Estuarine gillnet fishery catch rates decline during drought in eastern Australia

Jonathan Gillson [a,*], James Scandol [b], Iain Suthers [a]

[a] *Fisheries and Marine Environmental Research Facility, School of Biological, Earth and Environmental Sciences, University of New South Wales, NSW 2052, Australia*
[b] *New South Wales Department of Primary Industries, Cronulla Fisheries Research Centre of Excellence, Cronulla, NSW 2230, Australia*

ARTICLE INFO

ABSTRACT

Commercial catch-per-unit-effort (CPUE) data from nine estuaries were related to hydrological variation in eastern Australia. Relationships between drought declaration, rainfall, freshwater flow and fisheries catch rates were assessed from 1997 to 2007. Estuaries varied from 0.5 to 2.0×10^6 ML of mean freshwater inflow per annum. Monthly CPUE data from gillnetting were used to infer the abundance of yellowfin bream (*Acanthropagrus australis*), dusky flathead (*Platycephalus fuscus*), luderick (*Girella tricuspidata*), sand whiting (*Sillago ciliata*) and sea mullet (*Mugil cephalus*). CPUE for all species examined, except yellowfin bream, increased in proportion to freshwater flow and decreased during periods of drought. Freshwater flow may affect CPUE by stimulating migration and schooling due to salinity fluctuations altering habitat availability. Minimum and maximum flows were important determinants of CPUE. Freshwater flow *per se* may not be as important in influencing CPUE as extremes in the hydrological continuum. Seasonal flows were consistently the most important aspect of the flow regime that explained the highest proportion of variability in CPUE. Seasonal freshwater pulses proximate to critical reproductive periods may influence catchability by triggering seaward spawning migrations in estuarine-dependent fish.

© 2009 Elsevier B.V. All rights reserved.

1. Introduction

Freshwater flow is a critical landscape process that regulates the physical, chemical and biological properties of estuaries (Wolanski, 2007). Natural variability in freshwater flow, through its effects on environmental conditions in estuaries, has a pivotal role in determining fisheries production (Grimes, 2001). Despite consistent links between freshwater flow and estuarine fish communities the underlying causal mechanisms remain poorly understood. Several mechanisms have been proposed: (1) alterations to abundance and survival resulting from physical changes in habitat availability (Kimmerer, 2002); (2) changes to migration and schooling altering catchability (Loneragan and Bunn, 1999); (3) improved growth and survival due to nutrient enrichment increasing primary and secondary production (Darnaude et al., 2004); (4) increased immigration into estuaries owing to changes in olfactory concentration gradients (Whitfield, 1994); and (5) recruitment variability arising from alterations in water physicochemistry (North and Houde, 2003). These mechanisms are not, however, mutually exclusive and will operate over differing temporal and spatial scales.

Estuaries require a sufficient amount of freshwater to maintain biogeochemical processes (Skreslet, 1986). Nevertheless, there is still a false perception that freshwater is 'lost' when it enters estu-

aries or coastal systems (Gillanders and Kingsford, 2002). Without adequate freshwater flow, estuaries can become "arms" of the sea with high salinity and poor water quality (Scharler and Baird, 2005). Concern has been expressed about the ecological effects of reduced freshwater flow into estuaries (Benson, 1981). Reductions in freshwater flow can have major impacts on estuarine fish communities (Drinkwater and Frank, 1994). Under drought conditions, freshwater flow can cease into estuaries causing sedimentation at the mouth, thus creating a physical barrier at the estuary–coastal interface (Cooper, 1994). Although droughts can have deleterious effects on estuarine biota (Copeland, 1966), limited information is available on the impacts of drought on estuarine fisheries production.

Comparative examination of the effects of freshwater flow on fisheries catch rates from a range of eastern Australian estuaries with distinct hydrological characteristics has received little attention. Eastern Australia has an extreme hydrological nature (Finlayson and McMahon, 1988), with climatic shifts driving sporadic rainfall and unpredictable flow events (Power et al., 1999). Coastal rivers in this region are influenced by alternating flood and drought dominated regimes (Erskine and Warner, 1998).

Here, we analysed relationships between hydrological variation and fisheries catch rates from nine permanently open estuaries in eastern Australia. Our objectives were to determine the influence of hydrological variation on fisheries landings by using effort-adjusted data from gillnetting to infer the abundance of five commercially important species of estuarine-dependent fish. We aimed to: (i) examine relationships between drought declaration, rainfall, fresh-

* Corresponding author. Tel.: +61 0 2 9385 2118; fax: +61 0 2 9385 1558.
E-mail address: jonathan.gillson@unsw.edu.au (J. Gillson).

Table 1
Rainfall, freshwater flow and fisheries data sources and formats.

Variable	Source	Format	Period
Commercial fisheries catch	NSW DPI ComCatch	Monthly landings (kg) and catch-per-unit-effort (CPUE)	1997–2007
Drought declaration	NSW DPI Drought maps	Monthly drought declaration maps	1992–2007
Rainfall	BoM	Monthly, seasonal and annual rainfall	1992–2007
Freshwater flow	NSW DWE/SCA	Monthly, seasonal and annual river discharge	1992–2007

NSW DPI ComCatch = New South Wales Department of Primary Industries Commercial Catch and effort database; NSW DPI Drought maps = New South Wales Department of Primary Industries drought situation maps; BoM = Bureau of Meteorology; NSW DWE = New South Wales Department of Water and Energy Pinneena 9.1 database; SCA = Sydney Catchment Authority.

water flow and fisheries catch rates from 1997 to 2007 and (ii) investigate aspects of the freshwater flow regime that were most important in determining fisheries catch rates.

2. Methods

Freshwater flow data from monthly, annual (July–June) and seasonal time series were examined (Table 1). Seasons were selected to correspond with the spawning period of individual species because this time interval is postulated to be a key determi-nant of inter-annual variability in survival (Houde, 1987). For this purpose, austral seasons were defined as: winter (June–August), spring (September–November), summer (December–February) and autumn (March–May). Note that the summer period incor-porated data from the December of the previous year. Monthly CPUE data in nine estuaries were examined during drought declared and undeclared periods. Relationships between drought declara-tion, rainfall, freshwater flow and species-specific catch rates were investigated in the Clarence, Hunter and Hawkesbury River estuar-ies.
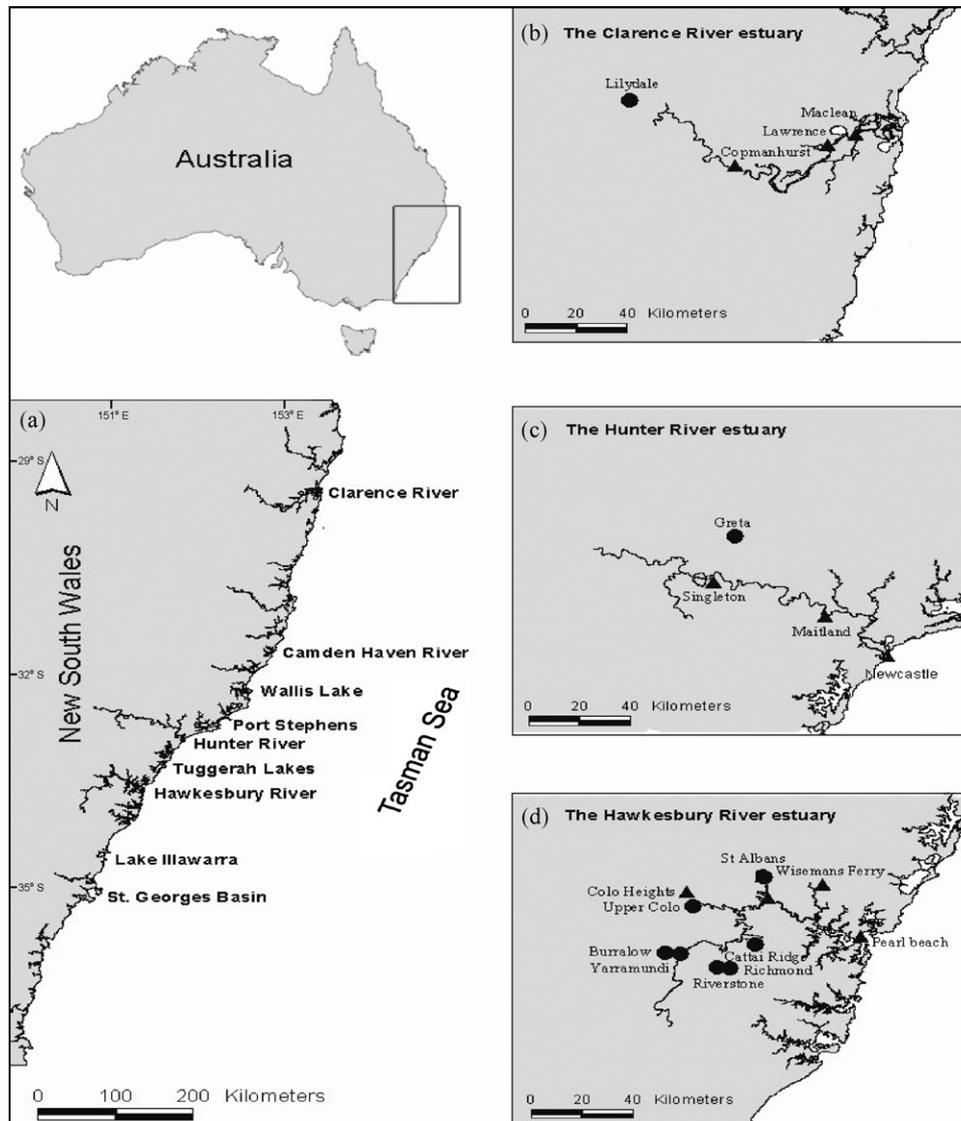

Fig. 1. Location of nine estuaries selected to investigate the impacts of freshwater flow on estuarine fisheries catch rates in eastern Australia (a). Rainfall [▲] and freshwater flow [●] gauging stations shown in relation to the estuarine reaches of the Clarence (b), Hunter (c) and Hawkesbury (d) River systems.

