south. There were gaps in age/size structure of our sample, however it was unclear whether this was sampling bias or evidence of interannual variation in recruitment. The maximum age of our sample was 19.2 years (sex not determined), 18 and 12.8 years for females and males respectively. We are also aware of a large (990 mm TL, 11.6 kg) Snapper that was caught in Tasmania that was aged at 27 years by the Fish Ageing Service. These age estimates however are still well below the maximum age reported for Snapper in Australia (41 years: East Coast Biological Stock, QLD/NSW). We found no statistical differences in the growth of Snapper between sexes or regions.

It was not possible to fit a logistic regression to estimate a size at 50% maturity (L_{50}) for Tasmanian Snapper. This was due to the relatively small sample of females with hydrated oocytes (n = 4) or spermiated males (n = 9), and individuals within the size range of 350–400 mm FL (n = 32, 7.3% of sample), coupled with difficulties discerning between virgin versus inactive gonads in larger fish. The smallest reproductively active female and male in our sample measured 315 mm and 310 mm FL respectively, where both individuals were approximately four years of age. We also found that the timing of reproduction in Tasmania is unique, occurring in the austral summer, evidenced by a high proportion of mature gonads, suggesting that Snapper in Tasmania may require the higher temperatures of early austral summer for the onset of spawning. While Snapper are potentially spawning in Tasmania, it is not clear whether the environmental conditions are suitable for survival of eggs and larvae, and whether the population is self-sustaining. However, SSTs in Tasmania are highest in January and February (i.e. late austral summer), with temperatures typically ranging between 18–20 °C, which may support a secondary peak in spawning, and also facilitate settlement and recruitment. Understanding potential sources of recruitment (i.e. local recruitment, sourced from the mainland, or a combination of both) could be resolved by the inclusion of a robust sample of Tasmanian fish in a molecular analysis of the population structure of Snapper in south-east Australia.

Diet analysis suggests Snapper in Tasmania are opportunistic generalist predators, consuming a range of benthic invertebrates and fish, with patterns in diet composition varying with fish size and demersal habitat as reported elsewhere in Australia and New Zealand. Bivalves, malacostracans (non-barnacle crustaceans), polychaetes, gastropods and fish were the five most dominant prey classes by both frequency of occurrence (bivalves: 49.38%, malacostracans: 44.44%, polychaetes 16.05%, gastropods 14.81%) and as mean proportion of stomach contents, together representing a mean of 82.36% of stomach contents by weight. Bivalves were predominantly comprised of Doughboy Scallops (*Mimachlamys asperrima*), which occurred in 49.38% of Snapper stomachs and comprised most stomach contents at the three northern sites (Wynyard, Devonport and Bridport). While there was a difference in diet for Snapper between the northern and southern regions, this difference was likely driven by the high abundance of Doughboy Scallops in the north, and the opportunistic feeding strategy of Snapper. Due to the species' broad and opportunistic demersal feeding habit, increasing abundance of Snapper (and especially large Snapper) could potentially provide some buffering effect against potential future introduction of non-native invertebrate species.

Spatial predictions of suitable environmental habitat show that there has been ‘suitable’ habitat for Snapper around the entire Tasmanian domain since at least 1998. There was evidence of seasonal variation in habitat suitability for Snapper, with more favourable habitat in the warmer seasons (i.e. summer and autumn), with evidence of a very small increase in suitable habitat from 1998–2018 within the north-east, south-east and west coasts. When comparing the averaged hindcasted (1998–2018) and forecasted (2036–2065) time periods, there was a consistent predicted increase in suitability in the winter across all regions ranging from 126.33–249.31 %, with the greatest amount of change predicted for the winter in the North-East-North (NEN) region. There was also a smaller increase in habitat suitability predicted for the summer (range: 21.01–40.27 %) and spring (range: 35–60.89 %) seasons. As Snapper at higher latitudes are spring/summer spawners, this increase in summer habitat suitability may facilitate successful spawning, recruitment and larval growth, and the subsequent increase in suitability in the winter may potentially lead to successful overwintering of new recruits or juveniles. Current satellite-derived mean summer SSTs in Tasmania range between 16.62 °C–19.12 °C which suggest potential for spawning in the north only. However, under future predictions, summer SSTs in all regions (except the west coast) exceeded 17.33 °C, which is above the reported temperature requirement for Snapper (~17 °C). This would suggest that spawning could occur further south than current predictions. Furthermore, it is predicted that autumn temperatures will exceed 15 °C in all regions, which is within the thermal range reported for egg and larval survival.

iv) King George Whiting

King George Whiting in the north-west of Tasmania were larger and older compared to those from mainland Australia. However, King George Whiting on the east coast had a smaller size composition. The east coast comprises several semi-enclosed bays, high in seagrass cover, which may represent a nursery area for King George Whiting. Fork length of King George Whiting sampled from Tasmania ranged from 53 to 619 mm and spanned ages 0 to 18.8 years. Samples from the north-west coast of Tasmania contained a greater proportion of larger (> 470 mm) fish that were older than 5 years of age. Growth models (pooled across all regions) for both sexes suggested that asymptotic size is attained at approximately 10 years of age, at a mean asymptotic size of 560 and 504 mm FL for females and males respectively. Across all regions in Tasmania, except for the east coast, there are differences in growth rates between sexes, where females had a greater asymptotic length than males with the magnitude of this difference increasing with increasing age

The smallest reproductively active female and male specimens measured 340 and 320 mm FL respectively and were both 2.9 years of age. Length and age at 50% maturity were estimated to be 407 and 403 mm FL, and 4.4 and 4.6 years for females and males respectively. Timing of gonadal development for both sexes was similar and evident from February to June. Seasonal changes in the proportions of reproductive stages and mean GSI values of “maturing adult” fish (≥ 3 years) were markedly different between regions. Furthermore, the large proportion of small and young fish on the east coast, and central north coast suggests that the sheltered waterways in these regions are likely to be areas for settlement and recruitment of juveniles. We also found recruitment to be variable, with strong cohorts in 2001, 2003, 2007, 2014, 2015 and 2017, potentially driven by changes in SST and increased flow rates of regional currents. This study provides additional evidence to support the distinction of two stocks in Tasmania. We hypothesize that a stock of adult King George Whiting resides in the waters off Flinders Island with their spawned eggs and larvae being dispersed in a southerly direction via a weakening East Australian Current during autumn and winter and Georges Bay functioning as the key nursery area for this stock. In contrast, eggs and larvae from adult King George Whiting spawning in Bass Strait are likely to be dispersed in an easterly direction by the dominant currents in the area in autumn/winter and the Tamar River and Port Sorell estuaries being the key nursery areas for this stock.

Diet analysis suggests that polychaetes and malacostracans were the most dominant prey classes among King George Whiting stomachs both in terms of frequency of occurrence (polychaetes: 68.8%, malacostracans: 55.19%) and mean proportion (84.8%) of prey, with no statistical differences detected across size classes or regions.

Spatial predictions of suitable environmental habitat from 1998 to 2018 show seasonal variation in the suitable habitat for King George Whiting, with more favourable habitat in the warmer seasons (i.e. summer and autumn versus winter and spring), but there was no increase in suitable habitat between 1998–2018. When comparing averaged hindcasted (1998–2018) and forecasted (2036-2065) time periods, the winter season is predicted to be more suitable under the IPCC RCP8.5 scenario, evidenced by the increase in proportional change of small (416 m^2) grid cells across the Tasmanian domain. This could potentially facilitate successful recruitment during the late autumn and winter (i.e. during their spawning season), and indicates King George Whiting may increase in abundance and range in future.

Implications for management

Modelling predicts an increase in habitat suitability across the state for Snapper and King George Whiting, and an increase in temporal persistence of suitable habitat for the seasonal migration of Yellowtail Kingfish along the north and east coasts of Tasmania. It was also predicted that there is a low chance of ecosystem collapse due to the increased presence of the three species assessed here. This provides a unique opportunity to proactively manage the emerging fisheries for these species. Due to the relatively short sampling period of this study (~2 years) we only have a snapshot of the population structure, and limited information on interannual recruitment. Therefore, for all three species, ongoing monitoring is critical not only to monitor population structure as well as spawning dynamics and recruitment trends, but to monitor these species at their range edge, to ensure understanding of how their populations are likely to continue increasing and establishing in Tasmania. However, should there be evidence of change (i.e. higher abundance, bigger individuals, occurrence earlier or later in the year), management strategies should be flexible and adaptive to change. Nevertheless, this project has provided baseline information on the life-history and biology of these range shifting species, useful for informed adaptive management.

i) Yellowtail Kingfish

For Yellowtail Kingfish, there is not enough information, or abundance of Yellowtail Kingfish to be included in the Tasmanian Fishery Stock Assessment. Therefore, we suggest ongoing monitoring via engagement with citizen science, fish frame collection, and fishery independent sampling. As the population is small, the current minimum legal size (MLS) of 450 mm TL, although below the size at 50% maturity reported on the mainland, means the species is available to the fishery. Therefore, if the MLS was increased there would be very few fish available to the fishery due to their small size in the state. The number of fish caught in Tasmania is unlikely to have a major impact on the Eastern Biological Stock (EBS) as a whole, but growth overfishing should be carefully monitored in the EBS assessment.

ii) Snapper

Species subject to high exploitation and variable recruitment such as Snapper are susceptible to age truncation, which can severely impact a population's capacity to respond to environmental change or recover from sustained or increased fishing pressure. Management strategies to avoid age truncation include protecting the older/larger cohorts and ensuring sufficient recruitment into the fishery. Victoria has adopted this strategy by limiting the harvest of larger individuals (> 40 cm) to 30% of the bag limit. Given that our models suggest that Snapper in Tasmania are most similar to those in Victoria (although slightly smaller and slower growing), we suggest that this strategy of limiting the harvest of larger individuals may also safeguard Snapper in Tasmania from age truncation, as the proportion of large individuals (> 550 mm FL) is low (12%). However, the current MLS in Victoria is 280 mm TL, which equates to approximately 2.5 years of age. Our results suggest that there is no sign of reproduction near the MLS in Tasmania of 300 mm TL (260 mm FL). At this size, Snapper in Tasmania are approximately 3 years old. Therefore, increasing the MLS to 380 mm TL (i.e. ~4 years) would offer at least one year of protection of spawning-capable individuals before recruiting into the fishery.

iii) King George Whiting

Our data suggest similar estimates of age and size at maturity of King George Whiting in Tasmania as populations in Western Australia and South Australia. Due to the genetic distinction between Tasmanian stocks and the Australian mainland, it appears that the observed increase in King George Whiting abundance in Tasmania is not a range extension from an Australian mainland stock, but rather a climate driven change in the coastal environment leading to more positive outcomes for the pre-existing Tasmanian adult stock's reproductive activities and subsequent larval survival and recruitment. King George Whiting in Tasmania had a similar mean asymptotic length as the west regions of Western Australia for both males and females, and was larger than those for South Australia, Victoria and the south regions of Western Australia. The MLS in Tasmania is the largest of all states at 350 mm TL. This is below the estimated size at 50% maturity identified in this study (~400 mm FL, 425 mm TL). If as expected, fishing pressure increases, a more precautionary management approach could include an increase in the MLS, and potentially temporal or spatial closures to afford greater protection of the spawning stock. More information and thus ongoing monitoring is needed to assess whether other management options such as maximum size limits and or temporal/spatial closures are needed in the future as the fishery develops.

Recommendations

While the data collection for this project has been successful in determining baseline information of key range shifting species into Tasmania, ongoing monitoring is necessary to determine potential changes to age/size structure, recruitment variability, and areas of potential spawning/recruitment. This is important to gain an accurate representation of population dynamics of these three species in Tasmania over time, and to be flexible and ready to implement proactive strategies to sustainably manage these emerging fisheries. Ongoing monitoring and stock assessment of King George Whiting in Tasmania would be particularly useful given there is evidence of a self-sustaining population, and fishing pressure is likely to continue to increase, perhaps as more traditional target species (e.g. Flathead) appear to be declining in coastal/inshore habitats. Lastly, for site-attached species, such as King George Whiting, it is also important to consider how their preferred habitat (i.e. seagrass) is likely to shift under future change.

For Snapper, the source of recruitment for the Tasmanian population remains unclear. Understanding genetic connectivity between fish caught in Tasmania versus the mainland would provide evidence of whether the population in Tasmania is self-replenishing, or simply persists as spill-over from one or more mainland stocks. Further, while correlative models such as SDMs provide a good estimate for habitat suitability, a limitation is that they cannot account for regional acclimation to temperature. Mechanistic studies to identify critical thermal minimums and optimums are necessary to gauge cold acclimation at the range edge.

Engaging with several citizen science initiatives was critical in the successful data collection/fish frame donation for this project. Underpinning this success, was the engagement across complimentary programs (i.e. Tassie Fish Frame Collection Program and Redmap) to reach a wider audience. An online hub or central source for all marine citizen science programs would be useful to maximise engagement and involvement by fishers in citizen science programs, and for the general public to engage and potentially provide information irrespective of their interests or abilities (i.e. photography, diving, fishing, education, clean-ups). Cross-pollination across different citizen science initiatives provides the opportunity to share resources, and cross-promotion ultimately maximizes community outreach and data collection efforts. However, the success of such citizen science programs requires appropriate and consistent support and resourcing.

Keywords

Yellowtail Kingfish, Snapper, King George Whiting, citizen science, species redistribution, range extension, emerging fisheries, life-history, species distribution model, Atlantis ecosystem model, diet, adaptive management

General Introduction

Globally, the oceans have buffered the world over the last century or so against greater impacts of climate change. The oceans have absorbed approximately 90% of the additional heat trapped in the atmosphere and 25% of the additional carbon emitted (IPCC 2019)—without this ‘service’ the impacts of climate change around the world would be much more severe. However, the associated physical and ecological changes in Australian coasts and oceans have been substantial, with major implications for marine systems. Since 1970, waters have warmed by 1–2 °C, are 26–30% more acidic (IPCC 2019), changes in atmospheric conditions have driven major shifts in the eddies, currents and upwellings of the oceans (Martinez-Moreno et al. 2021), and marine heatwaves have increased in frequency, duration and intensity (Laufkötter et al. 2020). However, there is large geographical variation in the rates of change in these parameters and processes. For example, the south-east of Australia is a ‘warming hotspot’, in the top 10% for rates of ocean warming (Hobday & Pecl 2014), primarily due to a strengthening of the East Australian Current (Oliver et al. 2015). It is also a region that has experienced several marine heatwaves in recent years (Holbrook et al. 2019), with peak intensities 1.5–3°C degrees above the long-term climatology in the Tasman Sea in 2015/2016, 2017/2018 and 2018/19 (Oliver et al. 2021).

The physical changes in oceans over recent decades have resulted in large changes in the biology and ecology of marine ecosystems, with the extent of these changes greater in regions that have warmed the most (Poloczanska et al. 2013). Approximately 45% of Australia’s coastal habitats have been extensively impacted so far by abrupt and extensive mortality of key habitat-forming organisms, primarily corals, kelps, seagrasses, and mangroves (Babcock et al. 2019), and including the loss of 95% of Tasmania’s giant kelp habitats (Butler et al. 2020). Moreover, in Tasmanian coastal systems, new occurrences or increased prevalence of disease, toxins and viruses have been recorded, including a paralytic shellfish toxin that resulted in global product recalls of mussels, scallops, oysters, abalone and Southern Rock Lobster, and a virulent virus causing Pacific Oyster Mortality Syndrome (POMS) that resulted in 90% of farmed oysters dying—both first detected during heatwave events (de Kantzow et al. 2017).

In addition to major climate-driven habitat changes, and new episodes of disease and virus outbreaks, extensive changes in the life-history, distribution and abundance of species have been observed, resulting in rapid shifts in the community composition, structure and integrity of Australian marine ecosystems. Substantial changes in the body size of species have been recorded with 55% of fish species around the Australian coastline smaller in warming waters, but 45% of species actually increasing in size with warming (Audzijonyte et al. 2020). These changes in body size equate to a rapid ~40% change in body size change per 1 °C of warming. The large change in body size—one of the most important life-history traits—combined with the variable responses of fish size to warming may lead to unexpected impacts on ecosystem restructuring, with potentially greater consequences than if all species were shrinking (Audzijonyte et al. 2020). At the same time as these major changes in the size-structure of communities, community composition itself is changing with one of the most pervasive responses to climate-driven warming being the redistribution of species as existing habitats at equatorward limits become too warm for survival and new habitats at poleward limits become warm enough for species to survive (Pecl et al. 2017). There is, however, great taxonomic and regional variation in the pace and magnitude of these shifts in species distributions, or ‘range shifts’.

The east coast of Tasmania is a hotspot for species range shifts, as well as warming, with almost 100 species documented thus far shifting either from the Victorian or New South Wales coast into Tasmanian waters, or from the north of Tasmania and shifting further south. Tasmania’s coastal ocean is associated with more records of range shifting species than any other region of Australia’s surrounding ocean, including algae, bivalves, gastropods, octopus, starfish, sea urchins, crustaceans, sharks and rays, ascidians, and fish (Gervais et al. 2021). However, there is a huge range in rates and magnitude of shifts documented, much of which can be explained by species traits, along with the local climate velocity and current speed and direction (Hunt et al in review). Off the east coast of Tasmania, greater extension distances are associated with pelagic species compared to demersal or benthic ones, and with carnivores or omnivores rather than herbivores. Furthermore, both current speed and alignment of ocean currents with thermal gradients influence the early stages of range extensions (Hunt et al in review).

Not all species shifting into new regions will result in ecosystem disturbance, but some species can have impacts equivalent to invasive species (Ling 2008) by altering food webs, competing for habitat with existing species, or other species interactions. Importantly, we don't yet know much about the net effect of many species shifting at the same time within a given region (Marzloff et al. 2016, Bonebrake et al. 2018), but we do know that wherever shifts within a particular region or for a given taxa are examined, between 10–90% of species are already shifting with climate (Melbourne-Thomas et al. 2021). The potential ecosystem implications of shifting species are largely unknown, especially for regions like the east coast of Tasmania with many species shifting all at the same time.

Potential implications of range shifting species for resource management is important to consider, especially as many stakeholder groups are already starting to adapt autonomously to the changes (Pecl et al. 2019). In Tasmania, charter operators are advertising trips for 'new' species, fishers have made changes to product handling and landing practices, and aquaculture operators have changed farming operations (Pecl et al. 2019). Assessment of key biological and ecological parameters of range shifting species, particularly in new areas of the range extensions, is of critical importance to underpin comprehensive understanding of species characteristics at the extending range edge, what novel species interactions may be occurring, and how 'new' species may be fitting into the changing ecosystem.

Range shifts into new regions occur as a process over time, with a series of linked stages of 'arrival', 'population increase' and then 'population persistence' (Bates et al. 2014). In this study, we focus on three key species whose range is extending into Tasmania and where they have become increasingly abundant in Tasmanian waters, providing new fishing opportunities for recreational and, to a lesser extent, commercial fishers. Species in this group include Yellowtail Kingfish, Snapper, King George Whiting. At this stage, it is unclear as to whether these species have or are likely to become established as self-sustaining populations in Tasmanian waters, or simply persist as spill-over from populations that are centered off mainland Australia. If the former is the case, it will be especially important to consider population attributes such as growth, mortality and reproductive dynamics relevant to the Tasmanian populations when developing and refining management arrangements to maximise the opportunities these 'new' species bring. In addition, the broader ecosystem impacts of such range extending species, including competition with resident species at similar trophic levels, are unknown but could have consequences for other recreationally and commercially important species. Understanding these relationships will have benefits for the assessment and management of the Tasmanian recreational fishery more generally.

Objectives

- Develop a program for ongoing collection of biological samples and data of key range shifting fish species using citizen science initiatives engaging with the recreational fishing community.

- Develop geographically discrete life-history parameters for key range shifting fish species in Tasmania to inform management decisions.

- Determine the diet composition of key range shifting fish species to refine parameterisation of an ecosystem model.

- Develop species distribution models that utilise oceanographic climate change projections to predict the future presence and persistence of the key target species in Tasmania.

- Utilise the Atlantis ecosystem model framework to predict ecological impacts of increasing abundance of key range shifting fish species in Tasmania.

General Methods

Objective 1–Develop a program for ongoing collection of biological samples and data of key range shifting fish species using citizen science initiatives engaging with the recreational fishing community

The Tassie Fish Frame Collection Program was launched by IMAS in December 2019 with the aim of creating a recognisable and ongoing fish frame collection program for Tasmania. The program has provided a platform to engage with the Tasmanian fishing community, and an opportunity for fishers to participate in citizen science through the donation of important biological samples and data (i.e. fish frames) for the current project. The program follows similar programs in other Australian states (e.g. *Send us your skeletons* in Western Australia) which have proved to be successful for a number of years.

To enable the Tassie Fish Frame Collection Program to operate on a state-wide scale, a network of 16 drop-off points was established to provide strategic spatial coverage along most of coastal Tasmania (Appendix A). These drop-off points were predominately tackle stores, which provide a natural point for knowledge sharing and communication around fishing for the target species. IMAS staff members regularly liaised with the drop-off points to organise pick-up and transport of frozen fish frames to IMAS laboratories in Launceston and Hobart for processing (Figure 1).

Figure 1. A 790 mm Yellowtail Kingfish caught in southern Tasmania by Will Thorpe (left), who donated the frame to the Tassie Fish Frame Collection Program via his local tackle store Fisherman’s Shed (right) who are one of 16 drop-off points for the program.

Engagement on the Tassie Fish Frame Collection Program has involved both online and on-ground components. Coinciding with the program launch in December 2019 was the Tassie Fish Frame Collection Program Facebook Page, enabling online engagement with a wide audience. On-ground engagement involved providing presentations at angling club nights, boat ramp signage promoting the program, and opportunistic face-to-face discussions between project team members and fishers at popular fishing spots, boat ramps and tackle stores.

The Tassie Fish Frame Collection Program has provided a significant number of fish frames and subsequent data for the current project on range extending species. The program has been largely successful due to our strong rapport with a core group of ~40 recreational anglers around Tasmania who regularly donated fish frames. Additionally, frames were also donated by commercial fishers and fish processors.

Within the project sampling period (July 2019–July 2021) a total 801 fish frames were donated from across Tasmania, which included 83 Yellowtail Kingfish, 264 Snapper and 454 King George Whiting (Table 1). This provided important spatial and temporal data coverage to compliment data collected by the project team.

In addition to donated frames, IMAS conducted fishery-independent sampling within the project sampling period yielding 259 additional fish (Table 1). The project team devoted sampling efforts to fishery-independent data, such as specimens below the legal size limit. All existing IMAS data (pre-July 2019) for these species was also collated for this project (Table 2). This included data on fish collected by IMAS for research and frames donated by the recreational and commercial sectors (Table 1).

Table 1. Sources and number of fish frames used for sampling for each study species during this FRDC project.

	Frame donation		**IMAS Research**	**Total**
	Recreational	**Commercial**		
2019-2020				
Yellowtail Kingfish	47 (15 anglers)	0	2	49
Snapper	163 (18 anglers)	0	46	209
King George Whiting	136 (11 anglers)	127	111	374
2020-2021				
Yellowtail Kingfish	36 (16 anglers)	0	2	38
Snapper	62 (15 anglers)	39	10	111
King George Whiting	143 (11 anglers)	48	88	279

Table 2. Number of fish frames used for sampling for each study species per year.

	2011	**2012**	**2013**	**2014**	**2015**	**2016**	**2017**	**2018**	**2019**	**2020**	**2021**	**Total**
Yellowtail Kingfish	0	1	0	0	59	60	32	19	21	39	38	269
Snapper	0	0	13	8	8	31	27	8	72	177	93	437
King George Whiting	3	28	112	96	46	53	157	516	199	386	185	1781

Objective 2–develop geographically discrete life-history parameters for key range shifting fish species in Tasmania to inform management decisions

Laboratory processing of samples

In the laboratory, frozen samples were thawed and total weight (TW ± 1 g; if whole fish was available), total length (TL ± 1 mm), fork length (FL ± 1 mm), sex, gonad weight (GW ± 1 g) and reproductive stage were recorded. For Yellowtail Kingfish and Snapper we used macroscopic staging keys outlined in West (1990), and for King George Whiting we used the species-specific macroscopic staging key outlined in Jenkins et al. (2016). If both gonad lobes were damaged, gonad weight was not recorded. If one gonad lobe was intact the weight of this lobe was recorded and doubled to represent total gonad weight. In cases where no gonads could be found, sex was not defined. When stomachs contained food items (post 2018) the whole stomach was removed and weighed, stomach contents were placed into jars containing 70% ethanol for later stomach content identification. The emptied stomach was reweighed to enable calculation of stomach content weights. Samples collected prior to 2018 were not processed to assess diet. Sagitta otoliths were extracted, cleaned and stored to dry during sample processing. Due to the opportunistic collections and various sources of data it is important to note that sampling effort cannot be standardized across capture years or regions. As such viewing absolute numbers between regions or years must be viewed with caution.

Otolith processing, reading and age determination

Whole sagittal otoliths were weighed, set in resin and photographed prior to being sectioned with a Gemmasta faceting saw. Two to four 300 µm sections closest to the otolith primordia were selected and mounted onto a microscope slide. Mounted sectioned otoliths were viewed under a stereo microscope using either reflected lighting (King George Whiting) or transmitted lighting (Snapper & Yellowtail Kingfish). During reading the number of complete opaque bands counted, the marginal edge classified, and the readability recorded (1st reading). Marginal edges were classified as a relative proportion of the penultimate zone (PZ = last fully completed translucent and opaque zones). Edge types were classified as narrow (translucent margin < 20% of PZ), intermediate (translucent margin between 20–50 % of PZ), wide (translucent margin > 50% of PZ) or edge (opaque margin). Opaque margins were not counted as complete. Readability of sectioned otoliths were assigned based on the number of age estimate interpretations within the sectioned otolith according to Table 3.

Table 3. Criteria for readability scores of sectioned otoliths.

Readability score	**Interpretation**
1	Sample has excellent readability, increments exceptionally clear throughout section
2	Sample is unambiguous, but some areas not as clear as 1
3	Sample may be subject to 2 interpretations
4	Sample is subject to multiple interpretations
5	unreadable

Otolith reading occurred without knowing the fish size, capture date or previous age estimate. Biological outliers and interpretation disparities between 1st and 2nd readings or edge classification were re-examined to assign a final count and edge type.

Modelling of growth

Growth rates were determined by fitting a von Bertalanffy growth model to size at age data:

$$L_t = L_\infty\left[1 - e^{-k(t-t_0)}\right]$$

where L_t is the fork length at age t, L_∞ is the asymptotic fork length, k is the growth rate and t_0 is the theoretical age at zero length. Model parameters were estimated by ordinary least squares regression. Models were fitted to both individual size at decimal age data and mean size at age data. Differences in mean size at age (regionally and by gender) were analysed using 2-tailed, equal variances *t*-tests, when variances were not equal, 2-tailed unequal variances *t*-tests were performed. *T*-tests were only performed when age classes had more than 10 individuals in each age class. Size at age data from indeterminate and unknown sex individuals were not included in data used for comparing sex-specific model parameters. Hedges G values were used to indicate the effect size when significant differences were detected, and adjusted to account for variable variance, and small (< 50) or variable *N* values (Hedges 1981, Glen 2021). Effect sizes of 0.2, 0.5 and 0.8 were defined as small, medium, or large. Differences in regional and sex-specific growth models were tested, using Analysis of Residual Sum of Squares (ARSS; Chen et al. 1992). To account for significant differences being detected due to base case models being influenced by available age ranges when fitting the models, ARSS were also performed on data constrained to a standardized age range. When differences were detected by the

ARSS, likelihood ratio tests were performed to determine which model parameter(s) were causing the difference as outlined in Haddon (2011).

Chi square (χ^2) tests were performed to assess if sex ratio in 20 mm fork length size classes or age classes were different from an expected 1:1 ratio. χ^2 tests were only performed when total sample numbers were greater than 10.

Size structure and mortality estimation

Length and age frequency data was used to describe the regional and sex-specific stock structure and generate age-based catch curves to estimate total mortality (*Z*). To identify strong or weak recruitment years, interannual age frequency progressions were examined. Where there was sufficient data to do so, sex- and region-specific mortality estimates were generated and examined, however due to either insufficient data or a failure to detect significant differences, data was pooled. Data selected for age-based catch curve analysis used the peak abundance value and all values after this point until the first age class with only one individual. Two methods were used to estimate whole stock total mortality rates, direct fitting of an exponential decay curve to age frequency data and an age-based catch curve on the logged abundance data.

Natural mortality (*M*) was estimated using four methods: an updated Hoenig$_{nls}$ estimator (Then et al. 2015):

$$M = 4.899 \ \times t_{max}{}^{-0.916}$$

Where t_{max} is the maximum observed age,

Tanka's (1960) 1% and 5% population survival to the maximum observed age (Kenchington 2014):

$$M = -Ln\left(\frac{\propto N_s}{t_{max}}\right)$$

Where $\propto N_s$ is the population survival proportion,

and Pauly's empirical natural mortality estimate (Pauly 1980),

$$M = \exp(-0.0152 - 0.279 Ln\ L_\infty) + 0.6543 Ln\ k + 0.4634\ Ln\ T$$

Where L_∞ and k are parameters derived from the von Bertalanffy growth function and T is the average annual sea surface temperature (°C) at the area of capture. As Snapper are distributed around Tasmania the mean annual temperature was set at 14.97 °C. As King George Whiting are restricted to northern Tasmania mean annual temperature was set at 15.94 °C (min. longitude:143°E, max. longitude: 150°E, max latitude: -42.1°S, min. latitude: -39.5°S). Annual mean sea surface temperature (SST) around Tasmania (min. longitude:143°E, max. longitude: 150°E, max latitude: -45°S, min. latitude: -39.5°S) was determined by aggregating satellite derived SST data to an annual temporal resolution and excluding values where depths exceed 200m (i.e. the continental shelf). SST data was sourced from Copernicus Marine Environment Monitoring Service, and gridded bathymetry data was sourced from The General Bathymetric Chart of the Oceans (for full details see Table 4).

Fishing mortality (*F*) was subsequently calculated by subtraction of natural mortality from total mortality, $F = Z - M$.

Reproductive biology

Length-weight (LW) relationships were generated from sex combined size and weight data for King George Whiting and Snapper (King George Whiting n = 416 individuals (FL range of 53–585 mm); Snapper n = 85 (FL range of 168–704 mm)). LW relationships were used to predict total weight (PTW) of each fish processed. The gonadosomatic index (GSI) was calculated as gonad weight /

PTW *100. Seasonal reproductive trends of "maturing adults" were examined by pooling month of capture across all sampling years and excluding all fish below the size of the first observed sexually mature individual for each sex and species.

Size and age at maturity was modelled using a logistic model:

$$P_{exp} = 1/[1+\exp\{-r^*(L_t-L_{m50})\}]$$

where L_t is the fork length at age t, r is the intrinsic rate of increase and L_{m50} is the fork length where 50% of individuals would be mature. For age at maturity modelling A_t and A_{m50} were substituted for the L_t and L_{m50} parameters. Model parameters were estimated using the same fitting routine for growth modelling based on the observed sex-specific proportions of mature individuals in 10 mm fork length size classes (or age class). Due to data limitations, size maturity models could only be developed for King George Whiting with data restricted to the spawning season (February–May).

Objective 3–Determine diet composition of key range shifting fish species to refine parameterisation of an ecosystem model

Stomachs were emptied into jars and stored in 70% ethanol for subsequent analysis. Stomach contents were then examined under compound dissecting scope. Contents were identified to the finest taxonomic resolution possible. Each prey taxon was weighed (wet mass) to the nearest 0.01 g to calculate proportional composition of stomachs (the mean percentage of each prey taxon's mass divided by mass of all identifiable prey taxa in each stomach). When it was not possible to separate individual prey items to weigh them due to the state of digestion of the sample, the percent weight of each stomach sample constituted by each prey item was visually estimated (typically to the nearest 5%, or nearest 1% for items estimated as less than 5% of the sample).

For each prey taxon identified in each species' stomach samples, the frequency of occurrence (number of stomachs in which a prey taxon occurred as a percent of the total stomachs containing identifiable food for each species) and mean proportional composition of prey taxa were summarised. To characterise coverage of prey taxa across species stomach samples, prey accumulation curves were calculated at the lowest highly resolved taxonomic level with the R package 'vegan' version 2.5–7 (Oksanen et al. 2020) with 1000 random permutations. The total number of prey taxa for each species in the survey area (i.e. gamma diversity) was extrapolated with a bootstrap estimator (Smith & van Belle 1984). Relationships between prey taxa were characterised with a correlation matrix of the pairwise Bray-Curtis dissimilarity of prey frequency of occurrence between samples. For each species, pairwise Bray-Curtis dissimilarity matrices were calculated across the frequency of occurrence of prey taxa. As Snapper and King George Whiting had fairly comprehensive sampling in multiple regions, their samples were stratified by region of collection (north versus south for Snapper, north vs east for King George Whiting), and permutational ANOVAs (McArdle & Anderson 2001) were used to test whether fish length and region were significant predictors of prey taxa occurrence across stomachs. As such these regions were defined as North TAS and South TAS for Snapper and North coast and East coast for King George Whiting for the diet analyses. Relationships between stomachs, prey taxon and significant predictors were visualised with nonmetric multidimensional scaling biplots.

Objective 4–Develop species distribution models that utilise oceanographic climate change projections to predict the future presence and persistence of the key target species in Tasmania

Species distribution models (SDM) have been used widely in ecology and conservation as a tool for exploring trends in species diversity (Graham et al. 2006) and predicting the effects of climate change on species redistributions (Araujo et al. 2005, Thomas & Ohlemuller 2006, Elith et al. 2010). SDMs perform the latter by determining the preferred habitat of a given species, and then using projected future climate data to predict the future location of preferred habitat for that species over time (Araujo et al. 2005, Elith et al. 2010). SDMs otherwise coined as 'habitat suitability models' (Keith et al. 2008), do this by relating occurrence data (presence or presence/absence) to environmental variables to predict the spatial distribution of a species' preferred habitat, and therefore estimating a species' distribution (Elith et al. 2006, Barbet-Massin et al. 2012). This approach is useful particularly for species which lack adequate direct observations (Araujo et al. 2005), or for species with commercial or recreational fishing value, as occurrence data can be readily obtained from fishing surveys (e.g. Brodie et al. 2015, Champion et al. 2018, Champion et al. 2019b, Jenkins et al. 2020), long-term monitoring programs (e.g. Reef Life Survey: RLS: Reef Life Survey - Reef Life Survey) or citizen science initiatives (e.g. Range Extension Data Base and Mapping Project: Redmap; www.redmap.org.au). While SDMs have been commonly used in terrestrial systems, there are relatively fewer examples of the use of SDMs for marine systems (but see Robinson et al. 2011), likely due to the high spatio-temporal variability of both the marine habitat itself (Franklin 2010) and data collection efforts (Brodie et al. 2015). Furthermore, there are difficulties in modelling the 3-dimensionality of marine habitats when many environmental data are available in only two dimensions (e.g. sea temperature data is available in latitude and longitude (at the surface), and not as a gradient in depth for many locations; Duffy & Chown 2017). Nevertheless, the use of SDMs to predict climate-driven shifts in marine systems are increasing in use (see Robinson et al. 2015, Champion et al. 2018, Champion et al. 2019b, Davis et al. 2021) and have accurately predicted the geographic distributions of species across a range of marine taxa (i.e. fish (Guinotte et al. 2006), temperate corals (Tittensor et al. 2009), invertebrates (Bentlage et al. 2009) and macroalgae (Verbruggen et al. 2009)).

Yellowtail Kingfish

Species distribution models (SDM) for the range extension for the pelagic Yellowtail Kingfish (*Seriola lalandi*) were completed in 2020 (Champion et al. 2018, Champion et al. 2019a, Champion et al. 2019b, Champion 2020).These models used environmental satellite-derived data (i.e. sea surface temperature, sea level anomaly, dissolved oxygen, and eddy kinetic energy) matched to occurrence records of Yellowtail Kingfish obtained from fish tagged by recreational anglers during a catch-and-release program run by the New South Wales Department of Primary Industries, and the Range Extension Database and Mapping Project (Redmap: www.redmap.org.au), to predict the preferred habitat of Yellowtail Kingfish through space and time. The resulting 'habitat suitability model' was then used to define core and range-edge habitat use, how this core, and range-edge habitats have shifted historically (Champion et al. 2018), and how they may shift under future climate change (Champion et al. 2019b). Furthermore, assessments of temporal persistence of suitable habitat (Champion et al. 2018, Champion et al. 2019b) and body condition (Champion et al. 2019a) were conducted–providing insight to the potential long-term establishment, and thus range-extension of Yellowtail Kingfish into Tasmania.

King George Whiting and Snapper

As both King George Whiting and Snapper are generally relatively site-attached (versus the pelagic Yellowtail Kingfish), and inhabit shallow reef environments, similar species distribution models (SDM) were developed.

Study extent and occurrence records

Occurrence records (latitude and longitude coordinates) for King George Whiting and Snapper from central and eastern Australia (including South Australia, Victoria, southern Queensland (Snapper only), New South Wales, and Tasmania) were obtained from the Atlas of Living Australia (Atlas of Living Australia 2021a,b), Range Extension Data Base and Mapping Project (Redmap;

www.redmap.org.au), Reef Life Survey (Edgar & Stuart-Smith 2014, Edgar & Stuart-Smith 2021), and the fish frame collection program for this FRDC project (see above). Although Snapper can occur in far north Queensland, and in the Northern Territory (ALA, 2021), we chose a cut off latitude at 25 °S (i.e. Bundaberg), to generate a thermal preference curve relevant to the mid–southern limit of the range of Snapper on the east coast. Collectively, these records ranged from 1897–present but were restricted to 1985–2021 to match the availability of satellite-derived environmental covariates. Modelling species at their range edges requires data from these range limits, and therefore Redmap is an ideal data source because its objective is to identify species outside of their historical distributions. Redmap is an Australian citizen science initiative which encourages members of the public to photograph and report new or uncommon species along the coast. As such, it is a useful source of data particularly for range extending species (Robinson et al. 2015, Champion et al. 2018, Champion et al. 2019b).

To account for spatio-temporal autocorrelation among species occurrence records, only one occurrence was included from the same day and location, and all other occurrences from the same day were only retained if they were greater than 0.10° apart for King George Whiting, and 0.20° apart for Snapper (Brodie et al. 2015, Champion et al. 2018). Once this was done, a total of 429 King George Whiting and 3662 Snapper records were available for model fitting and cross-validation (Figure 2). Pseudo-absence points were randomly generated at a ratio of 10 pseudo-absences:1 occurrence, as recommended for regression type analyses of species distributions (Barbet-Massin et al. 2012), throughout the temporal extent encompassed by species occurrence data. Pseudo-absences were only generated nearshore of the 200m isobath (the continent's shelf boundary) to characterise environmental variation prevalent within, and not beyond, the known distributions of the study species. Combining occurrence and pseudo-absences data produced a binomially distributed response variable for statistical modelling (Barbet-Massin et al. 2012). Large sampling of pseudo-absence points for generating background data have been shown to have high explanatory power and predictive skill when assessing range shifts or animal movement across wide spatial scales (Hazen et al. 2021), and was therefore deemed appropriate as our study extent extended as far north as Queensland (Snapper) and New South Wales (King George Whiting), with a maximum distance of 2,297 km (Figure 2).

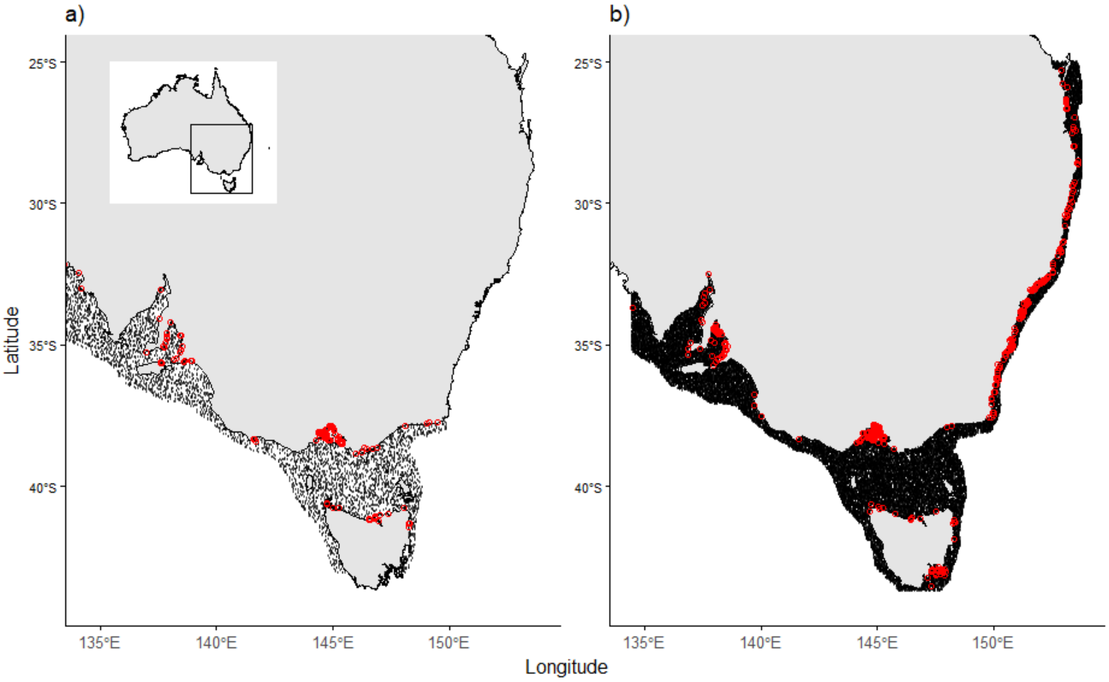

Figure 2. Presence (red) and pseudo absence (black) points used in optimal model for a) King George Whiting and b) Snapper.

Environmental habitat modelling and environmental predictors

To predict the spatial distribution of suitable habitat for King George Whiting and Snapper, individual 'habitat suitability models' for each species were used. Specifically, sea surface temperature (SST), depth (m), and distance (m) to seagrass habitat (King George Whiting only) were used for each habitat suitability model (Table 4), which have been shown to be significant predictors for both King George Whiting and Snapper occurrence (Jenkins et al. 2020). Satellite-derived sea surface temperature (SST) data was sourced from the Copernicus Marine Environment Monitoring Service (Table 4). Gridded bathymetry data measured from optical sensors was obtained from the General Bathymetric Chart of the Oceans (GEBCO Compilation Group 2020). Each presence and pseudo absence point were matched to day- and location-specific values for SST and depth. Seagrass habitat data was sourced from Seamap Australia (Butler et al. 2017), and the distance to seagrass was calculated by measuring the distance of each presence and pseudo absence point to the nearest seagrass polygon using the function *st_distance* in the "sf" package in R (version 1.0.2; Pebesma 2018).

Table 4. Descriptions of explanatory covariates for model selection for habitat suitability models for King George Whiting and Snapper.

Predictor	Description	Source	Spatial Resolution	Units
SST	Daily global sea surface temperature reprocessed (level 4) from Operational SST and Ice Analysis system	Copernicus Marine Monitoring Service (https://marine.copernicus.eu), product #010_011	0.05°	°C
Depth	Gridded bathymetry data measured by optical light sensor downloaded from the General Bathymetric Chart of the Oceans.	General Bathymetric Chart of the Oceans (GEBCO_2021 https://www.gebco.net/)	0.004 °	m
Distance to Seagrass	Distance to seagrass was measured by measuring the distance of each point to the nearest seagrass polygon from the Seamap Australia dataset.	Seamap Australia (https://seamapaustralia.org/) Downloaded from: https://data.gov.au FINALPRODUCT_Seamap Aus)	0.004 °	m
Year	Calendar year (random intercept term in mixed models)		-	-

Collinearity among predictors was assessed by comparing variance inflation factors (VIF) which are used to detect the severity of multicollinearity in the ordinary least squares regression (Thompson et al. 2017). VIFs for factors included in the optimal model were < 1.10 for both King George Whiting and Snapper (Table F1), indicating a low degree of dependence between the focal predictor (i.e. SST, depth, distance to seagrass) versus the other predictors in the model (i.e. SST relative to depth and distance to seagrass, etc; Thompson et al. 2017), and would therefore have little effect on model performance (Zuur et al. 2007). Individual generalised additive mixed models (GAMM) with a logit link function were developed for both King George Whiting and Snapper by relating the binomially distributed response variable (presence vs. pseudo-absence) to environmental predictors (Zuur et al. 2009). Due to the lack of consistent information of sampling effort in the Atlas of Living Australia database, Year was also included as a random effect in the model to account for intra-annual variability in sampling effort (Champion et al. 2018). Multiple models containing all reasonable combinations of model predictors were trialled and model selection was conducted by comparing Aikaike Information Criterion (AIC) values (see Table F2 and Table F3 for full model selection). To avoid model overfitting in the Snapper GAMM (see Figure F1), four knots were applied to the SST smoothing term, and although this model did not return the lowest AIC, we used the model using four knots due to the ecological realism in the thermal response (i.e. a unimodal thermal performance curve).

Spatial and Temporal Autocorrelation

To ensure our methods for accounting for spatial and temporal autocorrelation were sufficient to satisfy assumptions (i.e. removing occurrences from the same day and location, and occurrences within 0.1° and 0.2° for King George Whiting and Snapper respectively), spatial and temporal semivariograms were used to relate semi-variance of points to the space (degrees) and time (days) separating each occurrence record (Figure F2, Figure F3). Cut-off distances were chosen to reflect the spatial and temporal limits where autocorrelation is likely to arise (i.e. at relatively close distances in space (i.e. < 1°) and time (i.e. < 30 days). Semi-variograms were created by converting dates into Julian days to generate a cut off distance of 30 days to assess temporal autocorrelation and coordinates were used with a cut off distance of 111 kms (1.0°) to assess spatial autocorrelation.

Model Validation

To assess the predictive accuracy of the optimal models for each species, k-fold cross validations were used. This was done by randomly splitting the data set into five subsets (k = 5) containing an equal amount of occurrence points and randomly selected pseudo absences at a ratio of 10 pseudos:1 occurrence (Barbet-Massin et al. 2012, Brodie et al. 2015). Each model was then trained on each of four sets of subsetted data, and then tested against the 5th subset. Five folds were used as a conservative measure as there were relatively few occurrence data (Smith et al. 2017).

The area under the receiver operating characteristic curve (AUC) and the true skill statistic (TSS) were calculated to determine both model accuracy and predictive skill and are useful for models which predict spatial variation in species presences and absences (Allouche et al. 2006, Brodie et al. 2015). Rates of true positive predictions (sensitivity) and false positive predictions (1 - specificity) were used to calculate the mean AUC (range 0–1, where a value of 0.5 indicates poor prediction i.e. similar to random, and values > 0.8 indicate good predictive accuracy; Araujo et al. 2005). The AUC is a useful metric to assess the accuracy of species distribution models as it can differentiate between suitable and unsuitable habitat without assuming a cut-off probability (Elith et al. 2006). TSS is calculated as TSS = sensitivity + specificity - 1, and ranges between -1 to 1, where 0 indicates zero predictive skill. The optimal habitat suitability model for King George Whiting had a mean AUC (*SD*) of 0.9873 (0.0002) and a mean TSS (*SD*) of 0.9626 (0.0013). The optimal model for Snapper had a mean AUC (*SD*) of 0.9559 (0.0005) and a mean TSS (*SD*) of 0.8443 (0.0026).

Spatial Predictions of Habitat Suitability and Habitat Suitability Indices

To make spatial predictions of suitable habitat, satellite-derived sea surface temperature (SST) data were aggregated to an austral seasonal (i.e. spring: September–November, summer: December–February, autumn: March–May, winter: June–August) temporal resolution. Monthly spatial predictions were then stacked with depth and distance to seagrass (King George Whiting), as these predictors were assumed to be static, and bilinearly interpolated to a common resolution of 0.004° to account for small changes in depth to make predictions of environmental habitat suitability (Table 4). The predicted responses of each of the optimal models were then converted into a 'habitat suitability index' (following Champion et al. 2018). This was calculated by dividing all relative probability of occurrence predictions by the maximum relative probability predicted over the entire spatial domain and duration of the study. This was done because the relative probability of presence values are dependent on the ratio of occurrence to pseudo-absence data to fit the model (Pearce & Boyce 2006), and is therefore not consistent across the entire study domain. The habitat suitability index therefore ranged between 0 (not suitable) and 1 (very suitable).

Hindcast Analysis

Optimal models for King George Whiting and Snapper were used to create seasonally aggregated spatial predictions of environmental habitat suitability in south-eastern Australia from January 1998–December 2018. Spatial surfaces were resolved to the smallest common resolution to account for small changes in depth, and as a result, all predictions for future suitable habitat for King George Whiting and Snapper were resolved to 0.004°.

Future Projection Models

To assess the potential shift or increase in habitat suitability under future projections, future environmental data were obtained by downscaling sea surface temperature to a common resolution from five CMIP5 climate models (Table F4) forced under the IPCC RCP8.5 prediction scenario. This was done by applying the delta method (e.g. Morley et al. 2018, Navarro-Racines et al. 2020), which firstly involves calculating the difference (i.e. delta value) between seasonally aggregated SST data for the period 2036–2065 (centred on 2050) and a modelled historical baseline period encompassing 1993–2006 for each CMIP5 model forced under the RCP8.5 scenario. Secondly, delta value matrices were bilinearly interpolated from their native model resolution (~1°) to the finer resolution of observed ocean data (i.e. 0.05°), and adding to a satellite-derived seasonal climatology that encompassed the period 1993–2006.

Satellite-derived seasonal means were sourced from the Copernicus Marine Environment Monitoring Service, using sea surface temperature product #010_011. This procedure produced seasonally aggregated sea surface temperature, downscaled to a common 0.05° resolution from six CMIP5 models forced under RCP 8.5. This method was chosen as it has been shown to be robust to correct mean climate projections worldwide (Hawkins et al. 2013, Morley et al. 2018) and it has been useful in for providing downscaled mean climate conditions over shorter (i.e. decadal) time periods (Navarro-Racines et al. 2020).

To predict future oceanographic suitability for King George Whiting and Snapper, a model average of the five CMIP5 climate models was used. This model average was further interpolated to 0.004° to match the resolution of depth and distance to seagrass habitat (King George Whiting only) predictors (as they are assumed static). Future habitat projections created using these data were then compared to predictions created using observed environmental (i.e. SST) data for a 21-year period (averaged) of encompassing 1998–2018, as to compare two ~20-year averaged data sets centred on 2008 (hindcast) and 2050 (forecast).

Analysis for changes to habitat suitability through time

In the interest of understanding the emerging fishery of King George Whiting and Snapper in Tasmania and the potential spatial variation in predicted suitable habitat in this region, we divided Tasmania into six regions. These include, i) North-West (NW), extending north of Temma (41°12’S, 144°38’E), and extending just east of Devonport (41°09’S, 146°28’E), ii) North-East-North (NEN) which includes the Tamar River the Furneaux Islands and contours the North East coast to Eddystone Point (40°59’S, 148°20’E), iii) North-East (NE) which extends south of Eddystone, to Bicheno (41°52’S, 148°18’E), iv) East (E) which extends south of Bicheno to the Southern tip of Maria Island (Latitude = 41°44’S), v) South-East (SE) which extends south of Maria Island and west to Recherche Bay (43°34’S, 146°53’E), and lastly vi) West (W) which includes most of the West Coast; west of Recherche and south of Temma (Figure 3). As seagrass habitat isn’t yet mapped for the Furneaux

Islands, predicted habitat suitability for this region are not directly comparable with other regions, but assessing projected changes through time for this region remain robust.

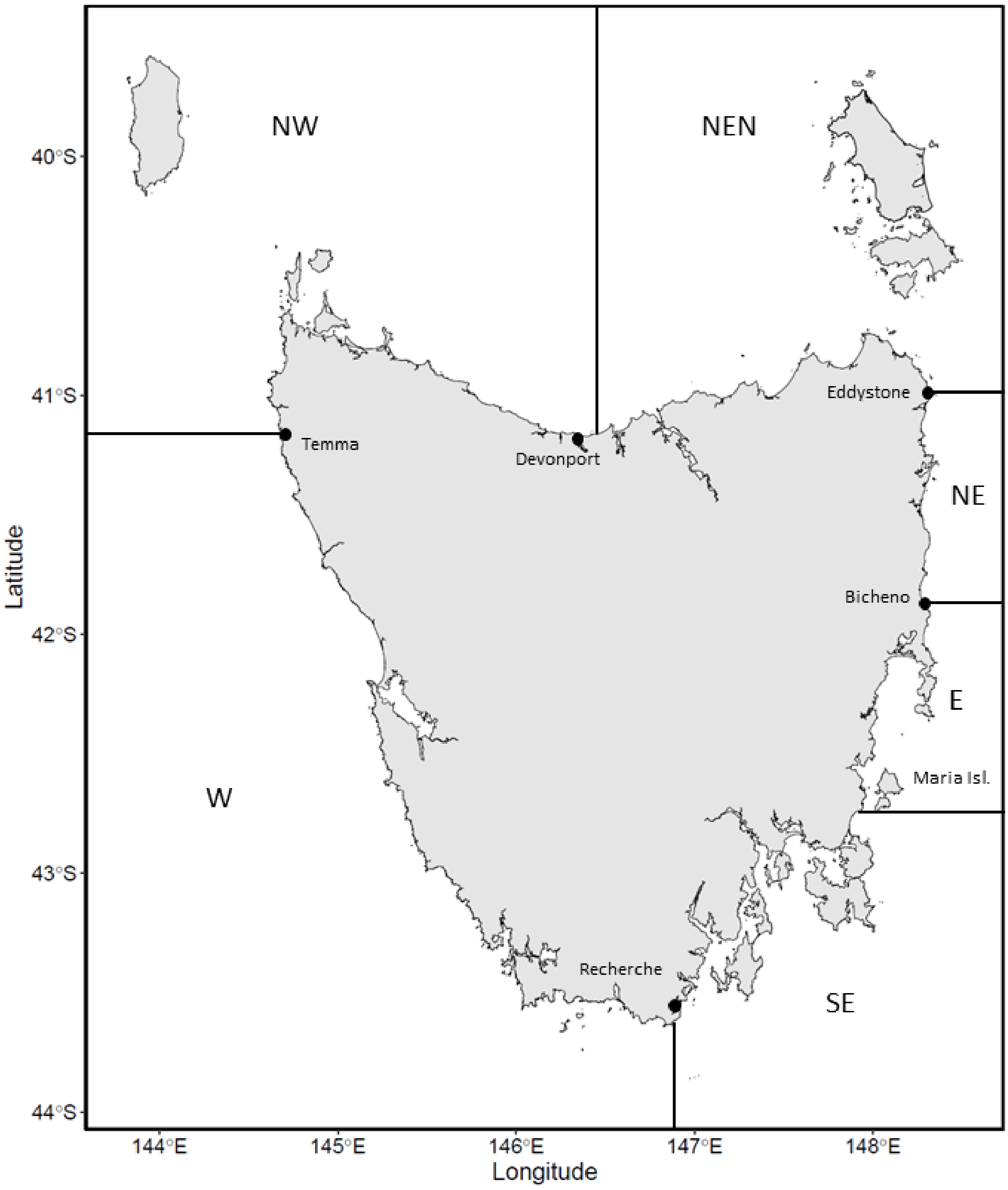

Figure 3. Map of Tasmania split into six regions used for analysis to assess regional differences in habitat suitability. North-West = NW, North-East-North = NEN, North-East = NE, East = E, South-East = SE, West = W.

To analyse differences in species environmental habitat suitability through time, we assessed: historical change in mean habitat suitability within each region, and proportional change of small areas (0.004° or 416 m^2) within each region.

Historical change in environmental habitat suitability within regions

To test whether there has been a change in habitat suitability in the last 20 years (i.e. from 1998–2018), we used a generalised linear model using a binomial error distribution with a logit link, where season, region and year were fixed factors, and habitat suitability from each grid cell was the response variable. AIC comparison was used to determine the best model, and interaction of terms (Table F5).

Analysis for King George Whiting used a multiplicative linear model (i.e.: suitability ~ year + season * region), and analysis for Snapper used a multiplicative linear model (i.e. suitability ~ year * season * region).

High resolution proportional change between historical and future periods

To assess the difference in habitat suitability between the hindcast (1998–2018) and forecast (2036–2065) time periods, we measured the proportional change within each 0.004° (416 m^2) grid cell within each region. This was done to account for the variation in habitat suitability within each region. We calculated the proportional change by subtracting the grid cells of the hindcast raster from the forecast raster and dividing by grid cells of the hindcast raster and multiplying each value by 100.

i.e. $((\text{raster}_{\text{forecast}} - \text{raster}_{\text{hindcast}}) / \text{raster}_{\text{hindcast}}) \times 100$

Linear models were used to assess differences in this high-resolution proportional change between regions, season and year (time), where region, season and year were used as fixed factors and a Gaussian error distribution was used. Data was assessed for normality and homogeneity of variance by assessing residual and Q-Q plots.

Statistical Analyses

All statistical analyses were conducted using the R Environment (version 4.0.3, R Core Team 2021). Spatial thinning of occurrence records was conducted using the ‘spThin’ package (version 0.2.0; Aiello-Lammens et al. 2015), generalised additive mixed models were fitted using the ‘gamm4’ package (version 0.2.6; Wood & Scheipl 2020), k-fold cross-validation was conducted using the ‘dismo’ package (version 1.3.3; Hijmans et al. 2020), generalised linear and linear models were conducted using the ‘lme4’ package (version 1.1.27.1; Bates et al. 2015). Where differences between factors were detected in linear and generalised linear models, pairwise comparisons between factors and were conducted using the ‘emmeans’ package (version 1.6.3; Lenth 2021). Spatial analyses (i.e. model averaging of downscaled climate models forced under RCP 8.5, averaging of environmental data between 1998–2018, and conducting spatial predictions using best habitat suitability model) were conducted using the ‘raster’ package (version 3.4.13; Hijmans 2021). Maps and plots were made using ‘raster’, ‘sf’, and ggplot2 within the ‘tidyverse’ (version 1.3.1; Wickham et al. 2019) packages.

Commercial Fisheries Data

To provide evidence of the potential range shift, or increase in abundance of the three target species, we compiled commercial Tasmanian fisheries data from the Department of Natural Resources and Environment Tasmania (NRE Tas) from July 1995–April 2021. Catch (kg) was further classified by gear type, and latitude of fishing block. Commercial catch was also compared to recent reports of recreational catch from the most recent Tasmanian State-wide Recreational Fishing survey which reports the presence Yellowtail Kingfish, Snapper and King George Whiting (Lyle et al. 2019). Retained catch estimates from this report were converted to weight by calculating the average weight of our sample for each species, and multiplying by the number of individuals reported in the recreational fishing survey Tasmanian State-wide Recreational Fishing survey.

Utilise the Atlantis ecosystem model framework to predict ecological impacts of increasing abundance of key range shifting fish species in Tasmania

Introduction

Marine ecosystem and multi-species models attempt to represent interactions between species—either through feeding or habitat use—and how these connections can be influenced by activities such as fishing or environmental drivers. There are a range of ecosystem modelling platforms available for use with each varying greatly in their structure (number of functional groups, size or age classes represented) and the level of complexity (the representation of space and time, ecological processes included). In addition, each model is designed for a particular question or purpose and as such often only focuses on (or has high confidence in) components of the ecosystem. For example, while models of intermediate complexity (MICE; Plagányi et al. 2011) provide tactical advice related to the effects of single perturbations on a small number of species over a large area, trophic models (such as Ecopath with Ecosim; Christensen & Walters 2004) or full system models (such as Atlantis; Fulton et al. 2011) are more useful strategic tools for characterising entire food webs and exploring the consequences of various biological, environmental or social-economic scenarios.

The utility of ecosystem models to project species distribution and their potential impacts on ecological components is only in its infancy. This category does not include SDMs or dynamic bioclimate envelope models (DBEM; Cheung et al. 2009) as while these models are resolved to species level, they but do not consider biotic interactions, population dynamics (e.g. density dependency), evolutionary change and species dispersal and assume that observed distributions are in equilibrium with their environment. Recent research has shown that very different projections are given between ecosystem models that do include trophic interactions and those that don’t (Fulton et al. 2018b, Pethybridge et al. 2020). However, for many ecosystem modelling platforms that do include trophic interactions, they are often not well resolved taxonomically, spatially or temporally and do not explicitly represent horizontal and vertical movements or migrations.

Atlantis is a whole of the system and process-based (or deterministic) model that includes dynamic, integrated representations of hydrodynamic flows, fished species, habitats and iconic species, major industries and land use, local demography and long term environmental and human drivers (Fulton et al. 2011, Audzijonyte et al. 2019). At the core of Atlantis is a biophysical sub-model, coarsely spatially resolved in three dimensions with vertical layers and horizontal polygons. The physical environment is represented explicitly, via a set of polygons matched to the major geographical and bioregional features of the simulated marine system. The model represents biochemical cycles and tracks nutrient (nitrogen and silica) flows through the main biological groups in the systems. Biological model components are replicated in each depth layer of each of these polygons. The primary ecological processes modelled are consumption, production waste, production, migration, predation, recruitment, habitat-dependency, and mortality. Movement between the polygons and layers are represented by advective transfer or by directed movements or migrations. Atlantis models are forced by temperature, salinity, and physical oceanography which affect the growth rates and habitat distribution of functional groups based on understanding from the literature. For each species or functional group, the model tracks area-specific and dynamic changes in the biomass, distribution and phenology (e.g. growth, condition, movement and recruitment) as well as the size and abundance at age.

In addition to the physical and biological processes major human activities and influences can be represented to varying degrees of detail as either simple pressure-impact-response formulations (e.g. fishing mortality rates) or as more detailed process-level representations of effort allocation, operations and the assessment-management decision making cycle. While fisheries and aquaculture can be represented in the greatest detail there is also the capacity to include shipping (including noise pollution), contaminants, seabed mining, infrastructure (including oil and gas platforms), coastal development, terrestrial run-off and terrestrial land-use.

Atlantis does have high complexity in that it uses a large number of parameters (> 1000), represents a large number of functional groups and ecological processes and is resolved spatially and temporally (Fulton et al. 2011, Pethybridge et al. 2018). This impacts the computing times but allows the model to be used to assess spatially explicit pressures and ecological consequences. The implementation of all these components is subject to a wide range of options, and modular construction of Atlantis allows for the implementation of system components and groups with various levels of details or simplicity. Details of the Atlantis modelling framework, including its main processes, operational aspects and mathematical equations can be found in the technical manual (Audzijonyte et al. 2017a,b). A schematic of the model's content is given in Figure 4.

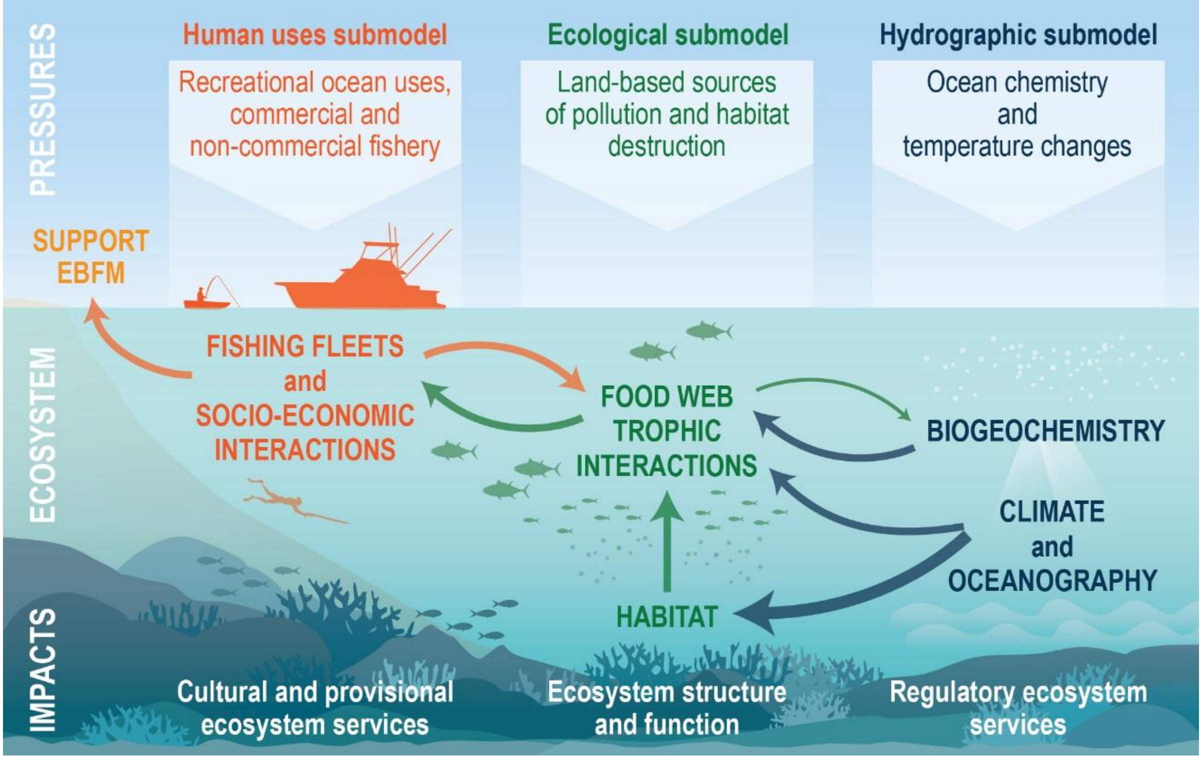

Figure 4. Schematic of the Atlantis modelling framework including the hydrodynamic forcing and the biophysical and exploitation sub-models (Pethybridge et al. 2019).

Modelling Objectives

The objectives for the ecological modelling components of this project were to:

1. Update an existing ecosystem (Atlantis) model for the South-East Australian region, including coastal and open ocean regions, with new biological and ecological data derived from the other components of this project.
2. Run ecosystem projections of the updated Atlantis model using the latest Ocean Forecasting Australian Models (latest climate projections) to examine:

(i) the extent to which climate-ecosystem model outputs can project species redistribution, and
(ii) the ecological implications (changes in trophic linkages and biomass flow) of the known or projected species redistribution of the three fish species of interest in Tasmanian waters.

In the process, we aimed to provide strategic insight to resource managers into the importance and function of specific trophic groups in the ecosystems in Tasmania and which groups would be most impacted by climate change and related changes in species distribution projected over the next 50 years.

Methods

Model Developments

The Atlantis model used for this study was based on the previously published South-East Australian model developed (SE Australia) under the South East Adaptation Program (Fulton et al. 2011, Hobday et al. 2011, Fulton et al. 2012). The model was originally developed by Fulton et al. (2007a) as the basis for a whole-of-ecosystem management strategy evaluation in support of a strategic restructuring of South-Eastern Australian Federal Fisheries. The model was then updated for various projects (Johnson 2012) including being used to look at general fisheries and climate-related questions, such as the implications of fishing small pelagic fishes (Smith et al. 2011). The model originally included the main ecological groups in the region, especially species of interest for fisheries or species assessed as vulnerable to climate change (Pecl et al. 2011, Fulton & Gorton 2014). de la Chesnais et al. (2019) the model to further divide cephalopods and their core predatory fish into more resolved functional groups.

Geography, model extent and design

The SEAust model represents 1.3 million km^2 of marine ecosystems extending from the central South Australian border to the New South Wales–Queensland border and encompasses the Tasmanian shelf and Tasman Sea in the South (Figure 5). The geography of the region is represented by 132 polygonal boxes, of which there are 19 boundary boxes and four islands. The model uses a telescoping spatial structure to maximise geographic details around fine scale coastal features (Johnson et al. 2011). The main features considered were bathymetry, the location of islands, geology, small-sized marine protected, ecological community boundaries and habitat type (e.g., rocky reef, mud flats, and sand flats). Within each box there are up to seven depth layers, depending on the total depth of the box, shallower boxes have fewer layers. The depth layers are 20 m, 50 m, 100 m, 250 m, and 4590 m; in addition, each polygon has a sediment layer. Each box was described with an average depth, but also with a vector giving the proportion of the box surface in a defined number of depth categories.

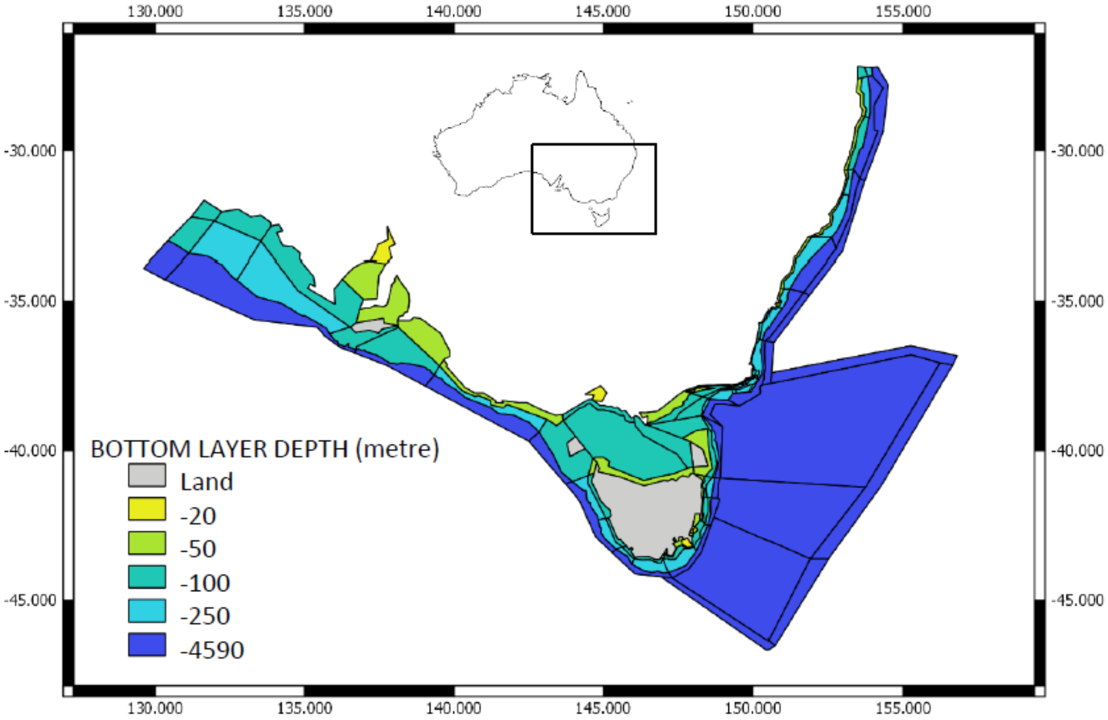

Figure 5. Map area of the model domain covering 1,329,953 km2. Outer blue boxes are boundary condition boxes, while others are dynamic boxes with different maximum depths.

Physical forcing and climate model projections

The hydrodynamic forcing files used by Atlantis to represent the physical environment and oceanic transports were derived from the Ocean Forecasting Australian Model version 3 (OFAM-v3; Matear et al. 2013, Oke et al. 2013). The OFAM-v3 model was coupled to a biogeochemical model representing nutrient flows and plankton components of the ocean food web to produce patterns of primary productivity, nutrient cycling and carbon fluxes that are consistent with observations. For the purpose of this study, we used OFAM-v3 projections from 2010 to 2050 for two scenarios taken from global ocean-atmosphere models used in the CMIP5 (Feng et al. 2017, Zhang et al. 2017), which set the context for the finer scale OFAM-v3 model, which focuses on the Australian region in more detail. This included: (i) the high “business as usual” emissions (RCP8.5) scenario without emissions, and (ii) a control scenario without emissions (control).

The OFAM-v3 simulations provided high-resolution (10 km, 0.1°) outputs that resolved important oceanographic features (e.g. eddies) and ocean state variables including temperature (ºC), base oxygen levels, aragonite saturation and current flows or vertical exchanges. For each spatial box and horizontal layer in the Atlantis model domain, an interpolated time-series of OFAM-v3 outputs were used (representing diurnal, 12 hours, changes). Each spatial cell of the model (i.e. one depth layer of one spatial box) is considered uniform in its environmental variables and hydrodynamic processes. To do this the original climate data was overlayed onto the geometry of the Atlantis models to extract mean monthly climate values for each spatial box and horizontal layer in the model domain. The monthly value per box and layer was then interpolated to create daily forcing data. The species and functional groups within Atlantis then responded to these conditions—both through physiological rates (e.g. growth) that are conditioned on ambient temperatures and via modifying spatial distributions if conditions were beyond their tolerance.

Biology

Throughout the various iterations of this SEAust Atlantis model, there have been between 24 to 67 functional groups represented. For this study, the SEAust model consisted of 54 functional groups in which species were aggregated based on similar size, diets, habitat preferences, migratory patterns, metabolic rates, and life-history strategies (Table E1). The biological components provide a representation of the entire food web, inshore and offshore, pelagic and demersal and from bacteria and phytoplankton up to top predators. This list already included Snapper. Two additional groups were added to explicitly represent King George Whiting and Yellowtail Kingfish. Overall, six plankton groups, two macrophytes, 15 invertebrate groups, 21 fish, and 10 large vertebrate groups were represented.

All the vertebrate groups were modelled using an age and size structured model—with typically 7–10 age classes (with each age class representing 1–8 years depending on the species). This representation tracks the abundance and weight-at-age (based on structural and reserve nitrogen) for each cohort of each group in each spatial cell. Cephalopods, rock lobster, urchins and abalone were represented using a fully age and size-structured model, similar to the one employed for vertebrates, which tracks average size, condition, growth and mortality of each age class in each spatial cell. All other invertebrate groups were treated as aggregated biomass pools.

Parameters for initial abundance estimates of all groups in the model were obtained from published sources or were model-derived (i.e. what biomass could be supported by the system in a stable state) where published data were unavailable. Data for other biological parameters—such as seasonal distribution, reproduction, growth and habitat preference—were obtained from a variety of sources (e.g. reef life surveys and survey data from regional experts), the literature, FISHBASE (www.fishbase.org) and SEALIFEBASE, or re-parameterised from ecosystem models that encompassed the study domain (Bulman et al. 2002, Fulton et al. 2007b, Fulton et al. 2007c, Watson et al. 2013).

This diet matrix in Atlantis represents the potential availability of a prey item to a predator should all else be in the predator’s favour. Predation occurs only if the predator and prey coincide temporally and spatially (given mobility, habitat preferences and habitat state), relative habitat dependency and the state of habitat refugia, the total amount of forage available (summing across prey groups) and whether the prey is of an appropriate size to be caught and consumed by the predator. Size and age structured groups have their diets split into juvenile and adult interaction matrices to represent the strength and rapidity of ontogenetic diet shifts that occur in these groups. The final values used were the result of estimates from these sources modified through model calibration so that the realised diet composition and biomass trajectories matched the available data.

