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Interannual variation in a freshwater recreational fishery under the influence of drought, bushfires, floods and a global pandemic

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ABSTRACT

Context. As social–ecological systems, recreational fisheries often vary temporally in response to environmental changes affecting ecological processes and human behaviour. Monitoring such variability in this ecosystem service can guide adaptive management measures for sustainability. **Aims.** This novel research for Australian, sought to quantify interannual changes in the freshwater recreational fisheries of five key (i.e. commonly caught) finfish species (Murray cod, *Maccullochella peelii*; golden perch, *Macquaria ambigua*; Australian bass, *Percalates novemaculeata*; brown trout, *Salmo trutta*; and rainbow trout, *Oncorhynchus mykiss*) in relation to a series of extreme climate-related events and the COVID-19 pandemic. **Methods.** Annual estimates during 2013–14, 2017–18 and 2019–20 of freshwater fishing effort and catch across New South Wales, Australia, were derived from off-site surveys and compared in relation to a severe drought period, the 'Black Summer' bushfires, widespread flooding and the COVID-19 pandemic, all of which affected fish productivity or human mobility. **Key results.** There were significant declines in fishing effort between 2013–14, the year preceding the extreme environmental events and the pandemic, and 2017–18 and 2019–20. Catch across the five species was also significantly lower in 2019–20. Catch of species such as golden perch and rainbow trout declined from 2013–14 to 2019–20. **Conclusions and implications.** This study can inform adaptive measures against societal and climate-related changes in weather by enabling scientists and managers to identify problematic trends.

Keywords: angling, cascading hazards, climate change, general linear mixed effects models, inland fisheries, Murray–Darling Basin, natural disasters, telephone-diary surveys.

Introduction

The world's fisheries are socio-ecological systems, representing a nexus between ecological processes and human behaviours (Arlinghaus *et al.* 2017; Taylor and Suthers 2021). Environmental changes can therefore threaten fisheries and their supporting ecosystems by influencing the productivity of fish stocks or by altering behaviours and decision-making among fishers (Dudgeon *et al.* 2006; Cooke *et al.* 2021). In freshwater ecosystems, anthropogenic environmental changes that threaten exploited species include over exploitation, pollution, habitat destruction and invasion by exotic species (Dudgeon *et al.* 2006; Bond *et al.* 2011; Forsyth *et al.* 2013; Burgin 2017). These ecosystems are also threatened by natural shifts in temperature, precipitation and runoff patterns (Dudgeon *et al.* 2006) which may manifest as extreme weather events (Morrongiello *et al.* 2011*a*; Scott *et al.* 2020). Although each of these environmental factors can have substantial independent effects on freshwater species, they can also act synergistically to have major impacts on freshwater fisheries (Leprieur *et al.* 2008; Rabalais *et al.* 2010).

At a global scale, the clearest example of this synergy is climate change, which has been accelerated over the past century by human-induced emission of green-house gases (Pachauri and Reisinger 2007; Füssel 2009). Climate change is estimated to threaten ~50% of global freshwater fish species by directly altering species distributions and survival rates (Parmesan and Matthews 2006; Bassar *et al.* 2016). Alterations to annual precipitation and temperature regimes due to climate change are also likely to amplify the impacts and frequency of localised extreme weather events such as droughts (Milly and Wetherald 2002; Trenberth 2005), storms, floods and wildfires (Füssel 2009; Reid *et al.* 2019); each of which can affect freshwater fish. Droughts, for example, cause declines in water levels, which can affect fish detrimentally through temperature and oxygen stress or introduction of disease (Morrongiello *et al.* 2011*b*; Lennox *et al.* 2019). This can result in spawning failures, reduced growth, assemblages crowded with predators and competitors and, in the Australian context, promote species invasion (Lennox *et al.* 2019; Sheldon *et al.* 2022). Changes in flows and water levels can also result in fish being isolated in fragmented pool habitats of poor water quality. When such extreme events happen in succession, their consequences on freshwater ecosystems and their services can be multiplicative (Alexandra and Finlayson 2020; Kemter *et al.* 2021). For freshwater stocks in developed countries, recreational fishing is a major ecosystem service (Cooke and Cowx 2006; Arlinghaus *et al.* 2017) and research documenting temporal changes in this service in relation to extreme weather events enables managers to undertake risk assessment and adaptation towards long-term fisheries sustainability (Hunt *et al.* 2016; Sainsbury *et al.* 2018). Although several international studies have explored these relationships in the context of broad climate change effects (Jones *et al.* 2013; Hunt *et al.* 2016; Jeanson *et al.* 2021), there are a limited number of Australian studies examining variation in freshwater recreational fisheries in relation to extreme climate-related events.

