Fisheries Research 181 (2016) 84–92

Contents lists available at ScienceDirect

Fisheries Research

journal homepage: www.elsevier.com/locate/fishres

Monitoring boat-based recreational fishing effort at a nearshore artificial reef with a shore-based camera

Krystle Keller[a,b,*], Aldo S. Steffe[c], Michael Lowry[d], Jeffrey J. Murphy[e], Iain M. Suthers[a,b]

[a] Ecology & Evolution Research Centre, School of Biological Earth & Environmental Sciences, University of New South Wales, Sydney 2052, Australia
[b] Institute of Marine Science, Chowder Bay Road, Mosman, NSW 2088, Australia
[c] Fishing Survey Solutions Pty. Ltd., 3 Payten Street, Kogarah Bay, NSW 2217, Australia
[d] Port Stephens Fisheries Institute, Locked Bag 1, Nelson Bay, NSW 2315, Australia
[e] NSW Department of Primary Industries, PO Box 5106, Wollongong, Sydney, NSW 2520, Australia

ARTICLE INFO

Article history:
Received 13 July 2015
Received in revised form 12 February 2016
Accepted 31 March 2016
Handled by Prof. George A. Rose.
Available online 13 April 2016

Keywords:
Fishing intensity
Fishing effort
Artificial reef
Digital imagery
Nearshore fishing

ABSTRACT

Recreational fishing effort was quantified at a 700 m^3 steel artificial reef (AR) off coastal Sydney with a shore-based camera (06:00–18:00) over a two-year period. Stratified random sampling was used to select days for analysis of fishing effort from digital images. Fishing effort estimates derived from the digital images were adjusted to account for visibility bias using information from a validation study. The levels of effort recorded in the first two seasons were low as the AR had been recently deployed and colonization of the AR by sessile organisms and fishes was still occurring. The effort intensity (fisher hours per square kilometer) at the Sydney AR was compared with three South Australian ARs and 14 estuarine fisheries in New South Wales (NSW) to provide context for the study. Effort intensity at the AR was found to be up to 12 times higher than that recorded from some estuarine fisheries in NSW. Conversely, the levels of effort intensity at two South Australian ARs were much higher compared to those at the Sydney AR site in both survey years. Effort intensity comparisons showed that the relative levels of usage at Australian ARs were higher than those recorded from estuarine fisheries. The Sydney AR provides diverse fishing opportunities that may be concentrated in a small area. Camera-based technologies can provide a solution for cost effective monitoring of AR sites, providing the accuracy of fishing effort information derived from camera images is validated. Our study has broad implications for other recreational ARs, including many future deployments planned for eastern Australia.

1. Introduction

A recent advance in technology for monitoring recreational fishing effort involves the use of shore-based camera imagery. This Internet Protocol (IP) webcam system can be used to monitor fishing effort at well-defined access points to a fishery (e.g. boat ramps, choke points) and fixed areas such as jetties, wharves, rock groynes (Ames and Schlindler, 2009; Smallwood et al., 2011, 2012) or the surface area above an artificial reef (AR). Camera imagery can reduce long-term program and monitoring costs and can provide a permanent record of activity which can be accessed and processed after the sampling period is complete (Ames and Schlindler, 2009; Smallwood et al., 2011, 2012). Shore-based cameras are becoming widely used for assessing nearshore recreational fishing. In the study by Ames and Schlindler (2009) two digital cameras were mounted to two separate towers, one adjacent to a jetty and the other at the jetty entrance to capture images of passing boats in Newport, Oregon. Smallwood et al. (2011, 2012) fixed cameras to four large groynes to observe recreational fishers in Perth, Western Australia. Similarly, Van Poorten et al. (2015) attached cameras to trees and other high stable permanent structures to record observations of angling effort in 49 small rural lakes in central British Columbia, Canada.

Monitoring recreational fishing effort at ARs is important for determining their effectiveness for enhancing recreational fishing. These structures are built from various materials to create fish habitat and enhance marine ecosystems as well as increase recreational fishing opportunities (Branden et al., 1994; Carr and Hixon, 1997; Cenci et al., 2011). An understanding of temporal patterns of fishing at ARs is also important because fishing effort is positively correlated to the levels of fishing-related mortality (i.e. harvest and

---

* Corresponding author at: Ecology & Evolution Research Centre, School of Biological Earth & Environmental Sciences, University of New South Wales, Sydney 2052, Australia.

E-mail address: k.keller@unsw.edu.au (K. Keller).

