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Using intervention analysis to evaluate the trends in release rates of recreational fisheries following extensive management changes

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ABSTRACT

Changes to management of a fisheries resource are often required to ensure ongoing sustainability. However, such changes can sometimes lead to unintended effects such as increased release rates and associated post-release mortality. These effects may be highly variable between species and areas. Recreational fishing management changes were introduced on the west coast of Australia in 2009/10 to recover stocks of demersal scalefish. Key changes included reducing mixed species bag limits across management zones and increasing the minimum size limit for one species in some management zones. The restrictive catch limits resulted in increased release rates of key demersal species. However, whether such increases are significant and sustained over time, and thus of management concern, have not been evaluated. We carried out intervention time series analysis to evaluate the impact of management changes on release rates of four key demersal species for the recreational sector in metropolitan and regional management zones covering ~8° latitude using an 18-year time series of charter recreational fishery data from July 2002 to January 2020. We observed varying responses in release rates by species and zones, the most common of which were a step increase, a ramp and a temporary increase that decayed. These responses may be related to targeted management changes which influenced fisher behaviour, perceived recreational value of some species and recruitment variation. Our study demonstrates that intervention analysis, which has seen limited use in this context, can assist in evaluating the impact of management changes on different species for recreational fisheries.

1. Introduction

Recreational fisheries are globally recognised as having high social, economic, and food security importance (Arlinghaus et al., 2019; Potts et al., 2020). In some multi-sector fisheries, a large proportion of the catch is taken by recreational fishers, which contributes substantially to overall fishing mortality (Radford et al., 2018). Fisheries management aims to ensure sustainable resources through implementation of regulations to prevent overfishing of stocks, or to promote recovery of overfished stocks (Garcia et al., 2018). This is typically achieved with methods such as input controls (e.g., restrictions on fishing gear or the timing and location of fishing activity) and output controls (e.g., limits on the size and number of fish that can be taken) (Morison, 2004; FAO, 2022; Potts et al., 2020). These controls are used internationally, especially in high income regions such as Australia, Canada, Europe and the United States of America (Radomski, 2011; Selig et al., 2017). In most jurisdictions, size limits on the retention of fish are promoted on the premise that under-size released fish will survive and contribute to future replenishment of the stock through subsequent growth and spawning (Woodward and Griffin, 2003). However, understanding whether such regulatory measures have any negative consequences is critical if they are to be effective in sustaining fish stocks. For example, more restrictive size and bag limits, and licensing management changes in the Canadian freshwater and marine recreational fisheries has led to high release rates. However, it was uncertain whether the increase in released catches between 1985 and 2010 was solely due to harvest regulations or conservation ethics of anglers (Brownscombe et al., 2014). More releases may result in high fishing mortality, especially for species that are susceptible to lower fitness from catch and release. If release mortality is high, then some management changes could have negative consequences for stock sustainability or recovery (Woodward and Griffin, 2003). While the total fishing mortality (including retained

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catches and post-release mortality) is ultimately the parameter of concern for fisheries management, post-release mortality is dependent on self-reported numbers and understanding the fate of released fish. For these reasons, released catches impact the accuracy of estimates of total fishing mortality because of uncertainty in the numbers, weights, lengths and proportions of fish that experience post-release mortality.

It is therefore important to understand how various fisheries management interventions influence release rates. For example, the proportion of released catch relative to the total catch (i.e. retained catch plus any releases, hereon referred to as the release rate) may decrease under some input controls, such as the mandated use of barbless or circle hooks, or effort restrictions, such as a temporal closures to fishing (Cooke and Suski, 2004). Conversely, release rates may increase under some output controls, such as decreasing bag limits or increasing size limits (Ferter et al., 2013). Another response to increased size limits is an increase in release rates for a short period of time after which fishers adjust their behaviour, by targeting different areas or species, eventually decreasing release rates (Näslund et al., 2010). Time series data are thus required pre- and post-management changes to understand the impact of such changes on release rates, but the interaction of multiple controls may confound the detection of specific effects. Such time series data are often only available from mandatory reporting or from a continuous survey program. However, the latter are typically cost-prohibitive.

Autoregressive integrated moving average (ARIMA) models are commonly used in analysing time series data in fisheries science and management (Craine, 2005; Selvaraj et al., 2020). These only require data on the time series in question and do not assume knowledge of any underlying model or structural relationships (Meyler et al., 1998; Box et al., 2015). However, instances such as policy changes, strikes, floods and price changes, can cause dramatic changes and affect the structural relationships in time series, which must be considered when modelling, but cannot be captured with ARIMA models alone. In general, there are three types of responses expected from such incidences: (i) an abrupt but permanent shift (step), (ii) a continuous increase or decrease in trend (ramp) or (iii) a temporary shift (pulse) (Box and Tiao, 1975; Matarise, 2011; Box et al., 2015). Box and Tiao (1975) proposed an intervention model of time series analysis (ARIMAX) to detect evidence for a change in the time series due to exogenous forces and to quantify the nature and magnitude of that change. These modelling procedures have been widely used, particularly for public health interventions, where "intervention" detection and estimation of impact are paramount (Bernal et al., 2017). However, there has been limited usage of intervention analysis in fisheries science (Madenjian et al., 1986; Fogarty and Miller, 2004).

The few studies that have applied intervention time series analyses to fisheries data have reported improved forecasted estimates of catch for various scenarios. For example, an intervention analysis provided important insights in assessing the impact of changes in recruitment and gear types and identifying the significance, magnitude, and form of structural shifts in the time series of trawl catches for 53 commercial species in the north-west Mediterranean Sea (Lloret et al., 2000). Intervention analysis was used to improve forecasts of the commercial catches of Western Rock Lobster *Panulirus cygnus* in Western Australia by incorporating the impact of a management change in 1993/94 and seasonality associated with migration (Craine, 2005).

Continuous time series of retained and released catches from recreational fisheries are rare and often rely on reconstructions from a variety of catch and effort data (Hyder et al., 2020). There are few published studies of intervention analysis in recreational fisheries (e.g., Cloutman and Jackson, 2003; Oh and Ditton, 2008). Pay-for-hire recreational fishing (i.e., charter fishing) often has rigorous reporting obligations on retained and released catches (Telfer, 2010; Gray and Kennelly, 2017b). In Western Australia, a formal licensing system for charter fishing was established in 2001/02. Mandatory catch reporting provides a long-term time series of retained and released catch data in the West Coast Bioregion (WCB), which extends along approximately 8° of latitude and 900 km of coastline from Kalbarri (27°S) to Augusta (115°30'E) (Wise et al., 2007). Four demersal species are particularly important to the recreational fishery in the WCB: West Australian dhufish *Glaucosoma hebraicum*, Snapper *Chrysophrys auratus*, Baldchin groper *Choerodon rubescens* and Breaksea cod *Epinephelides armatus*. Retained catches of these four species contribute 45% in 2020 to the retained catch from charter fishing in the WCB (Fairclough et al., 2021). The abundance and distribution of the key species along the latitudinal gradient of the West Coast Bioregion influence their spatial variation in catches (Ryan et al., 2022). For example, catches of *C. rubescens* are highly concentrated in the northern part of the West Coast Bioregion where it is most abundant (Hutchins, 2001).