In addition to affecting ecosystems that support fisheries, environmental changes often occur over a changing social, economic and political landscape. These societal changes can affect fisher behaviour and decision-making and therefore compound environmental impacts on fisheries (Arlinghaus *et al.* 2017; Taylor and Suthers 2021). For example, the emergence of the global pandemic caused by the SARS-CoV-2 virus (COVID-19), which has been partly attributed to human-induced environmental changes such as habitat encroachment and urban sprawl (Barouki *et al.* 2021; Kumar and Ayedee 2021), was an atypical event that had unprecedented impacts on economies, human mobility and use of aquatic ecosystems. It therefore affected the distribution of fishing effort and catch in Australian marine systems and other parts of the world (Coll *et al.* 2021; Ryan *et al.* 2021). However, just like the impacts of extreme whether events, there is no published work tracking freshwater recreational fishing in Australia in relation to the societal and behavioural changes caused by COVID-19.

In the Australian jurisdictions of New South Wales (NSW) and the Australian Capital Territory (ACT), off-site surveys are regularly used to track spatio-temporal patterns of recreational fishing activity (Ochwada-Doyle *et al.* 2021; Murphy *et al.* 2022). The freshwater systems of these jurisdictions are primarily recreational-only (Forbes *et al.* 2020; NSW Department of Primary Industries 2020) and support ~26% of the study area's total recreational fishing effort (West *et al.* 2015; Murphy *et al.* 2020, 2022), making the surveys a vital fisheries-monitoring tool. A standardised survey design was first implemented in NSW and ACT in 2013–14. The survey design was repeated for 2017–18 and 2019–20, a period when COVID-19 spread through NSW and ACT and a series of extreme weather events occurred in the region. These events were (1) the most severe drought recorded in the European history of Australia (Bureau of Meteorology 2020), (2) the 'Black Summer' bushfires, which burnt through more land than any fires in the past 25 years (Alexandra and Finlayson 2020; Rural Fire Service 2020) and (3) heavy rainfall and subsequent flooding that led to some of the highest river levels since 1992 (Kemter *et al.* 2021). This synchronicity created a unique opportunity to track changes in recreational fishing metrics alongside the occurrence of a series of atypical events. This work assessed (1) changes in total fishing effort and the total catch of finfish species in NSW and ACT freshwater systems among 2013–14, 2017–18 and 2019–20, and (2) changes in the species-specific catch of the region's five most sought-after freshwater finfish. This research aimed to monitor patterns of fishing across 3 years that happened to coincide with atypical environmental and social events, rather than elucidate conclusive causal relationships between the events and the fishery. This research encompassed all freshwaters of NSW and ACT, which can be divided into the following three geographic regions: the Eastern Basin (east of the Great Dividing Range), the Murray–Darling Basin (west of the Great Dividing Range including the ACT) and the South East Basin (the majority of the Snowy Mountains region) (Fig. 1; Murphy *et al.* 2022). The Murray–Darling Basin, which covers 14% of Australia's land, is of particular significance in terms of the nation's water, agriculture and fisheries management (Koehn 2015). It accounts for 66% of Australia's total agricultural water consumption, supports an agriculture industry valued at A$15 billion year^{-1}, provides habitat for 26 native fish species and supports 80% of the annual freshwater recreational fishing effort in NSW and ACT (Humphries *et al.* 1999; Koehn 2015; Murphy *et al.* 2022).

Materials and methods

Data collection

This study utilised recreational fishing data collected across all freshwater systems in New South Wales (NSW) and the Australian Capital Territory (ACT; Fig. 1). Data on recreational catch and effort were collected for fishing activity that occurred during the 2013–14, 2017–18 and 2019–20 periods. For all three 12-month periods, the data were collected using an off-site telephone-diary survey methodology that was developed to provide cost-effective, state-wide fishery information over