http://dx.doi.org/10.1016/j.fishres.2016.03.025
0165-7836/

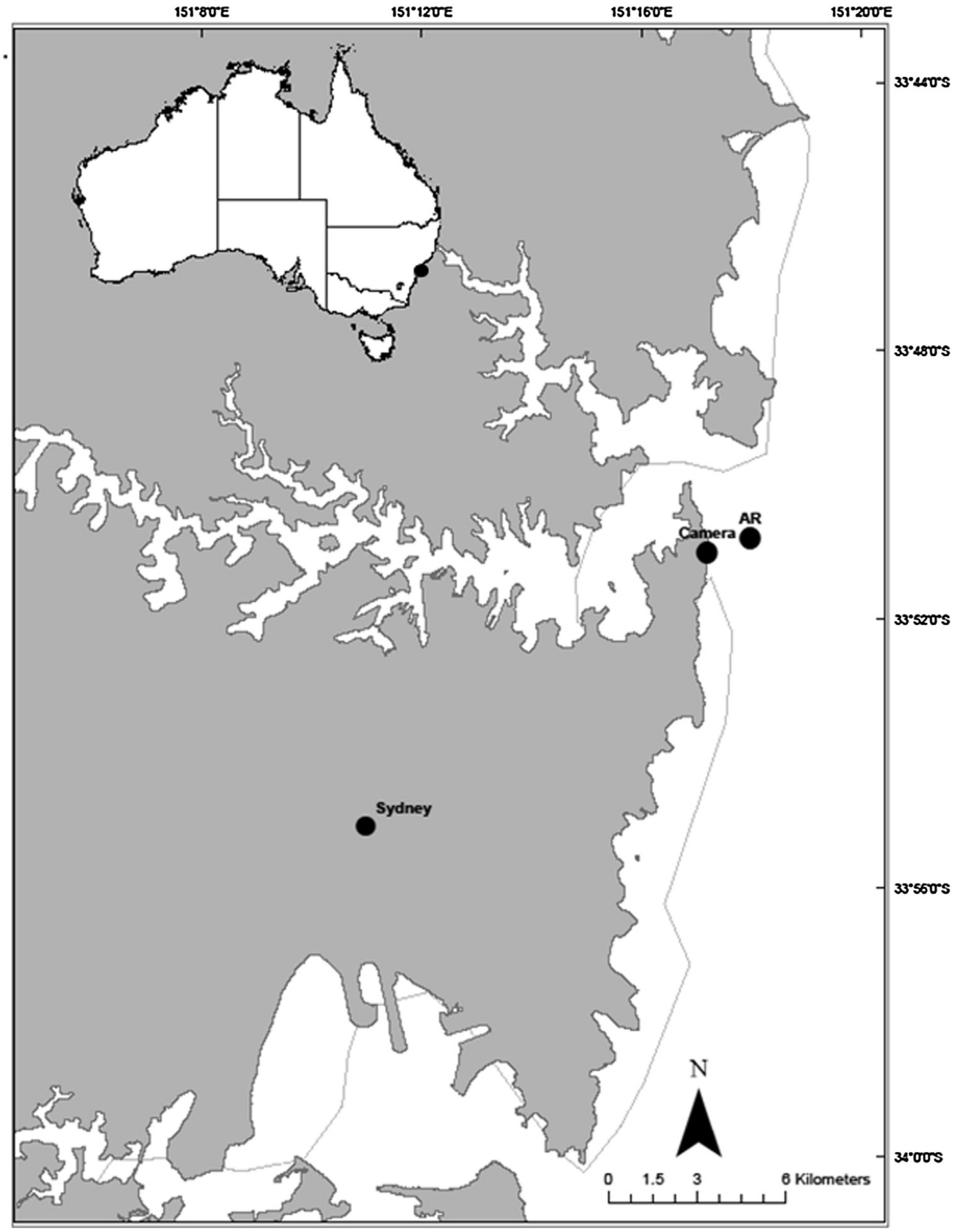

Fig. 1. Location of the Sydney Artificial Reef (AR) and the Old South Head Signal Station (indicated as camera).

release-induced mortality). The success of an AR depends on its ability to enhance fishing opportunities (i.e. increasing the number of fish that are available for capture) while also maintaining the community equilibrium. Few studies have assessed recreational fishing effort at ARs in relation to temporal fishing effort patterns within the region (Buchanan, 1973; McGlennon and Branden, 1994; Tinsman and Whitmore 2006).

Whether ARs enhance the production of fish biomass, or simply attract and aggregate fish is a controversial topic (Bohnsack and Sutherland, 1985; Solonsky, 1985; Bohnsack, 1989; Carr and Hixon, 1997; Folpp et al., 2013; Smith et al., 2015). However, ARs have been viewed as an important resource for preventing localized overfishing by reducing fishing pressure on nearby natural reefs (Bohnsack and Sutherland, 1985; Pickering and Whitmarsh, 1997; Santos and Monteiro, 1998; Folpp et al., 2013; Smith et al., 2015). Furthermore, studies have demonstrated that ARs may provide larger fishing catches compared to natural control reefs (Fabi and Fiorentini, 1994; Carr and Hixon, 1997; Santos and Monteiro, 1998; Whitmarsh et al., 2008; Bortone et al., 2011; Leitão, 2013).

The AR was deployed 1.5 km off the Sydney coast in October 2011, to enhance recreational fishing opportunities. The 42 ton steel structure was designed with many open void spaces and towers that are particularly attractive to fish. The design allows water flows that provide an enhanced supply of nutrient and plankton to the AR ecosystem and can promote the growth of sessile organisms and resident fishes (Connell and Anderson, 1999; Redman and Szedlmayer, 2009). Similar ARs are being deployed elsewhere around Australia, and the need for cost-effective solutions to monitor recreational fishing at these locations is imperative.

The main aim of this study is to estimate recreational boat-based fishing effort at the Sydney AR using a shore-based camera. We also quantify patterns of recreational boat-based fishing

effort at the AR for each season over a two year period, and standardize the fishing effort by area at the AR to allow comparisons

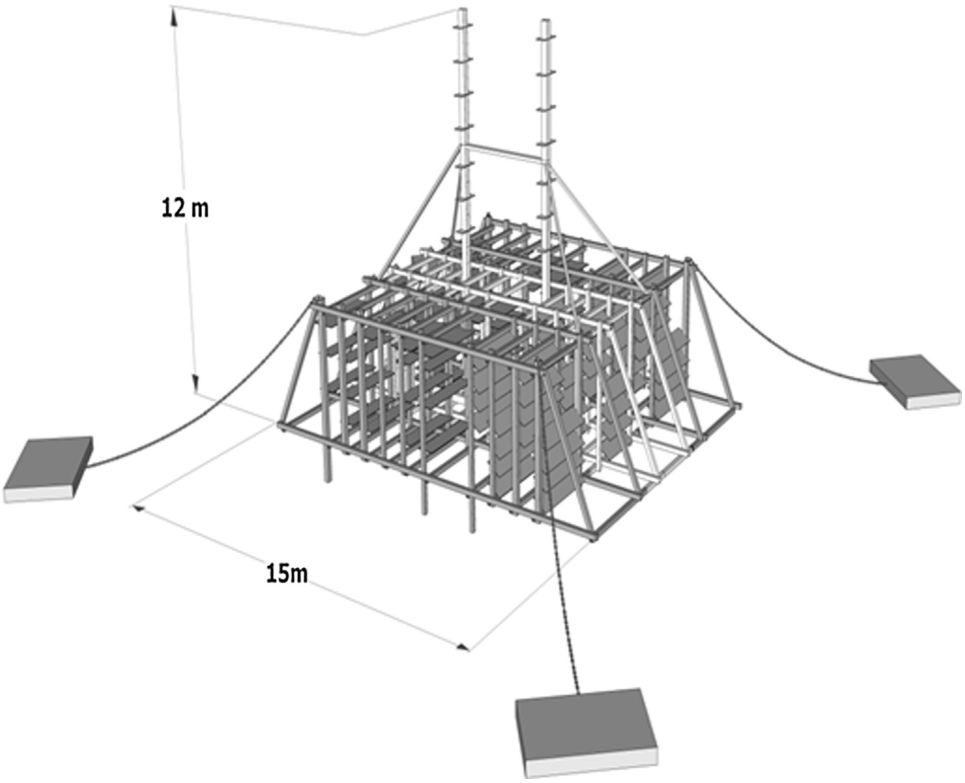

Fig. 2. Schematic of the 42-ton Sydney Artificial Reef (AR).

with other Australian ARs and estuarine fisheries. These comparisons are important for determining the usage by recreational anglers and for assisting managers to determine the economic benefits of implementing more ARs in Australia.

