LA TROBE UNIVERSITY

PRODUCTION CONDITION PREDICTIVE MODELLING

Part 1: River Murray, Hume to Wakool junction

Prepared for the Department of Planning and Environment NSW

October 2022

ENQUIRIES
Andre Siebers
La Trobe University
Victoria 3086

T 0481 733 464
E a.siebers@latrobe.edu.au
latrobe.edu.au/freshwater-ecosystems

Document history and status

VERSION	DATE ISSUED	REVIEWED BY	APPROVED BY	REVISION TYPE
Draft	24/02/2022	DPE & external review	AS	Science
Final Report	28/10/2022	AS, NB, DPE	AS	

Report citation

Siebers AR, Crook D, Silvester E, Bond N (2022) Production Condition Predictive Modelling. Part 1: River Murray, Hume to Wakool Junction. CFE Publication 281, October 2022, 76pp.

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.

Table of contents

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 River Murray (including the Edward/Kolety-Wakool system) and its associated floodplains from Hume Dam to the junction of the Murray and Wakool rivers under different flow options.

The ecosystem energetics modelling framework applied here first predicts floodplain inundation extent, or the integral of area and time inundated, across the modelled areas 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 differences in 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 constraints options. Model predictions were compared across two project areas: (i) the River Murray and its floodplains from Hume Dam to Yarrawonga Weir; and (ii) the River Murray and its floodplains from Yarrawonga Weir to the confluence of the Murray and Wakool rivers, including the Edward/Kolety-Wakool anabranch system. The different underlying flow option timeseries considered were reflective of different flow limits for water for the environment deliveries in each area: e.g., the base case, or Y15D25 option, representing the current 15,000 ML/day flow limit at Yarrawonga and 25,000 ML/day at Doctors Point; and a number of raised flow limit options (Y25D25, Y30D30, Y40D40, and Y45D40). Each was also compared to a flow option simulating discharge without development in the system.

Estimates of production potential for large native fish for the Y30D30, Y40D40, and Y45D40 flow options were frequently greater than in the base Y15D25 scenario, but only a small proportion (0.6 – 3.1%) of the net potential difference compared to without development, reflecting the substantial impacts of river regulation on overall productivity. To provide a prediction of the direct ecological benefits of each flow option, estimates of production potential for large native fish were therefore compared across years where the largest daily flow event was affected by potential flow limits at Doctors Point or Yarrawonga (e.g., 15,000 ML/day at Yarrawonga and 25,000 ML/day at Doctors Point for the Y15D25 flow option); i.e., years in which differences in production potential would be directly affected by flow limits rather than driven by unregulated flows. In these years, the energetics model predicted that the Y30D30, Y40D40, and Y45D40 flow options would still provide greater production potential more frequently than the Y15D25 base option, but that these increases represented a higher proportion (8 – 20%) of the potential difference from without development scenario in these years. In addition, the number of consecutive years where the Y30D30, Y40D40, and Y45D40 flow options exceeded the base case option (i.e., had higher estimates for 2 or more consecutive years) increased with higher flow limits. While the increase in production potential under different flow options may therefore be small in comparison

with overall estimates without development, the effect of increasing flow limits may be disproportionately greater in years where there are no unregulated flow events.

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 (Fig. 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).
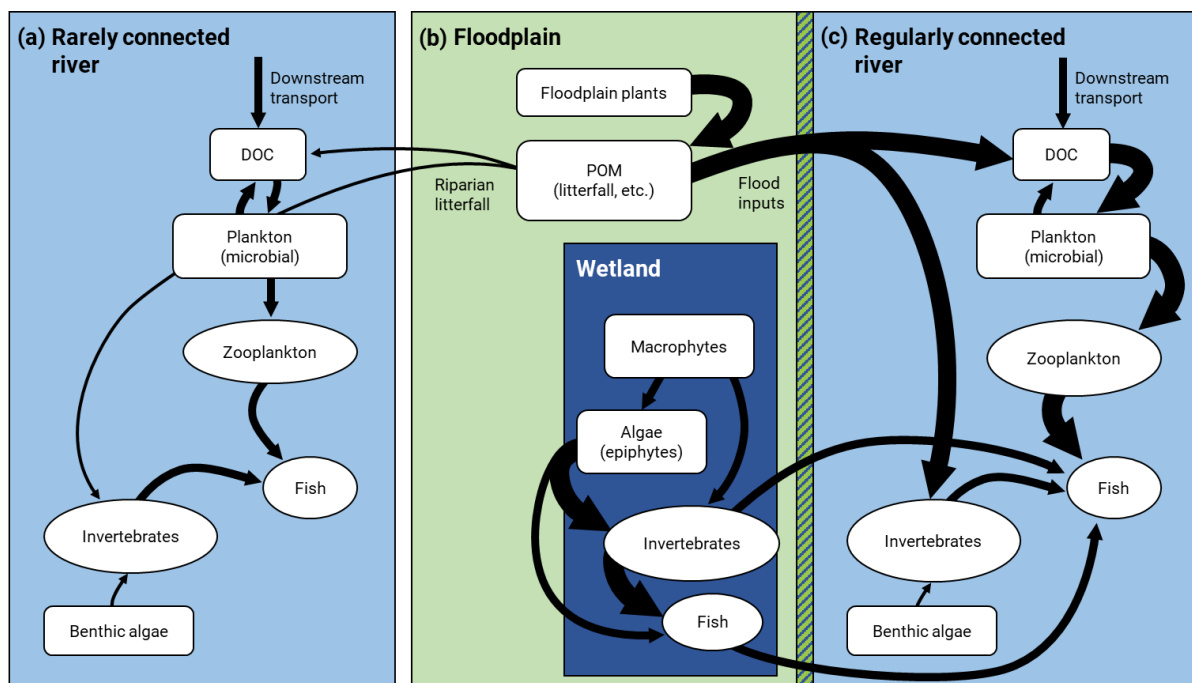

Figure 1.1 Conceptual differences in riverine food web function with respect to connections with floodplains. The size of arrows indicates different fluxes of energy (i.e., Carbon) across different food web components and environments: illustrating differences between (a) rarely connected rivers, where 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) in the Murray and Murrumbidgee River Valleys. 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 RRCP, 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).

The model developed by Bond et al. (in prep) 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 develop and adapt this operational modelling tool to the specific needs of the RRC program.

2. Project objectives

A key challenge in modelling medium to long-term production and food-web dynamics in response to changes in the flow regime is the variability in both (i) rates of productivity across different habitats and physico-chemical environments, and (ii) the effects of the physical environment, food quality, and food web structure on ecological efficiencies (i.e., what proportion of basal production is incorporated into food webs as opposed to respired or excreted). Modelled responses can be particularly sensitive to changes in the structure of the food web, especially when the addition of non-native species strongly modify pathways of energy (Bellmore et al. 2017).

For the purposes of this project, we have adopted an ecosystem energetics modelling framework similar to that described by Bond et al. (in prep). Ecosystem energetic approaches aim to quantify the total energy available to consumers and its flow through food webs. The approach uses the integral of inundated area and time as a broad estimate of total ecosystem production and models the distribution of energy through simulated food webs via an energetics model. Future changes in river and floodplain production and potential food web

responses, particularly the carrying capacity of top predators, can thus be mathematically forecast from time-series of predicted flows and inundation. An additional advantage of the approach is the ability to include (or exclude) invasive species into food-web structures, thereby assessing the difference in carrying capacity of native species with and without competitors (e.g. carp) and potentially informing the management of hydrological regimes to the advantage of native species.

The outcomes of the above modelling should provide insight into crucial factors that can constrain population sizes across time. However, it is noted that: (i) changes to the amount of food-based energy and the structure of food webs are not the sole driver of population dynamics in ecosystems, and so model outputs will not be an actual estimate of population size; (ii) the outcomes of the modelling approach cannot be easily measured or validated using conventional study methods (although c.f. Bond et al. in prep), and; (iii) the modelling approach is based on a coarse measure of productivity, and it may not be appropriate to apply the method at smaller spatial and temporal scales.

The primary objectives of the project were to:

1. Review recent literature (both domestic and international; see Appendix 2-4) with the aim of incorporating up-to-date information on basal production, ecological efficiencies and food web structure, particularly with respect to the influence of additional environmental drivers (e.g., temperature);

2. Modify the energetics models, based on these reviews, to incorporate the influence of these environmental drivers on rates of productivity, and;

3. Apply these revised models to modelled time-series of flow from the Murray RRC projects to evaluate the potential productivity benefits and changes to food-web energy flows associated with proposed flow limit options.

3. Methodology

3.1 MODEL INPUTS

Model inputs are approximately the same as described by Bond et al. (in prep). 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})

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)
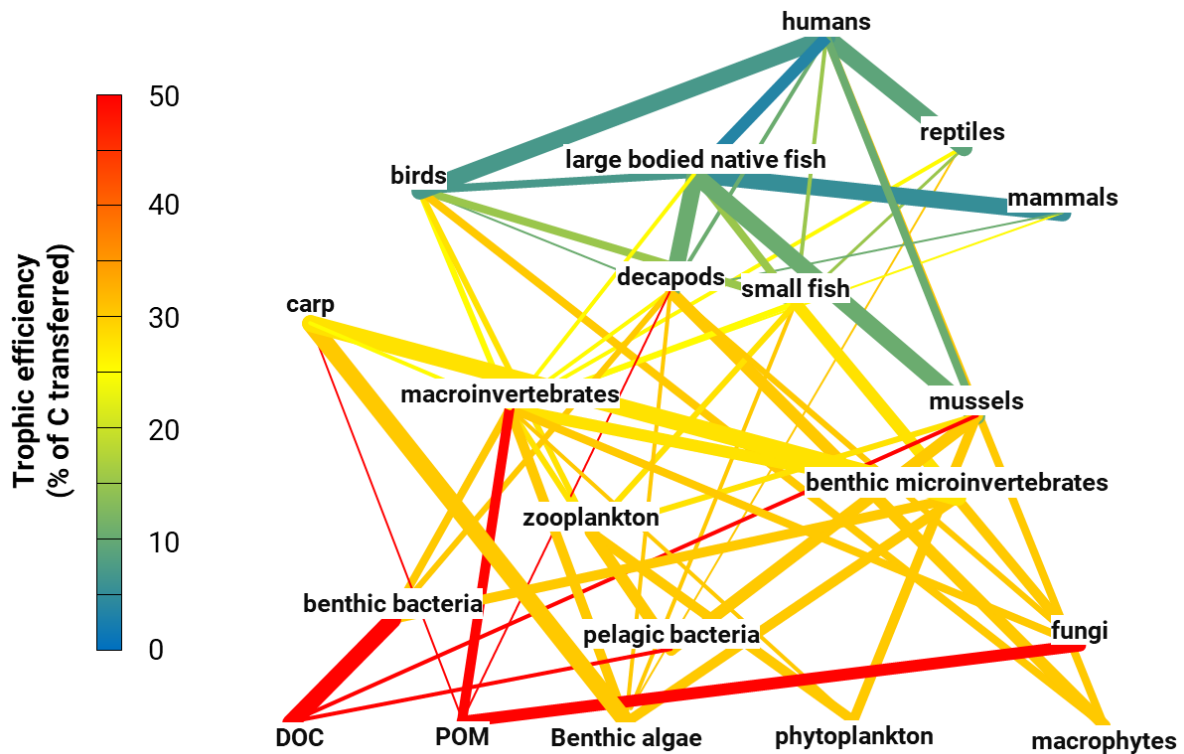

Figure 3.1 Modelled food web including carp (adapted from Bond et al., in prep). Lines between different organism groups and basal resources (a.k.a. nodes) indicate trophic connections (higher nodes consume lower nodes). Colour of lines indicates proportion of C at lower node which contributes to production 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 areas for the first phase of this project (River Murray: Hume to Wakool junction) as defined by DPE are given in Table 3.1. Similar to Bond et al. (in prep), the spatial inundation datasets used therefore include the River Murray Floodplain Inundation Model (RiMFIM) for the River Murray from Hume Dam to the junction of the Murray and Wakool Rivers (Overton et al., 2006), and the RiMFIM extension for the Edward/Kolety-Wakool anabranch system (Sims et al., 2014). Historical discharge timeseries were derived from flow gauge data used to calibrate each specific RiMFIM zones (Overton et al., 2006; Sims et al., 2014). Counter-factual discharge timeseries associated with different potential flow options were produced by DPE through eWater Source modelling (Carr and Podger, 2012).

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)	Southern Connected Basin (NSW/VIC, downstream of Hume to SA border)
Murray: Yarrawonga to Wakool	15,000 d/s Yarrawonga Weir 30,000 40,000 45,000	River Murray floodplain from Yarrawonga Weir to Wakool junction (NSW and VIC sides)	

3.2 MODIFICATIONS TO THE BASE MODEL

In the model of Bond et al. (in prep), productivity rates and food web structure were held constant across all model areas and timesteps. For this project we incorporated two major modifications over this structure with regards to productivity, based on our literature review: first, (i) we modelled the temperature-dependence of productivity by predicting variation in areal rates from seasonal changes in water temperature (see Appendix 1, section A1.1); and (ii) we specified different productivity rates for different, broad habitat types (perennial river channel, permanent floodplain wetlands, intermittent floodplain wetlands, and all other floodplain) across the River Murray and associated floodplains (see Appendix 1, section A1.2). An input of measured water temperature (in °C) corresponding with the discharge timeseries for at least two years is thus also required to model the temperature dependence of productivity rates (see Appendix 1, section A1.1). The food web structure remains that defined by Bond et al. (in prep) through a combination of literature review and expert opinion (see Appendix 1).

Finally, in the original model the duration of an inundation event for a given area, $Edays$ (i.e., "exceedance days", or duration of inundation), is set to 1 once discharge Q exceeds the inundation threshold of the area I. At each following timestep, t, $Edays_t$ is calculated as $Edays_{t-1}+1$ if $I > Q$. In the base model structure, $Edays$ is reduced to 0 at the next timestep when $I < Q$. This structure therefore does not account for water remaining on floodplain areas when discharge drops below the inundation threshold value. However, if these areas retain water and are

reconnected by subsequent, closely occurring flow pulses, they may potentially contribute strongly to energetics directly upon reconnection rather than requiring a lag time for productivity to occur (see e.g. Pettit et al. 2017).

To model the persistence of water on inundated floodplains, we kept the structure of increasing *Edays* values when *I* > *Q*. However, when discharge next drops below the inundation threshold, the decay of *Edays* for a given inundation threshold, *I*, is expressed as:

$$Edays_{I,t} = Edays_{I,t-1} - \frac{I}{Q_t}$$

Where *Edays* for a given *I* at time *t* is dependent on the preceding day's value (*Edays_{I,t-1}*) and is reduced by a factor related to the ratio of *I* to the current daily discharge (Q_t), i.e. the count of "exceedance days" decreases from the previous value rather than being reset immediately to zero, and decays more quickly when discharge drops more rapidly in relation to the inundation threshold.

3.3 MODEL WORKFLOW

The model workflow follows that of Bond et al. (in prep). First, the spatial dataset is split into the four broad habitat classes for which different productivity values are defined (see Appendix 1, section A1.2): perennial waterbodies, intermittent wetlands and anabranches, permanent off-channel wetlands, and all other floodplains. 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 (Fig. 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: Appendix 1, Bond et al., in prep). 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 derived from modelled temperature data timeseries provided by DPE (see Appendix 1). The productivity rate inputs, modified by monthly mean water temperature (see: Appendix 1), are then multiplied by appropriate area*days values (see Appendix 1) to provide estimates of potential C produced by these basal sources over the given time period (Fig. 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 (Fig. 3.2).
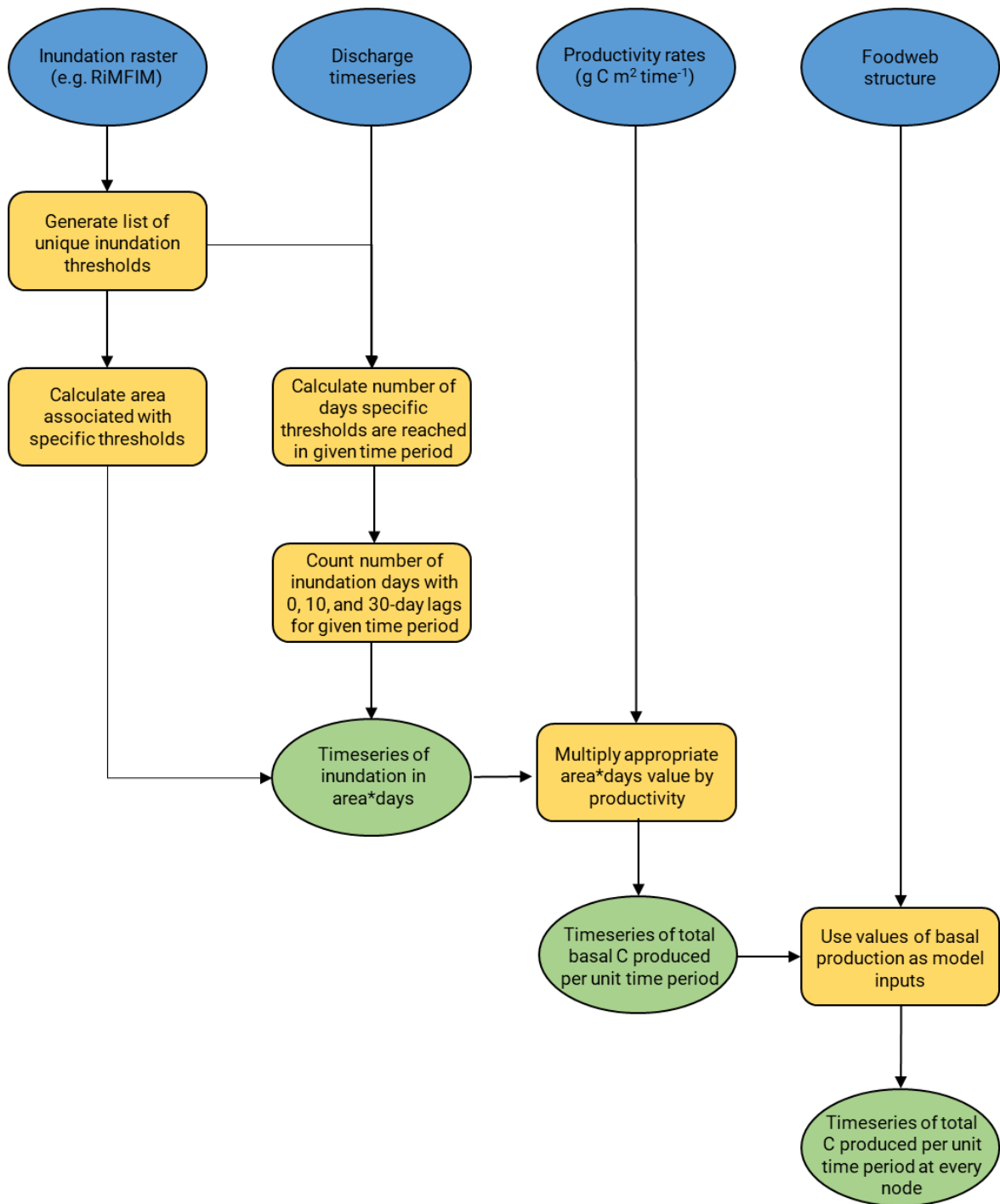
Figure 3.2 Flow chart showing the workflow associated with generating estimates of maximum energy (in g C) available for production at each foodweb node.

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 (Fig. 3.3).
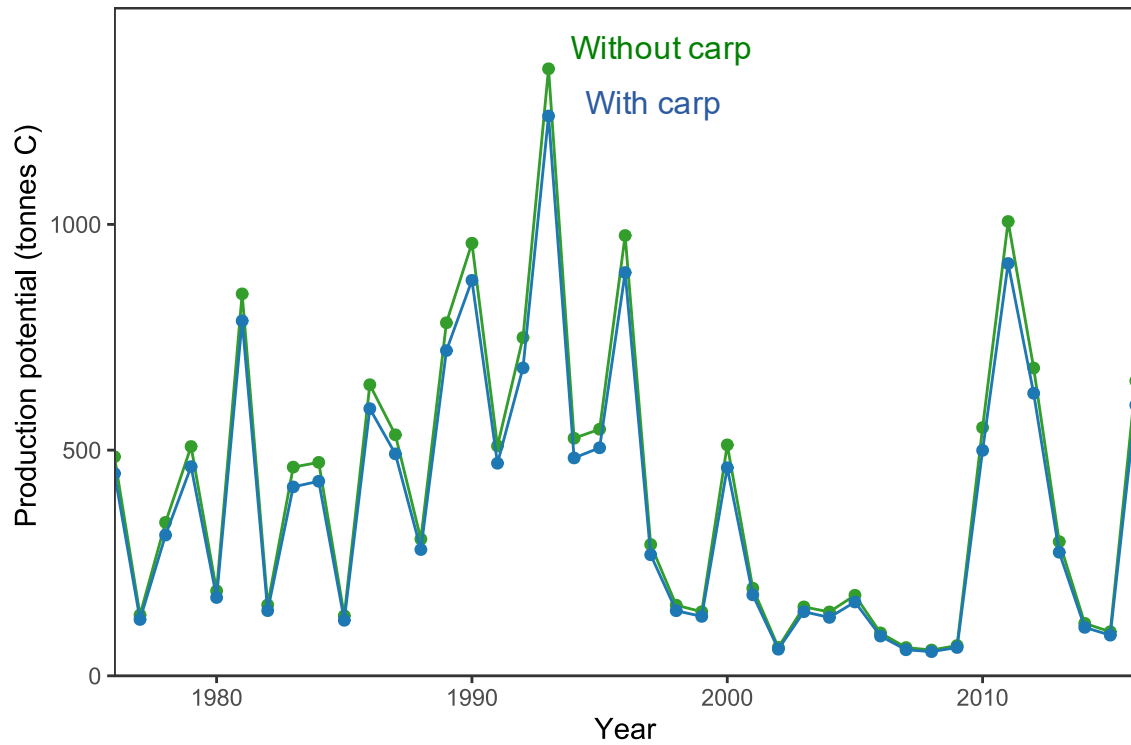
Figure 3.3 Comparison of predicted production potential of large native fish (tonnes C) from a model with carp included (blue) and excluded (green) from the food web structure. Model estimates are derived from spatial datasets on inundation from Koondrook-Perricoota (RiMFIM zone 7) and counterfactual discharge from the Y15D25 flow option (a.k.a. "base" option) (see: Appendix 1).

3.4 COMPARING FLOW OPTIONS

As above, modelled areas for the flow options comparison include all the mid-River Murray from Hume Dam to the Murray-Wakool junction, including the Edward/Kolety-Wakool (EK-W) anabranch system, including both NSW and VIC sides of the river (Fig. 3.4).
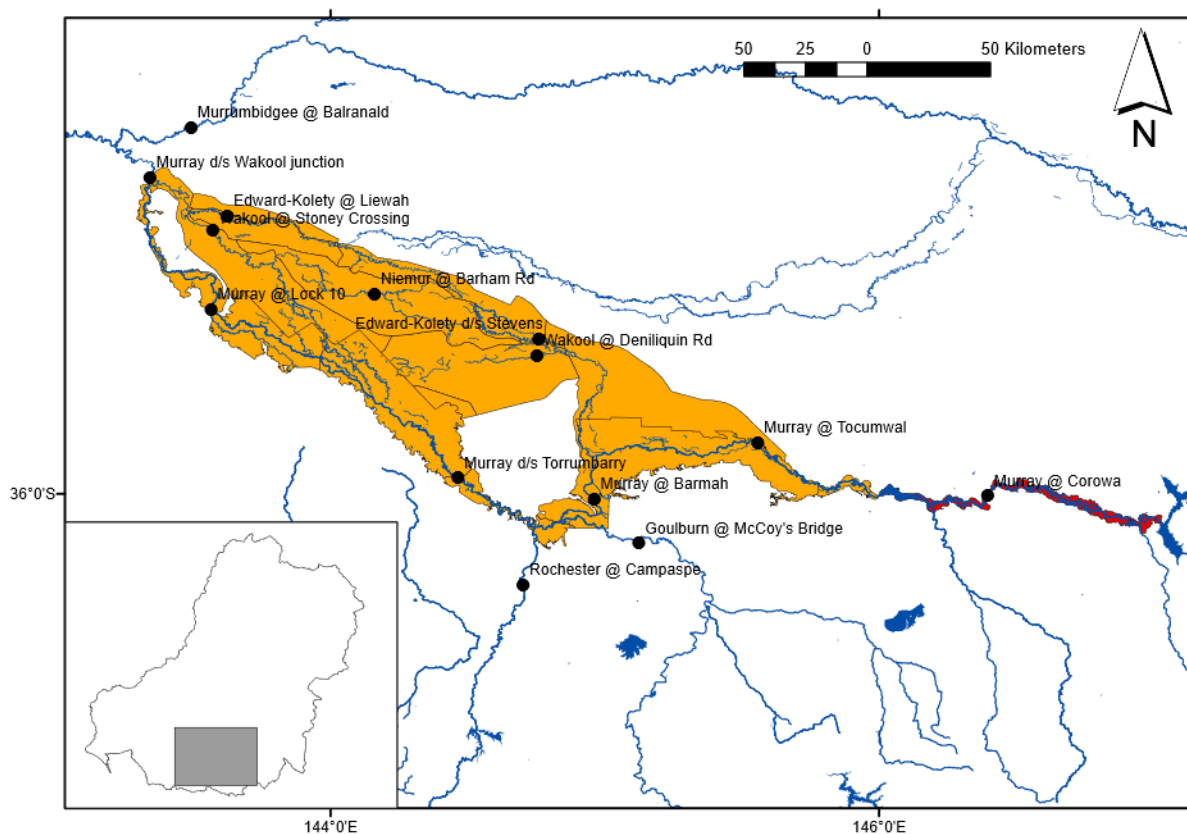
Figure 3.4 Extent of modelled areas, Hume to Yarrawonga (red filled area) and Yarrawonga to Wakool confluence (orange filled area), with RiMFIM zones outlined. Black dots show associated flow gauges. Inset map shows extent (grey area) within the Murray-Darling Basin boundaries.

The flow options to be assessed include the following combinations of constraints limits defined earlier in Table 3.1:

1. 25,000 ML/day at Doctors Point and 15,000 ML/day at Yarrawonga (henceforth: "Y15D25" option), a.k.a. the current "base" case scenario;

2. 25,000 ML/day at both gauges ("Y25D25" option);

3. 30,000 ML/day at both gauges ("Y30D30" option);

4. 40,000 ML/day at both gauges ("Y40D40" option); and

5. 40,000 ML/day at Doctors Point and 45,000 ML/day at Yarrawonga ("Y45D40" option).

In addition, we also considered an additional scenario where all flow regulators are considered open (henceforth: the "Without development" or "WOD" flow option).

For the Murray: Hume to Yarrawonga project area (henceforth: Area 1), the model inputs were therefore:

1. Inundation thresholds from RiMFIM Zone 1 (Hume Dam to Yarrawonga Weir); and,

2. Modelled discharge series for the Corowa gauge representing all the defined flow options; and,

3. Modelled temperature timeseries (section 3.3) associated with the above gauge data.

For the Murray: Yarrawonga to Wakool project area (henceforth: Area 2), model inputs were:

1. Inundation thresholds from RiMFIM River Murray Zones 2-9 (Yarrawonga Weir to Murray/Wakool junction) and Edward/Kolety-Wakool system (EK-W) Zones 1-7 (d/s Barmah to Murray/Wakool junction); and,

2. Modelled discharge timeseries representing all the defined flow options for the following RiMFIM-associated flow gauges (following Overton et al., 2006 and Sims et al., 2014):

   o Zone 2 (Yarrawonga Weir to Tocumwal): Tocumwal

   o Zone 3 (Tocumwal to downstream of Barmah Forest): Tocumwal

   o Zone 4 (d/s of Barmah to Goulburn junction): Barmah Choke

   o Zone 5 (Goulburn junction to Campaspe junction): Barmah plus Goulburn at McCoy's Bridge

   o Zone 6 (Campaspe junction to Torrumbarry Weir): Barmah plus Goulburn at McCoy's Bridge (with 1-day travel time) plus Rochester at Campaspe

   o Zone 7 (Torrumbarry to downstream of Gunbower): d/s of Torrumbarry Weir

   o Zone 8 (d/s of Gunbower to Swan Hill): d/s of Torrumbarry Weir

   o Zone 9 (Swan Hill to Wakool junction): Swan Hill

   o EK-W Zone 1 (Stevens Weir to Billabong Ck.): Edward/Kolety d/s Stevens Weir

   o EK-W Zone 2 (d/s Barmah to Stevens Weir, plus Tuppal and Bullatere Cks.): Tocumwal

   o EK-W Zones 3 and 3a (Billabong Ck. to Wakool junction): Edward/Kolety at Liewah

   o EK-W Zones 4 and 7 (Wakool origin to Merribit Ck.): Wakool at Deniliquin

   o EK-W Zone 5 (Merribit Ck. to Wakool junction): Wakool at Stoney's Crossing

   o EK-W Zone 6 (Niemur R.): Niemur at Barham Rd.

   and,

3. Modelled temperature timeseries (section 3.3) associated with the above gauges.

For all project areas, the remaining inputs were held in common:

1. Basal resource productivity rates (in g C m^{-2} yr^{-1}) modified for temperature and habitat-dependence (see Appendix 1); and,

2. Common food web structure (see: Figure 3.1).

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: Appendix 1). 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 WOD flow option (the 25th percentile), i.e., the proportion of years in which estimates for each flow option might more closely resemble unregulated conditions. 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 water 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 Y25D25 option – Y15D25 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 Y25D25 and Y15D25 options as a percentage of the Y15D25 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 Y25D25 option produced greater estimates than the Y15D25, 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 water years in which the maximum recorded daily flow at Yarrawonga Weir or Doctors Point (relative to the area being evaluated) was affected by constraints limits (e.g., 15,000 ML/day at Yarrawonga or 25,000 ML/day at Doctors Point for the Y15D25 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 WOD 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 AREA 1: RIVER MURRAY, HUME TO YARRAWONGA

Comparisons across all modelled years
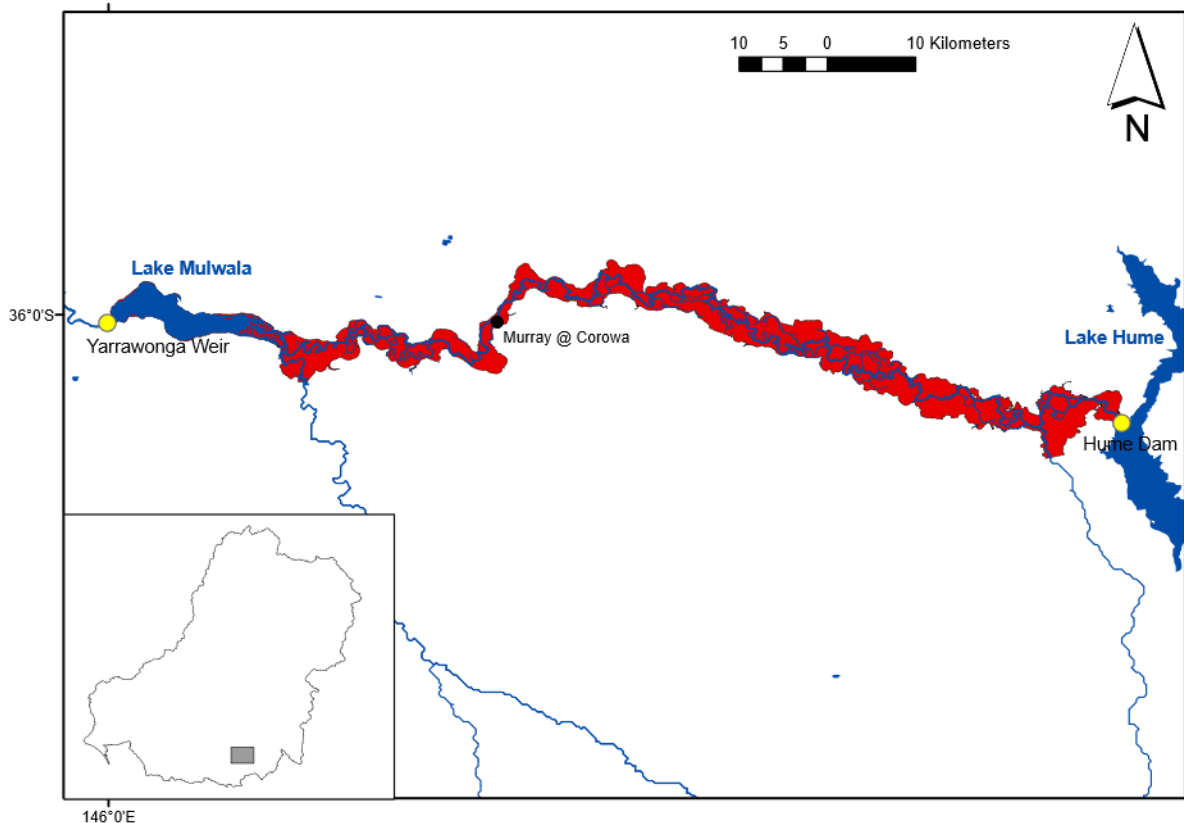
Figure 4.1 Extent of modelled area, Hume to Yarrawonga (red filled area), with RiMFIM zone outlined. Black dot shows associated flow gauge. Marked yellow circles indicate beginning and end of modelled area. Inset map shows extent (grey area) within the Murray-Darling Basin boundaries.

