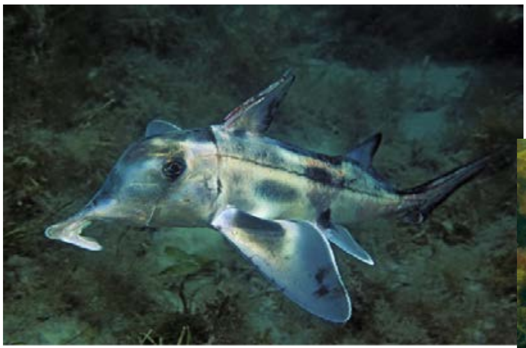
Understanding Western Port's past to manage its future:

Investigating the drivers of long-term change in key biological systems



A technical report prepared for Melbourne Water

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Executive Summary

Coastal and estuarine environments provide a range of valuable ecosystem services, including supporting commercial and recreational fisheries, nursery habitats for marine species, and filtering and detoxification services. These systems have, however, already been impacted by anthropogenic factors, which in conjunction with climate change will continue to threaten system viability. For many coastal and estuarine systems we currently lack the knowledge to link particular environmental drivers or events to observed biological changes. This understanding is essential if we are to sustainably manage coastal and estuarine environments and protect their valuable ecosystem services in a changing world.

Western Port is a large embayment in Victoria and supports recreational and commercial fisheries and a rich biodiversity. The bay is in relatively good condition, although we have a limited understanding of how its ecology responds to a range of local and regional environmental drivers. Developing such insight is of relevance to Melbourne Water, and other local natural resource managers, as it would help them manage the surrounding catchments, creeks and major drainage systems to promote Western Port's ecosystem health.

We proposed a two-phase project that will directly address the historical drivers of biological change across a range of key species in Western Port, and then use this insight to undertake scenario testing and prioritisation of management actions. This report represents the findings of Phase 1 where we undertook to investigate the drivers of long-term change in key Western Port fisheries to provide information with which to help manage the bay's future. First, we developed a series of conceptual models illustrating hypothesised links between environmental factors and the life history of three key fisheries species, pink snapper, King George whiting and elephant fish. Second, we collated a database of fisheries information, including commercial and recreational catch records, recruitment indices and two novel growth time series. Third, we used the conceptual models to identify potentially important local and regional environmental drivers. Fourth, we performed a series of sophisticated statistical analyses on the time series data to explore whether there are commonalities in biological variation through time across species, identify any change points present, and then relate common trends to environmental drivers.

We analysed 13 time series spanning the last 100 years in Western Port, including the development of two new growth chronologies. We successfully reduced this data into three common trends, identifying commonalities of biological responses across different species and life history metrics. Step changes (change points) in these trends were predominantly associated with El Niño and La Niña events, and to a lesser extent recruitment pulses and cessation of commercial netting.

The three common trends were associated with both local and regional drivers. On a local scale, nitrogen loads and Chlorophyll *a* concentrations affected fish through the food web and via

seagrass cover which provides essential habitat for juveniles. On a regional scale sea surface temperature in Bass Strait was important, especially in promoting catches of snapper and King George whiting.

Our analyses can be readily expanded to include biological data such as that from seabirds and other fishes to provide a more holistic view of the drivers of long term biological change in Western Port's marine environment, and the prioritisation of management actions (Phase 2).

Introduction

Project Background

Coastal and estuarine environments provide a range of valuable ecosystem services that support not only biodiversity, but also human food resources, commercial and recreation opportunities, filtering services and protection to coastlines (Barbier et al. 2011). These environments are, however, some of the most heavily utilised and impacted natural systems globally (Lotze et al. 2006, Halpern et al. 2008). Western Port and its estuaries have long supported valuable fisheries and a rich biodiversity. Whilst still in relatively good condition, over the last 100 years Western Port has experienced a number of significant natural and human-driven changes that have led to considerable shifts in highly valued social, economic and environmental assets such as fisheries, seabird populations and seagrass cover (Keough et al. 2011). Some of these environmental changes, such as the recent Millennium Drought (Dijk et al. 2013), can be measured on the scale of years, while others span multiple decades or will continue to have an impact for many years to come (e.g. increased urban development, river regulation, seagrass and mangrove loss, nutrient loading from stormwater, climate change; Lotze et al. 2006, Waycott et al. 2009, Gillanders et al. 2011).

Unfortunately, we often lack evidence linking specific events or types of environmental change to observed shifts and trends in coastal ecosystems. This is particularly relevant to Western Port, where very little research has been conducted since the 1970s compared to the neighbouring and much better studied Port Phillip Bay (Keough et al. 2011). This lack of empirical understanding hampers our ability to manage Western Port's environments, in particular the identification and prioritisation of measures that will ensure that its biological systems remain healthy and its fisheries remain viable, productive and sustainable.

A two-phase project has been proposed that will directly address the historical drivers of biological change across a range of key species in Western Port, and then use this insight to undertake scenario testing and prioritisation of management actions. This work closely aligns with Melbourne Water's responsibilities, including the protection and improvement of the health of Melbourne's waterways and bays. Here we report on phase 1 of the project which will focus on the drivers of long-term change for three key fishery species (Snapper, King George whiting, Elephant fish).

Project Objectives

Here, we directly addressed the historical drivers of biological change across three key fish species in Western Port (pink snapper *Chrysophrys auratus*, King George whiting *Sillaginodes punctatus*, elephant fish *Callorhinchus milii*, Keough et al. 2011, Jenkins and Conron 2015), that included:

1) Developing conceptual models describing how environmental drivers could influence each species;

2) Collating a database of fishery catch, recruitment and otolith-based growth time series;

3) Generating time series of environmental drivers affecting each species; and

4) Time series and change point analyses to explore commonalities and differences amongst species and quantify fish responses to environmental drivers.

Project Outcome

The successful completion of phase 1 of this project has resulted in the development of detailed conceptual models of how environmental drivers affect three key fisheries species, and a novel, integrated and long-term empirical appraisal of fish population and fishery responses to these drivers, including local and regional climate, regional weather, and water quality. Information from this study will support decision making regarding on-going investment in activities to protect and improve the health of Western Port. The results of phase 1 will also serve as a proof of concept for more extensive analyses to be conducted in phase 2, where we will focus on system-wide biological change (including birds, seagrass and other fish species) and the prioritisation of management actions.

Materials and Methods

Study Area

Western Port has been subdivided into segments based on physical characteristics (Marsden et al 1979): the Lower North Arm, Upper North Arm, Corinella Segment, Rhyll Segment and Western Entrance Segment (**Figure 1**). Western Port has a large area of intertidal mudflats (~ 1/3 of area) dissected by dendritic channels (depth to 30 m in the Western Entrance Segment) with strong tidal currents. The tidal range of 2 to 3 metres means that a large volume of water is exchanged between the bay and the offshore waters on each tidal cycle. Most of the freshwater input is in the north-east of the bay (**Figure 1**), and entrainment of sediments from the catchment leads to increased suspended sediments in this area (Keough et al. 2011). Circulation is primarily clockwise, meaning that freshwater inputs are mainly transported through the Corinella and Rhyll segments (Keough et al. 2011). The dominant habitat in Western Port is seagrass; the major species are *Zostera muelleri* on the intertidal flats, *Zostera nigricaulis* in lower intertidal and shallow sub-tidal areas, and *Amphibolis antarctica* in the oceanic Western Entrance Segment (Keough et al. 2011).

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Figure 1: Map of Western Port showing segments based on physical characteristics (from Marsden et al (1979))

Conceptual Model Development

Conceptual models for King George whiting (Figure 2), snapper (Figure 3) and Elephant fish (Figure 4) were developed to summarise the life histories of the species and important environmental drivers likely to be affecting their populations. These models were used to inform the selection of biological and environmental time series to use in the later analyses. King George whiting (Figure 2), spawn offshore from Western Port in the winter, most likely off the coast of far western Victoria to south-east South Australia (Jenkins et al. 2000). Larvae of 3-5 months age enter Western Port in the spring, and numbers vary widely from year to year (Jenkins 2005) depending on conditions in Bass Strait such as sea surface temperature (SST) and current strength (Jenkins and King 2006). Once in the bay, juveniles have a strong dependence on seagrass habitat, and are found in or near this habitat as they grow (Jenkins 2011). When nearing adulthood at an age of 4 to 5 years, older juveniles move back out into Bass Strait and return to the spawning areas. Key environmental drivers are expected to be variables affecting larvae such as SST and west to east current strength, and variables affecting seagrass habitat for juveniles such as water quality of catchment inflows.

Snapper (Figure 3) primarily spawn in Port Phillip Bay (PBB) with migrations to and from Western Port (Hamer et al. 2011). Some spawning of snapper is also likely to occur in Western Port but currently the extent of this spawning is unknown. In PPB, recruitment of young snapper to the population is highly variable from year to year and depends on water temperature and food quality for larvae (Murphy et al. 2012, 2013). Food quality (type of plankton), in turn depends on catchment inputs, particularly nutrients (Black et al. 2016). Snapper in Western Port live in deeper, often reef associated habitats (Jenkins and Conron 2015). Key environmental drivers would be expected to be water temperature and river (primarily nutrient) inputs.

Elephant fish (Figure 4) in Western Port are seasonal migrants from Bass Strait that come into the bay for spawning in February and March (Jenkins 2011). The annual catch was very low up to the 1980's but increased to a peak in the late 90's to 2000's. Females tend to lay their eggs in bare mud areas near seagrass, while neonates (juveniles) are thought to use seagrass habitat as a nursery area (although data on this is lacking) (Jenkins et al. 2015). Key environmental drivers are expected to be water quality from the catchment affecting the breeding area in the Rhyll basin, including effects on seagrass and sedimentary habitat.

