



FINAL REPORT

Ecological modelling of the impacts of water development in the Gulf of Carpentaria with particular reference to impacts on the Northern Prawn Fishery

Gulf of Carpentaria MICE (Models of Intermediate Complexity for Ecosystem assessments)

Éva Plagányi, Rob Kenyon, Laura Blamey, Michele Burford, Julie Robins, Annie Jarrett, Adrienne Laird, Justin Hughes, Shaun Kim, Trevor Hutton, Richard Pillans, Roy Deng, Toni Cannard, Emma Lawrence, Margaret Miller, Chris Moeseneder

March 2022

FRDC Project No **2018-079**

© 2022 Fisheries Research and Development Corporation.
All rights reserved.

ISBN: 978-1-925994-26-1

Ecological modelling of the impacts of water development in the Gulf of Carpentaria with particular reference to impacts on the NPF
[2018-079]

[2022]

Ownership of Intellectual property rights

Unless otherwise noted, copyright (and any other intellectual property rights, if any) in this publication is owned by the Fisheries Research and Development Corporation and The Commonwealth Scientific and Industrial Research Organisation (CSIRO)

This publication (and any information sourced from it) should be attributed to **Éva Plagányi, Rob Kenyon, Laura Blamey, Michele Burford, Julie Robins, Annie Jarrett, Adrienne Laird, Justin Hughes, Shaun Kim, Trevor Hutton, Richard Pillans, Roy Deng, Toni Cannard, Emma Lawrence, Margaret Miller, Chris Moeseneder, 2022. *Ecological modelling of the impacts of water development in the Gulf of Carpentaria with particular reference to impacts on the Northern Prawn Fishery, Final Report to FRDC, Canberra, March 2022, 611 pp, CC BY 3.0***

Creative Commons licence

All material in this publication is licensed under a Creative Commons Attribution 3.0 Australia Licence, save for content supplied by third parties, logos and the Commonwealth Coat of Arms.



Creative Commons Attribution 3.0 Australia Licence is a standard form licence agreement that allows you to copy, distribute, transmit and adapt this publication provided you attribute the work. A summary of the licence terms is available from creativecommons.org/licenses/by/3.0/au/deed.en. The full licence terms are available from creativecommons.org/licenses/by/3.0/au/legalcode.

Inquiries regarding the licence and any use of this document should be sent to: frdc@frdc.com.au

Disclaimer

The authors do not warrant that the information in this document is free from errors or omissions. The authors do not accept any form of liability, be it contractual, tortious, or otherwise, for the contents of this document or for any consequences arising from its use or any reliance placed upon it. The information, opinions and advice contained in this document may not relate, or be relevant, to a readers particular circumstances. Opinions expressed by the authors are the individual opinions expressed by those persons and are not necessarily those of the publisher, research provider or the FRDC.

The Fisheries Research and Development Corporation plans, invests in and manages fisheries research and development throughout Australia. It is a statutory authority within the portfolio of the federal Minister for Agriculture, Fisheries and Forestry, jointly funded by the Australian Government and the fishing industry.

Researcher Contact Details

Name: Dr Éva Plagányi
Address: QBP, 306 Carmody Road
St Lucia QLD 4067
Phone: 07 3833 5955
Fax:
Email: eva.plaganyi-lloyd@csiro.au

FRDC Contact Details

Address: 25 Geils Court
Deakin ACT 2600
Phone: 02 6285 0400
Fax: 02 6285 0499
Email: frdc@frdc.com.au
Web: www.frdc.com.au

In submitting this report, the researcher has agreed to FRDC publishing this material in its edited form.

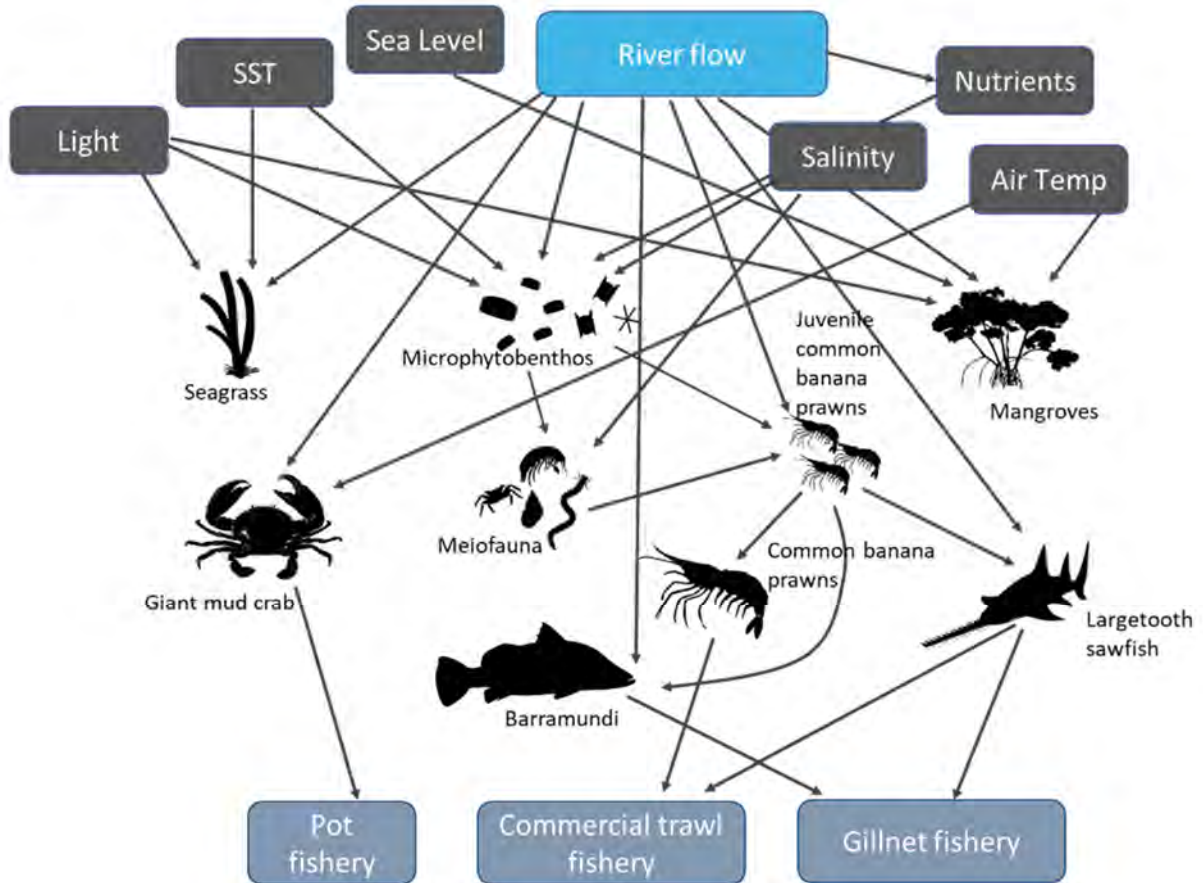
Foreword



Image: Queensland Department of Agriculture and Fisheries

River flow is crucial in the life cycle of common banana prawns that support the Northern Prawn Fishery (NPF), as well as iconic tropical species (e.g. Giant mud crab (*Scylla serrata*), barramundi (*Lates calcarifer*), grunter (*Pomadasys kaakan*), and king threadfin (*Polydactylus macrochir*)) of importance to commercial, recreational and Indigenous fisheries, and species with high conservation (e.g. largemouth sawfish (*Pristis pristis*) and cultural value. All rivers flowing into the Gulf of Carpentaria provide critical ecosystem services to riverine and estuarine reaches of the waterway. Several of these rivers have been flagged for the development of irrigated agriculture and other uses in their catchments and in the last decade extensive catchment-wide water and soil resource inventories have been undertaken. Water extraction from these rivers to support agriculture will modify natural flow regimes that currently support estuarine and coastal fisheries. The trade-offs associated with proposed water resource development have not previously been quantified using an integrated regional approach that considers the cumulative impacts on ecological assets, which are often linked across catchments. Building on previous work, this research aims to support decision making related to alternative strategies for managing water resources effectively for both agriculture (or mining) and marine production and biodiversity conservation. Quantifying these trade-offs entails evaluating how altered river flows might affect the downstream fisheries and ecological values. In remote northern Australia, most biological research to date has focussed on the basic biology of key species and their sustainable management as a fishery species. Likewise, hydrological research has monitoring catchment flows at key locations to gauge the water resource and develop a historical series of flow data. Only in the last decade have attempts been made to link river flow data with fishery and ecosystem productivity data to interpolate how the diversion of flows may impact GoC coastal productivity. However, water resource planning to date has failed to take a regional approach and consider the cumulative impacts on ecological assets that are often linked across catchments. There remains an ongoing need to strengthen the scientific basis that could be used to inform decisions about water resource development and underpin transparent consideration of trade-offs associated with different management options. Although previous and recent projects such as NAWRA have evaluated the qualitative impacts of changes in river flows on ecological assets, this study advances research through development of a spatial dynamic (model changes over time rather than a single snapshot) MICE (Models of Intermediate Complexity for Ecosystem assessments) to quantify impacts. The MICE uses validated and novel ways of linking river flow and other physical variables to key species, and then uses outputs of

sophisticated river models to evaluate rigorously what ecosystem changes could be expected in response to changes in flow regimes. The project therefore provides water resource managers with quantitative estimates such as the minimum water requirements to maintain ecosystem structure and functioning. Such analyses are also complicated by the fact that each catchment is different. Hence the MICE has sub-models tailored to be specific to each catchment area, plus they have been developed at the appropriate scale and incorporate key relationships in the ecosystem.



Key species – see text for details

Contents

Foreword	iii
Acknowledgments	xxxiv
Summary of findings & recommendations	1
Key findings.....	1
Considerations and recommendations	1
Executive Summary	3
Introduction	11
Objectives	15
Method	16
1. Project Plan and Scope	17
2. Gulf of Carpentaria MICE model overview	20
Key species.....	22
3. Water Resource Development overview and modelling of river flows in the Gulf of Carpentaria.....	25
4. Biological data collation: Qld and NT Fisheries	27
5. Physical data collation: Qld and NT.....	29
6. Stakeholder workshops.....	31
7. MICE sub-models.....	33
Overview of focus model species.....	33
Species modelled using a weekly time-step	34
Species modelled using a monthly time-step	39
Modelling of habitat and base productivity groups	43
Seagrass modelling.....	43
Mangrove Modelling.....	44
Estuarine microphytobenthos and meiofauna and habitat use by juvenile common banana prawns	45
Estimating catch as a function of river flow.....	46
Linking flow to biological components.....	47
8. Bio-economic modelling	50
9. MICE Modelling Process	53
10. Model Fitting	54
11. Ensemble of Alternative Models	55
12. Sensitivity Testing Trophic Links	58
13. Modelling Water Resource Development (WRD) scenarios.....	60
14. Biological and Fishery Risk	66
15. Economic Risk.....	68
Results	70
16. Modelling of river flows in Gulf of Carpentaria	70

17. Base Case MICE Results.....	74
Model ensemble: Prawns.....	74
River Portfolio Effect	94
Model ensemble: Barramundi	101
Model ensemble: Mud crab.....	110
Model ensemble: Sawfish	121
Model ensemble: Mangroves and Seagrass.....	128
18. Ecosystem overview	133
19. Water Resource Development (WRD) scenarios	139
20. Quantification of WRD influences on the aquatic ecosystem	143
Overview	143
Species comparisons and predicted impacts.....	144
Seagrass predicted responses under alternative WRDs.....	144
Mangroves predicted responses under alternative WRDs.....	146
Common banana prawns predicted responses under alternative WRDs	152
Barramundi predicted responses under alternative WRDs.....	161
Mud crabs predicted responses under alternative WRDs.....	168
Largetooth Sawfish predicted responses under alternative WRDs	177
21. Sensitivity – adding additional trophic linkages.....	184
22. Potential economic impacts of changes in river flow and timing due to water development on the Northern Prawn Fishery	188
23. Caveats and Model Limitations	191
24. Cross-checking model predictions of WRD impacts.....	194
Comparisons with other studies estimating the impact of water development on the catch of a commercial marine fishery	203
25. Biological and Fishery Risk Results.....	206
Discussion	218
Conclusion	224
Summary of key findings and recommendations.....	226
Implications.....	230
Recommendations	231
Further development	232
Extension and Adoption.....	233
Project coverage.....	234
Glossary	235
Project materials developed.....	238
Appendices	239
Appendix 1. List of researchers and key collaborators	239
Appendix 2. First Stakeholder Workshop.....	240
Workshop Agenda.....	240
Workshop Summary	242
Water Resource Development overview	242

List of Workshop 1 participants	249
Appendix 3 – Workshop Teleconferences	251
Prawn Workshop	251
<i>Text Box 1</i> –Teleconference 1 participant list	252
Mud crab Workshop	256
<i>Text Box 2</i> – Teleconference 2 participant list	257
Barramundi Workshop	259
<i>Text Box 3</i> – Teleconference 3 participant list	260
Appendix 4 – Final Stakeholder Workshop	263
Monday 9 August: Overview of Water Resource Development Scenarios being tested; focus on Prawns	263
Agenda	263
Key Objectives	263
Model Region Map (for reference).....	264
Participant list – Monday workshop	264
Presentation: Water Resource Development scenarios underpinning workshop outcomes – Rob Kenyon.....	265
Presentation: GoC Ecological MICE model Overview with Prawn Focus – Éva Plagányi	266
Presentation: Potential economic impacts on the NPF of reduced banana prawn productivity – Trevor Hutton.....	270
Next steps – Éva Plagányi	272
Tuesday 10 August: mudcrabs; barramundi; other components;next steps and towards recommendations	275
Agenda	275
Key Objectives	275
Participant list – Tuesday workshop	276
Presentation: Overview & Recap of previous day’s workshop – Rob Kenyon.....	276
Presentation: Modelling mud crabs in the Gulf of Carpentaria – Laura Blamey	276
Presentation: Barramundi model – Éva Plagányi	280
Presentation: Sawfish model preliminary results – Éva Plagányi	283
Presentation: Modelling habitat & the bottom of the food web – Éva Plagányi	284
Final Q&A Session.....	288
Appendix 5 – Water Resource Development Scenarios	290
Appendix 6 – Physical variables	296
Appendix 7 – Common Banana Prawns background information.....	314
Life History of banana prawns	314
Appendix 8 - Research on the underpinnings of banana prawn production in estuaries	318
Appendix 9 – Non-Spatial Prawn Model sub-model	322
Preliminary parameter settings, estimation and model outputs for prawns.....	325
Appendix 10 – Prawn Spatial Model sub-model	334
Appendix 11 – Mud crab sub-model component.....	344
Background	344
Environmental Drivers	348
Methods	349
Mud crab sub-model.....	350
Mud crab Model Equations	352
Modelling river flow relationships	356
Mud crab sub-model ensemble.....	357
Results	359
Discussion around model development.....	373

References	374
Appendix 12 – Barramundi Model sub-model.....	376
MICE representation of barramundi.....	378
Appendix 13 – Sawfish overview	388
Sawfish Model Equations.....	392
Results when testing alternative post-release survival proportion of sawfish.....	397
Appendix 14 – Microphytobenthos and meiofauna.....	405
Appendix 15 – Seagrass and Mangroves	409
Seagrass modelling.....	409
Comparing Seagrass Model outputs with data	410
Mangrove Modelling.....	413
Comparing Mangrove Model outputs with data.....	415
Modelling relative changes in seagrass and mangroves.....	417
Background Information Summary : Expert overview of GoC habitat groups.....	418
Appendix 16 – MICE equations, variable definitions and input values.....	427
Appendix 17 – MICE Ensemble Additional Model Results under baseline flows	455
Prawns	455
Barramundi.....	458
Mud crab	469
Sawfish.....	472
Additional Results – sensitivity to adding trophic linkages (Model 6)	473
Appendix 18 – MICE WRD Additional Model Results	477
Seagrass	477
Mangroves.....	480
Prawns	484
Barramundi.....	490
Mud crabs.....	497
Sawfish.....	504
Appendix 19 – MICE WRD All Scenarios Results	509
Appendix 20 – MICE WRD Additional Sensitivities: prawn fishing power, barramundi selectivity, sawfish model structure and mangrove growth assumptions	529
Appendix 21 – MICE WRD Risk Analysis	545
Appendix 22 – Literature Cited	565

Tables

Table 1. Summary of Biological data for the key species assessed in the MICE model.....	28
Table 2. Summary of physical variables used to drive components of the MICE model.	29
Table 3. Summary of MICE species or groups, together with time-step used in modelling their dynamics, form of equation and relative data availability. Species not discussed in detail in this report are shaded in grey.....	40
Table 4. Summary of the MICE ensemble (five model versions) highlighting key differences between the model versions for each MICE sub-model. See Appendix 16 for full equations and model inputs values.	56
Table 5. Water Resource Development configurations (extraction and offstream storage) deployed within the MICE model for the Mitchell River featuring two water resource allocations (GL/year), three	

thresholds of flow (ML/day) below which pumping cannot occur, and two pump rates (days to pump water).....	63
Table 6. Description of the 19 water resource development (WRD) scenarios for the Mitchell, Flinders and Gilbert Rivers tested using the MICE. WRD involved either the extraction of water or the impoundment of water (i.e. dams). For water extraction, a description of the annual allocation (gigalitres per annum), flow threshold (TH) and pump rate (PR) are provided. Low TH = 200 ML d ⁻¹ ; medium TH = 1200 ML d ⁻¹ ; high TH = 2000 ML d ⁻¹ ; low PR = 30 days to pump water allocation; high PR = 15 days to pump water allocation. Current WRD for Flinders and Gilbert Rivers involves minor WRD extraction for domestic-settlement use.	64
Table 7. Consequence score for ERAEF activities (Modified from Fletcher et al. 2002 and copied from Sporcic et al. 2020).....	66
Table 8. Summary of risk scoring criteria used for MICE outputs, noting that relatively smaller declines in abundance or catch are required for the same risk score when assessed at the regional rather than local scale (to account for the fact that regional influences account for a much broader spatial effect as well as having greater consequences).....	67
Table 9. Tropical Australian rivers subject to Water Resource Assessment and Management research to scope catchments suitable for irrigated agriculture in northern Australia. River flow allocations (GL per annum) and type of WRD are shown for various rivers from different project reports.	74
Table 10. MICE model parameter estimates, associated Hessian-based standard deviations (std), number of parameters estimated, negative log likelihood (-lnL) contributions and Akaike Information Criterion (AIC) scores across each of the five model versions for (1) common banana prawn, (2) barramundi and (3) mud crab. Parameter estimates in italics are fixed parameters. Bsp = spawning biomass (t); CAA = catch-at-age; SOI = southern oscillation index; Mitch = Mitchell River; Gilb = Gilbert River; Norm = Norman River; Flind = Flinders River.	81
Table 11. Summary of the top 15 <i>P. merguensis</i> catch years (using catch totals over reference week period 13-22) shown alongside the average annual flow anomalies for each river system. The yellow highlights show the river system with the highest positive flow anomaly in the corresponding year, followed by orange highlights. The red highlights show the two years that didn't make the top 5 catches in this list, compared with when sorting by total annual catch as per Table 7 following.....	99
Table 12. Summary of the top 15 <i>P. merguensis</i> catch years (using catch totals over entire year) shown alongside the average annual flow anomalies for each river system for the years (highlighted in green) that were not included in the top 15 list shown in Table 1. The yellow highlights show the river system with the highest positive flow anomaly in the corresponding year, followed by orange highlights.	100
Table 13. Summary of the 5 lowest <i>P. merguensis</i> catch years (using (A) catch totals over weeks 13-22 and (B) totals over entire year) shown alongside the average annual flow anomalies for each river system. The yellow highlights show the river system with the lowest flow anomaly (i.e. values less than one show that the flow that year was below average, sometimes as low as 1% or 10% of the average level).	101
Table 14. Summary of MICE ensemble current depletion levels (No. mature(2020)/No. mature (1970)) simulated for largemouth sawfish in each of the river catchment systems as shown, and intended to capture a broad range of plausible depletion scenarios for sawfish, as a basis for testing the influence of WRDs on this species.....	125
Table 15. Summary of the MICE sawfish ensemble variation in relative abundance in each region (Rgn) under the four alternative model representations Models 2-5) when compared with the Model version 1 abundance levels. The values in the Table are thus always 1 for Model version 1, and show differences in other model versions relative (Rel) to Model 1, for statistics as follows: the minimum (Min) and average (Mean) total number of animals (N), the minimum and average number of mature (breeding age) animals (Bsp) and the minimum and average number of new recruits i.e. 1-year old animals joining the population for the first time (Rec).	128

Table 16. Summary table of Model versions 1-5 showing minimum and mean biomass for seagrass and mangroves predicted under four water resource development scenarios (WRD1-WRD4) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.	149
Table 17. Summary table of Model versions 1-5 showing minimum and mean catch, commercially available biomass (Bcom) and spawning biomass (Bsp) for common banana prawns predicted under four water resource development scenarios (WRD1-WRD4) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.	158
Table 18. Summary table of Model versions 1-5 showing minimum and mean catch, commercially available biomass (Bcom) and spawning biomass (Bsp) for barramundi predicted under four water resource development scenarios (WRD1-WRD4) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.	165
Table 19. Summary table of Model versions 1-5 showing minimum and mean catch, commercially available biomass (Bcom) and spawning biomass (Bsp) for mud crabs predicted under four water resource development scenarios (WRD1-WRD4) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.	174
Table 20. Summary table of Model versions 1-5 showing minimum and mean number of individuals, and number of mature individuals (Bsp) for sawfish predicted under four water resource development scenarios (WRD1-WRD4) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.	181
Table 21. The economic impacts of the main WRD scenarios (the change in profit and the change in NPV compared to the estimates for an average biomass year). Estimates are provided for the minimum (Min) and mean (Mean) biomass predicted for each WRD scenario.	190
Table 22. Summary results using MICE Model version 1 showing the maximum and average catch losses and spawning biomass (Bsp) percentage decreases for common banana prawns predicted under three water resource development scenarios (WRD1, WRD2, WRD5) relative to base line flows for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.	205
Table 23. Example of fishery and population standardised risk scores for common banana prawns, barramundi and mud crabs for the Mitchell, Flinders and Gilbert catchments as well as Regions 2-6 combined. Results are based on Model version 1. Other model versions are shown in Appendix 21. Bcom = commercially available biomass, Bsp = spawning biomass.	208
Table 24. Example of population standardised risk scores for sawfish for the Mitchell, Flinders and Gilbert catchments as well as Regions 2-6 combined. Results are based on Model version 1. Other model versions are shown in Appendix 21. N = total numbers, Bsp = mature numbers.	210
Table 25. Example of habitat standardised risk scores for seagrass and mangroves for the Mitchell, Flinders and Gilbert catchments as well as Regions 2-6 combined. Results are based on Model version 1. Other model versions are shown in Appendix 21.	211
Table 26. Fishery and population standardised risk scores for WRDs 1-4 averaged across the MICE ensemble for Regions 2-6. These results are similarly shown in Fig. 1 but with standard deviations also.	212
Table 27. Comparison of common banana prawn economic risks in the baseline (no-WRD) scenario and under alternative WRD scenarios, evaluated using each of the five models comprising the MICE ensemble and for all years from 1970-2019 and across the entire Gulf.	216
Table A1. Economic profit and net present value (NPV) calculations for a range of WRD scenarios.	271

Table A2. Water resource development scenarios and related average catch, minimum catch and percentage of base catches.....	278
Table A3. Summary results for WRD scenarios as compared to Base (first row)	282
Table A4. Water Resource Management scenarios in the Flinders River catchment within scope (FGARA) and modelled by Bayliss <i>et al.</i> 2014. PLUS no WRD Baseline.	291
Table A5. Water Resource Management scenarios in the Gilbert River catchment within scope (FGARA) and modelled by Bayliss <i>et al.</i> 2014. PLUS no WRD Baseline.	291
Table A6. Water Resource Management scenarios in the Mitchell River catchment within scope and modelled during the NAWRA analyses (Pollino <i>et al.</i> 2018 a,b).	293
Table A7. Summary of physical variables used to drive components of the MICE model.....	296
Table A8. Model estimates of parameters A, B, C and D which represent respectively the amplitude, period, phase shift and vertical shift of a SST-week relationship fitted to three years' data.....	298
Table A9. Summary of BOM Stations and period for air temperature data including which model regions these data were used for.	302
Table A10. Model estimates of parameters A, B, C and D which represent respectively the amplitude, period, phase shift and vertical shift of a solar-week relationship fitted to five years' data.	305
Table A11. Summary of tropical cyclones recorded in the Gulf of Carpentaria from 1970-2019, including their estimated intensity (low or cyclone category 1-5) and allocation to MICE model region (1-8).....	308
Table A12. Summary of the parameters of the MICE non-spatial model for <i>P. esculentus</i> (P.e.), <i>P. semisulcatus</i> (P.s), and <i>P. merguensis</i> (P.m.).....	328
Table A13. Summary of model-estimated parameters and their corresponding Hessian-based standard deviations for <i>P. esculentus</i> (P.e.), <i>P. semisulcatus</i> (P.s), and <i>P. merguensis</i> (P.m.).....	329
Table A14. Several mud crab sub-models that were developed to assess which was best able to explain the observed mud crab catches in the GoC.	351
Table A15. Summary of the mud crab sub-model ensemble.....	358
Table A16. Negative log likelihoods (-lnL) for each sub-model 1 version (built up in step-wise process) and each region.....	362
Table A17. Negative ln likelihoods for mud crab sub-model when using a parabolic vs logistic flow relationship	365
Table A18. Model 1d estimated parameter values and associated Hessian-based 90% confidence intervals (CI), number of parameters estimated and the negative log likelihood (-lnL) for each model region.	370
Table A19. Sawfish listings under the EPBC legislation	388
Table A20. Population model parameters used for largetooth sawfish in the MICE, based on Pillans <i>et al.</i> (in press), and compared with parameters for a faster-growing species.....	390
Table A21. Detailed summary of life history parameters for <i>P. pristis</i> (extracted from Pillans <i>et al.</i> in press)	390
Table A22. Comparison of Model version 5 current depletion level (mature animals in 2020 compared with 1970 numbers) when using default setting of post-release survival (PRS) proportion as zero,	

compared with assuming increases in PRS from 25%, 75% to 90% of sawfish assumed to survive following release from incidental capture in each of the three regions as shown.	397
Table A23. Summary table of Sawfish sensitivity analyses (applied to Model version 5) and using different post-release survival proportions of 0, 0.25, 0.75 and 0.9. Values in the Table are minimum and mean number of individuals, and number of mature individuals (Bsp) for sawfish predicted under four water resource development scenarios (WRD1-WRD4) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.	401
Table A24. Model equations for common banana prawns (<i>psp</i> =prawns), barramundi (<i>sp</i> =barra), mud crabs (<i>sp</i> =mudcr) and sawfish (<i>sp</i> =saw) (definitions of variables and parameters are shown in Tables 2 & 3) and the log-likelihood function.	427
Table A25. Description of the variables of the MICE for common banana prawns (<i>psp</i> =prawns), barramundi (<i>sp</i> =barra), mud crabs (<i>sp</i> =mudcr) and sawfish (<i>sp</i> =saw)	436
Table A26. Input values and sources for parameter settings for common banana prawns (<i>psp</i> =prawns), barramundi (<i>sp</i> =barra), mud crabs (<i>sp</i> =mudcr) and sawfish (<i>sp</i> =saw) in MICE not included in Model Ensemble Tables.....	445
Table A27. Model equations for mangroves (<i>mangr</i>), seagrass (<i>seagr</i>), microphytobenthos (<i>Mb</i>), Meiofauna (<i>Mf</i>) (definitions of variables and parameters are shown in Table A28 and Table A29.....	449
Table A28. Description of the variables of the model for mangroves (<i>mangr</i>), seagrass (<i>seagr</i>), microphytobenthos (<i>Mb</i>), Meiofauna (<i>Mf</i>)	451
Table A29. Input values and sources for parameter settings not included in Model Ensemble Tables. ...	454
Table A30. MICE model parameter estimates, associated Hessian-based standard deviations (<i>std</i>), number of parameters estimated, negative log likelihood (-lnL) contributions and Akaike Information Criterion (AIC) scores across each of the five model versions for barramundi when comparing Model version 2 with Model version 6 which includes trophic linkages. Parameter estimates in italics are fixed parameters. Bsp = spawning biomass (t); CAA = catch-at-age; SOI = southern oscillation index; Mitch = Mitchell River; Gilb = Gilbert River; Norm = Norman River; Flind = Flinders River.	473
Table A31. MICE model parameter estimates, associated Hessian-based standard deviations (<i>std</i>), number of parameters estimated, negative log likelihood (-lnL) contributions and Akaike Information Criterion (AIC) scores across each of the five model versions for barramundi when comparing Model version 2 with Model version 6 which includes trophic linkages. Parameter estimates in italics are fixed parameters. Bsp = spawning biomass (t); CAA = catch-at-age; SOI = southern oscillation index; Mitch = Mitchell River; Gilb = Gilbert River; Norm = Norman River; Flind = Flinders River	508
Table A32. Summary table of Model version 1 showing minimum and mean biomass for seagrass and mangroves predicted under 19 water resource development scenarios (WRD1-WRD19) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.	509
Table A33. Summary table of Model version 1 showing minimum and mean catch, commercially available biomass (<i>Bcom</i>) and spawning biomass (<i>Bsp</i>) for common banana prawns predicted under 19 water resource development scenarios (WRD1-WRD19) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.	511
Table A34. Summary table of Model version 1 showing minimum and mean catch, commercially available biomass (<i>Bcom</i>) and spawning biomass (<i>Bsp</i>) for barramundi predicted under 19 water resource development scenarios (WRD1-WRD19) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.....	513

Table A35. Summary table of Model version 1 showing minimum and mean catch, commercially available biomass (Bcom) and spawning biomass (Bsp) for mud crabs predicted under 19 water resource development scenarios (WRD1-WRD19) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.....	515
Table A36. Summary table of Model version 1 showing minimum and mean number of individuals, and number of mature individuals (Bsp) for largemouth sawfish predicted under 19 water resource development scenarios (WRD1-WRD19) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.....	517
Table A37. Summary table of MICE ensemble showing minimum and mean biomass for seagrass and mangroves predicted under 19 water resource development scenarios (WRD1-WRD19) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.	519
Table A38. Summary table of MICE ensemble showing minimum and mean catch, commercially available biomass (Bcom) and spawning biomass (Bsp) for common banana prawns predicted under 19 water resource development scenarios (WRD1-WRD19) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.	521
Table A39. Summary table of Model version 1 showing minimum and mean catch, commercially available biomass (Bcom) and spawning biomass (Bsp) for barramundi predicted under 19 water resource development scenarios (WRD1-WRD19) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.....	523
Table A40. Summary table of MICE ensemble showing minimum and mean catch, commercially available biomass (Bcom) and spawning biomass (Bsp) for mud crabs predicted under 19 water resource development scenarios (WRD1-WRD19) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.....	525
Table A41. Summary table of MICE ensemble showing minimum and mean number of individuals, and number of mature individuals (Bsp) for largemouth sawfish predicted under 19 water resource development scenarios (WRD1-WRD19) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.....	527
Table A42. Additional sensitivity analysis MICE model parameter estimates, associated Hessian-based standard deviations (std), number of parameters estimated, negative log likelihood (-lnL) contributions and Akaike Information Criterion (AIC) scores for Model 5 of the Ensemble compared with additional sensitivity results from Model 7 for (1) common banana prawn, (2) barramundi. Parameter estimates in italics are fixed parameters. Bsp = spawning biomass (t); CAA = catch-at-age; SOI = southern oscillation index; Mitch = Mitchell River; Gilb = Gilbert River; Norm = Norman River; Flind = Flinders River. ...	538
Table A43. Summary table of Model sensitivity version 7 compared with Ensemble Model version 5 showing minimum and mean catch, commercially available biomass (Bcom) and spawning biomass (Bsp) for common banana prawns predicted under four water resource development scenarios (WRD1-WRD4) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.	540
Table A44. Summary table of Model sensitivity version 7 compared with Ensemble Model version 5 showing minimum and mean catch, commercially available biomass (Bcom) and spawning biomass (Bsp) for barramundi predicted under water resource development scenarios (WRD1-WRD4) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.	541
Table A45. Summary table of Model sensitivity version 7 compared with Ensemble Model version 5 showing minimum and mean number of individuals, and number of mature individuals (Bsp) for sawfish predicted under four water resource development scenarios (WRD1-WRD4) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.	543

Table A46. Summary table of Model sensitivity version 7 compared with Ensemble Model version 5 showing minimum and mean biomass for mangroves predicted under four water resource development scenarios (WRD1-WRD4) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.....	544
Table A47. Example of fishery and population standardised risk scores for banana prawns, barramundi and mud crabs for the Mitchell, Flinders and Gilbert catchments as well as Regions 2-6 combined. Results are based on Model version 2. Bcom = commercially available biomass, Bsp = spawning biomass.	545
Table A48. Example of fishery and population standardised risk scores for banana prawns, barramundi and mud crabs for the Mitchell, Flinders and Gilbert catchments as well as Regions 2-6 combined. Results are based on Model version 3. Bcom = commercially available biomass, Bsp = spawning biomass.	547
Table A49. Example of fishery and population standardised risk scores for banana prawns, barramundi and mud crabs for the Mitchell, Flinders and Gilbert catchments as well as Regions 2-6 combined. Results are based on Model version 4. Bcom = commercially available biomass, Bsp = spawning biomass.	549
Table A50. Example of fishery and population standardised risk scores for banana prawns, barramundi and mud crabs for the Mitchell, Flinders and Gilbert catchments as well as Regions 2-6 combined. Results are based on Model version 5. Bcom = commercially available biomass, Bsp = spawning biomass.	552
Table A51. Example of population standardised risk scores for sawfish for the Mitchell, Flinders and Gilbert catchments as well as Regions 2-6 combined. Results are based on Model versions 2-5. N = total numbers, Bsp = mature numbers.	555
Table A52. Example of habitat standardised risk scores for seagrass and mangroves for the Mitchell, Flinders and Gilbert catchments as well as Regions 2-6 combined. Results are based on Model versions 2-5.	557

Figures

Figure 1. Participants at the first stakeholder workshop (noting some participants joined by teleconference or only attended on the second day)	14
Figure 2. Schematic showing steps and processes involved in overall project	18
Figure 3. Schematic summary of Gulf of Carpentaria MICE modelling process and MICE structure. See glossary for technical terms used and model terminology	19
Figure 4. Schematic summary of the key species and drivers of ecosystem change in the Gulf of Carpentaria system	20
Figure 5. Spatial map of Gulf of Carpentaria study area showing sub-division into 8 connected spatial regions with major river catchments as follows (from East to West, with the vertical line between Region 6 and 7 representing the Qld-NT border): Region 1: Embley River (Weipa); Region 2: Mitchell River (Kowanyama); Region 3: Gilbert River; Region 4: Norman River (Karumba); Region 5: Flinders River; Region 6: border region; Region 7: Roper River; Region 8: Walker River. Model sub-components (common banana prawn, barramundi and mud crab) were fitted to catch data per region. For Region 2, barramundi and mud crab data were only used from the southern part of this region, indicated by dashed horizontal line, because data from the northern part of Region 2 weren't considered to be representative of the Mitchell River end-of-system flows.	21

Figure 6. Model schematic showing (A) the key species groups and linkages identified for modelling in the spatial Gulf of Carpentaria MICE. Freshwater river flow inputs are highlighted in light blue as they are a key driver and the other physical drivers are shown with dark grey colouring, whereas the fishery sectors are shaded blue-grey. (B) Focus species and linkages in the spatial Gulf of Carpentaria MICE that were included in the MICE when testing water resource development (WRDs) scenarios.....	24
Figure 7. Plot showing the correspondence between total fishing effort for tiger prawns and common banana prawns mapped with the three selectivity periods selected for the MICE model.	35
Figure 8. Distribution of effort targeted at the common banana prawns (green) and tiger prawns (blue) across the two fishing seasons in the NPF. Effort is summed boat days over the 2009-2018 fishing seasons as shown. Only locations with at least 8 days for tiger prawns and 6 days for common banana prawns over the year are shown. GoC- Gulf of Carpentaria; HS-hotspot. Modified from Pascoe et al. (2020).....	36
Figure 9. Fishing power estimates used to standardise nominal catch rates for common banana prawns and for the two species of tiger prawns in the MICE model.....	37
Figure 10. Summary of the relative abundance of the three prawn species across the 8 MICE model regions as shown.	37
Figure 11. The proportion of <i>P. merguensis</i> total catch that derives from each of the 8 MICE regions, based on an overall average across all years (1970-2019) compared with more recent years only (2002-2019).....	38
Figure 12. Comparison of the total common banana prawn <i>P. merguensis</i> catch over the period 1970 to 2019 when summed over reference week period 13-22 compared with over the entire year. The lower plots with fitted regression lines show how well the annual catch over weeks 13-22 predicts the total annual catch when (left) considering all years, and (right) over the period 1988-2019.....	39
Figure 13. Schematic summary of the representation of (A) mud crabs; (B) barramundi and (C) largemouth sawfish in each spatial region of the MICE. The combined sex symbol indicates that the model is not disaggregated by sex in these examples.	41
Figure 14. Summary of key management differences pertaining to (A) the mud crab fisheries and (B) barramundi fisheries based in the Northern Territory (MICE regions 7-8) compared with Queensland (MICE regions 1-6). Note that for the NT, the barramundi fishing area is restricted to waters seaward from the coast, river mouths and legislated closed lines (NT Department of Industry, Tourism and Trade, barramundi fishery management plan 1998).....	42
Figure 15. Schematic showing MICE logistic (or linear) function used to describe the relationship between the standardised flow and a population parameter such as recruitment, where the function yields a multiplier that describes recruitment relative to the maximum value.	49
Figure 16. Schematic showing MICE logistic function used to describe the relationship between the standardised flow and a population parameter such as recruitment of prawns or barramundi, where the function yields a multiplier that describes recruitment relative to the maximum value. In this example, one of the parameters is fixed such that there is a single free parameter that can be estimated to better describe the relationship for different model regions and/or species, noting that the estimated curve is based on flow anomalies. For mud crabs, a parabolic function is used, see Appendix 11.....	50
Figure 17. Summary of (A) the MICE modelling process and (B) adding complexity to the model.....	54
Figure 18. Comparison of the prey-interaction shape for different values of the steepness parameter H as shown.....	59
Figure 19. Examples of modelled flow estimates for selected Gulf of Carpentaria rivers from 1890 to 2018. Note the y-axis (vertical axis) is different across regions.....	71

Figure 20. Expanded view of modelled flow estimates for selected Gulf of Carpentaria rivers shown for the period corresponding to that for which historical fishery data are available for key species in the ecological model, namely 1970 to 2018. Note the y-axis (vertical axis) is different across regions.	72
Figure 21. Examples of intra-annual variation in flow compared between three illustrative years (1974, 2015 and 2018) for selected Gulf of Carpentaria rivers as shown. Note the y-axis (vertical axis) is different across regions.	73
Figure 22. MICE estimated logistic functions used to describe the relationship between the standardised flow and common banana prawn recruitment for each of the catchment systems as shown, where the function yields a multiplier that describes recruitment relative to the maximum value (noting the function asymptotes at 1 for flows greater than twice the median flow level).	76
Figure 23. Comparison of the observed and model-predicted annual common banana prawn catch (tonnes) estimated using only fishing effort (i.e. no flow linked to banana prawn population dynamics) for each of the eight model regions as shown.	78
Figure 24. Comparison of the observed and model-predicted annual common banana prawn catch (tonnes) estimated using Model version 1 (driven by baseline flows) and for each of the eight model regions as shown.	79
Figure 25. Comparison of the observed and model-predicted annual common banana prawn catch (tonnes) estimated using Model version 5 (driven by baseline flows, prawn survival linked with lagged overall index of primary productivity) and for each of the eight model regions as shown.	80
Figure 26. MICE ensemble outputs shown for the five alternative model configurations used to represent common banana prawns. Trajectories show the model-estimated commercially available biomass (Bcom) in units of tonnes shown from 1989 to 2019, for the different model regions and jurisdictions as indicated. See Table 4 for a description of the model versions.	90
Figure 27. MICE ensemble outputs shown for the five alternative model configurations used to represent common banana prawns. Trajectories show the model-estimated catch (tonnes) from 1989 to 2019, for the different model regions and jurisdictions as indicated. See Table 4 for a description of the model versions.	91
Figure 28. MICE ensemble outputs shown for the five alternative model configurations used to represent common banana prawns. Trajectories show the model-estimated spawning biomass (Bsp) plotted in each instance as relative to the 1989 spawning biomass (dashed line) estimate to highlight relative differences over time. See Table 4 for a description of the model versions.	92
Figure 29. MICE ensemble outputs shown for the five alternative model configurations used to represent common banana prawns. Trajectories show the model-estimated catch (tonnes) from 1989 to 2019, for the three key model regions as indicated, together with the combined catch for Regions 2-6. See Table 4 for a description of the model versions.	93
Figure 30. The relative contribution of the different model regions to the overall <i>P. merguensis</i> catch, when comparing the average over the entire time period (since 1970), with the average since 1980 and a more recent average over 2002-2018.	94
Figure 31. MICE model results showing (A) Model version 1 and (B) Model version 5 estimates of the relative contribution to <i>P. merguensis</i> recruitment (to the fishery) per model region of flow anomalies contributed by the different river systems. Areas 1 and 7 are assumed to only depend on the Embley and Roper Rivers respectively (hence why there is a single river shown as influencing recruitment in those areas), and the difference in the magnitudes reflect that the recruitment in the Embley area depends less on flow anomalies than is the case in the Roper River region. For regions 2-5, the model estimates relative contributions that are bounded between 0 and 1 (i.e. results suggest no effect of a river on that region's prawn recruitment, or varying influences). Hence for example, model results suggest that the Norman River is the dominant driver of prawn recruitment in model region 4 with some contribution from the	

Flinders River. The Norman River is also estimated to be an important driver of recruitment in the Gilbert River model region 3. In contrast, prawn recruitment in the Mitchell and Flinders region is predicted to be driven on average by a combination of flow anomalies across all four river systems. 96

Figure 32. MICE model results (version 5) showing for each of the four model regions, where region 2 corresponds to the Mitchell River catchment, 3 to the Gilbert River catchment, 4 to the Norman River catchment and 5 to the Flinders River catchment, what the relative contributions are to explaining variability in catches taken from that region, based on influences from end-of-system flows from each of the four major rivers. Hence for example, recruitment to the fishery in the Norman River region is influenced predominantly by the Norman River flows themselves, as well as influences from the Flinders River system, whereas variation in prawn catches taken from the Flinders River region are estimated to be influenced substantially by the Mitchell, Gilbert and Norman River flows. 97

Figure 33. MICE model results shown in Figure 31 combined with estimates of the average *P. merguensis* catch taken from each model region to standardise how these local recruitment fluctuations translate into overall contributions to total catch. Hence these results show the model-estimated relative contribution of each of the river systems (in turn mapped to model regions 1 to 8) to the total average *P. merguensis* recruitment and hence catches in the GoC. 98

Figure 34. MICE estimated base selectivity curves for the barramundi fishery applied to the northern Queensland region (Regions 1 and 2), southern Queensland regions (Regions 3-6) and Northern Territory (Region 7-8), as well as the “early” selectivity curve used for the Qld model regions prior to 1997 (see Appendix 16 for derivation). 102

Figure 35. MICE estimated age-dependent natural mortality (shown as annual values converted from the monthly mortality rates used in the model) showing the range of forms used in the model ensemble. The “base” *M* is used in Models 1,2 and 5. The dashed grey lines are simply for ease of comparison with the simpler assumption (not used in the MICE) that *M* is constant with age, with annual rates of 0.2 or 0.3 yr⁻¹ as commonly used in stock assessment. See Table 4 for a description of the model versions. 102

Figure 36. MICE estimated logistic functions used to describe the relationship between the standardised flow and barramundi recruitment for each of the catchment systems as shown, where the function yields a multiplier that describes recruitment relative to the maximum value. 104

Figure 37. Comparison of the observed and model-predicted annual barramundi catch (tonnes) estimated using Model version 1 (baseline flows, effort, fixed flow relationship parameters) and for each of the eight model regions as shown. 105

Figure 38. Comparison of the observed and model-predicted annual barramundi catch (tonnes) estimated using Model version 4 (baseline flows, effort, estimated flow relationship parameters and lower natural mortality rate) and for each of the eight model regions as shown. 106

Figure 39. Comparison of the observed and model-predicted annual commercial catch-at-age proportions of barramundi in each age-class as shown estimated using Model version 4 (baseline flows, estimated flow relationship parameters) for the Mitchell River catchment region from 2000 to 2019. Light blue (observed) and gold (predicted) bars indicate strong cohorts (otherwise known as year-classes) that can be followed over time through the catch-at-age proportions. 107

Figure 40. Comparison of the observed and model-predicted commercial catch-at-age proportions of barramundi in each age-class as shown estimated using Model version 4 (baseline flows, estimated flow relationship parameters) for the combined catchment Model regions 3-6 from 2000 to 2019. Light blue (observed) and gold (predicted) bars indicate strong cohorts (otherwise known as year-classes) that can be followed over time through the catch-at-age proportions. 108

Figure 41. MICE ensemble outputs shown for the five alternative model configurations used to represent barramundi. Trajectories show the model-estimated commercially available biomass (Bcom) in units of tonnes shown from 1989 to 2019, for the different model regions and jurisdictions as indicated. See Table 4 for a description of the model versions. 109

Figure 42. MICE ensemble outputs shown for the five alternative model configurations used to represent barramundi. Trajectories show the model-estimated catch (tonnes) from 1989 to 2019, for the different model regions and jurisdictions as indicated. See Table 4 for a description of the model versions..... 110

Figure 43. MICE estimated parabolic functions used to describe the relationship between the standardised flow and mud crab recruitment for each of the catchment systems as shown, where the function yields a multiplier that describes recruitment relative to the base value. 90% confidence intervals are showed by dashed lines, colour-coded for each region. 112

Figure 44. Comparison of the observed and model-predicted annual mud crab catch (tonnes) estimated using only fishing effort (i.e. no flow linked to mud crab population dynamics) for each of the eight model regions as shown. Note fishing for regions 1-6 only start in 1989. 114

Figure 45. Comparison of the observed and model-predicted annual mud crab catch (tonnes) estimated using Model version 1 (driven by baseline flows, effort and other environmental predictors e.g. SOI for regions 3-6 and temperature for region 7) and for each of the eight model regions as shown. Note fishing for regions 1-6 only start in 1989. 115

Figure 46. Comparison of the observed and model-predicted annual mud crab catch (tonnes) estimated using Model version 5 (driven by baseline flows, effort and other environmental predictors e.g. SOI for regions 3-6 and temperature for region 7, and a more conservative stock-recruitment steepness parameter) and for each of the eight model regions as shown. Note fishing for regions 1-6 only start in 1989. 116

Figure 47. MICE ensemble outputs shown for the five alternative model configurations used to represent mud crabs. Trajectories show the model-estimated commercially available biomass (Bcom) plotted as relative to the 1989 available biomass, for the different model regions and jurisdictions as indicated. See Table 4 for a description of the model versions. 117

Figure 48. MICE ensemble outputs shown for the five alternative model configurations used to represent mud crabs. Trajectories show the model-estimated catch (tonnes) from 1989 to 2019, for the different model regions and jurisdictions as indicated. See Table 4 for a description of the model versions..... 118

Figure 49. MICE ensemble outputs shown for the five alternative model configurations used to represent mud crabs. Trajectories show the model-estimated commercially available biomass (Bcom) in units of tonnes shown from 1989 to 2019, for the different model regions and jurisdictions as indicated. See Table 4 for a description of the model versions. 119

Figure 50. MICE ensemble outputs shown for the five alternative configurations used to represent mud crabs. Trajectories in these plots are the model-estimated catch (tonnes) plotted since start of the fishery (1989) for the three key model regions as indicated, together with the combined catch for Regions 2-6. See Table 4 for a description of the model versions. 120

Figure 51. Two illustrative alternative MICE representations of largemouth sawfish in the vicinity of the Mitchell River region, comparing depletion trajectories (relative to 1970 level) based on mature numbers with total numbers and new recruits, where the relative recruitment is plotted using the second vertical axis. The bars are the average of the standardised flow in the first 12 weeks of each year (note though that sawfish recruitment and juvenile survival uses the standardised flows as an input to an additional flow-recruitment relationship). Total recruitment depends on population size, as well as flows, with variable boom-bust dynamics also simulated..... 123

Figure 52. Example of model simulation for largemouth sawfish in the Roper River region to highlight the influence of flow (blue bars) on recruitment (red line, plotted against the second vertical axis.) and lag time before signal is seen in mature numbers (grey line)..... 124

Figure 53. Two illustrative MICE representations of largemouth sawfish in the vicinity of the Mitchell River (top) and Flinders River (bottom) region similar to previous figure, but showing changes in fishery interactions (based on effort in prawn trawl and inshore gillnet fisheries) over time as these influence population trajectories in addition to influences from changes in flow..... 124

Figure 54. MICE ensemble outputs shown for the five alternative configurations used to represent largemouth sawfish. The model trajectories represent the relative total number of animals plotted in each instance as relative to the 1989 reference numbers to highlight relative differences over time. 126

Figure 55. MICE ensemble outputs shown for the five alternative configurations used to represent largemouth sawfish. The model trajectories represent the total number of mature animals relative to the 1989 abundance level. Note that the label of “Bsp” is the closest comparable to the spawning biomass index used for the prawns, barramundi and mud crabs, but for sawfish (which do not spawn but produce a relatively small number of pups) the reproductively mature population is represented in units of numbers rather than biomass. Numbers are more meaningful for sawfish than biomass because the relationship between fecundity and length is not likely to be teleost like. 127

Figure 56. MICE ensemble outputs shown for the five alternative configurations used to represent relative changes in seagrass abundance (baseline flows) in all the model regions as shown. 129

Figure 57. Comparison of the changes in the abundance of seagrass, relative to the 1989 level, for each of the five model versions under baseline flows. 130

Figure 58. MICE ensemble outputs shown for the five alternative configurations used to represent relative changes in mangrove abundance (baseline flows) in all the model regions as shown. 131

Figure 59. Comparison of the changes in the abundance of mangroves, relative to the 1989 level, for each of the five model versions under baseline flows. 132

Figure 60. Example of MICE physical drivers and changes in the relative biomass and catch (tonnes) of the base model groups for Region 4 (Norman River) over 1980 to 2019. From top panel to bottom panel: flow, seagrass, mangroves, microphytobenthos, common banana prawn, barramundi, mud crab and sawfish. The relative influences of cyclones (black bars) and solar radiation (yellow line) on seagrass is shown by superimposing these physical variables on the seagrass biomass trajectory. Changes in air temperature (red dashed line) which can affect mangroves is also shown. 134

Figure 61. Example of MICE physical drivers and changes in the relative biomass and catch (tonnes) of the base model groups for Region 7 (Roper River) over 1980 to 2019. From top panel to bottom panel: flow, seagrass, mangroves, microphytobenthos, common banana prawn, barramundi, mud crab and sawfish. The relative influences of cyclones (black bars) and solar radiation (yellow line) on seagrass is shown by superimposing these physical variables on the seagrass biomass trajectory. Changes in air temperature (red dashed line) which can affect mangroves and mud crabs is also shown. 135

Figure 62. Comparison of GoC river flows using Region 2 (Mitchell River) as an example and changes in the intra-annual biomass during (left) a wet year (2009) and (right) a dry year (2015). From top panel to bottom panel: flow, seagrass, mangroves, microphytobenthos, common banana prawn, barramundi and mud crab. For common banana prawn, barramundi and mud crab, biomass is the commercially available biomass (divided by the long-term average) and catches (tonnes) are shown on the second vertical axis. 136

Figure 63. Comparison of GoC river flows using Region 5 (Flinders River) as an example and changes in the intra-annual biomass during (left) a wet year (2011) and (right) a dry year (2015). From top panel to bottom panel: flow, seagrass, mangroves, microphytobenthos, common banana prawn, barramundi and mud crab. For common banana prawn, barramundi and mud crab, biomass is the commercially available biomass (divided by the long-term average) and catches (tonnes) are shown on the second vertical axis. 137

Figure 64. Comparison of GoC river flows using Region 7 (Roper River) as an example and changes in the intra-annual biomass during (left) a wet year (2011) and (right) a dry year (2015). From top panel to bottom panel: flow, seagrass, mangroves, microphytobenthos, common banana prawn, barramundi and mud crab. For common banana prawn, barramundi and mud crab, biomass is the commercially available biomass (divided by the long-term average) and catches (tonnes) are shown on the second vertical axis. 138

Figure 65. Flow difference plots within the Flinders River between historical baseline unimpeded flows and one of the WRD scenarios to be deployed in the river (175 GL year⁻¹ water harvest and offshore storage). 140

Figure 66. Flow difference plots within the Gilbert River between historical baseline unimpeded flows and one of the WRD scenarios to be deployed in the river (172 GL year⁻¹ water impoundment, Green Hills dam). Note that while dams do have a dampening effect on peak flows, it is possible to get more flows (positive difference) at certain times under WRD involving dams due to a large surface area of water (dam), where there would otherwise be land..... 141

Figure 67. Flow difference plots within the Mitchell River between historical baseline unimpeded flows and one of the WRD scenarios to be deployed in the river (1313 GL year⁻¹ water impoundment, Nillinga and Pinnacles dams). Note that while dams do have a dampening effect on peak flows, it is possible to get more flows (positive difference) at certain times under WRD involving dams due to a large surface area of water (dam), where there would otherwise be land..... 142

Figure 68. MICE-predicted changes in seagrass relative biomass under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 1. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, and WRD1 tracks under WRD4..... 145

Figure 69. MICE-predicted changes in seagrass relative biomass under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 5. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4..... 146

Figure 70. MICE-predicted changes in mangrove relative biomass under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 1. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4..... 148

Figure 71. MICE-predicted changes to total annual common banana prawn commercially available biomass (Bcom, t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 1. 153

Figure 72. MICE-predicted changes to total annual common banana prawn catch (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 1. 154

Figure 73. MICE-predicted changes to total annual common banana prawn commercially available biomass (Bcom, t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 2 (no connectivity between systems). Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4. 155

Figure 74. MICE-predicted changes to total annual common banana prawn catch (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 2. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4..... 156

Figure 75. MICE-predicted changes to total common banana prawn catch (t) under baseline flow conditions compared with four alternative WRDs, shown for the Mitchell, Flinders and Gilbert catchment

systems, and when using best-fit Model version 5 which assumes a “flood productivity effect” and predicts relatively greater responses of prawn abundance and catch to changing river flows. 157

Figure 76. MICE-predicted changes to total annual barramundi catch (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 1. Note for the Mitchell River, BC trajectory tracks under WRD4 and for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2 tacks under WRD4. 162

Figure 77. MICE-predicted changes to total annual barramundi catch (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 2. Note for the Mitchell River, BC trajectory tracks under WRD4 and for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4..... 163

Figure 78. MICE-predicted changes to total annual barramundi spawning biomass (Bsp) (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 2. Note for the Mitchell River, BC trajectory tracks under WRD4 and for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4. 164

Figure 79. MICE-predicted changes to total annual mud crab catch (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 1. Note for the Mitchell River, the BC and all WRDs track under each other. For the other panels, the BC trajectory tracks under WRD3 and WRD2 tacks under WRD4. 171

Figure 80. MICE-predicted changes to total annual mud crab catch (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 2. Note for the Mitchell River, the BC and all WRDs track under each other. For the other panels, the BC trajectory tracks under WRD3 and WRD2 tacks under WRD4. 172

Figure 81. MICE-predicted changes to total annual mud crab catch (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 5 (best-fit model). Note for the Mitchell River, the BC and all WRDs track under each other. For the other panels, the BC trajectory tracks under WRD3 and WRD2 tacks under WRD4. 173

Figure 82. MICE-predicted changes to total largemouth sawfish numbers (relative to 1989 level) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 1. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4. 178

Figure 83. MICE-predicted changes to total largemouth sawfish mature adult numbers (Bsp) (relative to 1989 level) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 1. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4. 179

Figure 84. Comparison of predicted changes in response to WRDs when using Model version 4 (top) versus Model version 5 (bottom) incorporating different current depletion assumptions for largemouth sawfish. The left set of trajectories are total number whereas the right set are mature (Bsp) adult numbers (relative to 1989 level). WRDs scenarios are compared with the baseline flow conditions for the Mitchell, Flinders and Gilbert catchment systems. Model version 4 has baseline current depletion levels for the Mitchell, Gilbert and Flinders catchment systems of 45%, 19% and 6% compared with the following

levels for Model version 5: Mitchell 17% (less optimistic), Gilbert 4%(less optimistic) and Flinders (18% (more optimistic)).	180
Figure 85. Comparison of the barramundi commercially available biomass (Bcomm) in the (A) Mitchell catchment, (B) Gilbert River catchment and (C) Flinders River catchment regions under baseline flow conditions to compare the model version 2 estimated trajectory with Model version 6 (trophic link added to represent barramundi preying on common banana prawns).	185
Figure 86. MICE-predicted changes to total annual barramundi catch (t) using Model version 6 (trophic link added to represent barramundi preying on common banana prawns) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 2. Note for the Mitchell River, BC trajectory tracks under WRD4 and for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4.	186
Figure 87. Comparison of the largemouth sawfish relative abundance (total number) in the (A) Mitchell catchment, (B) Gilbert River catchment and (C) Flinders River catchment regions under baseline flow conditions to compare the model version 2 estimated trajectory with Model version 6 (trophic link added to represent barramundi preying on common banana prawns).	187
Figure 88. Comparison of predicted changes in relative abundance of largemouth sawfish in response to WRDs when using Model version 6 (additional trophic linkages added between largemouth sawfish and common banana prawns). WRDs scenarios are compared with the baseline flow conditions for the Mitchell, Flinders and Gilbert catchment systems.	188
Figure 89. Estimated total fishery profit in the NPF versus the initial common banana prawn biomass at the start of the banana prawn fishery season (Figure adapted from Hutton <i>et al.</i> 2022).	189
Figure 90. Predicted reductions in the Net Present Value (NPV) versus the simulated reduction in the common banana prawn biomass (based on model as specified in Hutton <i>et al.</i> (2022)).	190
Figure 91. (A) Example of baseline flows (m ³ /s) for Nov-Feb in a wet, intermediate and dry year for the Gilbert catchment (Model Region 3), with corresponding common banana prawn predicted catches (t) for these years showing percentage reduction when flow reduced. (B) Baseline flows can be significantly reduced under WRD1, and corresponding catches were also reduced under WRD1, although not to the same extent under varying baseline flows.	196
Figure 92. (A) Example of baseline flows (m ³ /s) for Nov-Feb in a wet, intermediate and dry year for the Flinders catchment (Model Region 5), with corresponding common banana prawn predicted catches (t) for these years showing percentage reduction when flow reduced. (B) Baseline flows can be significantly reduced under WRD1, and corresponding catches were also reduced under WRD1, although not to the same extent under varying baseline flows.	197
Figure 93. (A) Example of baseline flows (m ³ /s) for Nov-Feb in a wet, intermediate and dry year for the Gilbert catchment (Model Region 3), with corresponding barramundi predicted catches (t) for these years showing percentage reduction when flow reduced. (B) Baseline flows can be significantly reduced under WRD1, and corresponding catches were similarly reduced under WRD1.	199
Figure 94. (A) Example of baseline flows (m ³ /s) for Nov-Feb in a wet, intermediate and dry year for the Gilbert catchment (Model Region 3), with corresponding mud crab predicted catches (t) for these years showing percentage reduction when flow reduced. (B) Baseline flows can be significantly reduced under WRD1, and corresponding catches were similarly reduced under WRD1.	201
Figure 95. (A) Example of baseline flows (m ³ /s) for Nov-Feb in a wet, intermediate and dry year for the Flinders catchment (Model Region 5), with corresponding mud crab predicted catches (t) for these years showing percentage reduction when flow reduced. (B) Baseline flows can be significantly reduced under WRD1, and corresponding catches were similarly reduced under WRD1.	202

Figure 96. Extract of key results from Broadley et al. (2020) for the purpose of comparing with the MICE results, where the Broadley et al. (2020) high water extraction scenario compares most closely with the MICE water extraction scenario WRD5 and the low and medium water extraction scenarios are more similar to MICE WRD1 and WRD2 (see text for detail).	204
Figure 97. Comparison of average (with standard deviations) (A) Population risk and (B) Fishery Risk using the MICE ensemble to evaluate the impact of alternative WRDs on species and habitat groups as shown.....	213
Figure 98. Comparison of average (with standard deviations) (A) Population risk and (B) Fishery Risk across three river catchments using the MICE ensemble to evaluate the impact of alternative WRDs on species and habitat groups for these catchments. C. banana prawn = common banana prawn.	214
Figure 99. Comparison of Economic Risk when using Model version 1 to evaluate the impact of alternative WRDs on common banana prawns, and when using as the threshold risk annual catch level (A) 2000t versus (B) 1475t.	216
Figure 100. Comparison of average (with standard deviations) Economic Risk when using the MICE ensemble to evaluate the impact of alternative WRDs on common banana prawns.	217
Figure 101. Graphical comparison of the relative probability of occurrence of major risks (defined as risk of a bad year), severe risk (two successive bad years) and intolerable risk (fishery operations becoming unviable due to three or more consecutive bad years) predicted in response to alternative WRDs impacting the common banana prawn fishery.....	217
Figure A1. Map of GoC spatial regions used in MICE	264
Figure A2. Importance of high proportions of natural flows.....	266
Figure A3. Key biological processes in the MICE model	267
Figure A4. Understanding underpinning the prawn population model	267
Figure A5. Banana prawn population model.....	268
Figure A6. Scenarios for Mitchell, Gilbert, Norman, Flinders regions and Region 6.....	268
Figure A7. Model outputs based on effort, catch and profit.....	271
Figure A8. Net economic returns for actual, 8% reduction and low flows	271
Figure A9. Mud crab historical catch data plots.....	277
Figure A10. Lost catches under various water resource development scenarios	278
Figure A11. Biomass depletion of mud crabs	279
Figure A12. Workshop day 2 participants.....	280
Figure A13. Three spatially separate Barramundi stocks map	281
Figure A14. Barramundi model preliminary results based on flow and fishing effort.....	281
Figure A15. Barramundi catches relative to base flow	282
Figure A16. Sawfish catches: based on NPF and barramundi fishing effort.....	284
Figure A17. MICE GoC model foodweb	285

Figure A18. Important factors for modelling mangrove communities.....	286
Figure A19. Example of preliminary mangrove results for the Norman River.....	286
Figure A20. Example of preliminary seagrass results for the Roper Region	287
Figure A21. Comparing two WRD scenarios.....	287
Figure A22. Illustrative key risk statistics for the key species under various WRD scenarios	288
Figure A23. Mean sea level plotted against monthly SOI for the period 1994-2019.....	297
Figure A24. Water quality monitoring sites in the Norman River Estuary at which water quality, estuarine productivity and meiofauna abundance in the Norman River was measured in 2008/09 experiments. SST measurements are available from a temperature logger at Karumba. Figure taken from Duggan et al. 2014.	298
Figure A25. (A) Weekly observed vs. modelled SST values for 2016; (B) modelled temperature compared to observed temperature in the Norman River (temperature logger) and observed temperature (BOM Groote Eylandt station representative of model region 7). (C) Mean and maximum SST from the BOM compared with model estimates.	300
Figure A26. (A) Salinity from Duggan et al. (2014) as a function of flow and (B) salinity from the Karumba Wharf logger as a function of flow.....	301
Figure A27. Snapshot of mean maximum weekly air temps for three sites: Weipa (for Region 1), Sweers Island (for Regions 2-6) and Groote Eylandt (for Regions 7-8).....	303
Figure A28. Mean and minimum sea level measurements obtained from the BOM for Milner Bay, Groote Eylandt for the period 1994-2020.....	303
Figure A29. Example of the inter-annual and regional variability in the solar exposure for each month of the year, highlighting also the clear seasonal signal of less solar exposure during the austral winter.	304
Figure A30. Example of the solar exposure pattern for (A) 2016 and (B) 2017 compared for sites Karumba (K), Groote-Eylandt (GE), Weipa (W).....	304
Figure A31. Illustrative example of fitted solar exposure annual pattern compared with 2016 and 2020 data (after standardisation) from the Karumba site.	305
Figure A32. Flow diagram showing the linkages between river flow and prawn migration and production. Yellow highlighted boxes show those elements which were a focus of the study.	319
Figure A33. Contrasting effects of high and low flow on prawns in the estuary. Stars highlight new information.	319
Figure A34. Schematic to illustrate how sampling spatially-aggregated populations can influence overall abundance estimates.	324
Figure A35. Simplistic plot comparing a linear relationship between a CPUE index and the assumed relative population abundance, with a hyperstability relationship, using as illustrations a mild (hyps = 0.8) and more extreme (hyps = 0.5) examples.....	324
Figure A36. Quarterly selectivity patterns for each of the three selectivity periods (1970-1987; 1988-2001; 2002-current) shown for the three modelled prawn species: brown tiger prawn <i>P. esculentus</i> (<i>P.esc</i>), grooved tiger prawn <i>P. semisulcatus</i> (<i>P.semi</i>) and common banana prawns <i>Penaeus merguensis</i> (<i>P.merg</i>). The parameters estimated by the model are shown with diagonal shading, and the same parameters are assumed for the two tiger species except for the second quarter of years 2002-2020 (last column) when small catches of <i>P.esc</i> but not <i>P.semi</i> are recorded.	326

Figure A37. Model-estimated recruitment residuals for tiger prawn species as shown.	327
Figure A38. Model-estimated spawning biomass (t) trajectories for the three prawn species as indicated.	330
Figure A39. Comparison between observed total CPUE (t/day; blue) of brown tiger prawns (<i>P. esc</i>), grooved (<i>P. semi</i>) and common banana prawns (<i>P. merg</i>) per quarter of the year from 1970 to 2018, compared with model-estimated CPUE (red).....	330
Figure A40. Annual observed fishery independent survey biomass and recruitment indices with associated standard deviations (blue) and modelled survey biomass and recruitment indices (green) for brown tiger prawns (<i>Penaeus esculentus</i>), grooved tiger prawns (<i>P. semisulcatus</i>) and common banana prawns (<i>P. merguensis</i>) from 2002 to 2019.....	331
Figure A41. Annual modelled spawning stock biomass indices (relative to mean; blue) and the 2018 stock assessment spawning stock biomass indices (relative to mean; red) for brown tiger prawns (<i>Penaeus esculentus</i>) and grooved tiger prawns (<i>P. semisulcatus</i>) from 1971 to 2017.	332
Figure A42. Comparison of total annual observed catches (t) of common banana prawns (<i>Penaeus merguensis</i>) from 1970 to 2018, compared with model-estimated catches.	333
Figure A43. Comparison between observed total catches (t; blue) of common banana prawns (<i>Penaeus merguensis</i>) per quarter from 1970 to 2018, compared with model-estimated catches (red).	333
Figure A44. Example of the commercially exploitable biomass of each of the three prawn species over the period 1980 to 2019.....	334
Figure A45. Example of model-predicted tiger prawn abundance fitted to both the recruitment (REC) and spawning (SPW) survey data for MICE region 8.....	335
Figure A46. Comparison between actual observed catches and model-predicted common banana prawn catches over 1970 to 2019 for each of the MICE regions as shown, and when using a simple linear flow-recruitment formulation.....	336
Figure A47. Comparison between actual observed catches and model-predicted common banana prawn catches over 1970 to 2019 for each of the MICE regions as shown, and when using a logistic flow-recruitment formulation and incorporating offshore productivity.....	337
Figure A48. Overview of the Mud crab (<i>Scylla serrata</i>) life cycle and associated habitat use. Adapted from Alberts-Hubatsch et al. (2016).....	345
Figure A49. Mean (\pm SD) percentage of females from sampled catches pooled across years (1991-1995) and river systems (Adelaide, McArthur and Roper Rivers). Taken from Knuckey (1999).	346
Figure A50. Summary of some of the key similarities and differences between the NT and Qld mud crab fisheries.	348
Figure A51. Fishery reporting grids in the Guld of Carpentaria for Northern Territory (yellow, 1° x 1°) and Queensland (orange, 0.5° x 0.5°). Adapted from Robins et al. (2020).....	350
Figure A52. Schematic of mud crab model structure showing age classes for male and female crabs captured in model.	351
Figure A53. Three flow-recruitment relationships were initially considered when choosing how to link in river flow to mud crab population dynamics (via recruitment).....	357
Figure A54. Annual effort per model region in (A) the Queensland GoC mudcrab fishery (number of days per year) and (B) the Northern Territory GoC mud crab fishery (number of pots per year).....	360

Figure A55. Annual catch (tons) per model region from (A) the Queensland GoC mudcrab fishery and (B) the Northern Territory GoC mud crab fishery.....	361
Figure A56. Observed vs predicted catch (tons) for each MICE region, when effort is the only driver of model predicted catch (Sub-model 1a, i.e. no physical variables included in the model).....	363
Figure A57. Observed vs predicted catch (tons) for each MICE region, when effort and river flow, as well as temperature (region 7 only) are included as drivers for model predicted catch (Sub-model 1d).....	364
Figure A58. Model-estimated flow-recruitment relationships for some of the key GoC rivers using (A) a logistic function and (B) a parabolic function.	366
Figure A59. Observed (blue) vs predicted (red) catch (tons) for January-December over the period 1983-2019 for MICE region 3 (Gilbert). Note there were no catches prior to 1989 for this region and the model only fits to catch data from 1989.....	367
Figure A60. Observed (blue) vs predicted (red) catch (tons) for January-December over the period 1983-2019 for MICE region 5 (Flinders). Note there were no catches prior to 1989 for this region and the model only fits to catch data from 1989.....	368
Figure A61. Observed (blue) vs predicted (red) catch (tons) for January-December over the period 1983-2019 for MICE region 7 (Roper). Note there were no catch data available for 2019 in this region and hence the model does not fit to 2019.....	369
Figure A62. Annual modelled commercial biomass (tons; blue line) of mud crab (i.e. biomass available to be caught by the fishery) and annual standardised river flow (grey bars) for selected regions in the Queensland GoC.....	372
Figure A63. Annual modelled commercial biomass (tons; blue line) of mud crab (i.e. biomass available to be caught by the fishery) and annual standardised river flow (grey bars) for Region 7 in the Northern Territory GoC. Also shown is the annual commercial mud crab biomass (blue line) alongside mean Nov-Dec air temperature (orange line).....	373
Figure A64. Republished from Figure 6 (Jerry et al. 2013) genetically distinguishable subpopulations of barramundi based on analysis of 16 microsatellite loci; grey outline is the major freshwater fish bioregions.	376
Figure A65. Summary of the three stocks considered in the MICE, namely the Northern GoC stock (region 1), southern GoC stock (regions 2-6) and the Northern Territory stock (regions 7-8).....	378
Figure A66. Sum of barramundi catches (t) over years 1989 to 2019, taken from each of the MICE regions as shown.	379
Figure A67. Barramundi catch (t) show for each of the Qld model regions for selected years from 2010 to 2019.....	380
Figure A68. Schematic showing structure of barramundi model age classes, sex change and selectivity by the fishery.....	381
Figure A69. Length-at-age relationships used in the MICE for barramundi and the two sawfish species.....	381
Figure A70. Mass at age used in the MICE for barramundi.....	382
Figure A71. Schematic showing length-at-age for barramundi together with the Qld minimum and maximum size limits (indicated by vertical lines) which correspond respectively to 3 and 10 years.....	382
Figure A72. Schematic showing the virgin (pre-exploitation) relative numbers-at-age, split also into males and females, of barramundi assuming natural mortality $M = 0.2$	383

Figure A73. Schematic showing the virgin (pre-exploitation) relative total mass of each age class of barramundi when assuming natural mortality $M = 0.2$ compared with $M = 0.3$.	383
Figure A74. The proportion of males and females of ages as shown that are assumed to be sexually mature in the MICE.	384
Figure A75. The MICE formulation used to represent potential decreases in barramundi fertilisation success once the proportion of males decreases below the equilibrium proportion. The example assumes $M=0.2$ for which the equilibrium proportion is 0.3, but alternative formulations were also investigated.	384
Figure A76. Schematic summary of MICE structure used to represent the largemouth sawfish <i>Pristis pristis</i> .	391
Figure A77. Summary of the relative trends in the numbers of largemouth sawfish interactions with each of the two fishery sectors, trawl and gillnet, in each of the MICE spatial regions. These are hypothetical numbers of interactions used to drive plausible population trajectories for largemouth sawfish through accounting for changes in fishing effort in the two sectors as well as in the different spatial regions (although we don't model largemouth sawfish in model regions 1, 6 and 8). The total interactions are calculated by applying catch rates from Pillans et al. (in press) for the two sectors to actual fishing effort from 1970 to 2019 (prawns) and 1989-2019 (barramundi – gillnet sector), and assuming the 1989 gillnet effort level was constant back to 1970 given the absence of better information.	392
Figure A78. Summary of the relative trends in the numbers of largemouth sawfish assumed to die following interactions with fishing gear (trawl and gillnet combined), in each of the MICE spatial regions (although we don't model largemouth sawfish in model regions 1, 6 and 8), and when using the 3 sensitivity tests with post-release survival after 2000 assumed to be: 0%, 25%, 75%, 90%. As can be seen from the plots, as post-release survival rate is assumed to increase (with effect from year 2000), the incidental mortality (expressed as total annual number of animals which die) decreases in sync and can potentially decrease to very low levels relative to historical levels if post-release survival improves substantially.	398
Figure A79. Plots showing the change in the relative numbers of (A) all sawfish and (B) mature animals for each of the Flinders, Gilbert and Mitchell rivers when assuming different values of post-release survival. Top plot shows base case with PRS=0% compared with 25% and 90% whereas, for ease of discriminating the difference, the lower plot shows the two bounds of post-release survival, namely 0% and 90%.	400
Figure A80. Extract from Duggan et al. (2014) showing measures of changes in meiofauna (nematodes, copepods) and salinity during different seasons and flood conditions in the Norman River.	405
Figure A81. Summary of MICE biomass dynamic equations to model microphytobenthos in each week w of year y in model region a . The growth rate is shown to depend on a number of key physical drivers, including nutrient limitation, salinity, sea surface temperature and solar exposure.	406
Figure A82. Summary of MICE biomass dynamic equations to model meiofauna (principally nematodes and copepods) in each week w of year y in model region a . The growth rate is shown to depend on the microphytobenthos biomass as well as changes in salinity.	406
Figure A83. Preliminary validation of MICE salinity and meiofauna model components through comparison with observations of Duggan et al. (2014) from the Norman River region.	407
Figure A84. Model-estimated microphytobenthos trajectories showing differences between years and model regions.	407
Figure A85. Preliminary validation of MICE microphytobenthos model component through comparison with observations of Duggan et al. (2014) from the Norman River region.	408

Figure A86. Validation of seagrass model component through comparison with available seagrass data from Kenyon et al. (1997a) for the same region (region 7) and time period (November 1987 to July 1989).	412
Figure A87. Comparison of model-estimated seagrass (annual average) in model region 7 for years 1970 to 2019 shown together with average flow residuals for the region.....	412
Figure A88. Model-estimated seagrass (annual average) in model region 7 shown for year 1985 which corresponds to Cyclone Sandy which is indicated by red vertical bar and has intensity set at 4 in the model. The plot also shows the standardised flow residuals for the Roper River in 1985.....	413
Figure A89. Comparison of model-estimated seagrass (combined species and average over weeks 35 to 37) biomass with Karumba (model region 4) September long-term seagrass monitoring from 1994 to 2019.....	413
Figure A90. Example of preliminary validation of mangrove model component through comparison with available mangrove data from Conacher et al. (1996) for the same region and time period.....	416
Figure A91. Example of model weekly mangrove relative biomass in region 7 (Roper River region) over the period 2015 to 2016 shown together with changes in sea level and air temperature (see Appendix 6 for physical variable descriptions and sources).....	417
Figure A92. Example of MICE estimated relative biomass trajectories for seagrass and mangroves in model region 4 during selected year 2009.....	417
Figure A93. Comparison of the observed and model-predicted annual common banana prawn catch (tonnes) estimated using Model version 2 (driven by baseline flows, using different flow relationship parameters) and for each of the eight model regions as shown.....	455
Figure A94. Comparison of the observed and model-predicted annual common banana prawn catch (tonnes) estimated using Model version 3 (driven by baseline flows, with alternative lag between flow and effect on recruitment) and for each of the eight model regions as shown.....	456
Figure A95. Comparison of the observed and model-predicted annual common banana prawn catch (tonnes) estimated using Model version 4 (driven by baseline flows, with model starting values proportional to recent catch proportions) and for each of the eight model regions as shown.....	457
Figure A96. Comparison of the observed and model-predicted annual barramundi catch (tonnes) estimated using Model version 2 (baseline flows, with flow relationship parameter changed) and for each of the eight model regions as shown.....	458
Figure A97. Comparison of the observed and model-predicted annual barramundi catch (tonnes) estimated using Model version 3 (baseline flows, and with large natural mortality M) and for each of the eight model regions as shown.....	459
Figure A98. Comparison of the observed and model-predicted annual barramundi catch (tonnes) estimated using Model version 5 (baseline flows, and with no sex ratio influence on recruitment) and for each of the eight model regions as shown.....	460
Figure A99. Comparison of the observed and model-predicted commercial catch-at-age proportions of barramundi in each age-class as shown estimated using Model version 1 (baseline flows) for the Mitchell River catchment region from 2000 to 2019.....	461
Figure A100. Comparison of the observed and model-predicted commercial catch-at-age proportions of barramundi in each age-class as shown estimated using Model version 1 (baseline flows) for Regions 3-6 from 2000 to 2019.....	462

Figure A101. Comparison of the observed and model-predicted commercial catch-at-age proportions of barramundi in each age-class as shown estimated using Model version 2 (baseline flows, with flow parameter changed) for the Mitchell River catchment region from 2000 to 2019.	463
Figure A102. Comparison of the observed and model-predicted commercial catch-at-age proportions of barramundi in each age-class as shown estimated using Model version 2 (baseline flows, with flow parameter changed) for Regions 3-6 from 2000 to 2019.....	464
Figure A103. Comparison of the observed and model-predicted commercial catch-at-age proportions of barramundi in each age-class as shown estimated using Model version 3 (baseline flows, with larger natural mortality M) for the Mitchell River catchment region from 2000 to 2019.	465
Figure A104. Comparison of the observed and model-predicted commercial catch-at-age proportions of barramundi in each age-class as shown estimated using Model version 3 (baseline flows, with larger natural mortality M) for regions 3-6 from 2000 to 2019.	466
Figure A105. Comparison of the observed and model-predicted commercial catch-at-age proportions of barramundi in each age-class as shown estimated using Model version 5 (baseline flows, with no sex ratio influence on recruitment) for the Mitchell River catchment region from 2000 to 2019.	467
Figure A106. Comparison of the observed and model-predicted commercial catch-at-age proportions of barramundi in each age-class as shown estimated using Model version 5 (baseline flows, with no sex ratio influence on recruitment) for regions 3-6 from 2000 to 2019.....	468
Figure A107. Comparison of the observed and model-predicted annual mud crab catch (tonnes) estimated using Model version 2 (driven by baseline flows with an altered flow relationship parameter, and other environmental predictors e.g. SOI for regions 3-6 and temperature for region 7) and for each of the eight model regions as shown.....	469
Figure A108. Comparison of the observed and model-predicted annual mud crab catch (tonnes) estimated using Model version 3 (driven by baseline flows and other environmental predictors e.g. SOI for regions 3-6 and temperature for region 7, and a larger natural mortality M) and for each of the eight model regions as shown.	470
Figure A109. Comparison of the observed and model-predicted annual mud crab catch (tonnes) estimated using Model version 4 (driven by baseline flows and other environmental predictors e.g. SOI for regions 3-6 and temperature for region 7, and with model starting biomass doubled) and for each of the eight model regions as shown.....	471
Figure A110. Model estimated sawfish (<i>Pristis pristis</i>) interactions (number) with trawl and gillnet fishing gear used in all Model versions for each of the eight model regions as shown.....	472
Figure A111. Comparison of the observed and model-predicted annual barramundi catch (tonnes) estimated using Model version 6 (trophic-linked version) and for each of the eight model regions as shown	474
Figure A112. Comparison of the observed and model-predicted commercial catch-at-age proportions of barramundi in each age-class as shown estimated using Model version 6 (trophic-linked version) for Regions 3-6 from 2000 to 2019.....	475
Figure A113. Comparison of the observed and model-predicted commercial catch-at-age proportions of barramundi in each age-class as shown estimated using Model version 5 (baseline flows, with no sex ratio influence on recruitment) for the Mitchell River catchment region from 2000 to 2019.	476
Figure A114. MICE-predicted changes in seagrass relative biomass under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 2. Note for the	

Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, and WRD1 tracks under WRD4..... 477

Figure A115. MICE-predicted changes in seagrass relative biomass under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 3. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, and WRD1 tracks under WRD4..... 478

Figure A116. MICE-predicted changes in seagrass relative biomass under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 4. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, and WRD1 tracks under WRD4..... 479

Figure A117. MICE-predicted changes in mangrove relative biomass under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 2. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4..... 480

Figure A118. MICE-predicted changes in mangrove relative biomass under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 3. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4..... 481

Figure A119. MICE-predicted changes in mangrove relative biomass under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 4. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4..... 482

Figure A120. MICE-predicted changes in mangrove relative biomass under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 5. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4..... 483

Figure A121. MICE-predicted changes to total annual common banana prawn commercially available biomass (Bcom, t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 3. 484

Figure A122. MICE-predicted changes to total annual common banana prawn commercially available biomass (Bcom, t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 4. 485

Figure A123. MICE-predicted changes to total annual common banana prawn commercially available biomass (Bcom, t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 5. 486

Figure A124. MICE-predicted changes to total annual common banana prawn catch (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 3. 487

Figure A125. MICE-predicted changes to total annual common banana prawn catch (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 4.	488
Figure A126. MICE-predicted changes to total annual common banana prawn catch (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 5.	489
Figure A127. MICE-predicted changes to total annual barramundi spawning biomass (Bsp) (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 1. Note for the Mitchell River, BC trajectory tracks under WRD4 and for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4.	490
Figure A128. MICE-predicted changes to total annual barramundi catch (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 3. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2 tacks under WRD4.	491
Figure A129. MICE-predicted changes to total annual barramundi spawning biomass (Bsp) (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 3. Note for the Mitchell River, BC trajectory tracks under WRD4 and for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4.	492
Figure A130. MICE-predicted changes to total annual barramundi catch (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 4. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2 tacks under WRD4.	493
Figure A131. MICE-predicted changes to total annual barramundi spawning biomass (Bsp) (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 4. Note for the Mitchell River, BC trajectory tracks under WRD4 and for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4.	494
Figure A132. MICE-predicted changes to total annual barramundi catch (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 5. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2 tacks under WRD4.	495
Figure A133. MICE-predicted changes to total annual barramundi spawning biomass (Bsp) (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 5. Note for the Mitchell River, BC trajectory tracks under WRD4 and for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4.	496
Figure A134. MICE-predicted changes to mud crab spawning biomass (Bsp; t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 1. Note for the Mitchell River, the BC and all WRDs track under each other. For the other panels, the BC trajectory tracks under WRD3 and WRD2 tacks under WRD4.	497

Figure A135. MICE-predicted changes to mud crab spawning biomass (Bsp; t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 2. Note for the Mitchell River, the BC and all WRDs track under each other. For the other panels, the BC trajectory tracks under WRD3 and WRD2 tacks under WRD4. 498

Figure A136. MICE-predicted changes to total annual mud crab catch (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 3. Note for the Mitchell River, the BC and all WRDs track under each other. For the other panels, the BC trajectory tracks under WRD3 and WRD2 tacks under WRD4. 499

Figure A137. MICE-predicted changes to total annual mud crab catch (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 3. Note for the Mitchell River, the BC and all WRDs track under each other. For the other panels, the BC trajectory tracks under WRD3 and WRD2 tacks under WRD4. 500

Figure A138. MICE-predicted changes to mud crab spawning biomass (Bsp; t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 4. Note for the Mitchell River, the BC and all WRDs track under each other. For the other panels, the BC trajectory tracks under WRD3 and WRD2 tacks under WRD4. 501

Figure A139. MICE-predicted changes to total annual mud crab catch (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 4. Note for the Mitchell River, the BC and all WRDs track under each other. For the other panels, the BC trajectory tracks under WRD3 and WRD2 tacks under WRD4. 502

Figure A140. MICE-predicted changes to mud crab spawning biomass (Bsp; t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 5. Note for the Mitchell River, the BC and all WRDs track under each other. For the other panels, the BC trajectory tracks under WRD3 and WRD2 tacks under WRD4. 503

Figure A141. MICE-predicted changes to total largemouth sawfish mature numbers (Bsp; relative to 1989 level) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 2. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4. 504

Figure A142. MICE-predicted changes to total largemouth sawfish numbers (relative to 1989 level) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 2. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4. 505

Figure A143. MICE-predicted changes to total largemouth sawfish mature numbers (Bsp; relative to 1989 level) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 3. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4. 506

Figure A144. MICE-predicted changes to total largemouth sawfish numbers (relative to 1989 level) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using

Model version 3. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4.	507
Figure A145. Comparison of MICE fishing power trend assumed for common banana prawns compared with alternative series derived by Zhou et al. (2015).	529
Figure A146. Comparison of MICE-estimated commercial fishing selectivity for barramundi (Qld southern Gulf stock) for past (1989-1996) and recent (1997-current) periods compared with selectivity functions used in the stock assessment by Campbell et al. (2017).....	530
Figure A147. Comparison of the observed and model-predicted annual common banana prawn catch (tonnes) estimated using Model version 7 (driven by baseline flows, using alternative fishing power input series) and for each of the eight model regions as shown.	531
Figure A148. Comparison of the observed and model-predicted annual barramundi catch (tonnes) estimated using Model version 7 (baseline flows, alternative fixed selectivity input series) and for each of the eight model regions as shown.....	533
Figure A149. Comparison of the observed and model-predicted commercial catch-at-age proportions of barramundi in each age-class as shown estimated using Model version 7 (baseline flows, alternative fixed selectivity input series) for regions 3-6 from 2000 to 2019.....	534
Figure A150. Comparison of the observed and model-predicted commercial catch-at-age proportions of barramundi in each age-class as shown estimated using Model version 7 (baseline flows, alternative fixed selectivity input series) for the Mitchell River catchment region from 2000 to 2019.....	535
Figure A151. Example of differences in the relative depletion (relative to 1989 level) of largemouth sawfish in the (A) Gilbert River region and (B) Flinders River region under different structural assumptions used in Ensemble Model version 5 and sensitivity Model version 7 which assumes that recruitment success does not depend on river flows except in exceptional years.	536
Figure A152. Example of modelled difference in relative changes (since 1970) in mangrove biomass in the Roper River region when comparing Model versions 5 and 7.	537
Figure A153. Economic risk assessment for banana prawn fishery using MICE model version 1 showing risk of 1, 2 or 3+ bad catch years under four WRD scenarios relative to baseline.....	562
Figure A154. Economic risk assessment for banana prawn fishery using MICE model version 2 showing risk of 1, 2 or 3+ bad catch years under four WRD scenarios relative to baseline.....	562
Figure A155. Economic risk assessment for banana prawn fishery using MICE model version 3 showing risk of 1, 2 or 3+ bad catch years under four WRD scenarios relative to baseline.....	563
Figure A156. Economic risk assessment for banana prawn fishery using MICE model version 4 showing risk of 1, 2 or 3+ bad catch years under four WRD scenarios relative to baseline.....	563
Figure A157. Economic risk assessment for banana prawn fishery using MICE model version 5 showing risk of 1, 2 or 3+ bad catch years under four WRD scenarios relative to baseline.....	564

Acknowledgments

Funding provided by FRDC and CSIRO, Project number FRDC 2018-079. We wish to thank the following in particular for constructive inputs during the project: Danial Stratford (river flow modelling and scenarios), Peter Rothlisberg (ecological insights into all aspects of prawn modelling and Gulf of Carpentaria (GoC) oceanography), Sean Pascoe (contributions to economics discussions), Tonya van der Velde (insights into prawn ecology), Leo Dutra (contributions to discussions on mangroves), and Cuan Petheram (background information on WRD scenarios). Derek Fulton produced graphics and an animation to assist with project communication. The economic analyses were supported by parallel development work on the NPF bio-economic model funded by CSIRO as part of the CSIRO-AFMA MOU assessment project; with economic parameter estimates pertaining to that project being available as part of the original 2018 stock assessment.

We are grateful for inputs from all workshop attendees at the four project workshops in which the ideas, discussions and suggestions were input into the project results. We are particularly grateful to the following people who travelled to attend the first workshop in person to contribute broader perspectives - Zoe Williams of Northern Gulf Resource Management Group, John Clark of Kowanyama Aboriginal Land and Natural Resource Management Office and Stephen Eves (AFMA). Thanks also to the NPF fishery stakeholders who attended in person and also contributed to the first workshop – Ian Boot from Austfish, Phil Robson from Raptis, and Bryan van Wyk and Andy Prendergast from Austral Fisheries.

We are very grateful to the Griffith University providing the coastal environmental productivity data (Microphytobenthos and meiofauna data). Thanks also to Andrew Broadley who attended and presented at the first project workshop. The economics team benefitted from discussions with Jim Smart and are grateful for the opportunity to compare model results and assumptions

We would also like to extend our gratitude to the A. Raptis and Sons' who provided us privilege to access the wharf at their fishing fleet support facility in Karumba on the Norman River Estuary, south east GoC for collecting the Karumba logger data.

Thanks also must also go to NPMI and AFMA for granting us using the NPF commercial data.

We also wish to thank Department of Agriculture and Fisheries (Qld) and Department of Industry, Tourism and Trade (NT) for kind provision of the Barramundi and Mud crab commercial data.

We thank the following reviewers who provided constructive feedback on an earlier version of this report: Jacob Rogers (mathematical equations), Toby Patterson (sawfish), Rodrigo Bustamante and Alistair Hobday.

Finally, we are grateful that our project was able to build on considerable previous research conducted over several decades by a range of researchers and institutions, and most recently conducted as part of the following projects: FGARA, FRDC, NAWRA, NESP, RoWRA projects – see Glossary.

Summary of findings & recommendations

Our Gulf of Carpentaria MICE (Model of Intermediate Complexity for Ecosystem Assessments) links river flows with estuarine and marine systems and is the first integrated framework for quantifying the impacts of water resource development (WRD) across a range of ecological assets, catchment systems, scales and parameter settings. While there is scope to build on and refine this framework, we are nonetheless able to provide some considerations and recommendations with respect to water resource development implementation:

Key findings

- Changes from baseline flows due to WRDs had variable impacts on all species and catchment regions, with impacts ranging from minor through to extreme impacts under some scenarios.
- Our risk assessment classified the highest water allocation and multi-catchment WRD (WRD1) as the highest risk development, with moderate to intolerable risks predicted for all species and groups except for seagrass, and both in terms of population-level risk and fishery risk, followed by WRD2 and WRD4 (lower water allocation or single catchment WRD), both of which also predicted high risks to some populations and fisheries. WRD3 was assessed less risky because it assumes no development on the Flinders and Gilbert River catchments.
- Largetooth sawfish were predicted to show the greatest sensitivity to WRDs (due to their low productivity life-history characteristics) with risks ranked as intolerable across a broad range of alternative water extraction or impoundment scenarios.
- For common banana prawns, the Flinders River catchment emerged as the most sensitive to WRDs, consistent with previous findings from estuarine productivity studies (Burford and Faggotter 2021).
- We quantified the probability of alternative WRDs increasing the baseline economic risks to the common banana prawn sub-fishery of the Northern Prawn Fishery by computing the relative probability of occurrence of major risks (defined as risk of a bad or “unacceptable” year), severe risk (two successive bad years) and intolerable risk (fishery operations becoming unviable due to three or more consecutive bad years). We found that the risk of a bad year may more than double under some WRD scenarios.
- The Gilbert River catchment emerged as the riskiest scenario overall for barramundi abundance and catches.
- For mud crabs, the Flinders and Gilbert River catchments emerged as most vulnerable to WRDs with risks often as high as the severe risk category.
- The MICE predicted major to severe risks to mangrove habitats under some WRD scenarios, but the mangrove sub-model is more uncertain than the other groups as suitable data for validation were not available. The MICE predicted negligible risks to seagrass, although we did not account for potential increases in nutrient levels and turbidity that may be associated with WRDs.

Considerations and recommendations

- Timing of flows: Our model implicitly accounts for the fact that the timing of flows (and early wet season flows in particular) is critical for most species, and hence model results to evaluate the impact of alternative WRDs include integration of changes in intra-annual flow levels evaluated using either a weekly or monthly time step.

- Quantity of water allocated for extraction/impoundment: the amount of water either extracted or impounded will be important and should maintain a minimum ecosystem requirement. The accounting of water allocation may depend on the acceptable decline in species catch/biomass (or risk) that stakeholders are willing to accept. Guidance will be needed around water management that maintains estuarine functions which are critical to support high-value fisheries and other species. Water management targets should be amended to account for sustainability of economically valuable species.
- Number of WRDs: given there was major to substantial risk for all catchments tested, the number of catchments (or WRDs per catchment) should be considered in future planning. We found that WRD scenarios had significant impacts on the GoC ecosystem and in some cases a single WRD per catchment posed a high risk to most dependent species in the GoC.
- Types and settings of WRDs: Our MICE showed that types of WRD were important but vary depending on species. For example, extraction WRDs had greater impact on abundance of largemouth sawfish than impoundment scenarios (i.e. dams), whereas the case was opposite for barramundi, and both were equally impactful for mud crabs. The model also shed light on recommended settings for future WRDs if these were to be implemented. Specifically, we recommend an extraction duration as short as possible (i.e., a high pump rate) with a high river-flow level extraction threshold.
- Differences in catchments: the MICE corroborated that not all catchments have the same characteristics and species are likely to respond differently depending on the region. Overall, WRD on the Flinders and Gilbert rivers were predicted to pose the highest risk, although Mitchell WRDs also had a high level of risk for some species.
- Climate change: management of water extractions should also be considered in the context of changing climate, which has not been tested.
- Data and knowledge gaps: our study underscores the need for more empirical data, with key gaps highlighted including data on the natural mortality, status and connectivity of sawfish as is being addressed through planned close kin mark-recapture studies (Bravington et al. 2016). Ground-truthing data are also required to validate findings relating to mangrove and seagrass habitats. Extensions to our study, including adding additional species and fishery sectors (Indigenous, recreational) was constrained by the lack of suitable data. Our study was unable to address the critical need to involve Australian Indigenous peoples and include their values in water planning (Lyons and Barber 2021).
- Wider ecosystem components: the MICE has focussed on key species in the GoC system that are of economic or conservation importance and for which data were available. There are many other components of the ecosystem that we have not considered, including freshwater biota and many trophic links. We did not represent secondary impacts from WRDs – such as potential increases in nutrient and sediment loads and associated turbidity. Increased sediment loads could pose multiple problems, including light attenuation associated with turbidity but also the fine sediments associated with agricultural development may be an additional risk to estuarine biota through smothering of gills, reduced viable habitat area, etc. Thus, risk to the ecosystem could be greater than predicted and in ways other than modelled by the MICE.

Executive Summary

Over the last decade, Commonwealth and State Governments have developed plans and undertaken research (FGARA, FRDC, NAWRA, NESP, RoWRA projects – see Glossary) to develop the water resources in some of Australia's large-catchment northern rivers that feed into the Gulf of Carpentaria (GoC). Water extraction to support agriculture will modify the mostly natural flow regimes that currently support estuarine and coastal fisheries. River flow is crucial in the life cycle of prawns that support the Northern Prawn Fishery (NPF), as well as iconic tropical species (e.g. mud crab, barramundi, grunter, and threadfin salmons) of importance to commercial, recreational and Indigenous fisheries, and species with high conservation (e.g. sawfish) and cultural value.

CSIRO in conjunction with colleagues from NPF Industry Pty Ltd (NPFII), Griffith University and Queensland Department of Agriculture and Fisheries, have completed a modelling study to quantify the impacts and risks to the GoC ecosystem of water resource developments (WRD). A novel MICE (Models of Intermediate Complexity for Ecosystem assessments) ecosystem modelling approach (Plagányi et al. 2014) was applied to the Mitchell, Flinders and Gilbert catchments of northern Australia in particular. Our approach is the first to dynamically link changes in end of system flows and other physical drivers influencing common banana prawns, barramundi, mud crabs, largemouth sawfish as well as habitat-forming mangrove and seagrass communities, to quantify local and regional population and fishery risks to underpin management decisions.

Background

Across northern Australia, tropical rivers and their catchments are largely undeveloped and their monsoon-dependent ecology historically intact. Proposed water resource development for agriculture will extract or impound water away from natural flow regimes, and the marine ecological and economic impacts thereof are currently unknown within the wet/dry tropics. Hence, research is needed to understand the trade-offs and support decision making related to alternative strategies for managing water resources effectively for both agriculture and marine production and biodiversity conservation. To date, most modelling work in these northern catchments has focused on understanding the hydrological drivers rather than dynamic modelling of broader ecological aspects of water resource development when managers have made decisions about water allocation (and particularly how to quantify aspects such as the minimum water requirements for ecological components). Hence, there is a need to quantify WRD, both for consideration by affected commercial, recreational, Indigenous and other sectors, as well as to provide water resource managers with quantitative estimates such as the minimum water requirements to maintain ecosystem structure and functioning. In addition, a GoC-wide regional approach is required as there is often connectivity across catchments, catchment characteristics vary regionally and targeted outcomes will improve decision making over an area as extensive as multiple GoC catchments. To date, all water resource planning has failed to take a regional approach and consider the cumulative impacts on ecological assets that are often linked across catchments. This study provides rigorous data-driven outcomes to inform decision-making as to the potential impacts and risks from alternate WRD, including types of water development (extraction or impoundment), levels of water allocation and catchment development combinations inherent to WRD scenarios, and strategies to mitigate impact on species production and ecology from WRDs.

Aims/objectives

There remain considerable gaps in quantification of the trade-offs associated with proposed water resource development and disruption of ecosystem services within GoC estuaries. The current research aims to support decision making related to alternative strategies for managing water resources effectively for both new agriculture and current marine production and biodiversity conservation. Historical biological research was species focussed (prawns, predation, crabs, seagrass, mangroves) and regionally disparate (each river studied separately). Considerable previous effort has focused on the functioning of ecosystem

drivers in tropical estuaries but there remains a need for an integrated dynamic model that incorporates all available data and focuses on key aspects to inform on the complex non-linear functioning of the system.

The aim of this study was to develop a rigorous, data-driven spatial MICE of the GoC system that integrates existing data and understanding, and in consultation with stakeholders, to quantify the impacts on key marine species of alternative WRD scenarios. Using a dynamic ecosystem model, the project aimed to produce quantitative estimates of the impact of alternative flow regimes on the relative abundance of key fishery and other marine species in the GoC, as well as impacts on total fishery catches and value. Given the implications of this research for a broad range of stakeholders and managers, we also aimed to share the findings and engage in other relevant broader management fora.

Methodology

The project relied on extensive stakeholder consultation for input to the modelling, and review of model outputs. Although the project commenced pre-COVID-19, most of the project has overlapped with the peak COVID-19 period, which therefore impacted on aspects of project planning and resulted in some replanning and delays. The project included a number of workshops and teleconferences.

Principally, the project involved developing a spatial, dynamic MICE including key GoC ecosystem species and processes. The GoC MICE uses as inputs flow scenarios initially developed by the NAWRA project's river end-of-system flow modelling, with these models extended, updated and additional scenarios added specific to this project. For common banana prawns, mud crabs and barramundi, the model is formally statistically fitted to available data such as fisheries catch, effort and catch-at-age data, using methods similar to that used in stock assessments and implemented in AD Model Builder. The GoC MICE includes eight spatial regions, plus consideration of differences between freshwater, estuarine and marine environments. The project has also included collation and analysis of all available physical variables considered important drivers of system dynamics (spatially resolved to the extent possible). In addition to river flow, these include sea surface temperature, air temperature, salinity, sea level height and cyclones. We reviewed all tropical cyclones recorded in the Gulf of Carpentaria from 1970-2019, including their estimated intensity (low or cyclone category 1-5) and assignment to MICE model region.

Model components of the GoC MICE were built in a stepwise fashion, only adding additional complexity if needed (e.g. if statistically able to significantly better explain the available data) and if based on underlying hypotheses. For example, first we assessed the extent to which fishery effort data could explain observed changes in the spatially-resolved population trends of a species, before adding any hypothesised links with baseline river flow to try and improve model fits. Given there is uncertainty in the model structure and parameterisation, we used an ensemble modelling approach (as used in a number of ecosystem and climate modelling studies), with an ensemble comprised of five alternative GoC MICE versions to integrate project results over.

The MICE quantified changes in abundance and catch of the key species under the different WRDs for individual catchments (Gilbert, Mitchell and Flinders Rivers) as well as at a regional scale (south-eastern GoC, model regions 2-6). Overall, we tested 19 WRDs but focus on four key scenarios that simultaneously explore impacts on all three rivers being considered for future water resource developments. The four key WRDs range from high extraction scenarios on the Mitchell and Flinders rivers and two dams on the Gilbert River (WRD1) through to a moderate scenario for all three rivers (WRD2), a scenario with moderate extraction from the Mitchell and no WRDs on the Flinders and Gilbert (WRD3), and finally a lower impact scenario (WRD4 – no WRD on Mitchell River and moderate extraction on Flinders combined with a single dam on the Gilbert River). The other scenarios tested (WRD5-19) include alternative combinations, scenarios and technical specifications applied to the different rivers.

Finally, we summarised our findings in terms of risk to the respective fisheries and populations in these regions, including economic risk for the common banana prawn fishery, and provided suggestions on ways to mitigate impacts from WRD.

Baseline Model Results

Our results substantially advance understanding and quantification of relationships between key species and river flows because (1) we use a dynamic framework rather than static correlations, (2) our results are resolved for different spatial regions, (3) our results enable detailed investigation of fine-scale patterns and seasonal changes in flows because we use a weekly time-step (for prawns, mangroves, seagrass) and monthly time-step (for mud crabs, barramundi, sawfish), (4) we use an integrated framework when fitting to data which means that the model is able to simultaneously account for changes due to biological growth, trophic interactions, physical processes and river flow (both magnitude and intra-annual patterns), and (5) we account for uncertainties, thereby building confidence in the ability to correctly attribute the reasons for ecosystem changes. The GoC MICE therefore provides the most rigorous platform to date for use to simulate alternative plausible scenarios to support proactive planning and mitigation.

For the fished species, which were fitted to weekly or monthly historical catch data since 1970 (or 1989 as available), we found that linking physical drivers such as flows with population and fishery dynamics significantly improved the model's ability to estimate catches. The fitted relationships between flow and population dynamics and fishery catches were then used to predict plausible responses to altered flow scenarios.

For common banana prawns, each of the six major rivers that is explicitly represented in the MICE (i.e. including the Embley River which influences Region 1 and the Roper River which influences Region 7) is influential with the relative role and contributions of each river varying by region and by year. The combined portfolio of rivers thus acts to 'stabilise' or maintain the common banana population across the entire GoC, and reducing flows from one or more rivers will have non-linear effects on common banana prawns and other dependent species. Our study quantifies ways in which different river systems are important in different years and how the combination of these flow anomalies across the different systems ultimately determines the productivity of the GoC and catch that is available to be caught.

For barramundi, our model highlighted the challenges of linking flow relationships in an integrated model and representing a longer-lived species (>10 years) with a complex life history (including changing sex from male to female) and different management rules for the two legislative jurisdictions. Our modelling of mud crabs advances previous approaches and highlights their sensitivity to river flows, regional differences and corroborates the sometimes key role of other physical drivers influencing local populations. River flow improved model fit for most regions, particularly the Gilbert, Norman and Flinders catchments where a strong precise relationship with flow was estimated.

Our large-scale representation of largemouth sawfish is the first attempt to integrate the complex life history and past dynamics of this threatened euryhaline chondrichthyan and to quantify the relative impact of alternative WRDs on sawfish recruitment and survival. The results of the base case model ensemble, while highly uncertain due to a lack of historical data, uniformly support the notion of potential for large declines of sawfish in all catchments relative to historical levels. They also highlight the lack of information and the urgent need for improved information on the species abundance and life history.

GoC mangroves and seagrass are vital blue carbon assets and our model is the first large-scale attempt to model temporal changes in these key habitats (supporting fish, crustaceans, chondrichthyans and other species) in response to changes in river flows, light levels, cyclone impacts and other physical drivers. Although we had very limited data available at the scale required to model mangroves and seagrass, we based our model on available information and used a range of parameter settings and assumptions for these groups, drawing on previous observations of changes in mangrove and seagrass cover in response to changes in river flows and other factors.

Results under water resource development

Changes from baseline flows due to WRDs had variable impacts on all species and catchment regions, with impacts ranging from minor through to extreme impacts under some scenarios. Overall, we found that model-predicted catchment-system impacts increased with the greater volume of water extracted or impounded and number of rivers on which dams or WRD scenarios were deployed. Across all of the modelled species, water extraction (i.e. pumping) at a low river-flow threshold value caused a substantial negative impact on model-predicted catches and abundance compared with pumped extraction confined to higher river-flow levels; a result consistent with previous studies pointing to the need to maintain flows well above an ecosystem-minimum.

Common banana prawns

For common banana prawns, the MICE quantified a river portfolio effect across the Mitchell, Gilbert, Norman and Flinders Rivers, such that WRDs applied to a single river or different combinations of rivers had complex cumulative and synergistic effects on common banana prawn abundance and catches throughout this sub-region. This may be related to the offshore migration of common banana prawns from multiple estuaries. Simultaneous WRD across multiple catchments negatively affected common banana prawn populations from a moderate to a major degree. Lower water allocations lessened the impact, while WRD within a subset of catchments had the least impact. A significant decline in catch was associated with water extraction at low-levels of river flow, pointing to the need to maintain flows well above an ecosystem-minimum level. Our findings corroborate previous research exploring changes in freshwater inputs on the primary productivity of the same river estuaries (Burford and Faggotter 2021), underscoring the longer-term benefits to system productivity that result from ongoing productivity boosts driven by natural flow regimes – an effect we term the flood productivity boost effect. Our model predicted substantially greater local and regional decreases in common banana prawn catch and abundance if accounting for the flood productivity boost effect.

Barramundi

Barramundi were generally predicted to be most sensitive to the WRDs applied to the Gilbert River region, with both a single and two dams predicted to cause large declines to the local populations. Overall, the model ensemble suggested average catch would decrease up to about 20%, with a maximum decrease of around 27%, under WRDs 1 and 3.

Mud crabs

The MICE predicted substantial declines in mud crab abundance and catch for the Flinders and Gilbert catchments under both medium and high-impact WRDs (WRD2 and WRD1 respectively) and this result was consistent across all five model versions in the MICE ensemble. Almost no change was predicted for the Mitchell catchment, likely since the model estimated a very weak relationship with river flow for this region. Changes in catch under WRD for the Gilbert and Flinders catchment were similar to the magnitude of change observed in historical catches between wet, intermediate and dry years.

Large-tooth sawfish

Large-tooth sawfish showed high sensitivity and population impacts from almost all WRDs tested. Large-volumes of water for extraction and flow modification within multiple catchments (e.g. WRD1 and WRD2) was predicted to result in extremely large local population declines. Sawfish model results differed in a number of ways from those for prawns, barramundi and mud crabs as changes to WRD settings did not result in major differences in predicted impacts on sawfish (i.e. river flow extraction threshold value and pump rate). The result for sawfish contrasted with the case for the other species. For sawfish, anything other than very low water extractions were predicted to have a substantial negative impact on their population. The largest predicted population declines of sawfish corresponded to WRD scenarios for the Flinders River as this sub-population appeared to be the least resilient across a number of alternative scenarios.

Sawfish results suggested greater sensitivity to WRD scenarios involving water extraction compared with water impoundment i.e. dams (assuming free movement of the animals wasn't negatively impacted) with any scenario other than very low extraction volumes predicted to have substantial negative impacts on sawfish, across a range of alternative water extraction threshold and pump rate settings.

Our preliminary investigations for largemouth sawfish suggested that sawfish abundance in all catchments were highly sensitive to WRDs but also provides guidance as to which river systems are likely to be most sensitive to flow modification (e.g. Flinders River) and the type of impacts (extraction WRDs performed worse than impoundment scenarios (i.e. dams)). In addition, the models shed light on recommended settings for future WRDs if these were to be implemented.

Model sensitivities – trophic links

Model results were fairly robust to alternative model structures that included explicit representation of the dependence of barramundi and largemouth sawfish on estuarine prey availability (using common banana prawns as a proxy) albeit that this could worsen predicted impacts of WRDs in some scenarios.

Mangroves

Model results suggest that WRDs may have a dramatic effect on mangroves with water impoundment (dams) predicted to result in large declines in mangrove abundance in affected areas. This significant finding requires on-ground validation, but points to the need to consider the potential impacts of WRDs on mangroves. Our mesoscale mangrove community model component was based on research suggesting vegetative cover of mangrove-saltmarsh tidal wetlands increases with increasing average annual rainfall (and hence flow) and that periodic changes in rainfall trends (and hence flow) can result in encroachment or dieback of mangroves. Our results suggested that outcomes for mangroves under the same water extraction could be substantially worse if using a low river flow pump threshold (TH) and longer pump duration compared with a less impactful scenario using a medium river flow pump TH and shorter pump duration (pumping the same water extraction at a faster daily rate). A low river flow threshold allows the pumping of low-level flows whereby a large proportion of river flow is extracted and significantly less water passes downstream. In addition, a longer pump duration necessitates water extraction from flows other than peak flows, resulting in a higher proportion of the non-peak flows being extracted. Both pump routines disturb the pattern of river flow during low-level flows when water extraction disproportionately affects river flows and downstream ecosystem service provision.

Seagrass

In contrast to all the other MICE groups, seagrasses were predicted to marginally increase in abundance/distribution under some WRDs for some time periods, with fairly minor impacts (up to a 7% decline in seagrass abundance relative to base levels) across most scenarios. It is possible that lower flows reduce sediment loads and water turbidity in the nearshore zone, improving light-penetration conditions for seagrass.

Ecological and fisheries risk assessment

Our risk assessment classified WRD1 (highest water allocation and multi-catchment WRD) as the highest risk development, with moderate to intolerable risks predicted for all species and habitat groups except for seagrass, both in terms of population-level risk and fishery risk. This was followed by WRD2 and WRD4 (lower water allocation or single catchment WRD), both of which also predicted high risks to some populations and fisheries. WRD3 emerged as the least risky scenario but it should be noted that this scenario involved no development on the Flinders and Gilbert Rivers, and only WRD development on the Mitchell River. Largemouth sawfish were predicted to show the greatest sensitivity to WRDs (due to their low productivity life-history characteristics) with risks ranked as intolerable across a broad range of alternative water extraction or impoundment scenarios.

For common banana prawns, the Flinders River catchment emerged as the most sensitive to WRDs, consistent with previous findings from estuarine productivity studies (Burford and Faggotter 2021).

We quantified the probability of alternative WRDs increasing the baseline economic risks to the common banana prawn sub-fishery of the NPF by computing the relative probability of occurrence of major risks (defined as risk of a bad year), severe risk (two successive bad years) and intolerable risk (fishery operations becoming unviable due to three or more consecutive bad years). We found that the risk of a bad year may more than double under some WRD scenarios.

The Gilbert River catchment emerged as overall the riskiest scenario for barramundi abundance and catches, followed by scenarios for the Flinders and Mitchell rivers, while for mud crabs, the Flinders and Gilbert River catchments emerged as most vulnerable to WRDs with risks often as high as the severe risk category.

The MICE predicted major to severe risks to mangrove habitats under some WRD scenarios, but negligible risks to seagrass, although we did not account for potential increases in nutrient levels and turbidity that may be associated with WRDs.

Our spatial MICE linking river flows, estuarine and marine systems provides a useful framework for ongoing studies to improve understanding and quantification of predicted impacts of WRDs and climate drivers, and is readily extended to represent WRDs applied to other catchments, acknowledging that there remains considerable scope for improving our modelling approach as new information becomes available.

Implications for relevant stakeholders and future research

This project has a broad range of implications for industry, communities, resource managers and policy makers. Firstly, the study has enabled collation and integration of a vast amount of information (from previous and ongoing studies, plus consultation with experts), data (including physical variables from a range of sources and fisheries data from several different fisheries as well as three different jurisdictions) and has collated and analysed available information pertaining to potential WRD scenarios that may impact the GoC. Secondly, the MICE has integrated this available information and data, focussing on key aspects to keep it tractable, and hence is the first ecosystem model specifically tailored to serve as a reliable tool for answering complex multi-sector management questions (based on the best available current knowledge). Thirdly, the river modelling improvements and additions undertaken as part of this project are a valuable resource not only for this project but for other projects to quantify the end-of-system flow impacts of a range of potential WRD scenarios. Fourthly, the MICE has included detailed spatial modelling of several commercial species and a species of conservation concern, with such models significantly advancing the available toolbox of approaches to inform understanding and management – for example:

- (1) the common banana prawn component is the first fully dynamic spatially-resolved model with population dynamics simultaneously driven by a combination of river catchments;
- (2) the mud crab component is the first spatially-resolved age-structured model that simultaneously captures different management practices in Qld and NT and the impacts of physical variables on these stocks and will therefore be a useful resource for ongoing management of mud crabs both in GoC and elsewhere;
- (3) the spatially-resolved age-structured barramundi model linked to flows is useful to complement other modelling initiatives and represent both Qld and NT stocks;
- (4) the sawfish component is the first attempt to integrate the complex life history and past dynamics of this threatened euryhaline chondrichthyan and link river flow to sawfish recruitment and survival. The model can be refined over time as more data and information become available to assist with conservation efforts;
- (5) large-scale representation of GoC mangroves and seagrass linked with fishery species is the first such attempt to model changes in these key habitats– simultaneously vital blue carbon assets – in response to changes in river flows, light levels, cyclone impacts and other physical drivers;
- (6) the model simultaneously integrates a range of complex system connections and interactions in a single dynamic framework. The MICE includes representation of species of commercial and cultural importance – for example mud crabs and barramundi – and therefore provides insights for local communities as to potential future changes in the environment and dependent species; and
- Lastly but not least, the integrated results of the project may be used to inform trade-off decisions related to WRD scenarios or changes to managed outcomes from them.

It was beyond the scope of this project to refine some of the model components as much as could be achieved, and it is hoped that there will be future opportunities to continue to build on this work. For example, there is scope to gradually increase the complexity of the modelled systems such as to account for a range of inter-specific interactions, connectivity scenarios, other fishery sectors (e.g. recreational). Given the model has been specified, less resources would be required to re-run a series of alternative WRD scenarios in a future project. In particular, the MICE is an ideal framework for exploring climate change impacts and adaptation scenarios.

The MICE will also be useful for other future development scenarios that need to be tested, as well as alternative mitigation options. A separate project is currently underway to investigate impacts of WRD scenarios on the Roper River, and as this region is included in the MICE, it could also be used to quantify these impacts from an ecological perspective. An advantage of the MICE framework is that it can quantify cumulative impacts on the system. There is the potential to extend the MICE to include the entire NPF managed area, or greater.

Considerations and Recommendations

Our GoC MICE is the first integrated framework for quantifying the impacts of WRDs across a range of species, catchment systems, scales and parameter settings. Whereas there is scope to build on and refine this framework, we are nonetheless able to provide some considerations and recommendations with respect to water resource development implementation:

- **Timing of flows:** Our model implicitly accounts for the fact that the timing of flows (and early wet season flows in particular) is critical for most species, and hence model results to evaluate the impact of alternative WRDs include integration of changes in intra-annual flow levels evaluated using either a weekly or monthly time step.
- **Quantity of water allocated for extraction/impoundment:** the amount of water either extracted or impounded will be important and should maintain a minimum ecosystem requirement. The accounting of water allocation may depend on the acceptable decline in species catch/biomass (or risk) that stakeholders are willing to accept. Guidance will be needed around water management that maintains estuarine functions which are critical to support high-value fisheries and other species. Water management targets should be amended to account for sustainability of economically valuable species.
- **Number of WRDs:** given there was major to substantial risk for all catchments tested, the number of catchments (or WRDs per catchment) should be considered in future planning. We found that WRD scenarios had significant impacts on the GoC ecosystem: one WRD per catchment posed a high risk to most dependent species in the GoC.
- **Types and settings of WRDs:** Our MICE showed that types of WRD were important, but vary depending on species. For example, extraction WRDs had greater impact on abundance than impoundment scenarios (i.e. dams) for sawfish, whereas the case was opposite for barramundi, and both were equally impactful for mud crabs. The model also shed light on recommended settings for future WRDs if these were to be implemented. Specifically, we recommend an extraction duration as short as possible (i.e. a high pump rate) with a high river-flow level extraction threshold.
- **Differences in catchments:** the MICE corroborated that not all catchments have the same characteristics and species are likely to respond differently depending on the region. Overall, WRD on the Flinders and Gilbert rivers were predicted to pose the highest risk, although Mitchell WRDs also had a high level of risk for some species.
- **Climate change:** management of water extractions should also be considered in the context of changing climate, which has not been tested.
- **Data and knowledge gaps:** our study underscores the need for more empirical data, with key gaps highlighted including data on the natural mortality, status and connectivity of sawfish as is being addressed through planned close kin mark-recapture studies (Bravington et al. 2016, Bradford et al. 2018, Bruce et al. 2018, Hillary et al. 2018). Ground-truthing data are also required to validate findings relating to mangrove and seagrass habitats. Extensions to our study, including adding additional species and fishery sectors (Indigenous, recreational) was constrained by the lack of suitable data. Our study

was unable to address the critical need to involve Australian Indigenous peoples and include their values in water planning (Lyons and Barber 2021).

- Wider ecosystem components: the MICE has focussed on key species in the GoC system that are of economic or conservation importance and for which data were available. There are many other components of the ecosystem that we have not considered, including freshwater biota and many trophic links. We did not represent secondary impacts from WRDs – such as potential increases in nutrient and sediment loads and associated turbidity. Increased sediment loads could pose multiple problems, including light attenuation associated with turbidity but also the fine sediments associated with agricultural development may be an additional risk to estuarine biota through smothering of gills, reduced viable habitat area etc. Thus, risk to the ecosystem could be greater than predicted and in ways other than modelled by the MICE.

Keywords

Common (or white) banana prawns *Penaeus merguensis*; barramundi *Lates calcarifer*; giant mud crab *Scylla serrata*; largemouth sawfish *Pristis pristis*; mangroves; seagrass; catchments; cyclones; ecosystem modelling; estuarine species; fisheries modelling; marine fisheries; MICE; microphytobenthos; meiofauna; river modelling; water management



Image: Graphics and animation by Derek Fulton

Introduction

River flow is crucial in the life cycle of prawns that support the Northern Prawn Fishery (NPF), as well as iconic tropical species (e.g. giant mud crab, barramundi, grunter, and threadfin salmon) of importance to commercial, recreational and Indigenous fisheries, and species with high conservation (e.g. sawfish) and cultural value. Substantial interest in developing irrigated agriculture across northern Australia is reviewed in a recent FRDC report (Kenyon et al. 2018) and updates are provided in this report. Briefly, in northern Queensland, specific enterprises currently in-development (e.g. environmental assessment) or completed include the Three Rivers Irrigation Project (Flinders River), the Metro Mining (Skardon River) and Amrun Mines (west of Embley River, Weipa). Stanbroke Pastoral's 'Three Rivers' proposed cropland irrigation plans to extract ~150,000 ML y⁻¹ from the lower Flinders River by diversion structure or weir. It has been designated a 'Project of State Significance'.

The Northern Australia Water Resource Assessment (NAWRA) project (Petheram et al. 2018a) assessed the potential of the Mitchell River catchment to support irrigated agriculture. Four cost-effective dams in the Mitchell River catchment have been identified as capable of storing 2800 GL of water in 85% of years for irrigation use. As well, the Mitchell River catchment assessment was scoped possible aquaculture development associated with irrigation infrastructure or coastal water resources.

In the Northern Territory, research has also been investigating the landscape suitability for irrigated agriculture in seven tropical river catchments to the west and east of Darwin in the Top End. The Reynolds, Finnis, Blackmore, Adelaide, McKinlay, Mary, Wildman, and West Alligator rivers and their catchments empty into the Van Dieman Gulf and the Arafura Sea. The topography, soil, water, current infrastructure and cultural dimensions of these catchments have been quantified.

The NAWRA project (Petheram et al. 2018a) summarises water resource development (WRD) options which include surface water and groundwater options available in these catchments and landscapes. Surface water in-stream dams in the Finnis River and Adelaide River catchments could provide 436 GL water in 95% of years. Groundwater resources in the Darwin Rural Water Control District currently provide 25 GL of water with new groundwater resources further from Darwin able to provide another 35GL of water annually. Offstream dams in the Adelaide River catchment (Margaret River) and within the Mary River catchment (McKinlay and Mary rivers) are capable of providing 600 ML of water.

Water resource development to support agriculture will modify natural river flow regimes that support estuarine and coastal fisheries. The ecosystem-wide population, fisheries and economic trade-offs associated with proposed water resource development are currently unknown and research is needed to support decision making related to alternative strategies for managing water resources effectively for both agriculture, mining, marine production and biodiversity conservation. Quantifying these trade-offs entails evaluating how altered river flows might affect the fishery and ecological values. Most modelling work to date has focused on understanding the hydrological drivers rather than dynamic modelling of broader ecological aspects (and particularly how to quantify aspects such as the minimum water requirements for ecological components) as managers otherwise need to make decisions without sufficient research and under limited timeframes. Although previous and recent projects such as NAWRA (Pollino et al. 2018) have evaluated the qualitative impacts of changes in river flows on ecological assets, there is a need to quantify impacts both for consideration by affected commercial, recreational, indigenous and other sectors, as well as to provide water resource managers with quantitative estimates such as the minimum water requirements to maintain ecosystem structure and functioning. Such analyses are also complicated by the fact that each catchment is different, and hence models and the associated recommendations need to be tailored specifically to each catchment area, and there is currently no suitable ecosystem model at the appropriate scale incorporating key relationships.

This project aims to develop a "Models of Intermediate Complexity for Ecosystem assessment" (MICE) as a multi-species assessment tool (Plagányi et al. 2014). Context- and question-driven, MICE focus only on ecosystem components required to quantify specific impacts (and potentially mitigation and alternative solutions) on the ecosystem and on stakeholders of alternative water resource development scenarios.

Stakeholder participation and dialogue is an integral part of this process. MICE estimate parameters through fitting to data, use statistical diagnostic tools to evaluate model performance and account for a broad range of uncertainties. These models therefore address many of the impediments to greater use of ecosystem models in strategic and particularly tactical decision-making for marine resource management and conservation (Plagányi 2007, Morello et al. 2014, Plagányi et al. 2014, Collie et al. 2016, Tulloch et al. 2018).

The MICE focuses on prawns, barramundi and other species for which there are sufficient data, as well as species of conservation concern e.g. sawfish. It has the capability of incorporating river flow information in population models, at fine temporal scales (such as monthly or weekly) and spatial scales (e.g. the GoC can be broken down into regions based on river catchments). The MICE will use as inputs river flow scenarios developed by the NAWRA project's river end-of-system flow modelling (Hughes et al. 2017, Hughes et al. 2018a) and draw on an earlier FRDC study to collate information on proposed northern Australia water developments pertinent to the Northern Prawn Fishery (Kenyon et al. 2018). For species such as common banana and tiger prawns, barramundi and mud crabs, the model was formally statistically fitted to available data such as catch data or survey and CPUE data, using methods similar to that used in stock assessments and implemented in AD Model Builder (Fournier et al. 2012). Although the model is not a detailed mechanistic model, it incorporates some mechanistic representation of processes where necessary, and based on the latest available scientific research and data. As there are important differences in system structure and functioning in different regions of the Gulf of Carpentaria (GoC), the model has a spatial structure, whereby it is broken down into different spatial regions selected based on input from stakeholders at the first workshop. Within each region, there is a MICE ensemble which consists of alternative MICE versions and within each MICE version, there are sub-models for key species that capture the population dynamics of the species and associated environmental drivers (see Methods section 1). These key species and relevant drivers to be included in the model were similarly selected based on input from stakeholders. This project will focus on the GoC but could be extended to other regions as part of a future project.

Outputs such as predicted changes in future catches and species abundance under differing river flow regimes have also been used as the basis of analyses of economic impacts.

In the wet/dry tropics of northern Australia, 90% of the annual rainfall occurs from January to March, providing catchment-to-coast drivers that dominate the annual cycles in coastal ecosystems (Warfe et al. 2011). Historical rainfall and prawn catch and, more recent river flow and prawn catch analyses, over multiple-catchment scales have shown a strong relationship between the dimension of annual flood river flows and annual common banana prawn catch (Vance et al. 1985, Staples and Vance 1986). The effect of water development and the associated potential modifications to river flow have also been predicted for common banana prawn catch, based on a Bayesian Modelling approach (Duggan et al. 2019, Broadley et al. 2020). However, studies of the ecosystem services provided within estuarine habitats have shown that prawn growth, mortality due to predation and estuarine productivity all contribute, at the river or tributary scale, to population abundance of juvenile common banana prawns available to respond to the emigration cues provided by flood flows (Vance et al. 1985, Staples and Vance 1986, Vance et al. 1998).

There have been two projects focussed on understanding the underpinnings of common banana prawn production in estuaries. The first was a Commonwealth Government Tropical Rivers and Coastal Knowledge (TRaCK) consortium and FRDC funded project conducted in the Norman River, Gulf of Carpentaria in 2008-2010 (Burford et al. 2010). This project examined how river flows in the Norman River estuary affect prawn production and emigration. It was a two-year study, with a focus on the period October to March each year when postlarval and juvenile common banana prawns are in the estuary. That study measured several parameters that are also useful to inform modelling being undertaken in this project, including prawn densities and size metrics (main estuary and associated tidal creeks), meiofauna species composition and densities (mudflats, food supply for prawns), and water column physico-chemical parameters (temperature, salinity, DO, pH, TSS, total and dissolved nutrients, secchi).

There are two cues for prawns emigrating from the estuary in the wet season – low salinity causing stress, and a lack of food (low salinity removing meiofauna and benthic algae). Three months after wet season

flow, meiofauna and benthic algae had recruited back into estuary (Burford et al. 2010, Duggan et al. 2014). Mortality was high for prawns that did not emigrate out of the estuary (Burford et al. 2010).

Previous work showed that wetting of saltflats by wet season floods released significant loads of nutrients into estuary (Burford et al. 2016). However, concentrations of nutrients did not increase with wet season flow. The end-of-system nutrient loads were calculated for the Norman River (Burford et al. 2012). In the dry season there is net transport of nutrients back into the estuary and the loads were also calculated.

From 2012-2015, the National Environmental Research Programme's (NERP) Northern Australia Hub was funded by the Commonwealth government, followed by the National Environmental Science Program's (NESP) Northern Australian Environment Resources Hub (2015-2021). The NESP Northern Australian Hub supported a new three-year study led by Burford (Griffith University). Parallel funding for Kenyon's (CSIRO) involvement was provided by FRDC. A range of industry, government and local community stakeholders identified the need to examine the effect of water development in the Flinders, Gilbert and Mitchell Rivers on estuarine productivity with flow on effects to prawns. This project therefore built on previous findings in the Norman River estuary (Burford et al. 2020b, Burford et al. 2021, Smart et al. 2021) (see also <https://www.nespnorthern.edu.au/wp-content/uploads/2021/12/Finfish-catch-and-growth-in-the-Gulf-final-report.pdf>). The MICE model prawn and microphytobenthos sub-components (this study) will draw on results produced by the NESP project, for which there were a number of field investigations of all three estuaries in the wet and dry seasons when prawns were present in estuaries.

In conjunction, the NAWRA project modelled the effect on natural river flow regimes of water resource development, and how modified river flow would impact the provision of those ecosystem services to the riverine, estuarine and coastal crustacean and fish community. Under some water resource development, rivers can lose 80% of low-level flows while peak flood flows are impacted much less; 0-40%. The large, monsoon-season floods continue to support key ecosystem processes in tropical estuaries as a ~20% flow reduction delivers the same pulsed-drivers to the estuary as an un-modified flow. In contrast, low-level river flows are often associated with seasonal spawning and juvenile recruitment of key species during October to December, the late-dry season in the monsoon tropics. These flows are fickle, they precede the wet season, and a loss of 80% of their volume and duration is a critical loss to coastal estuaries during a time-window when the fish and crustacean community relies on facultative juvenile habitats.

The NESP project (<https://www.nespnorthern.edu.au/projects/nesp/links-gulf-rivers-coastal-productivity/>) measured the water quality, estuarine productivity, meiofaunal abundance and the juvenile common banana prawn community among three estuaries in the Gulf of Carpentaria with a view to determining the contribution of key estuaries to coastal fisheries and how modification of river flows due to extractive use would impact the sustainability of estuarine processes servicing fishery production. Results from the NESP project (M. Burford) for juvenile common banana prawns demonstrated that the maintenance of late-dry season river flow is crucial to providing a dynamic, brackish estuary that supports juvenile prawn abundance and early emigration of larger-sized cohorts. The cumulative support of early- and wet-season flows produces a strong contribution to later fishery catch.

Indeed, the least-well understood phase for many commercial species is the estuarine phase where the interaction of river flow with the ecosystem services provided by the estuary are critical to the fish and crustacean community. In combination, the NAWRA and NESP projects have shown the interplay between critical losses of seasonal river flows and the seasonal sustainability of productive estuarine habitats that support commercial fish and crustacean populations. The current MICE modelling project will quantify the relationship between flow and the estuarine populations of key fishery species, and hence catch. Collaboration between NAWRA river system modelers, NESP ecologists, CSIRO modelers and GoC regional stakeholders has brought together the latest data and knowledge to support the analyses (Figure 2). It will provide a matrix whereby the population dynamics of estuarine fish and crustaceans, estuarine productivity, together with river flow projections under water resource development scenarios can be modelled. This aims to elucidate water resource management policy that can sustain the ecosystem services in estuarine and nearshore habitats for downstream fisheries, while enabling the harvest of useful quota of water for extractive use.



Figure 1. Participants at the first stakeholder workshop (noting some participants joined by teleconference or only attended on the second day)

Objectives

1. Develop a MICE model that integrates existing data and understanding, and in consultation with stakeholders, to quantify the impacts on key marine species of alternative water extraction scenarios.
2. Produce quantitative estimates of the impact of alternative flow regimes on the relative abundance of key fishery and other marine species in the Gulf of Carpentaria, as well as impacts on total fishery catches and value.
3. Summarise findings in a technical report and non-technical reports to support sharing findings and engaging in other relevant broader management fora.

Method

This study included a number of methods and components as outlined below. As summarised in Figure 2, this project builds on considerable previous research. The various sub-components are briefly described in this section, with further detail and technical descriptions presented in accompanying Appendices to assist with readability of this report.

The various methods are outlined under the following sections:

1. Project Plan and Scope
2. Gulf of Carpentaria MICE model overview
3. Water Resource Development overview and modelling of river flows in the Gulf of Carpentaria
4. Biological data collation: Qld and NT
5. Physical data collation: Qld and NT
6. Stakeholder workshops:
 - a. Stakeholder scoping workshop 1
 - b. Stakeholder workshop 2: prawn focus
 - c. Stakeholder workshop 3: mud crab focus
 - d. Stakeholder workshop 4: barramundi focus
 - e. Stakeholder workshop 5: Final stakeholder workshop to present preliminary results
7. MICE sub-models
 - a. Overview of model species
 - b. Species modelled using a weekly time-step
 - c. Species modelled using a monthly time-step
 - d. Modelling of habitat and base productivity groups
 - e. Seagrass modelling
 - f. Mangrove modelling
 - g. Estuarine microphytobenthos and meiofauna and habitat use by juvenile common banana prawns
 - h. Estimating catch as a function of river flow
 - i. Linking flow to biological components
8. Bio-economic Modelling
9. MICE Modelling Process

10. Model Fitting
11. Ensemble of Alternative Models
12. Sensitivity Testing Trophic Links
13. Modelling Water Resource Development (WRD) Scenarios
14. Biological and Fishery Risk
15. Economic Risk

1. Project Plan and Scope

An overview of the project plan is shown in Figure 2. This includes as a starting point consideration of all previous related research, including published scientific studies (Robins et al. 2006, Robins and Ye 2007, Russell et al. 2015, Burford et al. 2016, Crook et al. 2017, Duggan et al. 2019), technical reports (e.g. Bayliss et al. 2014, Buckworth et al. 2014, Kenyon et al. 2018) and the recent suite of NAWRA reports (Petheram et al. 2018a, Pollino et al. 2018). Further project scoping and planning was done at Workshop 1 in consultation with stakeholders as a basis for commencing development of a quantitative MICE model. The model outputs will make explicit trade-offs between different scenarios, and these will be reviewed in consultation with stakeholders at a workshop planned for August 2021.

The project sought to build on the recent NAWRA project as well as related research. Figure 3 shows the scope of the current study which is more narrowly focused on the Gulf of Carpentaria (GoC) only and will focus on plausible water resource development (WRD) scenarios and quantify the impact of alternative scenarios on key marine resources. Previous studies have mostly focused on qualitative assessment of WRD impacts on marine assets. We used available ecological and fishery information and stakeholder input to first come up with a conceptual model on how the system works, which we then developed into a qualitative model that described the system and expected changes under WRD. After further consultation with stakeholders and model refinement, the qualitative model was then developed into a quantitative MICE model, with sub-models in each region representing each of the key species or groups (Figure 3). Each sub-model captures the population dynamics of a species/group (e.g. common banana prawn, barramundi, mud crab etc.), with some sub-models linked (Figure 3). The sub-models are dynamic (i.e. capture changes over time) and also capture non-linear relationships and feedbacks unlike static statistical models that have been used previously to model WRD e.g. Broadley et al. 2020. The sub-models make up the MICE and various input data are fed into the MICE including: updated river flow scenarios from the NAWRA project, physical variables, biological and ecological data, as well as fishery data. Where possible, the MICE is validated by fitting to some of these data, particularly fishery data. Additionally, we constructed alternative versions of the MICE creating a MICE ensemble (Figure 3). These alternative MICE versions help account for uncertainty in model structure, input parameters and underlying hypotheses (Figure 3). To capture spatial differences across the GoC, we developed a MICE ensemble for eight different regions (see section 2 and Figure 5). Some of the sub-models (e.g. prawns) were linked across some of the regions. Outputs from the MICE ensemble were then used to quantify changes in the abundance, catch and economics (NPF) of relevant species or groups and the ecological, fishery and economic risk under different WRD.

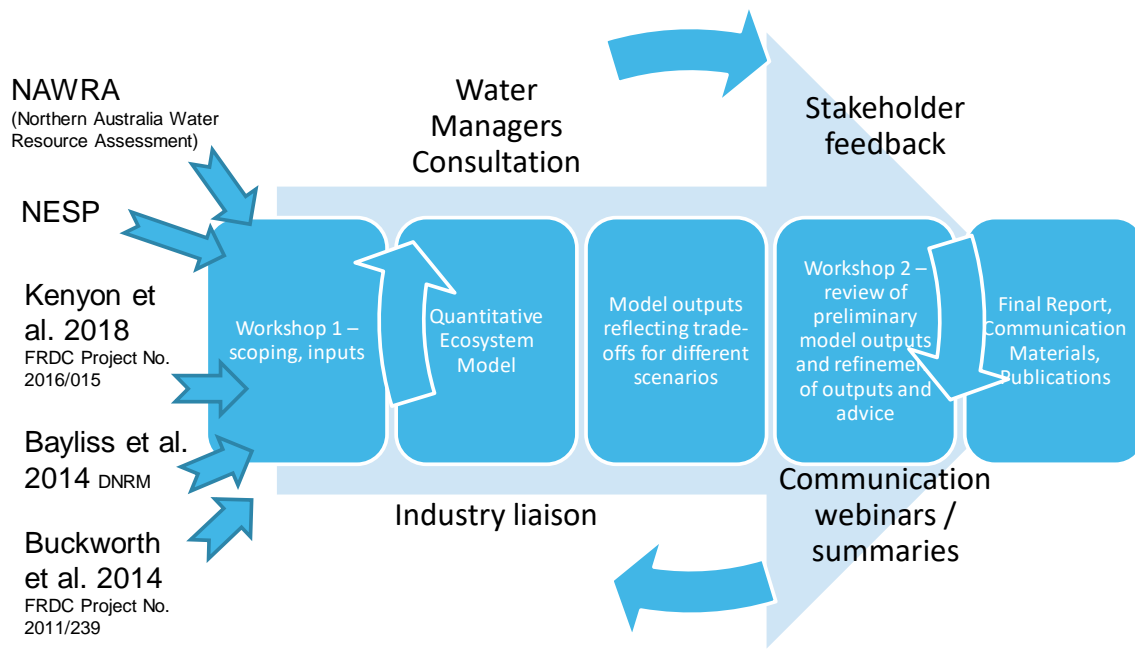


Figure 2. Schematic showing steps and processes involved in overall project

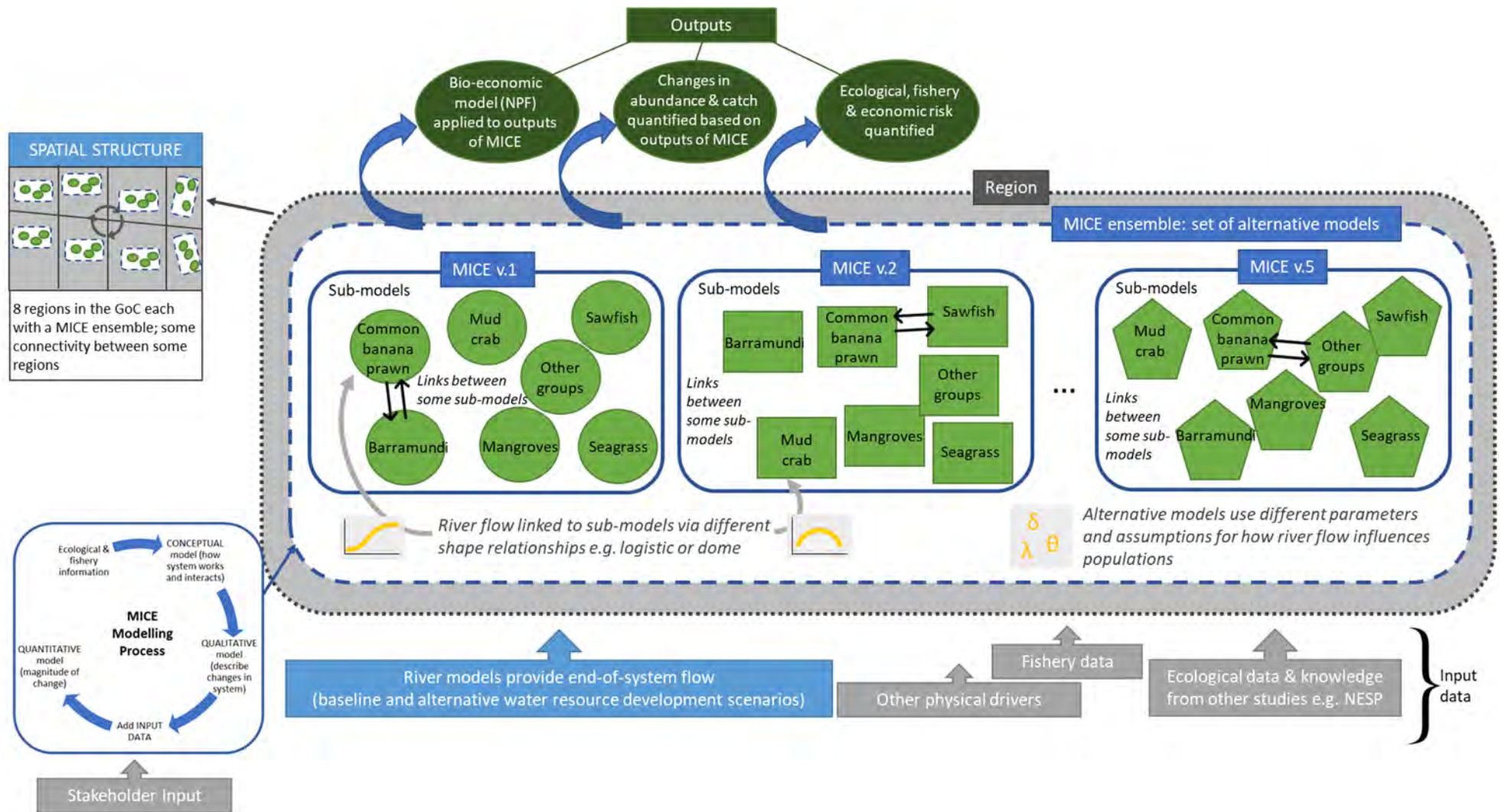


Figure 3. Schematic summary of Gulf of Carpentaria MICE modelling process and MICE structure. See glossary for technical terms used and model terminology

2. Gulf of Carpentaria MICE model overview

“Models of Intermediate Complexity for Ecosystem assessments” (MICE) have a tactical focus, including use as a multi-species assessment tool. Although MICE employ similar methods to single species stock assessment models, it is important to note they are not stock assessment models. Our MICE is therefore not intended to provide accurate estimates of the current status of different species as a basis for determining sustainable yields. Although the model is fitted to available data, for many of the model components there are limited (or no) data only and we therefore account for uncertainty by developing alternative models. The model is also used to evaluate changes in response to WRDs relative to the current no-development scenario, and it is these relative changes rather than the absolute biomass under any scenario that are of more relevance here. Context- and question-driven, MICE focus only on ecosystem components required to quantify specific impacts (and potentially mitigation and alternative solutions) on the ecosystem and stakeholders of alternative water resource development scenarios. MICE are typically constructed in a stepwise fashion by starting with focusing on the key species and processes of interest and then gradually adding complexity as necessary (Plagányi et al. 2022). Hence the first part of the project involved identifying the key species, namely prawns, barramundi, mud crabs and sawfish (Figure 4).

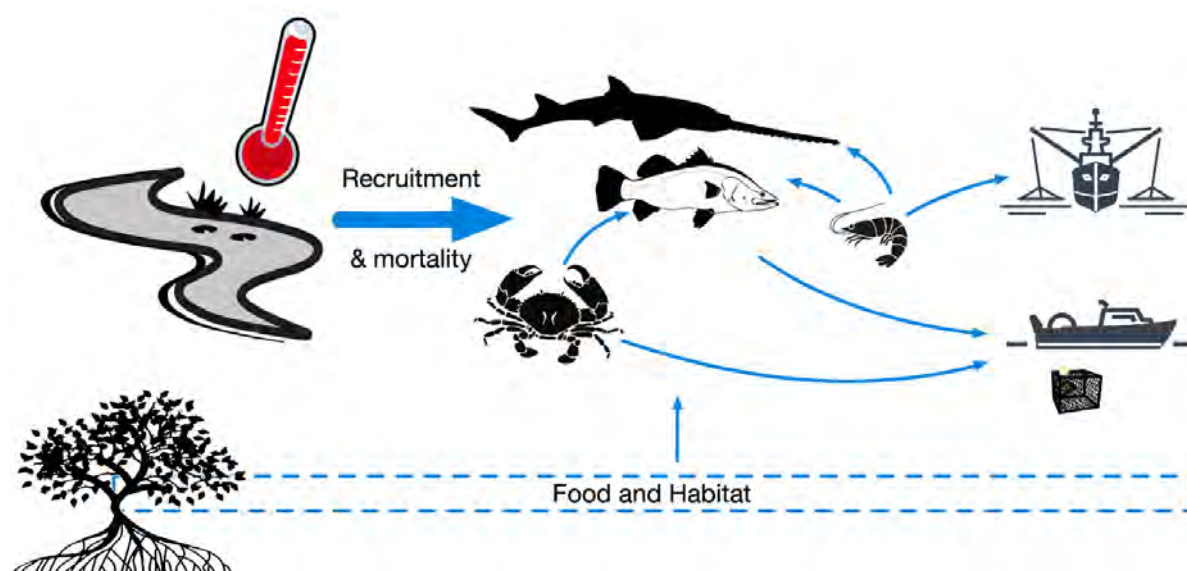


Figure 4. Schematic summary of the key species and drivers of ecosystem change in the Gulf of Carpentaria system

For species such as common banana prawns, the model has been formally statistically fitted to available data such as survey and catch data, using methods similar to that used in stock assessments and implemented in AD Model Builder. Although the model isn't a detailed mechanistic model, it incorporates some mechanistic representation of processes where necessary, and is based on the latest available scientific research and data. As there are important differences in system structure and functioning, as well as management differences between Queensland and the Northern Territory, a spatially structured model was developed.

The spatial divisions (Figure 5) were selected using the following criteria:

- Detailed focus on rivers most likely to be affected by WRD (Mitchell, Gilbert, Flinders)
- Major river basins and biogeographical regions

- Common banana prawn stock regions
- Barramundi stock regions
- Qld-NT state boundary (different management rules)
- Potential future uses of the model e.g. might want to model Roper River (region 7) in more detail hence adjusted regions 7 and 8 so that the Walker River basin is in region 8.
- The regions are complex polygons and so they can be used in spatial analysis (e.g. with barramundi data) (shapefile).

The model sub-structure considers freshwater, estuary and offshore dynamics operating differently. Moreover, alternative hypotheses regarding the patterns of connectivity in the offshore region can be explored in the model.

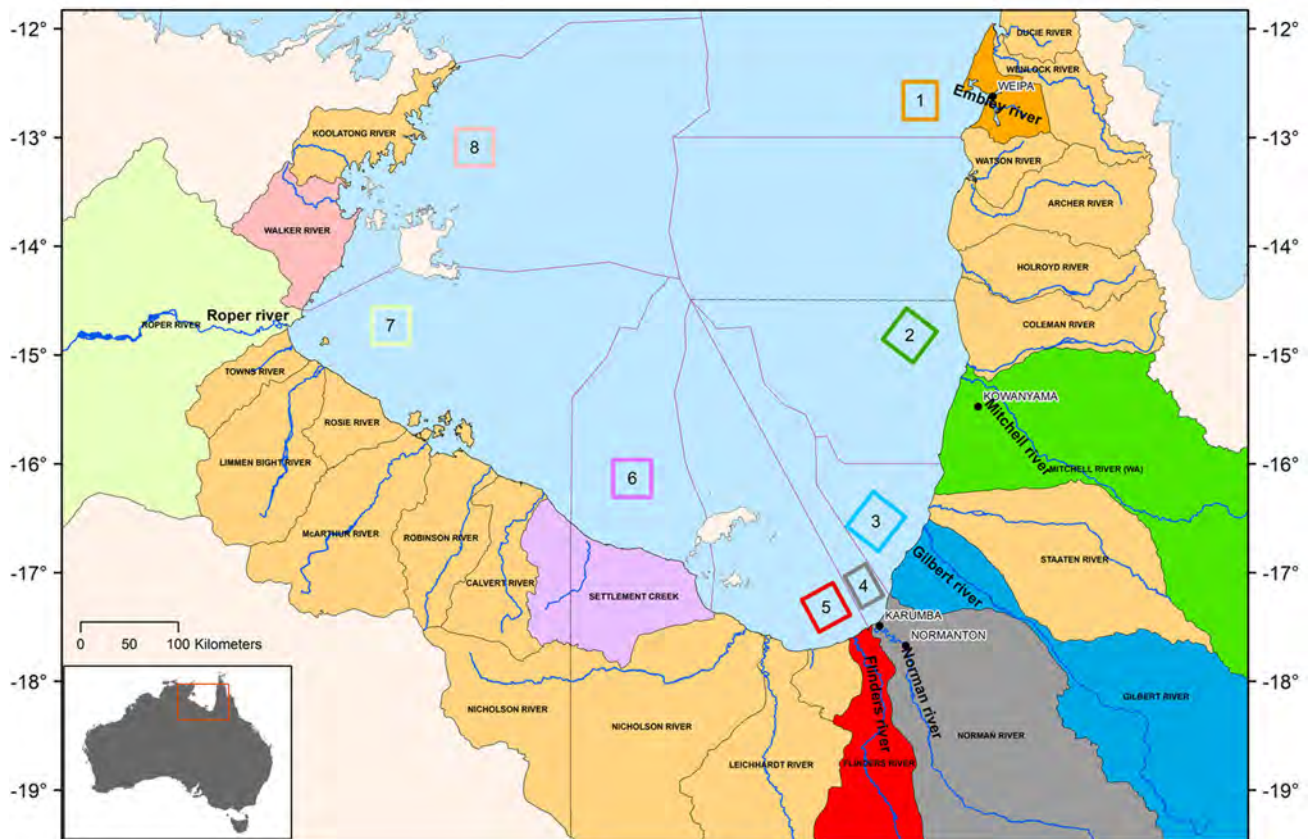


Figure 5. Spatial map of Gulf of Carpentaria study area showing sub-division into 8 connected spatial regions with major river catchments as follows (from East to West, with the vertical line between Region 6 and 7 representing the Qld-NT border): Region 1: Embley River (Weipa); Region 2: Mitchell River (Kowanyama); Region 3: Gilbert River; Region 4: Norman River (Karumba); Region 5: Flinders River; Region 6: border region; Region 7: Roper River; Region 8: Walker River. Model sub-components (common banana prawn, barramundi and mud crab) were fitted to catch data per region. For Region 2, barramundi and mud crab data were only used from the southern part of this region, indicated by dashed horizontal line, because data from the northern part of Region 2 weren't considered to be representative of the Mitchell River end-of-system flows.

Key species

The key species (groups) to be explicitly represented (with varying levels of complexity) in the model are as follows:

1. common (or white) banana prawns *Penaeus merguensis* (also known as *P. Fenneropenaeus merguensis* (Flegel 2007))
2. brown tiger prawn *P. esculentus*
3. grooved tiger prawn *P. semisulcatus*
4. barramundi *Lates calcarifer*
5. giant mud crab *Scylla serrata*
6. largemouth sawfish *Pristis pristis*
7. narrow sawfish *Anoxypristis cuspidate*
8. meiofauna
9. microphytobenthos
10. mangroves
11. seagrass

The key connections between these species and the rest of the system is summarised in Figure 6A. However for the purposes of this report we focus on a further subset of these species, namely those species and groups for which there are well-documented dependencies on end-of-system flows. Hence we do not present detail on tiger prawns or the narrow sawfish (Fig. 6B).

The Northern Prawn Fishery (NPF) is one of Australia's most valuable federally managed commercial fisheries and historically has regularly returned a profit (Rose and Kompas 2004). The fishery targets several species groups of prawns (banana, tiger, endeavour, and king) as well as other species of invertebrate, although the bulk of the revenue from the fishery is obtained from harvesting common banana prawns (*Penaeus merguensis*), grooved tiger prawns (*P. semisulcatus*), and brown tiger prawns (*P. esculentus*). However tiger prawns were not the focus of the current project and hence we focus on common banana prawns instead. Moreover, from the early days of the GoC common banana prawn fishery, a relationship between flow and catch was recognised. There are also important technical and economic interactions between these species, and they are in turn prey for a number of predators in the system, as summarised in Figure 6.

The fishery can be considered as two sub-fisheries which are spatially and temporally separate – a common banana prawn fishery and a mixed prawn fishery dominated by tiger prawns. The fishery has generally operated from April to November, with a mid-season closure from roughly June to August. After several industry and government funded buy-back schemes, there are now 52 vessels and 19 operators in the fishery. By comparison, more than 120 vessels operated in the fishery a decade ago, and over 300 vessels in the 1970s and 1980s (Punt et al. 2011).

Barramundi is a species of considerable commercial, recreational and cultural importance in the GoC. Populations of barramundi are likely to be highly responsive to river flows associated with any given river system because (Robins and Ye 2007): (i) limited migration between estuaries occurs, such that the estuarine population probably reflects local river flows; (ii) catadromy – migration upstream (by 0+ juveniles into freshwater habitats) and downstream (back to the estuary by adults) is facilitated by river flows; (iii) growth rates of juveniles and adults are significantly affected by river flows, probably through habitat access and/or food availability, which would impact upon juvenile survival rates and the subsequent abundance of adults; (iv) 'year-class strength' (which is an index of survival) of barramundi is significantly affected by river flows. However, relationships are unlikely to be simply linear, as there are flow thresholds and multi-year effects. Barramundi were therefore considered a fourth key species in the MICE. These fish spawn in the lower estuary and the larvae and small juvenile fish are estuarine. Juvenile fish however move upstream to freshwater riverine and floodplain aquatic habitats. Access to freshwater floodplain habitats is dependent on the level of large

river flows, overbank inundation, access to floodplain billabongs, and the creation of shallow floodplain aquatic habitats during the wet season, including floodplains in the coastal deltas of GoC rivers which are spatially extensive and probably offer expansive temporary habitat for young-of-the-year barramundi. As small adults, barramundi are usually males and they spend ~1-3 years in freshwater habitats. Subsequent high-level wet season flows reconnect the refugia and upstream barramundi emigrate downstream to the estuary where they mix with females to ensure they participate in annual spawning events in the lower estuary. Within the estuary, individual male barramundi transform to become females and they remain females for their ~20-25-year life. For barramundi, high level flows ensure connectivity of both floodplain and riverine habitats and riverine and estuarine habitats to support high population levels. Barramundi fishery catch is lagged to flood events by 1-6 years as it takes a number of years for barramundi to grow to a size that is available to the fishery and high-level flows as juveniles and male fish support palustrine and riverine habitat connectivity that sustains a large population which emigrates downstream to the estuarine fished population 3-6 years hence (Robins et al. 2005; Balston 2009; Crook et al. 2017).

Mud crabs, the fifth key species, are similarly important commercially, recreationally and culturally and are characterised by an inshore/offshore life history, however, unlike common banana prawns, both their juvenile and adult life history phases are estuarine (near coastal) residents. Egg-bearing adult female mud crabs emigrate offshore to release their eggs in marine waters and their pelagic larvae return inshore to reach their estuarine habitats where they settle to a benthic existence at the mangrove forest interface or within seagrass vegetation after about 3 months. Juvenile mud crabs are tolerant of low-salinity conditions within an estuary (3-45 ppt) and benefit from the river flows delivering nutrients and stimulating productivity within estuarine habitats. Adult crabs are less tolerant to low-salinity estuaries, though they thrive in brackish conditions. Mud crabs emigrate from estuaries that are subject to freshwater input during high-level floods and move to coastal littoral habitats. The relationship between river flows and mud crab populations is less well defined than for common banana prawns and barramundi, however, several researchers have found a positive correlation between flow and commercial mud crab catch (Loneragan 1999), Robins et al. (2005), (Meynecke and Meynecke 2010, Meynecke et al. 2011), and recent regional analyses of multiple environmental drivers have confirmed the influence of flow on catch, but with regional differences (Robins et al. 2020).

In Australia sawfish occur throughout the northern regions (Stevens et al. 2008, Morgan et al. 2011) and three species of sawfish are listed as “Vulnerable” under the Environment Protection and Biodiversity Conservation Act, 1999 (EPBC Act). These are largetooth (*Pristis pristis*), green (*P. zijsron*) and dwarf sawfish (*P. clavata*). A fourth species, the narrow sawfish (*Anoxypristis cuspidata*) is not listed as threatened, but as Migratory because of its listing on Appendix I and II on the Convention on the Conservation of Migratory Species, and therefore has similar protected status under the EPBC Act.

Sawfish have historically been taken in significant numbers by inshore gillnet and prawn trawl fisheries with ongoing catches of concern. In addition, they are threatened by alterations to the river and estuarine habitats they rely on for part of their life cycle. In this report we focus only on largetooth sawfish due to the dependency of their life cycles on river flows.

Additional model components have also been developed, but as these are considered less certain than those listed above, it is optional whether or not they are linked to the key species in any of the base model runs. The additional components are firstly microphytobenthos and meiofauna, which are included as indicators of the base productivity of the system, drawing on the extensive research of Burford, Duggan, Kenyon and colleagues. Next spatially-resolved models of the mangrove and seagrass communities were developed as no suitable existing models could be found that represented these key habitats at an appropriate scale and with an intermediate level of complexity, rather than detailed individual growth models of example.

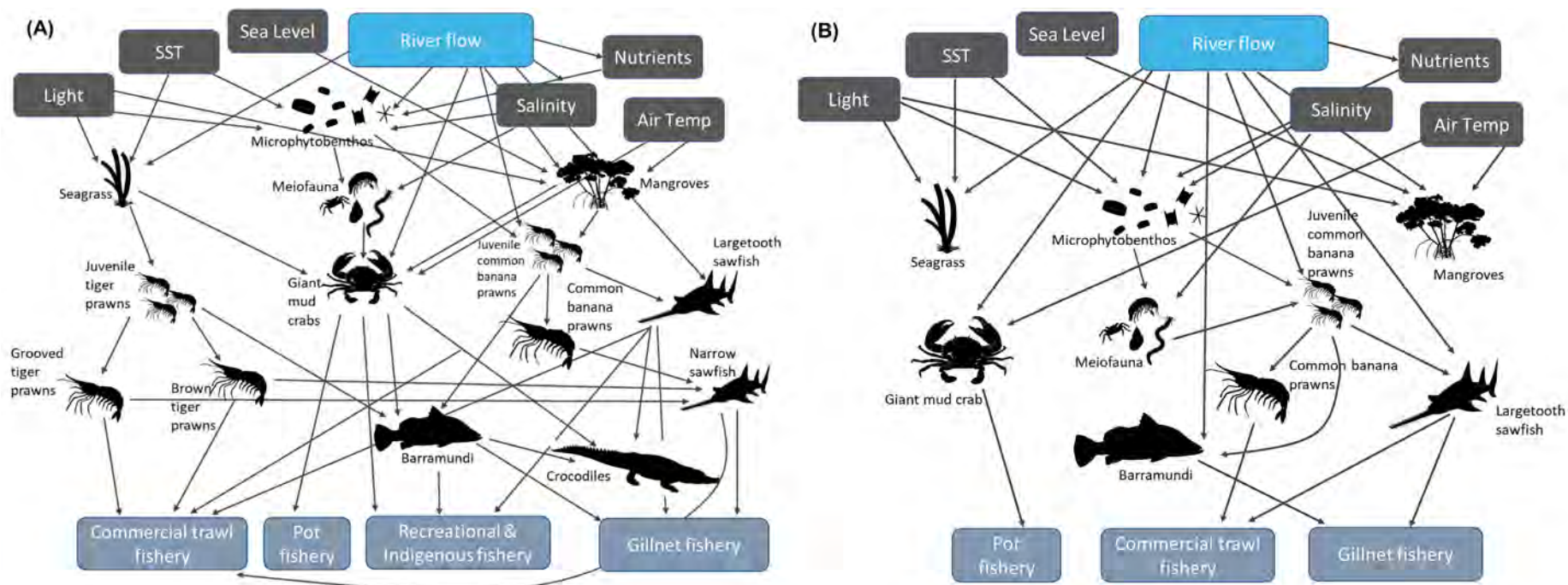


Figure 6. Model schematic showing (A) the key species groups and linkages identified for modelling in the spatial Gulf of Carpentaria MICE. Freshwater river flow inputs are highlighted in light blue as they are a key driver and the other physical drivers are shown with dark grey colouring, whereas the fishery sectors are shaded blue-grey. (B) Focus species and linkages in the spatial Gulf of Carpentaria MICE that were included in the MICE when testing water resource development (WRDs) scenarios.

3. Water Resource Development overview and modelling of river flows in the Gulf of Carpentaria

River flows were derived from water resource assessments undertaken across northern Australia in the last decade: the FGARA, NAWRA and Roper River Water Resource Assessment (RoWRA). End of system river flow data were produced using a calibrated AWRA-R river system model (Dutta et al. 2015) or model predecessors in the case of FGARA (see Lerat et al. 2013).

The Mitchell River end of system flows were produced using a calibrated AWRA-R model (Dutta et al. 2015). For the Mitchell River in particular, the model was built and calibrated as a part of the Northern Australia Water Resource Assessment (NAWRA). For further information see (Hughes et al. 2017, Hughes et al. 2018a). The intention of these models was to simulate river flow at various locations throughout the catchment at a daily scale, while also having utility to simulate flows for scenarios that include various future climates and irrigation development, in particular water harvest and larger in-stream dams.

The AWRA-R model accepts daily estimates of precipitation and potential evaporation aggregated to each residual catchment. These are derived from SILO data drill (Jeffrey et al. 2001). The initial Mitchell River model utilised inputs from January 1890 to September 2015. These were updated for this study, extending simulation dates to 31 January 2019.

Roper River end of system flows were also derived using a calibrated AWRA-R model. These were generated as part of the Roper River Water Resource Assessment (RoWRA). It should be noted that these data are preliminary and are subject to change with further model testing.

The Flinders and Gilbert simulations were based on river models built for the Flinders and Gilbert Agricultural Resource Assessment (FGARA). The model aims and uses were similar to those required for the NAWRA study. However, simulations were based upon the commercial, 'Source' modelling framework. For more information see Lerat et al. (2013) and Holz et al. (2013).

For most of the contributing area to the Gulf of Carpentaria, no river models are available. Within these areas, some stream gauge data are available. However, few of these are available at the respective catchment outlets and no gauge data are available across the entirety of the study period, i.e. no available gauge data matches the length of simulations from the available river models. This study utilised the AWRA-L model and outputs for the study area where no river model simulations were available.

AWRA-L is a grid based distributed water balance model that is conceptualised as a small catchment. It simulates the flow of water through the landscape from the rainfall entering the grid cell through the vegetation and soil and then out of the grid cell through evapotranspiration, surface water flow or lateral flow of groundwater to the neighbouring grid cells. Each grid cell is conceptualised as two separate hydrological response units (HRU), corresponding to deep rooted vegetation (trees) and shallow rooted vegetation (grass). The main difference between these two HRUs is that the shallow rooted vegetation has access to subsurface soil moisture in the two upper soil stores only, while the deep-rooted vegetation also has access to moisture in the deep store. The size of a grid cell is assumed to be large enough that hillslope processes are not important but small enough to assume homogeneity of the climate inputs (Viney et al. 2015). For this study climate inputs were available at 0.05 arc degrees (approximately 5 km), and thus, outputs including runoff were also produced at this scale. When all simulations for all grids within a catchment are aggregated, an estimate of runoff at the catchment outlet can be obtained.

The AWRA-L model has only one parameter set per region. This contrasts with conceptual rainfall-runoff models that operate on a catchment-by-catchment basis with a unique parameter set for individual gauged catchments. The AWRA-L model is calibrated with one parameter set to multiple catchments, and together with gridded physical inputs (e.g. soil information), is capable of robust

simulation in ungauged areas. For this study a ‘monsoon tropical’ regional parameter set was used (Viney et al. 2015). Estimates of runoff from simulation grids were aggregated to the Australian Water Resource Council (AWRC) watersheds to generate daily streamflow estimates at the end of each river basin in the Gulf area.

As a consequence of the AWRA-L model structure and its calibration routine, simulations tend to over-estimate low flows. To remedy this situation and correct any simulation bias, a quantile mapping transformation on the AWRA-L simulations was conducted. AWRA-L simulations were compared to runoff estimates from the Northern Australia Sustainable Yields Project (NASY). These used the Sacramento model and were calibrated to observed data in each of the catchments, and, hence, are a reliable source of information. Additionally, the NASY project dedicated many resources into data quality assurance and vetting. They were, however, for distinct locations within the AWRC catchments, rather than the end of system (Petheram et al. 2009). The transformation of daily flows compares the quantile values of runoff between AWRA-L values and NASY values to produce a transfer function. This transfer function is then applied to AWRA-L simulation to modify the flow distribution and bias to better match those from the NASY study.

$$F_r = H(F_a) \quad (1)$$

Where F_r is the vector of daily transformed flow estimates, H is the transfer function vector and F_a is the un-transformed AWRA-L flow vector. Each individual value in the transfer function is calculated as follows:

$$H_i = \frac{Q_{s_i}}{Q_{a_i}} \quad (2)$$

Where Q_{s_i} is the flow value of the NASY simulation for the quantile probability i and Q_{a_i} is the flow value of the AWRA-L simulation for the quantile probability i .

Summary climate and river flow data for the Flinders, Gilbert and Mitchell Rivers show that the mean and median annual rainfall for the Flinders, Gilbert and Mitchell catchments were 492 and 454 mm, 775 and 739 mm, and 996 mm (mean only), respectively. Of this 88%, 93% and 97% was calculated as falling during the wet season (1 November to 30 April) (Lerat et al. 2013, Charles et al. 2016). In each river catchment, the mean annual potential evaporation was >1800 mm, leaving a mean annual rainfall deficit > 600 mm. The median annual runoff spatially averaged across the Flinders, Gilbert and Mitchell catchments was 22, 100 and 229 mm. The percentage of runoff occurring in the wet season in the catchments was 95, 98 and 98%, respectively (Lerat et al. 2013, Hughes et al. 2017). Under the historical climate, the mean annual river flows in the Flinders, Gilbert and Mitchell River catchments were 2543 GL, 3719 GL and 15,570 GL respectively (Lerat et al. 2013, Hughes et al. 2017).

The initial Mitchell River model used inputs from January 1890 to September 2015, while for the MICE ecological model the data inputs were updated, extending simulations to 2019. The updated FGARA simulations were available for the time period 1 July 1889 to 31 January 2019; while the NAWRA and other simulations were updated to extend from 1 January 1900 to 17 November 2019. Daily flow value is given as cubic metres per second.

The following key was used to map flow estimates to the GoC 8 model regions Figure 5 Spatial map of Gulf of Carpentaria study area showing sub-division into 8 connected spatial regions with major river catchments as follows (from East to West, with the vertical line between Regions 6 and 7 representing the Qld-NT border): Region 1: Embley River (Weipa); Region 2: Mitchell River (Kowanyama); Region 3: Gilbert River; Region 4: Norman River (Karumba); Region 5: Flinders River; Region 6: border region; Region 7: Roper River; Region 8: Walker River.:

Region 1: Embley River (924)

Region 2: Mitchell River (919)

Region 3: Gilbert River (917)

Region 4: Norman River (916)

Region 5: Flinders River (915)

Region 6: Assume same as Flinders River

Region 7: Roper River (903)

Region 8: Assumed to correspond to Koolatong River (901) and Walker River (902)

The ecological model reads in the daily flow estimates for all historical years 1970 to 2018 as this period corresponds to that for which historical fishery (prawn) data are available.

4. Biological data collation: Qld and NT Fisheries

There are a lot of different data pertaining to the key model species, and the project team spent a considerable amount of time obtaining permission to use the data, which are variously owned by the Commonwealth, Qld and NT. The NT and Qld data are mainly used to inform on historical trends in barramundi and mud crab catches. The CSIRO team have access to the NPF prawn fishery and survey data for the key species: (1) common (or white) banana prawns *Penaeus merguensis* (2) brown tiger prawn *P. esculentus* and (3) grooved tiger prawn *P. semisulcatus*.

The fishery catch and effort data for the different fisheries (barramundi, mud crabs, prawns) correspond to fishing grids/blocks which have different spatial dimensions across fisheries, and hence these sub-areas were mapped to the MICE model regions. This was done by checking which region the centre of the fishing grid corresponded to, and then the data from those grids were subsequently assigned to that region. The data were also aggregated together to yield a total GoC series (i.e. a non-spatial version of the dataset) because a non-spatial prawn model version was developed in parallel to the spatial model version for the purpose of cross-comparing with the assessment model.

A summary of data sources and associated information is provided in Table 1.

Fishing power

For the tiger prawns, the same fishing power series were input as used in the stock assessment (Hutton et al. 2018). For common banana prawns, there are no comparably reliable fishing power series available and hence a series (with lower fishing power than applied for tiger prawns) was used based on outputs provided by Shijie Zhou (CSIRO), with a late update by Hutton et al. (2022) used as a sensitivity test.

Table 1. Summary of Biological data for the key species assessed in the MICE model

	Key species	Data Type	Temporal Scale	Spatial Scale	Data sources
1	Common banana prawns	Commercial catch	Weekly for all years since 1970	Per model region	NPFI, AFMA
2.	Common banana prawns	Recruitment abundance index	Annually from 2003	Aligned to model region	CSIRO
3	Common banana prawns	Spawner abundance index	Annually from 2002 up to 2010, then biennially from 2011	Aligned to model region	CSIRO
4	Brown tiger prawns	Commercial catch	Weekly for all years since 1970	Per model region	NPFI, AFMA
5	Brown tiger prawns	Recruitment abundance index	Annually from 2003	Aligned to model region	CSIRO
6	Brown tiger prawns	Spawner abundance index	Annually from 2002 up to 2010, then biennially from 2011	Aligned to model region	CSIRO
7	Grooved tiger prawns	Commercial catch	Weekly for all years since 1970	Per model region	NPFI, AFMA
8	Grooved tiger prawns	Recruitment abundance index	Annually from 2003	Aligned to model region	CSIRO
9	Grooved tiger prawns	Spawner abundance index	Annually from 2002 up to 2010, then biennially from 2011	Aligned to model region	CSIRO
4	Barramundi	Commercial catch, effort	Monthly from 1983 to 2019	Aligned to model region	DAF (data request DR2976), DITT
5	Mud crabs	Commercial catch, effort	Monthly from 1983 to 2018 (NT) or 1989 to 2019 (Qld)	Aligned to model region	DAF (data request DR2976), DITT
6	Largetooth Sawfish	Estimated catch from the by-catch from commercial fisheries.	Annually from 2002 to 2017	Aligned to model region in Qld	Rich Pillans (CSIRO)

5. Physical data collation: Qld and NT

The MICE includes a variety of physical variables which are used as drivers in the MICE model (Table 2). Further descriptions of the variables are provided in Appendix 6.

Table 2. Summary of physical variables used to drive components of the MICE model.

	Physical Variable	Temporal Scale	Spatial Scale	Impacted model groups
1	River flow	Daily for all years since 1900	Per model region	Prawns, mud crab, barramundi, sawfish, meiofauna, microphytobenthos, seagrass, mangroves
2	Southern Oscillation Index (SOI)	Monthly for all years since 1970	Index covers entire GoC	Mud crabs
3	Sea Surface Temperature (SST)	Weekly average	Selected sites	Seagrass, microphytobenthos
4	Salinity	Weekly average	Selected sites	Meiofauna, microphytobenthos
5	Air temperature	Daily, for various years depending on site	Selected sites	Mud crab, mangroves
6	Sea level	Monthly	One site	Mangroves, mud crab* (*not yet done, possible to include instead of SOI)
7	Solar exposure	Daily, for all years since 1990	Selected sites	Seagrass, mangroves, microphytobenthos
8	Cyclones	Since 1970	Region specific	Seagrass, mangroves

We drew on a variety of sources to obtain input values of physical variables for each week over the model time period, as well as disaggregated by area whenever possible. In some instances, we fitted additional models to available data for use in estimating values in the MICE. For example, to obtain a relationship between flow and salinity, we drew on Duggan et al. (2014) which included a time series and a total of 17 field measurements of salinity, temperature and a number of other variables at sites in the Norman River. We also fitted a sinusoidal curve to approximate the seasonal pattern of solar exposure, as described in the Appendices.

We used estimates of both sea surface temperature and air temperature as the latter can influence mangroves and mud crabs. Elevated temperatures or periods of heating can negatively affect some species in the GoC, e.g. as was recently seen with mangrove dieback in the 2015/2016 Extreme

Climatic Event (Duke et al. 2017, Babcock et al. 2019). Mud crabs are also susceptible to periods of heating during the summer months when temperatures are high and tides are low (Robins et al. 2020).

As anomalously low sea levels that are thought to influence mangroves (Duke et al. 2017) and prawn recruitment (Plagányi et al. 2020), we used observations of the minimum sea level per month to drive mangrove dynamics.

Recent increases in the frequency of Extreme Climate Events (ECEs) such as heatwaves and floods have been attributed to climate change, and could have pronounced ecosystem and evolutionary impacts because they provide little opportunity for organisms to acclimate or adapt (Babcock et al. 2019). ECEs have been documented as resulting in abrupt and extensive mortality of key marine habitat-forming organisms, such as corals, seagrasses and mangroves (Thomson et al. 2015, Hughes et al. 2018b, Babcock et al. 2019) and hence these impacts should ideally be incorporated in regional models.

Tropical cyclones of varying intensity occur in the GoC region, and can sometimes have significant negative effects on the mangrove and seagrass systems (Poiner et al. 1993). These littoral habitats are in turn critical nursery habitats for species such as tiger prawns and fish (Brewer et al. 1995, Loneragan et al. 1998). Cyclones that formed in or entered the Gulf of Carpentaria (GoC) from 1970 to 2019 were categorised on the basis of intensity and they were allocated to one of the eight GoC model regions used in the MICE (Appendix 6). Two sources of information were used to extract information.

Information on cyclones in the 1970s and 1980s was derived from a written document developed by the Bureau of Meteorology that described cyclones qualitatively from 1885 to 2004 (a summary prepared by Jeff Callaghan, BOM, 'Known tropical cyclone impacts in the Gulf of Carpentaria' (<https://www.australiasevereweather.com/cyclones/impacts-gulf.pdf>)). From 1885 to recent decades, the descriptions in the document increased in complexity and made greater use of observations from technological tools. The GoC was and continues to be a sparsely populated expanse of tropical Australia. In the early- to mid-1900s, reports were made from remote cattle stations or aboriginal townships (e.g. Ngukurr (Roper River Mission) and Pormpuraaw (Edward River Mission)) further north, and a few towns in the southern GoC catchments such as Normanton and Burketown. During the later decades of the 1900s, mining and fishing towns in the GoC such as Karumba (south east) Alyangula (Groote Eylandt, north west) and Weipa (north east) also provided detailed descriptions of cyclones, usually with the benefit of new technology. The descriptions were qualitative, but often detailed, and aspects such as the height/ depth/ extent of the storm surge and the damage/ demolition/ removal/ transport of various pieces of built-infrastructure were described. These descriptions assisted with allocating a wind-strength category and a point-of-crossing of the coast to the cyclone. However, the level of precision of a category interpreted from the early qualitative descriptions may be low.

From 1997 to 2021, remotely sensed images of Australian cyclone tracks have been documented by the BOM. They are presented as maps available on the BOM web site (<https://www.australiasevereweather.com/cyclones>). The tracks are accurately plotted and show the exact cyclone track and location where it crossed the coast. In addition, the tracks are colour-coded by wind strength category, so the cyclone wind strength category at the precise location where it crossed the coast is known.

From 1997 to 2004, both types of cyclone-documentation exist, so cross referencing of qualitative with quantitative descriptions was made. In some cases, the MICE region where the cyclone crossed the GoC coast was updated as the mapped tracks were more accurate than the qualitative on-ground observations.

The likely impact of a cyclone was estimated using two variables; cyclone intensity category, and the direction of impact of the cyclone track relative to the coast.

Cyclone intensity had six categories, a tropical low (category = 0.5) and then categories 1 to 5 as per the BOM definition of cyclonic wind strength. The direction of cyclone track had four categories:

- 0.25 – a cyclone crossing from land to water (often the case from the Coral Sea across Cape York to the GoC),
- 0.5 – a cyclone crossing the coast from water to land at a perpendicular angle,
- 0.75 – a cyclone crossing the coast from water to land at an oblique angle,
- 1 – a cyclone travelling parallel to and close to the coast before crossing from water to land at any angle.

On-ground experience in the Gulf of Carpentaria showed that the track of the eye of a cyclone relative to the coast is critical to the amount of destruction of littoral habitats. For example, in 1984 Cyclone Kathy crossed the Northern Territory coast at the Sir Edward Pellew Islands as a Category 5 cyclone and did not impact seagrass habitats west of the Vanderlin Islands (Poiner et al. 1987). In contrast, in 1985, Cyclone Sandy (Category 4) travelled parallel to the coast from the Sir Edward Pellew Islands (Vanderlin Island) to the Limmen Bight River/Roper River. The eye of the cyclone travelled along the coastline over the water/land interface and wind-driven wave action removed 183 km² of seagrass (Poiner et al. 1989), as well as severely damaging mangrove communities.

6. Stakeholder workshops

Workshop 1: Stakeholder engagement and scoping workshop (7-8 August 2019)

The first project workshop was held 7-8 August 2019, at CSIRO's Queensland Biosciences Precinct (QBP) lab in St Lucia, Brisbane. The agenda, list of participants and summary outcomes are shown in Appendix 2.

A diverse group of stakeholders was invited, including scientists, industry representatives, water managers, and community representatives. A number of background summary documents were prepared and circulated to participants before the workshop (see Appendix 3 in workshop report). The background documents summarised the current state of understanding and key considerations related to potential future water resource uses, as well as key species to be included in the model. Reference was also made to the considerable number of previous studies and how this project could build on those.

The workshop included a range of presentations by subject matter experts as well as several discussion sessions to accommodate additional comments as well as to collectively agree on the approach and modelling framework being used as part of the project.

In addition, there was a 1-day technical workshop held after the stakeholder workshop to refine some of the aspects raised during the workshop, and additional sub-group discussions in the weeks after the workshop with some stakeholders further informed these considerations.

A further three focussed teleconference workshops were held as follows, with detailed summaries given in Appendix 3. A final stakeholder workshop was held in August 2021 with details in Appendix 4.

Teleconference Workshop 1: Prawn focus (24 February 2021)

- overview of spatial, multispecies prawn model & fitting to data
- linking environmental drivers & quantifying impact of changes in flows – preliminary results
- update on WRD scenarios to be tested

Teleconference Workshop 2: Mud crab focus (23 March 2021)

- overview of mud crabs and modelling approach
- linking environmental drivers – preliminary results
- modelling mangrove & seagrass systems

Teleconference Workshop 3: Barramundi focus (24 April 2021)

- overview of barramundi & modelling approach to be used

Workshop 5: Final Stakeholder Workshop (9-10 August 2021)

The final stakeholder workshop was held 9-10 August 2021. The workshop was initially planned to be in-person and was delayed more than once in order to maximise the chances of holding an in-person (or hybrid style) workshop, but COVID-19 outbreaks meant it was eventually held virtually via videoconference. The full details of the workshop are provided as Appendix 4.

7. MICE sub-models

Overview of focus model species

Common banana prawns (*Penaeus merguensis*) are characterised by an annual inshore/offshore life history where the adults occupy relatively shallow coastal waters (10-30 m deep) and highly fecund females spawn large numbers of eggs which develop into pelagic larvae. During their juvenile phase however they are estuarine residents and over about 3-4 weeks, the larvae advect inshore on current and tides to move up the large estuaries of tropical Australian rivers to become benthic residents within the mudbank-mangrove forest matrix in estuarine tributaries. They reside within the estuary for 3-4 months and as they grow, they move to downstream estuarine habitats from where they emigrate offshore to grow to adults. Natural mortality and fish predation is high within estuarine habitats, and the population suffers mortality that can reach 20% per week. Forty years of biological research has demonstrated that large catches of common banana prawns in the offshore fishery are directly related to high-level flood flows that cue estuarine juvenile common banana prawns to emigrate offshore to marine habitats where mortality is lower, the population thrives, and they are harvested annually (Vance et al. 1998, Duggan et al. 2019, Broadley et al. 2020). Within-catchment water extraction or impoundment to support irrigated agriculture has the capacity to reduce the level and seasonality of annual river flows and hence reduce the contribution of the annual common banana prawn population to the offshore fishery.

Barramundi (*Lates calcarifer*, a large catadromous predator) are characterised by an inshore/ riverine life history and they occupy estuaries and close-inshore habitats as adults. They spawn in the lower estuary and the larvae and small juvenile fish are estuarine. Juvenile fish however move upstream to freshwater riverine and floodplain aquatic habitats. Access to freshwater floodplain habitats is dependent on the level of high river flows, overbank inundation, access to floodplain billabongs, and the creation of shallow floodplain aquatic habitats during the wet season. During the ensuing dry season, juvenile barramundi occupy permanent waterhole refugia in both the mostly dry river channels and on the floodplains or in some systems retreat to the estuary itself. As small adults, barramundi are predominantly males and they spend ~1-3 years in freshwaters habitats. Subsequent high-level wet season flows reconnect the refugia and male barramundi emigrate downstream to the estuary where they mix with females to ensure successful annual spawning events in the lower estuary. Within the estuary, individual male barramundi transform to become females and they remain females for their ~ 30-year life. For barramundi, high level flows ensure connectivity of both floodplain and riverine habitats and riverine and estuarine habitats to sustain high population levels. Fishery harvest of barramundi is lagged 1-6 years after preceding river flow events as there are short-term catchability/availability effects and longer term recruitment effects. Barramundi take between 2 and 5 years to grow to legal size and become 'available' to the fishery. (Robins et al. 2005; Balston 2009; Crook et al. 2017). Our MICE built on features of barramundi modelling included in the existing stock assessments of Campbell et al. (2017) for the Qld Southern GoC stock and Streipert et al. (2019) for the Queensland east coast stock.

Mud crabs (*Scylla serrata*) are characterised by an inshore/offshore life history, however, unlike common banana prawns, both their juvenile and adult phase are estuarine residents. In some regions e.g. the southern and southwestern GoC, they can be found along the extensive shallow coastal flats adjacent estuaries. Ovigerous adult female mud crabs emigrate offshore to extrude, incubate and hatch their eggs in marine waters and their pelagic larvae return inshore to reach their estuarine habitats where they settle to a benthic existence at the mangrove forest interface or within seagrass vegetation between 1 and 3 months, with development being temperature dependent. Juvenile mud crabs are tolerant of low-salinity conditions within an estuary (3-45 ppt) and benefit from the river flows delivering nutrients and stimulating productivity within estuarine habitats. Adult crabs are less tolerant to low-salinity estuaries, though they thrive in brackish conditions. Mud crabs emigrate from

estuaries that are subject to freshwater events during high-level floods and move to coastal littoral habitats. The relationship between flows and mud crab populations is less well defined than for common banana prawns and barramundi, however, several researchers have found a positive correlation between flow and commercial mud crab catch (Meynecke et al. 2010, 2012), and recent regional analyses of multiple environmental drivers have confirmed the influence of flow on catch, but with regional differences (Robins et al. 2020).

Characteristically for Australia's tropical estuaries, each year over the last four months of the dry season (September to December), the estuaries provide stable, albeit increasingly harsh (temperature and salinity) habitats for the settlement and benthic recruitment of juvenile fish and crustaceans following annual reproductive events. Estuarine macrophytes (mangroves, seagrass and algae) provide the juvenile phase protection from predation (Kenyon et al. 1995; Vance et al. 2002), while microphytobenthos sustains primary production and fuels estuarine foodwebs (Burford et al. 2012, 2016; Duggan et al. 2014). The onset of the wet season changes the estuarine environment and the habitats available to estuarine residents. Freshwater inflows transport sediments, deliver nutrients, lower salinities, create mechanical forces and scouring of channels and sediments inundate over-bank habitats, connect ephemeral habitats, and modify tidal inflows. Each species has evolved to use one or more of these annual opportunities to enhance their population or move to a habitat suitable for the next phase of their life history. The timing and volume of wet season flows and floods are critical to estuarine conditions and species response to the monsoon-driven river flows (Warfe et al. 2011).

Background and technical details on each of the following are provided in Appendices 7 to 15:

- a. Prawn model component
- b. Mud crab model component
- c. Barramundi model component
- d. Sawfish model component
- e. Microphytobenthos and meiofauna model components
- f. Seagrass and mangrove model components

The MICE has tailored the equations used and time step for each species and model group based on available information and data (Table 3). A weekly time-step was selected for prawns because the data were available on a weekly basis and the dynamics are known to be influenced by fairly short-term processes. Data for barramundi and mud crabs were available with a monthly time step and although few data were available for sawfish, their dynamics were also represented using a monthly time-step, in part because estimated catches of sawfish are based on the monthly fishery effort data. Moreover, the population dynamics of all of these species are influenced by river flows and hence use of a monthly time-step allows resolution of seasonal changes and impacts on these populations. In addition to being influenced by a common physical driver (river flow), there are also a number of important technical and biological interactions between these species, as shown in Figure 6.

The full set of mathematical equations, variable and parameter definitions and input values are provided in Appendix 16.

Species modelled using a weekly time-step

The MICE includes three species of prawn that are represented using a weekly time-step. Historical prawn catch, effort and CPUE data from the fishery have been input to the MICE, as well as spatially-disaggregated data. The fishery data have also been analysed to inform the minimum number of

selectivity periods that should be used in the model. These represent years when the level of fishing effort was reasonably comparable as well as the intra-annual fishing pattern (for example, which months of the year fishing occurred). In addition to viewing changes in fishing patterns over years, plus major changes in management, the total fishing effort (units of number of boat days) for all years since 1970 was plotted as shown in Figure 7 below. Based on these analyses, three fishing periods with different selectivities were defined as follows:

- (1) 1970-1987
- (2) 1988-2001
- (3) 2002-current

These periods are consistent with those used by Deng et al. (2020a) and are implemented in the model.

The intra-annual distribution of fishing effort has varied substantially for both tiger prawns and common banana prawns since the start of the fishery around 1970. Over the most recent period, common banana prawns dominate fishing activity in the first season (April-June) of the year (Figure 8), and most boats stay in this sub-fishery for most of the season (Pascoe et al. 2018). Depending on the level of fishing effort determined by the initial biomass and the opportunity cost of fishing for tiger prawns, some vessels leave early (Pascoe et al. 2016, Pascoe et al. 2020) with implications for changes for within-season changes in technical factors (Pascoe et al., 2017) which have not been taken into account in the current method for estimating the trigger. In the second season (August-November), most GoC boats target tiger prawns (Figure 8) and there are also incidental catches of endeavour prawns (not currently included in the model). There have also been alternative approaches (Kompas and Chu 2018) for estimating the stock levels and direct profit gains and losses in any given year. Zhou et al. (2015) applies a depletion method for estimating biomass each year but this has not been applied to the recent years and does not incorporate a stock recruitment relationship (i.e. estimation of one).

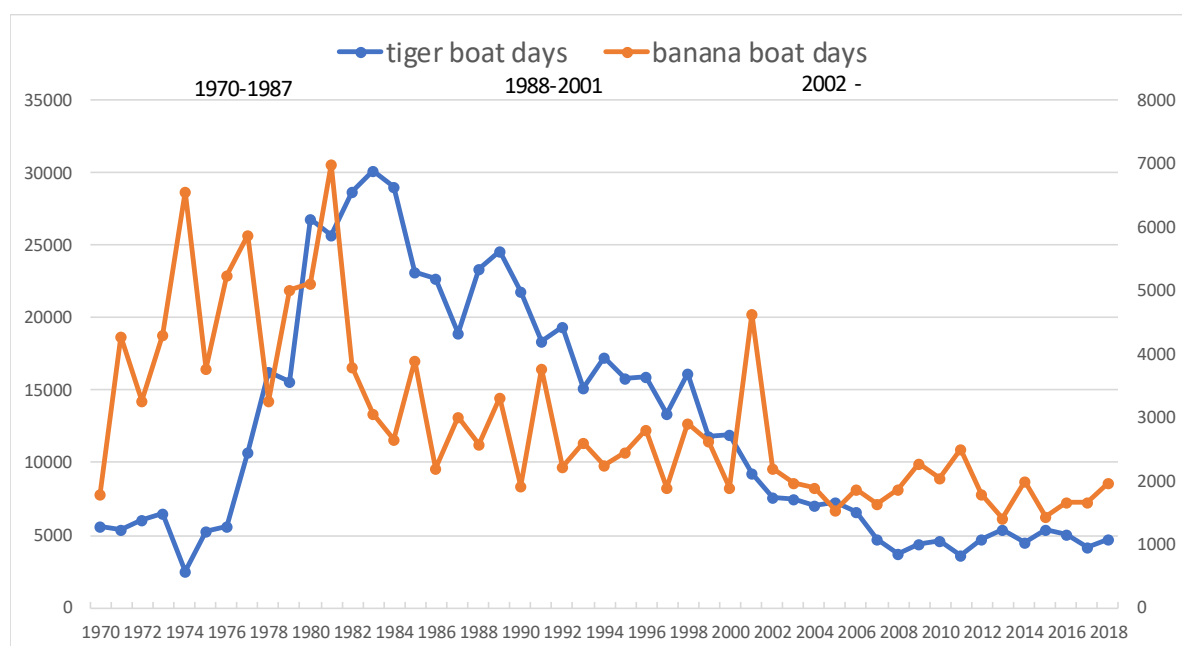


Figure 7. Plot showing the correspondence between total fishing effort for tiger prawns and common banana prawns mapped with the three selectivity periods selected for the MICE model.

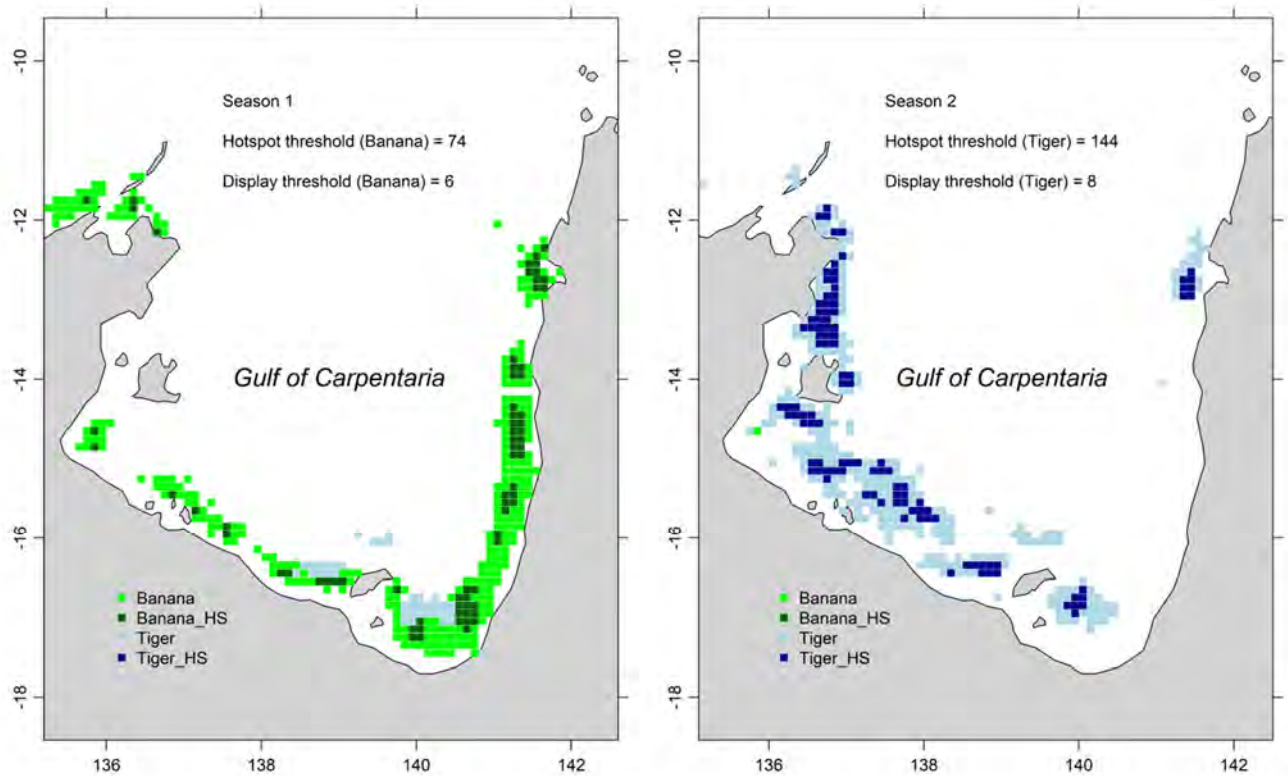


Figure 8. Distribution of effort targeted at the common banana prawns (green) and tiger prawns (blue) across the two fishing seasons in the NPF. Effort is summed boat days over the 2009-2018 fishing seasons as shown. Only locations with at least 8 days for tiger prawns and 6 days for common banana prawns over the year are shown. GoC- Gulf of Carpentaria; HS-hotspot. Modified from Pascoe et al. (2020)

The tiger prawn and common banana prawn fishing power series used in the model to standardise prawn catch rates are shown in Figure 9. The MICE used a simple trend assuming an annual increase of 2.3% per year for the period 1970 to 2011 (and a further increase thereafter), which compares with an estimate of 2.6% per year calculated for the period 1987 to 2011 by Zhou et al. (2015). However recent updated estimates for common banana prawn are lower and Appendix 21 shows the results of a sensitivity test to evaluate the influence of the assumed fishing power trend on model results. The MICE fishing power trend is roughly intermediate between that used for grooved and brown tiger prawns (Deng et al. 2020b), and in a recent analysis of common banana prawns (Hutton et al. 2022).

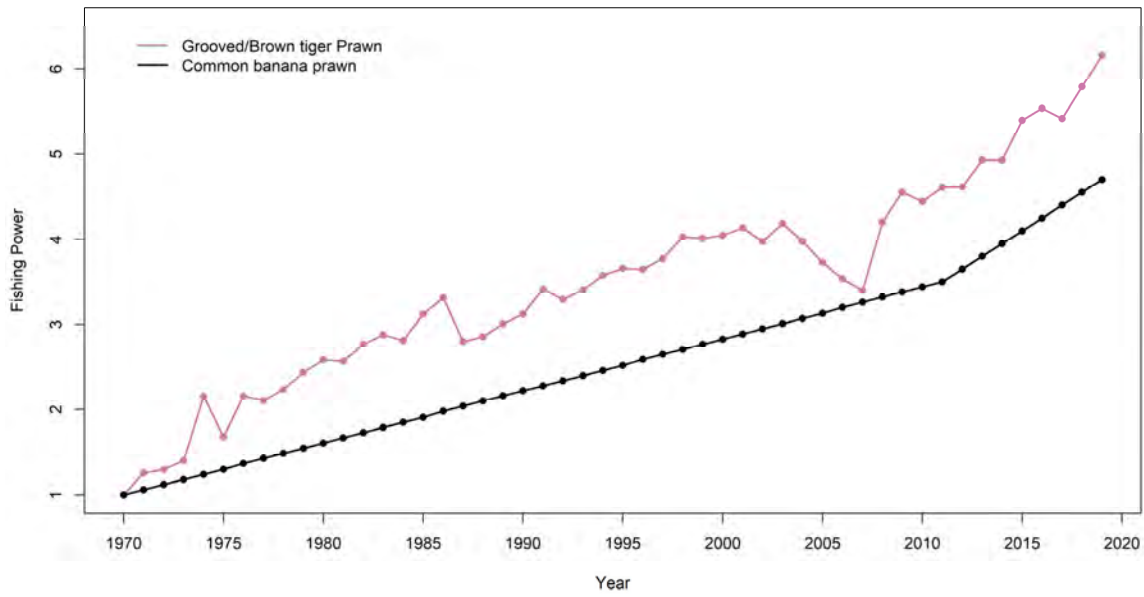


Figure 9. Fishing power estimates used to standardise nominal catch rates for common banana prawns and for the two species of tiger prawns in the MICE model

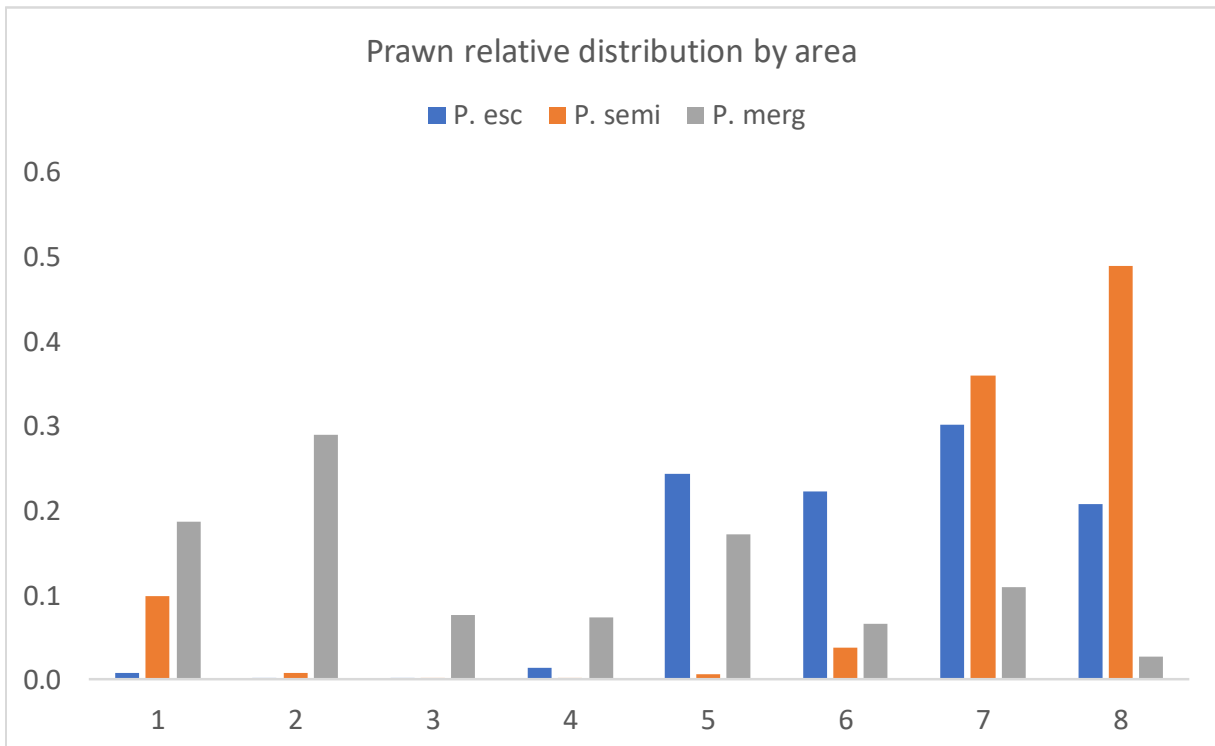


Figure 10. Summary of the relative abundance of the three prawn species across the 8 MICE model regions as shown.

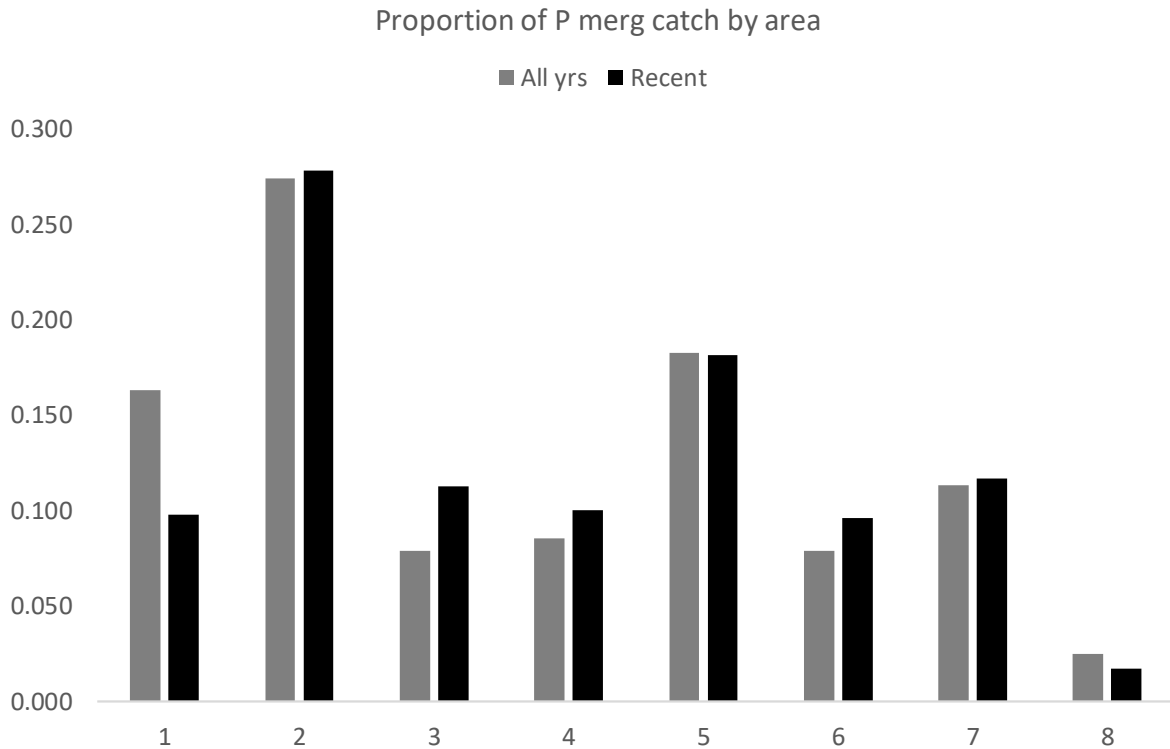


Figure 11. The proportion of *P. merguensis* total catch that derives from each of the 8 MICE regions, based on an overall average across all years (1970-2019) compared with more recent years only (2002-2019).

The common banana prawn *P. merguensis* component of the model is fitted to weekly catch data disaggregated into each of the 8 model regions over the period 1970 to 2019. To estimate the starting (1970) spawning biomass in the model, the relative distribution of each of the prawn species is assumed to be proportional to the relative catches. As shown in Figure 10, the relative distribution of the two tiger prawn species and the common banana prawn in each of the eight model regions differs considerably, with common banana prawns more evenly spread through the GoC whereas tiger prawns are caught mainly in the more western regions 5 to 8. Historically, the highest catches of common banana prawns have been taken from the Mitchell River region 2, followed by the Flinders River region 5 (Figure 11). There have also been changes over time in the relative common banana catch per region, with the most notable change being a decrease in the relative catch from the Embley River region 1 and an increase in relative catch from the Gilbert River region (Figure 11).

The model spatial domain is the GoC so these analyses ignore catches from outside this area. However, to reduce the numbers of missing data (i.e. zero catches per week), the model compares the predicted and observed sum of the catch over weeks 13-22 (Apr – mid-June) of each year (Figure 12). There is a very strong positive correlation between catches during this peak period when the fishery operates, and the total annual catch, apart from a few early years when the selectivity (timing of when fishing took place) was different (Figure 12).

The model includes 3 selectivity periods and the focus for projections is on the recent selectivity period. The integrated model also accounts for changes in effort and fishing power in estimating catches, as well as changes in spawning biomass/stock size due to natural drivers and impacts from fishing (the model estimates of fishing mortality per week are used as diagnostic tests).

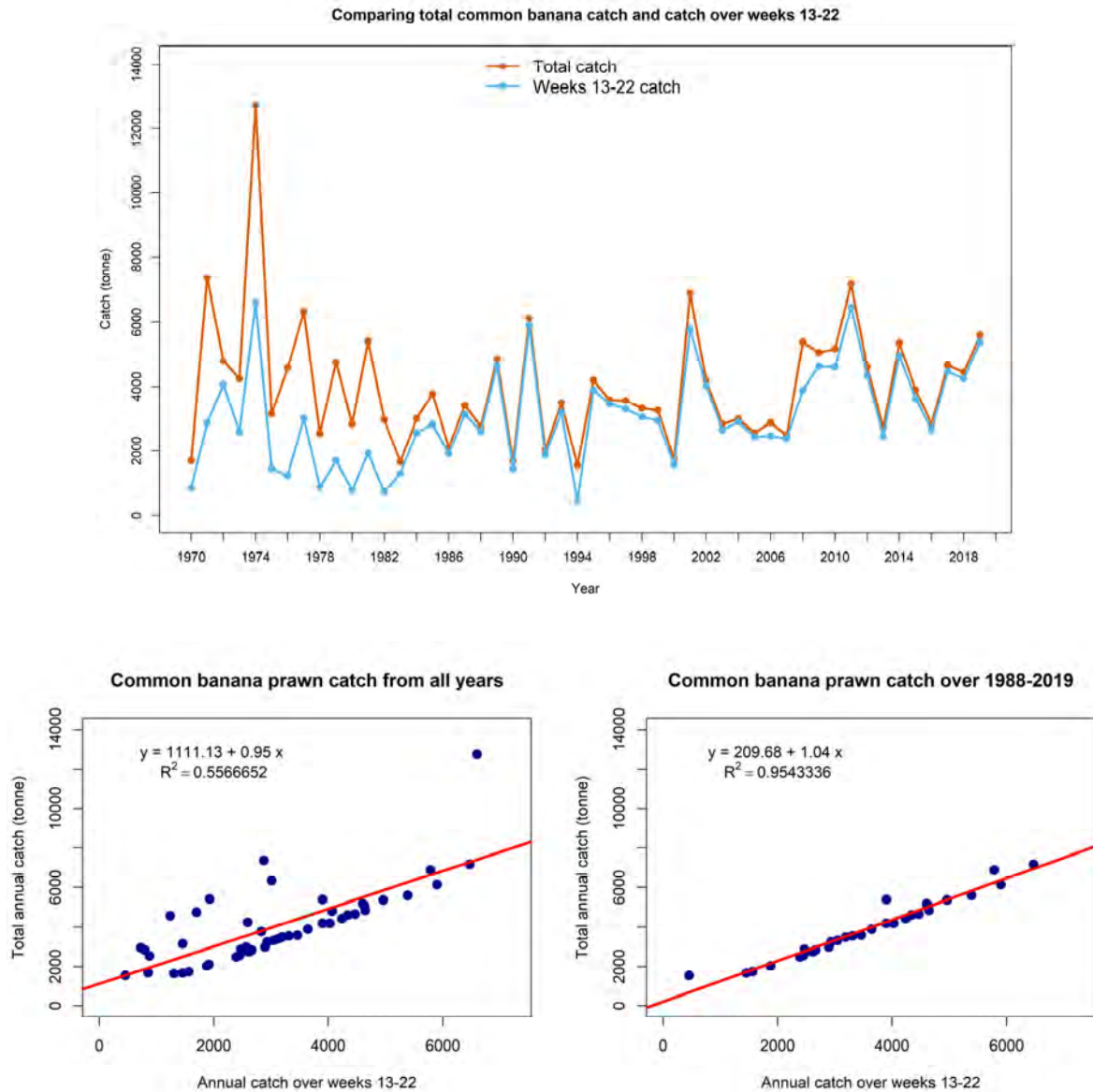


Figure 12. Comparison of the total common banana prawn *P. merguensis* catch over the period 1970 to 2019 when summed over reference week period 13-22 compared with over the entire year. The lower plots with fitted regression lines show how well the annual catch over weeks 13-22 predicts the total annual catch when (left) considering all years, and (right) over the period 1988-2019

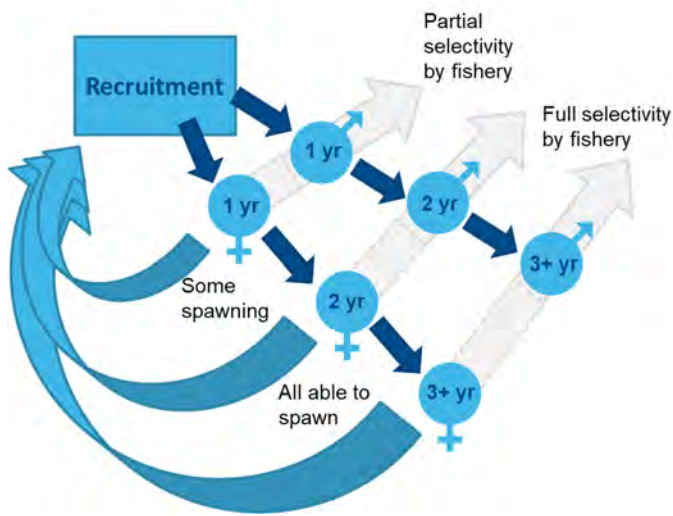
Species modelled using a monthly time-step

A summary of the age-structured approach used for mud crabs, barramundi and largemouth sawfish is given in Figure 13. Mud crabs are the only species in the MICE which includes separate representation of the male and female dynamics, which was necessary because the Qld mud crab fishery only takes males (Figure 14). However, as barramundi are protandrous hermaphrodites (Davis and Kirkwood 1984, Campbell et al. 2017), the MICE accounts for them changing sex from male to female at around 5 to 7 years, with the relative proportions of the two sexes used to modify the stock-recruitment relationship (see Appendix 12). There are a number of other differences between the NT and Qld fisheries management systems which have been taken into account in the MICE (Figure 14).

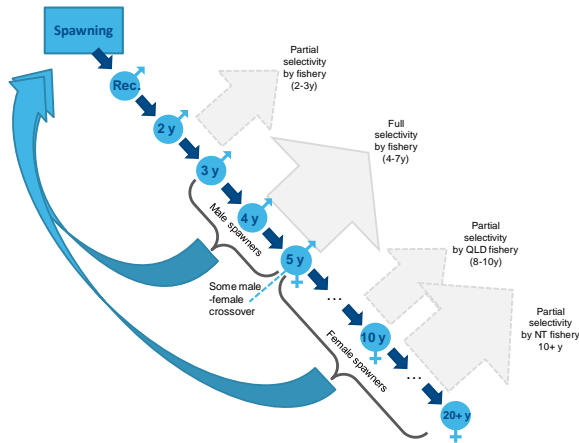
Table 3. Summary of MICE species or groups, together with time-step used in modelling their dynamics, form of equation and relative data availability. Species not discussed in detail in this report are shaded in grey.

No.	Model species/group	Time step	Equations	Data availability
1	Common banana prawns – <i>Penaeus merguensis</i>	weekly	Age-structured	High
2	Grooved tiger prawns – <i>Penaeus semisulcatus</i>	weekly	Age-structured	High
3	Brown tiger prawns – <i>Penaeus esculentus</i>	weekly	Age-structured	High
4	Giant mud crabs – <i>Scylla serrata</i>	monthly	Age-structured	Medium
5	Barramundi – <i>Lates calcarifer</i>	monthly	Age-structured	Medium-high
6	Largetooth (Freshwater) Sawfish – <i>Pristis pristis</i>	monthly	Age-structured	Low
7	Narrow Sawfish – <i>Anoxypristis cuspidate</i>	monthly	Age-structured	Low
8	Microphytobenthos	weekly	Lumped Biomass	Low
9	Meiofauna	weekly	Lumped biomass	Low
10	Seagrass	weekly	Lumped biomass	Very low
11	Mangroves	weekly	Lumped biomass	Very low

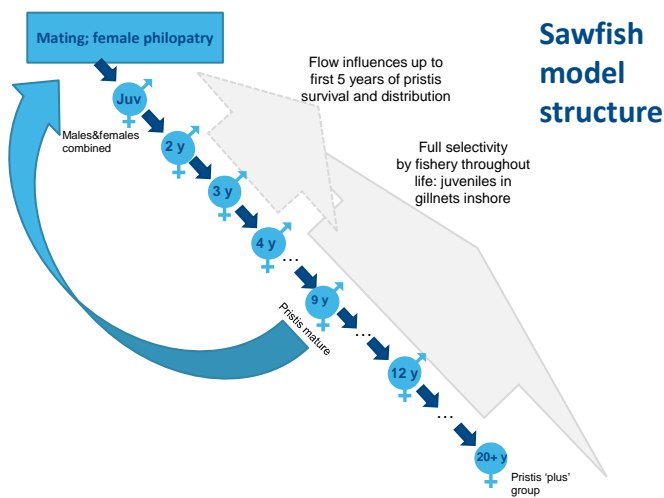
(A) Mud crabs



(B) Barramundi



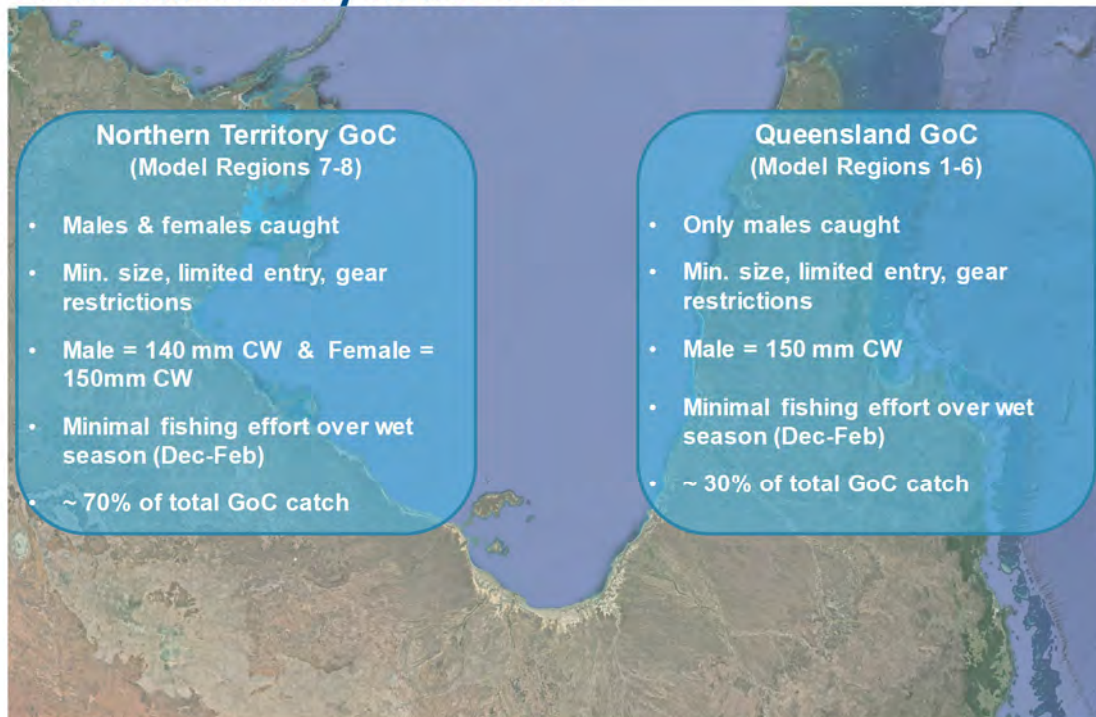
(C) Sawfish



Sawfish model structure

Figure 13. Schematic summary of the representation of (A) mud crabs; (B) barramundi and (C) largemouth sawfish in each spatial region of the MICE. The combined sex symbol indicates that the model is not disaggregated by sex in these examples.

Mud crab fishery in the GoC



Barramundi fishery in the GoC

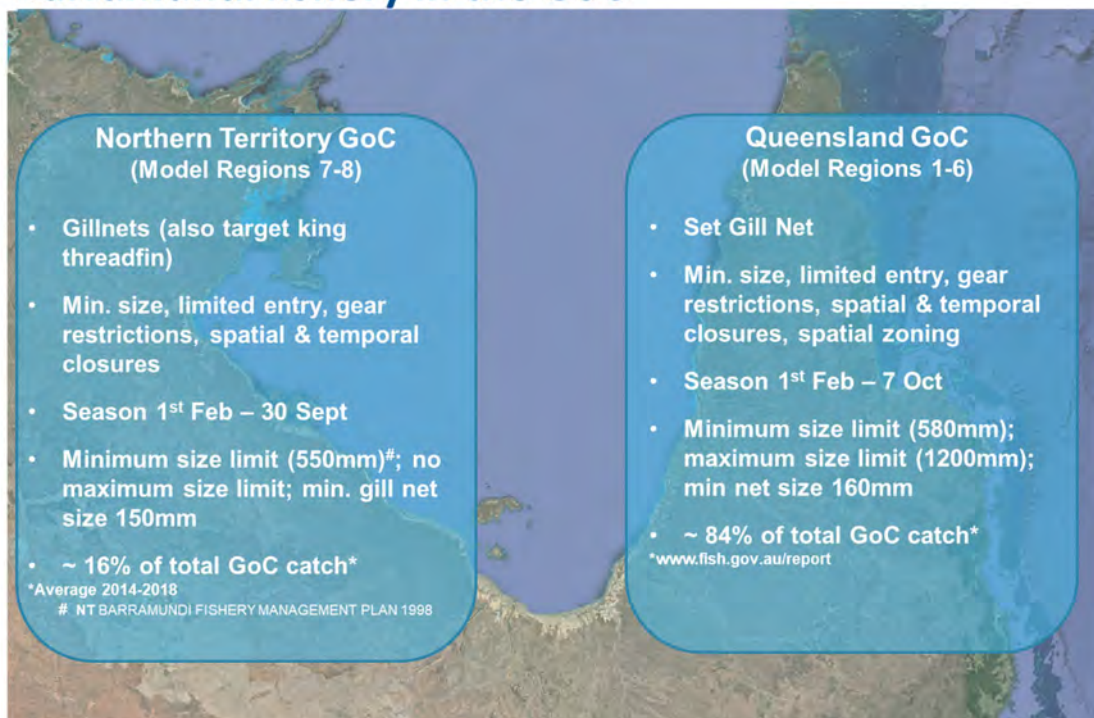


Figure 14. Summary of key management differences pertaining to (A) the mud crab fisheries and (B) barramundi fisheries based in the Northern Territory (MICE regions 7-8) compared with Queensland (MICE regions 1-6). Note that for the NT, the barramundi fishing area is restricted to waters seaward from the coast, river mouths and legislated closed lines (NT Department of Industry, Tourism and Trade, barramundi fishery management plan 1998).

Modelling of habitat and base productivity groups

Characteristically for Australia's tropical estuaries, each year over the last four months of the dry season (September to December), the estuaries provide stable habitats for the settlement and benthic recruitment of juvenile fish and crustaceans following annual reproductive events. Estuarine macrophytes (mangroves, seagrass and algae) provide the juvenile phase protection from predation (Kenyon et al. 1995, Vance et al. 2002), while microphytobenthos sustains primary production and estuarine foodwebs (Burford et al. 2012, Burford et al. 2016, Duggan et al. 2019). After occupying the stable estuary to survive and grow, the onset of the wet season changes the estuarine environment and the habitats available to estuarine residents. Freshwater inflows transport sediments, deliver nutrients, lower salinities, create mechanical forces and scouring, inundate over-bank habitats, connect ephemeral habitats, and modify tidal inflows. Each species has evolved to use one or more of these annual opportunities to enhance their population or move to a habitat suitable for the next phase of their life history. The timing and volume of wet season flows and floods are critical to estuarine conditions and species response to the monsoon-driven river flows.

Seagrass modelling

Seagrass are the base of the food web and a key food source in the tropics where phytoplankton biomass is extremely low (Adey 1998). They are the main diet of dugongs and green turtles which were beyond the scope of this study to model, and provide a habitat for many marine animals, including commercially important species such as prawns. They also help to keep coastal waters clear as stabilise sediment and absorb nutrients from coastal run-off. Seagrass and algae also provide critical nursery habitat for juvenile tiger prawns *Penaeus esculentus* and *Penaeus semisulcatus* (Haywood et al. 1995, Kenyon et al. 1997b, Loneragan et al. 1998, Dichmont et al. 2007, Haywood and Kenyon 2009). Seagrass are also important contributors to CO₂ mitigation and carbon storage, potentially accounting for as much as 30% of marine net primary productivity (NPP) that is buried in sediments (Duarte and Cebrián 1996).

The GoC has high seagrass species diversity and although we acknowledge that species-specific differences are important, we are unable to reliably capture these effects in our model and instead used a very simple model that represents combined seagrass biomass in each region of the GOC. Tropical seagrass dynamics are strongly influenced by both long term weather patterns and extreme flood and cyclone events (Carruthers et al. 2002). Other important drivers include turbidity and other restrictions to light availability, as well as temperature and inorganic nutrients (Lee et al. 2007, Collier and Waycott 2014). Salinity variations can affect the reproduction and distribution of seagrass species (Short et al. 2001) and is thus also included in the model (see Appendix 15).

Although there are several factors, including daily and seasonal variability, that will influence light attenuation affecting seagrass (Ralph et al. 2007), plus noting that different species have different sensitivities (Lee et al. 2007), here we use a much simpler starting relationship which consider only the average weekly solar exposure per area and major influences on coastal turbidity due to floods bringing additional sediments as well as cyclones mixing the water column and increasing turbidity. Turbidity and light attenuation are linearly related (MacDonald 2015). Here we assume also that river flow and turbidity are linearly related (a sensitivity to alternative formulations, such as assuming a step change in the relationship, could also readily be investigated). Hence our model assumes that seagrass light availability is inversely related to flow as well as inversely (linearly) related to cyclone category. We multiply the relative solar exposure by this value as shown in the equations below.

The seagrass additional mortality due to cyclone damage is modelled as directly proportional to cyclone category in the region corresponding to the landfall area, but we also multiply this by the evaluation of the cyclone impact.

We found few suitable data sets to validate our seagrass model component, and future work will seek to improve our initial modelling. However, we used available past data such as that described in

Kenyon et al. (1997a) who conducted a one-year study on the seasonal growth of *Enhalus acoroides* at Groote Eylandt, which grows in association with *Cymodocea serrulata* or *Thalassia hemprichii*. These seagrass beds support diverse communities of juvenile fish and decapods and provide macrofauna with food and shelter (Blaber et al. 1992, Brewer et al. 1995, Kenyon et al. 1995). We assessed the extent to which our model replicates observed patterns of seasonal growth in seagrass (Kenyon et al. 1997a), the response of seagrass to cyclones such as described in the Poiner et al. (1993) study which showed that cyclones such as 'Sandy' in 1985 had a severe effect on seagrass, plus we compared model results with data from the Karumba (model region 4) September long-term seagrass monitoring available from 1994 to 2019 (Scott and Rasheed 2021). Further detail is presented in Appendix 15.

Mangrove Modelling

Mangrove forests are a key nursery area for many species of fish and crustaceans (Robertson and Duke 1987, Vance et al. 1990, Vance et al. 2002), provide refuges from predators (Vance et al. 1996) and provide important food sources for the rest of the ecosystem (Robertson and Alongi 1992, Robertson and Blaber 1993). Common banana prawns, giant mud crabs and barramundi are all mangrove-associated commercial species (Manson et al. 2005). Posited reasons for the relationship between common banana prawns and mangroves include access to food sources, gaining protection from predators, and the favourable hydrodynamic retention capacity of mangroves (Vance et al. 2002, Nagelkerken et al. 2008, Vance and Rothlisberg 2020).

Mangroves provide essential ecosystem services but are at risk as have been declining worldwide with losses estimated at around 30-50% over the last century (Duke et al. 2007, Friess et al. 2019). The significant role of carbon sequestration in mangrove systems contributing to climate change mitigation has also recently been recognised (McLeod et al. 2011, Wylie et al. 2016) and approaches have been developed to support a blue carbon accounting method for Australia (Lovelock et al. 2022). We therefore included this vital habitat in our MICE, recognising that there are few suitable data to validate our model but that it would nonetheless be useful to make a start at developing a large-scale model capturing the key drivers of mangrove dynamics.

Similar simplifying assumptions are made when modelling mangroves as for seagrass. Relative changes in mangrove biomass are modelled as a function of changes in key drivers, namely flows, solar exposure, minimum sea level, air temperature and cyclone damage. The dependence of mangroves on salinity and water availability is modelled by assuming that the growth rate declines (relative to the maximum rate) when the salinity is too high (low flows used as a proxy) or water availability too low (flow, rainfall and sea level used as proxies).

Duke et al. (2019) confirmed that vegetative cover of mangrove-saltmarsh tidal wetlands increases with increasing average annual rainfall. Periodic changes in rainfall trends (and hence flow) trends can result in encroachment or dieback of mangroves. Our mesoscale mangrove community model component therefore assumes that the carrying capacity K of the mangroves in each model region varies linearly as a function of average annual flow. Asbridge et al (2016) showed seaward expansion of mangroves is due to prolonged inundation of low-lying coastal zone from tidal flows and floodwaters which pulse sediments to nearshore. Persistent freshwater inundation on the landward margins during the wet season made conditions more suitable for mangroves by reducing salinity and allowing the forests to compete with saltmarshes and expand landwards.

We found few suitable data sets to validate our mangrove model component, and future work will seek to improve our initial modelling (see also Section 23 on model caveats and further detail in Appendix 15). We compared our model results with the empirical study of Conacher et al. (1996) on three main mangrove communities in the Embley River Estuary from March 1993 to March 1994. During the Conacher et al. (1996) study period, the wet season was shorter than usual (December to March compared with November to May in some years) and rainfall was below average which resulted in environmental stress to the mangrove species. They used the rate of stipule fall of

Rhizophora and *Ceriops* to estimate the number of leaves produced, and we considered this a rough proxy of mangrove growth rate. The comparison between model outputs and observations is not straightforward but the model does capture some of the variation in mangrove growth rates, with the variability driven largely by changes in river flow. Future work can seek to improve validation of the mangrove model as this is considered uncertain.

The Asbridge et al. (2016) study suggests that the mangrove community has been accreting seaward over recent decades which is attributed to the combined effects of sea level rise and prolonged periods of tidal and freshwater inundation on coastal lowlands. Other studies have corroborated this finding (Eslami-Andargoli et al. 2009, Duke et al. 2019). It therefore seems reasonable to model mangrove growth as dependent on river flows, noting also that any reduction in flows would reduce the sedimentation that the mangroves are colonising.

Estuarine microphytobenthos and meiofauna and habitat use by juvenile common banana prawns

Australia's tropical estuaries are net-autotrophic with mangrove forests, water column plankton and epibenthic microalgae driving primary productivity (Burford et al. 2008, Burford et al. 2012, Duggan et al. 2014). Primary production in many Australian tropical estuaries is primarily supported by terrigenous inputs via riverine transport (Burford et al. 2012, Burford and Faggotter 2021).

In GoC estuaries, microphytobenthos (i.e. benthic microalgae) on mud- and sandflats contribute substantially to estuarine productivity (Burford et al. 2012; Burford and Faggotter 2021). In these tropical estuaries, the microphytobenthos feeds animals living in the mud, such as macrobenthos, that then feed common banana prawn juveniles (Duggan et al. 2014, Lowe et al. 2021).

Studies have shown that floods bring nutrients into estuaries and the nearshore, along with freshwater. In the short term, the freshwater reduces productivity, but after the salinity increases again at the end of the wet season, the nutrients are critical for fuelling primary productivity in the water and on the mudflats throughout the dry season (Duggan et al. 2014, Burford and Faggotter 2021, Lowe et al. 2021). Additionally, saltflats adjacent to the mangroves also provide a source of nutrients (Burford et al. 2016) (Burford et al. 2016). They are inundated only on the highest astronomical tides and during medium to high flow years.

Recently-settled postlarval and small juvenile penaeid prawns occupy a range of littoral habitats among vegetated and bare-substrate habitats. They forage on herbivorous and omnivorous food sources among the seagrass and estuarine mangroves that provide them shelter, refuge and sustenance (Wassenberg and Hill 1987, 1993, O'Brien 1994, Heales et al. 1996). Filamentous algae and unidentified plant detritus is common in the guts of both common banana prawns in estuarine mangrove forest habitats and tiger prawns in seagrass habitats. The proportion of animal food source increases with juvenile prawn size (Wassenberg and Hill 1993, O'Brien 1994), although small juvenile prawns (<~10 mm CL) commonly eat plant material. Both microphytobenthos and meiofauna in the mangrove forest and bare mud substrates within estuaries provide food resources for benthic postlarval and juvenile common banana prawns (Wassenberg and Hill 1993) and these resources are abundant on the mud-substrate banks of tropical rivers flowing into the Gulf of Carpentaria (Burford et al. 2012, Duggan et al. 2014, Burford and Faggotter 2021).

Juvenile common banana prawns forage on rising and high tides, particularly at night (Wassenberg and Hill 1993), presumably scouring the newly-submerged substrates as the tide floods. Both microphytobenthos and phytoplankton detritus are prolific on estuarine mudbanks, both within the mangrove forest and on the intertidal banks (Burford et al. 2012). These same habitats also support meiofauna such as copepods and nematodes (Duggan et al. 2014). The abundance of plant and animal food resources in these habitats demonstrates why tributary creek/mangrove forest habitats within coastal estuaries are the optimal habitat for recently-settled benthic and subsequent juvenile populations of common banana prawns (Vance et al. 1998, Vance et al. 2002).

Common banana prawn postlarvae recruit from their offshore planktonic stage to become estuarine benthic postlarvae and juveniles from about September to March annually. The critical recruitment time window to support the April prawn fishery is roughly September to January, as it gives the juvenile prawns time to grow within the estuary, then emigrate offshore and grow as sub-adults to become a commercial resource on the fishing grounds. September to December is the mid-to-end of the dry-season, with little to no significant rainfall. Ninety percent of rainfall in the wet/dry tropics occurs during January to March (Petheram et al. 2008, Petheram et al. 2012) and high river flows occur during this period (~80% of flows in 50% of rivers) and April.

During the recruitment time-window for juvenile common banana prawns, estuaries are stable autotrophic environments with only tidal currents suspending sediments and delivering nutrients. Small common banana prawns forage on meioflora and meiofauna within the mangrove forest/mudbank matrix in the upper tributaries of the estuarine creeks and rivers (Vance et al. 2002). The pelagic and benthic habitats support the plankton, microphytobenthos and mangroves that fuel the estuarine foodchain. Although effects on survivorship and population size due to food limitation has not been tested experimentally, there is no evidence that juvenile common banana prawn populations are limited by plant and animal resources on which to forage. High mortality due to fish predation is the major cause of population decline (Wang and Haywood 1999). Low-level river flows during the late dry-season enhance estuarine conditions by lowering salinity which often is hypersaline by October/November. Under brackish conditions, the micro- and meio-floral and faunal communities within the estuarine habitats flourish, prior to stressors associated with high-level freshwater floods (Burford et al. 2012, Duggan et al. 2014).

Recruitment is continuous to the estuary (Vance et al. 1998) and despite high mortality of juvenile prawns due to predation (Wang and Haywood 1999), the population of juveniles remains high throughout the late dry-season and into the wet-season (Staples 1980a, Vance et al. 1998). High rainfall during the annual monsoon season provides the freshwater emigration cue (riverine floodwaters) that are critical to sustain an abundant offshore common banana prawn spawning population. Volume of flow is the key determinant of resultant common banana prawn population in the nearshore and offshore zones within the GoC (Staples 1980b, Staples and Vance 1986, Vance et al. 1998, Duggan et al. 2019, Broadley et al. 2020). In addition, an estuarine freshwater habitat reduces the food resources of juvenile common banana prawns, and acts in concert with the floodwater emigration cue (Duggan et al. 2014). The larger the flood, the stronger the emigration event from the estuarine habitats to the nearshore zone (Staples 1980b, Staples and Vance 1986, Vance et al. 1998). In the nearshore, the floods deposit nutrients and sediments that probably enhance the food resources and survival of sub-adult prawns in the deeper waters they occupy as emigrants (Burford et al. 2012).

Importantly, once the freshwater influence within the estuaries subsides, microphytobenthos and meiofauna within the sediments recover to levels equal to pre-flood conditions and remain high for months (Duggan et al. 2014, Burford and Faggotter 2021, Lowe et al. 2021). The results demonstrate that inputs of nutrients and sediments are critical for primary and secondary productivity within the estuary during the subsequent dry-season (Burford and Faggotter 2021).

Estimating catch as a function of river flow

For some MICE components such as tiger prawns, the actual historical catches (after application of the species-splitting algorithm (Venables and Dichmont 2004)) are used directly in the population equations. The model is then fitted to available survey and/or CPUE data, and usually results in estimation of recruitment residuals. Additional analyses can then be used to try and explore whether part of the observed variation in recruitment can be explained by environmental drivers (Plagányi et al. 2019).

On the other hand, for the model groups common banana prawn, mud crabs and barramundi for which we have some confidence that recruitment (to the fishery) is driven in part by flows, we use a different

approach in the MICE. We use fishing effort (rather than catch) as an input to the model, and do not directly use the CPUE data. The model predicts the catch (either weekly or monthly depending on species) based on the fishing effort together with one or more catchability coefficients, which may be time-varying based on river flows for example. Hence the basic catch equation to predict catch is as follows (see Appendix 16):

$$\widehat{C}_{r,y,w}^{sp} = q_{seas}^{sp} qf_y^{sp} E_{r,y,w}^{sp} B_{r,y,w}^{exp-sp} \quad (3)$$

where $\widehat{C}_{r,y,w}^{sp}$ is the model-estimated catch for the species sp in region r , year y and week w (which is substituted for month m for barramundi and mud crabs);

q_{seas}^{sp} is the catchability coefficient for season seas for common banana prawns and which may vary per region r and year y and for mud crabs depends on river flows;

qf_y^{sp} is the fishing power (relative efficiency increase) per year y for species sp ;

$E_{r,y,w}^{sp}$ is the observed fishing effort in region r , year y and week w ;

$B_{r,y,w}^{exp-sp}$ is the commercially available or exploitable biomass (which depends also on selectivity) in region r , year y and week w .

The model is therefore fitted to the observed catches using standard log-likelihood fitting methods. Rather than estimating recruitment residuals, the MICE can be used to directly compute these based on a functional relationship with flow (and noting that the parameters of this relationship can then be estimated directly in the model fitting process). The next section summarises examples of the kinds of functional relationships that are assumed between recruitment and flow, noting that the term ‘recruitment’ is used here in the conventional fisheries stock assessment approach to denote those animals recruiting to the fishery (or some pre-defined age) for the first time.

If one assumes recruitment is density-dependent based on stock size, then one can straightforwardly use a Beverton-Holt or other stock-recruitment relationship to model recruitment. This would account for density-dependence in the stock recruitment relationship – for example, as the stock declines, there will be a decline in recruitment. Now rather than estimating stock-residuals to capture the observed variability about the stock-recruit curve, in the MICE we directly use the flow residuals – or some index based on these values as per examples in the next section – as the stock-recruit residuals. The variability about the average recruitment that would be expected at a particular stock size is therefore explained by the flow residuals. These therefore also implicitly incorporate the net outcomes of overall recruitment numbers and the survival of those young stages. The recruitment estimates in turn feed into the commercially available biomass estimates and therefore contribute to the biomass available to be caught by the fishery. In this way, the model is able to estimate parameters of a flow-recruitment relationship and evaluate whether the variability in flow contributes to the observed variation in catches.