2. Materials and methods

2.1. Artificial reef description

The Sydney AR (33°50.797′S, 151°17.988′E) was deployed in October 2011 in 38 m depth of water, approximately 1.2 km east of 'The Gap', near the southern headland of Sydney Harbor, New South Wales (NSW), Australia (Fig. 1). The steel structure is 12 × 15 m (a footprint of 180 m^2) and 12 m high with two 8 m tall pillars, resulting in a reef volume of 700 m^3 (Fig. 2). The reef is moored at each corner with chain and a 60 ton concrete block. The AR area monitored was calculated using time and position data with a boat Global Position System (GPS). We defined the effective fishing area at and adjacent to the AR so that it included the AR structure, its footprint and an adjacent area that would be used by anglers to target fishes associated with the AR. The effective fishing area adjacent to the AR was determined by considering the direction of prevailing currents and the fishing practices (e.g. drifting, trolling, berleying) used by recreational anglers in the AR area. This area was calculated to be 140 m (north to south) × 400 m (east to west), providing coverage of recreational fishing activity for about 0.056 km^2 (Fig. 3).

2.2. Camera imagery

Recreational fishing effort at the AR was assessed by using a shore-based camera system. A mobotix M24, 3 megapixel (2048 × 1536 pixel resolution, 8× digital zoom, 45° horizontal lens, 8 mm focal length, 2.0 aperture) twin-head IP camera (www.anso.com.au) was fixed to a vantage point at 85 m above sea level at the Old South Head Signal Station—a lighthouse (33°51′1.47″S, 151°17′12.41″E, Fig. 1), to capture digital images of the AR area, at a distance of 1.3 km from the reef. Photographic stills were recorded continuously every minute during a defined period of daylight (06:00–18:00) over two years from 1 June 2012–30 May, 2014 (total of 188,370 images). Images (400–800 KB each) were downloaded in JPEG format and stored on a 32 GB Micro-SD internal card. Images were downloaded on site via a laptop.

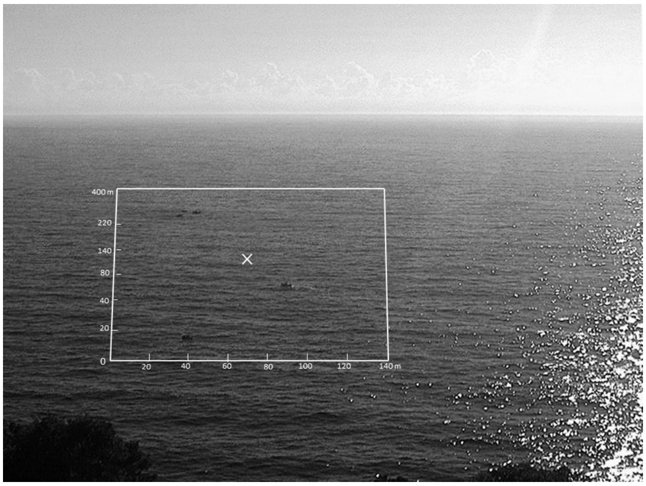

Fig. 3. Camera view of the 0.056 km^2 monitored AR area (140 m (north to south) × 400 m (east to west)). 'X' indicates location of the AR. Note the two different scale bars, the vertical scale bar indicates the increasing distance to 400 m.

A pilot study was conducted prior to the main study to determine the time it took for a non-fishing vessel to transit the AR area. From this study we classified any vessel that remained in the AR

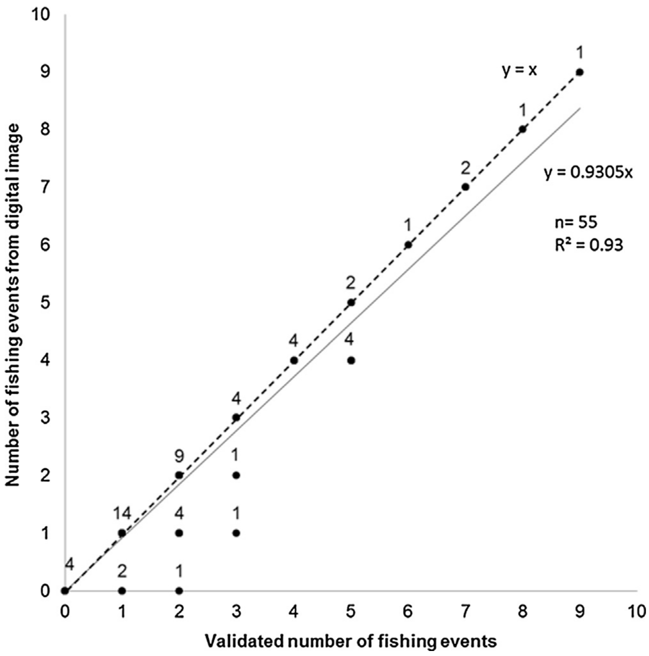

Fig. 4. Regression equation describing the relationship between the number of fishing events counted from camera digital images and from field validated observations at the AR. Number of overlying observations is provided above each data point. The dotted line denotes the y = x equation.

area for 5 min or longer to be fishing, whether they were drift fishing or trolling. A fishing event was therefore defined as a vessel remaining in the AR area for at least 5 min (i.e. 5 frames). All types of vessels, regardless of size, were counted and included in the estimation of fishing effort if they remained within the vicinity of the AR to ensure that all vessels fishing were included. The fishing effort data generated from the digital images was in units of fishing events and boat hours (i.e. the number of hours of boat-based fishing in the AR area). Two people were involved in the analysis of the digital images using Microsoft office picture manager. A reference set of digital images was used to train and standardize the image interpretation of the two readers.