Following an assessment of the West Coast Demersal Scalefish Resource in 2007 that demonstrated overfishing of key species (Wise et al., 2007), a recovery program was introduced between November 2007 and March 2010. The recovery program included policies to reduce retained catches of demersal species by the commercial and recreational sectors to no more than 50% of 2005/06 levels (see Table 1 for changes specific to the recreational sector). Most changes for the recreational sector were implemented in 2009 and early 2010. Recreational fishing regulations to support the recovery program included reductions to daily bag limits, increased minimum size limits for *C. auratus* in some management zones, a boat limit for *G. hebraicum*, an annual 2-month closure to fishing for demersal species in the WCB, and a statewide Recreational Boat Fishing Licence to provide access entitlement (Department of Fisheries Western Australia, 2010). Some of these changes had the potential to cause unintended consequences. For example, in addition to high retained catches, the resultant high release rates for species such as *C. auratus* and *G. hebraicum*, along with uncertainty regarding the extent of post-release mortality, poses an additional risk to their sustainability (DPIRD, 2021; Fairclough et al., 2021). It has been over 10 years since these changes were implemented and over 18 years since implementation of charter logbooks, which provides a suitable time series to quantify the impact of management changes on released catches.

In this study, we use intervention time series analysis to evaluate the impact of management changes on release rates of *G. hebraicum*, *C. auratus*, *C. rubescens* and *E. armatus* by charter fishers across different spatial management zones of the WCB. This case study uses 18 years of charter logbook catch and effort data from the West Coast Bioregion. The findings from the study would provide insights into the potential impact and estimation of post-release mortality in recreational fisheries at a time where management regulations are becoming increasingly important to ensure the sustainability of the fishery.

2. Methodology

2.1. Study area and data

This study used data from the Kalbarri, Mid-west and Metropolitan

Table 1
Key management changes for boat-based recreational fishing in the Kalbarri, Mid-west and Metropolitan zones of the West Coast Bioregion.

Year	Major recreational fishing rules changes
Early 2009	Halved daily bag limits for snapper (from 4 to 2). Increased minimum size limit applied south of 31° S for snapper from 41 to 45 cm. Four months closure to Snapper fishing from 1 October to 31 January in Cockburn and Warnbro Sounds.
Late 2009	Halved the daily bag limit for West Australian dhufish to 1, Baldchin groper, Breaksea cod and mixed demersal species to 2. A boat limit of two dhufish for private recreational fishers and six for charter fishers. Increased minimum size limit for snapper from 45 to 50 cm south of 31° S. Two months closure from October 15 to December 15 in the WCB to fishing for demersal species.
March 2010	Recreational Fishing from Boat licence introduced

management zones of the West Coast Bioregion, where most of the charter fishing in Western Australia occurs. Data for the Kalbarri and Mid-west zones were combined (hereon referred to as the Mid-west zone) due to the small number of charter operators in the Kalbarri zone and the comparable biology and management arrangements for recreationally caught species. The majority of recreational fishing in the West Coast Bioregion occurs in waters less than 250 m deep (Fairclough et al., 2021; Fig. 1).

Catches from charter fishing in Western Australia, expressed as the number of fish retained and released by species, have been recorded in mandatory daily logbooks (Telfer, 2010; DPIRD, 2021). Targeting of species is not recorded in charter logbooks and fishing effort is described by the number of vessels (ranging from 1 to 21 per month), the number of clients (ranging from 1 to 53 per day) and fishing duration (ranging from 1 to 20 h per day).

Release rates (*rr*) of *G. hebraicum*, *C. auratus*, *C. rubescens* and *E. armatus* from charter logbooks were the variables of interest in this study and were determined by the equation

$$rr_t = \frac{n_{released}(t)}{n_{retained}(t) + n_{released}(t)}$$

where $n_{released}(t)$and $n_{retained}(t)$ are the numbers of fish released and retained for each month (t) to determine a monthly release rate, rr_t, for the Mid-west and Metropolitan zones. Since 2009, there has been an annual closure to recreational (including charter) fishing from 15 October to 15 December. Therefore, the *rr* for October and December in all years were calculated using data only from Oct 1-Oct 14 and Dec 16-Dec 31, respectively. To ensure availability of complete time series, the monthly release rates for November were linearly interpolated using the na.interp function in the "*forecast*" package in R (Hyndman and Khandakar, 2008; Hyndman et al., 2022).

Release rates were logit transformed to constrain values in the interval [0,1]. To accommodate months where there were zero release events i.e. $rr = 0$, a $\log(x + 0.1)$ transformation was applied.

Release rates were determined for July 2002 to January 2020 with the intervention set at January 2009, corresponding to the commencement of management changes in the recreational sector of the West Coast Demersal Scalefish Resource which continued between 2009 and early 2010, and are expected to impact *rr* (Lai et al., 2019) (Table 1).

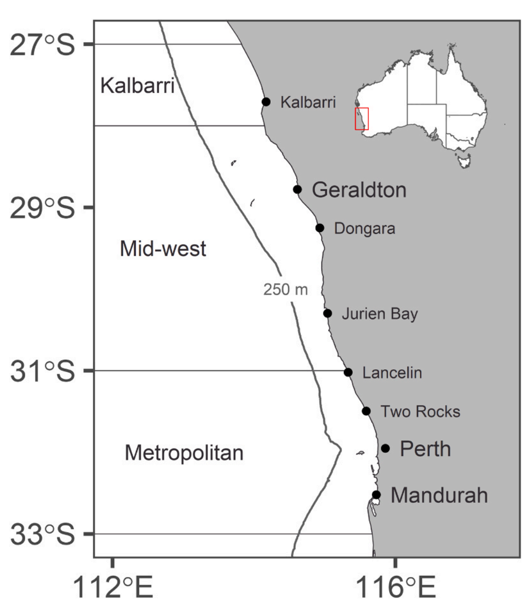


Fig. 1. Map of the study area showing the Kalbarri, Mid-west and Metropolitan management zones for recreational fishing in the West Coast Bioregion of Western Australia.

2.2. Intervention time series analysis

The 18 years of daily logbook data allowed us to effectively utilise intervention time series modelling across that time. This method is not appropriate in cases where intermittent data are available, such as via the on and off-site surveys of recreational fishing regularly used in Australia, including in Western Australia (e.g. Lai et al., 2019; Ryan et al., 2022).

