LA TROBE UNIVERSITY

CENTRE FOR FRESHWATER ECOSYSTEMS (CFE)
School of Life Sciences

PRODUCTION CONDITION PREDICTIVE MODELLING

Part 2: Murrumbidgee River

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Traditional Owner acknowledgement

The Centre for Freshwater Ecosystems, Albury–Wodonga is located on the land of the Wiradjuri people. We undertake work throughout the Murray–Darling Basin and acknowledge the traditional owners of this land and water. We pay respect to Elders past, present and future.

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Executive summary

Rivers and their floodplains are often disconnected by flow regulation, impacting aquatic productivity and food webs. Increasing the extent, or combination of area and time to which floodplains are inundated may thus result in increased aquatic production through greater nutrient cycling, organic matter production, and increased energy flow to aquatic animals. The Reconnecting River Country Program aims to increase the connectivity of rivers with their floodplains through relaxed flow constraints (increases to maximum daily discharge limits for water for the environment deliveries) in the River Murray and Murrumbidgee River. Relaxing these constraints is expected to result in positive ecological outcomes through an increased extent and duration of floodplain connectivity. However, how increased connectivity might affect the critical ecological processes of primary productivity and energy transfer through food webs, which can constrain the population sizes of organisms, is not well known within the Murray-Darling system. This report thus summarises the results of an ecosystem energetics model developed to predict the production potential –the potential maximum carrying capacity of food webs – of the Murrumbidgee River and its associated floodplains from Burrinjuck Dam to Hay Weir under different flow regimes.

The ecosystem energetics modelling framework applied here first predicts floodplain inundation extent, or the integral of area and time inundated, across the modelled area from (i) spatial inundation threshold models and (ii) modelled discharge time series associated with different flow options. From inundation extent, the model then predicts annual basal production (i.e., the amount of carbon or energy supplied to the base of food webs through primary production and terrestrial carbon subsidies) and simulates the transfer of this production through a model food web. As an indicator of potential differences in energetic carrying capacity of food webs across years, the production potential of large native fish (i.e., large predators such as cod and perch which reflect the dynamics of multiple, lower trophic levels) was used as a comparison measure across time series associated with different flow limit options. Model predictions were compared for several potential flow regime options, corresponding to different daily regulated flow limits at Wagga Wagga: a base a.k.a. W22 option (i.e., 22,000 ML/day limit at Wagga Wagga) was compared against W32, W36, and W40 flow limit options.

Across all modelled years, estimates of production potential for large native fish were slightly greater for the alternative flow option scenarios than for the base W22 option. Estimates of production potential for large native fish were also compared across years where the largest daily flow event was affected by potential flow limits at Wagga Wagga (e.g., 32,000 ML/day for the W32 flow option); i.e., years in which differences in production potential would be directly affected by flow limits rather than driven by unregulated flows. Under these restrictions, the difference between the alternative flow options and the base, W22 option was reduced. The topographic relief of the Murrumbidgee project area and a low frequency of intermediate flooding events in the modelled discharge timeseries, which may strongly skew annual production potential towards larger events, are the likely underlying causes of these patterns.

1. Background & context

Hydrologically connected rivers and floodplains are among the most productive ecosystems on Earth (Opperman et al., 2010). Inundation of low-lying benches, anabranches, and floodplains surrounding river channels can result in substantial pulses of terrestrial organic matter and nutrients into the aquatic environment, as well as large increases in aquatic plant and algal productivity, which can then potentially support production within riverine food webs for months following flood events (e.g., Rees et al., 2020). Consequently, the reduction of floodplain connectivity within large river systems, and the associated loss of production, can and has resulted in a substantial decline in biodiversity, population abundance, and ecological function across regulated systems (Figure 1.1; Kingsford, 2000). Restoration of floodplain production into the wider Murray-Darling Basin (MDB) river network should thus a be critical consideration in the design of flow management strategies aimed at increasing ecosystem health and function (Baldwin et al., 2016).

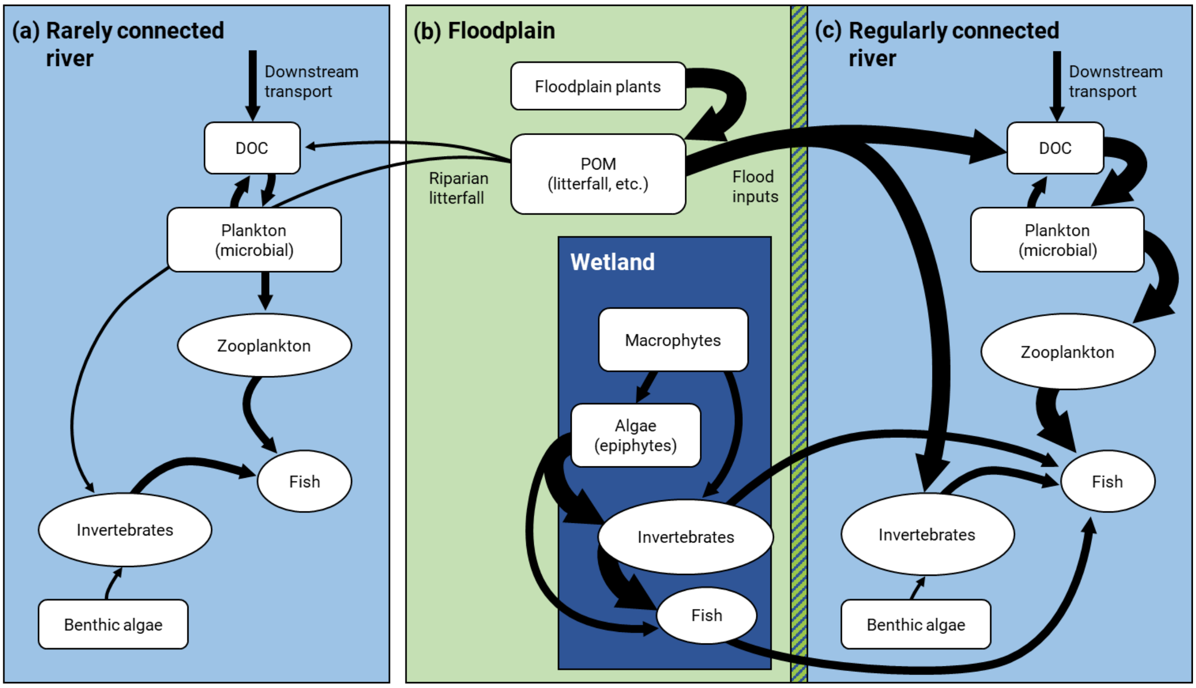

Figure 1.1 Conceptual differences in riverine food web function with respect to connections with floodplains. The size of arrows indicates different fluxes of energy (i.e., Carbon) across different food web components and environments: illustrating differences between (a) rarely connected rivers, where inputs from floodplains may largely be restricted to direct riparian litterfall, and energy fluxes are driven by patterns of in-river productivity or downstream transport of dissolved organic carbon (DOC); and (b) floodplain environments, where production of particulate organic matter (POM) on the terrestrial floodplain, and aquatic macrophytes and algae within floodplain wetlands, can be transferred to riverine environments by (c) regular connections of flooding which "blur" the boundaries between terrestrial and aquatic environments, allowing transport of large amounts of POM into rivers and movement of riverine organisms into floodplain environments; thus increasing both the flux and diversity of resources available to them (e.g., fish feeding on wetland resources as well as increased production of zooplankton) along with an increase in potential habitat availability.

The Reconnecting River Country Program (RRCP) aims to increase the ecological outcomes of environmental water delivery by relaxing constraints (including maximum discharge limits on environmental water delivery through the river system) across the MDB. Relaxing flow constraints therefore has the potential to ensure a greater frequency, duration, and extent of connectivity between rivers and floodplains. Enhanced connectivity should lead to positive ecological outcomes through an increase in habitat quality and nutrient cycling (Kahan et al., 2020). Primary production (i.e., the generation of new biomass by autotrophs), and its role in supporting populations of aquatic biota, is another essential ecosystem function that is expected to benefit from improved river flows under the RRC program, likely through an increased area of shallow aquatic habitat being generated and exposed to sunlight (Mulholland et al. 2001). Yet the literature on both the measured and predicted ecological effects of floodplain reconnection has previously given little consideration to how the processes of primary productivity and energy flow through food-webs, which strongly constrain the overall production of aquatic consumers (Power 1992), might result in positive ecological outcomes under flow restoration (Bellmore et al. 2017). A critical first step in predicting outcomes of relaxed constraints is thus incorporating these "ecosystem energetics" into the modelling frameworks which quantitatively forecast the benefits of flow restoration programs.

Assessing the expected outcomes from different flow regimes requires a predictive modelling approach capable of providing quantitative and spatially explicit predictions of long-term changes (i.e., over multiple years) in both basal production and food-web structure. Ecosystem energetics modelling approaches (which mechanistically model the total energy available to consumers, its quality, and flow through food webs) can be used in this context to assess the potential changes in food-web dynamics that occur with changes to the physical environment (Bellmore et al. 2017). This approach has previously been validated in the MDB by comparison of model predictions with historical estimates of large native fish carrying capacity (Bond et al., in prep), and has been further developed and adapted to predict the changes in aquatic production that might occur with differences in flow regime along the River Murray from Hume Dam to the confluence of the Murray and Wakool Rivers (Siebers et al. 2022).

The model developed by Bond et al. (in prep) and adapted further by Siebers et al. (2022) predicts the production potential of river food webs – the potential "upper ceiling" of production, broadly analogous to food web carrying capacity – by estimating the area and duration of floodplain inundation likely under a given flow regime, scaling this estimate of inundation by annual basal productivity rates, and modelling the transfer of energy through a simulated food web. This approach is thus ideal for forecasting the effects of altered flow regimes (e.g., timing and magnitude of floodplain connectivity) on the potential productivity of riverine systems. The aim of this project is to apply this operational modelling tool to estimate the effects of changing constraints limits on the Murrumbidgee River and its food webs.

2. Project objectives

Part 1 of this project used an ecological energetics model to predict: (i) the area and duration of floodplain inundation; and (ii) the production potential of aquatic food webs (analogous to upper carrying capacity) for the River Murray, and its associated floodplains, from Hume Dam to the confluence of the Murray and Wakool rivers (Siebers et al., 2022). In the process, the energetics models created by Bond et al. (in prep) were updated to include temperature-dependent terms for aquatic productivity, and habitat-specific productivity rates (e.g., differences in algal productivity between river channels and floodplains). This report details Part 2 of this project, which models inundation and production potential for the Murrumbidgee River (downstream of Burrinjuck Dam). The objectives of Part 2 are therefore to apply the ecological energetics models of Siebers et al. (2022) to evaluate the potential productivity benefits, and changes to food-web energy flows, associated with proposed flow options in the Murrumbidgee River.

3. Methodology

3.1 MODEL INPUTS

Model inputs are approximately the same as described by Siebers et al. (2022). Four data inputs are required to run the model:

1. Spatial dataset with defined inundation thresholds (daily discharge in ML estimated for inundation) assigned to given areas (in m^2)
2. Discharge timeseries (in ML day^{-1}) and associated water temperature timeseries (in mean °C day^{-1})
3. Productivity rates (in g C produced m^{-2} yr^{-1}) for seven food web basal resources: dissolved organic carbon (DOC), particulate organic matter (POM), benthic algae, benthic bacteria, phytoplankton, pelagic bacteria, and aquatic and emergent macrophytes (higher plants)
4. A defined food web structure, which includes nodes (taxonomic groups present), links (identification of herbivory or predator/prey relationships), trophic transfer efficiencies (a.k.a. ecological efficiencies) for each link (proportion of C at lower node which contributes to production at upper, linked node), and dominance values for each link (relative proportion of lower node available to upper node when competing links exist for the same resource; Figure 3.1)

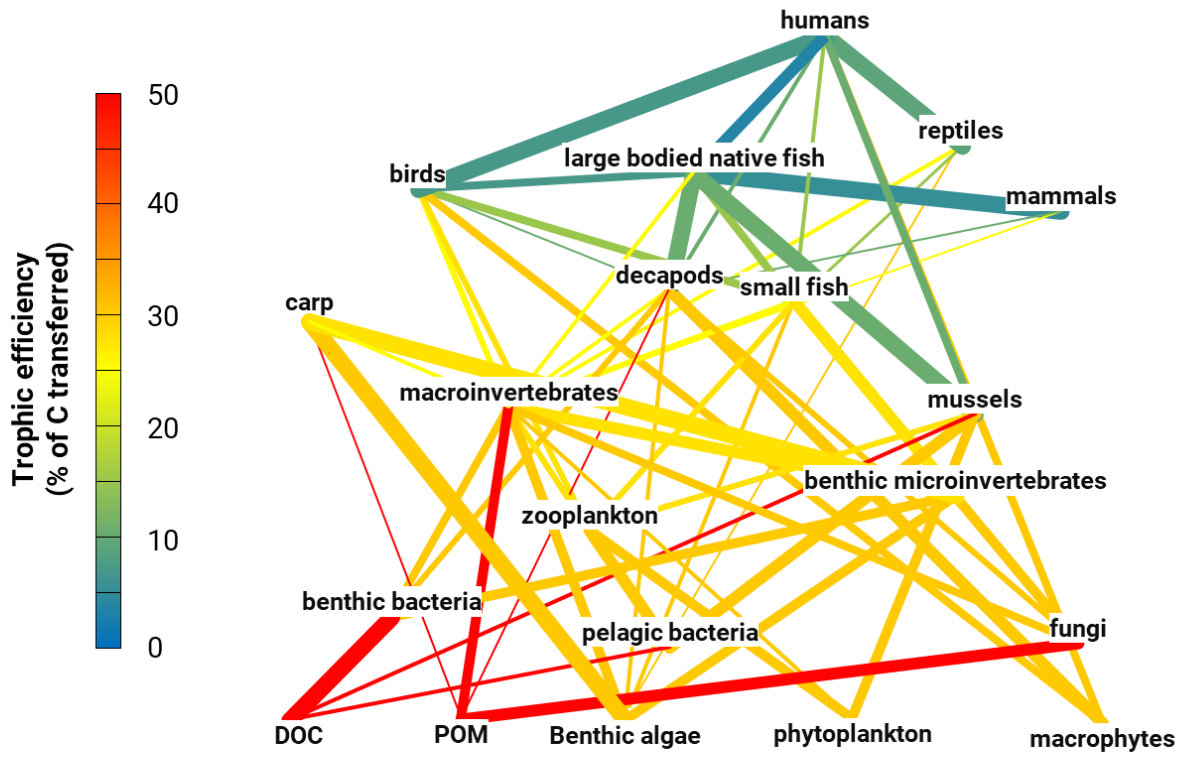

Figure 3.1 Modelled food web including carp (adapted from Bond et al., in prep). Lines between different organism groups and basal resources (a.k.a. nodes) indicate trophic connections (higher nodes consume lower nodes). Colour of lines indicates proportion of C at lower node which contributes to production at upper, linked node (a.k.a. trophic transfer efficiency). Relative width of lines indicates relative proportion of lower node transferred along each link where consumers compete for the same resource (a.k.a. dominance values).