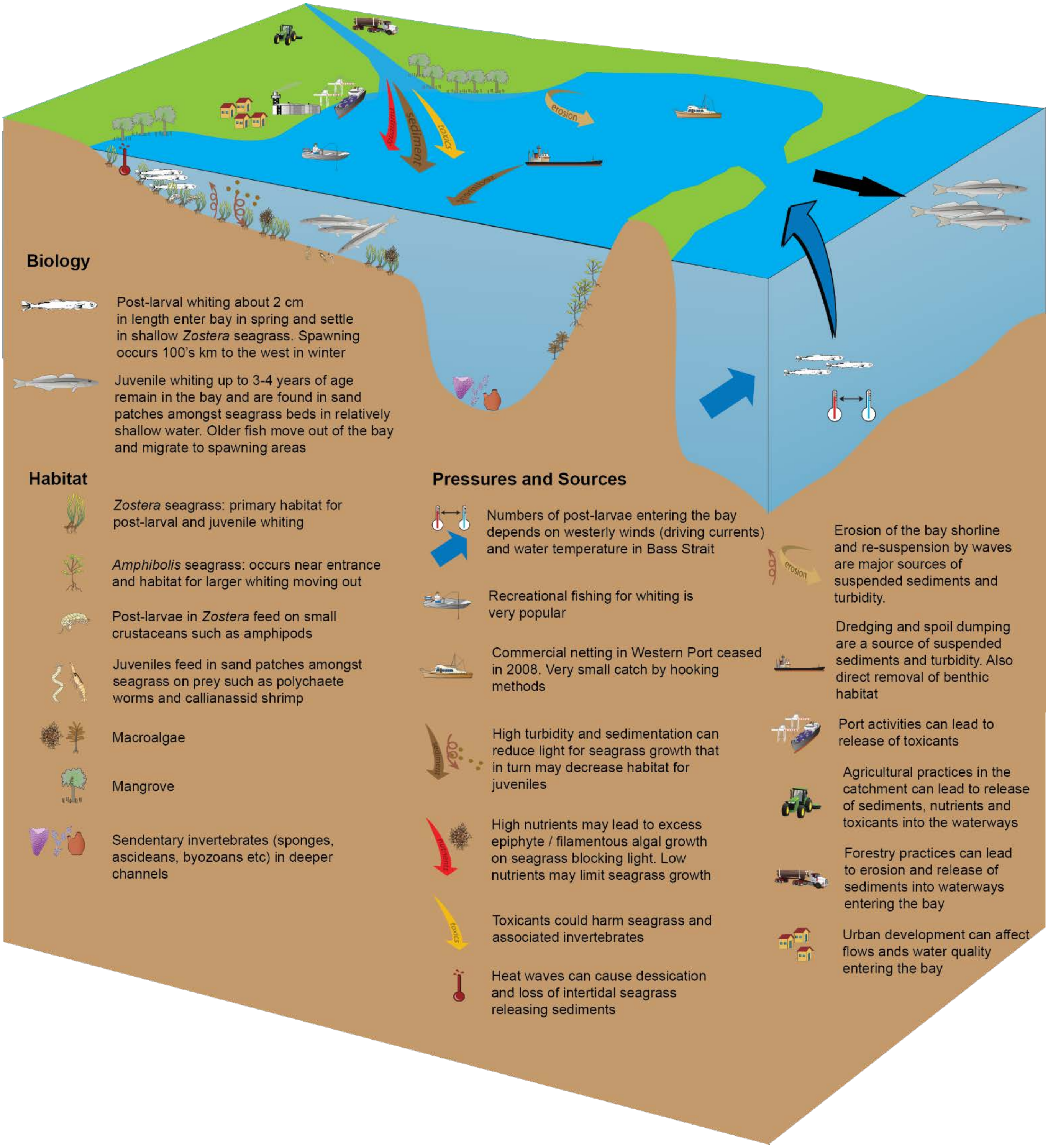

Figure 2: Conceptual model of King George whiting life history and important environmental drivers. Habitats without further comment are deemed of low importance for this species.

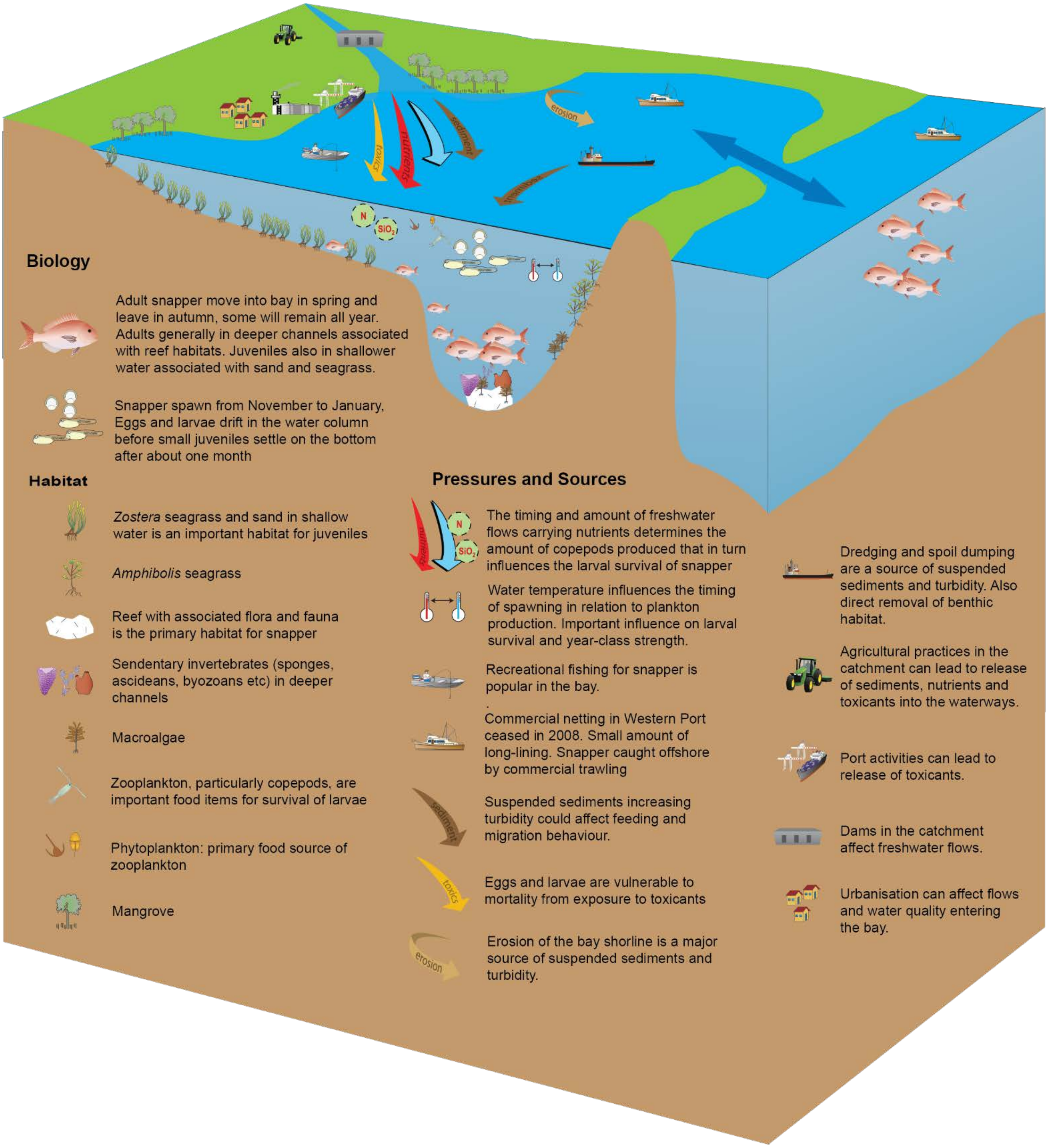

Figure 3: Conceptual model of snapper life history and important environmental drivers. Habitats without further comment are deemed of low importance for this species.

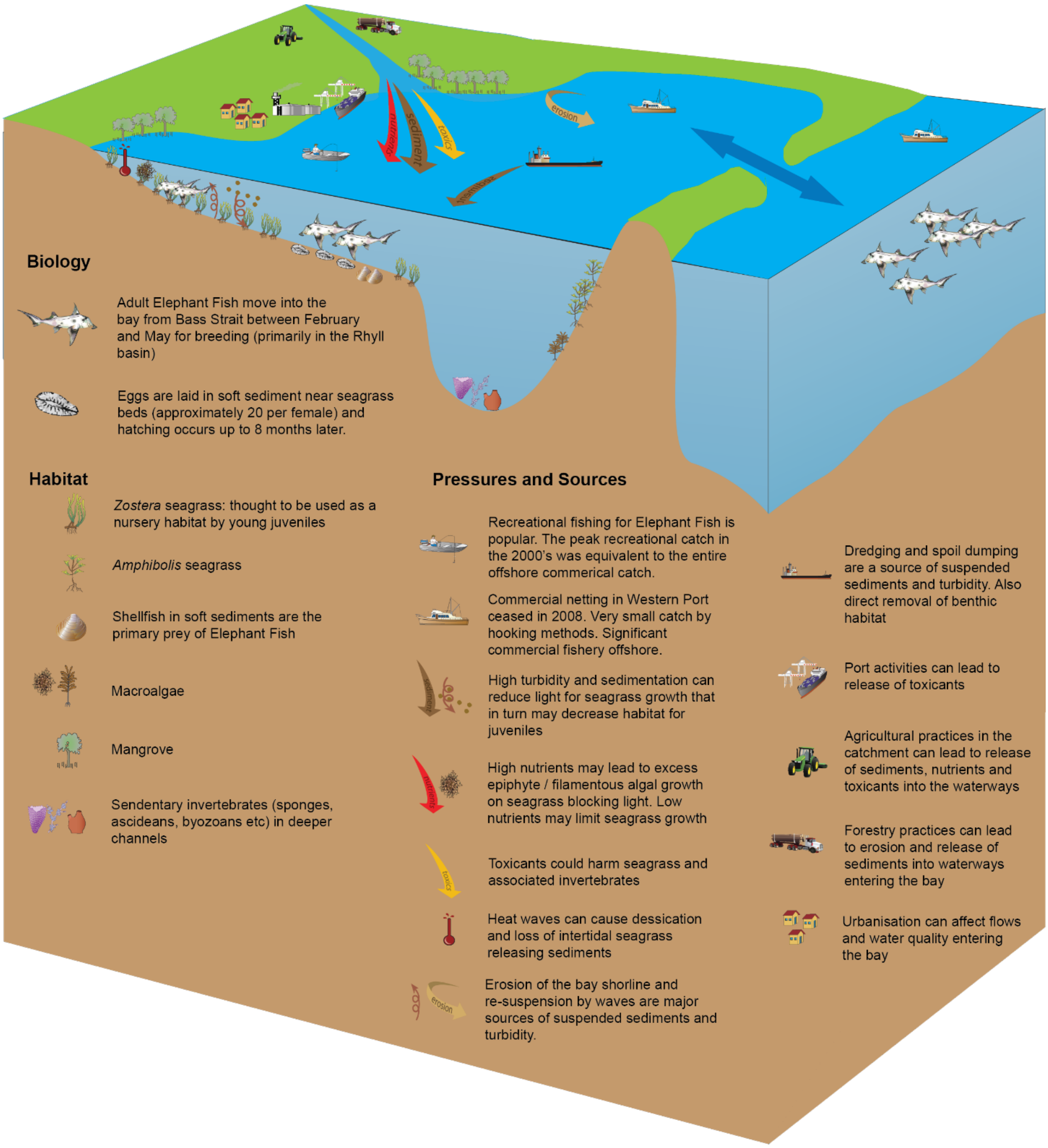

Figure 4: Conceptual model of elephant fish life history and important environmental drivers. Habitats without further comment are deemed of low importance for this species.