Linking flow to biological components

For each of the key model species whose dynamics are known to be influenced by river flows (common banana prawns, mud crabs, barramundi, sawfish), we developed relationships to describe the influence of flow on one or more of the following: recruitment, survival, availability (to the fishery). The simplest assumption is that there is a linear relationship between flow and its impact on

these population parameters. But it was also recognised based on previous work and discussions during the project that there are likely lower threshold flow values below which responses become non-linear, whereas for very large flows, there is an upper limit in terms of their impact on a population. We therefore set lower and upper threshold limits for flow. Next, we improved these formulations by using a logistic function to describe the relationship between flow and population processes such as recruitment (Figure 14) (or parabolic function for mud crabs – see Appendix 11). Hence two parameters could be set or estimated to describe the shape of this relationship, such as whether the relationship is near-linear, or increases more steeply after some lower threshold level. Another approach used was to fix one of the logistic parameters, so that it was only necessary to estimate one additional parameter to describe the relationship for different river systems and/or different species Figure 15.

The relationship between flow and recruitment can thus be described using a logistic function as per the example for prawns (Appendix 16):

$$\sigma_{r,y,w}^{flow_psp} = \sum_r X_r^{catchment} \lambda_{r,y,w}^{psp} \quad \text{where}$$

$$\lambda_{r,y,w}^{psp} = \frac{1}{1 + e^{-\frac{(flw_{r,y,w} - thresM^{sp})}{thres^{sp}}}}$$

$$\text{with } flw_{r,y,w} = \frac{fl_{r,y,w}}{fl\bar{w}_{r,w}} \quad \text{and} \quad fl\bar{w}_{r,w} = \frac{1}{n_yr} \sum_{y=1970}^{2019} fl_{r,y,w}$$

$$X_r^{catchment} = 0 \quad \forall \text{ catchment regions } r \text{ except current catchment if } r=1 \text{ or } r=7 \quad (4)$$

$$X_r^{catchment} \text{ estimated flow influence in catchments 2-5}$$

$$\text{from each catchment } r = 2 \text{ to } 5$$

$$\text{and } \sigma_{6,y,w}^{flow_psp} = \sigma_{5,y,w}^{flow_psp} \quad \text{if } r = 6$$

$$\sigma_{8,y,w}^{flow_psp} = \sigma_{7,y,w}^{flow_psp} \quad \text{if } r = 8$$

Where $thresM^{sp}$, $thres^{sp}$ are shape parameters: for example in Figure 14, $thresM^{sp}=0.8, thres^{sp}=0.3$ compared with an alternative formulation as shown in Figure 16. For mud crabs a parabolic function is used instead as described in Appendix 16.

In all cases we computed standardised flow anomalies by dividing the cumulative weekly (or monthly) end of system river flow per region and per week of each year by the average flow for that region and week (or month), based on data from all years over 1900-2019. For mud crabs, barramundi and sawfish, we assumed population dynamics were influenced by the standardised flow anomalies for each corresponding model region where they occurred. For common banana prawns, we assumed the population dynamics were potentially influenced by a combination of flows from rivers in the south-east region of the GoC.

For common banana prawns, which spawn offshore, it is assumed that total spawning biomass is shared amongst spatial regions. Recruitment residuals are computed directly based on flow anomalies (i.e. not estimated as is often the case in stock assessment models) and the relative contribution on average of different rivers is estimated (with associated STD) by fitting to the observed catch data (1971-2018)

$$\text{Recruit_residual}(wk,yr,area) = a \times \text{Mitchell_flow} + b \times \text{Gilbert_flow} + c \times \text{Norman_flow} + d \times \text{Flinders_flow}$$

Where flow variables are cumulative weekly end of system river flow per area and per week of each year. This is standardised to flow anomalies by dividing by average river flow, and this is also then

converted to actual impact on recruitment (defined as recruitment to fishery) after transforming using linear or logistic curve. This provides a rigorous basis for quantifying these relationships for forward projections under alternative scenarios.

It was also noted that in high flow years, there may be strong correlations between the Norman and Flinders Rivers which essentially become one big lake (Burford pers comm). The Gilbert River may also overflow into the Norman River catchment.

We therefore also tried alternative models that included this effect as an interaction term, i.e.

$$\text{Recruit_residual}(\text{wk}, \text{yr}, \text{area}) = a \times \text{Mitchell_flow} + b \times \text{Gilbert_flow} + c \times \text{Norman_flow} + d \times \text{Flinders_flow} + e \times \text{Norman_flow} \times \text{Flinders_flow}$$

The preliminary model wasn't able to estimate these additional parameters satisfactorily, but this warrants further investigation given the known mixing of floodwaters. However this is a secondary effect that is more relevant to high flows than low flows as is the focus here.

The Embley River was assumed to influence the local region only (Region 1), whereas the Roper River was assumed to influence model regions 7 and 8, hence simpler equations were used for these model regions:

$$\text{Recruit_residual}(\text{wk}, \text{yr}, \text{Embley}) = f \times \text{Embley_flow}$$

$$\text{Recruit_residual}(\text{wk}, \text{yr}, \text{Roper}) = g \times \text{Roper_flow}$$

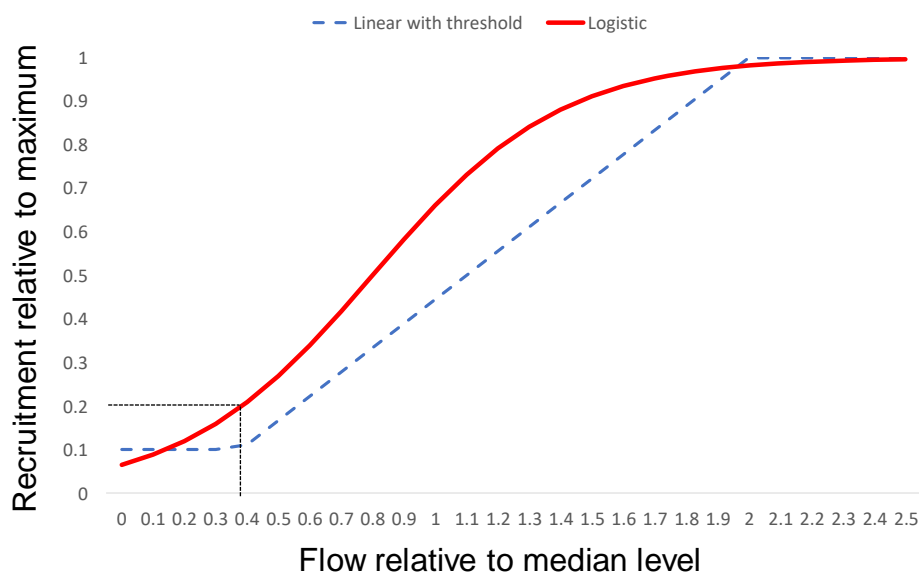


Figure 15. Schematic showing MICE logistic (or linear) function used to describe the relationship between the standardised flow and a population parameter such as recruitment, where the function yields a multiplier that describes recruitment relative to the maximum value.

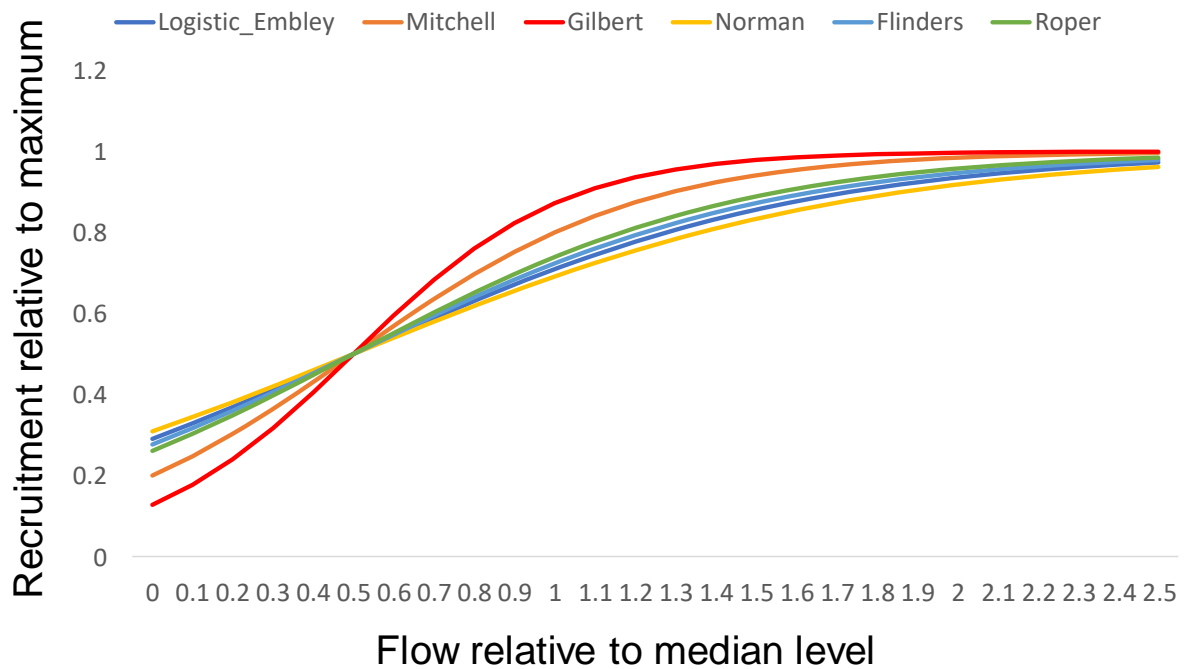


Figure 16. Schematic showing MICE logistic function used to describe the relationship between the standardised flow and a population parameter such as recruitment of prawns or barramundi, where the function yields a multiplier that describes recruitment relative to the maximum value. In this example, one of the parameters is fixed such that there is a single free parameter that can be estimated to better describe the relationship for different model regions and/or species, noting that the estimated curve is based on flow anomalies. For mud crabs, a parabolic function is used, see Appendix 11.

8. Bio-economic modelling

The Northern Prawn Fishery (NPF) is a multispecies fishery that operates over two seasons. In the first season (late March/early April to mid June) common banana prawns are targeted and in the second season (August to November) two species of tiger prawns and endeavour prawns are targeted (Larcombe and Bath 2017). Although harvests of these species are predominantly sequential, they do overlap. Other prawn species are also landed (in less quantity) as well as byproduct species. Economically, the common banana prawn fishery has provided, on average over half (55%) of the Gross Value of Production (GVP) for the NPF over a recent 11-year time period (2006/7 till 2016/2017) (Mobsby et al. 2018). The common banana prawn sub-fishery is currently managed by an in-season Maximum Economic Yield (MEY) catch rate trigger (Buckworth et al. 2014).

Given the known relationship between river catchment end-of-system flow and common banana prawn production, the assumption has always been that any changes to flow regimes and the timing thereof would impact on the common banana prawn fishery only. This is not the case as reduced catches of common banana prawns could shift effort on to the tiger prawn fishery as fishery participants seek to maximise their return on investment the tiger prawn fishery (Pascoe et al. 2016). Only a full re-specification of the tiger prawn fishery bio-economic model (Punt et al. 2010) would allow for the impacts on the whole NPF to be quantified. Previous derivations of bio-economic models of the NPF fishery have included both the banana and tiger prawn stocks; however these models were used to assess optimal season closures (e.g. (Somers and Wang 1997)) or optimal fleet size (Haynes and Pascoe 1988). A revised bio-economic model (Hutton et al. 2022) that selects effort levels to maximize the net present value (NPV) of the flow of profits over time (to 2050) for catches

of common banana, tiger, and endeavour prawns in order to calculate the MEY across the fishery over both seasons has been developed. We applied this revised model (Hutton et al. 2022) to estimate the potential economic impact of WRD scenarios on the whole fishery. The revised bio-economic model included: (i) a depletion model for the common banana prawn season as per Zhou et al. (2015) and (ii) the stock dynamics and economic components of the tiger prawn fishery as per the current assessment (Punt et al. 2011), within a bio-economic model that maximises profit over the whole fishery (Hutton et al. 2022). Below we provide a brief summary of the bio-economic model, including the fishing effort and price dynamics. We then outline how changes in profit and NPV were evaluated under four key WRD scenarios.

Bio-economic model and stock assessment

The current tiger prawn assessment uses a multispecies, weekly sex- and size-structured population model for each tiger prawn species and a Bayesian hierarchical biomass production model to the blue endeavour prawns. The results of the biological assessments are combined within a bio-economic model that calculates profit and hence the levels of effort by fishing fleet needed to achieve MEY (Punt et al. 2011, Deng et al. 2015). The NPF is almost globally unique in basing management advice on dynamic optimisation of MEY, which is in turn only possible as a result of considerable investment – in terms of data, stakeholder engagement and intellectual advances. Our study was therefore able to build on this foundation, but no similar in-depth data or analyses were available to inform on economic outcomes under WRDs for the other species in the model.

This approach requires predictions of the weekly pattern of fishing effort directed at tiger prawns, and assumptions about changes over time in costs and prices. The output from the bio-economic model determines the Total Allowable Effort (TAE) recommendation for the tiger prawn fishery and the associated costs and prices are only related to the tiger prawn component of the fishery.

The revised bio-economic model selects effort levels to maximize the net present value of the flow of profits over time (to 2050) for catches of common banana, tiger, and endeavour prawns to calculate MEY across the entire fishery over both seasons. Total effort each week is constrained by the number of vessels. The model allows the fleet (52 vessels) to potentially fish up to seven days a week (based on observed average fishing activity in the fishery), leading to a weekly effort limit of 364 boat days a week. Future catches are modelled based on forward projections of the assessment models in the case of tiger and endeavour prawns (as described above), with future stocks of common banana prawns included through a series of simulated alternative scenarios, with options for evaluating short-term impacts of changes in common banana prawn biomass or longer-term declines in common banana prawn biomass associated with WRD scenarios. The objective is to maximize total discounted profit (II) (i.e., net present value, or NPV) given the time-trajectory of effort by métier, accounting for contributions from common banana, tiger, and endeavour prawns.

Common banana prawn stock and effort dynamics within season

The common banana prawn component of the fishery is essentially a depletion fishery, with catch rates (and hence revenue per day) declining over the season (Pascoe et al., 2018). The available stock is fished down until it is no longer economically viable to continue fishing. That is, the point at which the marginal revenue per unit of effort (which decreases as the stock decreases) falls below the marginal cost of fishing. This is the principle underlying the trigger catch rate used to manage the fishery (Pascoe et al. 2018). This point will be reached by different vessels at different points in time as each vessel has different fishing costs. Further, the potential to fish for tiger prawns also introduces an additional opportunity cost of fishing for common banana prawns (Pascoe et al., 2016). Consequently, the number of vessels fishing for common banana prawns, and their related fishing effort, decreases over the season (Pascoe et al., 2018).

The heterogeneity in individual vessel fishing costs is not captured in the bio-economic model, so an effort allocation model was developed from the observed effort each week over the season. Effort declines as biomass declines, and catch per unit fishing effort and hence revenue per day would also

decline. Previous studies have found that the level of fishing effort applied to common banana prawns was a function of initial biomass and the relative price of banana and tiger prawns (when tiger prawns were available for fishing given the management constraints of the tiger prawn season only commencing in August) (Pascoe et al., 2016), thus price is included in the model. For the weeks when tiger prawn fishing is also available, some effort is diverted to tiger prawns, based on the relative prices of tiger and common banana prawns (see Hutton et al. 2022). More vessels leave the common banana prawn sub-fishery as catch rates continue to decline towards the end of season one. This is consistent with profit maximizing behaviour and was observed empirically by Pascoe et al. (2018). The effort model in Hutton et al. (2022) explains 86% of the total variance in observed fishing days. In addition, effort in the common banana prawn métier is also constrained by the catch rate trigger, this is set to zero in the model effort if the catch rate (catch per unit of effort) is less than a critical value determined on the basis of prices and fishing costs (Pascoe et al., 2018).

Common banana prawn price dynamics

Since 2008, the main market for common banana prawns has shifted from an export-dominated market to domestic supply. When supplying the export market, common banana prawns from the NPF made up only a negligible share of global supply, and prices were effectively exogenously determined. However, the domestic market is considerably smaller, and changes in the supply of common banana prawns from the sub-fishery can not only affect the prices received in the NPF, but those received in other sub-fisheries. Data are extremely limited, but preliminary estimates of the impact of changes in quantity supplied on price can be made. Hutton et al. (2022) applied simple regression models of price against quantity that revealed an own price flexibility of between -0.18 and -0.38. That is, a 1% increase in quantity supplied results in between a 0.18 and 0.38% decline in prices received (and vice versa). However these estimates are preliminary due to limited data as noted. Given the uncertainty in the estimates of price flexibility, the model used to evaluate WRD scenarios was run without price flexibility included.

Model base year

As in Hutton et al. (2022), the base year for the bio-economic model used here to evaluate changes in profit and NPV under WRD is 2018. Although, the most recent tiger prawn assessment was conducted in 2020 the economic situation in that year was distorted due to COVID-19, both in terms of the level of applied effort and changes in economic conditions (i.e., reduced access to international markets and a decrease in fuel prices during the banana season of up to 30%) and impacts on price. No assessment was undertaken in 2019 or 2021 as the assessment is biennial.

WRD scenarios evaluated

Changes in profit and NPV for the NPF under four key WRD scenarios (WRD1-WRD4, Table 6) were determined. The change in profit for year 1 was estimated as this provides an indication of the immediate impact. The change in NPV provides an indication of the potential longer-term impact on the fishery although the potential change presented herein does not account for any adjustment in fleet size.

The revised bio-economic model includes a common banana prawn component for the entire NPF and although most of the catch comes from the GoC, there are also some areas outside the GoC that are fished. However, when assessing changes in profit and NPV, we ran the bio-economic model using the MICE ensemble predicted GoC average and minimum commercially available biomass under the various WRDs and assumed this applied to common banana prawns across the NPF. We then compared this to the profit and NPV obtained when the bio-economic model was run using an average of 5000t of common banana prawn biomass year as per Hutton et al. (2022) and with no WRD. Our economic calculations may have over-estimated the impact of regional declines in biomass because we assume an average effect across the whole common banana prawn fishery, i.e. including areas outside the GoC. However, it is challenging to predict what might actually happen in response to a regional decline and the extent to which the fishery might spatially redistribute its effort, and hence

we kept these analyses as simple as possible in the absence of further information. This was also because of uncertainty regarding impacts of hypothetical WRDs that are not currently accounted for in our model but which may influence the fishery in future. If there was a major structural shift in the fishery, it's possible this could offset some losses but there are no data and low certainty to include these considerations in our modelling.

9. MICE Modelling Process

The MICE modelling process is summarised in Figure 17a. A first step in the process involved consulting with stakeholders to identify and agree on key species/components and processes to consider building into the model. Data were then gathered, and population models developed for each component. These population models aimed to capture the key life history stages for each species / component, and biological components can be linked where relevant. The detail we included in each population model was dependent on the data available. For example, species for which there is a commercial fishery (prawn, barramundi, mud crab) had more data available to parameterise a population model than species that are not harvested (e.g. seagrass, mangroves). River flow and other relevant environmental drivers were then added to the model. This additional complexity to the model was added only as needed, in a step-wise process, and had to be driven by *a priori* hypotheses (Figure 177b). For example, we began with the simplest approach in which we used fishing effort to explain the observed catches. The primary environmental variable (in this case flow) was then only added to components for which there was an underlying hypothesis. In the same way, other drivers were only added if needed (i.e. improved the model fit to the data) and if hypothesis-based.

Where possible, the model components were then validated against timeseries of observed data, which helped to provide some confidence in the model developed. Observed data included indices of abundance such as catch, catch-at-age, catch-per-unit-effort. Often these data were only available for harvested species, but sometimes there may be snapshots of information available for others e.g. through scientific surveys. The preliminary MICE model was then presented and discussed with stakeholders through a series of workshops. This process provided an additional validation step in which expert advice was considered and used to help fine-tune the model. Following this step, the model developed was now able to replicate historical observations (in this case, fishery catches) and could be used to predict outcomes under different WRD scenarios. Additional MICE model versions were also created given uncertainty in model structure and parametrisation (five model versions in total) and similarly outcomes were predicted under WRD scenarios. These predictions are then quantified and risk to various fisheries assessed and presented to stakeholders. Following feedback from the stakeholders, final adjustments to the model scenarios can be made and key sensitivity tests run.

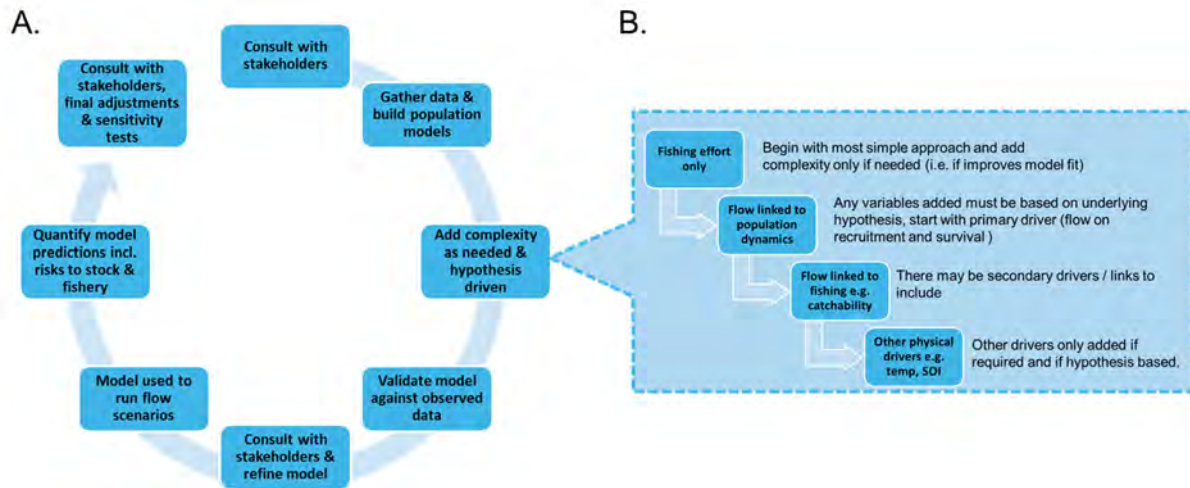


Figure 17. Summary of (A) the MICE modelling process and (B) adding complexity to the model

10. Model Fitting

Model parameter estimation was achieved using a staged approach and phased estimation of parameters. Fitting a complex ecosystem model with multiple species is not as straightforward as when fitting a stock assessment, but has the advantage that one is aiming for the best possible model given constraints plus trying to set up plausible alternative representations. As it is challenging and time consuming to fit ecosystem models, some parameter estimates estimated early in the process were fixed for later runs, and some parameters were held constant when refitting the model with different settings such as changes to flow relationships and depletion levels. Hence for example, for prawns, the initial non-spatial model version was used to estimate the total pristine (1970) spawning biomass and catchability parameters. Next we tried to not over-parameterise the model as although this might improve the model's ability to match fluctuations in data, there are insufficient data to support estimation of very many parameters. Thus, for example for common banana prawns we assumed the same catchability applies for all regions whereas for barramundi we assumed a common catchability applies to regions 3-6, a separate parameter to combined regions 7-8, and estimated separate catchability parameters for regions 1 and 2 because model results suggested different dynamics in these four broader regions. This means some compromise in the model's ability to exactly fit each of regions 3-6, but avoids the need to assume that catchability differs in sub-region 3-6 in the absence of additional information. On the other hand, for mud crabs we estimated a starting biomass for each region, but assumed a fixed common catchability for regions 1-6 in the Qld fishery and a different fixed common catchability for regions 7-8 in the NT fishery. Estimated flow parameters for mud crabs were also common between some regions e.g. 1-2, 3-6 and 7-8.

Not only did we apply common parameters for some regions to overcome the absence of information and over-parameterising of the model, but for some species there are also important jurisdictional differences between the Qld regions 1-6 and NT regions 7-8, which we needed to account for in our model. This was particularly the case for barramundi and mud crabs which have very different management regulations in the two jurisdictions (see Figure 14).

For most species we also drew on available information which suggested that the system dynamics of these four sub-regions (1; 2; 3-6; 7-8) can be expected to show differences. For example, the Mitchell River region differs in a number of respects from the Embley River to the north as well as the more southern rivers, plus the marine currents influence these regions differently (Wolanski 1993, Li et al. 2006).

There are also differences in other physical drivers of species population dynamics and ultimately catch between regions. For example, temperature (through a cumulative heat index) in addition to river flow was found to be important predictor for mud crab CPUE in the Roper River region (Robins et al. 2020), but not in the eastern GoC. On the other hand, mean sea level anomaly and cumulative rainfall (or an index of water stress) were the most important predictors of CPUE in the Flinders and Norman Rivers, and MJO and cumulative rainfall in the Staaten and Gilbert Rivers (Robins et al. 2020). Hence, we only included additional complexity (drivers of species abundance or catch) where necessary and for which we had underlying hypotheses (see section 9). This meant that models were slightly different between regions.

11. Ensemble of Alternative Models

Throughout the project, we conducted a number of sensitivity tests and implemented changes and improvements to the MICE based on feedback, but sticking to the philosophy of only adding extra complexity if it was sufficiently justified. As there was nonetheless a large amount of uncertainty associated with aspects of the MICE, we used an ensemble approach to generate final results so as to explore the robustness of the results to some key uncertainties. An ensemble approach uses two or more models with a similar but different set-up to account for uncertainty in e.g. model structure or parameterisation. This approach can help bound uncertainty in model outputs. Our ensemble comprised five model versions that variously included changes to model structure and parameterisation, loosely grouped into the following categories (Table 4):

Model 1: Base-case for reference purposes and considered the preferred model

Model 2: Alternative representations of flow-population dynamics representation

Model 3: Changes to production parameters such as growth rates, mortality or stock-recruitment steepness settings

Model 4: Changes to starting biomass or calibrated to change final depletion statistics

Model 5: System productivity settings such as whether prawn recruitment influenced by primary productivity, sea ratio assumed not to influence recruitment success in barramundi

A summary of the parameter estimates and corresponding likelihoods for each of the five key model versions is provided in Table 10. Further details on additional sensitivity analyses are provided in the Appendices for each species, but due to the practical limitations of so many results, we only show and discuss selected results, noting that preliminary versions of the MICE sub-models have been presented and reviewed at a number of project workshops.

Table 4. Summary of the MICE ensemble (five model versions) highlighting key differences between the model versions for each MICE sub-model. See Appendix 16 for full equations and model inputs values.

	Common banana prawn	Barramundi	Mud crab	Largetooth Sawfish	Seagrass	Mangroves
Model 1	Base model: estimate flow threshold parameters and river connection parameters for all combinations of Mitchell, Gilbert, Flinders and Norman Rivers	Base model: estimate M ; fix flow threshold parameters at same values as estimated for prawns Model 1	Base model: fixed $M = 0.1$; starting biomass estimated; parabolic flow relationship with fixed optimal flow and slope estimated; fixed steepness parameter $h = 0.6$	Base model: starting biomass tuned so sawfish currently heavily depleted as considered plausible	Base	Base
Model 2	Flow relationships modelled with no connections between systems so local effects only	Flow relationship parameters estimated for barramundi	Flow relationship parameter changed (optimal flow reduced by 20%)	Flow relationship parameters changed	Flows have bigger impact on light attenuation levels	Flow not assumed to influence growth rate but does influence carrying capacity
Model 3	Alternative lag between flow and effect on recruitment plus change 2 nd flow relationship parameter and estimate river connection parameters	Larger fixed M in combination with change to age-dependent slope parameter and flow relationship threshold value	Larger natural mortality M (20% increase)	Larger M assumed for sawfish but also higher juvenile survival and higher average pups per year	Larger mortality (1.5x) due to cyclones	Larger mortality (1.5x) due to cyclones

Model 4	Starting values proportional to recent rather than historical catch proportions	Assume more depleted in 1989 down to 20% instead of 30% in other versions	Model starting biomass doubled	Different starting numbers so current depletion estimates different	Pella-Tomlinson type production model used rather than Schaefer	Pella-Tomlinson type production model used rather than Schaefer
Model 5	Link to primary productivity index	No sex ratio influence on recruitment	Steepness parameter h reduced to 0.4	Larger boom-bust dynamics (0.6-1.4); change in threshold parameter	Growth rate reduced down to 80%	Growth rate reduced down to 80%

12. Sensitivity Testing Trophic Links

To ensure our model provides as rigorous a basis as possible for informing management decisions, we developed the model in a stepwise fashion and only added additional complexity where necessary. We therefore did not add all plausible trophic linkage in the MICE ensemble. However, here we present a sensitivity analysis with respect to explicitly representing the dependence of both barramundi and largemouth sawfish (predators) on common banana prawn (prey) abundance in the estuarine environment. In doing so, we assume that common banana prawns serve as an index of the overall productivity of penaeids and other smaller prawns. Other non-commercial species such as *Metapenaeopsis* spp. and *M. palmensis* are known to dominate the penaeid diet of fish species in the Gulf of Carpentaria (Salini et al. 1994) and the smaller *Acetes* sp., which are abundant in estuaries and fluctuate with flow cycles, are an important part of the diet of juvenile fish (JR, pers comm).

Barramundi and prey

In the Gulf of Carpentaria, barramundi rank with a few other fish species such as Spanish mackerel *Scomberomorus commerson* in being relatively abundant predators that eat relatively high numbers of juvenile penaeid prawns (Brewer et al. 1991, Brewer et al. 1995, Salini et al. 1998). There is a lot of variation in the abundance and composition of predator communities in different catchments of the GoC which has been attributed to variations in the diversity of habitat types – for example the Norman River estuary has high turbidity and mangrove-lined muddy bottom habitat (Blaber et al. 1989, Blaber et al. 1994, Salini et al. 1998). Salini et al. (1998) sampled a total of 2059 fish stomachs from the Norman River estuary during 1991-1992 from 54 predator species. They found that the giant queenfish *Scomberoides commersonianus* had the highest proportion of common banana prawns in its stomach, followed by barramundi, which had a percentage dry weight of 88.7%. Given several predators feed on juvenile common banana prawns in the GoC estuaries, and that other prawn species may also be important in the diet, we assumed in this model version that barramundi growth is influenced by common banana prawn abundance, but did not explicitly represent changes in the natural mortality of prawns due to barramundi. This will be considered in future model extensions that include a greater focus also on the other prawn species in the MICE.

Following Plagányi and Butterworth (2012), first we compute a predator-prey interaction term $f(B_{ar,y,m}^{prey})$ for each month m , year y and area ar , expressed in terms of common banana prawn prey depletion level $B_{ar,y,m}^{prey} / B_{ar,1970,m}^{prey}$ (see Appendix 16):

$$f(B_{a,y,m}^{prey}) = \frac{\beta^j \alpha^j B_{a,y,m}^{prey} / B_{a,1970,m}^{prey}}{\beta^j \alpha^j - 1 + B_{a,y,m}^{prey} / B_{a,1970,m}^{prey}} \quad (5)$$

Where the prey interaction parameter α^j for predator species j is computed from the relation:

$$\alpha^j = \frac{4H^j}{5H^j - 1} \quad (6)$$

Similar to the steepness parameter h (i.e. the parameter that controls the slope of the curve towards the asymptote) in stock-recruitment relationships, here the parameter H^j can be estimated or input and controls the shape (steepness) of the relationship between predator-prey net interaction outcome and prey abundance (Plagányi and Butterworth 2012). We scaled this relationship (hence the addition of scaling parameter β^j) so that the predator-prey interaction term is fixed at one when the prey

biomass is the same as the pre-exploitation 1970 biomass, but increases above 1 for larger biomass values. This is because the starting prawn biomass value is not a maximum but instead the biomass can be expected to fluctuate about this level for this highly variable prey species. We set β^j equal to 1.1 to constrain increases in predator growth rate to 10% in the first instance. As an example, Figure 18 compares how different settings for H^j influence how rapidly decreases in relative prey abundance start to impact negatively on a predator's reproduction, growth or survival, and conversely the increase in predator growth rate or productivity as prey biomass increases (noting this value tends to an asymptote).

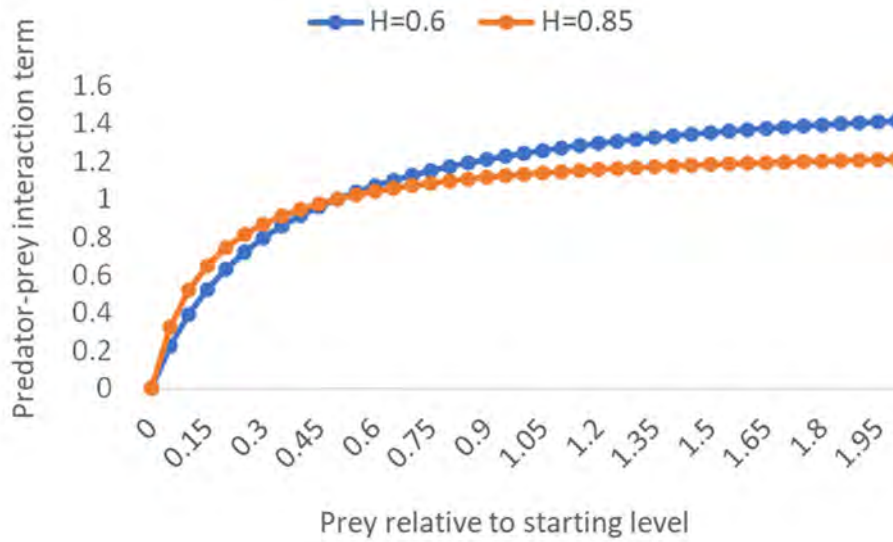


Figure 18. Comparison of the prey-interaction shape for different values of the steepness parameter H as shown.

For barramundi, we then assume that increases or decreases in common banana prawn prey modify the growth rate parameter k . This will in turn translate into the same number of fish having a larger or smaller biomass when caught, and similarly a larger spawning biomass.

The modified form of the von Bertalanffy curve used to calculate the length at age a , $l(a)$ is as follows:

$$l(a) = l_{\infty} \left(1 - \exp \left\{ -k * f(B_{ar,y,m}^{prey}) (a - a_0) \right\} \right) \quad (7)$$

Where the growth rate parameter k is set at 0.18 and the average asymptote parameter (i.e. average maximum length) l_{∞} is 1.297m, $a_0 = 0$, based on base case values for the southern Gulf barramundi stock fitted by Streipert et al. (2019) and growth is computed on a monthly rather than annual basis.

Length-at-age is then converted to mass-at-age using the relationship (Streipert et al. 2019)(see Appendix 16):

$$w_a = 0.0106(l_a)^{3.02} \quad (8)$$

Which in turn influences both the commercially available and spawning biomass computed at each time step in each of the model regions.

Largetooth Sawfish and prey

Stomach sampling of *Pristis pristis* suggested a varied diet comprising small teleost fish (freshwater, estuarine and marine species), *Penaeus* prawns, notably tiger prawns, as well as freshwater prawns (*Macrobrachium australiense*, *M. rosenbergi* and *M. handschii*) (Peverell 2005, 2010). Juvenile sawfish that move far upstream would not have access to common banana prawns in the estuarine nursery area, but when feeding in the estuarine environment, they are likely to feed on any available prawn species. Hence it isn't reasonable to assume any top-down control of estuarine common banana prawns by largetooth sawfish, but common banana prawns may in turn serve as an indicator of productivity available to sawfish. For illustrative purposes we simulate a dependence of juvenile sawfish survival and recruitment on common banana prawn abundance. We use a similar formulation as for barramundi to derive a predator-prey interaction term, but for largetooth sawfish we apply this term to the juvenile (age < 5 years) mortality rate $M_{r,y,m}^{sp}$ in month m of year y and in area r . Hence (see Appendix 16):

$$M_{r,y,m,a}^{sp} = \frac{\left[M_{base}^{sp} + \frac{M_{par}^{sp}}{a+1} \right] \cdot M_{pools}^{saw}}{f(B_{r,y,m}^{prey})} \quad \text{for saw with } a < 5 \text{ and if flow multiplier } \sigma_{r,y,1}^{flow_sp} < \text{thres}_{pools}^{saw} \text{ else } M_{pools}^{saw} = 1 \quad (9)$$

We did also test other non-trophic linkages e.g. mud crab abundance or survival linked with mangrove (juvenile mud crab) or seagrass (early benthic stage mud crab) habitat but this did not significantly improve our model fits and hence was not included in our MICE ensemble. However, there is scope to further explore this and other trophic linkages in the future.

13. Modelling Water Resource Development (WRD) scenarios

The initial river models developed as part of NAWRA and FGARA (see Methods Section 3) were updated where necessary for this project so that daily river flow (cubic meters per second) simulations were extended to 2019. River flow data for three river catchments (Gilbert, Flinders and Mitchell) that flow into the Gulf of Carpentaria were then modelled as a combination of water resource development (WRD) scenarios that affected natural river flows. Combinations of modified river flows from these three catchments were selected based on discussions with the project team and stakeholder input. These combinations were selected to illustrate the effects on downstream impacts from within-catchment WRD on a range of biota including fishery species, icon species and habitat-forming species. Combinations of water harvest quanta (allocation, GL yr⁻¹), river flow pump thresholds (TH, minimum flow rate before pumping can begin) and pump rates (PR, number of days to extract/pump out water) were modelled for the Mitchell River, together with WRD scenarios for the Flinders and Gilbert Rivers (Table 5).

A selection of WRD scenarios were modelled for the Mitchell River. A large array of possible WRD scenarios exist, including water harvest scenarios and impoundment behind constructed dam-wall infrastructure. Nineteen WRD scenarios were modelled (Table 6), however, for contrast, clarity and concise discussion, two water harvest quanta were described in detail (1000 GL per annum and 2000 GL per annum, both 85% reliability). Each WRD extraction was associated with selected river flow thresholds before which pumping cannot occur, and two pump duration variables representing the time taken in days for the water allocation to be harvested. These scenarios were compared with the historical baseline flows whereby only natural precipitation and resultant streamflows were evident, then projected into the future. Historical baseline flows were estimated using a catchment model

(using climate data) to determine runoff which then was applied to a river system model to estimate river flow (Hughes et al. 2018).

Within the Gilbert and Flinders Rivers, the selection of possible WRD scenarios were less diverse. For both rivers, historical baseline flows were estimated using catchment models (and climate data) to determine runoff which then was applied to a river system model to estimate river flow (Lerat et al. 2013). In the Gilbert River catchment, projected streamflow data following the placement of one or two dams were modelled (Petheram et al. 2013). The Green Hills Dam (227 GL capacity, 172 GL yield, 85% reliability) and the Dagworth Dam (498 GL capacity, 326 GL yield, 85% reliability) were the contenders. In the Flinders River, a streamflow model projected the changes to river flows due to the extraction of 160 GL of water per annum (~75% reliability) and 400 GL pa (~60% reliability) (Petheram et al. 2013a,b). No flow pump threshold (TH) or pump rate (PR) variables were added to the modelled WRD scenarios for the Gilbert and Flinders Rivers.

Within Australia's wet/dry tropics, the annual monsoon-driven river flows are highly variable and characterised by sequential low-level flows over some years. The 2000 GL pa extraction was characterised by 85% annual reliability (availability of water) (Petheram et al. 2018a) and was the largest extraction scenario modelled. The 1000 GL pa extraction was a lower level quantum of water extraction modelled; though not representing a negligible extraction.

River flow thresholds of 200 ML d⁻¹ and 1800 ML d⁻¹ were selected. River flow thresholds represent a level of down-river flow that would be allowed to pass pump-locations before pumped extraction would commence. Water pumping above a threshold of 200 ML d⁻¹ would extract water from low-flow streamflow levels. Low-level flows associated with late-dry season (early season) rainfall would be extracted and they would not reach downstream riverine and estuarine habitats. Dry-season habitat refugia in riverine pools may remain unconnected and estuarine habitats may not receive freshwater inputs to ameliorate a hypersaline estuary or provide upstream connectivity. In addition, in years of a poor monsoon and low rainfall, low-level wet season flows would be harvested. River flows are key drivers of connectivity and productivity in the wet-dry Australian tropics. The dynamic wet-season inputs to riverine and estuarine habitats after months of a stable, hot, dry, high-evaporation environment during the dry season would be debilitated. Following on from the low-level river flows, a large proportion of the trailing flows of a flood would be harvested.

Water pumping that began above a threshold of 1800 ML d⁻¹ would allow late-dry season flows to continue down river. Dry-season refuge riverine pools and some palustrine billabongs would be connected along the river. River flows would enter the estuary and re-establish connectivity allowing catadromous species to move upstream. A brackish ecotone within the estuary would benefit many estuarine species, especially the recent juvenile recruits of some marine species (Balston 2009, Alberts-Hubatsch et al. 2016). After the stress of near 40°C heat and evaporation during the dry season, low-level wet season flows continue down-river where they are critical to sustain riverine and estuarine habitat productivity. During strong wet seasons, river flows are high and support downstream productivity. However, in poor-monsoon years of low rainfall, all flows are critical to sustain ecosystem services in tropical catchments.

Pump rates were unitised as 15 or 30 days pump duration. This variable is a measure of the time-window required to extract the water allocation (1000 or 2000 GL pa). A pump rate of 15 days means it would take 15 days to pump the allocation of either 1000 or 2000 GL from the Mitchell River during each monsoon season. Likewise, a pump rate of 30 days means it would take 30 days to pump the water allocation each monsoon season. A pump rate of 15 days is linked to high capacity pumps that can harvest a volume of water in a short period of time. A pump rate of 30 days is linked to lower capacity pumps that take longer to harvest the same quantum of water. Pumping water over a short period would benefit ecosystem service provision that is supported by monsoon-driven river flows. Each year, the wet season river flows may be high-volume, but of limited, short duration e.g. floods. The ability to pump a large volume of water in a short period from peak flows of a large flood may have limited impact to the flood volume and duration. Only a relatively small proportion of a

high-volume flood peak would be removed. However, extended pumping would take a smaller proportion of the flood peak and as the flood declined, take a large proportion of the trailing flows that reduced their duration and downstream benefit.

A WRD scenario for the Mitchell River catchment used a 1000 GL pa water harvest combined with a flow threshold of 200 ML d⁻¹ and a pump rate of 15 days. This scenario represents water extraction from a broad range of river flows, including low-level floods, as extraction can occur at levels above a 200 ML d⁻¹ flow rate. Under this scenario, high capacity pumps would harvest the water over a relatively short duration of pumping - 15 days. To compare the effect of river flow thresholds, a comparable scenario also used 1000 GL pa water harvest together with a pump rate of 15 days. The contrast was provided by a river flow threshold of 1800 ML d⁻¹; a higher river flow level above which water harvest could occur. A higher river flow threshold would maintain downstream ecosystem services to riverine and estuarine habitats as low-level flows would continue downstream. Low-level floods establish long-stream and estuarine connectivity at the end of the dry season; replenishing riverine pool refugia after months of isolation, and allowing riverine/estuarine access.

In the Mitchell River catchment, the effect of different quanta of annual harvest was examined by contrasting two WRD scenarios, extraction of 1000 GL pa and 2000 GL pa. The 2000 GL pa scenario combined with a low river flow threshold (200 ML d⁻¹) and a long pump duration (30 d) to produce a scenario that was likely to strongly affect streamflows and the downstream ecosystem services that they provide. A higher level of extraction (2000 GL pa) would remove a larger proportion of the annual river flow volume. In addition, water would be extracted from the majority of flows, including low-level flows. Extraction from low-level flows results in a large proportion of the flow being removed from the natural flow. Less water would continue downstream and in the case of early-season flows or in years of low rainfall and poor runoff, the downstream progression of critical river flows would be impeded or cease, reducing the provision of ecosystem services to the lower catchment. A further reduction in proportional flow would result from an extended duration of pumping. A 30-day time window for water extraction would harvest water from not only the peak of the flow hydrograph, but the trailing river flows as flood levels decline. Water extraction from peak river flows removes a smaller proportion of the daily flood volume. Avoiding water extraction from the trailing edge of the flood hydrograph extends the duration of effective river flows and downstream ecosystem service provision.

The WRD scenarios modelled for the Gilbert River were the impoundment of river flows due to the placement of one (172 GL yield, 85% reliability) or two dams (498 GL yield, 85% reliability). No other variables, such as the release of environmental flows from the dams, were incorporated into the Gilbert River streamflow projections. The WRD scenarios modelled for the Flinders River were pumped extraction of 160 GL pa and 400 GL pa. No river flow pump-limit thresholds were available to be modelled. Neither were the effects of pump rate criteria on projected streamflows available. These variables were not modelled in the WRD scenarios applied to streamflow data derived from the original catchment and river systems models for the Flinders River catchment (Petheram et al. 2013).

Of the 19 WRD scenarios run, we have focused on four key scenarios. These range from volumes of water allocated above which annual reliability becomes problematic (WRD1) that simultaneously explore impacts on all three key rivers (high extraction rates on Mitchell and Flinders and two dams on the Gilbert) through to a moderate scenario (WRD2), a scenario (WRD3) with moderate extraction from the Mitchell and no WRDs on the Flinders and Gilbert, and finally a lower impact scenario (WRD4 – no WRD on Mitchell River and moderate extraction on Flinders combined with a single dam on the Gilbert River). Further details of these are provided below and summarised in (Table 6).

WRD1 is a WRD combination that would extract or impound quanta of water from the rivers that are near the upper-level of volume that is feasible while maintaining a reliability of 75-85% (except 400 GL extraction from the Flinders River, 60%) (Petheram et al. 2013, 2018) (Table 6). In addition, low river flow thresholds (200 ML d⁻¹) and extended pump periods (30 days) would have negative impacts on downstream ecosystem service provision.

WRD2 is a WRD combination that would extract or impound about half the quanta of water extracted by WRD1 (and build one dam on the Gilbert River as opposed to two) and increase extraction reliability to 75%-85% for all rivers (Table 6). In addition, higher river flow thresholds and shorter pump periods (15 days) would moderate the reduction to low-level flood flows by preventing water extraction from early flows that may occur during the late dry-season. Ecosystem service provision would be enhanced relative to WRD1 as pumping would not commence until river flows reached 1800 ML d⁻¹, and a high proportion of water would be extracted from flood peaks during the wet season.

WRD3 is a WRD combination that extracts the same quanta of water from the Mitchell as WRD2, and with the same extraction duration (15 days), but with a low river flow threshold for both rivers (200 ML d⁻¹). The low extraction thresholds provide a contrast between harvesting low-level flows and the loss of ecosystem services that they provide, or allowing late dry-season initial flows and low-level flows to continue downstream. After the environmental stressors of the dry season, low-level flows support the re-establishment of some ecosystem services within riverine and estuarine habitats. When pumped extraction of those flows is not prevented, then no or much-reduced river flows proceed downstream. Under WRD3, facilitated, early ecosystem service re-establishment before the onset of the wet season would not occur.

WRD4 is a WRD combination that combines no WRD on the Mitchell and Roper River, so the natural flow regime is maintained. However, WRD on the Gilbert and Flinders Rivers was modelled. Under the scenario, 160 GL pa was extracted from the Flinders River and one dam was placed on the Gilbert River. No river flow thresholds or pump rates are modelled.

The other scenarios tested (WRD5-19) include alternative combinations and scenarios applied to the different rivers. Scenarios WRD5 and WRD6 are considered more extreme but unlikely. Nonetheless they are included for purposes of comparing with predictions from other studies, and our preliminary simulations to try and bound the problem (Table 6).

Table 5. Water Resource Development configurations (extraction and offstream storage) deployed within the MICE model for the Mitchell River featuring two water resource allocations (GL/year), three thresholds of flow (ML/day) below which pumping cannot occur, and two pump rates (days to pump water).

Configuration	Flow threshold (TH, ML/day)	Pump rate (PR, days)	Catchment allocation (GL/year)
1	200	15	200
2	200	30	200
3	1200	15	200
4	1200	30	200
5	2200	15	200
6	2200	30	200
7	200	15	1000
8	200	30	1000

Configuration	Flow threshold (TH, ML/day)	Pump rate (PR, days)	Catchment allocation (GL/year)
9	1200	15	1000
10	1200	30	1000
11	2200	15	1000
12	2200	30	1000

Table 6. Description of the 19 water resource development (WRD) scenarios for the Mitchell, Flinders and Gilbert Rivers tested using the MICE. WRD involved either the extraction of water or the impoundment of water (i.e. dams). For water extraction, a description of the annual allocation (gigalitres per annum), flow threshold (TH) and pump rate (PR) are provided. Low TH = 200 ML d⁻¹; medium TH = 1200 ML d⁻¹; high TH = 2000 ML d⁻¹; low PR = 30 days to pump water allocation; high PR = 15 days to pump water allocation. Current WRD for Flinders and Gilbert Rivers involves minor WRD extraction for domestic-settlement use.

WRD	Mitchell	Flinders	Gilbert	Description
Base	Base line	Base line	Base line	Base flows with current water resource development
WRD1	High allocation (2000 GL pa, low TH, low PR)	400 GL pa allocation	2 Dams (498 GL yield)	High-allocation scenario for all three rivers (MSC_23+FSC_2+GSC_2) with low PR (Mitchell)
WRD2	Mid allocation (1000 GL pa, high TH, high PR)	160 GL pa allocation	1 Dam (172 GL yield)	Mid-range allocation scenario for all three rivers (MSC_11+FSC_1+GSC_1) high PR. (Mitchell)
WRD3	Mid allocation (1000 GL pa, low TH, high PR)	Base line	Base line	Contrast-allocation scenario for Mitchell River (MSC_7) with low TH, high PR. Current WRD for Flinders and Gilbert Rivers
WRD4	Base line	160 GL pa allocation	1 Dam (172 GL yield)	Base flows for Mitchell River and mid-allocation scenario for Flinders and Gilbert Rivers (FSC_1+GSC_1)
WRD5	5 Dams (2831 GL yield)	400 GL pa allocation	2 Dams (498 GL yield)	High-allocation scenarios for all three rivers (MSC_20+FSC_2+GSC_2). Mitchell River scenario unrealistic.
WRD6	Very High allocation (4800 GL pa, low TH, high PR)	400 GL pa allocation	2 Dams (498 GL yield)	Very high-allocation scenario for Mitchell River and high-allocation scenario for Flinders and Gilbert Rivers (MSC_13+FSC_2+GSC_2) Mitchell River scenario unreliable annually.

WRD	Mitchell	Flinders	Gilbert	Description
WRD7	1 Dam (Nullinga, 65 GL yield)	400 GL pa allocation	2 Dams (498 GL yield)	One dam for Mitchell River and high-allocation scenario for Flinders and Gilbert Rivers (MSC_21+FSC_2+GSC_2)
WRD8	Low allocation (200 GL pa, low TH, low PR)	Base line	Base line	Low-allocation scenario for Mitchell River (MSC_2) with low PR. Current WRD for Flinders and Gilbert Rivers.
WRD9	Low allocation (200 GL pa, high TH, low PR)	Base line	Base line	Low-allocation scenario for Mitchell River (MSC_6) with high TH, low PR. Current WRD for Flinders and Gilbert Rivers
WRD10	Mid allocation, med TH, low PR)	160 GL pa allocation	1 Dam (172 GL yield)	Mid-range allocation scenario for all three rivers with med TH and low PR (MSC_10+FSC_1+GSC_1)
WRD11	Mid allocation (1000 GL pa, low TH, low PR)	Base line	Base line	Mid-range allocation scenario for Mitchell River (MSC_8) with low TH, low PR. Current WRD for Flinders and Gilbert Rivers
WRD12	Mid allocation (1000 GL pa, med TH, high PR)	Base line	Base line	Mid-range allocation scenario for Mitchell River (MSC_9) with medium TH, high PR. Current WRD for Flinders and Gilbert Rivers
WRD13	Mid allocation (1000 GL pa, med TH, low PR)	Base line	Base line	Mid-range allocation scenario for Mitchell River (MSC_10) with medium TH, low PR. Current WRD for Flinders and Gilbert Rivers
WRD14	Mid allocation (1000 GL pa, high TH, high PR)	Base line	Base line	Mid-range allocation scenario for Mitchell River (MSC_11) with high TH, high PR. Current WRD for Flinders and Gilbert Rivers
WRD15	Mid allocation (1000GL pa, high TH, low PR)	Base line	Base line	Mid-range allocation scenario for Mitchell River (MSC_12) with high TH, low PR. Current WRD for Flinders and Gilbert Rivers
WRD16	2 Dams (1313 GL yield)	Base line	Base line	Two dams for Mitchell River (MSC_19) and current WRD for Flinders and Gilbert Rivers
WRD17	1 Dam (65 GL yield)	Base line	Base line	One dam for Mitchell River (MSC_21) and current WRD for Flinders and Gilbert Rivers

WRD	Mitchell	Flinders	Gilbert	Description
WRD18	High allocation (2000 GL pa, low TH, low PR)	Base line	Base line	High allocation scenario for Mitchell River with low TH, low PR (MSC_23) and current WRD for Flinders and Gilbert Rivers
WRD19	High allocation (2000 GL pa, high TH, low PR)	Base line	Base line	High allocation scenario for Mitchell River with high TH, low PR (MSC_27) and current WRD for Flinders and Gilbert Rivers

14. Biological and Fishery Risk

Several fisheries use risk assessment frameworks as a key decision tool to screen and quantify risk levels in a structured way. For example, the “Ecological Risk Assessment for Effects of Fishing” ERAEF method developed jointly by CSIRO Marine and Atmospheric Research and the Australian Fisheries Management Authority (Hobday et al. 2011) involves a hierarchical approach that progresses from a qualitative through to semi-quantitative and quantitative model-based approaches. For example, an assessment of the ecological impacts of the Northern Prawn Banana Prawn sub-fishery was undertaken in 2020 to assess the ecological risks arising from fishing (Sporcic et al. 2020). The ERAEF also considers the impact of external factors such as coastal development and within-catchment infrastructure placement, in this case concluding this may have a moderate impact given it “may have a detectable impact on banana prawn behaviour/movement as a result of altered flow regimes and changes to water/habitat quality” but noting there is low associated confidence as “little data available to demonstrate the effects of coastal development on prawn behaviour/movement” (Sporcic et al. 2020). The consequence scores used for ERAEF assessments are shown in Table 6 below.

Risk-based eco-hydrological approaches have also been developed to assess the influence of environmental flow regimes (McGregor et al. 2018). The risk to an ecological asset within a plan area is assessed based on the likelihood of flow-related opportunities and the frequency of exceedance of a consequence (McGregor et al. 2018). This approach also accords greater risk if multiple locations suffer simultaneous influences.

Table 7. Consequence score for ERAEF activities (Modified from Fletcher et al. 2002 and copied from Sporcic et al. 2020).

LEVEL	SCORE	DESCRIPTION
Negligible	1	Impact unlikely to be detectable at the scale of the stock/habitat/community
Minor	2	Minimal impact on stock/habitat/community structure or dynamics
Moderate	3	Maximum impact that still meets an objective (e.g. sustainable level of impact such as full exploitation rate for a target species).
Major	4	Wider and longer term impacts (e.g. long-term decline in CPUE)
Severe	5	Very serious impacts now occurring, with relatively long time period likely to be needed to restore to an acceptable level (e.g. serious decline in spawning biomass limiting population increase).
Intolerable	6	Widespread and permanent/irreversible damage or loss will occur-unlikely to ever be fixed (e.g. extinction)

Undertaking a full risk assessment is beyond the scope of this project, but we have applied a preliminary risk analysis to broadly classify risks under alternative WRDs to (A) the species habitat-forming groups sub-set used in our MICE and (B) dependent fisheries. In the latter case, for the banana prawn fishery, we also present a preliminary economic risk analysis. Future studies could build on our approach to refine and add to our risk statistics, as well as add risk statistics for recreational and indigenous fisheries, including values associated with social and cultural attributes. Based on the ERAEF scoring in Table 7, we developed modified scoring criteria for our model indicator outputs as shown in Table 8, with slightly different criteria used for regional averages as opposed to local (i.e. single catchment system) impacts. The negligible and minor categories were defined using semi-arbitrary small changes to indicators (< 10%). At the other extreme, intolerable risk was defined as local declines in excess of 50%. A 50% decline relative to the base-case was selected because well-managed fisheries (such as the NPF) aim to maintain stocks at a target level (MRAG 2017), with the Commonwealth Harvest Strategy defining half this level as the Limit Reference Point below which fisheries should be closed. We therefore considered that declines of between 30-50% could reasonably be considered as severe, and then semi-arbitrarily defined declines between 20-30% as major and between 10-20% as moderate. As regional losses are of even greater concern, we adjusted these scores such that regional declines of one-third (as opposed to half) are rated as “Intolerable” (Table 8).

We sought risk scorings that could be used consistently across all categories and species or habitat-forming groups. In addition, we evaluate risk as relative to the baseline (i.e. no future WRDs) scenario because estimates of current depletion are uncertain for some species and groups and hence we consider it more robust to evaluate outcomes relative to the current levels. Note that this may underestimate risk for species which are currently thought to be below target levels (such as sawfish) but our analysis provides a starting point for future more detailed risk analyses.

Table 8. Summary of risk scoring criteria used for MICE outputs, noting that relatively smaller declines in abundance or catch are required for the same risk score when assessed at the regional rather than local scale (to account for the fact that regional influences account for a much broader spatial effect as well as having greater consequences).

Risk Ratings	Scores	Criteria for local	Criteria for regional
Negligible	1	<5% locally	<5% locally
Minor	2	Minimal impact (<10%)	Minimal impact (<10%)
Moderate	3	At least 10% decrease in indicator (10-20%)	At least 10% decrease in indicator (10-15%)
Major	4	Wider and long-term impacts eg at least 20% decrease in indicator (20-30%)	Wider and long-term impacts eg at least 15% decrease in indicator (15%-25%)
Severe	5	Very serious impacts - decline of at least 30% (30-50%)	Very serious impacts - decline of at least 25% (25-33%)
Intolerable	6	Widespread and unacceptable loss - decline of at least 50%	Widespread and unacceptable - decline of at least 33% (i.e. one-third regionally)

Our scoring outlined above accounts for the consequences of different scenarios, both in terms of local impact on a system as well as whether the impact translates also into a larger scale regional impact. Regional scale also takes into account if multiple developments are implemented at the same time. To account for the likelihood of predicted impacts occurring, we generate results for each of our

five models and compute the average (and SD (standard deviation)) across the MICE ensemble. Note also that our analyses are based on a 50-year time series which has implicit within it the likelihood of changes occurring in any one year. We assign equal weights to the average change over the 50-year time period (which captures the average consequence of an impact) and the largest decline over the period (lower likelihood but greater consequence).

For each model version and for each species, we classified risks under alternative WRD scenarios. The spawning biomass or other available population abundance index was used as performance (output) statistics to evaluate risks to the population. For the fished species, we used relative changes to catch and commercially available biomass. We grouped “commercially available biomass” in the fishery risks category because this abundance measure is influenced by changes in catchability (which may in turn be influenced by flow – see mud crab example) and hence this is not a true measure of the underlying biomass. Although female mud crabs are not fished in the eastern GoC, we nonetheless used spawning biomass as a measure of population abundance as both male and female crabs would be impacted by river flow. In averaging results, we assigned equal weight to minimum and average changes such that standardised overall scores were the average of the individual risk scores. Finally, we compared the regional (model regions 2-6) average risk for each species under each WRD (averaged over model ensemble) to obtain an overall risk rating for each WRD. We also analysed relative risks per catchment region.

We used the colour coding scheme as shown in Table 8 to highlight categories of risk, and in particular where we found risks to be major, severe or intolerable.

15. Economic Risk

To complement the economic analysis described earlier in the report (see section 8. Bio-economic modelling), we also calculated some preliminary economic risk statistics for the common banana prawn GoC sub-fishery. Consistent with the inputs used in our MICE, we focused only on the common banana prawn component of the total banana prawn fishery (i.e. we used a species split algorithm to exclude catches of redleg banana prawns (*Penaeus indicus*)). We found that the GoC common banana prawn catch is 74% on average (1970-2019) of the total NPF banana prawn catch. The average NPF total and GoC common banana prawn catch over the past 10 years (2010-2019) is 4647t and 3606t respectively. These analyses could be extended to other species in future if additional data become available.

In theory, economic analyses (in consultation with fishing industry representatives) could be used to identify the profit break-even point for the common banana prawn fishery. Below this lower limit total fishery profits are reduced and given the crew payment system where their income is a proportion of catch, crew retention is compromised. Moreover, the impact of a “bad” year would be considerably exacerbated if it is repeated for a second year. This approach would be useful if doing forward projections, but is less useful in the context of this project because we rely on 50 historical years of data during which the fishery has undergone restructuring (Pascoe et al. 2012) and there have been substantial changes in prices and costs which the fishery is sensitive to.

As a first step towards evaluating the relative risk of an increase in the frequency of “bad” fishing years, we therefore used our MICE ensemble to establish what the catch total is below which the fishery drops 10% of the time, i.e. on average, under baseline flow conditions, there is a 10% risk that the fishery will catch less than this amount in any given year. This is consistent with the Commonwealth Harvest Strategy Policy (Resources 2018) which identified points beyond which the risk to a stock is regarded as unacceptably high – here we extend that logic to the risk to a fishery not achieving an “acceptable” catch. The Harvest Strategy Policy uses as a guideline that stocks should be above a biomass limit at least 90 per cent of the time, and hence we use a 10% risk of falling below the economic catch threshold value we identify as the “unacceptable” catch limit for the purposes of

these analyses (and acknowledging that the corresponding economic evaluations depend on a number of additional variables as explained above).

The model average “unacceptable” catch average for the GoC common banana prawn fishery is 2000t. This means that across our model versions, the fishery catches less than 2000t common banana prawns in the GoC once every ten years or so. We also consulted with industry representatives who confirmed that a catch below around 2000t (GoC common banana prawns) is a reasonable “unacceptable” catch limit to use for the purpose of these analyses focused on the long run economic environment. Anecdotal information also confirmed the sensitivity of the industry to changes in prices and costs. Moreover, we note that 2000t is 55% of the recent average annual GoC common banana prawn catch of 3606t i.e. it is approximately half the average catch which can reasonably be considered a bad year for most fisheries. Furthermore, to account for the likelihood of a series of “unacceptable catch” years, we define an additional risk statistic to evaluate the longer-term impacts on the ability of companies to attract and retain good crew. Industry have stated that in this situation their operations become unviable. Finally, we define “fishery operations unviable” as three or more consecutive years below a threshold value of 2000t of banana prawn catch based on anecdotal information from industry representatives that operations would become unviable if poor or “unacceptable” catches continued for three years.

We therefore used an annual catch of 2000t as the threshold “unacceptable” catch value and for ease of explanation, we term this a “bad” year. As above, this is a long run average guideline that is used as a reference level in our modelling to evaluate how this risk level might change under different scenarios. In the short-term, several other factors influence what might be considered a “bad” year economically. For all Model versions, we computed what the risk is of a bad year, two successive bad years, and risk of fishery operations becoming unviable, under the baseline flow scenario compared with alternative WRD scenarios. As we had 50 annual model estimates of catch (for the period 1970-2019) we computed risk probabilities as the number of occurrences divided by 50.

We also computed risk in addition to baseline risk (“additional risk”) as the risk minus the baseline risk level. For the purpose of comparability with our ecological risk assessment, we classify a bad year as a major risk to the fishery, two consecutive bad years as a severe risk and a series of bad years as an intolerable risk that should be avoided if at all possible.

WRD simulations are only assumed to impact some or all of the Mitchell, Gilbert and Flinders Rivers and hence our analysis ignores potential future impacts on other catchment systems that extend throughout the GoC and Top End region where the common banana prawn fishery operates i.e. catches in these other areas are assumed maintained. In addition, we acknowledge that averaging results over the scale of the entire GoC under-estimates regional risks such as lower catches available in the south-eastern corner of the GoC, potentially higher travel costs for fishers using local ports such as Karumba as well as the preferred fishing locations of individual fishers.

Results

16. Modelling of river flows in Gulf of Carpentaria

The river flows for use in modelling historical population dynamics in the ecological model were derived from water resource assessments undertaken across northern Australia in the last decade: the Flinders and Gilbert Agricultural Resource Assessment (FGARA), Northern Australia Water Resource Assessment (NAWRA) and Roper River Water Resource Assessment (RoWRA). End of system river flow data were produced using a calibrated AWRA-R river system model (Dutta et al. 2013) or model predecessors in the case of FGARA (see Lerat et al. 2013). The initial Mitchell River model used inputs from January 1890 to September 2015, while for the MICE ecological model the data inputs were updated, extending simulations to 2019. The intention of the river system model was to simulate river flow at locations throughout the catchment at a daily scale. The models also had the utility to simulate flows for scenarios that include various future climates and irrigation development, in particular, water harvest and off-stream storage and the placement of large in-stream dams.

The Mitchell and Roper River end of system river flows were produced using a calibrated AWRA-R model that was built and calibrated as a part of the NAWRA. The Flinders and Gilbert simulations were based on river models built for the FGARA. For most of the contributing area to the Gulf of Carpentaria, no river models are available, but there are some stream gauge data but not for the entire study area. This study thus utilised the AWRA-L model and outputs for the study area where no river model simulations were available.

Selected illustrative plots showing the full historical flow patterns as well as more recent (1970-2018) for the first 6 model regions are presented in Figure 19 and Figure 20 respectively. As evident from Figure 19, there is a lot of intra-annual variability in flow, and the magnitude also varies from year to year. This is illustrated by comparing flow patterns for a very wet year (1974) compared with a dry year (2015) and a recent year (2018) is also shown (Figure 21). The spatial and intra-annual variability are incorporated in the ecological model, is used to quantify relationships between flow and population dynamics based on past observations.

Over the last decade, Commonwealth and State Governments have prosecuted plans and scoping research to develop the water resources of Australia's large-catchment northern rivers. Investing millions of dollars, three major research projects have identified the potential water resource assets and catchment characteristics and ecology of rivers basins:

The Flinders and Gilbert Agricultural Resource Assessment (FGARA, Petheram 2013); the Northern Australia Water Resource assessment (NAWRA, Petheram 2018 a,b,c) and the Roper River Water Resource Assessment (RoWRA CSIRO 2019, current) have provided comprehensive water and arable land assessments of nine catchments in the wet-dry tropics of northern Australia. Hence, a series of large-catchment northern rivers are now in-scope to support irrigated agriculture (Table 9).

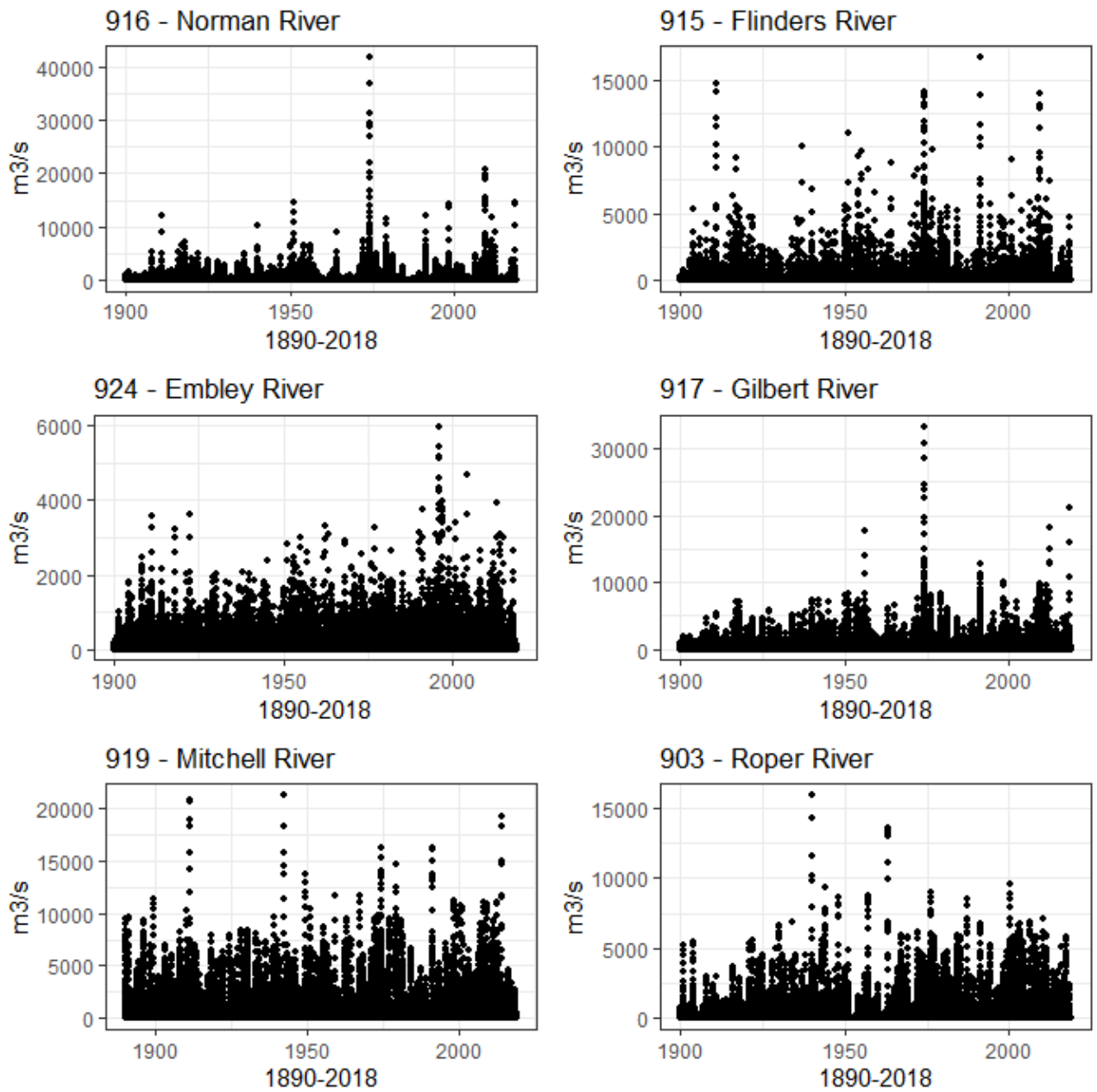


Figure 19. Examples of modelled flow estimates for selected Gulf of Carpentaria rivers from 1890 to 2018. Note the y-axis (vertical axis) is different across regions.

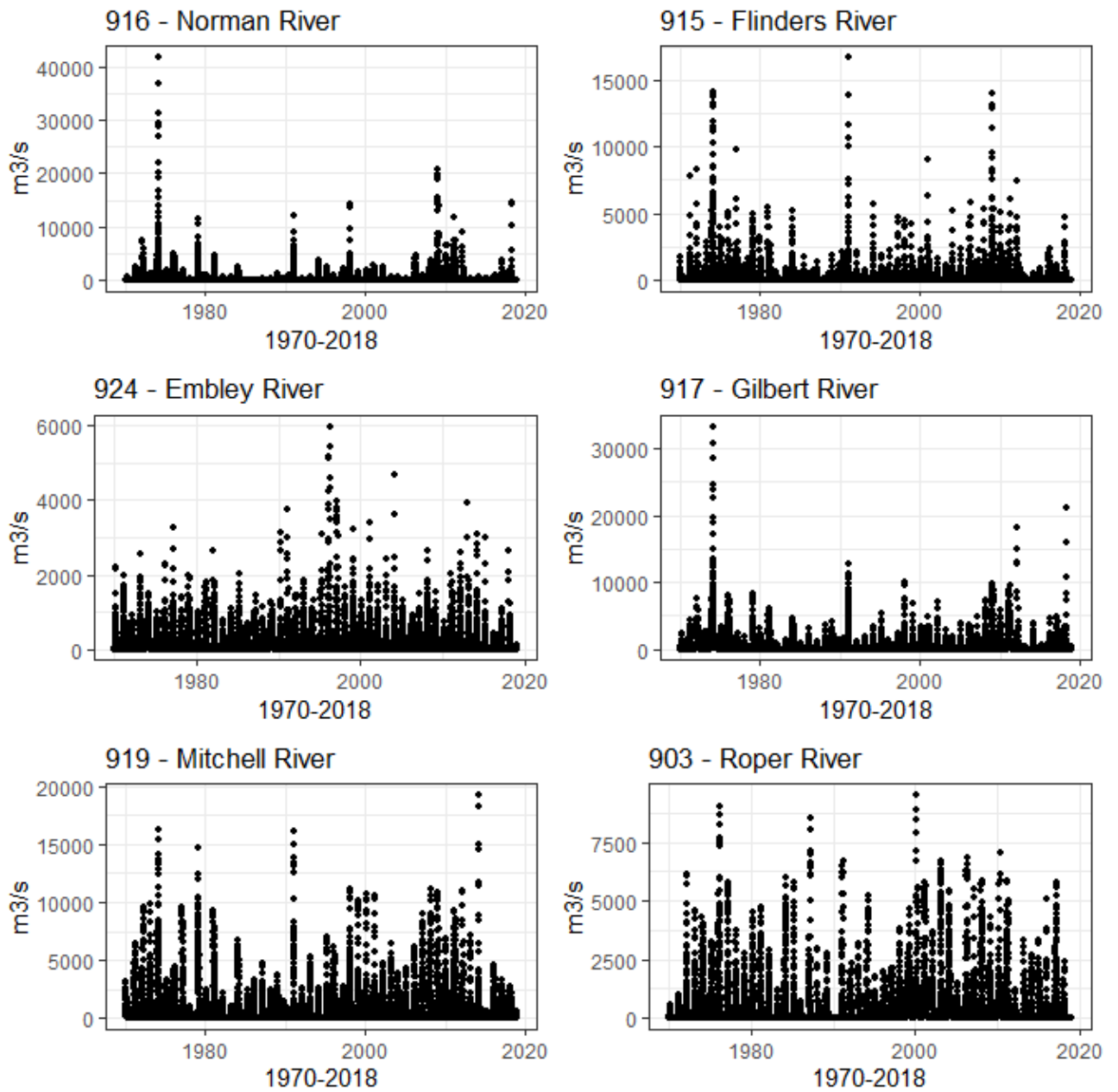


Figure 20. Expanded view of modelled flow estimates for selected Gulf of Carpentaria rivers shown for the period corresponding to that for which historical fishery data are available for key species in the ecological model, namely 1970 to 2018. Note the y-axis (vertical axis) is different across regions.

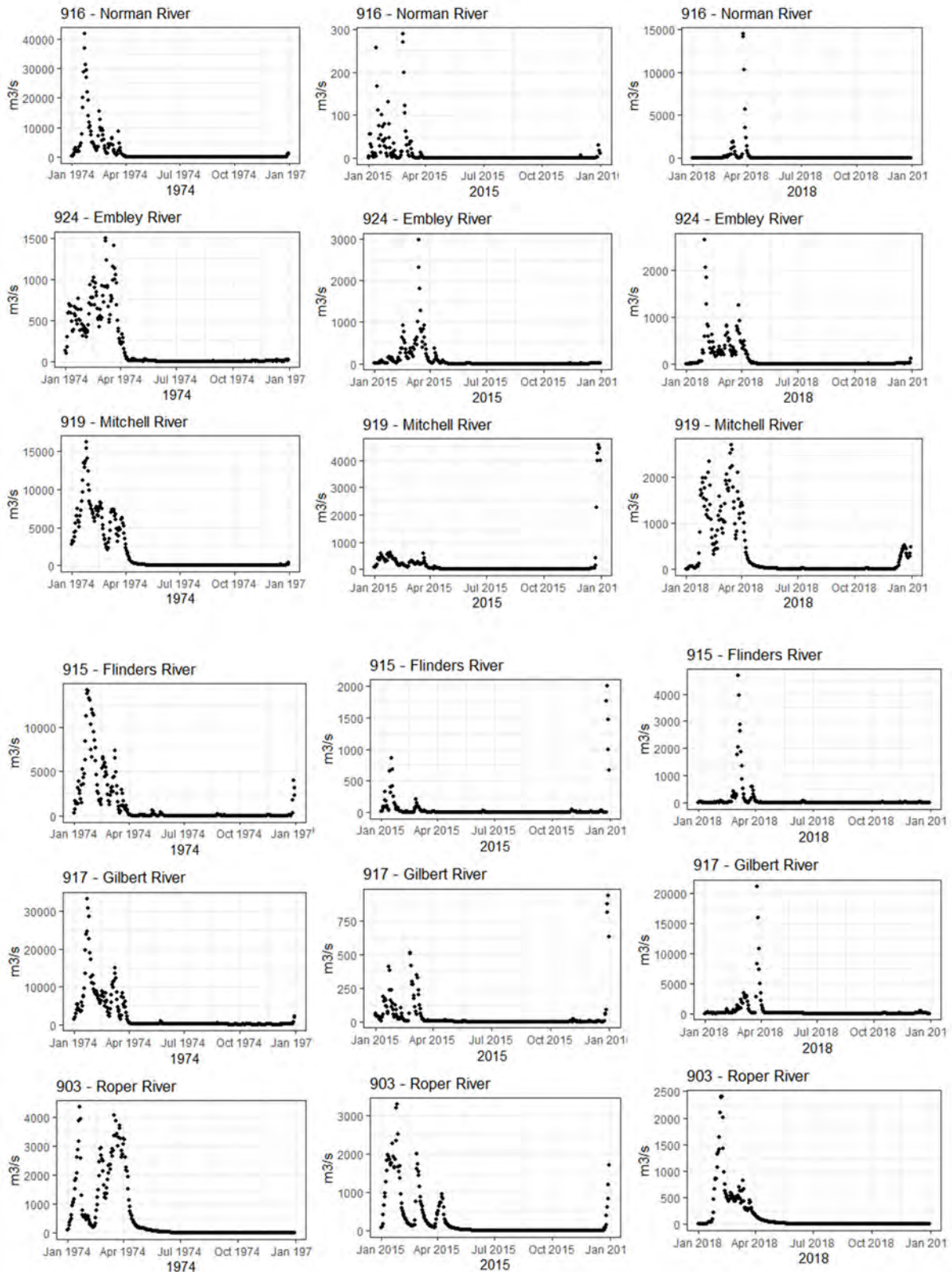


Figure 21. Examples of intra-annual variation in flow compared between three illustrative years (1974, 2015 and 2018) for selected Gulf of Carpentaria rivers as shown. Note the y-axis (vertical axis) is different across regions.

Table 9. Tropical Australian rivers subject to Water Resource Assessment and Management research to scope catchments suitable for irrigated agriculture in northern Australia. River flow allocations (GL per annum) and type of WRD are shown for various rivers from different project reports.

<i>Report</i>	<i>River</i>	<i>Flow GL pa</i>	<i>Water Resource Development Type</i>
FGARA	Flinders River (Qld)	2,543 GL	Off-stream storage
	Gilbert River (Qld)	3,706 GL	In-stream dam, off-stream storage
NAWRA	Adelaide River (NT)	2,413 GL	In-stream Dam
	Finnis River (NT)	1,436 GL	na
	Fitzroy River (WA)	6,586 GL	Off-stream storage
	Mary River (NT)	2,405 GL	na
	Mitchell River (Qld)	15,579	In-stream dam, off-stream storage
	Wildman River (NT)	905 GL	na
RoWRA	Roper River (NT)	2,269 GL	Off-stream storage

17. Base Case MICE Results

Model ensemble: Prawns

For common banana prawns, all model versions linked flow with subsequent recruitment to the population and fishery (i.e. number of 6-month old prawns in the marine environment), with the addition of flow as an explanatory variable significantly improving model fits (see below). There were insufficient data to support separate estimation of processes influencing recruitment of prawn larvae to estuaries versus the return emigration of juveniles recruiting to the adult population. Hence, the model estimates the overall net effect of all processes that result in recruitment of 6-month old prawns leaving the estuary and joining the population for the first time. These recruits are added to the existing prawn population in the marine environment (i.e. those that have survived natural mortality and fishing) and this combined population in turn comprises the exploitable biomass that is available to fishers. Hence the amount of prawns caught in each week of each year will be influenced by the amount of recruitment, which is in turn influenced by river flows.

The model tries to find the optimal flow-recruitment parameter combinations that best explain the observed catch data. For ease of viewing (and because there are too many plots otherwise), model fits are shown as the combined annual totals (over the key fishing period - weeks 13 to 22) but the model likelihood function sums the contribution of fits to the weekly catch data. As there are 10 key fishing weeks each year, and the model is fitted to 50 years from 1970 to 2019, there are 500 weekly likelihood contributions for each of the eight model regions, which reduces to 490 weekly catch comparisons per region (given zero effort in some years for some of these weeks), and hence a total of 3920 common banana prawn catch observations that the model is fitted to. The model structure and parameter combination therefore need to be broadly consistent with all these data inputs. This means

that the observed and predicted annual catch totals don't always match up perfectly, and in some cases this is because the underlying fits are perfect fits for some weeks but poor fits for other weeks. The model trades-off across all of these weekly catch rate fits, so that the 'amount' unexplained is minimised. It would not be possible to fit all data perfectly even with considerable further information. However the model has not had to draw on additional information or needed to add to model complexity in order to provide a reasonably good explanation of the observed data. In other words, we have been able to keep the model as simple and tractable as possible through dynamic statistical estimation of a limited number of parameters that capture how flow affects prawn recruitment. This has only been possible because of the long time series for prawns consistent with historical data that capture considerable variability in natural flows as well as fishing effort. These data therefore provide a solid basis for investigating the impacts of flow on common banana prawn recruitment and the use of a dynamic integrated spatial modelling approach (as compared to more static statistical analyses as done in other studies e.g. Broadley et al. 2019) improves the model power and ability to integrate understanding of the complex system dynamics.

The model also estimated the way in which flows across the entire south-eastern GoC sub-region (model regions 2-6) interact to explain observed common banana prawn catches that are caught offshore of each of the major rivers (Mitchell, Gilbert, Norman and Flinders Rivers) but are the combined result of different flows entering the GoC. This study is the first to quantify the relative contributions of different rivers – the “river portfolio effect” – to the overall abundance and catches of common banana prawns in the south-eastern sub-region in an integrated manner. Hence in some years the rivers act in concert to boost overall prawn recruitment whereas in other years one river with good flows may contribute more prawns than a neighbouring river where flows are less due to natural climate variability. We acknowledge that these complex interactions will vary from year to year and there may well be other processes at play in some years, but the model was nonetheless able to estimate the average interactions between these major river systems (Table 10; Models 1, 3-5) based on fitting to all the weekly catch data as explained above.

The river catchment interaction estimates scale overall recruitment relative to the actual flows in each individual catchment system, and hence this effect is not constant over time but depends on the standardised flow levels in each week of each year for each river system – in other words, the relative contribution of each river system during each week of the year is proportional to the standardised flow level of that river for that week (with constant of proportionality the interaction parameter). Model version 2 does not incorporate the river portfolio effect and instead explores sensitivity to assuming that river flows have more local system impacts only (Table 10 Model 2).

Based on previous studies pointing to the role of river flows in driving primary production and hence ultimately boosting prawn productivity and catches (Burford and Faggotter 2021), we incorporated this effect in our model (Model 5) by linking prawn survival with a lagged overall index of primary productivity, in turn modelled as a function of flow. We found that adding this additional effect significantly improved the model fits (Model 5 Table 10; Figure 24-Figure 25).

Early investigations suggested that a logistic rather than linear relationship is preferred when modelling the impact of flows on recruitment. The different model versions incorporate different parametrisations of this relationship (Table 10) – for example the models variously set different flow threshold values and Model 3 uses an average over the preceding four weeks rather than week by week changes to influence the recruitment. In terms of the logistic relationship describing the way in which flow influences prawn recruitment, one parameter was fixed in all model versions while the second parameter was estimated for each of the major catchments. The model reliably estimated these flow parameters with small and largely non-overlapping associated confidence intervals (Table 10). Model results therefore suggest that there are important differences in the rate and point at which decreases in flow impact on prawn recruitment. For example, the base model estimates that decreases in flow in the Gilbert River have the greatest relative impact whereas the Norman and Embley Rivers are estimated to have a more gradual impact as flow is reduced (cf steepest slope of decreasing limb of logistic curve for Gilbert Region compared with flatter slope for Norman River in Figure 22). The

next steepest relationship was estimated to apply to the Mitchell River, followed by the Roper and Flinders Rivers. Note that the estimation of the Gilbert River catchment system’s influence on prawn recruitment as particularly sensitive to reductions in flow is an emergent result that is well estimated within the model (narrow 90% confidence intervals), and hence provides a fairly robust basis for model simulations to explore the effect of alternative WRDs. It suggests the presence of thresholds which are different to those for the other river systems. For example, in the Roper, Embley, Flinders, and Norman Rivers, the relationship between recruitment and flow index is almost linear, whereas in the Gilbert River and to a lesser extent Mitchell River, the relationship is logistic - whereby at low and high flows, changes in flow have a modest effect on recruitment but at middle flows, changes in flow can have major consequence on recruitment. This result was also found to be robust to other model settings such as changing the second flow threshold parameter and recruitment spatial proportion (e.g. based on more recent versus historical relative catch per region) (Table 10).

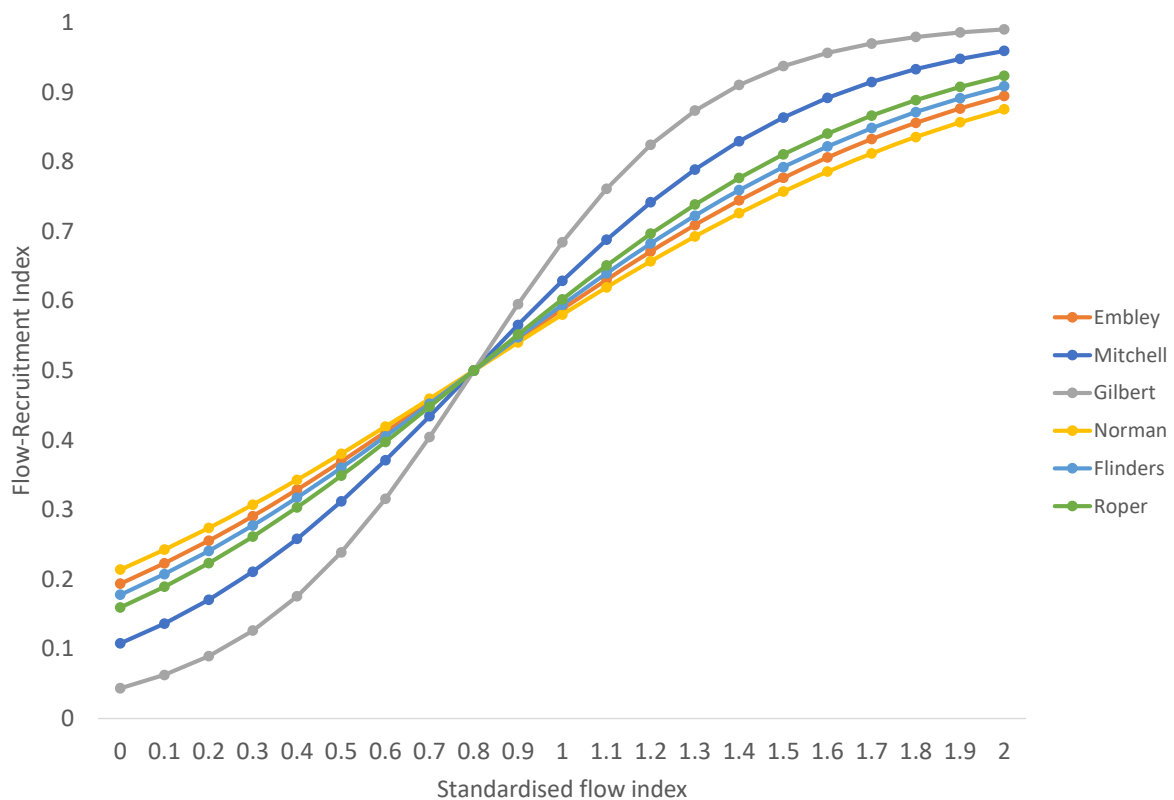


Figure 22. MICE estimated logistic functions used to describe the relationship between the standardised flow and common banana prawn recruitment for each of the catchment systems as shown, where the function yields a multiplier that describes recruitment relative to the maximum value (noting the function asymptotes at 1 for flows greater than twice the median flow level).

The final key sensitivity relates to the assumed relative abundance of prawns in each of the model regions. Given historical catches are likely to have been roughly in proportion to the relative abundance, we use the average historical relative catch to inform on the proportion of the pristine overall spawning biomass that is distributed in each region. Model 4 tests an alternative to this assumption and uses instead a more recent (2002-2019) relative catch proportion as an input (with one of the major differences being that the proportion of catches taken from model region 1 has reduced by almost half over the recent period).

A no-flow linked model (i.e. the catches estimated based on observed effort but with no accounting for the potential influence of flows), was seen to reasonably capture trends in catch in some regions such as the Embley and Flinders River catchment regions, but very poor fits were obtained (without adding additional complexity that is also not included in the with-flow model versions) for other regions such as the Gilbert and Norman River catchment regions (Figure 23). Moreover, for regions such as the Flinders and Roper River catchments, the fishing effort clearly explains a lot of the overall pattern of changes in catch over time, but the model did not adequately capture the amplitude of the large peaks in catches (Figure 23). The addition of flow as a second explanatory variable to estimate catches dramatically improved the ability of the model to replicate these very high catch years (Figure 24).

A comparison of the model fits for Model version 1 and Model version 5 (the best fit model) are shown in Figure 24 and Figure 25, with the full set of results shown in Appendix 17. This highlights the kind of trade-offs that result in the fit to the different regions when using different model structures and hence why it is useful to use an ensemble approach when evaluating the impacts of alternative WRDs.

The range of plausible population abundance and catch trajectories estimated under each model version is shown in Figure 26-Figure 29. Note that for prawns these trajectories extend back to 1970 but here we standardise outputs by starting in 1989 because that is the earliest year for which data were available for other species in the MICE. The ensemble of biomass and catch trajectories is shown first for all model regions, and combined Qld and NT jurisdictions (Figure 26-Figure 27), followed by a closer focus on regions 2,3 and 5 given these are the key focus of the current WRD simulations (Figure 28-Figure 29). In place of commercially available biomass, in Figure 28 we show relative spawning biomass. Summary statistics of the differences between all our model versions are presented in more detail in Appendix 17.

In the following section we present the results of predicted responses to changes in baseline flows under alternative WRDs using the base case model (Model 1), as well as differences between the model versions comprising our ensemble (Models 1-5), in terms of predicted responses under alternative WRDs.

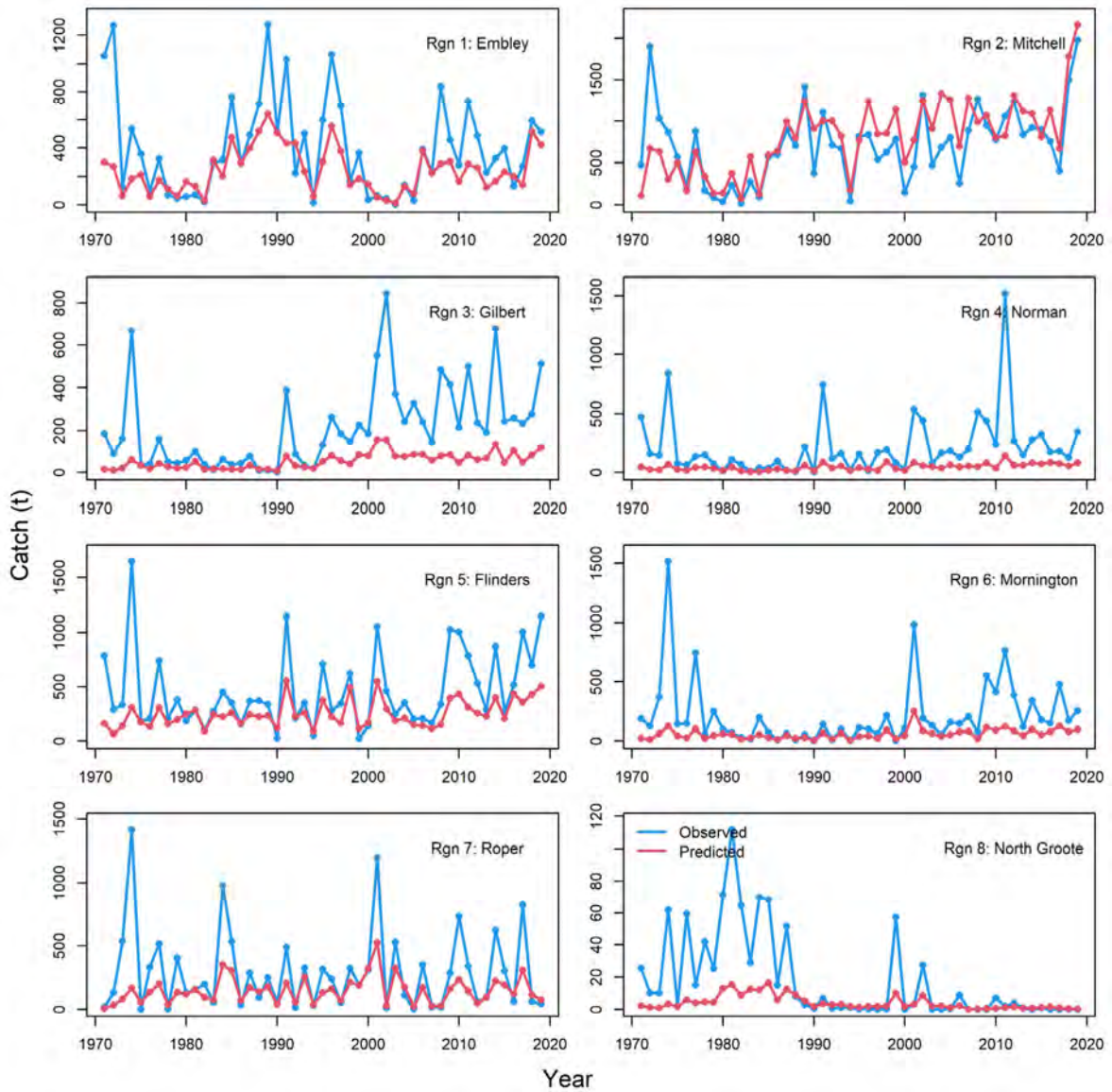


Figure 23. Comparison of the observed and model-predicted annual common banana prawn catch (tonnes) estimated using only fishing effort (i.e. no flow linked to banana prawn population dynamics) for each of the eight model regions as shown.

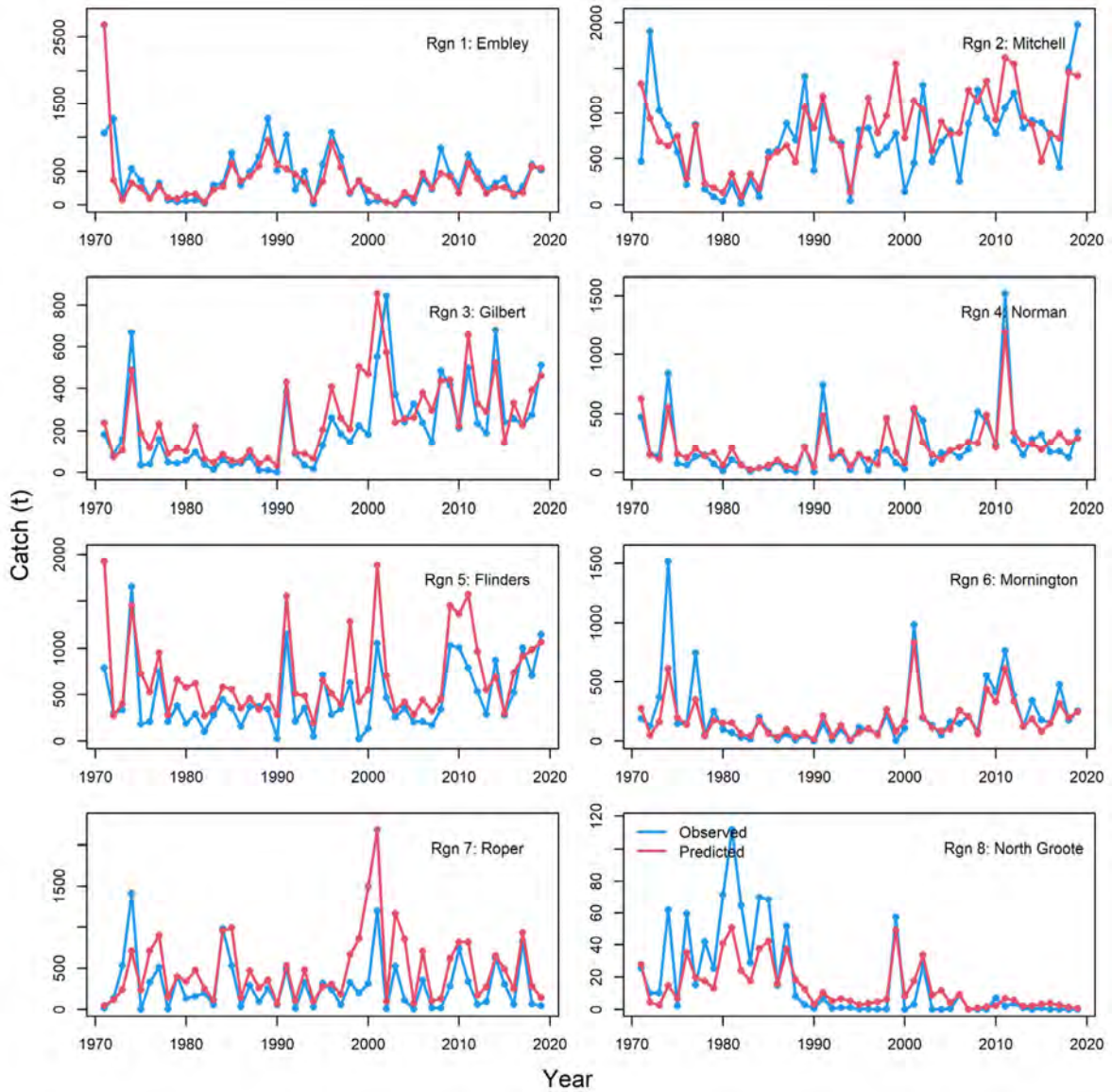


Figure 24. Comparison of the observed and model-predicted annual common banana prawn catch (tonnes) estimated using Model version 1 (driven by baseline flows) and for each of the eight model regions as shown.

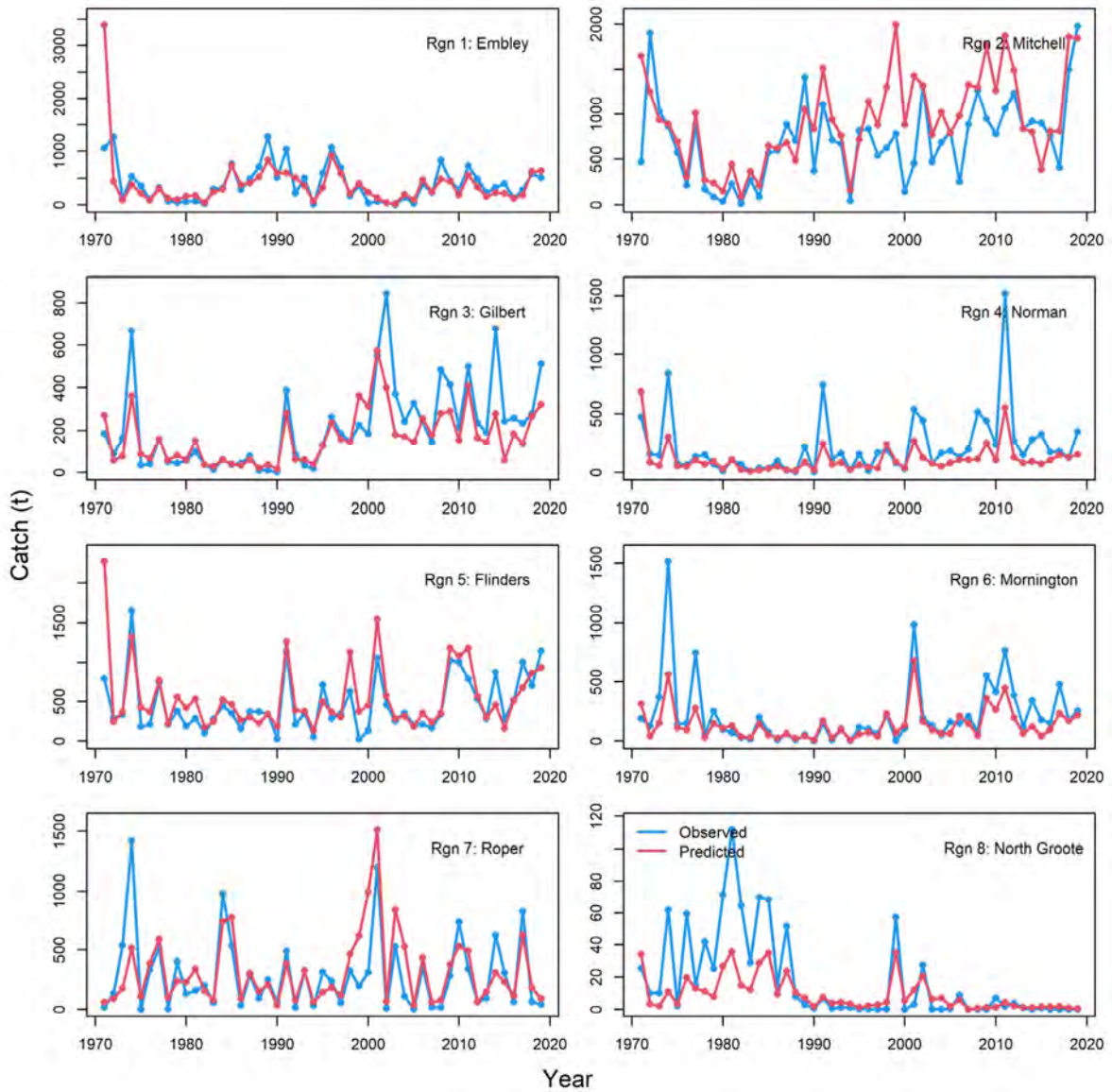


Figure 25. Comparison of the observed and model-predicted annual common banana prawn catch (tonnes) estimated using Model version 5 (driven by baseline flows, prawn survival linked with lagged overall index of primary productivity) and for each of the eight model regions as shown.

Table 10. MICE model parameter estimates, associated Hessian-based standard deviations (std), number of parameters estimated, negative log likelihood (-lnL) contributions and Akaike Information Criterion (AIC) scores across each of the five model versions for (1) common banana prawn, (2) barramundi and (3) mud crab. Parameter estimates in italics are fixed parameters. Bsp = spawning biomass (t); CAA = catch-at-age; SOI = southern oscillation index; Mitch = Mitchell River; Gilb = Gilbert River; Norm = Norman River; Flind = Flinders River.

Parameter	Description	<u>Model 1 - base</u>		<u>Model 2 - flow relationship</u>		<u>Model 3 - production</u>		<u>Model 4 - depletion</u>		<u>Model 5 -link productivity</u>	
		value	std	value	std	value	std	value	std	value	std
<i>(1) Common banana prawn</i>											
Kstart	Initialisation ln(Bsp)	5.000	0.003	5.200	0.003	<i>5.000</i>	–	5.265	0.003	5.400	0.026
Thres (1)	Flow threshold parameter	0.561	0.010	0.561	0.009	<i>0.561</i>	–	0.561	0.002	<i>0.561</i>	–
Thres (2)	Flow threshold parameter	0.379	0.032	0.379	0.004	<i>0.379</i>	–	0.379	0.013	<i>0.379</i>	–
Thres (3)	Flow threshold parameter	0.259	0.022	0.259	0.012	<i>0.259</i>	–	0.259	0.019	<i>0.259</i>	–
Thres (4)	Flow threshold parameter	0.615	0.017	0.615	0.019	<i>0.615</i>	–	0.615	0.018	<i>0.615</i>	–
Thres (5-6)	Flow threshold parameter	0.522	0.014	0.522	0.010	<i>0.522</i>	–	0.522	0.012	<i>0.522</i>	–
Thres (7-8)	Flow threshold parameter	0.481	0.044	0.481	0.021	<i>0.481</i>	–	0.481	0.010	<i>0.481</i>	–
Mitch1	River connection parameter	0.071	0.007	0.085	0.062	0.055	0.003	<i>0.071</i>	–	0.051	0.016
Mitch2	River connection parameter	0.349	0.041	–	–	0.405	0.038	<i>0.349</i>	–	0.229	0.028
Mitch3	River connection parameter	0.001	0.002	–	–	0.000	0.001	<i>0.001</i>	–	0.007	0.013
Mitch4	River connection parameter	0.055	0.016	–	–	0.047	0.013	<i>0.055</i>	–	0.055	0.014

Parameter	Description	<u>Model 1 - base</u>		<u>Model 2 - flow relationship</u>		<u>Model 3 - production</u>		<u>Model 4 - depletion</u>		<u>Model 5 -link productivity</u>	
		value	std	value	std	value	std	value	std	value	std
Gilb1	River connection parameter	0.031	0.011	0.450	0.645	0.022	0.008	<i>0.031</i>	–	0.041	0.009
Gilb2	River connection parameter	0.001	0.004	–	–	0.001	0.004	<i>0.001</i>	–	0.001	0.004
Gilb3	River connection parameter	0.003	0.007	–	–	0.002	0.006	<i>0.003</i>	–	0.003	0.006
Gilb4	River connection parameter	0.081	0.024	–	–	0.080	0.019	<i>0.081</i>	–	0.081	0.016
Norm1	River connection parameter	0.023	0.008	0.450	0.151	0.047	0.012	<i>0.023</i>	–	0.068	0.015
Norm2	River connection parameter	0.164	0.047	–	–	0.187	0.072	<i>0.164</i>	–	0.114	0.037
Norm3	River connection parameter	0.528	0.072	–	–	0.574	0.086	<i>0.528</i>	–	0.228	0.074
Norm4	River connection parameter	0.043	0.017	–	–	0.047	0.016	<i>0.043</i>	–	0.043	0.010
Flind1	River connection parameter	0.028	0.009	0.245	0.131	0.024	0.009	<i>0.028</i>	–	0.033	0.009
Flind2	River connection parameter	0.012	0.013	–	–	0.001	0.005	<i>0.012</i>	–	0.012	0.009

Parameter	Description	<u>Model 1 - base</u>		<u>Model 2 - flow relationship</u>		<u>Model 3 - production</u>		<u>Model 4 - depletion</u>		<u>Model 5 -link productivity</u>	
		value	std	value	std	value	std	value	std	value	std
Flind3	River connection parameter	0.053	0.028	–	–	0.007	0.010	0.053	–	0.053	0.021
Flind4	River connection parameter	0.190	0.027	–	–	0.242	0.047	0.190	–	0.120	0.016
Embley	Flow parameter	0.155	0.018	0.156	0.040	0.148	0.016	0.155	–	0.166	0.057
Roper	Flow parameter	0.336	0.013	0.306	0.263	0.566	0.020	0.336	–	0.223	0.040
$thresM^{PSP}$	Flow threshold parameter	0.8	–	0.8	–	1.500	0.101	0.8	–	0.8	–
No. parameters		26		13		19		7		19	
Likelihood contributions		value	sigma	value	sigma	value	sigma	value	sigma	value	sigma
-lnL: Catch (Region 1)		505.4	1.7	539.2	1.8	509.2	1.7	486.8	1.6	494.6	1.7
-lnL: Catch (Region 2)		393.8	1.4	369.6	1.3	399.8	1.4	396.7	1.4	397.7	1.4
-lnL: Catch (Region 3)		432.5	1.5	412.0	1.4	433.6	1.5	463.1	1.6	355.8	1.3
-lnL: Catch (Region 4)		468.8	1.6	472.2	1.6	472.1	1.6	480.1	1.6	371.0	1.3
-lnL: Catch (Region 5)		541.9	1.8	528.0	1.8	548.5	1.9	561.2	1.9	488.4	1.6
-lnL: Catch (Region 6)		412.4	1.4	405.5	1.4	416.7	1.4	398.0	1.4	373.0	1.3
-lnL: Catch (Region 7)		479.4	1.6	510.8	1.7	476.7	1.6	553.2	1.9	399.7	1.4
-lnL: Catch (Region 8)		254.1	1.0	267.6	1.0	253.0	1.0	280.2	1.1	233.7	1.0
-lnL: overall		3488.3	–	3505.0	–	3509.6	–	3619.3	–	3114.0	–
AIC		7028.5	–	7035.9	–	7057.3	–	7252.6	–	6266.0	–

Parameter	Description	<u>Model 1 - base</u>		<u>Model 2 - flow relationship</u>		<u>Model 3 - production</u>		<u>Model 4 - depletion</u>		<u>Model 5 -link productivity</u>	
		value	std	value	std	value	std	value	std	value	std
<u>Other fixed parameters</u>											
M_{base}^{psp}	Natural mortality base (wk)	0.045	–	0.045	–	0.045	–	0.045	–	0.045	–
Selectivity pars (from non-spatial model)											
Parameter	Description	<u>Model 1 - base</u>		<u>Model 2 - flow relationship</u>		<u>Model 3 - production</u>		<u>Model 4 - depletion</u>		<u>Model 5 -link productivity</u>	
(region)		value	std	value	std	value	std	value	std	value	std
(2)											
<i>Barramundi</i>											
M_{base}^{barra}	Natural mortality base	8.01E-03	6.11E-05	8.00E-03	1.32E-07	fixed	0.022	5.00E-03	2.44E-08	8.00E-03	2.57E-08
q_r^{barra} (1)	Catchability	6.04E-05	9.60E-06	1.63E-04	2.12E-05	1.04E-05	1.53E-06	1.75E-04	1.21E-05	5.40E-05	1.00E-05
q_r^{barra} (2)	Catchability	3.68E-05	3.34E-06	2.58E-05	1.89E-06	5.69E-06	8.09E-07	2.16E-05	1.83E-06	4.40E-05	4.10E-06
q_r^{barra} (3-6)	Catchability	6.51E-05	1.40E-06	7.87E-05	4.03E-06	1.84E-05	1.97E-06	1.01E-04	4.41E-06	8.05E-05	5.45E-06
q_r^{barra} (7-8)	Catchability	3.22E-04	4.72E-06	3.14E-04	6.42E-06	3.19E-04	5.44E-06	3.18E-04	5.44E-06	3.28E-04	4.81E-06
$thres^{barra}$ (1)	Flow threshold parameter	0.561	–	0.268	0.138	0.000	0.389	0.268	–	0.561	–
$thres^{barra}$ (2)	Flow threshold parameter	0.379	–	0.389	0.092	4.203	2.303	0.389	–	0.379	–
$thres^{barra}$ (3)	Flow threshold parameter	0.259	–	2.472	0.847	0.233	0.039	2.472	–	0.259	–
$thres^{barra}$ (4)	Flow threshold parameter	0.615	–	20.000	0.008	1.553	0.626	20.000	–	0.615	–

Parameter	Description	<u>Model 1 - base</u>		<u>Model 2 - flow relationship</u>		<u>Model 3 - production</u>		<u>Model 4 - depletion</u>		<u>Model 5 -link productivity</u>	
		value	std	value	std	value	std	value	std	value	std
$thres^{barra}$ (5-6)	Flow threshold parameter	0.522	–	5.535	4.063	4.478	2.168	5.535	–	0.522	–
$thres^{barra}$ (7-8)	Flow threshold parameter	0.481	–	20.000	0.014	20.000	0.001	20.000	–	0.481	–
$\delta_{r=3-6,1997-2019}$	Age-selectivity par (Southern)	5.71E-01	4.15E-02	4.63E-01	2.94E-02	0.399	0.023	0.500	0.033	0.455	0.029
$sfa_{r=3-6,1997-2019}$	Age-selectivity par (Southern)	1.00E+00	4.88E-05	1.88E-01	4.57E-02	0.928	0.444	1.000	0.000	0.426	0.215
$\delta_{r=1-2,1997-2019}$	Age-selectivity par (Northern)	1.00E+00	3.93E-04	1.00E+00	4.11E-04	1.000	0.000	1.000	0.001	1.000	0.000
$sfa_{r=1-2,1997-2019}$	Age-selectivity par (Northern)	1.00E+00	2.15E-04	1.00E-02	8.87E-05	1.000	0.001	0.034	0.031	1.000	0.000
$thresM^{barra}$	Flow threshold parameter	0.80	–	0.80	–	1.50	–	0.80	–	0.80	–
No. parameters		9		15		14		9		9	
Likelihood contributions		value	sigma	value	sigma	value	sigma	value	sigma	value	sigma
-lnL: Catch (Region 1)		96.8	0.9	79.3	0.9	90.3	0.9	77.1	0.9	96.1	0.9
-lnL: Catch (Region 2)		130.3	1.1	106.2	1.0	123.1	1.1	96.7	0.9	127.7	1.1
-lnL: Catch (Region 3)		135.4	1.1	152.6	1.2	152.6	1.2	101.9	1.0	126.2	1.1
-lnL: Catch (Region 4)		137.8	1.1	107.7	1.0	128.2	1.1	104.9	1.0	141.3	1.2
-lnL: Catch (Region 5)		125.9	1.1	99.8	1.0	133.1	1.1	101.4	1.0	114.6	1.0
-lnL: Catch (Region 6)		118.4	1.0	94.4	0.9	130.4	1.1	105.2	1.0	151.0	1.2

Parameter	Description	<u>Model 1 - base</u>		<u>Model 2 - flow relationship</u>		<u>Model 3 - production</u>		<u>Model 4 - depletion</u>		<u>Model 5 -link productivity</u>	
		value	std	value	std	value	std	value	std	value	std
-lnL: Catch (Region 7)		141.7	1.2	81.2	0.9	115.0	1.0	80.7	0.9	123.2	1.1
-lnL: Catch (Region 8)		128.9	1.1	105.9	1.0	128.7	1.1	92.3	0.9	138.6	1.1
-lnL: CAA (Northern)		38.9	0.2	45.5	0.2	65.1	0.3	38.2	0.2	56.4	0.2
-lnL: CAA (Southern)		41.1	0.2	58.2	0.2	50.2	0.2	51.8	0.2	38.7	0.2
-lnL: overall		1095.3	–	931.0	–	1116.7	–	850.3	–	1113.8	–
AIC		2208.6	–	1891.9	–	2261.4	–	1718.5	–	2245.6	–
<u>Other fixed parameters</u>											
M_{par}^{barra}	Slope of age-dependent mortality	0.045		0.045		0.06		0.045		0.045	
		Calculated		Calculated		Calculated		Calculated		Not applied	
<u>Sex ratio</u>											
Parameter	Description	<u>Model 1 - base</u>		<u>Model 2 - flow relationship</u>		<u>Model 3 - production</u>		<u>Model 4 - depletion</u>		<u>Model 5 -link productivity</u>	
(region)		value	std	value	std	value	std	value	std	value	std
(3) Mud crab											
M_{base}^{mudcr}	Natural mortality base	0.10	–	0.10	–	0.12	–	0.10	–	0.10	–
h^{mudcr}	Stock-recruitment steepness parameter	0.6	–	0.6	–	0.6	–	0.6	–	0.4	–
$B_{1,1970,1}^{spn,mudcr}$ (1)	Initialisation ln(Bsp)	1.5	0.05	2.2	0.05	1.6	0.05	2.2	–	1.76	0.05
$B_{2,1970,1}^{spn,mudcr}$ (2)	Initialisation ln(Bsp)	1.4	0.04	2.1	0.04	1.5	0.04	2.1	–	1.64	0.04
$B_{3,1970,1}^{spn,mudcr}$ (3)	Initialisation ln(Bsp)	3.3	0.09	2.9	0.05	3.3	0.09	4.0	–	3.51	0.08

Parameter	Description	<u>Model 1 - base</u>		<u>Model 2 - flow relationship</u>		<u>Model 3 - production</u>		<u>Model 4 - depletion</u>		<u>Model 5 -link productivity</u>		
		value	std	value	std	value	std	value	std	value	std	
$B_{4,1970,1}^{spn,mudcr}$ (4)	Initialisation ln(Bsp)	3.3	0.16	2.4	0.07	3.3	0.16	4.0	–	3.66	0.16	
$B_{5,1970,1}^{spn,mudcr}$ (5)	Initialisation ln(Bsp)	2.7	0.09	2.2	0.06	2.7	0.10	3.4	–	2.89	0.09	
$B_{6,1970,1}^{spn,mudcr}$ (6)	Initialisation ln(Bsp)	2.0	0.15	1.5	0.13	2.0	0.15	2.7	–	2.27	0.15	
$B_{7,1970,1}^{spn,mudcr}$ (7)	Initialisation ln(Bsp)	4.2	0.10	4.4	0.08	4.3	0.09	4.9	–	4.34	0.06	
$B_{8,1970,1}^{spn,mudcr}$ (8)	Initialisation ln(Bsp)	3.6	0.07	3.9	0.08	3.7	0.07	4.3	–	3.78	0.05	
$thres^q$ (3-6)	Catchability due to flow	2.36E+00	7.11E-01	2.05E+00	5.86E-01	2.84E+00	1.03E+00	1.36E+00	2.43E-01	2.35E+00	6.92E-01	
$thres^{mudcr}$ (1-2)	Recruitment parameter due to flow	-8.64E-02	1.78E-04	7.00E-03	1.39E-02	-8.64E-02	1.72E-04	1.00E+00	-	7.32E-05	-8.64E-02	1.37E-04
$thres^{mudcr}$ (3-6)	Recruitment parameter due to flow	-7.45E-01	2.44E-02	4.50E-02	3.31E-02	-7.22E-01	2.61E-02	-8.98E-01	1.18E-02	-6.59E-01	1.97E-02	
$thres^{mudcr}$ (7-8)	Recruitment parameter due to flow	-1.33E-01	4.73E-02	1.75E+00	3.78E-01	-1.32E-01	4.55E-02	-3.72E-01	1.67E-02	-4.60E-08	1.46E-04	
τ^{temp} (7)	Mortality parameter due to temperature	3.46E+00	3.61E-01	3.46E+00	3.46E-01	2.96E+00	3.12E-01	4.07E+00	3.21E-01	3.12E+00	3.24E-01	
τ^{SOI} (3-6)	Recruitment parameter due to SOI	6.95E-01	1.10E-01	9.92E-01	7.95E-02	7.01E-01	1.10E-01	1.51E-01	4.19E-02	7.97E-01	1.08E-01	
No. parameters		14		14		14		6		14		
Likelihood contributions		value	sigma	value	sigma	value	sigma	value	sigma	value	sigma	

Parameter	Description	<u>Model 1 - base</u>		<u>Model 2 - flow relationship</u>		<u>Model 3 - production</u>		<u>Model 4 - depletion</u>		<u>Model 5 -link productivity</u>	
		value	std	value	std	value	std	value	std	value	std
-lnL: Catch (Region 1)		92.8	0.8	107.2	0.9	88.5	0.8	103.2	0.8	88.6	0.8
-lnL: Catch (Region 2)		66.9	0.8	62.2	0.7	63.3	0.7	71.1	0.8	62.0	0.7
-lnL: Catch (Region 3)		43.1	0.7	55.8	0.7	40.4	0.7	60.6	0.7	18.1	0.6
-lnL: Catch (Region 4)		93.8	0.8	128.1	0.9	91.3	0.8	116.1	0.9	76.4	0.8
-lnL: Catch (Region 5)		94.5	0.8	124.8	0.9	89.5	0.8	93.9	0.8	96.0	0.8
-lnL: Catch (Region 6)		46.1	1.1	48.0	1.1	46.4	1.1	46.4	1.1	44.9	1.1
-lnL: Catch (Region 7)		-32.8	0.6	-36.2	0.6	-36.5	0.6	1.0	0.6	-12.9	0.6
-lnL: Catch (Region 8)		58.1	0.8	54.7	0.8	58.2	0.8	91.0	0.9	61.4	0.8
-lnL: overall		462.5	–	544.7	–	441.2	–	583.4	–	434.4	–
AIC		953.1	–	1117.4	–	910.4	–	1178.9	–	896.8	–
<u>Other fixed parameters</u>											
q_r^{mudcr} (1-6)	Catchability	5.00E-04	–	5.00E-04	–	5.00E-04	–	5.00E-04	–	5.00E-04	–
q_r^{mudcr} (7-8)	Catchability	1.00E-06	–	1.00E-06	–	1.00E-06	–	1.00E-06	–	1.00E-06	–
$thresM^{mudcr}$ (1-2)	Optimum flow (standardised)	2.5	–	2.0	–	2.5	–	2.5	–	2.5	–
$thresM^{mudcr}$ (3-6)	Optimum flow (standardised)	2.0	–	1.6	–	2.0	–	2.0	–	2.0	–
$thresM^{mudcr}$ (7-8)	Optimum flow (standardised)	2.5	–	2.0	–	2.5	–	2.5	–	2.5	–

Parameter	Description	<u>Model 1 - base</u>		<u>Model 2 - flow relationship</u>		<u>Model 3 - production</u>		<u>Model 4 - depletion</u>		<u>Model 5 -link productivity</u>	
		value	std	value	std	value	std	value	std	value	std
c_σ	Flow relationship scaler	3.5	–	3.5	–	3.5	–	3.5	–	3.5	–
c_q	Catchability-flow relationship scaler	0.5	–	0.5	–	0.5	–	0.5	–	0.5	–
$temp^{opt}$	Long-term mean air temperature for Nov-Dec	34.5	–	34.5	–	34.5	–	34.5	–	34.5	–
c^{temp}	Temperature relationship scaler	1.0	–	1.0	–	1.0	–	1.0	–	1.0	–
η	ElNino constant	-1.0	–	-1.0	–	-1.0	–	-1.0	–	-1.0	–
η	LaNina constant	1.0	–	1.0	–	1.0	–	1.0	–	1.0	–
Overall MICE Likelihood		value		value		value		value		value	
Total no. parameters		49		42		47		22		42	
-lnL: overall		5046.1		4980.6		5067.5		5053.0		4662.2	
AIC		10190.2		10045.2		10229.0		10150.0		9408.3	

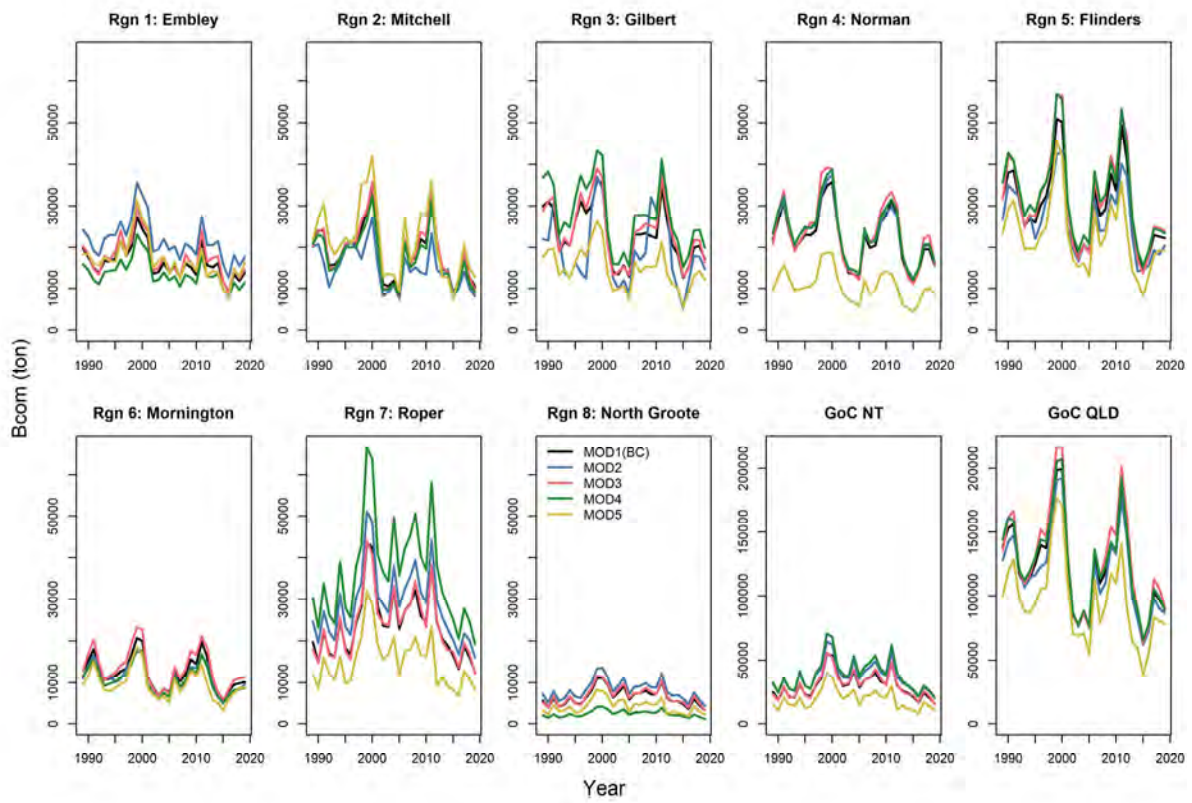


Figure 26. MICE ensemble outputs shown for the five alternative model configurations used to represent common banana prawns. Trajectories show the model-estimated commercially available biomass (Bcom) in units of tonnes shown from 1989 to 2019, for the different model regions and jurisdictions as indicated. See Table 4 for a description of the model versions.

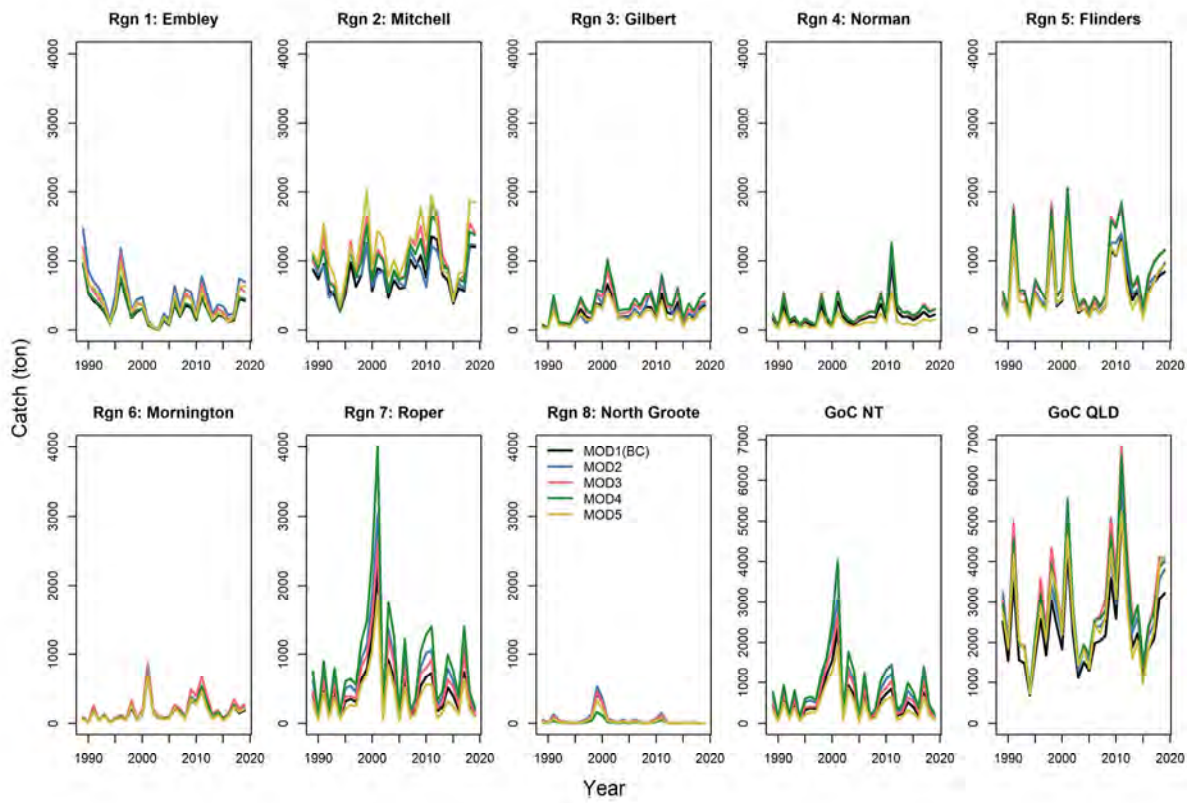


Figure 27. MICE ensemble outputs shown for the five alternative model configurations used to represent common banana prawns. Trajectories show the model-estimated catch (tonnes) from 1989 to 2019, for the different model regions and jurisdictions as indicated. See Table 4 for a description of the model versions.

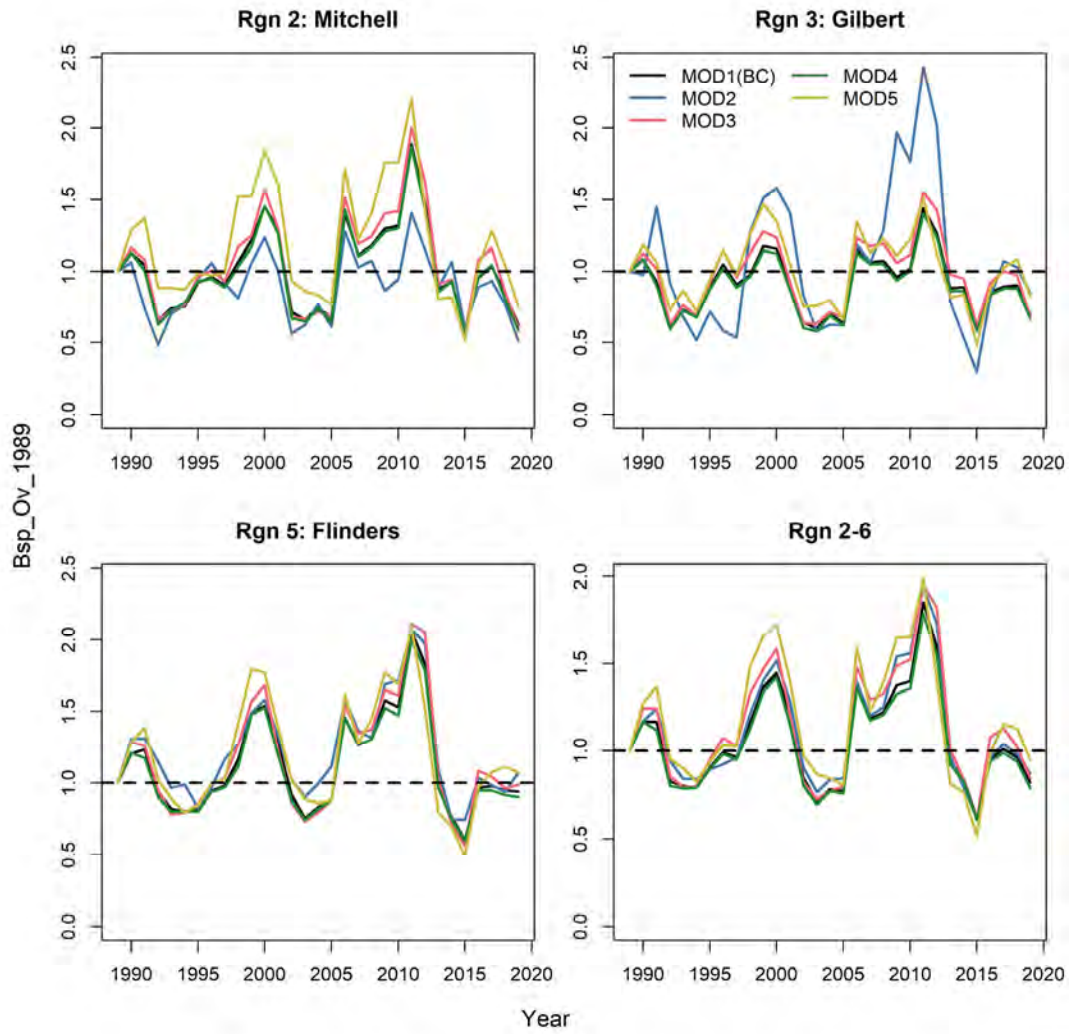


Figure 28. MICE ensemble outputs shown for the five alternative model configurations used to represent common banana prawns. Trajectories show the model-estimated spawning biomass (Bsp) plotted in each instance as relative to the 1989 spawning biomass (dashed line) estimate to highlight relative differences over time. See Table 4 for a description of the model versions.

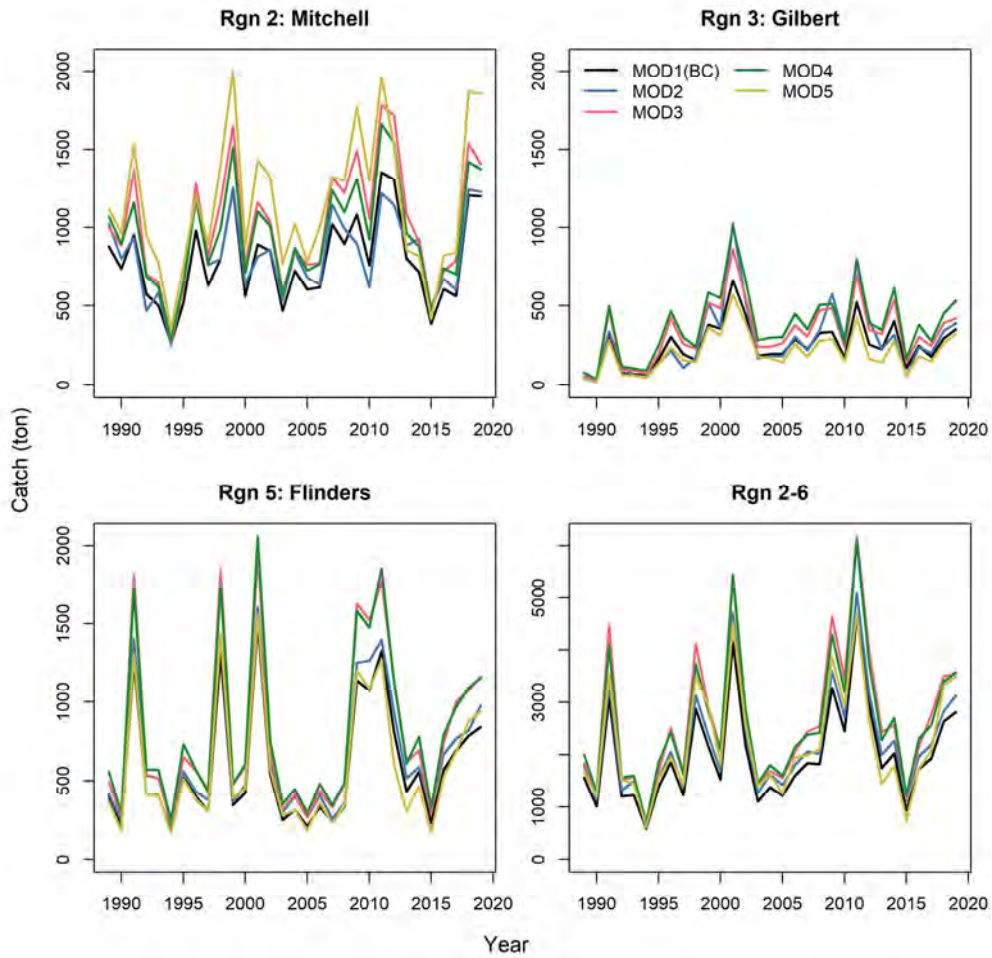


Figure 29. MICE ensemble outputs shown for the five alternative model configurations used to represent common banana prawns. Trajectories show the model-estimated catch (tonnes) from 1989 to 2019, for the three key model regions as indicated, together with the combined catch for Regions 2-6. See Table 4 for a description of the model versions.

River Portfolio Effect

The spatial structure of the MICE necessitated disaggregating historical common banana prawn catches to each of the eight model sub-regions. The model then uses observed fishing effort per region to estimate the total catch taken from each model region, and hence the model estimates both the magnitude and trends in the common banana catch taken from each region. As we demonstrate above, the model is able to more accurately estimate total catch per region if accounting for the influence of variable end-of-system flows. Fishing effort has also changed in intensity and spatial distribution over time such that the relative contribution of different areas to the overall common banana catch has changed over time – for example, Model region 1 (Embley River) currently (2002-2018) contributes an average of 6.5% to total catch versus a historical average contribution of 16%, in contrast to the Mitchell, Flinders and Roper River model regions which have increased their contributions to catches in recent years (Figure 30). However, given prevailing currents and connectivity in the system, the total biomass of prawns recruiting to the fishery in each region could well be influenced by neighbouring catchment systems. Apart from “non-connected” model version 2, all the ensemble model versions try and estimate the extent to which considering the south-eastern GoC river catchment systems - comprising the Mitchell, Gilbert, Flinders and Norman Rivers – as a shared portfolio of influence can improve estimation of the historical catches per region.

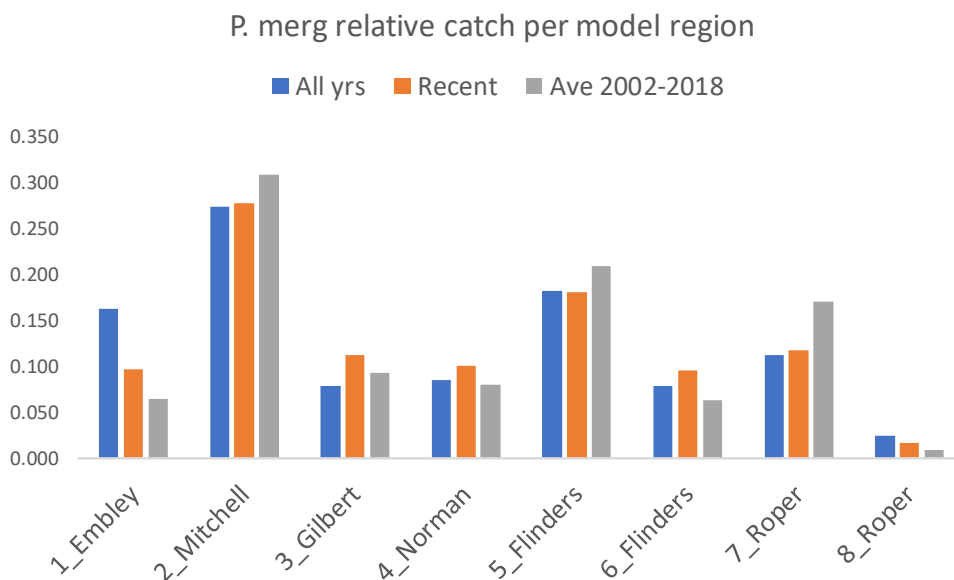


Figure 30. The relative contribution of the different model regions to the overall *P. merguensis* catch, when comparing the average over the entire time period (since 1970), with the average since 1980 and a more recent average over 2002-2018.

The integrated model accounts for the relative importance of early and late flows by comparing the flow in each region and each week with the average over the full time period 1900-2019. These so-called flow anomalies are then assumed to influence recruitment to the fishery (and hence weekly catches). Note that results do not necessarily imply that the recruits in a particular region of the GoC derive from juveniles moving from a particular river system: rather, these recruits are the integrated net outcome of a number of processes that eventuate in increases (or conversely decreases) in the mean catch taken from a region as a result of different combinations of flow across the suite of river systems. The relationship between flow and catch is not simply linear as a threshold value is also

estimated below which recruitment decreases (Figure 22), and there is an upper satiation limit above which additional flow doesn't make a huge difference.

For model regions 2-6 (Mitchell through to Flinders), the model estimates a relative contribution of each of the rivers to the observed catch i.e. formal statistical model fitting methods are used to estimate how influential each river systems are in determining the associated catch (Figure 31). The results suggest that all of the 6 major rivers (i.e. including Embley which influences region 1 and Roper River which influences region 7) are influential with the relative role and contributions of each varying by region and by year (Figure 31).

The model results suggested that the relative contributions of the different river systems to prawn recruitment (and hence catches) when averaged over the entire time period 1970-2018 is as shown in Figure 31. Figure 32 provides an alternative way of visualising how influential, on average, each of the different river systems is to predicting the weekly recruitment residual estimates that in turn determine at each time step the deviation from the average recruitment level (in turn a function of the spawning biomass in each region). Note that in any one year, the relative importance of the rivers (in terms of influencing variation in prawn recruitment) will vary depending on their flows. Model region 6 (i.e. Mornington, currently assumed similar to region 5 (Flinders River) and region 8 (Walker River, assumed similar to region 7 (Roper River)) are not shown as there are no specific flow estimates for these two regions.

The relative magnitudes of the bars in Figure 31 are similar to the magnitude of recruitment residuals estimated in a more conventional stock assessment model. Hence the relatively small magnitude of the residuals estimated for the Embley and Mitchell River systems suggests that recruitment in these systems is not as strongly influenced by end-of-system flows as is the case for the Roper River catchment for which larger residuals were estimated. Similarly, the Gilbert, Norman and Flinders River systems are all estimated as contributing substantially to variability in observed catches. There were also some differences estimated between our different model versions. When comparing Model version 5, which accounts for the ongoing flood productivity effect (Burford and Faggotter 2021), with model version 1, the Gilbert River emerged as relatively more influential in explaining variability of catches from that model region (Figure 31). In all cases, these south-eastern river catchment systems are estimated to have a combined influence on the total regional common banana prawn catch and variability.

The model flow residual estimates control the magnitude of the variability in common banana prawn recruitment as well as influences from neighbouring catchments. This variability is superimposed on an average regional abundance and recruitment and as we showed in Figure 30, the average total catch varies across spatial regions. Hence to quantify the model-estimated relative contribution of each of the river catchment systems (in turn mapped to model regions 1 to 8) to the total average common banana prawn recruitment and hence catches in the GoC, the flow residual estimates are combined with the average proportion of catches caught in each spatial region. This standardises how these local recruitment fluctuations translate into overall contributions to total catch on average (Figure 33).

The results were similar when using model versions 1 and 5. However comparing the MICE estimates of the relative contribution of the different river systems with the average spatial proportions of catch observed over 2002-2018 reveals a mismatch (Figure 33C). For example, the estimated contribution of the Norman River to regional catch totals is estimated to be much larger because large flows from this catchment have an influence on regional productivity and catches, i.e. it does not only influence the coastal waters immediately adjacent to the Norman River. The Flinders River is estimated to contribute a roughly similar amount as the Norman River to regional catches, followed by the Mitchell River catchment. The estimated increase in the relative contribution of the Gilbert River when using model version 5 compared with model version 1, is presumably because of the positive effect on the rest of the region of the flood productivity boost effect.

The combined portfolio of rivers thus acts to ‘stabilise’ or maintain the *P. merguensis* population across the entire GoC, and reducing flows from one or more rivers will have non-linear effects on the common banana prawn and other marine species as quantified by this project.

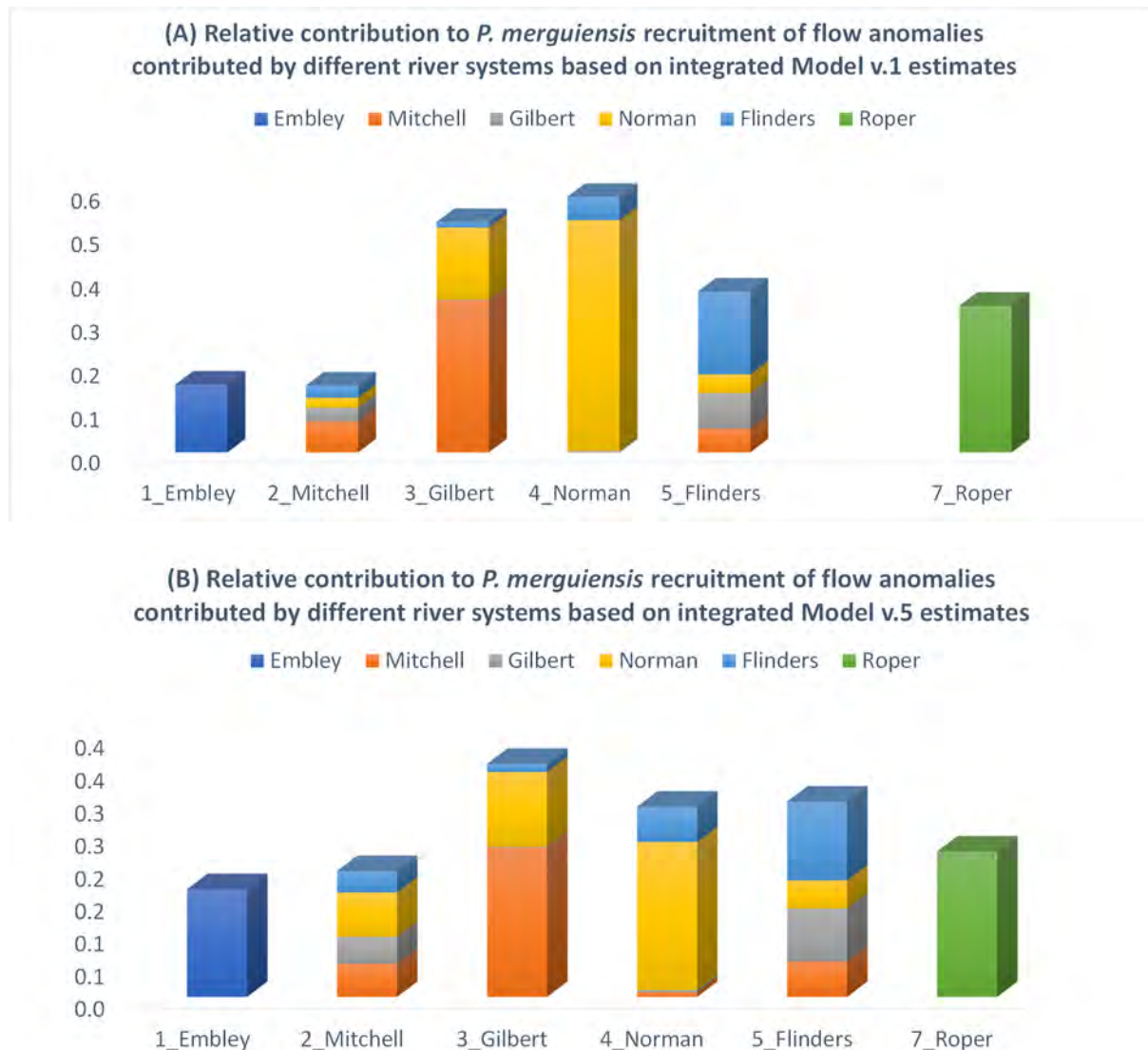


Figure 31. MICE model results showing (A) Model version 1 and (B) Model version 5 estimates of the relative contribution to *P. merguensis* recruitment (to the fishery) per model region of flow anomalies contributed by the different river systems. Areas 1 and 7 are assumed to only depend on the Embley and Roper Rivers respectively (hence why there is a single river shown as influencing recruitment in those areas), and the difference in the magnitudes reflect that the recruitment in the Embley area depends less on flow anomalies than is the case in the Roper River region. For regions 2-5, the model estimates relative contributions that are bounded between 0 and 1 (i.e. results suggest no effect of a river on that region’s prawn recruitment, or varying influences). Hence for example, model results suggest that the Norman River is the dominant driver of prawn recruitment in model region 4 with some contribution from the Flinders River. The Norman River is also estimated to be an important driver of recruitment in the Gilbert River model region 3. In contrast, prawn recruitment in the Mitchell and Flinders region is predicted to be driven on average by a combination of flow anomalies across all four river systems.

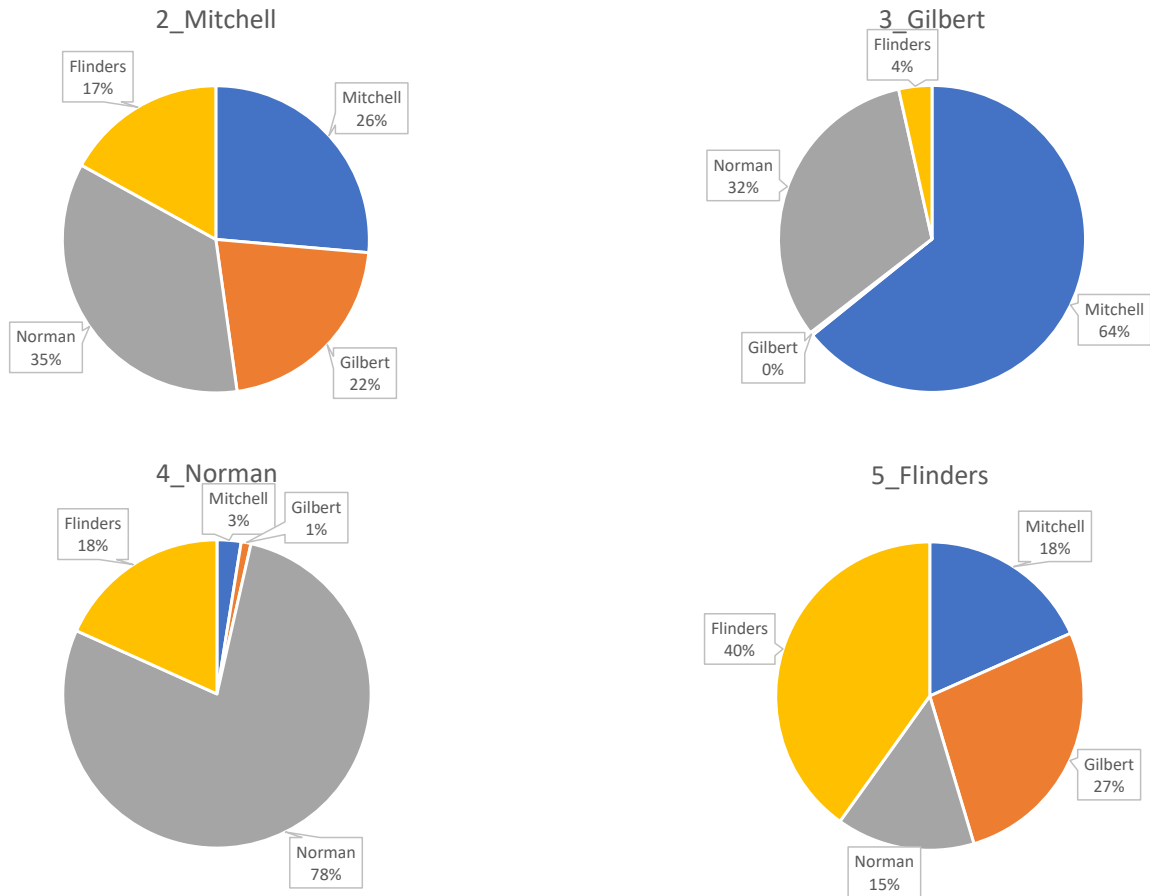
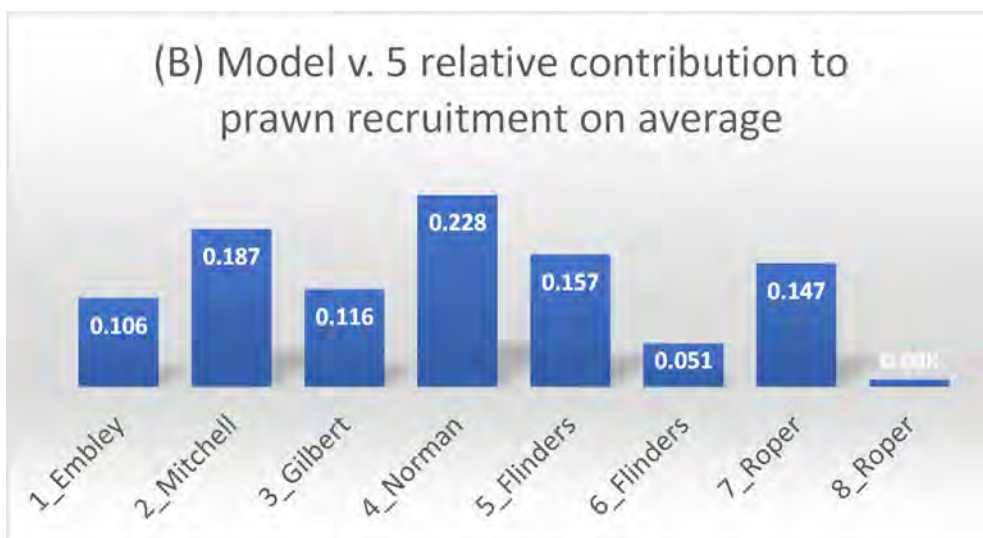
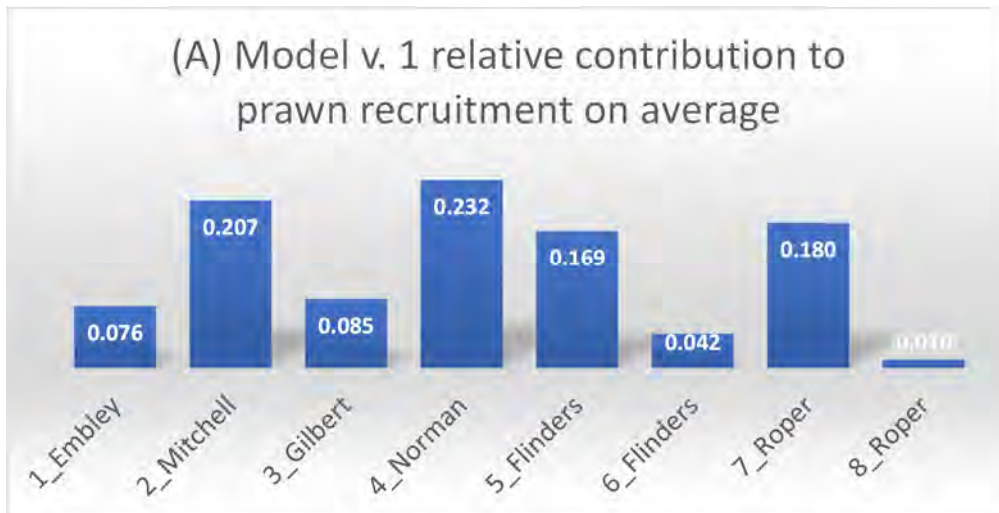


Figure 32. MICE model results (version 5) showing for each of the four model regions, where region 2 corresponds to the Mitchell River catchment, 3 to the Gilbert River catchment, 4 to the Norman River catchment and 5 to the Flinders River catchment, what the relative contributions are to explaining variability in catches taken from that region, based on influences from end-of-system flows from each of the four major rivers. Hence for example, recruitment to the fishery in the Norman River region is influenced predominantly by the Norman River flows themselves, as well as influences from the Flinders River system, whereas variation in prawn catches taken from the Flinders River region are estimated to be influenced substantially by the Mitchell, Gilbert and Norman River flows.



(C) Model estimation of the relative contribution of individual rivers as shown to the regional common banana prawn catch

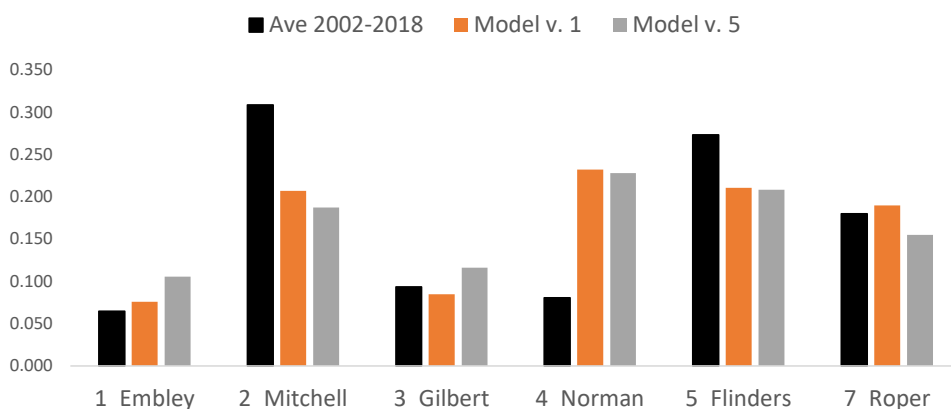


Figure 33. MICE model results shown in Figure 31 combined with estimates of the average *P. merguensis* catch taken from each model region to standardise how these local recruitment fluctuations translate into overall contributions to total catch. Hence these results show the model-estimated relative contribution of each of the river systems (in turn mapped to model regions 1 to 8) to the total average *P. merguensis* recruitment and hence catches in the GoC.

The model results suggest that the Norman River is particularly influential relative to the Mitchell, Gilbert and Flinders Rivers. As a preliminary way of seeking validation of this model result, we used the database to extract the total catches over weeks 13-22 and sorted these from largest historical observed catches to lowest as shown in Table 11. Highlights show that the highest catches do indeed correlate well with the Norman River flow anomalies, as well as reinforcing the points made above that different river systems are important in different years and that it is the combination of these flow anomalies across the different systems that ultimately determines the system productivity and catch that is available to be caught.

To further interrogate the data, we compared the top 15 catch years as above, with the top 15 catch years when using the total annual catch (i.e. all weeks instead of just 13-22) (Table 12). There was significant overlap as is expected (noting discussion above re correlation between catches over the current peak season versus full year) (Table 12). This was only not the case for early years (with different selectivity) 1973, 1979 and 1976.

Clearly the system dynamics are complex which is why it is helpful to use a dynamic integrated system model such as MICE, noting the model calculates the weekly catch per region based on fishing effort, fishing power, total stock size and spawning biomass and changes in flow from one or more linked river systems.

Table 11. Summary of the top 15 *P. merguensis* catch years (using catch totals over reference week period 13-22) shown alongside the average annual flow anomalies for each river system. The yellow highlights show the river system with the highest positive flow anomaly in the corresponding year, followed by orange highlights. The red highlights show the two years that didn't make the top 5 catches in this list, compared with when sorting by total annual catch as per Table 7 following.

Top catch yrs	P. merg	Flow relative to average					
	Total Catch (t) wks 13-22	Embley	Mitchell	Gilbert	Norman	Flinders	Roper
1974	7553.2	1.4	3.4	7.7	10.8	7.8	1.7
2011	5707.8	1.5	2.1	3.2	3.7	1.9	2.6
1991	5050.3	1.1	2.0	3.0	3.5	3.7	1.2
2001	4847.1	1.5	0.9	1.0	0.7	0.8	2.1
2009	4123.9	1.0	1.7	3.0	5.5	4.6	1.2
2014	4042.4	1.3	1.2	0.5	0.1	0.1	0.6
1972	3977.0	0.8	1.4	1.6	1.5	1.0	0.9
2010	3671.4	0.6	1.2	1.2	1.9	1.7	1.2
1989	3564.9	1.1	0.8	0.6	0.3	0.4	0.5
2008	3529.3	1.3	2.0	1.4	1.0	0.8	1.8
1977	3516.4	1.3	1.6	0.5	0.6	1.3	1.6
2018	3430.0	0.8	0.6	1.3	1.2	0.6	0.5
2017	3399.5	0.8	0.8	0.9	0.7	0.3	1.6
2002	3332.0	0.8	0.7	0.7	0.4	0.3	0.6
1971	3205.6	1.4	1.2	0.9	0.6	1.0	0.3

Table 12. Summary of the top 15 *P. merguensis* catch years (using catch totals over entire year) shown alongside the average annual flow anomalies for each river system for the years (highlighted in green) that were not included in the top 15 list shown in Table 1. The yellow highlights show the river system with the highest positive flow anomaly in the corresponding year, followed by orange highlights.

Top Total Catch yrs	Top Total Catch (t)	Flow relative to average					
		Embley	Mitchell	Gilbert	Norman	Flinders	Roper
1974	11847.3						
1971	6761.8						
2011	5947.7						
2001	5771.0						
1977	5757.6						
1991	5156.1						
1972	4375.2						
2009	4302.3						
2014	4130.4						
2010	3937.6						
1973	3899.7	1.2	1.2	0.8	0.5	0.6	0.9
2008	3730.6						
1989	3615.1						
1979	3608.4	1.3	2.6	1.9	2.8	2.2	0.5
1976	3607.1	1.2	1.4	2.5	1.9	1.9	2.5

It is also important to quantify the conditions leading to poor and very poor banana catch years. Table 13 shows the 5 lowest catch years (when summing catch over weeks 13-22 versus using total annual catch) and corresponding flow anomalies. This supports the model results that these are the result of synchronous low flows across the Mitchell, Gilbert, Norman and Flinders Rivers. What this means in terms of using an integrated MICE to quantify impacts of WRD scenarios, is that the model will replicate the non-linear effect on prawn catches of further reducing flows in one river, because it means the system is already under stress (and possibly close to tipping points).

These data analyses are deliberately simple as they focus on total catch and the average annual flow anomaly in each region, but note that the MICE model resolves the finer scale details by attempting to estimate the catch each week of each year and for each region based on weekly cumulative flows. The model is also set up so that it can switch to units of daily flow and convert to flow volume estimates, for example, to explore hypotheses related to threshold flow levels.

Table 13. Summary of the 5 lowest *P. merguensis* catch years (using (A) catch totals over weeks 13-22 and (B) totals over entire year) shown alongside the average annual flow anomalies for each river system. The yellow highlights show the river system with the lowest flow anomaly (i.e. values less than one show that the flow that year was below average, sometimes as low as 1% or 10% of the average level).

(A) Based on catch totals over weeks 13-22

Lowest catch yrs	P. merg	Flow relative to average					
	Total Catch (t) wks 13-22	Embley	Mitchell	Gilbert	Norman	Flinders	Roper
2000	950.8	1.1	1.3	1.0	0.7	1.1	2.0
1978	745.1	0.6	0.2	0.1	0.0	0.3	0.7
1980	650.5	0.7	0.6	0.5	0.3	0.6	1.1
1982	537.5	1.4	0.3	0.2	0.1	0.2	0.7
1994	179.0	0.8	0.4	0.4	0.5	0.8	1.0

(B) Based on catch totals over entire year

Lowest catch yrs	P. merg annual	Flow relative to average					
	Top Total Catch (t)	Embley	Mitchell	Gilbert	Norman	Flinders	Roper
1986	1285.9	0.5	0.4	0.3	0.01	0.1	0.1
2000	1086.8	1.1	1.3	1.0	0.7	1.1	2.0
1983	1037.7	0.4	0.2	0.2	0.1	0.2	0.1
1990	1007.4	0.8	0.1	0.2	0.1	0.5	0.0
1994	940.6	0.8	0.4	0.4	0.5	0.8	1.0

The MICE model provides a robust tool to support decision-making because it is able to integrate dynamically across all regions and years and through using log-likelihood criteria for model-fitting (equivalent to methods used in stock assessment models) it can output model estimates with associated confidence intervals that provide a solid basis for projecting under future alternative scenarios. Similar approaches that include a mix of explicitly representing mechanistic processes and using statistical relationships are being applied for the other species in the model.

Model ensemble: Barramundi

For barramundi, all model versions linked flow with subsequent recruitment to the population and fishery, as well as influenced the survival of fish aged 5 years and younger (due to enhanced estuarine productivity). The model did not successfully converge when additional complexity was added, such as linking flow with the growth rate. This is likely because growth and survival are highly correlated in that (i) both are related to flow and (ii) faster growth fish/year classes are likely to have better survival. Hence these model versions are excluded here due to confoundment but could be investigated further in future work. The numbers of new 1-year old barramundi recruiting to the population for the first time was assumed dependent on average flows the previous year, and the survival of this cohort plus animals up to age 5 was also influenced by flow and hence there was around a 3-5 year lag before changes in flows translated into changes in abundance and catches (with fishing selectivity estimated to increase steeply from around age 3 – Figure 34). The multiple age classes in the population and the fishery and smoothing of the flow-abundance signal over time considerably complicates estimation of flow-abundance relationships in barramundi compared with more short-lived species such as common banana prawns or mud crabs.

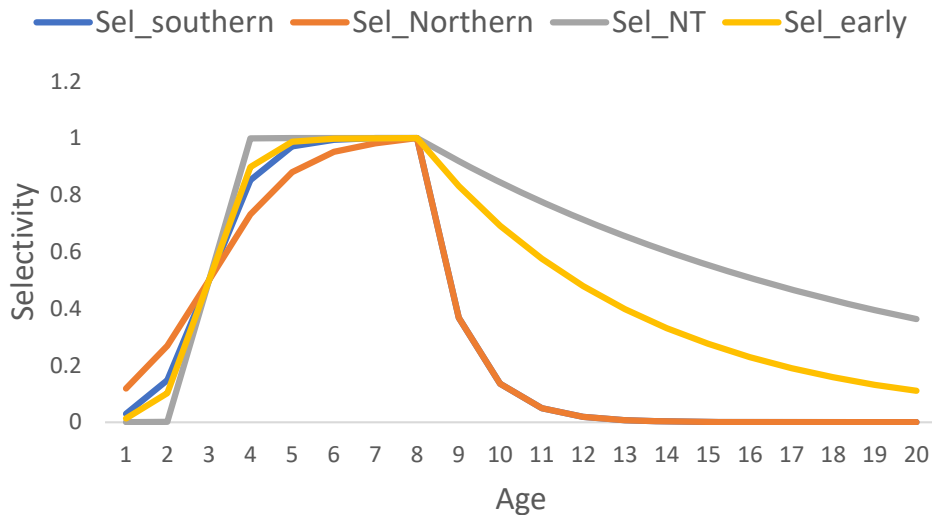


Figure 34. MICE estimated base selectivity curves for the barramundi fishery applied to the northern Queensland region (Regions 1 and 2), southern Queensland regions (Regions 3-6) and Northern Territory (Region 7-8), as well as the “early” selectivity curve used for the Qld model regions prior to 1997 (see Appendix 16 for derivation).

An age-dependent mortality form was found to better represent barramundi population dynamics and the model was able to reliably estimate the parameter controlling the magnitude of this relationship for different slope fixed values. The base-case estimate (Model 1) of barramundi natural mortality (with monthly estimates here translated to annual estimates for ease of comparison) suggested much higher mortality of younger animals, ranging from around 0.37/yr for age 1 and decreasing to around 0.2/yr for 4-year old animals and levelling at a lower rate of around 0.13/yr for older animals (Figure 35). The same natural mortality rate was applied to all regions and jurisdictions, but fishing mortality rates varied based on fishing effort recorded in each region. A sensitivity to higher mortality rates and a different slope of the age-dependent relationship were also included as part of the barramundi ensemble.

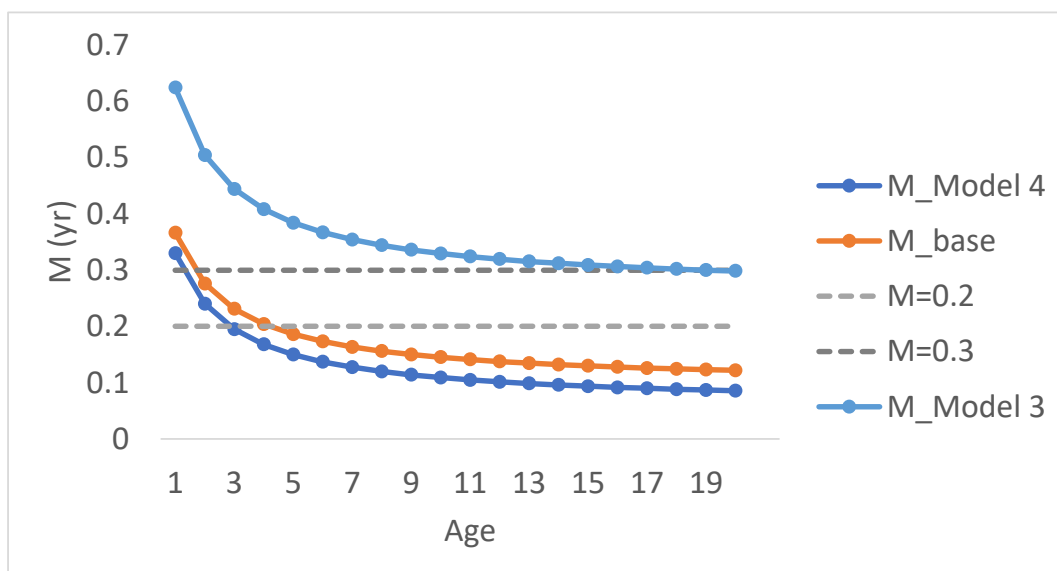


Figure 35. MICE estimated age-dependent natural mortality (shown as annual values converted from the monthly mortality rates used in the model) showing the range of forms used in the model ensemble. The “base” M is used in Models 1,2 and 5. The dashed grey lines are simply for ease of comparison with the simpler assumption (not used in the MICE) that M is constant with age, with annual rates of 0.2 or 0.3 yr⁻¹ as commonly used in stock assessment. See Table 4 for a description of the model versions.

Based on feedback during the project workshops, it was felt that the catch and effort data provided for barramundi in Region 2 likely did not represent the dynamics in the northern section of this large model region, and hence the model was fitted instead to the more southerly subset of the barramundi data, which was considered more likely influenced by the Mitchell River catchment system. The model was also fitted to catch-at-age data (CAA) (see Appendix 12 for details) available for the Qld south-eastern barramundi fishery (assumed to correspond to model regions 3-6 and labelled “southern” CAA) and Mitchell catchment (assumed to correspond to model region 2 and labelled “northern” CAA). The CAA data were informative in estimation of the age-dependent natural mortality rates as well as to track cohorts through time (with the model attempting to explain fluctuations over time as a response to changes in flow, with a lagged response as describe above). However, consistent with the philosophy of not over-parametrising the model, there were a number of trade-offs in the model trying to fit to both the monthly catch data per region as well as the annual catch-at-age data. Overall, the MICE was fitted to a large amount of data to inform parametrisation of the barramundi component, and this provided a strong basis to support model outputs. The model summed likelihood contributions from monthly catch data for each of seven months of the fishery open season (February to September inclusive) over a 30-year period (210 monthly comparisons for each of the model regions) in addition to fitting to the two annual CAA series.

As for common banana prawns, the addition of flow as an explanatory variable significantly improved model fits (see Appendix 12). For ease of viewing (and because there are too many plots otherwise), model fits are shown as the combined annual totals (over the key fishing period February to September) but the model likelihood function sums the contribution of fits to the monthly catch data (Table 10).

As was done for common banana prawns, a logistic relationship was used to model the influence of flows on recruitment and mortality. The different model versions incorporate different parametrisations of this relationship (Table 4). Model version 1 fixed these parameters at the same values as used for prawns, noting that instead of weekly standardised flow indices, the index for each month was computed as an average of the weekly standardised flows. In comparison, Model version 2 estimated the parameters of the flow relationship, with the corresponding parameter estimates for each model region having small associated standard deviation (i.e. good precision in estimates) and this model is the preferred model based on the AIC score (Table 10). Model results again suggested there are important differences in the rate and point at which decreases in flow impact barramundi population dynamics (Figure 36). For example, for barramundi, Model version 2 estimates that decreases in flow in the Embley and Mitchell Rivers have the greatest relative impact whereas for the Gilbert, Flinders and Roper Rivers changes in flow are estimated to have a more gradual linear effect (Figure 36).

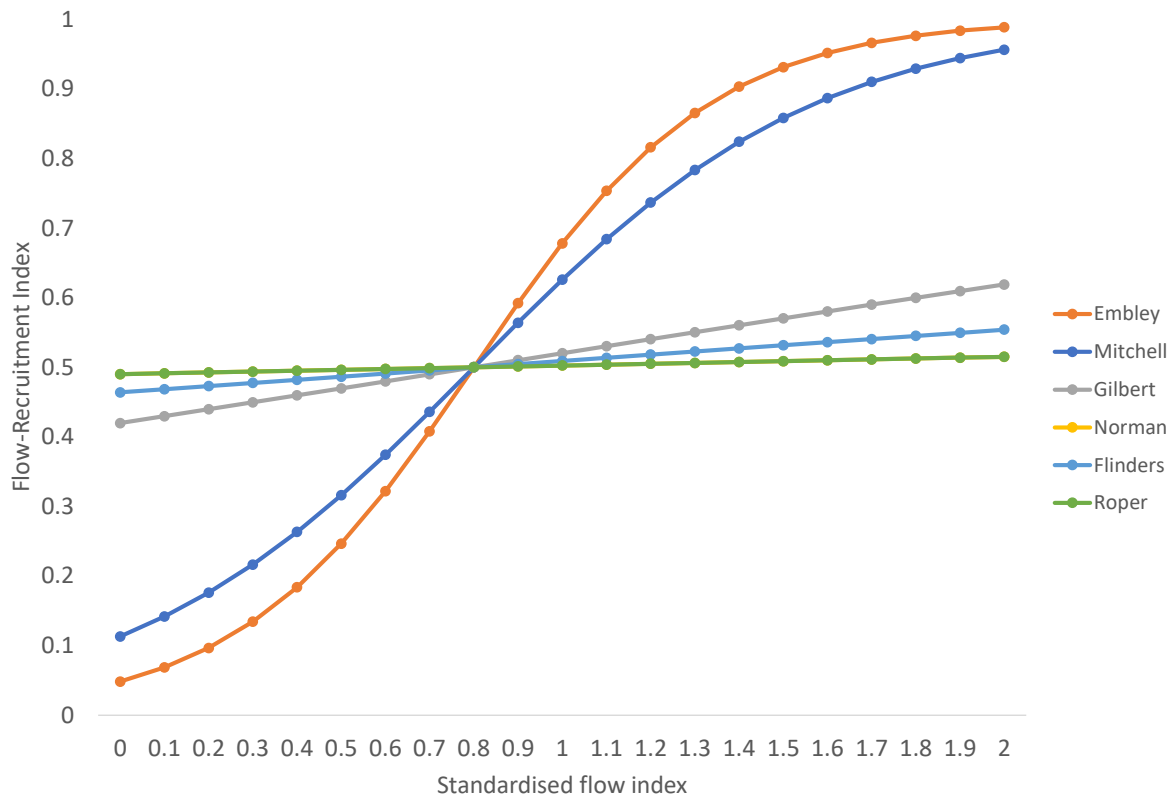


Figure 36. MICE estimated logistic functions used to describe the relationship between the standardised flow and barramundi recruitment for each of the catchment systems as shown, where the function yields a multiplier that describes recruitment relative to the maximum value.

Another key sensitivity for barramundi relates to the assumption that skewed sex ratios may negatively impact reproductive success (see Appendix 12). This formulation yielded a lower AIC score than models excluding it (Model version 5 – Table 10) and also improved the fit to the CAA data for the Mitchell region. As there were some differences in these alternative plausible representations of barramundi recruitment, both versions were included in the ensemble.

A comparison of the model fits for Model version 1 and Model version 4 (the best fit model) are shown in Figure 37 and Figure 38, with the full set of results shown in Appendix 17. Model version 4 fits to the CAA data for Region 2 and Regions 3-6 combined are shown respectively in Figure 39 and Figure 40, with the full set of results shown in Appendix 17. This again highlights the kind of trade-offs that result in the fit to the different regions and hence why it is useful to use an ensemble approach when evaluating the impacts of alternative WRDs.

The range of plausible population abundance and catch trajectories estimated under each model version is shown in Figure 41-Figure 42. The ensemble of commercially available biomass and catch trajectories is shown for regions 2, 3 and 5 given these are the key focus of the current WRD simulations, and then also for regions 2-6 combined (Figure 41-Figure 42). Summary statistics of the differences between all our model versions are presented in more detail in Appendix 17.

In the following section we present the results of predicted responses to changes in baseline flows under alternative WRDs using the base case model (Model 1), as well as differences between the model versions comprising our ensemble (Models 1-5), in terms of predicted responses of barramundi abundance and catches to alternative WRDs.

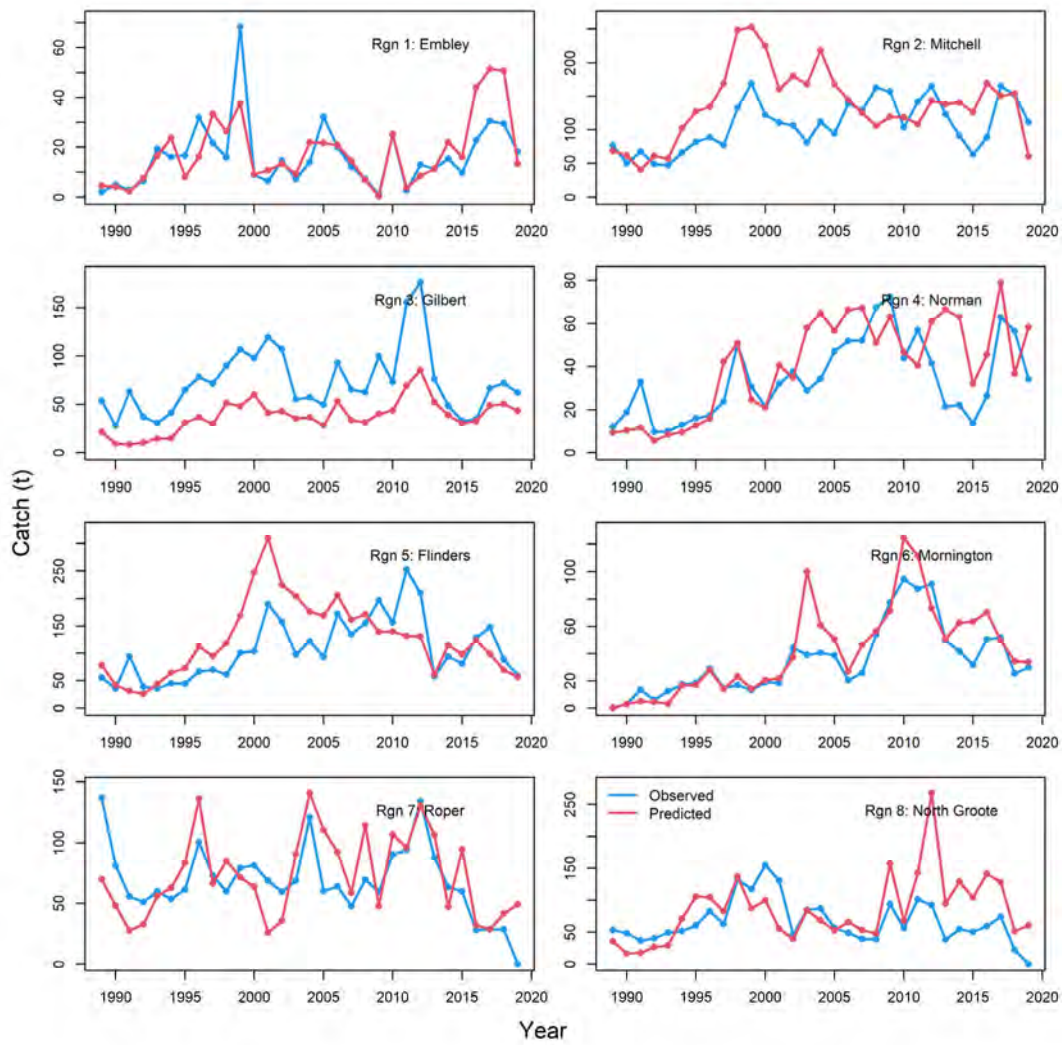


Figure 37. Comparison of the observed and model-predicted annual barramundi catch (tonnes) estimated using Model version 1 (baseline flows, effort, fixed flow relationship parameters) and for each of the eight model regions as shown.

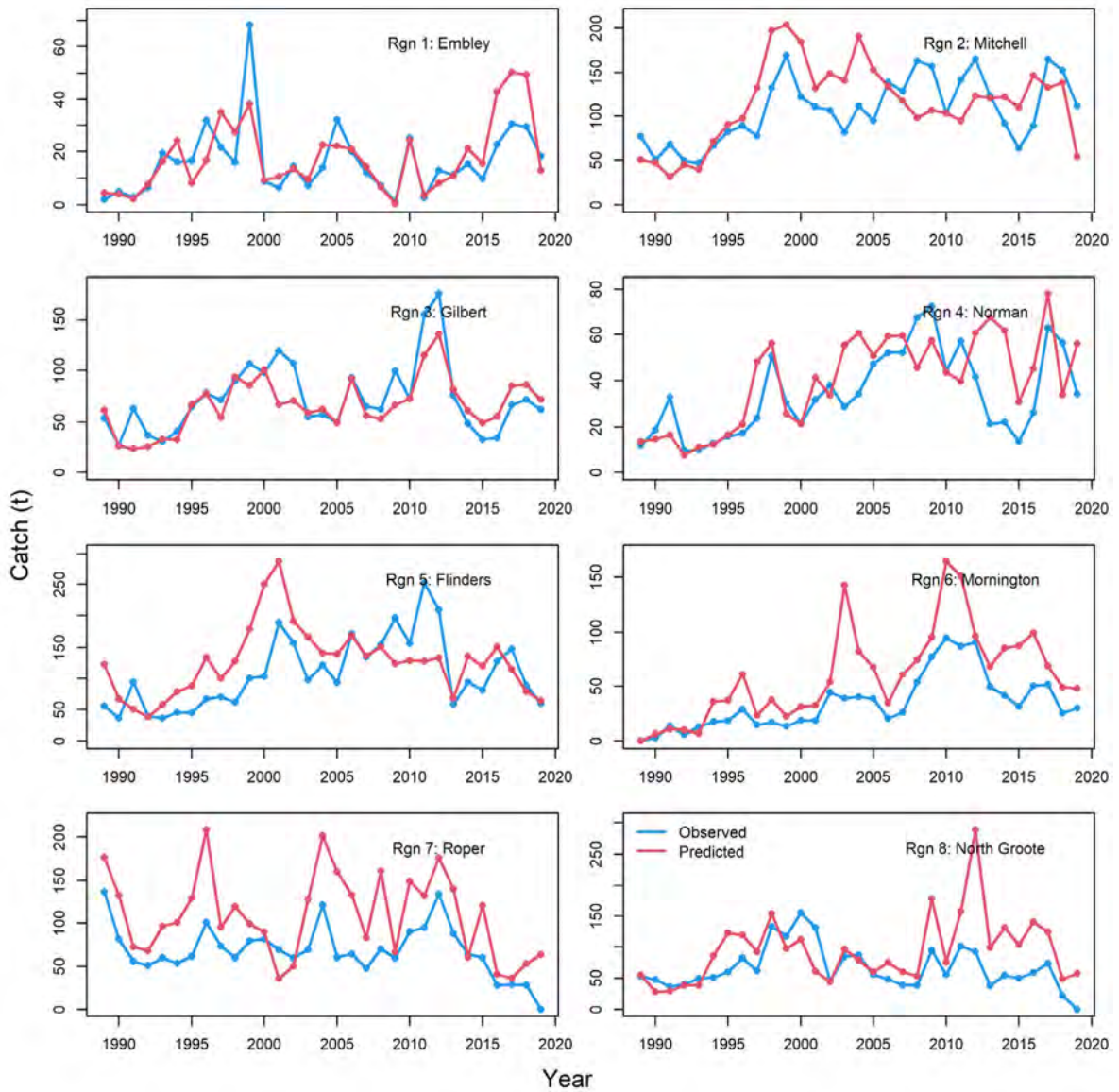


Figure 38. Comparison of the observed and model-predicted annual barramundi catch (tonnes) estimated using Model version 4 (baseline flows, effort, estimated flow relationship parameters and lower natural mortality rate) and for each of the eight model regions as shown.

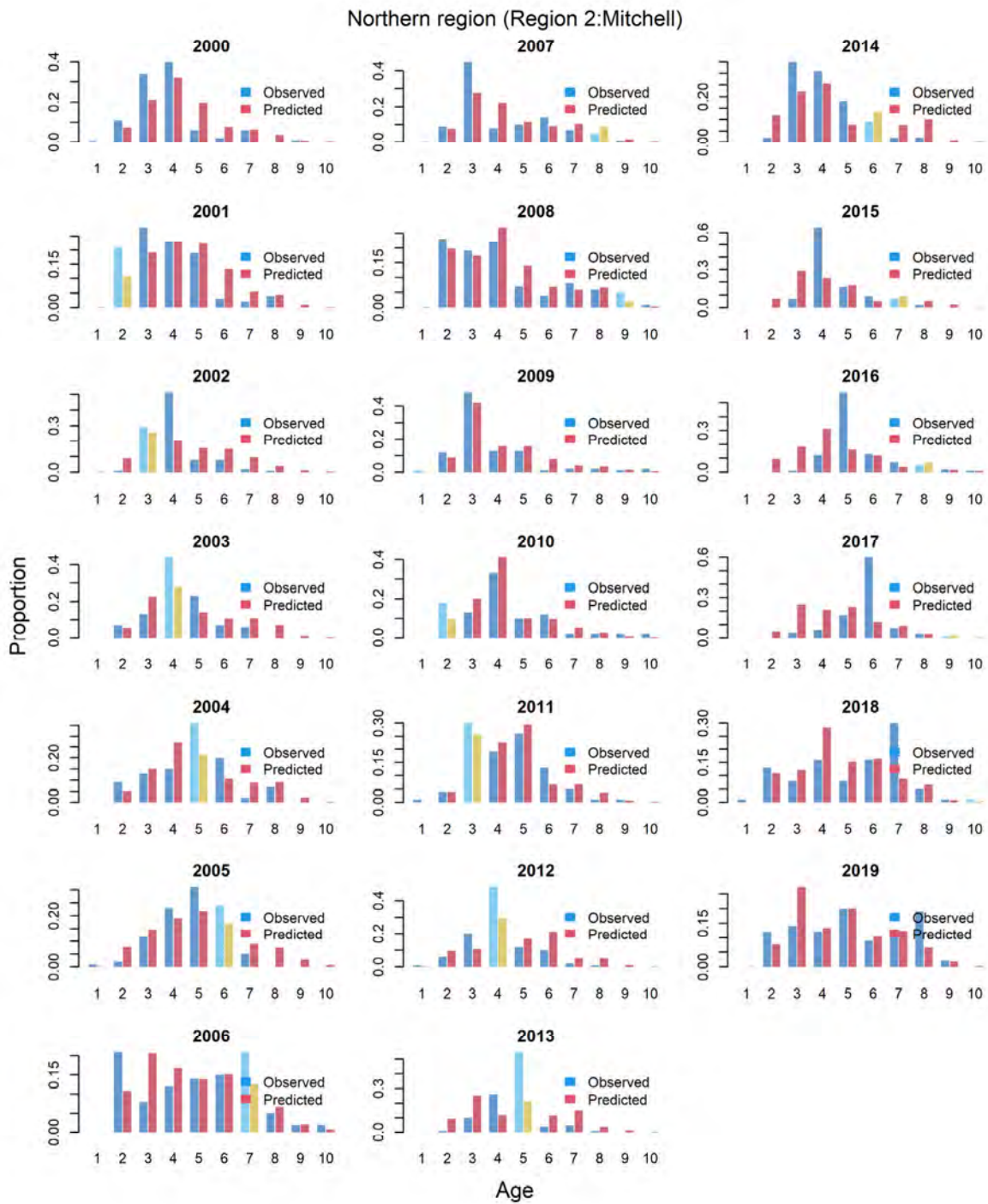


Figure 39. Comparison of the observed and model-predicted annual commercial catch-at-age proportions of barramundi in each age-class as shown estimated using Model version 4 (baseline flows, estimated flow relationship parameters) for the Mitchell River catchment region from 2000 to 2019. Light blue (observed) and gold (predicted) bars indicate strong cohorts (otherwise known as year-classes) that can be followed over time through the catch-at-age proportions.

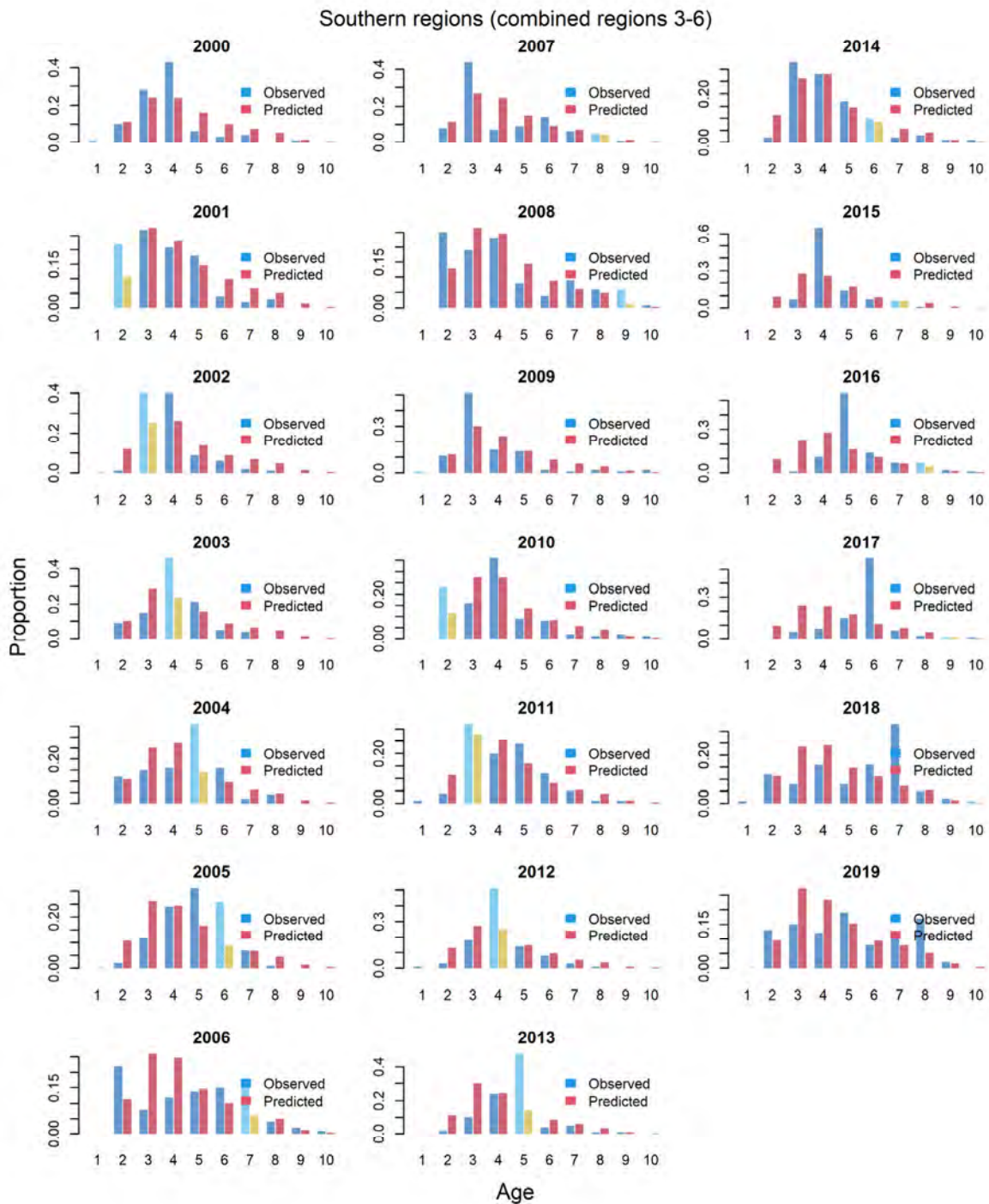


Figure 40. Comparison of the observed and model-predicted commercial catch-at-age proportions of barramundi in each age-class as shown estimated using Model version 4 (baseline flows, estimated flow relationship parameters) for the combined catchment Model regions 3-6 from 2000 to 2019. Light blue (observed) and gold (predicted) bars indicate strong cohorts (otherwise known as year-classes) that can be followed over time through the catch-at-age proportions.

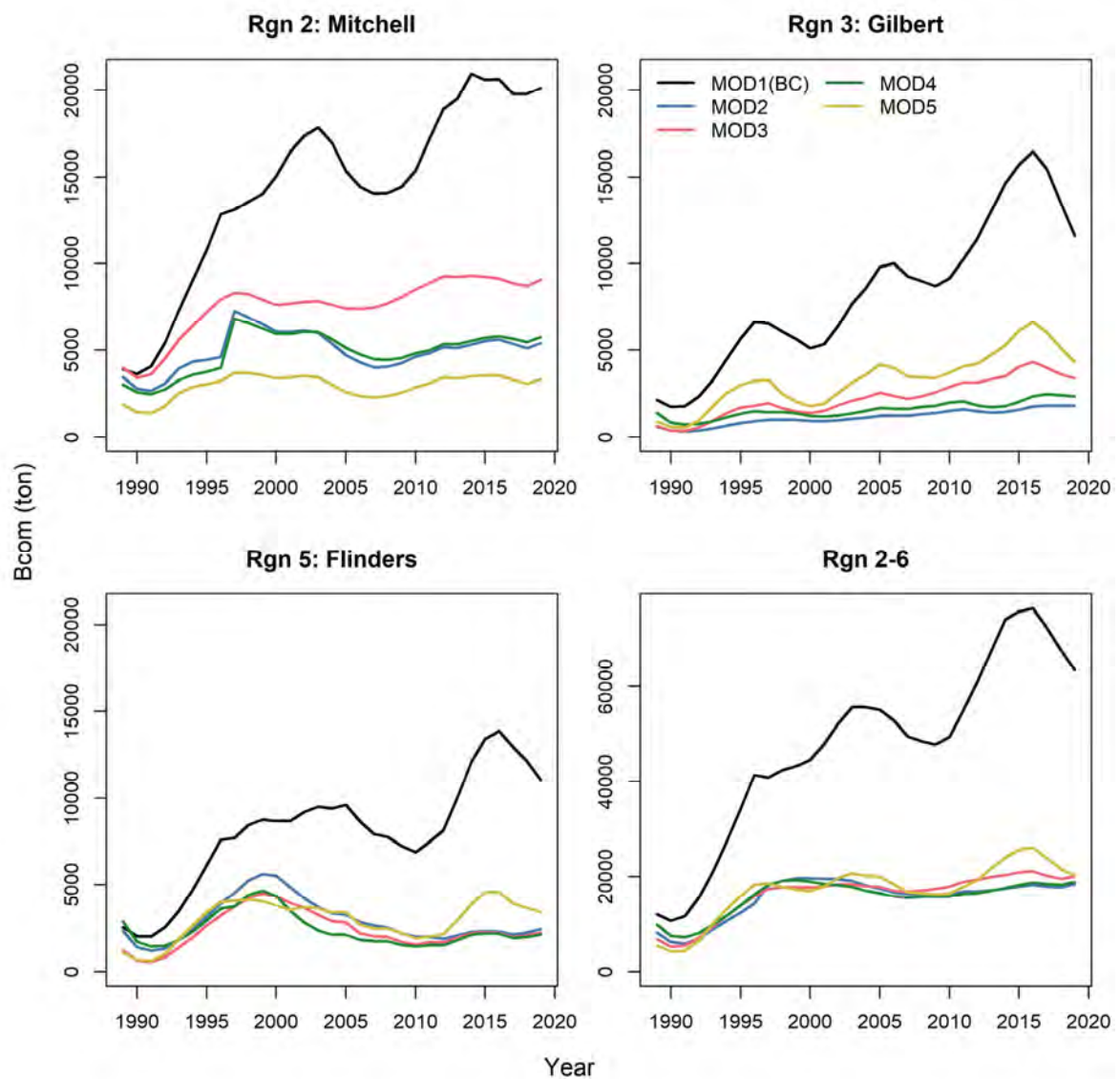


Figure 41. MICE ensemble outputs shown for the five alternative model configurations used to represent barramundi. Trajectories show the model-estimated commercially available biomass (Bcom) in units of tonnes shown from 1989 to 2019, for the different model regions and jurisdictions as indicated. See Table 4 for a description of the model versions.

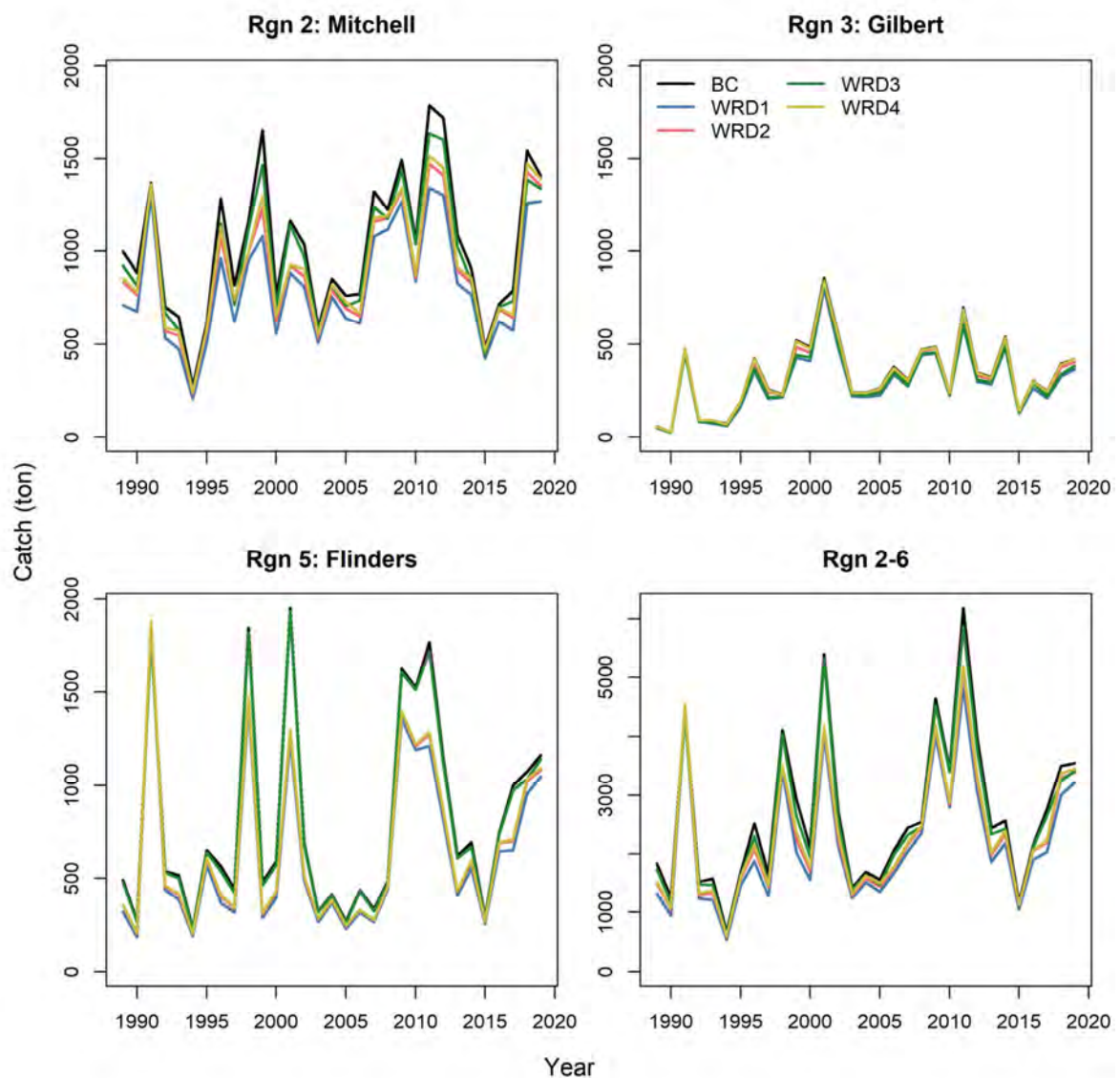


Figure 42. MICE ensemble outputs shown for the five alternative model configurations used to represent barramundi. Trajectories show the model-estimated catch (tonnes) from 1989 to 2019, for the different model regions and jurisdictions as indicated. See Table 4 for a description of the model versions.

Model ensemble: Mud crab

All five versions of the model linked flow to mud crabs, either through recruitment (all model regions) or also through catchability (model regions 3-6). Some model regions also included other secondary environmental drivers (in all model versions) e.g. SOI was included as an additional driver for model regions 3-6 and air temperature was included for model region 7. We added additional drivers in a step-wise process as needed and based on hypotheses (see section 9). As flow was considered the primary driver for fluctuations in mud crab abundance and catch, we have focused on this driver here.

River flow has been hypothesised to impact mud crabs in different ways. Increased river flow is thought to have a positive effect on mud crab abundance and catch, whereby it might increase the catchability of mud crabs by causing them to move downstream (away from low salinity) and into fishing grounds (Loneragan and Bunn 1999, Robins et al 2005, (Butcher et al. 2003, Meynecke et al. 2010). As such, less adults upstream would result in reduced competition for burrows and reduced cannibalism by larger crabs. This could lead to increased recruitment of juveniles into the fishery (i.e. more juveniles survive), possibly causing increased catches the following year (lag effect). However, it is also hypothesised that too much river flow could negatively impact mud crabs. Small catches in the GoC have been linked to high migration rates out of fishing areas and recruitment failure, as a consequence of extended periods of freshwater run-off (Helmke et al. 1998). Often the effects of heavy and prolonged flooding are evident with a 1-2 year lag affect (Meynecke et al. 2012a). These negative impacts are thought to be linked to die-off of seagrass beds, which are important habitat for newly settled and juvenile mud crabs (Meynecke et al. 2012a), but could also be the result of mud crab larvae not being able to move inshore and settle during prolonged heavy rains.

Based on these hypotheses, we linked end of system river flow to (1) boost or reduce recruitment (defined as number of crabs surviving and entering the 1-year old age class) the following year and/or (2) increase catchability of adult crabs. The former translated into either increased or decreased available biomass to be fished approximately 18 months after flows, while the latter resulted in increased catchability during the same period (month) as the experienced flow. We linked flow to mud crab recruitment by calculating a flow residual which either boosted or reduced recruitment based on a second-order polynomial (parabolic) relationship (see Appendix 16 equations). This dome-shape relationship (as opposed to a linear or logistic relationship) was considered the most likely because it increased mud crab recruitment at good/optimal flows (peak of the dome) and reduced mud crab recruitment when flows were below or above this optimum threshold (slope of the dome).

There is little information to inform the optimum flow for mud crabs in each region. We considered that optimal flows would likely be above the regional long-term average but couldn't be too elevated as extreme flood years were thought to negatively impact mud crabs. Thus, we consulted time series flow-catch plots for each region to observe levels of good flow and initially tested a range of flows above the long-term average. All model regions seemed to perform best when the optimum flow (dome shape relationship) was set at 2.0 - 2.5. We thus used this as our basis for the base-case model (Model 1), and estimated the strength (slope) of this relationship.

The mud crab component of the model was fit to monthly catch data from 8 model regions either from 1983 (regions 7-8, NT) or 1989 (regions 1-6, Qld). This resulted in the model fitting to 2376 observations. For almost all model regions there were adequate time series of data, except for perhaps region 6 (81 observations), where catches were low and sporadic. We nonetheless fit the model to catches for this region, but it is not a focus of this report. Similar to barramundi, model region 2 contained mostly catch and effort data from the southern part of the region around the Mitchell River catchment and hence it was decided for mud crabs to fit the model to the more southerly subset of the data, which was considered more likely influenced by the Mitchell River. Model fits are shown for annual catches (Figure 44-Figure 46) as there are too many plots to show monthly fits for each year in each region.

Under Model 1, effort alone was able to explain mud crab catches for some but not all the regions in the GoC. For example, predicted catch followed the general trend in observed catch over the model period for Regions 1, 2 and 8 but not for Regions 3, 4, 5 and 7 (Figure 44). In Regions 3,4,5 and 7 in particular, the model was not able to capture the sharp peaks in catches over the early 2000s and 2009-2013 periods (periods of good flow). The addition of flow linked to recruitment greatly improved the overall -ln likelihood from 737.8 (no flow linked, only effort used to explain the catch) to 553.7 (flow linked to recruitment) and almost all regions had substantially better likelihoods (Appendix 11). For some regions, these large peaks in catches could only be captured by the addition of flow or flow and an additional environmental explanatory variable. The addition of flow linked to catchability generally improved the likelihoods but not for all regions and thus we only included this

for regions 3-6. The addition of other drivers that were hypothesised to influence mud crab dynamics were also considered, such as SOI (regions 3-6) and temperature (region 7) and these resulted in a much better model fit (Figure 46) and overall -ln likelihood (Appendix 11), capturing the peaks and dips in catches relatively well. In some cases, fits might not appear very good in a particular year, but it could be a case of the model fitting really well in some months and poorly in other months. For example, for Region 7 the model fails to predict large catches in the 2000s which is amplified on the annual scale. However, if one considers the monthly fits to the data, for some months in the 2000s these peaks are captured better than in other months and so the net overall effect is slightly dampened (see Appendix 11).

The strength of the flow relationship varied between regions, with a weak flow relationship estimated for the Embley-Mitchell rives (regions 1-2), followed by the Roper river (regions 7-8) and a very strong flow relationship being estimated for the Gilbert-Norman-Flinders (regions 3-6) rivers (Figure 43). The relationships, particularly for the eastern GoC, were estimated with high precision shown by the narrow 90% confidence intervals in Figure 43. There was less precision (wider 90% CI) in the flow relationship for the Roper river relative to rivers in the eastern GoC. Given less precision in the Roper flow relationship and the model struggling to predict very large catches, future modelling work may need to consider sub-dividing this region as there may be slightly different dynamics at play across other catchments within Region 7.

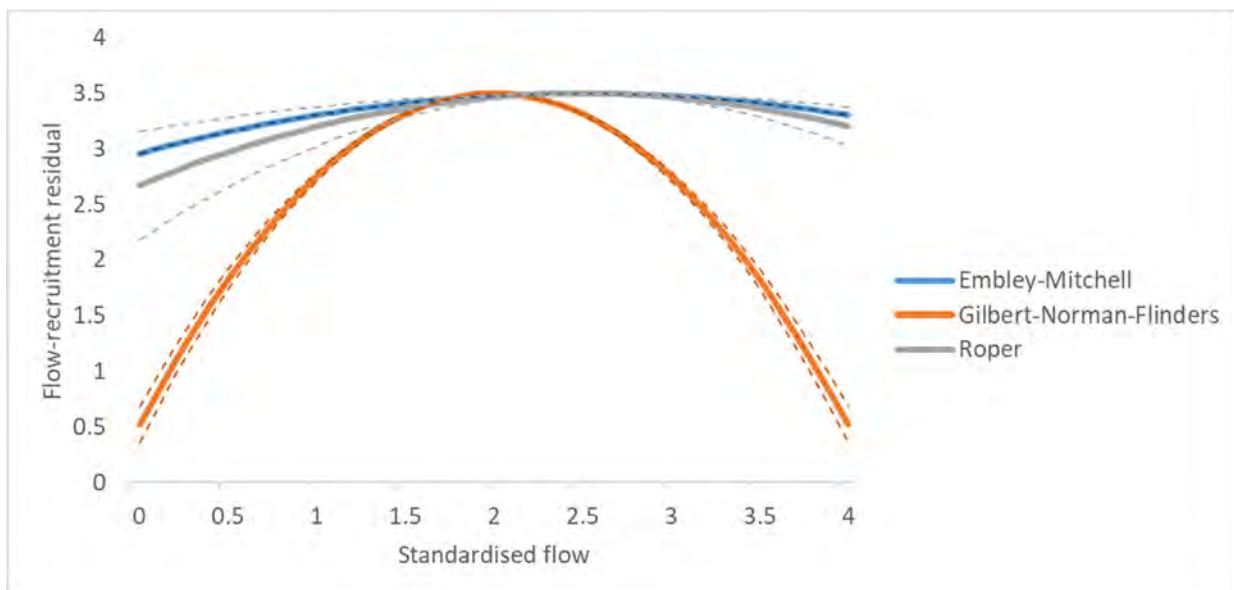


Figure 43. MICE estimated parabolic functions used to describe the relationship between the standardised flow and mud crab recruitment for each of the catchment systems as shown, where the function yields a multiplier that describes recruitment relative to the base value. 90% confidence intervals are showed by dashed lines, colour-coded for each region.

The model is likely to be quite sensitive to flow relationships and hence we included an alternative model version in which the optimal flow was reduced by 20% (Model 2). This resulted in a worse overall fit for the mud crab component of the model. Some regions had a substantially worse fit (region 1, 3-5) while others showed little difference (Regions 2, 6-8) (Table 10). We also considered a logistic relationship in which crab recruitment increased with flow and was maximised under an optimal flow and then remained at that level even under increased flows. However, based on the literature (summarised above) and expert opinion, we do not think that this relationship accurately

reflects mud crab-flow dynamics but nonetheless trialled it in the model and found that it resulted in a significantly worse fit for all regions (see Appendix 11).

Like many species, natural mortality of mud crabs is likely to vary across age, space and time but data are unavailable to reliably parametrise a variable mortality. A study by Knuckey (1999) used three different methods to calculate natural mortality for mud crabs from rivers in the western GoC. The average mortality from this study has been used by Grubert et al (2019) to model mud crabs in the western Gulf of Carpentaria mud crab fishery and similarly we used this mortality in Model 1. As one of the key sensitivities for mud crabs, we tested a larger natural mortality (20% increase) in Model 3. This model performed somewhat better than the base case model as indicated by the AIC score (Table 10), with slightly improved likelihoods for almost all regions. This is not surprising given that the base case model does not capture additional sources of mortality e.g. presumably larger mortality of juvenile crabs (which are subject to cannibalism from adult crabs), as well as increased mortality of female crabs that migrate offshore to spawn (Hill 1995).

Our base-case Model 1 estimated a small starting biomass for mud crabs at the beginning of the model period (1970). The model period between 1970 and the start of the mud crab fishery is considered a “burn-in” period in which the modelled biomass builds up prior to the fishery starting in 1983 (Regions 7 and 8) and 1989 (Regions 1-6). There is little information to inform what might have been an expected biomass in this period and thus we ran a model sensitivity by changing the start biomass. Additionally, for Region 7, we struggled to predict the large catches in the early 2000s and thus wanted to test whether a larger model starting biomass might help achieve those large catches. Hence, in Model 4 we doubled the start biomass. This sub-model version performed the worst of the ensemble with the largest AIC score (Table 10). However, an increased starting biomass did allow the model to better predict the peak catches in the 2000s for Regions 7-8 but with the trade-off that it over-predicts catches for Region 7 in 2011-2012 (Appendix 17) and thus the fit for this region was worse. Further refinements for this model region should be explored in future work.

Our final sensitivity for the mud crab component involved using a more conservative spawner-recruit steepness parameter h . In mud crab sub-model versions 1-4, we used a steepness parameter of $h = 0.6$, which takes into account that mud crab recruitment is defined as 1 year old crabs and that other density-independent factors affect mud crab dynamics during this first year (age 1). Given the multiple environmental drivers hypothesised to influence mud crab dynamics, in Model 5 we thus tested a more conservative value of $h = 0.4$, in which recruitment declines sooner in response to a reduction in spawning stock. Of the mud crab sub-model ensemble, this model version performed the best with the smallest AIC score and substantially improved model fits (likelihood scores) for Region 3 and Region 4, but little change or worse fits for Region 5, Region 6, and Region 7 (Figure 46). However, improvements to the fits, both for model 5 and model 3, compared with model 1, are not noticeable in the observed vs predicted catch plots (Figure 46 and Appendix 17) and in all instances, these three models are broadly consistent with the data and provide a similarly good representation of mud crab catch trends.

A comparison of the model fits for Model version 1 and Model version 5 (the best fit model) are shown in Figure 45 and Figure 46, with the full set of results shown in Appendix 17. This again highlights the kind of trade-offs that result in the fit to the different regions and hence why it is useful to use an ensemble approach when evaluating the impacts of alternative WRDs.

The range of plausible population abundance and catch trajectories estimated under each model version is shown in Figure 47-Figure 50. The ensemble of biomass and catch trajectories is shown first for all model regions, and combined Qld and NT jurisdictions (Figure 47-Figure 48), followed by a closer focus on regions 2,3 and 5 given these are the key focus of the current WRD simulations (Figure 49-Figure 50). Summary statistics of the differences between all our model versions are presented in more detail in Appendix 17.

In the following section we present the results of predicted responses to changes in baseline flows under alternative WRDs using the base case model (Model 1), as well as differences between the model versions comprising our ensemble (Models 1-5), in terms of predicted responses of mud crab abundance and catches to alternative WRDs.

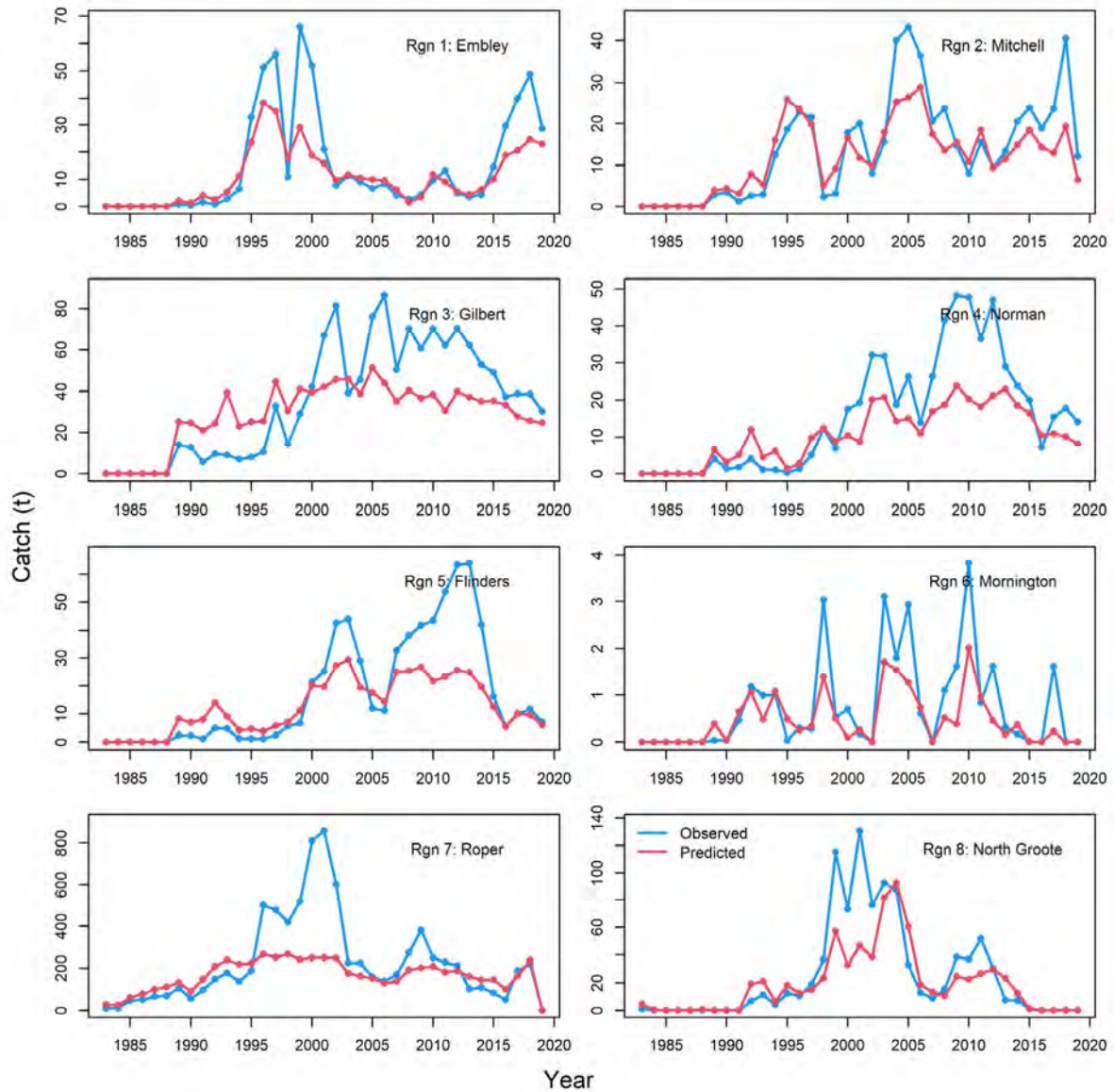


Figure 44. Comparison of the observed and model-predicted annual mud crab catch (tonnes) estimated using only fishing effort (i.e. no flow linked to mud crab population dynamics) for each of the eight model regions as shown. Note fishing for regions 1-6 only start in 1989.

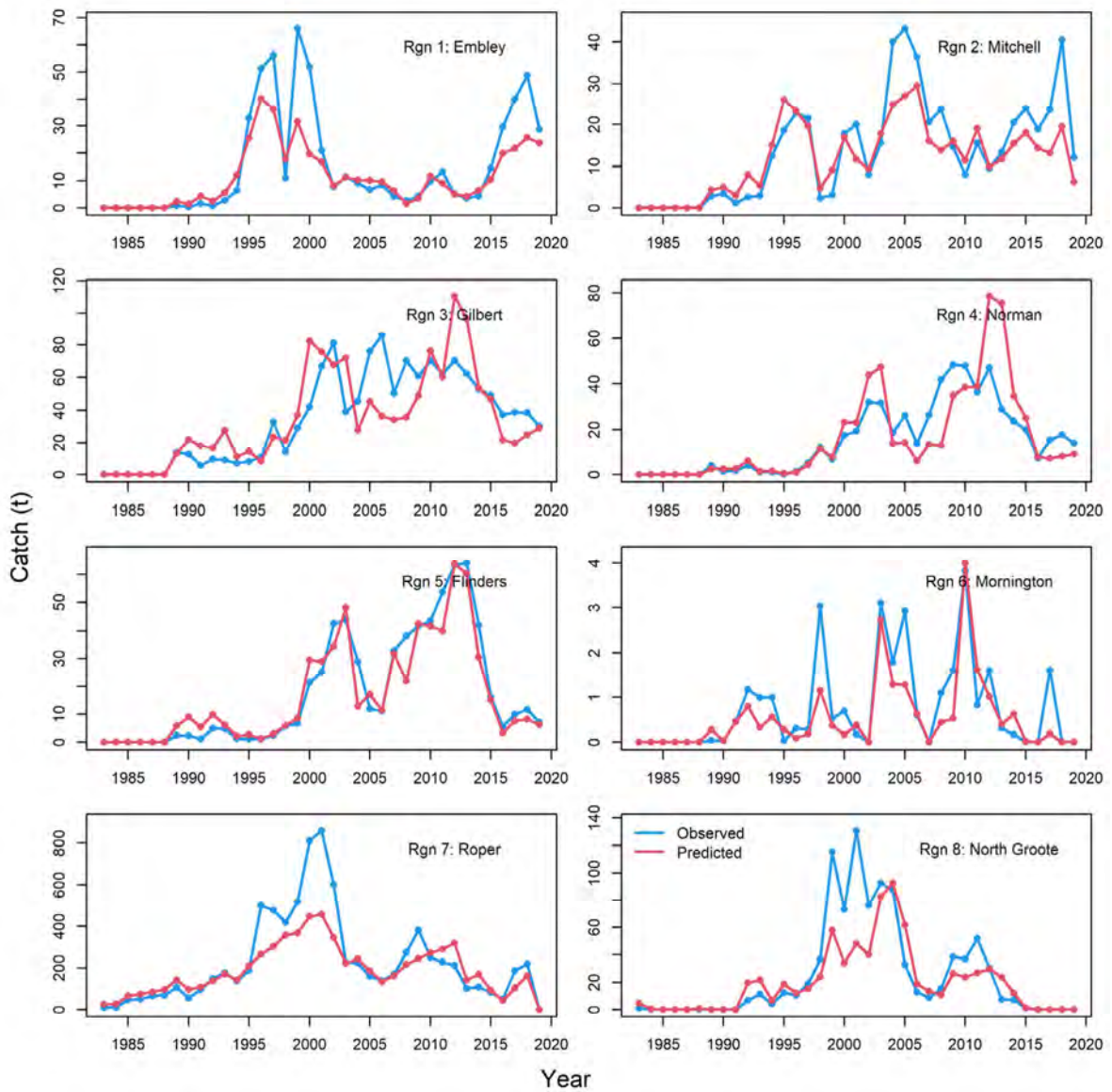


Figure 45. Comparison of the observed and model-predicted annual mud crab catch (tonnes) estimated using Model version 1 (driven by baseline flows, effort and other environmental predictors e.g. SOI for regions 3-6 and temperature for region 7) and for each of the eight model regions as shown. Note fishing for regions 1-6 only start in 1989.

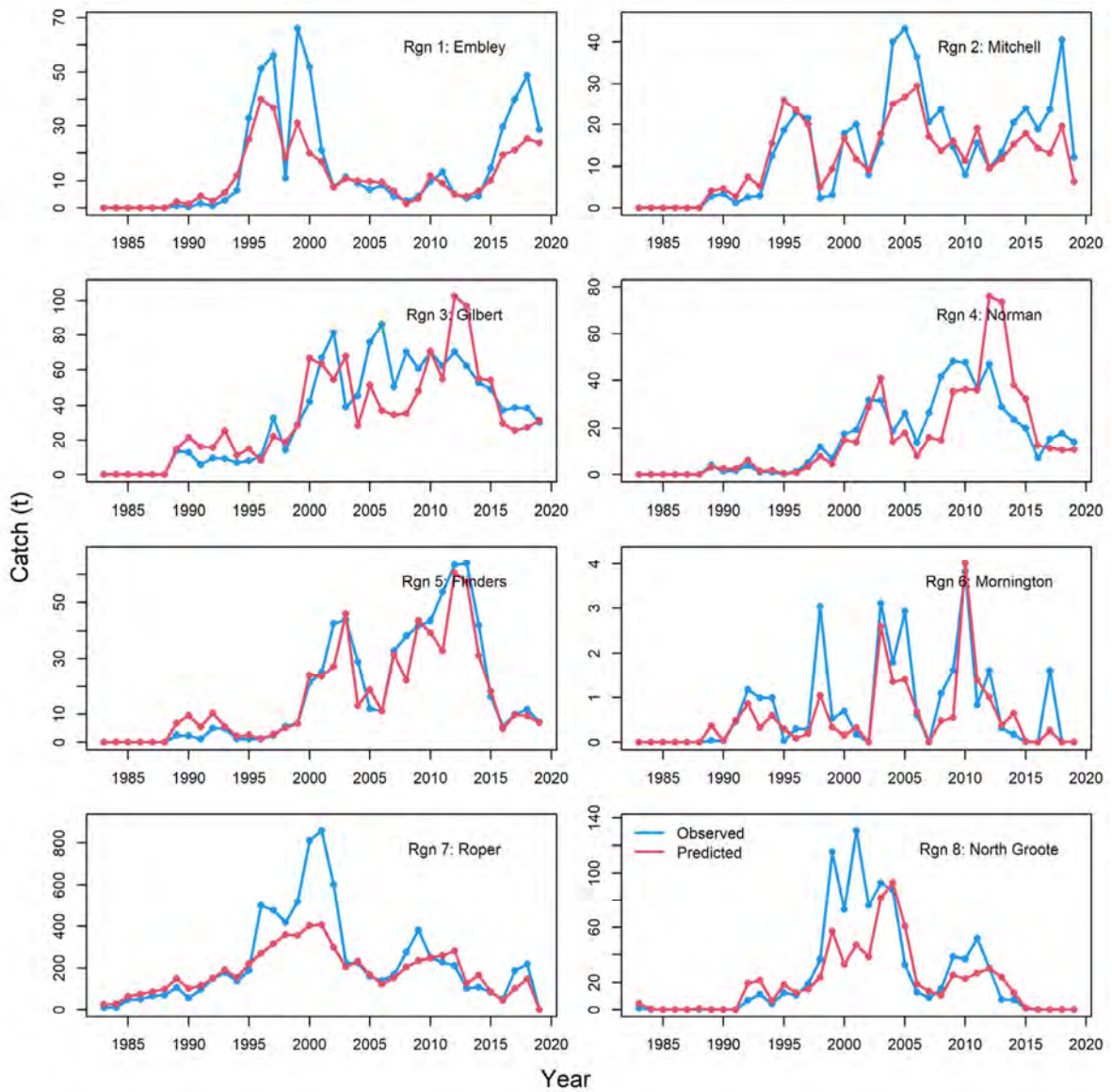


Figure 46. Comparison of the observed and model-predicted annual mud crab catch (tonnes) estimated using Model version 5 (driven by baseline flows, effort and other environmental predictors e.g. SOI for regions 3-6 and temperature for region 7, and a more conservative stock-recruitment steepness parameter) and for each of the eight model regions as shown. Note fishing for regions 1-6 only start in 1989.

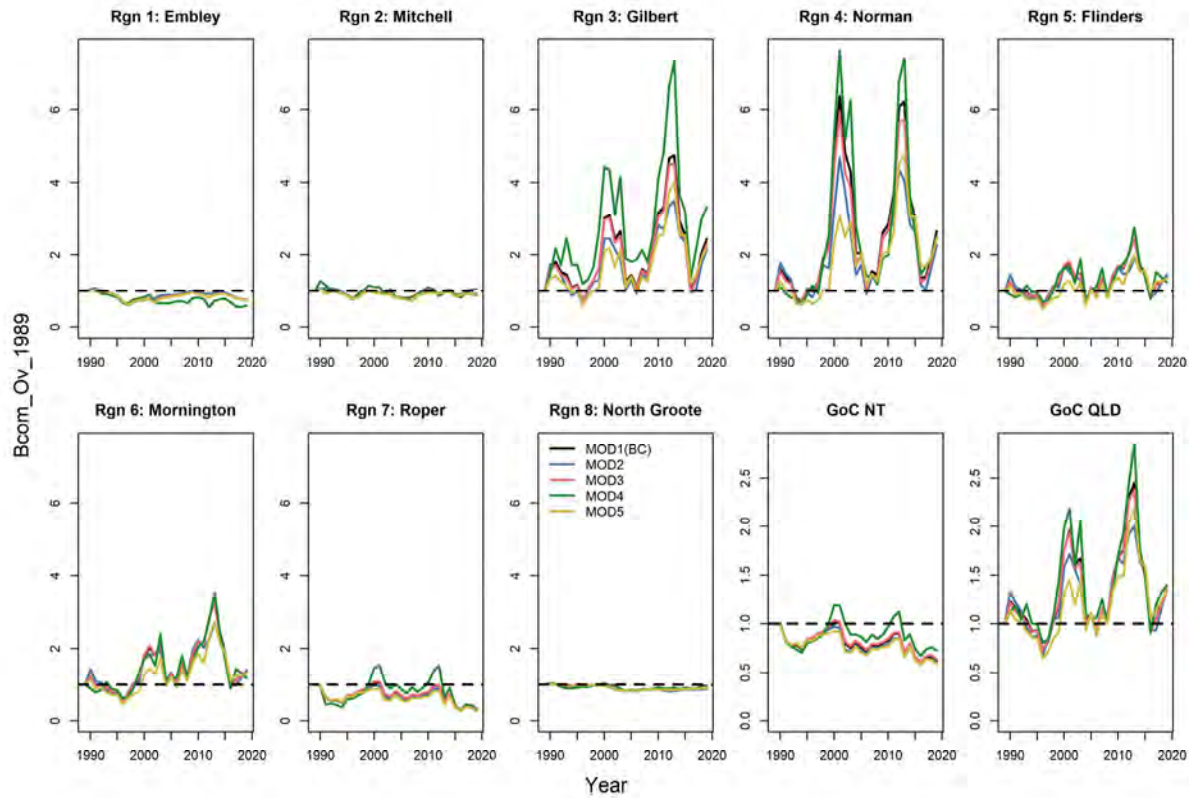


Figure 47. MICE ensemble outputs shown for the five alternative model configurations used to represent mud crabs. Trajectories show the model-estimated commercially available biomass (Bcom) plotted as relative to the 1989 available biomass, for the different model regions and jurisdictions as indicated. See Table 4 for a description of the model versions.

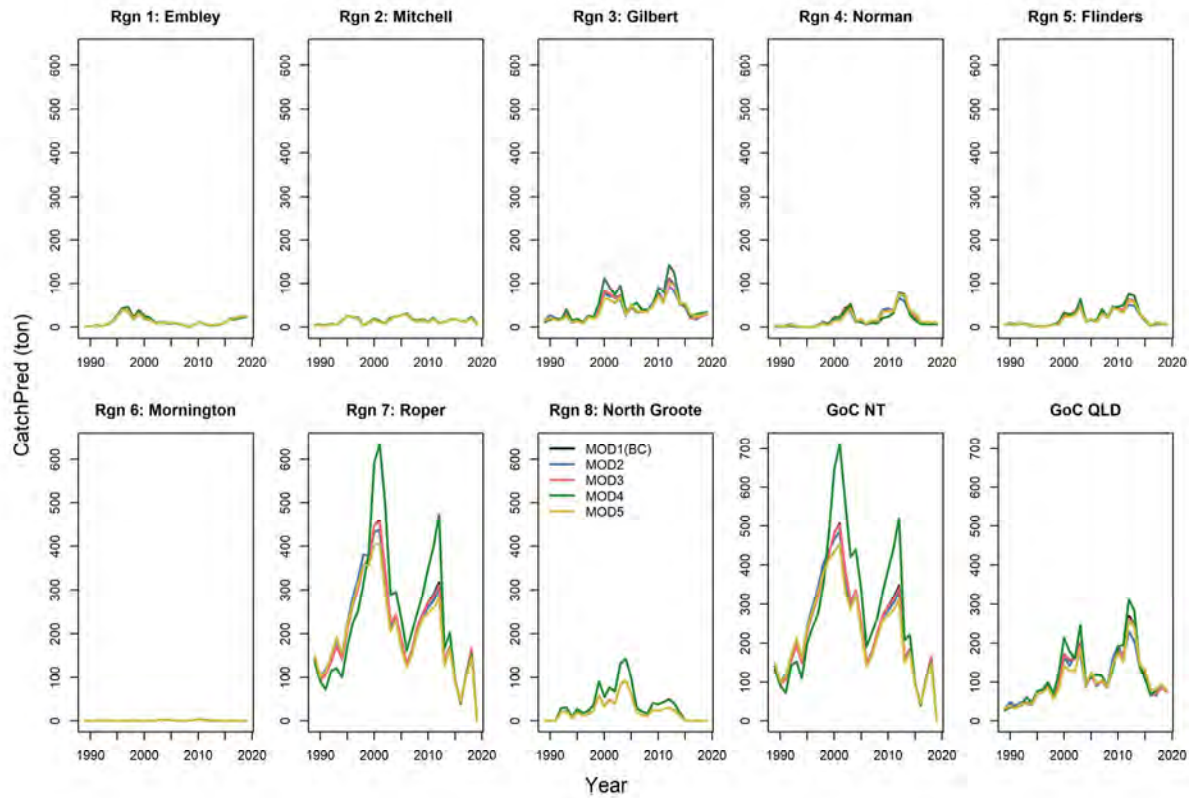


Figure 48. MICE ensemble outputs shown for the five alternative model configurations used to represent mud crabs. Trajectories show the model-estimated catch (tonnes) from 1989 to 2019, for the different model regions and jurisdictions as indicated. See Table 4 for a description of the model versions.

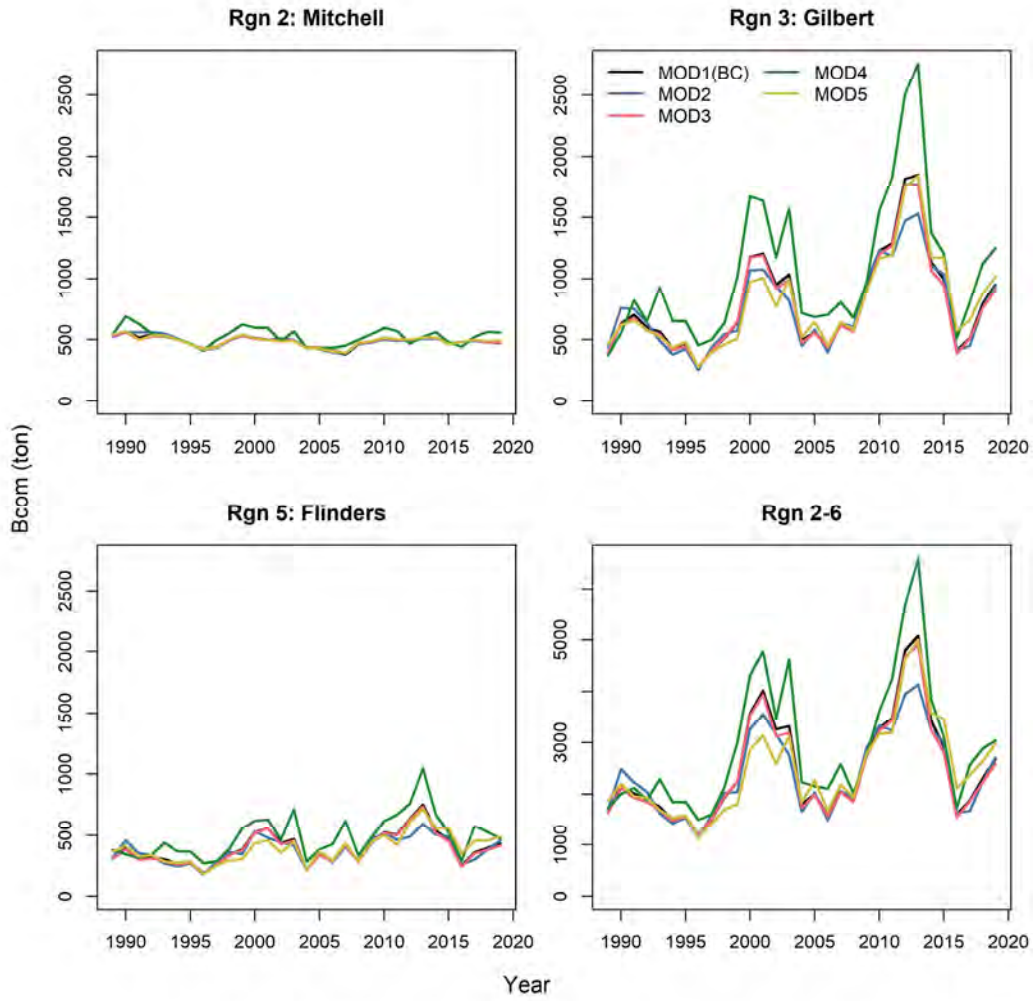


Figure 49. MICE ensemble outputs shown for the five alternative model configurations used to represent mud crabs. Trajectories show the model-estimated commercially available biomass (Bcom) in units of tonnes shown from 1989 to 2019, for the different model regions and jurisdictions as indicated. See Table 4 for a description of the model versions.