Table 2
Estuaries selected to investigate the impacts of freshwater flow on fisheries catch rates in eastern Australia.

Estuary	Latitude and longitude	Bioregion	Estuary type	Water area (km²)	Catchment area (km²)	Flow type	Flow regulation	Finfish production (t)
Clarence River	29°25′37.20″S, 153°22′19.20″E	Northern	Barrier river	89	22,400	High	Moderate	10,870
Hunter River	32°54′54.00″S, 151°48′03.59″E	Central	Barrier river	29	22,000	High	High	1,669
Hawkesbury River	33°34′10.20″S, 151°18′32.40″E	Central	Drowned river valley	100	21,500	High	High	3,997
Camden Haven River	31°38′09.59″S, 152°50′13.20″E	Northern	Barrier river	28	720	Medium	Low	1,526
Wallis Lake	32°10′26.40″S, 152°30′39.59″E	Central	Barrier lagoon	85	1,420	Medium	Low	4,577
Port Stephens and Myall Lakes	32°42′29.00″S, 152°11′45.59″E	Central	Drowned river valley	289	6,610	Medium	Low	5,794
Tuggerah Lakes	33°20′42.00″S, 151°30′14.40″E	Central	Barrier lagoon	70	760	Low	High	3,109
Lake Illawarra	34°32′38.40″S, 150°52′26.40″E	Central	Barrier lagoon	36	270	Low	Moderate	1,778
St. Georges Basin	35°11′06.00″S, 150°35′38.40″E	Central	Barrier lagoon	39	390	Low	Low	554

Bioregion refers to defined latitudinal estuarine regions (Pease, 1999). Estuary type describes geomorphological classification (Roy et al., 2001; Saintilan, 2004). Water area (km²) denotes the area of water comprising the estuary from the downstream estuarine limit to the upper limit of tidal influence (West et al., 1985). Flow type is a comparative theoretical measure of freshwater flow. Flow regulation symbolises the amount of regulation and extraction of freshwater within a catchment. Finfish production represents the summed total harvest of bream, dusky flathead, luderick, sand whiting and sea mullet in tonnes (t) from July 1997 to June 2007 for all examines estuaries, except St. Georges Basin. Commercial fishing ceased and a recreational fishing haven was declared in St. Georges Basin in May 2002.

2.1. Study areas

Nine estuaries along the Eastern Australian coastline were selected to evaluate the impacts of freshwater flow on fisheries catch rates (Fig. 1). The selection process was structured to obtain a suite of permanently open estuaries with differing geomorphologies, distinct freshwater inputs (three estuaries each with high, medium and low flow) and varying degrees of freshwater regulation (Table 2). Estuaries entering the Tasman Sea included: (1) the lower reaches of the Clarence River and adjoining Lake Woolooweyah; (2) Hunter River; (3) Hawkesbury River; (4) Camden Haven River; (5) Wallis Lake; (6) Port Stephens and adjoining Myall Lakes; (7) Tuggerah Lakes; (8) Lake Illawarra; and (9) St. Georges Basin. A detailed description of the geomorphological classification and physico-chemical characteristics of the estuaries is provided in Pease (1999), Roy et al. (2001) and Saintilan (2004).

2.2. Fisheries data

Monthly commercial fisheries catch and effort data were compiled from the New South Wales Department of Primary Industries (NSW DPI) ComCatch database from July 1997 to June 2007. Fisheries parameters consisted of landings (kg) and catch-per-unit-effort (CPUE, kg per day) from gillnetting. This passive gear provided a size specific measure of fisheries catch (≥80 mm mesh size) from which to assess the impacts of freshwater flow on CPUE, and presented the most consistent method available from which an index of abundance could be inferred. Estuarine fishers deploy gillnets throughout the upper limits of tidal influence. Fisheries landings were standardised for fishing effort by dividing the monthly or annual catch of each species from gillnetting by the total number of days fished with gillnetting in that month or year. Five species of estuarine-dependent finfish were selected for analysis: yellowfin bream (*Acanthopagrus australis*), dusky flathead (*Platycephalus fuscus*), luderick (*Girella tricuspidata*), sand whiting (*Sillago ciliata*) and sea mullet (*Mugil cephalus*). Yellowfin bream and black bream (*Acanthopagrus butcheri*) hybridise in eastern Australia (Roberts et al., 2009), and therefore are reported as a single species group in NSW DPI commercial catch records. Hereafter, we consider that yellowfin bream make the most significant contribution to our CPUE data. Yellowfin bream represent the majority (~95%) of commercial estuarine landings for this species group in New South Wales, with black bream only found in estuarine waters south of Myall Lakes. These species of estuarine-dependent fish were selected because they make the dominant contribution to commercial and recreational fisheries harvest. When the

five species studied were examined in combination, gillnet catch data for all five species were summed and then divided by total gillnet effort data (on either a monthly or annual timescale) to investigate the impacts of freshwater flow on multi-species catch rates.

2.3. Hydrological data

Monthly drought declaration maps from July 1992 to June 2007 were obtained from NSW DPI (2008). NSW DPI assesses climatic and agricultural factors to officially declare the drought-affected status of an area (http://www.dpi.nsw.gov.au/drought). Drought-affected areas were based on Rural Lands Protection Board districts, with the following estuaries co-located within the same district: the Hunter River and Tuggerah Lakes, and Wallis Lake and Port Stephens and Myall Lakes. Periods of drought declaration were examined for an area surrounding each estuary and formatted into a categorical variable with "0" and "1" representing the absence or presence of drought declaration, respectively.

Monthly rainfall from July 1992 to June 2007 was collated from the Bureau of Meteorology (BoM). Rainfall data from gauging stations in the upper, middle and lower reaches of the catchment were summed to calculate monthly rainfall. Gauging stations were located in Copmanhurst, Lawrence and Maclean for the Clarence River system; Singleton, Maitland and in Newcastle for the Hunter River system; and Colo Heights, St. Albans, Wisemans Ferry and Pearl beach for the Hawkesbury River system.

Monthly freshwater flow data were extracted from the Pinneena 9.1 database of the New South Wales Department of Water and Energy (NSW DWE) for the Clarence and Hunter Rivers; and obtained from the Sydney Catchment Authority (SCA) for the Hawkesbury River. Gauged freshwater flow data was provided for the Clarence River at Lilydale (~10 km upstream of the estuarine reaches); the Hunter River at Greta (~15 km upstream of the estuarine reaches); and the Hawkesbury River at Yarramundi, Burralow, Richmond, Cattai Ridge, Upper Colo, St. Albans and Riverstone (all <9 km upstream of the estuarine reaches) from July 1992 to June 2007. Several flow variables were selected to measure monthly, annual and seasonal variability in the flow regime: (i) total, minimum, mean and maximum river discharge; and (ii) total seasonal river discharge as a percentage of total annual river discharge (referred to as % total seasonal flow). Variables were categorized for their impacts on: (i) catchability (i.e. a relatively short-term response with no lags, coinciding with the seasonality of the fishery); and (ii) recruitment (i.e. a delayed response with lags equalling the appropriate age when a cohort recruits to the fishery). Lags of

up to five years were considered for all examined species (indicated by the notation $L - x$).

2.4. Data analysis

Fisheries, rainfall and freshwater flow variables were $\log_{10}$ transformed to normalise variances. Transformed variables were normally distributed (Lilliefors test) with no evidence of heteroscedasticity (standardised quantile plots).

Univariate analyses were performed to examine relationships between hydrological variables and CPUE. Initial exploratory analysis was undertaken with linear regression techniques and corrected with Bonferroni inequality adjustment (Sokal and Rohlf, 1995). A one-way analysis of variance (ANOVA) was used to compare rainfall, freshwater flow and CPUE during drought declared and undeclared periods. The ANOVA models applied to each estuary in Section 3.2 consisted of one fixed factor (drought) and two levels (declared and undeclared). Interactions between drought and estuary were not considered. Forward stepwise multiple regression analysis was used in Section 3.3 to identify hydrological variables that explained the highest proportion of variability in CPUE. The general equation used to predict CPUE from environmental variables was:

$$U_t = f(x_t) = \sum_{i=0}^{n} \beta_i x_{i,t} + e_t$$

where U_t is the CPUE (yellowfin bream, dusky flathead, luderick, sand whiting or sea mullet) at time t, $x_{i,t}$ the covariates that represent environmental factor i (drought declaration, rainfall or freshwater flow), t the unit of time (month, year and season), n the number of covariates, β_i the coefficient for covariate i and e_t is the residual term for observation t. The environmental factors were either continuous variables such as rainfall and freshwater flow, or categorical variables associated with multi-level factors such as drought declaration or season. Coefficient β_i is a weight that indicates how U_t responds to a change in x_t. Only significant β_i coefficients were considered ($P < 0.05$). Regression models were checked for statistical adequacy by examining the normality (Lilliefors test), independence (Durbin–Watson test) and heteroscedasticity (standardised quantile plots) of the residuals.