We analysed monthly values of *rr* as a time series. The series exhibited the typical time series features of autocorrelation and seasonality. In addition, the time series considered here were non-stationary in both mean and variance (i.e., statistical properties are changing through time). To assist with the identification of an intervention type we examined the de-seasonalised series derived from a classical seasonal decomposition using moving averages (Box et al., 2015). The de-seasonalised series was not used in subsequent analysis.

The time series of *rr* for each species in the Mid-west and Metropolitan zones were subdivided into a pre- and post-intervention series, covering release rates from 2002 to 2009 and 2010–2020, respectively. A Chow test was used to test and confirm a structural shift from this intervention in each zone using the R-function *sctest* in the *strucchange* package (Chow, 1960; Zeileis et al., 2002). A more temporally precise analysis of the individual management changes was not possible as they occurred at similar times.

To check for non-stationarity and assist with model selection, we used the Augmented Dickey-Fuller (ADF) test on the pre-intervention component of each scenario of *rr* (Dickey and Fuller, 1979).

We fitted seasonal Autoregressive Integrated Moving Average (ARIMA) models using the Box-Jenkins methodology (1976). A general seasonal ARIMA $(p, d, q)(P, D, Q)_s$, can be written as

$$rr_t = \frac{\theta(B)\Theta(B)}{\phi(B)\Phi(B)(1-B)^d(1-B^s)^D}e_t$$

where $\phi\ (B) = 1 - \phi_1 B - ... - \phi_p B^p$ is the autoregressive parameter polynomial, $\Phi\ (B) = 1 - \Phi_1 B^s - ... - \Phi_P B^{sP}$ is the seasonal autoregressive parameter polynomial, $\theta(B) = c + (1 + \theta_1 B + ... + \theta_q B^q)$ is the moving average parameter polynomial, $\Theta(B) = C + \Theta_1 B^s + ... + \Theta_Q B^{sQ}$ is the seasonal moving average parameter polynomial, B is the backshift operator defined by $B^a Y_t = Y_{t-a}$. The model includes the noise e_t occurring at time t (Williams, 2001; Schaffer et al., 2021).

We first assessed the performance of ARIMA models on the pre-intervention time series using the auto.arima function in the *forecast* package in R (Hyndman and Khandakar (2008); (2022). The auto.arima function searches for the best ARIMA model based on AIC. Using the orders of the ARIMA model for the pre-intervention period we forecasted the values (and 95% prediction intervals) of *rr* for a period of 36 months post intervention using the forecast.ts function, also in the *forecast* package, and compared forecasted to observed values. A forecast period of 36 months was chosen as longer-term forecasts from these models are typically unreliable (Box et al., 2015).

For the analysis of the long-term impact of the intervention an ARIMA transfer function (ARIMAX) model that allows for the fitting of multiple transfer functions simultaneously, as proposed by Chan (2020) was used. ARIMAX models have capacity to adequately capture structural shifts in the time series and are applicable to non-stationary series (Williams, 2001). Given a discrete release rate forecast rr_t and intervention variable m (step change, ramp or pulse), the transfer function terms for the external series $U_{i,t}, t = 1, \ldots, m$ are added to the right-hand side of the general ARIMA model,

$$rr_t = \frac{\theta(B)\Theta(B)}{\phi(B)|\Phi(B)(1-B)^d(1-B^s)^D}e_t + \sum_{i=1}^{3}\frac{\omega_i(B)}{\delta_i(B)}B^{k_i}U_{i,t}$$

where $\omega_i(B) = \theta_0^{\omega_i} + \theta_1^{\omega_i}B + ... + \theta_k^{\omega_i}B^k$ and $\delta_i(B)) = 1 - \phi_1^{\delta_i}B - .... - \phi_l^{\delta_i}B^l$ and

$$U_{i,t} = \begin{matrix} S_t & \text{if} & i = 1 \\ P_t & \text{if} & i = 2 \\ R_t & \text{if} & i = 3 \end{matrix}$$

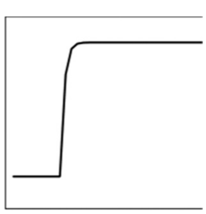
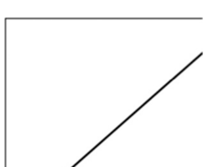
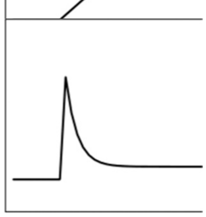
An a priori assessment of the intervention effect is used to select a transfer function. Transfer functions describe the relationship between the intervention and the outcome series Y_t. Data were initially visualised, and the most appropriate transfer function was tested. Parameters of transfer functions were selected by finding those with the lowest AIC (Schaffer et al., 2021). Transfer functions considered in this study were step, ramp and pulse functions or a combination thereof, as described in Table 2.

In the case of a pulse function, the value of δ_1, i.e., the denominator polynomial in the pulse function determines how fast the effect of the intervention decays to zero and is the key parameter in the function (Box and Tiao, 1975). To calculate this value, for each scenario, we fit the orders of the pre-intervention ARIMA model to the full time series for a range of values of δ_1 (i.e., 0–1) and selected the value for δ_1 which produced the model with the lowest AIC. Once $\delta_1 = 1$ the transfer function can be considered a step change where the time series indefinitely increases by ω_0 from the time of the intervention.

The numerator terms ω_i for a transfer function of a predictor series are similar to the moving average component of the ARIMA model for the noise series. The denominator δ_i terms are similar to the autoregressive component of the ARIMA model for the noise series. Denominator terms introduce exponentially weighted, infinite distributed lags into the transfer function. To model the entire series with ARIMAX, we use the ARIMA orders of the pre-intervention model and the transfer function(s) using an autoregressive and moving average filter of order (p,q). For example, if $p = 0$ and $q = 1$, this specifies a simple regression relationship and if $p = 0$ and $q = 0$, this specifies a step-change relationship (Chan and Ripley, 2020). The use of transfer functions is a complex topic and further descriptions are available (Helfenstein, 1991; Williams, 2001; Box et al., 2015).

The final transfer function model form was determined by an iterative procedure using varying p and q combinations that removed statistically insignificant transfer function parameters to find the model with the lowest AIC. The selected transfer function(s) are used to show the nature of the intervention across the observations after the intervention period.

Table 2
Base transfer functions used for intervention analysis in ARIMA.

Function	Form of response	Interpretation	Function
Step		The time series increases by ω_0 immediately following the intervention and remains at this new level for the duration of the study period.	$S_t = \begin{matrix} 0 & \text{if} & t < T \\ 1 & \text{if} & t \geq T \end{matrix}$
Ramp		The time series increases by ω_0 at each time point.	$R_t = \begin{matrix} 0 & \text{if} & t < T \\ t - T + 1 & \text{if} & t \geq T \end{matrix}$
Pulse		The time series increases by ω_0 immediately following the intervention and returns to base line immediately afterwards.	$P_t = \begin{matrix} 0 & \text{if} & t \neq T \\ 1 & \text{if} & t = T \end{matrix}$

3. Results

3.1. Overview of release rates

In all species-zone combinations, the mean *rr* in the post-intervention period was significantly greater than the pre-intervention period (Fig. 2 and Table 3). Additionally, a significant change in trends was detected at the intervention date for all cases (Chow test, $p < 0.05$).