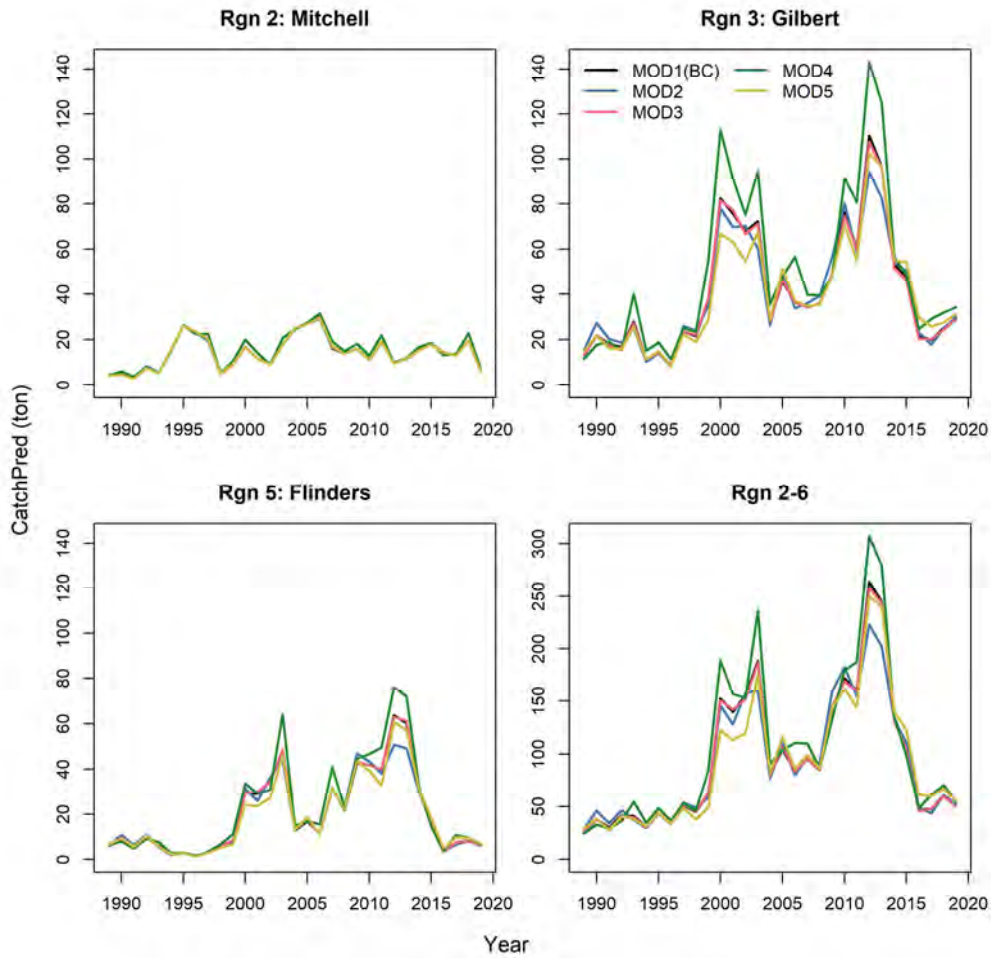


Figure 50. MICE ensemble outputs shown for the five alternative configurations used to represent mud crabs. Trajectories in these plots are the model-estimated catch (tonnes) plotted since start of the fishery (1989) for the three key model regions as indicated, together with the combined catch for Regions 2-6. See Table 4 for a description of the model versions.

Model ensemble: Sawfish

There are no available relative abundance indices to fit the MICE largemouth sawfish population component and hence this component was modelled based on the best available scientific information, pertaining both to parameter estimates and the influence of river flows. Hereafter, unless otherwise specified, we use “sawfish” to mean largemouth sawfish. The ensemble for sawfish focused on representing uncertainty in these estimates as well as in the current depletion levels.

To successfully complete their life cycle across both the marine and freshwater systems, all sawfish species require access to suitable upstream habitat areas such as is found in the Mitchell, Flinders, Gilbert, Roper and Archer Rivers. Sub-adult and adult largemouth sawfish have been recorded in river systems without extensive freshwater habitat upstream of the estuary, such as the Embley, Hey and Mission Rivers around Weipa (Pillans, pers obs). The lack of suitable upstream habitat in systems such as these makes them less important as nursery areas that given sawfish require extensive and accessible upstream habitat (Peeverell 2005, 2010) (Pillans, pers obs). Once they leave their upstream freshwater nurseries, some animals may spend time in other river systems, but as they are not thought to reproduce there, and there is no information on immigration rates, our model does not explicitly represent sawfish population dynamics in Model regions 1, 6 and 8 due to the lack of suitable rivers with extensive and accessible freshwater habitats known to or assumed to be important nursery areas. We note though that even though there may not be suitable breeding habitat in these other systems, this does not imply that water extraction won't have an impact on these systems through habitat reduction and modification.

The model integrates the two chief sources of variability influencing changes in sawfish population numbers, namely river flows and interactions (involving incidental capture by fisheries). The natural variability in rainfall and hence flows in each river system is known to result in boom-bust dynamics of sawfish (Lear et al. 2019) with alternative parameter settings to represent boom-bust dynamics incorporated as part of the sawfish ensemble. Fishing effort in the region has mostly declined since around the turn of the century (Appendix Figure A77), but there is still considerable fishing effort in both inshore and offshore sawfish habitat (Pillans et al. in press). As confirmed by the model, recovery is slow for a slow-growing species and because the populations are predicted to have already declined substantially by the turn of the century, it's harder for the population to bounce back to past levels.

The model is a useful tool as a start for constraining the range of plausible parametrisations, as some parameter combinations were found to not be viable as they either resulted in unrealistic population trends or resulted in unrealistically low (or extinct) final population levels. The interaction rate (similar to fishing mortality) was also used as a model diagnostic to scale the size of populations in each region because a slow-growing species such as sawfish would only be able to sustain relatively low interaction mortality rates over an extended time period (e.g. $< 0.1/\text{yr}$).

We acknowledge that the number of interactions (i.e. animals caught in fishing gear) doesn't always mean that all these animals die and hence that this straightforwardly translates into a mortality rate. However, there are currently no data on post-release survival of largemouth sawfish despite this being a key parameter that is needed to accurately model the population dynamics of a threatened species. Before they were afforded protection, it is likely that few captured sawfish were released or survived entanglement in fisheries gear. Additional threats included illegal shark finning operations (Putt and Anderson 2007). Protocols for correct handling of entangled animals to enhance post-release survival are also only more recently available (Kyne and Pillans 2014). Australia's *Environment Protection and Biodiversity Conservation Act* (the EPBC Act) came into effect in 1999, such that sawfish became better protected from 2000 and hence it is likely that post-release survival rates have increased over the past two decades. As there are no estimates to inform on recent post-release survival rates, we assume in the MICE ensemble that post-release survival is zero. This is also because there is considerable uncertainty regarding the catch rates we apply in the model, and hence adding an additional unknown parameter would only confound this uncertainty further. However, we

ran sensitivity analyses to evaluate how sensitive the model results are to assuming that post-release survival has improved since 2000. Given no reliable estimates of post-release survival, we used four alternative values to span a range of post-release survival examples : 0, 0.25, 0.75 & 0.9. The details and results of these sensitivity analyses are provided in Appendix 13. Model results suggest that improvements in post-release survival can reverse population declines or substantially improve population status even though it may take some time before their impact is seen in terms of total population size. However, accounting for post-release survival did not substantially change model predictions in terms of the impact and risks to sawfish as assessed under different WRD scenarios.

There are no suitable time series data to validate model predictions – indeed, we will never have accurate time series data as it was simply not recorded - therefore current modelling is bounded only by plausibility and current life history understanding, being based on the best available estimates of life history parameters and results are explored across a wide range of alternative parametrisations. The model uses actual fishing effort data from the NPF trawl and Qld and NT barramundi gillnet fisheries, disaggregated by model region and per month of each year (available from 1970 for the trawl fishery and 1989 for the gillnet fishery, with the gillnet fishing effort thus assumed constant over 1970-1989). The model applies the catchability estimates from Pillans et al. (in press) noting that these are based on a subset of regions (all Qld Gulf of Carpentaria and not NT) and short time period hence may not accurately represent past catchabilities, but are the only available estimates. We note that largemouth sawfish susceptibility to being caught during fishing operations is very high due to their toothed rostrum, but that catchability is presumably tightly linked with abundance and hence actual catch rates are likely to vary spatially and temporally based on abundance. The actual interaction estimates should ideally also be a function of sawfish density in each region and time step, however data on sawfish abundance/density are currently lacking in all regions. As a result of lack of data on abundance, this has not been implemented here to avoid introducing further uncertainties, and future work will further explore this aspect. The reconstructed interaction time-series per model region is shown in Appendix 13 and as it is highly uncertain, should not be considered reliable but rather a preliminary means of adding realism to the model by accounting for past variations in interactions with trawl and gillnet fisheries in each of the GoC model regions.

As female largemouth sawfish are at least likely to be maternally philopatric (Phillips et al. 2011, Feutry et al. 2015), no connectivity between regions is represented in the model, although some movement of older animals (i.e. once they leave their river nursery home) likely occurs and adults may move freely throughout the Gulf region. Hence the modelled sawfish numbers represent the recruitment that is specific to that river system (due to philopatry) as well as the resultant adults, but noting that some of these adults may move to neighbouring regions and interact with fishing gear there for example, which is beyond the scope of this model to capture. Given the lack of data on movement and connectivity and the fact that each river experiences unique flows, we therefore considered that the most appropriate method was to treat each river as a separate unit. We note also that modelling populations as separate units is generally the most precautionary setting

There are also few data to quantify the influence of river flow on sawfish population dynamics. Representation of this aspect in our model is based on the best available scientific knowledge and literature, in particular Lear et al. (2019) and we test sensitivity of results to a range of alternative plausible parametrisations. The model runs over a relatively long time period, since 1970, during which there have been massive natural variability in flow. As there are still extant sawfish in many of the Gulf rivers, model parametrisations have had to ensure that the impacts of changes in flow on sawfish recruitment variability is not such that past variability has wiped out the population, hence this is a useful diagnostic to check that parametrisation of modelled responses to changes in flow are not overly extreme. This provides some confidence in model predictions of the plausible impact of altered flows on sawfish populations. The modelled response is seen to vary depending on the unique characteristics of each river system in combination with different past levels of interactions. In all cases the same natural mortality and reproduction rates are assumed for sawfish in the different model regions, but the response to changes in flow varies across systems.

Model trajectories show a slow build-up of numbers into the mature age class category, noting that this includes animals older than 9 and the 20-year age class is a ‘plus’ group that includes any survivors older than 20. Figure 51 below illustrates how large fluctuations in sawfish recruitment – the numbers of 1-year old juvenile sawfish – becoming progressively dampened as they are added to the total population numbers and surviving individuals eventually move into the “mature” category.

There is also a lag before the effect of changes in river flows translate into changes in recruitment and ultimately into the number of mature animals (Figure 51). But recruitment depends also on the number of mature animals and hence when population numbers are low, recruitment will be relatively low regardless of flows (Figure 51). The MICE integrates the effect of both flows and fishery interactions on largemouth sawfish population dynamics with these two external drivers having cumulative or mitigating impacts at different points in time (Figure 52). This is illustrated further in Figure 52. There are also regional differences in the relative influence of these external drivers. The downward trend from 2004 in barramundi (which also serves here as a proxy for other gillnet fisheries) fishing effort in the Mitchell and Gilbert River regions is predicted to have facilitated a noticeable increase in largemouth sawfish in these regions, which were subsequently boosted further by good rainfall (and hence flows) over 2008 to 2011 (Figures 53-54). On the other hand, a decreasing trend in gillnet fishing effort in the Flinders River region only occurred after 2011 and hence although recruitment is predicted to have increased slightly over 2008-2011 (Figures 52-53), there is no noticeable increase in the number of mature animals until towards the end of the time series (Figure 54).

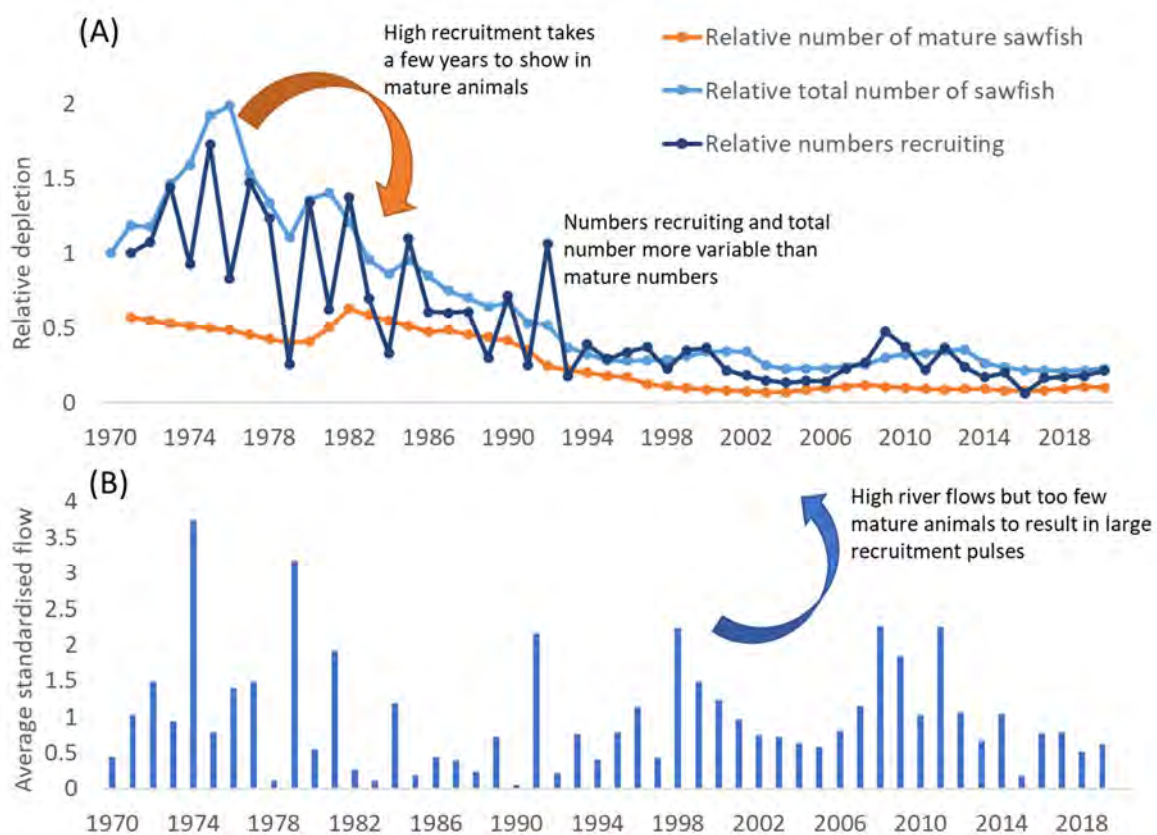


Figure 51. Two illustrative alternative MICE representations of largemouth sawfish in the vicinity of the Mitchell River region, comparing depletion trajectories (relative to 1970 level) based on mature numbers with total numbers and new recruits, where the relative recruitment is plotted using the second vertical axis. The bars are the average of the standardised flow in the first 12 weeks of each year (note though that sawfish recruitment and juvenile survival uses the standardised flows as an input to an additional flow-recruitment relationship). Total recruitment depends on population size, as well as flows, with variable boom-bust dynamics also simulated.

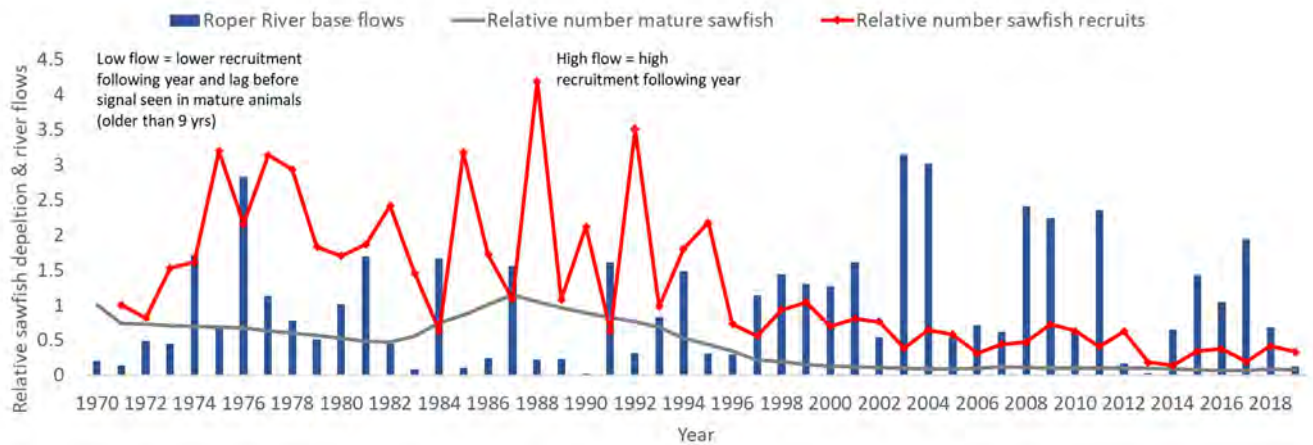


Figure 52. Example of model simulation for largemouth sawfish in the Roper River region to highlight the influence of flow (blue bars) on recruitment (red line, plotted against the second vertical axis,) and lag time before signal is seen in mature numbers (grey line).

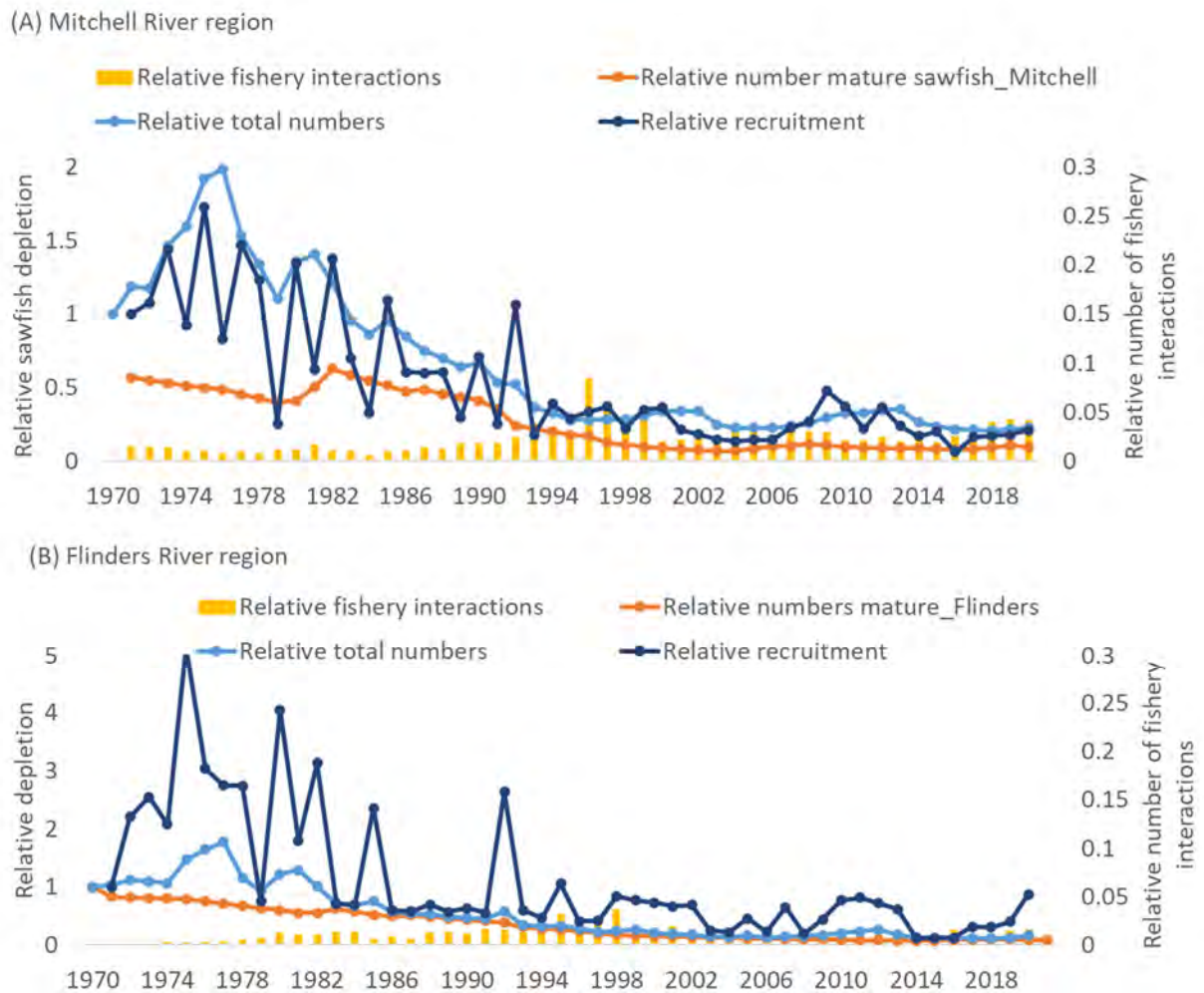


Figure 53. Two illustrative MICE representations of largemouth sawfish in the vicinity of the Mitchell River (top) and Flinders River (bottom) region similar to previous figure, but showing changes in fishery interactions (based on effort in prawn trawl and inshore gillnet fisheries) over time as these influence population trajectories in addition to influences from changes in flow.

As expected, the model is particularly sensitive to the natural mortality M estimate (adult and juvenile) hence improved estimates of M will considerably reduce the uncertainty with respect to modelling plausible rates of decline and recovery of sawfish. For species for which M is unknown, estimates as usually based on methods developed for teleosts there is a large degree of variability between the methods traditionally used regardless of the life history parameters (Zhou et al. 2021). The alternative model versions incorporate a range of alternative parametrisations related to mortality rates, boom and bust dynamics, productivity (reproduction parameters), depletion estimates and flow-recruitment relationship.

The total numbers of sawfish should not be considered accurate as there are no data to validate these estimates and they depend largely also on the reconstructed historical interaction series we used (i.e. effort in offshore prawn trawl and inshore gill net fisheries), which as noted above represents the best available information but is not considered accurate. The relative depletion trajectories are considered more reliable than simulated total numbers, as a range of alternative depletion estimates is represented (capturing some of the uncertainty) which is intended as a plausible representation of the changes over time in sawfish populations (Table 14-Table 15, Figure 54-Figure 55).

More confidence should be attached to the scenarios representing greater current depletion levels as these are considered most likely (by expert opinion and based on IUCN and EPBC Act status and e.g. (Dulvy et al. 2016, Simpfendorfer et al. 2016)), but given uncertainty, more optimistic scenarios are also included to provide a balanced representation (e.g. Model 2, Figure 54-Figure 55). This plausible set of sawfish population trajectories is therefore used as a basis for comparing the extent to which population may become further depleted or not under a range of alternative WRDs. Model results should therefore be interpreted as a plausible set of sawfish past population relative depletion trajectories and the extent to which these are altered under alternative WRDs. We note that there is a fair amount of consistency across these runs, which suggest that even given the massive uncertainty, the take home results are more or less similar.

Table 14. Summary of MICE ensemble current depletion levels (No. mature(2020)/No. mature (1970)) simulated for largemouth sawfish in each of the river catchment systems as shown, and intended to capture a broad range of plausible depletion scenarios for sawfish, as a basis for testing the influence of WRDs on this species.

	Mitchell	Gilbert	Norman	Flinders	Roper
Model 1	0.09	0.12	0.04	0.06	0.05
Model 2	0.10	0.08	0.12	0.09	0.07
Model 3	0.45	0.19	0.04	0.06	0.05
Model 4	0.45	0.19	0.04	0.06	0.05
Model 5	0.17	0.04	0.04	0.18	0.05

The model alternative representations of largemouth sawfish are not considered overly conservative as in some cases the model does not currently account for potential additional negative effects such as barrages or dams that obstruct the free movement of sawfish up and down river systems – as this is an essential component of their life history. The base-case model scenarios also doesn't include the additional predation mortality that sawfish pups may be subject to in response to recent increases in crocodile numbers in these same systems with increased impediments to upstream movement thought to exacerbate predation.

Our modelling results highlight the paucity of data and information available for the largemouth sawfish *P. pristis*. The largest uncertainty that has the most impact on modelling results is the natural mortality estimates. Close kin mark recapture (CKMR) studies have previously provided information

on M and abundance for other species (Bravington et al. 2016, Hillary et al. 2018). If sufficient tissue samples of sawfish are collected from bycaught animals, this method has the potential to provide updated estimates of M as well as population abundance, that will considerably improve the ability to model depletion and recovery scenarios for this threatened species under a range of alternative future scenarios. Additional information on population connectivity (a by-product of CKMR methods) will also improve the ability to more accurately model the connectivity between regions in future models.

There is also scope to considerably refine estimates of fishery interactions and post release survival, and to link these estimates to sawfish density per region. This will be considerably facilitated if reliable estimates of adult M and population abundance become available in future. Ongoing collection of observer data will also allow improved estimation of interaction rates as well as the extent to which bycatch reduction devices are successful in reducing mortality due to interactions in future.

There is an urgent need for ongoing information and data pertaining to the sensitivity of sawfish to changes in river flows, mortality rates due to dying in dried out waterholes as well as recruitment variability in response to flows (e.g. Lear et al. 2019) will all help to further reduce the uncertainty around modelling of the impact of changes in flow on population dynamics, and hence the use of this as a basis for modelling plausible impacts due to alternative WRDs. Research surveys and long term studies that incorporate acoustic telemetry are the most appropriate methods for demonstrating the influence of flows on movement, recruitment and mortality (Lear et al. 2019, Patterson and Pillans 2019, Pillans et al. 2020).

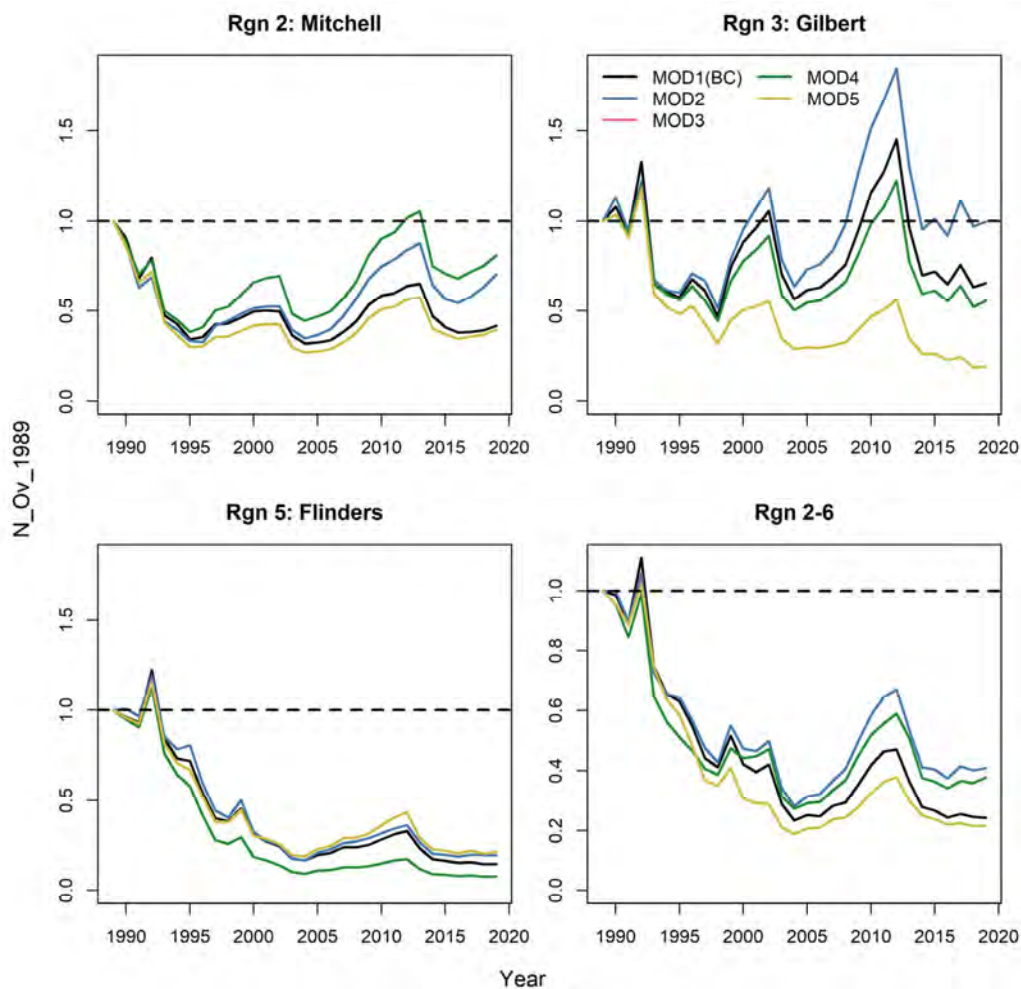


Figure 54. MICE ensemble outputs shown for the five alternative configurations used to represent largemouth sawfish. The model trajectories represent the relative total number of animals plotted in each instance as relative to the 1989 reference numbers to highlight relative differences over time.

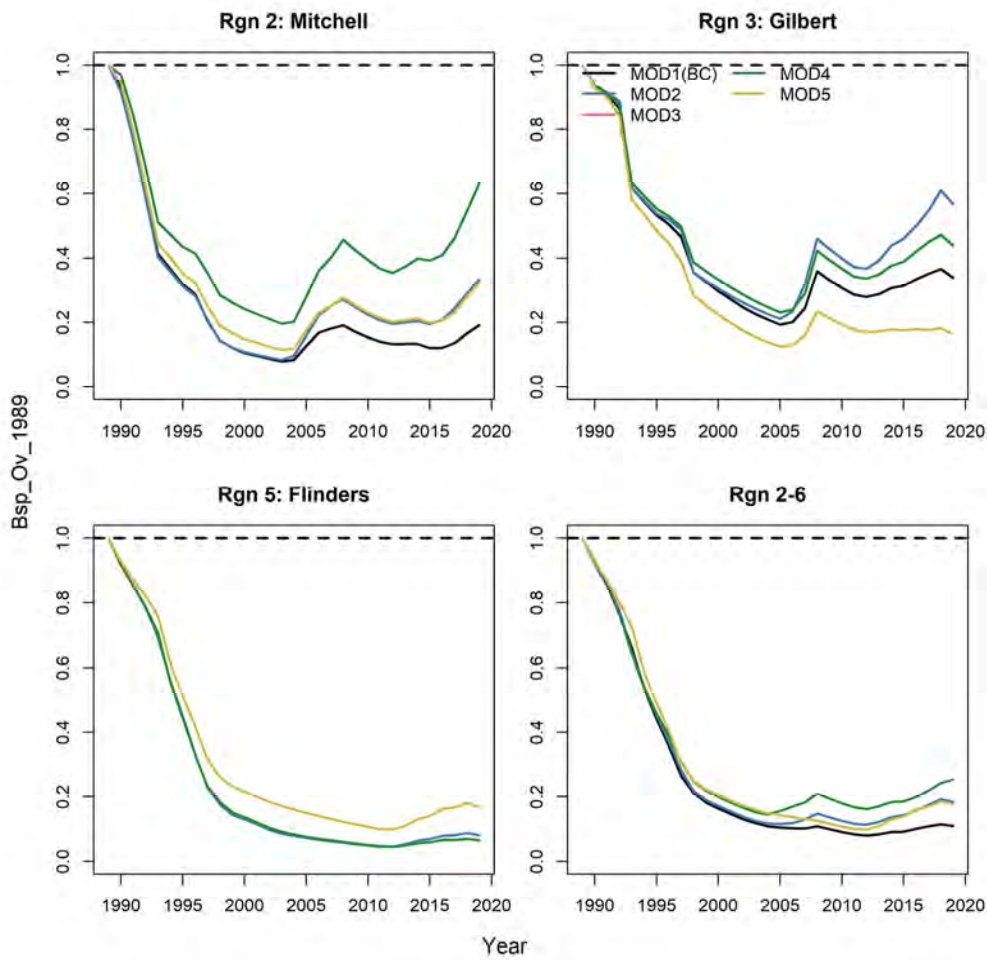


Figure 55. MICE ensemble outputs shown for the five alternative configurations used to represent largemouth sawfish. The model trajectories represent the total number of mature animals relative to the 1989 abundance level. Note that the label of “Bsp” is the closest comparable to the spawning biomass index used for the prawns, barramundi and mud crabs, but for sawfish (which do not spawn but produce a relatively small number of pups) the reproductively mature population is represented in units of numbers rather than biomass. Numbers are more meaningful for sawfish than biomass because the relationship between fecundity and length is not likely to be teleost like.

Table 15. Summary of the MICE sawfish ensemble variation in relative abundance in each region (Rgn) under the four alternative model representations Models 2-5) when compared with the Model version 1 abundance levels. The values in the Table are thus always 1 for Model version 1, and show differences in other model versions relative (Rel) to Model 1, for statistics as follows: the minimum (Min) and average (Mean) total number of animals (N), the minimum and average number of mature (breeding age) animals (Bsp) and the minimum and average number of new recruits i.e. 1-year old animals joining the population for the first time (Rec).

Model ver.	Rgn.N	Rgn.Name	Rel.N.Min	Rel.N.Mean	Rel.Bsp.Min	Rel.Bsp.Mean	Rel.Rec.Min	Rel.Rec.Mean
Model1	2	Mitchell	1.00	1.00	1.00	1.00	1.00	1.00
Model2	2	Mitchell	1.13	1.53	1.05	1.36	1.17	1.80
Model3	2	Mitchell	5.24	7.51	3.28	7.32	2.83	6.15
Model4	2	Mitchell	5.24	7.51	3.28	7.32	2.83	6.15
Model5	2	Mitchell	1.80	1.94	1.31	1.80	1.26	1.71
Model1	3	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
Model2	3	Gilbert	1.79	2.33	1.73	2.07	1.76	2.69
Model3	3	Gilbert	2.86	3.17	1.89	2.17	1.80	2.07
Model4	3	Gilbert	2.86	3.17	1.89	2.17	1.80	2.07
Model5	3	Gilbert	0.26	0.52	0.23	0.34	0.25	0.37
Model1	4	Norman	1.00	1.00	1.00	1.00	1.00	1.00
Model2	4	Norman	0.95	1.01	0.98	0.99	0.91	1.10
Model3	4	Norman	1.30	1.60	1.00	1.00	1.00	1.00
Model4	4	Norman	1.30	1.60	1.00	1.00	1.00	1.00
Model5	4	Norman	1.30	1.60	1.00	1.00	1.00	1.00
Model1	5	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
Model2	5	Flinders	1.03	1.17	0.93	1.02	1.05	1.40
Model3	5	Flinders	0.90	1.15	1.00	1.00	1.00	1.00
Model4	5	Flinders	0.90	1.15	1.00	1.00	1.00	1.00
Model5	5	Flinders	3.65	4.49	2.23	4.01	2.05	3.61
Model1	7	Roper	1.00	1.00	1.00	1.00	1.00	1.00
Model2	7	Roper	1.02	1.06	1.01	1.05	1.01	1.08
Model3	7	Roper	0.87	1.07	1.00	1.00	1.00	1.00
Model4	7	Roper	0.87	1.07	1.00	1.00	1.00	1.00
Model5	7	Roper	0.87	1.07	1.00	1.00	1.00	1.00
Model1	12	Rgn 2-6	1.00	1.00	1.00	1.00	1.00	1.00
Model2	12	Rgn 2-6	1.13	1.49	1.06	1.34	1.16	1.70
Model3	12	Rgn 2-6	2.02	2.84	1.26	1.99	1.28	1.97
Model4	12	Rgn 2-6	2.02	2.84	1.26	1.99	1.28	1.97
Model5	12	Rgn 2-6	2.12	2.65	1.80	2.28	1.31	1.75

Model ensemble: Mangroves and Seagrass

There were no suitable data available at the spatial and temporal scales needed to formally validate the mangrove and seagrass model trajectories. However, there were some limited observations that were useful in informing on the likely nature and magnitude of changes in these model groups, noting that in both cases the modelled relative biomass represents the average response of these communities to physical drivers. The seagrass and mangrove components were based on available scientific information and expert knowledge (Rob Kenyon in particular) and hence are considered plausible representations of these key habitats as a first step towards trying to quantify large-scale changes in supporting habitats in response to changing physical drivers such as flow.

The ensemble for both groups included alternative parameter settings for growth rates (Model 2), assuming cyclone-induced mortality was greater (Model 3) as well as using a Pella-Tomlinson (Model

4) as opposed to Schaefer model representation of the production function (Hilborn and Walters 2013). Alternative settings were also used across the ensemble to present the sensitivity of seagrass growth to light attenuation, as well as the amplitude of the changes in abundance due to flow.

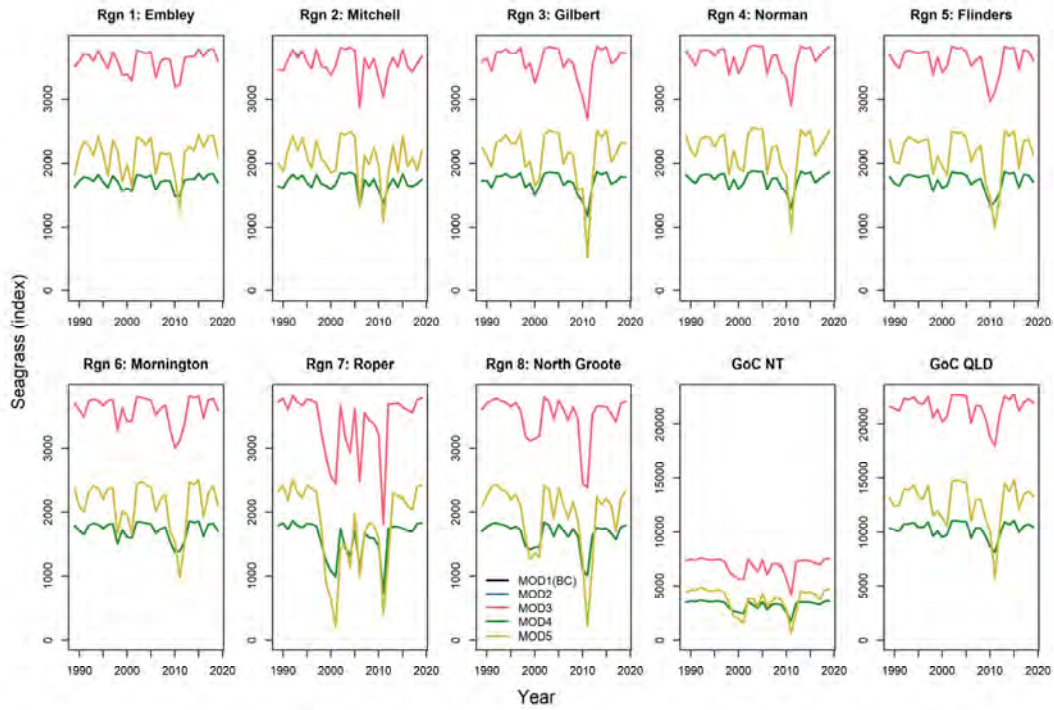


Figure 56. MICE ensemble outputs shown for the five alternative configurations used to represent relative changes in seagrass abundance (baseline flows) in all the model regions as shown.

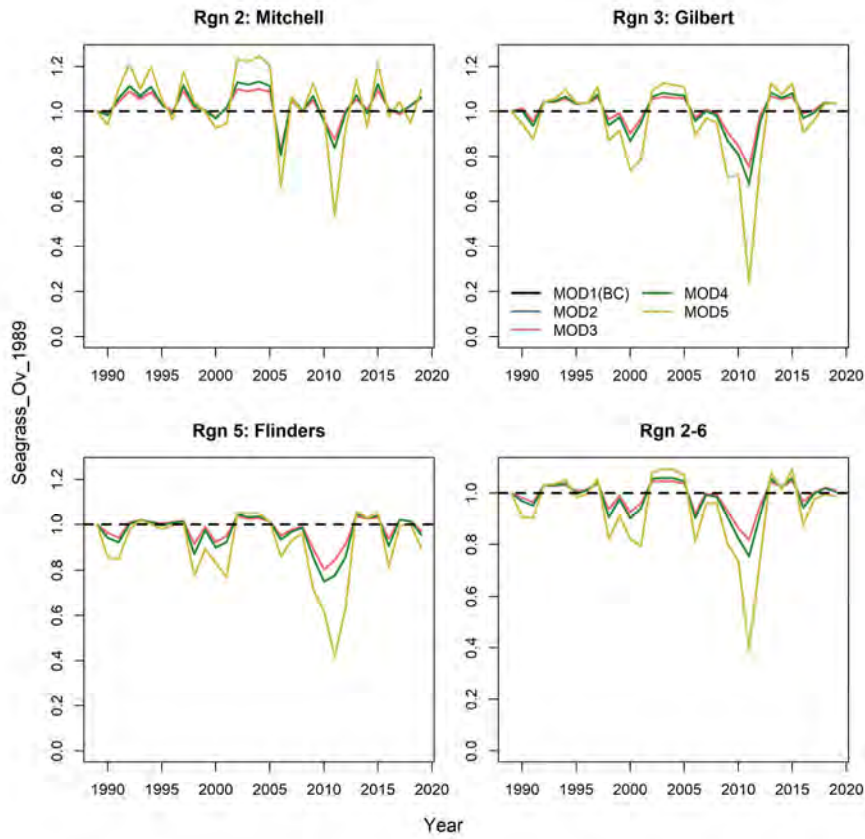


Figure 57. Comparison of the changes in the abundance of seagrass, relative to the 1989 level, for each of the five model versions under baseline flows.

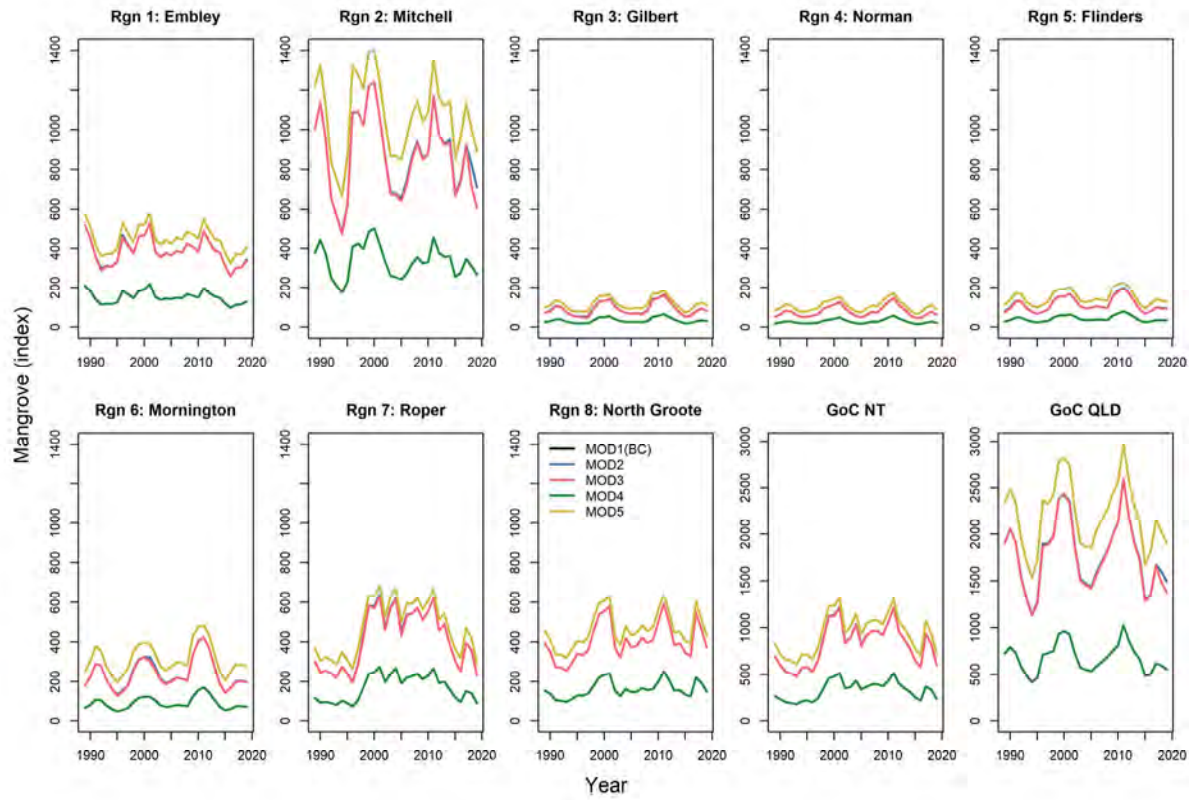


Figure 58. MICE ensemble outputs shown for the five alternative configurations used to represent relative changes in mangrove abundance (baseline flows) in all the model regions as shown.

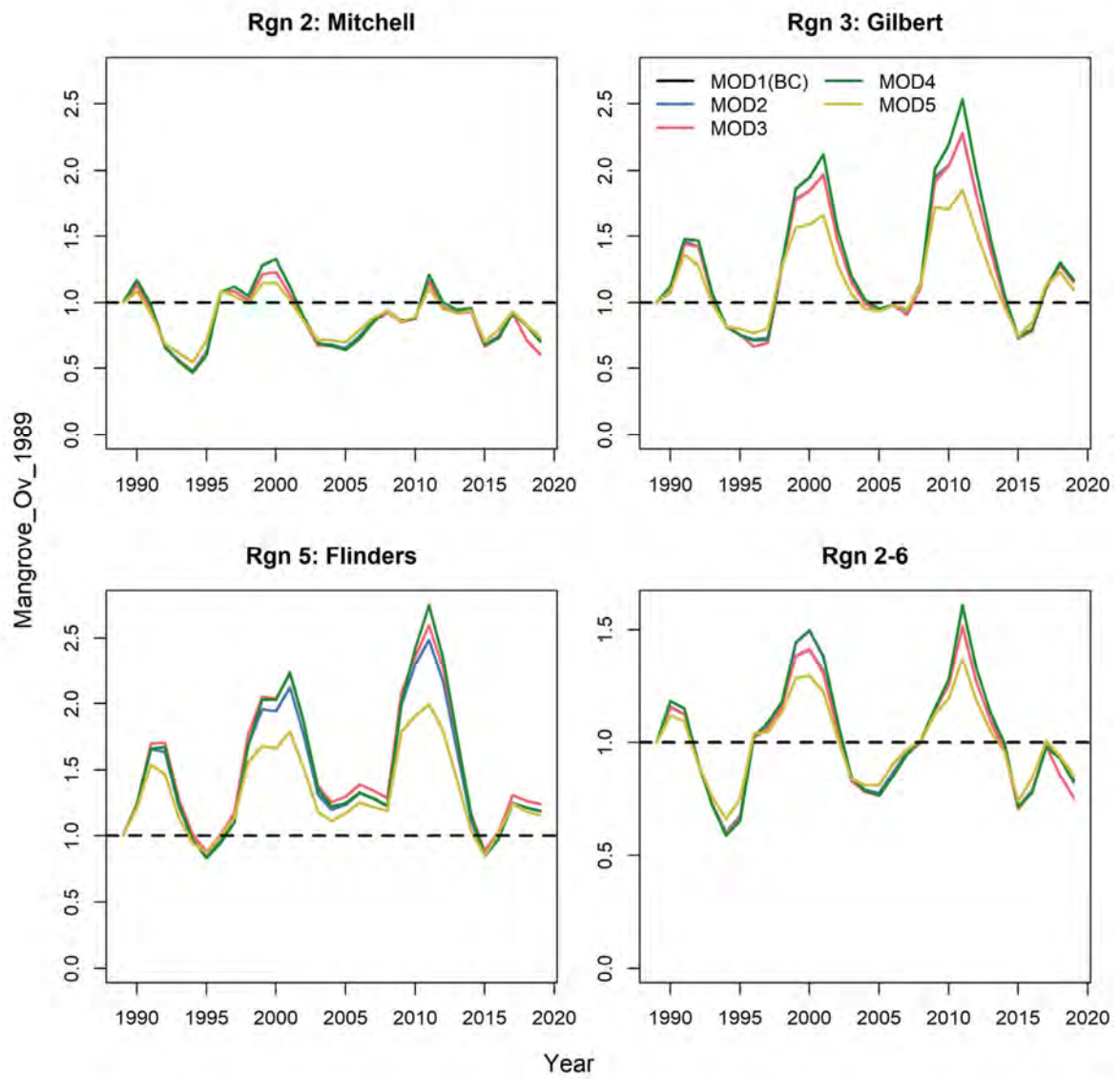


Figure 59. Comparison of the changes in the abundance of mangroves, relative to the 1989 level, for each of the five model versions under baseline flows.

18. Ecosystem overview

We used the MICE to quantify the weekly (or monthly for some species) changes in biomass of each of the model species or groups, whilst tracking changes in each of the underlying physical variables. To summarise the impact of flow (and other physical variables) on key components of the ecosystem, we provide some examples of overall populations trends relative to flow using Model version 1.

As shown in Figure 60, the overall population trends are driven principally by a combination of fishing and changes in river flows. There are a number of other drivers and interactions that also modify the model predictions of likely changes in biomass over time and in each spatial region, and some selected examples of these are also shown. The biomass and associated catch of key species that are subject to large inter-annual variation in flows – such as shown for the Norman River MICE region 4 – is seen to similarly be highly variable over time, with biomass (and subsequently catch) of a number of species/groups peaking around periods of good flow (Figure 60). Evidence from Australia and globally suggests that sawfish have declined significantly (Dulvy et al. 2016), which is attributed to overfishing and habitat loss (Yan et al. 2021). Boom-bust recruitment dynamics are therefore dampened in systems where the number of mature animals is low, plus there is a much longer lag time before recruits and juvenile sawfish become old enough to join the mature age class, further explaining the less variable trends in sawfish mature abundance as shown in Figure 59. Another example of the combined MICE-predicted changes in physical drivers and relative biomass of the base model groups, namely mangroves, seagrass, microphytobenthos, together with common banana prawns, barramundi, mud crabs and sawfish, is shown for Region 7 – Roper River (Figure 61). The relative influences of cyclones (black bars) and solar radiation (yellow line) on seagrass is shown by superimposing these physical variables on the seagrass biomass trajectory. Changes in air temperature (red dashed line) which can affect mangroves and mud crabs is also shown, noting that this plot shows annual averages only, whereas the model evaluates impacts on a weekly basis for the base groups and monthly for barramundi, mud crabs and sawfish.

The MICE outputs also highlight some of the system changes during ‘dry’ (low rainfall relative to past averages) versus ‘wet’ (high rainfall relative to past averages) years. For example, Figure 62 compares the intra-annual biomass trends in Region 2 (Mitchell River) during an illustrative wet year (2009) and dry year (2015). The plots highlight also the difference in magnitude of common banana prawn catches, and to a lesser extent mud crabs in this region, in dry compared with wet years. The impact of a cyclone during 2015 on Region 2 (Mitchell River) is also shown and can be compared with a scenario in Figure 61 that includes a more substantial cyclone impact. Figure 63 and Figure 64 show further examples of intra-annual population trends during a wet year (2011) and dry (2015) year in each of regions 5 (Flinders River) and 7 (Roper River) respectively. The low flows during 2015 are seen to result in substantial reductions in both common banana prawn, barramundi and mud crab catches from both the Flinders region as well as the Roper region.

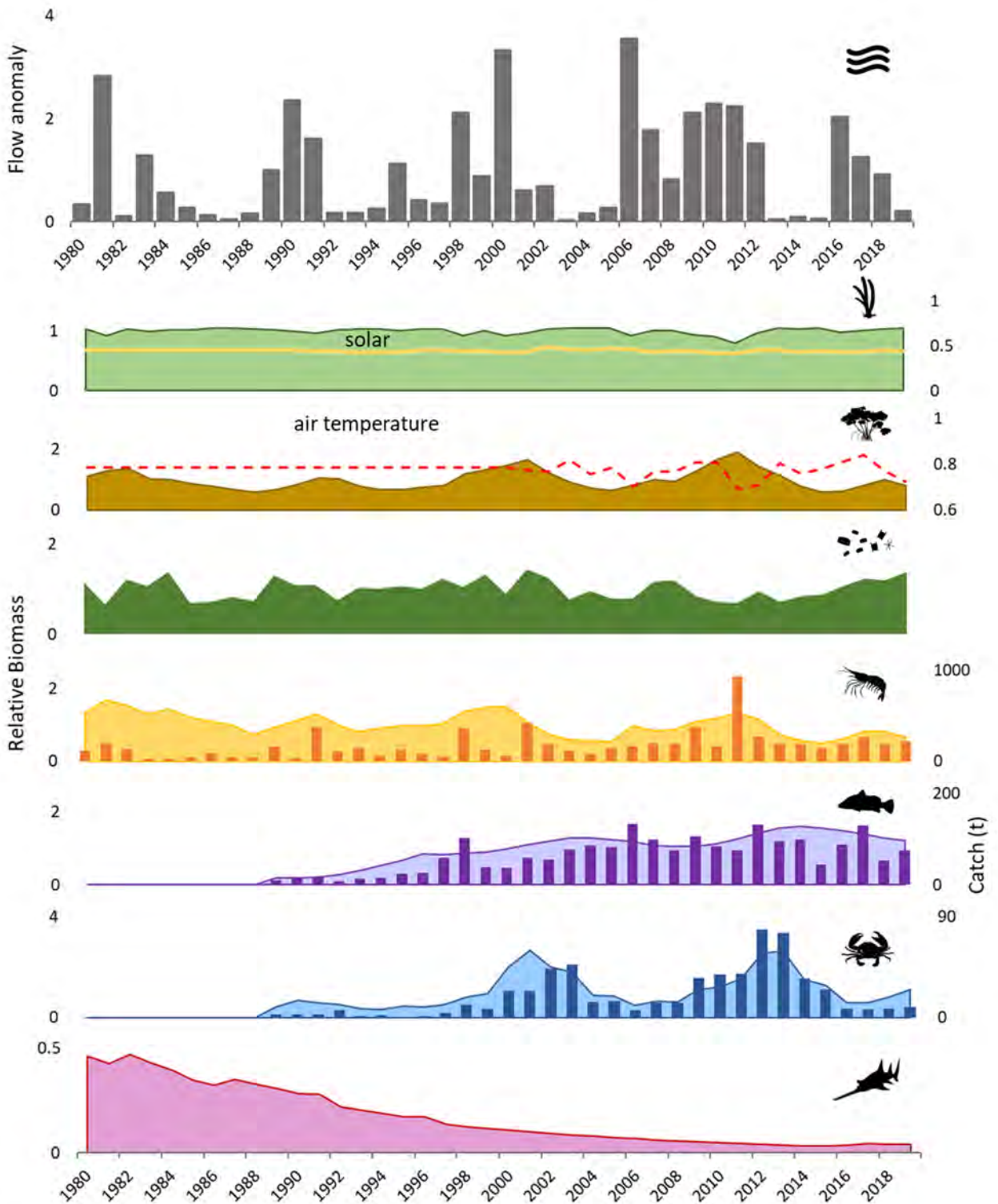


Figure 60. Example of MICE physical drivers and changes in the relative biomass and catch (tonnes) of the base model groups for Region 4 (Norman River) over 1980 to 2019. From top panel to bottom panel: flow, seagrass, mangroves, microphytobenthos, common banana prawn, barramundi, mud crab and sawfish. The relative influences of cyclones (black bars) and solar radiation (yellow line) on seagrass is shown by superimposing these physical variables on the seagrass biomass trajectory. Changes in air temperature (red dashed line) which can affect mangroves is also shown.

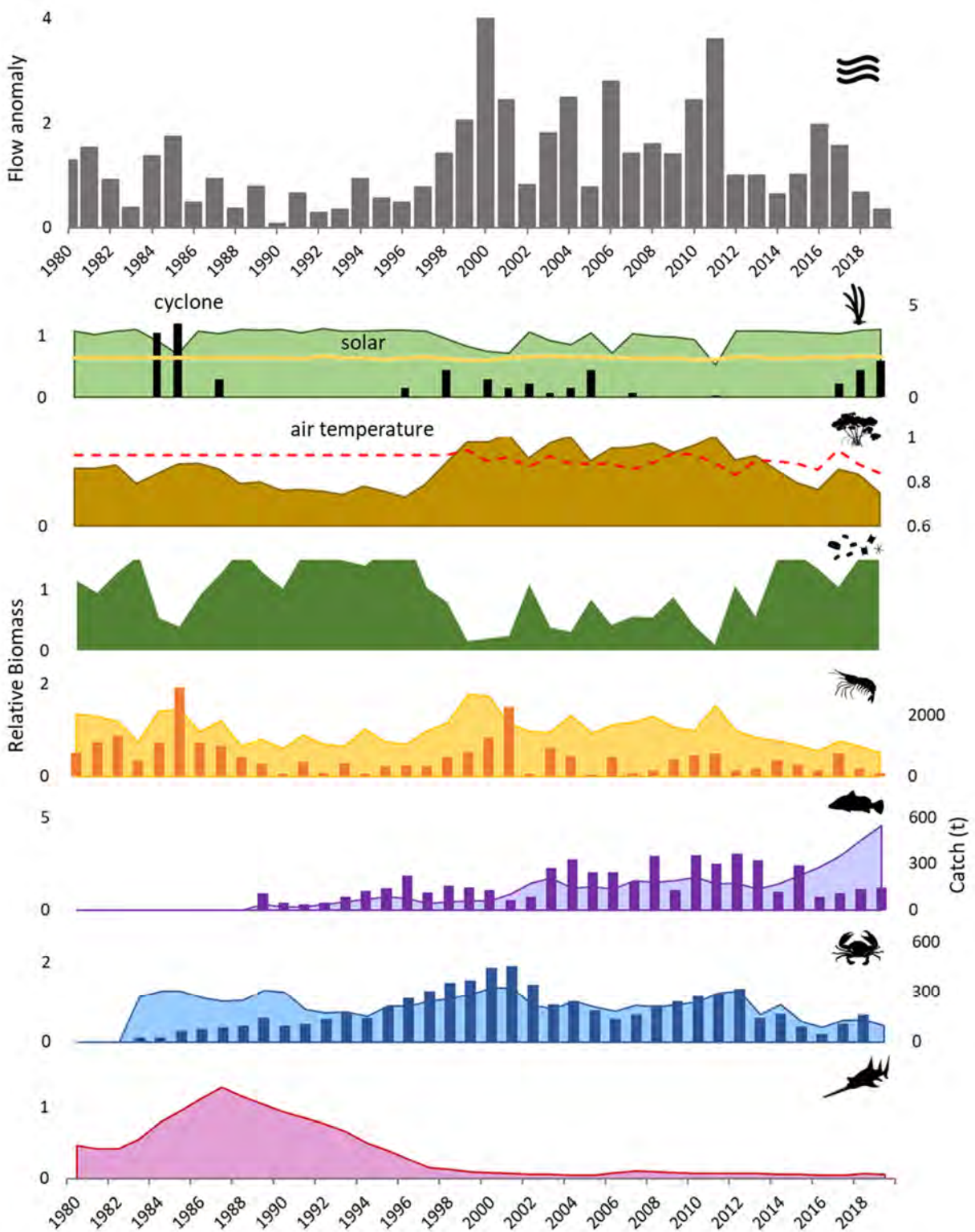


Figure 61. Example of MICE physical drivers and changes in the relative biomass and catch (tonnes) of the base model groups for Region 7 (Roper River) over 1980 to 2019. From top panel to bottom panel: flow, seagrass, mangroves, microphytobenthos, common banana prawn, barramundi, mud crab and sawfish. The relative influences of cyclones (black bars) and solar radiation (yellow line) on seagrass is shown by superimposing these physical variables on the seagrass biomass trajectory. Changes in air temperature (red dashed line) which can affect mangroves and mud crabs is also shown.

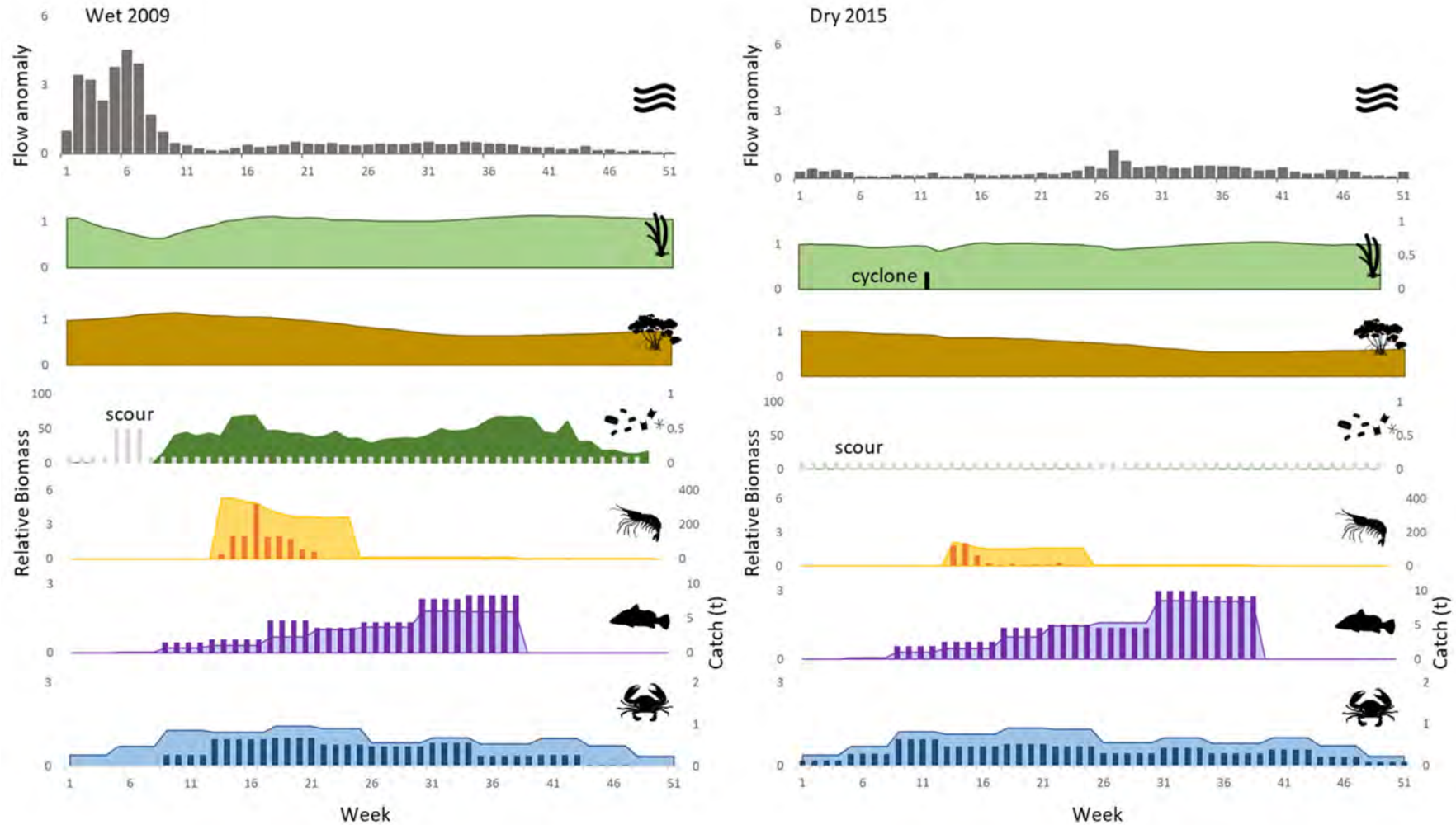


Figure 62. Comparison of GoC river flows using Region 2 (Mitchell River) as an example and changes in the intra-annual biomass during (left) a wet year (2009) and (right) a dry year (2015). From top panel to bottom panel: flow, seagrass, mangroves, microphytobenthos, common banana prawn, barramundi and mud crab. For common banana prawn, barramundi and mud crab, biomass is the commercially available biomass (divided by the long-term average) and catches (tonnes) are shown on the second vertical axis.

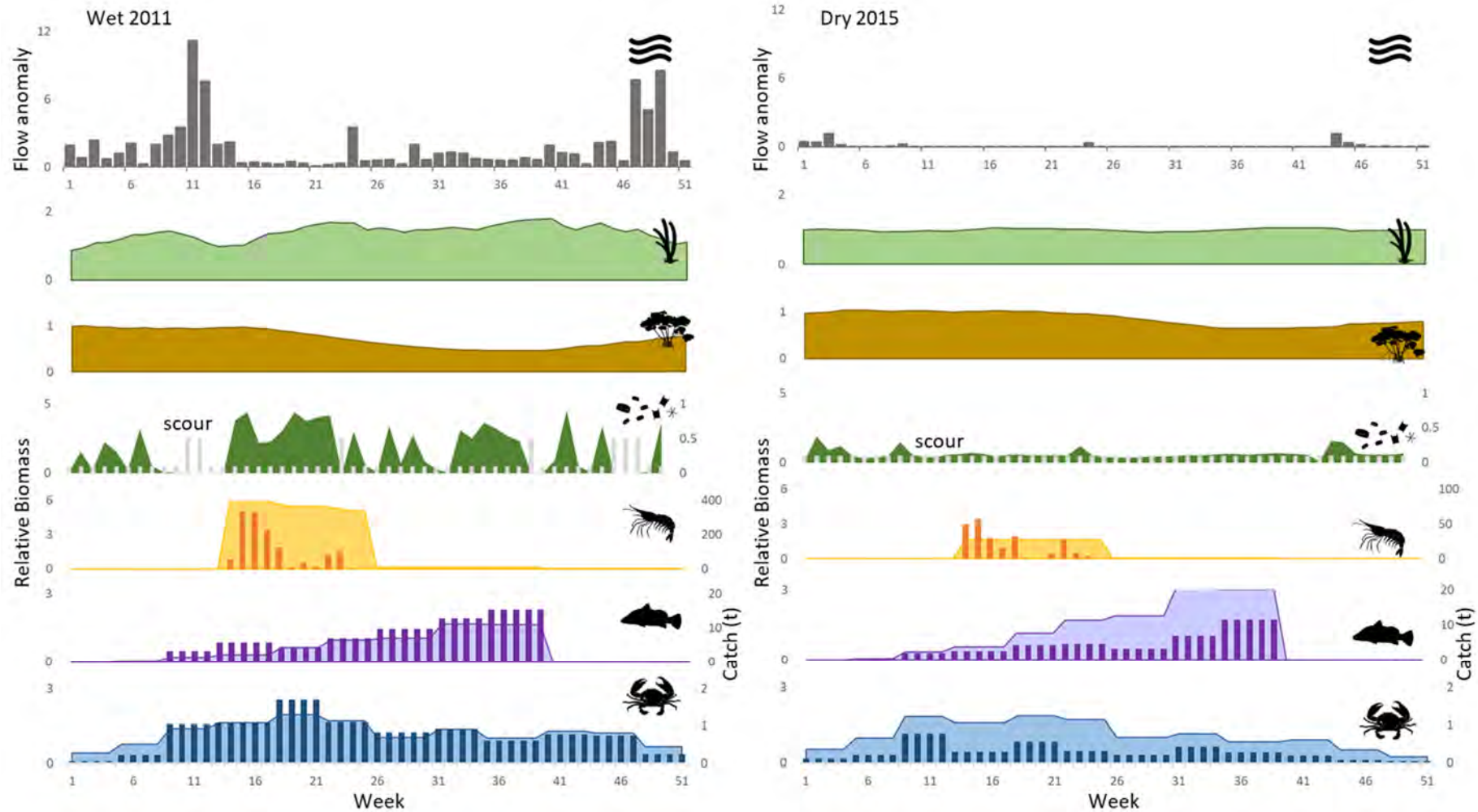


Figure 63. Comparison of GoC river flows using Region 5 (Flinders River) as an example and changes in the intra-annual biomass during (left) a wet year (2011) and (right) a dry year (2015). From top panel to bottom panel: flow, seagrass, mangroves, microphytobenthos, common banana prawn, barramundi and mud crab. For common banana prawn, barramundi and mud crab, biomass is the commercially available biomass (divided by the long-term average) and catches (tonnes) are shown on the second vertical axis.

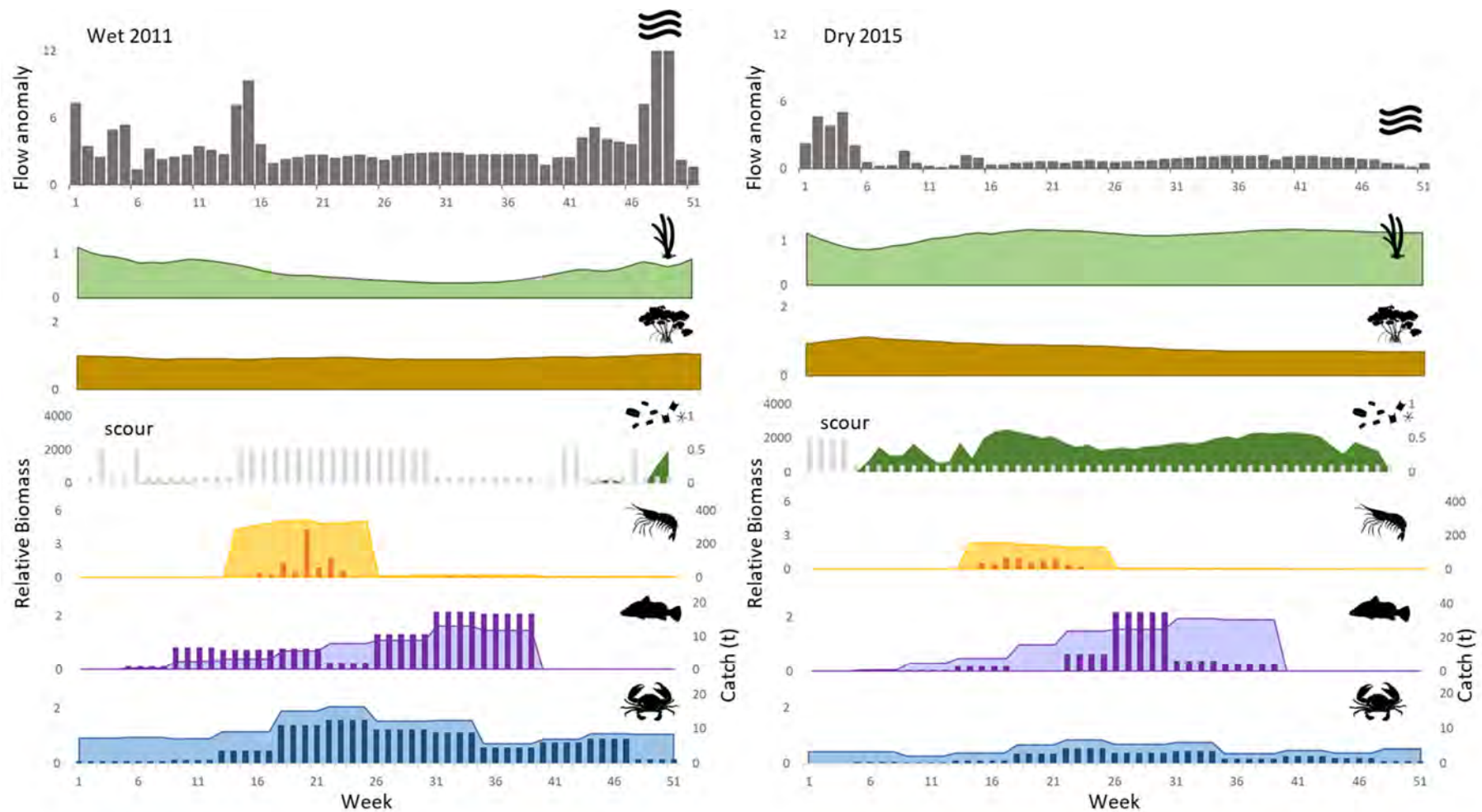


Figure 64. Comparison of GoC river flows using Region 7 (Roper River) as an example and changes in the intra-annual biomass during (left) a wet year (2011) and (right) a dry year (2015). From top panel to bottom panel: flow, seagrass, mangroves, microphytobenthos, common banana prawn, barramundi and mud crab. For common banana prawn, barramundi and mud crab, biomass is the commercially available biomass (divided by the long-term average) and catches (tonnes) are shown on the second vertical axis.

19. Water Resource Development (WRD) scenarios

Streamflow models were used to produce a number of scenarios of the impact of Water Resource Development (WRD) scenarios on historical natural river flows for the Mitchell, Gilbert and Flinders Rivers. We identified water resource development scenarios for the Flinders, Gilbert and Mitchell Rivers that were feasible given the topographical features of the respective catchments relative to proximity to arable soils. The hypothetical but feasible WRD scenarios were defined based on workshopped outcomes and discussions with scientists involved in the FGARA and NAWRA projects. The Flinders River catchment is characterised by low topographical relief over its majority extent and particularly in proximity to arable soils. Consequently, a water extraction and offstream storage scenario was featured. The topography of the Gilbert River catchment supported an instream dam upstream from arable soils in the vicinity of Georgetown, mid-catchment. The Mitchell River catchment was favourable to both instream dams in the upper- and mid-catchment and water extraction.

One of the WRD scenarios in the Flinders River catchment was 175 GL year⁻¹ (75% reliability) pump-extracted and stored offstream (Figure 33). Water harvest was distributed 10 sites across the catchment sited adjacent to arable soils. Water extraction from the Flinders River regularly caused 10 to 40 GL reductions in weekly flow compared to the historical baseline flows. However, the mean annual flow of the Flinders River is 2543 GL, so the percent quantum of flow reduction was low. Despite the low percent reduction, the seasonal timing of reduced flows may be critical and the percent reduction of early season flows may be high. Since 1970, 16 of the 2548 weeks were subject to flow reduction of approximately 40 GL, while in a majority of weeks flow was reduced by 5-10 GL.

One of the WRD scenarios in the Gilbert River (3706) was the Green Hills Dam (172 GL year⁻¹ at 85% reliability) upstream of Georgetown (Figure 34). The scenario diverted water to arable soils downstream. Water impoundment within the Gilbert River both increases and reduces up to 200 GL of water weekly, though 200 GL anomalies were infrequent. Water anomalies in the 50 to 100 GL range were frequent and reductions in weekly flow of that magnitude compared to the historical baseline flows were more common than increases. However, the mean annual flow of the Gilbert River is 3706 GL, so the percent quantum of flow reduction was low. Despite the low percent reduction, the seasonal timing of reduced flows may be critical and the percent reduction of early season flows may be high. Flow reductions of 50 GL in any week were common.

One of the WRD scenarios in the Mitchell River (15579) was two in-stream dams; the Nullinga Dam (65 GL year⁻¹ at 85% reliability) in the upper Mitchell River catchment and the Pinnacles Dam (1248 GL year⁻¹ at 85% reliability) in the mid-catchment area (west of Chillagoe). The two dams were modelled in series to divert water to arable soils downstream. The change in flows in the Mitchell River was predominantly flow reduction, though in a few years increases of ~ 1 GL were predicted. Flow reductions of approximately 10 GL occurred in only 10 weeks of the 2548 weeks since 1970. Weekly flow reductions of 5 GL were common, though not as common as flow reduction in either the Flinders or Gilbert Rivers. However, the mean annual flow of the Mitchell River is 15579 GL, so the percent quantum of flow reduction was very low (Figure 35). Despite the low percent reduction, the seasonal timing of reduced flows may be critical and the percent reduction of early season flows may be high.

The final set of WRD scenarios we used in our analyses is presented in Table 6 in the Methods section.

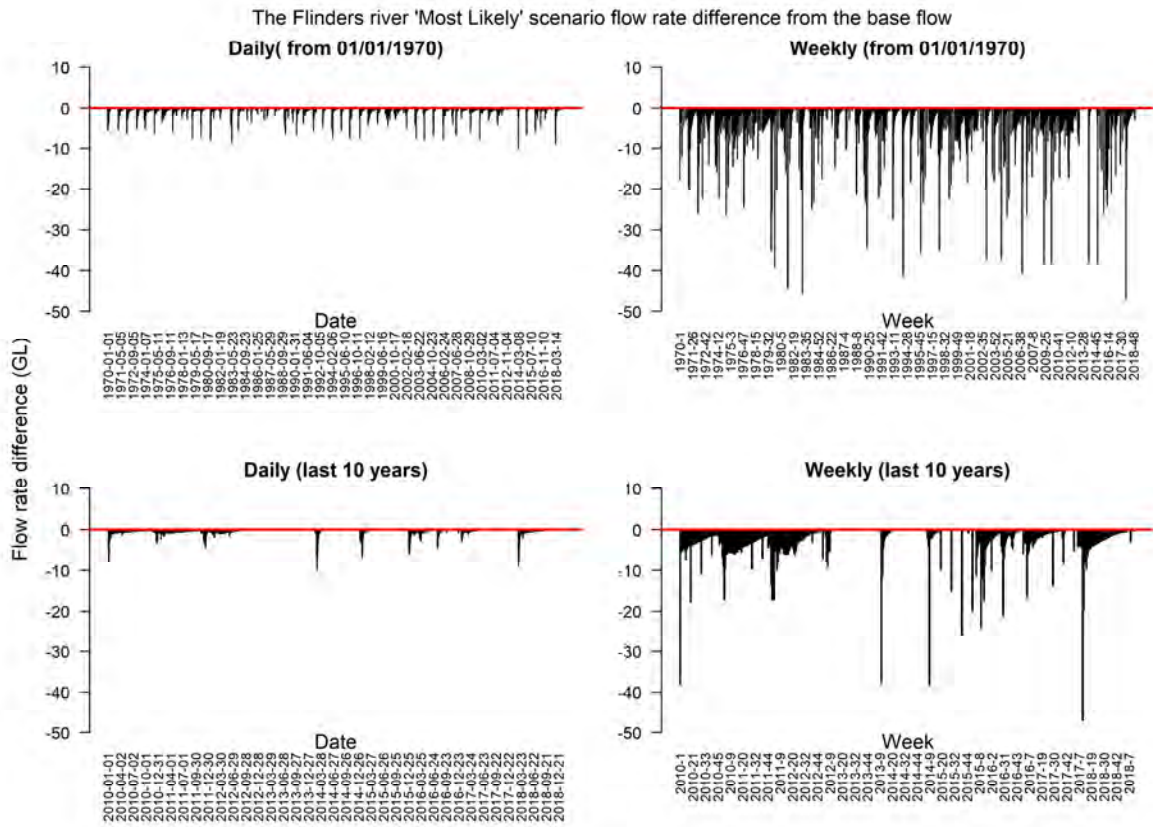


Figure 65. Flow difference plots within the Flinders River between historical baseline unimpeded flows and one of the WRD scenarios to be deployed in the river (175 GL year⁻¹ water harvest and offshore storage).

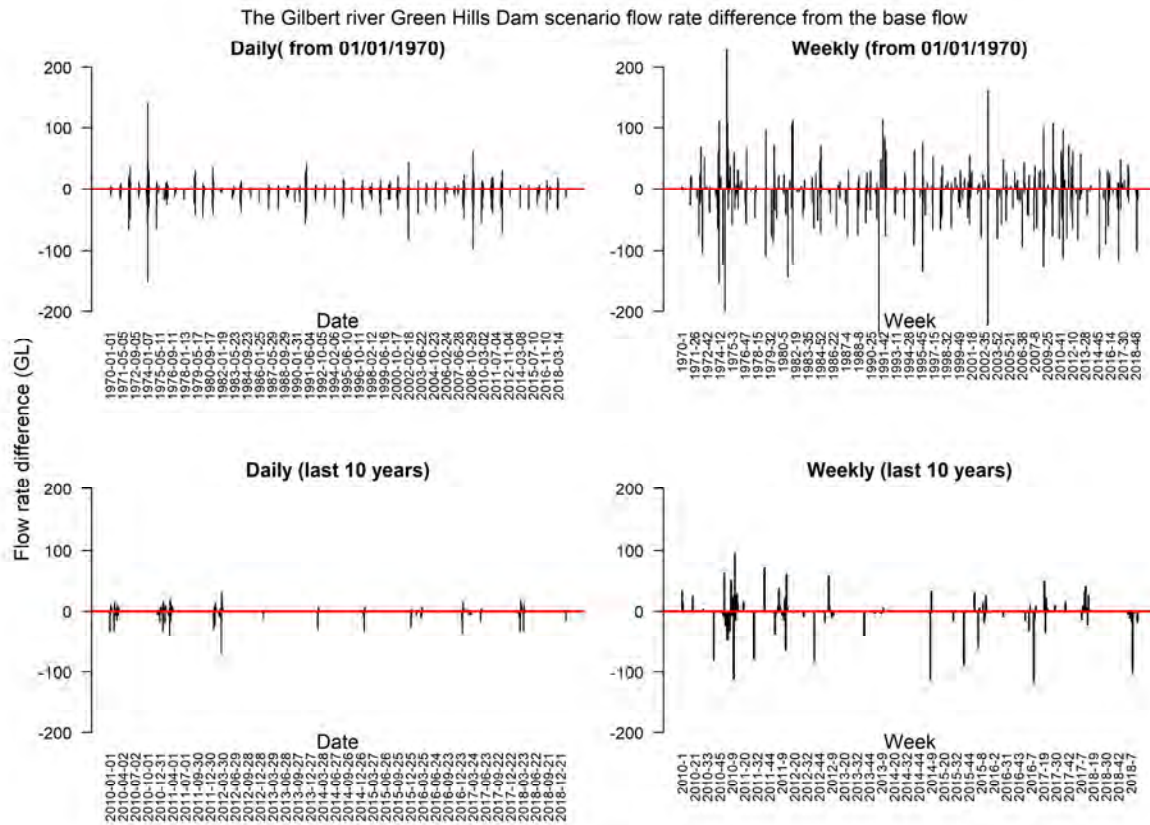


Figure 66. Flow difference plots within the Gilbert River between historical baseline unimpeded flows and one of the WRD scenarios to be deployed in the river (172 GL year⁻¹ water impoundment, Green Hills dam). Note that while dams do have a dampening effect on peak flows, it is possible to get more flows (positive difference) at certain times under WRD involving dams due to a large surface area of water (dam), where there would otherwise be land.

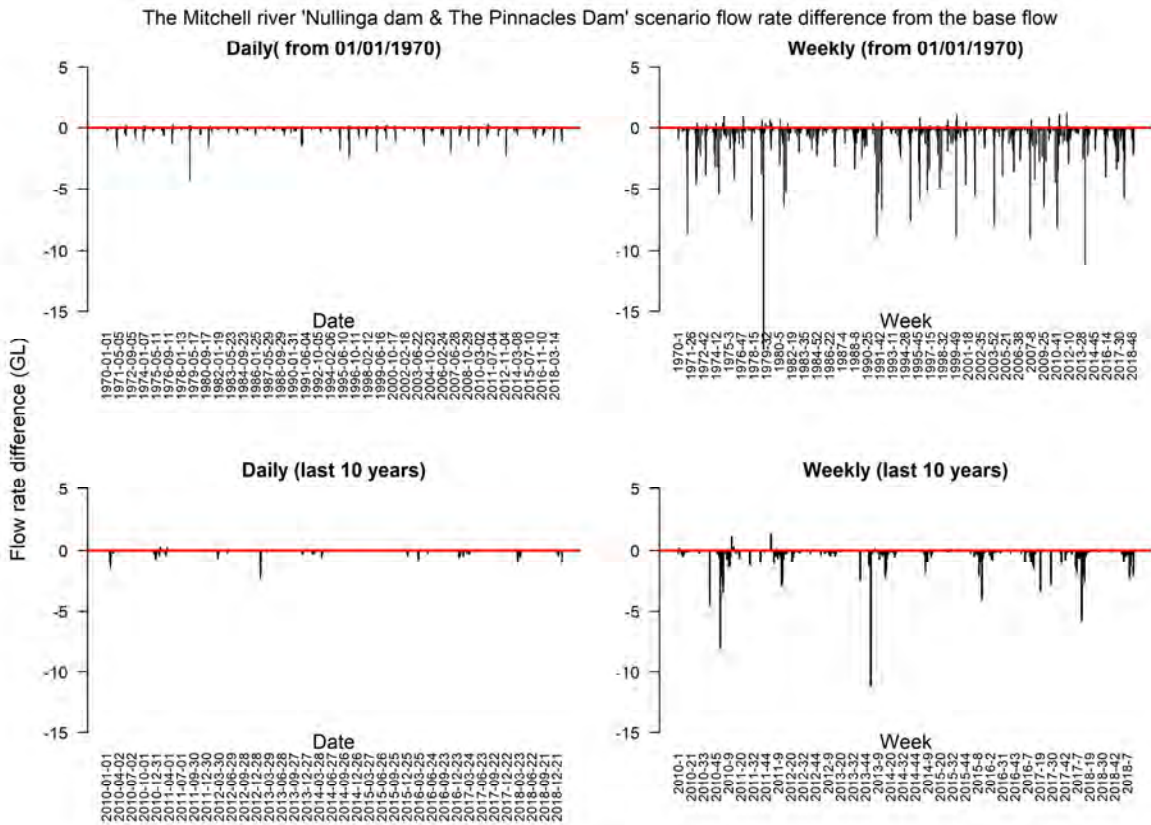


Figure 67. Flow difference plots within the Mitchell River between historical baseline unimpeded flows and one of the WRD scenarios to be deployed in the river (1313 GL year⁻¹ water impoundment, Nillinga and Pinnacles dams). Note that while dams do have a dampening effect on peak flows, it is possible to get more flows (positive difference) at certain times under WRD involving dams due to a large surface area of water (dam), where there would otherwise be land.

20. Quantification of WRD influences on the aquatic ecosystem

Overview

We found impacts of almost all WRDs on all species and catchment regions for which changes from baseline flows were modelled. However different species and regions had different magnitudes and types of response. There were major differences in the predicted impact of alternative WRDs on the indicator species and habitats within the ecosystem, ranging from minor changes relative to baseline flow conditions through to extreme impacts under some scenarios.

We produced full sets of results for all 19 WRDs, with summaries presented in this report, but electronic copies of outputs for each WRD scenario are available on request. To keep presentation of results tractable, we focus in more depth on the four key WRD scenarios (WRD1-WRD4) that we identified based on plausibility as well as sufficient contrast to assist in bounding the likely range of outcomes. In each case, we ran these four key WRDs using each of the five alternative model versions in order to test the robustness of model results under each scenario, as well as to quantify the model process uncertainty. The set of five key model versions comprise a subset of the full set of model sensitivity analyses that we ran (partly to inform choice of the key sensitivities). Selected details from the other sensitivity tests are provided in Appendix 17. Additional results for the four key WRD scenarios (for the five model versions) are shown in Appendix 18 and a summary of results from all 19 WRDs under Model version 1 and for the MICE ensemble are shown in Appendix 19.

Prawns were the only species in the model that included representation of connectivity between different model regions. This was both in terms of the combined spawning biomass influencing recruitment estimates, as well as a portfolio of rivers being estimated as influencing the overall regional recruitment. We did however, include a sensitivity to this assumption of a 'portfolio river effect' as one of our alternative models (Model 2). Results suggested that WRD scenarios that only affect individual rivers could nonetheless influence prawn recruitment and catches in neighbouring regions due to the assumed system connectivity for prawns. For all other species, localised impacts only of altered flows are predicted because we did not assume connectivity between regions in modelling these other species.

When evaluating the impact of an altered flow on the marine system, we quantify both the relative impact as well as the regional impact. The latter accounts for the fact that the different model regions contribute differently, in terms of both relative abundance and catches, to the overall regional or Gulf-wide catch and abundance. As the focus of these analyses is on potential WRDs implemented for the Mitchell, Gilbert and Flinders Rivers, we focus on presentation of results on these three model regions; as well as the combined Queensland eastern and south-eastern region stretching from Model region 2 to Model region 6 (which borders the NT). When reliable WRD end of system river flows become available for NT rivers such as the Roper River (region 7) these analyses can similarly be applied to that region.

In terms of the alternative WRDs we tested, the four key scenarios range from volumes of water allocated above which annual reliability becomes problematic (WRD1) that simultaneously explore impacts on all three key rivers (high extraction rates on Mitchell and Flinders and two dams on the Gilbert) through to a moderate scenario (WRD2), a scenario (WRD3) with moderate extraction from the Mitchell and no WRDs on the Flinders and Gilbert, and finally a lower impact scenario (WRD4 – no WRD on Mitchell River and moderate extraction on Flinders combined with a single dam on the Gilbert River). The other scenarios tested (WRD5-19) include alternative combinations and scenarios applied to the different rivers. Scenarios WRD5 and WRD6 are considered more extreme but unfeasible, but are included for purposes of comparing with predictions from other studies, and our preliminary simulations to try and bound the problem.

As predicted, when considering the four key WRDs relative to the baseline flow case, predicted catchment-system impacts increased with the greater volume of water extracted or impounded and number of rivers on which WRD scenarios were deployed.

Species comparisons and predicted impacts

In general, considering all modelled species, WRDs were predicted to have the greatest negative impacts when implemented on the Gilbert River, then the Flinders River compared with the Mitchell River. This is most plausibly explained by the fact the Mitchell River is a perennial system and thus less vulnerable to additional shocks. However, when evaluating the impacts of results on the marine ecosystem, it is also important to account for the relative abundance and local catches of species that are attributed to the Mitchell River system and hence its overall contribution to regional biomass and catch changes - for example, on average the largest proportion (28%) of common banana prawns is caught downstream of catchment region 2, and similarly for barramundi, with an average of 17% of catches located in region 2, and another 17% in Region 5. The type of water extraction also had a strong influence on model results – for example, constructing more than one impoundment dam on the Mitchell River generally had a much more negative impact on the system than water extraction scenarios from the same river. Hence, the proposed dam scenarios for the Gilbert River may also in part explain why the model predicted large impacts on the dependent catchment ecosystem. There were also substantial differences when evaluating extraction scenarios using medium versus low river flow threshold levels, with the latter resulting in large declines in dependent species, consistent with previous studies pointing to the need to maintain flows well above an ecosystem-minimum level (see eg WRD 16 vs WRD 17 in Appendix 19).

Our model results suggested freshwater sawfish may be the most sensitive species that we tested and seagrass the least sensitive. The former result was driven mainly by the likely poor current depletion status of *Pristis pristis* such that under some scenarios the additional pressures of altered river flows were predicted to lead to further declines or possibly crash of local sawfish populations (Table 20).

Seagrass predicted responses under alternative WRDs

Interestingly, modelled seagrass were predicted to occasionally do marginally better under some WRDs for some time periods. This is because seagrass growth in the MICE is assumed sensitive to light attenuation, which is in turn influenced by very high end-of-system flows that may increase littoral turbidity and hence decrease light attenuation. This was also the reason predicted changes in seagrass abundances that were compared across alternative WRDs were not consistent with the predictions for the other model species. Although on average there were fairly minor predicted changes in seagrass under alternative WRDs, our results nonetheless suggested that at times there could be up to a 7% decline in seagrass abundance relative to base levels (Table 16). Seagrass downstream of the Flinders River were most sensitive, followed by those located in the vicinity of the Gilbert River (Figure 68).

Model results were generally similar across the entire ensemble, although Model version 5 represented larger amplitudes in increases or decreases in seagrass growth in response to changes in flow, but with broadly similar responses (Figure 69).

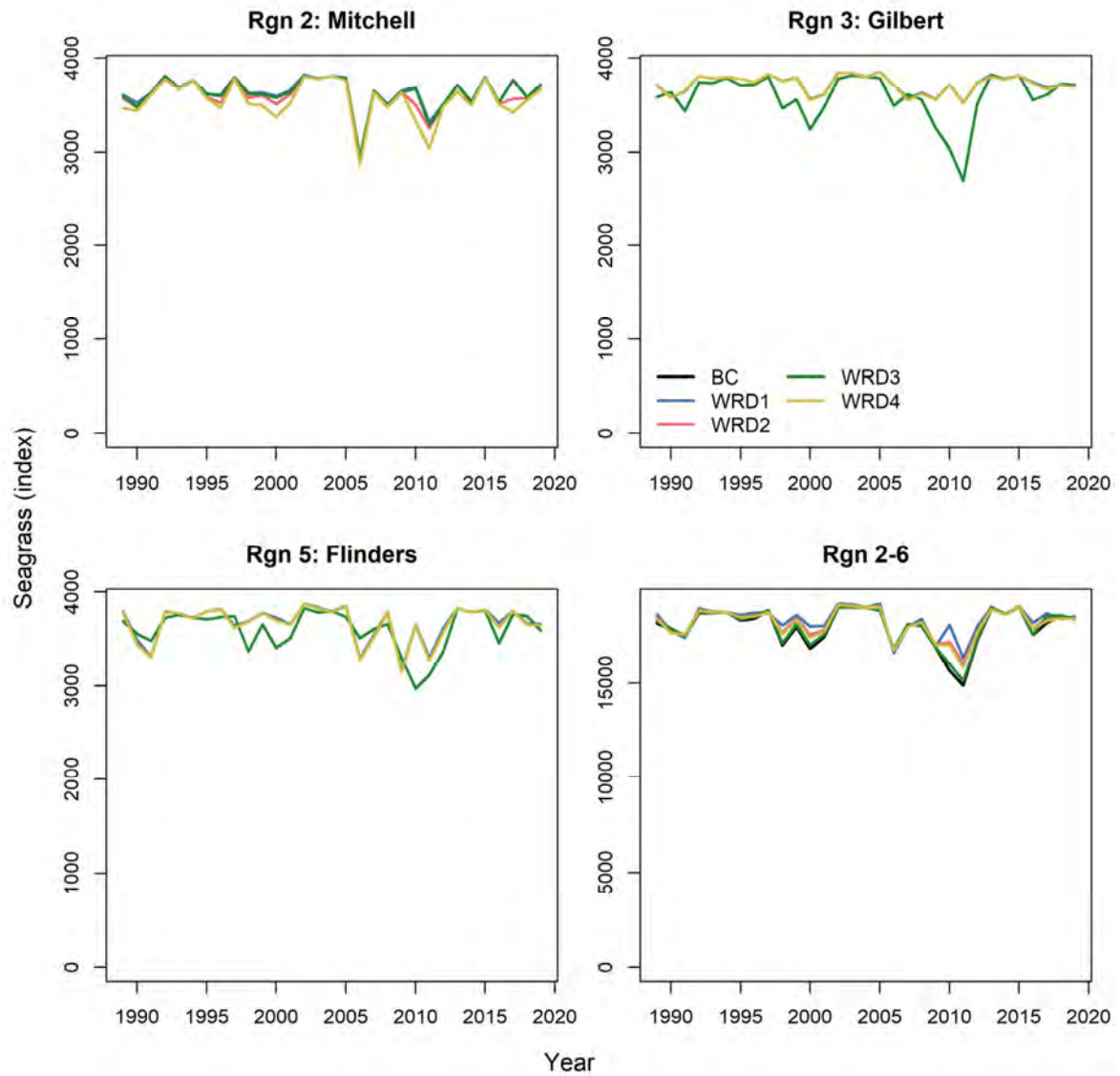


Figure 68. MICE-predicted changes in seagrass relative biomass under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 1. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, and WRD1 tracks under WRD4.

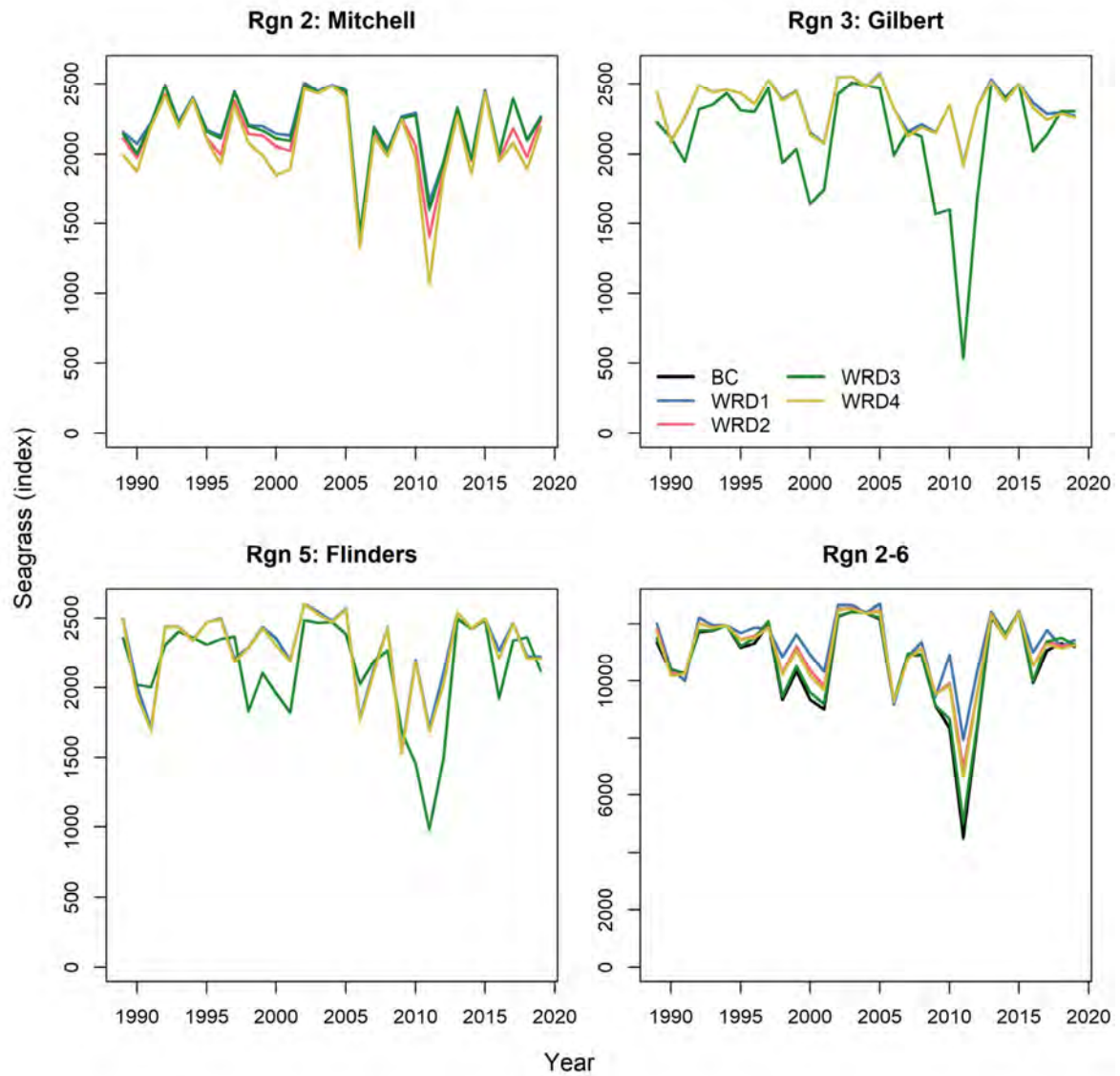


Figure 69. MICE-predicted changes in seagrass relative biomass under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 5. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4.

Mangroves predicted responses under alternative WRDs

Our model captured cumulative impacts on seagrass and mangroves, including from past cyclones. Mangroves were also modelled as sensitive to changes in sea surface height (Duke et al. 2017), for example as is associated with El Niño events. Hence when simulating likely impacts of WRDs by “rewriting” the past as though these had been in place, the combined effects of water extraction and other pressures were also simulated. Our results suggest that WRDs may have a dramatic effect on mangroves with the higher-volume WRD1 scenario predicted to result in average and maximum declines of 26% (max 44%), 44% (max 73%) and 28% (max 53%) for Mitchell, Gilbert and Flinders River systems, respectively (using Model version 1 (Figure 70, Table 16)). Building one dam

(WRD2) instead of two on the Gilbert River (WRD1) only marginally decreases the predicted impact on mangroves (40% decrease on average with max 71%). However, adding a single dam on the Mitchell River system (WRD17) had only a small effect whereas two dams (WRD16) was predicted to have a very large impact on mangroves in the Mitchell River with a 27% decline on average (Table 16). There were also large differences in performance for the same extraction rate but when comparing this combined with a low river flow pump TH and longer pump duration (WRD11) compared with a medium river flow pump TH and shorter pump duration (WRD12), with the former scenario resulting in a 22% average decline compared with 8% for the latter. A low river flow threshold allows the pumping of low-level flows whereby a large proportion of river flow is extracted and significantly less water passes downstream. In addition, a longer pump duration necessitates water extraction from flows other than peak flows, resulting in a higher proportion of the non-peak flows being extracted. Both pump routines disturb the pattern of river flow during low-level flows when water extraction disproportionately affects river flows and downstream ecosystem service provision. Asbridge et al. (2016) showed that estuarine mangrove habitats within the GoC replenish and expand via sediment loads on large river flows, hence a reduction in flow that reduces sediment delivery may limit or cause decline within estuarine deposition habitats.

Model results for mangroves were similar across the ensemble, with the Gilbert River catchment system consistently emerging as the most sensitive, followed by the Flinders River and finally Mitchell River.

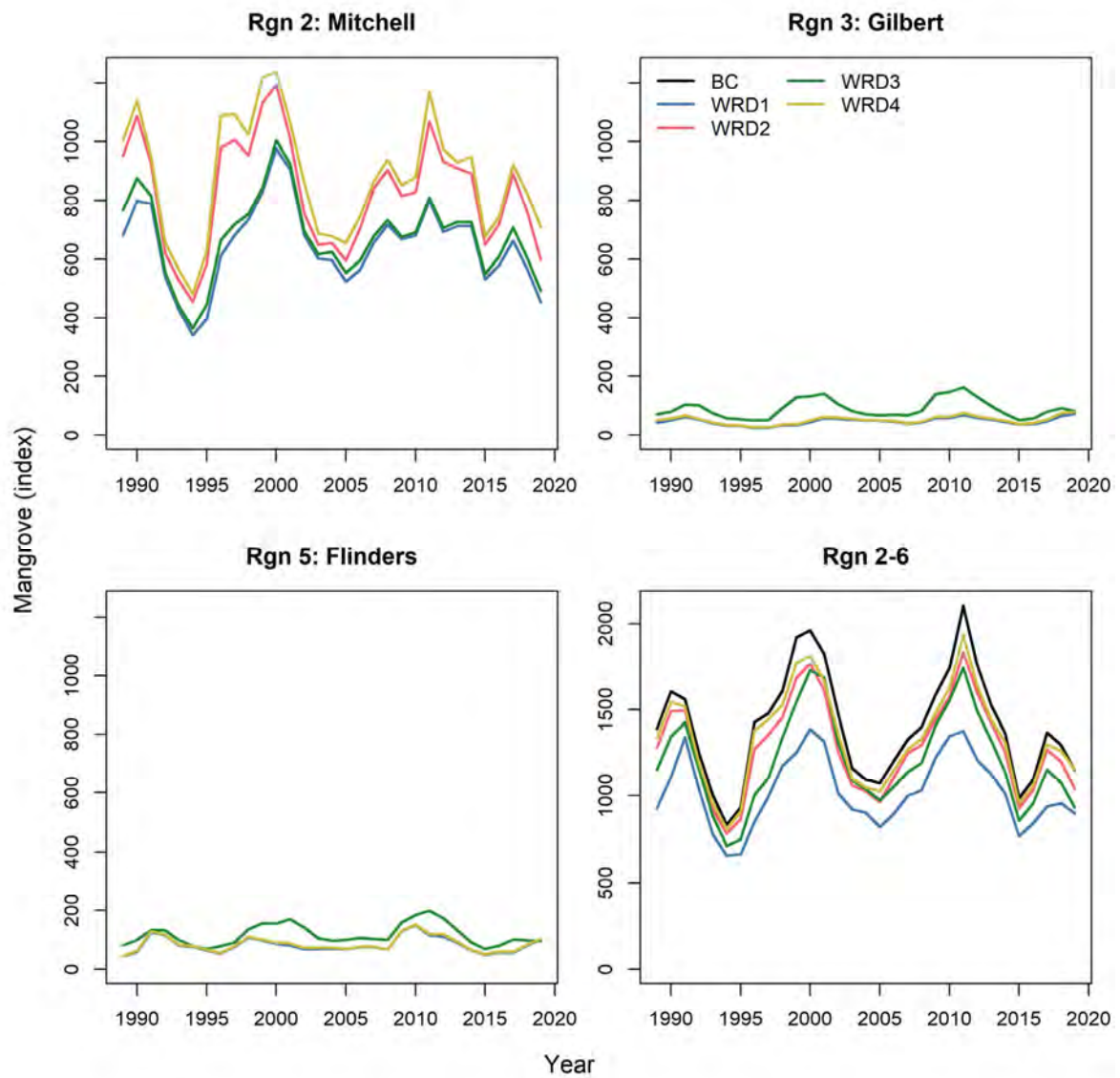


Figure 70. MICE-predicted changes in mangrove relative biomass under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 1. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4.

Table 16. Summary table of Model versions 1-5 showing minimum and mean biomass for seagrass and mangroves predicted under four water resource development scenarios (WRD1-WRD4) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.

Model version 1 - Seagrass and mangroves					
WRD Scenario	Rgn Name	Rel.Seagrass (Min)	Rel.Seagrass (Mean)	Rel.Mangrove (Min)	Rel.Mangrove (Mean)
BaseCase	Mitchell	1	1	1	1
BaseCase	Gilbert	1	1	1	1
BaseCase	Flinders	1	1	1	1
BaseCase	Rgn 2-6	1	1	1	1
WRD1	Mitchell	1	1.02	0.56	0.74
WRD1	Gilbert	0.98	1.04	0.27	0.56
WRD1	Flinders	0.94	1.02	0.47	0.72
WRD1	Rgn 2-6	0.99	1.02	0.59	0.74
WRD2	Mitchell	1	1.01	0.84	0.94
WRD2	Gilbert	0.98	1.04	0.29	0.6
WRD2	Flinders	0.93	1.02	0.5	0.74
WRD2	Rgn 2-6	0.99	1.01	0.85	0.92
WRD3	Mitchell	1	1.02	0.61	0.78
WRD3	Gilbert	1	1	1	1
WRD3	Flinders	1	1	1	1
WRD3	Rgn 2-6	1	1	0.7	0.86
WRD4	Mitchell	1	1	1	1
WRD4	Gilbert	0.98	1.04	0.29	0.6
WRD4	Flinders	0.93	1.02	0.5	0.74
WRD4	Rgn 2-6	0.99	1.01	0.91	0.95

Model version 2 - Seagrass and mangroves					
WRD Scenario	Rgn Name	Rel.Seagrass (Min)	Rel.Seagrass (Mean)	Rel.Mangrove (Min)	Rel.Mangrove (Mean)
BaseCase	Mitchell	1	1	1	1
BaseCase	Gilbert	1	1	1	1
BaseCase	Flinders	1	1	1	1
BaseCase	Rgn 2-6	1	1	1	1
WRD1	Mitchell	1	1.02	0.56	0.74
WRD1	Gilbert	0.98	1.04	0.27	0.56
WRD1	Flinders	0.94	1.02	0.47	0.72
WRD1	Rgn 2-6	0.99	1.02	0.59	0.74
WRD2	Mitchell	1	1.01	0.84	0.94
WRD2	Gilbert	0.98	1.04	0.29	0.6
WRD2	Flinders	0.93	1.02	0.5	0.74
WRD2	Rgn 2-6	0.99	1.01	0.85	0.92
WRD3	Mitchell	1	1.02	0.61	0.78
WRD3	Gilbert	1	1	1	1
WRD3	Flinders	1	1	1	1
WRD3	Rgn 2-6	1	1	0.7	0.86
WRD4	Mitchell	1	1	1	1

WRD4	Gilbert	0.98	1.04	0.29	0.6
WRD4	Flinders	0.93	1.02	0.5	0.74
WRD4	Rgn 2-6	0.99	1.01	0.91	0.95

Model version 3 - Seagrass and mangroves

WRD Scenario	Rgn Name	Rel.Seagrass (Min)	Rel.Seagrass (Mean)	Rel.Mangrove (Min)	Rel.Mangrove (Mean)
BaseCase	Mitchell	1	1	1	1
BaseCase	Gilbert	1	1	1	1
BaseCase	Flinders	1	1	1	1
BaseCase	Rgn 2-6	1	1	1	1
WRD1	Mitchell	1	1.02	0.56	0.74
WRD1	Gilbert	0.98	1.04	0.26	0.55
WRD1	Flinders	0.94	1.02	0.47	0.71
WRD1	Rgn 2-6	0.99	1.02	0.59	0.73
WRD2	Mitchell	1	1.01	0.82	0.94
WRD2	Gilbert	0.98	1.04	0.28	0.59
WRD2	Flinders	0.93	1.02	0.5	0.74
WRD2	Rgn 2-6	0.99	1.01	0.85	0.91
WRD3	Mitchell	1	1.02	0.61	0.77
WRD3	Gilbert	1	1	1	1
WRD3	Flinders	1	1	1	1
WRD3	Rgn 2-6	1	1	0.7	0.86
WRD4	Mitchell	1	1	1	1
WRD4	Gilbert	0.98	1.04	0.28	0.59
WRD4	Flinders	0.93	1.02	0.5	0.74
WRD4	Rgn 2-6	0.99	1.01	0.91	0.95

Model version 4 - Seagrass and mangroves

WRD Scenario	Rgn Name	Rel.Seagrass (Min)	Rel.Seagrass (Mean)	Rel.Mangrove (Min)	Rel.Mangrove (Mean)
BaseCase	Mitchell	1	1	1	1
BaseCase	Gilbert	1	1	1	1
BaseCase	Flinders	1	1	1	1
BaseCase	Rgn 2-6	1	1	1	1
WRD1	Mitchell	1	1.03	0.53	0.72
WRD1	Gilbert	0.99	1.06	0.25	0.53
WRD1	Flinders	0.93	1.03	0.44	0.7
WRD1	Rgn 2-6	0.99	1.03	0.57	0.71
WRD2	Mitchell	1	1.02	0.83	0.93
WRD2	Gilbert	0.98	1.06	0.27	0.57
WRD2	Flinders	0.93	1.03	0.47	0.72
WRD2	Rgn 2-6	0.99	1.02	0.84	0.91
WRD3	Mitchell	1	1.03	0.58	0.75
WRD3	Gilbert	1	1	1	1
WRD3	Flinders	1	1	1	1
WRD3	Rgn 2-6	1	1.01	0.68	0.84
WRD4	Mitchell	1	1	1	1
WRD4	Gilbert	0.98	1.06	0.27	0.57
WRD4	Flinders	0.93	1.03	0.47	0.72

WRD4	Rgn 2-6	0.99	1.02	0.91	0.95
Model version 5 - Seagrass and mangroves					
WRD Scenario	Rgn Name	Rel.Seagrass (Min)	Rel.Seagrass (Mean)	Rel.Mangrove (Min)	Rel.Mangrove (Mean)
BaseCase	Mitchell	1	1	1	1
BaseCase	Gilbert	1	1	1	1
BaseCase	Flinders	1	1	1	1
BaseCase	Rgn 2-6	1	1	1	1
WRD1	Mitchell	1	1.07	0.67	0.81
WRD1	Gilbert	0.99	1.19	0.43	0.68
WRD1	Flinders	0.85	1.09	0.55	0.8
WRD1	Rgn 2-6	0.98	1.08	0.7	0.81
WRD2	Mitchell	1	1.04	0.87	0.95
WRD2	Gilbert	0.98	1.18	0.45	0.71
WRD2	Flinders	0.85	1.08	0.59	0.81
WRD2	Rgn 2-6	0.99	1.05	0.88	0.94
WRD3	Mitchell	1	1.06	0.71	0.83
WRD3	Gilbert	1	1	1	1
WRD3	Flinders	1	1	1	1
WRD3	Rgn 2-6	1	1.01	0.79	0.9
WRD4	Mitchell	1	1	1	1
WRD4	Gilbert	0.98	1.18	0.45	0.71
WRD4	Flinders	0.85	1.08	0.59	0.81
WRD4	Rgn 2-6	0.99	1.04	0.93	0.96
Model Ensemble: Summary of average over ensemble					
WRD1	Mitchell	1.00	1.04	0.58	0.75
WRD1	Gilbert	0.99	1.07	0.30	0.58
WRD1	Flinders	0.92	1.04	0.48	0.73
WRD1	Rgn 2-6	0.99	1.04	0.61	0.75
WRD2	Mitchell	1.00	1.02	0.84	0.94
WRD2	Gilbert	0.98	1.07	0.32	0.62
WRD2	Flinders	0.91	1.03	0.52	0.75
WRD2	Rgn 2-6	0.99	1.02	0.85	0.92
WRD3	Mitchell	1.00	1.03	0.62	0.78
WRD3	Gilbert	1.00	1.00	1.00	1.00
WRD3	Flinders	1.00	1.00	1.00	1.00
WRD3	Rgn 2-6	1.00	1.01	0.71	0.86
WRD4	Mitchell	1.00	1.00	1.00	1.00
WRD4	Gilbert	0.98	1.07	0.32	0.62
WRD4	Flinders	0.91	1.03	0.52	0.75
WRD4	Rgn 2-6	0.99	1.02	0.91	0.95

Common banana prawns predicted responses under alternative WRDs

Common banana prawn biomass and catch were predicted to be negatively impacted across almost all WRD scenarios, with the predicted impacts on single river systems also predicted to negatively influence catches in surrounding regions when assuming system connectivity. The severity of impacts ranged from fairly minor under low extraction localised scenarios through to an average and maximum decline of 16% and 29% respectively of base catches under WRD1 and using Model version 1 (Figure 71-Figure 72). Under WRD1 the greatest impact was predicted for catches downstream of the Flinders River with a 19% decline on average extending to as much as a 37% decline in some years. As with all the modelled species, there was considerable temporal variability in the predicted impacts of alternative WRDs ranging from negligible effects in some years to very extreme declines in other years, highlighting the importance of evaluating impacts dynamically over a long time series as has been done in this study (Figure 71-Figure 74).

Construction of a single dam on the Mitchell River had a substantially lower impact than the two dam scenario (WRD17 vs WRD16), the latter predicted to cause a 12% on average (max 25%) decline in banana catches from that region alone and a regional (areas 2-6) total decline of 9% (max 17%) in banana catches (Table 17). Although average declines are useful for cross-comparing impacts, we note that the maximum modelled decline in any year is likely of more interest to fishing industry stakeholders because they rely on regular annual income and may not be able to withstand even a few years of low catch or periods when they are unable to break even economically (see economics discussion). The maximum predicted declines are also relevant for discussions around stock sustainability or conservation concerns related to the risk of species numbers dropping below levels considered viable to support ongoing population recruitment and replenishment.

Model results provide insights into possible mitigation strategies to reduce WRD impacts on individual species as well as the broader ecosystem. For example, common banana prawn catches and biomass were substantially more reduced under WRD scenarios using the same extraction amount but with a low versus medium flow threshold level (WRD18 vs WRD19). For example, the low flow threshold setting doubled the maximum annual decline in offshore catch (20% vs 10%, Table 17). The maintenance of low-level flows above 200 ML d⁻¹ supports the estuarine common banana prawn population by creating a brackish ecotone within a likely hypersaline estuary after the 9-month dry season without rainfall. In addition, early low-level floods cue emigration of larger resident juvenile common banana prawns from estuaries, though allowing ongoing recruitment of postlarvae and small juveniles that can survive low salinity levels. Similarly adjusting water extraction settings such as comparing a 1000GL scenario for the Mitchell River combined with 25 river flow TH and 15 days to pump, combined with WRD in the Gilbert and Flinders Rivers, (WRD2) with use of a low river flow TH and 15 days to pump, without WRD on the other rivers (WRD3) also substantially changed predicted outcomes, with the maximum local decline of 21% in banana catch under the first scenario reduced to 7% decline under the second scenario. In this case, the high river flow pump threshold did not support a higher prawn catch; probably due to reduction to the common banana prawn catch due to loss of flows and hence offshore catch from the adjacent Gilbert River.

When using Model version 2 with no “river portfolio effect”, there were some differences between the predicted decreases in catch from individual catchment systems, but the overall regional (regions 2-6) changes in catch and abundance were similar (Figure 73-Figure 74). For example, under WRD1 the no-connectivity Model version 2 predicted a bigger decrease in average catch (41% decrease with maximum decrease of 69%) of prawns caught in the Gilbert River catchment, but not the Mitchell and Flinders River catchments (which showed 15% decreases versus 18-19% average decreases under Model version 1) (Table 17). This was because the no-connectivity model version did not account for the plausible influence of changes to adjacent catchment systems in explaining the complex estuary-marine connections resulting in how much is caught from different regions.

The best-fit Model version 5 which incorporated the additional lagged boost that has been hypothesized to result after flood years (Burford and Faggotter 2021) resulted in considerably more

negative outcomes (Figure 75). For example, under WRD1, average catches for the Mitchell, Gilbert and Flinders Rivers all dropped to around three-quarters of the baseline level, with an overall regional catch decrease of 26%, and as much as a 39% decrease in the regional (regions 2-6) catch in some years. This model version provided the best explanation of observed common banana prawn catches per region, and hence corroborates research (Burford and Faggotter 2021) underscoring the longer-term benefits to system productivity that result from ongoing productivity boosts that result under natural flow regimes. Some similar results emerge when testing WRD2 that highlight the amplifying effect that system connectivity can have on regional increases or decreases in prawn abundance and catch. For example, under WRD2 which simulates a single dam scenario for the Gilbert River and moderate extractions for the Mitchell and Flinders Rivers, predicted mean catch decreases for each of these catchments are 4%, 11%, 15% under Model version 1 but drop further to 12%, 17% and 21% under the “flood productivity effect” Model version 5 (and see Table 17 for differences in maximum predicted decreases).

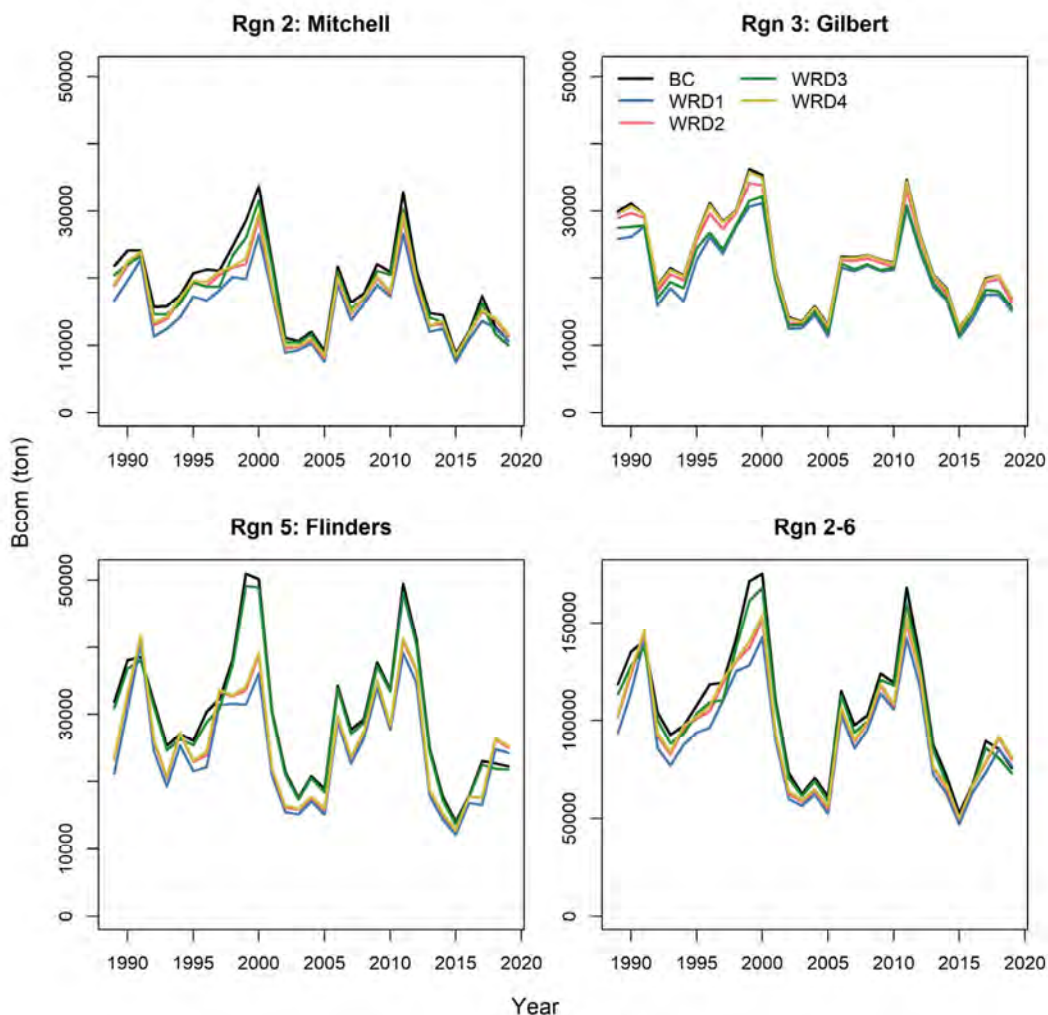


Figure 71. MICE-predicted changes to total annual common banana prawn commercially available biomass (Bcom, t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 1.

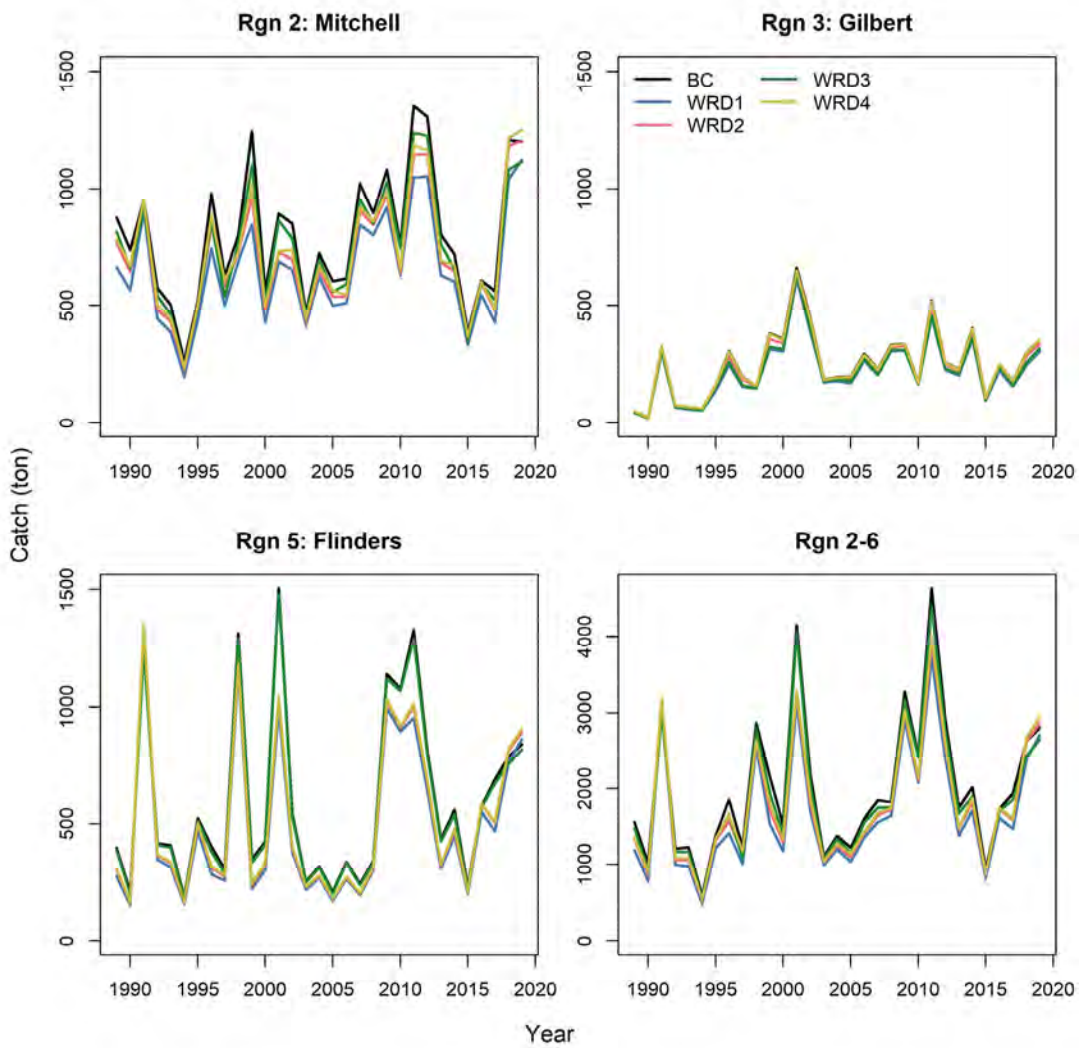


Figure 72. MICE-predicted changes to total annual common banana prawn catch (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 1.

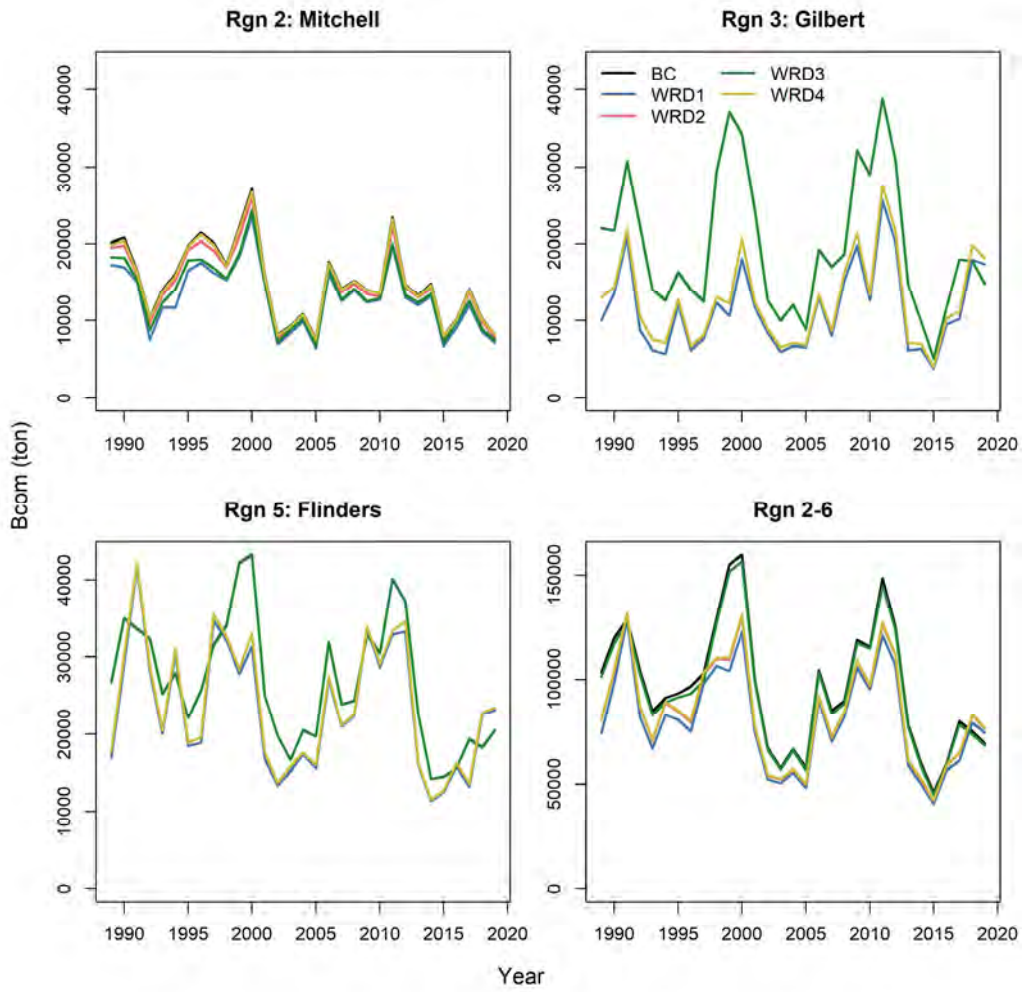


Figure 73. MICE-predicted changes to total annual common banana prawn commercially available biomass (Bcom, t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 2 (no connectivity between systems). Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4.

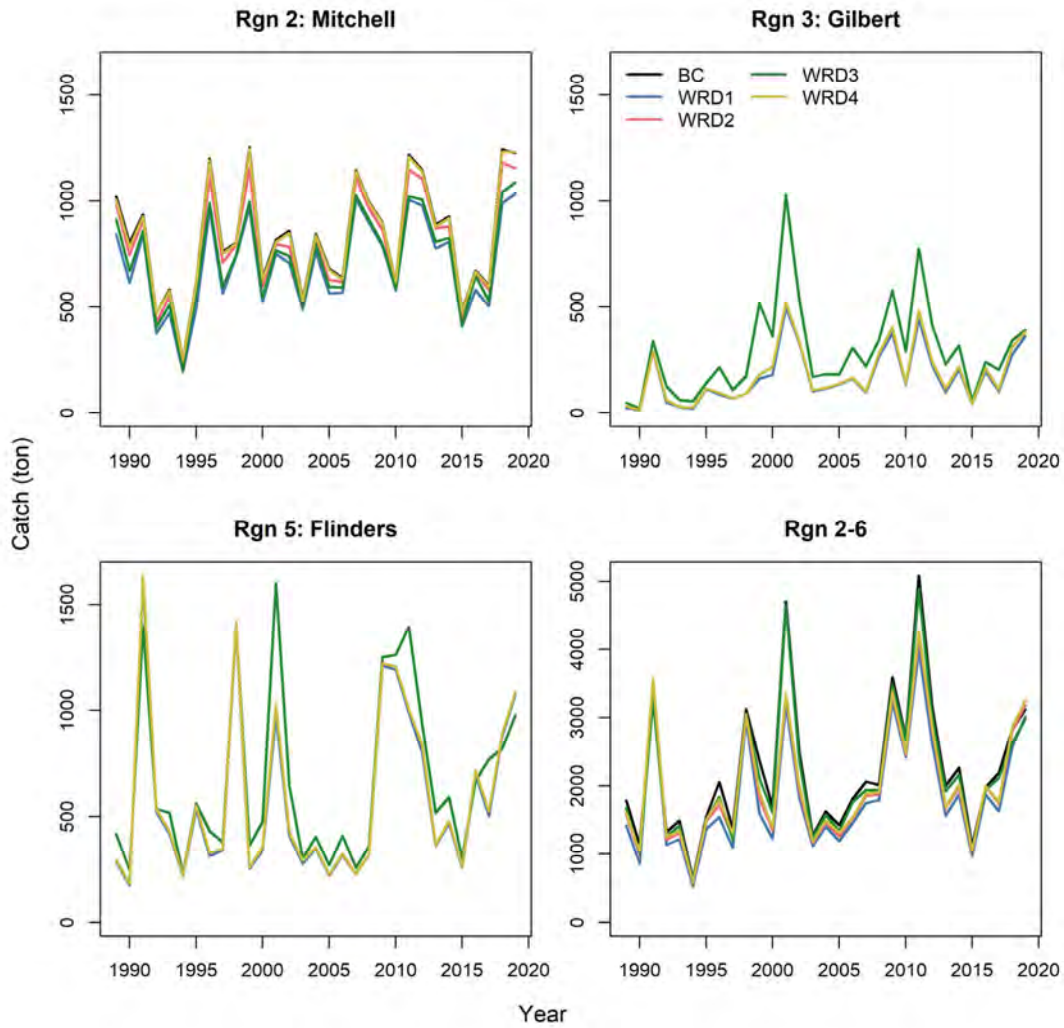


Figure 74. MICE-predicted changes to total annual common banana prawn catch (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 2. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4.

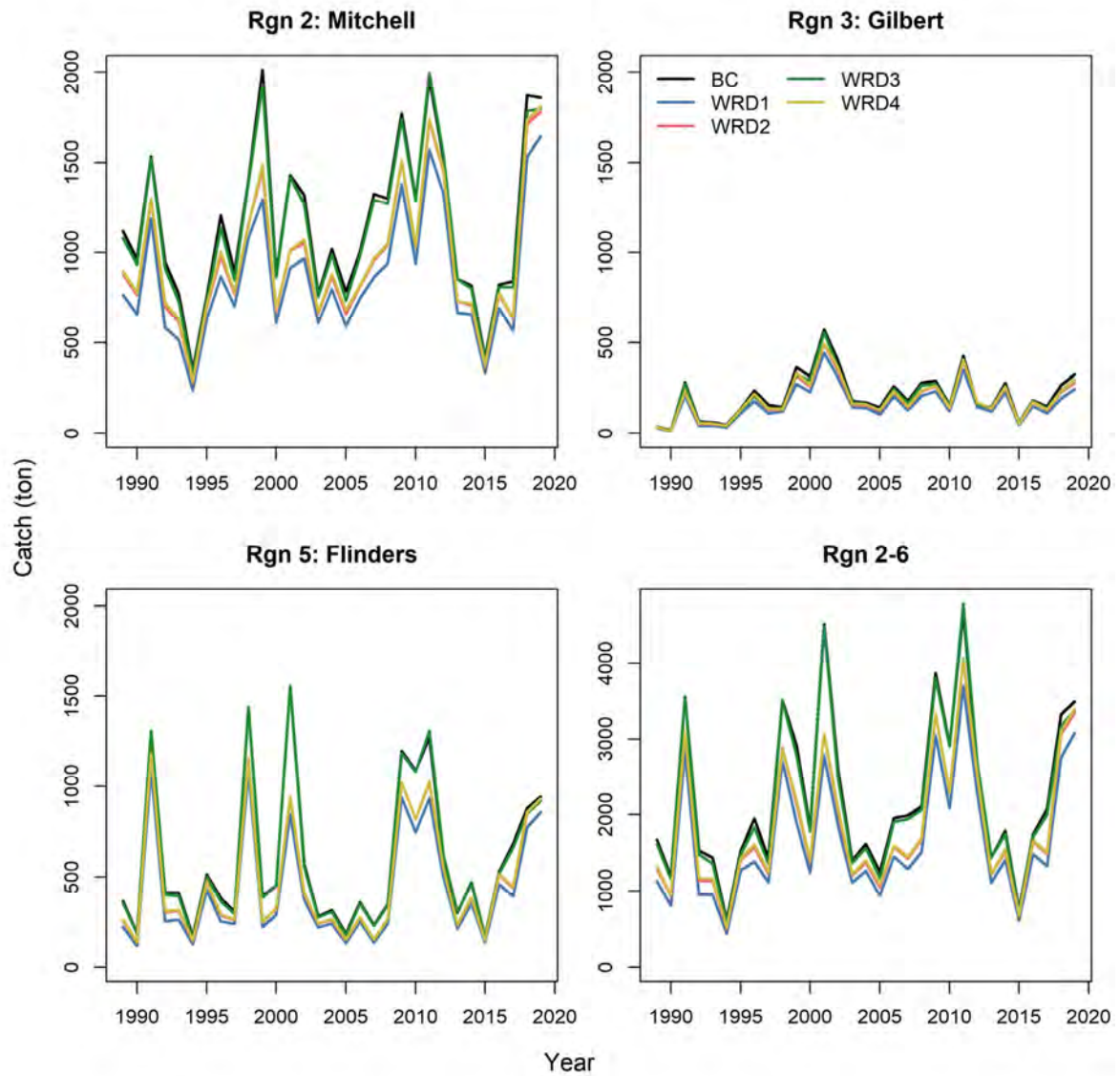


Figure 75. MICE-predicted changes to total common banana prawn catch (t) under baseline flow conditions compared with four alternative WRDs, shown for the Mitchell, Flinders and Gilbert catchment systems, and when using best-fit Model version 5 which assumes a “flood productivity effect” and predicts relatively greater responses of prawn abundance and catch to changing river flows.

Table 17. Summary table of Model versions 1-5 showing minimum and mean catch, commercially available biomass (Bcom) and spawning biomass (Bsp) for common banana prawns predicted under four water resource development scenarios (WRD1-WRD4) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions2-6 combined.

Model version 1 - common banana prawns							
WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)
BaseCase	Mitchell	1	1	1	1	1	1
BaseCase	Gilbert	1	1	1	1	1	1
BaseCase	Flinders	1	1	1	1	1	1
BaseCase	Rgn 2-6	1	1	1	1	1	1
WRD1	Mitchell	0.68	0.82	0.7	0.84	0.71	0.82
WRD1	Gilbert	0.79	0.88	0.81	0.89	0.77	0.89
WRD1	Flinders	0.63	0.81	0.62	0.83	0.63	0.81
WRD1	Rgn 2-6	0.71	0.84	0.75	0.87	0.76	0.86
WRD2	Mitchell	0.77	0.89	0.77	0.91	0.79	0.9
WRD2	Gilbert	0.93	0.96	0.94	0.97	0.94	0.97
WRD2	Flinders	0.68	0.85	0.66	0.87	0.67	0.85
WRD2	Rgn 2-6	0.79	0.9	0.8	0.92	0.81	0.91
WRD3	Mitchell	0.85	0.93	0.88	0.94	0.89	0.94
WRD3	Gilbert	0.82	0.91	0.85	0.92	0.86	0.92
WRD3	Flinders	0.94	0.97	0.95	0.98	0.95	0.98
WRD3	Rgn 2-6	0.88	0.95	0.92	0.96	0.92	0.96
WRD4	Mitchell	0.82	0.91	0.8	0.93	0.82	0.91
WRD4	Gilbert	0.98	0.99	0.99	0.99	0.98	0.99
WRD4	Flinders	0.69	0.86	0.67	0.88	0.68	0.86
WRD4	Rgn 2-6	0.79	0.91	0.82	0.93	0.83	0.92
Model version 2 - common banana prawns							
WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)
BaseCase	Mitchell	1	1	1	1	1	1
BaseCase	Gilbert	1	1	1	1	1	1
BaseCase	Flinders	1	1	1	1	1	1
BaseCase	Rgn 2-6	1	1	1	1	1	1
WRD1	Mitchell	0.74	0.85	0.73	0.86	0.68	0.86
WRD1	Gilbert	0.31	0.59	0.29	0.61	0.29	0.57
WRD1	Flinders	0.61	0.85	0.63	0.87	0.64	0.85
WRD1	Rgn 2-6	0.67	0.83	0.67	0.85	0.68	0.83
WRD2	Mitchell	0.91	0.95	0.93	0.96	0.94	0.96
WRD2	Gilbert	0.35	0.63	0.33	0.66	0.34	0.64
WRD2	Flinders	0.64	0.87	0.66	0.89	0.67	0.87
WRD2	Rgn 2-6	0.71	0.89	0.7	0.88	0.72	0.87
WRD3	Mitchell	0.78	0.89	0.84	0.9	0.85	0.9
WRD3	Gilbert	1	1	1	1	1	1
WRD3	Flinders	1	1	1	1	1	1
WRD3	Rgn 2-6	0.88	0.95	0.96	0.98	0.96	0.98
WRD4	Mitchell	0.98	0.99	0.98	0.99	0.98	0.99

WRD4	Gilbert	0.35	0.63	0.33	0.66	0.34	0.64
WRD4	Flinders	0.64	0.87	0.66	0.89	0.67	0.87
WRD4	Rgn 2-6	0.71	0.91	0.71	0.89	0.72	0.87

Model version 3 - common banana prawns

WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)
BaseCase	Mitchell	1	1	1	1	1	1
BaseCase	Gilbert	1	1	1	1	1	1
BaseCase	Flinders	1	1	1	1	1	1
BaseCase	Rgn 2-6	1	1	1	1	1	1
WRD1	Mitchell	0.65	0.8	0.68	0.82	0.69	0.81
WRD1	Gilbert	0.81	0.88	0.81	0.89	0.78	0.89
WRD1	Flinders	0.6	0.77	0.59	0.8	0.6	0.78
WRD1	Rgn 2-6	0.69	0.82	0.73	0.86	0.74	0.85
WRD2	Mitchell	0.74	0.87	0.75	0.89	0.77	0.88
WRD2	Gilbert	0.93	0.96	0.94	0.97	0.94	0.97
WRD2	Flinders	0.64	0.81	0.62	0.84	0.64	0.82
WRD2	Rgn 2-6	0.77	0.87	0.78	0.9	0.79	0.89
WRD3	Mitchell	0.87	0.94	0.9	0.95	0.91	0.95
WRD3	Gilbert	0.84	0.91	0.87	0.92	0.88	0.93
WRD3	Flinders	0.95	0.98	0.96	0.98	0.96	0.98
WRD3	Rgn 2-6	0.9	0.95	0.94	0.97	0.94	0.97
WRD4	Mitchell	0.79	0.9	0.78	0.91	0.79	0.89
WRD4	Gilbert	0.98	0.99	0.98	0.99	0.98	0.99
WRD4	Flinders	0.65	0.82	0.63	0.84	0.65	0.82
WRD4	Rgn 2-6	0.77	0.89	0.8	0.91	0.81	0.9

Model version 4 - common banana prawns

WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)
BaseCase	Mitchell	1	1	1	1	1	1
BaseCase	Gilbert	1	1	1	1	1	1
BaseCase	Flinders	1	1	1	1	1	1
BaseCase	Rgn 2-6	1	1	1	1	1	1
WRD1	Mitchell	0.68	0.82	0.7	0.84	0.72	0.82
WRD1	Gilbert	0.79	0.88	0.81	0.89	0.77	0.89
WRD1	Flinders	0.63	0.81	0.62	0.83	0.63	0.81
WRD1	Rgn 2-6	0.71	0.84	0.75	0.88	0.76	0.86
WRD2	Mitchell	0.77	0.89	0.77	0.91	0.79	0.9
WRD2	Gilbert	0.93	0.96	0.94	0.97	0.94	0.97
WRD2	Flinders	0.68	0.85	0.66	0.87	0.67	0.85
WRD2	Rgn 2-6	0.79	0.9	0.8	0.92	0.81	0.91
WRD3	Mitchell	0.85	0.93	0.88	0.94	0.89	0.94
WRD3	Gilbert	0.82	0.91	0.86	0.92	0.86	0.92
WRD3	Flinders	0.94	0.97	0.95	0.98	0.95	0.98
WRD3	Rgn 2-6	0.88	0.95	0.92	0.96	0.93	0.96
WRD4	Mitchell	0.82	0.91	0.8	0.93	0.82	0.91
WRD4	Gilbert	0.98	0.99	0.98	0.99	0.98	0.99
WRD4	Flinders	0.69	0.86	0.67	0.88	0.68	0.86

WRD4	Rgn 2-6	0.79	0.91	0.82	0.93	0.83	0.92
Model version 5 - common banana prawns							
WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)
BaseCase	Mitchell	1	1	1	1	1	1
BaseCase	Gilbert	1	1	1	1	1	1
BaseCase	Flinders	1	1	1	1	1	1
BaseCase	Rgn 2-6	1	1	1	1	1	1
WRD1	Mitchell	0.61	0.75	0.59	0.76	0.59	0.73
WRD1	Gilbert	0.63	0.77	0.61	0.77	0.61	0.76
WRD1	Flinders	0.54	0.72	0.54	0.73	0.55	0.69
WRD1	Rgn 2-6	0.61	0.74	0.6	0.76	0.59	0.73
WRD2	Mitchell	0.71	0.83	0.72	0.85	0.72	0.83
WRD2	Gilbert	0.79	0.88	0.8	0.88	0.8	0.88
WRD2	Flinders	0.6	0.79	0.61	0.81	0.63	0.78
WRD2	Rgn 2-6	0.68	0.83	0.72	0.84	0.73	0.83
WRD3	Mitchell	0.94	0.97	0.93	0.98	0.93	0.97
WRD3	Gilbert	0.87	0.93	0.88	0.94	0.89	0.93
WRD3	Flinders	0.95	0.98	0.95	0.99	0.95	0.98
WRD3	Rgn 2-6	0.94	0.97	0.94	0.98	0.94	0.97
WRD4	Mitchell	0.71	0.84	0.73	0.85	0.73	0.83
WRD4	Gilbert	0.82	0.9	0.8	0.9	0.81	0.9
WRD4	Flinders	0.6	0.8	0.62	0.82	0.63	0.79
WRD4	Rgn 2-6	0.68	0.84	0.73	0.85	0.73	0.83
Model Ensemble: Summary of average over ensemble							
WRD1	Mitchell	0.67	0.80	0.68	0.82	0.68	0.81
WRD1	Gilbert	0.67	0.79	0.67	0.81	0.64	0.79
WRD1	Flinders	0.60	0.79	0.60	0.81	0.61	0.79
WRD1	Rgn 2-6	0.68	0.81	0.70	0.84	0.71	0.82
WRD2	Mitchell	0.78	0.89	0.79	0.90	0.80	0.89
WRD2	Gilbert	0.78	0.88	0.79	0.89	0.79	0.88
WRD2	Flinders	0.65	0.84	0.64	0.86	0.65	0.83
WRD2	Rgn 2-6	0.74	0.88	0.76	0.89	0.77	0.88
WRD3	Mitchell	0.86	0.93	0.89	0.94	0.89	0.94
WRD3	Gilbert	0.87	0.93	0.89	0.94	0.90	0.94
WRD3	Flinders	0.96	0.98	0.96	0.98	0.96	0.98
WRD3	Rgn 2-6	0.90	0.96	0.94	0.97	0.94	0.97
WRD4	Mitchell	0.82	0.91	0.82	0.92	0.83	0.91
WRD4	Gilbert	0.82	0.90	0.82	0.91	0.82	0.90
WRD4	Flinders	0.65	0.84	0.65	0.86	0.66	0.84
WRD4	Rgn 2-6	0.75	0.89	0.77	0.90	0.78	0.89

Barramundi predicted responses under alternative WRDs

Barramundi were generally predicted to be most sensitive to the WRDs applied to the Gilbert River region, with both a single and two dams predicted to cause large declines to the local populations (Figure 76-Figure 77). In using the same flow threshold settings as for prawns (Model version 1), two dams (WRD1) almost halves the population and reduces the catches down to about a third of the base level, falling even lower (to 16% of baseline catch) in the worst year. The single dam scenario (WRD2) also predicts a biomass decrease of 42% on average and catch decrease of around 30%, with a max decrease of 46% (Table 18). However, these results are less extreme when using the model versions 2-4 with the flow threshold relationship refitted for barramundi. Under Models 2 and 3 for example, the predicted decrease in average barramundi catch is 16% and 57% respectively, although in some years catch is predicted to decline by as much as 34% (Model 2) (Table 18).

Using Model version 1, the Flinders River WRDs tested resulted in similar predicted decreases in barramundi abundance (ca. 20% decrease) associated with a 28% average and 60% maximum decline in catches (Table 18). However under Model version 2, much smaller declines were predicted for the Flinders River, with a decrease in average and minimum catch of around 5% down to 18% in the worst year under Model 2. As barramundi are long-lived, there is a lag before the effects of different WRDs translate into differences in the spawning biomass (Figure 78).

The sensitivity of barramundi to changes in flow results not only from assumed impacts on recruitment, but also on survival of the younger age classes, which is in turn based on the findings of previous studies. Juvenile barramundi rely on the inundation of palustrine, dry-riverine and supra-littoral estuarine habitats and connectivity between these habitats and estuarine adult spawning locations to support the movement of juveniles of this catadromous species to their freshwater (Crook et al. 2016, Roberts et al. 2019). Early wet season low-level flows re-establish connectivity between the estuary and riverine pool refugia that withstood the stress of the annual dry season, and high-level river flows inundate floodplain and saltflat habitats.

As the Mitchell River is a perennial system that exhibits less environmental extremes than the Gilbert and Flinders Rivers, WRDs are predicted to have a less extreme impact on the local population. Most WRD scenarios were predicted to result in a decrease ranging from fairly minor changes (decline in catch and abundance of 3-4%; WRD2) through more moderate impacts (decline in catch and abundance of 13%; WRD3) to the largest decreases predicted under the two dam scenario (decline in catch and abundance of 37% with a maximum of 50%; WRD16). As was the case for the other modelled species, use of a low flow pump TH value caused a much more substantial negative impact on barramundi catches and abundance than using a higher TH level (cf WRD18 vs WRD19). Overall, the model ensemble suggested average catch would decrease up to about 20%, with a maximum decrease of around 27%, under WRDs1 and 3, with more minor decreases predicted under WRDs 2 and 4.

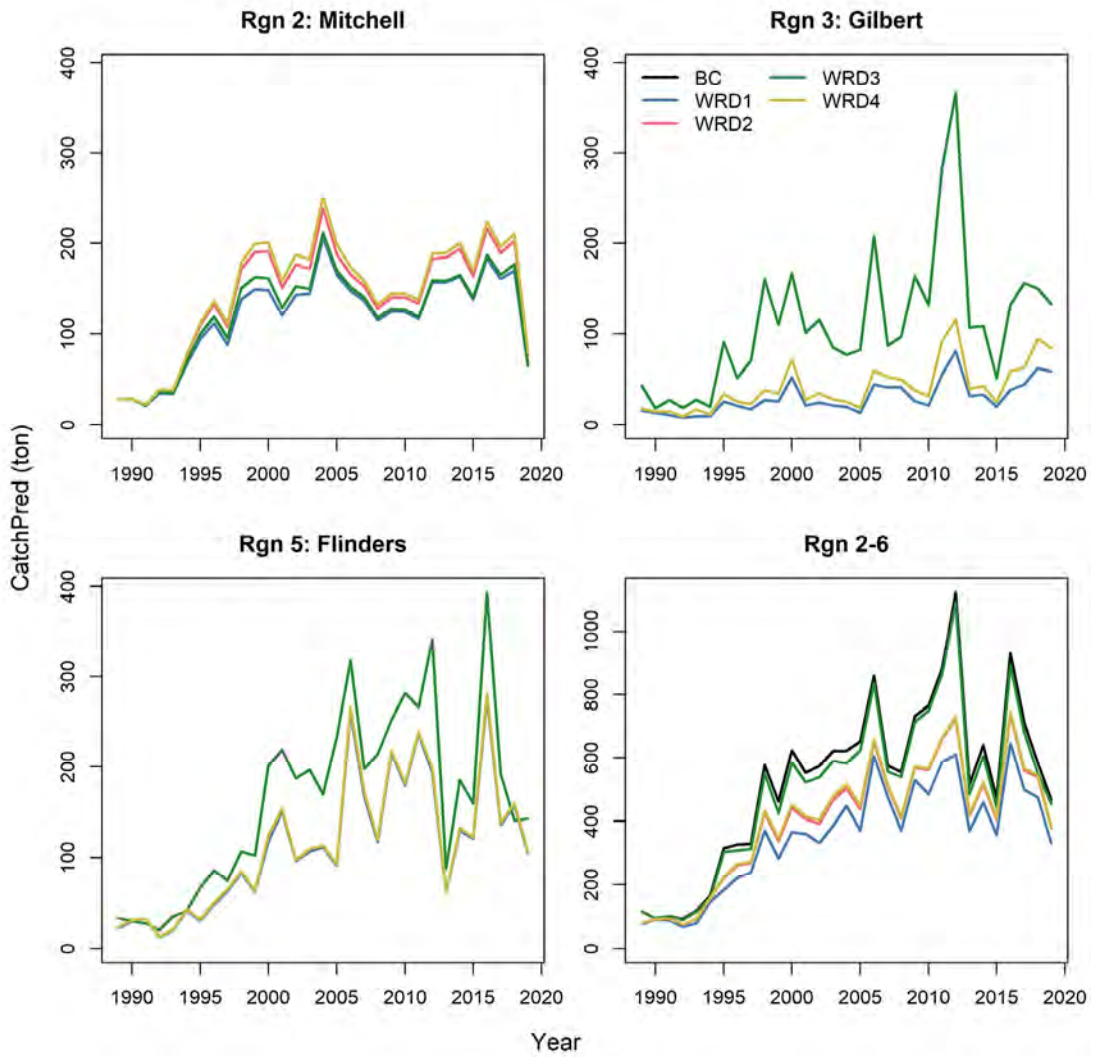


Figure 76. MICE-predicted changes to total annual barramundi catch (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 1. Note for the Mitchell River, BC trajectory tracks under WRD4 and for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2 tracks under WRD4.

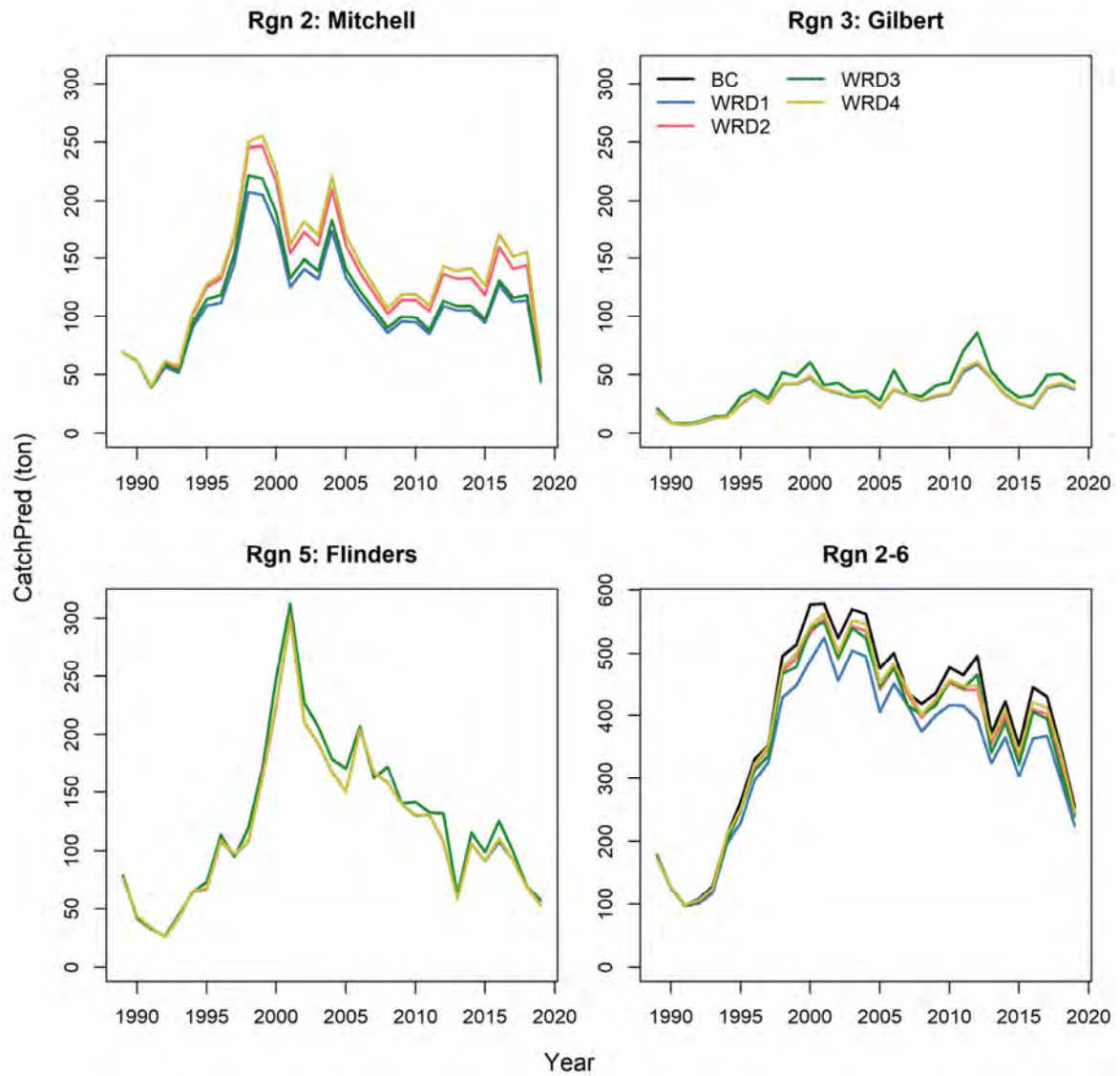


Figure 77. MICE-predicted changes to total annual barramundi catch (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 2. Note for the Mitchell River, BC trajectory tracks under WRD4 and for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4.

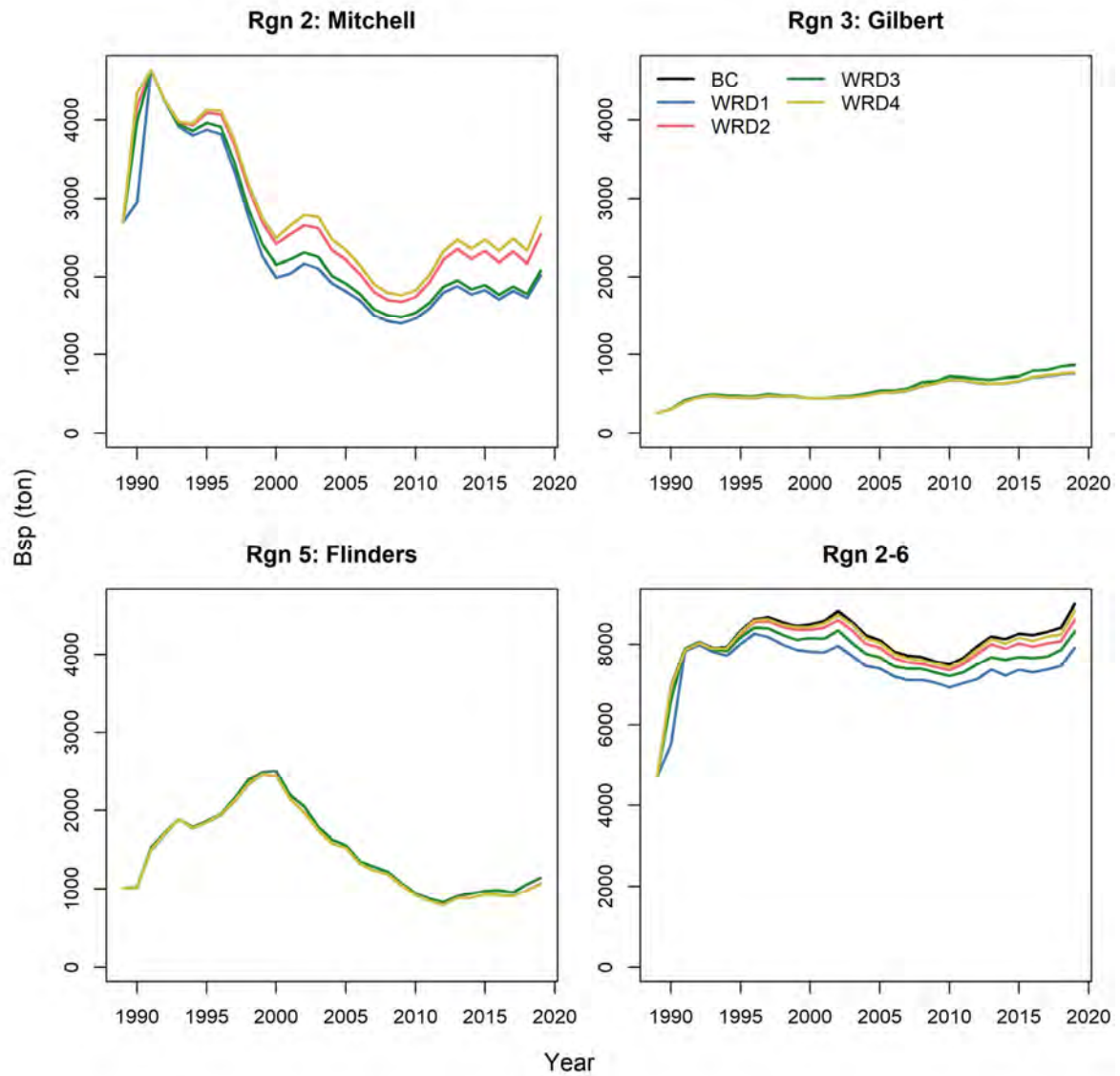


Figure 78. MICE-predicted changes to total annual barramundi spawning biomass (Bsp) (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 2. Note for the Mitchell River, BC trajectory tracks under WRD4 and for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4.

Table 18. Summary table of Model versions 1-5 showing minimum and mean catch, commercially available biomass (Bcom) and spawning biomass (Bsp) for barramundi predicted under four water resource development scenarios (WRD1-WRD4) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions2-6 combined.

Model version 1 - barramundi							
WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)
BaseCase	Mitchell	1	1	1	1	1	1
BaseCase	Gilbert	1	1	1	1	1	1
BaseCase	Flinders	1	1	1	1	1	1
BaseCase	Rgn 2-6	1	1	1	1	1	1
WRD1	Mitchell	0.74	0.84	0.74	0.84	0.78	0.85
WRD1	Gilbert	0.16	0.32	0.38	0.51	0.44	0.55
WRD1	Flinders	0.39	0.72	0.71	0.79	0.74	0.82
WRD1	Rgn 2-6	0.54	0.7	0.73	0.79	0.76	0.82
WRD2	Mitchell	0.94	0.97	0.94	0.97	0.95	0.97
WRD2	Gilbert	0.23	0.42	0.45	0.58	0.51	0.63
WRD2	Flinders	0.4	0.73	0.72	0.8	0.75	0.83
WRD2	Rgn 2-6	0.65	0.79	0.84	0.88	0.89	0.92
WRD3	Mitchell	0.81	0.87	0.81	0.87	0.83	0.88
WRD3	Gilbert	1	1	1	1	1	1
WRD3	Flinders	1	1	1	1	1	1
WRD3	Rgn 2-6	0.92	0.96	0.93	0.96	0.92	0.94
WRD4	Mitchell	1	1	1	1	1	1
WRD4	Gilbert	0.23	0.42	0.45	0.58	0.51	0.63
WRD4	Flinders	0.4	0.73	0.72	0.8	0.75	0.83
WRD4	Rgn 2-6	0.65	0.79	0.85	0.89	0.91	0.93
Model version 2 - barramundi							
WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)
BaseCase	Mitchell	1	1	1	1	1	1
BaseCase	Gilbert	1	1	1	1	1	1
BaseCase	Flinders	1	1	1	1	1	1
BaseCase	Rgn 2-6	1	1	1	1	1	1
WRD1	Mitchell	0.73	0.81	0.73	0.81	0.68	0.82
WRD1	Gilbert	0.66	0.84	0.83	0.92	0.88	0.95
WRD1	Flinders	0.82	0.95	0.92	0.96	0.92	0.97
WRD1	Rgn 2-6	0.8	0.89	0.87	0.92	0.79	0.92
WRD2	Mitchell	0.93	0.96	0.93	0.96	0.92	0.96
WRD2	Gilbert	0.69	0.85	0.84	0.93	0.89	0.95
WRD2	Flinders	0.83	0.95	0.92	0.96	0.92	0.97
WRD2	Rgn 2-6	0.89	0.96	0.95	0.98	0.96	0.98
WRD3	Mitchell	0.76	0.85	0.76	0.85	0.75	0.86
WRD3	Gilbert	1	1	1	1	1	1
WRD3	Flinders	1	1	1	1	1	1
WRD3	Rgn 2-6	0.89	0.95	0.93	0.95	0.92	0.96

WRD4	Mitchell	1	1	1	1	1	1
WRD4	Gilbert	0.69	0.85	0.84	0.93	0.89	0.95
WRD4	Flinders	0.83	0.95	0.92	0.96	0.92	0.97
WRD4	Rgn 2-6	0.9	0.97	0.97	0.99	0.98	0.99

Model version 3 - barramundi

WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)
BaseCase	Mitchell	1	1	1	1	1	1
BaseCase	Gilbert	1	1	1	1	1	1
BaseCase	Flinders	1	1	1	1	1	1
BaseCase	Rgn 2-6	1	1	1	1	1	1
WRD1	Mitchell	0.95	0.97	0.95	0.97	0.95	0.97
WRD1	Gilbert	0.25	0.43	0.48	0.61	0.49	0.64
WRD1	Flinders	0.58	0.82	0.76	0.86	0.62	0.89
WRD1	Rgn 2-6	0.6	0.8	0.83	0.89	0.89	0.93
WRD2	Mitchell	0.97	0.99	0.97	0.99	0.98	0.99
WRD2	Gilbert	0.31	0.51	0.52	0.66	0.53	0.69
WRD2	Flinders	0.59	0.83	0.77	0.86	0.62	0.89
WRD2	Rgn 2-6	0.66	0.84	0.88	0.93	0.93	0.95
WRD3	Mitchell	0.95	0.97	0.95	0.97	0.96	0.97
WRD3	Gilbert	1	1	1	1	1	1
WRD3	Flinders	1	1	1	1	1	1
WRD3	Rgn 2-6	0.98	0.99	0.98	0.99	0.97	0.98
WRD4	Mitchell	1	1	1	1	1	1
WRD4	Gilbert	0.31	0.51	0.52	0.66	0.53	0.69
WRD4	Flinders	0.59	0.83	0.77	0.86	0.62	0.89
WRD4	Rgn 2-6	0.67	0.84	0.89	0.93	0.93	0.96

Model version 4 - barramundi

WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)
BaseCase	Mitchell	1	1	1	1	1	1
BaseCase	Gilbert	1	1	1	1	1	1
BaseCase	Flinders	1	1	1	1	1	1
BaseCase	Rgn 2-6	1	1	1	1	1	1
WRD1	Mitchell	0.77	0.84	0.77	0.84	0.77	0.86
WRD1	Gilbert	0.65	0.84	0.82	0.93	0.93	0.97
WRD1	Flinders	0.84	0.95	0.93	0.97	0.96	0.98
WRD1	Rgn 2-6	0.8	0.9	0.87	0.92	0.91	0.95
WRD2	Mitchell	0.94	0.97	0.94	0.97	0.94	0.97
WRD2	Gilbert	0.68	0.86	0.83	0.94	0.94	0.97
WRD2	Flinders	0.84	0.95	0.93	0.97	0.96	0.98
WRD2	Rgn 2-6	0.89	0.95	0.95	0.98	0.98	0.99
WRD3	Mitchell	0.8	0.87	0.8	0.87	0.79	0.89
WRD3	Gilbert	1	1	1	1	1	1
WRD3	Flinders	1	1	1	1	1	1
WRD3	Rgn 2-6	0.93	0.96	0.94	0.96	0.95	0.97

WRD4	Mitchell	1	1	1	1	1	1
WRD4	Gilbert	0.68	0.86	0.83	0.94	0.94	0.97
WRD4	Flinders	0.84	0.95	0.93	0.97	0.96	0.98
WRD4	Rgn 2-6	0.89	0.96	0.97	0.99	0.99	0.99

Model version 5 - barramundi

WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)
BaseCase	Mitchell	1	1	1	1	1	1
BaseCase	Gilbert	1	1	1	1	1	1
BaseCase	Flinders	1	1	1	1	1	1
BaseCase	Rgn 2-6	1	1	1	1	1	1
WRD1	Mitchell	0.76	0.83	0.76	0.83	0.78	0.85
WRD1	Gilbert	0.21	0.39	0.52	0.62	0.59	0.77
WRD1	Flinders	0.45	0.78	0.77	0.86	0.85	0.94
WRD1	Rgn 2-6	0.56	0.72	0.75	0.81	0.78	0.86
WRD2	Mitchell	0.94	0.96	0.94	0.96	0.95	0.97
WRD2	Gilbert	0.29	0.49	0.57	0.68	0.65	0.81
WRD2	Flinders	0.45	0.79	0.77	0.86	0.85	0.94
WRD2	Rgn 2-6	0.7	0.82	0.86	0.91	0.91	0.95
WRD3	Mitchell	0.8	0.86	0.8	0.86	0.81	0.88
WRD3	Gilbert	1	1	1	1	1	1
WRD3	Flinders	1	1	1	1	1	1
WRD3	Rgn 2-6	0.92	0.96	0.96	0.98	0.99	0.99
WRD4	Mitchell	1	1	1	1	1	1
WRD4	Gilbert	0.29	0.49	0.57	0.68	0.65	0.81
WRD4	Flinders	0.45	0.79	0.77	0.86	0.85	0.94
WRD4	Rgn 2-6	0.7	0.83	0.87	0.91	0.91	0.96

Model Ensemble: Summary of average over ensemble

WRD1	Mitchell	0.79	0.86	0.79	0.86	0.79	0.87
WRD1	Gilbert	0.39	0.56	0.61	0.72	0.67	0.78
WRD1	Flinders	0.61	0.84	0.82	0.89	0.82	0.92
WRD1	Rgn 2-6	0.66	0.80	0.81	0.86	0.83	0.89
WRD2	Mitchell	0.95	0.97	0.95	0.97	0.95	0.97
WRD2	Gilbert	0.44	0.63	0.64	0.76	0.70	0.81
WRD2	Flinders	0.62	0.85	0.82	0.89	0.82	0.92
WRD2	Rgn 2-6	0.76	0.87	0.90	0.93	0.93	0.96
WRD3	Mitchell	0.82	0.88	0.82	0.88	0.83	0.90
WRD3	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD3	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD3	Rgn 2-6	0.93	0.96	0.95	0.97	0.95	0.97
WRD4	Mitchell	1.00	1.00	1.00	1.00	1.00	1.00
WRD4	Gilbert	0.44	0.63	0.64	0.76	0.70	0.81
WRD4	Flinders	0.62	0.85	0.82	0.89	0.82	0.92
WRD4	Rgn 2-6	0.76	0.88	0.91	0.94	0.94	0.97

Mud crabs predicted responses under alternative WRDs

Similar to barramundi, mud crabs were predicted to be most sensitive to water development on the Flinders and Gilbert rivers, with little impact to mud crab biomass or catches predicted for the Mitchell River system.

For the Flinders River, across all model versions, there wasn't a noticeable difference between a high-impact (e.g. WRD1) and a moderate-impact extraction (e.g. WRD2) (Figure 79, Table 19). Using Model version 1, catches reduced on average by 46% and 47% respectively, and the most extreme reduction by 86% and 87% (Table 19, Figure 79). Similarly, available biomass was, on average, predicted to halve under both moderate and high impact water development scenarios (Table 19). As with other species, there was temporal variability in catch reductions, with less impact on mud crab catches in years when catch is already low (presumably due to already lower flow in those years) and greater in years when catches are large (presumably due to more flow in those years) (Figure 79).

For both extraction scenarios in the Flinders, mud crab catch reductions were largest under Model 4, and smallest under Model 2 (Figure 80), with our base model (Model 1), best-fit model (Model 5, Figure 81) and Model 3 being somewhere in between (Table 19). Average catch reductions under a high-impact scenario ranged from 32% to 49% across all 5 models, while the largest catch reductions ranged from 75% to 89%. Similarly, under a moderate-impact scenario, average catch reductions ranged from 32% to 53% across all 5 models, while the largest catch reductions ranged from 75% to 88%. Model version 3 was most similar to our base model.

For the Gilbert River, a two-dam scenario (e.g. WRD1) had a more severe impact than a one-dam scenario (WRD2, WRD4) and this was seen across all models. Under Model 1 average catch was predicted to decline by 47% (WRD1) vs 36% (WRD2, WRD4) and with the largest reduction being 76% under a two-dam scenario and 68% under a one-dam scenario. Changes in predicted available biomass were similar (Table 19). The difference in predicted impacts between a one-dam and two-dam scenario was similar in some years, but greatest in years when catches were large. Although a one-dam scenario on the Gilbert River was predicted to have less impact than a Gilbert two-dam scenario and both Flinders River water extraction scenarios, it was still predicted to have substantial reductions in mud crab biomass and catch.

Mud crab catch reductions (and similarly available biomass) were largest under Model 4, and smallest under Model 2 (Figure 80), with our base model (Model 1) being somewhere in between (Figure 79, Table 19). Under a two-dam scenario, average catch reductions ranged from 38% to 54% across all 5 models (Table 19), while the largest catch reductions were predicted to be from 70% to 86%. Under a one-dam scenario, average catch reductions ranged from 28% to 43% across all five models, while the largest catch reductions were 56% to 81% (Table 19). For mud crabs, Model versions 3 and 5 (best-fit model) were most similar to Model version 1 (base model) with respect to biomass and catch reductions under the WRD scenarios (Table 19).

Water development scenarios for the Mitchell River had little to almost no impact on mud crab catch and biomass (Figure 79) and this was consistent across all model versions (Table 19). Using Model 1, a slight increase (1-2%) in catch was predicted to occur on average, and at most, catches were reduced by 1-2% e.g. under WRD16 (two-dam scenario for the Mitchell, Appendix 19). Across all model versions, average catches remained the same or slightly increased, ranging from 1-2% under a high-impact scenario e.g. WRD1 and 2-8% under a moderate-impact scenario e.g. WRD2. Model 4 appeared to be the most sensitive and although it predicted no decline in average catch, its largest reduction in catch was 15% under WRD3.

If we consider the Qld GoC region as a whole (Model Regions 1-6), on average catches were reduced by about 17-24% when considering water development on all three river systems discussed above (WRDs 1,2,4), with the lowest reduction being 40-46% (Table 19).

The effect of river flows on mud crabs is more difficult to define than for other species e.g. common banana prawns and may be compounded by other environmental variables. Mud crabs neither use freshwater riverine or palustrine habitats as juveniles or emigrate from their estuarine habitats to deep offshore waters where they are harvested as adults like common banana prawns do. However, female crabs emigrate downstream to release their eggs and some may end up offshore (e.g. Alberts-Hubatsch et al. 2016). Although positive relationships between flow and catches across GoC estuaries have been identified previously, a number of environmental parameters are correlated with catch (Robins et al. 2020). Experimentation has shown that juvenile mud crabs benefit from a brackish estuary where mortality is lower and growth is stronger (Ruscoe et al. 2004, Alberts-Hubatsch et al. 2016). Additionally, river flows may move adults downstream and thus reduce cannibalism on juvenile crabs and competition for burrows thereby enhancing survival of juvenile crabs (Robins et al. 2005). Hence, river flows to their estuarine habitats benefit the population and interrupted flows (especially wet season flows) would be expected to negatively affect biomass and catch. Adult crabs are euryhaline and tolerate a range of salinities within the estuary including near-freshwater (Meynecke et al. 2010). However, anecdotal reports from fishers have suggested both an increase in catch in the western GoC (noting peak catches in the western GoC are April to June) following a 'good wet season' in 2017; through to poor mud crab catches in the south eastern GoC in 2009, following a one-in-fifty-year flood caused the littoral habitats and inundated coasts of the gulf to become a giant shallow freshwater 'lake'. High-level flows may benefit the estuarine population via increased productivity due to nutrient loads delivered to estuarine and nearshore littoral habitats (Burford et al. 2012, 2016, Burford and Faggotter 2021). However, very large floods cause the loss of marine influence and may negatively impact inshore crab habitats in the year of the flood, though being beneficial in subsequent years due to medium-term productivity enhancement.

In the western and southern GoC, coastal flats adjacent to the river mouth are vast and shallow and because of the diurnal tidal regime, these areas can be exposed for long periods to high (low) temperatures over the summer (winter). This hot (cold) water is then moved upstream with the incoming tide. If some years have consistently high temperatures over the summer months (often as a result of no cloud cover to reduce solar radiation into the exposed coastal flats during the daytime low tide) a continual body of warm water being moved upstream may increase mud crab mortality (Robins et al. 2020). A combination of these conditions (extreme variability in rainfall with multiple hot and dry years) are most notable for the western and south-eastern GoC and less so for the eastern and northern GoC (Robins et al. 2020). Sea level anomalies may also impact mud crab abundance and catch (Robins et al. 2020), with anomalously low sea levels in 2015 being linked to mangrove die back (Duke et al 2016). Mangroves are an important habitat for mud crabs (Alberts-Hubatsch et al. 2016) and loss of habitat as well as disrupted connectivity between offshore (important for mud crab larvae) and intertidal estuarine habitats (newly settled and juvenile mud crabs) during periods of low sea level may impact mud crab abundance. While anomalous sea levels are likely to be experienced around the entire gulf, the combined effects of all these environmental variables is likely to be most notable in the western and south-eastern GoC due to the oceanographic circulation and topographic nature and layout of these rivers systems and catchments.

The stark contrast in impacts of WRD on mud crab catches in the Flinders and Gilbert Rivers, compared to the Mitchell River is likely due to the nature of the river systems (episodic vs perennial) and the way the model captured the relationships in flow and recruitment, which was reflected in biomass and ultimately catches. For example, in the Flinders and Gilbert rivers, flow and the SOI (proxy for sea level) were drivers for crab recruitment (1-year olds entering the fishery) and flow was also a driver for catchability of adult crabs, with a strong relationship estimated for the link to recruitment, and this played out in large biomass and catch reductions when flow was reduced. For the Mitchell River, we generally found little change in catch or available biomass across all models and this result is thought to be due to the perennial nature of the Mitchell River (and hence maintenance of an extensive brackish ecotone) and the relatively weak (almost no) relationship estimated between flow and mud crab populations (recruitment), as captured by our model ensemble. Indeed, we found that effort alone could explain the observed catches from this region (Figure 44) however, we nonetheless included a flow relationship, and this relationship was estimated to be very

weak (Table 10). Spatial variability across the GoC with regards to which environmental driver affected crab catch was also detected by Robins' et al. (2020) multi-variable analyses. The contrast in our results between rivers could also be that crab catchability was not impacted by flow for the Mitchell region, as per our best mud crab model (Appendix 11), and thus an additional flow-linked catchability parameter was not included for this region but was included for the Gilbert and Flinders regions. However, there were some instances when mud crab catch was predicted to increase very slightly (on average 1-2%) under WRD scenarios and this is likely due to the weak flow relationship used, where mud crab populations are hypothesised to be negatively impacted under very strong flows (as per literature) and thus if these flows are reduced somewhat under a WRD scenario to a more "optimal flow" for the Mitchell region, it could mean a slight, but negligible, increase in mud crab catch.

Although flow is thought to be the main driver of mud crab dynamics, there are also a number of other environmental variables that may drive abundance and catch (as summarised above and see Robins et al. 2020). The combination of some of these environmental variables, which will vary across regions, means that attributing any impacts to mud crab abundance (and hence catch) solely to flow could result in the model over estimating changes in biomass and catch when flows are reduced. For this reason, we also included other environmental drivers where there was a reasonable basis to do so (e.g. hypothesis driven). For example, in the Gilbert, Norman and Flinders regions, both flow and SOI were key drivers of mud crab abundance but only flow was alerted. Similarly, in the Roper region, both flow and temperature were the main drivers of mud crab dynamics (but no WRD scenarios are yet available for this region).

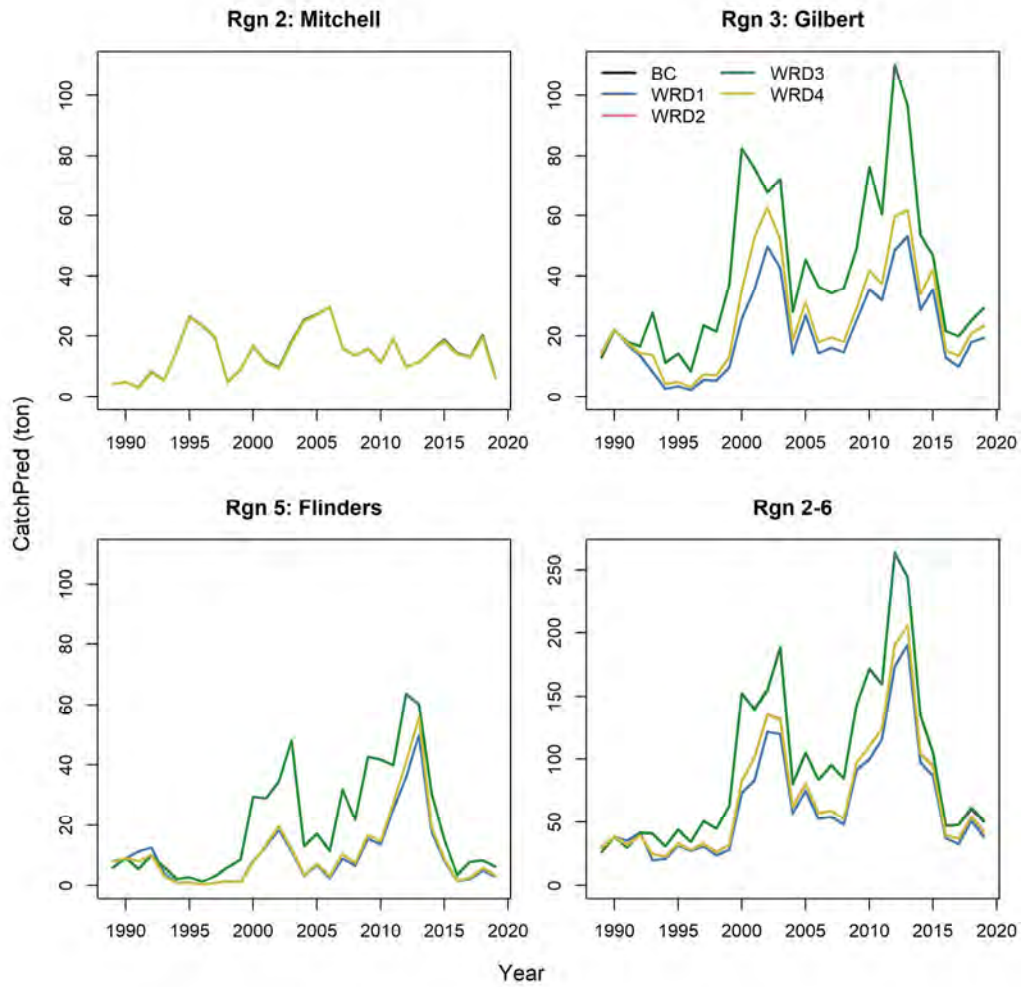


Figure 79. MICE-predicted changes to total annual mud crab catch (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 1. Note for the Mitchell River, the BC and all WRDs track under each other. For the other panels, the BC trajectory tracks under WRD3 and WRD2 tracks under WRD4.

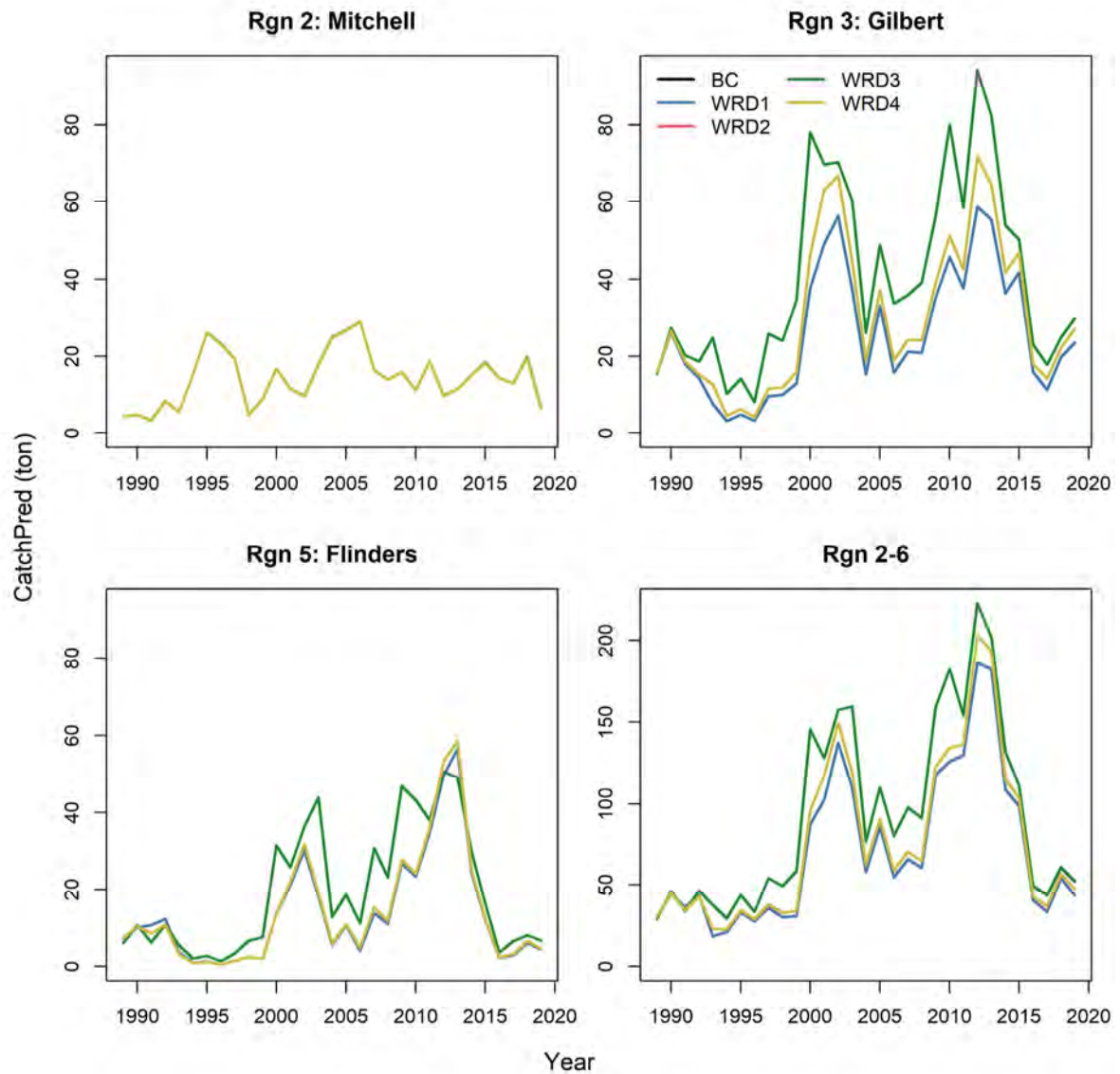


Figure 80. MICE-predicted changes to total annual mud crab catch (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 2. Note for the Mitchell River, the BC and all WRDs track under each other. For the other panels, the BC trajectory tracks under WRD3 and WRD2 tracks under WRD4.

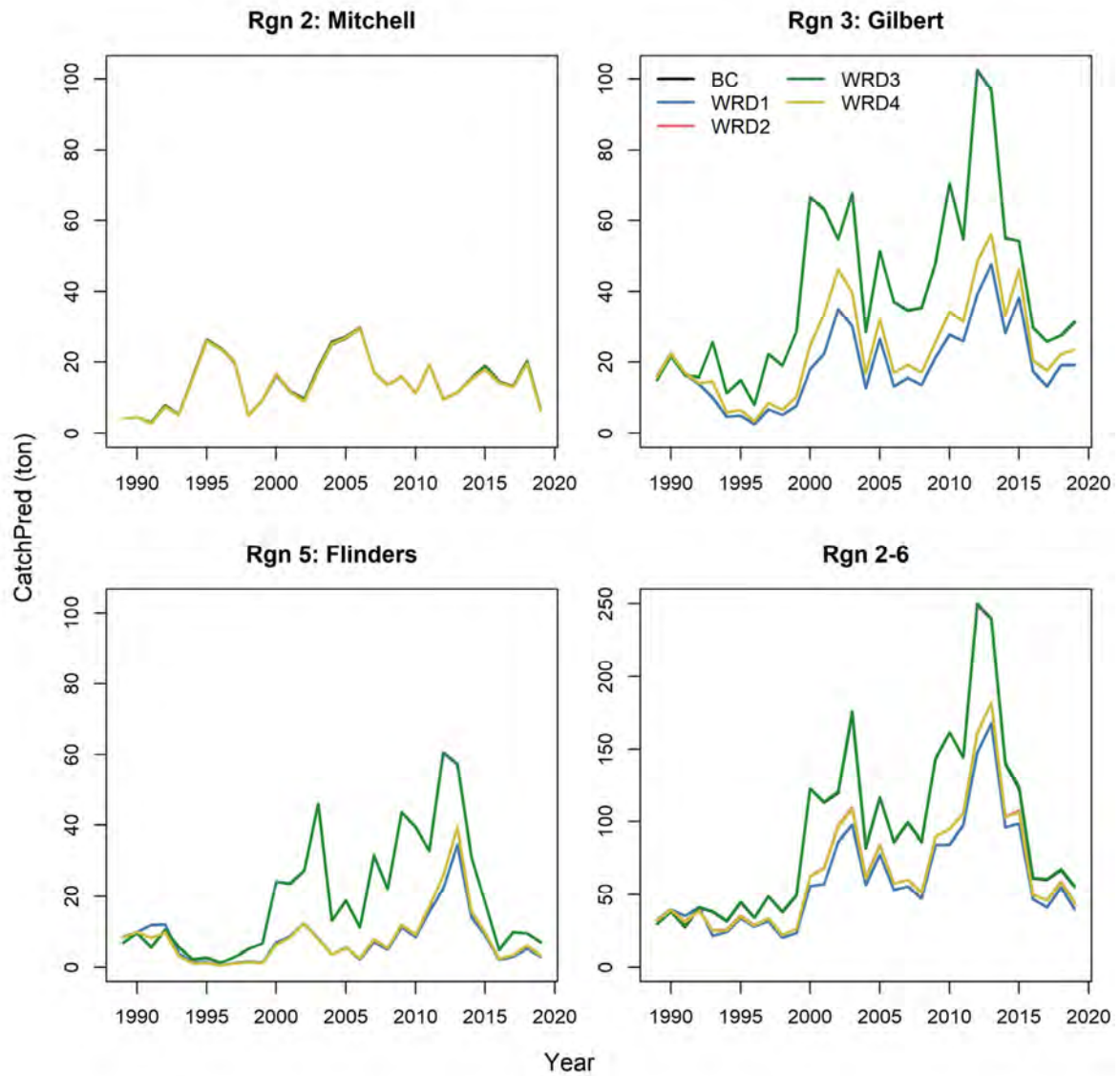


Figure 81. MICE-predicted changes to total annual mud crab catch (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 5 (best-fit model). Note for the Mitchell River, the BC and all WRDs track under each other. For the other panels, the BC trajectory tracks under WRD3 and WRD2 tracks under WRD4.

Table 19. Summary table of Model versions 1-5 showing minimum and mean catch, commercially available biomass (Bcom) and spawning biomass (Bsp) for mud crabs predicted under four water resource development scenarios (WRD1-WRD4) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions2-6 combined.

Model version 1 - mud crabs							
WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)
BaseCase	Mitchell	1	1	1	1	1	1
BaseCase	Gilbert	1	1	1	1	1	1
BaseCase	Flinders	1	1	1	1	1	1
BaseCase	Rgn 2-6	1	1	1	1	1	1
WRD1	Mitchell	0.99	1.01	0.99	1.01	0.99	1.01
WRD1	Gilbert	0.24	0.53	0.25	0.55	0.24	0.53
WRD1	Flinders	0.14	0.54	0.16	0.55	0.19	0.54
WRD1	Rgn 2-6	0.44	0.72	0.57	0.74	0.57	0.74
WRD2	Mitchell	1	1.02	1	1.02	1	1.02
WRD2	Gilbert	0.32	0.64	0.33	0.65	0.32	0.64
WRD2	Flinders	0.13	0.53	0.16	0.53	0.19	0.53
WRD2	Rgn 2-6	0.51	0.77	0.69	0.82	0.66	0.81
WRD3	Mitchell	0.99	1.01	0.99	1.01	1	1.01
WRD3	Gilbert	1	1	1	1	1	1
WRD3	Flinders	1	1	1	1	1	1
WRD3	Rgn 2-6	1	1	1	1	1	1
WRD4	Mitchell	1	1	1	1	1	1
WRD4	Gilbert	0.32	0.64	0.33	0.65	0.32	0.64
WRD4	Flinders	0.13	0.53	0.16	0.53	0.19	0.53
WRD4	Rgn 2-6	0.51	0.77	0.69	0.82	0.66	0.81
Model version 2 -mud crabs							
WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)
BaseCase	Mitchell	1	1	1	1	1	1
BaseCase	Gilbert	1	1	1	1	1	1
BaseCase	Flinders	1	1	1	1	1	1
BaseCase	Rgn 2-6	1	1	1	1	1	1
WRD1	Mitchell	1	1	1	1	1	1
WRD1	Gilbert	0.3	0.62	0.3	0.64	0.31	0.62
WRD1	Flinders	0.25	0.68	0.3	0.69	0.34	0.69
WRD1	Rgn 2-6	0.49	0.78	0.65	0.81	0.65	0.8
WRD2	Mitchell	1	1	1	1	1	1
WRD2	Gilbert	0.44	0.72	0.44	0.74	0.45	0.72
WRD2	Flinders	0.25	0.68	0.3	0.69	0.33	0.69
WRD2	Rgn 2-6	0.58	0.83	0.75	0.87	0.73	0.86
WRD3	Mitchell	1	1	1	1	1	1
WRD3	Gilbert	1	1	1	1	1	1
WRD3	Flinders	1	1	1	1	1	1
WRD3	Rgn 2-6	1	1	1	1	1	1
WRD4	Mitchell	1	1	1	1	1	1

WRD4	Gilbert	0.44	0.72	0.44	0.74	0.45	0.72
WRD4	Flinders	0.25	0.68	0.3	0.69	0.33	0.69
WRD4	Rgn 2-6	0.58	0.83	0.75	0.87	0.73	0.86

Model version 3 - mud crabs

WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)
BaseCase	Mitchell	1	1	1	1	1	1
BaseCase	Gilbert	1	1	1	1	1	1
BaseCase	Flinders	1	1	1	1	1	1
BaseCase	Rgn 2-6	1	1	1	1	1	1
WRD1	Mitchell	0.99	1.01	0.99	1.01	0.99	1.01
WRD1	Gilbert	0.27	0.55	0.27	0.57	0.26	0.54
WRD1	Flinders	0.16	0.56	0.19	0.57	0.23	0.57
WRD1	Rgn 2-6	0.46	0.73	0.59	0.75	0.58	0.75
WRD2	Mitchell	1	1.02	1	1.02	1	1.02
WRD2	Gilbert	0.34	0.66	0.36	0.67	0.34	0.65
WRD2	Flinders	0.16	0.55	0.18	0.56	0.22	0.56
WRD2	Rgn 2-6	0.52	0.78	0.7	0.83	0.68	0.82
WRD3	Mitchell	0.99	1.01	0.99	1.01	1	1.01
WRD3	Gilbert	1	1	1	1	1	1
WRD3	Flinders	1	1	1	1	1	1
WRD3	Rgn 2-6	1	1	1	1	1	1
WRD4	Mitchell	1	1	1	1	1	1
WRD4	Gilbert	0.34	0.66	0.36	0.67	0.34	0.65
WRD4	Flinders	0.16	0.55	0.18	0.56	0.22	0.56
WRD4	Rgn 2-6	0.52	0.78	0.7	0.83	0.67	0.82

Model version 4 - mud crabs

WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)
BaseCase	Mitchell	1	1	1	1	1	1
BaseCase	Gilbert	1	1	1	1	1	1
BaseCase	Flinders	1	1	1	1	1	1
BaseCase	Rgn 2-6	1	1	1	1	1	1
WRD1	Mitchell	0.77	0.95	0.79	0.96	0.83	0.95
WRD1	Gilbert	0.14	0.46	0.17	0.47	0.16	0.45
WRD1	Flinders	0.11	0.51	0.11	0.52	0.13	0.51
WRD1	Rgn 2-6	0.34	0.63	0.42	0.66	0.44	0.65
WRD2	Mitchell	0.95	1.04	0.96	1.04	0.98	1.04
WRD2	Gilbert	0.18	0.57	0.21	0.59	0.21	0.57
WRD2	Flinders	0.12	0.47	0.12	0.48	0.14	0.47
WRD2	Rgn 2-6	0.41	0.7	0.61	0.77	0.59	0.75
WRD3	Mitchell	0.8	0.95	0.82	0.95	0.85	0.96
WRD3	Gilbert	1	1	1	1	1	1
WRD3	Flinders	1	1	1	1	1	1
WRD3	Rgn 2-6	0.91	0.99	0.94	0.99	0.95	0.99
WRD4	Mitchell	1	1	1	1	1	1
WRD4	Gilbert	0.18	0.57	0.21	0.59	0.21	0.57
WRD4	Flinders	0.12	0.47	0.12	0.48	0.14	0.47

WRD4	Rgn 2-6	0.41	0.7	0.61	0.76	0.59	0.75
Model version 5 - mud crabs							
WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)
BaseCase	Mitchell	1	1	1	1	1	1
BaseCase	Gilbert	1	1	1	1	1	1
BaseCase	Flinders	1	1	1	1	1	1
BaseCase	Rgn 2-6	1	1	1	1	1	1
WRD1	Mitchell	0.98	1.02	0.98	1.02	0.98	1.02
WRD1	Gilbert	0.27	0.52	0.29	0.53	0.27	0.52
WRD1	Flinders	0.17	0.51	0.17	0.52	0.2	0.51
WRD1	Rgn 2-6	0.46	0.7	0.57	0.73	0.57	0.72
WRD2	Mitchell	1	1.03	1	1.03	1	1.03
WRD2	Gilbert	0.35	0.62	0.37	0.63	0.36	0.62
WRD2	Flinders	0.17	0.48	0.17	0.49	0.2	0.49
WRD2	Rgn 2-6	0.51	0.75	0.7	0.81	0.67	0.8
WRD3	Mitchell	0.98	1.02	0.99	1.02	0.99	1.02
WRD3	Gilbert	1	1	1	1	1	1
WRD3	Flinders	1	1	1	1	1	1
WRD3	Rgn 2-6	1	1	1	1.01	1	1.01
WRD4	Mitchell	1	1	1	1	1	1
WRD4	Gilbert	0.35	0.62	0.37	0.63	0.36	0.62
WRD4	Flinders	0.17	0.48	0.17	0.49	0.2	0.49
WRD4	Rgn 2-6	0.51	0.74	0.7	0.81	0.67	0.79
Model Ensemble:		Summary of average over ensemble					
WRD1	Mitchell	0.94	1.00	0.95	1.00	0.96	1.00
WRD1	Gilbert	0.24	0.54	0.26	0.55	0.25	0.53
WRD1	Flinders	0.17	0.56	0.19	0.57	0.22	0.56
WRD1	Rgn 2-6	0.44	0.71	0.56	0.74	0.56	0.73
WRD2	Mitchell	0.99	1.02	0.99	1.02	1.00	1.02
WRD2	Gilbert	0.33	0.64	0.34	0.65	0.34	0.64
WRD2	Flinders	0.17	0.54	0.19	0.55	0.22	0.55
WRD2	Rgn 2-6	0.51	0.77	0.69	0.82	0.66	0.81
WRD3	Mitchell	0.95	1.00	0.96	1.00	0.97	1.00
WRD3	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD3	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD3	Rgn 2-6	0.98	1.00	0.99	1.00	0.99	1.00
WRD4	Mitchell	1.00	1.00	1.00	1.00	1.00	1.00
WRD4	Gilbert	0.33	0.64	0.34	0.65	0.34	0.64
WRD4	Flinders	0.17	0.54	0.19	0.55	0.22	0.55
WRD4	Rgn 2-6	0.50	0.76	0.69	0.82	0.66	0.80

Largetooth Sawfish predicted responses under alternative WRDs

Sawfish were predicted to show high sensitivity to almost all WRDs tested except for low extraction scenarios (200GL/day, with the high TH scenario having less of an impact than the low TH scenario; WRD8-9). At the other extreme, more intensive WRDs such as WRD1 and WRD2 were predicted to result in extremely large local population declines (Table 20). The Flinders River was predicted to be most sensitive with an average decrease in abundance relative to the base simulation of around 95% (with the model predicting occasional complete local population collapse), compared with the Gilbert River system for which a 53% decline (with maximum of 67%) was predicted under both the single and double dam scenarios. WRDs simulated for the Mitchell River suggested an increasing severity of impact on sawfish with increasing extraction level – for example doubling the amount of water extracted changed the predicted average population decrease from 76% (max 98%; WRD2) to 80% (max 100%; WRD1). Changes to other WRD settings such as TH and pump rate did not result in major differences in predicted impacts on sawfish as was the case for some of the other species – rather anything other than very low extraction amounts were predicted to have a substantial negative impact on sawfish.

The predicted declines in sawfish populations are supported by recent research in the Fitzroy River, Western Australia which has shown that the recruitment and survival of largetooth sawfish to and within riverine freshwater habitats is critically dependent on large river flows. Sawfish recruitment to riverine habitats is dependent on extended periods of high-level flows (14 or more consecutive days in the 98th percentile of recorded water level) to support access to the upstream freshwater reaches of their juvenile habitats (Lear et al. 2019). Moreover, it is likely that certain rivers are stronghold nursery habitats for freshwater sawfish as some rivers support consistent and high numbers of recruits (Lear et al. 2021). Importantly, riverine pools are critical refugia for sawfish during the dry season. They lose body condition over the dry season; and following low-volume wet-season flows, the loss of body condition is greater than following high-volume wet season flows (Lear et al. 2021). The maintenance of depth and stability of river pools during the dry season is critical to the health of sawfish and the disruption to natural flows due to water impoundment or extraction has the capacity to impact their survival (Lear et al. 2020).

The Mitchell River two dam WRD scenario was predicted to result in a 41% decline in the local population, but for this species the dam scenario was predicted to have less of an effect than the water extraction scenarios. This result rests on the assumption that the planned dams would not restrict the movement of sawfish upstream and downstream but this could be further explored in future work depending on the associated planning proposed. In order to accurately evaluate the full range of impacts under various WRD scenarios, data on the distribution and movement of largetooth sawfish within each river system are required to enable an assessment of the various WRD scenarios.

Predicted changes in total sawfish numbers (i.e. when including the juvenile animals) were predicted to be much more variable than changes in mature adults numbers which are dampened to some extent because of the long time to reach maturity (Figure 51). Model results also suggested that the resilience of sawfish to negative impacts of WRDs depended critically on how depleted largetooth sawfish currently are in each catchment system (Figure 82 – Figure 83). While the level of depletion is not known, it is well accepted that globally sawfish have declined significantly (Dulvy et al. 2016). Within Australia, evidence from historical interviews and photographs indicate that sawfish have undergone significant depletions in the last 50 years. This highlights the need for accurate abundance estimates for sawfish. Improving the accuracy of demographic estimates for sawfish will also help reduce uncertainty in predictions of population responses to altered system flows. Overall, considering a range of alternative sensitivities incorporating different reproduction rates and current depletion levels, our MICE highlights that sawfish may be particularly vulnerable to negative impacts of WRDs and more work is needed to more accurately quantify the extent of predicted population impacts. Nonetheless, our preliminary investigations provide some guidance as to which river systems are likely to be most sensitive (e.g. Flinders River), the type of impacts (extraction WRDs

performed worse than dams) as well as recommended settings for future WRDs if these are to be implemented (extraction rate as low as possible with a high extraction threshold level).

A healthy population with lots of adults located offshore is able to withstand fairly high levels of natural variability in recruitment with occasional crashes, but as the mature population size decreases, the population may become less resilient and any additional reductions in flow are thus predicted to have a major or catastrophic impact on the population. Across almost all alternative model versions tested, any additional reductions in flow were predicted to have a not insubstantial effect on already depleted sawfish populations

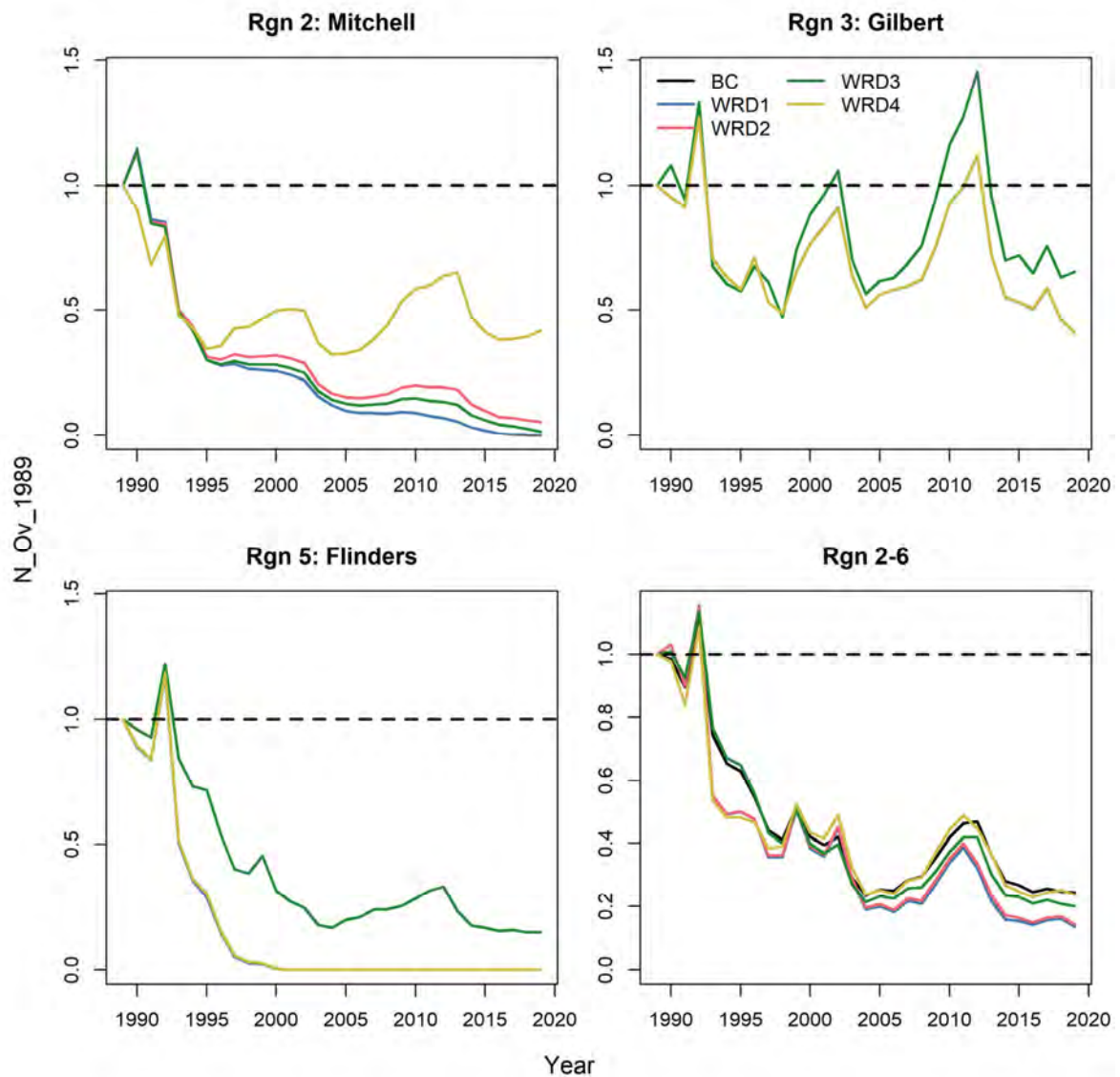


Figure 82. MICE-predicted changes to total largemouth sawfish numbers (relative to 1989 level) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 1. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4.

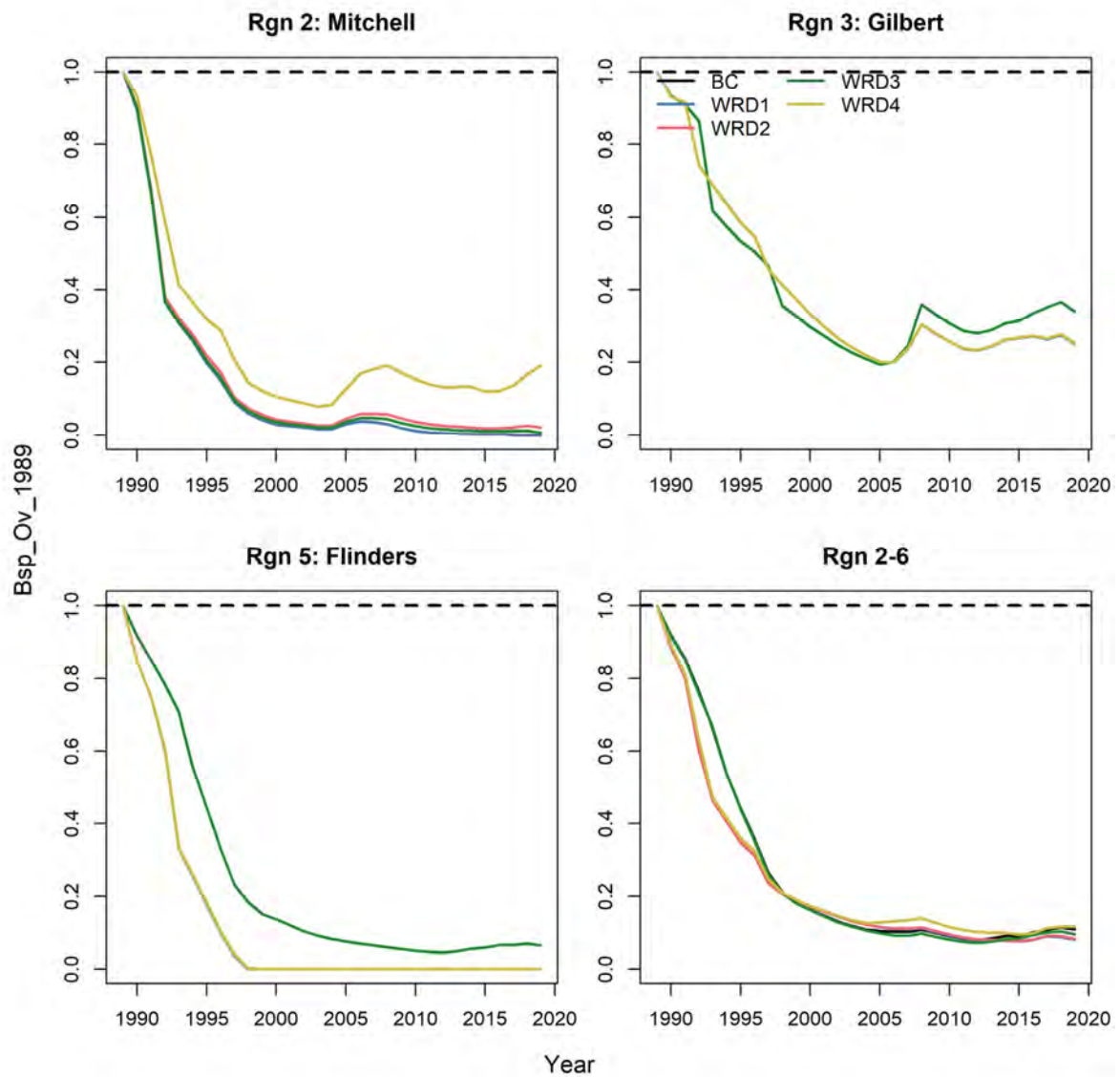


Figure 83. MICE-predicted changes to total largemouth sawfish mature adult numbers (Bsp) (relative to 1989 level) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 1. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4.

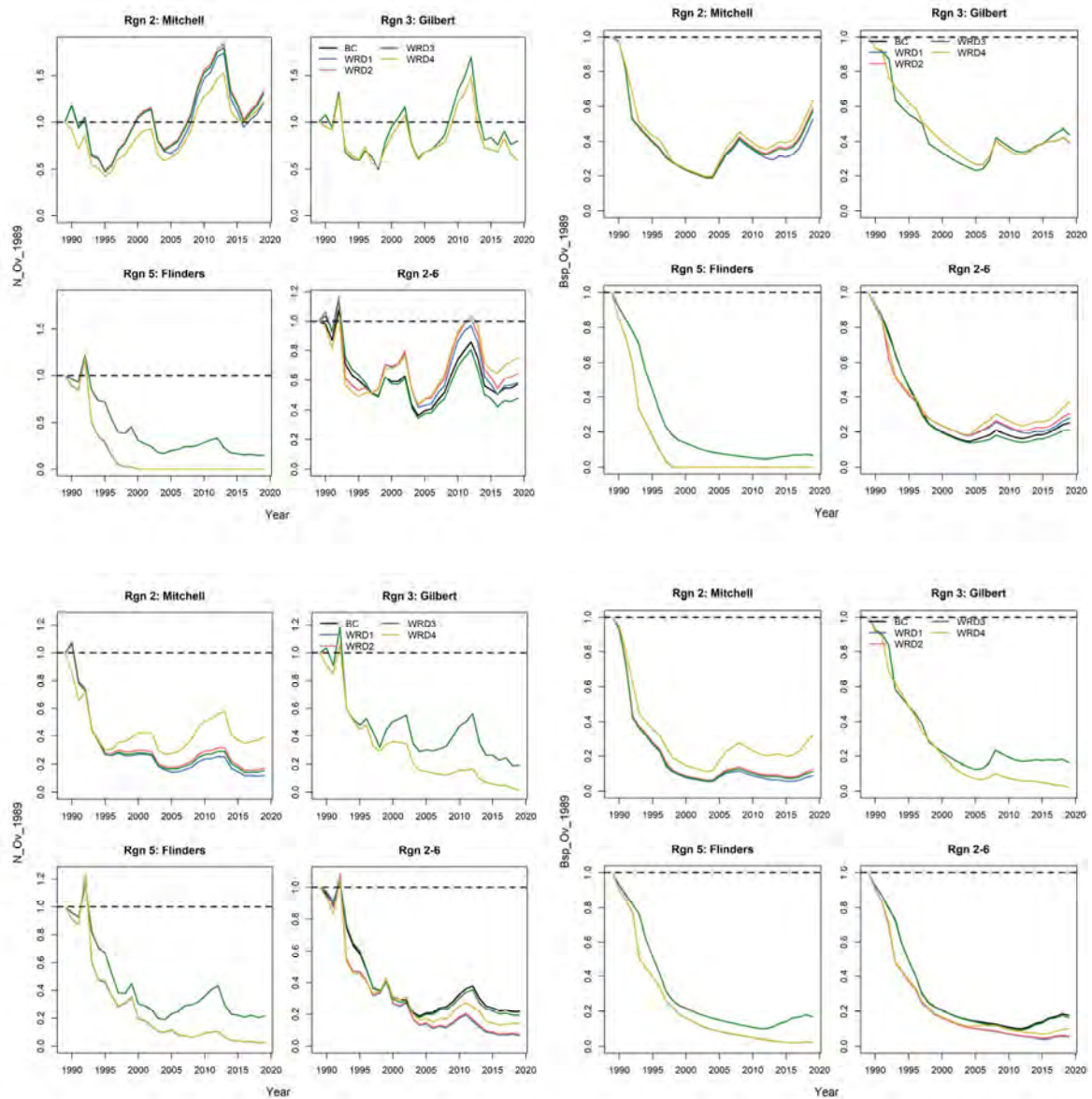


Figure 84. Comparison of predicted changes in response to WRDs when using Model version 4 (top) versus Model version 5 (bottom) incorporating different current depletion assumptions for target tooth sawfish. The left set of trajectories are total number whereas the right set are mature (Bsp) adult numbers (relative to 1989 level). WRDs scenarios are compared with the baseline flow conditions for the Mitchell, Flinders and Gilbert catchment systems. Model version 4 has baseline current depletion levels for the Mitchell, Gilbert and Flinders catchment systems of 45%, 19% and 6% compared with the following levels for Model version 5: Mitchell 17% (less optimistic), Gilbert 4%(less optimistic) and Flinders (18% (more optimistic)).

Table 20. Summary table of Model versions 1-5 showing minimum and mean number of individuals, and number of mature individuals (Bsp) for sawfish predicted under four water resource development scenarios (WRD1-WRD4) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.

Model version 1 - sawfish					
WRD Scenario	Rgn Name	Rel.N (Min)	Rel.N (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)
BaseCase	Mitchell	1	1	1	1
BaseCase	Gilbert	1	1	1	1
BaseCase	Flinders	1	1	1	1
BaseCase	Rgn 2-6	1	1	1	1
WRD1	Mitchell	0	0.2	0	0.17
WRD1	Gilbert	0.33	0.47	0.39	0.51
WRD1	Flinders	0	0.05	0	0.04
WRD1	Rgn 2-6	0.22	0.31	0.25	0.34
WRD2	Mitchell	0.06	0.28	0.07	0.25
WRD2	Gilbert	0.34	0.47	0.4	0.51
WRD2	Flinders	0	0.06	0	0.04
WRD2	Rgn 2-6	0.23	0.33	0.26	0.35
WRD3	Mitchell	0.02	0.24	0.02	0.21
WRD3	Gilbert	1	1	1	1
WRD3	Flinders	1	1	1	1
WRD3	Rgn 2-6	0.76	0.86	0.84	0.91
WRD4	Mitchell	1	1	1	1
WRD4	Gilbert	0.34	0.47	0.4	0.51
WRD4	Flinders	0	0.06	0	0.04
WRD4	Rgn 2-6	0.34	0.46	0.29	0.43

Model version 2 - sawfish					
WRD Scenario	Rgn Name	Rel.N (Min)	Rel.N (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)
BaseCase	Mitchell	1	1	1	1
BaseCase	Gilbert	1	1	1	1
BaseCase	Flinders	1	1	1	1
BaseCase	Rgn 2-6	1	1	1	1
WRD1	Mitchell	0.05	0.22	0.06	0.19
WRD1	Gilbert	0.35	0.46	0.4	0.5
WRD1	Flinders	0	0.05	0	0.04
WRD1	Rgn 2-6	0.23	0.33	0.26	0.36
WRD2	Mitchell	0.12	0.28	0.13	0.25
WRD2	Gilbert	0.36	0.46	0.4	0.5
WRD2	Flinders	0	0.05	0	0.04
WRD2	Rgn 2-6	0.25	0.34	0.27	0.36
WRD3	Mitchell	0.09	0.25	0.09	0.22
WRD3	Gilbert	1	1	1	1
WRD3	Flinders	1	1	1	1
WRD3	Rgn 2-6	0.76	0.86	0.85	0.91
WRD4	Mitchell	1	1	1	1

WRD4	Gilbert	0.36	0.46	0.4	0.5
WRD4	Flinders	0	0.05	0	0.04
WRD4	Rgn 2-6	0.34	0.47	0.3	0.45

Model version 3 - sawfish

WRD Scenario	Rgn Name	Rel.N (Min)	Rel.N (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)
BaseCase	Mitchell	1	1	1	1
BaseCase	Gilbert	1	1	1	1
BaseCase	Flinders	1	1	1	1
BaseCase	Rgn 2-6	1	1	1	1
WRD1	Mitchell	0.45	0.54	0.48	0.56
WRD1	Gilbert	0.42	0.55	0.49	0.59
WRD1	Flinders	0	0.04	0	0.04
WRD1	Rgn 2-6	0.38	0.48	0.33	0.46
WRD2	Mitchell	0.53	0.61	0.51	0.62
WRD2	Gilbert	0.42	0.55	0.5	0.59
WRD2	Flinders	0	0.04	0	0.04
WRD2	Rgn 2-6	0.39	0.51	0.34	0.49
WRD3	Mitchell	0.5	0.58	0.5	0.59
WRD3	Gilbert	1	1	1	1
WRD3	Flinders	1	1	1	1
WRD3	Rgn 2-6	0.7	0.81	0.76	0.85
WRD4	Mitchell	1	1	1	1
WRD4	Gilbert	0.42	0.55	0.5	0.59
WRD4	Flinders	0	0.04	0	0.04
WRD4	Rgn 2-6	0.48	0.69	0.41	0.64

Model version 4 - sawfish

WRD Scenario	Rgn Name	Rel.N (Min)	Rel.N (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)
BaseCase	Mitchell	1	1	1	1
BaseCase	Gilbert	1	1	1	1
BaseCase	Flinders	1	1	1	1
BaseCase	Rgn 2-6	1	1	1	1
WRD1	Mitchell	0.46	0.55	0.48	0.56
WRD1	Gilbert	0.42	0.52	0.49	0.59
WRD1	Flinders	0	0.05	0	0.04
WRD1	Rgn 2-6	0.37	0.46	0.33	0.46
WRD2	Mitchell	0.53	0.62	0.51	0.62
WRD2	Gilbert	0.42	0.53	0.5	0.59
WRD2	Flinders	0	0.06	0	0.04
WRD2	Rgn 2-6	0.38	0.5	0.34	0.49
WRD3	Mitchell	0.52	0.6	0.5	0.59
WRD3	Gilbert	1	1	1	1
WRD3	Flinders	1	1	1	1
WRD3	Rgn 2-6	0.71	0.82	0.76	0.85
WRD4	Mitchell	1	1	1	1
WRD4	Gilbert	0.42	0.53	0.5	0.59

WRD4	Flinders	0	0.06	0	0.04
WRD4	Rgn 2-6	0.47	0.67	0.41	0.64

Model version 5 - sawfish

WRD Scenario	Rgn Name	Rel.N (Min)	Rel.N (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)
BaseCase	Mitchell	1	1	1	1
BaseCase	Gilbert	1	1	1	1
BaseCase	Flinders	1	1	1	1
BaseCase	Rgn 2-6	1	1	1	1
WRD1	Mitchell	0.14	0.29	0.16	0.3
WRD1	Gilbert	0.05	0.28	0.07	0.31
WRD1	Flinders	0.04	0.16	0.04	0.17
WRD1	Rgn 2-6	0.11	0.24	0.09	0.22
WRD2	Mitchell	0.21	0.36	0.24	0.37
WRD2	Gilbert	0.05	0.28	0.07	0.31
WRD2	Flinders	0.04	0.17	0.04	0.18
WRD2	Rgn 2-6	0.12	0.25	0.1	0.23
WRD3	Mitchell	0.19	0.33	0.21	0.34
WRD3	Gilbert	1	1	1	1
WRD3	Flinders	1	1	1	1
WRD3	Rgn 2-6	0.85	0.91	0.89	0.94
WRD4	Mitchell	1	1	1	1
WRD4	Gilbert	0.05	0.28	0.07	0.31
WRD4	Flinders	0.04	0.17	0.04	0.18
WRD4	Rgn 2-6	0.25	0.33	0.16	0.28

Model Ensemble: Summary of average over ensemble

WRD1	Mitchell	0.22	0.36	0.24	0.36
WRD1	Gilbert	0.31	0.46	0.37	0.50
WRD1	Flinders	0.01	0.07	0.01	0.07
WRD1	Rgn 2-6	0.26	0.36	0.25	0.37
WRD2	Mitchell	0.29	0.43	0.29	0.42
WRD2	Gilbert	0.32	0.46	0.37	0.50
WRD2	Flinders	0.01	0.08	0.01	0.07
WRD2	Rgn 2-6	0.27	0.39	0.26	0.38
WRD3	Mitchell	0.26	0.40	0.26	0.39
WRD3	Gilbert	1.00	1.00	1.00	1.00
WRD3	Flinders	1.00	1.00	1.00	1.00
WRD3	Rgn 2-6	0.76	0.85	0.82	0.89
WRD4	Mitchell	1.00	1.00	1.00	1.00
WRD4	Gilbert	0.32	0.46	0.37	0.50
WRD4	Flinders	0.01	0.08	0.01	0.07
WRD4	Rgn 2-6	0.38	0.52	0.31	0.49

21. Sensitivity – adding additional trophic linkages

The model estimated a steep value for the interaction slope parameter $\beta_{H \cdot J}$ of 0.85 (std 0.00037) (Appendix 17, Table A30) suggesting that barramundi growth rates initially decrease gradually as the prey biomass is reduced, but then decrease substantially once prey abundance drops below around 0.3 of the reference level (see Methods Section 12 Figure 18). Adding a trophic link driven by common banana prawns resulted in a similar trend and abundance in barramundi biomass but with somewhat more variable estimates towards the end of the model years (Figure 85). The small differences between the trophic and non-trophic model versions is partly because fishing is assessed as having a bigger effect than predation in influencing population trends. However the trophic-linked model was not the best model based on the AIC scores.

As other authors have also noted, statistics derived from likelihood values to inform model selection should not be the only criterion considered when selecting models. This is because models with trophic linkages are structurally different to simpler models or may be preferred because the latter are misspecified and overfit the observations (Trijoulet et al. 2020). As highlighted by Trijoulet et al. (2020), single species models often fit the data better, but this does not necessarily indicate good model predictive performance.

The minor differences in the barramundi biomass trajectories estimated using the non-trophic linked model versus the trophic-linked model merits further investigation and validation, which is beyond the scope of this study. Future modelling may improve the representation of these predator-prey dynamics. In particular, sensitivity to the choice of the scaling parameter β could be explored as a next step. However, for current purposes, this sensitivity demonstrates that there are very minor differences (compared with the non-trophic model equivalent) in terms of predicted responses to alternative WRDs (Appendix 18, Figure 86).

Similarly adding a trophic link between largemouth sawfish and common banana prawns resulted in relatively small differences in the estimated abundance (Figure 87) and hence outcomes under alternative WRDs (Figure 88). These small differences translate into even smaller differences in the spawning biomass trajectories. As we demonstrate earlier in the text, sawfish are sensitive to any fluctuations in their numbers, and hence even though the differences between the estimated trajectories are very small, this translates into up to a 3% increase in the predicted average deletion estimates, so may be worth investigating further. There are no suitable data to inform on model selection and hence future work would need to explore a range of alternative plausible presentations of predator-prey dynamics, including the potentially increasing predation of sawfish pups by crocodiles, which are increasing in this region, with predation possibly exacerbated further through increased impediments to upstream movement thought to exacerbate predation by crocodiles.

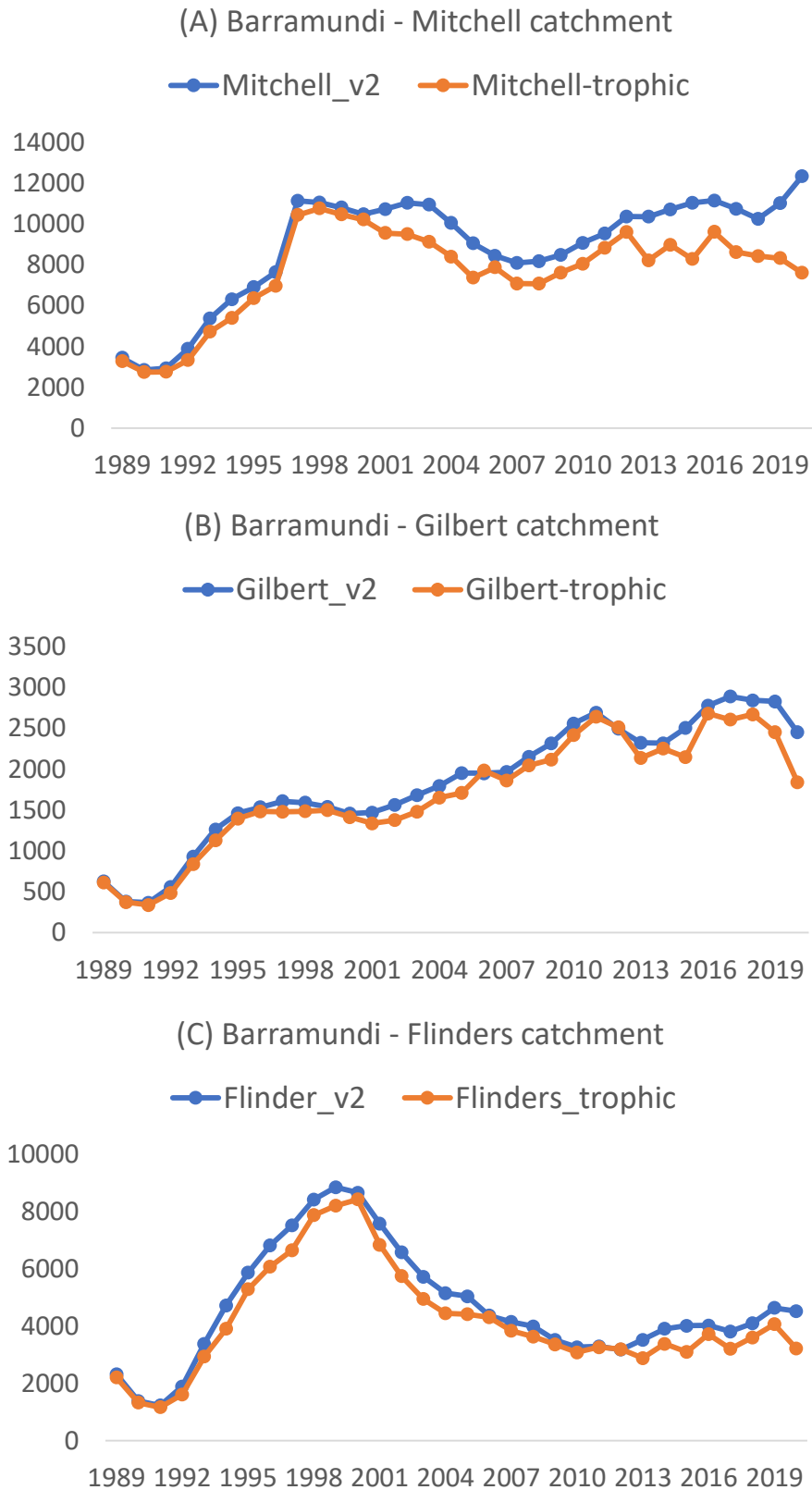


Figure 85. Comparison of the barramundi commercially available biomass (Bcomm) in the (A) Mitchell catchment, (B) Gilbert River catchment and (C) Flinders River catchment regions under baseline flow conditions to compare the model version 2 estimated trajectory with Model version 6 (trophic link added to represent barramundi preying on common banana prawns).

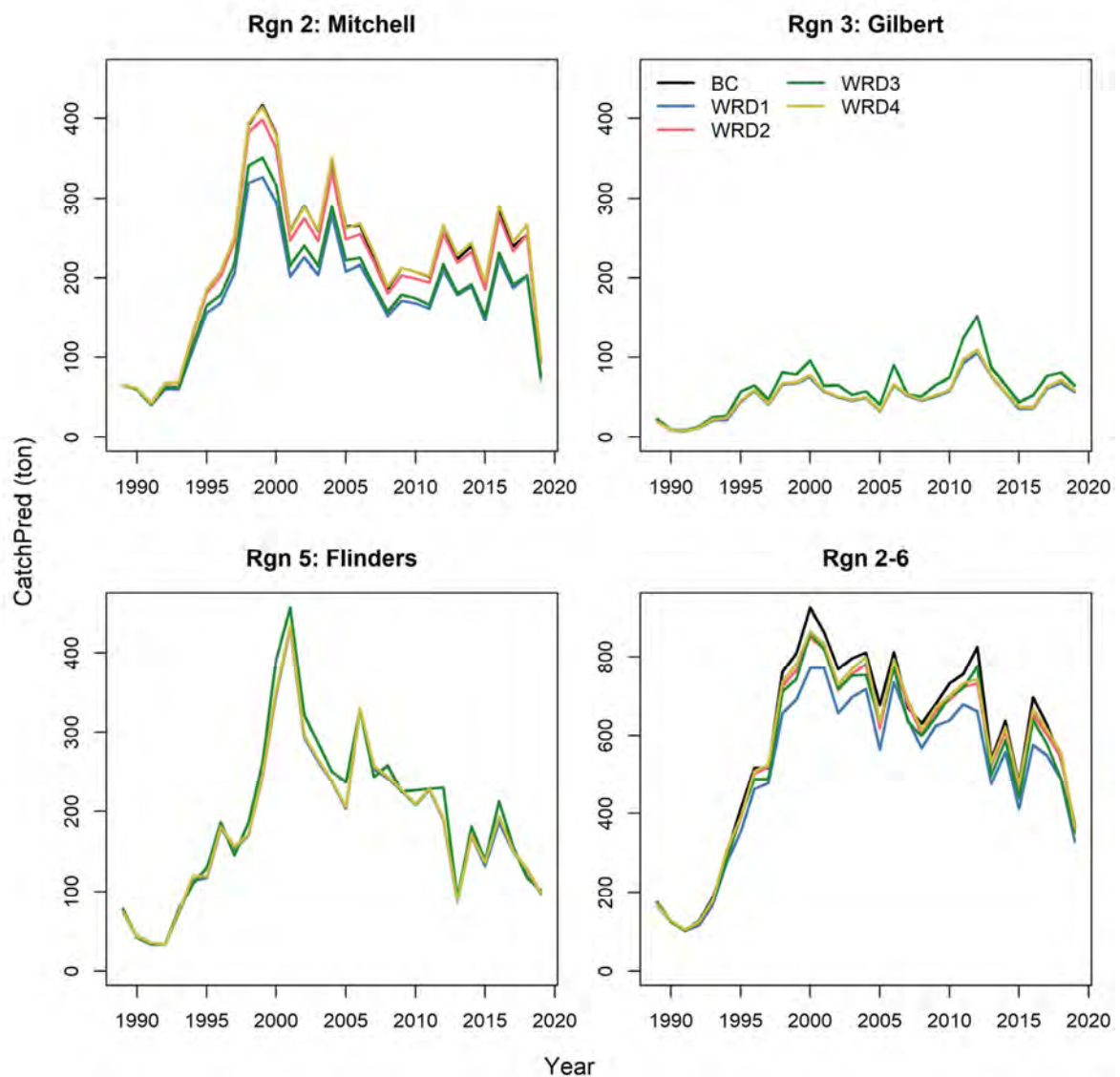


Figure 86. MICE-predicted changes to total annual barramundi catch (t) using Model version 6 (trophic link added to represent barramundi preying on common banana prawns) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 2. Note for the Mitchell River, BC trajectory tracks under WRD4 and for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4.

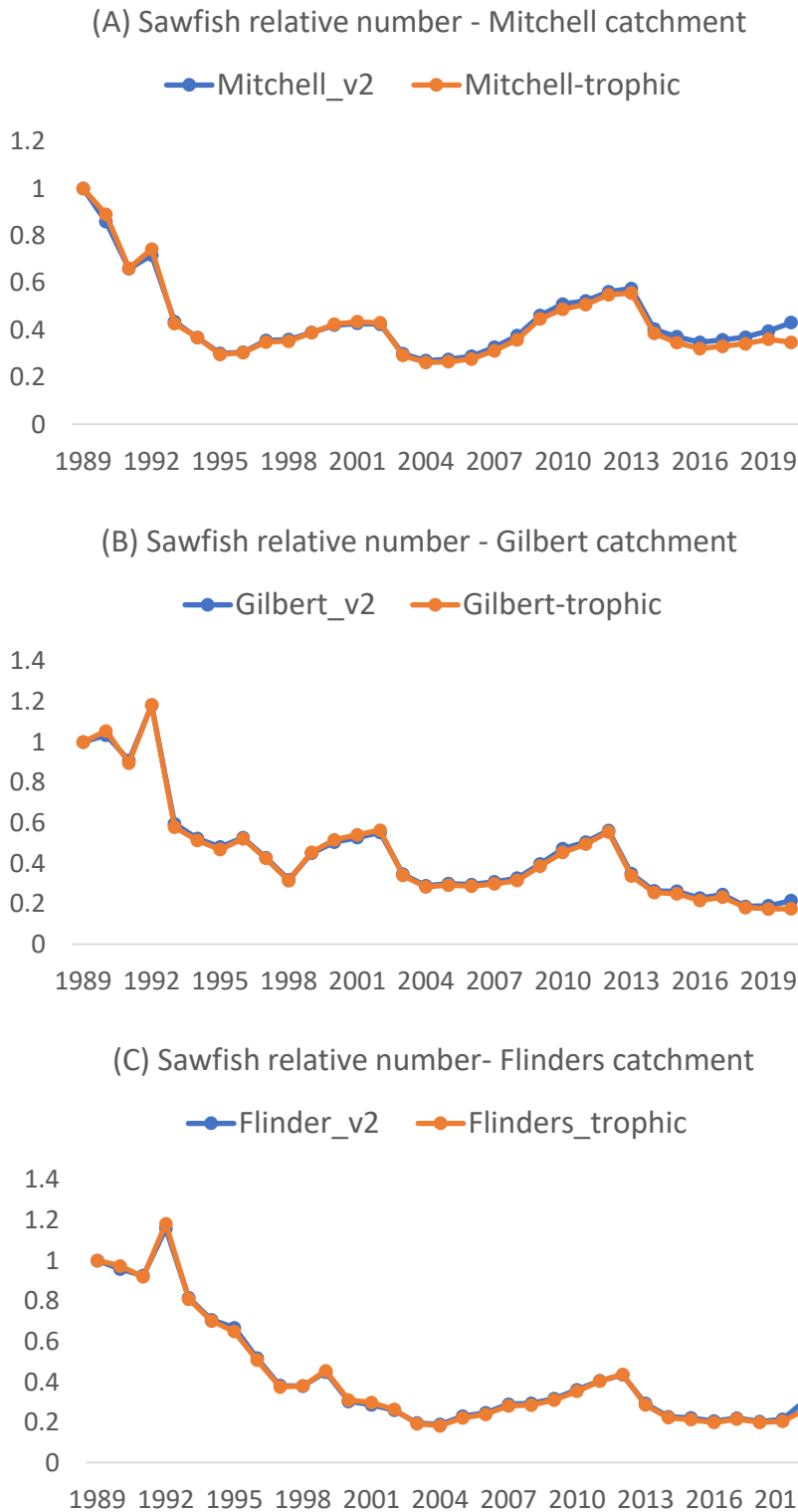


Figure 87. Comparison of the largemouth sawfish relative abundance (total number) in the (A) Mitchell catchment, (B) Gilbert River catchment and (C) Flinders River catchment regions under baseline flow conditions to compare the model version 2 estimated trajectory with Model version 6 (trophic link added to represent barramundi preying on common banana prawns).

