nature sustainability

Article

https://doi.org/10.1038/s41893-023-01238-x

Integrated assessment of river development on downstream marine fisheries and ecosystems

Received: 12 January 2023

Accepted: 28 September 2023

Published online: 02 November 2023

Check for updates

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Demands on freshwater for human use are increasing globally, but water resource development (WRD) has substantial downstream impacts on fisheries and ecosystems. Our study evaluates trade-offs between WRDs and downstream ecosystem functioning considering alternative dam and water extraction options, diverse eco-hydrological responses and catchment-to-coast connectivity. We used a data-driven ensemble modelling approach to quantify the impacts of alternative WRDs. WRD impacts varied from weakly positive to severely negative depending on species, scenario and cross-catchment synergies. Impacts on fishery catches and the broader ecosystem (including mangroves) increased with catchment developments and volume of water removed, or if flow reduced below a threshold level. We found complex, linked-catchment dependence of banana prawns on flow and floods. Economic risks for this important fishery more than doubled under some scenarios. Sawfish emerged as the most sensitive across a range of WRD scenarios. Our findings highlight the need to consider marine ecosystems and fisheries to inform sustainable management of the world's remaining free-flowing rivers.

Globally, many marine species and habitats rely on freshwater river flows and palustrine, riverine or estuarine environments for some or all of their life-history stages, with follow-on effects for fisheries[1,2]. The critical importance of river flows and network connectivity is well documented for temperate rivers and coasts[2,3], to support freshwater fish[4], as well as biodiversity conservation[5]. Humans have fundamentally modified the terrestrial water cycle resulting in substantial impacts on drainage basins, river systems and land-to-ocean linkages[6], as well as on the world's commercial fisheries[7]. Yet, studies of how flow influences marine species and downstream fishery catches are less common for tropical rivers and coastal fisheries[7,8]. There is a paucity of models and coordinated planning to quantify downstream impacts due to alterations to natural flows to meet water needs for agriculture and other industries[9], especially at the inter-catchment scale, across multiple parts of the ecosystem (Fig. 1a) and at basin-wide scales[4]. The growing pressure on limited freshwater sources threatens the

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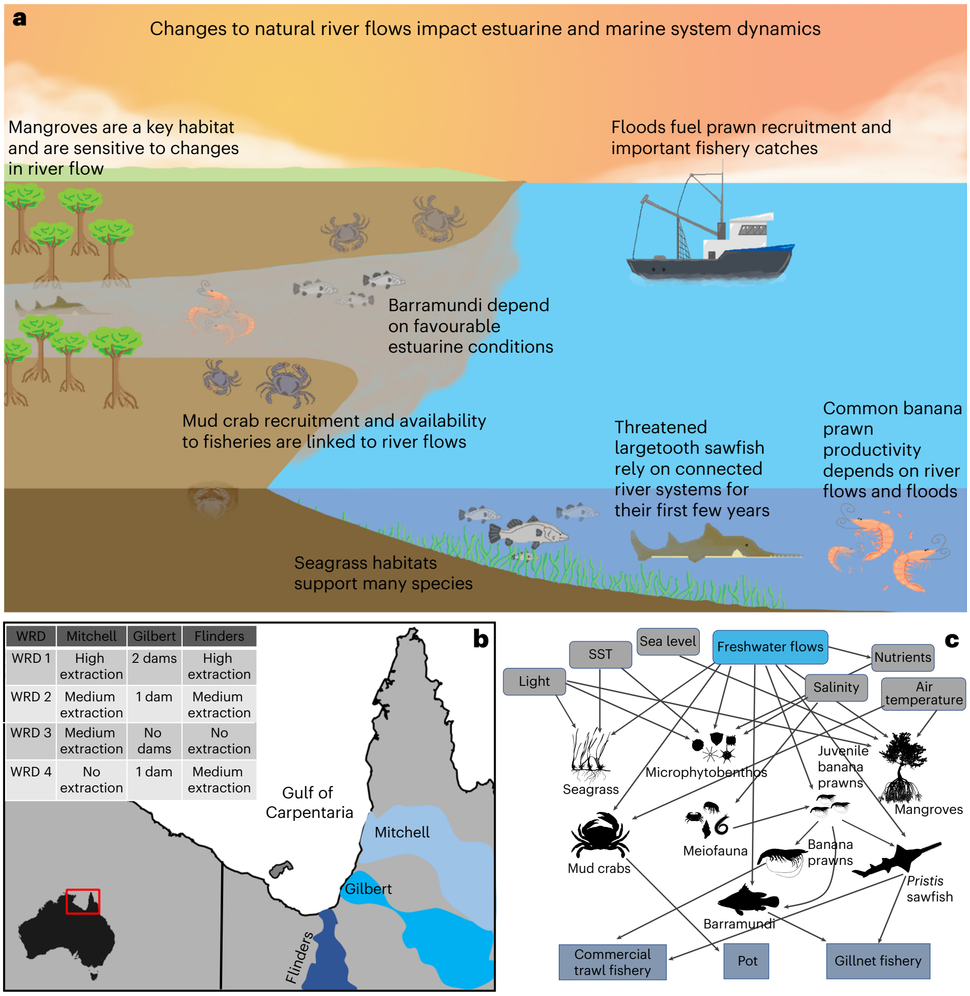

Fig. 1 | Influence of river flows on marine ecosystems and fisheries. **a**–**c**, Natural river-flow variability and periodic floods of rivers such as the Mitchell, Gilbert and Flinders rivers drive the productivity of marine species and habitats (**a**) in Australia's GoC, where alternative WRD scenarios have been scoped for these rivers (**b**), and a MICE used to quantify complex relationships between environmental variables such as flow and sea surface temperature (SST), and the recruitment, growth and survival of important fishery species, threatened species such as the largetooth sawfish, and supporting mangrove and seagrass habitats (**c**). Further details and sources of information are provided in Supplementary Information. Credit: mud crab (**c**), Freesvg.org; all other icons in **c**, PhyloPic. Illustration in **a** by James Chen.

future sustainability of not only river systems but also intricately connected estuarine and marine systems and the many livelihoods that depend on them. There is thus an urgent need for proactive, focused eco-hydrological studies rather than reactive approaches to socio-ecological disasters.

Water resource developments (WRDs)—such as dams and water extraction for agriculture—can disturb natural river flows, fragment river connectivity and associated ecosystem services, and modify flood-plains for aquatic species[10]. The negative impacts of large dams on the structure and functioning of downstream ecosystems have been documented by multiple studies[11,12]. Hydropower developments have also been flagged as in urgent need of rethinking to lessen disruptions to aquatic ecosystems and local livelihoods[4,13]. Increasing pressure on shared water resources[14], which may be exacerbated by climate change[15], means there is a pressing need for science to inform decision-making that balances human water needs between the socio-economic and ecological benefits that flowing rivers provide[16].

A study[4] shows that non-strategic dam-by-dam hydropower developments result in forgone ecosystem service benefits. In this study, a multi-objective optimization framework was proposed to evaluate trade-offs between energy production goals and environmental destruction. However, there is a concerning invisibility of the freshwater needs of marine ecosystems in most dam-planning processes. Our study addresses a need for proactive research before dam construction or before water extraction quotas are allocated, to serve as an assessment tool for informing trade-offs between upstream benefits and downstream biodiversity loss, habitat alteration and social and economic costs to existing food production industries (that is, fishers). Consideration of downstream impacts and how to mitigate them have lagged behind other issues. For example, a study[17] highlights that hydropower dam developers have not made sufficient efforts to compensate for the downstream social and environmental impacts of dams. Another study[16] points to a lack of transparency during dam approval processes and to dam construction projects often overestimating economic benefits and underestimating impacts on biodiversity and fisheries. More holistic assessments of new projects should incorporate basin-scale planning and evaluation of site selection to minimize biodiversity losses[16]. Based on a study of the socioecological impacts of a planned dam in the Brazilian Amazon[18], a call for more inclusive impact assessments was made given that the importance of tropical inland fisheries is often undervalued, despite them sustaining the livelihoods of millions of fishers and their families. Also, water management planning has mostly failed to effectively incorporate Indigenous values or account for the concept of cultural flows[19,20].

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Given the contentious nature of river developments and the need to quantify or compensate losses to downstream industries, a rigorous quantitative assessment of potential downstream impacts is desirable, particularly for transboundary river basins[21]. Moreover, there is a need to assess the combined effects on fish of multiple developments or cascades of dams[22], such that integrated basin-wide assessments are necessary to inform coordinated policies and strategies[4,23].

We investigated how upstream WRDs applied to multiple rivers can affect ecological functioning, biodiversity and livelihoods that are a considerable distance downstream, even extending into the ocean (Fig. 1a). We tested how much statistical support there is to infer likely impacts and trade-offs associated with alternative water resource planning (Fig. 1b). We analysed ways to lessen or mitigate impacts when considering a key set of representative species and habitats with species- and catchment-specific differences in eco-hydrological responses (Fig. 1c). We used end-of-system flow estimates from river system models to drive an ensemble of spatial multispecies Models of Intermediate Complexity for Ecosystem assessments (MICE)[24], an approach recognized as suited for informing operational decision-making[25]. Our tractable, integrated ecological model represents the population dynamics and dependence on freshwater flows of two habitat-forming groups (mangroves and seagrass) and four key indicator species (representing fisheries, recreation, cultural and conservation needs; Fig. 1a). For fishery species, we formally fitted the model to long data time series to estimate the relationship between changes in river flow and ecological functioning and fishery catches, as a basis to predict eco-hydrological responses under altered flow conditions. Our study goes beyond qualitative assessments and integrates data and ecological information from different fishery jurisdictions and resource sectors to quantify species-specific and catchment-specific responses to WRDs, account for uncertainty and test cumulative effects of multi-catchment WRDs. Moreover, we show the advantages of converting results into risk assessment metrics to inform decision-making.

Australia's Gulf of Carpentaria (GoC) is fed by multiple major river catchments (Extended Data Fig. 1) that are largely undeveloped in terms of freshwater storage and extraction. However, as with many of the world's remaining unregulated rivers, their catchments have been scoped for development, with the WRD alternatives currently hypothetical only[10,26,27]. Possible water resource infrastructure includes the placement of one or more public-resource dams with different storage capacities on a number of different river systems (Fig. 1b). In addition, legislated water allocations for irrigation within catchments could involve privately funded farm-scale water extraction via on-farm pumping and water storage (with different policy settings such as pump rate or flow threshold for the commencement and cessation of pumping).

These remote tropical regions contain several crustacean and fish species that inhabit estuaries during their life history and are high-value fishery species that are harvested seasonally[8,28,29]. There is marked variability in annual wet-season flows (Extended Data Fig. 2), which can, in turn, lead to low fishery catches with low associated economic value in years with low flows[8,28,30]. By contrast, the frequency of high-level wet-season flows ensures that bumper harvests occur regularly enough for the long-term economic viability of tropical fisheries[8,31]. In addition to the economic benefits of these fisheries, local Indigenous fishers rely on river flows for sustenance, and to maintain cultural practices and ecological knowledge developed over thousands of years[32]. As well, recreational fishers target species such as barramundi (*Lates calcarifer*) and giant mud crabs (*Scylla serrata*)[33]. Within-catchment irrigated agriculture has the capacity to reduce and modify the natural-flow regimes of these wet and dry tropical rivers, and hence modify the populations and fishery catch of several high-value fish and crustacean species. We therefore aimed to quantify the impacts and risks to the GoC coastal and nearshore ecosystems of WRDs, applied to three large river catchments with scoped WRD scenarios available, namely, the Mitchell, Flinders and Gilbert River catchments (Fig. 1b). This situation is emblematic of the global need for proactive, rigorous approaches to inform trade-offs and account for downstream livelihoods and dependent estuarine and marine systems when planning WRDs.

Results

River-flow influences on ecology and fishery catches

We successfully fitted five alternative MICE (Fig. 1c and Supplementary Fig. 1)—the ensemble approach (Extended Data Fig. 3)—to long-term environmental and fishery data for each of eight spatial regions. We found a statistically significant improved fit to past fishery catches when predicting these based on fishing effort and flow variability, compared with fishing effort alone (Fig. 2a). We focused on overall quality of model fits rather than over-parameterizing the model by improving the fits to each model region (Fig. 2a). By fitting to available data that captured past variability in fishery catches attributable to changes in historical unregulated flow (Fig. 2b and Supplementary Figs. 17–21), we were able to estimate statistically the parameters of functional forms (logistic and dome-shaped relations) describing how river flow influences fishery recruitment, survival and catchability (Extended Data Fig. 4, Supplementary Section 5 and Supplementary Table 8). A flow multiplier was computed for every week (or month) of every year before being applied to the relevant recruitment or population processes, noting, for example, that the timing of spawning and recruitment varies seasonally (Extended Data Fig. 2b,c). The ensemble was used to capture additional uncertainty in these eco-hydrological relationships (Extended Data Fig. 4) and to bound a range of alternative plausible representations for the more data-poor species and habitat groups. Eco-hydrological relationships were shown to explain considerable past interannual variability (Fig. 3), as well as intra-annual variability such as occurs most obviously between wet and dry years (Extended Data Fig. 5 and Supplementary Figs. 23 and 24).