The project area for the second phase of this project (Murrumbidgee: Burrinjuck to Hay Weir) as defined by DPE is given in Table 3.1. Originally, the project area was proposed to extend to the confluence of the Murrumbidgee and Murray Rivers. However, following construction of the original Burrinjuck Dam from 1907 to 1928, overbank flows to the Lower Murrumbidgee floodplain reduced significantly. Landholders lobbied for the construction of weirs and infrastructure to allow some inundation to still be provided to the Lower Murrumbidgee floodplain even under low river flows, as inundation was, and still is, critically important for supporting grazing enterprises on the floodplain. Two weirs were constructed in the 1940s: Maude Weir, which enables the inundation of the Gayini-Nimmie-Caira area, and Redbank Weir, which enables the inundation of South Redbank (Yanga National Park) and the North Redbank area. In most years, the area inundated in these sections of the floodplain is not able to be related to river flow levels: instead, water deliveries via infrastructure are likely to drive inundation. Because of this, reliable relationships between inundation and river discharge are likely unachievable for the area downstream of Hay Weir.

It should be noted that there are still portions of the floodplain below Hay Weir that are not able to be watered by infrastructure, and here the area inundated may still be related to river flow levels. These areas include:

- The floodplain from Hay to Maude
- The floodplain downstream of Maude through to Redbank Weir pool that is adjacent to the Murrumbidgee River. On the northern bank this area is large and includes a section of Juanbung/the Great Cumbung Swamp.
- The floodplain below Balranald, including the Junction Wetlands.

However, it is unfeasible to analyse the effect of river flows for these areas separately; the Murrumbidgee Source model is being further developed, with further work still needed on the gauge/flow relationships of the lower Murrumbidgee (including Maude and Balranald) to improve the reliability of flow estimates. Given the above limitations on estimating the effects of modelled flows in the Lower Murrumbidgee, the area was excluded from the core model results for ecosystem production potential. Brief descriptions of preliminary results from the Lower Murrumbidgee (using the currently available, low-sensitivity inundation models) are instead provided in the appendices (Appendix 2).

Similar to Siebers et al. (2022), a spatial inundation dataset was used specific to the mid-Murrumbidgee River (CARM model; provided by DPE). Counter-factual discharge timeseries for the flow gauge used to calibrate the inundation model, for each of the different potential flow constraints limits (Table 3.1), were produced by DPE through eWater Source modelling (Carr and Podger, 2012). Modelled temperature timeseries were produced for the same gauge by DPE through boosted regression models of historic flow, historic water temperature, and modelled air temperature data (Siebers et al., 2022).

Table 3.1 Constraints projects, flow limit options and area for productivity outcomes assessment

CONSTRAINTS MEASURES PROJECT AREA	FLOW LIMIT OPTIONS TO BE ASSESSED (ML/DAY)	AREA FOR PRODUCTIVITY OUTCOMES ASSESSMENTS	
		PROJECT AREA	BROADER AREA
Murray: Hume to Yarrawonga	25,000 @ Doctors Pt 30,000 40,000	River Murray floodplain from Hume Dam to Yarrawonga Weir (NSW and VIC sides)	
Murray: Yarrawonga to Wakool	15,000 d/s Yarrawonga Weir 30,000 40,000	River Murray floodplain from Yarrawonga Weir to Wakool junction (NSW and VIC sides)	Southern Connected Basin (NSW/VIC, downstream of Hume and Burrinjuck dams to SA border)
Murrumbidgee: Burrinjuck to Hay	22,000 @ Wagga Wagga 32,000 36,000 40,000	Murrumbidgee River floodplain from Burrinjuck Dam to Hay Weir	

3.2 MODEL WORKFLOW

The model workflow follows that of Siebers et al. (2022). First, the spatial dataset is split into the four broad habitat classes for which different productivity values are defined: perennial river channels, permanent floodplain wetlands, intermittent floodplain waterbodies, and all other floodplain (Siebers et al., 2022). Next, the spatial dataset on inundation thresholds is combined with the discharge timeseries to produce a timeseries of total inundation (area*days) over a given time period (Figure 3.2). Inundation area*days values are calculated for three distinct inundation time periods: all inundation periods (0-day lag); inundation periods where the area concerned has already been inundated for at least 10 days (10-day lag); and inundation periods where at least 30 days of inundation has already occurred (30-day lag). These lagged inundation periods are integrated into the model to account for the amount of time required for different basal producers to contribute to food web energetics (DOC and POM – 0 days; benthic algae and phytoplankton – 10 days; macrophytes – 30 days; see: Bond et al., in prep, Siebers et al., 2022). In the case of the 10-day and 30-day lag values, area*days values are therefore only calculated using days where the initial lag value has been exceeded. Inundation extent is then summed to a monthly time-step, and monthly mean water temperature is calculated from the modelled temperature data timeseries. The productivity rate inputs, modified by monthly mean water temperature (see: Siebers et al., 2022), are then multiplied by appropriate area*days values (see: Siebers et al., 2022) to provide estimates of potential C produced by these basal sources over the given time period (Figure 3.2). Each of these values is then passed through the given food web structure to produce estimates of "production potential", i.e. maximum potential C produced at each node (Figure 3.2).

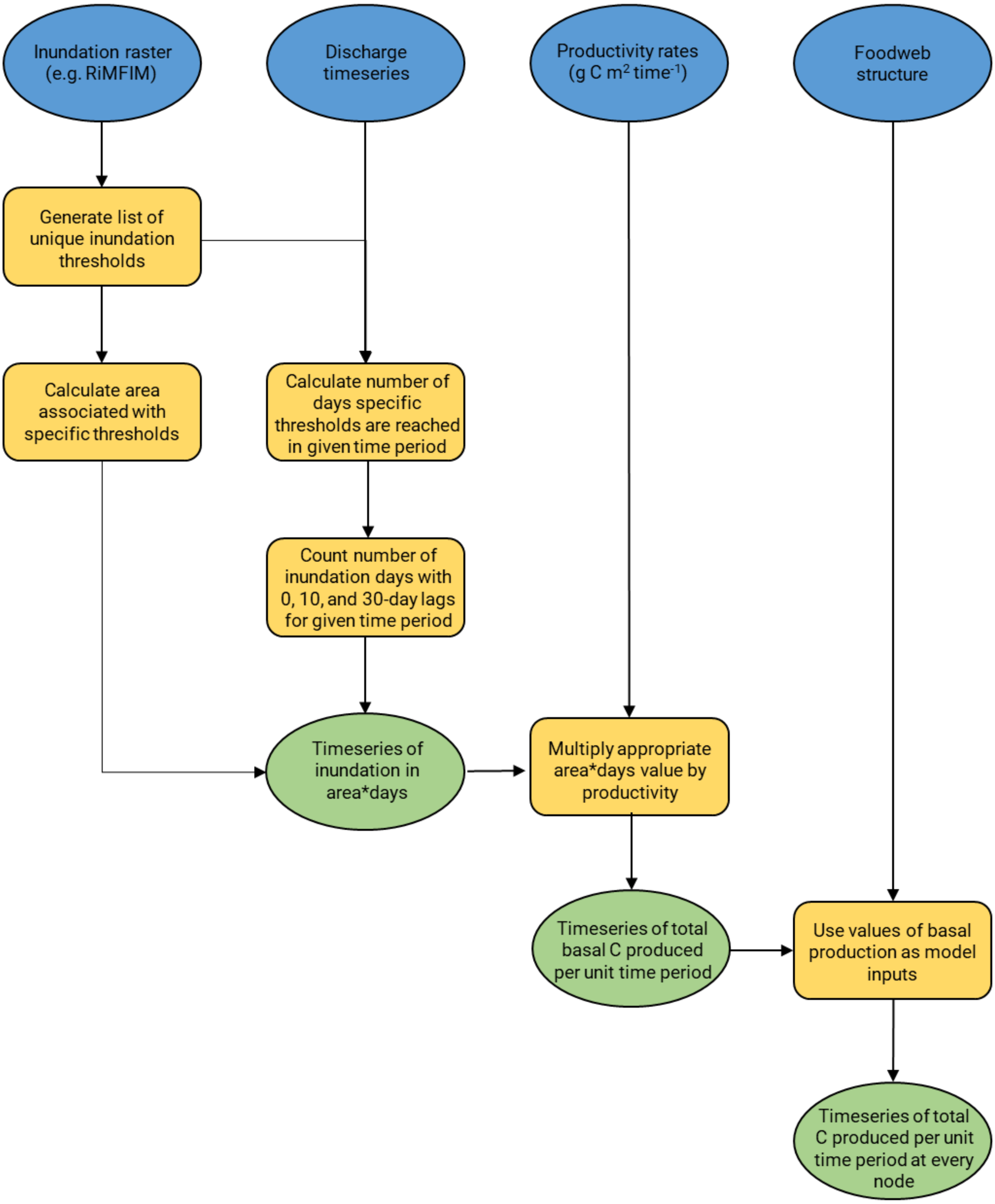

Figure 3.2 Flow chart showing the workflow associated with generating estimates of maximum energy (in g C) available for production at each foodweb node.

Outputs of the model are then summed to yearly values of maximum C potentially produced across all habitat classes, and used to compare different models, model modifications or inputs, e.g. food-web structures including or excluding carp (Figure 3.3).

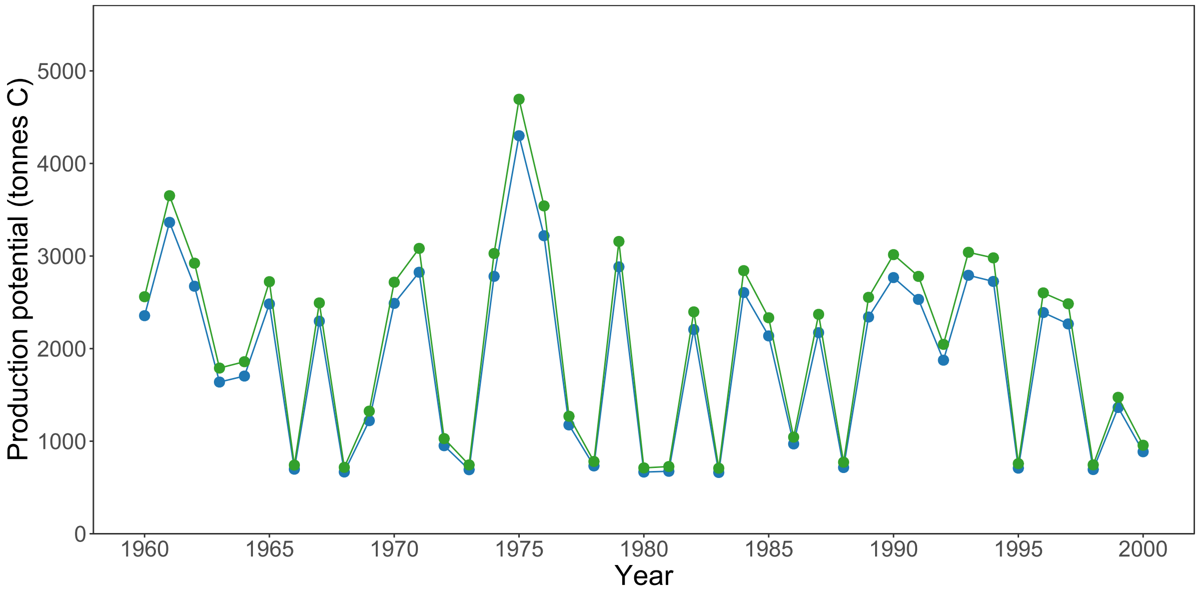

Figure 3.3 Comparison of predicted production potential of large native fish (tonnes C) from a model with carp included (blue) and excluded (green) from the food web structure. Model estimates are derived from spatial datasets on inundation from the mid-Murrumbidgee project area (see: Section 3.3) and counterfactual discharge from the W22 flow option (a.k.a. "base" option).

3.3 COMPARING FLOW OPTIONS

As above, the modelled area for the flow options comparison was the Murrumbidgee River catchment from Burrinjuck Dam to Hay Weir (Figure 3.4).

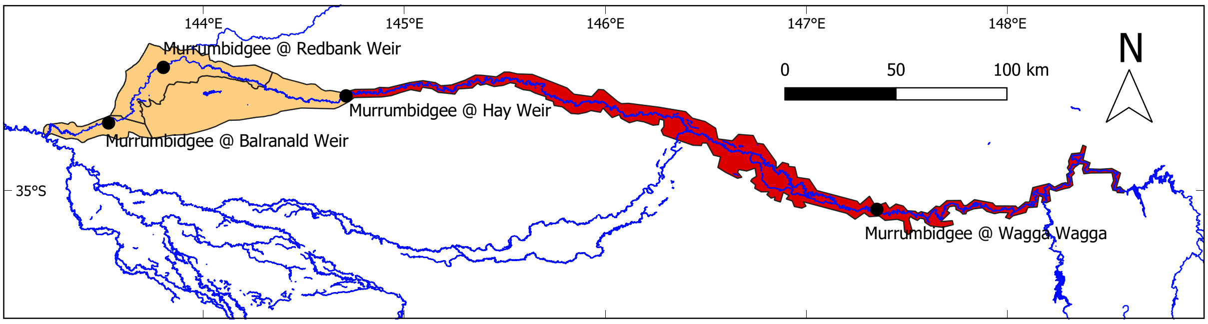

Figure 3.4 Extent of modelled areas, Murrumbidgee River from Burrinjuck to Hay Weir (red filled area), with CARM zones outlined. RiMFIM inundation model zones (see: Appendix 2; not included in core models) are shown for reference (orange filled areas). Black dots show associated flow gauges.

The flow options to be assessed are those defined earlier in Table 3.1 and follow the nominal constraints limits at Wagga Wagga for each flow option: for the Murrumbidgee project area, the 22,000 ML/day option (henceforth: "W22" option, a.k.a. the "base" option), 32,000 ML/day (henceforth: "W32"), 36,000 ML/day (henceforth: "W36"), and 40,000 ML/day (henceforth: "W40") flow options.

For the Murrumbidgee: Burrinjuck to Hay project area (henceforth: Area 3), the model inputs were therefore:

1. Inundation thresholds for the mid Murrumbidgee (Burrinjuck Dam to Hay Weir);
2. Modelled discharge and water temperature timeseries representing all of the defined flow options for the Wagga Wagga gauge (as inputs for the mid-Murrumbidgee CARM inundation model);
3. Basal resource productivity rates (in g C m^{-2} yr^{-1}) modified for habitat-dependence (Table 3.2); and,
4. Common food web structure (see: Figure 3.1).