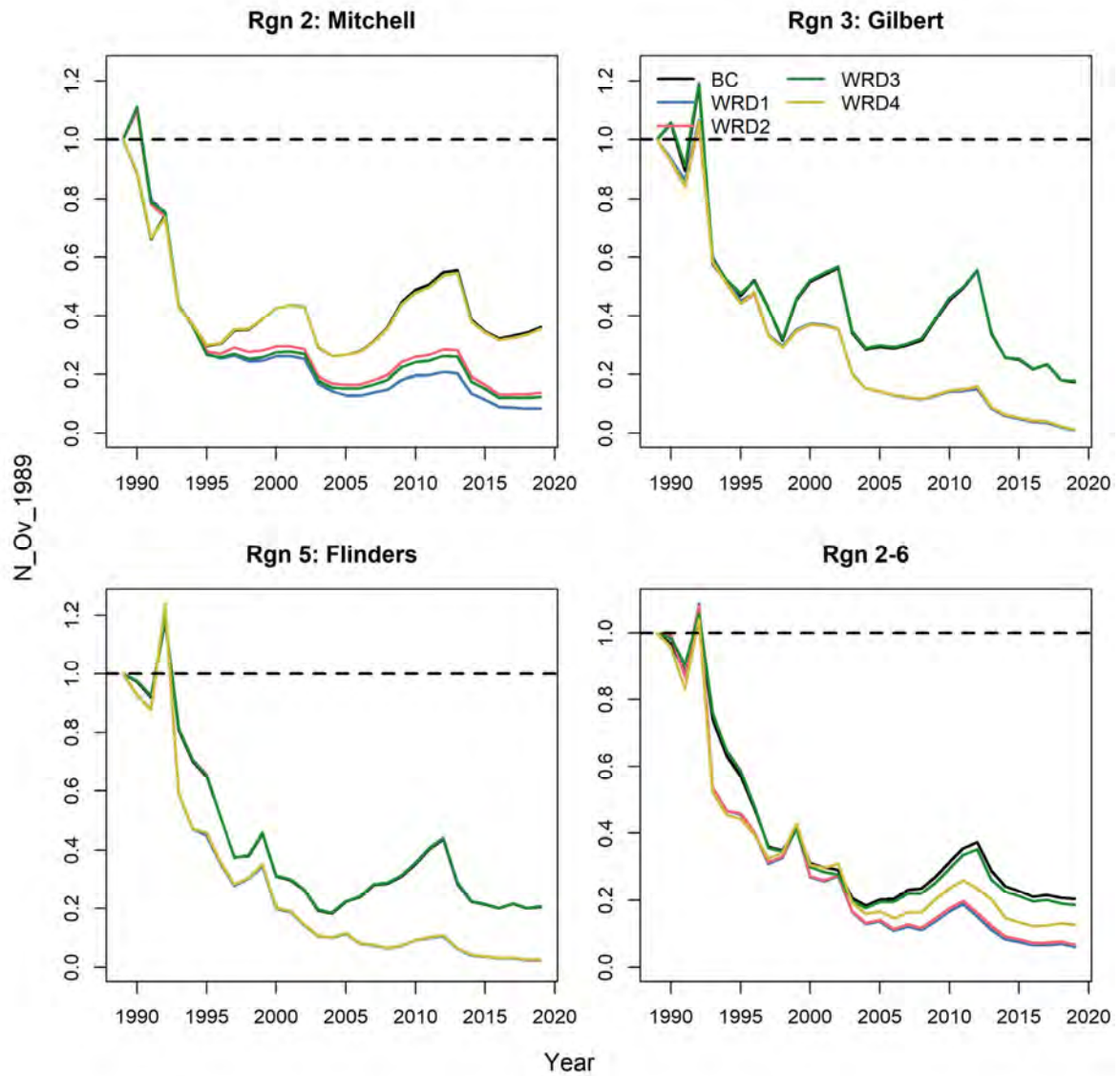


Figure 88. Comparison of predicted changes in relative abundance of largemouth sawfish in response to WRDs when using Model version 6 (additional trophic linkages added between largemouth sawfish and common banana prawns). WRDs scenarios are compared with the baseline flow conditions for the Mitchell, Flinders and Gilbert catchment systems.

22. Potential economic impacts of changes in river flow and timing due to water development on the Northern Prawn Fishery

In order to quantify potential economic impacts to the NPF due to WRD, we first had to quantify the economic contribution of common banana prawns to the NPF fishery. Here we provide estimates of total profit with common banana prawns included in the bio-economic model and estimate the break-even point in the fishery for a range of initial (start of season) biomass values for common banana prawns.

The estimated profit with common banana, tiger, and endeavour prawns is considerably greater than the estimated profit if only tiger and endeavour prawns are in the model (Figure 89) (approximately \$30 million versus approximately \$2 million), as might be expected given common banana prawns regularly account for 50% or more of total revenue. We were also able to estimate the break-even point of the fishery where total fishery profit is zero¹. This was estimated to occur when the start of season biomass of common banana prawns is approximately 1300t, with an estimated catch based on this biomass in the region of 800t (Figure 89).

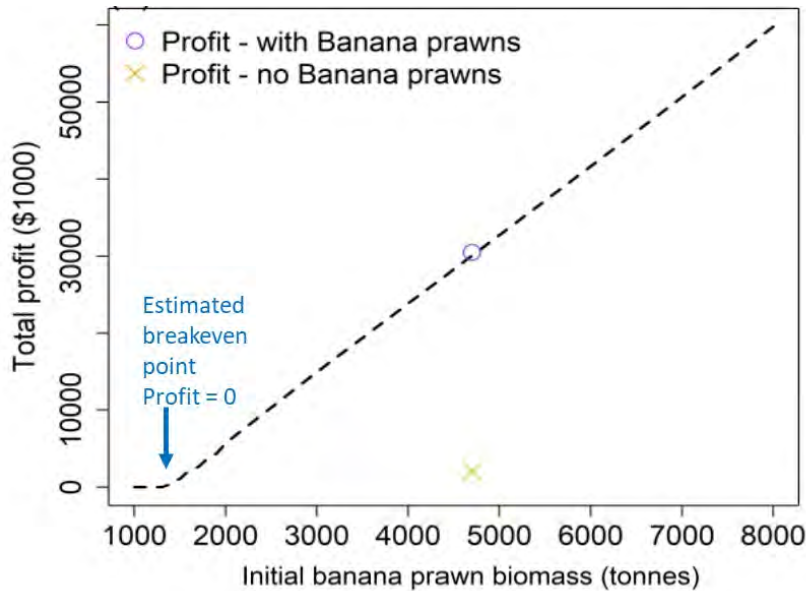


Figure 89. Estimated total fishery profit in the NPF versus the initial common banana prawn biomass at the start of the banana prawn fishery season (Figure adapted from Hutton *et al.* 2022).

These estimates are based on the 2018 bio-economic assessment model and additionally are sensitive to the input prices as well as all the costs (variable and fixed), however it provides an indication of the contribution of the common banana prawns to the total fleet profit.

The estimated longer-term impacts on the fishery of reductions in common banana prawn biomass are provided in Figure 90. These estimates are based on the assumption that there is no reduction in the size of the fleet. Reductions in banana biomass reduce the long-term profits to the fishery, for example, simulated long-term reductions in common banana prawn biomass in the region of 30% (relative to an average year) result in a 26% decline in the estimated Net Present Value (NPV) (in absolute terms a decline in NPV from \$782 million to \$582 million) (Figure 90).

¹ That is, the sum of revenues earned in the tiger prawn component of the fishery plus those of the banana prawn component equal the sum of fixed and variable costs over both components.

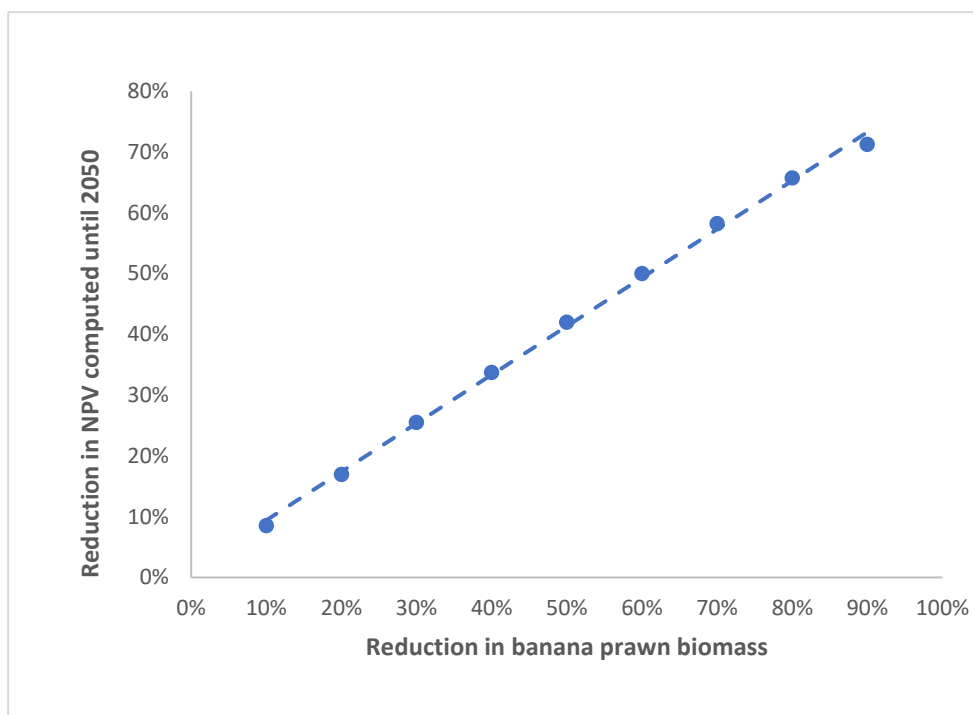


Figure 90. Predicted reductions in the Net Present Value (NPV) versus the simulated reduction in the common banana prawn biomass (based on model as specified in Hutton et al. (2022)).

Our bio-economic estimates assume there are no reductions in tiger prawn biomass, due to e.g. climate change and/or cyclones. The model was then run under each of the key four WRD scenarios and the results appear in Table 21. We provide estimates of the profit change in year 1, and the long-term profit change by estimating the NPV computed till 2050.

Table 21. The economic impacts of the main WRD scenarios (the change in profit and the change in NPV compared to the estimates for an average biomass year). Estimates are provided for the minimum (Min) and mean (Mean) biomass predicted for each WRD scenario.

Scenario	WRD1		WRD2		WRD3		WRD4	
	Min	Mean	Min	Mean	Min	Mean	Min	Mean
\$AU Million Profit Year 1	-9.4	-5.2	-7.3	-3.2	-2.1	-0.8	-6.9	-3.2
difference from average year								
(% change)	(-32.5)	(-17.8)	(-25.2)	(-10.9)	(-7.4)	(-2.9)	(-23.8)	(-10.9)
\$AU NPV Million	-155	-85	-120	-50	-36	-14.2	-113	-50
(difference from average year)								
(% change)	(-19.8)	(-10.9)	(-15.4)	(-6.4)	(-4.5)	(-1.8)	(-14.5)	(-6.4)

We found the largest change in profit under WRD1. The economic impacts for WRD3 are the least compared to the other WRD scenarios (Table 21). The average change in profit in year 1 across all the scenarios (WRD1-4) for the predicted decline in mean biomass is approximately 10.6% *ceteris paribus*. In absolute terms this is approximately an average \$3.1 million reduction in profit (across all the scenarios) in total fleet profit from about \$29.0 million to \$25.9 million.

23. Caveats and Model Limitations

Our model philosophy was to start as simple as possible and only add complexity as needed and preferably when supported by data or published literature. We therefore used the best available river models of end of system river flows under alternative WRDs as inputs to our MICE models and did not simulate additional potential negative impacts such as due to upstream developments impeding the free movement of animals upstream and downstream, as well as potential impacts from fertilisers, sediments or other factors associated with a WRD and that may alter water quality downstream. This is a particularly important caveat for seagrass because developments (either agriculture or construction activities) are known to deteriorate water quality and one of the major causes of seagrass losses globally relates to increased nutrient additions, soil erosion and sediment runoff (Waycott et al. 2009, Grech and Coles 2010). Increased sediment loads could pose multiple problems, including light attenuation associated with turbidity but also the fine sediments associated with agricultural development may be an additional risk to estuarine biota through smothering of gills, reduced viable habitat area etc. Stakeholders have shared observations that when flood waters had high sediment loads, the downstream estuary becomes devoid of biotic activity. Thus, risk to the ecosystem could be greater than predicted and in ways other than modelled by the MICE.

We also assumed in our analyses that there would be zero implementation error when implementing a WRD and ongoing compliance with any associated stipulations such as to reduce extractions in dry years. Our results may thus under-estimate the impacts of WRDs and should be interpreted as a first approximation of potential impacts, and in particular have utility in cross-comparing between different scenarios so as to inform on potential mitigation strategies.

Moreover, our analyses do not account for plausible future climate change (Fulton et al. 2018) but our framework provides an ideal platform for exploring this as part of future work. Climate projections for Australia's tropics are uncertain, some models project an increase in rainfall in northern ecosystems; whereas other models project decreasing rainfall, hence it is uncertain whether future changes will exacerbate or mitigate the predicted impacts of water extraction. But the oceans are continuing to warm even faster than originally predicted due to anthropogenic climate change (Cheng et al. 2019, Arias et al. 2021, IPCC 2021), and there is an increase in extreme temperature events (Oliver et al. 2018). Across large parts of Australia over recent years (2011-2017), extensive areas of habitat-forming coastal species have retreated due to the impacts of a warming climate: approximately half of Australia's littoral habitats have suffered, including coral reefs, mangroves, seagrass and kelp (Babcock et al. 2019). There are also additional climate factors which interact with river flows to influence estuarine and marine system dynamics, including that this region is experiencing some of Australia's most substantial changes in sea level (Church and White 2006), and future El Niño events are expected to intensify (Cai et al. 2018, Blamey et al. 2021).

Given data and capacity constraints, we focused in-depth on a few key species only as indicators of plausible system changes and our model tried to capture a representative range from vitally important commercial and recreational species, through species of conservation concern as well as the valuable mangrove and seagrass habitats. It should be noted though that there are very many other species in this system that are likely to show similar or even greater sensitivity to altered river flows. We focused on some of the most studied species in order to keep our analyses as rigorous as possible (i.e. data were available which allowed for model validation). We also focused on impacts on species that rely on both the freshwater or estuarine as well as marine component of the Gulf. We therefore did not consider any impacts on freshwater species in the modelled rivers because these have been the focus of previous studies and were beyond the scope of this study.

WRD simulations are only assumed to impact some or all of the Mitchell, Gilbert and Flinders Rivers and hence our analysis ignores potential future impacts on other catchment systems that extend throughout the GoC and Top End region where the common banana prawn fishery operates i.e. catches in these other areas and on other species or sub-fisheries of the NPF are assumed maintained. In addition, we acknowledge that averaging results over the scale of the entire GoC under-estimates

regional risks such as lower catches available in the south-eastern corner of the GoC, potentially higher travel costs for fishers using local ports such as Karumba as well as the preferred fishing locations of individual fishers. A similar situation applies to barramundi results in this project, where barramundi is a 'sub-fishery' of the inshore net fishery, with other finfish species contributing to the fishery, but WRD impacts on effort shift or other species are not explicitly modelled herein. Moreover, we only had sufficiently reliable economic information to apply an economic assessment to the common banana prawn results, but future studies could extend this to other species, or quantify the blue carbon value of mangroves.

Two of our model species – mud crabs and barramundi –also support highly important recreational and indigenous fisheries. The value of fisheries to the region's traditional owners is beyond the scope of this study to quantify. Moreover, the value of these recreational fisheries extending beyond direct economic impacts to flow-on impacts on local tourism and branding of the Gulf region's appeal as a unique remote destination. We did not have access to reliable data on recreational fishing levels but the impacts we model on commercial catches and species abundance may have a fair amount in common with likely impacts on regional recreational fisheries. The model has been set up so that recreational fishing could be added in future work

We were also unable to quantify or document the significant cultural importance of the Gulf to traditional owners and other stakeholders. Although we recognise that this is an equally or more important consideration than some of the other impacts we model, it was beyond the scope of this study to fully capture and the reader is referred to other sources (Douglas et al. 2019, Hart et al. 2019, Lyons and Barber 2021) for greater insight into this aspect. Importantly, Lyons and Barber (2021) highlight the critical need to involve Australian Indigenous peoples and include their values in water planning, but they draw attention also to the need to build capacity to strengthen communication across knowledge systems, enable knowledge co-creation and build trust across interest groups. These authors encourage consideration of relatedness and co-existence to drive new thinking about water planning processes (Lyons and Barber 2021). We had planned to extend our project consultation to visit local communities and traditional owners, but unfortunately COVID-19 impacted on our travel plans. However, future research could more explicitly link the impacts we model on species and catches to socio-cultural indices of performance. It may also be possible to draw on approaches such as the hydro-socio-ecological model developed by Douglas et al. (2019) to inform water resource development decisions.

There was a broad range associated with the uncertainty in modelling different species and hence the associated rigour of model results. This was due to both the availability of data as well as information on different species. In all cases we tried to incorporate the best available scientific information and data as well as include a number of alternative parametrisations and model structures. In the case of prawns and barramundi (and mud crabs to a lesser extent) which have previously been the focus of considerable research, it would not have been possible to model these species with as much detail and rigour as in a stock assessment and hence we tried to balance the need for rigour with a need to be pragmatic as well as cognisant of a tight timeframe and need for a model with a fast enough runtime to rapidly explore a very large number of alternative scenarios. We therefore stress that our MICE is not a stock assessment and is not intended for use in providing accurate species assessments and sustainable catch estimates, but rather its strength is its broader ecosystem focus, linking with physical drivers and use in quantifying the relative impacts of different WRDs relative to the base condition. On the other hand, our MICE is a considerably more rigorous and validated multispecies model than almost all other ecosystem models. We are therefore confident that it is currently the best available tool with an acceptable level of uncertainty, for use in quantifying regional impacts of WRDs on the GoC.

Our model results for the largemouth sawfish *P. pristis* need to be similarly considered in the context of being plausible but uncertain representations of this species given how little information is available. We used the best available information, including consultation with sawfish experts, and tested five different sawfish model structures, but do not claim that our model trajectories are an

accurate representation of the population fluctuations and declines over time. Rather we tried to capture and bound, using a set of models, the plausible range of impacts of alternative WRDs on sawfish as a first approximation to the problem. There is considerable scope to improve the representation of sawfish in the MICE as more data and information become available. Although we therefore accord less confidence than for other species to our estimates quantifying impacts of WRDs on sawfish, we nonetheless consider the range of results plausible given that there is sufficient research to date to be confident that sawfish depend critically on river flows to complete their life cycle.

We also had no data available at the scale required to model mangroves and seagrass, but have based our model on available information and used a range of parameter settings and assumptions for these groups. We used limited local-scale data from past studies as an initial step towards validating our large-scale mangrove and seagrass model based on previous observations of changes in mangrove and seagrass cover in response to changes in river flows and other factors (see Appendix 15). Our mangrove sub-component is therefore less reliable than other modelled components as is based on a mechanistic formulation (rather than an empirical analysis or statistically-validated model) in which interactions among ecological processes are represented using a mathematical model, which is then used to predict broader-scale properties of ecosystems (Rastetter et al. 2003). An advantage of using a mechanistic approach – as is used by many strategic ecosystem models (e.g. Atlantis (Fulton et al. 2011), OSMOSE (Shin et al. 2004)) - is that extensive fine-scale biological and ecological knowledge of ecosystem functioning can be represented (Rastetter et al. 2003). When drawing on available information from local studies, it is necessary to scale the model – for instance in our MICE we represent whole community mangrove productivity and abundance – plus there is a need to account for additional processes that act at the broader scale (Rastetter et al. 2003): for example, our MICE incorporates how the carrying capacity of mangroves might change in response to changes in rainfall, for which we in turn use changes in flow as a proxy. Mechanistic approaches are still the dominant means of incorporating changes in ecosystem properties in response to changes in the environment because statistical correlations are hard to define rigorously (Plagányi et al. 2019), they may break down (Myers 1998) and there are limits to how far they can be extrapolated which is necessary when the data needed to make an empirical projection do not yet exist (Rastetter et al. 2003).

We based our representation of mangrove large-scale dynamics on existing knowledge of key drivers such as solar exposure, cyclones, air temperature (which we assumed would more accurately reflect influences on mangroves than SST) and minimum sea level. We acknowledge that there is considerable variation in different mangrove species tolerances to changes in these variables (also due to differences in their spatial location) but aimed to capture large-scale average changes to the system. This set of drivers was constant across our MICE ensemble and was used to simulate realistic mangrove dynamics and consistent with the MICE approach, we accounted for some uncertainty in parametrisation of these relationships. We assumed constant nutrients in the base model given the challenges of parameterising this aspect, but future extensions could account for the likely changes in nutrient levels that may accompany WRDs (for example, due to upstream agriculture run-off). An important feature of our modelling of mangroves was therefore how we represented the influence on large-scale mangrove dynamics of changes in river flow. Here we drew on previous empirical research showing that mangroves produce less litter when subject to water and salinity stress (Conacher et al. 1996), they can be highly vulnerable to sea level variability and extreme low sea levels (Duke et al. 2017, Lovelock et al. 2017a), and variations in rainfall and sea level can have large influences on the landward and seaward extent of mangroves (Eslami-Andargoli et al. 2009, Asbridge et al. 2016, Duke et al. 2019). The amount of rainfall influences river flows and WRDs can in turn reduce flows (and for example, may dampen the influence of rainfall) and hence we used changes in river flows as a proxy for salinity stress, for the level of hydration or dehydration of mangroves (in combination with temperature), and the potential for expansion. As more data at appropriate scales become available, we may be able to better quantify the exact magnitude of fluctuations in mangrove communities that result from variability in the surrounding river flows and connection with the ocean that they depend on.

We were not highly confident of the parameter estimates or model structures we used to model primary producers, but tested a range of alternatives. We were able to successfully reproduce some past patterns and used an intermediate complexity approach to represent these components at the scale required for a very large marine ecosystem like the GoC. Due to the uncertainty in our representation of microphytobenthos and meiofauna, we only linked these components to the rest of the MICE in one model version (Model 5) and did so in a simple way, based on previous field studies (Duggan et al. 2014, Duggan et al. 2019), by assuming that microphytobenthos production is an index of system productivity in response to changes in river flow, salinity and other factors.

The only species in our MICE for which we accounted for likely connectivity effects was the common banana prawns, but we acknowledge that there may be varying degrees of connectivity between our model regions for the other species as well. Genetic analyses (Jerry R et al. 2013, Loughnan et al. 2019) provide some support for more localised populations of barramundi. Preliminary particle modelling highlights possible but limited mud crab larval exchange between some regions of the GoC (Robins et al 2020). While this is unlikely to impact results for our MICE Region 7 (as both river systems showing connectivity are within this Region), it could have implications for regions in the south-eastern GoC (MICE Regions 3-5) in that some regions could be subsidised by larvae from crabs migrating offshore from other regions.

As female largemouth sawfish are philopatric (Phillips et al. 2011, Feutry et al. 2015), we assumed population dynamics were primarily influenced by their homing river system, but we acknowledge that adults may range more widely throughout the GoC, and that fishery interactions may also not necessarily be restricted to the specific model regions we assigned them to for practical purposes. Once juveniles leave their natal river system, they may also colonise or spend time in neighbouring river systems (Rich Pillans, pers comm) and this aspect could be incorporated in future model versions if additional data become available.

The focus of this study has been on WRDs potentially impacting the Mitchell, Gilbert and Flinders Rivers and hence we focus first and foremost on these three model regions (regions 2, 3 and 5). However, we have also represented in equivalent detail the system dynamics in the other spatial regions because of the connectivity of the system, regional implications of changes to the production from one system as well as to draw on data from other regions to better inform our overall understanding of the impacts of flows on marine species and ecosystems. Our model is therefore set up to fairly rapidly test WRDs for other river systems in the near future, and in particular the Roper River for which work is ongoing as part of a separate project.

The scale, depth and amount of work associated with this study exceeded our expectations, plus was challenged to various degrees by COVID-19 impacts. We therefore consulted as widely as possible given constraints (see Appendices 3-4: summaries of project workshops) but regret that we were not able to engage with even more stakeholders as planned. In particular, the time and care needed to generate model results did not allow us to comprehensively translate these into a form that is readily implementable by water resource managers, but our team is committed to ongoing consultation to assist in translating any of our findings into the risk assessment approaches used to guide WRD decision-making. Our analyses do allow comparisons between the plausible impacts of alternative water development approaches (dams vs water extraction) as well as different rates and methods of water extraction.

24. Cross-checking model predictions of WRD impacts

Our model predictions of changes to the three fished species (prawns, barramundi and mud crab) under WRD scenarios are informed by fitting to historical data. Nonetheless, as there are a huge amount of data and information incorporated in the model, to build further confidence in our model

results, we present below some simpler comparisons to facilitate understanding of the basis for the model predictions.

First, we looked at changes in predicted catch across illustrative wet (large river flow), intermediate (intermediate river flow) and dry (little river flow) years for the baseline flow, and quantified reductions in catch between these years. For example, under baseline flow conditions, we assessed what had been the change in catch between a wet year and an intermediate year, and similarly between an intermediate year and a dry year.

Second, we looked for years in which a water development scenario (in this case we used WRD1 as an example) reduced the baseline flow from e.g. a wet year to an intermediate year, or an intermediate year to a dry year. We then quantified the model-predicted reduction in catch (Baseline vs WRD1) for this particular year to see how comparable the change was with what could be expected from wet-to-intermediate or intermediate-to-dry under baseline conditions (i.e. cyclical fluctuations).

Prawns

Baseline flows in the Gilbert River could be in excess of 100,000 m³/s over Nov-Feb in a wet year (e.g. 126,000 m³/s for Nov-Feb 2010/11) and reduced to around 5,000 m³/s for the same period in a dry year (e.g. 2014/15) (Figure 91) Model-predicted catches under baseline flow following a wet year (no lag) for example in 2011 (650t) could be reduced by 44% for an intermediate year (e.g. 2006 – 363t) and by 63% when considering intermediate to a dry year (e.g. 135t in 2015) (Figure 91A). Under WRD1, river flow in 2010/11 was reduced by more than half in most months over Nov-Feb, resulting in flows that corresponded more to an “intermediate” year. If we consider baseline vs WRD1 catch for the corresponding year, this was estimated to reduce by 15% under WRD1. Similarly, baseline river flow in an intermediate year (e.g. 2018/19) is reduced under WRD1 to a flow that could be expected in a dry year, and associated catch was predicted to decline by 14% (Figure 91). These reductions are somewhat less than what is observed in the fishery already when comparing common banana prawn catch from good, intermediate and poor flow years.

For the Flinders region, baseline flows were over 50,000 m³/s for Nov-Feb in 2010/11 (a wet year) and reduced to around 6,000 m³/s for the same period in a dry year (e.g. 2014/15) (Figure 92) Model predicted catches under baseline flow following a wet year (no lag) for example in 2011 (1622t) could be reduced by 74% for an intermediate year (e.g. 2006 – 420t) and by 32% when considering intermediate to a dry year (e.g. 285t in 2015) (Figure 92A). Under WRD1, river flow in 2010/11 was reduced by half or more in most months over Nov-Feb, resulting in flows that corresponded more to an “intermediate” year. If we consider baseline vs WRD1 catch for the corresponding year, this was reduced by 28% under WRD1. Similarly, baseline river flow in an intermediate year (e.g. 2005/06) is reduced under WRD1 to a flow that could be expected in a dry year, and associated catch was predicted to decline by 22% (Figure 92B). Again, these reductions for the Flinders region are less than what is observed in the fishery already when comparing prawn catch from good, intermediate and poor flow years. These results are not expected to match perfectly because the model also account for a range of other factors such as fishing effort, spawning biomass and fishing pressure, but they do illustrate that our model predictions are not too extreme.

Gilbert (Region 3)

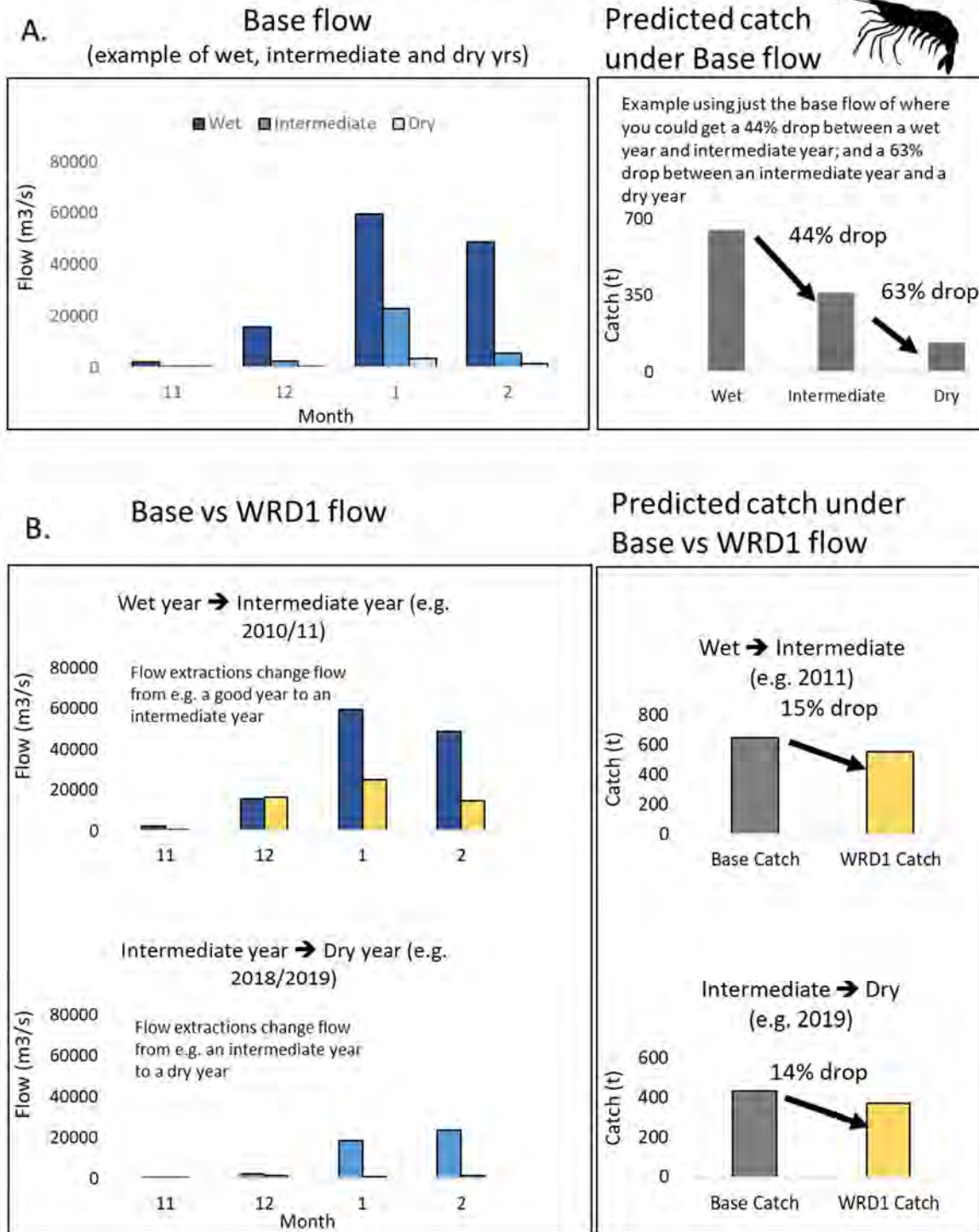


Figure 91. (A) Example of baseline flows (m³/s) for Nov-Feb in a wet, intermediate and dry year for the Gilbert catchment (Model Region 3), with corresponding common banana prawn predicted catches (t) for these years showing percentage reduction when flow reduced. (B) Baseline flows can be significantly reduced under WRD1, and corresponding catches were also reduced under WRD1, although not to the same extent under varying baseline flows.

Flinders (Region 5)

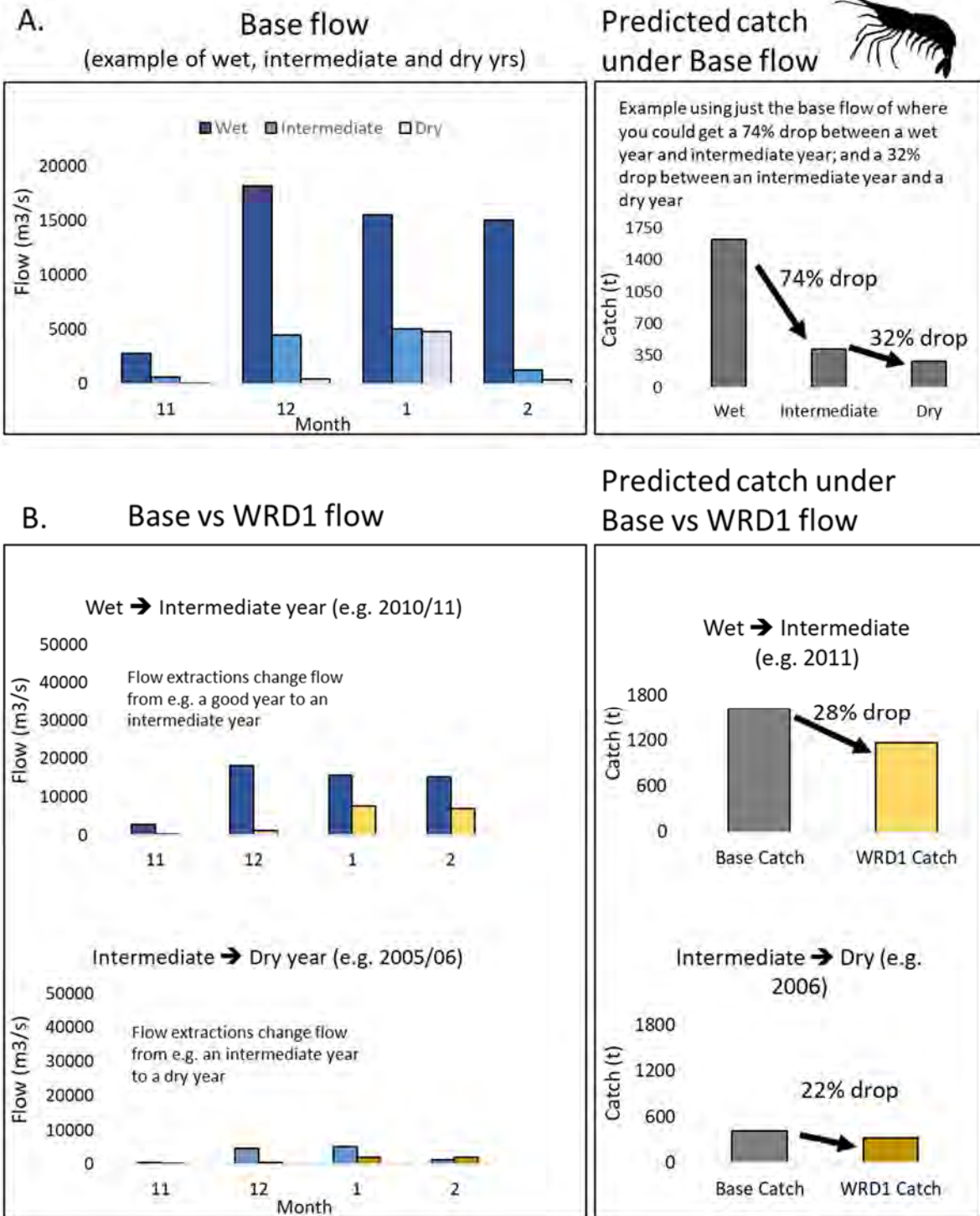


Figure 92. (A) Example of baseline flows (m³/s) for Nov-Feb in a wet, intermediate and dry year for the Flinders catchment (Model Region 5), with corresponding common banana prawn predicted catches (t) for these years showing percentage reduction when flow reduced. (B) Baseline flows can be significantly reduced under WRD1, and corresponding catches were also reduced under WRD1, although not to the same extent under varying baseline flows.

Barramundi

Comparisons for a longer-lived species like barramundi are less straightforward because there is also a lag effect before flows that impact recruitment and juvenile survival translate into a change in catch, plus effects operate on multiple age classes and hence may be more spread across years. For current purposes, we assume a 3-year lag effect between flows and catches. Baseline flows in the Gilbert region could be in excess of 100,000 m³/s over Nov-Feb in a wet year (e.g. 245,000 m³/s and 126,000 m³/s for Nov-Feb 2008/09 and 2010/11 respectively) and reduced to around 5,000 m³/s for the same period in a dry year (e.g. 2013/14 or 2014/15) (Figure 93A) Model predicted catches under baseline flow following a wet year (up to a 3-yr lag) for example in 2012 (227t) could be reduced by up to 71% for an intermediate year (e.g. 2008 – 66t) and by 48% when considering intermediate to a dry year (e.g. 35t in 2015) (Figure 93A).

Under WRD1, river flow in 2010/11 was reduced by up to half in most months over Nov-Feb, resulting in flows that corresponded more to an “intermediate” year. If we consider baseline vs WRD1 catch for the corresponding year, this was reduced by 72% under WRD1. Similarly, baseline river flow in an intermediate year (e.g. 2006/07) is reduced under WRD1 to a flow that could be expected in a dry year, and associated catch was predicted to decline by 80% (Figure 93B). Reductions in flow of a typical “wet” year are not too different from what is observed in the fishery already when comparing barramundi catch from good flow (wet) to intermediate flow years, taking into account lags between good/poor flow years and catch.

Gilbert (Region 3)

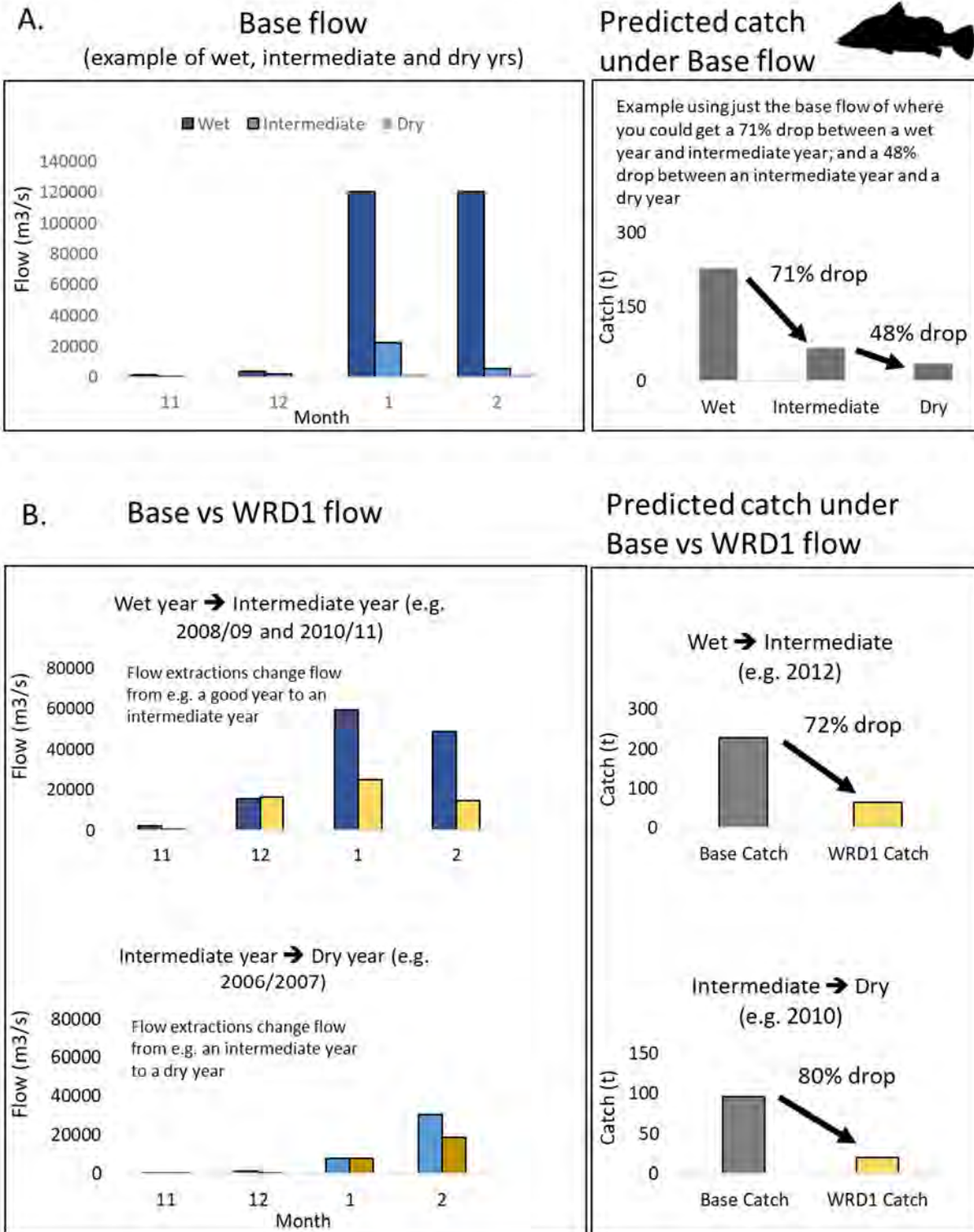


Figure 93. (A) Example of baseline flows (m³/s) for Nov-Feb in a wet, intermediate and dry year for the Gilbert catchment (Model Region 3), with corresponding barramundi predicted catches (t) for these years showing percentage reduction when flow reduced. (B) Baseline flows can be significantly reduced under WRD1, and corresponding catches were similarly reduced under WRD1.

Mud crabs

Baseline flows in the Gilbert region could be in excess of 100,000 m³/s over Nov-Feb in a wet year (e.g. 126,000 m³/s for Nov-Feb 2010/11) and reduced to around 5,000 m³/s for the same period in a dry year (e.g. 2014/15) (Figure 94A). Model predicted catches under baseline flow following a wet year (1-yr lag) for example in 2010 (76t) or 2012 (110t) could be reduced by up to 67% for an intermediate year (e.g. 2006 – 36t) and by 41% when considering intermediate to a dry year (e.g. 21t in 2016) (Figure 94A).

Under WRD1, river flow for the Gilbert River in 2010/11 was reduced by more than half in most months over Nov-Feb, resulting in flows that corresponded more to an “intermediate” flow year. If we consider baseline vs WRD1 catch for the corresponding year, this was reduced by 56% under WRD1. Similarly, baseline river flow in an intermediate year (e.g. 2018/19) is reduced under WRD1 to a flow that could be expected in a dry year, and associated catch was predicted to decline by 34% (Figure 94B). Hence, when comparing mud crab catch from good, intermediate and poor flow years, reductions under WRD1 are not too different from what is observed in the fishery already (Figure 94A).

For the Flinders region, baseline flows were over 50,000 m³/s for Nov-Feb in 2010/11 (a wet year) and reduced to around 6,000 m³/s for the same period in a dry year (e.g. 2014/15) (Figure 95A) Model predicted catches under baseline flow following a wet year (1-yr lag) for example in 2012 (64t) could be reduced by 50% for an intermediate year (e.g. 2007 – 31t) and by nearly 90% when considering intermediate to a dry year (e.g. 3t in 2016) (Figure 95A).

Under WRD1, river flow in 2010/11 was reduced by half or more in most months over Nov-Feb, resulting in flows that corresponded more to an “intermediate” year. If we consider baseline vs WRD1 catch for the corresponding year, this was reduced by 43% under WRD1. Similarly, baseline river flow in an intermediate year (e.g. 2005/06) is reduced under WRD1 to a flow that could be expected in a dry year, and associated catch was predicted to decline by 73% (Figure 95B). Again, these reductions for the Flinders region are similar to what is observed in the fishery already when comparing mud crab catch from good, intermediate and poor flow years.

Gilbert (Region 3)

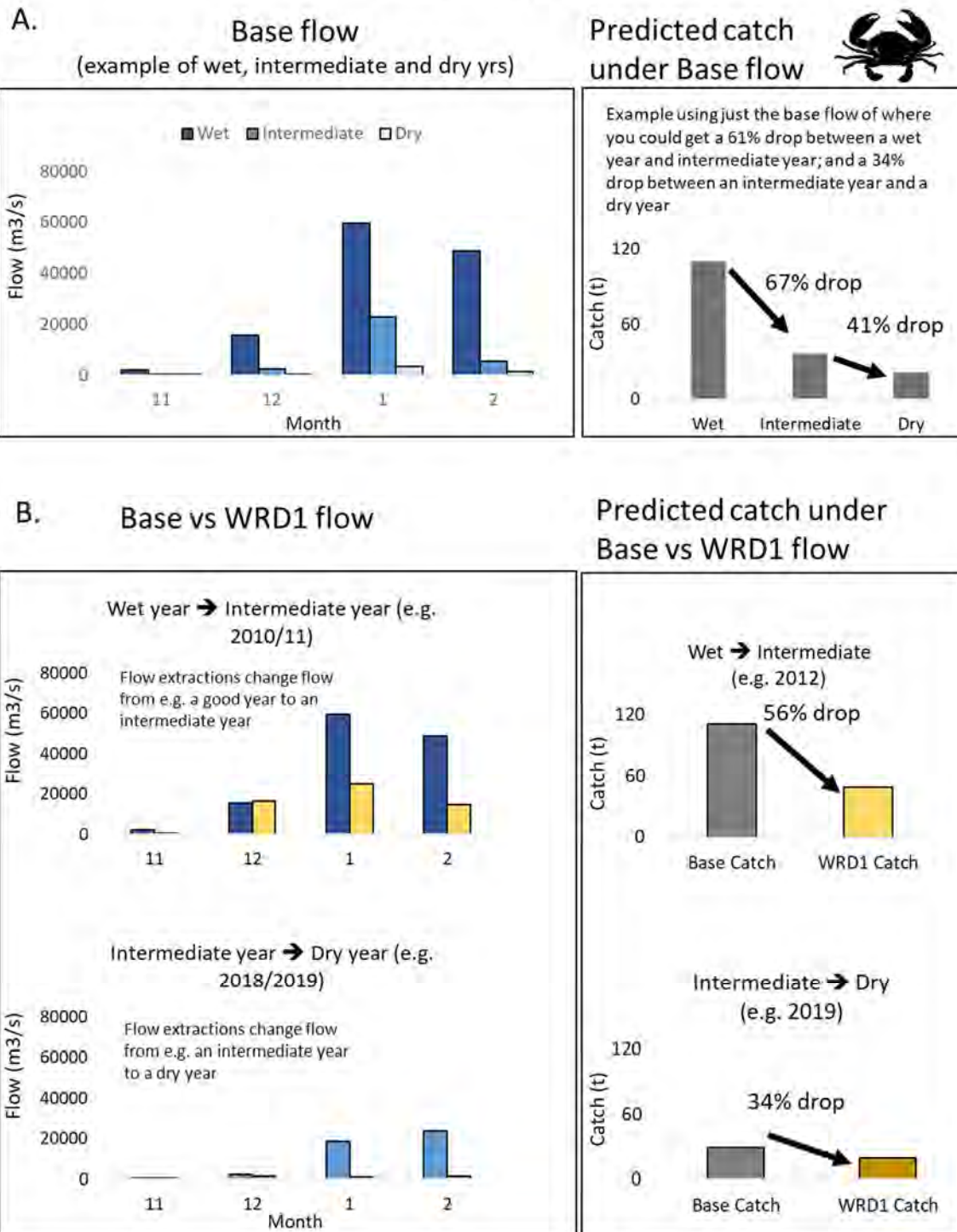


Figure 94. (A) Example of baseline flows (m3/s) for Nov-Feb in a wet, intermediate and dry year for the Gilbert catchment (Model Region 3), with corresponding mud crab predicted catches (t) for these years showing percentage reduction when flow reduced. (B) Baseline flows can be significantly reduced under WRD1, and corresponding catches were similarly reduced under WRD1.

Flinders (Region 5)

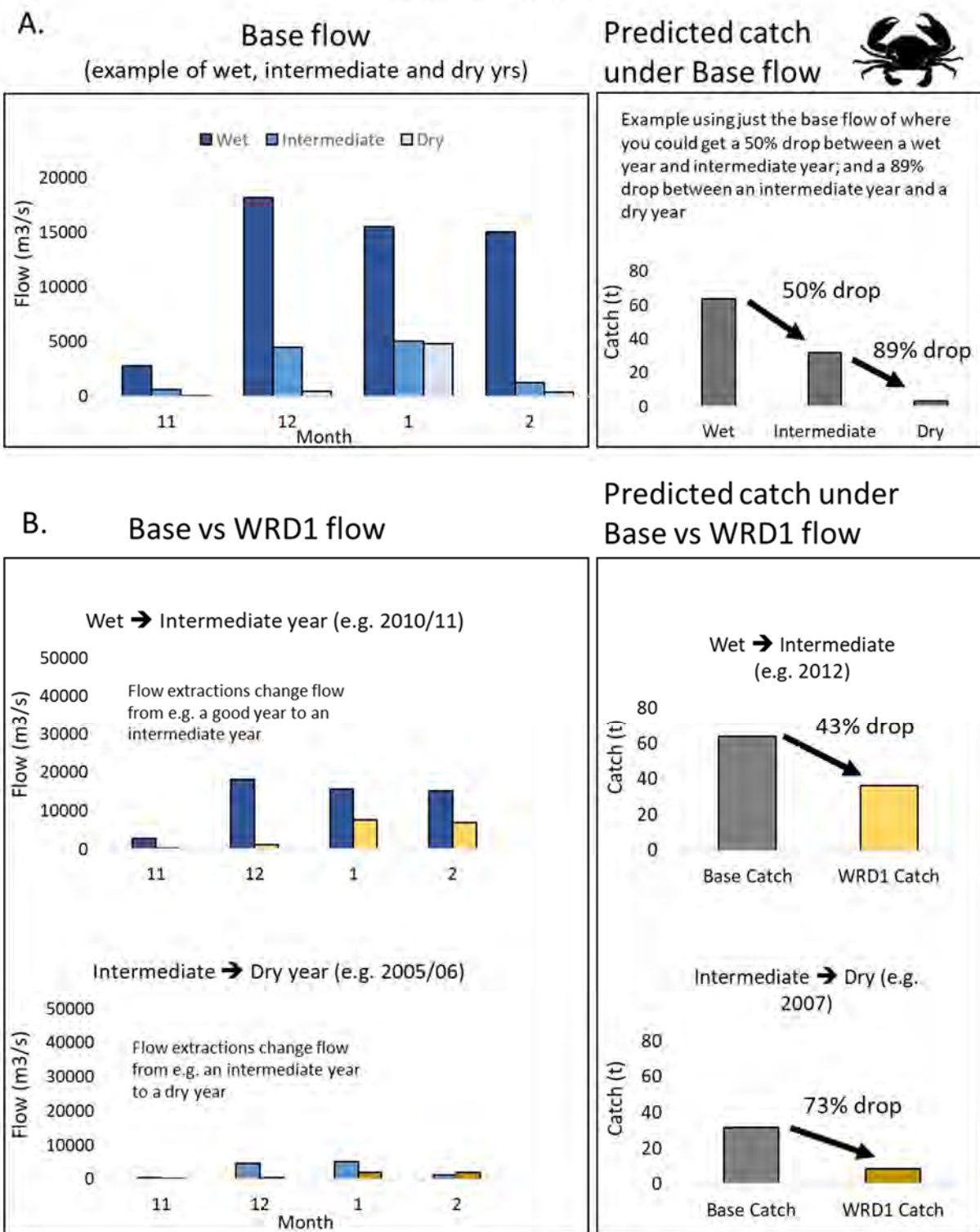


Figure 95. (A) Example of baseline flows (m³/s) for Nov-Feb in a wet, intermediate and dry year for the Flinders catchment (Model Region 5), with corresponding mud crab predicted catches (t) for these years showing percentage reduction when flow reduced. (B) Baseline flows can be significantly reduced under WRD1, and corresponding catches were similarly reduced under WRD1.

Comparisons with other studies estimating the impact of water development on the catch of a commercial marine fishery

Our study is the first fully dynamic (i.e. model changes over time and under varying conditions) model that covers the entire GoC region and is able to incorporate non-linear interactions, cumulative effects (for example, combined effects of changes in fishing effort, trophic interactions, additional environmental drivers) and account for spatial differences and overlaps in the population dynamics. Here we briefly compare our results with other available studies that have looked at catch reductions under altered river flow. To our knowledge, only Duggan et al. (2019) (applying a Bayesian belief network to the Norman River estuarine system) and Broadley et al. (2020) (using a hierarchical spatiotemporal Bayesian model applied to the Mitchell, Flinders and Gilbert systems and common banana prawn catch data over the period 1984 to 2011) have quantified this. Previous studies have documented and described the complex ways in which river flows influence the spawning success (larval survival) and inshore recruitment success of common banana prawns, as well as movements between the estuarine juvenile habitats and the offshore adult habitats (Vance et al. 1998, Vance and Rothlisberg 2020, van der Velde et al. 2021). Previous approaches have convincingly demonstrated relationships between common banana prawn catches and river flow, for example using correlation methods, general linear modelling approaches (Vance et al. 1985, Bayliss et al. 2014) and most recently, a spatiotemporal Bayesian model (with linear predictor functions) to predict how changes in prawn spatiotemporal catch vary with flows from the Mitchell, Gilbert and Flinders Rivers (Broadley et al. 2020).

Broadley et al. (2020) used three water extraction scenarios for the Flinders, Gilbert and Mitchell River catchments in their analysis. Their low and medium extraction scenarios corresponded to the following amounts of water extracted from each of these rivers respectively: low: 206;126;20 GL and medium 266; 489; 70 GL. These scenarios correspond roughly to our study's WRD2 and WRD1 for the Gilbert River (for which their medium scenario is most similar to the WRD1 which is the 2-dams scenario for the Gilbert River) and the Flinders low and medium scenarios fall between our WRD1 and WRD2 (cf our Table 22 and Table 1 in Broadley et al. (2020)). The Broadley et al. (2020) high extraction scenario (266; 489; 3425 GL for Flinders, Gilbert, Mitchell respectively) is much more extreme than our WRD5 scenario (Mitchell extraction amount 2831 GL) but slightly less so than our very high allocation scenario (WRD6: Mitchell extraction 4800 GL) – we did not include WRD5 and WRD6 in our core set of scenarios because of feedback from stakeholders that these extraction amounts were too high as would be unreliable for the Mitchell River. Nonetheless, we ran these scenarios to help bound potential impacts and also for purposes of comparison (Appendix 19).

Under the high extraction scenario, Broadley et al. (2020) found that during low-flow river conditions, the model predicted common banana prawn catches would decrease by about half or 586t. The proportional decline in catch was less for medium flow conditions (decline of around 25-30%; 371-426t) with a small (9%; 305-349t) proportional decline predicted in years of high flows (Figure 96). Our results are similar in terms of predicting variable impacts on annual catch depending on whether flow conditions are low or high – for example under WRD5 and using Model version 1, we found a 10% occurrence of catch decreases exceeding 568t, a 35% occurrence of catch decreases exceeding 371t and a 50% occurrence of catch losses exceeding 304t. Under WRD5, our model versions 1 and 5 predicted an average decrease in catch for this sub-region of 21% and 25%, with catch in some years decreasing by as much as 38%, 35% and 37% for the Mitchell, Flinders and Gilbert River catchment regions when using Model version 1 and even bigger decreases predicted using the preferred prawn model version 5: 40%, 43% and 47% respectively (Table 22). The corresponding spawning biomass was predicted by our model to decline by up to 40-52% under this scenario (Table 22).

Our results are consistent, if slightly less pessimistic, with the Broadley et al. (2020) proportional decline and reduced tonnage estimates such as those under their medium extraction scenario which suggested that during medium and high flow years catches would be reduced by 12-17%. Under our WRD1 scenario the MICE estimated regional catches would decrease by on average 16%, and by as much as 29% in some years. Our corresponding estimates under WRD2 were 10% average catch decrease down to 21% in some years (Table 22).

The two studies are not directly comparable because Broadley et al. (2020) used common banana prawn catch data for the period 1984 to 2011, whereas our study used catch data for the 50-year period 1970 to 2019. Hence Broadley et al. (2020) only used data from before the industry restructuring (Dichmont et al. 2010) which likely contributed to higher estimates of the impacts of WRDs than those in this study. There are a number of other differences also, including that Broadley et al. (2020) did not use a dynamic model to account for the underlying population dynamics, and did not have available updated end-of-system flows from river modelling conducted as part of this study. The Broadley et al. (2020) study provided useful information suggesting what the maximum common banana proportional catch loss might be under the most extreme scenarios (i.e. high proportional water allocation in poor monsoon years when flow would necessarily be low due to poor catchment runoff). Our model did not predict quite as extreme as a 50% decrease in catch (in a small number of years under a very high extraction scenario), but rather the MICE estimated maximum catch decreases of about one-third to 41% in some years and an average catch decrease of around 21% to 29% across all years under very high extraction scenario WRD5 (Table 22). Under Model version 5 and WRD5, the biggest decrease in catch was estimated for the Flinders River catchment, which estimated catch dropping to about half (47%) of that estimated under baseline flows (Table 22). Hence although the maximum MICE-predicted decrease in catch is slightly less than that of Broadley et al.'s Bayesian-based estimate, there are similarities for the predicted high-proportion catch decreases in low and medium flow years. Overall both approaches corroborate extensive previous work highlighting the critical dependence of common banana prawn abundance and catches on end-of-system river flows (Vance et al. 1985, Petheram et al. 2013b, Bayliss et al. 2014, Griffiths et al. 2014, Petheram et al. 2018b, Pollino et al. 2018, Burford and Faggotter 2021). Were the MICE to be expanded to model other catchments, the Embley River and Albatross Bay would provide a comprehensive suite of cross-life-history data to train the model and provide more precise insights on the contribution of flow modification upon each prawn stage that contributes to the eventual adult population: prawn spawning capacity, larval abundance, phytoplankton abundance, postlarval and juvenile abundance within the estuary, as well as emigration response and adult abundance in Albatross Bay (van der Velde et al. 2021).

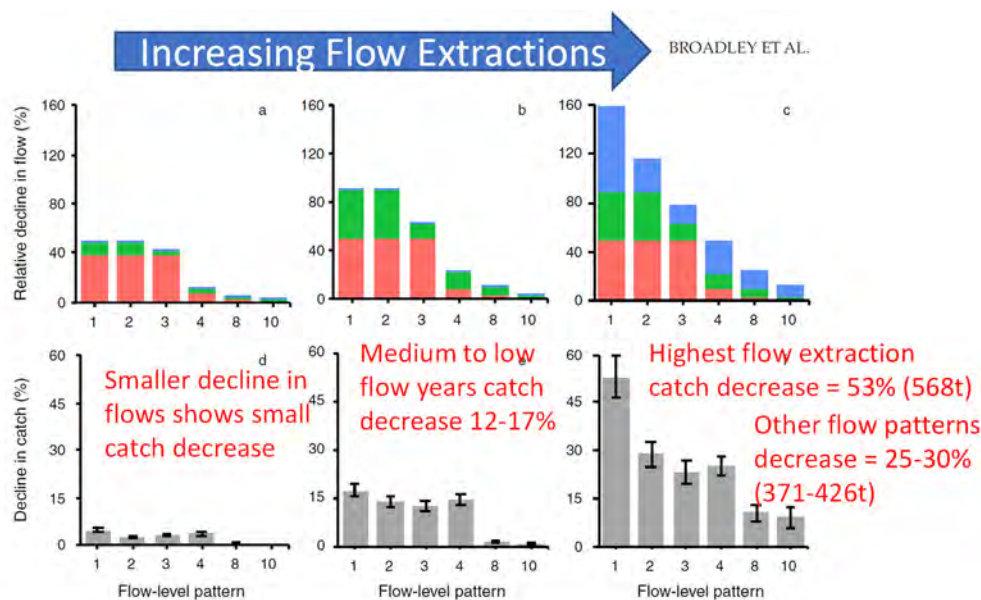


Fig. 6. The relative decline in flow as a percentage of mean end-of-system flow for the Flinders (red), Gilbert (green), and Mitchell (blue) rivers, and the predicted change in catch for flow-level patterns 1-4, 8, and 10 for water extraction scenarios: (a,d) Scenario A, flow was reduced by 206 GL, 126 GL, and 20 GL, respectively; (b,e) scenario B, flow was reduced by 266 GL, 489 GL, and 70 GL, respectively; and (c,f) scenario C, flow was reduced by 266, 489, and 3425 GL, respectively. The predicted decline in total banana prawn catch as a percentage of total catch with 95% confidence intervals (CI) is shown in (d, e, and f).

Figure 96. Extract of key results from Broadley et al. (2020) for the purpose of comparing with the MICE results, where the Broadley et al. (2020) high water extraction scenario compares most closely with the

MICE water extraction scenario WRD5 and the low and medium water extraction scenarios are more similar to MICE WRD1 and WRD2 (see text for detail).

Table 22. Summary results using MICE Model version 1 showing the maximum and average catch losses and spawning biomass (Bsp) percentage decreases for common banana prawns predicted under three water resource development scenarios (WRD1, WRD2, WRD5) relative to base line flows for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.

MICE Model version 1 – common banana prawn					
WRD Scenario	Rgn Name	Catch % decrease (Min)	Catch % decrease (Mean)	Bsp % decrease (Min)	Bsp % decrease (Mean)
WRD1	Mitchell	32%	18%	29%	18%
WRD1	Gilbert	21%	12%	23%	11%
WRD1	Flinders	37%	19%	37%	19%
WRD1	Rgn 2-6	29%	16%	24%	14%
WRD2	Mitchell	23%	11%	21%	10%
WRD2	Gilbert	7%	4%	6%	3%
WRD2	Flinders	32%	15%	33%	15%
WRD5	Mitchell	38%	25%	47%	32%
WRD5	Gilbert	35%	21%	41%	28%
WRD5	Flinders	37%	22%	40%	24%
WRD5	Rgn 2-6	33%	21%	34%	22%
MICE Model version 5 – common banana prawn					
WRD1	Mitchell	39%	25%	41%	27%
WRD1	Gilbert	37%	23%	39%	24%
WRD1	Flinders	46%	28%	45%	31%
WRD1	Rgn 2-6	39%	26%	41%	27%
WRD2	Mitchell	29%	17%	28%	17%
WRD2	Gilbert	21%	12%	20%	12%
WRD2	Flinders	40%	21%	37%	22%
WRD2	Rgn 2-6	32%	17%	27%	17%
WRD5	Mitchell	40%	29%	47%	35%
WRD5	Gilbert	43%	29%	51%	38%
WRD5	Flinders	47%	31%	52%	36%
WRD5	Rgn 2-6	41%	29%	47%	34%

Smart et al. (2021) predicted the economic impacts of reduced catches in the NPF due to water resource development scenarios. They mapped recent years, for which they could obtain economic data, to flow patterns that aligned with Broadley et al.'s (2020) scenarios. Smart et al. (2021) take a bottom-up data driven approach accounting for specific parts of the fishery under specific assumptions (e.g. no effort relocation versus effort relocation). This contrasts with the economic modelling applied in this project, which was broader in scope, less spatially specific and did not relate to any particular year.

For comparison, for the medium river flow extraction scenario (most similar in both economic analyses), Smart et al. (2021) results are an average of 10% reduction in boat business profit assuming no effort reallocation, and around 5% with effort reallocation (to areas outside of the GoC region e.g.

the Joseph Bonaparte Gulf). The bio-economic model that evaluates the MICE results is most similar to the Smart et al. (2021) “no effort relocation” given that the JBG has now been closed to fishing in the first season. Using this bio-economic model, we found for WRD2 (our medium extraction scenario) a 10.9% reduction in fleet profit for the change in mean biomass, similar to that found by Smart et al. (2021).

Although, there are similarities in our findings, there are potential differences in the accounting measures applied in each approach. It is unclear what the estimate of boat business profit includes in Smart et al.’s. (2021) analysis, and the degree to which they represent financial or economic costs, but they only looked at data post-restructuring of the fishery. Our bio-economic model includes economic depreciation (different from financial depreciation) and an opportunity cost of capital (which excludes interest payments which are not an economic cost). Despite these potential differences, the results are relatively consistent where overlapping scenarios can be identified.

25. Biological and Fishery Risk Results

Biological and Fishery Risk Results

The population and fishery risk scores for each species under alternative WRDs is shown in Table 23-Table 26 using the base-case Model version 1 as an example. There is a fair amount of consistency but also some variation in the risk scores across the full model ensemble and the full set of results are presented in Appendix 21. Comparing risk profiles across the Model versions highlights some of the uncertainties and also helps bound likely risk profiles. Common banana prawns and mud crabs showed more consistency (greater precision) between model versions than barramundi (Table 23 and Appendix 21).

Averaging results over the full ensemble and at the regional scale (model regions 2-6) provides an integrated way to evaluate and cross-compare the risks under alternative WRDs (Figure 97) while accounting for uncertainty in model structure. In general, WRD1 emerged (as expected) as the highest risk development, with moderate to intolerable risks predicted for all species and groups except for seagrass, and both in terms of population-level risk and fishery risk (Figure 97). Overall, WRD3 was predicted to pose lower risks overall compared with the other key WRD scenarios which is as expected given WRD3 simulates development applied to the Mitchell River but not Flinders or Gilbert Rivers (Figure 97). For fishery and habitat species, WRDs within multiple catchments poses the greatest risk to the marine community and species of economic and ecological importance; regardless of whether the allocation of water for extractive-use is set at a ‘moderate’ level, or double that level. Implementing WRDs within a subset of catchments has a much-reduced impact on the sustainability of the marine community and coastal ecosystem services.

When evaluating the relative risks of WRDs per catchment, we found that for prawns, the Flinders River catchment emerged as being the most sensitive to WRDs (Figure 98, Table 26). This finding is consistent with previous findings of (Burford and Faggotter 2021) who suggested that the Flinders estuary is more productive than the Gilbert and Mitchell estuaries and also the most vulnerable to excessive water development. This result is further corroborated by our model ensemble results for which Model version 5 predicted the highest relative risk for the Flinders catchment in response to water development (Appendix 21), presumably because Model version 5 accounts for the enhanced productivity effect proposed by Burford and Faggotter (2021) and Duggan et al. (2014) whereby even once the freshwater influence within the estuaries subsides, microphytobenthos and meiofauna within the sediments recover to levels equal to pre-flood conditions and remain high for months (Duggan et al. 2014, Burford and Faggotter 2021). The only model version for which the Flinders catchment didn’t rate as having relatively highest risk for prawns was Model version 2 which assumes no connectivity between the different catchments (Appendix 21).

For barramundi, the Flinders River catchment was also evaluated as having reasonably high risk to WRDs, but the Gilbert River catchment emerged as overall the most risky scenario for barramundi abundance and catches (Figure 98, Table 26). Note WRD3 assumed baseline flows for the Flinders

and Gilbert Rivers, and hence the risk is always higher for the Mitchell River catchment under this WRD, and in the case of barramundi posed higher risk than WRD2 and WRD4 (Figure 98, Table 26).

The patterns that emerged for mud crabs were once again slightly different, with the Flinders and Gilbert River catchments evaluated as roughly equally vulnerable with risks often as high as the severe risk category (Figure 98, Table 26). There was little risk for mud crabs in the Mitchell River catchment given a very weak flow relationship estimated for this region. On the other hand, risks to sawfish population abundance were almost always evaluated as severe to intolerable across WRDs, although risks tended to be relatively lower overall for the Mitchell River system (Figure 98, Table 26). When evaluating relative risks to key habitats such as mangroves, a different risk profile emerged yet again (Figure 98, Table 26). WRDs were evaluated as posing relatively higher risks for the Flinders and Gilbert systems than the Mitchell River system (Figure 98, Table 26). On the other hand, overall risks to seagrass were consistently estimated as negligible (Table 26) and hence these results aren't shown in Figure 98.

Table 23. Example of fishery and population standardised risk scores for common banana prawns, barramundi and mud crabs for the Mitchell, Flinders and Gilbert catchments as well as Regions 2-6 combined. Results are based on Model version 1. Other model versions are shown in Appendix 21. Bcom = commercially available biomass, Bsp = spawning biomass.

Model version 1 - common banana prawns											
WRD Scenario	Region Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)	Combined Risk Score	Combined Standardised Risk Score	Fishery Standardised Risk Score	Population Standardised Risk Score
WRD1	Mitchell	5	3	4	3	4	3	22	3.7	3.8	3.5
WRD1	Gilbert	4	3	3	3	4	3	20	3.3	3.3	3.5
WRD1	Flinders	5	3	5	3	5	3	24	4.0	4.0	4.0
WRD1	Rgn 2-6	5	4	4	4	4	3	24	4.0	4.3	3.5
WRD2	Mitchell	4	3	4	2	4	3	20	3.3	3.3	3.5
WRD2	Gilbert	2	2	2	2	2	2	12	2.0	2.0	2.0
WRD2	Flinders	5	3	5	3	5	3	24	4.0	4.0	4.0
WRD2	Rgn 2-6	4	3	4	2	4	2	19	3.2	3.3	3.0
WRD3	Mitchell	3	2	3	2	3	2	15	2.5	2.5	2.5
WRD3	Gilbert	3	2	3	2	3	2	15	2.5	2.5	2.5
WRD3	Flinders	2	2	2	1	2	1	10	1.7	1.8	1.5
WRD3	Rgn 2-6	3	2	2	1	2	1	11	1.8	2.0	1.5
WRD4	Mitchell	3	2	3	2	3	2	15	2.5	2.5	2.5
WRD4	Gilbert	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD4	Flinders	5	3	5	3	5	3	24	4.0	4.0	4.0
WRD4	Rgn 2-6	4	2	4	2	4	2	18	3.0	3.0	3.0
Model version 1 - barramundi											
WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)	Combined Risk Score	Standardised Risk Score	Fishery Standardised Risk Score	Population Standardised Risk Score
WRD1	Mitchell	5	3	5	3	5	3	24	4.0	4.0	4.0
WRD1	Gilbert	6	6	6	5	6	5	34	5.7	5.8	5.5
WRD1	Flinders	6	5	5	5	5	3	29	4.8	5.3	4.0

WRD1	Rgn 2-6	5	5	5	4	4	3	26	4.3	4.8	3.5
WRD2	Mitchell	2	1	2	1	2	1	9	1.5	1.5	1.5
WRD2	Gilbert	6	6	6	5	5	5	33	5.5	5.8	5.0
WRD2	Flinders	6	4	4	3	4	3	24	4.0	4.3	3.5
WRD2	Rgn 2-6	5	4	4	3	3	2	21	3.5	4.0	2.5
WRD3	Mitchell	3	3	3	3	3	3	18	3.0	3.0	3.0
WRD3	Gilbert	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD3	Flinders	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD3	Rgn 2-6	2	1	2	1	2	2	10	1.7	1.5	2.0
WRD4	Mitchell	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD4	Gilbert	6	6	6	5	5	5	33	5.5	5.8	5.0
WRD4	Flinders	6	4	4	3	4	3	24	4.0	4.3	3.5
WRD4	Rgn 2-6	5	4	4	3	2	2	20	3.3	4.0	2.0

Model version 1 - mud crabs

WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)	Combined Risk Score	Standardised Risk Score	Fishery Standardised Risk Score	Population Standardised Risk Score
WRD1	Mitchell	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD1	Gilbert	6	5	6	5	6	5	33	5.5	5.5	5.5
WRD1	Flinders	6	5	6	5	6	5	33	5.5	5.5	5.5
WRD1	Rgn 2-6	6	4	6	4	6	4	30	5.0	5.0	5.0
WRD2	Mitchell	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD2	Gilbert	6	5	6	5	6	5	33	5.5	5.5	5.5
WRD2	Flinders	6	5	6	5	6	5	33	5.5	5.5	5.5
WRD2	Rgn 2-6	6	4	5	4	5	4	28	4.7	4.8	4.5
WRD3	Mitchell	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD3	Gilbert	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD3	Flinders	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD3	Rgn 2-6	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD4	Mitchell	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD4	Gilbert	6	5	6	5	6	5	33	5.5	5.5	5.5

WRD4	Flinders	6	5	6	5	6	5	33	5.5	5.5	5.5
WRD4	Rgn 2-6	6	4	5	4	5	4	28	4.7	4.8	4.5

Table 24. Example of population standardised risk scores for sawfish for the Mitchell, Flinders and Gilbert catchments as well as Regions 2-6 combined. Results are based on Model version 1. Other model versions are shown in Appendix 21. N = total numbers, Bsp = mature numbers.

Model version 1 - sawfish							
WRD Scenario	Region Name	Rel.N (Min)	Rel.N (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)	Combined Risk Score	Population Standardised Risk Score
WRD1	Mitchell	6	6	6	6	24	6.0
WRD1	Gilbert	6	6	6	5	23	5.8
WRD1	Flinders	6	6	6	6	24	6.0
WRD1	Rgn 2-6	6	6	6	6	24	6.0
WRD2	Mitchell	6	6	6	6	24	6.0
WRD2	Gilbert	6	6	6	5	23	5.8
WRD2	Flinders	6	6	6	6	24	6.0
WRD2	Rgn 2-6	6	6	6	6	24	6.0
WRD3	Mitchell	6	6	6	6	24	6.0
WRD3	Gilbert	1	1	1	1	4	1.0
WRD3	Flinders	1	1	1	1	4	1.0
WRD3	Rgn 2-6	4	3	4	2	13	3.3
WRD4	Mitchell	1	1	1	1	4	1.0
WRD4	Gilbert	6	6	6	5	23	5.8
WRD4	Flinders	6	6	6	6	24	6.0
WRD4	Rgn 2-6	6	6	6	6	24	6.0

Table 25. Example of habitat standardised risk scores for seagrass and mangroves for the Mitchell, Flinders and Gilbert catchments as well as Regions 2-6 combined. Results are based on Model version 1. Other model versions are shown in Appendix 21.

Model version 1 - Seagrass and mangroves									
WRD Scenario	Region Name	Rel.Seagrass (Min)	Rel.Seagrass (Mean)	Combined Risk Score	Habitat Standardised Risk Score	Rel.Mangrove (Min)	Rel.Mangrove (Mean)	Combined Risk Score	Habitat Standardised Risk Score
WRD1	Mitchell	1	1	2	1.0	5	4	9	4.5
WRD1	Gilbert	1	1	2	1.0	6	5	11	5.5
WRD1	Flinders	1	1	2	1.0	6	4	10	5.0
WRD1	Rgn 2-6	1	1	2	1.0	6	4	10	5.0
WRD2	Mitchell	1	1	2	1.0	3	2	5	2.5
WRD2	Gilbert	1	1	2	1.0	6	5	11	5.5
WRD2	Flinders	1	1	2	1.0	4	4	8	4.0
WRD2	Rgn 2-6	1	1	2	1.0	4	2	6	3.0
WRD3	Mitchell	1	1	2	1.0	5	4	9	4.5
WRD3	Gilbert	1	1	2	1.0	1	1	2	1.0
WRD3	Flinders	1	1	2	1.0	1	1	2	1.0
WRD3	Rgn 2-6	1	1	2	1.0	5	3	8	4.0
WRD4	Mitchell	1	1	2	1.0	1	1	2	1.0
WRD4	Gilbert	1	1	2	1.0	6	5	11	5.5
WRD4	Flinders	1	1	2	1.0	4	4	8	4.0
WRD4	Rgn 2-6	1	1	2	1.0	2	2	4	2.0

Table 26. Fishery and population standardised risk scores for WRDs 1-4 averaged across the MICE ensemble for Regions 2-6. These results are similarly shown in Fig. 1 but with standard deviations also.

Fishery average standardised score (Regions 2-6)				
Model Ensemble Average				
Species	WRD1	WRD2	WRD3	WRD4
Common banana prawns	Major	Moderate	Negligible	Moderate
Barramundi	Major	Moderate	Negligible	Minor
Mud crabs	Severe	Major	Negligible	Major
Overall Score	Major	Moderate	Negligible	Moderate
Overall Ecosystem Score (avg of species scores)	4.6	3.8	1.5	3.7

Species	WRD1	WRD2	WRD3	WRD4
Banana prawns	4.5	3.6	1.8	3.5
Barramundi	4.0	3.0	1.6	2.8
Mud crabs	5.3	4.9	1.1	4.9

Population average standardised score (Regions 2-6)				
Model Ensemble Average				
Species	WRD1	WRD2	WRD3	WRD4
Common banana prawns	Major	Moderate	Negligible	Moderate
Barramundi	Minor	Negligible	Negligible	Negligible
Mud crabs	Severe	Major	Negligible	Major
Sawfish	Intolerable	Intolerable	Moderate	Severe
Mangroves	Severe	Minor	Moderate	Negligible
Seagrass	Negligible	Negligible	Negligible	Negligible
Overall Score	Major	Moderate	Minor	Moderate
Overall Ecosystem Score (avg of species scores)	4.1	3.3	2.0	3.1

Species	WRD1	WRD2	WRD3	WRD4
Common banana prawns	4.2	3.6	1.4	3.5
Barramundi	2.9	1.7	1.4	1.4
Mud crabs	5.2	4.6	1.1	4.6
Sawfish	6.0	6.0	3.4	5.9
Mangroves	5.2	2.7	3.9	1.9
Seagrass	1.0	1.0	1.0	1.0

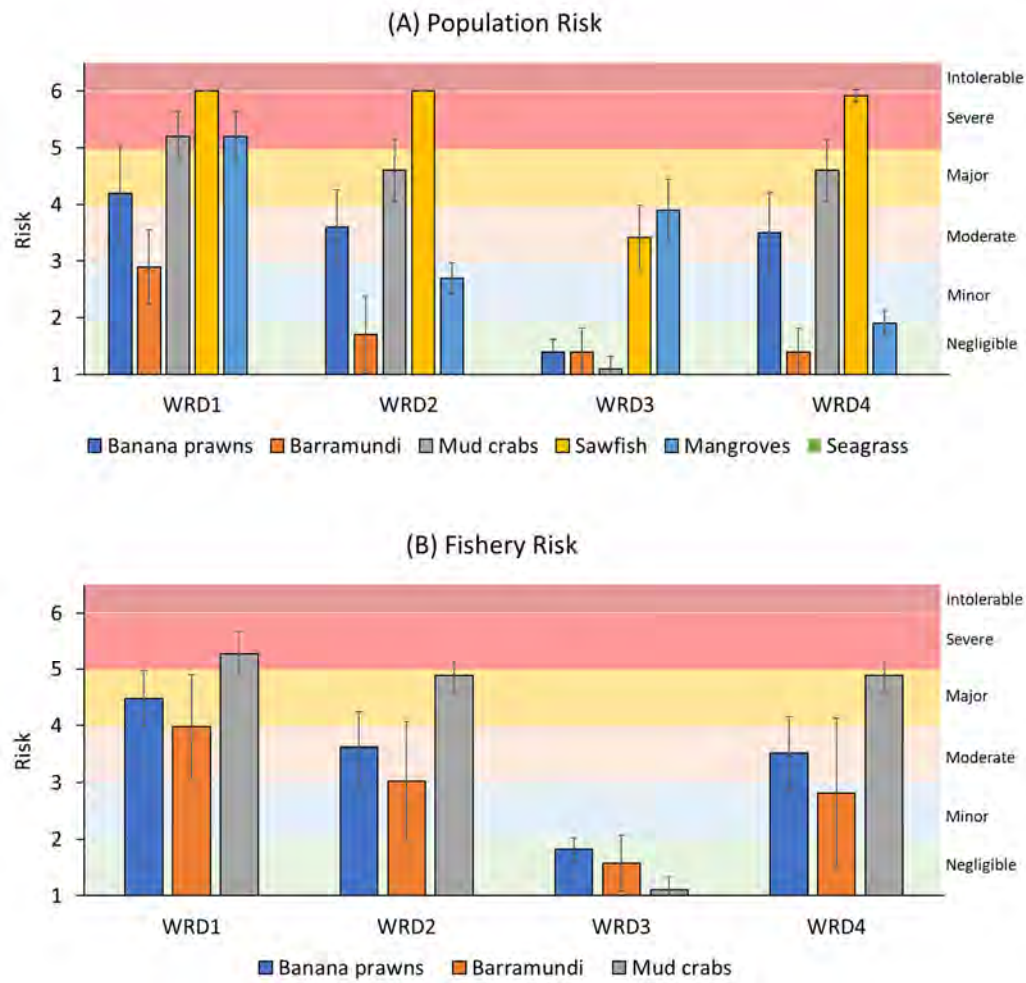


Figure 97. Comparison of average (with standard deviations) (A) Population risk and (B) Fishery Risk using the MICE ensemble to evaluate the impact of alternative WRDs on species and habitat groups as shown.

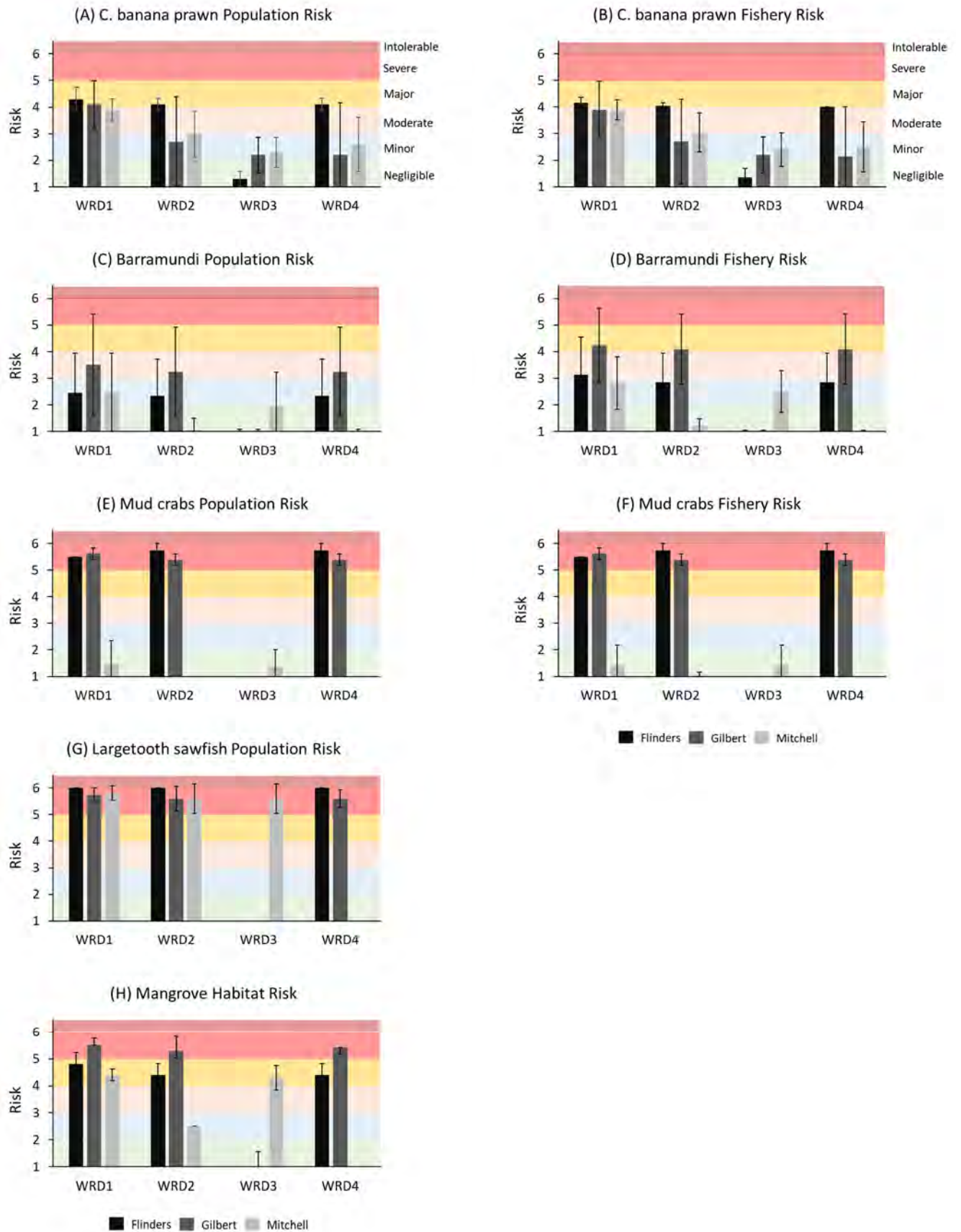


Figure 98. Comparison of average (with standard deviations) (A) Population risk and (B) Fishery Risk across three river catchments using the MICE ensemble to evaluate the impact of alternative WRDs on species and habitat groups for these catchments. C. banana prawn = common banana prawn.

Economic Risk Results

In general, we found substantial increases in economic risk under all WRDs and across the model ensemble, although there were differences in the magnitude of risk predicted under different models (Table 7). For Model version 1, we found that the risk of a bad year increased by a factor of 2.5 under WRD1 (WRD within all catchments) and more than doubled under WRDs 2 and 4 (WRD within some catchments), compared with a 50% increase under WRD3 ('moderate' WRD in the Mitchell River catchment (Fig. 3). The risk of successive bad years or fishery ruin also increased from the baseline zero risk case as WRD scenarios were implemented (Table 7, Fig 3). Similar patterns emerged when using the other model versions, but for example under Model version 5, the risks under some WRD scenarios increased substantially: for example, the risk of a bad year more than tripled under WRD1 (Table 7).

When using the alternative lower threshold catch value of 1475t, the economic risks were lower as expected, but still showed a similar pattern of doubling under WRD1 and increasing by 70% under WRD2 (Fig. 3(B)).

The average economic risks across the MICE ensemble are shown in Figs 4-5. Figure 5 compares the relative probability of occurrence of major risks (defined as risk of a bad year), severe risk (two successive bad years) and intolerable risk (fishery operations becoming unviable due to three or more consecutive bad years) predicted in response to alternative WRDs impacting the common banana prawn fishery. This highlights the probability of alternative WRD candidates increasing the baseline economic risks of different severity.

Table 27. Comparison of common banana prawn economic risks in the baseline (no-WRD) scenario and under alternative WRD scenarios, evaluated using each of the five models comprising the MICE ensemble and for all years from 1970-2019 and across the entire Gulf.

(A) Model version 1	Baseline	WRD1	WRD2	WRD3	WRD4
Risk of bad year	0.12	0.3	0.28	0.18	0.26
Risk of 2 bad years	0	0.08	0.06	0.04	0.04
Risk of fishery operations unviable	0	0.02	0.02	0.02	0.02
(B) Model version 2	Baseline	WRD1	WRD2	WRD3	WRD4
Risk of bad year	0.06	0.12	0.1	0.08	0.08
Risk of 2 bad years	0	0	0	0	0
Risk of fishery operations unviable	0	0	0	0	0
(C) Model version 3	Baseline	WRD1	WRD2	WRD3	WRD4
Risk of bad year	0.08	0.12	0.12	0.1	0.12
Risk of 2 bad years	0	0	0	0	0
Risk of fishery operations unviable	0	0	0	0	0
(D) Model version 4	Baseline	WRD1	WRD2	WRD3	WRD4
Risk of bad year	0.08	0.1	0.1	0.08	0.1
Risk of 2 bad years	0	0	0	0	0
Risk of fishery operations unviable	0	0	0	0	0
(E) Model version 5	Baseline	WRD1	WRD2	WRD3	WRD4
Risk of bad year	0.14	0.46	0.3	0.16	0.28
Risk of 2 bad years	0.02	0.16	0.08	0.02	0.04
Risk of fishery operations unviable	0	0.06	0.02	0	0.02
Average risk	Baseline	WRD1	WRD2	WRD3	WRD4
Risk of bad year	0.096	0.22	0.18	0.12	0.168
Risk of 2 bad years	0.004	0.048	0.028	0.012	0.016
Risk of fishery operations unviable	0	0.016	0.008	0.004	0.008

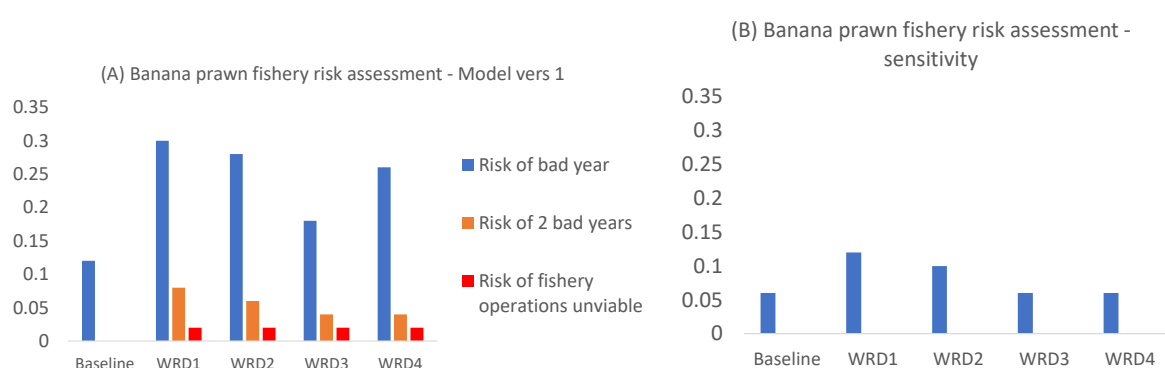


Figure 99. Comparison of Economic Risk when using Model version 1 to evaluate the impact of alternative WRDs on common banana prawns, and when using as the threshold risk annual catch level (A) 2000t versus (B) 1475t.

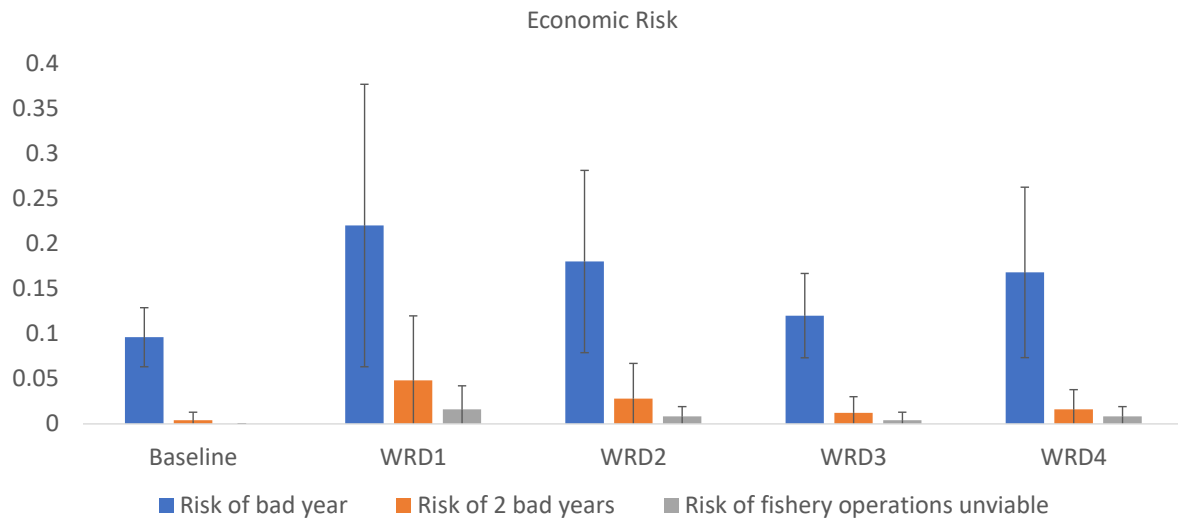


Figure 100. Comparison of average (with standard deviations) Economic Risk when using the MICE ensemble to evaluate the impact of alternative WRDs on common banana prawns.

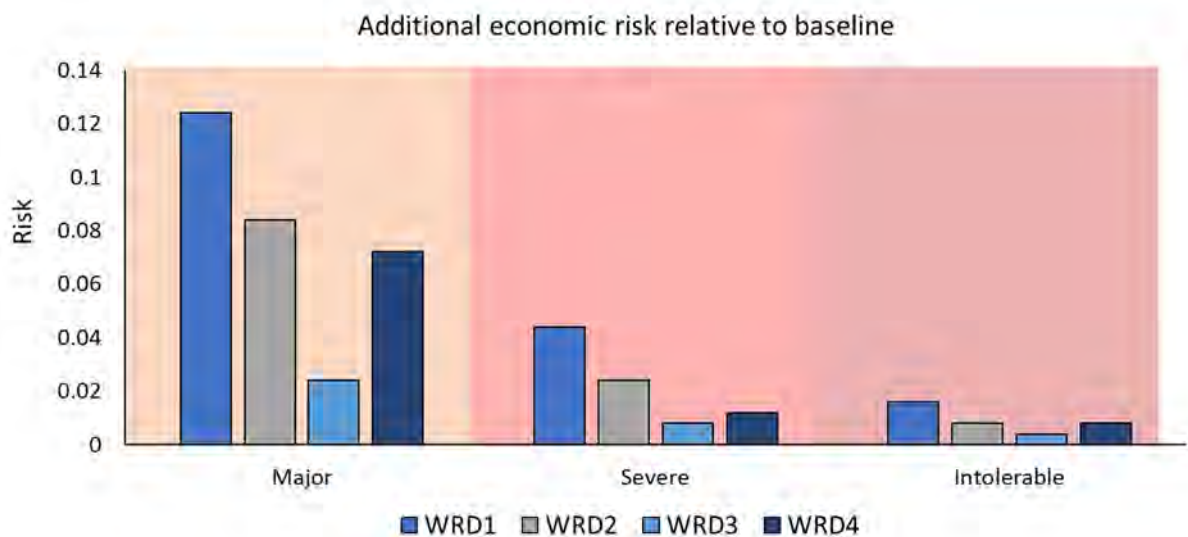


Figure 101. Graphical comparison of the relative probability of occurrence of major risks (defined as risk of a bad year), severe risk (two successive bad years) and intolerable risk (fishery operations becoming unviable due to three or more consecutive bad years) predicted in response to alternative WRDs impacting the common banana prawn fishery.

Discussion

The aim of this study was to develop a MICE (Model of Intermediate Complexity) (Plagányi et al. 2014) that integrates existing data and understanding of key species and processes in the Gulf of Carpentaria, and this has been achieved, noting that there is scope for improving and expanding the approach in future work. Developing a tractable model of intermediate complexity that focuses only on key aspects has been an extremely complex task. The GoC is an extremely complex system which makes it challenging to capture the dynamics in a model: the Gulf is highly seasonal with changes in monsoonal winds and rainfall in turn driving substantial changes in currents, phytoplankton productivity and recruitment and survival of coastal marine species (Rothlisberg and Burford 2016). The life history of species such as the common banana prawn *Penaeus merguensis* is also extremely complex due to the combination of their offshore spawning behaviour, inshore mangrove-lined nursery grounds and movements between these habitats (Vance et al. 1998, Vance and Rothlisberg 2020, van der Velde et al. 2021). Our MICE model uses modelled historical flow estimates to produce quantitative estimates of the impact of alternative flow regimes on the relative abundance of key fishery and other marine species in the Gulf of Carpentaria, as well as impacts on total fishery catches and value. The model is validated using available data and we account for structural and parameter uncertainty by using an ensemble comprising five alternative model configurations, in order to provide as rigorous a basis as possible for model estimates. We also computed risk statistics to show what the risks and trade-offs are under alternative scenarios in order to link with the risk-based ecohydrological approach adopted in Queensland (McGregor et al. 2018). This process includes measurement endpoint (often these are a suitable surrogate) linked to a threshold of concern; these become the ecohydrological rules. Our study will assist in informing on ecological assets such as barramundi, mud crabs, sawfish and prawns, as well as to refine definitions of a threshold of concern for some species.

There are good catch and effort data available for key fishery species and we analysed for explanatory trends in these data, such as the extent to which we are able to explain spatially-disaggregated catches based on fishing effort in that region. As part of this project, we have also compiled (complete and part-complete) series of environmental (ocean-climate) data that affect the GoC ecosystem, including the key fishery species. Historical analyses have shown that some of these environmental variables are correlated with catch, recruitment and growth of particular species. In addition, we have updated and expanded river catchment flow scenarios from streamflow models developed to explore the water resources of several GoC catchments as part of the FGARA, NAWRA and RoWRA catchment land resource and water resource assessments (Petheram et al. 2018a, Petheram et al. 2018b).

For fishery species, we built the model up in a stepwise fashion, for example, by first using effort data to explain catches taking account also of the underlying population dynamics. Next, river flow (and other environmental drivers where relevant) were added to the model to assess whether they improved the model fit and thereby helped explain the observed variability in the fishery data. Our banana prawn, barramundi and mud crab models all demonstrated good predictive power using river flow as a driver of variability in fishery catch. These models were therefore a reliable basis to investigate the likely impact of flow reductions using alternative end-of-system flow scenarios. We also assessed plausible impacts (albeit with less rigour due to fewer data being available) on largemouth sawfish and the two habitat-forming groups – mangrove and seagrass communities. The MICE assisted in integrating and quantifying the complex non-linear responses of threatened largemouth sawfish to further anthropogenic modification of the GoC. The MICE was able to simultaneously evaluate these complex trophic, technical and environmental interactions in a single integrated framework that also accounts for spatial differences in impacts. At the same time, for prawn species, the MICE was able to account for a degree of mixing between spatial regions. The outputs from the models are intended for use to help inform decision making and management of the GoC water resources.

The MICE has also considerably advanced attempts to model key habitat groups, including mangroves and seagrass as a first step to quantifying potential changes in these supporting habitats in response to changes in flow regime. Mangroves and seagrass are themselves important for conservation, as key

habitat and as sources of blue carbon (Sasmito et al. 2019). Mangroves, sea grass and saltmarshes are known as blue carbon ecosystems because of their ability to sequester significant amounts of organic carbon compared with other ecosystems, and hence they are globally important systems (McLeod et al. 2011, Lovelock and Duarte 2019). We based our representation of mangrove large-scale dynamics on existing knowledge of key drivers such as solar exposure, cyclones, air temperature and minimum sea level. Moreover, to represent the influence on large-scale mangrove dynamics of changes in river flow, we used a mechanistic approach and drew on previous empirical research showing that mangroves produce less litter when subject to water and salinity stress (Conacher et al. 1996), they can be highly vulnerable to sea level variability and extreme low sea levels (Duke et al. 2017, Lovelock et al. 2017a), and variations in rainfall and sea level can have large influences on the landward and seaward extent of mangroves (Eslami-Andargoli et al. 2009, Asbridge et al. 2016, Duke et al. 2019). We used changes in river flows as a proxy for salinity stress, for the level of hydration or dehydration of mangroves (in combination with temperature), and the potential for landward or seaward expansion. As more data at appropriate scales become available, we may be able to improve quantification of large-scale influences of changes in river flows on mangroves.

The MICE also produced some outputs that considerably advance our ability to quantify the relative contributions of different river catchments to common banana prawn productivity and yield. Previous approaches have convincingly demonstrated relationships between common banana prawn catches and river flow, for example using correlation methods, general linear modelling approaches (Vance et al. 1985, Bayliss et al. 2014) and most recently, a spatiotemporal Bayesian model (with linear predictor functions) to predict how changes in prawn spatiotemporal catch vary with flows from the Mitchell, Gilbert and Flinders Rivers (Broadley et al. 2020). Our study is the first fully dynamic (i.e. models changes over time and under varying conditions) model that covers the entire GoC region and is able to incorporate non-linear interactions, cumulative effects (for example, combined effects of changes in fishing effort, trophic interactions, additional environmental drivers) and account for spatial differences and overlaps in the population dynamics.

Previous estimates of what the maximum common banana catch loss might be under extreme scenarios were provided based on a Bayesian statistical analysis by Broadley et al. (2020), who used a more restricted subset of data and different flow scenarios. Our model did not predict quite as extreme as a 50% decrease in catch (in a small number of years under a very high extraction scenario), but rather the MICE estimated maximum catch decreases of about one-third to 41% in some years and an average catch decrease of around 21% to 29% across all years under very high extraction scenario WRD5. The biggest decrease in catch was estimated for the Flinders River catchment - down to about half (47%) of that estimated under baseline flows. Overall, both approaches corroborate extensive previous work highlighting the critical dependence of common banana prawn abundance and catches on end-of-system flows (Vance et al. 1985, Bayliss et al. 2014, Petheram et al. 2018b, Pollino et al. 2018, Burford and Faggotter 2021).

For common banana prawns, we found that all of the six major rivers that are explicitly represented in the MICE (i.e. including the Embley River which influences Region 1 and the Roper River which influences Region 7) are influential with the relative role and contributions of each varying by region and by year. The combined portfolio of rivers thus acts to 'stabilise' or maintain the common banana prawn population across the entire GoC and reducing flows from one or more rivers will have non-linear effects on common banana prawns and other marine species which this project is quantifying. Different river systems are important in different years and the combination of these flow anomalies across the different systems ultimately determines the system productivity and catch that is available to be caught. For prawns, the preferred model version 5 accounted for the enhanced productivity effect proposed by Burford and Faggotter (2021) and Duggan et al. (2014) whereby even once the freshwater influence within the estuaries subsides, microphytobenthos and meiofauna within the sediments recover to levels equal to pre-flood conditions and remain high for months (Duggan et al. 2014, Burford and Faggotter 2021).

In terms of economic consequences predicted on the Northern Prawn Fishery, we have established that the breakeven point for the fishery as a whole is ~1300t of banana prawn biomass at the start of the season as this is where total fishery profit tends to zero. Total fishery profits increasing linearly with banana prawn biomass after 1300t, suggesting that banana prawn stocks are the key driver of overall profits in the fishery.

We also estimated the longer-term effects of changes in banana prawn biomass on the NPF under the assumption that the fleet size is constant. As expected, reductions in banana biomass reduce the long-term profits to the fishery. For example, simulated reductions in common banana prawn biomass in the region of 30% (relative to an average year) resulted in a 26% decline in estimated Net Present Value (NPV)(in absolute terms a decline in NPV from \$782 million to \$582 million). These estimates are sensitive to the optimal level of boats in the fishery and assumes there are no impacts on tiger prawn biomass associated with any other factors (e.g. climate change and/or cyclones).

Mud crabs (*Scylla serrata*) are also characterised by an inshore/offshore life history, however, unlike banana prawns, both their juvenile and adult phase are estuarine residents. A number of studies have focused on trying to understand environmental and fisheries factors causing changes in mud crabs in the GoC (Robins et al. 2020) and our model complements these efforts because of its ability to couple the dynamics to environmental drivers across the entire spatial scale of the GoC. For the mud crab model component we found that accounting for river flow significantly improves the fit of the model when comparing observed versus predicted mud crab catches. Our model was successfully able to quantify the effect of changes in river flows on both recruitment and catchability. Moreover, in some regions, linking air temperature to mud crab mortality rates further significantly improved the model fit.

Barramundi (*Lates calcarifer*, a large catadromous predator) are characterised by an inshore/ riverine life history and occupy estuaries and close-inshore habitats as adults. Previous research has shown that growth rates of juvenile and adult barramundi are significantly affected by river flows (Tanimoto et al. 2012), probably through habitat access and/or food availability and hence 'year-class strength' (which is an index of survival) of barramundi is significantly affected by river flows (Robins et al. 2006, Halliday et al. 2010, Streipert et al. 2019). The MICE has drawn on previous research and existing models to develop a tailored spatial and age-structured barramundi sub-model in the MICE.

In terms of the alternative WRDs, streamflow models were used to produce a number of scenarios of the impact of WRD scenarios on historical natural river flows for the Mitchell, Gilbert and Flinders Rivers. We identified water resource development scenarios for the Flinders, Gilbert and Mitchell Rivers that were feasible given the topographical features of the respective catchments relative to proximity to arable soils. The hypothetical but feasible WRD scenarios were defined based on workshop outcomes and discussions with scientists involved in the FGARA and NAWRA projects and stakeholders, with:

- One of the WRD scenarios in the Flinders River catchment was 175 GL year⁻¹ (75% reliability) pump-extracted and stored off-stream, with water harvest being distributed 10 sites across the catchment sited adjacent to arable soils.
- One of the WRD scenarios in the Gilbert River was the Green Hills Dam (172 GL year⁻¹ at 85% reliability) upstream of Georgetown. The scenario diverts water to arable soils downstream. Water impoundment within the Gilbert River both increases and reduces up to 200 GL of water weekly, though 200 GL anomalies were infrequent.
- One of the WRD scenarios in the Mitchell River was two in-stream dams; the Nullinga Dam (65 GL year⁻¹ at 85% reliability) in the upper Mitchell River catchment and the Pinnacles Dam (1248 GL year⁻¹ at 85% reliability) in the mid-catchment area (west of Chillagoe). The two dams were modelled in series to divert water to arable soils downstream. The change in flows in the Mitchell River was predominantly flow reduction, though in a few years increases of ~ 1 GL were predicted. Flow reductions of approximately 10 GL occurred in only 10 weeks of the

2548 weeks since 1970. Weekly flow reductions of 5 GL were common, though not as common as flow reduction in either the Flinders or Gilbert Rivers. However, the mean annual flow of the Mitchell River is 15579 GL, so the percent quantum of flow reduction was very low.

We modified and applied a risk analysis method to broadly classify risks under alternative WRDs to (A) the species habitat-forming groups sub-set used in our MICE and (B) dependent fisheries. In the latter case, for the banana prawn fishery, we also conducted a preliminary economic risk analysis. We averaged results over the full ensemble and at the regional scale (model regions 2-6) to provide an integrated way to evaluate and cross-compare the risks under alternative WRDs while accounting for uncertainty in model structure. In general, WRD1 emerged (as expected) as the highest risk development, with moderate to intolerable risks predicted for all species and groups except for seagrass, and both in terms of population-level risk and fishery risk. Overall, WRD3 was predicted to pose lower risks overall compared with the other key WRD scenarios because it simulates development applied to the Mitchell River but not Flinders or Gilbert Rivers (Fig. 1). For fishery and habitat species, WRDs within multiple catchments poses the greatest risk to the marine community and species of economic and ecological importance; regardless of whether the allocation of water for extractive-use is set at a 'moderate' level, or double that level. Implementing WRDs within a subset of catchments has a much-reduced impact on the sustainability of the marine community and coastal ecosystem services.

When evaluating the relative risks of WRDs per catchment, we found that for prawns, the Flinders River catchment emerged as being the most sensitive to WRDs, which was consistent with previous findings of (Burford and Faggotter 2021) who suggested that the Flinders estuary is more productive than the Gilbert and Mitchell estuaries and also the most vulnerable to excessive water development. The only model version for which the Flinders catchment didn't rate as having relatively highest risk for prawns was Model version 2 which assumes no connectivity between the different catchments.

Based on our economic analysis of the profit break-even point for the fishery (and consultation with fishing industry representatives), we defined a "bad" year for common banana prawns as one where the annual catch is 2000t or less. Below this lower limit total fishery profits are reduced and given the crew payment system where their income is a proportion of catch, crew retention is compromised. Moreover, the impact of a "bad" year would be considerably exacerbated if it is repeated for a second year. We found substantial increases in economic risk under all WRDs and across the model ensemble. For Model version 1, we found that the risk of a bad year increased by a factor of 2.5 under WRD1 (WRD within all catchments) and more than doubled under WRDs 2 and 4 (WRD within some catchments), compared with a 50% increase under WRD3 ('moderate' WRD in the Mitchell River catchment). Our results also predicted an increase (from zero) in the risk of successive bad years or fishery ruin.

For barramundi, the Flinders River catchment was evaluated as having reasonably high risk to WRDs, but the Gilbert River catchment emerged as overall the most risky scenario for barramundi abundance and catches. Note WRD3 assumed baseline flows for the Flinders and Gilbert Rivers, and hence the risk is always higher for the Mitchell River catchment under this WRD, and in the case of barramundi posed higher risk than WRD2 and WRD4.

The patterns that emerged for mud crabs were once again slightly different, with the Flinders and Gilbert River catchments evaluated as roughly equally vulnerable with risks often as high as the severe risk category (Fig. 2, Table 6). There was little risk for mud crabs in the Mitchell River catchment given a very weak flow relationship estimated for this region. On the other hand, risks to sawfish population abundance were almost always evaluated as severe to intolerable across WRDs, although risks tended to be relatively lower overall for the Mitchell River system (Fig. 2, Table 6).

When evaluating relative risks to key habitats such as mangroves, a different risk profile emerged. The MICE predicted major to severe risks to mangrove habitats under some WRD scenarios, but negligible

risks to seagrass, although we did not account for potential increases in nutrient levels and turbidity that may be associated with WRDs.

Based on the best available scientific information (Lear et al. 2019, Kyne et al. 2021, Lear et al. 2021, Pillans et al. in press), we modelled sawfish recruitment as dependent on wet-season flows, plus accounted for boom-bust recruitment dynamics and increases in mortality in dry years (when pools die). Across a range of alternative parametrisations, we found that decreases in river flows translated into substantial decreases in population abundance because the life-history characteristics of largemouth sawfish mean they are unable to compensate or rapidly bounce back in response to decreases in their productivity. Life-history characteristics are recognised as key determinants of the resilience of species to external pressures influencing their reproduction and survival, and hence extinction risk (Hutchings 2002, Hutchings et al. 2012, Siple et al. 2019). The most likely explanation for largemouth sawfish emerging in our model results as highly sensitive to alterations in flow is because these chondrichthyans have life-history traits that make them more vulnerable to extinction risk than crustaceans (prawns, mud crabs) and bony fish (barramundi). They have relatively much lower productivity as have infrequent small litters, slow growth rates and late sexual maturity and hence are less able to compensate for any increases to their mortality rate (Stevens et al. 2000, Salini et al. 2007, García et al. 2008, MacArthur and Wilson 2016). In particular, they have a late age at maturity (ca. 8-10 years – (Kyne et al. 2021, Pillans et al. in press)) – considerably longer than for the other species we modelled – and age at maturity is considered a universal predictor of extinction risk regardless of taxonomic affinity (ie. in fish, chondrichthyans and mammals) (Hutchings et al. 2012).

Our findings corroborate those of Lear et al. (2021) who demonstrated that the body condition of *Pristis pristis* in the NT's Fitzroy River is tied to patterns in flooding and drought, with energy intake curtailed in years with low wet season flood volumes. These authors also caution that WRDs have the potential to worsen the effects of droughts or create drought-like conditions which could also lengthen the period when pools sawfish rely on become disconnected or stagnant (Bunn et al. 2006). Our study therefore adds to a growing body of work highlighting that WRDs and climate change both pose serious threats to threatened and endangered euryhaline species in arid freshwater environments, such as the largemouth sawfish.

One of the key issues when dealing with sawfish is the lack of quantitative data on abundance, movement/habitat use and the response of individuals and populations to variability in environmental flows. Research is urgently required to gain a better understanding of largemouth sawfish population status and ecology in rivers that are likely to be impacted by water extraction and upstream dams. Targeted field-based research on sawfish could significantly improve our understanding of abundance and movement, as has been done for the Critically Endangered speartooth shark (*Glyphis glyphis*) in the Wenlock and Ducie Rivers in Queensland (see Box 1 in Appendix 13 for further details).

There is considerable scope to improve and expand the MICE – indeed no model is ever truly complete. However, our focus was on capturing first order effects driving changes in the population dynamics before adding secondary effects in a stepwise fashion to evaluate whether they substantially improved the model's predictive ability. When possible, we did not include detailed mechanistic and other processes in the model if we did not have some basis to validate or inform these additions. When adding additional trophic links between barramundi, largemouth sawfish and common banana prawns, we found that this merits further investigation (preferably informed by data) but that our model results were robust to excluding this additional complexity.

Accurately representing ecosystem dynamics, species with different life histories, multiple users and jurisdictions, as well as the impacts of WRDs is complex as there are often more than one driver at play. We therefore acknowledge that our MICE framework has limitations such as not fully representing connectivity between all regions, not explicitly modelling the oceanographic and wind-driven dynamics of the GoC, not including multiple other species that are trophically-linked to our key species and may be similarly impacted by flows (eg threadfin salmon, grunter), not accounting for species composition and different life history characteristics of the habitat-forming groups and using relatively simple relationships to represent complex mechanistic processes. As more observational data

become available – particularly for the shallow waters of the GoC which have received less attention – it may be possible to reduce the uncertainty associated with representation of these aspects.

Our spatial MICE linking river flows, estuarine and marine systems provides a useful framework for ongoing studies to improve understanding and quantification of predicted impacts of WRDs and climate drivers, and is readily extended to represent WRDs applied to other catchments, acknowledging that there remains considerable scope for improving our modelling approach as new information becomes available. It was beyond the scope of this project to refine some of the model components as much as could ideally be done, and it is hoped that there will be future opportunities to continue to build on this work. For example, there is scope to gradually increase the complexity of the modelled systems such as to account for a range of inter-specific interactions, connectivity scenarios, other fishery sectors (e.g. recreational). Given the model has been specified, less resources would be required to re-run a series of alternative WRD scenarios in a future project. In particular, the MICE is an ideal framework for exploring climate change impacts and adaptation scenarios. The MICE will also be useful in case there are other future development scenarios that need to be tested, as well as alternative mitigation options. A separate project is currently underway to investigate impacts of WRD scenarios on the Roper River, and as this region is included in the MICE, it could also be used to quantify these impacts from an ecological perspective. An advantage of the MICE framework is that it can quantify cumulative impacts on the system. There is also the potential to extend the MICE to include the entire NPF area, or an even greater area.

Conclusion

The impacts of natural streamflow modification within northern rivers on the habitats, habits and populations has the potential to impact key fishery species; such as three high-value species, common banana prawns, barramundi and mud crabs. Water resource development will modify the level and timing of historical flow regimes that are characteristic of tropical Australian rivers (Pollino et al. 2018). Water extraction or impoundment reduces flow volumes, consequentially reducing annual ecological cues, crustacean and fish populations, and population biomass available to be fished (Mallen-Cooper and Zampatti 2020). Sequential years of reduced flows due to water diversion for irrigation may increase the frequency of low-level flow years and hence increase the years of uneconomic catches to a level where the long-term economic viability of particular fisheries is compromised (Broadley et al. 2020). In Australia's wet/dry tropics, species have evolved life-strategies to match the highly variable volume and timing of historical flows, and flows that vary from these seasonal and annual patterns likely debilitate long-term population stability.

Changes to the timing of flows, particularly early season flows that enter the estuary at the end of the annual dry season, can modify flow-cued behaviours that are critical to the survival of life-history stages of particular species: emigration to nearshore habitats for banana prawns, and upstream migration for largemouth sawfish. As well, reduced flow volumes modify the delivery of nutrient loads to the estuary and nearshore habitats (Burford et al. 2012). Medium to long-term effects on tropical estuaries from flow modification are reduction of nutrient transport and flux in estuarine and coastal habitats (Burford et al. 2020a), and changes to the deposition regime within the estuary and to coastal soft-sediment geo-morphology (Asbridge et al. 2016).

This project has brought together a wealth of information and data to synthesize and quantify current understanding of the links between river flows and the Gulf of Carpentaria marine ecosystem. This is in turn integrated into a MICE (Model of Intermediate Complexity for Ecosystem assessments) (Plagányi et al. 2014, Collie et al. 2016) which focuses on key species and key processes to address management questions pertaining to the system. The MICE is a dynamic spatial model that is fitted to available data in order to identify drivers and produce projections that provide as rigorous management advice as possible.

MICE components were built up in a stepwise fashion, only adding additional complexity if needed or if justified due to statistical testing demonstrating that the addition of further complexity is able to significantly better explain the available data. For example, first we assessed the extent to which fishery effort data could explain observed changes in the spatially-resolved population trends of a species. We then added a link with river flow to try and improve model fits, and finally added other identified key drivers, such as other physical variables or trophic interactions. We used innovative methods to link river flows and other physical variables to population dynamics. Our results are consistent with previous studies highlighting the strong relationships between river flows and tropical fishery dynamics, particularly for banana prawns, mud crabs and barramundi (Vance et al. 1985, Robins et al. 2005, Robins et al. 2006, Robins and Ye 2007, Meynecke et al. 2012b, Broadley et al. 2020). Our results substantially advance understanding and quantification of these relationships because:

- we use a dynamic framework rather than static correlations,
- our results are resolved for different spatial regions,
- our results enable detailed investigation of fine-scale patterns and seasonal changes in flows because we use a weekly time-step (for prawns) and monthly time-step (for mud crabs, barramundi and sawfish),

- we use of an ensemble approach and an integrated framework with fitting to data means that the model is able to simultaneously account for changes due to biological growth, trophic interactions, physical processes and river flow (both magnitude and intra-annual patterns). thereby building confidence in the ability to correctly attribute the reasons for ecosystem changes.

The summation of our approach enables the model to provide the most rigorous platform to date for use to simulate alternative plausible scenarios to support proactive planning and mitigation.

The MICE developed in this study includes a number of novel aspects targeted to selected species, and providing coverage for all of the species,

- the model is the first spatial, dynamic model fitted to data and linked to physical drivers at the scale of the entire GoC.
- The model is coupled with the latest outputs of complex river models.
- The common banana prawn component has the most sophisticated representation to date of catch estimates per week and per region as a function of a suite of river catchment flows.
- The mud crab model component is the first fully age-structured, spatially-structured and sex-disaggregated model for the entire GoC.
- The barramundi model component includes novel representation of the protandrous life history and is the first age-structured model with linked river flows for the entire GoC.
- The sawfish model component is the first fully age- and spatially-structured model for the entire GoC that captures the different life history characteristic of two sawfish species.
- The mangrove and seagrass model components are also unique in being intermediate complexity representations of broad-scale changes in these groups in response to major physical drivers such as river flow, sea level height, cyclones, temperature and light in each of the model spatial regions.

The MICE has also developed the first large-scale spatial model to represent changes in microphyto-benthos and meiofauna which sustain primary production and estuarine foodwebs (Burford et al. 2012, Burford et al. 2016, Duggan et al. 2019). This latter component is still preliminary but provides a useful starting point for ongoing work to quantify how the onset of the wet season changes the estuarine environment and the habitats available to estuarine residents. Freshwater inflows to river channels and estuaries transport sediments, deliver nutrients, lower salinities, create physical forces and scouring, inundate over-bank habitats, connect ephemeral habitats, and modify tidal inflows. Each species has evolved to use one or more of these annual opportunities to enhance their population or move to a habitat suitable for the next phase of their life history. The timing and volume of wet season flows and floods are critical to estuarine conditions and species response to the monsoon-driven river flows.

The results of the MICE provide novel insights into system dynamics. the MICE enabled quantification of a river portfolio effect across the Mitchell, Gilbert, Norman and Flinders Rivers, such that WRDs applied to a single river or different combinations of rivers had complex cumulative and synergistic effects on common banana prawn abundance and catches throughout this sub-region. Different river systems are important in different years and the combination of identified flow anomalies across the different catchments ultimately determines the GoC littoral productivity and the populations of fishery species that are available to be caught.

For mud crabs, the MICE was able to successfully quantify the influence of flows in determining fishery catches, plus was able to demonstrate that significantly better model fits could be achieved for some areas when accounting for the impact of air temperature on mud crab dynamics.

The results from evaluating impacts of alternative WRDs are intended for use in informing recommendations on the relative impacts and risk assessment of alternative WRD scenarios, as well as to explore potential mitigation strategies for reducing risks. Mitigation strategies can be deployed in consultation with water resource managers to reduce impacts on the natural GoC ecosystem and best deploy catchment water resources to achieve both agricultural and fishery production.

Summary of key findings and recommendations

- Changes from baseline flows due to WRDs had variable impacts on all species and catchment regions, with impacts ranging from minor through to extreme impacts under some scenarios
- Model-predicted catchment-system impacts increased with the greater volume of water extracted or impounded and number of rivers on which dams or WRD scenarios were deployed
- For common banana prawns, the MICE enabled quantification of a river portfolio effect across the Mitchell, Gilbert, Norman and Flinders Rivers, such that WRDs applied to a single river or different combinations of rivers had complex cumulative and synergistic effects on common banana prawn abundance and catches throughout this sub-region. Simultaneous WRD across multiple catchments negatively affected banana prawn populations from a moderate to a major degree. Lower water allocation WRDs lessened the impact, while WRD within a subset of catchments had the least impact
- Model results corroborated previous research exploring changes in freshwater inputs on the primary productivity of the same river estuaries (Burford and Faggotter 2021) underscoring the longer-term benefits to system productivity that result from ongoing productivity boosts driven by natural flow regimes. Our model predicted substantially greater local and regional decreases in banana prawn catch and abundance if accounting for the flood productivity boost effect.
- We found the largest change in profit for the common banana prawn sub-fishery of the NPF under WRD1. The economic impacts for WRD3 were the least compared to the other WRD scenarios. The average change in profit in year 1 across all the scenarios (WRD1-4) for the predicted decline in mean biomass is approximately 10.6% *ceteris paribus*. In absolute terms this is approximately an average \$3.1 million reduction in profit (across all the scenarios) in total fleet profit from about \$29.0 million to \$25.9 million
- Barramundi were generally predicted to be most sensitive to the WRDs applied to the Gilbert River region, with both a single and two dams predicted to cause large declines to the local populations. Overall, the model ensemble suggested average catch would decrease up to about 20%, with a maximum decrease of around 27%, under WRDs 1 and 3
- For mud crabs, substantial negative declines in abundance and catch were predicted for the Flinders and Gilbert catchments under both medium and high-impact WRDs (WRD2 and WRD1 respectively) and for all five model versions in the MICE ensemble, but almost no change was predicted for the Mitchell catchment
- Changes in catch under WRDs were similar to the magnitude of change observed in historical catches between wet, intermediate and dry years, particularly for barramundi and mud crabs

- Across each of the modelled species, the incorporation of a low river flow extraction threshold value (TH) as part of the pumped-water extraction regime caused a much more substantial negative impact on model-predicted catches and abundance. A significant decline in catch associated with water extraction at low-levels of river flow was consistent with previous studies pointing to the need to maintain flows well above an ecosystem-minimum level. Maintaining low-level flows, especially early season flows, enhances the estuarine ecosystem by re-establishing estuarine/riverine connectivity and ameliorating estuarine conditions after prolonged dry season stressors
- As the Mitchell River is a perennial system that exhibits less environmental extremes than the Gilbert and Flinders Rivers, WRDs for the Mitchell River were predicted to have a less extreme impact on the local populations of barramundi, mud crabs and prawns
- Our model implicitly accounts for the fact that the timing of flows (and early wet season flows in particular) is critical for most species, and hence model results to evaluate the impact of alternative WRDs include integration of changes in intra-annual flow levels evaluated using either a weekly or monthly time step
- Across a range of alternative parametrisations, we found that decreases in river flows translated into substantial decreases in population abundance of largemouth sawfish because their life-history characteristics mean they are unable to compensate or rapidly bounce back in response to decreases in their productivity.
- Largemouth sawfish were predicted to show high sensitivity to almost all WRDs tested with more large allocation of water for extraction and flow modification within multiple catchments (e.g. WRD1 and WRD2) predicted to result in extremely large local population declines. Sawfish model results differed in a number of ways from those for prawns, barramundi and mud crabs as changes to WRD settings such as river flow extraction threshold value and pump rate did not result in major differences in predicted impacts on sawfish as was the case for some of the other species – rather anything other than very low extraction amounts were predicted to have a substantial negative impact on sawfish. The largest predicted population declines of sawfish corresponded to WRD scenarios for the Flinders River because this sub-population appeared to be the least resilient across a number of alternative scenarios.
- Results for largemouth sawfish suggested greater sensitivity to WRD scenarios involving water extraction compared with water impoundment i.e. dams (assuming free movement of the animals wasn't negatively impacted) with anything other than very low extraction amounts predicted to have substantial negative impacts on sawfish, across a range of alternative water extraction threshold and pump rate settings.
- Our preliminary investigations for largemouth sawfish suggested that sawfish abundance in all catchments were highly sensitive to WRDs but also provides guidance as to which river systems are likely to be most sensitive to flow modification (e.g. Flinders River) and the type of impacts (extraction WRDs performed worse than impoundment scenarios (i.e. dams)) In addition, the models shed light on recommended settings for future WRDs if these were to be implemented
- Model results were fairly robust to alternative model structures that included explicit representation of the dependence of barramundi and largemouth sawfish on estuarine prey availability (using common banana prawns as a proxy) albeit that this could worsen predicted impacts of WRDs in some scenarios
- In contrast to all the other MICE groups, seagrasses were predicted to marginally increase in abundance/distribution under some WRDs for some time periods, with fairly minor impacts (up to a 7% decline in seagrass abundance relative to base levels) across most scenarios.

- Model results suggest that WRDs may have a dramatic effect on mangroves with water impoundment (dams) predicted to result in large declines in mangrove abundance in affected areas. Our results suggested that outcomes for mangroves under the same extraction amount could be substantially worse if using a low river flow pump threshold (TH) and longer pump duration compared with a less impactful scenario using a medium river flow pump TH and shorter pump duration. A low river flow threshold allows the pumping of low-level flows whereby a large proportion of river flow is extracted and significantly less water passes downstream. In addition, a longer pump duration necessitates water extraction from flows other than peak flows, resulting in a higher proportion of the non-peak flows being extracted. Both pump routines disturb the pattern of river flow during low-level flows when water extraction disproportionately affects streamflows and downstream ecosystem service provision. Our representation of the influences of changes in river flows on mangrove large-scale dynamics requires further validation but were based on previous empirical research showing that mangroves produce less litter when under water and salinity stress (Conacher et al. 1996), they can be highly vulnerable to sea level variability and extreme low sea levels (Duke et al. 2017, Lovelock et al. 2022), and variations in rainfall and sea level can have large influences on the landward and seaward extent of mangroves (Eslami-Andargoli et al. 2009, Asbridge et al. 2016, Duke et al. 2019).
- For commercially fished species, model results suggested minor to large changes in average catches. Economic analyses corroborated that the maximum modelled decline in any year is likely to be of more interest to fishing industry stakeholders because they rely on regular annual income and may not be able to withstand even a few years of low catch or periods when they are unable to break even economically
- Our MICE framework is a useful tool for quantifying the impacts of WRDs across a range of species, catchment systems, time-scales and parameter settings. The MICE could readily be extended to include other species or systems. There is considerable scope to gradually increase the complexity of the modelled systems such as to account for a range of inter-specific interactions, connectivity scenarios, other fishery sectors (e.g. recreational); as well as to investigate what the additional effects of climate change might impose on this ecosystem (beyond the scope of this study but see (Fulton et al. 2018))
- Accurately representing ecosystem dynamics, species with different life histories, multiple users and jurisdictions, as well as the impacts of WRDs is complex as there are often more than one driver at play. We therefore acknowledge that our MICE framework has limitations such as not fully representing connectivity between all regions, not explicitly modelling the oceanographic and wind-driven dynamics of the GoC, not including multiple other species that are trophically-linked to our key species and may be similarly impacted by flows (eg threadfin salmon, grunter), not accounting for species composition and different life history characteristics of the habitat-forming groups and using relatively simple relationships to represent complex mechanistic processes. Although additional complexity can be added to the MICE, the current version is consistent with the MICE philosophy of finding the ‘sweet spot’ at which the uncertainty in policy indicators is minimised
- Research is urgently required to gain a better understanding of largemouth sawfish population status and ecology in rivers that are likely to be impacted by water extraction and upstream dams. Our study underscores the need for more population parameter data, highlighting key gaps highlighted including data on the natural mortality, status and connectivity of sawfish as is being addressed through planned close kin mark-recapture studies, and we also highlight the need for further targeted field-based research.

Ecological and fisheries risk assessment

- Our risk assessment classified WRD1 (highest water allocation and multi-catchment WRD) as the highest risk development, with moderate to intolerable risks predicted for all species and groups except for seagrass, and both in terms of population-level risk and fishery risk, followed by WRD2 and WRD4 (lower water allocation or single catchment WRD), both of which also predicted high risks to some populations and fisheries. WRD3 was assessed less risky because it assumes no development on the Flinders and Gilbert River catchments
- Largemouth sawfish were predicted to show the greatest sensitivity to WRDs (due to their low productivity life-history characteristics) with risks ranked as intolerable across a broad range of alternative water extraction or impoundment scenarios
- For common banana prawns, the Flinders River catchment emerged as the most sensitive to WRDs, consistent with previous findings from estuarine productivity studies (Burford and Faggotter 2021)
- We quantified the probability of alternative WRDs increasing the baseline economic risks to the common banana prawn sub-fishery of the NPF by computing the relative probability of occurrence of major risks (defined as risk of a bad year), severe risk (two successive bad years) and intolerable risk (fishery operations becoming unviable due to three or more consecutive bad years). We found that the risk of a bad year may more than double under some WRD scenarios
- The Gilbert River catchment emerged as the riskiest scenario overall for barramundi abundance and catches
- For mud crabs, the Flinders and Gilbert River catchments emerged as most vulnerable to WRDs with risks often as high as the severe risk category
- The MICE predicted major to severe risks to mangrove habitats under some WRD scenarios, but negligible risks to seagrass, although we did not account for potential increases in nutrient levels and turbidity that may be associated with WRDs

Our spatial MICE linking river flows, estuarine and marine systems provided a useful framework for ongoing studies to improve understanding and quantification of predicted impacts of WRDs and climate drivers, and is readily extended to represent WRDs applied to other catchments, acknowledging that there remains considerable scope for improving our modelling approach as new information becomes available

Implications

This project has a broad range of implications for industry, communities, managers and policy makers. First, the study has enabled collation and integration of a vast amount of information (from previous and ongoing studies, plus consultation with experts), data (including physical variables from a range of sources and fisheries data from several different fisheries as well as three different jurisdictions) and has collated and analysed available information pertaining to potential WRD scenarios that may impact the GoC. Second, the MICE has integrated all this available information and data, focussing on key aspects to keep it tractable, and hence is the first ecosystem model specifically tailored to serve as a reliable tool for answering complex multi-sector management questions (based on the best available current knowledge). Third, the river modelling improvements and additions done as part of this project are a valuable resource not only for this project but for other projects to quantify the end-of-system flow impacts of a range of potential WRD scenarios. Fourth, the MICE has included detailed spatial modelling of several commercial species and a species of conservation concern, with such models significantly advancing the available toolbox of approaches to inform understanding and management – for example, (1) the common banana prawn component is the first fully dynamic spatially-resolved model with population dynamics simultaneously driven by a combination of river catchments; (2) the mud crab component is the first spatial age-structured model that simultaneously captures different management practices in Qld and NT and the impacts of physical variables on these stocks and will therefore be a useful resource for ongoing management of mud crabs both in GoC and elsewhere; (3) the spatial barramundi model linked to flows is useful to complement other modelling initiatives; (4) the sawfish component is the first attempt to integrate the complex life history and past dynamics of this threatened euryhaline chondrichthyan and link river flow to sawfish recruitment and survival. The model can be refined over time if more data and information become available to assist with conservation efforts and highlights the need for an integrated plan to improve broadscale (i.e. over multiple river systems) knowledge of *P. pristis* in a rigorous fashion; (5) large-scale representation of GoC mangroves and seagrass is the first such attempt to model changes in these key habitats – simultaneously vital blue carbon assets – in response to changes in river flows, light levels, cyclone impacts and other physical drivers, and (7) the model simultaneously integrates a range of complex system connections and interactions in a single dynamic framework. The MICE includes representation of species of commercial and cultural importance – for example mud crabs and barramundi – and therefore provides insights for local communities as to potential future changes in the environment and dependent species. Last but not least, the integrated results of the project may be used to inform trade-off decisions related to WRD scenarios or changes to these.

It was beyond the scope of this project to refine some of the model components as much as could ideally be done, and it is hoped that there will be future opportunities to continue to build on this work. For example, there is scope to gradually increase the complexity of the modelled systems such as to account for a range of inter-specific interactions, connectivity scenarios, other fishery sectors (e.g. recreational). Given the model has been specified, less resources would be required to re-run a series of alternative WRD scenarios in a future project. In particular, the MICE is an ideal framework for exploring climate change impacts and adaptation scenarios. The MICE will also be useful in case there are other future development scenarios that need to be tested, as well as alternative mitigation options. A separate project is currently underway to investigate impacts of WRD scenarios on the Roper River, and as this region is included in the MICE, it could also be used to quantify these impacts from an ecological perspective. An advantage of the MICE framework is that it can quantify cumulative impacts on the system. There is also the potential to extend the MICE to include the entire NPF area, or an even greater area.

Recommendations

Our GoC MICE is the first integrated framework for quantifying the impacts of WRDs across a range of species, catchment systems, scales and parameter settings. While there is scope to build on and refine this framework we are nonetheless able to provide some considerations and recommendations with respect to water resource development going forward:

- **Timing of flows:** Our model implicitly accounts for the fact that the timing of flows (and early wet season flows in particular) is critical for most species, and hence model results to evaluate the impact of alternative WRDs include integration of changes in intra-annual flow levels evaluated using either a weekly or monthly time step.
- **Quantity of water allocated for extraction/impoundment:** the amount of water either extracted or impounded will be important and should maintain a minimum ecosystem requirement. The accounting of water allocation may depend on the acceptable decline in species catch/biomass (or risk) that stakeholders are willing to accept. Guidance will be needed around water management that maintains estuarine functions which are critical to support high-value fisheries and other species. Water management targets should be amended to account for sustainability of economically valuable species.
- **Number of WRDs:** given there was major to substantial risk for all catchments tested, the number of catchments (or WRDs per catchment) should be considered in future planning. We found that WRD scenarios had significant impacts on the GoC ecosystem: the MICE showed that one WRD per catchment posed a high risk to most dependent species in the GoC.
- **Types and settings of WRDs:** Our MICE showed that types of WRD were important, but vary depending on species. For example, extraction WRDs had greater impact on abundance than impoundment scenarios (i.e. dams) for sawfish, whereas the case was opposite for barramundi, and both were equally impactful for mud crabs. The model also shed light on recommended settings for future WRDs if these were to be implemented. Specifically, we recommend an extraction duration as short as possible (i.e. a high pump rate) with a high river-flow level extraction threshold.
- **Differences in catchments:** the MICE corroborated that not all catchments have the same characteristics and species are likely to respond differently depending on the region. Overall, WRD on the Flinders and Gilbert rivers were predicted to pose the highest risk, although Mitchell WRDs also had a high level of risk for some species.
- **Climate change:** management of water extractions should also be considered in the context of changing climate, which have not been tested.
- **Data and knowledge gaps:** our study underscores the need for more empirical data, with key gaps highlighted including data on the natural mortality, status and connectivity of sawfish as is being addressed through planned close kin mark-recapture studies (Bravington et al. 2016, Hillary et al. 2018). Extensions to our study, including adding additional species and fishery sectors (Indigenous, recreational) was constrained by the lack of suitable data. Our study was unable to address the critical need to involve Australian Indigenous peoples and include their values in water planning (Lyons and Barber 2021).
- **Wider ecosystem components:** the MICE has focussed on key species in the GoC system that are of economic or conservation importance and for which data were available. There are many other components of the ecosystem that we have not considered, including freshwater biota and many trophic links. We did not represent secondary impacts from WRDs – such as

potential increases in nutrient loads and turbidity. Thus, risk to the ecosystem could be greater than predicted by the MICE.

Further development

Some recommendations for next steps are provided in the final report. These include:

- Ongoing consultation and communication with interested and affected stakeholders is important to validate and improve model results, align model outputs with stakeholder and manager needs and to provide a basis for informing water planning and assessment.
- COVID-19 limited the planned project consultation after project results were available, and it may be possible to further extend consultation, particularly to include greater engagement with water resource managers, Indigenous peoples, recreational fishers as well as NT and Qld fishery managers
- The MICE could readily be extended to include other species or systems. There is considerable scope to gradually increase the complexity of the modelled systems such as to account for a range of inter-specific interactions, connectivity scenarios and other fishery sectors (e.g. recreational) including greater consideration of economics and social aspects.
- The MICE could be expanded to other catchments where past research has provided a comprehensive suite of cross-life-history data to enable insights on the contribution of flow modification upon each prawn life history stage. For example, a six-year historical study in the Embley River estuary and Albatross Bay (van der Velde et al. 2021): prawn spawning capacity, larval abundance, phytoplankton abundance, postlarval and juvenile abundance within the estuary, predation and mortality, as well as emigration response and adult abundance in Albatross Bay.
- The MICE is a useful tool that can be used to investigate what the additional effects of climate change might impose on this ecosystem. As summarised in the latest IPCC report (Arias et al. 2021), anthropogenic-mediated CO₂ emissions are driving warming of the upper ocean and global acidification of the surface open ocean. Recent decades have all been warmer than the preceding decades with recent increases in ocean temperatures approaching one degree Celsius. The average rate of global mean sea level is also increasing at a faster rate than previously, namely 3.7 [3.2 to 4.2] mm yr⁻¹ between 2006 and 2018 (high confidence) (Arias et al. 2021). There are also observed and predicted increases in the frequency of extreme events and oxygen levels have dropped in many upper ocean regions. The MICE can readily be projected forwards to quantify likely impacts on the GoC as well as to explore the efficacy of alternative resilience-building and adaptation strategies (see e.g. Fulton et al. 2018).
- Research is urgently required to gain a better understanding of largemouth sawfish population status and ecology in rivers that are likely to be impacted by water extraction and upstream dams. Our study underscores the need for more population parameter data for largemouth sawfish, highlighting key gaps including data on the natural mortality, status and connectivity of sawfish. Some of these data gaps are being addressed for other species (narrow sawfish) through planned close kin mark-recapture studies. We also highlight the need for further targeted field-based research such as has been conducted in the freshwater reaches of the Fitzroy River system (Whitty et al. 2017, Lear et al. 2019).

Extension and Adoption

The first project workshop was held 7-8 August 2019, at CSIRO's Queensland Biosciences Precinct (QBP) lab in St Lucia, Brisbane and included a diverse group of stakeholders. Thereafter, regular updates (including detailed presentations on all project aspects) and opportunities for input were provided at a number of NPF RAG and management meetings as well as three focussed teleconference workshops as follows:

1. NPRAG 7th-8th Nov 2019 (in person)
2. NPRAG 20th-21st May 2020 (tele/video-conference)
3. NPRAG 6th Aug 2020 (tele/video-conference)
4. NORMAC 1st Oct 2020 (tele/video-conference)
5. NPRAG 30th Nov-1st Dec 2020 (tele/video-conference)
6. Stakeholder Teleconference Workshop I: Prawn focus (24 February 2021)
7. Stakeholder Teleconference Workshop II: Mud crab focus (23 March 2021)
8. Stakeholder Teleconference Workshop III: Barramundi focus (24 April 2021)
9. NPRAG 12th-13th May 2021 (in person)
10. NPRAG Aug 2021 (tele/video-conference)
11. Final Stakeholder Workshop 9-10th August 2021 (held via teleconference due to COVID-19)

COVID-19 meant that we have not yet had the opportunity to travel and meet with additional stakeholders in northern Australia (for example, Darwin, Karumba, Cairns) as part of this project, although some of the project team have separately been in discussions with stakeholders in the region that have referred to components of this work. We will be looking for further opportunities to possibly travel and further engage with other managers, researchers, industry and communities.

In terms of outreach, this work was presented at the World Fisheries Congress, 20-24 September 2021, which provided an ideal opportunity for broad communication of our key research findings: *Plagányi et al. Using MICE to quantify trade-offs between fisheries, agriculture production and biodiversity objectives*. We also presented this work at the *Fisheries science and knowledge to meet the challenges of a changing Northern Territory* symposium hosted by the Research Institute for the Environment and Livelihoods, Charles Darwin University on 18 November 2021. In addition to journal articles, we are also planning an article for broader dissemination, most likely to be submitted to *The Conversation*. We are also developing a short video animation to support our outreach efforts, and will use social media, other media and CSIRO channels to further communicate our research outcomes.

Complementary projects investigating estuarine productivity and water quality (NESP 1.4) and juvenile banana prawn abundance (FRDC 2016/047) in the Mitchell, Gilbert and Flinders Rivers took three once-a-year snapshots of ecological productivity and estuarine extent that contribute to offshore fishery resources in the Gulf of Carpentaria (FRDC 2016/047, Addressing knowledge gaps for studies of the effect of water resource development on the future of the NPF; NESP 1.4, Links between Gulf Rivers and Coastal Productivity). MICE outputs will enable the extension of the snapshot of field data to a time series generated by regional models that investigate the contribution of river flow and ecological components of each of the three rivers to fishery productivity. The regional models relate to the individual rivers, allowing estimates of juvenile banana prawn production from estuaries of the Mitchell, Gilbert and Flinders Rivers to be compared to regional catch.

Project coverage

A number of additional articles are planned for the next few months.

Based on this project and related research, CSIRO submitted a number of recommendations for consideration in the National Water Reform 2020 Productivity Commission Draft Report 2020, 24 March 2021, CSIRO Submission 21/748.

These recommendations included the following:

Suggested inclusion 1 - The scope of the report could be broadened to:

- better represent northern systems and connections with marine systems;
- provide greater recognition of the critical role of estuaries;
- provide guidance around water management to maintain estuarine functions that are critical to support high-value fisheries and other species; and
- include information specifically for tropical ecosystems.

Suggested inclusion 2 - Differences in catchments and regional climate projections could be accounted for in the report.

Suggested inclusion 3 - Water management targets could be amended to account for sustainability of economically valuable species.

Suggested inclusion 4 - A chapter dedicated to matching water resource management with the environmental water requirements of rivers in Australia's wet/dry tropics is critical to the final NWR 2020 report to address the potential development of irrigated agriculture in Australia's northern tropical river catchments over the next 20 years.

Glossary

AFMA: Australian Fisheries Management Authority

Allocation: Water resource allocations (GL/year) which are either extracted by pumping or impounded i.e. via dams.

Baseline flow: Base river flows which include current water development but no future water development.

Bio-economic model: Separate model for Northern Prawn Fishery to which MICE outputs are applied.

CCA: Catch-at-age – catch data recorded per age class of fish (in this case barramundi).

CKMR: Close Kin Mark Recapture

CPUE: Catch-per-unit-effort

CSIRO: Commonwealth Scientific and Industrial Research Organisation

Ecosystem model: a multispecies model that models part or all of the ecosystem. There are a diverse range of these models, ranging from those that model the entire ecosystem and often include uncertainty and hence are used for strategic purposes, to those that are more focused on key parts of the ecosystem for which there are data to validate the model and hence can be used for more tactical use (see rigorous ecosystem model below).

Ensemble: A group or set of models with different specifications, used to address the same question

ENSO: El Niño Southern Oscillation

EPBC: Environment Protection and Biodiversity Conservation Act 1999 (EPBC Act) (Australia)

ERA: Ecological Risk Assessment

ERAEF: Ecological Risk Assessment for Effects of Fishing

FGARA: Flinders and Gilbert Agricultural Resource Assessment

GoC: Gulf of Carpentaria

IUCN: The International Union for Conservation of Nature – a global authority on the status of the natural world and the measures needed to safeguard it.

Mechanistic model: model structure in which interactions among ecological processes are represented using a mathematical model, often after scaling up small-scale processes before being used to predict broader-scale properties of ecosystems.

Métier: fishing activities that target a particular species by a certain gear type in a given area.

MEY: Maximum Economic Yield

MICE: Models of Intermediate Complexity for Ecosystem assessments

MICE ensemble: A set of alternative MICE model versions that are similar but may differ in model structure or use different input parameters, or may have different underlying assumptions e.g. how river flow affects population dynamics.

MICE sub-models: population dynamics sub-model for each species or group, which may be linked. These are DYNAMIC as model changes over time, plus capture NON-LINEAR relationships and feedbacks unlike static statistical models e.g. correlations, general linear models (GLMs).

MJO: Madden Julian Oscillation

NAWRA: Northern Australia Water Resource Assessment

NESP: Northern Environmental Science (NESP) hub on Northern Australian Environment Resources, funded by the Commonwealth Government.

NPF: Northern Prawn Fishery

NPFI: NPF Industry Pty Ltd

NPV: Net Profit Value

NT: Northern Territory

PR: Pump rate – river flow extraction pump rate measured as the number of days taken to pump water.

Qualitative model: A model that describes how a system or process works and e.g. the direction of change rather than magnitude of change.

Quantitative model: A model that quantifies the system and processes and e.g. the magnitude of a change.

Rigorous ecosystem model: a multispecies model, rarely a stock assessment model, that has been validated by fitting to data or has adequately accounted for uncertainty and hence provides more reliable outputs to inform tactical or strategic management advice than is typical of most ecosystem models (used in a strategic context).

River flow: End of system river flow computed using river models; standardised flow measures or flow residuals are the flow amounts divided by the average or median flow.

River model: External set of river models updated and extended to provide river flow as input to MICE (this study).

RoWRA: Roper River Water Resource Assessment

SOI: Southern Oscillation Index

Spatial structure: The way in which the MICE was set up to capture spatial differences for species/groups (i.e. sub-models) and associated drivers. In this case, the Guld of Carpentaria was divided into different spatial regions, with a separate MICE ensemble for each region and with some connectivity between sub-models (prawns) across some regions.

Strategic model: A model that aims to capture the ‘big-picture’ and is often used for contextual purposes and direction-setting. See e.g. Plagányi et al (2014).

Stock assessment model: A model that is used to evaluate how a fishery impacts the status and surplus production of a particular fished species to underpin sustainable management decisions. Usually a statistical integrated model that is fitted to fishery and/or survey data and combines likelihood contributions into a single objective function that is then used to estimate reference points and inform tactical management decisions such as setting of a TAC.

Tactical model: A model focused on management actions over short time scales. See e.g. Plagányi et al (2014).

TAC: Total Allowable Catch – in reference to a catch quota set by fisheries management.

TH: Threshold – river flow extraction threshold value (ML/day). Water can only be extracted when river flow surpasses this value.

Qld: Queensland

Validation: the process of fitting a model to data to check the extent to which the model is a plausible or best explanation of the data.

WRD: Water Resource Development – the anthropogenic alteration of natural river flow regimes implemented by allocating water for either extraction or impoundment (i.e. dams) for use in agriculture and/or other industries.

Project materials developed

The project has created the following fact sheets and scientific papers.

Factsheets:

Ecological Modelling of impacts of Water Development, CSIRO Factsheet, 8 pp.

(In prep) Summary Recommendations based on Ecological Modelling of impacts of Water Development, CSIRO Factsheet

Visualisation material:

Animation series to illustrate life history and system dynamics connecting estuarine and freshwater habitats, and impact of flows

Scientific papers:

A number of planned and potential papers will be generated from this project. Below are the first three papers currently being drafted:

Plagányi et al. In prep. Complex impacts on marine ecosystems of river water extraction and damming

Blamey et al. In prep. Modelling environmental and fishery drivers of mud crab population dynamics

Kenyon et al. In prep. Overview of Water Resource Development scenarios potentially influencing Northern Australian marine ecosystems

Appendices

Appendix 1. List of researchers and key collaborators

CSIRO (O&A, St Lucia)

Éva Plagányi (PI)

Rob Kenyon (Co-investigator)

Trevor Hutton (Co-investigator)

Laura Blamey

Roy Deng

Toni Cannard

Margaret Miller

Chris Moeseneder

Richard Pillans

Derek Fulton

Tonya van der Velde

Sean Pascoe

Leo Dutra

Peter Rothlisberg

CSIRO (Data61, ESP)

Emma Lawrence

CSIRO (L&W, Black Mountain)

Justin Hughes

Shaun Kim

Cuan Petheram

Danial Stratford

DAF

Julie Robins (Co-investigator)

Griffiths University

Michele Burford (Co-investigator)

NPFI

Annie Jarrett (Co-investigator)

Adrienne Laird (Co-investigator) (current affiliation: FRDC)

Appendix 2. First Stakeholder Workshop

Workshop Agenda

Ecological Modelling of Impacts of Water Development 2019

Workshop 7-8 August, QBP, St Lucia

Project FRDC/CSIRO: 2018-079 Ecological modelling of the impacts of water development in the Gulf of Carpentaria with particular reference to impacts on the Northern Prawn Fishery

Project Duration: May 2019 -Jan 2021

Principal Investigator: Éva Plagányi (CSIRO)

Co-investigators: Rob Kenyon, Trevor Hutton (CSIRO), Michele Burford (Griffiths University), Julie Robins (QDAF), Annie Jarrett/CEO (NPFI), Adrienne Laird/Industry Liaison Officer (NPFI).

Project Facilitator: Toni Cannard

Wednesday 7 August

9:30 am Welcome, acknowledgement of Traditional Owners and Introductions (Éva)

9:40-10:00 am Background to project (Éva)

10:00- 10:30 am Overview of Water Extraction Plans (Rob)

10:30-11 am Capacity for managing water (presentation/discussion session with input from water managers, local council and other managers)

11-11:30 am Morning tea

11:30-12:30 pm River system modelling inputs (flows based on different scenarios) (Justin, Shaun)

12:30-12:45 pm Brief summary of existing statistical and qualitative model outputs (Broadley et al)

12:45-1:30 pm Lunch

1:30-1:50 pm Discussion of inputs and concerns from Industry (Annie/Adrienne/NPFI rep)

1:50-2:10 pm Discussion of inputs and concerns related to Recreational and Indigenous sectors (Toni to chair)

2:10-2:20 pm Species of conservation concern (Rich)

2:20-2:50 pm Prawns (Rob et al) - what data, existing models, capability? Relationship between production and flow, descriptive models, predictive power, data available

2:50-3:20pm Barramundi (Julie) what data, existing models, capability? Relationship between production and flow, descriptive models, predictive power

3:20-3:40pm Mud crabs (Julie) what data, existing models, capability? Relationship between production and flow, descriptive models, predictive power

3:40-4 pm Next highest priority species and any key trophic interactions?

4-4:30 pm Ecological model overview (Éva)

4:30pm Close – day 1

Thursday 8 August

9-9:30 am Summary of progress on Day 1 and conceptual overview of model (Éva)

9:30-10 am Prawn monitoring indices and relationship to flow (Rob)

10-10:30am Overview of estuarine productivity and drivers with links to prawns (Michele)

10:30am-11 Spatial structure of model

11-11:30am Morning tea

11:30-11:45am Economic considerations (Trevor)

11:45-12:45 Group inputs to conceptual model

12:45-1:30pm Lunch

1:30-3:30pm Group inputs to conceptual model continued

3:30-4pm Summary and next steps (Éva and Co-investigators)

4pm Next meeting and communication plan

4:30pm Meeting close

Workshop Summary

Water Resource Development overview

The workshop commenced with an overview of water resource development (WRD) plans and relevant legislation in tropical northern Australia. The Mitchell, Gilbert and Flinders rivers and their catchments empty into the Gulf of Carpentaria. The topography, soil, water, current infrastructure and cultural dimensions of these catchments have been quantified during the Flinders and Gilbert Agricultural Resource Assessment (FGARA) and the Northern Australia Water Resource Assessment (NAWRA) projects. The research has scoped potential WRD in the tropical Queensland catchments and some scenarios can provide reliable water supply in >70-80% of years (Petheram et al. 2013, Petheram et al. 2018b).

In northern Queensland, specific enterprises currently in-development (e.g. environmental assessment) or recently completed include the Three Rivers Irrigation Project (Flinders River), the Metro Mining (Skardon River) and Amrun Mines (west of Embley River, Weipa) are the projects of focus in Queensland. Stanbroke Pastoral's 'Three Rivers' proposed cropland irrigation plans to extract ~150,000 ML y^{-1} from the lower Flinders River by diversion structure or weir. It has been designated a 'Project of State Significance'.

CSIRO's FGARA assessment mapped and modelled catchment flows and floodplain inundation, as well as mapped soils and undertook case studies of prospective crops, given dimensions of the local climate in the Flinders (109,000 km²) and Gilbert (46,000 km²) River catchments (Dutta et al. 2013, Lerat et al. 2013, Petheram and Yang 2013). In addition, the assessment identified key water issues for the indigenous population in the two catchments and their concerns, preferences and goals for engagement with types of WRD (Barber 2013). The study identified offstream storage (350 GL potential, delivering 175 GL pa) as an optimum method for water storage in the Flinders River catchment (Petheram et al. 2013a) and this is reflected by the commercial investment in the Three River Irrigation Project. In-stream dams were identified as capable of delivering a combined 80 GL of water for irrigation. The total water storage capacity could support 10,000–20,000 ha of year-round irrigation in 70–80% of years. Growing one crop per year would increase the area of land that could be irrigated to ~40,000 ha. The addition of 20,000 ha of irrigated agriculture would increase the irrigated area of northern Australia by ~15%. Water Resource Development would divert 14% of the mean and 28% of the median annual flow from the Flinders River's end-of-system output. Most of the soils in the Flinders River catchment (~8 million ha) were classified as moderately suitable for irrigation (class 3 soils). Class 2 'suitable' soils are limited to ~300,000 ha in the catchment. Both the best soils and moderately suitable soils are limited for production due to distance from the river channel, thus the

much-reduced estimate of 10,000-40,000 ha of suitable soils lying an economically viable distance from the water source.

The FGARA assessment identified instream dams as the most efficient and reliable form of water storage in the Gilbert River catchment. Two sites were located in the catchment for in-stream dams of approximately 500 GL and 220 GL capacity, delivering ~320 and 170 GL of water at 85% reliability (Petheram et al. 2013). The water could support 20,000–30,000 ha of irrigation in 85% of years. The scale of the development is larger than the existing Ord River Irrigation Area and could produce a profit margin of \$60 million yr⁻¹. As the proposed dams were located in the middle reaches of the catchment, Water Resource Development would divert 14% of the mean and 20% of the median annual flow from the Flinders River's end-of-system output. About 2 million ha of moderately suitable soils (Class 3) dominate the area of the Gilbert River catchment. Class 2 'suitable' soils cover only 50,000 to 300,000 ha of the catchment. For both dams, the areas of suitable soils are ~70 km downstream of the dam walls, resulting in ~25% loss of water during delivery downstream. A large proportion of the catchment's class 2 and class 3 soils are too distant from the river to be economically viable. The capital costs of the dams would be about \$1 billion and would need to be funded by centralised capital.

The NAWRA project (Petheram et al. 2018b) assessed the potential of the Mitchell River catchment (72,000 km²) to support irrigated agriculture. It mapped and modelled catchment hydrology, as well as mapped soils and undertook case studies of prospective crops, given the dimensions of the local climate (Petheram et al. 2018a, Taylor et al. 2018). As well, the project undertook a cost/benefit assessment of mosaic water harvesting and a large dam in the Mitchell River catchment and considered legal, regulatory, social and cultural aspects ensuing from irrigation development (Stokes et al. 2017, Macintosh et al. 2018). Specific cultural dimensions of indigenous peoples with country in the catchment were assessed, identifying residential patterns, land management representation, key water issues, water resource rights and development objectives, and priorities and goals for engagement with types of WRD (Barber and Woodward 2018). Four cost-effective dams in the Mitchell River catchment have been identified as capable of storing 2800 GL of water in 85% of years for irrigation use. 140,000 ha of irrigated agriculture could be supported. Offstream storage using ringtanks and a mosaic of irrigated lands could deliver 2000 GL of water with 85% reliability. As well, the Mitchell River catchment assessment was scoped possible aquaculture development associated with irrigation infrastructure or coastal water resources.

Of the 2800 GL of WRD, 65% could be delivered by two potential dams, the Pinnacles dam site on the Mitchell River (2316 GL capacity) and Rookwood dam site on the Walsh River (1288 GL storage). These would yield 1248 GL and 575 GL, respectively, to agriculture in 85% of years. Case studies match soils with selected crops and suggest that irrigated cropping could generate \$720 million annual

income, though some crop/soil combinations were less lucrative. Water Resource Development would divert about 22% of the mean and 24% of the median annual flow from the Mitchell River. About 3 million ha of the soils in the Mitchell River catchment are classified as class 3 moderately suitable soils, though the type of crop grown and the method of irrigation reduces the potential viable area of soil. About 300,000 ha of soils are the better 'suitable' class 2 soils. A large proportion of the catchment's class 2 and class 3 soils are too distant from the river to be economically viable. The capital cost of the dams would be about \$2.75 billion. The NAWRA report outlines the provision of reliable year-round water for irrigated agriculture in the Mitchell River catchment, but Stokes et al. (2017) suggest that the achievement of favourable long-term economic returns from agriculture in these remote regions will require rigorous consideration and planning.

The different water acts and water resource plans are summarised in Appendix 3 of the first workshop report and is available on request.

The project is committed to providing recommendations that can support water planning decisions. Hence the workshop participants discussed risk assessment approaches used to assess environmental flow regimes, and in particular the risk-based ecohydrological approach adopted in Queensland (McGregor et al. 2018). Bill Senior (Qld DES) explained the process includes measurement endpoint (often these are a suitable surrogate) linked to threshold of concern; these become the ecohydrological rules. The assessment is focussed on a particular ecological asset and linked to the life history of the asset, e.g. barramundi. The year class strength (YCS) of barramundi is assembled and this creates the YCS index. It was noted that there is a need to revise the barramundi model as all parts of flow regime are important. Another important gap identified was the need to define a threshold of concern for banana prawns. Finally, it will be important to consider differences between NT and Qld.

Gareth Jones (DAFF northern regional office) explained there is good data to understand the impacts of agricultural development and previous projects have also provided extraction limits information. There are flow past points from which water cannot be taken unless sufficient flows have already gone past that point.

The background material, presentations (shared via Dropbox and as Appendix 3) and workshop discussions presented a lot of detailed information on prawns, barramundi and crabs that will be used to inform development of the MICE. The exact use and specification of population dynamics and interactions within the model will be described in future reports. The quantity and quality of data available to inform on trends in the different fishery sectors was discussed.

Barramundi fishing occurs mostly out of Burketown, Karumba and Weipa, otherwise fishermen must travel long distances to catch fish. Barramundi caught is often in the 3-7 years age class, but can live to approximately 22 years. Barramundi uses freshwater and estuaries with monitoring data available from

2000 and standardised by 2006. Analysis of otoliths has shown that fish growth slows in winter. Microchemistry using Strontium isotopes is used on the otoliths. Barramundi use freshwater and associated habitats opportunistically (i.e., non-obligatory). Whilst a certain proportion of the population appear to spend their entire life in estuarine conditions, the population does benefit from access to or the consequence of river flows to estuaries. The details of the link between population benefit and flows is still being resolved (e.g., upstream access, increased growth, increased survival).

The prawn presentations stressed that river flow and catch certainly affect post-larvae because of the offshore life cycle. Flows can influence how many spawners get back into the estuarine reaches. Spawning occurs in the dry season and there are two opportunities: the first if the early rain is low enough for post-larval emigration and offshore exchange, then 2 weeks afterwards they move into the estuary based on environmental cues, namely the pressure on the prawn due to the amount of water above it. Once sensed this cue triggers prawn behaviour to move vertically upwards into the faster velocity waters and be 'taken' inshore on the currents. The prawns are prolific egg layers, producing 2-3 million eggs per year.

Andrew Broadley (Griffiths University) presented a brief summary of existing statistical and qualitative model outputs, relating offshore prawn production to a sub-set of catchments. Michele Burford provided a summary of estuarine productivity and drivers with links (for more details refer to 2 page summary in Appendix 3 in workshop report – available on request). The importance of the salt flats and the coastal zone to the overall productivity of the system was stressed. It was noted polychaetes might be an interesting indicator species since they are the extreme survivors. One of the main conclusions is that there is not a single one of the river systems (Mitchell, Flinders, Gilbert) that is more important than the other two in contributing to fisheries productivity. The three rivers function as a wider ecological system.

The Flinders upper reaches experiences increased evaporation but all GoC rivers systems have relatively low rainfall for many months of the year and then some wet seasons are 'wetter' than others. Major flooding events turn the rivers into one major system in which coastal tidal banks are submerged and the floodwaters create an extensive flood plume visible from the air. It was also noted that Gilbert River has less monitoring and a complex system of breakouts during high flood flows.

In terms of technical interactions, it was discussed that the following are worth considering:

1. banana prawn and tiger prawn fishery interactions;
2. some development scenarios need a feedback loop to recreational fishing;
3. sawfish – fishery interactions.

It was decided to include both species of sawfish given that they are differentially affected by river flows and fishery impacts. The sawfish presentation highlighted the importance of river flows and access to upstream environments for juvenile Largetooth sawfish. Nursery areas (up rivers) provide more food and less predation for smaller sawfish. Entrapment of sawfish in isolated stretches can result in increased predation. Both crocodile and bull sharks are known to prey on sawfish within rivers. There is some known avoidance of bull sharks through movement behaviour. Once the sawfish grow to around 3 metres they then move downstream.

The West GoC mud crab fishery catch collapsed in 2016 (Grubert 2018), noting NT allows female crabs to be taken whereas Qld does not. There have been considerable price changes from \$8/kg in 2000 to now approximately \$31/kg. Increases in catches over the last 10 years are most likely to primarily be driven by increasing in fishing effort.

Model outputs (in the form of biological, economic, ecosystem, socio-cultural trade-offs) will thus be aligned with water resource management frameworks in terms of any outcomes as to how to improve trade-offs under alternative scenarios. The project also aims to quantify trade-offs to inform understanding, improve transparency and as a basis for discussions and management decisions. In addition, use of a dynamic framework encompassing non-linear and indirect interactions will facilitate identifying high risk scenarios / situations that may require additional action.

Some key model Performance Statistics were identified and included:

- Median projected biomass of species and risk of depletion below reference points
- Median projected average catch by different sectors
- Economic performance statistics (direct impacts)
- Ecosystem integrity index
- Spatial performance statistics – to discuss impacts of changes in spatial availability of resources/species
- How to capture non-extractive values; indigenous aspirations, social and cultural impacts

Fishery sectors

Discussion focused on the major fishery sectors that rely on catches of prawns, barramundi and crabs. But there was also discussion around considerations pertaining to indigenous and recreational fisheries. Concerns around lack of indigenous engagement prior to decision-making were raised and also discussion of certain sacred places where ceremonies are conducted are believed to increase fishing success e.g. barramundi, and ‘mermaid place’. Fishing industry representatives shared the way that they have sought engagement at various levels of government to trigger projects such as this current one. There are long-standing measures in the Gulf and particularly southern Gulf to reduce

sector conflict - as an example - commercial fisheries are closed on all rivers – see Department of Employment, Economic Development and Innovation, 2010²). Sectors have worked collaboratively on bycatch measures across all the fisheries (Indigenous, recreational and commercial) (see (QLD 2004). Recreational fishers targeting high end species (valuable as trophy fish) have attempted to engage with local indigenous clans to offer mixed packages (catch fish and cultural – indigenous experience). Recent policy initiatives by Qld state government which governs the southern end of the Gulf have more openly supported indigenous fishing: that is “Maintain Aboriginal peoples and Torres Strait Islanders’ access for traditional fishing - Access to traditional fishing is important to many Indigenous fishers as a way of remaining connected to culture and providing a source of food. The purpose of this objective is to ensure that Aboriginal peoples and Torres Strait Islanders’ access to fisheries resources is recognised in Queensland and Indigenous communities are involved in the sustainable management of fisheries” (Queensland Department of Agriculture and Fisheries – Sustainable Fisheries Strategy 2017-2027) (State of Queensland 2017). This has led to transfer of rights to catch mud crab from non-indigenous commercial fishers to indigenous commercial fishers; potentially putting this “new” sector in conflict with recreational fishers targeting this species. However, there is continual greater recognition of indigenous rights, with participation by indigenous representatives mandated on state and federal management committees. Documentation and publication of historical indigenous fishing is assisting with the knowledge flow about their rights which potentially reduces conflict between sectors as it clearly establishes people’s rights (National Oceans Office 2004)..

Recreational fishers are aware of and support efforts to ensure good environmental flows so that they have high confidence that when they go out to fish, they will get a catch. The value of being able to enjoy catching a fish is very different to the table price of a fish. In other words, a decrease in ability to catch a fish can induce a larger than expected decrease in value. Recreational fishing is a considerable source of socioeconomic value to towns like Karumba where fishers also target grey and Spanish mackerel as well as Barramundi.

Additional discussions were held about the higher value attached to the ‘wild’ charter fishing experience as compared to the general recreational fishing community. The value in this case is a combination of the enjoyment of fishing and experiencing fishing in these Wild areas of GoC. Reference was made fishers targeting trophy fish versus finfish.

Some other questions and issues raised by participants included:

² Department of Employment, Economic Development and Innovation, 2010. Management arrangements for the Gulf of Carpentaria Inshore Fin Fish Fishery. Regulatory Impact Statement.

- Are downstream ecological issues related to extremely high flood flows and velocities (e.g. stream and riverbed scouring, riparian vegetation impacts, erosion resulting in very high total suspended solid (TSS) and nutrient loads) that can result from large dam releases e.g. Brisbane 2011 floods example.
- Will model incorporate tendency for dams to increase water permanence in area local to dam?
- Could lessons be learnt from the Ord development, e.g. impacts on environmental flows, fishing (all sectors), etc.
- Will emigration blockages result from either additional localised water harvesting and dams?
- How to avoid a Murray-Darling situation again? Is it too risky based on that experience which is still unfolding?
- Technology to gauge water extraction may be part of failure in Murray-Darling, how could that be explored or guarded against? Thought to be outside the scope of this project.
- Discussion about mine impacts and examples of MacArthur River mine diverting water and effect on fishing in the Vanderlins.
- Weipa mine impacts and the potential implications
- Fishing industry representatives reported extensive river side clearing in Gilbert River 2-3 years ago with multiple D9 dozers, and impacts of widespread riparian vegetation, e.g. erosion, sediments.

In summary, participants agreed on choice of the key species whilst acknowledging that it's not going to be easy to model all aspects, especially as there are limited high quality data on abundance trends. The importance of mangroves in contributing to blue carbon (Alongi 2012, Kelleway et al. 2016) was recognised. The workshop group expressed several common concerns regarding WRD and the need to model a range of potential options plus consider flexibility to rapidly inform on new developments. Discussions confirmed that model outputs could be aligned with the risk-based ecohydrological approach to assessing environmental flow regimes.

List of Workshop 1 participants

Name	Organisation	Attendance	In person/webex
Éva Plagányi	CSIRO Oceans & Atmosphere	Whole workshop	In person
Rob Kenyon	CSIRO Oceans & Atmosphere	Whole workshop	In person
Trevor Hutton	CSIRO Oceans & Atmosphere	Whole workshop	In person
Michele Burford	Griffith University	Day 2 only	In person
Julie Robins	QDAF	Whole workshop	In person
Tonya van der Velde	CSIRO Oceans & Atmosphere	Day 2 only	In person
Zoe Williams	Northern Gulf Resource Management Group	Day 2 only	In person
Toni Cannard	CSIRO Oceans & Atmosphere	Whole workshop	In person
Stephen Faggotter	Griffith University	Day 1 only	In person
Stephen Eves	AFMA	Whole workshop	In person
Shaun Kim	CSIRO Land & Water	Whole workshop	Webex
Roy Deng	CSIRO Oceans & Atmosphere	Whole workshop	In person
Rik Buckworth	NT Fisheries	Whole workshop	Webex
Richard Pillans	CSIRO Oceans & Atmosphere	PM sessions both days	Webex
Phil Robson	Raptis	Whole workshop	In person
Marg Miller	CSIRO Oceans & Atmosphere	Whole workshop	In person
Leo Dutra	CSIRO	Whole workshop	In person
Laura Blamey	CSIRO Oceans & Atmosphere	Whole workshop	In person
Justin Hughes	CSIRO Land & Water	Whole workshop	Webex
John Clark	Kowanyama Aboriginal Land and Natural Resource Management Office	Whole workshop	In person
Ian Boot	Austfish	Whole workshop	In person
Gareth Jones	DAFF	Day 2 only	Webex
Emma Lawrence	CSIRO Oceans & Atmosphere	Whole workshop	In person
Danial Stratford	CSIRO Land & Water	Whole workshop	In person
Chris Moeseneder	CSIRO Oceans & Atmosphere	Day 2 only	In person
Bryan Van Wyk	Austral Fisheries	Whole workshop	In person
Bill Senior	Qld DES	Whole workshop	In person
Annie Jarrett	NPF Industry Pty Ltd	Whole workshop	In person
Andy Prendergast	Austral Fisheries	Whole workshop	In person
Andrew Broadley	Griffith University	Whole workshop	In person
Adrienne Laird	NPF Industry Pty Ltd	Whole workshop	In person
Mark & Julianne Grunske	Mud Crabs Direct	Apologies	
Adam West	QDAF	Apologies	
Alison King	Charles Darwin University	Apologies	
Carmel Pollino	CSIRO/NAWRA	Apologies	
Claudine Ward	Commercial fisher	Apologies	
Cuan Petheram	CSIRO/NAWRA	Apologies	
David Power	AFMA	Apologies	
Desmond Yin Foo	Water Resources, Northern Territory, Environment and Natural Resources (DENR)	Apologies	
Gavin New	Carpentaria Barra & Sport Fishing Charters	Apologies	
Glenn McGregor	QDES	Apologies	
Jed Matz	CRCNA CEO	Apologies	
Jo & Belinda Berwick	Eclipse FNQ Charters	Apologies	
Jonathan Marshall	QDES	Apologies	
Mardi Miles	DENR	Apologies	
Peter Kyne	Charles Darwin University, RIEL	Apologies	

Rolfe Ellem	Etheridge Shire Council	Apologies
Stephenie Hogan	DNRM	Apologies
Steve Farnsworth	Recreational fisher	Apologies
Thor Saunders	NT Fisheries	Apologies

Appendix 3 – Workshop Teleconferences

Prawn Workshop

WORKSHOP AGENDA: 24 February 2021

Stakeholders via teleconference, 9:30-12:30 AEST

Project Team meeting afterwards until 2pm

Kindly note the meeting will be recorded for internal project notetaking purposes only, but please let us know if you have any concerns

9:30 Welcome and Introductions

9:40 Brief overview of project followed by presentation focused on prawn component: development of spatial, multispecies model, fitting to data, linking environmental drivers and quantifying impact of changes in flows (Éva Plagányi)

10:25 5 minute break

10:30 Questions, comments and discussion re prawn modelling

11:20 10 minute break

11:30 Brief update on recent NESP prawn research (Michele Burford)

11:45 Update re WRD scenarios to be tested in model (Rob Kenyon, Justin Hughes, Shaun Kim)

12:00 Next steps re project teleconferences, workshop and finalising report (Éva)

Key Objectives

The MICE model is the first model to specify a multi-species prawn model that is spatial, weekly and related to key environmental variables and the key objective of the meeting was to obtain peer review from project scientists and stakeholders (see Text Box 1 for list of participants) on the model fits; that is the observed abundance indices versus model estimates of biomass. Additional detail points to make about the meeting are as follows:

- Presentation of food web (with tiger prawns, banana prawns, sawfish, barramundi, mud crabs, and seagrass and mangroves) and assumptions and role of meiofauna
- Preliminary results of banana prawn biomass based on flow modelling
- Summary provided of main technical interactions across fishery and possible inclusion of economic parameters and estimation of economic impact
- Review presented of alternative WRD scenarios (presentation provided by Flow Modeller)

Text Box 1 –Teleconference 1 participant list

Zoe Williams	Northern Gulf Resource Mgmt
Toni Cannard	CSIRO O&A
Stephen Mackay	Water Manager
Rik Buckworth	NT Fisheries
Leo Dutra	CSIRO O&A
Marg Miller	CSIRO O&A
Justin Hughes	CSIRO Land & Water
Emma Lawrence	CSIRO O&A
Danial Stratford	CSIRO Land & Water
Chris Moeseneder	CSIRO O&A
Bill Senior	Qld DES
Annie Jarrett	NPF Industry Pty Ltd
Andy Prendergast	Austral Fisheries
Adrienne Laird	NPF Industry Pty Ltd
Darci Wallis	AFMA
Sean Pascoe	CSIRO
Peter Rothlisberg	CSIRO
Ryan Downie	CSIRO
Bill Senior	DNRME
Michaelie Pollard	DNRME
Amelia Desbiens	CSIRO
Rodrigo Bustamante	CSIRO
Stephen Eves	AFMA
Shije Zhou	CSIRO
Phil Robson	Raptis
Bryan Van Wyk	Austral Fisheries

Multi-species model has 8 spatial areas, freshwater, estuary and offshore components as well with a 1 week time step. Relationship included of salinity and flow and flow – meiofauna growth rates, and biomass equations for microphytobenthos and meiofauna. Floods are shown to drive offshore productivity and catch of common banana prawns and evidence thereof presented over full history of fishery. Model for common banana prawns has three periods with difference selectivity. Plots shown of fit of monitoring survey to model biomass for tiger prawns.

Recruitment residual for common banana prawns linked directly to flow of main river flow associated with each spatial unit via an equation that can be parameterised (and an interaction flow – that occurs between spatial units offshore). This fitted to offshore catch data (that includes selectivity, effort and fishing power and biomass based on complex flow equation). Option included for an offshore productivity effect, that improves the fits of model and catch (with many R-squared values in the region of 0.75). Presentation of contribution of each main catchment shown relative to offshore region. Main result indicates combination of effects of each river act as a portfolio like situation which each contributing differently over time, maintaining some stability in the system; yet highly vulnerable to impacts if they are constant (low flow for many years) or geographically consistent (over more than one catchment). Catches associated to some catchments could be reduced by up to half.

Outcomes of Meeting

Both scientists and stakeholders were impressed with the model fits and were able to ask important questions and gain further clarity on the methodology of how the parameters (of movement of prawns between spatial units) were estimated.

Questions

Question - Scientist. How does one estimate or calculate linkages and connections between the areas (four or five rivers) and does the strength of those connections vary from year to year (based on flow and other factors).

- Where prawns originate is based on mixing, however considering relationship which considers probability of which was source, and it is estimated over all the data from 1970 to 2018, for each week, year, area and the parameters (a, b, c, d) are those estimated to be explain observed catches. It is integrated estimate/ average estimate. Varies by year. Considers flow in each flow from each source. Users all available data. Contributes to recruitment and thus catches, estimates contributions. Similar to stock assessment that capture net effect and matches observed data.

Question – Scientist. What the process that affect that mixing and the ‘reach’ of each river, and actual connections.

- Need more information to understand underlying biological processes. We have a combination of statistical model and mechanistic model. Would require hydrodynamics – but could in future use model to explore hypotheses. We need to trade-off against not over-parameterising. We have non-linear effects due to recruitment assumptions and logistic relationships, and extra effect of productivity, and fishing effort – which are all integrated into the model.

Comment – Scientist. Could talk about offshore productivity in Albatross Bay and offshore larvae survival.

- Is on list of processes to potential capture, taking into account how to deal with incorporation of this in terms of tractably.

Question – Scientist. Fitting to banana catch data looks quite good. Is the model basically a single species model + one environmental variable (river flow)? No interaction with other species (mud crab, sawfish, meiofauna, etc)?

- Tiger prawn, mud crabs and barramundi. Predation by barramundi on prawns to be included (if it is justified) and technical interactions.
- Integration is step wise
- Will show at next workshop impacts of flow change on mud-crabs

Question - Industry. Any comments/considerations for banana prawn contributions to our fishery from other rivers? Particularly in the Vanderlin areas (Calvert, Robinson, McArthur, Rose and Limen Bight). Is any of this accounted for in the models? Plus, we would like to know about smaller rivers. We assume Roper is a “region”.

- Lack of data for smaller rivers – thus model capturing intermediate complexity. Assume small river near large rivers where data exists – are capturing background effects. Have to work with data we have.
- Roper is a proxy for region.
- Slides will be made available / taking into account this is all preliminary.

Question – Scientist and clarification sort from Industry. Is flow the total annual flow? Is timing important? Assume flow volume in, or degrees – low, medium and high. Clarification is flow by week, (?) and consider potential time lags and which time lag is best (?)

- The flow is daily in each river system, - model users cumulated weekly flow. The timing is critically important. Takes into account spawning timing.
- Has absolute flow, and average flow anomalies. Model can produce any flow statistics that are required. Flow is by week, year, area. Considers salinity, prawn recruitment, fishing selectivity. Includes mortality and population dynamics – as per stock assessment model. Estimate parameters within recruitment residual equation, with no explicit error term. Model has flexibility.

Question/Comment – Industry (after prompting by Eva about spatial impacts on banana prawns). Dam on river like Flinders would be major catastrophe for NPF, and reason is NPF is set up that if you take away that catch, you decrease the viability of the season and if the first season is not viable it puts pressure on the second season as more resources are put into that, getting the boats ready. Normally it can compensate when not ok in one system one year, then ok the next year, with compensation from other river systems. However, if catch reduced permanently in one system and then low flows in others – it will have a big impact. Government would have to buy out 50% of effort to compensate.

- Modelling is showing that flow-on effects. Preliminary economic analyses show impacts on banana prawn season are large, as it provides catch that essentially keeps fishing viable. Greater proportional impact as fixed costs have to be met. Below catch of 1400 t (preliminary estimate) the fishery is not viable. Could run model with reduced fixed costs (reduce number of boats).

Question/Comment - Industry. As well as economic impacts, we know from past experience that if we shift a lot of effort from bananas to tigers in the first season would have a disastrous impact on the productivity of the tiger prawn fishery

- Work in progress is to consider how it could impact on Tiger prawns as technical interactions.

Question/Comment – Scientist/Project participant. Difference between dams (in-stream) and water harvesting (pumping and off-stream). Cannot put dent in big flows with pumps. Maybe two scenarios where one is impact of in-stream dams and other where we have off-stream impacts.

- Other studies use total dam volume as an extraction scenario. Ok, if testing model, however more variable than this. Lot of interest in Flinders, couple on Gilbert. Three large projects in Mitchell. However, one is up-stream. The large project – the Pinnacles, is not likely. We must consider this (see his presentation for detail for *in system requirement*, and future climate impacts).
- Unsure of process of selecting scenarios, possibility is a scenario with one dam in each catchment; and/or a scenario or set of scenarios with off-stream storage in all. Can be varied in Mitchell (the in-stream and off-stream options). Require finite list of scenarios so can interpret outputs. Be good to have bounded extremes – but have most like scenarios. Can run model for others and have in background. Model can be re-run, but good to have set now. Many possibilities but require sensible combinations. What are the ways we can reduce the impacts, by mitigation measures or triggers (as a separate analysis). Potentially many scenarios. Climate impact is important scenario to run; as a bound to potential impact.

Question/Comment – Scientist/Project participant. State government should provide likely scenarios. Studies that consider the top of catchment scenarios are not able to easily consider impact, as claims impacts are minimal.

- It is question of water balance and where it rains over large catchments.

Question/Comment – Government. What about 1000GL extractions and how does timing of the extraction impact on any flow in a system.

- Require large events to get this amount, and low years mean cannot extract amount and extraction is halted

Question/Comment – Scientist. Good to speak to Queensland hydrology experts. They get input data and water allocations, and thus capture likely scenarios and what is planned.

- Models use existing users and extraction; but good suggestion to consider.

Question/Comment - Industry. How are scenarios going to be chosen?

- Scenarios were listed and being circulated and given suggestions today – range of pumping rates. Be good to get feedback from Canberra experts.
- Run range and begin testing now; and see what is likely; and how it fits in range. Model does run quick – but set needs to be tractable and what set can inform our thinking and importantly mitigation scenarios.
- Is it going to a case of one river having many projects, or a project on each river. We need to consider what is in ‘pipeline’, which ones have gone to feasibility studies and which ones the ‘soil has been turned’.
- Next few weeks/months going to consider impacts on mud crabs, barramundi. Runs on each, and need to consider all the performance statistics
- Evaporation is a critical factor to consider and has large impact on water balance. There are plans to irrigate in some years and low years no crop. Feasibility studies are considering the variability.

Question/Comment – Scientist/Project participant. Do consider uncertainty in models but end up with lots of data. Our studies are pre-feasibility.

Question/Comment Industry. Concerned about impact on other stocks and impact on whole ecosystem.

Question/Comment – Industry Main Representative. Thank you to Project Lead and all the presenters - it was a really good session and very informative. Excellent work everyone.

Mud crab Workshop

WORKSHOP AGENDA: 23 March 2021

Stakeholders via teleconference, 9:30-12:30 AEST

Kindly note the meeting will be recorded for internal project notetaking purposes only, but please let us know if you have any concerns

9:30 Welcome and Introductions (*Éva Plagányi*)

9:45 Brief overview of mud crabs (*Julie Robins*)

10:00 Overview of approach for modelling GoC mud crab dynamics (*Laura Blamey*)

10:25 5 minute break

10:30 Mud crabs continued – linking with environmental drivers (*Laura Blamey*)

10:50 Questions and Discussion

11:20 10 min break

11:30 Update re modelling of mangrove and seagrass systems (*Éva Plagányi*)

11:45 Questions and Discussion

12:00 Next steps re project and opportunity for further questions, discussions, suggestions (*Éva & Laura*)

12:30 Stakeholder Workshop Close

Key Objectives

The meeting was held to review mud crab ecology and modelling thus two presentations were provided one on each – ecology and modelling. This is the first model to include mud crabs based on stock assessment primary data to be included in a MICE model; and related to key environmental variables and the key objective of the meeting was to obtain peer review from project scientists and stakeholders (see Text Box 2 for list of participants) on the model fits; that is the observed abundance indices versus model estimates of biomass. Additional detail points to make about the meeting are as follows:

- Presentation of mud crab life cycle; use of habitat and movement
- Summary presented of the NT and Qld fisheries (and summary of project FRDC 2017/047)
- Model structure – 8 regions, monthly time step with 3 age-classes, with Length-Weight relationship in for each sex, and assumption sexual maturity (1.5 years approx.) and assumption of spawning during summer
- Provided summary of flow and recruitment assumed linkage
- A presentation was also provided by Rob Kenyon on cyclones and effects on the seagrass and a presentation by Eva Plaganyi on potential modelling of mangroves

Text Box 2 – Teleconference 2 participant list

Eva Plaganyi	CSIRO O&A
Trevor Hutton	CSIRO O&A
Julie Robins	QDAF
Laura Blamey	CSIRO O&A
Rob Kenyon	CSIRO O&A
Emma Lawrence	CSIRO Data61
Richard Pillans	CSIRO O&A
Peter Rothlisberg	CSIRO O&A
Jacob Rogers	PhD candidate
Rodrigo Bustamante	CSIRO O&A
Linda Merrin	CSIRO L&W
Rik Buckworth	Sea Sense Australia
Michele Burford	Griffith University

Outcomes of Meeting

Both scientists and stakeholders were impressed with the model fits and were able to ask important questions and gain further clarity on the methodology and suggest modifications for contribution of rivers in region 7 and 8; leading to change in assumption applied model.

Questions:

First presentation by Julie Robins

- Scientist asked about number of species of mud crab. Julie clarified only one dominates catch – as only that one is the one that gets to legal size (other do not confound catch data)
- Scientist noted that two species appear in catches in Darwin area – and the rarer species mostly in the recreational catches
- Scientist mentioned hydrodynamics and offshore water movement and asked whether the assumption that all life stages are going to the bottom is being made as it affects the predictions of the models
- There was also mention of work by John Church and others that showed change in sea level on different side of the Gulf due to dominant winds. Higher on Eastern side with North winds blow and higher tides on Western side when SE winds blow

Second presentation by Laura Blamey

- Biomass two regions – looks very high; is it flow or other factors. Flow is impacting model recruitment. If reduced does not support high catches. Julie noted that areas are not of equal size. Made point later that model boundary is different than catch reported boundaries (i.e. blue mud bay is in region 8). Could modify catch in model; so it links to that spatial resolution; or at least acknowledge that is why there is a difference. Walters said 80-90% of biomass removed. With such a competitive species; remove animals results in higher turnover.
- Also - a monthly time step in model and F cap on month is 1; but will calibrate further. F should be higher in NT regions; maybe need to sort out availability vector. Massive catch year in 2000, large rain and flow. Fishery was developing at that stage as well. Really fished hard. Did not catch to same extent in 2003. Not catch that magnitude again due to size restrictions.

- Let's consider dome shaped flow/recruitment relationship. Should this be different per region, and the slopes of run-off per catchment. How does this relationship shift to the left or the right? Mitchell and Flinders are very different.
- Comment from Julie; need to think about where juvenile habitat is in each system. Crabs don't come out of river; they are coming from offshore. Not sure if this applies to Roper. Scientist - need to know extent of juvenile habitat. They are obligatory estuarine habitat individuals. How do post-larvae get into that habitat; and flow pushes them out. This may not be same for mud crabs as it is like that for prawns (point made by Scientist).
- I am interested that you picked a Feb/Mar birthdate. Based on spawning in Oct? Answer – November and December in model. I suggest shift it back a bit. Larvae need higher salinity water and may fix your two-year old problem. Model has limited spawning throughout the year which helps. Fishermen note that there are small crabs in pots all the time and have to align observations of large numbers of juveniles with model. End of dry season is the time. They disappear in NT in November and December. Eva mentioned it could be refined; as focus is not on Roper; but can be extended after.
- Air temp talked about; and have three regions or could use heat index. Also trying to model mangroves and they are sensitive to air temp; and link mangroves to biomass of mud-crabs. Julie – they are linked, but is it about leave cover is it about algae cover on root structure. What are the mechanisms? If mangroves decrease; will we see a decrease in mud-crabs? Still working on it. Most of mangrove die-back died on fringe and very small part of mangrove extent in Gulf.
- Michele Burford made the comment, that we don't have a food limited system; and not seen any evidence that we have a food limited system (for e.g. shorebirds). Coastal mangroves not impacted by flow.
- Last point was about mud-crabs spending time in mud and being able to spend a long time buried in mud.
- Last point – look at heat stress on western side. May need a parabolic relationship with an optimal.

Third presentation by Rob Kenyon and then Eva Plaganyi

- Cyclones affect many factors of the benthos. Large amounts of 'prawn killers' found after Cyclone event. Answer: we have seagrass "mortality" after Cyclone and is quantified as Cyclone Intensity versus amount of mortality.
- Rob Kenyon – does it cross coast and runs parallel – will affect very different amount of hectares of seagrass.
- Discussed MICE slot on NPRAG meeting.
- Emma Lawrence mentioned – that can discuss in Cairns about seagrass mortality.
- Linda Merrin – have you looked at impact of rainfall from Cyclone. Question to Rob, nothing in remote sensing. It is in their notes on BOM site. Made comments about spatial aspects of NAWRA work.
- Scientist – Indian Ocean Dipole? Have not thought that far yet. It does affect barramundi growth. It can affect wind speed and direction in Gulf. Gulf is a basin, and could be a negative or positive effect, and that tilts the sea-level in the Gulf.

Barramundi Workshop

WORKSHOP AGENDA: 28 April 2021

via teleconference, 9:30-12:30 AEST

Kindly note the meeting will be recorded for internal project notetaking purposes only, but please let us know if you have any concerns

9:30 Welcome and Introductions (Éva Plagányi)

9:45 Brief overview of barramundi (Julie Robins)

10:10 Overview of approach for modelling GoC barramundi (Éva Plagányi)

10:30 10 minute break

10:40 Barramundi continued – linking with environmental drivers (Éva Plagányi)

11:00 Questions and Discussion

11:30 10 min break

11:40 Update re modelling of other model components (Éva Plagányi, Laura Blamey, Rob Kenyon)

12:00 Next steps re project and opportunity for further questions, discussions, suggestions (All)

12:30 Stakeholder Workshop Close

Key Objectives

The meeting was held to review barramundi ecology and modelling thus two presentations were provided one on each – ecology and modelling. This is the first model to include barramundi based on stock assessment primary data to be included in a MICE model; and related to key environmental variables and the key objective of the meeting was to obtain peer review from project scientists and stakeholders (see Text Box 2 for list of participants) on assumptions and options for modelling. Additional detail points to make about the meeting are as follows:

- Presentation of barramundi stock a structure and life cycle; use of habitat and movement
- Summary presented of the NT and Qld fisheries
- Provided summary of flow and barramundi growth assumed linkage (observed and hypothesis)
- Summary provided of science behind otolith microchemistry
- In GOC, Mitchell, Gilbert and Flinders are key rivers for barramundi production
- Modelling of barramundi considers predation, and is a delay difference model; with eight regions, includes historic catch data, size limits, 20 year plus group, monthly time step, combined sexes (with male-female cross-over), modified Beverton and Holt recruitment relationship, selectivity assumptions
- Illustrative results of model provided for discussion

- Provided summary of flow and barramundi growth assumed linkage (here model assumption based in pint above)
- Additional data noted and being included.

Text Box 3 – Teleconference 3 participant list

Eva Plaganyi	CSIRO O&A
Trevor Hutton	CSIRO O&A
Julie Robins	QDAF
Laura Blamey	CSIRO O&A
Rob Kenyon	CSIRO O&A
Toni Cannard	CSIRO O&A
Michele Burford	Griffith University
Emma Lawrence	CSIRO Data61
Sean Pascoe	CSIRO O&A
Keller Kopf	Charles Darwin University
Jacob Rogers	PhD candidate
Ryan Downie	CSIRO O&A Hobart
	Qld Department of Regional Development,
Michaelia Pollard	Manufacturing and Water

Outcomes of Meeting

Both scientists and stakeholders were impressed with the presentations and progress made with the modelling of barramundi in the MICE model. Peer review was extensive and included good suggestions for modifications which have been taken into account.

Questions

Leo: Could sea level rise drive creation of new habitat for barramundi or is this unrealistic to expect?
E.g. in next 50yrs

Julie: Qld fisheries not looking that far in future, usually only in 5-10 years. Hard to predict but could be potential for expansion of wetlands.

Sean: In terms of recreational fisheries, are the fishers barramundi specialists or do they catch variety species?

Julie - Inshore net fishery, so will target what is available and what brings in money. At start of year everyone targeting barramundi (season just opened), but later in the year will target threadfin or black jew (not by net fish though), will also take blue threadfin – bladders bring in good money. Some will also put in crab pots but difficult to work multiple gears. Few people have private contracts, lots of fishers sell locally – all domestic. Fishers come from Weipa and Karumba – not very many people fishing out of Weipa. Karumba is the main port and most fishers live there. They may go away during the wet, but mostly local (you have to be set up for it). On NT side, they mostly net fishers based locally although new license has effort coming out of Darwin (steaming round). Most are local gulf fishers – need to know the system well as very different to East Coast.

Rec fishers – charter fishers out of Weipa or Karumba (not many). Most recs located around Karumba because further you move away from this area, it gets logistically challenging. Most recs operate within the rivers or coastal creeks – coastal foreshore can have challenging weather conditions.

East coast: 60%-40% for comm-rec catch

Gulf: about 17% of catch – but maybe not that big anymore.

Eva – we do want to capture rec fishery value to see how this may be impacted by flow?

Julie: rec effort likely to be correlated to comm effort depending on whether it's a good/bad year. Rec effort could radiate from Karuma with greatest effort there and then diminishes further out.

Sean: would be great to get rec fishery values

Indigenous fishery: – different for different regions – some communities have strong association with foreshores, but others don't and more on the river. Julie not certain on these and would need more info from those who have better knowledge. Eva: seems unlikely we will capture this – maybe a stretch objective / gap that needs more data.

Eva – Barramundi Model:

Need clarity on size limit – appears NT doesn't have upper size limit

Region 1 does have very small catches (Julie confirmed) – also remember that model regions are different sizes, so would expect that catches are larger than in larger regions.

Extent of depletion at start of model – were many of the large fish depleted at start of model in 1989?

Julie: fishery was different in the 1970s – family boats, fillet fishery. Medium sized fish were probably easier to handle – large fish difficult to handle and fillets tougher. But because upper size limit came in, there must have been pressure on larger fish. Depletion level of 0.2 was set across all ages/sizes.

Probably set depletion level as was done in second stock assessment. This allowed model to fit and, with the upper size limit, allowed stock to recover.

NT not as depleted as Qld.

Julie: have size at sex data. Eva want to fix proportion that are male and female

Sex ratio and spawning:

Model has knife-edge protandry from male to female but could re-visit this if data to inform?

Julie: would be better to have probability distribution for sex change e.g. proportion that are male vs female (at the moment it is very black and white – which is unlikely to be the case). Eva – could maybe add some variability around ages 4 and 6 as well. Want to try and keep it simple and not get too complicated.

Rodrigo – do you care about the male vs female proportions for this model?

Eva – captures BOFF females using average mass which then captures larger biomass of females contributing more to recruitment.

Maturity at age: Julie says that mature males switch to fully mature females. Shouldn't have proportion of mature females less than 1. Julie can send data so that can work out proportion of mature male at various ages and thus females.

Barramundi spawn in groups – may have a number of males with lesser number of females in spawning groups. Eva – can't find any evidence that sex ratio is modulated based on proportion available (like you find in other sex changing species).

Might be able to pick up a signal of low proportion of males from the data Julie has.

Questions Julie interested in:

- Were too many fish taken out and thus this male problem
- Did flow impact proportion of male-female

Level of depletion – was only anecdotal evidence, and so very hard to quantify.

Eva's suggestion: as a start, assume sex ratio not an issue, but based on data we get from Julie can see if this holds or if signal suggesting something else.

Julie: Doesn't think water temp is a problem for them and spawning.

Natural mortality:

$M = 0.2$ (standard assumption used in stock ass. but is it correct...might be more like 0.3?) – is there any reason why $M = 0.2$?

Julie: no – in the model we couldn't estimate M and q_{inc} , so they fixed M but no reason why it should be 0.2.

Selectivity

Qld – two selectivity periods

NT – are they catching fish older than 10? (Julie – thinks unlikely but she can try find out). Large fish hard to handle and no one really wants those fillets.

Age frequency of catch – waiting for these data from Julie

Fitting to catch data: ding similar as for prawn and mud crabs.

Relationships with flows:

Julie - Flinders have had huge flows, much larger than others (2009, 2019), so relationship for Finders probably parabolic compared to other regions? Although maybe logistic is more realistic – more important to capture what happens at low flow years as that is what water extraction will be speaking to.

Michele: in 2019 floods – barramundi fisher's giving up because nothing to catch and habitat trashed, and their view was it would take a while for things to return to "normal" – so can have long lasting effects. Julie: massive amount of sediment that came down in 2019 flood and after 18 months still there.

Eva: are we going to get artificial enhancements of flow when they release huge amounts of water in years when there wouldn't have been big flows? And thus, do we need to get those extreme flow ends correct?

Julie – thinks unlikely. Prob better to focus on lower end of spectrum (low flow years).

Julie: Will be important to note that there are events in the system that the MICE model just can't capture.

Rodrigo – how much complexity are you capturing in each component. Seems to be more than intermediate complexity?

Appendix 4 – Final Stakeholder Workshop

Stakeholder Teleconference Workshop (9-10 August 2021)

Monday 9 August: Overview of Water Resource Development Scenarios being tested; focus on Prawns

WORKSHOP AGENDA: 9-10 August 2021

via teleconference, 9:30-12:30 AEST both days

webex: <https://csiro.webex.com/csiro/j.php?MTID=m71e4785e7c6f8885607f6c8a3e1d592f> phone:

+61 2 6246 4433 ; quick dial: [+61262464433,1659048576%23%23](tel:+61262464433,1659048576%23%23)

recording available: By request to Éva Plagányi

Agenda

9:30 Welcome and Introductions (Éva Plagányi)

9:45 Overview of Water Resource Development modelling and scenarios being tested (Justin Hughes, Shaun Kim, Rob Kenyon)

10:30 10 minute break

10:40 Overview of prawn modelling (Éva Plagányi)

11:40 Prawn economics discussion (Trevor Hutton)

12:00 Industry and stakeholder perspectives

12:20 Open Discussion session (All)

12:30 Workshop Close of Day 1

Key Objectives

With a focus on **prawns**, participants were invited to join our final project workshop where we will discuss results and seek your feedback on our Fisheries Research and Development Corporation (FRDC) funded project titled **‘Ecological modelling of the impacts of water development in the Gulf of Carpentaria with particular reference to impacts on the Northern Prawn Fishery’ (NPF)**. The project is focussed on the Mitchell, Gilbert and Flinders Rivers of Cape York as a template for modelling the impacts of water extraction on the maintenance of ecosystem services in tropical estuarine habitats, and recreational and commercial fish and crustaceans.

Model Region Map (for reference)

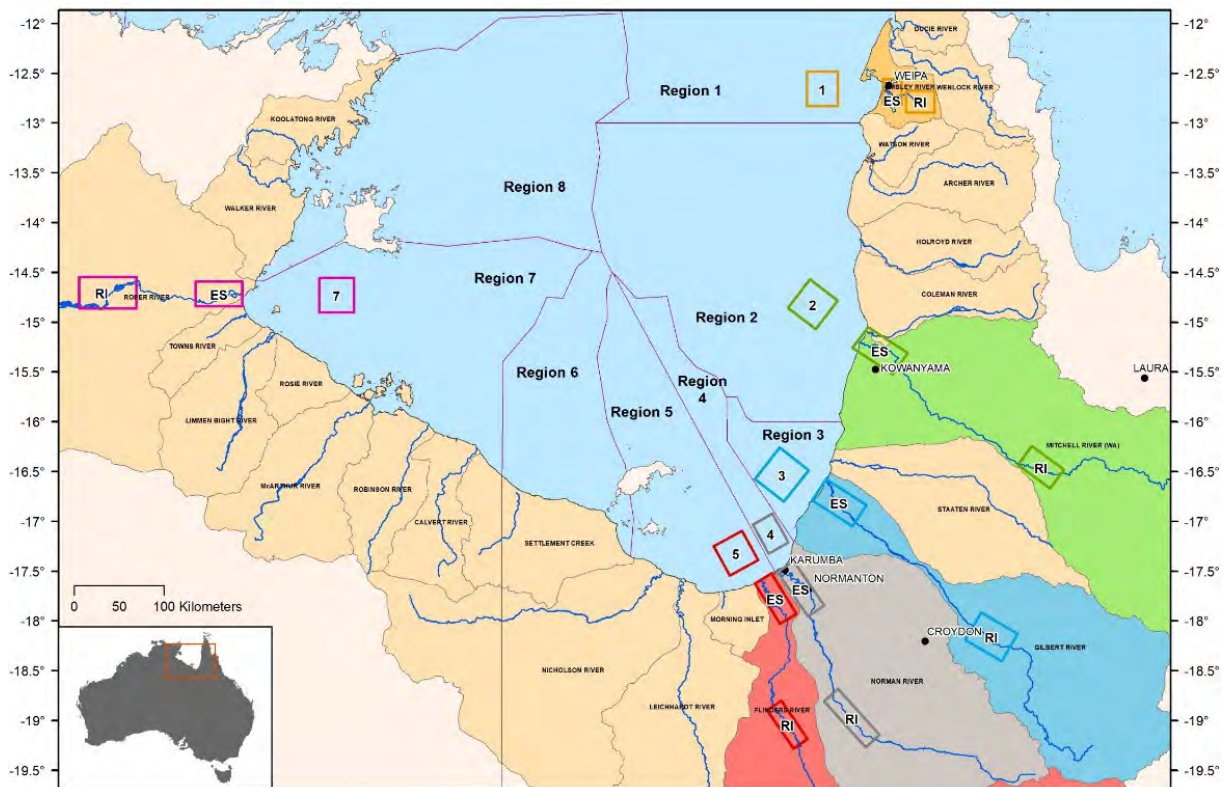


Figure A1. Map of GoC spatial regions used in MICE

Participant list – Monday workshop

Name	Organisation	Online / phone
Zoe Williams	Northern Gulf NRM	Y
Linda Merrin	CSIRO L&W	Y
Rik Buckworth	Consultant	Y
Bryan van Wyk	Austral Fisheries	Y
Chris Chilcott	CSIRO L&W	Y
Darci Wallis	AFMA	Y
Elissa Mastroianni	AFMA	Y
Ian Knuckey	Fishwell Consulting	Y
Ian Watson	CSIRO Ag & Food	Y
Rodrigo Bustamante	CSIRO O&A	Y
Cuan Petheram	CSIRO L&W	Y
Linda Merrin	CSIRO L&W	Y
Marcus Barber	CSIRO L&W	Y
Michaelie Pollard	CSIRO L&W	Y
John Ritchie	DNRME	Y
Daniel Stratford	CSIRO L&W	Y
Éva Plagányi	CSIRO O&A	Y
Rob Kenyon	CSIRO O&A	Y
Laura Blamey	CSIRO O&A	Y
Annie Jarrett	NPF Industry Pty Ltd	Y
Adrienne Laird	NPF Industry Pty Ltd	Y
Michele Burford	Griffith University	Y

Name	Organisation	Online / phone
Julie Robins	QDAF	Y
Trevor Hutton	CSIRO O&A	Y
Roy Deng	CSIRO O&A	Y
Justin Hughes	CSIRO L&W	Y
Margaret Miller	CSIRO O&A	Y
Chris Moeseneder	CSIRO O&A	Y
Emma Lawrence	CSIRO Data61	Y
Sean Pascoe	CSIRO O&A	Y
Phil Robson	Raptis	Y
Austral Fisheries - Andy 467****58	Austral Fisheries ??	Y

*Abbreviations: CSIRO Land & Water (L&W), CSIRO Oceans & Atmosphere (O&A), CSIRO Agriculture & Food (A&F), Australian Fisheries Management Authority (AFMA). **Apologies** from: Shaun Kim, CSIRO L&W.

Presentation: Water Resource Development scenarios underpinning workshop outcomes – Rob Kenyon

Slide pack entitled MICE_GoC_NPF_WRDrepresentations_rk.pdf

The environment provides significant refugia but it is nutrient limited, very hot and stressed (9mths of the year) and each year invigorated by river flows which are vital for productivity and connectivity (~3 months of the year). Phytoplankton and microalgae drive the majority of productivity. Hypersaline to brackish when flows occur and flows also provide connectivity through the catchment that result in emigration cues.

A quick recap on life cycles of prawns including how the larvae and juveniles use these systems (prawns and barramundi lifecycles were shown) and we were reminded of the importance to both species of connection through the catchments. The extensive mangrove systems in various rivers are supportive of these key species and others. The flow characteristics for Mitchell, Gilbert and Flinders Rivers were shown and effects on how these flows result in habitat access changes from year to year were explained.

For the interim results shown in this workshop a number of scenarios were run (but not all possible scenarios or combination of scenarios) included FGARA work baseline vs moderate levels vs high-level WRD scenario for Flinders and Gilbert (explained) and for Gilbert feasibility study for Greenhills dam already published. For the Mitchell River more options were considered, 4 extraction levels with several triggers for pumping explained – to retain lower flows; plus 2 dams or 5 dams scenarios.

An explanation of flood flow given patters of late dry versus wet season flows and how these flows affect the prawns, crabs and barramundi reproduction cues and recruitment to the fishery were explained, see Figure A2.

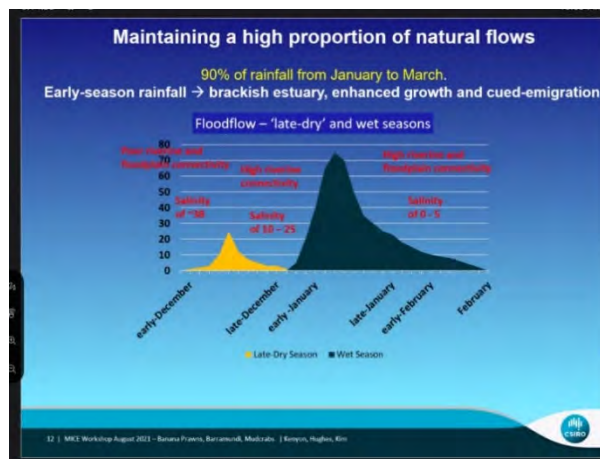


Figure A2. Importance of high proportions of natural flows

When wet seasons comes arrives some of the estuaries become freshwater systems and this cues banana prawn emigration. However, taking the top off high season flows should be done in a way that maintains the connectivity in the floodplain for extended periods. [extra notes here from RK notes or recording].

Q&A Session

- There were no questions however strong feedback was provided that ‘invigorating’ is definitely the right term to capture the importance of flows.
- Brief discussion of seasonal catch and importance of rain the previous year as well as the current year to the catch volumes.





Presentation: GoC Ecological MICE model Overview with Prawn Focus – Éva Plagányi

Slide pack entitled *MICE_Economic impacts on prawns_9th_AUG_final_update.pdf*

Eva reminded everyone that the spatial map on the MICE model was provided with the workshop agenda, as during the presentation different region numbers will be referenced. Important to note that the region map uses the Qld/NT border because different management in NT as compared to Qld.

Summaries collected through stakeholder engagement were used to capture these key processes. An overview of the key biological processes used in our MICE model are shown in Figure A3.

Example of key biological processes captured in the MICE population models

Processes captured in biological model				
Stock-recruit relationship (density dependence)	✓	✓	✓	✓ (simple)
Prop. of spawning each week/month	✓	✓	✓	January
Growth (in weight) each week/month	✓	✓	✓	Numbers
Natural mortality	✓	✓	✓	✓
Age-dependent natural mortality		✓		✓
Age/size able to spawn	✓	✓	✓	✓
Age/size selected by fishery	✓	✓	✓	✓
Availability to fishery	✓	✓	✓	
Sex change		✓		
Sex modelled separately			✓	

13 | NPF Ecological Modelling - Eva Pálagany et al.

Figure A3. Key biological processes in the MICE model

Given MICE models capture intermediate complexity we start with a simple model. Then we add complexity as needed, but always hypothesis based. The question is asked ‘what is the next most important thing for this species?’, and then once included in the model checks are done to make sure it does improve model fit and also reflect well with historical data. Then the next step is to validate the model against the observed data and then reiterate with stakeholders to look for further improvements. This becomes the base model.

Once the base model is constructed, then we run the model with different WRD. So then stepwise model validation.

An example is provided for prawns in Figure A4. Understanding underpinning the prawn population model. Fishing effort is the input, and catch is the output – then link flow to population dynamics, and we do know that including flow improves catch, gradually adding in the further complexity.

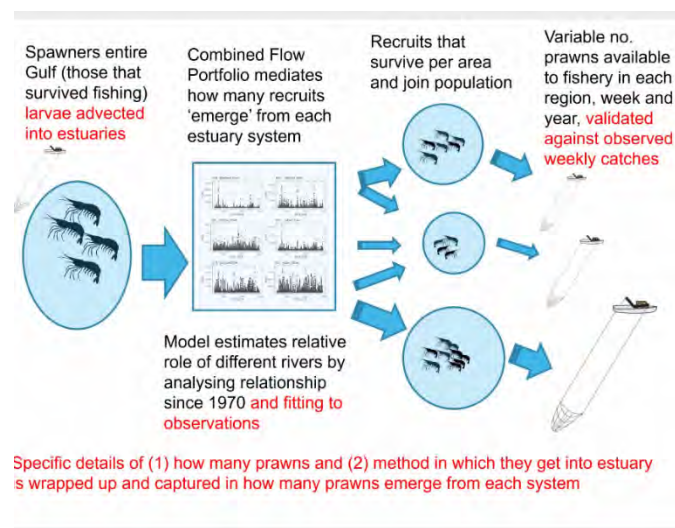


Figure A4. Understanding underpinning the prawn population model

While explaining the figure, Eva noted the different dynamics of prawns – model starts with spawning biomass (those that survived fishing the previous year). We include the flows from river each week, each area, and output such that blue is the observed catch, red the predicted average catch per region (see Figure A5) – this includes effect of rainfall in the previous year.

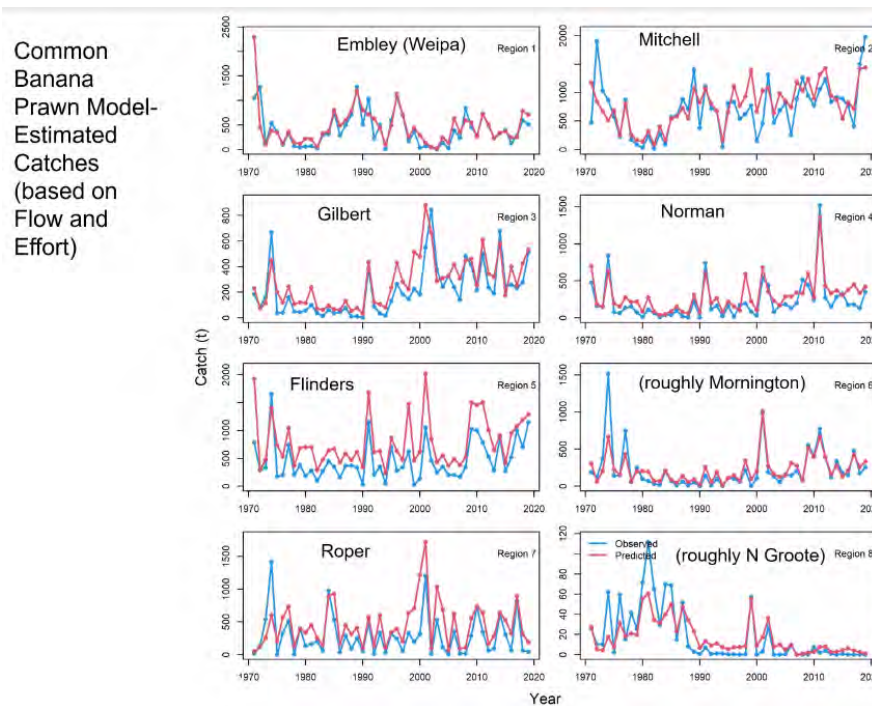


Figure A5. Banana prawn population model

The prawn model does a good job of fitting the data. This is basic model used to compare catches once flows change in each area. Note Mitchell variation explanation 53% not great but very good in other regions over 70%.

Dynamics is influenced by the portfolio of Rivers – this is important to note and understand.

We also look at the commercially available biomass, and how does spawning biomass for each species change under different WRD scenarios. An explanation and discussion of WRD1 – plot below in Figure A6 shows catch as a proportion of the base flow catch for the different areas – dark blue circle/line is the Mitchell region.

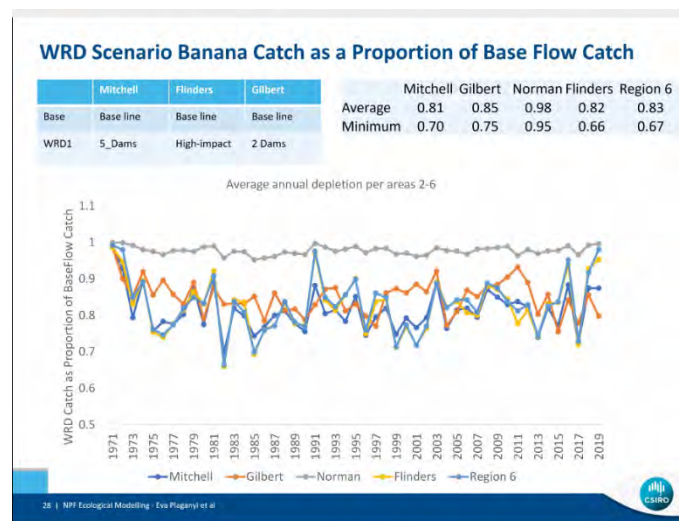


Figure A6. Scenarios for Mitchell, Gilbert, Norman, Flinders regions and Region 6

Q&A session

Q: Is there a correlation between percentage drop in catch and annual size (amount) of catch? i.e. do you lose the same amount of catch on average regardless of whether it is a good year or not?

A: Eva clarified that the question is around which has more effect big flow years versus good catch years. Most reliable basis is portfolio affect for coming up with these numbers – these numbers are at the less conservative end, noting there is a fair amount of rigour but we still need to look at the sensitivities of the model in more detail before final results and the remainder of the scenarios are run.

.....

Q: Asking about model fit – have you tried calibrating data with first half of data and then assessing it with the second half of the data (time series).

A: We have not done that because it would not necessarily produce a better model fit. However we could try a version of that proposal once we get the updated flows if perhaps we were to run the model to 2019 and see if data to could predict the last 5 years data.

Discussion around these issues with some noting fitting to catch/and difficulty with breaking the timeseries is that you have events that come at various times in the catch series, so knowing where the timeseries can be broken without just missing one of the big flow events say after many years of low flows would produce a different result to having that big flow event in the series. Another noted a particularly large flow in Flinders as another example of data challenges re atypical years/result. How else to ensure how representative the model fit is moving forward and another alternative to assess the model fit were discussed.

Takeaway points: Eva notes the model is fitting to every week of every year with likelihood function, but we are aware that collating environmental data is challenging so next step after this project is to look at future climate scenarios. We are using historical data to come up with a model that explains past data and then you use that to project forward, and we check other diagnostics as well, for example, catch as proportion of biomass.

from Ian (Guest): Do we have an idea which of the scenarios would be preferable for extractors? I meant for the people extracting the water

Discussion around disincentives and logic that the more water you seek, the lower reliability that you can achieve that harvest each year. Yes, various scenarios were evaluated in NWARA 2000 G / same for dam scenario as well. Noted that 5 dams scenario is high and some thought probably unrealistic.

A: We are busy quantifying all the other scenarios and combinations of scenarios that we will run with the model. Remembering that It's not just on average how much catch is foregone, but also potentially sustained low catches, and hearing that industry are interested in answering the question of how bad might it get.

Q: Reliability of the Embley – is there a correlation with mining industry water extraction on the Embley, notes mine in Weipa already extract a high proportion – would that have impacted on the proportion of what you would expect to catch off there. A: Eva will check if we have data for mine extraction – we could then check model outputs in region 1.

Discussion noted that there are a range of stakeholders – mining, commercial fishing, environmental and indigenous - but adding crops, means that graziers and their associated communities could become the main stakeholders. The Ord irrigation scheme was mentioned and the ensuing environmental impacts. It was noted that mining is a relatively small user of water – 10G for large coal mine and taken during high flows is arguably noise in system but in dry times could have more impact on ecosystems.

Q: Is the point more about consecutive low flow years, and the need for some hard decisions at those times? Also noted that it seemed like the preliminary results shown have some alarmingly big catch reductions associated with water development in very key areas for the NPF industry.

A: The results shown are still preliminary and we need to run the other scenarios and other combinations of scenarios as well as further sensitivity testing and additional flow data.

Observation: Would hate to imagine what it would look like adding these reductions to an already poor year like 2020. It's hard to find crew as it is! Flow on impacts shifting a lot more effort to top end areas and making it hard to operate out of Karumba also concerning.

More queries as to whether the Qld mining was on special agreement mining acts, developed prior to water management Act i.e. they were not restricted by flow levels. John Ritchie will chase up the information and will circulate to the project team for sending to all participants.

Further discussion around model fit for Embley River area and in estimating the relationships – Eva answered if we knew about mining extractions, we could get an idea of how it affects. Noted that rainfall runoff model and calibration not recent so it might not be picking up those extractions. Eva reminded everyone that following the workshop we will look at some differing scenarios.

Presentation: Potential economic impacts on the NPF of reduced banana prawn productivity – Trevor Hutton

Slide pack entitled MICE_Economic impacts on prawns_9th_AUG_final_update.pdf

CSIRO funded associated research on the flow-on effects to the economics of the NPF. The analysis makes an assumption of fixed costs which is related to catch ratio.

One benefit of an integrated model is that total profit is maximised as per Commonwealth harvest policy. Also shows most profit is from banana prawns.

Projections using tiger prawn stock assessment model which sets the TAC for the fishery, every second year – with profit in region of \$2-5 m. So we needed to include common banana prawns as a depletion model (too variably as influenced too much), and an effort model for banana prawns. We estimate a price flexibility relationship for banana prawns.

There are some issues of Christmas peak demand for prawns, plus issues of imports.

Now achieved but the fixed costs are at the fishery level with this new model. Thanks to industry for providing the non-confidential economics – large sample size of 52 vessels, as many as 37 vessels. Results would be different if finer scale economic costs of fishing were available.

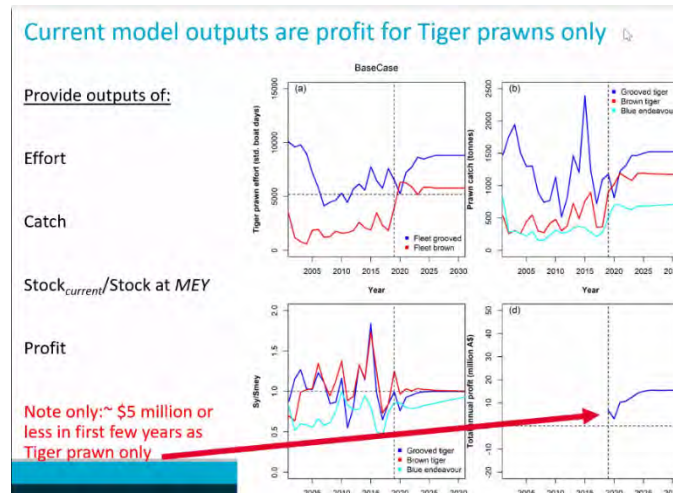


Figure A7. Model outputs based on effort, catch and profit

We are able to cross check with biomass out of the MICE model. Two metrics are used, the first is short term profit, and the second is net present value (NPV) of the NPF.

Results for first year – profit is \$30m in 2018 as profit for the fishery with bananas with 1500t common banana prawns give 7% profit based on investment and fixed costs, see Figure A7. Noting that in Figure A8 below the changes from actual returns at either an 8% reduction or under low flows.

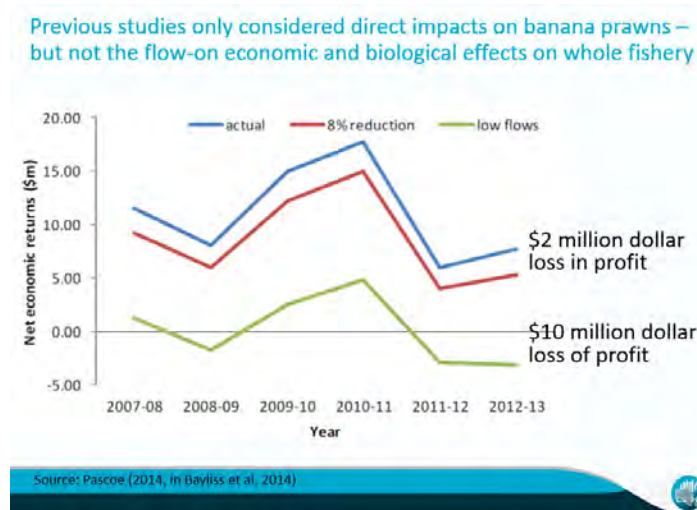


Figure A8. Net economic returns for actual, 8% reduction and low flows

As seen in Table A1 below, the profit lost under various WRD scenarios ranges between \$2 - \$4 m while the reduction in NPV ranges from \$9.8 – 68.6m.

Table A1. Economic profit and net present value (NPV) calculations for a range of WRD scenarios

Case	Base	WRD1	WRD2	WRD3	WRD4	WRD5
Profit (\$millions)	32.7	29.2	28.7	30.3	29.8	30.7
difference from base		3.5	4.0	2.4	2.9	2.0
NPV (\$millions)	799.7	740.6	731.1	759.8	789.9	765.7
difference from base		59.1	68.6	39.9	9.8	34.0

Final integration with MICE model is still to occur and that will determine the sensitivity of break-even point to economic parameters.

Q&A Session

Q: wondering what impact of ponded aquaculture might be on the economic forecast ... could be for irrigated agriculture or prawns?

A: No not at this point but there are moves in the GoC around Weipa and there are indigenous communities to do aquaculture ponds (small sized) – noted that there is a risk of this becoming a complicating factor. Also in JBG there were plans for aquaculture with production eventually 5000 t. Sean notes we looked at aggregate level of imports and the effect of aquaculture on wild caught prawn fisheries economics – notes 10% increase in domestic aquaculture would decrease wild caught prawn fishery by 1% when considering the national situation. But if large scale aquaculture is an option, then note currently 10% imports reduces wild catches

Comment: noting zero net discharge is wishful thinking rather than reality.

Next steps – Éva Plagányi

The next steps for the project are to:

- Compile full set of WRD scenario runs
- Continue refining model with stepwise addition of complexity if justified –including trialling links to other model components
- Refine economics
- Continue testing robustness of model results to alternative assumptions / model parameterisation
- Collate risk statistics across all species groups
- Focused discussions with key stakeholders
- Present summary talk at World Fisheries Congress, 20-24 Sept
- Final Report 15 October + communication materials including high level summary for managers
- Scientific journal papers

Comment: One fisher notes last couple of year, 2000t just going back to Trevor's valuing of the industry struggle with economics as % of fleet might have made some money but a lot of private fleet struggling with low catches and lack of market access. Big issue with whole river system is quite political at times, and some of the rich pastoral companies have a lot of say and don't think that down the track about the possibility of not being able to avoid consequences to the fishery. Those wanting building dams may seek to dodge responsibility for monetary compensation re changes. The % losses for the fishery do not have to be large to see changes in fishing habits. The effects would be far more extreme – even small retrospective damage from proposals like Seafarms (aquaculture) and Cubbie station etc. There was mention of other consequences e.g. white spot prawn contamination, other unknown consequences. Previous prices are not economical but this year's price was slightly better. Industry does a lot of work as far as seeking to be ecologically friendly with bycatch reduction and lessening environmental impact, and then have to deal with impacts of other industries.

Trevor notes we only get fishing cost data averages from Tom Kompas and without the semi-confidential data we can't do much better than current estimates; agrees that industry are struggling. One fisher pointed out that filling crew positions is very challenging – partly from COVID but more from the poor fishing year last year.

Comment: notes relative importance of river flows for fishery, context is when NESP project started, they were asked which Rivers are less important to the fishery. But even with noted caveats and uncertainties showing any one River is more important may lead to ignoring the effect of the portfolio of rivers – this could be used to say one particular River is less important so we can do more water development there. Also notes the importance of helping industry understand estimates are conservative since there are some impacts that in the long term may be significantly more if a range of analyses are done.

A: We agree cumulative effects over the long term. MB we can't quantify sediment and nutrients effects on productivity which will have flow on effects on the fishery.

Q: NPF Industry thanked the researchers for their excellent work and asked if presentations could be made available after the workshop.

Action: Presentations to be circulated following the workshop.

Comment: relative importance of the rivers means it would be judicious to reconsider naming regions without referring to specific rivers, in case it is misconstrued – noting that prawn catches are offshore versus barramundi caught inshore. It would be good to see differing context of coastal versus offshore impacts. Somewhat surprised Norman has such big impact on Gilbert – but noting different spatial size of each region. Surprised Gilbert didn't have a bigger impact on lowering prawn catches.

Comment: With catchment clearing extra sediment will be coming down the rivers especially true for land use conversion to grazing land. Evidence in SEQ that such sedimentation caused issues with mud crab catch and led to fishery collapse. Also note the for the Embley/Weipa when in 2000 you see drop in prawn harvest corresponding dearth of mud crab harvest, and the mud crabs seemed to disappear for a number of years, plus at same time some air freighting issues due to Ansett Airlines collapse.

Comment: Oceanographic issues, currents may sweep up rather than coming down into SE corner of GoC.

Comment: Great piece of work and like the way its drawing together all the knowledge.

A: We can look at small scale but some of higher results are difficult to interpret – Gilbert is important in consideration of variability so we will tease that out as averages over all the years can be difficult to see.

Extra questions after the workshop closing time continued, as follows:

Q: Are you considering how each river system influences the others? A: No, not how the river systems influences the other river systems but how all these influence how much prawns are available to be caught.

Q: Fisher asked whether region 5 takes in Leichardt and Nichols River systems. A: No, we aren't able to do that, the difficulty is lack of data.

Comment: Fisher notes that catches from region 5 is not the way they look at it as a fishery, noting that region 4 is narrow and thought Norman spilled out into NW and the NE and Flinders more influential north of Leichardt and Nichols that propagates.

A: we have the observed catches using grid data and if you take one of them for say region 2 we can explain variability in catch not just in Mitchell but also how the other rivers explain the variability ... sometimes they do but sometimes they do not.

Q: As region 5 feeds into SW corner it is hard to see how you see Flinders could influence without Leichardt and Nichols.

A: Could be included if we had more flow data – however we have to work with what we have. We are more interested in how things change relative to the base, so its not quite as bad as if we were trying to predict catches like a stock assessment model. Eva reinforced we looked at low catch years and high catch years but they were consistent with when the Norman was flooding, so we need to think of SE corner as combined.

Q: Fisher noting rainstorms crossing over the areas can be patchy, as one area can completely miss out on the rainfall. A: Eva notes rainfall patchiness is very important and by having spatial structure in the model but it is currently not finely resolved enough and maybe that could be further developed down the line.

Eva notes the scale issue and the comments around fishers profitability, that while some fishers will be ok there are a lot battling, and the variability means that some operators are missing out.

Comment: Big companies like Austral will get better price but have to cold store for many months, and these companies have higher corporate costs. Ultimately fishers are still basically a winner or loser based on what they catch. The 76t average but variation 20-50t. note high catches come at higher costs.

Comment: Very impressed to see the cross river systems modelled as up until now we've tended to think of Mitchell River driving a considerable proportion of catches, whereas the portfolio approach is very good – as it captures the cumulative impacts and correlated effects.

Tuesday 10 August: mudcrabs; barramundi; other components; next steps and towards recommendations

Agenda

9:30 Short welcome (Rob Kenyon)

9:40 Mud crab model results, followed by discussion (Laura Blamey)

10:20 10 minute break

10:30 Barramundi model results, followed by discussion (Éva Plagányi)

11:10 Other species (sawfish, habitat components)

11:30 Integrated overview of Water Resource Development Scenarios, combined species results and risk statistics (Justin Hughes, Shaun Kim, Rob Kenyon, Éva Plagányi, Laura Blamey, Roy Deng)

11:50 Industry and stakeholder perspectives and open discussion (All)

12:10 Next steps and towards recommendations

12:30 Workshop Close

Key Objectives

With a focus on **mudcrabs, barramundi and sawfish**, participants were invited to join our final project workshop where we will discuss results and seek your feedback on our Fisheries Research and Development Corporation (FRDC) funded project titled **'Ecological modelling of the impacts of water development in the Gulf of Carpentaria with particular reference to impacts on the Northern Prawn Fishery'** (NPF). The project is focussed on the Mitchell, Gilbert and Flinders Rivers of Cape York as a template for modelling the impacts of water extraction on the maintenance of ecosystem services in tropical estuarine habitats, and recreational and commercial fish and crustaceans.

Participant list – Tuesday workshop

Name	Organisation	Online / phone
Zoe Williams	Northern Gulf NRM	Y
Linda Merrin	CSIRO L&W	N
Rik Buckworth	Consultant	Y
Bryan van Wyk	Austral Fisheries	Y
Chris Chilcott	CSIRO L&W	N
Darci Wallis	AFMA	Y
Elissa Mastroianni	AFMA	Y
Ian Knuckey	Fishwell Consulting	N
Ian Watson	CSIRO Ag & Food	Y
Rodrigo Bustamante	CSIRO O&A	N
Cuan Petheram	CSIRO L&W	Y
Linda Merrin	CSIRO L&W	N
Marcus Barber	CSIRO L&W	N
Michaelie Pollard	CSIRO L&W	N
John Ritchie	DNRME	N
Daniel Stratford	CSIRO L&W	N
Éva Plagányi	CSIRO O&A	Y
Rob Kenyon	CSIRO O&A	Y
Laura Blamey	CSIRO O&A	Y
Annie Jarrett	NPF Industry Pty Ltd	Y
Adrienne Laird	NPF Industry Pty Ltd	N
Michele Burford	Griffith University	Apologies
Julie Robins	QDAF	Y
Trevor Hutton	CSIRO O&A	Y
Roy Deng	CSIRO O&A	Y
Justin Hughes	CSIRO L&W	Y
Margaret Miller	CSIRO O&A	Y
Chris Moeseneder	CSIRO O&A	Y
Emma Lawrence	CSIRO Data61	N
Sean Pascoe	CSIRO O&A	N
Phil Robson	Raptis	Y
Austral Fisheries - Andy	Austral Fisheries	N

Presentation: Overview & Recap of previous day's workshop – Rob Kenyon

A brief summary of the talks on the previous day reminded participants of the reinvigoration that the river systems provide given that for 9 months of most years these are self-contained ecosystems and microalgae driving productivity and limited tidal exchange.

Presentation: Modelling mud crabs in the Gulf of Carpentaria – Laura Blamey

Slide pack entitled NPF Ecological Model_Mud crabs_August2021 – Copy.pdf

Adult crabs live in estuarine habitats, thought that females spawn offshore during wet season, larvae move back in shore and then will when juvenile moves into mangrove estuarine habitat.

Mud crab fishery in NT vs Qld, slightly different management regime, in Qld no females can be retained if caught. Qld catch around 30% of whole GoC, and the rest NT. Both states' fishery management regime includes gear restrictions.

Explained the effort and catch from 1990 up to 2019 – effort peaked in early 2000s and then declined afterwards, catches followed same trend with decline in catches since.

The mud crab catch data are mapped across our model regions is shown in the plots of Figure A9.



Figure A9. Mud crab historical catch data plots

Mud crab model structure explained, see slide and the environmental factors – flow, air temperature and potentially all SOI. Crabs are cannibalistic. Prolonged and heavy flows may reduce availability of crabs because they've moved offshore, or perhaps seagrass die off or juveniles not able to get into estuary.

Very complex system, probably the closest fit we can achieve as we've kept it to just a few drivers. Showed the correlation plots and pointed out the variability in model ability to match to the observed data.

Explained the WRD scenarios currently used and the model has been re-run and outputs are the proportion of the catch relative to the base case versus the scenarios (more scenarios still to be run). Note 60-70% reduction in catch for the high scenario. Note Flinders and Gilbert quite different – but the Roper shows some reduction in catch 80-90% of base case. What's the minimum the catch goes to across the modelled prior – note it can dip down as low as 50% in the extreme 5 dam scenario. But for Flinders can go down as much as 10% of catch. For Roper not showing so much reduction in catch.

Table A2. Water resource development scenarios and related average catch, minimum catch and percentage of base catches

	Mitchell	Flinders	Gilbert	Roper	AVERAGE CATCH (t); MINIMUM CATCH (t) for entire GoC	% OF BASE CATCH
Base	Base line	Base line	Base line	Base line	425 ; 79	
WRD1	5_Dams	High-impact	2 Dams	Base line	367 ; 50	86%
WRD2	High-impact	High-impact	2-Dams	Base line	369 ; 50	87%
WRD3	Moderate impact	Moderate impact	1-Dams	Base line	377 ; 57	89%
WRD4	Base line	Base line	Base line	20% reduction	399 ; 79	94%
WRD5	Base line	Base line	Base line	50% reduction	339 ; 79	80%

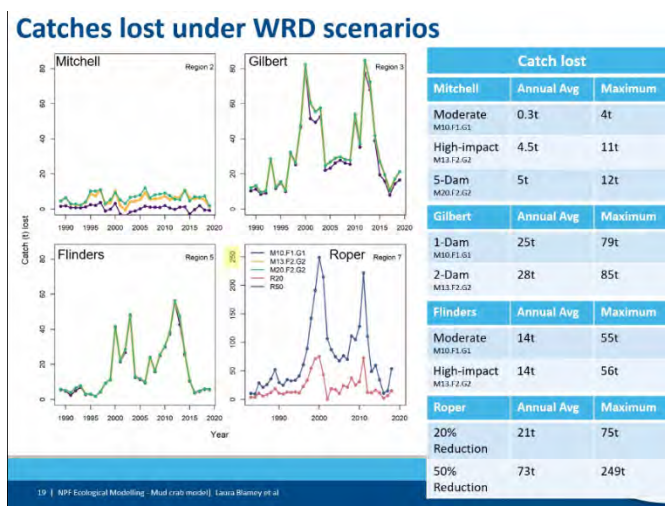


Figure A10. Lost catches under various water resource development scenarios

Remembering that large catches coming from the NT side. Also potentially large losses predicted for the Gilbert in extreme (high catch) years ~80t.

We also look to see how depleted the biomass is over time – for Mitchell under moderate scenarios not too far below baseline, but the Gilbert and Flinders depletion is an issue, see Figure A11.

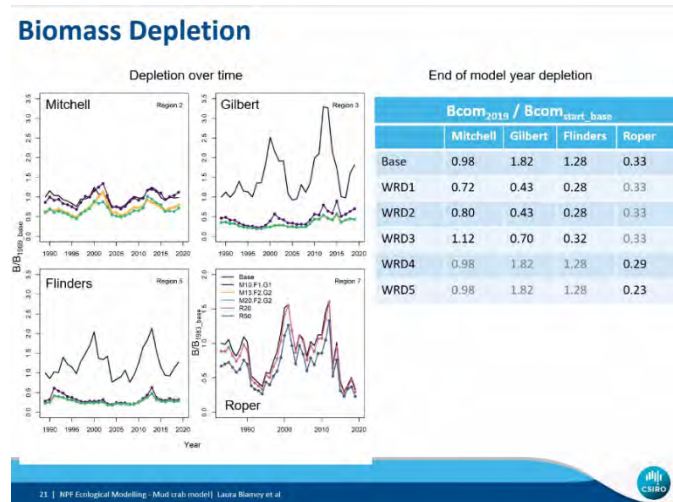


Figure A11. Biomass depletion of mud crabs

We are also thinking about how to link to habitat, as well as potential impacts of sea level rise/southern oscillation index (SOI) but that is not currently in the model.