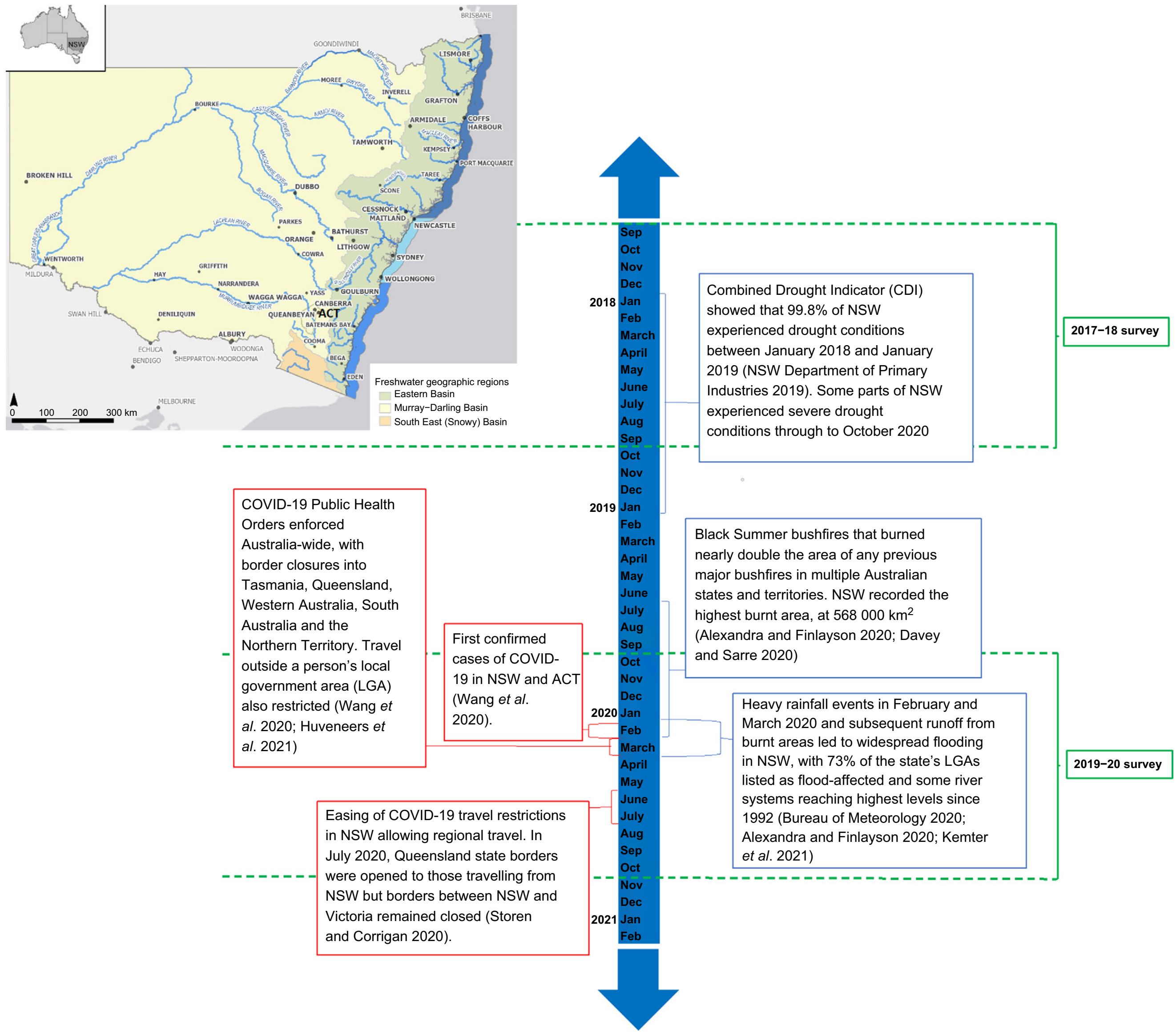

Fig. 1. Australian map, indicating the relative sizes and locations of New South Wales (NSW) and the Australian Capital Territory (ACT). Included is a timeline indicating the sequence of social and environmental events that took place within NSW and ACT during the 2017–18 and 2019–20 recreational fishing surveys. Note that a preceding 2013–14 survey is not included in the timeline depicted here (NSW Department of Primary Industries 2019; Alexandra and Finlayson 2020; Bureau of Meteorology 2020; Davey and Sarre 2020; Storen and Corrigan 2020; Wang *et al.* 2020; Kemter *et al.* 2021; Huveneers *et al.* 2021).

a large spatial scale (Lyle *et al.* 2002; Henry and Lyle 2003; West *et al.* 2015).

The telephone–diary approach involved two phases, namely an initial *Screening Phase* followed by an intensive *Diary Phase*. These phases are described in greater detail in Murphy *et al.* (2020) and Murphy *et al.* (2022). In brief, the *Screening Phase* was conducted from March to May 2013, from September to October 2017 and from August to October 2019. This phase was conducted as a structured telephone interview of a randomly selected sample of 1–3-year licence holders listed within the NSW Recreational Fishing Licence (RFL) database (Murphy *et al.* 2020, 2022). The shortcomings of broad-scale offsite surveys that rely on fisher-based databases such as the RFL are discussed in Taylor and Ryan (2020) and Ochwada-Doyle *et al.* (2021). They include reliance on a sample that is dominated by moderate and highly avid fishers, rather than those who only fish a few times per year. Such designs can also preclude robust estimation of species-specific effort or catch rate (catch per unit effort, CPUE), because of their broad assessment of multiple and varied fishing methods for many species and an inability to disaggregate effort that leads to kept catch *versus* released catch (Ochwada-Doyle *et al.* 2022). Furthermore, off-site surveys are generally limited in accuracy and precision when it comes to estimating the catch of infrequently caught species or those caught as part of 'niche' recreational fisheries. During the *Screening Phase*, 1882, 1960 and 1990 RFL households were selected and contacted as part of the net sample for 2013–14, 2017–18 and 2019–20 respectively (Murphy *et al.* 2020, 2022). The number of RFL households that responded to the complete screening survey in 2013–14 was 1686, compared with 1618 for 2017–18 and 1608 in 2019–20. These households answered questions regarding their intention to fish in the ensuing 12 months to determine their eligibility for the *Diary Phase* (Murphy *et al.* 2020, 2022). Profiling information, including age, gender and fishing avidity (measured as a function of estimated fishing frequency in the previous 12 months for residents aged ≥5 years), was also collected for each resident of a sampled household during the *Screening Phase* (Murphy *et al.* 2020).