For common banana prawns (*Penaeus merguiensis*), the most parsimonious model (lowest Akaike information criterion (AIC); Supplementary Table 8) was the version incorporating a flood-induced productivity effect based on historical research within the currently unregulated catchments, suggesting that nutrient inputs from floods fuel estuarine primary productivity[34]. However, sediment trapped by upstream infrastructure may have additional major impacts on estuarine geomorphology and productivity[35] in ways beyond the scope of this study.

River portfolio influences prawn recruitment and catches

The MICE quantified a river portfolio effect across four regional rivers: the Mitchell, Gilbert, Norman and Flinders rivers, and was able—through fitting to data—to estimate the relative contributions of the different rivers in explaining observed prawn catches (Fig. 2c and Supplementary Figs. 14–16). Hence, we found that WRDs applied to a single river or different combinations of rivers had complex cumulative and synergistic effects on prawn abundance and catches (Fig. 2c). This effect was significant in the south-east region of the GoC, where these rivers are adjacent to one another (Fig. 1b). This result arose owing to our approach quantitatively translating recruitment fluctuations per region into overall contributions to total catch in each subregion of the GoC. Our model was able to estimate with adequate statistical rigour (Supplementary Table 8 and Supplementary Fig. 16) the relative contributions of different spatial regions to explaining subregion total observed catches.

Estimating influences of altered flows

Changes from baseline flows due to WRDs (Fig. 2b and Extended Data Fig. 6) had variable impacts on all species and catchment regions, ranging from minor through to extreme under some scenarios (Fig. 4). Overall, we found that model-predicted catchment-system impacts increased with the greater volume of water extracted or

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Fig. 2 | Quantifying how river flows influence fishery catches. a,b, Observed variability (black points) in the total annual commercial catch (t) of common banana prawns, barramundi and mud crabs from each of five catchment systems in the GoC; blue lines show the best model fits achievable when estimating catches based on fishing effort but ignoring river flows, compared with the improved ability of the model to fit the observed catches (red lines) (**a**) when linking end-of-system flow variability to the system dynamics, using weekly or monthly flow inputs with an example plot showing interannual variability in flows (**b**). The Mornington region was assumed influenced by the adjacent Flinders River flows. **c**, For prawns, the model also estimates the importance of different rivers influencing recruitment (to the fishery) per model region based on flow anomalies contributed by the different river systems. The model estimates relative contributions that are bounded between 0 and 1 (that is, results suggest no effect of a river on that region's prawn recruitment, or varying influences; values shown are model version 5 estimates; see Supplementary Table 8 for associated standard deviations). Hence, for example, model results suggest that the Norman River is the dominant driver of prawns caught directly offshore of the Norman River model region, with some contribution from the Flinders River. The Norman River is also estimated to be an important driver of prawn catches in the neighbouring Gilbert River model region. By contrast, prawn catches in the Mitchell and Flinders model regions are predicted to be driven on average by a combination of flow anomalies across all four river systems (see Supplementary Table 8 for details of model fits). Credit: species icons, PhyloPic.

impounded and the number of rivers on which dams were deployed. Across all modelled species, water extraction (that is, 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 (Table 1, Extended Data Fig. 7, Supplementary Figs. 25 and 26, and Supplementary Tables 14–19). Results for several species suggested that limiting water extraction to higher river-flow levels than ecosystem-sustaining flow thresholds and extracting water from short-duration peak flows may reduce the impacts of anthropogenic use.

Biomass and catches of the common banana prawn were predicted to decrease by 4% to 40% depending on the extent of water extraction from the Mitchell, Gilbert and Flinders rivers (Table 1 and Fig. 4a,b). The MICE predicted that local and regional decreases in prawn abundance and catches were larger if accounting for the flood-induced productivity effect (Supplementary Table 9).

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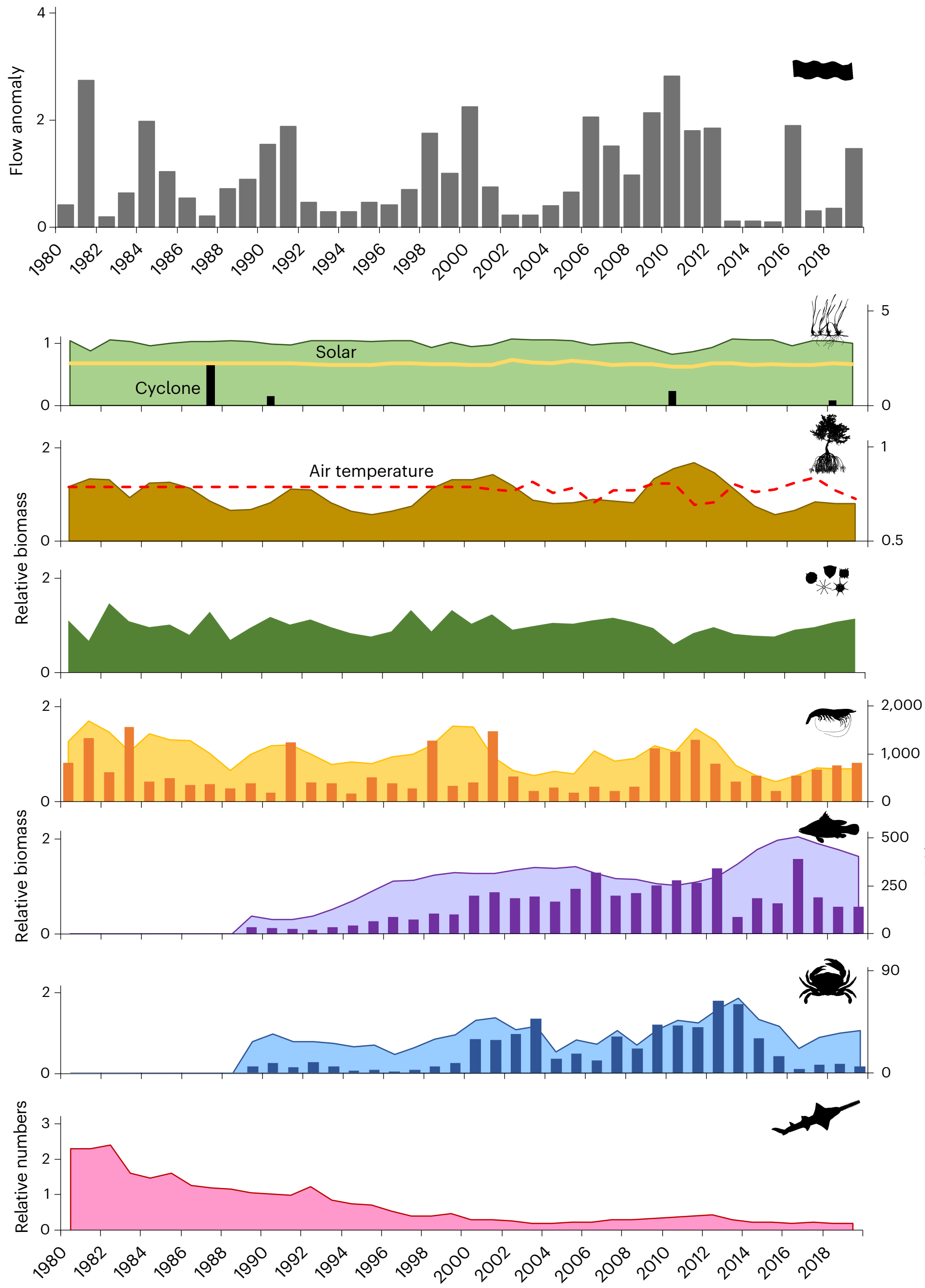

Fig. 3 | Influence of flow and environmental variables on population and fishery responses. Example of MICE physical drivers and changes in the relative biomass and catch (tonnes) of the base model groups for region 5 (Flinders River) from 1980 to 2019. From the top panel to the bottom panel: flow, seagrass, mangroves, microphytobenthos, common banana prawns, barramundi, mud crabs and sawfish. Biomass shown as shaded area and catch as vertical bars for fished species. The relative influences of cyclones (black bars) and solar radiation (yellow line) on seagrass are shown by superimposing these physical variables on the seagrass biomass trajectory. Changes in air temperature (red dashed line) that can affect mangroves are also shown. Credit: species icons, PhyloPic.

Our integrated model, incorporating the complex non-obligatory catadromous life history of barramundi[36] and naturally variable mud crabs, extends previous studies linking growth and catches of barramundi[31,36] and mud crab[28] to rainfall or river flow. For barramundi, we found that biomass and catch decreased by 4–61% under WRDs 1–4 (Table 1 and Fig. 4). With the exception of perennial rivers such as the Mitchell River (and Roper), our study predicted substantial influences of WRDs on mud crabs, with catch decreasing up to 83% in some years (Table 1 and Fig. 4a,b).

Although uncertain due to a lack of historical data, ensemble results consistently supported the notion that WRDs have the potential to cause large declines of largetooth sawfish (*Pristis pristis*) in all catchments relative to historical levels (Fig. 4). Model results were robust to alternative model structures that included explicit representation of the dependence of barramundi and largetooth sawfish on declines in estuarine prey abundance (using common banana prawns as a proxy), albeit this could slightly worsen predicted impacts of WRDs in some scenarios (Supplementary Table 19).

Model results suggested that WRDs may cause large declines in mangrove abundance in affected catchments (Fig. 4a). In contrast to all other MICE groups, seagrasses were predicted to marginally increase in abundance under some WRDs (up to 7% relative to base levels), with minor impacts (up to a 9% decline) across most scenarios (Table 1). When comparing the relative impact of different water

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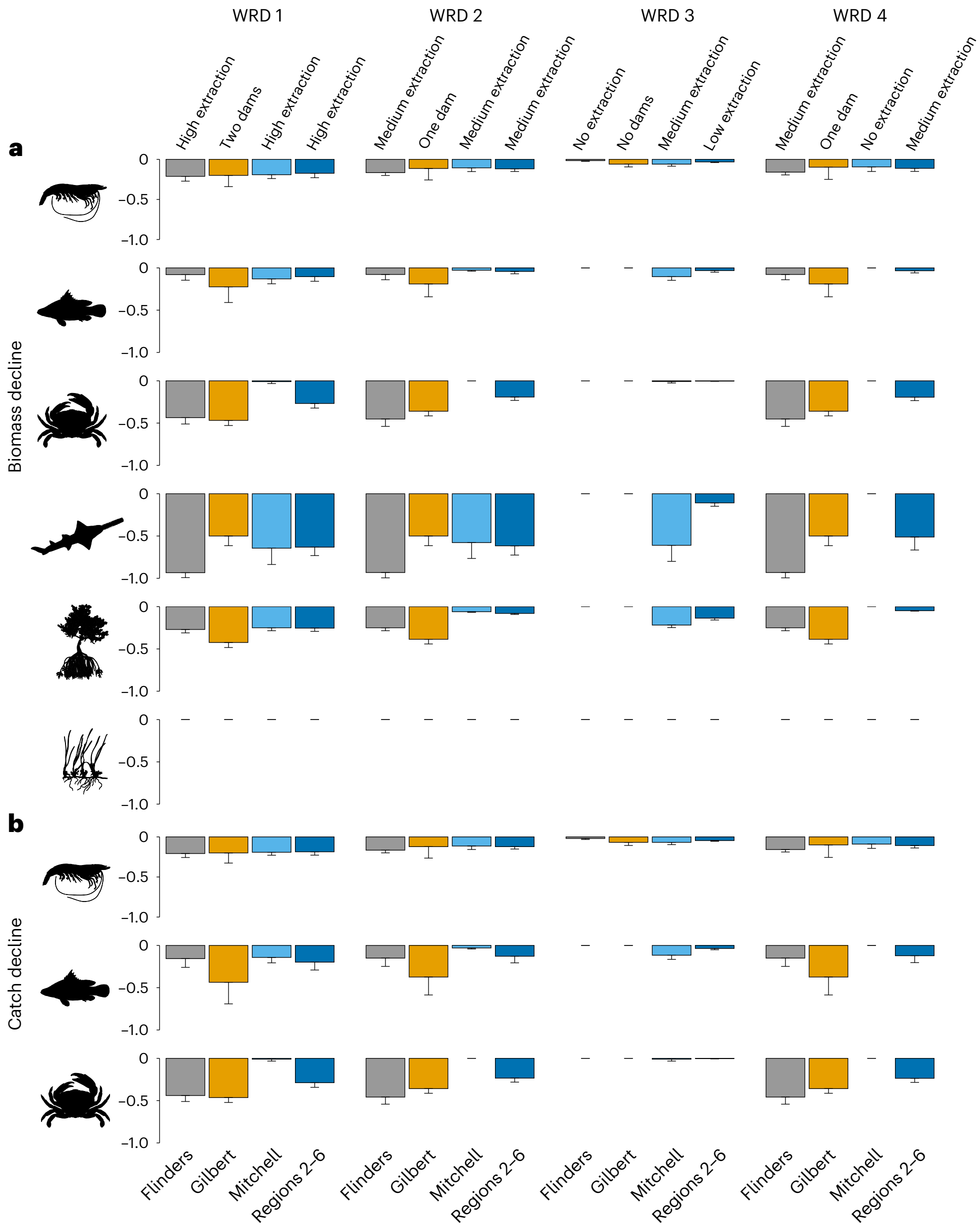

Fig. 4 | Modelled ecosystem responses to illustrative WRD scenarios. a, MICE ensemble average decline (relative to the baseline flow scenario, with standard deviation shown as error bars) of biomass indicators for (from top to bottom) common banana prawns, barramundi, mud crabs, largetooth sawfish (numbers), mangroves and seagrass under alternative illustrative water development scenarios (Fig. 1b) applied to three river systems and shown for the south-east subsection of the GoC. **b**, MICE ensemble average decline in units of fishery catches, which generally were of similar magnitude to biomass, although relatively larger in the case of barramundi. The sample size for each bar is $n = 155$. Credit: species icons, PhyloPic.

extraction scenarios using the Mitchell River as an example, higher extractions had a more negative impact on fishery catches, sawfish numbers and mangrove biomass (Extended Data Fig. 6a–d). However, the same water allocation amount had a more negative impact when using a lower river-flow pumping threshold value in all cases (Extended Data Fig. 6a–d). For prawns, barramundi and mangroves, the medium allocation scenarios with a low pump-initiation threshold setting had a more negative impact than a scenario with double the water allocation by volume but a high threshold value. Sawfish had worse outcomes for higher extraction amounts although outcomes were also sensitive to the river-flow threshold setting (Extended Data Fig. 6d). For prawns, the worst estimated outcomes were for scenarios that also included WRDs applied to the Flinders and Gilbert rivers (Extended Data Fig. 6a), consistent with the model estimates of a river portfolio effect (Fig. 2c). This highlights the important role of the threshold setting as a mitigation measure, including the need to consider cumulative regional impacts of WRDs.