In addition to sediment-based habitat cover (reef, flat sands, or soft muds and silts), Atlantis can represent functional group dependencies for living habitat types. For this model, these living habitat types included seagrass, macroalgae, shallow filter feeder and deep filter feeders. Each habitat dependent group indicates which of these geological and living habits it prefers (or avoids). These preferences act to restrict the spatial domain of groups that are associated only with particular habitat types. This representation also allows for non-trophic interactions (e.g. competition for space and habitat use).

Movement, both horizontal and vertical, is another process that can be explicitly handled by Atlantis. For this SEAust model, migration patterns were represented for three large vertebrate groups: seabirds, baleen whales and Southern Bluefin Tuna. Within domain horizontal movement also occurred seasonally or based on the distribution of prey fields. The model also includes vertical movement (i.e. diel vertical migration). By incorporating vertical stratification of physical properties and vertical migrations of certain biological components the interactions of hydrodynamic and biological processes that vary with depth were represented with sufficient detail to reproduce the gross properties of vertical profiles.

For King George Whiting, Snapper and Yellowtail Kingfish parameterisation of growth and diet composition come directly from research undertaken within this project while other parameters for distribution, abundance and recruitment were taken from various sources of literature (Table G1, Table G2, Table G3), FishBase and expert elicitation.

Fisheries

While Atlantis has the capacity to incorporate dynamic fishing fleets, this study only used a static fishing mortality-based representation of fishing pressure to focus on the strategic questions related to species distribution. This simple representation is less responsive to realised catch rates and other behavioural drivers such as profitability, market price, and social connections, and it thus removes some of the noise associated with variations in fishers' behaviour that a dynamic fishing model can impose. For fished groups in the model domain, fishing mortality was combined with estimates of linear mortality with values based on estimates of the 2010 rates of fishing by both Federal and State fleets. The final values used were modified to a small degree in the calibration process, in order to allow for a stable biomass (i.e. no evidence of numerical instability) under constant conditions and for biomass trajectories that matched observed trajectories in the system over the past 10 years given observed catches.

Calibration strategy

The updated SEAust Atlantis model was constructed for the initial year of 2010, to coincide with initial year of the hydrodynamic models and when a lot of effort was utilised to parameterise the model based on empirical data stemming from the SEAP project (Hobday et al. 2011). We calibrated the model to obtain a stable system state with long run biomasses within approximately 20% of the initial biomass values. Where available, time-series trajectories of both biomass and abundance of groups, taken from the stock assessments, technical reports and published literature were also used to calibrate the model. Initial abundances (numbers), weights in nitrogen, and vertical and horizontal migration were also checked through the calibration process to ensure that the model was as good as possible.

Scenario descriptions and model assessment

Status Quo

The OFAM-v3 projections for the "business as unusual" greenhouse emissions (RCP8.5) scenario were used to run the ecological projections of the base or *Status Quo* model in which scenarios were then compared to.

For the baseline scenario population densities for the three species of interest were even across all boxes with median depth < 100 m except in the Bass Strait and around Tasmania where these boxes were only given 10% density compared to mainland boxes (Figure 6).

For all the Atlantis simulations, all scenarios ran for the first 10 years (2010–2020) without change from the *Status Quo* conditions to allow for consistent model 'burn-in', as is consistent with Atlantis best practice (Pethybridge et al. 2019), so that transient effects of the initial conditions in the system do not unduly influence the projections. The scenario-imposed perturbations were then applied for a further 40 years (called the projection period between 2020–2060), a total run time of 60 years. The burn-in period is discarded and not included in the analysis of model results. For all simulations we focused on examining medium-term (2050) ecological impacts. To ensure that we weren't reflecting inter-annual variability (i.e. the results were not skewed by the coincidence of a "poor" year with a reporting window), we took 2-year averages. The 30-year time span was used as this is of most interest to regional planners and others with a strategic view of a system.

In each scenario cases, the biomasses were compared to those of the *Status Quo* simulation to assess the potential influence of the drivers in the scenario on the SEAust ecosystem.

Ocean warming

To explore the effects of projected climate change on the Tasmanian marine ecosystem, we ran the model under the OFAM-v3 control (no greenhouse emissions) scenario out to 2060 and compared it to the *Status Quo* run under the RCP8.5 climate projections. The impact of climate change on relative changes in biomass at any given time was calculated by:

$$\text{Climate impacts} = \left(\frac{\text{RCP8.5} - \text{Control}}{\text{Control}} \cdot 100\right)_t - \left(\frac{\text{RCP8.5} - \text{Control}}{\text{Control}} \cdot 100\right)_{2010}$$

Species redistribution

Three scenarios were developed to explore species redistribution for each of the species of interest to reflect the trends of increasing population toward the south. For each of the three species of interest, three different distributions were represented by the model (Figure 6), including:

A. NorthTas: Density is even across boxes with median depth < 100 m, broader northern Tasmania was populated at 50% of mainland density, while the species are not present in the rest of Tasmania.
B. North and SouthTas: Density is even across all boxes with a median depth < 100 m off the north of Tasmania, but only half the density off the east and south Tasmania.
C. ALLTas: Present in every box off the north, east and southern parts off Tasmania with a median depth < 100m.

For scenarios B and C, King George Whiting distribution didn’t extend around the west to south-west of Tasmania due to restricted distribution of preferable habitat (seagrass and sand or mud flats). For Yellowtail Kingfish, scenarios B and C were represented by changes in seasonal migration patterns (Figure 6) as this was more realistic than a greater poleward shift in distribution.

For the species distribution scenarios, change in relative biomass is the main diagnostic used with model output. The change for variable x (Δ_x) is calculated as:

$$\Delta_x = 100 \cdot \left(\frac{\bar{x}_T}{\bar{x}_{SQ}} - 1\right)$$

where $\bar{x}_T$ is the average value over the final 5 years of the scenario (treatment) projection period and $\bar{x}_{SQ}$ is the average value over the final 5 years of the *Status Quo* simulation.

Spatial changes in biomass over the model domain were visually assessed. We also assessed relative biomass changes in Tasmanian boxes (Figure 6).

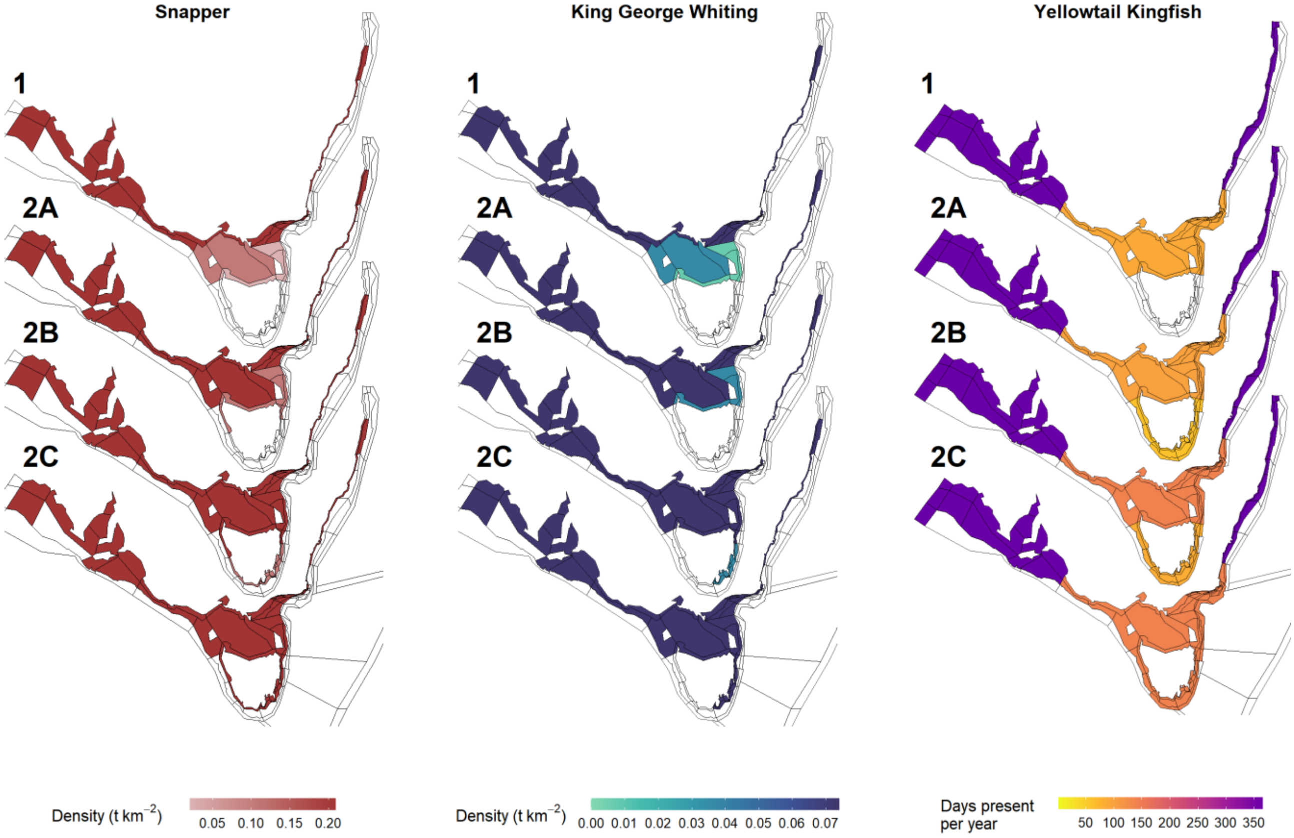

Figure 6. Distribution maps of density or days present for the *Status Quo* (1) and the species distribution scenarios (2A, 2B, 2C) for Snapper, King George Whiting, and Yellowtail Kingfish.

Results

Status Quo outputs to 50-year climate change (RCP8.5) projections

The updated SEAust Atlantis model produced reasonable ecosystem structure and function, with biomasses typically of the correct order of magnitude and plausible dynamics. During the entire projection period under the *Status Quo* simulation invertebrates dominate the biomass with most showing relative stability (relative changes within 20% of initial conditions) throughout the projection period. The trophic groups that accounted for the highest proportion of total system biomass were large, micro and meso-zooplankton (18–25%), gelatinous zooplankton (11–15%), benthic carnivores (5–8%), and deposit feeders (1–4%). Throughout the entire simulation period there was a strong seasonal cycle displayed for the planktonic groups including pico-phytoplankton, diatoms and micro-zooplankton. For the three species of interest, biomass projections for all age-groups over the whole model domain were reasonably stable after the initial 20 years (burn-in period), with changes less than 10% in either direction.

The Atlantis model outputs of diet composition for the three species of interest showed variability throughout the projection period (Figure 7). The diet of King George Whiting consisted primarily of benthic carnivores and grazers with some additional contributions of seagrass, filter feeders and crabs. The diet of Yellowtail Kingfish consisted primarily of benthic filter feeders, crabs and shallow demersal fish. The diet of Snapper initially consisted of deposit feeders before switching to a more diverse intake of benthic carnivores, crabs, and filter feeders. The main predators of Snapper were sharks, shallow demersal and piscivorous fish, and marine mammals. King George Whiting was mainly preyed on by shallow piscivorous fish, shallow demersal fish, seabirds, skates and rays, and marine mammals. Yellowtail Kingfish were predominantly preyed on by large piscivorous fish, sharks, tunas, and marine mammals.

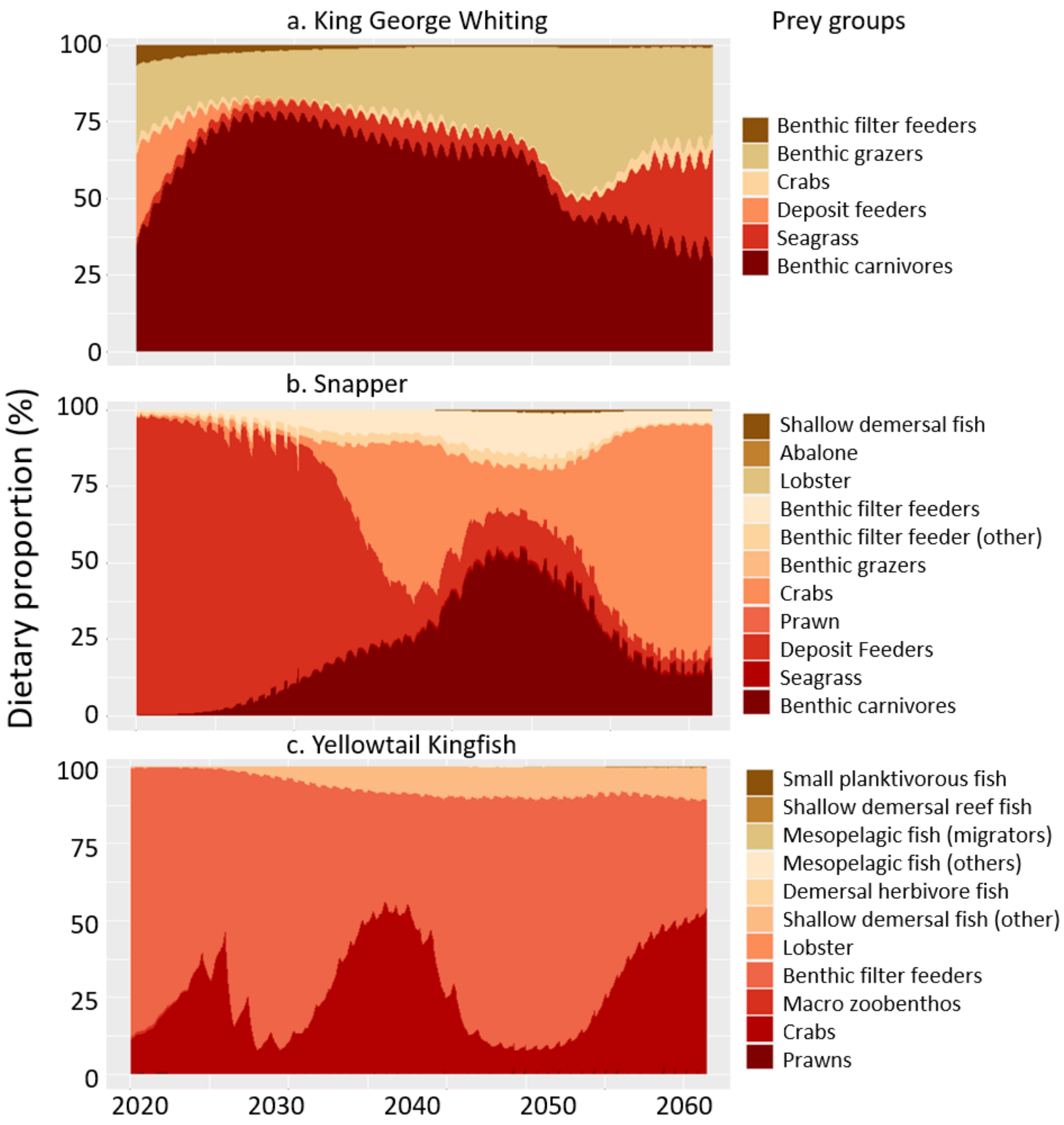

Figure 7. Model outputs for diet composition of (a) King George Whiting, (b) Snapper, and (c) Yellowtail Kingfish, over the model period for the *Status Quo* simulation.

Scenario outputs: Ocean warming

There was a high degree of temporal variability in sea surface temperature and primary production projected by the climate model under both the control and RCP8.5 scenarios. Across the entire model domain, sea surface temperature (in top 20 m) was projected to increase by about 1.4 °C with increases ranging from 0.9 °C in the north to 1.75 °C in the more southern model polygons.

The three study species also showed interesting temporal responses to climate change (Figure 8). A clear temporal response was observed for King George Whiting with biomass projected to increase by up to 60% in Tasmania and within the whole model domain. For Snapper and Yellowtail Kingfish, more dynamic responses were observed with both displaying decadal cycles with biomass peaking at 2060. For Snapper over the whole model domain, biomass reduced by 5% in 2040 before increasing by 5% in 2060 with a slightly higher increase in biomass projected in Tasmania alone (15% by 2060). For Yellowtail Kingfish biomass over the whole model domain declined by as much as 30% in both 2025 and 2040, with slight increases in 2030 and 2060. In Tasmania alone, a more reduced change in biomass of kingfish was observed with biomass projected to increase by 5% in 2030 and then 13% in 2060.

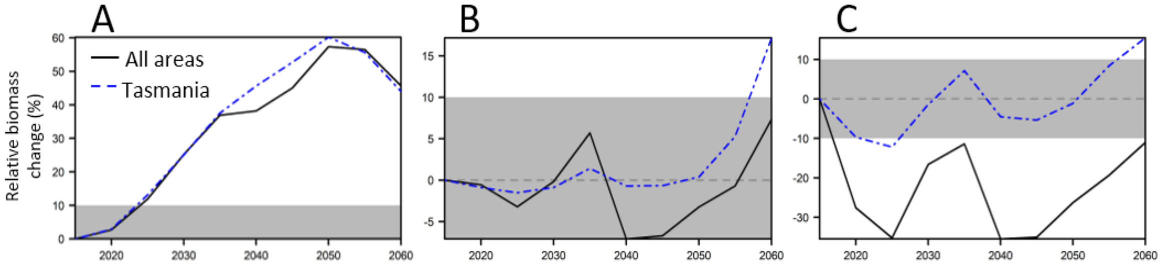

Figure 8. Relative changes (%) in the total biomass of (A) King George Whiting, (B) Snapper, and (C) Yellowtail Kingfish under the RCP8.5 ocean warming scenarios in Atlantis calculated relative to the *Status Quo* simulation.

Climate change (RCP8.5) was projected to impact some functional groups positively and negatively, while most maintained a constant biomass over both the entire model domain and in Tasmania (Figure 9). Among the vertebrates most positively impacted (showing relative biomass increases of more than 20%) over the whole model domain and in Tasmania, were Warehou and Trevalla, Blue Grenadier, King George Whiting, and wrasses. Other positively impacted invertebrate and macrophyte groups included seagrass, prawns, benthic carnivores, crabs, and abalone. More negatively impacted groups included urchins, shallow demersal herbivores, flatheads, and shallow piscivorous fish. While we acknowledge the increase in range and abundance of urchin *Centrostephanus rodgersii* in Tasmania, the decline in urchins predicted is likely due to a combination of Atlantis modelled pH effects including pH mediated predator-prey interactions and the projected decline in macroalgae. There were a number of groups that had opposite trends between the short-term (2030) and long-term (2060) projections, including flatheads, pinnipeds, shallow demersal reef fish, Striped Tuna, shallow piscivorous fish, Blue Grenadier and abalone. There were also three groups that showed very different regional changes with Tasmania seeing unique declines in prawns and microzooplankton and increases in mesopelagic fish.

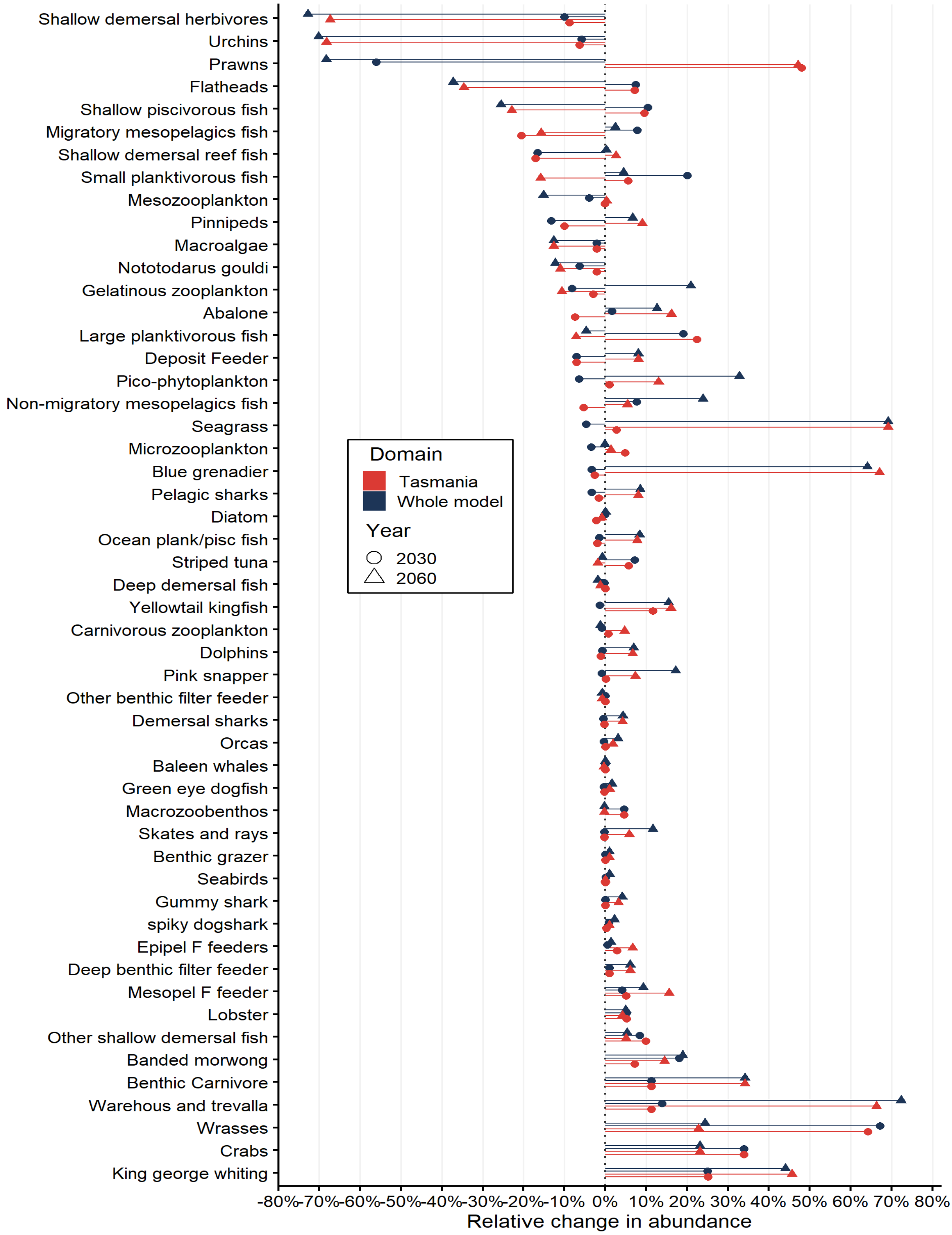

Figure 9. Relative changes (%) in the total biomass of all functional groups under the ocean warming scenarios in Atlantis calculated relative to the *Status Quo* simulation.

Scenario outputs: Species redistribution

The responses of different functional groups to the species redistribution scenario were typically much more reduced than that to climate change. Relative changes in biomass rarely exceeded 10% with slightly larger negative responses than positive ones. For each of the three species, most of the groups that showed negative responses were often dominant prey items while groups that showed positive responses were predators, though there were some surprises also. For each species the scenarios were run for, there were also groups that showed mixed responses (with responses going in both directions) between years, including Warehou and Trevalla, cephalopods, small planktivorous fish, and pico-plankton.

For King George Whiting, two groups (deposit feeders and shallow benthic filter feeders) showed clear and consistent declines between scenarios and years (Figure 10). There were eight groups that showed small, yet consistent, positive change in biomass included shallow demersal reef and piscivorous fish, skates, sharks, prawns, gelatinous zooplankton, seabirds and urchins.

For Yellowtail Kingfish, there was a longer list of groups that showed consistent negative declines in biomass, although often at reduced magnitudes relative to King George Whiting and Snapper, including shallow demersal herbivores, prawns, shallow and small piscivorous fish, macrozoobenthos, and crabs (Figure 11). There were only four groups that showed slight increases in biomass, including pinnipeds, large epipelagic and mesopelagic fish. The biomass of large planktivorous fish declined under scenario A (north Tasmania only) but increase under scenarios B and C (north-east and south of Tasmania).

For Snapper, the only functional groups to show declines were shallow benthic filter feeders and deposit feeders (Figure 12). There were seven groups that showed a slight positive change including macrozoobenthos, shallow piscivorous fish, pinnipeds, and migratory mesopelagic fish.

When all species were included together under the B and C scenarios, there were four groups that showed clear declines (across both scenarios and years; Figure 13). There were also four groups that showed small yet consistent increases in biomass, including macrozoobenthos, pinnipeds, large planktivorous fish and urchins. Another eight groups showed mixed responses between scenarios (pico-phytoplankton, gelatinous zooplankton and prawns) or years (crabs, cephalopods, Striped Tuna, and Blue Grenadier) or due to a combination of scenario and year effects. Most groups that showed > 2% changes in biomass were identified in the single species scenarios, though there were some obvious exceptions including macroalgae and deep benthic filter feeders. There were also a number of groups that showed interesting multi-species effects, including shallow piscivorous fish that showed a mixed year response despite showing clear negative responses for Yellowtail Kingfish and positive response for King George Whiting and Snapper.

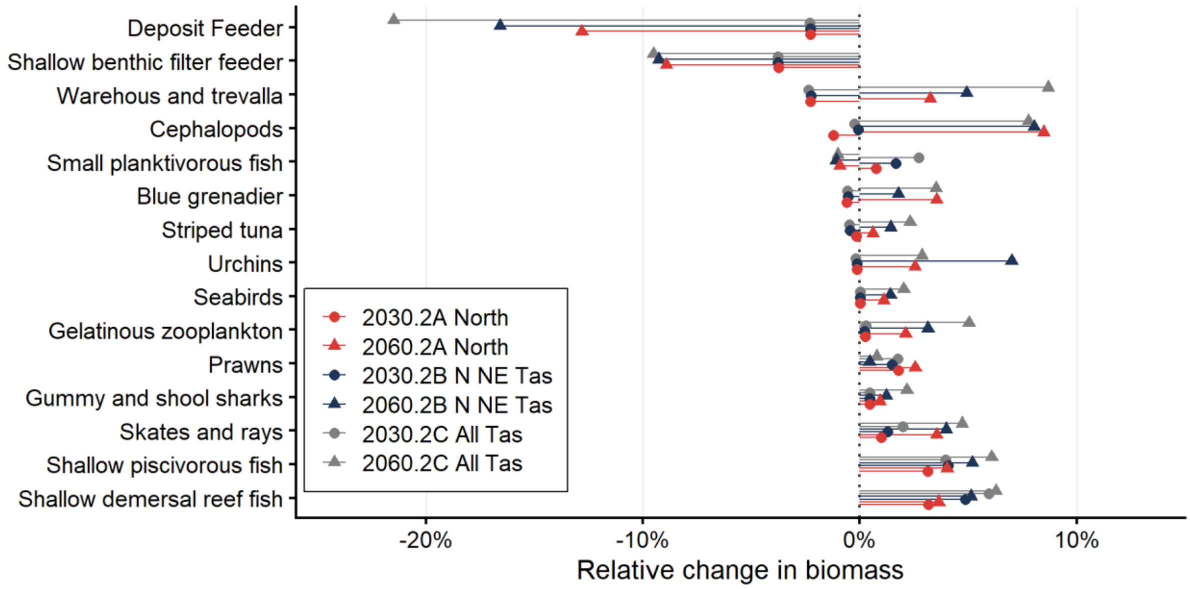

Figure 10. Relative changes (%) in the total biomass of all impacted functional groups (with changes > 2%) under the species distribution scenarios for King George Whiting at 2030 and 2050—calculated relative to the *Status Quo* simulation in Tasmanian polygons boxes only.

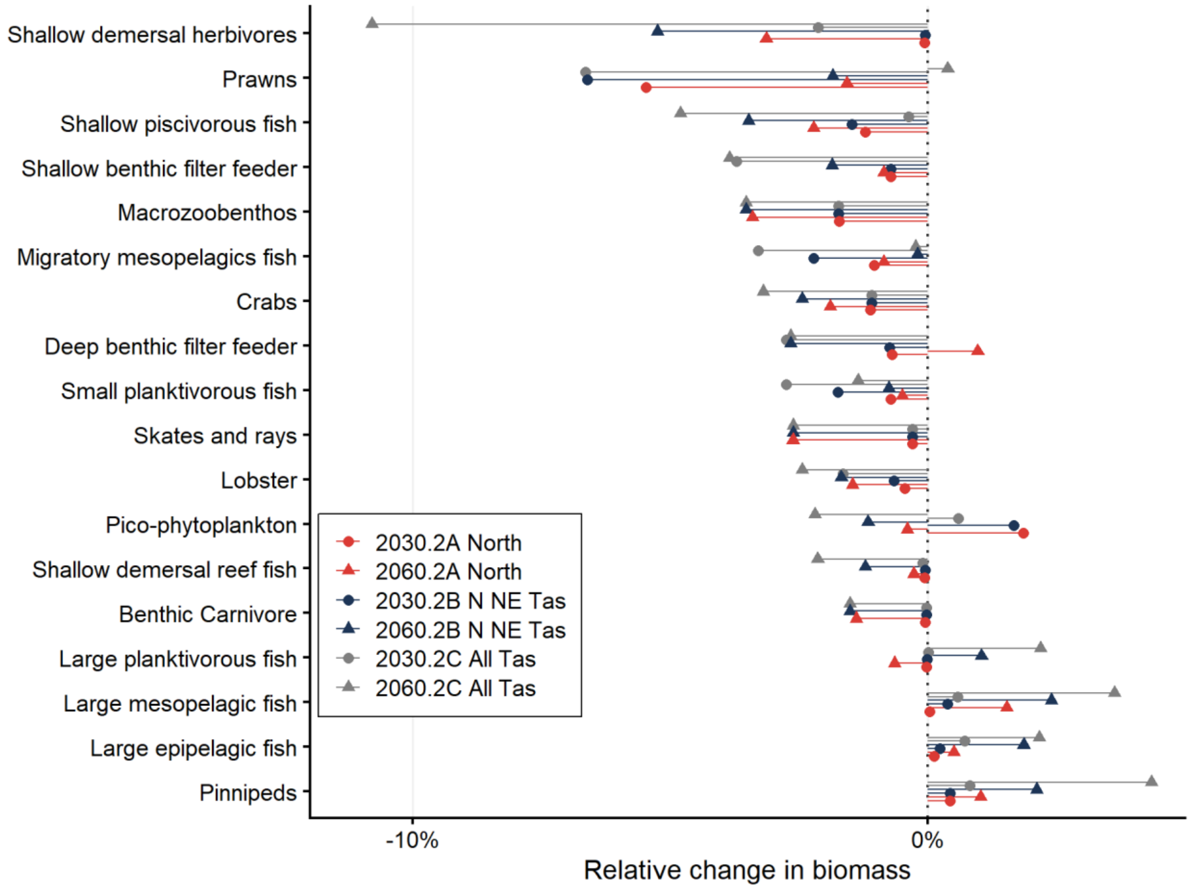

Figure 11. Relative changes (%) in the total biomass of all impacted functional groups (with changes > 2%) under species distribution scenarios for Yellowtail Kingfish at 2030 and 2050—calculated relative to the *Status Quo* simulation in Tasmanian polygons boxes only.

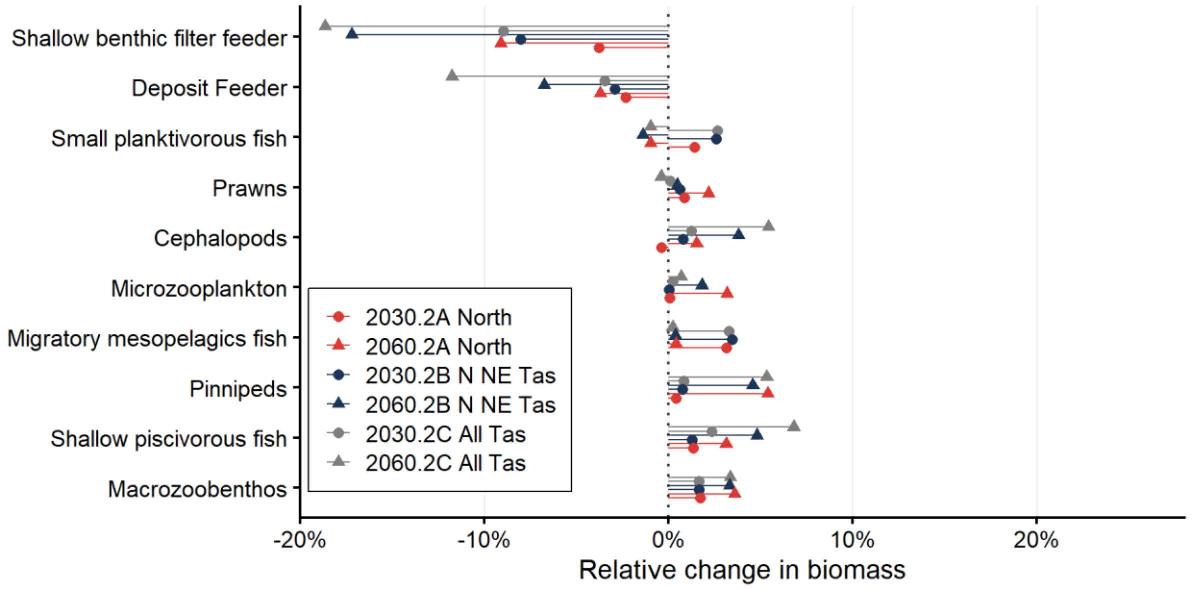

Figure 12. Relative changes (%) in the total biomass of all impacted functional groups (with changes > 2%) under species distribution scenarios for Snapper at 2030 and 2050—calculated relative to the *Status Quo* simulation in Tasmanian polygons boxes only.

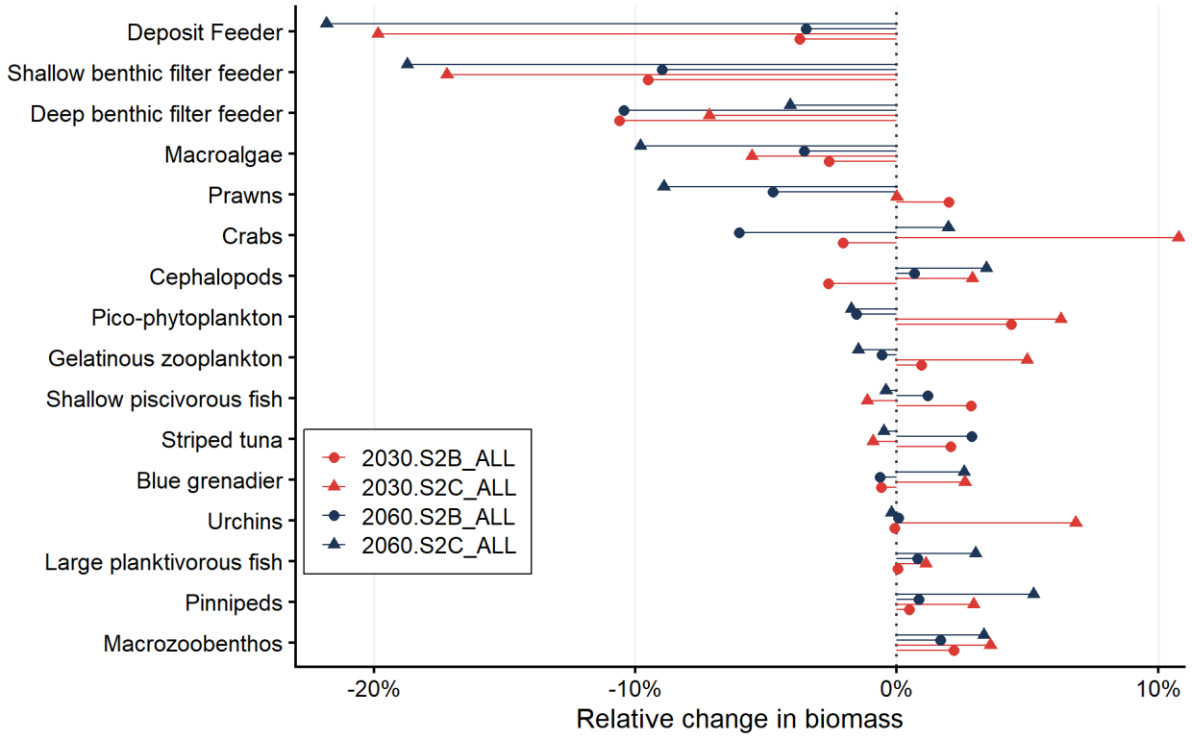

Figure 13. Relative changes (%) in the total biomass of all impacted functional groups (with changes > 2%) under species distribution scenario C for all 3 species of interest at 2030 and 2060—calculated relative to the *Status Quo* simulation in Tasmanian polygons boxes only.

Discussion

The Atlantis ecosystem modelling scenario results suggest that the SEAust model is very responsive to climate change as shown by the Atlantis Great Australian Bight (Fulton et al. 2018a) and the larger Atlantis South East Australia models (Fulton et al. 2011). Ecopath models of the area, including for the Tasmanian Eastern Shelf and slopes (Bulman et al. 2002), the whole of Tasmania (Watson et al. 2013) and the Great Australian Bight (Goldsworthy et al. 2013) have also been shown to be very responsive to both climate and anthropogenic drivers, and therefore represent a dynamic and responsive ecosystem. In these systems, strong annual cycles of phytoplankton dictate to a large degree the productivity of the entire system with clearly observed differences between coastal and offshore areas. In particular, the gulfs, inlets and bays in the model had some of the largest biomasses of macroalgae and seagrass which had flow on effects to many habitat-dependent functional groups including Snapper and King George Whiting. In other areas, and particularly over deeper shelf, slope and open ocean waters, the main driver of biomass changes were plankton with strong seasonal dynamics shown for all seven groups represented. These findings highlight the importance of getting the climate projections right for sensitive and dynamic groups such as plankton and macrophytes.

Over the last decade there has been an increase in the use of regional and global ecosystem models to project the impacts of climate change on various marine ecosystems. This includes projections from global ecosystem models (Tittensor et al. 2018) including for the most recent IPCC report (Cheung et al. 2016) and from 13 marine ecosystem models for regions around Australia (Fulton et al. 2018a, Pethybridge et al. 2020). These studies have shown that the direction of response to climate drivers seems to be system and species dependent in that there will be winners and losers in every system (Fulton et al. 2007a). The study by Fulton et al. (2018b) highlighted that 60–70% of target species in Australia are sensitive to climate change in some way and that, under the latest available physical environmental information, many species are projected to decline in abundance. This is particularly the case for models that don’t include trophic interactions, such as species distribution models, that are unable to represent how predators or competitors are more strongly affected by projected environmental shifts and resulting food web feedbacks (Pethybridge et al. 2020).

Consistent with Fulton et al. (2018b) the SEAust Atlantis model in this study projected that both shallow-water species from benthic and coastal systems are likely to be more strongly affected by climate change than pelagic and open-ocean systems. These findings are in line with previous vulnerability analyses (Pecl et al. 2014, Fulton et al. 2020), and past climate-ecosystem model assessments (Brown et al. 2010, Fulton 2011) of Australian ecosystems. Among the groups projected by the SEAust Atlantis model to increase with ocean warming, was King George Whiting, and Snapper and Yellowtail Kingfish to a lesser degree, along with crabs and other pelagic or demersal fish groups (morwong, trevalla & wrasses). These increases are related to a range of complex factors represented in the model such as the widening of temperature ranges for spawning and increased habitat availability with the relative biomass of seagrass also increasing in the southern polygons.

While the influence of climate change on the system was clearly observed, the degree to which the SEAust Atlantis model could project redistribution (based on climate projections alone), or the ecological impacts of species redistribution (scenario 2), was questionable. Under ocean warming projections, Yellowtail Kingfish was the only study species that showed an increase in abundance in the Tasmanian region compared to other areas across the model domain. While the biomass of King George Whiting and Snapper increased in the north of Tasmania, there were no responses for these species in the east or south of Tasmania. This result is not surprising given the way that distribution parameters are set up under the Atlantis framework (Audzijonyte et al. 2017a, b). Indeed, for each group the numbers and densities (proportion of biomass) are initially set by polygon and depth layer. Animals can then only move between available polygons given a set of other dependent factors such as food availability, habitat dependencies and environmental ranges.

The Atlantis results for the species distribution scenarios showed a mix of expected responses, where there were direct changes in abundance to predators or prey, and some more unusual outcomes. Groups that showed consistent response trends (of > ±5% change in abundance) between the scenarios (A, B, and C) and or years mostly occurred for dominant prey or predators' groups, which was to be expected from a trophic model and are likely to best reflect change due to species distribution. Other groups to respond, such as prawns, cephalopods, and small planktivorous fish, did not show consistent trends over time or with changes in distribution and were either found to be responsive to climate or showed large seasonal dynamics in the initial model run. These groups seem to be governed by primary producers such as pico-plankton, macroalgae and seagrass, that were among the most responsive, suggesting complex interaction between top-down and bottom-up controls.

The more unusual ecosystem model projections reported here highlight some clear caveats of this work such as remaining structural and parameter uncertainty and pitfalls in recycling models (Essington & Plagányi 2014). Indeed, the SEAust model has traditionally been used to project the impacts of fishing (Fulton et al. 2007a) or better representing or understanding the role of particular functional groups (Fulton 2013a, Fulton 2013b, de la Chesnais et al. 2019). As such, much less attention, and thus assessments of quality of fits between data and model outputs, has been undertaken for some functional groups (such as urchins which showed an unexpected decrease in abundance to climate change). Clearly in respect to modelling species redistribution with an ecosystem model like Atlantis, further improvements are needed in respect to getting the magnitude of change right and understanding the form of that responsiveness to which ecosystem changes are moderated by trophic interactions or habitat dependencies. In the future, such models should seek to incorporate or better represent our growing understanding of other processes, such as a species ability to adapt to changing condition (it's evolution), larvae production and distribution, and the complex interactions between environmental conditions and animal behaviours (including immigration) and physiology.

Quantifying biomass changes in different species, and the complex ways in which resource-dependent communities will need to respond to external and internal drivers, is inherently difficult. Multi-model or ensemble ecosystem modelling exercises, such as that performed by Fulton et al. (2018b) have improved our understanding of uncertainty about the structure, function and responses of marine ecosystems. These works have shown that highly resolved spatial dynamics are particularly hard to compare between models as many ecosystem models lack spatial resolution or don't represent key movement or ecophysiology parameters. It is largely for this reason that we have seen limited use of large ecosystem models used to project the impacts of species redistribution on ecosystem dynamics. In contrast, species distribution models that represent physiological responses, do provide the highly resolved spatial information about when, where, and how species will move. However, these SDM's do not include trophic interactions, which have been shown to severely impact their ability to predict response to climate change (Fulton et al. 2018b). Accordingly, the next step to better understand the larger scale impacts of species redistribution, is to combine (couple) these, or at least aspects, of these two modelling approaches.

Yellowtail Kingfish

Introduction

Yellowtail Kingfish (*Seriola lalandi*) are a mobile benthopelagic species that inhabit tropical and temperate waters globally (Nugroho et al. 2001, Sepulveda & Gonzalez 2017). Maximum size for Yellowtail Kingfish range between 1900–2500 mm TL, and ~70 kg (Smith 1987, Kailola et al. 1993, Gomon et al. 2008) and their lifespan can exceed 20 years (Stewart et al. 2001, Holdsworth et al. 2016). Yellowtail Kingfish are generalist mesopredators that consume a variety of small pelagic and demersal fish, including Redbait (*Emmelichthys nitidus*), Australian Herring (*Arripis georgianus*), King George Whiting (*Sillanginoides punctatus*), and small crustaceans (McGlennon 1997, Fowler et al. 2003).

Distribution

Within Australia, Yellowtail Kingfish occur as far north as North Island in Queensland (23°S) and contour the southern coast of mainland Australia, including the north and east coast of Tasmania, to Trigg Island, Western Australia (32°S), and are common around Lord Howe and Norfolk Islands (Love & Langenkamp 2003, Bray 2018). Yellowtail Kingfish have been recorded in Tasmania well before ~1947 (Roughley 1957). At the time, this species was called 'Yellowtail' in Victoria, South Australia, and Tasmania, but 'Yellowtail Kingfish' in NSW, but to avoid confusion with the related smaller 'yellowtail' (*Trachurus declivis*), the name was changed to 'Yellowtail Kingfish' across Australia (*Regificola grandis*; Roughley 1957). Further, large Yellowtail Kingfish (~16 kgs) have been recorded in Tasmania, as far back as 1967 in the historical game fishing records (Gamefish Association Australia: https://gfaa.asn.au/records-info/). Anecdotal evidence from anglers suggests that numbers of Yellowtail Kingfish are increasing around Port Phillip Bay in Victoria (Bray 2018, Green et al. 2020), and participation in citizen science initiatives have identified Yellowtail Kingfish 200 km south of the previous southernmost record from a suite of open access data bases including the Atlas of Living Australia (http://www.ala.org.au/), Australian Faunal Directory (https://biodiversity.org.au/afd/), FishBase (http://www.fishbase.org/), Fishes of Australia (http://www.fishesofaustralia.net.au/), Global Biodiversity Information Facility (http://www.gbif.org/), Ocean Biogeographic Information System http://www.iobis.org/), and Reef Life Survey (http://reeflifesurvey.imas.utas.edu.au/portal/home (Stuart-Smith et al. 2018), suggesting a potential range extension into the south-east coast of Tasmania (Champion et al. 2018, Champion et al. 2019b). Species distribution modelling of Yellowtail Kingfish in eastern Australia has identified rapid poleward shifts (94.4 km per decade for core habitat use) and an increase in the temporal persistence (by approx. 3 months) of suitable habitat over the past 22 years (Brodie et al. 2015, Champion et al. 2018), with these predicted shifts largely driven by the increase in sea surface temperature, and the southern extension of the Eastern Australian Current (Ridgeway 2007).

Connectivity/Stock Structure

There is still some debate whether *Seriola* spp. are a singular species with recent divergence into three distinct populations (Premachandra et al. 2017), or whether they are separated into three distinct species (*S. dorsalis* in the NE Pacific (USA and Mexico), *S. aureovittata* in the NW Pacific (Japan), and *S. lalandi* in only the Southern Hemisphere: Martinez-Takeshita et al. 2015)). However, there is agreement that *S. lalandi* in Australia (excluding Western Australia) and New Zealand are genetically distinct (Miller et al. 2011). Network analysis of mtDNA has recently confirmed that there is no differentiation between populations in Victoria, New South Wales, South Australia or Tasmania, but evidence of differentiation of Yellowtail Kingfish populations in Western Australia (Green et al. 2020). However, comparison of microsatellites suggests significant, but small genetic variation between the Tasmanian and Victorian populations (Green et al. 2020). Therefore, in Australia, Yellowtail Kingfish are separated into the “Eastern Australia” biological stock which includes populations from Queensland, NSW, Victoria, South Australia, and Tasmania and the “Western Australia” biological stock which includes populations in Western Australia.

Fisheries and Management

Due to their high meat quality and large size, Yellowtail Kingfish are a highly prized species in both commercial and recreational fishing sectors, where annual recreational catches commonly exceed commercial catches in Queensland, New South Wales, South Australia, Western Australia, and estimates from recreational fishing surveys suggest this trend is emerging in Victoria and Tasmania (Table 6). Yellowtail Kingfish are also a primary target in game fisheries in SE Australia, where they have been one of the most frequently caught species in both the SE Australian Land Based Game Fishery (mean catch rate: 0.039 ± 0.045 fish per hour; Griffiths 2012) and in the NSW boat-based fishery (mean catch rate: 0.05 ± 0.18 fish per hour; Lowry et al. 2006).

Stock assessment modelling of the Eastern Australian stock and catch curve analyses indicates that current harvest across all sectors is under the maximum sustainable yield, and that fishing mortality rates have been similar to natural mortality for the past 10 years (Hughes & Stewart 2020). However, stock assessments for the entire Eastern Biological stock have been limited to information gathered from NSW only, as information on Yellowtail Kingfish from other states is lacking (Hughes & Stewart 2020). Using a weight-of-evidence approach based on temporal stability of size frequency distributions, estimated spawning stock biomass, and estimated mortality (natural and fishing), both Western and Eastern Australian stocks have been deemed ‘sustainable’ (Hughes et al. 2021). However, information on historical size distributions, population connectivity, movement, and spawning are required to improve the accuracy of these assessments (Hughes et al. 2021).

Movements

Yellowtail Kingfish are generally reef-associated, but can also be found offshore and at depths exceeding 300 m (Stewart and Hughes 2008). They can be found schooling in single- or mixed-species groups, often with Silver Trevally (*Pseudocaranx dentex*) and Southern Bluefin Tuna (*Thunnus maccoyi*). Yellowtail Kingfish have been documented travelling large distances (> 500 km) within Australia (Gillanders et al. 2001, Green et al. 2020, Hughes & Stewart 2020), and between Australia, Lord Howe Island and New Zealand (Gillanders et al. 2001, Holdsworth & Sippel 2016). Recent otolith oxygen isotope analyses suggest a potential seasonal migration between Victoria and NSW, where smaller younger fish live in warmer waters off NSW, and then move into cooler waters of Victoria as they age (Green et al. 2020), which is consistent with a recent tagging study which identified reciprocal movement between NSW and SA (Hughes & Stewart 2020). However, tag and recapture and acoustic telemetry studies indicate that Yellowtail Kingfish can also be relatively sedentary, remaining within a 50 km radius for an extended period of time (> one year: Gillanders et al. 2001, Hutson et al. 2007, Brodie et al. 2015). In South Australia, larger individuals have been caught within 5 km of tagging, indicating that movement can also be limited, or they return to specific locations (e.g. Coffin Bay) every year (Hutson et al. 2007). Homing has been identified for this species, although the mechanisms for this behaviour remain unclear (Dempster & Kingsford 2003).

Reproduction

Age at maturity varies between females and males where 50% maturity is reached at 3–4 years (830 mm FL), or < 1 year (470 mm FL) for females and males respectively (Gillanders et al. 2001). The development of multiple oocytes suggests that there can be multiple spawning events within a season (Poortenaar et al. 2001). Spawning occurs in austral spring and summer (Gillanders et al. 1999b, Poortenaar et al. 2001, Moran et al. 2007), when water temperatures exceed 17 °C (Moran et al. 2007).

While the specific spawning locations around Australia remain relatively unknown, a major spawning site of the Eastern Australia stock is thought to occur offshore near Lord Howe Island and Elizabeth and Middleton Reefs where a high proportion of gravid females are found during the spring/summer months (Patterson & Swearer 2008). Evidence of gravid females, and aggregations of large mature individuals have also been found near Port Augusta in South Australia (McGlennon 1997). While spawning has yet to be observed in the wild, spawning behaviour of wild-caught brood stock in New Zealand has been described (Moran et al. 2007). Courtship behaviour consists of one male and one female, where the male nips at the females gonoduct, followed by stalling and nipping and frenzied behaviour for 0.5–1.5 hours. In 50% of cases another male joins and releases gametes, with the spawning event itself lasts only ~22 seconds (Moran et al. 2007).

Early life-history

Yellowtail Kingfish are pelagic spawners, and their eggs range from 1.33–1.5 mm in diameter (Moran et al. 2007). An embryo forms 27.5 hours post fertilization (hpf), and hatching of larvae occurs between 103–108 hpf, and are approximately 4.8 mm in length (Moran et al. 2007). The development of a digestive system is quick (Chen et al. 2006), with mouth opening and exogenous feeding commencing around 2–15 days post hatch (dph), and the development of a full juvenile at 30 dph (Martínez-Montaño et al. 2016). Juveniles can be found either in large schools or solitary in coastal rocky environments or offshore to depths exceeding 300m (Stewart & Hughes 2008). Controlled laboratory studies suggest that both preferred and optimal temperatures of juvenile Yellowtail Kingfish are ~26 °C (Abbnik et al. 2012, Larios-Soranio et al. 2021), where growth, metabolic rate and activity have been shown to increase (Abbnik et al. 2012, Laubenstein et al. 2018, Larios-Soranio et al. 2021). In laboratory experiments, critical thermal minimums and maximums have been identified at 10.9–17.3 °C and 34.2–36.9 °C, respectively, for juveniles (Larios-Soranio et al. 2021), however reductions in feed intake have been identified in temperatures outside of 18–29 °C. Therefore, sustained temperatures > 18 °C are likely to be required for the successful establishment of Yellowtail Kingfish populations in Tasmanian waters.

Given the commercial and recreational value of Yellowtail Kingfish, information on the basic biology and ecology of *S. lalandi* have been well documented in New Zealand (McKenzie et al. 2014, Holdsworth et al. 2016), and mainland Australia (Gillanders et al. 1999a, b, 2001, Stewart et al. 2001, Stewart et al. 2004, Green et al. 2020). However, the majority of information on Australian populations has been inferred from NSW (see: Steffe et al. 1996, Gillanders et al. 1999a, b, 2001, Stewart et al. 2001, Stewart et al. 2004, Brodie et al. 2015, Hughes & Stewart 2020, Hughes et al. 2021), with basic biological data limited from Victoria (but see: Green et al. 2020), South Australia (Hughes et al. 2021), Western Australia (Gaughan & Santoro 2018) and absent from Tasmania. Given the evidence of a potential range shift into Tasmania, regional biological data is essential for effective management of this emerging fishery.

Results

Biological Traits

Sample collections, age validation and interpretations

From a total of 270 fish processed, 195 fish were subject to age analysis. Obtaining samples was highly dependent on season with approximately half (51%) of the fish processed being captured in February, and 92% of samples obtained being from January to March.

Although opaque zones on transverse sections could be observed, interpretation of annual banding was difficult with most fish (73%) having more than one interpretation during reading and nine fish having otoliths that were deemed to be unreadable. As such the mean readability was poor (3.3) with an inter-reading agreement of 67.2% (Figure 14A). A large proportion (48%) of the inter-reader differences were due to a difference in edge classifications (i.e. reader 1 assigned 2 years with a wide margin vs reader 2 assigning 3 years with a narrow for the same fish). Despite these difficulties, the optical characteristics of the edges of sectioned otoliths suggested an annual pattern of increment formation with the proportion of opaque edges being highest in November (Figure 14B), however due to absent or low sample numbers across most months a full seasonal profile to assign a robust opaque zone closure date was not performed. Therefore, band counts were not adjusted to account for their otolith edge classification with their date of capture and coupled with a lack of reproductive information to assign a birth date, a decimal age could not be determined. Narrow translucent edges were present in all sampled months, with their proportions being highest between December and February, thereafter wide translucent edges became the dominant edge type. Despite the difficulty in otolith interpretation some otoliths did display a clear pattern of banding as demonstrated by the largest and oldest Yellowtail Kingfish encountered (Figure 14C) and the grey value profile plot of the transect line placed across the sectioned otolith (Figure 14D). This fish was a female measuring 910 mm FL and had an estimated age of 7 yrs.

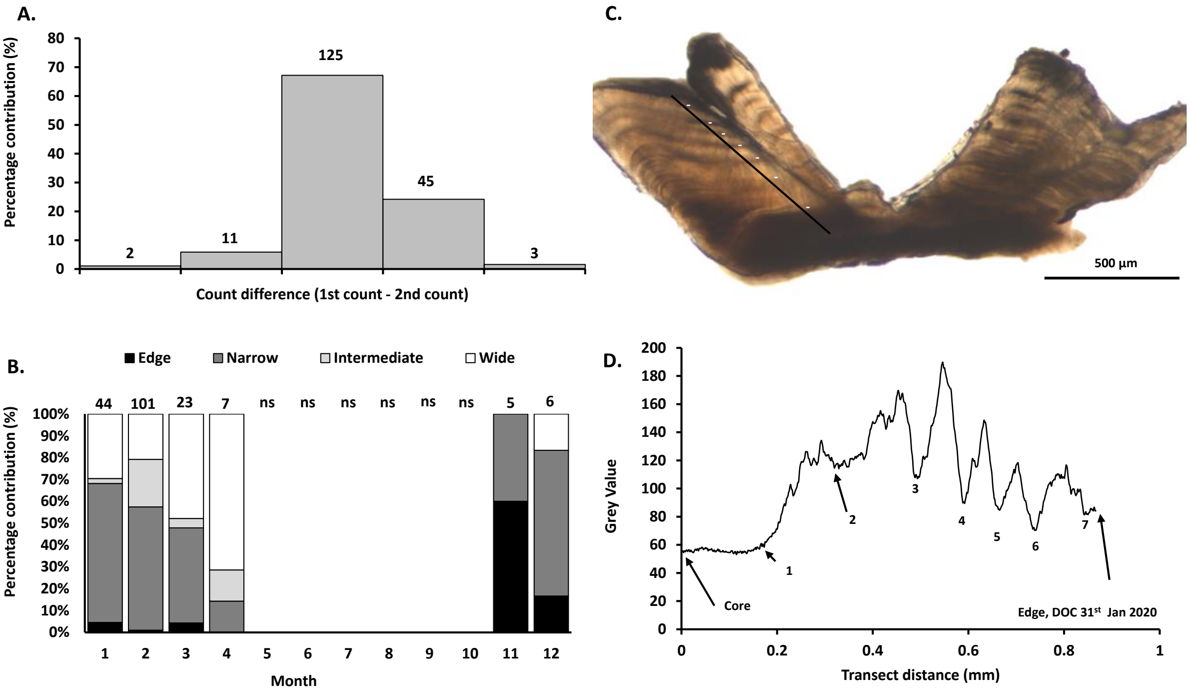

Figure 14. Inter-reading agreement (A), seasonal changes in otolith edge classification (B), photomicrograph using transmitted lighting of a transverse otolith section of a Yellowtail Kingfish (*Seriola lalandi*) estimated to be 7 years of age (C) and the resulting grey value profile plot of the transect laid in C, (where, 0 = black and 250 = white and all other values are shades of grey). Note; ns = no sample.

Sex ratios, length, age frequencies and growth

A total of 114 fish frames possessed gonadal material, which included one indeterminate juvenile (480 mm FL), 58 females (400–910 mm FL) and 55 males (415–620 mm FL). The remaining 134 fish frames lacked gonadal material and were recorded as unknown sex (280–783 mm FL) (Figure 15). The overall trends in length frequencies were similar between sexes with most fish (80%) measuring between 380 and 570 mm FL. Females and males had an average fork length (± 95% CI) of 524 ± 21 mm and 510 ± 12 mm respectively (*t*-test $p = 0.27$). Where valid χ^2 tests were possible, there were no significant biases away from a 1:1 sex ratio in any of the size classes or age classes tested (Figure 16). The oldest fish was female and estimated to be seven years of age and measured 910 mm FL. The oldest male fish encountered was estimated to be five years of age and measured 537 mm FL. The overall trends in age frequencies were similar between sexes with most fish (77%) being either 2 or 3 years of age. Females and males had an average age (± 95% CI) of 3.0 ± 0.2 years and 2.7 ± 0.2 years respectively (*t*-test $p = 0.054$). Size at age scatter plots for sex-specific size at age and all data pooled at age are presented in Figure 17. *T*-tests on the sex-specific mean size at age failed to detect evidence of sex-specific variations in growth. Due to the marked age truncation of the sample, growth models could not be developed as all attempts to model growth produced unrealistic and biologically inappropriate outputs. Moreover, observed increases in mean size between ages groups (2–5 years) were relatively small or not always evident.

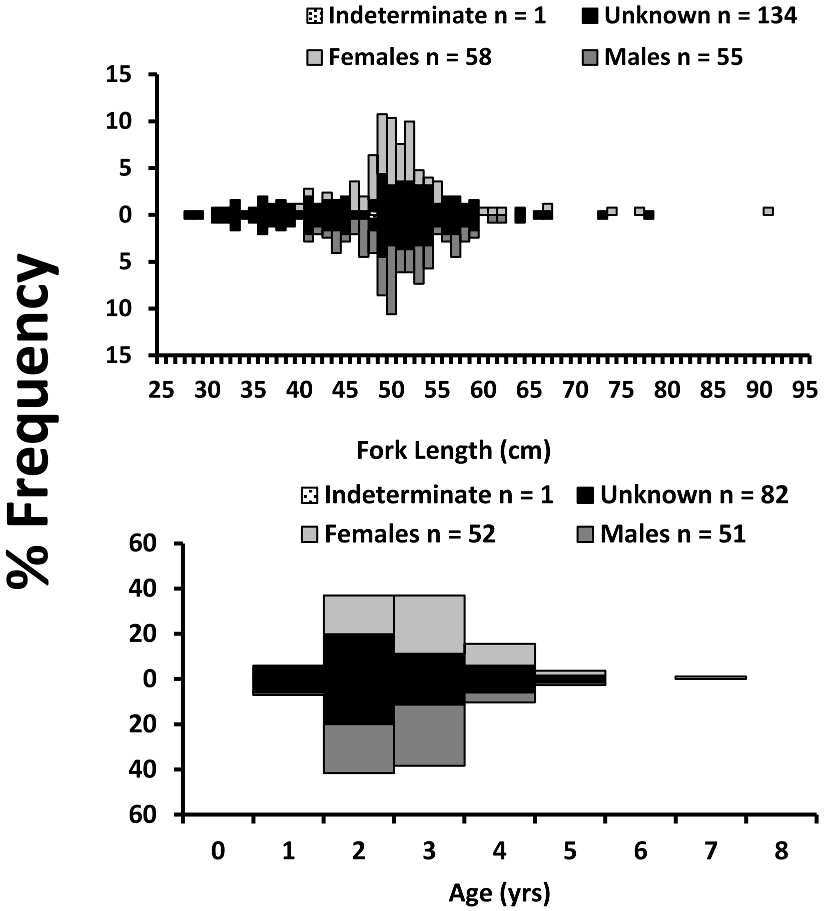

Figure 15. Comparisons of the length and age percentage contributions for male, female, indeterminate, and unknown sex Yellowtail Kingfish (*Seriola lalandi*) from Tasmania. Note values for indeterminate and unknown sex were shared between the upward and downward directions with a 1:1 ratio across all size and age classes.

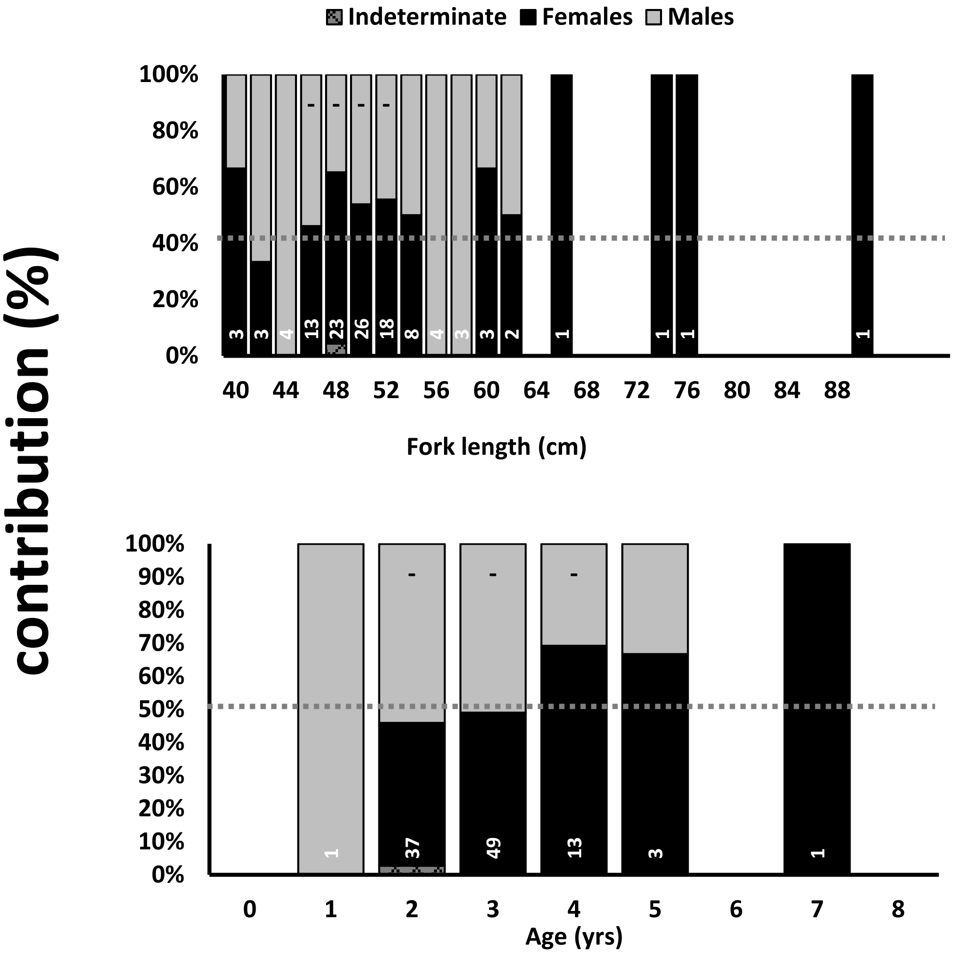

Figure 16. Percentage contribution of indeterminate, male and female, Yellowtail Kingfish (*Seriola lalandi*) from Tasmania in relation to fork length (20 mm) and age. Note: "-" indicates χ^2 square test failed to detect a difference of the male:female sex ratio away from 1:1 with a *P* value set at 0.05, values without annotation were not tested due to low *n* values (< 10) as indicated as numbers in the base of the columns in white. Note: the indeterminate individual was excluded in the χ^2 square test for this age class.

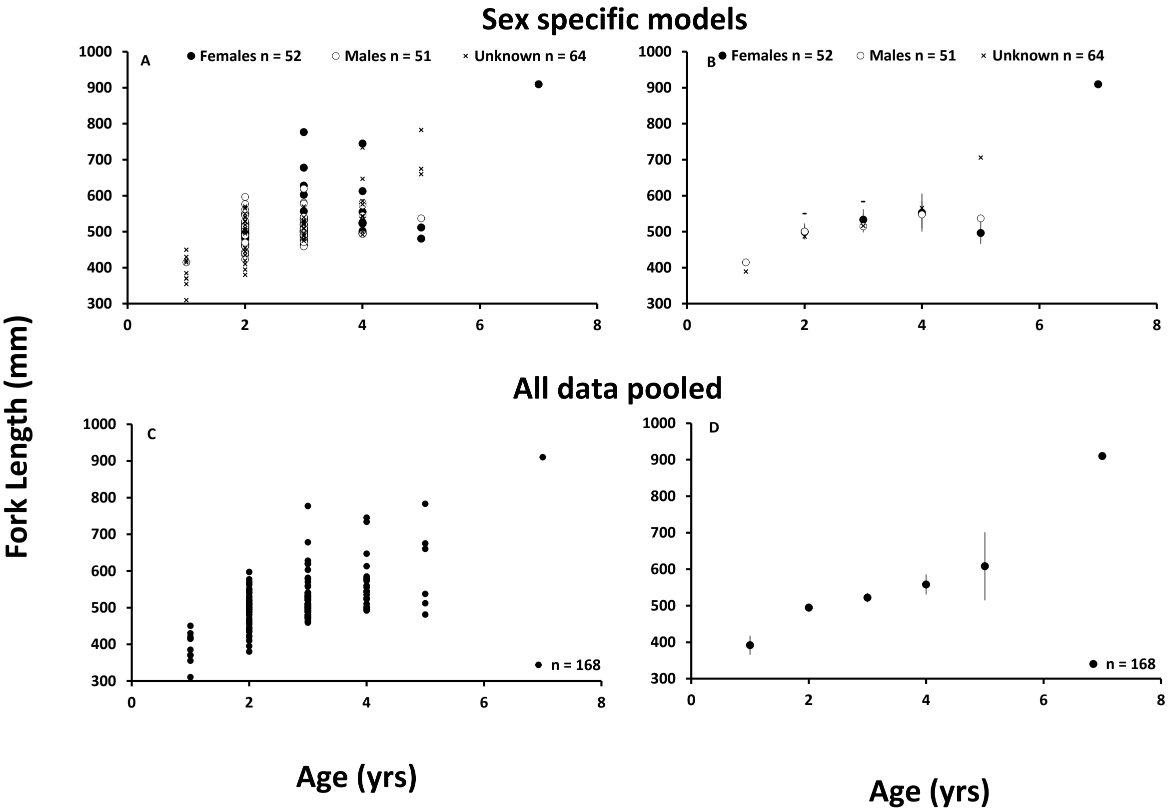

Figure 17. Sex-specific and whole stock size at age scatter plots for Yellowtail Kingfish (*Seriola lalandi*). Solid black circles and open circles represent data for females and males respectively in A and B. Paired means without annotation above were excluded from *t*-tests due to low *n* values, "-" denotes, a *t*-test was performed but no difference was detected. Error bars are 95% CI.

Reproductive biology

The only reproductively active fish encountered was a large female measuring 910 mm FL. However, despite possessing vitellogenic ovaries that weighed 84.5 grams the overall GSI value was relatively low (0.79%) when compared to the estimated weight (~10.6 kg) of the individual (Figure 18). All other fish were reproductively inactive with very small gonads with low GSI values (0.03–0.45) irrespective of age or size.

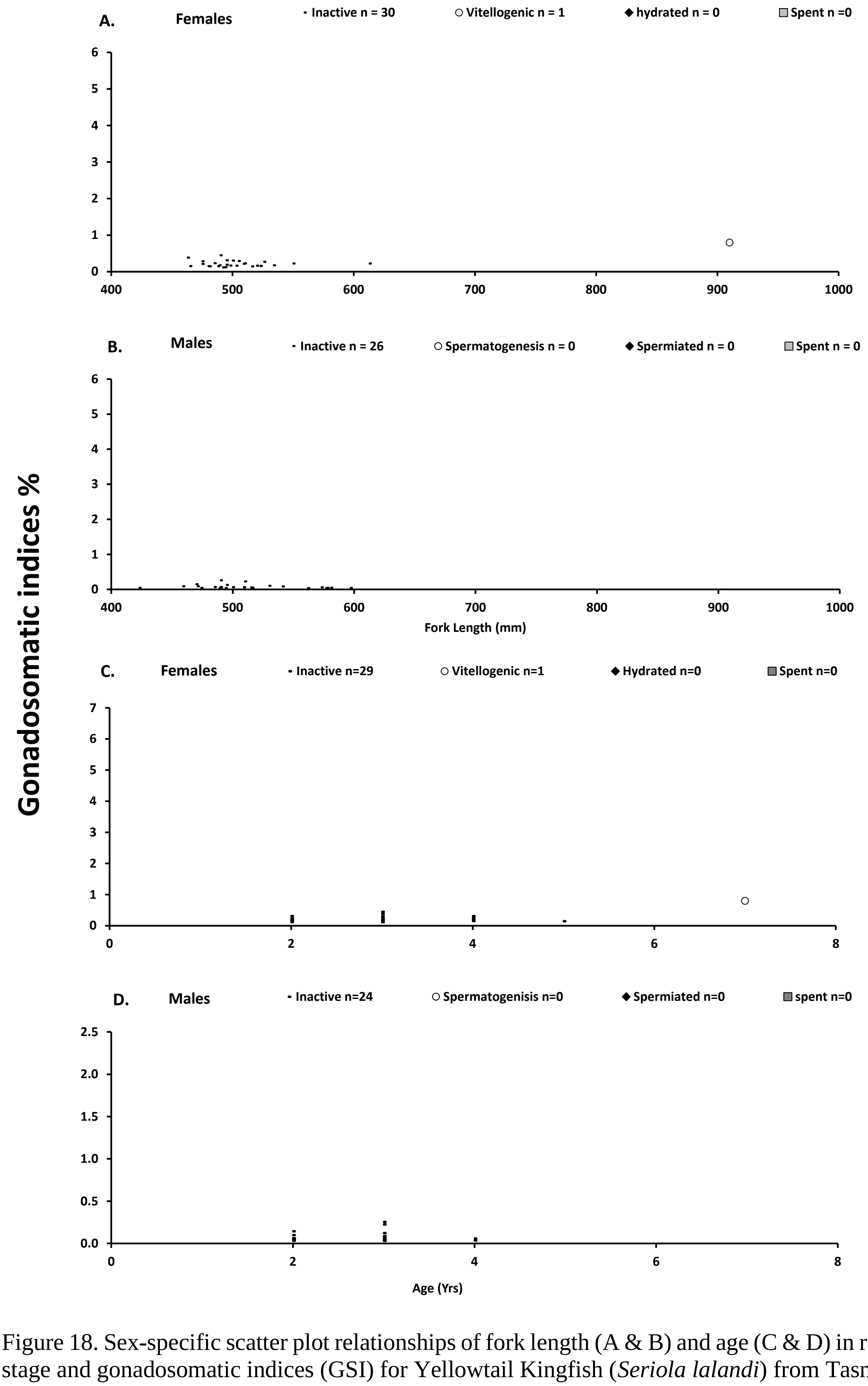

Figure 18. Sex-specific scatter plot relationships of fork length (A & B) and age (C & D) in relation to gonadal-stage and gonadosomatic indices (GSI) for Yellowtail Kingfish (*Seriola lalandi*) from Tasmania.

Diet

Twenty Yellowtail Kingfish stomachs that contained prey were recovered from seven sites (Figure 19). Prey consisted of two taxonomic classes, Actinopterygii (fish) and Malacostraca (crabs; Table 5). Most of the fish prey were in a highly digested state and generally couldn't be taxonomically resolved. However, there was one Jack Mackerel (*Trachurus* sp.) of 200 mm TL identified in one of the stomachs, and another stomach contained 25 clupeiformes of approximately 70 mm TL. The crabs that appeared in the stomachs of two fishes were the 12 mm long pelagic megalopa larvae of the red rock crab *Plagusia chabrus*. The fish with stomachs containing these larvae (18 and 6 respectively) were caught in Marion Bay in the south-east of Tasmania.

Table 5. Prey taxa identified in Yellowtail Kingfish (*Seriola lalandi*) stomach contents by the percent frequency of occurrence (% of non-empty stomachs containing each item) and mean proportion (± *SD*) of stomach contents.