Q&A session

Q: Asking about differences in model sensitivity noted results in the Roper, using extreme flow scenarios but doesn't seem to be proportionally affecting all Rivers with Gilbert/Norman much more affected.

A: Laura notes flow is linked in region 7 but model links to catch. Also got temperature linked in for Gilbert/Norman and maybe temperature is making a difference to the effects of flows. Model has a more extreme estimate for the eastern side of Gulf.

Comment: Roper region includes Macarthur, so flows from Macarthur & Robinson, so it would be like pooling regions on Qld side. Plus no water resource development forecast on NT side. So a step going forward would be to split region 7 to help understand these effects, given Macarthur is dominant source of mud crab catch.

Comment: Confirms comment about Macarthur is correct plus strong influence of temperature, as a larger explanatory factor than just flows alone.

Comment: Mud crab have a narrow range of temperature and salinity tolerance as compared to other species.

Eva notes that some other steps might be able to split the northern area.

Comment: For our purposes we do not need to split region 2 for prawns but might be good to do that for mud crabs. Our responsibility to check plausibility given major impacts on biomass.

Laura notes we have no mixing between regions for larval stage or when they are spawning. The MICE model assumes each population sticks within its region.

Comment: Pathway for replenishment of recruits might give more variability on what's happening in each region.

Eva notes we will not be able to do connectivity for mud crabs in this project but the strong effect of flow does driving mud crab available to the fishery, and there is a difference between mud crab

biomass and the mud crab biomass that is actually available to the commercial fishery. When we look at the whole GoC (despite some regional differences) which will disproportionately effect some of the stakeholders.

Q: is there a way to add various scenarios iteratively?

A: Eva answered that we consider each species per region – but then we look at the cumulative effects of multiple WRD in different regions. But for prawns which are only species we assume are connected across the regions – so it is more complicated as many different combinations considered.

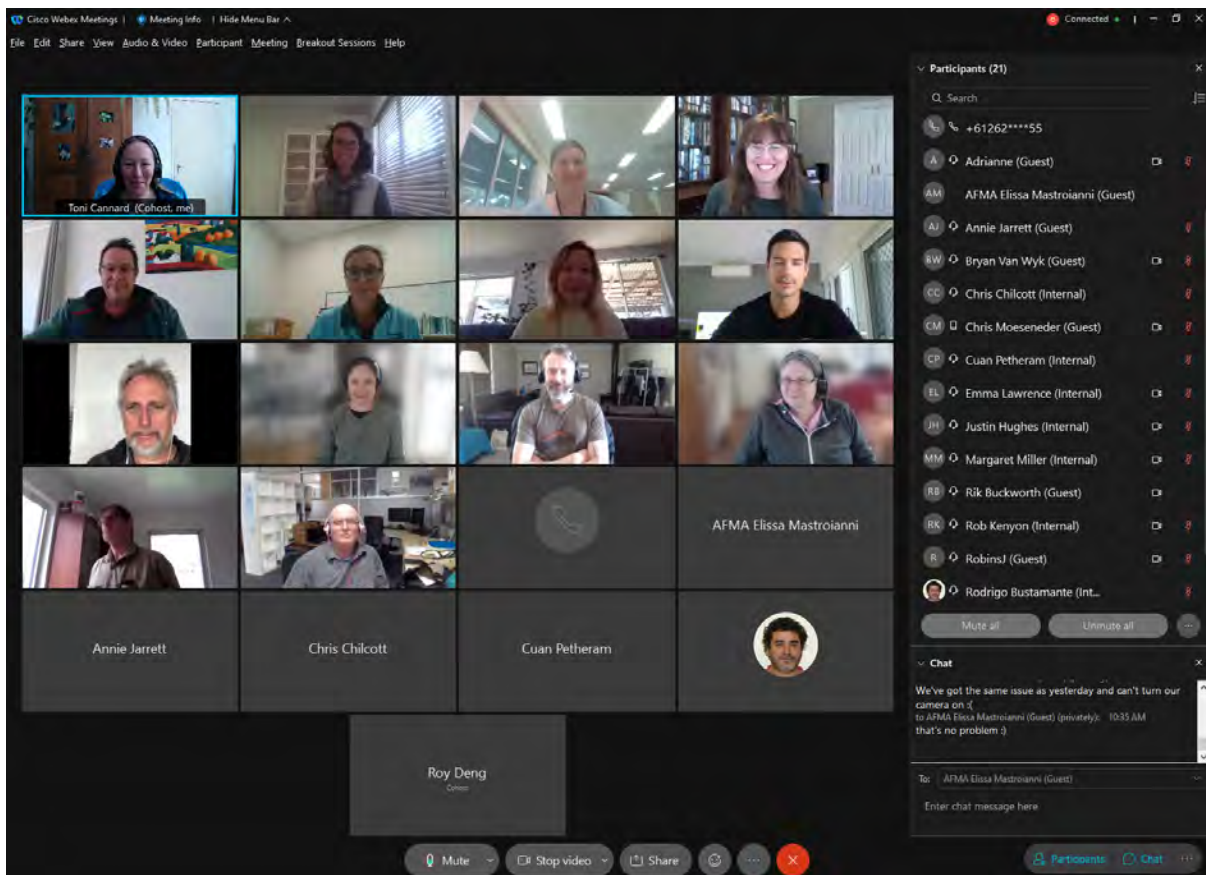


Figure A12. Workshop day 2 participants

Presentation: Barramundi model – Éva Plagányi

Slide pack entitled MICE_Economic impacts on prawns_9th_AUG_final_update.pdf from page 31 onwards

Overview of Barramundi provided noting that it is a very difficult species to model as they have dynamic regional differences, they change sex and there are 3 different stocks in the GoC (see Figure A13). Some differences between Qld and NT management – but similar season and gillnets in NT whereas there are set gillnets in Qld.

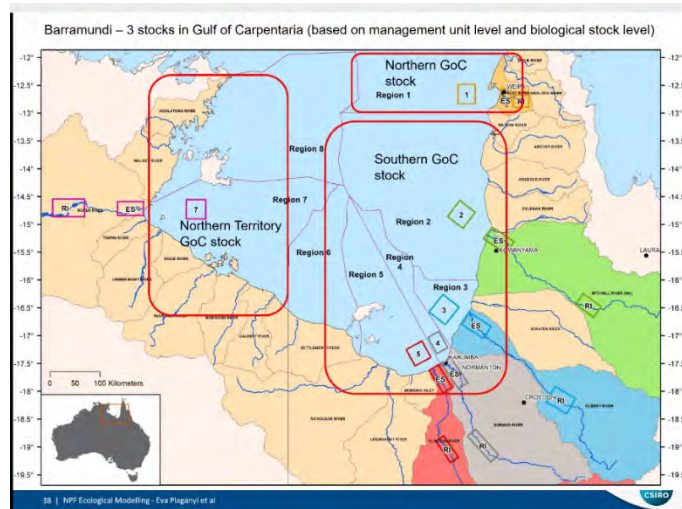


Figure A13. Three spatially separate Barramundi stocks map

An overview of Barramundi model structure was explained – the model is using monthly time-step but take the fish out to 20 years, with the sex change from male to female around 3-5 years, most large fish will be females and in the model sex ratio is skewed. Ages are also captured in the model.

Eva showed model results with the current averages fitted to monthly catch data (see Figure A14) – not perfect but in some regions ok e.g. region 1, but region 2 might have some issues. First the base model is fitted.

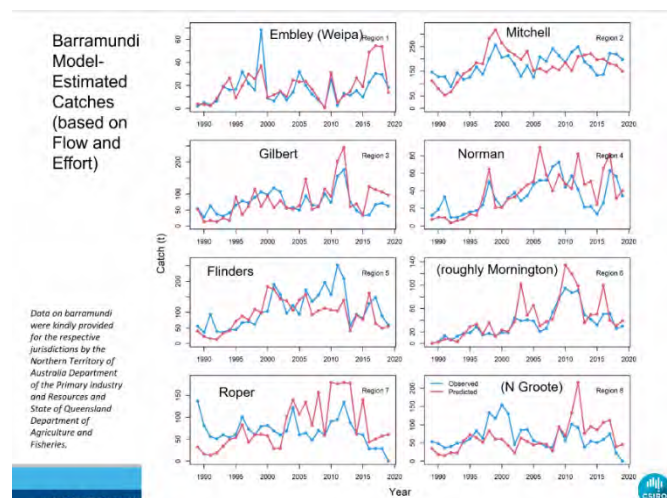


Figure A14. Barramundi model preliminary results based on flow and fishing effort

Several age classes and size distributions of the catch for which we have calculated the catch at age data for the southern region (our regions 3-6) versus the northern region, then showed the proportion of the variation we are able to explain with the model. Combined all fish in regions 3-6 and compare age distributions, blue is observed age distributions, highlighted with light blue and yellow pbar to see how we try to track cohorts in the model.

Second model fit shown for northern region (our region 2) and again track cohorts and trying to estimate the selectivity, mortality parameters and calculate the stock sizes. Note age-dependent mortality for barramundi, so young fish have a higher mortality rate than older fish.

Displayed and explained the preliminary results, if you look at the total for whole of GoC and the colours are the different WRD scenarios. Differences in catch are detected – and then we do the same for total biomass. Estimates of lost barramundi catch as much as 424 tonnes but plenty of variability.

Discussed various results from each of the WRD scenarios that have been run so far, see Figure A15.

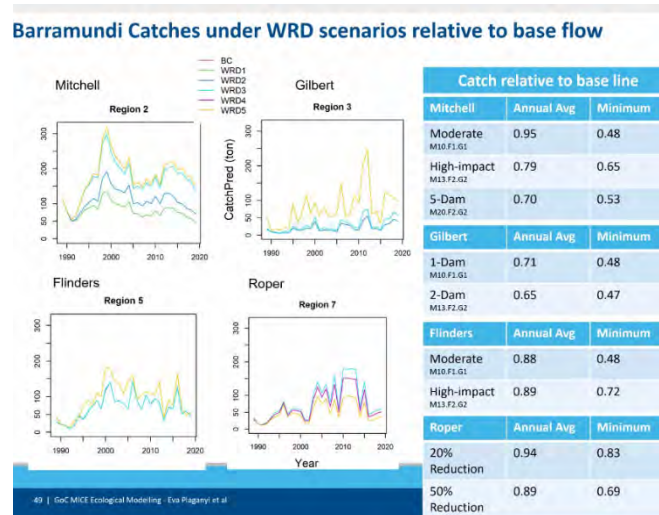


Figure A15. Barramundi catches relative to base flow

Then Eva explained the overall results for the whole GoC annual average catch – see Table A3 below.

Table A3. Summary results for WRD scenarios as compared to Base (first row)

	Mitchell	Flinders	Gilbert	Roper	AVERAGE CATCH (t); MINIMUM CATCH (t)	% OF BASE CATCH
Base	Base line	Base line	Base line	Base line	616 ; 137	
WRD1	5_Dams	High-impact	2 Dams	Base line	455 ; 123	74%
WRD2	High-impact	High-impact	2-Dams	Base line	483 ; 125	78%
WRD3	Moderate impact	Moderate impact	1-Dams	Base line	539 ; 130	87%
WRD4	Base line	Base line	Base line	20% reduction	597 ; 134	97%
WRD5	Base line	Base line	Base line	50% reduction	579 ; 133	94%

Note that the Roper region catch is only dropping 3-6%, but note that the devil is in the detail – as we are just taking off a constant percentage.

Q&A Barramundi session

Comment: Results look intuitively quite good. Make sure we don't have movement/recruitment between regions, note Flinders and Gilbert we do have movements between the two. Maybe consider for the report perhaps think about reporting under the governance, i.e. NT vs Qld.

Q: do you fit to age-fit and also to catch? A: Eva notes we are checking both.

Comment: Maybe weight the age-fit more heavily than the catch weightings – might be missing some strong cohorts coming through as fisheries is size selective rather than age-selected. For Qld side under 2000t so perhaps more availability to the fishery but also can cause mortality in upstream reaches when the environment dries out and cuts the connectivity through the catchment.

Eva noted that the big events do have non-linear effects on flow and survival – i.e. at high flows it is the same but then it flattens out, but as flow decreased it affects survival and when very low flow experience the barramundi mortality rises again. A lot of the data was received much later than ideal so we have not had as much time as we would have like to work with the data.

Comment: NPF good years help the fishers hold out through the bad years, so it is not profit in any single year that determines whether fishers can continue operating.

Eva noting Trev will look at some economics on barramundi/mud crab re revenue impacts. Trev notes though that we don't know about cross-licence details, when we looked at the fishing cost data the main issue was two types of gear and not a single species fishery though barramundi might make up a high proportion of the catch there are still 30 other species targeted.

Comment: Notes that BDo? did some recent report using data from last year, and surveys of fishers at the end of 2018/19 – although barramundi only considered threadfins do use estuary.

Trev suggested that being the case we might start with looking at revenue reduction rather than production reduction. Eva reminded all that Barramundi is our key species, and we will ensure that threadfins will also be mentioned in the report. For Barramundi we might be able to create an index for recreational fishery for GoC and how that might change based on flow impacts from WRD scenarios.

Comment: Ensure awareness of small number of operators in NT only gillnet, barramundi and king threadfin. So Barramundi is the key economic driver. You could ask NT fisheries to check economic data available.

Eva asked how depleted the NT barramundi fishery was prior to 1989 as compared to Qld. Eva says the model assumes 20% depleted by 1989 on Qld side. Rik noted that up to 1980s barramundi but notes in 2001 areas like Roper and Macarthur Rivers were approximately 10% depleted.

Julie notes fishing in rivers banned but on Qld side they can fish a distance upstream 30-50 nm. Mitchell south arm however is closed to all forms of fishing except indigenous, but a lot of the fast young fish you see in Qld. They are interested in changes from long term baseline – but they can manage how much you can take, and when you can fish, so looking at overall trends of minor to major impacts from the various WRD scenarios will be informative but not necessarily precise predictions as you cannot predict everything.

Presentation: Sawfish model preliminary results – Éva Plagányi

Slide pack entitled MICE_Economic impacts on prawns_9th_AUG_final_update.pdf from page 47 onwards

Sawfish are a protected species EPBC listed and we need to account for these species by showing how their dynamics are affected by flow. Note we are not presenting sawfishes results today as the team are still discussing these. We do not have detailed data like for the other species which is on a monthly timestep. Sawfishes are a long-lived and slow growing species. In the MICE model these separated by each region as we know sawfishes will come back to the same river to release pups, the pups then swim upstream. Note that sawfishes reach sexual maturity at 9 and 3yrs for Largemouth and Narrow sawfish respectively and produce on average 4 and 12 pups per year respectively.

To model the current population trends over time, we use estimates of the two species – large-mouth sawfish is blue line and narrow is the red (see Figure A16). At the start of 1989 based on barramundi fishing levels we can come up with a relationship between barramundi fishing.

Age-dependent natural mortality results are currently under review – and it is important to get this right as there are significant implications, e.g. offsets. This MICE model is not a stock assessment approach, instead it is taking populations and bycatch over time with a range of different assumptions into account.

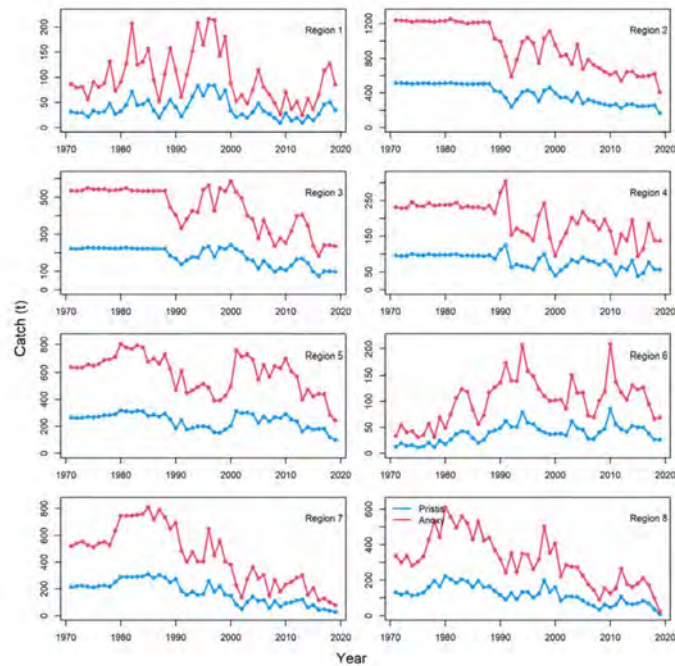


Figure A16. Sawfish catches: based on NPF and barramundi fishing effort

Q&A Session Sawfishes

Q: Noted % changes to flow but instead flow impact on connectivity would not be a linear relationship given they can be isolated in river pools and billabongs.

A: Eva agrees important point, doing it in a simple way in the model, when flows drop low the mortality is steep using a proxy but we will need to look at the

In the plot it shows the number of sawfish caught for each species – not Catch tonnes as shown on plot (axis label error).

Eva points out there is a close kin mark recapture (CKMR) study about to happen but results won't be available for a few years, for this model we cannot wait for those results.

Presentation: Modelling habitat & the bottom of the food web – Éva Plagányi

Slide pack entitled *MICE_Economic impacts on prawns_9th_AUG_final_update.pdf* from page 52 onwards

Note that physical variables are driving the system dynamics, and role of air temperature is linked to mud crabs. Some base habitat categories have been modelled as dependent of the base variables. In the base model we have not linked everything up. This way we can say if you don't like

seagrass/mangroves then you can look at the base model without that part of the model. Cyclone event data is included and a resource we are happy to share to get a more integrated picture.

Meiofauna and microphytobenthos (refer to Figure A17) – for these groups the model was calibrated by comparing with Burford/Dugan? et al studies in the Norman River for the copepods and nematode abundance from empirical studies. In the model we link to flow to work out the productivity in the base of the food web.

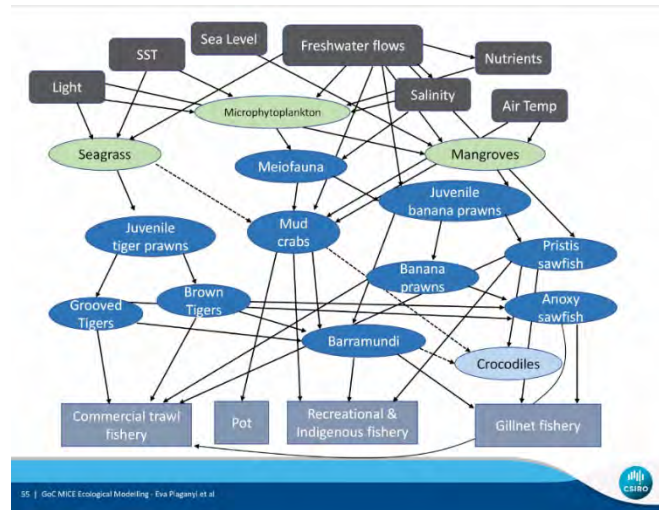


Figure A17. MICE GoC model foodweb

We have prepared seagrass and mangrove models because none found at the scale that we needed for the model i.e. regions. We asked how does the whole seagrass areas change due to sediments and cyclones.

For Seagrass: showed plot of very large flows for Roper region (see presentation slides), high flows in the model shows drops in seagrass coverage – proportionally there are effects of sediments, turbidity, and light attenuation. One of the WRD scenarios annual average showed increases in seagrass when high flows were dampened, likely because reducing associated turbidity with big flows is favourable for seagrass beds.

For Mangroves: we are trying to capture broadscale changes in mangroves over time, important assets in the region as they store blue carbon. However we are still looking for temporal series data at the same spatial scales we are looking at in the MICE model. Extra points on mangrove community modelling is noted in Figure A18 below.

Modelling GoC mangrove community

- Mangroves are important for a number of species, incl juvenile prawns, fish, mudcrabs; also as predation refuge (Robertson & Blaber 1992)
- Useful to try and capture broad seasonal and inter-annual changes in mangrove biomass (grouped together in MICE)
- Relative changes in growth influenced by flow, solar exposure, cyclones, air temperature (AT used instead of SST) and minimum sea level; we assume constant nutrients in base model given challenges of parameterising this aspect
- Extracting old data that can be used to help validate model



Figure A18. Important factors for modelling mangrove communities

Currently the model shows mangroves are fairly sensitive (whole of GoC) with proportional reductions to ~80% of the base levels due to reductions in the flows.

Our draft report shows the summary MICE outputs (see Figure A19) top shows flow anomaly 1980 to 2018. Base case of the model is with no WRD scenario. Bars are the catches over time whereas the shaded areas are biomass changes.

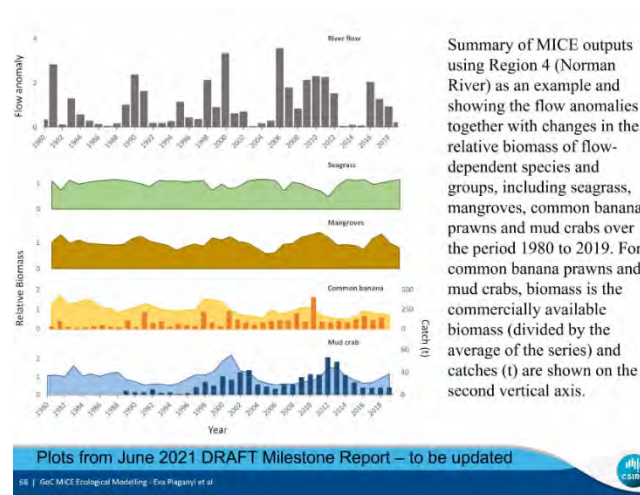


Figure A19. Example of preliminary mangrove results for the Norman River

Then Eva displayed the monthly time steps for a single wet year, and a dry year – and looked at 2015 in the low sea levels (as SOI might be a predictor). We can see reduced flows and low sea level results in more extreme decreases in mangroves. Plus there is a feedback effect on seagrass in high flow years.

The impact of significant cyclones (using indices to every week for GoC thanks to Rob), not just to flows but to seagrass and mangrove habitats is shown in Figure A20.

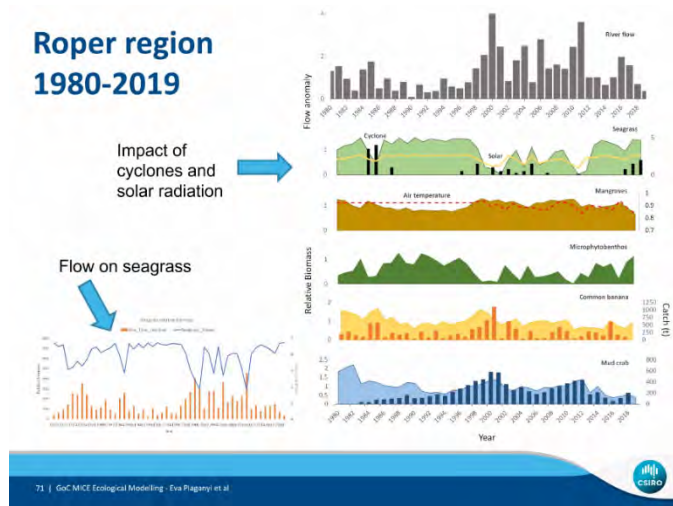


Figure A20. Example of preliminary seagrass results for the Roper Region

Differences in catch in common banana prawns in 2009 after incidence of early season flows, versus dry years for Mitchell and Flinders noting differences in seagrass and mangroves, and in both species of prawns. More results will be available when we run all the other WRD scenarios to show impacts on species in each of the regions.

Plus showing below how multiple regions are affected noting that a single region is only a small contributor to the whole GoC (see below). We can conclude that you will see decreased prawn catch off the Gilbert (Figure A21). See other preliminary results in the presentation slides.

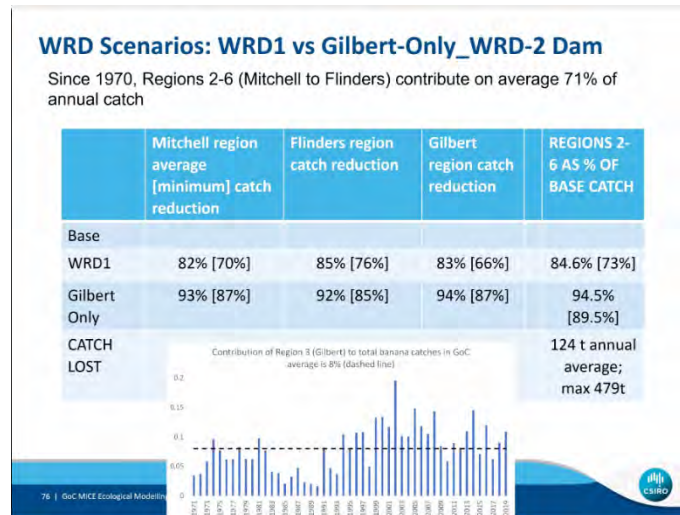


Figure A21. Comparing two WRD scenarios

For all different species we will provide risk statistics – see Figure A22 below.

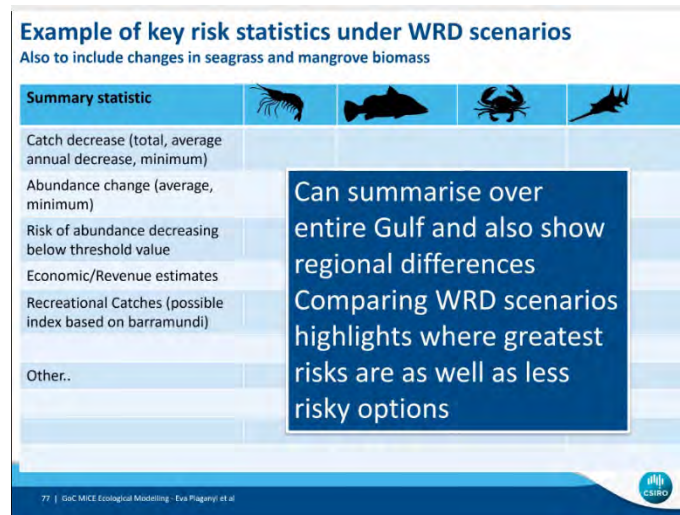


Figure A22. Illustrative key risk statistics for the key species under various WRD scenarios

Once we have the models refined and have run all the WRD scenarios, we will write up the results in the final report.

Thanks to all of the team and to our collaborators and co-investigators. Thanks to all the stakeholders who have interacted with us and provided insightful comments and feedback.

Final Q&A Session

Comment: Noted that ABARES biennial have GVP and production data published for Barramundi – so maybe you could get some price data.

Comment: Also point out the importance on how we report the results so that any one region is not view to be any less important than the other region/s. Flag also for common banana prawns and possibly the other 2 species as well – it would be good for report not to look at the averages as there is high variability (so averaging can be fraught with danger essentially losing the cream of ‘high yield years’ plus need to see the immediate impacts of lower flows in low catch years).

Q: Wondering whether or not we will compare the results from this project with the work that Michele Burford has done to see whether those results align?

A: Eva notes Michele’s work is an input into the model already, that is how we calibrate the base layers and from that we work out the effect of that on productivity. However, we plans to refine the way the data is captured in the model next week.

Eva also notes Peter Rothlisberg was keen for us to include the Embley based on the van der Velde et al paper – however while chlorophyll changes are important it is just too complicated for us to captured reliability in our MICE models. While it helps us with mechanistic understanding it can’t be fully included in the model as just too complex for a MICE model, and also we do not have the data to achieve that at this point.

Comment: NPMI thanked all the presenters for such a comprehensive session.

Trevor has been comparing our economic results to the Smart et al paper.

Eva notes Broadley et al took a statistical approach to look at catch reductions – they say declines could be up to 53% in one region. Our model smooths out some processes but we can explain the

differences from model results with the other published studies. Rob notes Broadley paper in a low flow saw half the catch 500t and he also showed in high flow years but it dropped 9% i.e. 500t. Eva points out that we will show both high and low flow years in our results.

Comment: Concerned that if low production in a range of other regions in the fishery that reduction in flow and catch then creates a greater impact.

Trev noted we only have average economic costs per vessels so that means only when the average profit values, though quite a number of operators could be in the negative range. Sean worked on these issues years ago, Trev notes currently 52 boats in the NPF and some are there for the good banana years – in MEY analysis we could consider fleet size but that is outside the scope of this project. That being the case though, we could do with knowing how many bad years in a row would mean that fishers are likely to take the decision to leave the fishery.

Eva thanked everyone for their attendance, participation, questions and comments.

Rob noted that it has been two really productive workshops and we want to acknowledge our co-investigators and their contributions, thanks to Julie Robins and Michele Burford. We also acknowledge the NESP teams working the freshwater regions of the rivers. They have been showing the importance of connectivity through catchments and how the flows and floodplains interact and link to productivity.

Meeting closed.

Appendix 5 – Water Resource Development Scenarios

FGARA WRD impacts on estuarine habitats and processes.

The FGARA WRD assessment did not treat the estimation of WRD impacts on riverine and estuarine flora and fauna in as comprehensive a manner as NAWRA conducted.

Hindsight showed that from an ‘impact on catchment-to-coast biological and bioeconomic assets’ perspective, FGARA was a ‘learning experience’. FGARA did include an ecological component and fishing industry description within the catchment reports, but no modelling of impact of reduced streamflow on biota or fish and fisheries due to WRD was undertaken (Petheram et al. 2013). NAWRA incorporated an estuarine WRD component (i.e. ‘effects on natural biota and exploited primary resources’) as part of the comprehensive study’s primary body of work (see Pollino et al. 2018 a,b). In contrast, FGARA added the biological-impact component in the form of a separate report as an addendum to the FGARA Reports, with a strong focus on estuarine species (Bayliss et al. 2014). The Bayliss et al. (2014) report was undertaken by O&A staff at CSIRO Brisbane.

Because of the non-core treatment of estuarine impacts, the explanations of the WRD scenarios are not as well delineated or explained in the FGARA reports. FGARA undertook Case Studies that explored the growing of alternative crops in a region and the water needs to support those crops. The water came from instream or offstream storage in proximity to the agricultural expanse. The case studies were targeted to the (water) needs that were required to grow the crop, rather than ‘exploration of the impact of stepped-scenario WRD water-loss downstream of irrigated agriculture’. Scenarios for the Flinders River involved both instream (dams) and water extraction for offstream storage; scenarios for the Gilbert River involved instream dams only.

However, models of the impact of individual dam construction and a range of water extraction scenarios on downstream flows were made (Table A4; Table A5). Bayliss et al. (2014) used these scenarios in their exploration of the impact of streamflow reduction on estuarine species in qualitative and quantitative predictions (common banana prawns and barramundi were modelled quantitatively). FGARA used dam yield (Flinders and Gilbert Rivers) and 5 levels of water extraction (Flinders River only) as WRD scenarios. Bayliss et al. (2014) used dam yield (at ~85% annual reliability), and dam capacities (all dam water), as well as the lower (80 ML y⁻¹) and upper (560 ML y⁻¹) water extraction quantities, as low-level and ‘contrast extreme’ scenarios of WRD usage. These streamflow data should be available from Canberra colleagues.

Table A4. Water Resource Management scenarios in the Flinders River catchment within scope (FGARA) and modelled by Bayliss *et al.* 2014. PLUS no WRD Baseline.

<i>River</i>	<i>Source report</i>	<i>GL extracted</i>	<i>Comments (rel = reliability)</i>
<i>Flinders B.1</i>	FGARA and Bayliss	40 (yield, 85% rel)	Cave Hill Dam
<i>Flinders B.2</i>	FGARA and Bayliss	34 (yield, 85% rel)	O'Connell Creek offstream Dam
<i>Flinders B.3</i>	FGARA and Bayliss	80 (entitlement 85%)	Minimum offstream extraction
<i>Flinders</i>	FGARA	160 (175, 70% - 80%)	Case study: 175 GL, 70% - 80% rel
<i>Flinders</i>	FGARA	240 (ent; 70% rel)	
<i>Flinders</i>	FGARA	400 (ent; 60% rel)	
<i>Flinders B.3 max</i>	FGARA and Bayliss	560 (entitlement)	Maximum offstream extraction
<i>Flinders B.1 max</i>	Bayliss et al. 2014	248 (capacity)	Cave Hill Dam
<i>Flinders B.2 max</i>	Bayliss et al. 2014	127 (capacity)	O'Connell Creek Dam
<i>Flinders B.3 max</i>	Bayliss et al. 2014	560 (entitlement)	All possible offstream extraction summed

For the Gilbert River, Bayliss et al. (2014) used dam yield (at ~85% annual reliability) as well as dam capacities (all dam water) as the extremes of scenario WRD usage.

Table A5. Water Resource Management scenarios in the Gilbert River catchment within scope (FGARA) and modelled by Bayliss *et al.* 2014. PLUS no WRD Baseline.

<i>River</i>	<i>Source report</i>	<i>GL extracted</i>	<i>Comments (rel = reliability)</i>
<i>Gilbert B.1</i>	FGARA and Bayliss	172 (yield, 85% rel)	Green Hills Dam
<i>Gilbert B.2</i>	FGARA and Bayliss	498 (yield, 85% rel)	Dagworth Dam (326 GL) & Green Hills Dam (172 GL)
<i>Gilbert B.3</i>	FGARA and Bayliss	17 (yield, 85% rel)	Kidston Dam – 2 m wall raise
<i>Gilbert B.1 max</i>	Bayliss et al. 2014	227 (capacity)	Green Hills Dam
<i>Gilbert B.2 max</i>	Bayliss et al. 2014	725 (capacity)	Dagworth Dam (498 GL) & Green Hills Dam (227 GL)
<i>Gilbert B.3 max</i>	Bayliss et al. 2014	25 (capacity)	Kidston Dam – 2 m wall raise
<i>Gilbert + Flinders</i>	Bayliss et al. 2014	additive of above	E.g. Cave Hill and Green Hills

NAWRA WRD impacts on estuarine habitats and processes.

The NAWRA WRD assessment did treat the estimation of WRD impacts on riverine and estuarine flora, fauna and fisheries in a comprehensive manner as a core component of the reports (Pollino et al. 2018 a,b).

The NAWRA WRD scenarios explored a greater array of water use protocols. For the Mitchell River, streamflow scenarios for both dams (seven possible sites) and water extraction (seven levels of extraction) were modelled (Table A3). In addition, facets such as trigger levels for pumping and pump capacity were also modelled, as well as single and cumulative dam impoundment scenarios.

Thresholds

Thresholds or trigger levels of streamflow above which pumping can be initiated have the capacity to maintain low-level streamflows in a river. Below a threshold, extractive pumping is not allowed to commence, or must cease. These thresholds are particularly important for low-level flows that might be characteristic of October to December flows that are less common but are important in the wet-dry tropics. Thresholds also preserve low-level flows in years of a failed monsoon. For the Mitchell River, NAWRA modelled streamflows with thresholds of 200 ML d⁻¹ and 1800 ML d⁻¹ below which pumping could not occur. The 200 ML d⁻¹ level was low and allowed extraction scenarios that did not preserve flows that had the capacity to modify the salinity regime of an estuary or provide ecosystem services (such as emigration cues) to an estuary, prior to the onset of the monsoon rainfall and high-level flows.

The 1800 ML d⁻¹ threshold preserved flows that could provide ecosystem services to the estuary, particularly modification of estuarine salinity that both caused a brackish estuary and cued emigration of banana prawns.

Pump extraction rates

Pump rates relate to the time taken to extract an allocation of water; and the characteristic of flow from which water might be extracted. Superimposed on NAWRA WRD scenarios (Hughes et al. 2017), a fifteen (15) day pump window and a 30 day pump (No: GL d⁻¹ pump rate) window were explored. High pumping rates that allow water allocation to be extracted over 5 days enables water to be taken from peak flows only, with a lesser level of impact on stream flow downstream of the offtake point (i.e. only 5% of a peak flood might be extracted). For example, a 5 day (No: GL d⁻¹ pump rate) extraction may mean a large flood peak is diminished minimally, with little downstream impact. In contrast, low pump rates that prolong the period of water allocation extraction to 30 days requires water to be taken from both the peak flow and most likely from the trailing end of a flood event after the peak-flows pass downriver. A 30-day extraction would have little impact on the peak flow level, but may mean a significant reduction in the levels and duration of the trailing end of the floodflow. Streamflow would be lower sooner, and the flood event would cease sooner. If water management allowed a combination of both a low trigger level for pump-extraction and extended periods of pump extraction, streamflow may cease much sooner than an unimpeded flood as the trailing end of a floodflow may be extracted in its entirety.

Both trigger levels and pump-rates were explored for each of the seven WRD extraction allocations investigated for the Mitchell River catchment (i.e. 300 - 6000 GL y⁻¹) (Table A6).

In the Mitchell River catchment, four instream dams were modelled individually. In conjunction, an additive series of dams were modelled, beginning with the Pinnacles Dam, then the Pinnacles and Rookwood Dams (a plausible high water storage capacity dam combination) and then adding five more dams sequentially.

To maintain environmental flows (e.g. low level flows), dams require offtake structural engineering that allows flows to pass the dam wall. Offtake outlets need to be staged at a gradation of vertical levels within the dam wall, to allow good quality water to be accessed to flow downstream. Stored water in a dam can become stratified; perhaps being cold and de-oxygenated in deep portions of a dam storage. As single offtake at the base of the dam wall would risk deterioration of the water quality along the downstream riverine and estuarine habitats.

Streamflow scenarios under both a drier and a wetter climate than current climate were also modelled.

Table A6. Water Resource Management scenarios in the Mitchell River catchment within scope and modelled during the NAWRA analyses (Pollino *et al.* 2018 a,b).

<i>River</i>	<i>Source report</i>	<i>GL extracted</i>	<i>Comments (rel = reliability)</i>
<i>Mitchell River</i>	NAWRA	300 (200) (85% rel)	Water extraction
	NAWRA	600 (yield, 85% rel)	Low-level trigger
	200 ML d	1200 (1000)	
	Trigger level (Threshold)	2400	
	@ both 5 (15) GL day	3600	
	and 30 GL day	4800	
	extraction	6000	
<i>Mitchell River</i>	NAWRA	300 (yield, 85% rel)	Water extraction
		600	Higher-level trigger
	1800 (1200, 2100) ML d	1200 (1000)	
	Trigger level (Threshold)	2400	
	@ both 5 (15) GL day	3600	
	and 30 GL day	4800	
	extraction	6000	
<i>Mitchell River</i>	Pinnacles Dam	1248 (yield, 85% rel)	Individual dams
	Rookwood Dam	575 (yield, 85% rel)	
	Palmer River Dam	553 (yield, 85% rel)	
	Elizabeth Creek Dam	55 (yield, 85% rel)	

Mitchell River	Pinnacles Dam	1248 (yield, 85% rel)	
	Pinnacles + Nulinga	1313	1248 + 65
	Pinnacles, Rookwood Dams	1823 (yield, 85% rel)	(1248 + 575)
	Pinnacles, Rookwood, Palmer River Dams	2376 (yield, 85% rel)	Three dams (1823 + 553)
	Pin, Rookwood, Palmer River, Lynd Dams	2883 (yield, 85% rel)	Four dams (2376 + 507)
	P, R, Palmer River, Lynd, Chillagoe Dams	3271 (yield, 85% rel)	Five dams (2883 + 388)
	P, R, PR, L, Chillagoe, Elizabeth Creek Dams	3326 (yield, 85% rel)	Six dams (3271 + 55)
	P, R, PR, L, C, Elizabeth Creek, Nulinga Dams	3391 (yield, 85% rel)	Seven dams (3326 + 65)
Mitchell River	Wetter climate		Climate scenario
	Current climate		
	Drier climate		Climate scenario

References

- Bayliss P, Buckworth R and Dichmont C (Eds) (2014) Assessing the water needs of fisheries and ecological values in the Gulf of Carpentaria. Final Report prepared for the Queensland Department of Natural Resources and Mines (DNRM). CSIRO, Australia.
- Petheram C, Watson I and Stone P (eds) (2013) Agricultural resource assessment for the Flinders catchment. A report to the Australian Government from the CSIRO Flinders and Gilbert Agricultural Resource Assessment, part of the North Queensland Irrigated Agriculture Strategy. CSIRO Water for a Healthy Country and Sustainable Agriculture flagships, Australia.
- Petheram C, Watson I and Stone P (eds) (2013) Agricultural resource assessment for the Gilbert catchment. A report to the Australian Government from the CSIRO Flinders and Gilbert Agricultural Resource Assessment, part of the North Queensland Irrigated Agriculture Strategy. CSIRO Water for a Healthy Country and Sustainable Agriculture flagships, Australia.
- Pollino CA, Barber E, Buckworth R, Cadiegues M, Deng A, Ebner B, Kenyon R, Liedloff A, Merrin LE, Moeseneder C, Morgan D, Nielsen DL, O'Sullivan J, Ponce Reyes R, Robson BJ, Stratford DS, Stewart-Koster B, and Turschwell M (2018) Synthesis of knowledge to support the assessment of impacts of water resource development to ecological assets in northern Australia: asset analysis. A technical report to the Australian Government from the CSIRO Northern Australia Water Resource Assessment, part of the National Water Infrastructure Development Fund: Water Resource Assessments. CSIRO, Australia

Pollino CA, Barber E, Buckworth R, Cadiegues M, Deng A, Ebner B, Kenyon R, Liedloff A, Merrin LE, Moeseneder C, Morgan D, Nielsen DL, O'Sullivan J, Ponce Reyes R, Robson BJ, Stratford DS, Stewart-Koster B, and Turschwell M (2018) Synthesis of knowledge to support the assessment of impacts of water resource development to ecological assets in northern Australia: asset descriptions. A technical report to the Australian Government from the CSIRO Northern Australia Water Resource Assessment, part of the National Water Infrastructure Development Fund: Water Resource Assessments. CSIRO, Australia.

Appendix 6 – Physical variables

The MICE includes a variety of physical variables which are used as drivers in the MICE model (Table A7). Further descriptions of the variables are provided below.

Table A7. Summary of physical variables used to drive components of the MICE model.

	Physical Variable	Temporal Scale	Spatial Scale	Impacted model groups
1	River flow	Daily for all years since 1990	Per model region	Prawns, mud crab, barramundi, sawfish, meiofauna, microphytobenthos, seagrass, mangroves
2	Southern Oscillation Index (SOI)	Monthly for all years since 1970	Index covers entire GoC	Mud crab
3	Sea Surface Temperature (SST)	Weekly average	Selected sites	Seagrass, microphytobenthos
4	Salinity	Weekly average	Selected sites	Meiofauna, microphytobenthos
5	Air temperature	Daily, for various years depending on site	Selected sites	Mud crab, mangroves
6	Sea level	Monthly	One site	Mangroves, mud crab* (*not yet done, possible to include later)
7	Solar exposure	Daily, for all years since 1990	Selected sites	Seagrass, mangroves, microphytobenthos
8	Cyclones	Since 1970	Region specific	Seagrass, mangroves

1) River flow

See main text and Appendix 5.

2) Southern Oscillation Index (SOI)

A monthly Southern Oscillation Index is available from the Australian Bureau of Metrology (BOM) for the period 1970 to 2020. Given SOI and sea level are correlated (White et al. 2014; Figure A23), albeit with some scatter, in the absence of sea level data, SOI could be used as a proxy (e.g. Plaganyi et al. 2020). Presently, we do not include SOI in the MICE given we have some actual sea level data (see (6) below).

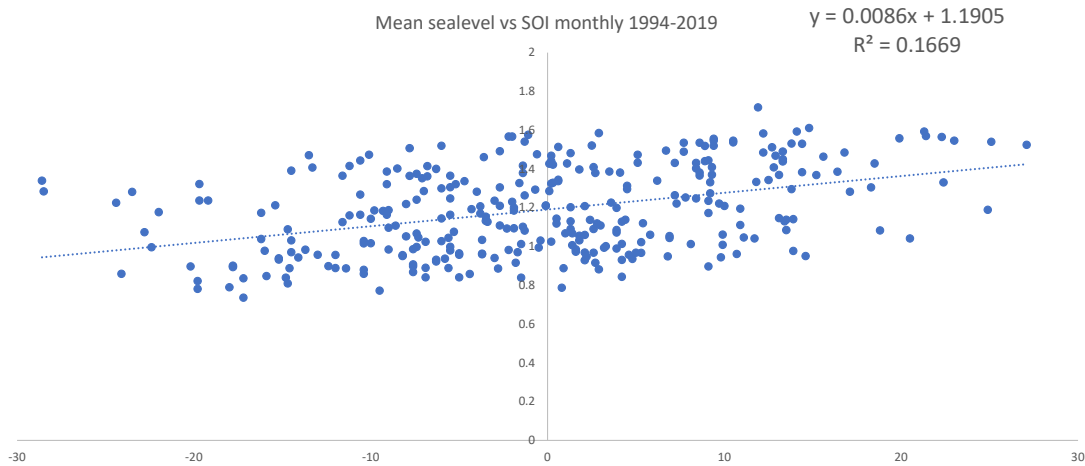


Figure A23. Mean sea level plotted against monthly SOI for the period 1994-2019.

3) Sea surface temperature (SST)

For two decades, CSIRO has located a salinity, temperature and depth logger on the A. Raptis and Sons’ wharf at their fishing fleet support facility in Karumba on the Norman River Estuary, south east GoC. The logger configuration has changed over the years with a YSI logger installed in July 2014 and continuing to 2021. The logger sensor is suspended from the wharf to be 1.5 m below the water level at low spring tides and about 1 m above the sediment. At high tide, the logger sensor is 3-4 m below the estuary water’s surface. The recording unit was located on a rigid post on the wharf. As it was fixed to a wharf, the logger is about 5 to 8 metres from the water’s edge, depending on tide level. The sediments below the logger sensor were always submerged.

To capture roughly the seasonal changes in SST, we extracted all available SST measurements from a logger attached to the Karumba wharf for the three years 2016 to 2018. Data are available on a fine-scale of every few minutes during the day, but this needed to be aggregated to match the model time step, and hence weekly averages were computed for each year.

The logger is located about 3 km upstream from the mouth of the Norman River and 2 km downstream from sites at which water quality, estuarine productivity and meiofauna abundance in the Norman River was measured in 2008/09 experiments (see Burford et al. 2012 and Duggan et al. 2014; Figure A24). The logger is located in a similar position laterally on the riverbank as the site where the experiments to investigate flux within the estuarine community was located. In addition, in 2008/09 estuarine water quality, productivity and meiofauna abundance also were measured at sites approximately 25 to 30 km upstream from the logger. Due to tidal influence, the estuarine waters measured at the logger passed upstream to influence environmental conditions at water quality field sites. The logger location is characteristic of environmental parameters measured at the experimental sites of Burford et al. (2012) and Duggan et al. (2014), and provides a continuous series of multiple observations per hour for several years.

Seasonal change in SST can be approximated by a sinusoidal curve. A sine curve as follows was thus fitted to the 119 available weekly SST measurements from the Karumba Wharf over 2016 to 2018.

$$\widehat{SST} = A \cdot \sin(B(\text{week} + C)) + D \quad (1)$$

Where A, B, C and D are respectively parameters representing the amplitude, period, phase shift and vertical shift. The model parameter estimates and Standard deviations are shown in Table A8.

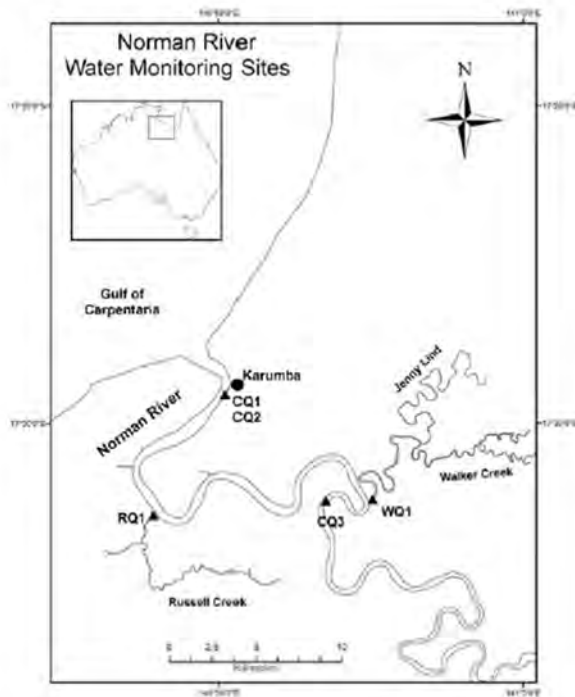


Figure A24. Water quality monitoring sites in the Norman River Estuary at which water quality, estuarine productivity and meiofauna abundance in the Norman River was measured in 2008/09 experiments. SST measurements are available from a temperature logger at Karumba. Figure taken from Duggan et al. 2014.

Table A8. Model estimates of parameters A, B, C and D which represent respectively the amplitude, period, phase shift and vertical shift of a SST-week relationship fitted to three years' data.

Parameter	Value	STD
A	4.73	0.09
B	0.15	0.00
C	4.50	0.37
D	26.08	0.07

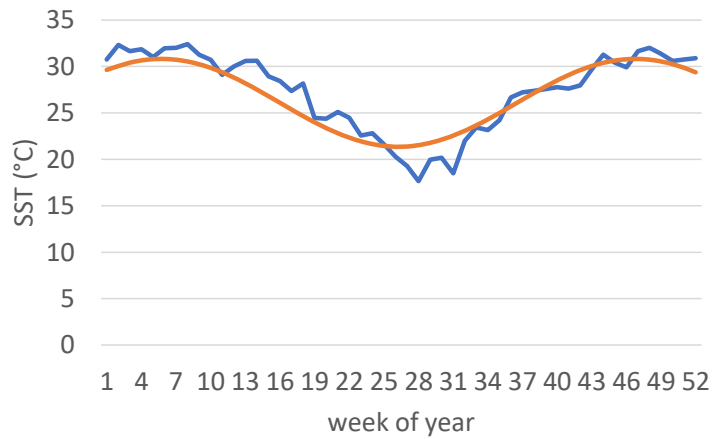
The model-estimated SST values are shown together with the actual observations for 2016 in Figure A25(A) below. To further validate use of this relationship to estimate SST, Equation (1) above was used to estimate the SST corresponding to each of the field observations of Duggan et al. (2014) taken on the Norman River, and after matching the sampling date to week of the year. Equation (1) is seen to reasonably estimate the SST as measured at the data sampling sites on the mudflats (Figure A25(B)).

Climate-related increases in temperature (specifically maximum values) can be readily incorporated by increasing the parameter A in Equation (1) hence it is a useful equation also for simulating alternative climate scenarios. Similarly, the model assumes the same temperature relationship for all eight regions but this could be adjusted if more data become available.

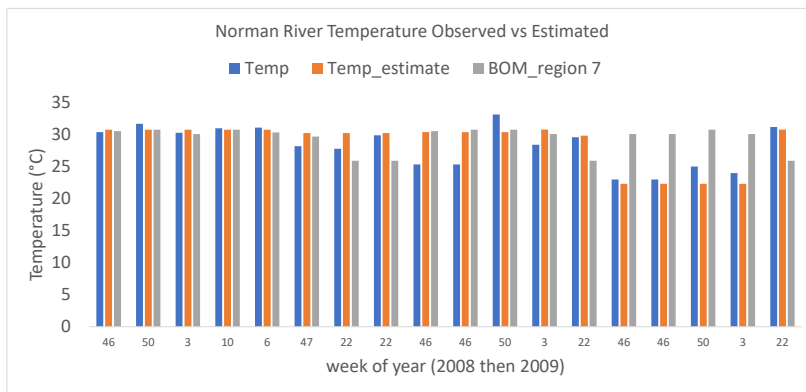
There are also monthly water temperature data available from BOM (<http://www.bom.gov.au/oceanography/projects/abslmp/data/monthly.shtml#table>) for Milner Bay, Groote Eylandt, which corresponds to Region 7 in the MICE. These data are useful not only because they provide monthly means from 1994 to 2020, but they also include the minimum and maximum values as these might be more relevant than the mean in influencing the dynamics of selected species groups.

The mean BOM observations were compared with the Norman River observations, and the predictions using Equation (1) above, as shown in Figure A25(C). The BOM data are broadly representative of temperatures measured during corresponding months at the Norman River, but over-estimate the SST in some periods. This may be due to a number of factors. The BOM SST data series was also compared with the Equation (1) model estimates and the equation is seen to reproduce the seasonal variation in SST, but with a different lower SST (Figure A25(C)). Equation (1) in its current form captures 83.8% of the variation in the BOM SST. Equation (1) can be refitted using the Groote Eylandt data, but the preferred approach here is to use Equation (1) SST model estimates because it more closely simulates the Norman River observations, which in turn are considered representative of the SST that model organisms and juvenile prawns will be subject to. However, an alternative scenario will also be run using the BOM mean SST observations, as well as the maximum observed values.

(A)



(B)



(C)

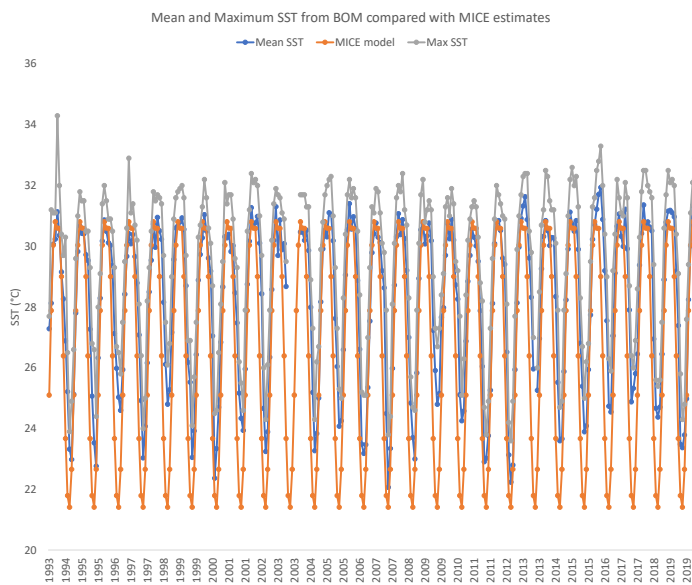


Figure A25. (A) Weekly observed vs. modelled SST values for 2016; (B) modelled temperature compared to observed temperature in the Norman River (temperature logger) and observed temperature (BOM Groote Eylandt station representative of model region 7). (C) Mean and maximum SST from the BOM compared with model estimates.

4) Salinity

The Duggan et al. (2014) study includes a time series and a total of 17 field measurements of salinity, temperature and a number of other variables at sites in the Norman River (see section 3 SST above). The sampling date of each observation was matched with the river model end of system flow estimates for that week, and the corresponding average flow was computed and converted to a standardised flow anomaly by dividing by the average flow for that week (based on all years 1900 to 2019). As expected, salinity is seen to decrease as flow volume increases given that more freshwater is added from the river to the estuarine environment. This relationship can be represented using a fitted exponential equation as shown in Figure A26(A). Note that this is a rough average only as in each instance these values can be expected to vary based on distance from the river mouth as well as factors such as temperature, but this is considered adequate for current purposes which need to capture the steep drops in salinity that occur during high flows.

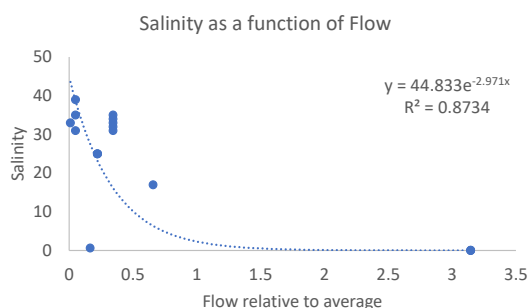
The simple salinity-flow relationship is as follows:

$$\widehat{sal} = \varphi \cdot e^{-\omega V} \quad (2)$$

Where estimates of the fitted parameters φ and ω are 44.833 and 2.971 respectively, and V is the standardised weekly flow.

To further validate the use of Equation (2) to estimate salinity based on standardised flow for each week in the model, we compared model-predicted salinities with 411 actual observations from the Karumba wharf for years 2017 to 2018, as shown in Figure A26(B). Equation (2) is considered to provide an adequate approximation to represent changes in salinity as a function of flow.

(A)



(B)

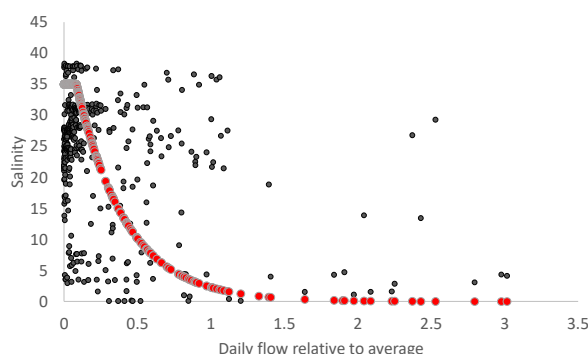


Figure A26. (A) Salinity from Duggan et al. (2014) as a function of flow and (B) salinity from the Karumba Wharf logger as a function of flow.

5) Air temperature data

Elevated temperatures or periods of heating can negatively affect some species in the GoC, e.g. as was recently seen with mangrove dieback in the 2015/2016 Extreme Climatic Event (Duke et al. 2016; Babcock et al. 2019). Mud crabs are also susceptible to periods of heating during the summer months when temperatures are high and tides are low (Robins et al. 2020).

In many parts of the GoC estuaries there are no direct measurements of water temperature, and SST measurements from the river mouth may not be useful when relating it to organisms that live elsewhere in the estuary e.g. mud crabs, mangroves. Robins et al. (2020) found that air temperature was a reasonable proxy for water temperature in the estuaries. We therefore used air temperature as one of the physical drivers for mud crabs and mangroves.

Daily maximum air temperature data are available from BOM for various sites and time periods around the GoC (<http://www.bom.gov.au/climate/data/>). Many of these sites do not have a complete timeseries available for the model period, or those that do, are from two nearby, but different, stations. Hence, we extracted data for stations that were considered representative of model regions and that had data available for more recent time periods i.e. up until present day. These are summarised in Table A9.

Table A9. Summary of BOM Stations and period for air temperature data including which model regions these data were used for.

Site (BOM station number)	Period	Represented Model Region	Comments
Weipa Aero (027045)	1994-2021	1	
Sweers Island (029139)	2001-2021	2-6	These data currently used to represent the south east GoC in the model (regions 2-6)
Mornington Island Airport (029182)	2013-2021	2-6	Shorter time series, suggest don't use this site and instead use Sweers Island.
Centre Island (014703)	1975-2021	7	Good time series length but Groote Eylandt might be more representative of this region? Centre Island temperatures were more similar to sites in SE hence have not used these.
Groote Eylandt Airport (014518)	1999-2021	7-8	These data currently used to represent the the western GoC in the model (regions 7-8)

The average maximum temperature for each week and month of the year was calculated from the daily data to create a weekly and monthly time series for each site as a representative for that model Region(s) (Table A9). Weekly and monthly climatologies (long-term means) for each site were used to replace gaps in the time series, particularly for the earlier periods in the model where data were missing. A comparison of mean maximum weekly air temperatures for a randomly selected period showed that, as expected, temperatures are different for different parts of the GoC, e.g. there is less variability at Weipa (north-east GoC) compared to Groote Eylandt and Sweers Island further south (Figure A27). We thus selected stations for which data were available and were considered to be representative of the model region (Figure A27).

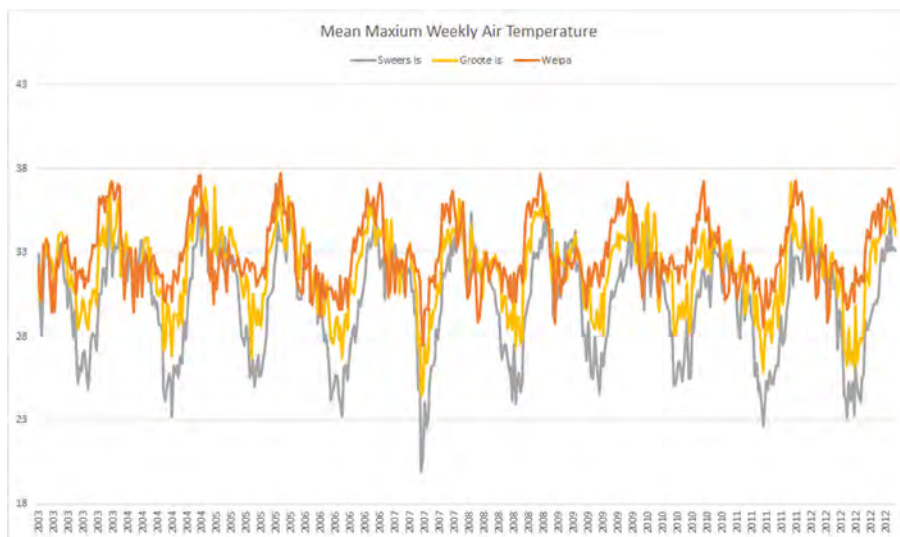


Figure A27. Snapshot of mean maximum weekly air temps for three sites: Weipa (for Region 1), Sweers Island (for Regions 2-6) and Grootte Eylandt (for Regions 7-8).

6) Sea Level

Monthly sea level data are available from BOM for Grootte Eylandt (Figure A28). As it is the anomalously low sea levels that are thought to influence mangroves (Duke et al. 2017, Lovelock et al. 2017a) and prawn recruitment (Plaganyi et al. 2020), we use the minimum sea level per month as the model inputs.

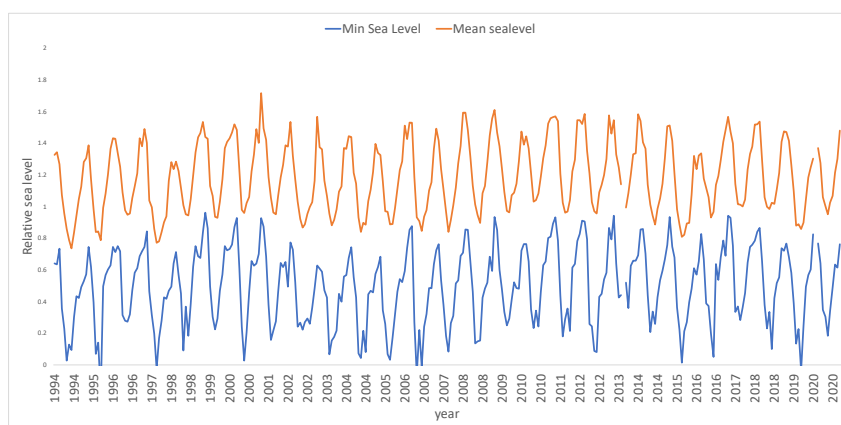


Figure A28. Mean and minimum sea level measurements obtained from the BOM for Milner Bay, Grootte Eylandt for the period 1994-2020.

7) Solar Exposure

To capture seasonal and other changes in the amount of solar exposure that is available to support the growth of phytoplankton and plant species, we use BOM measurements of daily solar exposure that measure the total solar energy falling on a horizontal surface from midnight to midnight (<http://www.bom.gov.au/climate/data/index.shtml?bookmark=193>). Given regional differences, we selected three sites: Weipa (representing area 1 in our model), Karumba airport (representing model

areas 2-6) and Groote Eylandt (models areas 7-8) (Figure A29;Figure A30). Data were available from 1990 to present day, and were used to compute weekly averages for each of these three areas over all years since 1990. The 1990 averages were used for years pre-1990. For the few instances with missing data, we calculated a representative value for that week and area as the average of the equivalent values from each of the two preceding years.

The maximum daily global solar exposure (MJ/m*m) from these data was 31.1 and hence we standardized weekly solar exposure averages so that they represent the proportion of the maximum observed solar exposure.

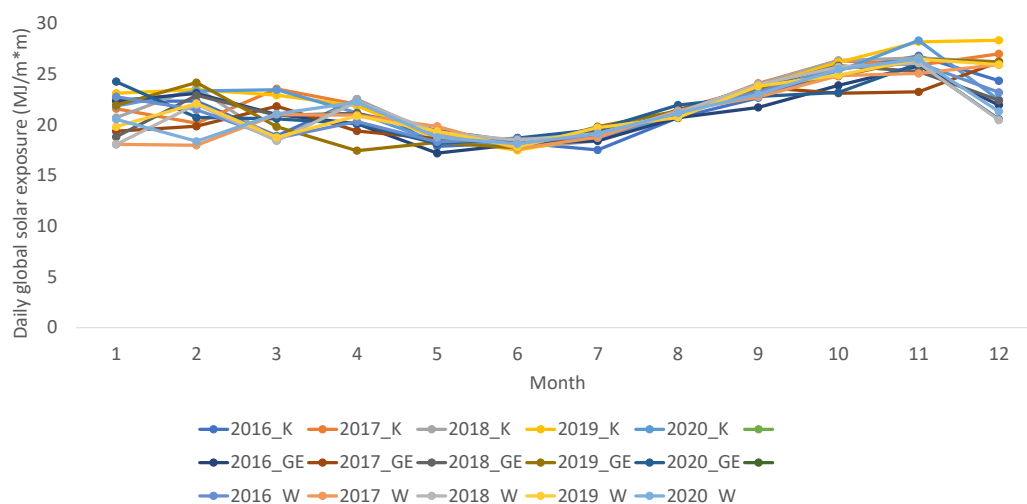


Figure A29. Example of the inter-annual and regional variability in the solar exposure for each month of the year, highlighting also the clear seasonal signal of less solar exposure during the austral winter.

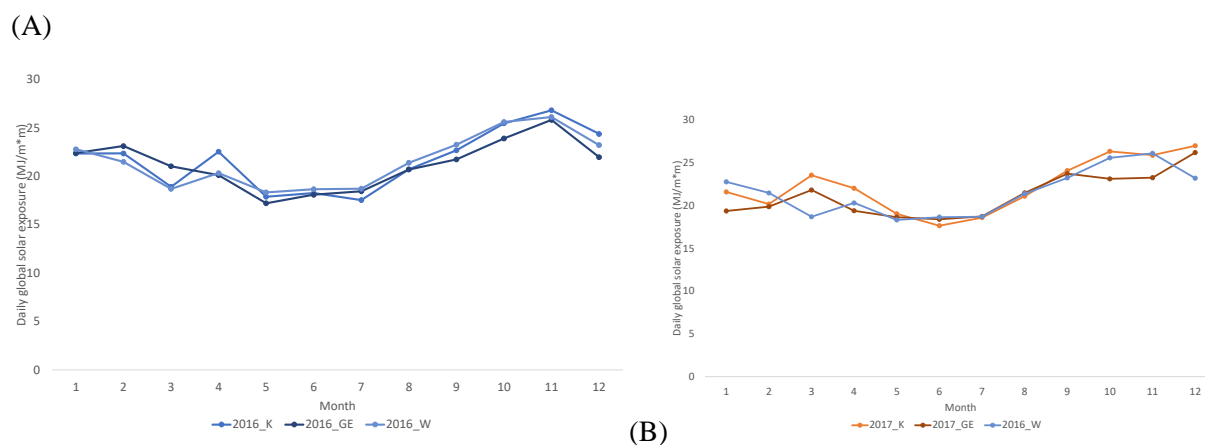


Figure A30. Example of the solar exposure pattern for (A) 2016 and (B) 2017 compared for sites Karumba (K), Groote-Eylandt (GE), Weipa (W).

We use the actual observations (after computing standardised weekly averages) as inputs to our model, because there is a fairly substantial amount of variation in these values (e.g. see scatter in Figure A31). However for the purposes of forward projecting the model, we used data for the period 2016 to 2020 and fitted a sinusoidal curve to approximate the seasonal pattern of solar exposure. The formula for the sine curve was follows and the example here is fitted to 260 weekly standardised averages:

$$\widehat{solar} = A \cdot \sin(B(\text{week} + C)) + D \quad (1)$$

Where A, B, C and D are respectively parameters representing the amplitude, period, phase shift and vertical shift. The model parameter estimates and standard deviations are shown in Table A10.

Table A10. Model estimates of parameters A, B, C and D which represent respectively the amplitude, period, phase shift and vertical shift of a solar-week relationship fitted to five years' data.

Parameter	Value	STD
A	0.41	5.75
B	0.05	0.37
C	76.83	812
D	1.00	5.80

The model-estimated solar values are shown together with the actual estimates for two selected series, 2016 and 2020 in Figure A31 below.

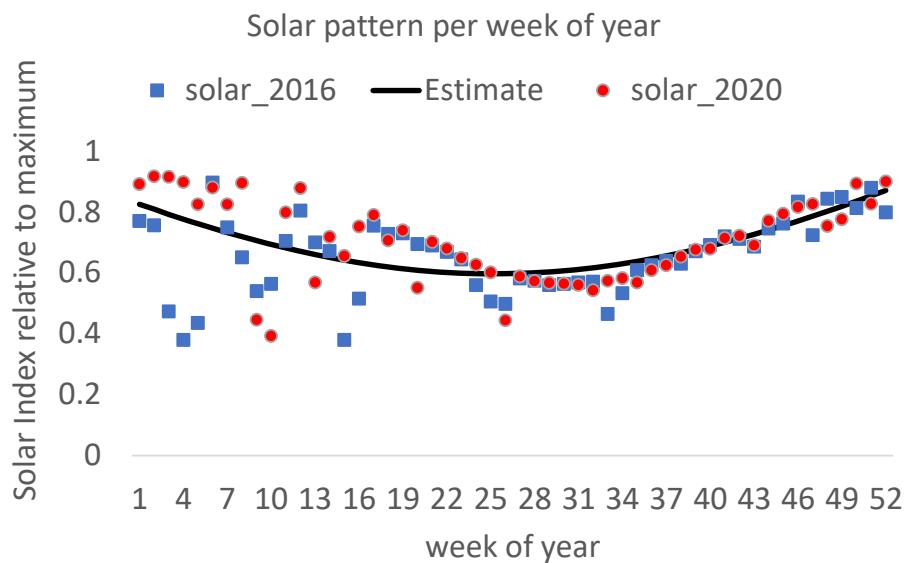


Figure A31. Illustrative example of fitted solar exposure annual pattern compared with 2016 and 2020 data (after standardisation) from the Karumba site.

8) Cyclones

Tropical cyclones of varying intensity occur in the GoC region, and can sometimes have significant negative effects on the mangrove and seagrass systems (Poiner et al., 1989). These littoral habitats are in turn critical nursery habitats for species such as tiger prawns and fish (Brewer et al. 1995, Loneragan et al. 1998). Cyclones that formed in or entered the Gulf of Carpentaria (GoC) from 1970

to 2019 were categorised on the basis of intensity and they were allocated to one of the eight GoC model regions used in the MICE (Table A11). Two sources of information were used to extract information.

Information on cyclones in the 1970s and 1980s was derived from a written document developed by the Bureau of Meteorology that described cyclones qualitatively from 1885 to 2004 (a summary prepared by Jeff Callaghan, BOM, 'Known tropical cyclone impacts in the Gulf of Carpentaria' (<https://www.australiasevereweather.com/cyclones/impacts-gulf.pdf>)). From 1885 to recent decades, the descriptions in the document increased in complexity and made greater use of observations from technological tools. The GoC was and continues to be a sparsely populated expanse of tropical Australia. In the early- to mid-1900s, reports were made from remote cattle stations or aboriginal townships (e.g. Ngukurr (Roper River Mission) and Pormpuraaw (Edward River Mission)) further north, and a few towns in the southern GoC catchments such as Normanton and Burketown. During the later decades of the 1900s, mining and fishing towns in the GoC such as Karumba (south east) Alyangula (Groote Eylandt, north west) and Weipa (north east) also provided detailed descriptions of cyclones, usually with the benefit of new technology. The descriptions were qualitative, but often detailed, and aspects such as the height/ depth/ extent of the storm surge and the damage/ demolition/ removal/ transport of various pieces of built-infrastructure were described. These descriptions assisted with allocating a wind-strength category and a point-of-crossing of the coast to the cyclone. However, the level of precision of a category interpreted from the early qualitative descriptions may be low.

From 1997 to 2021, remotely sensed images of Australian cyclone tracks have been documented by the BOM. They are presented as maps available on the BOM web site (<https://www.australiasevereweather.com/cyclones>). The tracks are accurately plotted and show the exact cyclone track and location where it crossed the coast. In addition, the tracks are colour-coded by wind strength category, so the cyclone wind strength category at the precise location where it crossed the coast is known.

From 1997 to 2004, both types of cyclone-documentation exist, so cross referencing of qualitative with quantitative descriptions was made. In some cases, the MICE region where the cyclone crossed the GoC coast was updated as the mapped tracks were more accurate than the qualitative on-ground observations.

The likely impact of a cyclone was estimated using two variables; cyclone intensity category, and the direction of impact of the cyclone track relative to the coast.

Cyclone intensity had six categories, a tropical low (category = 0.5) and then categories 1 to 5 as per the BOM definition of cyclonic wind strength. The direction of cyclone track had four categories:

- 0.25 - a cyclone crossing from land to water (often the case from the Coral Sea across Cape York to the GoC),
- 0.5 - a cyclone crossing the coast from water to land at a perpendicular angle,
- 0.75 - a cyclone crossing the coast from water to land at an oblique angle,
- 1 - a cyclone travelling parallel to and close to the coast before crossing from water to land at any angle.

References

Babcock, R. C., R. H. Bustamante, E. A. Fulton, D. J. Fulton, M. D. Haywood, A. J. Hobday, R. Kenyon, R. J. Matear, E. E. Plaganyi, and A. J. Richardson. 2019. Severe continental-scale impacts of

climate change are happening now: extreme climate events impact marine habitat forming communities along 45% of Australia's coast. *Frontiers in Marine Science*:411.

Brewer, D., S. Blaber, J. Salini, and M. Farmer. 1995. Feeding ecology of predatory fishes from Groote Eylandt in the Gulf of Carpentaria, Australia, with special reference to predation on penaeid prawns. *Estuarine, Coastal and Shelf Science* 40:577-600.

Burford, M., I. Webster, A. Reville, R. Kenyon, M. Whittle, and G. Curwen. 2012. Controls on phytoplankton productivity in a wet-dry tropical estuary. *Estuarine, Coastal and Shelf Science* 113:141-151.

Duggan, M., R. Connolly, M. Whittle, G. Curwen, and M. Burford. 2014. Effects of freshwater flow extremes on intertidal biota of a wet-dry tropical estuary. *Marine Ecology Progress Series* 502:11-23.

Duke, N. C., J. M. Kovacs, A. D. Griffiths, L. Preece, D. J. Hill, P. Van Oosterzee, J. Mackenzie, H. S. Morning, and D. Burrows. 2017. Large-scale dieback of mangroves in Australia's Gulf of Carpentaria: a severe ecosystem response, coincidental with an unusually extreme weather event. *Marine and Freshwater Research* 68:1816-1829.

Loneragan, N. R., R. A. Kenyon, D. J. Staples, I. R. Poiner, and C. A. Conacher. 1998. The influence of seagrass type on the distribution and abundance of postlarval and juvenile tiger prawns (*Penaeus esculentus* and *P. semisulcatus*) in the western Gulf of Carpentaria, Australia. *Journal of Experimental Marine Biology and Ecology* 228:175-195.

Lovelock, C. E., I. C. Feller, R. Reef, S. Hickey, and M. C. Ball. 2017. Mangrove dieback during fluctuating sea levels. *Scientific reports* 7:1-8.

Plagányi, É., R. A. Deng, T. Hutton, R. Kenyon, E. Lawrence, J. Upston, M. Miller, C. Moeseneder, S. Pascoe, and L. Blamey. 2021. From past to future: understanding and accounting for recruitment variability of Australia's redleg banana prawn (*Penaeus indicus*) fishery. *ICES Journal of Marine Science* 78:680-693.

Poiner, I., C. Conacher, N. Loneragan, and I. Sonters. 1993. Effects of cyclones on seagrass communities and penaeid prawn stocks of the Gulf of Carpentaria.

Robins, J. B., A. R. Northrop, M. Grubert, and R. C. Buckworth. 2020. Understanding environmental and fisheries factors causing fluctuations in mud crab and blue swimmer crab fisheries in northern Australia to inform harvest strategies.

White, N. J., I. D. Haigh, J. A. Church, T. Koen, C. S. Watson, T. R. Pritchard, P. J. Watson, R. J. Burgette, K. L. McInnes, and Z.-J. You. 2014. Australian sea levels—Trends, regional variability and influencing factors. *Earth-Science Reviews* 136:155-174.

Table A11. Summary of tropical cyclones recorded in the Gulf of Carpentaria from 1970-2019, including their estimated intensity (low or cyclone category 1-5) and allocation to MICE model region (1-8).

Year	Month	Date	Name	Category	Region	Direction	Storm Surge	Comments (landfall)
				Estimated				
1970	February	11 th	Dawn	1	1	unknown	unknown	Weipa, 52 knot gusts
1970	March	17 th to 18 th	Cindy	1/2	7	perpendicular	unknown	Port Roper, 910 mm rainfall- 3 days
1971	February	2 nd to 3 rd	Aggie	1/2	7	unknown	unknown	Vanderlin Island, gusts to 61 knots
1971	February	19 th	Fiona	3/4	2	perpendicular	0.9 to 3 m	Nassau River, infrastructure scattered like confetti
1972	January	6 th	Bronwyn	1	1	unknown	unknown	Mapoon, no damage
1972	April	13 th	Faith	1	1	unknown	unknown	Weipa, gale report, 256 rain 24 h
1973	January	29 th	Adeline	4/5	7	unknown	unknown	Vanderlin Island (970 hPa)
1973	March	1 st	Leah	1	7	unknown	unknown	Groote Eylandt/Numbulwar, minor damage
1973	March	5 th to 6 th	Madge	2/3	2 and 7	perpendicular	unknown	Groote Eylandt, Numbulwar (985 hPa), buildings damaged
1973	March	25 th	Bella	1	7	unknown	unknown	Edward Pellew Islands , 998 hPa.
1974	February	11 th to 14 th	Yvonne	1/2	4 and 7	unknown	unknown	Centre Island, Borroloola, some damage
1974	March	17 th	Jenny	1	8	unknown	unknown	Gove, 995 hPa.

Year	Month	Date	Name	Category	Region	Direction	Storm Surge	Comments (landfall)
1975	December	10 th	Kim	1	2	unknown	unknown	Edward River Mission (Pormpuraaw)
1976	December	19 th	Ted	4/5	5	perpendicular	2 - 3 m, 30 km inland	Mornington Island (95% building damage) metal telephone posts bent to ground level
1977	March	7 th	Otto	1	2	unknown	unknown	between Aurukun and Weipa, no damage,
1978	February	27 th	Gwen	2/3	2	perpendicular	waves over dunes	Edward River Mission, wind gusts to 70 knots
1978	April	7 th	Hal	1	2	perpendicular	unknown	80 km south of Aurukun, no damage
1978	December	31 st	Peter	1	2	perpendicular	unknown	Edward River Mission (Pormpuraaw)
1979	January	10 th	Greta	1	1	perpendicular	unknown	20 km south of Weipa, 42 knot wind gusts
1979	February	26 th	Rosa	2/3	7	unknown	~ 2.0 m	30 km NNW of Bing Bong; wind gusts 91 knots
1979	April	14 th	Stan	1/2	1	perpendicular	0.5 m	40 km N of Weipa, 80 - 200 mm rainfall- Cape York
1980	March	17 th	Doris	1/2	8	perpendicular	unknown	between Gove and Groote Eylandt, minor damage
1981	February	10 th to 11 th	Eddie	1/2	6	unknown	1.5 m	Old/NT border, gusts to 77 knots
1981	December	20 th	unnamed	1/2	8	unknown	unknown	Gove, average winds 64 knots.
1982	April	7 th	Dominic	2/3	2	perpendicular	1.5 m	Cape Keerweer

Year	Month	Date	Name	Category	Region	Direction	Storm Surge	Comments (landfall)
1984	March	10 th	Jim	2/3	7	unknown	1.5 m	Port Roper
1984	March	23 rd	Kathy	5	7	perpendicular	3 m to 7 km inland	Crosses Centre Island, Borroloola, av. 100 knot wind, gusts 125 knots
1985	February	22 nd	Rebecca	1/2	1	perpendicular	unknown	north of Weipa, trees (60 cm dia) blown down
1985	March	24 th	Sandy	4	7	parallel	3 to 3.5 m	Pellews, parallel to coast to Roper River, 130 knot gusts, 90 knot av., 12 m swell sheltered waters
1987	January	20 th	Irma	1/2	7	perpendicular		crossed coast at Roper Bar, 60 knot winds
1987	February	9 th to 13 th	Jason	2/3	8 and 5	oblique	2.04 m	Groote Eylandt, Baniyalla (Blue Mud Bay), reformed GoC to Burketown, av. Wind 64 knots
1989	December	15 th	Felicity	2/3	3	perpendicular	2.5 m	Gilbert River, SE GoC
1990	March	5 th	Greg	1	5	perpendicular	unknown	Normanton Sweers Island, 50 knit wind gusts
1992	January	10 th	Mark	1/2	1	perpendicular	unknown	Weipa, 63 knot winds
1992	December	25 th	Nina	2/3	2	perpendicular	unknown	Cape Keerweer
1994	January	30 th	Sadie	1/2	2	perpendicular	unknown	Inkerman Station (Nassau River, Staaten River)
1995	March	6 th	Warren	2/3	6	perpendicular	1.5 - 2.5 m	Mornington Island, eye over Gununa;

Year	Month	Date	Name	Category	Region	Direction	Storm Surge	Comments (landfall)
1996	January	6 th	Barry	2/3	3	perpendicular	> 4 m, 7 km inland	between Staaten and Gilbert Rivers
1996	January	28 th	Jacob	1	7	perpendicular	unknown	Bing Bong
1996	March	9 th	Ethel	2/3	1	perpendicular	1.18 m	Duyfken Point/ Weipa
1996	December	26 th	Phil	1	8	unknown	nil	Groote Eylandt, 35 knot winds, 116 mm rainfall
1997	December	27 th to 28 th	Sid	1	6	perpendicular	unknown	Mornington Island as weak system, 340 mm rain
1998	January	25 th	Les	2	7	perpendicular	1 m	Groote Eylandt (gust 90 knots) / Numbulwar
1998	February	26 th	May	1	6	perpendicular	unknown	Mornington Island, 1065 mm rain-7 d Burketown
1998-1999	Dec/Jan	27 th - R7; 29 th - R6	Low #2	low	6 and 7	parallel	unknown	SW GoC as a Tropical Low
1999	February	26 th	199916	low	na	nil landfall	unknown	Cape Wessels to central GOC, no landfall
1999	February	12 th	Rona	low	3	perpendicular	unknown	Cape York from the east, did not enter GoC
1999	April	24 th	Low #4	low	8	perpendicular	unknown	crossed northern GoC and made landfall at Gove
2005	February	7 th	Harvey	Cat 3	7	perpendicular	unknown	Old/ NT border tracking south from central GoC.
2005	March	10 th to 12 th	Ingrid	2 and 4	2 and 8	perpendicular	unknown	crossed Cape York to Aurukun (2) to Gove (4 to 5)

Year	Month	Date	Name	Category	Region	Direction	Storm Surge	Comments (landfall)
2006	April	20 th to 23 rd	Monica	2 and 5	2 and 8	perpendicular	unknown	crossed Cape York to Aurukun (2) to Gove (Cat 5)
2007	February	5 th to 7 th	Nelson	Cat 2	7 and 3	perpendicular	unknown	Arnhemland to Vanderlins to Gilbert River
2008	January	6 th	Helen	low	8 and 2	perpendicular	unknown	Arnhemland, Blue Mud Bay, to south of Aurukun
2009	January	12 th	Charlotte	Cat 1	3	perpendicular	unknown	Mornington to Vanderlins, east to Gilbert River
2010	January	18 th	Neville	low	1	perpendicular	unknown	Wessels/ NW GoC, to Pera Head, Weipa
2010	January	29 th	Olga	Cat 1	5	oblique	unknown	Vanderlins to Mornington Island/ west of Karumba
2010	March	30 th	Paul	Cat 1	8	perpendicular	unknown	Arafura Sea, Nhulunbuy to Blue Mud Bay
2011	December	28 th	Grant	low	7	perpendicular	unknown	Groote Eylandt from the west
2013	January	21 st	Oswald	low	6 and 2	perpendicular	unknown	Qld/NT border then reversed to Kowanyama.
2013	March	11 th to 13 th	Tim	low	8 and 1	perpendicular	unknown	Arnhemland via Blue Mud Bay to Cape York
2013	April/May	2 nd May	Zane	low	1	perpendicular	unknown	northern tip of Cape York
2013	November	26 th to 28 th	Alessia	low	8 and 7	oblique	unknown	Arnhemland via Blue Mud Bay to Vanderlins
2014	March	10 th to 12 th	Gillian	low	2	parallel	unknown	west Cape York coast (parallel)
2015	February	15 th	Lam	low	1	perpendicular	unknown	Cape York from the east at Weipa to Arafura Sea

Year	Month	Date	Name	Category	Region	Direction	Storm Surge	Comments (landfall)
2015	March	21 st to 22 nd	Nathan	1 and 2	2 and 8	perpendicular	unknown	Cape York from the east, Aurukun to Nhulunbuy
2016	March	16 th	201616	1	3	perpendicular	unknown	Bold Point
2017	January	22 nd	201703	low	7	perpendicular	unknown	Limmen Bight
2017	February	17 th to 21 st	Alfred	low	6 and 7	parallel	unknown	Mornington Island, veered to Roper River, then to Qld/NT border
2018	March	24 th to 25 th	Nora	3 and low	2 and 5	parallel	unknown	Pormpuraaw/Kowanyama (parallel), then reversed to Mornington Island.
2018	December	11 th to 14 th	Owen	3/2	7 and 2	perpendicular	unknown	Limmen Bight, then reversed to Gilbert River
2019	January	1 st	Penny	1	1	perpendicular	unknown	Weipa
2019	February	1 st	Oma	low	6	perpendicular	unknown	NT/Qld border and crossed the GoC coast
2019	March	21 st to 23 rd	Trevor	2 and 4	2 and 7	perpendicular	unknown	Cape York from the east, Pera Head to Vanderlins
2019	May	16 th	Ann	low	1	perpendicular	unknown	Cape York from the east at Weipa to Wessels

Appendix 7 – Common Banana Prawns background information

Life History of banana prawns

The larval life history of common banana prawns renders them critically dependent on estuarine juvenile habitats (Dall et al. 1990). After being spawned at sea, developing postlarvae typically advect inshore using a combination of tidal currents and behaviour that optimises inshore movement. Pelagic banana prawn postlarvae settle in the upper reaches of estuarine tributary habitats where they accumulate at high densities within the ‘mudbank/mangrove forest matrix’ that exists in tropical estuaries (Vance et al. 1998, Kenyon et al. 2018, Vance and Rothlisberg 2019). In northern Australia, immigration to coastal estuaries occurs from September to March; the first cohorts entering estuarine habitats from September to December prior to the wet season when estuarine waters are often hypersaline and $> 30^{\circ}\text{C}$. Planktonic and epibenthic algae and meiofauna form components of the foodweb which sustains the first-settled benthic-postlarval and juvenile population (Burford et al. 2012, Duggan et al. 2014). Internal nutrient cycles drive estuarine primary production in rivers in the wet/dry tropics due to minor terrigenous-sourced nutrient inputs.

Juvenile common banana prawns reside within the estuary for 2-4 months; residence time often dependent of freshwater inflows to the estuary. Within the estuary, growth and natural mortality is dependent on environmental conditions. Both laboratory experimentation and temporal analyses of field-sampled populations have shown that survival and growth is lower in either hypersaline or freshwater conditions, as well as outside moderate temperatures ($\sim 20\text{-}28^{\circ}\text{C}$), and at high individual densities (Staples and Heales 1991; Wang and Haywood 1999). In addition, juvenile populations suffer very high mortality due to fish predation (initially up to 90% wk⁻¹, $\sim 3\text{-}5$ mm CL), reducing as juveniles grow ($\sim 20\%$ wk⁻¹ at 15 mm CL). Wet-season river flows cue emigration of the juvenile population from estuarine to marine habitats (Vance et al. 1998, Kenyon et al. 2018) where mortality is lower (Lucas et al. 1979, Gwyther 1982) and a flood plume nutrient dump supports nearshore production (Burford et al. 2012) and hence the emigrant prawn population. In addition, estuarine river flows reduce the abundance of benthic algae and meiofauna, modifying the food resources of juvenile common banana prawns and providing further impetus to emigrate (Duggan et al. 2014, 2019).

From the early days of the Gulf of Carpentaria (GoC) banana prawn fishery, a relationship between flow and catch was recognised. A one in a 100 year flood and 12000 tonne catch in 1974 remains the most significant environmental cue and resultant catch in the history of the fishery; a catch that cued interest in the impact of environmental drivers on common banana prawns and a vision of monetary profit.

Historical attempts to explore the links between annual river flow in tropical rivers and adjacent banana prawn catch have used catchment rainfall as a proxy for river hydrology and river flow (Vance and Staples 1985, Vance et al. 2003, Buckworth et al., 2014). In part, the aim of the models was to predict catch for the upcoming fishing season and gauged river flow data were available only after fishing occurred; too late for predictive modelling (Venables et al. 2011). Models were constructed using monthly or seasonal rainfall data; attempting to include a temporal dimension of floodflow which may assist in interpreting the impact of flow on the early or later estuarine phase of juvenile prawns.

For much of the history of the fishery, hydrological data for the river catchments were poor. However, recent river modelling for the FGARA and NAWRA assessments have made available robustly-modelled river flow data series (1890 – 2011 or 2015, Lerat et al. 2013, Hughes et al. 2018) to fishery modelers and over the last three years, new models have deployed these data to explore

temporal and dimensional aspect of annual flows. Daily discharge data are available, grouped weekly, monthly and seasonally.

Catch Prediction

In the 1980s and 2000s, Vance et al. (1985 and 2003) demonstrated that total annual flow and seasonal components of flow (seasonal or monthly steps) was significant in driving commercial banana prawn catch. The relationship between flow and catch varied regionally; high-level wet season flows are the most significant drivers of catch across most regions (up to 70%), but prior to the wet season, 'spring' flows and November flows also contribute significantly to some regional catches (Vance et al. 1985, 2003). The temperature of Gulf of Carpentaria waters and wind speed and direction during juvenile recruitment are also contributory factors to eventual commercial catch.

Buckworth et al. (2014) refined a model developed by Venables et al. (2011) where estimated banana prawn catch tracked actual catch with an R-squared of 0.926. The primary driver of the model was rainfall, with 'early' and 'late' seasonal components which corresponded to pre-wet and wet season rainfall (weighted monthly).

During the mid-to-late 2010s, river flow data became available for key GoC catchments and Bayliss et al. (2014) and Broadley et al. (2020) used estimated river flow from recently-developed catchment hydrological models to predict reduction in catch due to the loss of the natural flow regime due to water extraction for use in irrigated agriculture. In the Gilbert and Flinders Rivers, under a maximum extraction scenario, the mean estimated reduction in fishery catch was 11% (3–13%) for common banana prawns and 16% (4–19%) for barramundi in the Gilbert River (Bayliss et al. 2014). A prominent extraction scenario estimated a mean 7% (2–8%) likely catch reduction for prawns and a mean 10% (3–12%) likely catch reduction for barramundi in the Gilbert River. Importantly, Broadley et al. (2020) demonstrated a much greater impact on prawn catch when water is extracted from low-level flows (50% impact) than high level flows (9%). They highlight a key point that a likely low prawn catch in a dry year of low flow would be proportionally severely reduced, perhaps to levels that are uneconomic to fish.

Habitat use by juvenile common banana prawns within estuaries

Despite tributary creek/mangrove forest habitats within coastal estuaries being the optimal habitat for recently-settled benthic common banana prawns (Vance et al. 1998, 2002), in tropical Australia the often-encountered hypersaline, tropical warm-water estuaries are not optimal conditions for survival and growth (Vance and Rothlisberg 2019).

Importantly, early season flows (~ October to December) can ameliorate estuarine conditions, reducing salinity to optimal 20-25, reducing temperature by a few degrees and motivating movement by juvenile common banana prawns from the uppermost tributaries, downstream where they may reside at lesser densities. Early season freshwater inputs create a brackish estuary that enhances the growth and survival of the prawns (Staples and Heales 1991) and supports the foodweb (Haliday et al. 2012, Duggan et al. 2014, Welch et al. 2014); despite prawn mortality due to predation by fish being high (Haywood and Staples 1993, Wang and Haywood 1999). Low-level wet-season flows continue to support benthic algae, meiofauna and resident common banana prawns within the estuary.

The ability to explore the relationship between seasonal and dimensional aspects of hydrological floodflow data and banana prawn catch is critical to investigate the impact of the offtake of river flows for irrigated agriculture. Under water resource management regimes, water can be impounded or extracted as required by irrigation demand. Water extraction conflicts with natural river flow regimes that historically have sustained ecological services downstream of water resource infrastructure. The intricate life history of common banana prawns requires seasonal flows which both condition estuaries prior to the wet season to sustain resident juvenile populations, and to provide

an emigration cue in the form of wet-season floods (Kenyon et al. 2018, Vance and Rothlisberg 2019). Wet season river flows do not elevate nutrients or productivity with GoC estuaries and hence do not stimulate estuarine prawn growth or biomass (Burford et al. 2012). However, wet-season rainfall and over-banks flows stimulate desiccated algal crusts on extensive saltflat floodplains adjacent to the estuaries and through-flows transport released nutrients to the estuary and nearshore (Burford et al. 2016). Flood flows cause both the emigration of banana prawn juveniles offshore, and dump terrigenous nutrients in the flood plume which sustain nearshore productivity for the recent emigrants (Vance et al. 1998, Burford et al. 2012, 2016).

References

Bayliss, P., Buckworth, R., Dichmont, C., 2014. Assessing the water needs of fisheries and ecological values in the Gulf of Carpentaria. Final report prepared for the Queensland Department of Natural Resources and Mines. CSIRO, Australia. 321p.

Broadley, Andrew; Stewart-Koster, Ben; Kenyon, Rob; Burford, Michele; Brown, Christopher. (2020). Reduced river flow impacts the catch of a commercial marine fishery. *Ecosphere* 11:e03194.

Buckworth, R. C., Venables, W.N., Lawrence, E., Kompas, T., Pascoe, S., Chu, L., Hill, F., Hutton, T., Rothlisberg, P.C., 2014. Incorporation of predictive models of Banana Prawn catch for MEY-based harvest strategy development for the Northern Prawn Fishery. Final Report to the Fisheries Research and Development Corporation, Project 2011/239. CSIRO Marine & Atmospheric Research, Brisbane, Australia. 115p.

Burford, M.A., Webster, I.T., Reville, A.T., Kenyon, R.A., Whittle, M. and Curwen, G., 2012. Controls on phytoplankton productivity in a wet-dry tropical estuary. *Estuarine Coastal and Shelf Science* 113:141-151.

Burford, M.A., Valdez, D., Curwen, G., Faggotter, S.J., Ward, D.P. O'Brien, K.R., 2016. Inundation of saline supratidal mudflats provides an important source of carbon and nutrients in an aquatic system. *Marine Ecology Progress Series* 545:21-33.

Dall, W., Hill, B.J., Rothlisberg, P.C., Staples, D.J., 1990. The biology of the Penaeidae. *Advances in Marine Biology* 27. 498pp. Blaxter, JHS & Southward, AJ (eds) Academic Press, London.

Duggan, M., Connolly, R.M., Whittle, M., Curwen, G., Burford, M.A., 2014. Effects of freshwater flows extremes on intertidal biota of a wet-dry tropical estuary. *Marine Ecology Progress Series* 502:11-23.

Duggan, M., Bayliss, P., Burford, M.A. 2019. Predicting the impacts of freshwater-flow alterations on prawn *Penaeus merguensis* catches. *Fisheries Research* 215:27-37.

Gwyther, D., 1982. Yield estimates for the Banana Prawn (*Penaeus merguensis*) in the Gulf of Papua prawn fishery. *Journal du Conseil/ Conseil Permanent International pour l'Exploration de la Mer* 40:245-258.

Halliday, I., Saunders, T., Sellin, M.J., Allsop, Q., Robins, J., McLennan, M., Kurnoth, P., 2012. Flow impacts on estuarine finfish fisheries of the Gulf of Carpentaria. FRDC Project No. 2007/002.

Hughes J, Yang A, Wang B, Marvanek S, Seo L, Petheram C and Vaze J (2018) River model simulation for the Fitzroy, Darwin and Mitchell catchments. A technical report from the CSIRO Northern Australia Water Resource Assessment to the Government of Australia. CSIRO, Australia.

Kenyon R., Plaganyi E., Miller M., Moeseneder C, Buckworth, R. CSIRO Oceans and Atmosphere. 2018. *Proposed northern Australian water developments pertinent to the Northern Prawn Fishery*. Brisbane, April. CC BY 3.0.

- Lerat J, Egan C, Kim S, Gooda M, Loy A, Shao Q and Petheram C (2013). Calibration of river models for the Flinders and Gilbert catchments. A technical report to the Australian Government from the CSIRO Flinders and Gilbert Agricultural Resource Assessment, part of the North Queensland Irrigated Agriculture Strategy. CSIRO Water for a Healthy Country and Sustainable Agriculture flagships, Australia.
- Lucas, C., Kirkwood, G., Somers, I., 1979. An assessment of the stocks of the Banana Prawn *Penaeus merguensis* in the Gulf of Carpentaria. *Marine and Freshwater Research* 30:639-652.
- Staples, D., Heales, D., 1991. Temperature and salinity optima for growth and survival of juvenile Banana Prawns *Penaeus merguensis*. *Journal of Experimental Marine Biology and Ecology* 154:251-274.
- Vance, D.J., Rothlisberg, P.C. - in review. The Biology and Ecology of the Banana Prawns *Penaeus merguensis* De Man and *P. indicus* H. Milne Edwards.
- Vance, D.J., Staples, D.J., Kerr, J.D., 1985. Factors affecting year-to-year variation in the catch of Banana Prawns (*Penaeus merguensis*) in the Gulf of Carpentaria, Australia. *Journal du Conseil/ Conseil Permanent International pour l'Exploration de la Mer* 42: 83-97.
- Vance, D. J., Haywood, M.D.E., Heales, D.S., Kenyon, R.A., Loneragan, N.R., 1998. Seasonal and annual variation in abundance of postlarval and juvenile Banana Prawns *Penaeus merguensis* and environmental variation in two estuaries in tropical northeastern Australia: a six year study. *Marine Ecology Progress Series* 163:21-36.
- Vance, D.J., Bishop, J., Dichmont, C.M., Hall, N., McInnes, K., Taylor, B.R., 2003. Management of common Banana Prawn stocks of the Gulf of Carpentaria: separating the effects of fishing from the environment. CSIRO Report 1998/0716.
- Vance, D., Haywood, M., Heales, D., Kenyon, R., Loneragan, N., Pendrey, R., 2002. Distribution of juvenile penaeid prawns in mangrove forests in a tropical Australian estuary, with particular reference to *Penaeus merguensis*. *Marine Ecology Progress Series* 228:165-177.
- Venables, W., Hutton, T., Lawrence, E., Rothlisberg, P., Buckworth, R., Hartcher, M., and Kenyon, R. (2011). Prediction of Common Banana Prawn Potential Catch in Australia's Northern Prawn Fishery. . *Unpublished Report to the Australian Fisheries Management Authority*.
- Wang, Y.-G., Haywood, M., 1999. Size-dependent natural mortality of juvenile Banana Prawns *Penaeus merguensis* in the Gulf of Carpentaria, Australia. *Marine and Freshwater Research* 50:313-317.
- Welch, D., Robins, J., Saunders, T., 2014. Implications of climate change impacts on fisheries resources of northern Australia. Part 2: species profiles. Final report for FRDC project 2010/565, James Cook University, Townsville.

Appendix 8 - Research on the underpinnings of banana prawn production in estuaries

Michele Burford, Australian Rivers Institute, Griffith University

There have been two projects focussed on understanding the underpinnings of banana prawn production in estuaries. The first was a Commonwealth Government Tropical Rivers and Coastal Knowledge (TRaCK) hub and FRDC funded project conducted in the Norman River, Gulf of Carpentaria in 2008-2010 (Burford et al. 2010). It was a collaboration between Griffith University (Burford et al.) and CSIRO (Kenyon, Revill, Webster). This project examined how river flows in the Norman River estuary affect prawn production and emigration. It was a two-year study, with a focus on the period October to March each year when postlarval and juvenile common banana prawns are in the estuary. Parameters measured in study were:

- Prawn densities and size metrics (main estuary and associated tidal creeks)
- Meiofauna species composition and densities (mudflats, food supply for prawns)
- Stable isotope values (C, N) for prawns and meiofauna
- Benthic chlorophyll *a* concentrations (areal values, mudflats)
- Sediment carbon concentrations (areal values, mudflats)
- Water column physico-chemical parameters (temperature, salinity, DO, pH, TSS, total and dissolved nutrients, secchi)
- Water column chlorophyll *a* concentrations
- Measures of primary production in water column, mudflat, saltflat (less frequent)

The key elements examined in the project are outlined below (Figure A32; Figure A33, see highlighted boxes to determine which components were specifically studied).

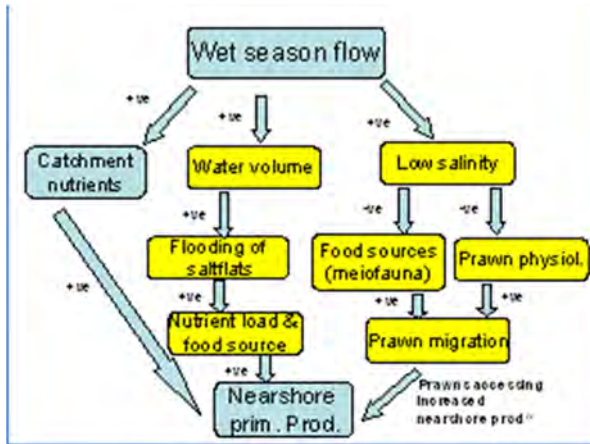


Figure A32. Flow diagram showing the linkages between river flow and prawn migration and production. Yellow highlighted boxes show those elements which were a focus of the study.

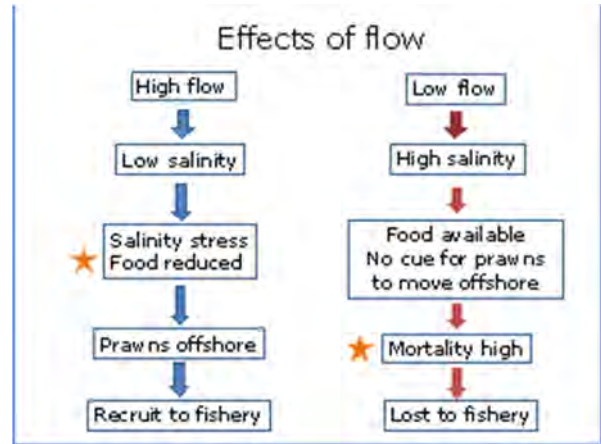


Figure A33. Contrasting effects of high and low flow on prawns in the estuary. Stars highlight new information.

Key findings of the study were:

- There are two cues for prawns emigrating from the estuary in the wet season – low salinity causing stress, and a lack of food (low salinity removing meiofauna and benthic algae). Three months after wet season flow, meiofauna and benthic algae had recruited back into estuary (Burford et al. 2010, Duggan et al. 2014).
- Mortality was high for prawns that did not emigrate out of the estuary (Burford et al. 2010). Presumably this is predation. Essentially this means they do not contribute to the next generation of prawns.
- Wetting of saltflats by wet season floods released significant loads of nutrients into estuary (Burford et al. 2016).
- Concentrations of nutrients did not increase with wet season flow. The end-of-system nutrient loads were calculated for the Norman River (Burford et al. 2012). In the dry season there is net transport of nutrients back into the estuary and the loads were also calculated.
- The effect of water development and the associated potential modifications to flow were predicted for banana prawn catch, based on a Bayesian Modelling approach (Duggan et al. 2019).

In 2015, a Northern Environmental Science (NESP) hub on Northern Australian Environment Resources was funded by the Commonwealth Government. This supported a new three year study lead by Burford. Parallel funding for Kenyon's (CSIRO) involvement was provided by FRDC. A range of industry, government and local community stakeholders identified the need to examine the effect of water development in the Flinders, Gilbert and Mitchell Rivers on estuarine productivity with flow on effects to prawns. This project therefore built on previous findings in the Norman River estuary.

For this study there were a number of field campaigns examining all three estuaries in the wet and dry seasons when prawns were in estuaries. The focus has been on:

- Prawn densities and size metrics
- Macrobenthos densities and species composition (food supply for prawns)
- Benthic chlorophyll *a* concentrations and primary productivity
- Sediment grain size, total nitrogen, phosphorus, carbon
- Water column chlorophyll *a* concentrations
 - Physico-chemical parameters (Water column physico-chemical parameters (temperature, salinity, DO, pH, turbidity, total and dissolved nutrients, secchi))
- Areal extent of mangroves, mudflats

Sample and data analysis are not yet completed for these parameters but it is anticipated that this will be completed by the end of the year. To date findings have supported those in the Norman River, i.e. high flow causes prawn emigration, and removes the benthos and benthic algae from the estuary.

This project is also quantifying the temporal changes in wetland hotspots in the three river system using a remote sensing approach (Ndehedehe, Griffith University). Economic tradeoffs between the prawn fishery and agricultural development are also being examined (Smart et al. Griffith University).

References

Burford, M., Kenyon, R., Whittle, M., Curwen, G. 2010. *River flow impacts on estuarine prawns in the Gulf of Carpentaria*. Fisheries Research and Development Corporation final report 2007/003.

Burford, M.A., Webster, I.T., Reville, A.T., Kenyon, R.A., Whittle, M., Curwen, G. 2012. Controls on phytoplankton productivity in a wet-dry tropical estuary. *Estuarine, Coastal and Shelf Science*, 113, 141-151.

Duggan, M., Connolly, R.M., Whittle, M., Curwen, G., Burford, M.A. 2014. Effects of freshwater flow extremes on intertidal biota of a wet-dry tropical estuary. *Marine Ecology Progress Series*, 502, 11-23.

Burford, M.A., Valdez, D., Curwen, G., Faggotter, S.J., Ward, D.P., O'Brien, K.R. 2016. Inundation of saline supratidal mudflats provides an important source of carbon and nutrients in an aquatic system. *Marine Ecology Progress Series*, 545, 29-33.

Duggan, M., Bayliss, P., Burford, M.A. 2019. Predicting the impacts of freshwater alterations on prawn (*Penaeus merguensis*) catches. *Fisheries Research* 215, 27-37.

Appendix 9 – Non-Spatial Prawn Model sub-model

Banana boils and an index of abundance

Commercial fishers targeting common banana prawns take advantage of the visibility of prawn aggregations or boils from the air. They contract small aircraft that include their pilots and trained spotters of banana prawn boils. Organisations use their own rules of thumb based on what has worked in the past. Spotter planes are deployed once the fishing vessels are in the fishing grounds for the early part of the season (2-4 weeks) with mostly a single plane. One organisation reporting using two planes in the first 4 weeks and then retained a single plane is used for the final two or three weeks. Seven weeks was the longest spotter plane deployment reported to our researchers. When two planes are operating two disparate areas can be scanned separately.

The ability of spotter planes to find boils often within relatively short vicinity to the fishing vessels was reported multiple times. Stories like: ‘In some cases, a spotter may find a boil 20 nm away from a vessel and the vessel hangs on the assessment of the spotter whether it’s worth a physical investigation.’ However, spotter planes are expensive, and one rule of thumb reported to us was that catches must be greater than 5 tonne per day to be viable on a cost/benefit basis. Spotters will often start from where the vessel skippers have steamed to, locations where experience suggest that prawns might be found based on fishery knowledge plus the conditions of the tide/moon/wind, etc. Spotter planes will do a morning flight in the vicinity of the vessel (or vessels) that are already actively looking for mud boils.

Vessel ‘behaviour’ then becomes highly dependent on the spotter. There was significant discussion about the skill of the spotter and how they relay good usable information to the skipper. It sounded complicated; and difficult to describe in words but suffice to say that we heard that the ‘clever skippers’ explicitly respond to and exploit the spotter’s instructions. When spotter aircraft are not working the crew are up in the mast rigging all day looking for mud boils.

CPUE as long included the technological advantage that sounders provide to commercial fishers. To date though it has not explicitly included the advantage won by using spotter planes. In the case of banana prawns, what may not look like a viable percentage of prawn in a boil on a sounder can be verified to be worth trawling by the spotters, and vice versa.

Fisheries that target spatial aggregations are thought to exhibit hyperstability, in which CPUE remains elevated as stock abundance declines (Harley et al. 2001, Erisman et al. 2011). This can be due to fishers maintaining high CPUE despite stock abundance dropping because they are able to efficiently target aggregations. Similarly, collapses of stocks such as the cod fishery have been attributed in part

“..rapid learning, and technological adjustments which were mistaken as increases in stock size in the calculation of a standardized catch per unit of effort index” (Walters and Maguire 1996).

The banana prawn fishery also operates by efficiently targeting aggregations or boils, and hence it seems prudent to assume that CPUE is not linearly proportional to stock biomass. As a simple illustration of this concept, Figure A34 shows a hypothetical area with a true average relative abundance of prawns of 1t per unit area. There are no dedicated surveys available for the banana prawn fishery, but a randomly stratified survey would nonetheless not perform too badly in estimating the underlying prawn relative abundance – in the illustrative example, the estimate is 1.5t/area. On the other hand, if the CPUE data were based on selective sampling of only the three largest boils, then as per the illustrative example, the biomass may be substantially overestimated – in this case yielding an estimate of 3.33t/area. One method to account for this is to assume a hyperstable relationship between exploitable biomass ($B_{ex}(y)$) and CPUE in year y (C/E), where a parameter $hyps$ controls the extent of hyperstability as follows:

$$\left(\frac{C}{E}\right)_y = q_{merg} q_y^{fp,merg} (B_y^{ex})^{hyps} \quad (8)$$

Where q_{merg} is the catchability constant and $q_y^{fp,merg}$ is the fishing power estimate in year y .

The preliminary MICE model uses parameter setting of $hyps = 0.8$ but this will be explored further in ongoing work. This is important because there are few survey-based estimates of relative abundance and hence the CPUE data (and Catch data – see below) are primarily used to inform on estimates of relative stock abundance. Figure A35 shows an example of two alternative hyperstability parameter settings compared with an assumption of a linear relationship between CPUE and biomass.

The preliminary MICE model is fitted to all available CPUE data since 1970, but ongoing work will investigate whether it is preferable to only use CPUE data for the first few weeks, together with other considerations.

Accounting for hyperstability when using CPUE data for fishery with aggregations

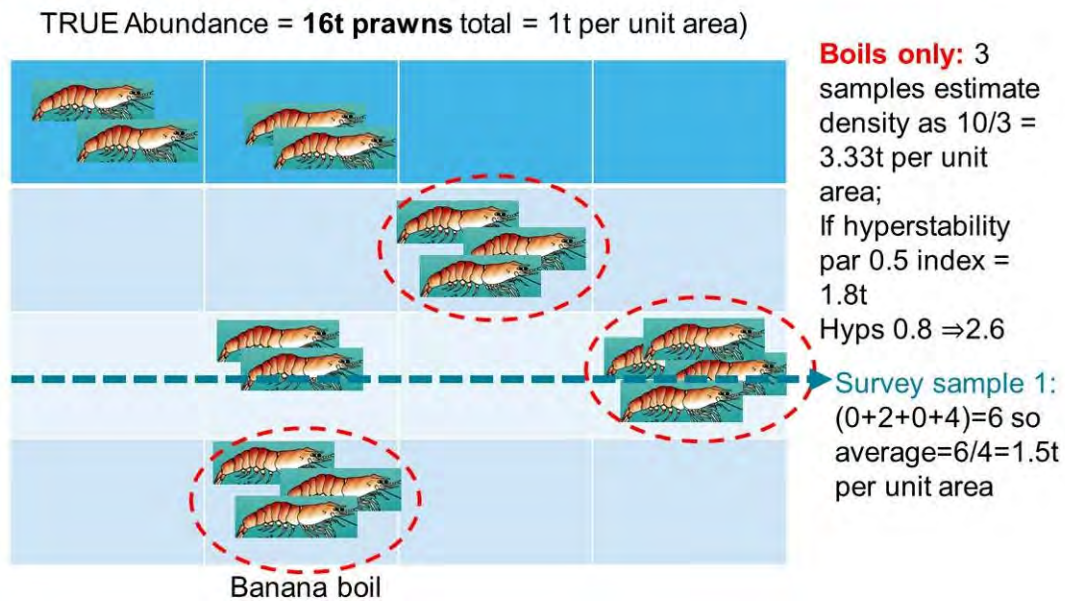


Figure A34. Schematic to illustrate how sampling spatially-aggregated populations can influence overall abundance estimates.

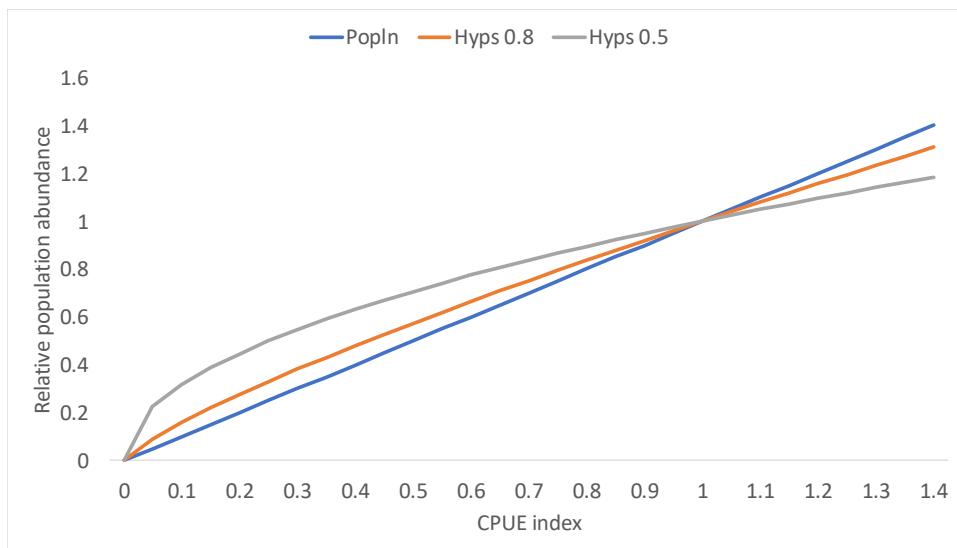


Figure A35. Simplistic plot comparing a linear relationship between a CPUE index and the assumed relative population abundance, with a hyperstability relationship, using as illustrations a mild (hyps = 0.8) and more extreme (hyps = 0.5) examples.

Preliminary parameter settings, estimation and model outputs for prawns

The preliminary non-spatial version of the MICE model was fitted to the available CPUE and survey data and estimated a total of 107 parameters including the 1970 spawning biomass of each of the tiger prawn species, steepness estimates for the tiger prawn species, 7 selectivity parameters and 48 recruitment residuals for each of the tiger prawn species for years 1971 to 2018. The parameter settings, estimates, and associated confidence intervals are shown in Table A12 and Table A13, except for the estimated recruitment residuals which are plotted instead (Figure A37). The model-estimated spawning biomass trajectories for the three species are shown in Figure A38.

Estimated annual CPUE for each quarter reflects observed CPUE reasonably well (Figure A39). MICE survey recruitment indices also closely correspond to fishery independent survey (FIS) recruitment indices for the two tiger prawn species (Figure A40). Annual MICE spawning biomass indices lie closely with FIS spawning biomass indices for the two tiger prawn species (Figure A40). Modelled annual spawning biomass is consistent with 2018 stock assessment spawning biomass (SSB) for the two tiger species respectively (Figure A41).

The banana prawn population model was also fitted to the available CPUE (Figure A39) and survey indices although the survey indices are not representative of the entire stock and this aspect will be explored further in ongoing work. MICE survey recruitment indices closely roughly to fishery independent survey (FIS) recruitment indices for common banana prawns (Figure A40). Annual MICE spawning biomass indices do not closely fit the FIS spawning biomass indices (Figure A40). Comparison between the observed and model-predicted total (Figure A42) and quarterly (Figure A43) catch totals for common banana prawns suggests that the model performs well and the recruitment-rainfall index can be used to project future changes in banana prawn abundance in response to changes in this key environmental driver. Future work, such as in refining some model parameter estimates and inputs, could focus on improving these results further.

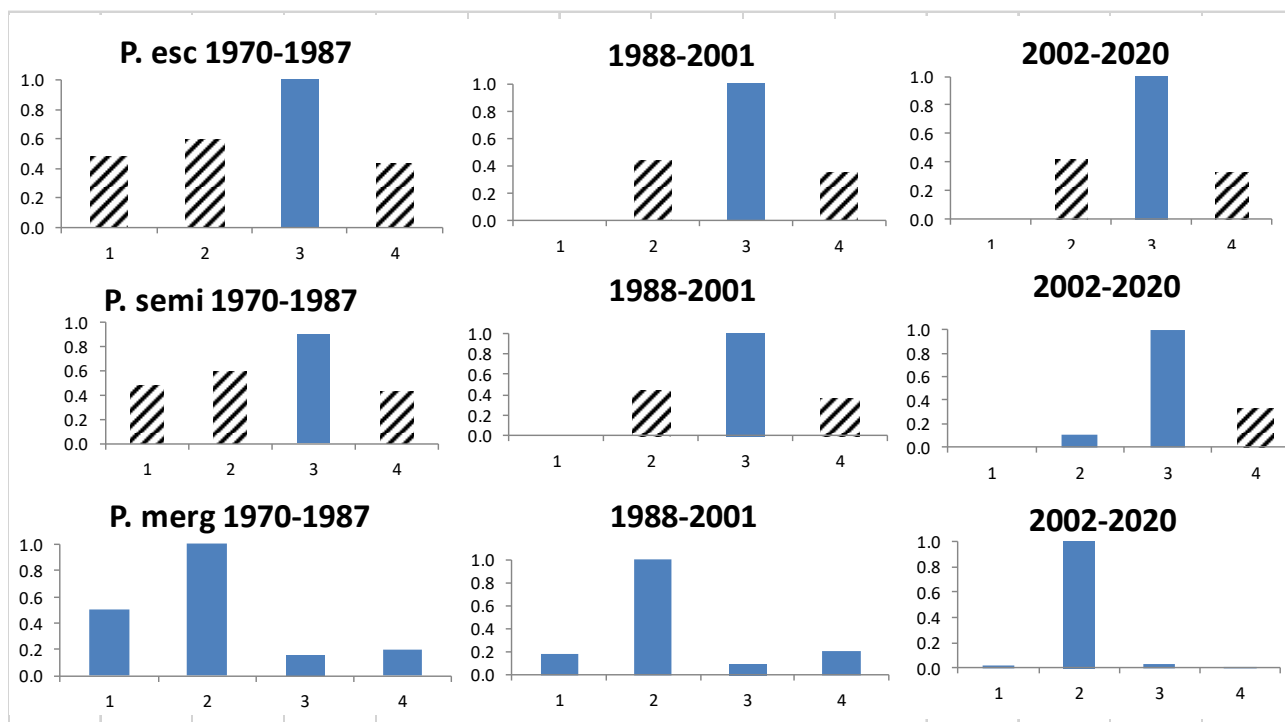
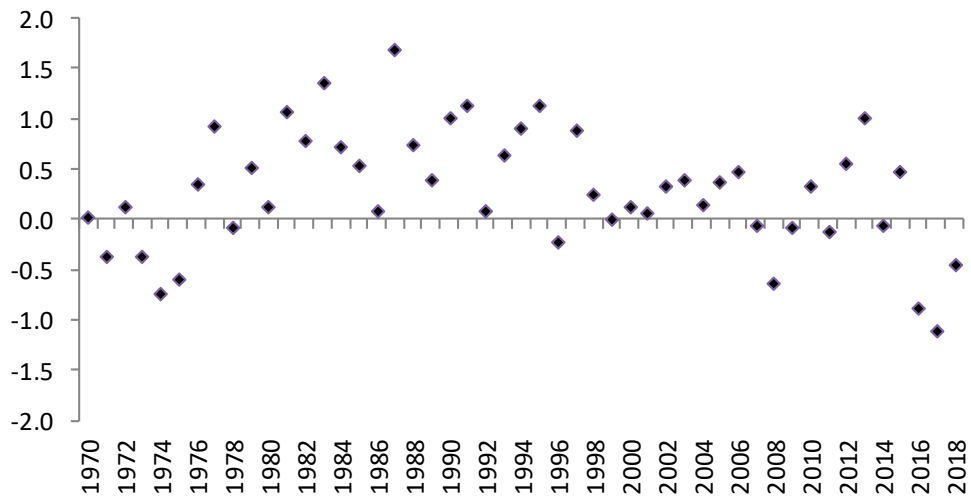


Figure A36. Quarterly selectivity patterns for each of the three selectivity periods (1970-1987; 1988-2001; 2002-current) shown for the three modelled prawn species: brown tiger prawn *P. esculentus* (*P.esc*), grooved tiger prawn *P. semisulcatus* (*P.semi*) and common banana prawns *Penaeus merguensis* (*P.merg*). The parameters estimated by the model are shown with diagonal shading, and the same parameters are assumed for the two tiger species except for the second quarter of years 2002-2020 (last column) when small catches of *P.esc* but not *P.semi* are recorded.

P. esc model-estimated residuals



P. semi model-estimated residuals

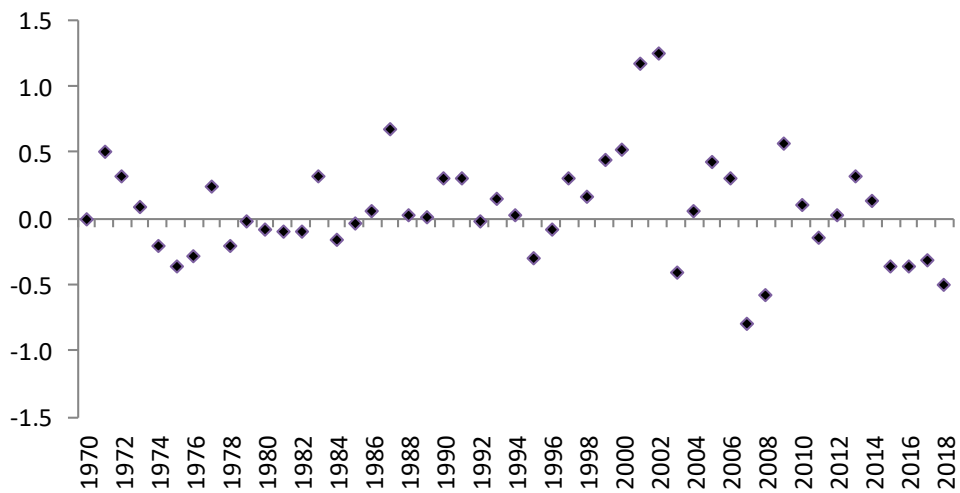


Figure A37. Model-estimated recruitment residuals for tiger prawn species as shown.

Table A12. Summary of the parameters of the MICE non-spatial model for *P. esculentus* (P.e.), *P. semisulcatus* (P.s), and *P. merguensis* (P.m.)

Parameter	Treatment
Pre-exploitation equilibrium spawning biomass, K_{1970}^{sp} (P.e.; P.s.; P.m.)	Estimate $K_{1970,1}^{sp}$ for first quarter, compute values for other quarters using equilibrium assumptions, and set $K_{1970}^{sp} = \sum_{seas} K_{1970,seas}^{sp}$
Natural mortality, M	Fixed at 0.05 wk ⁻¹
<i>Recruitment and spawning</i>	
“Steepness”, h , of the stock-recruitment relationship	Estimated for P.e and P.s but fixed for P.m
Recruitment residuals, R_s	Estimated – 48 pars for 1971-2018 for P.e. and P.s.
Proportion of recruited stock that spawn each quarter, f_s	Assumed known [0.05; 0.4;0.5;0.05] P.e.;P.s [0.05; 0.5;0.4;0.05] P.m.
Stock-recruitment relationship parameters, α, β	Computed using estimated values of K_{1970}^{sp} and h
Variance in recruitment, σ_r	Fixed at 0.9 (sensitivities tested)
<i>Fishing mortality related</i>	
Catchability – q ($\times 10^{-4}$)	Computed
Availability during each quarter $A_{y,s}$	Estimated
<i>Growth parameters</i>	
Von Bertalanffy growth curve parameters	Assumed known & same as stock assessment model
Length-weight regression	Assumed known
<i>The observation model</i>	
Observation error variance, σ	Estimated

Table A13. Summary of model-estimated parameters and their corresponding Hessian-based standard deviations for *P. esculentus* (P.e.), *P. semisulcatus* (P.s), and *P. merguensis* (P.m).

Parameter	Value	90% Confidence Interval	
<i>B</i> (1970) ^{sp} (<i>P. esc</i>) (<i>Q1</i>)	863	260	1466
<i>B</i> (1970) ^{sp} (<i>P. semi</i>) (<i>Q1</i>)	693	371	1014
<i>h</i> (<i>P.esc</i>)	0.46	0.44	0.48
<i>h</i> (<i>P.semi</i>)	0.49	0.47	0.52
<i>Sel</i> (1st period) <i>Q1 P.e;P.s</i>	0.48	0.41	0.54
<i>Sel</i> (1st period) <i>Q2 P.e;P.s</i>	0.59	0.51	0.67
<i>Sel</i> (1st period) <i>Q4 P.e;P.s</i>	0.43	0.37	0.48
<i>Sel</i> (2nd period) <i>Q2 P.e;P.s</i>	0.44	0.37	0.51
<i>Sel</i> (2nd period) <i>Q4 P.e;P.s</i>	0.36	0.31	0.40
<i>Sel</i> (3rd period) <i>Q2 P.e;P.s</i>	0.42	0.36	0.48
<i>Sel</i> (3rd period) <i>Q4 P.e;P.s</i>	0.33	0.29	0.37
<u>Current Bsp(2018) total</u>			
<i>P. esc</i>	3229.9	1058.7	5401.1
<i>P. semi</i>	4810.2	3277.7	6342.7
<i>P. merg</i>	3094.2	3094.2	3094.2
<u>Likelihood contributions</u>		<u>sigma</u>	
-lnL:CPUE (P.e)	-143.55	0.24	
-lnL:Survey (P.e)	-24.61		
-lnL:CPUE (P.s)	-23.95	0.51	
-lnL:Survey (P.s)	-31.95		
-lnL:CPUE (P.m)	78.39	1.27	
-lnL:Survey (P.m)	1.79		
-lnL:RecRes (P.e)	12.85		
-lnL:RecRes (P.m)	4.73		
No. parameters estimated	107		
'-lnL:overall	-126.298		
AIC	-38.596		

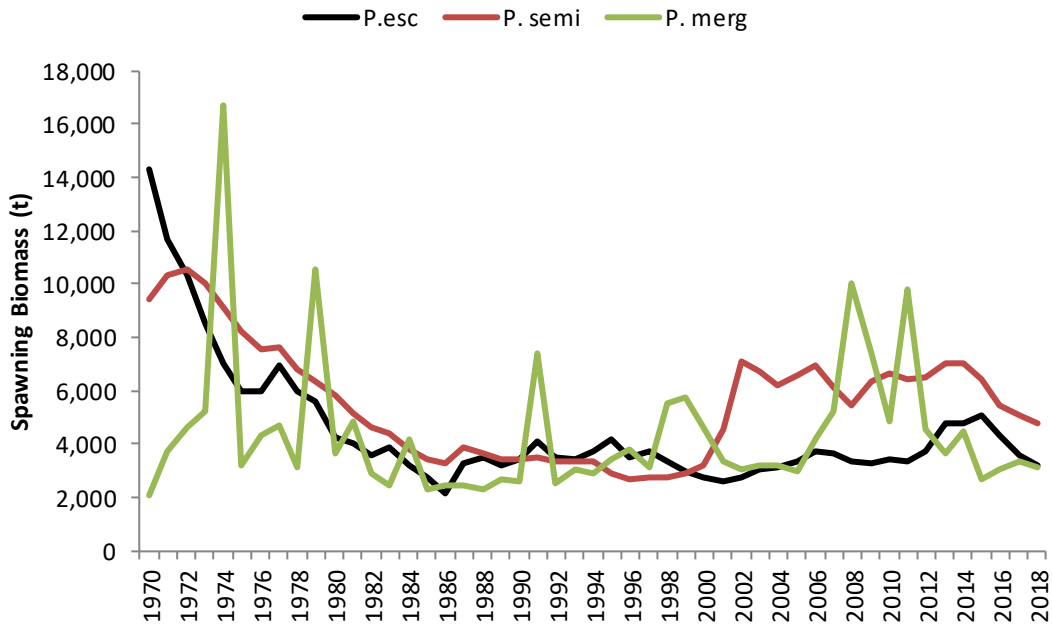


Figure A38. Model-estimated spawning biomass (t) trajectories for the three prawn species as indicated.

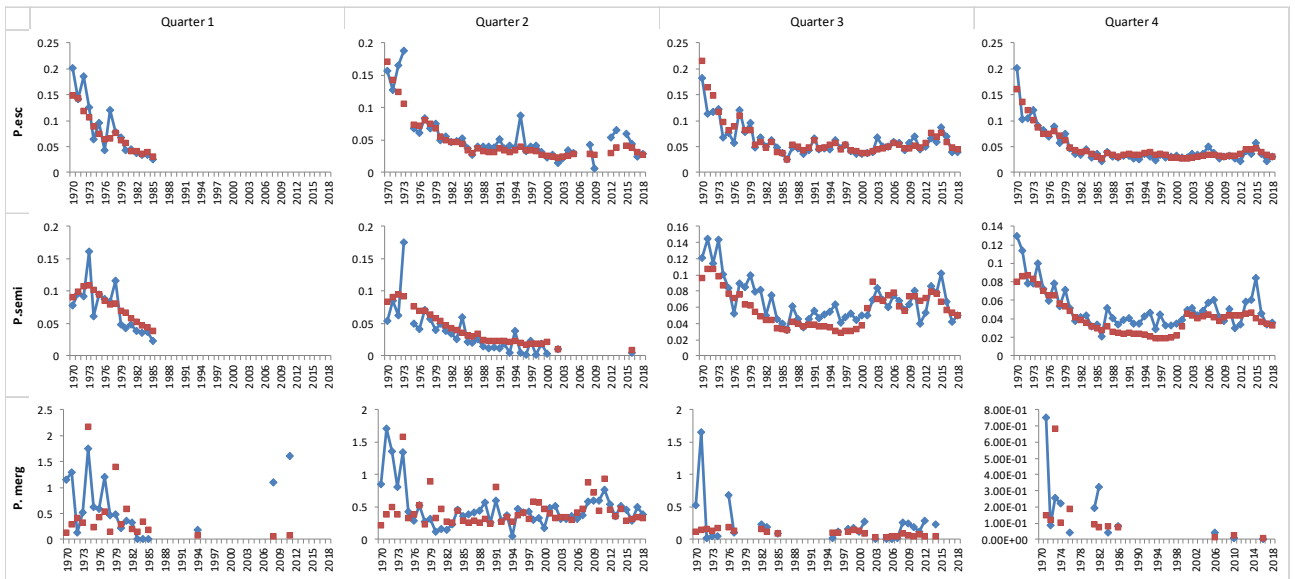


Figure A39. Comparison between observed total CPUE (t/day; blue) of brown tiger prawns (*P. esc*), grooved (*P. semi*) and common banana prawns (*P. merg*) per quarter of the year from 1970 to 2018, compared with model-estimated CPUE (red).

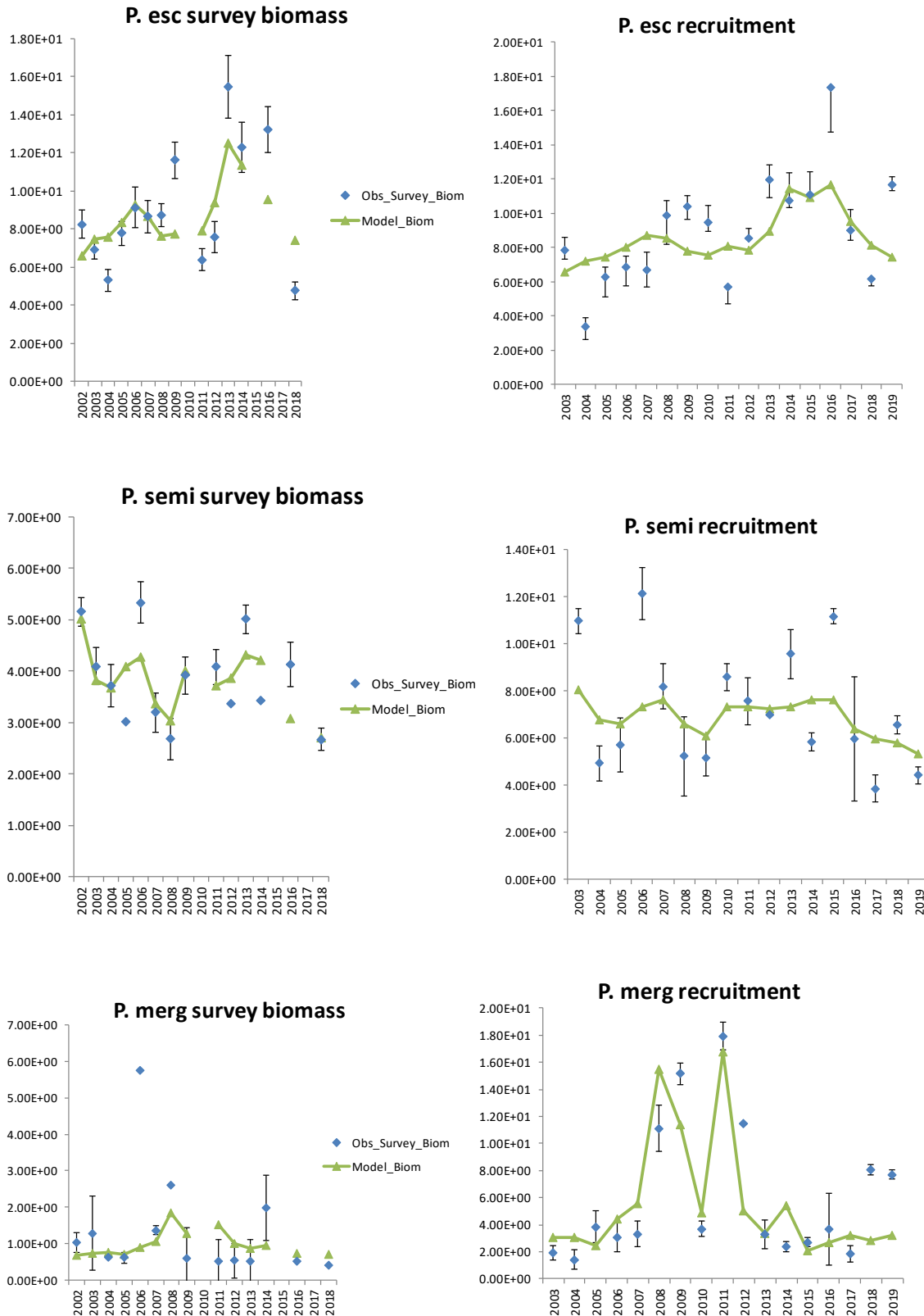


Figure A40. Annual observed fishery independent survey biomass and recruitment indices with associated standard deviations (blue) and modelled survey biomass and recruitment indices (green) for brown tiger prawns (*Penaeus esculentus*), grooved tiger prawns (*P. semisulcatus*) and common banana prawns (*P. merguensis*) from 2002 to 2019.

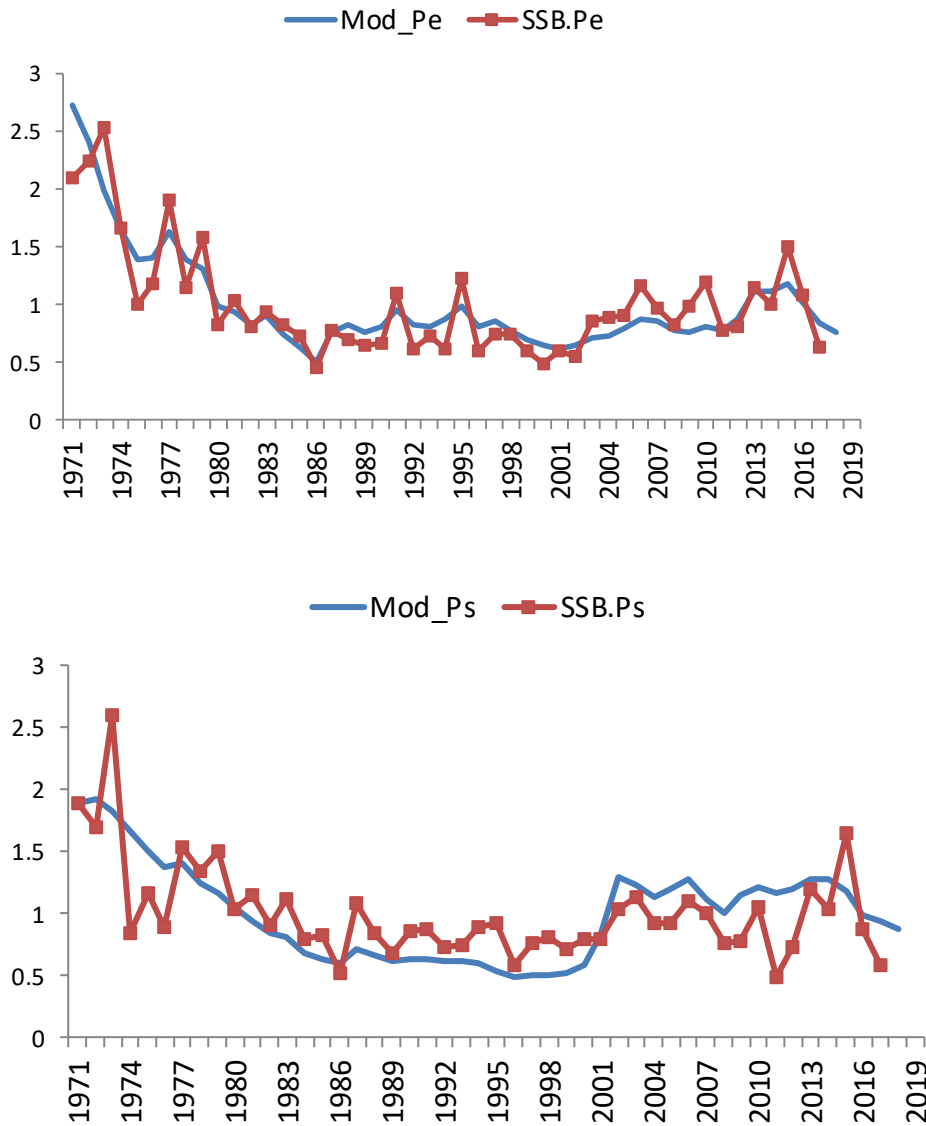


Figure A41. Annual modelled spawning stock biomass indices (relative to mean; blue) and the 2018 stock assessment spawning stock biomass indices (relative to mean; red) for brown tiger prawns (*Penaeus esculentus*) and grooved tiger prawns (*P. semisulcatus*) from 1971 to 2017.

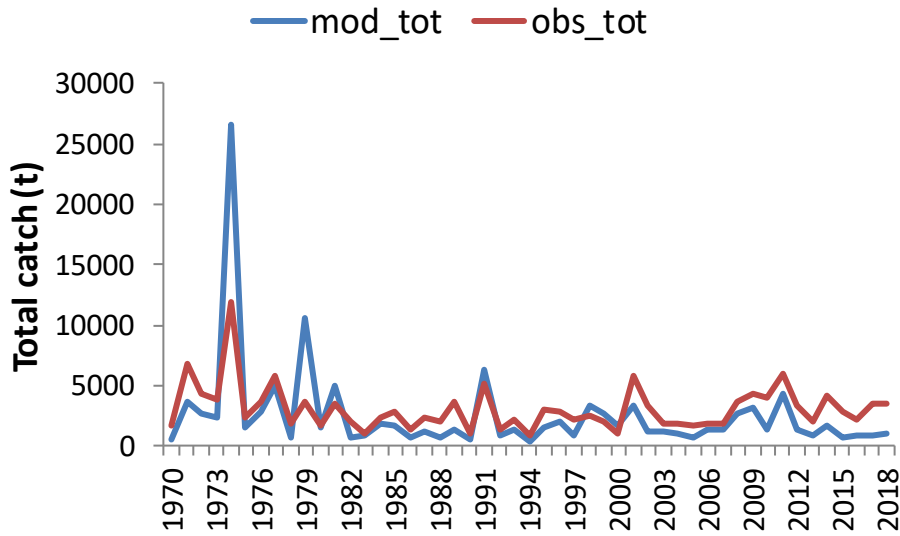


Figure A42. Comparison of total annual observed catches (t) of common banana prawns (*Penaeus merguensis*) from 1970 to 2018, compared with model-estimated catches.

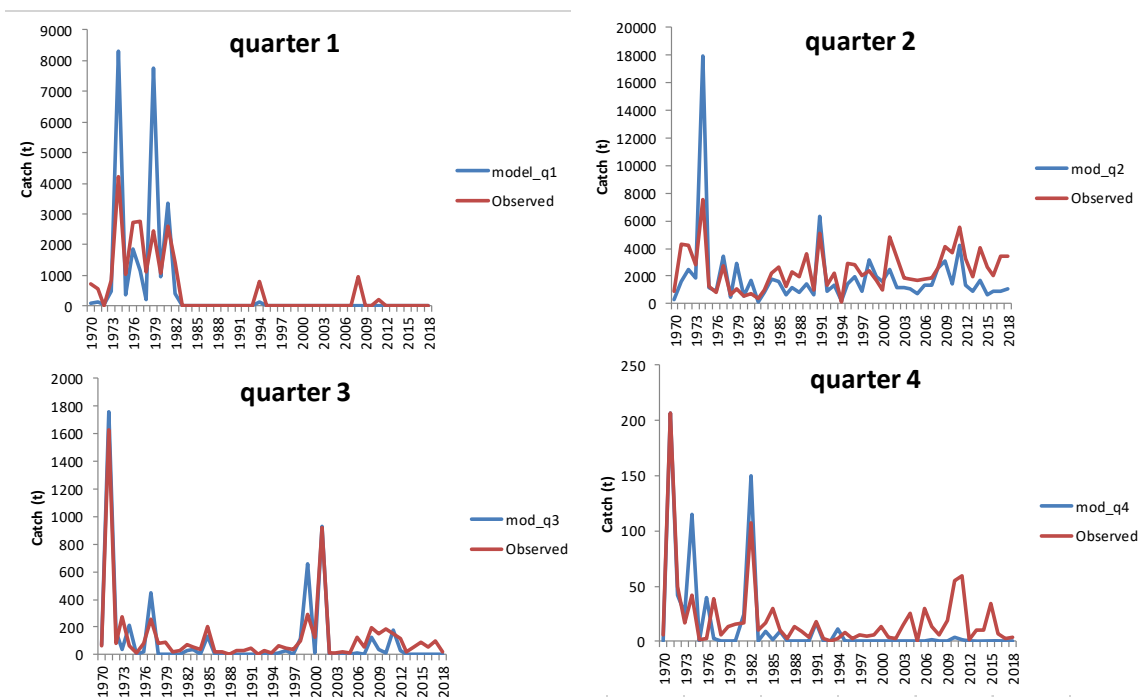


Figure A43. Comparison between observed total catches (t; blue) of common banana prawns (*Penaeus merguensis*) per quarter from 1970 to 2018, compared with model-estimated catches (red).

Appendix 10 – Prawn Spatial Model sub-model

The basic equations use dot model the prawn species are shown at the end of this Appendix. As explained in the main text, there are substantial differences in the relative abundance and catches of the three prawn species by region, and hence the MICE outputs very different trajectories for each model region, as illustrated for example by Figure A44 below.

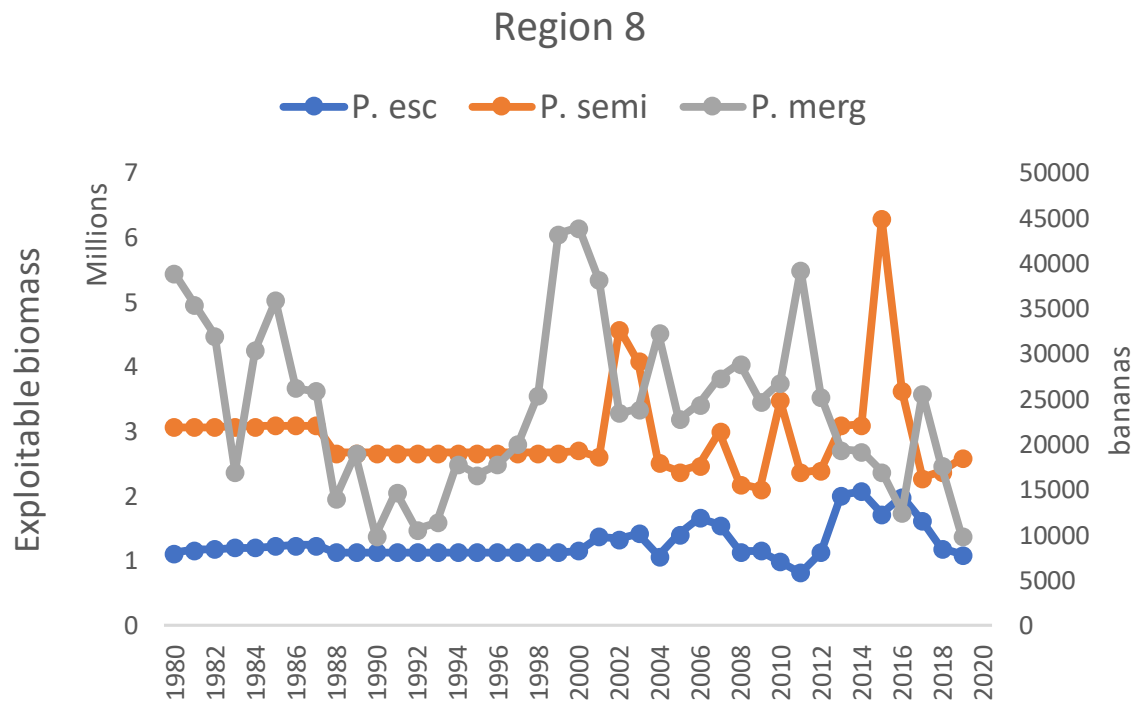


Figure A44. Example of the commercially exploitable biomass of each of the three prawn species over the period 1980 to 2019.

In some model regions, the modelled tiger prawn abundance fitted fairly well to the available survey (recruitment and spawning survey series) data (Figure A45), but for other regions more work is needed to refine the model fits. The base-case common banana prawn model component did not include fitting to the survey data as it wasn't possible to reconcile these data with the other information, and it is well known that the surveys are focussed on tiger prawns and therefore do not provide an accurate reflection of banana prawn abundance.

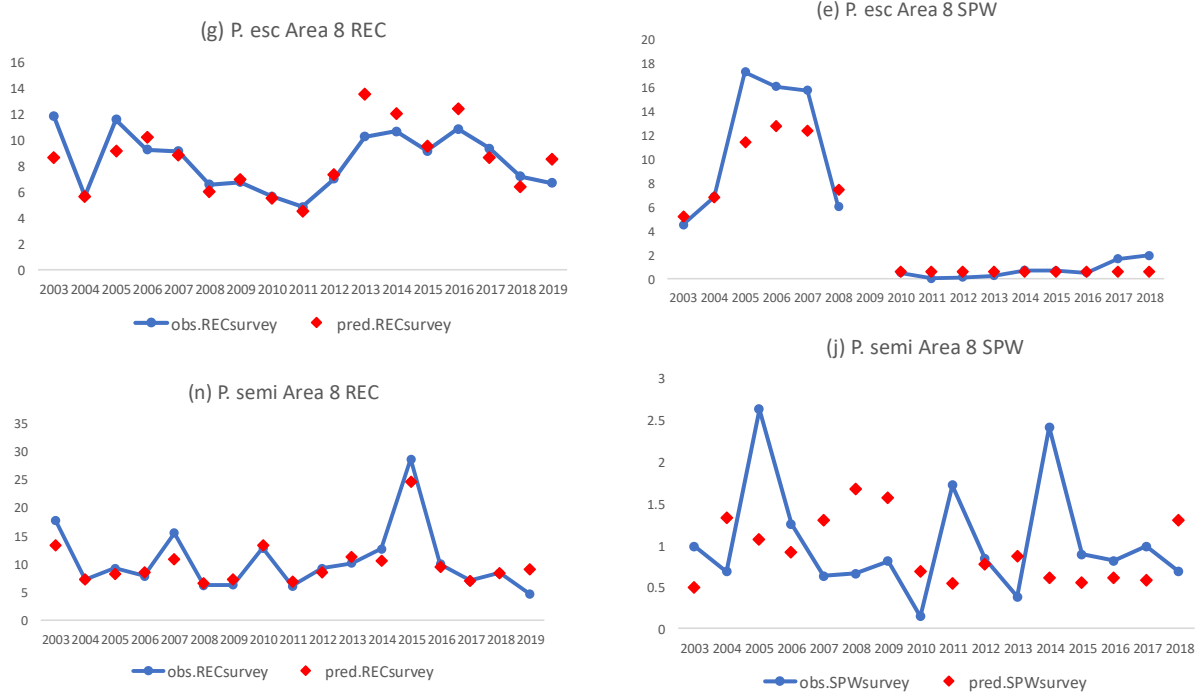


Figure A45. Example of model-predicted tiger prawn abundance fitted to both the recruitment (REC) and spawning (SPW) survey data for MICE region 8.

The common banana prawn model was built in a stepwise fashion by first estimating catches based on fishing effort alone, followed by use of a simple linear flow-recruitment relationship (Figure A46) and finally refining the model fit by using a logistic flow-recruitment relationship together with incorporating an offshore productivity effect (Figure A47).

Linear Rec-Flow relationship and no offshore productivity effect

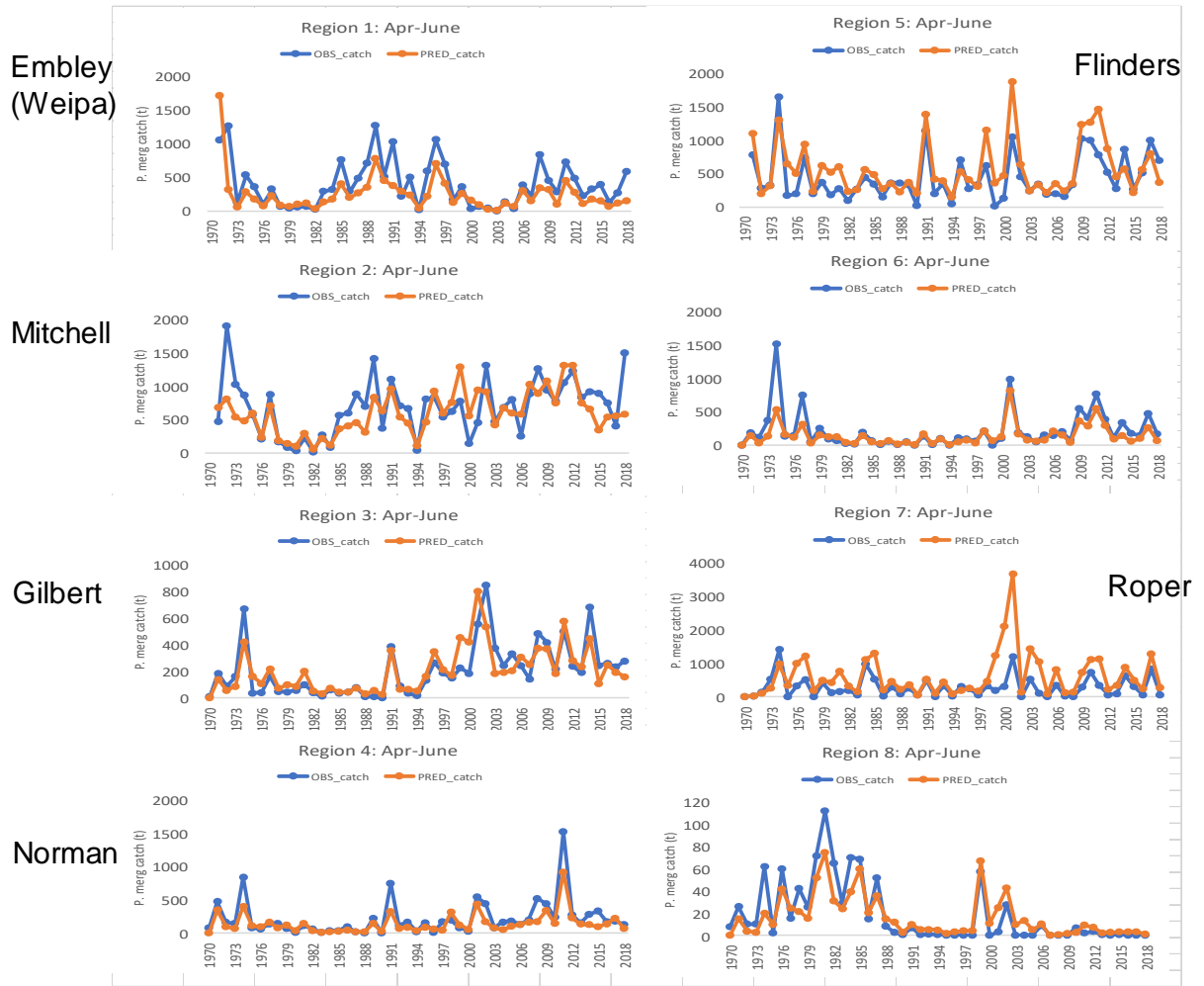


Figure A46. Comparison between actual observed catches and model-predicted common banana prawn catches over 1970 to 2019 for each of the MICE regions as shown, and when using a simple linear flow-recruitment formulation.

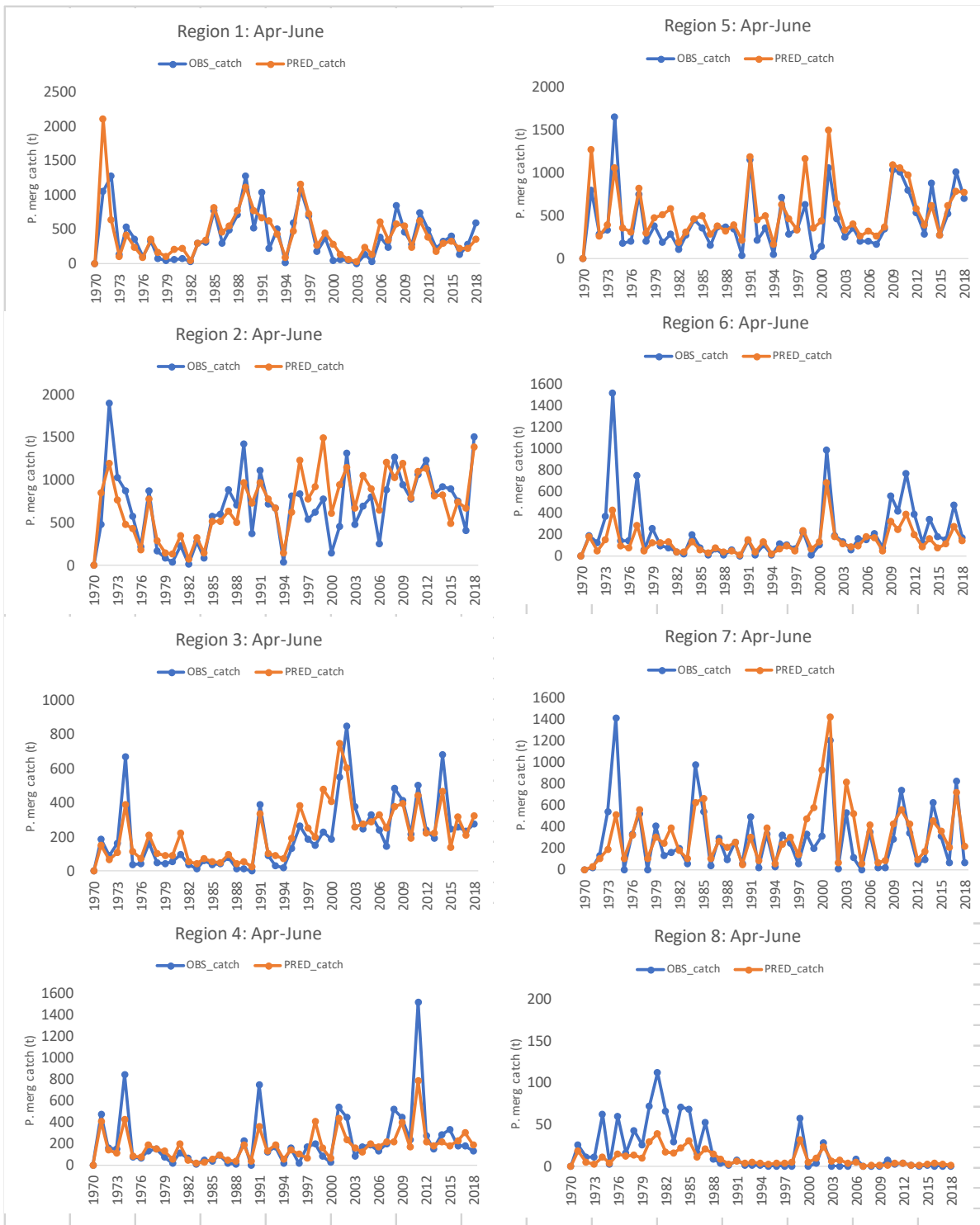


Figure A47. Comparison between actual observed catches and model-predicted common banana prawn catches over 1970 to 2019 for each of the MICE regions as shown, and when using a logistic flow-recruitment formulation and incorporating offshore productivity.

Prawn Equations

Numbers-at-age

The following discrete equations, based on the redleg banana prawn (*Penaeus indicus*) assessment model of (Plagányi et al. 2012), were applied to each of the three prawn species: 1. grooved tigers (*Penaeus semisulcatus*), 2. brown tigers (*P. esculentus*), and 3. common banana prawns (*Penaeus merguensis*) over the period 1970-2019. The model time-step s is weekly, with the number of prawns of species p in region r in year y and week s $N_{r,y,s}^p$ given by:

$$N_{r,y,s+1}^p = N_{r,y,s}^p e^{-M_{r,s}^p} - C_{r,y,s}^p + R_{r,y,s+1}^p \quad \text{for } s = 1 \text{ to } 51 \quad (1)$$

and

$$N_{r,y+1,1}^p = N_{r,y,52}^p e^{-M_{r,52}^p} - C_{r,y,52}^p + R_{r,y+1,1}^p \quad \text{for } s = 52 \quad (2)$$

where

$N_{r,y,s}^p$ is the number of recruited and mature prawns (those corresponding to a size large enough to be fished) of species p in region r and week s of year y (which refers to a calendar year),

$R_{r,y,s}^p$ is the number of recruits (number of 6-month old prawns) which are added to the population of species p in region r at the end of each week s in year y ,

$M_{r,y}^p$ denotes the natural mortality rate of species p in region r during week s , and computed by multiplying the weekly background natural mortality estimate by factors representing additional sources of density-dependent and density-independent mortality as described below; and

$C_{r,y,s}^p$ is the predicted number of prawns of species p caught in region r during week s in year y .

Given catches are recorded in units of mass, the predicted number of prawns of species p caught in region r during week s in year y is computed from the following relationship:

$$C_{r,y,s}^p = A_{r,y,s}^p F_{r,y,s}^p N_{r,y,s}^p e^{-M_{r,y}^p} \quad (3)$$

where

$A_{r,y,s}^p$ is the relative availability of species p in region r for week s and for year y , with four different availability vectors applied over the period 1970-2019;

$F_{r,y,s}^p$ is the fished proportion of species p in region r in week s and year y of a fully selected age class.

The fished proportion reflects the catch by mass ($C_{r,y,s}^{mass,p}$) in region r in week s and year y as a proportion of the exploitable (“available”) component of biomass:

$$F_{r,y,s}^p = \frac{C_{r,y,s}^{mass,p}}{B_{r,y,s}^{ex,p}} \quad (4)$$

with

$$B_{r,y,s}^{ex,p} = w_s^p N_{r,y,s}^p e^{-M_{r,y}^p} A_{r,y,s}^p \quad (5)$$

where

w_s^p is the average mass of prawns of species p during week s .

The Reference Case model used the female (because the male growth is too slow on its own) von Bertalanffy growth parameters and assumed that individual mass increases through the year. An average length and mass of prawns was thus calculated for each week, assuming a median birth date of October.

In terms of recruitment, we assumed that there was mixing of the prawns across model regions and hence we first calculated a combined spawning biomass and then proportion in each region.

The number of recruits of species p for the entire region R at the end of week s in year y is assumed to be related to the spawning stock size six months previously by a modified Beverton-Holt stock-recruitment relationship (Beverton and Holt, 1957), such that density-dependent changes in spawning apply equally to the entire stock. The recruitment to each region r is then assumed to be a proportion

$\rho_{r,y,s}^p$ that can be modelled as year and weekly dependent based on changes in underlying physical (density-independent) processes. In addition, it is assumed that density-independent processes that impact successful recruitment (e.g. changes in advection envelopes) operate differently in each region and hence fluctuations about the deterministic recruitment relationship are computed per region and species for selected periods of the year whereas in some weeks the residuals are set to zero:

$$\begin{aligned} R_{r,y,s+1}^p &= \rho_{r,y,s}^p \cdot \frac{\alpha^p B_{R,y,s-25}^{sp,p}}{\beta^p + (B_{R,y,s-25}^{sp,p})} e^{(\zeta_{r,y,s}^p - (\sigma_R^p)^2/2)} & 25 < s < 52 \\ R_{r,y,s+1}^p &= \rho_{r,y,s}^p \cdot \frac{\alpha^p B_{R,y-1,s+26}^{sp,p}}{\beta^p + (B_{R,y-1,s+26}^{sp,p})} e^{(\zeta_{r,y,s}^p - (\sigma_R^p)^2/2)} & s \leq 25 \\ R_{r,y,s}^p &= \rho_{r,y,s-1}^p \cdot \frac{\alpha^p B_{R,y,s-26}^{sp,p}}{\beta^p + (B_{R,y,s-26}^{sp,p})} e^{(\zeta_{r,y,s}^p - (\sigma_R^p)^2/2)} & s = 52 \end{aligned} \quad (6)$$

where

α^p and β^p are spawning biomass-recruitment relationship parameters for species p ,

$\zeta_{r,y,s}^p$ reflects fluctuation about the expected recruitment for species p in region r in year y and week s , which is assumed to be normally distributed with standard deviation σ_R^p (which is input in the applications considered here and considered constant for each species); for the two tiger prawn species, these residuals are treated as estimable parameters in the model fitting process, and a single set of residuals is estimated for the second half of the year because almost all recruitment is assumed to occur during this half of the year and is assumed driven by the same environmental influences each

year; for the banana prawns, recruitment fluctuations are assumed to be driven by flow as described below.

$B_{R,y,s}^{sp,p}$ is the total spawning biomass summed over all regions r for species p at the start of week s in year y ,

$B_{r,y,s}^{sp,p}$ is the spawning biomass of species p in region r at the start of week s in year y , computed as:

$$B_{r,y,s}^{sp,p} = f_s^p \cdot w_s^p \cdot N_{r,y,s}^p \quad (7)$$

where

f_s^p is a relative index of the amount of spawning of species p during week s .

For common banana prawns in particular, it is recognised that there is a weak or non-existent stock-recruitment relationship (Rothlisberg and Okey 2007) and hence, although an underlying stock-recruitment curve will still be assumed for this species, the σ_R^p will have a large value, and the variability in recruitment is a function of flow. In contrast to the two tiger prawn species for which recruitment residuals are estimated by fitting to available data, a flow-recruitment function is thus applied for banana prawns.

In order to work with estimable parameters that are more meaningful biologically, the stock-recruitment relationship is re-parameterised in terms of the pre-exploitation equilibrium spawning biomass, $B_0^{sp,p}$, and the “steepness”, h^p , of the stock-recruitment relationship, which is the proportion of the virgin recruitment that is realized at a spawning biomass level of 20% of the virgin spawning biomass (Equation (6) can be rewritten in terms of the “steepness” h^p , defined as the fraction of pristine recruitment R_0^p that results when spawning biomass drops to 20% of its pristine level, i.e.:

$$h^p R_0^p = R^p (0.2 B_0^{sp,p}) \quad (8)$$

which yields the following for the deterministic component of the formulation:

$$R^p (B_{r,y,s}^{sp,p}) = \frac{4h^p \cdot R_0^p \cdot B_{r,y,s}^{sp,p}}{B_0^{sp,p} (1-h^p) + B_{r,y,s}^{sp,p} (5h^p - 1)} \quad (9)$$

It follows that the total spawner stock size and recruitment for of species p in calendar year y are given respectively by:

$$B_{p,y}^{sp} = \sum_s B_{p,y,s}^{sp} \quad (10)$$

$$R_{p,y} = \sum_s R_{p,y,s} \quad (11)$$

The resource is assumed to be at the deterministic equilibrium (corresponding to an absence of harvesting) at the start of 1970, the initial year considered here.

Likelihood function

The model is fitted to weekly CPUE and survey data for the two tiger prawn species. The likelihood contribution is calculated assuming that the observed abundance index is log-normally distributed about its expected value:

$$I_y^s = \hat{I}_y^s e^{\varepsilon_y^s} \quad \text{or} \quad \varepsilon_y^s = \ln(I_y^s) - \ln(\hat{I}_y^s) \quad (14)$$

where I_y^s is the abundance index (with fishing power effect added) for year y and quarter s ,

$\hat{I}_y^s = q^s B_{y,s}^{ex}$ is the corresponding model estimated value, where $B_{y,s}^{ex}$ is the model value for exploitable resource biomass corresponding to quarter s , given by equation (5).

q is the constant of proportionality which is assumed to be the same for each of the quarters, and

$$\varepsilon_y^s \quad \text{from} \quad N\left(0, (\sigma_y^s)^2\right).$$

The contribution to the negative of the log-likelihood function (after removal of constants) is given then by:

$$-\ln L = \sum_y \left[\sum_s \ln \sigma_y^s + (\varepsilon_y^s)^2 / 2(\sigma_y^s)^2 \right] \quad (15)$$

with the standard deviation of the residuals for the logarithms of the abundance series assumed to be independent of y , and set in the fitting procedure by its maximum likelihood value:

$$\hat{\sigma}^s = \sqrt{\frac{1}{n} \sum_y \sum_s (\ln I_y^s - \ln \hat{I}_y^s)^2} \quad (16)$$

where n is the number of data points across all years and quarters.

The catchability coefficient q is also estimated using maximum likelihood:

$$\ln \hat{q} = \frac{1}{n} \sum_y \sum_s (\ln I_{y,s}^s - \ln \hat{B}_{y,s}^{ex}) \quad (17)$$

Stock-recruitment function residuals

The stock-recruitment residuals for the 3 tiger prawn species are assumed to be log-normally distributed. Thus, the contribution of the recruitment residuals to the negative of the (now penalised) log-likelihood function is given by:

$$-\ln L^{pen} = \sum_{y=y1+1}^{y2} \frac{(R_{y,s})^2}{2\sigma_R^2} \quad (18)$$

where

σ_R is the standard deviation of the log-residuals, which is input.

References

- Beverton, R. J., and Holt, S. J. 1957. On the dynamics of exploited fish populations. Fisheries Investigation Series 2, volume 19, UK Ministry of Agriculture, Fisheries, and Food, London, UK.
- Deng, R., Hutton, T., Buckworth, R., Zhou, S., and Punt, A. 2020. Modification of the NPF Endeavour prawn stock assessment models to accommodate management changes. Unpublished Document. O&A Brisbane, St Lucia. 29 pages.
- Dichmont, C., Punt, A., Deng, A., Dell, Q., and Venables, W. 2003. Application of a weekly delay-difference model to commercial catch and effort data for tiger prawns in Australia's Northern Prawn Fishery. *Fisheries Research*, 65: 335-350.
- Erisman, B. E., Allen, L. G., Claisse, J. T., Pondella, D. J., Miller, E. F., and Murray, J. H. 2011. The illusion of plenty: hyperstability masks collapses in two recreational fisheries that target fish spawning aggregations. *Canadian Journal of Fisheries and Aquatic Sciences*, 68: 1705-1716.
- Fournier, D. A., Skaug, H. J., Ancheta, J., Iannelli, J., Magnusson, A., Maunder, M. N., Nielsen, A., et al. 2012. AD Model Builder: using automatic differentiation for statistical inference of highly parameterized complex nonlinear models. *Optimization Methods and Software*, 27: 233-249.
- Harley, S. J., Myers, R. A., and Dunn, A. 2001. Is catch-per-unit-effort proportional to abundance? *Canadian Journal of Fisheries and Aquatic Sciences*, 58: 1760-1772.
- Haywood, M. D. E., Heales, D. S., Kenyon, R. A., Loneragan, N. R., and Vance, D. J. 1998. Predation of juvenile tiger prawns in a tropical Australian estuary. *Marine Ecology Progress Series*, 162: 201-214.
- Hilborn, R. 2010. Pretty good yield and exploited fishes. *Marine Policy*, 34: 193-196.
- Kompas, T., and Chu, L. 2018. MEY for a short-lived species: A neural network approach. *Fisheries Research*, 201: 138-146.
- Myers, R. A., Barrowman, N. J., Hilborn, R., and Kehler, D. G. 2002. Inferring Bayesian Priors with Limited Direct Data: Applications to Risk Analysis. *North American Journal of Fisheries Management*, 22: 351-364.
- Pascoe, S., Hutton, T., Coglean, L., and Nguyen, V. Q. 2018. Implications of efficiency and productivity change over the season for setting MEY-based trigger targets. *Australian Journal of Agricultural and Resource Economics*, 62: 199-216.

- Pascoe, S., Hutton, T., Plagányi, É., Deng, R. A., Miller, M., Moeseneder, C., and Eves, S. 2020. Influence of environment and economic drivers on fishing effort in Australia's redleg banana prawn fishery. *Fisheries Research*, 227: 105555.
- Pascoe, S. D., Sharp, J. A., and Buckworth, R. C. 2016. Modelling effort levels in a sequential fishery. *ICES Journal of Marine Science*, 73: 503-511.
- Plaganyi, E., Punt, A., Hillary, R., Morello, E., Thebaud, O., Hutton, T., Pillans, R., et al. 2012. Multi-species fisheries management and conservation: tactical applications using models of intermediate complexity. *Fish and Fisheries*, DOI: 10.1111/j.1467-2979.2012.00488.x.
- Plagányi, É., Upston, J., Miller, M., Bishop, J., Deng, R., Buckworth, R., and Hutton, T. 2012. Assessment of the Joseph Bonaparte Gulf Red-legged Banana Prawn (*Penaeus indicus*) Fishery and preliminary TAC estimate for 2012. CSIRO Marine and Atmospheric Research technical report.
- Punt, A. E., Deng, R. A., Dichmont, C. M., Kompas, T., Venables, W. N., Zhou, S., Pascoe, S., et al. 2010. Integrating size-structured assessment and bioeconomic management advice in Australia's northern prawn fishery. *ICES Journal of Marine Science*, 67: 1785-1801.
- Rothlisberg, P. C., and Okey, T. A. 2007. Variation in banana prawn catches at Weipa: a comprehensive regional study. Final Report. Technical report FRDC project no. 2004/024, CSIRO Marine and Atmospheric Science.
- Walters, C., and Maguire, J.-J. 1996. Lessons for stock assessment from the northern cod collapse. *Reviews in Fish Biology and Fisheries*, 6: 125-137.
- Zhou, S., Buckworth, R. C., Ellis, N., Deng, R. A., and Pascoe, S. 2015. Getting all information out of logbooks: estimating banana prawn fishable biomass, catchability, and fishing power increase, with a focus on natural mortality. *ICES Journal of Marine Science*, 72: 54-61.

Appendix 11 – Mud crab sub-model component

Background

The mud crab *Scylla serrata* is a large portunid crab that is widely distributed across the Indo-Pacific and found in mangrove or estuarine habitats. It is a fast-growing, short-lived crab with a longevity of 3-4 years (Heasman and Fielder 1977, Knuckey 1999).

In Australia it is found in mangrove and estuarine habitats in northern and eastern Australia, although their distribution is spatially variable. Juvenile mud crabs (up to 80 mm CW) use the intertidal zone in among the mangroves. The first juvenile stage (3-4mm CW) might settle in the upper sub-tidal before migrating to the intertidal mangrove areas (Alberts-Hubatsch 2015). Sub-adults and adults use the intertidal zone at high tide and retreat to the sub-tidal zone at low tide.

Reproduction

Across the Indo-Pacific, mud crabs have similar-timed reproductive cycles. Most of the mating takes place between August – November, but there is evidence that it occurs throughout the year in some regions (e.g. SE Qld). Mating occurs between mature, intermoult males and soft-shelled females, thus the timing of moulting and temporal coordination is important (Knuckey 1999). As such, females are thought to moult middle of the year during mating season, and males probably moult in Jan-Feb the following year (Knuckey 1999). In general (i.e. averaged across the year), the sex ratio appears to be 1:1, but it does vary, and in some months, females can be more or less abundant (Knuckey 1999).

Female crabs move offshore to spawn in September-November (Figure A48). Reasons behind the migrations are not fully understood, but are thought to be linked to the more stable temperature and salinity environments of the open ocean given larvae are intolerant to low-salinity conditions and prefer intermediate-to-warm temperatures (Hill 1984, Baylon 2010). They may also be linked to maximising the dispersal of eggs/larvae via ocean currents (Hill 1994). Peak spawning is thought to occur between September and December, although it's thought there is some spawning throughout the year (Knuckey 1999). Some females are thought to migrate back to the estuary after spawning (Hyland et al. 1984), and on average, 20% of females with spent ovaries were caught in traps over the summer season in Moreton Bay, SE Queensland (Heasman et al. 1985). After spawning, eggs hatch into free-swimming zoeal larvae approximately 20-40 days after fertilisation. There is then a planktonic larval stage of ~ 1 month, after which larvae move inshore as they moult through 4 zoeal stages into the demersal megalopal stage (Fielder and Heasman 1978). Over the next 1-2 years, crabs develop into subadults and adults. At approximately 12 months after hatching, crabs recruit to the fishery (around Feb-March) (Knuckey 1999). The timing of spawning is similar for both western and eastern Gulf of Carpentaria (GoC), and indeed many other regions.

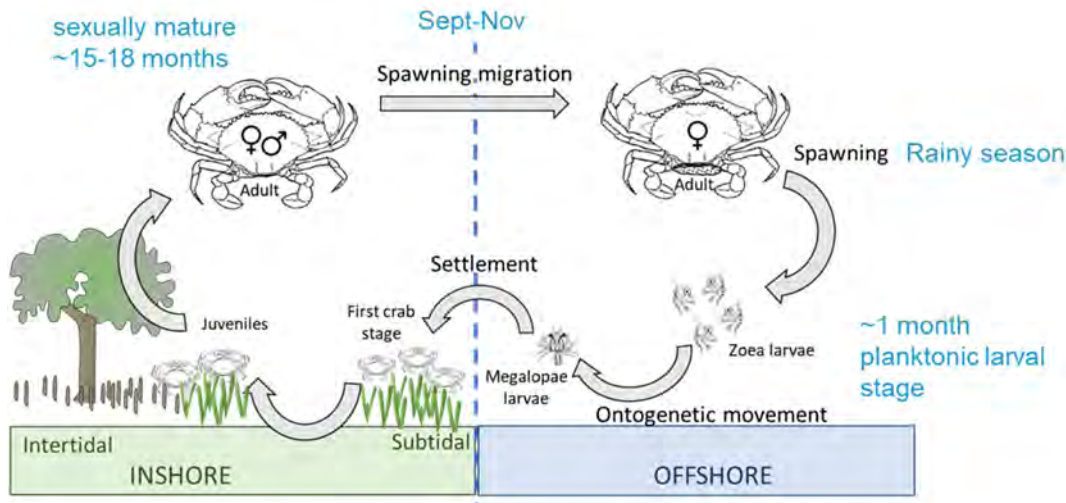


Figure A48. Overview of the Mud crab (*Scylla serrata*) life cycle and associated habitat use. Adapted from Alberts-Hubatsch et al. (2016).

Growth

Male crabs grow faster than females (Sara 2010) and length-weight (male and female) and von Bertalanffy (combined sexes) estimates are available from Knuckey (1999) for the western GoC.

Length-weight functions:

$$w = 1.4148 \times 10^{-5} l^{3.54960} \quad \text{for males}$$

$$w = 2.8914 \times 10^{-4} l^{2.88207} \quad \text{for females}$$

where w is the weight and l is the carapace width.

Carapace width (CW) can be computed using the von Bertalanffy growth function:

$$L_a = 193.6(1 - e^{-1.14(a-0)}) \quad \text{combined for sexes}$$

where L_a is the carapace width at age a .

Mortality

Knuckey (1999) obtained 3 estimates of natural mortality M using three different methods following Beverton and Holt (1957); Pauly (1980) and Rikhter and Efanov (1976). These were 1.31, 1.16 and 1.18 per year respectively, providing a mean of $M = 1.21$ per year.

Total mortality (Z) estimates for the Adelaide River, Roper River and McArthur River (western side of GoC) were 2.55, 2.92 and 3.65 per year respectively (Knuckey 1999). Thus, estimates of fishing mortality F for the Adelaide River, Roper River and McArthur River were initially estimated as 1.34,

1.71 and 2.44 per year (respectively) based on catch-curve analysis. However, Knuckey (1999) also used the Leslie method (Leslie and Davis 1939) for estimating stock size and applied this method to all the data (i.e. not individual river data) and found that F averaged at 2.15 per year with a max of 3.77 per year in 1990 and a min of 1.18 per year in 1991. Based on assumptions of the Leslie method, Knuckey (1999) suggests these estimates may be inflated.

Mud crab Fishery

In the GoC, mud crab fisheries fall under the Northern Territory and Queensland jurisdictions. Below and in Figure A50 we summarise the similarities and differences in the mud crab fishery for each state/territory, which we then try and capture in the mud crab model.

Northern Territory (Western GoC)

In the Western Gulf of Carpentaria Mud Crab Fishery (WGOCMCF), catches are taken from three main rivers: Roper River, McArthur River and Adelaide River (Knuckey 1999). Both males and females are caught and *S. serrata* make up more than 99% of the catch in the commercial fishery (Hay and Calogeras 2001). Recreational catch is only available for 3 periods and is thought to have contributed to <5% of commercial catch.

There is a skewed sex ratio in catches, which changes throughout the year (Figure A49; (Knuckey 1999)). However, sexes are not separated in the reported catches.

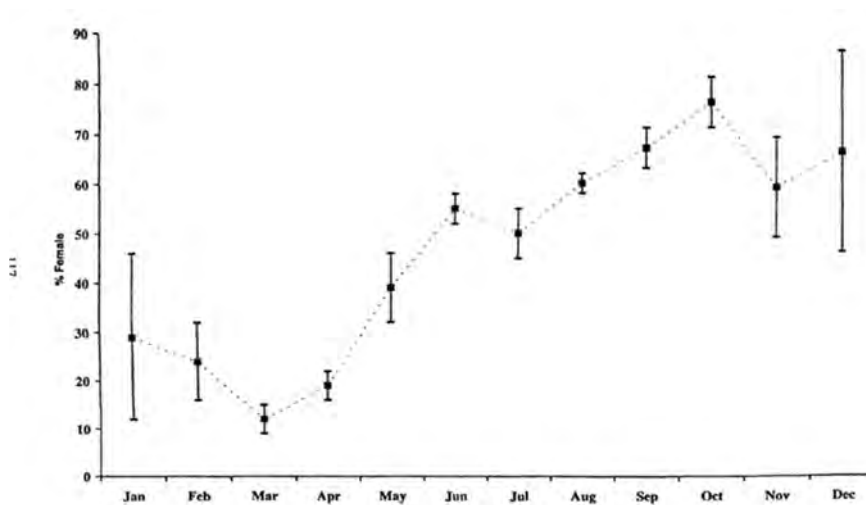


Figure A49. Mean (\pm SD) percentage of females from sampled catches pooled across years (1991-1995) and river systems (Adelaide, McArthur and Roper Rivers). Taken from Knuckey (1999).

Mud crabs are primarily caught using baited pots (traps). Most commercial fishers (crabbers) operate from land-based camps using a small dinghy with an outboard motor. Pots are laid out 50-100m apart, with most of the fishing effort taking place outside of the wet season. There is minimal fishing effort between December-February (Knuckey 1999), which is when females migrate offshore to spawn and summer rains make fishing conditions unfavourable.

A management plan for mud crabs caught in Northern Territory was first introduced in 1985, where the minimum legal size (MLS) was introduced at 130mm carapace width (CW) for both sexes. It is thought however, that gear selectivity overruled this size and likely only caught crabs around 140mm CW (Knuckey 1999). Female MLS increased to 140 mm CW in 1996 and in 2001 fishers were now

required to return soft-shelled crabs. In 2006, the minimum legal size (MLS) increased again, to 140mm CW for males and 150mm CW for females (the current MLS). In 2018, the compulsory use of escape vents was initiated.

Fishery input controls include a minimum size and restrictions on effort (number of licenses and pots per license). Crabs take approx. 12 months to reach 130 mm CW and 16 months to reach 150 mm CW, the latter is when they are fully vulnerable to fishing gear.

Queensland (Eastern GoC)

Only males are caught in the Queensland mud crab fishery, where the MLS has been set at 150mm CL since before the 1980s. Crabs were caught from rigid pots before the 1990s. Since 1990s, commercial crabbers now use collapsible mesh pots. High catches are common in Nov-May, with low catches occurring Apr-Oct. Catch data are available from QFB and commercial fishers' logbooks (CFISH) from QDPI for 1988-present. There have been changes in the way catch and effort data have been reported – in the early 1990s, cumulative catches were reported over weeks and even months with no corresponding effort data. Since late 1990s this type of reporting has diminished and thus not thought to impact cpue from late 1990s onwards (Brown 2010). Effort creep is thought to be a problem though. Although there is less variability in catches from the Queensland GoC side compared to the WGOCMCF (Northern Territory GoC), there is thought to be much more uncertainty in the catch data. Northrop et al. (2019) suggest there is 30% over-reporting in catch data since 2003. Brown (2010) found a bias in catch rates from logbooks and that the number of pots operated and lifted was not accurately recorded and thus unreliable (Wang et al. 2012).

Recreational catch is estimated to be ~12% of commercial catch in GoC (Northrop et al. 2019), based on recreational fishing surveys that were done in 1997, 1999, 2002, 2005.

Fishery input controls include minimum size and restrictions on effort. However, restrictions on effort are less restrictive in Qld compared to NT (Knuckey 1999). The MLS has remained unchanged but changes in the C1 license occurred in 2016. With an increased pot allowance (individuals may now hold 2 licenses). QDAF is currently exploring the possibility of an ITQ system for the future and a TAC (Northrop et al. 2019).

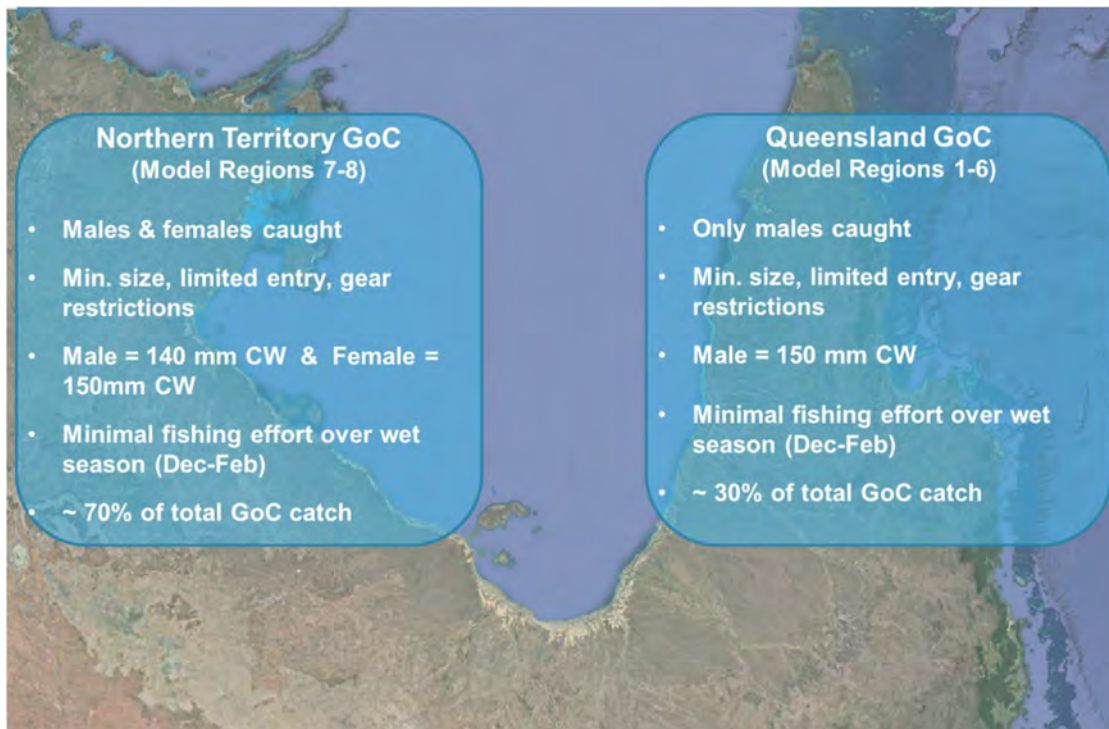


Figure A50. Summary of some of the key similarities and differences between the NT and Qld mud crab fisheries.

Environmental Drivers

Offshore environments

Temperature and salinity are thought to have the greatest impact on the life cycle of mud crabs, specifically the larval and juveniles stages (Hill 1984). Adult mud crabs migrate offshore to spawn, likely due to the larvae requiring an environment with more stable temperatures and salinity.

Inshore environments

Fluctuations in catch rate and abundance are thought to be impacted by environmental variables. Mud crabs are hypothesized to be strongly influenced by the ENSO, particularly La Nina cycles which bring cooler temperatures and increased rainfall and can positively affect the occurrence and reproduction of mud crabs (Meynecke et al. 2006, Meynecke and Lee 2011, Meynecke et al. 2012a, Meynecke et al. 2012b). Both Robins et al. (2005) and Meynecke et al. (2012a) have shown that mud crab catches in central-to-northern Australia are correlated with rainfall or river flow (sometimes approximated using the Southern Oscillation Index – SOI).

In south-east Queensland, Loneragan and Bunn (1999) suggested that river flow might increase the catchability of mud crabs by causing them to move downstream (away from low salinity) and into fishing grounds. As such, less adults upstream would result in reduced competition for burrows and reduced cannibalism by larger crabs. This could lead to increased recruitment of juveniles into the

fishery (i.e. more juveniles survive), possibly causing increased catches the following year (lag effect).

Robins et al. (2005) found that annual catches of mud crabs in central/northern Queensland were positively correlated with summer river flow. Similarly, two hypotheses were put forward, namely the catchability and recruitment hypotheses:

- (1) increased catchability due to more downstream movement of crabs from increased flow; and/or
- (2) increased survival of juveniles through reduced cannibalism and/or competition for burrows due to emigration downstream of adults.

They found that mud crab catch was positively correlated with autumn rain, with a 2-year lag for the Fitzroy Region, possibly supporting the recruitment hypothesis (Robins et al. 2005).

Meynecke et al. (2012a) found that 30-40% of the variation in CPUE could be explained by rainfall and SOI. Peaks in catches coincided with La Nina cycles. Both 1-year and 2-year time-lags (matching the life cycle of mud crabs) were also significantly correlated with SOI and rainfall. A good wet season resulted in high catches in the following dry season and the best environmental predictor for fluctuations in *S. serrata* catch variability was SOI with a 6-9 month lag (Meynecke et al. 2012a). The increased river flow during the rainy season likely enhances activity of adult mud crabs and thus triggers their movement away from low salinities (i.e. downstream) and hence increases catchability (Butcher et al. 2003, Meynecke et al. 2010, Alberts-Hubatsch et al. 2016)

Small catches in the GoC have been linked to high migration rates out of fishing areas and recruitment failure, as a consequence of extended periods of freshwater run-off (Helmke et al. 1998). Often the effects of heavy and prolonged flooding are evident with a 1-2 year lag affect (Meynecke et al. 2012a). These negative impacts are thought to be linked to die-off of seagrass beds, which are important habitat for newly settled and juveniles mud crabs (Meynecke et al. 2012a). Would expect that mangrove habitat loss would also negatively impact juvenile crabs – need to see if any links to 2016 mangrove die-off recorded?

In south-east Queensland, catches have been shown to be correlated with temperature but not salinity (Hill 1980, Williams and Hill 1982). The optimal water temperature for *Scylla serrata* is between 28 and 32 C with survival significantly reduced when temperatures drop below 20 C (Heasman et al. 1985, Ruscoe et al. 2004, Robertson 2011). Fisherman have seen drastic drops in catches when water temperatures fall below 20 C or go above 35 C (Meynecke et al. 2010).

Methods

Data

Mud crab catch and effort data were provided by the Northern Territory of Australia Department of the Primary Industry and Resources and the State of Queensland Department of Agriculture and Fisheries.

Northern Territory Data

Northern Territory fishery data from the GoC used in the mud crab component of the model included monthly catch (kg) and effort (days fished per month) for the years 1983-2018. These data were available at a spatial resolution of 1° x 1° grids. Data from each grid were allocated to a model region based on where the centre of the grid fell (i.e. in model region 7 or model region 8). The only

exception was grid 1435 (Figure A51) whose centre fell in model region 8 but the data were mostly obtained from the lower section of the grid (Roper River region), and thus data from this grid were allocated model region 7. Catch data are not validated (Robins et al. 2020) but are thought to be fairly representative of mud crab abundance (Knuckey 1999).

Queensland Data

Queensland fishery data from the GoC used in the mud crab component of the model included monthly catch (kg) and effort (days fished per month) for the years 1989-2019. Additional forms of effort include number of pots and pot lifts, however these were considered unreliable and thus not included. Catch and effort were allocated to model regions 1-6 and where no allocation could be made, these records were excluded (< 50 out of approx. 140 000 records). Offshore mud crab catch and effort were also excluded, as was any crab product description that wasn't recorded as "whole dead" (< 45 records).

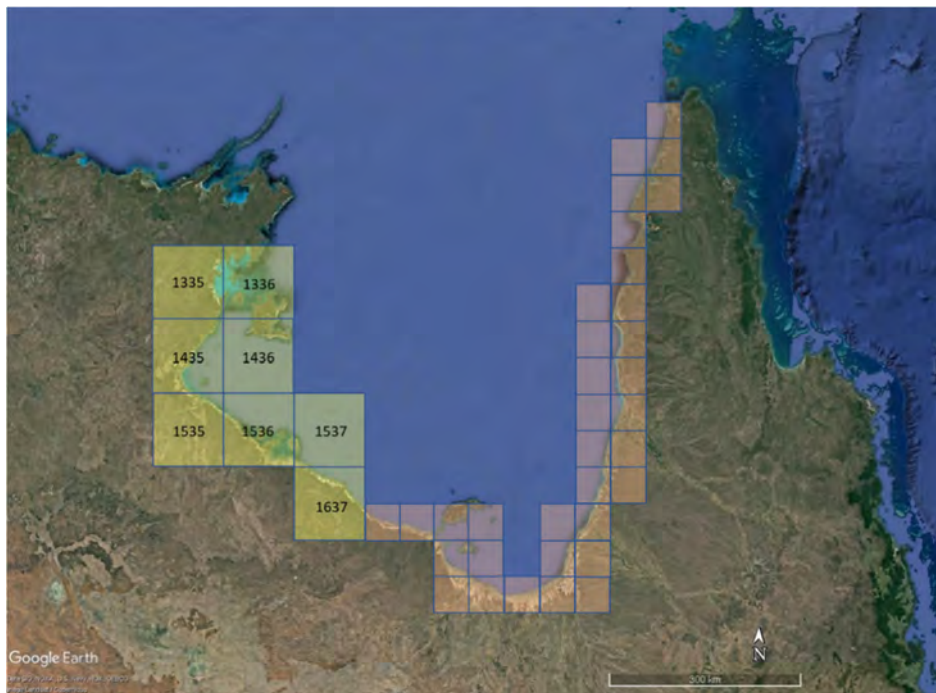


Figure A51. Fishery reporting grids in the Gulf of Carpentaria for Northern Territory (yellow, 1° x 1°) and Queensland (orange, 0.5° x 0.5°). Adapted from Robins et al. (2020).

Mud crab sub-model

For each of the eight model regions, male and female mud crabs are modelled separately using an age-structured population model with a monthly time-step and three ages classes (Figure A52).

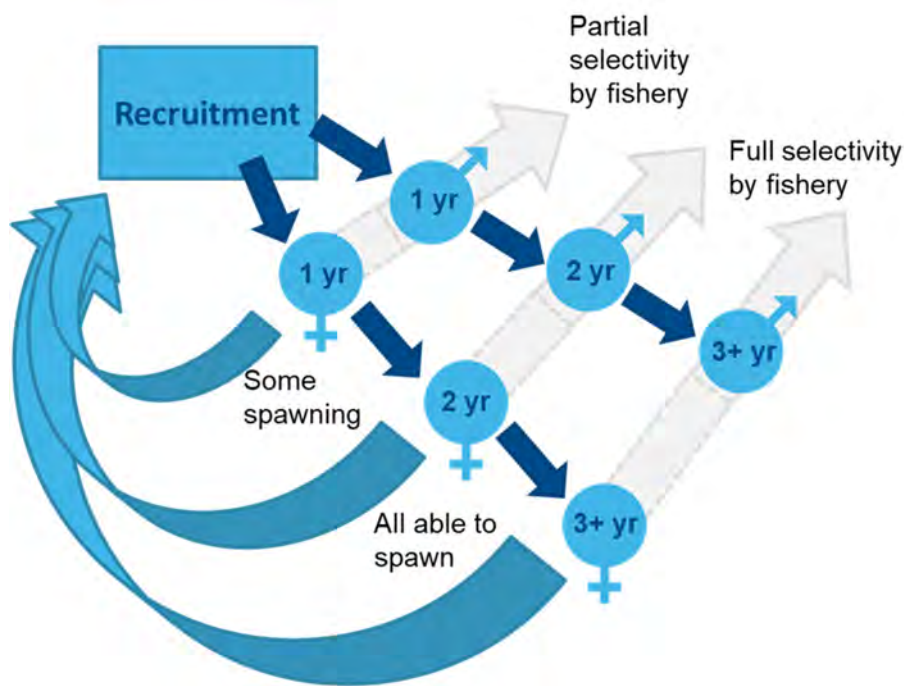


Figure A52. Schematic of mud crab model structure showing age classes for male and female crabs captured in model.

Step-by-step build-up of mud crab sub-model

Using the model structure from Fig A30, we developed a mud crab sub-model by starting with the simplest way to explain observed catches (Model 1a; Table A14) and added complexity to the model (Models 1b-1d; Table A14) in a step-wise process to assess which sub-model would best explain mud crab catches in each region of the GoC.

Table A14. Several mud crab sub-models that were developed to assess which was best able to explain the observed mud crab catches in the GoC.

Sub-Model	Environmental drivers included
1a	None (only effort used to explain observed catch)
1b	Flow linked to mud crab recruitment
1c	Flow linked to mud crab recruitment and catchability
1d	Flow linked to mud crab recruitment and catchability; SOI linked to mud crab recruitment in Regions 3-6 and temperature linked to mud crab mortality in Region 7 only.

The sub-model that best explained the catches in the GoC was then chosen as the mud crab sub-model for MICE version 1. Four other versions of this sub-model were then developed to account for uncertainty in parametrisation (i.e. mud crab sub-models in MICE versions 2-4, see Section 11 in main report).

Mud crab Model Equations

Numbers-at-age

$$N_{r,y+1,1,1}^{mudcr,sex} = R_{r,y+1,1}^{mudcr} \quad a = 1; m = 1 \quad (1)$$

$$N_{r,y,m+1,1}^{mudcr,sex} = N_{r,y,m,1}^{mudcr,sex} e^{-M_{r,y,m,1}^{mudcr}} - C_{r,y,m,1}^{mudcr,sex} + R_{r,y,m+1}^{mudcr} \quad a = 1; m = 2 \text{ to } 11$$

$$N_{r,y,m+1,a}^{mudcr,sex} = N_{r,y,m,a}^{mudcr,sex} e^{-M_{r,y,m,a}^{mudcr}} - C_{r,y,m,a}^{mudcr,sex} \quad 2 \leq a < z; m = 1 \text{ to } 11 \quad (2)$$

$$N_{r,y+1,1,a+1}^{mudcr,sex} = N_{r,y,m,a}^{mudcr,sex} e^{-M_{r,y,m,a}^{mudcr}} - C_{r,y,m,a}^{mudcr,sex} \quad m = 12$$

$$N_{r,y+1,1,z}^{mudcr,sex} = \left(N_{r,y,12,z}^{mudcr,sex} e^{-M_{r,y,m,z}^{mudcr}} - C_{r,y,12,z}^{mudcr,sex} \right) + \left(N_{r,y,12,z-1}^{mudcr,sex} e^{-M_{r,y,m,z-1}^{mudcr}} - C_{r,y,12,z-1}^{mudcr,sex} \right) \quad a = z; m = 12 \quad (3)$$

where

$N_{r,y,m,a}^{mudcr,sex}$ are the number of mud crabs (sex disaggregated), at age a in year y and month m in region r .

$R_{r,y,m}^{mudcr}$ is the mud crab recruitment at the start of month m in year y for region r

$M_{r,y,m,a}^{mudcr}$ is the mud crab natural mortality rate (assumed to be the same for male and female crabs),

where $M_{r,y,m,a}^{mudcr} = M_{base}^{mudcr}$ except for a temperature dependent form when $r = 7$ and

$temp_{r,y,m} > temp^{opt}$, then $M_{r,y,m,a}^{mudcr} = M_{base}^{mudcr} \cdot M_{r,y,m}^{mudcr,temp}$. See Eq 13 for further details on temperature-related mortality.

$C_{r,y,m,1}^{mudcr,sex}$ is the numbers of mud crabs (sex disaggregated) of age a caught in month m of year y in region r

z is the plus group (for mud crabs this is age 3+ years)

Recruitment

Recruitment depends on the female spawning biomass. We assume that half the recruitment will be male and half the recruitment will be female. We also assume spawning takes place each month, with the main spawning event in September and October (J. Robins pers comm), and recruitment is defined as crabs entering the 1-year old age class, which are spawned 14 months prior. Recruitment is also assumed to be influenced by flow (and in regions 3-6, also SOI) through a recruitment multiplier as follows:

$$\begin{aligned}
R_{r,y,m+1}^{mudcr} &= 0.5 \frac{\alpha_r^{mudcr} B_{r,y-1,m-1}^{spn,mudcr}}{\beta_r^{mudcr} + B_{r,y-1,m-1}^{spn,mudcr}} (\sigma_{r,y-1,m+1}^{flow_mudcr} + \sigma_{y-1,m+1}^{SOI_mudcr}) \quad \text{for } 1 < m < 12 \\
R_{r,y+1,m}^{mudcr} &= 0.5 \frac{\alpha_r^{mudcr} B_{r,y-1,m+10}^{spn,mudcr}}{\beta_r^{mudcr} + B_{r,y-1,m+10}^{spn,mudcr}} (\sigma_{r,y,m}^{flow_mudcr} + \sigma_{y,m}^{SOI_mudcr}) \quad \text{for } m = 1 \\
R_{r,y+1,2}^{mudcr} &= 0.5 \frac{\alpha_r^{mudcr} B_{r,y-1,12}^{spn,mudcr}}{\beta_r^{mudcr} + B_{r,y-1,12}^{spn,mudcr}} (\sigma_{r,y,2}^{flow_mudcr} + \sigma_{y,2}^{SOI_mudcr}) \quad \text{for } m = 12 \\
\text{with } \sigma_{y,m}^{SOI_mudcr} &= 0 \quad \text{for } r = 1, 2, 7, 8
\end{aligned} \tag{4}$$

See Eqs 10 and 12 below for details on the recruitment multiplier.

$$B_{r,y,m}^{spn,mudcr} = \sum_{a=1}^z f_{m,a}^{mudcr} \cdot W_{m,a}^{mudcr,sex} \cdot N_{r,y,m,a}^{mudcr,fem} \tag{5}$$

Where $B_{r,y,m}^{spn,mudcr}$ is the spawning biomass of females in month m of year y in region r , $f_{m,a}^{mudcr}$ is the proportion of females of age a , spawning in each month and $W_{m,a}^{mudcr,sex}$ is the weight at age for female crabs.

Stock recruitment relationship

For each model region, the stock-recruitment relationship is re-parameterised in terms of the model start year (1970) equilibrium spawning biomass and the “steepness”, h^{mudcr} , of the stock-recruitment relationship, which is the proportion of the virgin recruitment that is realized at a spawning biomass level of 20% of the virgin spawning biomass.

$$\begin{aligned}
\beta_r^{mudcr} &= \frac{(1 - h^{mudcr}) \sum_m B_{r,1970,m}^{spn,mudcr}}{5h^{mudcr} - 1} \\
\alpha_r^{mudcr} &= \frac{\beta_r^{mudcr} + \sum_m B_{r,1970,m}^{spn,mudcr}}{SPR_r^{virg,mudcr}} \quad \text{where } SPR_r^{virg,mudcr} = \sum_m B_{r,1970,m}^{spn,mudcr}
\end{aligned} \tag{6}$$

Catches

Predicted catches (mass) of mud crabs is computed as:

$$\hat{C}_{r,y,m}^{mudcr,sex} = q_r^{mudcr} q_{r,y,m}^{flow_mudcr} E_{r,y,m}^{mudcr} B_{r,y,m}^{exp_mudcr_sex} \tag{7}$$

where

q_r^{mudcr} is the catchability coefficient

$q_{r,y,m}^{flow_mudcr}$ is the catchability of crabs due to river flow in any given month m in year y and region r (see Eqn 11).

$E_{r,y,m}^{mudcr}$ is the monthly fishing effort for each year in each region

and $B_{r,y,m}^{exp_mudcr,sex}$ is the commercially exploitable biomass of female and male mud crabs in each month, year and region calculated as follows:

$$B_{r,y,m}^{exp_mudcr,sex} = \sum_{a=1}^z S_{r,y,m,a}^{mudcr,sex} A_{r,m}^{mudcr,sex} W_{m,a}^{mudcr,sex} N_{r,y,m,a}^{sp,sex} e^{-M_{r,y,m,a}^{sp}} \quad (8)$$

where $A_{r,m}^{mudcr,sex}$ is the availability of female and male mud crabs in month m for region r . Note that $A_{r,m}^{mudcr,sex}$ omits subscript y for simplicity but uses adjusted values for NT with $y < 2001$ (See Appendix 16, Table A26).

$S_{r,y,m,a}^{mudcr,sex}$ is the selectivity for female and male mud crabs (See Appendix 16, Table A26)

The fished proportion of mud crabs can be calculated as follows:

$$F_{r,y,m}^{sp,sex} = \widehat{C}_{r,y,m}^{sp,sex} / B_{r,y,m}^{exp_sp,sex} \quad (9)$$

Environmental Drivers

The main physical drivers of mud crabs currently included in the model include river flow, SOI and air temperature.

River flow

River flow is hypothesised to increase survival of younger crabs given that adults are assumed to move downstream with good flows, thereby decreasing cannibalism on younger crabs and reducing competition for burrows (Robins et al. 2005). At very heavy flows, it is possible that survival of younger crabs is reduced. We therefore linked river flow to (1) boost or reduce recruitment (defined as number of crabs surviving and entering the 1-year old age class) and (2) increase catchability of adult crabs.

River flow model outputs were available at a weekly scale and summed up to a monthly timeseries for input into the mud crab model component.

Flow was linked to mud crab recruitment by calculating a flow residual $\sigma_{r,y,m}^{flow_mudcr}$ which either boosted or reduced recruitment (see Eq 4) based on the following relationship:

$$\sigma_{r,y,m}^{flow_mudcr} = thres^{mudcr} \left(flm_{r,y,m} - thresM^{mudcr} \right)^2 + c_{\sigma} \quad (10)$$

where

$thres^{mudcr}$ is a parabolic parameter for the $mudcr$ parabolic equation, estimated in the model

$flm_{r,y,m}$ is the standardised monthly end-of-system flow in catchment region r in year y in month m (based on average or cumulative weekly standardised flows)

$thresM^{mudcr}$ is the optimal flow parameter, which maximises recruitment (see Appendix 16, Table A26)

c_{σ} is a recruitment-related flow scaling parameter

Flow was also linked to mud crab catchability through an additional catchability term $q_{r,y,m}^{flow_mudcr}$ that was modelled using a logistic relationship:

$$q_{r,y,m}^{flow_mudcr} = \frac{1}{1 + c_q + e^{-\frac{(flm_{r,y,m} - thresM^{SP})}{thres^q}}} \quad (11)$$

where

$thres^q$ is a catchability-related flow parameter, estimated in the model

c_q is a catchability-related flow scaling parameter

Southern Oscillation Index (SOI)

The SOI was also linked to mud crab recruitment (regions 3-6) by calculating a SOI recruitment multiplier $\sigma_{r,y,m}^{SOI_mudcr}$ which either boosted or reduced recruitment (see Eq 4) based on the following relationship:

$$\text{If } r = 3-6: \sigma_{y,m}^{SOI_mudcr} = \eta \cdot \tau^{SOI} \quad (r=1,2,7,8: \sigma_{y,m}^{SOI_mudcr} = 0) \quad (12)$$

Where η is a scaling parameter and $\eta = -1$ if $SOI < -7$ or $\eta = 1$ if $SOI > 7$ and τ^{SOI} is a recruitment-related SOI parameter that is estimated in the model.

Air temperature

Mud crab survival is thought to be impacted by heat stress in some regions (Robins et al. 2020). Hence, for region 7, we included an additional mortality term that when air temperature exceeded the long-term mean, $temp^{opt}$, in any given month, then $M_{r,y,m,a}^{mudcr} = M_{base}^{mudcr} \cdot M_{r,y,m}^{mudcr,temp}$ otherwise at all other times (and for all other regions) $M_{r,y,m,a}^{mudcr} = M_{base}^{mudcr}$

for which

M_{base}^{mudcr} is the base monthly mortality, set at 0.1 (Knuckey 1999) and

$$M_{r,y,m}^{mudcr,temp} = \tau^{temp} \cdot temp_{r,y,m} + c^{temp} \quad \text{for } r=7 \quad (13)$$

where

τ^{temp} is a temperature parameter, estimated in the model

$temp_{r,y,m}$ is the monthly air temperature. Observed air temperatures were taken from a relevant BOM station in that region (or from a proxy station thought to be representative of the region – see Appendix 6).

c^{temp} is a temperature scaling parameter

The Likelihood Function

See Appendix 16, Table A24 for details on the Likelihood Function.

Input parameters

See Appendix 16, Table A26 for a list of input parameters for the mud crab sub-model.

Modelling river flow relationships

River flow is hypothesised to increase survival of younger crabs given that adults are assumed to move downstream with good flows, thereby decreasing cannibalism on younger crabs and reducing competition for burrows (Robins et al. 2005). At very heavy flows, it is possible that survival of younger crabs is reduced (Robins et al. 2005). We therefore linked river flow to (1) boost or reduce recruitment (defined as number of crabs surviving and entering the 1-year old age class) and (2) increase catchability of adult crabs. We did this by linking in river flow to mud crab recruitment by calculating a flow-related recruitment multiplier which then boosted or reduced recruitment based on a dome-shaped parabolic relationship (Figure A53).

Other relationships we considered included a linear relationship, in which mud crab recruitment increases linearly with river flow (Figure A53). However, this relationship would fail to capture any negative impacts on mud crabs under extreme river flows e.g. flood events and we do not consider it plausible for mud crab recruitment to be maximised under the most extreme river flows. For this reason, we did not explore this relationship further.

We also considered using a logistic-type relationship (Figure A53), in which mud crab recruitment could be boosted by river flow up until a certain point (an optimal flow) after which, it plateaued (i.e. no additional increase in river flow impacted mud crab recruitment). This relationship seemed more plausible than a linear relationship, although still does not capture the fact that very large flood flows may negatively impact mud crabs (Robins et al. 2005). Nonetheless we explore this option to see if it could adequately and plausibly capture variability in mud crab abundance.

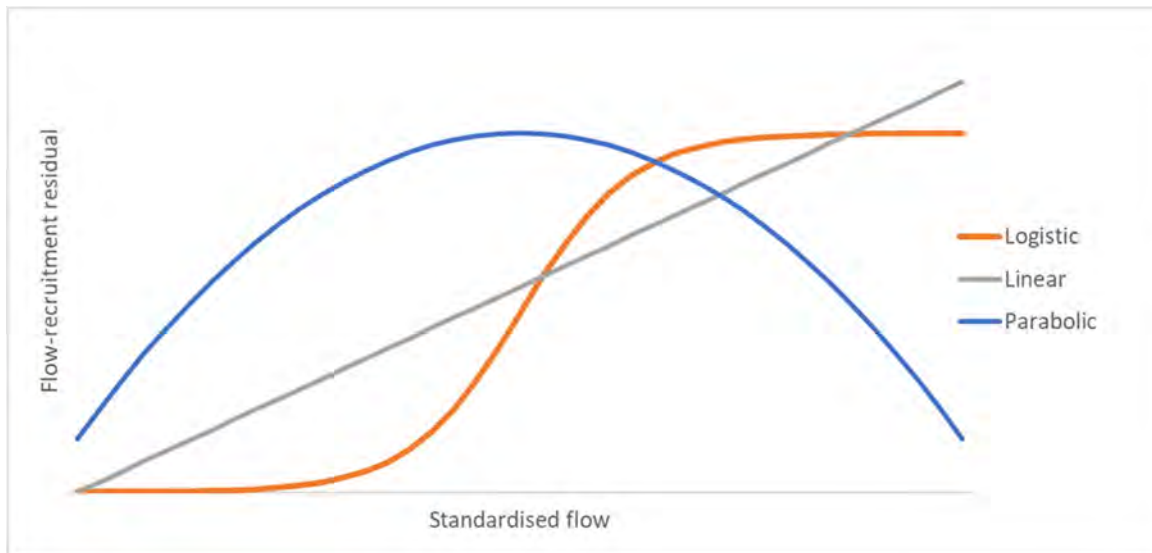


Figure A53. Three flow-recruitment relationships were initially considered when choosing how to link in river flow to mud crab population dynamics (via recruitment).

Mud crab sub-model ensemble

A summary of the mud crab sub-model ensemble is provided in Table A15. The base mud crab sub-model (Model version 1) used a fixed natural mortality of $M = 0.1$, which is an average of three estimates from Knuckey (1999) and also the value previously used in Grubert et al. (2019). Model start year (1970) spawning biomass was estimated in this model and flow was linked to mud crab population dynamics via a parabolic flow relationship (see Eq 10 and section below on modelling river flow relationships). We felt that this was the most plausible relationship to capture hypotheses around increased survival of younger crabs (following adult movement downstream) with good flows, and reduced survival of younger crabs under very heavy (e.g. large or prolonged flood) flows. This version of the model also assumed a stock-recruitment steepness parameter of $h = 0.6$ in the Beverton-Holt stock-recruitment relationship. We acknowledge that h can be quite high for some crustacean species, including crabs, but several crab species assessments use low values for h too ranging from 0.24 – 0.8 (O'Neill et al. 2010, Froehlich et al. 2017, Lovett et al. 2020)) and for prawns in the NPF e.g. $h = 0.3$ for tiger and endeavour prawns (Deng et al. 2020b) and $h = 0.6$ for redleg banana prawns (Plagányi et al 2021). Additionally, we felt our choice of h was reasonable because we define mud crab recruitment as 1-yr old crabs entering the population and thus assumed it accounts for additional density-dependent factors in the first year, plus other density-independent factors influencing mud crab recruitment, which we assume dominates density-dependent factors.

In our Model 2 version, we kept the model the same as in Model 1 except changed the optimal flow parameter by reducing it by 20%. This allowed us to capture some uncertainty around what this optimal flow value was, but still allowed it to be somewhat more than the long-term mean flow. We decided not to change the entire flow relationship given we felt the parabolic relationship used was most plausible, and also provided the best model fit compared to e.g. a logistic relationship (see Table A17).

In Model 3, we adjusted the mud crab natural mortality by increasing it by 20%. This value was still within the range estimated by Knuckey (1999).

In Model 4, we doubled the starting biomass of mud crabs given there a very small spawning biomass was estimated for the model start year (although allows for a 'burn in' period before the fishery

formally begins in 1983 (NT) and 1989 (Qld)). We also chose to use a much larger starting biomass to see if this would help the model capture some of the peaks in catches in the Roper region (region 7).

In Model 5 we used a more conservative steepness parameter of $h = 0.4$. We acknowledge that this is perhaps lower than might be expected, but we found an improved fit to the data when h was reduced rather than increased. In fact, Model 5 had the best fit of all mud crab sub-model versions.

Table A15. Summary of the mud crab sub-model ensemble

Model version	Description
Model 1	Base model: fixed $M = 0.1$; starting biomass estimated; parabolic flow relationship with fixed optimal flow and slope estimated; fixed steepness parameter $h = 0.6$
Model 2	As in Model 1 except that flow relationship parameter changed (optimal flow reduced by 20%)
Model 3	As in Model 1 except that a larger natural mortality M (20% increase) used
Model 4	As in Model 1 except that model starting biomass not estimated and instead is doubled from Model 1 version.
Model 5	As in Model 1 except that the Stock-recruitment steepness parameter h is reduced to 0.4

Results

Fishing effort

Along the Queensland side of the GoC, effort is recorded as the number of pots lifted, the number of pots used and number of days fished. The number of pots lifted and used is thought to be unreliable (JR pers comm.) and so we have focused on number of days fished per month and/or year.

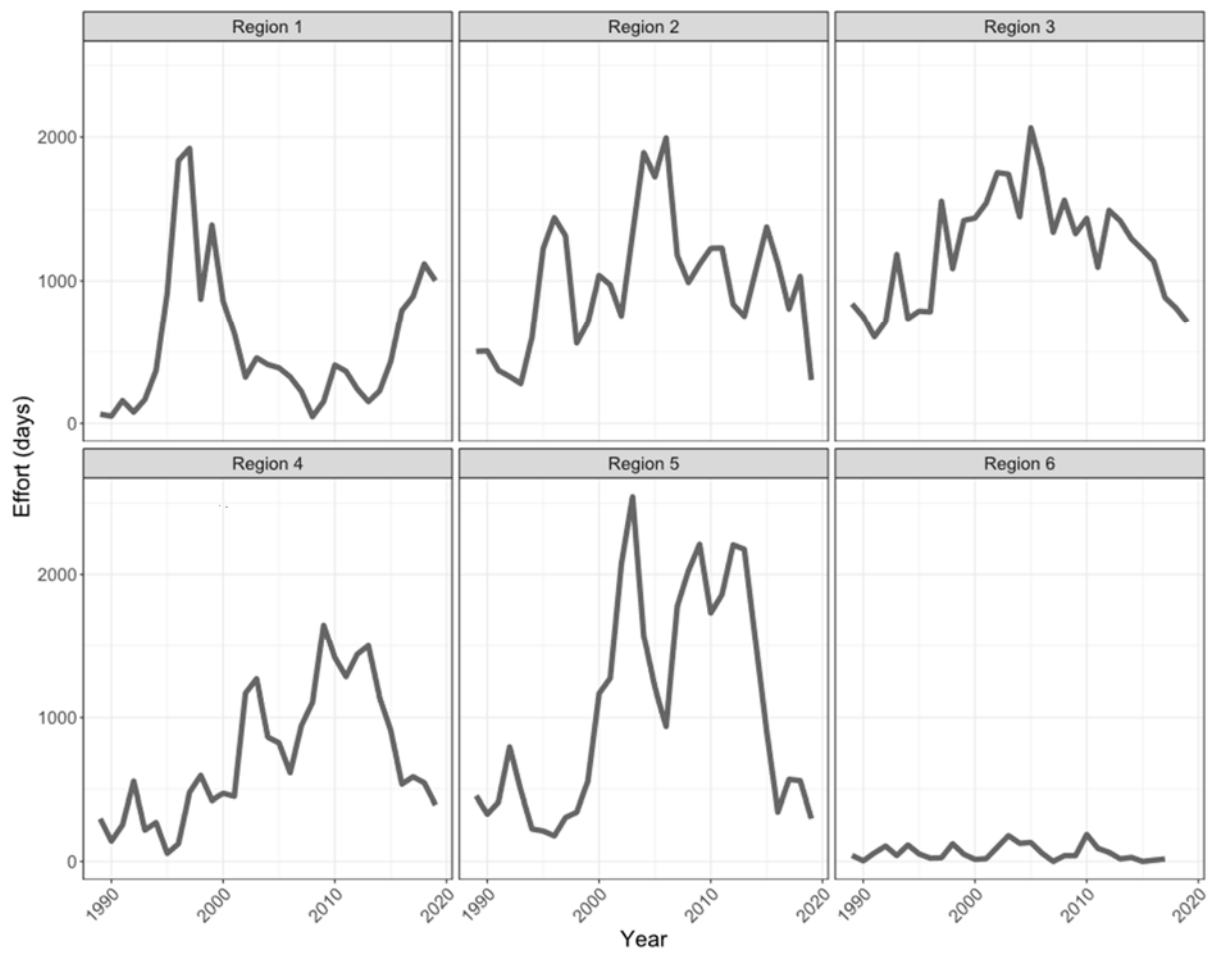
Effort has increased since formal records of the fishery began in the late 1980s, with peaks in effort ranging from 1500 to 2500 days per year between the mid-1990s (Region 1) to mid-to-late 2000s (Regions 4 and 5) (Figure A54(A)). Much of the effort in the Queensland GoC has been from Regions 2-5, with very little effort in Region 6 (Figure A54(A)).

Along the NT side of the GoC, much of the effort is concentrated around the Roper River area (Region 7) (Figure A54(B)). Effort (number of pots) has increased since the early 1980s, peaking in the early 2000s at around 700 000 pots per year. Since then effort has declined, with a slight rise in 2010, followed by another decline and then a more recent increase since 2016 (Figure A54(B)). In contrast, there has been much less effort in Region 8 (less than 100 000 pots per year until the early 2000s), with a peak in effort in the mid-2000s, followed by a decline (Figure A54(B)).

Catch

Mud crab catches have increased since the 1980s, following a rise in effort in both the Queensland and Northern Territory GoC fisheries (Figure A55). Peaks in catches have generally followed peaks in fishing effort, although there were cases where extremely large catches occurred with average fishing effort (e.g. Region 3 and Region 7 in the early 2000s), suggesting additional drivers of increased catch. Catches from the Queensland side of the GoC reached only 80 tons in some years (Region 3) and were often an order of magnitude smaller than some of the large catches (>750 tons) from the Northern Territory (Region 7) side (Figure A55). Catches from Region 6 were small compared to all other regions, similar to patterns in effort (Figure A54(A)). Catches from Region 8 were comparable to catches from the Queensland GoC but far less than the neighbouring Region 7.

(A)



(B)

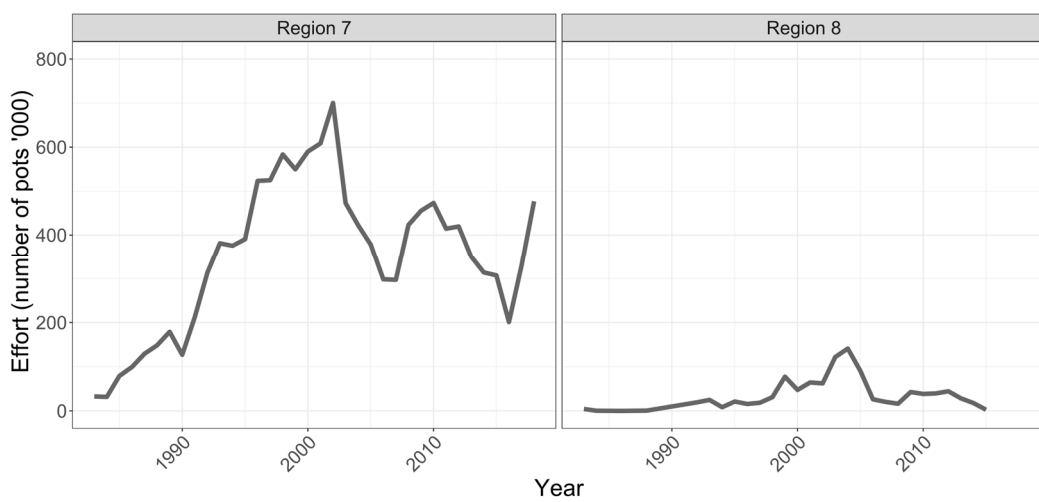
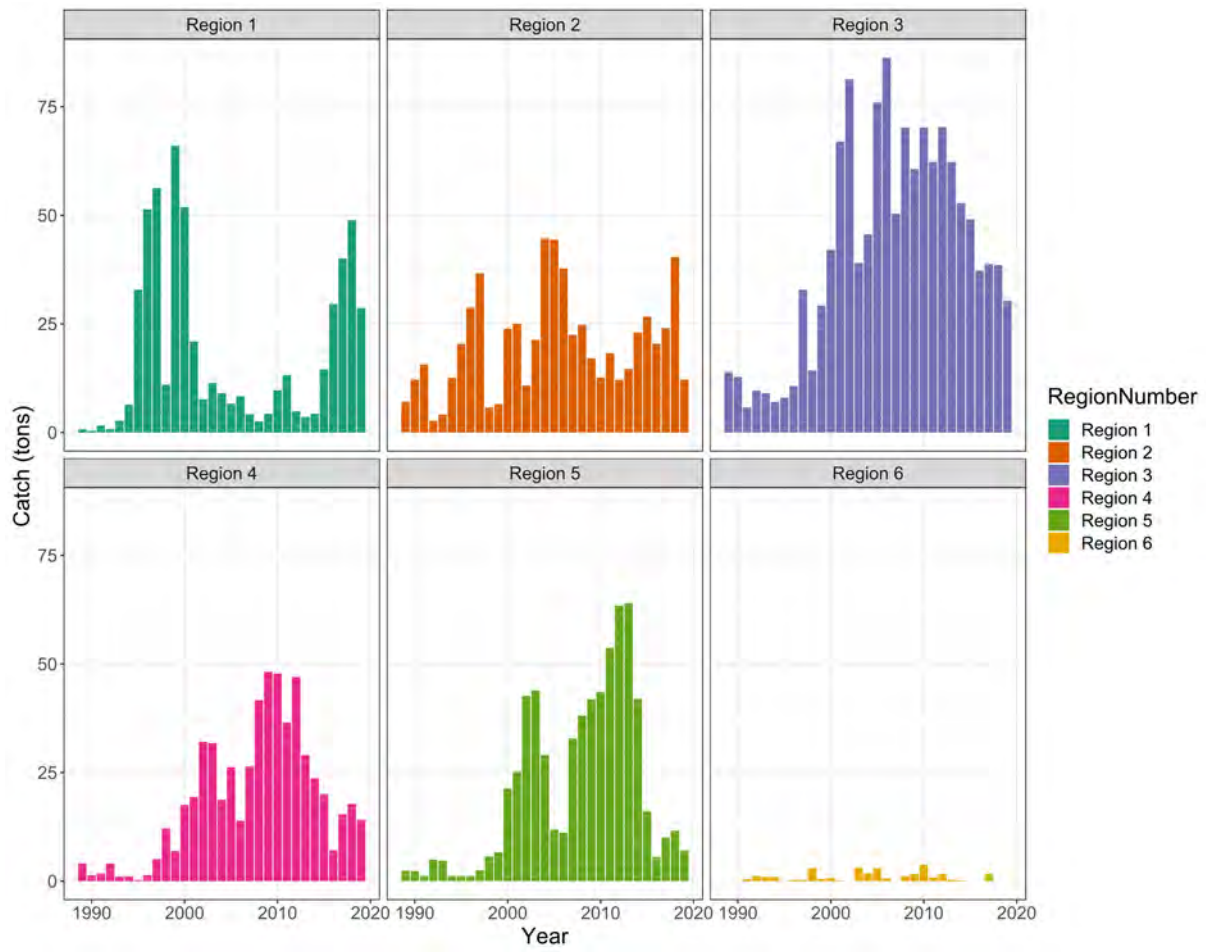


Figure A54. Annual effort per model region in (A) the Queensland GoC mudcrab fishery (number of days per year) and (B) the Northern Territory GoC mud crab fishery (number of pots per year).

(A)



(B)

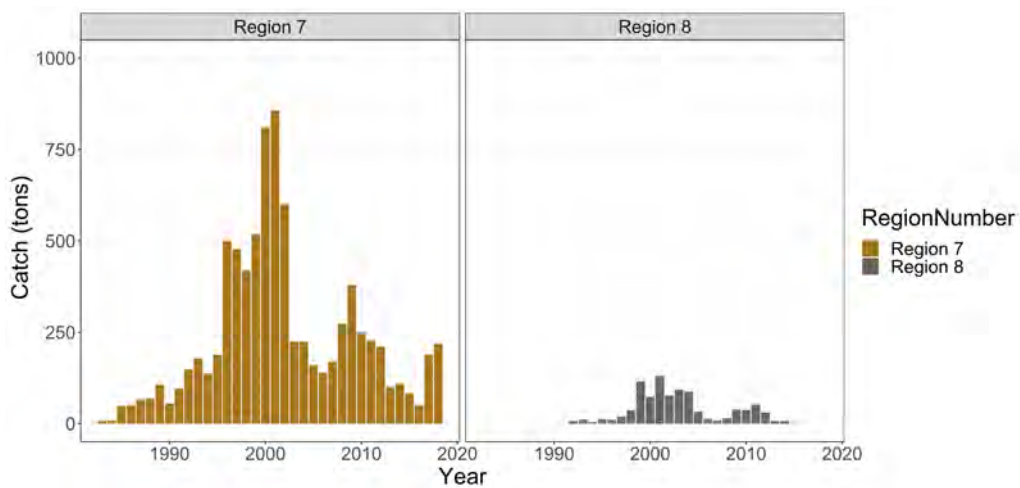


Figure A55. Annual catch (tons) per model region from (A) the Queensland GoC mudcrab fishery and (B) the Northern Territory GoC mud crab fishery.

Stepwise build-up of mud crab sub-model

Model fits overall

Effort alone (sub-model 1a) was able to explain mud crab catches for some but not all the regions in the GoC. For example, predicted catch followed the general trend in observed catch over the model period for Regions 1, 2 and 8 but not for Regions 3, 4, 5 and 7 (Figure A56). In Regions 4, 5 and 7 in particular, the model was not able to capture the sharp peaks in catches over the early 2000s and 2009-2013 periods (periods of good flow).

The addition of river flow as a driver of mud crab abundance and hence catch (sub-model 1b), improved some of the model fits e.g. for Regions 3, 4, and 5 in which the negative log likelihoods were significantly improved (Table A16, plots not shown) but not for other regions e.g. Regions 1, 2, 7 and 8 (Table A16).

Adding flow as a driver of recruitment and catchability again improved the model fits for Regions 3, 4 and 5 (Table A16) but not for Regions 1, 2, 7 and 8. Despite some of the negative log likelihoods not improving when flow was added (e.g. Region 7), the addition of the physical driver did boost catch over the 2000s period (not shown, but can be seen under sub-model 1d in Figure A57) – without the addition of flow, some of these larger catches couldn't be explained.

The addition of flow (linked to recruitment and catchability) as well as temperature (linked to mud crab mortality; sub-model 1d) resulted in a significant improvement of the negative log likelihood for Region 7 (Table A16) and a pattern in predicted catch that was much more similar to that of the observed catch, including some of the peaks (Figure A57). However, this model still didn't manage to capture the extreme peak in observed catch with predicted catch being slightly less that year. Nonetheless, this model was adequate in capturing the overall patterns in catch.

Table A16. Negative log likelihoods (-lnL) for each sub-model 1 version (built up in step-wise process) and each region.

-ln Likelihoods for MICE Regions	Sub-model 1a (no flow)	Sub-model 1b (flow linked to recruitment)	Sub-model 1c (flow linked to recruitment & catchability)	Sub-model 1d (flow linked to recruitment – Regions 1-8; flow linked to catchability – Regions 3-6); SOI linked to recruitment – Regions 3-6; air temperature linked to mortality in Region 7 only)
Region 1	86.5	92.8	92.1	92.8
Region 2	61.3	66.9	71.6	66.9
Region 3	90.8	58.9	54.8	43.1
Region 4	165.8	106.2	104.1	93.8
Region 5	168.5	99.5	94.8	94.5
Region 6	49.3	48.1	46.8	46.1
Region 7	52.6	23.5	16.9	-32.8

Region 8	63.2	57.7	58.0	58.1
Total -ln L	737.8	553.7	539.0	462.5

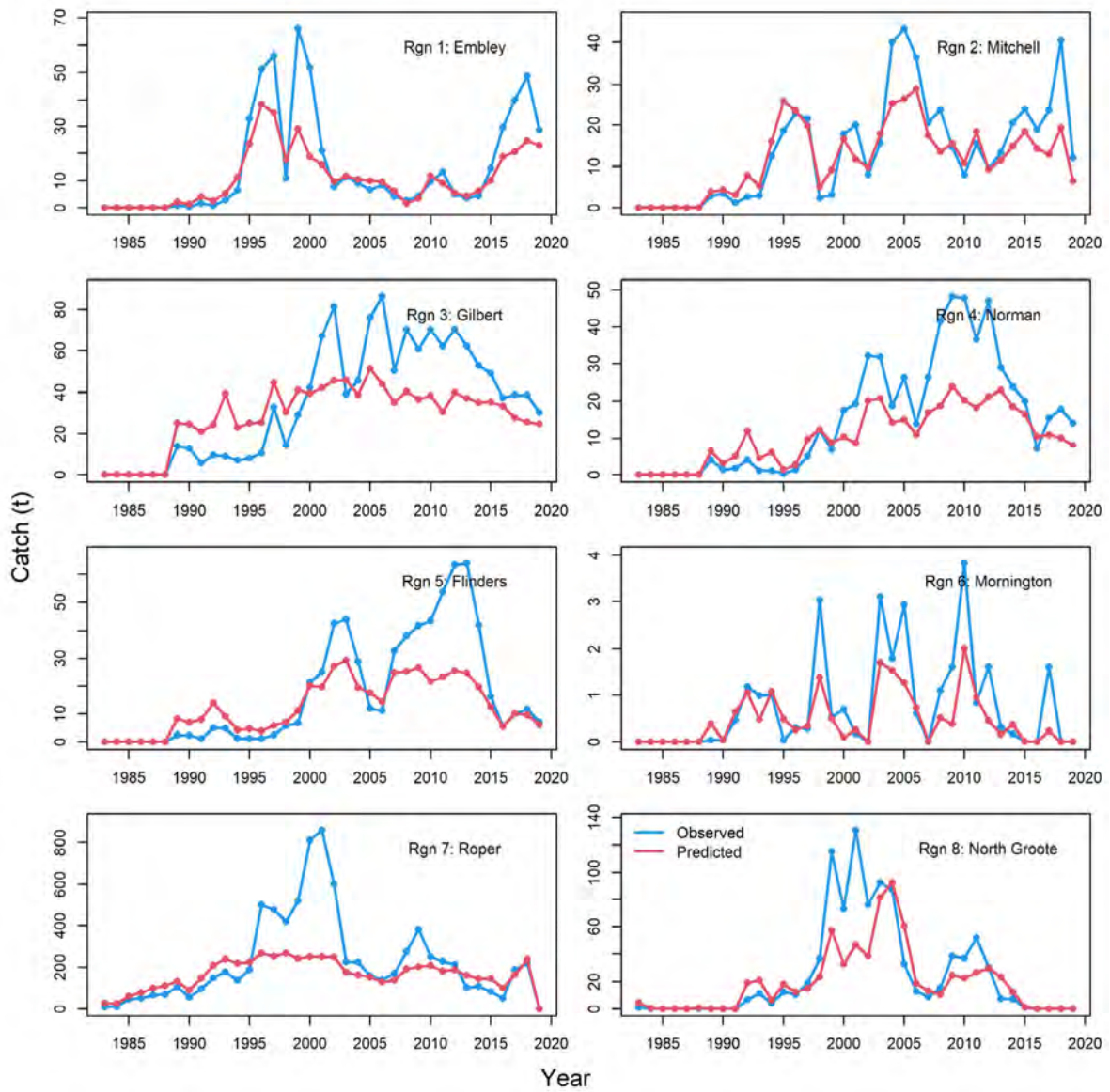


Figure A56. Observed vs predicted catch (tons) for each MICE region, when effort is the only driver of model predicted catch (Sub-model 1a, i.e. no physical variables included in the model).