Fisheries time series

We accessed, or developed, 13 biological time series for three key Western Port fisheries species. These included estimates of annual total commercial catch, estimates of abundance (commercial and recreational catch per unit effort (CPUE)), recruitment time series (from Port Phillip Bay), and novel somatic growth time series based on otolith increment measurements. We provide a brief summary of these 13 fisheries time series in **Table 1**, followed by more detailed variable descriptions.

Table 1: Summary of 13 fisheries time series used in analyses of long term change in Western Port Bay.

Species	Time series	Year range	Data source
Pink snapper	Commercial catch (tonnes)	1914-1993	Fisheries Victoria
	Commercial CPUE (kg/200 hook lifts)	1978-2013 (2 missing years)	Fisheries Victoria
	Recreational CPUE (fish/angler hour)	1998-2014	Fisheries Victoria
	Recruitment (Port Philip Bay; num. 0+/ trawl survey)	1993-2015	Fisheries Victoria
	Somatic growth rate (mm otolith growth/ year)	1996-2013	This study
King George whiting	Commercial catch (tonnes)	1914-1993	Fisheries Victoria
	Commercial Seine CPUE (kg/shot)	1978-2007	Fisheries Victoria
	Commercial mesh CPUE (kg/day)	1978-2007	Fisheries Victoria
	Recreational CPUE (fish/angler hour)	1998-2014	Fisheries Victoria
	Recruitment (Port Philip Bay; av. num. recruits/site)	1996-2015	Fisheries Victoria
	Somatic growth rate (mm otolith growth/ year)	1994-2014	This study
Elephant fish	Commercial mesh CPUE (kg/day)	1978-2007	Fisheries Victoria
	Recreational CPUE (fish/angler hour)	1998-2013 (1 missing year)	Fisheries Victoria

Historical commercial catch (tonnes) Western Port supported a significant commercial fishery from the early 1900s until a commercial netting buyout in 2007. Annualised measures (fiscal year) of total catch (across all capture methods) for snapper and King George whiting were available from Fisheries Victoria for the period 1914-1993. No total catch data was available for elephant fish over the same period.

Commercial catch per unit effort (CPUE) CPUE standardises catch (here kg) by the amount of effort used to capture those fish, and is an indirect measure of the abundance of species. When fish are abundant, more can be caught with minimal effort compared to when fish are scarce. Different fishing methods are used to target snapper, King George whiting and Elephant fish. We therefore calculated species-specific CPUEs for the primary method of capture using fisheries log book data (snapper: long line [fish/200 hooks]; King George whiting: beach seine [kg/shot] and mesh nets [kg/day]; Elephant fish: mesh nets [kg/day]).

Recreational catch per unit effort (CPUE) Relative estimates of abundance were also estimated from recreational fisher creel surveys. Recreational anglers can operate in different locations, at different times and target different species/ life history stages compared to commercial fishers and therefore provide a complementary index of abundance. Creel survey data (fish/angler hour) was collected by interviewers at boat ramps from 1998 to 2015 (Conron et al. 2016).

Recruitment indices Western Port Snapper and King George whiting are from the same stocks as fish in Port Phillip Bay (Conron et al. 2016) and each species will likely display similar temporal patterns in recruitment across the two bays. We therefore used the long-term recruitment monitoring data for both species collected by Fisheries Victoria in Port Phillip as a proxy for Western Port recruitment. These data are derived from targeted, fisheries independent seine net surveys for King George whiting (average Oct-Nov abundance at 3 sites) and beam trawl surveys for snapper (average March/April abundance at 7 sites). No recruitment data was available for Elephant fish.

Somatic growth rate Fish possess structures called otoliths ('ear stones') which are biogenic carbonate structures located in the inner ear of fish. Otoliths assist fish in determining orientation and detecting movement, and grow proportionally to overall somatic growth through the periodic deposition of material. It is this unique characteristic that makes otoliths a valuable proxy for individual-level growth rates across a fish's life time.

Annual growth increments (**Figure 5**) were measured from 363 snapper and 694 King George whiting otoliths archived by Fisheries Victoria. Snapper were collected from 2010-2014 and King George whiting from 1997-2015. We adopted a stratified sampling regime to ensure all age classes were represented across the length of the chronology to avoids issues of growth estimate bias through time (Morrongiello et al. 2012). We used the mixed effects modelling technique described by Morrongiello and Thresher (2015) to partition otolith growth into its intrinsic (age and sex dependent) and extrinsic (environmental) components. These models enabled us to develop 18 and 21 years of growth indices (mm otolith growth/year) for snapper and King George whiting respectively.

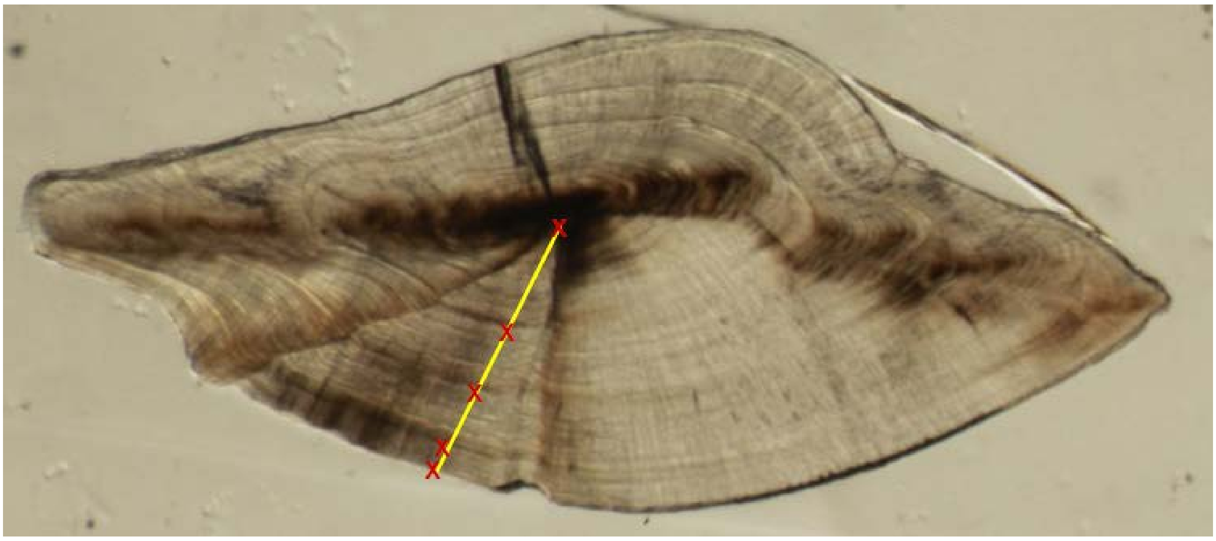

Figure 5: King George whiting otolith showing annual growth increments (under 1.6x magnification). The yellow line spans from the core to the edge of the otolith through the sulcus and the red crosses indicate the point of each annual increment.

Environmental data

We developed eight environmental variables (**Table 2**) hypothesised to influence the abundance, recruitment and growth of snapper, King George whiting and elephant fish in Western Port (**Figures 2-4**). These variables account for local and regional scale environmental processes. There was insufficient toxicant or heavy metal data available (only point estimates in 3-8 months per year, 1990-1998) to be included in formal analyses.

Table 2: Summary of environmental parameters used to explore drivers of biological change in in Western Port.

Variable	Description	Time period available	Data source
Local			
Total Flow (ML)	Total discharge of Bass River, Bunyip River and Cardinia Creek; affects salinity, nutrient levels and turbidity	1972-2015	Melbourne Water
Western Port SST (^{o}C)	Average daily SST from satellite imagery, calculated over area -38.23 to -38.53 and 145.04 to 145.06; local measure of temperature	1981-2015	NOAA OI SST V2 High Resolution Dataset*
Total Nitrogen (mg/L)	Average daily total nitrogen, based on measurements from three water stations in Western Port; measure of nutrient inputs	1990-2015	Environment Protection Authority
Secchi depth (m)	Secchi depth measured at three stations in Western Port; measure of turbidity	1990-2015	Environment Protection Authority
Chlorophyll *a* (ug/L)	Average daily chlorophyll *a*, based on spectrophotometry measurements of water sample from three stations in Western Port; measure of primary productivity	1990-2015	Environment Protection Authority
Regional			
Bass Strait SST (^{o}C)	Average daily SST from satellite imagery, calculated over area -37.5.0 to -41.27 and 140.97 to 150.3; regional measure of temperature	1981-2015	NOAA OI SST V2 High Resolution Dataset*
Southern Oscillation index (SOI)	Average monthly SOI index; important regional weather driver	1972-2015	Bureau of Meteorology#
Zonal Westerly Wind (ZWW)	Summed east-west wind vector (U) from Cape Otway lighthouse (Hamer et al. 2010); important weather driver of oceanic processes in SE Australia. Likely to be an indicator of the strength of west-east currents in Bass Strait	1972-2015	Bureau of Meteorology

* downloaded from http://www.esrl.noaa.gov/psd/data/gridded/data.noaa.oisst.v2.highres.html
# downloaded from http://www.bom.gov.au/climate/current/soi2.shtml

Data Analysis

Common trends

Dynamic factor analysis (DFA) is a multivariate time series analysis technique that allows us to look for a set of underlying common trends from a larger set of time series (Zuur et al. 2003b). DFA is ideally suited to analysing time series of differing lengths, with missing values, or displaying non-stationarity (i.e. temporal trend). These attributes make it a much better tool to reduce the dimensions of time series data compared to principle components analysis (PCA) or canonical correspondence analysis (CCA) that are not specifically designed for this purpose (Zuur et al. 2003a). The general aim of DFA is to model as few common trends as possible that explain modes of variation whilst also giving a reasonable model fit. Common trends are related back to the original data by multiplying them by time series-specific factor loadings. Stronger factor loadings (positive or negative) indicate that a given trend explains a greater proportion of a particular time series' variation; weaker factor loadings indicate that a trend explains less variation.