Prey Class	**Frequency of occurrence**	**Mean Proportion**
Crab (*Plagusia chabrus*)	10%	7.5% ± 5.47%
Fish (Actinopterygii)	95%	92.5% ± 5.47%

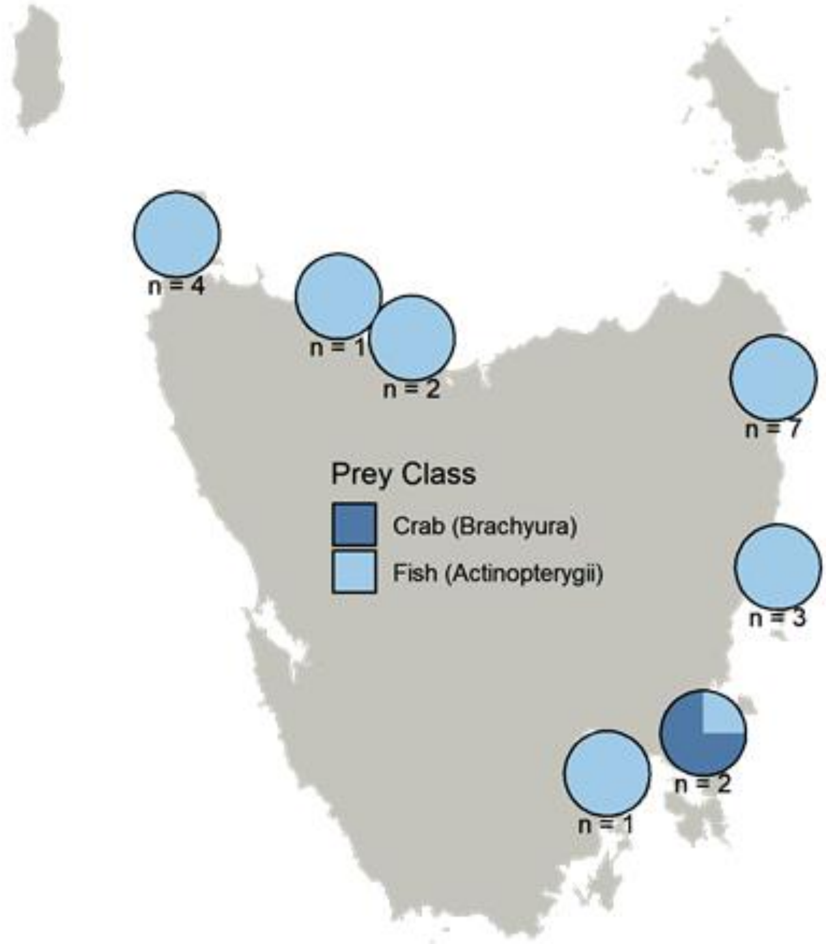

Figure 19. Mean proportional stomach contents by prey class and samples sizes (*n*) from Yellowtail Kingfish (*Seriola lalandi*) stomachs that contained prey, by region of collection (*clockwise*: Woolnorth, Cam River, Devonport, Georges Bay, Marion Bay, North-West Bay).

Fisheries Data

There was a cyclical pattern in the annual catch of the Tasmanian commercial sector (NRE Tas) between 1995–2020, however demonstrating a general decline (1243.69 kg in 1999 to 24.44 kg in 2020) in overall catch across all gear types (Figure 20a, b). Between 1995–2020 there was shift in gear types where between 2000–2005 the majority of the catch was attributed to troll gear (35–78%), and between 2005–2017 the majority of the catch was attributed to gill nets (10–100%), and recently handlining comprises the majority of the catch at 57% and 89% of the total catch for 2018 and 2019 respectively (Figure 20b). Recreational catch in 17/18 (~1700 kg) exceeded the annual commercial catch across all years (Figure 20b). Weighted linear models of catch versus latitude through time indicate an increase in latitude of 0.0447° yr^{-1} ($p < 0.0001$, $R^2 = 0.28$), equivalent to a shift of 49.7 km south per decade (Figure 20c).

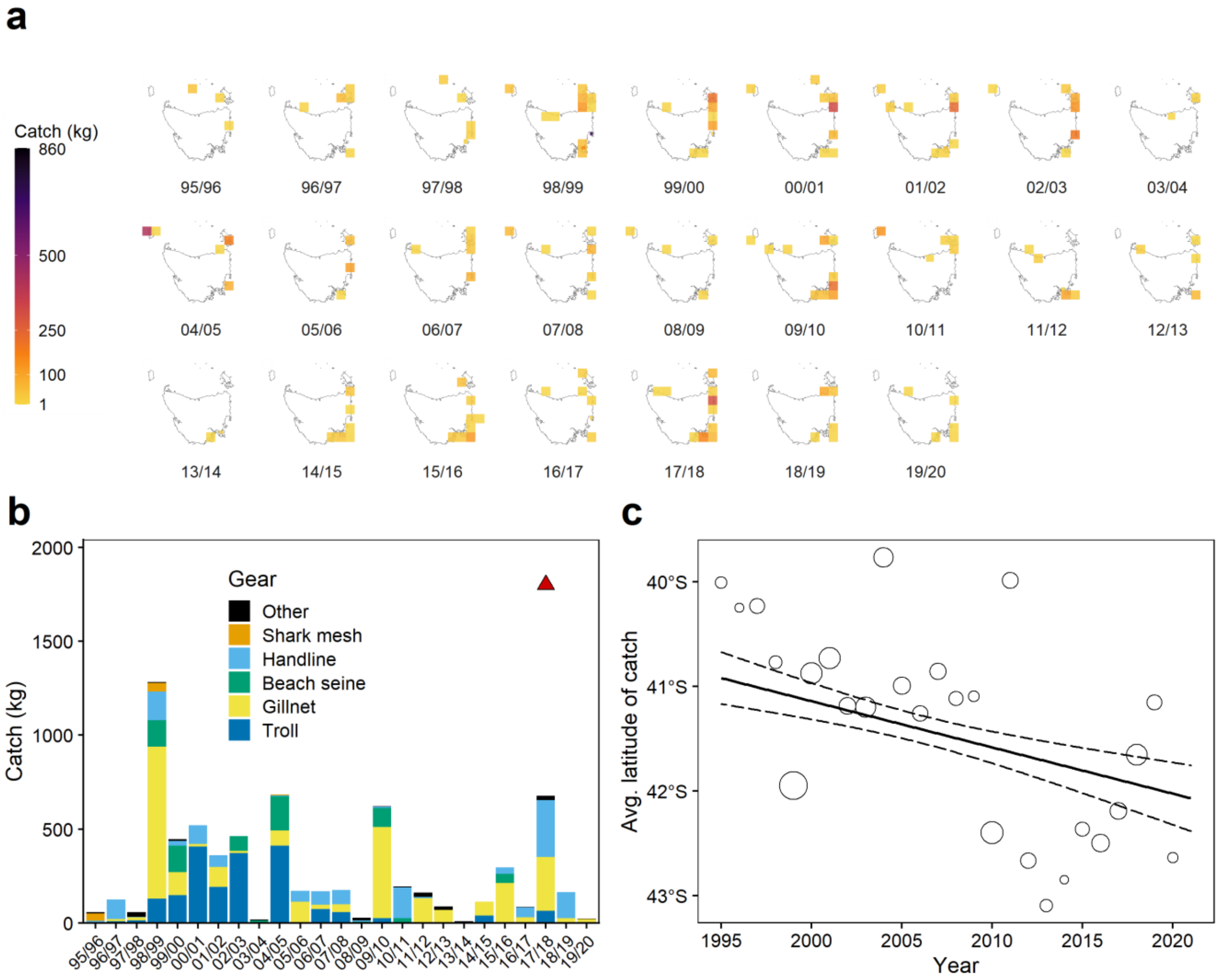

Figure 20. a) Commercial catch (kg) of Yellowtail Kingfish (*Seriola lalandi*) from Jul 1995–Apr 2020 in Tasmania and the Bass Strait. Catch by reporting block in each financial year, b) Total catch per year, by management body The triangle represents the total recreational retained catch in 2017/2018 (Lyle et al. 2019). c) Mean latitude of commercial Yellowtail Kingfish catch (reported by block) per year. The trendlines were estimated with a weighted linear model of the latitude of catch reports over time (weighted by reported catch weight).

Discussion

Biological summary

Our results suggest that Yellowtail Kingfish caught around Tasmania, particularly on the east coast are generally small, immature individuals. Unfortunately, due to the small sample size, and small proportion of older/larger individuals we were unable to model growth, mortality, or age at maturity. However, size/age composition of landings are similar in NSW and Victoria, where the NSW population is also dominated by immature individuals < 640 mm FL (2–3 years; Stewart et al. 2001, Stewart et al. 2004), where approximately 90% are < 7 years (Stewart & Hughes 2008). Similarly, the majority of landings in Victoria are < 1000 mm TL, and < 6 years (Green et al. 2020). Yellowtail Kingfish were absent from May to October, suggesting that temperatures are too cold to overwinter, and that the population is transient and seasonal.

Mortality

Due to the low sample size, we were unable to model mortality rates. As Yellowtail Kingfish do not overwinter and recruit in Tasmania, using traditional catch curve analysis to estimate mortality is inappropriate as this method assumes constant spawning and recruitment and that a proportion of the population reaches the maximum size or age. It is likely that natural mortality of Yellowtail Kingfish within Tasmania would be attributed to predation, resource limitation and disease, rather than old age. Rates of natural (M), fishing (F), and total (Z) mortality (where $Z = M + F$) have been estimated for populations in NSW (Gillanders et al. 2001, Stewart et al. 2004, Stewart & Hughes 2008), and New Zealand (McKenzie et al. 2014). Catch curve analyses estimate values of Z to range between 0.43–0.79 (Stewart et al. 2004). Rates of M were estimated to range between 0.2 and 0.14 assuming 1% and 5% of the population reach the maximum age (T_{max} = 25 years) respectively (summarised by Stewart & Hughes 2008), and 0.12 based on the rate of decay of tag returns (Gillanders et al. 2001). The most recent NSW stock assessment used multiple empirical methods to estimate Z = 0.33–0.60 and M = 0.2–0.3, resulting in an estimated F of 0.17–0.27 for individuals ranging between 2–9 years of age, noting that fishing pressure may not be the same for larger older individuals (Hughes & Stewart 2020).

Age/length

Fork length of Yellowtail Kingfish sampled from Tasmania ranged from 280 to 910 mm and spanned ages 1 to 7 yrs. Overall trends in length-frequencies were similar between sexes with most fish (80%) measuring between 380 and 570 mm FL, with a mean length of 524 mm (range 400–910 mm) and 510 mm (range: 415–620 mm) for females and males respectively. The majority (134) of individuals were of indeterminate sex with fork lengths ranging between 280–783mm. Similar to NSW and Victoria, the majority (77%) of our subsample were between the ages of 2–3. Size estimates from landings indicate that the NSW population is also dominated by small, and mostly immature individuals < 640–850 mm FL (2–3 years; Stewart et al. 2001, Stewart et al. 2004, Hughes et al. 2021) where ~90% of individuals are < 7 years old (Stewart & Hughes 2008), similar to Victoria where the majority of landings are < 1000 mm TL (< 6 years age; Green et al. 2020). However, the range in age of landings in NSW, Victoria, and New Zealand are greater than in Tasmania, with ages ranging from 0–11 (Victoria; Green et al. 2020) and 19 (NSW; Stewart et al. 2001), and 29 (New Zealand; Holdsworth et al. 2016). Commercial and recreational catch records confirm that the NSW Yellowtail Kingfish fishery has been dominated by a younger population (i.e. 2–3 years) since ~1975 (Steffe et al. 1996, Gillanders et al. 2001, Steffe & Murphy 2011), however the size and age composition of landings has remained constant, suggesting stable and consistent recruitment (Stewart et al. 2001, Hughes & Stewart 2020), and that the stock may be replenished outside of NSW (Green et al. 2020).

In addition to the absence of older individuals from our sample, there was also a lack of smaller, and younger individuals (< 2 years). While this is not surprising as we only identified one reproductive female, this confirms that juveniles are not able to recruit during the summer in Tasmania, and that it is more common to wait until ~2 years to seasonally migrate south. In contrast, the New Zealand population is dominated by an older, larger cohort (5–7 years, 550–1470 mm FL; Holdsworth et al. 2016), and a wider age range (4–23 years; McKenzie et al. 2014).

Growth

Due to the low sample size, we were also unable to model growth for Yellowtail Kingfish caught in Tasmania. However, von Bertalanffy growth parameters have been estimated for populations in NSW (incl. Lord Howe Island), Victoria and New Zealand, where there is little evidence to support sex-specific growth between males and females (Table G1). Estimated asymptotic length values (mm FL) range from 1250.2 (Gillanders et al. 1999a) and 1840 (Stewart et al. 2004) in NSW, 1350.84 in Victoria (Green et al. 2020) and 1400.58 in New Zealand (McKenzie et al. 2014). The asymptotic length of 1840 estimated by Stewart et al. (2004) has been suggested to be the most accurate as this estimate is closest to the maximum recorded length which exceeds 1900 mm TL (Smith 1987, Kailola et al. 1993, Hughes et al. 2021). However, this model assumes an age at length zero at -4.4 years (Stewart et al. 2004). Nevertheless, all growth models agree that growth of Yellowtail Kingfish is fast, where they reach 600 mm TL (the minimum legal size (MLS) for Vic, SA, and WA) within 2–3 years (Stewart et al. 2004).

Maturity

As most of our Yellowtail Kingfish samples were either of indeterminate sex, or immature males and females, we had insufficient numbers of mature individuals to perform robust size at maturity curves. However, the one reproductively mature female in our sample was 910 mm FL, and the largest non-reproductive female was 610 mm FL, which is reflective of established size at maturity estimates in the NSW and Victorian populations of Yellowtail Kingfish (Table G1: Gillanders et al. 1999b, Green et al. 2020), noting that size at maturity estimates for Yellowtail Kingfish in Victoria are slightly bigger than estimates from NSW. As such, females reach 50% maturity between 4–5 years at 834 mm FL in NSW (Gillanders et al. 1999b), and 843 mm FL in Victoria (Green et al. 2020), but can be mature as small as 680–700 mm FL (approx 3 years; Gillanders et al. 1999b, Stewart & Hughes 2008). Males mature sooner, at < 1 year, where 50% maturity occurs at 471 mm and 693 mm FL in NSW and Victoria respectively (Gillanders et al. 1999b, Green et al. 2020), but can be as small as 360 mm FL (NSW; Gillanders et al. 1999b). The males in our subsample were all reproductively inactive, and as old as five years. New Zealand populations mature at a later age, and larger size (Table G1: Poortenaar et al. 2001, McKenzie et al. 2014).

Seasonality of spawning

As our sample only consisted of one reproductively mature female, there was no evidence of a currently active reproductive population in Tasmania. Spawning generally occurs in the austral spring and summer: November–February (NSW; Gillanders et al. 1999b) and October–January (New Zealand; Poortenaar et al. 2001), where temperatures exceed 17 °C (New Zealand; Moran et al. 2007). Ninety-two percent of our samples were obtained during the mainland spawning season (i.e. from January to March). SSTs on the east coast of Tasmania range from 13–15 °C in November to 17–19 °C in February, where temperatures can exceed 20 °C in anomalous years (IMOS 2021). While the SDMs predict an increase in the persistence of suitable habitat (i.e. increase in months per year), it still remains unclear whether spawning in Tasmania is limited by temperature, as temperatures can exceed 17°C. It may be that successful spawning and recruitment requires an extended period of warmer temperatures (Larios-Soranio et al. 2021), and the presence of mature fish, which are absent from our sample.

Recruitment variability

Yellowtail Kingfish were absent from our sample from May to October (Figure 14), suggesting that temperatures are too cold for overwintering and population establishment in Tasmania and that fishing for Yellowtail Kingfish during this period is limited. The spawning potential ratio (SPR) is a useful indicator for the capacity of a population to produce enough eggs for sufficient recruitment and population turnover, calculated as the proportion of potential eggs to be spawned from a fished versus an unfished population. Most recent modelling estimates the SPR of Yellowtail Kingfish in NSW is 20% (Hughes & Stewart 2020), at the typical SPR threshold for pelagic species (Goodyear 1993). However, this method violates assumptions of uniform fishing mortality across all ages, as older and larger fish are less vulnerable to the fishery (Hughes & Stewart 2020). This could explain why there has been little evidence of recruitment failure, as the age/length composition of historical landings (since mid-1970s) has remained consistent on the mainland, comprising mainly of 2–3 year-old juveniles (Steffe et al. 1996, Gillanders et al. 2001, Steffe & Murphy 2011) indicating consistent recruitment to the Eastern Australian biological stock (Hughes & Stewart 2020).

Diet

Due to the growing interest in the aquaculture of Yellowtail Kingfish in Australia, the majority of nutritional analyses has focussed on artificial diets to maximise growth at a low cost (e.g. Moran et al. 2009, Booth et al. 2010, Booth et al. 2013). There is limited empirical information on the wild diet of Australian Yellowtail Kingfish, with the exception of a study in South Australia, where Redbait (*Emmelichthys nitidus*), Australian Herring (*Arripis georgianus*), King George Whiting (*Sillanginoides punctatus*), and small crustaceans have been found in the stomach (McGlennon 1997, Fowler et al. 2003). The results of our study suggest that Yellowtail Kingfish in Tasmania are also generalist and opportunistic carnivores, where most stomachs in our sample contained highly digested fish which couldn’t be taxonomically resolved beyond the level of class. But where identifiable, there was one Jack Mackerel (*Trachurus* sp.) and 25 clupeiformes. Furthermore, a small proportion (10%) of stomachs contained long megalopa larvae of the red rock crab (*Plagusia chabrus*). Interestingly, these were found in individuals caught in Marion Bay, where conditions are shallow and protected. This diet profile is similar with a recent study on the Californian congener, the Yellowtail Amberjack (*Seriola dorsalis*), where a large proportion (61.5%) of identifiable stomach contents comprised of a variety of small pelagic fish species (i.e. Pacific Sardine (*Sardinops sagax*), Northern Anchovy (*Engraulis mordax*), Pacific Chub Mackerel (*Scomber japonicus*), and Jack Mackerel (*Trachurus symmetricus*), but also the Pelagic Red Crab (*Pleuroncodes planipes*) was the most abundant species, appearing in 22.7% of all examined stomachs (Ben-Aderet et al. 2020). Furthermore Ben-Aderet et al. (2020) also identified an ontogenetic shift in diet, where larger fish (> 610 mm FL) had a proportionally more diverse diet than smaller fish, likely driven by an ontogenetic shift in habitat. Due to small sample size, and little variation in size composition of our sample (i.e. < 610 mm FL, and immature), it is unclear if the same pattern occurs in Australian *Seriola lalandi*. However, a generalist diet is positively correlated with the capacity of a species to undergo range shifts (Sunday et al. 2015), which may be a facilitating factor for Yellowtail Kingfish at its range edge.

Species Distribution Models

The SDMs for Yellowtail Kingfish predicted rapid poleward shifts of 94.4 km per decade for core habitat use, and 108.8 km per decade of range-edge habitat use, and an increase in temporal persistence (months per year) of suitable habitat off eastern Tasmania of approximately three months over the past 22 years (Champion et al. 2018). Furthermore, results suggest that future climate change may lead to a decline in the temporal persistence of suitable habitat in the low-latitude regions encompassing the species distribution off south-eastern Queensland (i.e. range contraction) and a concurrent increase in annual persistence of suitable habitat in higher latitude (southernmost) fishing regions of south-eastern Australia (Champion et al. 2019b). Lastly, individual fishes sampled from areas of relatively high habitat suitability were in overall better condition than those collected from areas of low habitat suitability, meaning that potential poleward shifts in preferred oceanographic habitat may mean that the body condition of Yellowtail Kingfish is not necessarily compromised as a range extension occurs (Champion et al. 2019a).

Using satellite-derived environmental data to hindcast areas of suitable habitat of Yellowtail Kingfish on the east coast of Tasmania, temporal persistence of suitable habitat increased by 0.11 months per year between 1996–2016, equating to an increase from an average of 0–1 months per year where habitat is suitable in 1996 to ~3 months per year in 2016 (Champion et al. 2019b). When forced under the IPCC RCP8.5 warming scenario, the persistence of suitable habitat was also predicted to increase, but at a slightly slower rate (0.07 months yr^{-1}), increasing from an average of 1–2 months per year in 2016 to 3–4 months in 2040 (Champion et al. 2019b). The majority (92%) of the samples for this study (2012–2021) were collected between January to March (i.e. approx. 3 months), which aligns with predictions of suitable habitat from SDMs of Yellowtail Kingfish which were undertaken in 2018 (Champion et al. 2018, Champion et al. 2019a, Champion et al. 2019b). The SDMs suggest that even under the worst-case warming scenario, it will still not be warm enough to sustain overwintering of Yellowtail Kingfish off eastern Tasmania by 2040, and therefore the establishment of a reproductive population in Tasmania. Laboratory studies on juvenile *Seriola* spp. suggest that temperatures which exceed 18 °C are required for successful growth and development (Larios-Soranio et al. 2021).

Fisheries Summary

The commercial catch for the entire Eastern Australian biological stock (i.e. NSW, QLD, SA, Victoria, Tasmania) has declined from an annual catch of 311.04 t in 2012 to 155.59 t in 2013, but has remained stable between 91.18 t–175.68 t from 2013–2019 (Hughes et al. 2021). Most of the information determining the stock status of the Eastern Australia stock is derived from NSW. Reported annual catch of the NSW commercial sector has also declined by ~500 t between the 1980s and the 1990s following the ban of pelagic fish traps (Stewart et al. 2001, Stewart & Hughes 2008), and the fishery has since remained relatively stable at ~85–160 t per year (Pecl et al. 2011, Hughes & Stewart 2020). However, the catch per unit effort (CPUE) in handlining (the main method in the commercial sector) has increased by ~12 kg day^{-1} from 1997 to 2002, but since then has remained relatively stable at ~45 kg day^{-1} (Hughes & Stewart 2020). Interestingly, the use of handlines in the Tasmanian commercial sector has increased since 2005, where most of the catch in 2019 was from handlines (Figure 20). Reported recreational catch in NSW has also remained relatively stable since 2000–2001 (144–129 t; Henry & Lyle 2003), with a potential small increase in catch (2013–2014: 120 t, 2017–2018; 129 t; Murphy et al. 2020). However, as methodologies between recreational fisher surveys differ, they are not directly comparable.

Similar patterns are found across other mainland states. In Victoria, commercial landings also decreased from 2–12 t from 1980–1993 to less than 1 t annually until ~2013. Similarly, recreational catch also declined during this time, presumably due to the decrease in both the size and availability of Yellowtail Kingfish (Hughes et al. 2021). However, since ~2010 there has been an increase in popularity of Yellowtail Kingfish among recreational anglers, potentially indicating that their availability is increasing, however, the majority of reported catch in Victoria are small (i.e. < 100 cm) and immature (< 6 years; Green et al. 2020). Annual reported commercial and recreational catch of Yellowtail Kingfish in Queensland remains relatively low, where annual commercial catch has varied between 3–14 t between 2004–2009 (QFish 2020), and annual recreational catch was 9 t in 2013–2014 (Webley et al. 2015). In SA, Yellowtail Kingfish makes a small proportion (< 3 t) of the annual catch in the Marine Scalefish Fishery and is usually retained as bycatch. Conversely, Yellowtail Kingfish is an increasingly popular recreational fishing target in SA, with reported catch from recreational fishing surveys increasing from 62 t in 2000–2001 (Henry & Lyle 2003), to 100 t in 2007–2008 (Jones 2009), to 199 t in 2013–2014 (Giri & Hall 2015).

Commercial catch for the Western Australia stock remains considerably lower than the Eastern biological stock (total annual catch of 2.89 t in the 2018–2019 financial year), with some years reporting an annual catch of zero. Commercial records of the WA stock commenced in 2014 with an annual catch of 1.75 t in the 2013–2014 financial year, with no reported catch for 2015, and 2016, after which the annual catch increased to 3.62 t in the 2017–2018 financial year (Hughes et al. 2021). The recreational catch of Yellowtail Kingfish in Western Australia also exceeds records of commercial catch with 7 t recorded as harvested in the last recreational fisher survey (Ryan et al. 2019). Similarly, in NSW, SA, QLD, and Tasmania recreational catch exceeded commercial catch (Table 6).

Long-term patterns within commercial and recreational catch of Yellowtail Kingfish in Tasmania are similar to those apparent for mainland populations, albeit at a proportion of the volume, where commercial catch has remained relatively stable since ~2005 and recreational catch exceeds commercial catch (Figure 20). Furthermore, the mean latitude of commercial catch (NRE Tas) has shifted south by ~50 km per decade from 1995–2020. This spatial redistribution of catch may be indicative of Yellowtail Kingfish undergoing a poleward range shift. However, disentangling this trend from possible spatial redistributions in fishing effort is complicated. Currently there is not enough information to include Yellowtail Kingfish in the Tasmanian stock assessment reports. Specific catch and size limits for Yellowtail Kingfish were introduced in 2015, with a bag limit of five, and an MLS of 450 mm TL. Prior to this there were no size limits and a general scalefish possession limit of 15 was applied.

Table 6. Total annual harvest (tonnes (t)) from commercial and recreational catch of Yellowtail Kingfish (*Seriola lalandi*).

	Queensland	**New South Wales**	**Victoria**	**South Australia**	**Western Australia**	**Tasmania**
Commercial[1]	7.97 t range 3–14 between 2004–2019[2]	75.62 t[3]	< 1 t[1]	1.82 t[1]	2.89 t[1]	< 1 t since 1995–96[1]
Recreational	9 t[4] 2013–14	129 t[5] 2017–18	< 1 t[6] 2000–01	199 t[7] 2013–14	7 t[8] 2017–18	1.7 t* 2017–18[9] 1000 individuals

*estimated from average size from Tasmanian subsample

[1] Hughes et al. 2021.
[2] QFish 2020.
[3] Hughes & Stewart 2020.
[4] Webley et al. 2015.
[5] Murphy et al. 2020.
[6] Henry & Lyle 2003.
[7] Giri & Hall 2015.
[8] Ryan et al. 2019.
[9] Lyle et al. 2019.

Implications for Management

The Minimum Legal Size (MLS) controls the length of fish that can legally be targeted and are typically based on the size of an individual at first maturity (PIRSA 2021). Currently the MLS for Yellowtail Kingfish varies between states with Tasmania being the smallest at 450 mm TL (NRE Tas 2022a) compared to 600 in Queensland (Queensland Government 2021), Victoria (VFA 2021b), SA

(PIRSA 2021) and 650 mm TL in NSW (NSW DPI 2021). An increase from 600 to 650 mm in MLS was implemented in NSW after Stewart et al. (2004) determined they were growth overfished. However, an MLS of 650 mm still equates to individuals ~2–3 years, and mostly immature (Gillanders et al. 1999b). As Yellowtail Kingfish reach ~500 mm FL after 1 year (Stewart et al. 2001), the current MLS in Tasmania allows the targeting of virgin individuals < 1 year old, though males from other states reach 50% maturity between 471 mm and 693 mm FL. Therefore, the current size limit in other states may offer some protection to mature male Yellowtail Kingfish. But as fish > 650 mm TL were rare in our sample, this MLS of 450 mm ensures access of recreational anglers to Yellowtail Kingfish in Tasmania. Given that Yellowtail Kingfish are likely absent from May to October, the fishing pressure on smaller individuals in Tasmania is for a relatively short (~4 months) amount of time. However, if catches do increase, an alternative strategy may include reducing the bag limit, rather than the MLS. As the population in Tasmania is seasonal and transient, evidence of growth-overfishing would be found in the Victorian and NSW populations. Therefore, the regularity in the age/length composition of landings in NSW does not provide evidence of growth-overfishing, suggesting that the small MLS in Tasmania is having little effect on the Eastern Australian biological stock.

However, recent yield per recruit analyses suggests that under current exploitation levels ($F = 0.23$), the yield per recruit is maximised at approximately 740 mm TL (650 mm FL), 100 mm above the current MLS in NSW, suggesting that the current management controls may limit optimum recruitment of Yellowtail Kingfish in NSW (Stewart & Hughes 2008). Furthermore, the high rates of recapture of tagged individuals (12–15%) suggests high levels of concentrated fishing effort and exploitation (Hughes & Stewart 2020). The current annual total harvest of Yellowtail Kingfish in NSW is 200–330 t, which is well below the maximum sustainable yield of 475 t (Hughes & Stewart 2020). This consideration, coupled with long-term stability in age/size structure (Steffe et al. 1996, Gillanders et al. 2001, Steffe & Murphy 2011), and anecdotal evidence of increased availability of Yellowtail Kingfish in Victoria (Green et al. 2020), supports a ‘sustainable’ stock-status classification.

Most of the information used in the Eastern Biological Stock assessment is derived from data from the NSW fishery, likely due to its proximity to the continental shelf, large recreational and game fishing community (Griffiths 2012) and successful tag and recapture programmes (Gillanders et al. 2001). As such, data collection efforts within the other states remains minimal (Hughes & Stewart 2020, Hughes et al. 2021). However, a recent assessment from Victoria provided useful information on size-structure and genetic connectivity (Green et al. 2020). Currently there is not enough information or frequency of occurrence to include Yellowtail Kingfish in the Tasmanian Fishery Stock Assessment. However, ongoing monitoring and data collection should continue to support the stock assessments of the Eastern Biological Stock, and to determine changes to the ‘Tasmanian’ population at its range edge.

Snapper

Introduction

Snapper (*Chrysophrys auratus* Forster, in Bloch and Schneider 1801, formerly *Pagrus auratus*) are a member of the family Sparidae confined to temperate and subtropical waters of Australia and New Zealand. Snapper are a demersal generalist predator that occur in estuarine and coastal habitats to 200 m depth (MacDonald 1982, Kailola et al. 1993). Snapper live to 40 years across much of their range and can reach total lengths of 130 cm and 20 kg (Gomon et al. 2008, Norriss & Crisafulli 2010).

Distribution

In Australia, the species has a continuous distribution from Hinchinbrook Island, QLD, to Barrow Island, WA (Kailola et al. 1993) and also occurs around Norfolk Island and Lord Howe Island (Paulin 1990). In New Zealand, Snapper occur around the North Island, northern part of the South Island and only occasionally further south (Crossland 1981). The southern distributional limit across the species' range led to the inference that an environmental factor, likely temperature, was a limiting factor to their distribution (Johnston 1882, Crossland 1981, MacDonald 1982).

Available evidence suggests the Australian population is undergoing a poleward range extension into Tasmania (Robinson et al. 2015). Historically, Snapper were noted to rarely occur in northern Tasmania, which Johnston (1882) attributed to an affinity for warmer water, and a lack of preferred deep fringing reef along the northern coast of Tasmania where conditions otherwise were assumed suitable for the species. Snapper was included in subsequent checklists of Tasmanian fishes with no specific mention of distribution, but excluded from a list of species brought to market in Tasmania (Johnston 1890, Lord 1922). Bones identified as Snapper have been found in archaeological excavations of the South-Eastern Tasmanian Lagoon Bay and Adventure Bay whaling stations, which were active between 1829–1854 (Lawrence & Tucker 2013). The authors note however that the fish may have been caught elsewhere during whaling operations, and further misidentification of small bones of more common similar species (e.g. *Acanthopagrus butcheri*) is presumably a possibility. Lord and Scott (1924) noted the species occasionally was caught along both the northern and eastern coasts of Tasmania. Snapper were recorded among steam trawl catches in the vicinity of Great Oyster Bay, eastern Tasmania in December 1939 (Klaer 1991). More recently, Last et al. (2011) noted that Snapper were uncommon from north-eastern Tasmania to Bruny Island in the south-east in the 1980s, but present in this region as of 2009. Further, the authors assumed Snapper to be absent from the west and south-west of the state from a lack of evidence. Since 2008, small amounts of catch have been reported by Commonwealth commercial fishers off the west coast of Tasmania (< 260 kg annually, NRE Tas).

Connectivity/Stock structure

Within the continuous distribution of Snapper around the southern half of Australia, the population structure of Snapper is complex, with nine biological stocks recognised. Relevant to the range extension in the south-east are the adjacent Western Victoria stock, Eastern Victoria stock, and East Coast stock. The East Coast stock is genetically distinct from the population to the south, with a breakpoint near Eden, around which admixture appears to occur over several hundred kms (Morgan et al. 2018). To the south, conventional tagging and otolith chemistry analyses demonstrate that the population is structured into Western and Eastern Victoria stocks, due to limited connectivity across Wilson's Promontory (Sanders 1974, Coutin et al. 2003, Hamer et al. 2011). The Western Victoria stock, which extends from Wilson's Promontory west to the mouth of the Murray River, is largely dependent on recruitment from Port Philip Bay (Hamer et al. 2011), while recruitment into the Eastern Victoria and East Coast stocks originates from local coastal spawning (Gillanders 2002, Hamer &

Jenkins 2004). It follows that for north-west Tasmania and King Island, the nearest mainland source of recruitment and corresponding stock is Port Philip Bay/Western Victoria stock. For Tasmania, east of the Tamar River as well as the Furneaux Island Group, the Eastern Victoria stock and recruitment from spawning along the coast of eastern Victoria and southern New South Wales is closest in proximity. A study of the genetic structure of Snapper on the east and south-east coasts of Australia found that samples from Tasmania (*n* = 29) clustered with the Victorian population south of Eden, however all but four of the samples were collected from the north-west and mid-north coast of Tasmania and the authors noted that the Tasmanian sample may have included more than one genetic stock (Morgan et al. 2018).

Fisheries and Management

Snapper are managed as twelve units in Australia, typically at the scale of biological stocks. An exception are the continuous biological stocks along the west and east coasts of Australia, which are monitored in separate units at regional (Shark Bay Oceanic, West Coast, South Coast) and jurisdictional (Queensland and New South Wales) levels, respectively. While Snapper in the continuous west and east coast stocks are genetically similar, mixing is limited by distance and demographic characteristics vary with latitude (Wakefield et al. 2015, 2016, Stewart et al. 2020). Recruitment and patterns of depletion in these stocks have also been observed to be localised, (Gillanders 2002, Wortmann et al. 2018), further supporting management of these stocks at the regional scale.

Currently, of the twelve management units, six are considered ‘sustainable’, four (Queensland, Gulf St. Vincent, Spencer Gulf/West Coast SA, and Shark Bay Oceanic) are ‘depleted’, the West Coast is ‘recovering’, and the East Victoria stock is ‘undefined’ (Fowler et al. 2021b). Commercial catch of Snapper has declined across Australia over the past decade, from a high of approximately 1800 t in 2011, to 619 t in 2019. Snapper are an important recreational fishery target, with recreational catch, often on par with or exceeding commercial catch (Ford & Gilmour 2013, Giri & Hall 2015). In Tasmania, over 3900 Snapper (± 33%) were estimated to have been caught by recreational fishers in 2017–2018, with over 60% released (Lyle et al. 2019). Since 2004–2005, annual commercial catch by these gears has remained under 200 kg annually, and annual catch has remained under 500 kg per financial year. Total reported commercial catch of Snapper has averaged approximately 300 kg per financial year in Tasmanian commercial managed fisheries (i.e. NRE Tas) since 1995.

Movements

Snapper demonstrate a range of movement patterns across age classes, habitats, and individuals. Snapper often exhibit site attachment, especially when associated with reef habitats. For example, the majority of Snapper tagged in a New South Wales reserve remained resident to a ~2 km^2 reef patch for up to several years (Harasti et al. 2015). A similar pattern of generally high residency has been documented among New Zealand reef-associated Snapper (Willis et al. 2001, Parsons et al. 2003, Egli & Babcock 2004). Much of the work on reef-associated Snapper movements has been derived from acoustic telemetry tagging within marine reserves, which appear to encourage hyper-residency among Snapper (Parsons et al. 2010). Similar patterns have been demonstrated with conventional tagging of Snapper on unprotected reefs in the Harukai Gulf, New Zealand, which were recaptured a median of 0.7 km distance from tagging locations over the two years following tagging. Snapper can also demonstrate restricted movements in estuarine habitats (Hartill et al. 2003). For example, 85% of Snapper tagged in the vicinity of Moreton Bay (~28°S) were recaptured within 1 km of their release location (Sumpton et al. 2003).

Broader movements of tens of kilometres have been demonstrated among sediment-associated Snapper in bays, often seasonal or related to spawning (Crossland 1976, 1982). In the northern Gulf St Vincent, Snapper readily make episodic movements of > 10 km per day between habitat features, to which many remain temporarily or seasonally site-attached (Fowler et al. 2017). While juvenile Snapper remain resident on reefs in Port Philip Bay year-round, adult Snapper migrate from coastal Victorian waters into Port Philip Bay in September and October for the duration of the spawning season, a pattern that is repeated across years (Hamer & Mills 2017). Long-distance movement may not be uncommon among Snapper in some areas, for example Sanders (1974) reported that of 1080 Snapper tagged near Mallacoota, Victoria, 71% (44 of 62) of recaptures of known location were > 200 km away, all toward lower latitude along the east coast. Worth noting is that among studies of largely site-attached Snapper, there is often a contingent proportion (~5–10%) documented that exhibit movements over much greater scales of 10s to 100s of kilometres (Sumpton et al. 2003, Parsons et al. 2011, Harasti et al. 2015, Stewart et al. 2020).

Ontogenetic movements of juvenile Snapper from estuaries to coastal habitats is a key process for coastal replenishment of Snapper. This emigration can occur locally (Gillanders 2002), but among higher latitude stocks, it often occurs across the regional scale. Otolith chemistry of 9+ year class Snapper sampled across 2000 km of South Australia’s coastline was consistent with a single nursery origin from which Snapper emigrate at between 3+ and 5+ years of age (Fowler et al. 2005). A similar analysis in Victoria demonstrated that by the 4+–5+ year classes, over > 70% of Snapper along 800 km of western and mid-Victoria coastline originate from Port Philip Bay (Hamer et al. 2011).

Assuming the Bass Strait (~50–80 m deep) is not a barrier to movement for Snapper, the distances over which the occasional long-distance movements of adults and ontogenetic movement patterns of 3+–5+ age juveniles are known to occur suggest movements of these life stages into Tasmania are possible from the East Victoria or West Victoria stocks.

Reproduction

Age at maturity varies considerably across the Australian range of Snapper, with median total length at maturity ranging from 260 to 600 mm, and age of maturity ranging from 1.7–7 years (Stewart et al. 2010, Wakefield et al. 2015). Snapper are multiple batch spawners capable of spawning daily (Scott et al. 1993, Wakefield 2010, Saunders et al. 2012). While spawning can occur in shallow open coastal waters along the east coast of mainland Australia (MacDonald 1982, Hamer & Jenkins 2004), in the southern portion of Australia spawning occurs in aggregations in protected embayments, such as Cockburn Sound, Spencer Gulf and Port Philip Bay (Wakefield 2010, Hamer et al. 2011, Saunders et al. 2012). These spawning aggregations are thought to be typically made up of transient Snapper from coastal areas, which has been confirmed in Port Philip Bay with acoustic tracking of adults annually migrating into the bay during spawning season (Hamer & Mills 2017). Similar habitats exist in Tasmania, such as Storm Bay and its tributaries in the south-east (43°S), which presumably could support Snapper spawning aggregations. While protected embayments that may be suitable for spawning aggregations exist in the extended range, successful spawning would still be contingent on suitable water temperatures, as spawning is highly temperature dependent.

Seasonality of spawning varies markedly across the range of Snapper due to seasonal water temperatures. At lower latitudes on both the east (21–24°S; Ferrell & Sumpton 1997) and west coasts of Australia (25°S; Jackson et al. 2010, Wakefield et al. 2015) spawning occurs from May to September, coinciding with the decline of seasonal water temperatures toward the seasonal minimum of 20–21 °C. At higher latitudes, the onset of spawning coincides with increasing day length and seasonal water temperatures in mid to late spring.

Early life-history

Snapper eggs hatch after 36 hours at 21 °C, and 48 hours at 18 °C (Cassie 1956). At hatching, larvae are 3.1 mm total length, and start feeding as soon as three to four days post-hatch (Battaglene & Talbot 1992, Fielder et al. 2005). Snapper have a pelagic larval duration of 18–32 days (Francis 1994, Fowler & Jennings 2003), and protected estuaries are preferred settlement habitats (Trnski 2002), but may not be essential (Parsons et al. 2014). Once tail flexion has developed late in the pelagic larval duration period, Snapper larvae are competent swimmers, and can swim 5–10 km without feeding (Kingsford & Atkinson 1995, Clark et al. 2005). Larvae respond to olfactory cues from preferred settlement habitats and are thought to be capable of capitalising on tidal flows and wind-driven currents for transport (Trnski 2002, Radford et al. 2012, Sim-Smith et al. 2013). As spawning in the East Victoria stock occurs along the coast (Hamer & Jenkins 2004), it may be possible for competent larvae to be advected by the East Australia Current to the east coast of Tasmania, where they may be capable of locating and moving into suitable settlement habitats. However, even if hydrodynamic conditions could facilitate recruitment into Tasmania, survival of Snapper larvae is dependent on prey availability and suitable water temperatures (Murphy et al. 2013). Year-class strength of Snapper in temperate areas is highly variable, fluctuating 7–20x from year to year (Hamer & Jenkins 2004). This in part appears to be due to the length suitable temperatures are available over the spawning and larval settlement season. Survival of larvae decreases outside of 15–24 °C and growth increases with temperature over this range, given sufficiently high prey density (Fielder et al. 2005). Larval abundance in Port Philip Bay, Victoria was greatest in years with high copepod nauplii prey abundance and higher relative temperatures (up to 21 °C; Murphy et al. 2013). Temperatures less than about 18 °C are thought to result in very high larval mortality (Parsons et al. 2014), although high prey abundances may be able to offset the deleterious effect of slightly lower temperatures (Zeldis et al. 2005).

Post-settlement Snapper are associated with soft sediment habitats, perhaps due to food availability and low water movement (Fowler & Jennings 2003). Such habitats with biogenic structures like seagrass meadows, which likely provide refuge, are especially favoured (Jackson et al. 2007, Parsons et al. 2014). After settlement once entering estuaries, larvae are likely capable of selecting adventageous habitat through post-settlement migration at 30 mm standard length (SL: Hamer & Jenkins 2004). Older, 1- and 2-year-old juveniles prefer higher structural complexity (Thrush et al. 2002). Growth and survival of juvenile Snapper is highly temperature dependent, which likely influences year-class variability. At winter temperatures < 14 °C, Francis (1994) reported growth of 0+ age Snapper substantially declined and ceased entirely during winter for the 2+ and 3+ age classes. Early 0+ individuals put all available energy into growth rather than into energetic storage reserves until autumn, thus too short of a period of temperatures favourable for growth may not allow sufficient reserves to be accumulated to prevent over-winter starvation. Thus, even if hydrodynamic and thermal conditions allow for successful spawning and settlement of larvae into extended-range nursery habitats, survival of post-settlement 0+ Snapper requires a sufficiently long period post-settlement of temperatures warm enough to permit growth (> 14 °C at minimum) prior to winter to permit persistence in range extended areas.

Methods

Due to the high abundance of samples of Snapper collected across the state, we were able to compare life-history characteristics between the north and south coasts of Tasmania. The northern stratum included Snapper caught in St. Helens (41.3°S; inclusive, n = 293) and northward. The southern stratum, included Snapper caught in waters south of St. Helens (41.3°S; exclusive, n = 144).

Results

Biological Traits

Age validation and interpretations

From a total of 437 fish, 424 were subjected to age analysis. Opaque zones on transverse sections were easily discernible with an inter-reading agreement of 93% (Figure 21A). Eight fish had unreadable otoliths. The optical characteristics of the edges of sectioned otolith displayed a clear seasonal pattern, irrespective of capture region. Opaque edges were present from October to April with a peak occurrence in December. Narrow translucent edges were present from November to April with occurrence highest from January to March. Intermediate edges were present from December to June with a peak occurrence in May. Thereafter, wide translucent edges dominated between June to November. These observations were used to confirm the banding periodicity was annual and to assign an opaque zone closure date of 1st January (Figure 21B) for all regions. Alternating bands were easily recognisable as demonstrated by the image in Figure 21C and the grey value profile plot of the transect line placed across the sectioned otolith (Figure 21D). The fish depicted was the oldest Snapper encountered and had an estimated decimal age of 19.2 years.

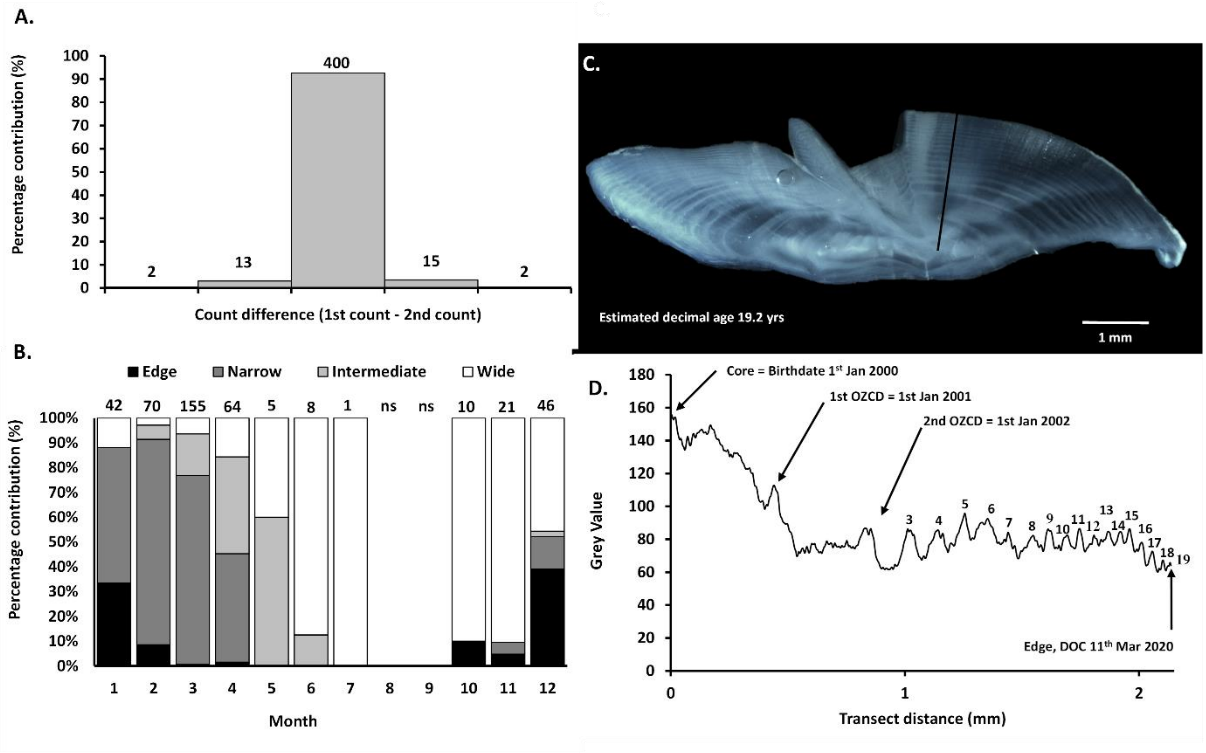

Figure 21. (A) Inter reading agreement, (B) seasonal changes in otolith edge classification, (C) photomicrograph of a transverse otolith section from a Snapper (*Chrysophrys auratus*) estimated to be 19.19 years of age and (D) the resulting grey value profile plot of the transect laid in C, (where, 0 = black, 250 = white and all other values are shades of grey). Note birth date (core) and opaque zone closure date (OZCD) were assigned to 1st January, and date of capture was 11th March 2020. ns = no sample.

Sex ratios, length and age frequencies

A total of 316 fish frames possessed gonadal material, which included 45 indeterminate juveniles (150–316 mm FL), 152 females (241–700 mm FL) and 119 males (233–633 mm FL). The remaining 122 fish frames lacked gonadal material and were recorded as unknown sex (268–779 mm FL) (Figure 22). Although more females than males were encountered, the overall trends in length frequencies were similar between sexes within each region. Females and males reached similar maximum sizes with an average fork length (± 95% CI) for legal sized (≥ 261 mm FL) fish being 445 ± 18 mm and 434 ± 19 mm respectively (t-test: $p = 0.418$). Where valid χ^2 tests were possible within each 20 mm size class, few significant biases away from a 1:1 sex ratio could be detected. Moreover, the few significant findings detected were relatively scattered and most likely indicate a sample effect rather than a true biological effect (Figure 23). There were markedly different size compositions between regions, with many of the larger fish ≥ 500 mm FL being obtained from the North Coast (87% of $n = 107$), and most of the fish between 270–340 mm FL being obtained from the South Coast

(81% of $n = 90$). As a result, the mean size of legal sized (≥ 261 mm FL) fish between region was significantly different (*t*-test: $p < 0.001$) with the average fork length (± 95% CI) from the North Coast being 478 ± 15 mm and 373 ± 19 mm for the South Coast.

The oldest fish was estimated to be 19.2 years and measured 627 mm FL (sex unknown). The oldest female and male Snapper were estimated to be 18.0 years and 12.8 years old respectively. Analysis of age frequencies reflected very similar findings to the length frequency distributions, no significant differences between mean age in relation to sex (females = 7.3 ± 0.5 years and males = 7.0 ± 0.5 years; *t*-test: $p = 0.350$) and significant differences between regions (North = 8.1 ± 0.4 years and South = 5.6 ± 0.6 years; *t*-test: $p < 0.001$). Where valid χ^2 tests were possible a few significant biases away from a 1:1 sex ratio were detected. The few significant findings detected were isolated and most likely indicate a sample effect rather than a true biological effect.

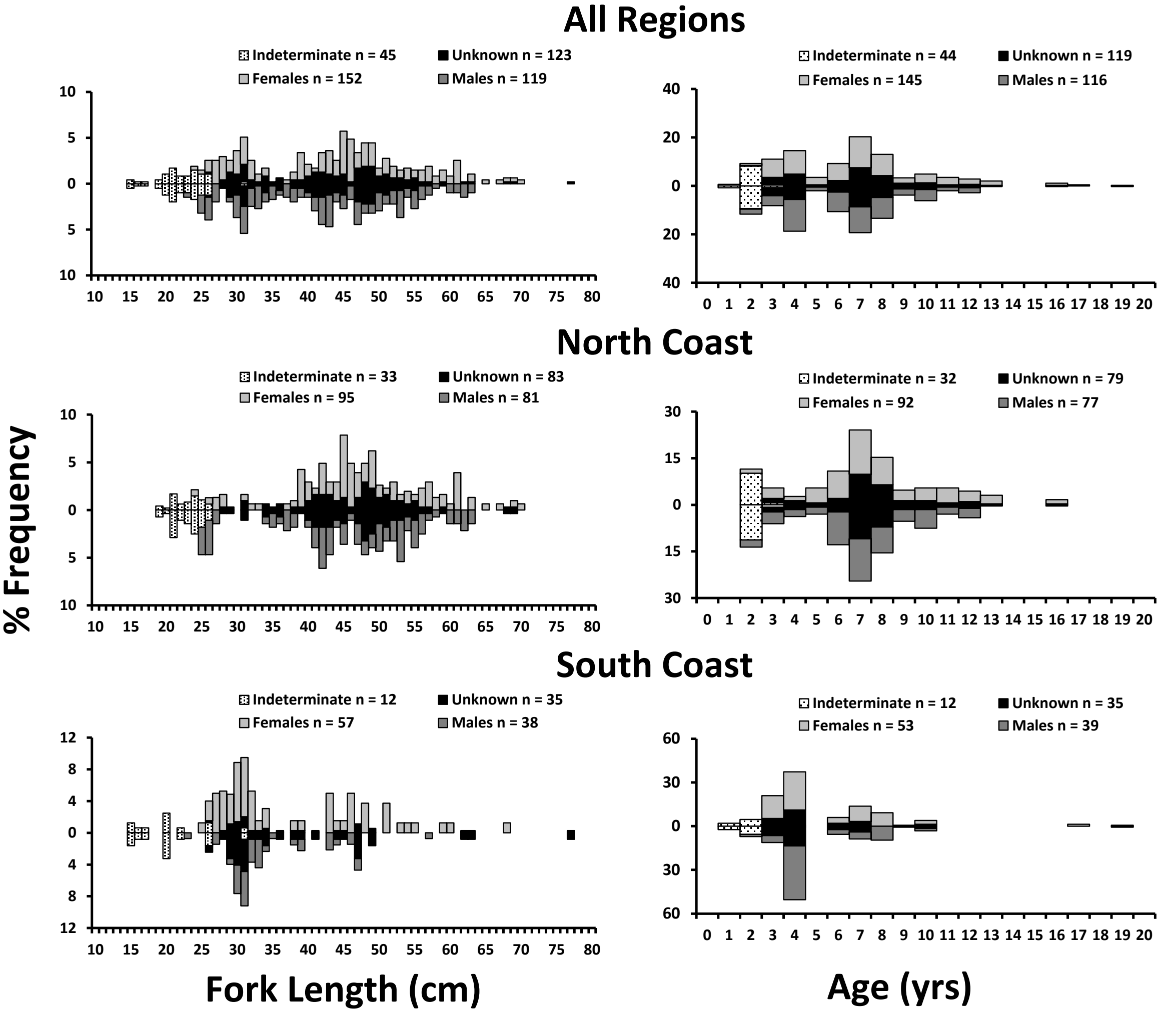

Figure 22. Spatial comparisons of the length and age percent contributions of male, female, indeterminate and unknown sex Snapper (*Chrysophrys auratus*) from the North and South coasts of Tasmania. Note: *y*-axis maximums are not equal values. Sample numbers for indeterminate and unknown sex were shared between the upward and downward directions with a 1:1 ratio across all size and age classes.

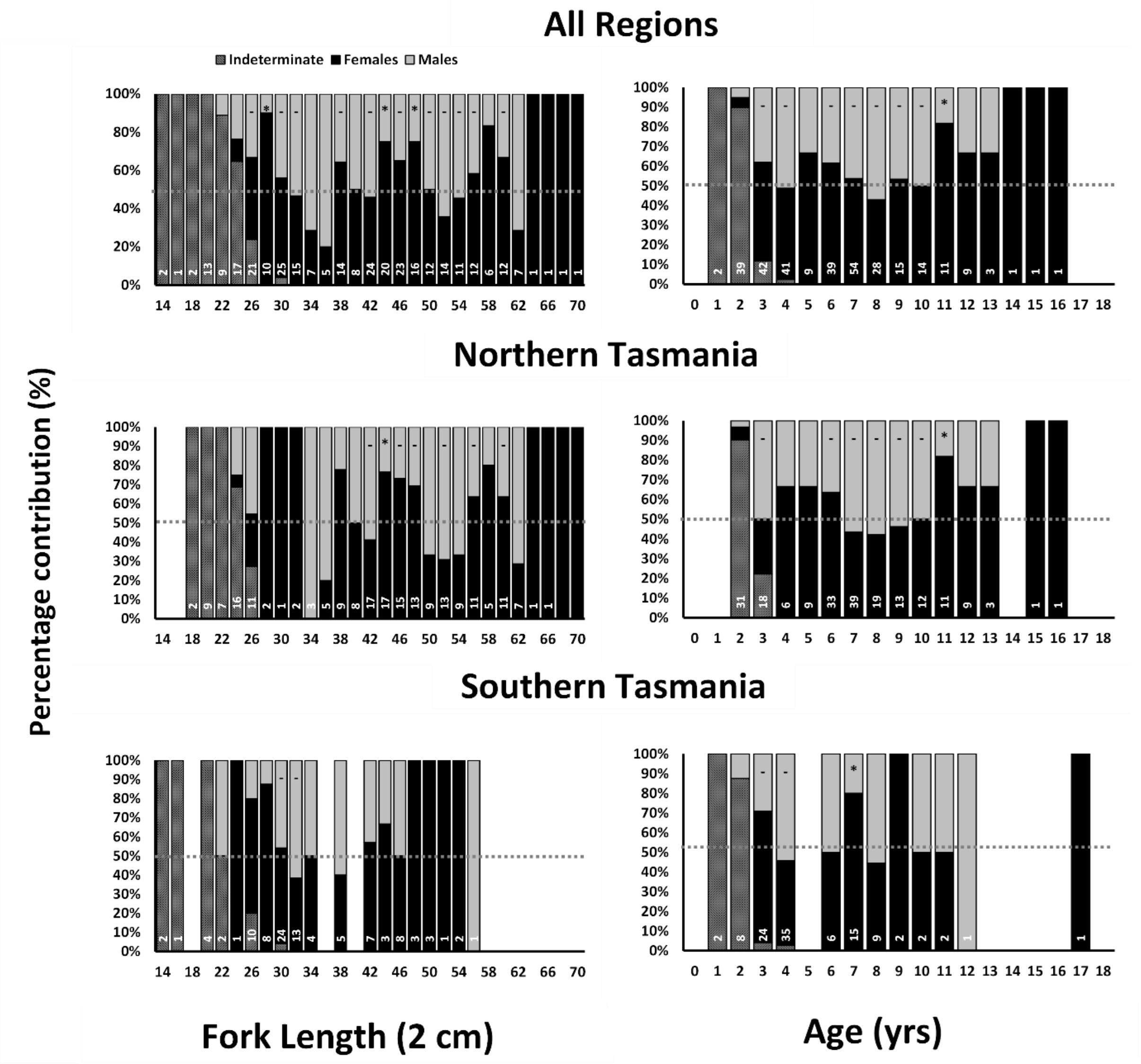

Figure 23. Spatial comparisons of the percentage contribution of indeterminate, male and female Snapper (*Chrysophrys auratus*) from the North and South coasts of Tasmania in relation to fork length (20 mm) and age. Note: * indicate a significant difference ($p < 0.05$) of the male: female ratio away from 1:1, "-" indicates χ^2 test failed to detect a difference, values without annotation were not tested due to low sample size (< 10) where sample size is indicated as numbers in the base of the columns in white.

Size structure and mortality estimation

Due to sample size limitations, total mortality estimates by gender could only be examined with all regions pooled and indicated that gender specific total mortality rates were not significantly different (ARSS, $F = 1.18_{1,11}$, $P = 0.34$; Slopes test, $t = 0.74_{1,11}$, $p = 0.47$). Similarly, mortality rates of pooled sexes between region were also not different ($F = 0.44_{1,12}$, $P = 0.66$; Slopes test, $t = 0.94_{1,12}$, $p = 0.37$). Given the lack of gender and regional differences in total mortality, data was pooled (whole stock) where total mortality rates (age range 7–19 years) were estimated to be 0.37 yr^{-1} (R^2 0.94) for the age-based catch curve method and 0.52 yr^{-1} (R^2 0.95) when directly fitting an exponential decay curve to the pooled age frequency data (Figure 24). Natural mortality estimates were variable depending on the estimation method used. Hoenig$_{nls}$ estimator produced the highest estimate of natural mortality with Tanka's (1960) 5% survival producing the lowest estimate (see Table 7). As, the current observed T_{max} for Tasmania is quite low for the species, natural mortality estimates using a 40 yr T_{max} for the species have also been estimated. These natural mortality estimates indicate that fishing mortality may range from 0.04 to 0.45 yr^{-1} depending on the total mortality and natural mortality estimates used.

Table 7. Total and Natural Mortality estimates for Tasmanian Snapper (*Chrysophrys auratus*).

Estimation technique	**Mortality estimate**
Total Mortality (Z)	
Z age-based catch curve	0.37
Z direct fit of exponential decay curve	0.52
Natural Mortality (M)	
M Hoenig est (observed Tas. T_{max} = 19.19)	0.33
M Hoenig est (observed species T_{max} = 40)	0.17
M Pauly’s method mean size @ age (14.97°C)	0.15
M Pauly’s method decimal age (14.97 °C)	0.12
M 5% survival to T_{max} = 19.19	0.16
M 1% survival to T_{max} = 19.19	0.24
M 5% survival to T_{max} = 40	0.07
M 1% survival to T_{max} = 40	0.12

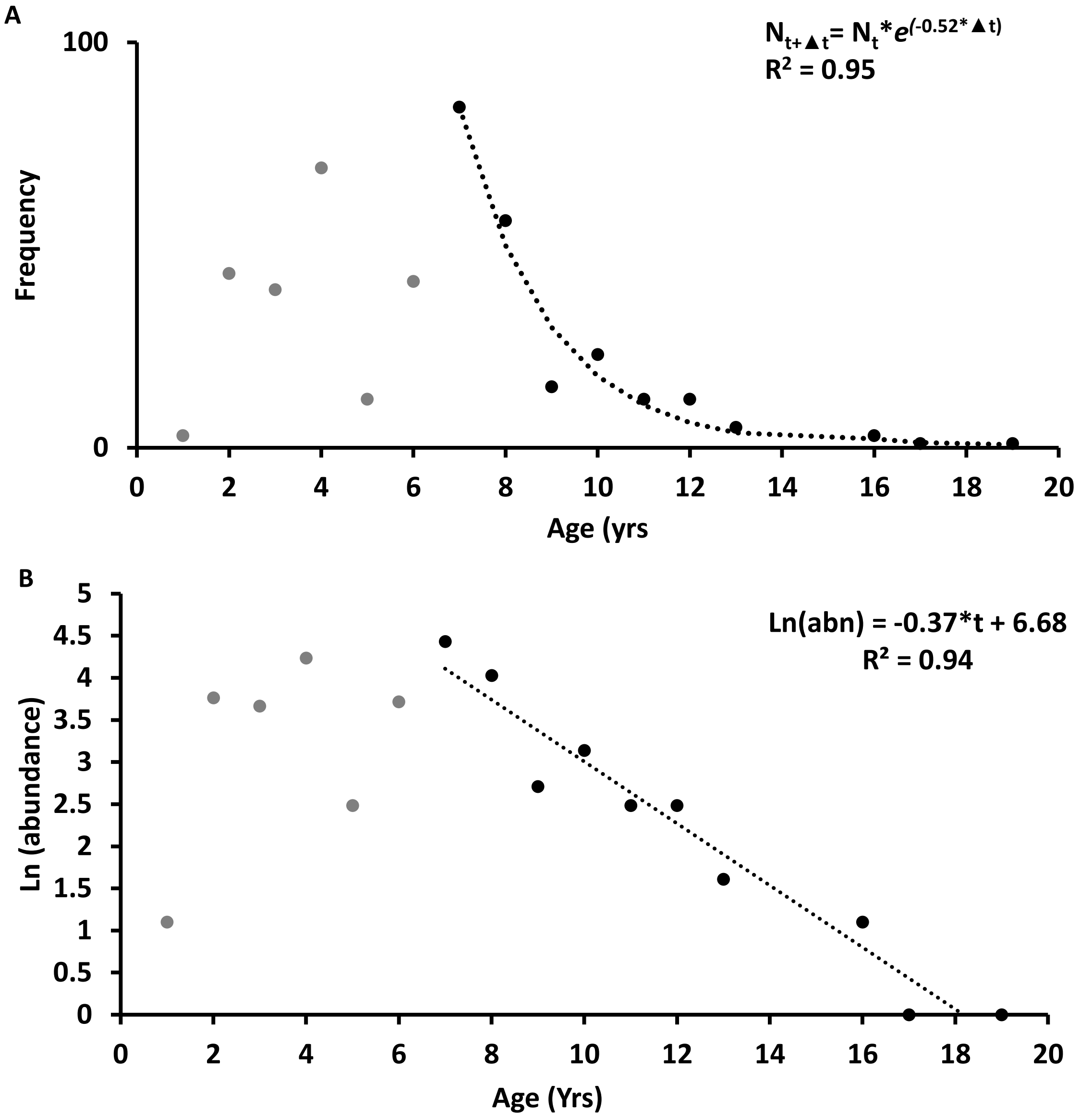

Figure 24. (A) Whole stock age- frequency with a fitted exponential decay curve and (B) age-based catch curve for Snapper (*Chrysophrys auratus*). Black markers represent data selected for model fitting. Grey markers were excluded from model fitting due to selectivity issues.

Growth

Regional and sex-specific von Bertalanffy growth parameters, and statistical tests on model comparisons are presented in Table 8. As differential growth models between sex were not evident, all other growth modelling used data with sex pooled. Regional comparisons of growth models displayed little evidence of variable growth, except when testing unconstrained individual size at decimal age between regions (Figure 25). However, this effect could be attributed to a modelling artefact due to differences in the data ranges between regions. When an ARSS was performed on data constrained to standardized size ranges to compare models, this significant difference was lost. Additional support to suggest growth in Snapper does not display differential growth between sexes or regions in Tasmania, comes from the *t*-test results on mean size at age. Where all *t*-test performed failed to detect a significant difference between the respective age classes being tested. Despite the lack of a regional or sex-specific effect, we have chosen to present all models generated to demonstrate the similarity in growth between sex and region and allow readers’ choice in interpretation between our findings and other studies (Figure 25).

Table 8. Regional and gender specific comparisons of von Bertalanffy growth curve parameters of Snapper (*Chrysophrys auratus*) using mean size at age, and individual size at decimal age data. Asterisks indicate a significant base case ARSS result (in bold) between models. In these cases, an additional ARSS was performed on a standardized data range (2–10 yrs of age) to investigate if the finding was of a statistical or biological basis.

Mean size at integer age VBGF model parameters

		Sex-specific		**Regional specific**	
	ARSS	F=0.06$_{1,19}$, P=0.98		F=1.17$_{1,18}$, P=0.35	
	All Tas	**Females**	**Males**	**North**	**South**
L_∞ (mm)	725	809	826	775	720
k (yrs^{-1})	0.13	0.10	0.10	0.11	0.13
t_0 (yrs)	-0.76	-1.44	-1.35	-1.23	-0.84
n	18	16	16	11	9
R^2	0.99	0.98	0.98	0.99	0.98

Individual size at decimal age VBGF model parameters

		Sex-specific		**Regional specific***	
	ARSS	F=0.23$_{1,254}$, P=0.87		**F=3.88$_{1,369}$, P=0.009**	
	All Tas	**Females**	**Males**	**North**	**South**
L_∞ (mm)	791	795	906	785	773
k (yrs^{-1})	0.11	0.11	0.08	0.10	0.11
t_0 (yrs)	-0.98	-0.78	-1.28	-1.20	-0.71
n	873	466	265	235	206
R^2	0.88	0.87	0.84	0.86	0.91

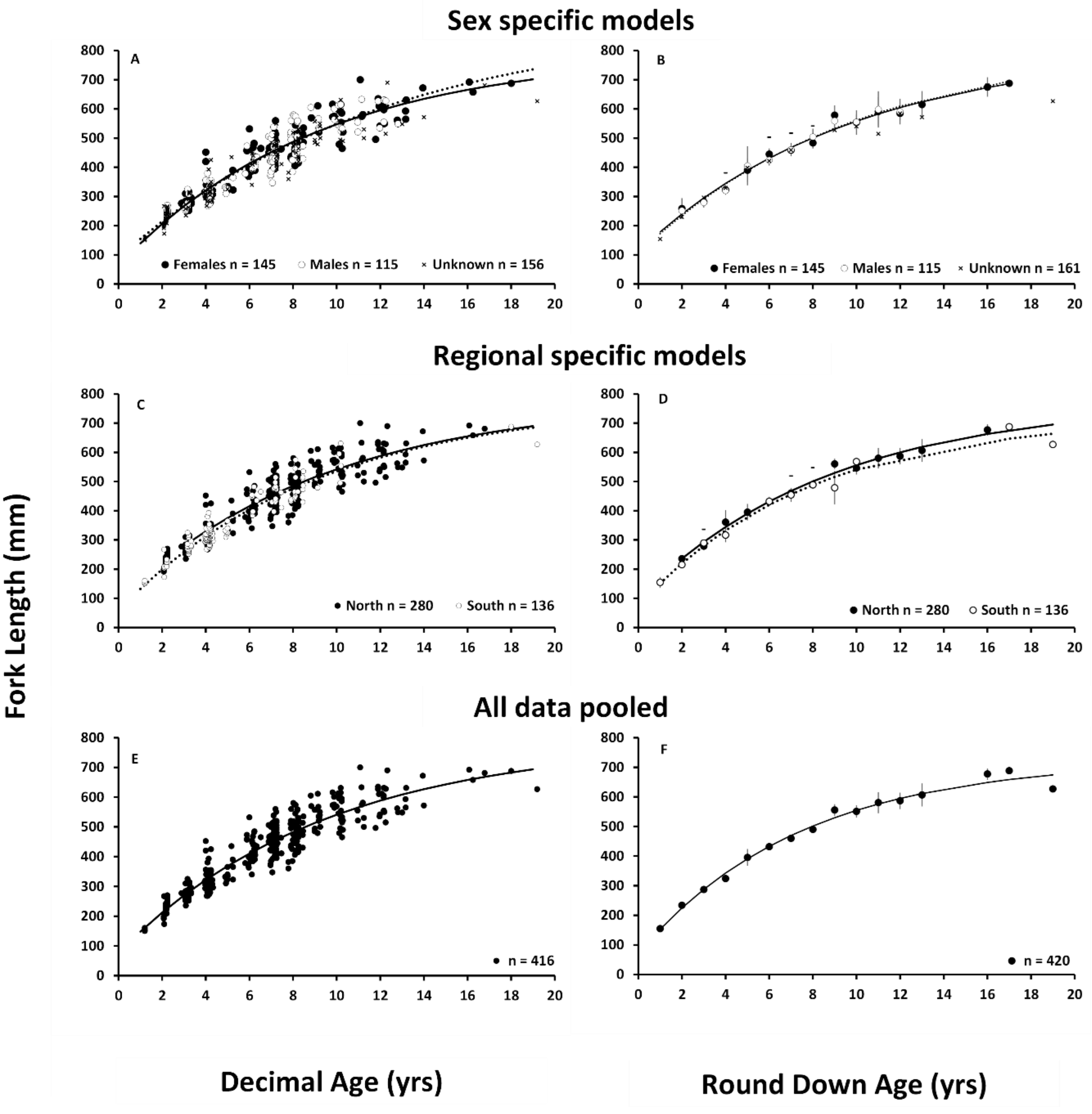

Figure 25. Size at age scatter plots and von Bertalanffy growth models for Snapper (*Chrysophrys auratus*) in relation to sex and region. Sex is pooled in the regional comparisons. Solid black lines and dotted lines represent models for females and males respectively in A and B and for North and South respectively in graphs C and D. Paired means without annotation above were excluded from *t*-tests due to low sample size, "-" denotes, a *t*-test was performed but no difference was detected. Error bars are 95% CI.

Reproductive biology

There was a clear relationship between reproductive stage and mean GSIs in both females and males with GSI values being low in reproductively inactive and spent fish and higher in fish that were reproductively active. The smallest reproductively active female and male measured 315 mm and 310 mm FL respectively (Figure 26A, B). These lengths were subsequently used as the adult boundaries for seasonal reproductive trends. Both individuals were approximately four years of age. Due to low sample sizes reproductive trends need to be viewed with caution. However, despite these low sample sizes, seasonal changes in the proportions of reproductive stages and mean GSI values did display a clear seasonal pattern of reproductive activity (Figure 27). Female GSI profiles rose to a peak in December, with a possible secondary peak in February. Vitellogenic females (n = 48) were present from October to April and females possessing hydrated oocytes were present in November, December, and February. Spent females first appeared in January and were present until June.

Proportions of reproductively inactive females were highest in June. Male GSI profiles peaked in November which was associated with a relatively large proportion of spermiated males. Reproductively active (spermatogenic) males were present from October to April with spermiated males being present from November to January, and one individual in April. Spent males appeared in March and April. In May and June all males were classified as reproductively inactive.

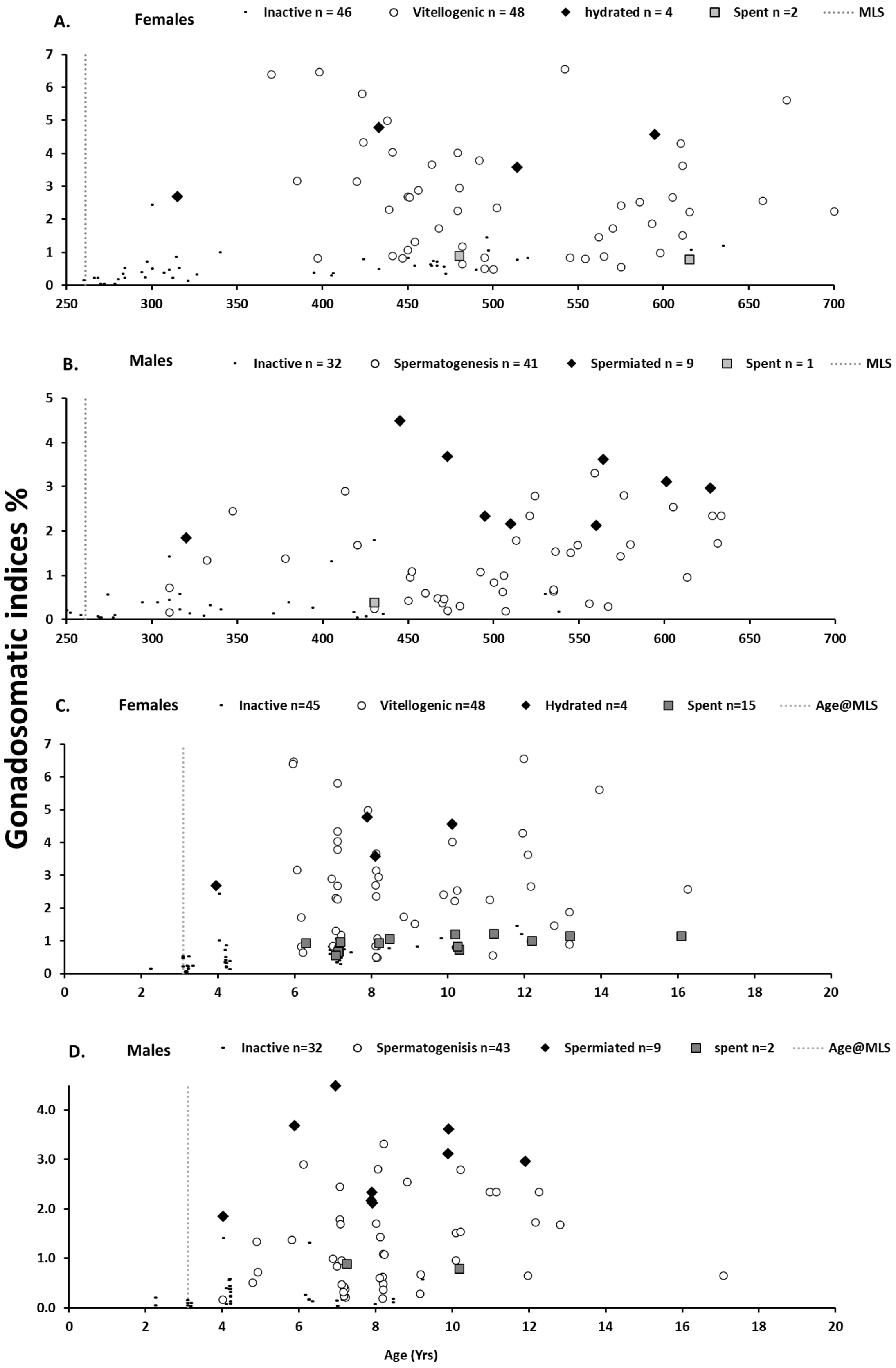

Figure 26. Sex-specific scatter plot relationships of fork length (A & B) and age (C & D) in relation to gonadal-stage and gonadosomatic indices (GSI) for Snapper (*Chrysophrys auratus*) from Tasmania. Broken grey lines represent the modelled fork length and age at the current Tasmanian minimum legal size (MLS) of 30 mm total length.