The ADF tests performed on the pre-intervention time series supported a conclusion of non-stationarity and a need for differencing in each case, except *G. hebraicum, C. rubescens* and *E. armatus* in the Metropolitan zone (Table 3).

The overall average *rr* was highest for *C. auratus* compared with the other species in both the Mid-west (0.40) and Metropolitan (0.42) zones between July 2002 and January 2020. For all species, the overall average *rr* was higher in the Metropolitan zone than the Mid-west zone, except *C. rubescens* where *rr* was higher in the Mid-west. The average monthly *rr* for *G. hebraicum*, *C. auratus* and *E. armatus* were highly periodic with several underlying trends across the time series. Seasonality was evident for all species and management zones (Fig. S1). The high values of *rr*, e.g., for *G. hebraicum*, *C. auratus* and *C. rubescens* in April 2020, August 2020 and November 2017, respectively, and for *C. rubescens* in November 2019, were instances of limited records (Fig. S1).

3.2. Trends in release rate

The ACF and PACF plots indicate that *rr* were structurally different with respect to species and the two management zones. Generally, seasonality was weaker in the Metropolitan than in the Mid-west zone (Figs. S1:S5).

There was considerable variation in the trends of *rr* among species and management zones identified from decomposition analyses (Fig. 3). Across all species and management zones there were no knife-edge or immediate responses to the intervention, with an increasing trend often occurring before the intervention that continued across the intervention period.

The pre-intervention trend of *rr* for *G. hebraicum* in the Mid-west zone increased markedly in 2008, while in the Metropolitan zone *rr* was stable with a slight increase in trend starting in 2009. The post-intervention trend for *rr* of *G. hebraicum* in the Mid-west and Metropolitan zones increased shortly after the intervention, before declining several years later. The pre-intervention trend for *rr* of *C. auratus* was similar in the Mid-west and Metropolitan zones. The post-intervention trend for *rr* of *C. auratus* in both the Mid-west and Metropolitan zones peaked in late 2009 and late 2010, respectively, and then declined until 2017. The trends of monthly *rr* of *C. rubescens* in the Mid-west increased gradually after the intervention date. In contrast, the monthly *rr* in the Metropolitan zone were low before the intervention and showed an initial decrease after the intervention, before increasing markedly after that time. There was an increasing trend for *rr E. armatus* in the Mid-west zone before the intervention that changed to a declining trend for around 5–6 years after the intervention. Release rates of *E. armatus* from the Metropolitan zone increased throughout the time series with four periodic pulses (Fig. 3).

3.3. Forecasts in the absence of intervention

The ARIMA model fitted to the pre-intervention time series of data for all scenarios provided adequate fit based on AIC, which allowed

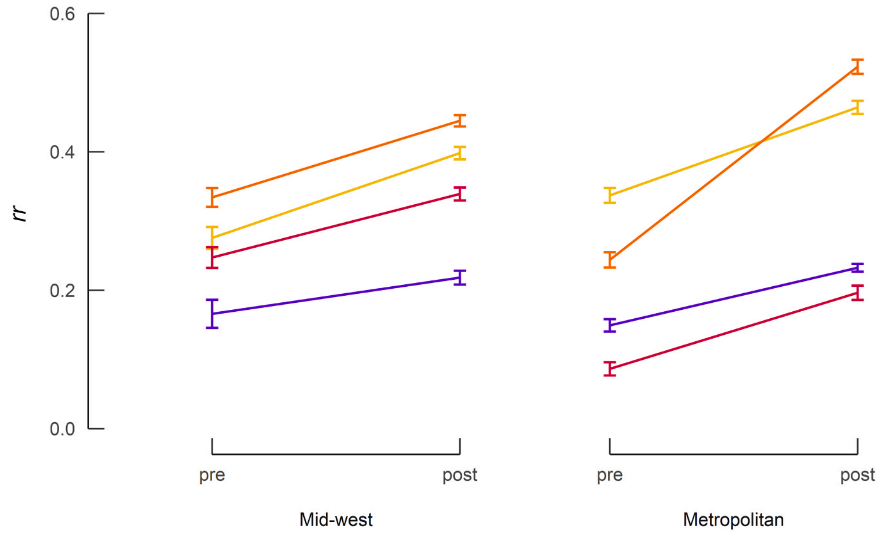


Fig. 2. Mean release rates (*rr*) pre- and post-intervention for *G. hebraicum* (yellow), *C. auratus* (orange), *C. rubescens* (red) and *E. armatus* (purple). Error bars represent 95% confidence intervals.

Table 3
Chow test results for pre- and post-intervention series and results of Augmented Dickey-Fuller test results for pre-intervention series of mean monthly release rates from Charter daily logbooks. One star for p-values below 0.05, two for those below 0.01, three for those below 0.001.

		Chow		ADF	
		Test statistic	p-value	Test statistic	p-value
G. hebraicum	Mid-west	276.471	< 0.001 * * *	-2.713	0.284
	Metropolitan	238.407	< 0.001 * * *	-3.503	0.047 *
C. auratus	Mid-west	222.979	< 0.001 * * *	-2.575	0.34
	Metropolitan	192.585	< 0.001 * * *	-2.729	0.278
C. rubescens	Mid-west	211.727	< 0.001 * * *	-3.031	0.154
	Metropolitan	198.195	< 0.001 * * *	-3.762	0.025 *
E. armatus	Mid-west	282.245	< 0.001 * * *	-3.353	0.069
	Metropolitan	164.947	< 0.001 * * *	-4.430	0.01 * *

short term forecasts of 36 months and provided visual support for the presence of the structural break evidenced by the Chow test. Non-seasonal differencing, i.e. $d = 1$, was required for all scenarios, except for *C. rubescens* in the Metropolitan zone and *E. armatus* in the Mid-west and Metropolitan zones. Different ARIMA orders were found for each pre-intervention time series with *C. auratus* in the Mid-west zone requiring the highest number of ARIMA orders. The order for any given ARIMA was no more than 2, except for *E. armatus* in the Metropolitan zone. No model required non-seasonal autoregressive terms ($p = 0$). The model for *C. auratus* in the Mid-west zone had two and three moving average terms, i.e. $q = 2$. Seasonal differencing ($D = 1$) was found for each model. There was no seasonal autoregressive term for any scenarios i.e. ($P = 0$), except for *C. auratus* in the Mid-west zone ($P = 1$), *C. rubescens* in the Metropolitan zone ($P = 1$) and *E. armatus* in the Metropolitan zone ($P = 2$) (Table S1).