2.3. *Validation of data derived from digital images*

We investigated the potential bias in the data derived from digital images by comparing them with independent counts of fishing events made by observers. A group of marine rescue volunteers based at the old signal station were trained to observe and record the number of fishing events during daylight hours (06:00–18:00) in the AR monitored area. All volunteers used a pair of standard binoculars (7 × 50 Tasco marine series, model 222YRZ) to observe fishing events. Volunteers were trained to observe and count the number of all vessels that remained at least 5 min within the same monitored area. Observers also recorded the time each vessel arrived and departed. The location of the AR was easily observed from the old signal station due to calibrated markings (previously calculated using GPS distance) on the window facing directly east. The observations of fishing effort reported by the volunteer observers were regarded as accurate measures of fishing effort in the AR area. Observer counts were regularly quality controlled by comparing current observations with those of the volunteers during onsite visits to download the data from the camera. These volunteer observations were used to calculate fishing events only. Volunteer observations of fishing effort were done on 55 randomly selected days in the period November 2012 to June 2014 that covered all types of weather conditions. Volunteers also noted when no vessels were observed.

Table 1
The number of days sampled (n) and stratum sizes (N) within day-type, weekdays (WD) and weekend days (WE), during the survey period.

Season/Year	Day-Type	Days sampled (*n*)	Number of days in stratum (*N*)
Winter 2012	WD	10	64
	WE	10	28
Spring 2012	WD	15	64
	WE	15	27
Summer 2012–13	WD	15	60
	WE	15	30
Autumn 2013	WD	15	63
	WE	15	29
Winter 2013	WD	15	64
	WE	15	28
Spring 2013	WD	15	64
	WE	15	27
Summer 2013–14	WD	15	61
	WE	15	29
Autumn 2014	WD	15	62
	WE	15	30

A linear regression forced through the origin was fitted using digital image estimates of daily fishing events on the y-axis and the observer-validated number of daily fishing events on the x-axis. This regression analysis can provide evidence of bias in the digital image generated estimates of fishing events if the slope of the regression line differs significantly from 1.0. To determine whether there was any significant bias, a two-tailed *t*-test was used to test whether the sample value of b (i.e. slope) was different from the expected value of 1 (i.e. when there is no bias) (Sokal and Rohlf, 1981). When bias is detected, it is possible to derive a correction factor using the regression coefficient and its variance derived from the "variance of a quotient" equation (Blumenfeld, 2001). This correction factor was used to adjust the estimates of fishing effort (fishing events) for all strata. See Appendix A for detailed calculations.

2.4. *Effort estimation from digital images*

Stratified random sampling methods were used to select a sample of daily digital images for processing. Days were the primary sampling unit for all strata. By definition, a survey day started at 6 am and ended at 6 pm. Each year was stratified into austral seasons and day-types within seasons (weekdays and weekend days). Public holidays were classified as weekend days. Sample sizes for each base level stratum are given in Table 1. Whenever possible, each month within a season was allocated an equal number of each day-type. Mean daily fishing effort values (i.e. daily mean of the mean number of fishing events per day) and variances for each day-type stratum within each season and within each survey year were calculated. The first survey year spans from June 2012 (when the reef was 8 months old) to May 2013 and the second survey year spans from June 2013 to May 2014. The total fishing effort for each day-type stratum was estimated using a direct expansion method to account for the unsampled fraction of the stratum (Pollock et al., 1994; Cochran, 1977; Steffe and Chapman, 2003). This was calculated by multiplying the mean daily fishing effort values by the number of possible sample days (*N*) in each day-type stratum. Day-type stratum totals were added together to obtain seasonal totals and the seasonal totals were summed to obtain annual estimates. The variances for the base level strata are independent estimates therefore they were summed to obtain seasonal variances. Annual variances were obtained by adding seasonal vari-

Table 2
Study site survey periods, habitat types, distance from shore, depth, location (latitude and longitude) and measured areas (km^2).

Survey location	Survey period	Habitat type	Distance from shore & depth	Latitude	Longitude	Area (km^2)	Source
Sydney artificial reef	Jun 2012–May 2013; Jun 2013–May 2014	Untreated steel designed reef	1.2 km, 38 m	33°50.80′S	151°17.99′E	0.06	This study
Grange artificial reef	Sep 1990–Aug 1991	Tyre modules (1200)	4.3 km, 15 m depth	34°55.1′S	138°24′E	0.08	McGlennon and Branden (1994)
Glenelg artificial reef	Sep 1990–Aug 1991	Tyre modules (900)/sunken vessels (2)	5 km, 18 m depth	34°58.8′S	138°26.4′E	0.19	McGlennon and Branden (1994)
Port Noarlunga artificial reef	Sep 1990–Aug 1991	Tyre modules (650)	2.5 km, 18 m depth	35°05.2′S	138°26.5′E	0.07	McGlennon and Branden (1994)
Northern Lake Macquarie	Mar 1999–Feb 2000; Dec 2003–Nov 2004	All estuarine habitats	–	33°02.0′S	151°37.0′E	60.73	Steffe et al. (2005b)
Southern Lake Macquarie	Mar 1999–Feb 2000; Dec 2003–Nov 2004	All estuarine habitats	–	33°06.0′S	151°35.0′E	43.10	Steffe et al. (2005b)
Swansea channel	Mar 1999–Feb 2000; Dec 2003–Nov 2004	All estuarine habitats	–	33°04.35′S	151°38.40′E	3.23	Steffe et al. (2005b)
Tweed River	Mar 1994–Feb 1995	All estuarine habitats	–	28°14.38′S	153°32.42′E	20.25	Steffe et al. (1996)
Richmond River	Mar 1994–Feb 1995	All estuarine habitats	–	28°52.24′S	153°32.7′E	25.85	Steffe et al. (1996)
Clarence River	Mar 1994–Feb 1995	All estuarine habitats	–	29°27.35′S	153° 9.39′E	101.37	Steffe et al. (1996)
Brunswick River	Mar 1994–Feb 1995	All estuarine habitats	–	28°31.95′S	153°32.0′E	1.58	Steffe et al. (1996)
Sandon River	Mar 1994–Feb 1995	All estuarine habitats	–	29°41.05′S	153°18.17′E	1.49	Steffe et al. (1996)
Wooli River	Mar 1994-Feb 1995	All estuarine habitats	–	29°57.31′S	153°09.09′E	2.17	Steffe et al. (1996)
Mooball Creek	Mar 1994–Feb 1995	All estuarine habitats	–	28°25.75′S	153°33.33′E	0.40	Steffe et al. (1996)
Tuross estuary	Mar 1999–Feb 2000; Dec 2003–Nov 2004	All estuarine habitats	–	36°03.80′S	150° 6.08′E	14.47	Steffe et al. (2005a)
Hawkesbury estuary	Mar 2007–Feb 2008; Mar 2008–Feb 2009	All estuarine habitats	–	33°33.0′S	151°20.15′E	120.81	Steffe and Murphy (2011)
Port Hacking estuary	Mar 2007–Feb 2008; Mar 2008–Feb 2009	All estuarine habitats	–	34°04.31′S	151°09.30′E	11.51	Steffe and Murphy (2011)
Manning River	Mar 2007–Feb 2008; Mar 2008–Feb 2009	All estuarine habitats	–	31°53′14′S	152°39′13′E	25.35	Bucher (2006)

ances together. A correction factor developed during the validation study was applied to the effort and variance estimates to adjust for the under-estimation of fishing effort caused by weather related visibility issues (Blumenfeld, 2001; refer to Appendix A for further details of equations used). This correction factor was applied in units of fishing events.