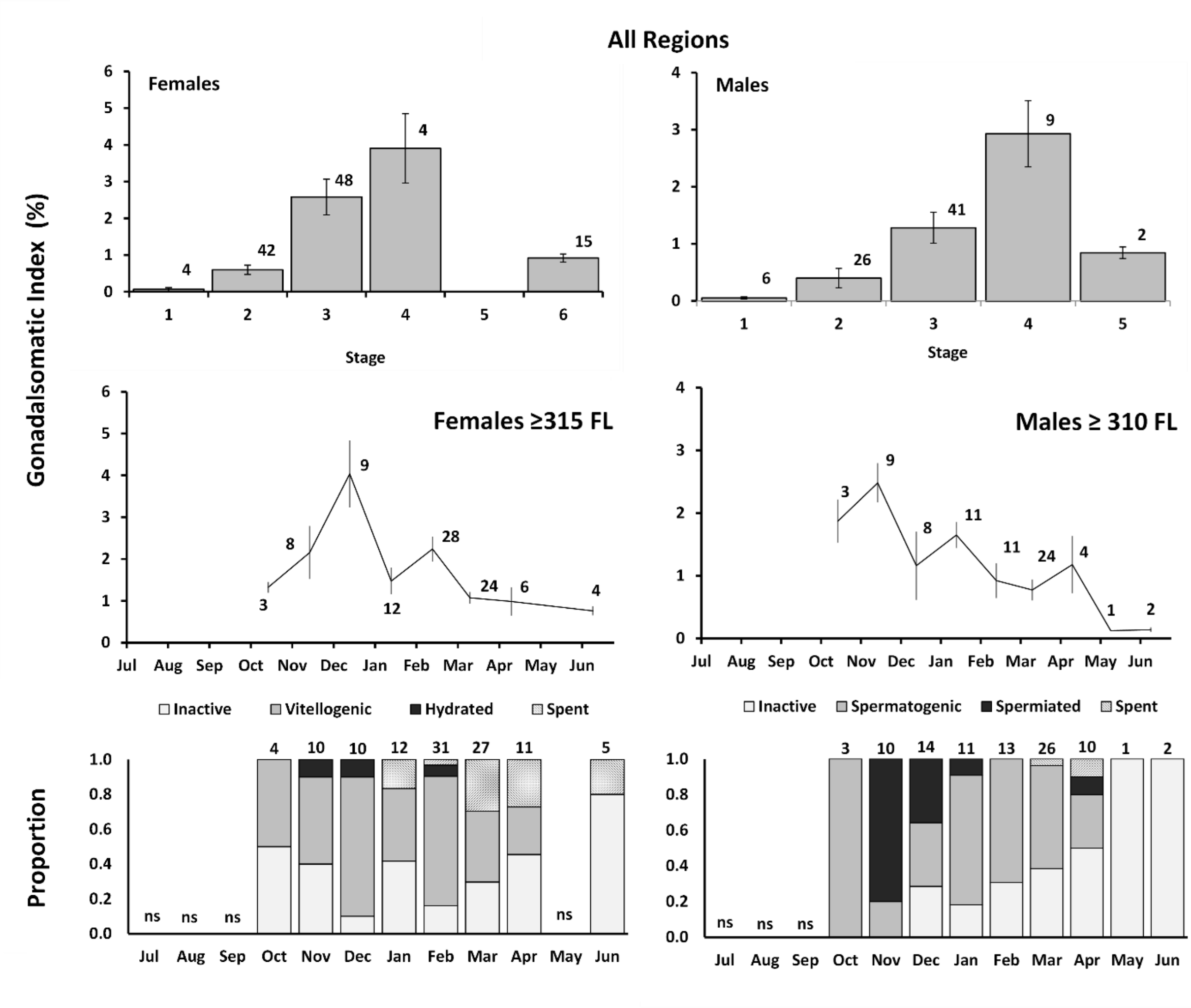

Figure 27. Reproductive characteristics of female and male Snapper (*Chrysophrys auratus*) in relation to reproductive stage, seasonal changes in monthly mean gonadosomatic indices (GSI) (± *SE*) and seasonal proportions of reproductive stages. Data has been pooled across years and regions and constrained to be from the smallest observed mature female (≥ 315 mm FL) and male (≥ 310 mm FL). Sample sizes (*n*) are indicated as values next to error bars or on top of columns. ns = no sample.

Diet

Prey taxa was identified in 77 Snapper stomachs collected from eight sites across Tasmania (Figure 28). A mean (± *SE*) of 2.13 ± 1.45 distinct taxa per stomach were identified (range: 1–8). Class was the lowest taxonomic level with high resolution, with identified prey taxa representing 16 total taxonomic classes (Table 9). The prey accumulation curve analysis estimated an asymptotic class richness of 18.2 ± 1.16 classes, demonstrating the 16 classes identified from Snapper stomachs span most of the estimated gamma diversity of Snapper diet in the region at this taxonomic level (Figure 29).

Table 9. Prey taxa identified in Snapper (*Chrysophrys auratus*) stomach contents by the percent frequency of occurrence (%FO) and mean proportion of stomach contents by weight (%Contents).

Prey Taxon	%FO	%Contents
Bivalvia	49.38%	35.95 ± 5.0%
Pectinida	37.04%	33.04 ± 5.1%
Anomia trigonopsis	1.23%	0.06 ± 0.06%
Mimachlamys asperrima	35.80%	32.97 ± 5.1%
Cardiida	7.41%	1.38 ± 0.74%
Fulvia tenuicostata	4.94%	0.8 ± 0.53%
Theora lubrica	2.47%	0.58 ± 0.52%
Mytiloida	2.47%	0.9 ± 0.72%
Arcuatula senhousia	2.47%	0.8 ± 0.62%
Other Bivalvia	4.94%	0.63 ± 0.41%
Malacostraca	44.44%	20.55 ± 3.83%
Decapoda	37.04%	20.22 ± 3.83%
Jasus sp.	1.23%	0.37 ± 0.37%
Anomura	6.17%	1.66 ± 1.0%
Galatheoidea	2.47%	0.14 ± 0.12%
Galathea australiensis	1.23%	0.12 ± 0.12%
Other Galatheoidea	1.23%	0.02 ± 0.02%
Paguroidea	6.17%	1.52 ± 0.97%
Brachyura	27.16%	16.21 ± 3.65%
Grapsidae-like	4.94%	4.14 ± 2.09%
Metacarcinus novaezelandiae	2.47%	2.46 ± 1.73%
Dromiidae	2.47%	1.95 ± 1.41%
Austrodromidia australis	1.23%	1.21 ± 1.21%
Other Dromiid	1.23%	0.74 ± 0.74%
Leucosidae	2.47%	1.39 ± 0.98%
Bellidilia spp.	1.23%	0.77 ± 0.77%
Other Leucosidae	1.23%	0.62 ± 0.62%
Litocheira bispinosa	1.23%	0.25 ± 0.25%
Naxia aurita	1.23%	1.23 ± 1.23%
Notomithrax spp.	1.23%	0.14 ± 0.14%
Caridea	9.88%	1.11 ± 0.6%
Alpheus novazealandae	1.23%	0.12 ± 0.12%
Alpheus richardsoni	2.47%	0.07 ± 0.06%
Alpheus spp.	1.23%	0.19 ± 0.19%
Philocheras spp.	1.23%	0.01 ± 0.01%
Penaeoidea	1.23%	0.86 ± 0.86%
Amphipoda	9.88%	0.23 ± 0.1%
Isopoda	3.70%	0.08 ± 0.05%
Stomatopoda	1.23%	0.01 ± 0.01%
Polychaeta	16.05%	9.29 ± 2.71%
Terebellidae	8.64%	5.2 ± 2.07%
Aphroditdae	1.23%	0.25 ± 0.25%
Sabellariidae	1.23%	1.23 ± 1.23%
Other Polychaeta	7.41%	2.61 ± 1.08%
Gastropoda	14.81%	5.23 ± 2.01%
Philine angasi	4.94%	1.35 ± 0.76%
Sinum zonale	2.47%	0.65 ± 0.46%
Littorinimorpha	2.47%	0.03 ± 0.03%
Volutidae	1.23%	0.04 ± 0.04%
Nudibranchia	1.23%	0.68 ± 0.68%
Fissurelloidea	2.47%	2.47 ± 1.73%
Actinopterygii	13.58%	11.34 ± 3.33%
Neosebastes	1.23%	0.54 ± 0.54%
Stigmatopora nigra	1.23%	1.21 ± 1.21%
Contusus brevicaudus	1.23%	1.23 ± 1.23%
Tetractenus glaber	1.23%	1.23 ± 1.23%
Other Actinopterygii	8.64%	7.12 ± 2.68%
Crinoidea	7.41%	4.86 ± 2.22%
Comatulida	7.41%	4.86 ± 2.22%
Ophiuroidea	7.41%	2.41 ± 1.03%
Amphilepidida	3.70%	1.49 ± 0.85%
Amphiura elandiformis	2.47%	0.99 ± 0.71%
Other Amphiuridae	1.23%	0.49 ± 0.49%
Ophiurida	3.70%	0.92 ± 0.61%
Cephalopoda	6.17%	3.74 ± 1.89%
Octopoda	3.70%	3.11 ± 1.8%
Sepiida	1.23%	0.62 ± 0.62%
Other Cephalopod	1.23%	0.01 ± 0.01%
Holothuroidea	6.17%	3.16 ± 1.57%
Cucumariidae	1.23%	0.25 ± 0.25%
Other Holothuroidea	4.94%	2.91 ± 1.55%
Anthozoa	1.23%	0.03 ± 0.03%
Ascidiacea	1.23%	0.46 ± 0.46%
Asteroidea	1.23%	1.23 ± 1.23%
Asterias amurensis	1.23%	1.23 ± 1.23%
Maxillopoda	1.23%	0.09 ± 0.09%
Balanomorpha	1.23%	0.09 ± 0.09%
Sipuncula	1.23%	1.23 ± 1.23%
Seagrass	8.64%	0.32 ± 0.15%
Macroalgae	1.23%	0.1 ± 0.1%

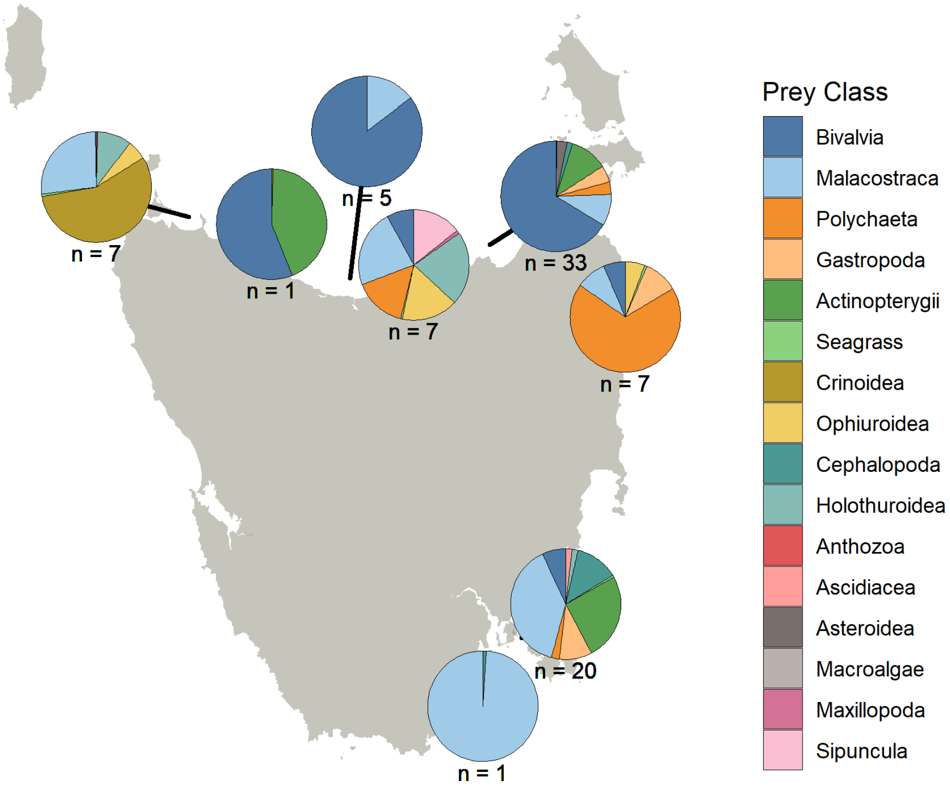

Figure 28. Mean proportion of Snapper (*Chrysophrys auratus*) stomach contents by prey taxa class and samples sizes (*n*) of stomachs that contained prey, by region of collection. *Clockwise, from top left*: Smithton, Wynyard, Devonport, Tamar, Bridport, St. Helens, together comprising the 'North' stratum for analysis; and South-East, South, combined into the 'South' stratum.

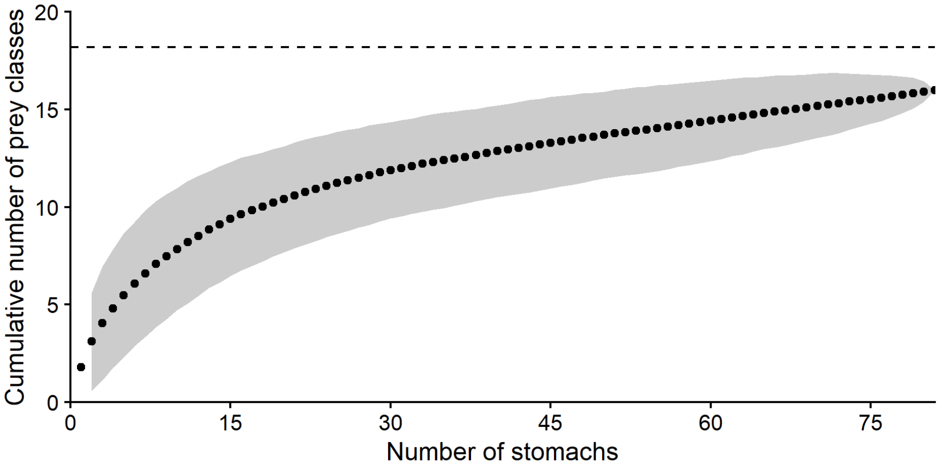

Figure 29. Snapper (*Chrysophrys auratus*) prey taxa accumulation curve estimated from prey taxon presence in 1000 random permutations of stomach samples. Dotted line indicates bootstrap estimated asymptotic number of prey classes of snapper in the sampled region.

Bivalves, malacostracans (non-barnacle crustaceans), polychaetes, gastropods and fish were the five most dominant prey classes by both frequency of occurrence and as mean proportion of contents (Figure 30), together representing a mean of 82.36% of stomach contents by weight. Bivalves were predominantly comprised of doughboy scallops (*Mimachlamys asperrima*), which occurred in 49.38% of Snapper stomachs and comprised most stomach contents at the three northern sites (Wynyard, Devonport and Bridport, Figure 28). Major malacostracans taxa included brachyurans (crabs, 27.2% of stomachs), amphipods (9.88%), caridids (pistol and crangonid shrimp, 9.88%), and anomurans (hermit crabs and squat lobsters, 6.17%). Worth noting was the occurrence of a juvenile spiny lobster (*Jasus* sp.) in a single stomach from the south-east site. Several non-native species occurred in Snapper stomachs, including Asian date mussel (*Arcuatula senhousia*), pie crust crab (*Metacarcinus novaezelandiae*), and an entire 140 g North Pacific sea star (*Asterias amurensis*).

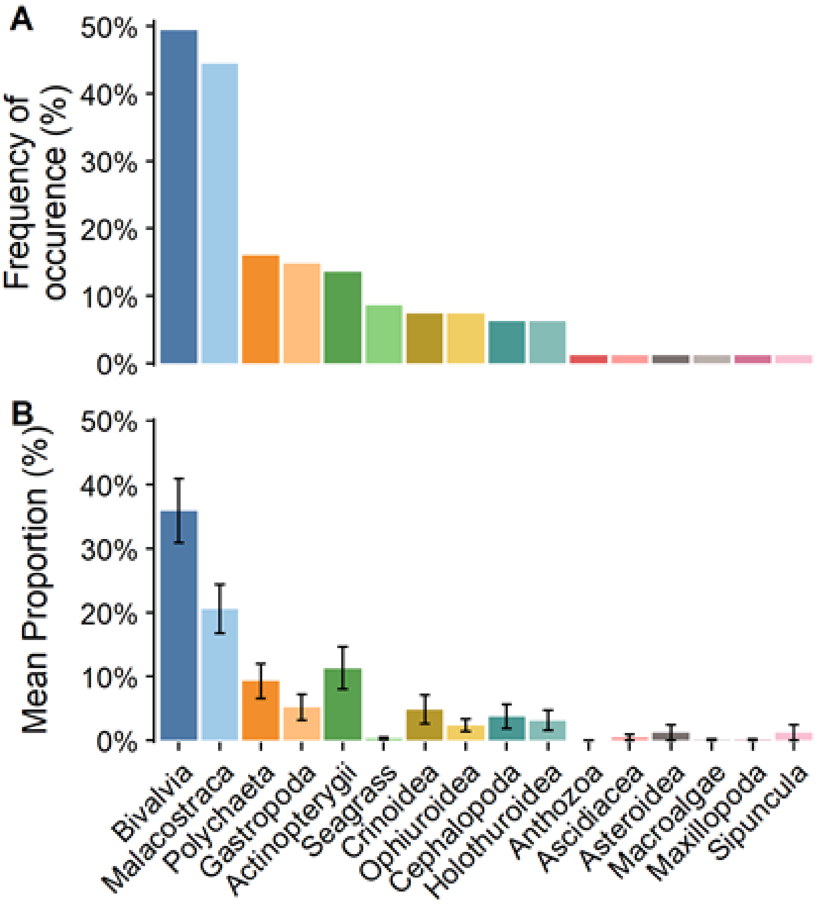

Figure 30. Snapper (*Chrysophrys auratus*) prey taxa (class) by mean proportion (%) of Snapper stomach contents (A) and frequency of occurrence (%) across stomachs (B). Error bars are one *SE*.

Occurrence of polychaetes, seagrass, and ophiuroids (brittle stars) in Snapper stomachs were significantly positively correlated (r_{phi} = 0.262–0.416), as was occurrence of ophiuroids and holothuroids (sea cucumbers, r_{phi} = 0.319; Figure 31). Occurrence of bivalves was significantly negatively correlated with occurrence of polychaetes, ophiuroids, cephalopods, and holothuroids in Snapper stomachs (r_{phi} = 0.230–0.279). Permutational ANOVA revealed both fork length of Snapper and region of capture (North/South) had significant effects ($p < 0.001$) on prey occurrence in stomachs, explaining R^2 = 0.117 and 0.22 of the dissimilarity of prey occurrence across stomachs, respectively (Table 10). The interaction term between region and fork length was not significant (p = 0.095) and was dropped from the model.

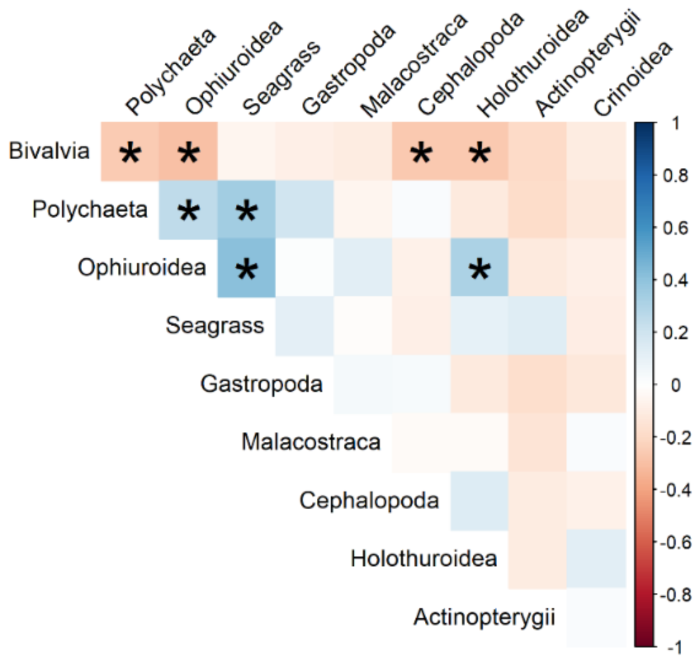

Figure 31. Correlation of occurrence of prey classes among Snapper (*Chrysophrys auratus*) stomachs. Asterisks indicate significant correlations between prey classes. Note that only prey classes that appeared in more than one stomach are represented.

Table 10. Permutational ANOVA model summary for Bray-Curtis pairwise dissimilarity between prey class occurrence in Snapper (*Chrysophrys auratus*) stomachs. FL = Fork length, Region = North or South TAS.

Model: Region + FL + Region:FL	***df***	***SS***	***R^2***	***F***	**Pr(> *F*)**
Region	7	7.359	0.311	4.792	0.001
FL	1	0.622	0.026	2.835	0.029
Region:FL	5	1.626	0.069	1.483	0.097
Residual	64	14.040	0.594	-	-
Total	77	23.647	1.000	-	-
Model: Region + FL	***df***	***SS***	***R^2***	***F***	**Pr(> *F*)**
FL	1	2.769	0.117	12.197	0.001
Region	7	5.212	0.220	3.279	0.001
Residual	69	15.667	0.662	-	-
Total	77	23.647	1.000	-	-

Nonmetric multidimensional scaling suggests that despite considerable similarity in prey occurrence between regions, the increasing occurrence of bivalves (largely doughboy scallops *Mimachlamys asperrima*) among Snapper stomachs collected from North TAS drove prey occurrence dissimilarity (Figure 32A, B). Differences in the presence of prey items among region were independent of Snapper size (e.g., brittle stars and crinoids only appearing in North TAS stomachs, cephalopods predominantly occurring in South TAS stomachs, Figure 32C). Prey class varied among region by proportion of stomach contents as well. While bivalves were proportionally the most important prey class among samples in the north, followed by malacostracans and polychaetes, in the south malacostracans were the most important prey class, followed by fish and cephalopods (mostly small octopus; Figure 32).

While only stomach contents were systematically analysed, the contents of hindguts from several frames were opportunistically examined and contained several species that didn't occur in stomach samples, including the Heart Urchin *Echinocardium cordatum*, the Venerid Clam *Tawera gallinula*, and the crustaceans *Ovalipes australiensis* and a sphaeromatid isopod of 15 mm width.

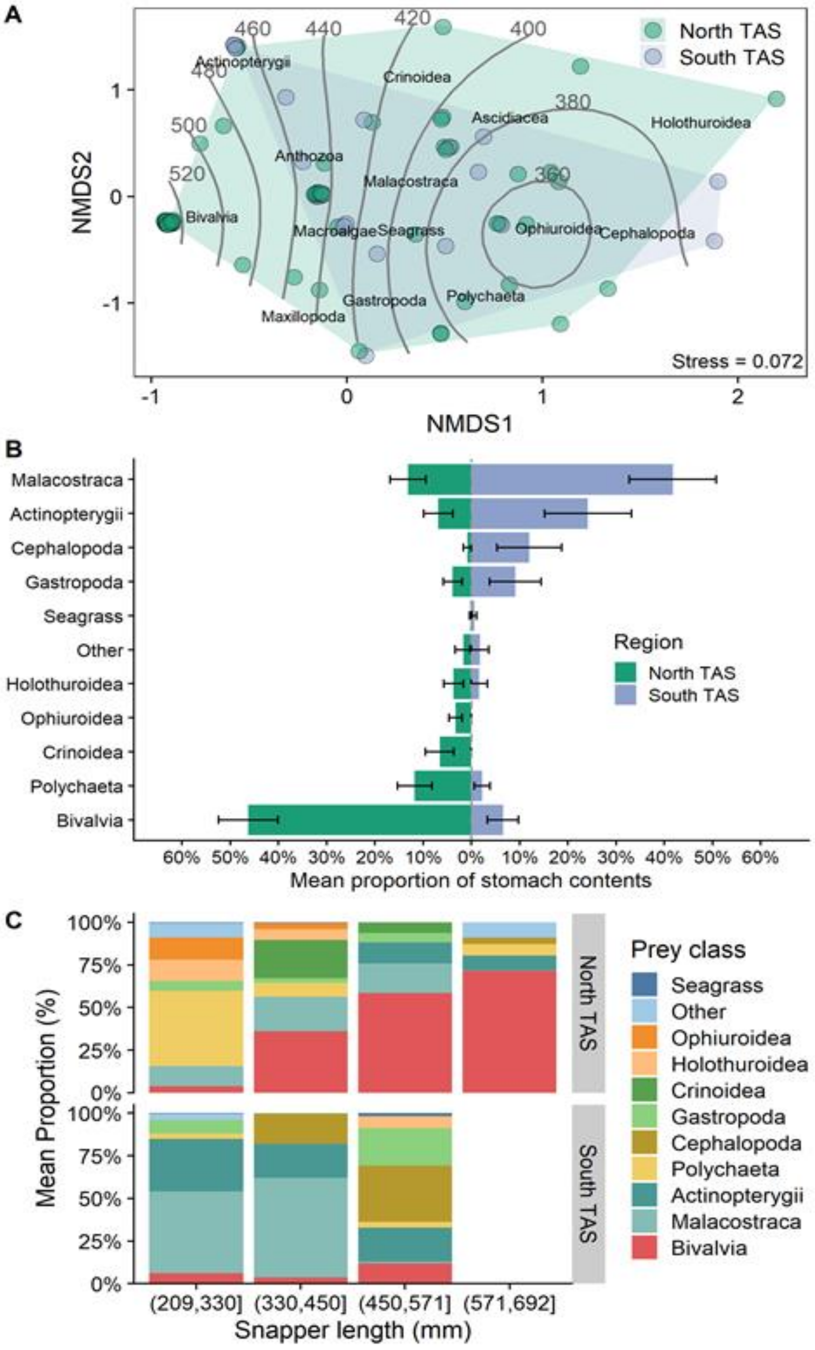

Figure 32. (A) Nonmetric multidimensional scaling scores of Snapper (*Chrysophrys auratus*) stomach (circles) prey class presence dissimilarity, by region of collection, and scores of prey classes in ordinal space. Contour lines represent the smoothed surface of fork length regressed on the ordination axes ($p < 0.001$). (B) Mean proportion of Snapper stomach contents, by region of collection. bars represent one *SE*. C. Mean proportion of Snapper stomach contents, by binned fork length of individual Snapper and region of collection.

Species Distribution Model

Environmental habitat suitability model

The optimal model for Snapper environmental habitat included sea surface temperature (SST: °C) and depth (m) (Table 11).

$$\text{Response} \sim s(\text{SST}, k=4) + s(\text{depth}) + (1|\text{year})$$

Where response is the relative probability of Snapper occurrence as a function of SST and depth, 's' denotes a smoothing term.

The effect of SST on Snapper occurrence was non-linear with a peak effect at approximately 20°C (Figure 33a, Table 11). Depth was also significantly non-linear but displayed a general positive effect at shallower depths (i.e. < -50m; Figure 33b).

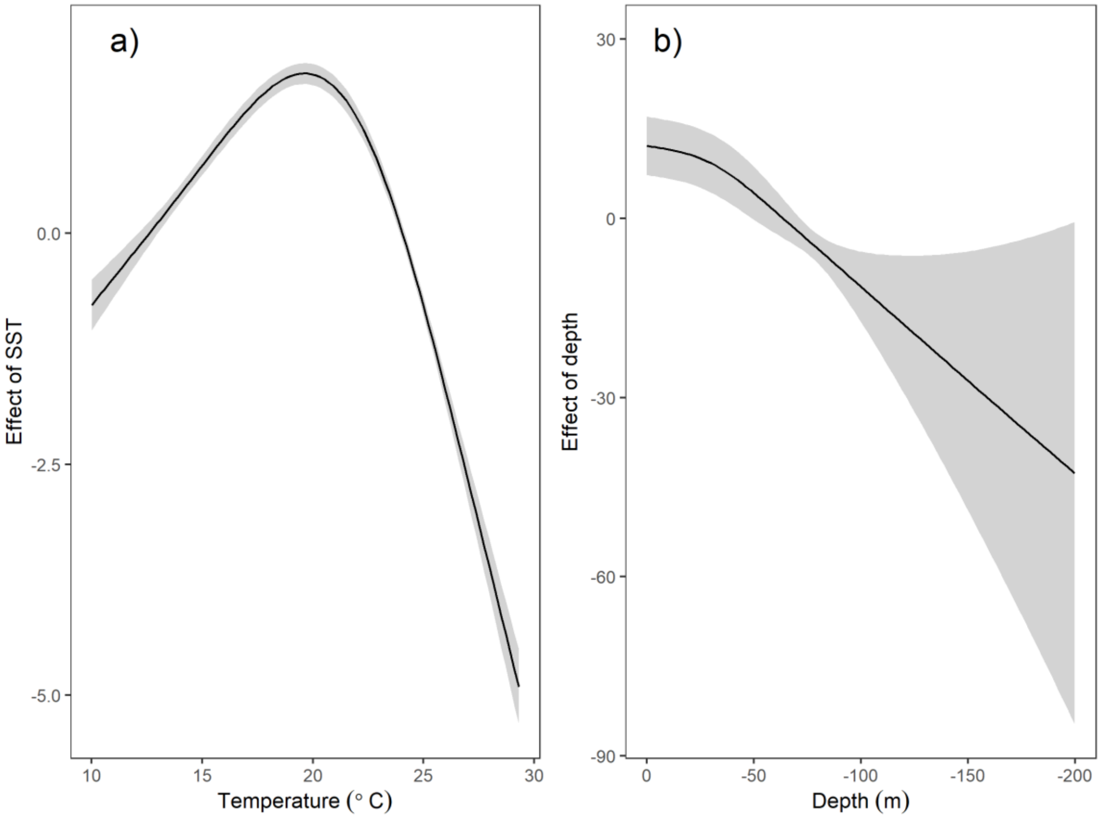

Figure 33. Partial effects of a) sea surface temperature (SST), and b) depth (m) on the fitted values of the optimal habitat suitability model for Snapper (*Chrysophrys auratus*) ± 95% confidence intervals (shaded in grey).

Table 11. Summary of results for the optimal model for suitable habitat of Snapper (*Chrysophrys auratus*). Smoothing terms are denoted by an 's'.

Factor	**Effective degrees of freedom (edf)**	**Coefficient estimate**	***p*-value**
s(SST)	2.989	-1.8118	< 0.001*
s(depth)	2.815	8.0932	< 0.001*
$\text{Year}_{\text{intercept}}$	-	-14.45	< 0.001*

Environmental Habitat Suitability

Hindcast (1998–2018)

There was relatively little change in habitat suitability of Snapper throughout the hindcast period (when pooled across whole domain and seasons: z value = 0.524, $p = 0.60$), with the exception of a small increase in habitat suitability in the NE, SE and W Regions in the summer season (Figure 34, Table F6). The greatest increase in habitat suitability values occurred in the summer in the SE at a rate of 0.02 per year. When holding year at the median, summer was consistently more suitable than the other seasons (Figure 34, Table F7), and predicted mean habitat suitability was consistently greater in the summer and autumn versus the spring and winter across all regions, ranging from 1.83–1.99 times greater in the autumn and summer in the W region to 2.64–3.67 times greater in the autumn and summer in the SE region compared to the winter (Figure 34, Table F7).

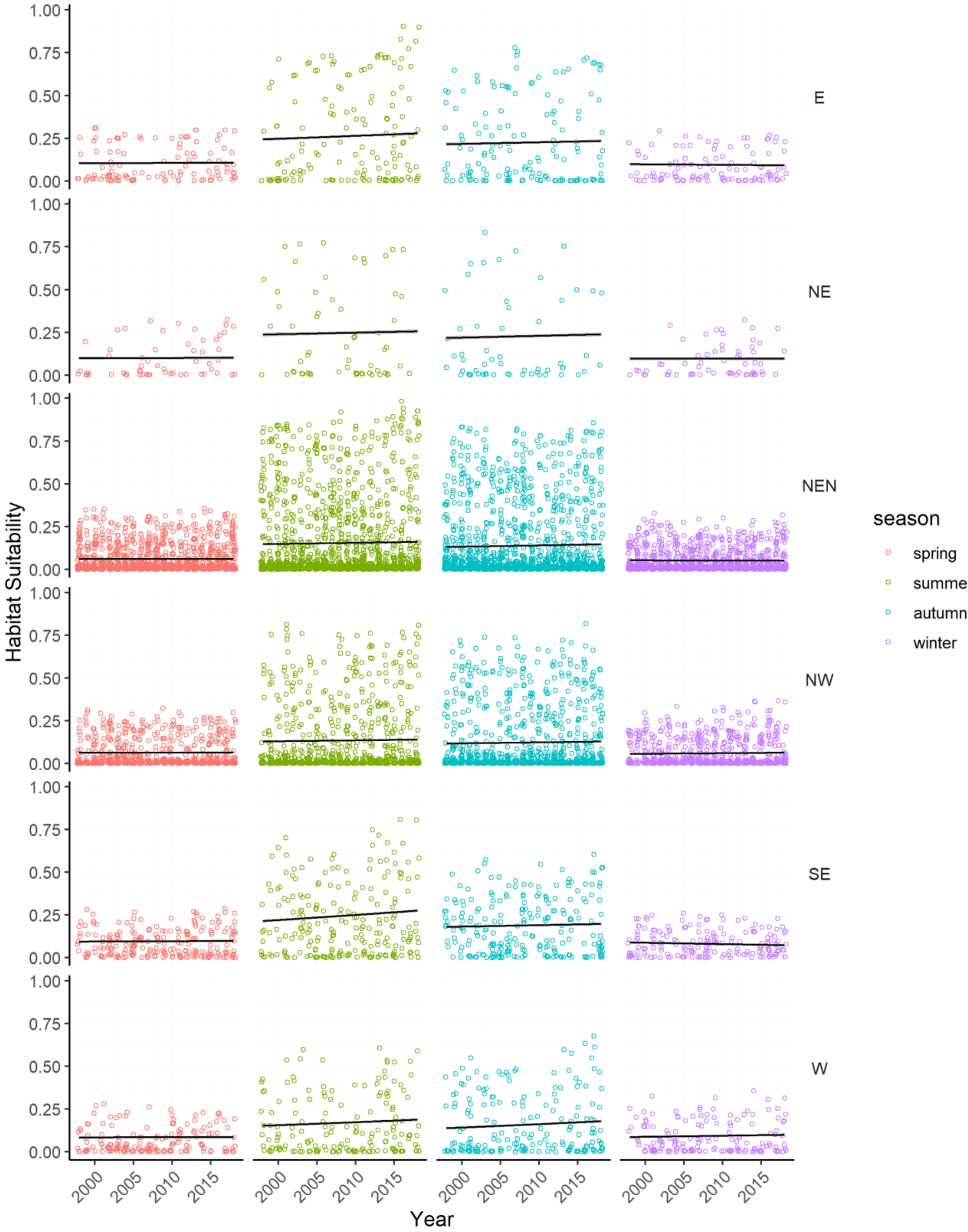

Figure 34. Predicted habitat suitability of seasonally aggregated environmental data per year from 1998–2018 of Snapper (*Chrysophrys auratus*) across four seasons and within six regions around Tasmania. Black lines denote the result of generalised linear additive models (suitability ~ year * season * region) ± 95% confidence intervals (shaded in grey). NB: Points are a subset of 10,000 out of 4,234,136 grid cells for legibility.

Proportional change at a resolution of 0.004°

Forecast (1998–2018 vs 2036–2065)

Proportional change in habitat suitability of Snapper within each grid cell (0.004° or 416 m^2), throughout each region was greatest in the winter, with an average percent increase (estimated marginal model mean (± *SE*)) ranging from 126.33 (± 0.29) % in the NE region to 249.31 (± 0.06) % in the NEN region (Figure 35, Table F8). The greatest proportional increase in habitat suitability was 296.58% in the NEN region in the winter (Figure 35, Figure 36). There was also a small increase in suitability in the spring and summer seasons across all regions ranging from an increase of 35 (± 0.08)–60.89 (± 0.14) % in the spring (NW and SE regions respectively) and 21.01 (± 0.06)–40.27 (± 0.13) % in the summer (NEN and SE regions respectively: Figures 35–37). Proportional change remained low and consistent in the autumn across all regions (Figures 35–37, Figures F4–F7).

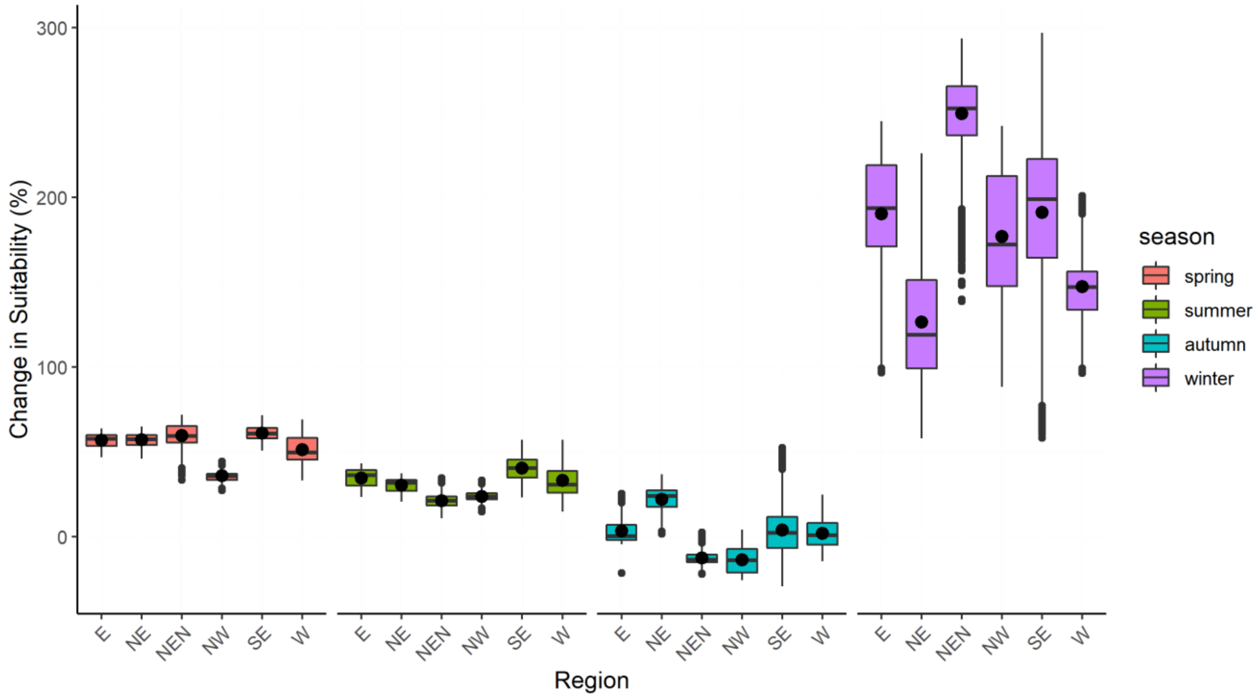

Figure 35. Proportional change (%) in predicted habitat suitability of 416 m^2 areas within six regions of Tasmania, comparing habitat suitability predictions for Snapper (*Chrysophrys auratus*) created using seasonally aggregated environmental data averaged across 20-year historical (1998–2018) and future (2036–2065) periods within six regions of Tasmania. Boxplots show the median and inner quantiles, points are means ± *SE*.

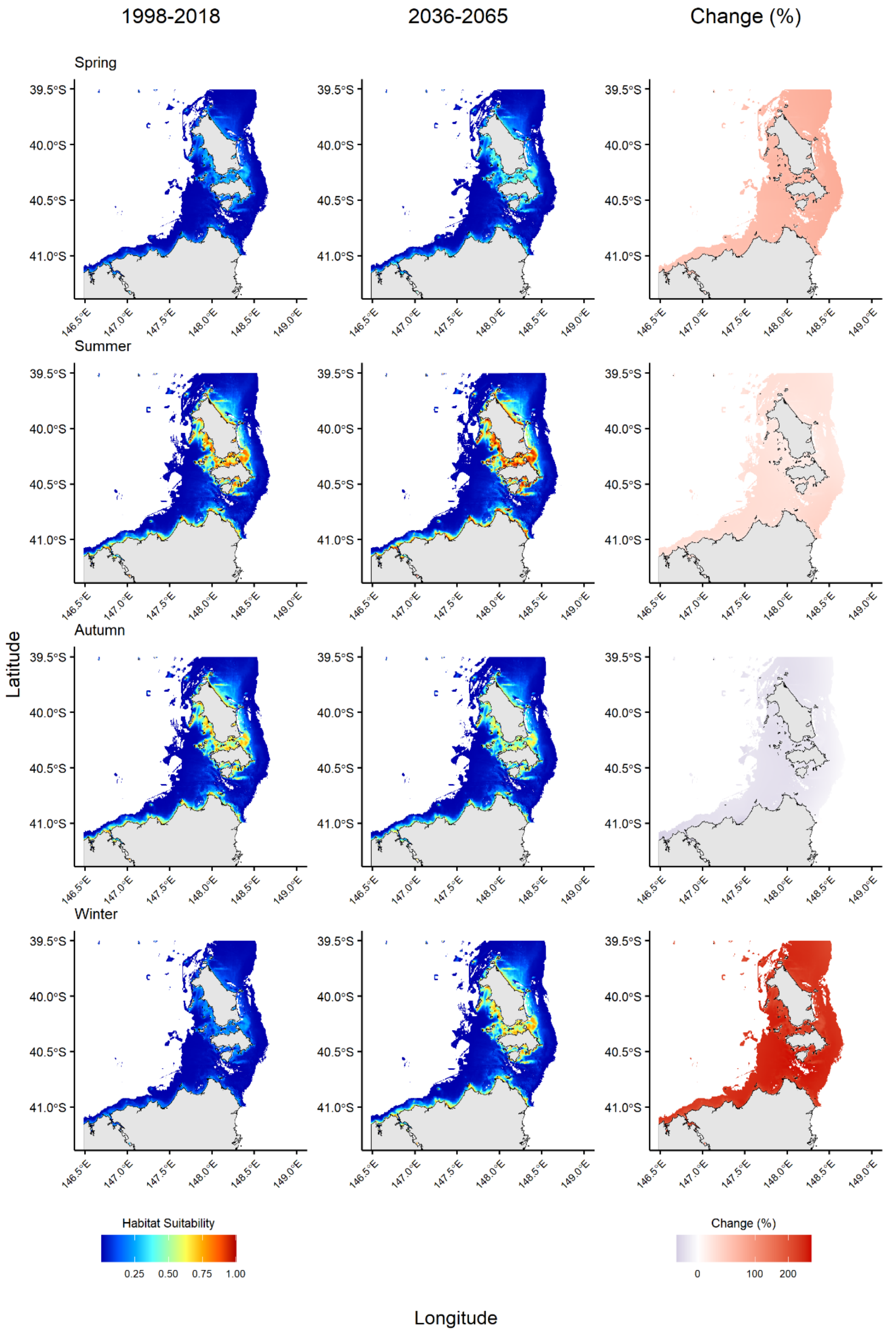

Figure 36. Seasonally aggregated spatial predictions of Snapper (*Chrysophrys auratus*) oceanographic habitat suitability for the North-East-North (NEN) Coast of Tasmania, and the percent change in suitability between 20-year averaged hindcasted (1998–2018) and forecasted (2036–2065) predictions.

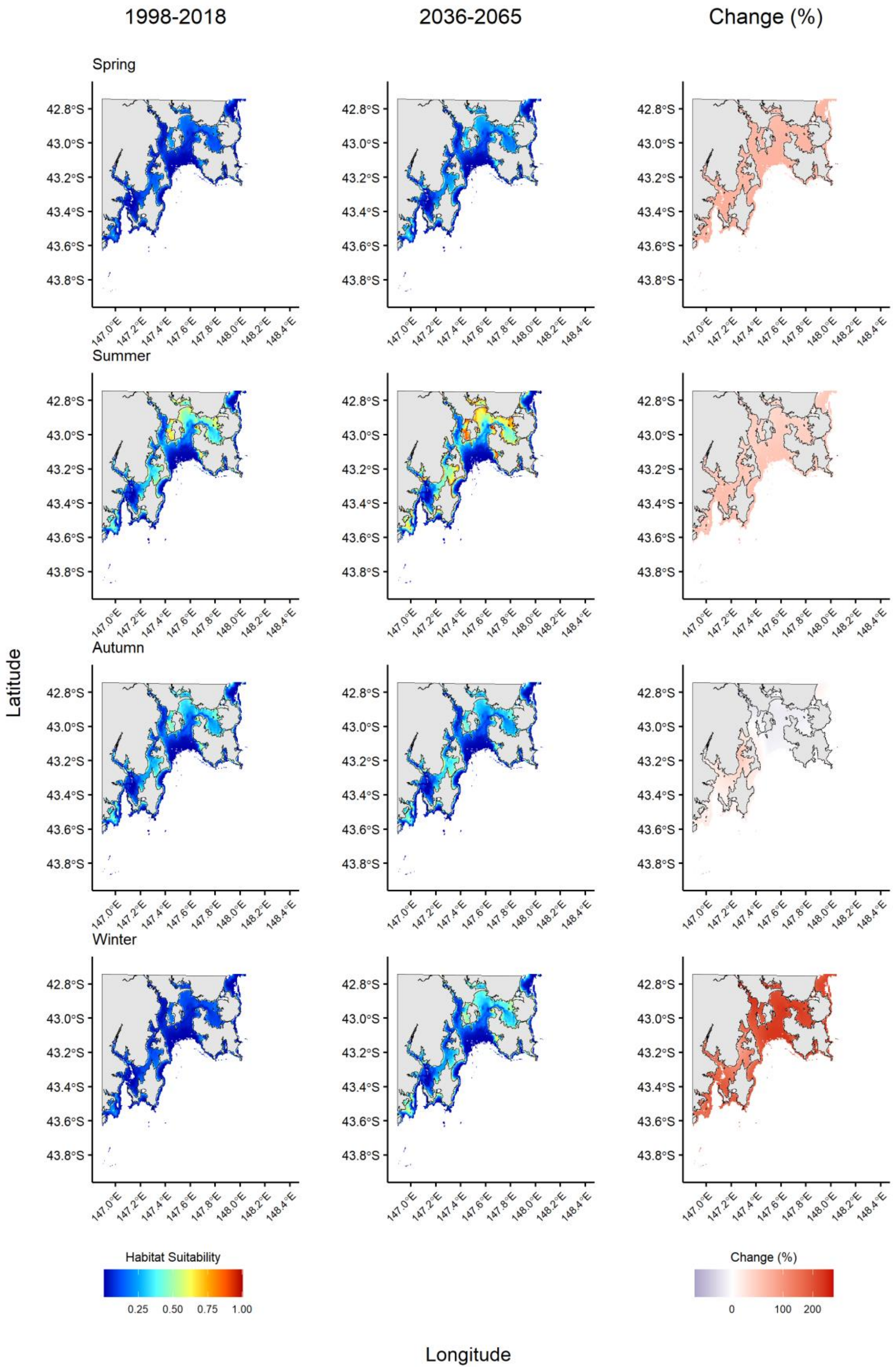

Figure 37. Seasonally aggregated spatial predictions of Snapper (*Chrysophrys auratus*) oceanographic habitat suitability for the South-East (SE) Coast of Tasmania, and the percent change in suitability between 20-year averaged hindcasted (1998–2018) and forecasted (2036–2065) predictions.

Fisheries Data

Total reported commercial catch of Snapper has averaged approximately 300 kg per financial year in Tasmanian commercial managed fisheries (i.e. NRE Tas) since 1995. Gear types with > 1000 kg of Snapper catch reported since 1995 include bottom longline, shark longline, handline, and shark mesh (Figure 38b). Tasmanian commercial catch peaked from the late 1990s–early 2000s (Figure 38a, b), largely due to bottom longline and shark longline gear catch in the Bass Strait. Since 2004–2005, annual catch by these gears has remained under 200 kg annually, and annual catch has remained under 500 kg per financial year. Recreational catch in 2017–2018 (~2650 kg) exceeded the annual commercial catch across all years (Figure 38b). Weighted linear models of catch versus latitude through time indicate an increase in latitude of 0.0280° yr^{-1} ($p < 0.0001$, $R^2 = 0.09$), equivalent to a shift of 31.1 km south per decade (Figure 38c).

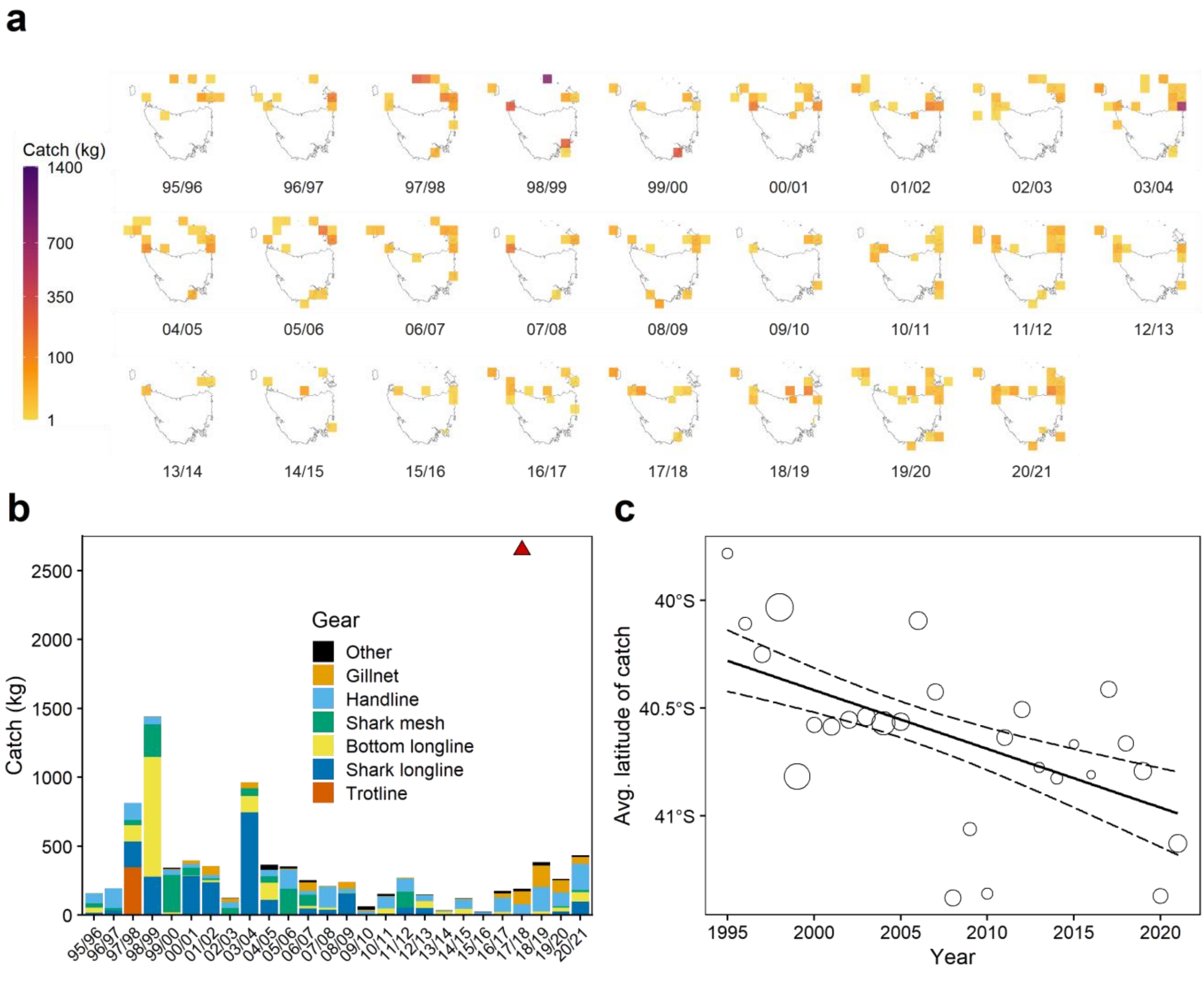

Figure 38. a) Commercial catch (kg) of Snapper (*Chrysophrys auratus*) from Jul 1995–Apr 2020 in the region 39°S–44°S, 143°E–150°E, Catch by reporting block in each financial year. b) Total catch per year by gear type. The triangle represents the total recreational retained catch in 2017–2018. c) Mean latitude of commercial Snapper catch (reported by block) per year. The trendlines were estimated with a weighted linear model of the latitude of catch reports over time (weighted by reported catch weight).

Discussion

Biological Summary

Snapper in our study reached a maximum size of 799 mm FL, and ranged in fork length from 241–700 mm FL for females, 233–633 mm FL for males, and 150–316 mm FL for juveniles of indeterminate sex, and 268–779 mm FL for individuals lacking gonadal material. While we did not determine a difference in the size composition between males and females, we did find regional differences in size, with larger individuals in the North versus the South. The maximum age of our sample was 19.2 years (indeterminate sex) and was 18 and 12.8 years for females and males respectively. Models of growth show a relatively smaller asymptotic length (L_∞) for Tasmanian Snapper versus those on the mainland (Table G2), where very few of our sample reached L_∞. A high proportion of hydrated oocytes in females and spermiated males in the austral summer confirms a reproductively mature population of Snapper in Tasmania. The timing of reproduction is unique, as many high latitude populations spawn mid-late austral spring, our results suggest that Snapper in Tasmania may require the higher temperatures of early austral summer for the onset of spawning. Patchiness within the age-size class structure of our sample suggest potential variability in interannual recruitment as seen in SA and Victoria which has implications for age truncation and subsequent management.

Mortality

Our models show variable total, natural and fishing mortality depending on which model was used. Total mortality of Tasmanian Snapper was estimated using both an age-based catch curve, and an exponential decay curve which resulted in 0.37 and 0.53 for each method respectively, for Snapper aged between 7–19 years. Data was pooled across both sex and region as they were statistically similar. Total mortality estimates generated from age-based catch curves assumes consistent recruitment over time and that all age groups are equally susceptible to the fishing gear (King 2013), and as such this method may violate this assumption as juvenile/sub-adult Snapper can either be targeted, but are also often caught as bycatch by anglers using generic (i.e. not species-specific) methods. Total mortality estimates can also be highly variable between regions within a state, as these estimates are dependent on the age class by which an individual is 'fully recruited' into the fishery or the youngest age used in the regression (Ferrell & Sumpton 1997). As age structures can differ across small spatial scales, total mortality estimates have varied between 0.28–0.8 across six regions between Sydney, NSW and Fraser Island, QLD (Ferrell & Sumpton 1997), assuming a minimum age of 2 and 3, and omitting the oldest 1% of samples (age 10+). The minimum age used in our mortality estimates was 7 years, due to variation in abundance of younger age cohorts. This variation is likely driven by interannual recruitment variability, however, our time series (i.e. 2 years) is not long enough to confirm this assumption. This highlights that initial data collected from a species at the range edge or an emerging fishery may violate assumptions of traditional fisheries assessments and should be interpreted with caution.

Assuming different ages of T_{max} resulted in considerable differences in natural mortality. For example, using the Tanaka (1960) method, and assuming 5% of the population reach the T_{max} observed in our sample (i.e. 19.19 years) resulted in an estimated natural mortality of 0.16 yr^{-1}, which is nearly double than assuming 5% reach the T_{max} observed for the species across its range of 40 years, estimating a natural mortality of 0.07 yr^{-1}. A similar pattern was found using the $Hoenig_{nls}$ method, assuming T_{max} of 19.19 resulted in an estimated natural mortality of 0.33 yr^{-1} vs only 0.17 when assuming T_{max} of 40 years. Recruitment dynamics of temperate Snapper stocks on the mainland tend to be patchy and variable (Hamer & Jenkins 2004, Saunders et al. 2012), often resulting in populations dominated by relatively few age classes (McGlennon et al. 2000, Fowler et al. 2020). As such, it is possible that our study may have underestimated T_{max} due to the relatively short data collection period of this project (i.e. 2 years). Further sampling is needed to understand the variability of the age class structure of the Tasmanian Snapper population to adequately determine the T_{max} for this region. Therefore, assuming a T_{max} of 40 years when estimating mortality may be more appropriate. Similarly, estimated natural mortality was also substantially higher when using a T_{max} of 31 years (0.21) versus 41 years (0.167) in Queensland Snapper (Then et al. 2015, Wortmann et al. 2018), and significantly impacted modelled spawning potential ratio (i.e. higher spawning ratios with higher mortality), and estimated fishing mortality. Therefore, uncertainties in natural mortality estimation, particularly when overall exploitation rate is low, impacts model outputs of predicted biomass, and spawning potential (Fowler et al. 2013).

Due to the variability in estimates for both total and natural mortality, estimated fishing mortality for Tasmanian Snapper were also highly variable, ranging between 0.04–0.45 yr^{-1}, exceeding estimates of natural mortality under some scenarios. Fishing mortality estimates which exceed natural mortality estimates, coupled with few older fish (i.e. > 10 years) in the population suggest a heavily depleted stock, as seen in WA (Wise et al. 2007). However, fishing mortality is likely to be relatively low in this emerging fishery, although continued monitoring is necessary to assess the potential variation in recruitment and subsequent age-size class structure to fully understand the population dynamics of Tasmanian Snapper.

Growth

We found little evidence for differential growth of Snapper in Tasmania both between sexes and regions. There was a small effect between region only when individual size was unconstrained, however this is likely a modelling artifact due to differences in the size distribution between regions. When all data were pooled, the mean asymptotic length (L_∞) was 725 mm FL, where $k = 0.13$, and $t_0 = -0.75$. Furthermore, the von Bertalanffy growth curves derived in this study displayed only a small proportion of individuals close to attaining their asymptotic length. von Bertalanffy growth parameters have been well documented across NSW, Queensland, SA, Victoria and WA (Table G2). Asymptotic length for Snapper in Tasmania is relatively smaller than that of Victoria, NSW, SA and WA where estimated L_∞ generally exceed 800 mm FL, with the exception of some regions within WA and SA which have been estimated to be as small as 563 mm FL (Wakefield et al. 2016) and 673 mm FL (Fowler et al. 2013) respectively. Such variability in growth can be caused by environmental factors, resource availability, driven by location, or inconsistencies in age estimation (Hyndes et al. 1998, McGlennon et al. 2000, Murphy et al. 2013).

Rates of early growth are useful for management, as they provide insight on early life-history strategies, and can therefore help determine an appropriate MLS. SA supports the fastest growing Snapper, where von Bertalanffy growth models predict at the age of three, fork length can range between 394.18 and 427.81 mm (McGlennon et al. 2000, McGlennon 2003). Similarly, Snapper in WA, and NSW are also relatively fast growing, reaching 3 years at 320.21, 327.36, and 312.24 mm FL in the north and south of WA, and in NSW respectively (Jackson et al. 2010, Wortmann et al. 2018). Interestingly, our Tasmanian sample demonstrates similar early growth (reaching year 3 at 279.73 mm FL, using the pooled model) as Snapper in both Victoria and QLD, where von Bertalanffy growth curves predict an age of 3 at 284.81 and 285.45 mm FL respectively (Sumpton 2001, Coutin et al. 2003). Due to variation in age at maturity, Snapper in QLD and WA reach their MLS of 350 mm TL (~300 mm FL) and 410 or 500 mm TL (356 or 410 mm FL) respectively, or between 3.5–5 years of age which allows for at least one year of spawning before recruiting to the fishery. However, Snapper in Tasmania, SA, NSW, and Victoria (for juvenile ‘pinkys’ only) reach their MLS between ~2.5–3 years of age (NSW DPI 2021, PIRSA 2021, VFA 2021b, NRE Tas 2022a), which may not offer protection to enough spawning individuals, and could lead to the population to being recruitment-overfished, whereby there are insufficient spawning adults to sustain adequate recruitment into the fishery (Stewart 2011).

Age/length

Snapper in this study ranged in length from 241–711 mm FL for females, 233–633 mm FL for males, and 150–316 mm FL for juveniles of indeterminate sex, and trends in size between males and females were similar, however we did find regional differences in Snapper age/size structure where individuals were larger and older in the north versus in the south. The age of Snapper in this study exhibited a multi-modal structure, with modes in the north stratum at two and six to eight years of age, and individuals from the south stratum predominantly from the year four age class. This pattern may partially have arisen due to sampling. For example, as at least one recreational fisher anecdotally reported that Snapper ~350 mm FL are commonly prepared whole for eating (i.e. whole baked fish) and thus frames in that size class may be under-represented in the frame donations, which could contribute to the relative dearth of Snapper of that size class in the north stratum. However, the pattern is even more marked across age rather than size classes, e.g., the absence of five-year-old Snapper in the south stratum despite fish of age four being the modal age, which suggests a biological cause rather than a size bias from recreational anglers, as 350 mm FL fish can range between four to five years. The multimodal age structure is similar to that described from commercial longline catches from Port Philip Bay in the 1990s, in which a strong 1991 year classes in the middle of several years of particularly weak year classes formed a mode of increasing size as it aged through the fishery (Coutin et al. 2003). Highly variable year classes, with strong classes occurring every 1–10+ years also occur in South Australia (McGlennon et al. 2000, Fowler et al. 2020).

As a long-lived, commercially exploited species, Snapper are susceptible to age truncation. For example, in the late 2000s, at a time the stock was considered growth overfished, where more than 95% of commercially landed Snapper in NSW were less than five years of age (Stewart 2011). In the higher latitude fisheries in Port Phillip Bay and northern Spencer Gulf, the majority of commercially caught Snapper had routinely been older than ten years of age in the past several decades with modal classes of 10–20+ years of age, depending on the timing of recent strong year classes (McGlennon et al. 2000, Coutin et al. 2003, Hamer & Conron 2016). The results of the current study, with a median age of six, 10% of individuals older than ten years and a maximum age of 19, places Tasmanian Snapper somewhere between these left-shifted (i.e. age truncated) and right-shifted (i.e. driven by isolated aging year classes) patterns.

Maturity

The smallest reproductively active female and male measured in our sample were 315 mm and 310 mm FL respectively, where both individuals were approximately four years of age. Due to the relatively small sample of females with hydrated oocytes (n = 4) or spermiated males (n = 9), and individuals within the size range of 350–400 mm FL (n = 32, 7.3% of sample), coupled with difficulties discerning between virgin versus inactive gonads in larger fish, it was not possible to fit a logistic regression to estimate a size at 50% maturity (L_{50}) for Tasmanian Snapper. However, it is apparent that there is no sign of reproduction near the MLS in Tasmania of 260 mm FL.

Sexual maturation of Snapper varies markedly around Australia, depending on the region (Table G2). In QLD and northern NSW, individuals are quick to mature, with the youngest mature at 1.8 years of age and small with L_{50} values ranging from 218–220 mm FL, and 100% of the population mature at 330 mm FL at 5+ years (Table G2; Stewart et al. 2010). As such the MLS in NSW (260 mm FL) and QLD (303 mm FL) exceed the L_{50} of the population. In Western Australia, the trend is the opposite, fish mature later in life, where females mature at an older age and larger size (i.e. Female L_{50} = 348–508 mm FL at 3.2–5.7 years versus male L_{50} = 243–491 mm FL, at 1.6–5.6 years: Lenanton et al. 2009, Female L_{50} = 348–508 mm FL at 3.2–5.7 years versus male L_{50} = 243–491 mm FL, at 1.6–5.6 years: Jackson et al. 2010, Wakefield et al. 2015). As such the MLS for Snapper is higher in WA ranging between 410 mm TL (356 mm FL) in regions north of 31°S, and 500 mm TL (434 mm FL) in regions south of 31°S (DPIRD 2021), allowing for most of the population to reach at least 50% maturity.

Our results are most similar to those reported for Victoria, where the smallest mature individual in Port Phillip Bay was 270 mm FL, and females in the population reached 50% maturity at 363 mm FL at 4.9 years, and 100% of the population reached maturity at 480 mm FL (Coutin et al. 2003). However, the management of Snapper in Victoria only allows for harvesting 3 individuals > 400 mm TL (347 mm FL), and juveniles > 280 mm TL (243 mm FL) for a total bag limit of 10 (VFA 2021b). Such a strategy is useful to limit age truncation within the older age cohorts, however, the current MLS of 280 mm TL (243 mm FL) for juveniles, which is below L_{50}, may limit spawning and recruitment potential. As such, a similar management strategy would be appropriate for Tasmania (i.e. limiting harvesting of older individuals), however increasing the MLS to 380 mm TL (330 mm FL, ~4 years) would ensure at least one year of spawning for 50% of the population.

Seasonality of spawning

We found evidence for seasonality in spawning in Snapper caught in Tasmania, where there was a clear relationship between both the proportions of reproductive stages, and mean GSI values versus time of year. For example, for females GSI peaked in December, with a potential second peak in February, correlating with a high proportion of hydrated oocytes in November, December and February. Similarly, for males, GSI peaked slightly earlier in November, with a large proportion of spermiated males from November to January. Vitellogenic females and spermatogenic males were present between October to April. However, the first spent female occurred in January, whereas the first spent male occurred in March.

Due to temporal variation in water temperature, the time and duration of spawning of Snapper varies latitudinally, where Snapper from higher latitudes (SA, Victoria) spawn mid-late austral spring (i.e. Sept–Nov; Saunders et al. 2012, Wakefield et al. 2015), and Snapper at lower latitudes (21–24°S) on the East and West coasts of Australia spawn in the austral winter (i.e. May–Sept; Jackson et al. 2010, Wakefield et al. 2015), coinciding with a narrow thermal window for spawning between 18–21 °C. In the North Spencer Gulf, the gonadosomatic index (GSI) increases in October at water temperatures of ~17 °C, and the onset of spawning occurred around 18 °C, typically in late November (Saunders et al. 2012). Similarly in south-west Australia (~35 °S), females with macroscopically developed/ripe ovaries appeared in September at a water temperature of 15.6 °C, peaked in November at 18 °C (48% of females), and declined again by December (7%) at 21 °C (Wakefield et al. 2015). In Port Philip Bay, macroscopically ripe ovaries appeared in October at water temperatures 14–15 °C (Coutin et al. 2003). Spawning occurred in November and December, peaking at 18.5 °C, and declining well before the seasonal maximum temperature of ~20.6 °C. Conversely with the Tasmanian sample, the peak in GSI for males in November coincided with water temperatures varying between 13–15 °C in 2019 and 2020 (IMOS 2021), and peak GSI for females in December coincided with temperatures varying between 13.5–16.5 °C in 2019 and 2020 around the whole Tasmanian domain (IMOS 2021). However, satellite derived mean SST is likely to underestimate local variation in temperature, particularly in shallow embayments where temperate Snapper are known to spawn (Crossland 1977, Coutin et al. 2003, Hamer & Jenkins 2004), which may reach maximum temperatures exceeding 18 °C in November and December. However, in Tasmania, SSTs are highest in January and February with temperatures typically range between 18–20 °C (IMOS 2021) which may support a secondary peak in spawning, but also facilitate settlement and recruitment.

The apparent 18 °C trigger for temperate Snapper spawning onset has also been noted among captive Australian Snapper (Battaglene & Talbot 1992). However, in the Harukai Gulf, New Zealand (36°S) the onset of spawning occurs around October in 15–16 °C water, concluding when temperatures reach 18.8–21 °C around January (Crossland 1977). While this indicates the species is capable of spawning at these temperatures, spawning among high latitude Australian Snapper still appears to be limited when thermal conditions are unfavourable. For example, in south-west Australia, where winter water temperatures typically reach a minimum of 15 °C, spawning appears to have been skipped entirely during an anomalously cold spring in 2005 (Wakefield et al. 2015). Therefore, regional thermal acclimation is likely a driver for optimum spawning temperatures and reproduction in Snapper.

Recruitment variability

It is possible the patchiness of age classes of Snapper analysed is due to highly variable interannual recruitment, as is seen in other states (Coutin et al. 2003). Water temperature and environmental factors that promote high prey (e.g., copepod nauplii) productivity appear to contribute to year class strength by allowing greater larval Snapper growth and survival (McGlennon et al. 2000, Murphy et al. 2013). However, while it appears Tasmanian snapper are genetically linked to either the west or east Victoria stocks (Morgan et al. 2018), the exact sources of recruitment to the north coast and south-east of Tasmania are not currently known. In the absence of more definitive data, comparing year class structure of Snapper caught in Tasmania over time to the strength of year classes from potential sources of recruitment on the mainland could provide insight, although of the three most likely sources of recruitment (Port Phillip Bay, Eastern Victoria, or self-recruitment), 0+ age class abundance data are currently only available from Port Phillip Bay (Conron et al. 2020). However, while variable, recruitment in Port Phillip Bay has increased in recent years, with 2018 having had the highest recorded abundance of 0+ recruits (Conron et al. 2020). Therefore, should Port Phillip Bay year classes reflect recruitment patterns to Tasmania, there should be an increase of ~4+ individuals into the fishery in 2022.

If Tasmania is subject to variable interannual recruitment like nearby temperate Snapper stocks, avoiding age truncation may be important to maintain the nascent fishery between replenishment with sporadic strong recruitment years. As an example, in northern Spencer Gulf in 1994, despite no strong year classes recruiting to the longline fishery for the 14 years prior, the fishery was still supported by 1979 and 1973 year classes (i.e. fish 15 and 21 years old at time), which respectively made up 27% and 15% of the catch in that year (McGlennon et al. 2000). By contrast, the major year classes of 1999 and 1997, which dominated the age structure of catch in the region until 2015, were depleted by 2017 (i.e. at 18 and 20 years of age) so the vast majority of fish caught were five or less years of age and the oldest Snapper aged was 11, indicating severe age truncation (Fowler et al. 2020). Due to the bourgeoning nature of a population at the range edge, the Tasmanian population may be highly susceptible to overfishing of large fish, if they are contributing to self-recruitment before the stock is fully established. Molecular analysis or otolith chemistry would therefore enable us to determine whether the stock is self-replenishing, or persists as a sink population from mainland Australia.

Diet

As reported elsewhere in Australia and New Zealand, Snapper in Tasmania are opportunistic generalist predators (Godfriaux 1969, Colman 1972, Godfriaux 1974, Parry et al. 1995, Coutin et al. 2003) with patterns in diet composition varying with Snapper size and demersal habitat. While only stomach contents were systematically analysed, the contents of hindguts from several frames were also opportunistically examined and contained several species that didn’t occur in stomach samples, including the Heart Urchin *Echinocardium cordatum*, the Venerid Clam *Tawera gallinula*, and the crustaceans *Ovalipes australiensis* and a sphaeromatid isopod of 15 mm width.

Regional differences in diet composition

Snapper from south-east Tasmania, predominantly collected near reef features and over soft sediments in the greater Storm Bay area, consumed a variety of prey including crustaceans, fish, cephalopods, gastropods, bivalves and polychaetes. This is consistent, but slightly broader than the prey of Snapper in the similar habitats of Port Philip Bay, where Coutin et al. (2003) found Snapper prey consisted predominantly of crustaceans, followed by bivalves, cephalopods and fish. Similarly, Parry et al. (2005) found that among Port Philip Bay Snapper collected on soft sediments distant from reefs, prey was composed of crustaceans (28–63% by volume), molluscs (0–42%) and fish (21–26%).

In the north Tasmania stratum, in which Snapper stomachs analysed were collected predominantly from coastal sites (46 vs 14 from the estuarine St. Helens and Tamar River), stomach contents were dominated by bivalves, mainly doughboy scallops (*Mimachlamys asperrima*) and smaller proportions of crabs, polychaetes, and fish. In contrast, Snapper from coastal Victoria predominantly consumed (> 60% of stomach contents) fish or cephalopods among the western stock and eastern stock (i.e. west/east of Wilson's Promontory), respectively (Coutin et al. 2003). However, coastal samples in the aforementioned Victoria study came from a range of commercial gears (traps, hook and line, trawl and seine) and charter vessels, while stomachs analysed in the present study were targeted at specific locations by recreational fishers which may explain the differences in diet across studies. While sampling differences could explain the predominance of *M. asperrima* in the diet of north Tasmania Snapper, the deeper portions of the Bass Strait are known to have a large population *M. asperrima* (Edgar 2008), which would likely provide an important food resource for Snapper in this region.

There was also evidence of fine-scale differences in Snapper prey within regions. For example, the positive correlation of seagrass with brittle stars and polychaetes and polychaetes with sea cucumbers and brittle stars appears to be a signature of feeding in seagrass beds, which are noted to be important nursery habitats for Snapper (Jackson et al. 2007, Parsons et al. 2014). While sampling wasn't extensive enough to test for differences in overall diet composition at a fine spatial scale, there were notable differences at the level of sampling site. For example, among the eight Snapper collected at Smithton and Wynyard, six (75%) had crinoids in their stomachs, a prey class absent at any other site (nor reported, to our knowledge, in any previous studies of Snapper diet).

Ontogenetic changes in diet composition

Gape limitation (along with changes in dentition and jaw musculature) are likely responsible for much of the significant shift in diet composition with increasing Snapper size. Among the Snapper frames analysed, mouth width increased hyper-allometrically with fork length. In other words, while all individuals < 350 mm FL had mouth widths under 35 mm, at 400–450 mm FL, (the length at which *M. asperrima* began to appear in Snapper stomach contents), most individuals had mouth widths greater than 50 mm. Stomachs containing *M. asperrima* typically had numerous large (> 20–25 mm fragments) of the scallop's shell, suggesting the anatomical capacity to crush the scallops is required to consume them. Similarly, in the south Tasmania stratum, cephalopods, small octopus of 60–80 mm mantle length were only present in the top third largest individual's stomachs, likely due to gape limitation.

Ecological implications of Snapper predation

In the present study, virtually all prey identified were benthic species. While generally considered a demersal predator, Snapper have been reported to consume pelagic species as well. In Shark Bay, WA a morphologically distinct semi-pelagic group of Snapper co-occur inshore with resident benthos-associated Snapper during spawning season (Moran et al. 1999). In the Hauraki Gulf, New Zealand there is a similar co-occurrence of a semi-pelagic ecotype, and stable isotope analysis indicates they have a more pelagic, higher trophic level diet (Parsons et al. 2016). Bottom-trawled (and thus presumably benthos-associated) Snapper are also reported to feed on pelagic prey. For example, occasional Pilchard (*Sardinops neopilchardus*), Anchovy (*Engraulis australis*), and Horse Mackerel (*Trachurus novaezelandiae*) were reported from Snapper stomachs that often also contained benthic teleost prey, suggesting opportunistic consumption when the baitfish were low in the water column (Godfriaux 1969, Colman 1972). While no pelagic teleosts were identified among Snapper prey species in this study, taxonomic resolution of teleosts was poor overall due to the advanced state of digestion of smaller teleost prey (i.e. consistent with small bait) and it is plausible that pelagic baitfish are opportunistically consumed by Snapper in Tasmania. Regardless, these results indicate that Snapper predation predominantly impacts benthic communities.