ARIMA model forecasts of *rr*, using the pre-intervention time series of data suggested that in the absence of the intervention, *rr* of *G. hebraicum* would be higher, with an increase in the Mid-west zone and fluctuation around a constant level in the Metropolitan zone. The forecast *rr* of *C. auratus* increased compared with the observed *rr* in the Mid-west zone, while in the Metropolitan zone the observed *rr* was higher than the forecast *rr*. The forecasted *rr* of *C. rubescens* was comparable to the observed *rr* in the Mid-west zone and slightly lower and is similar in some months, but highly variablein the Metropolitan zone. In contrast, forecasted monthly *rr* of *E. armatus* from the Mid-west zone were slightly higher than observed *rr* and increasing. In the absence of an intervention, the forecasted monthly *rr* of *E. armatus* from the Metropolitan zone were comparable with the observed *rr* with the only difference being a gradual increase more evident in the later years (Fig. 4).

3.4. Modelling the impact of the intervention

For *G. hebraicum* in the Mid-west and Metropolitan zones and *C. rubescens* in the Mid-west zone, the transfer function that provided the best fit for use in the ARIMAX models in terms of AIC was a step function, which indicates the intervention effect on *rr* was ongoing. In contrast, the best fitting model for *rr* of *C. auratus* in the Mid-west and Metropolitan zones included both step and pulse transfer functions with δ_1 values of 0.68 and 0.98, respectively. The presence of a pulse indicates the intervention effect on *rr* was temporary, noting that the coefficients for the pulse parameter was far greater in the Metropolitan zone (Table S2).

ARIMAX models for *C. rubescens* in the Metropolitan zone and *E. armatus* in the Mid-west and Metropolitan zones had the lowest AIC values with both step and ramp transfer functions, which indicates the intervention effect on *rr* was ongoing and increasing for *C. rubescens* in the Metropolitan zone. However, the negative coefficients for *E. armatus* indicate a negatively sloped ramp (Table 4 & Table S2).

The series produced from the ARIMAX models aligned well with the observed series in all cases. The fit for *G. hebraicum* and *C. auratus* in the Metropolitan zone after the intervention had lower variation in the predicted *rr* compared with the observed *rr*. The predicted *rr* for *E. armatus* in the Metropolitan zone had lower variation than the observed values throughout the time series (Fig. 5).

4. Discussion

In this study, we assessed the impact of restrictive management changes implemented in 2009 and early 2010 on release rate trends in a major recreational fishery in Western Australia. These changes limited both access to fishing and retained catches more than prior to management arrangements (Lai et al., 2019). The intervention time series analysis applied in this study provided a statistical approach to evaluate the extent of impacts of changes in resource management. This is particularly beneficial in recreational fisheries, where aperiodic changes are implemented to achieve various sustainability objectives. To the best of our knowledge, this is the first study that has used intervention analysis on changes in release rates in a marine recreational fishery.

There was an increase in the release rates and a significant change in

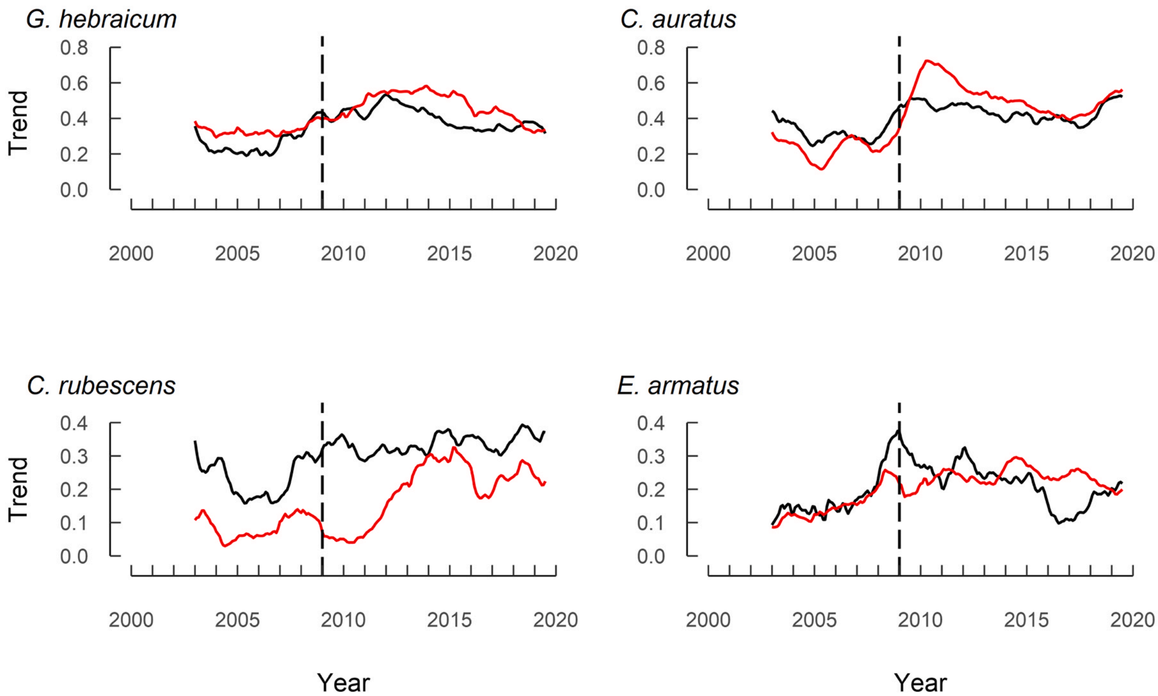


Fig. 3. Trend component from classical decomposition of monthly release rates for *G. hebraicum*, *C. auratus*, *C. rubescens* and *E. armatus* from the Mid-west (black) and Metropolitan (red) management zones. Dashed line at start of 2009 represents period of management changes.

trends detected for all species in each zone post management changes. The impact of the regulatory changes on release rates was greater in the Metropolitan management zone for all species. Four types of responses to the intervention were identified, including pulse, ramp, step, and a combined step-ramp response. However, the response types differed among species and management zones, except for *C. auratus* where a pulse was detected in both management zones.

Time-series analysis of fisheries data has been an important tool for fisheries management and decision-making and has been predominantly applied to commercial fisheries (Parsons and Colbourne, 2000; Lloret et al., 2000; Prista et al., 2011; Ryan and Meyer, 2020; Selvaraj et al., 2020; Sanz-Fernández et al., 2022). In contrast there has been limited application in recreational fisheries management (Farmer and Froeschke, 2015). Examples of intervention analyses in recreational fisheries are limited to those focusing on economic impacts and catch per unit effort in inland fisheries (Cloutman and Jackson, 2003; Oh and Ditton, 2008). Our study focused on highly prized key demersal species from a charter fishery which, like commercial fisheries in Western Australia, maintain mandatory logbooks. These provide time series data that allow annual monitoring of trends in key catch and effort metrics, including release rates and are suitable for modelling intervention effects. Many recreational fisheries are considered to be data poor and, despite their importance in Western Australia, the periodic nature of surveys employed to obtain catch information cannot be used in such models. Using ARIMAX, we extended the capabilities of ARIMA modelling (Box and Jenkins, 1976) to adequately describe the types of changes in release rates and assess their significance due to changes in management. For example, in this study, a sustained impact of the intervention occurred in both the Metropolitan and Mid-west zones for all species except *C. auratus*. Additionally, the temporary increase in release rates for *C. auratus* in the Mid-west and Metropolitan zones was best described using pulse and step transfer functions which quantified the transient nature of the impact. This contrasts with the step change identified for *G. hebraicum* in the Mid-west and Metropolitan zones that confirmed a permanent increase in release rates.