Fishing effort was converted from fishing events into boat hours and finally into fisher hours, for comparisons with other studies (see Appendix A). We multiplied the estimated total stratum effort (i.e. number of fishing events) independently for each stratum by the daily mean of the mean number of boat hours per fishing event for that stratum (derived from digital images) to obtain estimates of fishing effort in units of boat hours. We then used data from a survey of coastal marine fishing outside the Port Hacking estuary during the period March 2008 to February 2009 (Steffe and Murphy, 2011) to obtain estimates of the daily mean of the mean number of fishers per boat (Appendix A). The conversion of fishing effort from boat hours to fisher hours was done by multiplying the boat hour estimates to the daily mean of the mean number of fishers per boat within each base level stratum. Variances were also adjusted for this unit conversion (Appendix A).

Pairwise comparisons of fishing effort (fisher hours) were made between seasons and years. We used the standard method recommended by Schenker and Gentleman (2001) to calculate an interval for each pairwise comparison. The standard method works by calculating a new interval for each pairwise comparison based on the nominal 95% confidence intervals of the point estimates being compared. The null hypothesis is tested at the convention of $P = 0.05$ and is rejected if and only if the interval does not contain 0 (Schenker and Gentleman, 2001).

2.5. Standardized comparisons of effort intensity

Standardized comparisons of effort intensity per unit of area were made between the AR and three ARs in the coastal waters of South Australia (McGlennon and Branden, 1994; Table 2), and

14 estuarine fisheries in NSW (Steffe et al., 1996, 2005a, 2005b; Bucher, 2006; Steffe and Murphy, 2011; Table 2). We calculated standardized values of effort intensity for each fishery by dividing the total effort (in fisher hours) by area in square kilometers. The point estimates of effort intensity for the boat-based fishery at the AR were used to benchmark this fishery against these other recreational fisheries. These standardized comparisons provide relative measures of recreational usage across different fisheries, with the assumption that the patterns of fishing effort and the average number of fishers per boat per day within the study area had not changed. When necessary, survey areas were calculated in ArcGIS (i.e. Steffe et al., 1996, 2005a, 2005b; Steffe and Murphy, 2011).

2.6. *Comparative coastal fishing effort data from the greater Sydney region*

We wanted to compare the seasonal fishing effort estimates from the Sydney AR in our study to the coastal fishing effort from the Sydney region. Total coastal fishing effort (angling trips ± SE) was calculated using unpublished data from a survey of coastal marine fishing originating from four large waterways in the Sydney area (Hawkesbury, Port Hacking, Botany Bay, Sydney Harbour) from March 2007 to February 2009 (Steffe and Murphy, 2011). Counts of boats returning from the sea were made by observers located on the headlands to these sites. All counts started one hour after sunrise and ended at sunset. The trailer boat data used were corrected to account and remove non-fishing, spearfishing and diving trips (Steffe and Murphy, unpublished data) Random stratified sampling was used. Each survey year was stratified into seasons and day types within season (weekend and weekday strata). Public holidays were regarded as part of the weekend day stratum. Days were the primary sampling unit. Sampling was done on 9 weekdays and 9 weekend days within each season at each site. Further analyses of these data were conducted for the present study to describe the seasonal pattern of coastal fishing effort within the Sydney region.

3. Results

The validation study provided evidence that the data derived from the camera images was significantly biased (two-tailed *t*-test, t = −3.174, df = 53, P < 0.01; Fig. 4). Therefore effort estimates derived from camera images were adjusted to compensate for the underestimation caused by weather-related changes in detectability of vessels. We found that the visibility bias resulted in an underestimate of about 7.5% in the levels of fishing effort. All effort estimates and their measures of precision have been adjusted to correct this bias.

We estimated about 1765 and 2460 fisher hours of fishing effort were expended annually at the AR site during the two survey years (Fig. 5). These annual estimates of effort did not differ significantly between years (P > 0.05, Table 3). However, we found some seasonal differences in fishing effort (Table 3). The level of fishing effort in spring year 2 was significantly greater (P < 0.05) than that of spring year 1. All other same season comparisons between years were not significantly different (Table 3). The first survey year was characterized by low levels of fishing effort in the winter and spring seasons (Fig. 5). We found that in this first survey year that the fishing effort levels recorded during the autumn and summer seasons were significantly greater than those recorded in the winter and spring (Table 3). We found no significant differences in fishing effort among seasons in the second survey year (Table 3).

The Sydney AR received 31,525 and 44,116 fisher hours per square kilometer during the two survey years respectively (Fig. 6). Effort intensity comparisons between the three South Australian ARs (McGlennon and Branden, 1994) and the Sydney AR showed that effort intensity was 2.1 times higher at the Grange AR than at the Sydney AR in the first survey year and 1.5 times higher than at the Sydney AR in the second survey year. Effort intensity was also 1.8 times higher at the Glenelg AR than at the Sydney AR in the first survey year and 1.3 times higher than at the Sydney AR in the second survey year. However, effort intensity was higher at the Sydney AR than at the Port Noarlunga AR during both survey years; fishing intensity at the Sydney AR was 1.7 times and 2.4 times higher than at the Port Noarlunga AR in the first and second survey years respectively.