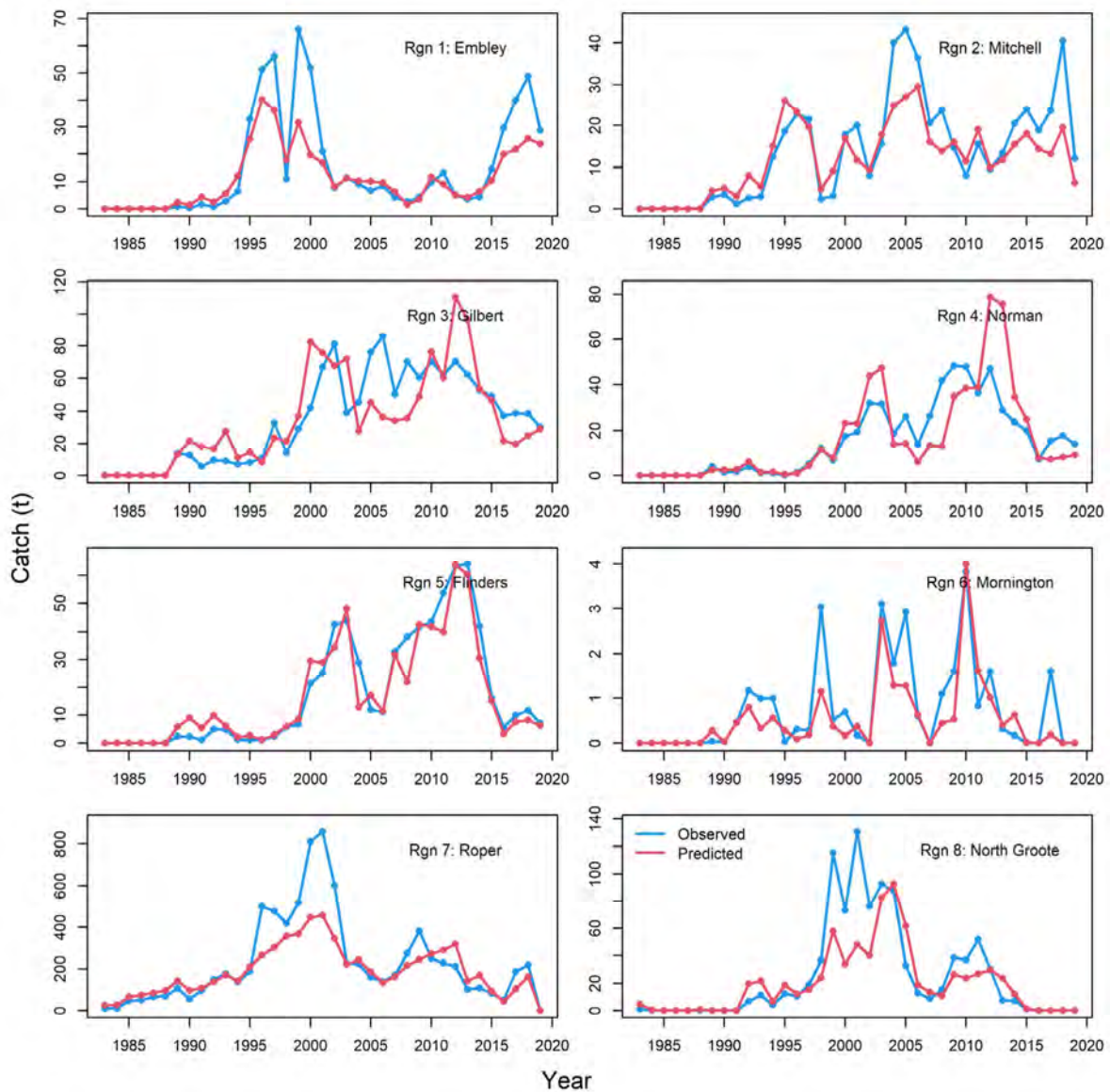


Figure A57. Observed vs predicted catch (tons) for each MICE region, when effort and river flow, as well as temperature (region 7 only) are included as drivers for model predicted catch (Sub-model 1d).

Modelling flow relationships

We trialled modelling of flow relationships with mud crab recruitment using a parabolic function and logistic function. The parabolic function provided a significantly better fit to the data for all regions except region 8 (Table A17). Additionally, when using a logistic function to link flow to mud crab recruitment, the model could only capture trends in the catch (index of abundance) for model regions 1-6 by estimating a very strong logistic relationship, which we felt was perhaps too extreme (Figure A58), compared to a more gradual change in recruitment under various flows as when using a parabolic relationship (Figure A58). Hence, for these reasons we used the parabolic relationship to capture the relationship between flow and mud crab abundance.

Table A17. Negative ln likelihoods for mud crab sub-model when using a parabolic vs logistic flow relationship

-ln Likelihoods for MICE Regions	Sub-model 1d with Parabolic flow relationship	Sub-model 1d with Logistic flow relationship
Region 1	92.8	138.1
Region 2	66.9	91.4
Region 3	43.1	82.1
Region 4	93.8	143.0
Region 5	94.5	142.0
Region 6	46.1	52.0
Region 7	-32.8	-25.1
Region 8	58.1	57.1
Total -ln L	462.5	680.5

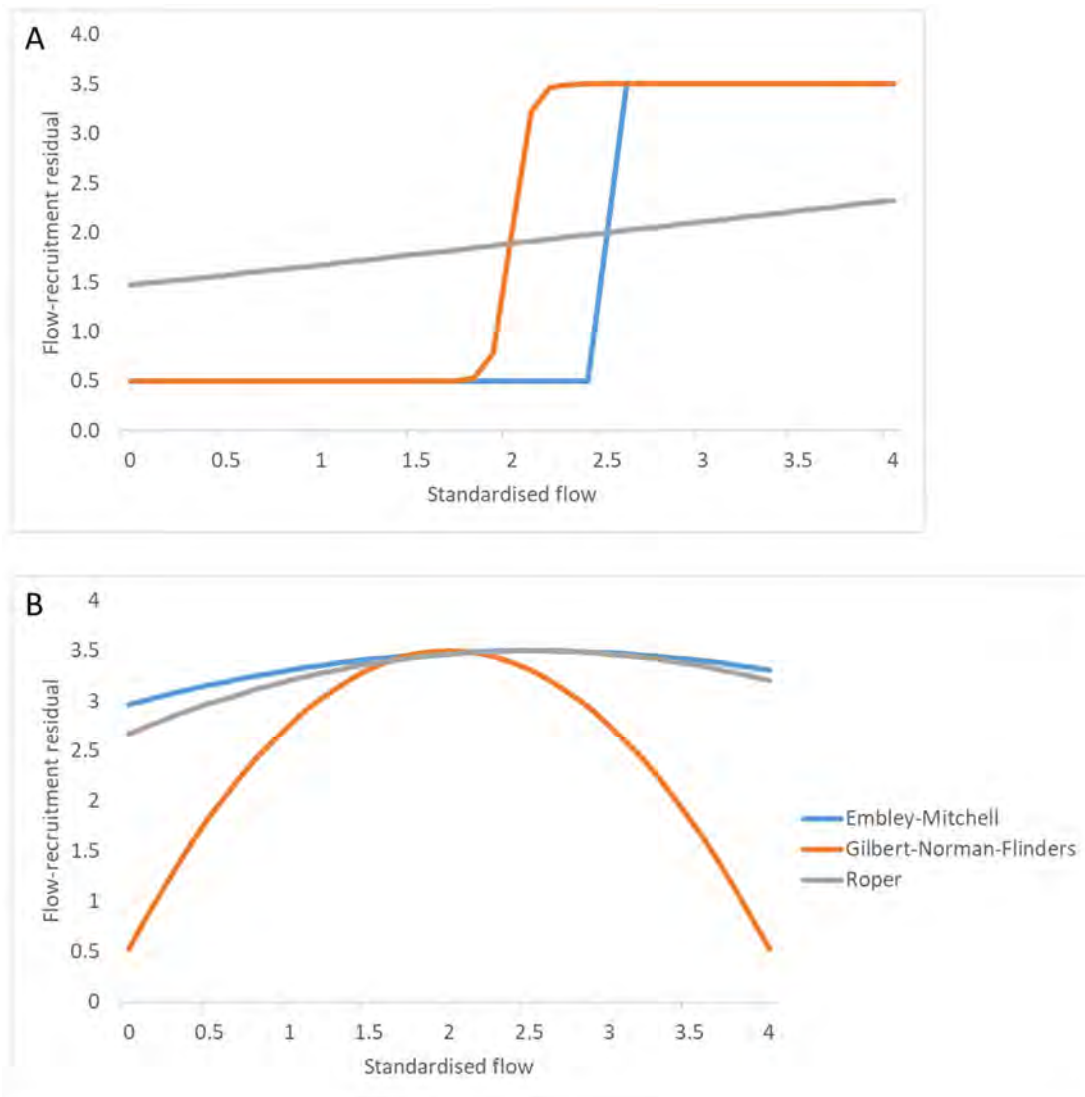


Figure A58. Model-estimated flow-recruitment relationships for some of the key GoC rivers using (A) a logistic function and (B) a parabolic function.

Model fits from selected months

The mud crab sub-model was fitted to monthly catch data which resulted in fitting to over 2300 data points. As such, we only show annual model fits and in some cases, fits might not appear very good in a particular year. This could be a case of the model fitting really well in some months and poorly in other months and so we have included here the monthly fits for some key regions e.g. Gilbert, Flinders and Roper rivers (Figure A59-Figure A61). For the Gilbert (Region 3), the model managed to capture the general trend in mud crab catches, but in some months e.g. January and February, failed to capture some of the peaks (Figure A59). For the Flinders (Region 5), the model captured the variability in catches for most months, particularly over the March-October period (Figure A60). For the Roper (Region 7) the model failed to predict large catches in the 2000s which is amplified on the annual scale (see Results Section 17 mud crabs in main report). However, if one considers the monthly fits to the data, for some months in the 2000s these peaks are captured better than in other months (Figure A61).

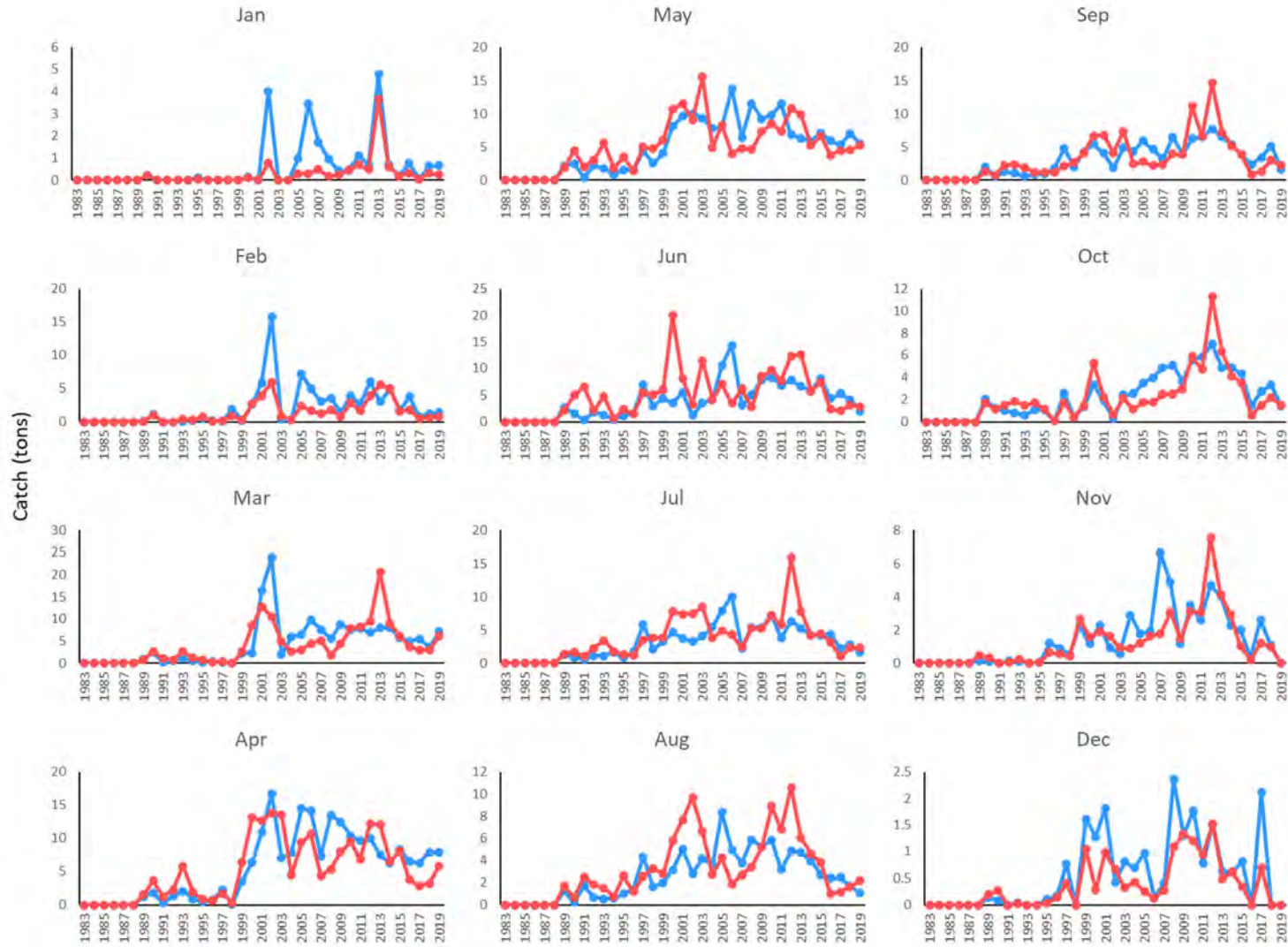


Figure A59. Observed (blue) vs predicted (red) catch (tons) for January-December over the period 1983-2019 for MICE region 3 (Gilbert). Note there were no catches prior to 1989 for this region and the model only fits to catch data from 1989.

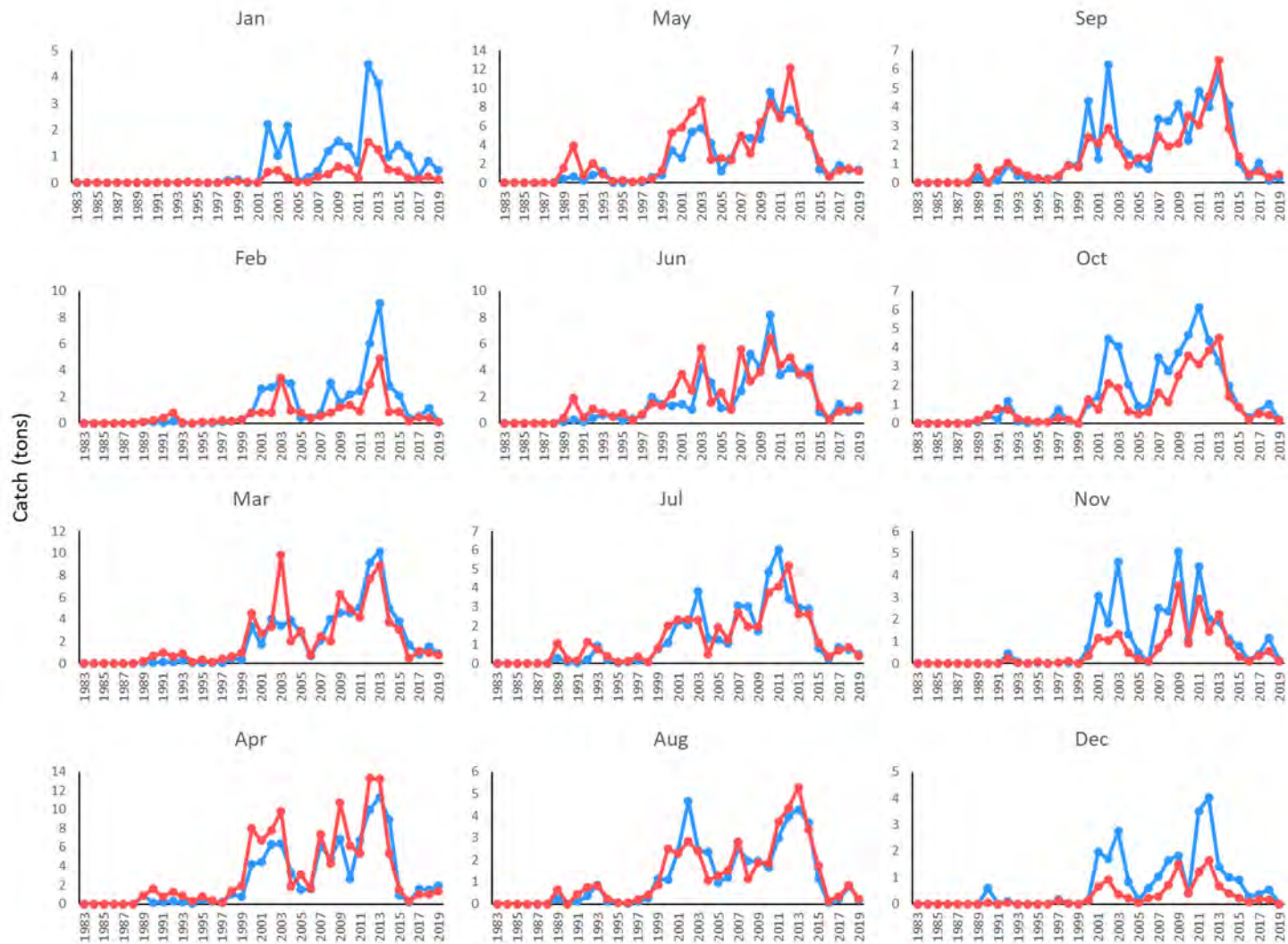


Figure A60. Observed (blue) vs predicted (red) catch (tons) for January-December over the period 1983-2019 for MICE region 5 (Flinders). Note there were no catches prior to 1989 for this region and the model only fits to catch data from 1989.

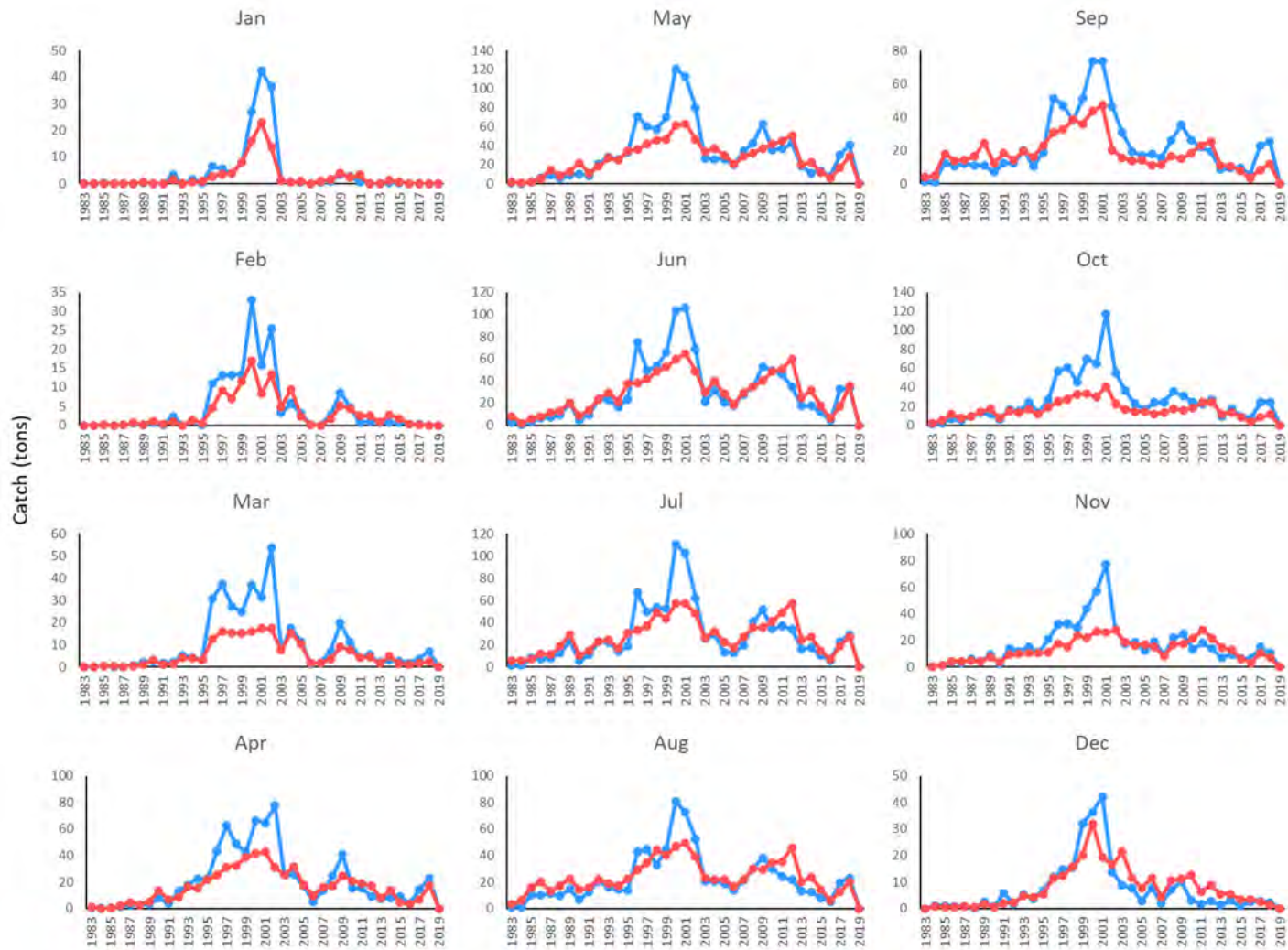


Figure A61. Observed (blue) vs predicted (red) catch (tons) for January-December over the period 1983-2019 for MICE region 7 (Roper). Note there were no catch data available for 2019 in this region and hence the model does not fit to 2019.

Model parameter estimates

In the mud crab sub-model, fourteen parameters were estimated – a model start biomass $B_{r,1970,1}^{spn,mudcr}$ for each of the eight regions, one flow parameters $thres^q$ related to catchability (Regions 3-6); three flow parameters $thres^{mudcr}$ related to recruitment (one for Regions 1-2; one for Regions 3-6; one for Region 7-8), one SOI parameter related to recruitment τ^{SOI} and a temperature parameter τ^{temp} related to additional mortality (Region 7 only). Parameters were fairly well estimated, as shown by 90% confidence intervals (Table A18). Starting biomasses for the model regions in the Queensland GoC ranged from around 4 tons to 30 tons, while in the Northern Territory GoC, mud crab biomass in the start year was much larger at approximately 70 tons in Region 7 and just under 40 tons in Region 8 (Table A18).

Estimated parameters and likelihoods for mud crab sub-model for all five MICE models are shown in the main report in Table 10.

Table A18. Model 1d estimated parameter values and associated Hessian-based 90% confidence intervals (CI), number of parameters estimated and the negative log likelihood (-lnL) for each model region.

Parameter Name	Parameter	Estimate	90% CI
Female crab spawning biomass (t) for each region in start year (1970) month 1	$B_{1,1970,1}^{spn,mudcr}$	4.4 t	4.12 – 4.81 t
	$B_{2,1970,1}^{spn,mudcr}$	3.9 t	3.63 – 4.17 t
	$B_{3,1970,1}^{spn,mudcr}$	27.1 t	23.44 – 31.34 t
	$B_{4,1970,1}^{spn,mudcr}$	28.2 t	21.68 – 36.78 t
	$B_{5,1970,1}^{spn,mudcr}$	14.5 t	12.42 – 16.95 t
	$B_{6,1970,1}^{spn,mudcr}$	7.4 t	5.79 – 9.46 t
	$B_{7,1970,1}^{spn,mudcr}$	69.1 t	58.78 – 81.14 t
	$B_{8,1970,1}^{spn,mudcr}$	37.8 t	33.85 – 42.22 t
Catchability parameter due to flow	$thres^q$	2.36	1.19 – 3.53
Recruitment parameter due to flow (regions)	$thres^{mudcr}$ (1-2)	-0.086	-0.087 – -0.086
	$thres^{mudcr}$ (3-6)	-0.74	-0.78 – -0.70
	$thres^{mudcr}$ (7-8)	-0.13	-0.21 – -0.05

Recruitment parameter due to SOI	τ^{SOI}	0.69	0.51 – 0.88
Mortality parameter due to air temperature	τ^{temp}	3.46	2.87 – 4.06
Number of parameters estimated	14		
-ln Likelihood per region			
Model region 1	92.8		
Model region 2	66.9		
Model region 3	43.1		
Model region 4	93.8		
Model region 5	94.5		
Model region 6	46.1		
Model region 7	-32.8		
Model region 8	58.1		

Model predicted biomass

In the mud crab sub-model, biomass was driven by a combination of fishing effort and river flow, as well as SOI (Regions 3-6) and air temperature (Region 7 only). Below we provide an overview of model-predicted mud crab biomass in some of the key mud crab fishing regions.

In Region 2, modelled mud crab commercial (available) biomass didn't appear to show much response to flow (Figure A62). This is perhaps not surprising given the best model appeared to be the one in which effort explained the catch.

In Region 3, lower river flows in the 1980s corresponded with a decline in available biomass and low river flows in the first half of the 1990s similarly corresponded with a low biomass (Figure A62). An increase in flow in late 1990s/early 2000s matched up with an increase in mud crab biomass. Good flow years from 2006-2012 corresponded with increasing biomass, which declined sharply from 2013 when there was very low flow. Biomass then picked up slightly following good flow in 2017.

Very high flows didn't always correspond to an increased mud crab abundance in Region 4 – in fact too much flow has been suggested to reduce abundance and/or catch (Robins et al. 2005) and this is something that we accounted for in the relationship with flow, which is likely what we see in Region 4. Low flows do correspond to low biomass though, which was shown to pick up after a few good flow years (Figure A62).

In Region 5, low flow years corresponded to dips in mud crab biomass but good flow years didn't correspond with sharp increases, although they did correspond with a turn around and some increase.

Very low flow years in 2014-2015 had a corresponding decline in biomass, which then picked up after a good flow year in 2016 (Figure A62).

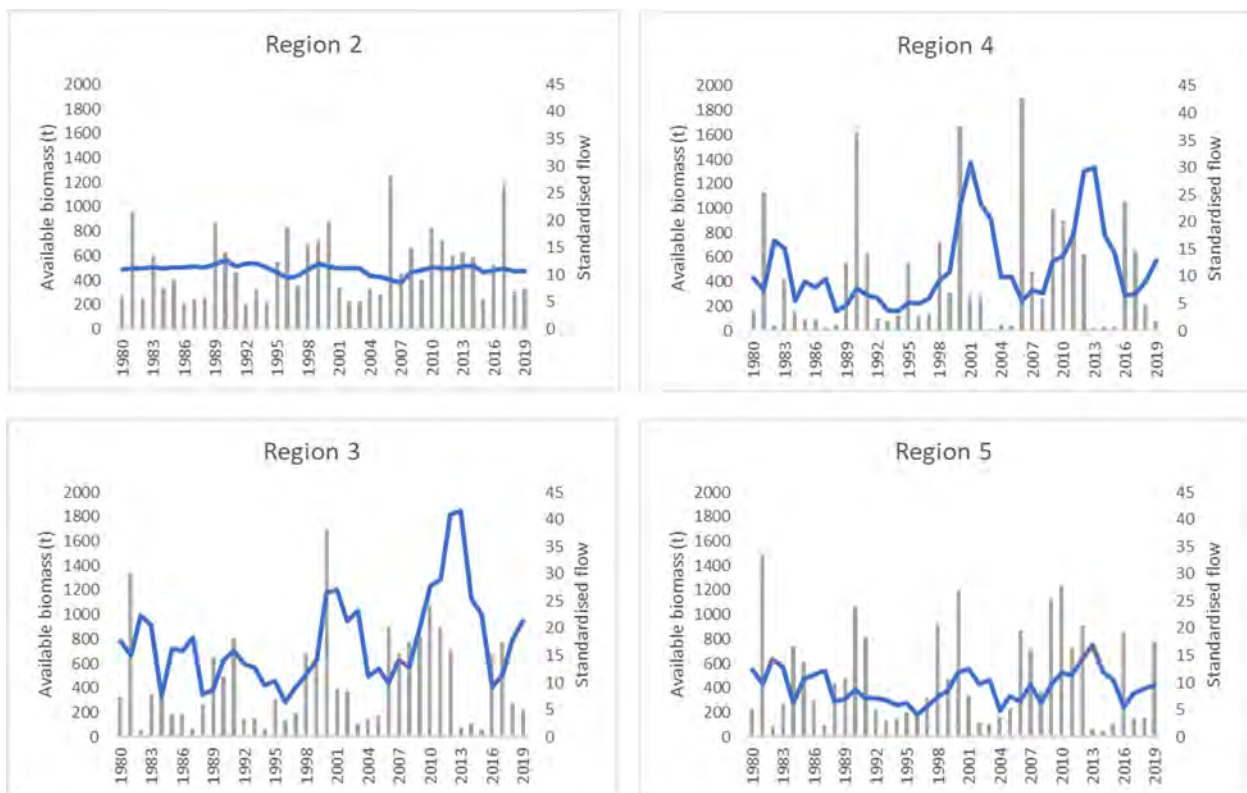


Figure A62. Annual modelled commercial biomass (tons; blue line) of mud crab (i.e. biomass available to be caught by the fishery) and annual standardised river flow (grey bars) for selected regions in the Queensland GoC.

In Region 7, years with lower flow in the late 80s/early 1990s matched with a declining mud crab biomass. However, this could also be because the fishery was taking off and catches were increasing. Good flow years in late 1990s matched with a biomass increase and this then declines briefly every time there is a low flow year, and then picks up again when flows are better e.g. 2009/10. Low flow years in 2013-2014 correspond with a decline in biomass but biomass continued to decline even though flows were fairly good in 2016-2017. One explanation for this may be the increasing trend in mean summer air temperature, which corresponds with a decline in mud crab biomass – particularly from 2014, when summer temperatures were warm during 2015-2016 El Niño years and mud crab biomass appears to have dropped off sharply. This suggests temperature may have been a more dominant driver over this period, despite there being some fairly good flow (Figure A63). It is worth noting that sea level may have also played a role in a decline in crab biomass over this period, by preventing crab larvae from entering the estuary, reduced habitat available and/or increased mortality. This could be explored further in the future.

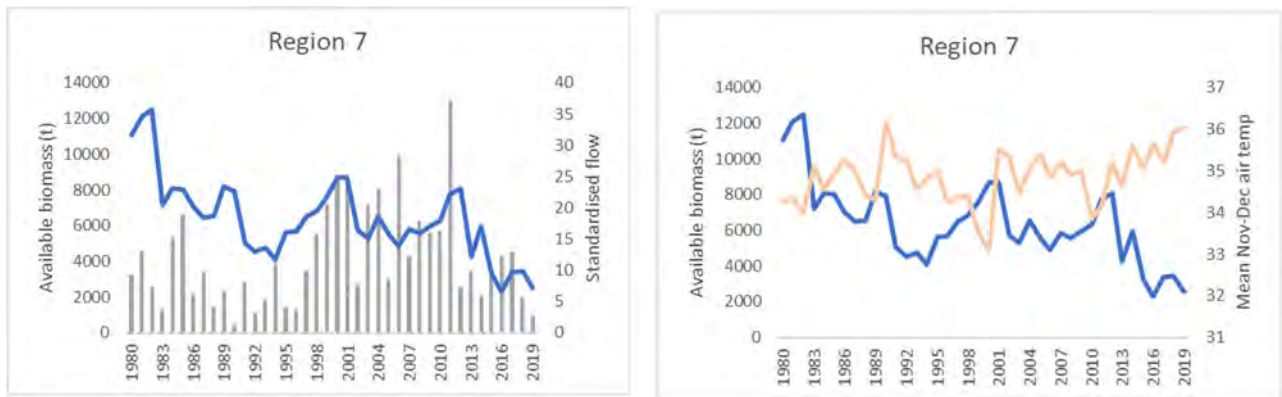


Figure A63. Annual modelled commercial biomass (tons; blue line) of mud crab (i.e. biomass available to be caught by the fishery) and annual standardised river flow (grey bars) for Region 7 in the Northern Territory GoC. Also shown is the annual commercial mud crab biomass (blue line) alongside mean Nov-Dec air temperature (orange line).

Discussion around model development

- There have been significant fluctuations in mud crab catch and catch rates in the GoC. Part of this can be explained by changes in fishing effort and management restrictions (Robins et al. 2020) but the importance of environmental variables in driving mud crab abundance and catch has also been recognised (Robins et al. 2005, Meynecke et al. 2012a, Meynecke et al. 2012b, Grubert et al. 2019, Robins et al. 2020).
- The most important physical variables likely driving mud crab dynamics are flow, temperature, and sea level. Recent work by Robins et al. (2020) have been shown that the relative importance and timing of these variables may vary across regions in the GoC.
- Our mud crab sub-model is the first spatial- and age-structured mud crab model for the GoC region, in which sexes are also modelled separately.
- We captured relationships between mud crabs and the environment using novel methods that account for the dynamic nature of the system.
- Our model structure and novel ways of incorporating environmental drivers allowed us to capture (1) differences between the Queensland and Northern Territory GoC fisheries, (2) real-time differences between fishing and physical/environmental drivers of mud crab dynamics and (3) regional differences in physical/environmental drivers.
- We found that fishing effort alone was able to explain trends in mud crab catches in some regions, but not all regions.
- River flow improved model predicted catches in some regions and was able to adequately capture the broad patterns in catch for these regions, including peak catches.
- Region 7 (Roper River) is a significant mud crab fishing region and was different to other regions with temperature likely an important factor in explaining mud crab abundance. This is consistent with recent work by Robins et al 2020. However, the model failed to capture some peak catches in the 2000s and there was greater uncertainty in the flow relationship used. There is scope to refine the model for Region 7 (Roper), by possibly sub dividing this region in case the environmental variables at play differ somewhat across the larger region and thus are not fully captured in the model.

- Sea level is likely to also be an important predictor of mud crab abundance in Region 7 and this could be explored going forward to better understand its role relative to other physical variables, and particularly to understand when flow is likely to be important.
- It will be important to understand how low flow years and added temperature or sea level stress will impact mud crab biomass and catches, particularly under changing climates.

References

- Alberts-Hubatsch, H. 2015. Movement patterns and habitat use of the exploited swimming crab *Scylla serrata* (Forskål, 1775). Universität Bremen.
- Alberts-Hubatsch, H., S. Y. Lee, J.-O. Meynecke, K. Diele, I. Nordhaus, and M. Wolff. 2016. Life-history, movement, and habitat use of *Scylla serrata* (Decapoda, Portunidae): current knowledge and future challenges. *Hydrobiologia* **763**:5-21.
- Baylon, J. C. 2010. Effects of salinity and temperature on survival and development of larvae and juveniles of the mud crab, *Scylla serrata* (Crustacea: Decapoda: Portunidae). *Journal of the World Aquaculture Society* **41**:858-873.
- Beverton, R. J., and S. J. Holt. 1957. On the dynamics of exploited fish populations. *Fishery Investigations, Series II*, 19. London: Her Majesty's Stationary Office.
- Brown, I. 2010. Taking female mud crabs (*Scylla serrata*): assessments of risk and benefits., Department of Employment, Economic Development and Innovation, Southern Fisheries Centre, Brisbane
- Butcher, P., A. Boulton, and S. Smith. 2003. Mud crab (*Scylla serrata*: Portunidae) populations as indicators of the effectiveness of estuarine marine protected areas. *in* World Congress on Aquatic Protected Areas.
- Deng, R., T. Hutton, A. E. Punt, J. Upston, M. Miller, C. Moeseneder, and S. Pascoe. 2020. Status of the Northern Prawn Fishery Tiger Prawn Fishery at the end of 2019 with estimated TAEs for 2020 and 2021. CSIRO, Brisbane.
- Fielder, D., and M. P. Heasman. 1978. The mud crab. Queensland Museum.
- Froehlich, H. E., T. E. Essington, and P. S. McDonald. 2017. When does hypoxia affect management performance of a fishery? A management strategy evaluation of Dungeness crab (*Metacarcinus magister*) fisheries in Hood Canal, Washington, USA. *Canadian Journal of Fisheries and Aquatic Sciences* **74**:922-932.
- Grubert, M. A., C. J. Walters, R. C. Buckworth, and S. S. Penny. 2019. Simple Modeling to Inform Harvest Strategy Policy for a Data-Moderate Crab Fishery. *Marine and Coastal Fisheries* **11**:125-138.
- Hay, T., and C. Calogeras. 2001. NT mud crab fishery: summary of assessment information 1996–1999. Northern Territory Department of Primary Industry and Fisheries, *Fishery Report* **53**.
- Heasman, M., and D. Fielder. 1977. The management and exploitation of the Queensland mud crab fishery [*Scylla serrata*]. *Australian Fisheries*.
- Heasman, M., D. Fielder, and R. Shepherd. 1985. Mating and spawning in the mudcrab, *Scylla serrata* (Forskål)(Decapoda: Portunidae), in Moreton Bay, Queensland. *Marine and Freshwater Research* **36**:773-783.
- Helmke, S., N. Gribble, and S. Gould. 1998. Status of Guld of Carpentaria mud crab stocks: establishing a benchmark for Albatross Bay, Weipa. 22, Queensland Department of Primary Industries, Brisbane.
- Hill, B. 1980. Effects of temperature on feeding and activity in the crab *Scylla serrata*. *Marine Biology* **59**:189-192.
- Hill, B. 1984. Aquaculture of the mudcrab. The Potential for Aquaculture in Queensland, Queensland Department of Primary Industries Publication:29-45.

- Hill, B. 1994. Offshore spawning by the portunid crab *Scylla serrata* (Crustacea: Decapoda). *Marine Biology* **120**:379-384.
- Hyland, S., B. Hill, and C. Lee. 1984. Movement within and between different habitats by the portunid crab *Scylla serrata*. *Marine Biology* **80**:57-61.
- Knuckey, I. 1999. Mud crab (*Scylla serrata*) population dynamics in the Northern Territory, Australia and their relationship to the commercial fishery. Northern Territory University, Darwin.
- Leslie, P., and D. Davis. 1939. An attempt to determine the absolute number of rats on a given area. *The Journal of Animal Ecology*:94-113.
- Loneragan, N., and S. Bunn. 1999. River flows and estuarine ecosystems: implications for coastal fisheries from a review and a case study of the Logan River, southeast Queensland. *Australian Journal of Ecology* **24**:431-440.
- Lovett, R., M. O'Neill, and A. Garland. 2020. Stock assessment of Queensland east coast blue swimmer crab (*Portunus armatus*).
- Meynecke, J.-O., M. Grubert, J. M. Arthur, R. Boston, and S. Y. Lee. 2012a. The influence of the La Niña-El Niño cycle on giant mud crab (*Scylla serrata*) catches in Northern Australia. *Estuarine, Coastal and Shelf Science* **100**:93-101.
- Meynecke, J.-O., M. Grubert, and J. Gillson. 2012b. Giant mud crab (*Scylla serrata*) catches and climate drivers in Australia—a large scale comparison. *Marine and Freshwater Research* **63**:84-94.
- Meynecke, J.-O., and S. Y. Lee. 2011. Climate-coastal fisheries relationships and their spatial variation in Queensland, Australia. *Fisheries Research* **110**:365-376.
- Meynecke, J.-O., S. Y. Lee, N. C. Duke, and J. Warnken. 2006. Effect of rainfall as a component of climate change on estuarine fish production in Queensland, Australia. *Estuarine, Coastal and Shelf Science* **69**:491-504.
- Meynecke, J., S. Lee, M. Grubert, I. Brown, S. Montgomery, N. Gribble, D. Johnston, and J. Gillson. 2010. Evaluating the environmental drivers of mud crab (*Scylla serrata*) catches in Australia. Final Report 2002/012.
- Northrop, A., M. F. O'Neill, and J. Robins. 2019. Towards an Initial Quota for the Queensland Mud Crab Fishery.
- O'Neill, M. F., A. B. Campbell, I. W. Brown, and R. Johnstone. 2010. Using catch rate data for simple cost-effective quota setting in the Australian spanner crab (*Ranina ranina*) fishery. *ICES Journal of Marine Science* **67**:1538-1552.
- Pauly, D. 1980. On the interrelationships between natural mortality, growth parameters, and mean environmental temperature in 175 fish stocks. *ICES Journal of Marine Science* **39**:175-192.
- Rikhter, V., and V. Efanov. 1976. On One of the Approaches to Estimation of Natural mortality of Fish Populations. ICNAF Res. Doc. 79/VI/8.
- Robertson, W. 2011. Abundance, population structure and size at maturity of *Scylla serrata* (Forsk.) (Decapoda: Portunidae) in Eastern Cape estuaries, South Africa. *South African Journal of Zoology* **31**:177-185.
- Robins, J. B., I. A. Halliday, J. Staunton-Smith, D. G. Mayer, and M. J. Sellin. 2005. Freshwater-flow requirements of estuarine fisheries in tropical Australia: a review of the state of knowledge and application of a suggested approach. *Marine and Freshwater Research* **56**:343-360.
- Robins, J. B., A. R. Northrop, M. Grubert, and R. C. Buckworth. 2020. Understanding environmental and fisheries factors causing fluctuations in mud crab and blue swimmer crab fisheries in northern Australia to inform harvest strategies.
- Ruscoe, I. M., C. C. Shelley, and G. R. Williams. 2004. The combined effects of temperature and salinity on growth and survival of juvenile mud crabs (*Scylla serrata* Forskål). *Aquaculture* **238**:239-247.
- Sara, L. 2010. Study on the size structure and population parameters of mud crab *Scylla serrata* in Lawele Bay, Southeast Sulawesi, Indonesia. *Journal of Coastal Development* **13**:133-147.
- Wang, Y.-G., M. O'Neill, I. Brown, and J. McGilvray. 2012. Assessment of the utility of Queensland commercial mud crab logbook data.
- Williams, M., and B. Hill. 1982. Factors influencing pot catches and population estimates of the portunid crab *Scylla serrata*. *Marine Biology* **71**:187-192.

Appendix 12 – Barramundi Model sub-model

Summary of current state of research on the topic of recent understanding of the contribution of flow to population dynamics and ecosystem interactions – Julie Robins, Qld Dept. Agriculture and Fisheries

Barramundi – for reviews see Russell (2014), Meynecke et al., (2014), Lawson et al., (2014); Griffiths et al., (2014)

SUMMARY (modified from Robins & Ye 2007): Populations of barramundi are likely to be highly responsive to river flows associated with any given river system because: (i) limited migration between estuaries occurs, such that the estuarine population probably reflects local river flows; (ii) catadromy -migration upstream (by 0+ juveniles into freshwater habitats) and downstream (back to the estuary by adults) is facilitated by river flows; (iii) growth rates of juveniles and adults are significantly affected by river flows, probably through habitat access and/or food availability, which would impact upon juvenile survival rates and the subsequent abundance of adults; (iv) ‘year-class strength’ (which is an index of survival) of barramundi is significantly affected by river flows. However, relationships are unlikely to be simply linear, as there are flow thresholds and multi-year effects.

Supporting Information:

Multiple genetically distinguishable sub-populations in the GoC (Figure A64; Jerry et al. 2013; Loughnan et al. 2019).



Figure A64. Republished from Figure 6 (Jerry et al. 2013) genetically distinguishable subpopulations of barramundi based on analysis of 16 microsatellite loci; grey outline is the major freshwater fish bioregions.

- Genetic analysis concurs with the available tag-recapture information (Infofish 2014). Most GoC tagged-recaptured barramundi were within the same river catchment. A few had significant movement. i.e., Norman to Nicholson and Mitchell to Norman.
- Barramundi are relatively long lived –up to 22 years old in the GoC; up to 35 years old on the Queensland east coast (Fisheries Qld monitoring fishery monitoring data).
- Protandrous hermaphrodite – matures as male first at 2 to 5 yrs, changes sex to become mature females at 5 to 7 years, although there are some primary females (Moore 1979; Davis 1982; Schipp et al. 2007; Grey 1987; Streipert et al. 2019).

- Non-obligatory catadromy – spawn in salt water and may migrate to freshwater habitats. Eggs require high salinity (i.e., 20 to 30 ppt) to survive (Davis 1985). Spawning assumed to take place in lower estuary and coastal foreshores (supported by anecdotal reports from fishers). Barramundi can use numerous habitats, from marine to freshwater. The duration and locality (i.e., distance) of freshwater residency is variable between years and rivers (Pender and Griffin 1996; Milton et al. 2008; Halliday et al., 2012; Crook et al., 2017).
- Spawning occurs during spring and summer (November to March, peak in December; Davis 1985), with the timing and duration of spawning thought to depend on water temperature, lunar and tidal cycles; spawning activity peaks during new and full moon periods (Grey 1987).
- Growth is variable between rivers (Davis and Kirkwood 1984) and between years (given variable length-at-age over time – Fisheries Queensland age-length data, Streipert et al. 2019), and is most likely a consequence of individual genetics as well as habitat/environmental conditions.

Flow-specific information

- Catch has been related to flow (Halliday et al., 2012; Lawson et al., 2014; Bayliss et al., 2014) and incorporates immediate effects of flow allowing fish to move downstream to become available to the fishery as well as more ‘catchable’. Catch also incorporates multi-year impacts of flow on population biomass as the fishery is comprised of about 10 age-classes (i.e., on average ages 2 to 11 but does vary between years). Age-classes in the fishery is also a reflection of fishery regulations
 - (Qld: minimum legal size of 580 mm; maximum legal size 1200 mm) as well netting practices (e.g., mesh size and set location) and market forces (i.e., GOC barramundi usually landed as frozen fillets).
 - ‘year-class strength’ is used as an index of ‘relative recruitment’. In the absence of annual pre-recruit sampling, it is estimated retrospectively using the age-frequencies of harvested fish (Maceina 1997). Has been applied to Qld east coast (Staunton Smith et al. 2004); GoC (Halliday et al. 2012; Saunders 2014; Bayliss et al. 2014; Robins NESP unpublished).
 - Growth – either derived from tag-recapture studies (e.g., Robins et al., 2005, Russell et al. 2014) or otolith increment-width analysis (Roberts et al., 2019; Leahy & Robins NESP unpublished).
 - Flow/freshwater access thought to enhance growth rates, but is likely to be habitat/river specific -Russell et al. (2014), based on tag-recapture growth noted that growth in east coast wet tropics rivers was greater in estuarine and coastal areas than in fish inhabiting lower freshwater river habitats.
 - Movement – flow facilitates upstream movement, especially in GoC rivers, where connectivity is dependent on the wet season flows. Based on either tag-recapture (Infofish 2014), acoustic tracking (Crook et al. 2017) or inferred from otolith microchemistry (Crook et al. 2017; Robins & Sellin NESP unpublished)

Qld barramundi fishery:

- Limited entry commercial fishery (N3 symbol).
- Set gill net fishery, size limits and net restrictions.

- Commercial daily logbook (1989 to present) – catch and effort, recorded at 30 nm grids (0.5 degree).
- Quantitative stock assessments (Campbell et al., 2017; Streipert et al., 2019).
- Southern GoC stock (13°S to NT border) shows long term recovery from heavy fishing in the 1970's and regulation changes in 1980 and 1992.
- Tanimoto et al., 2007 fitted a quantitative population model to the Fitzroy catchment (Qld east coast, where flow quantitatively affected recruitment and growth).
- The complexity of the GoC (larger scale, multiple rivers, limited tag recapture data for growth analysis) has hampered efforts to explicitly include flow effects in the stock assessment. However, in both Campbell et al., 2017 and Streipert et al., 2019, recruitment deviations are estimated during the model fitting process, somewhat including flow effects in the results.

MICE representation of barramundi

The MICE represents barramundi using three different stocks as shown in Figure A65, based on differences in management and biological properties including genetics (Campbell et al. 2017).

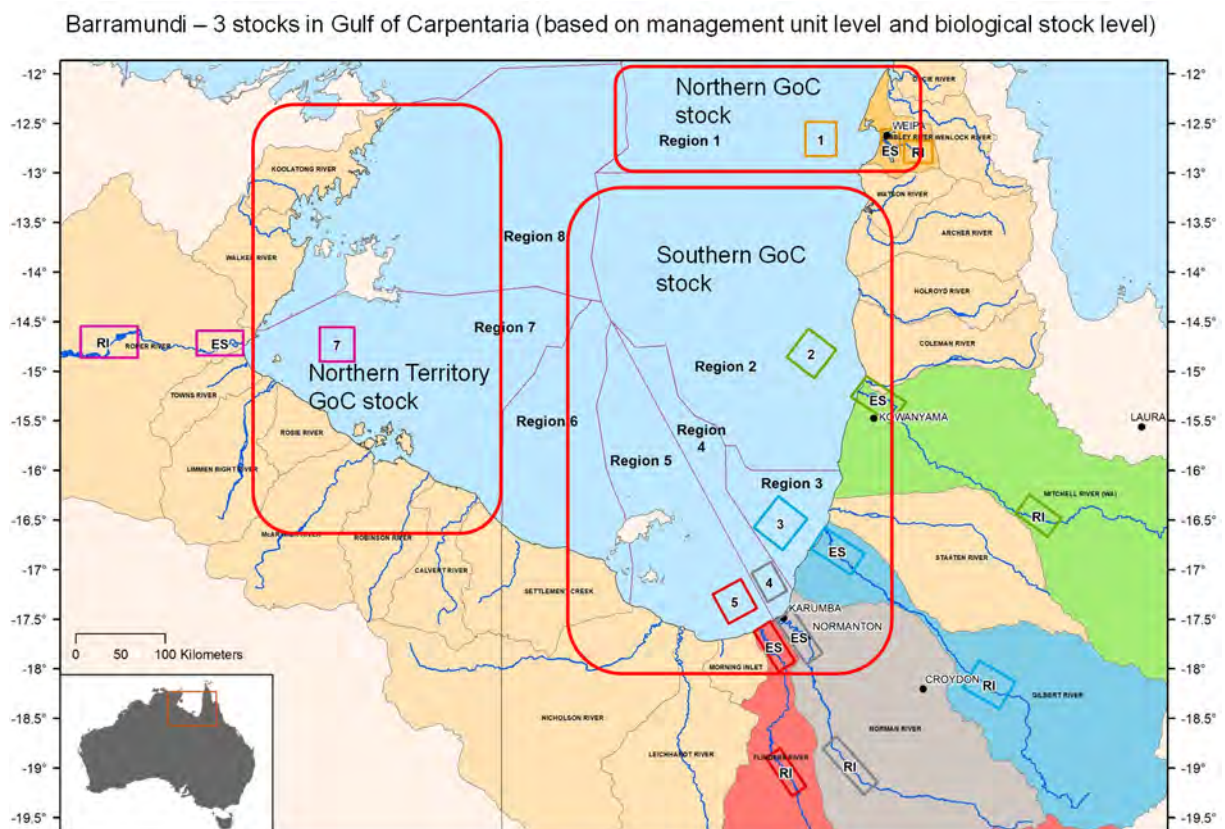


Figure A65. Summary of the three stocks considered in the MICE, namely the Northern GoC stock (region 1), southern GoC stock (regions 2-6) and the Northern Territory stock (regions 7-8).

There are a number of differences in management between the Qld and NT jurisdictions (Fig. 14). These include differences in minimum mesh net sizes and in minimum and maximum fish size limits, which are accounted for in the model. For the NT, the fishing area is restricted to waters seaward from the coast, river mouths and legislated closed lines (NT Department of Industry, Tourism and Trade, barramundi fishery management plan 1998) (see also <https://nt.gov.au/marine/commercial-fishing/fishery-licenses/barramundi-fishery-and-licences>). There are also minor differences in the commercial fishing seasons which are from 1 February to 30 September for NT and 1 February to 7 October for Qld. Given the model uses a monthly time step, the model accounts for the closed spawning season by assuming zero selectivity for the months October to January and 50% selectivity for February.

There have also been a number of changes in management over time (1981 endorsements; a mesh size limit in 1989, then a specified maximum legal size limit in 1992) (Campbell et al. 2017). To account for changes in size limits over time and differences between regions, we used different selectivity parameters as explained below.

The GoC Stock was seriously depleted following the high fishing effort in the 1970s and early 1980s (Campbell et al. 2017). Model catches start in 1989 and hence it is necessary to assume a starting level of stock depletion. We based our approach on that of Streipart et al. (2019) who assumed a depletion rate of 0.2 in 1989 for the Queensland Gulf stocks. These authors noted that this is a significant (but necessary) assumption. We assumed the barramundi stock was less depleted in model regions 1, 2, 7 and 8 than in the regions 3-6 in the Qld sector. We therefore set the 1989 depletion level as 50% of the starting 1970 spawning biomass level for these former regions and at 30% for regions 3-6. However, we also included a model version as part of the ensemble (Table 4) with depletion set at 20% instead for regions 3-6 to account for uncertainty in this important model assumption. Our choice of 30% as the base depletion rate was because a depletion level of 20% resulted in estimates of natural mortality M which were lower and less consistent with previous estimates of M for GoC barramundi (e.g. Campbell et al. 2017).

A comparison of the total barramundi catches taken from the NT (model regions 7-8) compared with Qld (model regions 1-6) is shown in Figure A66. Figure A67 shows examples for selected years over 2010 to 2019 of the total catch in each of the Qld model regions, highlighting that there is considerable variability in terms of the spatial distribution of catches: for example, region 2 is consistently a large catch region whereas the relative contributions from regions 4 to 6 is much more variable inter-annually.

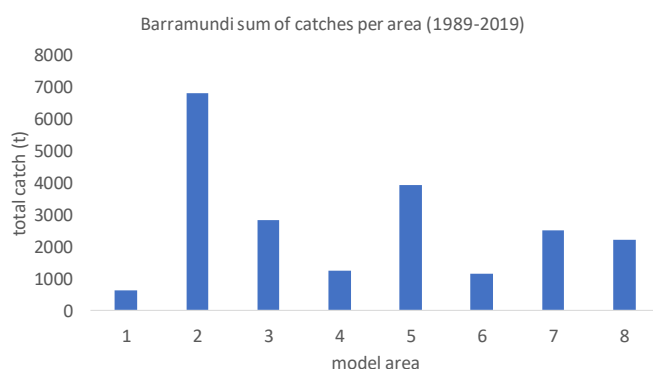


Figure A66. Sum of barramundi catches (t) over years 1989 to 2019, taken from each of the MICE regions as shown.

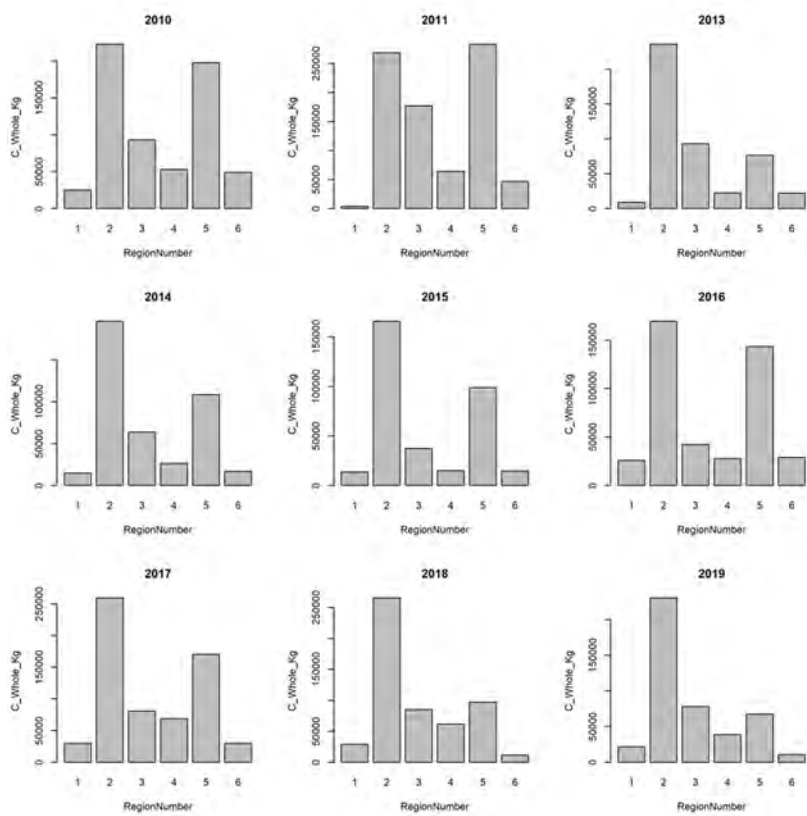


Figure A67. Barramundi catch (t) show for each of the Qld model regions for selected years from 2010 to 2019.

Age-structured equations are used to represent barramundi in each of the MICE spatial regions. It is assumed in the initial model version that there is no mixing between model regions. The monthly age-structured equations used to model the barramundi are similar to those used for mud crabs except that there are many more age classes included (20 ages represented, with the final age being a plus group capturing all animals 20 years and older) and instead of the separate representation of sexes as in the mud crab example, barramundi change sex with age as shown in Figure A68. As explained in the previous section, barramundi are protandrous hermaphrodites.

The length-at-age and mass-at-age relationships for barramundi are shown in Figure A69 and Figure A70 respectively. Figure A71 has superimposed the minimum and maximum Qld size limits. For Qld it is assumed that only fish 10 years and younger are caught, whereas no such limit is applied to the NT model regions.

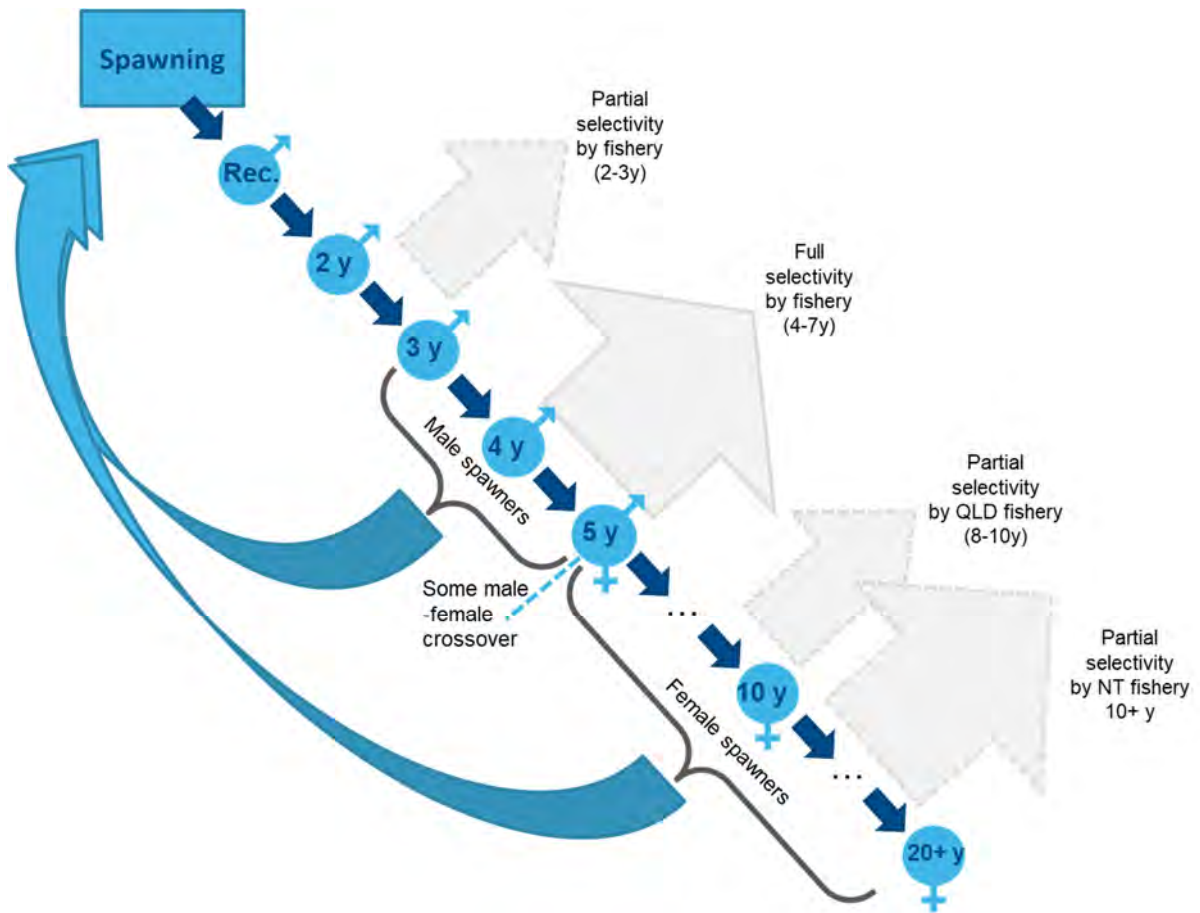


Figure A68. Schematic showing structure of barramundi model age classes, sex change and selectivity by the fishery.

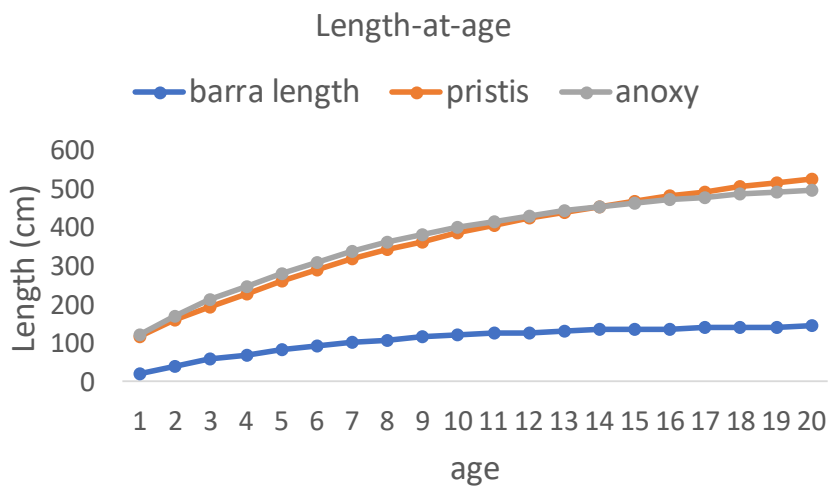


Figure A69. Length-at-age relationships used in the MICE for barramundi and the two sawfish species.

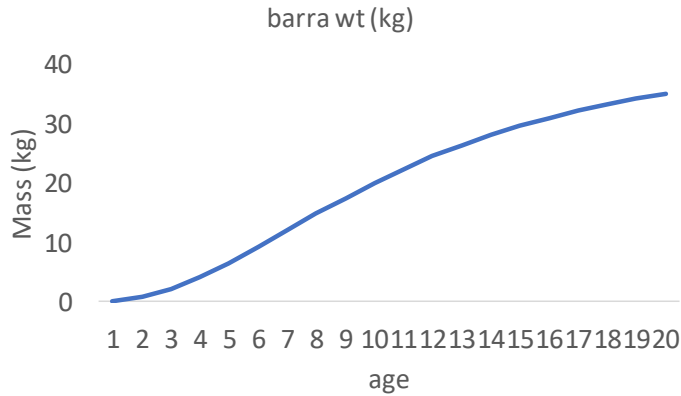


Figure A70. Mass at age used in the MICE for barramundi.

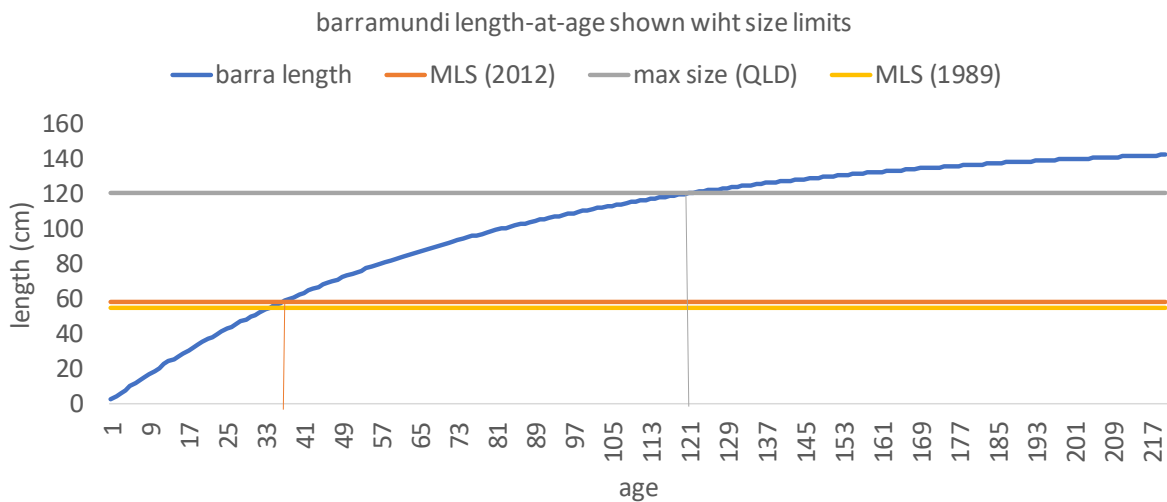


Figure A71. Schematic showing length-at-age for barramundi together with the Qld minimum and maximum size limits (indicated by vertical lines) which correspond respectively to 3 and 10 years.

A preliminary analyses was done to inform choice of an appropriate natural mortality estimate M for barramundi. This suggested that the default M of 0.2/yr (as used in Campbell et al. 2017) may be too low when considering the relative proportions of males versus females (Figure A72) in the population as well as the expected proportion of older, larger animals in the population (Figure A73). For example, if $M=0.2$, this means 43% of the stock biomass is >10 yrs versus if $M=0.3$ 24% is >10 y which was considered more plausible and hence the preliminary model assumed base $M=0.3$. However this was later updated to an age-dependent mortality relationship (see Appendix 16).

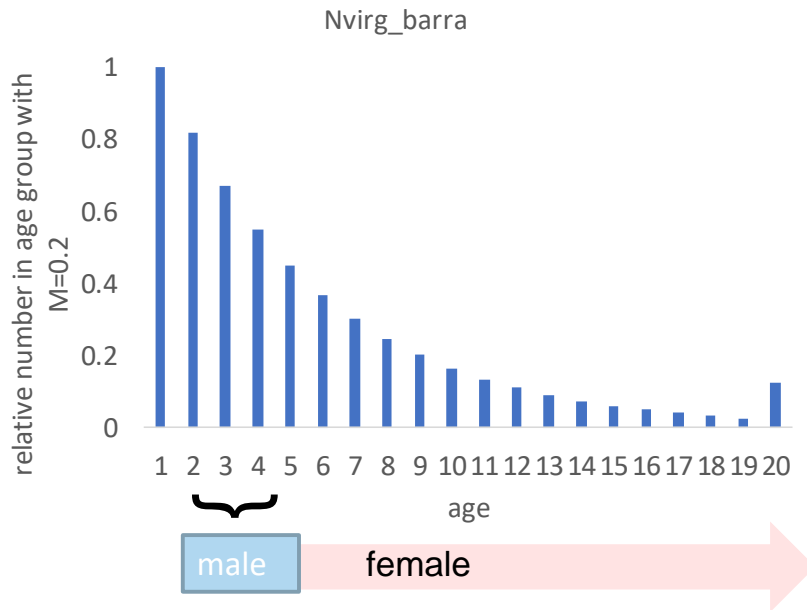


Figure A72. Schematic showing the virgin (pre-exploitation) relative numbers-at-age, split also into males and females, of barramundi f assuming natural mortality $M = 0.2$.

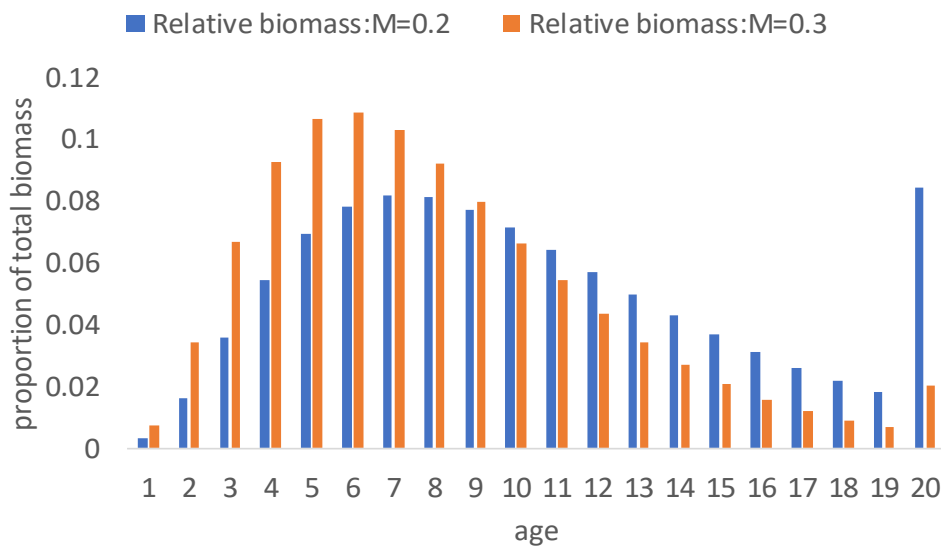


Figure A73. Schematic showing the virgin (pre-exploitation) relative total mass of each age class of barramundi when assuming natural mortality $M = 0.2$ compared with $M = 0.3$.

In both the Qld and NT, spawning is assumed to only occur during the months October to January. The proportion of each age class of males and females that is assumed sexually mature is as shown in Figure A74. Below follows a summary of the method used to represent spawning of barramundi so as to account for the protandrous behaviour of barramundi. Given a value of natural mortality M , it is possible to compute the equilibrium proportion of mature males in the population. Here we assume that a greater than equilibrium proportion of males has no effect on reproductive success, but that reproductive success declines exponentially if ratio is less than this (Figure A75). Sensitivity analyses could investigate the impact of alternative non-linear formulations.

Further details on the equations and settings used for barramundi are presented in Appendix 16.

barramundi maturity by sex at age as shown

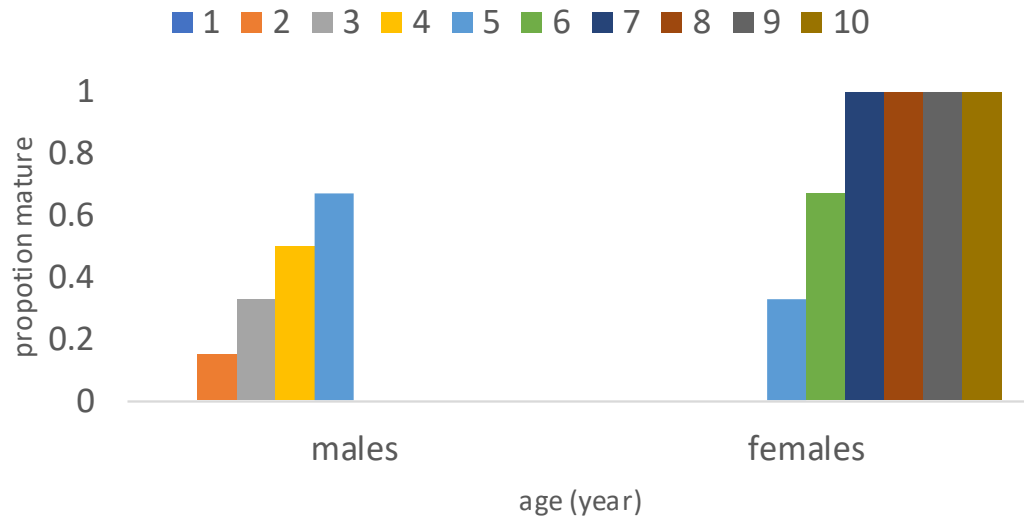


Figure A74. The proportion of males and females of ages as shown that are assumed to be sexually mature in the MICE.

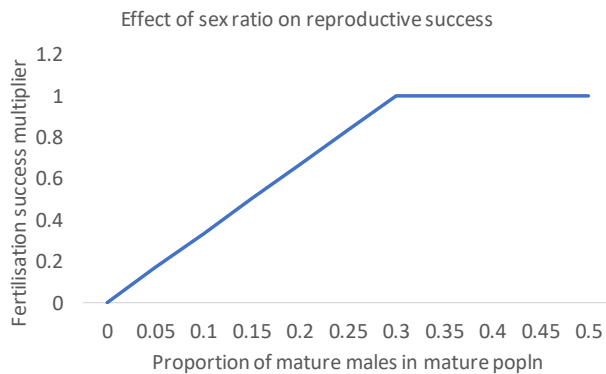


Figure A75. The MICE formulation used to represent potential decreases in barramundi fertilisation success once the proportion of males decreases below the equilibrium proportion. The example assumes $M=0.2$ for which the equilibrium proportion is 0.3, but alternative formulations were also investigated.

The model differentiates between the NT and Qld barramundi stocks as well as between the two genetic stocks that are split at around 13°S – so-called Northern GoC and Southern GoC stocks (Campbell et al. 2017). As detailed in Campbell et al. (2017), the Southern GoC stock are taken commercially as part of the Gulf of Carpentaria multispecies Inshore Fin Fish Fishery (GOCIFFF), which stretches roughly from the Cape York Peninsula region through to the Qld/NT border. It is part of an inshore (N3 symbol) (out to seven nautical miles) commercial net fishery that also targets species such as kind threadfin, as opposed to an offshore (N9 symbol) commercial net fishery that targets offshore species such as shark and grey mackerel. In this report we focus on commercial barramundi fishing, but note that there are also significant recreational and indigenous harvests that could be better accounted for in future work.

Campbell et al. (2017) used a three-parameter selectivity curve in their assessment of the southern GoC stock. Appendix 20 shows the results of using the Campbell et al. (2017) selectivity values directly as inputs to the MICE. In this study we use simpler approach to construct selectivity-at-age curves based on estimating relationships for Qld by fitting to data and capturing differences for the different regions by modifying the curve used for the NT regions. We use a logistic curve with two parameters to describe the ascending limb of the selectivity curve up until age 8 and then use a negative exponential function to describe the decline in selectivity at older ages:

$$S_{r,y,a}^{barra} = \frac{1}{1 + e^{-\frac{(a-a_{50})}{\delta_{r,y}}}} \quad a < 8$$

$$S_{r,y,a}^{barra} = e^{-\frac{(sfa_{r,y}) * (a-8)}{\delta_{r,y}}} \quad a \geq 8$$

Where $S_{r,y,a}^{barra}$ is the fishery selectivity of barra of age a , in region r and year y . In all instances we fix the age-at-50% (a_{50}) selectivity at 3 years. We used three different settings for $r=1-2$; 3-6; 7-8

determined by the two selectivity parameters $\delta_{r,y}$, $sfa_{r,y}$ which respectively represent the steepness of the ascending limb of the selectivity curve and the rate of decrease in selectivity at older ages.

Different parameter settings are used for each of the 3 regions. As we had catch-at-age data available for the southern GoC and northern GoC Qld stocks, we estimated each of the two parameters for each of these two regions (see Table 10 in main document). For the NT regions 7-8, we used a similar

shape but adjusted the $\delta_{r,y}$ value to reflect that younger animals may be caught and adjusted the $sfa_{r,y}$ value to reflect that animals older than age 10 are also caught (Appendix 16). For the regions, we used a different selectivity for $y < 1997$ (as per the approach of Campbell et al. 2017), with the parameters fixed so that the decline in selectivity at older ages was much less steep for this early period. An example (using Model 1) of the selectivity curves for the early period and the three barramundi groups is shown in Fig. 34. The model ensemble included different selectivity parameter estimates.

The influence of variable rainfall on barramundi catches has been well recognised for several decades (Williams 2002, Gribble *et al.* 2005, Halliday *et al.*, 2012; Lawson *et al.*, 2014; Bayliss *et al.*, 2014). Seasonal flows and flooding influences the relative recruitment of young-of-the-year barramundi (Staunton-Smith *et al.* 2004, Halliday *et al.* 2012), catchability (Campbell *et al.* 2017) as well as growth rates (Robins *et al.* 2006). The first two aspects are included in the MICE structure, but linking the effect of flows on growth did not produce satisfactory results so is not included in these models. Instead, changes in flow are assumed to affect the survival of young barramundi. As presented in Appendix 16, the average standardised monthly end-of-system flow multiplier value (based on a logistic relationship) is applied to the recruitment function in place of estimating stock-recruit residuals. The mortality rate of barramundi aged 5 or younger is assumed to be inversely proportional to the standardised monthly end-of-system flow multiplier value (with this constrained to fall between 0.1 and 2.0). In addition, flow is assumed to influence catchability of barramundi in all regions except model regions 1 and 2, because model fits were worse when this assumption was applied to regions 1 and 2.

For full model equations and specifications, please see Appendix 16.

References

Bayliss, P., Robins, J., Buckworth, R., Dichmont, C., Deng, R., and Kenyon, R. (2014). Part 3 Quantitative risk assessment for banana prawns and barramundi. In: Assessing the water needs of

- fisheries and ecological values in the Gulf of Carpentaria. Final Report prepared for the Qld DNRM. Eds: Bayliss, P., Buckworth, R., and Dichmont, C. CSIRO, Australia. Pp 203-254.
- Campbell, A., Robins, J., and O'Neill, M. (2017). Assessment of the barramundi (*Lates calcarifer*) fishery in the Southern Gulf of Carpentaria, Queensland, Australia. Qld DAF. Brisbane. 44 pp.
- Crook, D.A., Buckle, D.J., Allsop, Q., Baldwin, W., Saunders, T.M., Kyne, P.M., Woodhead, J.D., Mass, R., Roberts, B., and Douglas, M.M. (2017). Use of otolith chemistry and acoustic telemetry to elucidate migratory contingents in barramundi *Lates calcarifer*. MFR, 68, 1554-1566.
- Davis, T.L.O. (1982). Maturity and sexuality in barramundi, *Lates calcarifer* (Bloch), in the Northern Territory and South-eastern Gulf of Carpentaria. AJMFR 33, 529-45.
- Davis, T.L.O. (1985). Seasonal changes in gonad maturity and abundance of larvae and early juveniles of Barramundi, *Lates calcarifer* (Bloch), in Van Dieman Gulf and the Gulf of Carpentaria. AJMFR 36, 177-190.
- Davis, T.L.O. and Kirkwood, G.P. (1984). Age and growth studies on Barramundi *Lates calcarifer* (Bloch), in northern Australia. AJMFR 35, 673-389.
- Grey, D. (1987). An overview of *Lates calcarifer* in Australia and Asia. In: Management of Wild and Cultured Sea Bass/Barramundi. Proceedings of an International Workshop held in Darwin, N.T. Australia, 24-30 September 1986. J.W. Copeland and D.L. Grey (eds). Pp 15-20. ACIAR, Canberra.
- Gribble, N., Welch, D., Garrett, R., Higgs, J. and Whybird, O. (2005). Draft 1981 - 2004 assessment of the barramundi fishery in Queensland. Department of Primary Industries and Fisheries, Queensland, Cairns. 62 pp
- Griffiths, S., Kenyon, R., Poloczanska, E., Deng, R., Milton, D., Rothlisberg, P., Burford, M., Coman, F., and Miller, M. 2014. Qualitative risk assessment of the potential effects of altered river flows in the Flinders and Gilbert catchments. In: Assessing the water needs of fisheries and ecological values in the Gulf of Carpentaria. Final Report prepared for the Qld DNRM. Eds: Bayliss, P., Buckworth, R., and Dichmont, C. CSIRO, Australia. pp 25-202.
- Halliday, I., Saunders, T., Sellin, M.J., Allsop, Q.A., Robins, J.B., McLennan, M. and Kurnoth, P. (2012). Flow impacts on estuarine finfish fisheries of the Gulf of Carpentaria. Final Report for FRDC Project No. 2007/002. Qld DAFF, Brisbane. 76 pp.
- Infofish (2014). Tagging in the Gulf of Carpentaria Gilbert River to Flinders River 1985-2014. Infofish Australia. 11 pps.
- Jerry, D.R., Smith-Keune, C., Hodgson, L., Pirozzi, I., Carton, A.G., Hutson, K.S., Braxenor, A.K., Gonzalez, A.T., Gamble, S., Collins, G., Van Der Wal, J. (2013). Variability of an iconic Australian finfish (barramundi – *Lates calcarifer*) and aligned industries to climate change across tropical Australia. Final Report to the FRDC for Project No 2010/521. James Cook University, Townsville. 222 pps.
- Lawson, E., Saunders, T. and Robins, J. (2014) Barramundi. In: Welch, D.J., Robins, J., and Saunders, T., (eds). Implications of climate change on fisheries resources of northern Australia. Part 2: Species profiles. Final Report to the FRDC for Project number 2010/565. James Cook University. pp, 97-111.
- Loughnan, S., Smith-Keune, C., Beheregaray, L.B., Robinson, N.A., and Jerry, D.R. (2019). Population genetic structure of barramundi (*Lates calcarifer*) across the natural distribution range in Australia informs fishery management and aquaculture practices. *Marine and Freshwater Research*, 70(11), pp.1533-1542

- Maceina, M.J. (1997). Simple application of using residuals from catch curve regressions to assess year-class strength in fish. *Fisheries Research*, 32, 115-121.
- Meynecke, J.O., Robins, J., and Balston, J. (2014). Climate effects on the recruitment and catch effort of *Lates calcarifer*. In: *Biology and Culture of Asian Seabass*. Ed D. Jerry. CRC Press. Pp 31-66
- Milton, D., Halliday, I., Sellin, M., Marsh, R., Staunton-Smith, J., and Woodhead, J. (2008). The effect of habitat and environmental history on otolith chemistry of barramundi *Lates calcarifer* in estuarine populations of a regulated tropical river. *Estuarine, Coastal and Shelf Science* 78, 301-315.
- Moore, R. (1979). Natural sex inversion in the giant perch (*Lates calcarifer*). *AFMFR* 30, 808-813.
- Pender, P.J., and Griffin, R.K. (1996). Habitat history of barramundi *Lates calcarifer* in a north Australian river system based on barium and strontium levels in scales. *Transactions of the American Fisheries Society*, 125, 679-689.
- Roberts, B.H., Morrongiello, J.R., King, A.J., Morgan, D.L., Saunders, T.M., Woodhead, J., and Crook, D.A. (2019). Migration to freshwater increase growth rates in a facultatively catadromous tropical fish. *Oecologia* 191(2): 253-260.
- Robins, J., Mayer, D.G., Staunton-Smith, J., Halliday, I., Sawynok, B. and Sellin, M.J. (2006). Variable growth rates of the tropical estuarine fish barramundi *Lates calcarifer* (Bloch) under different freshwater flow conditions. *J. Fish Biol.* 69, 379-391.
- Robins, J. and Ye, Q. (2007). Relationships between freshwater flow and fisheries (or secondary) production in Australian estuaries: a review. Literature review for e-Water CRC.
- Russell, D.J. (2014). *Lates calcarifer* wildstocks: their biology, ecology and fishery. In: *Biology and Culture of Asian Seabass*. Ed D. Jerry. CRC Press. Pp 77-101.
- Russell, D.J., Thomson, F.E., Thuesen, P.A., Power, T.N., and Mayer, R.J. (2015). Variability in the growth, feeding and condition of barramundi (*Lates calcarifer* Bloch) in a northern Australian coastal river and impoundment. *MFR*, 66, 928-941.
- Saunders, T. (2014) Barramundi. In: Welch, D.J., Saunders, T., Robins, J., Harry, A., Johnson, J., Maynard, J., Saunders, R., Pecl, G., Sawynok, B., and Tobin, A. (2014). Implications of climate change on fisheries resources of northern Australia. Part 1: Vulnerability assessment and adaptation options. James Cook University. Final Report to FRDC for Project No 2010/565. Pp37-36, 96-100.
- Schipp, G., Bosmans, J., Humphrey, J. (2007). Northern Territory barramundi farming handbook. NT Fisheries, Darwin. 80 pps.
- Staunton-Smith, J., Robins, J.B., Mayer, D.G., Sellin, M.J. and Halliday, I. (2004). Does the quantity and timing of fresh water flowing into a dry tropical estuary affect year-class strength of barramundi (*Lates calcarifer*)? *MFR* 55, 787-797.
- Streipert, S., Filar, J., Robins, J.B., O'Neill, M.F., and Whybird, O. (2019). Stock assessment of the barramundi (*Lates calcarifer*) fishery in Queensland, Australia. Qld DAF, Brisbane. 103 pps.
- Tanimoto, M., Robins, J.B., O'Neill, M.F., Halliday, I. and Campbell, A.B. (2012). Quantifying the effects of climate change and water abstraction on a population of barramundi (*Lates calcarifer*), a diadromous estuarine finfish. *MFR* 63, 715-726.
- Williams, L.E. (2002). *Queensland's Fisheries Resources. Current condition and recent trends 1988-2000*. Queensland Department of Primary Industries, Brisbane. 180 pp.

Appendix 13 – Sawfish overview

In Australia there are four species of sawfish (family Pristidae) that occur in tropical and sub-tropical areas of northern Australia (Stevens et al 2008; Last and Stevens, 2009; Morgan et al 2011). While abundance of sawfish in northern Australia has undoubtedly declined over the past 40 years, Australia has some of the last remaining viable populations of sawfish in the world. The largetooth sawfish (*Pristis pristis*) (formerly known as the freshwater sawfish), green (*P. zijsron*) and dwarf sawfish (*P. clavata*) are listed as “Vulnerable” under the Environment Protection and Biodiversity Conservation Act, 1999 (EPBC Act) (Table A19). The narrow sawfish (*Anoxypristis cuspidata*) is listed as Migratory because of its listing on Appendix I and II on the Convention on the Conservation of Migratory Species, and therefore has similar protection status under the EPBC Act.

Table A19. Sawfish listings under the EPBC legislation

<i>Pristis pristis</i>	Largetooth Sawfish	Vulnerable
<i>Pristis clavata</i>	Dwarf Sawfish	Vulnerable
<i>Pristis zijsron</i>	Green Sawfish	Vulnerable
<i>Anoxypristis cuspidata</i>	Narrow Sawfish	Listed migratory species

Sawfish are elasmobranchs and exhibit classical K-selected (Pianka, 1970) life histories (low fecundity, long-lived) resulting in low population productivity (Table A20). These characteristics combined with a toothed rostrum that is easily entangled in nets make them extremely vulnerable to overexploitation (Dulvy et al. 2014; 2016). Sawfish are amongst the most threatened marine taxa (Dulvy et al. 2016) due to their overlap with coastal fisheries and high susceptibility to capture in gillnet and trawl fisheries. Australia's National Recovery Plan for the largetooth sawfish has identified the following as principal threats to sawfish and river shark species: “fishing activities including: being caught as by-catch in the commercial and recreational sectors; through Indigenous fishing; and illegal, unreported and unregulated fishing; habitat degradation and modification”

Historically, sawfish species have been captured incidentally in significant numbers by inshore gillnet and offshore prawn trawl fisheries with increasing concern of their status and limited fishery dependent data on which to base assessments. Available data sources include that from a number of crew-member and scientific observer programs which are analysed as part of bycatch sustainability assessments (Fry et al. 2018). These programs include collections of catch data on ‘Threatened, Endangered and Protected’ species, sawfish species and ‘at risk’ elasmobranch, teleost and invertebrate bycatch species.

The narrow sawfish (*Anoxypristis cuspidata*) has the highest productivity of all four sawfish species (Table A20) and has been recorded in both inshore and offshore set net fisheries as well as prawn trawl fisheries. The largetooth sawfish (*Pristis pristis*) inhabits both freshwater, and estuarine and marine environments and is caught mainly by the inshore fisheries in the monsoonal wet season (February to April) when juveniles and sub-adults move downstream (Peverell 2004).

In addition, largetooth sawfish are threatened by alterations to the riverine and estuarine habitats that they rely on for parts of their life cycle. Two sawfish species with different habitat and life history characteristics were considered for inclusion in the MICE but in this report we focus only on the largetooth sawfish (*Pristis pristis*) due to its strong dependency of freshwater river flows. There was inadequate scientific information available to reliably parametrise how flows may influence the dynamics of narrow sawfish as they are not confined to estuaries/rivers for any part of their life and data on the distribution of juveniles and juveniles are relatively poorly understood with small juveniles captured in both shallow inshore areas as well as offshore in prawn trawls (Pillans pers comm). Narrow sawfish are thus not considered further in this report which focuses instead on the largetooth sawfish which depends critically on a combination of freshwater, estuarine and marine habitats. Hereafter, we therefore use “sawfish” to mean largetooth sawfish. Nursery areas (up rivers)

provide more food and less predation for smaller largemouth sawfish. Entrapment of sawfish in isolated stretches can result in increased predation. Both crocodile and bull sharks are known to prey on sawfish within rivers. There is some known avoidance of bull sharks through movement behaviour. Once the sawfish grow to around 3 metres they then move downstream. One possible extension to the MICE described in this report is to represent the greater vulnerability of younger largemouth sawfish to crocodiles by modelling predation mortality as a simple linearly increasing function of crocodile abundance. Alternative models could also explore adjusting crocodile predation mortality as a function of flow rates.

As explained in the main text, our model does not explicitly represent sawfish population dynamics in Model regions 1, 6 and 8 due to the lack of suitable rivers with extensive and accessible freshwater habitats known to or assumed to be important nursery areas.

Pillans et al. (in press) collated data on catches and catch rates of sawfish caught as bycatch by gillnet and trawl fisheries. The trawl fishery mainly catches adults and large sub-adults that have left the river/estuary hence we assume no juveniles are caught by the trawl fishery. The NPF trawl fishery estimates were based on largemouth CPUE from years of scientific observer data. In comparison, the Pillans et al. (in press) estimates of gillnet catch are based on old CPUE data that are not nearly as robust as the NPF data.

As a first step in the MICE, we use these estimates of catch rates by the different fishery sectors and multiply by monthly estimates of fishing effort per sector to obtain rough estimates of largemouth sawfish interactions per model region (subject to a number of caveats as outlined in Pillans et al. in press). There is limited information on selectivity of largemouth sawfishes by different fishery sectors, however potentially, they can be caught at all life-history stages, so we assume selectivity is equal for all ages. We also use as the MICE starting values the demographic parameter estimates collated by Pillans et al. (in press) (Table A21).

We acknowledge that the number of interactions (i.e. animals caught in fishing gear) doesn't always mean that all these animals die and hence that this straightforwardly translates into a mortality rate. However, there are currently no data on post-release survival of largemouth sawfish (or any other sawfish in Australia) despite this being a key parameter that is needed to accurately model the population dynamics of a threatened species. Before they were afforded protection in 2000 (EPBC Act, 1999), it is likely that few captured sawfish were released or survived entanglement in fisheries gear. Additional threats included illegal shark finning operations (Putt and Anderson 2007). Protocols for correct handling of entangled animals to enhance post-release survival have been published since 2010 (https://www.daf.qld.gov.au/data/assets/pdf_file/0005/49109/Sawfish-Guide-Final-Nov-2010.pdf), however uptake by industry has not been assessed. Australia's *Environment Protection and Biodiversity Conservation Act* (the EPBC Act) came into effect in 1999, such that sawfish became better protected from 2000 and hence it is likely that post-release survival rates have increased over the past two decades. As there are no estimates to inform on recent post-release survival rates, we assume in the MICE ensemble that post-release survival is zero. This is also because there is considerable uncertainty regarding the catch rates we apply in the model, and hence adding an additional unknown parameter would only confound this uncertainty further. However, we ran sensitivity analyses to evaluate how sensitive the model results are to assuming that post-release survival has improved since 2000. Given no reliable estimates of post-release survival, we used four alternative values to span a range of post-release survival examples : 0, 0.25, 0.75 & 0.9

We use a monthly time step to model sawfish. This is because monthly effort data are available for the different fishery sectors to enable estimation of seasonal patterns of catches. Monthly time steps also enable resolution of fine scale responses of sawfish to changes in flow regime. For example, during the breeding season in wet years, pups will move much further upstream of rivers than during the dry season. Indeed, there are records of sawfish found greater than 350 km upstream (Pillans, pers comm). *Pristis pristis* is philopatric such that females release pups into the same river system where the mother was born (Phillips et al. 2011; Feutry et al. 2015). Although adult sawfish may move across

the model regions that we divided the GoC into, we assume that the population dynamics of each species are linked to the same river system.

Given the lack of information on mortality rates and abundance of sawfish species, specialist sawfish biologists have identified that the only likely technique that ‘in principle’ would be able to estimate adult sawfish abundance and mortality would be close-kin mark-recapture (CKMR) (Bravington *et al.*, 2016; Hillary *et al.*, 2018).

P. pristis is a relatively long-lived slow-growing species as is thought to live for a maximum of 35-80 years compared with species such as *A. cuspidata* that are thought to live to a maximum 9-12 years. We therefore use 20 ‘one-year’ age classes, with the final age 20+ being a plus group (i.e. it includes all animals 20 years and older) for *P. pristis*. The base natural mortality monthly rates for *P. pristis* is set at 0.0042month⁻¹. Additional mortality is added based on assumed relationships between flow and survival, which primarily impacts juvenile animals due to their reliance on riverine and estuarine habitats. A schematic of the model structure is shown in Figure A76.

Table A20. Population model parameters used for largetooth sawfish in the MICE, based on Pillans *et al.* (in press), and compared with parameters for a faster-growing species.

Parameter	Symbol	<i>A. cuspidata</i>	<i>P. pristis</i>
Maximum age	a_{max}	11	80
Age 50 maturity	a_{50}	2.5	8
Age 95 maturity	a_{95}	4	10
Reproductive Frequency	τ	1	2
Mean pups.female ⁻¹	$\tilde{\alpha}$	9	8
Av. pups.female.yr ⁻¹	α	9	4
Steepness	h	0.68	0.32
prop(Fem)	f	0.5	0.5
Initial B	B_0	1	1

Table A21. Detailed summary of life history parameters for *P. pristis* (extracted from Pillans *et al.* in press)

<i>Pristis pristis</i>	Value
size at birth	72-90 cm TL (Peverell, 2008)
size at maturity	300 cm TL (Thorson, 1976; Thorburn <i>et al.</i> 2007; Peverell 2008; Whitty <i>et al.</i> 2008)
age at maturity	8-10 (Thorson, 1982; Peverell, 2008)
max age	35 – 80 (Peverell, 2008)
litter size	4 (Peverell, 2005; 2008)

<i>Pristis pristis</i>	Value
	7.3 (Thorson, 1976)
	20 (Nunes et al. 2016)
Reproduction periodicity	1-2 (Thorson, 1976; Peverell, 2008)
L_{∞}	638 (Peverell, 2008)
K	0.08 (Peverell, 2008)
t_0	-1.55 (Peverell, 2008)
Juvenile mortality 0+ (Adelaide River)	0.866 (NESP, unpublished data)

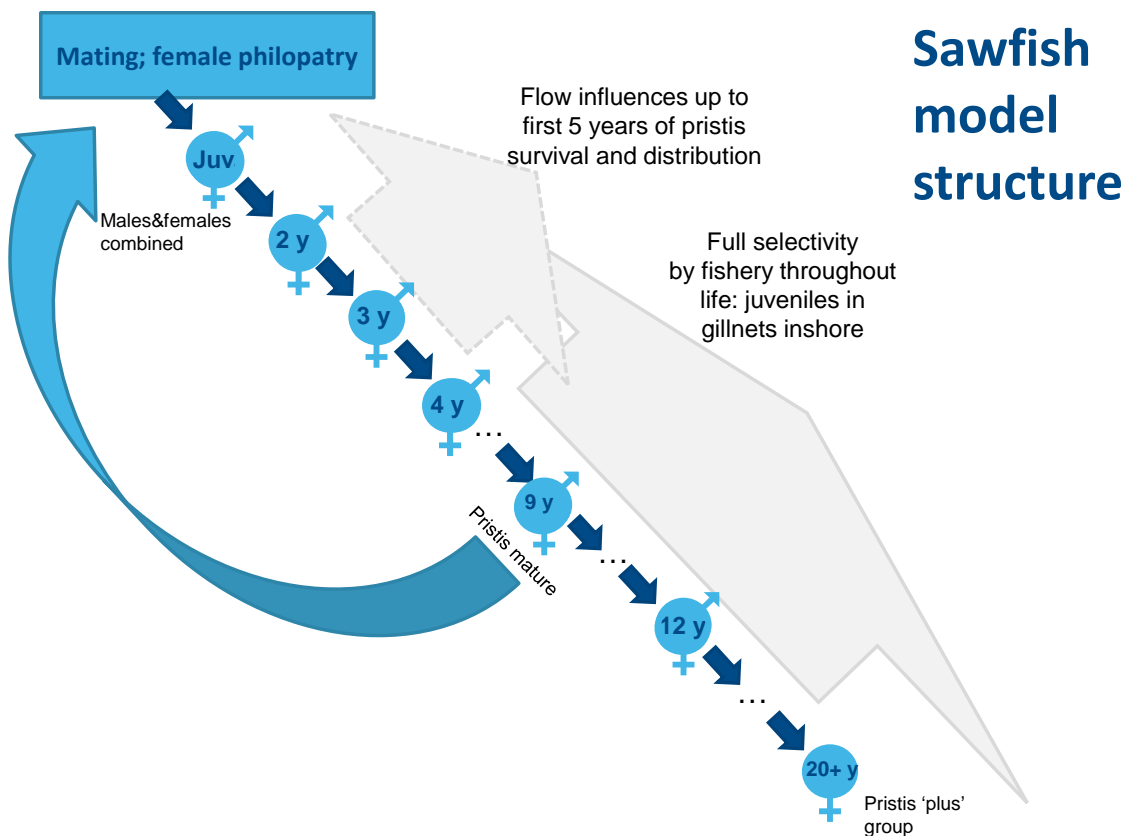


Figure A76. Schematic summary of MICE structure used to represent the largemouth sawfish *Pristis pristis*.

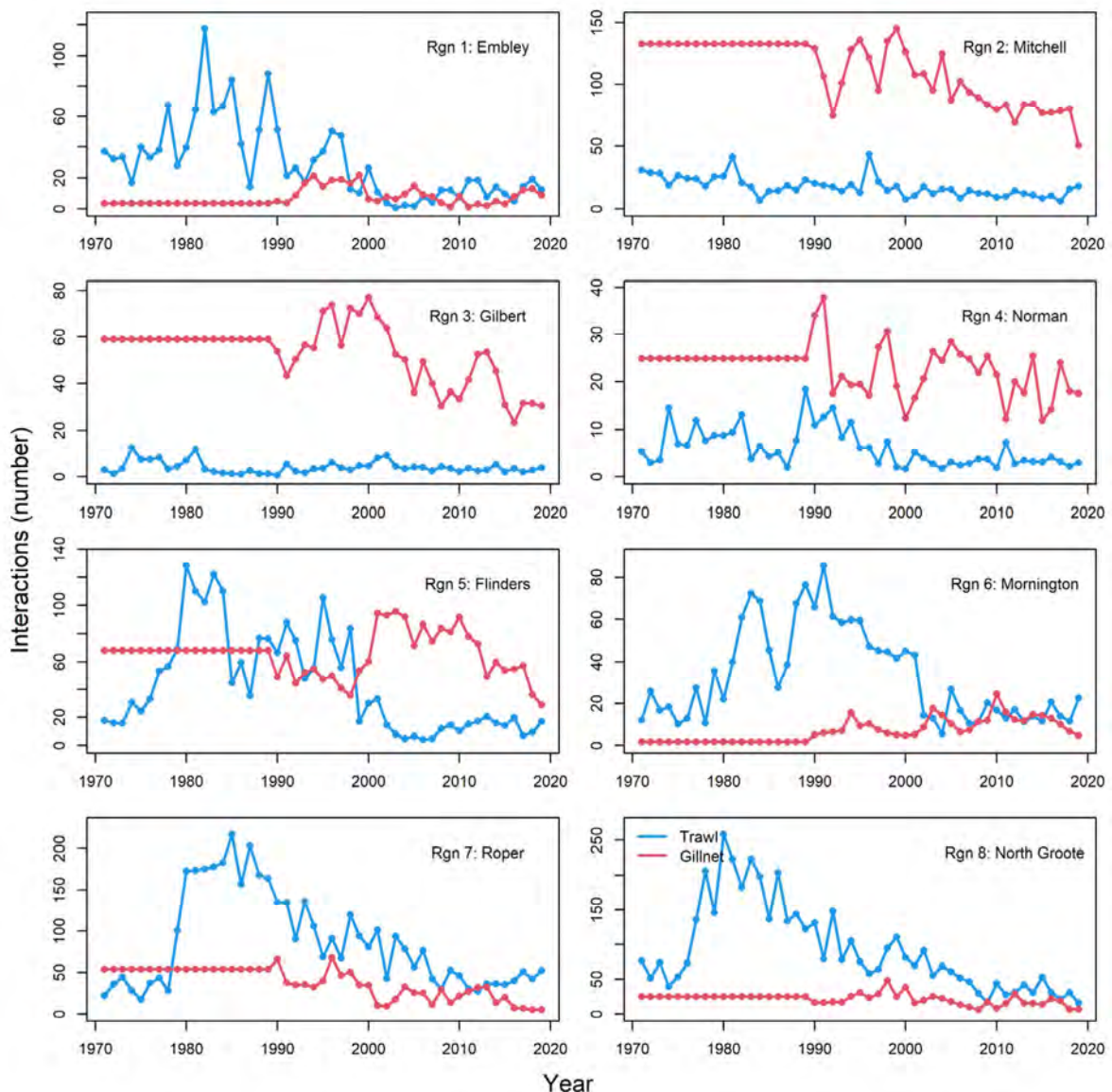


Figure A77. Summary of the relative trends in the numbers of largemouth sawfish interactions with each of the two fishery sectors, trawl and gillnet, in each of the MICE spatial regions. These are hypothetical numbers of interactions used to drive plausible population trajectories for largemouth sawfish through accounting for changes in fishing effort in the two sectors as well as in the different spatial regions (although we don't model largemouth sawfish in model regions 1, 6 and 8). The total interactions are calculated by applying catch rates from Pillans et al. (in press) for the two sectors to actual fishing effort from 1970 to 2019 (prawns) and 1989-2019 (barramundi – gillnet sector), and assuming the 1989 gillnet effort level was constant back to 1970 given the absence of better information.

Sawfish Model Equations

The *Pristis pristis* sawfish species is modelled using a monthly time-step, 20 age-classes (with a 20+ plus group that includes animals older than 20) and assuming separate sub-populations per model region, but with some common parameters estimated. The base natural mortality is assumed to be age-dependent and is fixed as a common parameter across all regions. Equations are not sex-disaggregated and an even sex ratio is assumed. We assume an even sex ratio given no indications to the contrary, but note that this isn't observed in some other river shark species such as *Glyphis glyphis* (Patterson et

al. in press). As female largemouth sawfish are philopatric (Phillips et al. 2011, Feutry et al. 2015), no connectivity between regions is represented in the model, although some movement of older animals (i.e. once they leave their river nursery home) likely occurs and adults may move freely throughout the Gulf region. Hence the modelled sawfish numbers represent the recruitment that is specific to that river system (due to philopatry) as well as the resultant adults, but noting that some of these adults may move to neighbouring regions and interact with fishing gear there for example, which is beyond the scope of this model to capture.

To estimate catches, the model uses as input monthly data on prawn (all species combined) and barramundi fishing effort and estimates the total catch per month, per year and per region based on catch rates in Pillans et al. (in press). There is equal selectivity across all ages and months, although monthly catches depend on fishing selectivity patterns.

Recruitment is defined as the number of 1-year old sawfish and is modelled using a saturating stock-recruitment relationship to account for density-dependent recruitment effects.

Numbers-at-age

$$N_{r,y,m,1}^{saw} = R_{r,y,m}^{saw} \quad (1)$$

$$N_{r,y,m+1,a}^{saw} = N_{r,y,m,a}^{saw} e^{-M_a^{saw}} - C_{r,y,m,a}^{saw} \quad 1 < a < z; m = 1 \text{ to } 11 \quad (2)$$

$$N_{r,y+1,1,a+1}^{saw} = N_{r,y,m,a}^{saw} e^{-M_a^{saw}} - C_{r,y,m,a}^{saw} \quad m = 12$$

$$N_{r,y+1,1,z}^{saw} = \left(N_{r,y,12,z}^{saw} e^{-M_z^{saw}} - C_{r,y,12,z}^{saw} \right) + \left(N_{r,y,12,z-1}^{saw} e^{-M_{z-1}^{saw}} - C_{r,y,12,z-1}^{saw} \right) \quad a = z; m = 12 \quad (3)$$

where

$N_{r,y,m,a}^{saw}$ is the number of sawfish of age a in year y and month m in region r ;

$R_{r,y,m}^{saw}$ is the sawfish recruitment at the start of month m (with base-case setting this to zero for all months except January) in year y for region r ;

M_a^b is the age-dependent monthly natural mortality rate for sawfish computed based on a fixed scaling parameter and model estimated base value M_{base}^b (0.00208 per month) as follows:

$$M_a^s = M_{base}^s + \frac{\psi}{a+1} \quad \text{where } \psi = 0.02 \quad (4)$$

$C_{r,y,m,a}^b$ is the numbers of sawfish of age a caught in month m of year y in region r ;

z is the plus group (age 20+ years)

Recruitment

Recruitment is defined as the number of 1-year old sawfish joining the population for the first time i.e. surviving pups and juveniles, and this is assumed to occur once a year only during January (to simplify computations).

Flow is linked to sawfish recruitment through a recruitment residual $\sigma_{r,y,m}^{flow}$ (see eqn for further details) as follows:

$$R_{r,y,1}^{saw} = \frac{\left(\frac{N_{r,y-1,1}^{sp_saw}}{K_r^{saw}} \right)}{1 + \beta^{saw} \left(\frac{N_{r,y-1,1}^{sp_saw}}{K_r^{saw}} \right)} \cdot \omega^{saw} \cdot \frac{1}{\tau^{saw}} \cdot pups^{saw} \cdot N_{r,y-1,1}^{sp_saw} \cdot S_{juv}^{saw} \cdot \sigma_{r,\bar{y}}^{flow} \quad (5)$$

where

$R_{r,y,m}^{saw}$ is the number of sawfish 1-year old recruits in each region r in month $m=1$ of year y ;

$N_{r,y-1,1}^{sp_saw}$ is the total breeding population (animals at or older than the age at sexual maturity) in each region r in month $m=1$ of year y :

$$N_{r,y,1}^{sp_saw} = \sum_{a=1}^z f_a^{saw} N_{r,y,1,a}^{saw} \quad \text{where} \quad f_a^{saw} = 1 \text{ if } a \geq agemat^{saw} \text{ else } f_a^{saw} = 0$$

β^{saw} is a density-dependent controlling parameter (fixed at 0.8 in base-case);

$agemat^{saw}$ is the age-at-first maturity for sawfish of species saw , with a knife-edge maturity function assumed;

f_a^{saw} is the proportion of sawfish of age a that are mature, and is fixed at 1 for all ages at or older than $agemat^{saw}$ else is zero;

ω^{saw} is the sawfish sex ratio (fixed at 0.5);

τ^{saw} is the sawfish reproductive frequency (with value 2 forargetooth sawfish)

$pups^{saw}$ is the average number of pups per female sawfish per year;

S_{juv}^{saw} is the base juvenile survival proportion applied to pups that survive the first few months of the year, and is computed as an equilibrium parameter that keeps the population in balance in the absence of external sources of mortality.

The model start year (1970) assumes the sawfish are at carrying capacity K_r^{saw} , i.e.

$$N_{r,sty,1}^{sp_saw} = K_r^{saw} \quad (6)$$

The base-case model assumes the same K_r^{saw} for all model regions except for Region 6 which is set lower due to being a small area with smaller rivers. The lower bound for K_r^{saw} is set at a sufficiently large value so that it doesn't result in extinction of the sawfish in the most heavily fished region, and can be adjusted upwards to simulate alternative current depletion scenarios (or downwards for other regions).

Starting values for biomass trajectories

The resource is assumed to be at deterministic equilibrium (corresponding to an absence of harvesting) at the start of 1970, the initial year considered here. Given a value for the pre-exploitation total breeding population size K_r^{saw} together with the assumption of an initial equilibrium age structure, and converting monthly mortality rates to annual values, it follows that for each region:

$$K_r^{saw} = R_0^{saw} \cdot \left[\sum_{a=agemat^{saw}}^{z-1} f_a^{saw} \exp\left(-\sum_{a'=1}^{a-1} 12 \times M_{a'}^{saw}\right) + \frac{\exp\left(-\sum_{a'=1}^{z-1} 12 \times M_{a'}^{saw}\right)}{1 - \exp\left(-12 \times M_z^{saw}\right)} \right] \quad (7)$$

which can be solved for the pristine sawfish recruitment number R_0^{saw} . This is in turn used to solve for S_{juv}^{saw} as follows (after substituting in Eqn (5)):

$$S_{juv}^{saw} = R_0^{saw} \cdot \frac{(1 + \beta^{saw}) \cdot \tau^{saw}}{\omega^{saw} \cdot pups^{saw} \cdot K_r^{saw}} \quad (8)$$

Solving yields base-case estimates of juvenile survival of S_{juv}^{saw} of 0.130/yr for largetooth sawfish.

The initial numbers at age corresponding to deterministic equilibrium, are:

$$\begin{aligned} N_{r,0,1,1}^{saw} &= R_0^{saw} \\ N_{r,0,1,a+1}^{saw} &= N_{r,0,1,a}^{saw} e^{-12 \times M_a^{saw}} \quad 1 < a < z-1 \\ N_{r,0,1,z}^{saw} &= \frac{N_{r,0,1,z-1}^{saw} e^{-12 \times M_{z-1}^{saw}}}{1 - e^{-12 \times M_z^{saw}}} \quad a = z \end{aligned} \quad (9)$$

Numbers-at-age for subsequent months and years are then computed by means of Equations (1) to (5).

Interactions – captures due to fishery interactions

Sawfish ($C_{r,y,m,a}^{saw}$) interactions are modelled as catches which are removed from the population and computed as:

$$C_{r,y,m,a}^{saw} = S_a^{saw} A_m^{saw} F_{r,y,m}^{saw} N_{r,y,m,a}^{saw} e^{-M_a^{saw}} \quad (10)$$

where

A_m^{saw} is the availability of sawfish to interactions in month m which in the base-case is assumed the same across all regions and set at 1 for all months;

S_a^{saw} is the selectivity to interactions of sawfish of age a , assumed the same across regions and in the base-case is set at 1 for all ages 1 and older;

$F_{r,y,m}^{saw}$ is the interaction proportion of a fully selected age class of sawfish, in month m and year y in region r and is calculated as follows:

$$F_{r,y,m}^{saw} = \widehat{C}_{r,y,m}^{saw} / B_{r,y,m}^{exp_saw} \quad (11)$$

Where the predicted interactions $\widehat{C}_{r,y,m}^{saw}$ for sawfish is computed as:

$$\widehat{C}_{r,y,m}^{saw} = (1 - PRS) \cdot [q_{trawl}^{saw} E_{r,y,m}^{trawl} + q_{net}^{saw} E_{r,y,m}^{net}] \quad (12)$$

$q_{trawl}^{saw}; q_{net}^{saw}$ are the catchability coefficients for sawfish of species saw , that are caught respectively by the trawl sector and net-fishing sector;

PRS is the post-release survival proportion, which is set at 0 in the model ensemble but sensitivity tests are included with a range of higher values applied to interactions post-2000;

$E_{r,y,m}^{trawl}; E_{r,y,m}^{net}$ are the monthly fishing effort in each region r and year y for the trawl (sum of tiger prawn and common banana prawn fishing effort) and net-fishing (using barramundi effort as an index) sectors;

$B_{r,y,m}^{exp_saw}$ is the total number of sawfish susceptible (i.e. exploitable numbers) to be caught in each month, year and region and is calculated as follows:

$$B_{r,y,m}^{exp_saw} = \sum_{a=1}^z S_a^{saw} A_m^{saw} N_{r,y,m,a}^{saw} e^{-M_a^{saw}} \quad (13)$$

Note that the estimates of catch are computed based on catch rates in Pillans et al. (in press) applied to observed fishing effort, and hence are independent of the sawfish population size in order to maintain consistency in the first instance with previous studies. This could be modified in future work by linking with the sawfish available numbers in the model. Note also that the sawfish populations are

modelled in units of numbers only, but this could be converted to biomass estimates in future if considered necessary.

Environmental Drivers

The main physical driver of sawfish currently included in the model is river flow. As detailed in Appendix 16, based on the available scientific literature, changes in river flow are modelled as influencing sawfish breeding reproduction, survival of juvenile animals as well as leading to occasional large increases in mortality in response to pools drying out. We use primarily river flows during the wet season peak period (January to March) to represent the risk of seasonal fluctuations in water level resulting in sections of rivers becoming small pools and billabongs that cannot adequately support the food and space requirements of sawfish for lengthy periods (Thorburn et al. 2007). An additional sensitivity test is presented in Appendix 20 to test the effect on model results of changing the model structure and assuming that river flows don't influence largetooth sawfish recruitment success except in exceptional years.

Results when testing alternative post-release survival proportion of sawfish

We used Model version 5 to investigate the sensitivity of model results to using each of the four alternative values to span a range of post-release survival (PRS) examples: 0, 0.25, 0.75 & 0.9. We ran both the baseline flow model and for each post-release survival value, we ran each of the four WRDs to evaluate the model sensitivity to unknown post-release survival. Model version 5 includes a range of alternative current depletion levels to ensure we tested this effect for lower and higher depletion levels (Table A22). The corresponding reduction in the number of animals that are assumed to die from fishery interactions for alternative assumptions of post-release survival is shown in Figure A78.

Table A22. Comparison of Model version 5 current depletion level (mature animals in 2020 compared with 1970 numbers) when using default setting of post-release survival (PRS) proportion as zero, compared with assuming increases in PRS from 25%, 75% to 90% of sawfish assumed to survive following release from incidental capture in each of the three regions as shown.

	Mitchell River	Gilbert River	Flinders River
PRS0	0.17	0.04	0.18
PRS25	0.19	0.05	0.19
PRS75	0.22	0.07	0.20
PRS90	0.24	0.08	0.21

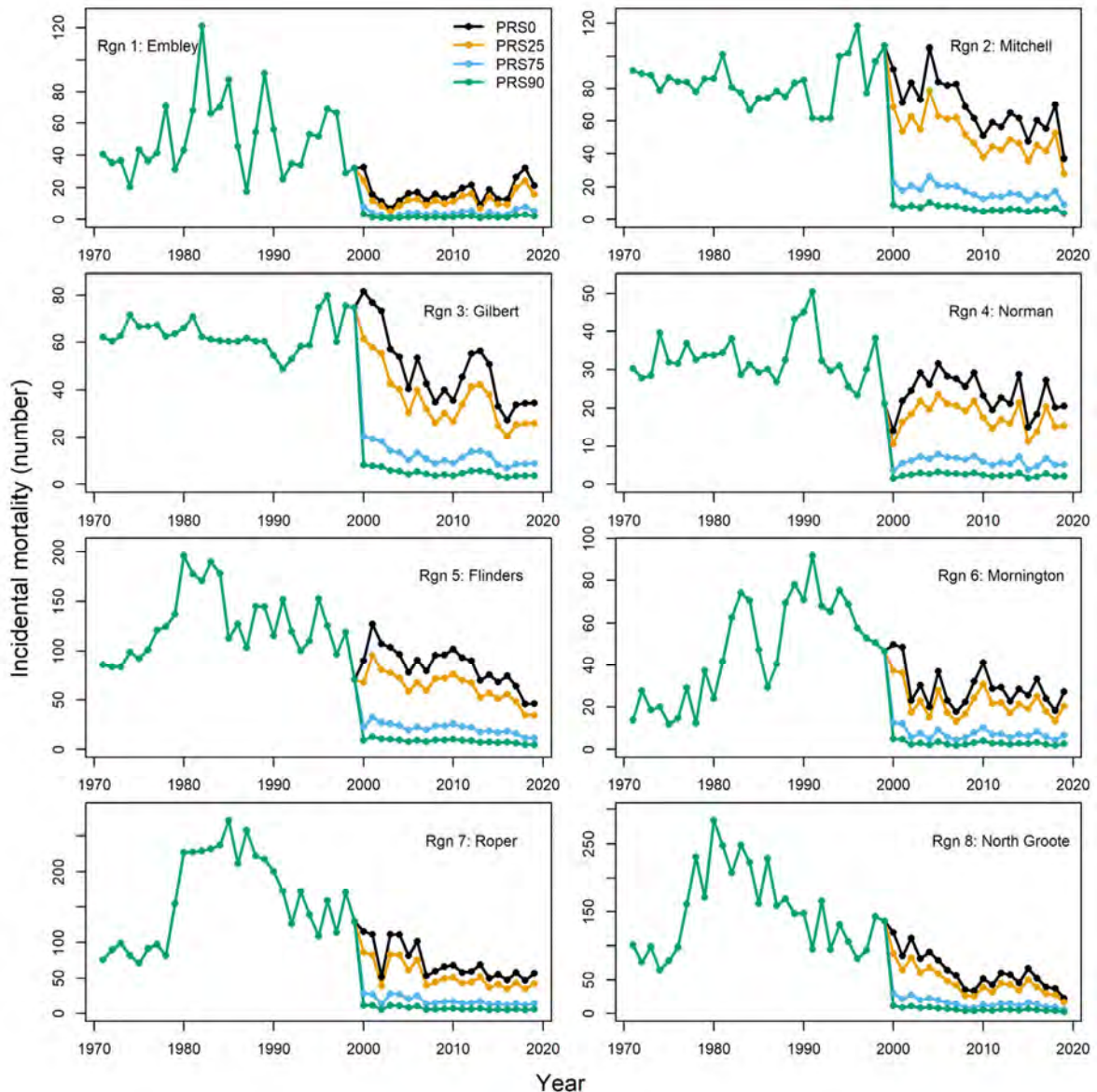


Figure A78. Summary of the relative trends in the numbers of largemouth sawfish assumed to die following interactions with fishing gear (trawl and gillnet combined), in each of the MICE spatial regions (although we don't model largemouth sawfish in model regions 1, 6 and 8), and when using the 3 sensitivity tests with post-release survival after 2000 assumed to be: 0%, 25%, 75%, 90%. As can be seen from the plots, as post-release survival rate is assumed to increase (with effect from year 2000), the incidental mortality (expressed as total annual number of animals which die) decreases in sync and can potentially decrease to very low levels relative to historical levels if post-release survival improves substantially.

As shown in Table A22, increases in the assumed post-release survival rate led to minor (1%) changes in the 2020 population status predictions when using a post-survival proportion of 25% whereas the optimistic scenario that assumes 90% survival after release, led to much larger changes in the modelled current depletion level. As these changes in post-release survival are assumed to have only come into effect in 2000, there is a lag time before the benefits are seen in the number of mature animals, as illustrated by comparing examples of the range of outcomes when plotting total sawfish population numbers versus mature animals only (Figure A79). These results support that under baseline conditions, increases in post-release survival can reverse population declines or substantially improve population status even though it may take some time before their impact is seen in terms of

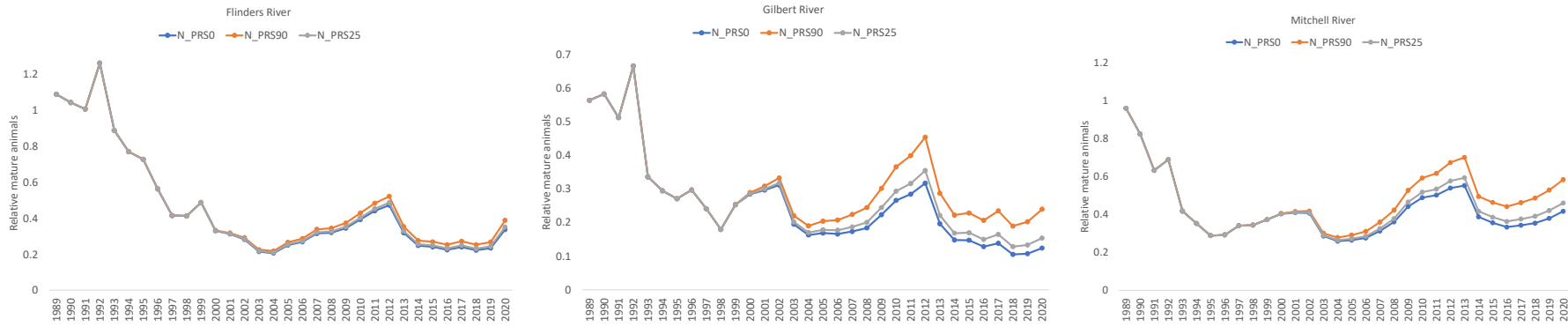
total population size. Over a longer time period, the benefits will become even greater than shown in Figure A78. Reducing this source of mortality on largemouth sawfish also has a substantial effect in reducing the risk that population numbers will decrease to very low levels in some years, as shown in Table A20. These results highlight the urgent need for improved information on post-release survival as well as mitigation measures – such as adopted by NPFI to improve post-release survival.

The results from evaluating the impact of alternative post-release survival rates on predicted outcomes under different WRDs are shown in Table A23. Hence, for example, the base-case model with PRS=0% estimates that WRD1 will result in the total numbers of sawfish reliant on the Mitchell River to decrease down to 29% on average of the comparable level under a no-development scenario. This estimated depletion level increases to 30%, 31% and 31% respectively for PRS=25%, 75% and 90%, so there is a minor difference only in model predictions of the impact of WRD1. For the Gilbert River system, changes in PRS result in bigger increases in the average depletion, with range from 28% to 38%, as well as a substantial change in the minimum depletion predicted over all years (Table A22). The results for the Flinders River system are similar too – for example the range of predicted outcomes (average depletion level – mature animals) under WRD1 is 0.16 (PRS=0%) to 0.20 (PRS=90%) compared with a no-development scenario.

These results show that the model ensemble results are fairly robust to the assumption of zero post-release survival – this is because even with this effect accounted for, the model predicts very large decreases in largemouth sawfish abundance under some WRD scenarios. Moreover, the extent of the decreases is such that regardless of PRS setting used in the model, the outcomes still mostly correspond to the intolerable risk rating based on our scoring system. Our sensitivity analysis therefore supports results from the MICE ensemble suggesting high sensitivity of sawfish population dynamics to WRDs.

As mentioned above, saltwater crocodile numbers have increased in northern Australia since 1970, and hence there is the possibility that any recent improvements in post-release survival rate are offset by increases in predation mortality of young animals. This aspect could be investigated in future model versions, and we note that our modelling of largemouth sawfish remains uncertain due to the urgent need for more information on these threatened animals.

(A) Total population numbers



(B) Total mature animals

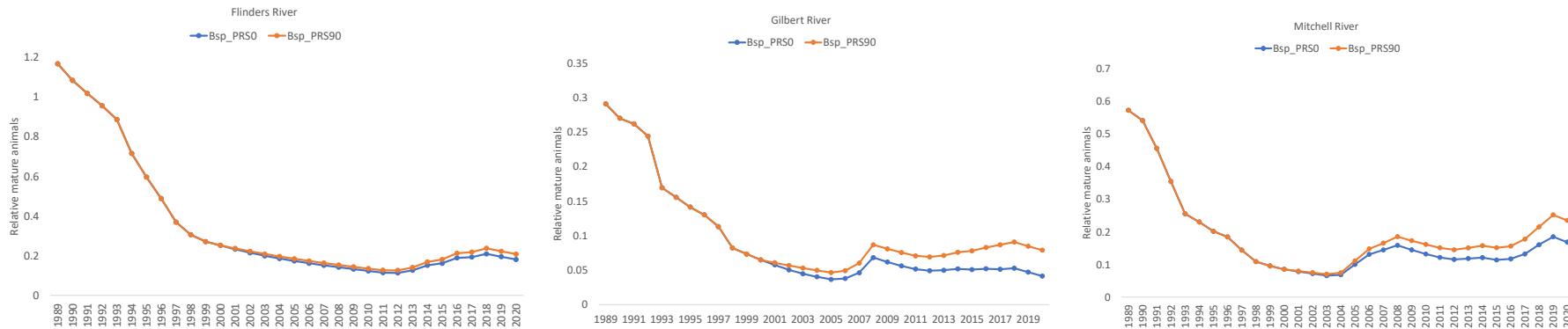


Figure A79. Plots showing the change in the relative numbers of (A) all sawfish and (B) mature animals for each of the Flinders, Gilbert and Mitchell rivers when assuming different values of post-release survival. Top plot shows base case with PRS=0% compared with 25% and 90% whereas, for ease of discriminating the difference, the lower plot shows the two bounds of post-release survival, namely 0% and 90%.

Table A23. Summary table of Sawfish sensitivity analyses (applied to Model version 5) and using different post-release survival proportions of 0, 0.25, 0.75 and 0.9. Values in the Table are minimum and mean number of individuals, and number of mature individuals (Bsp) for sawfish predicted under four water resource development scenarios (WRD1-WRD4) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.

Model version 5 - sawfish - PRS = 0.0						Model v.5 - PRS = 0.25				Model v.5 - PRS = 0.75				Model v.5 - PRS = 0.9			
WRD Scenario	Rgn Name	Rel.N (Min)	Rel.N (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)	Rel.N (Min)	Rel.N (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)	Rel.N (Min)	Rel.N (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)	Rel.N (Min)	Rel.N (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)
BaseCase	Mitchell	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
BaseCase	Gilbert	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
BaseCase	Flinders	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
BaseCase	Rgn 2-6	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
WRD1	Mitchell	0.14	0.29	0.16	0.3	0.15	0.30	0.18	0.31	0.18	0.31	0.21	0.33	0.19	0.31	0.21	0.33
WRD1	Gilbert	0.05	0.28	0.07	0.31	0.10	0.30	0.13	0.33	0.21	0.36	0.27	0.39	0.26	0.38	0.32	0.41
WRD1	Flinders	0.04	0.16	0.04	0.17	0.05	0.17	0.05	0.18	0.09	0.19	0.09	0.21	0.11	0.20	0.10	0.21
WRD1	Rgn 2-6	0.11	0.24	0.09	0.22	0.13	0.24	0.10	0.22	0.16	0.26	0.13	0.24	0.17	0.27	0.14	0.25
WRD2	Mitchell	0.21	0.36	0.24	0.37	0.23	0.36	0.26	0.38	0.24	0.37	0.28	0.39	0.25	0.37	0.29	0.39
WRD2	Gilbert	0.05	0.28	0.07	0.31	0.10	0.30	0.13	0.33	0.21	0.36	0.27	0.40	0.26	0.38	0.32	0.42
WRD2	Flinders	0.04	0.17	0.04	0.18	0.06	0.18	0.06	0.19	0.09	0.20	0.09	0.21	0.11	0.20	0.10	0.22
WRD2	Rgn 2-6	0.12	0.25	0.1	0.23	0.14	0.25	0.11	0.23	0.17	0.27	0.14	0.25	0.19	0.28	0.15	0.25
WRD3	Mitchell	0.19	0.33	0.21	0.34	0.20	0.33	0.23	0.35	0.22	0.34	0.25	0.36	0.23	0.35	0.26	0.37
WRD3	Gilbert	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
WRD3	Flinders	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
WRD3	Rgn 2-6	0.85	0.91	0.89	0.94	0.85	0.91	0.89	0.94	0.84	0.91	0.90	0.94	0.84	0.91	0.90	0.94
WRD4	Mitchell	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1	1
WRD4	Gilbert	0.05	0.28	0.07	0.31	0.10	0.30	0.13	0.33	0.21	0.36	0.27	0.40	0.26	0.38	0.32	0.42
WRD4	Flinders	0.04	0.17	0.04	0.18	0.06	0.18	0.06	0.19	0.09	0.20	0.09	0.21	0.11	0.20	0.10	0.22
WRD4	Rgn 2-6	0.25	0.33	0.16	0.28	0.26	0.34	0.17	0.29	0.29	0.36	0.20	0.30	0.29	0.36	0.21	0.31

The need for further field-based research on sawfish

One of the key issues when dealing with sawfish is the lack of quantitative data on abundance, movement/habitat use and the response of individuals and populations to variability in environmental flows. Research is urgently required to gain a better understanding of largetooth sawfish population status and ecology in rivers that are likely to be impacted by water extraction and upstream dams. Targeted field-based research on sawfish could significantly improve our understanding of abundance and movement, as has been done for the Critically Endangered speartooth shark (*Glyphis glyphis*) in the Wenlock and Ducie Rivers in Queensland (see Box 1 below).

Box 1. Example of how targeted field research can be used to address knowledge gaps.

The speartooth shark is a euryhaline species found in a few river systems in Australia and Papua New Guinea (Pillans et al. 2009; White et al. 2017). Prior to 2009 virtually nothing was known about the species, however intensive research by CSIRO in the Wenlock and Ducie River has provided estimates of adult population size and detailed observations of the long-term movement and habitat use of juveniles. Using genetic samples from juveniles in a close-kin mark recapture model, the adult population has been estimated at approximately 900 individuals with as few as 50 females breeding each year (Patterson et al. in press). Genetics was also used to demonstrate that the Wenlock River population is genetically distinct from populations in PNG and the Northern Territory (Feutry et al. 2014).

Over 300 juveniles have been tagged with acoustic tags and monitored by an array of acoustic receivers from the mouth to 90 km upstream including major tributaries of both rivers. Data from detections of tagged animals has provided long term data on movement and habitat use showing that juveniles change their distribution seasonally (Lyon et al. 2017; Pillans et al. in press). During the dry season animals are 30–60 km upstream and move downstream in response to increased river flows at the onset of the monsoon. During the wet season, animals are between 0–30 km upstream of the river mouth with occasional movements into Port Musgrave in years of prolonged and high freshwater influx. This research has demonstrated partial segregation of age-classes with animals moving further downstream as they age/grow. However, all juveniles are confined to the river and estuary for 7 – 9 years and then move into Port Musgrave before moving offshore. Recent research has also demonstrated that the mortality rates of speartooth sharks between 0–2 y is around 60% (Patterson et al. in press). These are amongst the highest estimates of mortality recorded in an elasmobranch and suggests that a combination of natural and fishing mortality (from commercial and recreational fisheries) results in very few animals surviving to adults. A mark-recapture experiment has recently been completed and estimates of the juvenile population will be available shortly. Only three adults have been captured in Australia with all of these in Port Musgrave. All of these animals were tagged with satellite tags. One animal moved ~ 600 km NW across the Gulf of Carpentaria before the tag detached and another spent 3 months within Port Musgrave before the tag detached (CSIRO, unpublished data).

References

- Bigelow, HB. and Schroeder. 1953. *Fishes of the western North Atlantic Pt. 2. Sawfishes, guitarfishes, skates and rays. Chimaeroids*, New Haven, Conn, Sears Foundation for Marine Research, Yale Univ.
- Bravington, MV, Skaug, HJ, Anderson, EC and others. 2016. "Close-Kin Mark-Recapture." *Statistical Science* 31 (2). Institute of Mathematical Statistics: 259–74.

- Dulvy, NK, Davidson, LNK, Kyne, PM, Simpfendorfer, CA, Harrison, LR, Carlson, JK and Fordham, SV. 2016. "Ghosts of the Coast: Global Extinction Risk and Conservation of Sawfishes." *Aquatic Conservation: Marine and Freshwater Ecosystems* 26 (1):134–53.
- Dulvy, NK, Sarah L Fowler, John A Musick, Rachel D Cavanagh, Peter M Kyne, Lucy R Harrison, John K Carlson, et al. 2014. "Extinction Risk and Conservation of the World's Sharks and Rays." *Elife* 3. eLife Sciences Publications, Ltd.
- Feutry P, Kyne PM, Pillans RD, Chen X, Marthick J, Morgan DL, Grewe PM (2015) Whole mitogenome sequencing refines population structure of the Critically Endangered sawfish *Pristis pristis*. *Mar Ecol Prog Ser* 533:237-244. <https://doi.org/10.3354/meps11354>
- Fry, G., Laird, A., Lawrence, E., Miller, M. and Tonks, M. (2018) Monitoring interactions with bycatch species using crew-member observer data collected in the Northern Prawn Fishery: 2014 – 2016. Final Report to AFMA; R2015/0812. June 2018. CSIRO, Australia. Pp. 236
- Last PR, Stevens JD (2009) *Sharks and rays of Australia*, 2nd edn. CSIRO Division of Fisheries, Melbourne
- Hillary, RM, Bravington, MV, Patterson, TA, Grewe, P, Bradford, R, Feutry, P, Gunasekera, R, Peddemors, V, Werry, J, Francis, MP and Duffy, CAJ. (2018). Genetic relatedness reveals total population size of white sharks in eastern Australia and New Zealand. *Scientific Reports*, 8(1), p.2661.
- Kyne, P.M. and Pillans, R.D., 2014. *Protocols for Surveying and Tagging Sawfishes and River Sharks. National Environmental Research Program Marine Biodiversity Hub, Hobart.*
- Nunes, JLS, Rincon, G, Piorski, NM and Martins, APB. 2016. Near-term embryos in a *Pristis pristis* (Elasmobranchii: Pristidae) from Brazil. *Journal of Fish Biology* 89: 1112- 1120
- Peverell, SC. 2005. "Distribution of Sawfishes (Pristidae) in the Queensland Gulf of Carpentaria, Australia, with Notes on Sawfish Ecology." *Environmental Biology of Fishes* 73 (4). Springer: 391–402.
- Pianka, ER. 1970. "On R-and K-Selection." *The American Naturalist* 104 (940). University of Chicago Press: 592–97.
- Phillips, N.M., Chaplin, J.A., Morgan, D.L. et al. Population genetic structure and genetic diversity of three critically endangered *Pristis* sawfishes in Australian waters. *Mar Biol* **158**, 903–915 (2011). <https://doi.org/10.1007/s00227-010-1617-z>
- Pillans, R.D., Hillary, R., Patterson, T. In Press. Assessing the effect of ongoing bycatch of sawfish in northern Australia.
- Putt, J. and Anderson, K. (2007). A national study of crime in the Australian fishing industry. Research & public policy series no. 76. Australian Institute of Criminology. Canberra.
- Stevens, JD, McAuley, RB, Simpfendorfer, CA and Pillans, RD. 2008. "Spatial Distribution and Habitat Utilisation of Sawfish (*Pristis* Spp) in Relation to Fishing in Northern Australia." *A Report to Department of the Environment, Water, Heritage and the Arts* 31.
- Thorburn, DC, Morgan, DL, Rowland, AJ and Gill, HS. 2007. "Freshwater Sawfish *Pristis microdon* Latham, 1794 (Chondrichthyes: Pristidae) in the Kimberley Region of Western Australia." *Zootaxa* 1471 (2).
- Thorson, TB. 1976. "Observations on the Reproduction of the Sawfish, *Pristis perotteti*, in Lake Nicaragua, with Recommendations for Its Conservation."

Thorson, TB. 1982. "Life History Implications of a Tagging Study of the Largetooth Sawfish, *Pristis Perotteti*, in the Lake Nicaragua-Rio San Juan System." *Environmental Biology of Fishes* 7 (3). Springer: 207–28.

Whitty, J.M., D.L. Morgan, D.C. Thorburn, T. Fazeldean & S.C. Peverell (2008). Tracking the movements of Freshwater Sawfish (*Pristis microdon*) and Northern River Sharks (*Glyphis* sp. C) in the Fitzroy River. Whitty, J.M., N.M. Phillips, D.L. Morgan, J.A. Chaplin, D.C. Thorburn & S.C. Peverell, eds. *Habitat associations of Freshwater Sawfish (Pristis microdon) and Northern River Sharks (Glyphis sp. C): including genetic analysis of P. microdon across northern Australia*. Centre for Fish & Fisheries Research (Murdoch University) report to the Department of the Environment, Water, Heritage and the Arts, Australian Government.

Appendix 14 – Microphytobenthos and meiofauna

Microphytobenthos and meiofauna equations were developed as part of the MICE to capture relationships between salt flat production, nutrients and prawn production in the southern GoC estuaries.

These groups represent the base of the Food Web, which ultimately supports prawns, mud crabs, barramundi, shorebirds and other species, and hence can be used as an index of ecosystem health.

We understand from NESP work (Burford, Kenyon, Duggan, Faggotter and others) how salinity, primary productivity and meiofaunal abundance change during wet-dry seasons and floods (Figure A50, Duggan et al. 2014), and therefore a start has been made to try and capture this understanding dynamically in the MICE model.

A brief summary of the equations used to model these components is shown in Figure A80-Figure A85.

The NESP work has also shown river systems are nutrient poor but large floods recruit nutrients from flooded adjacent land and deposits will shift offshore and boost the following year's productivity and hence banana prawn catches (Burford et al. 2020). The MICE therefore computes a productivity boost after flooding which is in turn assumed to influence the prawn survival.

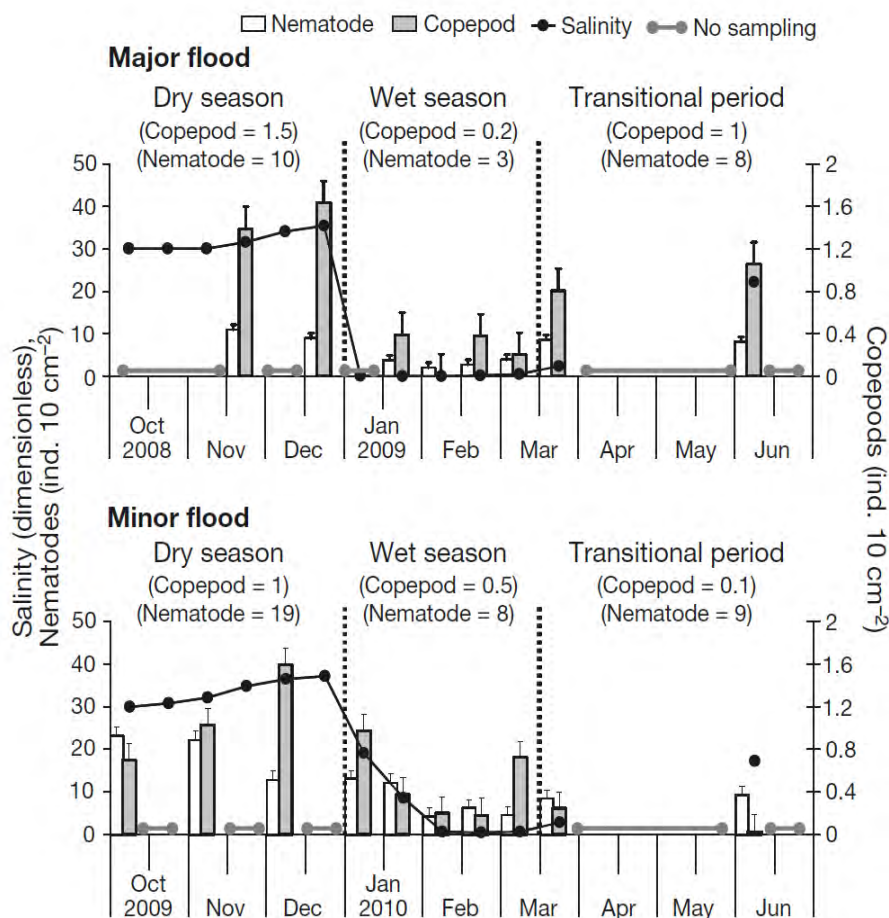


Figure A80. Extract from Duggan et al. (2014) showing measures of changes in meiofauna (nematodes, copepods) and salinity during different seasons and flood conditions in the Norman River.

Microphytobenthos Equations: revision

New biomass

$$B_{a,y,w+1} = B_{a,y,w} + G_{a,y,w} B_{a,y,w} - M_{a,y,w}^n B_{a,y,w} - M_{a,y,w}^{scour} B_{a,y,w}$$



Growth

$$G_{a,y,w} = \mu \cdot \delta_a^N \cdot \delta_{a,y,w}^{sal} \cdot \delta_{a,y,w}^{SST} \cdot \delta_{a,y,w}^{solar}$$

Max growth rate Nutrient limitation or primary productivity potential parameter scaled for different river systems Salinity (based on flow) Sea Surface Temperature (SST) (currently assumed same for years and areas) Solar exposure

* Can be refined for different flood levels and river systems. but substantiation needed so starting simple

Figure A81. Summary of MICE biomass dynamic equations to model microphytobenthos in each week w of year y in model region a . The growth rate is shown to depend on a number of key physical drivers, including nutrient limitation, salinity, sea surface temperature and solar exposure.

Meiofauna Equations

New biomass

Similar equation but growth rate depends on microphytobenthos which is a proxy for productivity; SST not included in growth term as is already in microphytobenthos term

Growth

$$G_{a,y,w} = \mu \cdot \delta_a^{Chla} \cdot \delta_{a,y,w}^{sal}$$

Max growth rate Microphytobenthos biomass influences growth rate Salinity (based on flow)

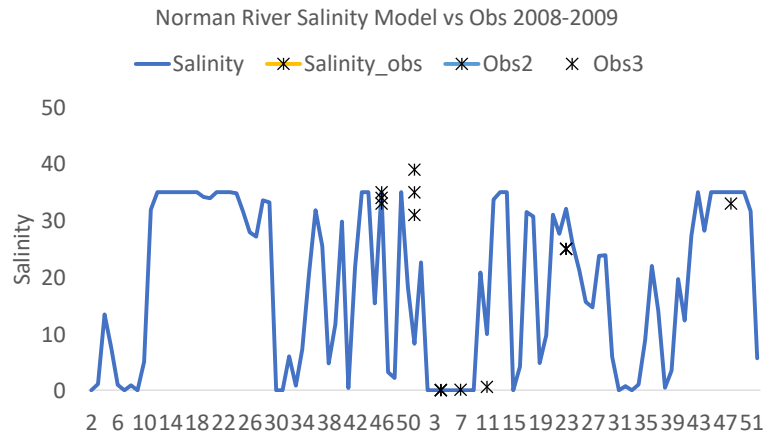
$$\delta_a^{Chla} = 0.01 \cdot B_{a,y,w} \cdot \left(1 - \frac{B_{a,y,w}}{B_{a,l,l}} \right)$$

Figure A82. Summary of MICE biomass dynamic equations to model meiofauna (principally nematodes and copepods) in each week w of year y in model region a . The growth rate is shown to depend on the microphytobenthos biomass as well as changes in salinity.

Norman River

Validation using data from Duggan et al. 2014

Use Salinity-Flow relationship and compare Model vs Observed



Compare modelled meiofauna with observed copepod and nematode abundance

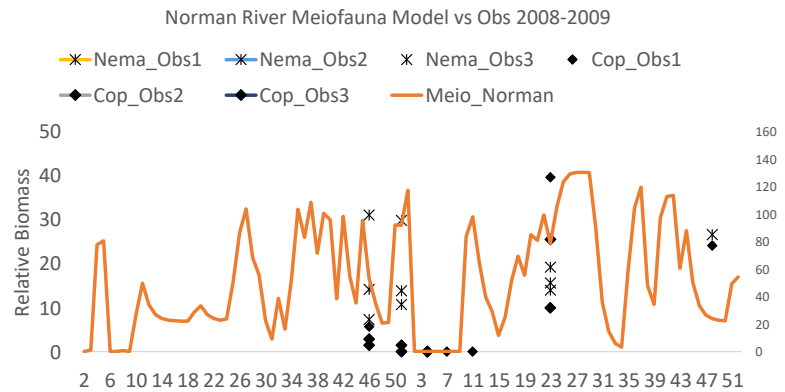


Figure A83. Preliminary validation of MICE salinity and meiofauna model components through comparison with observations of Duggan et al. (2014) from the Norman River region.

Microphytobenthos Example Trajectories

Comparing Gilbert vs Flinders in 2009, 2015 and 2018

Flinders calibrated to be more productive

Different patterns in wet vs dry years and seasons

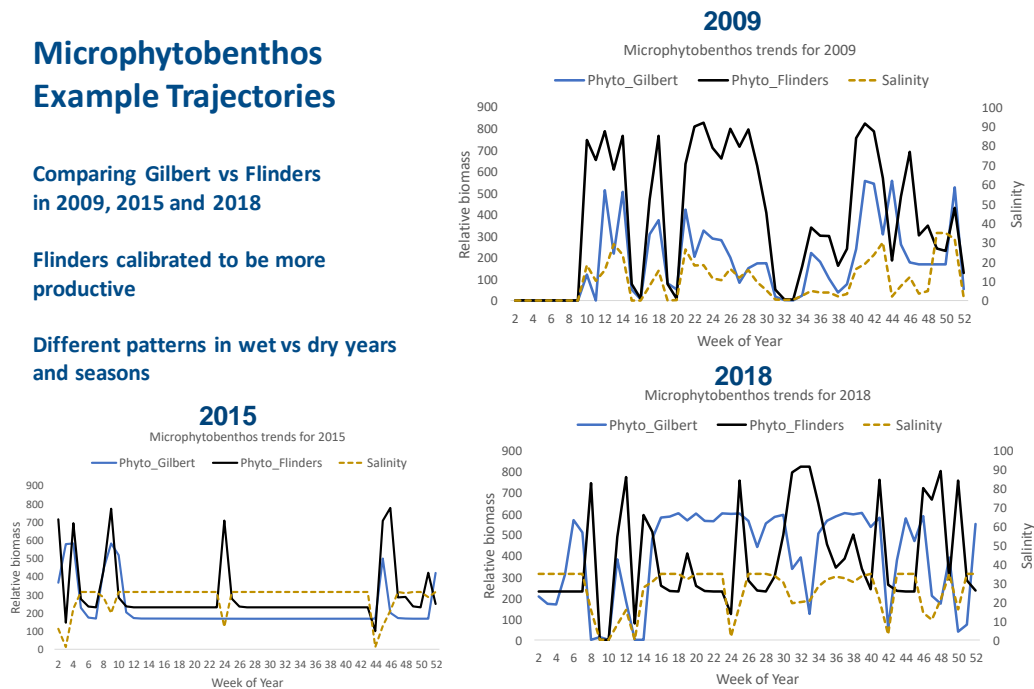


Figure A84. Model-estimated microphytobenthos trajectories showing differences between years and model regions.

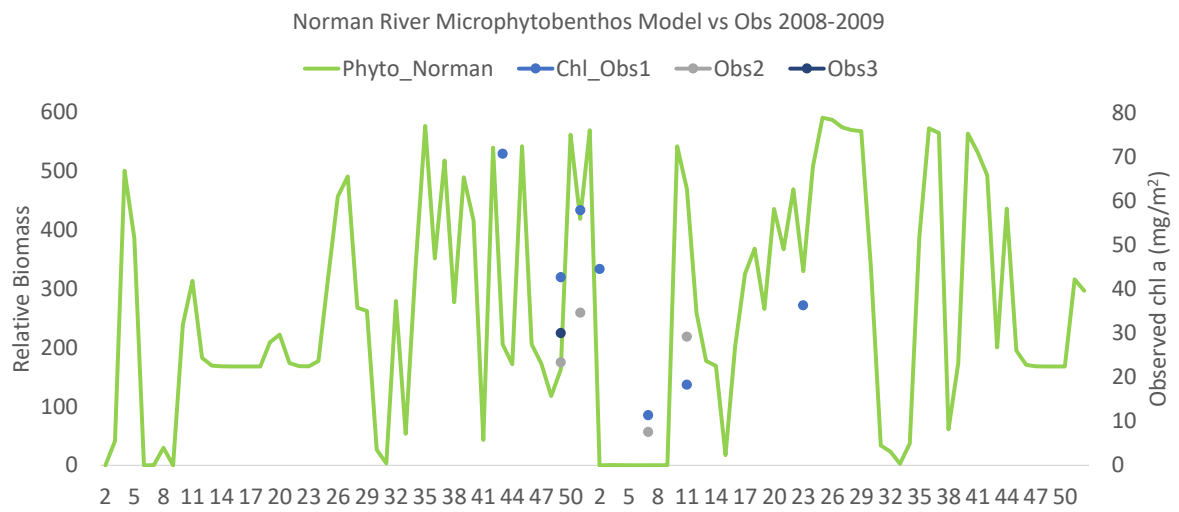


Figure A85. Preliminary validation of MICE microphytobenthos model component through comparison with observations of Duggan et al. (2014) from the Norman River region

Appendix 15 – Seagrass and Mangroves

Seagrass modelling

Seagrass are the base of the food web and a key food source in the tropics where phytoplankton biomass is extremely low (Adey 1998). They are the main diet of dugongs and green turtles which were beyond the scope of this study to model, and provide a habitat for many marine animals, including commercially important species such as prawns. They also help to keep coastal waters clear as stabilise sediment and absorb nutrients from coastal run-off. Seagrass and algae also provide critical nursery habitat for juvenile tiger prawns *Penaeus esculentus* and *Penaeus semisulcatus* (Haywood et al. 1995, Kenyon et al. 1997b, Loneragan et al. 1998, Dichmont et al. 2007, Haywood and Kenyon 2009). After spawning in deep water, the planktonic tiger prawn larvae are transported to shallow coastal area. Postlarvae settle on beds of seagrass and algae, three to four weeks after the eggs are released from the females (Haywood et al. 1995, Liu and Loneragan 1997, Haywood and Kenyon 2009).

The GoC has high seagrass species diversity and although we acknowledge that species-specific differences are important, we are unable to reliably capture these effects in our model and instead used a very simple model that represents combined seagrass biomass in each region of the GOC. Tropical seagrass dynamics are strongly influenced by both long term weather patterns and extreme flood and cyclone events (Carruthers et al. 2002). Other important drivers include turbidity and other restrictions to light availability, as well as temperature and inorganic nutrients (Lee et al. 2007, Collier and Waycott 2014). Salinity variations can affect the reproduction and distribution of seagrass species (Short et al. 2001) and is thus also included in the model (see equations below). As for microphytobenthos, we assume a parabolic relationship between SST and growth rate as well as salinity and growth rate, although the parameters of this relationship can be modified to better represent seagrass if more information becomes available.

Although there are several factors, including daily and seasonal variability, that will influence light attenuation affecting seagrass (Ralph et al. 2007), plus noting that different species have different sensitivities (Lee et al. 2007), here we use a much simpler starting relationship which consider only the average weekly solar exposure per area and major influences on coastal turbidity due to floods bringing additional sediments as well as cyclones mixing the water column and increasing turbidity. Turbidity and light attenuation are linearly related (MacDonald 2015). Here we assume also that river flow and turbidity are linearly related (a sensitivity to alternative formulations, such as assuming a step change in the relationship, could also readily be investigated). Hence our model assumes that seagrass light availability is inversely related to flow as well as inversely (linearly) related to cyclone category. We multiply the relative solar exposure by this value as shown in the equations below.

The seagrass additional mortality due to cyclone damage is modelled as directly proportional to cyclone category in the region corresponding to the landfall area, but we also multiply this by the evaluation of the cyclone impact.

The discrete updating equation for the biomass of seagrass $B_{a,y,w}^{seagr}$ in area a and week w of year y is:

$$B_{a,y,w+1}^{seagr} = B_{a,y,w}^{seagr} + G_{a,y,w}^{seagr} B_{a,y,w}^{seagr} \cdot \left(1 - B_{a,y,w}^{seagr} / K_a^{seagr}\right) - M_{a,y,w}^{seagr} B_{a,y,w}^{seagr}$$

where

K_a^{seagr} is the seagrass carrying capacity in area a ;

$M_{a,y,w}^{seagr}$ is the total seagrass mortality rate in area a and week w of year y computed using a base fixed mortality rate M_{base}^{seagr} and an additional mortality rate $M_{a,y,w}^{cyclone}$ that depends on the product of cyclone category $Cycl_{a,y,w}^{category}$ and impact score $Cycl_{a,y,w}^{impact}$, with constant of proportionality ξ^{seagr} :

$$M_{a,y,w}^{seagr} = M_{base}^{seagr} + M_{a,y,w}^{cyclone}$$

$$M_{a,y,w}^{cyclone} = \xi^{seagr} \cdot Cycl_{a,y,w}^{category} \cdot Cycl_{a,y,w}^{impact}$$

$G_{a,y,w}^{seagr}$ is the seagrass growth rate in area a and week w of year y computed as:

$$G_{a,y,w}^{seagr} = \mu^{seagr} \cdot \delta_{a,y,w}^{SST} \cdot \delta_{a,y,w}^{light}$$

Where μ^{seagr} is a fixed constant that is input;

$$\delta_{a,y,w}^{SST} = -0.01 \cdot (SST_w - 26)^2 + 1 \quad \text{and} \quad SST_w = 4.73 \cdot \sin(0.15(w + 4.5)) + 26.08$$

$$\delta_{a,y,w}^{light} = \sqrt{solar_{a,y,w}} \times \frac{1}{flow_{a,y,w}^{std}} \cdot Cycl_{a,y,w}$$

$$\text{if } flow \geq 1 \text{ else } \delta_{a,y,w}^{light} = \sqrt{solar_{a,y,w}}$$

$$\text{unless } flow < 1 \text{ and } Cycl_{a,y,w} > 0 \text{ then } \delta_{a,y,w}^{light} = \sqrt{solar_{a,y,w}} \times \frac{1}{(1 + Cycl_{a,y,w})}$$

Comparing Seagrass Model outputs with data

We found few suitable data sets to validate our seagrass model component, and future work will seek to improve our initial modelling. However, Kenyon et al. (1997a) conducted a one-year study on the seasonal growth of *Enhalus acoroides*, a tropical seagrass, that occurs in monospecific stands at Groote Eylandt and in association with *Cymodocea serrulata* or *Thalassia hemprichii*. These seagrass beds support diverse communities of juvenile fish and decapods and provide macrofauna with food and shelter (Blaber et al. 1992, Brewer et al. 1995, Kenyon et al. 1995). Kenyon et al. (1995) found leaf growth was positively correlated with water temperature and mean minimum water depth. Their results confirmed that growth varies with season – for example, leaves were longest in November and shortest in July and August, and the production of new shoots was greatest when water temperatures increased. We therefore confirmed that our model similarly replicates these patterns of seasonal growth in seagrass.

Kenyon et al. (1995) measured the mean growth per leaf of *E. acoroides* from November 1987 to July 1989. We used this time-series of seagrass growth to compare with our modelled seagrass trend over the same period and using model region 7 (noting that our model seagrass represents a grouping of seagrass species). As we didn't have absolute biomass estimates for seagrass, our model represents changes in relative seagrass biomass over time (i.e. relative to a starting 1970 reference level). Figure A86 shows the comparison between our model seagrass trajectory and the data from Kenyon et al. (1995). Our model broadly captured some of the seasonal variation in seagrass growth, including decreases during the austral winter months.

We based our modelling of the response of seagrass to cyclones on the Poiner et al. (1993) study which showed that cyclones such as 'Sandy' in 1985 had a severe effect on seagrass. Our model

similarly predicts a severe effect on seagrass (Figure A88) but additional data and analyses may be needed to accurately replicate the subsequent recovery speed.

As another informal check of our seagrass modelling, we plotted our model results to see whether they adequately matched trends expected based on ecological observations. For example, Rob Kenyon (pers comm) noted that there are no seagrass immediately offshore from the Roper River mouth (probably in an arc about 30 km probably due to turbidity/tidal suspension), but heaps outside that arc along the shallow coastline, as well as directly offshore from the Limmen Bight River mouth but more away from the mouth. Our model successfully simulated substantial decreases in seagrass relative biomass in response to large flows (Figure A87, see also Figures 61-64), and was therefore consistent with the expectation that widespread turbidity as a result of large flows has a negative impact on seagrass. Although there may be some benefit from nutrient inputs in the plume, turbidity would likely override in the short term (Kenyon, pers comm). Due to limited data, our model assumes constant nutrient inputs but this could be extended in future work.

To evaluate inter-annual changes in seagrass, we used data from the Karumba (model region 4) September long-term seagrass monitoring available from 1994 to 2019 (Scott and Rasheed 2021). Seagrasses have been monitored annually in the Port of Karumba since 1994. Annual monitoring is conducted at Alligator Bank between the Norman and Bynoe Rivers to assess changes in biomass (density), distribution (area), species composition, and reproductive capacity (seed bank, fruits and flowers) (Scott and Rasheed 2021). This seagrass meadow comprises two seagrass species: *Halophila ovalis* and *H. uninervis*. The authors used a visual estimate of biomass technique to measure above-ground seagrass biomass. We read the biomass measurements off Figure 7 of Scott and Rasheed (2021) and compared this with our model-estimated relative seagrass biomass trajectory (combined species and average over weeks 35 to 37) for our model region 4 and from 1994 to 2019 (Figure A89). The model does not accurately replicate the observed trends but nor was the model formally fitted to these data because they are from a sub-area of a single model region. Future work could refine seagrass modelling to more closely match available seagrass biomass (and area) estimates, particularly if these are available at larger scales.

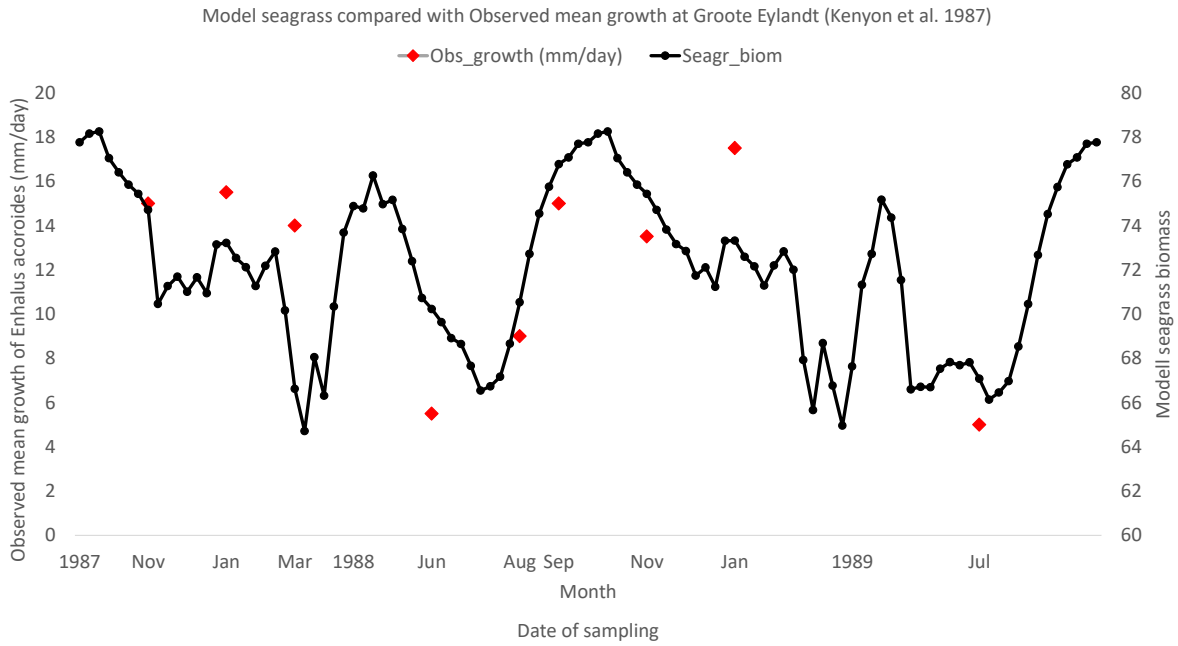


Figure A86. Validation of seagrass model component through comparison with available seagrass data from Kenyon et al. (1997a) for the same region (region 7) and time period (November 1987 to July 1989).

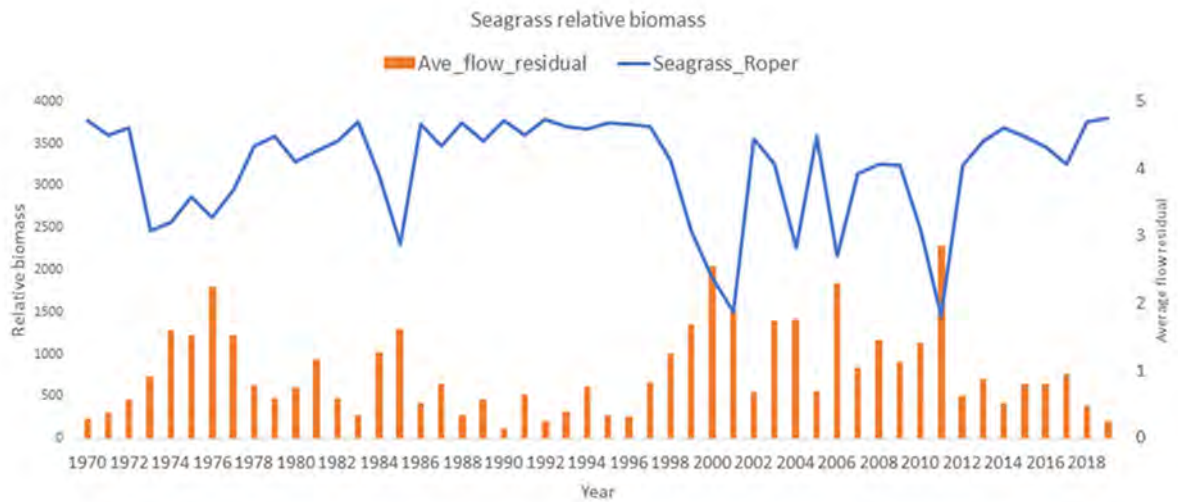


Figure A87. Comparison of model-estimated seagrass (annual average) in model region 7 for years 1970 to 2019 shown together with average flow residuals for the region.

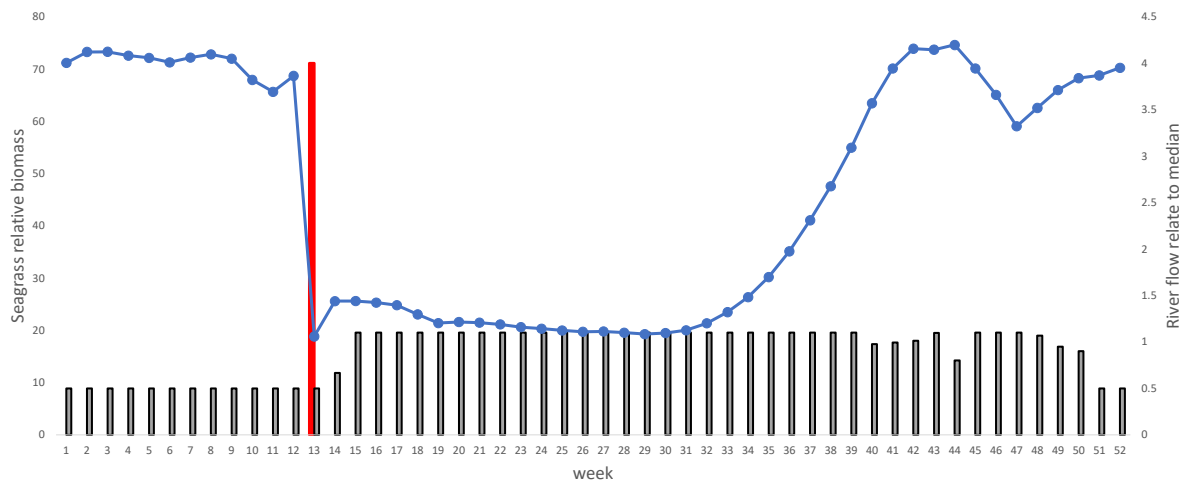


Figure A88. Model-estimated seagrass (annual average) in model region 7 shown for year 1985 which corresponds to Cyclone Sandy which is indicated by red vertical bar and has intensity set at 4 in the model. The plot also shows the standardised flow residuals for the Roper River in 1985.

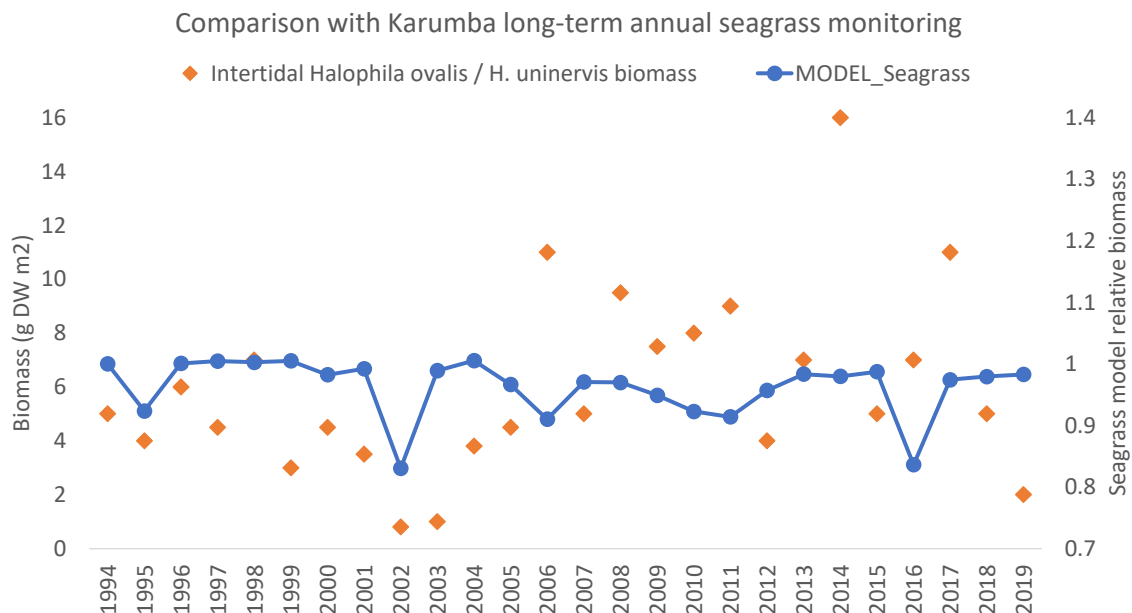


Figure A89. Comparison of model-estimated seagrass (combined species and average over weeks 35 to 37) biomass with Karumba (model region 4) September long-term seagrass monitoring from 1994 to 2019.

Mangrove Modelling

Mangrove forests are a key nursery area for many species of fish and crustaceans (Vance et al. 1990, Vance et al. 2002), provide refuges from predators (Vance et al. 1996) and provide important food sources for the rest of the ecosystem (Robertson and Alongi 1992, Robertson and Blaber 1993). Mangroves provide essential ecosystem services but are at risk as they have been declining worldwide with losses estimated at around 30-50% over the last century (Duke et al. 2007, Friess et al. 2019). The significant role of carbon sequestration in mangrove systems contributing to climate change mitigation has also recently been recognised (McLeod et al. 2011, Wylie et al. 2016) and approaches have been

developed to support a blue carbon accounting method for Australia (Kelleway et al. 2020, Lovelock et al. 2022). We therefore included this vital habitat in our MICE, recognising that there are few suitable data to validate our model but that it would nonetheless be useful to make a start at developing a large-scale model capturing the key drivers of mangrove dynamics.

Similar simplifying assumptions are made when modelling mangroves as for seagrass. The model aims to capture relative changes in mangrove biomass in the coastal and river margins due to extreme changes in physical drivers. Hence to simplify the model, we assume constant nutrient and light levels for all regions. Relative changes in mangrove biomass are modelled as a function of changes in other key drivers, namely flows, air temperature, solar exposure, minimum sea level and cyclone damage. The dependence of mangroves on salinity and water availability is modelled by assuming that the growth rate declines (relative to the maximum rate) when the salinity is too high (low flows used as a proxy) or water availability too low (flow, rainfall and sea level used as proxies).

Across large parts of Australia over recent years (2011-2017), extensive areas of habitat-forming coastal species have retreated due to the impacts of a warming climate: approximately half of Australia's littoral habitats have suffered, including coral reefs, mangroves, seagrass and kelp (Babcock et al. 2019). Other regions such as the Caribbean also experienced severe coral bleaching due to heat-stress events which were associated with the El Niño (with a lag of 6-12 months) (Muñiz-Castillo et al. 2019). In late 2015-2016, there was an unprecedented dieback of mangroves along approximately 1000km of coastline to the east of the JBG, in the Gulf of Carpentaria (Duke et al. 2017). The dieback was attributed to extended arid, hot conditions, low rainfall and a temporary fall in sea level, with Duke et al. (2017) also noting that the Southern Oscillation Index (SOI) is a notable correlate of these weather and sea level variables. Though not as extensive, the same conditions were measured and causality attributed when they simultaneously impacted mangroves in the Coral Bay region in the dry tropics of Western Australia; soil salinity and a lack of freshwater causing dieback (Lovelock et al. 2017b).

As explained in Appendix 6, SOI and sea level are correlated (White et al. 2014), hence given we have actual historical sea level data, we not include SOI, but instead capture the documented influences of El Niño on mangroves (Duke et al. 2017, Duke et al. 2021) by linking growth with sea level, flow and temperature changes.

Duke et al. (2019) confirmed that vegetative cover of mangrove-saltmarsh tidal wetlands increases with increasing average annual rainfall. Periodic changes in rainfall trends (and hence flow) trends can result in encroachment or dieback of mangroves. Our mesoscale mangrove community model component therefore assumes that the carrying capacity K of the mangroves in each model region varies linearly as a function of average annual flow. Asbridge et al (2016) showed seaward expansion of mangroves is due to prolonged inundation of low-lying coastal zone from tidal flows and floodwaters which pulse sediments to nearshore. Persistent freshwater inundation on the landward margins during the wet season made conditions more suitable for mangroves by reducing salinity and allowing the forests to compete with saltmarshes and expand landwards.

The MICE assumes the relative changes in mangrove community growth are influenced by flow, solar exposure, cyclones, air temperature (AT used instead of SST) and minimum sea level; we assume constant nutrients in base model given challenges of parameterising this aspect. There were limited old data available that could be used to help validate the model. An additional model sensitivity test is shown in Appendix 20 to investigate the influence on model results of not linking AT and mangrove growth.

Below follow the equations used to describe the mangrove model group.

The discrete updating equation for the biomass of mangroves $B_{a,y,w}^{mangr}$ in area a and week w of year y is:

$$B_{a,y,w+1}^{mangr} = B_{a,y,w}^{mangr} + G_{a,y,w}^{mangr} B_{a,y,w}^{mangr} \cdot \left(1 - B_{a,y,w}^{mangr} / K_{a,y}^{mangr}\right) - M_{a,y,w}^{mangr} B_{a,y,w}^{mangr}$$

where

$K_{a,y}^{mangr}$ is the time-varying seagrass carrying capacity in year y and area a , which is computed by multiplying the base carrying capacity value by the flow anomaly averaged over the past year (as a proxy for average annual rainfall), and with a constraint added that the carrying capacity can't vary by more than 10% up or down (sensitivity to other values can also be tested);

$M_{a,y,w}^{mangr}$ is the total mangrove mortality rate in area a and week w of year y computed using a base fixed mortality rate M_{base}^{mangr} and an additional mortality rate $M_{a,y,w}^{cyclone}$ (as above, product of cyclone category $Cycl_{a,y,w}^{category}$ and impact score $Cycl_{a,y,w}^{impact}$ with constant of proportionality ξ^{mangr}) as well as mortality term that is non-zero when sea level (SL) is below a minimum level :

$$\begin{aligned} M_{a,y,w}^{mangr} &= M_{base}^{mangr} + M_{a,y,w}^{cyclone,mangr} + M_{y,w}^{sealevel} \\ M_{a,y,w}^{cyclone,mangr} &= \xi^{mangr} \cdot Cycl_{a,y,w}^{category} \cdot Cycl_{a,y,w}^{impact} \\ M_{y,w}^{sealevel} &= 0.5 \text{ if } SL < 0.1 \text{ else } M_{y,w}^{sealevel} = 0 \end{aligned}$$

$G_{a,y,w}^{mangr}$ is the mangrove growth rate in area a and week w of year y computed as:

$$G_{a,y,w}^{mangr} = \mu^{mangr} \cdot \delta_{a,y,w}^{AT} \cdot flow_{a,y,w}^{anom} \cdot solar_{a,y,w}$$

Where μ^{mangr} is a fixed constant that is input;

$\delta_{a,y,w}^{AT}$ is a growth modifier based on Air Temperature (AT) in area a and week w of year y

$flow_{a,y,w}^{anom}$ is a growth modifier based on the flow anomaly in area a and week w of year y , with lower and upper values capped at 0.9 and 1.1 respectively

$solar_{a,y,w}$ is a growth modifier based on solar exposure in area a and week w of year y

Comparing Mangrove Model outputs with data

We found few suitable data sets to validate our mangrove model component, and future work will seek to improve our initial modelling. However, Conacher et al. (1996) conducted a study of the three main mangrove communities in the Embley River Estuary from March 1993 to March 1994. The communities in the Embley River Estuary included *Rhizophora* forest in the main river channel and a small creek, while other small creeks contained *Ceriops* forest and *Avicennia* forest (Conacher et al. 1996). During the Conacher et al. (1996) study period, the wet season was shorter than usual (December to March compared with November to May in some years) and rainfall was below average which resulted in environmental stress to the mangrove species. They used the rate of stipule fall of *Rhizophora* and *Ceriops* to estimate the number of leaves produced, and hence here we consider this a rough proxy of mangrove growth rate, and also compare their observations (after reading off their

Figure 3) with the modelled mangrove biomass (Figure A90). The model represents a combined species group of mangroves whereas there were differences in the observations for *Rhizophora* and *Ceriops*, due also to greater sustained inundation of the *Rhizophora* forest. The comparison between model outputs and observations is not straightforward but the model does capture some of the variation in mangrove growth rates. The variability is driven largely by changes in river flow. Future work can seek to improve validation of the mangrove model as this is considered uncertain. For this reason the MICE ensemble includes model versions with different parameter settings (such as for growth rate) as well as different assumptions, such as assuming different intensity of impact of cyclones on mangroves, using a different growth production function (Pella-Tomlinson instead of Schaefer production function (Hilborn and Walters 2013)) and assuming flow does not influence growth rate (as is the case in the base-case and other model versions) but does influence carrying capacity.

The Asbridge et al. (2016) study suggests that the mangrove community has been accreting seaward over recent decades which is attributed to the combined effects of sea level rise and prolonged periods of tidal and freshwater inundation on coastal lowlands. It therefore seems reasonable to model mangrove growth as dependent on river flows, noting also that any reduction in flows would reduce the sedimentation that the mangroves are colonising.

Figure A90 to Figure A92 provide examples of the changes in modelled mangroves over time and in response to changing physical drivers. These plots also show the relatively greater sensitivity of seagrass than mangroves to cyclones. Finally, Figure A91 shows the model mangrove trajectory together with decreases in sea level observed during late 2015 to 2016. The model estimates that the average 2016 mangrove biomass is about 25% less than the 2015 starting level for this local region. The modelled decline in mangrove biomass over this period is consistent with the Duke et al. (2017) observation of a severe and widespread decline in mangroves: these authors found the dieback affected more than 7400ha or 6% of mangrove vegetation between the Roper River and Karumba regions (regions 4-7 in our model). However further model validation is required to increase confidence in the extent of the modelled decline.

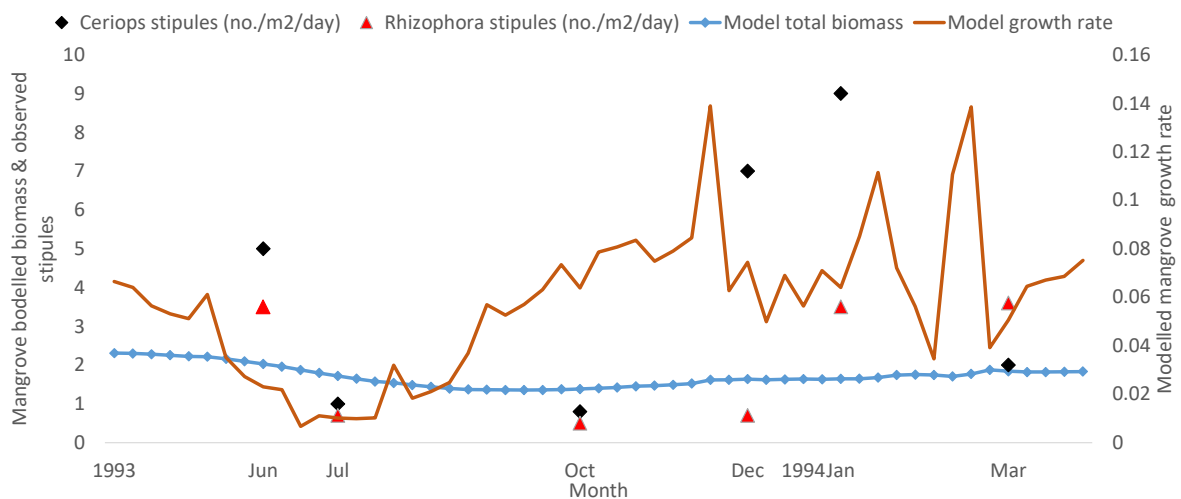


Figure A90. Example of preliminary validation of mangrove model component through comparison with available mangrove data from Conacher et al. (1996) for the same region and time period.

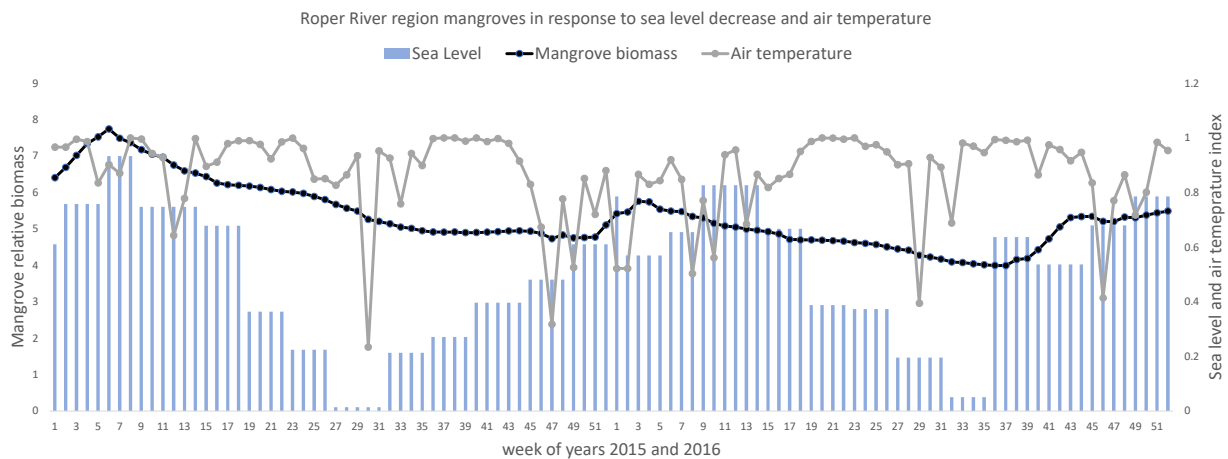


Figure A91. Example of model weekly mangrove relative biomass in region 7 (Roper River region) over the period 2015 to 2016 shown together with changes in sea level and air temperature (see Appendix 6 for physical variable descriptions and sources).

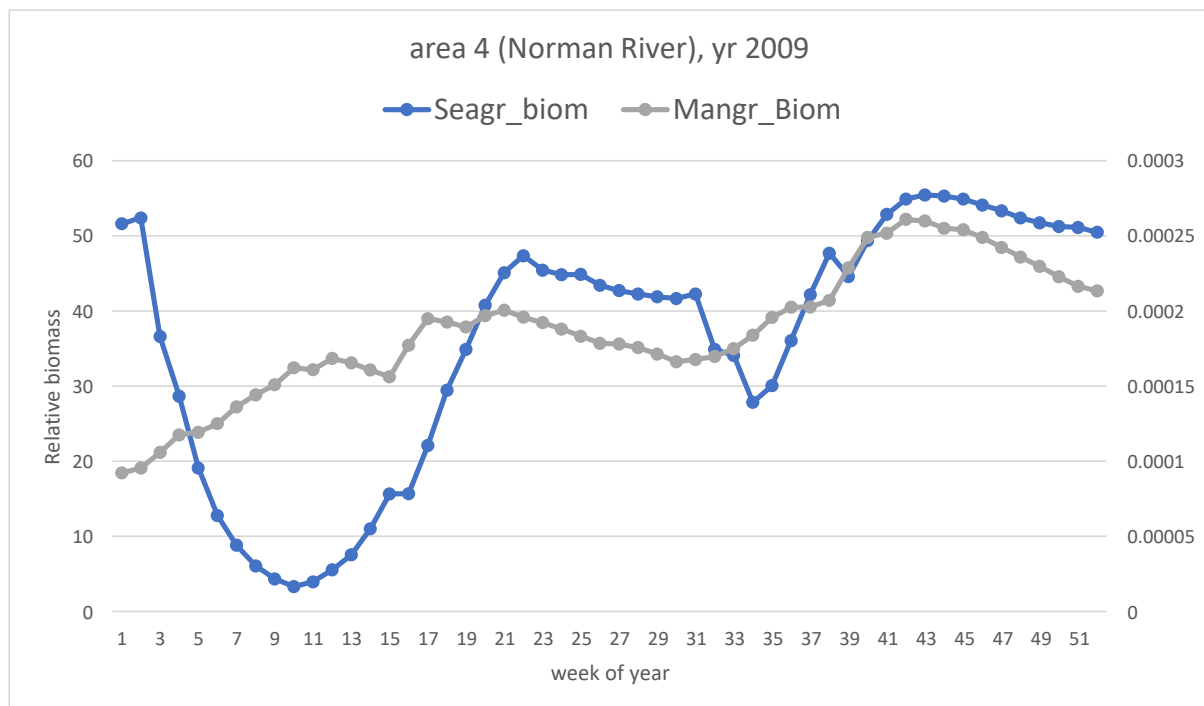


Figure A92. Example of MICE estimated relative biomass trajectories for seagrass and mangroves in model region 4 during selected year 2009.

Modelling relative changes in seagrass and mangroves

Representing spatial and temporal variation in the habitat-forming seagrass and mangrove community groups was challenging because both these groups include several different species, all of which have different life history characteristics as well as being adapted to slightly different environmental conditions. We were thus only able to represent the average features of these groups and use this to simulate approximate changes in these groups in response to changes in physical drivers. We acknowledge also that there is considerable spatial variation in seagrass and mangrove abundance and

composition. For example, there are much more extensive seagrass beds along the western versus the eastern GoC. We drew on the literature and extensive expert knowledge of team member Rob Kenyon (see summary below) to broadly characterise the different GoC spatial sub-regions to inform our modelling. Although the beach coastlines between the river estuaries in some of the model spatial regions are wave-washed and relatively turbid and so do not support seagrass communities, we nonetheless model relative changes in seagrass and mangroves in all model regions, irrespective of whether seagrass density is sparse or dense. Our model therefore does not provide any absolute estimates of seagrass or mangrove abundance, only changes relative to a starting 1970 base level.

As shown in Figure A92, our model settings mean that seagrass are modelled as more variable than mangroves, but as per the model equations, the two groups are driven by different physical drivers. We note also that in the tropics, seagrasses and macroalgae grow close to their thermal limits and may thus become increasingly stressed in future due to more frequent and longer sublethal and lethal temperatures (Koch et al. 2013). For these species to survive, they will have to up-regulate their stress-response systems (Koch et al. 2013).

Background Information Summary : Expert overview of GoC habitat groups

based on Rob Kenyon's field knowledge dating back to the 1980s

Region 1.

Seagrass

Rob has never undertaken field studies in the area apart from the Embley River estuary at Weipa. Within the Embley estuary, seagrass communities are prolific and are composed mostly of *Enhalus acoroides*, *Halophila ovalis* and *Halodule uninervis*. Seagrass density and shoot morphology data have been published for Embley River seagrass communities as part of CSIRO prawn ecology studies (juvenile tiger prawn habitat) (search Vance and Haywood - authors). TropWater (JCU) survey the littoral zones of Albatross Bay regularly and have documented similar species in inshore shallow habitats elsewhere within the Bay.

North of Albatross Bay, the region is more 'oceanic-influence' with sandy beaches, clearer waters and probably coral in places. JCU researchers described "Fringing reef areas, open sandy beaches and less turbid water north of Aurukun".

South of Albatross Bay, similar habitats as per to the north also probably exist. A new port was constructed at Pera Head south of Albatross Bay and coral patches were common (from an EIS report).

Mangroves

Mangroves are prolific within the Embley River estuary and within the Mission and Pine River estuaries. They probably form the majority of the coastal vegetation along a proportion of the Albatross Bay coastline. They would be prolific in the Wenlock River estuary (Port Musgrave) further north

To the north and south along the open coastline, sandy beaches dominate.

Coral reefs

Coral fringing reefs and patch-reefs are probably common in this Region in the subtidal shallows.

JCU via helicopter observed - "Fringing reef areas, open sandy beaches and less turbid water was found north of Aurukun."

Region 2.

Seagrass

Rob has never undertaken field studies in the area apart from the Mitchell River 'delta' estuary location with a focus on juvenile banana prawns. TropWater (JCU) surveyed the littoral zones of the western Cape York coast in the early 2000s and documented seagrasses in the estuaries of rivers along the coast. The Archer River estuary (Arukun) and the Kirke River lakes (barrier estuary?) were purported to support *Halophila ovalis* and *Halodule uninervis* communities. TropWater may have documented a patch of seagrass just north of the Mitchell River 'delta', though the scale of the map makes it difficult to determine. Comments over the years and observations by JCU suggest that the beach coastlines between the river estuaries are wave-washed, relatively turbid waters that do not support seagrass communities.

Mangroves

Mangroves probably are sparse and patchy along most of the exposed beach coastline of western Cape York. However, they are prolific within the river estuaries and wetland/saltpan fringes that are extensive in some areas. Near the coastline, many rivers bifurcate and produce a multitude of channels and gutters that support mangroves. Mangroves dominate along the exposed coastline adjacent to these 'delta-estuaries' (e.g. the Mitchell River complex) as beach habitats are not dominant. Mangroves may form a fringe along the exposed GOC coastline in more 'sheltered locations'.

Coral reefs

I doubt there are corals in this region. There may be small clumps on sparse rocky areas here and there.

Region 3.

Seagrass

Rob has never undertaken field studies in the area apart from the Gilbert River estuary with a focus on juvenile banana prawns. TropWater (JCU) surveyed the littoral zones of the western Cape York coast in the early 2000s and documented seagrasses in the littoral zone along the coast in the southern portion of Region 3 (*Halodule uninervis* and *Halophila ovalis* community). However, the seagrasses cover a small proportion of the Region's coastline. Comments over the years and observations by JCU suggest that the beach coastlines are wave-washed, relatively turbid waters that do not support seagrass communities.

Mangroves

Mangroves are prolific within the river estuaries and wetland/saltpan fringes that are extensive in some areas. Near the coastline, many rivers bifurcate and produce a multitude of channels and gutters that support mangroves. Mangroves dominate along the exposed coastline adjacent to these 'delta-estuaries' and low-relief coastal extents. In the southern Portion of Region Three, beach habitats are less dominant and mangroves may form a fringe

along the exposed GOC coastline. They may form robust stands in front of sand beaches along portions of the exposed beach coastline of western Cape York.

Coral reefs

I doubt there are corals in this region.

Region 4.

Seagrass

Rob has undertaken field studies in the area within the Norman River estuary with a focus on juvenile banana prawns.

TropWater (JCU) surveyed the littoral zones of the western Cape York coast in the early 2000s and documented seagrasses in the littoral zone along the coast of Region 4. As well, they regularly survey the seagrass community on the vicinity of the Norman River as part of the Port of Karumba' dredging EIS studies. There are extensive *Halophila ovalis* and *Halodule uninervis* communities to the west and east of the Norman River estuary on the intertidal and shallow subtidal sea-flats. However, the seagrasses are not as species rich, nor do they extend to the depths with species zonation as do the seagrass communities of the western GoC.

Mangroves

The 'terrestrial' coastline in Region 4 is dominated by saltflats intersected by mangrove creeks. Mangroves are prolific within the river estuaries and wetland/saltpan fringes that are very extensive in all areas. Near the coastline, many rivers bifurcate and produce a multitude of channels and gutters that support mangroves. Mangroves dominate along the exposed coastline adjacent to the low-relief coastal extents. Where evident, beach habitats are less dominant and mangroves may form a fringe along the exposed GOC coastline. They may form robust stands in front of sand beaches along portions of the exposed beach coastline.

Coral reefs

There are no corals in this region.

Region 5.

Seagrass

Rob has undertaken field studies in the area within the Flinders River estuary with a focus on juvenile banana prawns.

TropWater (JCU) surveyed the littoral zones of the southern Gulf coast in the early 2000s and documented seagrasses in the littoral zone along the western portion of the coast of Region 5. They denote seagrass to the west of the Albert River, but detail is not provided. A guess would suggest sparse *Halophila ovalis* and *Halodule uninervis* communities on this coastline on the intertidal and shallow subtidal sea-flats.

In the 1980s, the seagrasses within the bays of Mornington Island were surveyed by Coles and Lee Long and seagrass was found in the bays of the eastern coastline. The seagrass

community is probably extensive within these shallow zones in the vicinity of Mornington Island and perhaps the other island in Region 5.

Mangroves

Large river estuaries dominate Region Five and mangroves are extensive within them. The 'terrestrial' coastline in Region 5 is dominated by saltflats intersected by mangrove creeks. Mangroves are prolific within the river estuaries and wetland/saltpan fringes that are very extensive and link many of the large estuaries in the area. Near the coastline, many rivers bifurcate and produce a multitude of channels and gutters that support mangroves. Mangroves dominate along the exposed coastline adjacent to the low-relief coastal extents. Where evident, beach habitats are less dominant and mangroves may form a fringe along the exposed GOC coastline.

Coral reefs

I doubt there are corals along the mainland coastline in this region. There probably are fringing or clumped corals along the north-east coast of Mornington Island. There may be small clumps on sparse rocky areas here and there around the other islands in Region Five. There are corals surrounding a few Islands north of Mornington Island (clear water habitat, e.g. Man-o-War Island).

Region 6.

Seagrass

Rob has undertaken extensive field studies in the Region. Seagrass surveys involved diving and quantitative sampling along the mainland coastline of Region 6.

CSIRO quantitatively surveyed the seagrass community along the GoC coastline in the 1980s. Seagrass species and zonation with depth were identified. Seagrass beds were extensive from the intertidal to about 3-5 km offshore (~5 m deep). Due to the region boundary half way between Mornington and Sir Edward Pellew Islands, roughly 50% of '56 km² of seagrass along 91 km of coastline' was identified (to low-tide depths about 5 m). *Syringodium isoetifolium*, *Cymodocea serrulata*, *Enhalus acoroides*, *Halophila ovalis*, *Halophila spinulosa* and *Halodule uninervis* were common within the seagrass community.

TropWater (JCU) surveyed the littoral zones of the south-western Gulf coast in the early 2000s and documented seagrasses in the littoral zone along the portions of the coast of Region 6. They probably found *Halophila ovalis* and *Halodule uninervis* communities in the intertidal and shallow subtidal zones observed by helicopter. Their seagrass survey was not as comprehensive as the 1980's CSIRO survey. They do not document the extent of the depths of the seagrass community or the species zonation with depth as found in the western GoC.

TropWater found seagrasses on the mainland coastline in the lee of Mornington Island, an area not surveyed by CSIRO in the 1980s. There probably are sparse seagrasses (*Halophila ovalis* and *Halodule uninervis*) in the shallows around the small islands in the lee of Mornington Island.

Mangroves

In the lee of Mornington Island, the 'terrestrial' coastline in Region 6 is dominated by saltflats intersected by mangrove creeks. Mangroves are prolific within the creek estuaries and wetland/saltpan fringes that are extensive in the area. Mangroves dominate along the exposed coastline adjacent to the low-relief coastal extents. Further west, beach habitats

become more dominant and mangroves less prolific. They may form a fringe along the less-exposed GOC coastline. Several large river estuaries dominate the mainland coastline of Region Six and mangroves are extensive within them. Near the coastline, many rivers bifurcate and produce a multitude of channels and gutters that support mangroves. Extensive saltflats support fringing mangroves adjacent to rivers behind low coastal dunes and swales.

Coral reefs

I doubt there are corals along the mainland coastline in this region, I never encountered corals. There probably are sparse fringing or clumped corals along the south-west coast of Mornington Island and the other small islands in its lee. There would be corals on 'bommies' in the deeper waters of the southern GoC within Region 6. But they would be decimated by Trawler anchoring over the years

Region 7.

Seagrass

Rob has undertaken extensive field studies in the Region. Seagrass surveys involved diving and quantitative sampling along the mainland coastline of Region 7.

CSIRO quantitatively surveyed the seagrass community along the GoC coastline in the 1980s. Seagrass species and zonation with depth were identified. Seagrass beds were extensive from the intertidal to about 3-7 km offshore (~5 m deep). The seagrass community of the western Gulf of Carpentaria has been described by CSIRO researchers. Seagrass density and shoot morphology data have been published for the seagrass community in the vicinity of the McArthur River Mine port facility west of the Sir Edward Pellew Islands (search Poiner and Kenyon - authors). Roughly 600 km² of seagrass along 290 km of coastline was identified (to low-tide depths about 5 m). *Syringodium isoetifolium*, *Cymodocea serrulata*, *Enhalus acoroides*, *Halophila ovalis*, *Halophila spinulosa* and *Halodule uninervis* were common within the seagrass community. In the lee of the Sir Edward Pellew Islands, seagrasses are prolific in the shallows.

TropWater (JCU) surveyed the littoral zones of the south-western Gulf coast in the early 2000s and documented seagrasses in the littoral zone along the portions of the coast of Region 7. They probably found *Halophila ovalis* and *Halodule uninervis* communities in the intertidal and shallow subtidal zones observed by helicopter. Their seagrass survey was not as comprehensive as the 1980's CSIRO survey. They do not document the extent of the depths of the seagrass community or the species zonation with depth as found in the western GoC.

The seagrass species listed by TropWater on the mainland coastline and in the lee of the Sir Edward Pellew Islands are the shallow intertidal seagrasses (*Halophila ovalis* and *Halodule uninervis*) easily accessed in the shallows by helicopter.

Mangroves

Large river estuaries dominate the Region Seven coastline and mangroves are extensive within their estuaries. Beach habitats become more dominant between the rivers and mangroves are less prolific on these coasts. However, behind the beach dunes and swales, low-lying landforms are dominated by saltflats for several kilometres inland and the flats are intersected by mangrove creeks. Mangroves are prolific within the river estuaries and

wetland/saltpan fringes that in places are extensive. In the lee of the Sir Edward Pellew Islands, mangroves are prolific at the land/sea interface and within the saltflat zones landward.

The Limmen Bight estuary is a mangrove haven.

North of the Roper River the coastline features beaches and some dune areas. Mangroves are confined to the river estuaries. Beaches dominate along the exposed coastline adjacent to the low-relief coastal extents. Where evident, beach habitats are less dominant and mangroves may form a fringe along the exposed GOC coastline.

Coral reefs

Corals are abundant in Region 7. Fringing reefs surround the Sir Edward Pellew Islands.

Patch reefs are common along the coastline to the west of the Sir Edward Pellew Islands, their extent has never been surveyed. Rob observed them in many places along this coastline; interspersed among the seagrass community and adjacent to coastal hard-substrate features. Patch corals were prolific around Maria Island, adjacent to the Limmen and Roper Rivers; and in shallow water habitats offshore from Numbulwar.

Region 8.

Seagrass

Rob has undertaken extensive field studies in the Region. Seagrass surveys involved diving and quantitative sampling along the mainland coastline of Region 8.

Seagrasses are abundant in the bays surrounding Groote Eylandt and within the smaller bays within Blue Mud Bay. Seagrass species and zonation with depth were identified. Expansive seagrass communities behind fringing coral reefs were identified. Roughly 110 km² of seagrass along 140 km of coastline was identified (to low-tide depths about 5 m).

Syringodium isoetifolium, *Cymodocea serrulata*, *Enhalus acoroides*, *Halophila ovalis*, *Halophila spinulosa* and *Halodule uninervis* were common within the seagrass community. Seagrass density and shoot morphology data have been published for Groote Eylandt and Blue Mud Bay seagrass communities as part of CSIRO prawn ecology studies (juvenile tiger prawn habitat) (search Loneragan and Kenyon - authors)

TropWater (JCU) surveyed the littoral zones of the western Gulf coast in the early 2000s and documented seagrasses in the littoral zone along the portions of the coast of Region 8. They found *Halophila ovalis* and *Halodule uninervis* communities in the intertidal and shallow subtidal zones observed by helicopter. They found *Enhalus acoroides* in Caledon Bay, north of Blue Mud Bay. Their seagrass survey was not as comprehensive and the 1980's CSIRO survey (e.g. they did not survey Groote Eylandt). They do not document the extent of the depths of the seagrass community or the species zonation with depth as found in the western GoC.

Mangroves

Beach habitats dominant the coastline of Region Eight, apart from Blue Mud Bay. Large river estuaries do not dominate Region 8, so the extent of mangroves is restricted. However, they are prolific within the embayments throughout the region. Mangroves are prolific within the river estuaries and sheltered portions of embayments.

Blue Mud Bay is a mangrove haven. Both the coastline and small estuaries within the Bay are dominated by mangroves

North of the Blue Mud Bay the coastline features beaches and extensive dune areas. Mangroves are sparse to absent (perhaps in Port Bradshaw).

Coral reefs

Corals are abundant in Region 8. Fringing reefs surround portions of the coastline of Groote Eylandt. Often a mangrove stand, a broad seagrass community and reef-edge of expansive fringing reefs are characteristic of embayment coasts on Groote Eylandt.

Patch reefs are common along the coastline of Groote Eylandt, their extent has never been surveyed. Rob observed them in many places along this coastline.

There are corals on 'bommies' in the deeper waters surrounding Groote Eylandt within Region 8. But they have been decimated by Trawler anchoring over the years

References

- Adey, W. H. 1998. Coral reefs: algal structured and mediated ecosystems in shallow, turbulent, alkaline waters. *Journal of Phycology* **34**:393-406.
- Asbridge, E., R. Lucas, C. Ticehurst, and P. Bunting. 2016. Mangrove response to environmental change in Australia's Gulf of Carpentaria. *Ecology and evolution* **6**:3523-3539.
- Babcock, R. C., R. H. Bustamante, E. A. Fulton, D. J. Fulton, M. D. E. Haywood, A. J. Hobday, R. Kenyon, R. J. Matear, E. E. Plagányi, A. J. Richardson, and M. A. Vanderklift. 2019. Severe Continental-Scale Impacts of Climate Change Are Happening Now: Extreme Climate Events Impact Marine Habitat Forming Communities Along 45% of Australia's Coast. *Frontiers in Marine Science* **6**.
- Blaber, S., D. Brewer, J. Salini, J. Kerr, and C. Conacher. 1992. Species composition and biomasses of fishes in tropical seagrasses at Groote Eylandt, northern Australia. *Estuarine, Coastal and Shelf Science* **35**:605-620.
- Brewer, D., S. Blaber, J. Salini, and M. Farmer. 1995. Feeding ecology of predatory fishes from Groote Eylandt in the Gulf of Carpentaria, Australia, with special reference to predation on penaeid prawns. *Estuarine, Coastal and Shelf Science* **40**:577-600.
- Carruthers, T., W. Dennison, B. Longstaff, M. Waycott, E. Abal, L. McKenzie, and W. Long. 2002. Seagrass habitats of northeast Australia: models of key processes and controls. *Bulletin of Marine Science* **71**:1153-1169.
- Collier, C. J., and M. Waycott. 2014. Temperature extremes reduce seagrass growth and induce mortality. *Marine Pollution Bulletin* **83**:483-490.
- Conacher, C., C. O'Brien, J. Horrocks, and R. Kenyon. 1996. Litter production and accumulation in stressed mangrove communities in the Embley River estuary, North-eastern Gulf of Carpentaria, Australia. *Marine and Freshwater Research* **47**:737-743.
- Dichmont, C. M., N. R. Loneragan, D. T. Brewer, and I. R. Poiner. 2007. Partnerships towards sustainable use of Australia's Northern Prawn Fishery. *Fisheries Management: Progress Toward Sustainability*.
- Duarte, C. M., and J. Cebrián. 1996. The fate of marine autotrophic production. *Limnology and Oceanography* **41**:1758-1766.
- Duke, N. C., C. Field, J. R. Mackenzie, J.-O. Meynecke, and A. L. Wood. 2019. Rainfall and its possible hysteresis effect on the proportional cover of tropical tidal-wetland mangroves and saltmarsh-saltpans. *Marine and Freshwater Research* **70**:1047-1055.

- Duke, N. C., J. M. Kovacs, A. D. Griffiths, L. Preece, D. J. Hill, P. Van Oosterzee, J. Mackenzie, H. S. Morning, and D. Burrows. 2017. Large-scale dieback of mangroves in Australia's Gulf of Carpentaria: a severe ecosystem response, coincidental with an unusually extreme weather event. *Marine and Freshwater Research* **68**:1816-1829.
- Duke, N. C., J.-O. Meynecke, S. Dittmann, A. M. Ellison, K. Anger, U. Berger, S. Cannicci, K. Diele, K. C. Ewel, and C. D. Field. 2007. A world without mangroves? *Science* **317**:41-42.
- Friess, D. A., K. Rogers, C. E. Lovelock, K. W. Krauss, S. E. Hamilton, S. Y. Lee, R. Lucas, J. Primavera, A. Rajkaran, and S. Shi. 2019. The state of the world's mangrove forests: past, present, and future. *Annual Review of Environment and Resources* **44**:89-115.
- Haywood, M., D. Vance, and N. Loneragan. 1995. Seagrass and algal beds as nursery habitats for tiger prawns (*Penaeus semisulcatus* and *P. esculentus*) in a tropical Australian estuary. *Marine Biology* **122**:213-223.
- Haywood, M. D., and R. A. Kenyon. 2009. Habitat shifts by decapods—an example of connectivity across tropical coastal ecosystems. Pages 229-269 *Ecological connectivity among tropical coastal ecosystems*. Springer.
- Kenyon, R., C. Conacher, and I. Poiner. 1997a. Seasonal growth and reproduction of *Enhalus acoroides* (Lf) Royle in a shallow bay in the western Gulf of Carpentaria, Australia. *Marine and Freshwater Research* **48**:335-342.
- Kenyon, R., N. Loneragan, and J. Hughes. 1995. Habitat type and light affect sheltering behaviour of juvenile tiger prawns (*Penaeus esculentus* Haswell) and success rates of their fish predators. *Journal of Experimental Marine Biology and Ecology* **192**:87-105.
- Kenyon, R., N. Loneragan, J. Hughes, and D. Staples. 1997b. Habitat Type Influences the Microhabitat Preference of Juvenile Tiger Prawns (*Penaeus esculentus* Haswell and *Penaeus semisulcatus* De Haan). *Estuarine, Coastal and Shelf Science* **45**:393-403.
- Koch, M., G. Bowes, C. Ross, and X. H. Zhang. 2013. Climate change and ocean acidification effects on seagrasses and marine macroalgae. *Global change biology* **19**:103-132.
- Lee, K.-S., S. R. Park, and Y. K. Kim. 2007. Effects of irradiance, temperature, and nutrients on growth dynamics of seagrasses: a review. *Journal of Experimental Marine Biology and Ecology* **350**:144-175.
- Liu, H., and N. Loneragan. 1997. Size and time of day affect the response of postlarvae and early juvenile grooved tiger prawns *Penaeus semisulcatus* De Haan (Decapoda: Penaeidae) to natural and artificial seagrass in the laboratory. *Journal of Experimental Marine Biology and Ecology* **211**:263-277.
- Loneragan, N. R., R. A. Kenyon, D. J. Staples, I. R. Poiner, and C. A. Conacher. 1998. The influence of seagrass type on the distribution and abundance of postlarval and juvenile tiger prawns (*Penaeus esculentus* and *P. semisulcatus*) in the western Gulf of Carpentaria, Australia. *Journal of Experimental Marine Biology and Ecology* **228**:175-195.
- Lovelock, C. E., M. F. Adame, D. W. Butler, J. J. Kelleway, S. Dittmann, B. Fest, K. J. King, P. I. Macreadie, K. Mitchell, and M. Newnham. 2022. Modeled approaches to estimating blue carbon accumulation with mangrove restoration to support a blue carbon accounting method for Australia. *Limnology and Oceanography*.
- Lovelock, C. E., I. C. Feller, R. Reef, S. Hickey, and M. C. Ball. 2017. Mangrove dieback during fluctuating sea levels. *Scientific Reports* **7**:1680.
- Macdonald, R. K. 2015. Turbidity and light attenuation in coastal waters of the Great Barrier Reef. James Cook University.
- McLeod, E., G. L. Chmura, S. Bouillon, R. Salm, M. Björk, C. M. Duarte, C. E. Lovelock, W. H. Schlesinger, and B. R. Silliman. 2011. A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO₂. *Frontiers in Ecology and the Environment* **9**:552-560.
- Muñiz-Castillo, A. I., A. Rivera-Sosa, I. Chollett, C. M. Eakin, L. Andrade-Gómez, M. McField, and J. E. Arias-González. 2019. Three decades of heat stress exposure in Caribbean coral reefs: a new regional delineation to enhance conservation. *Scientific Reports* **9**:11013.
- Poiner, I., C. Conacher, N. Loneragan, and I. Sonters. 1993. Effects of cyclones on seagrass communities and penaeid prawn stocks of the Gulf of Carpentaria.

- Ralph, P. J., M. J. Durako, S. Enríquez, C. J. Collier, and M. A. Doblin. 2007. Impact of light limitation on seagrasses. *Journal of Experimental Marine Biology and Ecology* **350**:176-193.
- Robertson, A., and S. Blaber. 1993. Plankton, epibenthos and fish communities. *Coastal and estuarine studies*:173-173.
- Robertson, A. I., and D. M. Alongi. 1992. Tropical mangrove ecosystems.
- Scott, A., and M. Rasheed. 2021. Port of Karumba Long-term Annual Seagrass Monitoring 2020, Centre for Tropical Water & Aquatic Ecosystem Research Publication Number 21/05, James Cook University, Cairns, 28pp.
- Short, F. T., R. G. Coles, and C. Pergent-Martini. 2001. Global seagrass distribution. *Global seagrass research methods* **5**:30.
- Vance, D., M. Haywood, D. Heales, R. Kenyon, N. Loneragan, and R. Pendrey. 1996. How far do prawns and fish move into mangroves? Distribution of juvenile banana prawns *Penaeus merguensis* and fish in a tropical mangrove forest in northern Australia. *Marine Ecology Progress Series* **131**:115-124.
- Vance, D., M. Haywood, D. Heales, R. Kenyon, N. Loneragan, and R. Pendrey. 2002. Distribution of juvenile penaeid prawns in mangrove forests in a tropical Australian estuary, with particular reference to *Penaeus merguensis*. *Marine Ecology Progress Series* **228**:165-177.
- Vance, D., M. Haywood, and D. Staples. 1990. Use of a mangrove estuary as a nursery area by postlarval and juvenile banana prawns, *Penaeus merguensis* de Man, in northern Australia. *Estuarine, Coastal and Shelf Science* **31**:689-701.
- Wylie, L., A. E. Sutton-Grier, and A. Moore. 2016. Keys to successful blue carbon projects: Lessons learned from global case studies. *Marine Policy* **65**:76-84.

Appendix 16 – MICE equations, variable definitions and input values

Table A24. Model equations for common banana prawns (*psp*=prawns), barramundi (*sp*=barra), mud crabs (*sp*=mudcr) and sawfish (*sp*=saw) (definitions of variables and parameters are shown in Tables 2 & 3) and the log-likelihood function.

Model section	Equation	No.
Weekly Numbers-at-age		
Weekly updates (prawns)	$N_{r,y,w+1}^{psp} = N_{r,y,w}^{psp} e^{-M_{r,y,w}^{psp}} - C_{r,y,w}^{psp} + R_{r,y,w+1}^{psp} \quad w = 1 \text{ to } 51$ $N_{r,y+1,1}^{psp} = N_{r,y,w}^{psp} e^{-M_{r,y,w}^{psp}} - C_{r,y,w}^{psp} + R_{r,y+1,1}^{psp} \quad w = 52$	1a
Monthly Numbers-at-age		
Annual recruits (saw)	$N_{r,y,m,1}^{sp,sex} = R_{r,y,m}^{sp} \quad a = 1; m = 1$ $R_{r,y,m}^{sp} = 0 \quad m > 1$	1b
Monthly recruits (barra, mudcr, saw)	$N_{r,y+1,1,1}^{sp,sex} = R_{r,y+1,1}^{sp} \quad a = 1; m = 1$ $N_{r,y,m+1,1}^{sp,sex} = N_{r,y,m,1}^{sp,sex} e^{-M_{r,y,m,1}^{sp}} - C_{r,y,m,1}^{sp,sex} + R_{r,y,m+1}^{sp} \quad a = 1; m = 2 \text{ to } 11$	1c
Monthly updates (barra, mudcr, saw)	$N_{r,y,m+1,a}^{sp,sex} = N_{r,y,m,a}^{sp,sex} e^{-M_{r,y,m,a}^{sp}} - C_{r,y,m,a}^{sp,sex} \quad 2 \leq a < z; m = 1 \text{ to } 11$ $N_{r,y+1,1,a+1}^{sp,sex} = N_{r,y,m,a}^{sp,sex} e^{-M_{r,y,m,a}^{sp}} - C_{r,y,m,a}^{sp,sex} \quad m = 12$ <p>Noting this is only sex disaggregated for mudcr, and for saw represents fishery interactions (assuming 100% mortality) rather than catch <i>per se</i></p>	1d
Plus group (barra, mudcr, saw)	$N_{r,y+1,1,z}^{sp,sex} = \left(N_{r,y,12,z}^{sp,sex} e^{-M_{r,y,m,z}^{sp}} - C_{r,y,12,z}^{sp,sex} \right) + \left(N_{r,y,12,z-1}^{sp,sex} e^{-M_{r,y,m,z-1}^{sp}} - C_{r,y,12,z-1}^{sp,sex} \right) \quad a = z; m = 12$	1e
Mortality terms		
Weekly mortality rate (prawns)	$M_{r,y,w}^{psp} = M_{base}^{psp} \quad \text{for Model versions } < 5$ $M_{r,y,w}^{psp} = M_{base}^{psp} / prod_y \quad \text{for Model version 5 with boosted productivity effect, where}$ $prod_y = \prod_{r=2}^5 \sum_w scour_{r,w} \cdot B_{r,y,w}^{Mb} / prod \quad \text{with } scour=0.5 \text{ if } \sigma_{r,y,w}^{flow_sp} > 3 \text{ else } scour=0.1; \text{ average } prod \text{ computed in model}$	1f
Monthly mortality rate (mudcr)	$M_{r,y,m,a}^{mudcr} = M_{base}^{mudcr} \quad \text{for mudcr except for temperature-dependent form when } r = 7 \text{ and } temp_{r,y,m} > temp^{opt} :$ $M_{r,y,m,a}^{mudcr} = M_{base}^{mudcr} \cdot M_{r,y,m}^{mudcr,temp} \quad \text{where } M_{r,y,m}^{mudcr,temp} = \tau^{temp} \cdot temp_{r,y,m} + c^{temp}$	1g

<p>Monthly mortality rate (barra, saw)</p>	$M_{r,y,m,a}^{sp} = \left[M_{base}^{sp} + \frac{M_{par}^{sp}}{a+1} \right] / \sigma_{r,y,m}^{flow_sp} \quad \text{for barra with } a \leq 5 \text{ and flow multiplier bounded } 0.5 < \sigma_{r,y,m}^{flow_sp} < 2$ $M_{r,y,m,a}^{sp} = M_{base}^{sp} + \frac{M_{par}^{sp}}{a+1} \quad \text{for barra with } a > 5 \text{ and for saw with } a \geq 5$ $M_{r,y,m,a}^{sp} = \left[M_{base}^{sp} + \frac{M_{par}^{sp}}{a+1} \right] \cdot M_{pools}^{saw} \quad \text{for saw with } a < 5 \text{ and if flow multiplier } \sigma_{r,y,1}^{flow_sp} < \text{thres}_{pools}^{saw} \text{ else as above}$	<p>1h</p>
<p>Weekly Recruitment</p>		
<p>Prawn spawning biomass</p>	$B_{r,y,w}^{spn,psp} = f_w^{psp} \cdot w_w^{psp} \cdot N_{r,y,w}^{psp}$	<p>2a</p>
<p>Prawn recruitment</p>	$R_{r,y,w+1}^{psp} = \rho_r^{psp} \frac{\alpha^{psp} B_{r,y-1,w+27}^{psp}}{\beta^{psp} + B_{r,y-1,w+27}^{psp}} \sigma_{r,y,w}^{flow_psp} \quad \text{for } 1 \leq w \leq 25$ $R_{r,y,w+1}^{psp} = \rho_r^{psp} \frac{\alpha^{psp} B_{r,y,w-25}^{psp}}{\beta^{psp} + B_{r,y,w-25}^{psp}} \sigma_{r,y,w}^{flow_psp} \quad \text{for } 26 \leq w < 52$ $R_{r,y,w}^{psp} = \rho_r^{psp} \frac{\alpha^{psp} B_{r,y,w-26}^{psp}}{\beta^{psp} + B_{r,y,w-26}^{psp}} \sigma_{r,y,w}^{flow_psp} \quad \text{for } w = 52$	<p>2b</p>
<p>Annual/Monthly Recruitment</p>		
<p>Total breeding numbers (saw)</p>	$N_{r,y,1}^{br,saw} = \sum_{a=1}^{agemat^{saw}} N_{r,y,1,a}^{saw}$	<p>2c</p>
<p>Spawning biomass (barra)</p>	$B_{r,y,m}^{spn,barra} = \sum_{a=1}^z \zeta_{r,a}^{barra} \cdot f_a^{barra,female} \cdot \lambda_m^{barra} \cdot w_{r,y,m,a}^{sp} \cdot N_{r,y,m,a}^{barra}$	<p>2d</p>
<p>Spawning biomass (mudcr)</p>	$B_{r,y,m}^{spn,mudcr} = \sum_{a=1}^z f_{m,a}^{mudcr} \cdot w_{m,a}^{mudcr,sex} \cdot N_{r,y,m,a}^{mudcr,fem}$	<p>2e</p>

Recruitme nt (barra)	$R_{r,y+1,m}^{barra} = \frac{\alpha_r^{barra} B_{r,y,m}^{spn,barra}}{\beta_r^{barra} + B_{r,y,m}^{spn,barra}} \sigma_{r,y,m}^{flow_mudcr} \quad \text{if } \omega_{r,y,m} > 0.5$ $R_{r,y+1,m}^{barra} = \frac{\alpha_r^{barra} \sqrt{\omega_{r,y,m}} B_{r,y,m}^{spn,barra}}{\beta_r^{barra} + \sqrt{\omega_{r,y,m}} B_{r,y,m}^{spn,barra}} \sigma_{r,y,m}^{flow_mudcr} \quad \text{if } \omega_{r,y,m} \leq 0.5$ $\text{where } \omega_{r,y,m} = \frac{\sum_{a=1}^z (1 - \zeta_{r,a}^{barra}) \cdot f_a^{barra,male} \cdot \lambda_m^{barra} \cdot N_{r,y,m,a}^{barra}}{\sum_{a=1}^z [(1 - \zeta_{r,a}^{barra}) \cdot f_a^{barra,male} + \zeta_{r,a}^{barra} \cdot f_a^{barra,female}] \cdot \lambda_m^{barra} \cdot N_{r,y,m,a}^{barra}}$	2f
Recruitme nt (mudcr)	$R_{r,y,m+1}^{mudcr} = 0.5 \frac{\alpha_r^{mudcr} B_{r,y-1,m-1}^{spn,mudcr}}{\beta_r^{mudcr} + B_{r,y-1,m-1}^{spn,mudcr}} (\sigma_{r,y-1,m+1}^{flow_mudcr} + \sigma_{y-1,m+1}^{SOI_mudcr}) \quad \text{for } 1 < m < 12$ $R_{r,y+1,m}^{mudcr} = 0.5 \frac{\alpha_r^{mudcr} B_{r,y-1,m+10}^{spn,mudcr}}{\beta_r^{mudcr} + B_{r,y-1,m+10}^{spn,mudcr}} (\sigma_{r,y,m}^{flow_mudcr} + \sigma_{y,m}^{SOI_mudcr}) \quad \text{for } m = 1$ $R_{r,y+1,2}^{mudcr} = 0.5 \frac{\alpha_r^{mudcr} B_{r,y-1,12}^{spn,mudcr}}{\beta_r^{mudcr} + B_{r,y-1,12}^{spn,mudcr}} (\sigma_{r,y,2}^{flow_mudcr} + \sigma_{y,2}^{SOI_mudcr}) \quad \text{for } m = 12$ <p>with $\sigma_{y,m}^{SOI_mudcr} = 0$ for $r = 1, 2, 7, 8$</p>	2g
Stock- recruitme nt parameter s	$\beta^{psp} = \frac{(1 - h^{sp}) \sum_r \sum_w B_{r,1970,w}^{spn,psp}}{5h^{sp} - 1}$ $\alpha^{psp} = \frac{\beta^{psp} + \sum_r \sum_w B_{r,1970,w}^{spn,psp}}{SPR^{virg,psp}} \quad \text{where } SPR^{virg,psp} = \sum_r \sum_w B_{r,1970,w}^{spn,psp}$ $\beta_r^{sp} = \frac{(1 - h^{sp}) \sum_m B_{r,1970,m}^{spn,sp}}{5h^{sp} - 1}$ $\alpha_r^{sp} = \frac{\beta_r^{sp} + \sum_m B_{r,1970,m}^{spn,sp}}{SPR_r^{virg,sp}} \quad \text{where } SPR_r^{virg,sp} = \sum_m B_{r,1970,m}^{spn,sp}$	2h

<p>Recruitment (saw)</p>	$R_{r,y,1}^{saw} = \frac{\left(N_{r,y-1,1}^{br,saw} / K_r^{saw} \right)}{1 + \beta^{saw} \left(N_{r,y-1,1}^{br,saw} / K_r^{saw} \right)} \cdot \omega^{saw} \cdot \frac{1}{\tau^{saw}} \cdot pups^{saw} \cdot N_{r,y-1,1}^{br,saw} \cdot S_{juv}^{saw} \cdot \sigma_{r,\bar{y}}^{flow}$ <p>if $\sigma_{r,y-1,m}^{flow} > 2$ for any of $m = 1$ to 3,</p> <p>then $boom_{r,y} = \frac{1}{3} \sum_{m=1}^3 \sigma_{r,y-1,m}^{flow}$ with maximum $boom_max$</p> <p>and conversely, if $\sigma_{r,y-1,m}^{flow} < 1$ for any of $m = 1$ to 3,</p> <p>then $bust_{r,y} = \frac{1}{3} \sum_{m=1}^3 \sigma_{r,y-1,m}^{flow}$ with minimum $bust_min$, else</p> <p>if $1 \leq \sigma_{r,y-1,m}^{flow} \leq 2$ $boom_{r,y} = 1$, $bust_{r,y} = 1$</p> $R_{r,y,1}^{saw} = boom_{r,y} \cdot bust_{r,y} \cdot R_{r,y,1}^{saw}$	<p>2i</p>
<p>Age-length-mass relationships</p>		
<p>Weekly von Bertalanffy growth equation (prawns)</p>	$\ell_w^{psp} = \ell_{\infty}^{psp} \left[1 - e^{-K^{psp} (w - w_0^{psp})} \right]$	<p>3a</p>
<p>Mass-at-week (prawns)</p>	$w_w^{psp} = a^{psp} (\ell_w)^{b^{psp}} / 1e6$	<p>3b</p>
<p>Modified monthly von Bertalanffy growth equation</p>	$\ell_{r,y,m,t}^{sp} = \ell_{\infty}^{sp} \left[1 - e^{-\left(\sigma_{r,y,m}^{flow-sp} K^{sp} / 12 \right) (t - (t_0^{sp} * 12))} \right]$ <p>where $\sigma_{r,y,m}^{sp} = 1$ in all cases except one barra model version</p>	<p>3c</p>
<p>Mass-at-age</p>	$w_{r,y,m,a}^{sp} = a^{sp} (\ell_{r,y,m,a})^{b^{sp}} \quad (\text{barra})$ $w_{m,a}^{mudcr,sex} = a^{sp} (\ell_{m,a})^{b^{sp}} \quad (\text{mudcr})$	<p>3d</p>
<p>Flow relationships</p>		

Flow multiplier (prawns)	$\sigma_{r,y,w}^{flow_psp} = \sum_r X_r^{catchment} \lambda_{r,y,w}^{psp} \text{ where}$ $\lambda_{r,y,w}^{psp} = \frac{1}{1+e^{-\frac{(flw_{r,y,w}-thresM^{sp})}{thres^{sp}}}}$ <p>with $flw_{r,y,w} = \frac{fl_{r,y,w}}{flw_{r,w}}$ and $flw_{r,w} = \frac{1}{n_yr} \sum_{y=1970}^{2019} fl_{r,y,w}$</p> $X_r^{catchment} = 0 \forall \text{ catchment regions } r \text{ except current catchment if } r=1 \text{ or } r=7$ $X_r^{catchment} \text{ estimated flow influence in catchments 2-5 from each catchment } r = 2 \text{ to } 5$ <p>and $\sigma_{6,y,w}^{flow_psp} = \sigma_{5,y,w}^{flow_psp}$ if $r = 6$</p> $\sigma_{8,y,w}^{flow_psp} = \sigma_{7,y,w}^{flow_psp}$ if $r = 8$	4a
Flow multiplier (barra,saw ,mudcr)	$\sigma_{r,y,m}^{flow_sp} = \left(\frac{1}{\sum_m Z_m^{sp}} \right) \sum_{m=1}^{12} Z_m^{sp} \lambda_{r,y,m}^{sp} \text{ where}$ $\lambda_{r,y,m}^{sp} = \frac{1}{1+e^{-\frac{(flm_{r,y,m}-thresM^{sp})}{thres^{sp}}}}$ for barra,saw $\lambda_{r,y,m}^{sp} = thres^{sp} \left(flm_{r,y,m} - thresM^{sp} \right)^2 + c_\sigma$ for mudcr <p>with $flm_{r,y,m} = \frac{1}{n_wk} \sum_w flw_{r,y,w}$</p> <p>$Z_m^{sp} = 1 \forall$ months included in flow multiplier, else 0, with:</p> $Z_m^{barra} = 1$ (recruitment) for current month m and months 3 to 9 $Z_m^{barra} = 1$ (mortality; growth) for current month m $Z_m^{saw} = 1$ (recruitment; "boom-bust" recruitment) for $m = 1$ to 3 $Z_m^{saw} = 1$ ("pool dry" mortality) for $m = 1$ $Z_m^{mudcr} = 1$ (recruitment) for current month m	4b
Flow multiplier (mudcr catchability)	$q_{r,y,m}^{flow_mudcr} = \frac{1}{1+c_q + e^{-\frac{(flm_{r,y,m}-thresM^{sp})}{thres^q}}}$	4c
Southern Oscillation Index (SOI) relationship		
SOI multiplier (mudcr)	<p>If $r = 3-6$: $\sigma_{y,m}^{SOI_mudcr} = \eta \cdot \tau^{SOI}$ ($r=1,2,7,8$: $\sigma_{y,m}^{SOI_mudcr} = 0$)</p> <p>where $\eta = -1$ if $SOI < -7$ or $\eta = 1$ if $SOI > 7$</p>	4d
Catches and Fishery Interactions		
Predicted catch	$\widehat{C}_{r,y,w}^{psp} = q_{seas}^{psp} q_y^{psp} E_{r,y,w}^{psp} B_{r,y,w}^{exp_psp}$	5a

(mass) of prawns	where $seas1: w \leq 13; seas2: 13 < w \leq 25; seas3: 25 < w \leq 39; seas4: w > 39$	
Predicted catch (mass) of barra	$\widehat{C}_{r,y,m}^{sp} = q_r^{sp} q_f^{sp} \lambda_{r,y,m-1}^{sp} E_{r,y,m}^{sp} B_{r,y,m}^{exp_sp}$ where $\lambda_{r,y,m-1}^{sp} = 1$ for $r=1,2$ and $q_f^{sp} = qinc^{y-1989}$	5b
Predicted catch (mass) of mudcr	$\widehat{C}_{r,y,m}^{sp,sex} = q_r^{sp} q_{flow_mudcr}^{sp} E_{r,y,m}^{sp} B_{r,y,m}^{exp_mudcr_sex}$	5c
Predicted saw fishery interactions	$\widehat{C}_{r,y,m}^{saw,I} = (1 - PRS) \cdot [q_{trawl}^{saw} E_{r,y,m}^{trawl} + q_{net}^{saw} E_{r,y,m}^{net}]$ where $PRS=0$ except for sensitivity tests as detailed in Appendix (note this is independent of saw abundance)	5d
Commercially exploitable biomass (prawns)	$B_{r,y,w}^{exp_psp} = S_{y,w}^{psp} W_{r,y,w}^{psp} N_{r,y,w}^{psp} e^{-M_{r,y,w}^{psp}}$	5e
Number of saw susceptible to being caught	$B_{r,y,m}^{exp_saw,I} = \sum_{a=1}^z S_a^{saw,I} A_m^{saw} N_{r,y,m,a}^{saw} e^{-M_a^{saw}}$ (for each fishery sector I)	5f
Commercially exploitable biomass (barra)	$B_{r,y,m}^{exp_sp} = \sum_{a=1}^z S_{r,y,a}^{barra} A_{r,m}^{sp} W_{r,y,m,a}^{sp} N_{r,y,m,a}^{sp} e^{-M_{r,y,m,a}^{sp}}$ where $S_{r,y,a}^{barra} = \frac{1}{1 + e^{-(a-a_{50})/\delta_{r,y}}}$ $a < 8$ $S_{r,y,a}^{barra} = e^{-(sfa_{r,y} * (a-8))}$ $a \geq 8$ with $a_{50}=3$	5g
Commercially exploitable biomass (barra,mudcr)	$B_{r,y,m}^{exp_mudcr,sex} = \sum_{a=1}^z S_{r,y,m,a}^{mudcr,sex} A_{r,m}^{mudcr,sex} W_{m,a}^{mudcr,sex} N_{r,y,m,a}^{sp,sex} e^{-M_{r,y,m,a}^{sp}}$ Where $A_{r,m}^{mudcr,sex}$ omits subscript y for simplicity but uses adjusted values for NT with $y < 2001$	5h
Fishery proportion	$F_{r,y,m}^{sp} = \widehat{C}_{r,y,m}^{sp} / B_{r,y,m}^{exp_sp}$ (barra) $F_{r,y,m}^{sp,sex} = \widehat{C}_{r,y,m}^{sp,sex} / B_{r,y,m}^{exp_sp,sex}$ (mudcr)	5i

	$F_{r,y,w}^{psp} = \widehat{C}_{r,y,w}^{psp} / B_{r,y,w}^{exp-psp}$ (prawns)	
Fishery interaction proportion	$F_{r,y,m}^{saw,l} = \widehat{C}_{r,y,m}^{saw,l} / B_{r,y,m}^{exp-sp,l}$ (saw) Where sector $l=1$ is the gillnet fishery and $l=2$ is the trawl fishery	5j
Catch weekly numbers for prawns	$C_{r,y,w}^{psp} = S_w^{psp} F_{r,y,w}^{psp} N_{r,y,w}^{psp} e^{-M_{r,y,w}^{psp}}$	5k
Interactions-at-age (numbers) for saw	$C_{r,y,m,a}^{saw,l} = \sum_{l=1}^2 S_a^{saw,l} A_m^{saw,l} F_{r,y,m}^{saw,l} N_{r,y,m,a}^{saw,l} e^{-M_a^{saw,l}}$	5l
Catch-at-age (numbers) for barra	$C_{r,y,m,a}^{barra} = S_{r,y,a}^{sp} A_{r,m}^{sp} F_{r,y,m}^{sp} N_{r,y,m,a}^{sp} e^{-M_{r,y,m,a}^{sp}}$	5m
Catch-at-age (numbers) for mudcr	$C_{r,y,m,a}^{mudcr,sex} = S_{r,y,m,a}^{sp,sex} A_{r,m}^{sp,sex} F_{r,y,m}^{sp,sex} N_{r,y,m,a}^{sp,sex} e^{-M_{r,y,m,a}^{sp}}$	5n
Trophic relationships		
Predator-prey interaction term	$f(B_{a,y,m}^{prey}) = \frac{\beta^j \alpha^j B_{a,y,m}^{prey} / B_{a,1970,m}^{prey}}{\beta^j \alpha^j - 1 + B_{a,y,m}^{prey} / B_{a,1970,m}^{prey}}$	6a
Interaction steepness formulation	$\alpha^j = \frac{4H^j}{5H^j - 1}$	6b
The likelihood function		
Observed vs estimated weekly	$C_{r,y,w}^{psp,obs} = \widehat{C}_{r,y,w}^{psp} e^{\varepsilon_{r,y,w}^{psp}}$ or $\varepsilon_{r,y,w}^{psp} = \ln(C_{r,y,w}^{psp,obs}) - \ln(\widehat{C}_{r,y,w}^{psp})$ and $\varepsilon_{r,y,w}^{psp}$ from $N(0, (\sigma_r^{psp})^2)$	7a

catch (prawns)		
Contribution to negative of log-likelihood function (prawns)	$-\ln L_r^{psp,reg} = \sum_y \left[\sum_w \ln \sigma_r^{psp} + (\varepsilon_{r,y,w}^{psp})^2 / 2(\sigma_r^{psp})^2 \right]$ $-\ln L^{psp} = \sum_r -\ln L_r^{psp,reg}$ <p>for $r = 1-8$; $y = 1970-2019$; $w = 13-22$ & when $C_{r,y,w}^{psp,obs} > 0, \hat{C}_{r,y,w}^{psp} > 0$</p>	7b
Standard deviation of the residuals for the logarithms of the catch estimates (prawns)	$\hat{\sigma}_r^{psp} = \sqrt{\frac{1}{n_r^{psp}} \sum_y \sum_w (\ln C_{r,y,w}^{psp,obs} - \ln \hat{C}_{r,y,w}^{psp})^2}$ <p>assumed to be independent of y and w, and set in the fitting procedure by its maximum likelihood value</p>	7c
Observed vs estimated monthly catch (barra, mudcr)	$C_{r,y,m}^{sp,obs} = \hat{C}_{r,y,m}^{sp} e^{\varepsilon_{r,y,m}^{sp}} \quad \text{or} \quad \varepsilon_{r,y,m}^{sp} = \ln(C_{r,y,m}^{sp,obs}) - \ln(\hat{C}_{r,y,m}^{sp})$ <p>and $\varepsilon_{r,y,m}^{sp}$ from $N(0, (\sigma_r^{sp})^2)$</p> <p>and for <i>mudcr</i></p> $\hat{C}_{r,y,m}^{sp} = \hat{C}_{r,y,m}^{sp,male} \quad \text{for } r = 1-6 \text{ (only males caught)}$ $\hat{C}_{r,y,m}^{sp} = \hat{C}_{r,y,m}^{sp,male} + \hat{C}_{r,y,m}^{sp,female} \quad \text{for } r = 7-8$	7d
Contribution to negative of log-likelihood function (barra, mudcr)	$-\ln L_r^{sp,reg} = \sum_y \left[\sum_m \ln \sigma_r^{sp} + (\varepsilon_{r,y,m}^{sp})^2 / 2(\sigma_r^{sp})^2 \right]$ $-\ln L^{sp} = \sum_r -\ln L_r^{sp,reg}$ <p>for $r = 1-8$; $y = 1989-2019$; $m = 3-9$ (<i>barra</i>); $m = 1-12$ (<i>mudcr</i>)</p> <p>& when $C_{r,y,m}^{sp,obs} > 0, \hat{C}_{r,y,m}^{sp} > 0$</p> <p>Note for <i>mudcr</i>: for $r = 7-8$; $y = 1983-2018$ and for $r = 6$; $m = 1, 4-10$</p>	7e
Standard deviation of the residuals for the logarithms of the catch estimates	$\hat{\sigma}_r^{sp} = \sqrt{\frac{1}{n_r^{sp}} \sum_y \sum_m (\ln C_{r,y,m}^{sp,obs} - \ln \hat{C}_{r,y,m}^{sp})^2}$ <p>assumed to be independent of y and m, and set in the fitting procedure by its maximum likelihood value</p>	7f

(barra, mudcr)		
Predicted catch-at-age proportion (barra)	$\hat{p}_{sth,y,a}^{barra} = \frac{\sum_{r=3}^6 \hat{C}_{r,y,6,a}^{barra}}{\sum_{r=3}^6 \sum_{a=1}^{20} \hat{C}_{r,y,6,a}^{barra}}$ <p style="text-align: center;">where area A is southern (sth) or northern (nth)</p> $\hat{p}_{nth,y,a}^{barra} = \frac{\hat{C}_{2,y,6,a}^{barra}}{\sum_{a=1}^{20} \hat{C}_{2,y,6,a}^{barra}}$ <p>and for $\hat{p}_{sth,y,a}^{barra} > 0$</p>	7g
CAA likelihood contribution (barra)	$-\ln L_A^{CAA} = \sum_{y=2000}^{2019} \sum_{a=1}^{20} \left[\ln \left(\frac{\sigma_A^{CAA}}{\sqrt{p_{A,y,a}^{obs,barra}}} \right) + \frac{p_{A,y,a}^{obs,barra} \left(\ln(p_{A,y,a}^{obs,barra} + \delta^{CAA}) - \ln(\hat{p}_{A,y,a}^{barra} + \delta^{CAA}) \right)^2}{2(\sigma_A^{CAA})^2} \right]$ <p>where $\delta^{CAA} = 0.01$, and for $p_{A,y,a}^{obs,barra} > 0$; $\hat{p}_{A,y,a}^{barra} > 0$</p>	7h
Standard deviation associated with the catch-at-age data for region A (barra)	$\sigma_A^{CAA} = \sqrt{\frac{1}{(y \cdot a)} \sum_{y=2000}^{2019} \sum_{a=1}^{20} p_{A,y,a}^{obs,barra} \left(\ln(p_{A,y,a}^{obs,barra} + \delta^{CAA}) - \ln(\hat{p}_{A,y,a}^{barra} + \delta^{CAA}) \right)^2}$	7i

Table A25. Description of the variables of the MICE for common banana prawns (*psp*=prawns), barramundi (*sp*=barra), mud crabs (*sp*=mudcr) and sawfish (*sp*=saw)

Parameter	Description	Input, computed, estimated
Species		
$N_{r,y,w}^{psp}$	number of prawn species <i>psp</i> in region <i>r</i> in year <i>y</i> and week <i>w</i>	Computed from Eq. 1a
$N_{r,y,m,a}^{sp,sex}$	number of species <i>sp</i> in region <i>r</i> in year <i>y</i> and month <i>m</i> of age <i>a</i> and sex is either male or female or combined sexes (no sex-disaggregation represented for sawfish)	Computed from Eq. 1b
$N_{r,y,1}^{br,saw}$	Number of mature (breeding) sawfish in January (breeding assumed to be a pulse event) in region <i>r</i> and year <i>y</i> , month 1	Computed from Eq. 2c
K_r^{saw}	Carrying capacity or ‘pristine’ number of sawfish in 1970 in region <i>r</i> which is used to tune to different depletion levels	Input tuned to produce different current depletion estimates
$M_{r,y,w}^{psp}$	Mortality rate of prawn species <i>psp</i> in region <i>r</i> in year <i>y</i> and week <i>w</i>	Computed from Eq. 1f
$M_{r,y,m,a}^{sp}$	Mortality rate of species <i>sp</i> in region <i>r</i> in year <i>y</i> and month <i>m</i> for age <i>a</i> (but age-independent form used for mud crabs)	Computed from Eq. 1g
$C_{r,y,m,a}^{sp,sex}$	Total catch (number) of species <i>sp</i> in region <i>r</i> in year <i>y</i> and month <i>m</i> of age <i>a</i> and sex	Computed from Eq. 5n
$R_{r,y,w}^{psp}$	Number of recruiting prawns (6-month old) of species <i>psp</i> in region <i>r</i> in year <i>y</i> and week <i>w</i>	Computed from Eq. 2b
$R_{r,y,m}^{sp}$	Number of recruits (defined respectively as 1-year old & 14-month old for species <i>barra</i> , <i>mudcr</i> in region <i>r</i> in year <i>y</i> and month <i>m</i>	Computed from Eq. 2f-g
$R_{r,y,1}^{saw}$	Number of <i>saw</i> recruits (defined as 1-year old, juveniles in region <i>r</i> in year <i>y</i> and month <i>m</i> =1 (set at 0 for <i>m</i> >1)	Computed from Eq. 2h
Z^{sp}	Largest age considered (i.e. corresponding to a “plus group”)	Input (Table 3)

ℓ_w^{psp}	Length of prawn species psp in week w	Input (Table A26)
ℓ_∞^{psp}	Maximum length of prawn species psp	Input (Table A26)
K^{psp}	Growth rate parameter for prawn species psp	Input (Table A26)
t_0^{psp}	growth parameter for prawn species psp	Computed using Eq. 3c
$\ell_{r,y,m,a}^{sp}$	length of species sp of age a	Input (Table A26)
ℓ_∞^{sp}	Maximum length of species sp	Input (Table A26)
K_a^{sp}	growth rate parameter for species sp	Input (Table A26)
t_0^{sp}	growth parameter for species sp	Input (Table A26)
w_w^{psp}	wet mass of prawn species psp in week w , converted to units of tonnes in the model	Computed using Eq. 3b
$w_{r,y,m,a}^{sp}$	wet mass of species sp (<i>barra</i>) for age a in region r in year y and month m of age a , converted to units of tonnes in the model	Computed using Eq. 3b
$w_{m,a}^{mudcr,sex}$	wet mass of <i>mudcr</i> according to <i>sex</i> in month m and age a (assumed constant by region r and over years y), converted to units of tonnes in the model	Computed using Eq. 3b
$a^{psp}; b^{psp}$ $a^{sp}; b^{sp}$	Mass-at-age constants for prawn species psp or species sp	Input (Table A26)
h^{sp}	stock-recruitment steepness parameter for species sp which is the proportion of the virgin recruitment that is realized at a spawning biomass level of 20% of the virgin spawning biomass	Input (Table A26)
$\alpha^{psp}; \beta^{psp}$ $\alpha_r^{sp}; \beta_r^{sp}$	Stock-recruitment parameters for prawn species psp assumed constant across regions but region-specific for species sp and computed assuming the stock is at equilibrium in the first model year	Computed using Eq. 2h
$SPR^{virg,psp}; SPR_r^{virg,sp}$	The virgin (1970 starting year) spawning biomass for prawn species psp (modelled as combined over all r) and region-specific for species sp	Computed using Eq. 2h

$agemat^{saw}$	Age-at-maturity for sawfish	Input (Table A26)
Flow relationships		
$fl_{r,y,w}$	The cumulative weekly end-of-system (EOS) flow in catchment region r in year y and week w	Calculated from daily EOS flow from river models
$fl\bar{w}_{r,w}$	The average weekly end-of-system flow in catchment region r in week w (average of historical flows for that week used to standardise flow inputs)	Computed using Eq. 4a
$flm_{r,y,m}$	The standardised monthly end-of-system flow in catchment region r in year y in month m (based on average or cumulative weekly standardised flows)	Computed using Eq. 4b
$\lambda_{r,y,m}^{sp}$	The standardised flows converted into a logistic multiplier value for species sp , year y , region r and month m	Computed using Eq. 4b
$\sigma_{r,y,m}^{flow_sp}$	The average standardised monthly end-of-system flow multiplier value for species sp in catchment region r in year y and month m	Computed using Eq. 4b
$thresM^{sp}$	The flow multiplier logistic function maximum threshold parameter for species sp or optimal flow parameter if using parabolic equation as for $mudcr$	Estimated or fixed – see Table 10
$thres^{sp}$	The flow multiplier logistic function second threshold parameter for species sp or 2 nd parabolic parameter for $mudcr$ parabolic equation	Estimated or fixed – see Table 10
$thres^q$	A catchability-related flow parameter (for mud crabs)	Estimated or fixed – see Table 10
C_q	A catchability-related flow scaling parameter (for mud crabs)	Estimated or fixed – see Table 10
C_σ	A recruitment-related flow scaling parameter (for mud crabs)	Fixed – Table A26
$q_{r,y,m}^{flow_mudcr}$	The standardised monthly end-of-system flow multiplier value for mudcrab catchability in catchment region r in year y and month m	Estimated or fixed – see Table 10
Southern Oscillation Index (SOI) relationship		
$\sigma_{y,m}^{SOI_mudcr}$	The SOI multiplier value for mudcr recruitment in year y and month m	Computed using Eq. 4d

τ^{SOI}	A recruitment-related SOI parameter (for mud crabs)	Estimated
η	A SOI scaling parameter that takes the value -1 or 1 to either reduce or enhance mud crab recruitment.	Fixed. See Eq. 4d
Mortality terms		
$M_{base}^{psp}; M_{base}^{sp}$	Base weekly (for prawn species <i>psp</i>) and monthly (other species <i>sp</i>) natural mortality rate	Input (Table A26)
$M_{r,y,w}^{psp}$	Weekly mortality rate for prawn species <i>psp</i> in region <i>r</i> in year <i>y</i> and week <i>w</i>	Computed using Eq. 1f
$prod_y; \overline{prod}$	Productivity boost multiplier in year <i>y</i> ; average productivity value	Computed using Eq. 1f
$B_{r,y,w}^{Mb}$	Relative biomass of microphytobenthos <i>Mb</i> in region <i>r</i> in year <i>y</i> and week <i>w</i>	(See Table A27)
<i>scour</i>	Proportion of material displaced by scouring due to large floods	Computed using Eq. 1f
$M_{r,y,m}^{mudcr,temp}$	Additional mortality of mudcr due to temperature in region <i>r</i> (only included for <i>r</i> =7 and temp anomaly > temp_opt) in year <i>y</i> and month <i>m</i>	Computed using Eq. 1g
$temp_{r,y,m}$	Monthly air temperature anomaly (i.e. the difference between the observed temperature and the long-term monthly mean) in region <i>r</i> in year <i>y</i> and month <i>m</i>	BOM station air temperature data
$temp^{opt}$	Optimal air temperature (°C) for mud crabs	Input (Table A26)
$\tau^{temp}; c^{temp}$	Temperature-related parameter for <i>mudcr</i> mortality estimated; temperature-related scaling parameter fixed	Estimated; fixed – see Table A26
M_{par}^{sp}	A scaling parameter for species <i>sp</i> to control slope of age-dependent relationship	Input (Table A26)
M_{pools}^{saw}	Additional mortality for sawfish dying in pools in a dry year	Input (Table A26)
$thres_{pools}^{saw}$	Threshold standardised flow below which pools assumed to dry	Input (Table A26)
Biomass terms		
$B_{r,y,w}^{spn,psp}$	Spawning (<i>spn</i>) biomass of prawn species <i>psp</i> in region <i>r</i> in year <i>y</i> and week <i>w</i>	Computed using Eq. 2a

f_w^{psp}	Proportion of prawns spawning in week w	Input (Table A26)
$f_{m,a}^{mudcr}$	Proportion of mudcr of age a mature in month m	Input (Table A26)
$B_{r,y,m}^{spn,sp}$	Spawning biomass of species sp in region r in year y and month m	Computed using Eq. 2d-e
$\xi_{r,a}^{barra}$	Proportion of barra, population of age a that are female, with different values used for regions $r=1-2$ and $r=3-8$	Input (Table A26)
λ_m^{barra}	Determines whether spawning occurs in month m , hence set at 1 for $m=1,10,11,12$ and 0 otherwise	Input
$f_a^{barra,sex}$	Proportion of barra of age a and sex sex that are mature	Input (Table A26)
$\omega_{r,y,m}$	Barramundi sex ratio (proportion male) of mature animals in region r in year y and month m	Computed using Eq. 2f
ω^{saw}	Sex ratio (proportion female) for saw (set at 0.5)	(Table A26)
τ^{saw}	Sawfish average reproductive interval (2 years)	(Table A26)
$pups^{saw}$	Average number of pups per saw female per year	Input (Table 10)
S_{juv}^{saw}	Juvenile base survival rate which applies to pups that survive the first year of life, and is computed as an equilibrium parameter that keeps the population in balance in the absence of external sources of mortality (i.e. solve equilibrium value when substituting pristine sawfish recruitment number $R_{r,1970,1}^{saw}$ in Eqn 2i)	Input (Table 10)
β^{saw}	Density-dependent shape parameter for saw recruitment (0.8)	Input
Sawfish boom-bust dynamics		
$boom_{r,y}$	The “boom” recruitment multiplier for sawfish in region r and year y , which scales up breeding success in high flow years	Computed using Eq. 2i
$boom_max$	Maximum value of boom recruitment multiplier	Table 10
$bust_{r,y}$	The “bust” recruitment multiplier for sawfish in region r and year y , which scales down breeding success in low flow years	Computed using Eq. 2i
$bust_min$	Minimum value of bust recruitment multiplier	Table 10

Predator-prey interactions		
$B_{a,y,m}^{prey}$	Number of common banana prawn prey using as proxy the average number of recruits in region r in year y and month m (i.e. prey summed over weeks in month)	Computed using Eq. 6a
H^j	Parameter controlling steepness of interaction relationship between prawns and predator j (barramundi or sawfish)	Table 10
α^j	Interaction curve shape parameter	Computed using Eq. 6b
β^j	Interaction curve scaling parameter set at 1.1	Input
Catches and fishery interactions		
$\hat{C}_{r,y,w}^{psp}$	Predicted total prawn species psp catch in region r and year y , week w	Computed using Eq. 5a
$\hat{C}_{r,y,m}^{saw,I}$	Predicted total number of fishery interactions for saw in region r and year y , month m but noting that interaction rate is different to interaction mortality rate as not all sawfish interactions with fishing gear result in mortality of the animal as we assume here (due to lack of information – see text for details)	Computed using Eq. 5d
$\hat{C}_{r,y,m}^{sp,sex}$	Predicted total catch of species sp (barra, mudcr) in region r and year y , month m (disaggregated by sex for mudcr)	Computed using Eq. 5b-c
q_{seas}^{psp}	Catchability coefficient for prawn species psp in season $seas$ defined as follows: $seas1: w \leq 13; seas2: 13 < w \leq 25;$ $seas3: 25 < w \leq 39; seas4: w > 39$	Fixed based on area-aggregated initial model
$q_{trawl}^{saw}; q_{net}^{saw}$	Catchability coefficients for sawfish of species saw , that are caught respectively by the trawl sector and net-fishing sector	Input from Pillans et al. (in press)
PRS	Post-release survival proportion (0 in ensemble but sensitivity to post year 2000 settings of 0.25, 0.75 and 0.9 tested)	See sawfish Appendix
q_r^{sp}	Catchability coefficient for species sp in region r	Estimated – Table 10
$qf_y^{psp}; qf_y^{sp}$	Fishing power (relative efficiency increase) per year y for prawns of species psp and for $sp=barra$	Input
qrc	Fishing power annual increase rate for $sp=barra$	Input

$E_{r,y,w}^{psp}$	Weekly fishing effort for prawn species psp in each region r and week w of year y	Fishery effort data from NPFI
$E_{r,y,m}^{sp}$	Monthly fishing effort for species sp (barra, mudcr) in each region r and month m of year y	Fishery effort data from Qld and NT
$E_{r,y,m}^{trawl}, E_{r,y,m}^{net}$	Monthly fishing effort in each region r and year y for the trawl (sum of tiger prawn and common banana prawn fishing effort) and net-fishing (using barramundi effort as an index) sectors	Fishery effort data from Qld and NT
$B_{r,y,w}^{exp_psp}$	Weekly exploitable biomass of prawn species psp in each region r and week w of year y	Computed using Eq. 5e
$B_{r,y,m}^{exp_saw,l}$	Total number of sawfish susceptible (i.e. exploitable numbers) to be caught in each month m , year y and region r	Computed using Eq. 5f
$S_{y,w}^{psp}$	Fishing selectivity of prawn species psp in week w for each of 3 time periods as follows: (1) $y \leq 1987$; (2) $1987 < y < 2001$; (3) $y \geq 2001$	Input (Table A26)
$B_{r,y,m}^{exp_sp_sex}$	Monthly exploitable biomass of species sp (barra, mudcr) caught in each month m , year y and region r	Computed using Eq.5g-h
$F_{r,y,w}^{psp}, F_{r,y,m}^{sp}$	Proportion caught (as target catch) of prawns psp or species sp in region r and year y , and week w or month m (noting this is disaggregated by sex for mudcr)	Computed using Eq. 5i
$F_{r,y,m}^{saw,l}$	Proportion captured (fishery interaction) of saw by sector l in region r and month m of year y	Computed using Eq. 5j
$C_{r,y,m,a}^{saw,l}$	Total number of saw of each age a that interact with fisheries sector l in region r and year y , month m	Computed using Eq. 5l
A_m^{saw}	Availability of saw to interactions in month m which is assumed the same across all regions and set at 1 for all months	Input – see Table A26
$S_a^{saw,l}$	Selectivity to interactions with sector l of sawfish of age a , assumed the same across regions and for sector l =gillnet set at 1 for all ages 1 and older and sector l =trawl, zero for juveniles and 1 for ages ≥ 4	Input – see Table A26
$C_{r,y,m,a}^{sp,sex}$	Total number of sp of each age a caught in region r and year y , month m and in the case of mudcr for different sex	Computed using Eq 5m-n

$A_{r,m}^{sp}$	Availability to the fishery of species sp , in month m in region r	Input – see Table A26
$A_{r,m}^{mudcr,sex}$	Availability to the fishery of <i>mudcr</i> according to <i>sex</i> in month m in region r (with availability based on (a) historical sex ratios to help inform when mud crabs are predominantly caught throughout the year vs when they are unavailable e.g. females moved offshore to spawn and (b) a 25-30% decrease in availability set for $r=7,8$ when $y<2001$ and over moulting period $m=8-10$ as soft shell crabs had to be returned)	Input – see Table A26
$S_{r,y,a}^{barra}$	Fishery selectivity of <i>barra</i> of age a , in region r and year y ; uses a different selectivity for $y<1997$ and also three different settings for $r=1-2$; 3-6; 7-8 determined by the two selectivity parameters $sfa_{r,y}, \delta_{r,y}$ which are estimated for each of the 3 regions and 2 time periods.	Computed using Eq. 5g; see estimates Table 10
$S_{r,y,m,a}^{sp,sex}$	Fishery selectivity of <i>mudcr</i> of age a , in month m and year y in region r , (with selectivity based on average size each month) with value 0 for females in Qld ($r=1-6$), whereas females in NT ($r=7-8$) value is zero for $m=1$, 50% for $m=2-3$, then 1 (before 2006) or zero for $m=1-3$, 50% for $m=4-5$, then 1 (from 2006); <i>sex</i> =male full selectivity except month 1(NT) and months 1-3 (Qld)	Input – see Table A26
Likelihood equations		
$C_{r,y,w}^{psp,obs}$	Observed prawn species <i>psp</i> catch in region r and year y , week w	Qld & NT data input
$\mathcal{E}_{r,y,w}^{psp}$	Catch estimate residual when assuming that the observed catches are log-normally distributed	Computed using Eq. 7a
$\hat{\sigma}_r^{psp}$	Standard deviation of the residuals for the logarithms of the prawn <i>psp</i> catch series for region r (assumed independent of y and w)	Computed using Eq. 7c
n_r^{psp}	Total number of data points for prawn <i>psp</i> in region r for all years y and w included in likelihood term	Computed using Eq. 7c
$-\ln L_r^{psp,reg}$	The regional (<i>reg</i>) contribution to the negative of the log-likelihood function (after removal of constants) for prawn <i>psp</i> in region r	Computed using Eq. 7b
$-\ln L^{psp}$	The total contribution to the negative of the log-likelihood function from fitting to catch data for prawn <i>psp</i> (equations including fitting to survey data not shown here as survey data not considered adequately representative for common banana prawns)	Computed using Eq. 7b

$C_{r,y,m}^{sp,obs}$	Observed species sp catch in region r and year y , month m which for <i>mudcr</i> constitutes males only in Qld and both sexes combined for NT	Qld & NT data input
$\varepsilon_{r,y,m}^{sp}$	Catch estimate residual for species sp when assuming that the observed catches are log-normally distributed	Computed using Eq. 7d
$\hat{\sigma}_r^{sp}$	Standard deviation of the residuals for the logarithms of the sp catch series for region r (assumed independent of y and m)	Computed using Eq. 7f
n_r^{sp}	Total number of data points for sp in region r for all years y and months m included in likelihood term	Computed using Eq. 7f
$-\ln L_r^{sp,reg}$	The regional (<i>reg</i>) contribution to the negative of the log-likelihood function (after removal of constants) for species sp in region r	Computed using Eq. 7e
$-\ln L^{sp}$	The total contribution to the negative of the log-likelihood function from fitting to catch data for species sp (noting as below that for <i>barra</i> , the total likelihood also includes a contribution from fitting to catch-at-age data)	Computed using Eq. 7e
$-\ln L_A^{CAA}$	The total contribution to the negative of the log-likelihood function from fitting to catch data for <i>barra</i> in $A=nth$ ($r=2$) or $A=sth$ ($r=3-6$)	Computed using Eq. 7h
$p_{A,y,a}^{obs,barra}$	The observed proportion of <i>barra</i> of age a in combined area A in year y	Qld data input
$\hat{p}_{A,y,a}^{barra}$	The predicted proportion of <i>barra</i> of age a in combined area A in year y	Computed using Eq. 7g
σ_A^{CAA}	The standard deviation associated with the catch-at-age data for <i>barra</i> in $A=nth$ ($r=2$) or $A=sth$ ($r=3-6$)	Computed using Eq. 7f

Table A26. Input values and sources for parameter settings for common banana prawns (*psp*=prawns), barramundi (*sp*=barra), mud crabs (*sp*=mudcr) and sawfish (*sp*=saw) in MICE not included in Model Ensemble Tables.