Discharge timeseries from the flow options produced relatively similar patterns in cumulative yearly discharge over time for the Hume-Yarrawonga area (Fig. 4.2). Estimates of inundation extent over each year were also similar between the options, although the frequency of moderately-sized inundation years was noticeably higher in the without development scenario. However, the inclusion of Lake Mulwala (i.e. a large, permanently maintained water body) in the inundation estimates also resulted in relatively high base levels of inundation across all years for any of the constraints options (Fig. 4.2). Production potential for all flow options were therefore also similar across years, with large flood events consistently increasing yearly estimates to approximately twice that of the production potential from the permanent channel network and Lake Mulwala. There was, however, a greater frequency of moderately increased production potential years in the without development scenario (Fig. 4.2).

Figure 4.2 For the Hume Dam to Yarrawonga Weir area, from 1896-2018, comparisons of annual cumulative (a-f) discharge at Do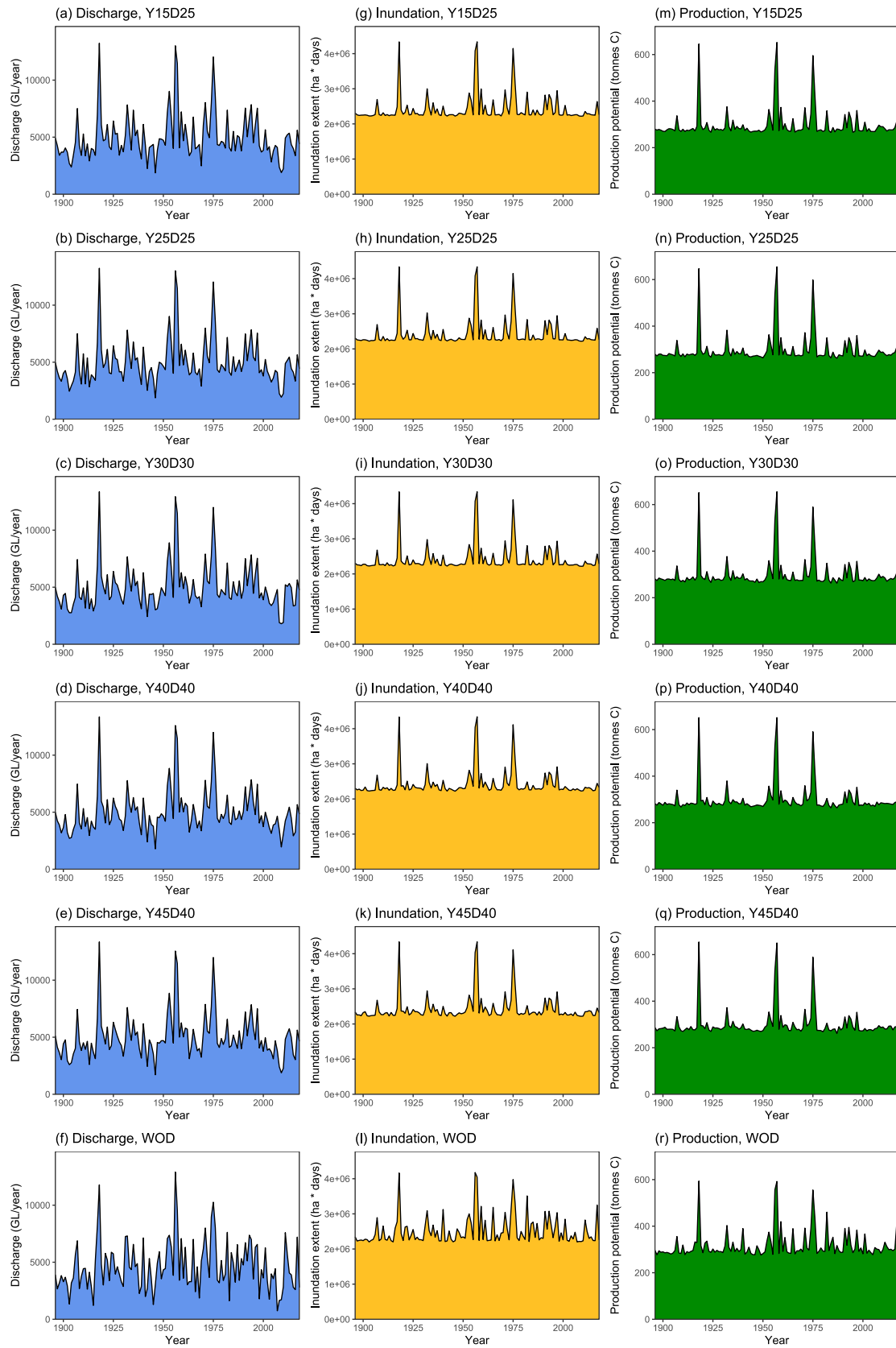ctors Point with (g-l) inundation extent (in ha * days) and (m-r) production potential of large native fish (tonnes C) for each of the modelled constraints scenario timeseries, as well as the without development timeseries ("WOD").

For the Hume to Yarrawonga area, the proportion of years in which the five modelled flow option scenarios produced estimates of large native fish production potential above the 25th percentile of without development estimates was around 28-39% (Figure 4.3). There was thus a small increase (up to 11.4%) in the proportion of years that might provide greater outcomes for production potential under the higher constraints relative to the base option (Y15D25) (Figure 4.3).
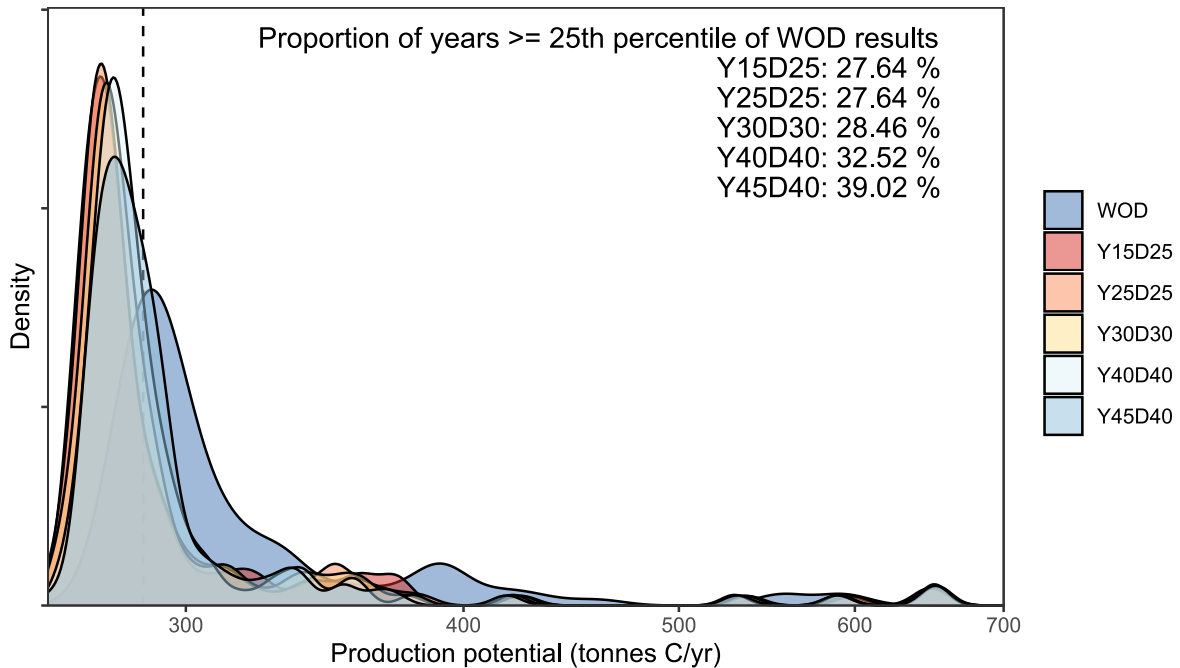
Figure 4.3 Density distributions of model results for the production potential of large native fish from the six different modelled timeseries, across all years (1986-2018), for the Hume Dam to Yarrawonga Weir area. Dashed line indicates 25th percentile of without development ("WOD") results (i.e., value above which 75% of yearly estimates from the WOD 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 WOD results. Note the use of nonlinear scale on the x axis (pseudo-log$_{10}$ transformation) to aid in discrimination between density distributions.

There was therefore only a small increase (1.5%) in the 25th percentile of production potential estimates from the Y15D25 base option (273 tonnes C) to the Y45D40 option (277 tonnes C), with the 25th percentile of WOD scenario estimates higher than any of the flow options (Figure 4.4). The median yearly production potential across flow options also slightly increased (2.1%) from the Y15D25 base option (278 tonnes C) to the Y45D40 option (284 tonnes C), with the WOD scenario having slightly higher estimated median production potential again (Figure 4.4).

Figure 4.4 Median and 25th percentile (i.e., "Q1") of annual production potential estimates, across all years, for the different flow options associated with the Hume Dam to Yarrawonga Weir area. Numbers above the Y25D25, Y30D30, Y40D40, T45D40, and WOD statistics indicate the difference between that value and the corresponding statistic for the Y15D25 option (a.k.a. "base" option).

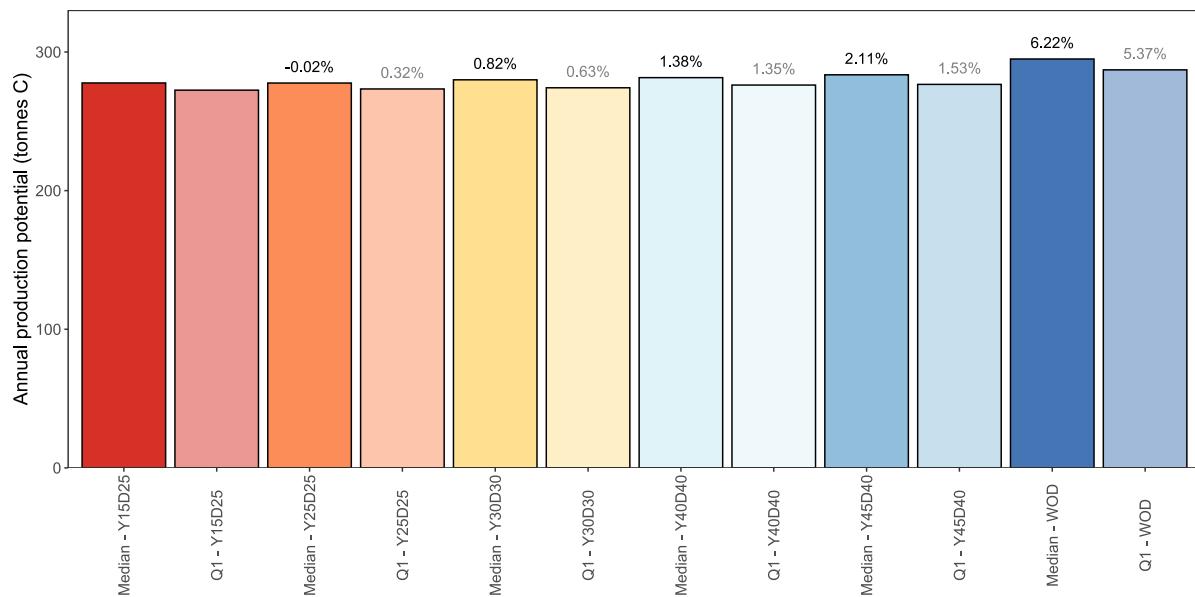Relative to the Y15D25 "base" option, the alternative flow option models produced greater estimates of production potential more often (Figure 4.5). The model with a higher constraints limit produced greater estimates ranging from 54% (Y25D25) to 62% (Y40D40) of all years in comparison with the Y15D25 model (Table 4.1). However, there were also a number of years in which the Y15D25 model produced greater production potential estimates than any of the other flow options (approx. 38 – 46% of total years), and often to a level that was noticeably higher than the maximum differences generated in favour of the alternative option (Figure 4.5). For example, while the mean difference between the Y15D25 and Y25D25 model estimates was approx. 2.3 tonnes C/yr when the Y25D25 model produced higher estimates, the mean difference was 3.4 tonnes C/yr when the Y15D25 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. 0.8 – 2.7% mean difference in production potential for the flow option with the higher constraint limit, and 1.1 – 2.4% mean difference in production potential for the base flow option (Table 4.1). Yet across all comparisons, the flow option with the higher constraint produced greater estimates in consecutive years more often (55 – 61 total years) and for longer on average (mean 3.0 – 3.6 years of consecutive periods) than the base option (25 – 55 years; mean 2.2 – 3.5 year periods). Consequently, while there were a number of years in which the Y15D25 base option produced notably higher estimates than the flow options with higher constraints, the alternative flow options maintained periods of higher production potential for longer.

Figure 4.5 Difference between model estimates for production potential (tonnes C), for the Hume Dam to Yarrawonga Weir area, from the Y15D25 (a.k.a. "base") flow option and (a) the Y25D25 (i.e., Y25D25 - Y15D25 estimate), (b) Y30D30, (c) Y40D40, and (d) Y45D40 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 Hume to Yarrawonga area (Area 1), summary statistics for differences in pairwise comparisons of model estimates between the Y15D25 (a.k.a. base option) and other flow options. Shown for the subset of results in which one option produced greater differences (e.g., when the Y25D25 model produced higher estimates than the Y15D25) are the mean differences in 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 produced higher estimates, the total number of years in which the given flow option produced higher estimates across consecutive years (i.e., >= 2 years), and the mean length of those consecutive periods. Results for comparison with the WOD model are also given.

COMPARISON	CONSTRAINT WITH HIGHER PRODUCTION	DIFFERENCE IN PRODUCTION (TONNES C; %)	PROPORTION OF YEARS	TOTAL LENGTH OF CONSECUTIVE PERIODS (YR)	MEAN LENGTH OF CONSECUTIVE PERIODS (YR)
Y15D25 vs Y25D25	Y25D25	2.27 (0.78%)	53.7%	55	3.06
	Y15D25	-3.43 (-1.17%)	46.3%	38	3.45
Y15D25 vs Y30D30	Y30D30	3.54 (1.26%)	59.3%	56	2.95
	Y15D25	-4.73 (-1.56%)	40.7%	25	2.5
Y15D25 vs Y40D40	Y40D40	5.78 (2.08%)	61.8%	61	3.59
	Y15D25	-7.54 (-2.44%)	38.2%	29	2.23
Y15D25 vs Y45D40	Y45D40	7.54 (2.73%)	59.3%	58	3.05
	Y15D25	-7.08 (-2.28%)	40.7%	28	2.55
Y15D25 vs WOD	WOD	23.11 (7.91%)	95.1%	117	16.71
	Y15D25	-27.64 (-5.12%)	4.9%	0	0

In all other comparisons between flow options for the Hume to Yarrawonga area, the basic pattern was much the same as for the comparisons with the base Y15D25 option. Across all years, the flow option with the higher constraints produced greater estimates of production potential more often (56 − 65% of years) than the option with the lower constraint (35 − 44% of years) (Table 4.2), but the flow option with the lower constraint had higher peak differences in production potential (Table 4.2) (Figure 4.6). However, these peaks in favour of the flow option with lower constraints likely occurred during years in which production potential estimates were high overall: the mean differences in production potential were approximately equivalent between flow options (2.6 − 6.0 tonnes C/yr compared with 3.2 − 6.2 tonnes C/yr), and these differences were also only 0.9 − 1.9% of the lower absolute production potential estimate being compared (as opposed to 1.1 − 2.5% for the alternative flow option) in those given years (Table 4.2). Again, however, the flow option with the higher constraints limit also produced greater estimates in consecutive years more often (42 − 72 total years) and for longer on average (mean 2.6 − 3.6 years of consecutive periods) than the alternative flow option (26 − 42 years; mean 2.3 − 3.0 year periods).
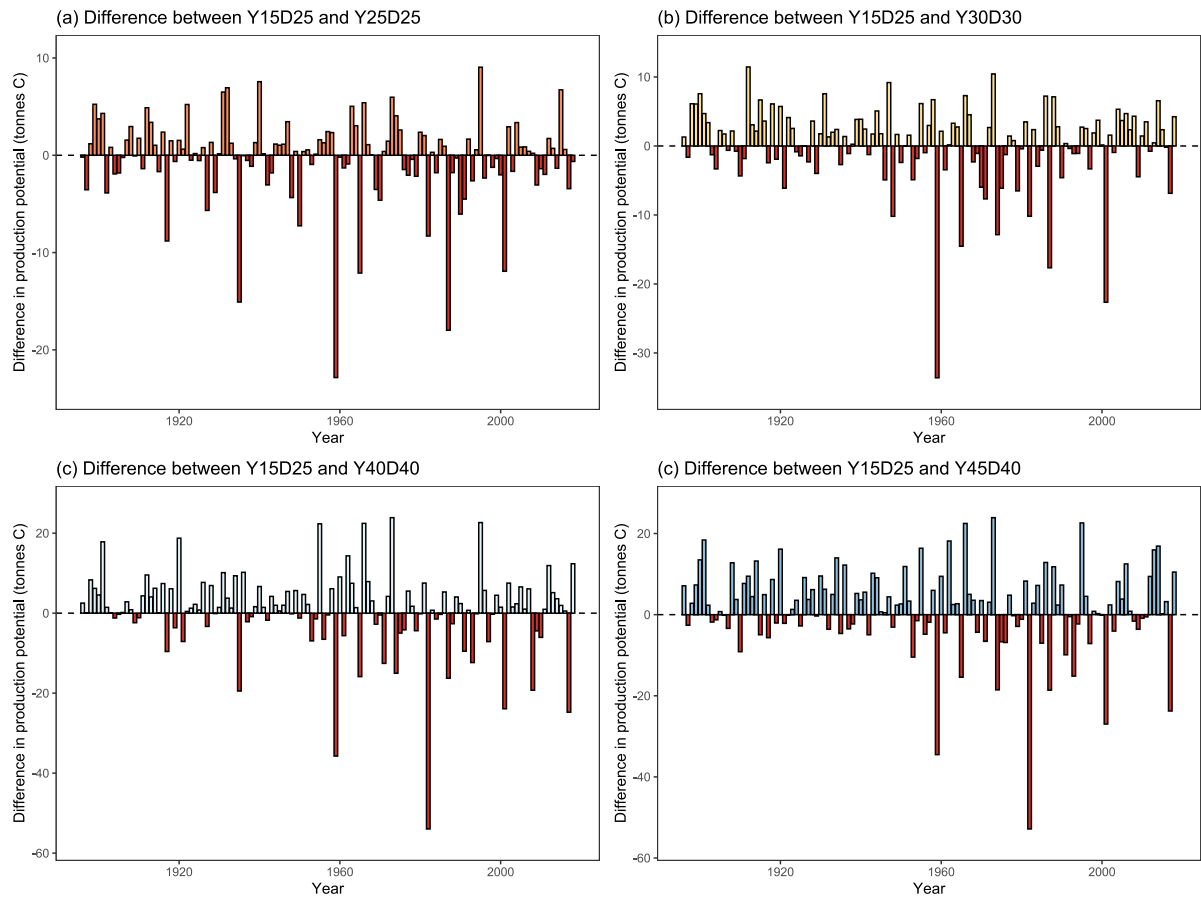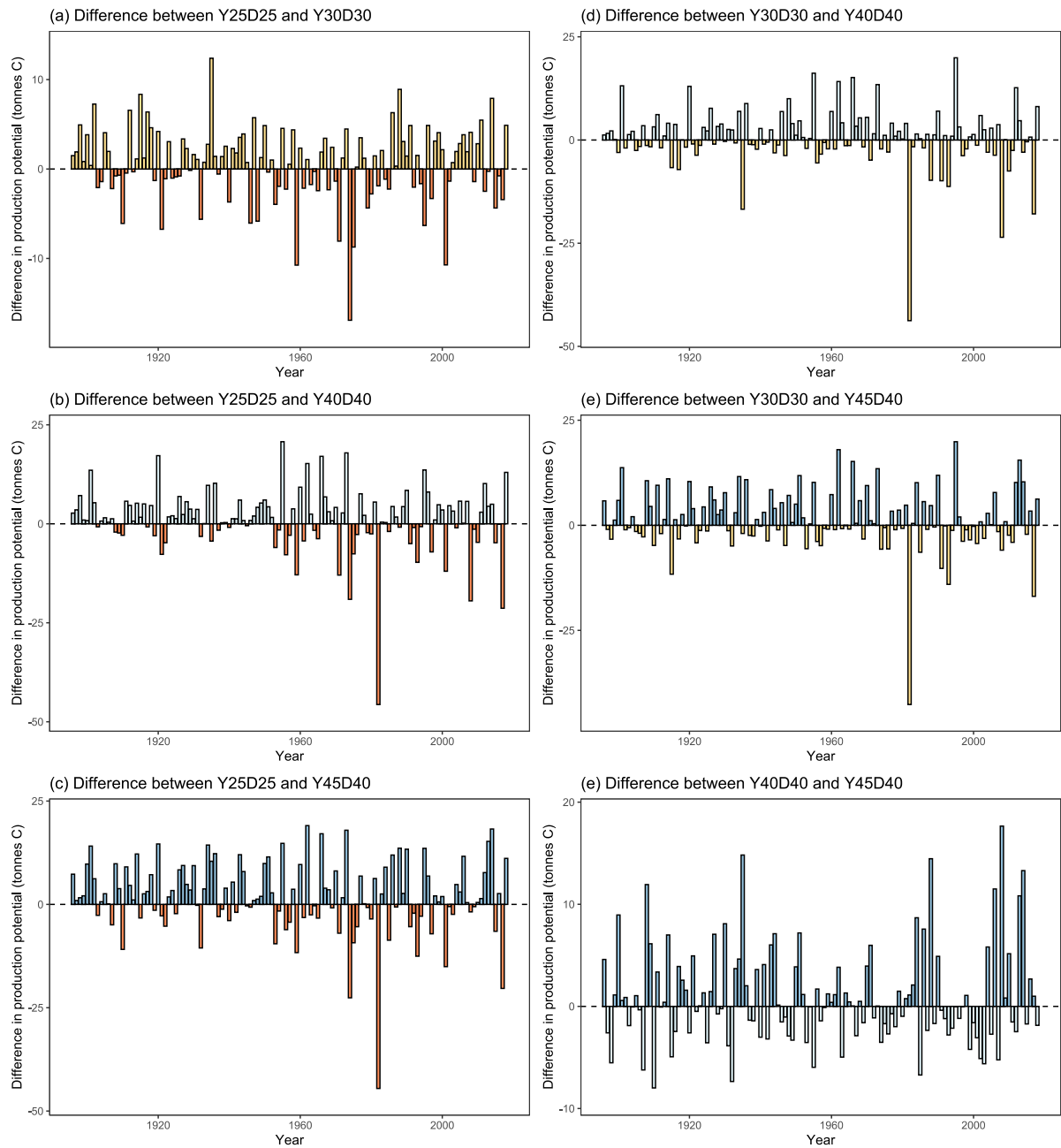
Figure 4.6 Difference between model estimates for production potential (tonnes C) for the Hume Dam to Yarrawonga Weir area, across all modelled years, from flow options not including the "base" model: comparisons are between the Y25D25 and (a) Y30D30 (i.e., Y30D30 - Y25D5 estimates), (b) Y40D40, and (c) Y45D40 model estimates; between the Y30D30 and (d) Y40D40 and (e) Y45D40 estimates; and (f) Y40D40 and Y45D45 estimates. Positive values indicate that estimates are greater for the flow option with the higher constraints limit, and *vice versa*.

Table 4.2 For the Hume to Yarrawonga area (Area 1), summary statistics for differences in pairwise comparisons of model estimates between flow options not including the Y15D25 option (a.k.a. base option). Shown for the subset of results in which one option produced greater differences (e.g., when the Y30D30 model produced higher estimates than the Y25D25) are the mean differences in 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 produced higher estimates, the total number of years in which the given flow option produced higher estimates across consecutive years (i.e., >= 2 years), and the mean length of those consecutive periods. Results for comparison with the WOD model are also given.

COMPARISON	CONSTRAINT WITH HIGHER PRODUCTION	DIFFERENCE IN PRODUCTION (TONNES C; %)	PROPORTION OF YEARS	TOTAL LENGTH OF CONSECUTIVE PERIODS (YR)	MEAN LENGTH OF CONSECUTIVE PERIODS (YR)
Y25D25 vs Y30D30	Y30D30	3.18 (1.13%)	58.5%	57	3.35
	Y25D25	-3.17 (-1.02%)	41.5%	34	2.43
Y25D25 vs Y40D40	Y40D40	4.87 (1.76%)	65.0%	72	3.6
	Y25D25	-6.04 (-1.91%)	35.0%	26	2.6
Y25D25 vs Y45D40	Y45D40	6.84 (2.47%)	61.8%	64	3.37
	Y25D25	-5.92 (-1.85%)	38.2%	29	2.42
Y25D25 vs WOD	WOD	23.33 (8%)	95.9%	118	19.67
	Y25D25	-33.8 (-6.13%)	4.1%	0	0
Y30D30 vs Y40D40	Y40D40	4.47 (1.61%)	56.1%	49	2.58
	Y30D30	-4.55 (-1.51%)	43.9%	28	2.33
Y30D30 vs Y45D40	Y45D40	6.23 (2.25%)	52.0%	42	2.8
	Y30D30	-3.8 (-1.24%)	48.0%	36	2.77
Y30D30 vs WOD	WOD	23.13 (7.9%)	94.3%	115	19.17
	Y30D30	-23.72 (-4.31%)	5.7%	2	2
Y40D40 vs Y45D40	Y45D40	4.31 (1.52%)	50.4%	45	2.81
	Y40D40	-2.55 (-0.87%)	49.6%	42	3
Y40D40 vs WOD	WOD	23.99 (8.19%)	90.2%	108	12
	Y40D40	-17.46 (-3.83%)	9.8%	2	2
Y45D40 vs WOD	WOD	23.41 (7.96%)	88.6%	106	9.64
	Y45D40	-14.93 (-3.28%)	11.4%	2	2

Comparisons across years with constraints operating

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 five modelled flow scenarios produced estimates of large native fish production potential above the 25th percentile of without development estimates was more variable than that across all years, at approximately 7-30% (Figure 4.7). In comparison with the modelled results across all years, there were larger differences in how often the base model (Y15D25) reached the 25th percentile of WOD estimates (6.8%) relative to the flow options with higher constraints limits, with progressively higher proportions of years reaching this nominal target for the Y25D25 (8.0% of years) up to the Y45D40 option (29.6% of years; Figure 4.6).
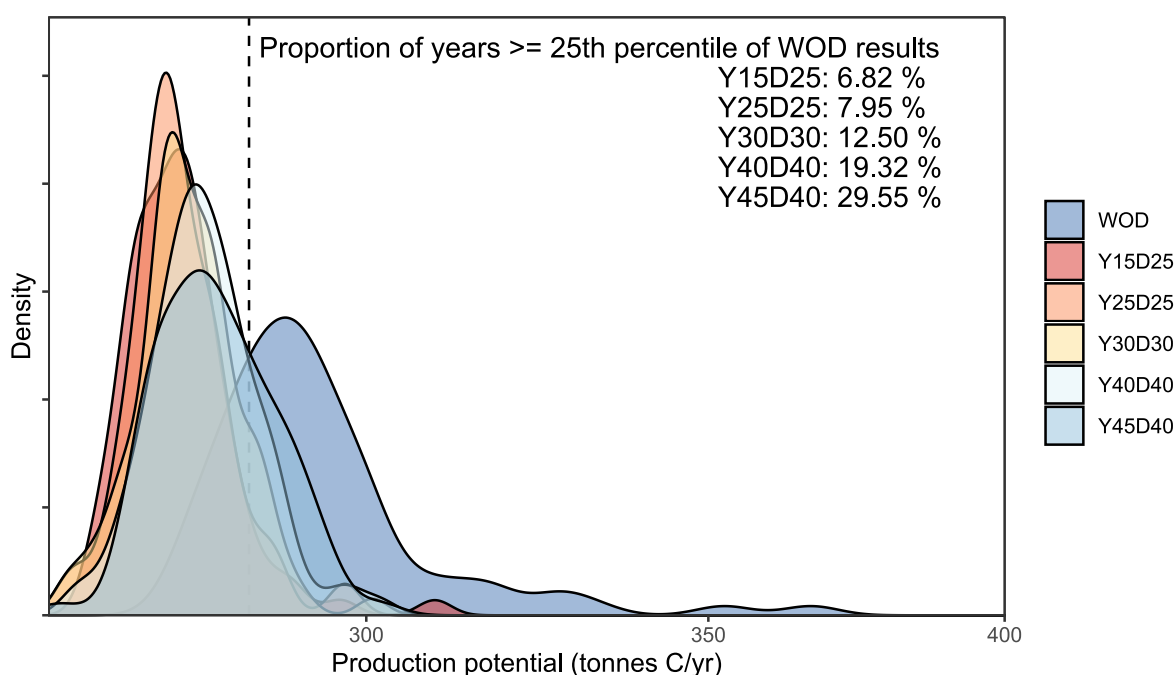
Figure 4.7 Density distributions of model results for the production potential of large native fish from the six different modelled timeseries, across years in which constraints were operating (i.e., any year where a flow option had a modelled maximum daily discharge below its designated constraints limit), for the Hume Dam to Yarrawonga Weir area. Dashed line indicates 25th percentile of without development ("WOD") results (i.e., value above which 75% of yearly estimates from the WOD 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 WOD results. Note the use of nonlinear scale on the x axis (pseudo-log$_{10}$ transformation) to aid in discrimination between density distributions.