Table A1.2 Comparison of basal productivity rates (in mean g C m^2 yr^{-1}) used for model inputs across different habitat types. See Siebers et al. (2022) for derivation of values.

HABITAT	DOC	POM	BENTHIC ALGAE	BENTHIC BACTERIA	PHYTO-PLANKTON	PELAGIC BACTERIA	MACROPHYTES
River channel	6.8	57	0.03	0.04	101.7	166.3	39.6
Floodplain	32.8	189.5	5	3.2	101.7	166.3	39.6
Intermittent wetland	32.8	189.5	7.9	5	185.5	729.3	72.3
Permanent wetland	22.8	189.5	39.1	25.1	185.5	729.3	121.7

Although we also produced estimates of large native fish production potential for modelled food webs excluding carp, these model estimates were consistently around 5% greater than those from models including carp, regardless of the flow option being modelled: these results likely reflect the consistency of trophic transfer estimates within our modelled food web, and our inability to parameterise these more accurately given a dearth of empirical evidence to inform model parameters (see: Siebers et al. 2022). Given this consistency across modelled timeseries, for this report we thus only compare results from modelled food webs including carp.

To compare flow options for each of the areas, we compared the density distributions of large native fish production potential estimates from each option (i.e., the proportion of all yearly estimates which reached progressively increasing values of production potential) against each other, for each water year (July to July) from modelled years 1896 to 2018. To provide context for these results against each other, we compared the proportion of years in which each flow option reached a nominal threshold from the W22 flow option (the 25th percentile), i.e., the proportion of years in which estimates for each flow option provided greater estimates of production potential than might be commonly observed under the "base" option. We also provide a summary of the overall median (and 25th percentile) of production potential across all years, under each different flow option.

We then calculated the difference between all pairwise comparisons of flow options in estimates of production potential of large native fish (tonnes C) for each year. To assess the magnitude of differences and how often each flow option produced greater estimates of production potential, we summarised each pairwise comparison using the following metrics:

1. The mean difference in production potential between the two flow options being compared (e.g., the mean value of the W32 option – W22 option, and *vice versa*);

2. The above difference in production potential expressed as a percentage of the lower estimate (e.g., the difference between the W32 and W22 options as a percentage of the W22 estimate); and

3. The proportion of years in which each flow option produced greater estimates than the other (e.g., the number of years the W32 option produced greater estimates than the W22, as a percentage of total years modelled).

We also calculated the following metrics to provide context on the consistency and frequency with which each flow option provided greater estimates than the option it was being compared against:

4. The total number of years in which a given flow option estimated consecutive periods of greater production potential (i.e., a continuous period of at least two years) than the flow option it was being compared against; and

5. The mean length of these consecutive periods.

Finally, we repeated these analyses on estimates of production potential for large native fish (g C) only across years in which the maximum recorded daily flow at Wagga Wagga was predicted to be affected by constraints limits (e.g., 22,000 ML/day for the W22 flow option). In practice, we filtered the dataset to years where maximum flow/day values were less than the constraints limit + 500 ML/day in the modelled discharge timeseries to account for maximum values just above these limits (i.e., years in which constraints limits may have a direct effect on the extent of floodplain inundation, rather than inundation being driven by large, unregulated flow events). Again, we (i) compared density distributions of large native fish production potential across constraints years (any year in which a constraint was modelled: i.e., any year in which a given flow option was restricted to maximum daily discharge below its nominated constraints limit), and compared these to the 25th percentile of W22 results in those years; and (ii) calculated the difference between flow options in estimates of large native fish production potential for each constraints year (for pairwise comparisons, each year in which the maximum daily discharge was below the constraints limit for either option being compared).

4. Results

4.1 COMPARISON OF CONSTRAINTS ACROSS ALL YEARS

Discharge timeseries from the flow options produced relatively similar patterns in cumulative yearly discharge over time for the Burrinjuck-Hay area (Figure 4.1). Estimates of inundation extent over each year were also similar between the options. Production potential for all of the flow options were therefore also similar across years, with large flood events consistently increasing yearly estimates to levels greater that of the base level of production potential from the permanent channel network (Figure 4.1).

For the Burrinjuck-Hay area, the proportion of years in which the three alternative flow option scenarios (i.e., the W32, W36, and W40 flow options) produced estimates of large native fish production potential above the 25th percentile of the base W22 option was around 78-80% (Figure 4.2). There was thus only a small, equivalent increase (4.7%) in the proportion of years that might provide greater outcomes for production potential under the W32 and W36 options relative to the base scenario (W22), with the lowest increase for the W40 (3.1%) flow option scenario (Figure 4.2).

There was only a small increase (1.24%) in the 25th percentile of production potential estimates from the W22 base option (207 tonnes C) to the W36 option (209 tonnes C), with a lower increase (0.4%) for the W40 option (Figure 4.3). However, the median yearly production potential across flow options increased by a greater proportion (9.3%) from the W22 base option (221 tonnes C) to the W36 option (242 tonnes C), while the W40 option produced a greater median estimate (11.2%) again (Figure 4.3).

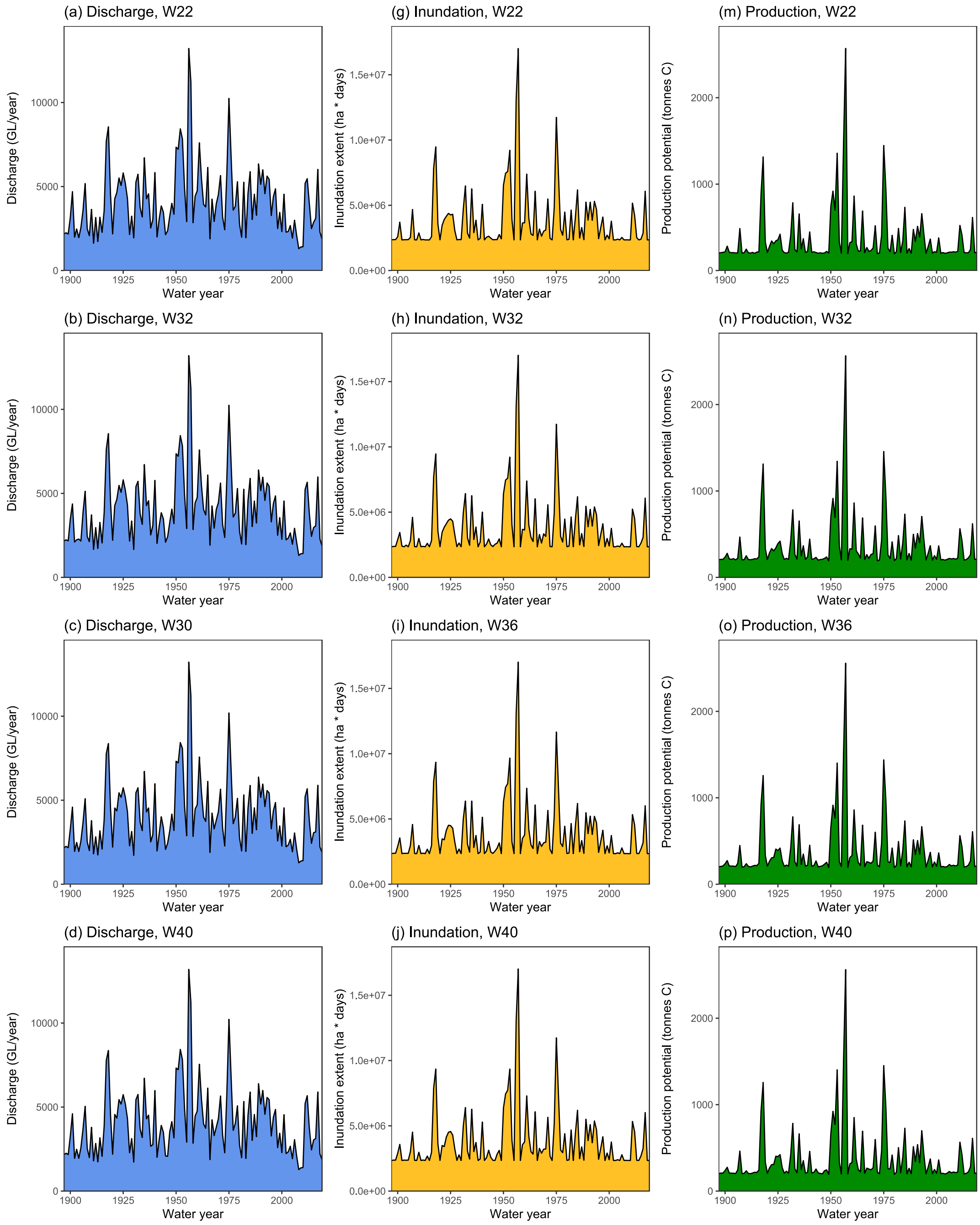

Figure 4.1 For the Burrinjuck Dam to Hay Weir area, for each year from 1896-2018, comparisons between each of the different modelled flow options (W22, W32, W36, and W40) for (a-d) annual cumulative discharge at Wagga Wagga; (e-j) annual estimated inundation extent (in ha*days); and (m-p) estimated production potential of large native fish (tonnes C).

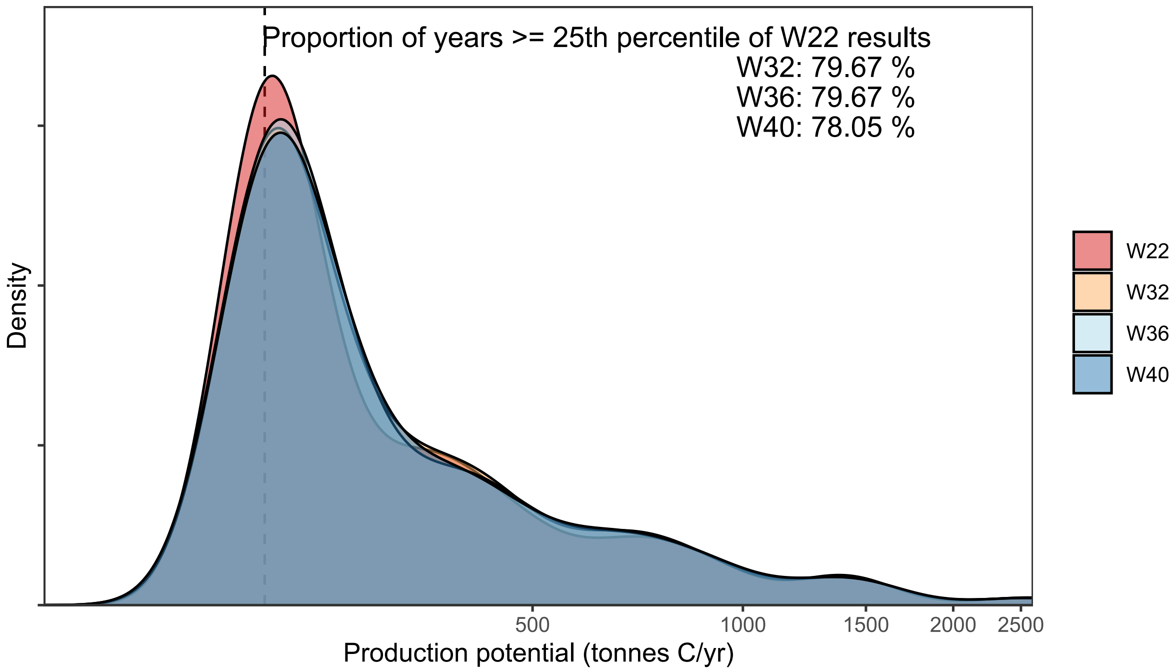

Figure 4.2 Density distributions of model results for the production potential of large native fish from the four different modelled timeseries, across all years (1986-2018), for the Burrinjuck Dam to Hay Weir area. Dashed line indicates 25th percentile of W22 results (i.e., value above which 75% of yearly estimates from the W22 model occur). Statistics shown in the top right indicate the percentage of years for each of the modelled flow option scenarios where estimates exceeded the 25th percentile of W22 results. Note the use of nonlinear scale on the x axis (pseudo-$\log_{10}$ transformation) to aid in discrimination between density distributions.

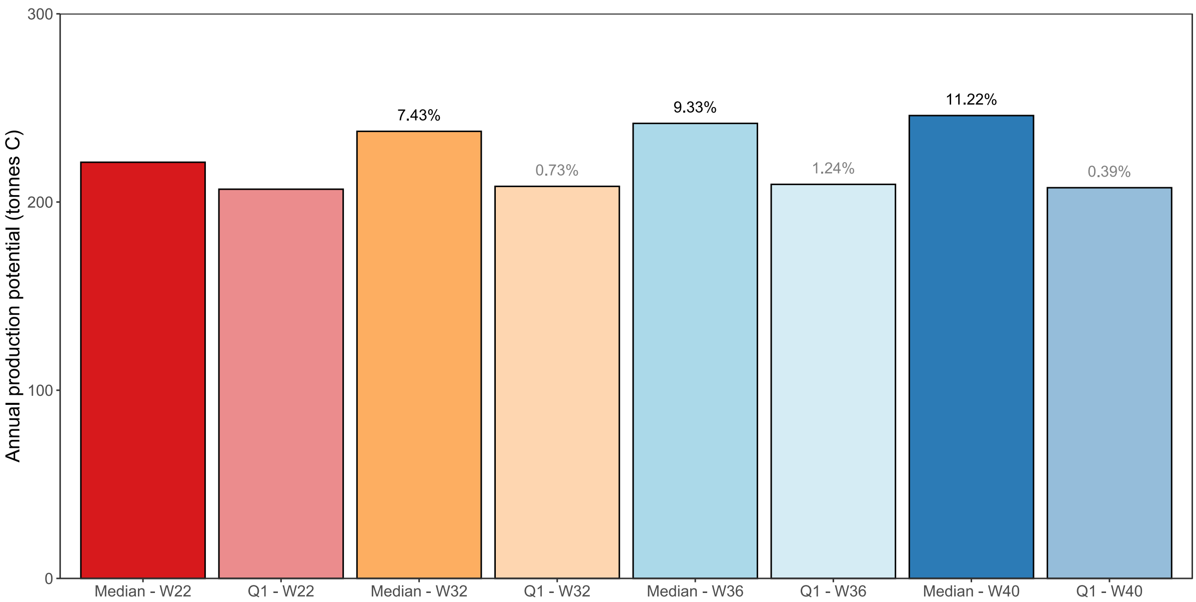

Figure 4.3 Median and 25th percentile (i.e., "Q1") of annual production potential estimates, across all years, for the different flow options associated with the Burrinjuck-Hay area. Numbers above the W32, W36, and W40 statistics indicate the difference between that value and the corresponding statistic for the W22 option (a.k.a. "base" option).