4.1. Understanding changes in release rates

This study demonstrates significant changes in release rates following targeted management changes. Targeted management changes for *G. hebraicum* and *C. auratus*, two of the most prized recreational species, included reductions in daily bag limits, introduction of boat limits for *G. hebraicum* and an increase in the minimum size limits for *C. auratus*. The minimum size limit for *C. auratus* in the southern part of the West Coast Bioregion (south of 31°S), including the Metropolitan zone, increased from 410 to 450 mm by early 2009, with a further increase to 500 mm in late 2009. This would most likely have contributed to the increase in release rates for *C. auratus* immediately following the change being greater in the Metropolitan zone compared to the Mid-west zone (where the size limit did not change). This aligns with findings from surveys of private boat-based recreational fishers in the West Coast Bioregion, that showed release rates of *C. auratus* were 10% higher in the Metropolitan zone than the Mid-west zone in 2009/10 and remained higher in subsequent years (Lai et al., 2019; Ryan et al., 2022). These findings are also consistent with a review of catch-and-release angling mortality in the United States where an increase in the release rate from 34% to 58% was attributed to lower bag limits and increased size limits from 1981 to 1999 (Bartholomew and Bohnsack, 2005). Similarly, an increase in release rates for sea bass in the United Kingdom was attributed to the introduction of size-based harvest regulations (Pickett et al., 1995). Our modelling approach was not able to identify the impact of the individual regulation changes such as the boat limit of 6 individual *G. hebraicum* per charter vessel or the size limit changes for *C. auratus*. However, an evaluation of charter boat regulations in New York, for example, found that boat limits were not effective at limiting retained catch unless they were very low (Powell et al., 2010). Furthermore, changes in release rates may also follow changes in fisher behaviour and perceived recreational value of some species, as well as biological changes (Arlinghaus et al., 2013).

Interannual variation in recruitment from varying intensity of spawning events and variation in environmental productivity can result in subsequent variation in catches by recreational fishers. New recruits would be below any size limit for retention for a period while they are

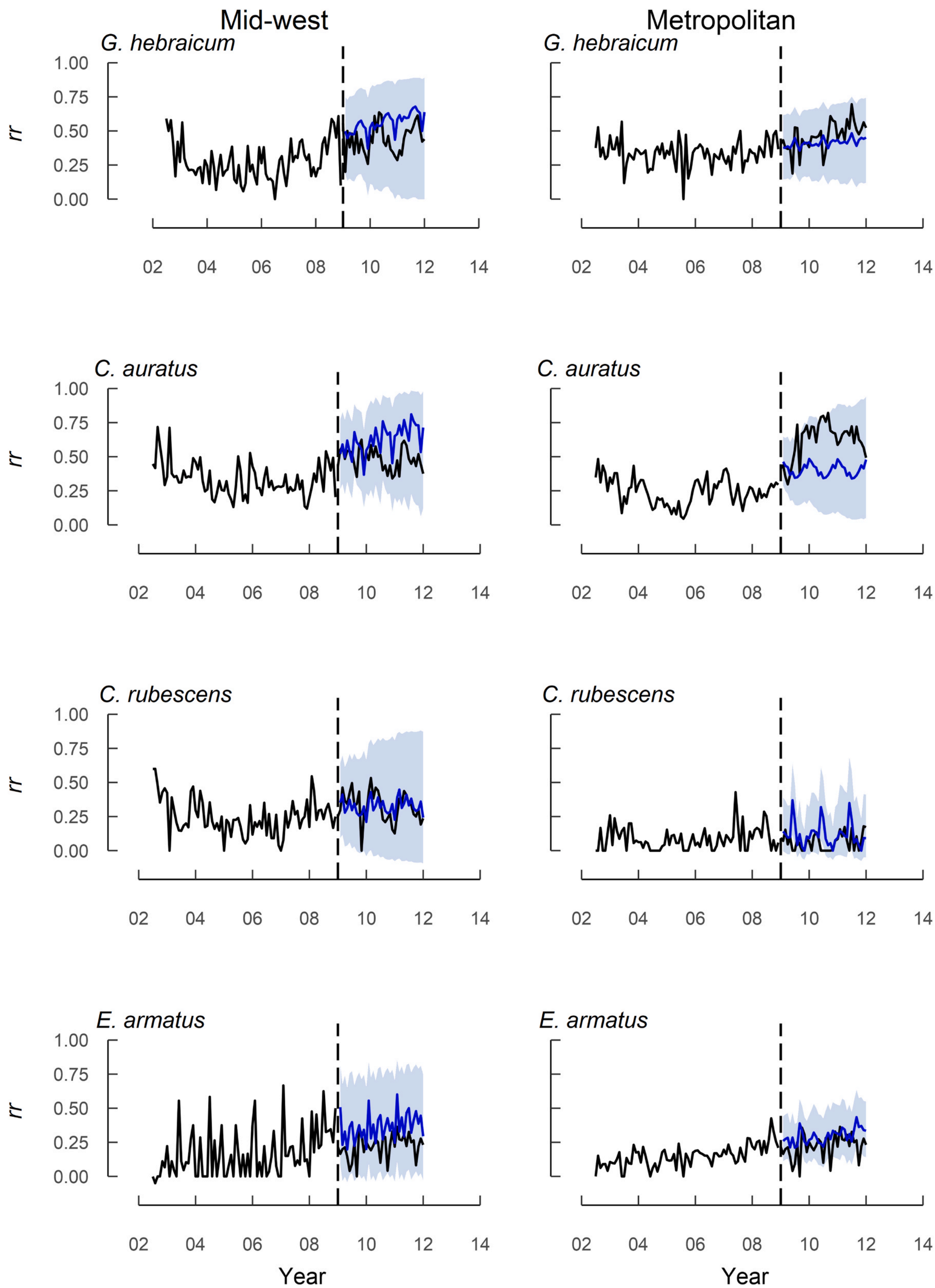


Fig. 4. Forecast of monthly release rates (*rr*) with associated 95% prediction interval (blue line with shaded ribbon) from model fitted to pre-intervention data. Observed pre- and post-intervention in the Mid-west and Metropolitan management zones for *G. hebraicum*, *C. auratus*, *C. rubescens* and *E. armatus* (black line). Dashed line indicates time of management intervention.