However, the percentage increase in the 25th percentile of production potential estimates from the Y15D25 base option (271 tonnes C) to the Y45D40 option (276 tonnes C) was still low (2.0%) when compared in years with constraints operating (Figure 4.8). The median yearly production potential across flow options also only showed a slight increase (1.7%) from the Y15D25 base option (275 tonnes C) to the Y45D40 option (280 tonnes C). In comparison, the WOD scenario had higher estimated 25th percentile (284 tonnes C) and median values (289 tonnes C) for production potential estimates in years with constraints operating (Figure 4.8).

Figure 4.8 Median and 25th percentile (i.e., "Q1") of annual production potential estimates, across years in which constraints were operating (i.e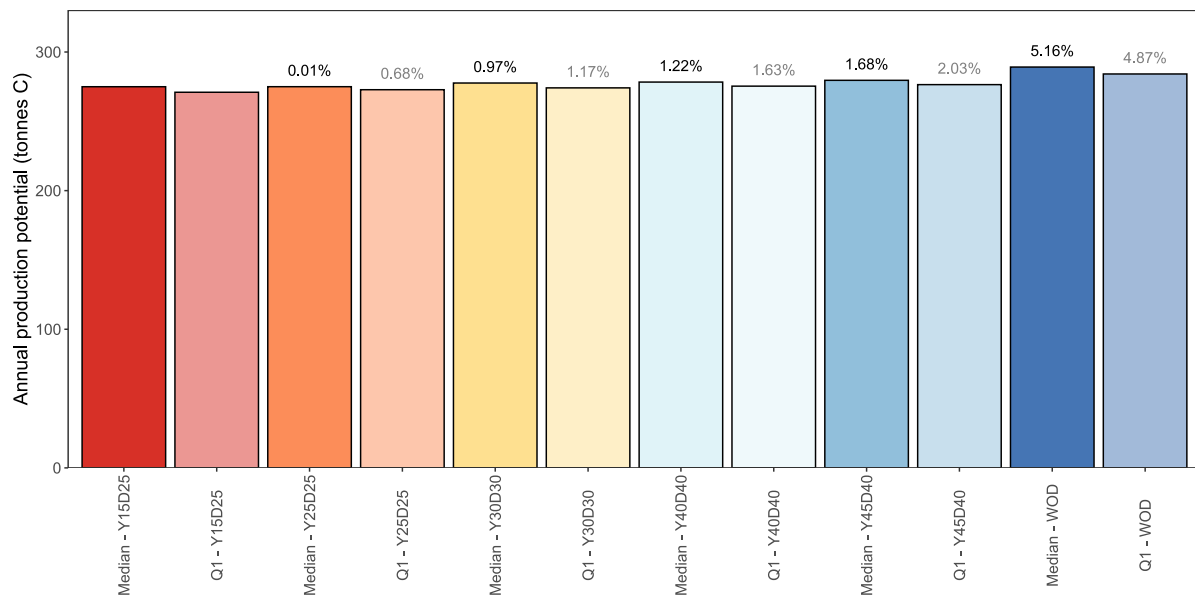., any year where a flow option had a modelled maximum daily discharge below its designated constraints limit), for the different flow options associated with the Hume Dam to Yarrawonga Weir area. Numbers above the Y25D25, Y30D30, Y40D40, T45D40, and WOD statistics indicate the difference between that value and the corresponding statistic for the Y15D25 option (a.k.a. "base" option)

When limited only to years in which constraints might operate, the alternative flow option models again produced greater estimates of production potential more often than the Y15D25 option (Figure 4.9). The flow option with a higher constraints limit produced greater estimates ranging from 57% (Y25D25) to 74% (Y45D40) of regulated years in comparison with the Y15D25 model (Table 4.3). In contrast, the Y15D25 model sometimes only produced notably greater production potential estimates than the Y25D25 flow option (Figure 4.9). Relative to the lowest total estimate produced by either flow option in any given year, these differences correspond to an approx. 0.8 − 2.9% mean difference in production potential for the flow option with the higher constraint limit, and 0.8 − 1.7% mean difference in production potential for the base flow option (Table 4.3). The flow option with the higher constraint also produced greater estimates in consecutive years more often (23 − 45 total years) and for longer on average (mean 3.1 − 3.8 years of consecutive periods) than the base option (10 − 12 years; mean 2.2 − 2.5 year periods). In contrast to the comparison across all years, therefore, the flow options with higher constraints limits not only maintained periods of higher production potential for longer than the Y15D25 base option but also (c.f. the Y25D25 option) largely produced consistently greater estimates of production potential.
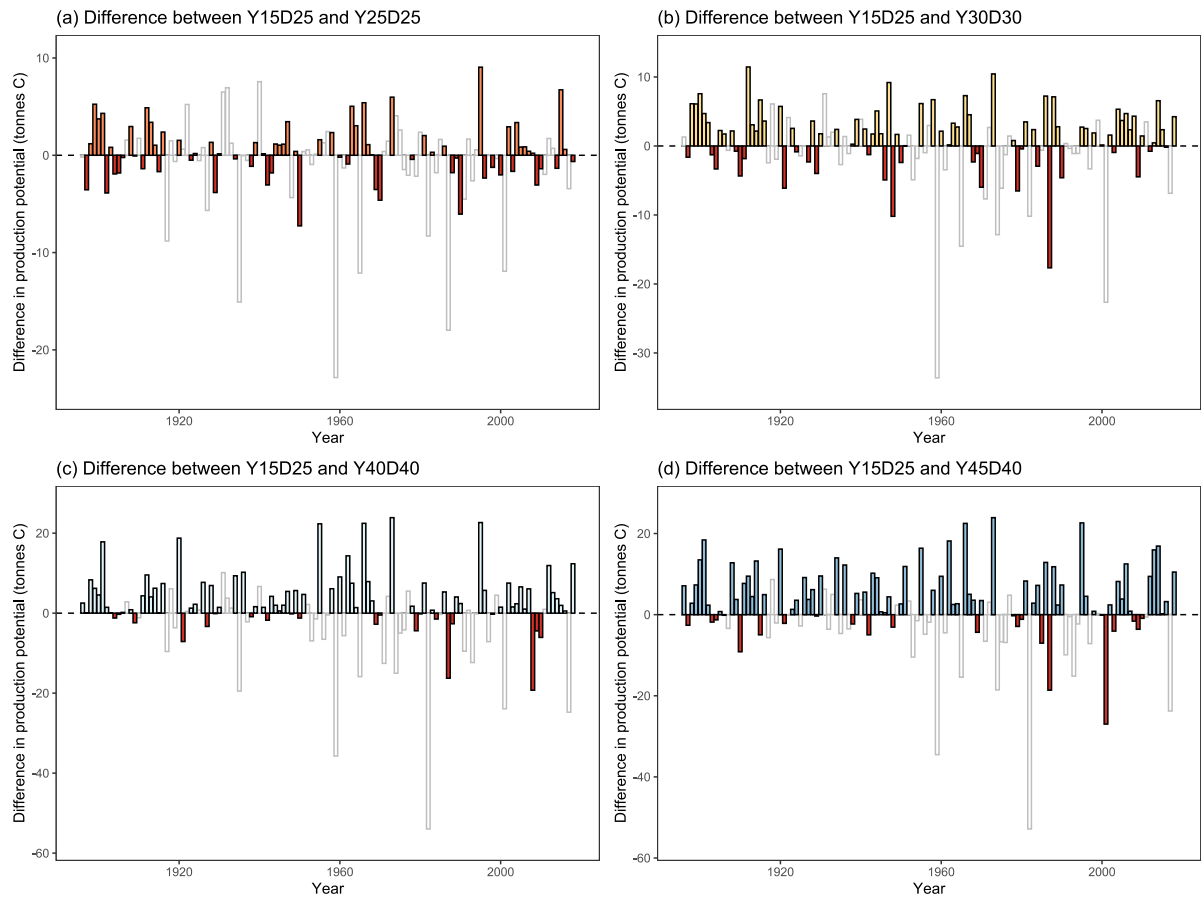
Figure 4.9 Difference between model estimates for production potential (tonnes C) from the Y15D25 (a.k.a. "base") flow option and (a) the Y25D25 (i.e., Y25D25 - Y15D25 estimate), (b) Y30D30, (c) Y40D40, and (d) Y45D40 model estimates, across years in which constraints were operating (i.e., any year where a flow option scenario had a modelled maximum daily discharge below the nominal constraints limit), for the Hume Dam to Yarrawonga Weir area. Positive values indicate that estimates are greater for the flow option model with the higher constraints limit, and *vice versa*. Greyed-out columns indicate results from years where either constraints limit was exceeded during that year.

Table 4.3 For the Hume to Yarrawonga area (Area 1), summary statistics for differences in pairwise comparisons of model estimates between the Y15D25 (a.k.a. base option) and other flow options, 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 Y25D25 model produced higher estimates than the Y15D25) are the mean differences in 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 produced higher estimates, the total number of years in which the given flow option produced higher estimates across consecutive years (i.e., >= 2 years), and the mean length of those consecutive periods.

COMPARISON	CONSTRAINT WITH HIGHER PRODUCTION	DIFFERENCE IN PRODUCTION (TONNES C; %)	PROPORTION OF YEARS	TOTAL LENGTH OF CONSECUTIVE PERIODS (YR)	MEAN LENGTH OF CONSECUTIVE PERIODS (YR)
Y15D25 vs Y25D25	Y25D25	2.30 (0.85%)	56.9%	23	3.29
	Y15D25	-2.06 (-0.75%)	43.1%	12	2.4
Y15D25 vs Y30D30	Y30D30	3.77 (1.37%)	68.7%	34	3.09
	Y15D25	-3.58 (-1.32%)	31.3%	10	2.5
Y15D25 vs Y40D40	Y40D40	6.27 (2.3%)	72.9%	38	3.8
	Y15D25	-3.34 (-1.22%)	27.1%	11	2.2
Y15D25 vs Y45D40	Y45D40	8.01 (2.93%)	73.6%	45	3.21
	Y15D25	-4.54 (-1.65%)	26.4%	10	2.5
Y15D25 vs WOD	WOD	15.48 (5.65%)	96.9%	77	5.13
	Y15D25	-3.39 (-1.19%)	3.1%	0	0

In all other comparisons between flow options for the Hume to Yarrawonga area, the flow option with the higher constraints limit produced greater estimates of production potential notably more often (57 – 82% of years) than the alternative flow option (18 – 44% of years; Table 4.10; Figure 4.8). The flow option with the higher constraint limit also produced greater estimates in consecutive years more often (24 – 58 total years) and for longer on average (mean 2.4 – 3.5 years of consecutive periods) than the alternative option (9 – 18 years; mean 2.0 – 2.6 year periods; Table 4.4).
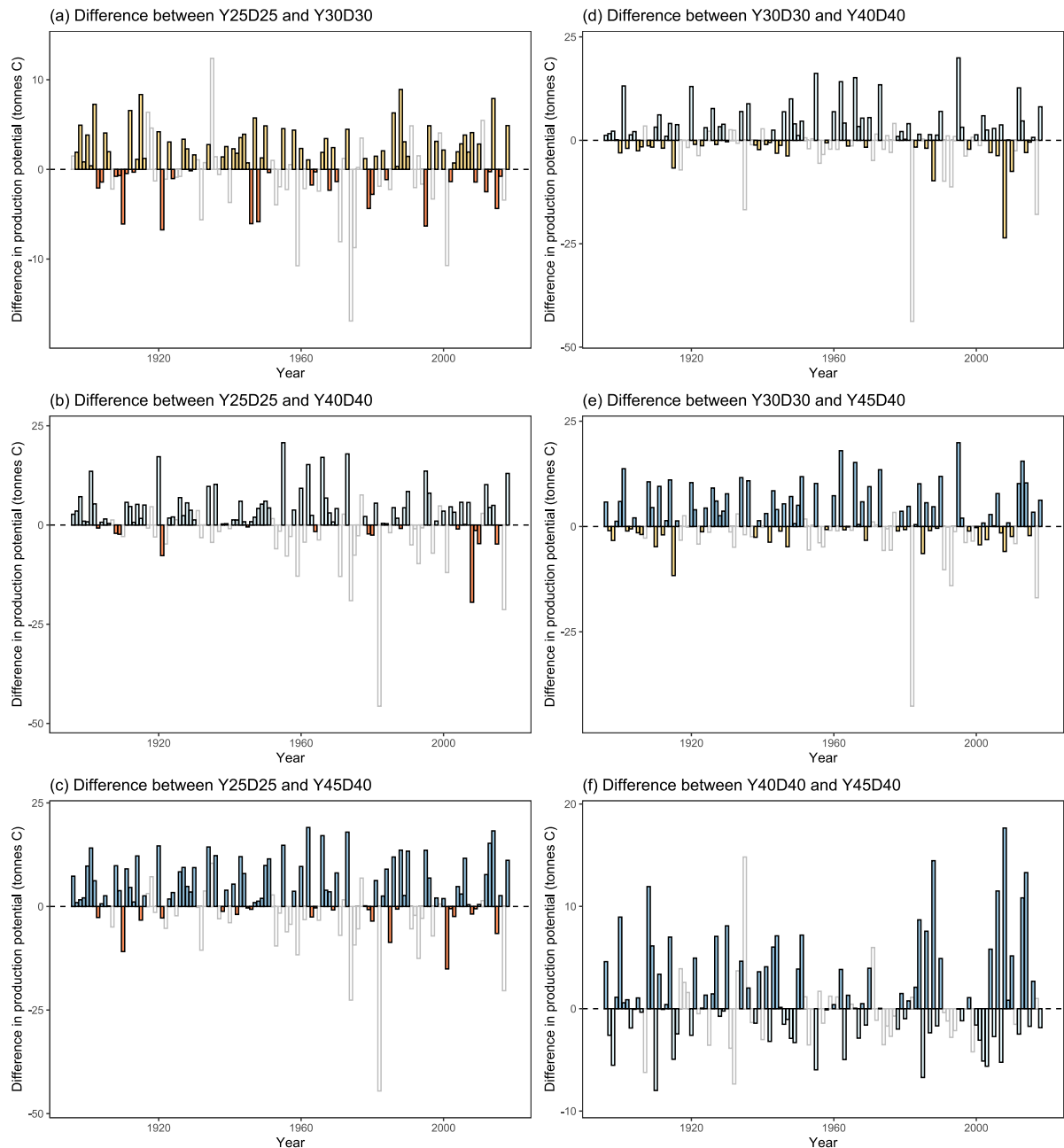
Figure 4.10 Difference between model estimates for production potential (tonnes C) for the Hume Dam to Yarrawonga Weir area, across years in which constraints were operating (i.e., any year where a flow option scenario had a modelled maximum daily discharge below the nominal constraints limit), from flow options not including the "base" model: comparisons are between the Y25D25 and (a) Y30D30 (i.e., Y30D30 - Y25D5 estimates), (b) Y40D40, and (c) Y45D40 model estimates; between the Y30D30 and (d) Y40D40 and (e) Y45D40 estimates; and (f) Y40D40 and Y45D45 estimates. Positive values indicate that estimates are greater for the flow option with the higher constraints limit, and *vice versa*. Greyed-out columns indicate results from years where either constraints limit was exceeded during that year.

Table 4.4 For the Hume to Yarrawonga area (Area 1), summary statistics for differences in pairwise comparisons of model estimates between flow options not including the Y15D25 option (a.k.a. base option), 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 Y30D30 model produced higher estimates than the Y25D25) are the mean differences in 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 produced higher estimates, the total number of years in which the given flow option produced higher estimates across consecutive years (i.e., >= 2 years), and the mean length of those consecutive periods. Comparisons with the WOD model are shown for reference.

COMPARISON	CONSTRAINT WITH HIGHER PRODUCTION	DIFFERENCE IN PRODUCTION (TONNES C; %)	PROPORTION OF YEARS	TOTAL LENGTH OF CONSECUTIVE PERIODS (YR)	MEAN LENGTH OF CONSECUTIVE PERIODS (YR)
Y25D25 vs Y30D30	Y30D30	3.19 (1.16%)	67.5%	35	3.5
	Y25D25	-2.33 (-0.86%)	32.5%	14	2.33
Y25D25 vs Y40D40	Y40D40	5.13 (1.87%)	82.4%	58	3.41
	Y25D25	-3.47 (-1.25%)	17.6%	9	2.25
Y25D25 vs Y45D40	Y45D40	7.2 (2.62%)	76.1%	47	3.36
	Y25D25	-3.23 (-1.17%)	23.9%	11	2.2
Y25D25 vs WOD	WOD	14.89 (5.43%)	98.6%	78	5.57
	Y25D25	-2.86 (-1.01%)	1.4%	0	NaN
Y30D30 vs Y40D40	Y40D40	5.26 (1.91%)	62.8%	33	2.54
	Y30D30	-3.04 (-1.1%)	37.2%	12	2
Y30D30 vs Y45D40	Y45D40	6.71 (2.44%)	65.9%	33	2.75
	Y30D30	-2.39 (-0.86%)	36.4%	12	2
Y30D30 vs WOD	WOD	16.47 (6.01%)	96.4%	77	5.5
	Y30D30	-1.72 (-0.6%)	3.6%	2	2
Y40D40 vs Y45D40	Y45D40	4.56 (1.65%)	56.8%	24	2.4
	Y40D40	-2.58 (-0.93%)	43.2%	18	2.57
Y40D40 vs WOD	WOD	15.24 (5.53%)	90.6%	71	5.46
	Y40D40	-6.25 (-2.21%)	9.4%	0	0
Y45D40 vs WOD	WOD	16.03 (5.79%)	88.5%	70	5.38
	Y45D40	-5.22 (-1.83%)	11.5%	0	0

4.2 AREA 2: RIVER MURRAY, YARRAWONGA TO WAKOOL CONFLUENCE

Comparison across all modelled years
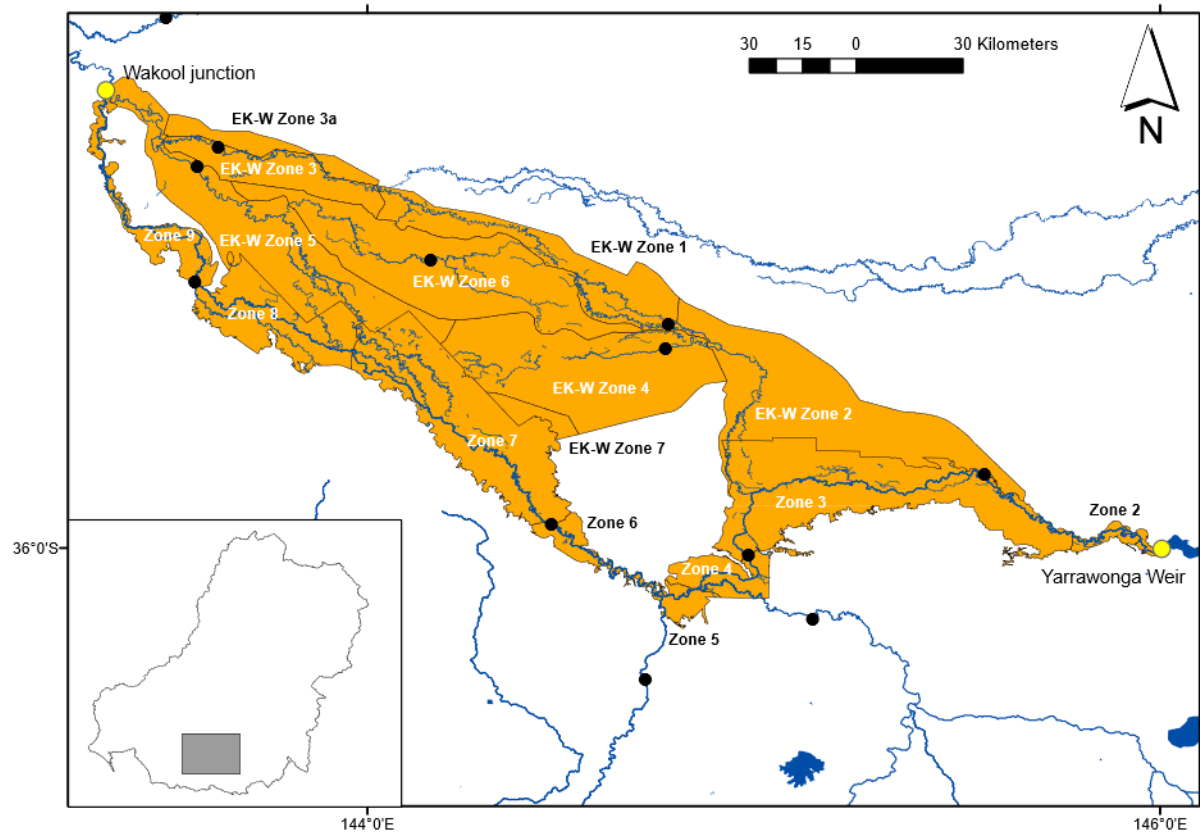
Figure 4.11 Extent of modelled area, Yarrawonga to Wakool confluence (orange filled area), with RiMFIM zones outlined and named. Black dots show associated flow gauges (see Fig. 3.3 for details). Marked yellow circles indicate beginning and end of modelled area. Inset map shows extent (grey area) within the Murray-Darling Basin boundaries.

As with Area 1, discharge timeseries from the different flow options produced relatively similar patterns in cumulative yearly discharge over time for the Yarrawonga-Wakool area (Figure 4.12). Estimates of inundation extent over each year were also similar between the options, although the frequently of moderately sized inundation years was noticeably higher in the without development scenario. Production potential for all of the flow options were therefore again similar across years, although high discharge years increased production potential by an order of magnitude over base levels regardless of the flow option being considered. Again, there was also a greater frequency of moderately increased production potential years in the without development scenario (Figure 4.12).

Figure 4.12 For the Yarrawonga Weir to Wakool junction area, from 1896-2018, comparisons of annual cumulative (a-f) discharge d/s Yarrawonga Weir with (g-l) inundation extent (in ha * days) and (m-r) production potential of large native fish (tonnes C) for each of the modelled flow option scenarios, as well as the without development timeseries ("WOD").

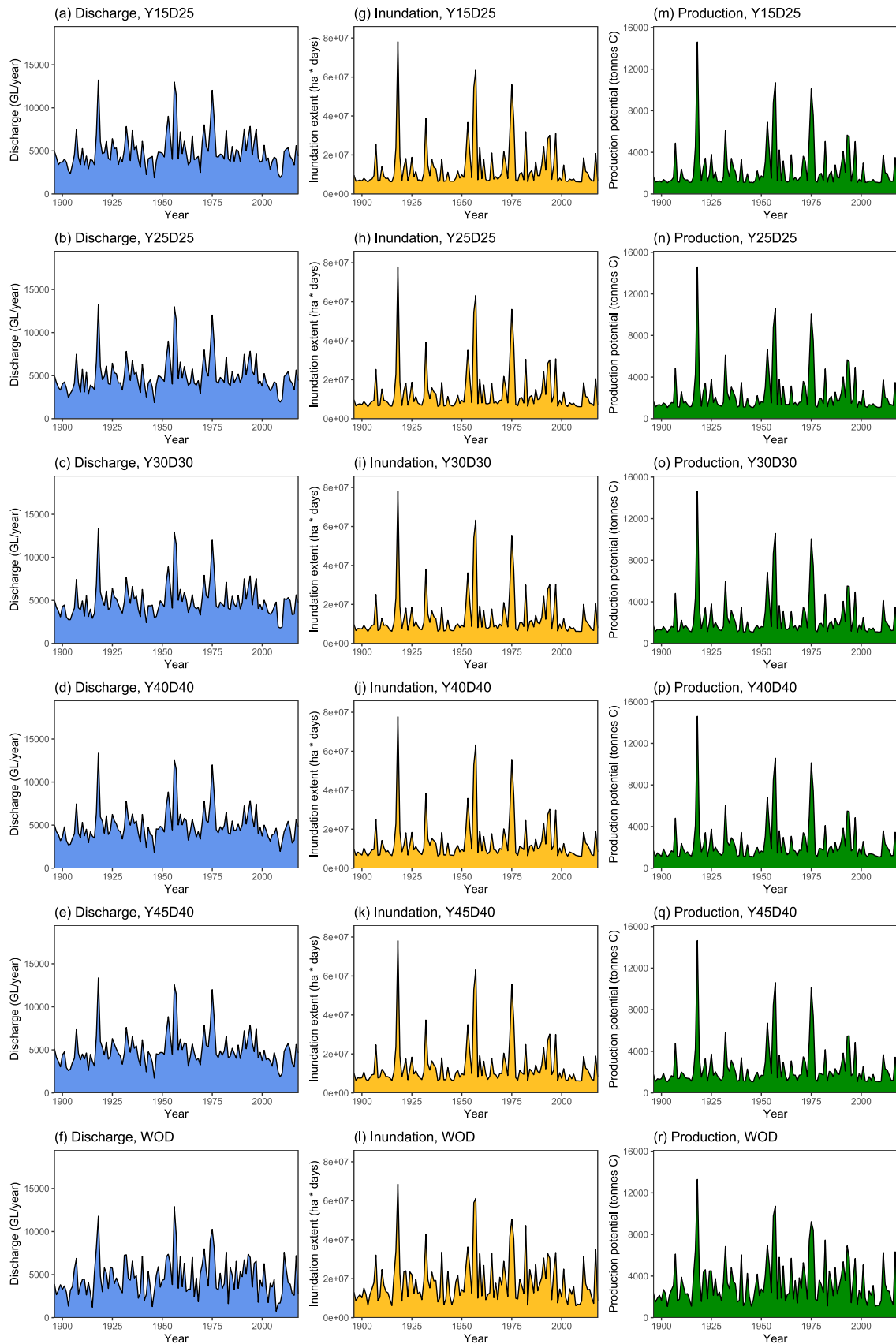In the Yarrawonga to Wakool area, the proportion of years in which the five modelled flow option scenarios produced estimates of large native fish production potential above the 25th percentile of without development estimates was around 39-46% (Figure 4.13). There was thus only a small increase (4.7%) in the proportion of total, modelled years where the models predicted greater outcomes for production potential under the flow options with higher constraints relative to the base option (Y15D25) (Figure 4.13).

Proportion of years >= 25th percentile of WOD results
Y15D25: 39.84 %
Y25D25: 41.46 %
Y30D30: 42.28 %
Y40D40: 45.53 %
Y45D40: 45.53 %
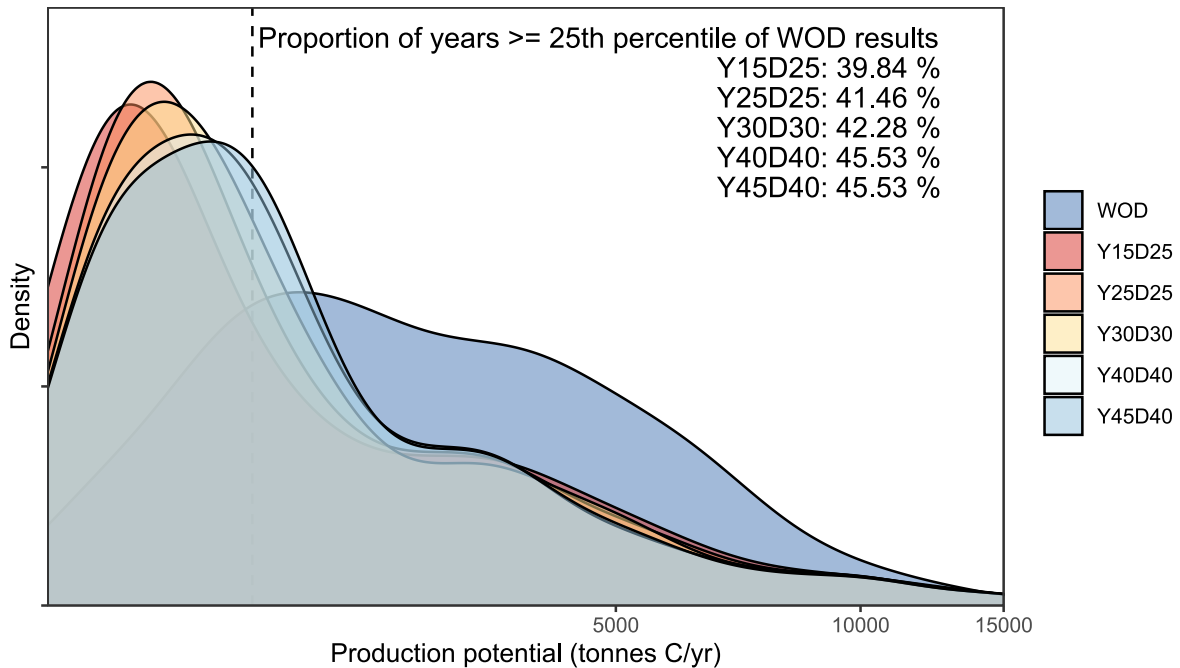
Figure 4.13 Density distributions of model results for the production potential of large native fish from the six different modelled timeseries, across all years (1986-2018), for the Yarrawonga Weir to Wakool junction area. Dashed line indicates 25th percentile of without development ("WOD") results (i.e., value below which lie the lowest 25% of yearly estimates from the WOD model). 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 WOD results. Note the use of nonlinear scale on the x axis (pseudo-log$_{10}$ transformation) to aid in discrimination between density distributions.