We used the 'MARSS' package in R (Holmes et al. 2012) to perform DFA on our 13 fisheries time series. First, we standardised the time series by subtracting its mean and dividing by its standard deviation. Then, we used Aikakie's Information Criterion corrected for small sample size (AICc) (Burnham and Anderson 2002) to select among competing models fit with different numbers of underlying trends (m: 1-5) and covariance matrix **R** (same variance and no covariance; different variance and no covariance; same variance and same covariance). We only looked for up to five trends as any more would suggest that there is little commonality amongst fisheries time series. Models with ΔAICc <2 (i.e. difference between a given model's AICc and that of the best model) are considered to have similar support (Burnham and Anderson 2002). DFA models were run with up to 100,000 iterations to ensure convergence.

Change-point analyses

Bayesian change point (BCP) analysis was used to investigate where change points, a partition in a time series, exist in each data set. These partitions are assumed to form among unknown blocks which are characterised by a constant mean value. Analysing change points in a Bayesian framework allowed us to estimate the probability of a change point occurring at a certain location and thus an indication of the level of uncertainty around the presence of a given change point. One limitation of change point analysis is that most methods assume a step change (i.e. change in mean) rather than a change in the trend (Rodionov 2005, Andersen et al. 2009, but see Thomson et al. 2010). Biological time series are often characterised by underlying trends and change point analysis can incorrectly identify one or multiple step changes during an increasing or decreasing value period (e.g. Möllmann and Diekmann 2012). Nonetheless, this technique has been successfully applied to time series data from Port Phillip Bay (Parry and Hirst 2016).

We used the 'bcp' package in R (Erdman and Emerson 2007) to estimate the posterior probability of a change point for a given year (indicating that step change occurred in the following year) for each time series. We used a burn-in period of 10,000 iterations, and then ran 50,000 iterations to estimate the sample posterior distribution. Convergence diagnostics were performed using the 'coda' package in R (Plummer et al. 2006).

Environmental relationships

We used multiple regression to explore the relationship between the common trends from DFA and a suite of local and regional environmental predictors on both annual and summer seasonal scales (Table 2). Water quality data was only available from 1990 onwards so we truncated trend data to the period 1990-2015. Annual and summer variables, local and regional temperature, and nitrogen and secchi disk measures were modelled separately due to co-linearity. Competing models for each trend were ranked using AICc.

Results

Fisheries time series

We collated Western Port fisheries data spanning 100 years (1914-2015), with the average length of any one time series being 32 years. Plots of the 13 fisheries time series are shown in **Figure 6**. A total of 2768 growth increments from 363 snapper and 1036 growth increments from 694 King George whiting were used in models to estimate somatic growth variation through time. The snapper growth chronology spanned 1996-2013 and was characterised by an increase in growth rate from about 2004 to the current (**Figure 7**). The King George whiting growth chronology spanned 1994-2014 and was characterised by an initial increase in growth up to 1999 followed by two years of below average growth, then relatively constant (above average) growth from 2002 until there was a drop in growth rate around 2010 followed by above average growth in the most recent period (**Figure 8**).

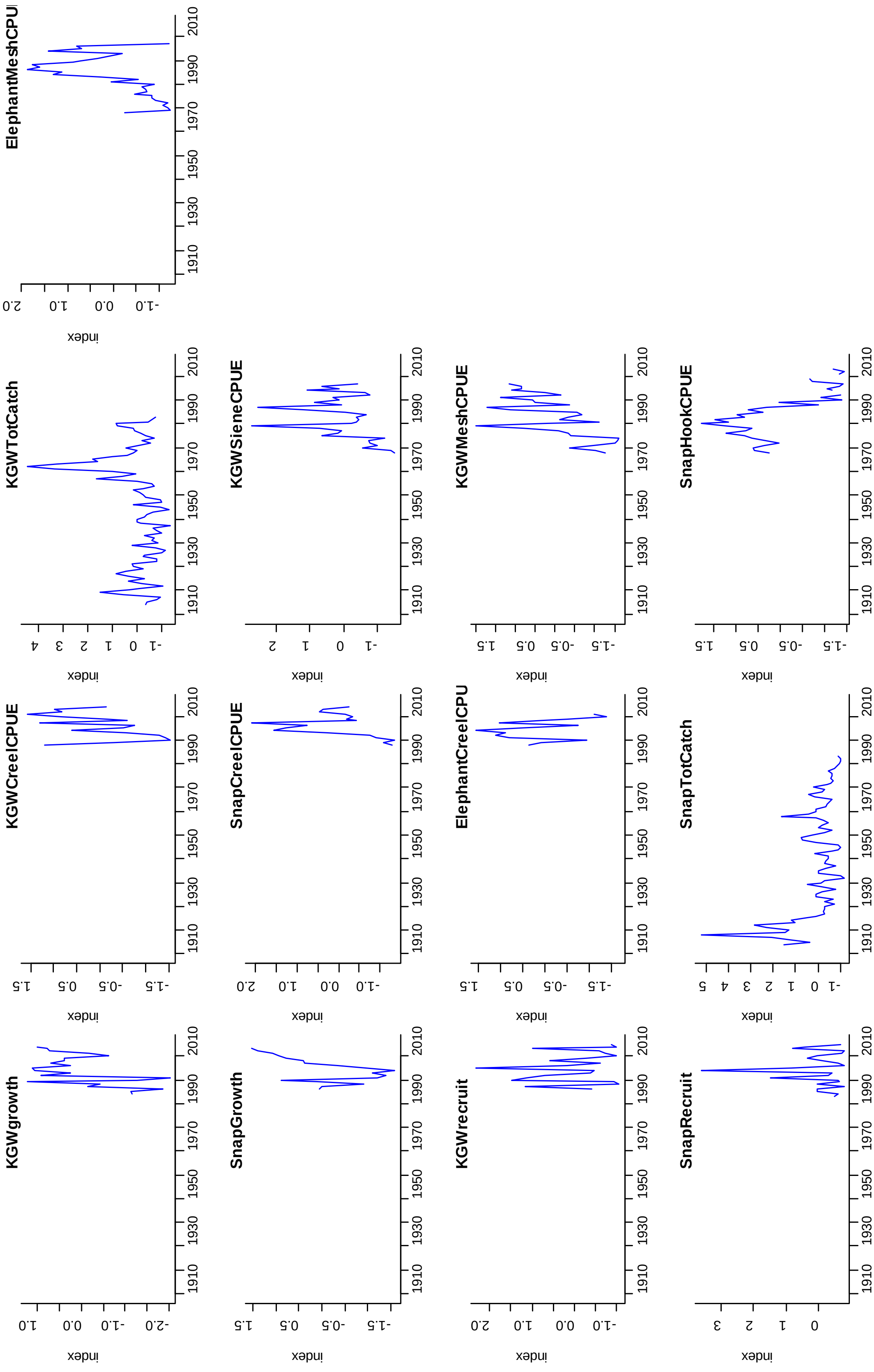

Figure 6: Time series of 13 biological parameters for three species in Western Port

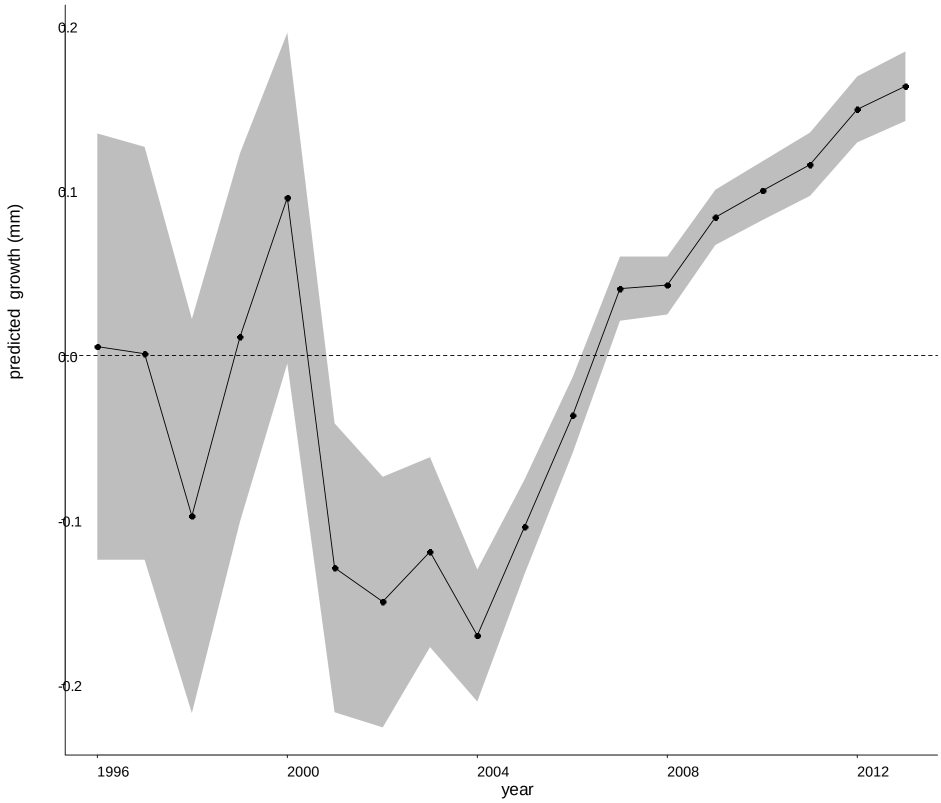

Figure 7: Novel growth time series for pink snapper in Western Port. Shown are Year random effect conditional modes (best linear unbiased predictors BLUPs) which provide an estimate (± 95% CI) of predicted annual growth rate associated with environmental drivers. The horizontal dotted line represents the long-term average (fixed effect intercept), with points above this line indicative of good growth years whilst those below poor growth years.