Ecological and economic risk assessments

Our risk assessment (Supplementary Table 12) classified WRD 1 (highest water allocation and multi-catchment WRD) as the highest-risk scenario with moderate to intolerable risks predicted for all species and habitat groups except seagrass, both in terms of population-level risk and fishery risk (Fig. 5a,b). This was followed by WRD 2 and WRD 4 (lower water allocation or single-catchment WRD), both of which also predicted high risks to some populations and fisheries. WRD 3 emerged as the least-risky scenario; this scenario had no development (relative to base) on the Flinders and Gilbert rivers but had development on the Mitchell River (Table 1). Sawfish were predicted to show the greatest sensitivity to WRDs (owing to their low-productivity life-history

Table 1 | Key WRD scenario combinations

WRD	Properties of WRD scenarios: Number of rivers affected	Dams (yield (Gl yr^{-1}))	Allocation amount (Gl yr^{-1})	Threshold for pumping (Mitchell)	Pump rate (Mitchell)	Median (90% range) of annual regional catch (t) difference and frequency of lost catch (%) (and catchment with greatest difference): Common banana prawn (t)	Barramundi (t)	Giant mud crab (t)
Base	None	None	0	–	–	Observed (since 1971): 1,580 t	Observed (since 1989): 329 t	Observed (since 1989): 108 t
WRD 1	3 (Mitchell, Flinders, Gilbert)	2 on Gilbert (498 Gl yr^{-1})	High (Mitchell: 2,000 Gl yr^{-1}; Flinders: 400 Gl yr^{-1})	Low	Low	−363 t (−855 t; −168 t) Frequency (loss): 100% (Flinders)	−63 t (−117 t; −7 t) Frequency (loss): 92% (Gilbert)	−25 t (−79 t; 2 t) Frequency (loss): 88% (Flinders)
WRD 2	3 (Mitchell, Flinders, Gilbert)	1 on Gilbert (172 Gl yr^{-1})	Medium (Mitchell: 1,000 Gl yr^{-1}; Flinders: 160 Gl yr^{-1})	High	High	−263 t (−662 t; −109 t) Frequency (loss): 100% (Flinders)	−34 t (−74 t; 1 t) Frequency (loss): 83% (Gilbert)	−19 t (−69 t; 2 t) Frequency (loss): 90% (Flinders)
WRD 3	1 (Mitchell)	None	Medium (Mitchell: 1,000 Gl yr^{-1})	Low	High	−134 t (−303 t; −59 t) Frequency (loss): 98% (Mitchell)	3 t (−22 t; 56 t) Frequency (loss): 79% (Mitchell)	0 t (−1 t; 0.4 t) Frequency (loss): 35% (Mitchell)
WRD 4	2 (Flinders, Gilbert)	1 on Gilbert (172 Gl yr^{-1})	Medium (Flinders: 160 Gl yr^{-1})	NA	NA	−225 t (−603 t; −99 t) Frequency (loss): 100% (Flinders)	−28 t (−68 t; 6 t) Frequency (loss): 81% (Gilbert)	−20 t (−69 t; 2 t) Frequency (loss): 90% (Flinders)

WRDs tested for the Mitchell, Gilbert and Flinders rivers relative to 'Base' (baseline flows including any existing water development). Also shown are MICE ensemble median annual catch loss or increase (t), with 5th and 95th percentiles: 90% of catch difference in this range, relative to 'Base' (row shows median observed catch (t)), for model regions 2–6 combined; frequency of years for which catch loss was estimated; and catchment with greatest decline. Gl, gigalitres, which is the total volume of water extracted per year. NA, not applicable. See Supplementary Table 11 for a description of all 19 WRDs tested.

characteristics[37]) with risks ranked as extreme across a broad range of alternative WRDs (Fig. 5). Our economic risk assessment for common banana prawns suggested that the risk of an uneconomic 'bad' catch year for industry may more than double under some WRD scenarios (Extended Data Fig. 8).

Discussion

We found species-specific and catchment-specific differences in how flow modifies downstream ecology, with alterations to flow resulting in impacts that varied from weakly positive to severely negative depending on the species and scenario. We predicted the highest sensitivity for critically endangered[37] largetooth sawfish and also found that some WRDs have substantial negative impacts on important fishery catches and habitat-forming species. Moreover, our modelling quantified the critical ecological role of floods in enhancing aquatic productivity[34]. Use of a statistical ecosystem model fitted to empirical data enabled quantifying for the first time the relative contributions of a portfolio of rivers in explaining observed marine fishery prawn catches. A river portfolio effect whereby a portfolio of rivers collectively reduces the interannual variability of returns by the Bristol Bay sockeye fishery has previously been shown in a previous study[38]. Our quantitative estimates advance on previous research[8,39] that hypothesized that juvenile banana prawns from the Mitchell River may be transported in a southerly direction because tidal and wind-driven currents acting in concert with salinity-driven (due to rivers) currents move water in a southerly direction until the trapped coastal water mass is ejected offshore during summer[8,40]. Our finding that a combined portfolio of rivers act to 'stabilize' or maintain the GoC common banana prawn population underscores the need to quantify cumulative impacts that result from multiple developments across catchments. We identify a need for coordinated WRD planning across multiple catchment systems based on our estimates of the individual contributions of a set of adjacent catchments that collectively contribute to the recruitment success of a connected prawn population. Our modelling thus suggests that reducing river flows from one or more catchments will have complex synergistic rather than simple additive effects on the common banana prawn population.

We converted model outputs to ecological (Fig. 5) and economic (for prawns) risk statistics. This highlights development combinations that are high risk or not sustainable to the downstream ecology and fisheries. The risk assessment can inform on preferred water storage or extraction settings (Extended Data Fig. 8) and location choices that may assist in offsetting regional impacts.

The world's rapidly growing population means demand for food and pressures on natural ecosystems are ever increasing, such that integrated cross-sector planning and management is required[5,41]. Our results highlight the conflicting needs to use rivers to support land-based agriculture versus downstream ecosystem function, fisheries, flow-dependent species and coastal habitats[42]. Our findings underscore the need for coupling marine and freshwater scientific understanding and approaches to improve infrastructure planning and flow management.

Our spatial MICE linking river flows and estuarine and marine systems focuses on key species and processes (Fig. 1c) to provide a reliable basis to predict how these ecosystem components are likely to respond to different types, locations and combinations of WRDs. This tailored modelling approach facilitated capturing species-specific differences in eco-hydrological responses to changes in flow, which is important for understanding how alternative WRDs may impact system biodiversity, sustainability and productivity as well as dependent livelihoods. By fitting to actual long-term fishery catch data, we were able to not only validate how future changes in flow might impact fishery yields, but also separate the relative influences of different anthropogenic activities (that is, fishing and WRDs) on natural resources. We drew on available information on the complex life history of largetooth sawfish to quantify the impact of alternative WRDs, and model results suggested that sawfish may be particularly sensitive to WRDs (Fig. 5) and hence may be a high-priority species for more detailed assessment. Our model showed that there are significant differences in response among catchments for downstream ecology impacted by WRDs (Fig. 4). This could inform trade-off decisions around locations and types of WRDs.

We found that long-term impacts of water extraction influenced fish and crustacean species to varying degrees depending on the extent and nature of the WRD. Threshold settings allow the pumping of freshwater to commence once a certain rate of flow of water has flowed past the most downstream gauge. Lower threshold settings increase the reliability for potential users to extract an allocation of water upstream but substantially reduce the amount of water that flows to the end of the system[26]. In addition, lower total pump capacity necessitates

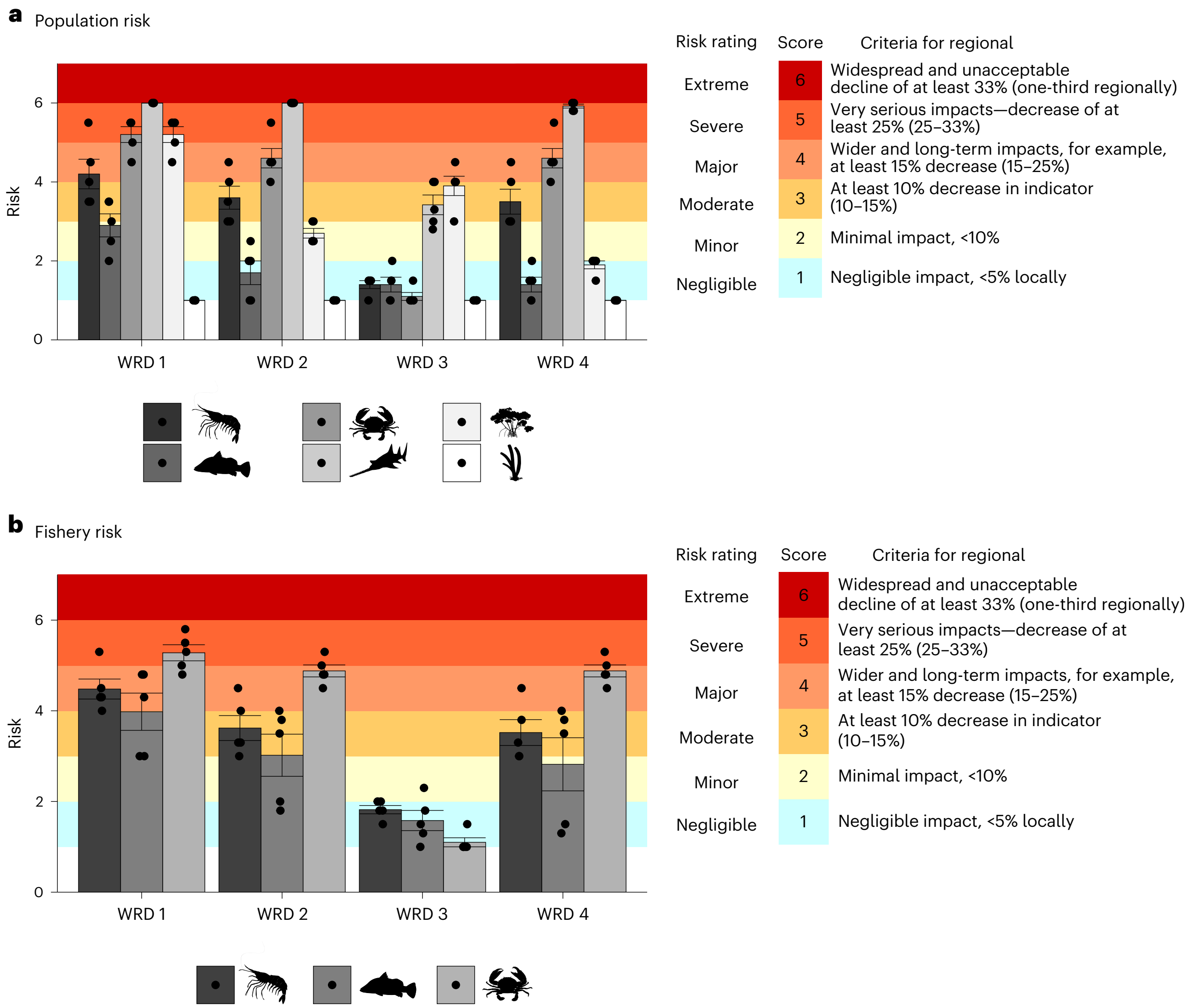

Fig. 5 | Quantifying the risks to marine species and fishery catches of WRD. **a**,**b**, Comparison of average (with standard error deviations) population risk (**a**) and fishery risk (**b**) using the MICE ensemble to evaluate the impact of alternative WRDs on species and habitat groups as shown. Corresponding data points are overlaid as dot plots with a sample size $n = 5$ for each bar. Risk ratings correspond to regional decline categories as shown. Credit: species icons, PhyloPic.

water extraction from flows other than peak flows, resulting in a higher proportion of non-peak flows being extracted. We found that pump routines that disturb the pattern of river flow during low-level flows have a proportionately larger impact on downstream ecosystem service provision because the system is already under stress when flows are low (Extended Data Fig. 7). The need to maintain flows well above an ecosystem-sustaining minimum has previously been recognized[43], and our study advances approaches to quantify river-flow threshold settings below which ecological functioning becomes compromised.