3. Results

3.1. Temporal trends in freshwater flow

Monthly rainfall and freshwater flow were positively correlated in the Clarence ($r^2 = 0.331$, $P < 0.0001$, $n = 120$), Hunter ($r^2 = 0.259$, $P < 0.0001$, $n = 120$) and Hawkesbury ($r^2 = 0.316$, $P < 0.0001$, $n = 120$) River systems from 1997 to 2007. There were distinct seasonal and inter-annual trends in freshwater flow (Fig. 2). Most freshwater flow in the Clarence River corresponded with summer ($r^2 = 0.571$, $P < 0.05$, $n = 10$) and autumn ($r^2 = 0.422$, $P < 0.05$, $n = 10$) rainfall. This seasonal periodicity was confirmed by ANOVA, with a significant difference between high flows in summer and autumn periods and low flows in winter and spring periods ($P < 0.05$). Freshwater flow in the Hunter and Hawkesbury Rivers was relatively aseasonal with no significant difference between flows in winter, spring, summer or autumn. Sustained summer and autumn peak flows occurred in all river systems from 1999 to 2001. Relatively low summer and autumn flows occurred from 2003 and 2005.

Annual variability in freshwater flow was significantly higher in the Clarence River compared to the Hunter and Hawkesbury Rivers from 1997 to 2007 (ANOVA, $P < 0.05$). Mean annual flow in the Clarence River was 2.00 million ML with minimum and maximum
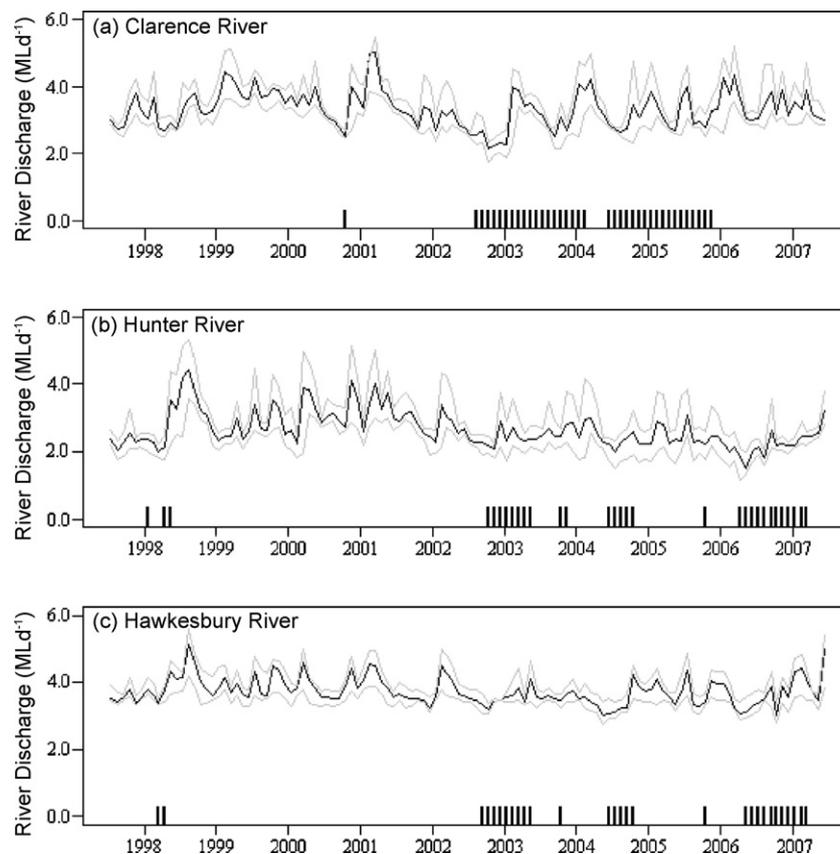

Fig. 2. Temporal trends in freshwater flow for the Clarence (a), Hunter (b) and Hawkesbury (c) River systems from July 1997 to June 2007. $\log_{10}$ monthly minimum (lower grey line), mean (middle black line) and maximum (upper grey line) flow. Vertical black lines indicate periods of drought declaration.

annual flows of 0.50 million ML and 6.96 million ML, respectively. Highest annual flows occurred in 2001 and the lowest in 2002. Mean annual flow in the Hunter River was 0.50 million ML with minimum and maximum annual flows of 0.09 million ML and 1.67 million ML, respectively. Total monthly flow decreased by 10% from 1997 to 2007 ($r^2 = 0.098$, $P < 0.0001$, $n = 120$). Highest annual flows occurred in 2007 and the lowest in 2005. Mean annual flow in the Hawkesbury River was 0.70 million ML with minimum and maximum annual flows of 0.25 million ML and 1.76 million ML, respectively. Highest annual flows were in 1998 and the lowest in 2002.

Monthly rainfall was significantly lower in the Clarence ($F = 7.36$, df = 119, $P < 0.001$), Hunter ($F = 12.81$, df = 119, $P < 0.001$) and Hawkesbury ($F = 12.54$, df = 119, $P < 0.001$) Rivers during periods of drought declaration from 1997 to 2007. Total monthly freshwater flow also exhibited significant reductions in the Clarence ($F = 13.96$, df = 119, $P < 0.0001$), Hunter ($F = 18.92$, df = 119, $P < 0.0001$) and Hawkesbury ($F = 21.16$, df = 119, $P < 0.0001$) Rivers during periods of drought declaration. The nine-month period from September 2002 to May 2003 was characterised by prolonged drought, with flow magnitudes that were frequently below the long-term (ten years) monthly means and considerably less than the Clarence (−49%), Hunter (−84%) and Hawkesbury (−65%) River estuaries usually receive.

3.2. Relationships between drought declaration and CPUE

Monthly CPUE was significantly different in all nine estuaries during drought declared and undeclared periods from 1997 to 2007 (Fig. 3). Not only was monthly CPUE significantly lower during periods of drought but CPUE also exhibited less variability around the median in the Clarence River, Hunter River, Hawkesbury River, Camden Haven River, Wallis Lake, Port Stephens and Myall Lakes, and St. Georges Basin ($P < 0.05$). Contrasting relationships were identified in Tuggerah Lakes and Lake Illawarra, with significantly higher monthly CPUE during periods of drought ($P < 0.05$). There was considerable deviation from the long-term (ten years) monthly mean CPUE during periods of drought. Negative deviation ranged from a minimum of −11% in the Hunter River to a maximum of −34% in the Camden Haven River, while positive deviation ranged from a minimum of +9% in Tuggerah Lakes to a maximum of +22% in Lake Illawara.
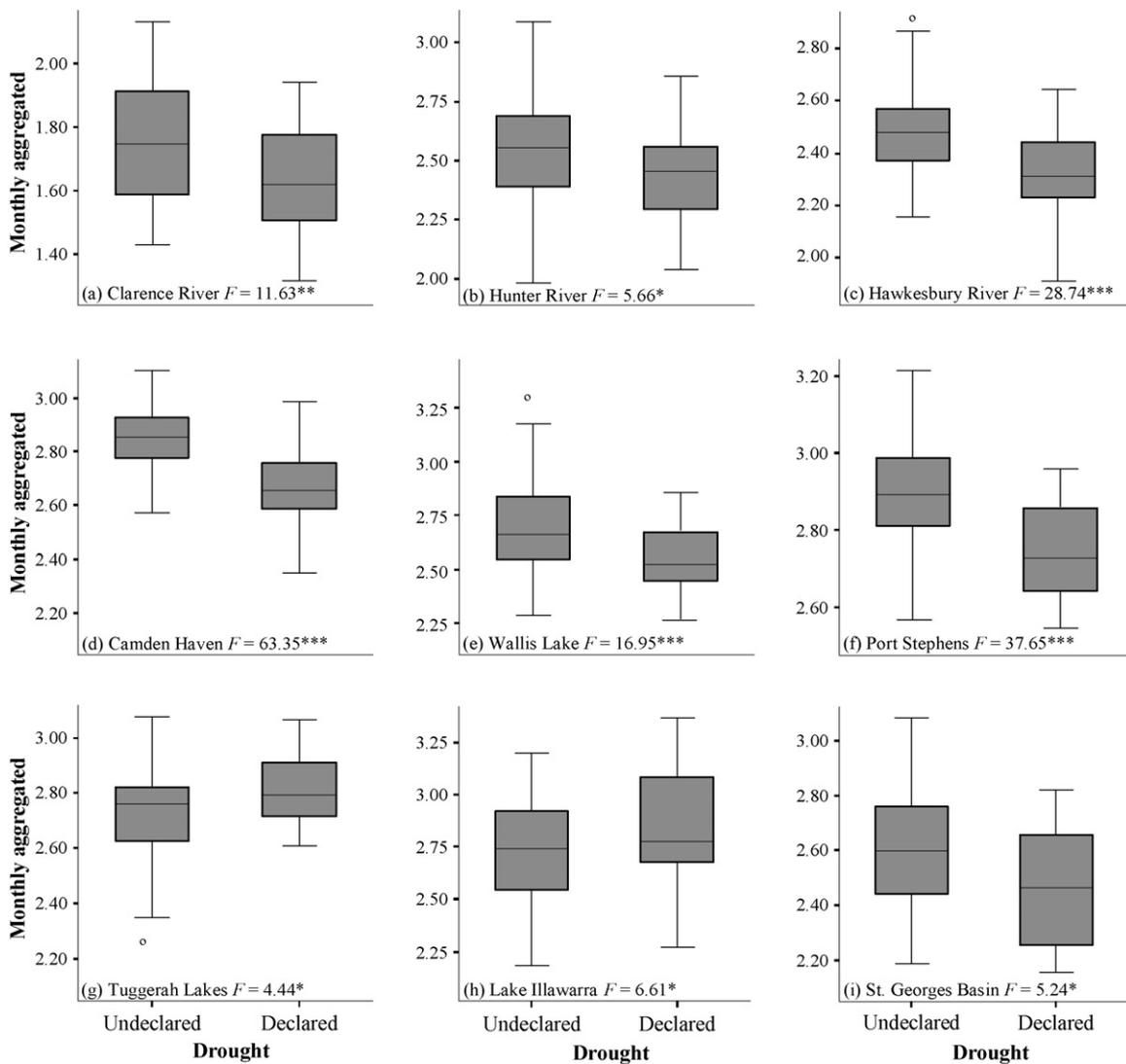

Fig. 3. Box and whisker plots illustrating significant differences in $\log_{10}$ monthly aggregated CPUE for nine estuaries in eastern Australia during drought declared and undeclared periods from 1997 to 2007 (one-way ANOVA; *$P < 0.05$; **$P < 0.01$; ***$P < 0.001$; df = 119; minimum, 25% quartile; median, 75% quartile and maximum ranges). Monthly aggregated refers to the $\sum$ Catch/$\sum$ Effort for all species combined (bream, dusky flathead, luderick, sand whiting and sea mullet). Monthly periods of drought declaration for the Clarence River were 38 months out of a total of 120 months analysed, Hunter River 31/120, Hawkesbury River 29/120, Camden Haven 30/120, Wallis Lake 32/120, Port Stephens and Myall Lakes 32/120, Tuggerah Lakes 31/120, Lake Illawarra 35/120 and St. Georges Basin 8/58.