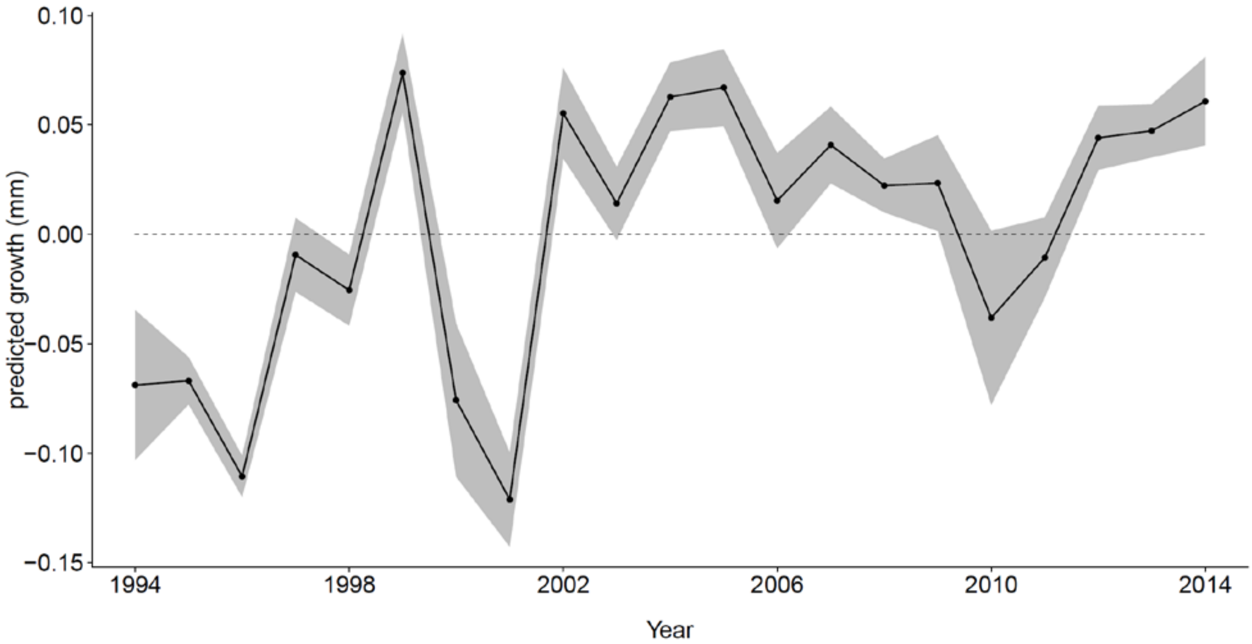

Figure 8: Novel growth time series for King George whiting in Western Port. Shown are Year random effect conditional modes (best linear unbiased predictors BLUPs) which provide an estimate (± 95% CI) predicted annual growth rate associated with environmental drivers. The horizontal dotted line represents the long-term average (fixed effect intercept), with points above this line indicative of good growth years whilst those below poor growth years.

DFA: investigating common trends in fisheries time series

We fitted 15 competing models with different covariance structures and up to 5 common trends (**Table 3**) to the 13 fisheries time series. The overwhelmingly best model estimated there to be three common trends and had a diagonal and unequal covariance matrix (**Table 3**). There was no support for any of the other models. The best model did a good job predicting each individual time series, with strong concordance between individual time series (blue lines) and model predictions (black lines) (**Figure 9**). The DFA allows for the hindcasting of individuals time series based on modelled relationships to other biological data.

Table 3: Model selection results for dynamic factor analysis performed on 13 fisheries time series from Western Port (1910-2015). Best model is highlighted in bold

Covariance matrix (R)*	Number of trends (m)	logLik	ΔAICc
diagonal and equal	1	-542.68	126.66
diagonal and equal	2	-493.47	54.80
diagonal and equal	3	-471.49	36.69
diagonal and equal	4	-463.92	46.39
diagonal and equal	5	-449.04	40.16
diagonal and unequal	1	-524.15	116.17
diagonal and unequal	2	-462.87	21.87
diagonal and unequal	**3**	**-438.16**	**0.00**
diagonal and unequal	4	-431.64	13.48
diagonal and unequal	5	-426.96	29.30
equalvarcov	1	-542.49	128.43
equalvarcov	2	-492.48	55.12
equalvarcov	3	-471.22	38.57
equalvarcov	4	-460.09	41.29
equalvarcov	5	-456.05	56.87

* Diagonal and equal: same variances and no covariance; diagonal and unequal: different variances and no covariance; equalvarcov: same variances and covariance

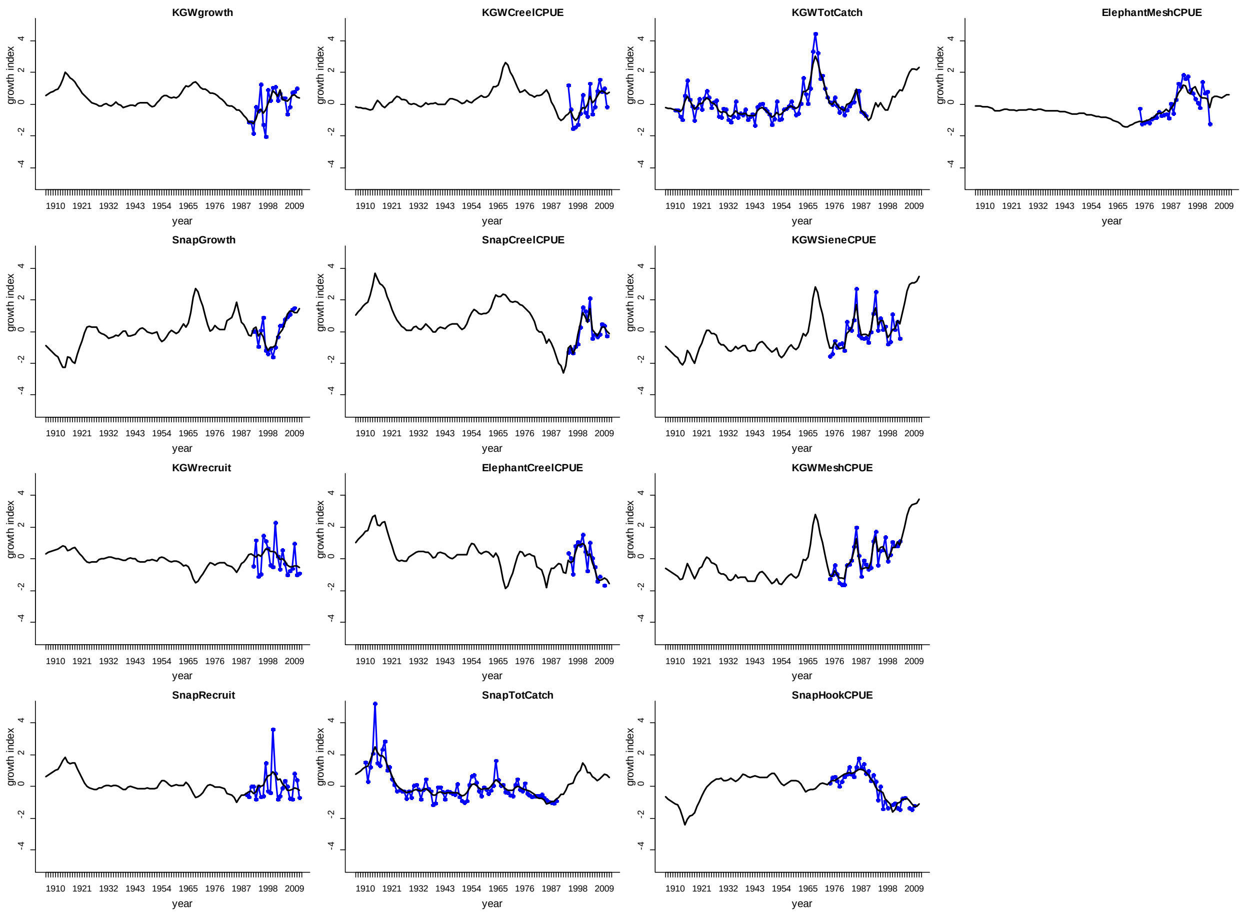

Figure 9: Visual assessment of best DFA model performance. Plots of 13 fisheries time series (blue lines) with model fits (black lines) from a model including three common trends and a diagonal and unequal variance-covariance matrix.

The first common trend (trend 1) shows an increase from 1975 to 1996, followed by a slight decline then remaining stable until 2015 (**Figure 10a**). The second common trend (trend 2) declines from 1970 to 1979, remains stable until about 2000 then increases steadily to 2015 (**Figure 10b**). The third common trend (trend 3) has large peak in the early 1900s, remains stable until 1975, then declines to 1990 followed by a rapid increase to the early 2000s (**Figure 10c**). All common trends were estimated with increased uncertainty prior to 1970 due to the paucity of time series data contributing to this earlier period (just snapper and King George whiting total catch).