Snapper are known to be able to alter soft sediment communities surrounding reefs through predation. In north-east New Zealand, reserves with higher densities of Snapper and Southern Rock Lobster (*Jasus edwardsii*) had lower densities of myochamid and venerid bivalves (Langlois et al. 2005), the latter of which were identified from the gut contents of a Snapper from patchy reef habitat in the current study. The presence of Snapper may also induce behavioural changes in juvenile *J. edwardsii*, like decreased movement which may reduce the lobsters' feeding rates (Mislan & Babcock 2008). Thus, while only a single *J. edwardsii* was identified across the 77 stomachs examined (and was found in a 310 mm FL Snapper from the south Tasmania stratum), an increased abundance of Snapper could potentially result in both lethal and sub-lethal impacts on juvenile rock lobster.

The Doughboy Scallop (*M. asperrima*), which has recreational significance in Tasmania, was the only scallop species identified in Snapper stomachs, and they were consumed up to approximately 50 mm shell height. However, Snapper are known to consume other scallop species, including *Pecten sp*. (Coutin et al. 2003), so it is plausible that Snapper in Tasmania may prey on small commercial scallop (*Pecten fumatus*) in habitats that weren't sampled in the present study.

While not important prey items by frequency, occurrence of two non-native pest species in Snapper stomachs are notable. Four arms and the central disk of a northern Pacific seastar (*Asterias amurensis*) were identified in the stomach of a 615 mm FL Snapper collected off Bridport. *A. amurensis* was introduced to Tasmania in the 1980s, and its high densities in some areas have detrimental impacts on native benthic assemblages, including commercial species (Ross et al. 2002). There are currently very few predators of *A. amurensis* in Australia (Parry 2017), so while there was only one observed in stomach contents in this study, confirming large Snapper will prey on even fully grown *A. amurensis* is worth note and may have implications for biocontrol of the species. The Asian date mussel (*Arcuatula senhousia*), another recent introduction to Tasmania that can reach overwhelming densities in its non-native range (Slack-Smith & Brearley 1987, Edgar 2008) was identified among prey in both the north and south strata. Thus, due to the species' broad and opportunistic demersal feeding habit, increasing abundance of Snapper (and especially large Snapper) may provide some buffering effect against current and future introduction of non-native invertebrate species.

Species Distribution Model

Spatial predictions of suitable environmental habitat show that there has been 'suitable' habitat for Snapper around the entire Tasmanian domain since 1998. SDMs did not provide evidence of a southern range shift in suitable habitat, as predictions were highly resolved to nearly every coastline around the state. However, there was evidence of seasonal variation in the suitable habitat for Snapper, with more favourable habitat in the warmer seasons (i.e. summer and autumn), and there was evidence of a very small increase (i.e. maximum increase 0.002 units per year in the summer in the SE) in suitable habitat from 1998–2018. When comparing the averaged hindcasted (1998–2018) and forecasted (2036–2065) time periods, there was a consistent increase in suitability in the winter across all regions ranging from 126.33–249.31 %, with the greatest amount of change predicted for the winter in the North-East-North region. There was also a smaller increase in habitat suitability predicted for the summer (range: 21.01–40.27 %) and spring (range: 35–60.89 %) seasons.

As the SDM provided evidence of suitable habitat across the entire Tasmanian domain, mechanisms which enable recruitment or overwintering may be limiting their persistence, particularly in the south, and south-west of Tasmania, where reported catch is considerably less than in the north (though this may also be correlated with also less fishing effort and reporting). Firstly, sufficient spawning temperatures are necessary for a successful spawning event, and should temperatures be below the threshold, spawning will not occur (Wakefield et al. 2015). Studies across Australia suggest a period of consistent temperatures exceeding 17–18 °C are necessary for spawning (Saunders et al. 2012, Wakefield et al. 2015), and therefore at higher latitudes, this occurs mid-late austral spring (Wakefield et al. 2015). Secondly, elevated temperatures must be sustained post-spawning for maximum larval growth and settlement to ensure survival over the winter months. Interestingly, Fielder et al. (2005) did not find a difference in survival of new Snapper recruits between 15 °C and 24 °C, however growth increased with temperature by 600%, and swim-bladder inflation was significantly higher between 20–24 °C. Therefore, while new recruits may survive at 15 °C, their fitness may be compromised. Current mean summer SSTs in Tasmania, range between 16.62 °C (W)–19.12 °C (NEN), which suggest potential for spawning in the north, however under future predictions SSTs in all regions (except the west coast) exceed 17.33°C followed by autumns which exceed 15°C in all regions, which may be sufficient to enable adequate spawning, settlement and therefore recruitment of Snapper in most regions of Tasmania under future warming. Furthermore, as SST data, and predictions are temporally aggregated (i.e. by season), our estimates do not account for the potential maximum and minimums which may occur during the spawning season, which may further enable successful spawning and recruitment. Lastly, successful settlement and juvenile growth is also largely influenced by resource/food availability (Murphy et al. 2013), which was not accounted for in our SDM.

Port Phillip Bay is a hotspot in Victoria for Snapper spawning and recruitment (Coutin et al. 2003, Hamer & Jenkins 2004), which may be the most similar in thermal habitat to regions in Tasmania. Using the SDM to predict habitat suitability within the hindcast period within Port Phillip Bay, we determined the mean habitat suitability for this region ranged between 0.11 in the winter to 0.47 in the summer, across an area of ~34.69 km^2, and maximum suitability values exceeding 0.95 (Figure 39), a suitability value not found in Tasmania. Even under future projections, mean suitability of the North-East (NE), East (E), and South-East (SE) coasts are 0.30, 0.33, and 0.32 in the summer, across areas of 39.64 km^2, 64.55 km^2 and 93.72 km^2, respectively, ~60% of the habitat suitability of a known hotspot for recruitment. Therefore, while Tasmania may not be as optimal for nursery habitat as Port Phillip Bay, the SDM suggests that under future warming, habitat is likely to become more suitable, and future temperature projections likely to enable overwintering of new recruits. By using the majority of distribution of Snapper in eastern Australia to determine the response of the distribution to temperature, we were unable to account for regional acclimation or adaptation to temperature. Therefore as cold-acclimation at the range edge for Snapper is possible (Wolfe et al. 2020), our SDM may actually under-represent habitat suitability for the species at its range edge. Integrating *in situ* behaviour and physiological data from mechanistic studies into correlative models such as SDMs would improve our understanding of how regional variation in thermal tolerance determines the range edge.

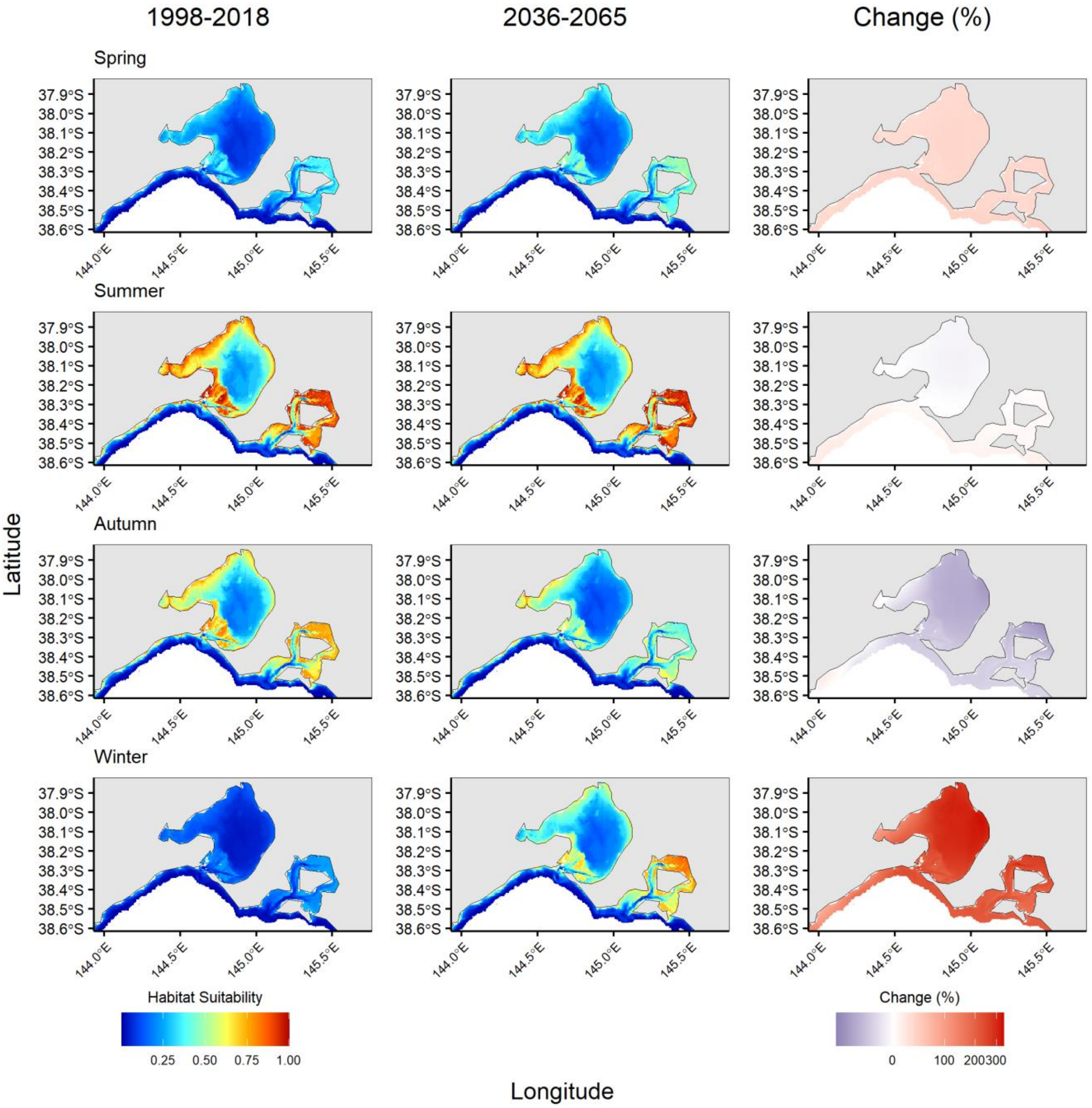

Figure 39. Seasonally aggregated spatial predictions of Snapper (*Chrysophrys auratus*) oceanographic habitat suitability in Port Phillip Bay, Victoria, and the percent change in suitability between 20-year averaged hindcasted (1998–2018) and forecasted (2036–2065) predictions.

Fisheries Summary

The centroid of the reported Tasmanian commercial catch has shifted from across King Island, the Kent Group and the Furneaux Group islands in the Bass Strait (39–39.5°S) to along the north coast of Tasmania, equivalent to 30.1 km per year. However, whether this is due to a shift in the distribution of Snapper or reflects a redistribution of fishing effort in the area remains unclear. While catch of Snapper has been reported in the southern portion (> 42°S) of Tasmania since 1998 and on the west coast since 2008, commercial catch does not appear to have discernibly expanded in these regions, with catches typically < 25 kg and < 250 kg per year, respectively. Consistent low catches across the state since the mid-2000s are unlikely to reflect high exploitation rates, as our estimates for fishing mortality were variable, ranging from 0.04–0.45 yr^{-1}. Under Tasmania's Fisheries (Scalefish) Rules 2015, commercial license holders are subject to a combined possession limit of 250 kg of Snapper, Striped Trumpeter and Yellowtail Kingfish. By contrast, Snapper catch exceeded 50 kg in only 3.1% of catch reports since 1995–1996, and only a single report in 1998 exceeded 250 kg. Furthermore, Snapper is largely a bycatch product of the Commonwealth fishery (not assessed here) and catch within the shark gillnet fishery in the Bass Strait were recently non-negligible at > 2 t per year.

Commercial catch of Snapper in Tasmanian waters and the Bass Strait is considerably lower than in neighbouring states, where reported catch for the 2018–2019 financial year in Tasmania was 0.383 t versus the annual catch on the mainland ranging from 30.09 in Victoria to 280.90 t in South Australia (Table 12). However, unlike other states, reported annual commercial catch in Tasmania doesn't seem to be undergoing a recent decline in catch. Indeed, there was a 98% decline in Tasmanian commercial catch from 1998 (1,444.37 kg) to 2016 (25 kg)–although there may be some uncertainty in historical data recording quality—but following 2016, there has been a relatively large increase (17.36-fold) in catch as 434.16 kg were reported for the 2020–2021 financial year. Conversely, across the mainland, annual reported commercial catch has been declining from 2010–2019 ranging from a 44% decline in NSW to an 92% decline in Western Victoria (Fowler et al. 2021b). As such, biological stocks in QLD, the Spencer Gulf/West Coast and the Gulf St. Vincent of SA, and Shark Bay in WA have been deemed 'depleted' in the most recent stock assessments, likely driven by overfishing and shifts and increases in efficiency of gear types, coupled with reduced recruitment in recent years (Fowler et al. 2021b). Shifts to management have been implemented to reduce fishing pressure in these regions, however the effects of these measures are still in their infancy and have yet to influence these biological stocks (Fowler et al. 2021b). Assessments of catch, CPUE, age composition, mortality and modelling of spawning potential and estimated biomass deem the NSW, Western Victoria, Shark Bay Freycinet Estuary, South Coast (WA), Shark Bay (Inshore Denham South: WA), and Shark Bay (Inshore Eastern Gulf: WA) biological stocks as 'sustainable' (summarised by Fowler et al. 2021b).

Similar to QLD, Victoria and SA, where recreational catch of Snapper exceeds commercial catch (Table 12), the estimated retained recreational catch of Snapper in Tasmania in December 2017–November 2018 of 2.65 t was approximately 6.97-fold greater than the total commercial catch at the time (0.38 t), Figure 38b). Relatively similar catch between the recreational and commercial sectors have been reported for NSW and WA (including charters: Table 12).

Table 12. Total annual harvest (tonnes) from commercial and recreational catch of Snapper (*Chrysophrys auratus*).

	Queensland	**New South Wales**	**Victoria**	**South Australia**	**Western Australia**	**Tasmania**
Commercial	44 t 2018[1]	160 t 2018–19[2]	30.9 t 2018[3]	280.9 t 2018[4]	141 t 2018–19[5]	0.38 t 2018–19[6]
Recreational	80k individuals; 118 t[¶] 2019–20[7,8]	106 t 2017-2018[9]	670 t W. Vic + 30 t E. Vic stock[¥] 2007–08[4,10]	332 t 2013-2014[11]	78.6 t + 22 t charter fleet 2017–18[5, 12]	2.65 t* 1.5k individuals 2017–18[13]

[¥] calculated from the individual total of 612,202 fish retained and the mean fish weight listed in Ryan et al. (2009), and Lyle et al. (2003) of 0.7 kg per fish.

*calculated from an average predicted weight of our Tasmanian sample of 1,768 g (± 33% *SE*), excluding Bridport which had a high size selection from angler bias.

[1] Heaven 2020.

[2] Stewart 2020.

[3] Conron et al. 2020.

[4] Fowler et al. 2019.

[5] Fairclough & Walters 2020.

[6] NRE Tas reported catch

[7] Fowler et al. 2021b.

[8] Teixeira et al. 2021.

[9] Murphy et al. 2020.

[10] Ryan et al. 2009.

[11] Giri & Hall 2015.

[12] Ryan et al. 2019.

[13] Lyle et al. 2019.

Implications for management

The MLS, bag and possession limits for Snapper vary considerably around Australia. NSW is the most liberal state, with the smallest MLS of 300 mm TL, and the largest bag limit of 10 (NSW DPI 2021). In Victoria, the bag limit is also 10, however it is limited to only 3 individuals > 400 mm TL, and the rest, juvenile/'pinkys' > 280 mm TL (VFA 2021b). In comparison, QLD is relatively conservative with a higher MLS of 350 mm TL, and a possession limit of 4, and limit of one fish over 700 mm TL (Queensland Government 2022). South Australia is the most restrictive with an MLS of 380 mm, bag limit of one, and a possession limit of three in the south-east, and a total closure on the West Coast/Spencer Gulf/Gulf St. Vincent waters until 2023 (PIRSA 2021). Fishing regulations in Tasmania are most similar to NSW, with an MLS of 300 mm TL, bag limit of five and possession limit of 10 (NRE Tas 2022a). However, in contrast to NSW, where 50% of the population is mature at, or close to the MLS (i.e. 251–320 mm TL, 218–278 mm FL, 1.7–2+ years; Stewart et al. 2010), our results suggest an MLS of 300 mm TL does not offer protection to at least 50% of the spawning population (i.e. L_{50}), thus may limit recruitment potential in Tasmania.

Species experiencing high exploitation and variable recruitment such as Snapper are susceptible to age truncation, which can severely impact a population's capacity to respond to environmental change or recover from sustained or increased fishing pressure (Stewart 2011). As such, following years of poor recruitment, coupled with increased/sustained exploitation led to the collapse of the Snapper fishery in western SA, leading to a complete closure of the area until 2023 (PIRSA 2021). Similarly, age truncation has been identified in both the commercial and recreational sectors in QLD, (2007–2019), with a decline in older and larger fish, coupled with variable abundance of pre-recruits (Bessell-Browne et al. 2020) rendering the stock recruitment impaired (Fowler et al. 2021b). Age truncation was also identified in NSW in 1997, where most of the stock was less than five years old (Ferrell & Sumpton 1997). In response to this assessment, the NSW MLS was increased from 28–30 cm TL (Wortmann et al. 2018), and most recent analyses of age/size composition of landings suggest an increase in the average size since ~2008 (Wortmann et al. 2018). Therefore, under appropriate management, recovery from severe age truncation is possible.

The age/size composition of Tasmanian Snapper from this study also features relatively few older and larger fish (likely reflecting the fact that it is an increasing population rather than overfishing) but did have a mean age of ~7 years for both males and females, therefore not as severely truncated as in depleted populations. Although there were a few mature fish in our sample, due to the patchiness in the age/size structure, the source of recruitment for Snapper in Tasmania is unclear. Larval duration of Snapper is relatively long, typically lasting 2.5–3 weeks (Francis 1994, Fowler & Jennings 2003), and as such, could facilitate larval advection from Victoria. Interannual recruitment is extremely variable for Snapper in Port Phillip Bay, and Western Victoria (Coutin et al. 2003), and therefore, if sourced from these locations, similar variation to age/size structure would persist in Tasmania. However due to the relatively short (i.e. 2 years) sampling period, recruitment variability, and therefore risk for age truncation in the Tasmanian population remains unclear. Therefore, we suggest ongoing monitoring of this population, by engaging with citizen science initiatives and the recreational fishing community in order to identify potential sources for recruitment and temporal shifts to age class/structure.

Management strategies to avoid age truncation include protecting the older/larger cohorts and ensuring sufficient recruitment into the fishery (Stewart 2011). Victoria has adopted this strategy by limiting collection of larger individuals (> 40 cm) to 30% of the bag limit (i.e. three out of a bag limit of 10: VFA 2021b). Given that our models suggest that Snapper in Tasmania are most similar to those in Victoria (although slightly smaller and slower growing), we suggest that this strategy of limiting the harvest of larger individuals may also safeguard Snapper in Tasmania from age truncation, as the proportion of large individuals (> 550 mm FL) is low (12%), and the population is likely still growing. Furthermore, under current management, Snapper in Tasmania are approximately 3 years at the current MLS (i.e. 30 cm TL), and reproductively immature as our smallest mature fish for males and females were 315 mm (i.e. 357.0 mm TL) and 310 mm FL (i.e. 362.8 mm TL) respectively. Increasing the MLS to 380 mm TL (i.e. ~4 years) would offer spawning-sized fish at least one year of protection before recruiting to the fishery. Furthermore, as Snapper live at depths exceeding 100 m, release weights may increase the potential for post-release survival (Butcher et al. 2012). Currently in Tasmania, the majority of Snapper are caught in reasonably shallow water, therefore the risk of barotrauma may not be as high as in regions where the continental shelf is close, such as NSW. However, it is becoming more common for fishers in Tasmania to find and target Snapper in deeper water. As such release weights should be actively considered in Tasmania as the fishery develops.

King George Whiting

Introduction

King George Whiting, *Sillaginodes punctatus* (Cuvier 1829) is the largest and longest-lived member of the Sillaginidae family, capable of growing to 800 mm in total length (Haddy, pers obs), 4.8 kg in weight and a maximum reported longevity of 22 years (Fowler et al. 2021a). Juvenile King George Whiting are inclined to aggregate in large schools of similar-sized fish and inhabit shallow (1–5 m) sand and seagrass habitats in sheltered coastal embayments and estuaries (Fowler et al. 2000a, Gillanders et al. 2003, Hamer et al. 2004, Jenkins 2005, Sulin 2012). Having tubular shaped mouths with protrusile jaws they are benthic carnivores that opportunistically feed on invertebrates such as worms, crustaceans and molluscs (Hyndes et al. 1998).

Distribution

King George Whiting are endemic to the southern regions of Australia extending from Western Australia to New South Wales, including Tasmania (Atlas of Living Australia 2021c). Commonly regarded as southern Australia’s most valuable inshore fin fish species (Fowler et al. 1999), they have been extensively targeted by recreational and commercial fishers, particularly in South Australia and Victoria. While they have been reported to occur in Tasmanian waters for at least 100 years (Lord & Scott 1924, Roughley 1957) they were mainly restricted to being seasonally common in the waters surrounding Flinders Island and rare south of Bass Strait (Last et al. 1983). Available evidence suggests that King George Whiting may be undergoing a poleward range extension into eastern Tasmania (Robinson et al. 2015), likely driven by a warming trend of central southern Bass Strait (Burnie) where the rate of warming is greatest in austral autumn (March–April) and spring (Nov–Dec: BOM 2021).

Connectivity / Stock Structure

Otolith microchemistry and genetic analyses confirm separate genetic stocks within each state’s jurisdiction (i.e. Victoria, SA, WA), with evidence of a small amount of genetic mixing between SA and Victoria (Jenkins et al. 2016). Furthermore, assessments of movement, spawning, nursery grounds, and hydrodynamics suggest that the biological stocks within SA, can be further subdivided into the regions of Gulf St. Vincent, Spencer Gulf, and the West Coast of SA (Fowler et al. 2000a, Fowler et al. 2000b, Fowler et al. 2002). While uncertain, it is assumed that there is a singular biological stock in Victoria (Jenkins et al. 2016), and WA (Hyndes et al. 1998, Sulin 2012, Brown et al. 2013).

Genetic analysis of the Tasmanian stock indicated that Tasmanian King George Whiting were genetically distinct from mainland stocks and that two genetically distinct stocks exist between the north-west coast and east coasts of Tasmania (Jenkins et al. 2016). Nicholls (2018) sampled from 2014–2018 and expanded sampling to include the emerging central north coast recreational fisheries of the Tamar River and Port Sorell estuary to provide preliminary descriptions of stock structure, growth, mortality and reproduction of King George Whiting from Tasmania. However, the lack of research and understanding with respect to Tasmanian King George Whiting is a significant issue for fisheries management.

Fisheries and Management

King George Whiting are managed as five units in Australia, at the scale of the biological stocks (i.e. three biological stocks in SA, single stocks in Victoria and WA). Recent stock assessments confirm that all stocks within SA, Victoria and WA are ‘sustainable’ based on key indicators of catch, catch per unit effort (CPUE), age structure, biomass and pre-recruitment surveys (Fowler et al. 2021a).

Stock assessment modelling in 2014 determined that the Gulf St. Vincent and Spencer Gulf biological stocks in SA as 'transitional-depleting' following a declining trend in CPUE and total catch between 1990–2012 (Fowler et al. 2014). As such, an increase in the minimum legal size (MLS) from 310–320 mm was implemented, in addition to reductions in bag (10 per person) and boat limits (30 with three or more people on board) and a month-long closure during the spawning season (PIRSA 2019a). These methods have proven effective with increases in CPUE (20–21%), and total catch (41%; Steer et al. 2018).

In Tasmania, commercial catches of King George Whiting date back to 1995, when compulsory logbooks were introduced. Historical timeline trends in commercial catches, although relatively small, demonstrate an increase in commercial catch volumes and the numbers of fishers reporting King George Whiting catches to a peak of ~3.5t in 2016–2017 (Fraser et al. 2021). Due to relatively low commercial catches the species has not been subject to annual stock assessments until 2021, which currently classifies the stock as "sustainable" (Fraser et al. 2021). Due to targeted recreational fishing on King George Whiting, these fish are now ranked as the 5th most retained marine fish species by recreational fishers in Tasmania (Lyle et al. 2019). In addition, Tasmania's north-west coast is now being regarded as Australia's last mecca for trophy size (> 1kg) whiting in Australia. The observed increase in catch rates of King George Whiting in Tasmania are not fully understood and it remains unclear whether catch trends are from an emerging abundance of King George Whiting, an increase in targeted fishing effort, or a combination of both.

Movements

Tag and recapture studies suggest differential movement between younger (< 4 years) and older (> 4 years, > 300 mm TL) King George Whiting, where younger fish tend to move less, within sheltered, shallow seagrass beds, while older fish tend to move greater distances as they migrate to deeper rocky reefs (Fowler et al. 2002). This ontogenetic shift occurs when young recruits (~1.5 years, 250 mm TL) migrate away from where they've settled (i.e. shallow seagrass beds < 1.5 m) into marine bays and estuaries (2–6 m; Hyndes et al. 1998). When individuals reach maturity (3–4 years) they then migrate into deeper habitats (up to 200 m; Fowler et al. 2002, Bray 2021). This ontogenetic shift may be driven by differences in resource availability (i.e. diet; Hyndes et al. 1997), to avoid intraspecific competition (Fowler et al. 2002), or predation (Hindell et al. 2002) and to spawn (6–50 m; Hyndes et al. 1998). Differences in the thermal environment may also benefit each life-history stage. For example, juveniles may maximise growth in warm shallow water, then move to cooler deeper water to take advantage of the low metabolic cost once fully grown (Macpherson & Duarte 1991). However, tag and recapture records of the South Australian population have indicated differential movement patterns between northern and southern populations, where spawning populations (i.e. in the south) did not display any directional movement, and non-spawning populations (i.e. in the Northern Gulf St. Vincent and the Spencer Gulf) travelled distances exceeding 100 km (Fowler et al. 2002). Such differential movement results in differences in the age/size structure of the regional fisheries, and as such has implications for management. Therefore, it is important to understand the potential spawning locations or ontogenetic movement of King George Whiting in Tasmania for effective management of this species at its range edge.

Reproduction

Between the ages of 3–4 years King George Whiting mature and migrate offshore (6–50 m; Hyndes et al. 1998) to spawn in the autumn months where there is a decrease in day length and water temperatures. On the mainland, wild fish have been observed to spawn between February and June when water temperatures range from 14.8–19.1 °C (Ham & Hutchinson 2003). After spawning, fish return to shallower coastal habitats in small schools or as solitary individuals to inhabit sand and broken ground within reef and seagrass habitats (Jenkins & Wheatley 1998, Hamer et al. 2004, Sulin 2012, Brown et al. 2013). Due to differences in both minimum temperature and variation in temperature in Tasmania there are many uncertainties relating to ontogenetic migration patterns, size at maturity and the onset, duration and peak of spawning activity and spawning locations.

Early life-history

King George Whiting eggs hatch after approximately 2.5 days and develop through a long temperature dependant larval phase between 80 and 146 days over the cooler austral autumn/winter months (Fowler & Short 1996). Despite an observed spawning range between February and June and a post-larval settlement range of June to November, it is not uncommon for most recruits (52–71%) to have a hatching date within a 3-week period in May (Rogers et al. 2019). Variations in recruitment strength resulting in strong and weak cohorts have been documented in WA, SA and Victoria (Jenkins & May 1994, Fowler & Short 1996, Hyndes et al. 1998), with fluctuations in ocean currents, temperature, prevailing winds and environmental river flows being identified as influential drivers of recruitment variability (Jenkins & Wheatley 1998, Fowler et al. 2000a). While these environmental variables play a significant role in larval dispersal, survival and recruitment, availability of high-quality seagrass beds (i.e. nursery habitats) is also important for juvenile fish growth and survival (Otway & Macbeth 1999, Fowler et al. 2000b).

Despite extensive studies being conducted on mainland stocks of King George Whiting, biological information on Tasmanian King George Whiting is very limited with only two recently conducted biological studies (Jenkins et al. 2016, Nicholls 2018). Jenkins et al. (2016) sampled fish between 2012 and 2015. This study was the first to confirm King George Whiting were spawning in Tasmania (North-West Coast only) and captured a fish (head only; aged 19 yrs) in 2014 with an estimated year of birth from 1995. Therefore, this research will focus on continuing the work of Jenkins et al. (2016) and Nicholls (2018) and aims to collate their data with ongoing sampling and present detailed spatial descriptions of the population dynamics of Tasmanian King George Whiting from three coastal regions, The North-West Coast (NWC), The Central-North Coast (CNC) and the East Coast (EC). area.

Materials and Methods

Samples and data were collected from multiple sources. Data from 2012–2015 was sourced from Jenkins et al. (2016), data and samples from 2014–2018 were sourced from Nicholls (2018), post 2018 samples were collected as part of this project. Sampling consisted of fishery independent research trips mainly targeting undersized fish (rod and line and seine nets), and fishery dependent sampling involving opportunistic donations of King George Whiting frames from recreational (rod and line) and commercial (gillnet and rod and line) fishers. Whole fish and frames were frozen and labelled with capture information (date, location, method) and stored for later biological processing. Samples were collected from three regions around the Tasmanian coast, North-West Coast (NWC; *n* = 929), Central North Coast (CNC; *n* = 484) and East Coast (EC; *n* = 366: Figure 40; Table 13). The most notable sample collection characteristics include a large increase in recreational frame donations and regional expansion to include the CNC initiated in November 2017. Commercial samples were only obtained from the North-West Coast of Tasmania. Due to a change in the fisher behaviour the number of commercial samples donated in 2018 was very low. Research sampling using a fine mesh beach seine to target Young-of-the-year (YOY) and 1-year-old fish was only performed on the East Coast in 2021 due to logistical and time constraints. As fishery independent sampling targeting undersized young fish was limited on NWC, mean size at age data for fish aged 2–3 years old from this region has an upward bias due to legal size constraints. Table 13 details the total sample numbers by region and year, to assist in interpretations.

Table 13. Regional sample numbers of King George Whiting (*Sillaginodes punctatus*) by capture year.

	Region		
Capture Year	**North-West Coast**	**Central-North Coast**	**East Coast**
2012	-	-	28
2013	89	-	23
2014	79	-	-
2015	46[¥]	-	-
2016	14	-	-
2017	130	27	-
2018	190	195	131
2019	65	92	42
2020	212	120	46
2021	63	34	88

[¥] samples were not subject to age determination

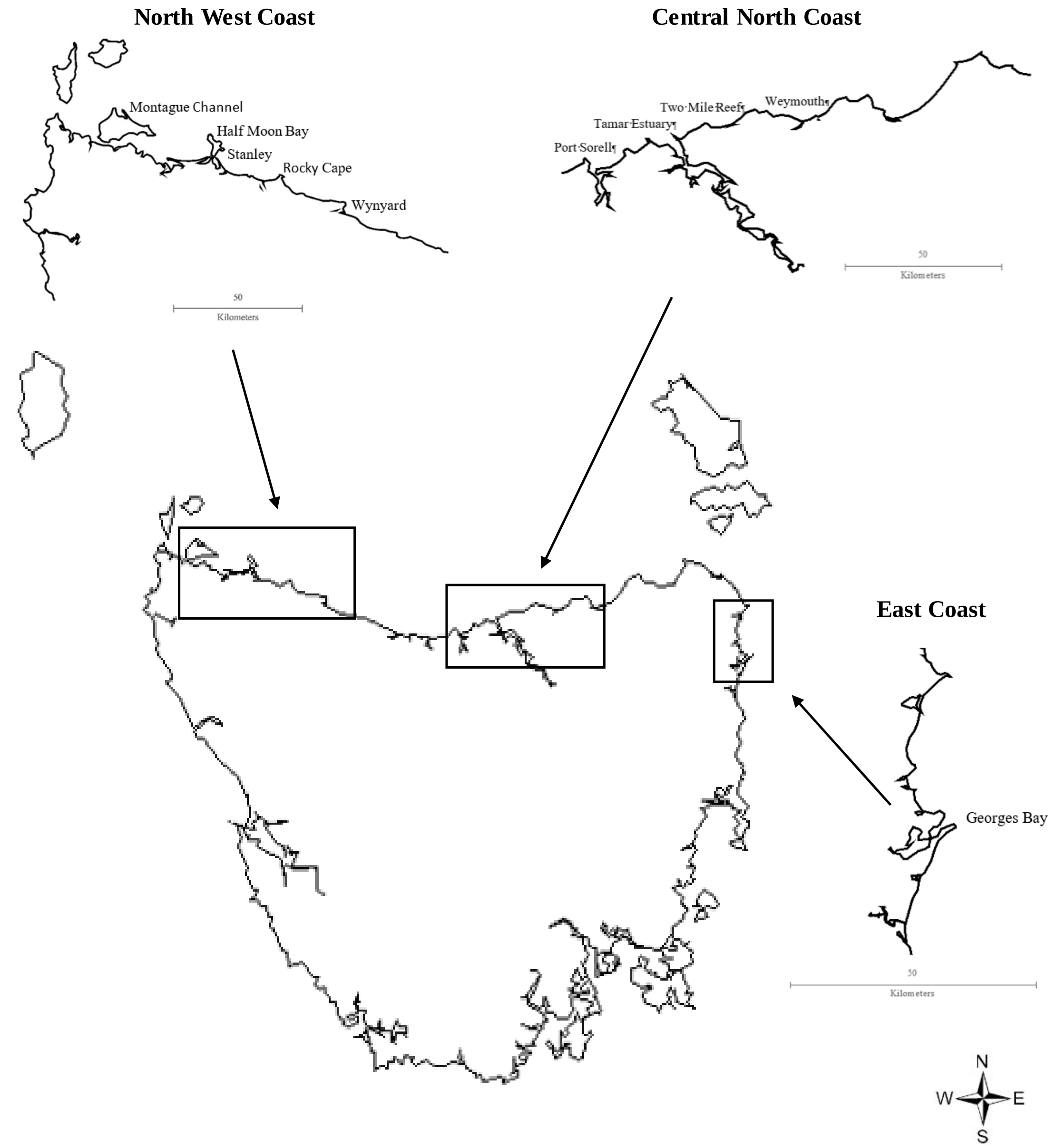

Figure 40. Coastal sampling sites for King George Whiting (*Sillaginodes punctatus*) along the North-West, Central-North Coast and East Coast of Tasmania.

Otolith processing, reading and age determination

Validation of annual increment formation was confirmed through seasonal changes in otolith margin classification (Figure 41) and used to assign an opaque zone closure date (OCZD) of the 1st of January of their capture year. The 1st May was defined to be the birth date with the first band not being visible in their first year (visible after ~20 months; Fowler & Short 1997). Date of capture (DOC) coupled with the marginal edge classification was used to adjust and re-allocate individuals to their correct cohort age. Individuals displaying edge or wide otolith margin classifications in January and February had increment counts adjusted by +1 (*n* = 12). Fish displaying narrow otolith edge classifications from August–December were adjusted by -1 (*n* = 41). A final decimal age was determined by adding the adjusted count + edge age (in decimal years; DOC – OZCD) + a first ring age adjustment value of 0.67 yrs (1st May to 31st Dec, *n* = 244 days; Fowler & Short 1997) ea. When samples did not have capture dates (*n* = 56) a decimal age could not be determined. Year of birth was calculated by subtraction of the decimal age from the DOC. To accurately present cohort frequencies by capture year, fish captured between Jan–April had +1 added to their round-down age to assign their birthday age on the 1st May of their capture year.

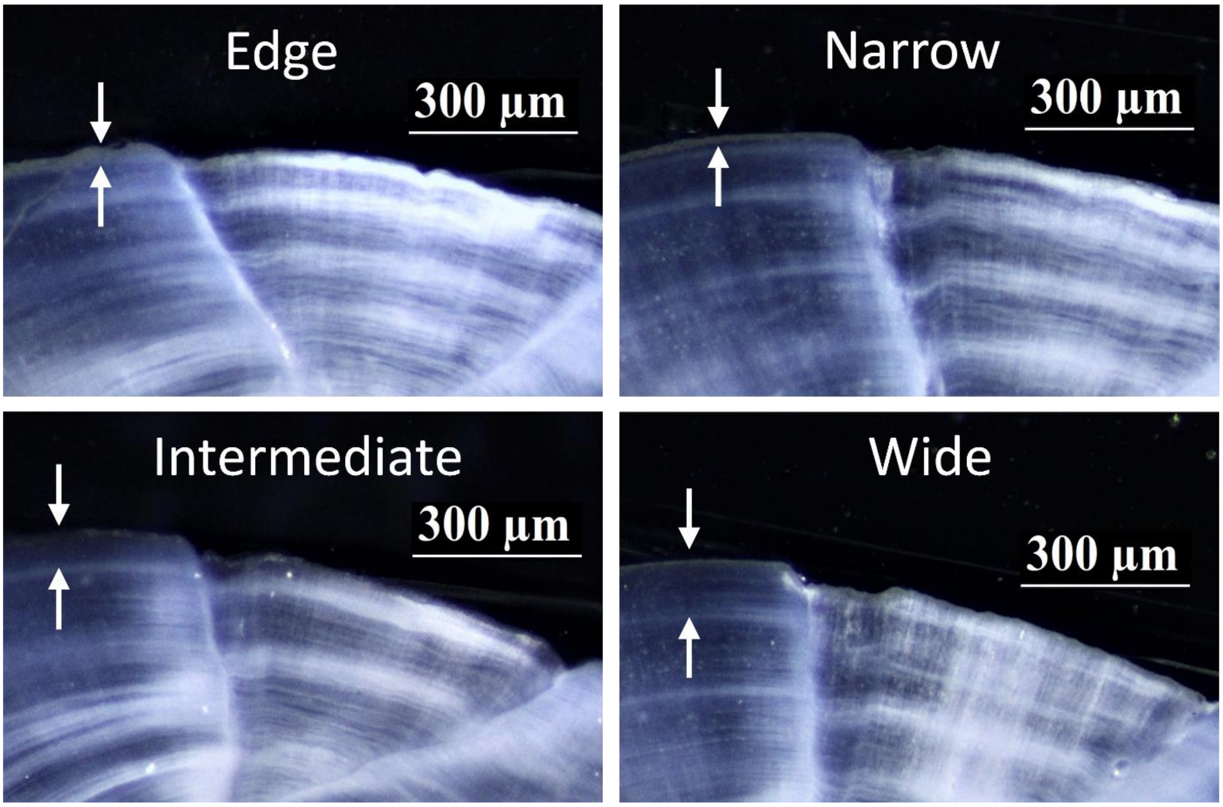

Figure 41. Example photomicrographs of different otolith margin classifications in sectioned otoliths of King George Whiting (*Sillaginodes punctatus*).

Results

Biological traits

Age validation and interpretations

From a total of 1787 fish, 1698 fish were subjected to age analysis. Opaque zones on transverse sections were easily discernible with an average readability score of 2.4 and an inter-reading agreement of 93% (based on a subset of 38% of all aged fish) (Figure 42A). The optical characteristics of the edges of sectioned otolith displayed a clear seasonal pattern, irrespective of capture region. Opaque edges were present from September to January with a peak occurrence in November. Narrow translucent edges were present from October to March with occurrence highest from December to February. Intermediate edges were present from January to July with a peak occurrence in March. Thereafter, wide translucent edges dominated between April to October. These observations were used to confirm the banding periodicity was annual and to assign an opaque zone closure date of 1st January (Figure 42B) for all regions. Alternating bands were easily recognisable as demonstrated by the image in Figure 42C and the grey value profile plot of the transect line placed across the sectioned otolith (Figure 42D). This fish was estimated to be 18.79 years old.

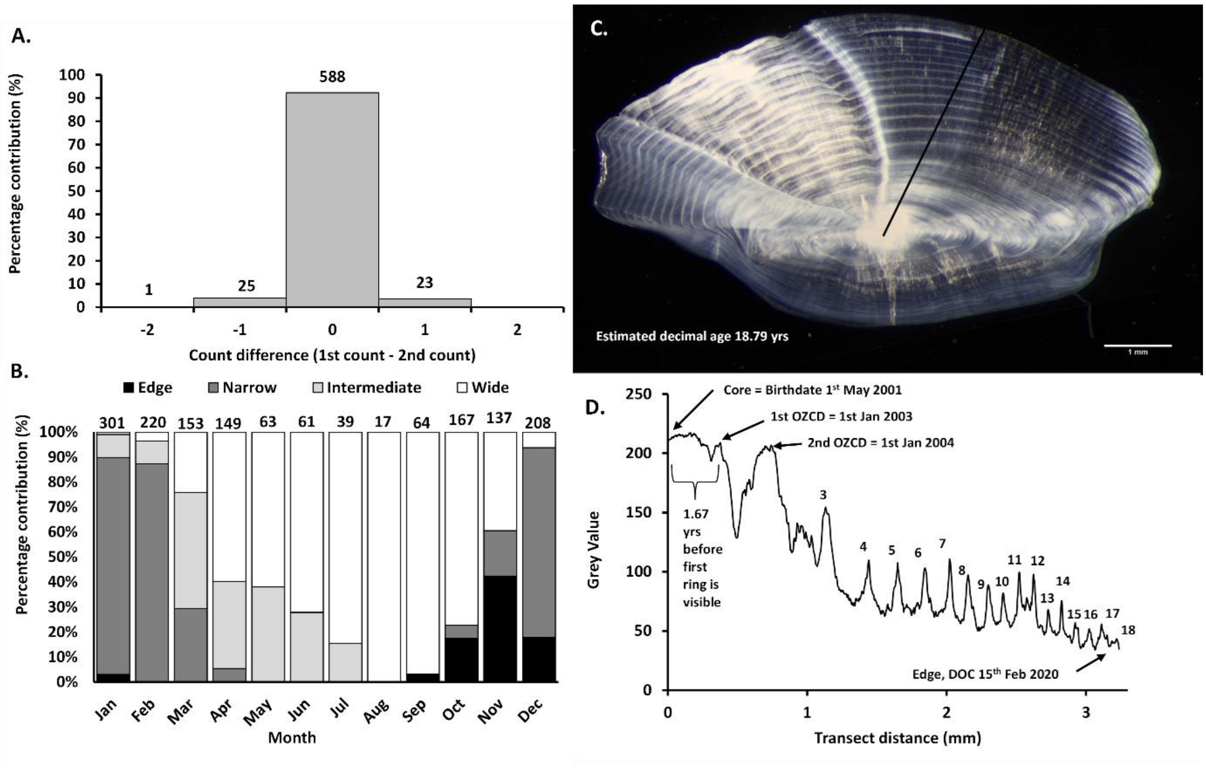

Figure 42. (A) Inter reading agreement, (B) seasonal changes in otolith edge classification, (C) photomicrograph of a transverse otolith section from a King George Whiting (*Sillaginodes punctatus*) estimated to be 18.79 years of age, and (D) the resulting grey value (where 0 = black and 250 = white and all values between are shades of grey) profile plots of the transect laid in C. Note birth date (core) was assigned as 1st May, opaque zone closure date (OZCD) is 1st January, and date of capture was 15th February 2020. When the 1st ring is visible the fish is ~1.67 years old.

Sex ratios, length and age frequencies

A total of 1664 fish possessed gonadal material, which included 49 indeterminate juveniles (FL= 53–165 mm), 955 females (FL 133–619 mm) and 660 males (FL 130–532 mm). The remaining 122 fish frames lacked gonadal material and were recorded as unknown sex (FL 265–585 mm) (Figure 43). Length frequency distributions were markedly different between regions, with many of the larger fish ≥ 400 mm FL obtained from the NWC (83% of *n* = 818). The average fork length (± 95% CI) for legal sized (≥ 32 FL) female and male fish from the NWC was 445 ± 5 mm and 427 cm ± 6 mm respectively. In contrast, the average female and male fork lengths for individuals ≥ 320 cm was 379 ± 7 mm, 370 ± 5 mm from the CNC, and 371 ± 5 and 363 ± 6 mm from the EC respectively.

Although females were more prevalent than males the overall trends in length frequencies were similar between sexes within each region, with the exception that large fish (> 500 mm FL) were more likely to be female (77%; *n* = 103 female: 31 male) (Figure 43). Where valid χ^2 tests were possible, significant bias away from a 1:1 sex ratio could be detected above 360 mm FL, however, all but one of these significant sex ratio biases could be associated with the samples collected from the NWC (Figure 44). The presence of the juvenile length modes between 50–80 mm FL and 130–170 mm FL on the EC was due to targeted fishery independent research sampling using a fine mesh beach seine which only occurred on the EC.

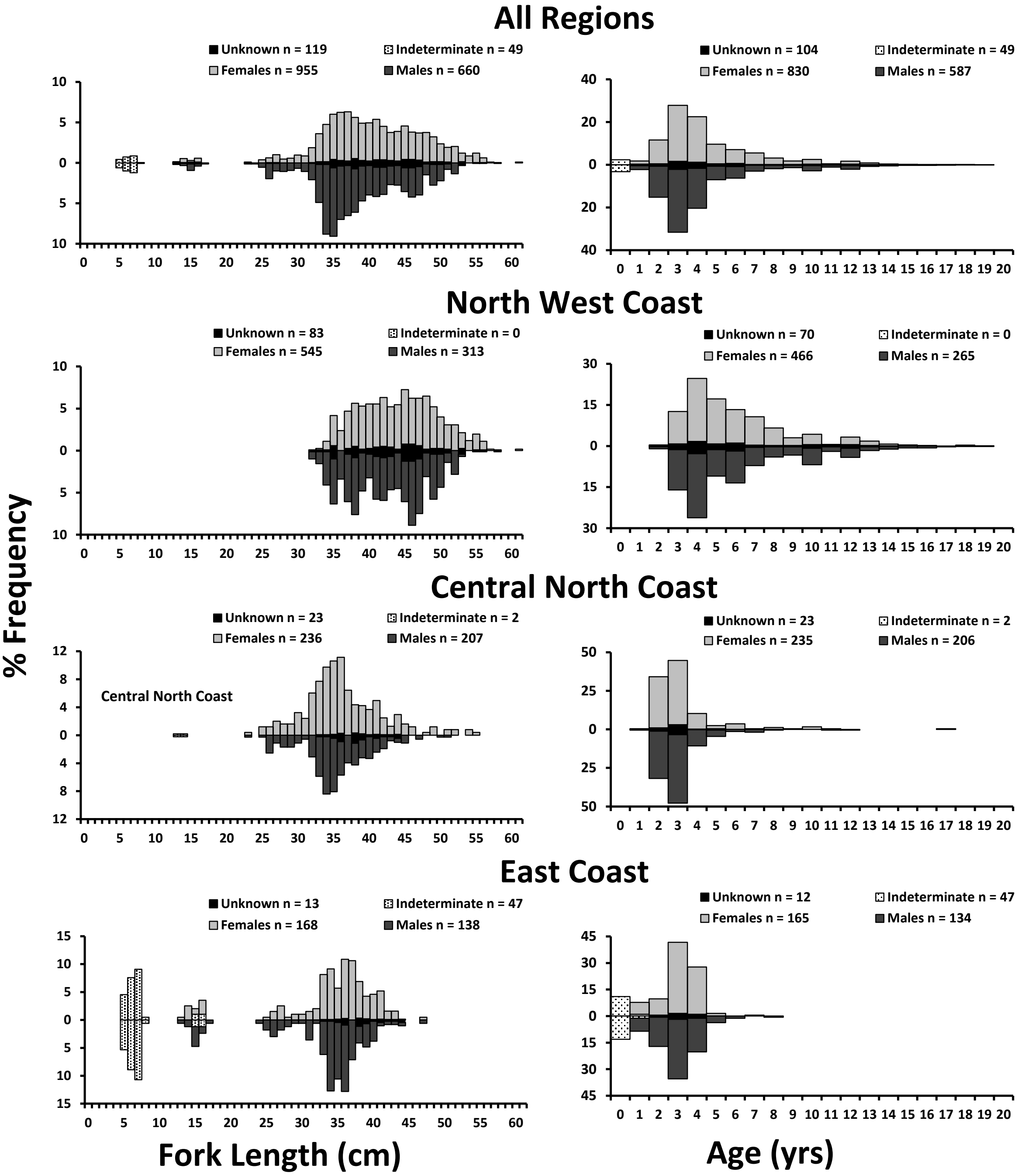

Figure 43. Spatial comparisons of the length and age percent contributions for male, female, indeterminate and unknown sex King George Whiting (*Sillaginodes punctatus*) from the North-West, Central-North and East Coasts of Tasmania. Note: data pooled across sampling years (2012–2021) and *y*-axis maximums are not equal values. Sample numbers for indeterminate and unknown sex were shared between the upward and downward directions with a 1:1 ratio across all size and age classes.

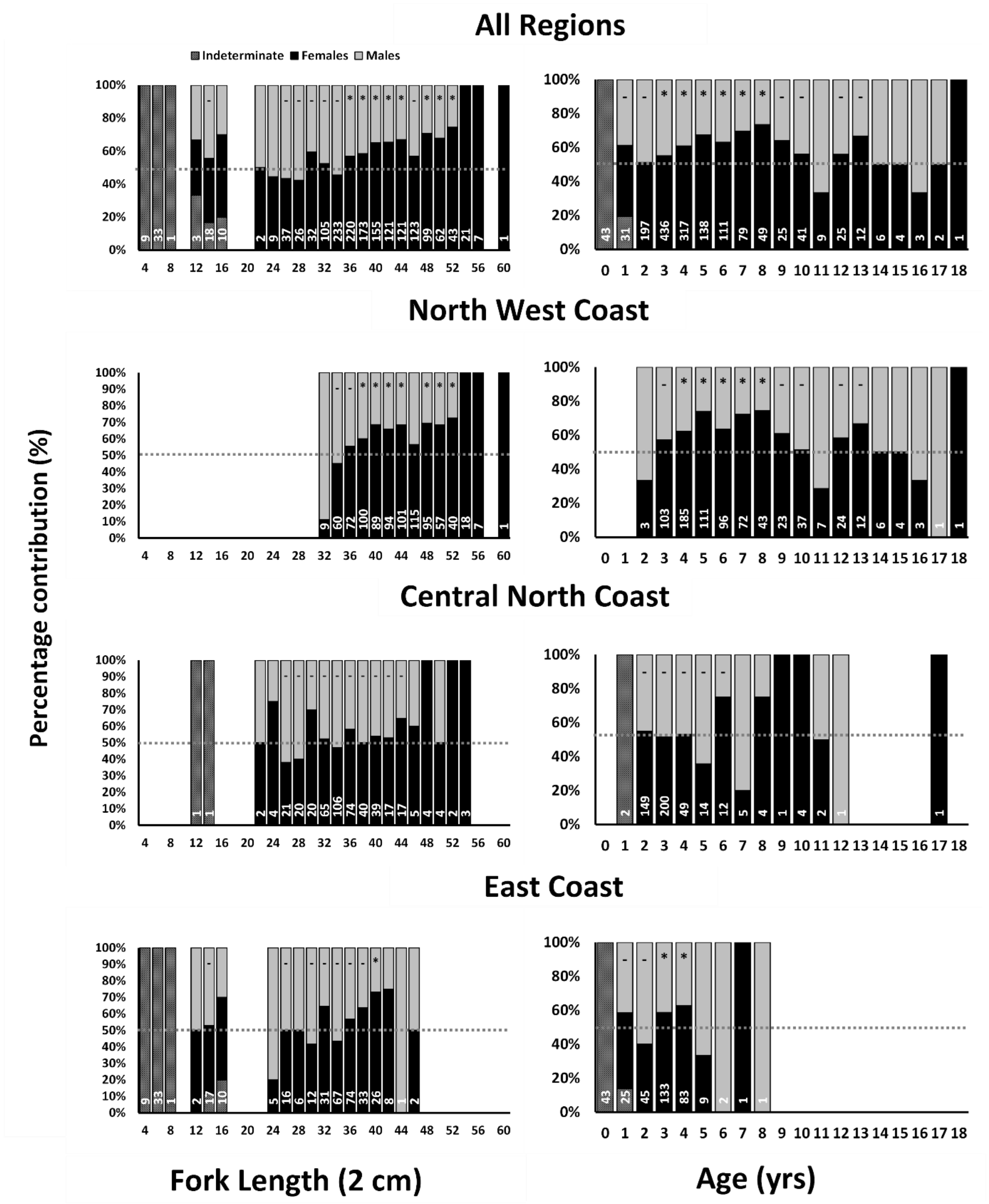

Figure 44. Spatial comparisons of the percentage contribution of indeterminate, male and female King George Whiting (*Sillaginodes punctatus*) from the North-West, Central-North and East Coasts of Tasmania in relation to fork length (2 cm) and age. Note: Data pooled across sampling years (2012–2021). *Indicate a significant difference (p < 0.05) of the male: female ratio away from 1:1, "-" indicates χ^2 square test failed to detect a difference, values without annotation were not tested due to low n values (< 10) as indicated in white.

The oldest female and male fish were estimated to be 18.8 years and 17.1 years old respectively. Age frequencies reflected marked differences between regions, with fish ≥ 5 years of age typically coming from the NWC (89% of n = 542) and the CNC and EC consisting mainly of fish aged between 2 and 4 years of age. In contrast to the dominance of females in the larger size classes, females were not proportionately dominant in older age classes. In addition, sex ratio bias with age was only detected in the younger age classes between 3 and 8 years of age.

Spatially pooled inter annual age frequencies demonstrated the progression of multiple strong cohorts born in the years 2001, 2003, 2007, 2014, 2015 and 2017 (Figure 45). There was a marked increase in young fish aged 2–5 years in 2017 and onwards due to the increased collection of recreational samples and a lack of older fish ≥ 5 years due to reduced commercial donations for 2018. The 2014 and 2015 birth year cohorts were relatively abundant across all three regions, however the recently strong 2017 recruits were not relatively abundant in catches from the NWC (n = 3). In addition to these observed strong year classes there is also evidence of weak recruitment occurring in 2002 (n = 1), 2006 (n = 1), 2016 (n = 9) and 2018 (n = 2).

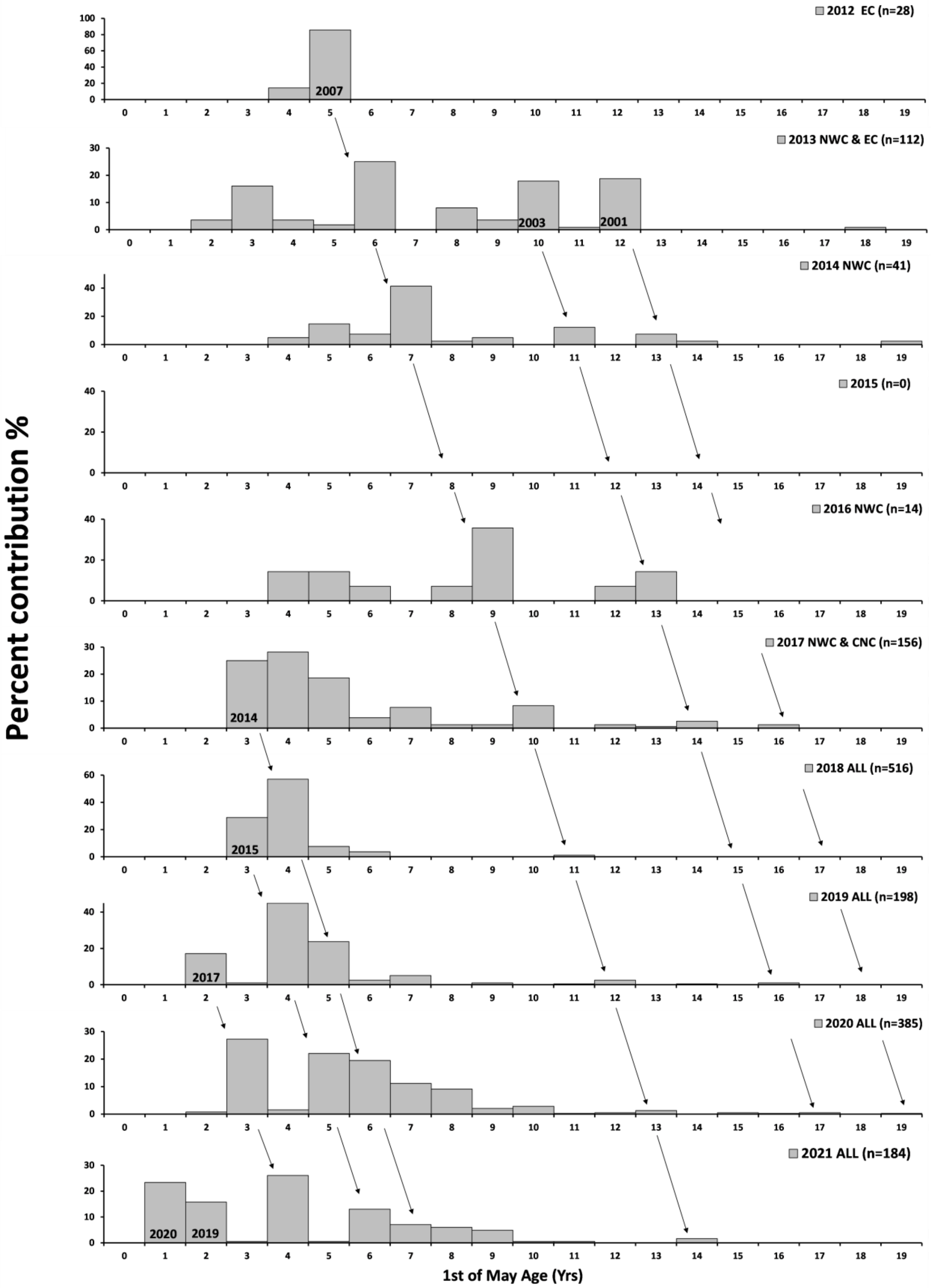

Figure 45. Interannual age frequency variations of King George Whiting (*Sillaginodes punctatus*) from Tasmania between 2012 and 2021. Years in column bases are the back calculated year of birth with lines indicating the progression of that birth year. Note—regional sampling was not consistent across all years, and age data was not available for 2015.

Size structure and mortality estimation

Total mortality estimates between gender could only be examined from the NWC or with all regions pooled. In both cases gender-specific total mortality rates were not significantly different (NWC: $F = 2.65_{1,24}$, $P = 0.09$; Regions pooled: $F = 0.77_{1,26}$, $P = 0.48$, Slopes test NWC: $t = 1.75_{1,22}$, $p = 0.09$; Slopes test Regions pooled: $t = 1.18_{1,24}$, $p = 0.25$). Similarly, mortality rates of pooled sexes between all regions and the NWC were also not different ($F = 0.95_{1,27}$, $P = 0.40$). Due to these results whole stock total mortality rates (age range: 3–19 years) were estimated to be 0.37 yr^{-1} (R^2 0.97) for the age-based catch curve method and 0.45 yr^{-1} (R^2 0.96) when directly fitting an exponential decay curve to the pooled age frequency data (Figure 46). Natural mortality estimates were variable depending on the estimation method used. Hoenig$_{nls}$ estimator produced the highest estimate of natural mortality with Tanka's (1960) 5% survival producing the lowest estimate (Table 14). These natural mortality estimates indicate that fishing mortality may range from 0.04 to 0.21 yr^{-1} depending on the total mortality and natural mortality estimates used. Given these values, the exploitation ratio is likely to range between 0.1 and 0.57.

Table 14. Total and Natural Mortality estimates for Tasmanian King George Whiting (*Sillaginodes punctatus*).

Estimation technique	**Mortality estimate**
Total Mortality (Z)	
Z age-based catch curve	0.37
Z direct fit of exponential decay curve	0.45
Natural Mortality (M)	
M Hoenig est	0.33
M Pauly's method mean size @ age (15.94°C)	0.26 f; 0.31 m
M Pauly's method decimal age (15.94 °C)	0.26 f; 0.32 m
M 5% survival to T_{max}	0.16
M 1% survival to T_{max}	0.24

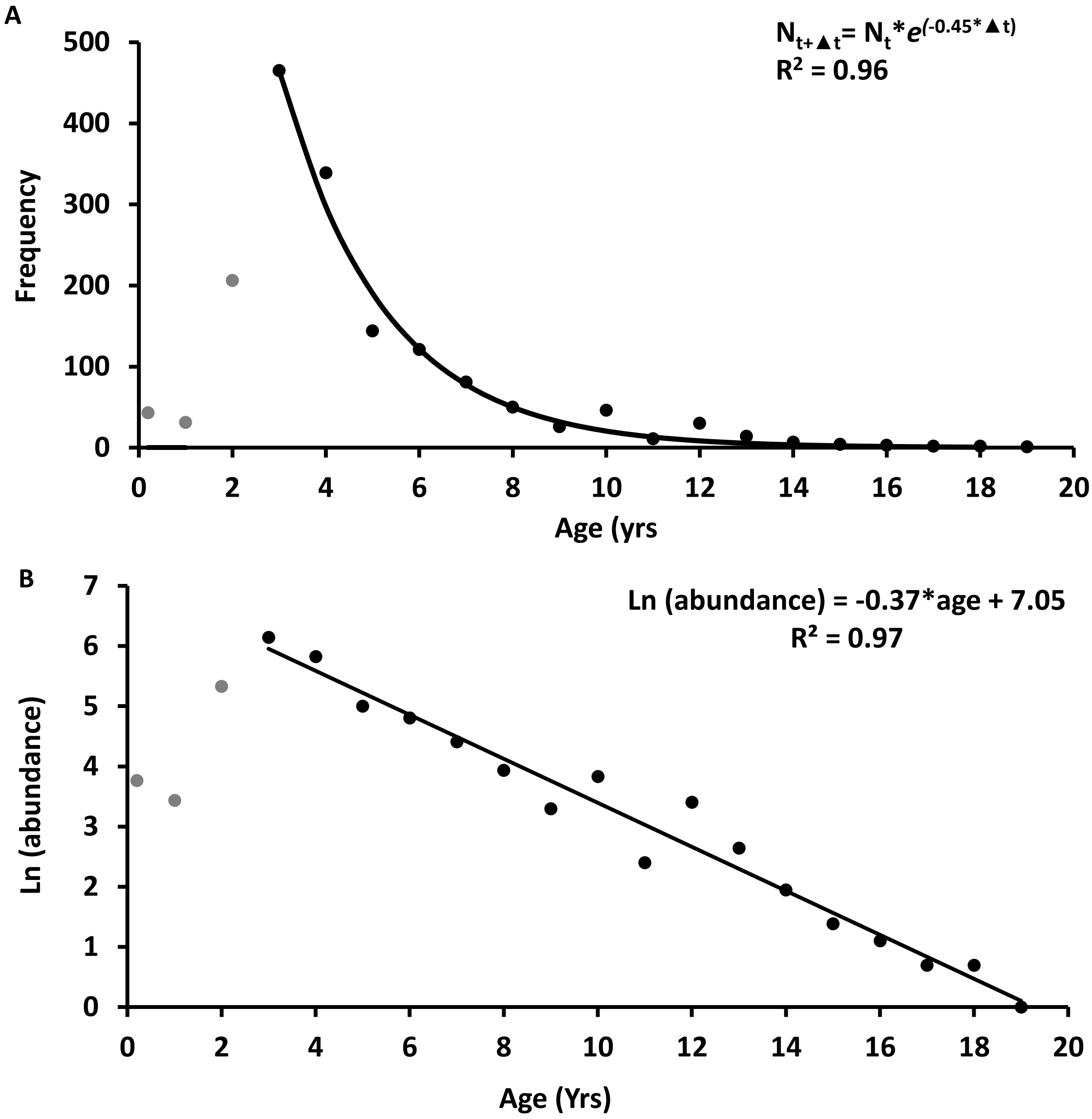

Figure 46. (A) Whole stock age-frequency with a fitted exponential decay curve and (B) age-based catch curve for King George Whiting (*Sillaginodes punctatus*). Black markers represent data selected for model fitting. Grey markers were excluded from model fit.

Growth

Regional and sex-specific von Bertalanffy growth models and parameters, and gender-specific statistical tests on model comparisons are presented in Figure 47 and Table 15. Significant differences in growth rates between sexes were detected from all regions except the EC when assessing models fitted to the mean size at age data. These differences were generally attributable to female models having larger values for the L_∞ parameter. Gender-specific differences in the other two von Bertalanffy growth parameters were less consistent and only detected when modelling decimal age data from pooled regions (k and t_0) and for the CNC (k only). In contrast, pairwise comparisons between mean size at age indicated that gender-based differences in growth could be detected between the ages of three and four years. When sufficient data was available, Hedges G values indicated that the magnitude of this difference increased with increasing age. Significant differences in sex-specific mean size at age could also be detected between regions. In the age ranges of 2–5 years where reliable pairwise comparisons of mean size at age could be performed there was a consistent trend that both female and male fish from the CNC were larger than those from the NWC, and EC respectively

(Figure 48). It is important to note that due to minimum legal size requirements fish from the NWC are likely to have an upward size bias for fish aged between 2 and 3 years old. This differential size at age could be detected by 2 years in females and 3 years in males between the CNC and EC, and by four years of age in both sexes between the CNC and NWC. ARSS on regional sex-specific growth models highlighted significant differences in base-case growth models between regions (Table 16). Pairwise post hoc likelihood ratio tests using unconstrained decimal age data, suggested all three von Bertalanffy parameters contributed to these differences for all regions tested. However, when tests were performed on mean size at age or on standardized age ranges with refitted models, several of these effects were lost.

Table 15. Regional comparisons of gender specific von Bertalanffy growth curve parameters of King George Whiting (*Sillaginodes punctatus*) using mean size at age, and individual size at decimal age data. Asterisks on regions indicate a significant ARSS result between sexes. Asterisks between values are post hoc likelihood ratio tests indicating which parameters were significantly different at $p < 0.05$ between sexes.

Mean size at integer age von Bertalanffy growth model parameters

	Regions pooled*		North-West Coast*		Central-North Coast*		East Coast	
ARSS	$F = 10.05_{1,29}$, $P < 0.001$		$F=16.88_{1,26}$, $P < 0.001$		$F=5.85_{1,14}$, $P=0.008$		$F = 0.48_{1,7}$, $P = 0.708$	
	Females	**Males**	**Females**	**Males**	**Females**	**Males**	**Females**	**Males**
L_∞ (mm)	560	504*	606	521*	560	534	485	471
k (yrs^{-1})	0.27	0.65	0.13	0.21	0.21	0.22	0.46	0.44
t_0 (yrs)	-0.50	-0.20	-4.36	-2.60	-2.03	-2.00	0.12	0.01
n	18	17	16	16	11	9	6	7
R^2	0.96	0.98	0.91	0.97	0.99	0.99	0.99	0.97

Individual size at decimal age VBGF model parameters

	Regions pooled*		North-West Coast*		Central-North Coast*		East Coast*	
ARSS	$F = 50.48_{1,1474}$, $P < 0.001$		$F = 67.78_{1,725}$, $P < 0.001$		$F = 6.24_{1,435}$, $P = 0.001$		$F = 3.29_{1,293}$, $P = 0.022$	
	Females	**Males**	**Females**	**Males**	**Females**	**Males**	**Females**	**Males**
L_∞ (mm)	538	492	566	514	533	487	456	426
k (yrs^{-1})	0.30	0.38	0.22	0.26	0.31	0.41	0.55	0.64
t_0 (yrs)	0.10	0.37	-0.98	-0.63	-0.12	0.33	0.99	1.04
n	873	607	466	265	235	206	165	134
R^2	0.90	0.89	0.87	0.89	0.85	0.84	0.92	0.91

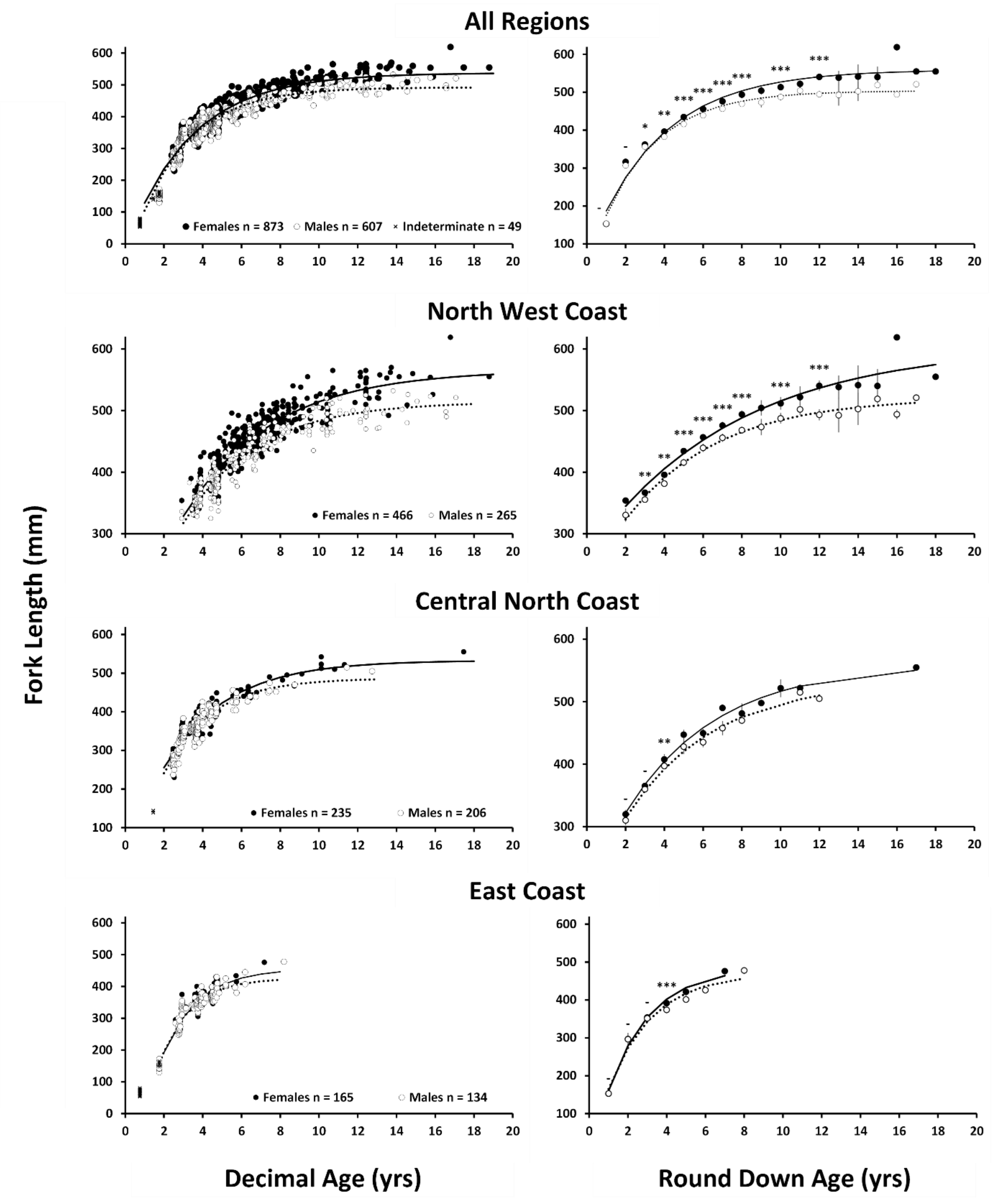

Figure 47. Sex-specific length at age and von Bertalanffy growth models of King George Whiting (*Sillaginodes punctatus*) caught on the North-West, Central-North and East Coasts of Tasmania. Solid black lines and dotted lines represent models for females and males respectively. Paired means without annotation above were excluded from *t*-tests due to low sample size, "-" denotes a *t*-test was performed but no difference was detected. Asterisks indicate significant difference between means with the number of asterisks indicating Cohen's D effect size of small *, medium ** and large *** respectively. Error bars are 95% confidence intervals.

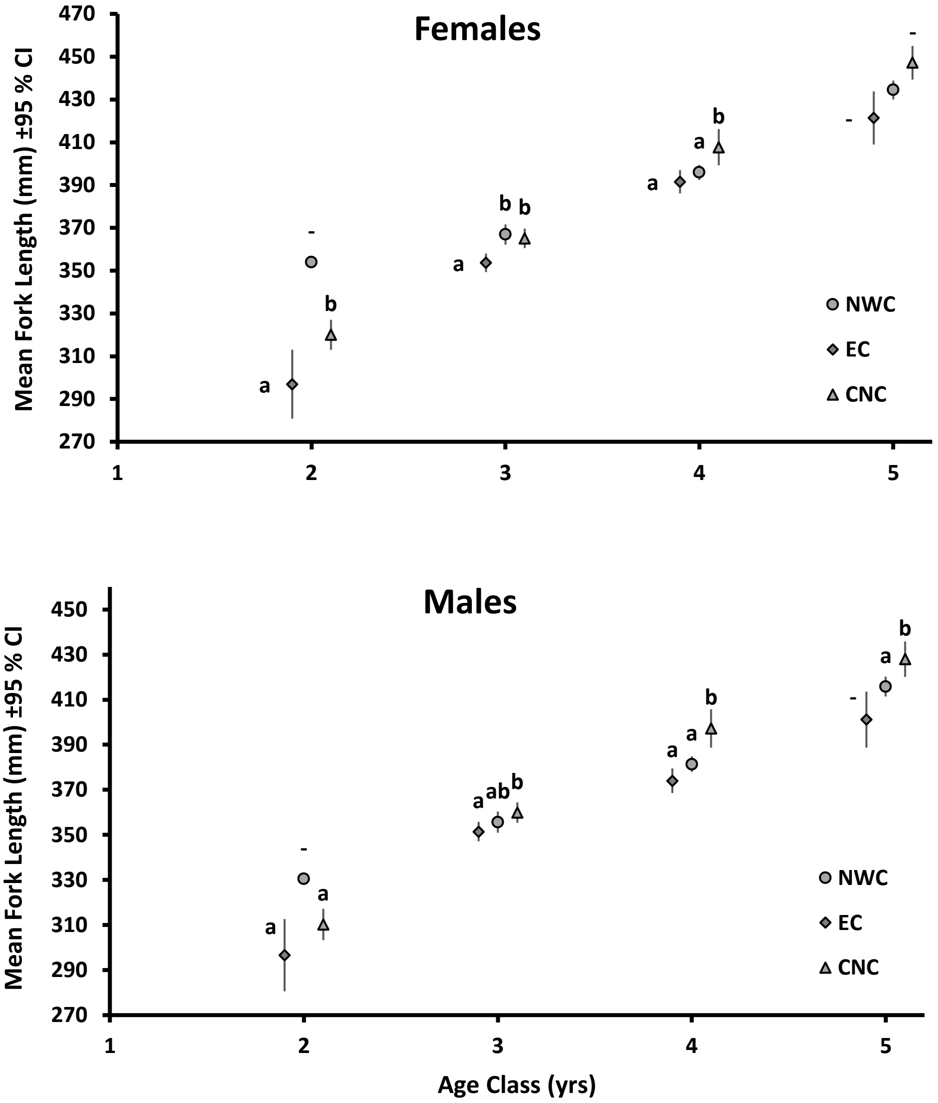

Figure 48. Regional comparisons of mean size at age of female and male King George Whiting (*Sillaginodes punctatus*) between the ages of 2 and 5. Error bars are ± 95% confidence intervals. Values that share common letters within the specific age class are not significantly different (*t*-tests, $p < 0.05$). Mean size at age values with "-" were not included in statistical tests due to low sample size.