Species-specific differences in monthly CPUE were identified during drought declared and undeclared periods from 1997 to 2007 (Fig. 4). CPUE was significantly lower for dusky flathead, luderick and sea mullet during periods of drought in the Clarence River ($P < 0.05$). Similar relationships were identified in the Hunter River with significantly lower monthly CPUE for dusky flathead, luderick, sand whiting and sea mullet under an equivalent scenario ($P < 0.05$). Monthly CPUE of sea mullet was significantly lower in the Hawkes-
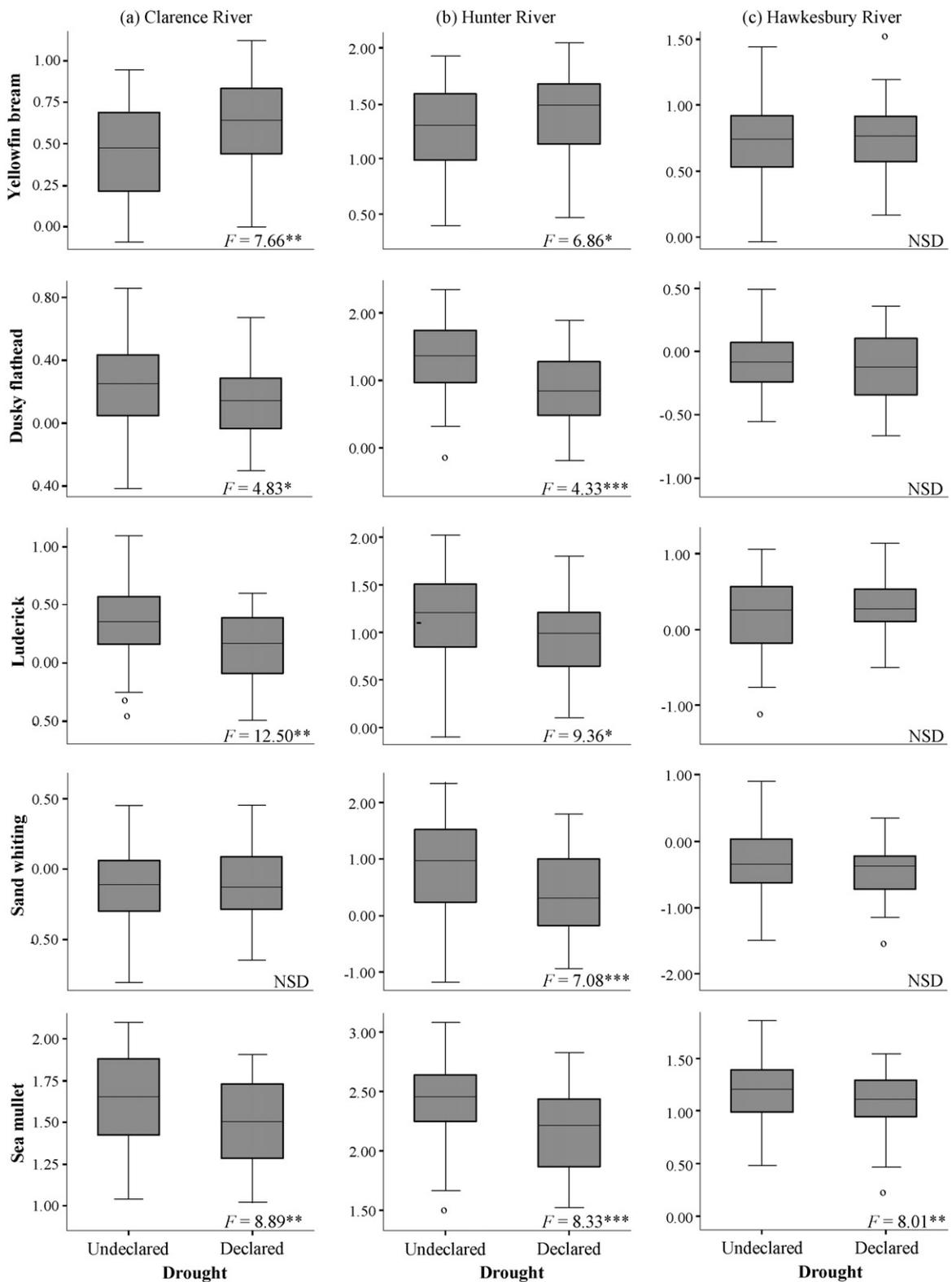

Fig. 4. Box and whisker plots illustrating species-specific differences in $\log_{10}$ monthly CPUE for the Clarence (a), Hunter (b) and Hawkesbury (c) River estuaries during drought declared and undeclared periods from 1997 to 2007 (one-way ANOVA; *$P < 0.05$; **$P < 0.01$; ***$P < 0.001$; df = 119; minimum, 25% quartile; median, 75% quartile and maximum ranges). Monthly periods of drought declaration for the Clarence River were 38 months out of a total of 120 months analysed, Hunter River 31/120 and Hawkesbury River 29/120. NSD indicates no significant difference ($P > 0.05$) between monthly CPUE during drought declared and undeclared periods.
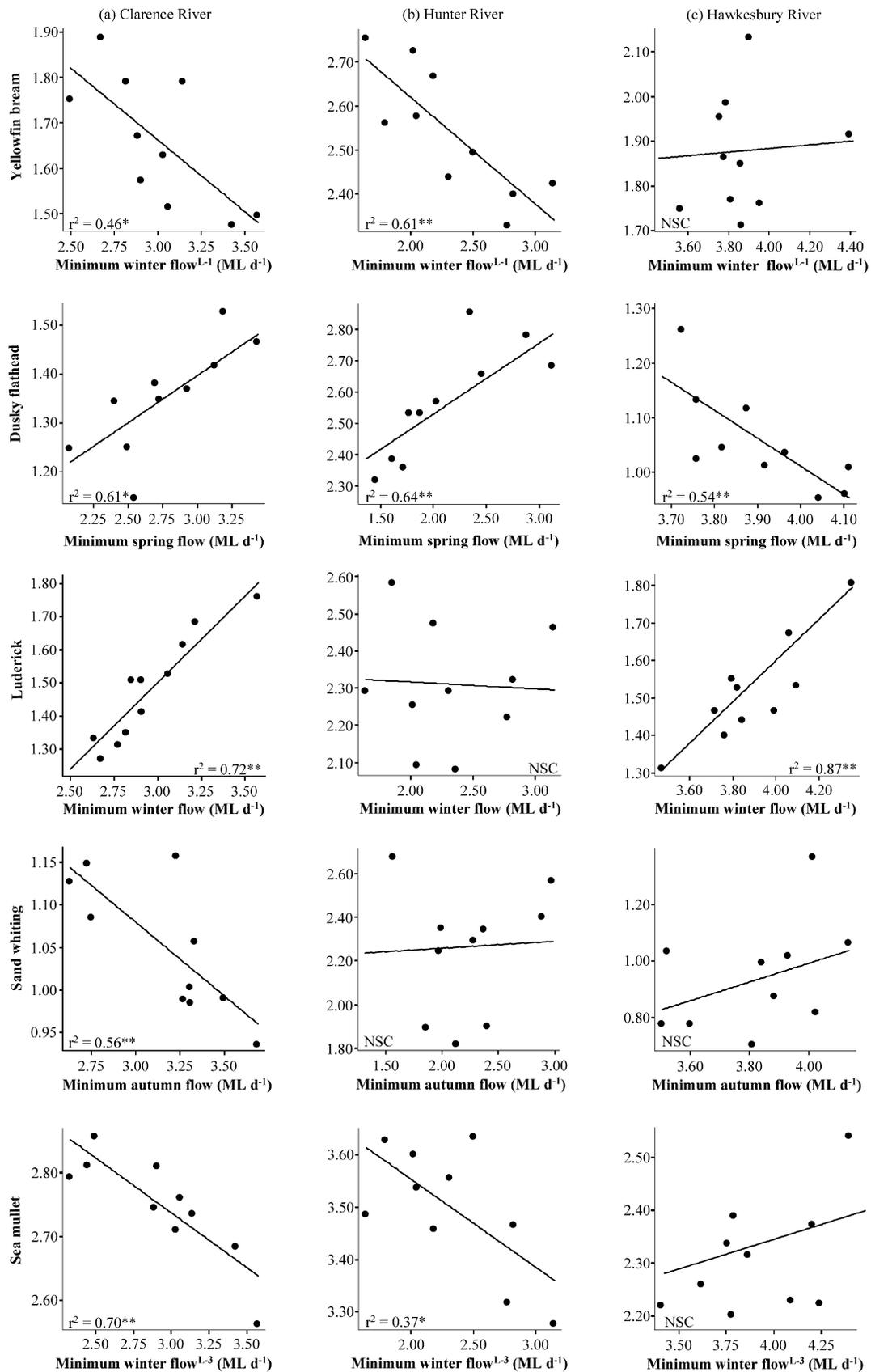
Fig. 5. Examples of relationships between $\log_{10}$ annual CPUE and minimum seasonal flow variables in the Clarence (a), Hunter (b) and Hawkesbury (c) Rivers. $^{*}P < 0.05$; $^{**}P < 0.01$; $n = 10$. Minimum flow indicates the lowest flow per season. A lag period of one and three years is represented by L − 1 and L − 3, respectively. NSC indicates non-significant correlations ($P > 0.05$).