Relative to the W22 "base" option, the alternative flow option models predicted greater estimates of production potential more often (Figure 4.4). The model with a higher constraints limit produced greater estimates ranging from 54% (W32) to 57% (W40) of all years in comparison with the W22 model (Table 4.1). There were thus consistently more years in which the alternative flow option models produced greater production potential estimates than the W22 base option, and often to a level that was noticeably higher than the maximum differences generated in favour of the base W22 option (Figure 4.3). For example, while the mean difference between the W22 and W36 model estimates was approx. 20 tonnes C/yr when the W36 model produced higher estimates, the mean difference was 10 tonnes C/yr when the W22 model produced higher estimates (Table 4.1). Relative to the lowest total estimate produced by either flow option in any given year, these differences correspond to an approx. 5.0 – 6.4% mean difference in production potential for the flow option with the higher constraint limit, and 1.0 – 2.6% mean difference in production potential for the base flow option (Table 4.1). Across all comparisons, the alternative flow option models also produced greater estimates in consecutive years more often (50 – 55 total years) and for longer on average (mean 2.9 – 3.4 years of consecutive periods) than the base W22 option (32 – 37 years; mean 2.5 – 2.9 year periods). Consequently, all of the alternative flow option models consistently estimated greater production potential for longer periods than the base scenario.

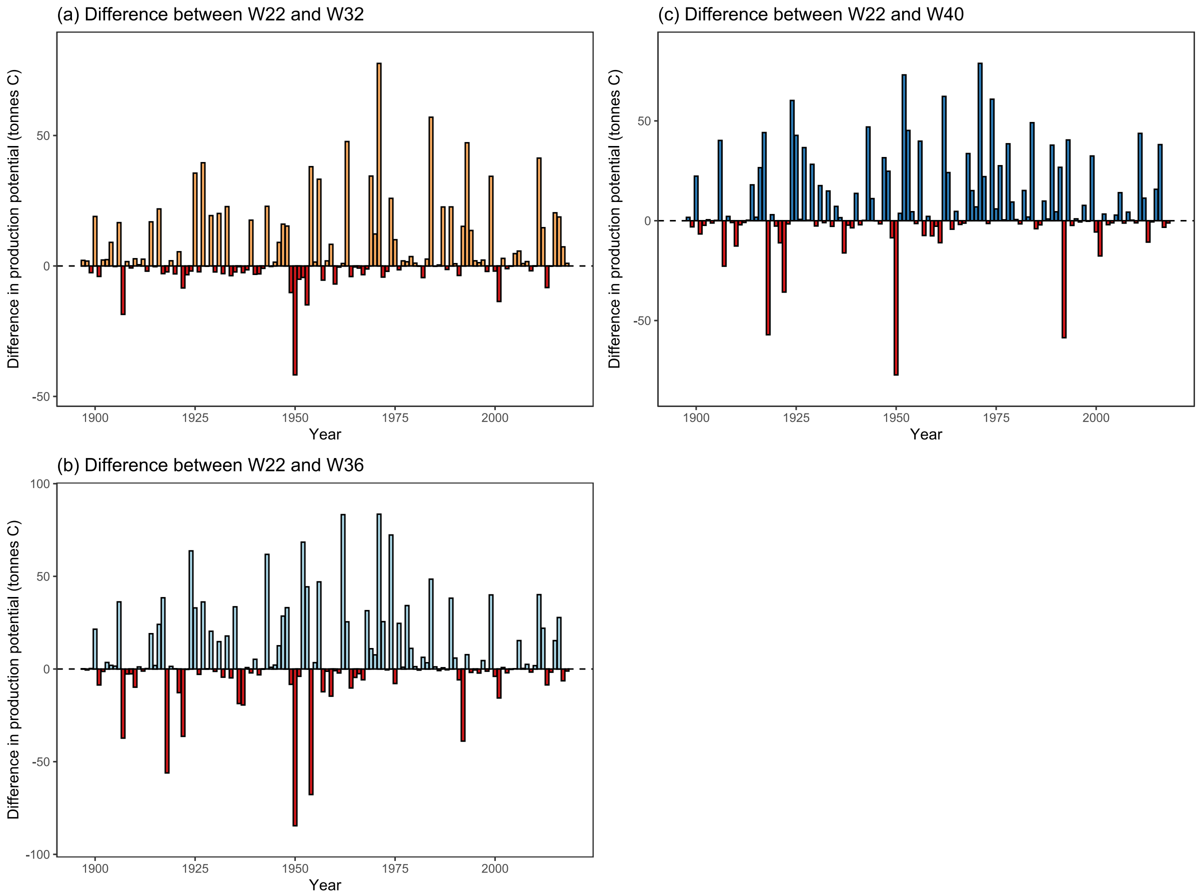

Figure 4.4 Difference between model estimates for production potential (tonnes C), for the Burrinjuck Dam to Hay Weir area, from the W22 (a.k.a. "base") flow option and (a) the W32 (i.e., W32 – W22 estimate), (b) W36, and (c) W40 model estimates, across all modelled years. Positive values indicate that estimates are greater for the flow option with the higher constraints limit, and *vice versa*.

In the other comparisons between flow options for the Burrinjuck to Hay area, however, estimates were not always greater for the flow option with the nominally higher constraints limit (Figure 4.5). Across all years, the W32 flow option produced greater estimates of production potential slightly less often (48% of years) than the W36 and W40 options (52% of years) (Table 4.1), and the W36 and W40 flow options also had higher peak differences in production potential (Figure 4.5) and higher mean differences (13 tonnes C/yr; corresponding to 3.7% of the lower estimate; Table 4.1). In turn, the W40 option also had greater peak (Figure 4.4) and mean differences (6 tonnes C/yr; 1.8% of lower estimate) than the W36 flow option (4 tonnes C/yr; 1.3% of lower estimate), and these greater estimates occurred across a greater proportion of years (54 vs. 46% of years) (Table 4.1). Yet the W32 option produced consecutive years with greater estimates more often (50 years) although for similar periods (2.9 – 3.1 year periods), than the W36 and W40 flow option models (45 years; 3.0 – 3.2 year periods), while the W40 option produced greater estimates more often (54 years) and for longer periods (3 year periods) than the W36 option (41 years; 2.6 year periods; Table 4.1). Consequently, although the W40 flow option produced greater estimates of production potential more often than the W36 option, the W32 option produced greater estimates of production potential more often than the W36 and W40 flow options.

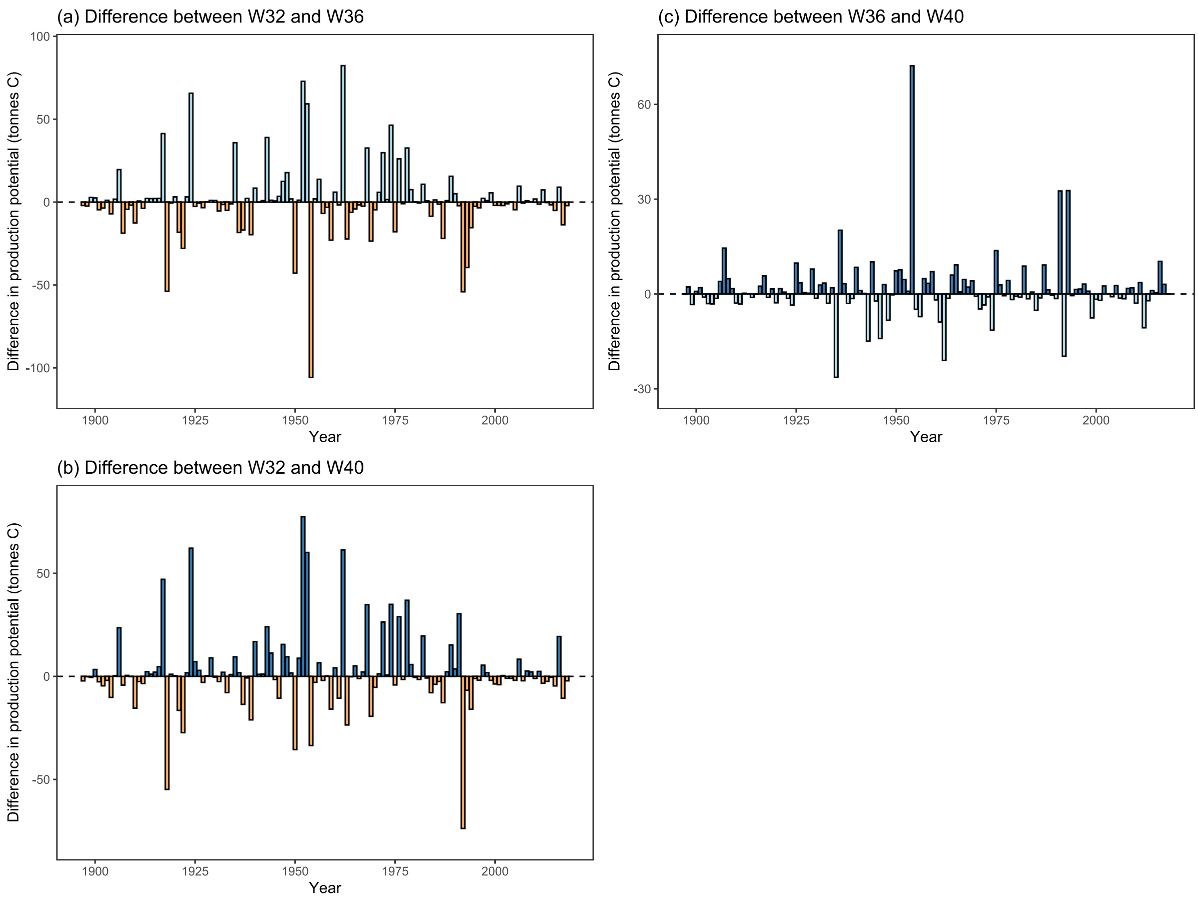

Figure 4.5 Difference between model estimates for production potential (tonnes C), for the Burrinjuck Dam to Hay Weir area, from the W32 flow option and (a) the W36 (i.e., W36 – W32 estimate) and (b) W40 model estimates; and (c) difference between W36 and W40 flow option model estimates, across all modelled years. Positive values indicate that estimates are greater for the flow option with the higher constraints limit, and *vice versa*.

Table 4.1 For the Burrinjuck Dam to Hay Weir area, summary statistics for differences in pairwise comparisons of estimates between flow option estimates for all modelled years. Shown for the subset of results in which one option produced greater differences (e.g., when the W32 flow option model produced higher estimates than the W22 model) are the mean differences in large native fish production potential estimates (in tonnes C/yr for both absolute values, and percentage of difference relative to the lower model estimate), proportion of years (%) in which the given flow option model produced higher estimates, the total number of years in which the given flow option model produced higher estimates across consecutive years (i.e., >= 2 years), and the mean length of those consecutive periods.

COMPARISON	OPTION WITH HIGHER PRODUCTION	DIFFERENCE IN PRODUCTION (TONNES C; %)	PROPORTION OF YEARS	TOTAL LENGTH OF CONSECUTIVE PERIODS (YR)	MEAN LENGTH OF CONSECUTIVE PERIODS (YR)
W22 vs W32	W32	15 (4.95%)	54.1%	50	2.94
	W22	-4 (-1.04%)	45.9%	37	2.85
W22 vs W36	W36	20 (6.37%)	55.7%	55	3.24
	W22	-10 (-2.56%)	44.3%	38	2.71
W22 vs W40	W40	20 (6.26%)	57.4%	54	3.38
	W22	-8 (-2.08%)	42.6%	32	2.46
W32 vs W36	W36	13 (3.68%)	48.4%	45	3.21
	W32	-11 (-2.95%)	51.6%	50	3.12
W32 vs W40	W40	13 (3.65%)	48.4%	45	3
	W32	-8 (-2.4%)	51.6%	50	2.94
W36 vs W40	W40	6 (1.79%)	54.1%	54	3
	W36	-4 (-1.29%)	45.9%	41	2.56

4.2 COMPARISONS WITHIN REGULATED YEARS

When limited only to years in which constraints were operating (i.e., any year where a flow option scenario had a modelled maximum daily discharge below the designated constraints limit), the proportion of years in which the three alternative flow scenarios (i.e., the W32, W36, and W40 flow options) produced estimates of large native fish production potential above the 25th percentile of the base case, W22 option estimates was only slightly greater than that across all years, at approximately 79 – 82% (Figure 4.6). In contrast with the modelled results across all years, however, the W32 option produced greater estimates more often than the W36 and W40 flow options (Figure 4.6).

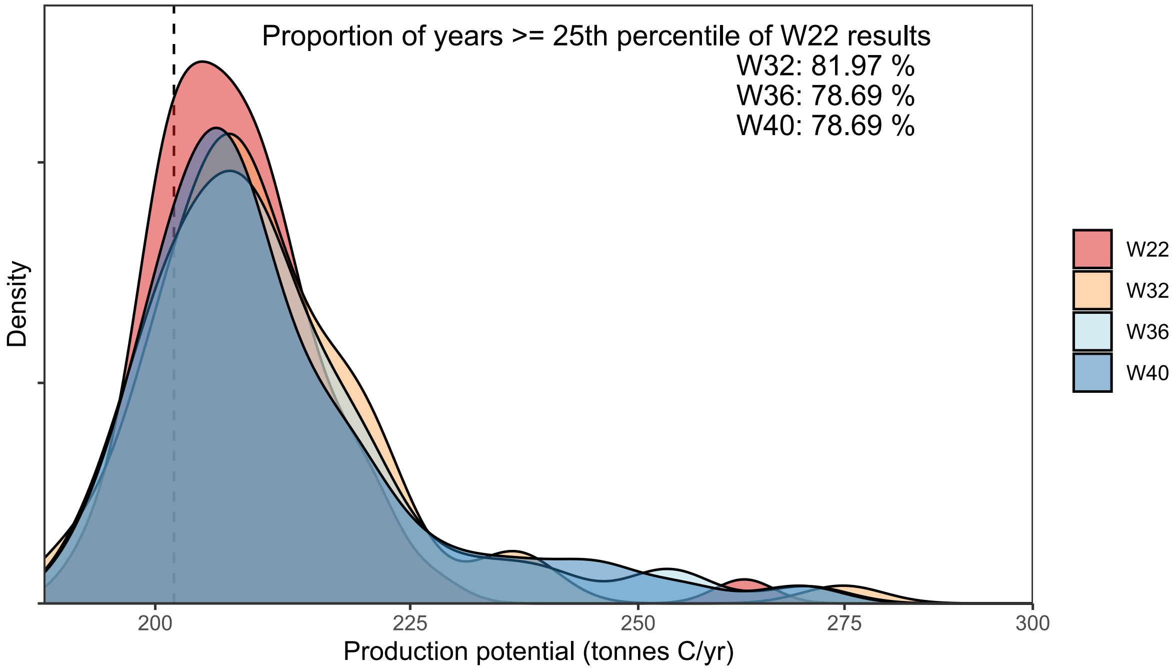

Figure 4.6 Density distributions of model results for the production potential of large native fish from the four different modelled timeseries, across years in which constraints were operating (i.e., any year where a flow option had a modelled maximum daily discharge below its designated constraints limit), for the Burrinjuck Dam to Hay Weir area. Dashed line indicates 25th percentile of W22 results (i.e., value above which 75% of yearly estimates from the W22 model occur). Statistics shown in the top right indicate the percentage of years for each of the modelled flow option scenarios where estimates exceeded the 25th percentile of W22 results. Note the use of nonlinear scale on the x axis (pseudo-$\log_{10}$ transformation) to aid in discrimination between density distributions.