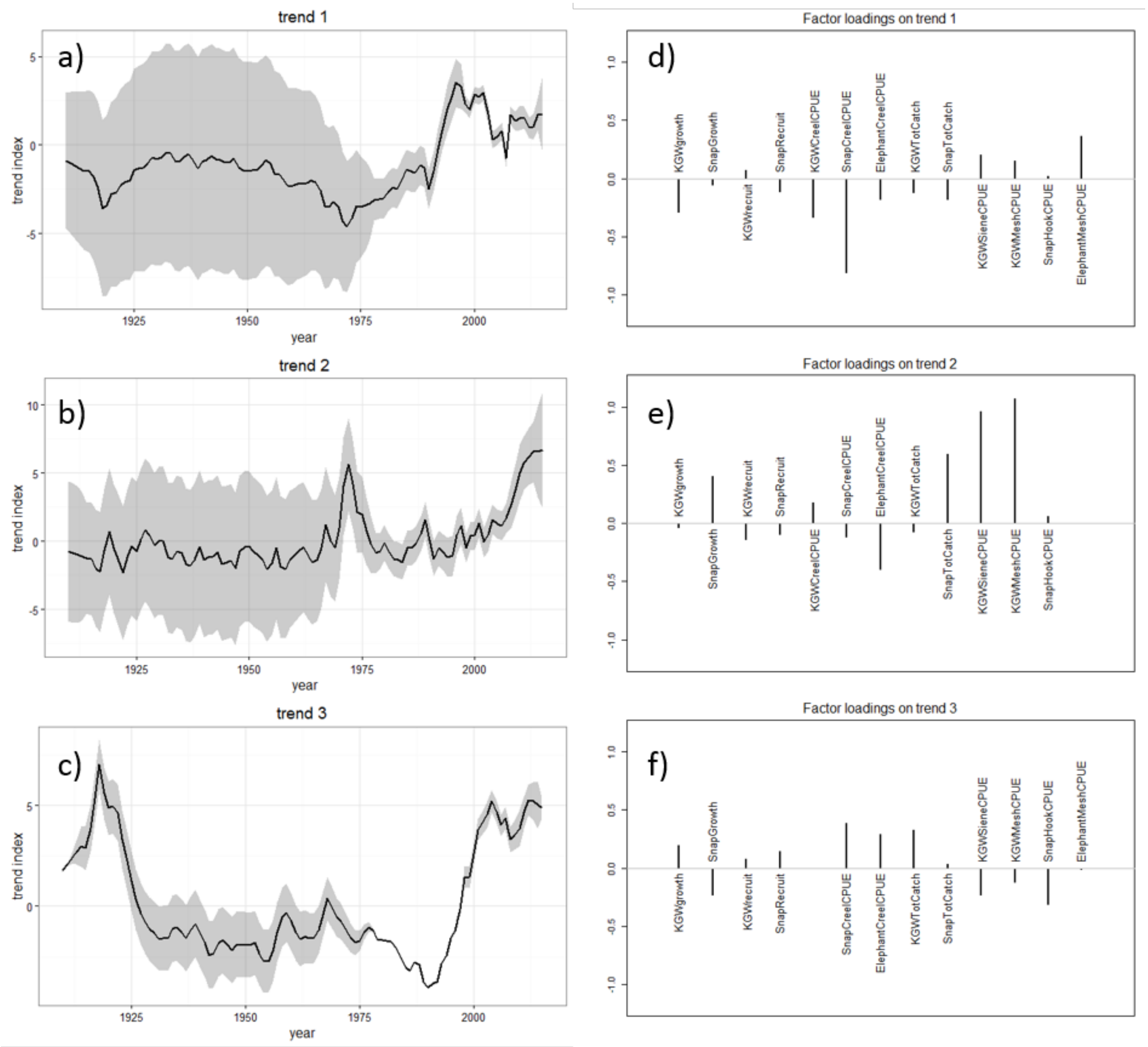

Figure 10: Common trends (± 95% CI) and factor loadings from the best DFA model describing variation in 13 fisheries time series from Western Port

Snapper recreational CPUE (SnapCreelCPUE) was strongly and negatively related to trend 1, as was to a lesser extent King George whiting growth (KGWgrowth) and King George whiting recreational CPUE (KGWCreelCPUE) (**Figure 10d**). This means that all these variables in part display the inverse of trend 1. Conversely, elephant fish commercial CPUE (ElephantMeshCPUE) was positively related to trend 1. Snapper growth (SnapGrowth), snapper total catch (SnapTotCatch), and King George whiting commercial catch (KGWSieneCPUE and KGWMeshCPUE) were all strongly and positively related to trend 2. Elephant fish recreational CPUE (ElephantCreelCPUE) displayed the opposite pattern (**Figure 6e**). There was a mix of positive and negative responses to trend 3 (**Figure 10f**).

Bayesian change-point analyses

We truncated the three common trends due to high levels of uncertainty in earlier trend estimation (**Figure 10**) and performed change point analyses on the period 1970-2015 (**Figure 11**). We detected evidence of 10 change points with a posterior probability >0.5 in the three common trends. There was strong evidence of a change point occurring in and around 1991-1993, 2003 and 2007 in trend 1 (**Figure 11a**). Change points occurred for trend 2 after 1970, 1973, and 2009 (**Figure 11b**). Change points were detected in trend 3 in 1994, 1997 and 2000, associated with the strong directional increase in this metric (**Figure 11c**). During this period there were 13 El Niño events, seven of which were classified as moderate to very strong, and seven La Niña events, six of which were classified as moderate to strong (**Table 4**) (Bureau of Meteorology 2016). Eight of the 10 change points were associated with El Niño events and the remaining two associated with La Nina events (**Table 4**). Despite this apparent relationship, no change point occurred during the 1980s despite the very strong 1982-83 El Niño event, or post 2009. A change point for trend 3 in 1984 had a posterior probability of 0.49, and a smaller probability for the same year for trend 1.

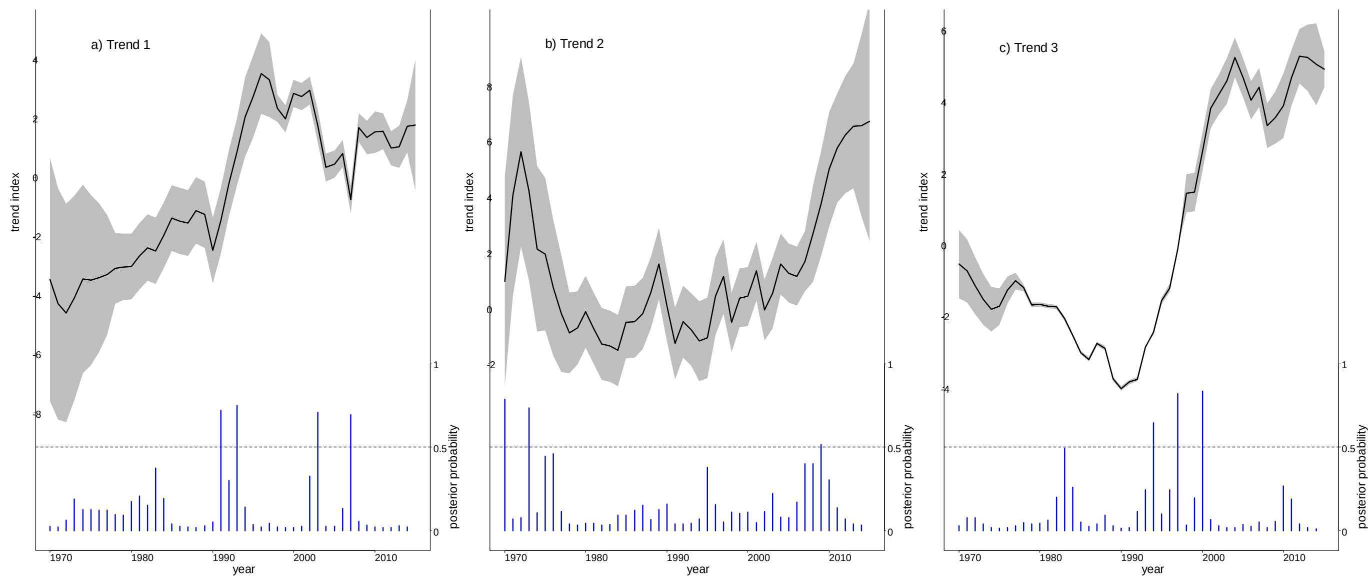

Figure 11: Bayesian change point posterior probabilities for three common trends (1970-2015). Trends (black solid line) and their 95% CIs (shaded area) are shown in the upper portion of each panel and the posterior probabilities for a change point occurring following each year (blue lines) are shown in the lower portion of each panel. The dotted horizontal line indicates those change points with >0.5 posterior probability of occurring.

Table 4: El Niño, La Niña and other notable events, with detected change points for period 1970-2015

year	phase	strength	Change point year*	Change point	Other events
1969-70	El Niño	weak	1971	Trend 2	
1970-72	La Niña	moderate	1971	Trend 2	Major seagrass decline
1972-73	El Niño	strong	1973	Trend 2	Major seagrass decline
1973-76	La Niña	strong	1973	Trend 2	
1977-78	El Niño	weak	-	-	
1982-83	El Niño	very strong	-	-	
1987-88	El Niño	moderate	-	-	
1988-89	La Niña	strong	-	-	
1991-92	El Niño	strong	1991	Trend 1	
1993-94	El Niño	weak	1993	Trend 1	
1994-95	El Niño	moderate	1994	Trend 3	Strong whiting recruitment
1997-98	El Niño	very strong	1997	Trend 3	Start of Millennium drought
1998-01	La Niña	moderate	2000	Trend 3	
2002-03	El Niño	moderate	2003	Trend 1	Strong snapper recruitment
2006-07	El Niño	weak	2007	Trend 1	Recruitment of strong 2005 whiting year class to fishery; end of commercial netting
2007-08	La Niña	moderate	-	-	
2008-09	La Niña	moderate	2009	Trend 2	End of Millennium drought
2009-10	El Niño	moderate	-	-	
2010-12	La Niña	weak	-	-	
2015-16	El Niño	very strong	-	-	

* change points with a posterior probability >0.5

Fish responses to environmental drivers

Trends 1, 2 and 3 were related to a mixture of local and regional scale environmental variables, over both annual and summer time periods (**Table 5**). Trend 1 was weakly and positively related to summer total nitrogen levels (**Figure 12**). Trend 2 was related to three variables: at a regional scale it increased with average annual Bass Strait sea surface temperature (**Figure 13a**), whilst at a local level it was negatively related to annual chlorophyll *a* levels (**Figure 13b**) but positively related to annual nitrogen levels (**Figure 13c**). Trend 3 was related positively to Bass Strait annual sea surface temperature (**Figure 14**).

Table 5: Environmental drivers of fisheries variation in Western Port (1990-2015). Shown are the seasons, parameters and total variance explained (R^2) for each trend's best model (based on ΔAICc).

Trend	**Season**	**Best model parameters**	**Multiple R^2**
trend 1	summer	total nitrogen	0.15
trend 2	annual	chlorophyll *a* + total nitrogen + Bass Strait SST	0.61
trend 3	annual	Bass Strait SST	0.46

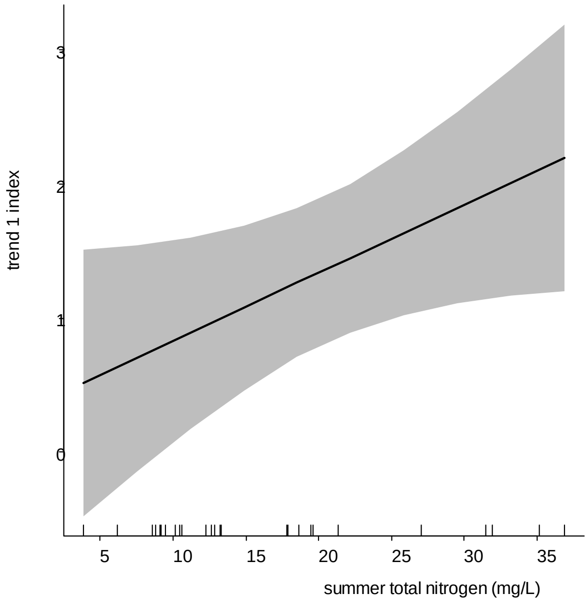

Figure 12: Predicted relationship between common trend 1 and summer nitrogen (± 95% CI). Short vertical lines on the horizontal axis illustrate observed nitrogen levels.