There was therefore also a small increase (11.7%) in the 25th percentile of production potential estimates from the Y15D25 base option (1,204 tonnes C) to the Y45D40 option (1,345 tonnes C), although the 25th percentile of WOD scenario estimates was much higher than for estimates from any of the flow options (Figure 4.14). The median yearly production potential across flow options increased by more (14.7%) from the Y15D25 base option (1,498 tonnes C) to the Y45D40 option (1,717 tonnes C), although again the WOD scenario had much higher estimated median production potential (Figure 4.14).
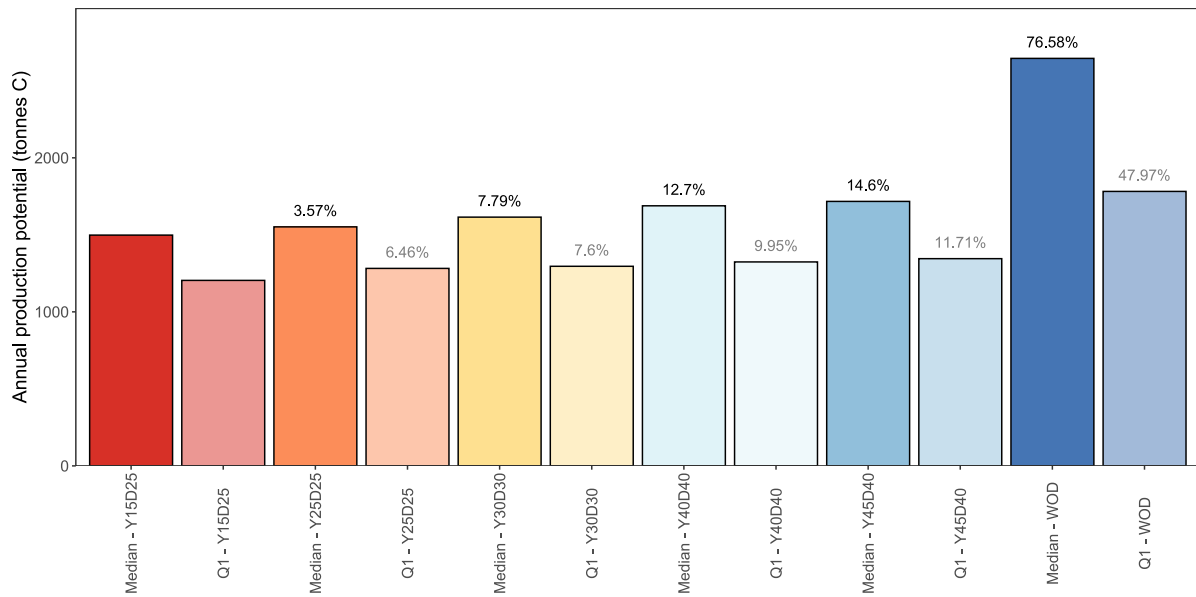
Figure 4.14 Median and 25th percentile (i.e., "Q1") of annual production potential estimates, across all years, for the different flow options associated with the Yarrawonga Weir to Wakool junction area. Numbers above the Y25D25, Y30D30, Y40D40, T45D40, and WOD statistics indicate the difference between that value and the corresponding statistic for the Y15D25 option (a.k.a. "base" option).

As in Area 1, the flow options being assessed against the Y15D25 option produced greater estimates of production potential more often (51 − 54% of years; as opposed to 46 − 49% of years in favour of the Y15D25 model) (Table 4.5), but differences in favour of the Y15D25 flow option were often of greater peak magnitude than differences in favour of the alternative flow option (Figure 4.15). However, the mean differences in favour of the Y15D25 model estimates were lower in comparison with all other flow options, and, as with Area 1, likely occurred in higher-potential years overall given that the differences were lower proportions in any given year (-3.4 − -5.7% vs 6.7 to 14.5% of the lower estimated production potential) (Table 4.5). The flow option with the higher constraint also produced greater estimates in consecutive years more often (41 − 49 total years), although not necessarily longer on average (mean 2.7 − 3.3 years of consecutive periods), than the base option (34 − 41 years; mean 2.7 − 3.2 year periods) (Table 4.5). All of the alternative flow options therefore produced greater estimates of production potential more often than the base Y15D25 flow option.

Figure 4.15 Difference between model estimates for production potential (tonnes C), for the Yarrawonga Weir to Wakool junction area, from the Y15D25 (a.k.a. "base") flow option and (a) the Y25D25 (i.e., Y25D25 - Y15D25 estimate), (b) Y30D30, (c) Y40D40, and (d) Y45D40 model estimates, across all modelled years. Positive values indicate that model estimates are greater for the flow option with the higher constraints limit, and *vice versa*.

Table 4.5 For the Yarrawonga to Wakool area (Area 2), summary statistics for differences in pairwise comparisons of model estimates between the Y15D25 (a.k.a. base option) and other flow options. Shown for the subset of results in which one option produced greater differences (e.g., when the Y25D25 model produced higher estimates than the Y15D25) are the mean differences in 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 produced higher estimates, the total number of years in which the given flow option produced higher estimates across consecutive years (i.e., >= 2 years), and the mean length of those consecutive periods. Results for comparison with the WOD model are also given.

COMPARISON	CONSTRAINT WITH HIGHER PRODUCTION	DIFFERENCE IN PRODUCTION (TONNES C; %)	PROPORTION OF YEARS	TOTAL LENGTH OF CONSECUTIVE PERIODS (YR)	MEAN LENGTH OF CONSECUTIVE PERIODS (YR)
Y15D25 vs Y25D25	Y25D25	91.16 (6.68%)	52.8%	49	3.27
	Y15D25	-88.92 (-3.33%)	47.2%	41	3.15
Y15D25 vs Y30D30	Y30D30	140.36 (10.06%)	52.8%	43	2.87
	Y15D25	-119.26 (-4.42%)	47.2%	34	2.83
Y15D25 vs Y40D40	Y40D40	190.27 (13.92%)	51.2%	41	2.73
	Y15D25	-150.91 (-5.22%)	48.8%	36	3
Y15D25 vs Y45D40	Y45D40	200.69 (14.54%)	53.7%	46	2.88
	Y15D25	-166.2 (-5.7%)	46.3%	35	2.69
Y15D25 vs WOD	WOD	1064.3 (56.44%)	95.1%	117	16.71
	Y15D25	-380.02 (-4.6%)	4.9%	0	0

In all other comparisons between flow options for the Yarrawonga to Wakool area, the flow option with the higher constraints limit produced greater estimates of production potential at approximately the same frequency (48 – 58% of years) than the alternative flow option (42– 52% of years; Table 4.6; Figure 4.16). The absolute mean difference in production potential was slightly greater for the flow option with the higher constraints limit (73.5 – 142.0 tonnes C) than the alternative option (-54.6 – 121.0 tonnes C), although the proportional difference relative to the lower production potential estimate was again greater for the flow option with the higher constraint (4.8 – 8.0%) than the alternative flow option being compared (-3.1 – -5.0%). The flow option with the higher constraint limit also produced greater estimates in consecutive years at a slightly higher frequency (38 – 52 total years) than the alternative option (23 – 47 years), while the mean lengths of consecutive periods were similar between flow options (mean 2.4 – 3.3 year vs. mean 2.3 – 2.9 year periods; Table 4.4).

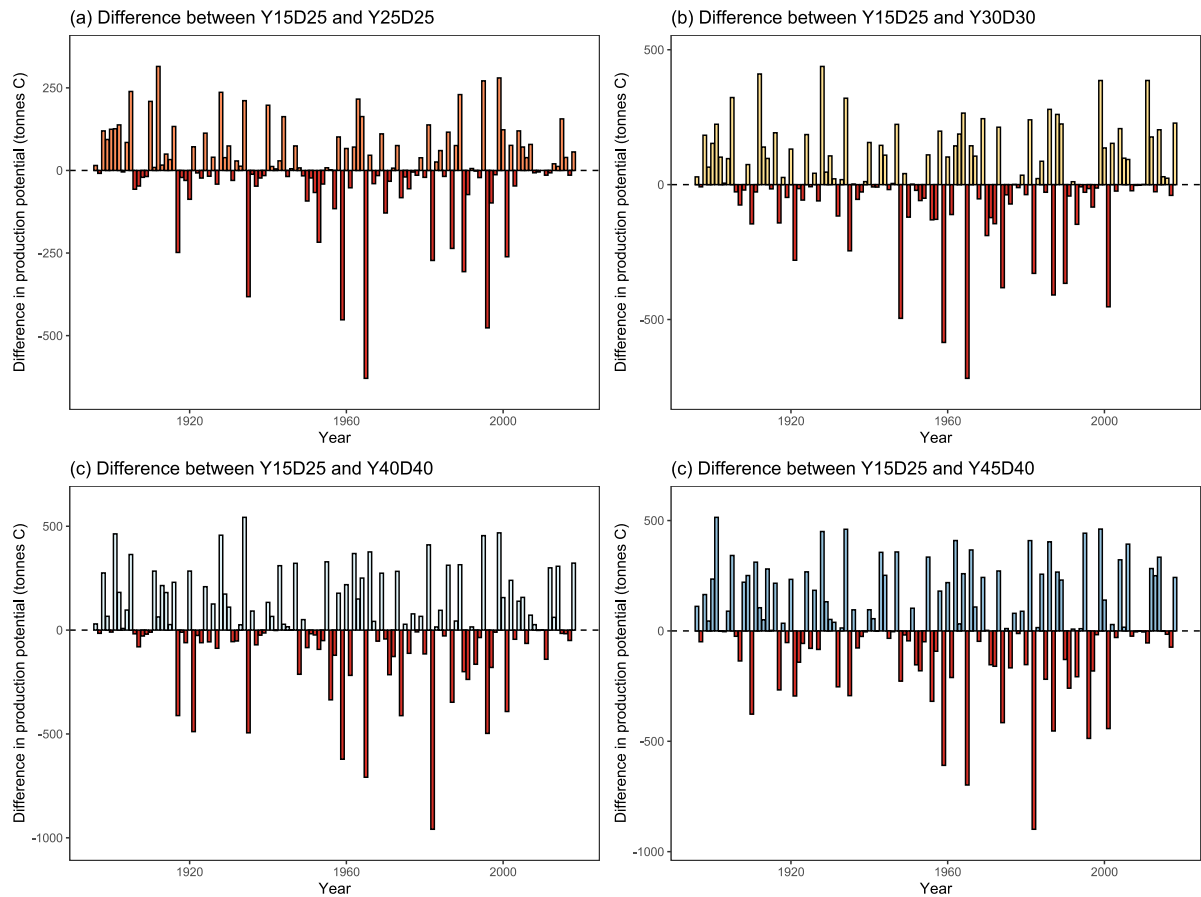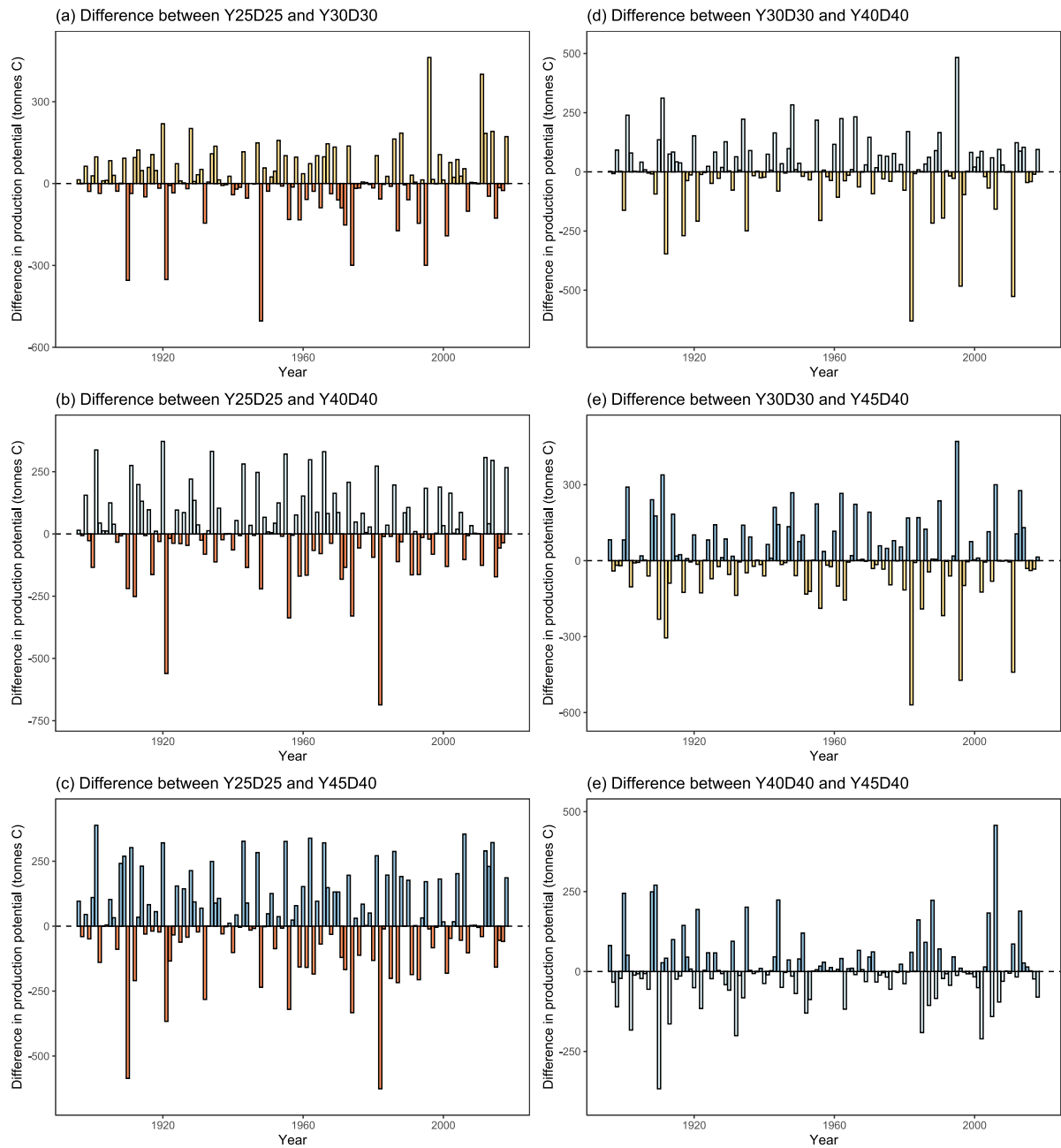**Figure 4.16** Difference between model estimates for production potential (tonnes C) for the Yarrawonga Weir to Wakool junction area, across all modelled years, from flow options not including the "base" model: comparisons are between the Y25D25 and (a) Y30D30 (i.e., Y30D30 - Y25D5 estimates), (b) Y40D40, and (c) Y45D40 model estimates; between the Y30D30 and (d) Y40D40 and (e) Y45D40 estimates; and (f) Y40D40 and Y45D45 estimates. Positive values indicate that estimates are greater for the flow option model with the higher constraints limit, and *vice versa*.

Table 4.6 For the Yarrawonga to Wakool area (Area 2), summary statistics for differences in pairwise comparisons of model estimates between flow options not including the Y15D25 option (a.k.a. base option). Shown for the subset of results in which one option produced greater differences (e.g., when the Y30D30 model produced higher estimates than the Y25D25) are the mean differences in 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 produced higher estimates, the total number of years in which the given flow option produced higher estimates across consecutive years (i.e., >= 2 years), and the mean length of those consecutive periods. Results for comparison with the WOD model are also given.

COMPARISON	CONSTRAINT WITH HIGHER PRODUCTION	DIFFERENCE IN PRODUCTION (TONNES C; %)	PROPORTION OF YEARS	TOTAL LENGTH OF CONSECUTIVE PERIODS (YR)	MEAN LENGTH OF CONSECUTIVE PERIODS (YR)
Y25D25 vs Y30D30	Y30D30	79.35 (4.78%)	57.7%	52	3.25
	Y25D25	-80.68 (-3.79%)	42.3%	30	2.5
Y25D25 vs Y40D40	Y40D40	119.6 (7.98%)	54.5%	40	3.08
	Y25D25	-104.43 (-4.22%)	45.5%	30	2.31
Y25D25 vs Y45D40	Y45D40	142 (9.44%)	55.3%	43	2.39
	Y25D25	-120.95 (-5.00%)	44.7%	23	2.3
Y25D25 vs WOD	WOD	1058.75 (53.25%)	95.1%	117	16.71
	Y25D25	-399.95 (-6.84%)	4.9%	0	0
Y30D30 vs Y40D40	Y40D40	82.41 (5.26%)	56.9%	52	3.06
	Y30D30	-95.14 (-4.23%)	43.1%	32	2.46
Y30D30 vs Y45D40	Y45D40	108.26 (7.03%)	50.4%	38	2.53
	Y30D30	-84.37 (-3.64%)	49.6%	41	2.28
Y30D30 vs WOD	WOD	1045.09 (51.22%)	95.1%	117	16.71
	Y30D30	-373.22 (-4.11%)	4.9%	0	0
Y40D40 vs Y45D40	Y45D40	73.46 (4.57%)	48.0%	41	2.73
	Y40D40	-54.61 (-3.12%)	52.0%	47	2.94
Y40D40 vs WOD	WOD	1099.93 (52.88%)	90.2%	110	10
	Y40D40	-231.89 (-5.35%)	9.8%	0	0
Y45D40 vs WOD	WOD	1053.3 (49.94%)	93.5%	115	12.78
	Y45D40	-332.3 (-6.25%)	6.5%	0	0

Comparisons across years with constraints operating

In contrast with the density distribution of model estimates across all years, the distributions of results from years in which constraints would be operating (i.e., any year where a flow option scenario had a modelled maximum daily discharge below the designated constraints limit) were notably different for each flow option being assessed (Figure 4.17). Consequently, the proportion of years where flow options with higher constraints crossed the 25th percentile of WOD results progressively increased by approximately 20% from the Y15D25 base model (8.6%) to the Y40D40 model (29.3%), although this proportion decreased slightly with the Y45D40 flow option (27.6%) (Figure 4.17).
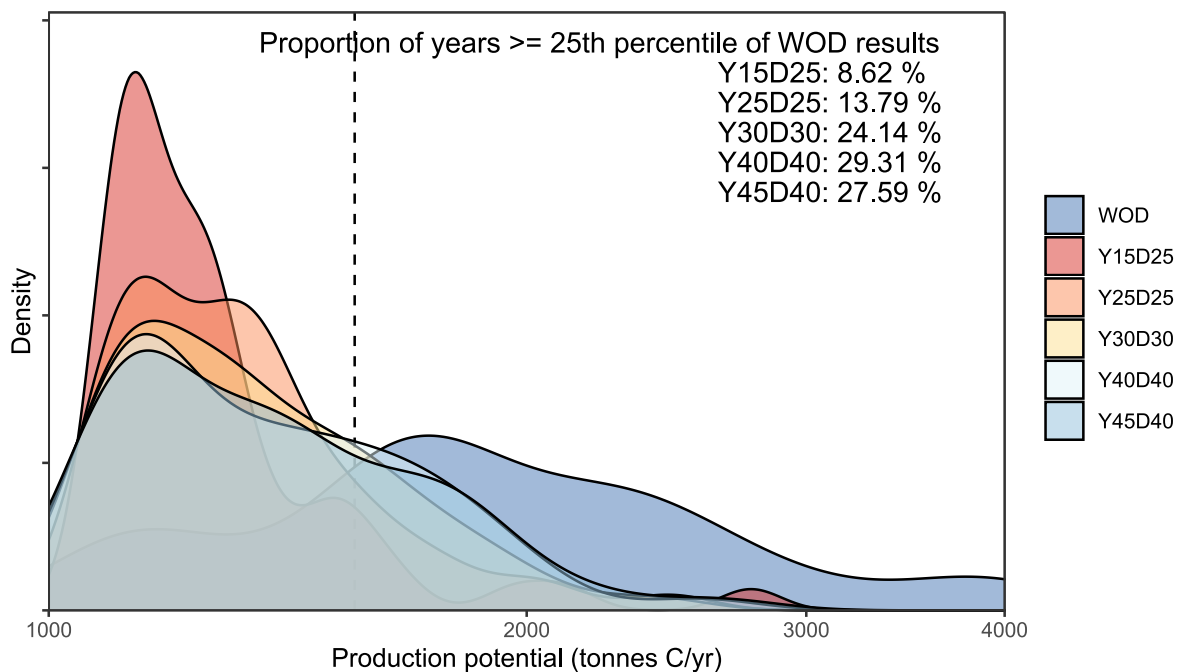
Figure 4.17 Density distributions of model results for the production potential of large native fish from the six different modelled timeseries, across years in which constraints were operating (i.e., any year where a flow option scenario had a modelled maximum daily discharge below the nominal constraints limit), for the Yarrawonga Weir to Wakool junction area. Dashed line indicates 25th percentile of without development ("WOD") results (i.e., value above which 75% of yearly estimates from the WOD 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 WOD results. Note the use of nonlinear scale on the x axis (pseudo-$\log_{10}$ transformation) to aid in discrimination between density distributions.

The increase in the 25th percentile of production potential estimates from the Y15D25 base option (1,072 tonnes C) to the Y40D40 option (1,076 tonnes C) was very low (0.3%), although the 25th percentile of WOD scenario estimates was also lower (1,113 tonnes C) in comparison with the entire set of years (Figure 4.18). The median yearly production potential across flow options also increased by only a small proportion (0.7%) from the Y15D25 base option (1,075 tonnes C) to the Y40D40 option (1,083 tonnes C), while the WOD scenario had slightly higher estimated median production potential in comparison (Figure 4.18).

Figure 4.18 Median and 25th percentile (i.e., "Q1") of annual production potential estimates, across years in which constraints were operating (i.e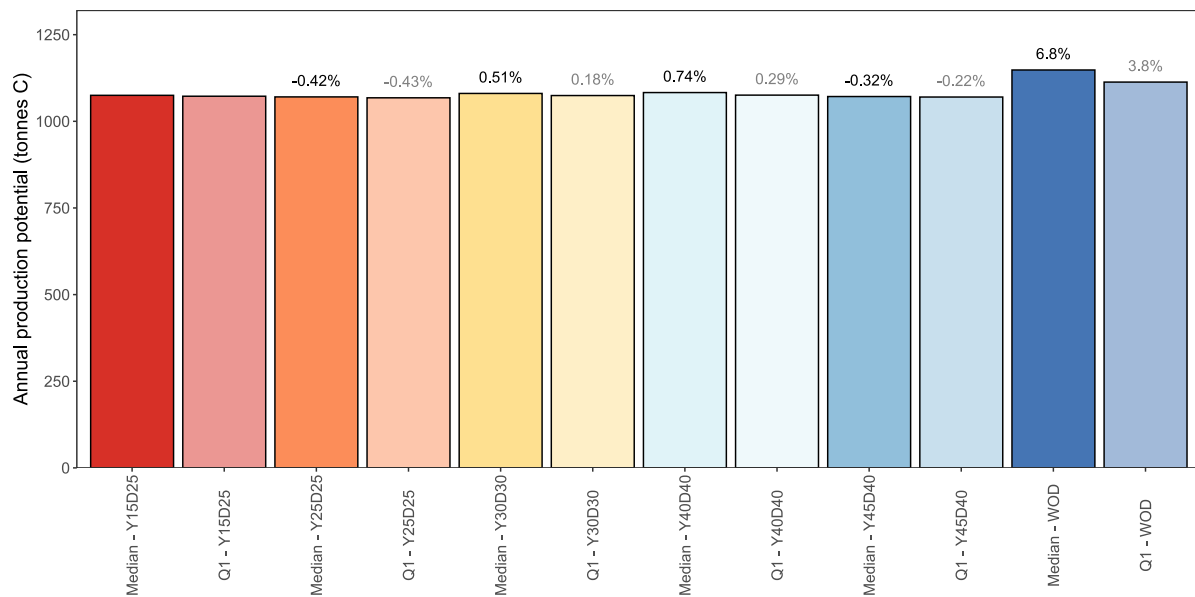., any year where a flow option scenario had a modelled maximum daily discharge below the nominal constraints limit), for the different flow options associated with the Yarrawonga Weir to Wakool junction area. Numbers above the Y25D25, Y30D30, Y40D40, T45D40, and WOD statistics indicate the difference between that value and the corresponding statistic for the Y15D25 option (a.k.a. "base" option).

In years where flow constraints might be operating, the flow options being assessed against the Y15D25 option produced greater estimates of production potential more often (45 – 67% of years) than the base option (33 – 55% of years) (Table 4.7), and differences in favour of the Y15D25 flow option were noticeably of lower peak magnitude than in the comparisons across all years (Figure 4.19). While there was again little difference in whether the Y15D25 or Y25D25 model produced greater estimates, mean differences in favour of the flow option with the higher constraint increased substantially from the Y30D30 option (105.1 tonnes C/yr) to the Y45D40 option (195.1 tonnes C/yr), with little change in the difference in favour of the Y15D25 option (-14.0 – -26.3 tonnes C/yr; Table 4.7). The proportional difference in favour of the flow options with higher constraints was also notably higher (3.1 – 14.9%) than the proportional differences when the Y15D25 option produced higher estimates (-1.3 – -2.0%) (Table 4.7). The flow option with the higher constraint also produced greater estimates in consecutive years more often (2 – 25 total years) than the base option (2 – 12 years), although the mean length of consecutive periods was similar for the Y15D25 option (2.0 – 2.5 years) when compared against the alternatives (2.0 – 2.5 years) (Table 4.7).
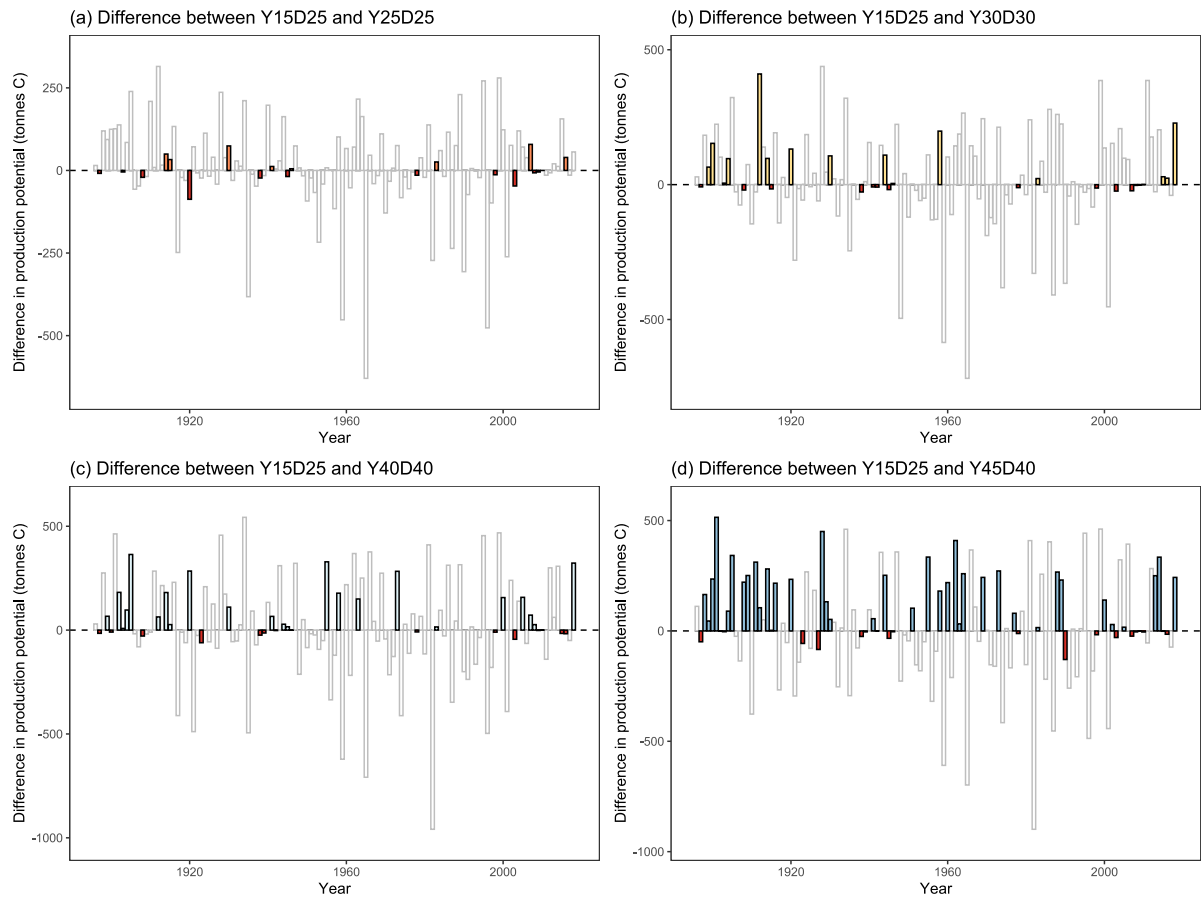
Figure 4.19 Difference between model estimates for production potential (tonnes C) from the Y15D25 (a.k.a. "base") flow option and (a) the Y25D25 (i.e., Y25D25 - Y15D25 estimate), (b) Y30D30, (c) Y40D40, and (d) Y45D40 model estimates, across years in which constraints were operating (i.e., any year where a flow scenario had a modelled maximum daily discharge below the nominal constraints limit), for the Yarrawonga Weir to Wakool junction area. Positive values indicate that estimates are greater for the modelled flow option with the higher constraints limit, and *vice versa*.

Table 4.7 For the Yarrawonga Weir to Wakool junction area (Area 2), summary statistics for differences in pairwise comparisons of model estimates between the Y15D25 (a.k.a. base option) and other flow options, in years where constraints might be 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 Y25D25 model produced higher estimates than the Y15D25) are the mean differences in 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 produced higher estimates, the total number of years in which the given flow option produced higher estimates across consecutive years (i.e., >= 2 years), and the mean length of those consecutive periods.