Table 16. Statistical tests results on sex-specific growth models between regions for King George Whiting (*Sillaginodes punctatus*). ARSS tested for differences in growth models between regions (3-way comparison). Likelihood ratio tests (LRTs) were used as post hoc pairwise comparisons, significant findings are in bold. Pairwise *t*-test were performed on data pairs between 2 and 5 where sample numbers were ≥ 10. Asterisks indicate significant differences between mean size at age with the number of asterisks indicating Hedges G effect sizes of small *, medium ** and large ***. Values without annotation were not significantly different.

Females

Mean size at integer age models

	Unconstrained age range comparison			Standardised age range comparison		
ARSS	$F=6.83_{1,24}$, $P=0.002$			$F=1.03_{1,5}$, $P=0.454$ (3–7 yrs)		
LRT	NWC v CNC	CNC v EC	NWC v EC	NWC v CNC	CNC v EC	NWC v EC
Age range	All ages			3–11	2–7	3–7
L_∞	0.099	**0.004**	**0.012**	0.675	0.990	0.330
k	0.083	**0.001**	**0.001**	0.688	0.961	0.386
t_0	0.081	**0.000**	**0.000**	0.806	0.746	0.606
Pairwise *t*-tests on mean size at age where $n > 10$				3,4**	2**,3**,4**	3**,4

Individual size at decimal age models

	Unconstrained age range comparison			Standardised age range comparison		
ARSS	$F = 42.12_{1,857}$, $P < 0.001$			$F = 7.27_{1,637}$, $P < 0.001$ (3–7 yrs)		
LRT	NWC v CNC	CNC v EC	NWC v EC	NWC v CNC	CNC v EC	NWC v EC
Age range	All ages			3–11	2–7	3–7
L_∞	**0.000**	**0.000**	**0.000**	0.453	0.447	0.222
k	**0.003**	**0.000**	**0.000**	0.452	0.288	0.124
t_0	**0.008**	**0.000**	**0.000**	0.588	0.488	**0.048**

Males

Mean size at integer age models

	Unconstrained age range comparison			Standardised age range comparison		
ARSS	$F = 9.65_{1,23}$, $P < 0.001$			$F = 6.08_{1,8}$, $P = 0.019$ (3–8 yrs)		
LRT	NWC v CNC	CNC v EC	NWC v EC	NWC v CNC	CNC v EC	NWC v EC
Age range	All ages			3–7	2–8	3–8
L_∞	0.325	**0.030**	**0.027**	0.127	**0.006**	**0.001**
k	0.757	**0.017**	**0.002**	0.294	**0.005**	**0.001**
t_0	0.382	**0.004**	**0.000**	0.302	**0.016**	**0.002**
Pairwise *t*-tests on mean size at age where $n > 10$				3,4***,5***	2,3*,4***	3,4

Individual size at decimal age models

	Unconstrained age range comparison			Standardised age range comparison		
ARSS	$F = 42.12_{1,857}$, $P < 0.001$			$F = 7.27_{1,637}$, $P < 0.001$ (3–7 yrs)		
LRT	NWC v CNC	CNC v EC	NWC v EC	NWC v CNC	CNC v EC	NWC v EC
Age range	All ages			3–11	2–7	3–7
L_∞	**0.029**	**0.000**	**0.000**	0.797	0.866	0.691
k	**0.001**	**0.001**	**0.000**	0.994	0.333	0.649
t_0	**0.003**	**0.000**	**0.000**	0.691	0.401	0.581

Reproductive biology

There was a clear relationship between reproductive stage and mean GSIs for females and males with GSI values being low < 1.36% in reproductively inactive and spent fish and higher in fish that were reproductively active (Figure 49). The smallest reproductively active female (vitellogenic) and male (spermatogenic) specimens measured 340 and 320 mm FL respectively and were both 2.9 years of age. Size and age at 50% maturity were estimated to be 407 and 403 mm FL, and 4.4 and 4.6 years for females and males respectively. Fish that possessed gonadal stages indicating imminent spawning (hydrated or spermiated) were relatively rare (hydrated *n* = 6; FL range 420–560 mm; spermiated *n* = 1; FL = 362 mm). Although a female GSI value as high as 15.5% was recorded, only 23% of the reproductively active females had GSI values above 6%. Male testes were markedly smaller than female ovaries with a peak GSI value of 2.1% and over all very few individuals with GSI values greater than 1.5%.

Seasonal changes in the proportions of reproductive stages and mean GSI values of "maturing adults" fish (≥ 3 years) were markedly different between regions (Figure 49, Figure 50). Although female GSI profiles did not display a clear seasonal pattern on the EC or CNC and that reproductively active females were relatively rare (EC *n* = 6; CNC *n* = 7), they were present from March–April on the EC and February–April on the CNC. Despite no hydrated fish being encountered on the EC or CNC, the only fully spermiated male encountered had a GSI value of 1.3% and was captured in May on the EC. In addition, two fish classified as spent were present in August from the CNC. In contrast to the EC and CNC, female GSI profiles from the NWC displayed a clearer seasonal pattern of reproductive activity with an increasing GSI in February, a peak value in March before a decline to low values by June. Reproductively active fish were present between February and June with hydrated females present between February to May and spent fish appearing in May and disappearing in September. An important sampling characteristic to note is that despite attempts, no samples were obtained in April from the NWC. In addition, only 1 fish ≥ 4 years of age with an April capture data was recorded, from the Tamar River on the CNC. In contrast to the female reproductive trends, male GSI profiles in all three regions displayed clear seasonal variations with peak GSI values in February, March and April for the NWC, CNC and EC respectively (Figure 50, Figure 51). These trends could be associated with increased numbers of spermatogenic males between February and May.

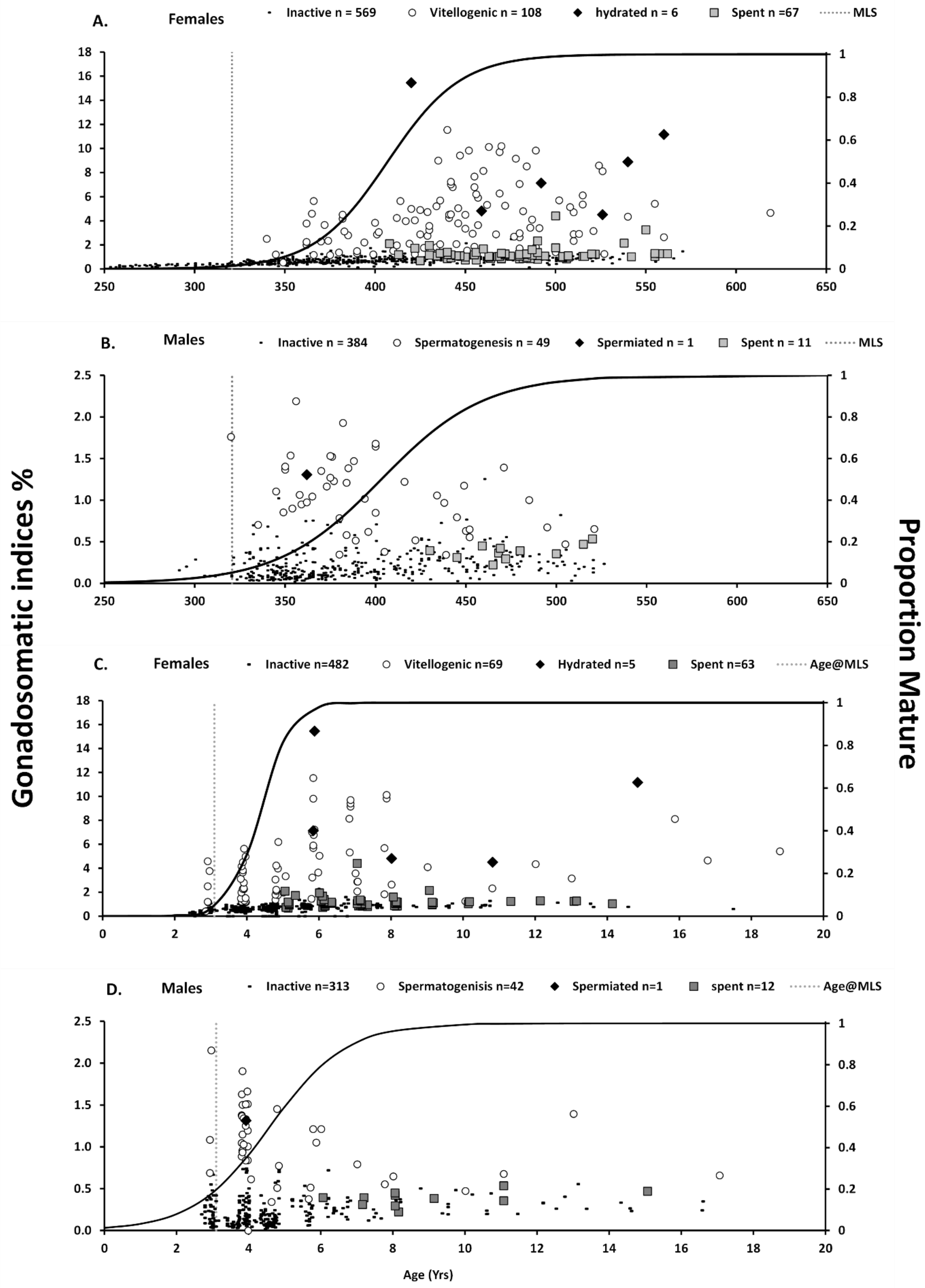

Figure 49. Sex-specific scatter plot relationships of fork length (A & B) and age (C & D) in relation to gonadal-stage and gonadosomatic indices (GSI) for King George Whiting (*Sillaginodes punctatus*) from Tasmania. Broken grey lines represent the modelled fork length and age at the current Tasmanian minimum legal size (MLS) of 35 cm total length. Solid black lines are the size and age at maturity models.

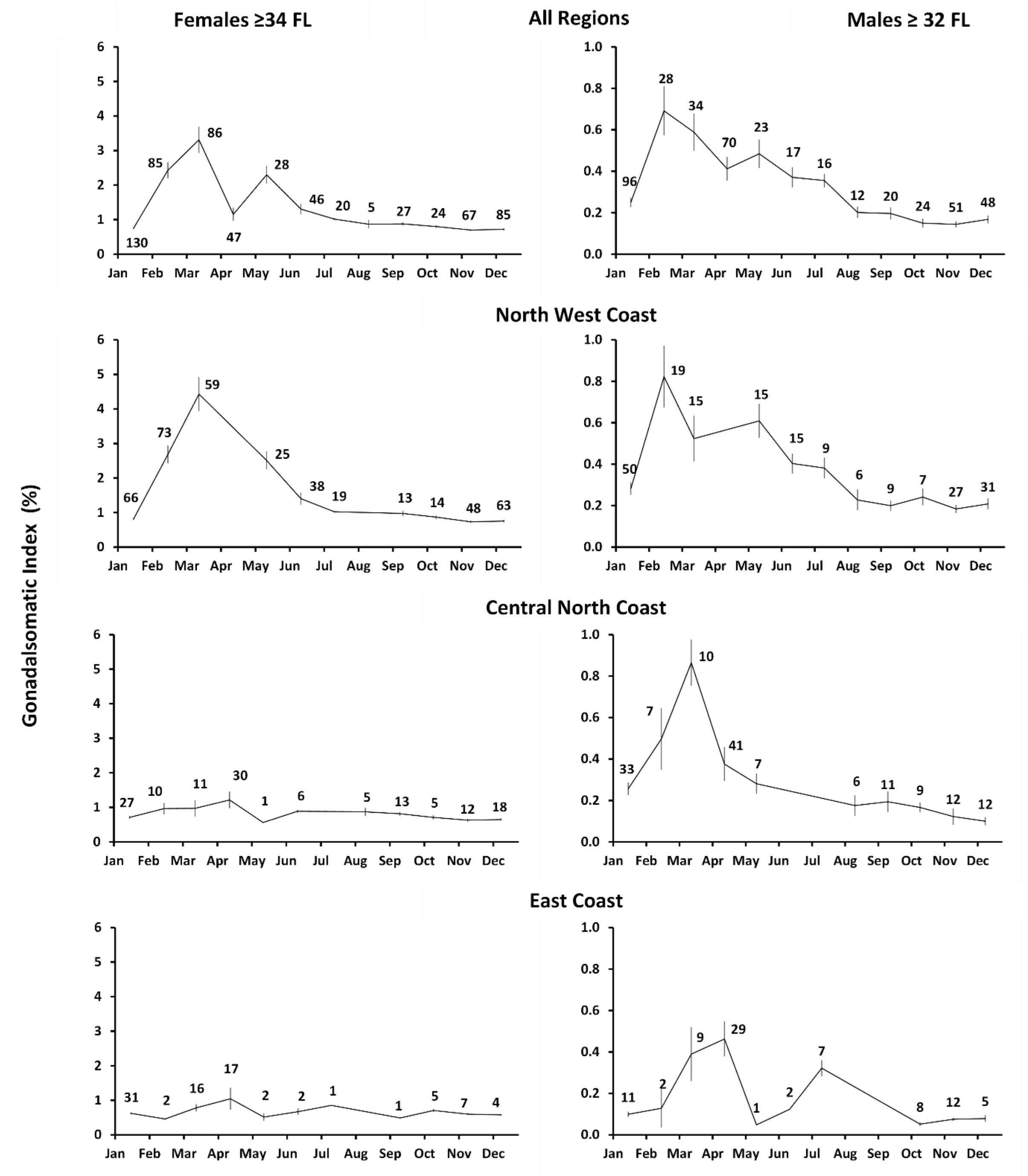

Figure 50. Seasonal changes in mean monthly gonadosomatic indices (GSI ± *SE*) of female and male King George Whiting (*Sillaginodes punctatus*). Data has been pooled across years, sampling sites (northern coast pooled) and constrained to be from the smallest observed mature female (≥ 34 cm FL) and male (≥ 33 cm FL). Sample sizes (*n*) are indicated above error bars.

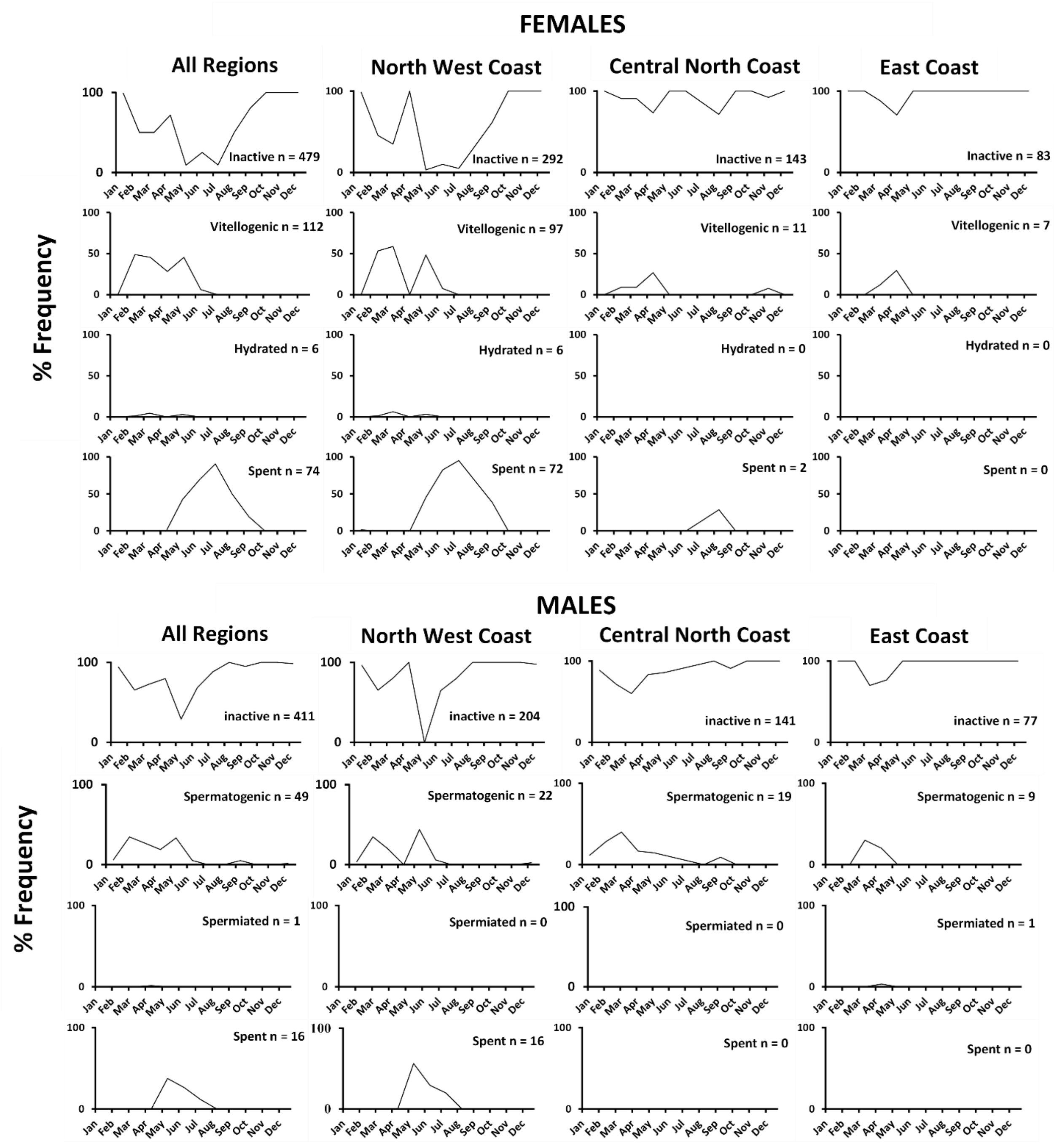

Figure 51. Annual variations in macroscopic stage frequency percentages of female and male King George Whiting (*Sillaginodes punctatus*). Data constraints are captioned in Figure 50.

Diet

Prey taxa were identified in 154 King George Whiting stomachs from eight sites across north and north-east Tasmania (Figure 52). A mean (± *SE*) of 1.81 ± 0.07 distinct taxa per King George Whiting stomach were identified (range: 1–4). Class was the lowest taxonomic level with high resolution, with identified prey taxa representing nine taxonomic classes (Table 17). The prey accumulation curve analysis estimated an asymptotic class richness of 9.6 ± 0.7 classes, demonstrating sampling provided high coverage of King George Whiting diet in the region at this taxonomic level (Figure 53).

Table 17. Prey taxa identified in King George Whiting (*Sillaginodes punctatus*) stomach contents by the frequency of occurrence (%FO) and mean (± *SE*) proportion of stomach contents (% Contents). Taxa named 'Other' are stomach contents that could not be resolved to a finer taxonomic level.

Prey Taxon	%FO	% Contents
Bivalvia	**3.90%**	**0.55 ± 0.26%**
Pectinidae	0.65%	0.16 ± 0.16%
Theora lubrica	0.65%	0.03 ± 0.03%
Tellinidae	0.65%	0.06 ± 0.06%
Other Bivalvia	1.95%	0.29 ± 0.19%
Cephalopoda	**0.65%**	**0.19 ± 0.19%**
Octopoda	0.65%	0.19 ± 0.19%
Gastropoda	**9.09%**	**5.53 ± 1.63%**
Sinum sp.	0.65%	0.13 ± 0.13%
Naticidae	1.30%	0.39 ± 0.36%
Littorinimorpha	0.65%	0.58 ± 0.58%
Other Gastropoda	1.30%	1.04 ± 0.76%
Stomatella impertusa	1.95%	1.04 ± 0.73%
Fissurellidae	3.25%	2.35 ± 1.1%
Holothuroidea	**8.44%**	**6.06 ± 1.78%**
Malacostraca	**55.19%**	**32.71 ±**
Amphipoda	7.79%	1.4 ± 0.77%
Decapoda	49.35%	29.38 ± 3.22%
Brachyura	11.04%	5.3 ± 1.52%
Dromeiidae	0.65%	0.62 ± 0.62%
Grapsidae	0.65%	0.32 ± 0.32%
Majidae	0.65%	0.42 ± 0.42%
Other Brachyura	9.09%	3.94 ± 1.31%
Caridea	3.90%	2.32 ± 1.14%
Alpheidae	1.95%	0.73 ± 0.65%
Alpheus sp.	1.30%	0.68 ± 0.65%
Other Alpheidae	0.65%	0.05 ± 0.05%
Other Caridea	1.95%	1.59 ± 0.95%
Paguroidea	0.65%	0.16 ± 0.16%
Thalassinidea	36.36%	21.6 ± 2.91%
Callianassidae	35.06%	20.31 ± 2.82%
Other	1.30%	1.3 ± 0.92%
Isopoda	3.90%	1.12 ± 0.73%
Stomatopoda	1.95%	0.81 ± 0.58%
Polychaeta	**68.18%**	**52.08 ±**
Pectinariidae	0.65%	0.65 ± 0.65%
Terribellidae	0.65%	0.65 ± 0.65%
Other Polychaeta	66.23%	50.27 ± 3.52%
Polyplacophora	**1.30%**	**0.44 ± 0.34%**
Chitonida	1.30%	0.44 ± 0.34%
Seagrass	**16.23%**	**1.16 ± 0.5%**
Sipuncula	**1.30%**	**1.27 ± 0.89%**

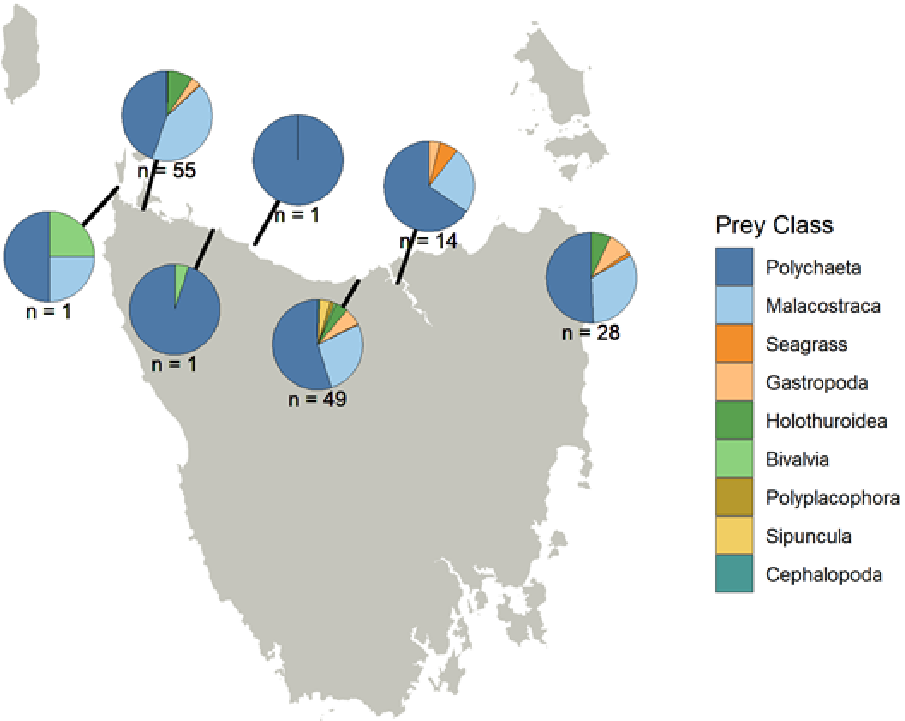

Figure 52. Mean proportional King George Whiting (*Sillaginodes punctatus*) stomach contents by prey taxa class and sample sizes (*n*) of stomachs that contained prey, by region of collection (from left to right: Woolnorth, Montagu, Rocky Cape, Wynyard, Port Sorell, Tamar, St. Helens).

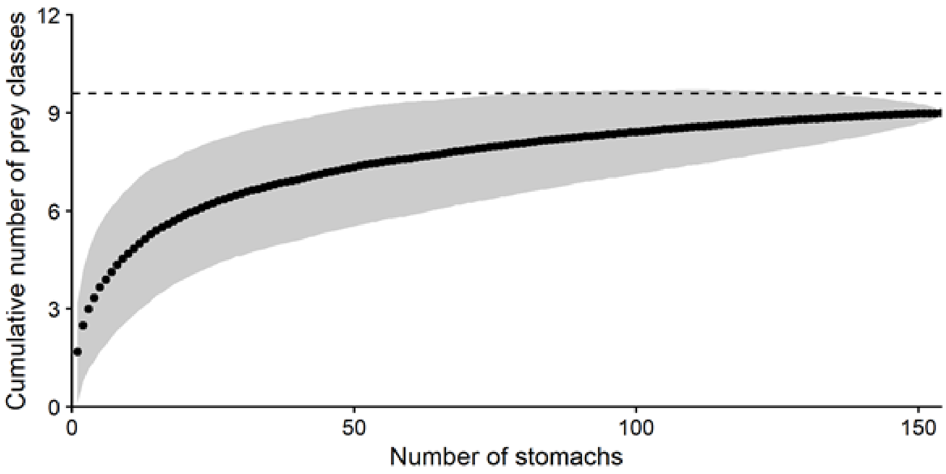

Figure 53. King George Whiting (*Sillaginodes punctatus*) prey taxa accumulation curve estimated from prey taxon presence in 1000 random permutations of stomach samples. Dotted line indicates bootstrap estimated asymptotic number of prey classes of Snapper in the sampled region.

Polychaetes, and malacostracans (non-barnacle crustaceans), were the most dominant prey classes among King George Whiting stomachs by both frequency of occurrence and mean proportion, together constituting a mean proportion of 84.80% of King George Whiting prey (Figure 54). Seagrass was present in 16.23% of King George Whiting stomachs, but only constituted a mean of 1.16 ± 0.50% of stomach contents by weight. Resolution of polychaete prey was largely limited to class level, while malacostracans taxa largely consisted of Callianassid ghost shrimp (35.06% of stomachs), crabs (11.04%) and amphipods (7.79%).

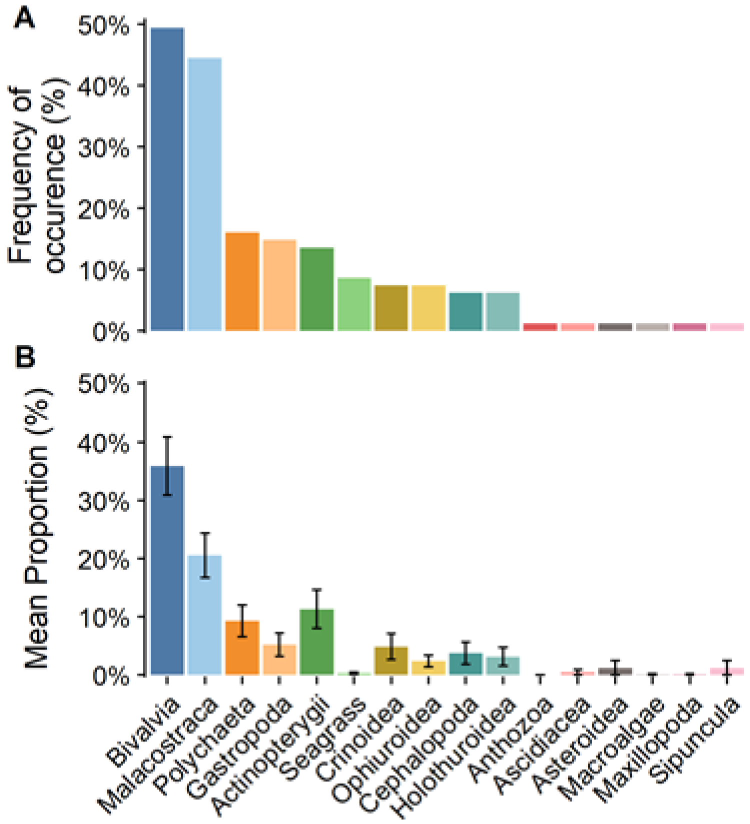

Figure 54. King George Whiting (*Sillaginodes punctatus*) prey class by mean proportion (%) of individual stomach contents (A) and frequency of occurrence (%) across King George Whiting stomachs (B). Error bars are one *SE*.

Occurrence of bivalves and holothurioids (sea cucumbers) in King George Whiting stomachs were significantly but weakly correlated ($r_{phi} = 0.180$; Figure 55), as of the six stomachs which contained bivalves, two also contained sea cucumbers. Occurrence of polychaetes in King George Whiting stomachs was significantly negatively correlated with occurrence of holothuroids ($\mathrm{r}_{\mathrm{phi}} = -0.194$), sipunculids ($r_{phi} = -0.168$), and malacostracans ($r_{phi} = -0.335$). This suggests the dominant prey classes malacostracans (largely ghost shrimp) and polychaetes, which occurred in 144 of 159 total King George Whiting stomachs, have limited overlap in habitat or are foraged selectively. Permutational ANOVA revealed neither fork length ($p = 0.138$) of individual King George Whiting or region of capture (North/East coast, $p = 0.909$) had discernible effects on prey occurrence in stomachs (Table 18).

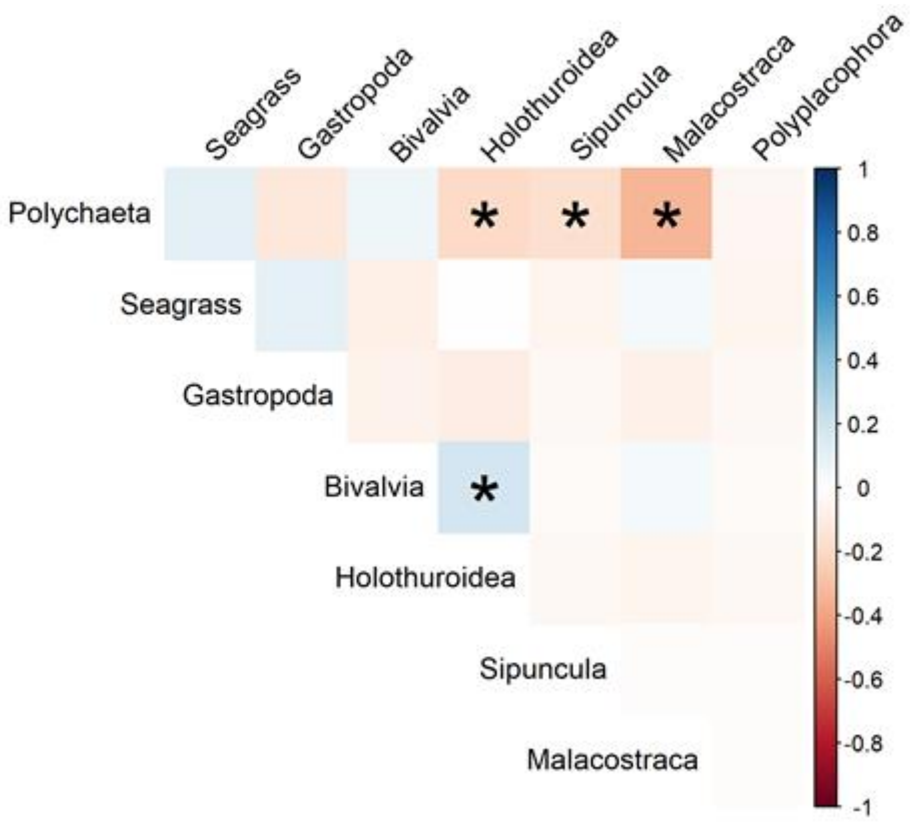

Figure 55. Correlation of occurrence of prey classes among King George Whiting (*Sillaginodes punctatus*) stomachs. Asterisks indicate significant correlations between prey classes. Cephalopoda, which was only identified in one King George Whiting stomach, was excluded.

Table 18. Permutational ANOVA model summary for Bray-Curtis pairwise dissimilarity between prey class occurrence in King George Whiting (*Sillaginodes punctatus*) stomachs. FL = Fork length (mm), Region = North or East coast.

Model: Region + FL + Region:FL	***df***	***SS***	***R^2***	***F***	**Pr(> *F*)**
Region	6	0.6057	0.0202	0.4982	0.901
FL	1	0.0699	0.0023	0.3448	0.708
Region:FL	3	1.3675	0.0456	2.2497	0.035
Residual	138	27.9612	0.9319		
Total	148	30.0042	1.0000		
Model: Region + FL	***df***	***SS***	***R^2***	***F***	**Pr(> *F*)**
FL	1	0.4080	0.0136	1.9613	0.157
Region	6	0.2676	0.0089	0.2144	0.990
Residual	141	29.3286	0.9775		
Total	148	30.0042	1.0000		

Nonmetric multidimensional scaling reveals occurrence of prey in stomachs is consistent between regions, and on the first ordinal axis appears to be largely driven by the presence of either or both of the negatively correlated two main prey classes, malacostracans and polychaetes, followed by occurrence of secondary prey classes, especially holothurids and gastropods on the second ordinal axis (Figure 56A). The ordinal axes were not significant predictors of fork length, so the smoothed surface was not included on the plot. Mean proportional composition of stomach contents by prey class was similarly consistent between region (Figure 56B) and across individual King George Whiting fork length (Figure 56C).

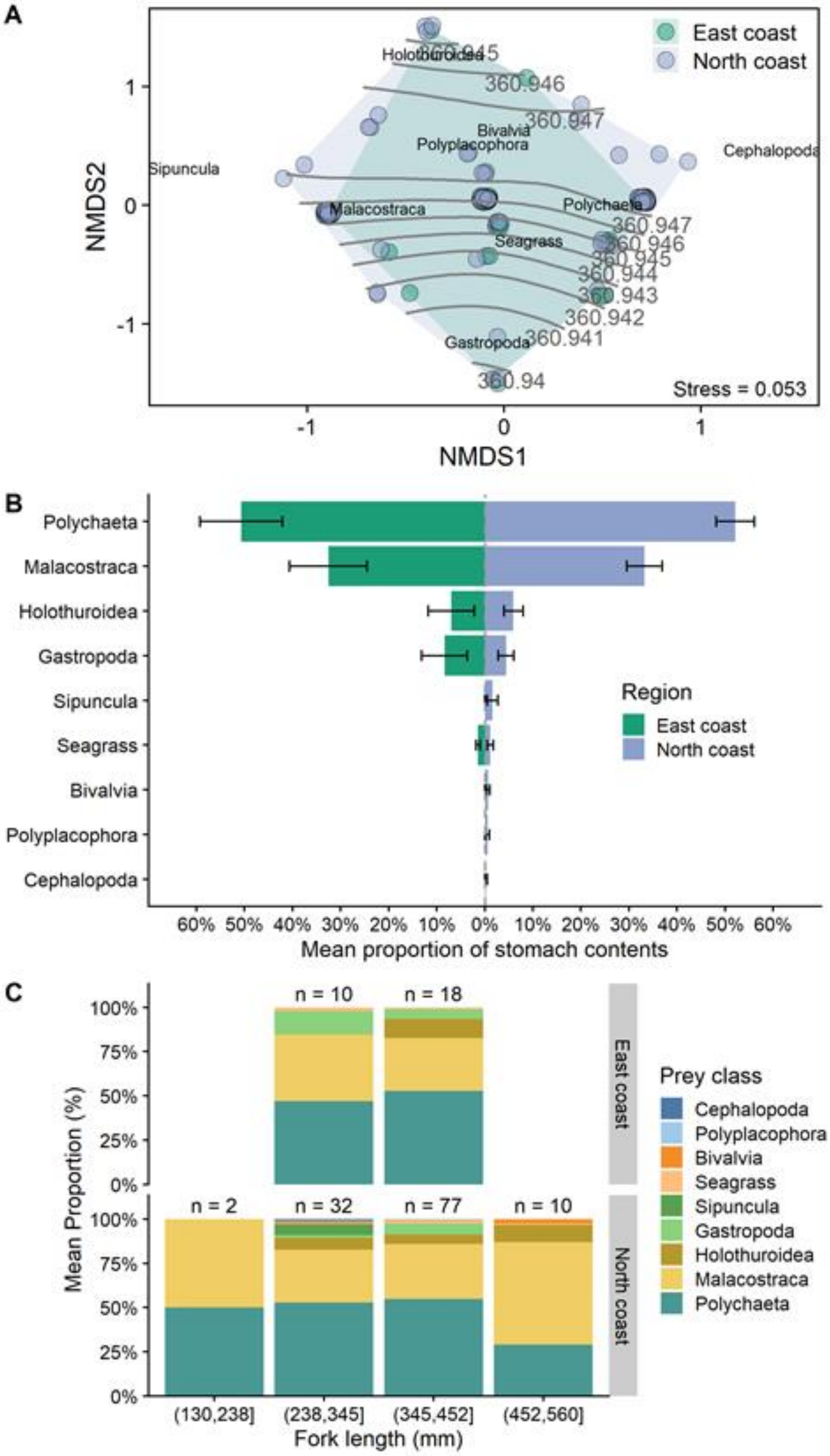

Figure 56. (A) Nonmetric multidimensional scaling scores of King George Whiting (*Sillaginodes punctatus*) stomach (circles) prey class presence dissimilarity, by region of collection, and scores of prey classes in ordinal space. Contour lines represent the smoothed surface of fork length regressed on the ordination axes ($p = 0.814$). (B) Mean proportion of King George Whiting stomach contents, by region of collection. bars represent one *SE*. C. Mean proportion of King George Whiting stomach contents, by binned fork length of individual King George Whiting and region of collection.

Species Distribution Model

Environmental habitat suitability model

The optimal model (based on AIC comparisons) for King George Whiting environment habitat included sea surface temperature (SST: °C), distance to seagrass (m), and depth (m) (Table F2):

$$\text{Response} \sim s(\text{SST}) + \text{depth} + \text{distance to seagrass} + (1|\text{year})$$

Where: Response is the relative probability of King George Whiting occurrence as a function SST, depth and distance to seagrass, ‘s’ denotes a smoothing term.

The effect of SST on King George Whiting occurrence was non-linear and peaked at approximately 18 °C (Figure 57a). Both depth and distance to seagrass were significant linear predictors of King George Whiting occurrence, where the effect on King George Whiting occurrence declining with increasing depth and distance to seagrass (Figure 57b, c, Table 19).

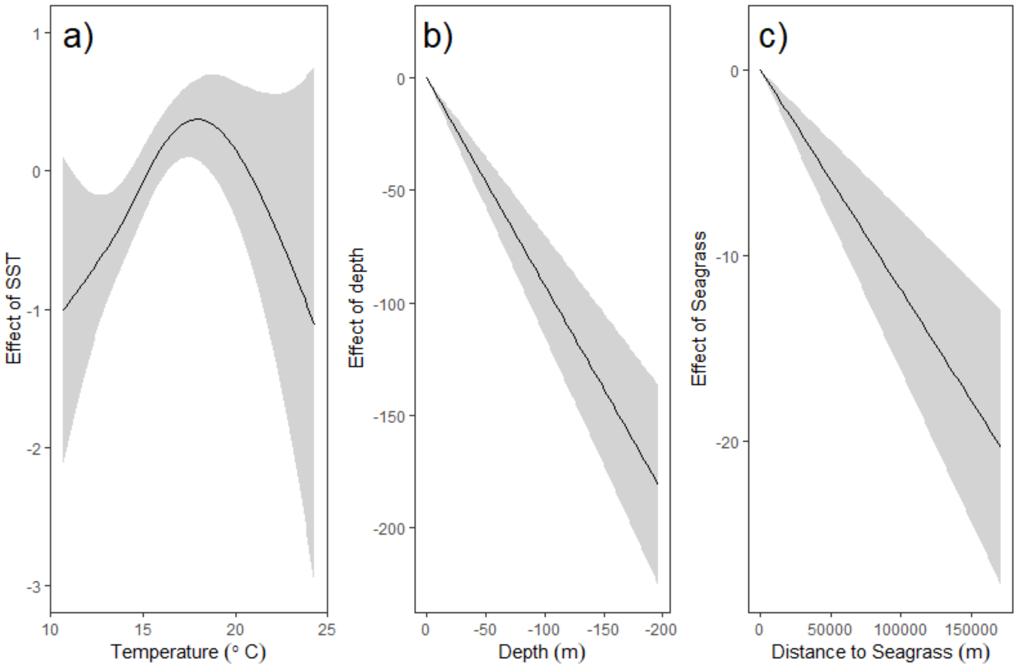

Figure 57. Partial effects of a) sea surface temperature (SST), b) depth c) distance to seagrass on the fitted values of the optimal King George Whiting (*Sillaginodes punctatus*) habitat suitability model ± 95% confidence intervals (shaded in grey).

Table 19. Summary of results for the optimal model for suitable habitat of King George Whiting (*Sillaginodes punctatus*). Smoothing terms are denoted by an ‘s’.

Factor	**Effective degrees of freedom (edf)**	**Coefficient estimate**	***p*-value**
s(SST)	2.461	-0.1759	0.0216*
distance_Seagrass	-	$-1.187\ e^{-4}$	< 0.00001*
depth	-	0.9204	< 0.00001*
$\text{Year}_{\text{intercept}}$	-	2.004	< 0.00001*

Habitat Suitability

Hindcast (1998–2018)

There was no change in habitat suitability throughout the hindcast period (Figure 58, Table F9). Predicted mean habitat suitability was consistently greater in the summer and autumn versus the spring and winter across all regions, ranging from 1.25–1.26 times greater in the autumn and summer in the NEN region to 1.42–1.52 times greater in the autumn and summer in the East region compared to the winter (Figure 58, Table F9, Table F10). The East region yielded the highest mean suitability with an estimated marginal model mean (± *SE*) of 0.53 (± 0.003), 0.62 (± 0.003), 0.61 (± 0.003) and 0.52 (± 0.003) averaged across all years in the spring, summer, autumn and winter respectively (Figure 58).

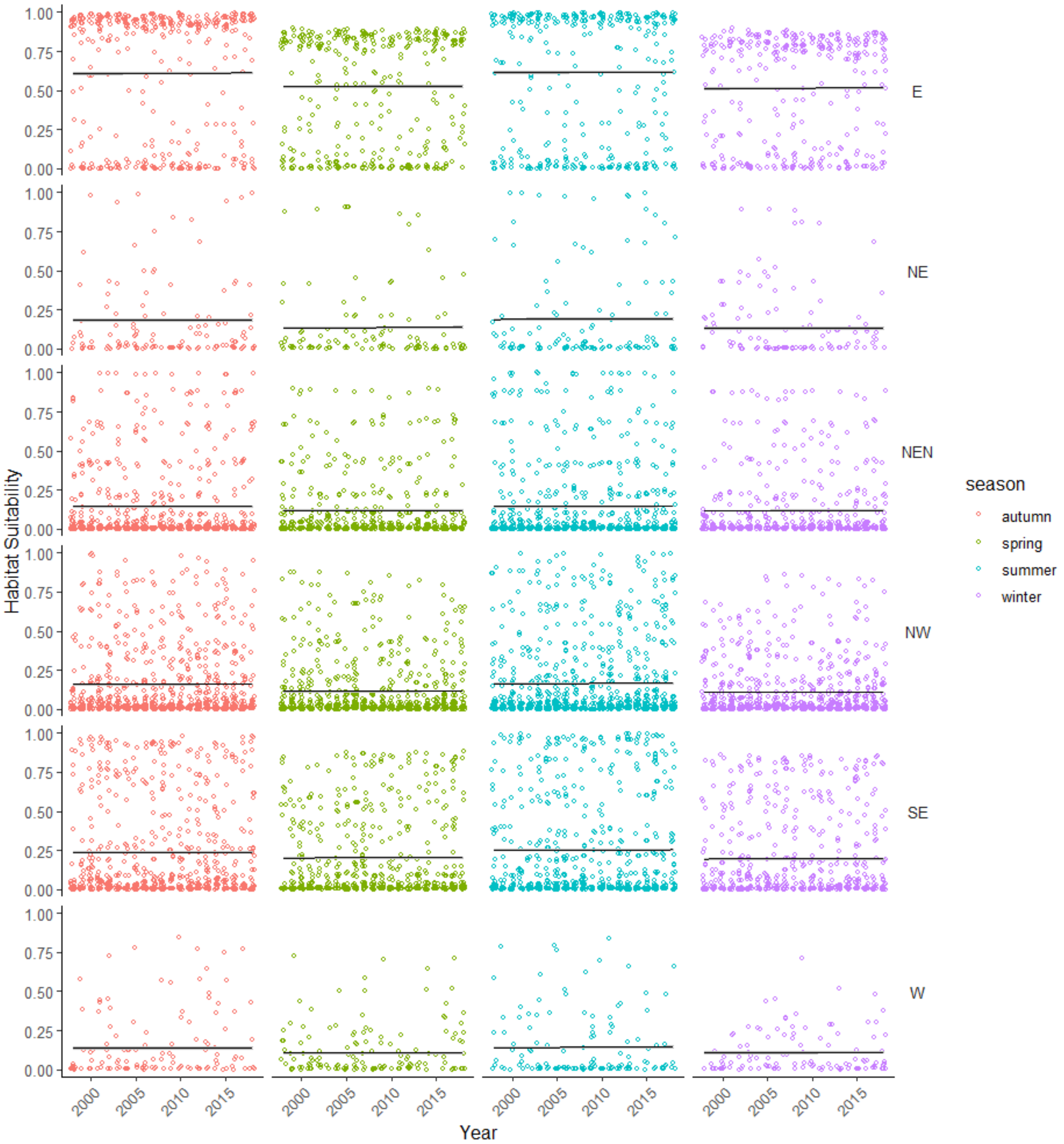

Figure 58. Predicted habitat suitability per year from 1998–2018 of King George Whiting (*Sillaginodes punctatus*) across four seasons and within six regions around Tasmania. Black lines denote result of generalised linear additive models (suitability ~ year + season + region) ± 95% confidence intervals (shaded in grey). NB: Points are a subset of 10,000 out of 975,539 data points for legibility.

Proportional change at a resolution of 0.004°

Forecast (1998–2018 vs 2036–2065)

Proportional change in habitat suitability within each grid cell (0.004° or 416 m^2), throughout each region was greatest in the winter, with an average percent increase (estimated marginal model mean (± *SE*)) ranging from 54.9 (± 0.51) % in the East, to 132 (± 0.43) % in the North-East North (NEN: Figure 59, Figure F9, Table F11). The greatest proportional increase in habitat suitability was 189% in the SE in the winter (Figure 59, Figure 60). There was also increased variance in the proportional change within each region during the winter due to some grid cells increasing in suitability while others remained unchanged (Figure 59, Figures F8–F12). There was also an increase in suitability in the spring across all regions ranging from 20.66 (± 0.50) % in the East to 40.22 (± 0.90) % in the North-East (Figure 59, Figures F10–F11). Change remained low and consistent in the summer and autumn (Figure 59).

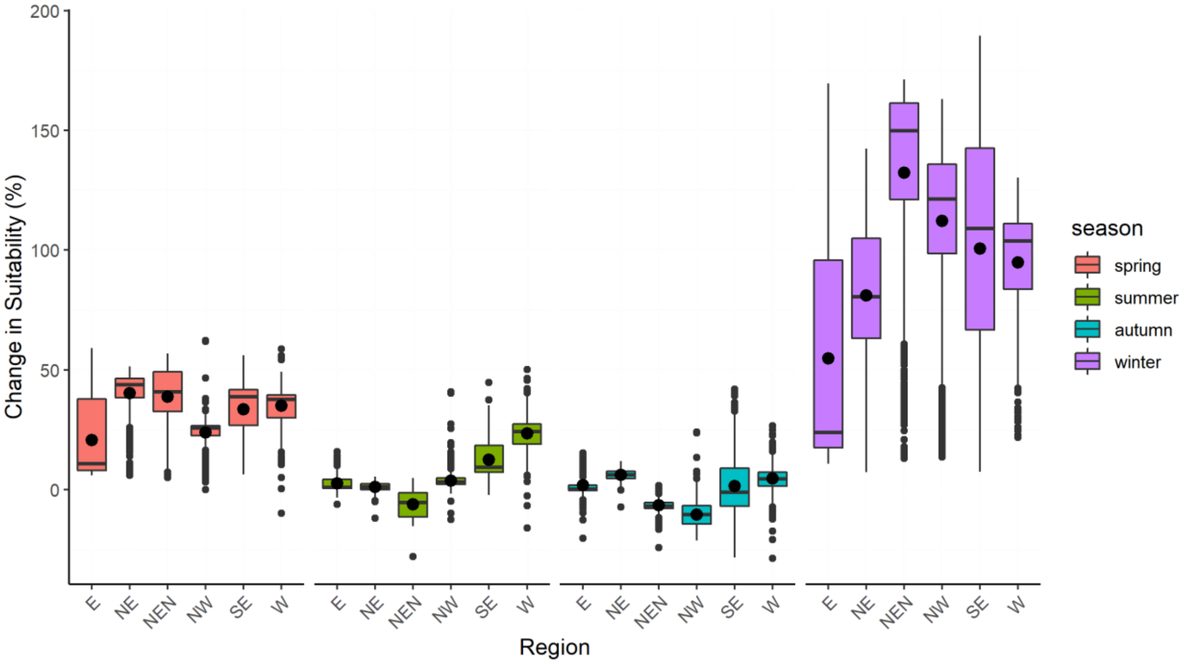

Figure 59. Proportional change (%) in predicted habitat suitability of 416 m^2 areas within six regions of Tasmania, comparing seasonally aggregated environmental data averaged across 20 years: hindcasted (1998–2018), and forecasted (2036–2065) periods across six regions of Tasmania, predicting oceanographic suitable habitat of King George Whiting (*Sillaginodes punctatus*). Boxplots show the median and inner quantiles, points are means ± *SE*.

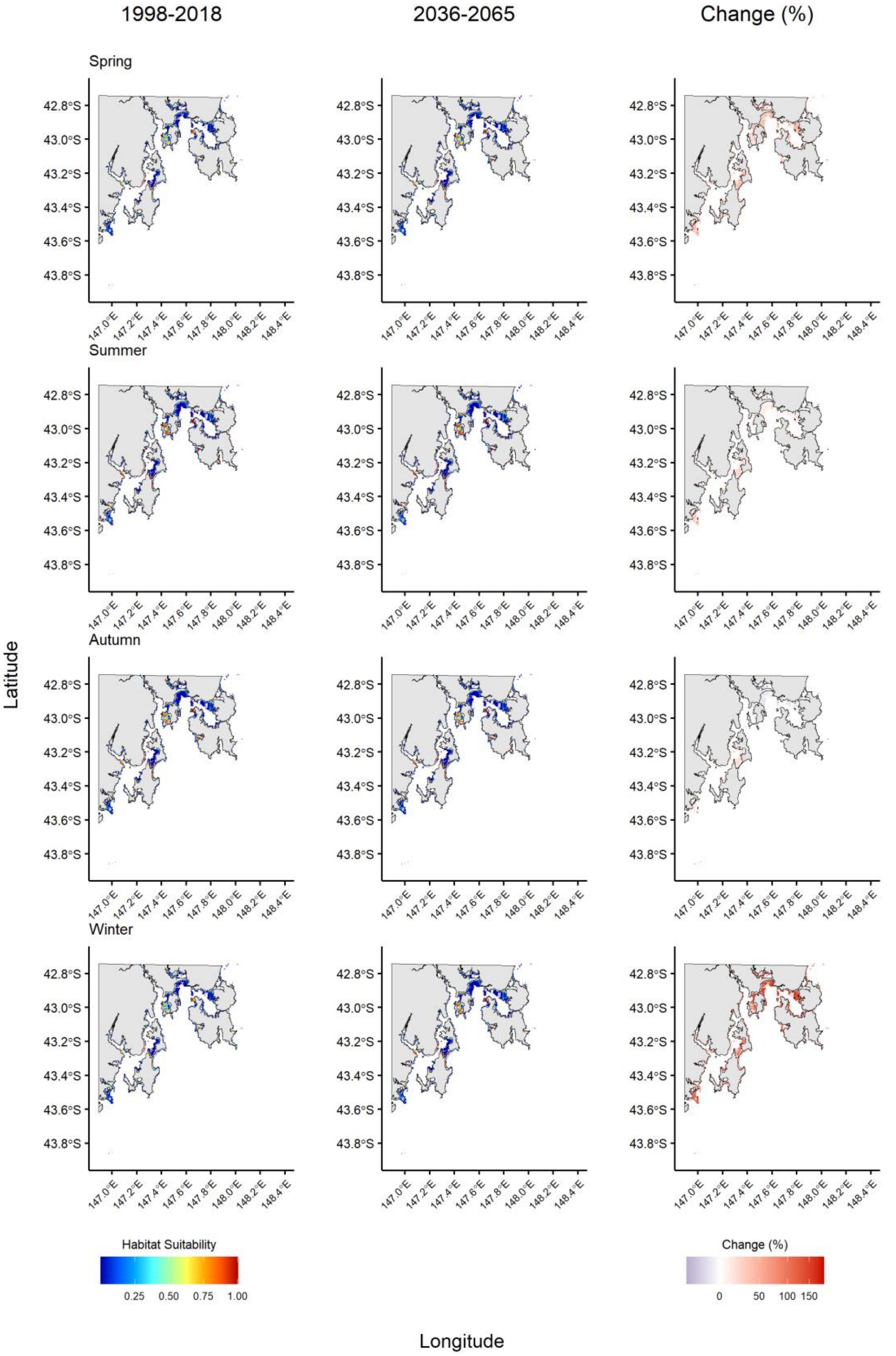

Figure 60. Spatial predictions of habitat suitability of King George Whiting (*Sillaginodes punctatus*) between 20-year averaged environmental data for hindcast (1998–2018) and forecast (2036–2065) periods in the SE region, and the proportional change (%) at a resolution of 0.004°.

Fisheries Data

There was a general increase (22-fold) in Tasmanian commercial catch (NRE Tas) of King George Whiting from 1995 (114.4 kg) to 2016 (2,552.7 kg) across all gear types (Figure 61a, b), with the exception of a peak in catch of 2,617.6 kg in 2006 (Figure 61b). Between 2016 to 2021 however, there has been a 38% decline in total annual catch (1,586.3 kg; Figure 61b). The predominant gear type between 1995–2021 for targeting King George Whiting was the beach seine, accounting for 12–95% of the total catch across all years (Figure 61b). Gear types within the 'other' category include: fish traps, bottom longlines, troll nets, dip nets, squid jigs, spear, ring nets (lampara) and small mesh nets and when pooled together, accounted for 3.3–64.4% of the total catch across all years (Figure 61b). Recreational catch in 2017–2018 (7,200 kg) exceeded the annual commercial catch across all years (range 20–2,617 kg; Figure 61b). Weighted linear models of catch versus latitude through time indicate a decrease in latitude of 0.0060° yr^{-1}, equivalent to a weak trend of 6.7 km north per decade (Figure 61c).

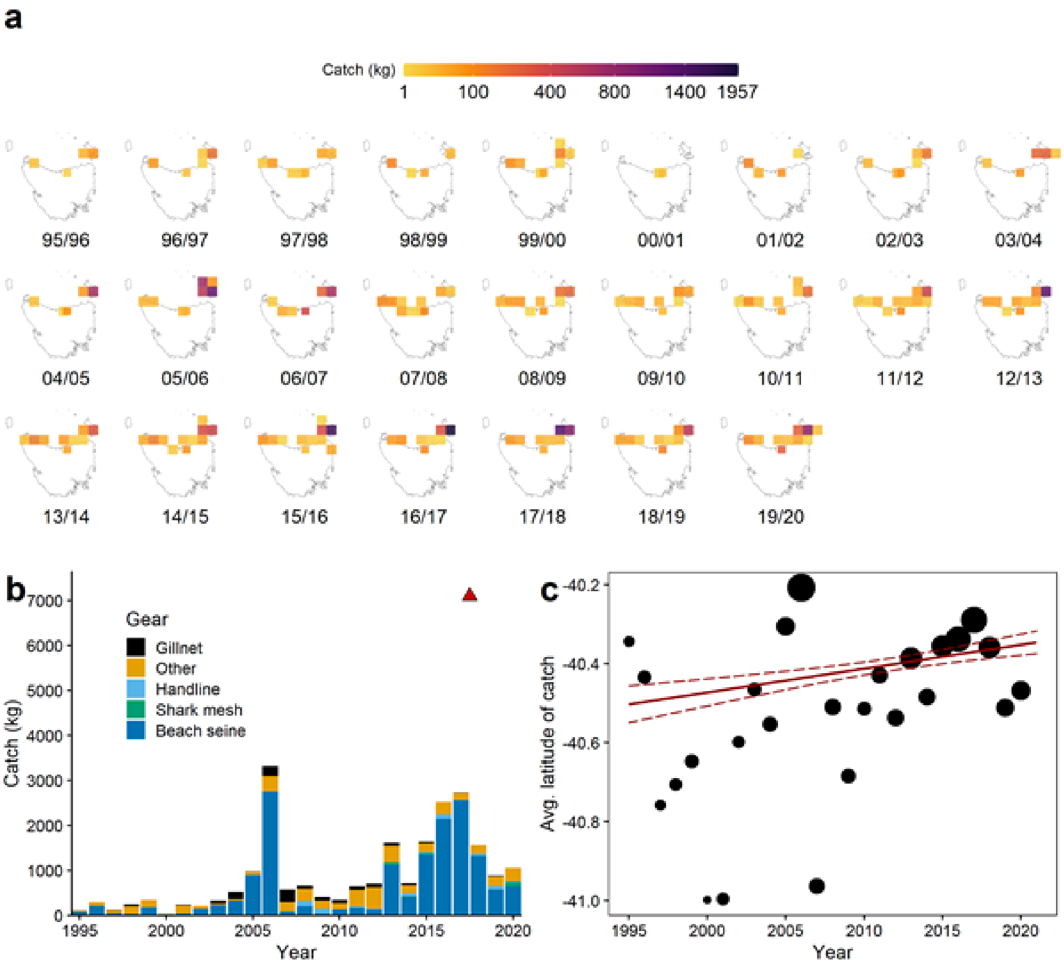

Figure 61. Commercial catch of King George Whiting (*Sillaginodes punctatus*) from July 1995–June 2021. a) Catch (kg) by reporting block in each financial year (all reports were from Tasmania NRE Tas managed fisheries). b) Total catch per year by gear type. c) Mean latitude of King George Whiting catch by year. The trendline was calculated with a linear model of the latitude of catch reports over time (weighted by weight (kg) reported).

Discussion

Biological Summary

King George Whiting in Tasmania can reach ages up to 19 years—with males and females attaining similar maximum ages. However, we found marked differences in size and age structures between regions, and evidence of differential growth between sexes and regions. The smallest observed mature female and male measured 340 and 320 mm fork length respectively and were both 2.9 years of age. Size and age at 50% maturity were estimated to be 407 and 403 mm FL and 4.4 and 4.6 years for females and males respectively. We also found evidence of variable recruitment with strong cohorts in 2001, 2003, 2007, 2014, 2015 and 2017, and potentially weak recruitment in 2002, 2006, 2016 and 2018. Seasonal trends in reproductive activity suggest spawning between February and May and there may be regional timing differences for the onset of spawning.

This study provides the first comprehensive investigation into the temporal and spatial variations of the population dynamics of the emerging King George Whiting fisheries in Tasmania. It has confirmed that Tasmanian King George Whiting are reproductively active and likely to be self-perpetuating. As the species is known to have been present in Tasmania for more than 100 years (Lord & Scott 1924, Roughley 1957), we hypothesise that the observed changes in Tasmanian coastal SST in autumn (cooling later) and spring (warming sooner: BOM 2021) are key factors in influencing adult spawning activities and post-larval recruitment success in Tasmania. This scenario supports Jenkins et al. (2016) findings that two genetically distinct stocks exist in Tasmania, and that these stocks are distinct from mainland stocks. The alignment of Tasmania's coastal environmental parameters with the species' biological thresholds for reproduction, larval survival and post-larval recruitment could explain the observed increase in abundance over the past decade. This suggests that King George Whiting recruitment success in Tasmania is shifting from sporadic and isolated events to a more frequent occurrence as environmental conditions become more favourable. For example, early in this current study the stock was characterised by 3 strong cohorts with birth years 7 years apart (2001, 2003 and 2007), however later in this study it appeared that strong recruitment had occurred more frequently (2014, 2015, 2017 and likely 2019 and 2020). Interannual variation in post-larval abundance and hence recruitment strength is known to be highly variable in King George Whiting with complex environmental interactions such as wind strength and direction, sea-surface temperatures and regional current strength being linked to recruitment variability (Jenkins 2005, Jenkins & King 2006, Hamer et al. 2010). The continued warming of the Bass Strait and southward extension of the East Australian Current along the south-eastern Australian coast is well documented and is the primary facilitating factor responsible for southward trends in species range shifts (Cai et al. 2005, Ramos et al. 2018). However, due to the genetic distinction between Tasmanian stocks and the Australian mainland, it appears that the observed increase in King George Whiting abundance in Tasmania is not a range extension from an Australian mainland stock, but rather a climate-driven change of the coastal environment leading to more positive outcomes for the pre-existing Tasmanian adult stock's reproductive activities and subsequent larval survival and recruitment within Tasmania.

Mortality

Our models show variable total and natural mortality depending on which model was used. Total mortality of the whole stock was estimated using an age-based catch curve from pooled data for both sexes, and across all regions, estimated to be 0.37 yr^{-1} (ages 3–19), and 0.45 yr^{-1} when directly fitting an exponential decay curve to the pooled age frequency data. However, total mortality estimates generated from age-based catch curves assumes consistent recruitment over time and that all age groups are equally susceptible to the fishing gear (King 2013). Due to the increased levels of sampling resulting in elevated numbers of 2–4 year-olds, mortality estimates are potentially biased and will require further sampling over time to derive more accurate estimates. Brown et al. (2013) also highlights the difficulties in obtaining a representative sample of the population resulting from the unidirectional offshore migratory movements of mature individuals. Estimates of natural mortality of Tasmanian King George Whiting were variable depending on which method was used, ranging from 0.16 yr^{-1} (Tanaka's (1960) 5% population survival to T_{max}) to 0.33 yr^{-1}(Hoenig$_{nls}$ estimator), which is most similar to estimates of natural mortality from south-western Australia (0.30 yr^{-1},using Hoenig's (1983) mortality equation; Fisher et al. 2014). By using our variable estimates of natural mortality (0.16–0.33 yr^{-1}), and total mortality (0.37–0.45 yr^{-1}), fishing mortality for Tasmanian King George Whiting could range between 0.04 and 0.29 yr^{-1}, which is markedly lower than the Fisher et al. (2014) estimate from inshore waters of Western Australia where $F = 0.55$ yr^{-1}.

Growth

Although von Bertalanffy growth model parameters are sensitive to the numbers or parameters modelled, the size measurement used (TL vs FL), the modelling method employed and the size at age data range, the asymptotic lengths (up to 606 mm FL ~ 653 mm TL) reported for King George Whiting here are the largest to have been reported for the species. Growth models for both sexes suggested that asymptotic size is attained at approximately 10 years of age. von Bertalanffy growth models indicated sexual dimorphism with females having larger L_∞ values and smaller k and t_0 values than males. The closest growth parameters from a mainland stock are from the west coast bioregion in WA where L_∞, k and t_0 values modelled by mean total length at age (years) data produced parameters of 565 mm TL, 0.36 $yr^{-1}$and 0.01 yrs for females and 536 mm TL, 0.39 yr^{-1} and 0.00 yrs for males (Table G3: Brown et al. 2013). In contrast, the largest growth models from the Spencer Gulf region in SA (individual size at monthly age model) are markedly smaller with L_∞ and k values (note: $t_0 = 0$) of 492 mm TL and 0.49 yr^{-1} for females and 41.6 mm TL and 0.77 yr^{-1} for males (McGarvey & Fowler 2002). Although some of these differences can be explained by the modelling techniques used, they cannot explain the full magnitude of these reported differences. One explanation is these differences are due to an inherit biological difference associated with the genetic distinction between the Tasmanian stocks and mainland stocks. Alternatively, it could also be a data effect where the relative abundance of old large individuals in the NWC stock is producing biological growth parameters more realistic for the species as growth models have been generated prior to a marked age/size truncation of the stock from fishing pressure. The observed sexual differences in growth models in King George whiting observed in the current study is consistent with the species' known biology and is well documented (Hyndes et al. 1998, McGarvey & Fowler 2002, Brown et al. 2013). However, our results demonstrate that divergent growth occurs prior to sexual maturity as it could be detected by 3 years of age with the effects magnitude increasing with increasing age. Although the detection of spatial differences in growth models can in part can be explained by the variable data ranges between regions, some effects still remained significant once standardized.

The spatial differences in mean size at age of fish aged between 2 and 5 between regions provides additional suggestion that growth in King George Whiting is regionally different in Tasmania. Under a species' normal temperature range, low temperatures will slow growth, while warmer temperatures promote faster growth (Handeland et al. 2008). As water temperatures from these three regions are typically warmest on the EC and coldest on the NWC, the differential trends in mean size at age do not align with a traditional temperature effect. Temperature and photoperiod are well known environmental cues in regulating reproductive activity in fish (Pankhurst 2016) with King George Whiting reported to spawn when daylength is shortening and temperatures are falling (19.1 to 14.8 °C; Ham & Hutchinson 2003). As the EC stock is genetically distinct to the NWC stock the differences in growth could be an intrinsic genetic effect. Alternatively, variations in environmental conditions between the NWC and EC stocks may be influencing the onset of spawning or main recruitment period between these 2 separate stocks. In particular, we suspect that a later post-larval recruitment period on the EC is the main driver of the smaller size at age of EC fish. Whether the observed regional differences in mean size at age are due to intrinsic or environmental influences on growth rates or simply reflect differences in reproductive events between regions or a combination of factors remains unclear. As such, future studies should clarify regionally specific birthdates, key recruitment periods and the timing of the first annuli between regions to ensure an individual's decimal age is regionally appropriate.

Age/Length

The marked regional differences in the length and age frequencies in Tasmania is most likely associated with ontogenetic habitat choices of King George Whiting. The dominance of young individuals (≤ 4 yrs) in the estuaries of Port Sorell, Tamar River and Georges Bay is similar to the age and size structures of King George Whiting from known nursery areas found in the mainland states, which suggests that these sheltered, and seagrass rich waterways are functioning as post-larval settlement areas and nursery habitats in Tasmania. In addition, the collection of a single post-larval King George Whiting in December 2014 (~31 mm TL) and numerous young of the year whiting in February 2021 (54–83 mm TL) in Georges Bay indicates the increased abundance of King George Whiting in Tasmania is more likely due to successful post-larval settlement in these areas rather than juvenile migration (Jenkins et al. 2016). These observations are consistent with previous observations of King George Whiting life-history, in which successful post-larval settlement (Jenkins & May 1994, Fowler & Short 1996, Jenkins et al. 2000) is followed by approximately four years spent in protected waterways and embayment's before an offshore autumn migration during the onset of sexual maturity (Hyndes et al. 1998, Fowler et al. 2000a). The abrupt change to age/size structure and disappearance of fish older than four years in Georges Bay is likely to be due to the topography of the narrow and shallow barway of Georges Bay being unfavourable for adults to return to Georges Bay after their offshore autumn spawning migration. In contrast, the Tamar Estuary has a wide (~3 km) and deep (60 m) opening which is more suited for the return of post-spawning adult fish, which is consistent with a recent capture of an 80 cm King George Whiting and numerous fish older than 5 years of age being present in the Tamar River. Although no samples were obtained, anecdotal reports of recreational fishers catching female King George Whiting with hydrated oocytes in the Tamar River in April 2017, suggest that the deep waters of the Tamar River mouth may be a potential spawning site for the species. The occurrence of juveniles and adult fish in the Tamar River indicates that this area possesses all the required habitats across the ontogenetic habitat preferences for King George Whiting. In contrast, the age structure of King George Whiting from the NWC had a distinctive lack of young fish, with the population comprising mainly of adult fish. This adult-biased age structure is similar to the identified spawning areas in South Australia and Western Australia (Brown et al. 2013, Fowler et al. 2014). It also suggests that the abundance of whiting in this area is due to migration of sub/early adult fish from either a yet to be identified nursery area or a westerly migration of King George Whiting from the Port Sorell and Tamar River estuaries. The population of fish from the NWC has a greater proportion of large and old fish compared to other spawning areas from mainland stocks in SA or WA. We hypothesise, that the relatively low exploitation rate in nursey areas prior to 2010 allowed sporadic recruitment events to establish into a significant standing stock of adult fish that did not experience significant fishing pressure while inhabiting their nursery areas. Although ongoing stock monitoring from the 1990s show no obvious evidence of size or age truncation in SA or WA offshore stocks, the relative abundance of large adults (> 45 cm or > 7 yrs) in these areas typically represent a small percentage of the catch (Brown et al. 2013, Fowler et al. 2014). The contrasts between the age/size structures from mainland Australia and the NWC of Tasmania, has previously been suggested as a potential sampling method effect (Jenkins et al. 2016). However, this is unlikely as the standardised abundances of age classes between capture methods (gillnet vs hook and line) from the NWC are very similar (unpublished data). Alternatively, it appears that the stock structure from the NWC in the present study may reflect a population that has experienced little age/size truncation due to the relatively low historical fishing pressure on this stock. However, over the past decade (half the longevity of King George Whiting) targeted fishing pressure has rapidly increased in both nursery (virgins) and coastal (adults) areas in Tasmania. The impact of this increased fishing pressure on the age/size structure of the stock into the future will need ongoing monitoring as the fishery develops in Tasmania, especially as the increase in fishing pressure is yet to be reflected across the full longevity of the species.

Maturity

Fork length of King George Whiting sampled from Tasmania ranged from 53 to 619 mm and spanned ages 0 to 18.8 years. The mean asymptotic lengths calculated for King George Whiting in Tasmania were higher than for known spawning populations in South Australia (McGarvey & Fowler 2002) and Western Australia (Hyndes et al. 1998). The smallest mature female and male observed in this study were 340 and 320 mm FL respectively. Size and age at 50% maturity were estimated to be 407 and 403 mm fork length and 4.4 and 4.6 years for females and males respectively, which is reflective of established maturity estimates in the SA and WA populations of King George Whiting (Hyndes et al. 1998, Fowler et al. 2000a, Jenkins et al. 2016). However, fish that possessed gonadal stages indicating imminent spawning (hydrated or spermiated) were relatively rare (hydrated: n = 6; spermiated: n = 1).

Seasonality of spawning

The reproductive trends observed in the current study are consistent with the known reproductive biology of the species (Hyndes et al. 1998, Fowler et al. 1999, Jenkins et al. 2016) with maturity typically occurring in the fourth year of life and spawning taking place between February and May. Although mean GSI values peaked in March and February for females and males respectively it is likely that these values are smaller than the true population mean, due to the suspected offshore migration of maturing adults leaving the inshore waters. Moreover, we suspect the true peak in GSI to be April, due to the confounding effects of a significant lack of fish older than four years of age being captured in this month (n=1), and their reappearance in the shallow waters as spent adults in May–July. Difficulties in obtaining samples of spawning whiting is common (Cockrum & Jones 1992) with the only known spawning sites typically being very restricted in space (Hyndes et al. 1998, Fowler et al. 1999, Fowler et al. 2000b). In the current study, all six hydrated females obtained were captured within a 50 km length of coastline on the NWC. In addition, all but one fish were captured in the shallow waters of coastal headlands with a close proximity to deep water drop-offs. However, whether Tasmanian King George Whiting simply have a dispersed offshore spawning migration to the deep waters throughout northern and eastern Tasmania or undergo a specific spawning migration and aggregation to site-specific spawning grounds remains uncertain. Despite this uncertainty, this study confirms the current environmental ocean conditions around northern Tasmania is conducive to reproduction and that this location can now be defined as an active spawning location for King George Whiting.

Recruitment variability

Annual recruitment of juvenile King George Whiting across southern Australia is highly variable likely due to complex environmental interactions (Jenkins & King 2006). The strength of regional wind indices and increased sea surface temperatures resulting from increased flow rates of regional currents are examples linked to variabilities in King George Whiting recruitment (Jenkins 2005, Jenkins & King 2006, Hamer et al. 2010). The continued warming and southward extension of the East Australian Current along the south-eastern Australian coast is well documented and is the primary facilitating factor responsible for southward trends in species range shifts (Ramos et al. 2018) As waters continue to warm and extend southward into Victoria and Tasmania, elevations in water temperature may be contributing to positive outcomes for growth and survival rates for King George Whiting (Cai et al. 2005, Jenkins & King 2006). Fluctuations in the mean monthly SST observed for the NWC (Burnie) during strong recruitment events in 2001, 2003 and 2007 are comparable to findings in known King George Whiting spawning locations (Jenkins 2005, Jenkins & King 2006, Hamer et al. 2010). Years on the NWC that recorded SSTs of 15, 16 and 17 ºC for May with maximum temperatures observed as high as 18º C experienced strong recruitment years. In contrast, years where recruitment was poor, particularly in 2006 the average SST for the same region was significantly lower at 13 ºC (BOM 2021).

The alignment of Tasmania's coastal environmental parameters with the species' biological thresholds for reproduction, larval survival and post-larval recruitment could explain the observed increase in their abundance over the past decade. This suggests that King George Whiting recruitment success in Tasmania is shifting from sporadic and isolated events to a more frequent occurrence as environmental conditions become more favourable. For example, early in this current study the stock was characterised by 3 strong cohorts born with birth years seven years apart (2001, 2003 and 2007), however later in this study it appeared that strong recruitment had occurred more frequently (2014, 2015, 2017 and likely 2019 and 2020). Over recent years (1992–current) the autumn SST of central-southern Bass Strait (Burnie) has shown a variable but general warming trend, where the rate of warming is greatest in the months March-April and Nov–Dec (BOM 2021). This effect results in the autumn months not cooling as rapidly as they historically did, which may be beneficial to onset of spawning for King George Whiting spawning and early larval survival. In addition, water temperatures in the spring are warming more quickly than previously, thus reducing overwintering time for recently settled recruits.