At the opposite extreme, during periods of extremely high flow or floods, our study accentuates that large volumes of water flowing out to sea are not wasted water and provides rigorous substantiation of previous research in GoC estuaries showing the long-term benefits to ecosystem productivity that result from ongoing primary productivity boosts reliant on natural floods[34]. Our results suggested that WRDs that dampen floods have both an immediate and longer-term negative influence on fishery yields as well as on sawfish[37].

Rivers support habitats such as mangroves that are sensitive to water development, highlighting the need to consider intricate connections between marine and freshwater ecosystems rather than basing management decisions solely on sector-specific analyses. Model results suggesting mangroves may be sensitive to WRDs require empirical validation, but they point to the need to consider the potential impacts of WRDs on habitat-forming species, as well as the need for quantitative monitoring. Habitat impacts predicted for the GoC may be exacerbated owing to the extreme seasonality of freshwater inputs (Extended Data Fig. 2a) and the harsh coastal environment.

Modelled seagrass increases were attributed to improved nearshore light penetration—critical for seagrass[44]—in response to lower flows reducing sediment loads and water turbidity[45].

Our risk assessments under a range of WRD scenarios highlight situations when the need to balance competing uses is critical; otherwise, water resource exploitation tips the ecosystem towards decline or results in fishery ruin. Instances include observed collapses of fish and fisheries[46,47], including prawn fisheries that have narrow salinity tolerances[48] and rely on enhanced estuarine and coastal productivity from naturally flowing rivers[34,49]. Understanding impacts and ways to mitigate such impacts can help decision makers determine ecologically responsible and equitable WRD.

Extensions to our study, including adding additional species and fishery sectors (Indigenous and recreational), were 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[19,20,50], but we encourage future studies to consider these aspects. Water management planning developed with genuine Indigenous partnerships, drawing on Indigenous expertise of ecological resilience and adaptive management, can improve the equity and effectiveness of water planning[19,51].

The approach used was tailored to current exploration of WRDs across northern Australia but has global relevance showing the following to achieve a balance between ongoing water development and environmental sustainability: (1) the need to proactively 'quantify and make transparent' downstream impacts of a range of alternative WRDs (such as the number, location and size of development, dam or water extraction type, and operational settings), (2) the need for rigorous integrated approaches such as MICE (which draw on established methods used in fisheries and ecosystem modelling that are coupled with river system models that can reliably quantify impacts and trade-offs to inform decision-making), (3) representation of key species and habitats to capture the likely variability in responses and impact types (for example, biodiversity, conservation values, fishery catches, fishery viability), (4) use of risk assessment metrics to effectively communicate which development proposals or settings likely pose unacceptable risks to the ecosystem and (5) consideration of land-to-sea connections as well as cross-catchment synergies and cumulative impacts to inform a holistic assessment of trade-offs between the development of one sector influencing the sustainability of others. Whereas few systems have long-term time-series data as used in this study, we also provide examples of applying the approach to more data-poor species and habitat groups, and promote the use of an ensemble approach to bound uncertainties. In addition, the intermittent nature of low-level dry-season flows in the wet and dry tropics may render these rivers more vulnerable to WRD than perennial rivers in other regions, although WRDs can substantially reduce flow through usually perennial river mouths[52].

Conserving natural ecosystems while optimizing human needs is a challenging task that can only be accomplished using integrated approaches that consider the catchment-to-coast continuum and cumulative impacts on supporting ecosystems. To achieve the goals of equity and sustainability of the United Nations Decade of Ocean Science for Sustainable Development[53], it is essential to reliably quantify and predict impacts from multiple stressors on ocean ecosystems, to quantify connections between land and sea, and to develop solutions for equitable and sustainable development, biodiversity conservation and food production.

Methods

Overview of study site and ecology

The GoC tropical rivers (Extended Data Fig. 1) are mostly non-perennial with a short wet season (Extended Data Fig. 2) and large estuaries as major components of the coastal tropical ecosystem[10,54]. They are critical habitats for key life-history stages of many species, including globally threatened largetooth sawfish[37] and commercially important common banana prawns[55], giant mud crabs[28] and barramundi[31].

Connectivity between freshwater and marine systems, and the dependence of marine species on the brackish estuarine environments where they intersect, have received extensive attention in tropical northern Australia[15,29,34,39,56,57]. Earlier studies used methods such as regression analyses[39] and linear statistical approaches to investigate relationships between flow and downstream prawn biomass[8], and otolith biochronology to quantify the relationship between river flow and barramundi growth rates[31]. Our study draws on this research to provide an integrated framework to quantify relationships between flow, population abundance and productivity, extending across catchments and jurisdictions, to provide a holistic basis for evaluating WRD impacts.

River system models and WRD scenarios

End-of-system flow outputs (hereafter referred to as flow) represent streamflow estimates that are the outputs from river models[27] at a river's estuary. Freshwater from catchments flows out to the sea, and here we use flow at the most downstream node in the river model to drive population dynamics based on underlying hypotheses. The GoC has multiple catchments, and we used natural-flow model estimates (available from 1900) for the key major rivers: Mitchell, Gilbert, Flinders, Norman, Embley and Roper (Supplementary Table 10). We assumed the Mornington region's system dynamics were also driven by Flinders River flows. Although the GoC river flows are all highly seasonal (Extended Data Fig. 2a), the Mitchell River is perennial with continuous, albeit low, flows throughout the year compared with the greater dry-season variability shown by the Gilbert, Norman and Flinders rivers (Supplementary Fig. 8).

Although there are other rivers in the GoC, we assumed that the key rivers were an adequate proxy for the variability in flow in all of the spatial regions in our integrated ecosystem model because they contribute an average of 65% to total GoC river flows (Supplementary Fig. 9). Their combined interannual flow patterns are generally synchronous with total GoC flow volumes (Supplementary Fig. 10). We focused in particular on three large rivers (Mitchell, Flinders, Gilbert) that have been scoped for potential WRDs because the other rivers, not subject to extensive WRD, will continue to 'perform' naturally with temporal variability in flow regimes (Supplementary Section 4).

To bound the potential impacts of a range of alternative WRDs, we used WRD-altered model flow estimates from 19 alternative scenarios (Supplementary Table 11) for the Mitchell, Gilbert and Flinders rivers. We selected an illustrative range of WRD scenarios from a larger set that have been scoped, to bound the problem by quantifying effects ranging from intensive development scenarios to more restricted development (Fig. 1b). We also incorporated examples representing possible dams as well as water extraction whereby different amounts may be allocated for purposes such as pumping for irrigation. 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. Of the 19 WRD scenarios run, we focused on four key scenarios for contrast, clarity and concise discussion (Table 1). These range from volumes of water allocated above which annual reliability becomes problematic (WRD 1) 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 (WRD 2), a scenario (WRD 3) with moderate extraction from the Mitchell and no WRDs on the Flinders and Gilbert, and a scenario (WRD 4) with no WRD on Mitchell River and moderate extraction on Flinders combined with a single dam on the Gilbert River (Table 1 and Fig. 1b). The WRD scenarios remain hypothetical, and no WRDs of similar scale are currently planned.

Stakeholder consultation

Our approach was developed in consultation with stakeholders via a series of workshops. The first in-person workshop in August 2019 was attended by a diverse group of 31 representatives, including scientists and fishery managers, fishing industry representatives, water managers and community representatives[58]. The subsequent teleconference workshops included three species-specific workshops and a final stakeholder workshop in August 2021 as COVID-19 prevented in-person meetings, visits and further consultation with local communities[58]. We first developed a conceptual model (Fig. 1a) to inform a quantitative MICE (Fig. 1c) built in a stepwise fashion[59]. The key species (groups) identified to be explicitly represented (with varying levels of complexity) in the model were common banana prawns, barramundi, giant mud crabs, largetooth sawfish and aggregate groups: meiofauna, microphytobenthos, mangroves and seagrass (Fig. 1c). The key species represent important ecological, conservation, commercial and cultural interests, and encompass a range of dependencies on freshwater and estuarine systems and river flows (Fig. 1a). The MICE therefore focuses on this subset that are intimately linked to the estuaries and near coastal region, and are the focus of key regional fisheries.

Development of a spatial ecological MICE

We developed a MICE[24,25] as a multispecies assessment tool (Supplementary Methods, Supplementary Fig. 1 and Supplementary Tables 1–6). Our MICE is an integrated model, meaning it uses all available data in a single analysis and has likelihood functions that allow for the propagation of uncertainty to final model outputs[60]. Context and question driven, MICE focus only on ecosystem components required to quantify specific impacts of WRDs (and potentially mitigation and alternative solutions; Fig. 1c). Stakeholder participation and dialogue are an integral part of this process[24].

We tailored equations for each model species and group based on available data and life-history characteristics[24]. To align with data availability, we used a weekly time step for prawns, but monthly time steps for barramundi, mud crabs and sawfish. We extended population dynamics equations to explicitly account for various ways in which environmental drivers influence parameters in time (that is, we modelled both intra-annual and interannual variability) and space (spatial structure; Supplementary Table 1).

The full set of mathematical equations, variable and parameter definitions, and input values for all model species groups are provided in Supplementary Tables 1–6. As the key species are not all trophically linked, this was not a key focus of the model, but we explored sensitivity to plausible trophic interactions, such as both sawfish and barramundi being predators of common banana prawns. We did not explicitly represent consumption but assumed that changes in prey abundance translate into changes in predator survival or growth rate (Supplementary Section 10).

MICE estimate parameters through fitting to data, use statistical diagnostic tools to evaluate model performance and account for a broad range of uncertainties[24,25]. The MICE was used to estimate parameters describing eco-hydrological relationships and to evaluate alternative functional forms (Extended Data Fig. 3). To bound uncertainties, we used an ensemble comprising five model versions with different parameter and structural uncertainty, and averaged final ensemble results. We also conducted additional sensitivity tests to gauge the robustness of model results to alternative plausible assumptions.

Linking flow, population dynamics and fishery catches

For each of the key model species, we developed relationships to describe the influence of flow on recruitment, survival and availability (to the fishery). The timing of peak spawning and breeding for each species was based on available literature (Extended Data Fig. 2 and Supplementary Table 3). Weekly flow totals were standardized relative to the average (1970 to 2019) for that same week (Supplementary Table 1). Most fishery stock assessment models either assume that recruitment is a constant or estimate recruitment residuals. The latter represent the differences between expected recruitment (based on an underlying functional form describing how recruitment is related to population spawning biomass) and the actual observed differences. These differences are attributed to some (unknown) environmental driver. We test direct use of flow anomalies as recruitment residuals—in other words, we test whether changes in flow can help explain the past observed variability in recruitment of key species (Fig. 3).

Previous studies[39,57] have suggested that there are lower threshold flow values below which population responses become nonlinear, whereas for very large flows, there is an upper limit constrained by life history. We used a logistic or parabolic function to describe the relationship between flow and population processes such as recruitment (equations (4a) to (4c) in Supplementary Table 1). The underlying data or fixed parameter settings determine whether the relationship is near linear or increases more steeply after some lower threshold level (Extended Data Fig. 4, Supplementary Section 5 and Supplementary Figs. 12 and 13).

For prawns, we computed modified weekly flow-influence values using an equation with flexibility to estimate the relative contributions of each of the four major catchments to their own model region as well as the other three regions (Supplementary Section 4). The MICE either estimated zero contribution of adjacent catchments in explaining fishery catches per model region or substantiated the contributions of changes in flow in adjacent regions to influence prawn catches in a spatial region, supporting the hypothesis that there is a cross-catchment river portfolio effect (Supplementary Section 6). To bound uncertainty, we included a structurally different model version in the ensemble that assumed no cross-catchment connectivity for prawns (Supplementary Table 7). For barramundi, mud crabs and sawfish, we assumed that the dynamics in each of the model spatial regions were influenced only by local catchment flows.