COMPARISON	CONSTRAINT WITH HIGHER PRODUCTION	DIFFERENCE IN PRODUCTION (TONNES C; %)	PROPORTION OF YEARS	TOTAL LENGTH OF CONSECUTIVE PERIODS (YR)	MEAN LENGTH OF CONSECUTIVE PERIODS (YR)
Y15D25 vs Y25D25	Y25D25	35.3 (3.12%)	45.0%	2	2
	Y15D25	-22.66 (-2%)	55.0%	2	2
Y15D25 vs Y30D30	Y30D30	105.1 (8.56%)	55.2%	6	2
	Y15D25	-14.02 (-1.26%)	44.8%	5	2.5
Y15D25 vs Y40D40	Y40D40	132.62 (10.64%)	63.2%	10	2.5
	Y15D25	-18.08 (-1.6%)	36.8%	4	2
Y15D25 vs Y45D40	Y45D40	195.14 (14.9%)	67.2%	25	2.5
	Y15D25	-26.31 (-1.83%)	32.8%	12	2.4
Y15D25 vs WOD	WOD	204.74 (18.23%)	100.0%	38	3.17
	Y15D25	0 (0%)	0%	0	0

In all other comparisons between flow options for the Yarrawonga to Wakool area, in years where constraints might be operating, differences in the frequency to which the flow option with the higher constraints limit produced greater estimates of production potential were similar (47 – 64% of years) to those in the compared flow option (36 – 53% of years) (Table 4.8) (Figure 4.16). However, the magnitude of differences between the flow option models was higher and varied much more across the options with higher constraints (44.1 – 141.4 tonnes C/yr) than in the options with lower constraints limits being compared (-34.8 to -58.3 tonnes C/yr), and was also greater as a proportion of the lower estimate (3.5 – 10.1% compared with -3.0 – -4.8%;Table 4.8). Differences between the consistency and frequency of these greater production potential years were much closer across comparisons, however, in both the total number of consecutive years (5 – 16 years vs. 2 – 16 years) and mean length of consecutive periods (mean 2.3 – 3.5 year periods vs. 2.0 – 2.7 year periods; Table 4.8).
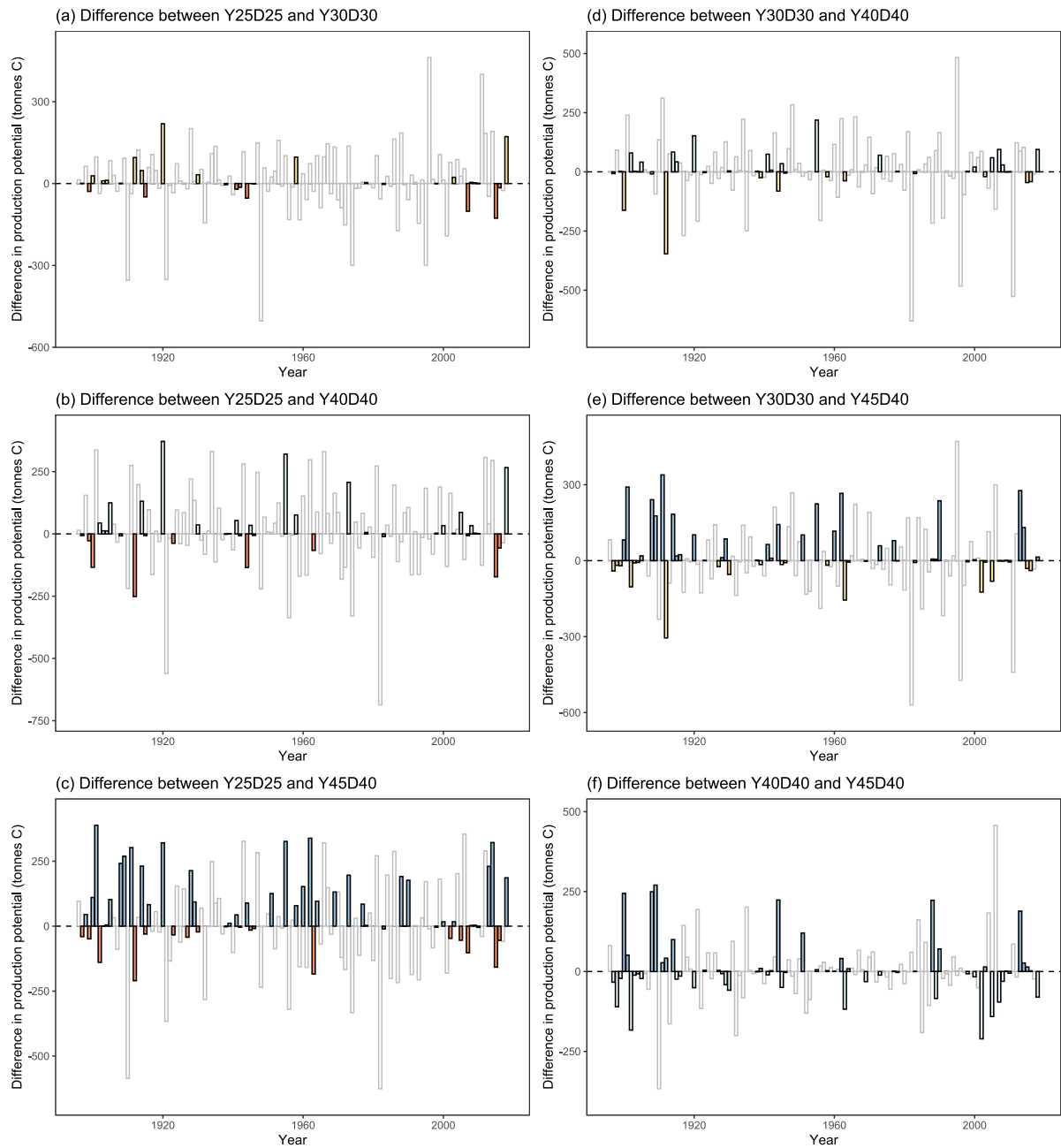
Figure 4.20 Difference between model estimates for production potential (tonnes C) for the Yarrawonga Weir to Wakool junction area, across years in which constraints were operating (i.e., any year where a flow option scenario had a modelled maximum daily discharge below the nominal constraints limit), from flow options not including the "base" model: comparisons are between the Y25D25 and (a) Y30D30 (i.e., Y30D30 - Y25D5 estimates), (b) Y40D40, and (c) Y45D40 model estimates; between the Y30D30 and (d) Y40D40 and (e) Y45D40 estimates; and (f) Y40D40 and Y45D45 estimates. Positive values indicate that estimates are greater for the modelled flow option with the higher constraints limit, and *vice versa*.

Table 4.8 For the Yarrawonga to Wakool area (Area 2), summary statistics for differences in pairwise comparisons of model estimates between flow options not including the Y15D25 option (a.k.a. base option), in years where constraints might be 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 Y30D30 model produced higher estimates than the Y25D25) are the mean differences in 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 produced higher estimates, the total number of years in which the given flow option produced higher estimates across consecutive years (i.e., >= 2 years), and the mean length of those consecutive periods.

COMPARISON	CONSTRAINT WITH HIGHER PRODUCTION	DIFFERENCE IN PRODUCTION (TONNES C; %)	PROPORTION OF YEARS	TOTAL LENGTH OF CONSECUTIVE PERIODS (YR)	MEAN LENGTH OF CONSECUTIVE PERIODS (YR)
Y25D25 vs Y30D30	Y30D30	44.08 (3.45%)	58.60%	5	2.5
	Y25D25	-34.82 (-2.97%)	41.40%	7	2.33
Y25D25 vs Y40D40	Y40D40	84.63 (6.51%)	57.90%	7	3.5
	Y25D25	-58.26 (-4.76%)	42.10%	4	2
Y25D25 vs Y45D40	Y45D40	141.43 (10.14%)	63.80%	18	2.25
	Y25D25	-57.93 (-4.69%)	36.20%	4	2
Y25D25 vs WOD	WOD	288.18 (25.49%)	85%	41	3.15
	Y25D25	-70.62 (-6.45%)	15%	0	0
Y30D30 vs Y40D40	Y40D40	48.65 (3.65%)	60.50%	10	2.5
	Y30D30	-54.13 (-4.31%)	39.50%	2	2
Y30D30 vs Y45D40	Y45D40	106.85 (7.33%)	53.40%	16	2.29
	Y30D30	-40.85 (-3.23%)	46.60%	16	2.29
Y30D30 vs WOD	WOD	371.65 (30.44%)	86.20%	40	2.86
	Y30D30	-15.13 (-1.37%)	13.80%	0	0
Y40D40 vs Y45D40	Y45D40	72.16 (5.41%)	46.60%	10	2.5
	Y40D40	-47.86 (-3.8%)	53.40%	16	2.67
Y40D40 vs WOD	WOD	521.07 (41.13%)	78.90%	34	2.83
	Y40D40	-33.22 (-2.97%)	21.10%	0	0
Y45D40 vs WOD	WOD	627.46 (42.63%)	91.40%	38	3.17
	Y45D40	-42.48 (-3.12%)	8.60%	0	0

5. Discussion

For both project areas, the Y30D30, Y40D40, and Y45D40 flow options provided consistent increases in production potential estimates over the base Y15D25 option, although these differences were relatively small in comparison to the potential production of large native fish without regulation ("WOD" option) (Figure 5.1a, c). However, these calculations included years with unregulated high flow events, which have a disproportionate effect on the calculation of mean annual values and are unlikely to vary across flow option scenarios (as they exceed the highest constraint anyway). Examining the years where maximum flows fall below the designated constraints options for each flow option may thus provide a better estimate of when differences between those options might have a direct effect on the production potential of large native fish. In this respect, the Y30D30, Y40D40, and Y45D40 flow option models produced greater mean estimates of production potential, relative to the base option, in years where constraints might be operating than across all years (Figure 5.1b, d). These differences correspond to 8 – 20% of the mean increase in potential production without development for those years (Figure 5.1b, d). When compared to the median production potential of the base Y15D25 scenario, the net difference in favour of the RRC flow options (from the Y25D25 to Y45D40 scenario) corresponds to 0.01 – 1.7% for the Hume to Yarrawonga area (median of Y15D25: 275.2 tonnes C) and 4.4 – 10.1% for the Yarrawonga to Wakool area (median of Y15D25: 1210.9 tonnes C), for years in which constraints might operate.

Years in which flows are disproportionately high or low affect the ability to assess differences between constraints options in a number of ways. As above, unregulated high flow events have a disproportionate effect on the calculation of mean annual values. Very high flows are also likely to provide a relatively high proportion of the annual sum of inundation and production potential in any given year, and thus make it difficult to assess variation between options when flows are lower for the rest of the year. Low flow years will also skew mean values and are likely to provide equivalent estimates for different constraints options (especially where maximum daily values are well below the lowest constraints level). It must be stressed again that the estimates here are analogous to the maximum carrying capacity – the potential energy that could be available to fuel production of large native fish – and are not predictions of production itself (e.g., biomass or abundance of large fish). Yet, it is during these lower-flow years when maintenance of threshold levels of basal production and energy flows might be critically important in maintaining populations in the real world, e.g., in sustaining viable population sizes, buffering populations against disturbance events, and supporting an appropriate diversity of food resources in the periods between high-flow, "boom" events.

In this context, another important result from comparisons of the different flow options is that consecutive periods of increased production over the base Y15D25 option occur more frequently moving from the Y30D30 up to the Y45D40 options, rather than fluctuating between years where the production potential of large native fish alternately exceeds or falls short of the current, base scenario (see section 4). Further, while the absolute difference in production potential of large native fish may only increase to around 20% of the potential without development in the latter flow option scenarios, the total number of years which encompassed consecutive periods of greater differences over the base scenario were above 50% of those in the WOD option for these same scenarios. While production potential in these flow options may thus be lower than the potential without development, they may be able to sustain this level of production potential over the base scenario relatively consistently. In terms of ecological dynamics, fluctuations in the carrying capacity of populations – and consequently, the probability of populations being suddenly restricted by energy supply – might thus be reduced under the alternative flow options, despite a relatively low carrying capacity overall.
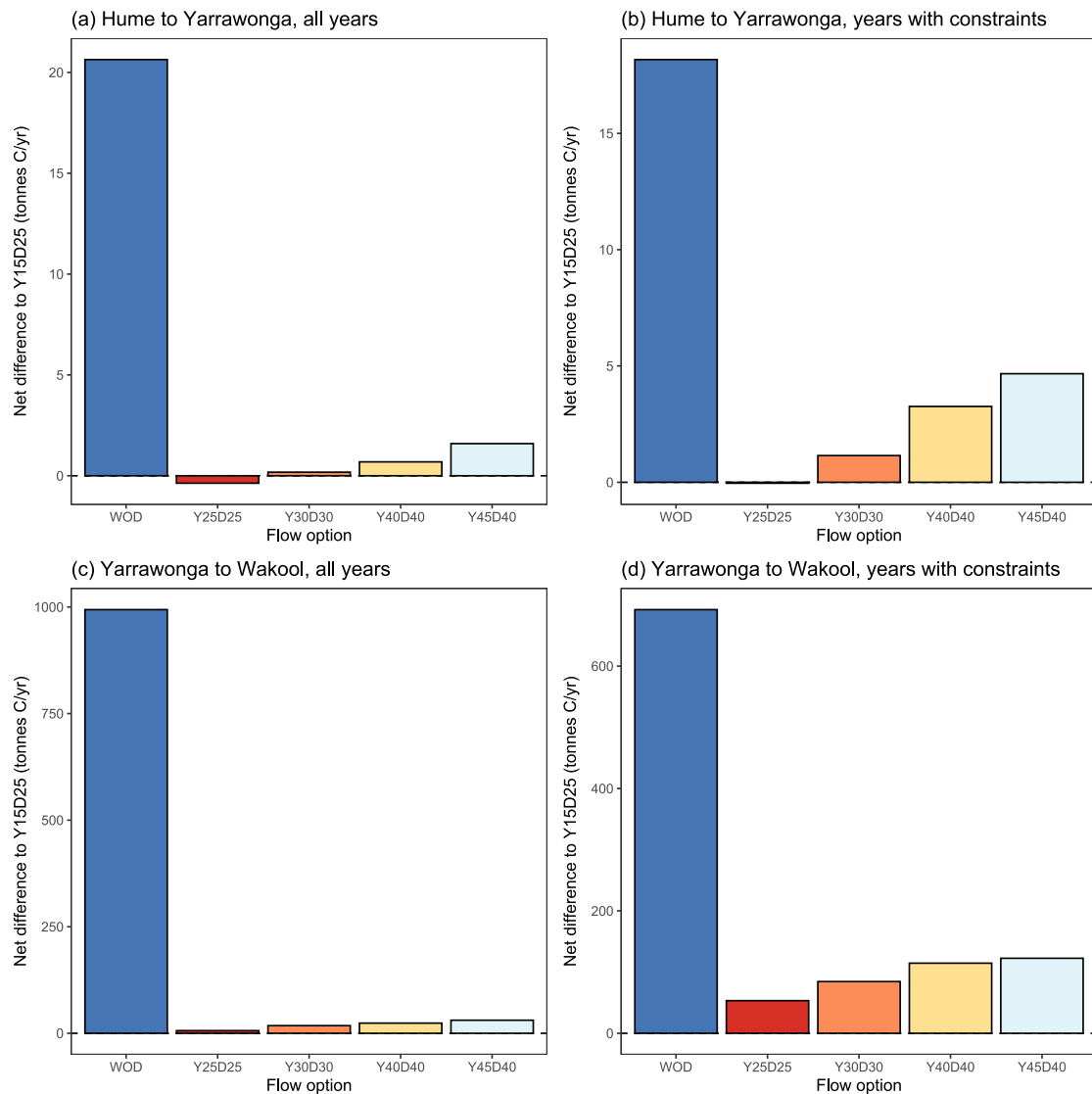
Figure 5.1 Summary of results examining the mean, net difference in production potential estimates between the base Y15D25 flow option and each other option (mean of compared option estimate – Y15D25 estimate), for both project areas, for either all modelled years (a, c) or years in which maximum daily discharge was below the designated constraints limit for either flow option (b, d) (or, for the WOD scenario, for any flow option).

5.1 EVALUATION OF CONSTRAINTS OPTIONS

We chose to present comparisons of estimated production potential across flow options at two levels: (i) comparison of the distribution of responses across all years and; (ii), comparisons in years where the maximum daily flow was affected by constraints limits. In this respect, we sought to predict both the overall effect of changing flow options and the effect in years where river regulation might directly affect the magnitude of the floodplain inundation. These analyses focus on mean responses, rather than cumulative responses or long-term trends, as those alternative approaches are likely not appropriate for the underlying data structure.

First, comparisons of the cumulative differences in production potential across all years (i.e., the sum of potential added to the sum of all previous years) are problematic given the non-stationary nature of the time series. There is also little difference in these overall estimates (e.g., ca. 0.3-0.7% differences in cumulative production potential from 1896-2018 for the Yarrawonga to Wakool area) due to the high proportion of years in which the flow option with the lower constraints limit estimates greater production potential (see e.g. Figures

4.4 and 4.11) and the overriding effect of unregulated flow events on estimate sums (see above). Comparison of long-term trends across the time series (e.g., comparing generalised additive models using cross-validation) would also be problematic, as the production potential data are noisy and so do not fit this modelling approach well: e.g., additive model fits across the flow option estimates for the Yarrawonga to Wakool area only predict ca. 5-10% of the deviance in production potential by time. Our approach is therefore also dictated to a large extent by the nature of the underlying discharge time series. Since the flow options relax the limits on regulated flow events rather than increasing the cumulative flow being delivered to the system, differences in production potential between years may often be "cancelled out" by the compared flow option receiving more discharge, or a longer duration of peak flows, in the following year.

5.2 FUTURE OPTIONS: INUNDATION MODELLING

We incorporated a single addition to the structure of inundation modelling developed by Bond et al. (in prep) by estimating the persistence of surface water when daily discharge drops below inundation thresholds (see section 3.2). Our approach takes a scaling relationship between the difference in discharge and inundation threshold, as well as the preceding number of inundated days, to model broadly applicable patterns in surface water persistence. We consider this approach appropriate here for two reasons. First, surface water is likely to persist for longer if the area has already been inundated for a longer time, as bank storage and groundwater levels have likely been increased and will result in longer conservation of surface water if discharge drops (Jolly & Rassam 2009). Secondly, more rapid drops in discharge below an inundation threshold are likely to result in drying of surface water more quickly because (i) "ungauged" discharge, or a reduction in flow measurements without a drop in water levels that still connect rivers and floodplains, is more likely to occur when discharge is higher overall, and; (ii) there is less likely to be sub-surface connectivity between the river channel and inundated off-channel areas which may help maintain surface water (Jolly & Rassam 2009). In reality, surface water persistence in off-channel habitats is likely to be a more complex factor based on local topography, antecedent rainfall patterns, and soil moisture and groundwater dynamics (e.g., Karim et al. 2020). However, our approach provides a relatively simple method of estimating surface water persistence that is a substantial improvement over no consideration, in which drops in discharge below inundation thresholds unrealistically "reset" algal and macrophyte production (see section 3). It is worth noting, however, that the contribution of standing water on floodplains to production within the river system as a whole is both largely unknown (see: Appendix 4) and is likely to be highly dependent on the temporal and spatial patterns of hydrological (and biological) connectivity, which are currently out of scope for us to parameterise (see: 5.3 below).

A future alteration to the inundation modelling approach could focus on improved consideration of low flows. At present, all flows below the lowest inundation threshold in RiM-FIM (typically 3,000 ML/day) are scaled up to that threshold. Variability in production potential under low flows is therefore not modelled, and all areas under the lowest threshold are assumed to be perennially inundated. Our approach therefore does not consider any changes to productivity and food web energy flows that may occur as channels experience low flows, which may have substantial implications for ecosystem energetics (Rolls et al. 2012). Further, we do not currently model zero flows or drying, and so our model continues predicting production even when channels might be dry. However, our modelling approach strongly reflects the structure of the underlying inundation models. Both the base RiM-FIM and Edward-Wakool RiM-FIM extensions are constrained by the resolution and frequency of imagery used to calibrate the models (Overton et al. 2006, Sims et al. 2014), which results in lower model accuracy when surface water extent is low (and thus a "lowest" inundation threshold that does not reflect the potential lowest discharge or no flows). In addition, neither inundation model considers hysteresis in surface water persistence. Yet at the present time, more detailed consideration of low flows in our approach is unlikely to affect the evaluation of constraints options here, given that unregulated flow events contribute so disproportionately to overall, annual estimates of production potential (see above). In addition, the aggregation of sections of the River Murray with very different geomorphological settings (e.g., Barmah-Millewa and Koondrook-Perricoota vs. constrained sections of the river between these areas) means that local differences

in the modelling of inundation under both low and high-flow conditions is likely averaged out at the scales we model. We caution that any modelling approach which seeks to mechanistically estimate population demographics, abundance, or species interactions during low flows will therefore need to provide much more detail in this respect than we currently do.

5.3 FUTURE OPTIONS: PRODUCTIVITY AND FOOD WEBS

Here, we estimated production potential at an annual scale. This approach was taken to "average out" several ecological processes that would have otherwise required a much more complicated model, particularly with respect to hysteresis in productivity and trophic interactions (i.e., dependence on antecedent conditions). There are several processes we do not currently model explicitly that might be considered most important in this respect: (i) productivity rates, for both autotrophic and heterotrophic basal resources, are likely to be affected by the recent history of inundation, drying, and discharge in any given area (Humphries et al. 2014); (ii) basal productivity may not be transferred to progressively higher trophic levels within the annual timescale (particularly where basal productivity occurs in the latest months of the year), instead potentially requiring months for primary and secondary consumers to grow and ultimately support the largest predators; and (iii) organisms can move between habitats, and water can flow between areas, resulting in a network of ecological interactions which depend upon where inflows are coming from and the productivity and trophic state of the areas from which they originated (i.e., downstream effects; Loreau et al. 2003). We could therefore consider incorporating more detailed inundation time-productivity relationships (e.g., Burford et al. 2012), lag times for transfer of production through the modelled food web, organism movement between areas (and barriers to this movement), and upstream effects on downstream systems to the model structure in the future. Given the difficulty we faced in finding appropriate studies to update model parameters for a number of food web properties, however (see: Appendix 2-4), we suggest that further complicating the energetics model by incorporating hysteresis would likely be restricted by a lack of information on how these processes are regulated by their history and position in the landscape.

Food web structure and the associated dominance and efficiency values we assigned to our modelled web, are the most complex part of our model overall. They therefore also represent the aspect which might benefit most from more exact parameterisation. Further updates to productivity rates and environmental effects on those rates may be a good place to start, for several reasons. First, (i) the habitat classification system we use is relatively broad, given that the ANAE classification system we base ours on includes over 40 different classes for freshwater and floodplain ecosystems (see: Appendix 1). In particular, the dominant vegetation type in ephemeral wetlands and floodplains is likely to substantially affect nutrient cycling and organic matter lability, as well as light and substrate availability, and thus productivity (e.g., Zhang et al. 2021). Light and temperature availability are also likely to vary substantially across aquatic habitats, particularly across inundated floodplains, and thus affect primary productivity to a potentially large degree (Mulholland et al. 2001), but our model currently has little consideration of how these parameters might both affect primary productivity rates and the quality (i.e., nutritional value or trophic efficiencies associated thereof) of primary production. In a related issue, (ii) the modelling of particulate organic matter (POM) and dissolved organic carbon (DOC) in our model treats the entire resource as a homogenous entity, with a single dominance and efficiency value assigned to links with this resource. In reality, POM and DOC are a highly heterogenous mixture of differing substances with varying energy and nutrient value to consumers (Baldwin 1999). A corollary to this issue is that little is known of how accumulation and decomposition of plant litter varies under different floodplain vegetation types, which in turn will affect its integration into food webs upon inundation. Finally, (iii) the modelling of temperature effects on productivity could be modelled more specifically. For example, productivity and assimilation efficiency are linked in photosynthetic algae (García-Carreras et al., 2018), and the effect of the same increase in temperature on productivity may therefore increase by different scaling factors (see: Appendix 1) at different temperature ranges (e.g., a 10°C increase from 10-20°C may scale productivity by a factor of 1.066, but an increase from 20-30°C may scale productivity by 1.07). Again, however, the paucity of

studies from which we were able to obtain updated model parameters in this respect (see: Appendix 2-4) is likely to make any attempt to further update the energetics models a difficult task.

6. References

Adame, M.F., Pettit, N.E., Valdez, D., Ward, D., Burford, M.A. and Bunn, S.E., 2017. The contribution of epiphyton to the primary production of tropical floodplain wetlands. Biotropica 49, 461-471.

AETG: Aquatic Ecosystems Task Group, 2012. Aquatic ecosystems toolkit: module 2 interim Australian national aquatic ecosystem classification framework. Australian Government Department of Sustainability, Environment, Water, Population and Communities, Canberra

Andersen, K.H., Beyer, J.E. and Lundberg, P., 2009. Trophic and individual efficiencies of size-structured communities. Proceedings of the Royal Society B: Biological Sciences 276, 109-114.

Anderson, T.R. and Tang, K.W., 2010. Carbon cycling and POC turnover in the mesopelagic zone of the ocean: Insights from a simple model. Deep Sea Research Part II: Topical Studies in Oceanography 57, 1581-1592.

Armstrong, G. and Booth, D.T., 2005. Dietary ecology of the Australian freshwater turtle (Elseya sp.: Chelonia: Chelidae) in the Burnett River, Queensland. Wildlife Research, 32, 349-353.

Baumgartner L.J., 2007. Diet and feeding habits of predatory fishes upstream and downstream of a low-level weir. Journal of Fish Biology, 70, 879−894.

Balcombe, S.R. and Humphries, P., 2006. Diet of the western carp gudgeon (Hypseleotris klunzingeri Ogilby) in an Australian floodplain lake: the role of water level stability. Journal of Fish Biology, 68, 1484-1493.

Baldwin, D.S., 1999. Dissolved organic matter and phosphorus leached from fresh and 'terrestrially'aged river red gum leaves: implications for assessing river−floodplain interactions. Freshwater Biology, 41, 675-685.

Baldwin, D.S., Colloff, M.J., Mitrovic, S.M., Bond, N.R. and Wolfenden, B., 2016. Restoring dissolved organic carbon subsidies from floodplains to lowland river food webs: a role for environmental flows?. Marine and Freshwater Research 67, pp.1387-1399.

Beesley, L., Price, A., King, A., Gawne, B., Nielsen, D., Koehn, J., Meredith, S., Vilizzi, L. and Hladyz, S., 2011. Watering floodplain wetlands in the Murray−Darling Basin for native fish. Waterlines report, National Water Commission, Canberra.

Bellmore, J.R., J. R. Benjamin, M. Newsom, J. A. Bountry, and D. Dombroski, 2017. Incorporating food web dynamics into ecological restoration: a modeling approach for river ecosystems. Ecological Applications 27, 814-832.

Bond, N.R., Thomson, J., Yen, J.D.L., Butler, G.L., Crook, D., Hoenberg, D., Humphries, P., Kennard, M.J., K. Kopf, K.J.D, McCasker, N.G,. Morrongiello, J., Nielsen, D. and Reich, P., in prep. Quantifying the effects of altered hydrology and food-web structure on ecosystem carrying capacity in a large floodplain river.

Briggs, S.V. and Maher, M.T., 1985. Limnological studies of waterfowl habitat in south-western New South Wales. II. Aquatic macrophyte productivity. Marine and Freshwater Research 36, pp.707-715.

Brown, J.H., Gillooly, J.F., Allen, A.P., Savage, V.M. and West, G.B., 2004. Toward a metabolic theory of ecology. Ecology 85, pp.1771-1789.

Bunn, S.E. and Boon, P.I., 1993. What sources of organic carbon drive food webs in billabongs? A study based on stable isotope analysis. Oecologia 96, 85-94.

Bunn, S.E., Thoms, M.C., Hamilton, S.K. and Capon, S.J., 2006. Flow variability in dryland rivers: boom, bust and the bits in between. River Research and Applications, 22, 179-186.

Burford, M.A., Cook, A.J., Fellows, C.S., Balcombe, S.R. and Bunn, S.E., 2008. Sources of carbon fuelling production in an arid floodplain river. Marine and Freshwater Research, 59, 224-234.

Burford, M.A., Revill, A.T., Palmer, D.W., Clementson, L., Robson, B.J. and Webster, I.T., 2011. River regulation alters drivers of primary productivity along a tropical river-estuary system. Marine and Freshwater Research, 62, 141-151.

Burford, M.A., Webster, I.T., Revill, A.T., Kenyon, R.A., Whittle, M. and Curwen, G., 2012. Controls on phytoplankton productivity in a wet−dry tropical estuary. Estuarine, Coastal and Shelf Science, 113, 141-151.

Burford, M.A., Valdez, D., Curwen, G., Faggotter, S.J., Ward, D.P. and Brien, K.O., 2016. Inundation of saline supratidal mudflats provides an important source of carbon and nutrients in an aquatic system. Marine Ecology Progress Series 545, 21-33.

Burns, A. and Walker, K.F., 2000. Biofilms as food for decapods (atyidae, Palaemonidae) in the river Murray, south australia. Hydrobiologia, 437, 83-90.

Butcher, J.B. and Covington, S., 1995. Dissolved-oxygen analysis with temperature dependence. Journal of Environmental Engineering 121, 756-759.

Cadwallader, P.L. and Eden, A.K., 1979. Observations on the food of Macquarie perch, Macquaria australasica (Pisces: Percichthyidae), in Victoria. Marine and Freshwater Research, 30, 401-409.

Cadwallader, P.L., Eden, A.K. and Hook, R.A., 1980. Role of streamside vegetation as a food source for Galaxias olidus Gnnther (Pisces: Galaxiidae). Marine and Freshwater Research, 31, 257-262.

Carr, R. and Podger, G., 2012. eWater source—Australia's next generation IWRM modelling platform. In 34th Hydrology and Water Resources Symposium (pp. 742-749).

Chessman, B.C., 1984. Food of the snake-necked turtle, Chelodina longicollis (Shaw)(Testudines: Chelidae) in the Murray Valley, Victoria and New South Wales. Wildlife Research, 11, 573-578.

Chester, H. and Norris, R., 2006. Dams and flow in the Cotter River, Australia: effects on instream trophic structure and benthic metabolism. Hydrobiologia, 572, 275-286.

Deegan, B.M. and Ganf, G.G., 2008. The loss of aquatic and riparian plant communities: implications for their consumers in a riverine food web. Austral Ecology, 33, 672-683.

Deegan, B.M., Lamontagne S., Aldridge K.T. and Brookes, J.D., 2009. Trophodynamics of the Coorong. CSIRO: Water for a Healthy Country National Research Flagship.

Del Giorgio, P.A. and Cole, J.J., 1998. Bacterial growth efficiency in natural aquatic systems. Annual Review of Ecology and Systematics 29, 503-541.

Dell, A.I., Pawar, S. and Savage, V.M., 2011. Systematic variation in the temperature dependence of physiological and ecological traits. Proceedings of the National Academy of Sciences 108, 10591-10596.

Demars, B.O., Russell Manson, J., Olafsson, J.S., Gislason, G.M., Gudmundsdottir, R., Woodward, G.U.Y., Reiss, J., Pichler, D.E., Rasmussen, J.J. and Friberg, N., 2011. Temperature and the metabolic balance of streams. Freshwater Biology 56, 1106-1121.

Driver, P.D., Harris, J.H., Closs, G.P. and Koen, T.B., 2005. Effects of flow regulation on carp (Cyprinus carpio L.) recruitment in the Murray−Darling Basin, Australia. River Research and Applications 21, 327-335.

Duffy, R.E., Godwin, I., Purvis, I. and Nolan, J., 2013. The contribution of naturally occurring food items to the diet of Bidyanus bidyanus when fed differing formulated diets. Journal of Applied Aquaculture, 25, 206-218.

Ebner, B., 2006. Murray cod an apex predator in the Murray River, Australia. Ecology of Freshwater Fish, 15, 510-520.

Ebner, B.C., Donaldson, J.A., Marshall, J., Starrs, D. and Freeman, A.B., 2021. Diving beetles strip eel to the bone. Food Webs, 27, p.e00188.

Ernest, S.M., Enquist, B.J., Brown, J.H., Charnov, E.L., Gillooly, J.F., Savage, V.M., White, E.P., Smith, F.A., Hadly, E.A., Haskell, J.P. and Lyons, S.K., 2003. Thermodynamic and metabolic effects on the scaling of production and population energy use. Ecology Letters 6, 990-995.

Faggotter, S.J., Webster, I.T. and Burford, M.A., 2013. Factors controlling primary productivity in a wet–dry tropical river. Marine and Freshwater Research 64, 585-598.

Field, C.B., Behrenfeld, M.J., Randerson, J.T. and Falkowski, P., 1998. Primary production of the biosphere: integrating terrestrial and oceanic components. Science 28, 237-240.

García-Carreras, B., Sal, S., Padfield, D., Kontopoulos, D.G., Bestion, E., Schaum, C.E., Yvon-Durocher, G. and Pawar, S., 2018. Role of carbon allocation efficiency in the temperature dependence of autotroph growth rates. Proceedings of the National Academy of Sciences 115, E7361-E7368.

Gawne, B., Merrick, C., Williams, D.G., Rees, G., Oliver, R., Bowen, P.M., Treadwell, S., Beattie, G., Ellis, I., Frankenberg, J. and Lorenz, Z., 2007. Patterns of primary and heterotrophic productivity in an arid lowland river. River Research and Applications 23, 1070-1087.

Giling, D., Reich, P. and Thompson, R.M., 2009. Loss of riparian vegetation alters the ecosystem role of a freshwater crayfish (Cherax destructor) in an Australian intermittent lowland stream. Journal of the North American Benthological Society, 28, 626-637.