Table 4
Transfer function polynomials for release rates from Charter daily logbooks. P = Pulse, S = Step and R= Ramp. Numbers represent the autoregressive order for each transfer function.

Mid-west	Metropolitan
S_1	S_1
P_0 and S_0	P_1 and S_1
S_2	S_0 and R_0
S_2 and R_0	S_0 and R_1

young and potentially being surplus to needs or above bag limits once they exceed the minimum size limit. The extent of interannual and spatial recruitment variation differs for *G. hebraicum* and *C. auratus* (Wakefield, 2010; Fairclough et al., 2021). For example, the variation in annual recruitment of *G. hebraicum* differs between the management zones examined in this study, and they are largely independent in terms of larval recruitment (Berry et al., 2012; Lewis, 2015). Wakefield (2010) found the intensity of spawning events of *C. auratus* was significantly different between years in a major marine embayment in the Metropolitan zone of Western Australia. This was reflected in an increasing trend in the release rates of *C. auratus* in the Metropolitan zone around the intervention period (2009–2011), which began with a strong cohort in 2007 and the increased minimum size limit in 2009, evidenced by the presence of strong cohorts in recent age structures of *C. auratus* (Fairclough et al., 2021). Additionally, changes in the release rates could be linked to the limited mixing of breeding stocks of *C. auratus* between the Mid-west and Metropolitan zones, as indicated in genomic, otolith microchemistry and mark-recapture studies (Fairclough et al., 2013; Crisafulli et al., 2019; Bertram et al., 2022). While there is evidence of relatively stronger and weaker year classes of *G. hebraicum* in age compositions from the Metropolitan zone, the interannual variation was presumably not strong enough to be clearly reflected in changes in release rates (Fairclough et al., 2014). The release rates for *G. hebraicum* in the Metropolitan zone trended downwards a few years after the intervention and this could be linked to the gradual decline in abundance of a strong cohort from 1999 that was fully recruited into the fishery along the southern part of the west coast (Fairclough et al., 2021). Similarly, elevated release rates of Atlantic cod, *Gadus morhua* in German waters in 2009/10 appeared to be an indication of a large recruitment event in 2008 (Strehlow et al., 2012). Indeed, variability in recruitment of exploited fish populations is the primary contributor to variability in population growth for many commercially important species (Thorson et al., 2014). A simulation study by Allen and Pine (2000) indicated that variable recruitment could strongly affect the ability of fishery managers to detect effects of introducing or changing a size limit.

Management changes aiming to reduce the overall catch usually result in changes in fisher behaviour (Arlinghaus et al., 2013). For example, when strict mixed species bag limits apply, fishers are more likely to release lesser valued species or smaller fish of the valued species, known as high grading (Arlinghaus et al., 2013). This could partly explain the increase in release rates of *C. rubescens* and *E. armatus* (which are less prized than *G. hebraicum* and *C. auratus*) after the management change. For instance, the release rate for *C. rubescens* in the Metropolitan zone tripled after the intervention. These higher release rates can contribute to increases in fishing mortality, particularly for species which are more susceptible to post-release mortality (Coleman et al., 2004; Haase et al., 2022). Additionally, fishers may change their behaviour by adjusting the locations fished and intensity of their fishing effort after the reduction of bag limits. This may have occurred in the West Coast Bioregion, evidenced by the generally greater annual number of fishing days from charter fishing in coastal waters adjacent to regional towns such as Jurien Bay (Fig. 1) from around 2010/11–2020/21 (Fairclough et al., 2021).

4.2. Management implications

The importance of understanding the extent of released catches in recreational fisheries is well documented (Cooke and Cowx, 2004; Fairclough et al., 2021; Shertzer et al., 2022). Release rates are important as the accuracy of total fishing mortality is dependent on understanding released catches. Post-release mortality can contribute substantially to the total fishing mortality and, if not accounted for, can bias estimates of biomass from assessments (Ferter et al., 2013). For example, a simulation study found that additional fishing mortality from released catches increased in accordance with increasing size limits for both short- and long-lived species and can lead to recruitment overfishing (Coggins et al., 2007). This finding was supported in our study in terms of the increase in release rates of *C. auratus* in the Metropolitan zone following increases in the minimum size limit from 410 to 500 mm and given the likely influence of a strong year class in the stock, resulting in large numbers of undersize fish. Furthermore, many undersize *C. auratus* are caught in deeper waters, where they are more susceptible to barotrauma-related mortality (Lenanton et al., 2009; Wakefield, 2010).

In addition to higher release rates of *G. hebraicum* and *C. auratus* in the Metropolitan management zone than the Mid-west zone, the average weight of the retained catch for these species was also higher in the former zone. While the average weight of released fish is unknown, the greater weight-at-length of retained fish from the Metropolitan zone, in addition to higher released catches, will contribute a greater overall proportion of released catches (by weight) being lost to post-release mortality (Crisafulli et al., 2022). Furthermore, increased release rates of *G. hebraicum* and *C. auratus* occurred in intermittent surveys of private boat-based recreational fishers after the intervention (Lai et al., 2019; Fairclough et al., 2021; Ryan et al., 2022).

There is consensus regarding the importance of accounting for release mortality in stock assessment models that account for all sources of fishing mortality (Cook, 2019; Shertzer et al., 2022). The most common approaches for including release mortality in assessments of commercial fisheries are to combine dead discards with landings, assign discards as a separate fleet or to model discards with a retention function, the latter being considered more precautionary in the 2021 assessments of *G. hebraicum* and *C. auratus* (Fairclough et al., 2021; Shertzer et al., 2022). The best approach depends on the relative contributions of observation and process errors in discarding, and further research is needed to provide general guidance on model selection (Shertzer et al., 2022).

There is value in modelling the impact of future management changes on release rates of key species and making forecasts. The result of the 2021 assessment of *G. hebraicum* and *C. auratus* demonstrated that stocks had not recovered as expected and total fishing mortality from the recreational sector, including post-release mortality from released catches, for some species, such as *C. auratus*, *C. rubescens* and *G. hebraicum,* was above catch limits (Fairclough et al., 2021; Fairclough and Walters, 2021). However, it is difficult to distinguish the effects of different rules on release rates and mortality. Thus, future time series analyses need to incorporate extra information around targeting, changes in population abundance (e.g., due to recruitment), reasons for release (e.g., undersize, too small) and whether bag/boat limits were reached into such modelling to allow researchers and managers to better disentangle the effects of specific rules changes (e.g., bag limits vs boat limits) when multiple changes are implemented at the same time. The reliability of this information would benefit from validation by an onboard scientific observer program which can help identify reporting issues of species diversity and collect more quantitative information around the released catches (Gray and Kennelly, 2017a; Gray and Kennelly et al., 2017b).