Given limited data, we based the modelling of sawfish on research describing boom–bust dynamics of sawfish in response to natural variability in rainfall[61]. Hence, we developed sawfish recruitment equations that depended on flow and yielded recruitment booms (or busts) under high (or low) flows (equation (2i) in Supplementary Table 1). We also accounted for additional mortality whenever flow dropped below a threshold level, to capture the relatively higher mortality (for example, due to thermal stress, dehydration, predation pressure[61]) associated with shallower or disconnected river refuge pools in low-flow years[61] (equation (1h) in Supplementary Table 1).

Flow impacts on key habitat

There were limited data available at the spatial and temporal scale for mangroves and seagrass, so the model ensemble incorporated alternative parameter settings and assumptions (Supplementary Table 7), drawing on previous observations of mangrove and seagrass cover in response to changes in river flows and other factors (Supplementary Figs. 2–5).

Seagrass are submerged and sensitive to reductions in light availability[45]. Intense rainfall and run-off events lead to sedimentation and resuspension, thereby reducing light penetration and causing seagrass decline[45,62]. Based on studies[45,62] showing a negative relationship between light attenuation and flows, we assumed an inverse relationship between flow and the light attenuation term[45] in the growth rate equation (equations (8a) and (10c) in Supplementary Table 4).

For mangroves, we applied standardized weekly flow-influence values (equation (2) in Supplementary text) to the growth rate (equations (8b) and (10d) in Supplementary Table 4). In addition, our mesoscale mangrove community sub-model was based on research[63] suggesting that the vegetative cover of mangroves increases with increasing average annual rainfall (and hence flow) and that periodic changes in rainfall trends can result in encroachment or die-back of mangroves[63]. We captured this effect by modelling mangrove carrying capacity as a function of average flow over the preceding year (equation (8c) in Supplementary Table 4).

Flow, salinity and floods

We used changes in river flows as a proxy for salinity stress[57,63] and directly included salinity as a variable influencing the growth rate of microphytobenthos and meiofauna[57] (equations (10e) and (10f) in Supplementary Table 4). Large floods result in severe scouring whereby microphytobenthos is flushed out into coastal waters, enhancing subsequent productivity[34]. We used microphytobenthos biomass as a proxy to simulate how nutrient inputs from floods fuel estuarine primary productivity[34] (Supplementary Fig. 6). Based on existing research[34,57], we assumed (in one model version) that the natural mortality of prawns was inversely proportional to relative changes in the influx of scoured microphytobenthos biomass to estuaries (equation (1f) in Supplementary Table 1 and equation (10f) in Supplementary Table 4). We defined large floods as flows exceeding three times the average flow. We used this formulation to test whether accounting for a so-called flood productivity boost effect would provide a better explanation of past observed prawn catches, and included this in ensemble model version 5 (Supplementary Table 7).

Model fitting and use of an ensemble to capture uncertainty

We coded the model in AD Model Builder[64] and used maximum-likelihood techniques to fit the ecosystem model, analogous to methods used in fisheries stock assessment modelling[60]. We fitted the model to extensive data for key fishery species including a 50 year weekly catch-and-effort time series for common banana prawns and 30 year monthly time-series data for barramundi and mud crabs (Fig. 2), disaggregated into eight spatial regions (Extended Data Fig. 1). For example, for prawns, the model likelihood contribution was based on observed versus model-predicted catch for each of weeks 13 to 22 (the main fishing period) for 1970 to 2019 (equation (7b) in Supplementary Table 1). For mud crabs, the model was fitted to male-only catches for Queensland model regions versus catches derived from both sexes for the Northern Territory (equation (7d) in Supplementary Table 1). For barramundi for which (some) age composition data were available, we were also able to add a likelihood contribution based on how well the model fitted the age composition data (Supplementary Figs. 20 and 21, and equations (7e) to (7i) in Supplementary Table 1).

Through fitting to these data, the MICE can rigorously quantify how altering river flows may influence system productivity and fishery catches, plus how floods and system connectivity influence outcomes. To reduce uncertainty in model structure and parameterization, and because data were limited for some species, we used an ensemble comprising five MICE with alternative parameterizations and structural assumptions and used RStudio to analyse and plot model outputs (Supplementary Fig. 1 and Supplementary Table 7). Supplementary Table 8 shows the associated fixed and model-estimated parameters, together with Hessian-based standard deviations. The fitted relationships between flow and population abundance and fishery catches were possible because we used data that informed observed historical changes in response to changing environmental conditions (Fig. 3), as well as intra-annual differences between wet and dry years (Extended Data Fig. 5 and Supplementary Figs. 23 and 24).

Quantifying the influences of anthropogenic changes to flow

For the fishery species, the best-fitting eco-hydrological model parameter estimates were fixed when the model was re-run with different WRD scenarios; that is, the model predicted how population abundance and catches would change had the past flows been reduced according to the altered flow estimates under alternative WRDs. We estimated catchment-specific historical changes in abundance (and catches of some species) in response to baseline flows as well as under 19 alternative WRD scenarios encompassing different combinations of hypothetical water extraction and/or dam placements for the Mitchell, Flinders and Gilbert rivers (Table 1 and Supplementary Table 11) and assuming no anthropogenic alterations to the flows of the other GoC catchments.

Ecological and economic risk assessments

We defined population and fishery risk respectively based on average declines in abundance and catches (under WRDs relative to baseline flows) mapped to risk categories we defined (Supplementary Section 8 and Supplementary Table 12). We quantified economic risks to the common banana prawn sub-fishery of the Northern Prawn Fishery (NPF) under alternative WRDs by computing the relative probability of occurrence of major risks (defined as risk of a 'bad' year of 2,000 t catch corresponding to very poor economic return), severe risk (two successive 'bad' years) and extreme risk (fishery operations becoming unviable due to three or more consecutive 'bad' years; Supplementary Methods Section 8).

Capturing key system dynamics and acknowledging shortcomings

The MICE advances previous approaches but has limitations. It does not fully represent connectivity between all GoC regions, nor explicitly model the oceanographic and wind-driven dynamics[65]. It does not include several other species that are trophically linked to our key species[66], and we do not account for species and life-history differences of the habitat-forming groups. We used relatively simple relationships to represent complex mechanistic processes[24]. We did not represent secondary impacts from WRDs and agriculture, such as potential increases in nutrient and sediment loads (turbidity) from disturbed and tilled soils[45]. Increased sediment loads could pose multiple problems, including risk to estuarine biota through smothering of gills and reduced viable habitat area[67,68]. Once fine sediments enter the estuarine system, their tidal resuspension causes irreparable changes in seasonally turbid–clear systems to permanently turbid systems[67]. Irrigated agriculture, and its associated run-offs of nutrients, herbicides, pesticides, fungicides and pollutants[69], and future changes in precipitation, including extreme events, may also increase the risk of eutrophication leading to downstream impacts[42,70]. The river models did not consider broader effects such as disrupting migration routes or changing sediment loads[12,67]. Thus, the risk to the ecosystem could be greater than predicted (and permanent) and in ways other than modelled by the MICE. At present, there is little to no irrigated development in the study area catchments although current land uses (pastoral industries and limited mining) are likely to have altered water quality relative to a pre-European state.

As more observational data become available—particularly for the GoC shallow coastal waters—it may be possible to reduce this uncertainty. The MICE captured first-order effects that drive changes in population dynamics before adding secondary effects in a stepwise fashion to evaluate whether they substantially improved the model's predictive ability[59]. We did not include detailed mechanistic and other processes in the model if we did not have a basis to validate or inform these additions.

In the context of the existing water resource infrastructure and licensing, the WRDs examined herein are relatively large and would signify notable changes to current management practices for these catchments[26,27]. Impacts of smaller development scenarios are reported in Supplementary Tables 14–18. Ecosystem impacts may be slightly overestimated if there are large, neighbouring unmodified rivers that also influence regional dynamics (Supplementary Fig. 9). Furthermore, other than pumping thresholds, the WRDs do not consider mitigating strategies such as operation of sophisticated environmental flow rules[10]. As none of the WRDs have been implemented, we did not assume additional impacts of WRDs such as species migration barriers or reservoir retention times[22]. Future work should further explore trade-offs between agriculture and fishery industries, as well as conservation, including the discovery of the best mitigation strategies that minimize fishery impacts while adequately supporting agricultural development.

Ethics

The project did not involve any collection of human data, small group notes or interviews. The project workshops were used to inform local researchers and stakeholders and invite feedback on locally relevant research. Participants were recruited based on experience and network connections for the key fisheries as well as key local researchers and water managers, drawing on long-term involvement in the region's fisheries and research by the project team, several of whom have formal representative roles on local fisheries management advisory committees as detailed further in the Nature Portfolio Reporting Summary.

Reporting summary

Further information on research design is available in the Nature Portfolio Reporting Summary linked to this article.

Data availability

The data for the river system models are available at https://nawra-river.shinyapps.io/river/. Permission to obtain the raw fishery data needs to be granted by the relevant data custodians: Commonwealth Australian Fisheries Management Authority and NPFI (under co-management arrangements, NPFI is the delegate) for NPF data, as well as relevant state fisheries departments from both the Queensland and Northern Territory jurisdictions for barramundi and mud crabs. Data access contacts and request numbers can be provided on request to É.P., and raw model data input files will also be provided subject to relevant data agreements being in place. The environmental driver datasets are publicly available (mostly derived from http://www.bom.gov.au/), and our collated time series, together with the cyclone history and impact scores we developed, are available at https://www.dropbox.com/sh/yirnujdmv22qgoe/AAAyDrOET1j6YkGbAFUbqIiva?dl=0. Source data are provided with this paper.

Code availability

The code for the river system models is available at https://nawra-river.shinyapps.io/river/.

The following model output files are available at https://www.dropbox.com/sh/yirnujdmv22qgoe/AAAyDrOET1j6YkGbAFUbqIiva?dl=0.

The file flow_std.csv shows the standardized weekly flow (flowstd) and model 1 flow multiplier (prod) for all weeks and years since 1971 and all areas.

The files MonthFlow.csv and MonthFlow_mudcrab.csv are as above but with monthly standardized flow and the model 1 flow multipliers for barramundi and mud crab, respectively.

For model ensemble versions 1 to 5 and all species and groups, model output.csv files show the change in abundance and catches relative to the base case for WRDs 1 to 4.

Using the model output files as above, Microsoft Excel file Risk_stats_ensemble_working.xls shows the calculations used in the risk assessment.

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Acknowledgements

We thank the very many scientists for their hard work and numerous earlier studies that provided the foundation of our study. We thank all workshop attendees from a range of sectors at the four project workshops for their ideas, discussions and suggestions. In particular, we thank the following people for discussions: S. Pascoe, P. Rothlisberg, D. Stratford, C. Petheram, L. Dutra, T. van der Velde, A. Broadley and J. Smart. We thank T. Patterson, A. Hobday, R. Bustamante, J. Rogers and R. Buckworth for providing constructive comments on the paper. We thank Griffith University for providing the coastal environmental productivity data (microphytobenthos and meiofauna data). We also thank the Department of Agriculture and Fisheries (Queensland) and the Department of Industry, Tourism and Trade (Northern Territory) for their provision of the barramundi and mud crab commercial data, and we thank NPF Industry Pty Ltd (NPFI) and AFMA for granting use of the NPF commercial data. We thank J. Chen for producing Fig. 1a. Funding was provided by Fisheries Research Development Corporation grant 2018-079 (E.P.), with co-funding provided by the CSIRO.

Author contributions

Conceptualization: É.P., R.K., A.J., T.H., M.B., J.R., R.P.; data curation: M.M., R.A.D., C.M.; methodology (data analysis): R.A.D., M.M., R.K., L.B., É.P., T.H., C.M.; methodology (modelling analysis): É.P., L.B., R.A.D.; methodology (river models): J.H., S.K.; stakeholder engagement and workshop convening: É.P., T.C.; visualization: L.B., R.A.D., É.P.; funding acquisition: É.P.; project administration: É.P.; writing—original draft: É.P., R.K., L.B.; writing—review and editing: J.R., M.B., R.P., T.H., J.H., S.K., R.A.D., T.C., A.J., A.L., E.L., M.M., C.M.

Funding

Open access funding provided by CSIRO Library Services.

Competing interests

A.J. is the CEO of NPF Industry Pty Ltd (NPFI), and A.L. was (until the end of 2021) NPFI project manager. The other authors declare that they have no competing interests.

Additional information

Extended data is available for this paper at https://doi.org/10.1038/s41893-023-01238-x.

Supplementary information The online version contains supplementary material available at https://doi.org/10.1038/s41893-023-01238-x.

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Peer review information *Nature Sustainability* thanks Thiago Couto and the other, anonymous, reviewer(s) for their contribution to the peer review of this work.