The *Diary Phase* comprised a longitudinal panel survey that monitored the fishing activity of all residents (aged 5 years or older) within recruited households between June 2013 and May 2014, between October 2017 and September 2018 and between November 2019 and October 2020 (Murphy *et al.* 2020, 2022). For 2013–14, 87% of eligible households completed the *Diary Phase*, 89% completed this phase for 2017–18, and 75% completed it for 2019–20 (Murphy *et al.* 2020, 2022). A single diarist within each household that participated in each year's *Diary Phase* recorded basic information after each fishing event undertaken by any household resident (aged ≥5 years) including unlicenced members of the household and those with short-term licences. This information included date, location, start and finishing times and numbers of kept and released fish by species or species group (Murphy *et al.* 2020, 2022). Throughout the diary period, trained interviewers telephoned diarists at regular, repeated intervals to conduct structured interviews that collected more detailed data on each fishing event (with each fishing event being defined as a single fishing trip made on a single day). These data included fishing method, fishing platform (boat-based or shore-based), waterbody type (freshwater, estuarine or oceanic) and target species. The numbers of fish that were retained (i.e. harvested and kept by anglers) and released (i.e. discarded for any reason) by species or species group were also verified during these interviews (Murphy *et al.* 2020; Murphy *et al.* 2022).

Fig. 1 illustrates the sequence, duration, severity and spatial extent of three extreme weather events (wide-spread drought, bushfires and heavy rainfall resulting in flooding) as well as the COVID-related restrictions relative to the timing of the 2017–18 and 2019–20 recreational fishing surveys. This study, involving human participants, was approved by the New South Wales Department of Primary Industries Human Ethics process (INT20/76587).

Data analysis

All analyses and graphing described hereafter were executed within the statistical programming software R (ver. 3.61 32 bit, R Foundation for Statistical Computing, Vienna, Austria, see https://www.r-project.org/). Details on the analytical expansion of the core survey data can be found in West *et al.* (2015) and Murphy *et al.* (2020). This analysis was based on a stratified random survey design that used single-stage cluster sampling, with the RFL holders representing the primary sampling unit (n = sample size in terms of RFL holders) within a household where the activity of all resident fishers was surveyed. Expansion of information collected from each sampled household to population estimates relied on an integrated approach that calibrated against population benchmarks and adjusted for non-response using logistic generalised linear models (Lyle *et al.* 2010; West *et al.* 2015). Expansion and calibration to attain population estimates of catch and associated standard errors (s.e.) were completed using the *Survey* (Lumley 2010) package, on the basis of the instructions detailed in Lyle *et al.* (2010). Refer to Lyle *et al.* (2010) and Lumley (2010) for equations.

The native species Murray cod (*Maccullochella peelii*), golden perch (*Macquaria ambigua*) and Australian bass (*Percalates novemaculeata*), and the introduced species brown trout (*Salmo trutta*) and rainbow trout (*Oncorhynchus mykiss*) are among the five most commonly caught finfish species in freshwater river and lake or dam environments of NSW and ACT (Murphy *et al.* 2022). The expanded estimates of retained and released catch (numbers) of these species were calculated for freshwater environments during each survey period. Both species-specific estimates of catch and total catch estimates across these five species were calculated. It should be noted that seasonal closures to harvesting Australian bass, Murray cod, rainbow trout and brown trout apply in NSW and ACT rivers. Because the closures for each species are applied at the same time each year, annual catch estimates should not be affected by the closures from one year to the next. Expanded estimates of total recreational fishing effort (fisher days) were also calculated for freshwater environments during each survey period.

Using the glmer.nb() function in the *lme4* package (ver. 1.1-26, see https://CRAN.R-project.org/package=lme4; Bates *et al.* 2015, 2020), generalised linear mixed-effects models (GLMMs) assuming a negative binomial distribution were utilised to examine whether species-specific catch, total catch and effort changed among the three survey years, which had three levels, namely 2013–14, 2017–18 and 2019–20 (Maunder and Punt 2004; Coelho *et al.* 2020). The use of GLMMs enabled us to account for any non-independence among primary sampling units through inclusion of random-effects terms for individual persons and households (Zuur *et al.* 2009; Morrongiello and Thresher 2015). For catch data from each survey period, separate analyses were conducted for kept and released catch data. The GLMMs took the following form:

$$Y = \beta_0 + \beta_1 x_{1,ij} + a_{1,i} + a_{2,j} + \varepsilon_{ij} \quad (1)$$