When limited to years in which constraints were potentially operating, there was only a small increase (1.2%) in the 25th percentile of production potential estimates from the W22 base option (202 tonnes C) to the W32 option (204 tonnes C), with lower increases for the W36 (0.7%) and W40 (0.5%) option, respectively (Figure 4.4). In contrast to the comparison across all years, the median yearly production potential across flow options only increased by a very small proportion (0.6%) from the W22 base option (206.5 tonnes C) to the W36 option (207.7 tonnes C), while the increase in median production potential of the W36 (0.2%) and W40 (0.1%) options was smaller again (Figure 4.7).

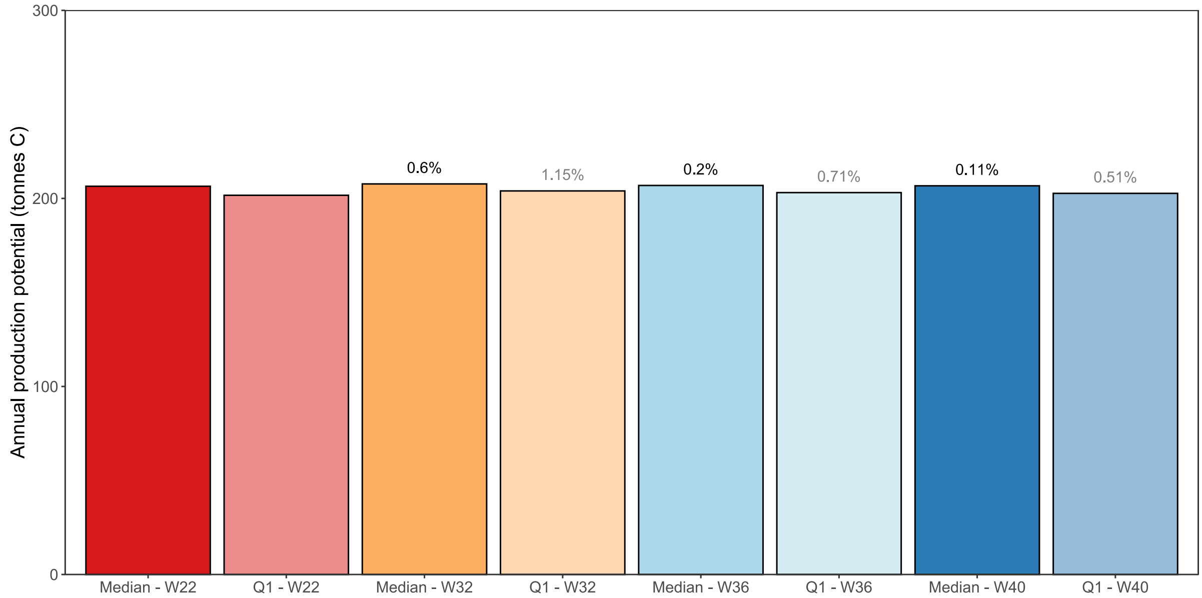

Figure 4.7 Median and 25th percentile (i.e., “Q1”) of annual production potential estimates, across in which constraints might be operating (i.e., any year where a flow option scenario had a modelled maximum daily discharge below the nominal constraints limit), for the different flow options associated with the Burrinjuck-Hay area. Numbers above the W32, W36, and W40 statistics indicate the difference between that value and the corresponding statistic for the W22 option (a.k.a. “base” option).

When limited only to years in which constraints might operate, the alternative flow option models had greater estimates of production potential approximately as often as the base, W22 option model (Figure 4.8), with proportions ranging from 49 – 51% of years (Table 4.2). However, the mean difference between options was greater (5 – 10 tonnes C/yr; 2.7 – 4.9% of lower estimate) in favour of the alternative flow option models in comparison with the base W22 option (2 tonnes C/yr and 1.0 – 1.1%; Table 4.2). The alternative flow options produced greater estimates in consecutive years more often (13 – 15 total years) and for longer on average (mean 2.2 – 3.0 years of consecutive periods) than the W22 flow option (8 - 10 years; mean 2 year periods). In contrast to the River Murray project areas, therefore (see: Siebers et al. 2022), the alternative flow options produced greater estimates less often (and to a lower magnitude) in comparison with the base option when the focus was on years where constraints might be operating, rather than the entire dataset.

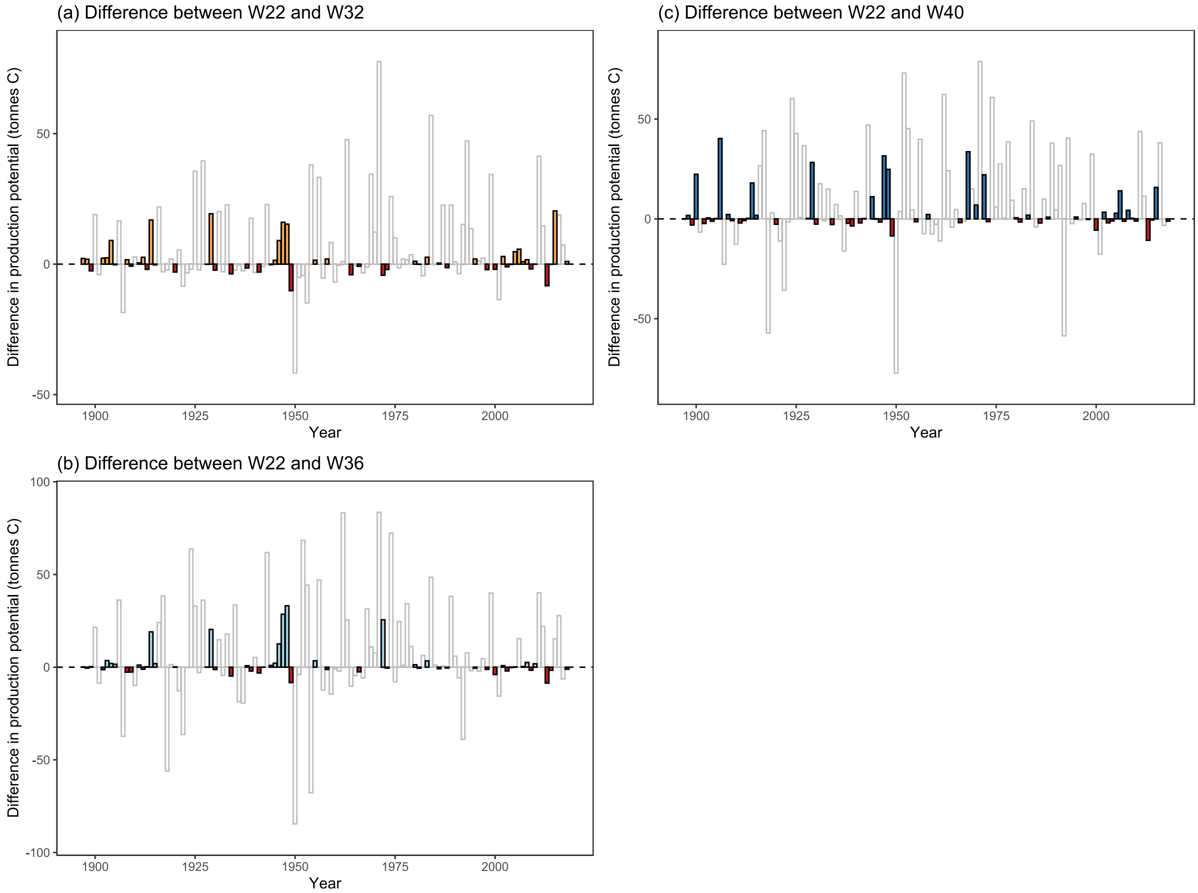

Figure 4.8 Difference between model estimates for production potential (tonnes C), for the Burrinjuck Dam to Hay Weir area, from the W22 (a.k.a. "base") flow option and (a) the W32 (i.e., W32 – W22 estimate), (b) W36, and (c) W40 model estimates, across years in which constraints might be operating (i.e., any year where a flow option scenario had a modelled maximum daily discharge below the nominal constraints limit). Positive values indicate that estimates are greater for the flow option with the higher constraints limit, and *vice versa*.

The W36 flow option produced greater estimates of production potential more often (54% of years) than both the W32 and W40 models (46% of years; Table 4.2), although these differences were not always of greater peak magnitude (Figure 4.9). The mean and proportional difference in comparisons between flow options were therefore only slightly in favour of the flow option with the nominally higher constraints limit (2 – 4 tonnes C/yr; and 1.0 – 2.8 % of the lower estimate) rather than the alternative (-2 – -3 tonnes C/yr and -1 – -1.4 %; Table 4.4). Differences in the consistency and frequency of these greater production potential years were also very similar across comparisons, in both the total number of consecutive years (14 – 17 years) and mean length of consecutive periods (mean 2.0 – 3.4 year periods; Table 4.4). Consequently, while the W36 flow option had slightly higher estimates and number of years with greater production potential, these differences were not large (Figure 4.9, Table 4.4).

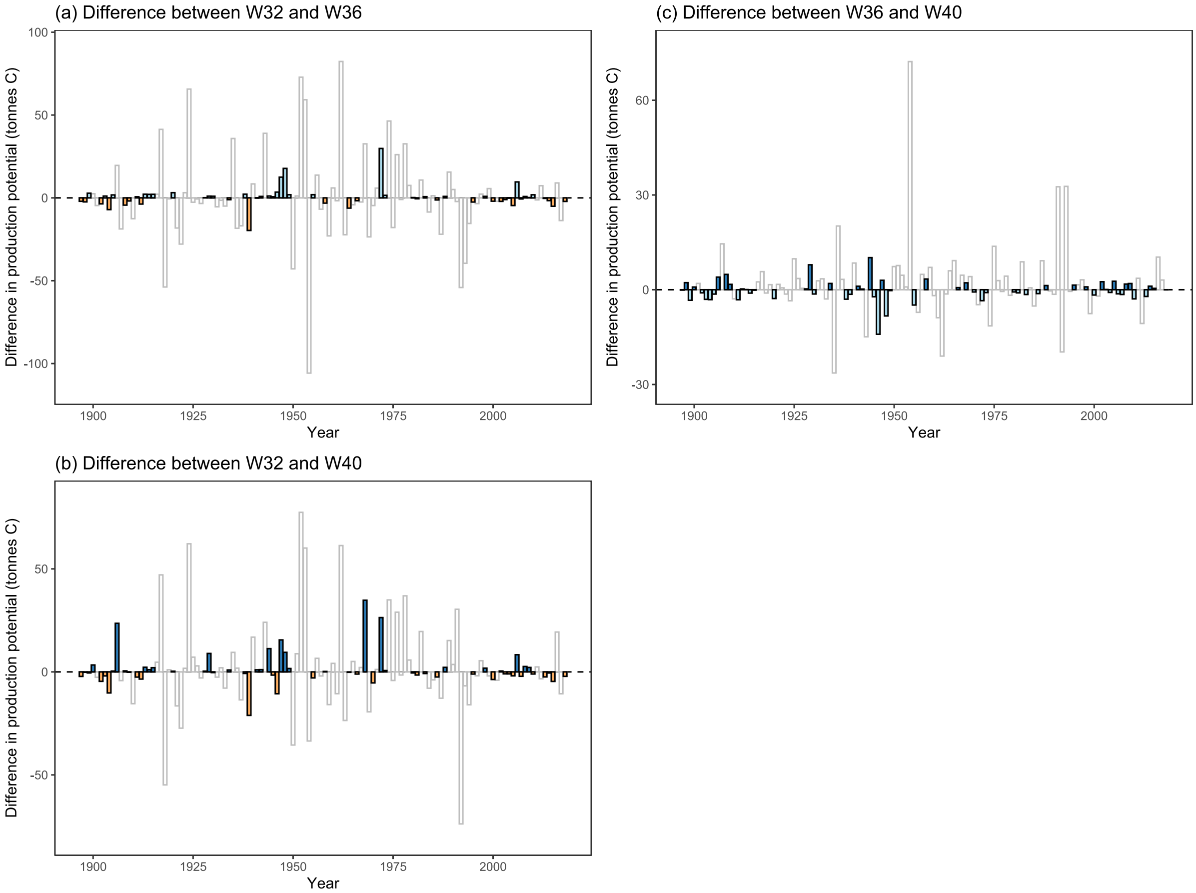

Figure 4.9 Difference between model estimates for production potential (tonnes C), for the Burrinjuck Dam to Hay Weir area, from the W32 flow option and (a) the W36 (i.e., W36 – W32 estimate) and (b) W40 model estimates; and (c) difference between W36 and W40 flow option model estimates, across years in which constraints might be operating (i.e., any year where a flow option scenario had a modelled maximum daily discharge below the nominal constraints limit). Positive values indicate that estimates are greater for the flow option with the higher constraints limit, and *vice versa*.

Table 4.2 For the Burrinjuck Dam to Hay Weir area, summary statistics for differences in pairwise comparisons of estimates between flow option estimates, in years where constraints are operating (i.e., either timeseries has a maximum daily discharge below the constraints limit for that year). Shown for the subset of results in which one option produced greater differences (e.g., when the W32 flow option model produced higher estimates than the W22 model) are the mean differences in large native fish production potential estimates (in tonnes C/yr for both absolute values, and percentage of difference relative to the lower model estimate), proportion of years (%) in which the given flow option model produced higher estimates, the total number of years in which the given flow option model produced higher estimates across consecutive years (i.e., >= 2 years), and the mean length of those consecutive periods.

COMPARISON	OPTION WITH HIGHER PRODUCTION	DIFFERENCE IN PRODUCTION (TONNES C; %)	PROPORTION OF YEARS	TOTAL LENGTH OF CONSECUTIVE PERIODS (YR)	MEAN LENGTH OF CONSECUTIVE PERIODS (YR)
W22 vs W32	W32	5 (2.67%)	50.0%	15	3
	W22	-2 (-1.03%)	50.0%	8	2
W22 vs W36	W36	6 (3.03%)	50.9%	15	3
	W22	-2 (-1.02%)	49.1%	8	2
W22 vs W40	W40	10 (4.85%)	49.2%	13	2.17
	W22	-2 (-1.1%)	50.8%	10	2
W32 vs W36	W36	4 (1.71%)	53.6%	17	3.4
	W32	-3 (-1.44%)	46.4%	11	2.75
W32 vs W40	W40	6 (2.84%)	45.0%	16	2.29
	W32	-3 (-1.34%)	55.0%	20	2.5
W36 vs W40	W40	2 (1.04%)	45.8%	14	2
	W36	-2 (-1.08%)	54.2%	18	2.25

5. Discussion

For the Murrumbidgee project area, there were increases in estimates of large native fish production potential from the base (W22) to all of the alternative flow options, increasing to approx. 11% greater median estimates under the highest (W40) constraints scenario. However, the difference between flow options in lower estimates (25^{th} percentiles) was minor. Further, the difference in production potential estimates between different flow option time series (whether median or 25^{th} percentile) also became inconsequential when estimates of production potential were compared only across years in which unregulated events would not have impacted estimates (i.e., maximum daily discharge was below the constraints limits being compared). These results suggest that unregulated flow events may have been contributing significantly to differences in median estimates between the flow options.