Table 3
Statistically significant linear regressions (r^2) for the association between monthly CPUE and monthly hydrological variables in the Clarence (CR), Hunter (HU) and Hawkesbury (HK) River estuaries.

Estuary	Species	Rainfall	Total flow	Min. flow	Mean flow	Max. flow
CR	Bream	−0.040[*]	−0.038[*]	−0.041[*]	−0.039[*]	−0.037[*]
HU	Bream	–	–	–	–	–
HK	Bream	−0.050[**]	−0.029[*]	−0.052[**]	−0.049[**]	−0.042[*]
CR	Dusky flathead	–	–	–	–	–
HU	Dusky flathead	–	0.033[*]	0.147[***]	–	–
HK	Dusky flathead	–	–	–	–	–
CR	Luderick	0.092[**]	–	0.113[***]	–	–
HU	Luderick	0.025[*]	–	–	–	–
HK	Luderick	0.060[**]	–	0.028[*]	0.021[*]	0.024[*]
CR	Sand whiting	–	–	–	–	–
HU	Sand whiting	–	0.045[*]	0.046[*]	0.044[*]	0.037[*]
HK	Sand whiting	–	–	–	–	–
CR	Sea mullet	0.108[***]	0.150[***]	0.151[***]	0.114[***]	0.125[***]
HU	Sea mullet	–	–	–	–	–
HK	Sea mullet	0.035[*]	0.098[**]	0.118[***]	0.028[*]	0.037[*]
CR	All species	0.046[*]	0.171[***]	0.184[***]	0.123[***]	0.171[***]
HU	All species	0.031[*]	0.015[*]	0.129[***]	0.015[*]	0.020[*]
HK	All species	0.044[*]	0.029[*]	0.142[***]	0.051[**]	0.065[**]

Regression details provided for $\log_{10}$ transformed data; $n = 120$. Non-independent residuals (Durbin–Watson test) did not account for observed relationships between monthly fisheries CPUE and hydrological variables. All species refers to the $\sum$Catch/$\sum$Effort for bream, dusky flathead, luderick, sand whiting and sea mullet combined. Min. and max. flow represent the lowest and highest flows per month, respectively. Note that 1 ML d^{-1} equals ~0.0116 m^3 s^{-1}.

[*] $P < 0.05$.
[**] $P < 0.01$.
[***] $P < 0.001$.

bury River during periods of drought ($P < 0.05$). Yellowfin bream was the only species that demonstrated higher monthly CPUE (+20%) and higher variability around the median during periods of drought in the Clarence and Hunter Rivers ($P < 0.05$).

3.3. Relationships between freshwater flow and CPUE

Numerous significant relationships between CPUE and hydrological variables were identified in the Clarence, Hunter and Hawkesbury River estuaries from 1997 to 2007 (see Fig. 5, Tables 3–5). Forward stepwise multiple regression indicated that the best predictors of variation in monthly CPUE were drought declaration, monthly rainfall, minimum monthly flow and maximum monthly flow (Table 4). Other hydrological variables were relatively less important in explaining variation in monthly CPUE. Non-significant ($P > 0.05$) or statistically inadequate regression models (autocorrelation or non-normal residuals) relating monthly CPUE and freshwater flow did not account for observed relationships. After considering the variation in CPUE accounted for by month, freshwater flow explained from 2% to 17% of the variation in the residuals for the CPUE and month relationship ($P < 0.05$). Freshwater flow and residuals were significantly correlated, indicating that, even after removing temporal effects CPUE was higher during months of higher flow.

3.3.1. Yellowfin bream
Monthly CPUE of yellowfin bream was negatively correlated with monthly rainfall, total monthly flow, minimum monthly flow,

Table 4
Significant stepwise multiple regression models for monthly fisheries CPUE and hydrological variables and temporal components for the Clarence (CR), Hunter (HU) and Hawkesbury (HK) River estuaries.

Estuary	Species	Model	r^2
CR	Bream	Year + season + drought + rainfall (mm)	0.344[***]
HU	Bream	Drought	0.021[**]
HK	Bream	Min. flow (ML d^{-1}) + drought	0.245[***]
CR	Dusky flathead	Season + drought	0.158[***]
HU	Dusky flathead	Min. flow (ML d^{-1}) + drought	0.208[***]
CR	Luckerick	Rainfall (mm) + min. flow (ML d^{-1}) + season	0.321[***]
HU	Luckerick	Drought + season + month	0.162[***]
HK	Luderick	Season + month + rainfall (mm)	0.209[***]
CR	Sand whiting	Season	0.155[***]
HU	Sand whiting	Season + drought + year + max. flow (ML d^{-1})	0.364[***]
HK	Sand whiting	Season + year + min. flow (ML d^{-1})	0.123[*]
CR	Sea mullet	Season + min. flow (ML d^{-1}) + rainfall (mm)	0.408[***]
HU	Sea mullet	Drought + season	0.218[***]
HK	Sea mullet	Min. flow (ML d^{-1}) + total flow (ML) + year	0.192[***]
CR	All species	Season + min. flow (ML d^{-1})	0.460[***]
HU	All species	Min. flow (ML d^{-1}) + season	0.183[***]
HK	All species	Min. flow (ML d^{-1}) + drought	0.258[***]

Models with the highest correlation coefficients presented for $\log_{10}$ transformed data; $n = 120$. Fisheries parameters and hydrological variables were from monthly time series. Interaction terms between hydrological variables were not fitted. All species refers to the $\sum$Catch/$\sum$Effort for all species combined (bream, dusky flathead, luderick, sand whiting and sea mullet). Min. and max. flow indicate the lowest and highest flows per month, respectively. Drought denotes the drought declared status for an area surrounding each estuary. Season represents winter, spring, summer and autumn.

[*] $P < 0.05$.
[**] $P < 0.01$.
[***] $P < 0.001$.

Table 5
Statistically significant linear regressions (r^2) for the association between annual CPUE and seasonal hydrological variables in the Clarence (CR), Hunter (HU) and Hawkesbury (HK) River estuaries.

Estuary	Species	Season	Rainfall	% flow	Total flow	Min. flow	Mean flow	Max. flow
CR	Bream	Winter	–	0.513*	–	−0.457* (L − 1)	–	–
HU	Bream	Winter	–	0.478*	–	−0.614** (L − 1)	–	–
HK	Bream	Winter	–	0.499*	–	–	–	–
CR	Dusky flathead	Spring	0.593**	–	0.503*	0.608**	0.305*	0.487*
	Dusky flathead	Spring	–	–	−0.446** (L − 3)	−0.362* (L − 3)	−0.447* (L − 3)	–
HU	Dusky flathead	Spring	0.343*	–	0.812***	0.636**	0.801***	0.718**
HK	Dusky flathead	Spring	−0.422*	–	−0.580**	−0.543**	−0.507*	−0.325*
CR	Luderick	Winter	–	–	–	0.865**	–	–
HU	Luderick	Winter	–	−0.352*	–	–	–	–
HK	Luderick	Winter	–	–	–	0.721**	–	–
CR	Sand whiting	Autumn	−0.613**	–	−0.481*	−0.555**	–	−0.472*
	Sand whiting	Spring	–	–	−0.456* (L − 3)	−0.334* (L − 3)	−0.455* (L − 3)	−0.532** (L − 3)
HU	Sand whiting	Autumn	–	–	–	–	–	–
HK	Sand whiting	Autumn	–	–	–	–	–	–
CR	Sea mullet	Winter	–	–	−0.752** (L − 3)	−0.695** (L − 3)	−0.715** (L − 3)	−0.753** (L − 3)
HU	Sea mullet	Winter	–	–	−0.408* (L − 3)	−0.370* (L − 3)	−0.282* (L − 3)	−0.409* (L − 3)
HK	Sea mullet	Winter	–	–	–	–	–	–

Regression details provided for $\log_{10}$ transformed data; $n = 10$. Non-independent residuals (Durbin–Watson test) did not account for observed relationships between annual fisheries CPUE and seasonal hydrological variables. Min. and max. flow represent the lowest and highest flows per season, respectively. % flow refers to total seasonal flow as a percentage of total annual flow. A lag period of one and three years is indicated by (L − 1) and (L − 3), respectively. Note that 1 ML d^{-1} equals ~0.0116 m^3 s^{-1}.

* $P < 0.05$.
** $P < 0.01$.
*** $P < 0.001$.

mean monthly flow and maximum monthly flow in the Clarence and Hawkesbury Rivers (Table 3). Stepwise multiple regression identified five alternative models that explained between 2% and 34% of the variation in the monthly CPUE of yellowfin bream (Table 4). The most parsimonious models that contained a hydrological variable were year, season and drought declaration in the Clarence River ($r^2 = 0.302$, $P < 0.001$, $n = 120$); drought declaration in the Hunter River ($r^2 = 0.021$, $P < 0.01$, $n = 120$) and minimum flow in the Hawkesbury River ($r^2 = 0.197$, $P < 0.001$, $n = 120$). Annual CPUE of yellowfin bream was positively correlated with % total winter flow in the Clarence, Hunter and Hawkesbury Rivers (Table 5). Negative correlations between the annual CPUE of yellowfin bream and minimum winter flow L − 1 were identified in the Clarence and Hunter Rivers (Fig. 5).