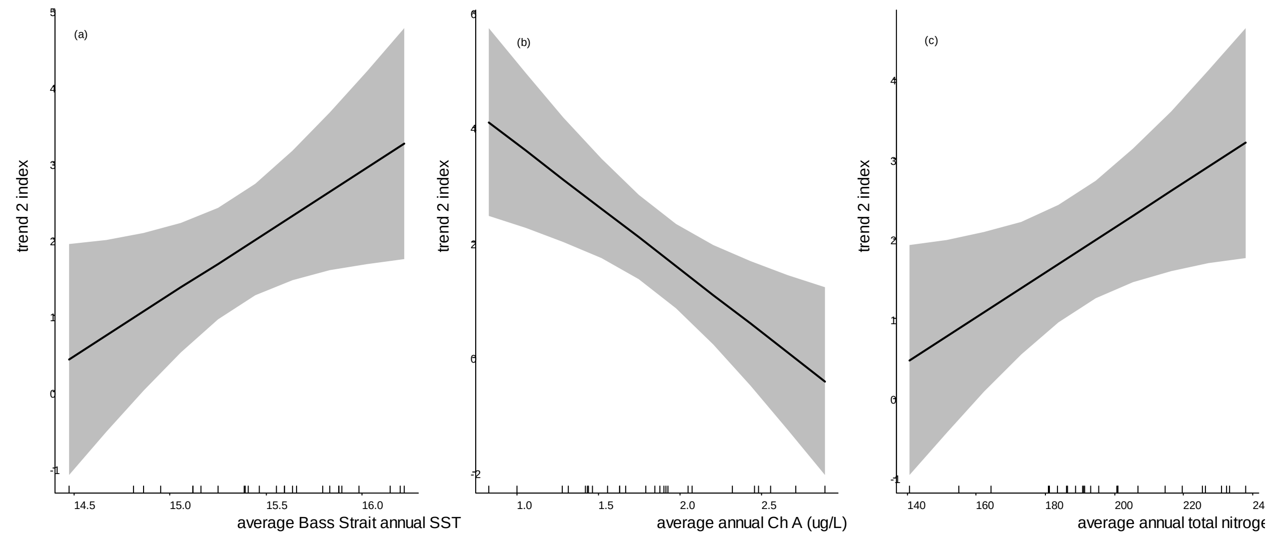

Figure 13: Predicted relationship between common trend 2 and environmental variables (± 95% CI). a) average annual Bass Strait sea surface temperature (SST ^{o}C); b) average annual chlorophyll *a* levels; and c) average annual total nitrogen. Short vertical lines on the horizontal axis illustrate observed environmental variable levels.

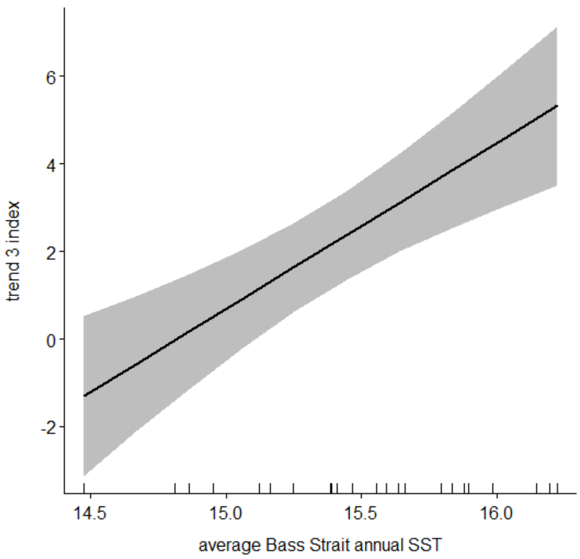

Figure 14: Predicted relationship between common trend 3 and average annual Bass Strait sea surface temperature (SST ^{o}C) (± 95% CI). Short vertical lines on the horizontal axis illustrate observed temperature levels.

Discussion

We show that there is clear commonality in biological variation in Western Port over the last 100 years, with three common trends best explaining the temporal variation in thirteen fisheries time series (1914-2015, average 32 years in length) analysed here. We also show that these three common trends are related in a logical way to a set of local and regional scale environmental drivers. Further, we have developed two new biological time series (snapper and King George whiting growth) that will be useful to future studies.

Snapper and King George whiting growth

We developed two novel growth time series for snapper and King George whiting spanning 18 and 21 years respectively. Snapper growth was depressed in the period 2001-04, followed by a steady increase in annual growth to the present. There were big snapper year classes in 2000/01 and 2003/04 and it is possible that high competition due to large cohort sizes during this time led to depressed growth. Similar patterns have been observed in other temperate Australian fish (e.g. Whitten et al. 2013). The steady increase in growth from 2004 might reflect a growth compensation associated with reduced density dependence caused by natural and fishing mortality of these year classes. It is not until 2009 that growth rates return to the 2000 high, but of note is the steady increase thereafter suggesting that other environmental drivers like temperature are also important (Thresher et al. 2007, Morrongiello and Thresher 2015). King George whiting growth was depressed in 2000/01 and again in 2010, two periods characterised by dominant La Nina conditions (**Table 4**). High rainfall in these years could lead to greater runoff and thus increased nutrient loads and turbidity above some required optimum (Hirst et al. 2016, Hirst and Jenkins 2017), which might reduce seagrass cover and therefore availability of habitat with negative flow on effects for whiting growth. King George whiting growth was above average for much of 2000s, a period characterised by record low rainfall (Dijk et al. 2013). This period may have had improved water clarity through reduced freshwater runoff and associated turbidity, leading to an increased cover of seagrass with positive flow on effects for whiting growth. Water temperatures during the 2000's were also above average, potentially leading to an extended period of increased growth.

Common trends

The three common trends each displayed a different temporal pattern of biological variability through time (**Figure 10**). Due to a scarcity of available data, we were unable to estimate these trends with much certainty prior to about 1970. We therefore focussed on the last 45 years of data and explored unique properties of each trend and how they were related to environmental variables.

Historical evidence provides essential information and context for developing models to predict the biological impacts of environmental change (Morrongiello et al. 2012). Whilst such long-term data sets are relatively common for terrestrial taxa and environments, it is sparse for aquatic systems (Richardson and Poloczanska 2008). This greatly hinders our ability to understand how local and regional drivers affect marine systems (see Keough et al. 2011 for Western Port-specific discussion) and in turn limits our ability to make meaningful management decisions to mitigate threats and increase system resilience. A unique property of our DFA models is that they allow us to hindcast the values for particular time series beyond the observational record, and also put observed values into a more historical context (**Figure 9**). This hindcasting is possible because each individual time series is related to the common trends by factor loadings, and longer time series facilitate the generation of common trends further back in time. The combination of common trends then becomes a proxy for a particular time series. For example, the model predicts that snapper growth would have been high in the early 1970s and late 1980s, and that in general the observed level of variability in each trend fits within the potential historical range. The validity of these predictions requires further targeted analysis, but raises the potential of developing a more in-depth understanding of long term change in Western Port.

It is important to acknowledge that DFA-derived common trends are not the property of any one species, rather an integration of variability across inputted data (Zuur et al. 2003a, Zuur et al. 2003b). Further, a given time series can be positively or negatively related to a trend, with the strength and direction of association estimated via factor loadings (**Figure 10**). Together, this can make it difficult to interpret patterns in common trends in terms of species-specific time series, unless there are strong factor loadings. Nonetheless, it is variation in the abundance data (commercial and recreational CPUE) and to a lesser extent total catch that is best represented by the common trends (factor loadings in **Figure 10**). This could be because processes that govern the abundance of fish (e.g. survival rates, attraction to region, recruitment into the fishery) are more similar across species than processes governing somatic growth and juvenile recruitment. It might also be reflective of there being only two each of recruitment and growth time series and thus their influence on the overall common trends is relatively limited. That said, many of the change points in the three trends appear related to recruitment and growth.

Change points

Ecological and biological processes can change from one state to another once particular thresholds have been surpassed (Beckage et al. 2007, Thomson et al. 2010). Two types of change points in time series can exist: abrupt or unusual changes in absolute values (step changes), or changes in the rates of change in absolute values (trend changes) (Beckage et al. 2007). Our analyses (Erdman and Emerson 2007) focussed on detecting 'step changes' as 'trend change' analyses can be computationally difficult to estimate (Thomson et al. 2010). Change points, and associated 'regime shifts' (more sudden, dramatic, and long lasting changes to ecosystem structure and function: Mollmann et al. 2015), have been detected in a growing number of marine and estuarine environments and linked to anthropogenic and climate drivers (e.g. Petersen et al. 2008, Moellmann et al. 2009, Thomson et al. 2010, Parry and Hirst 2016). For example, in Port Phillip Bay, change points in fish assemblage data were related to the arrival of the invasive northern Pacific seastar and drought (Parry and Hirst 2016).

We found good evidence (posterior probability >0.5) for a number of change points occurring across the three common trends during a 45 year period (1970-2015). The majority of these change points were associated with El Niño events, and to a lesser extent La Niña events (**Table 4**). In south east Australia, El Niño periods cause below average rainfall across the region whereas La Niña periods cause above average rainfall. It is possible that these discrete climatic events are causing changes in the state of Western Port's biological system via freshwater inflows.

More broadly, evidence for ENSO effects on SE Australian fishes is equivocal, despite the strong effect this large scale climate process has on the region's weather. For example, Morrongiello and Thresher (2015) and Morrongiello etal (2014) found no evidence for an ENSO effect on the growth of coastal and estuarine fishes, whereas others have implicated ENSO in driving patterns of marine fish abundance and recruitment (Harris et al. 1988). Further, ENSO showed a significant positive relationship with King George whiting catch in Port Phillip Bay (but not in Western Port) (Jenkins 2005). It was hypothesised that higher rainfall led to increased nutrient runoff, which in turn promoted seagrass and thus provided good habitat for juvenile King George whiting in Port Phillip Bay. The disparity amongst studies suggests that ENSO might have a threshold effect on biological systems, and thus is only evident at the event, rather than full time series scale as seen here.