Parameter	Input Value (units)	Justification /Reference
Species		
M_{base}^{psp}	0.045 wk ⁻¹	Deng et al. (2020)
M_{base}^{sp}	0.00167 mth ⁻¹ (barra); 0.1 mth ⁻¹ (mudcr); 0.0042 mth ⁻¹ (saw)	See also Ensemble Table; Knuckey (1999)
Z^{sp}	20 y (barra, saw); 3 (mudcr)	Fisheries Qld monitoring fishery monitoring data
K^{psp}	0.089	Deng et al. (2020)
ℓ_{∞}^{psp}	35.58 mm	Deng et al. (2020)
t_0^{psp}	0	Deng et al. (2020)
K_a^{sp}	0.18 (barra); 1.14 (mudcr)	Streipert et al. (2019); Knuckey (1999)
ℓ_{∞}^{sp}	129.7 cm (barra); 193.6 mm carapace width (mudcr)	Robins et al. (2005); Streipert et al. (2019); Knuckey (1999)
t_0^{sp}	0 (all)	Streipert et al. (2019)
$a^{psp}; b^{psp}$ $a^{sp}; b^{sp}$	0.001709; 2.7775 (prawns) 0.0106; 3.02 (barra) 0.00001418; 3.5496 (mudcr male) 0.00028914; 2.8821 (mudcr female)	Deng et al. (2020); Streipert et al. (2019); Knuckey (1999)
h^{sp}	0.9 (prawns); 0.8 (barra); 0.6 (mudcr)	High steepness due to environmental driver; lower setting for mudcr
$agemat^{saw}$	9 y	Pillans et al. (in press)
i_{thresM}^{sp}	<i>Prawns, barra</i> : set at 0.8 or 1.5 <i>Mudcr</i> : set at 2 or 2.5	See Ensemble Table 10

	<i>Saw</i> : range of values tested 0.1, 0.4, 0.7	
$thres^{sp}$	<i>Saw</i> : range of values tested 0.2, 0.6, 0.7	See Ensemble Table 10
c_{σ}	<i>Mudcr</i> : fixed at 3.5	Model scaling setting
c_q	<i>Mudcr</i> : fixed at 0.5	Model scaling setting
$temp^{opt}$	<i>Mudcr</i> : Fixed at 34.5 (mean air temperature for Nov-Dec)	Selected for Nov-Dec period based on Robins et al. (2020)
$\tau^{temp}; c^{temp}$	<i>Mudcr</i> : first parameter estimated; scaling parameter fixed at 1	See Ensemble Table 10
M_{par}^{sp}	0.045	Selected to provide reasonable spread in mortality at age
M_{pools}^{saw}	100	Set to substantially increase mortality in dry pools based on Lear et al. (2019)
$thres_{pools}^{saw}$	0.2	Low flow value
<i>boom_max</i>	1.2 or 1.5	See Ensemble Table 10
<i>bust_min</i>	0.8	See Ensemble Table 10
f_w^{psp}	Prawns: relative spawning proportion per weeks 1-52: 0.0459 0.0459 0.0459 0.0459 0.0459 0.0101 0.0101 0.0101 0.0101 0.0112 0.0112 0.0112 0.0112 0.0112 0.0112 0.0783 0.0783 0.0783 0.0783 0.0028 0.0028 0.0028 0.0028 0.0034 0.0034 0.0034 0.0034 0.0022 0.0022 0.0022 0.0022 0.0447 0.0447 0.0447 0.0447 0.0447 0.0168 0.0168 0.0168 0.0168 0.0045 0.0045 0.0045 0.0045 0.0078 0.0078 0.0078 0.0078 0.0224 0.0224 0.0224 0.0224	Based on Crocos and Kerr (1983)
$f_{m,a}^{mudcr}$	0 for $m=1-5$ when $a=1$, otherwise 1	Mature from 18 months (Knuckey 1999)
$\zeta_{r,a}^{barra}$	<i>Barra</i> : proportion females for ages 1 to 20: r=1-2: 0.092 0.131 0.183 0.251 0.332 0.426 0.525 0.622 0.710 0.785 0.845 0.890	Robins pers comm; based on Moore 1979; Davis 1982; Schipp et al. 2007;

	0.924 0.947 0.964 0.976 0.984 0.989 0.993 0.995 r=3-8: 0.053 0.076 0.110 0.155 0.215 0.289 0.377 0.474 0.574 0.667 0.749 0.816 0.869 0.908 0.936 0.956 0.970 0.980 0.986 0.991	Grey 1987; Campbell et al. 2017
λ_m^{barra}	1 for $m=1,10,11,12$ and 0 otherwise	Davis (1985); Campbell et al. 2019
$f_a^{barra,sex}$	<i>Barra</i> males: 0 for $a=1$ and $a>5$; 0.15 ($a=2$); 0.33 ($a=3$); 0.5 ($a=4$); 0.6 ($a=5$) <i>Barra</i> females: 0 for $a<5$; 0.4 ($a=5$); 1 ($a>5$)	Based on Streipert & Filar (2019)
ω^{saw}	0.5	Even sex ratio
τ^{saw}	2 y	Pillans et al. (in press)
$pups^{saw}$	9	Pillans et al. (in press)
β^{saw}	0.8	Model setting
H^j	Estimated (0.85)	See Ensemble Table 10
α^j	Calculated from H^j	See Eqn. 6b
β^j	1.1	Model scalar
q_{seas}^{psp}	For prawns in quarters 1-4 of year: 0.00041; 0.00038; 0.00255; 0.01685	Based on fitting non-spatial version of model
$S_{y,w}^{psp}$	For prawns in quarters 1-4 of year: y≤1987: 0.998 0.873 0.000 0.000 1987<y<2001 1.000 0.276 1.000 0.002 y≥2001 1.000 0.683 0.000 0.000	Model estimated
$q_{trawl}^{saw}; q_{net}^{saw}$	0.02; 0.008	Pillans et al. (in press)
$qf_y^{psp}; qf_y^{sp}$	See sources	Prawns use stock assessment settings; Barra based on Streipert et al. (2019)
$qinc$	1.02 (for Qld); 1 (i.e. no effect) for NT	Based on Campbell et al. (2017) but simplified annual rate since 1989
A_m^{saw}	1	Pillans (pers comm)

$\delta_{r,y}$	Selectivity parameter for barramundi that is estimated for post 1997 period for Qld southern GoC and northern GoC; for NT fixed at 0.15 and for early (pre-1997) period fixed at 0.46	Model estimated or fixed based on available information
$sfa_{r,y}$	Selectivity parameter for barramundi that is estimated for post 1997 period for Qld southern GoC and northern GoC; for NT fixed at 0.085 and for early (pre-1997) period fixed at 0.184	Model estimated or fixed based on available information
$S_a^{saw,l}$	l =gillnet set at 1 for all ages 1 and older and sector l =trawl, zero for juveniles and 1 for ages \geq 4	Pillans (pers comm)
$A_{r,m}^{sp}$	<i>Barra</i> : 0 for $m=1,10,11,12$; 0.5 for $m=2$; else 1	Based on fishery closed seasons
$A_{r,m}^{mudcr,sex}$	<p><i>Mudcr</i>: availability for $m=1-12$</p> <p>For sex=female, $r=1-6$:</p> <p>0 for all months</p> <p>For sex=male, $r=1-6$:</p> <p>0.25, 0.5, 1.0, 1.0, 1.0, 1.0, 0.8, 0.8, 0.8, 0.8, 0.5, 0.25</p> <p>For sex=female, $r=7-8$:</p> <p>0.25, 0.5, 1.0, 1.0, 1.0, 1.0, 0.8, 0.8, 0.8, 0.8, 0.5, 0.25</p> <p>For sex=male, $r=7-8$:</p> <p>0.25, 0.25, 0.25, 0.5, 0.75, 1.0, 1.0, 1.0, 1.0, 0.75, 0.5, 0.5</p> <p>(Note: a 25-30% decrease in availability set for $r=7,8$ when $y>2001$ and over moulting period $m=8-10$ as soft shell crabs had to be returned)</p>	Based on catch sex ratio (assumed known from Knuckey 1999) and catch size throughout the year, taking into account females begin to move offshore in last quarter of year.
$S_{r,y,m,a}^{sp,sex}$	<i>Mudcr</i> : selectivity: value 0 for sex =females in Qld ($r=1-6$), whereas sex =females in NT ($r=7-8$) value is zero for $m=1$, 50% for $m=2-3$, then 1 (before 2006) or zero for $m=1-3$, 50% for $m=4-5$, then 1 (from 2006); sex =male full selectivity except month 1(NT) and months 1-3 (Qld)	Based on average size at which mud crabs become vulnerable to fishing gear

Table A27. Model equations for mangroves (mangr), seagrass (seagr), microphytobenthos (Mb), Meiofauna (Mf) (definitions of variables and parameters are shown in Table A28 and Table A29

Model section	Equation	No.
Weekly Biomass update		
Weekly updates (seagr)	$B_{r,y,w+1}^{seagr} = B_{r,y,w}^{seagr} + G_{r,y,w}^{seagr} B_{r,y,w}^{seagr} \cdot \left(1 - B_{r,y,w}^{seagr} / K_r^{seagr}\right)^\gamma - M_{r,y,w}^{seagr} B_{r,y,w}^{seagr}$ <p>where $\gamma = 1$ for base Schaefer model and $\gamma = 3$ for Pella-Tomlinson</p>	8a
Weekly updates (mangr)	$B_{r,y,w+1}^{mangr} = B_{r,y,w}^{mangr} + G_{r,y,w}^{mangr} B_{r,y,w}^{mangr} \cdot \left(1 - B_{r,y,w}^{mangr} / K_{r,y}^{mangr}\right)^\gamma - M_{r,y,w}^{mangr} B_{r,y,w}^{mangr}$ <p>where $\gamma = 1$ for base Schaefer model and $\gamma = 3$ for Pella-Tomlinson</p>	8b
Mangr carrying capacity	$K_{r,y}^{mangr} = K_r^{mangr} \cdot \frac{\sum flow_{r,y,w}}{52}$ <p>where average flow is computed over the previous 52 weeks, and the carrying capacity multiplier is constrained to fall between 0.9 and 1.1</p>	8c
Weekly updates for group <i>g</i> (Mb, Mf)	$B_{r,y,w+1}^g = B_{r,y,w}^g + G_{r,y,w}^g B_{r,y,w}^g - M_{r,y,w}^g B_{r,y,w}^g - M_{r,y,w}^{scour} B_{r,y,w}^g$	8d
Weekly mortality rates		
Mortality rate (seagr)	$M_{r,y,w}^{seagr} = M_{base}^{seagr} + M_{r,y,w}^{cyclone}$ $M_{r,y,w}^{cyclone} = \xi^{seagr} \cdot Cycl_{r,y,w}^{category} \cdot Cycl_{r,y,w}^{impact}$	9a
Mortality rate (mangr)	$M_{r,y,w}^{mangr} = M_{base}^{mangr} + M_{r,y,w}^{cyclone,mangr} + M_{y,w}^{sealevel}$ $M_{r,y,w}^{cyclone,mangr} = \xi^{mangr} \cdot Cycl_{r,y,w}^{category} \cdot Cycl_{r,y,w}^{impact}$ $M_{y,w}^{sealevel} = 0.5 \text{ if } SL < 0.1 \text{ else } 0$	9b
Scour	$M_{r,y,w}^{scour} = 0.5 \text{ if } flow_{r,y,w}^{anom} > 3 \text{ else } 0.1$	9c
Growth rates		
Growth rate (seagr)	$G_{r,y,w}^{seagr} = \mu^{seagr} \cdot \delta_{r,y,w}^{SST} \cdot \delta_{r,y,w}^{light}$	10a
Temperature effect on	$\delta_{r,y,w}^{SST} = -0.01 \cdot (SST_{r,w} - 26)^2 + 1 \quad \text{and} \quad SST_{r,w} = 4.73 \cdot \sin(0.15(w + 4.5)) + 26.08$ <p>Where 26°C is the median SST assumed to be the optimal temperature at which function has value 1, with an inverted parabola fitted with fixed slope input value such</p>	10b

growth (seagr, Mb)	that 1°C change in optimal SST results in approx. 1% change in multiplier, 2°C change results in ca. 4% change etc	
Light effect on seagr	$\delta_{r,y,w}^{light} = \sqrt{solar_{r,y,w}} \times \frac{1}{flow_{r,y,w}^{std}} \cdot Cycl_{r,y,w}$ <p>if $flow \geq 1$ else $\delta_{r,y,w}^{light} = \sqrt{solar_{r,y,w}}$</p> <p>unless $flow < 1$ and $Cycl_{r,y,w} > 0$ then $\delta_{r,y,w}^{light} = \sqrt{solar_{r,y,w}} \times \frac{1}{(1 + Cycl_{r,y,w})}$</p>	10c
Growth rate (mangr)	$G_{r,y,w}^{mangr} = \mu^{mangr} \cdot \delta_{r,y,w}^{AT} \cdot \delta_{r,y,w}^{solar} \cdot flow_{r,y,w}^{anom}$	10d
Air Temperature multiplier function (mangr)	$\delta_{r,y,w}^{AT} = -0.02 * (AT_{r,y,w} - 31.93)^2 + 1$ <p>Where 31.93°C is the median temperature assumed to be the optimal temperature at which function has value 1, with an inverted parabola fitted with fixed slope input value such that 1°C change in optimal AT results in approx. 2% change in multiplier, 2°C change results in ca. 7% change etc</p>	10e
Growth rate (Mb)	$G_{r,y,w}^{Mb} = \mu^{Mb} \cdot \delta_r^N \cdot \delta_{r,y,w}^{sal} \cdot \delta_{r,y,w}^{SST} \cdot solar_{r,y,w}$	10f
Growth rate (Mf)	$G_{r,y,w}^{Mf} = \mu^{Mf} \cdot \delta_{r,y,w}^{pp} \cdot \delta_{r,y,w}^{sal}$ $\delta_{r,y,w}^{pp} = 0.01 \cdot B_{r,y,w}^{Mb} \cdot \left(1 - \frac{B_{r,y,w}^{Mb}}{B_{r,1,1}^{Mb}} \right)$	10g
Salinity multiplier function	$\delta_{r,y,w}^{sal} = \left(-0.047 * (Sal_{r,y,w})^2 + 1.794 \cdot Sal_{r,y,w} - 0.66 \right) / 14.5$ <p>With minimum value 0.001 (Appendix 14)</p>	10e
Flood productivity boost multiplier	$prod_y = \frac{\sum_{r=2}^5 \sum_w scour_{r,w} \cdot B_{r,y,w}^{Mb}}{prod}$ <p>with scour=0.5 if $\sigma_{r,y,w}^{flow-sp} > 3$ else scour=0.1; average $prod$ computed in model</p>	10f

Table A28. Description of the variables of the model for mangroves (mangr), seagrass (seagr), microphytobenthos (Mb), Meiofauna (Mf)

Parameter	Description	Input, computed, estimated
Group		
$B_{r,y,w}^{seagr}$	Relative biomass of seagr in region r and week w of year y (not absolute value as relative to arbitrary 1970 starting value)	Computed from Eq. 8a
$B_{r,y,w}^{mangr}$	Relative biomass of mangr in region r and week w of year y (not absolute value as relative to arbitrary 1970 starting value)	Computed from Eq. 8a
$B_{r,y,w}^g$	Relative biomass of group g (Mb or Mf) in region r and week w of year y (not absolute value as relative to arbitrary 1970 starting value)	Computed from Eq. 8a
K_r^{seagr}	Seagr carrying capacity in region r	Fixed input
$K_{r,y}^{mangr}$	Time-varying mangrove carrying capacity in year y and region r , which is computed by multiplying the base carrying capacity value by the flow anomaly averaged over the past year (as a proxy for average annual rainfall), and with a constraint added that the carrying capacity can't vary by more than 10% up or down	Computed from Eq. 8c
Mortality rates		
$M_{r,y,w}^{seagr}, M_{r,y,w}^{mangr}$	The total seagr or mangr mortality rate in region r and week w of year y	Computed from Eqns 9a, 9b
$M_{base}^{seagr}, M_{base}^{mangr}$	A base fixed mortality rate for seagr or mangr	Input (Table A29)
$M_{a,y,w}^{cyclone}, M_{a,y,w}^{cyclone}$ $M_{r,y,w}^{cyclone}, M_{r,y,w}^{cyclone,mangr}$	Additional mortality rate for seagr or mangr that depends on cyclone category and impact	Computed from Eq. 9a
$Cycl_{r,y,w}^{category}$	Cyclone category score in region r and week w of year y	Appendix 6
$Cycl_{a,y,w}^{impact}, Cycl_{r,y,w}^{impact}$	Cyclone impact score in region r and week w of year y	Appendix 6
ξ^{seagr}	Constant of proportionality for cyclone mortality function	Input
$M_{y,w}^{sealevel}$	Mortality term due to sea level changes	Computed from Eq. 9b

$M_{r,y,w}^g$	Mortality rate for group g (Mb or Mf) region r and week w of year y	Input (Table A29)
Growth rates		
$G_{r,y,w}^{seagr}$	The seagr growth rate in region r and week w of year y	Computed from Eq. 10a
$G_{r,y,w}^{mangr}$	The mangr growth rate in region r and week w of year y	Computed from Eq. 10d
$G_{r,y,w}^{Mb}$	The Mb growth rate in region r and week w of year y	Computed from Eq. 10f
$G_{r,y,w}^{Mf}$	The Mf growth rate in region r and week w of year y	Computed from Eq. 10g
μ^{Mb}, μ^{Mf}	Fixed constant used as base growth rate for Mb or Mf	Input (Table A29)
μ^{seagr}, μ^{mangr}	Fixed constant used as base growth rate for seagr or mangr	Input (Table A29)
$\delta_{r,y,w}^{SST}$	Growth modifier that depends on Sea Surface Temperature (SST) for seagr	Computed from Eq. 10b
$\delta_{r,y,w}^{light}$	Growth modifier for seagr that depends on relative light attenuation level	Computed from Eq. 10c
$solar_{r,y,w}$	Solar exposure in region r and week w of year y	BOM measurements of daily solar exposure that measure the total solar energy falling on a horizontal surface from midnight to midnight
$\delta_{r,y,w}^{AT}$	A growth modifier for mangr based on Air Temperature (AT) in region r and week w of year y	Daily maximum air temperature data are available from BOM
$\delta_{r,y,w}^{sal}$	A growth modifier for Mb and Mf based on salinity level, which is computed based on a fitted relationship with flow, as described in Appendix 6	Data and fitted function
$\delta_{r,y,w}^{pp}$	A primary production (pp) growth modifier for Mf which uses as an index a relationship based on Mb abundance	Eqn.10g

$flow_{r,y,w}^{anom}$	growth modifier based on the flow anomaly in region r and week w of year y , with lower and upper values capped at 0.9 and 1.1 respectively, or at alternative settings in ensemble to allow bigger influence on dynamics eg 0.4 and 1.3	Based on end-of-system flow inputs
$prod_y; \overline{prod}$	Productivity boost multiplier in year y ; average productivity value	Computed using Eq. 10f and links to Eqn 1f in Model version 5

Table A29. Input values and sources for parameter settings not included in Model Ensemble Tables.

Parameter	Input Value (units)	Justification /Reference
Species		
$M_{a,y,w}^{seagr}$	0.15 wk ⁻¹	Roughly calibrated as per Appendix 15 but ensemble tests sensitivities
$M_{a,y,w}^{mangr}$	0.05 wk ⁻¹	Roughly calibrated as per Appendix 15 but ensemble tests sensitivities
K_a^{seagr}	Arbitrary set to 100 as each region relative to this amount	Relative changes only of interest
$K_{a,y}^{mangr}$	Starting values in year 1970 for each region: 20 56.85 7.23 7.23 9.39 20 20 20	Arbitrary relative values based on satellite area estimates available for some regions (Burford (pers comm))
μ^{seagr}	0.8	Set higher than mangr and adjusted to reasonably represent observed variation in seagrass
μ^{mangr}	0.2	Set lower than seagr and adjusted to reasonably represent observed variation in mangr
$M_{r,y,w}^{Mb}$	0.8	High weekly turnover rate assumed for Mb
$M_{r,y,w}^{Mf}$	0.5	Slightly lower than Mb weekly turnover rate assumed for Mf

Appendix 17 – MICE Ensemble Additional Model Results under baseline flows

Model fits (under baseline flows) for prawns, barramundi and mud crabs for model versions 2-5, are shown in the figures below. Estimated trajectories of sawfish interactions from trawl and gillnet fishing gear follow.

Prawns

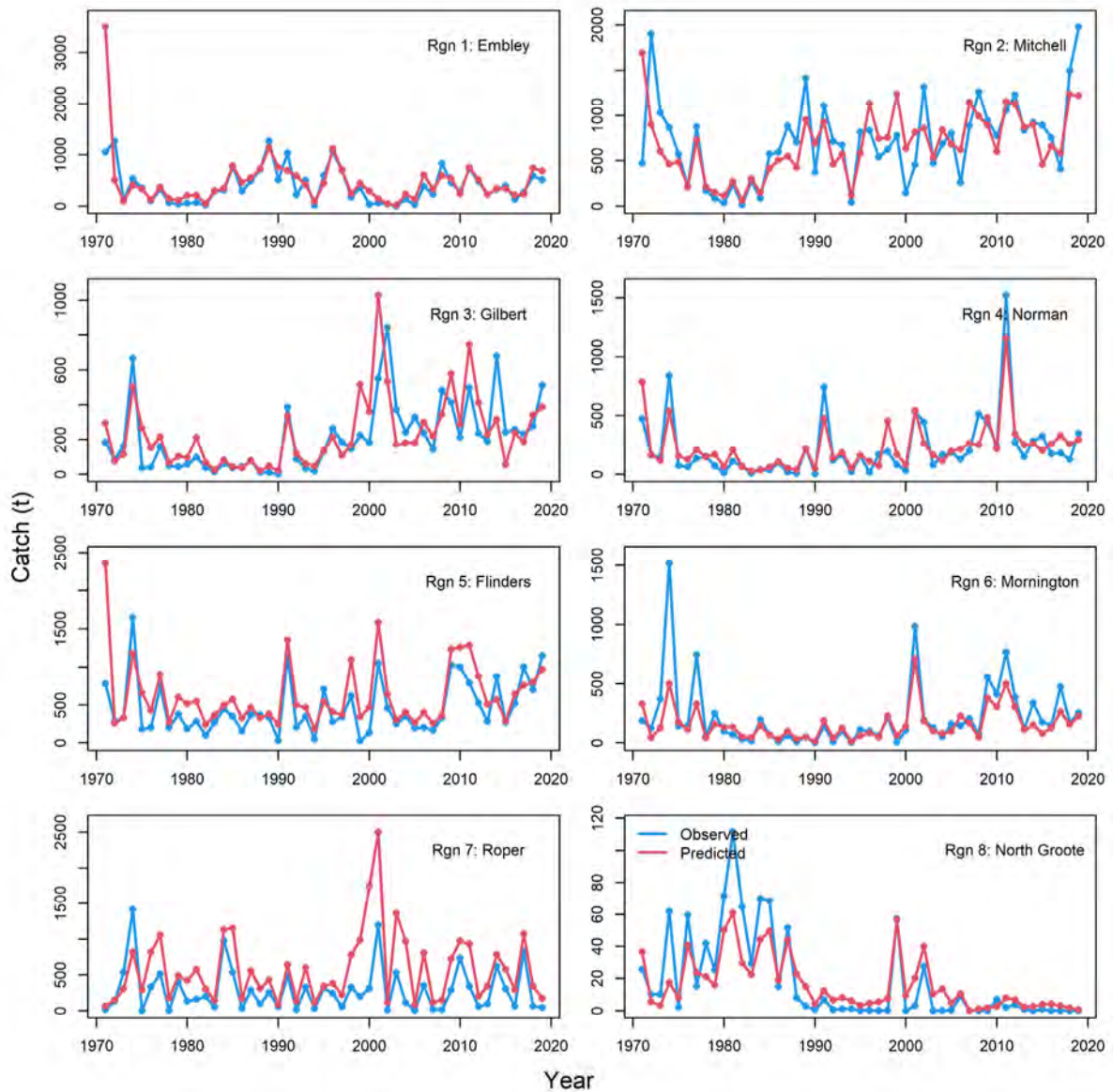


Figure A93. Comparison of the observed and model-predicted annual common banana prawn catch (tonnes) estimated using Model version 2 (driven by baseline flows, using different flow relationship parameters) and for each of the eight model regions as shown.

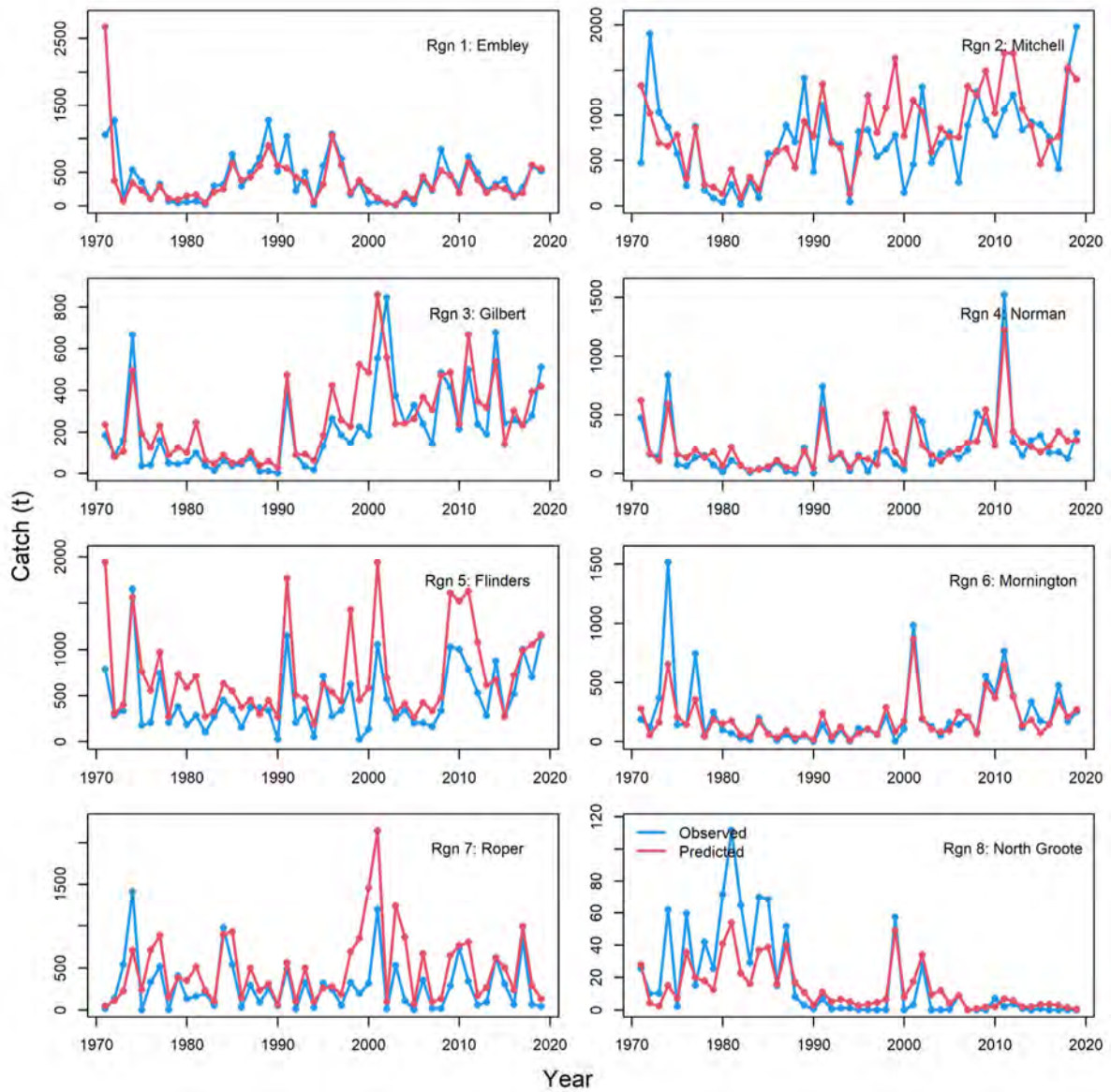


Figure A94. Comparison of the observed and model-predicted annual common banana prawn catch (tonnes) estimated using Model version 3 (driven by baseline flows, with alternative lag between flow and effect on recruitment) and for each of the eight model regions as shown.

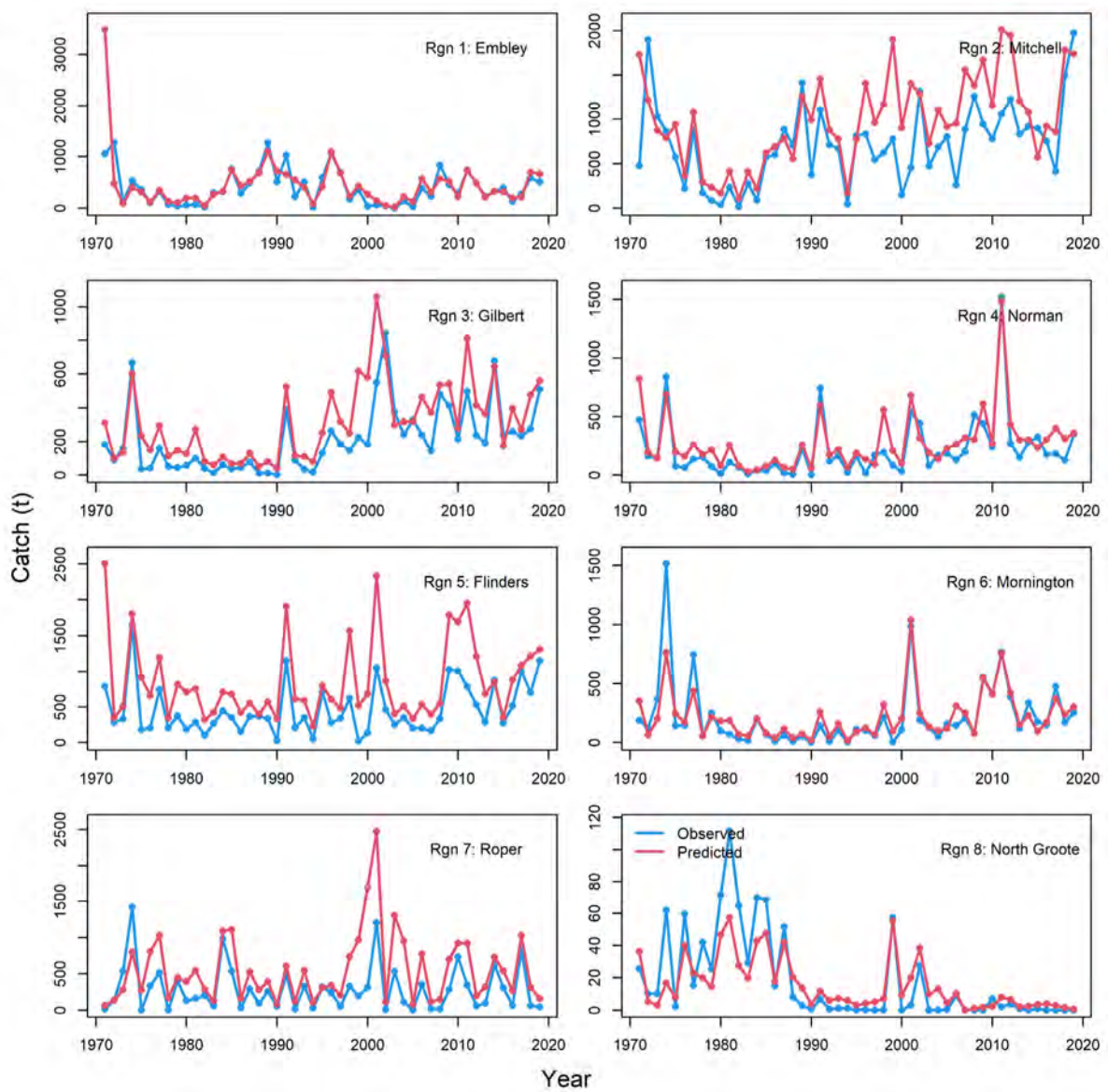


Figure A95. Comparison of the observed and model-predicted annual common banana prawn catch (tonnes) estimated using Model version 4 (driven by baseline flows, with model starting values proportional to recent catch proportions) and for each of the eight model regions as shown.

Barramundi

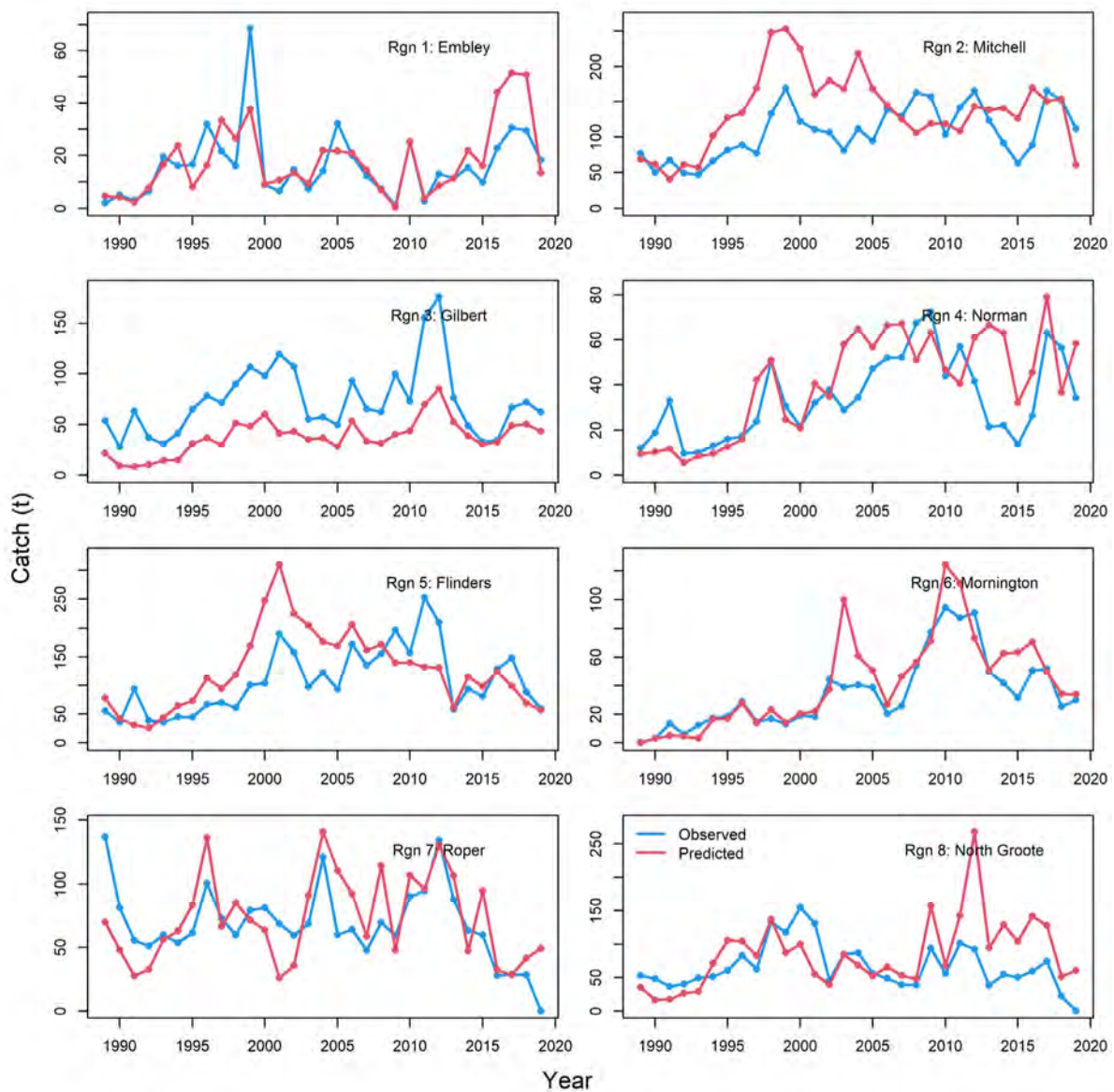


Figure A96. Comparison of the observed and model-predicted annual barramundi catch (tonnes) estimated using Model version 2 (baseline flows, with flow relationship parameter changed) and for each of the eight model regions as shown.

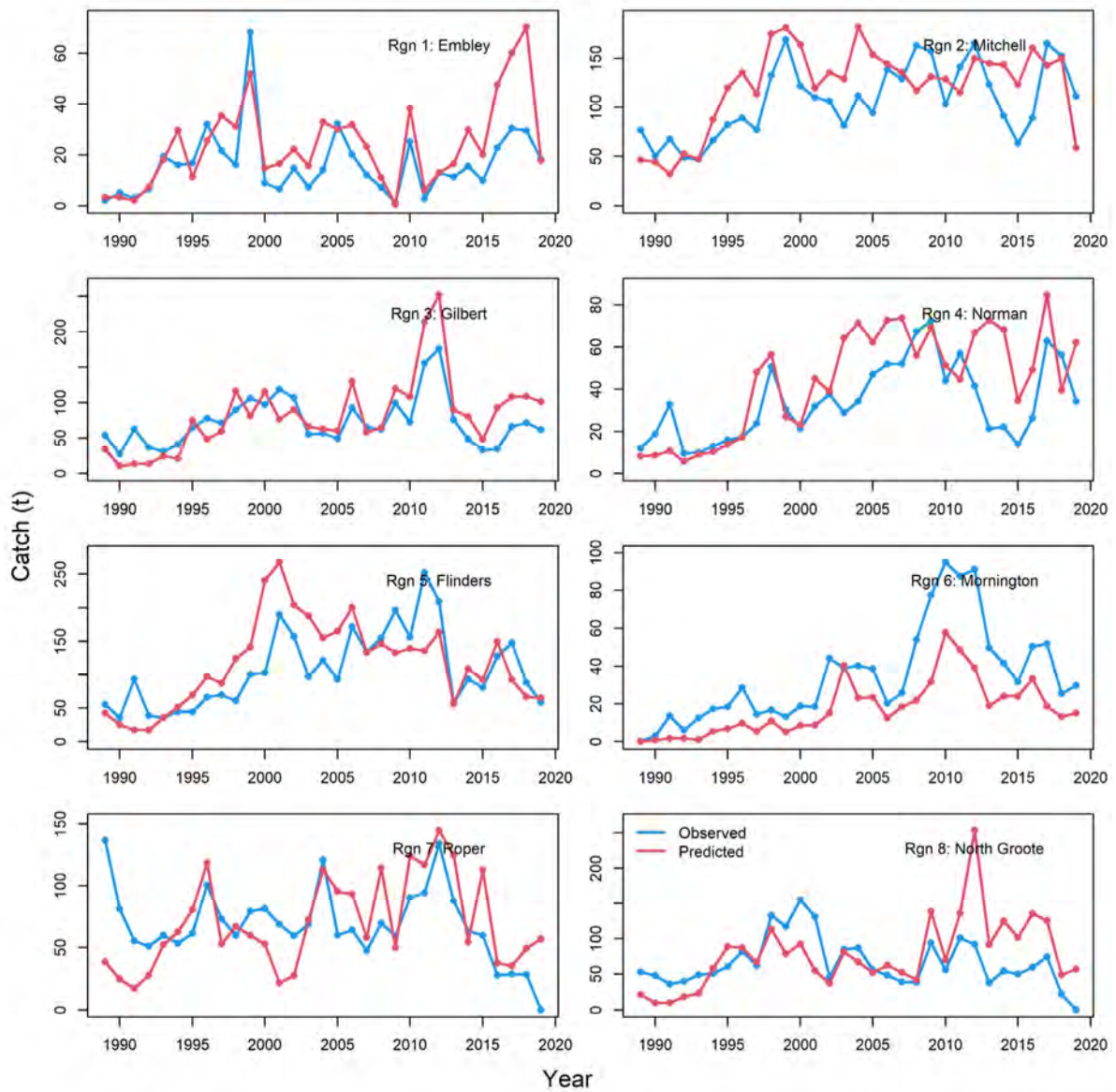


Figure A97. Comparison of the observed and model-predicted annual barramundi catch (tonnes) estimated using Model version 3 (baseline flows, and with large natural mortality M) and for each of the eight model regions as shown.

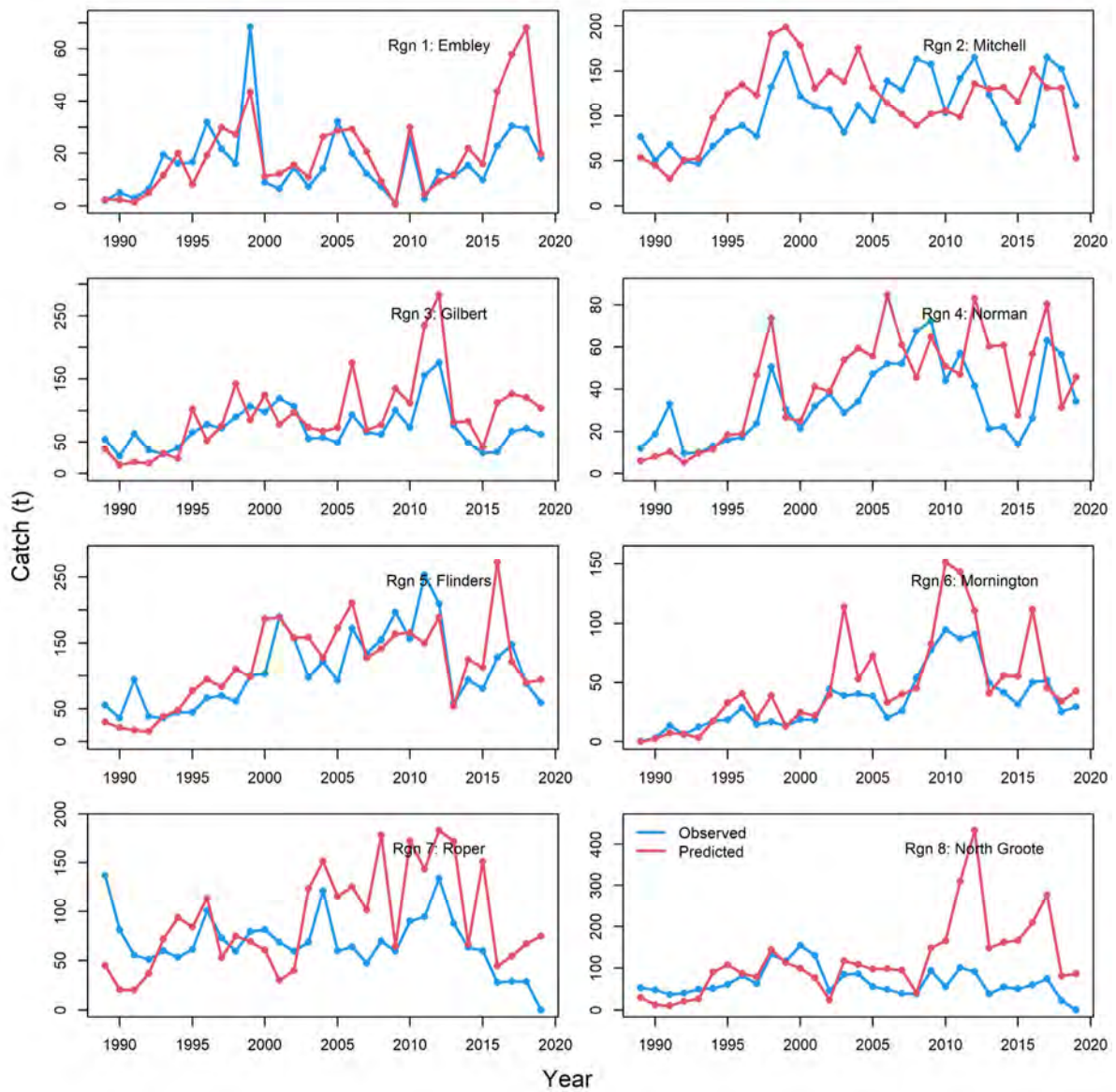


Figure A98. Comparison of the observed and model-predicted annual barramundi catch (tonnes) estimated using Model version 5 (baseline flows, and with no sex ratio influence on recruitment) and for each of the eight model regions as shown.

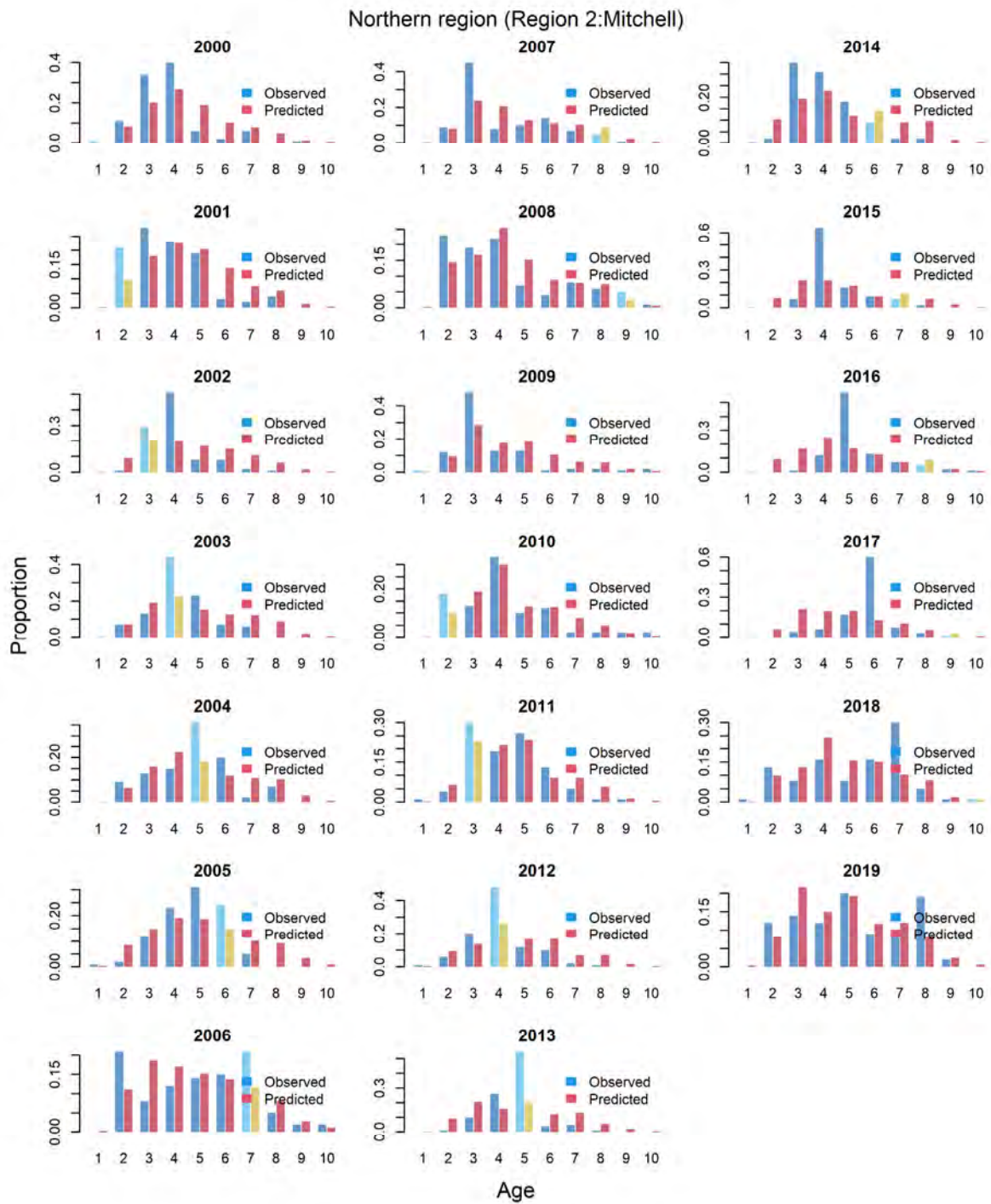


Figure A99. Comparison of the observed and model-predicted commercial catch-at-age proportions of barramundi in each age-class as shown estimated using Model version 1 (baseline flows) for the Mitchell River catchment region from 2000 to 2019.

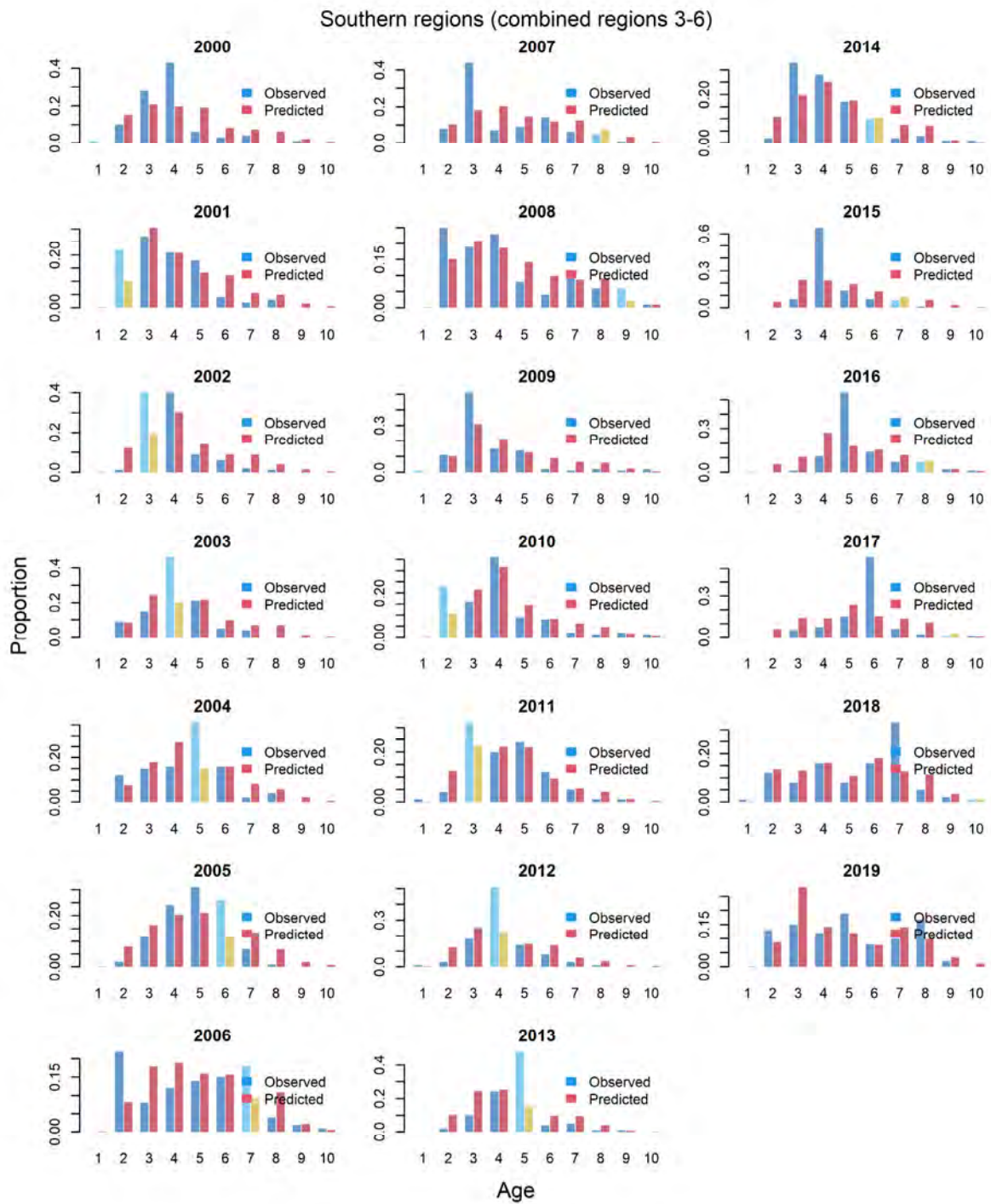


Figure A100. Comparison of the observed and model-predicted commercial catch-at-age proportions of barramundi in each age-class as shown estimated using Model version 1 (baseline flows) for Regions 3-6 from 2000 to 2019.

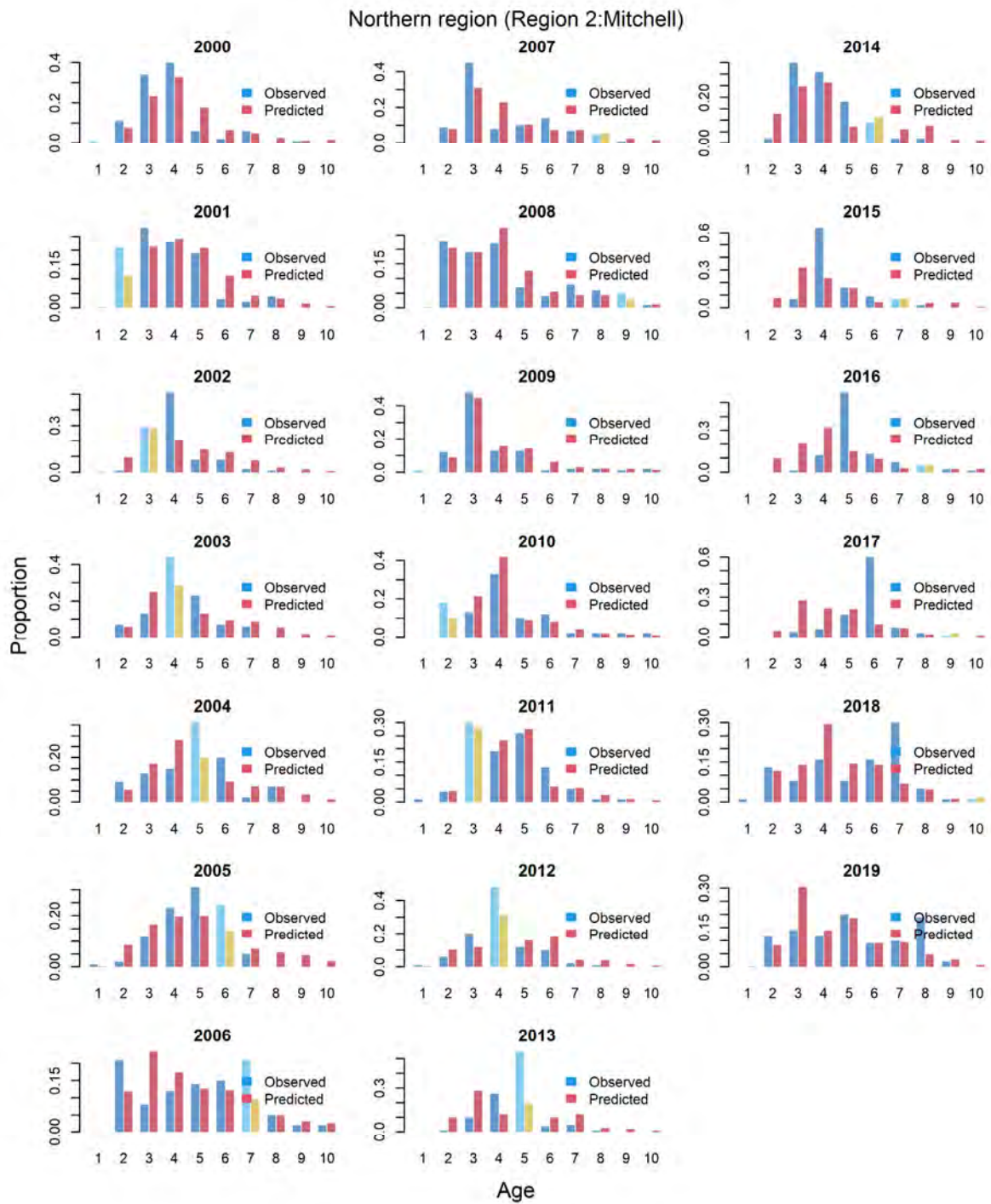


Figure A101. Comparison of the observed and model-predicted commercial catch-at-age proportions of barramundi in each age-class as shown estimated using Model version 2 (baseline flows, with flow parameter changed) for the Mitchell River catchment region from 2000 to 2019.

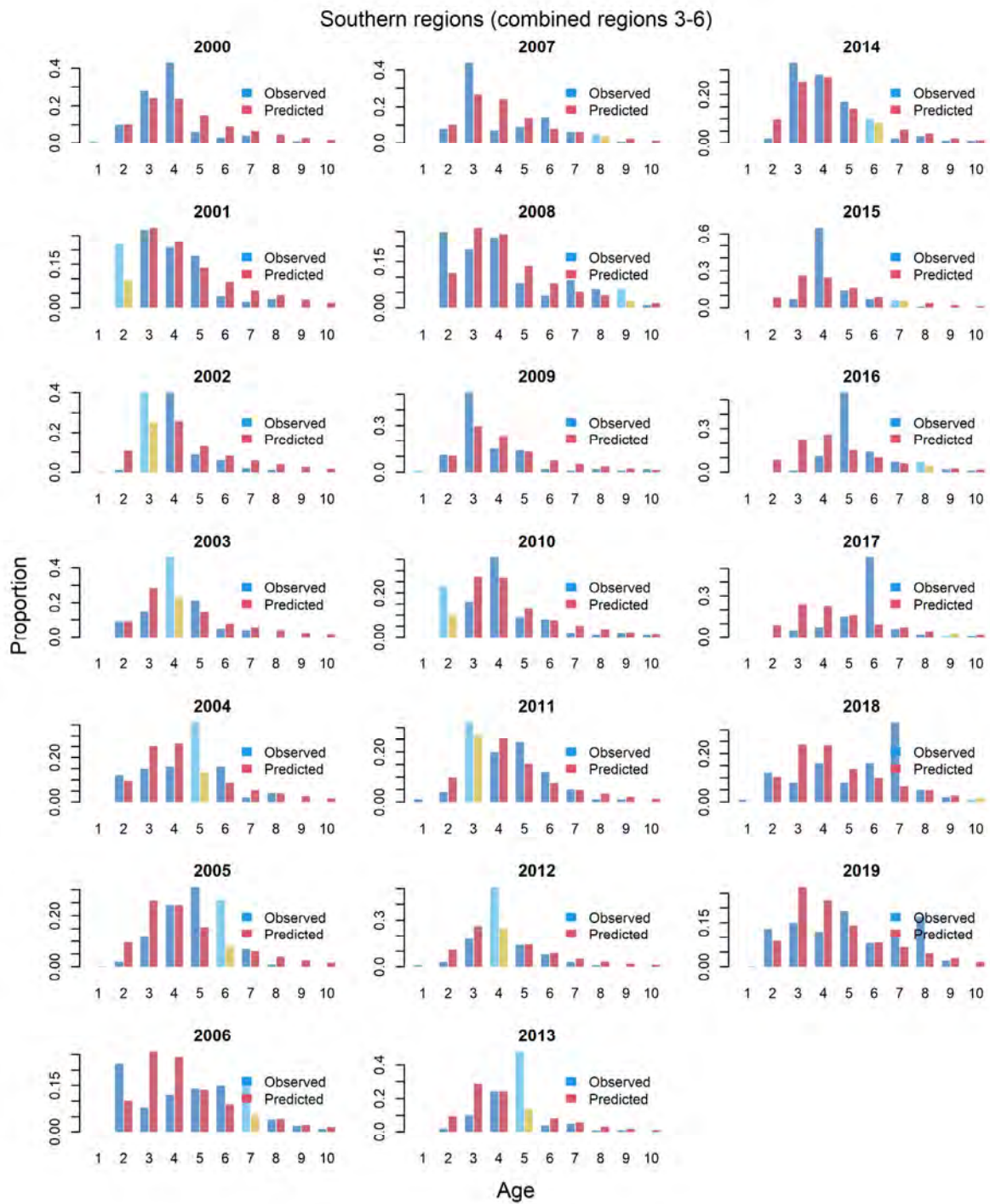


Figure A102. Comparison of the observed and model-predicted commercial catch-at-age proportions of barramundi in each age-class as shown estimated using Model version 2 (baseline flows, with flow parameter changed) for Regions 3-6 from 2000 to 2019.

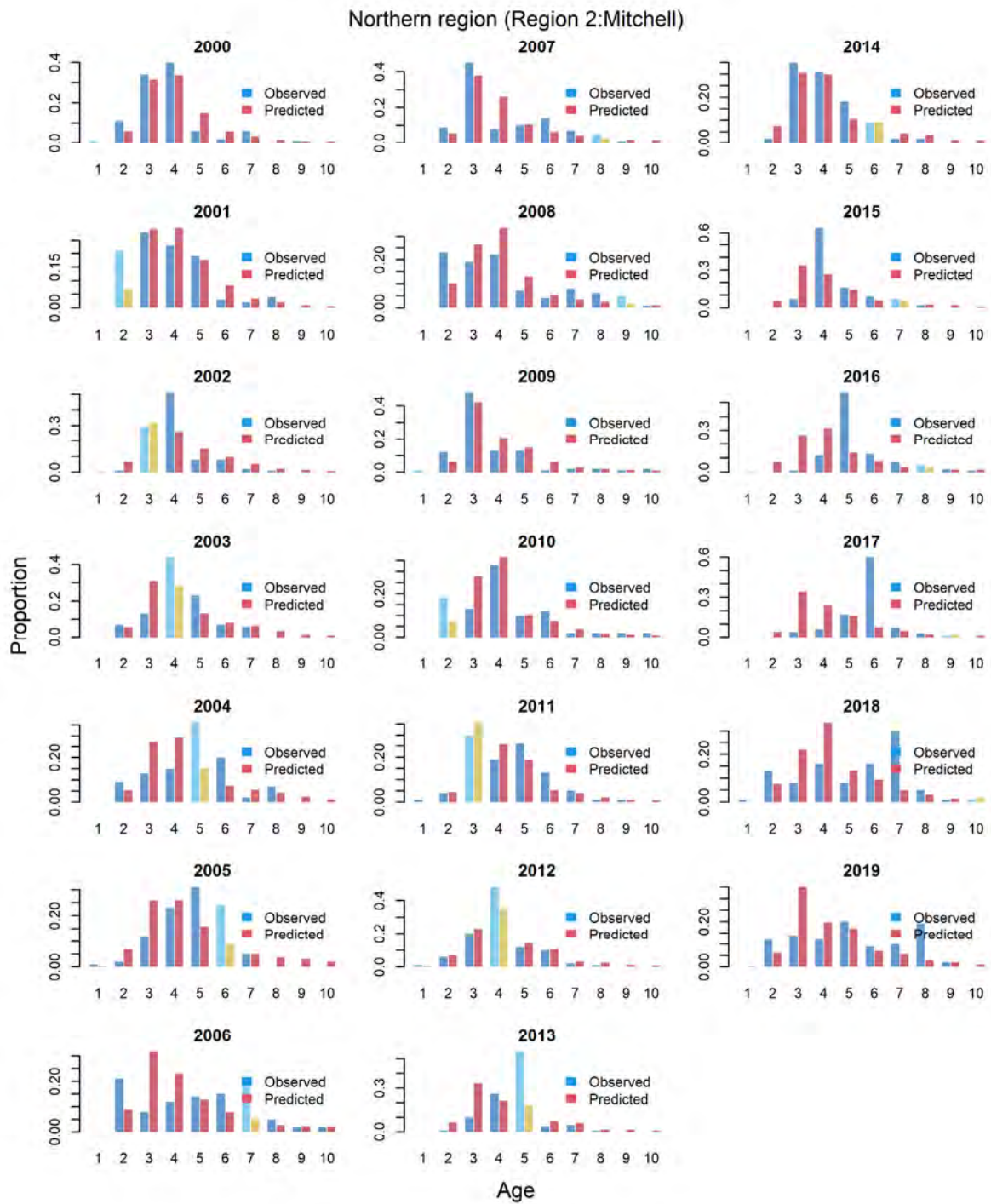


Figure A103. Comparison of the observed and model-predicted commercial catch-at-age proportions of barramundi in each age-class as shown estimated using Model version 3 (baseline flows, with larger natural mortality M) for the Mitchell River catchment region from 2000 to 2019.

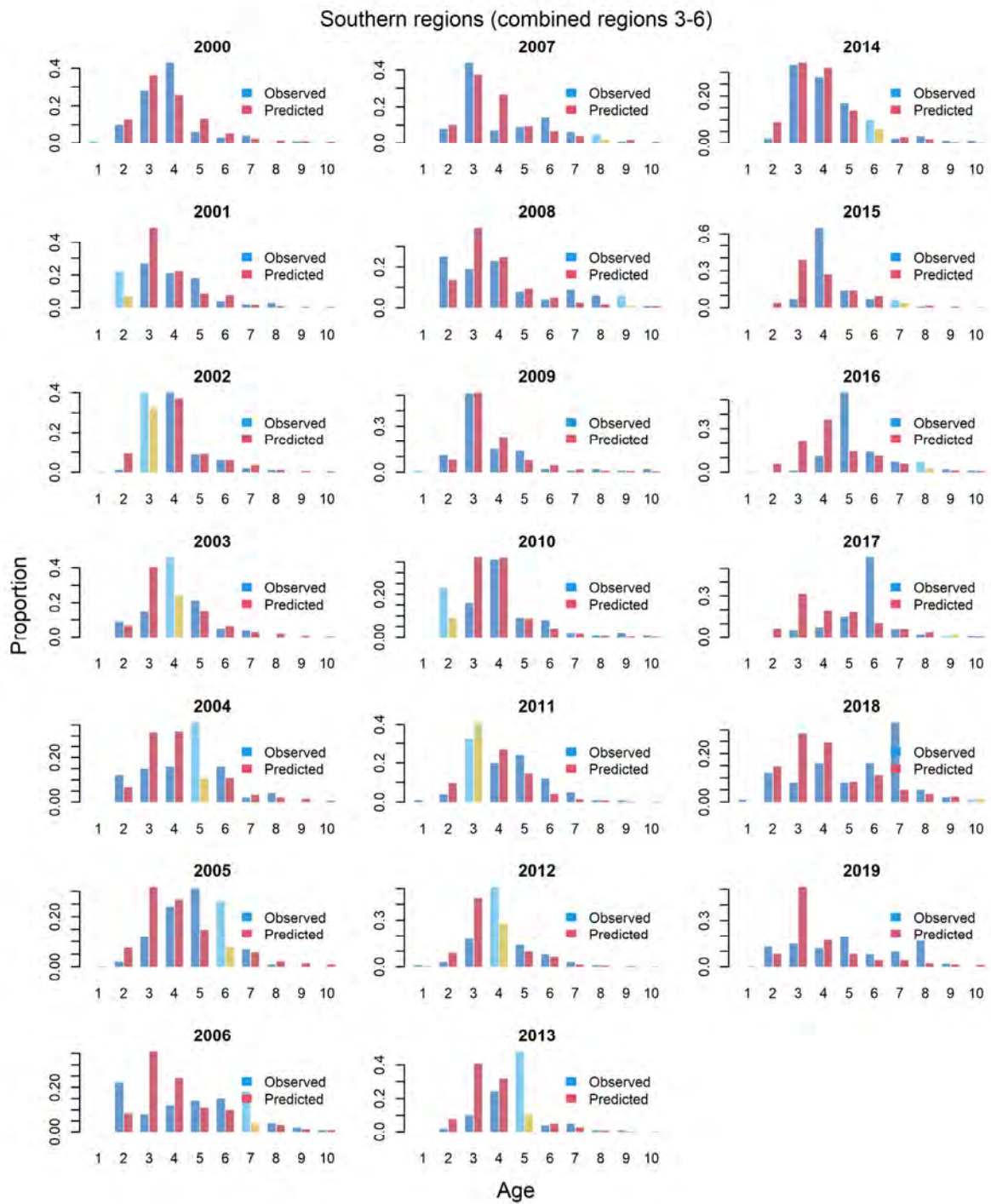


Figure A104. Comparison of the observed and model-predicted commercial catch-at-age proportions of barramundi in each age-class as shown estimated using Model version 3 (baseline flows, with larger natural mortality M) for regions 3-6 from 2000 to 2019.

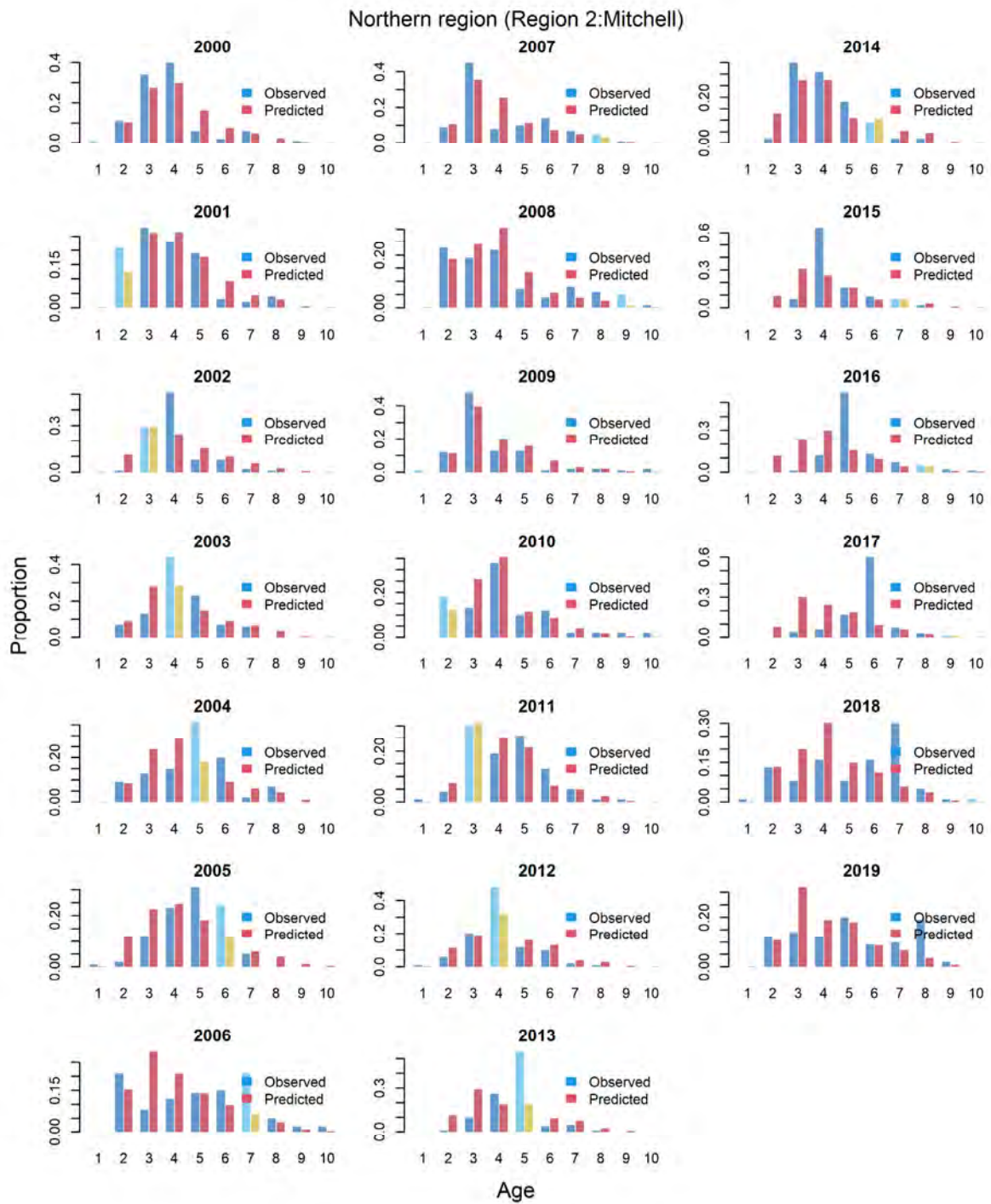


Figure A105. Comparison of the observed and model-predicted commercial catch-at-age proportions of barramundi in each age-class as shown estimated using Model version 5 (baseline flows, with no sex ratio influence on recruitment) for the Mitchell River catchment region from 2000 to 2019.

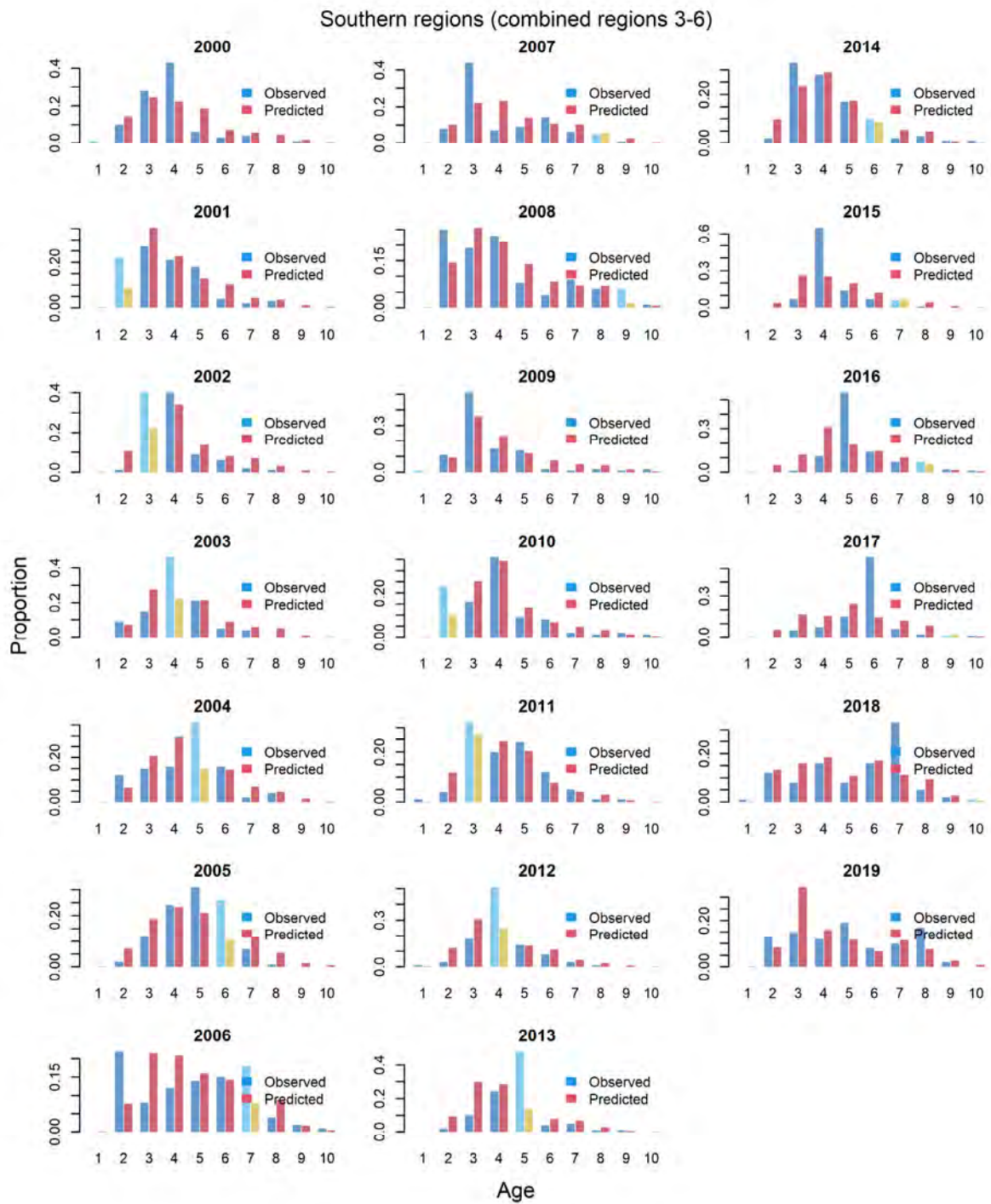


Figure A106. Comparison of the observed and model-predicted commercial catch-at-age proportions of barramundi in each age-class as shown estimated using Model version 5 (baseline flows, with no sex ratio influence on recruitment) for regions 3-6 from 2000 to 2019.

Mud crab

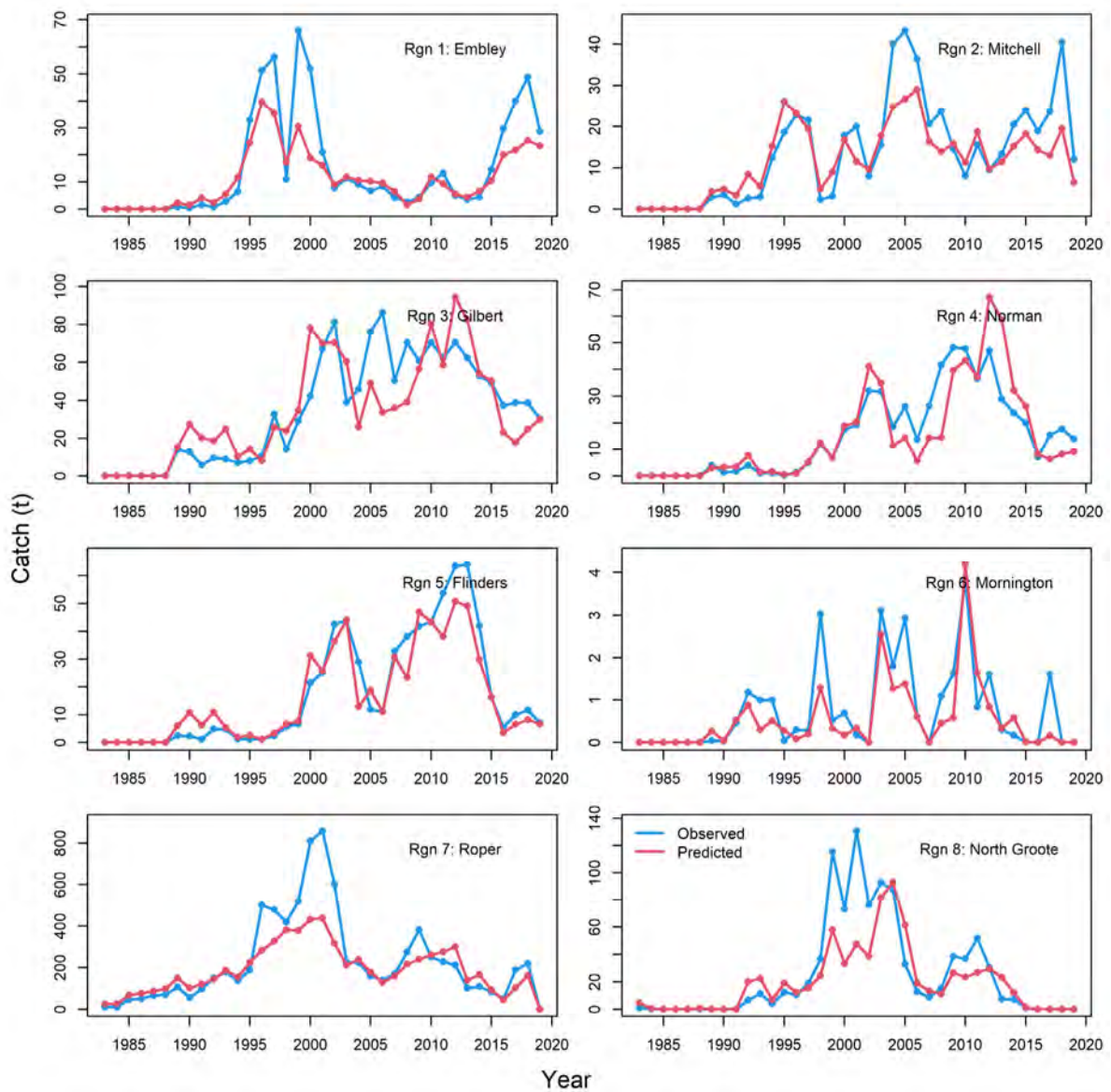


Figure A107. Comparison of the observed and model-predicted annual mud crab catch (tonnes) estimated using Model version 2 (driven by baseline flows with an altered flow relationship parameter, and other environmental predictors e.g. SOI for regions 3-6 and temperature for region 7) and for each of the eight model regions as shown.

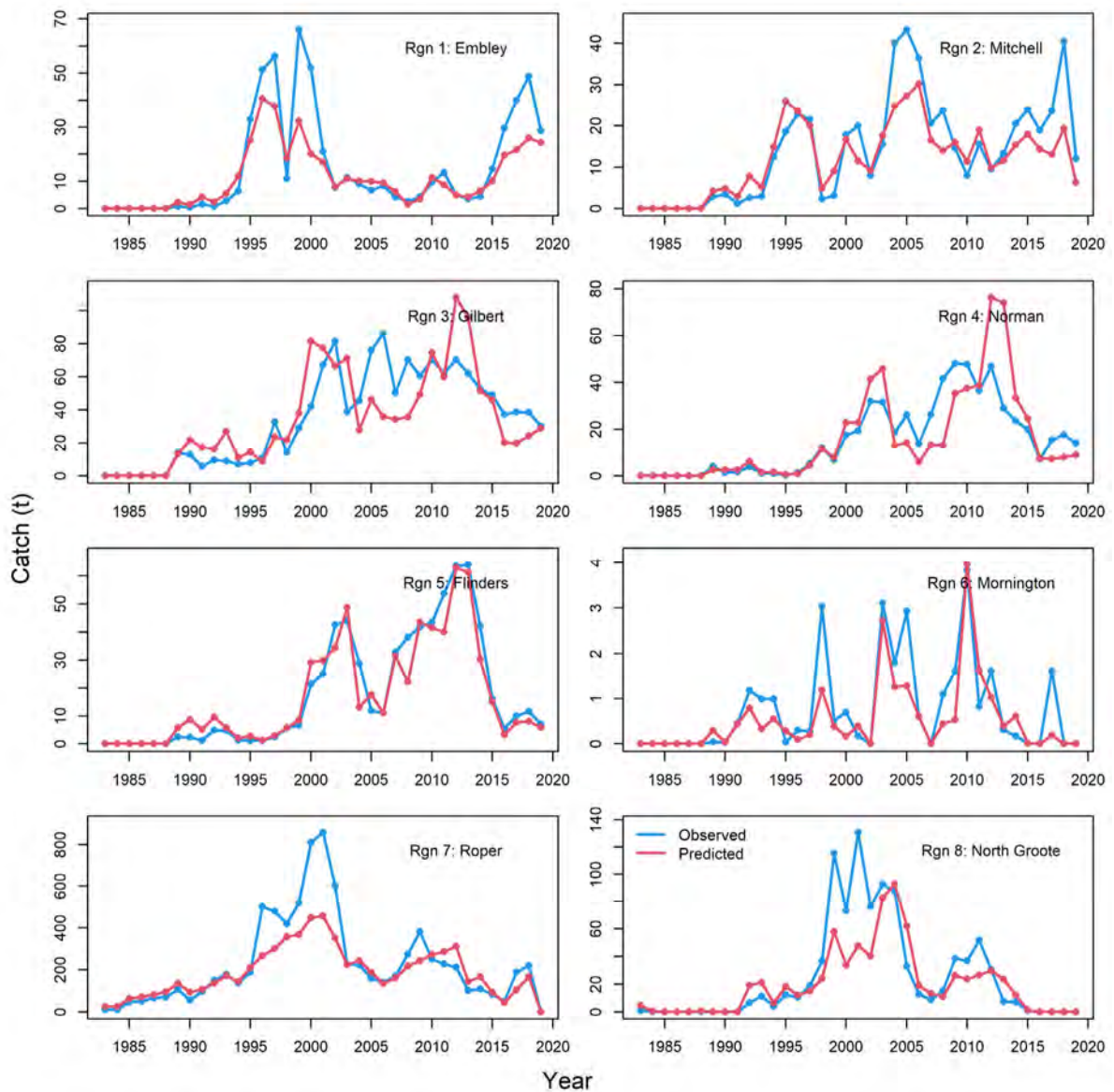


Figure A108. Comparison of the observed and model-predicted annual mud crab catch (tonnes) estimated using Model version 3 (driven by baseline flows and other environmental predictors e.g. SOI for regions 3-6 and temperature for region 7, and a larger natural mortality M) and for each of the eight model regions as shown.

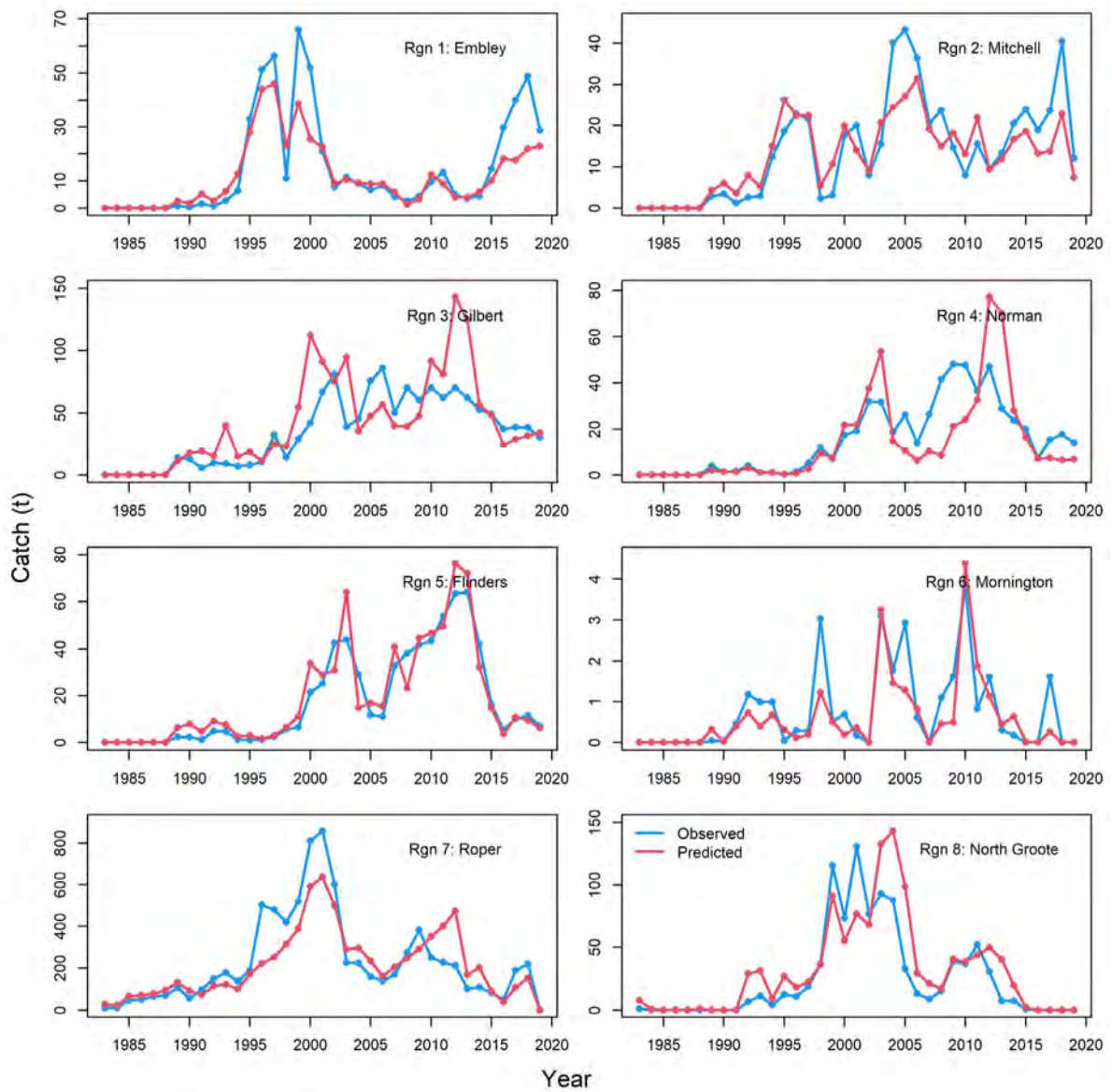


Figure A109. Comparison of the observed and model-predicted annual mud crab catch (tonnes) estimated using Model version 4 (driven by baseline flows and other environmental predictors e.g. SOI for regions 3-6 and temperature for region 7, and with model starting biomass doubled) and for each of the eight model regions as shown.

Sawfish

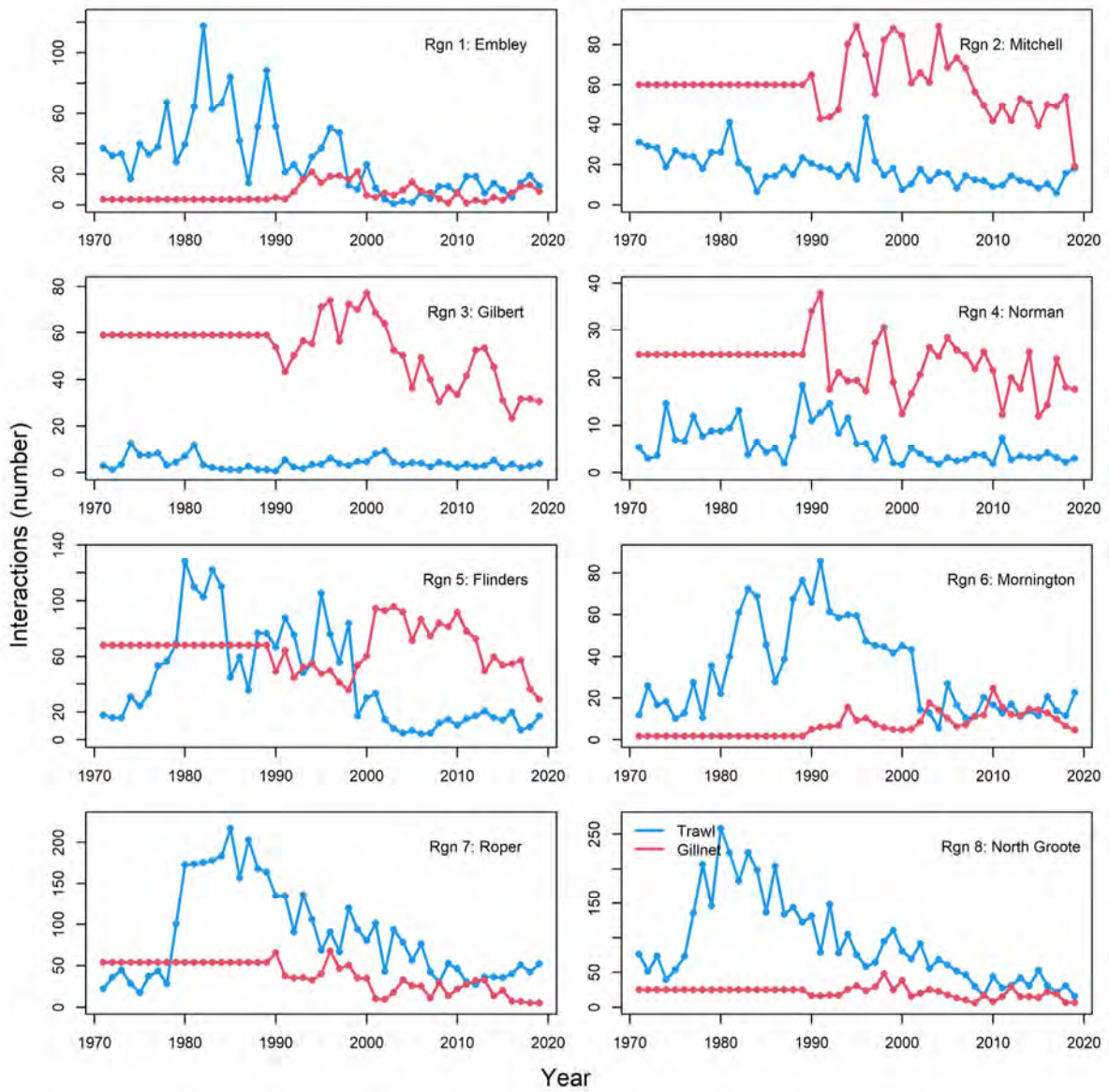


Figure A110. Model estimated sawfish (*Pristis pristis*) interactions (number) with trawl and gillnet fishing gear used in all Model versions for each of the eight model regions as shown.

Additional Results – sensitivity to adding trophic linkages (Model 6)

Below are some additional MICE results from additional sensitivity test (Model 6) on adding trophic linkages.

Table A30. MICE model parameter estimates, associated Hessian-based standard deviations (std), number of parameters estimated, negative log likelihood (-lnL) contributions and Akaike Information Criterion (AIC) scores across each of the five model versions for barramundi when comparing Model version 2 with Model version 6 which includes trophic linkages. Parameter estimates in italics are fixed parameters. Bsp = spawning biomass (t); CAA = catch-at-age; SOI = southern oscillation index; Mitch = Mitchell River; Gilb = Gilbert River; Norm = Norman River; Flind = Flinders River.

Parameter	Description	<u>Model 2 - flow relationship</u>		<u>Model6 - trophic</u>	
		value	std	value	std
(2) Barramundi					
Mbase	Natural mortality base	8.00E-03	1.32E-07	8.00E-03	1.32E-07
q(barra) (1)	Catchability	1.63E-04	2.12E-05	1.63E-04	2.12E-05
q(barra) (2)	Catchability	2.58E-05	1.89E-06	2.58E-05	1.89E-06
q(barra) (3-6)	Catchability	7.87E-05	4.03E-06	7.87E-05	4.03E-06
q(barra) (7-8)	Catchability	3.14E-04	6.42E-06	3.14E-04	6.42E-06
Thres (1)	Flow threshold parameter	0.268	0.138	0.268	0.138
Thres (2)	Flow threshold parameter	0.389	0.092	0.389	0.092
Thres (3)	Flow threshold parameter	2.472	0.847	2.472	0.847
Thres (4)	Flow threshold parameter	20.000	0.008	20.000	0.008
Thres (5-6)	Flow threshold parameter	5.535	4.063	5.535	4.063
Thres (7-8)	Flow threshold parameter	20.000	0.014	20.000	0.014
del_barra_S	Age-selectivity par (Southern)	4.63E-01	2.94E-02	4.63E-01	2.94E-02
sfa_barra_S	Age-selectivity par (Southern)	1.88E-01	4.57E-02	1.88E-01	4.57E-02
del_barra_N	Age-selectivity par (Northern)	1.00E+00	4.11E-04	1.00E+00	4.11E-04
sfa_barra_N	Age-selectivity par (Northern)	1.00E-02	8.87E-05	1.00E-02	8.87E-05
MaxThres	Flow threshold parameter	<i>0.80</i>	–	<i>0.80</i>	–
<i>hpred</i>	Predator-prey interaction parameter	-	-	0.85	3.67e-04
No. parameters		15		16	
Likelihood contributions		value	sigma	value	sigma
-lnL: Catch (Region 1)		79.3	0.9	83.4	0.9
-lnL: Catch (Region 2)		106.2	1.0	139.3	1.2
-lnL: Catch (Region 3)		152.6	1.2	124.3	1.1
-lnL: Catch (Region 4)		107.7	1.0	113.9	1.0
-lnL: Catch (Region 5)		99.8	1.0	131.1	1.1
-lnL: Catch (Region 6)		94.4	0.9	96.1	0.9
-lnL: Catch (Region 7)		81.2	0.9	96.3	0.9
-lnL: Catch (Region 8)		105.9	1.0	119.9	1.1
-lnL: CAA (Northern)		45.5	0.2	42.9	0.2
-lnL: CAA (Southern)		58.2	0.2	60.0	0.2
-lnL: overall		931.0	–	1007.2	–

AIC		1891.9	-	2046.4	-
<u>Other fixed parameters</u>					
Mcon	Slope of age-dependent mortality	0.045		0.045	
Sex ratio		Calculated		Calculated	

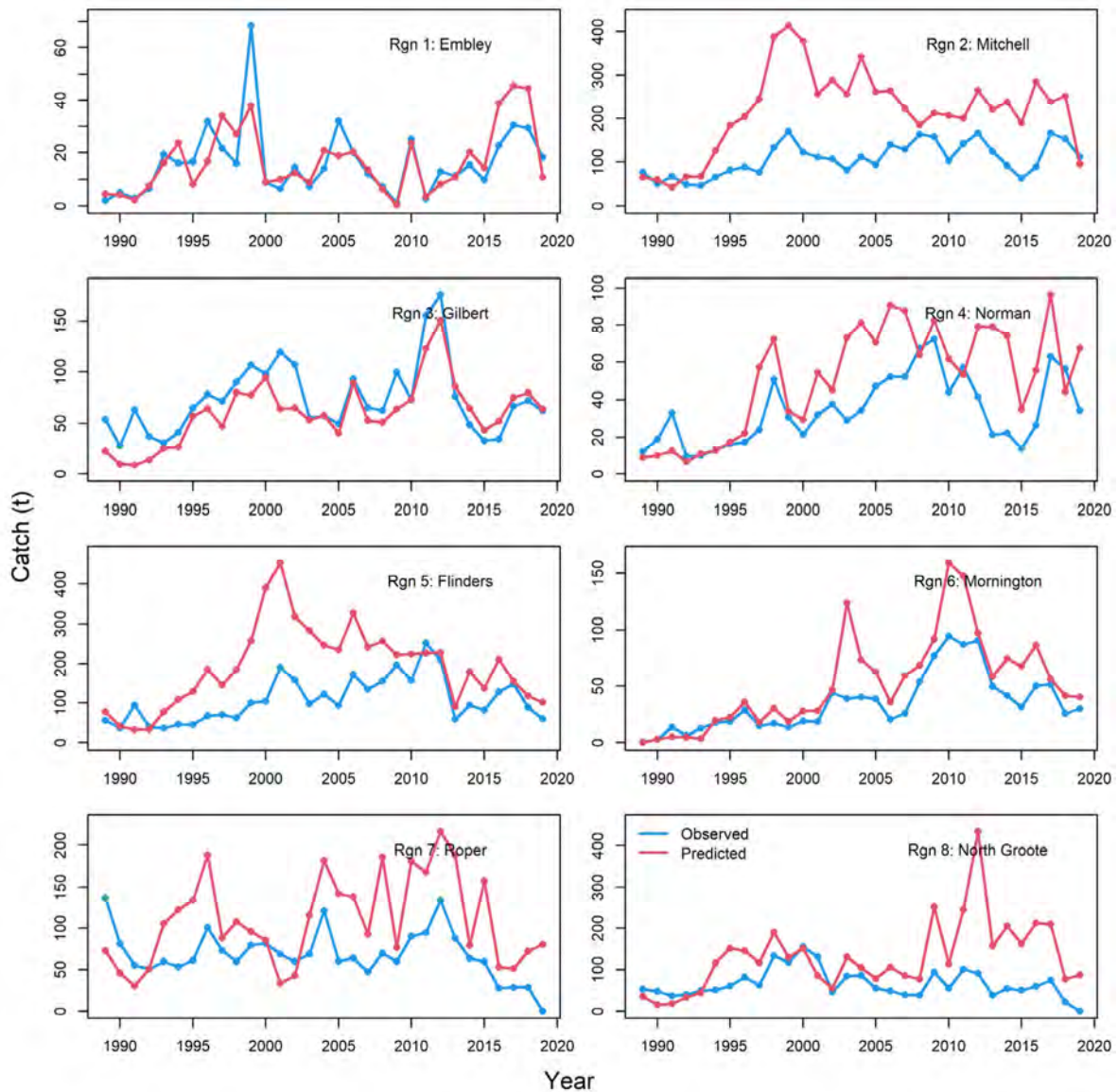


Figure A111. Comparison of the observed and model-predicted annual barramundi catch (tonnes) estimated using Model version 6 (trophic-linked version) and for each of the eight model regions as shown

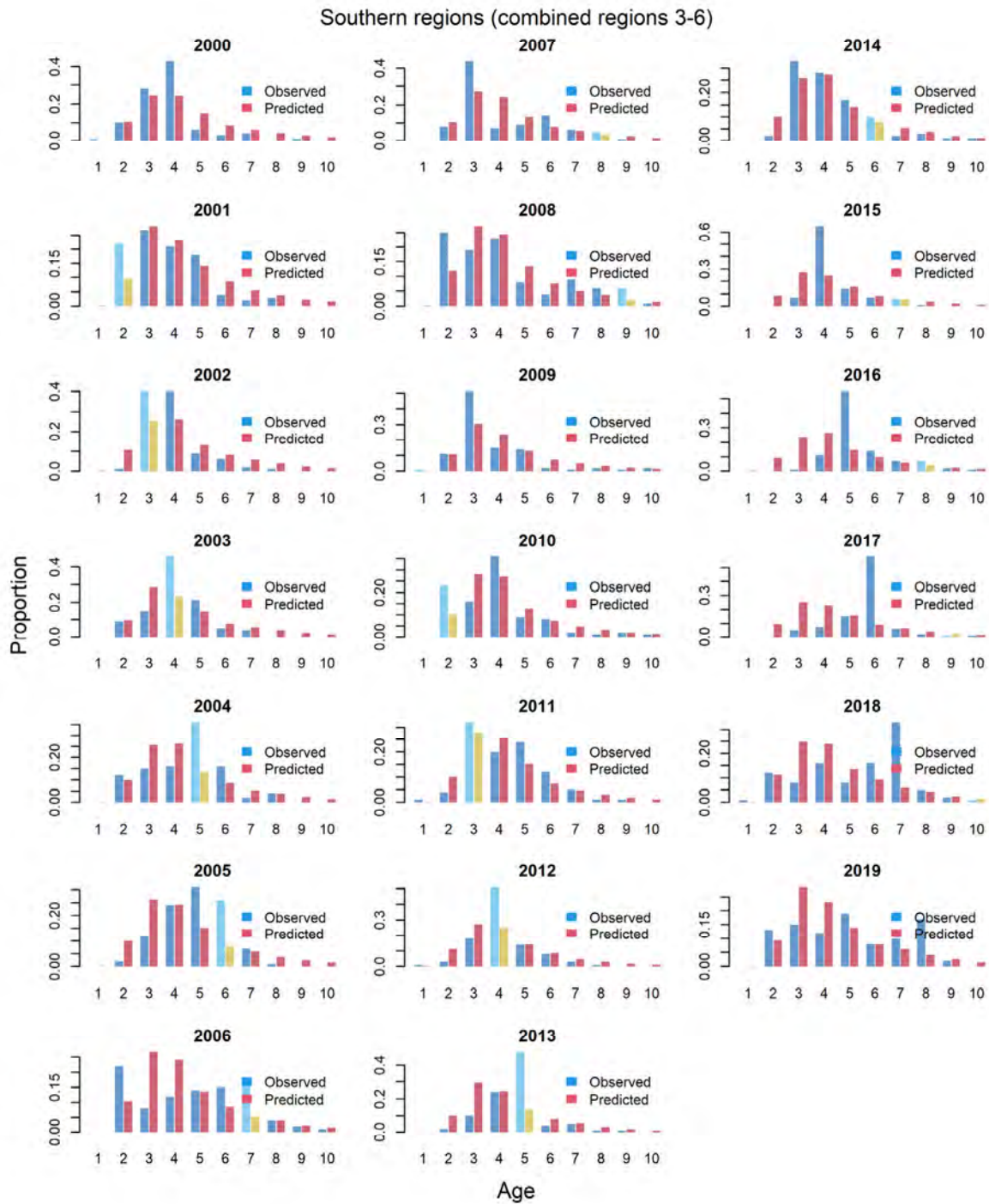


Figure A112. Comparison of the observed and model-predicted commercial catch-at-age proportions of barramundi in each age-class as shown estimated using Model version 6 (trophic-linked version) for Regions 3-6 from 2000 to 2019.

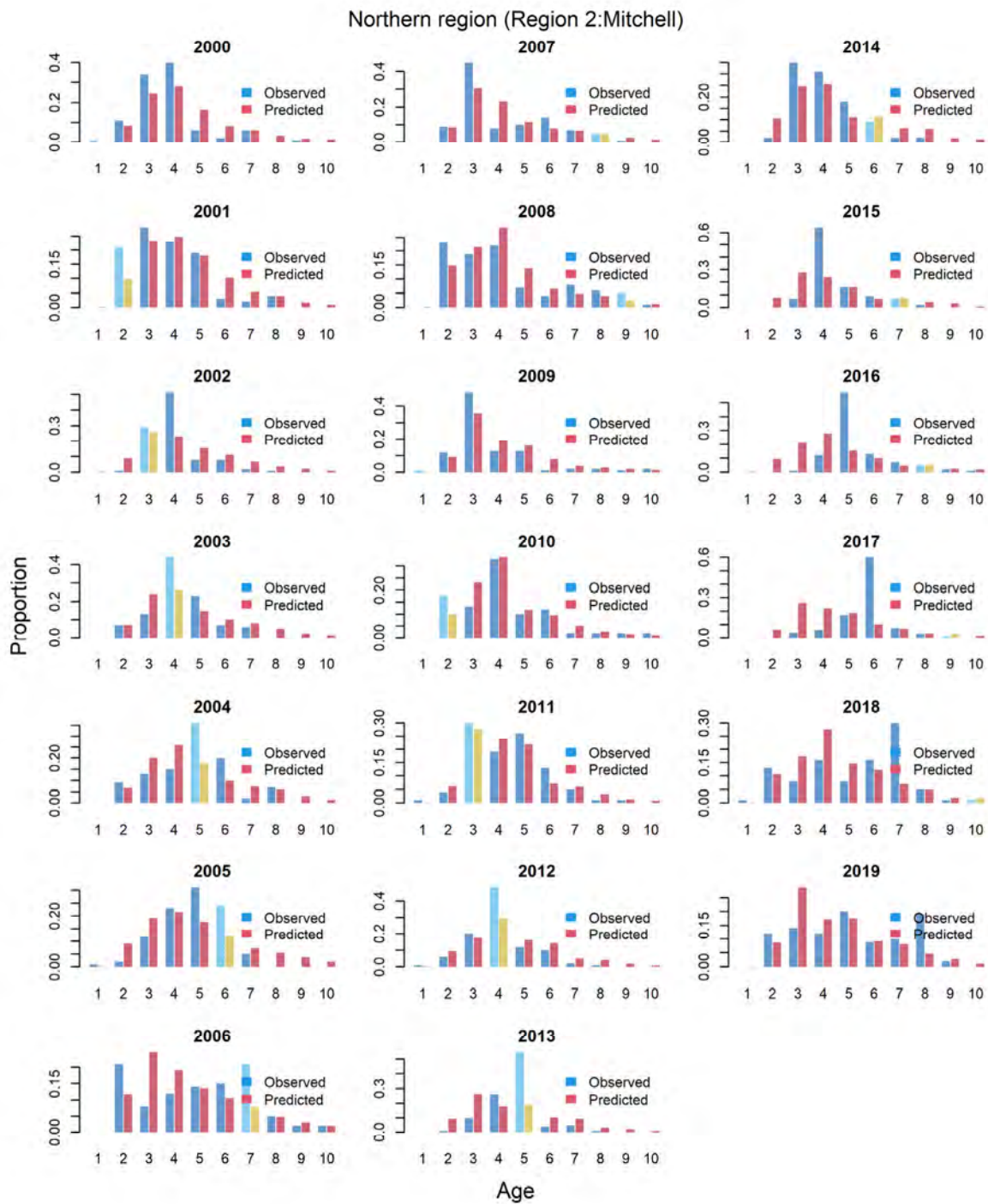


Figure A113. Comparison of the observed and model-predicted commercial catch-at-age proportions of barramundi in each age-class as shown estimated using Model version 5 (baseline flows, with no sex ratio influence on recruitment) for the Mitchell River catchment region from 2000 to 2019.

Appendix 18 – MICE WRD Additional Model Results

Impacts of WRDs 1-4, on seagrass, mangrove, prawns, barramundi, mud crabs and sawfish, using alternative model versions not presented in the main report, are shown in the figures below.

Seagrass

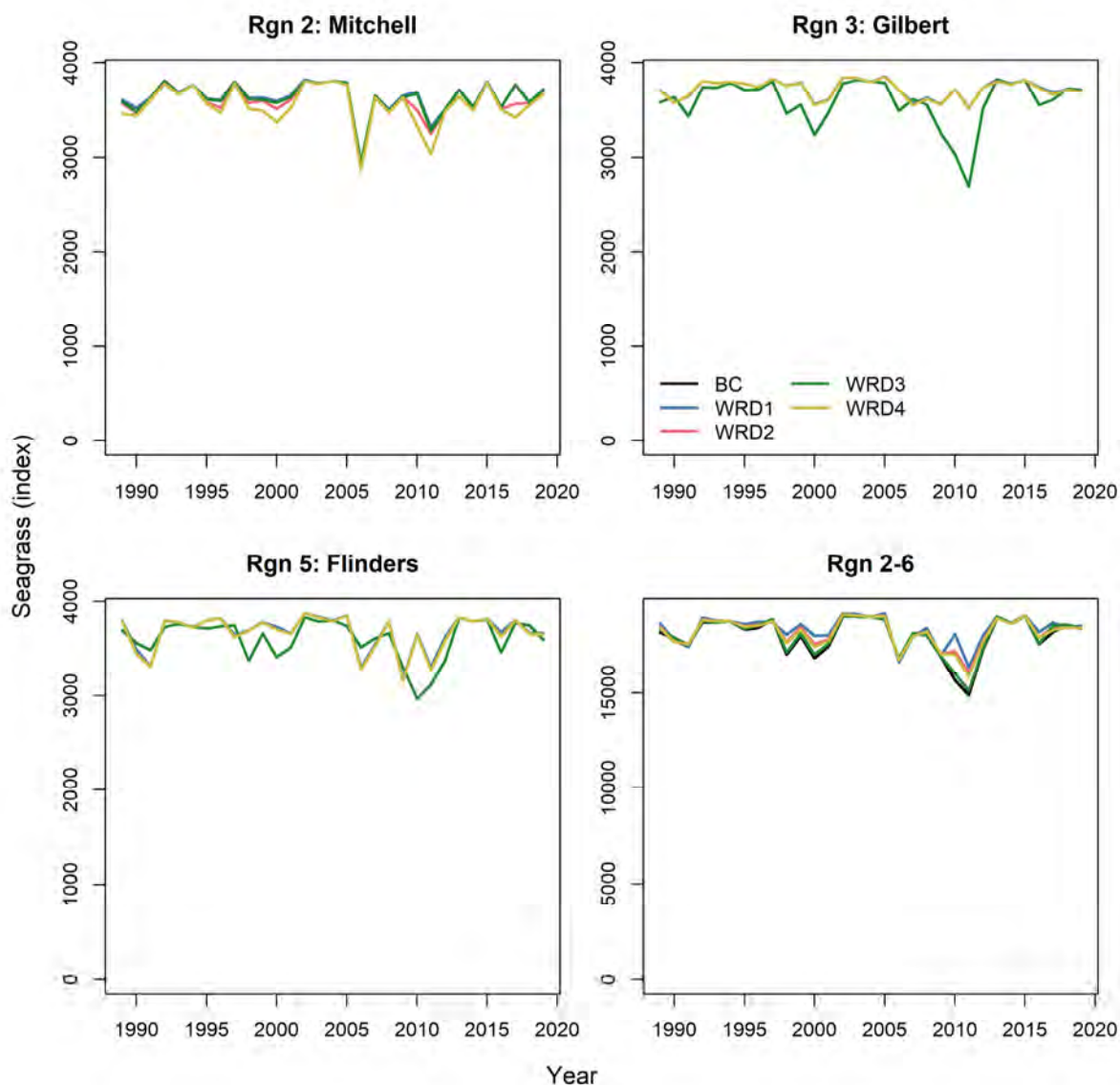


Figure A114. MICE-predicted changes in seagrass relative biomass under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 2. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, and WRD1 tracks under WRD4.

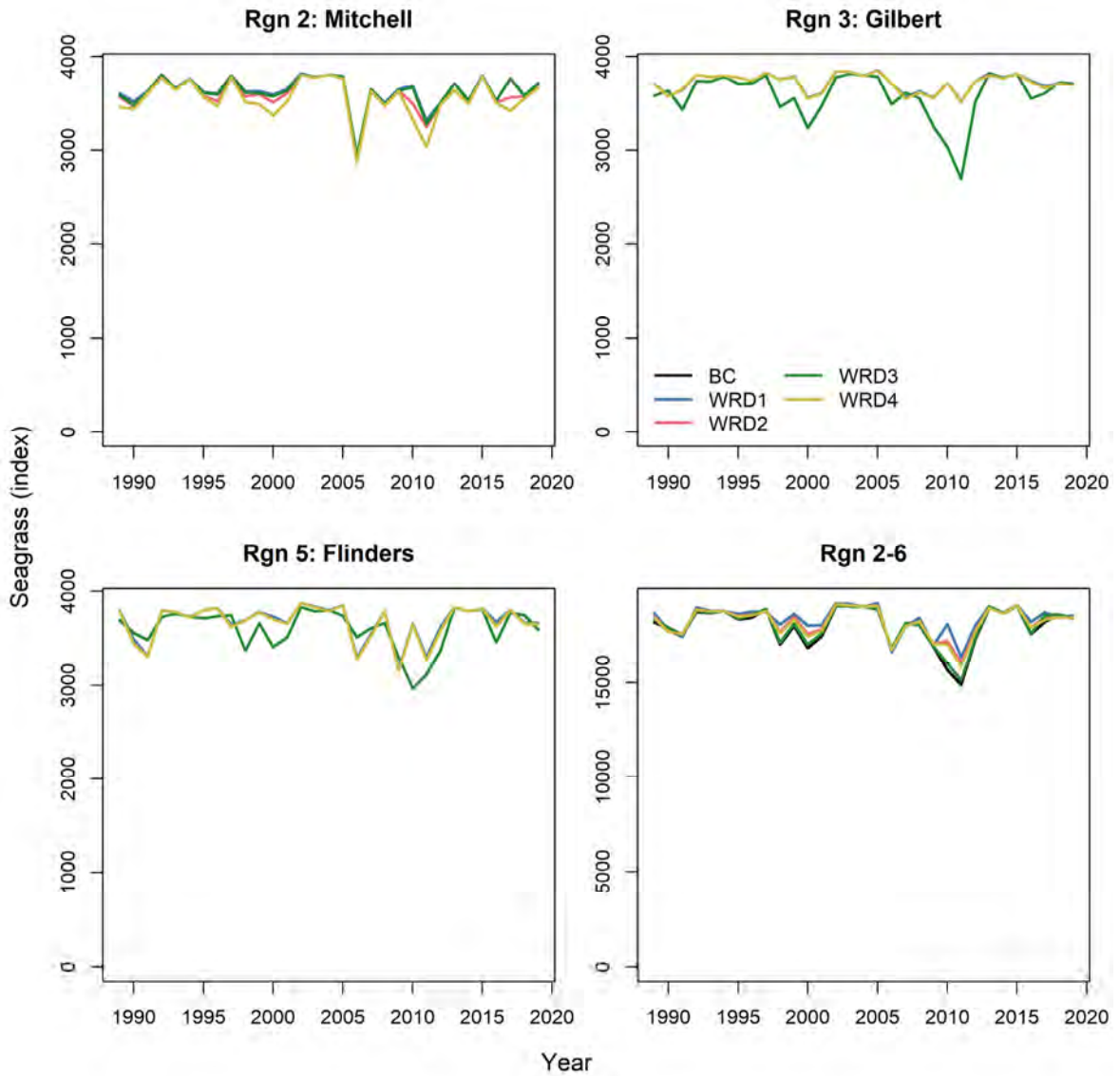


Figure A115. MICE-predicted changes in seagrass relative biomass under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 3. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, and WRD1 tracks under WRD4.

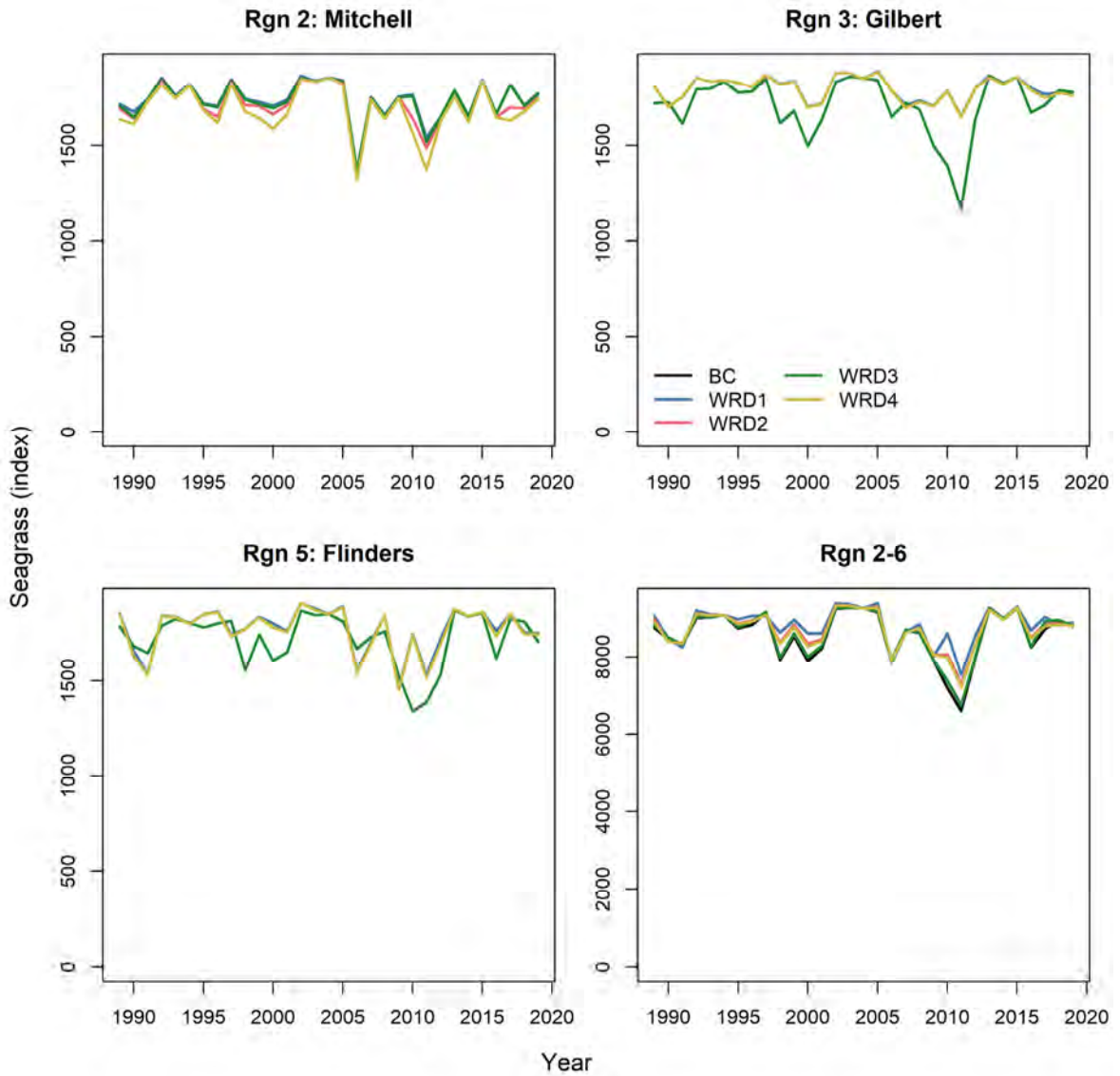


Figure A116. MICE-predicted changes in seagrass relative biomass under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 4. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, and WRD1 tracks under WRD4.

Mangroves

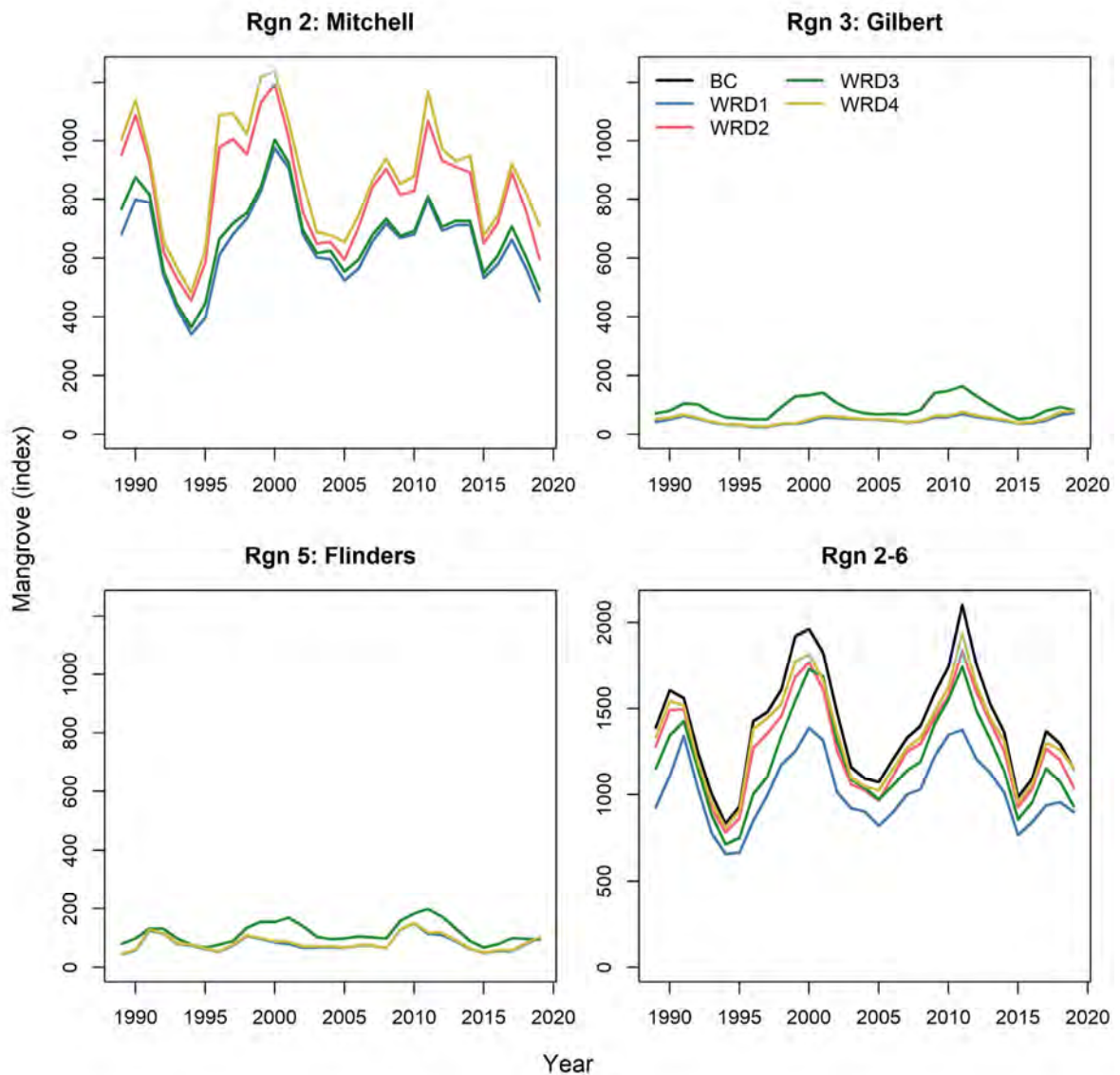


Figure A117. MICE-predicted changes in mangrove relative biomass under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 2. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4.

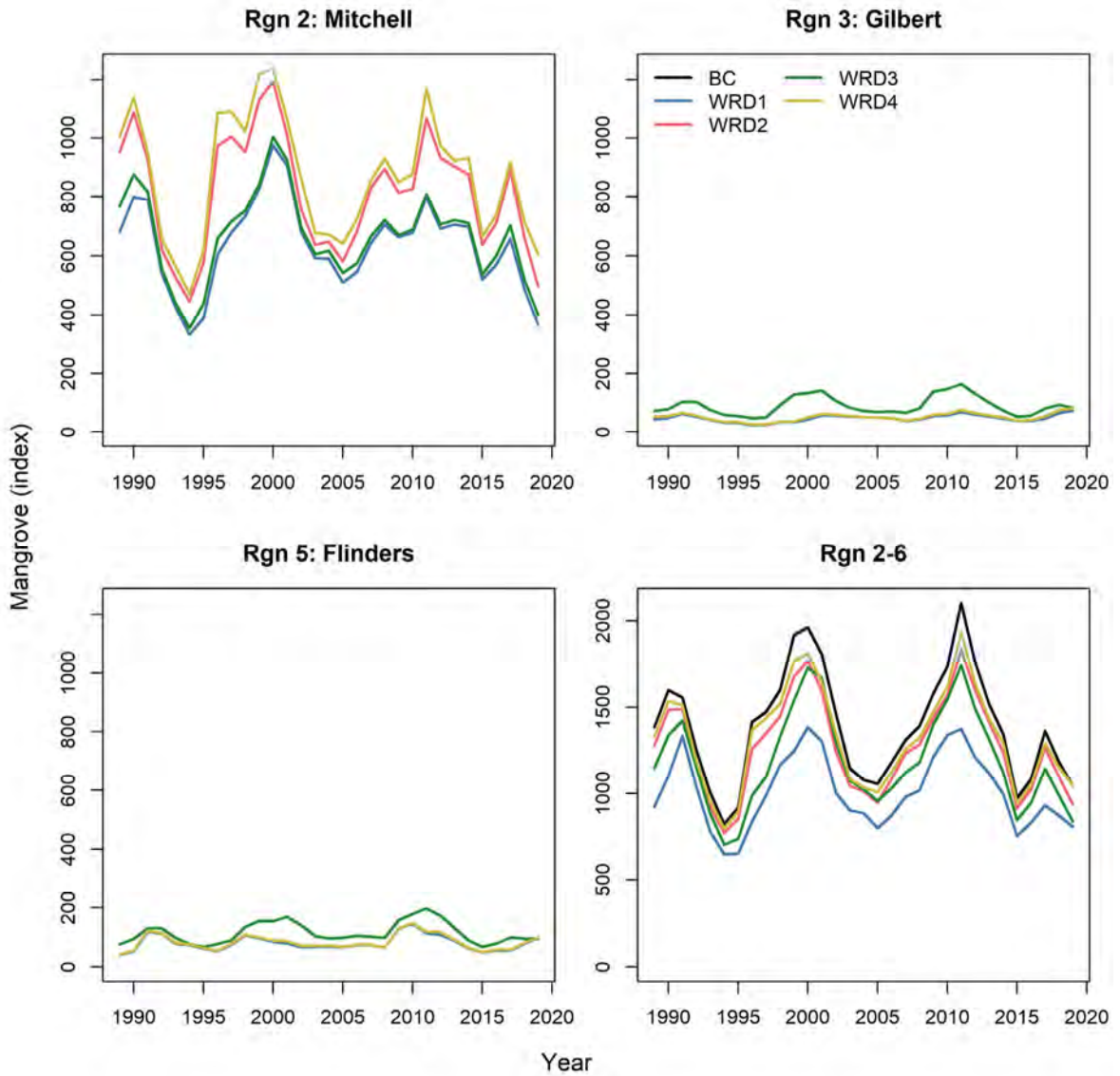


Figure A118. MICE-predicted changes in mangrove relative biomass under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 3. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4.

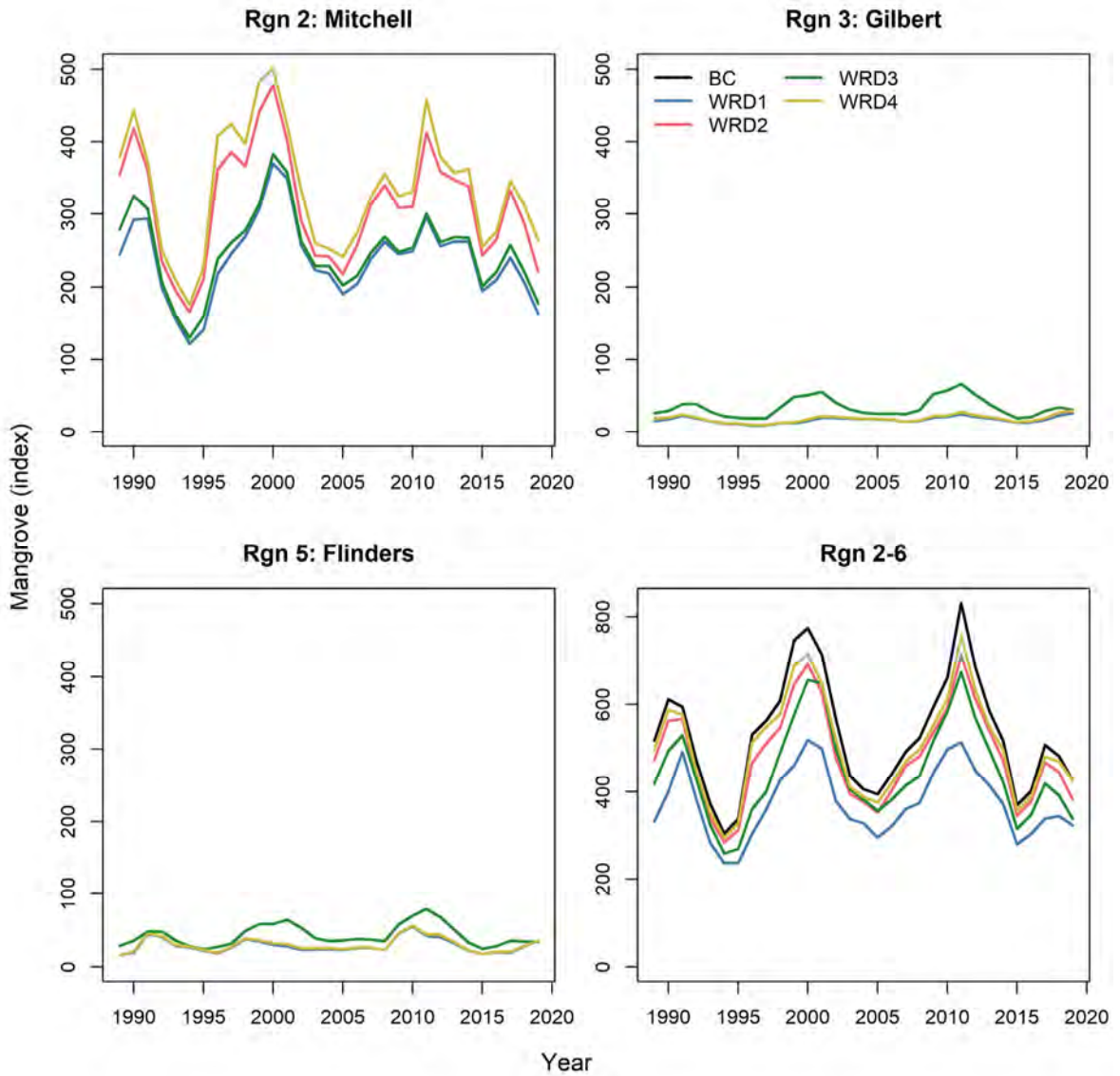


Figure A119. MICE-predicted changes in mangrove relative biomass under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 4. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4.

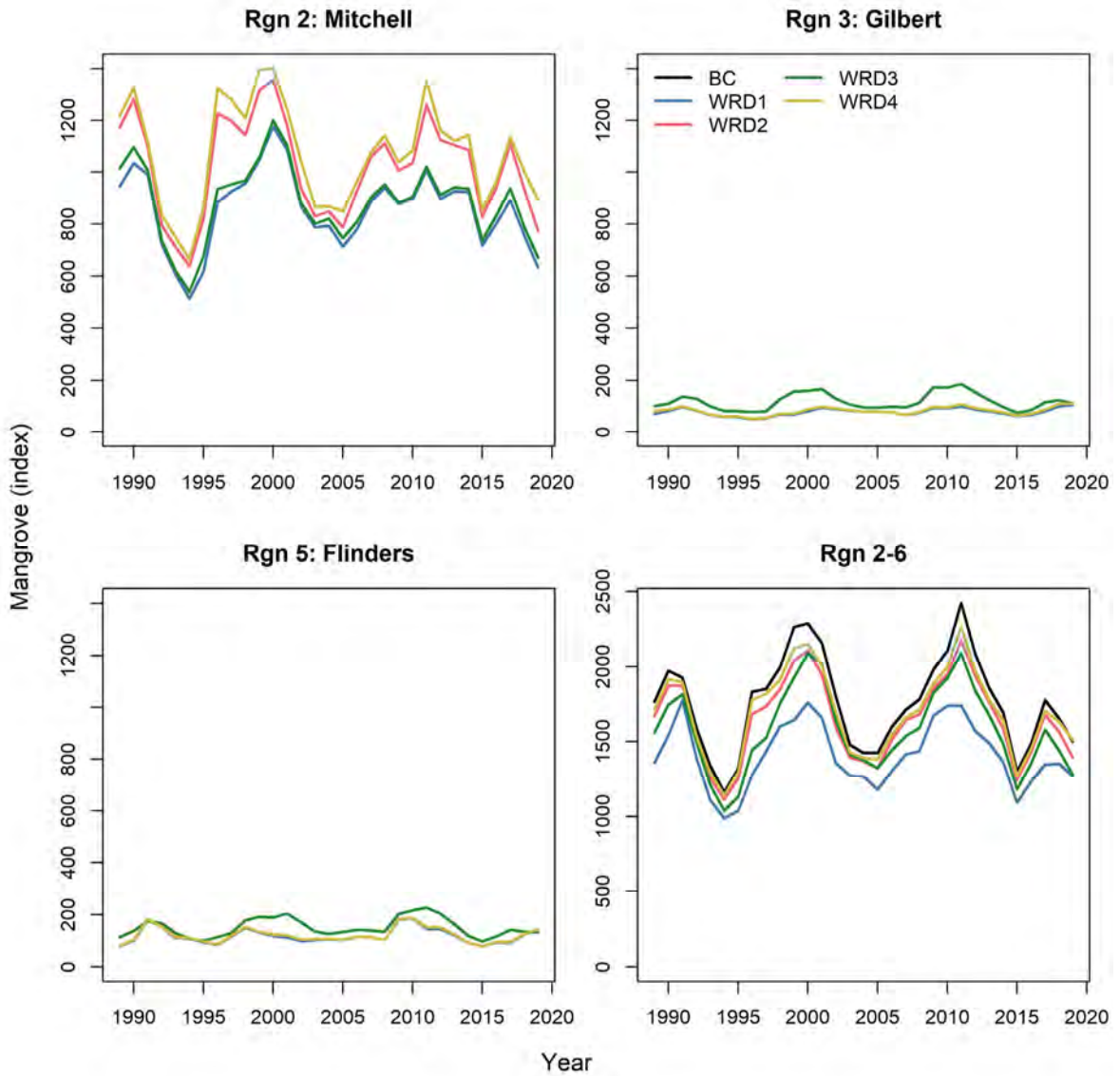


Figure A120. MICE-predicted changes in mangrove relative biomass under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 5. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4.

Prawns

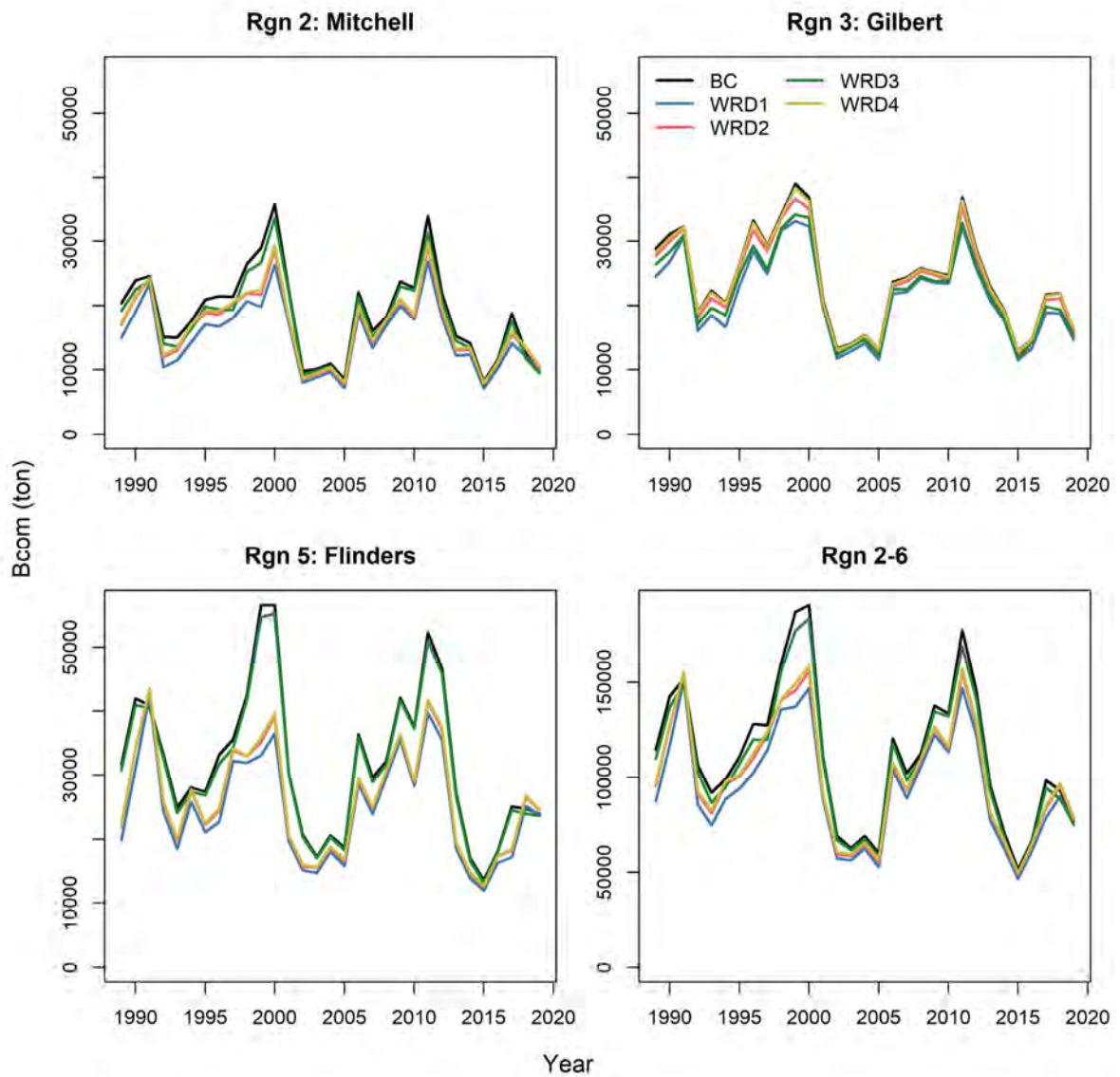


Figure A121. MICE-predicted changes to total annual common banana prawn commercially available biomass (Bcom, t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 3.

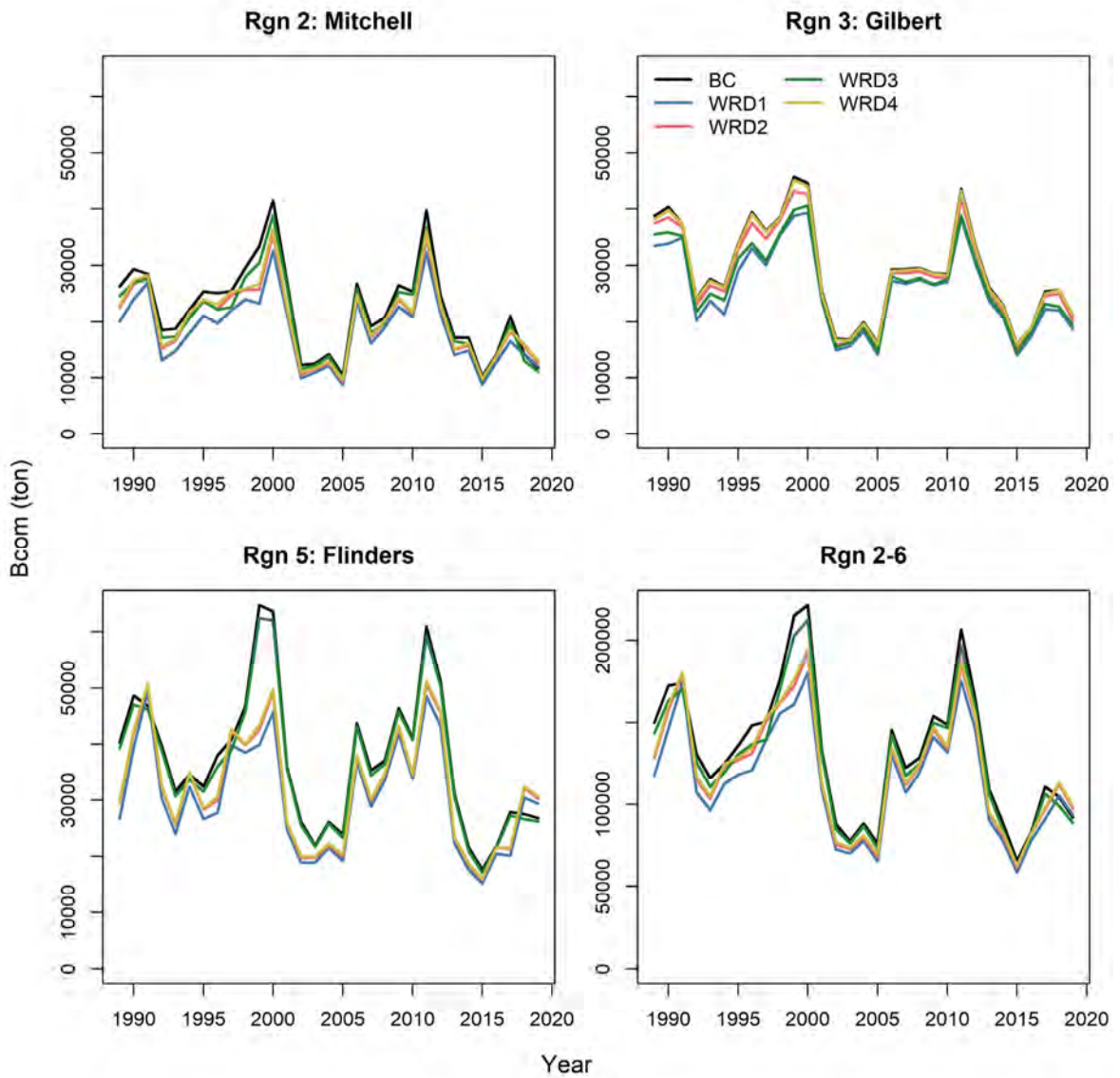


Figure A122. MICE-predicted changes to total annual common banana prawn commercially available biomass (Bcom, t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 4.

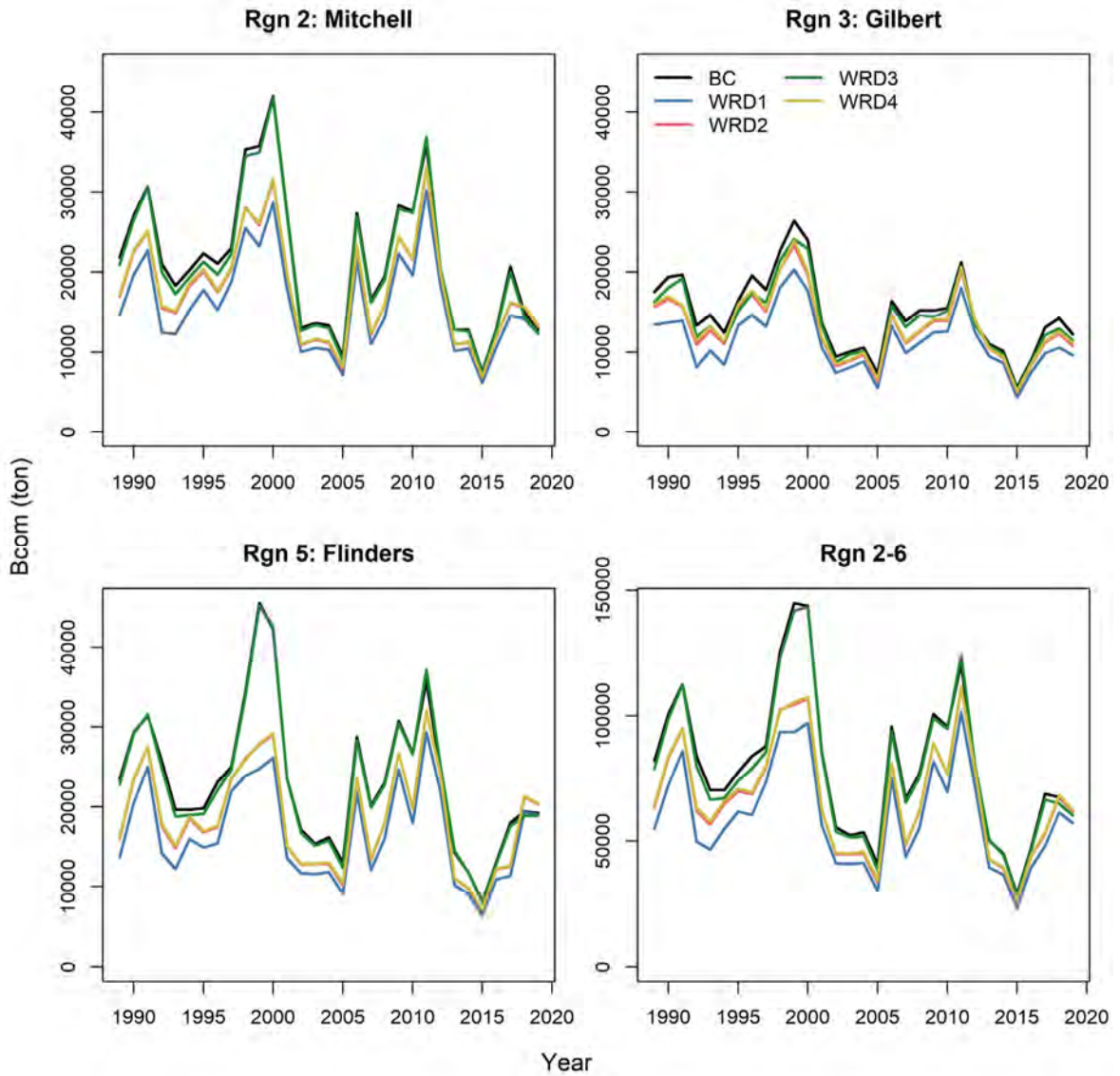


Figure A123. MICE-predicted changes to total annual common banana prawn commercially available biomass (Bcom, t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 5.

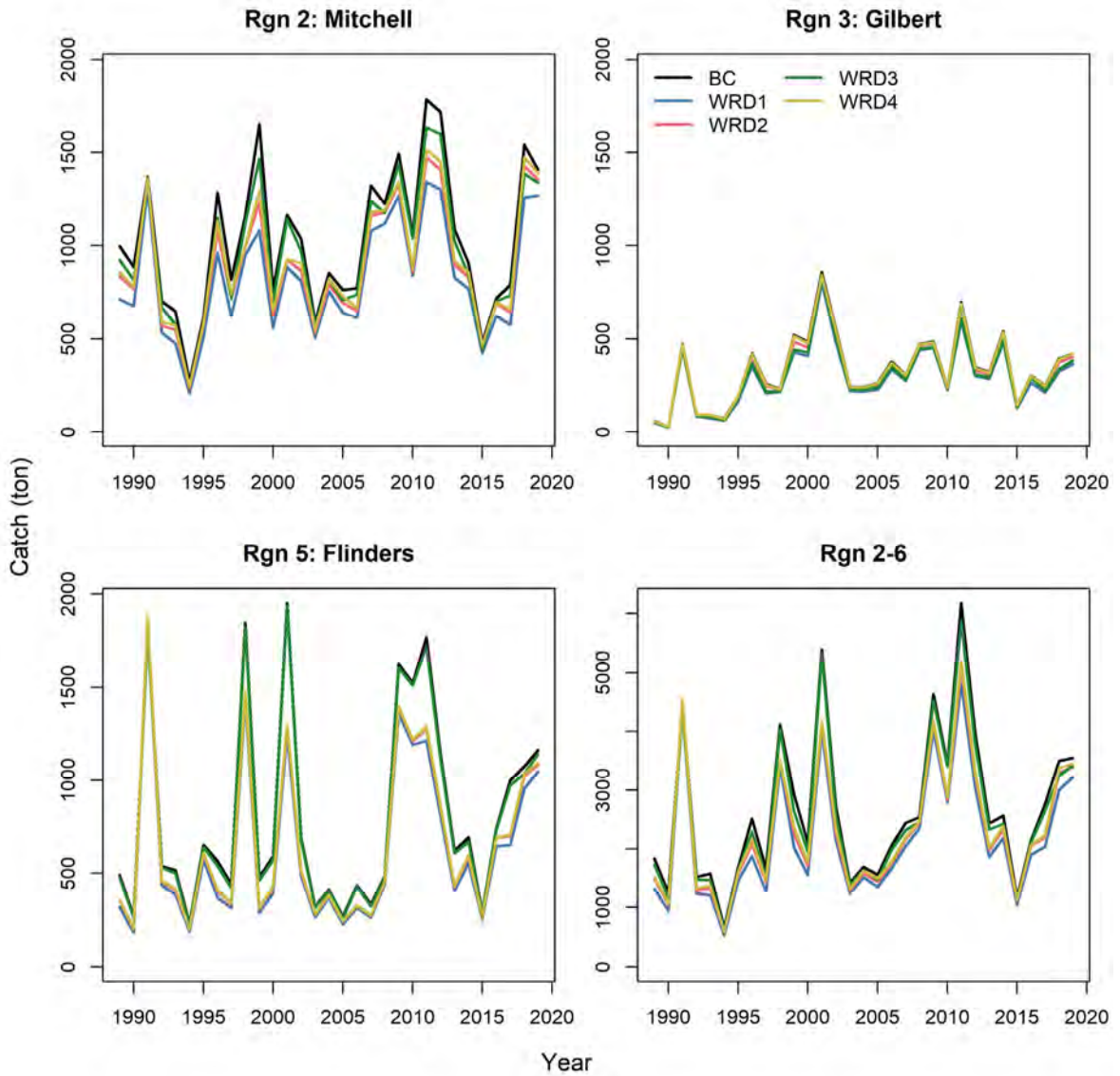


Figure A124. MICE-predicted changes to total annual common banana prawn catch (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 3.

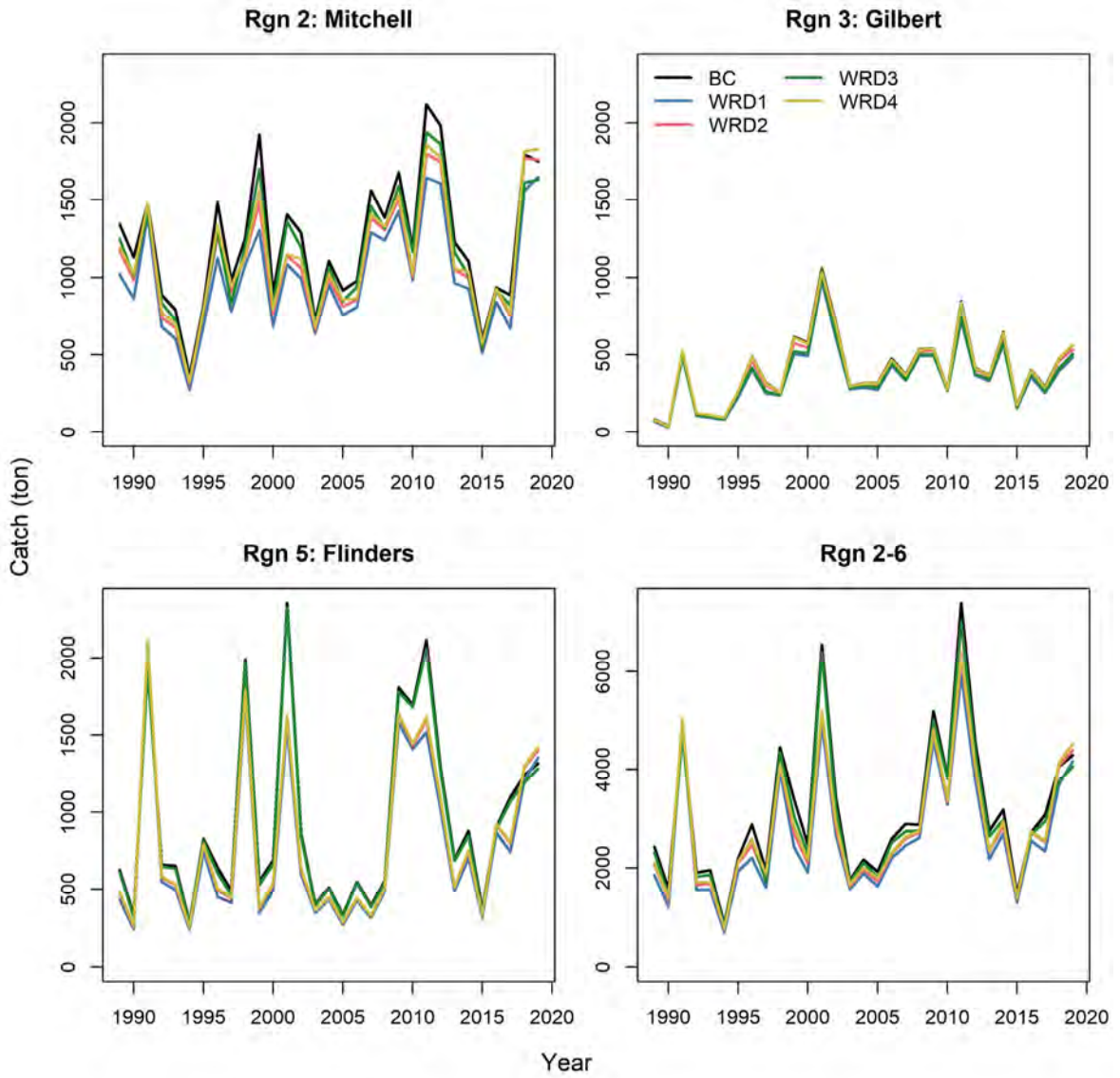


Figure A125. MICE-predicted changes to total annual common banana prawn catch (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 4.

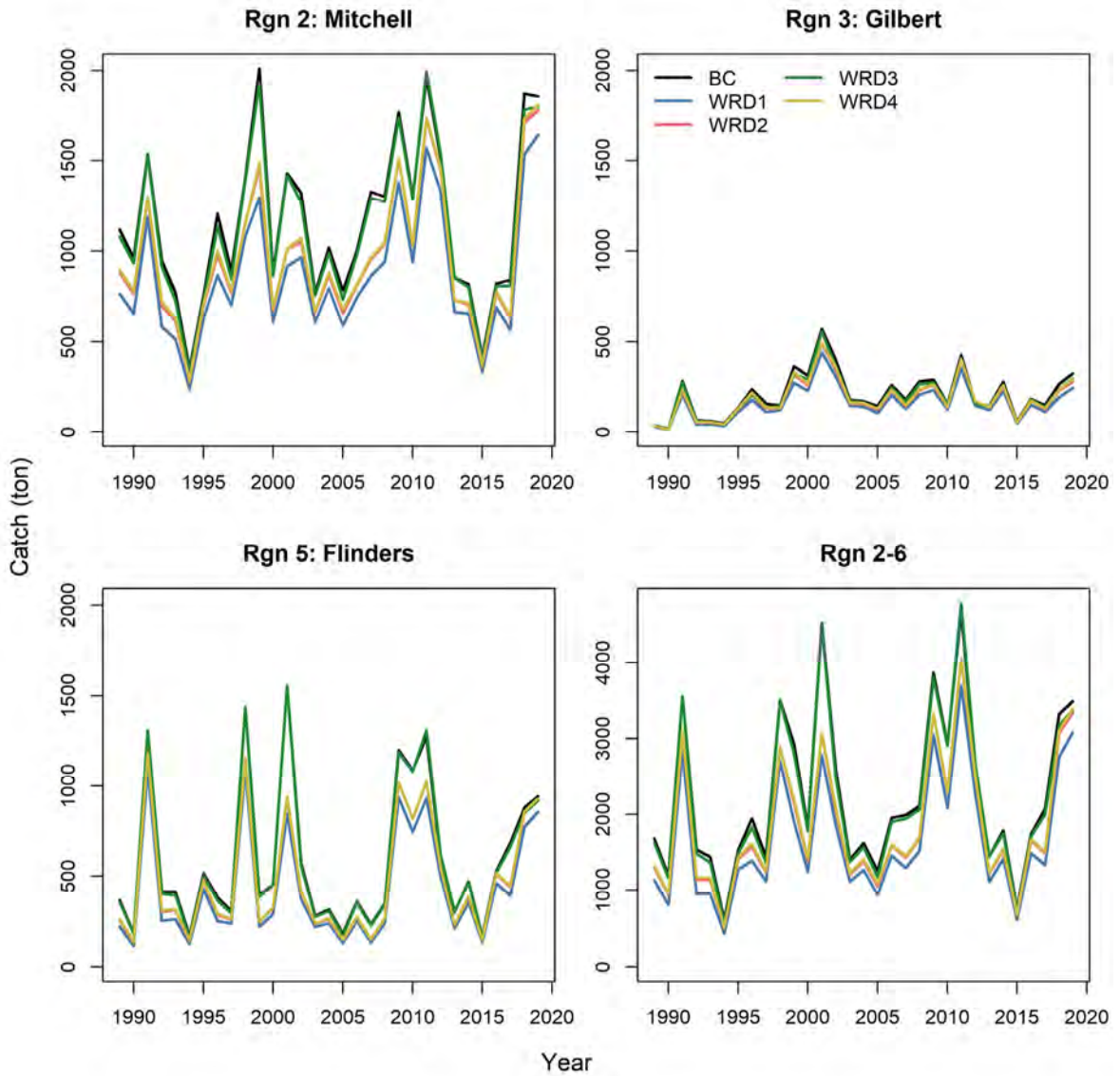


Figure A126. MICE-predicted changes to total annual common banana prawn catch (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 5.

Barramundi

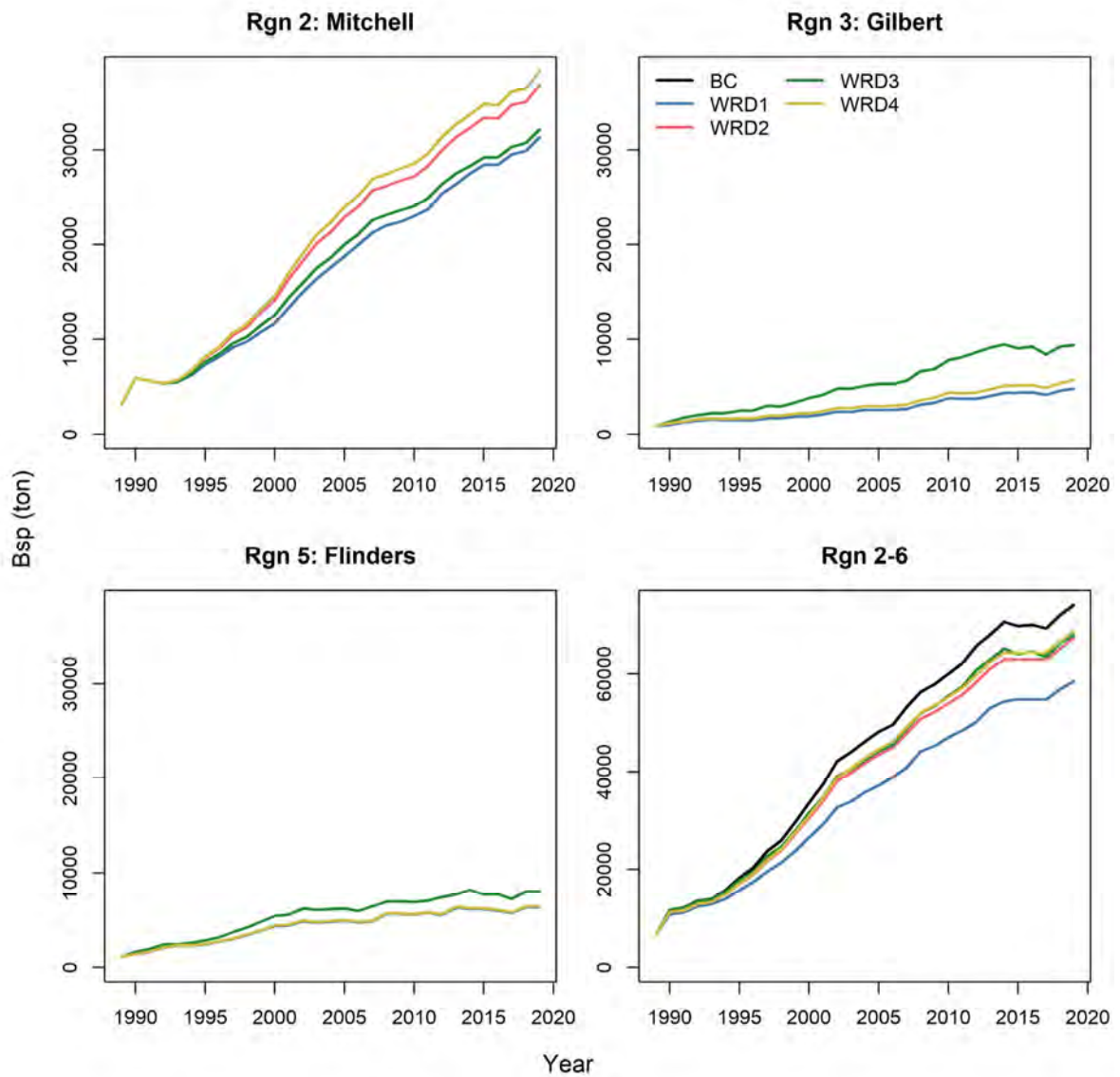


Figure A127. MICE-predicted changes to total annual barramundi spawning biomass (Bsp) (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 1. Note for the Mitchell River, BC trajectory tracks under WRD4 and for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4.

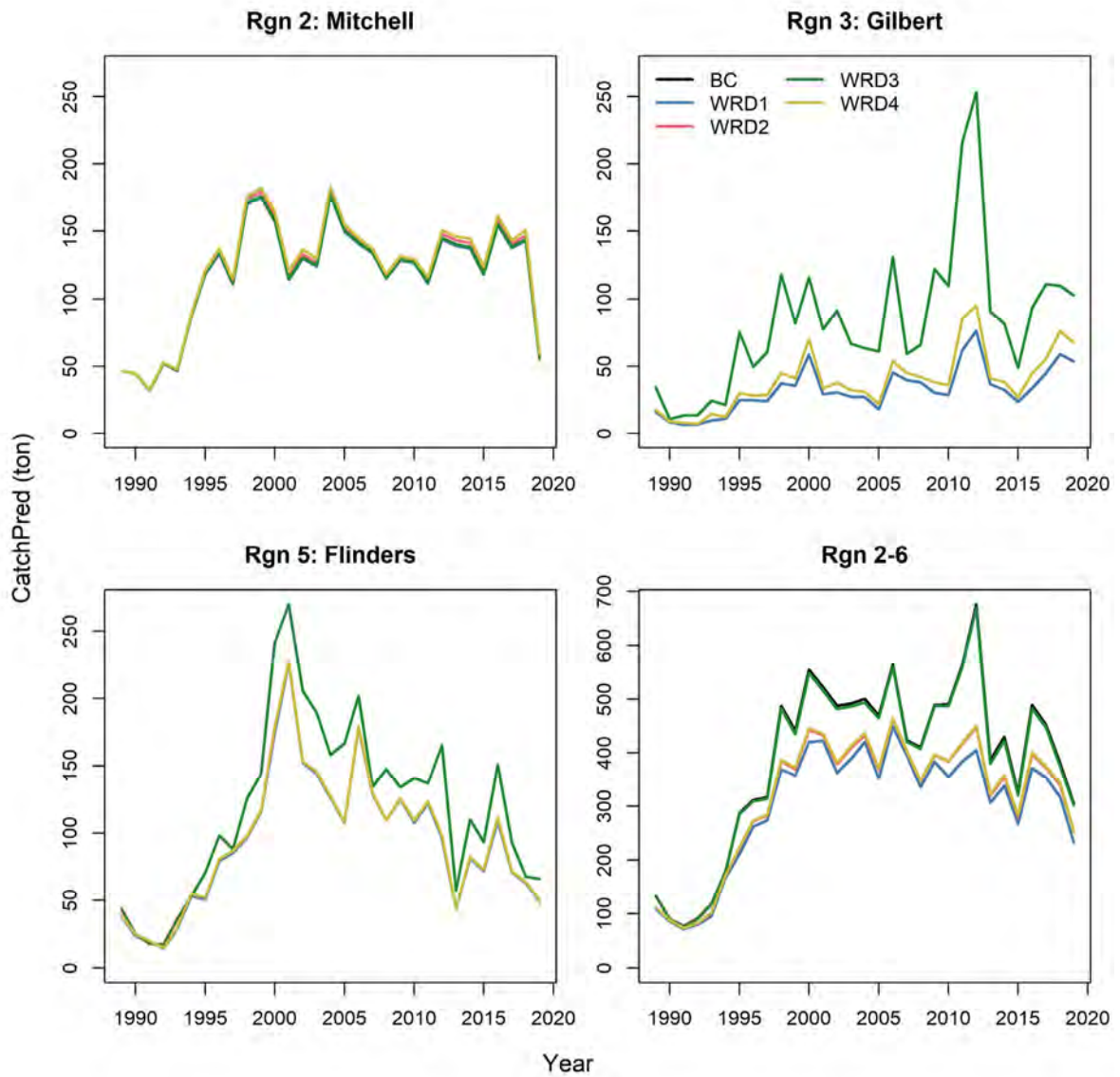


Figure A128. MICE-predicted changes to total annual barramundi catch (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 3. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2 tracks under WRD4.

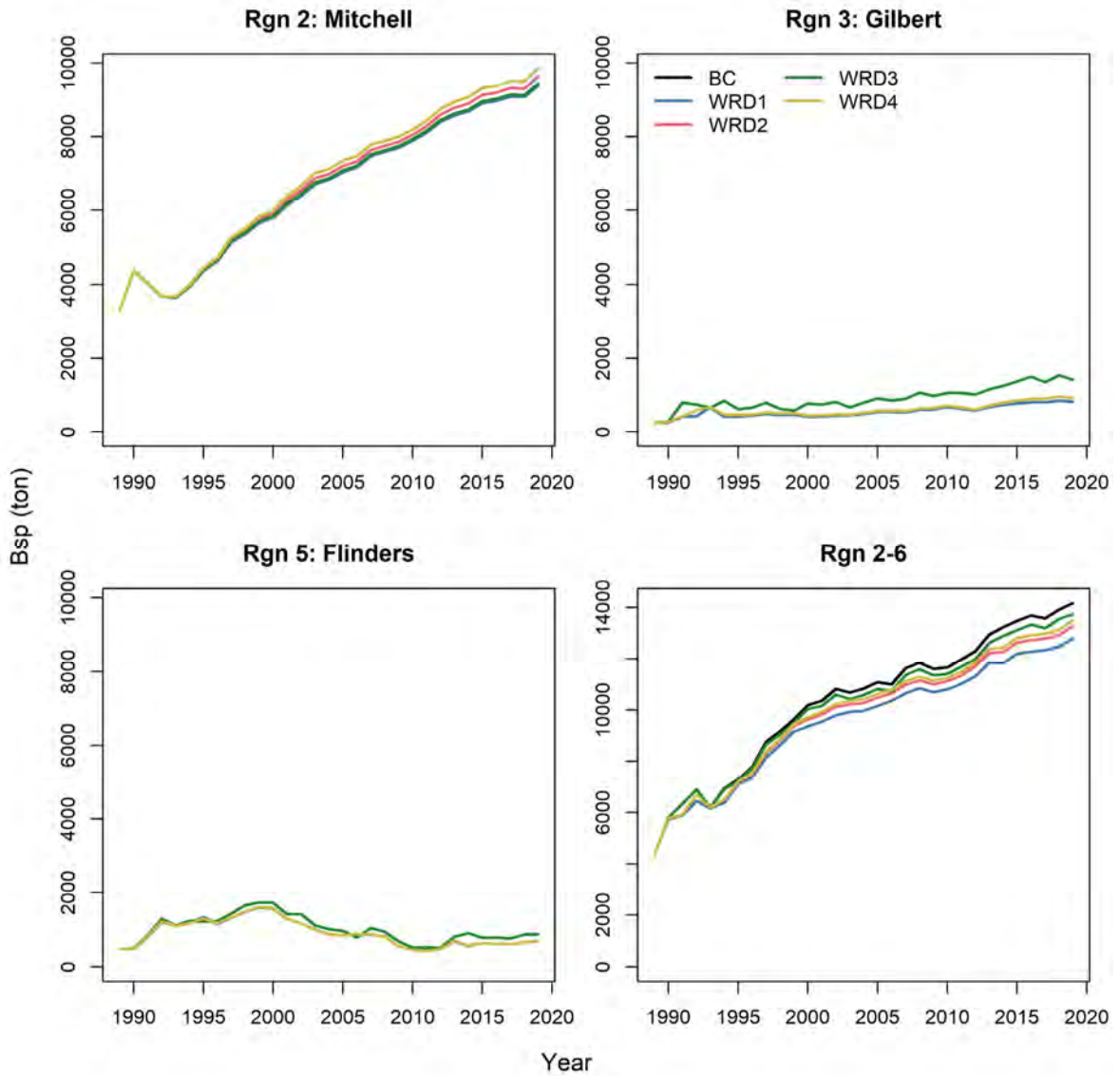


Figure A129. MICE-predicted changes to total annual barramundi spawning biomass (Bsp) (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 3. Note for the Mitchell River, BC trajectory tracks under WRD4 and for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4.

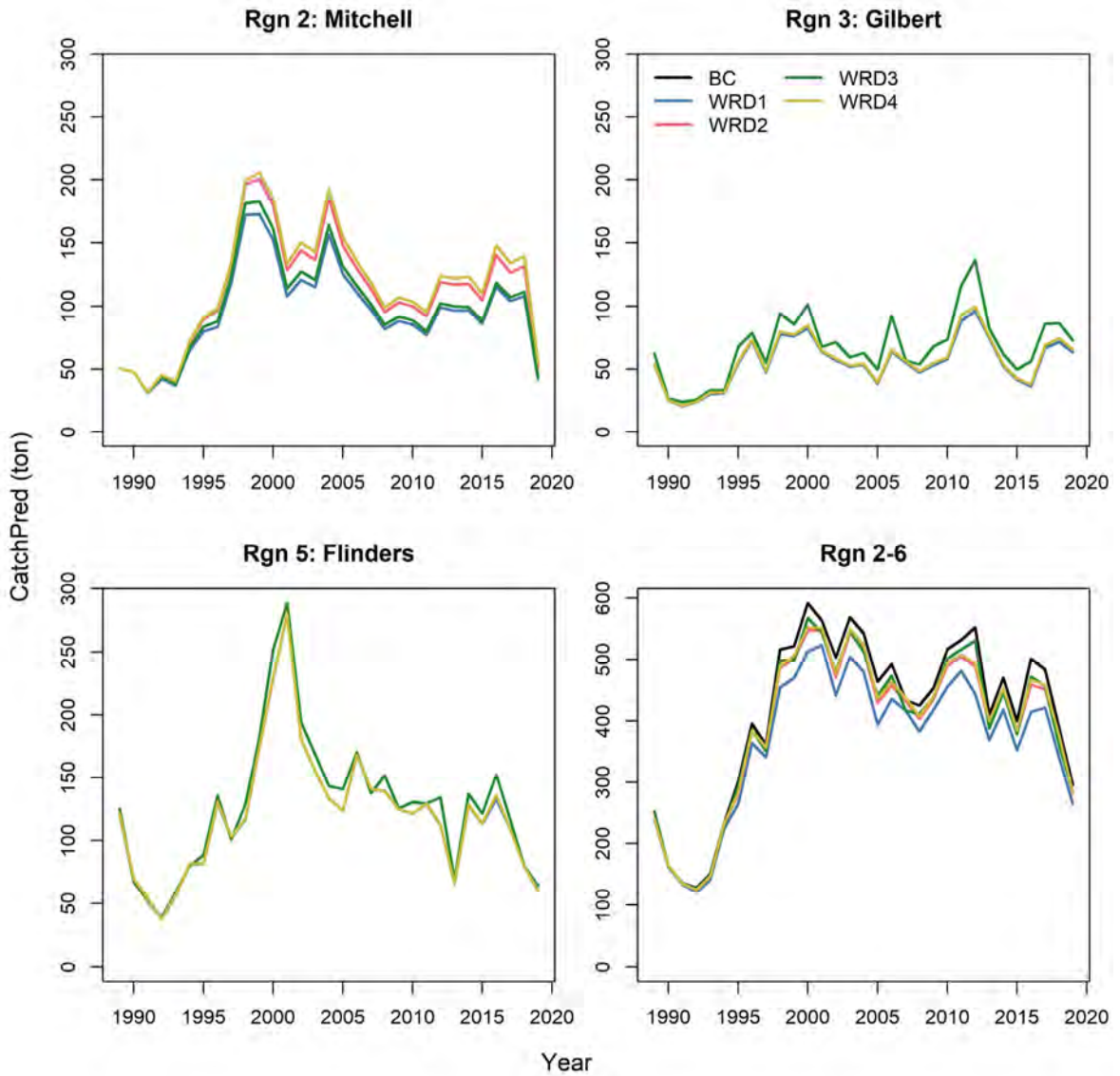


Figure A130. MICE-predicted changes to total annual barramundi catch (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 4. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2 tracks under WRD4.

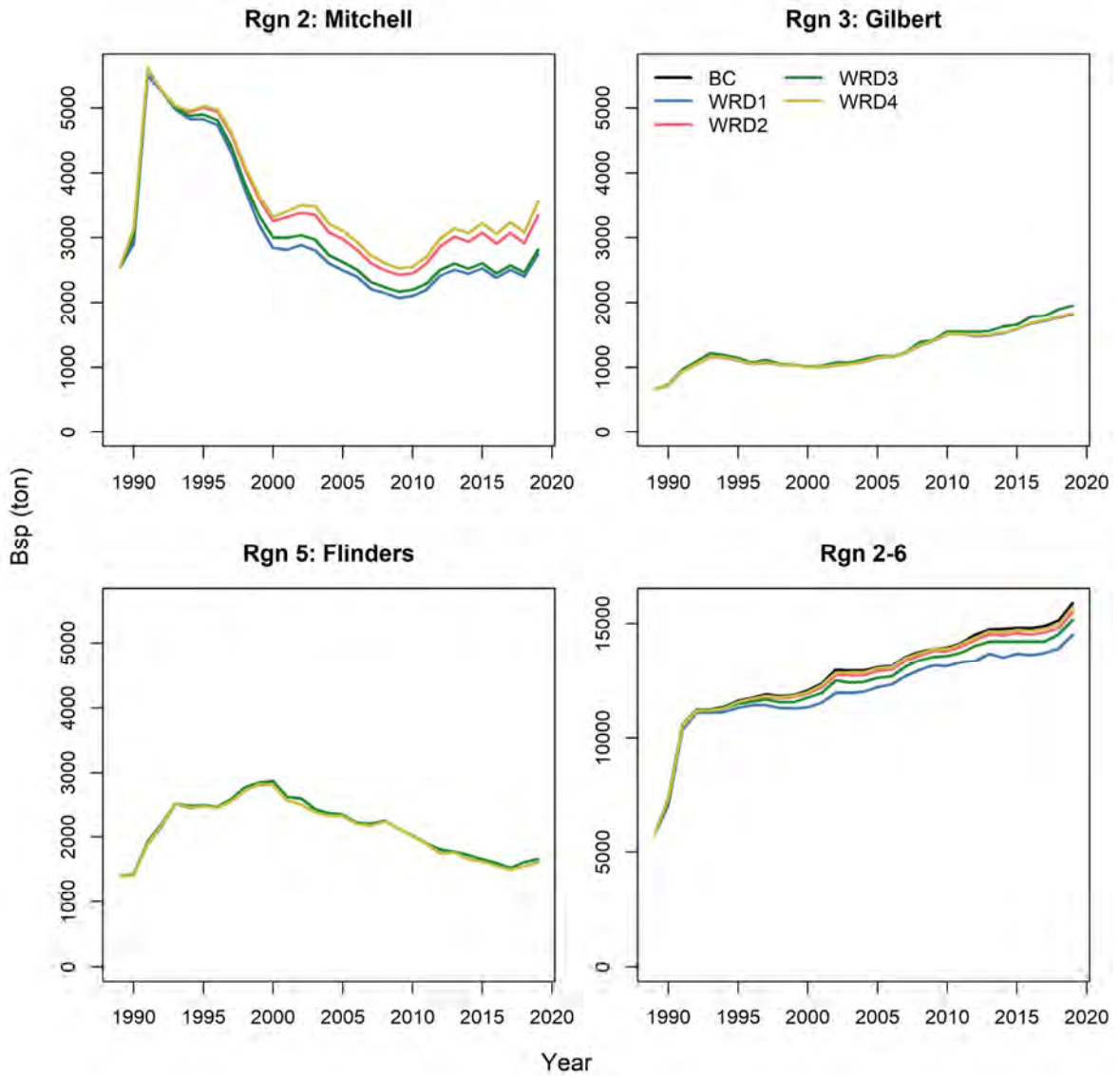


Figure A131. MICE-predicted changes to total annual barramundi spawning biomass (Bsp) (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 4. Note for the Mitchell River, BC trajectory tracks under WRD4 and for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4.

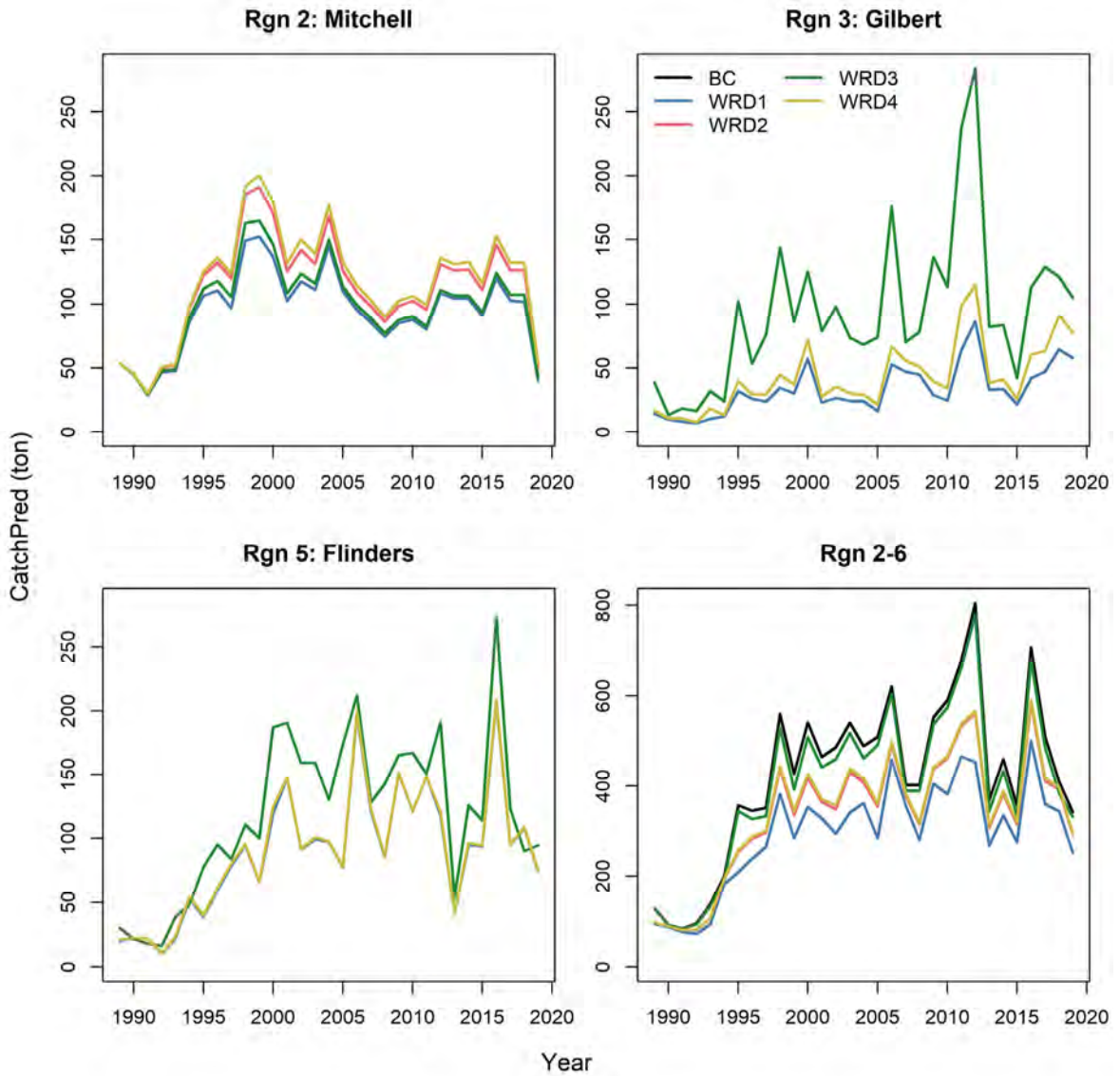


Figure A132. MICE-predicted changes to total annual barramundi catch (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 5. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2 tracks under WRD4.

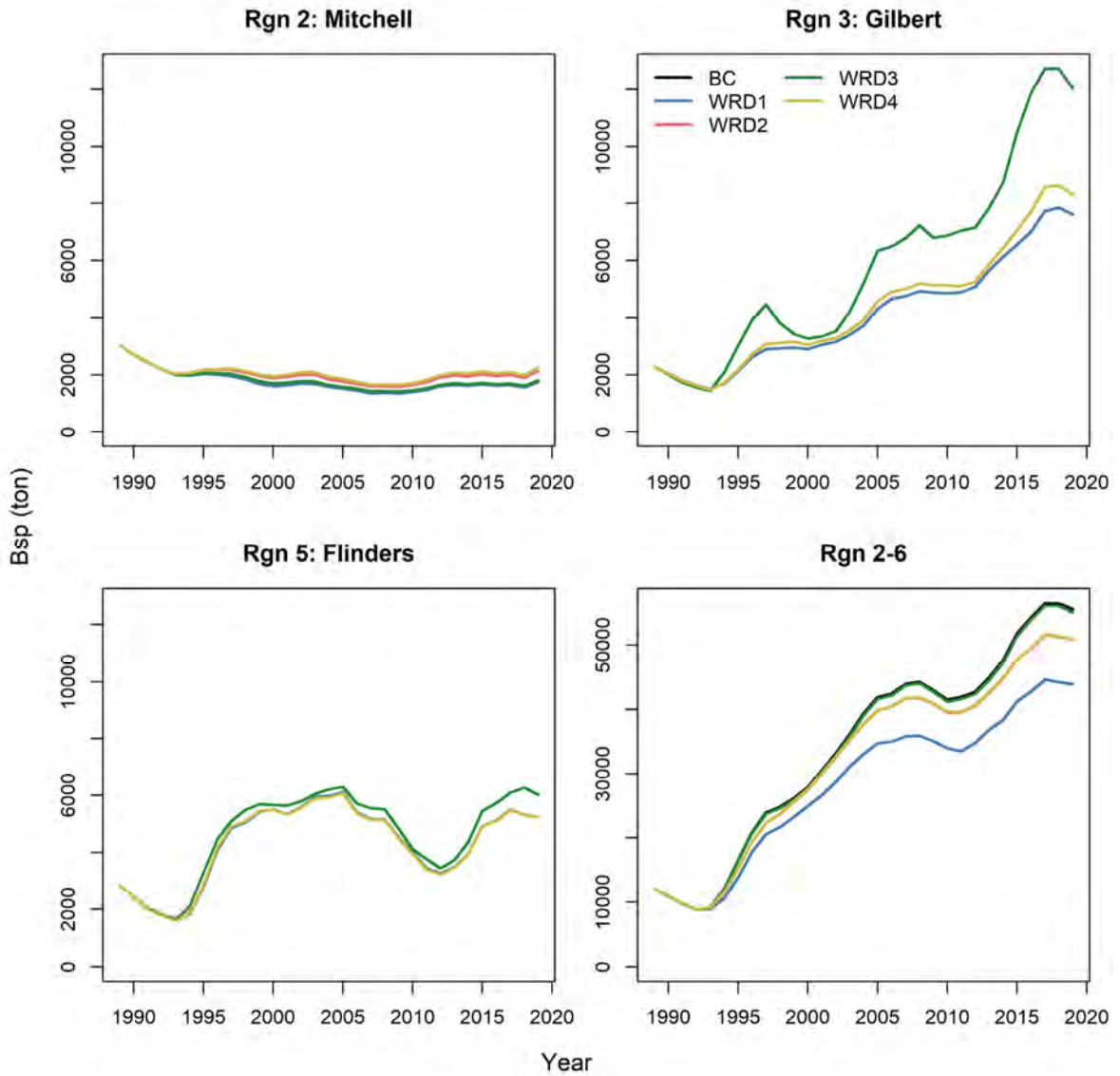


Figure A133. MICE-predicted changes to total annual barramundi spawning biomass (Bsp) (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 5. Note for the Mitchell River, BC trajectory tracks under WRD4 and for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4.

Mud crabs

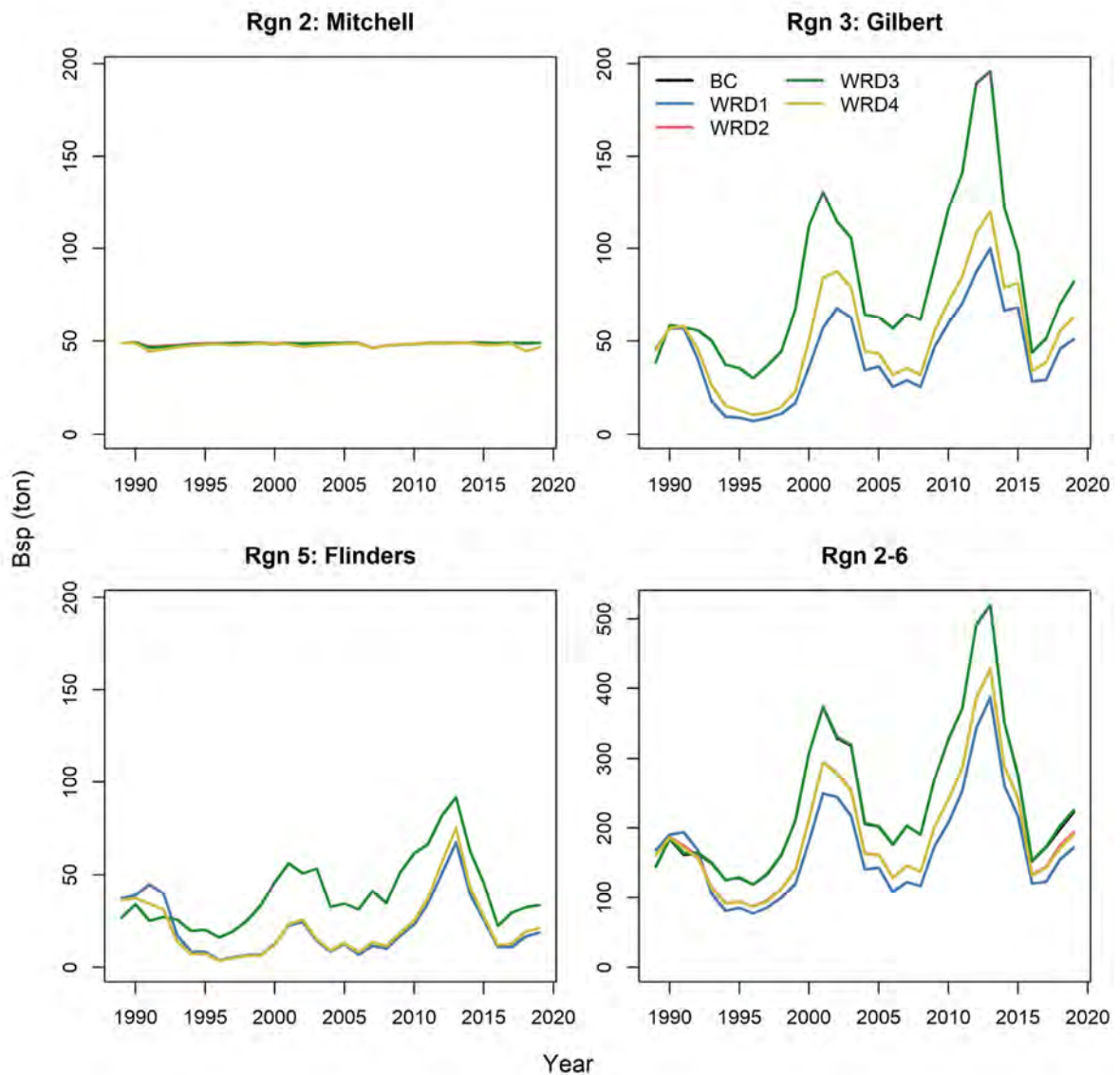


Figure A134. MICE-predicted changes to mud crab spawning biomass (Bsp; t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 1. Note for the Mitchell River, the BC and all WRDs track under each other. For the other panels, the BC trajectory tracks under WRD3 and WRD2 tracks under WRD4.

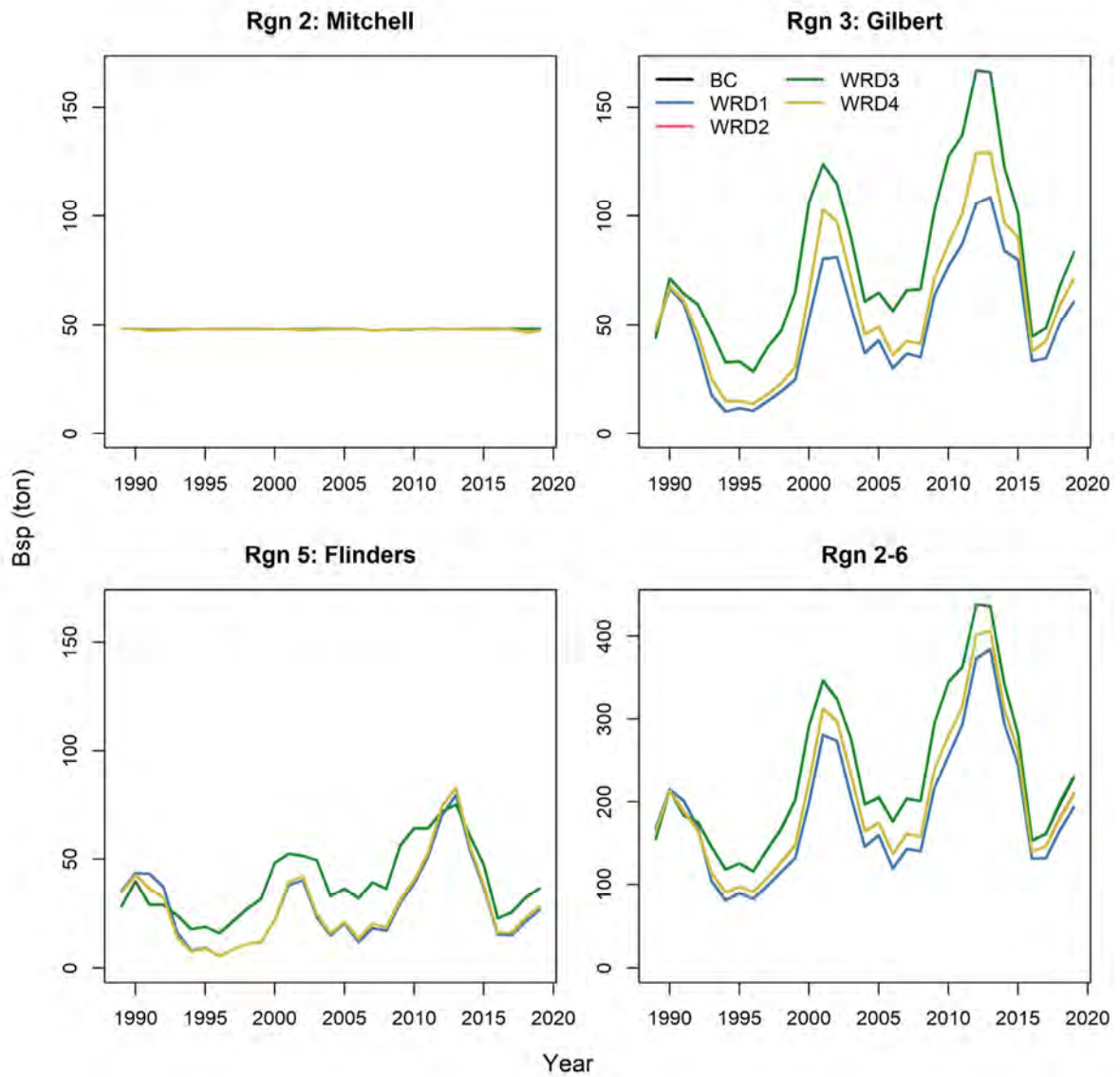


Figure A135. MICE-predicted changes to mud crab spawning biomass (Bsp; t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 2. Note for the Mitchell River, the BC and all WRDs track under each other. For the other panels, the BC trajectory tracks under WRD3 and WRD2 tracks under WRD4.

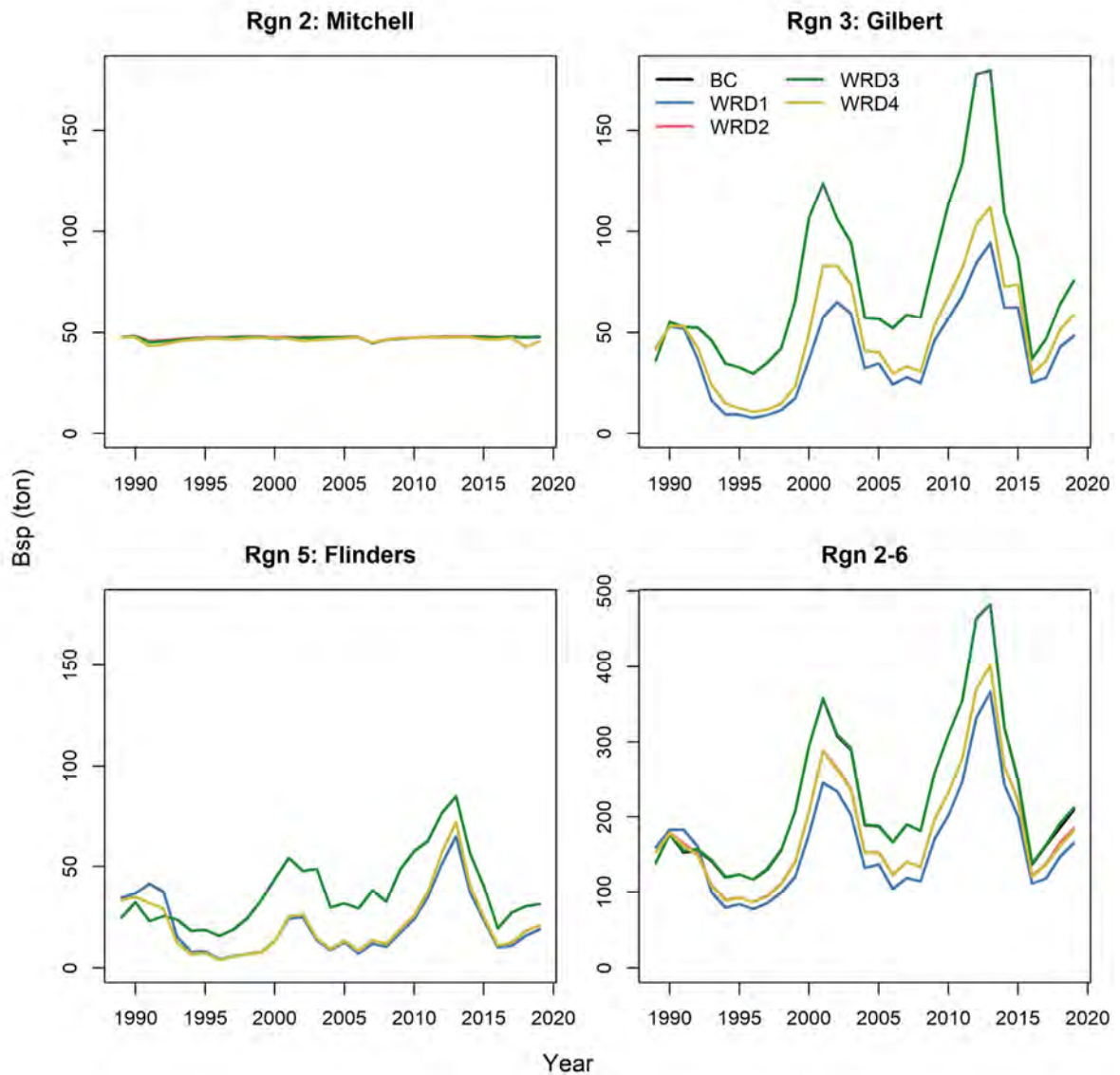


Figure A136. MICE-predicted changes to total annual mud crab catch (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 3. Note for the Mitchell River, the BC and all WRDs track under each other. For the other panels, the BC trajectory tracks under WRD3 and WRD2 tracks under WRD4.

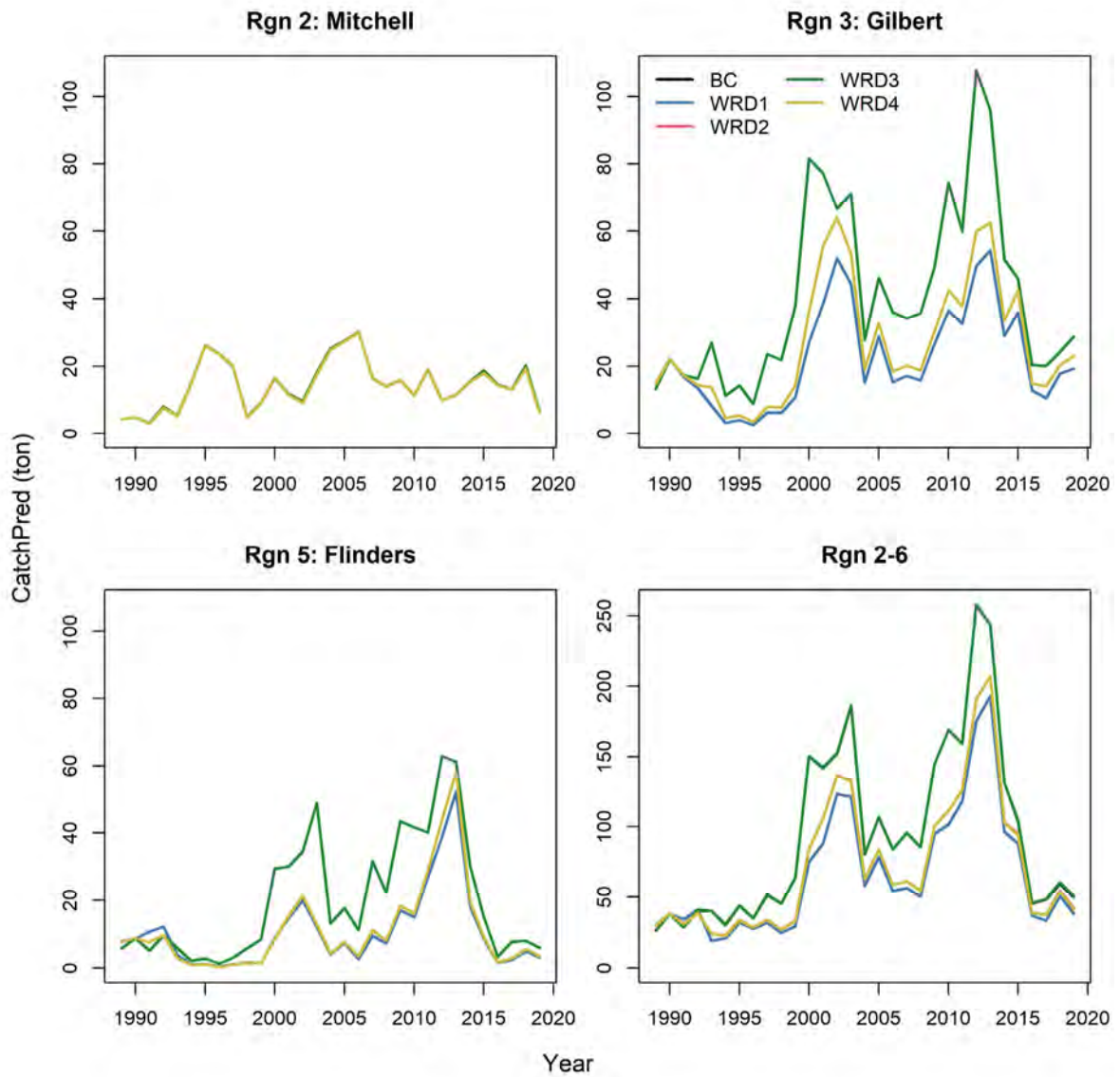


Figure A137. MICE-predicted changes to total annual mud crab catch (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 3. Note for the Mitchell River, the BC and all WRDs track under each other. For the other panels, the BC trajectory tracks under WRD3 and WRD2 tracks under WRD4.

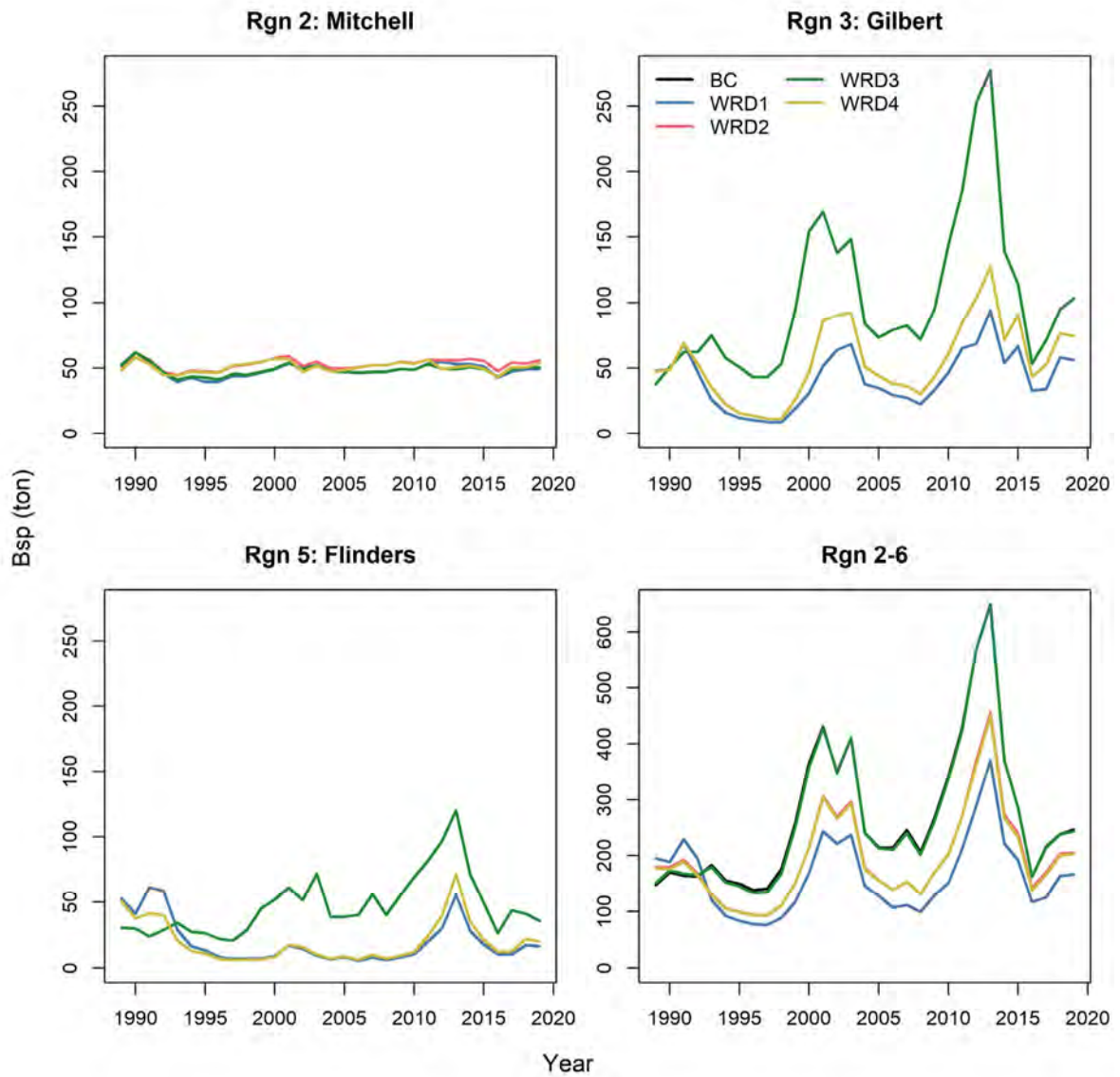


Figure A138. MICE-predicted changes to mud crab spawning biomass (Bsp; t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 4. Note for the Mitchell River, the BC and all WRDs track under each other. For the other panels, the BC trajectory tracks under WRD3 and WRD2 tracks under WRD4.

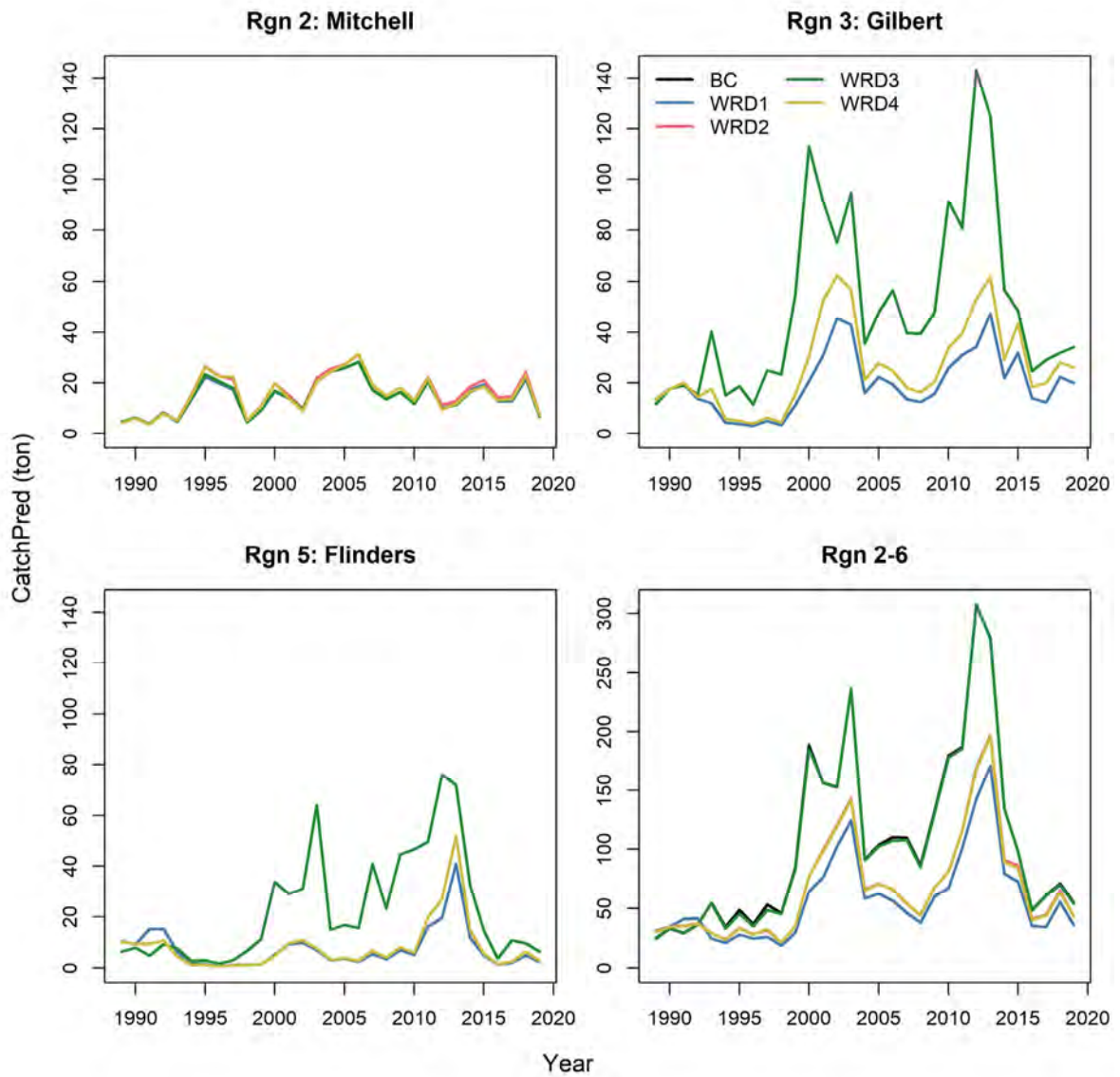


Figure A139. MICE-predicted changes to total annual mud crab catch (t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 4. Note for the Mitchell River, the BC and all WRDs track under each other. For the other panels, the BC trajectory tracks under WRD3 and WRD2 tracks under WRD4.

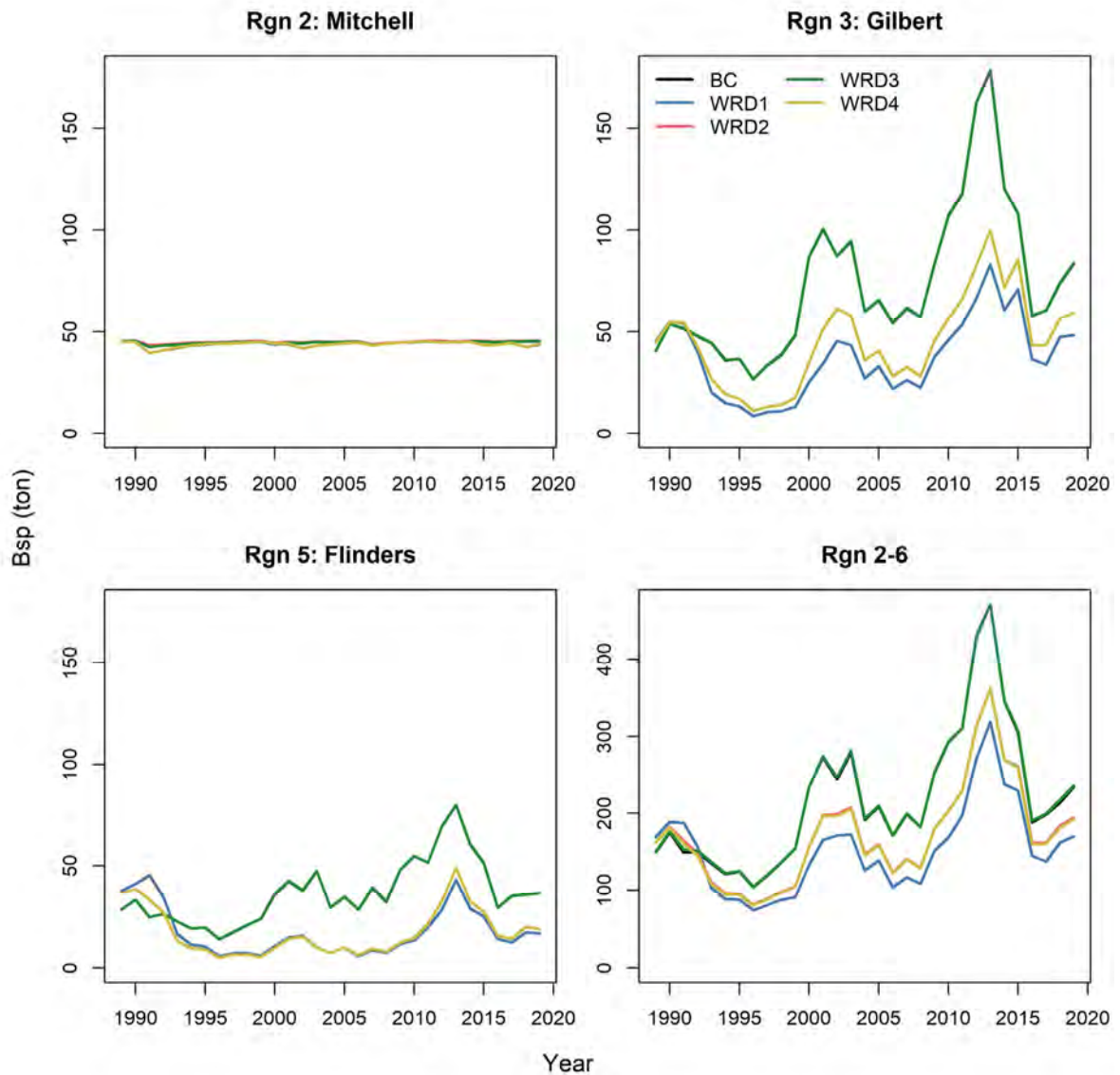


Figure A140. MICE-predicted changes to mud crab spawning biomass (Bsp; t) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 5. Note for the Mitchell River, the BC and all WRDs track under each other. For the other panels, the BC trajectory tracks under WRD3 and WRD2 tracks under WRD4.

Sawfish

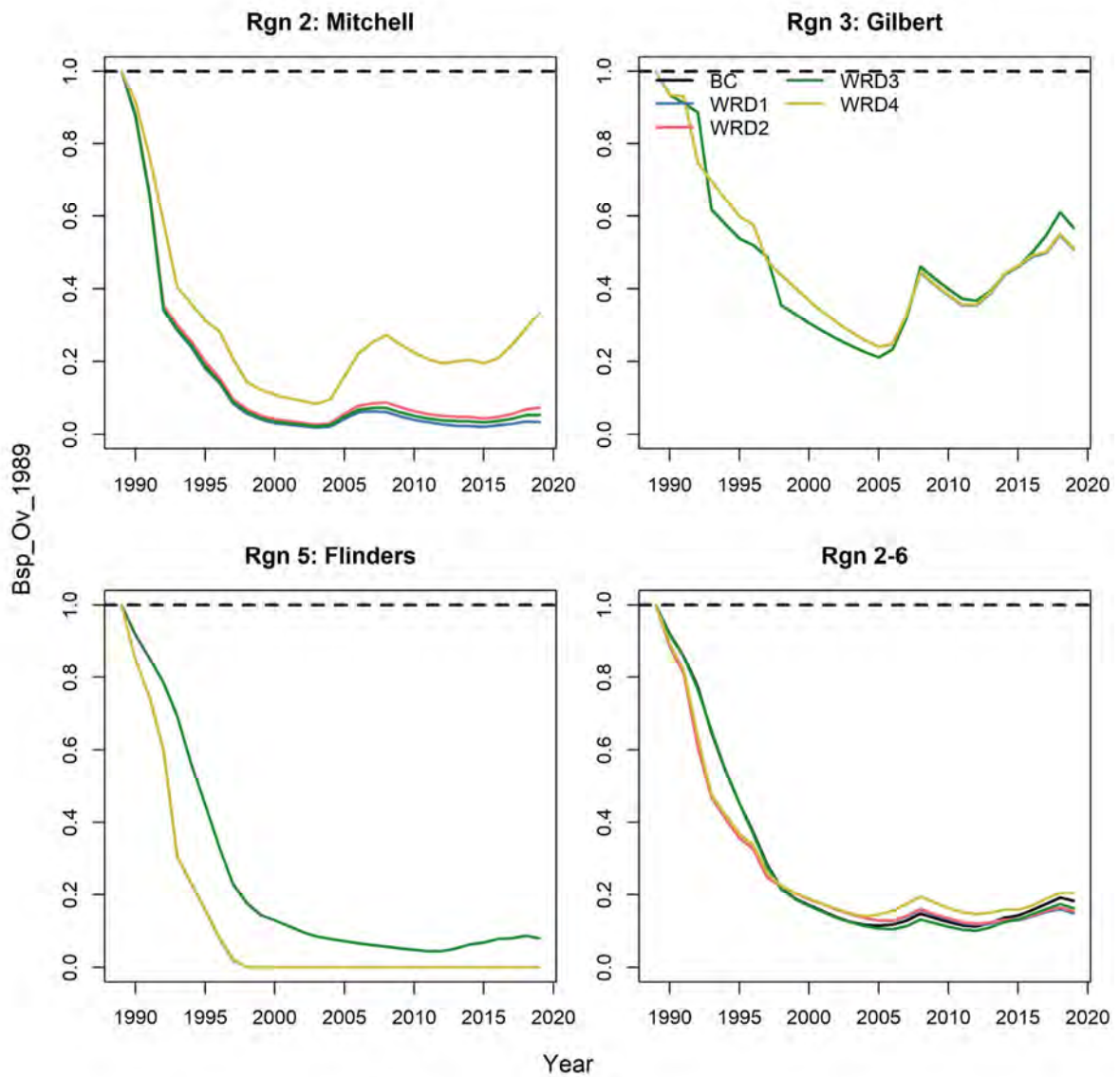


Figure A141. MICE-predicted changes to total largemouth sawfish mature numbers (Bsp; relative to 1989 level) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 2. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4.