Diet

Polychaetes and malacostracans were the dominant prey classes in King George Whiting stomachs of our sample both in terms of frequency of occurrence (polychaetes: 68.8%, malacostracans: 55.19%) and mean proportion (84.8%) of prey. These results are consistent with previous diet/stomach content studies of adult King George Whiting in Western Australia, where the frequency of occurrence of crustaceans and polychaetes are 69.7% and 60.7% respectively (Potter et al. 1996). However, in contrast to previous studies, we did not find differences in prey items across different size classes or between the North and North-East regions. Previous studies suggest that small (0+ years, 10–44 mm) King George Whiting feed on harpacticoids and gammarid amphipods (Robertson 1977, Jenkins et al. 2011), and when they reach 1+ years (275 mm TL), there is a shift in diet to opisthobranch gastropods, algae, errant polychaetes, and the Ghost Prawn (*Callianassa australiensis*, Robertson 1977, Potter et al. 1996), and large sipunculid worms have been found in stomachs of larger fish (~545 mm: Potter et al. 1996). It is likely that we didn't find any ontogenetic shifts in diet due to low sample size/representation at the smaller (2 individuals < 238 mm FL) and larger size classes (10 individuals 452–560 mm FL), to detect any potential shifts in diet. Furthermore, the consistency of prey classes in the diet across the size of King George Whiting may also be due to a range of sizes of prey at the class taxonomic level. For example, polychaetes identified in King George Whiting stomachs ranged from several millimeters to 300 mm in length. Similarly, the size of the larger claw of callianasid Ghost Shrimp identified in King George Whiting stomachs, typically the largest hard parts, ranged over an order of magnitude in size, and were consumed by individuals ranging from 230 mm to 560 mm fork length.

The results of our study also found no evidence of differential diet between the North and North-East coasts of Tasmania. However, previous studies have found diet to be site- and habitat-specific. For example, Jenkins et al. (2011) found that King George Whiting recruits consumed much higher quantities of prey in unvegetated versus seagrass habitats, but that prey diversity/composition was related to the proximity to Port Phillip Bay, presumably driven by differential levels of sedimentation and turbulence. Such differences in diet are at relatively fine spatial scales compared to our broad North vs North-East coast comparison. As the reporting of the location for the 'Tassie Fish Frame Collection Program' was relatively coarse (i.e. 'St Helens' etc), we were therefore able to determine whether there were habitat- and site-specific differences in diet. However, given that meiofaunal communities, and thus prey availability significantly differs between habitat type and location (Jenkins & Hamer 2001), fine scale differences in diet of King George Whiting in Tasmania is likely.

Species Distribution Model

Spatial predictions from 1998 to 2018 show seasonal variation in the suitable habitat for King George Whiting, with more favourable habitat in the warmer seasons (i.e. summer and autumn versus winter and spring), but there was no increase in suitable habitat through the hindcast period (1998–2018). When comparing averaged environmental data between 1998–2018 (hindcast) and 2036–2065 (forecast), the winter season was predicted to be more suitable under the IPCC RCP8.5 scenario, evidenced by the proportional increase in suitability of small (416 m^2) grid cells across the Tasmanian domain. There was negligible difference in suitability in the summer and autumn seasons between the hindcast and forecast time periods, suggesting that predicted warming during the warmer seasons may be at, near, or beyond the peak of the thermal preference curve for King George Whiting and that further warming (i.e. > 2050), may lead to a decline in suitability in the warmer seasons. However, the increase in suitability in the winter season may be beneficial for successful recruitment and overwintering of King George Whiting in Tasmania. As spawning generally occurs in the autumn, when temperatures and day length decrease (Ham & Hutchinson 2003), warmer winters will therefore be beneficial particularly for larval and juvenile stages, thereby enabling successful recruitment and overwintering of these critical life-history stages in Tasmania. Current (i.e. 2018) average winter temperatures in the north and east coast of Tasmania (as far south as Bicheno) range from 13.3–13.7 °C (Copernicus Marine Monitoring Service 2018). However, under future warming (RCP8.5 scenario), winter temperatures are predicted to increase by as much as 3 °C in the north-east regions of Tasmania, predicted winter temperatures ranging from 16.8–17.7 °C. To date, the critical thermal minimum for larval and juvenile performance of King George Whiting still remains unclear, as the majority of physiological studies focus on determining optimum performance (e.g. Ham & Hutchinson 2003) or are from the mainland and use regional winter temperatures as the minimum (i.e. ~16 °C: Mazloumi 2015, Drew et al. 2020). However, egg development (Drew et al. 2020) and both adult and juvenile aerobic metabolism, and swimming performance (Mazloumi 2015, 2017) are sustained during the South Australian winter (i.e. 16 °C), albeit at a slower rate than at 22–26 °C, a potential optimum, providing evidence for enhanced performance on the mainland, and potential for areas where recruitment and overwintering is possible around Tasmania, under future warming.

In addition to temperature, distance to seagrass was a significant linear predictor for King George Whiting occurrence. Seagrass habitat provides protection from both physical disturbance (Bostrom & Mattila 1999), and predation (Flynn & Ritz 1999, Hindell et al. 2000, 2002), and increases food availability (Connolly 1994, Edgar 1999, Jenkins et al. 2002), and are therefore important nursery areas for many juvenile fishes (Jackson et al. 2001), including King George Whiting (Jenkins et al. 1995). In our current SDM we assumed the presence of seagrass to be static under future change, and we acknowledge that is a limitation of our current model. However, using regional habitat predictors such as proximity to seagrass is encouraged, as this increases the predictive capacity of SDMs (Kaplan et al. 2016). Some seagrass populations have already undergone a redistribution, specifically, contracting at the warm-edge of their range, or extending at the cold-edge of their range (Duarte et al. 2018). Given the temperate locality of Tasmania, future warming may lead to range extensions of temperate seagrasses which are currently limited to the north, and north-east coasts *(i.e. Posidonia australis* and *Amphibolis antarctica*, Rees 1993), or increase performance/growth for seagrass communities at the centre or cold-edge of their distribution *(e.g. Zostera mulleri, Heterozostera tasmanica, Halophila australis*, Rees 1993), which may be beneficial for future recruitment of King George Whiting.

George's Bay, on the north-east coast of Tasmania, may be a hotspot for recruitment, as it is a popular fishing spot, where anecdotal evidence from recreational anglers suggests that King George Whiting abundance is increasing. This was evidenced by 360 individuals, resulting in 20% of our subsample of King George Whiting caught within that region during 2012–2021. SDMs confirm that this area is of relatively high habitat suitability due to the shallow depth, abundance of seagrass, and preferable temperatures within this region. Between the hindcast and forecast time-periods, there is a predicted average increase in habitat suitability of 59.07% (± 0.25 *SD*), with a maximum increase in suitability of 108% in some grid cells/areas within George's Bay in the winter season (Figure 62), with a predicted mean winter temperature under RCP8.5 of 15.81 °C.

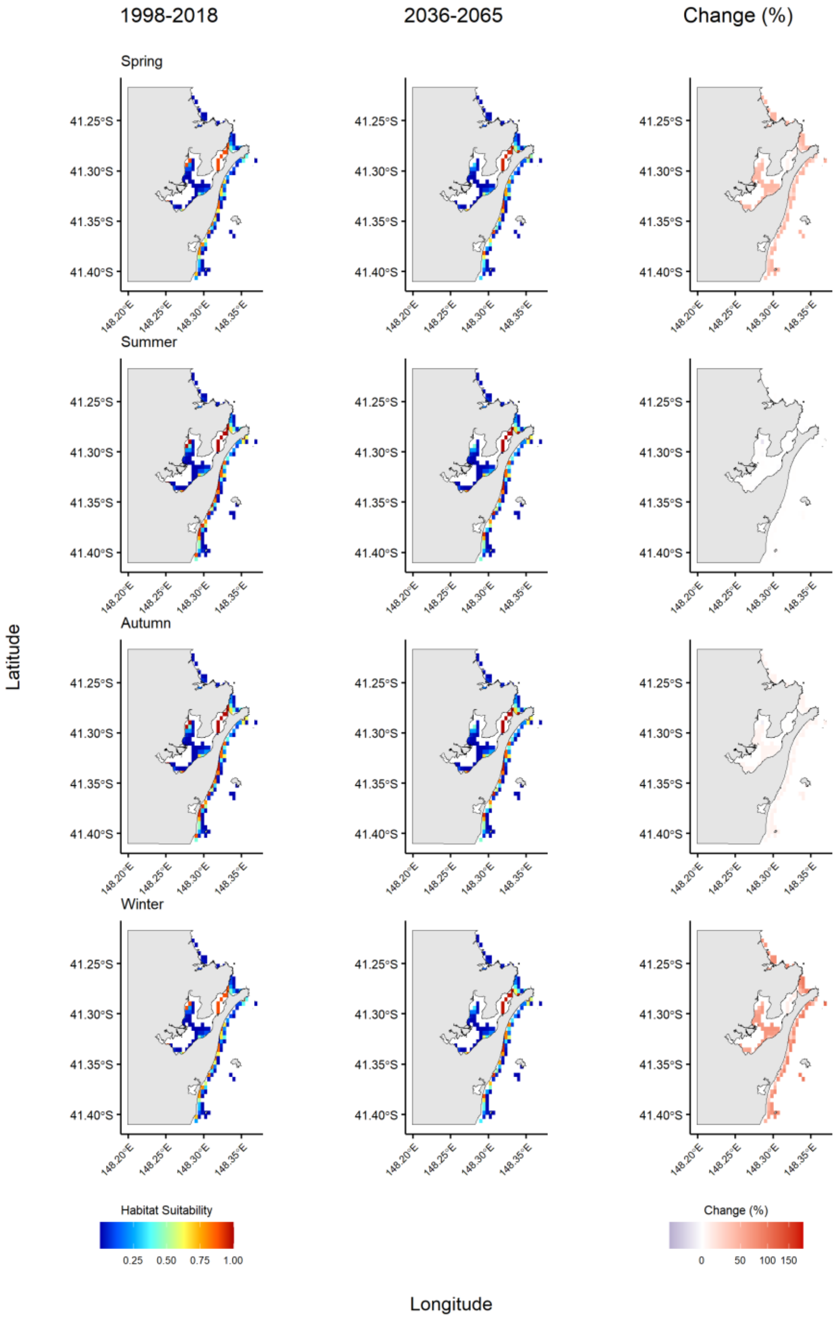

Figure 62. Spatial predictions of habitat suitability of King George Whiting (*Sillaginodes punctatus*) between 20-year averaged environmental data for hindcast (1998–2018) and forecast (2036–2065) periods for George's Bay, within the NE region, and the proportional change (%) at a resolution of 0.004°.

Fisheries Summary

There was on overall increase in annual catch of King George Whiting from the Tasmanian (NRE Tas) commercial sector from the mid-1990s to 2016, followed by a slight (38%) decline in 2020. The beach seine is the dominant method used by commercial vessels in Tasmania, and we found a weak trend of decreasing latitude in catch, of 6.7 km north per decade, coinciding with increased catch around the Furneaux Islands. Commercial catches of King George Whiting in Tasmania date back to 1995, when compulsory logbooks were introduced. Historical timeline trends in commercial catches, although relatively small, demonstrate an increase in commercial catch volumes and the numbers of fishers reporting King George Whiting catches to a peak of ~3.5 t in 2016–2017 (Fraser et al. 2021). The majority of this catch is landed by beach seine around Flinders Island and a lesser volume by gillnets on the north-west coast, however, due to a reduction in beach seine effort on Flinders Island in recent years the commercial catch has dropped to 1.6 t in 2019–2020. Due to relatively low commercial catches the species has not been subject to annual stock assessments until 2021, which currently classifies the stock as "sustainable" (Fraser et al. 2021). Similarly, recreational catches have also rapidly expanded over the past decade and are considerably larger than the commercial catch with an estimated 7.2 tonnes (~82% of total estimated catch) retained in 2018 (Lyle et al. 2019).

This trend in annual catch is the opposite to the commercial sector in South Australia where annual catch was considerably higher in the 1980s and 1990s than the 2000s, where there was a significant decline in annual catch across all three biological stocks. This was related to a significant decline in the number of fishers and effort, but fisheries assessment modelling suggested reduced recruitment within the Gulf St. Vincent and Spencer Gulf biological stocks (Steer et al. 2018). As such, these stocks were considered 'transitional-depleting', whereby the minimum legal size was increased (310 mm to 320 mm TL), bag and boat limits were reduced to 10 per person, and 30 max per boat with more than three people (PIRSA 2019a), and spawning closures were imposed during May from 2017–2019 (PIRSA 2019b). Since then, catch per unit effort (CPUE) has increased by 20% and 21% for the Gulf St. Vincent and Spencer Gulf biological stocks respectively (Steer et al. 2018). Furthermore, modelling suggests increased recruitment across all biological stocks in South Australia, in concert with reduced fishing exploitation, resulting in increased fishable biomass of King George Whiting in South Australia.

Similarly, commercial fishing effort in Victoria for King George Whiting has also declined since 1999 as a result of a reduction in licensed fishers and change in gear types (VFA 2017). Specifically, fishing effort for the haul seine, the main method of fishing in Victoria has declined substantially in Port Phillip Bay (PPB) since 2015–2016 and is expected to be eliminated from this region by 2022. Therefore, CPUE may not be the best method to determine stock status in Victoria. Due to the ontogenetic shift in habitat of King George Whiting (Hamer et al. 2004), the Victorian biological stock is highly dynamic, and the age/size structure of this population is limited to a few age classes at any given time (Fowler et al. 2021a). Therefore, post-larvae surveys may provide better estimates of recruitment, and recent surveys have shown increased recruitment in Victoria between 2016–2019, with a peak in 2017, which is expected to increase CPUE in the near future (Conron et al. 2020). Further, reported annual commercial catch has increased from a low of 54.08 t in the 2017–2018 financial year to ~117.65 t in the 2018–2019 financial year (Fowler et al. 2021a). In contrast, annual commercial catch in WA has been variable between 1976 to 2000, ranging between 35–150 t (Fowler et al. 2021a). Since 2000, annual commercial catch in WA has remained relatively stable between 20–40 t, which coincides with limited change in fishing effort (Fowler et al. 2021a).

Across Victoria, SA, WA, and Tasmania, the recreational catch of King George Whiting consistently exceeds the commercial catch (Table 20), where the difference between commercial and recreational catch ranges between 1.22 times greater in South Australia to 4.39 times greater in Tasmania, noting that recreational catch in Victoria was taken from the national recreational fisher survey in 2003 (Henry & Lyle 2003), and is not routinely quantified.

Table 20. Total annual harvest (tonnes (t)) from commercial and recreational catch of King George Whiting (*Sillaginodes punctatus*).

	Victoria	**South Australia**	**Western Australia**	**Tasmania**
Commercial	117.65 t[1]	299.77 t[1]	17.06 t[1]	1.64 t[2]
Recreational	214.57 t[3] 2000–2001	367 t[4] 2013–2014	29 t[5] 2017–2018	7.2 t[6] 2017–2018

[1] Fowler et al. 2021a.
[2] Fraser et al. 2021.
[3] Henry & Lyle 2003.
[4] Giri & Hall 2015.
[5] Ryan et al. 2019.
[6] Lyle et al. 2019.

Implications for management

This study provides additional evidence to support the distinction of two stocks in Tasmania. We hypothesize that a stock of adult King George Whiting resides in the waters off Flinders Island with their spawned eggs and larvae being dispersed in a southerly direction via a weakening East Australian current during autumn and winter and Georges Bay functioning as the key nursery area for this stock. In contrast, eggs and larvae from adult King George Whiting spawning in Bass Strait are likely to be dispersed in an easterly direction by the dominant currents in the area in autumn/winter and the Tamar River and Port Sorell estuaries being the key nursery areas for this stock. Under these two scenarios we believe that over the past few decades a climate change increase in the environmental suitability for spawning and post-larval recruitment success has allowed 2 relatively small endemic Tasmanian stocks to establish into significant adult populations with relatively little fishing mortality. However, despite this increase in total egg production and post-larval recruitment success, their increasing abundance has only become apparent to the recreational fishing sector over the past decade. The rapid increase in targeted fishing on King George Whiting is concerning as it is uncertain whether over the long term the relative abundance of large old fish will be maintained given the increase in fishing mortality on these stocks. Although the large King George Whiting around the Flinders Island area are partially protected due to the isolated nature of this area, the relative ease with which both the juvenile and adult stock in the north can be targeted is a concern. If this fishing pressure is too heavy, we may start to see the size and age structure of this stock mimic those seen on mainland Australia, where fish over the size of 450 mm TL or age of seven become uncommon.

Currently the MLS for King George Whiting varies between all states with Tasmania being the largest (PIRSA 2019a, VFA 2021a, NRE Tas 2022b, WA DoF 2022). The current MLS in Tasmania is 350 mm TL (NRE Tas 2022b). The smallest mature female and male King George Whiting captured in this study was 362 and 355 mm TL at approximately three years of age for both sexes which is larger than previous studies for King George Whiting in Southern Australia (Fowler & Short 1997, Hyndes et al. 1997, Jenkins et al. 2016). The current MLS in Tasmania currently allows fishers to target virgin fish. Increasing the MLS closer to the size at 50% maturity (i.e. ~400 mm FL, 425 mm TL) would allow a proportion of the population to spawn at least once. The migratory behaviours of larger individuals involve movement to deeper offshore waters that typically receive less fishing pressure and are therefore are less exploitable (mainly due to predominant adverse weather conditions). Additionally, these fish exhibit modified feeding behaviours during the spawning season (Fowler & Short 1997, Hyndes et al. 1997). In combination, these behaviours effectively impose a biological closure on the fishery thus limiting the catchability of spawning individuals. However, to avoid outcomes seen in the South Australian King George Whiting fisheries where the mean fish size has decreased significantly as a result of technological advances enabling increased targeting of spawning aggregations, continued monitoring of both the stock and fisher behaviour is recommended. In 2016, South Australia implemented a series of fisheries management controls aimed at addressing these issues. Consideration of such measures by Tasmania is recommended if future evidence demonstrates that exploitation of spawning fish in increasing. Imposing a seasonal closure in Tasmanian waters would assist in the protection of spawning aggregations, and therefore future recruitment into the fishery. More information and thus ongoing monitoring is needed to assess whether other management options such as maximum size limits and or temporal/spatial closures are needed in the future as the fishery develops.

Conclusions and Implications

Objective 1 - Develop a program for ongoing collection of biological samples and data of key range shifting fish species using citizen science initiatives engaging with the recreational fishing community.

Here we have presented discrete life-history parameters for three key range extending species in Tasmania. Underpinning these results is a successful citizen science project for collecting fish frames across the state, as well as opportunistic and fisheries independent sampling. As the 'Tassie Fish Frame Collection Program' has been so successful, Striped Trumpeter, Sand Flathead and Silver Trevally have also been added to the program to provide critical and ongoing data collection for these species, as information on these species is limited. Over 500 fish frames of Yellowtail Kingfish, Snapper and King George Whiting were donated from recreational anglers between 2019–2021, demonstrating strong engagement with this sector. Such engagement is critical, as the recreational sector is the dominant users for all three species.

Collecting data on range extending species is difficult as it is largely dependent on where they are in terms of their range extension (i.e. between arrival, persistence and establishment: Bates et al. 2014). As species at the edge of their range typically aren't as abundant and may not have the same life-history attributes, knowledge surrounding movement and spawning is limited, and critical information regarding where to find them and how to catch them may not be as well understood. Therefore, a fundamental component in collecting data on range-extending species is collaboration across many sectors both within and outside of the citizen science space. For example, collaboration between Redmap and the present study was fundamental in enhancing the success of both initiatives. Collaboration bolstered data collection (fish frame donations, photos, geographic information), helped both programs reach a broader audience and extended science communication about range-extending species in Tasmania to a broader audience. Furthermore, the Redmap data set was a major component for the SDMs as it provides critical occurrence data for species in new environments.

In addition, engaging with industry also provides an opportunity to collect data. Due to the very nature of range edges, range-extending species are likely to be caught as bycatch or emerge as targets of commercial fishers. As such, the commercial sector was particularly helpful in the donation of a large proportion of the King George Whiting frames. However, fishery-independent sampling is also critical to collect data outside of fish size limits (i.e. small/juvenile), and to account for potential angler bias (i.e. targeting only the upper limit of the MLS, or baking fish whole (Snapper), thus limiting a size class for fish frame donation). Therefore, while engaging with industry, citizen science and the recreational fishing community contributes to the bulk of data collection efforts, targeted sampling by the research sector is also critical to obtain the whole size/age spectrum for species at the range edge. Open access databases such as the Atlas of Living Australia, Reef Life Survey and Seamap Australia are also fundamental for ecological modelling and predicting potential shifts in habitat. Therefore, not only is it important to engage across sectors for data collection, but it is also equally important to make the data available to inform larger ecosystem / climate models for a more holistic prediction of future change.

Objective 2: Develop geographically discrete life-history parameters for key range shifting fish species in Tasmania to inform management decisions.

Although we successfully estimated discrete life-history parameters for the three key range shifting species, this data provides only baseline information for King George Whiting, Yellowtail Kingfish and Snapper in Tasmania. Due to the opportunistic nature of the fish frame collection program, and the relatively short sampling period there are several gaps in the data that cannot be disentangled between sampling design or natural variation in recruitment and age/size structure. Quantifying the life-history of a species at the edge of their distribution is difficult as they are only starting to increase in abundance, and some sources for recruitment remain unclear. Further, any increases in sampling may shift this baseline, due to predicted increases in abundance and ongoing climate change, making this problem dynamic. As such, our limited data violates assumptions of traditional fishery assessment methods which typically require a large and robust sample. For example, as the Tasmanian sample is relatively young, and therefore still growing, our estimates of mortality are likely to be different as their maximum age (T_{max}) increases through time. Estimates of natural mortality also assume consistent recruitment, of which we have little information. Therefore, there is a need for ongoing monitoring to identify the source of patchiness in the data, as there is evidence of interannual recruitment variability on the mainland, however we do not have the time-series to confirm this as a trait or simply product of sampling design. Nevertheless, the information derived from this project provides a useful baseline for life-history characteristics of Yellowtail Kingfish, Snapper and King George Whiting in Tasmania.

The SDMs and the Atlantis ecosystem model suggest increased habitat suitability across the Tasmanian domain with limited impact on ecosystem function. Furthermore, SDMs suggest less limitation for overwintering of new recruits for both Snapper and King George Whiting, with suitability predicted to increase the most in the winter. Therefore, it is likely that Snapper and King George Whiting will become more abundant, and move southward across the Tasmanian domain, having little ecological impact as they expand their range. This provides Tasmania with a unique opportunity to manage these emerging fisheries. Most fisheries management is geared towards managing an established fishery in order to preserve current stocks. Or fisheries management can be reactive to declines in recruitment or abundance. Here, we are presented with the opposite—providing scope for proactive strategies to grow these emerging fisheries sustainably. As such, using the life-history information from this study as a baseline, we were able to make informed suggestions for management–noting that these populations are in their infancy, and are likely to continue to change. As such, we should be ready to be flexible, and apply adaptive management accordingly.

Yellowtail Kingfish

We determined Yellowtail Kingfish adjacent to the east coast of Tasmania were generally small (< 600 mm FL), immature fish (2–3 years) which were seasonally present from November to April. Due to the small sample size, and age truncation of the samples, we were unable to model growth rates. Also, given the size of the fish, it is highly unlikely that Yellowtail Kingfish spawn in Tasmanian waters as there was only one reproductively active female in our sample. Furthermore, SDMs show that even under extreme warming (RCP8.5), it is unlikely that it will be warm enough for Yellowtail Kingfish to become a self-sustaining population in Tasmania. Currently there is not enough information, or abundance of Yellowtail Kingfish to be included in the Tasmanian Stock Assessment. Therefore, we suggest ongoing monitoring via engagement with citizen science, fish frame collection, and fishery independent sampling. However, should there be evidence of change (i.e. higher abundance, bigger individuals, occurrence earlier or later in the year), management strategies should be flexible and adaptive to change. As the population is so small, the current MLS of 450 mm TL keeps the recreational fishery available to fishers in Tasmania, and if catches do increase, an alternative strategy may include reducing the bag limit, rather than the MLS. Furthermore, evidence of the population being growth overfished should be evident when assessing the Eastern Biological Stock as a whole.

Snapper

Snapper in our study reached a maximum size of 799 mm FL, and ranged in fork length from 241–700 mm FL for females, 233–633 mm FL for males, and 150–316 mm FL for juveniles of indeterminate sex, and 268–779 mm FL for individuals lacking gonadal material. There were no differences in age/size structure between sexes, however we did find regional differences in Snapper age/size structure where individuals were larger and older in the north versus the south. There were gaps in age/size structure of our sample, however it was unclear whether this was sampling bias or evidence of interannual variation in recruitment. The maximum age of our sample was 19.2 years (indeterminate sex) and was 18 and 12.8 years for females and males respectively. We are also aware of a large (990 mm TL, 11.6 kg) Snapper that was caught in Tasmania that was aged at 27 years by the Fish Ageing Service. These age estimates, however are still well below the maximum age reported for Snapper in Australia (41 years: Table G2). We found no statistical differences in the growth of Snapper both between sexes and regions. Due to the relatively small sample of females with hydrated oocytes (n = 4), and spermiated males (n = 9), and individuals within the size range of 350–400 mm FL (n = 32, 7.3% of sample), coupled with difficulties discerning between virgin versus inactive gonads in larger fish, it was not possible to fit a logistic regression to estimate a size at 50% maturity (L_{50}) for Tasmanian Snapper. However, there was evidence of Snapper in spawning condition with the smallest reproductively active female and male measured in our sample were 315 mm and 310 mm FL respectively, where both individuals were approximately four years of age.

We also found that the timing of reproduction of Snapper in Tasmania is unique, occurring in the austral summer, evidenced by a high proportion of mature gonads, suggesting that Snapper in Tasmania may require the higher temperatures of early austral summer for the onset of spawning. Current mean summer SSTs in Tasmania, range between 16.62°C (W)–19.12 °C (NEN), which suggest potential for spawning in the north only, however under future predictions SSTs in all regions (except the west coast) exceed 17.33 °C followed by autumns which exceed 15 °C in all regions, which may be sufficient to enable adequate spawning, settlement and therefore recruitment of Snapper in most regions of Tasmania under future warming. Understanding potential sources of recruitment could be resolved by the inclusion of a robust sample of Tasmanian fish into a molecular analysis of the population structure of Snapper in south-east Australia.

Species of high exploitation and variable recruitment such as Snapper are susceptible to age truncation, which can severely impact a population’s capacity to respond to environmental change or recover from sustained or increased fishing pressure. Management strategies to avoid age truncation include protecting the older/larger cohorts and ensuring sufficient recruitment into the fishery. Victoria has adopted this strategy by limiting harvest of larger individuals (> 40 cm) to 30% of the bag limit. Given that our models suggest that Snapper in Tasmania are most similar to those in Victoria (although slightly smaller and slower growing), we suggest that this strategy of limiting the harvest of larger individuals may also safeguard snapper in Tasmania from age truncation, as the proportion of large individuals (> 550 mm FL) was low (12%). However, the current MLS in Victoria is 280 mm TL, which equates to approximately 2.5 years. Our results suggest that there is no sign of reproduction near the MLS in Tasmania of 260 mm FL (300 mm TL). At this size, Snapper in Tasmania are approximately 3 years. Therefore, increasing the MLS to 380 mm TL (i.e. ~4 years) would offer some protection of the adult spawning population for at least one year before recruiting into the fishery.

King George Whiting

King George Whiting in this study ranged between 53–619 mm FL reaching a maximum age of 18.8 years–with males and females reaching similar maximum ages. However, we found marked differences in size and age structures between regions, and evidence of differential growth between sexes and regions. King George Whiting in Tasmania had a similar mean asymptotic length as the west regions of Western Australia for both males and females, and was larger than those for South Australia and Victoria and the southern regions of Western Australia. The smallest observed mature female and male measured 340 and 320 mm FL respectively and were both 2.9 years of age. Size and age at 50% maturity was estimated to be 407 and 403 mm FL and 4.4 and 4.6 years for females and males respectively. The MLS for King George Whiting in Tasmania is the largest of all states at 350 mm TL, which is below the estimated size at 50% maturity identified in this study (~400 mm FL, 425 mm TL).

We also found evidence of variable recruitment with strong cohorts in 2001, 2003, 2007, 2014, 2015 and 2017, and potentially weak recruitment in 2002, 2006, 2016 and 2018, potentially driven by changes in SST and increased flow rates of regional currents. Seasonal trends in reproductive activity suggest spawning between February to May and there are indications that there may be regional timing differences for the onset of spawning. Our data suggests similar maturity estimates of King George Whiting in Tasmania as populations in Western Australia and South Australia. Due to the genetic distinction between Tasmanian stocks and the Australian mainland, it appears that the observed increase in King George Whiting abundance in Tasmania is not a range extension from an Australian mainland stock, but rather a climate driven change on the coastal environment leading to more positive outcomes for the pre-existing Tasmanian adult stocks' reproductive activities and subsequent larval survival and recruitment within Tasmania.

The rapid increase in targeted fishing on King George Whiting is concerning as it is uncertain whether over the long term the relative abundance of large old fish will be maintained given the increase in fishing mortality on these stocks. If this fishing pressure is too large we may start to see the size and age structure of this stock mimic those seen on mainland Australia, where fish over the size of 450 mm TL or age of seven become uncommon. Therefore, continued monitoring is recommended. In 2016, South Australia implemented a series of fisheries management controls aimed at addressing increased fishing pressure including imposing a seasonal spawning closure, and is a recommendation of this study that Tasmania considers such measures if future evidence demonstrates that spawning individuals become exploited. Imposing a seasonal spawning closure in Tasmanian waters would assist in the protection of spawning aggregations, and therefore future recruitment into the fishery. The current MLS for King George Whiting is below the estimated size at 50% maturity of this study, and, if as expected, fishing pressure increases, a more precautionary management approach could include an increase in the MLS, and potentially temporal or spatial closures to afford greater protection of the spawning stock. More information and thus ongoing monitoring is needed to assess whether other management options such as, maximum size limits and or temporal/spatial closures are needed or possible in the future as the fishery develops.

There is also an ongoing debate for the management of species whose biological stock crosses state jurisdictions. This is particularly relevant for these range-extending species, where their biological stock is shared between mainland states. For example, the Yellowtail Kingfish in our study are small, immature and only present in Tasmania from November to April. Therefore, while continued monitoring is necessary in Tasmania, this information may be best used as the southern limit of the Eastern Biological Stock. As such, monitoring information of Yellowtail Kingfish in Tasmania is regularly provided in the Status of Australian Fish Stocks Report (Hughes et al. 2021).

Objective 3: Determine the diet composition of key range shifting fish species to refine parameterisation of an ecosystem model.

Diet analyses were completed for the three range shifting species. Cumulative prey density curves suggest that our sampling was sufficient to describe and quantify the diets of the three species adequately. Interestingly, all three species displayed a relatively generalist diet, meaning that food choice or resource availability may not be a limiting factor for these species at their range edge (Sunday et al. 2015). While there was a difference in diet between the north and southern stratums of Snapper, this difference was likely driven by the high abundance of doughboy scallops in the north, and the opportunistic feeding strategy by Snapper. Such opportunistic feeding regimes of all three study species may provide some buffering effect against current and future introduction of non-native invertebrate and vertebrate species which may also be extending their range.

Generally, our results align closely to what has been documented for these species on the mainland or congeners in the Atlantic. However, there is a lack of updated information on the basic biology, including diet of these three key species. Updated information from all jurisdictions would be useful as the climate is and has been changing, likely to cause shifts in habitat or resource use. Such information is critical as it forms the basis of larger ecosystem models such as Atlantis and would improve model accuracy. As such, the results from our diet analyses were used to parameterise the Atlantis South-East Ecosystem Model. And the results of this model suggest little ecosystem impact should these species increase their distribution across the Tasmanian domain. However, applying molecular methods of diet analysis may be useful to overcome some of the issues associated with the relatively small sample size of recreationally caught fish, and to taxonomically resolve the highly digested material beyond the level of class.

Objective 4: Develop species distribution models that utilise oceanographic climate change projections to predict the future presence and persistence of the key target species in Tasmania.

Here we have used species distribution models using occurrence records from both open access data bases and citizen science programs to identify areas of suitable habitat and predict how the suitability of this habitat may change under future warming for both a pelagic species, and two relatively site-attached, coastal species. As such our models predict an increase in habitat suitability for all three species, however with slightly different outcomes. For example, the temporal persistence of suitable habitat is predicted to increase for Yellowtail Kingfish—i.e. habitat is suitable in Tasmania for a longer period. For Snapper and King George Whiting, SDM's identified suitable habitat around the state since 1998, however the quality (i.e. suitability) of the habitat is predicted to increase under future change, more so in the winter for both King George Whiting and Snapper which has implications for the successful overwintering of new recruits, and therefore establishment into Tasmania.

The manner by which SDMs have been used in this study—by matching occurrence records to environmental data, and climate projections—have traditionally been used for highly migratory species in terrestrial systems or coastal-pelagic species in marine systems. These species groups use thermal habitat across large spatial scales, making them particularly suited to the application of SDMs. As such, in a recent review of Australian marine species redistributions, Gervais et al. (2021) highlighted that there is a bias towards coastal-pelagic fishes (such as: Black Marlin, Yellowtail Kingfish, Yellowfin Tuna, Southern Bluefin Tuna and Dolphinfish) as long-term (i.e. > 20 years) fishery-dependant data is abundant, and offshore habitats are ideal for the use of satellite-derived environmental covariates to predict spatial shifts. Here, using similar methodology, we went beyond the means of this more traditional use of SDMs to predict the potential shift in suitable habitat for coastal species. As such our results were much more highly resolved to the coastline, and data had to be interpolated to a smaller resolution (i.e. 0.004°) to account for the sensitivity of these species to small changes in depth. Nevertheless, using SDMs for these species was useful in identifying areas of suitable habitat and predicting potential shifts in this suitability, proving the usefulness of using SDMs for coastal species at the range edge. Furthermore, the partial plots for the effect of temperature derived from the SDMs provide a good reference to match to the documented preferred thermal habitat for young recruits and juveniles. Such information is useful as this life-history stage is often limiting to the persistence and establishment of a species.

Adding local environmental habitat predictors in SDMs has been suggested to improve the model's predictive accuracy but these improvements are often difficult to quantify (Hazen et al. 2021). By calculating the distance to a key habitat for King George Whiting—seagrass, from open access mapped data, we were able to include proximity to seagrass as a predictor in the King George Whiting SDM. Furthermore, by rasterizing these distances by grid cell we were able to stack this variable with the oceanographic variables to make spatial predictions in habitat suitability. This method is novel in its use, and through model selection and validation we have demonstrated that using a local environmental predictor is not only possible but improves predictive accuracy and skill. However, a limitation to this method is the assumption that this habitat will remain stable under future change. Many marine foundation species (i.e. macroalgae, kelps, seagrass, corals) have already begun to shift their distribution in response to change (Pecl et al. 2017). Therefore, the next logical step in building a more comprehensive SDM is to include projected spatial change in habitat types, in addition to projected climate change. Lastly, while the SDMs of all three species predict an increase in habitat suitability within the Tasmanian domain under future climate change, our models only account for predicted changes to sea surface temperature (Yellowtail Kingfish, Snapper and King George Whiting), eddy kinetic energy and sea level anomaly (Yellowtail Kingfish), and as such they do not account for other environmental changes associated with climate change, which may dampen predicted increases to habitat suitability. More complex models which include predicted changes to ocean biogeochemistry, which affect an organism's performance (i.e. pCO_2, salinity, and oxygen) would provide more comprehensive predictions of habitat suitability, but were beyond the scope of this study.

Objective 5: Utilise the Atlantis ecosystem model framework to predict ecological impacts of increasing abundance of key range shifting fish species in Tasmania.

Marine ecosystem models have been demonstrated to be useful tools to describe complex interrelationships between species and how ecosystems can be impacted by climate change. Their extension to look at ecological impacts of changes in species distributions is relatively new, due to challenges related to tracking fine-scaled spatial dynamics. This study used a biogeochemical and whole of ecosystem Atlantis model of south-eastern Australia to run a series of climate change and species distribution scenarios, representing incremental biomass changes down the east and southern parts of Tasmania. The Atlantis South-East Australian model was very responsive to climate change with biomass projected to greatly increase for King George Whiting with more variable spatial dynamics observed for Yellowtail Kingfish and Snapper. This result compliments that of the SDMs, where habitat suitability is also predicted to increase across all of the state for King George Whiting and Snapper, and the temporal persistence of suitable habitat in south-east Tasmania is predicted to increase for Yellowtail Kingfish under IPCC RCP8.5. Most of the temporal and spatial dynamics were determined by trends in the primary producers including picoplankton and macrophytes. The effects of species redistribution of the three focal species to the whole of ecosystem model were less pronounced than those of projected climate change. Single species redistribution scenarios typically only showed changes in relative abundance of less than 5% (in either direction) of trophic groups or species, with the species most negatively impacted including dominant prey items or potential competitors. Therefore, under both climate change and species redistribution scenarios, there wasn't evidence of any ecosystem collapse. However, there were unusual interactions between the three species, with scenarios of species distribution for Yellowtail Kingfish contrasting somewhat to those for Snapper and King George Whiting. Therefore, a level of caution is warranted when interpreting outputs of ecosystem models to examine the broader ecological effects of species distribution. This is particularly true for highly responsive models that represent bottom-up trophic processes and tightly connected systems.

Recommendations

While the data collection for this project has been successful in determining baseline information for key range shifting species in Tasmania, ongoing monitoring is necessary to determine potential changes to age/size structure, recruitment variability, and areas of potential spawning/recruitment. Not only is this important to gain a more accurate representation of population dynamics of these three species in Tasmania, but to be ready and flexible to implement proactive (rather than reactive) strategies to sustainably manage these emerging fisheries.

Ongoing monitoring would therefore require continued engagement with the citizen science community. Redmap is particularly successful in identifying species at the range-edge or novel environment, as a ‘canary in the coalmine,’ raising an alert and an awareness surrounding a new species in a new environment. However, interest/engagement wavers once the novelty wears off, or knowledge surrounding this species becomes commonplace. Therefore, collaborating and working alongside programs where data collection is more routine, such as the ‘Tassie Fish Frame Collection Program’, is useful for sustained engagement and data collection of those species which are known to be in Tasmania, but for which data is still limited (i.e. Snapper, King George Whiting, Yellowtail Kingfish, Striped Trumpeter, Sand Flathead, Silver Trevally). Therefore, the complementarity between these two citizen science initiatives can improve the quality and scope of data collection of range extending species.

Further scope to engage across a suite of citizen science programs would be beneficial for collecting data for range extending species beyond the recreational fishing community. Rapid advances in technology allows for anyone with an interest to be able to sensor their environment and collect data. An online hub/central source for all marine citizen science programs would be useful for the general public to engage and provide information irrespective of their interests or abilities (i.e. photography, diving, fishing, education, clean-ups). Such a central hub would not only reach a wider audience but allow for different institutions/initiatives to collaborate and identify gaps within their own program which could be supplemented within programs elsewhere. Cross-pollination across different citizen science initiatives provides the opportunity to share resources, and ultimately maximizes community outreach and data collection efforts.

To improve the existing ‘Tassie Fish Frame Collection Program’, making fish frame donations as easy and simple as possible would not only attract a greater audience/community, but could improve the quality of the fish frames donated. For example, if each drop off point had its own freezer capacity, it would take the responsibility away from the fisher to freeze their samples, and as such would be inclusive to those who do not have access to a freezer. Improved communication on processing of samples where possible to retain any visceral material would be beneficial to maximise tissue samples for analyses. And lastly feedback is important to retain engagement with the fishing community. Increased scope to effectively communicate with the existing participants/donors, to regularly keep participants informed of the data that they have provided would not only increase scientific awareness, but also foster a sense of community and purpose amongst participants.

While engagement with citizen science and the recreational fishing community has been fundamental to the success of this project, the importance of fisheries independent sampling cannot be overlooked. The major caveat around fisheries-dependant sampling arises from size limits, and angler bias. For example, engagement with experienced anglers who only target larger fish may lead to a large size bias in our sample. Or, minimum size limits restricts the community from donating small or juvenile fish. Lastly, there is a need to sample from a range of areas to get an understanding of preferred habitat, and potential spawning/recruitment areas, especially for species at the range edge. These areas often include difficult to access locations, marine reserves, and during spawning closures. Therefore, fisheries independent sampling is essential to fill these gaps. As such we suggest further netting for juvenile King George Whiting in sheltered embayments along the north and north-east coast of Tasmania, to identify potential hotspots for recruitment. As we found observed regional differences in mean size at age of King George Whiting, it is unclear whether these are due to intrinsic or environmental influences on growth rates or simply reflect differences in reproductive events between regions or a combination of factors. As such, future studies should clarify regionally specific birthdates, key recruitment periods and the timing of the first annuli between regions to ensure an individual's decimal age is regionally appropriate.

Understanding genetic connectivity between fish caught in Tasmania, versus the mainland would help resolve many issues in jurisdictional management, and sources for recruitment. While evidence from mature gonads suggests that Snapper are reproductively active, it remains unclear whether these populations in Tasmania are a viable source of recruitment. Furthermore, as the climate continues to change, connectivity between populations is also likely to be a moving feast. Therefore, the first logical step is to determine current genetic connectivity between Tasmanian and mainland populations as a baseline understanding of stock structure, followed by continued analysis as the Tasmanian population is likely to grow, and extend southward.

Extension and Adoption

Online extension

The Tassie Fish Frame Collection Program and associated Facebook page was launched in December 2019 with the intention of providing an online engagement point with the Tasmanian recreational fishing community. The page allows us to efficiently push out information on how to get involved, project news, interesting findings, as well as requests for frames or anecdotal information to fill knowledge gaps (see Appendix B for examples). The page also allows two-way communication with the public via post comments and private messages and helps build positive rapport with fishers.
Over time the Facebook page has grown in popularity, with significant numbers of people engaging with the page and the various posts. Aided by posts shares from project partners (e.g. NRE Tas, TARFish, IMAS, Redmap) and relevant Facebook groups (e.g. Tasmanian Snapper, Tasmanian King George Whiting and Tassie Kingfish), a following of over 1,000 Facebook users has been reached (Table 21). Upon the completion of this report, information about the major findings will be released parts via the Facebook page to feed information back to the fishing community.

Table 21. Indicators of engagement on the Tassie Fish Frame Collection Facebook page (December 2019–September 2021).

Total number of page followers	Total number of post engagements (reactions, comments, shares, clicks)	Total number of post reactions (e.g. Like, Love, Haha, Wow, Sad or Angry)	Total number of comments	Total number of post shares
1,077	22,593	2,970	734	394

On-ground and face-to-face extension

In order to broaden the reach of the Tassie Fish Frame Collection Program, the project team have conducted on-ground and face-to-face extension. Shortly after the program launched, promotional signage was placed at a number of boat ramps around the state detailing the target species, how to get involved and how the data would be used (Appendix C). The team also conducted opportunistic, face-to-face conversations with recreational fishers at popular fishing locations, boat ramps and in tackle stores.

In addition, the project team also presented at angling club meetings for the Van Dieman Fly Fishers Club in Launceston and the Break O’Day Sports Angling Club in St Helens (Appendix C). These sessions allowed us to network and engage face-to-face with anglers by showing them interesting insights from the project. It also allowed us to demonstrate the value of frames donated by anglers to the project and promote the further donation of frames. Overall, we received very positive feedback from those attending.

Project coverage

The project had radio, newspaper and free-to-air television coverage. Project leader, and Primary Investigator Associate Professor Sean Tracey was interviewed by Leon Compton on the ABC Radio

Hobart Morning show on 6/2/2020. This provided the opportunity to promote the project and encourage participation by recreational fishers.

Tasmanian newspaper The Mercury ran an article titled ‘Your frames paint a picture’ by local fishing identity Craig Hyland on 12/3/2021. The article promoted the Tassie Fish Frame Collection Program and included a step-by-step guide on how to get involved (Appendix D).

National fishing identity Paul Worsteling featured a piece about the Tassie Fish Frame Collection Program on Series 15 Episode 25 which aired on 10 BOLD on 18/04/2021. In the episode, Paul fishes with well-known Tasmanian angler Jamie Harris to catch King George Whiting and promote how to prepare frames for donation. The IFISH w TACKLEWORLD Facebook followed up with posts about the data collected from the donated frames, reaching an audience of 500,000+ followers.

Project materials developed

Peer-reviewed publications

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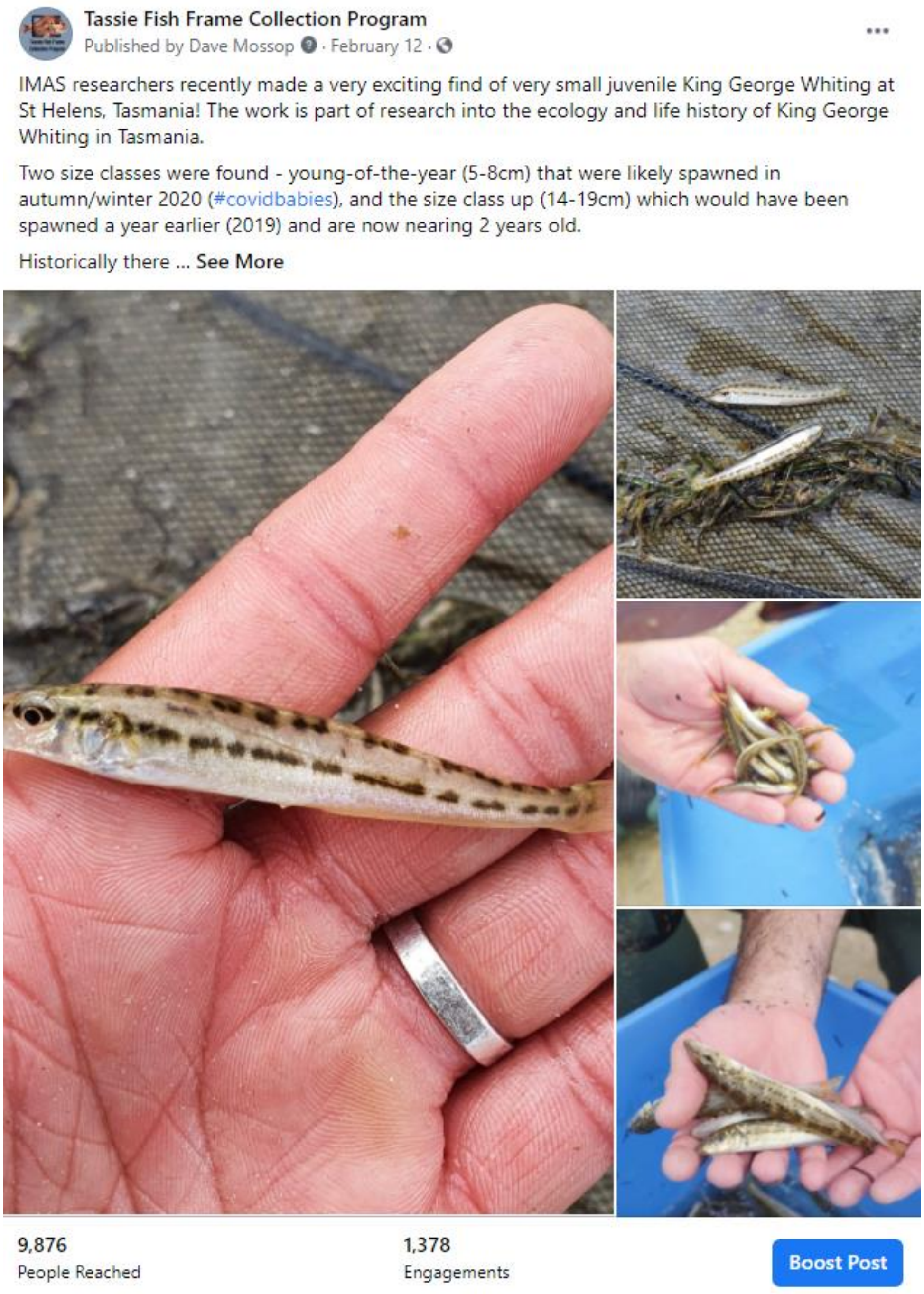
Appendix A: Tassie Fish Frame Collection Program drop-off locations around Tasmania

Table A1. Locations and drop-off points for the Tassie Fish Frame Collection Program.

North & North-West	South	East
• Tackleworld (Burnie) • Coastal Marine (Somerset) • Deegan Marine (Ulverstone) • Seamaster (Devonport) • BCF (Devonport) • Tamar Marine (Launceston) • BCF (Launceston) • Anaconda (Launceston)	• Fisherman's Shed (Kingston) • Spot On (Hobart) • Franklin Marine (Franklin) • Anaconda (Cambridge) • BCF (Cambridge)	• St Helens Slipway & Marine Shop • Morris' Store (Swansea) • Freycinet National Park Visitor Centre (Freycinet)

Appendix B: Online extension through the Tassie Fish Frame Collection Program Facebook page

The following Facebook post (12/02/2021) details the exciting find by the project team of juvenile King George Whiting (*Sillaginodes punctatus)*, which is the first time such numbers of young recruits had ever been recorded in Tasmanian waters.



Tassie Fish Frame Collection Program

Published by Dave Mossop · February 12 ·

IMAS researchers recently made a very exciting find of very small juvenile King George Whiting at St Helens, Tasmania! The work is part of research into the ecology and life history of King George Whiting in Tasmania.

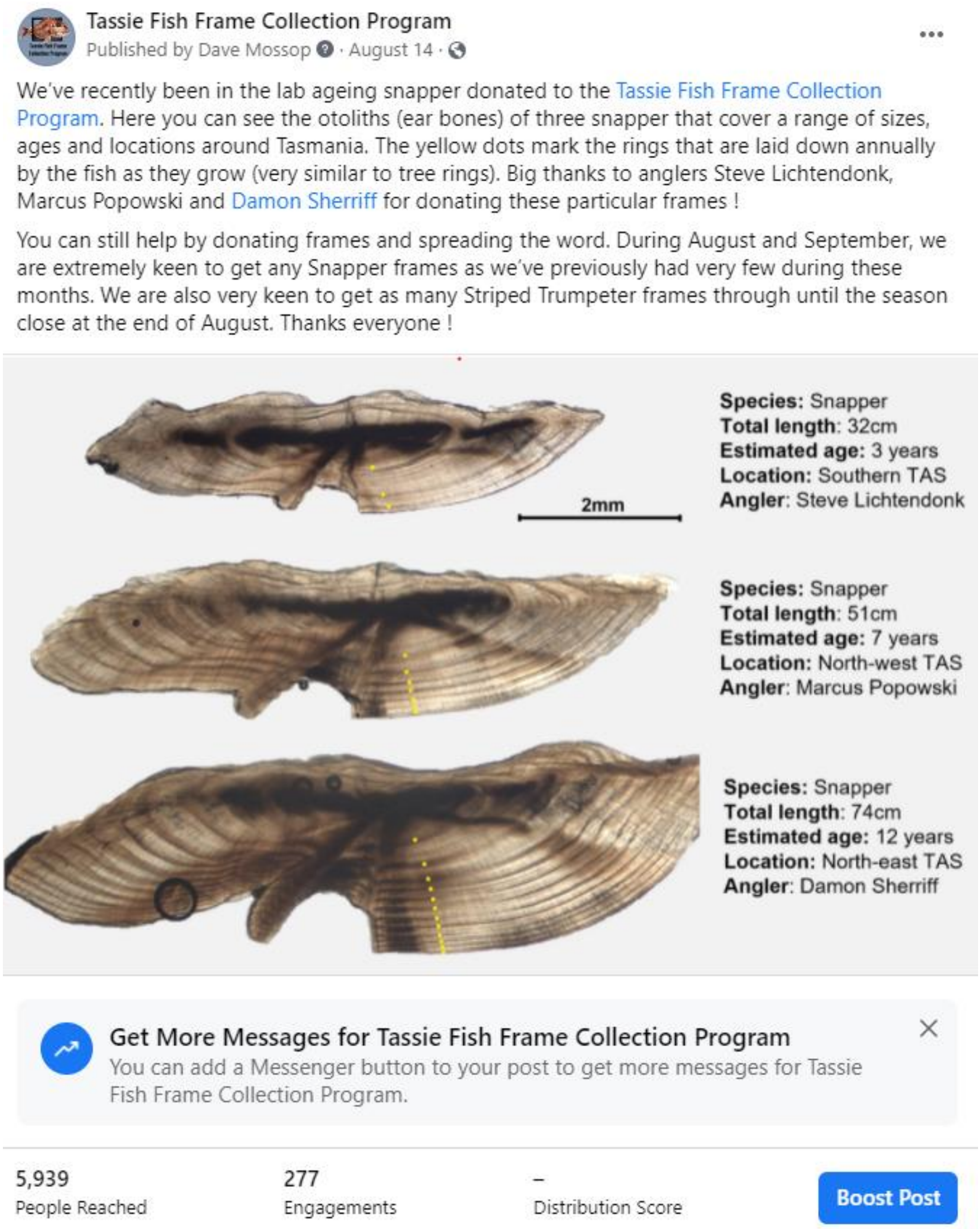
Two size classes were found - young-of-the-year (5-8cm) that were likely spawned in autumn/winter 2020 (#covidbabies), and the size class up (14-19cm) which would have been spawned a year earlier (2019) and are now nearing 2 years old.

Historically there ... **See More**

9,876
People Reached

1,378
Engagements

Figure B1. Facebook post (12/02/2021) of juvenile King George Whiting (*Sillaginodes punctatus)*, in Tasmanian waters.



Tassie Fish Frame Collection Program

Published by Dave Mossop · August 14 ·

We've recently been in the lab ageing snapper donated to the Tassie Fish Frame Collection Program. Here you can see the otoliths (ear bones) of three snapper that cover a range of sizes, ages and locations around Tasmania. The yellow dots mark the rings that are laid down annually by the fish as they grow (very similar to tree rings). Big thanks to anglers Steve Lichtendonk, Marcus Popowski and Damon Sherriff for donating these particular frames !

You can still help by donating frames and spreading the word. During August and September, we are extremely keen to get any Snapper frames as we've previously had very few during these months. We are also very keen to get as many Striped Trumpeter frames through until the season close at the end of August. Thanks everyone !

Species: Snapper
Total length: 32cm
Estimated age: 3 years
Location: Southern TAS
Angler: Steve Lichtendonk

Species: Snapper
Total length: 51cm
Estimated age: 7 years
Location: North-west TAS
Angler: Marcus Popowski

Species: Snapper
Total length: 74cm
Estimated age: 12 years
Location: North-east TAS
Angler: Damon Sherriff

Get More Messages for Tassie Fish Frame Collection Program

You can add a Messenger button to your post to get more messages for Tassie Fish Frame Collection Program.

5,939	277	–
People Reached	Engagements	Distribution Score

Figure B2. Facebook post (14/08/2021) shows the otoliths and respective ages from three fish donated by fishers in various locations in Tasmania.

Appendix C: On-ground and face-to-face extension

The poster below was used as signage various at various boat ramps around Tasmania to promote the Tassie Fish Frame Collection Program. It includes information on the target species, how to donate the frames and where to drop them off. *Note: Striped Trumpeter is an additional species of scientific interest to IMAS but not part of the current project.*

★★★ WANTED ★★★

We need your help to collect samples!

We are studying the age, growth & distribution of these species in Tasmanian waters

- If you catch and keep any of these species, we would like your fish frames (preferably with gut and gonads intact)
- Frames will be used to determine age, size, sex, reproductive stage
- Freeze frames in a plastic bag with a label inside the bag detailing:
  1) Angler contact details 2) Date of capture 3) Approx. area of capture

FRAME DROP OFF LOCATIONS*:

NORTH

- Tackleworld (Burnie)
- Coastal Marine (Somerset)
- Deacon Marine (Ulverstone)
- BCF (Devonport)
- CH Smith (Launceston)
- Tamar Marine (Launceston)
- BCF (Launceston)
- Anaconda (Launceston)

SOUTH

- Fisherman's Shed (Kingston)
- Spot On (Hobart)
- Franklin Marine (Franklin)
- Anaconda (Cambridge)
- BCF (Cambridge)

EAST COAST

- East Coast Tackle (St Helens)
- Slipway & Marine Shop (St Helens)
- Morris' Store (Swansea)
- Freycinet National Park Visitor Centre (Freycinet)

*drop off during business hours only

For more info, take a photo of this poster and contact:

- See 'Tassie Fish Frame Collection Program' Facebook page
- St Helens & north TAS: James.Haddy@utas.edu.au 6324 3828
- Anywhere south of St Helens: David.Mossop@utas.edu.au 6226 1839

Tassie Fish Frame Collection Program

UNIVERSITY of TASMANIA

IMAS INSTITUTE FOR MARINE & ANTARCTIC STUDIES

Fish illustrations by Peter Gouldthorpe. Copyright DPIPWE.

Dr James Haddy (below) presenting to the Break O’Day Sports Angling Club in St Helens on how donated fish frames provide important data for the project, using Snapper as an example.

Appendix D: The Mercury newspaper article 'Your frames paint a picture' by Craig Hyland

Your frames form a picture

CARL HYLAND

RECREATIONAL fishers are important citizen scientists that can hugely help researchers by donating fish frames from their catch to the Institute for Marine and Antarctic Studies, who are currently investigating how warming waters might be leading these species' further south, potentially increasing recreational fishing opportunities in Tasmanian waters.

Researchers are interested in the diet and reproduction of kingfish, snapper, whiting and other species from Tasmania, so fishers are encouraged to donate frames with the stomach and gonads intact if possible.

This year IMAS has added one of Tasmania's favourite fish to the Tassie Fish Frame Collection Program – the Sand Flathead. It is a species that researchers are keen to understand more about across its Tasmanian range, and you can help by providing fish frames.

Similar to other target fish frame species, the flattie collection program works as follows:

1. Save the frames of sand flathead you catch and keep within Tasmania (everything leftover after filleting – head, body and organs attached).
2. This includes any legal-size (32cm+) sand flathead (i.e. not just your biggest, all sizes are important).
3. Put frames in a plastic bag along with a label (labels also available at drop off locations) The information required on the label includes – the angler's name and contact details (phone or email); location caught (can include nearest boat ramp); date caught; and the average water depth that fish were caught.
4. Drop the frames off at one of 16 drop-off locations around Tasmania for the full list go to https://www.facebook.com/TassieFishFrameCollectionProgram/photos pcb.270497457762944/270496334429723/

IMAS scientists will then process the frames to determine the age, size and sex of fish to help understand more about sand flathead populations in Tasmania.

Disposal of fish frames and other matter is often an issue pondered by fishers when at the boat ramp and one idea that seems to be taking off is recycling the normally discarded skin, bones and offal. This can be done by collecting all of your fish scraps after each outing and freezing them in manageable chunks, then putting them through the garden shredder or a friend's shredder if allowed. The resulting chips are an absolutely awesome product for use in berley but go even better in the garden when added to the compost or mulch. Add a small amount of fish oil and refreeze and you have one of the best fish attracting baits about.

A word of warning with the fish frames if frozen, if placing through the chipper frozen they can become missiles if bouncing around on top of the blades. Plus if you want to ensure the process remains smell free an application of WD-40 will help and also ensure it doesn't rust (WD-40 is mainly fish oil).

Some use shredder or mincers on the side of their boat to apply berley straight into the water but this is a time-consuming process and others have choppers mounted on the back of their bigger boats for tuna frames and the like. These are chopped up usually when fishing for sharks and the resulting oil and mash is a great attractor.

Some add chicken pellets or even fish pellets to their mix and that seems to work well. Some fish processors will sell blocks of berley frozen to fishers and they are well worth approaching if you want bulk berley. Obviously, a freezer would be required to store frozen blocks.

One thing I will say about berley is that it will attract the small pickers, so be prepared to catch gurnard, toad fish and other unwanted by catch.

Oliver Wolfert caught this bream — measuring 38cm — off a soldier crab, at Dolphin Sands in the river.

Appendix E: Atlantis Ecosystem model supplemental material

Table E1. The species composition of all functional groups used in the Atlantis ecosystem model.

Group Name	**Group composition**	**Model code**
Large planktivorous fish	*Trachurus declivis, Scomber australasicus, Emmelichthys nitidus, Trachurus novaezelandiae*	FPL
Blue grenadier	*Macruronus novazelandiae*	FBG
Small planktivorous fish	*Sardinops sagax, Engraulis australis, Hyperlophus vittatus, Spratelloides robustus,Trachurus novaezelandiae, Argentina australiae*	FPS
Ocean plank/pisc fish	Exocoetidae, Scomberesocidae	FVD
Shallow piscivorous fish	*Arripis trutta, Thyrsites, Dinolestes lewini, Arripis georgianus, Pomatomus saltatrix, Sphyraena, Euthynnus affinis, Atractoscion aequidens, Sarda australis, Coryphaena hippurus, Argyosomus hololepidotus, Cheilodactylus spectabilis*	FVS
Shallow demersal reef fish	*Trachinops caudimaculatus,* Labridae, Monacanthidae, *Atypichthys latus, Scorpis aequipinnis, Enoplosus armatus, Caesioperca lepidoptera, Pempheris multiradiata, Scorpis lineolata*	FDR
Yellowtail kingfish	***Seriola lalandi***	**FVT**
Epipelagic fish feeders	*Auxis thazard, Thunnus albacares, Kadjikia audax*	FTE
Mesopelagic fish feeder	*Thunnus alalunga, Xiphias gladius, Thunnus obesus*	FTM
Flatheads	Platycephalidae	FFH
Migratory mesopelagics fish	Myctophids	FMM
Non-migratory mesopelagics fish	Sternophychids, cyclothene (lightfish)	FMN
King Gorge Whiting	***Sillaginodes punctatus***	**FKG**
Deep demersal fish	Zeidae, Cyttidae, *Genypterus blacodes, Rexea solandri, Polyprion oxygeneios, Paraulopus nigripinnis, Rexea antefurcata, Coelorinchus, Oreosomatidae*, Macrouridae, Zenopsis, Centroberyx, cardinalfish, *Mora moro*	FDD
Shallow demersal herbivores	*Kyphosus sydneyanus, Girella elevata, Hyporhamphus melanochir, Girella tricuspidata, Dactylophora nigricans*, Aplodactylidae, Mugilidae	FDH
Other shallow demersal fish	Paralichthyidae, Pleuronectidae, Triglidae, Rhabdosargus sarba, Pseudophycis bachus, Lotella, Pseudophycis, Acanthopagrus butcheri, Pseudocaranax georgianus, Acanthopagrus australis, Sillago, Zeus faber, Helicolenus percoides, Hime purpurissatus, Batrachoidiformes, *Cheilodactylus nigripes,* Nemadactylus, Sebastes alutus, Pterygotrigla, Uranoscopidae, *Scolecenchelys breviceps*	FDS
Shallow territorial demersal fish	Hippocampus, *Phycodurus eques, Phyllopteryx taeniolatus*, Stigmatopora, Gobiidae, Pomacentridae, *Monodactylus argenteus*	FDT
wrasse	*Notolabrus tetricus, Pseudolabrus fucicola*	FBW
Snapper	***Pagrus auratus***	**FSN**
Warehous and trevalla	*Seriolella, Hyperoglyphe Antarctica*	FWT
Striped tuna	*Katsuwonus pelamis*	FST
Coastal sharks	*Squalus megalops, Mustelus antarcticus*	SHG
Demersal sharks	*Heterodontus portusjacksoni*, Scyliorhinidae, Orectolobidae *Centrophorus spp.*	SHD
Pelagic sharks	*Prionace glauca, Isurus oxyrunchus, Carcharodon carcharias, Carcharhinus*	SHP

Group Name	**Group composition**	**Model code**
Skates and rays	Rajidae, Dasyatidae	SSK
Seabirds	Diomedeidae, *Puffinus*, Laridae, *Morus*, *Eudyptula minor*	SB
Urchins		BGU
Pinnipeds	*Arctocephalus pusillus doriferus, Arctocephalus forsteri*	PIN
Abalone	*Halioteuthis laevigata*, *Halioteuthis rubra*	BGA
Baleen whales	*Megaptera novaeangliae, Balaenoptera, Eubalaena australis*	WHB
Dolphins	Delphinidae	WHS
Orcas	*Orcinus orca*	WHT
Lobster	*Jasus edardsii, Jasus verreauxi*	BRL
Cephalopods	*Ommastrephes bartramii, Todarodes filippovae, Stenoteuthis oualaniensis, Sepioteuthis australis, Nototodarus gouldi, Sepia apama*	OMM
Octopods	*Octopus pallidus, Octopus tetricus,*	OCT
Shallow benthic filter feeders	mussels, oysters, sponges, corals	BFF
Deep benthic filter feeders	holothurians, echinoderms, burrowing, bivalves	BFD
Macrozoobenthos	stomatopods, asteroids, gastropods, crustaceans	BMD
Crabs		BMS
Prawns		PWN
Carnivorous zooplankton		ZL
Deposit Feeder		BD
Macroalgae	Kelp	MA
Seagrass		SG
Benthic Carnivore	Polychaetes	BC
Gelatinous zooplankton		ZG
Diatom		PL
Pico-phytoplankton		PS
Mesozooplankton	Copepods	ZM
Microzooplankton	Flagellates	ZS
Pelagic Bacteria		PB
Sediment Bacteria		BB
Meiobenthos		BO

Appendix F: SDM supplemental material

General Methods

Table F1. Variance inflation factors for predictors of best ocean suitability model for King George Whiting (*Sillaginodes punctatus*) and Snapper (*Chrysophrys auratus*) (VIFs < ~1.5 indicates that collinearly is unlikely to affect model performance).

Species	SST	Depth	Distance to seagrass
King George Whiting	1.007	1.007	1.002
Snapper	1.083	1.083	

Table F2. Aikaike Information Criterion (AIC) and degrees of freedom (df) used for model selection for the best generalised additive mixed model for to predict the preferred habitat for King George Whiting (*Sillaginodes punctatus*) where SST = sea surface temperature, distance_Seagrass = distance to seagrass (m), and depth (m). All models use year as a random factor to account for variation in catch between years, and all models are fitted with a binomial distribution with a logit link. Best model indicated in bold.

Model	*df*	AIC
pa ~ s(SST) + s(depth) +s(distance_Seagrass)	8	447.5259
pa ~ s(SST) + s(depth)	6	504.0626
pa ~ s(depth) +s(distance_Seagrass)	6	449.4738
pa ~ s(SST) + depth + distance_Seagrass	**6**	**443.5259**

Table F3. Aikaike Information Criterion (AIC) and degrees of freedom (df) used for model selection for the best generalised additive mixed model for to predict the preferred habitat for Snapper (Chrysophrys auratus) where SST = sea surface temperature, and depth (m). All models use year as a random factor to account for variation in catch between years, and all models are fitted with a binomial distribution with a logit link. Best model indicated in bold.

Model	*df*	AIC
pa ~ s(SST) + s(depth)	6	6976.15
pa ~ s(SST, k=4) + s(depth)	**6**	**7173.58***
pa ~ s(SST) + depth	5	7103.90
pa ~ s(SST)	4	13995.63

Table F4. Details of CMIP5 models downscaled (0.05°) to support projections of suitable habitat for King George Whiting (*Sillaginodes punctatus*) and Snapper (*Chrysophrys auratus*).

Model	Institution	Native ocean resolution (°)
ACCESS1.0	CSIRO-BOM, Australia	1.0×1.0
CanESM2	CCCMA, Canada	1.4×0.9
CNRM-CM5	CNRM-CERFACS, France	1.0×0.8
GFDL-ESM2M	NOAA, GFDL, USA	1.0×1.0
HadGEM2-CC	MOHC, UK	1.0×1.0

Table F5. Model selection for generalised linear models to assess change in habitat suitability of a) King George Whiting (*Sillaginodes punctatus*) and b) Snapper (*Chrysophrys auratus*) from 1998–2018 across four seasons within each region. Best model indicated in bold.

Species	Model	*df*	AIC
a) King George Whiting	Year * season * region	48	810958.4
	Year + season * region	**25**	**810927.9****
	Year * season + region	13	811110.8
	Year + season + region	10	811108.1
b) Snapper	**Year * season * region**	**48**	**4824659****
	Year + season * region	25	4826810
	Year * season + region	13	4833121
	Year + season + region	10	4833895

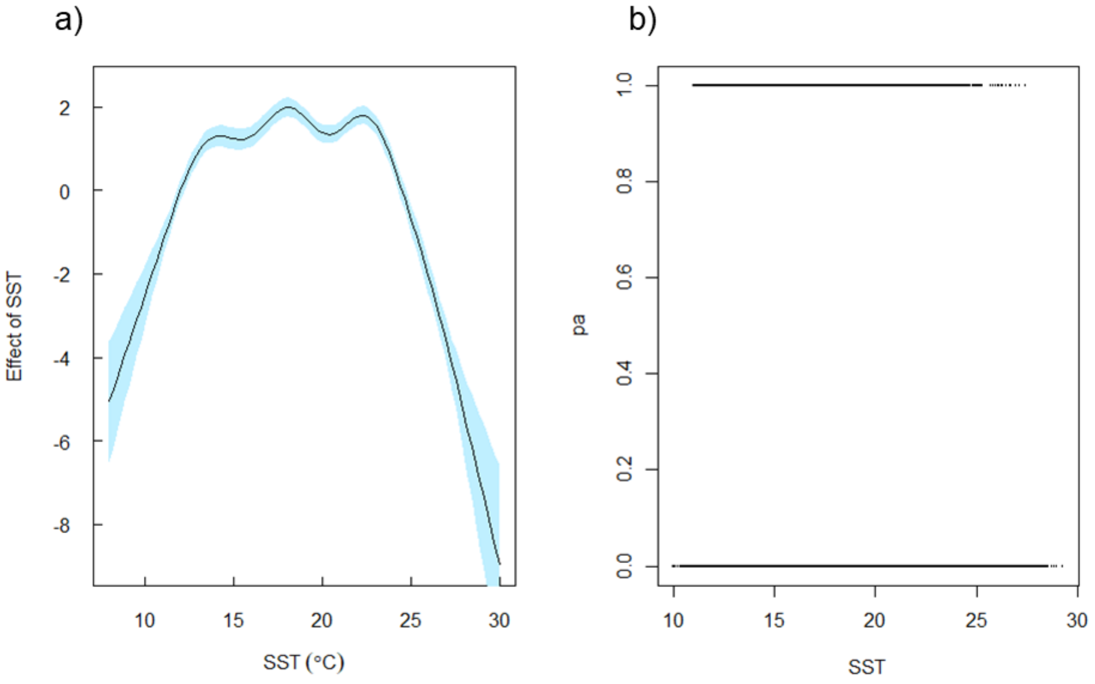


Figure F1. Evidence of model overfitting in GAMM without knots applied to the SST smoothing term for the Snapper SDM: pa ~ s(SST) + s(depth) + (1|year). a) SST Partial plot from Snapper GAMM, and b) proportion of species presence (pa=1), and pseudoabsences (pa=0), across the SST range.

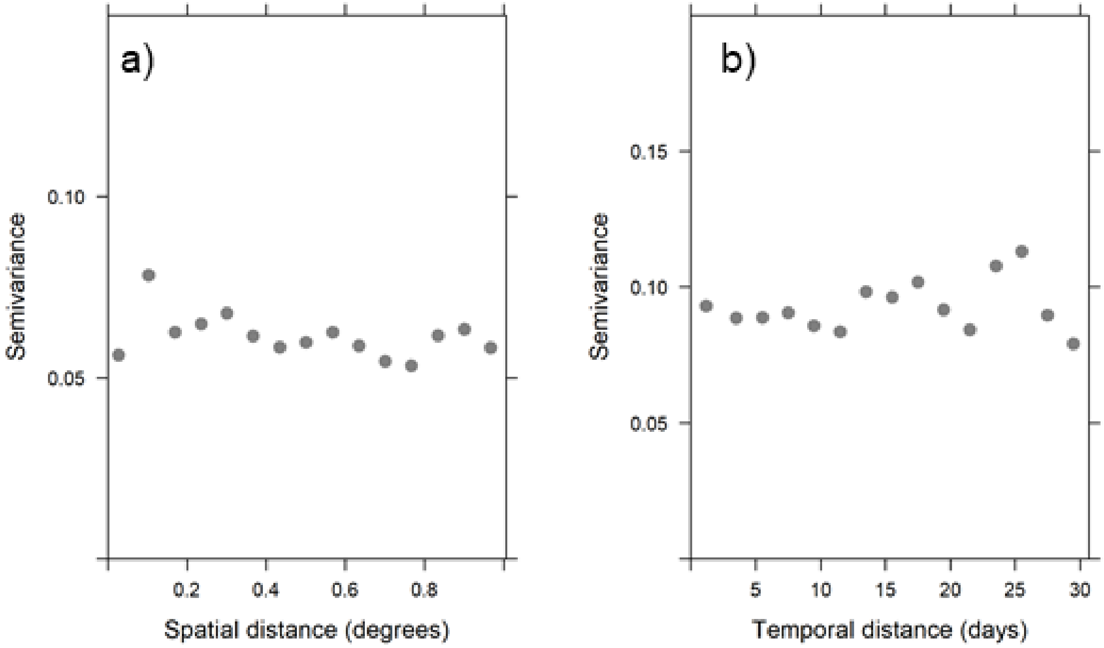


Figure F2. Spatial and temporal semivariograms to assess a) spatial and b) temporal autocorrelation for the data used for the optimal habitat suitability model for King George Whiting (*Sillaginodes punctatus*) (i.e. applying spatial thinning of 10 km per day).

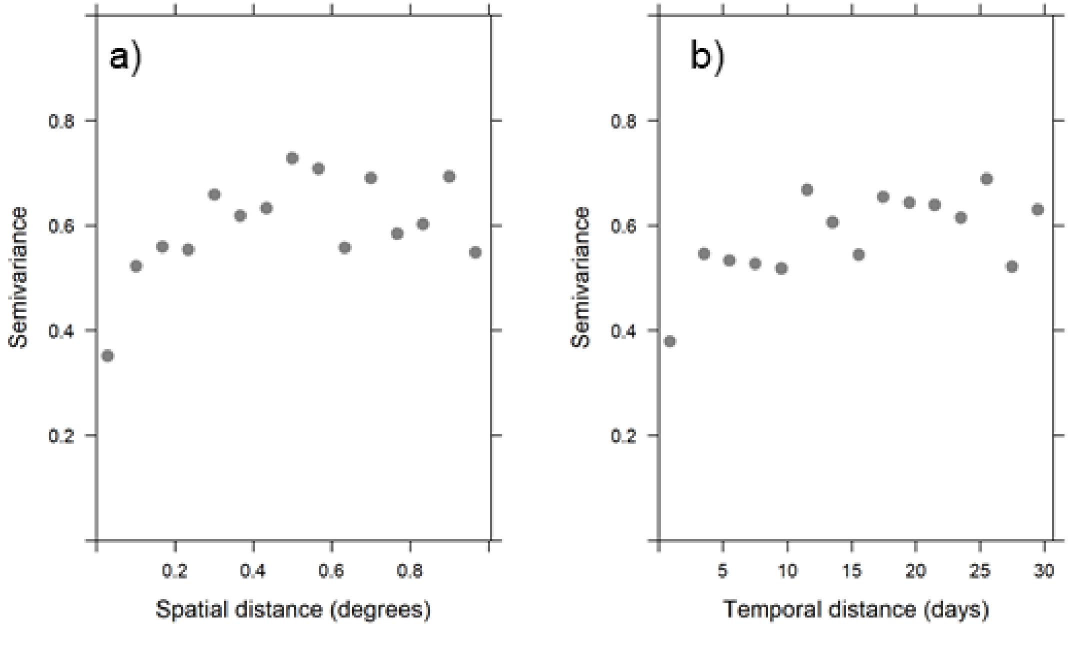


Figure F3. Spatial and temporal semivariograms to assess a) spatial and b) temporal autocorrelation for the data used for the optimal habitat suitability model for Snapper (*Chrysophrys auratus*) (i.e. applying spatial thinning of 20 km per day).