3.3.2. Dusky flathead

Monthly CPUE of dusky flathead was positively correlated with total monthly flow and minimum monthly flow in the Hunter River (Table 3). No significant correlations between the monthly CPUE of dusky flathead and flow variables were identified in the Clarence and Hawkesbury Rivers. Stepwise multiple regression provided three alternative models that explained from 16% to 21% of the variation in the monthly CPUE of dusky flathead (Table 4). Hydrological factors that explained the highest proportion of variability in the monthly CPUE of dusky flathead were drought declaration ($r^2 = 0.058$, $P < 0.001$, $n = 120$) and minimum flow ($r^2 = 0.147$, $P < 0.001$, $n = 120$) in the Clarence and Hunter Rivers, respectively. No significant relationships between the monthly CPUE of dusky flathead and hydrological variables were identified in the Hawkesbury River. Annual CPUE of dusky flathead was positively correlated with minimum annual flow in the Clarence ($r^2 = 0.591$, $P < 0.01$, $n = 10$), Hunter ($r^2 = 0.532$, $P < 0.01$, $n = 10$) and Hawkesbury ($r^2 = 0.637$, $P < 0.01$, $n = 10$) Rivers. Positive correlations between the annual CPUE of dusky flathead and total spring rainfall, total spring flow, minimum spring flow, mean spring flow and maximum spring flow were identified in the Clarence and Hunter Rivers (Table 5). Negative correlations between the annual CPUE of dusky flathead and total spring rainfall, total spring flow, minimum spring flow, mean spring flow and maximum spring flow were identified in the Hawkesbury River (Table 5). Annual CPUE of dusky flathead was negatively correlated with total spring flow

L − 3, mean spring flow L − 3 and minimum spring flow L − 3 in the Clarence River ($P < 0.05$).

3.3.3. Luderick

Monthly CPUE of luderick was positively correlated with monthly rainfall, minimum monthly flow, mean monthly flow and maximum monthly flow (Table 3). Stepwise multiple regression identified six alternative models that explained from 16% to 32% of the variation in the monthly CPUE of luderick (Table 4). Primary factors that explained the highest proportion of variability in the monthly CPUE of luderick were rainfall in the Clarence River ($r^2 = 0.202$, $P < 0.001$, $n = 120$); drought declaration in the Hunter River ($r^2 = 0.091$, $P < 0.001$, $n = 120$); and season, month and rainfall in the Hawkesbury River ($r^2 = 0.209$, $P < 0.001$, $n = 120$). Annual CPUE of luderick was positively correlated with total annual rainfall, total annual flow, mean annual flow and maximum annual flow in the Hawkesbury River (all $P < 0.05$). Positive correlations between the annual CPUE of luderick and minimum winter flow were identified in the Clarence and Hawkesbury Rivers (Fig. 5). There was a negative correlation between the annual CPUE of luderick and % total winter flow in the Hunter River (Table 5).

3.3.4. Sand whiting

Monthly CPUE of sand whiting was positively correlated with total monthly flow, minimum monthly flow, mean monthly flow and maximum monthly flow in the Hunter River (Table 3). No significant correlations between the monthly CPUE of sand whiting and hydrological variables were identified in the Clarence and Hawkesbury Rivers. Stepwise multiple regression provided five alternative models that explained between 12% and 36% of the variation in the monthly CPUE of sand whiting (Table 4). The most parsimonious models were season in the Clarence River ($r^2 = 0.155$, $P < 0.001$, $n = 120$); season and drought declaration in the Hunter River ($r^2 = 0.267$, $P < 0.001$, $n = 120$); and season, year and minimum flow in the Hawkesbury River ($r^2 = 0.123$, $P < 0.005$, $n = 120$). Annual CPUE of sand whiting was negatively correlated with total autumn rainfall, total autumn flow, minimum autumn flow, maximum autumn flow, total spring flow L − 3, minimum spring flow L − 3, mean spring flow L − 3 and maximum spring flow L − 3 in the Clarence River (Table 5).

3.3.5. Sea mullet

Monthly CPUE of sea mullet was positively correlated with monthly rainfall, total monthly flow, minimum monthly flow, mean monthly flow and maximum monthly flow in the Clarence and Hawkesbury Rivers (Table 3). Stepwise multiple regression identified six alternative models that explained between 12% and 41% of the variation in the monthly CPUE of sea mullet (Table 4). The most parsimonious models were season and minimum flow in the Clarence River ($r^2 = 0.364$, $P < 0.001$, $n = 120$); drought declaration in the Hunter River ($r^2 = 0.163$, $P < 0.001$, $n = 120$); and minimum flow in the Hawkesbury River ($r^2 = 0.118$, $P < 0.001$, $n = 120$). Annual CPUE of sea mullet was negatively correlated with total winter flow $L - 3$, minimum winter flow $L - 3$, mean winter flow $L - 3$ and maximum winter flow $L - 3$ in the Clarence and Hunter Rivers (Table 5).

3.4. Monthly CPUE summed for all species

Monthly CPUE summed for all species was positively correlated with monthly rainfall, total monthly flow, minimum monthly flow, mean monthly flow and maximum monthly flow (Table 3). Stepwise multiple regression provided three alternative models that explained between 18% and 46% of the variation in monthly CPUE summed for all species (Table 4). The most parsimonious models were season and minimum flow in the Clarence River ($r^2 = 0.460$, $P < 0.001$, $n = 120$); minimum flow in the Hunter River ($r^2 = 0.123$, $P < 0.001$, $n = 120$); and minimum flow in the Hawkesbury River ($r^2 = 0.162$, $P < 0.001$, $n = 120$). No significant correlations between CPUE summed for all species and annual or seasonal hydrological variables were identified (Table 5).

4. Discussion

Commercial fisheries and hydrological datasets from nine estuaries with a range of freshwater inputs revealed the effects of river flow on fisheries CPUE. Catch rate/flow relationships were species, season and estuary specific. Freshwater enhancement of fisheries production was evident (see also Loneragan and Bunn, 1999; Grimes, 2001; Robins et al., 2005), with increased CPUE in months with higher flow. Monthly fisheries–flow relationships were often highly significant ($P < 0.01$), but yielded relatively low correlation coefficients. Once seasonality in river discharge was factored into linear regression analysis, freshwater flow explained a higher proportion (28–87%) of variability in annual CPUE (Table 5). Some of the variability underlying relationships between freshwater flow and CPUE may be related to factors such as bioregion (Pease, 1999), estuary type (Saintilan, 2004), degree of freshwater regulation within the catchment (Drinkwater and Frank, 1994) and the life history of individual species (Robins et al., 2005).

Anthropogenic modification of freshwater flow often forces estuarine processes to deviate from natural successionary patterns (Whitfield, 2005). Our results suggest that river regulation dampens hydrological extremes, decouples fisheries–flow relationships and hinders identification of important aspects of the flow regime that affect CPUE. Correlation coefficients (r^2) between rainfall, freshwater flow and CPUE were consistently higher for the less regulated Clarence River than the highly regulated Hunter or Hawkesbury Rivers (Tables 3–5). Significant relationships between the annual CPUE of dusky flathead and spring flow variables in the Hawkesbury River produced contrasting regression slopes compared to the Clarence and Hunter Rivers (Fig. 5). Divergent fisheries–flow relationships may be attributed to freshwater regulation altering connections between the delivery of seasonal flows and the timing of recurring lifecycle events in estuarine-dependent fish (Drinkwater and Frank, 1994). Dusky flathead spawn in estuaries and coastal waters, during spring and summer (Kailola et al., 1993). Freshwater regulation may alter synchronisation between spring

flows and the reproductive movements of dusky flathead in the Hawkesbury River. Fisheries and environmental attributes have a similar ordination in these river systems (Pease, 1999). Thus, differences in river geomorphology were unlikely to account for observed disparity in fisheries–flow relationships. Determining the freshwater flow requirements of estuarine fisheries is likely to be more difficult in highly regulated river systems, where variability in the flow regime is managed for human requirements.

4.1. Impacts of drought on CPUE

Drought declaration in catchments associated with the river-dominated estuaries corresponded with lower rainfall, freshwater flow and CPUE. Significant reductions in monthly CPUE summed for all species during periods of drought were identified in all examined estuaries, except Tuggerah Lakes and Lake Illawarra. Drought may lower fisheries production by reducing food availability (Bennett et al., 1995), deteriorating water quality (Attrill and Power, 2000), forcing seaward migration of estuarine residents (Dolbeth et al., 2008), or by increasing predation pressure from marine species (Martinho et al., 2007). Tuggerah Lakes and Lake Illawarra are shallow (≤ 3 m) poorly flushed coastal barrier lagoons with a limited tidal range (0.1–1.0 m) and relatively low freshwater inputs. Hydrodynamics are largely controlled by runoff from highly urbanised catchments in these estuaries (Scanes et al., 2007). Drought may result in higher fisheries catch rates in freshwater deprived estuaries due to increased catchability resulting from short-term improvements in water quality. Lower rainfall reduces runoff limiting concentrations of land-based pollutants leached into running waters (Niemczynowicz, 1999). Drought-induced low flows may lower fisheries catch rates in river-dominated estuaries owing to unfavourable water quality characteristics forcing the emigration of estuarine-dependent fish into coastal waters. Reductions in freshwater flow can alter the abundance and distribution of estuarine fish communities by increasing salinity and decreasing concentrations of dissolved oxygen (Sklar and Browder, 1998). Fish actively avoid drought associated conditions by using behavioural adaptations to limit physiological costs associated with unfavourable water quality characteristics (Magoulick and Kobza, 2003).