We also found evidence for a change point associated with the cessation of commercial netting in Western Port in 2007. This result is intriguing as it is one of the first in the coastal region of Victoria to implicate fisheries management decisions on the broader ecosystem (Jenkins and Conron 2015, Parry and Hirst 2016) (but see Foster et al. 2015 for Bass Strait example). A similar commercial netting ban is to be implemented in Port Phillip Bay from 2018-2022 with the stated aim of improving recreational fishing opportunities. Interestingly, the 2007 change point on trend 1 was for this index to dramatically increase; however snapper recreational catches (which are strongly and negatively associated with this trend) actually declined. Further, more detailed analysis of this event is warranted.

Environmental drivers of trends

A range of local and regional drivers are hypothesised to have an impact on Western Port's snapper, King George whiting and elephant fish fisheries (conceptual models, **Figures 2-4**). We successfully simplified 13 time series associated with these three species into just three common trends, and found evidence for regional temperature and local nutrients and primary productivity being major drivers of change.

Trends 1 and 2 were significantly correlated to nitrogen and chlorophyll levels in Western Port, on both seasonal (summer) and annual time scales. This is important as both variables are associated with catchment processes (nutrient runoff) and subsequent seagrass and zooplankton dynamics in the bay (conceptual models). It is important to remember that whilst trend 2 is positively related to nitrogen, species that have strong negative factor loadings will actually display the opposite effect (**Figure 10**). Thus, it appears that snapper abundance is negatively, but King George whiting and elephant fish abundance positively, associated with nitrogen levels in Western Port. The negative relationship for snapper is consistent with research in Port Phillip Bay. There, high river flows and associated elevated nutrient loads shift the plankton assemblage composition such that it becomes unsuitable for larval snapper to eat, reducing larval survival and thus depressing juvenile recruitment (Black et al. 2016). In contrast, research on seagrass in Port Phillip Bay has found that large areas of seagrass in the bay are nutrient limited, and seagrass cover increases under conditions of higher nutrient loads (Hirst et al. 2016, Hirst and Jenkins 2017). Therefore, it is possible that increased nutrient loads to Western Port, up to some threshold point, may lead to increased seagrass growth with positive flow on effects to King George whiting and elephant fish (Jenkins et al. 2000, Jenkins and Conron 2015), consistent with the positive relationship in our analyses. On very high flows, excess nutrients lead to algal blooms (measured here by chlorophyll A) and increased turbidity, limiting seagrass growth. Of note, the major spawning area for elephant fish is the Rhyll basin (**Figure 1**) which is exposed to relatively nutrient rich waters from the north east of Western Port.

Significantly less is known about nutrient cycling in Western Port compared to Port Phillip Bay (Keough et al. 2011). Questions have been raised as to whether our understanding is transferable given the former is dominated by seagrass beds and mudflats whereas the latter is dominated by sub tidal sediments (Keough et al. 2011). More broadly, research into nutrient cycling in Western Port and the relative importance of turbidity, freshwater flow and temperature on driving plankton and seagrass is required.

Both trend 2 and 3 were correlated with average annual Bass Strait temperature. There was a strong co-linearity between Bass Strait temperature and Western Port temperature ($r=0.95$), so these two variables are essentially measuring the same process. The marine environment of south east Australia has undergone significant change over the last 50 years, directly attributable to a climate change-induced strengthening of the East Australian Current (Ridgway 2007). The rate of warming is 3-4 times the global average (Hobday and Pecl 2014) and has resulted in major changes to species' distributions, recruitment patterns and somatic growth (Thresher et al. 2007, Morrongiello and Thresher 2015, Sunday et al. 2015). Our results indicate that Western Port's key fisheries are climate sensitive and all else being equal, future warmer waters will result in increased King George whiting and snapper abundance. Such a prediction is supported by previous work that showed Bass Strait temperature is a key factor in determining King George whiting recruitment dynamics, while increased water temperature within the Port Phillip Bay has a positive influence on snapper recruitment (Jenkins and King 2006, Murphy et al. 2013).

Trend 2 is strongly and positively related to King George whiting catch and abundance, and appears sensitive to seagrass cover decline in the early 1970s (from change point analysis). King George whiting are known to be strongly dependent on seagrass habitat (Jenkins 2011, Jenkins and Conron 2015). It is possible that seagrass cover would be a good predictor of this trend's variability, although we were unable to test this hypothesis as time series are not currently available. The lack of association between this Trend and snapper is consistent with the lower dependence of this species on seagrass habitat (Jenkins and Conron 2015). This may also apply to elephant fish; however there is little information on the degree of dependence of this species on seagrass (Jenkins and Conron 2015). Interestingly, there were almost no elephant fish caught in Western Port in the 1970s (Jenkins and Conron 2015), indicating that in that period fish may not have been migrating into the bay to breed.

Interestingly, we found no evidence for zonal westerly winds (ZWW) or river discharge affecting common modes of variation. ZWW is thought to be a proxy for the strength of west-east currents in Bass Strait and is considered to be very important for whiting recruitment (Jenkins 2005) and other south-east Australian fishes (e.g. Wayte 2013). It must be noted that our common trends did not successfully explain variation in the two recruitment metrics included in this study (**Figure 9**), and this may have limited our ability to detect a ZWW effect. In Port Phillip Bay, freshwater inflows have been linked to pink snapper and sand flathead recruitment via a nutrient input pathway (Jenkins 2010), and elsewhere discharge affects black bream and estuary perch growth and recruitment (Morrongiello et al. 2014, Jenkins et al. 2015). In Western Port, the relationship between flow and bay nutrients was weak (r=0.22) likely due to the strong tidal mixing of this Bay (few days to 3 months, Harris and Robinson 1979, Longmore 1997, Keough et al. 2011), compared to other systems where freshwater inputs can have more influence on nutrient levels (e.g. Port Phillip Bay is ~270 days Walker 1999).

Future directions

This report covers Phase 1 of our investigation into the drivers of long-term change in Western Port. We deliberately focussed on snapper, King George whiting and elephant fish as these are three of the most important commercial and recreational fisheries species in the bay. In Phase 2 we propose to expand analyses to generate a more holistic understanding of how local and regional drivers affect a range of species, including sea birds, seagrass and other fish species. Our analyses here would have greatly benefited from time series of seagrass cover and turbidity. CSIRO (Scott Wilkinson funded by Melbourne Water) are developing a remote sensing method but the technique is only at the proof of concept stage and historical time series are not yet available. We would look to include this new environmental data in future modelling.

We have summed flow data from rivers entering both the north and south east of Western Port. The waters of Western Port display strong clockwise circulation, and the Bay itself is split into a number of discrete basins due to the position of Flinders and French Islands (**Figure 1**) (Keough et al. 2011). Given the characteristics, it is possible that separate north (Bunyip and Cardinia) and south east (Bass) flow variables might elucidate greater understanding of freshwater inputs into the Bay.

Conclusions and management implications

We have collated and analysed a data set of 13 biological and fisheries time series spanning the last 100 years in Western Port, including the development of two new growth chronologies. Using a series of sophisticated analyses, our study has identified clear commonalities in key Western Port fishery metrics related to both local and regional-scale drivers. Step changes in the three common trends are predominantly associated with El Niño and La Niña events, and to a lesser extent recruitment pulses and cessation of commercial netting. Temporal changes in the trends are associated with nutrient loads (negative for snapper abundance, positive for whiting and elephant fish abundance) and Chlorophyll levels (negative for whiting and elephant fish abundance) which likely affect fish through the food web and also in mediating seagrass cover (nutrients could promote growth, turbidity associated with algal blooms could limit growth) which provides essential habitat for juveniles. Sea surface temperature in Bass Strait was also related to the common trends, corroborating findings from previous work on snapper and whiting.

Estuarine and coastal environments are often characterised by a rich and diverse biota and provide multiple ecosystem services to humans. These environments are, however, under threat from a range of human-induced stressors that operate on local, regional, and global scales, including intensified urbanisation, altered hydrology and stormwater, resource exploitation and habitat modification, and climate change (Kingsford and Gillanders 2002, Moser et al. 2012). The effective management of estuarine and coastal environments is therefore a challenge that is only going to get more acute in the future.

Regional to global scale processes (drought, ENSO and sea surface temperature) played a major role in driving fisheries change in Western Port. From a regional management perspective, little can be done to ameliorate their direct current and future impact (Harley et al. 2006). We predict that continued oceanic warming and intensification of extreme El Nino events (Cai et al. 2014) will perturbate Western Port's environment and result in more biological change points and continued directional change in species associated with trend 2 and 3. Regional management can, however, promote system resilience to these larger environmental changes by addressing more local-scale stressors such as water quality and freshwater flow modification, nearshore habitat degradation and fishing pressure (Brown et al. 2013).

We found good evidence that nutrient and primary productivity levels in Western Port had important influences on fisheries that were both positive and negative. We speculate that nutrient loads can affect fish via a number of direct and indirect pathways (i.e. food chain productivity, habitat modification), although more research is needed to elucidate the actual causal mechanisms and these effects are likely to be context dependent (e.g. specific species involved, other environmental conditions). Such work is of high priority and will play an important role in informing management of nutrient inputs into the Bay from the surrounding urban and rural catchment. We found evidence that a change in fisheries management (cessation of commercial netting) caused a noticeable step-change in trend 1 which provides support for the efficacy of local management interventions. Surprisingly, we did not detect a direct effect of river inflows on the species studied. Perhaps this is due to flow variability being captured by larger climatic processes (e.g. ENSO), the geographic location of rivers, or because flow is related in a non-linear way to the fish studied. Nonetheless, flow can play a vital role in the delivery of nutrients and sediment, although less so in Western Port compared to Port Philip Bay. Appropriate flow management is still an essential part of estuarine and coastal environmental management (Kingsford and Gillanders, 2002) as it affects other species using Western Port (e.g. Crook et al. 2014, Jenkins et al. 2015, Koster et al. 2016), and thus is essential to maintaining or increasing system resilience to regional and global environmental change.

Importantly, the analyses presented here can be readily expanded to include other biological time series available, such as sea bird and penguin counts and other fish species, which would result in a more holistic view of the drivers of long-term biological change in Western Port's marine environment (Phase 2).

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