Results: Snapper SDM

Table F6. Model summary for generalised linear model assessing the effect of year, season and region to the habitat suitability of Snapper (*Chrysophrys auratus*) between 1998–2018. (suitability ~ year * season * region) Results are on the link scale.

Factor	**Estimate**	***SE***	***z* value**	***p***
(Intercept)	-3.687	2.947	-1.251	0.211
year	0.001	0.001	0.524	0.600
Summer	-15.490	3.533	-4.386	<.001*
Autumn	-8.367	3.596	-2.327	0.012*
Winter	9.138	4.277	2.136	0.033*
NE	-1.242	5.450	-0.228	0.820
NEN	-0.571	3.210	-0.178	0.859
NW	-2.285	3.359	-0.680	0.496
SE	-2.927	3.756	-0.779	0.436
W	-1.163	4.057	-0.287	0.774
year:summer	0.008	0.002	4.697	<.001*
year:autumn	0.005	0.002	2.577	0.01*
year:winter	-0.005	0.002	-2.163	0.031
year:NE	0.001	0.003	0.218	0.827
year:NEN	0.000	0.002	-0.011	0.991
year:NW	0.001	0.002	0.518	0.604
year:SE	0.001	0.002	0.752	0.452
year:W	0.000	0.002	0.230	0.818
Summer:NE	8.964	6.529	1.373	0.170
Autumn:NE	0.044	6.611	0.007	0.995
Winter:NE	-6.444	7.836	-0.822	0.411
Summer:NEN	7.642	3.832	1.994	0.046*
Autumn:NEN	-2.186	3.899	-0.561	0.575
Winter:NEN	-5.702	4.668	-1.221	0.222
Summer:NW	10.380	4.013	2.586	0.01*
Autumn:NW	1.351	4.080	0.331	0.740
Winter:NW	-19.080	4.866	-3.921	<.001*
Summer:SE	-12.190	4.506	-2.704	0.007*
Autumn:SE	2.212	4.613	0.479	0.632
Winter:SE	16.250	5.486	2.962	0.003*
Summer:W	-6.525	4.918	-1.327	0.185
Autumn:W	-19.370	4.991	-3.880	<.001*
Winter:W	-22.100	5.761	-3.836	<.001*
year:Summer:NE	-0.004	0.003	-1.376	0.169
year:Autumn:NE	0.000	0.003	0.005	0.996
year:Winter:NE	0.003	0.004	0.831	0.406
year:Summer:NEN	-0.004	0.002	-2.009	0.045*
year:Autumn:NEN	0.001	0.002	0.565	0.572
year:Winter:NEN	0.003	0.002	1.212	0.225
year:Summer:NW	-0.005	0.002	-2.655	0.008*
year:Autumn:NW	-0.001	0.002	-0.377	0.707
year:Winter:NW	0.010	0.002	3.930	<.001*
year:Summer:SE	0.006	0.002	2.707	0.007*

Factor	**Estimate**	***SE***	***z* value**	***p***
year:Autumn:SE	-0.001	0.002	-0.505	0.614
year:Winter:SE	-0.008	0.003	-2.975	0.003*
year:Summer:W	0.003	0.002	1.263	0.207
year:Autumn:W	0.010	0.002	3.840	<.001*
year:Winter:W	0.011	0.003	3.871	<.001*

Table F7. Results of pairwise contrasts of generalised linear model assessing the effect of year, season and region to the habitat suitability of Snapper (*Chrysophrys auratus*) between 1998–2018. (suitability ~ year * season * region). Seasonal contrasts are averaged across regions, and regional contrasts are averaged across season. All contrasts hold year at the median (2008). Results are on the response scale.

Contrast	**Region**	**Odds Ratio**	***SE***	***df***	**z.ratio**	***p***
spring / summer	E	0.33	0.00	Inf	-102.96	<.0001*
spring / autumn	E	0.41	0.00	Inf	-83.03	<.0001*
spring / winter	E	1.12	0.01	Inf	8.80	<.0001*
summer / autumn	E	1.22	0.01	Inf	22.99	<.0001*
summer / winter	E	3.36	0.04	Inf	109.54	<.0001*
autumn / winter	E	2.76	0.03	Inf	90.26	<.0001*
spring / summer	NE	0.34	0.01	Inf	-65.09	<.0001*
spring / autumn	NE	0.38	0.01	Inf	-58.31	<.0001*
spring / winter	NE	1.05	0.02	Inf	2.43	<.0001*
summer / autumn	NE	1.11	0.01	Inf	7.88	<.0001*
summer / winter	NE	3.09	0.05	Inf	66.87	<.0001*
autumn / winter	NE	2.78	0.05	Inf	60.19	<.0001*
spring / summer	NEN	0.35	0.00	Inf	-232.91	<.0001*
spring / autumn	NEN	0.40	0.00	Inf	-202.18	<.0001*
spring / winter	NEN	1.17	0.01	Inf	27.66	<.0001*
summer / autumn	NEN	1.13	0.00	Inf	37.19	<.0001*
summer / winter	NEN	3.32	0.02	Inf	252.79	<.0001*
autumn / winter	NEN	2.93	0.01	Inf	223.66	<.0001*
spring / summer	NW	0.44	0.00	Inf	-143.00	<.0001*
spring / autumn	NW	0.49	0.00	Inf	-123.17	<.0001*
spring / winter	NW	1.08	0.01	Inf	10.47	<.0001*
summer / autumn	NW	1.11	0.00	Inf	23.68	<.0001*
summer / winter	NW	2.45	0.01	Inf	151.93	<.0001*
autumn / winter	NW	2.20	0.01	Inf	132.51	<.0001*
spring / summer	SE	0.33	0.00	Inf	-131.79	<.0001*
spring / autumn	SE	0.46	0.00	Inf	-89.94	<.0001*
spring / winter	SE	1.21	0.01	Inf	18.08	<.0001*
summer / autumn	SE	1.39	0.01	Inf	47.12	<.0001*
summer / winter	SE	3.67	0.03	Inf	145.22	<.0001*
autumn / winter	SE	2.64	0.02	Inf	105.46	<.0001*
spring / summer	W	0.46	0.00	Inf	-76.04	<.0001*
spring / autumn	W	0.50	0.01	Inf	-66.83	<.0001*
spring / winter	W	0.91	0.01	Inf	-7.93	<.0001*
summer / autumn	W	1.09	0.01	Inf	10.11	<.0001*
summer / winter	W	2.00	0.02	Inf	69.07	<.0001*
autumn / winter	W	1.83	0.02	Inf	59.64	<.0001*

Table F8. Results of pairwise contrasts of linear model assessing the difference in proportional change (%) of habitat suitability of Snapper (*Chrysophrys auratus*) at a resolution of 0.004° between hindcasted predictions (1998–2018) and forecasted predictions (2036–2065) across six regions of Tasmania and four seasons. Results are on the response scale.

Region	**Contrast**	**Estimate**	***SE***	***df***	***t* ratio**	***p***
North West	spring - summer	12.34	0.11	522230.00	112.85	<.0001*
	spring - autumn	49.64	0.11	522230.00	451.19	<.0001*
	spring - winter	-140.93	0.12	522230.00	-1215.21	<.0001*
	summer - autumn	37.31	0.10	522230.00	362.82	<.0001*
	summer - winter	-153.27	0.11	522230.00	-1404.01	<.0001*
	autumn - winter	-190.57	0.11	522230.00	-1734.40	<.0001*
North East North	spring - summer	38.61	0.09	522230.00	452.46	<.0001*
	spring - autumn	72.30	0.09	522230.00	839.65	<.0001*
	spring - winter	-189.68	0.09	522230.00	-2118.71	<.0001*
	summer - autumn	33.68	0.08	522230.00	411.19	<.0001*
	summer - winter	-228.30	0.09	522230.00	-2669.87	<.0001*
	autumn - winter	-261.98	0.09	522230.00	-3036.74	<.0001*
North East	spring - summer	26.51	0.40	522230.00	67.08	<.0001*
	spring - autumn	35.08	0.40	522230.00	88.32	<.0001*
	spring - winter	-69.44	0.41	522230.00	-170.30	<.0001*
	summer - autumn	8.57	0.38	522230.00	22.30	<.0001*
	summer - winter	-95.95	0.40	522230.00	-242.76	<.0001*
	autumn - winter	-104.52	0.40	522230.00	-263.14	<.0001*
East	spring - summer	22.35	0.26	522230.00	86.25	<.0001*
	spring - autumn	53.61	0.26	522230.00	206.31	<.0001*
	spring - winter	-133.45	0.27	522230.00	-499.95	<.0001*
	summer - autumn	31.26	0.25	522230.00	123.55	<.0001*
	summer - winter	-155.80	0.26	522230.00	-598.55	<.0001*
	autumn - winter	-187.07	0.26	522230.00	-716.62	<.0001*
South East	spring - summer	20.62	0.20	522230.00	103.95	<.0001*
	spring - autumn	57.06	0.20	522230.00	286.14	<.0001*
	spring - winter	-130.00	0.20	522230.00	-643.71	<.0001*
	summer - autumn	36.44	0.20	522230.00	185.46	<.0001*
	summer - winter	-150.62	0.20	522230.00	-756.57	<.0001*
	autumn - winter	-187.06	0.20	522230.00	-934.60	<.0001*
West	spring - summer	18.18	0.22	522230.00	81.02	<.0001*
	spring - autumn	49.22	0.22	522230.00	219.11	<.0001*
	spring - winter	-96.03	0.23	522230.00	-420.52	<.0001*
	summer - autumn	31.05	0.22	522230.00	141.14	<.0001*
	summer - winter	-114.21	0.22	522230.00	-510.39	<.0001*
	autumn - winter	-145.26	0.22	522230.00	-648.25	<.0001*

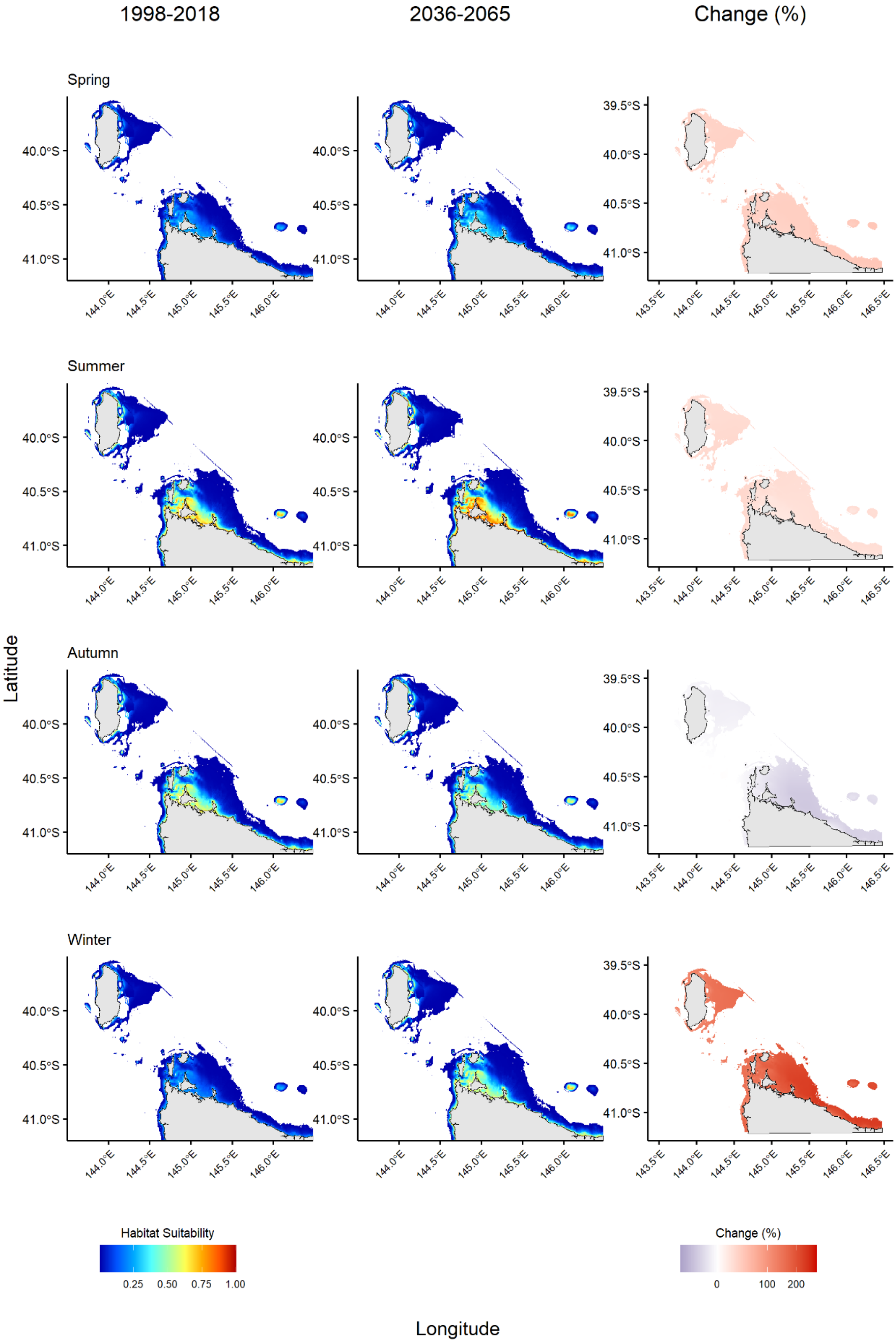


Figure F4. Seasonally aggregated spatial predictions of Snapper (*Chrysophrys auratus*) oceanographic habitat suitability for the North-West Coast of Tasmania, and the percent change in suitability between 20-year averaged hindcasted (1998–2018) and forecasted (2036–2065) predictions.

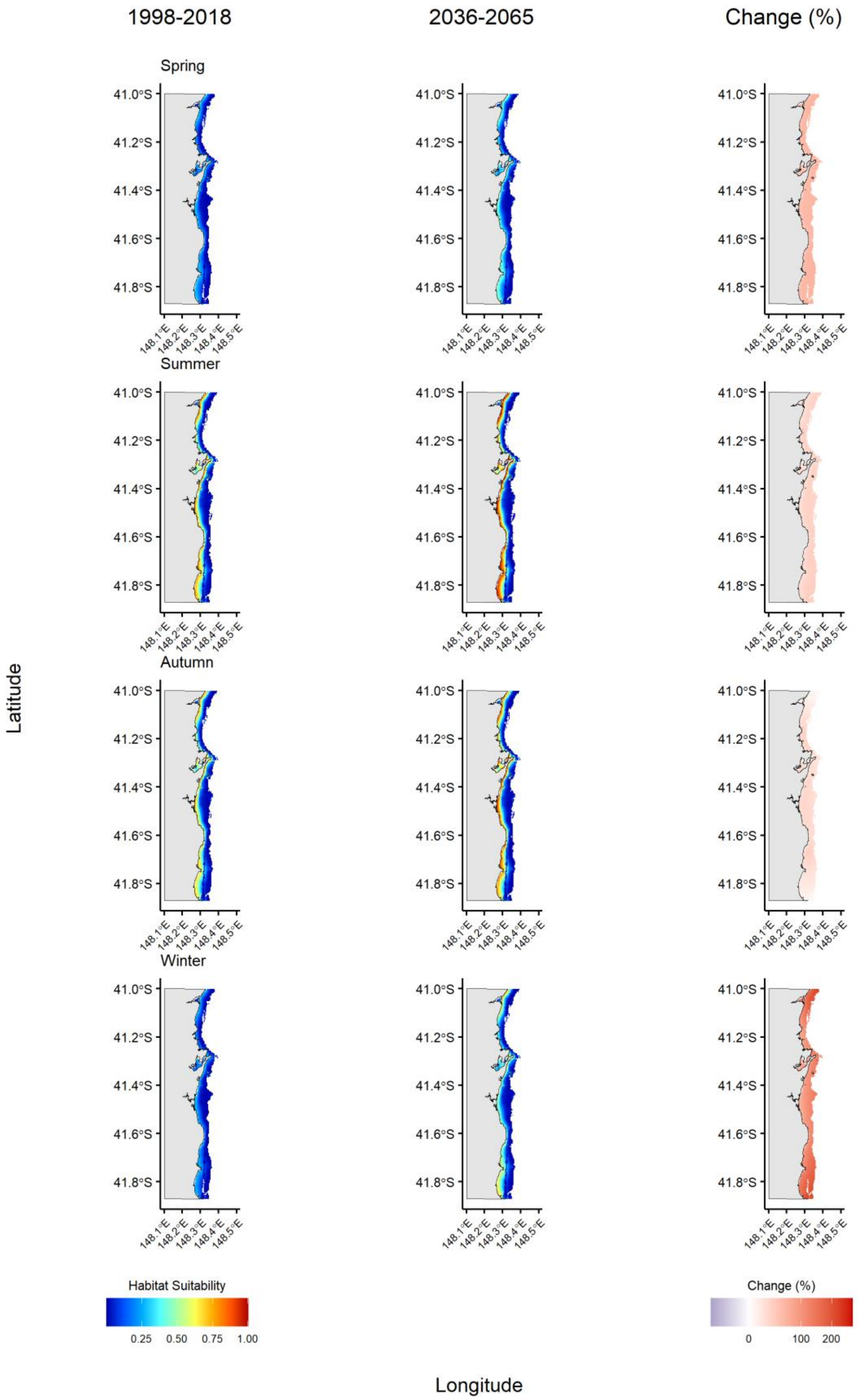


Figure F5. Seasonally aggregated spatial predictions of Snapper (*Chrysophrys auratus*) oceanographic habitat suitability for the North East (NE) Coast of Tasmania, and the percent change in suitability between 20-year averaged hindcasted (1998–2018) and forecasted (2036–2065) predictions.

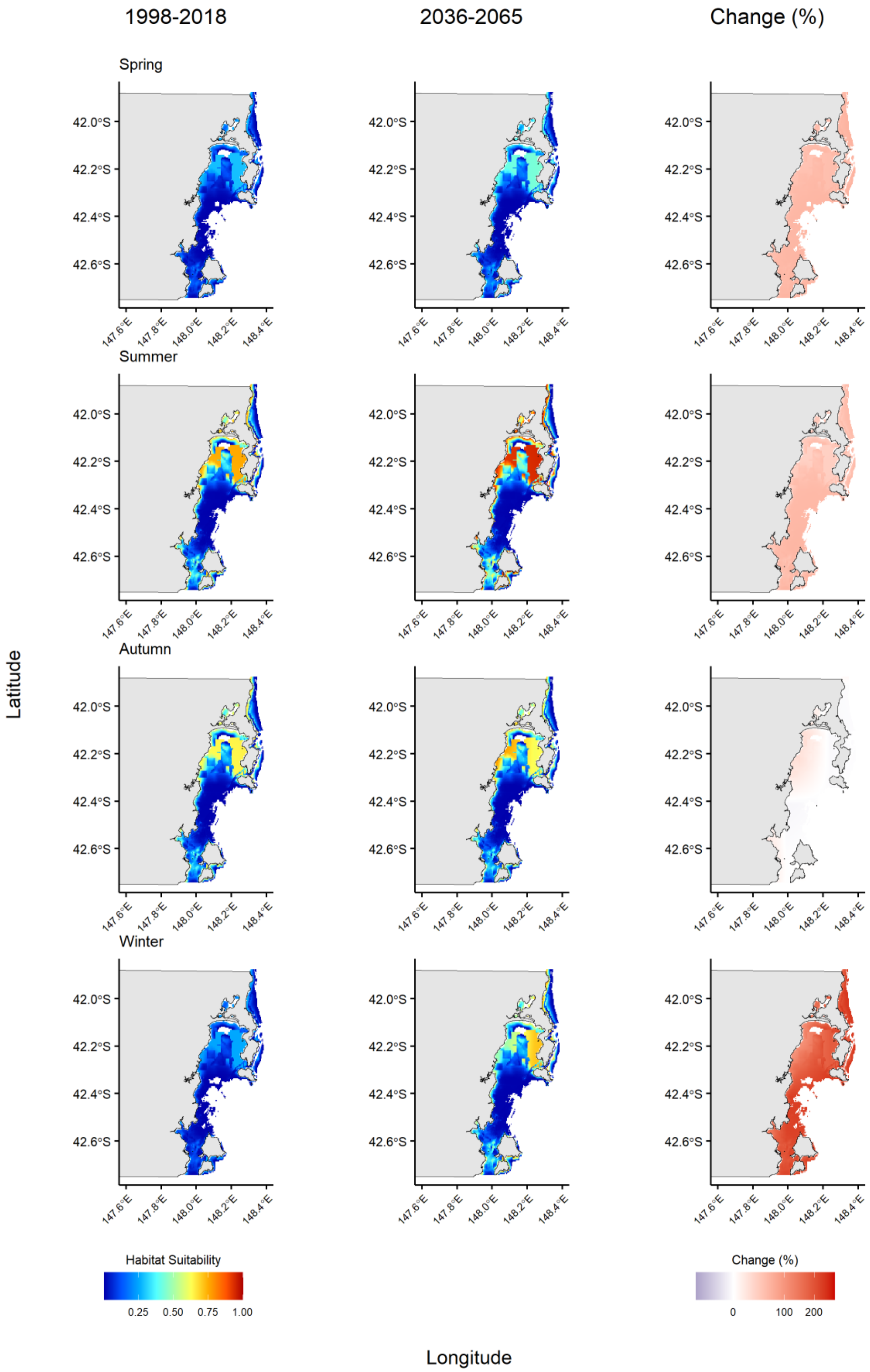


Figure F6. Seasonally aggregated spatial predictions of Snapper (*Chrysophrys auratus*) oceanographic habitat suitability for the East E) Coast of Tasmania, and the percent change in suitability between 20-year averaged hindcasted (1998–2018) and forecasted (2036–2065) predictions.

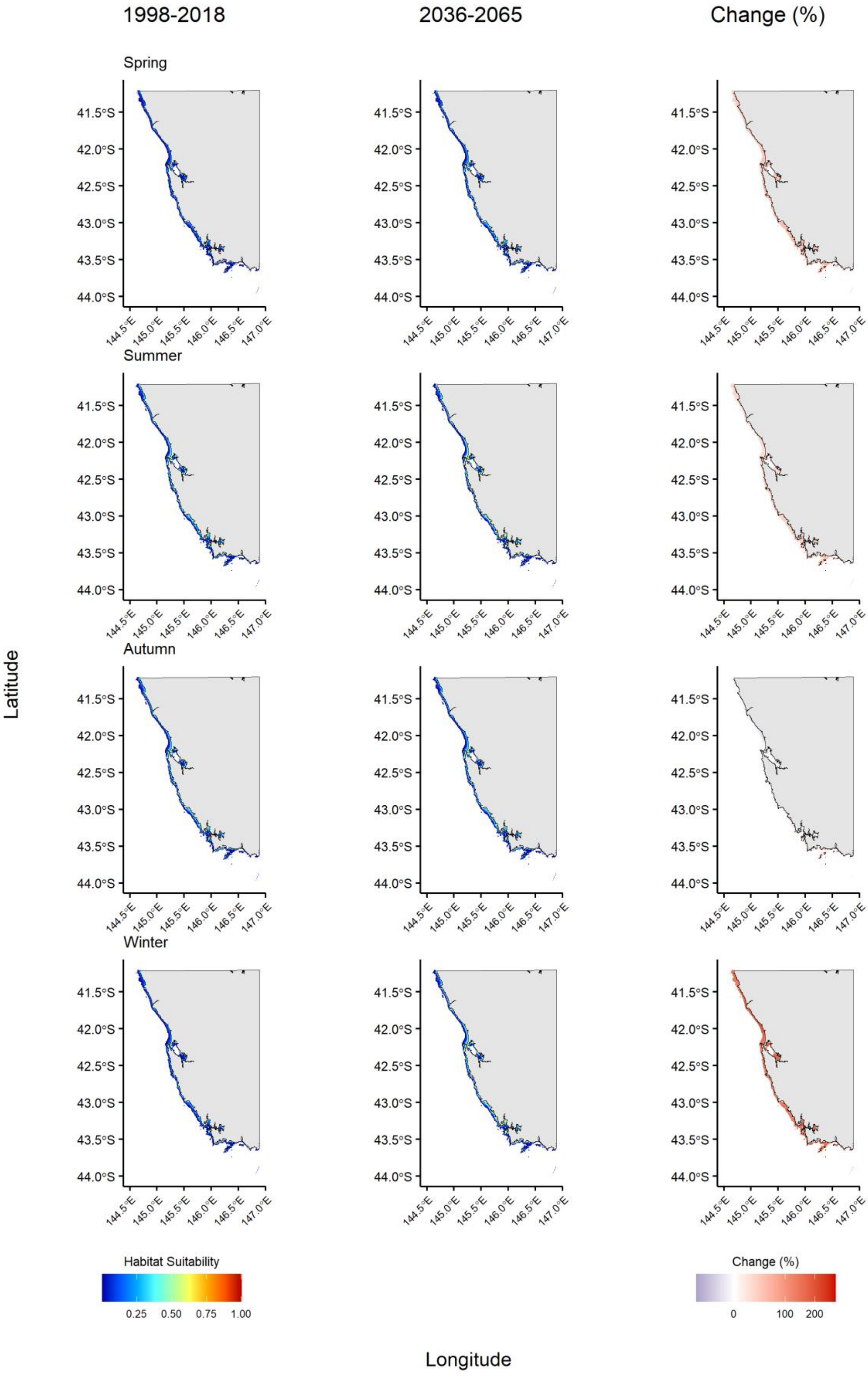


Figure F7. Seasonally aggregated spatial predictions of Snapper (*Chrysophrys auratus*) oceanographic habitat suitability for the West (W) Coast of Tasmania, and the percent change in suitability between 20-year averaged hindcasted (1998–2018) and forecasted (2036–2065) predictions.

SDM Results: King George Whiting

Table F9. Model summary for generalised linear model assessing the effect of year, season and region to the habitat suitability of King George Whiting (*Sillaginodes punctatus*) between 1998–2018. (suitability ~ year + season * region). Results are on the link scale

Factor	**Estimate**	***SE***	**z value**	***p***
Intercept (Autumn)	-1.138	0.8695942	-1.309	0.1905
Year	0.00079	0.0004330	1.831	0.0671
Spring	-0.3489	0.0153059	-22.793	< .001*
Summer	0.02870	0.0153339	1.872	0.0612
Winter	-0.3905	0.0153267	-25.481	< .001*
NE	- 1.9250153	0.0257404	-74.786	< .001*
NEN	- 2.2284660	0.0157956	-141.082	< .001*
NW	- 2.1089470	0.0145609	-144.836	< .001*
SE	-1.6163787	0.0145758	-110.895	< .001*
W	-2.2731912	0.0278550	-81.608	< .001*
Spring * NE	-0.0247605	0.0393956	-0.629	0.5297
Summer * NE	-0.0093142	0.0362598	-0.257	0.7973
Winter * NE	-0.0012834	0.0395751	-0.032	0.9741
Spring * NEN	0.1246477	0.0236129	5.279	< .001*
Summer * NEN	-0.0223527	0.0222897	-1.003	0.3159
Winter * NEN	0.1596334	0.0238120	6.704	< .001*
Spring * NW	-0.0311761	0.0216154	-1.442	0.1492
Summer * NW	0.0094786	0.0205028	0.462	0.6439
Winter * NW	-0.0468925	0.0218194	-2.149	0.0316*
Spring * SE	0.1464610	0.0211303	6.931	< .001*
Summer * SE	0.0524071	0.0204601	2.561	0.0104*
Winter * SE	0.1439810	0.0212772	6.767	< .001*
Spring * W	0.0401679	0.0426308	0.942	0.3461
Summer * W	0.0004479	0.0391764	0.011	0.9909
Winter * W	0.1293489	0.0420471	3.076	< .001*

Table F10. Results of pairwise contrasts assessing the differences in habitat suitability of King George Whiting (*Sillaginodes punctatus*) across season and region, while holding year at the median for seasonally aggregated yearly predictions of habitat suitability between 1998–2018. Results are on the response scale.

Region	**Contrast**	**Odds Ratio**	***SE***	***z* ratio**	***p***
NEN	autumn / spring	1.251	0.0225	12.471	< .001*
	autumn / summer	0.994	0.01607	-0.393	0.9795
	autumn / winter	1.260	0.02296	12.671	< .001*
	spring / summer	0.794	0.01424	-12858	< .001*
	spring / winter	1.007	0.01993	0.337	0.9868
	summer / winter	1.268	0.02304	13.053	< .001*
NE	autumn / spring	1.453	0.05275	10.293	< .001*
	autumn / summer	0.981	0.03223	-0.590	0.9351
	autumn / winter	1.480	0.05399	10.739	< .001*
	spring / summer	0.675	0.02441	-10.868	< .001*
	spring / winter	1.018	0.04022	0.461	0.9676
	summer / winter	1.509	0.05484	11.312	< .001*
E	autumn / spring	1.417	0.02170	22.793	< .001*
	autumn / summer	0.972	0.01490	-1.872	0.2401
	autumn / winter	1.478	0.02265	25.481	< 0.001*
	spring / summer	0.686	0.01048	-24.693	< 0.001*
	spring / winter	1.043	0.01593	2.726	0.0325
	summer / winter	1.521	0.02329	27.381	< .001*
SE	autumn / spring	1.224	0.01784	13.895	< .001*
	autumn / summer	0.922	0.01249	-5.988	< .001*
	autumn / winter	1.280	0.01889	16.706	< .001*
	spring / summer	0.753	0.01082	-19.736	< .001*
	spring / winter	1.045	0.01622	2.845	0.0230*
	summer / winter	1.388	0.02020	22.506	< .001*
W	autumn / spring	1.362	0.054	7.759	< .001*
	autumn / summer	0.0971	0.03502	-0.809	0.8504
	autumn / winter	1.298	0.05084	6.671	< .001*
	spring / summer	0.713	0.02823	-8.536	< .001*
	spring / winter	0.954	0.04046	-1.120	0.6772
	summer / winter	1.337	0.5206	7.456	< .001*
NW	autumn / spring	1.362	0.05418	7.759	< .001*
	autumn / summer	0.971	0.03502	-0.809	0.8504
	autumn / winter	1.298	0.05084	6.671	< .001*
	spring / summer	0.713	0.02823	-8.536	< .001*
	spring / winter	0.954	0.04046	-1.120	0.6772
	summer / winter	1.337	0.05206	7.456	< .001*

Table F11. Results of pairwise contrasts of linear model assessing the difference in proportional change (%) of habitat suitability for King George Whiting (*Sillaginodes punctatus*) at a resolution of 416 m between hindcasted predictions (1998–2018) and forecasted predictions (2036–2065) across six regions of Tasmania and four seasons. Results are on the response scale.

Region	**Contrast**	**Estimate**	***SE***	***df***	***t* ratio**	***p***
NW	spring - summer	20.15	0.48	45702	41.91	< .0001*
	spring - autumn	34.30	0.48	45702	70.92	< .0001*
	spring - winter	-88.28	0.48	45702	-177.56	< .0001*
	summer - autumn	14.16	0.48	45702	30.21	< .0001*
	summer - winter	-108.43	0.48	45702	-224.70	< .0001*
	autumn - winter	-122.59	0.49	45702	-252.53	< .0001*
NEN	spring - summer	44.90	0.56	45702	79.64	< .0001*
	spring - autumn	45.42	0.56	45702	80.30	< .0001*
	spring - winter	-93.43	0.60	45702	-156.34	< .0001*
	summer - autumn	0.52	0.54	45702	0.97	< .0001*
	summer - winter	-138.33	0.57	45702	-242.91	< .0001*
	autumn - winter	-138.84	0.59	45702	-243.03	< .0001*
NE	spring - summer	39.08	1.25	45702	31.27	< .0001*
	spring - autumn	33.98	1.25	45702	27.11	< .0001*
	spring - winter	-40.86	1.28	45702	-31.80	< .0001*
	summer - autumn	-5.11	1.22	45702	-4.19	< .0001*
	summer - winter	-79.94	1.25	45702	-63.95	< .0001*
	autumn - winter	-74.83	1.25	45702	-59.70	< .0001*
E	spring - summer	17.95	0.70	45702	25.53	< .0001*
	spring - autumn	18.778	0.71	45702	26.58	< .0001*
	spring - winter	-34.24	0.71	45702	-48.00	< .0001*
	summer - autumn	0.82	0.70	45702	1.18	< .0001*
	summer - winter	-52.19	0.70	45702	-74.08	< .0001*
	autumn - winter	-53.01	0.71	45702	-74.90	< .0001*
SE	spring - summer	20.94	0.56	45702	37.46	< .0001*
	spring - autumn	32.07	0.56	45702	56.98	< .0001*
	spring - winter	-67.09	0.58	45702	-115.56	< .0001*
	summer - autumn	11.13	0.54	45702	20.43	< .0001*
	summer - winter	-88.02	0.56	45702	-156.36	< .0001*
	autumn - winter	-99.15	0.56	45702	-174.96	< .0001*
W	spring - summer	11.51	1.21	45702	9.43	< .0001*
	spring - autumn	30.30	1.22	45702	24.75	< .0001*
	spring - winter	-59.97	1.25	45702	-48.02	< .0001*
	summer - autumn	18.79	1.18	45702	15.81	< .0001*
	summer - winter	-71.48	1.21	45702	-58.88	< .0001*
	autumn - winter	-30.27	1.21	45702	-74.10	< .0001*

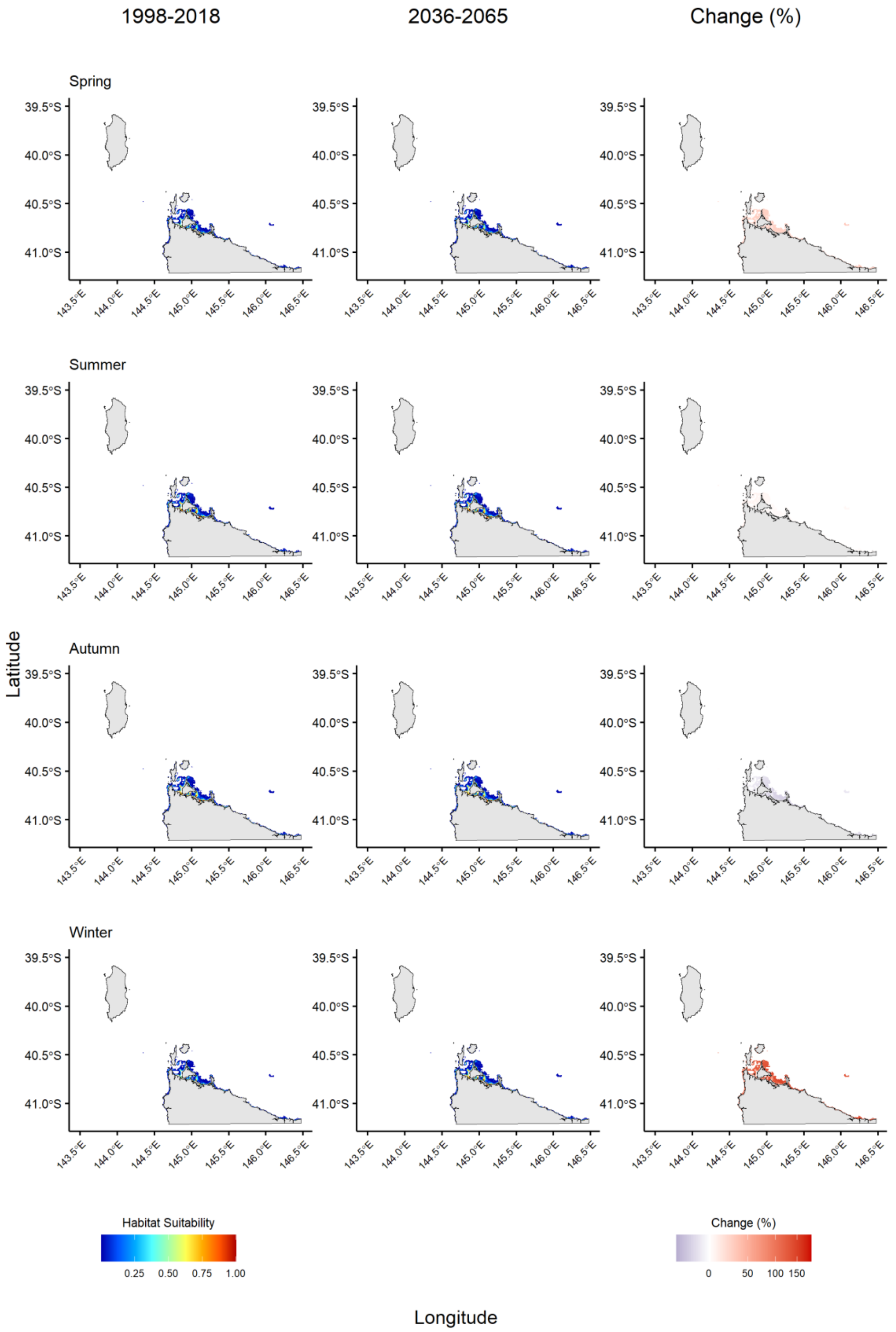


Figure F8. Spatial predictions of habitat suitability of King George Whiting (*Sillaginodes punctatus*) between 20-year averaged environmental data for hindcast (1998–2018) and forecast (2036–2065) periods in the NW region, and the proportional change (%) at a resolution of 0.004°.

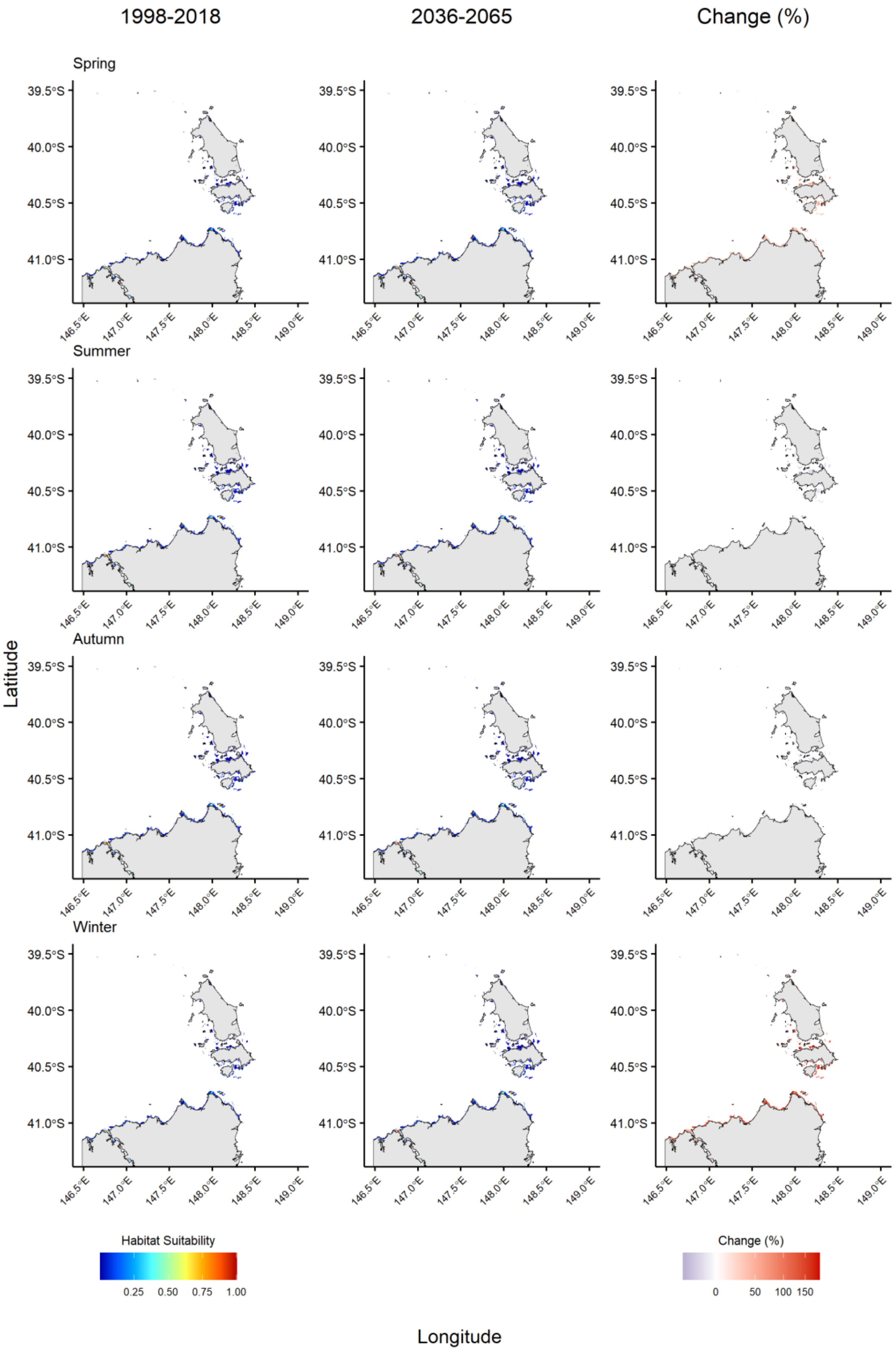


Figure F9. Spatial predictions of habitat suitability of King George Whiting (*Sillaginodes punctatus*) between 20-year averaged environmental data for hindcast (1998–2018) and forecast (2036–2065) periods in the NEN region, and the proportional change (%) at a resolution of 0.004°.

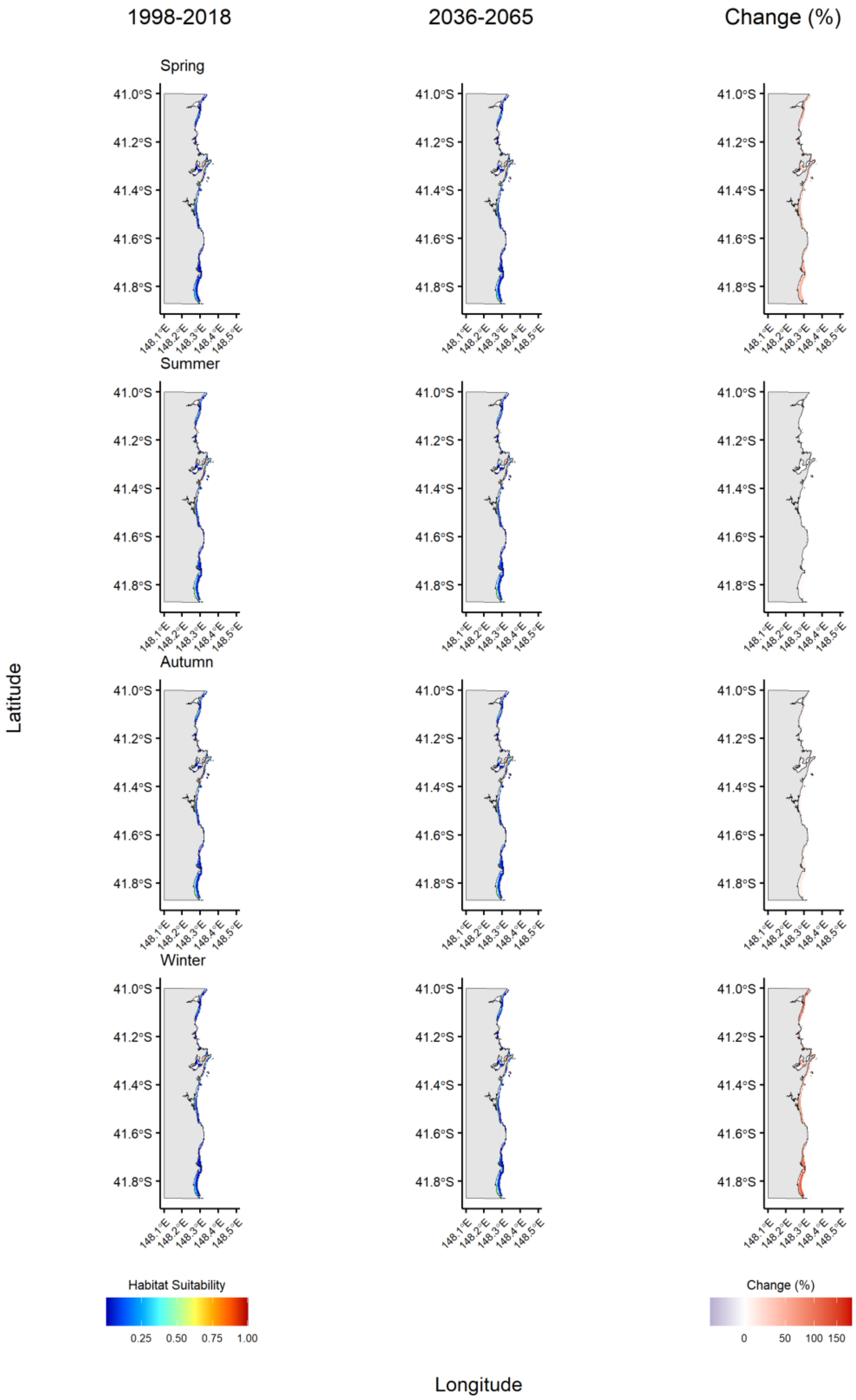


Figure F10. Spatial predictions of habitat suitability of King George Whiting (*Sillaginodes punctatus*) between 20-year averaged environmental data for hindcast (1998–2018) and forecast (2036–2065) periods in the NE region, and the proportional change (%) at a resolution of 0.004°.

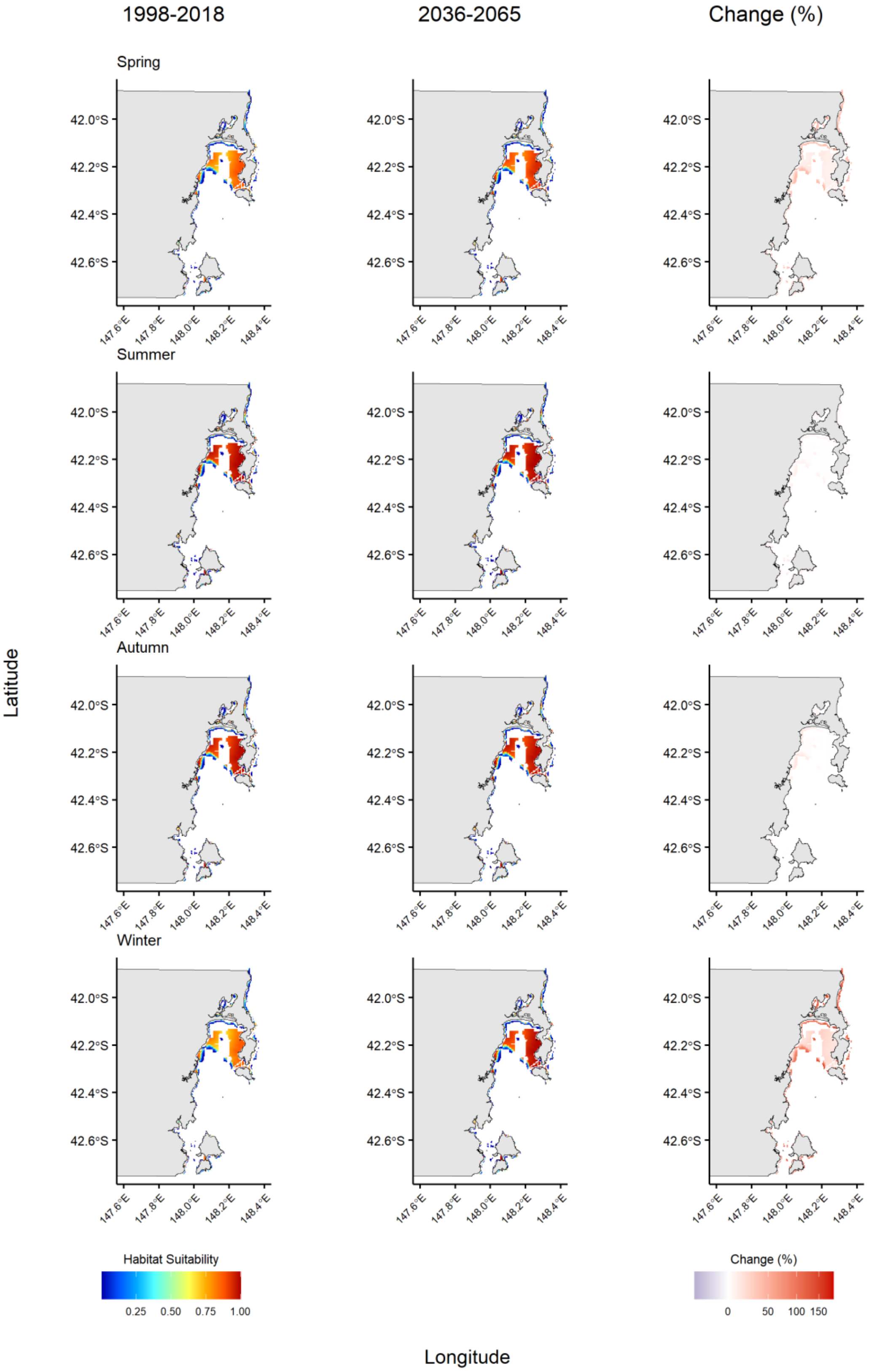


Figure F11. Spatial predictions of habitat suitability of King George Whiting (*Sillaginodes punctatus*) between 20-year averaged environmental data for hindcast (1998–2018) and forecast (2036–2065) periods in the East region, and the proportional change (%) at a resolution of 0.004°.

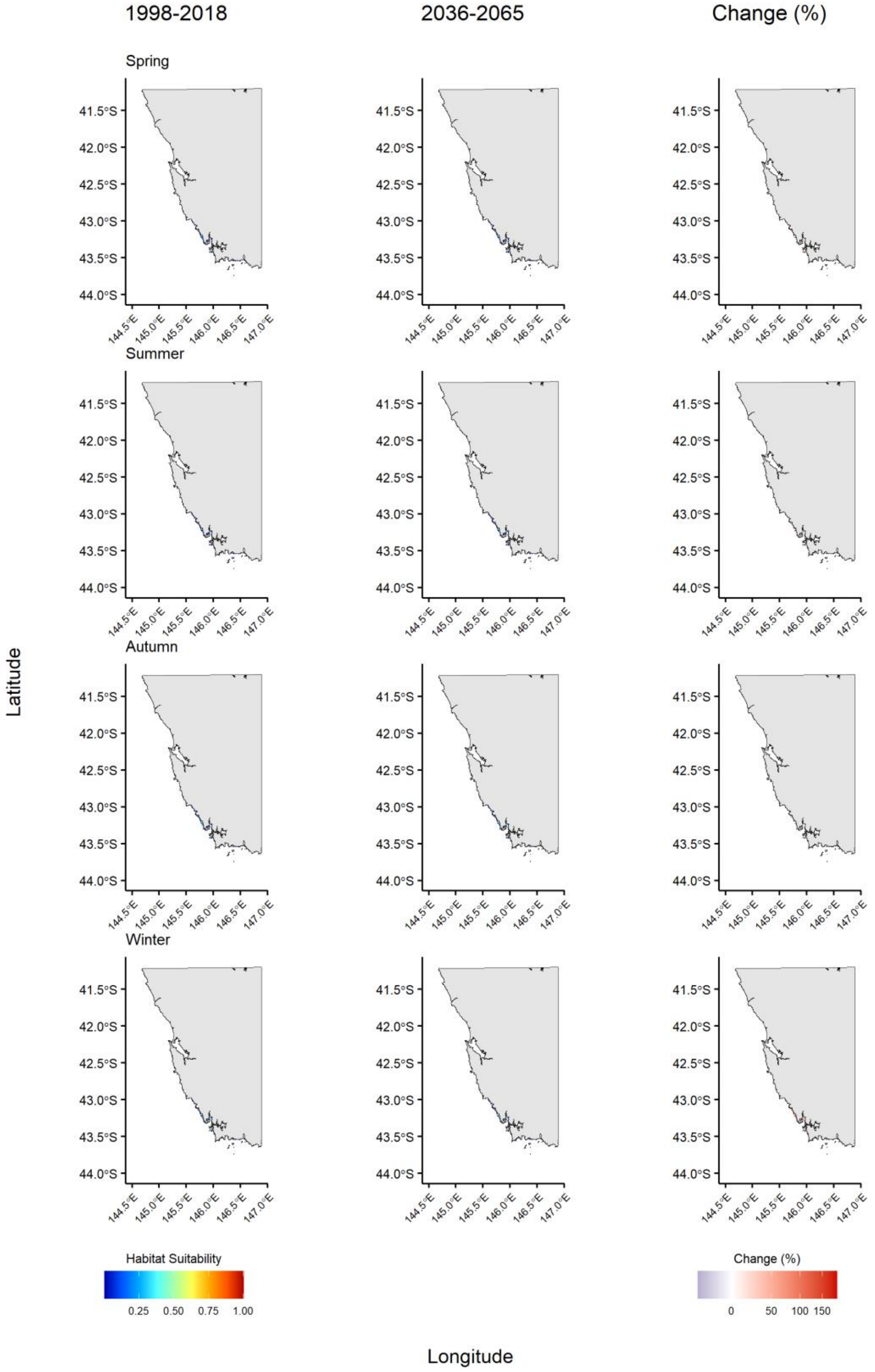


Figure F12. Spatial predictions of habitat suitability of King George Whiting (*Sillaginodes punctatus*) between 20-year averaged environmental data for hindcast (1998–2018) and forecast (2036-2065) periods in the West region, and the proportional change (%) at a resolution of 0.004°.

Appendix G: Biological Comparison Tables

Table G1. Parameters for the Bertalanffy growth curve: $\boldsymbol{L_t = L_\infty\left[1 - e^{-k(t-t_0)}\right]}$, sizes (Fork Length (FL: mm), rates of mortality (Z = total, M = natural, F = fishing) age and length at maturity, maximum length (L_{MAX}) and age (A_{MAX}) for yellowtail kingfish (*Seriola lalandi*) from Australian and New Zealand populations. Where: $\boldsymbol{L_t}$ = length at age t, $\boldsymbol{L_\infty}$ = asymptotic length, $\boldsymbol{k}$ = growth coefficient (the rate at which the increase in length decreases) and $\boldsymbol{t_0}$ = theoretical age at zero length.

Region	L_∞	k	t_0	Z	M	F	**Spawning season**	**Smallest Mature (FL mm)**	**Age at smallest mature**	**50% Maturity (FL mm)**	**Age at 50% maturity**	**100% Maturity (FL mm)**	L_{MAX} **(FL mm)**	A_{MAX}
New South Wales														
(Gillanders et al. 1999b)	125.2	0.189	-0.7				Nov–Feb	M: 360 F: 698	M: < 1 F: 3	M: 471 F: 834	M: < 1 F: 4–5			9
(Gillanders et al. 1999a)									M: 4 F: 4		M: 5 F: 7			
(Stewart et al. 2001)				0.43–0.79	0.12	0.31–0.67				800–1250	5-10		1400	21
(Stewart & Hughes 2008)				0.43	0.2	0.23		F:700 (~800 TL) M: (~550TL)						
(Stewart et al. 2004)	184	0.054	-4.4	0.43–0.79	0.12	0.31–1.38								21
(Hughes & Stewart 2020)				0.47 (0.33–0.60)	0.2–0.3	0.17–0.27								
(Steffe et al. 1996)													1200	
Victoria														
(Green et al. 2020)	135.84	0.149	-1.23							F: 843 (952 TL) M: 693 (784 TL)			1300 (TL)	

Region	L_∞	***k***	t_0	***Z***	***M***	***F***	**Spawning season**	**Smallest Mature (FL mm)**	**Age at smallest mature**	**50% Maturity (FL mm)**	**Age at 50% maturity**	**100% Maturity (FL mm)**	L_{MAX} **(FL mm)**	A_{MAX}
New Zealand														
(McGregor 1995)										580–670		700		
(McKenzie et al. 2014)	M: 131.370 F: 149.95 B: 140.58	M: 0.105 F: 0.086 B: 0.096	M: -1.314 F: -1.434 B: -1.339	0.29	0.19					M: 828.9 F: 968.6		M: 8+ F: 10+	1470	23
(Poortenaar et al. 2001)							Oct–Jan	M: 750 F: 755	M: 4 F: 4	M: 812 F: 944	M: 5 F: 7–8	M: 925 F: 1275		
(Moran et al. 2007)							Austral spring–summer							
(Holdsworth et al. 2016)													1340	29

Table G2. Parameters for the Bertalanffy growth curve: $L_t = L_\infty\left[1 - e^{-k(t-t_0)}\right]$, sizes (Fork Length (FL: mm), rates of mortality (Z = total, M = natural, F = fishing) age and length at maturity, maximum length (L_{MAX}) and age (A_{MAX}) for snapper (*Chrysophrys auratus*) from Australian and New Zealand populations. Where: L_t = length at age t, L_∞ = asymptotic length, k = growth coefficient (the rate at which the increase in length decreases) and t_0 = theoretical age at zero length.

Region	L_∞	k	t_0 **(yr)**	Z	M	F	**Spawning season**	**Smallest Mature (FL mm)**	**Age at smallest mature**	**50% Maturity (FL mm)**	**Age at 50% maturity**	**100% Maturity (FL mm)**	**Age at 100% maturity**	L_{MAX} **(FL mm)**	A_{MAX}
East Coast															
NSW/QLD (Ferrell & Sumpton 1997)				0.28–0.80			Jun–Sep		1.8	220	2+	F: 330	5+		
QLD (Sumpton 2001)	792	0.082	-2.45				Jun–Oct								
NSW/QLD (Wortmann et al. 2018)*	879.9	0.078	-2.548		0.163										41
NSW (Stewart et al. 2010)										248	2.5	350			
North NSW (30.3°S) (Stewart et al. 2010)										218	1.7				
Mid. NSW (31.5-35°S) (Stewart et al. 2010)										273	3.0				
South Australia															
(Stewart et al. 2010)*										280					
(McGlennon et al. 2000)	930.2	0.144	0.828												
(McGlennon 2003)	900.0	0.197	-2.742				Nov–Feb								

Region	L_∞	k	t_0 (yr)	Z	M	F	Spawning season	Smallest Mature (FL mm)	Age at smallest mature	50% Maturity (FL mm)	Age at 50% maturity	100% Maturity (FL mm)	Age at 100% maturity	L_{MAX} (FL mm)	A_{MAX}
(Saunders et al. 2012)							Nov–Jan								
N. Spencer Gulf	975	0.11	-0.699		0.05										36
S. Spencer Gulf	673	0.17	-0.435												30
N. Gulf St. Vincent	926	0.14	-0.075												35
S. Gulf St. Vincent	1045[†]	0.08	-0.996												32
South east (S of 36°S)	880	0.065	-2.735												29
(Fowler et al. 2013)															
Western Australia															
(Smallwood et al. 2013)				0.72–0.76											
(Wise et al. 2007)					0.12	0.37–0.80									
Upper west[§] (23.5–26.5°S)							May–Sep	(257 TL)	3.1	(353 TL)	3.9	(484 TL)[‡]	6.2[‡]		
(Wakefield et al. 2015)								(270 TL)	3.2	(378 TL)	4.0	(482 TL)[‡]	6.0[‡]		

Region	L_∞	k	t_0 (yr)	Z	M	F	Spawning season	Smallest Mature (FL mm)	Age at smallest mature	50% Maturity (FL mm)	Age at 50% maturity	100% Maturity (FL mm)	Age at 100% maturity	L_{MAX} (FL mm)	A_{MAX}
Upper west[§] (23.5–26.5°S) (Wakefield et al. 2016)	589 M: (648) F: (691) (681)	0.22 (0.25) (0.21) (0.22)	-0.23 (-0.25) (-0.33) (-0.42)											(840 TL) (864 TL)	22.1 30.0
Freycinet Estuary 26.5°S (Jackson et al. 2010)	770 M: 766 F: 773	0.17 0.17 0.17	0.1 0.08 0.13				Jun–Oct			M: 330 F: 420	M: 2.7 F: 4.5	M: 465[‡] F: 566[‡]	M: 5.9[‡] F: 8.4[‡]	M: 790 F: 785	M:31 F: 29
Denham Sound 26°S (Jackson et al. 2010)	728 M: 660 F: 762	0.15 0.18 0.14	-0.08 -0.06 -0.03				May–Sep			M: 276 F:401	M: 2.7 F: 5.5	M:730[‡] F:750[‡]	M:7.1[‡] F:9.9[‡]	M:730 F:750	M:17 F:19
Eastern Gulf 26°S (Jackson et al. 2010)	755 M: 751 F: 755	0.17 0.17 0.18	-0.02 -0.07 0.04				May–Sep			M: 243 F:348	M: 1.6 F: 3.2	M: 800[‡] F: 752[‡]	M: 4.2[‡] F: 5.6[‡]	M: 800 F: 752	M:15 F: 17
Midwest (26.5–31°S) (Lenanton et al. 2009)	(786) M: (803) F: (746)	(0.15) (0.15) (0.17)	(-1.37) (-1.30) (-0.93)					M: (319) F: (405)	M: 2.2 F: 3.7	M: (426) F: (487)	M: 3.8 F: 5.5	M: (629)[‡] F: (682)[‡]	M: 9.3[‡] F: 11.1[‡]	M: (947) F: (832)	M: 18.4 F: 26.2

Region	L_∞	k	t_0 (yr)	Z	M	F	Spawning season	Smallest Mature (FL mm)	Age at smallest mature	50% Maturity (FL mm)	Age at 50% maturity	100% Maturity (FL mm)	Age at 100% maturity	L_{MAX} (FL mm)	A_{MAX}
Midwest (26.5–31°S) (Wise et al. 2007)	681.9	0.149	-1.256												
Lower west (31–33°S) (Wakefield et al. 2015)							Oct–Mar	M: (407) F: (375)	M: 2.9 F: 2.8	M: (566) F: (585)	M: 5.6 F: 5.7	M: (730)‡ F: (752)‡	M: 8.5‡ F: 8.0‡		
Lower west (31–33°S) (Wakefield et al. 2016)	1004 (1136) M: (1127) F: (1150)	0.12 (0.12) (0.12) (0.12)	(-0.23) (-0.42) (-0.46) (-0.41)											(1056) (1051)	28.8 24.8
South west (33°S-115.5°E) (Lenanton et al. 2009)	(925) M: (902) F: (918)	(0.15) M: (0.14) F: (0.14)	(-1.16) M: (-0.98) F: (-0.84)					M: (429) F: (457)	M: 3.2 F: 4.1					M: (993) F: (879)	M: 32.6 F: 13.3
South west (33°S-115.5°E) (Wise et al. 2007)	804.9	0.129	-1.047												
South coast (115.5°E-125°E) (Wakefield et al. 2015)							Nov–Dec	M: 369 F: 466	M: 3.1 F: 3.8	M: (586) F: (600)	M: 6.5 F: 7.0	M: (838)‡ F: (899)‡	M: 11.5‡ F: 15.5‡		

Region	L_∞	k	t_0 (yr)	Z	M	F	Spawning season	Smallest Mature (FL mm)	Age at smallest mature	50% Maturity (FL mm)	Age at 50% maturity	100% Maturity (FL mm)	Age at 100% maturity	L_{MAX} (FL mm)	A_{MAX}
South coast (115.5°E-125°E) (Wakefield et al. 2016)	876 (986) M: (950) F: (1013)	0.11 (0.11) M: (0.13) F: (0.11)	-1.15 (-1.22) M: (-0.61) F: (-0.94)											(999) (1083)	37.8 29.4
Victoria															
Port Philip Bay/ Port Albert (Coutin et al. 2003)	856.5 M: 867.5 F: 915	0.1190 M: 99 F: 0.092	0 M: -0.76 F: -0.81				Nov–Dec	270		F: 363 F:(422)	F: 4.9	480			
(Hamer & Jenkins 2007)	836.1	0.100	-1.147											840	27
New Zealand															
West North Island	528	0.21	-0.28												
Hauraki Gulf	631	0.07	-1.94												
Bay of Plenty	565	0.10	-1.64												
East Northland	478	0.11	-2.12												
(Davies et al. 2003)															
Golden/Tasman Bays (Francis et al. 1992)	720	0.106	-0.75												

Region	L_∞	k	t_0 (yr)	Z	M	F	Spawning season	Smallest Mature (FL mm)	Age at smallest mature	50% Maturity (FL mm)	Age at 50% maturity	100% Maturity (FL mm)	Age at 100% maturity	L_{MAX} (FL mm)	A_{MAX}
Golden/Tasman Bays (Harley & Gilbert 2000)	696	0.122	-0.71		0.075										
Hauraki Gulf (Scott et al. 1993)							Oct–Feb/Mar								
Hauraki Gulf (Crossland 1977)							Oct–Jan	230		237		300			
Hauraki Gulf (Paul 1976)									3				5		
East Cape (Paul & Tarring 1980)				0.09											

***** References that report values from primary sources that are unavailable: Stewart et al. 2010 cites “Primary Industries and Resources SA” for the SA L_{50} figure, Wortmann et al. 2018 refers to the Campbell et al. 2009 stock assessment for the listed von Bertalanffy parameters, however the reference is to a summary of the stock assessment that contain the values.

§ Excluding Inner Shark Bay (e.g. Freycinet Estuary, Denham Sound, Eastern Gulf)

† source notes large fish were underrepresented, L_∞ not well estimated

‡ 95% Maturity not 100%

() denote values are TL not FL

Table G3. Parameters for the Bertalanffy growth curve: $\boldsymbol{L_t = L_\infty\left[1 - e^{-k(t-t_0)}\right]}$, sizes (Fork Length (FL: mm), rates of mortality (Z = total, M = natural, F = fishing) age and length at maturity, maximum length (L_{MAX}) and age (A_{MAX}) for King George Whiting (*Sillaginodes punctatus*) from Australian populations. Where: $\boldsymbol{L_t}$ = length at age t, $\boldsymbol{L_\infty}$ = asymptotic length, $\boldsymbol{k}$ = growth coefficient (the rate at which the increase in length decreases) and $\boldsymbol{t_0}$ = theoretical age at zero length.**NB: Gulf St. Vincent = GSV, Spencer Gulf = GS and the West Coast of South Australia = WC.

Region	L_∞	k	t_0	Z	M	F	**Spawning season**	**Smallest Mature (FL mm)**	**Age at smallest mature**	**50% Maturity (FL mm)**	**Age at 50% maturity**	**100% Maturity (FL mm)**	L_{MAX} **(FL mm)**	A_{MAX}
South Australia														
(Fowler et al. 2000b)										(300–350)	3–4 yrs		(590)	22
(Fowler et al. 1999)												3+ yrs		
(Cockrum & Jones 1992)										320				
(McGarvey & Fowler 2002)	GSV M: 467 F: 418 SG M: 416.1 F: 492.6 WC M: 387.0 F: 545.4	GSV M: 0.61 F: 0.48 SG M: 0.77 F: 0.49 WC M: 1.17 F: 0.70	0 (fixed)											
(McKay 1992)													(722)	
(Ham & Hutchinson 2003)							Feb–July (14–19 °C)							
Victoria														
(Hamer et al. 2004)	52.6 cm	0.297	-0.101										(600)	11
(Robertson 1977)							May–July							

Region	L_∞	k	t_0	Z	M	F	Spawning season	Smallest Mature (FL mm)	Age at smallest mature	50% Maturity (FL mm)	Age at 50% maturity	100% Maturity (FL mm)	L_{MAX} (FL mm)	A_{MAX}
Western Australia														
(Hyndes et al. 1998, Sulin 2012)	M: (500) F: (532)	M:0.53 F: 0.47	M: 0.16 F: 0.13				June–Sept			M: F: (413)	3–4 yrs	End of 4th year (majority)	M: (555) F: (596)	M: 13 F: 14
(Potter et al. 1996)	M: (501.6) F: (535.0)	M: 0.52 F: 0.46	M: 0.15 F: 0.13				June–Sept		3 yrs	(400)	4 yrs		(530)	13
(Gaughan et al. 2006)	South M: (414.89) F: (410.15) West M: (571.90) F: (562.60)	South M: 0.415 F: 0.425 West M: 0.345 F: 0.359								(518.20)				
(Fisher et al. 2014)	M: (526.5) F: (543.1)	M: 0.514 F: 0.446	M: 0.568 F: 0.568			Inshore: 0.55 Offshore: 0.06								
(Brown et al. 2013)	South M: (536) F: (565) West M: (536) F: (565)	South M: 0.39 F: 0.36 West M: 0.56 F: 0.53	South M: 0.12 F: 0.10 West M: 0.00 F: 0.01				July–Sept	South M: (273) F: (391) West M: (327) F: (401)	South M: F: West M: 4.3 F: 4.2	South M: (427)* F: (399)* West M: (437) F: (440)		West > (500)	(586)	F:13.9 M: 13.3
Tasmania														

Region	L_∞	*k*	t_0	*Z*	*M*	*F*	Spawning season	Smallest Mature (FL mm)	Age at smallest mature	50% Maturity (FL mm)	Age at 50% maturity	100% Maturity (FL mm)	L_{MAX} (FL mm)	A_{MAX}
(Nicholls 2018)	M: (51.9) cm F: (57) cm	M: 0.24 F: 0.2	M:-1.45 F: -1.58		0.16					400 (TL)				
(Jenkins et al. 2016)	519	0.347	-0.419										NW Tas: 585 Georges Bay: 445	

* estimated

( ) denotes values are TL not FL

Appendix H: Ancillary analyses for Snapper

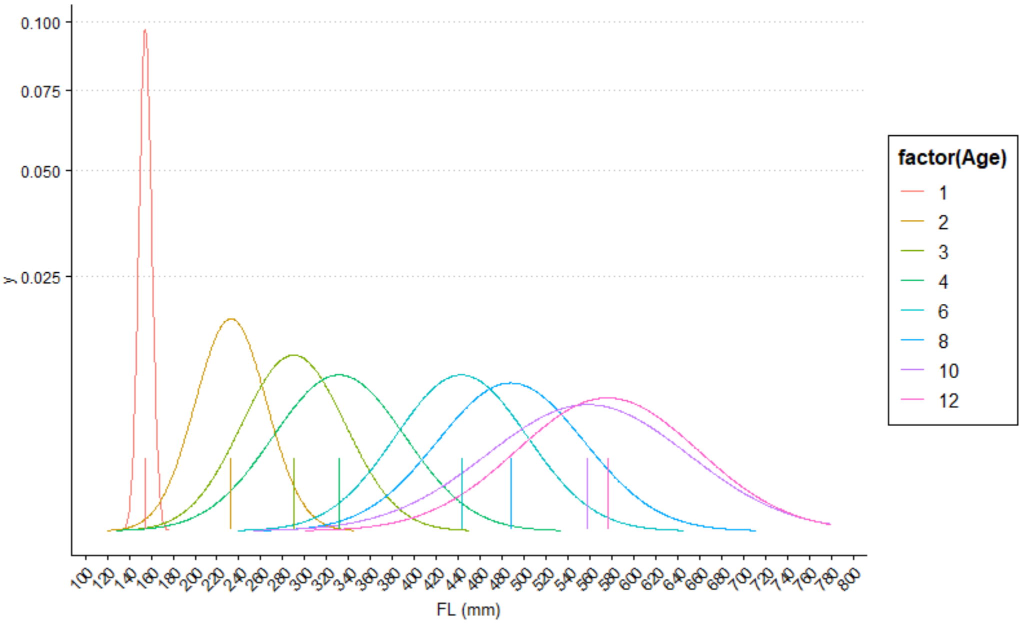


Figure H1. Length and age frequency distributions of Tasmanian Snapper (*Chrysophrys auratus*), based on the mean and standard deviation of each age class. *NB age 1 has an n of 3, which is driving the strong density response.

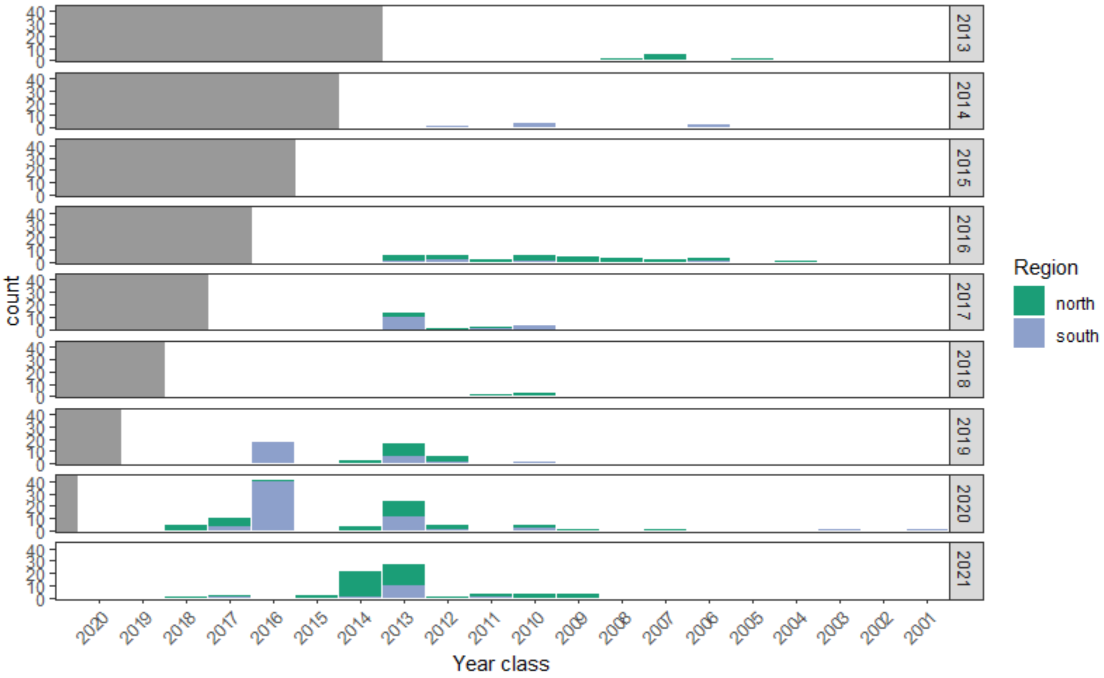


Figure H2. Interannual age frequency variations of Snapper (*Chrysophrys auratus*) from Tasmania between 2001 and 2020. Years in column bases are the back calculated year of birth with lines indicating the progression of that birth year. Note—regional sampling was not consistent across all years.