Species-specific reductions in monthly CPUE were evident for dusky flathead, luderick, sand whiting and sea mullet during periods of drought. Sea mullet dominated estuarine finfish harvest ($\geq 65\%$) providing the greatest contribution to significant differences in monthly aggregated CPUE between drought declared and undeclared periods. Yellowfin bream was the only species with significantly higher monthly CPUE during periods of drought. Yellowfin bream complete their lifecycle within estuarine and inshore coastal waters (Blaber and Blaber, 1980), predominately inhabiting marine and brackish regions but can also penetrate the inter-tidal freshwater reaches of coastal rivers (West and King, 1996). Drought may result in the increased catchability of yellowfin bream owing to higher salinity waters stimulating seaward migration. Estuarine fishers suggest that yellowfin bream migrate downstream into the estuary mouth during periods of low flow, thereby increasing their catchability by passive fishing gears, such as gillnets.

4.2. Impacts of freshwater flow on CPUE

High freshwater flow results in increased catchability by restricting distribution or stimulating movement into areas where estuarine-dependent fish are more readily caught (Loneragan and Bunn, 1999). Comparison of positive slope values with stepwise multiple regression suggested that the catch rates of sea mullet were highly responsive to flow variability (Tables 3 and 4). Salinity strongly influences habitat selection by sea mullet with individuals

actively seeking optimum habitat conditions to minimise osmoregulatory costs and maximise growth rates (Cardona, 2000, 2006; Chang et al., 2004). Freshwater flow may have a marked influence on the catch rates of sea mullet due to salinity fluctuations stimulating migration into preferred habitat.

Increased freshwater flow lowers salinity altering habitat availability for estuarine-dependent fish (Jassby et al., 1995). Yellowfin bream may respond negatively to high flows due to the seaward displacement of preferred habitat. Yellowfin bream and black bream are not conspecific, and have markedly different life history characteristics (Roberts et al., 2009). Nevertheless, similar responses have been reported for black bream in southeastern Australia, where considerable numbers of fish left the Hopkins River in Victoria for sheltered coastal habitats after the salt wedge was flushed seaward by heavy freshwater discharge (Sherwood and Backhouse, 1982). Black bream can be flushed out to sea during extreme flow events but return to the natal estuary once the rate of freshwater discharge declines (Chaplin et al., 1998). Freshwater flow may regulate fisheries catch rates by altering the spatial distribution of estuarine-dependent fish due to salinity fluctuations stimulating migration and schooling.

Minimum and maximum flows were important determinants of CPUE (Table 3). Freshwater flow *per se* may not be as important in determining estuarine fisheries production as extremes in the hydrological continuum. Episodic flow events are natural perturbations that maintain biological productivity in estuaries (Whitfield, 2005). Protecting natural fluxes in the flow regime may represent the most reliable management strategy to maintain fisheries production in Australian estuaries. Management of environmental flows has evolved from solely concentrating on the protection of minimum flows to recognition of the importance of episodic flow events for maintaining the structure and function of aquatic ecosystems (Sparks and Spink, 1998).

Aquatic species have evolved life history strategies in direct response to natural flow regimes (Bunn and Arthington, 2002). Our analyses indicated that the highest correlation coefficients (r^2) between freshwater flow and CPUE coincided with spawning periods. High seasonal flows may trigger spawning migrations in estuarine-dependent fish (Stevens and Miller, 1983). Yellowfin bream undertake annual migrations to spawn in coastal surf zones near the mouths of estuaries, typically during winter (Pollock, 1982a,b; Kailola et al., 1993). Annual CPUE of yellowfin bream was positively correlated with percentage total winter flow. Minimum winter flow explained a substantial proportion of the variation in the annual CPUE of luderick in the Clarence (87%) and Hawkesbury Rivers (72%). Luderick spawn in coastal surf zones from August to December in central and northern New South Wales (Kailola et al., 1993). The protracted spawning period of luderick corresponds with late winter, spring and early summer. High winter flow rates may result in increased catchability by triggering the seaward spawning migration of yellowfin bream and luderick.

Dusky flathead spawn in the lower reaches of estuaries and nearshore coastal waters, typically during spring and summer (Kailola et al., 1993). Positive relationships between the annual CPUE of dusky flathead and spring flow variables could, therefore, reflect increased catchability due to high flows inducing the pre-spawning migration. Sand whiting aggregate to spawn in the lower reaches of estuaries and nearshore coastal waters, typically during summer and autumn (Morton, 1985; Burchmore et al., 1988). Annual CPUE of sand whiting was negatively correlated with autumn flows in the Clarence River. Accordingly, our results suggest that seasonal freshwater pulses create windows of opportunity for estuarine-dependent fish to undertake spawning migrations when suitable hydrological conditions arise. Reproductive movements may be synchronised with seasonal environmental cues such as changes in freshwater flow, water temperature, salinity, turbidity and dissolved oxygen (Bjorgo et al., 2000; Dahl et al., 2004; Nicholson et al., 2008).

Relationships between freshwater flow and CPUE lagged by age at first maturity indicated a possible recruitment effect. Sexual maturity for dusky flathead, sand whiting and sea mullet occurs after approximately three years (Morton, 1985; Burchmore et al., 1988; Kailola et al., 1993), which coincides with negative relationships between annual CPUE and seasonal flow variables with a three-year lag in the Clarence River. Freshwater flow can enhance recruitment by increasing offshore concentrations of land-based cues that stimulate fish larvae to immigrate into estuarine nursery grounds (Vinagre et al., 2007). The results presented here, however, suggest that recruitment effects were unlikely to result from olfactory cues increasing estuarine immigration. High freshwater flows lower salinity creating a barrier to recruitment, reducing available nursery habitat and limiting the immigration of marine larvae (Loneragan and Bunn, 1999; Strydom et al., 2002; Shoji et al., 2006). Negative relationships between CPUE and lagged flows may result from lower salinity reducing estuarine immigration and forcing the larvae of marine stragglers (e.g. dusky flathead and sand whiting) to find temporary refuge in the sea. Another possible mechanism for catadromous species (e.g. sea mullet) is that high freshwater flows flush larvae out of estuaries into adjacent coastal waters. Increased freshwater flow may create unfavourable habitat conditions that force the seaward dispersion of larvae and result in subsequent reductions in CPUE.

The results from this study suggest that freshwater flow influences the catchability of estuarine-dependent fish by stimulating migration and schooling due to salinity fluctuations altering habitat availability (Loneragan and Bunn, 1999). Correlative analyses undertaken in this study were insufficient to make inferences about possible food chain effects. Freshwater flow can influence fish abundance by altering nutrient delivery and trophic conditions within an estuary (Livingston et al., 1997). It has, however, been alleged that freshwater flow is more likely to determine the abundance of estuarine organisms by altering habitat availability rather than trophic dynamics (Kimmerer, 2002). There was no evidence to suggest that high freshwater flow stimulated larval immigration into estuaries. Negative relationships between CPUE and lagged flows may be attributed to lower salinity reducing available habitat and forcing the seaward dispersion of larvae.

5. Conclusion

Estuarine-dependent fish are influenced by variation in freshwater flow in ways that affect CPUE via changes in recruitment and catchability. Freshwater flow may regulate fisheries catch rates by stimulating migration and schooling due to salinity fluctuations altering the availability of habitat. If the true effects of freshwater flow on estuarine fish communities are to be identified, future work should examine the influence of flows during critical reproductive periods. With climatic warming and greater hydrological extremes predicted in Australia (Hughes, 2003), improved knowledge of the freshwater flow requirements of estuarine fisheries is important to devise effective management strategies. Understanding how interactions between hydrology and salinity produce such a marked effect on estuarine-dependent fish is essential when considering the wider implications of altered flow regimes on estuarine fisheries.

References

Attrill, M.J., Power, M., 2000. Effects on invertebrate populations of drought-induced changes in estuarine water quality. Mar. Ecol. Prog. Ser. 203, 133–143.

Bennett, W.A., Ostrach, D.J., Hinton, D.E., 1995. Larval striped bass condition in a drought-stricken estuary: evaluating pelagic food-web limitation. Ecol. Appl. 5, 680–692.

Benson, N.G., 1981. The freshwater-inflow-to-estuaries issue. Fisheries 6, 8–10.

Bjorgo, K.A., Isely, J.J., Thomason, C.S., 2000. Seasonal movement and habitat use by striped bass in the Combahee River, South Carolina. T. Am. Fish. Soc. 129, 1281–1287.

Blaber, S.J.M., Blaber, T.G., 1980. Factors affecting the distribution of juvenile estuarine and inshore fish. J. Fish. Biol. 17, 143–162.

Bunn, S.E., Arthington, A.H., 2002. Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. Environ. Manage. 30, 492–507.

Burchmore, J.J., Pollard, D.A., Middleton, M.J., Bell, J.D., Pease, B.C., 1988. Biology of four species of whiting (Pisces: Sillaginidae) in Botany bay, New South Wales. Aust. J. Mar. Fresh. Res. 39, 709–727.

Cardona, L., 2000. Effects of salinity on the habitat selection and growth performance of Mediterranean flathead grey mullet Mugil cephalus (Osteichthyes, Mugilidae). Estuar. Coast Shelf S. 50, 727–737.

Cardona, L., 2006. Habitat selection by grey mullets (Osteichthyes: Mugilidae) in Mediterranean estuaries: the role of salinity. Sci. Mar. 70, 443–455.

Chang, C.W., Iizuka, Y., Tzeng, W.N., 2004. Migratory environmental history of the grey mullet Mugil cephalus as revealed by otolith Sr:Ca ratios. Mar. Ecol. Prog. Ser. 269, 277–288.

Chaplin, J.A., Baudains, G.A., Gill, H.S., McCulloch, R., Potter, I.C., 1998. Are assemblages of black bream (Acanthopagrus butcheri) in different estuaries genetically distinct? Int. J. Salt Lake Res. 6, 303–321.