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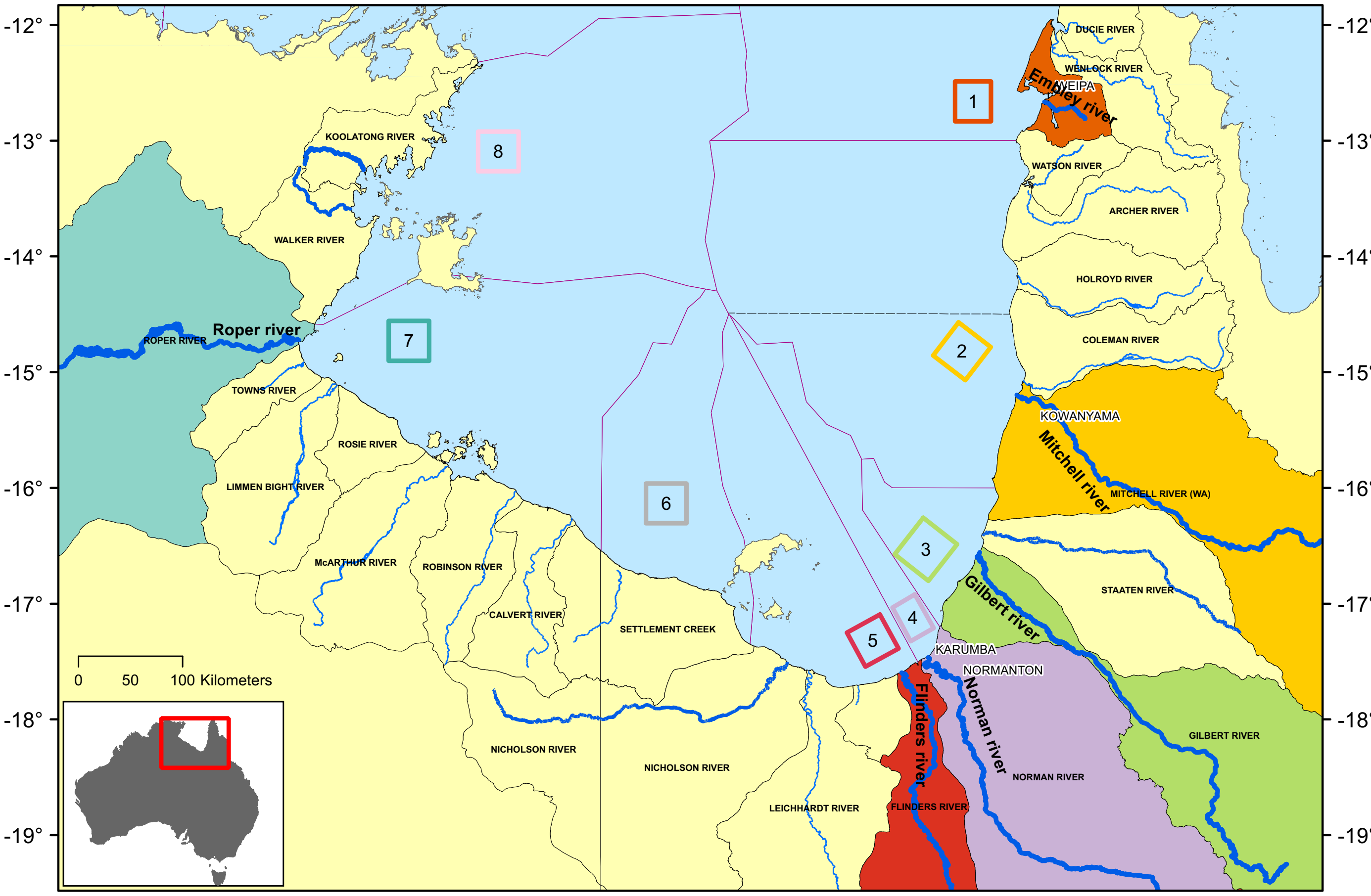
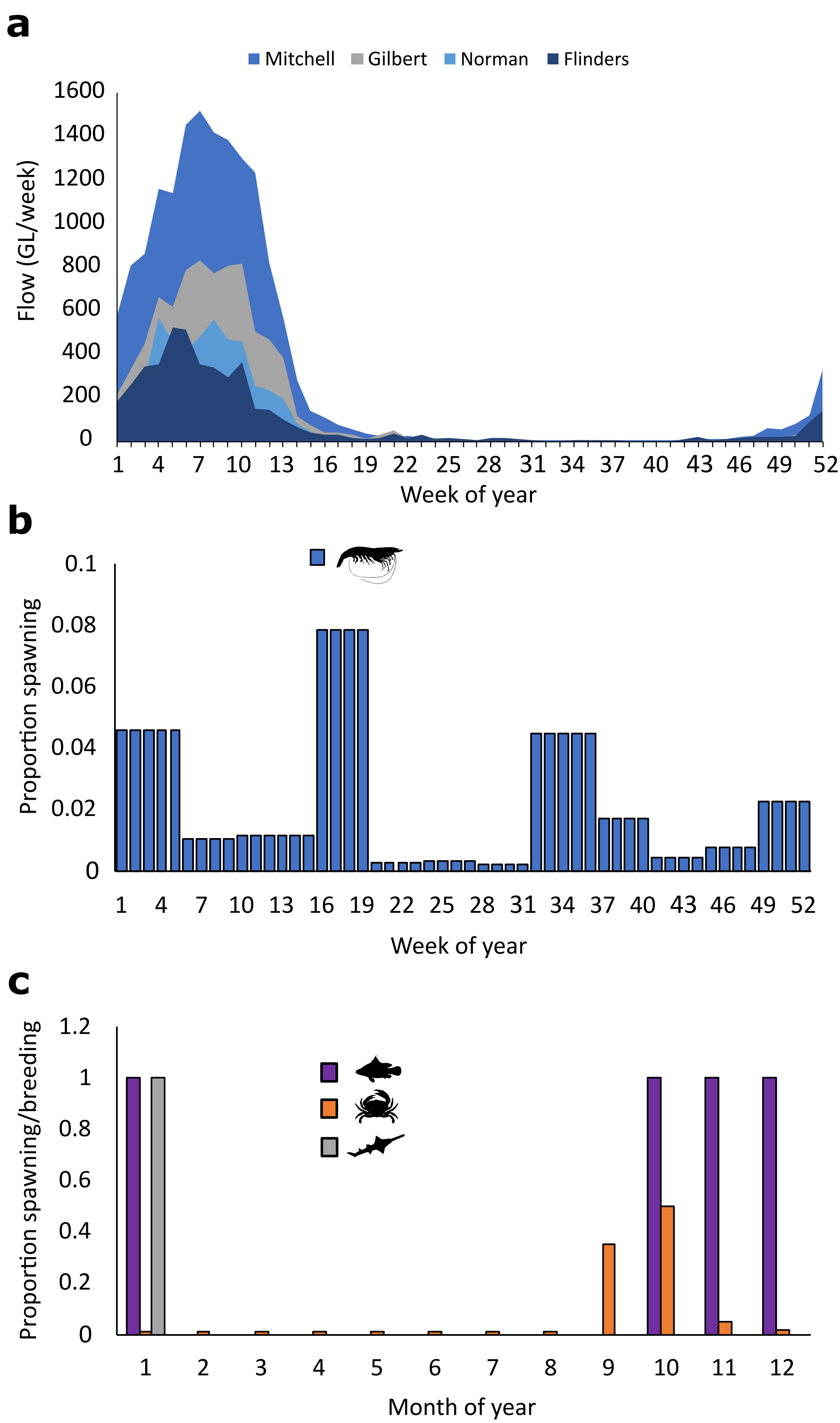


Extended Data Fig. 1 | Spatial map of Gulf of Carpentaria study area showing sub-division into 8 connected spatial regions. Major river catchments are as follows (from East to West, with the vertical line between Region 6 and 7 representing the Queensland-Northern Territory border): Region 1: Embley River; Region 2: Mitchell River; Region 3: Gilbert River; Region 4: Norman River; Region 5: Flinders River; Region 6: border region; Region 7: Roper River; Region 8: Walker River. The width of the blue river shading represents the relative average flow (1970–2019) of each 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. See Supplementary Information for figure details and sources. Main map adapted from Supplementary Refs. 58,59, Geoscience Australia. Inset map from Freesvg.org.



Extended Data Fig. 2 | See next page for caption.

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Extended Data Fig. 2 | Seasonal variability in natural hydrological flows and population processes. (**a**) Weekly average (1970–2018) end-of-system flow (GL/week) for each of the major south-eastern Gulf of Carpentaria rivers including the Norman River and the three rivers for which WRD scenarios are modelled to evaluate the influences of alterations to the natural flows on population dynamics of key species; (**b**) Relative weekly spawning pattern assumed for common banana prawns, noting there is a 6-month lag until recruitment, with the latter thus timed to coincide with high flow periods; (**c**) Monthly patterns of spawning used in model for barramundi and mud crabs, whereas the single grey bar represents that sawfish pupping occurs as a discrete event early in the year. Credit: species icons, PhyloPic.

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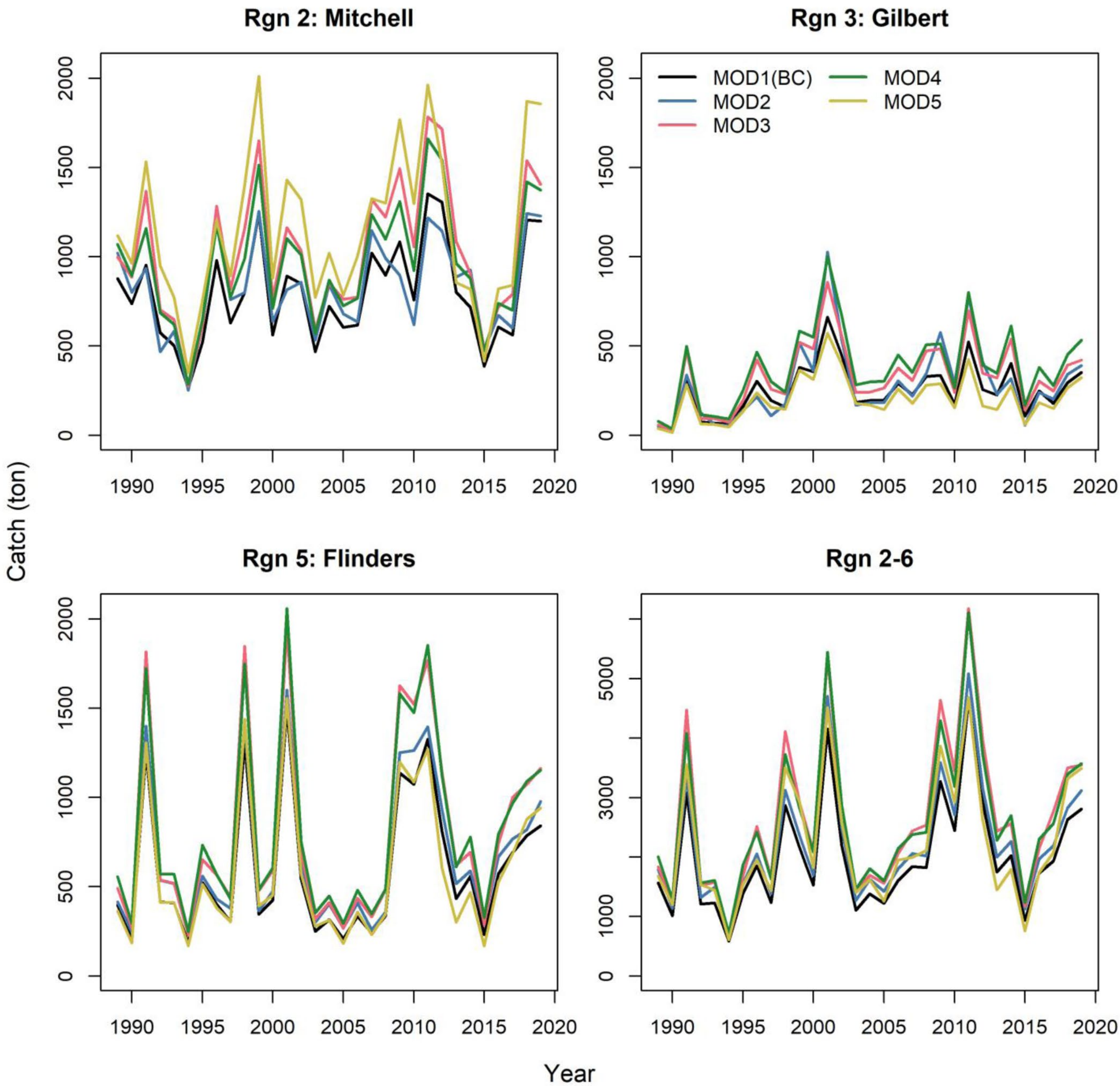

Extended Data Fig. 3 | MICE ensemble outputs shown for the five alternative model configurations used to represent common banana prawns. Trajectories show the model-estimated catch (tonnes) from 1989 to 2019, for the three key model regions as indicated, together with the combined catch for Regions 2–6. See Supplementary Table 7 for a description of the model versions.