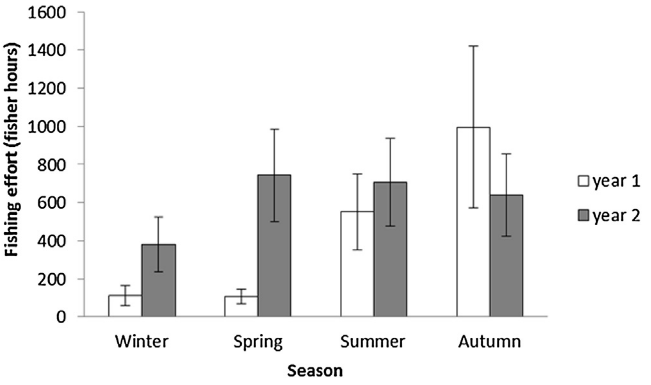

Fig. 5. Seasonal fishing effort (fisher hours ± SE) at the AR for each survey year (year 1 = June 2012–May 2013, year 2 = June 2013–May 2014).

Table 3
Pairwise comparisons of effort between years and seasons over the two year survey period (NS: not significant, p > 0.05; all other comparisons are significant, p < 0.05).

Comparisons	Interval	Result
Total effort (year 1 and 2)	−539.30 to 1949.44	NS
Winter (year 1 and 2)	−34.53 to 569.14	NS
Spring (year 1 and 2)	156.57 to 1117.22	Spring Y2 > Spring Y1
Summer (year 1 and 2)	−442.22 to 752.21	NS
Autumn (year 1 and 2)	−1286.84 to 578.61	NS
Winter and spring (year 1)	−135.85 to 120.92	NS
Winter and summer (year 1)	33.31 to 840.92	Summer Y1 > Winter Y1
Winter and autumn (year 1)	42.82 to 1719.54	Autumn Y1 > Winter Y1
Spring and autumn (year 1)	53.17 to 1724.13	Autumn Y1 > Spring Y1
Summer and autumn (year 1)	−474.96 to 1363.08	NS
Spring and summer (year 1)	46.80 to 842.37	Summer Y1 > Spring Y1
Winter and spring (year 2)	−190.44 to 914.69	NS
Winter and summer (year 2)	−208.77 to 858.39	NS
Winter and autumn (year 2)	−248.40 to 767.92	NS
Spring and autumn (year 2)	−736.92 to 532.19	NS
Summer and autumn (year 2)	−683.15 to 553.04	NS
Spring and summer (year 2)	−692.40 to 617.77	NS

Annual effort intensity was higher at the Sydney AR compared to most estuarine fisheries (Fig. 6). Effort intensity at the AR during both survey years was between 5.8 and 9.1 times higher than effort intensity at northern Lake Macquarie during both survey years and between 4.1 and 6.8 times higher than at southern Lake Macquarie during both survey years (Steffe et al., 2005b). Similarly, effort intensity at the AR in survey year one and two was 2.9 and 4.1 times higher respectively than at the Tweed River; 5.7 and 8 times higher respectively than at the Richmond River; 8.3 and 11.7 times higher respectively than at the Clarence River; 6.7 and 9.4 times higher respectively than at the Sandon River; 2.3 and 3.3 times higher respectively than at the Wooli River; 2.4 and 3.3 times higher respectively than at the Mooball Creek (Steffe et al., 1996), and 5.5 and 7.7 times higher respectively than at the Manning River (Bucher 2006). Effort intensity at the AR during both survey years was also between 5.5 and 9.5 times higher than at the Tuross Estuary during both survey years (Steffe et al., 2005a); between 7.3 and 10.4 times higher than at the Hawkesbury Estuary during both survey years (Steffe and Murphy, 2011) and between 3.8

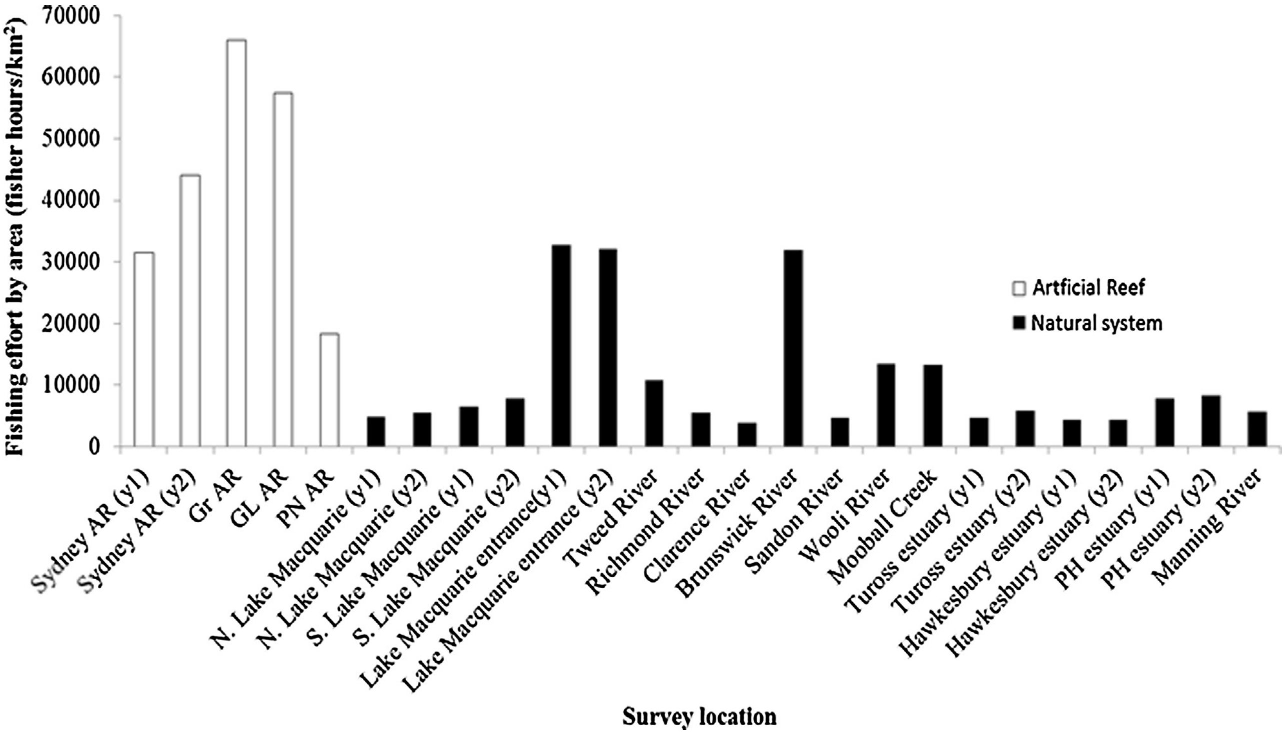

Fig. 6. Comparison of annual fishing effort by area (fisher hours/km^2) between study locations. Sydney AR (this study); South Australia ARs (Gr AR = Grange artificial reef, Gl AR = Glenelg artificial reef, PN AR = Port Noarlunga artificial reef, (McGlennon and Branden, 1994); N. Lake Macquarie = Northern Lake Macquarie, S. Lake Macquarie = Southern Lake Macquarie, Lake Macquarie entrance = Swansea channel (Steffe et al., 2005b); Tweed River, Richmond River, Clarence River, Brunswick River, Sandon River, Wooli River and Mooball Creek (Steffe et al., 1996); Tuross estuary (Steffe et al., 2005a); Hawkesbury estuary (Steffe and Murphy, 2011); PH estuary = Port Hacking estuary (Steffe and Murphy, 2011); Manning River (Bucher, 2006). For details of survey periods see Table 2.