where Y represents the catch (numbers kept or released) or effort (fisher days) of a sampled angler after it has been expanded to represent the catch or effort of all anglers in the whole known population that are represented by that sampled angler, β_0 is the vertical intercept, β_1 is the regression coefficient for the independent parameter Year ($x_{1,ij}$); $a_{1,i}$ and $a_{2,j}$ represent the random variables associated with Person i and Household j, and ε_{ij} represents the error term (Quinn and Keough 2002; Coelho *et al.* 2020). Models were initially fitted assuming both a Poisson and a negative binomial distribution family. Akaike information criteria (AICs) were then generated for each model by using the aictab() function, and were used to select the model that had most appropriate distribution family, whereby the model with the lowest AIC values was deemed most appropriate (Quinn and Keough 2002). The AIC values consistently showed that the negative binomial distribution provided the best fitting models. For each negative binomial model, partial tests within analyses of deviance tables (generated using the Anova() function) were then used to assess the influence of year on catch and effort, whereby Wald tests ($\alpha = 0.05$) examined the null hypothesis that $\beta_1 = 0$ (Quinn and Keough 2002; Bates *et al.* 2015; Duursma and Powell 2016). Where the influence of year in a model was shown to be significant, the summary() function was used to evaluate whether differences between pairs of years were significant at $\alpha = 0.05$. The relevel() function allowed us to switch reference levels in the models, thereby enabling determination of exactly where significant differences among years lay.

Results

The sample sizes (n = number of RFL holders) used in the survey expansion procedure to estimate the catch or effort for a particular year are shown in Table 1. This also represents the sample size for each level of year in the GLMMs. The models showed that year had a statistically significant relationship with the total effort (in fisher days) exerted by recreational fishers within the freshwaters of NSW and ACT ($n = 1225$; $\chi^2 = 8.86$; d.f. = 2; $\Pr(>\chi^2) = 0.01$). Effort was highest in 2013–14 and similarly low in 2017–18 and 2019–20 (Fig. 2). Total retained and released catch across the five key species studied were also significantly different among the 3 years, with both components of catch being similarly high in 2013–14 and 2017–18 and lowest in 2019–20 (Fig. 3, Table 2).

Analysis of the species-specific data showed variation among the five species in terms of the relationship between year and retained or released catch. In terms of retained catch, golden perch and rainbow trout showed significant differences among the years (Fig. 3*a*, Table 2). The retained catch numbers for golden perch were highest in 2013–14 and lowest in 2019–20 (Fig. 3*a*, Table 2). The retained catch numbers for rainbow trout were highest in 2013–14 and 2017–18 and lowest in 2019–20 (Fig. 3*a*, Table 2). Golden perch, Murray cod and rainbow trout showed a significant difference in the numbers released among the 3 years (Fig. 3*b*, Table 2). The released numbers of golden

Table 1. The number of long-term recreational fishing licence holders used in the survey expansion procedure to generate an estimate of catch or effort for a particular year.

Item	2013–14	2017–18	2019–20
Effort data			
Total effort	536	414	275
Catch data			
Australian bass	43	67	26
Brown trout	42	43	34
Golden perch	209	141	68
Murray cod	222	174	108
Rainbow trout	47	50	30
Key finfish combined	375	307	188

This also represents the sample size for each level of year in the GLMMs.

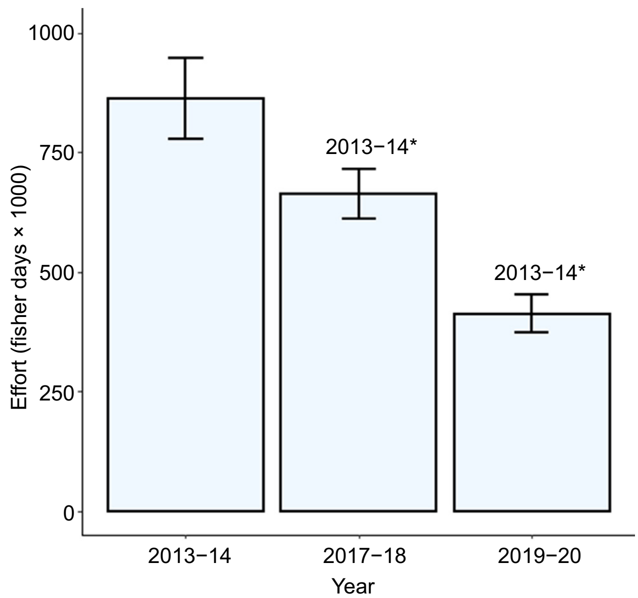

Fig. 2. Estimated total effort (number of fisher days × 1000) exerted through recreational fishing activity in NSW and ACT freshwaters in 2013–14, 2017–18 and 2019–20. Error bars represent 1 s.e. of the total estimated effort. Significant *P*-values ($\alpha = 0.05$) from pairwise comparisons of years are indicated by an asterisk (*), with the associated year in the comparison being labelled.