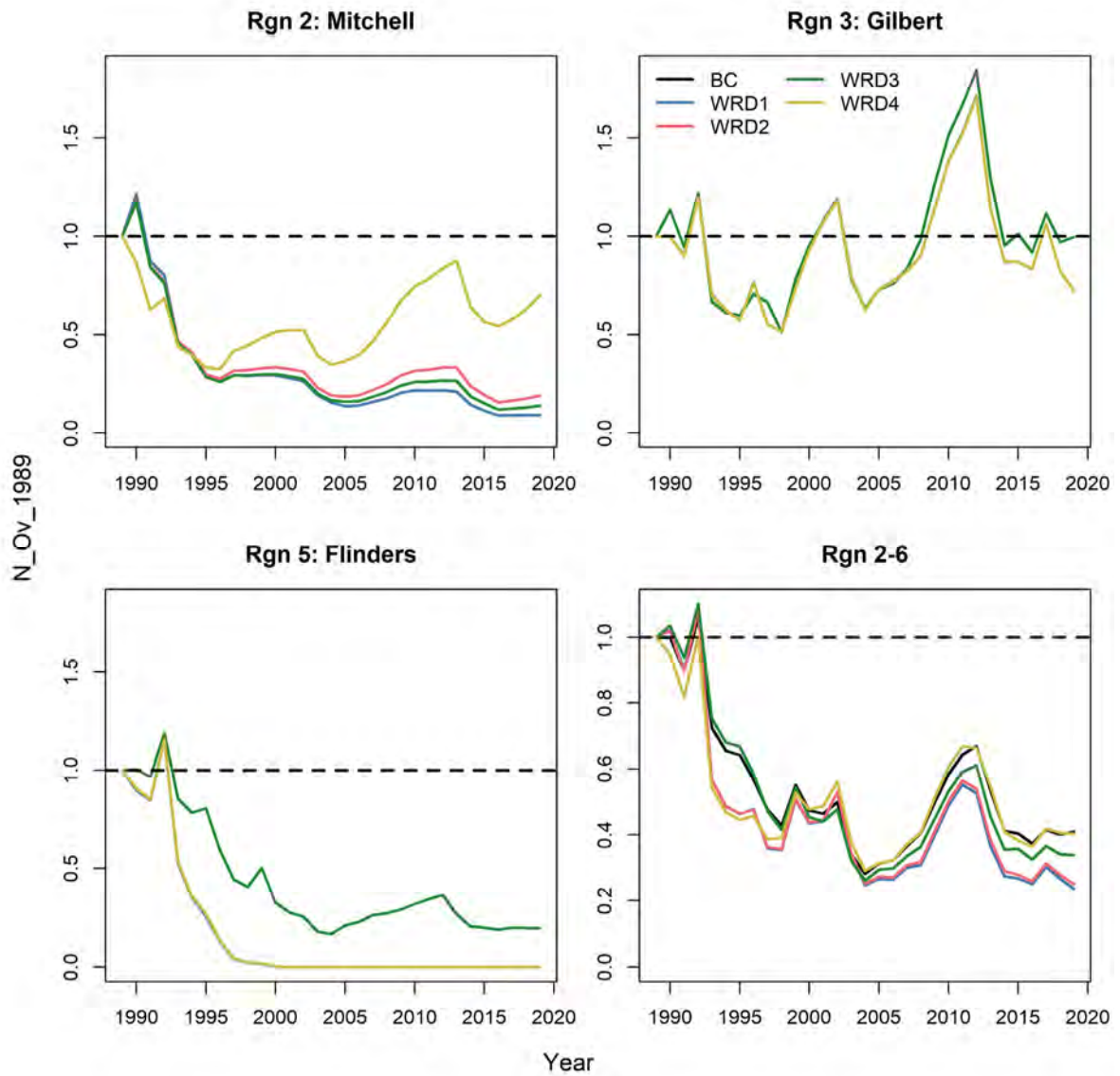


Figure A142. MICE-predicted changes to total largemouth sawfish numbers (relative to 1989 level) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 2. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4.

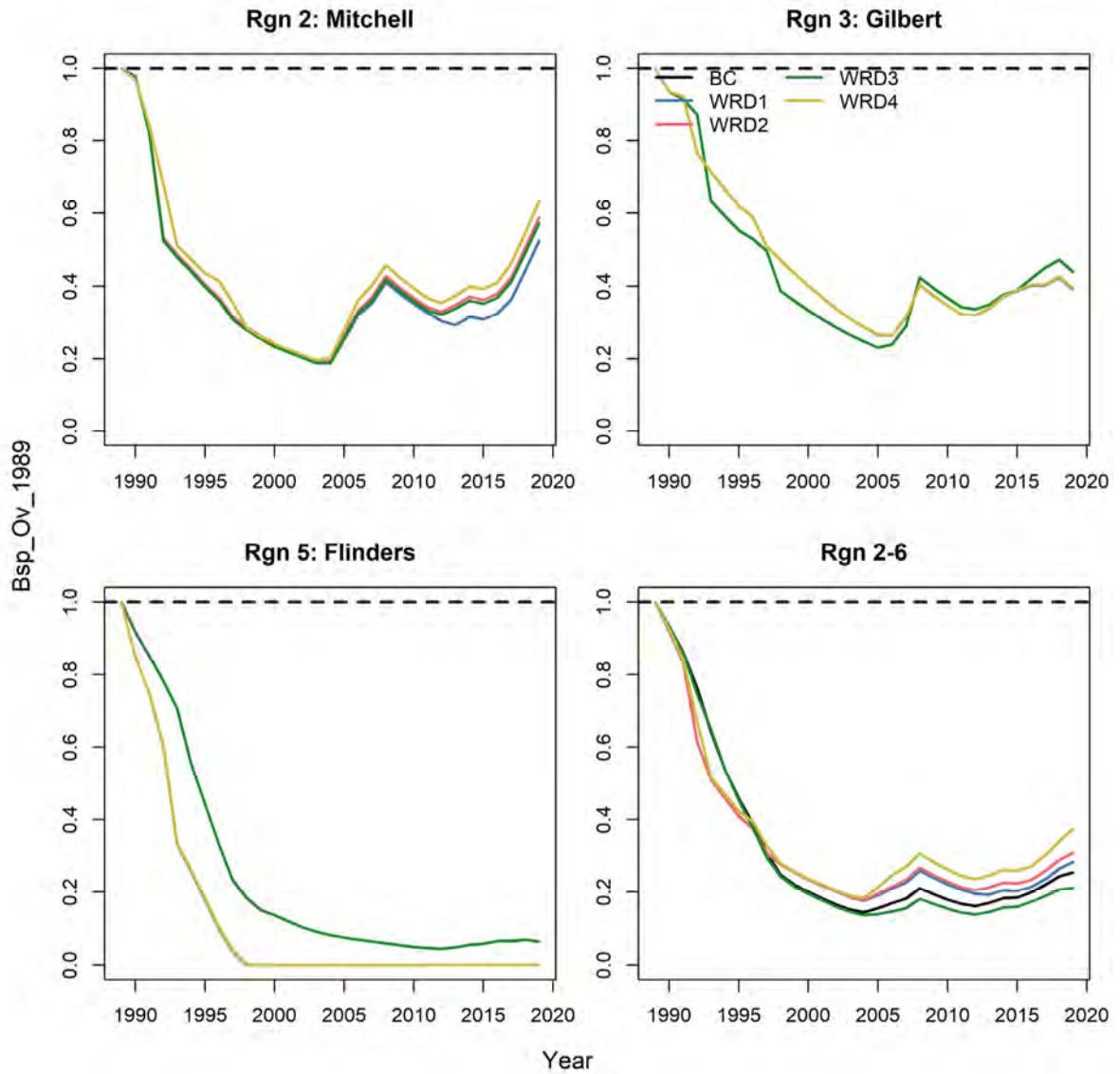


Figure A143. MICE-predicted changes to total largemouth sawfish mature numbers (Bsp; relative to 1989 level) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 3. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4.

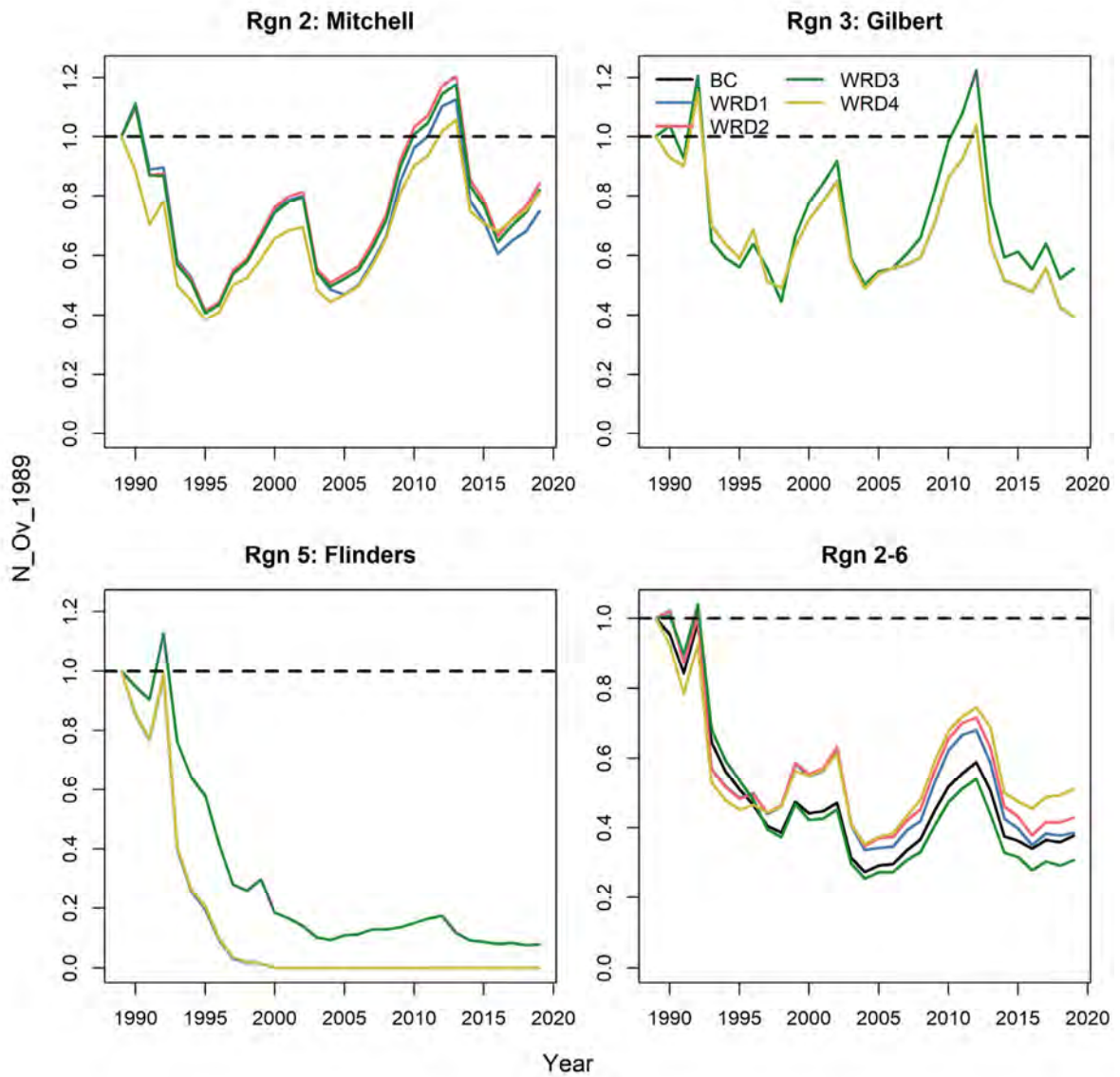


Figure A144. MICE-predicted changes to total largemouth sawfish numbers (relative to 1989 level) under baseline (BC) flow conditions compared with four alternative WRDs, shown for the Mitchell (Region 2), Gilbert (Region 3) and Flinders (Region 5) catchment systems, as well as Regions 2-6 combined using Model version 3. Note for the Gilbert and Flinders Rivers, the BC trajectory tracks under WRD3 and WRD2, WRD1 tracks under WRD4.

Table A31. MICE model parameter estimates, associated Hessian-based standard deviations (std), number of parameters estimated, negative log likelihood (-lnL) contributions and Akaike Information Criterion (AIC) scores across each of the five model versions for barramundi when comparing Model version 2 with Model version 6 which includes trophic linkages. Parameter estimates in italics are fixed parameters. Bsp = spawning biomass (t); CAA = catch-at-age; SOI = southern oscillation index; Mitch = Mitchell River; Gilb = Gilbert River; Norm = Norman River; Flind = Flinders River

Parameter	Description	<u>Model 2 - flow relationship</u>		<u>Model6 - trophic</u>	
		value	std	value	std
(2) Barramundi					
Mbase	Natural mortality base	8.00E-03	1.32E-07	8.00E-03	1.32E-07
q(barra) (1)	Catchability	1.63E-04	2.12E-05	1.63E-04	2.12E-05
q(barra) (2)	Catchability	2.58E-05	1.89E-06	2.58E-05	1.89E-06
q(barra) (3-6)	Catchability	7.87E-05	4.03E-06	7.87E-05	4.03E-06
q(barra) (7-8)	Catchability	3.14E-04	6.42E-06	3.14E-04	6.42E-06
Thres (1)	Flow threshold parameter	0.268	0.138	0.268	0.138
Thres (2)	Flow threshold parameter	0.389	0.092	0.389	0.092
Thres (3)	Flow threshold parameter	2.472	0.847	2.472	0.847
Thres (4)	Flow threshold parameter	20.000	0.008	20.000	0.008
Thres (5-6)	Flow threshold parameter	5.535	4.063	5.535	4.063
Thres (7-8)	Flow threshold parameter	20.000	0.014	20.000	0.014
del_barra_S	Age-selectivity par (Southern)	4.63E-01	2.94E-02	4.63E-01	2.94E-02
sfa_barra_S	Age-selectivity par (Southern)	1.88E-01	4.57E-02	1.88E-01	4.57E-02
del_barra_N	Age-selectivity par (Northern)	1.00E+00	4.11E-04	1.00E+00	4.11E-04
sfa_barra_N	Age-selectivity par (Northern)	1.00E-02	8.87E-05	1.00E-02	8.87E-05
MaxThres	Flow threshold parameter	<i>0.80</i>	-	<i>0.80</i>	-
hpred	Predator-prey interaction parameter	-	-	0.85	3.67e-04
No. parameters		15		16	
Likelihood contributions		value	sigma	value	sigma
-lnL: Catch (Region 1)		79.3	0.9	83.4	0.9
-lnL: Catch (Region 2)		106.2	1.0	139.3	1.2
-lnL: Catch (Region 3)		152.6	1.2	124.3	1.1
-lnL: Catch (Region 4)		107.7	1.0	113.9	1.0
-lnL: Catch (Region 5)		99.8	1.0	131.1	1.1
-lnL: Catch (Region 6)		94.4	0.9	96.1	0.9
-lnL: Catch (Region 7)		81.2	0.9	96.3	0.9
-lnL: Catch (Region 8)		105.9	1.0	119.9	1.1
-lnL: CAA (Northern)		45.5	0.2	42.9	0.2
-lnL: CAA (Southern)		58.2	0.2	60.0	0.2
-lnL: overall		931.0	-	1007.2	-
AIC		1891.9	-	2046.4	-
<u>Other fixed parameters</u>					
Mcon	Slope of age-dependent mortality	0.045		0.045	
Sex ratio		Calculated		Calculated	

Appendix 19 – MICE WRD All Scenarios Results

A full list of all 19 WRDs relative to baseline flows and the associated changes in biomass and/or catch for each species/group under each WRD is shown in the tables below. These results are based on Model version 1.

Table A32. Summary table of Model version 1 showing minimum and mean biomass for seagrass and mangroves predicted under 19 water resource development scenarios (WRD1-WRD19) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.

Model version 1 - seagrass and mangrove					
WRD Scenario	Rgn Name	Rel.Seagrass (Min)	Rel.Seagrass (Mean)	Rel.Mangrove (Min)	Rel.Mangrove (Mean)
BaseCase	Mitchell	1.00	1.00	1.00	1.00
BaseCase	Gilbert	1.00	1.00	1.00	1.00
BaseCase	Flinders	1.00	1.00	1.00	1.00
BaseCase	Rgn 2-6	1.00	1.00	1.00	1.00
WRD1	Mitchell	1.00	1.02	0.56	0.74
WRD1	Gilbert	0.98	1.04	0.27	0.56
WRD1	Flinders	0.94	1.02	0.47	0.72
WRD1	Rgn 2-6	0.99	1.02	0.59	0.74
WRD2	Mitchell	1.00	1.01	0.84	0.94
WRD2	Gilbert	0.98	1.04	0.29	0.60
WRD2	Flinders	0.93	1.02	0.50	0.74
WRD2	Rgn 2-6	0.99	1.01	0.85	0.92
WRD3	Mitchell	1.00	1.02	0.61	0.78
WRD3	Gilbert	1.00	1.00	1.00	1.00
WRD3	Flinders	1.00	1.00	1.00	1.00
WRD3	Rgn 2-6	1.00	1.00	0.70	0.86
WRD4	Mitchell	1.00	1.00	1.00	1.00
WRD4	Gilbert	0.98	1.04	0.29	0.60
WRD4	Flinders	0.93	1.02	0.50	0.74
WRD4	Rgn 2-6	0.99	1.01	0.91	0.95
WRD5	Mitchell	1.00	1.04	0.41	0.66
WRD5	Gilbert	0.98	1.04	0.27	0.56
WRD5	Flinders	0.94	1.02	0.47	0.72
WRD5	Rgn 2-6	1.00	1.02	0.51	0.73
WRD6	Mitchell	1.00	1.04	0.41	0.62
WRD6	Gilbert	0.98	1.04	0.27	0.56
WRD6	Flinders	0.94	1.02	0.47	0.72
WRD6	Rgn 2-6	1.00	1.02	0.55	0.70
WRD7	Mitchell	1.00	1.00	0.98	0.99
WRD7	Gilbert	0.98	1.04	0.27	0.56
WRD7	Flinders	1.00	1.00	1.00	1.00
WRD7	Rgn 2-6	0.99	1.01	0.85	0.92

WRD8	Mitchell	1.00	1.01	0.65	0.84
WRD8	Gilbert	1.00	1.00	1.00	1.00
WRD8	Flinders	1.00	1.00	1.00	1.00
WRD8	Rgn 2-6	1.00	1.00	0.73	0.90
WRD9	Mitchell	1.00	1.00	0.95	0.98
WRD9	Gilbert	1.00	1.00	1.00	1.00
WRD9	Flinders	1.00	1.00	1.00	1.00
WRD9	Rgn 2-6	1.00	1.00	0.96	0.99
WRD10	Mitchell	1.00	1.02	0.83	0.93
WRD10	Gilbert	0.98	1.04	0.29	0.60
WRD10	Flinders	0.93	1.02	0.50	0.74
WRD10	Rgn 2-6	0.99	1.01	0.84	0.91
WRD11	Mitchell	1.00	1.02	0.61	0.78
WRD11	Gilbert	1.00	1.00	1.00	1.00
WRD11	Flinders	1.00	1.00	1.00	1.00
WRD11	Rgn 2-6	1.00	1.00	0.71	0.86
WRD12	Mitchell	1.00	1.02	0.82	0.92
WRD12	Gilbert	1.00	1.00	1.00	1.00
WRD12	Flinders	1.00	1.00	1.00	1.00
WRD12	Rgn 2-6	1.00	1.00	0.89	0.95
WRD13	Mitchell	1.00	1.02	0.83	0.93
WRD13	Gilbert	1.00	1.00	1.00	1.00
WRD13	Flinders	1.00	1.00	1.00	1.00
WRD13	Rgn 2-6	1.00	1.00	0.89	0.95
WRD14	Mitchell	1.00	1.01	0.84	0.94
WRD14	Gilbert	1.00	1.00	1.00	1.00
WRD14	Flinders	1.00	1.00	1.00	1.00
WRD14	Rgn 2-6	1.00	1.00	0.90	0.96
WRD15	Mitchell	1.00	1.01	0.85	0.94
WRD15	Gilbert	1.00	1.00	1.00	1.00
WRD15	Flinders	1.00	1.00	1.00	1.00
WRD15	Rgn 2-6	1.00	1.00	0.91	0.96
WRD16	Mitchell	1.00	1.03	0.49	0.73
WRD16	Gilbert	1.00	1.00	1.00	1.00
WRD16	Flinders	1.00	1.00	1.00	1.00
WRD16	Rgn 2-6	1.00	1.01	0.61	0.82
WRD17	Mitchell	1.00	1.00	0.98	0.99
WRD17	Gilbert	1.00	1.00	1.00	1.00
WRD17	Flinders	1.00	1.00	1.00	1.00
WRD17	Rgn 2-6	1.00	1.00	0.99	1.00
WRD18	Mitchell	1.00	1.02	0.56	0.74
WRD18	Gilbert	1.00	1.00	1.00	1.00
WRD18	Flinders	1.00	1.00	1.00	1.00
WRD18	Rgn 2-6	1.00	1.00	0.66	0.84
WRD19	Mitchell	1.00	1.02	0.79	0.92
WRD19	Gilbert	1.00	1.00	1.00	1.00
WRD19	Flinders	1.00	1.00	1.00	1.00
WRD19	Rgn 2-6	1.00	1.00	0.87	0.95

Table A33. Summary table of Model version 1 showing minimum and mean catch, commercially available biomass (Bcom) and spawning biomass (Bsp) for common banana prawns predicted under 19 water resource development scenarios (WRD1-WRD19) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions2-6 combined.

WRD Scenario	Rgn Name	Model version 1 - prawn					
		Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)
BaseCase	Mitchell	1.00	1.00	1.00	1.00	1.00	1.00
BaseCase	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
BaseCase	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
BaseCase	Rgn 2-6	1.00	1.00	1.00	1.00	1.00	1.00
WRD1	Mitchell	0.68	0.82	0.70	0.84	0.71	0.82
WRD1	Gilbert	0.79	0.88	0.81	0.89	0.77	0.89
WRD1	Flinders	0.63	0.81	0.62	0.83	0.63	0.81
WRD1	Rgn 2-6	0.71	0.84	0.75	0.87	0.76	0.86
WRD2	Mitchell	0.77	0.89	0.77	0.91	0.79	0.90
WRD2	Gilbert	0.93	0.96	0.94	0.97	0.94	0.97
WRD2	Flinders	0.68	0.85	0.66	0.87	0.67	0.85
WRD2	Rgn 2-6	0.79	0.90	0.80	0.92	0.81	0.91
WRD3	Mitchell	0.85	0.93	0.88	0.94	0.89	0.94
WRD3	Gilbert	0.82	0.91	0.85	0.92	0.86	0.92
WRD3	Flinders	0.94	0.97	0.95	0.98	0.95	0.98
WRD3	Rgn 2-6	0.88	0.95	0.92	0.96	0.92	0.96
WRD4	Mitchell	0.82	0.91	0.80	0.93	0.82	0.91
WRD4	Gilbert	0.98	0.99	0.99	0.99	0.98	0.99
WRD4	Flinders	0.69	0.86	0.67	0.88	0.68	0.86
WRD4	Rgn 2-6	0.79	0.91	0.82	0.93	0.83	0.92
WRD5	Mitchell	0.62	0.75	0.59	0.74	0.53	0.68
WRD5	Gilbert	0.65	0.79	0.63	0.77	0.59	0.72
WRD5	Flinders	0.63	0.78	0.60	0.79	0.60	0.76
WRD5	Rgn 2-6	0.67	0.79	0.69	0.82	0.66	0.78
WRD6	Mitchell	0.62	0.75	0.59	0.77	0.58	0.75
WRD6	Gilbert	0.66	0.80	0.59	0.80	0.55	0.79
WRD6	Flinders	0.63	0.79	0.61	0.80	0.63	0.78
WRD6	Rgn 2-6	0.65	0.79	0.72	0.83	0.69	0.82
WRD7	Mitchell	0.85	0.93	0.85	0.93	0.85	0.92
WRD7	Gilbert	0.98	0.99	0.98	0.99	0.98	0.99
WRD7	Flinders	0.83	0.92	0.83	0.92	0.83	0.92
WRD7	Rgn 2-6	0.88	0.94	0.90	0.96	0.90	0.95
WRD8	Mitchell	0.90	0.95	0.90	0.96	0.91	0.96
WRD8	Gilbert	0.87	0.94	0.88	0.94	0.89	0.94
WRD8	Flinders	0.96	0.98	0.96	0.98	0.96	0.98
WRD8	Rgn 2-6	0.92	0.97	0.94	0.97	0.94	0.97
WRD9	Mitchell	0.98	0.99	0.98	0.99	0.99	0.99
WRD9	Gilbert	0.97	0.99	0.98	0.99	0.98	0.99
WRD9	Flinders	0.99	1.00	0.99	1.00	0.99	1.00
WRD9	Rgn 2-6	0.98	0.99	0.99	1.00	0.99	1.00

WRD10	Mitchell	0.75	0.88	0.76	0.90	0.77	0.89
WRD10	Gilbert	0.90	0.95	0.92	0.96	0.92	0.96
WRD10	Flinders	0.67	0.85	0.65	0.87	0.67	0.85
WRD10	Rgn 2-6	0.78	0.89	0.79	0.92	0.80	0.91
WRD11	Mitchell	0.85	0.93	0.88	0.94	0.89	0.94
WRD11	Gilbert	0.82	0.91	0.85	0.92	0.86	0.92
WRD11	Flinders	0.94	0.97	0.95	0.98	0.95	0.98
WRD11	Rgn 2-6	0.88	0.95	0.92	0.96	0.92	0.96
WRD12	Mitchell	0.94	0.97	0.95	0.97	0.95	0.98
WRD12	Gilbert	0.91	0.96	0.93	0.96	0.93	0.97
WRD12	Flinders	0.98	0.99	0.98	0.99	0.98	0.99
WRD12	Rgn 2-6	0.94	0.98	0.97	0.98	0.97	0.98
WRD13	Mitchell	0.94	0.97	0.95	0.98	0.95	0.98
WRD13	Gilbert	0.91	0.96	0.93	0.97	0.93	0.97
WRD13	Flinders	0.98	0.99	0.98	0.99	0.98	0.99
WRD13	Rgn 2-6	0.94	0.98	0.97	0.98	0.97	0.99
WRD14	Mitchell	0.95	0.98	0.96	0.98	0.97	0.98
WRD14	Gilbert	0.94	0.97	0.95	0.97	0.96	0.98
WRD14	Flinders	0.98	0.99	0.98	0.99	0.99	0.99
WRD14	Rgn 2-6	0.96	0.98	0.98	0.99	0.98	0.99
WRD15	Mitchell	0.95	0.98	0.97	0.98	0.97	0.98
WRD15	Gilbert	0.94	0.97	0.95	0.97	0.95	0.98
WRD15	Flinders	0.98	0.99	0.98	0.99	0.99	0.99
WRD15	Rgn 2-6	0.96	0.98	0.98	0.99	0.98	0.99
WRD16	Mitchell	0.75	0.88	0.74	0.86	0.68	0.81
WRD16	Gilbert	0.71	0.85	0.70	0.82	0.67	0.78
WRD16	Flinders	0.91	0.95	0.89	0.95	0.87	0.93
WRD16	Rgn 2-6	0.83	0.91	0.84	0.91	0.80	0.89
WRD17	Mitchell	0.99	1.00	0.99	1.00	0.99	1.00
WRD17	Gilbert	0.99	1.00	0.99	1.00	0.99	1.00
WRD17	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD17	Rgn 2-6	0.99	1.00	0.99	1.00	0.99	1.00
WRD18	Mitchell	0.83	0.91	0.85	0.92	0.81	0.92
WRD18	Gilbert	0.80	0.88	0.82	0.90	0.78	0.89
WRD18	Flinders	0.94	0.97	0.94	0.97	0.94	0.97
WRD18	Rgn 2-6	0.86	0.94	0.91	0.95	0.89	0.95
WRD19	Mitchell	0.93	0.96	0.94	0.97	0.95	0.97
WRD19	Gilbert	0.90	0.95	0.92	0.96	0.93	0.96
WRD19	Flinders	0.97	0.99	0.98	0.99	0.98	0.99
WRD19	Rgn 2-6	0.95	0.97	0.96	0.98	0.97	0.98

Table A34. Summary table of Model version 1 showing minimum and mean catch, commercially available biomass (Bcom) and spawning biomass (Bsp) for barramundi predicted under 19 water resource development scenarios (WRD1-WRD19) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.

Model version 1 - barramundi							
WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)
BaseCase	Mitchell	1.00	1.00	1.00	1.00	1.00	1.00
BaseCase	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
BaseCase	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
BaseCase	Rgn 2-6	1.00	1.00	1.00	1.00	1.00	1.00
WRD1	Mitchell	0.74	0.84	0.74	0.84	0.78	0.85
WRD1	Gilbert	0.16	0.32	0.38	0.51	0.44	0.55
WRD1	Flinders	0.39	0.72	0.71	0.79	0.74	0.82
WRD1	Rgn 2-6	0.54	0.70	0.73	0.79	0.76	0.82
WRD2	Mitchell	0.94	0.97	0.94	0.97	0.95	0.97
WRD2	Gilbert	0.23	0.42	0.45	0.58	0.51	0.63
WRD2	Flinders	0.40	0.73	0.72	0.80	0.75	0.83
WRD2	Rgn 2-6	0.65	0.79	0.84	0.88	0.89	0.92
WRD3	Mitchell	0.81	0.87	0.81	0.87	0.83	0.88
WRD3	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD3	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD3	Rgn 2-6	0.92	0.96	0.93	0.96	0.92	0.94
WRD4	Mitchell	1.00	1.00	1.00	1.00	1.00	1.00
WRD4	Gilbert	0.23	0.42	0.45	0.58	0.51	0.63
WRD4	Flinders	0.40	0.73	0.72	0.80	0.75	0.83
WRD4	Rgn 2-6	0.65	0.79	0.85	0.89	0.91	0.93
WRD5	Mitchell	0.45	0.58	0.45	0.58	0.49	0.62
WRD5	Gilbert	0.16	0.32	0.38	0.51	0.44	0.55
WRD5	Flinders	0.39	0.72	0.71	0.79	0.74	0.82
WRD5	Rgn 2-6	0.50	0.64	0.70	0.75	0.66	0.74
WRD6	Mitchell	0.60	0.71	0.60	0.71	0.63	0.73
WRD6	Gilbert	0.16	0.32	0.38	0.51	0.44	0.55
WRD6	Flinders	0.39	0.72	0.71	0.79	0.74	0.82
WRD6	Rgn 2-6	0.56	0.68	0.73	0.79	0.74	0.79
WRD7	Mitchell	0.99	0.99	0.99	0.99	0.99	0.99
WRD7	Gilbert	0.16	0.32	0.38	0.51	0.44	0.55
WRD7	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD7	Rgn 2-6	0.70	0.83	0.82	0.87	0.88	0.91
WRD8	Mitchell	0.85	0.90	0.85	0.90	0.87	0.90
WRD8	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD8	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD8	Rgn 2-6	0.94	0.97	0.95	0.97	0.93	0.95
WRD9	Mitchell	0.98	0.99	0.98	0.99	0.98	0.99
WRD9	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD9	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD9	Rgn 2-6	0.99	1.00	0.99	1.00	0.99	0.99

WRD10	Mitchell	0.93	0.96	0.93	0.96	0.94	0.96
WRD10	Gilbert	0.23	0.42	0.45	0.58	0.51	0.63
WRD10	Flinders	0.40	0.73	0.72	0.80	0.75	0.83
WRD10	Rgn 2-6	0.64	0.78	0.84	0.88	0.88	0.91
WRD11	Mitchell	0.81	0.87	0.81	0.87	0.83	0.88
WRD11	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD11	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD11	Rgn 2-6	0.92	0.96	0.93	0.96	0.92	0.94
WRD12	Mitchell	0.93	0.96	0.93	0.96	0.94	0.96
WRD12	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD12	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD12	Rgn 2-6	0.97	0.99	0.98	0.99	0.97	0.98
WRD13	Mitchell	0.93	0.96	0.93	0.96	0.94	0.96
WRD13	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD13	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD13	Rgn 2-6	0.97	0.99	0.98	0.99	0.97	0.98
WRD14	Mitchell	0.94	0.97	0.94	0.97	0.95	0.97
WRD14	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD14	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD14	Rgn 2-6	0.98	0.99	0.98	0.99	0.98	0.99
WRD15	Mitchell	0.94	0.97	0.94	0.97	0.95	0.97
WRD15	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD15	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD15	Rgn 2-6	0.98	0.99	0.98	0.99	0.98	0.99
WRD16	Mitchell	0.52	0.63	0.52	0.63	0.55	0.67
WRD16	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD16	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD16	Rgn 2-6	0.80	0.89	0.83	0.89	0.78	0.84
WRD17	Mitchell	0.99	0.99	0.99	0.99	0.99	0.99
WRD17	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD17	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD17	Rgn 2-6	0.99	1.00	1.00	1.00	0.99	1.00
WRD18	Mitchell	0.74	0.84	0.74	0.84	0.78	0.85
WRD18	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD18	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD18	Rgn 2-6	0.89	0.95	0.91	0.95	0.89	0.93
WRD19	Mitchell	0.93	0.96	0.93	0.96	0.94	0.96
WRD19	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD19	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD19	Rgn 2-6	0.97	0.99	0.98	0.99	0.97	0.98

Table A35. Summary table of Model version 1 showing minimum and mean catch, commercially available biomass (Bcom) and spawning biomass (Bsp) for mud crabs predicted under 19 water resource development scenarios (WRD1-WRD19) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.

Model version 1 - mud crab							
WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)
BaseCase	Mitchell	1.00	1.00	1.00	1.00	1.00	1.00
BaseCase	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
BaseCase	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
BaseCase	Rgn 2-6	1.00	1.00	1.00	1.00	1.00	1.00
WRD1	Mitchell	0.99	1.01	0.99	1.01	0.99	1.01
WRD1	Gilbert	0.24	0.53	0.25	0.55	0.24	0.53
WRD1	Flinders	0.14	0.54	0.16	0.55	0.19	0.54
WRD1	Rgn 2-6	0.44	0.72	0.57	0.74	0.57	0.74
WRD2	Mitchell	1.00	1.02	1.00	1.02	1.00	1.02
WRD2	Gilbert	0.32	0.64	0.33	0.65	0.32	0.64
WRD2	Flinders	0.13	0.53	0.16	0.53	0.19	0.53
WRD2	Rgn 2-6	0.51	0.77	0.69	0.82	0.66	0.81
WRD3	Mitchell	0.99	1.01	0.99	1.01	1.00	1.01
WRD3	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD3	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD3	Rgn 2-6	1.00	1.00	1.00	1.00	1.00	1.00
WRD4	Mitchell	1.00	1.00	1.00	1.00	1.00	1.00
WRD4	Gilbert	0.32	0.64	0.33	0.65	0.32	0.64
WRD4	Flinders	0.13	0.53	0.16	0.53	0.19	0.53
WRD4	Rgn 2-6	0.51	0.77	0.69	0.82	0.66	0.81
WRD5	Mitchell	0.98	1.00	0.98	1.00	0.98	1.00
WRD5	Gilbert	0.24	0.53	0.25	0.55	0.24	0.53
WRD5	Flinders	0.14	0.54	0.16	0.55	0.19	0.54
WRD5	Rgn 2-6	0.45	0.72	0.66	0.79	0.63	0.77
WRD6	Mitchell	0.99	1.02	0.98	1.02	0.98	1.02
WRD6	Gilbert	0.24	0.53	0.25	0.55	0.24	0.53
WRD6	Flinders	0.14	0.54	0.16	0.55	0.19	0.54
WRD6	Rgn 2-6	0.45	0.72	0.66	0.79	0.63	0.77
WRD7	Mitchell	1.00	1.00	1.00	1.00	1.00	1.00
WRD7	Gilbert	0.24	0.53	0.25	0.55	0.24	0.53
WRD7	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD7	Rgn 2-6	0.52	0.79	0.72	0.81	0.70	0.81
WRD8	Mitchell	0.98	1.00	0.98	1.00	0.99	1.00
WRD8	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD8	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD8	Rgn 2-6	1.00	1.00	1.00	1.00	1.00	1.00
WRD9	Mitchell	1.00	1.00	1.00	1.00	1.00	1.00
WRD9	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD9	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD9	Rgn 2-6	1.00	1.00	1.00	1.00	1.00	1.00
WRD10	Mitchell	1.00	1.01	1.00	1.01	1.00	1.01

WRD10	Gilbert	0.32	0.64	0.33	0.65	0.32	0.64
WRD10	Flinders	0.13	0.53	0.16	0.53	0.19	0.53
WRD10	Rgn 2-6	0.51	0.77	0.69	0.82	0.66	0.81
WRD11	Mitchell	0.99	1.01	0.99	1.01	0.99	1.01
WRD11	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD11	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD11	Rgn 2-6	1.00	1.00	1.00	1.00	1.00	1.00
WRD12	Mitchell	1.00	1.02	1.00	1.02	1.00	1.02
WRD12	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD12	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD12	Rgn 2-6	1.00	1.00	1.00	1.00	1.00	1.00
WRD13	Mitchell	1.00	1.01	1.00	1.01	1.00	1.01
WRD13	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD13	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD13	Rgn 2-6	1.00	1.00	1.00	1.00	1.00	1.00
WRD14	Mitchell	1.00	1.02	1.00	1.02	1.00	1.02
WRD14	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD14	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD14	Rgn 2-6	1.00	1.00	1.00	1.00	1.00	1.00
WRD15	Mitchell	1.00	1.01	1.00	1.01	1.00	1.01
WRD15	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD15	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD15	Rgn 2-6	1.00	1.00	1.00	1.00	1.00	1.00
WRD16	Mitchell	0.98	1.00	0.98	1.00	0.98	0.99
WRD16	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD16	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD16	Rgn 2-6	1.00	1.00	1.00	1.00	1.00	1.00
WRD17	Mitchell	1.00	1.00	1.00	1.00	1.00	1.00
WRD17	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD17	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD17	Rgn 2-6	1.00	1.00	1.00	1.00	1.00	1.00
WRD18	Mitchell	0.99	1.01	0.99	1.01	0.99	1.01
WRD18	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD18	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD18	Rgn 2-6	1.00	1.00	1.00	1.00	1.00	1.00
WRD19	Mitchell	1.00	1.02	1.00	1.02	1.00	1.02
WRD19	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD19	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD19	Rgn 2-6	1.00	1.00	1.00	1.00	1.00	1.00

Table A36. Summary table of Model version 1 showing minimum and mean number of individuals, and number of mature individuals (Bsp) for largetooth sawfish predicted under 19 water resource development scenarios (WRD1-WRD19) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.

Model version 1 – largetooth sawfish					
WRD Scenario	Rgn Name	Rel.N (Min)	Rel.N (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)
BaseCase	Mitchell	1.00	1.00	1.00	1.00
BaseCase	Gilbert	1.00	1.00	1.00	1.00
BaseCase	Flinders	1.00	1.00	1.00	1.00
BaseCase	Rgn 2-6	1.00	1.00	1.00	1.00
WRD1	Mitchell	0.00	0.20	0.00	0.17
WRD1	Gilbert	0.33	0.47	0.39	0.51
WRD1	Flinders	0.00	0.05	0.00	0.04
WRD1	Rgn 2-6	0.22	0.31	0.25	0.34
WRD2	Mitchell	0.06	0.28	0.07	0.25
WRD2	Gilbert	0.34	0.47	0.40	0.51
WRD2	Flinders	0.00	0.06	0.00	0.04
WRD2	Rgn 2-6	0.23	0.33	0.26	0.35
WRD3	Mitchell	0.02	0.24	0.02	0.21
WRD3	Gilbert	1.00	1.00	1.00	1.00
WRD3	Flinders	1.00	1.00	1.00	1.00
WRD3	Rgn 2-6	0.76	0.86	0.84	0.91
WRD4	Mitchell	1.00	1.00	1.00	1.00
WRD4	Gilbert	0.34	0.47	0.40	0.51
WRD4	Flinders	0.00	0.06	0.00	0.04
WRD4	Rgn 2-6	0.34	0.46	0.29	0.43
WRD5	Mitchell	0.11	0.36	0.12	0.30
WRD5	Gilbert	0.39	0.71	0.39	0.51
WRD5	Flinders	0.00	0.07	0.00	0.04
WRD5	Rgn 2-6	0.30	0.48	0.26	0.35
WRD6	Mitchell	0.00	0.13	0.00	0.11
WRD6	Gilbert	0.33	0.47	0.39	0.51
WRD6	Flinders	0.00	0.05	0.00	0.04
WRD6	Rgn 2-6	0.21	0.30	0.25	0.33
WRD7	Mitchell	0.99	0.99	0.99	0.99
WRD7	Gilbert	0.33	0.47	0.39	0.51
WRD7	Flinders	1.00	1.00	1.00	1.00
WRD7	Rgn 2-6	0.80	0.88	0.81	0.89
WRD8	Mitchell	0.77	0.98	0.98	0.99
WRD8	Gilbert	1.00	1.00	1.00	1.00
WRD8	Flinders	1.00	1.00	1.00	1.00
WRD8	Rgn 2-6	0.97	1.00	1.00	1.00
WRD9	Mitchell	0.77	0.97	0.97	0.98
WRD9	Gilbert	1.00	1.00	1.00	1.00
WRD9	Flinders	1.00	1.00	1.00	1.00
WRD9	Rgn 2-6	0.97	1.00	1.00	1.00

WRD10	Mitchell	0.06	0.29	0.07	0.25
WRD10	Gilbert	0.34	0.47	0.40	0.51
WRD10	Flinders	0.00	0.06	0.00	0.04
WRD10	Rgn 2-6	0.23	0.33	0.26	0.35
WRD11	Mitchell	0.02	0.25	0.02	0.21
WRD11	Gilbert	1.00	1.00	1.00	1.00
WRD11	Flinders	1.00	1.00	1.00	1.00
WRD11	Rgn 2-6	0.76	0.86	0.84	0.91
WRD12	Mitchell	0.06	0.28	0.07	0.25
WRD12	Gilbert	1.00	1.00	1.00	1.00
WRD12	Flinders	1.00	1.00	1.00	1.00
WRD12	Rgn 2-6	0.77	0.87	0.85	0.92
WRD13	Mitchell	0.06	0.29	0.07	0.25
WRD13	Gilbert	1.00	1.00	1.00	1.00
WRD13	Flinders	1.00	1.00	1.00	1.00
WRD13	Rgn 2-6	0.77	0.87	0.85	0.92
WRD14	Mitchell	0.06	0.28	0.07	0.25
WRD14	Gilbert	1.00	1.00	1.00	1.00
WRD14	Flinders	1.00	1.00	1.00	1.00
WRD14	Rgn 2-6	0.77	0.87	0.85	0.92
WRD15	Mitchell	0.06	0.28	0.07	0.25
WRD15	Gilbert	1.00	1.00	1.00	1.00
WRD15	Flinders	1.00	1.00	1.00	1.00
WRD15	Rgn 2-6	0.77	0.87	0.85	0.92
WRD16	Mitchell	0.44	0.59	0.44	0.56
WRD16	Gilbert	1.00	1.00	1.00	1.00
WRD16	Flinders	1.00	1.00	1.00	1.00
WRD16	Rgn 2-6	0.86	0.92	0.91	0.95
WRD17	Mitchell	0.99	0.99	0.99	0.99
WRD17	Gilbert	1.00	1.00	1.00	1.00
WRD17	Flinders	1.00	1.00	1.00	1.00
WRD17	Rgn 2-6	1.00	1.00	1.00	1.00
WRD18	Mitchell	0.00	0.20	0.00	0.17
WRD18	Gilbert	1.00	1.00	1.00	1.00
WRD18	Flinders	1.00	1.00	1.00	1.00
WRD18	Rgn 2-6	0.76	0.86	0.84	0.91
WRD19	Mitchell	0.01	0.24	0.01	0.21
WRD19	Gilbert	1.00	1.00	1.00	1.00
WRD19	Flinders	1.00	1.00	1.00	1.00
WRD19	Rgn 2-6	0.76	0.86	0.84	0.91

A full list of all 19 WRDs relative to baseline flows and the associated changes in biomass and/or catch for each species/group under each WRD is shown in the tables below. These results are based on the MICE ensemble (average of all five models). Individual model results are stored electronically and available on request.

Table A37. Summary table of MICE ensemble showing minimum and mean biomass for seagrass and mangroves predicted under 19 water resource development scenarios (WRD1-WRD19) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.

MICE Ensemble – seagrass and mangrove					
WRD Scenario	Rgn Name	Rel.Seagrass (Min)	Rel.Seagrass (Mean)	Rel.Mangrove (Min)	Rel.Mangrove (Mean)
BaseCase	Mitchell	1.00	1.00	1.00	1.00
BaseCase	Gilbert	1.00	1.00	1.00	1.00
BaseCase	Flinders	1.00	1.00	1.00	1.00
BaseCase	Rgn 2-6	1.00	1.00	1.00	1.00
WRD1	Mitchell	1.00	1.04	0.58	0.75
WRD1	Gilbert	0.99	1.07	0.30	0.58
WRD1	Flinders	0.92	1.04	0.48	0.73
WRD1	Rgn 2-6	0.99	1.04	0.61	0.75
WRD2	Mitchell	1.00	1.02	0.84	0.94
WRD2	Gilbert	0.98	1.07	0.32	0.62
WRD2	Flinders	0.91	1.03	0.52	0.75
WRD2	Rgn 2-6	0.99	1.02	0.85	0.92
WRD3	Mitchell	1.00	1.03	0.62	0.78
WRD3	Gilbert	1.00	1.00	1.00	1.00
WRD3	Flinders	1.00	1.00	1.00	1.00
WRD3	Rgn 2-6	1.00	1.01	0.71	0.86
WRD4	Mitchell	1.00	1.00	1.00	1.00
WRD4	Gilbert	0.98	1.07	0.32	0.62
WRD4	Flinders	0.91	1.03	0.52	0.75
WRD4	Rgn 2-6	0.99	1.02	0.91	0.95
WRD5	Mitchell	1.00	1.05	0.42	0.67
WRD5	Gilbert	0.99	1.07	0.30	0.58
WRD5	Flinders	0.92	1.04	0.48	0.73
WRD5	Rgn 2-6	1.00	1.03	0.53	0.74
WRD6	Mitchell	1.00	1.06	0.42	0.63
WRD6	Gilbert	0.99	1.07	0.30	0.58
WRD6	Flinders	0.92	1.04	0.48	0.73
WRD6	Rgn 2-6	1.00	1.03	0.57	0.71
WRD7	Mitchell	1.00	1.00	0.98	0.99
WRD7	Gilbert	0.99	1.07	0.30	0.58
WRD7	Flinders	1.00	1.00	1.00	1.00
WRD7	Rgn 2-6	0.99	1.02	0.86	0.92
WRD8	Mitchell	1.00	1.02	0.66	0.84

WRD8	Gilbert	1.00	1.00	1.00	1.00
WRD8	Flinders	1.00	1.00	1.00	1.00
WRD8	Rgn 2-6	1.00	1.00	0.74	0.90
WRD9	Mitchell	1.00	1.00	0.95	0.98
WRD9	Gilbert	1.00	1.00	1.00	1.00
WRD9	Flinders	1.00	1.00	1.00	1.00
WRD9	Rgn 2-6	1.00	1.00	0.96	0.99
WRD10	Mitchell	1.00	1.02	0.83	0.93
WRD10	Gilbert	0.98	1.07	0.32	0.62
WRD10	Flinders	0.91	1.03	0.52	0.75
WRD10	Rgn 2-6	0.99	1.02	0.85	0.91
WRD11	Mitchell	1.00	1.03	0.63	0.79
WRD11	Gilbert	1.00	1.00	1.00	1.00
WRD11	Flinders	1.00	1.00	1.00	1.00
WRD11	Rgn 2-6	1.00	1.01	0.72	0.86
WRD12	Mitchell	1.00	1.02	0.82	0.92
WRD12	Gilbert	1.00	1.00	1.00	1.00
WRD12	Flinders	1.00	1.00	1.00	1.00
WRD12	Rgn 2-6	1.00	1.00	0.89	0.95
WRD13	Mitchell	1.00	1.02	0.83	0.93
WRD13	Gilbert	1.00	1.00	1.00	1.00
WRD13	Flinders	1.00	1.00	1.00	1.00
WRD13	Rgn 2-6	1.00	1.00	0.90	0.95
WRD14	Mitchell	1.00	1.02	0.84	0.94
WRD14	Gilbert	1.00	1.00	1.00	1.00
WRD14	Flinders	1.00	1.00	1.00	1.00
WRD14	Rgn 2-6	1.00	1.00	0.90	0.96
WRD15	Mitchell	1.00	1.02	0.85	0.94
WRD15	Gilbert	1.00	1.00	1.00	1.00
WRD15	Flinders	1.00	1.00	1.00	1.00
WRD15	Rgn 2-6	1.00	1.00	0.91	0.96
WRD16	Mitchell	1.00	1.04	0.50	0.73
WRD16	Gilbert	1.00	1.00	1.00	1.00
WRD16	Flinders	1.00	1.00	1.00	1.00
WRD16	Rgn 2-6	1.00	1.01	0.62	0.83
WRD17	Mitchell	1.00	1.00	0.98	0.99
WRD17	Gilbert	1.00	1.00	1.00	1.00
WRD17	Flinders	1.00	1.00	1.00	1.00
WRD17	Rgn 2-6	1.00	1.00	0.99	1.00
WRD18	Mitchell	1.00	1.04	0.58	0.75
WRD18	Gilbert	1.00	1.00	1.00	1.00
WRD18	Flinders	1.00	1.00	1.00	1.00
WRD18	Rgn 2-6	1.00	1.01	0.68	0.84
WRD19	Mitchell	1.00	1.02	0.79	0.92
WRD19	Gilbert	1.00	1.00	1.00	1.00
WRD19	Flinders	1.00	1.00	1.00	1.00

WRD19	Rgn 2-6	1.00	1.00	0.87	0.95
-------	---------	------	------	------	------

Table A38. Summary table of MICE ensemble showing minimum and mean catch, commercially available biomass (Bcom) and spawning biomass (Bsp) for common banana prawns predicted under 19 water resource development scenarios (WRD1-WRD19) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.

WRD Scenario	Rgn Name	MICE Ensemble – Common banana prawn					
		Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)
BaseCase	Mitchell	1.00	1.00	1.00	1.00	1.00	1.00
BaseCase	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
BaseCase	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
BaseCase	Rgn 2-6	1.00	1.00	1.00	1.00	1.00	1.00
WRD1	Mitchell	0.67	0.80	0.68	0.82	0.68	0.81
WRD1	Gilbert	0.67	0.79	0.67	0.81	0.64	0.79
WRD1	Flinders	0.60	0.79	0.60	0.81	0.61	0.79
WRD1	Rgn 2-6	0.68	0.81	0.70	0.84	0.71	0.82
WRD2	Mitchell	0.78	0.89	0.79	0.90	0.80	0.89
WRD2	Gilbert	0.78	0.88	0.79	0.89	0.79	0.88
WRD2	Flinders	0.65	0.84	0.64	0.86	0.65	0.83
WRD2	Rgn 2-6	0.74	0.88	0.76	0.89	0.77	0.88
WRD3	Mitchell	0.86	0.93	0.89	0.94	0.89	0.94
WRD3	Gilbert	0.87	0.93	0.89	0.94	0.90	0.94
WRD3	Flinders	0.96	0.98	0.96	0.98	0.96	0.98
WRD3	Rgn 2-6	0.90	0.96	0.94	0.97	0.94	0.97
WRD4	Mitchell	0.82	0.91	0.82	0.92	0.83	0.91
WRD4	Gilbert	0.82	0.90	0.82	0.91	0.82	0.90
WRD4	Flinders	0.65	0.84	0.65	0.86	0.66	0.84
WRD4	Rgn 2-6	0.75	0.89	0.77	0.90	0.78	0.89
WRD5	Mitchell	0.57	0.70	0.53	0.68	0.49	0.63
WRD5	Gilbert	0.55	0.73	0.53	0.71	0.49	0.66
WRD5	Flinders	0.58	0.75	0.56	0.76	0.55	0.73
WRD5	Rgn 2-6	0.62	0.74	0.63	0.76	0.60	0.73
WRD6	Mitchell	0.57	0.71	0.53	0.72	0.52	0.70
WRD6	Gilbert	0.56	0.73	0.50	0.73	0.48	0.72
WRD6	Flinders	0.59	0.75	0.58	0.77	0.58	0.75
WRD6	Rgn 2-6	0.60	0.75	0.66	0.78	0.65	0.76
WRD7	Mitchell	0.82	0.89	0.82	0.89	0.82	0.88
WRD7	Gilbert	0.82	0.89	0.81	0.89	0.81	0.88
WRD7	Flinders	0.82	0.90	0.82	0.90	0.83	0.89
WRD7	Rgn 2-6	0.82	0.90	0.84	0.90	0.84	0.90
WRD8	Mitchell	0.86	0.91	0.87	0.92	0.88	0.92
WRD8	Gilbert	0.90	0.95	0.91	0.95	0.91	0.95
WRD8	Flinders	0.94	0.96	0.95	0.97	0.95	0.96
WRD8	Rgn 2-6	0.90	0.94	0.92	0.95	0.92	0.95
WRD9	Mitchell	0.93	0.95	0.94	0.95	0.94	0.95

WRD9	Gilbert	0.97	0.98	0.97	0.98	0.97	0.98
WRD9	Flinders	0.97	0.97	0.97	0.97	0.97	0.97
WRD9	Rgn 2-6	0.95	0.96	0.96	0.96	0.96	0.96
WRD10	Mitchell	0.72	0.83	0.74	0.85	0.75	0.84
WRD10	Gilbert	0.74	0.85	0.75	0.86	0.75	0.86
WRD10	Flinders	0.62	0.81	0.62	0.83	0.63	0.80
WRD10	Rgn 2-6	0.70	0.83	0.72	0.85	0.73	0.84
WRD11	Mitchell	0.82	0.89	0.85	0.90	0.86	0.90
WRD11	Gilbert	0.86	0.92	0.88	0.93	0.89	0.93
WRD11	Flinders	0.93	0.96	0.94	0.96	0.94	0.96
WRD11	Rgn 2-6	0.87	0.92	0.91	0.94	0.91	0.94
WRD12	Mitchell	0.89	0.92	0.91	0.93	0.91	0.93
WRD12	Gilbert	0.92	0.95	0.93	0.96	0.94	0.96
WRD12	Flinders	0.95	0.96	0.96	0.97	0.96	0.97
WRD12	Rgn 2-6	0.91	0.94	0.94	0.95	0.94	0.95
WRD13	Mitchell	0.89	0.93	0.91	0.93	0.91	0.94
WRD13	Gilbert	0.92	0.96	0.94	0.96	0.94	0.97
WRD13	Flinders	0.96	0.97	0.96	0.97	0.96	0.97
WRD13	Rgn 2-6	0.91	0.95	0.94	0.96	0.94	0.96
WRD14	Mitchell	0.91	0.93	0.92	0.94	0.93	0.94
WRD14	Gilbert	0.94	0.96	0.95	0.97	0.96	0.97
WRD14	Flinders	0.96	0.97	0.96	0.97	0.96	0.97
WRD14	Rgn 2-6	0.93	0.95	0.95	0.96	0.95	0.96
WRD15	Mitchell	0.91	0.93	0.92	0.94	0.93	0.94
WRD15	Gilbert	0.94	0.96	0.95	0.97	0.96	0.97
WRD15	Flinders	0.96	0.97	0.96	0.97	0.96	0.97
WRD15	Rgn 2-6	0.93	0.95	0.95	0.96	0.95	0.96
WRD16	Mitchell	0.71	0.83	0.69	0.81	0.64	0.76
WRD16	Gilbert	0.76	0.87	0.75	0.85	0.72	0.81
WRD16	Flinders	0.89	0.94	0.88	0.93	0.87	0.92
WRD16	Rgn 2-6	0.79	0.88	0.83	0.90	0.79	0.87
WRD17	Mitchell	0.95	0.95	0.94	0.95	0.94	0.95
WRD17	Gilbert	0.98	0.99	0.98	0.99	0.98	0.99
WRD17	Flinders	0.97	0.98	0.97	0.97	0.97	0.97
WRD17	Rgn 2-6	0.96	0.96	0.96	0.97	0.96	0.97
WRD18	Mitchell	0.80	0.87	0.80	0.88	0.76	0.88
WRD18	Gilbert	0.84	0.90	0.84	0.91	0.81	0.91
WRD18	Flinders	0.92	0.95	0.93	0.95	0.92	0.95
WRD18	Rgn 2-6	0.85	0.91	0.89	0.93	0.87	0.93
WRD19	Mitchell	0.88	0.92	0.90	0.93	0.91	0.93
WRD19	Gilbert	0.91	0.95	0.92	0.96	0.94	0.96
WRD19	Flinders	0.95	0.96	0.96	0.97	0.96	0.97
WRD19	Rgn 2-6	0.91	0.94	0.94	0.95	0.94	0.96

Table A39. Summary table of Model version 1 showing minimum and mean catch, commercially available biomass (Bcom) and spawning biomass (Bsp) for barramundi predicted under 19 water resource development scenarios (WRD1-WRD19) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.

WRD Scenario	Rgn Name	MICE Ensemble - Barramundi					
		Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)
BaseCase	Mitchell	1.00	1.00	1.00	1.00	1.00	1.00
BaseCase	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
BaseCase	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
BaseCase	Rgn 2-6	1.00	1.00	1.00	1.00	1.00	1.00
WRD1	Mitchell	0.79	0.86	0.79	0.86	0.79	0.87
WRD1	Gilbert	0.39	0.56	0.61	0.72	0.67	0.78
WRD1	Flinders	0.61	0.84	0.82	0.89	0.82	0.92
WRD1	Rgn 2-6	0.66	0.80	0.81	0.86	0.83	0.89
WRD2	Mitchell	0.95	0.97	0.95	0.97	0.95	0.97
WRD2	Gilbert	0.44	0.63	0.64	0.76	0.70	0.81
WRD2	Flinders	0.62	0.85	0.82	0.89	0.82	0.92
WRD2	Rgn 2-6	0.76	0.87	0.90	0.93	0.93	0.96
WRD3	Mitchell	0.82	0.88	0.82	0.88	0.83	0.90
WRD3	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD3	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD3	Rgn 2-6	0.93	0.96	0.95	0.97	0.95	0.97
WRD4	Mitchell	1.00	1.00	1.00	1.00	1.00	1.00
WRD4	Gilbert	0.44	0.63	0.64	0.76	0.70	0.81
WRD4	Flinders	0.62	0.85	0.82	0.89	0.82	0.92
WRD4	Rgn 2-6	0.76	0.88	0.91	0.94	0.94	0.97
WRD5	Mitchell	0.46	0.65	0.59	1.07	0.56	1.77
WRD5	Gilbert	0.29	0.46	0.54	0.69	0.56	0.71
WRD5	Flinders	0.47	0.78	0.70	0.89	0.68	0.85
WRD5	Rgn 2-6	0.50	0.69	0.72	0.93	0.74	1.04
WRD6	Mitchell	0.56	0.78	0.69	1.32	0.65	2.24
WRD6	Gilbert	0.29	0.46	0.54	0.69	0.56	0.71
WRD6	Flinders	0.47	0.78	0.70	0.89	0.68	0.85
WRD6	Rgn 2-6	0.54	0.73	0.74	1.00	0.75	1.14
WRD7	Mitchell	0.84	1.05	0.97	1.85	0.93	3.23
WRD7	Gilbert	0.29	0.46	0.54	0.69	0.56	0.71
WRD7	Flinders	0.84	0.97	0.87	1.05	0.86	0.95
WRD7	Rgn 2-6	0.64	0.86	0.79	1.15	0.82	1.36
WRD8	Mitchell	0.74	0.96	0.88	1.67	0.84	2.88
WRD8	Gilbert	0.87	1.01	0.90	1.13	0.88	1.02
WRD8	Flinders	0.84	0.97	0.87	1.05	0.86	0.95
WRD8	Rgn 2-6	0.81	0.95	0.87	1.19	0.88	1.35
WRD9	Mitchell	0.83	1.05	0.97	1.84	0.92	3.21
WRD9	Gilbert	0.87	1.01	0.90	1.13	0.88	1.02

WRD9	Flinders	0.84	0.97	0.87	1.05	0.86	0.95
WRD9	Rgn 2-6	0.85	0.98	0.90	1.24	0.90	1.42
WRD10	Mitchell	0.79	1.02	0.93	1.79	0.89	3.10
WRD10	Gilbert	0.34	0.54	0.57	0.76	0.60	0.76
WRD10	Flinders	0.48	0.79	0.70	0.90	0.69	0.85
WRD10	Rgn 2-6	0.61	0.83	0.81	1.15	0.83	1.35
WRD11	Mitchell	0.71	0.93	0.85	1.62	0.81	2.78
WRD11	Gilbert	0.87	1.01	0.90	1.13	0.88	1.02
WRD11	Flinders	0.84	0.97	0.87	1.05	0.86	0.95
WRD11	Rgn 2-6	0.80	0.95	0.86	1.18	0.87	1.32
WRD12	Mitchell	0.79	1.01	0.93	1.78	0.88	3.09
WRD12	Gilbert	0.87	1.01	0.90	1.13	0.88	1.02
WRD12	Flinders	0.84	0.97	0.87	1.05	0.86	0.95
WRD12	Rgn 2-6	0.83	0.97	0.89	1.22	0.89	1.39
WRD13	Mitchell	0.79	1.02	0.93	1.79	0.89	3.10
WRD13	Gilbert	0.87	1.01	0.90	1.13	0.88	1.02
WRD13	Flinders	0.84	0.97	0.87	1.05	0.86	0.95
WRD13	Rgn 2-6	0.83	0.97	0.89	1.23	0.89	1.40
WRD14	Mitchell	0.80	1.03	0.94	1.80	0.90	3.13
WRD14	Gilbert	0.87	1.01	0.90	1.13	0.88	1.02
WRD14	Flinders	0.84	0.97	0.87	1.05	0.86	0.95
WRD14	Rgn 2-6	0.84	0.97	0.89	1.23	0.89	1.40
WRD15	Mitchell	0.81	1.03	0.94	1.81	0.90	3.14
WRD15	Gilbert	0.87	1.01	0.90	1.13	0.88	1.02
WRD15	Flinders	0.84	0.97	0.87	1.05	0.86	0.95
WRD15	Rgn 2-6	0.84	0.98	0.89	1.23	0.89	1.41
WRD16	Mitchell	0.51	0.70	0.64	1.17	0.61	1.96
WRD16	Gilbert	0.87	1.01	0.90	1.13	0.88	1.02
WRD16	Flinders	0.84	0.97	0.87	1.05	0.86	0.95
WRD16	Rgn 2-6	0.71	0.88	0.81	1.04	0.79	1.13
WRD17	Mitchell	0.84	1.05	0.97	1.85	0.93	3.23
WRD17	Gilbert	0.87	1.01	0.90	1.13	0.88	1.02
WRD17	Flinders	0.84	0.97	0.87	1.05	0.86	0.95
WRD17	Rgn 2-6	0.85	0.98	0.90	1.25	0.90	1.43
WRD18	Mitchell	0.68	0.90	0.82	1.56	0.77	2.66
WRD18	Gilbert	0.87	1.01	0.90	1.13	0.88	1.02
WRD18	Flinders	0.84	0.97	0.87	1.05	0.86	0.95
WRD18	Rgn 2-6	0.78	0.94	0.85	1.16	0.83	1.29
WRD19	Mitchell	0.79	1.01	0.93	1.78	0.89	3.09
WRD19	Gilbert	0.87	1.01	0.90	1.13	0.88	1.02
WRD19	Flinders	0.84	0.97	0.87	1.05	0.86	0.95
WRD19	Rgn 2-6	0.83	0.97	0.89	1.22	0.89	1.39

Table A40. Summary table of MICE ensemble showing minimum and mean catch, commercially available biomass (Bcom) and spawning biomass (Bsp) for mud crabs predicted under 19 water resource development scenarios (WRD1-WRD19) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.

MICE Ensemble – Mud crabs							
WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)
BaseCase	Mitchell	1.00	1.00	1.00	1.00	1.00	1.00
BaseCase	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
BaseCase	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
BaseCase	Rgn 2-6	1.00	1.00	1.00	1.00	1.00	1.00
WRD1	Mitchell	0.94	1.00	0.95	1.00	0.96	1.00
WRD1	Gilbert	0.24	0.54	0.26	0.55	0.25	0.53
WRD1	Flinders	0.17	0.56	0.19	0.57	0.22	0.56
WRD1	Rgn 2-6	0.44	0.71	0.56	0.74	0.56	0.73
WRD2	Mitchell	0.99	1.02	0.99	1.02	1.00	1.02
WRD2	Gilbert	0.33	0.64	0.34	0.65	0.34	0.64
WRD2	Flinders	0.17	0.54	0.19	0.55	0.22	0.55
WRD2	Rgn 2-6	0.51	0.77	0.69	0.82	0.66	0.81
WRD3	Mitchell	0.95	1.00	0.96	1.00	0.97	1.00
WRD3	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD3	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD3	Rgn 2-6	0.98	1.00	0.99	1.00	0.99	1.00
WRD4	Mitchell	1.00	1.00	1.00	1.00	1.00	1.00
WRD4	Gilbert	0.33	0.64	0.34	0.65	0.34	0.64
WRD4	Flinders	0.17	0.54	0.19	0.55	0.22	0.55
WRD4	Rgn 2-6	0.50	0.76	0.69	0.82	0.66	0.80
WRD5	Mitchell	0.94	0.97	0.94	0.97	0.94	0.97
WRD5	Gilbert	0.24	0.54	0.26	0.55	0.25	0.53
WRD5	Flinders	0.17	0.56	0.19	0.57	0.22	0.56
WRD5	Rgn 2-6	0.44	0.71	0.65	0.78	0.62	0.76
WRD6	Mitchell	0.95	1.00	0.95	1.00	0.94	1.00
WRD6	Gilbert	0.24	0.54	0.26	0.55	0.25	0.53
WRD6	Flinders	0.17	0.56	0.19	0.57	0.22	0.56
WRD6	Rgn 2-6	0.44	0.72	0.65	0.79	0.63	0.77
WRD7	Mitchell	1.00	1.00	1.00	1.00	1.00	1.00
WRD7	Gilbert	0.24	0.54	0.26	0.55	0.25	0.53
WRD7	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD7	Rgn 2-6	0.53	0.79	0.70	0.81	0.69	0.80
WRD8	Mitchell	0.96	0.98	0.96	0.98	0.96	0.98
WRD8	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD8	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD8	Rgn 2-6	0.99	1.00	0.99	1.00	0.99	1.00
WRD9	Mitchell	0.99	1.00	0.99	1.00	0.99	1.00
WRD9	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00

WRD9	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD9	Rgn 2-6	1.00	1.00	1.00	1.00	1.00	1.00
WRD10	Mitchell	0.98	1.01	0.98	1.01	0.99	1.01
WRD10	Gilbert	0.33	0.64	0.34	0.65	0.34	0.64
WRD10	Flinders	0.17	0.54	0.19	0.55	0.22	0.55
WRD10	Rgn 2-6	0.50	0.76	0.69	0.82	0.66	0.81
WRD11	Mitchell	0.95	0.99	0.96	0.99	0.96	1.00
WRD11	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD11	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD11	Rgn 2-6	0.98	1.00	0.99	1.00	0.99	1.00
WRD12	Mitchell	0.98	1.02	0.99	1.02	1.00	1.02
WRD12	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD12	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD12	Rgn 2-6	0.99	1.00	1.00	1.00	1.00	1.00
WRD13	Mitchell	0.98	1.01	0.98	1.01	0.99	1.01
WRD13	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD13	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD13	Rgn 2-6	0.99	1.00	1.00	1.00	1.00	1.00
WRD14	Mitchell	0.99	1.02	0.99	1.02	1.00	1.02
WRD14	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD14	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD14	Rgn 2-6	1.00	1.00	1.00	1.00	1.00	1.00
WRD15	Mitchell	0.99	1.01	0.99	1.01	1.00	1.02
WRD15	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD15	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD15	Rgn 2-6	0.99	1.00	1.00	1.00	1.00	1.00
WRD16	Mitchell	0.95	0.97	0.95	0.97	0.94	0.97
WRD16	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD16	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD16	Rgn 2-6	0.98	0.99	0.99	1.00	0.98	0.99
WRD17	Mitchell	1.00	1.00	1.00	1.00	1.00	1.00
WRD17	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD17	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD17	Rgn 2-6	1.00	1.00	1.00	1.00	1.00	1.00
WRD18	Mitchell	0.94	1.00	0.95	1.00	0.96	1.00
WRD18	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD18	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD18	Rgn 2-6	0.98	1.00	0.99	1.00	0.99	1.00
WRD19	Mitchell	0.99	1.02	0.99	1.02	1.00	1.02
WRD19	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD19	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD19	Rgn 2-6	1.00	1.00	1.00	1.00	1.00	1.01

Table A41. Summary table of MICE ensemble showing minimum and mean number of individuals, and number of mature individuals (Bsp) for largemouth sawfish predicted under 19 water resource development scenarios (WRD1-WRD19) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.

MICE Ensemble – Largemouth sawfish					
WRD	Rgn.Name	Rel.N.Min	Rel.N.Mean	Rel.Bsp.Min	Rel.Bsp.Mean
BaseCase	Mitchell	1.00	1.00	1.00	1.00
BaseCase	Gilbert	1.00	1.00	1.00	1.00
BaseCase	Flinders	1.00	1.00	1.00	1.00
BaseCase	Rgn 2-6	1.00	1.00	1.00	1.00
WRD1	Mitchell	0.22	0.36	0.24	0.36
WRD1	Gilbert	0.31	0.46	0.37	0.50
WRD1	Flinders	0.01	0.07	0.01	0.07
WRD1	Rgn 2-6	0.26	0.36	0.25	0.37
WRD2	Mitchell	0.29	0.43	0.29	0.42
WRD2	Gilbert	0.32	0.46	0.37	0.50
WRD2	Flinders	0.01	0.07	0.01	0.07
WRD2	Rgn 2-6	0.27	0.39	0.26	0.39
WRD3	Mitchell	0.26	0.40	0.26	0.39
WRD3	Gilbert	1.00	1.00	1.00	1.00
WRD3	Flinders	1.00	1.00	1.00	1.00
WRD3	Rgn 2-6	0.75	0.85	0.82	0.89
WRD4	Mitchell	1.00	1.00	1.00	1.00
WRD4	Gilbert	0.32	0.46	0.37	0.50
WRD4	Flinders	0.01	0.07	0.01	0.07
WRD4	Rgn 2-6	0.37	0.52	0.31	0.49
WRD5	Mitchell	0.36	0.52	0.33	0.46
WRD5	Gilbert	0.35	0.57	0.37	0.50
WRD5	Flinders	0.01	0.08	0.01	0.07
WRD5	Rgn 2-6	0.34	0.46	0.26	0.39
WRD6	Mitchell	0.12	0.29	0.09	0.27
WRD6	Gilbert	0.34	0.52	0.37	0.50
WRD6	Flinders	0.01	0.07	0.01	0.07
WRD6	Rgn 2-6	0.23	0.37	0.24	0.34
WRD7	Mitchell	0.99	1.06	0.95	0.98
WRD7	Gilbert	0.34	0.52	0.37	0.50
WRD7	Flinders	0.96	1.02	0.98	0.99
WRD7	Rgn 2-6	0.84	0.95	0.81	0.89
WRD8	Mitchell	0.85	1.04	0.94	0.97
WRD8	Gilbert	0.98	1.09	0.92	0.97
WRD8	Flinders	0.96	1.02	0.98	0.99
WRD8	Rgn 2-6	0.98	1.06	0.97	0.99
WRD9	Mitchell	0.84	1.04	0.93	0.97
WRD9	Gilbert	0.98	1.09	0.92	0.97
WRD9	Flinders	0.96	1.02	0.98	0.99

WRD9	Rgn 2-6	0.98	1.06	0.97	0.98
WRD10	Mitchell	0.31	0.47	0.28	0.42
WRD10	Gilbert	0.34	0.52	0.37	0.50
WRD10	Flinders	0.01	0.08	0.01	0.07
WRD10	Rgn 2-6	0.32	0.43	0.26	0.38
WRD11	Mitchell	0.28	0.44	0.25	0.39
WRD11	Gilbert	0.98	1.09	0.92	0.97
WRD11	Flinders	0.96	1.02	0.98	0.99
WRD11	Rgn 2-6	0.75	0.91	0.79	0.88
WRD12	Mitchell	0.31	0.47	0.28	0.42
WRD12	Gilbert	0.98	1.09	0.92	0.97
WRD12	Flinders	0.96	1.02	0.98	0.99
WRD12	Rgn 2-6	0.77	0.92	0.80	0.89
WRD13	Mitchell	0.31	0.47	0.28	0.42
WRD13	Gilbert	0.98	1.09	0.92	0.97
WRD13	Flinders	0.96	1.02	0.98	0.99
WRD13	Rgn 2-6	0.77	0.92	0.81	0.89
WRD14	Mitchell	0.31	0.47	0.28	0.42
WRD14	Gilbert	0.98	1.09	0.92	0.97
WRD14	Flinders	0.96	1.02	0.98	0.99
WRD14	Rgn 2-6	0.77	0.92	0.80	0.89
WRD15	Mitchell	0.31	0.47	0.28	0.42
WRD15	Gilbert	0.98	1.09	0.92	0.97
WRD15	Flinders	0.96	1.02	0.98	0.99
WRD15	Rgn 2-6	0.77	0.92	0.81	0.89
WRD16	Mitchell	0.58	0.74	0.54	0.67
WRD16	Gilbert	0.98	1.09	0.92	0.97
WRD16	Flinders	0.96	1.02	0.98	0.99
WRD16	Rgn 2-6	0.87	0.98	0.87	0.93
WRD17	Mitchell	0.99	1.06	0.95	0.98
WRD17	Gilbert	0.98	1.09	0.92	0.97
WRD17	Flinders	0.96	1.02	0.98	0.99
WRD17	Rgn 2-6	1.01	1.07	0.97	0.99
WRD18	Mitchell	0.23	0.40	0.23	0.35
WRD18	Gilbert	0.98	1.09	0.92	0.97
WRD18	Flinders	0.96	1.02	0.98	0.99
WRD18	Rgn 2-6	0.73	0.89	0.78	0.87
WRD19	Mitchell	0.25	0.43	0.25	0.38
WRD19	Gilbert	0.98	1.09	0.92	0.97
WRD19	Flinders	0.96	1.02	0.98	0.99
WRD19	Rgn 2-6	0.74	0.90	0.78	0.88

Appendix 20 – MICE WRD Additional Sensitivities: prawn fishing power, barramundi selectivity, sawfish model structure and mangrove growth assumptions

Below we summarise some additional MICE results from additional sensitivity tests (Model 7) considered important. These include sensitivity to prawn fishing power, barramundi selectivity, sawfish model structure and mangrove growth assumptions. These additional tests focused in particular on selected sub-components of the MICE, with results thus presented here for this subset only.

Methods for Model 7 sensitivity tests

1. Common banana prawn additional sensitivity: replacing the fishing power trend used as an input for common banana prawns with an alternative series derived by Zhou et al. (2015) and Hutton et al. (2022) as shown in Figure A145 below. The sensitivity was compared with Ensemble Model version 5.

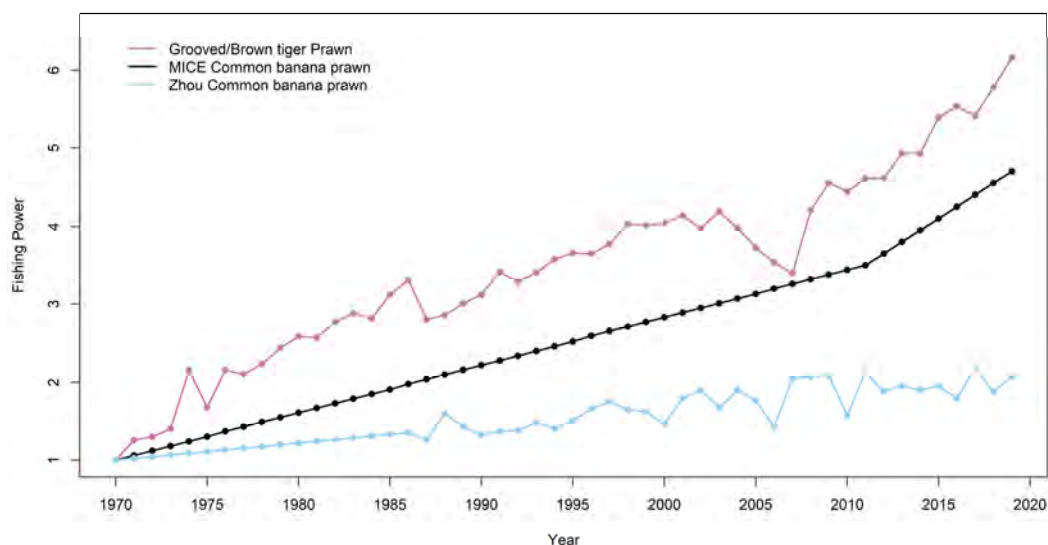


Figure A145. Comparison of MICE fishing power trend assumed for common banana prawns compared with alternative series derived by Zhou et al. (2015).

2. Barramundi additional sensitivity: The MICE estimates selectivity curves for the northern and southern GoC barramundi stocks for the recent period as well as an earlier period. Based on these estimated curves and the lack of a maximum size class for barramundi in the NT, a separate selectivity curve is assumed for the NT, but that has less relevance for the current project which is focused more on WRDs applied to Qld. Campbell et al. (2017) estimated selectivity curves for the Qld southern GoC barramundi stock assessment. As per Figure A146, these differ considerably from those estimated in the MICE versions. This is not surprising given population models can be very sensitive to assumptions regarding selectivity and the models here (MICE barramundi sub-component vs model in Campbell et al. (2017)) have very different model structures – and in particular the use of an age-dependent M in the MICE versus age-independent M in Campbell et al. (2017). To assist in making the model runs as comparable as possible, the sensitivity run also used the same von Bertalanffy growth parameters as Campbell et al. (2017) based on length-at-age observations for Southern Gulf of Carpentaria barramundi. Length-at-

age observed from LTMP data (all years) for Southern Gulf of Carpentaria barramundi, with $L_{\infty} = 150$ cm, $a_0 = -0.5$ and $\kappa = 0.16$ (standard). The sensitivity was compared with Ensemble Model version 5 with base M fixed as that is most similar (but still very different) to the constant $M=0.2$ used by Campbell et al. (2017), and MICE model 5 does not apply a sex ratio modified to the stock-recruit relationship.

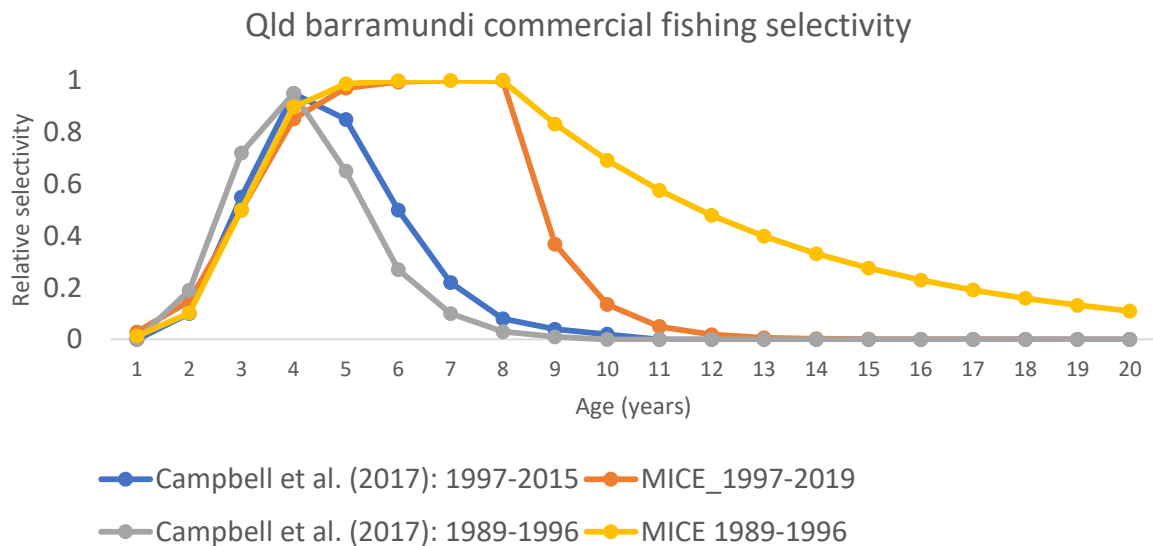


Figure A146. Comparison of MICE-estimated commercial fishing selectivity for barramundi (Qld southern Gulf stock) for past (1989-1996) and recent (1997-current) periods compared with selectivity functions used in the stock assessment by Campbell et al. (2017)

3. Sawfish additional sensitivity: Largetooth sawfish are predicted to be very sensitive to WRDs across all of the ensemble model versions. Flow is assumed to influence their population dynamics based on available information in the literature, with full equations shown in Appendix 16. Here we tested what the influence on impacts from WRDs would be if we removed the assumed relationship between flows and recruitment success, and only included boom and bust effects on recruitment in very good and poor years respectively. The sensitivity was compared with Ensemble Model version 5.
4. Mangroves additional sensitivity: the MICE assumed that changes in air temperature can influence the growth rates of mangroves. This interaction is uncertain and may operate as more of an indirect link correlated with other variables such as El Niño (C. Lovelock, pers commn). We therefore ran an additional sensitivity test which excluded the air temperature modifier function (see Appendix 16 for equations) from the mangrove growth function. The sensitivity was compared with Ensemble Model version 5.

Results for Model 7 sensitivity tests

1. Common banana prawns: the Model 7 sensitivity with the alternative fishing power input series converged successfully and parameter estimates are shown in Table A42. The total likelihood contribution from the common banana prawn component improves slightly when using the alternative fishing power compared with Model version 5, with the likelihood contributions improving most for the fit to the Embley, Mitchell and Flinders regions (Table A42). However

solely relying on likelihood scores when selecting amongst more complex ecosystem models – or deciding which model structures should be included in an ensemble – isn't necessarily always wise. For example, in this instance as evident from Figure A147, the much lower fishing power trend leads to the MICE under-estimating recent catches, particularly for the Gilbert and Flinders regions. As these regions are focus regions in the current study, the alternative fishing power series was therefore not included in the ensemble but future work could focus on better reconciling these approaches. This decision was also based to some extent on the MICE common banana prawn fishing power trend being more similar to the much higher tiger prawn fishing power trends (based on detailed information and analyses).

The results of testing the influence of alternative WRDs using Model version 7 sensitivity (compared with Ensemble version 5) are shown in Table A43. Overall the results are not too different although under most scenarios, Model version 7 predicts less severe declines by up to about 4%. Although slightly more optimistic, the results from this sensitivity scenario are broadly consistent with the ensemble and risk assessment results.

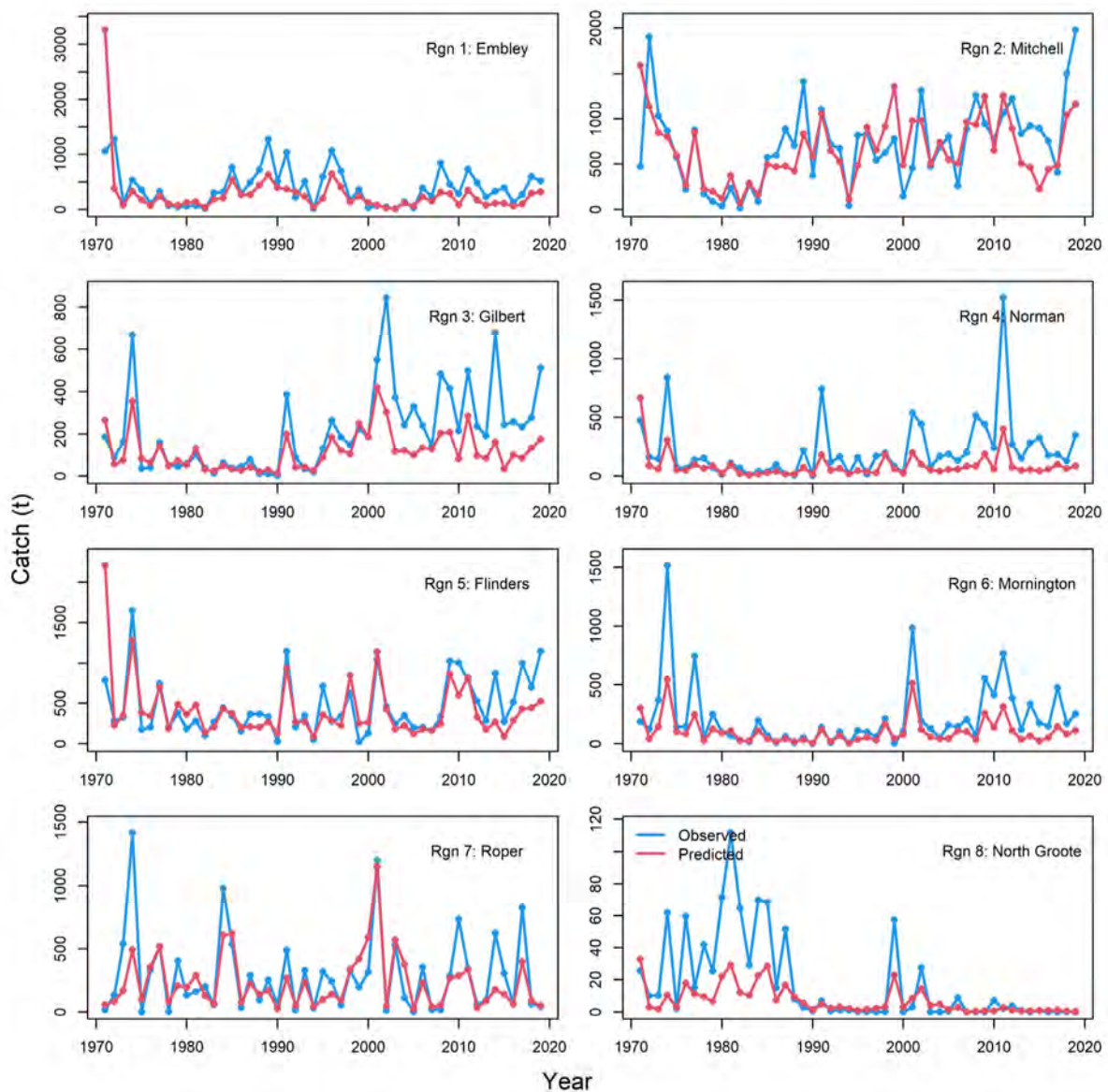


Figure A147. Comparison of the observed and model-predicted annual common banana prawn catch (tonnes) estimated using Model version 7 (driven by baseline flows, using alternative fishing power input series) and for each of the eight model regions as shown.

2. Barramundi: the Model 7 sensitivity with the alternative fixed selectivity input series converged successfully and parameter estimates are shown in Table A42 below. The total likelihood contribution from the barramundi component was significantly worse when using the alternative selectivity settings compared with Model version 5, including the likelihood contributions based on fitting to catch-at-age data for the southern and northern Gulf stocks (Table A42). As shown in Figure A148, the model fit to the Gilbert region and arguably the Norman regions did seem to improve slightly, but at the expense of the fit to the Flinders region. Model 7 did not result in an improved fit to the southern Gulf catch-at-age data (Figure A149) and also not to the northern Gulf catch-at-age data when assuming the Campbell et al. (2017) selectivity curve applies also to the northern stock (Figure A150). The fits to the NT regions 7 and 8 are not relevant here as the Campbell et al. (2017) selectivity curves apply only to the Qld stocks. Model 7 was also not a preferred model formulation because it is not good practice or internally consistent to use the outputs of a stock assessment model as inputs to another model with different structure if this can be avoided.

Nonetheless, given the different barramundi population dynamics estimated using Model version 7, we evaluated what the influence of the four key WRDs would be when using this model version. The results are shown in Table A44. In general, results for the Mitchell River aren't too different but Model version 7 estimates that on average barramundi catches from the Gilbert (84%) and Flinders River (95%) regions will be less heavily reduced under WRD1 than is the case when using Model version 5 which estimates reductions of 62% and 86% respectively (Table A44). These differences are similar across the different WRDs and categories as shown in Table A44. Full results are stored electronically and available on request. Given the sensitivity of model results to alternative selectivity assumptions, this aspect merits further investigation in future work.

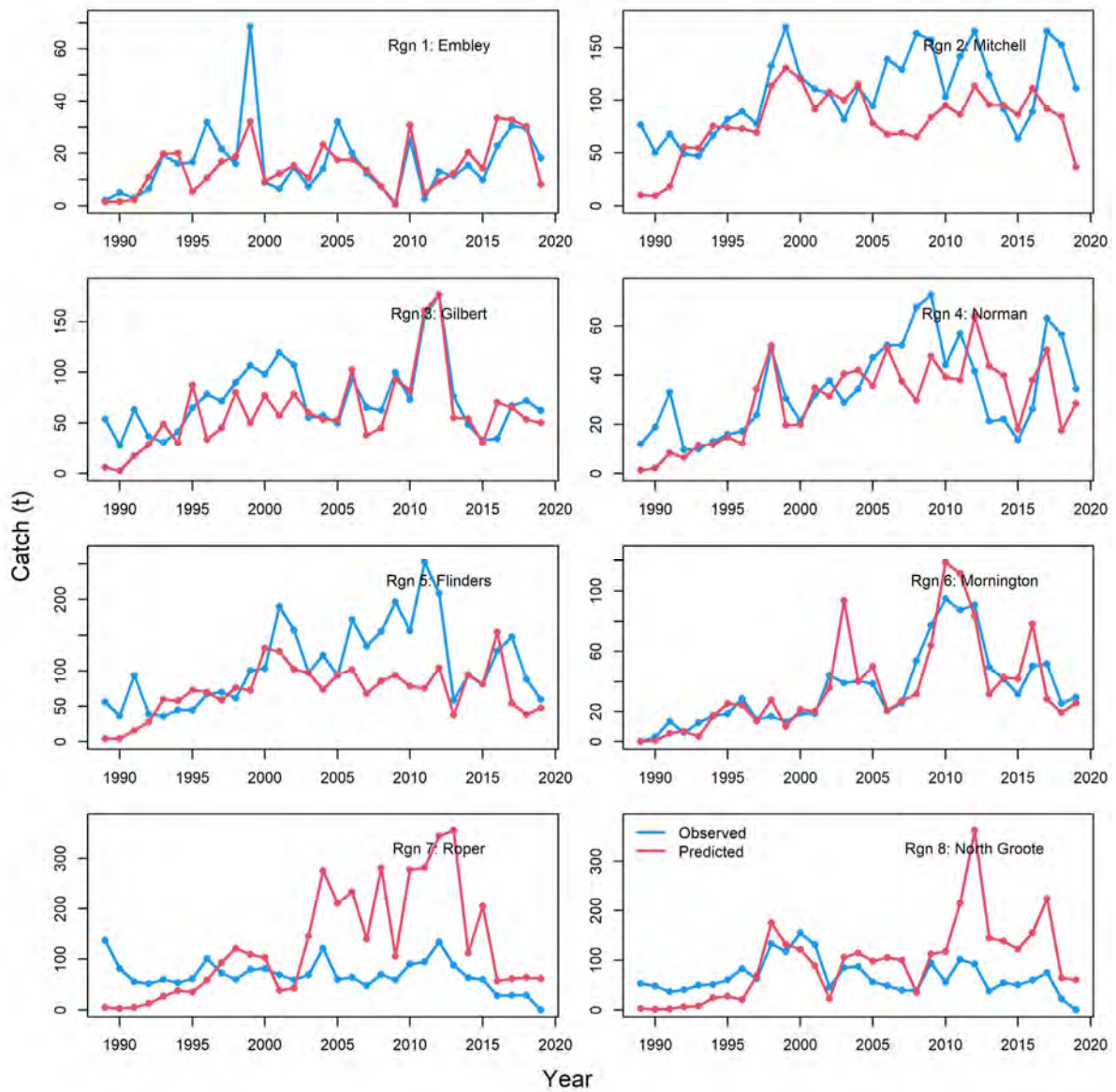


Figure A148. Comparison of the observed and model-predicted annual barramundi catch (tonnes) estimated using Model version 7 (baseline flows, alternative fixed selectivity input series) and for each of the eight model regions as shown.

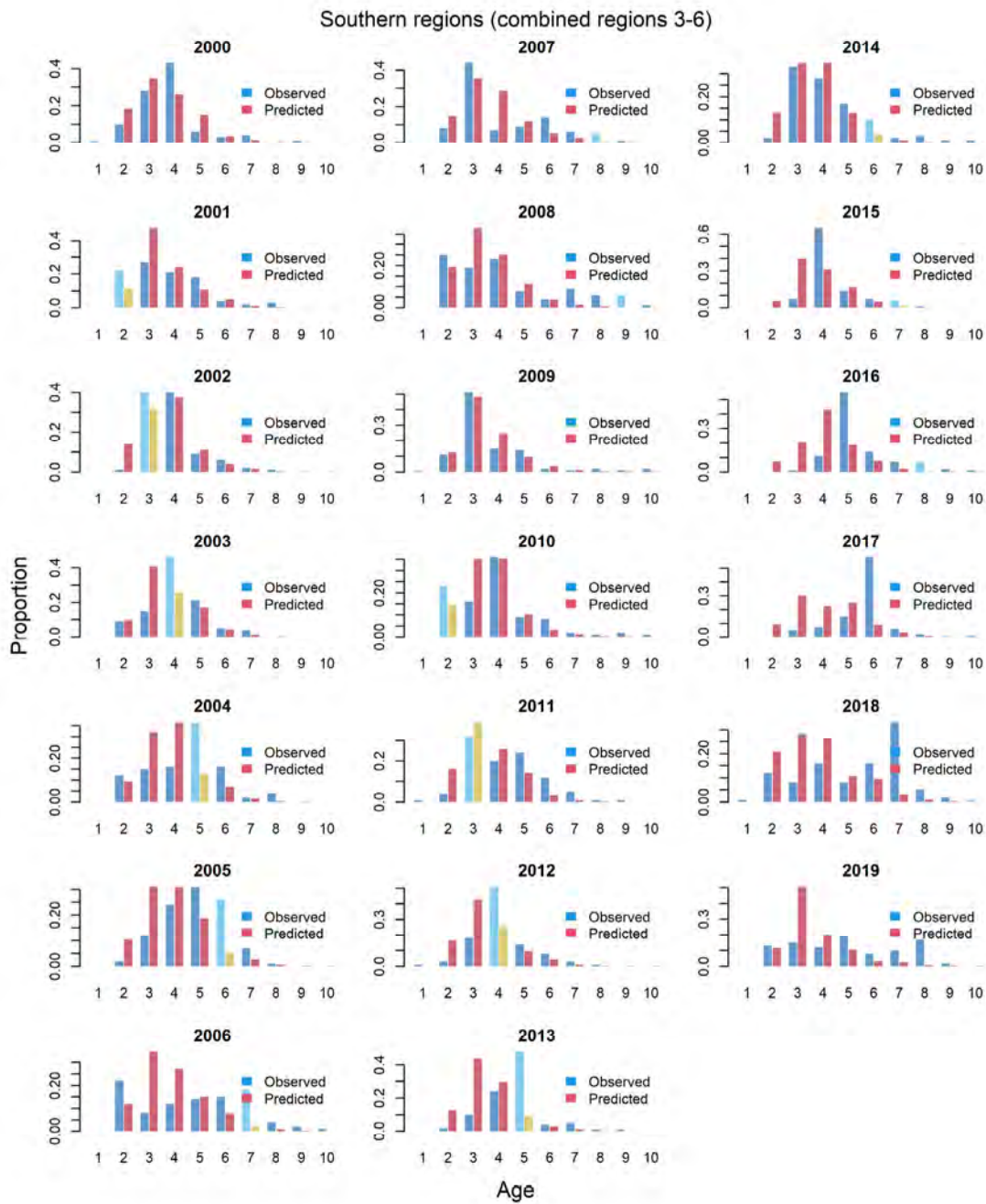


Figure A149. Comparison of the observed and model-predicted commercial catch-at-age proportions of barramundi in each age-class as shown estimated using Model version 7 (baseline flows, alternative fixed selectivity input series) for regions 3-6 from 2000 to 2019.

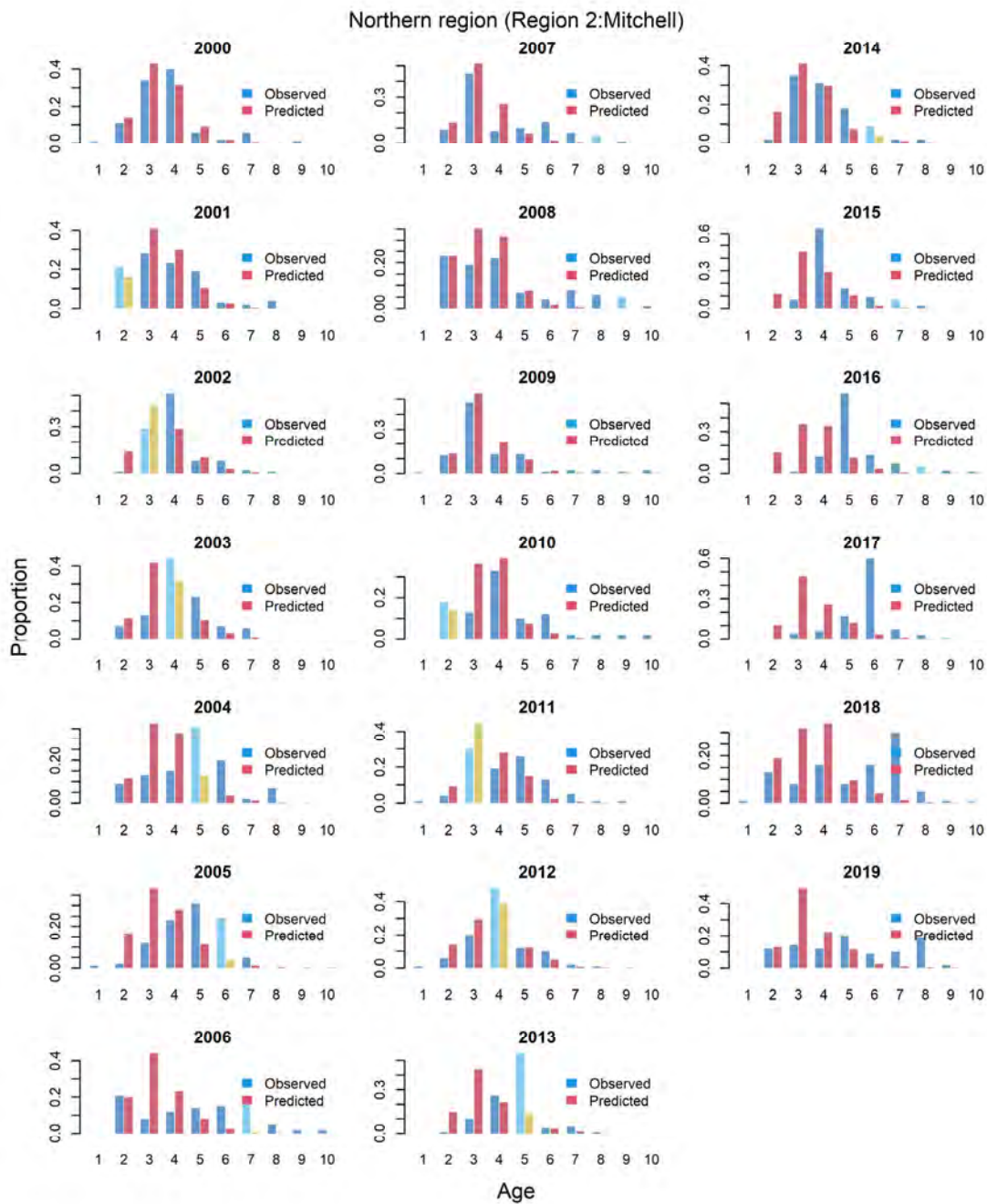


Figure A150. Comparison of the observed and model-predicted commercial catch-at-age proportions of barramundi in each age-class as shown estimated using Model version 7 (baseline flows, alternative fixed selectivity input series) for the Mitchell River catchment region from 2000 to 2019.

3. Sawfish: Model version 7 highlighted the sensitivity of the model to assumptions related to how flows influence largemouth sawfish population dynamics. When assuming that river flows do not influence the recruitment success (i.e. numbers of 1-year old largemouth sawfish that recruit to the population) the model estimated that over the recent period sawfish in the Gilbert River region would be less depleted than estimated under Model 5 (which linked flows and recruitment

success based on the best available scientific understanding). This difference was less in other regions such as the Flinders River region (Figure A151). When using Model version 7 for evaluating the impact of alternative WRDs, the model still estimated substantial declines in largemouth sawfish (Table A45). The declines were less than those predicted using Model version 5 (Table A45). However relative to the Model version 7 base case (no-WRD scenario), the MICE still predicted that declines in sawfish could be as much as 60% or more under WRD1, 57% or more under WRD2, 59% or more in the Mitchell region under WRD3 and 65% or more under WRD4 (except for Mitchell region). These estimated reductions, although less than estimated under Model version 5, would still result in classification (using our system as defined in Table 8) of risks to sawfish as being in the severe to intolerable range. These results therefore support the robustness of the MICE results that largemouth sawfish may be particularly sensitive to WRDs due to their life history characteristics, as these results hold even when substantially changing model assumptions.

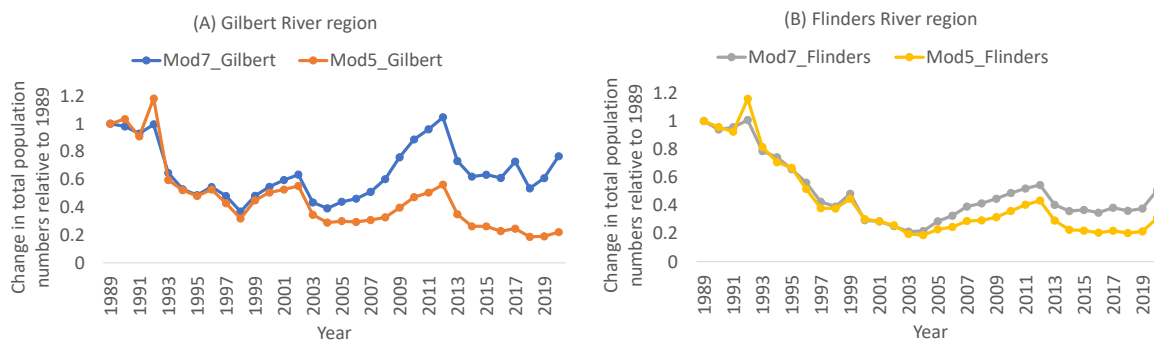


Figure A151. Example of differences in the relative depletion (relative to 1989 level) of largemouth sawfish in the (A) Gilbert River region and (B) Flinders River region under different structural assumptions used in Ensemble Model version 5 and sensitivity Model version 7 which assumes that recruitment success does not depend on river flows except in exceptional years.

4. Mangroves: When comparing Model version 5 (with air temperature (AT) assumed to influence mangrove growth rate) and Model version 7 (no influence of AT on mangrove growth rate) there were some small but noticeable differences in the trend and magnitude of the estimated trajectories over time (Figure A152). Model 7 consistently estimated a slightly higher mangrove biomass than Model 5 which adjusted growth based on changes in AT. This is presumably due to the cumulative effect of AT combined with other driving variables. When evaluating the influence of WRDs using Model version 7, the estimated declines due to WRDs were less than those predicted using Model version 5 (Table A46). For example, under WRD1, Model version 7 estimated an average decline of 11% for Regions 2 to 6 compared with 19% predicted under Model version 5. Comparable declines under WRD2 were 4% compared with 6% for Model versions 7 and 5 respectively. However Model 5 estimates more variability in mangrove responses – for example the maximum declines are considerably greater under some scenarios using Model 5. Overall these results suggest that the MICE shows some sensitivity to the assumption of AT influencing growth such that refinement of the mangrove model component will assist in better quantifying likely WRD impacts. The extent of predicted average declines in mangroves did not however change substantially under this sensitivity test.

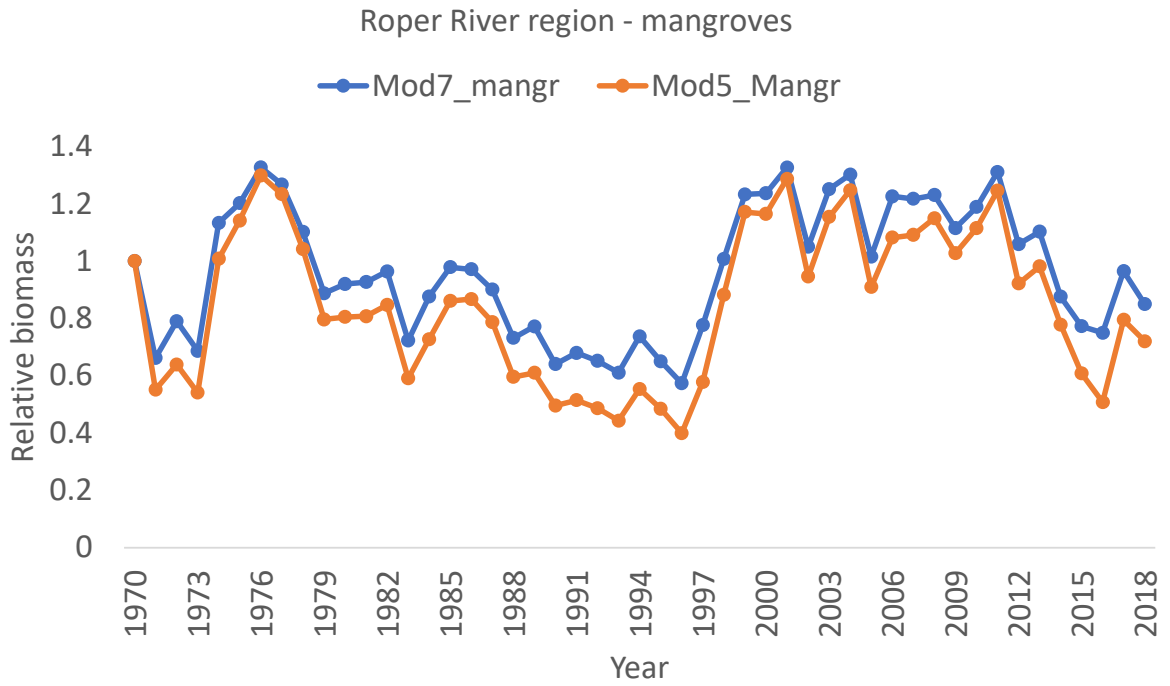


Figure A152. Example of modelled difference in relative changes (since 1970) in mangrove biomass in the Roper River region when comparing Model versions 5 and 7.

Table A42. Additional sensitivity analysis MICE model parameter estimates, associated Hessian-based standard deviations (std), number of parameters estimated, negative log likelihood (-lnL) contributions and Akaike Information Criterion (AIC) scores for Model 5 of the Ensemble compared with additional sensitivity results from Model 7 for (1) common banana prawn, (2) barramundi. Parameter estimates in italics are fixed parameters. Bsp = spawning biomass (t); CAA = catch-at-age; SOI = southern oscillation index; Mitch = Mitchell River; Gilb = Gilbert River; Norm = Norman River; Flind = Flinders River.

Parameter	Description	<u>Model 5 -link productivity</u>		<u>Model 7 – additional sensitivity</u>	
		value	std	value	std
<i>(1) Common banana prawn</i>					
Kstart	Initialisation ln(Bsp)	5.400	0.026	<i>5.400</i>	-
Thres (1)	Flow threshold parameter	<i>0.561</i>	-	<i>0.561</i>	-
Thres (2)	Flow threshold parameter	<i>0.379</i>	-	<i>0.379</i>	-
Thres (3)	Flow threshold parameter	<i>0.259</i>	-	<i>0.259</i>	-
Thres (4)	Flow threshold parameter	<i>0.615</i>	-	<i>0.615</i>	-
Thres (5-6)	Flow threshold parameter	<i>0.522</i>	-	<i>0.522</i>	-
Thres (7-8)	Flow threshold parameter	<i>0.481</i>	-	<i>0.481</i>	-
Mitch1	River connection parameter	0.051	0.016	0.049	0.009
Mitch2	River connection parameter	0.229	0.028	0.239	0.033
Mitch3	River connection parameter	0.007	0.013	0.008	0.014
Mitch4	River connection parameter	0.055	0.014	0.060	0.013
Gilb1	River connection parameter	0.041	0.009	0.038	0.010
Gilb2	River connection parameter	0.001	0.004	0.001	0.004
Gilb3	River connection parameter	0.003	0.006	0.003	0.011
Gilb4	River connection parameter	0.081	0.016	0.079	0.017
Norm1	River connection parameter	0.068	0.015	0.065	0.011
Norm2	River connection parameter	0.114	0.037	0.118	0.034
Norm3	River connection parameter	0.228	0.074	0.243	0.039
Norm4	River connection parameter	0.043	0.010	0.046	0.013
Flind1	River connection parameter	0.033	0.009	0.030	0.006
Flind2	River connection parameter	0.012	0.009	0.013	0.012

Parameter	Description	<u>Model 5 -link productivity</u>		<u>Model 7 – additional sensitivity</u>	
		value	std	value	std
Flind3	River connection parameter	0.053	0.021	0.056	0.032
Flind4	River connection parameter	0.120	0.016	0.122	0.019
Embley	Flow parameter	0.166	0.057	0.149	0.019
Roper	Flow parameter	0.223	0.040	0.226	0.011
$thresM^{psp}$	Flow threshold parameter	0.8	–	0.8	–
No. parameters		19		19	
Likelihood contributions		value	sigma	value	sigma
-lnL: Catch (Region 1)		494.6	1.7	476.9	1.6
-lnL: Catch (Region 2)		397.7	1.4	386.1	1.3
-lnL: Catch (Region 3)		355.8	1.3	348.4	1.2
-lnL: Catch (Region 4)		371.0	1.3	367.4	1.3
-lnL: Catch (Region 5)		488.4	1.6	472.2	1.6
-lnL: Catch (Region 6)		373.0	1.3	369.6	1.3
-lnL: Catch (Region 7)		399.7	1.4	383.3	1.3
-lnL: Catch (Region 8)		233.7	1.0	235.1	1.0
-lnL: overall		3114.0	–	3038.9	–
AIC		6266.0	–	6115.9	–
<u>Other fixed parameters</u>					
M_{base}^{psp}	Natural mortality base (wk)	0.045	–	0.045	–
(2)					
<i>Barramundi</i>					
M_{base}^{barra}	Natural mortality base	8.00E-03	2.57E-08	8.00E-03	-
q_r^{barra} (1)	Catchability	5.40E-05	1.00E-05	3.5E-04	5.0E-05
q_r^{barra} (2)	Catchability	4.40E-05	4.10E-06	7.3E-05	8.8E-06
q_r^{barra} (3-6)	Catchability	8.05E-05	5.45E-06	1.4E-04	8.7E-06
q_r^{barra} (7-8)	Catchability	3.28E-04	4.81E-06	8.8E-05	2.0E-05
$thres^{barra}$ (1)	Flow threshold parameter	0.561	–	0.572	0.137
$thres^{barra}$ (2)	Flow threshold parameter	0.379	–	0.376	0.249
$thres^{barra}$ (3)	Flow threshold parameter	0.259	–	0.284	0.036

Parameter	Description	<u>Model 5 -link productivity</u>		<u>Model 7 – additional sensitivity</u>	
		value	std	value	std
<i>thres</i> ^{barra} (4)	Flow threshold parameter	0.615	–	0.595	0.322
<i>thres</i> ^{barra} (5-6)	Flow threshold parameter	0.522	–	0.553	0.070
<i>thres</i> ^{barra} (7-8)	Flow threshold parameter	0.481	–	0.489	0.064
del_barra_S	Age-selectivity par (Southern)	0.455	0.029	-	-
sfa_barra_S	Age-selectivity par (Southern)	0.426	0.215	-	-
del_barra_N	Age-selectivity par (Northern)	1.000	0.000	-	-
sfa_barra_N	Age-selectivity par (Northern)	1.000	0.000	-	-
<i>thresM</i> ^{barra}	Flow threshold parameter	0.80	–	0.80	–
No. parameters		9		11	
Likelihood contributions		value	sigma	value	sigma
-lnL: Catch (Region 1)		96.1	0.9	112.4	1.0
-lnL: Catch (Region 2)		127.7	1.1	151.4	1.2
-lnL: Catch (Region 3)		126.2	1.1	160.1	1.3
-lnL: Catch (Region 4)		141.3	1.2	160.9	1.3
-lnL: Catch (Region 5)		114.6	1.0	157.3	1.3
-lnL: Catch (Region 6)		151.0	1.2	135.5	1.1
-lnL: Catch (Region 7)		123.2	1.1	208.2	1.6
-lnL: Catch (Region 8)		138.6	1.1	214.2	1.6
-lnL: CAA (Northern)		56.4	0.2	121.9	0.4
-lnL: CAA (Southern)		38.7	0.2	90.2	0.3
-lnL: overall		1113.8	–	1512.1	–
AIC		2245.6	–	3046.2	–
<u>Other fixed parameters</u>					
Sex ratio		Not applied		Not applied	

Table A43. Summary table of Model sensitivity version 7 compared with Ensemble Model version 5 showing minimum and mean catch, commercially available biomass (Bcom) and spawning biomass (Bsp) for common banana prawns predicted under four water resource development scenarios (WRD1-WRD4)

relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.

Model version 7 sensitivity - common banana prawns							
WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)
BaseCase	Mitchell	1	1	1	1	1	1
BaseCase	Gilbert	1	1	1	1	1	1
BaseCase	Flinders	1	1	1	1	1	1
BaseCase	Rgn 2-6	1	1	1	1	1	1
WRD1	Mitchell	0.63	0.78	0.62	0.79	0.63	0.78
WRD1	Gilbert	0.65	0.80	0.64	0.81	0.65	0.80
WRD1	Flinders	0.58	0.74	0.57	0.76	0.58	0.73
WRD1	Rgn 2-6	0.63	0.78	0.62	0.79	0.63	0.78
WRD2	Mitchell	0.73	0.85	0.74	0.86	0.75	0.85
WRD2	Gilbert	0.80	0.90	0.81	0.90	0.82	0.91
WRD2	Flinders	0.64	0.81	0.64	0.83	0.65	0.81
WRD2	Rgn 2-6	0.71	0.85	0.74	0.86	0.75	0.85
WRD3	Mitchell	0.93	0.97	0.94	0.98	0.94	0.97
WRD3	Gilbert	0.87	0.93	0.89	0.94	0.89	0.94
WRD3	Flinders	0.95	0.98	0.95	0.99	0.96	0.98
WRD3	Rgn 2-6	0.93	0.97	0.94	0.98	0.94	0.97
WRD4	Mitchell	0.74	0.86	0.75	0.87	0.76	0.86
WRD4	Gilbert	0.83	0.92	0.82	0.92	0.83	0.93
WRD4	Flinders	0.64	0.82	0.64	0.84	0.66	0.81
WRD4	Rgn 2-6	0.71	0.86	0.74	0.87	0.75	0.86
Model version 5 - common banana prawns							
WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)
WRD1	Mitchell	0.61	0.75	0.59	0.76	0.59	0.73
WRD1	Gilbert	0.63	0.77	0.61	0.77	0.61	0.76
WRD1	Flinders	0.54	0.72	0.54	0.73	0.55	0.69
WRD1	Rgn 2-6	0.61	0.74	0.6	0.76	0.59	0.73
WRD2	Mitchell	0.71	0.83	0.72	0.85	0.72	0.83
WRD2	Gilbert	0.79	0.88	0.8	0.88	0.8	0.88
WRD2	Flinders	0.6	0.79	0.61	0.81	0.63	0.78
WRD2	Rgn 2-6	0.68	0.83	0.72	0.84	0.73	0.83
WRD3	Mitchell	0.94	0.97	0.93	0.98	0.93	0.97
WRD3	Gilbert	0.87	0.93	0.88	0.94	0.89	0.93
WRD3	Flinders	0.95	0.98	0.95	0.99	0.95	0.98
WRD3	Rgn 2-6	0.94	0.97	0.94	0.98	0.94	0.97
WRD4	Mitchell	0.71	0.84	0.73	0.85	0.73	0.83
WRD4	Gilbert	0.82	0.9	0.8	0.9	0.81	0.9
WRD4	Flinders	0.6	0.8	0.62	0.82	0.63	0.79
WRD4	Rgn 2-6	0.68	0.84	0.73	0.85	0.73	0.83

Table A44. Summary table of Model sensitivity version 7 compared with Ensemble Model version 5 showing minimum and mean catch, commercially available biomass (Bcom) and spawning biomass (Bsp)

for barramundi predicted under water resource development scenarios (WRD1-WRD4) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions2-6 combined.

Model version 7 - sensitivity - barramundi							
WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)
BaseCase	Mitchell	1	1	1	1	1	1
BaseCase	Gilbert	1	1	1	1	1	1
BaseCase	Flinders	1	1	1	1	1	1
BaseCase	Rgn 2-6	1	1	1	1	1	1
WRD1	Mitchell	0.77	0.84	0.77	0.84	0.77	0.86
WRD1	Gilbert	0.65	0.84	0.82	0.93	0.93	0.97
WRD1	Flinders	0.84	0.95	0.93	0.97	0.96	0.98
WRD1	Rgn 2-6	0.8	0.9	0.87	0.92	0.91	0.95
WRD2	Mitchell	0.94	0.97	0.94	0.97	0.94	0.97
WRD2	Gilbert	0.68	0.86	0.83	0.94	0.94	0.97
WRD2	Flinders	0.84	0.95	0.93	0.97	0.96	0.98
WRD2	Rgn 2-6	0.89	0.95	0.95	0.98	0.98	0.99
WRD3	Mitchell	0.8	0.87	0.8	0.87	0.79	0.89
WRD3	Gilbert	1	1	1	1	1	1
WRD3	Flinders	1	1	1	1	1	1
WRD3	Rgn 2-6	0.93	0.96	0.94	0.96	0.95	0.97
WRD4	Mitchell	1	1	1	1	1	1
WRD4	Gilbert	0.68	0.86	0.83	0.94	0.94	0.97
WRD4	Flinders	0.84	0.95	0.93	0.97	0.96	0.98
WRD4	Rgn 2-6	0.89	0.96	0.97	0.99	0.99	0.99
Model version 5 - barramundi							
WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)
WRD1	Mitchell	0.78	0.87	0.75	0.83	0.75	0.83
WRD1	Gilbert	0.46	0.62	0.24	0.43	0.53	0.66
WRD1	Flinders	0.73	0.86	0.47	0.79	0.68	0.87
WRD1	Rgn 2-6	0.76	0.82	0.56	0.74	0.74	0.82
WRD2	Mitchell	0.96	0.98	0.94	0.97	0.94	0.97
WRD2	Gilbert	0.52	0.70	0.31	0.54	0.58	0.73
WRD2	Flinders	0.74	0.87	0.47	0.80	0.70	0.87
WRD2	Rgn 2-6	0.86	0.90	0.68	0.83	0.87	0.92
WRD3	Mitchell	0.86	0.90	0.81	0.87	0.81	0.87
WRD3	Gilbert	1.00	1.00	1.00	1.00	1.00	1.00
WRD3	Flinders	1.00	1.00	1.00	1.00	1.00	1.00
WRD3	Rgn 2-6	0.97	0.98	0.92	0.96	0.96	0.98
WRD4	Mitchell	1.00	1.00	1.00	1.00	1.00	1.00
WRD4	Gilbert	0.52	0.70	0.31	0.54	0.58	0.73
WRD4	Flinders	0.74	0.87	0.47	0.80	0.70	0.87
WRD4	Rgn 2-6	0.86	0.91	0.69	0.84	0.87	0.93

Table A45. Summary table of Model sensitivity version 7 compared with Ensemble Model version 5 showing minimum and mean number of individuals, and number of mature individuals (Bsp) for sawfish predicted under four water resource development scenarios (WRD1-WRD4) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.

Model version 7 – sensitivity - sawfish					
WRD Scenario	Rgn Name	Rel.N (Min)	Rel.N (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)
BaseCase	Mitchell	1	1	1	1
BaseCase	Gilbert	1	1	1	1
BaseCase	Flinders	1	1	1	1
BaseCase	Rgn 2-6	1	1	1	1
WRD1	Mitchell	0.30	0.39	0.33	0.39
WRD1	Gilbert	0.25	0.34	0.30	0.35
WRD1	Flinders	0.11	0.23	0.08	0.23
WRD1	Rgn 2-6	0.20	0.29	0.13	0.26
WRD2	Mitchell	0.35	0.43	0.39	0.43
WRD2	Gilbert	0.25	0.34	0.30	0.35
WRD2	Flinders	0.11	0.23	0.08	0.23
WRD2	Rgn 2-6	0.21	0.30	0.14	0.26
WRD3	Mitchell	0.33	0.41	0.37	0.41
WRD3	Gilbert	1.00	1.00	1.00	1.00
WRD3	Flinders	1.00	1.00	1.00	1.00
WRD3	Rgn 2-6	0.83	0.90	0.86	0.93
WRD4	Mitchell	1.00	1.00	1.00	1.00
WRD4	Gilbert	0.25	0.34	0.30	0.35
WRD4	Flinders	0.11	0.23	0.08	0.23
WRD4	Rgn 2-6	0.31	0.40	0.22	0.33
Model version 5 - sawfish					
WRD Scenario	Rgn Name	Rel.N (Min)	Rel.N (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)
WRD1	Mitchell	0.14	0.29	0.16	0.3
WRD1	Gilbert	0.05	0.28	0.07	0.31
WRD1	Flinders	0.04	0.16	0.04	0.17
WRD1	Rgn 2-6	0.11	0.24	0.09	0.22
WRD2	Mitchell	0.21	0.36	0.24	0.37
WRD2	Gilbert	0.05	0.28	0.07	0.31
WRD2	Flinders	0.04	0.17	0.04	0.18
WRD2	Rgn 2-6	0.12	0.25	0.1	0.23
WRD3	Mitchell	0.19	0.33	0.21	0.34
WRD3	Gilbert	1	1	1	1
WRD3	Flinders	1	1	1	1
WRD3	Rgn 2-6	0.85	0.91	0.89	0.94
WRD4	Mitchell	1	1	1	1
WRD4	Gilbert	0.05	0.28	0.07	0.31
WRD4	Flinders	0.04	0.17	0.04	0.18
WRD4	Rgn 2-6	0.25	0.33	0.16	0.28

Table A46. Summary table of Model sensitivity version 7 compared with Ensemble Model version 5 showing minimum and mean biomass for mangroves predicted under four water resource development scenarios (WRD1-WRD4) relative to base line flows (BaseCase) for the Mitchell, Gilbert and Flinders rivers, as well as MICE regions 2-6 combined.

Model version 7 – sensitivity - mangroves			
WRD Scenario	Rgn Name	Rel.Mangrove (Min)	Rel.Mangrove (Mean)
BaseCase	Mitchell	1	1
BaseCase	Gilbert	1	1
BaseCase	Flinders	1	1
BaseCase	Rgn 2-6	1	1
WRD1	Mitchell	0.81	0.89
WRD1	Gilbert	0.61	0.80
WRD1	Flinders	0.66	0.88
WRD1	Rgn 2-6	0.81	0.89
WRD2	Mitchell	0.92	0.97
WRD2	Gilbert	0.62	0.82
WRD2	Flinders	0.68	0.89
WRD2	Rgn 2-6	0.92	0.96
WRD3	Mitchell	0.83	0.91
WRD3	Gilbert	1.00	1.00
WRD3	Flinders	1.00	1.00
WRD3	Rgn 2-6	0.90	0.94
WRD4	Mitchell	1.00	1.00
WRD4	Gilbert	0.62	0.82
WRD4	Flinders	0.68	0.89
WRD4	Rgn 2-6	0.95	0.98
Model version 5 – mangroves			
WRD Scenario	Rgn Name	Rel.Mangrove (Min)	Rel.Mangrove (Mean)
WRD1	Mitchell	0.67	0.81
WRD1	Gilbert	0.43	0.68
WRD1	Flinders	0.55	0.8
WRD1	Rgn 2-6	0.7	0.81
WRD2	Mitchell	0.87	0.95
WRD2	Gilbert	0.45	0.71
WRD2	Flinders	0.59	0.81
WRD2	Rgn 2-6	0.88	0.94
WRD3	Mitchell	0.71	0.83
WRD3	Gilbert	1	1
WRD3	Flinders	1	1
WRD3	Rgn 2-6	0.79	0.9
WRD4	Mitchell	1	1
WRD4	Gilbert	0.45	0.71
WRD4	Flinders	0.59	0.81
WRD4	Rgn 2-6	0.93	0.96

Appendix 21 – MICE WRD Risk Analysis

Risk analysis results for each species/group, based on model versions 2-5, under WRDs 1-4 are presented in the tables below.

Table A47. Example of fishery and population standardised risk scores for banana prawns, barramundi and mud crabs for the Mitchell, Flinders and Gilbert catchments as well as Regions 2-6 combined. Results are based on Model version 2. Bcom = commercially available biomass, Bsp = spawning biomass.

Model version 2 - common banana prawns												
WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)	Combined Risk Score	Fishery + Population Standardised Risk Score	Fishery Standardised Risk Score	Population Standardised Risk Score	
WRD1	Mitchell	4	3	4	3	5	3	22	3.7	3.5	4.0	
WRD1	Gilbert	6	5	6	5	6	5	33	5.5	5.5	5.5	
WRD1	Flinders	5	3	5	3	5	3	24	4.0	4.0	4.0	
WRD1	Rgn 2-6	5	4	5	4	5	4	27	4.5	4.5	4.5	
WRD2	Mitchell	2	2	2	1	2	1	10	1.7	1.8	1.5	
WRD2	Gilbert	5	5	6	5	6	5	32	5.3	5.3	5.5	
WRD2	Flinders	5	3	5	3	5	3	24	4.0	4.0	4.0	
WRD2	Rgn 2-6	5	3	5	3	5	3	24	4.0	4.0	4.0	
WRD3	Mitchell	4	3	3	3	3	3	19	3.2	3.3	3.0	
WRD3	Gilbert	1	1	1	1	1	1	6	1.0	1.0	1.0	
WRD3	Flinders	1	1	1	1	1	1	6	1.0	1.0	1.0	
WRD3	Rgn 2-6	3	2	1	1	1	1	9	1.5	1.8	1.0	
WRD4	Mitchell	1	1	1	1	1	1	6	1.0	1.0	1.0	
WRD4	Gilbert	6	5	5	5	6	5	32	5.3	5.3	5.5	
WRD4	Flinders	5	3	5	3	5	3	24	4.0	4.0	4.0	
WRD4	Rgn 2-6	5	2	5	3	5	3	23	3.8	3.8	4.0	

Model version 2 - barramundi

WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)	Combined Risk Score	Fishery + Population Standardised Risk Score	Fishery Standardised Risk Score	Population Standardised Risk Score
WRD1	Mitchell	4	3	4	3	5	3	22	3.7	3.5	4.0
WRD1	Gilbert	5	3	3	2	3	2	18	3.0	3.3	2.5
WRD1	Flinders	3	2	2	1	2	1	11	1.8	2.0	1.5
WRD1	Rgn 2-6	4	3	3	2	4	2	18	3.0	3.0	3.0
WRD2	Mitchell	2	1	2	1	2	1	9	1.5	1.5	1.5
WRD2	Gilbert	5	3	3	2	3	2	18	3.0	3.3	2.5
WRD2	Flinders	3	2	2	1	2	1	11	1.8	2.0	1.5
WRD2	Rgn 2-6	3	1	2	1	1	1	9	1.5	1.8	1.0
WRD3	Mitchell	4	3	4	3	4	3	21	3.5	3.5	3.5
WRD3	Gilbert	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD3	Flinders	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD3	Rgn 2-6	3	2	2	2	2	1	12	2.0	2.3	1.5
WRD4	Mitchell	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD4	Gilbert	5	3	3	2	3	2	18	3.0	3.3	2.5
WRD4	Flinders	3	2	2	1	2	1	11	1.8	2.0	1.5
WRD4	Rgn 2-6	2	1	1	1	1	1	7	1.2	1.3	1.0

Model version 2 - mud crab

WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)	Combined Risk Score	Fishery + Population Standardised Risk Score	Fishery Standardised Risk Score	Population Standardised Risk Score
WRD1	Mitchell	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD1	Gilbert	6	5	6	5	6	5	33	5.5	5.5	5.5
WRD1	Flinders	6	5	6	5	6	5	33	5.5	5.5	5.5
WRD1	Rgn 2-6	6	4	5	4	5	4	28	4.7	4.8	4.5
WRD2	Mitchell	1	1	1	1	1	1	6	1.0	1.0	1.0

WRD2	Gilbert	6	4	6	4	6	4	30	5.0	5.0	5.0
WRD2	Flinders	6	5	6	5	6	5	33	5.5	5.5	5.5
WRD2	Rgn 2-6	6	4	5	3	5	3	26	4.3	4.5	4.0
WRD3	Mitchell	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD3	Gilbert	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD3	Flinders	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD3	Rgn 2-6	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD4	Mitchell	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD4	Gilbert	6	4	6	4	6	4	30	5.0	5.0	5.0
WRD4	Flinders	6	5	6	5	6	5	33	5.5	5.5	5.5
WRD4	Rgn 2-6	6	4	5	3	5	3	26	4.3	4.5	4.0

Table A48. Example of fishery and population standardised risk scores for banana prawns, barramundi and mud crabs for the Mitchell, Flinders and Gilbert catchments as well as Regions 2-6 combined. Results are based on Model version 3. Bcom = commercially available biomass, Bsp = spawning biomass.

Model version 3 - common banana prawns											
WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)	Combined Risk Score	Fishery + Population Standardised Risk Score	Fishery Standardised Risk Score	Population Standardised Risk Score
WRD1	Mitchell	5	3	5	3	5	3	24	4.0	4.0	4.0
WRD1	Gilbert	3	3	3	3	4	3	19	3.2	3.0	3.5
WRD1	Flinders	5	4	5	3	5	4	26	4.3	4.3	4.5
WRD1	Rgn 2-6	5	4	5	3	5	3	25	4.2	4.3	4.0
WRD2	Mitchell	4	3	4	3	4	3	21	3.5	3.5	3.5
WRD2	Gilbert	2	1	2	1	2	1	9	1.5	1.5	1.5
WRD2	Flinders	5	3	5	3	5	3	24	4.0	4.0	4.0
WRD2	Rgn 2-6	4	3	4	2	4	3	20	3.3	3.3	3.5
WRD3	Mitchell	3	2	2	2	2	2	13	2.2	2.3	2.0

WRD3	Gilbert	3	2	3	2	3	2	15	2.5	2.5	2.5
WRD3	Flinders	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD3	Rgn 2-6	2	2	2	1	2	1	10	1.7	1.8	1.5
WRD4	Mitchell	4	2	4	2	4	3	19	3.2	3.0	3.5
WRD4	Gilbert	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD4	Flinders	5	3	5	3	5	3	24	4.0	4.0	4.0
WRD4	Rgn 2-6	4	3	4	2	4	2	19	3.2	3.3	3.0

Model version 3 - barramundi

WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)	Combined Risk Score	Fishery + Population Standardised Risk Score	Fishery Standardised Risk Score	Population Standardised Risk Score
WRD1	Mitchell	2	1	2	1	2	1	9	1.5	1.5	1.5
WRD1	Gilbert	6	6	6	5	6	5	34	5.7	5.8	5.5
WRD1	Flinders	5	3	4	3	5	3	23	3.8	3.8	4.0
WRD1	Rgn 2-6	6	4	4	3	3	2	22	3.7	4.3	2.5
WRD2	Mitchell	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD2	Gilbert	6	5	5	5	5	5	31	5.2	5.3	5.0
WRD2	Flinders	5	3	4	3	5	3	23	3.8	3.8	4.0
WRD2	Rgn 2-6	6	4	3	2	2	2	19	3.2	3.8	2.0
WRD3	Mitchell	2	1	2	1	1	1	8	1.3	1.5	1.0
WRD3	Gilbert	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD3	Flinders	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD3	Rgn 2-6	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD4	Mitchell	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD4	Gilbert	6	5	5	5	5	5	31	5.2	5.3	5.0
WRD4	Flinders	5	3	4	3	5	3	23	3.8	3.8	4.0
WRD4	Rgn 2-6	5	4	4	2	2	1	18	3.0	3.8	1.5

Model version 3 - mud crab												
WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)	Combined Risk Score	Fishery + Population Standardised Risk Score	Fishery Standardised Risk Score	Population Standardised Risk Score	
WRD1	Mitchell	1	1	1	1	1	1	6	1.0	1.0	1.0	
WRD1	Gilbert	6	5	6	5	6	5	33	5.5	5.5	5.5	
WRD1	Flinders	6	5	6	5	6	5	33	5.5	5.5	5.5	
WRD1	Rgn 2-6	6	4	6	5	6	5	32	5.3	5.3	5.5	
WRD2	Mitchell	1	1	1	1	1	1	6	1.0	1.0	1.0	
WRD2	Gilbert	6	5	6	5	6	5	33	5.5	5.5	5.5	
WRD2	Flinders	6	5	6	5	6	5	33	5.5	5.5	5.5	
WRD2	Rgn 2-6	6	4	5	4	5	4	28	4.7	4.8	4.5	
WRD3	Mitchell	1	1	1	1	1	1	6	1.0	1.0	1.0	
WRD3	Gilbert	1	1	1	1	1	1	6	1.0	1.0	1.0	
WRD3	Flinders	1	1	1	1	1	1	6	1.0	1.0	1.0	
WRD3	Rgn 2-6	1	1	1	1	1	1	6	1.0	1.0	1.0	
WRD4	Mitchell	1	1	1	1	1	1	6	1.0	1.0	1.0	
WRD4	Gilbert	6	5	6	5	6	5	33	5.5	5.5	5.5	
WRD4	Flinders	6	5	6	5	6	5	33	5.5	5.5	5.5	
WRD4	Rgn 2-6	6	4	5	4	5	4	28	4.7	4.8	4.5	

Table A49. Example of fishery and population standardised risk scores for banana prawns, barramundi and mud crabs for the Mitchell, Flinders and Gilbert catchments as well as Regions 2-6 combined. Results are based on Model version 4. Bcom = commercially available biomass, Bsp = spawning biomass.

Model version 4 - common banana prawns												
WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)	Combined Risk Score	Fishery + Population Standardised Risk Score	Fishery Standardised Risk Score	Population Standardised Risk Score	

WRD1	Mitchell	5	3	4	3	4	3	22	3.7	3.8	3.5
WRD1	Gilbert	4	3	3	3	4	3	20	3.3	3.3	3.5
WRD1	Flinders	5	3	5	3	5	3	24	4.0	4.0	4.0
WRD1	Rgn 2-6	5	4	4	3	4	3	23	3.8	4.0	3.5
WRD2	Mitchell	4	3	4	2	4	2	19	3.2	3.3	3.0
WRD2	Gilbert	2	1	2	1	2	1	9	1.5	1.5	1.5
WRD2	Flinders	5	3	5	3	5	3	24	4.0	4.0	4.0
WRD2	Rgn 2-6	4	2	4	2	4	2	18	3.0	3.0	3.0
WRD3	Mitchell	3	2	3	2	3	2	15	2.5	2.5	2.5
WRD3	Gilbert	3	2	3	2	3	2	15	2.5	2.5	2.5
WRD3	Flinders	2	1	2	1	2	1	9	1.5	1.5	1.5
WRD3	Rgn 2-6	3	2	2	1	2	1	11	1.8	2.0	1.5
WRD4	Mitchell	3	2	3	2	3	2	15	2.5	2.5	2.5
WRD4	Gilbert	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD4	Flinders	5	3	5	3	5	3	24	4.0	4.0	4.0
WRD4	Rgn 2-6	4	2	4	2	4	2	18	3.0	3.0	3.0

Model version 4 - barramundi

WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)	Combined Risk Score	Fishery + Population Standardised Risk Score	Fishery Standardised Risk Score	Population Standardised Risk Score
WRD1	Mitchell	4	3	4	3	4	3	21	3.5	3.5	3.5
WRD1	Gilbert	5	3	3	2	2	1	16	2.7	3.3	1.5
WRD1	Flinders	3	2	2	1	1	1	10	1.7	2.0	1.0
WRD1	Rgn 2-6	4	3	3	2	2	2	16	2.7	3.0	2.0
WRD2	Mitchell	2	1	2	1	2	1	9	1.5	1.5	1.5
WRD2	Gilbert	5	3	3	2	2	1	16	2.7	3.3	1.5
WRD2	Flinders	3	2	2	1	1	1	10	1.7	2.0	1.0
WRD2	Rgn 2-6	3	2	2	1	1	1	10	1.7	2.0	1.0
WRD3	Mitchell	3	3	3	3	4	3	19	3.2	3.0	3.5
WRD3	Gilbert	1	1	1	1	1	1	6	1.0	1.0	1.0

WRD3	Flinders	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD3	Rgn 2-6	3	1	2	1	2	1	10	1.7	1.8	1.5
WRD4	Mitchell	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD4	Gilbert	5	3	3	2	2	1	16	2.7	3.3	1.5
WRD4	Flinders	3	2	2	1	1	1	10	1.7	2.0	1.0
WRD4	Rgn 2-6	3	1	1	1	1	1	8	1.3	1.5	1.0

Model version 4 - mud crab

WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)	Combined Risk Score	Fishery + Population Standardised Risk Score	Fishery Standardised Risk Score	Population Standardised Risk Score
WRD1	Mitchell	4	2	4	1	4	2	17	2.8	2.8	3.0
WRD1	Gilbert	6	6	6	6	6	6	36	6.0	6.0	6.0
WRD1	Flinders	6	5	6	5	6	5	33	5.5	5.5	5.5
WRD1	Rgn 2-6	6	6	6	5	6	5	34	5.7	5.8	5.5
WRD2	Mitchell	2	1	1	1	1	1	7	1.2	1.3	1.0
WRD2	Gilbert	6	5	6	5	6	5	33	5.5	5.5	5.5
WRD2	Flinders	6	6	6	6	6	6	36	6.0	6.0	6.0
WRD2	Rgn 2-6	6	5	6	4	6	5	32	5.3	5.3	5.5
WRD3	Mitchell	4	2	3	2	4	1	16	2.7	2.8	2.5
WRD3	Gilbert	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD3	Flinders	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD3	Rgn 2-6	2	1	2	1	2	1	9	1.5	1.5	1.5
WRD4	Mitchell	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD4	Gilbert	6	5	6	5	6	5	33	5.5	5.5	5.5
WRD4	Flinders	6	6	6	6	6	6	36	6.0	6.0	6.0
WRD4	Rgn 2-6	6	5	6	4	6	5	32	5.3	5.3	5.5

Table A50. Example of fishery and population standardised risk scores for banana prawns, barramundi and mud crabs for the Mitchell, Flinders and Gilbert catchments as well as Regions 2-6 combined. Results are based on Model version 5. Bcom = commercially available biomass, Bsp = spawning biomass.

Model version 5 - common banana prawns												
WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)	Combined Risk Score	Fishery + Population Standardised Risk Score	Fishery Standardised Risk Score	Population Standardised Risk Score	
WRD1	Mitchell	5	4	5	4	5	4	27	4.5	4.5	4.5	
WRD1	Gilbert	5	4	5	4	5	4	27	4.5	4.5	4.5	
WRD1	Flinders	5	4	5	4	5	5	28	4.7	4.5	5.0	
WRD1	Rgn 2-6	6	5	6	4	6	5	32	5.3	5.3	5.5	
WRD2	Mitchell	4	3	4	3	4	3	21	3.5	3.5	3.5	
WRD2	Gilbert	4	3	3	3	3	3	19	3.2	3.3	3.0	
WRD2	Flinders	5	4	5	3	5	4	26	4.3	4.3	4.5	
WRD2	Rgn 2-6	5	4	5	4	5	4	27	4.5	4.5	4.5	
WRD3	Mitchell	2	1	2	1	2	1	9	1.5	1.5	1.5	
WRD3	Gilbert	3	2	3	2	3	2	15	2.5	2.5	2.5	
WRD3	Flinders	2	1	2	1	2	1	9	1.5	1.5	1.5	
WRD3	Rgn 2-6	2	1	2	1	2	1	9	1.5	1.5	1.5	
WRD4	Mitchell	4	3	4	3	4	3	21	3.5	3.5	3.5	
WRD4	Gilbert	3	2	3	2	3	2	15	2.5	2.5	2.5	
WRD4	Flinders	5	3	5	3	5	4	25	4.2	4.0	4.5	
WRD4	Rgn 2-6	5	4	5	4	5	4	27	4.5	4.5	4.5	

Model version 5 - barramundi

WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)	Combined Risk Score	Fishery + Population Standardised Risk Score	Fishery Standardised Risk Score	Population Standardised Risk Score
WRD1	Mitchell	4	3	4	3	4	3	21	3.5	3.5	3.5
WRD1	Gilbert	6	6	5	5	5	4	31	5.2	5.5	4.5
WRD1	Flinders	6	4	4	3	3	2	22	3.7	4.3	2.5
WRD1	Rgn 2-6	6	5	4	4	4	3	26	4.3	4.8	3.5
WRD2	Mitchell	2	1	2	1	2	1	9	1.5	1.5	1.5
WRD2	Gilbert	6	6	5	5	5	3	30	5.0	5.5	4.0
WRD2	Flinders	6	4	4	3	3	2	22	3.7	4.3	2.5
WRD2	Rgn 2-6	5	4	3	2	2	2	18	3.0	3.5	2.0
WRD3	Mitchell	3	3	3	3	3	3	18	3.0	3.0	3.0
WRD3	Gilbert	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD3	Flinders	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD3	Rgn 2-6	2	1	1	1	1	1	7	1.2	1.3	1.0
WRD4	Mitchell	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD4	Gilbert	6	6	5	5	5	3	30	5.0	5.5	4.0
WRD4	Flinders	6	4	4	3	3	2	22	3.7	4.3	2.5
WRD4	Rgn 2-6	5	4	3	2	2	1	17	2.8	3.5	1.5

Model version 5 - mud crab

WRD Scenario	Rgn Name	Rel.Catch (Min)	Rel.Catch (Mean)	Rel.Bcom (Min)	Rel.Bcom (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)	Combined Risk Score	Fishery + Population Standardised Risk Score	Fishery Standardised Risk Score	Population Standardised Risk Score
WRD1	Mitchell	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD1	Gilbert	6	5	6	5	6	5	33	5.5	5.5	5.5
WRD1	Flinders	6	5	6	5	6	5	33	5.5	5.5	5.5
WRD1	Rgn 2-6	6	5	6	5	6	5	33	5.5	5.5	5.5
WRD2	Mitchell	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD2	Gilbert	6	5	6	5	6	5	33	5.5	5.5	5.5
WRD2	Flinders	6	6	6	6	6	6	36	6.0	6.0	6.0
WRD2	Rgn 2-6	6	5	5	4	5	4	29	4.8	5.0	4.5
WRD3	Mitchell	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD3	Gilbert	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD3	Flinders	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD3	Rgn 2-6	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD4	Mitchell	1	1	1	1	1	1	6	1.0	1.0	1.0
WRD4	Gilbert	6	5	6	5	6	5	33	5.5	5.5	5.5
WRD4	Flinders	6	6	6	6	6	6	36	6.0	6.0	6.0
WRD4	Rgn 2-6	6	5	5	4	5	4	29	4.8	5.0	4.5

Table A51. Example of population standardised risk scores for sawfish for the Mitchell, Flinders and Gilbert catchments as well as Regions 2-6 combined. Results are based on Model versions 2-5. N = total numbers, Bsp = mature numbers.

Model version 2 - sawfish							
WRD	Region Name	Rel.N (Min)	Rel.N (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)	Combined Risk Score	Population Standardised Risk Score
WRD1	Mitchell	6	6	6	6	24	6.0
WRD1	Gilbert	6	6	6	6	24	6.0
WRD1	Flinders	6	6	6	6	24	6.0
WRD1	Rgn 2-6	6	6	6	6	24	6.0
WRD2	Mitchell	6	6	6	6	24	6.0
WRD2	Gilbert	6	6	6	6	24	6.0
WRD2	Flinders	6	6	6	6	24	6.0
WRD2	Rgn 2-6	6	6	6	6	24	6.0
WRD3	Mitchell	6	6	6	6	24	6.0
WRD3	Gilbert	1	1	1	1	4	1.0
WRD3	Flinders	1	1	1	1	4	1.0
WRD3	Rgn 2-6	4	3	3	2	12	3.0
WRD4	Mitchell	1	1	1	1	4	1.0
WRD4	Gilbert	6	6	6	5	23	5.8
WRD4	Flinders	6	6	6	6	24	6.0
WRD4	Rgn 2-6	6	6	6	6	24	6.0

Model version 3 - sawfish							
WRD	Region Name	Rel.N (Min)	Rel.N (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)	Combined Risk Score	Population Standardised Risk Score
WRD1	Mitchell	6	5	6	5	22	5.5
WRD1	Gilbert	6	5	6	5	22	5.5
WRD1	Flinders	6	6	6	6	24	6.0
WRD1	Rgn 2-6	6	6	6	6	24	6.0
WRD2	Mitchell	5	5	5	5	20	5.0
WRD2	Gilbert	5	5	5	5	20	5.0
WRD2	Flinders	6	6	6	6	24	6.0
WRD2	Rgn 2-6	6	6	6	6	24	6.0
WRD3	Mitchell	5	5	5	5	20	5.0
WRD3	Gilbert	1	1	1	1	4	1.0
WRD3	Flinders	1	1	1	1	4	1.0
WRD3	Rgn 2-6	5	4	4	3	16	4.0
WRD4	Mitchell	1	1	1	1	4	1.0
WRD4	Gilbert	6	5	5	5	21	5.3
WRD4	Flinders	6	6	6	6	24	6.0
WRD4	Rgn 2-6	6	5	6	6	23	5.8

Model version 4 - sawfish

WRD	Region Name	Rel.N (Min)	Rel.N (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)	Combined Risk Score	Population Standardised Risk Score
WRD1	Mitchell	6	5	6	5	22	5.5
WRD1	Gilbert	6	5	6	5	22	5.5
WRD1	Flinders	6	6	6	6	24	6.0
WRD1	Rgn 2-6	6	6	6	6	24	6.0
WRD2	Mitchell	5	5	5	5	20	5.0
WRD2	Gilbert	6	5	5	5	21	5.3
WRD2	Flinders	6	6	6	6	24	6.0
WRD2	Rgn 2-6	6	6	6	6	24	6.0
WRD3	Mitchell	5	5	5	5	20	5.0
WRD3	Gilbert	1	1	1	1	4	1.0
WRD3	Flinders	1	1	1	1	4	1.0
WRD3	Rgn 2-6	5	4	4	3	16	4.0
WRD4	Mitchell	1	1	1	1	4	1.0
WRD4	Gilbert	6	5	5	5	21	5.3
WRD4	Flinders	6	6	6	6	24	6.0
WRD4	Rgn 2-6	6	5	6	6	23	5.8

Model version 5 - sawfish

WRD	Region Name	Rel.N (Min)	Rel.N (Mean)	Rel.Bsp (Min)	Rel.Bsp (Mean)	Combined Risk Score	Population Standardised Risk Score
WRD1	Mitchell	6	6	6	6	24	6.0
WRD1	Gilbert	6	6	6	6	24	6.0
WRD1	Flinders	6	6	6	6	24	6.0
WRD1	Rgn 2-6	6	6	6	6	24	6.0
WRD2	Mitchell	6	6	6	6	24	6.0
WRD2	Gilbert	6	6	6	6	24	6.0
WRD2	Flinders	6	6	6	6	24	6.0
WRD2	Rgn 2-6	6	6	6	6	24	6.0
WRD3	Mitchell	6	6	6	6	24	6.0
WRD3	Gilbert	1	1	1	1	4	1.0
WRD3	Flinders	1	1	1	1	4	1.0
WRD3	Rgn 2-6	4	2	3	2	11	2.8
WRD4	Mitchell	1	1	1	1	4	1.0
WRD4	Gilbert	6	6	6	6	24	6.0
WRD4	Flinders	6	6	6	6	24	6.0
WRD4	Rgn 2-6	6	6	6	6	24	6.0

Table A52. Example of habitat standardised risk scores for seagrass and mangroves for the Mitchell, Flinders and Gilbert catchments as well as Regions 2-6 combined. Results are based on Model versions 2-5.

Model version 1 - Seagrass and mangroves									
WRD Scenario	Region Name	Rel.Seagrass (Min)	Rel.Seagrass (Mean)	Combined Risk Score	Standardised Risk Score	Rel.Mangrove (Min)	Rel.Mangrove (Mean)	Combined Risk Score	Standardised Risk Score
WRD1	Mitchell	1	1	2	1.0	5	4	9	4.5
WRD1	Gilbert	1	1	2	1.0	6	5	11	5.5
WRD1	Flinders	1	1	2	1.0	6	4	10	5.0
WRD1	Rgn 2-6	1	1	2	1.0	6	4	10	5.0
WRD2	Mitchell	1	1	2	1.0	3	2	5	2.5
WRD2	Gilbert	1	1	2	1.0	6	5	11	5.5
WRD2	Flinders	1	1	2	1.0	4	4	8	4.0
WRD2	Rgn 2-6	1	1	2	1.0	4	2	6	3.0
WRD3	Mitchell	1	1	2	1.0	5	4	9	4.5
WRD3	Gilbert	1	1	2	1.0	1	1	2	1.0
WRD3	Flinders	1	1	2	1.0	1	1	2	1.0
WRD3	Rgn 2-6	1	1	2	1.0	5	3	8	4.0
WRD4	Mitchell	1	1	2	1.0	1	1	2	1.0
WRD4	Gilbert	1	1	2	1.0	6	5	11	5.5
WRD4	Flinders	1	1	2	1.0	4	4	8	4.0
WRD4	Rgn 2-6	1	1	2	1.0	2	2	4	2.0

Model version 2 - Seagrass and mangroves

WRD Scenario	Region Name	Rel.Seagrass (Min)	Rel.Seagrass (Mean)	Combined Risk Score	Standardised Risk Score	Rel.Mangrove (Min)	Rel.Mangrove (Mean)	Combined Risk Score	Standardised Risk Score
WRD1	Mitchell	1	1	2	1.0	5	4	9	4.5
WRD1	Gilbert	1	1	2	1.0	6	5	11	5.5
WRD1	Flinders	2	1	3	1.5	6	4	10	5.0
WRD1	Rgn 2-6	1	1	2	1.0	6	5	11	5.5
WRD2	Mitchell	1	1	2	1.0	3	2	5	2.5
WRD2	Gilbert	1	1	2	1.0	5	5	10	5.0
WRD2	Flinders	2	1	3	1.5	5	4	9	4.5
WRD2	Rgn 2-6	1	1	2	1.0	3	2	5	2.5
WRD3	Mitchell	1	1	2	1.0	5	4	9	4.5
WRD3	Gilbert	1	1	2	1.0	1	1	2	1.0
WRD3	Flinders	1	1	2	1.0	1	1	2	1.0
WRD3	Rgn 2-6	1	1	2	1.0	5	3	8	4.0
WRD4	Mitchell	1	1	2	1.0	1	1	2	1.0
WRD4	Gilbert	1	1	2	1.0	6	5	11	5.5
WRD4	Flinders	2	1	3	1.5	5	4	9	4.5
WRD4	Rgn 2-6	1	1	2	1.0	2	2	4	2.0

Model version 3 - Seagrass and mangroves

WRD Scenario	Region Name	Rel.Seagrass (Min)	Rel.Seagrass (Mean)	Combined Risk Score	Standardised Risk Score	Rel.Mangrove (Min)	Rel.Mangrove (Mean)	Combined Risk Score	Standardised Risk Score
WRD1	Mitchell	1	1	2	1.0	5	4	9	4.5
WRD1	Gilbert	1	1	2	1.0	6	5	11	5.5
WRD1	Flinders	2	1	3	1.5	6	4	10	5.0
WRD1	Rgn 2-6	1	1	2	1.0	6	5	11	5.5
WRD2	Mitchell	1	1	2	1.0	3	2	5	2.5
WRD2	Gilbert	1	1	2	1.0	6	5	11	5.5
WRD2	Flinders	2	1	3	1.5	5	4	9	4.5
WRD2	Rgn 2-6	1	1	2	1.0	3	2	5	2.5
WRD3	Mitchell	1	1	2	1.0	5	4	9	4.5
WRD3	Gilbert	1	1	2	1.0	1	1	2	1.0
WRD3	Flinders	1	1	2	1.0	1	1	2	1.0
WRD3	Rgn 2-6	1	1	2	1.0	5	3	8	4.0
WRD4	Mitchell	1	1	2	1.0	1	1	2	1.0
WRD4	Gilbert	1	1	2	1.0	6	5	11	5.5
WRD4	Flinders	2	1	3	1.5	5	4	9	4.5
WRD4	Rgn 2-6	1	1	2	1.0	2	2	4	2.0

Model version 4 - Seagrass and mangroves

WRD Scenario	Region Name	Rel.Seagrass (Min)	Rel.Seagrass (Mean)	Combined Risk Score	Standardised Risk Score	Rel.Mangrove (Min)	Rel.Mangrove (Mean)	Combined Risk Score	Standardised Risk Score
WRD1	Mitchell	1	1	2	1.0	5	4	9	4.5
WRD1	Gilbert	1	1	2	1.0	6	5	11	5.5
WRD1	Flinders	2	1	3	1.5	6	4	10	5.0
WRD1	Rgn 2-6	1	1	2	1.0	6	5	11	5.5
WRD2	Mitchell	1	1	2	1.0	3	2	5	2.5
WRD2	Gilbert	1	1	2	1.0	6	5	11	5.5
WRD2	Flinders	2	1	3	1.5	6	4	10	5.0
WRD2	Rgn 2-6	1	1	2	1.0	4	2	6	3.0
WRD3	Mitchell	1	1	2	1.0	5	4	9	4.5
WRD3	Gilbert	1	1	2	1.0	1	1	2	1.0
WRD3	Flinders	1	1	2	1.0	1	1	2	1.0
WRD3	Rgn 2-6	1	1	2	1.0	5	4	9	4.5
WRD4	Mitchell	1	1	2	1.0	1	1	2	1.0
WRD4	Gilbert	1	1	2	1.0	6	5	11	5.5
WRD4	Flinders	2	1	3	1.5	6	4	10	5.0
WRD4	Rgn 2-6	1	1	2	1.0	2	2	4	2.0

Model version 5 - Seagrass and mangroves									
WRD Scenario	Region Name	Rel.Seagrass (Min)	Rel.Seagrass (Mean)	Combined Risk Score	Standardised Risk Score	Rel.Mangrove (Min)	Rel.Mangrove (Mean)	Combined Risk Score	Standardised Risk Score
WRD1	Mitchell	1	1	2	1.0	5	3	8	4.0
WRD1	Gilbert	1	2	3	1.5	6	5	11	5.5
WRD1	Flinders	3	1	4	2.0	5	3	8	4.0
WRD1	Rgn 2-6	1	1	2	1.0	5	4	9	4.5
WRD2	Mitchell	1	1	2	1.0	3	2	5	2.5
WRD2	Gilbert	1	2	3	1.5	6	4	10	5.0
WRD2	Flinders	3	1	4	2.0	5	3	8	4.0
WRD2	Rgn 2-6	1	1	2	1.0	3	2	5	2.5
WRD3	Mitchell	1	1	2	1.0	4	3	7	3.5
WRD3	Gilbert	1	1	2	1.0	1	1	2	1.0
WRD3	Flinders	1	1	2	1.0	1	1	2	1.0
WRD3	Rgn 2-6	1	1	2	1.0	4	2	6	3.0
WRD4	Mitchell	1	1	2	1.0	1	1	2	1.0
WRD4	Gilbert	1	2	3	1.5	6	4	10	5.0
WRD4	Flinders	3	1	4	2.0	5	3	8	4.0
WRD4	Rgn 2-6	1	1	2	1.0	2	1	3	1.5

Economic risk analysis results for the common banana prawn fishery, based on model versions 2-5, under WRDs 1-4 are presented in the figures below.

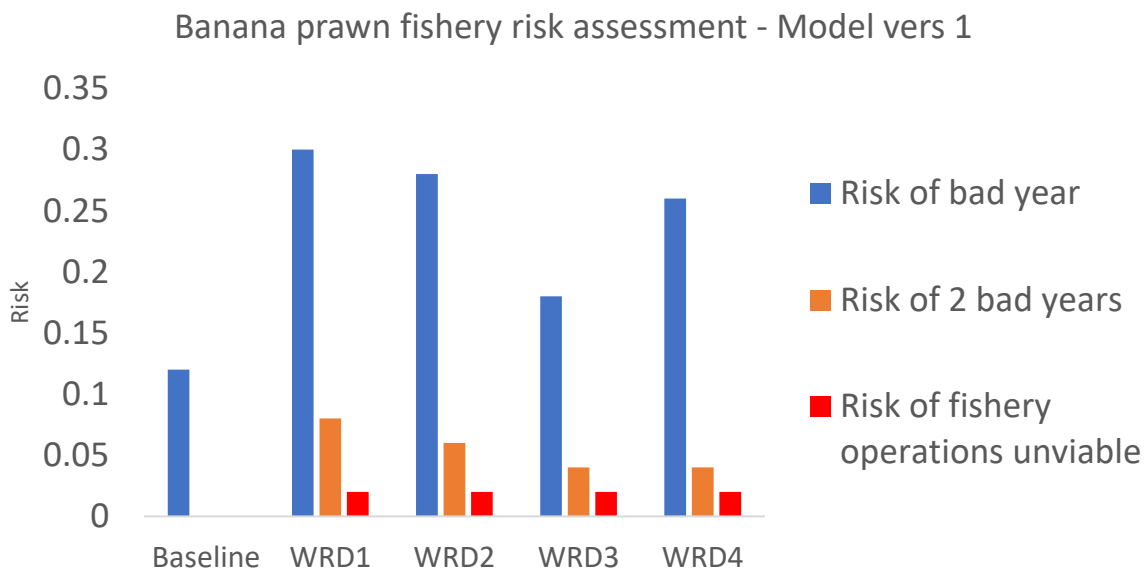


Figure A153. Economic risk assessment for banana prawn fishery using MICE model version 1 showing risk of 1, 2 or 3+ bad catch years under four WRD scenarios relative to baseline.

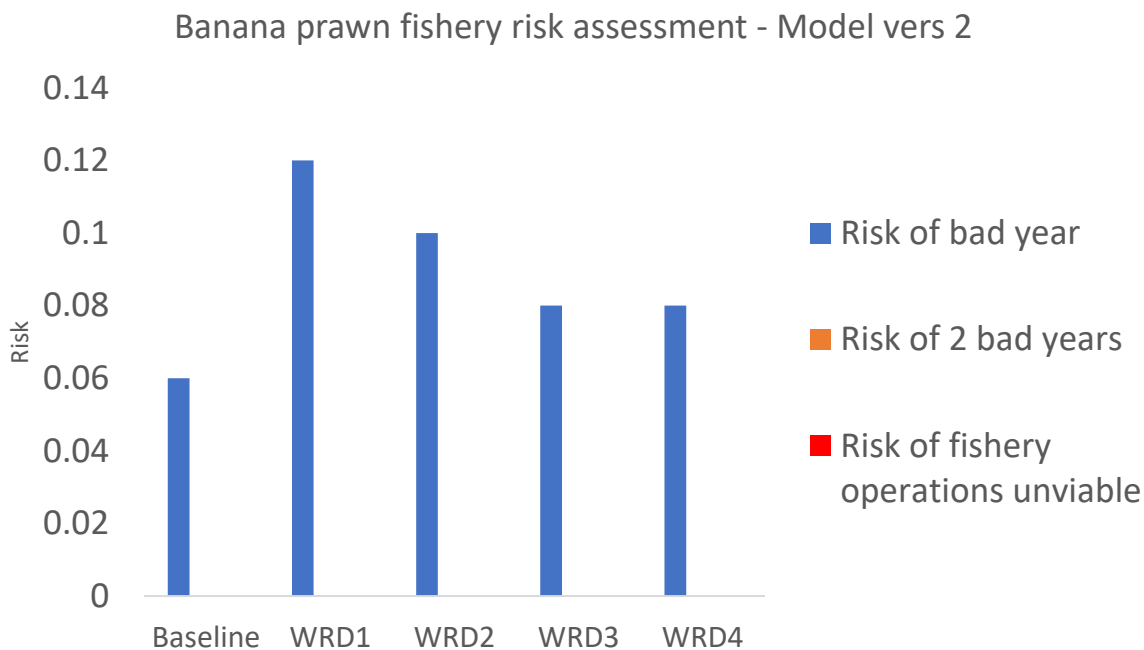


Figure A154. Economic risk assessment for banana prawn fishery using MICE model version 2 showing risk of 1, 2 or 3+ bad catch years under four WRD scenarios relative to baseline.

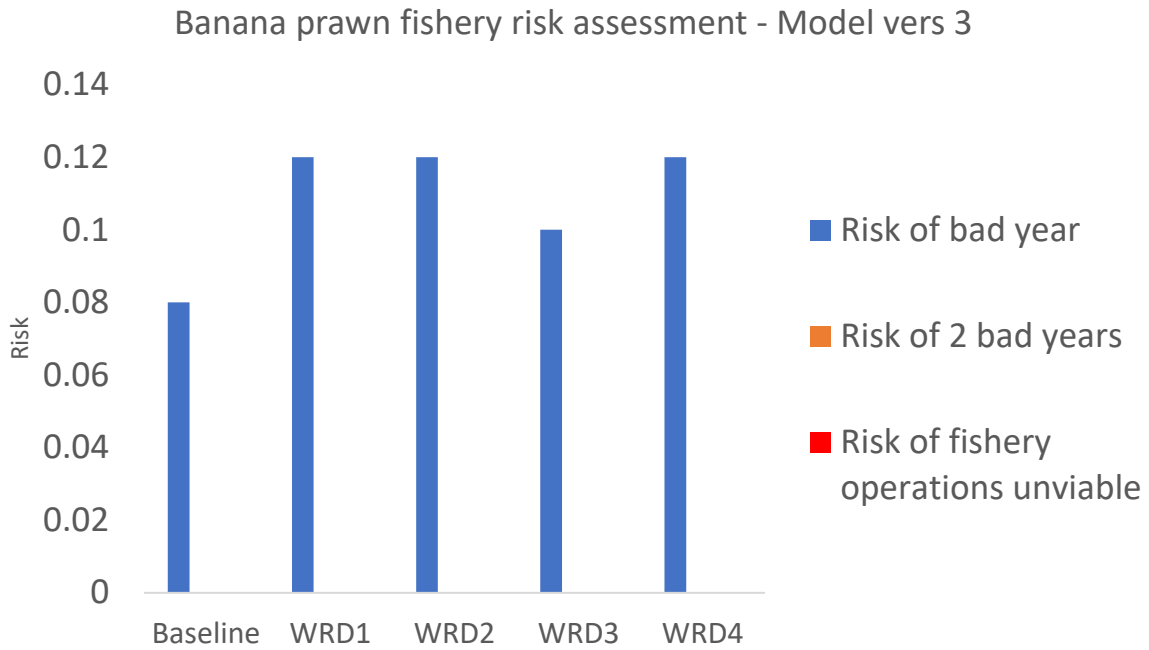


Figure A155. Economic risk assessment for banana prawn fishery using MICE model version 3 showing risk of 1, 2 or 3+ bad catch years under four WRD scenarios relative to baseline.

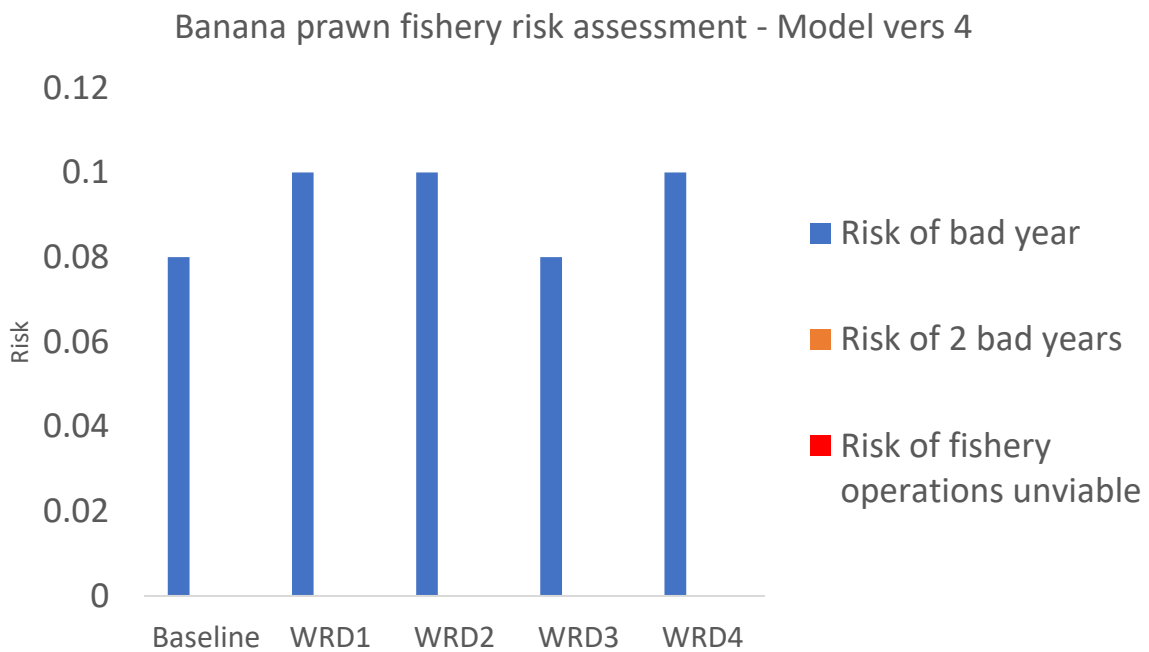


Figure A156. Economic risk assessment for banana prawn fishery using MICE model version 4 showing risk of 1, 2 or 3+ bad catch years under four WRD scenarios relative to baseline.

Banana prawn fishery risk assessment - Model vers 5

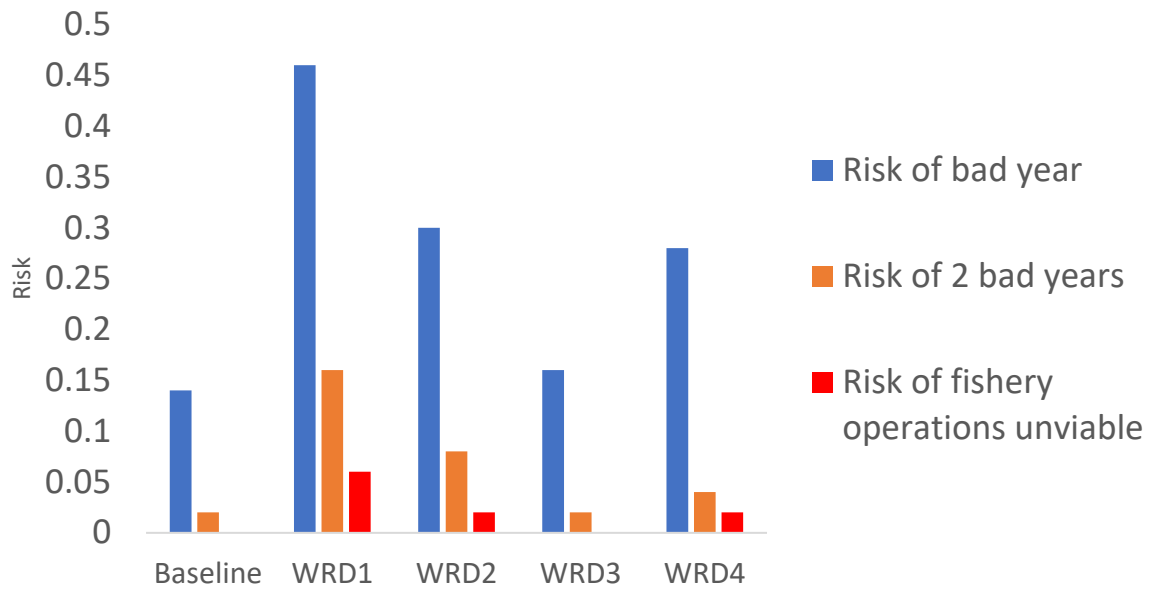


Figure A157. Economic risk assessment for banana prawn fishery using MICE model version 5 showing risk of 1, 2 or 3+ bad catch years under four WRD scenarios relative to baseline.

Appendix 22 – Literature Cited

- Adey, W. H. 1998. Coral reefs: algal structured and mediated ecosystems in shallow, turbulent, alkaline waters. *Journal of Phycology* **34**:393-406.
- Alberts-Hubatsch, H. 2015. Movement patterns and habitat use of the exploited swimming crab *Scylla serrata* (Forsk., 1775). Universität Bremen.
- Alberts-Hubatsch, H., S. Y. Lee, J.-O. Meynecke, K. Diele, I. Nordhaus, and M. Wolff. 2016. Life-history, movement, and habitat use of *Scylla serrata* (Decapoda, Portunidae): current knowledge and future challenges. *Hydrobiologia* **763**:5-21.
- Alongi, D. M. 2012. Carbon sequestration in mangrove forests. *Carbon management* **3**:313-322.
- Arias, P., N. Bellouin, E. Coppola, R. Jones, G. Krinner, J. Marotzke, V. Naik, M. Palmer, G.-K. Plattner, and J. Rogelj. 2021. Climate Change 2021: The Physical Science Basis. Contribution of Working Group 14 I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change; Technical Summary.
- Asbridge, E., R. Lucas, C. Ticehurst, and P. Bunting. 2016. Mangrove response to environmental change in Australia's Gulf of Carpentaria. *Ecology and evolution* **6**:3523-3539.
- Babcock, R. C., R. H. Bustamante, E. A. Fulton, D. J. Fulton, M. D. Haywood, A. J. Hobday, R. Kenyon, R. J. Matear, E. E. Plaganyi, and A. J. Richardson. 2019. Severe continental-scale impacts of climate change are happening now: Extreme climate events impact marine habitat forming communities along 45% of Australia's coast. *Frontiers in Marine Science* **6**:411.
- Balston, J. 2009. Short-term climate variability and the commercial barramundi (*Lates calcarifer*) fishery of north-east Queensland, Australia. *Marine and Freshwater Research* **60**:912-923.
- Barber, M. 2013. Indigenous water values, rights and interests in the Flinders and Gilbert catchments. A technical report to the Australian Government from the CSIRO Flinders and Gilbert Agricultural Resource Assessment, part of the North Queensland Irrigated Agriculture Strategy. CSIRO Water for a Healthy Country and Sustainable Agriculture flagships, Australia.
- Barber, M., and E. Woodward. 2018. Indigenous water values, rights, interests and development objectives in the Fitzroy catchment. A technical report to the Australian Government from the CSIRO Northern Australia Water Resource Assessment, part of the National Water Infrastructure Development Fund: Water Resource Assessments. CSIRO, Australia.
- Bayliss, P., R. Buckworth, and C. Dichmont. 2014. Assessing the Water Needs of Fisheries and Ecological Values in the Gulf of Carpentaria: Final Report Prepared for the Queensland Department of Natural Resources and Mines (DNRM). Commonwealth Scientific and Industrial Research Organisation, Brisbane.
- Baylon, J. C. 2010. Effects of salinity and temperature on survival and development of larvae and juveniles of the mud crab, *Scylla serrata* (Crustacea: Decapoda: Portunidae). *Journal of the World Aquaculture Society* **41**:858-873.
- Beverton, R. J., and S. J. Holt. 1957. On the dynamics of exploited fish populations. *Fishery Investigations, Series II*, 19. London: Her Majesty's Stationary Office.
- Blaber, S., D. Brewer, and J. Salini. 1989. Species composition and biomasses of fishes in different habitats of a tropical northern Australian estuary: their occurrence in the adjoining sea and estuarine dependence. *Estuarine, Coastal and Shelf Science* **29**:509-531.
- Blaber, S., D. Brewer, J. Salini, J. Kerr, and C. Conacher. 1992. Species composition and biomasses of fishes in tropical seagrasses at Groote Eylandt, northern Australia. *Estuarine, Coastal and Shelf Science* **35**:605-620.
- Blaber, S. J., D. T. Brewer, and J. P. Salini. 1994. Diet and dentition in tropical ariid catfishes from Australia. *Environmental Biology of Fishes* **40**:159-174.
- Blamey, L. K., É. E. Plagányi, T. Hutton, R. A. Deng, J. Upston, and A. Jarrett. 2021. Redesigning harvest strategies for sustainable fishery management in the face of extreme environmental variability. *Conservation Biology*.
- Bradford, R., R. Thomson, M. Bravington, D. Foote, R. Gunasekera, B. Bruce, D. Harasti, N. Otway, and P. Feutry. 2018. A close-kin mark-recapture estimate of the population size and trend of east coast grey nurse shark. Report to the National Environmental Science Program, Marine Biodiversity Hub. CSIRO Oceans & Atmosphere, Hobart, Tasmania.
- Bravington, M. V., H. J. Skaug, and E. C. Anderson. 2016. Close-kin mark-recapture. *Statistical Science* **31**:259-274.

- Brewer, D., S. Blaber, and J. Salini. 1991. Predation on penaeid prawns by fishes in Albatross Bay, Gulf of Carpentaria. *Marine Biology* **109**:231-240.
- Brewer, D., S. Blaber, J. Salini, and M. Farmer. 1995. Feeding ecology of predatory fishes from Groote Eylandt in the Gulf of Carpentaria, Australia, with special reference to predation on penaeid prawns. *Estuarine, Coastal and Shelf Science* **40**:577-600.
- Broadley, A., B. Stewart-Koster, R. A. Kenyon, M. A. Burford, and C. J. Brown. 2020. Impact of water development on river flows and the catch of a commercial marine fishery. *Ecosphere* **11**:e03194.
- Brown, I. W. 2010. Taking Female Mud Crabs (*Scylla Serrata*): Assessment of Risks and Benefits. Fisheries Research and Development Corporation.
- Bruce, B., R. Bradford, M. Bravington, P. Feutry, P. Grewe, R. Gunasekera, D. Harasti, R. Hillary, and T. Patterson. 2018. A national assessment of the status of white sharks. National Environmental Science Program Marine Biodiversity Hub.
- Buckworth, R., W. Venables, E. Lawrence, T. Kompas, S. Pascoe, L. Chu, F. Hill, T. Hutton, and P. Rothlisberg. 2014. Incorporation of predictive models of banana prawn catch for MEY-based harvest strategy development for the Northern Prawn Fishery. Final Report to the Fisheries Research and Development Corporation, Project 2011/239. CSIRO Marine & Atmospheric Research, Brisbane, Australia. 115p.
- Bunn, S. E., M. C. Thoms, S. K. Hamilton, and S. J. Capon. 2006. Flow variability in dryland rivers: boom, bust and the bits in between. *River Research and Applications* **22**:179-186.
- Burford, M., S. Faggotter, and R. Kenyon. 2020a. Contribution of three rivers to floodplain and coastal productivity in the Gulf of Carpentaria: Component 1 final report. Griffith University, Brisbane.
- Burford, M., R. Kenyon, M. Whittle, and G. Curwen. 2010. River flow impacts on estuarine prawns in the Gulf of Carpentaria. Fisheries Research and Development Corporation final report 2007/003. 1921760222.
- Burford, M., D. Valdez, G. Curwen, S. Faggotter, D. Ward, and K. O. Brien. 2016. Inundation of saline supratidal mudflats provides an important source of carbon and nutrients in an aquatic system. *Marine Ecology Progress Series* **545**:21-33.
- Burford, M., I. Webster, A. Revill, R. Kenyon, M. Whittle, and G. Curwen. 2012. Controls on phytoplankton productivity in a wet-dry tropical estuary. *Estuarine, Coastal and Shelf Science* **113**:141-151.
- Burford, M. A., A. J. Cook, C. S. Fellows, S. R. Balcombe, and S. E. Bunn. 2008. Sources of carbon fuelling production in an arid floodplain river. *Marine and Freshwater Research* **59**:224-234.
- Burford, M. A., and S. J. Faggotter. 2021. Comparing the importance of freshwater flows driving primary production in three tropical estuaries. *Marine Pollution Bulletin* **169**:112565.
- Burford, M. A., S. J. Faggotter, and R. Kenyon. 2020b. Project 1.4: Contribution of rivers to the productivity of floodplains and coastal areas of the southern Gulf of Carpentaria – Component 1. NESP Northern Australia Environmental Resources Hub final report.
- Burford, M. A., S. J. Faggotter, V. Lowe, M. Venarsky, C. Frid, M. Ronan, R. Bush, and D. Edwards. 2021. Project 3.6: The vulnerability of food supplies for migratory shorebirds to altered flow in the southern Gulf of Carpentaria. NESP Northern Australia Environmental Resources Hub final report.
- Butcher, P., A. Boulton, and S. Smith. 2003. Mud crab (*Scylla serrata*: Portunidae) populations as indicators of the effectiveness of estuarine marine protected areas. *in* World Congress on Aquatic Protected Areas.
- Cai, W., G. Wang, B. Dewitte, L. Wu, A. Santoso, K. Takahashi, Y. Yang, A. Carréric, and M. J. McPhaden. 2018. Increased variability of eastern Pacific El Niño under greenhouse warming. *Nature* **564**:201-206.
- Campbell, A. B., J. Robins, and M. O'Neill. 2017. Assessment of the barramundi (*Lates calcarifer*) fishery in the Southern Gulf of Carpentaria, Queensland, Australia.
- Carruthers, T., W. Dennison, B. Longstaff, M. Waycott, E. Abal, L. McKenzie, and W. Long. 2002. Seagrass habitats of northeast Australia: models of key processes and controls. *Bulletin of Marine Science* **71**:1153-1169.
- Charles, S., C. Petheram, A. Berthet, G. Browning, G. Hodgson, M. Wheeler, A. Yang, S. Gallant, J. Vaze, and B. Wang. 2016. Climate data and their characterisation for hydrological and agricultural scenario modelling across the Fitzroy, Darwin and Mitchell catchments. A technical report to the Australian Government from the CSIRO Northern Australia Water Resource Assessment, part of the National Water Infrastructure Development Fund: Water Resource Assessments. CSIRO, Australia. A technical report to the Australian Government from the CSIRO Northern ...
- Cheng, L., J. Abraham, Z. Hausfather, and K. E. Trenberth. 2019. How fast are the oceans warming? *Science* **363**:128-129.

- Church, J. A., and N. J. White. 2006. A 20th century acceleration in global sea-level rise. *Geophysical research letters* **33**.
- Collie, J. S., L. W. Botsford, A. Hastings, I. C. Kaplan, J. L. Largier, P. A. Livingston, É. Plagányi, K. A. Rose, B. K. Wells, and F. E. Werner. 2016. Ecosystem models for fisheries management: finding the sweet spot. *Fish and Fisheries* **17**:101-125.
- Collier, C. J., and M. Waycott. 2014. Temperature extremes reduce seagrass growth and induce mortality. *Marine Pollution Bulletin* **83**:483-490.
- Conacher, C., C. O'Brien, J. Horrocks, and R. Kenyon. 1996. Litter production and accumulation in stressed mangrove communities in the Embley River estuary, North-eastern Gulf of Carpentaria, Australia. *Marine and Freshwater Research* **47**:737-743.
- Crook, D., D. Buckle, Q. Allsop, W. Baldwin, T. Saunders, P. Kyne, J. Woodhead, R. Maas, B. Roberts, and M. Douglas. 2017. Use of otolith chemistry and acoustic telemetry to elucidate migratory contingents in barramundi *Lates calcarifer*. *Marine and Freshwater Research* **68**:1554-1566.
- Davis, T., and G. Kirkwood. 1984. Age and growth studies on barramundi, *Lates calcarifer* (Bloch), in northern Australia. *Marine and Freshwater Research* **35**:673-689.
- Deng, R., T. Hutton, R. Buckworth, S. Zhou, and A. Punt. 2020a. Modification of the NPF Endeavour prawn stock assessment models to accommodate management changes. Unpublished Document. O&A Brisbane, St Lucia. 29 pages.
- Deng, R., T. Hutton, A. E. Punt, J. Upston, M. Miller, C. Moeseneder, and S. Pascoe. 2020b. Status of the Northern Prawn Fishery Tiger Prawn Fishery at the end of 2019 with estimated TAEs for 2020 and 2021. CSIRO, Brisbane.
- Deng, R. A., A. E. Punt, C. M. Dichmont, R. C. Buckworth, and C. Y. BurrIDGE. 2015. Improving catch prediction for tiger prawns in the Australian northern prawn fishery. *ICES Journal of Marine Science* **72**:117-129.
- Dichmont, C. M., N. R. Loneragan, D. T. Brewer, and I. R. Poiner. 2007. Partnerships towards sustainable use of Australia's Northern Prawn Fishery. *Fisheries Management: Progress Toward Sustainability*.
- Dichmont, C. M., S. Pascoe, T. Kompas, A. E. Punt, and R. Deng. 2010. On implementing maximum economic yield in commercial fisheries. *Proceedings of the national academy of sciences* **107**:16-21.
- Douglas, M. M., S. Jackson, C. A. Canham, S. Laborde, L. Beesley, M. J. Kennard, B. J. Pusey, R. Loomes, and S. A. Setterfield. 2019. Conceptualizing hydro-socio-ecological relationships to enable more integrated and inclusive water allocation planning. *One Earth* **1**:361-373.
- Duarte, C. M., and J. Cebrián. 1996. The fate of marine autotrophic production. *Limnology and Oceanography* **41**:1758-1766.
- Duggan, M., P. Bayliss, and M. A. Burford. 2019. Predicting the impacts of freshwater-flow alterations on prawn (*Penaeus merguensis*) catches. *Fisheries Research* **215**:27-37.
- Duggan, M., R. Connolly, M. Whittle, G. Curwen, and M. Burford. 2014. Effects of freshwater flow extremes on intertidal biota of a wet-dry tropical estuary. *Marine Ecology Progress Series* **502**:11-23.
- Duke, N. C., C. Field, J. R. Mackenzie, J.-O. Meynecke, and A. L. Wood. 2019. Rainfall and its possible hysteresis effect on the proportional cover of tropical tidal-wetland mangroves and saltmarsh-saltpans. *Marine and Freshwater Research* **70**:1047-1055.
- Duke, N. C., L. B. Hutley, J. R. Mackenzie, and D. Burrows. 2021. Processes and factors driving change in mangrove forests: An evaluation based on the mass dieback event in Australia's Gulf of Carpentaria. Pages 221-264 *Ecosystem Collapse and Climate Change*. Springer.
- Duke, N. C., J. M. Kovacs, A. D. Griffiths, L. Preece, D. J. Hill, P. Van Oosterzee, J. Mackenzie, H. S. Morning, and D. Burrows. 2017. Large-scale dieback of mangroves in Australia's Gulf of Carpentaria: a severe ecosystem response, coincidental with an unusually extreme weather event. *Marine and Freshwater Research* **68**:1816-1829.
- Duke, N. C., J.-O. Meynecke, S. Dittmann, A. M. Ellison, K. Anger, U. Berger, S. Cannicci, K. Diele, K. C. Ewel, and C. D. Field. 2007. A world without mangroves? *Science* **317**:41-42.
- Dulvy, N. K., L. N. Davidson, P. M. Kyne, C. A. Simpfendorfer, L. R. Harrison, J. K. Carlson, and S. V. Fordham. 2016. Ghosts of the coast: global extinction risk and conservation of sawfishes. *Aquatic Conservation: marine and freshwater ecosystems* **26**:134-153.
- Dutta, D., F. Karim, C. Ticehurst, S. Marvanek, and C. Petheram. 2013. Floodplain inundation mapping and modelling in the Flinders and Gilbert Catchments. A Technical Report to the Australian Government from the CSIRO Flinders and Gilbert Agricultural Resource Assessment, Part of the North Queensland Irrigated Agriculture Strategy. CSIRO Water for a Healthy Country and Sustainable Agriculture flagships.

- Dutta, D., S. Kim, J. Hughes, J. Vaze, and A. Yang. 2015. AWRA-R v5.0 Technical Report. CSIRO Land and Water, Australia.
- Erisman, B. E., L. G. Allen, J. T. Claisse, D. J. Pondella, E. F. Miller, and J. H. Murray. 2011. The illusion of plenty: hyperstability masks collapses in two recreational fisheries that target fish spawning aggregations. *Canadian Journal of Fisheries and Aquatic Sciences* **68**:1705-1716.
- Eslami-Andargoli, L., P. Dale, N. Sipe, and J. Chaseling. 2009. Mangrove expansion and rainfall patterns in Moreton Bay, Southeast Queensland, Australia. *Estuarine, Coastal and Shelf Science* **85**:292-298.
- Feutry, P., P. M. Kyne, R. D. Pillans, X. Chen, J. R. Marthick, D. L. Morgan, and P. M. Grewe. 2015. Whole mitogenome sequencing refines population structure of the Critically Endangered sawfish *Pristis pristis*. *Marine Ecology Progress Series* **533**:237-244.
- Fielder, D., and M. P. Heasman. 1978. The mud crab. Queensland Museum.
- Flegel, T. 2007. The right to refuse revision in the genus *Penaeus*. *Aquaculture* **264**:2-8.
- Fournier, D. A., H. J. Skaug, J. Ancheta, J. Ianelli, A. Magnusson, M. N. Maunder, A. Nielsen, and J. Sibert. 2012. AD Model Builder: using automatic differentiation for statistical inference of highly parameterized complex nonlinear models. *Optimization Methods and Software* **27**:233-249.
- Friess, D. A., K. Rogers, C. E. Lovelock, K. W. Krauss, S. E. Hamilton, S. Y. Lee, R. Lucas, J. Primavera, A. Rajkaran, and S. Shi. 2019. The state of the world's mangrove forests: past, present, and future. *Annual Review of Environment and Resources* **44**:89-115.
- Froehlich, H. E., T. E. Essington, and P. S. McDonald. 2017. When does hypoxia affect management performance of a fishery? A management strategy evaluation of Dungeness crab (*Metacarcinus magister*) fisheries in Hood Canal, Washington, USA. *Canadian Journal of Fisheries and Aquatic Sciences* **74**:922-932.
- Fulton, E. A., A. J. Hobday, H. Pethybridge, J. Blanchard, C. Bulman, I. Butler, W. Cheung, L. Dutra, R. Gorton, and T. Hutton. 2018. Decadal scale projection of changes in Australian fisheries stocks under climate change.
- Fulton, E. A., J. S. Link, I. C. Kaplan, M. Savina-Rolland, P. Johnson, C. Ainsworth, P. Horne, R. Gorton, R. J. Gamble, A. D. M. Smith, and D. C. Smith. 2011. Lessons in modelling and management of marine ecosystems: the Atlantis experience. *Fish and Fisheries* **12**:171-188.
- García, V. B., L. O. Lucifora, and R. A. Myers. 2008. The importance of habitat and life history to extinction risk in sharks, skates, rays and chimaeras. *Proceedings. Biological sciences* **275**:83-89.
- Grech, A., and R. Coles. 2010. An ecosystem-scale predictive model of coastal seagrass distribution. *Aquatic Conservation: marine and freshwater ecosystems* **20**:437-444.
- Griffiths, S., R. Kenyon, E. Poloczanska, R. Deng, D. Milton, P. Rothlisberg, M. Burford, F. Coman, and M. Miller. 2014. Qualitative risk assessment of the potential effects of altered river flows on ecosystems in the Flinders and Gilbert river catchments. In *Assessing the Water Needs of Fisheries and Ecological Values in the Gulf of Carpentaria*. Queensland, AU: CSIRO Oceans and Atmosphere Flagship.
- Grubert, M. 2018. Giant mud crab *Scylla serrata*. Status of key northern territory fish stocks report.
- Grubert, M. A., C. J. Walters, R. C. Buckworth, and S. S. Penny. 2019. Simple Modeling to Inform Harvest Strategy Policy for a Data-Moderate Crab Fishery. *Marine and Coastal Fisheries* **11**:125-138.
- Halliday, I. A., J. B. Robins, D. G. Mayer, J. Staunton-Smith, and M. J. Sellin. 2010. Freshwater flows affect the year-class strength of 'Barramundi lates calcarifer' in the Fitzroy River estuary, Central Queensland. *Proceedings of the Royal Society of Queensland, The* **116**:1-10.
- Harley, S. J., R. A. Myers, and A. Dunn. 2001. Is catch-per-unit-effort proportional to abundance? *Canadian Journal of Fisheries and Aquatic Sciences* **58**:1760-1772.
- Hart, B., E. O'Donnell, and A. Horne. 2019. Sustainable water resources development in northern Australia: the need for coordination, integration and representation. *International Journal of Water Resources Development*.
- Hay, T., and C. Calogeras. 2001. NT mud crab fishery: summary of assessment information 1996–1999. Northern Territory Department of Primary Industry and Fisheries, Fishery Report **53**.
- Haynes, J., and S. Pascoe. 1988. A policy model of the northern prawn fishery. 0644078170, Australian Bureau of Agricultural and Resource Economics, Canberra.
- Haywood, M., D. Vance, and N. Loneragan. 1995. Seagrass and algal beds as nursery habitats for tiger prawns (*Penaeus semisulcatus* and *P. esculentus*) in a tropical Australian estuary. *Marine Biology* **122**:213-223.
- Haywood, M. D., and R. A. Kenyon. 2009. Habitat shifts by decapods—an example of connectivity across tropical coastal ecosystems. Pages 229-269 *Ecological connectivity among tropical coastal ecosystems*. Springer.

- Heales, D., D. Vance, and N. Loneragan. 1996. Field observations of moult cycle, feeding behaviour, and diet of small juvenile tiger prawns *Penaeus semisulcatus* in the Embley River, Australia. *Marine Ecology Progress Series* **145**:43-51.
- Heasman, M., and D. Fielder. 1977. The management and exploitation of the Queensland mud crab fishery [*Scylla serrata*]. Australian Fisheries.
- Heasman, M., D. Fielder, and R. Shepherd. 1985. Mating and spawning in the mudcrab, *Scylla serrata* (Forsk.) (Decapoda: Portunidae), in Moreton Bay, Queensland. *Marine and Freshwater Research* **36**:773-783.
- Helmke, S., N. Gribble, and S. Gould. 1998. Status of Guld of Carpentaria mud crab stocks: establishing a benchmark for Albatross Bay, Weipa. 22, Queensland Department of Primary Industries, Brisbane.
- Hilborn, R., and C. J. Walters. 2013. Quantitative fisheries stock assessment: choice, dynamics and uncertainty. Springer Science & Business Media.
- Hill, B. 1980. Effects of temperature on feeding and activity in the crab *Scylla serrata*. *Marine Biology* **59**:189-192.
- Hill, B. 1984. Aquaculture of the mudcrab. The Potential for Aquaculture in Queensland, Queensland Department of Primary Industries Publication:29-45.
- Hill, B. 1994. Offshore spawning by the portunid crab *Scylla serrata* (Crustacea: Decapoda). *Marine Biology* **120**:379-384.
- Hill, B. 1995. Offshore spawning by the portunid crab *Scylla serrata* (Crustacea: Decapoda). *Oceanographic Literature Review* **5**:399.
- Hillary, R., M. Bravington, T. Patterson, P. Grewe, R. Bradford, P. Feutry, R. Gunasekera, V. Peddemors, J. Werry, and M. Francis. 2018. Genetic relatedness reveals total population size of white sharks in eastern Australia and New Zealand. *Scientific reports* **8**:1-9.
- Holz, L., S. Kim, C. Petheram, G. Podger, J. Hughes, M. Kehoe, D. Aramini, S. Podger, J. Lerat, and P. Poulton. 2013. River system modelling for the Flinders and Gilbert Agricultural Resource Assessment case study analysis. A technical report to the Australian Government from the CSIRO Flinders and
- Hughes, J., A. Yang, B. Wang, S. Marvanek, L. Carlin, L. Seo, C. Petheram, and J. Vaze. 2017. Calibration of river system and landscape models for the Fitzroy, Darwin and Mitchell catchments. A technical report to the Australian Government from the CSIRO Northern Australia Water Resource Assessment, part of the National Water Infrastructure Development Fund: Water Resource Assessments.
- Hughes, J., A. Yang, B. Wang, S. Marvanek, L. Seo, C. Petheram, and J. Vaze. 2018a. River model simulation for the Fitzroy, Darwin and Mitchell catchments. A technical report to the Australian Government from the CSIRO Northern Australia Water Resource Assessment, part of the National Water Infrastructure Development Fund: Water Resource Assessments.
- Hughes, T., J. Kerry, and T. Simpson. 2018b. Large-scale bleaching of corals on the Great Barrier Reef. Wiley Online Library.
- Hutchings, J. 2002. Life Histories of Fish. Handbook of Fish Biology and Fisheries, vol. 1. Fish Biology. Blackwell Science, Oxford.
- Hutchings, J. A., R. A. Myers, V. B. García, L. O. Lucifora, and A. Kuparinen. 2012. Life-history correlates of extinction risk and recovery potential. *Ecological Applications* **22**:1061-1067.
- Hutton, T., R. A. Deng, E. Plaganyi, S. Pascoe, M. Miller, J. Upston, A. Punt, C. Moeseneder, T. Kompas, D. Sterling, and E. Lawrence. 2018. Northern Prawn Fishery Assessments 2015-18. Final Report. Report to the Australian Fisheries Management Authority – Project 2015/0811. Brisbane.
- Hutton, T., S. Pascoe, R. A. Deng, A. E. Punt, and S. Zhou. 2022. Effects of re-specifying the Northern Prawn Fishery bioeconomic model to include banana prawns. *Fisheries Research* **247**:106190.
- Hyland, S., B. Hill, and C. Lee. 1984. Movement within and between different habitats by the portunid crab *Scylla serrata*. *Marine Biology* **80**:57-61.
- IPCC. 2021. Climate Change 2021: The Physical Science Basis. Contribution of Working Group I to the Sixth Assessment Report of the Intergovernmental Panel on Climate Change [Masson-Delmotte, V., P. Zhai, A. Pirani, S.L. Connors, C. Péan, S. Berger, N. Caud, Y. Chen, L. Goldfarb, M.I. Gomis, M. Huang, K. Leitzell, E. Lonnoy, J.B.R. Matthews, T.K. Maycock, T. Waterfield, O. Yelekçi, R. Yu, and B. Zhou (eds.)]. Cambridge University Press.
- Jeffrey, S. J., J. O. Carter, K. B. Moodie, and A. R. Beswick. 2001. Using spatial interpolation to construct a comprehensive archive of Australian climate data. *Environmental Modelling & Software* **16**:309-330.

- Jerry R, D., C. Smith-Keune, L. Hodgson, I. Pirozzi, A. G. Carton, K. S. Hutson, A. K. Brazenor, A. Trujillo Gonzalez, S. Gamble, and G. Collins. 2013. Vulnerability of an iconic Australian finfish (barramundi-Lates calcarifer) and aligned industries to climate change across tropical Australia. Fisheries Research and Development Corporation.
- Kelleway, J. J., N. Saintilan, P. I. Macreadie, C. G. Skilbeck, A. Zawadzki, and P. J. Ralph. 2016. Seventy years of continuous encroachment substantially increases 'blue carbon' capacity as mangroves replace intertidal salt marshes. *Global change biology* **22**:1097-1109.
- Kelleway, J. J., O. Serrano, J. A. Baldock, R. Burgess, T. Cannard, P. S. Lavery, C. E. Lovelock, P. I. Macreadie, P. Masqué, M. Newnham, N. Saintilan, and A. D. L. Steven. 2020. A national approach to greenhouse gas abatement through blue carbon management. *Global Environmental Change* **63**:102083.
- Kenyon, R., C. Conacher, and I. Poiner. 1997a. Seasonal growth and reproduction of *Enhalus acoroides* (Lf) Royle in a shallow bay in the western Gulf of Carpentaria, Australia. *Marine and Freshwater Research* **48**:335-342.
- Kenyon, R., N. Loneragan, and J. Hughes. 1995. Habitat type and light affect sheltering behaviour of juvenile tiger prawns (*Penaeus esculentus* Haswell) and success rates of their fish predators. *Journal of Experimental Marine Biology and Ecology* **192**:87-105.
- Kenyon, R., N. Loneragan, J. Hughes, and D. Staples. 1997b. Habitat Type Influences the Microhabitat Preference of Juvenile Tiger Prawns (*Penaeus esculentus* Haswell and *Penaeus semisulcatus* De Haan). *Estuarine, Coastal and Shelf Science* **45**:393-403.
- Kenyon, R., C. Moeseneder, E. Plaganyi, M. Miller, and R. Buckworth. 2018. Proposed northern Australia water developments pertinent to the Northern Prawn Fishery: collation and review. FRDC Final Report, project no. 2016/015
- Knuckey, I. 1999. Mud crab (*Scylla serrata*) population dynamics in the Northern Territory, Australia and their relationship to the commercial fishery. Northern Territory University, Darwin.
- Koch, M., G. Bowes, C. Ross, and X. H. Zhang. 2013. Climate change and ocean acidification effects on seagrasses and marine macroalgae. *Global change biology* **19**:103-132.
- Kompas, T., and L. Chu. 2018. MEY for a short-lived species: A neural network approach. *Fisheries Research* **201**:138-146.
- Kyne, P. M., M. Oetinger, M. I. Grant, and P. Feutry. 2021. Life history of the Critically Endangered largetooth sawfish: a compilation of data for population assessment and demographic modelling. *Endangered Species Research* **44**:79-88.
- Larcombe, J., and A. Bath. 2017. Northern Prawn Fishery. Australian Bureau of Agricultural and Resource Economics and Sciences (ABARES).
- Lear, K. O., A. C. Gleiss, J. M. Whitty, T. Fazeldean, J. Albert, N. Green, B. C. Ebner, D. C. Thorburn, S. J. Beatty, and D. L. Morgan. 2019. Recruitment of a critically endangered sawfish into a riverine nursery depends on natural flow regimes. *Scientific reports* **9**:1-11.
- Lear, K. O., D. L. Morgan, J. M. Whitty, S. J. Beatty, and A. C. Gleiss. 2021. Wet season flood magnitude drives resilience to dry season drought of a euryhaline elasmobranch in a dry-land river. *Science of The Total Environment* **750**:142234.
- Lee, K.-S., S. R. Park, and Y. K. Kim. 2007. Effects of irradiance, temperature, and nutrients on growth dynamics of seagrasses: a review. *Journal of Experimental Marine Biology and Ecology* **350**:144-175.
- Lerat, J., C. Egan, S. Kim, M. Gooda, A. Loy, Q. Shao, and C. Petheram. 2013. Calibration of river models for the Flinders and Gilbert catchments. A Technical Report to the Australian Government from the CSIRO Flinders and Gilbert Agricultural Resource Assessment, Part of the North Queensland Irrigated Agriculture Strategy.
- Leslie, P., and D. Davis. 1939. An attempt to determine the absolute number of rats on a given area. *The Journal of Animal Ecology*:94-113.
- Li, F., E. Paraschivoui, and C. P. Dyt. 2006. Modelled seabed response to possible climate change scenarios over the next 50 years in the Australian North. CSIRO Report 06-002. 76 pps.
- Liu, H., and N. Loneragan. 1997. Size and time of day affect the response of postlarvae and early juvenile grooved tiger prawns *Penaeus semisulcatus* De Haan (Decapoda: Penaeidae) to natural and artificial seagrass in the laboratory. *Journal of Experimental Marine Biology and Ecology* **211**:263-277.
- Loneragan, N., and S. Bunn. 1999. River flows and estuarine ecosystems: implications for coastal fisheries from a review and a case study of the Logan River, southeast Queensland. *Australian Journal of Ecology* **24**:431-440.

- Loneragan, N. R. 1999. River flows and estuarine ecosystems: implications for coastal fisheries from a review and a case study of the Logan River, southeast Queensland. *Australian journal of ecology* **24**:431-440.
- Loneragan, N. R., R. A. Kenyon, D. J. Staples, I. R. Poiner, and C. A. Conacher. 1998. The influence of seagrass type on the distribution and abundance of postlarval and juvenile tiger prawns (*Penaeus esculentus* and *P. semisulcatus*) in the western Gulf of Carpentaria, Australia. *Journal of Experimental Marine Biology and Ecology* **228**:175-195.
- Loughnan, S. R., C. Smith-Keune, L. B. Beheregaray, N. A. Robinson, and D. R. Jerry. 2019. Population genetic structure of barramundi (*Lates calcarifer*) across the natural distribution range in Australia informs fishery management and aquaculture practices. *Marine and Freshwater Research* **70**:1533-1542.
- Lovelock, C. E., M. F. Adame, D. W. Butler, J. J. Kelleway, S. Dittmann, B. Fest, K. J. King, P. I. Macreadie, K. Mitchell, and M. Newnham. 2022. Modeled approaches to estimating blue carbon accumulation with mangrove restoration to support a blue carbon accounting method for Australia. *Limnology and Oceanography*.
- Lovelock, C. E., and C. M. Duarte. 2019. Dimensions of blue carbon and emerging perspectives. *Biology Letters* **15**:20180781.
- Lovelock, C. E., I. C. Feller, R. Reef, S. Hickey, and M. C. Ball. 2017a. Mangrove dieback during fluctuating sea levels. *Scientific reports* **7**:1-8.
- Lovelock, C. E., I. C. Feller, R. Reef, S. Hickey, and M. C. Ball. 2017b. Mangrove dieback during fluctuating sea levels. *Scientific Reports* **7**:1680.
- Lovett, R., M. O'Neill, and A. Garland. 2020. Stock assessment of Queensland east coast blue swimmer crab (*Portunus armatus*).
- Lowe, V., C. L. Frid, M. Venarsky, and M. A. Burford. 2021. Responses of a macrobenthic community to seasonal freshwater flow in a wet-dry tropical estuary. *Estuarine, Coastal and Shelf Science*:107736.
- Lyons, I., and M. Barber. 2021. Relatedness and co-existence in water resource assessments: Indigenous water values, rights and interests in the Mitchell catchment, North Queensland. *Australasian Journal of Water Resources*:1-13.
- MacArthur, R. H., and E. O. Wilson. 2016. *The theory of island biogeography*. Princeton university press.
- MacDonald, R. K. 2015. *Turbidity and light attenuation in coastal waters of the Great Barrier Reef*. James Cook University.
- Macintosh, A., M. Waschka, J. Jones, and A. Wood. 2018. Legal, regulatory and policy environment for development of water resources in northern Australia: Summary report. A technical report to the Australian Government from the CSIRO Northern Australia Water Resource Assessment, part of the National Water Infrastructure Development Fund: Water Resource Assessments. CSIRO, Australia.
- Mallen-Cooper, M., and B. P. Zampatti. 2020. Restoring the ecological integrity of a dryland river: Why low flows in the Barwon–Darling River must flow. *Ecological Management & Restoration* **21**:218-228.
- Manson, F. J., N. R. Loneragan, B. D. Harch, G. A. Skilleter, and L. Williams. 2005. A broad-scale analysis of links between coastal fisheries production and mangrove extent: A case-study for northeastern Australia. *Fisheries Research* **74**:69-85.
- McGregor, G. B., J. C. Marshall, J. S. Lobegeiger, D. Holloway, N. Menke, and J. Coysh. 2018. A risk-based ecohydrological approach to assessing environmental flow regimes. *Environmental management* **61**:358-374.
- McLeod, E., G. L. Chmura, S. Bouillon, R. Salm, M. Björk, C. M. Duarte, C. E. Lovelock, W. H. Schlesinger, and B. R. Silliman. 2011. A blueprint for blue carbon: toward an improved understanding of the role of vegetated coastal habitats in sequestering CO₂. *Frontiers in Ecology and the Environment* **9**:552-560.
- Meynecke, J.-O., M. Grubert, J. M. Arthur, R. Boston, and S. Y. Lee. 2012a. The influence of the La Niña-El Niño cycle on giant mud crab (*Scylla serrata*) catches in Northern Australia. *Estuarine, Coastal and Shelf Science* **100**:93-101.
- Meynecke, J.-O., M. Grubert, and J. Gillson. 2011. Giant mud crab (*Scylla serrata*) catches and climate drivers in Australia—a large scale comparison. *Marine and Freshwater Research* **63**:84-94.
- Meynecke, J.-O., M. Grubert, and J. Gillson. 2012b. Giant mud crab (*Scylla serrata*) catches and climate drivers in Australia—a large scale comparison. *Marine and Freshwater Research* **63**:84-94.
- Meynecke, J.-O., and S. Y. Lee. 2011. Climate-coastal fisheries relationships and their spatial variation in Queensland, Australia. *Fisheries Research* **110**:365-376.

- Meynecke, J.-O., S. Y. Lee, N. C. Duke, and J. Warnken. 2006. Effect of rainfall as a component of climate change on estuarine fish production in Queensland, Australia. *Estuarine, Coastal and Shelf Science* **69**:491-504.
- Meynecke, J.-O., and J.-O. Meynecke. 2010. Evaluating the environmental drivers of mud crab (*Scylla serrata*) catches in Australia. Fisheries Research and Development Corporation.
- Meynecke, J., S. Lee, M. Grubert, I. Brown, S. Montgomery, N. Gribble, D. Johnston, and J. Gillson. 2010. Evaluating the environmental drivers of mud crab (*Scylla serrata*) catches in Australia. Final Report 2002/012.
- Mobsby, D., A. Bath, and R. Curtotti. 2018. Fisheries: Outlook to 2022-23. *Agricultural Commodities* **8**:128-137.
- Morello, E. B., É. E. Plagányi, R. C. Babcock, H. Sweatman, R. Hillary, and A. E. Punt. 2014. Model to manage and reduce crown-of-thorns starfish outbreaks. *Marine Ecology Progress Series* **512**:167-183.
- Morgan, D., J. Whitty, N. Phillips, D. Thorburn, J. Chaplin, and R. McAuley. 2011. North-western Australia as a hotspot for endangered elasmobranchs with particular reference to sawfishes and the Northern River Shark. *Journal of the Royal Society of Western Australia* **94**:345-358.
- Muñiz-Castillo, A. I., A. Rivera-Sosa, I. Chollett, C. M. Eakin, L. Andrade-Gómez, M. McField, and J. E. Arias-González. 2019. Three decades of heat stress exposure in Caribbean coral reefs: a new regional delineation to enhance conservation. *Scientific Reports* **9**:11013.
- Myers, R. A. 1998. When do environment–recruitment correlations work? *Reviews in Fish Biology and Fisheries* **8**:285-305.
- Nagelkerken, I., S. Blaber, S. Bouillon, P. Green, M. Haywood, L. Kirton, J.-O. Meynecke, J. Pawlik, H. Penrose, and A. Sasekumar. 2008. The habitat function of mangroves for terrestrial and marine fauna: a review. *Aquatic botany* **89**:155-185.
- National Oceans Office. 2004. Living on Saltwater Country: Southern Gulf of Carpentaria. Sea Country Management, Needs and Issues.
- Northrop, A., M. F. O'Neill, and J. Robins. 2019. Towards an Initial Quota for the Queensland Mud Crab Fishery.
- O'Brien, C. 1994. Ontogenetic changes in the diet of juvenile brown tiger prawns *Penaeus esculentus*. *Marine ecology progress series*. Oldendorf **112**:195-200.
- O'Neill, M. F., A. B. Campbell, I. W. Brown, and R. Johnstone. 2010. Using catch rate data for simple cost-effective quota setting in the Australian spanner crab (*Ranina ranina*) fishery. *ICES Journal of Marine Science* **67**:1538-1552.
- Oliver, E. C., M. G. Donat, M. T. Burrows, P. J. Moore, D. A. Smale, L. V. Alexander, J. A. Benthuisen, M. Feng, A. S. Gupta, and A. J. Hobday. 2018. Longer and more frequent marine heatwaves over the past century. *Nature communications* **9**:1-12.
- Pascoe, S., L. Coglán, A. E. Punt, and C. M. Dichmont. 2012. Impacts of vessel capacity reduction programmes on efficiency in fisheries: the case of Australia's multispecies northern prawn fishery. *Journal of Agricultural Economics* **63**:425-443.
- Pascoe, S., T. Hutton, L. Coglán, and V. Q. Nguyen. 2018. Implications of efficiency and productivity change over the season for setting MEY-based trigger targets. *Australian Journal of Agricultural and Resource Economics* **62**:199-216.
- Pascoe, S., T. Hutton, É. Plagányi, R. A. Deng, M. Miller, C. Moeseneder, and S. Eves. 2020. Influence of environment and economic drivers on fishing effort in Australia's redleg banana prawn fishery. *Fisheries Research* **227**:105555.
- Pascoe, S. D., J. A. Sharp, and R. C. Buckworth. 2016. Modelling effort levels in a sequential fishery. *ICES Journal of Marine Science* **73**:503-511.
- Patterson, T. A., and R. D. Pillans. 2019. Designing acoustic arrays for estimation of mortality rates in riverine and estuarine systems. *Canadian Journal of Fisheries and Aquatic Sciences* **76**:1471-1479.
- Pauly, D. 1980. On the interrelationships between natural mortality, growth parameters, and mean environmental temperature in 175 fish stocks. *ICES Journal of Marine Science* **39**:175-192.
- Petheram, C., J. Hughes, C. Stokes, I. Watson, S. Irvin, D. Musson, S. Philip, C. Turnadge, P. Poulton, and L. Rogers. 2018a. Case studies for the Northern Australia Water Resource Assessment. A report to the Australian Government from the CSIRO Northern Australia Water Resource Assessment, part of the National Water Infrastructure Development Fund: Water Resource Assessments. CSIRO, Australia.
- Petheram, C., T. A. McMahon, and M. C. Peel. 2008. Flow characteristics of rivers in northern Australia: implications for development. *Journal of Hydrology* **357**:93-111.

- Petheram, C., P. Rustomji, T. R. McVicar, W. Cai, F. H. Chiew, J. Vleeshouwer, T. G. Van Niel, L. Li, R. G. Cresswell, and R. J. Donohue. 2012. Estimating the impact of projected climate change on runoff across the tropical savannas and semiarid rangelands of northern Australia. *Journal of Hydrometeorology* **13**:483-503.
- Petheram, C., P. Rustomji, and J. Vleeshouwer. 2009. Rainfall-runoff modelling across northern Australia. A report to the Australian Government from the CSIRO Northern Australia Sustainable Yields Project. CSIRO Water for a Healthy Country Flagship, Australia. 1835-095X.
- Petheram, C., I. Watson, C. Bruce, and C. Chilcott, (eds), . 2018b. Water resource assessment for the Mitchell catchment. A report to the Australian Government from the CSIRO Northern Australia Water Resource Assessment, part of the National Water Infrastructure Development Fund: Water Resource Assessments. CSIRO, Australia. Final Report. Prepared for the Department of the Environment and Energy.
- Petheram, C., I. Watson, and P. Stone, (eds). 2013a. Agricultural resource assessment for the Flinders catchment. A report to the Australian Government from the CSIRO Flinders and Gilbert Agricultural Resource Assessment, part of the North Queensland Irrigated Agriculture Strategy. CSIRO Water for a Healthy Country and Sustainable Agriculture flagships, Australia.
- Petheram, C., I. Watson, and P. Stone, (eds). 2013b. Agricultural resource assessment for the Gilbert catchment. A report to the Australian Government from the CSIRO Flinders and Gilbert Agricultural Resource Assessment, part of the North Queensland Irrigated Agriculture Strategy. CSIRO Water for a Healthy Country and Sustainable Agriculture flagships, Australia.
- Petheram, C., and A. Yang. 2013. Climate data and their characterisation for hydrological and agricultural scenario modelling across the Flinders and Gilbert catchments. CSIRO Water for a Healthy Country and Sustainable Agriculture flagships. A technical report to the Australian Government from the CSIRO Flinders and Gilbert Agricultural Resource Assessment, part of the North Queensland Irrigated Agriculture Strategy.
- Peeverell, S. C. 2005. Distribution of sawfishes (Pristidae) in the Queensland Gulf of Carpentaria, Australia, with notes on sawfish ecology. *Environmental Biology of Fishes* **73**:391-402.
- Peeverell, S. C. 2010. Sawfish (Pristidae) of the Gulf of Carpentaria, Queensland, Australia. James Cook University.
- Phillips, N. M., J. A. Chaplin, D. L. Morgan, and S. C. Peeverell. 2011. Population genetic structure and genetic diversity of three critically endangered *Pristis* sawfishes in Australian waters. *Marine Biology* **158**:903-915.
- Pillans, R., G. Fry, A. Steven, and T. Patterson. 2020. Environmental influences on long-term movement patterns of a euryhaline Elasmobranch (*Carcharhinus leucas*) within a subtropical estuary. *Estuaries and Coasts* **43**:2152-2169.
- Pillans, R., R. Hillary, and T. Patterson. in press. Assessing the effect of ongoing bycatch of sawfish in northern Australia. CSIRO Report for Department of the Environment and Energy, Australia.
- Plagányi, É., R. A. Deng, T. Hutton, R. Kenyon, E. Lawrence, J. Upston, M. Miller, C. Moeseneder, S. Pascoe, and L. Blamey. 2020. From past to future: understanding and accounting for recruitment variability of Australia's redleg banana prawn (*Penaeus indicus*) fishery. *ICES Journal of Marine Science*.
- Plagányi, É., J. Upston, M. Miller, J. Bishop, R. Deng, R. Buckworth, and T. Hutton. 2012. Assessment of the Joseph Bonaparte Gulf Red-legged Banana Prawn (*Penaeus indicus*) Fishery and preliminary TAC estimate for 2012. CSIRO Marine and Atmospheric Research technical report
- Plagányi, É. E. 2007. Models for an ecosystem approach to fisheries. Food & Agriculture Org.
- Plagányi, É. E., L. K. Blamey, J. G. Rogers, and V. J. Tulloch. 2022. Playing the detective: Using multispecies approaches to estimate natural mortality rates. *Fisheries Research* **249**:106229.
- Plagányi, É. E., and D. S. Butterworth. 2012. The Scotia Sea krill fishery and its possible impacts on dependent predators: modeling localized depletion of prey. *Ecological Applications* **22**:748-761.
- Plagányi, É. E., M. D. Haywood, R. J. Gorton, M. C. Siple, and R. A. Deng. 2019. Management implications of modelling fisheries recruitment. *Fisheries Research* **217**:169-184.
- Plagányi, É. E., A. E. Punt, R. Hillary, E. B. Morello, O. Thébaud, T. Hutton, R. D. Pillans, J. T. Thorson, E. A. Fulton, and A. D. Smith. 2014. Multispecies fisheries management and conservation: tactical applications using models of intermediate complexity. *Fish and Fisheries* **15**:1-22.
- Poiner, I., C. Conacher, N. Loneragan, and I. Sonters. 1993. Effects of cyclones on seagrass communities and penaeid prawn stocks of the Gulf of Carpentaria.
- Pollino, C., E. Barber, R. Buckworth, M. Cadiegues, A. Deng, B. Ebner, R. Kenyon, A. Liedloff, L. Merrin, C. Moeseneder, D. Morgan, D. Nielsen, J. O'Sullivan, R. Ponce Reyes, B. Robson, D. Stratford, B.

- Stewart-Koster, and M. Turschwell. 2018. Synthesis of knowledge to support the assessment of impacts of water resource development to ecological assets in northern Australia: asset descriptions. A technical report to the Australian Government from the CSIRO Northern Australia Water Resource Assessment, part of the National Water Infrastructure Development Fund: Water Resource Assessments. .
- Punt, A. E., R. Deng, S. Pascoe, C. M. Dichmont, S. Zhou, É. E. Plagányi, T. Hutton, W. N. Venables, R. Kenyon, and T. Van Der Velde. 2011. Calculating optimal effort and catch trajectories for multiple species modelled using a mix of size-structured, delay-difference and biomass dynamics models. *Fisheries Research* **109**:201-211.
- Punt, A. E., R. A. Deng, C. M. Dichmont, T. Kompas, W. N. Venables, S. Zhou, S. Pascoe, T. Hutton, R. Kenyon, and T. Van der Velde. 2010. Integrating size-structured assessment and bioeconomic management advice in Australia's northern prawn fishery. *ICES Journal of Marine Science* **67**:1785-1801.
- QLD. 2004. By-catch Action Plan 2004 For the Gulf of Carpentaria Indigenous, Recreational and Commercial Fisheries. Page 50 *in* Q. Government, editor.
- Ralph, P. J., M. J. Durako, S. Enríquez, C. J. Collier, and M. A. Doblin. 2007. Impact of light limitation on seagrasses. *Journal of Experimental Marine Biology and Ecology* **350**:176-193.
- Rastetter, E. B., J. D. Aber, D. P. C. Peters, D. S. Ojima, and I. C. Burke. 2003. Using Mechanistic Models to Scale Ecological Processes across Space and Time. *BioScience* **53**:68-76.
- Resources, D. o. A. a. W. 2018. Commonwealth Fisheries Harvest Strategy Policy, Canberra, June. CC BY 4.0. .
- Rikhter, V., and V. Efanov. 1976. On One of the Approaches to Estimation of Natural mortality of Fish Populations. *ICNAF Res. Doc.* 79/VI/8.
- Robertson, A., and S. Blaber. 1993. Plankton, epibenthos and fish communities. *Coastal and estuarine studies*:173-173.
- Robertson, A., and N. Duke. 1987. Mangroves as nursery sites: comparisons of the abundance and species composition of fish and crustaceans in mangroves and other nearshore habitats in tropical Australia. *Marine Biology* **96**:193-205.
- Robertson, A. I., and D. M. Alongi. 1992. Tropical mangrove ecosystems.
- Robertson, W. 2011. Abundance, population structure and size at maturity of *Scylla serrata* (Forsk.) (Decapoda: Portunidae) in Eastern Cape estuaries, South Africa. *South African Journal of Zoology* **31**:177-185.
- Robins, J., D. Mayer, J. Staunton-Smith, I. Halliday, B. Sawynok, and M. Sellin. 2006. Variable growth rates of the tropical estuarine fish barramundi *Lates calcarifer* (Bloch) under different freshwater flow conditions. *Journal of Fish Biology* **69**:379-391.
- Robins, J., and Q. Ye. 2007. Relationships between freshwater flow and fisheries (or secondary) production in Australian estuaries: a review. Department of Primary Industries and Fisheries.
- Robins, J. B., I. A. Halliday, J. Staunton-Smith, D. G. Mayer, and M. J. Sellin. 2005. Freshwater-flow requirements of estuarine fisheries in tropical Australia: a review of the state of knowledge and application of a suggested approach. *Marine and Freshwater Research* **56**:343-360.
- Robins, J. B., A. R. Northrop, M. Grubert, and R. C. Buckworth. 2020. Understanding environmental and fisheries factors causing fluctuations in mud crab and blue swimmer crab fisheries in northern Australia to inform harvest strategies.
- Rose, R., and T. Kompas. 2004. Management options for the Australian Northern Prawn Fishery: an economic assessment. ABARE.
- Rothlisberg, P. C., and M. A. Burford. 2016. Biological Oceanography of the Gulf of Carpentaria, Australia: A Review. Pages 251-260 *in* P. M. Glibert and T. M. Kana, editors. *Aquatic Microbial Ecology and Biogeochemistry: A Dual Perspective*. Springer International Publishing, Cham.
- Rothlisberg, P. C., and T. A. Okey. 2007. Variation in banana prawn catches at Weipa: a comprehensive regional study. Final Report. Technical report FRDC project no. 2004/024, CSIRO Marine and Atmospheric Science.
- Ruscoe, I. M., C. C. Shelley, and G. R. Williams. 2004. The combined effects of temperature and salinity on growth and survival of juvenile mud crabs (*Scylla serrata* Forskål). *Aquaculture* **238**:239-247.
- Russell, D., F. Thomson, P. Thuesen, T. Power, and R. Mayer. 2015. Variability in the growth, feeding and condition of barramundi (*Lates calcarifer* Bloch) in a northern Australian coastal river and impoundment. *Marine and Freshwater Research* **66**:928-941.

- Salini, J., S. Blaber, and D. Brewer. 1994. Diets of trawled predatory fish of the Gulf of Carpentaria, Australia, with particular reference to predation on prawns. *Marine and Freshwater Research* **45**:397-411.
- Salini, J., D. Brewer, and S. Blaber. 1998. Dietary studies on the predatory fishes of the Norman River Estuary, with particular reference to penaeid prawns. *Estuarine, Coastal and Shelf Science* **46**:837-847.
- Salini, J., R. McAuley, S. Blaber, R. Buckworth, J. Chidlow, N. Gribble, J. Oviden, S. Peverell, R. Pillans, and J. Stevens. 2007. Northern Australian sharks and rays: the sustainability of target and bycatch species, phase 2. CSIRO.
- Sara, L. 2010. Study on the size structure and population parameters of mud crab *Scylla serrata* in Lawele Bay, Southeast Sulawesi, Indonesia. *Journal of Coastal Development* **13**:133-147.
- Sasmito, S. D., P. Taillardat, J. N. Clendenning, C. Cameron, D. A. Friess, D. Murdiyarso, and L. B. Hutley. 2019. Effect of land-use and land-cover change on mangrove blue carbon: A systematic review. *Global Change Biology* **25**:4291-4302.
- Scott, A., and M. Rasheed. 2021. Port of Karumba Long-term Annual Seagrass Monitoring 2020, Centre for Tropical Water & Aquatic Ecosystem Research Publication Number 21/05, James Cook University, Cairns, 28pp.
- Shin, Y.-J., L. Shannon, and P. Cury. 2004. Simulations of fishing effects on the southern Benguela fish community using an individual-based model: learning from a comparison with ECOSIM. *African Journal of Marine Science* **26**:95-114.
- Short, F. T., R. G. Coles, and C. Pergent-Martini. 2001. Global seagrass distribution. *Global seagrass research methods* **5**:30.
- Simpfendorfer, C. A., P. M. Kyne, T. H. Noble, J. Goldsbury, R. K. Basiita, R. Lindsay, A. Shields, C. Perry, and D. R. Jerry. 2016. Environmental DNA detects Critically Endangered largemouth sawfish in the wild. *Endangered Species Research* **30**:109-116.
- Siple, M. C., T. E. Essington, and É. E. Plagányi. 2019. Forage fish fisheries management requires a tailored approach to balance trade-offs. *Fish and Fisheries* **20**:110-124.
- Smart, J. C. R., J. M. McMahon, G. Curwen, and M. Burford. 2021. Estimating the economic impact of potential extractions from Gulf rivers on the banana prawn fishery, Technical report, Griffith University, Brisbane.
- Somers, I., and Y. G. Wang. 1997. A simulation model for evaluating seasonal closures in Australia's multispecies northern prawn fishery. *North American Journal of Fisheries Management* **17**:114-130.
- Staples, D. 1980a. Ecology of juvenile and adolescent banana prawns, *Penaeus merguensis*, in a mangrove estuary and adjacent off-shore area of the Gulf of Carpentaria. I. Immigration and settlement of postlarvae. *Marine and Freshwater Research* **31**:635-652.
- Staples, D. 1980b. Ecology of juvenile and adolescent banana prawns, *Penaeus merguensis*, in a mangrove estuary and adjacent off-shore area of the Gulf of Carpentaria. II. Emigration, population structure and growth of juveniles. *Marine and Freshwater Research* **31**:653-665.
- Staples, D., and D. Vance. 1986. Emigration of juvenile banana prawns *Penaeus merguensis* from a mangrove estuary and recruitment to offshore areas in the wet-dry tropics of the Gulf of Carpentaria, Australia. *Marine Ecology Progress Series* **27**:52.
- State of Queensland. 2017. Queensland Sustainable Fisheries Strategy 2017-2027. Queensland Department of Agriculture and Fisheries, Brisbane. <https://www.daf.qld.gov.au/business-priorities/fisheries/sustainable/sustainable-fisheries-strategy/sustainable-fisheries-strategy-overview> [Accessed 19 December 2019].
- Stevens, J., R. McAuley, C. Simpfendorfer, and R. Pillans. 2008. Spatial distribution and habitat utilisation of sawfish (*Pristis* spp) in relation to fishing in northern Australia. A report to Department of the Environment, Water, Heritage and the Arts **31**.
- Stevens, J. D., R. Bonfil, N. K. Dulvy, and P. A. Walker. 2000. The effects of fishing on sharks, rays, and chimaeras (chondrichthyans), and the implications for marine ecosystems. *ICES Journal of Marine Science* **57**:476-494.
- Stokes, C., J. Addison, A. Macintosh, D. Jarvis, A. Higgins, A. Doshi, M. Waschka, J. Jones, W. Asmi, and N. Horner. 2017. Costs, benefits, institutional and social considerations for irrigation development. A technical report to the Australian Government from the CSIRO Northern Australia Water Resource Assessment, part of the National Water Infrastructure Development Fund: Water Resource Assessments. CSIRO, Australia.
- Streipert, S., J. Filar, J. Robins, and O. Whybird. 2019. Stock assessment of the barramundi (*Lates calcarifer*) fishery in Queensland, Australia. May 2019.

- Tanimoto, M., J. B. Robins, M. F. O'Neill, I. A. Halliday, and A. B. Campbell. 2012. Quantifying the effects of climate change and water abstraction on a population of barramundi (*Lates calcarifer*), a diadromous estuarine finfish. *Marine and Freshwater Research* **63**:715-726.
- Taylor, A., R. Doble, R. Crosbie, K. Barry, G. Harrington, P. Davies, and M. Thomas. 2018. Hydrogeological assessment of the Bulimba Formation - Mitchell catchment, Queensland. A technical report to the Australian Government from the CSIRO Northern Australia Water Resource Assessment, part of the National Water Infrastructure Development Fund: Water Resource Assessments. CSIRO, Australia.
- Thomson, J. A., D. A. Burkholder, M. R. Heithaus, J. W. Fourqurean, M. W. Fraser, J. Statton, and G. A. Kendrick. 2015. Extreme temperatures, foundation species, and abrupt ecosystem change: an example from an iconic seagrass ecosystem. *Global Change Biology* **21**:1463-1474.
- Trijoulet, V., G. Fay, and T. J. Miller. 2020. Performance of a state-space multispecies model: What are the consequences of ignoring predation and process errors in stock assessments? *Journal of Applied Ecology* **57**:121-135.
- Tulloch, V. J., É. E. Plagányi, R. Matear, C. J. Brown, and A. J. Richardson. 2018. Ecosystem modelling to quantify the impact of historical whaling on Southern Hemisphere baleen whales. *Fish and Fisheries* **19**:117-137.
- van der Velde, T. D., W. N. Venables, P. J. Crocos, S. Edgar, F. Evans, and P. C. Rothlisberg. 2021. Seasonal, interannual and spatial variability in the reproductive dynamics of *Penaeus merguensis*. *Marine Ecology Progress Series* **658**:117-133.
- Vance, D., M. Haywood, D. Heales, R. Kenyon, and N. Loneragan. 1998. Seasonal and annual variation in abundance of postlarval and juvenile banana prawns *Penaeus merguensis* and environmental variation in two estuaries in tropical northeastern Australia: a six year study. *Marine Ecology Progress Series* **163**:21-36.
- Vance, D., M. Haywood, D. Heales, R. Kenyon, N. Loneragan, and R. Pendrey. 1996. How far do prawns and fish move into mangroves? Distribution of juvenile banana prawns *Penaeus merguensis* and fish in a tropical mangrove forest in northern Australia. *Marine Ecology Progress Series* **131**:115-124.
- Vance, D., M. Haywood, D. Heales, R. Kenyon, N. Loneragan, and R. Pendrey. 2002. Distribution of juvenile penaeid prawns in mangrove forests in a tropical Australian estuary, with particular reference to *Penaeus merguensis*. *Marine Ecology Progress Series* **228**:165-177.
- Vance, D., M. Haywood, and D. Staples. 1990. Use of a mangrove estuary as a nursery area by postlarval and juvenile banana prawns, *Penaeus merguensis* de Man, in northern Australia. *Estuarine, Coastal and Shelf Science* **31**:689-701.
- Vance, D. J., and P. C. Rothlisberg. 2020. The biology and ecology of the banana prawns: *Penaeus merguensis* de Man and *P. indicus* H. Milne Edwards. Pages 1-139 *Advances in Marine Biology*. Elsevier.
- Vance, D. J., D. J. Staples, and J. D. Kerr. 1985. Factors affecting year-to-year variation in the catch of banana prawns (*Penaeus merguensis*) in the Gulf of Carpentaria, Australia. *ICES Journal of Marine Science* **42**:83-97.
- Venables, W. N., and C. M. Dichmont. 2004. A generalised linear model for catch allocation: an example from Australia's Northern Prawn Fishery. *Fisheries Research* **70**:409-426.
- Viney, N., J. Vaze, R. Crosbie, B. Wang, W. Dawes, and A. Frost. 2015. AWRA-L v5. 0: Technical description of model algorithms and inputs. 2015. CSIRO, Australia.
- Walters, C., and J.-J. Maguire. 1996. Lessons for stock assessment from the northern cod collapse. *Reviews in Fish Biology and Fisheries* **6**:125-137.
- Wang, Y.-G., and M. Haywood. 1999. Size-dependent natural mortality of juvenile banana prawns *Penaeus merguensis* in the Gulf of Carpentaria, Australia. *Marine and Freshwater Research* **50**:313-317.
- Wang, Y.-G., M. O'Neill, I. Brown, and J. McGilvray. 2012. Assessment of the utility of Queensland commercial mud crab logbook data.
- Warfe, D. M., N. E. Pettit, P. M. Davies, B. J. Pusey, S. Hamilton, M. J. Kennard, S. A. Townsend, P. Bayliss, D. Ward, and M. Douglas. 2011. The 'wet-dry' in the wet-dry tropics drives river ecosystem structure and processes in northern Australia. *Freshwater Biology* **56**:2169-2195.
- Wassenberg, T., and B. Hill. 1987. Natural diet of the tiger prawns *Penaeus esculentus* and *P. semisulcatus* [in northern Australian waters]. *Australian Journal of Marine and Freshwater Research (Australia)*.
- Wassenberg, T., and B. Hill. 1993. Diet and feeding behaviour of juvenile and adult banana prawns *Penaeus merguensis* in the Gulf of Carpentaria, Australia. *MARINE ECOLOGY-PROGRESS SERIES* **94**:287-287.

- Waycott, M., C. M. Duarte, T. J. Carruthers, R. J. Orth, W. C. Dennison, S. Olyarnik, A. Calladine, J. W. Fourqurean, K. L. Heck, and A. R. Hughes. 2009. Accelerating loss of seagrasses across the globe threatens coastal ecosystems. *Proceedings of the national academy of sciences* **106**:12377-12381.
- White, N. J., I. D. Haigh, J. A. Church, T. Koen, C. S. Watson, T. R. Pritchard, P. J. Watson, R. J. Burgette, K. L. McInnes, and Z.-J. You. 2014. Australian sea levels—Trends, regional variability and influencing factors. *Earth-Science Reviews* **136**:155-174.
- Whitty, J. M., J. Keleher, B. C. Ebner, A. C. Gleiss, C. A. Simpfendorfer, and D. L. Morgan. 2017. Habitat use of a Critically Endangered elasmobranch, the largetooth sawfish *Pristis pristis*, in an intermittently flowing riverine nursery. *Endangered Species Research* **34**:211-227.
- Williams, M., and B. Hill. 1982. Factors influencing pot catches and population estimates of the portunid crab *Scylla serrata*. *Marine Biology* **71**:187-192.
- Wolanski, E. 1993. Water circulation in the Gulf of Carpentaria. *Journal of Marine Systems* **4**:401-420.
- Wylie, L., A. E. Sutton-Grier, and A. Moore. 2016. Keys to successful blue carbon projects: Lessons learned from global case studies. *Marine Policy* **65**:76-84.
- Yan, H. F., P. M. Kyne, R. W. Jabado, R. H. Leeney, L. N. K. Davidson, D. H. Derrick, B. Finucci, R. P. Freckleton, S. V. Fordham, and N. K. Dulvy. 2021. Overfishing and habitat loss drive range contraction of iconic marine fishes to near extinction. *Science Advances* **7**:eabb6026.
- Zhou, S., R. C. Buckworth, N. Ellis, R. A. Deng, and S. Pascoe. 2015. Getting all information out of logbooks: estimating banana prawn fishable biomass, catchability, and fishing power increase, with a focus on natural mortality. *ICES Journal of Marine Science* **72**:54-61.
- Zhou, S., R. A. Deng, M. R. Dunn, S. D. Hoyle, Y. Lei, and A. J. Williams. 2021. Evaluating methods for estimating shark natural mortality rate and management reference points using life-history parameters. *Fish and Fisheries*.