and 5.6 times higher than at the Port Hacking estuary during both survey years (Steffe and Murphy, 2011). In contrast, effort intensity was similar between the Lake Macquarie entrance (Swansea channel), Brunswick River and the AR. Effort intensity at the Lake Macquarie entrance in both survey years was 1.1 times higher than at the AR in the first survey year (Steffe et al., 2005b). However effort intensity at the AR in survey year two was between 1.3 and 1.4 times higher than at the Lake Macquarie entrance during both survey years (Steffe et al., 2005b). Similarly, effort intensity at the Brunswick River was 1.1 times higher than at the AR in survey year one, although effort intensity at the AR in survey two was 1.4 times higher than effort intensity at the Brunswick River (Steffe et al., 1996).

4. Discussion

This study demonstrates that shore-based camera systems are effective for monitoring changes in fishing effort in a recreational fishery that is likely to be enhanced by creating an AR. We have shown the importance of validating data derived from digital camera images because changes in weather conditions (i.e. fog, rainfall, wind speed), can affect the detection of vessels, leading to bias in the data. Similarly, Smallwood et al. (2011, 2012) indicated that the fishing effort estimates from their camera study may have been underestimated due to a visibility bias when calculating fishing activity, since they were unable to identify all people fishing. By incorporating a validation study, we found that the visibility bias inherent in our digital image data would have produced an underestimate of about 7.5% in the level of fishing effort if not corrected. Therefore, future studies that rely on camera technologies to capture effort information for recreational fisheries should routinely include a validation component. The minimal additional cost required for the inclusion of a validation study far outweighs the potential risk of basing important management decisions on less accurate information.

The utility of camera systems for monitoring recreational fishing effort can be expected to increase as technological advances occur. Higher quality lenses may become available at lower cost thereby providing enhanced digital image quality and better resolution of fishing events. Future surveys and monitoring programs may be able to more accurately record daytime fishing and possibly even extend coverage into the night, which was not possible to be quantified with the camera system in this study.

The seasonal pattern of fishing effort observed during the two year survey period was influenced by (a) the length of elapsed time since the deployment of the AR structure; and (b) normally occurring seasonal patterns in the activity of recreational fishers in this region.

The Sydney AR was deployed in October 2011 and the monitoring of fishing effort at this site commenced just 8 months later. It is known that fish colonization on ARs can occur rapidly after initial deployment and that this can continue to increase for a period of about 5 years until all elements of the reef ecosystem are established (Bohnsack and Sutherland 1985; Bohnsack et al., 1994; Scott et al., 2015). Adult fish recruitment at ARs generally occurs within the first year post-deployment and species richness is highest in the austral summer and autumn (Walsh 1985; Folpp et al., 2011; Lowry et al., 2014). Hence, it is likely that a resident assemblage of fish was not fully established at the AR site during the first two seasons of the monitoring period. This may have either discouraged recreational fishers initially or they were unaware of the AR, which could have contributed to the relatively low levels of fishing effort recorded at the AR site during the first winter and spring seasons (11% of the maximum). The pattern of seasonal fishing effort recorded after this initial period at the AR site closely resembled the known pattern of coastal fishing effort within this region. That

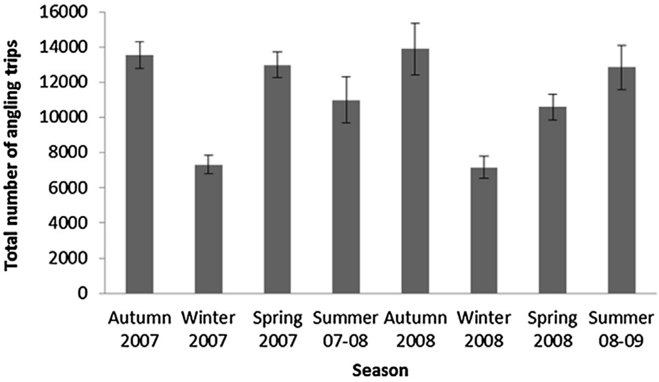

Fig. 7. Total seasonal fishing effort (angling trips ±SE) for line fishing in four greater Sydney coastal systems from March 2007 to February 2009 (Hawkesbury, Port Hacking, Botany Bay, Sydney Harbour).

is, fishing effort levels tend to be lowest in the winter season and higher in the spring, summer and autumn seasons (Fig. 7).

We found that the effort intensity recorded at the AR site was 31,525 and 44,116 fisher hours per square kilometer for years 1 and 2 respectively. This level of usage was up to 12 times more than that recorded from many estuarine fisheries in NSW (Fig. 6) and of similar size to the estuarine fisheries in the Brunswick River and the Lake Macquarie Channel, a relatively shallow but productive area that connects the coastal lagoon to the ocean.

In comparison to the effort intensity at three ARs in South Australia (McGlennon and Branden, 1994; Fig. 5), the level of effort intensity at the Sydney AR was higher than that reported for the Port Noarlunga AR (18,310 fisher hours per square kilometer). However, the two ARs at Grange (66,046 fisher hours per square kilometer) and Glenelg (57,505 fisher hours per square kilometer) had much higher levels of effort intensity than those observed at the Sydney AR site. This indicates that ARs may concentrate effort in a small area. Fish density is often much higher on ARs compared to natural reefs (Bohnsack and Sutherland, 1985; Ambrose and Swarbrick, 1989). Therefore, ARs can be expected to concentrate fishing effort in the vicinity as anglers target the fish assemblages near them.