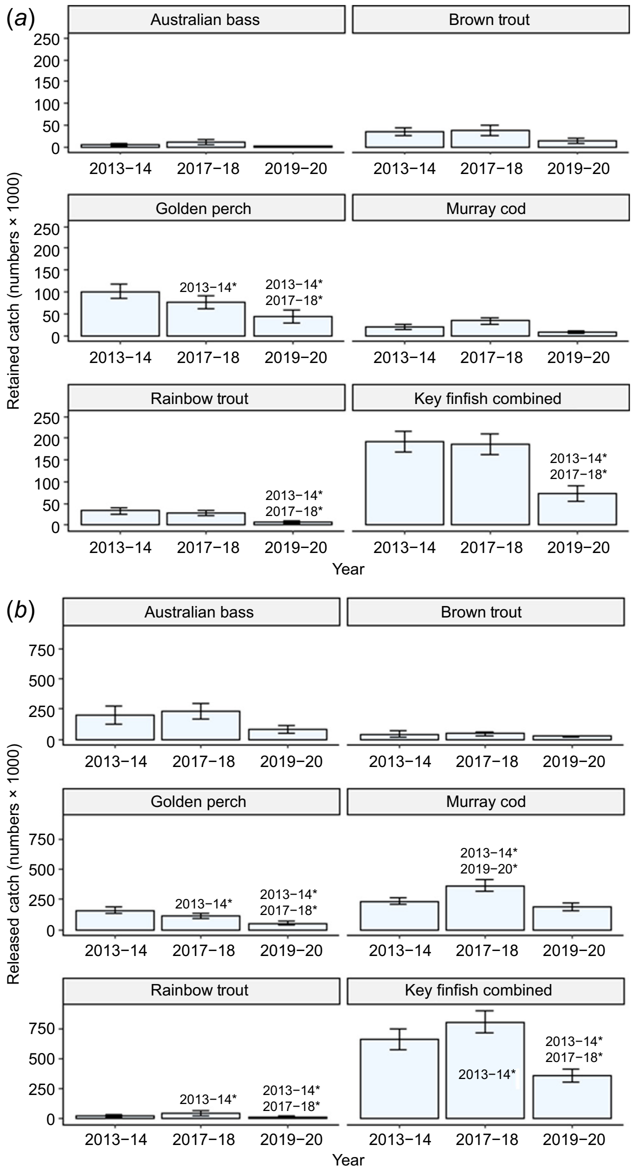

Fig. 3. Estimated freshwater catch (numbers × 1000) of Australian bass, brown trout, golden perch, Murray cod and rainbow trout (*a*) retained and (*b*) released through recreational fishing activity in NSW and ACT in 2013–14, 2017–18 and 2019–20. Also shown is the total catch estimated across these five key finfish species. Error bars represent 1 s.e. of the total estimated effort. Significant *P*-values (α = 0.05) from pairwise comparisons of years are indicated by asterisk (*), with the associated year in the comparison being labelled.

perch were highest in 2013–14 and lowest in 2019/20 (Fig. 3*b*, Table 2). For Murray cod, the released catch numbers were highest in 2017–18 and low during 2013–14 and 2019–20 (Fig. 3*b*, Table 2). For rainbow trout, numbers released were highest in 2017–18 and lowest in 2019–20 (Fig. 3*b*, Table 2). This study's raw data are provided in the Supplementary Tables S1–S7.

Table 2. Analysis of deviance table, showing the results of the generalised linear mixed-effects models used to examine the influence of year (three levels: 2013–14, 2017–18 and 2019–20) on the estimated number of Australian bass, brown trout, golden perch, Murray cod and rainbow trout retained and released through recreational fishing activity in NSW freshwaters during 2013–14, 2017–18 and 2019–20.

Item	Retained catch			Released catch		
	χ^2	d.f.	Pr(>χ^2)	χ^2	d.f.	Pr(>χ^2)
Australian bass (*n* = 136)						
Year	1.72	2	0.42	0.40	2	0.82
Brown trout (*n* = 119)						
Year	2.43	2	0.30	0.53	2	0.77
Golden perch (*n* = 418)						
Year	79.59	2	<2.00E^{-16}*	32.79	2	7.58E^{-08}*
Murray cod (*n* = 504)						
Year	1.38	2	0.50	22.47	2	1.32E^{-05}*
Rainbow trout (*n* = 127)						
Year	7.04	2	0.03*	31.06	2	1.80E^{-07}*
Key finfish combined (*n* = 870)						
Year	27.88	2	8.85E^{-07}*	13.82	2	9.99E^{-04}*

The models, which assumed a negative binomial distribution, applied the Wald test (α = 0.05) to examine the null hypothesis that a $\beta_i = 0$, where β is were the regression coefficients for year. Significant *P*-values are indicated by an asterisk (*).