Given that the River Murray models produced noticeable (if still relatively small) differences between different flow options in potential "constraints years" (Siebers et al. 2022), the underlying nature of the spatial inundation models or modelled discharge time series might be the driver of the lack of differences here. In the CARM model for the mid-Murrumbidgee (see: Section 3, Figure 3.4), 22.1% of the total area has an inundation threshold of between 22,000 and 40,000 ML/day, compared with 4.8% of the area with an inundation threshold below 22,000 ML/day. Consequently, flows between the proposed constraints limits may have a relatively smaller influence than in the River Murray models, where – for comparison – the area of Barmah-Millewa Forest inundated by flows between 15,000 and 40,000 ML/day is estimated to be approximately 8.4 times that which is inundated by flows less than 15,000 ML/day (Overton et al. 2006). Similarly, the proportion of the mid-Murrumbidgee project area that has an inundation threshold of above 40,000 ML/day (i.e., the highest proposed constraint limit at Wagga Wagga) is 73.1%. Under the modelled flow options being assessed for the Murrumbidgee River, there are noticeable differences between the number of days exceeding this limit for each of the scenarios (Figure 5.1).

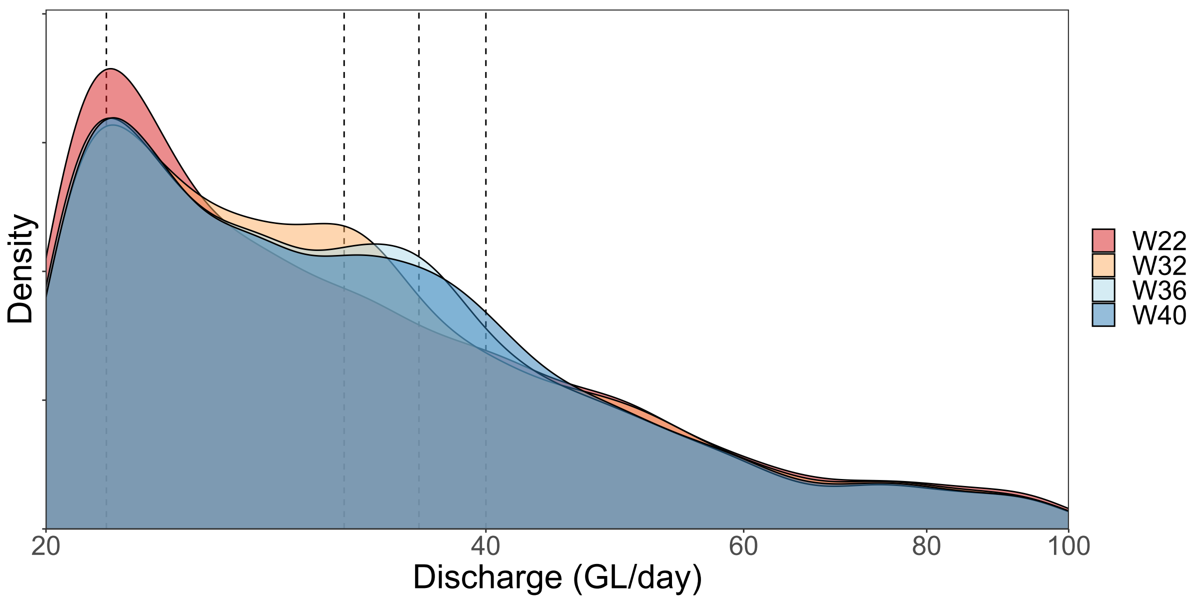

Figure 5.1 Density distributions of modelled discharge (in GL/day, for all days where flow exceeded 22 GL/day), across all years (1986-2018), for the Wagga Wagga flow gauge: flows under 20 GL/day are excluded here to emphasise the differences in high flow events. Dashed lines indicate 22, 32, 36, and 40 GL/day (i.e., different constraints limits at Wagga Wagga). Note the use of nonlinear scale on the x axis (pseudo-$\log_{10}$ transformation) to aid in discrimination between density distributions.

Given that (i) all other drivers being held constant, higher discharge flow events inundate a wider area for longer periods (Siebers et al. 2022); (ii) the area inundated above the highest constraints limit proposed here is relatively large, and; (iii) the difference in production potential estimates between flow options here was typically small; the magnitude and frequency to which each flow option predicted greater production potential than the alternatives is thus likely to have been strongly influenced by a greater magnitude of unregulated flow events. These patterns are reflected in the slightly greater frequency of production potential estimates for the W36 and W40 flow options within the middle of the range (approximately 500 – 700 tonnes C/yr) of the comparisons across all years (see: Figure 4.2). In this context, it is notable that this pattern is not as clear in potentially regulated years (see: Figure 4.6).

The River Murray project identified several areas in which the modelling approach described here might be improved (Siebers et al., 2022), e.g.:

- Improved modelling of surface water persistence on floodplains. The current approach incorporates a universal decay factor to discharge-inundation relationships, but the actual persistence of surface water might vary considerably based on antecedent flow patterns and local topography.
- Improved consideration of low flows: particularly with respect to intermittent cessation of flow or drying events, which are not currently modelled.
- Consideration of the timing or seasonality of flow events, as well as previous patterns of inundation, which can have substantial effects on the type of primary production and whether is transferred from basal resources up through food webs. Currently, only general seasonal effects on rates of productivity (i.e., temperature dependence) are modelled.
- More mechanistic information on ecological efficiencies (the transfer of energy through food web links), which are likely to vary across different dominant vegetation types, light and temperature regimes, and decomposition processes through effects on the composition (a.k.a., “quality”) of basal resources.

For a full discussion of these potential improvements, we refer to the River Murray report (Siebers et al., 2022). For the Murrumbidgee project area, however, the most relevant area to improve on may be that there is no hysteresis built into production potential estimates, i.e., production is assumed to be instantaneously generated and available to all components of a food web upon inundation (Siebers et al., 2022). In practice, the variation in these processes is likely to be averaged out when production potential is assessed at an annual timescale, particularly given the overwhelming influence of large, unregulated flow events on annual sums of inundation and production (Bond et al. in prep). However, our model is thus unable to assess the likelihood that production generated during a flood pulse will continue to support aquatic food webs for the months or years following the event. The low definition of spatial inundation models at lower discharge (< 3,000 – 5,000 ML/day for the River Murray models; < 16,000 ML/day for the mid-Murrumbidgee model) also greatly constrains our ability to predict variation in production at low flows (Siebers et al. 2022), which may be of great importance in maintaining aquatic populations in the periods between flood pulses. We therefore stress again here that: (i) the estimates we give are of a potential upper ceiling of annual energy availability, rather than a corollary to population abundance; (ii) delivering more flow, and for longer durations, will provide the greatest net gain in aquatic production; but (iii) maintaining production during times of the year when floodplain connectivity might be more ecologically important (e.g., maintaining primary production during lower flow periods, timing of flows for fish spawning) will also be of vital importance to the ecological integrity of the Southern Connected Basin.

6. References

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Appendices

APPENDIX 1: SUMMARY OF RIVER MURRAY MODELLING (PART 1)

Part 1 of this project (Siebers et al. 2022) applied an ecosystem energetics modelling approach to estimate (i) the extent of floodplain inundation, or integral of area and time inundated, and (ii) the production potential of riverine food webs, or potential upper limit of energy available for production, for the River Murray and its associated floodplains from Hume Dam to the confluence of the Murray and Wakool rivers. To do so, inundation was first modelled from several, contrasting modelled daily discharge timeseries (in ML/day, representing different flow options tailored to variable constraints limits at Hume Dam and Yarrawonga Weir) in combination with spatial datasets on daily discharge thresholds associated with areas of the River Murray catchment (the River Murray Floodplain Inundation Model, or RiM-FIM; Overton et al., 2006). These inundation estimates were multiplied by areal rates of productivity (g C produced per m^2 per year) for several food web basal resources (see: Section 3.1), derived from the literature on aquatic productivity for the Murray-Darling Basin and Australian rivers more generally (Siebers et al. 2022), to provide annual estimates of basal production. These estimates were then passed into a modelled food web (see: Section 3.1) which simulates the transfer of energy (as carbon, C) between different taxonomic and functional groups within the food web. Finally, estimates of production potential were extracted for a single group (large native fish) as an indicator of the overall effects of variation in the underlying discharge timeseries on food web dynamics.

Estimates of production potential for large native fish for the flow options with nominally higher constraints limits were frequently greater than in the base scenario, but only a small proportion (0.6 – 3.1%) of the net potential difference without development, reflecting the substantial influence of large, unregulated flow events on annual estimates of floodplain inundation. However, differences between flow option scenarios were more pronounced in years where maximum daily discharge was constrained under both of the flow options being compared; i.e., years in which differences in production potential would be directly affected by constraints limits rather than driven by unregulated flows. In these years, the energetics model predicted that the alternative flow options would still produce greater estimates of production potential more frequently than the base option, but that these estimates represented a higher proportion (8 – 20%) of the potential difference without development in these years. Overall, therefore, the model results emphasised that while the increase in production potential under different flow options may be small in comparison with overall estimates without development, the effect of increasing constraints limits may be disproportionately greater in years where there are no unregulated flow events.

APPENDIX 2: SUMMARY OF LOWER MURRUMBIDGEE MODELLING

The following figures and tables summarise the results of modelling in the Lower Murrumbidgee area (Hay Weir to Murrumbidgee/Murray confluence; see: Figure 3.4), based on the Murrumbidgee extension to the River Murray Floodplain Inundation Model (RiMFIM; Sims et al. 2014), which were excluded from core results due to low specificity in discharge-inundation relationships (see: Section 3).

ALL MODELLED YEARS

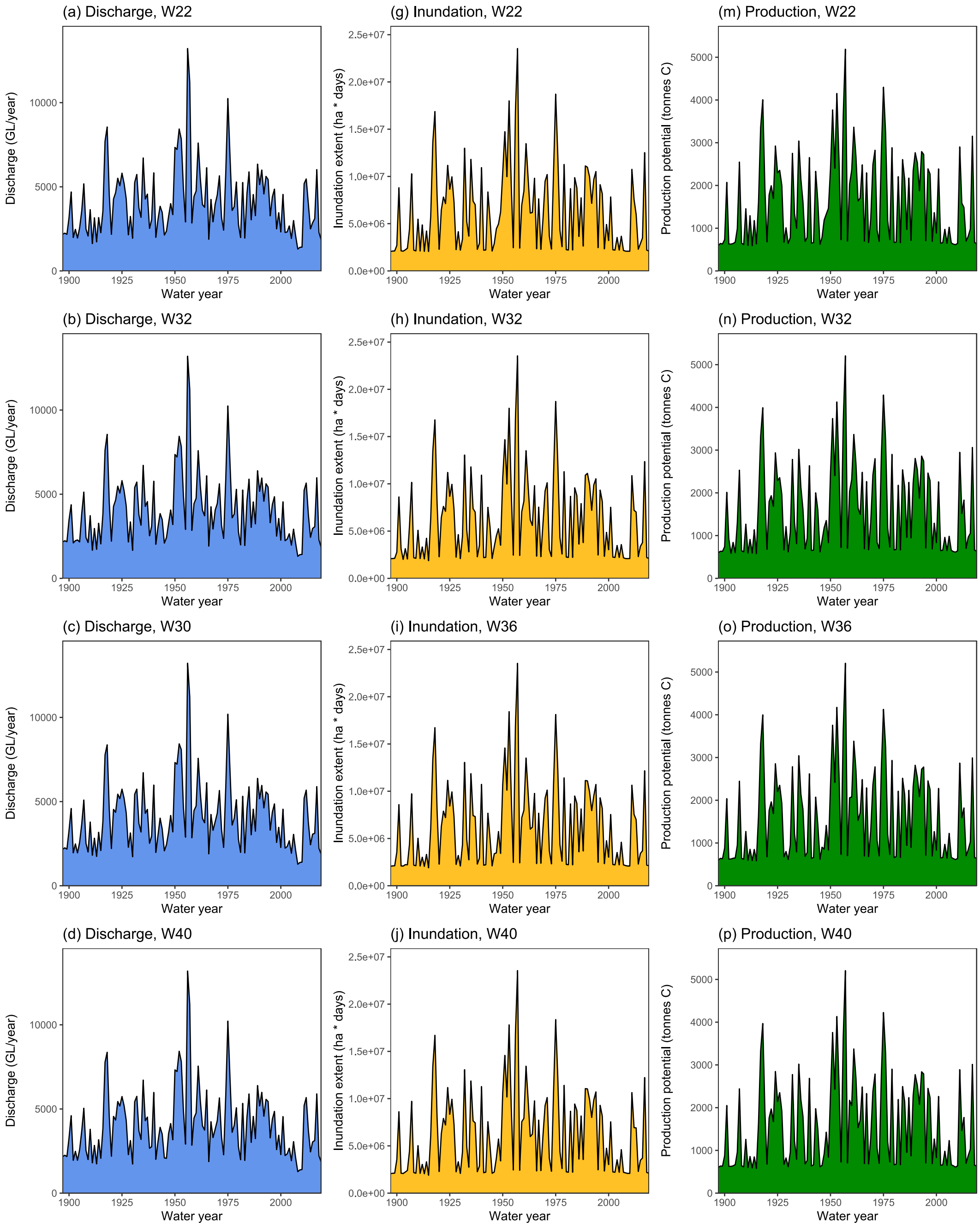

Figure A2.1 For the Hay Weir to Murray confluence area, for each year from 1896-2018, comparisons between each of the different modelled flow options (W22, W32, W36, and W40) for (a-d) annual cumulative discharge at Wagga Wagga; (e-j) annual estimated inundation extent (in ha*days); and (m-p) estimated production potential of large native fish (tonnes C).