Gordon, B.A., Dorothy, O. and Lenhart, C.F., 2020. Nutrient retention in ecologically functional floodplains: A review. Water, 12, 2762.

Grace, M.R. and Imberger, S.J., 2006. Stream metabolism: performing & interpreting measurements. Water Studies Centre Monash University, Murray Darling Basin Commission and New South Wales Department of Environment and Climate Change 204, pp.1-204.

Graham, R. and Harris, J.H., 2005. Floodplain inundation and fish dynamics in the Murray-Darling Basin. Current concepts and future research: a scoping study. CRC for Freshwater Ecology, 51.

Growns, I., Ryder, D. and Frost, L., 2020a. The basal food sources for Murray cod (Maccullochella peelii) in wetland mesocosms. Journal of Freshwater Ecology, 35, 235-254.

Growns, I., Ryder, D., McInerney, P., Bond, N., Holt, G., Lester, R. and Thompson, R., 2020b. The use of fatty acids to identify food sources of secondary consumers in wetland mesocosms. Journal of Freshwater Ecology, 35, 173-189.

Hadwen, W.L., Fellows, C.S., Westhorpe, D.P., Rees, G.N., Mitrovic, S.M., Taylor, B., Baldwin, D.S., Silvester, E. and Croome, R., 2010. Longitudinal trends in river functioning: patterns of nutrient and carbon processing in three Australian rivers. River Research and Applications, 26, 1129-1152.

Hardy, C.M., Krull, E.S., Hartley, D.M. and Oliver, R.L., 2010. Carbon source accounting for fish using combined DNA and stable isotope analyses in a regulated lowland river weir pool. Molecular Ecology, 19, 197-212.

Hladyz, S., Nielsen, D.L., Suter, P.J. and Krull, E.S., 2012. Temporal variations in organic carbon utilization by consumers in a lowland river. River Research and Applications, 28, 513-528.

Humphries, P., Keckeis, H. and Finlayson, B., 2014. The river wave concept: integrating river ecosystem models. BioScience, 64, 870-882.

Irvin, S., Coman, G., Musson, D., Doshi, A. and Stokes, C., 2018. Aquaculture viability. A technical report to the Australian Government from the CSIRO Northern Australia Water Resource Assessment, part of the National Water Infrastructure Development Fund: Water Resource Assessments. CSIRO, Australia.

Jennings, S., Warr, K.J. and Mackinson, S., 2002. Use of size-based production and stable isotope analyses to predict trophic transfer efficiencies and predator-prey body mass ratios in food webs. Marine Ecology Progress Series 240, 11-20.

Jolly, I.D. and Rassam, D.W., 2009. A review of modelling of groundwater-surface water interactions in arid/semi-arid floodplains. In 18th World IMACS Congress and MODSIM09 International Congress on Modelling and Simulation. Modelling and Simulation Society of Australia and New Zealand and International Association for Mathematics and Computers in Simulation (Vol. 3088, p. 3094).

Jones, P.L. and De Silva, S.S., 1997. Apparent nutrient digestibility of formulated diets by the Australian freshwater crayfish Cherax destructor Clark (Decapoda, Parastacidae). Aquaculture Research, 28, 881-891.

Jones, P.L. and De Silva, S.S., 1998. Comparison of internal and external markers in digestibility studies involving the Australian freshwater crayfish, Cherax destructor Clark (Decapoda, Parastacidae). Aquaculture research, 29, 487-493.

Jones, P.L., Chavez, J.R. and Mitchell, B.D., 2002. Production of Australian freshwater crayfish in earthen-based systems using pelleted diets and forage crops as food. Aquaculture international, 10, 157-175.

Kahan, G., Colloff, M. and Pittock, J., 2020. Using an ecosystem services approach to re-frame the management of flow constraints in a major regulated river basin. Australasian Journal of Water Resources, pp.1-12.

Karim, F., Marvanek, S., Merrin, L.E., Nielsen, D., Hughes, J., Stratford, D. and Pollino, C., 2020. Modelling Flood-Induced Wetland Connectivity and Impacts of Climate Change and Dam. Water, 12, 1278.

Kattel, G., Gell, P., Perga, M.E., Jeppesen, E., Grundell, R., Weller, S., Zawadzki, A. and Barry, L., 2015. Tracking a century of change in trophic structure and dynamics in a floodplain wetland: integrating palaeoecological and palaeoisotopic evidence. Freshwater Biology, 60, 711-723.

Kattel, G.R., Eyre, B.D. and Gell, P.A., 2020. Integration of palaeo-and-modern food webs reveal slow changes in a river floodplain wetland ecosystem. Scientific Reports, 10, 1-13.

Kelleway, J., Mazumder, D., Wilson, G.G., Saintilan, N., Knowles, L., Iles, J. and Kobayashi, T., 2010. Trophic structure of benthic resources and consumers varies across a regulated floodplain wetland. Marine and Freshwater Research, 61, 430-440.

Kennett, R. and Tory, O., 1996. Diet of two freshwater turtles, Chelodina rugosa and Elseya dentata (Testudines: Chelidae) from the wet-dry tropics of northern Australia. Copeia, 409-419.

King, A.J., 2005. Ontogenetic dietary shifts of fishes in an Australian floodplain river. Marine and Freshwater Research, 56, 215-225.

Kingsford, R.T., 2000. Ecological impacts of dams, water diversions and river management on floodplain wetlands in Australia. Austral Ecology 25, pp.109-127.

Kobayashi, T., Gibbs, P., Dixon, P.I. and Shiel, R.J., 1996. Grazing by a river zooplankton community: importance of microzooplankton. Marine and Freshwater Research, 47, 1025-1036.

Kobayashi, T., Ralph, T.J., Ryder, D.S. and Hunter, S.J., 2013. Gross primary productivity of phytoplankton and planktonic respiration in inland floodplain wetlands of southeast Australia: habitat-dependent patterns and regulating processes. Ecological research, 28, 833-843.

Kobayashi, T., Ralph, T.J., Ryder, D.S., Hunter, S.J., Shiel, R.J. and Segers, H., 2015. Spatial dissimilarities in plankton structure and function during flood pulses in a semi-arid floodplain wetland system. Hydrobiologia, 747, 19-31.

Koehn, J.D., Raymond, S.M., Stuart, I., Todd, C.R., Balcombe, S.R., Zampatti, B.P., Bamford, H., Ingram, B.A., Bice, C.M., Burndred, K. and Butler, G., 2020. A compendium of ecological knowledge for restoration of freshwater fishes in Australia's Murray–Darling Basin. Marine and Freshwater Research 71, 1391-1463.

Likens, G., Benbow, M., Burton, T., Van Donk, E., Downing, J. and Gulati, R., 2009. Encyclopedia of inland waters.

Lindholm, M., Hessen, D.O., Mosepele, K. and Wolski, P., 2007. Food webs and energy fluxes on a seasonal floodplain: the influence of flood size. Wetlands, 27, 775-784.

Loreau, M., Mouquet, N. and Holt, R.D., 2003. Meta-ecosystems: a theoretical framework for a spatial ecosystem ecology. Ecology Letters, 6, 673-679.

Manzoni, S., Taylor, P., Richter, A., Porporato, A. and Ågren, G.I., 2012. Environmental and stoichiometric controls on microbial carbon-use efficiency in soils. New Phytologist 196, 79-91.

Marchant, R., 1978. The energy balance of the Australian brine shrimp, Parartemia zietziana (Crustacea: Anostraca). Freshwater Biology, 8, 481-489.

Mazumder, D., Johansen, M., Saintilan, N., Iles, J., Kobayashi, T., Knowles, L. and Wen, L., 2012. Trophic shifts involving native and exotic fish during hydrologic recession in floodplain wetlands. Wetlands, 32, 267-275.

Mazumder, D., Saintilan, N., Wen, L., Kobayashi, T. and Rogers, K., 2017. Productivity influences trophic structure in a temporally forced aquatic ecosystem. Freshwater biology, 62, 1528-1538.

McInerney, P.J., Holt, G., Lester, R.E., Thompson, R.M., Robson, B., Ryder, D.S., Bond, N.R., Baldwin, D.S., Gawne, B. and Petrie, R., 2020. Basal resource quality and energy sources in three habitats of a lowland river ecosystem. Limnology and Oceanography, 65, 2757-2771.

Medeiros, E.S. and Arthington, A.H., 2008. The importance of zooplankton in the diets of three native fish species in floodplain waterholes of a dryland river, the Macintyre River, Australia. Hydrobiologia, 614, 19-31.

Medeiros, E.S. and Arthington, A.H., 2011. Allochthonous and autochthonous carbon sources for fish in floodplain lagoons of an Australian dryland river. Environmental Biology of Fishes, 90, 1-17.

Medeiros, E.S. and Arthington, A.H., 2014. Fish diet composition in floodplain lagoons of an Australian dryland river in relation to an extended dry period following flooding. Environmental biology of fishes, 97, 797-812.

Mehner, T., Lischke, B., Scharnweber, K., Attermeyer, K., Brothers, S., Gaedke, U., Hilt, S. and Brucet, S., 2018. Empirical correspondence between trophic transfer efficiency in freshwater food webs and the slope of their size spectra. Ecology 99, 1463-1472.

Mitrovic, S.M., Westhorpe, D.P., Kobayashi, T., Baldwin, D.S., Ryan, D. and Hitchcock, J.N., 2014. Short-term changes in zooplankton density and community structure in response to different sources of dissolved organic carbon in an unconstrained lowland river: evidence for food web support. Journal of Plankton Research, 36, 1488-1500.

Molinari, B., Stewart-Koster, B., Adame, M.F., Campbell, M.D., McGregor, G., Schulz, C., Malthus, T.J. and Bunn, S., 2021. Relationships between algal primary productivity and environmental variables in tropical floodplain wetlands. Inland Waters, 1-11.

Moy, K.G., Wilson, G.G. and Ellison, T.L., 2018. Life history and habitat preference in the Darling hardyhead, Craterocephalus amniculus (Teleostei, Atherinidae) in the northern Murray-Darling Basin, Australia. Austral Ecology, 43, 476-487.

Mulholland, P.J., Fellows, C.S., Tank, J.L., Grimm, N.B., Webster, J.R., Hamilton, S.K., Martí, E., Ashkenas, L., Bowden, W.B., Dodds, W.K. and McDowell, W.H., 2001. Inter-biome comparison of factors controlling stream metabolism. Freshwater biology, 46, 1503-1517.

Ocock, J.F., Brandis, K.J., Wolfenden, B.J., Jenkins, K.M. and Wassens, S., 2019. Gut content and stable isotope analysis of tadpoles in floodplain wetlands. Australian Journal of Zoology, 66, 261-271.

Oliver, R.L. and Merrick, C.J., 2006. Partitioning of river metabolism identifies phytoplankton as a major contributor in the regulated Murray River (Australia). Freshwater Biology, 51, 1131-1148.

Opperman, J.J., Luster, R., McKenney, B.A., Roberts, M. and Meadows, A.W., 2010. Ecologically functional floodplains: connectivity, flow regime, and scale 1. JAWRA Journal of the American Water Resources Association 46, pp.211-226.

Overton, I.C., McEwan, K. and Sherrah, J.R., 2006. The River Murray Floodplain Inundation Model – Hume Dam to Lower Lakes. CSIRO Water for a Healthy Country Technical Report 2006. CSIRO: Canberra.

Paschke, K., Agüero, J., Gebauer, P., Díaz, F., Mascaró, M., López-Ripoll, E., Re, D., Caamal-Monsreal, C., Tremblay, N., Pörtner, H.O. and Rosas, C., 2018. Comparison of aerobic scope for metabolic activity in aquatic ectotherms with temperature related metabolic stimulation: a novel approach for aerobic power budget. Frontiers in physiology 9, 1438.

Pettit, N.E., Naiman, R.J., Warfe, D.M., Jardine, T.D., Douglas, M.M., Bunn, S.E. and Davies, P.M., 2017. Productivity and connectivity in tropical riverscapes of northern Australia: ecological insights for management. Ecosystems 20, 492-514.

Petrov, K., Lewis, J., Malkiewicz, N., Van Dyke, J.U. and Spencer, R.J., 2018. Food abundance and diet variation in freshwater turtles from the mid-Murray River, Australia. Australian Journal of Zoology, 66, 67-76.

Petrov, K., Spencer, R.J., Malkiewicz, N., Lewis, J., Keitel, C. and Van Dyke, J.U., 2020. Prey-switching does not protect a generalist turtle from bioenergetic consequences when its preferred food is scarce. BMC ecology, 20, 1-12.

Piola, R.F., Suthers, I.M. and Rissik, D., 2008. Carbon and nitrogen stable isotope analysis indicates freshwater shrimp Paratya australiensis Kemp, 1917 (Atyidae) assimilate cyanobacterial accumulations. Hydrobiologia, 608, 121-132.

Pollino, C., Thompson, R. and Flett, D., 2020. Flow-MER Basin-scale Annual Report 2020. Flow-MER Program. Commonwealth Environmental Water Office (CEWO): Monitoring, Evaluation and Research Program, Department of Agriculture, Water and the Environment, Australia.

Power, M.E., 1992. Top-down and bottom-up forces in food webs: do plants have primacy. Ecology 73, 733-746.

Pusey, B.J., Jardine, T.D., Beesley, L.S., Kennard, M.J., Ho, T.W., Bunn, S.E. and Douglas, M.M., 2020. Carbon sources supporting Australia's most widely distributed freshwater fish, Nematalosa erebi (Günther)(Clupeidae: Dorosomatinae). Marine and Freshwater Research, 72, 299-299.

Rana, A., Kobayashi, T. and Ralph, T.J., 2021. Planktonic Metabolism and Microbial Carbon Substrate Utilization in Response to Inundation in Semi-Arid Floodplain Wetlands. Journal of Geophysical Research: Biogeosciences, 126, e2019JG005571.

Rees, G.N., Cook, R.A., Ning, N.S., McInerney, P.J., Petrie, R.T. and Nielsen, D.L., 2020. Managed floodplain inundation maintains ecological function in lowland rivers. Science of The Total Environment, 727, p.138469.

Reid, D.J., Quinn, G.P., Lake, P.S. and Reich, P., 2008. Terrestrial detritus supports the food webs in lowland intermittent streams of south-eastern Australia: a stable isotope study. Freshwater Biology, 53, 2036-2050.

Reid, M.A., Delong, M.D. and Thoms, M.C., 2012. The influence of hydrological connectivity on food web structure in floodplain lakes. River Research and Applications, 28, 827-844.

Robertson, A.I., Bunn, S.E., Boon, P.I. and Walker, K.F., 1999. Sources, sinks and transformations of organic carbon in Australian floodplain rivers. Marine and Freshwater Research 50, 813-829.

Robertson, A.I., Bacon, P. and Heagney, G., 2001. The responses of floodplain primary production to flood frequency and timing. Journal of Applied Ecology, 38, 126-136.

Robertson, A.I., Burns, A. and Hillman, T.J., 2016. Scale-dependent lateral exchanges of organic carbon in a dryland river during a high-flow experiment. Marine and Freshwater Research, 67, 1293-1301.

Rogers, K., 2011. Vegetation. In: Rogers, K. and Ralph, T, (eds), Floodplain Wetland Biota in the Murray–Darling Basin: Water and Habitat Requirements. CSIRO Publishing: Melbourne.

Rolls, R.J., Leigh, C. and Sheldon, F., 2012. Mechanistic effects of low-flow hydrology on riverine ecosystems: ecological principles and consequences of alteration. Freshwater Science, 31, 1163-1186.

Rowland, S.J., 2009. Review of aquaculture research and development of the Australian freshwater fish silver perch, Bidyanus bidyanus. Journal of the World Aquaculture Society, 40, 291-324.

Ryder, D.S., 2004. Response of epixylic biofilm metabolism to water level variability in a regulated floodplain river. Journal of the North American Benthological Society, 23, 214-223.

Saintilan, N., Kelleway, J.J., Mazumder, D., Kobayashi, T. and Wen, L., 2021. Incorporation of local dissolved organic carbon into floodplain aquatic ecosystems. Aquatic Ecology, 55, 779-790.

Schulze, D.J. and Walker, K.F., 1997. Riparian eucalypts and willows and their significance for aquatic invertebrates in the River Murray, South Australia. Regulated Rivers: Research & Management: An International Journal Devoted to River Research and Management, 13, 557-577.

Sheldon, F. and Walker, K.F., 1997. Changes in biofilms induced by flow regulation could explain extinctions of aquatic snails in the lower River Murray, Australia. Hydrobiologia, 347, 97-108.

Silva, T.S., Melack, J.M. and Novo, E.M., 2013. Responses of aquatic macrophyte cover and productivity to flooding variability on the Amazon floodplain. Global change biology, 19, 3379-3389.

Sims N., Warren G., Overton I., Austin J., Gallant J., King D., Merrin L., Donohue R., McVicar T., Hodgen M., Penton D., Chen Y., Huang C. and Cuddy S. 2014. RiM-FIM floodplain inundation modelling for the Edward-Wakool, Lower Murrumbidgee and Lower Darling River systems. CSIRO publications. CSIRO, Clayton.

Spencer, R.J., Thompson, M.B. and Hume, I.D., 1998. The diet and digestive energetics of an Australian short-necked turtle, Emydura macquarii. Comparative Biochemistry and Physiology Part A: Molecular & Integrative Physiology 121, 341-349.

Spencer, R.J., Georges, A., Lim, D., Welsh, M. and Reid, A.M., 2014. The risk of inter-specific competition in Australian short-necked turtles. Ecological Research, 29, 767-777.

Sternberg, D., Balcombe, S., Marshall, J. and Lobegeiger, J., 2008. Food resource variability in an Australian dryland river: evidence from the diet of two generalist native fish species. Marine and Freshwater Research, 59, 137-144.

Taylor, I.R. and Schultz, M.C., 2008. Breeding season diets of egrets in southeast Australia: implications for future population viability. Waterbirds, 31, 593-601.

Tockner, K., Lorang, M.S. and Stanford, J.A., 2010. River flood plains are model ecosystems to test general hydrogeomorphic and ecological concepts. River research and applications, 26, 76-86.

Tonkin, Z.D., Humphries, P. and Pridmore, P.A., 2006. Ontogeny of feeding in two native and one alien fish species from the Murray-Darling Basin, Australia. Environmental Biology of Fishes, 76, 303-315.

Tucker, A.D., Guarino, F. and Priest, T.E., 2012. Where lakes were once rivers: contrasts of freshwater turtle diets in dams and rivers of southeastern Queensland. Chelonian Conservation and Biology, 11, 12-23.

Villarreal, H., 1991. A partial energy budget for the Australian crayfish Cherax tenuimanus. Journal of the World Aquaculture Society, 22, 252-259.

Vucic-Pestic, O., Ehnes, R.B., Rall, B.C. and Brose, U., 2011. Warming up the system: higher predator feeding rates but lower energetic efficiencies. Global Change Biology 17, 1301-1310.

Watkins, S.C., Nielsen, D., Quinn, G.P. and Gawne, B., 2011. The influence of leaf litter on zooplankton in floodplain wetlands: changes resulting from river regulation. Freshwater Biology, 56, 2432-2447.

Wedderburn, S.D., Bice, C.M. and Barnes, T.C., 2014. Prey selection and diet overlap of native golden perch and alien redfin perch under contrasting hydrological conditions. Australian Journal of Zoology, 62, 374-381.

Welsh, M.A., Doody, J.S. and Georges, A., 2017. Resource partitioning among five sympatric species of freshwater turtles from the wet−dry tropics of northern Australia. Wildlife Research, 44, 219-229.

Westhorpe, D.P., Mitrovic, S.M., Ryan, D. and Kobayashi, T., 2010. Limitation of lowland riverine bacterioplankton by dissolved organic carbon and inorganic nutrients. Hydrobiologia, 652, 101-117.

Wetzel, R.G., 2001. Limnology. Lake and River Ecosystems, third ed. Academic Press, SanDiego.

Winemiller, K.O., 2004. Floodplain river food webs: generalizations and implications for fisheries management. In Proceedings of the second international symposium on the management of large rivers for fisheries (Vol. 2, pp. 285-309). FAO Regional Office for Asia and the Pacific, Bangkok, Thailand.

Whiterod, N.S., Meredith, S.N., Humphries, P., Sherman, B.S., Koehn, J.D., Watts, R.J., Ingram, B.A. and Ryan, T., 2018. Flow alteration and thermal pollution depress modelled growth rates of an iconic riverine fish, the Murray cod Maccullochella peelii. Ecology of Freshwater Fish, 27, 686-698.

Whitworth, K.L. and Baldwin, D.S., 2016. Improving our capacity to manage hypoxic blackwater events in lowland rivers: the Blackwater risk assessment tool. Ecological Modelling 320, 292-298.

Zhang, X., Smernik, R., Doolette, A., Walters, S. and Mosley, L.M., 2021. Phosphorus speciation and release from different plant litters on a River Murray (Australia) floodplain. Plant and Soil, 1-16.

Appendices

APPENDIX 1: WORKING EXAMPLE OF MODEL WITH MODIFICATIONS

Here, we use the Koondrook-Perricoota Forest floodplain area (RiMFIM zone 7; River Murray system from Torrumbarry Weir to downstream of Gunbower) and associated historical discharge regime (measured at Torrumbarry Weir from 1975-2021) (Fig. A1.1) as a working example of model structure and the effects of potential modifications to the underlying energetics model.
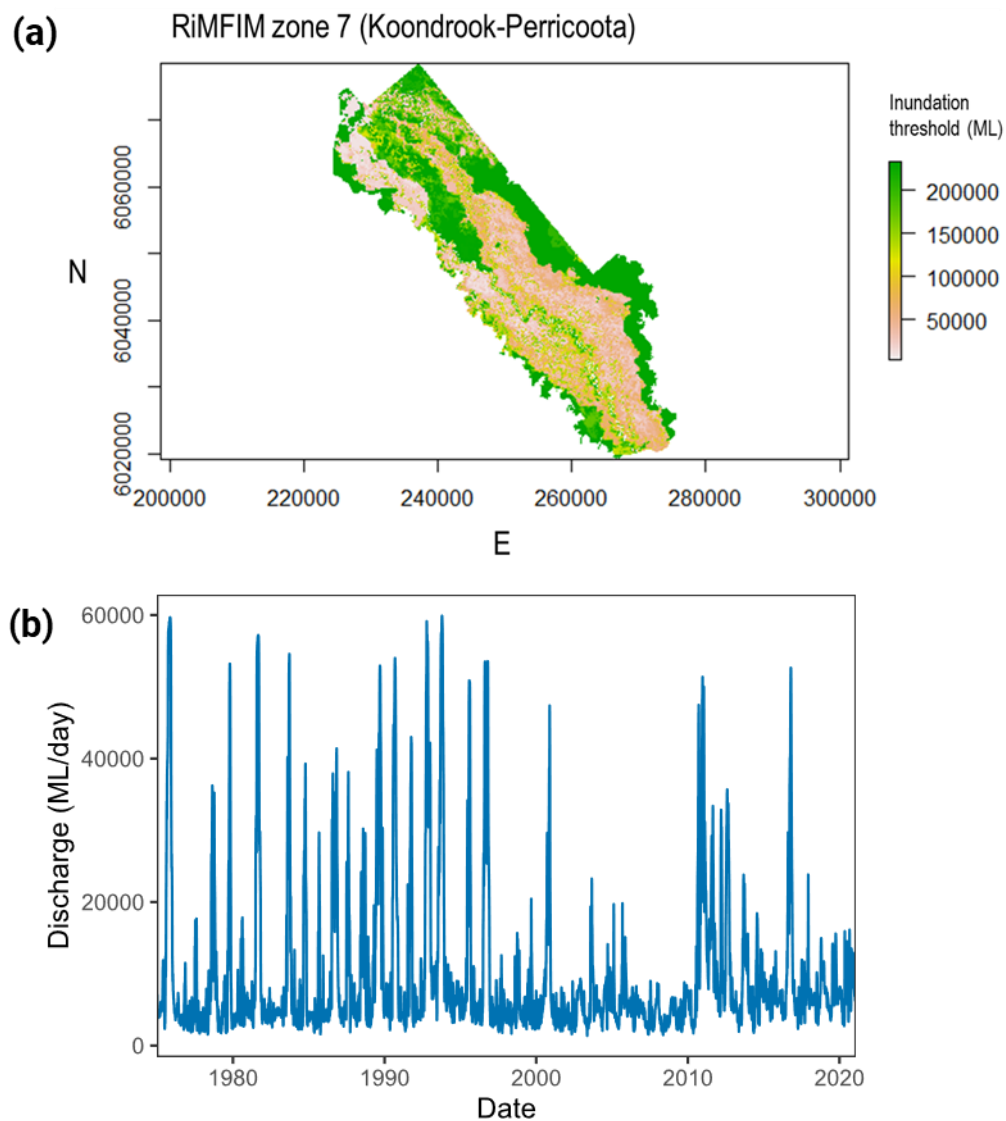

Figure A1.1 (a) Inundation threshold data for the Koondrook-Perricoota area (RiMFIM zone 7). (b) Historical (1975-2020) daily discharge measured at Torrumbarry Weir.

SECTION A1.1 MODELLING THE TEMPERATURE-DEPENDENCE OF PRODUCTION

Productivity of both heterotrophic and autotrophic organisms in aquatic ecosystems varies with water temperature (Wetzel, 2001). In addition, fundamental trophic relationships such as metabolic activity are strongly moderated by temperature (Brown et al., 2004). The model of Bond et al. (in prep) considers these factors constant. Consequently, we explored the potential for temperature-dependence to be credibly incorporated into the model structure through the influence of water temperature on (i) basal productivity rates and (ii) the efficiency of carbon transfer through food web links.

As starting model inputs, productivity rates were derived from literature values and standardised to consistent units (g C m^2 yr^{-1}). POM, benthic algae, benthic bacteria, phytoplankton, pelagic bacteria, and macrophyte productivity were derived from values measured by Gawne et al. (2006) at several sites along the River Murray (Appendix 3). DOC productivity was estimated as per Whitworth and Baldwin (2016), where the daily release of DOC is a first-order rate process proportionate to the mass of POM and soil per unit area. We used updated constants for maximum release per unit biomass and rate production of DOC from Silvester et al. (2020), assuming that the release of DOC is proportionate to POM production per day (where POM is approx. 0.4 g C per g dry mass) plus the release of DOC from soil (assuming leachable soil carbon of 1500 g/m^2) at the average water temperature (16.6 °C) during measurements by Gawne et al. (2006). The food web structure is that defined by Bond et al. (in prep) through a combination of literature review and expert opinion.

Modelling water temperature

Water temperature data is less consistently measured than discharge data across the MDB gauging network. Consequently, predicted temperature timeseries were created by DPIE from a boosted regression model of (i) historic flow measurements, (ii) historic temperature measurements, and (iii) historic climate models. We took these model timeseries and averaged temperature (in °C) to each month of the year (to broadly account for mean seasonal variation) and used these monthly averages as predicted values for all following temperature conversion factors.

Temperature dependence of productivity rates

Productivity rates should theoretically increase with temperature (Brown et al., 2004), but general scaling relationships are often complicated by co-limitation from light and nutrients (which can also vary greatly in spatio-temporal availability across river reaches), as well as organism adaptations to specific temperature regimes (Wetzel, 2001). However, there is also a broad correlation between microbial production and respiration (Del Giorgio and Cole, 1998). A well-defined temperature dependence factor of 1.072 for ecosystem-level heterotrophic respiration (Grace and Imberger, 2006) was therefore used to estimate the dependence of heterotrophic productivity rates on variation in temperature, such that:

$$productivity_{temperature-modified} = productivity_{base} * (1.072^{(temp_i - temp_{avg})})$$

Where $temp_i$ is the mean modelled water temperate for month i, and $temp_{avg}$ is the mean water temperature (16.7°C) during the base productivity rate measurements made by Gawne et al. (2007). Autotrophic productivity rates were similarly modified, with the only exception to the equation above that of the temperature dependence factor (1.066) (Butcher and Covington, 1995). These temperature-dependence factors essentially assume that heterotrophic productivity doubles and autotrophic productivity increases by 90% for each 10°C increase in temperature (Wetzel, 2001; Demars et al,. 2011). The effect of including temperature-dependent

productivity rates on the model is to increase modelled rates from October-April and decrease rates from May-Sept (Fig. A1.2).
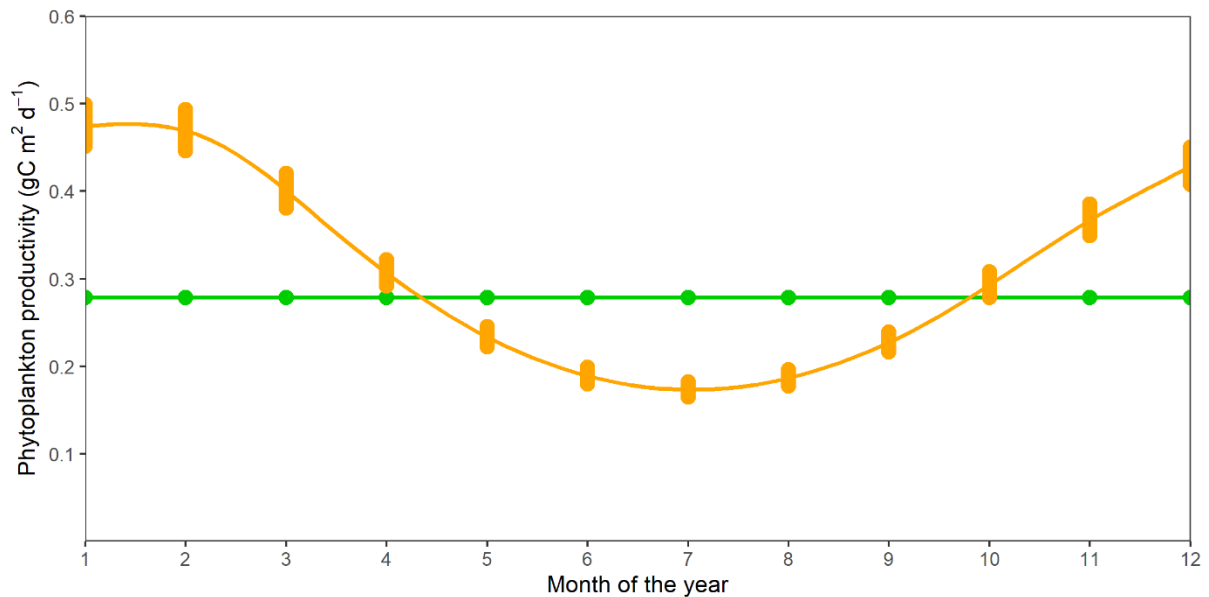

Figure A1.2 Example comparison of modelled productivity rates across months for phytoplankton (gC m^2 day^{-1}) in the base model (green) and model adjusted for temperature-dependence of productivity (orange). Temperature values used to modify productivity rates are fitted values derived (via GAM, see above) from historical water temperature measurements at Torrumbarry Weir (1993-2021).