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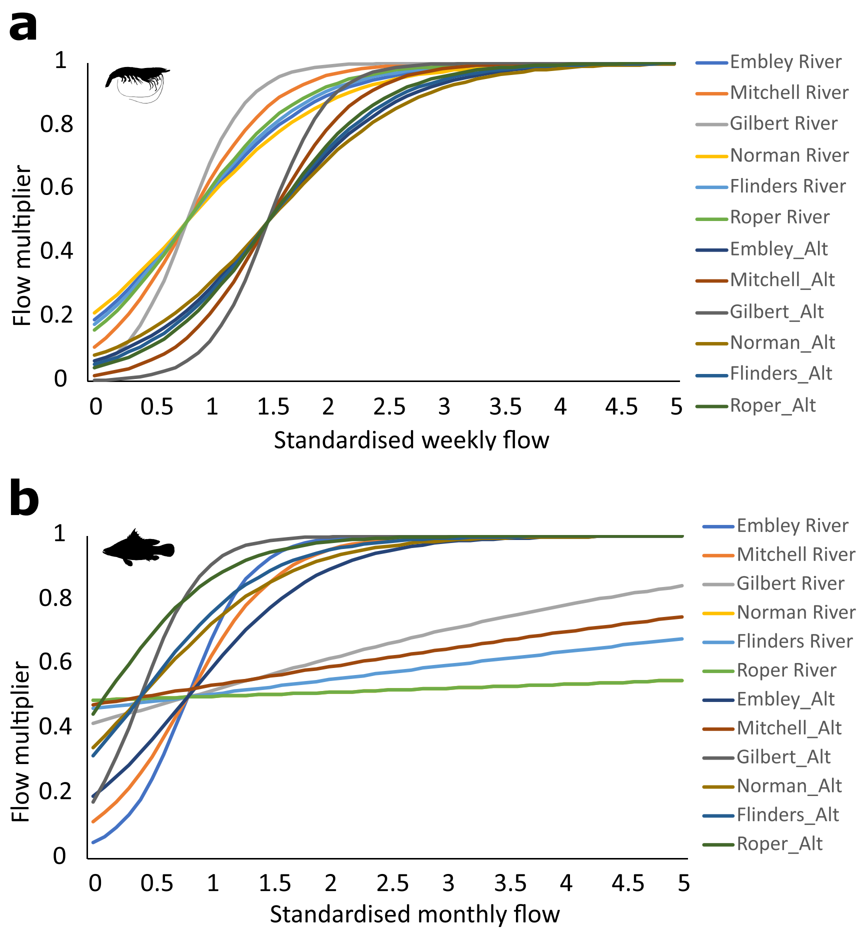

Extended Data Fig. 4 | Functional relationship between flow and population parameters. Model-estimated logistic relationship between standardised flow and a population parameter such as recruitment of (**a**) prawns or (**b**) barramundi, where the function yields a multiplier that describes recruitment relative to the maximum value. The flow multiplier is computed for every week of every year before being applied to the relevant recruitment or population processes, noting for example that the timing of spawning and recruitment varies seasonally. The ensemble uses a range of alternative fixed values and estimated values (See Supplementary Table 8), with two alternatives shown here for each species. Weekly cumulative flow totals are standardised by dividing by the average flow for the same week over all years 1970 to 2019, and similarly for monthly standardised flow. For mud crabs, a parabolic function is used, see Supplementary Table 1. Credit: species icons, PhyloPic.

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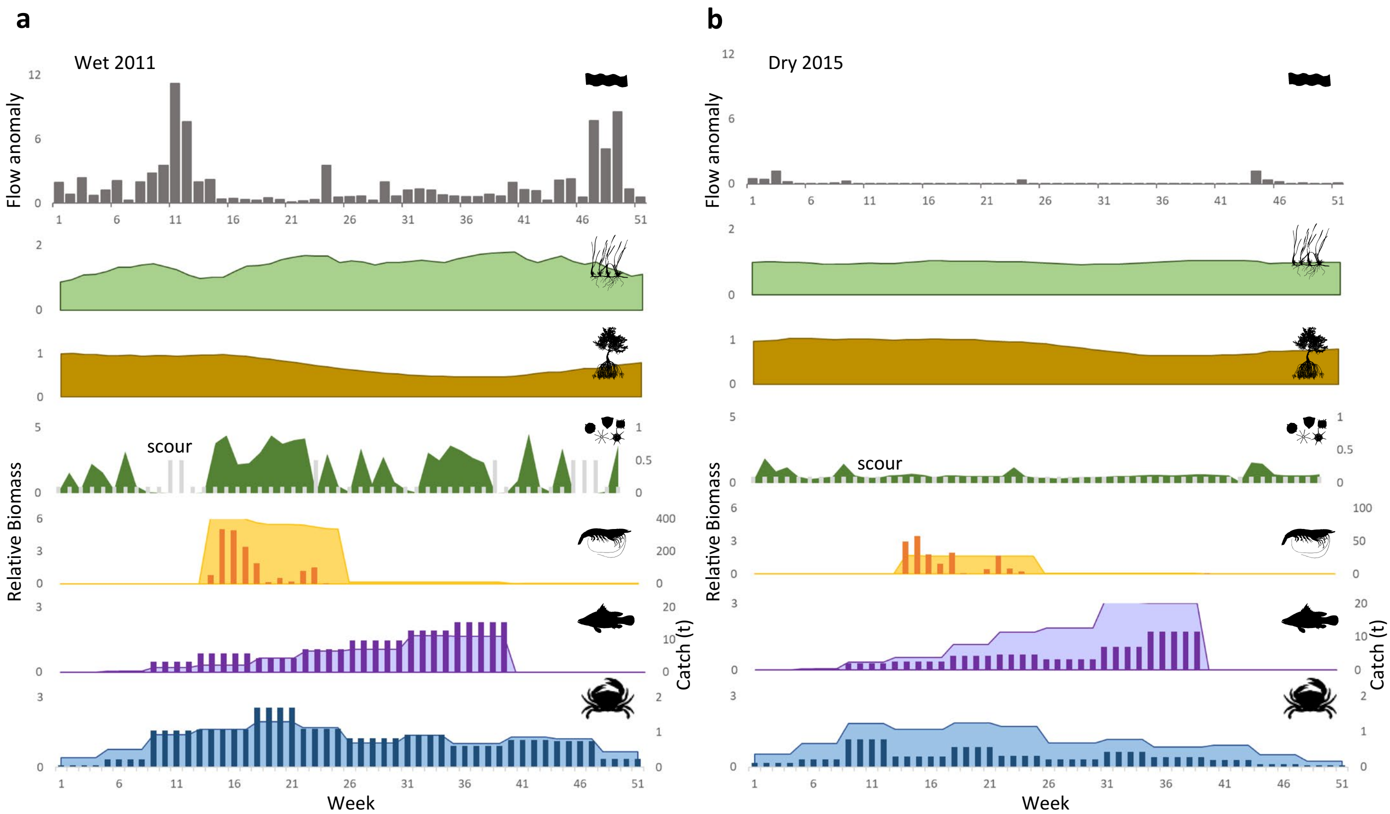

Extended Data Fig. 5 | Cascading effects of wet or dry years on components of the marine ecosystem. An example using the Flinders River to show influence of modelled end of system flows on the intra-annual biomass (shaded areas) during (**a**) a wet year (2011) and (**b**) a dry year (2015). From top 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 depicted as bars and shown on the second vertical axis. Credit: species icons, PhyloPic.

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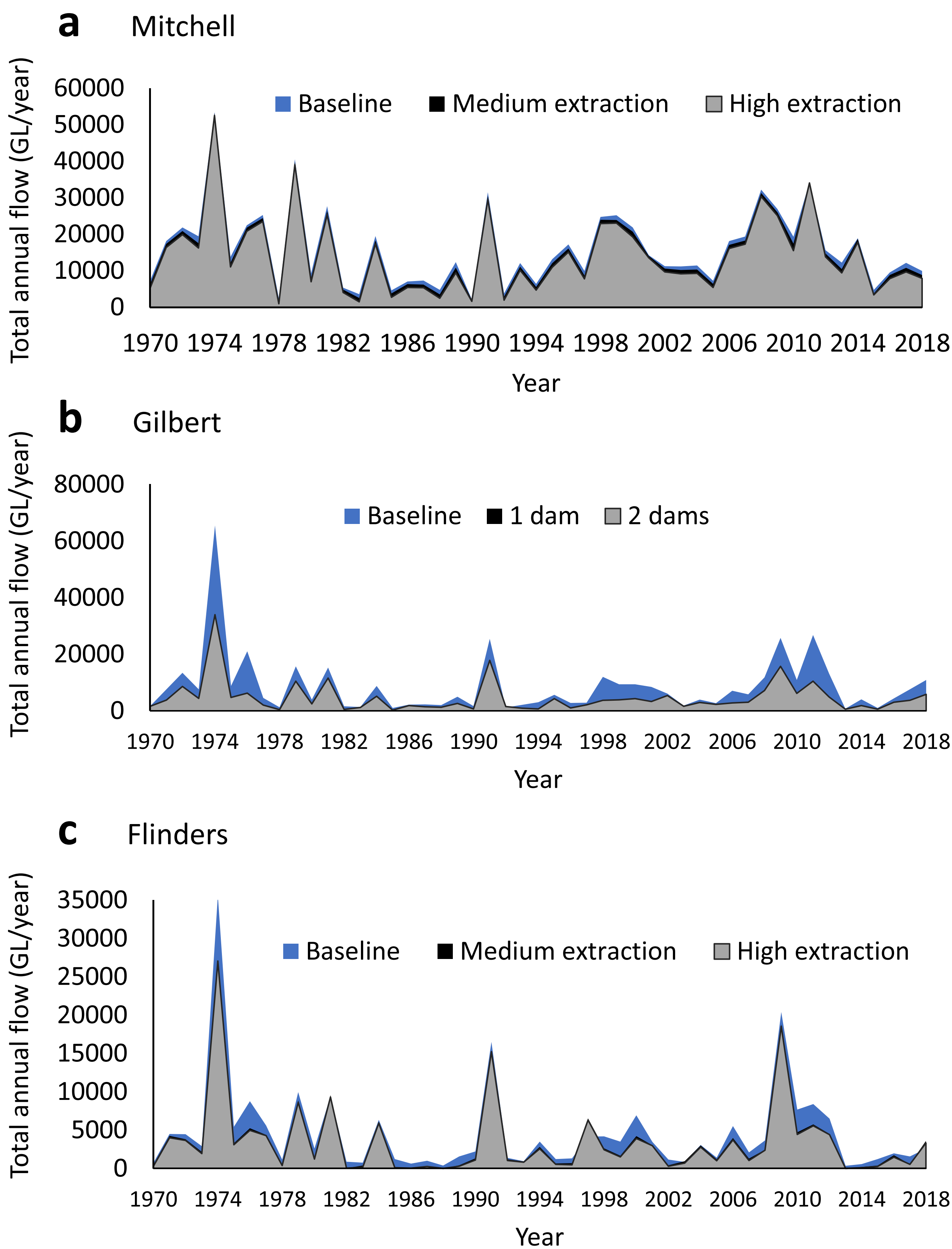

Extended Data Fig. 6 | Baseline river flows compared with water resource development scenarios. Total annual end-of-system flow (GL/year) for (**a**) Mitchell River, (**b**) Gilbert River and (**c**) Flinders River, comparing baseline flows with alternative WRD settings (see Fig. 1b) used to evaluate the influences of alterations to the natural flows on population dynamics and catches of key species (see Supplementary Table 11 for full list of WRDs tested). Note that the model inputs weekly flows and hence the annual aggregated plots do not show shorter time scale variability that is used to inform the MICE. The vertical axes have different scales in these plots to be able to see differences more easily when compared with baseline levels.