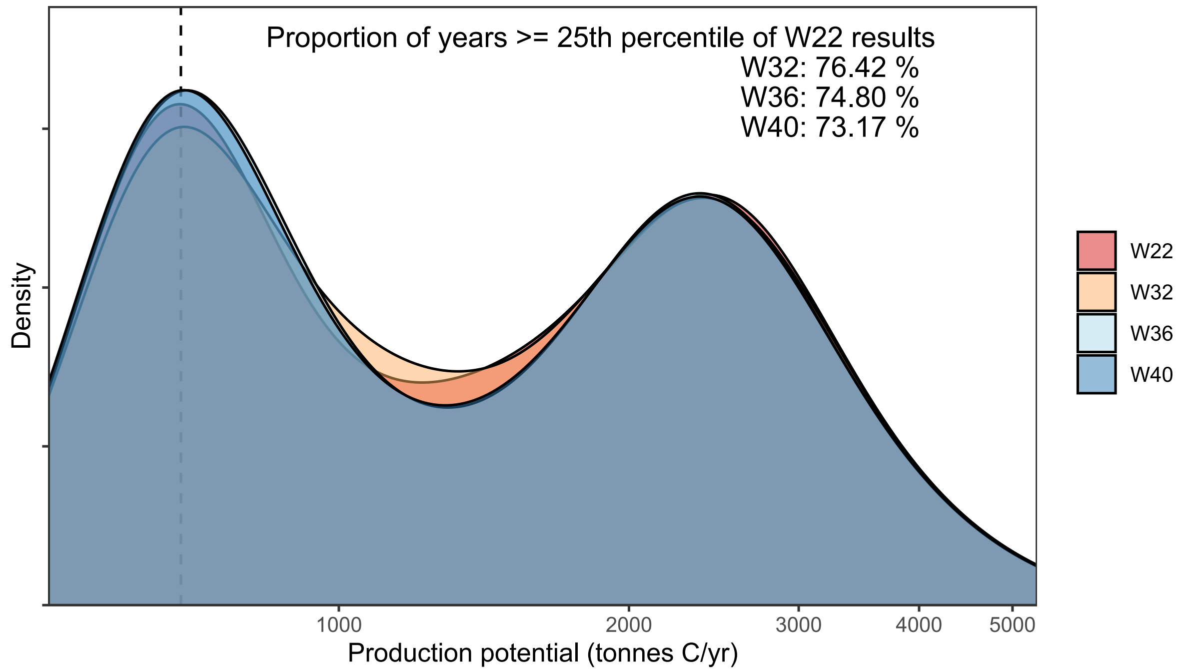

Figure A2.2 Density distributions of model results for the production potential of large native fish from the four different modelled timeseries, across all years (1986-2018), for the Hay Weir to Murray confluence area. Dashed line indicates 25th percentile of W22 results (i.e., value above which 75% of yearly estimates from the W22 model occur). Statistics shown in the top right indicate the percentage of years for each of the modelled flow option scenarios where estimates exceeded the 25th percentile of W22 results. Note the use of nonlinear scale on the x axis (pseudo-$\log_{10}$ transformation) to aid in discrimination between density distributions.

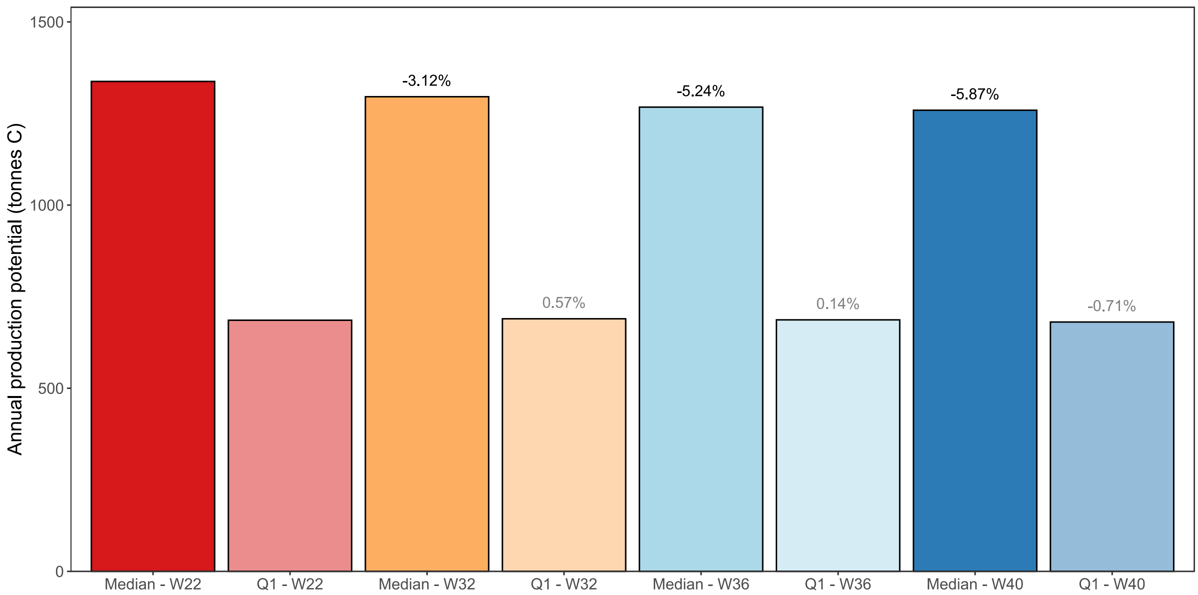

Figure A2.3 Median and 25th percentile (i.e., “Q1”) of annual production potential estimates, across all years, for the different flow options associated with the Hay-Murray area. Numbers above the W32, W36, and W40 statistics indicate the difference between that value and the corresponding statistic for the W22 option (a.k.a. “base” option).

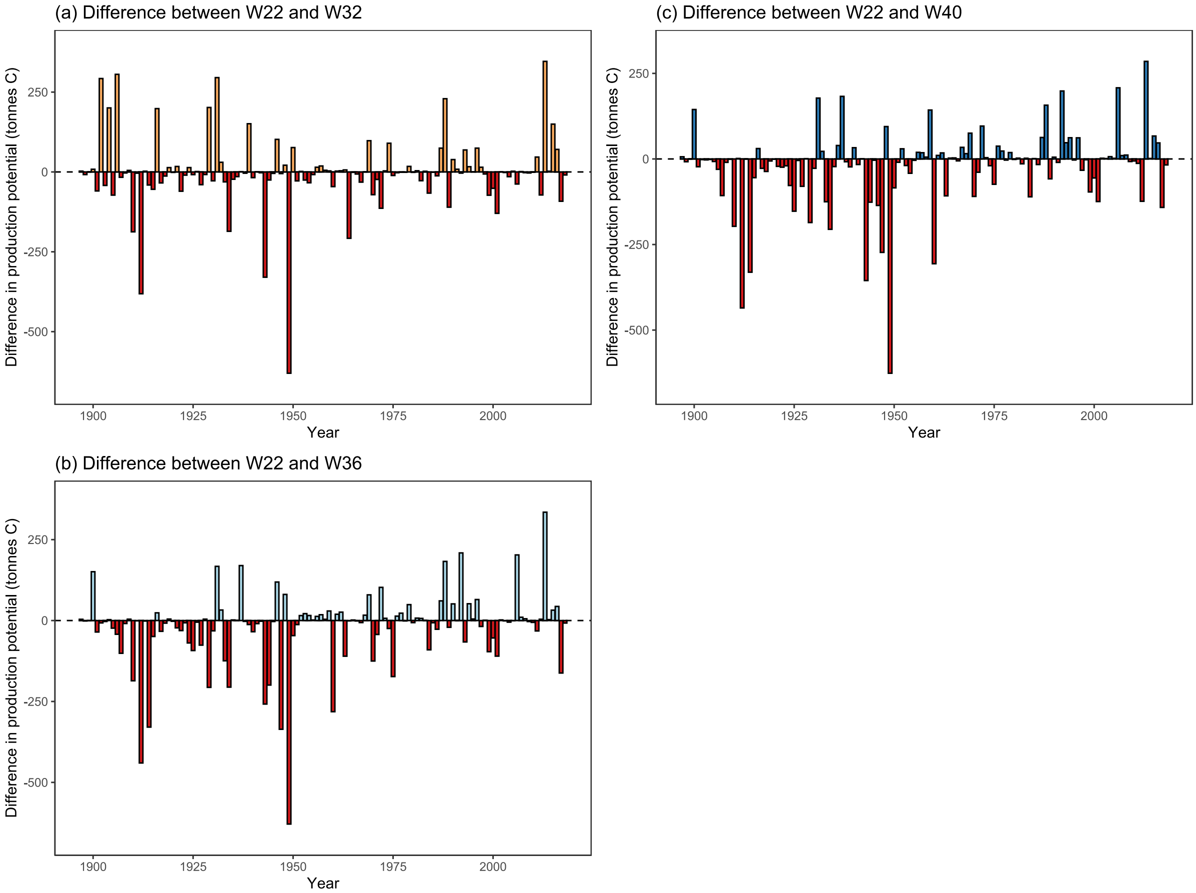

Figure A2.4 Difference between model estimates for production potential (tonnes C), for the Hay Weir to Murray confluence area, from the W22 (a.k.a. "base") flow option and (a) the W32 (i.e., W32 – W22 estimate), (b) W36, and (c) W40 model estimates, across all modelled years. Positive values indicate that estimates are greater for the flow option with the higher constraints limit, and *vice versa*.

Table A2.1 For the Hay Weir to Murray confluence area, summary statistics for differences in pairwise comparisons of estimates between flow option estimates for all modelled years. Shown for the subset of results in which one option produced greater differences (e.g., when the W32 flow option model produced higher estimates than the W22 model) are the mean differences in large native fish production potential estimates (in tonnes C/yr for both absolute values, and percentage of difference relative to the lower model estimate), proportion of years (%) in which the given flow option model produced higher estimates, the total number of years in which the given flow option model produced higher estimates across consecutive years (i.e., >= 2 years), and the mean length of those consecutive periods.

COMPARISON	OPTION WITH HIGHER PRODUCTION	DIFFERENCE IN PRODUCTION (TONNES C; %)	PROPORTION OF YEARS	TOTAL LENGTH OF CONSECUTIVE PERIODS (YR)	MEAN LENGTH OF CONSECUTIVE PERIODS (YR)
W22 vs W32	W32	63 (6.29%)	43.4%	26	2.89
	W22	-53 (-3.68%)	56.6%	51	2.83
W22 vs W36	W36	47 (4.02%)	43.4%	40	3.08
	W22	-75 (-4.96%)	56.6%	62	3.26
W22 vs W40	W40	51 (4.09%)	41.0%	35	2.92
	W22	-76 (-5.09%)	59.0%	60	4
W32 vs W36	W36	36 (2.84%)	50.8%	50	2.63
	W32	-76 (-5.49%)	49.2%	45	3
W32 vs W40	W40	41 (3.29%)	42.6%	34	2.62
	W32	-68 (-5.06%)	57.4%	56	3.29
W36 vs W40	W40	16 (1.02%)	49.2%	49	2.88
	W36	-20 (-1.38%)	50.8%	53	2.79

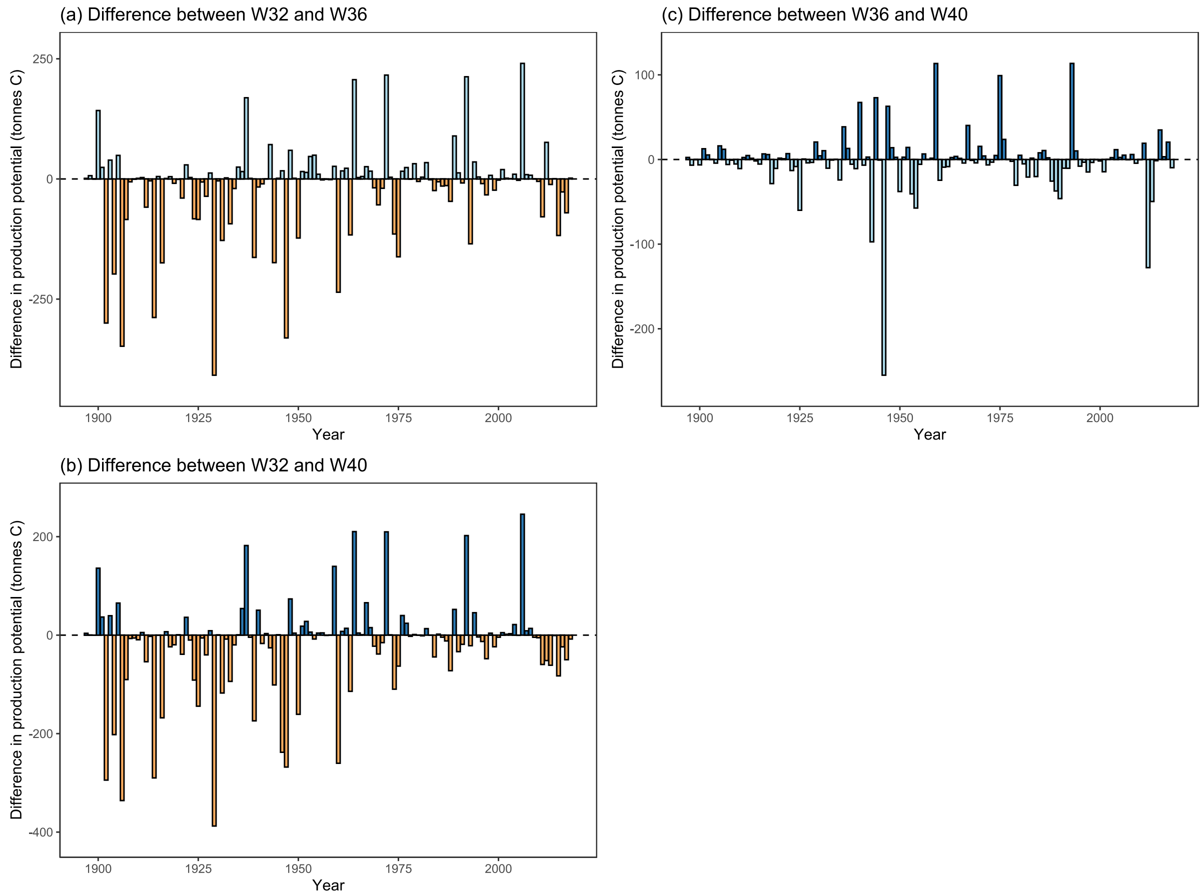

Figure A2.5 Difference between model estimates for production potential (tonnes C), for the Hay Weir to Murray confluence area, from the W32 flow option and (a) the W36 (i.e., W36 – W32 estimate) and (b) W40 model estimates; and (c) difference between W36 and W40 flow option model estimates, across all modelled years. Positive values indicate that estimates are greater for the flow option with the higher constraints limit, and *vice versa*.