When temperature-dependence is modelled into productivity rates, estimates of maximum potential C produced by large native fish followed approximately the same pattern as the base model (Fig. A1.3). However, estimates from the temperature-dependent model are sometimes noticeably higher or lower than those from the base model, with no consistent correlation with the magnitude of estimates. These differences reflect the timing of inundation events, as monthly differences in estimated C production between the temperature-dependent and base model are positive from October-April and negative from May-Sept (i.e., inundation events in summer are estimated to produce more C in the temperature-dependent than base model, and *vice versa* in winter) (Fig. A1.4). These patterns mirror general relationships between season and productivity which can be observed across the MDB (Pollino et al., 2020). As a result, we included temperature-dependent productivity rates into the model structure.

Figure A1.3 Comparison of predicted maximum C able to be produced by large native fish from the base model (blue) and model adjusted for temperature-dependence of basal productivity (orange). Model estimates are derived from spatial datasets on inundation from Koondrook-Perricoota (RiMFIM zone 7) and historical discharge timeseries from Torrumbarry Weir (1993-2020).

Figure A1.4 Difference in estimated maximum carbon produced by large native fish between the base model and model adjusted for temperature-dependence of productivity rates, by month, in the Koondrook/Perricoota region from 1993 to 2020. Dashed line indicates no difference between compared model predictions.

Temperature dependence of ecological efficiencies

The ecological efficiency of a trophic link is defined as the proportion of biomass production at the lower node which contributes to biomass production of the consumer (Anderson et al., 2009). Numerous empirical and theoretical approaches have shown that ecological efficiencies average approximately 5-10% across most food

webs (Jennings et al., 2002), which suggests that a consistent scaling factor of temperature could be applied to ecological efficiency values. However, ecological efficiencies integrate a number of potential factors on which temperature can, and is known to, act:

1. Feeding efficiency, or the proportion of basal production that is ingested by a consumer (a.k.a. exploitation efficiency). Feeding efficiency can be highly temperature-dependent via metabolic constraints on movement, which further incorporates the availability of energy (for ectotherms) counterbalanced with changes in respiration rates;

2. Assimilation efficiency, or the proportion of ingested production which is retained within consumers i.e., not excreted. Assimilation efficiency can be temperature-dependent through the sensitivity of enzyme-catalysed reactions; and,

3. Production efficiency, or the proportion of assimilated production which is utilised for biomass production rather than respiration. Production efficiency can be temperature-dependent through both respiration rates and the rate of enzyme-catalysed reactions.

The problem herein is that temperature has different effects on different efficiencies; simply, an increase in temperature increases the activity of processes that increase production (feeding, digestion) but concurrently increases those which consume energy (activity rates, respiration). A number of studies have investigated the effects of temperature on these efficiency values in isolation, but individual efficiencies are complicated by interactions with predator/prey body mass ratios (Mehner et al., 2018), resource/consumer nutrient stoichiometry (Manzoni et al., 2012), and variation in growth rate and metabolism among different taxa at the same trophic level (Ernest et al., 2003). With respect to this last point, very few studies have been conducted on taxa that exist in the MDB (Appendix 2), and even the most comprehensive are limited in their scope. For example, Spencer et al. (1998) define the temperature-dependence of feeding rates and assimilation efficiency for a single taxa, Macquarie turtles, but not production efficiency or feeding efficiency. Consequently, attempting to assign temperature-dependent ecological efficiencies for individual links within the model would likely have to be based on unsupported assumptions.

As a starting point, therefore, we applied a general temperature-dependent relationship to transfer efficiencies based on the metabolic theory of ecology (Brown et a., 2004), and particularly on the scaling rate of "overall" organism metabolism with temperature. Taking a similar approach to the temperature-dependence of basal productivity, therefore, we defined temperature-dependent ecological/trophic transfer efficiencies (TTE) as:

$$TTE_{temperature-modified} = TTE_{base} - (TTE_{base} * (1.087^{(temp_i - temp_{max})})) + (0.2 * TTE_{base})$$

Where $temp_i$ is the mean modelled water temperate for month i, and $temp_{max}$ is the theoretical maximum temperature (40°C) at which energy and respiratory demands outweigh feeding rates in ectothermic metabolism (Paschke et al., 2018). The scaling factor assumes that metabolic rates increase by approx. 2.3 for every 10°C increase in temperature (Dell et al., 2011), i.e. that metabolic activity scales allometrically with temperature (Brown et al., 2004), and that the overall effect of increasing metabolism is to exponentially decrease ecological efficiency towards a minimum value of ecological efficiency (20 % of max efficiency) at maximum temperature tolerance (Vucic-Pestic et al., 2011).

When temperature-dependence is modelled into both productivity rates and ecological efficiency, estimates of maximum potential C produced by large native fish were approximately the same as those from the base model (Fig. A1.5). Once again, monthly differences in estimated C production between the temperature-dependent and base model were positive from October-April and negative from May-Sept (i.e., inundation events in summer are estimated to produce more C in the temperature-dependent than base model, and *vice versa* in winter) (Fig. Fig A1.6).
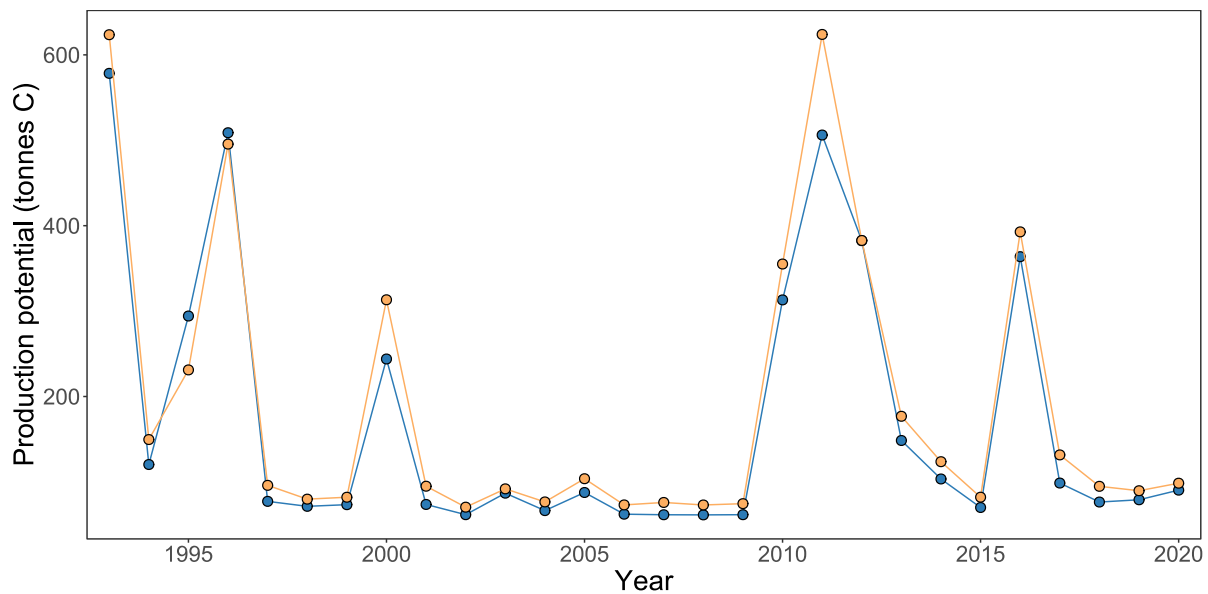
Figure A1.5 Comparison of predicted maximum C able to be produced by large native fish from the base model (blue) and model adjusted for temperature-dependence of both basal productivity and trophic transfer efficiencies (dark orange). Model estimates are derived from spatial datasets on inundation from Koondrook-Perricoota (RiMFIM zone 7) and historical discharge timeseries from Torrumbarry Weir (1993-2020).
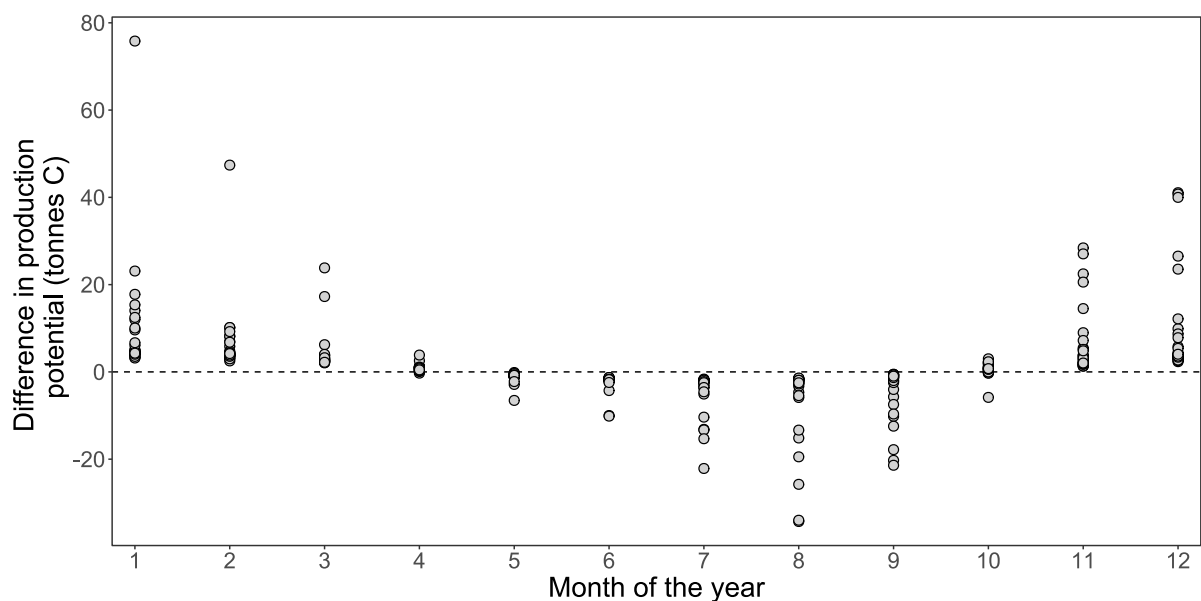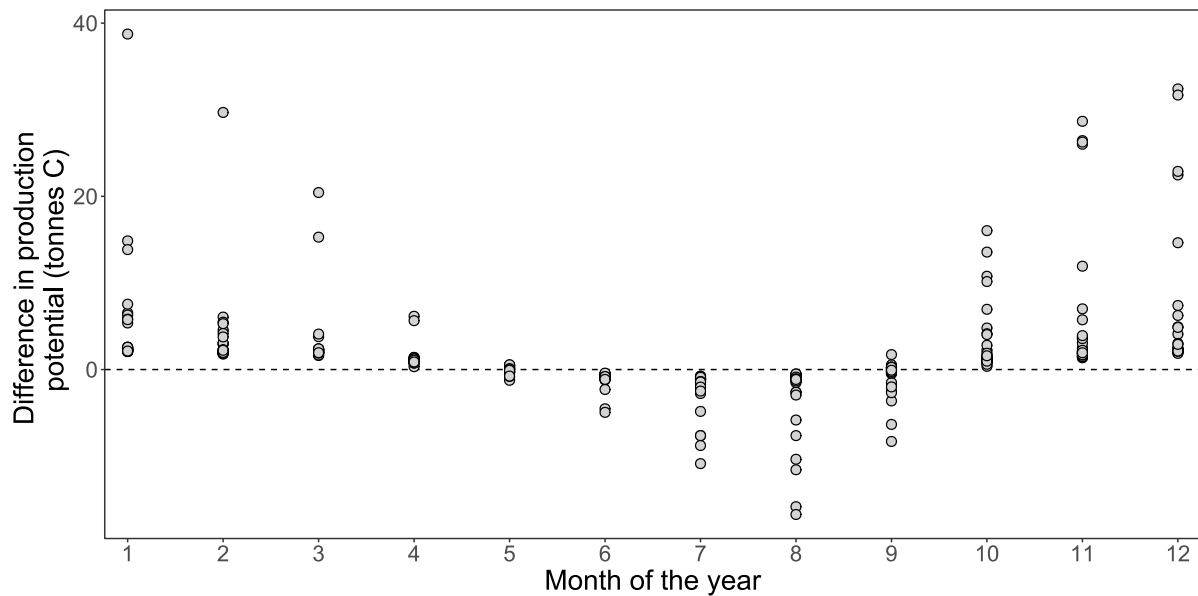

Figure A1.6 Difference in estimated maximum carbon produced by large native fish between base model and model adjusted for temperature-dependence of productivity rates, by month, in the Koondrook/Perricoota region from 1993 to 2020. Dashed line indicates no difference between compared model predictions.

Interestingly, although the seasonal aspect of the model with temperature-dependence of both productivity and ecological efficiency was dampened over including temperature-dependence of productivity alone, the difference between models with carp included or excluded in food-web structures is substantial (Fig. A1.7). In the latter case, the exclusion of carp resulted in much higher estimates of potential fish production than in the base model without temperature-dependence.

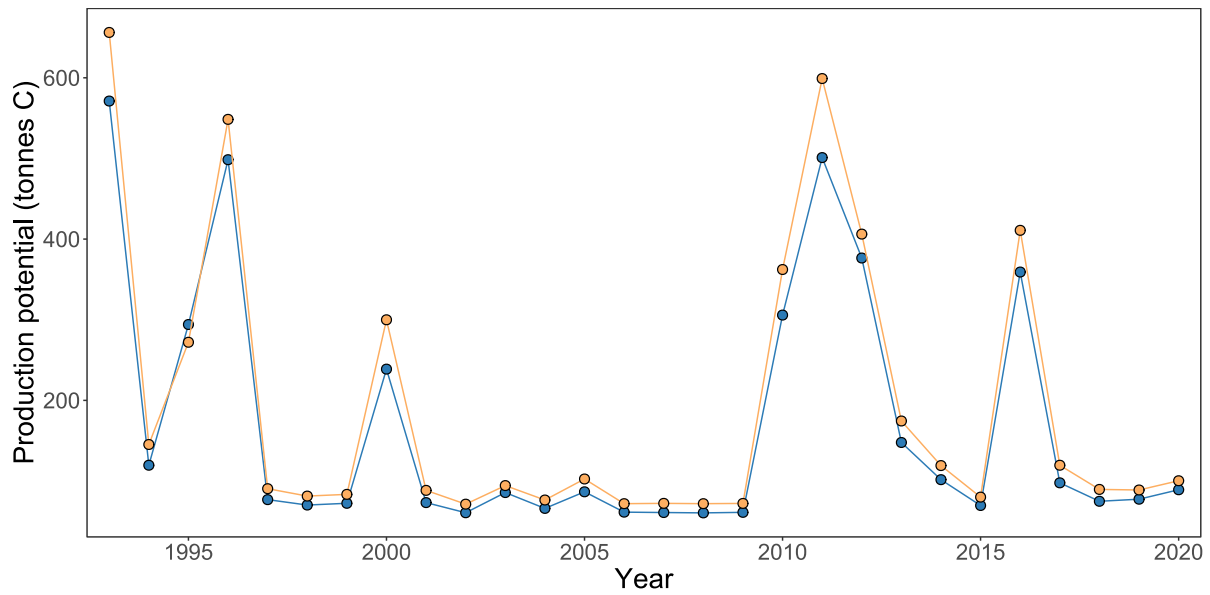**Figure A1.7** Comparison of predicted maximum C able to be produced by large native fish from the model with temperature-dependent productivity rates and ecological efficiencies (blue) and the same model but with carp excluded from the food web structure (green). Model estimates are derived from spatial datasets on inundation from Koondrook-Perricoota (RiMFIM zone 7) and historical discharge timeseries from Torrumbarry Weir (1993-2020).
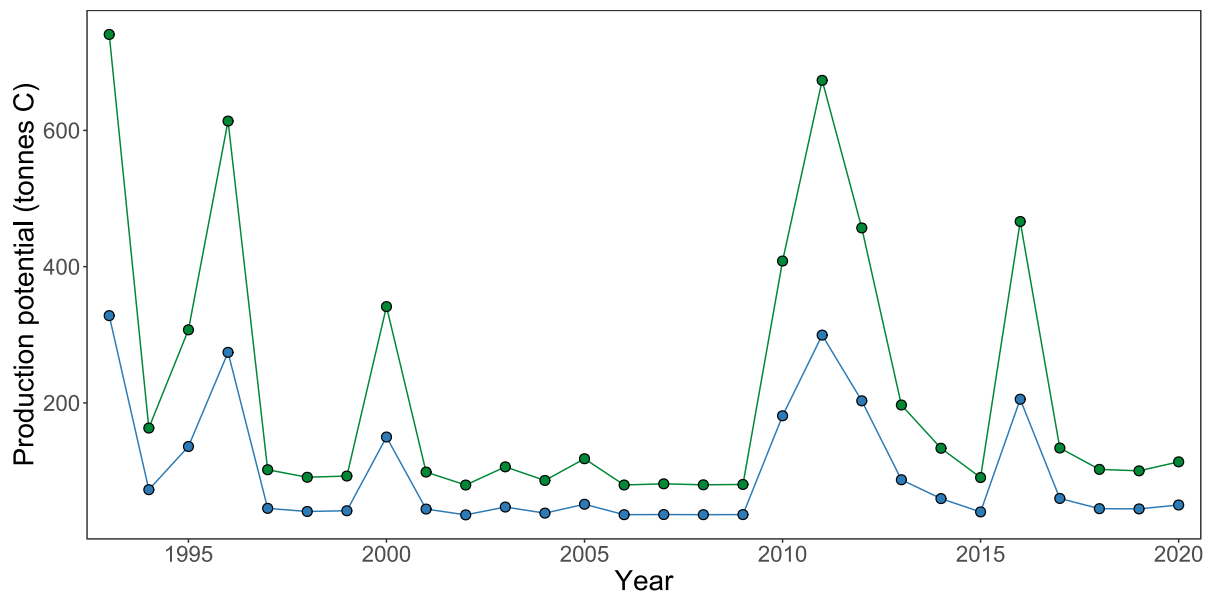
Although the model with temperature-dependence of both productivity rates and ecological efficiencies emphasises a negative correlation between presence of carp and large native fish biomass, we believe that inclusion of temperature-dependent ecological efficiencies as modelled here may not be warranted for several reasons. First, (i) the pronounced effect of carp-exclusion on the model may match field abundance or biomass data, but carp likely contribute to reduced native fish biomass strongly through environmental degradation (both resilience to and creation of; Driver et al., 2005) as well as the effect on resource distribution through food webs modelled here. The direct effect of temperature on ecological efficiency is thus likely overestimated in our model. Second, (ii) the seasonal variation in production associated with timing of inundation events is dampened, which does not correlate with field observations (e.g. Beesley et al., 2011); and third (iii), a general lack of empirical data on ecological efficiencies in the MDB hampers our ability to mechanistically model temperature-dependence. Consequently, we recommend that this modification not be incorporated until empirical data on trophic transfer efficiencies is widely available for MDB taxa.

SECTION A1.2 MODELLING HABITAT SPECIFICITY IN PRODUCTIVITY AND FOOD WEB STRUCTURE

In general, production in permanent floodplain wetlands can be an order of magnitude higher than that of river channels or floodplains (Likens et al., 2009). While much of this additional production can be attributed to high densities of aquatic macrophytes not likely to be directly utilised by aquatic food webs, macrophytes can potentially contribute indirectly as substrate for algal production (e.g. Adame et al., 2017; Molinari et al., 2021) and as a source of organic detritus for bacterial production (e.g. Bunn and Boon, 1993). Floodplain wetlands may also exhibit different relative proportions and identity of links between food web nodes to river channels (Winemiller, 2004). As with temperature, we therefore explored the potential for habitat-specific (i) productivity rates and (ii) food web structures to be credibly incorporated into the model.

Identifying habitat divisions

We used the Australian National Aquatic Ecosystem (ANAE) classification (AETG, 2012) as a starting point to sub-divide inundation spatial layers into specific habitats. The ANAE includes 45 freshwater ecosystem and floodplain classes which incorporate both lotic and lentic habitats along a range of flow permanence. However, a lack of empirical data on MDB floodplain productivity rates in general, and particularly across different floodplain or wetland classes, constrains our ability to assign plausible rates to all ANAE classes (Appendix 3). Empirical data in other locations (e.g. large Australian rivers in the wet-dry tropics) is more readily available but still too scarce to assign differences across ANAE-specific habitats. Nontheless, these studies have suggested macrophyte presence/absence and growth form as the most important controls on productivity rates within floodplain habitats (Adame et al., 2017; Molinari et al., 2021). In the MDB, these attributes can be broadly ascribed to the frequency of inundation, with permanent floodplain wetlands associated with aquatic (submerged and floating) macrophytes, temporary wetlands associated with rushes and reeds, and the more rarely inundated areas of the floodplain generally lacking aquatic or emergent macrophytes (Rogers, 2011). The ANAE also broadly classifies off-channel habitats into these groups through persistence of water (permanent vs. temporary vs. other).

Consequently, we created simplified habitat maps for the river and floodplain areas included in the project. The map took existing datasets (ANAE v3, 2021; and GeoScience Australia (GA) surface hydrology data, 2021) and grouped them into four categories:

- Permanent streams/rivers
- Permanent off-channel waterbodies
- Temporary waterbodies
- All other floodplain

The base dataset was the ANAE data; the attribute table with descriptors was used to categorise all polygons in ANAE into one of the above four categories. As the ANAE does not cover the whole project area, the GA HydroPolys dataset was used to fill in gaps under a similar classification: both datasets combined provided enough information to categorise the majority of the floodplain into the above categories; e.g. notes on whether they are temporary or permanent waterbodies, and whether they are on the mainstream of the Murray or off-channel. The combined dataset thus covered most of the project area. However, there were some areas on the high floodplain (i.e., inundation values too high to be regularly inundated) that were not covered. These areas were given a blanket classification of 'All other floodplain'. For the HydroPolys dataset there were also some reservoirs that had to be individually classified, as they were part of the stream or other inconsistencies (Table A1.1).

Table A1.1 Waterbodies which were individually classified or re-classified from the HydroPolys dataset.

HABITAT CLASSIFICATION	HYDROPOLYS CLASSIFICATION OR WATERBODY NAME
Permanent stream/river	Lake Hume
	Berembed weir
	Gogeldrie weir
	Lake Talbot
	Lake Burrinjuck
	Maude weir
	Blowering reservoir
	Torrumbarry weir
	Lake Mulwala
	Watercourse; perennial
	Canal area
	Reservoir area
Permanent off-channel waterbodies	Lake Victoria
	Mungabarina lagoon
	Sheep Dip lagoon
	Lake; perennial
	Farm Dam Area
	Swamp
Temporary waterbodies	Croppers lagoon
	Matangry lagoon
	Cooks lagoon
	Deep lagoon
	Lake Wetherell
	Tombullen storage
	Browns tank
	Gum Creek tank
	Johnstons tank
	Dip tank
	Weirs tank

Finally, the spatial inundation datasets (i.e., RiMFIM) were clipped by the habitat datasets, generating separate inundation threshold data tables for areas associated with each habitat class to be used as inputs to the model workflow.

Assigning habitat-specific productivity rates

River channel productivity inputs were largely the same as those for the base model. However, three terms were modified: (i) DOC productivity was adjusted to remove the term associated with soil DOC release upon inundation; (ii) benthic algae productivity was adjusted to reflect the assumptions of Bond et al. (in prep) that only large woody debris provides surface area for benthic algae in river channels (1.2% of total area, with only

half available for photosynthesis); and (iii) benthic bacteria productivity rates were also adjusted to reflect a reliance on large woody debris as a substrate (1.2% of total area).

As above, empirical data on MDB floodplain productivity is scarce. However, some values could still be derived from MDB-specific studies (see: Appendix 3). POM productivity for wetlands and floodplain was estimated from the average of values reviewed by Robertson et al. (1999). DOC was derived as per the base model (per unit POM and soil release) using the updated POM rates for floodplains and intermittent wetlands, while for permanent wetlands we removed the soil term (as for river channels). Phytoplankton, pelagic bacteria, and macrophytes were assumed consistent with river channels (Gawne et al., 2007) for floodplain areas. Macrophyte productivity was derived from values measured by Robertson et al. (2001) for emergent macrophytes (intermittent wetlands) and Briggs and Maher (1985) for submerged and floating macrophytes (permanent wetlands).

For productivity rates which could not be extracted from literature on the MDB, we used data measured in Australian floodplain rivers from the wet-dry tropics as the closest possible analogue (see: Appendix 3). Benthic algal productivity for wetlands was estimated using values measured by Adame et al. (2017) and Molinari et al. (2021) for epiphytic algae inhabiting different macrophyte classes in northern Australian floodplain wetlands. Permanent wetland rates were estimated assuming 80% areal cover of submerged and floating macrophytes, plus 20% areal cover at base model algal productivity rates. Intermittent wetland rates were estimated assuming 40% areal cover of emergent macrophytes and grasses, plus 60% areal cover at base model algal productivity rates. Benthic bacteria rates were estimated assuming the scaling factor of bacterial to algal production in Gawne et al. (2007) remains constant (0.64). Phytoplankton productivity rates for both permanent and intermittent wetlands were averaged from the values of Burford et al. (2016) and Fagotter et al. (2012) measured in off-channel wetlands. Pelagic bacterial productivity for wetlands was estimated from the average of values reviewed by Robertson et al. (1999) for central Australian waterholes. We adjusted all productivity rates derived from northern Australian field studies by a factor of 0.5, i.e. assuming that productivity halves with the latitudinal distance from the wet-dry tropics to the MDB (Field et al., 1998). The overall result of these modifications is to increase variability in most productivity rates across habitats (Table A1.2).

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

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

Finally, we adjusted the time-lag dependence of productivity in permanent wetland habitats to account for their pre-wetted condition. We removed the time-lag dependence of algal and macrophyte production as an assumption that already wetted habitats do not require a "lead-up" time for productivity rates to reach their maximum, i.e. algal and macrophyte production are now calculated from 0-day lag inundation area*days values in permanent wetlands.

The overall effect of incorporating habitat-specific productivity on model estimates is to amplify estimates of large native fish production potential during larger flow events (Fig. A1.8).
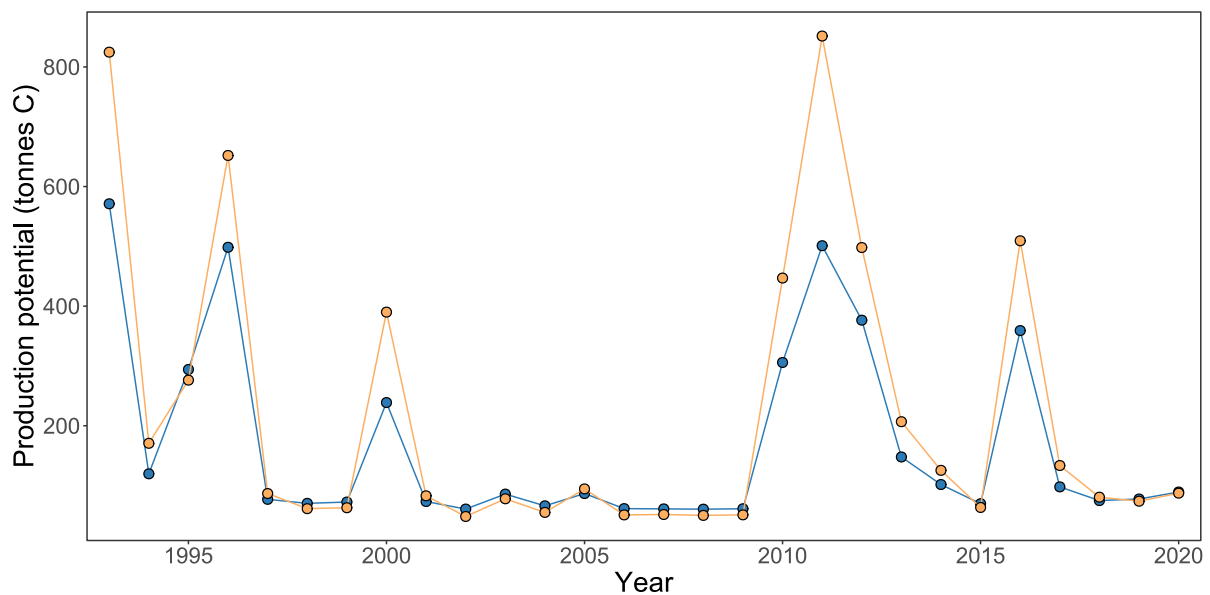
Figure A1.8 Comparison of predicted maximum C able to be produced by large native fish from the base model (blue) and model adjusted for temperature-dependence and habitat-specificity of basal productivity (dark orange). Model estimates are derived from spatial datasets on inundation from Koondrook-Perricoota (RiMFIM zone 7) and historical discharge timeseries from Torrumbarry Weir (1976-2020).