Discussion

Recreational fishing is a dominant ecosystem service within freshwater bodies in many industrialised nations (Cooke *et al.* 2015; Hunt *et al.* 2016) and understanding how this fishery may be influenced by environmental change under projected climate scenarios is imperative for adaptive management to improve sustainability (Jeanson *et al.* 2021). This study uncovered interannual variability in an expansive freshwater recreational fishery that occurred in conjunction with a series of extreme climate-related environmental events as well as a global pandemic, which altered human behaviour and economies and was linked to anthropogenic environmental changes (Barouki *et al.* 2021; Kumar and Ayedee 2021).

Using examples from this study, the following discussion demonstrates how recreational fisheries may be governed by a unique complex of socio-ecological parameters in a fluctuating world (Arlinghaus *et al.* 2017). However, it must be noted that unequivocal causal relationships cannot be inferred here due to the absence of a randomly selected set of 'control sites' where recreational fishing activity was monitored over the same time period in the absence of the extreme weather events and the societal disruptions in question. Such sites may have enabled application of a before–after–control–impact (BACI) design to the study's analyses (Underwood 1992). Many of the putative relationships and mechanistic processes described below are therefore exploratory in nature.

Changing climates in arid regions such as Australia are anticipated to increase temperatures and decrease rainfall and river flows, resulting in more frequent and extended drought conditions (Reid *et al.* 2019). The past two decades have seen south-eastern Australia experience some of its worst historical droughts (Bond *et al.* 2008; NSW Department of Primary Industries 2019). In freshwater bodies, the combination of elevated temperatures, low oxygen concentrations, increased stratification and low-flow or stagnant conditions observed during drought can lead to isolation of populations within poor or toxic conditions, causing extensive fish-kills (Bond *et al.* 2008; Vertessy *et al.* 2019; Sheldon *et al.* 2022). In the summer periods of 2018–19 and 2019–20, for example, fish-kill events that lead to the loss of millions of native fish in the Darling–*Baaka* River were attributed to hypoxia in a protracted period of low-flow associated with climatic events (Stocks *et al.* 2022). Species observed among the fish-kills included golden perch and Murray cod (Stocks *et al.* 2022). Although the drought-related fish-kills initially affected populations of these species, no significant changes in golden perch and Murray cod abundances were observed 18 months after the fish-kills. The fish-kills would have, therefore, affected the numbers of golden perch and Murray cod caught only for a short period and are unlikely to have caused the fishery declines observed between 2017–18 and 2019–20 (Stocks *et al.* 2022). Even so, the fish-kill events of 2019–20 could still have contributed to some immediate reductions in the recreational catch of these species during that year. Unlike for golden perch, which is kept and released at numbers relatively similar to those of Murray cod (Murphy *et al.* 2020, 2022), catch-and-release practices are common and increasing in popularity for Murray cod because of restrictive regulations (e.g. slot limit of 55–75 cm) and changing attitudes towards voluntary release (Douglas *et al.* 2010; Murphy *et al.* 2020, 2022). For Murray cod, any potential fishery impacts may have, therefore, been easier to detect in the released catch.

In many parts of the world, the frequency and intensity of bushfires has increased in recent years and is projected to escalate with expected climatic and land-use changes (Bixby *et al.* 2015; Nunes *et al.* 2017). In NSW and ACT ($\sim$816 000 km^2), the 'Black Summer' bushfires of 2019–20 burned through $\sim$57 000 km^2. This was nearly double the area of any previous bushfires (Alexandra and Finlayson 2020; Davey and Sarre 2020). Fires may alter freshwater ecosystems by destroying riparian and wetland vegetation, resulting in nutrient mobilisation, changed microclimatic regimes, modified biogeochemistry, and increased runoff, erosion, ash and sediment inputs into waterways (Bixby *et al.* 2015; Nunes *et al.* 2017). Each of these processes may harm aquatic fish and limit availability of prey items (periphyton, phytoplankton and macroinvertebrates) (Bixby *et al.* 2015; Nunes *et al.* 2017). In a North American study, Rosenberger *et al.* (2015) found that older rainbow trout was least abundant in streams within burned watersheds and most abundant in streams with unburned watersheds. Rainbow trout from burned watersheds also had faster growth, earlier maturity and lower lipid content than did those in unburned watersheds (Rosenberger *et al.* 2015). Although our comparisons encompass a shorter period, similar processes may have contributed to the reduced retained and released catch observed for rainbow trout in 2019–20 compared with 2013–14 and 2017–18.