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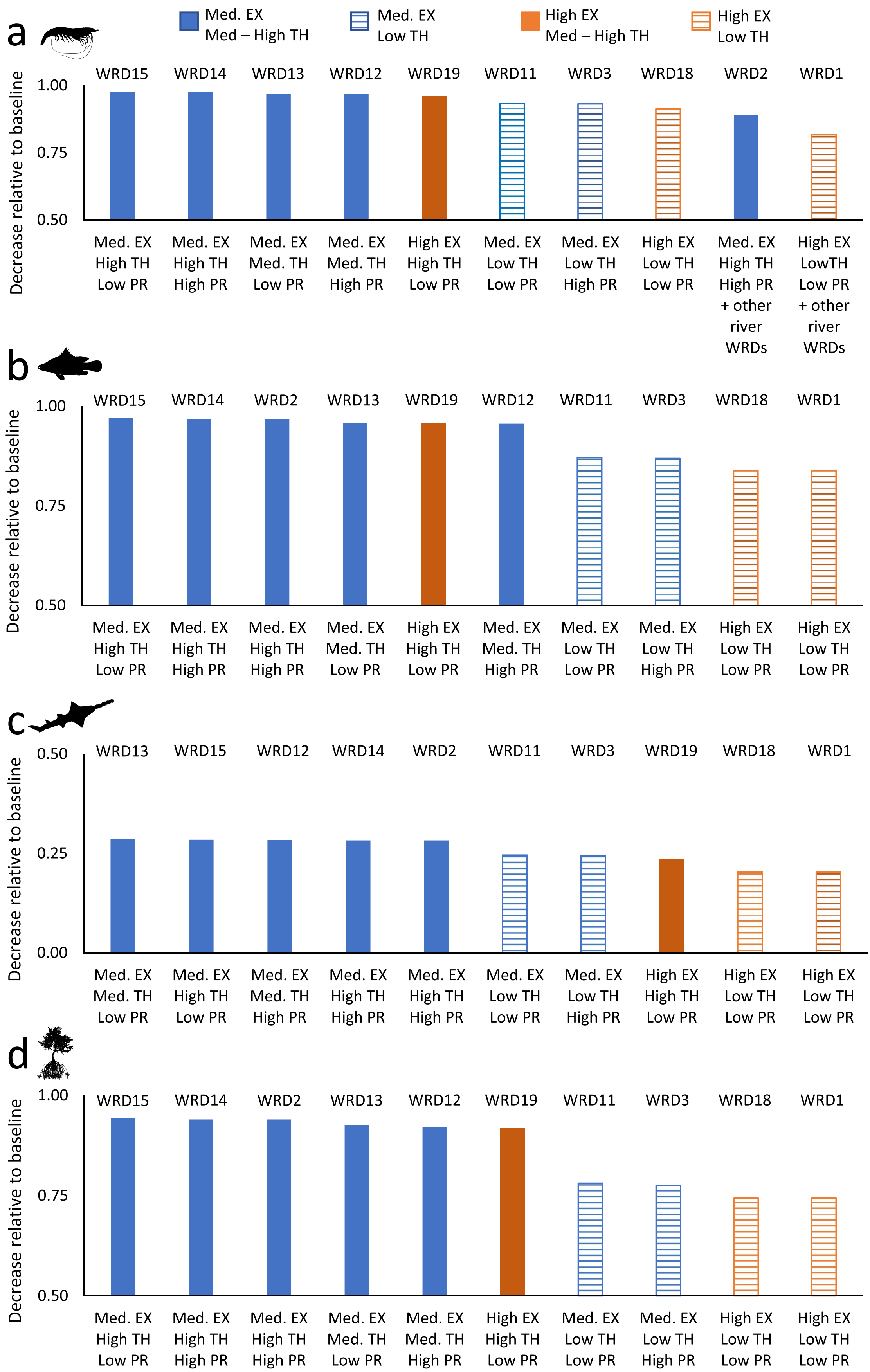

Extended Data Fig. 7 | See next page for caption.

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Extended Data Fig. 7 | Comparison of relative impact of alternative water resource developments and associated settings. Comparison of MICE Model version 1 estimated relative average decline in (**a**) prawn catch, (**b**) barramundi catch, (**c**) sawfish population numbers and (**d**) mangrove biomass under alternative WRD scenarios applied to the Mitchell River. Results not shown for mud crabs and seagrass as they weren't estimated to be sensitive to changes in Mitchell River flows. Results are sorted from least impact on the left to greatest estimated declines to the right. Water extraction (EX) WRDs vary based on the annual allocation (gigalitres per year) (high allocation shaded in orange; 2000 GL/year and medium allocation shaded in blue; 1000 GL/year), flow threshold (TH) and pump rate (PR). Low TH = 200 ML d^{-1} (patterned shading); 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 (see also Supplementary Table 11 for further details and Supplementary Tables 15–18 for full ensemble results). Note that the vertical axes scales differ for ease of viewing. Credit: species icons, PhyloPic.

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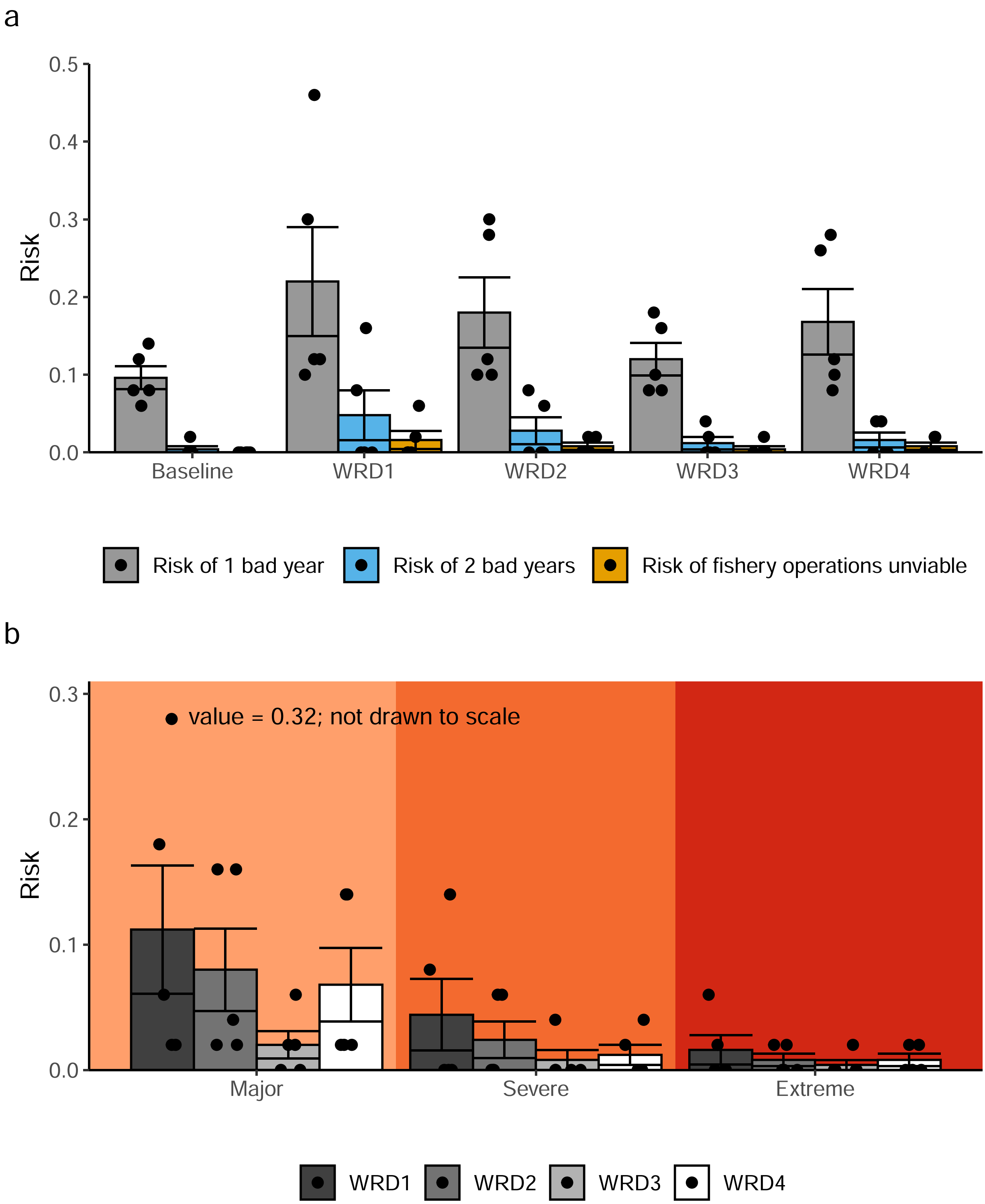

Extended Data Fig. 8 | Model-estimated economic risk to the common banana prawn sub-fishery of Australia's Northern Prawn Fishery. (**a**) The MICE ensemble average (with standard deviation errors) economic risk evaluated under alternative WRDs as summarised in Table 1b. (**b**) Graphical comparison of the average (with standard errors) relative probability of occurrence of major risks (defined as risk of a bad year), severe risk (two successive bad years) and extreme 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. Corresponding data points are overlaid as dot plots with a sample size n = 5 for each bar. See Supplementary section 8 for full details of risk assessment method.

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Our web collection on statistics for biologists contains articles on many of the points above.

Software and code

Policy information about availability of computer code

Data collection	No data collected as part of this study - used existing data
Data analysis	Analysed some existing data using RStudio 2022.02.1 , Excel Microsoft 365 vers 2302 and MICE model developed using AD Model Builder vers 12.0-0

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The data for the river system models are available at: https://nawra-river.shinyapps.io/river/. Permission to obtain the raw fisheries data needs to be granted by the relevant data custodians: Commonwealth Australian Fisheries Management Authority and NPFI (under co-management arrangements NPFI is the delegate) for NPF data, as well as relevant State Fisheries Departments from both the Queensland and Northern Territory jurisdictions for barramundi and mud crabs. Data access contacts and request numbers can be provided on request to EP, and raw model data input files will also be provided subject to relevant data agreements being in place. The environmental driver datasets are publicly available (mostly derived from http://www.bom.gov.au/) and our collated time series, together with the cyclone history and impact scores we developed are available at: https://www.dropbox.com/sh/yirnujdmv22qgoe/AAAyDrOET1j6YkGbAFUbqIiva?dl=0

Human research participants

Policy information about studies involving human research participants and Sex and Gender in Research.

Reporting on sex and gender	The project Principal Investigator is female and the project included 2 male co-investigators and 3 female co-investigators plus a diverse team representing a mix of genders and career stages. In terms of stakeholders participating in workshop discussions, approximately two-thirds were male and one-third female
Population characteristics	NA
Recruitment	Participants at the stakeholder workshops were recruited based on the experience and network connections for the key fisheries as well as key local researchers, drawing on long-term involvement in the region's fisheries and research by the project team, several of whom (including the Principal Investigator) have formal representative roles on local fisheries management advisory committees. Co-investigator AJ was the Northern Prawn Fishery (NPF) industry liaison lead and all NPF industry participants involved in the project were recruited and nominated by NPF Industry Pty Ltd, the industry organisation which represents NPF SFR holders. Co-investigator JR is with the Department of Agriculture and Fisheries, Queensland – the agency responsible for regulating and managing the GoC fisheries for inshore fish (including barramundi) and mud crabs, plus is the fisheries scientist on the FQ Mud Crab Working Group and the FQ GoC Inshore Finfish Fishery Working Group. These groups are equivalent to management advisory committees. Participants that she suggested, were based on her own experience and network connections, informed also by suggestions of other staff who have a responsibility in the space. Persons recommended had extensive experience in GoC fisheries for barramundi and/or mud crabs. Many of the CSIRO and GU project team have conducted research in the study region for two or more decades and assisted in identifying stakeholders who might be interested in hearing about the research or providing feedback, plus we extended invitations more broadly to anyone interested.
Ethics oversight	The research project was approved by CSIRO and the funding agency FRDC. The project did not involve any collection of human data, small group notes or interviews. The project workshops were used to inform stakeholders and invite feedback on the research and workshops were conducted consistent with The National Statement on Ethical Conduct in Human Research (2007) (National Statement (2007)) and the Privacy Act 1988. As above, we invited a broad group of interested stakeholders to one in-person workshop and three teleconferences, obtained their written consent to attend, explained how discussions were informing the research, provided an opportunity for everyone to review the meeting summary notes (with comments other than the project team anonymised), and we included (with consent) as co-authors on this paper everyone who made a substantial (or unpublished) contribution to the research. All participation was on a voluntary basis, with participants who partook in the meetings informed that the meetings would be recorded and participants agreed to this approach. The research is locally relevant and included local researchers and stakeholders throughout the research process.

Note that full information on the approval of the study protocol must also be provided in the manuscript.

Field-specific reporting

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Ecological, evolutionary & environmental sciences study design

All studies must disclose on these points even when the disclosure is negative.

Study description	We developed an ecological ecosystem model called MICE that represented the population dynamics plus the influences of fishing and the environment on a set of key species and habitats. We fitted the models to available fishery data and explored alternative feasible ways in which the species/habitats were influenced by river flow. We used the end of system flows from river models (weekly, from 1970 to 2019 for each of the major rivers in 8 spatial regions) as an input to the MICE. We selected 5 different MICE models that differed in their parametrization and representation of how flow influences system dynamics, to create an ensemble to account for uncertainty, and we used the average or median of the model outputs, plus associated standard deviation when presenting results. Once we had confidence in the ensemble models, we used these together with the baseline river flows as a baseline to compare with alternative water resource development scenarios (WRDs) whereby flow was altered due to dams or water harvesting. We then ran all the models with the WRD-altered flows to estimate what the impact would be on the abundance and catches of selected species and habitats.
Research sample	We built a model that used existing fisheries catch and effort data for common banana prawn (Penaeus merguiensis), barramundi (Lates calcarifer) and mud crab (Scylla serrata) in Australia's Gulf of Carpentaria. For other species, we used parameters and

information from published sources and technical reports.

Sampling strategy	N/A
Data collection	N/A
Timing and spatial scale	N/A
Data exclusions	N/A
Reproducibility	N/A (models reproducible as the same baseline model is used each time when testing a different water resource development (WRD) scenario
Randomization	N/A
Blinding	N/A

Did the study involve field work? ☐ Yes ☒ No

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Materials & experimental systems

n/a	Involved in the study	
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☒	☐	Eukaryotic cell lines
☒	☐	Palaeontology and archaeology
☒	☐	Animals and other organisms
☒	☐	Clinical data
☒	☐	Dual use research of concern

Methods

n/a	Involved in the study	
☒	☐	ChIP-seq
☒	☐	Flow cytometry
☒	☐	MRI-based neuroimaging