However, despite higher rates of algal and bacterial productivity, the contribution of both permanent and intermittent wetlands does not appear to be disproportionate to that of other floodplain areas (Fig. A1.9). This likely reflects the overall dominance of "other floodplain" area in Koondrook-Perricoota, where this habitat class makes up 92% of the RiMFIM raster as opposed to 3.9% for intermittent wetlands, 1.3% for permanent wetlands, and 3.2% for perennial river channels, and the slightly higher productivity rates assigned to "other floodplain" habitats over the base model (Table A1.1). When examining the range of discharges more likely to be considered under flow constraints options (< 30,000 ML/day), the difference between habitats in contributions to overall production potential is smaller (Fig. A1.10). However, floodplain production still contributes the majority of production potential at this lower range of inundation thresholds.
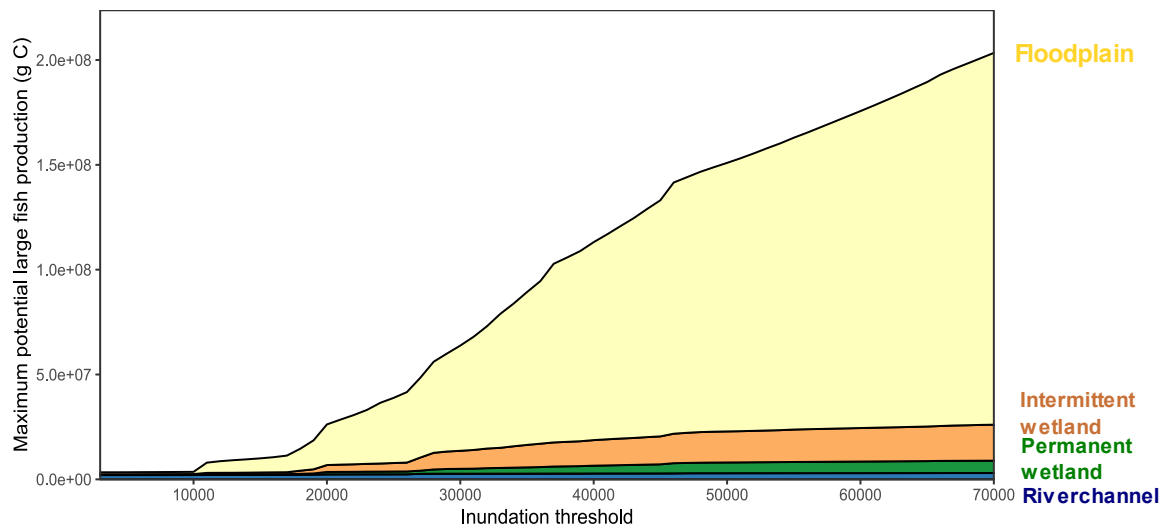
Figure A1.9 Comparison of maximum C able to be produced by large native fish along a gradient of discharge (i.e., inundation threshold reached) up to the maximum discharge reached historically from Torrumbarry Weir (1976-2020). Different colours show contribution of different habitats to overall production potential: river channel (blue), permanent wetlands (green), intermittent wetlands (peach), and floodplain (light yellow). Model estimates are derived from spatial datasets on inundation from Koondrook-Perricoota (RiMFIM zone 7).
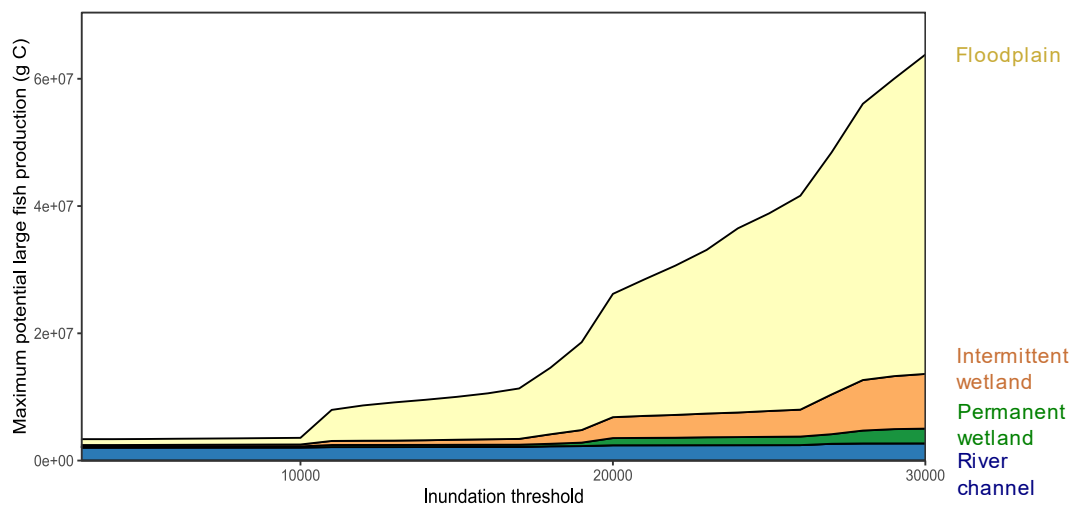
Figure A1.10 Comparison of maximum C able to be produced by large native fish along a gradient of discharge (i.e., inundation threshold reached) up to a proposed constraints limit of 30,000 ML/day. Different colours show contribution of different habitats to overall production potential: river channel (blue), permanent wetlands (green), intermittent wetlands (peach), and floodplain (light yellow). Model estimates are derived from spatial datasets on inundation from Koondrook-Perricoota (RiMFIM zone 7).

While the input productivity rates for permanent and intermittent wetlands are substantially higher in some areas (particularly bacterial and algal productivity) than those for other floodplain areas, they do not appear to disproportionately affect overall model estimates. Instead, the contributions of permanent and intermittent wetlands to overall production potential are only emphasised in situations where they would make up a

substantial proportion of inundated area (e.g., at lower discharges when only intermittent anabranches and low-lying floodplain areas are inundated), which correlates well with results from field studies on the productivity of large rivers and floodplain wetlands (Pettit et al., 2017). As a result, we included habitat-specific productivity rates into the model structure from this point forward.

Habitat-specific food webs

In all likelihood, the structure of food webs (i.e., the identity of nodes, and identity and strength of links) differs across in-channel and floodplain habitats (Winemiller, 2004). Yet, a lack of empirical data on MDB taxa again limits our ability to assign credible differences to food web structures across different habitats (see: Appendix 4). The "black box" structure of food webs inherited from Bond et al. (in prep), where broad functional and morphological groups rather than specific taxa are considered as food web nodes, also decreases our ability to modify food web structures. For example, many freshwater fish may be attributed to the "large native fish" category but can have very different habitat and feeding requirements, particularly in terms of their use of floodplain habitats (Koehn et al., 2020). Yet expanding the food web structure to include more nodes would, while increasing our ability to conceptualise differences between habitats in terms of the presence/absence of food web links, also significantly increase the errors associated with (i) ecological efficiencies (see above) and (ii) dominance structures for which little empirical data is available. Specific taxa may derive benefits from floodplain inundation periodically, seasonally, or at particular flow thresholds, but when (and for which taxa) these relationships apply is also still unclear (Graham and Harris, 2005; Koehn et al., 2020). Applying data from other large floodplain systems as an analogue (as for productivity rates) is problematic, as (i) the feeding habits of freshwater organisms from other regions can be very different to those of MDB taxa, and (ii) different taxa in general may be adapted to very different flooding regimes (Graham and Harris, 2005). As with the temperature-dependence of ecological efficiencies (see above), modifications to food web structures across different habitats may potentially have large effects which cannot be validated against empirical data. Consequently, we retained the base model food web structure with a single modification: the addition of a single, low dominance value link between benthic algae and reptiles (see: Appendix 4) based on the characterisation of this link within recent literature (Petrov et al. 2018, 2020).

APPENDIX 2: TROPHIC TRANSFER EFFICIENCIES REVIEW

We conducted literature searches to identify studies quantifying any aspect of trophic efficiencies (feeding, assimilation, and/or production efficiencies) between all nodes included in the modelled food web across all included habitat types (in-channel, wetland, and floodplain) in (i) the Murray-Darling Basin (MDB), and (ii) all other areas of Australia. For both (i) and (ii), we also included studies from in-channel waterholes and billabongs (typically, dry-season in the northern MDB or northern and central Australia), which may be considered equivalent to perennial and intermittent wetlands.

We chose to restrict our results to studies which were either conducted under field conditions or in mesocosms reproducing those same conditions, even though expanding our review to aquaculture studies would have resulted in a wider knowledge base to draw from (see e.g., Irvin et al. 2018). In particular, the assimilation and production efficiencies of decapods (both shrimp and crayfish) (e.g., Villareal 1991, Marchant 1978, Jones and De Silva 1997, 1998, Jones et al. 2002) and silver perch (see review in Rowland 2009) have been relatively well documented under differing diets and living conditions. However, animals raised in captivity and/or on a diet of commercial feed will likely have different trophic efficiencies to those living under natural conditions (Duffy et al. 2013). While these studies therefore provide a reference to the potential trophic efficiencies we might expect, they are likely not analogous to field conditions and are very specific to certain taxa popular in aquaculture (crayfish and silver perch in particular).

Our review therefore identified very few studies where trophic efficiencies were defined for Australian taxa under natural conditions (Table A2.1). Spencer et al. (1998) measured the assimilation efficiencies of short-necked turtles from the River Murray which were utilising different food sources (filamentous algae and carrion), while Kobayashi et al. (1996) estimated the feeding efficiency of zooplankton using measures of community biomass in the Hawkesbury-Nepean River. Consequently, there is very little Australian-specific information available to evaluate our model assumptions, particularly for the indicator taxa we use (large native fish). For example, no measured data on trophic efficiencies exists for Murray cod (Whiterod et al. 2018), despite its status as one of the most widely-studied native fish in the MDB (Koehn et al. 2020). Further, even studies which measure the overall metabolic demand of large native fish require the use of theoretical trophic efficiency values in their energetics models (e.g. Whiterod et al. 2018), and there are no whole-community estimates of trophic efficiencies available from comprehensive studies of multiple trophic levels (e.g., Mehner et al. 2018). We therefore conclude that there is no evidential basis for modification of our modelled trophic efficiencies at present.

Table A2.1 Australian studies quantifying trophic efficiencies (feeding, assimilation, and/or production efficiencies) for trophic links within the initially defined food web model structure here. Only studies conducted under natural or mesocosm (i.e., simulated natural) conditions are included (i.e., no aquaculture studies).

TAXA ASSESSED	EFFICIENCIES MEASURED	REFERENCE
Reptiles	Assimilation	Spencer et al. (1998)
Zooplankton	Feeding	Kobayashi et al. (1996)

APPENDIX 3: PRODUCTIVITY RATES REVIEW

We conducted literature searches to identify studies quantifying productivity rates (g C produced per unit area per unit time) for all basal resources included in the model (DOC, POM, benthic algae, benthic bacteria, phytoplankton, planktonic bacteria, macrophytes, and synonyms for all) across all included habitat types (in-channel, wetland, and floodplain) in (i) the Murray-Darling Basin, and (ii) all other areas of Australia. For (ii), we also included studies from in-channel waterholes and billabongs, which may be considered equivalent to perennial and intermittent wetlands. Studies only quantifying biomass, chlorophyll content, or abundance of organisms were not included, as translating these measurements into rate values requires conversion constants that are often not empirically validated or universal (e.g. Anderson and Tang, 2011). Finally, we searched for literature in all other parts of the world that quantified productivity rates for the floodplains of large river systems.

In the Murray-Darling Basin (MDB), direct measurements of productivity (via oxygen consumption/production or assimilation of tracers) have largely been conducted in in-channel habitats (Table A3.1). Of these studies, Gawne et al. (2007) provides the most comprehensive suite of measurements, where a number of different basal components were sampled seasonally (for 15 months) at multiple sites along the River Murray. Oliver & Merrick (2006) sampled similar sites for phytoplankton and benthic biofilm productivity, while Ryder (2004) conducted an experimental study of biofilm metabolism at different water levels in the Murrumbidgee River.

Many of the studies conducted in wetland or floodplain habitats of the MDB quantified productivity through changes in biomass (Table A3.1). The exceptions include Kobayashi et al. (2013, 2015) and Rana et al. (2021), but these studies were conducted in the Macquarie Marshes and may thus not be wholly transferable to the southern MDB rivers modelled here. Of the remaining studies, the review of Robertson et al. (1999) is the most widely applicable, combining (most frequently) biomass data from a number of studies across the MDB and Australia to estimate floodplain and floodplain wetland productivity rates for a number of basal components. However, these estimates are therefore also not specific to the southern MDB rivers and not generally direct measures of productivity *per se*. Macrophyte productivity rates are likely the most accurate of those derived from biomass measures, for which Briggs & Maher (1985) also separated out the contributions of floating and submerged macrophytes to overall macrophyte productivity.

Overall, Gawne et al. (2007) and Robertson et al. (1999) likely provide the most accurate estimates for model productivity rate parameters, as also concluded by Bond et al. (in prep). However, these studies also prove most applicable for in-channel and bulk floodplain areas rather than specifically for floodplain wetlands.

Table A3.1 Studies quantifying productivity rates for specific basal resources in the Murray-Darling Basin. Abbreviations: EM = ecosystem metabolism, POM = particulate organic matter, DOC = dissolved organic carbon.

HABITAT	TARGET AND APPROACH	REFERENCE
In-channel	Phytoplankton (light/dark bottles)	Gawne et al. (2007)
	Phytoplankton (light/dark bottles, EM)	Oliver & Merrick (2006)
	Phytoplankton (light/dark bottles)	Kobayashi et al. (2015)
	Bacterioplankton (thymidine)	Gawne et al. (2007)
	Benthic biofilms (chambers)	Gawne et al. (2007)
	Benthic biofilms (chambers)	Oliver & Merrick (2006)

HABITAT	TARGET AND APPROACH	REFERENCE
Wetland	Emergent macrophytes (biomass)	Gawne et al. (2007)
	POM (litter traps)	Gawne et al. (2007)
	Biofilms (chambers)	Ryder (2004)
	Benthic algae (biomass)	Robertson et al. (2001)
	Benthic bacteria (biomass)	Robertson et al. (2001)
	Phytoplankton (light/dark bottles)	Kobayashi et al. (2013)
	Floating macrophytes (biomass)	Briggs & Maher (1985)
	Submerged macrophytes (biomass)	Briggs & Maher (1985)
Floodplain	Benthic biofilms (biomass)	Robertson et al. (1999)
	Phytoplankton (light/dark bottles)	Kobayashi et al. (2013)
	Phytoplankton (light/dark bottles)	Kobayashi et al. (2015)
	Phytoplankton (light/dark bottles)	Rana et al. (2021)
	DOC (mass)	Robertson et al. (1999)
	Macrophytes (biomass)	Robertson et al. (1999)
	Macrophytes (biomass)	Robertson et al. (2001)
	POM (biomass)	Robertson et al. (1999)

Less productivity studies have been conducted in other areas of Australia than in the MDB specifically, although with a greater frequency of direct rate measurements than estimates from biomass changes (Table A3.2). In-channel measurements during flow have received lower attention, with only Burford et al. (2011) providing direct estimates of a specific basal component with their measurement of phytoplankton productivity in a tidal estuary of northern Australia. However, Fagotter et al. (2013) provide estimates of phytoplankton productivity from both in and off-channel waterholes during non-flowing phases. Studies from off-channel billabongs and wetlands of the northern wet/dry tropics are the most comprehensive when compared with those from the MDB. Adame et al. (2016) and Molinari et al. (2021) provide direct field measurements of productivity for epiphytic algae on different aquatic macrophytes of floodplain wetlands, building on earlier studies which estimated productivity hotspots from satellite imagery and algal biomass (see Pettit et al., 2017). Both studies use the highly accurate ^{13}C-bicarbonate method to complement more conventional dissolved oxygen-based approaches.

Bulk floodplain areas have been less specifically studied across Australia. Robertson et al. (1999) again provides generally applicable productivity rates, although with the prior caveat of the estimates generally being derived from biomass measurements. Burford et al. (2012, 2016) measured phytoplankton and benthic algal productivity with the ^{13}C-bicarbonate method, but again in tidal or estuarine environments which are likely not analogous enough to the southern MDB rivers for comparison. Given that (i) the major gaps in MDB-based

studies are found in estimates of benthic algae productivity for wetlands and floodplains, and for all components in wetlands more generally, and that (ii) more accurate methods have generally been used in these studies than in the MDB, the studies of northern Australian wetland and floodplains by Fagotter et al. (2013), Adame et al. (2016), Burford et al. (2016), and Molinari et al. (2021) are likely the most useful for transferring rate parameters across to the southern MDB.

Table A3.2 Studies quantifying productivity rates for specific basal resources in non-MDB areas of Australia. Abbreviations: EM = ecosystem metabolism, POM = particulate organic matter, DOC = dissolved organic carbon.

HABITAT	TARGET AND APPROACH	REFERENCE
In-channel	Phytoplankton (bicarbonate uptake)	Burford et al. (2011)
	Planktonic bacteria (biomass)	Robertson et al. (1999)
Waterhole/billabong	Phytoplankton (bicarbonate uptake)	Fagotter et al. (2013)
	Littoral algae (biomass, EM)	Robertson et al. (1999)
	Epipelic algae (biomass, EM)	Robertson et al. (1999)
Wetland	Epiphytic algae (bicarbonate uptake)	Adame et al. (2016)
	Epiphytic algae (bicarbonate uptake)	Molinari et al. (2021)
	Macrophytes (biomass, EM)	Molinari et al. (2021)
Floodplain	Phytoplankton (bicarbonate uptake)	Burford et al. (2012)
	Planktonic bacteria (biomass)	Robertson et al. (1999)
	POM (biomass)	Robertson et al. (1999)
	Benthic algae (bicarbonate uptake)	Burford et al. (2016)
	Biofilms (biomass)	Robertson et al. (1999)
	DOC (mass)	Robertson et al. (1999)
	Macrophytes (biomass)	Robertson et al. (1999)

Studies of floodplains overseas are biased towards ecological restoration efforts in temperate regions of North America and Europe, and almost overwhelmingly rely on estimates of primary productivity derived from macrophyte biomass (Gordon et al., 2020). Even where studies are conducted outside of these areas, for example on the Amazonian tropical floodplain (Silva et al., 2013), macrophyte biomass remains the estimation measure of choice. Bacterial production is largely limited to estimations of terrestrial POM decomposition

(Tockner et al., 2010). A notable exception to these studies is that of Lindholm et al. (2007), who measured phytoplankton productivity (via. ^{13}C-bicarbonate uptake) and anaerobic bacterial metabolism in floodplains of the semi-arid Okavango Delta, Botswana. However, Australian studies of phytoplankton productivity are both more numerous and indicate relatively similar rates (see Burford et al., 2016). In-channel measures of productivity are much more common than those for floodplains but are likely too site and climate-specific to directly utilise as parameters in MDB models (Mulholland et al., 2011). Overseas studies were therefore excluded from our model parameter estimates, especially considering the scarcity of studies overall (Table A3.3).

Table A3.3 Studies quantifying productivity rates for specific basal resources outside of Australia (floodplain-only).

HABITAT	TARGET AND APPROACH	REFERENCE
Floodplain	Phytoplankton (bicarbonate)	Lindholm et al. (2007)
	POM (decomposition constant)	Tockner et al. (2010)
	Macrophytes (biomass)	Silva et al. (2013)
	Macrophytes (biomass)	Gordon et al. (2013)

APPENDIX 4: REVIEW OF FOOD WEB STRUCTURE

We conducted literature searches to identify studies quantifying food web linkages (resources identified as contributing to the diets of consumers) between all nodes included in the modelled food web across all included habitat types (in-channel, wetland, and floodplain) in (i) the Murray-Darling Basin (MDB), and (ii) all other areas of Australia. For both (i) and (ii), we also included studies from in-channel waterholes and billabongs (typically, dry-season in the northern MDB or northern and central Australia), which may be considered equivalent to perennial and intermittent wetlands.

Most of the studies we identified in our literature review (65%) supported the structure of our initial modelled food web. Of these studies, the diets of small native fish, macroinvertebrates, and large native fish were the most frequently evaluated (table A4.1). Consumers in river channel and wetland habitats (both permanent and ephemeral) were evaluated to largely the same extent, although wetlands were studied slightly more often (likely reflecting our characterisation of waterholes and billabongs as "wetlands"). The diets of zooplankton, decapods, and carp are also relatively well characterised in river and wetland habitats across the MDB (table A4.1). Because of the large number of studies available in the MDB, we do not consider it necessary to catalogue the wider Australian literature for support of our initial food web structure (although we do discuss potential variations in this literature below).

Table A4.1 Studies quantifying food web linkages in the Murray-Darling Basin where trophic links (resources identified as contributing to the diets of consumers) supported the initially defined food web model structure here.

HABITAT	CONSUMER TAXA INVOLVED	REFERENCES
In-channel	Zooplankton	Westhorpe et al. (2010), Mitrovic et al. (2014)
	Macroinvertebrates	Schulze & Walker (1997), Sheldon & Walker (1997), Chester & Norris (2006), Reid et al. (2008), Hadwen et al. (2010), Hladyz et al. (2012), McInerney et al. (2020), Rees et al. (2020)
	Decapods	Giling et al. (2009), Mazumder et al. (2017), Saintilan et al. (2021)
	Small fish	Cadwallader et al. (1980), King (2005), Hardy et al. (2010), Hladyz et al. (2012), Mazumder et al. (2017), Moy et al. (2018), McInerney et al. (2020), Saintilan et al. (2021)
	Carp	Mazumder et al. (2017), Saintilan et al. (2021)
	Large native fish	Cadwallader & Eden (1979), King (2005), Ebner (2006), Baumgartner (2007), Mazumder et al. (2017), Wedderburn et al. (2014)
Wetland	Zooplankton	Medeiros & Arthington (2011), Watkins et al. (2011), Reid et al. (2012), Kattel et al. (2015)

HABITAT	CONSUMER TAXA INVOLVED	REFERENCES
Wetland	Macroinvertebrates	Bunn & Boon (1993), Reid et al. (2012), Kattel et al. (2015), McInerney et al. (2020)
	Decapods	Bunn & Boon (1993), Burns & Walker (2000), Kelleway et al. (2010)
	Small fish	Sternberg et al. (2008), Medeiros & Arthington (2008), Kelleway et al. (2010), Mazumder et al. (2012), Medeiros & Arthington (2011), Reid et al. (2012), Growns et al. (2020a), Kattel et al. (2020), McInerney et al. (2020)
	Carp	Kelleway et al. (2010), Mazumder et al. (2012), Reid et al. (2012)
	Large native fish	Sternberg et al. (2008), Growns et al. (2020a)

Several imbalances in the literature constrain our ability to evaluate our assumptions about food web structure, however. The bias towards fish is reflected in the underrepresentation of mussels, benthic microinvertebrates, mammals, and birds in dietary studies of the MDB. Reptile diets have been studied more frequently, although these do not fully support our initial food web model (see discussion below). Studies of trophic dynamics in wetlands were also largely conducted in dryland river waterholes, reflecting one of the earliest and most productive research environments in the Australian food web literature (see e.g., Bunn et al. 2006), while waterholes or billabongs in the southern basin are less highly represented. The results of these studies may thus not be wholly applicable to the areas modelled here. Conversely, we did not include studies of estuarine environments of the River Murray, the trophic dynamics of which have been widely studied (e.g., Deegan et al. 2008), because we considered these environments to be too dissimilar to the freshwater southern basin rivers and wetlands. These studies may provide clues to variation in food web trophic links across different environments if evaluated further. Finally, we could not find studies that specifically examined food web structure or trophic links on inundated floodplains, as opposed to floodplain wetlands. Floodplain food webs therefore represent a largely unexplored area of research and one in which we cannot find information to challenge many of our assumptions.

Our literature review also identified several studies (19) which we considered might challenge the structure of our initially modelled food web (table A4.2). These exceptions to the food web structure defined by Bond et al. (in prep) are discussed below:

i. Ocock et al. (2019) analysed the dietary composition of tadpoles in floodplain wetlands of the Murrumbidgee River, and found that benthic biofilms, algae, macrophytes, and benthic microcrustaceans were all consumed. Tadpoles may also represent a food resource for fish. The initial food web model of Bond et al. (in prep) did not list amphibians as a node, likely because they largely do not feed in the aquatic environment as adults and all of the other nodes are assumed to be adults (see also (vi), below). Yet tadpoles may be a limited resource in both space (probably only found in wetlands, rather than riverine habitat) and time (pulsed recruitment events, with adults emerging from the aquatic environment). We concluded that there is currently not enough evidence to assign other food web properties (especially dominance values) to these potential links, notwithstanding the current limitation of the modelled food web to adult consumer stages.

ii.   Taylor and Schultz (2008), in a study of the breeding season diets of Eastern great egrets, found that the diet of these birds consisted largely of carp with a small occurrence of amphibians (adult frogs). However, the carp were all juvenile (< 1 year old) and therefore do not conform to the assumption of our modelled web that all consumers and sources are adult. Similarly, the amphibians consumed were likely not fully aquatic in their feeding habits (see (i) above) and thus do not integrate fully into an "aquatic" food web.

iii.   A striking example is that of predatory Dystiscid diving beetles, which may often include fish in their diets. Although this usually occurs through consumption of carrion, there is evidence that diving beetles may actively predate on vertebrates. For example, Ebner et al. (2021) provide anecdotal evidence of packs of diving beetles attacking and consuming relatively large fish in fyke nets within waterholes of northern Queensland (Table A4.2. Although there is no published evidence of this behaviour in wetlands of the southern MDB, the possibility that this feeding behaviour can occur might potentially be high given that Dytiscidae have been found across all MDB river systems. However, adding a link for macroinvertebrates feeding on fish to the modelled foodweb is likely unjustified at this stage: besides the aforementioned lack of evidence of this behaviour occurring in the southern MDB, (i) Dystiscidae only represent a very small proportion of macroinvertebrate taxa overall; (ii) Dytiscidae are specialised to certain habitats (backwaters and permanent wetlands with abundant floating macrophyte cover) that are not widespread in the southern MDB; and (iii) other pathways by which higher trophic levels may provide energy to lower levels, particularly through the consumption or decomposition of dead animals or excreta (see also e.g. Burford et al. 2008), are currently assumed within the food web structure through the use of ecological efficiency values derived from mass balance rather than measured energetics (as above; see also: Appendix 2).

iv.   Petrov et al. (2018, 2020) found that Murray River short-necked turtles in oxbow lakes of northern Victoria can obtain a large proportion of their diets from filamentous algae (Table A4.2). However, these diets were highly taxon-specific, with other turtles (*Chelodina* spp.) being almost entirely carnivorous. The strength of the link (i.e., dominance values) is thus likely low overall. These results are also backed up by the study of freshwater turtle diets by Tucker et al. (2012) in south-eastern Queensland, which provides further documentation of the algae → turtles link in riverine habitats, as do Spencer et al. (2014) in waterholes of the northern MDB and Armstrong and Booth (2005) in the Burnett River, QLD. Tucker et al. (2012) and Amrstrong and Booth (2005) also document feeding by turtles on terrestrial POM (fruits) and aquatic macrophytes, as do Kennett & Tory (1996) in the wet-dry tropics, although these behaviours are species-specific and may also shift in time as turtles exhibit facultative omnivory (e.g. Welsh et al., 2017). We suggest that more evidence from Murray-Darling species is required to back up inclusion of all of these links in a modelled food web.

v.   Chessman (1984) documented high abundance of decapods in the diet of snake-necked turtles within the River Murray, and specifically within river channels. Turtles within wetlands appeared to feed on macroinvertebrates instead, which is the only documented case we could find of differing food web links between habitats in the MDB. Decapods made up a small proportion of stomach contents by volume, however, and we could not find evidence of other turtle species exhibiting these feeding behaviours. Turtles also exhibited a high proportion of carrion in their diets, which contradicts the assumption that dead organic matter links are not included in our modelled food web (see (iii) above).

vi.   King (2005) documented a high abundance of zooplankton in the stomach contents of larval to juvenile Murray cod. A number of subsequent studies have also emphasised the role of zooplankton in the development of large native fish (see Koehn et al. 2020). However, zooplankton likely do not contribute greatly (if at all) to the diet of large adult fish. Given the assumption implicit in the current food web of adult stages at each consumer link (see (i), above), addition of this link may not be justified.

vii.   Tonkin et al. (2006) documented a link between zooplankton and juvenile carp in experimental mesocosms. However, this finding comes under the same consideration as (vi) above where we cannot confirm that this behaviour is exhibited in adults.

viii.   Balcombe and Humphries (2006) documented the gut contents of Carp gudgeon in a billabong along the River Murray and found that shrimp were occasionally consumed by the fish. However, (i) this occurred in only three of the 24 subjects examined; (ii) shrimp were only consumed by the largest specimens; and (ii) shrimp did not make up a large proportion of the stomach contents when they were found. A link between decapods and small fish may thus not be justified based on this single study.

ix.   Pusey et al. (2020) comprehensively surveyed studies of the diets of Bony bream across Australia (including Medeiros & Arthington 2014) and consistently identified detritus (particularly terrestrial plant detritus) as the major component of stomach contents. However, while there is no direct POM to small fish link in our modelled food web, Pusey et al. (2020) acknowledge that fungi and bacteria on POM likely contribute more to the assimilated diets of Bony bream. As fungi to small fish is already included as a link in our modelled food web, we therefore see no reason to add a link to POM as well.

x.   Piola et al. (2008) documented a link between cyanobacterial accumulations and shrimp diets using stable isotope analysis of food webs in Myall Lake, NSW. This link may also occur in the MDB, and River Murray in particular, due to the occurrence of cyanobacterial blooms in the weir pools and lakes of the system. However, it is yet to be documented in the Basin, and may only be restricted to those times of year when blooms occur.

xi.   Deegan and Ganf (2008) conducted a stable isotope study of macroinvertebrate shredders and their food sources along the Finniss River in South Australia. Their results indicated that macroinvertebrates in unregulated reaches consumed semi-emergent aquatic macrophytes as a regular part of their diet. However, these results directly contrast with macroinvertebrates from regulated reaches, where terrestrial POM and filamentous algae formed the base of macroinvertebrate diets. These results are therefore not likely to be generally transferable to the River Murray, although they do point to potential shifts in food web dynamics with greater deregulation of the system or expansion of the permanent anabranch and floodplain wetland network.

Table A4.2 Studies quantifying food web linkages in the Murray-Darling Basin, and across Australia in general, which documented either (i) food web linkages not initially included in the model structure here or (ii) food web linkages which were habitat-specific (i.e., occurring in one type of aquatic habitat but not another). The specific case (either linkage not defined or habitat-specificity) is highlighted with red text. Linkages are denoted by resource → consumer.

HABITAT	LINKAGE DEFINED	INCLUDED IN INITIAL WEB	HABITAT-SPECIFIC?	REFERENCES
In-channel	Zooplankton → Large fish	No	No	King (2005), others cited in Koehn et al. (2020)
	Macrophytes → Macroinvertebrates	No	No	Deegan & Ganf (2008)
	Benthic algae → Reptiles	No	No	Tucker et al. (2012)
	Macrophytes → Reptiles	No	No	Tucker et al. (2012)
		No	No	Armstrong & Booth (2005)
	POM → Reptiles	No	No	Kennett & Tory (1996)
		No	No	Tucker et al. (2012)
		No	No	Armstrong & Booth (2005)

HABITAT	LINKAGE DEFINED	INCLUDED IN INITIAL WEB	HABITAT-SPECIFIC?	REFERENCES
Wetland	Decapods → Reptiles	**No**	**Yes**	Chessman (1984)
	Pelagic bacteria → Decapods	**No**	No	Piola et al. (2008)
	Zooplankton → Carp	**No**	No	Tonkin et al. (2006)
	Benthic algae → Reptiles	**No**	No	Petrov et al. (2018)
		No	No	Petrov et al. (2020)
		No	No	Spencer et al. (2014)
	Large fish → Macroinvertebrates	**No**	No	Ebner et al. (2021)
	Benthic bacteria → Amphibians	**No**	No	Ocock et al. (2019)
	Benthic algae → Amphibians	**No**	No	Ocock et al. (2019)
	Macrophytes → Amphibians	**No**	No	Ocock et al. (2019)
Wetland	Benthic microinvertebrates → Amphibians	**No**	No	Ocock et al. (2019)
	Decapods → Small fish	**No**	No	Balcombe & Humphries (2006)
	POM → Small fish	**No**	No	Medeiros & Arthington (2014), Pusey et al. (2020)
	Carp → Birds	**No**	No	Taylor & Schultz (2008)
	Amphibians → Birds	**No**	No	Taylor & Schultz (2008)
	POM → Reptiles	**No**	No	Welsh et al. (2017)

Overall, therefore, food web studies in the MDB and across Australia emphasise a single fully justifiable change to the food web model of Bond et al. (in prep): (i) benthic algae to reptiles, with a low dominance value (1) indicating taxon-specificity. This change was thus incorporated into the model structure for this project. Changes to the dietary links of fish and amphibians are likely not justified given the broad scope and limits on the current modelled food web (particularly the assumption that all nodes are adult populations). However, we suggest that future model revisions should consider whether ontogenic shifts in diet should be taken into account, although this would likely require substantial expansion of the number of modelled nodes and links – particularly with respect to cross-ecosystem energy subsidies (i.e., from terrestrial to aquatic systems and *vice versa*). Another notable gap is that many of the studies documenting potential variations in food web structure

occurred in floodplain wetlands or billabongs: this points to potential differences in the diets of certain species between river and wetland habitats (Winemiller, 2004). However, none of the studies referenced here directly compared taxa across river channels and wetlands in the MDB (although c.f. Pusey et al., 2020, who do compare diets of Bony bream across habitats in northern Australia), and there is therefore no data to inform model assumptions in this respect.