Intensified water cycling associated with a mean global temperature increase of as little as 1.5°C is anticipated to increase the magnitude of floods (Talbot *et al.* 2018). Accordingly, many eastern Australian river systems experienced the highest water levels recorded in the past 30 years following recent flood events (Alexandra and Finlayson 2020; Bureau of Meteorology 2020; Kemter *et al.* 2021). Because flood duration and magnitude are correlated with the availability of suitable fish habitat (King *et al.* 2003) and the density and diversity of piscivore prey species (Luz-Agostinho *et al.* 2009), floods may influence the abundance of fish within important freshwater systems such as the Murray–Darling Basin (Harris and Gehrke 1994; King *et al.* 2003). Using experiments in ponds, Gehrke *et al.* (1993) demonstrated how the lethal concentrations of chemical leachates from riparian vegetation and the hypoxic, acidic conditions that follow flooding can limit habitat suitability and affect freshwater fish such as larval and juvenile golden perch. Even so, it remains unknown whether these adverse conditions affect the presence of adult fish of catchable size to a similar extent. Many other contributing factors may be linked to the significantly lower annual recreational catch numbers observed for golden perch in 2019–2020 than in 2017–18. For example, flood events can affect access to fishing areas, which may decrease angler satisfaction and cause a shift in angling effort (Cahill *et al.* 2018; Birdsong *et al.* 2021).

Following the drought of 2018–19, the bushfires in NSW and ACT were the second step in an entire cascade of adverse environmental impacts on freshwater ecosystem services (Kemter *et al.* 2021). The subsequent heavy rainfall events in 2020 triggered increased surface runoff, transported ash and eroded soil, enhancing sedimentation and deterioration of water quality in freshwater systems (Nunes *et al.* 2017; Silva *et al.* 2020; Legge *et al.* 2022). This link between sequential drivers has been widely described as a 'cascade' of environmental events; characterised by an initial impact that triggers other destructive impacts (Alexandra and Finlayson 2020; Kemter *et al.* 2021). For example, Verkaik *et al.* (2013) demonstrated how the response of freshwater communities to fire is often mediated by interactions with preceding droughts or subsequent flood events. Here, the significantly reduced numbers of retained and released recreational catch across all key finfish species and for specific species such as golden perch in 2019–20 may have been the overall result of cascading environmental impacts (Zampatti *et al.* 2022).

COVID-19 introduced another factor in the cascade of events affecting freshwater ecosystem services and may have influenced fisher behaviour. As governments tried to control the spread of the disease, measures were taken that restricted human movement and led to temporary cessation of non-essential activities, such as tourism and social gatherings (Storen and Corrigan 2020; Fernández-González *et al.* 2021). Although recreational fishing was allowed in NSW and ACT, enforceable public health orders prohibited travel outside an individual's local government area. This reduced intra-state tourism and limited fishing to waterbodies within 5–10 km of a fisher's residence. For metropolitan areas of some Australian jurisdictions, a reduction in fishing participation was subsequently reported (Ryan *et al.* 2021). Similar observations were reported from Canada during the early phases of the pandemic (Howarth *et al.* 2021). State border closures largely precluded inter-state fishers from fishing in NSW and ACT and this was expected to reduce fishing effort further. However, freshwater recreational effort did not decrease significantly between the period directly before the pandemic (2017–18) and the period following its spread (2019–20). In fact, the percentage of freshwater fishing effort that was attributed to fishers from bordering states (Queensland and Victoria) was lower in 2017–18 (~15%) than in 2019–10 (~27%) (Murphy *et al.* 2022). Furthermore, the estimated total number of people that participated in freshwater fishing declined by ~12% from 2013–14 (119 502 ± s.e. 5473 fishers) to 2017–18 (105 613 ± s.e. 5128 fishers), and then declined by only ~8% from 2017–18 to 2019–20 (97 135 ± s.e. 6712 fishers). This suggests a general decline in freshwater recreational fishing activity within NSW and ACT over recent years that may be unrelated to the pandemic and, perhaps to a lesser extent, independent of the extreme weather events. This decline is commensurate with the catch and effort trends observed for recreational fisheries in other Australian jurisdictions (Ryan *et al.* 2019) and for commercial fisheries within NSW (NSW Department of Primary Industries Commercial Fisheries, unpubl. data). The fact that a significant reduction in effort was detected only between 2013–14 and each of 2017–18 and 2019–20 for this study suggests that the effect of the cascade of extreme weather events that occurred after 2013–14 may have had a larger impact on freshwater recreational effort than did the pandemic.

Conclusions and implications for management

The utility of off-site survey data in monitoring interannual variation in recreational fishing, alongside extreme environmental and societal changes was clearly demonstrated here. Environmental change may affect freshwater catch and effort through ecological fluctuations in fish productivity driven by climate-related events, and through alterations to fishers' behaviours resulting from socio-economic changes (van Putten *et al.* 2017; Cooke *et al.* 2021). The overall ongoing threat of such changes to aquatic systems and the way we use them necessitate adaptive initiatives to manage inland recreational fisheries (Hunt *et al.* 2016; Howarth *et al.* 2021; Jeanson *et al.* 2021). Studies such as this facilitate such adaptation because they enable policy makers and scientists to identify problematic trends and make projections that assist in building strategies to address future ecological and societal stressors (Arlinghaus *et al.* 2019; Jeanson *et al.* 2021).

Supplementary material

Supplementary material is available online.

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