Cooper, J.A.G., 1994. Lagoons and microtidal coasts. In: Carter, R.W.C., Woodroffe, C.S. (Eds.), Coastal Evolution. Cambridge University Press, Cambridge, pp. 219–265.

Copeland, B.J., 1966. Effects of decreased river flow on estuarine ecology. Water Pollut. Control Fed. 38, 1831–1839.

Dahl, J., Dannewitz, J., Karlsson, L., Petersson, E., Löf, A., Ragnarsson, B., 2004. The timing of spawning migration: implications of environmental variation, life history, and sex. Can. J. Zool. 82, 1864–1870.

Darnaude, A.M., Salen-Picard, C., Polunin, N.V.C., Harmelin-Vivien, M.L., 2004. Trophodynamic linkage between river runoff and coastal fishery yield elucidated by stable isotope data in the Gulf of Lions (NW Mediterranean). Oecologia 138, 325–332.

Dolbeth, M., Martinho, F., Viegas, I., Cabral, H., Pardal, M.A., 2008. Estuarine production of resident and nursery fish species: conditioning by drought events? Estuar. Coast Shelf S. 78, 51–60.

Drinkwater, K.F., Frank, K.T., 1994. Effects of river regulation and diversion on marine fish and invertebrates. Aquat. Conserv. 4, 135–151.

Erskine, W.D., Warner, R.F., 1998. Further assessment of flood- and drought-dominated regimes in south-eastern Australia. Aust. Geogr. 29, 257–261.

Finlayson, B.L., McMahon, T.A., 1988. Australia vs the world: a comparative analysis of streamflow characteristics. In: Warner, R.F.S. (Ed.), Fluvial Geomorphology of Australia. Sydney Academic Press, Sydney, pp. 17–40.

Gillanders, B.M., Kingsford, M.J., 2002. Impact of changes in flow of freshwater on estuarine and open coastal habitats and the associated organisms. Oceanogr. Mar. Biol. 40, 233–309.

Grimes, C.B., 2001. Fishery production and the Mississippi River discharge. Fisheries 26, 17–26.

Houde, E.D., 1987. Fish early life dynamics and recruitment variability. Am. Fish. Soc. Symp. 2, 17–29.

Hughes, L., 2003. Climate change and Australia: trends, projections and impacts. Austral Ecol. 28, 423–443.

Jassby, A.D., Kimmerer, W.J., Monismith, S.G., Armor, C., Cloern, J.E., Powell, T.M., Schubel, J.R., Vendlinski, T.J., 1995. Isohaline position as a habitat indicator for estuarine populations. Ecol. Appl. 5, 272–289.

Kailola, P.J., Williams, M.J., Stewart, P.C., Reichelt, R.E., McNee, A., Grieve, C., 1993. Australian Fisheries Resources. Bureau of Resource Sciences. In: Department of Primary Industries and Energy. Fisheries Research and Development Cooperation, Canberra, Australia.

Kimmerer, W.J., 2002. Effects of freshwater flow on abundance of estuarine organisms: physical effects or trophic linkages? Mar. Ecol. Prog. Ser. 243, 39–55.

Livingston, R.J., Niu, X., Lewis III, F.G., Woodsum, G.C., 1997. Freshwater input to a Gulf Estuary: long-term control of trophic organization. Ecol. Appl. 7, 277–299.

Loneragan, N.R., Bunn, S.E., 1999. River flows and estuarine ecosystems: implications for coastal fisheries from a review and a case study of the Logan River, southeast Queensland. Aust. J. Ecol. 24, 431–440.

Magoulick, D.D., Kobza, R.M., 2003. The role of refugia for fishes during drought: a review and synthesis. Freshw. Biol. 48, 1186–1198.

Martinho, F., Leitão, R., Viegas, I., Dolbeth, M., Neto, J.M., Cabral, H.N., Pardal, M.A., 2007. The influence of an extreme drought event in the fish community of a southern Europe temperate estuary. Estuar. Coast Shelf S. 75, 537–546.

Morton, R.M., 1985. The reproductive biology of summer whiting, Sillago ciliata (C. and V.), in northern Moreton Bay, Queensland. Aust. Zool. 21, 491–502.

New South Wales Department of Primary Industries, 2008. Drought Situation Maps., http://www.dpi.nsw.gov.au/drought.

Nicholson, G., Jenkins, G.P., Sherwood, J., Longmore, A., 2008. Physical environmental conditions, spawning and early-life stages of an estuarine fish: climate change implications for recruitment in intermittently open estuaries. Mar. Freshw. Res. 59, 735–749.

Niemczynowicz, J., 1999. Urban hydrology and water management—present and future challenges. Urban Water 1, 1–14.

North, E.W., Houde, E.D., 2003. Linking ETM physics, zooplankton prey and fish early-life histories to striped bass Moore saxatilis and white perch M. americana recruitment. Mar. Ecol. Prog. Ser. 260, 219–236.

Pease, B.C., 1999. A spatially oriented analysis of estuaries and their associated commercial fisheries in New South Wales, Australia. Fisheries 42, 67–86.

Pollock, B.R., 1982a. Spawning period and growth of yellowfin bream, Acanthopagrus australis (Günther), in Moreton Bay, Australia. J. Fish. Biol. 21, 349–355.

Pollock, B.R., 1982b. Movements and migrations of yellowfin bream, Acanthopagrus australis, in Moreton Bay, Queensland as determined by tag recoveries. J. Fish. Biol. 20, 245–252.

Power, S., Casey, T., Folland, C., Colman, A., Mehta, V., 1999. Inter-decadal modulation of the impact of ENSO in Australia. Clim. Dynam. 15, 319–324.

Roberts, D.G., Gray, C., West, R., Ayre, D.J., 2009. Evolutionary impacts of hybridization and interspecific gene flow on an obligatory estuarine fish. J. Evol. Biol. 22, 27–35.

Robins, J.B., Halliday, I.A., Staunton-Smith, J., Mayer, D.G., Sellin, M.J., 2005. Freshwater-flow requirements of estuarine fisheries in tropical Australia: a review of the state of knowledge and application of a suggested approach. Mar. Freshw. Res. 56, 343–360.

Roy, P.S., Williams, R.J., Jones, A.R., Yassini, I., Gibbs, P.J., Coates, B., West, R.J., Scanes, P.R., Hudson, J.P., Nichol, S., 2001. Structure and function of South-east Australian Estuaries. Estuar. Coast Shelf S. 53, 351–384.

Saintilan, N., 2004. Relationships between estuarine geomorphology, wetland extent and fish landings in New South Wales. Estuar. Coast Shelf S. 61, 591–601.

Scanes, P., Coade, G., Doherty, M., Hill, R., 2007. Evaluation of the utility of water quality based indicators of estuarine lagoon condition in NSW, Australia. Estuar. Coast Shelf S. 74, 306–319.

Scharler, U.M., Baird, D., 2005. A comparison of selected ecosystem attributes of three South African estuaries with differing freshwater inflow regimes, using network analysis. J. Mar. Syst. 56, 283–308.

Sherwood, J.E., Backhouse, G.N., 1982. Hydrodynamics of salt wedge estuaries—implications for successful spawning in black bream (Acanthopagrus butcheri). Warrnambool Institute of Advanced Education, Faculty of Applied Science and Technology, Research Report.

Shoji, J., Ohta, T., Tanaka, M., 2006. Effects of river flow on larval growth and survival of Japanese seaperch Lateolabrax japonicus (Pisces) in the Chikugo River estuary, upper Ariake Bay. J. Fish. Biol. 69, 1662–1674.

Sklar, F.H., Browder, J.A., 1998. Costal environmental impacts brought about by alterations to freshwater flow in the Gulf of Mexico. Environ. Manage. 22, 547–562.

Skreslet, S., 1986. The Role of Freshwater Outflow in Coastal Marine Ecosystems. Springer-Verlag, Berlin.

Sokal, R.R., Rohlf, J., 1995. Biometry: The principles and Practice of Statistics in Biological Research, Third edition. Freeman WH, New York, New York.

Sparks, R.E., Spink, A., 1998. Disturbance, succession and ecosystem processes in rivers and estuaries: effects of extreme hydrologic events. Regul. Rivers Res. Manage. 14, 155–159.

Stevens, D.E., Miller, L.W., 1983. Effects of river flow on abundance of young chinook salmon, American shad, longfin smelt, and delta smelt in the Sacramento-San Joaquin River system. North Am. J. Fish. Manage. 3, 425–437.

Strydom, N.A., Whitfield, A.K., Paterson, A.W., 2002. Influence of altered freshwater flow regimes on abundance of larval and juvenile Gilchristella aestuaria (Pisces: Clupeidae) in the upper reaches of two South African estuaries. Mar. Freshw. Res. 53, 431–438.

Vinagre, C., Costa, M.J., Cabral, H.N., 2007. Impact of climate and hydrodynamics on sole larval immigration towards the Tagus estuary. Portugal. Estuar. Coast Shelf S. 75, 516–524.

West, R.J., King, R.J., 1996. Marine, brackish, and freshwater fish communities in the vegetated and bare shallows of an Australian coastal river. Estuaries 19, 31–41.

West, R.J., Thorogood, C.A., Walford, T.R., Williams, R.J., 1985. An estuarine inventory for New South Wales, Australia. Fisheries Bulletin No. 2. Department of Agriculture, New South Wales.

Whitfield, A.K., 1994. Abundance of larval and 0+ juvenile marine fishes in the lower reaches of three southern African estuaries with differing freshwater inputs. Mar. Ecol. Prog. Ser. 105, 227–267.

Whitfield, A.K., 2005. Fishes and freshwater in southern African estuaries—a review. Aquat. Living Resour. 18, 275–289.

Wolanski, E., 2007. Estuarine Ecohydrology. Elsevier Science, Amsterdam, Netherlands.