YEARS WITH CONSTRAINTS OPERATING

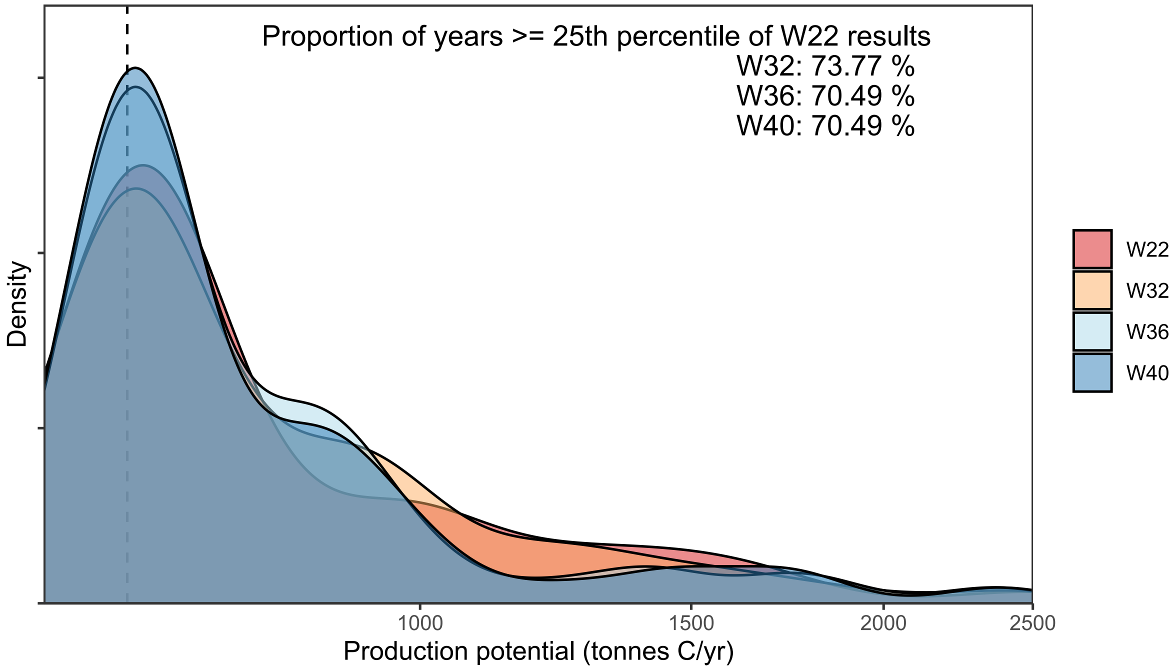

Figure A2.6 Density distributions of model results for the production potential of large native fish from the four different modelled timeseries, across years in which constraints were operating (i.e., any year where a flow option had a modelled maximum daily discharge below its designated constraints limit), for the Hay Weir to Murray confluence area. Dashed line indicates 25th percentile of W22 results (i.e., value above which 75% of yearly estimates from the W22 model occur). Statistics shown in the top right indicate the percentage of years for each of the modelled flow option scenarios where estimates exceeded the 25th percentile of W22 results. Note the use of nonlinear scale on the x axis (pseudo-$\log_{10}$ transformation) to aid in discrimination between density distributions.

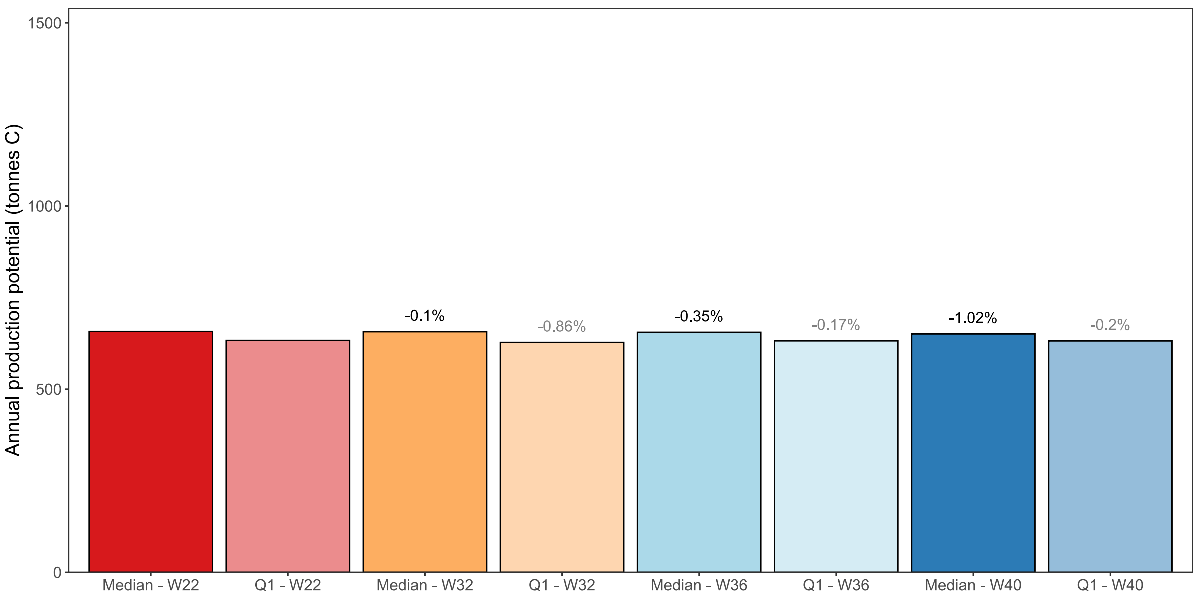

Figure A2.7 Median and 25th percentile (i.e., "Q1") of annual production potential estimates, across in which constraints might be operating (i.e., any year where a flow option scenario had a modelled maximum daily discharge below the nominal constraints limit), for the different flow options associated with the Hay-Murray area. Numbers above the W32, W36, and W40 statistics indicate the difference between that value and the corresponding statistic for the W22 option (a.k.a. "base" option).

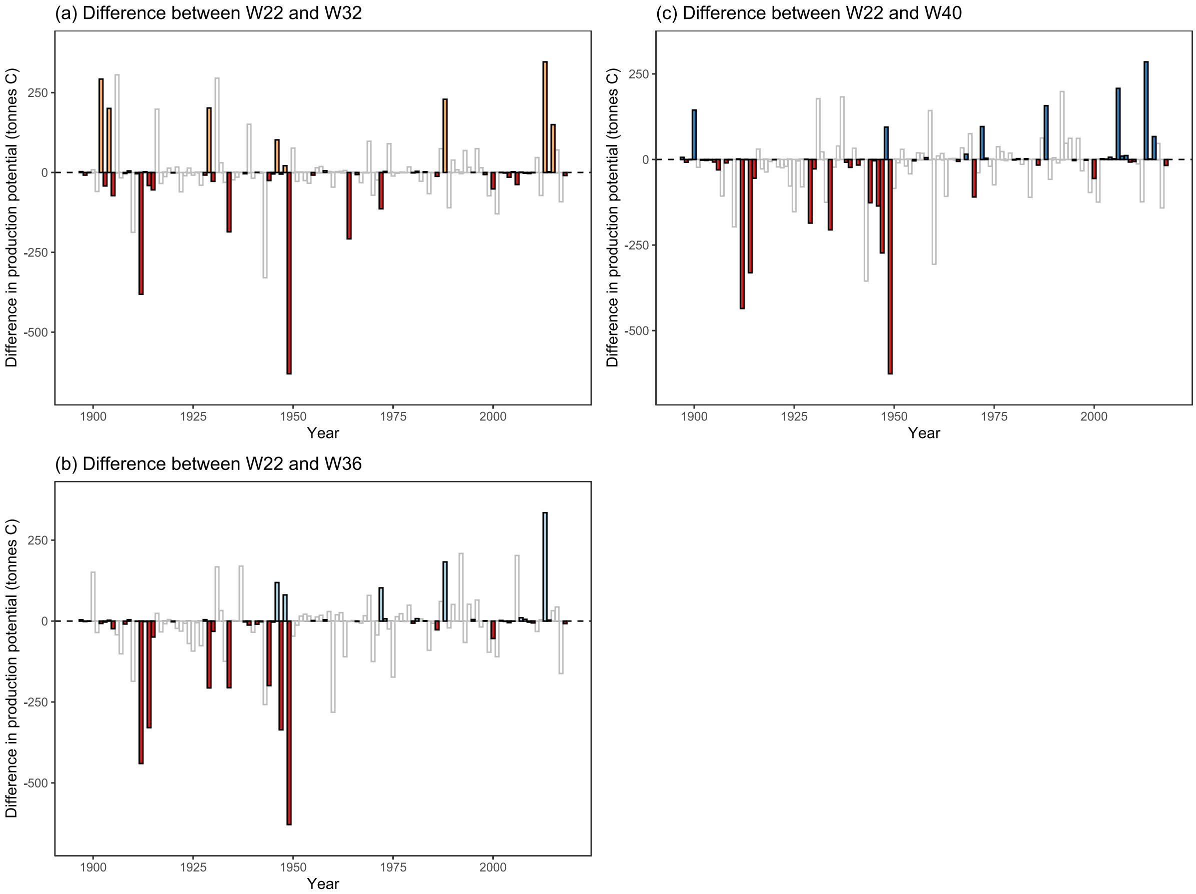

Figure A2.6 Difference between model estimates for production potential (tonnes C), for the Hay Weir to Murray confluence area, from the W22 (a.k.a. "base") flow option and (a) the W32 (i.e., W32 – W22 estimate), (b) W36, and (c) W40 model estimates, across years in which constraints might be operating (i.e., any year where a flow option scenario had a modelled maximum daily discharge below the nominal constraints limit). Positive values indicate that estimates are greater for the flow option with the higher constraints limit, and *vice versa*.

Table A2.2 For the Hay Weir to Murray confluence area, summary statistics for differences in pairwise comparisons of estimates between flow option estimates, in years where constraints are operating (i.e., either timeseries has a maximum daily discharge below the constraints limit for that year). Shown for the subset of results in which one option produced greater differences (e.g., when the W32 flow option model produced higher estimates than the W22 model) are the mean differences in large native fish production potential estimates (in tonnes C/yr for both absolute values, and percentage of difference relative to the lower model estimate), proportion of years (%) in which the given flow option model produced higher estimates, the total number of years in which the given flow option model produced higher estimates across consecutive years (i.e., >= 2 years), and the mean length of those consecutive periods.

COMPARISON	OPTION WITH HIGHER PRODUCTION	DIFFERENCE IN PRODUCTION (TONNES C; %)	PROPORTION OF YEARS	TOTAL LENGTH OF CONSECUTIVE PERIODS (YR)	MEAN LENGTH OF CONSECUTIVE PERIODS (YR)
W22 vs W32	W32	75 (9.09%)	38.9%	3	3
	W22	-60 (-5.28%)	61.1%	13	2.17
W22 vs W36	W36	44 (4.48%)	37.7%	6	2
	W22	-79 (-6.79%)	62.3%	22	2.44
W22 vs W40	W40	49 (5.11%)	39.0%	12	4
	W22	-76 (-6.65%)	61.0%	23	2.88
W32 vs W36	W36	31 (3.32%)	54.0%	15	2.5
	W32	-84 (-7.93%)	46.4%	7	2.33
W32 vs W40	W40	40 (4.44%)	45.0%	12	2.4
	W32	-81 (-7.68%)	55.0%	17	2.43
W36 vs W40	W40	11 (1.19%)	49.2%	17	2.83
	W36	-15 (-1.66%)	50.8%	17	2.12

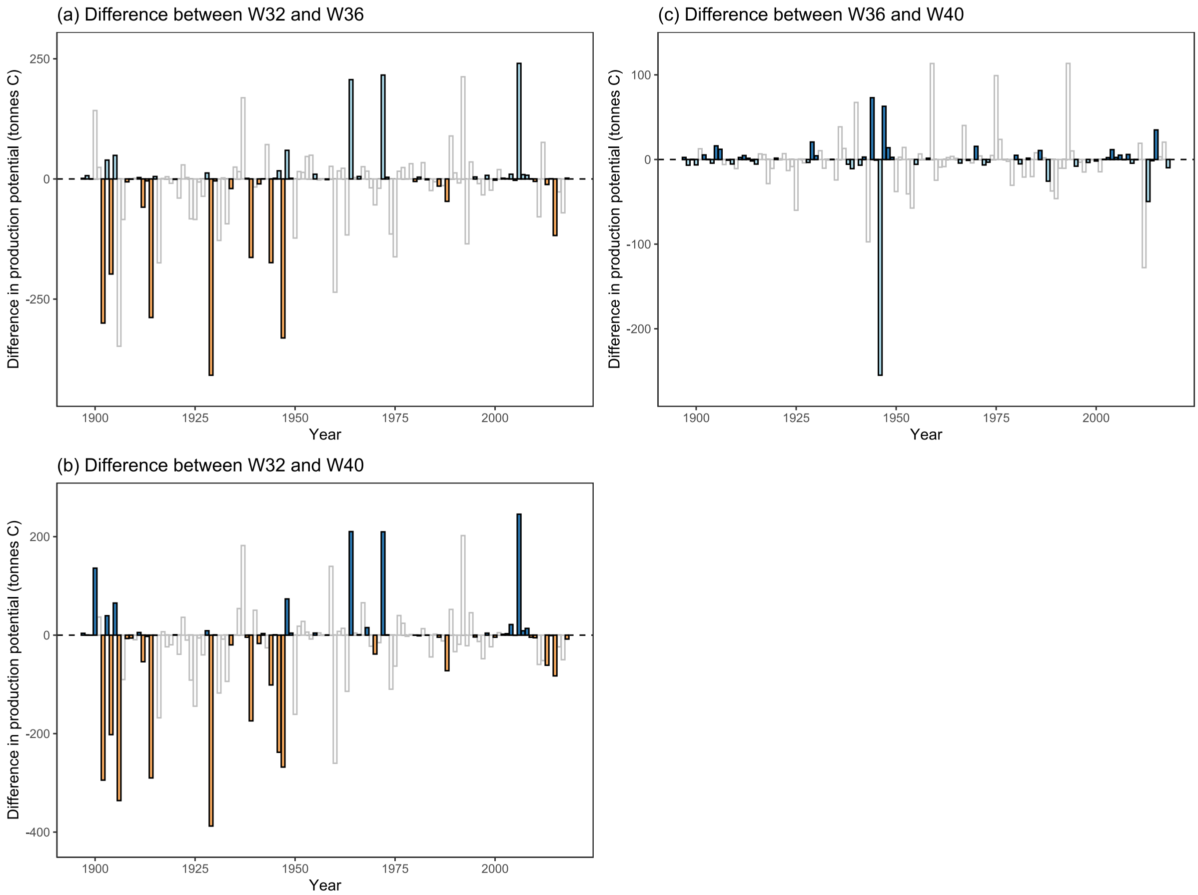

Figure A2.7 Difference between model estimates for production potential (tonnes C), for the Hay Weir to Murray confluence area, from the W32 flow option and (a) the W36 (i.e., W36 – W32 estimate) and (b) W40 model estimates; and (c) difference between W36 and W40 flow option model estimates, across years in which constraints might be operating (i.e., any year where a flow option scenario had a modelled maximum daily discharge below the nominal constraints limit). Positive values indicate that estimates are greater for the flow option with the higher constraints limit, and *vice versa*.