This study has demonstrated that release rates differ spatially, temporally and among species in the charter component of a recreational fishery. The observed release rates from charter logbooks differed

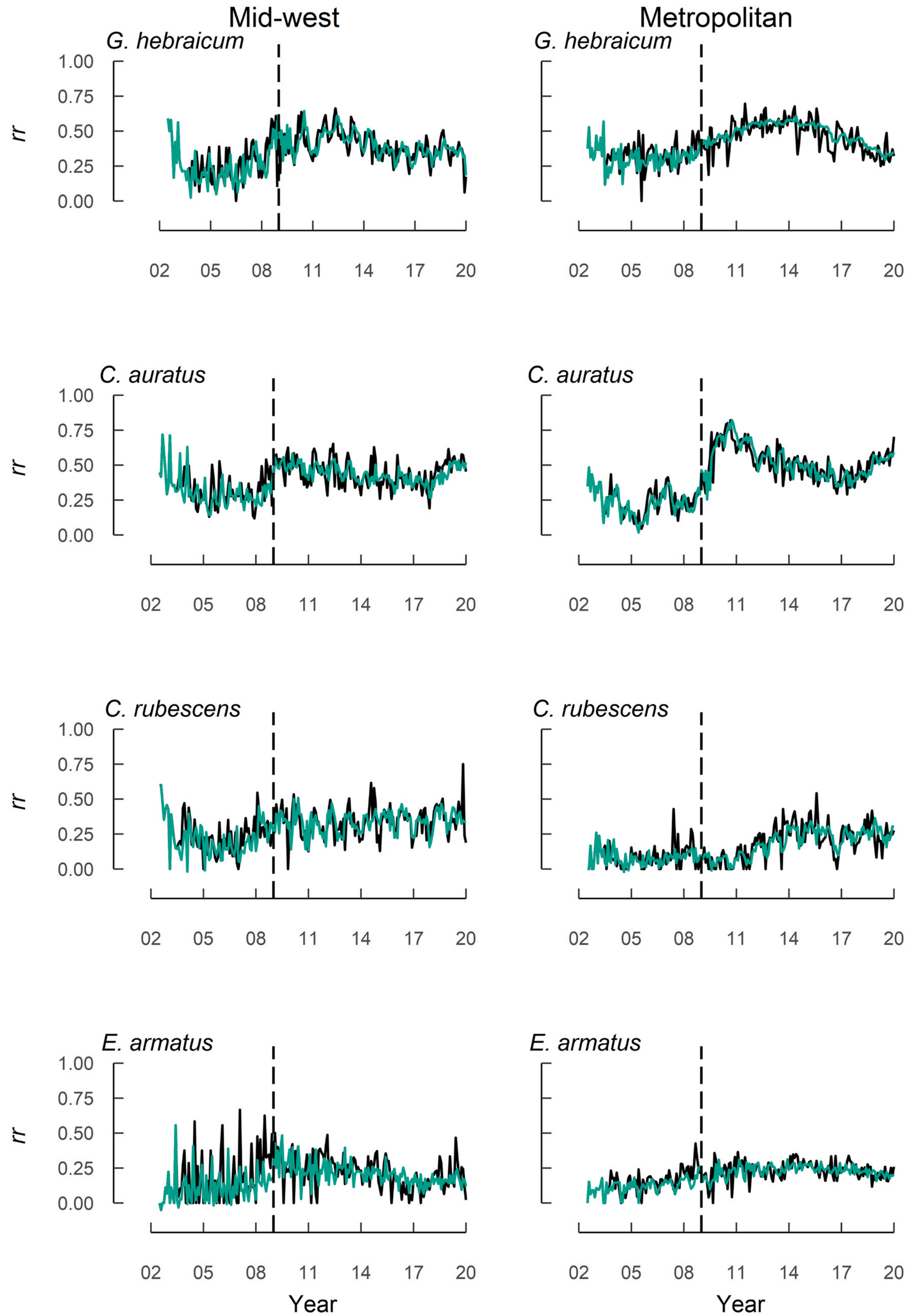


Fig. 5. ARIMAX model fits (green) and observed values (black) for the time series of monthly release rates (*rr*) of *G. hebraicum*, *C. auratus*, *C. rubescens* and *E. armatus* from 2002 to 2020 in the Mid-west and Metropolitan management zones. Dashed line indicates time of management intervention.

to those observed from private boat-based recreational fishing surveys for species and management zones investigated in this study where comparisons could be made (Ryan et al., 2022). For example, release rates for Snapper in the Metropolitan zone were higher from private boat-based recreational fishing (75% in 2020/21) compared with the 56% observed from charter fishing in this study (Ryan et al., 2022). This finding highlights the differences between for-hire (charter) and private boat-based fisheries and is useful given the transition to new, more restrictive, management regulations in 2023, which may potentially exacerbate these differences (DPIRD, 2022). Intermittent surveys of private boat-based recreational fishing collect information around the reasons for release, which shows that most released catches are for fish being 'under size', however, charter fishers are not required to record this information (Ryan et al., 2022). It would be expected that the reasons for release of fish differ between charter and private boat-based fishers due to the differing targeting, experience, motivation and management arrangements. Indeed, a different response to management interventions between recreational fisheries was observed in the Baltic Sea where distances travelled across Germany by charter anglers decreased after a management change, but distances travelled by private boat and shore fishers remained constant (Lewin et al., 2021). Assessment and management of recreational fisheries need to consider the unique human dimensions influencing catch variation and releases (Cooke and Cowx, 2004; Arlinghaus et al., 2007).

5. Conclusions

Our study has shown that intervention analysis using ARIMA and ARIMAX models have utility in capturing and describing the effect of management changes on release rates of four key species in a multi-species recreational fishery. Different transfer functions were required to describe intervention effects for different species and management zone combinations, highlighting the complexities in understanding external factors that influence release rates. This indicates that management changes alone are not likely to explain all temporal changes. However, with additional information, such as targeting, this approach would be useful to understand how key recreational fishing metrics, such as release rates, respond to various management changes, such as species-specific minimum size and daily bag limits. The application of intervention time series modelling approaches has the potential to support evaluation of impacts of policy settings to inform management decision-making.

CRediT authorship contribution statement

Brett Crisafulli: Conceptualization, Methodology, Formal analysis, Visualization, Data curation, Writing – original draft. **Ebenezer Afrifa-Yamoah:** Conceptualization, Investigation, Methodology, Supervision, Writing - review & editing. **Ute Mueller:** Conceptualization, Resources, Project administration, Methodology, Supervision, Writing - review & editing. **Karina Ryan:** Conceptualization, Investigation, Data curation, Supervision, Writing - review & editing. **David Fairclough:** Conceptualization, Investigation, Resources, Supervision, Writing - review & editing. **Johnny Lo:** Conceptualization, Supervision, Methodology, Writing – review & editing, Validation.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Data Availability

Data will be made available on request.

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Appendix A. Supporting information

Supplementary data associated with this article can be found in the online version at doi:10.1016/j.fishres.2023.106818.

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