It is important to note that fishing effort is not homogenous in estuarine fisheries and can also be concentrated in certain areas. Our standardized estimates of effort in these fisheries are calculated on the whole fishery because we did not have data for smaller spatial scales. It is likely that some of the disparity in the fishing intensity comparisons among fisheries can be attributed to these spatial differences.

The relatively high levels of recreational fishing effort per square kilometer observed in both survey years at the AR site indicates that recreational fishing opportunities are likely to have been enhanced at this site. This may be due to a variety of factors which include (a) an increase in biomass as a consequence of additional food being provided by the AR substrate; (b) fish attraction and/or (c) fish movements from adjacent habitats (Bohnsack, 1989; Cresson et al., 2014). Similarly, Santos and Monteiro (1997) found that the local fishing yields were higher at ARs and that fish biomass was enhanced, especially at protection reefs which provide shelter for fish.

ARs have been shown to provide a diverse range of opportunities for recreational fisheries (Bohnsack, 1989; Milon, 1989; Whitmarsh et al., 2008). AR deployments are currently being planned for eastern Australia and are expected to increase in the future, thus making it increasingly important to monitor the activities of recreational fishers at these sites. Camera-based technologies provide a solution for cost-effectively monitoring these AR fisheries.

5. Conclusions

Management of coastal resources at ARs especially in the vicinity of cities needs long-term data on resource use to determine their relative success for enhancing recreational fisheries. As digital imagery improves and costs decline, camera systems will provide this long-term data. Better image analysis technologies are needed to provide cost-effective solutions for monitoring. ARs are popular with recreational fishers and are likely to concentrate effort in a small area. Effort intensity comparisons revealed that ARs received higher levels of recreational usage than many natural estuarine fisheries. Camera-based technologies provide a solution for cost-effective monitoring of AR sites, however it is essential to validate the accuracy of data derived from digital images. Our study has broad application to many other recreational ARs around the world.

Acknowledgements

We would like to thank the NSW recreational fishing trust for providing funds for this project. We would also like to thank Molly Scott for assistance with analyzing the camera imagery, as well as the marine rescue volunteers at the Port Jackson station for assisting with the collection of validation data. This research was funded by an Australian Research Council Linkage Project (LP120100592). K. Keller was also supported by an Australian Postgraduate Award and stipend.

Appendix A. Supplementary data

Supplementary data associated with this article can be found, in the online version, at http://dx.doi.org/10.1016/j.fishres.2016.03.025.

References

Ambrose, R.F., Swarbrick, S.L., 1989. Comparison of fish assemblages on artificial and natural reefs off the coast of southern California. Bull. Mar. Sci. 44 (2), 718–733.

Ames, R.T., Schlindler, E., 2009. Video Monitoring of Ocean Recreational Fishing Effort. Oregon Department of Fish and Wildlife, Oregon, USA, Final Report 05-163 pp. 15.

Blumenfeld, D., 2001. Means and variances. In: Blumenfeld, D. (Ed.), Operations Research Calculations Handbook. CRC Press LLC, Boca Raton, Florida, pp. 6–7.

Bohnsack, J.A., Sutherland, D.L., 1985. Artificial reef research: a review with recommendations for future priorities. Bull. Mar. Sci. 37 (1), 11–39.

Bohnsack, J.A., Harper, D.E., McClellan, D.B., Hulsbeck, M., 1994. Effects of reef size on colonization and assemblage structure of fishes at artificial reefs off southeastern Florida, USA. Bull. Mar. Sci. 55 (3), 796–823.

Bohnsack, J.A., 1989. Are high densities of fishes at artificial reefs the result of habitat limitation or behavioral preference? Bull. Mar. Sci. 44 (2), 631–645.

Bortone, S.A., Brandini, F.P., Otake, S., 2011. Artificial Reefs in Fisheries Management. CRC Press, Boca Raton.

Branden, K.L., Pollard, D.A., Reimers, H.A., 1994. A review of recent artificial reef developments in Australia. Bull. Mar. Sci. 55 (3), 982–994.

Buchanan, C.C., 1973. Effects of an artificial habitat on the marine sport fishery and economy of MurrellsInlet, South Carolina. Mar. Fish. Rev. 35 (9), 15–22.

Bucher, D.J., 2006. Spatial and temporal patterns of recreational angling effort in a warm-temperate Australian Estuary. Geogr. Res. 44 (1), 87–94.

Carr, M.H., Hixon, M.A., 1997. Artificial reefs: the importance of comparisons with natural reefs. Fisheries 22 (4), 28–33.

Cenci, E., Pizzolon, M., Chimento, N., Mazzoldi, C., 2011. The influence of a new artificial structure on fish assemblages of adjacent hard substrata. Est. Coast. Shelf Sci. 91, 133–149.

Cochran, W.G., 1977. Sampling Techniques. John Wiley and Sons, New York.

Connell, S.D., Anderson, M.J., 1999. Predation by fish on assemblages of intertidal epibiota: effects of predator size and patch size. J. Exp. Mar. Biol. Ecol. 241 (1), 15–29.

Cresson, P., Ruitton, S., Harmelin-Vivien, M., 2014. Artificial reefs do increase secondary biomass production: mechanisms evidenced by stable isotopes. Mar. Ecol. Prog. Ser. 509, 15–26.

Fabi, G., Fiorentini, L., 1994. Comparison between an artificial reef and a control site in the Adriatic Sea: analysis of four years of monitoring. Bull. Mar. Sci. 55 (2(3)), 538–558.

Folpp, H., Lowry, M., Gregson, M., Suthers, I.M., 2011. Colonization and community development of fish assemblages associated with estuarine artificial reefs. Braz. J. Oceanogr. 59, 55–67 (special issue 1 (CARAH)).

Folpp, H., Lowry, M., Gregson, M., Suthers, I.M., 2013. Fish assemblages on estuarine artificial reefs: natural rocky-reef mimics or discrete assemblages? PLoS One 8 (6), e63505.

Leitão, F., 2013. Artificial reefs: from ecological processes to fishing enhancement tools. Braz. J. Oceanogr. 61, 77–81.

Lowry, M.B., Glasby, T.M., Boys, C.A., Folpp, H., Suthers, I., Gregson, M., 2014. Response of fish communities to the deployment of estuarine artificial reefs for fisheries enhancement. Fish. Manag. Ecol. 21 (1), 42–56.

McGlennon, D., Branden, K.L., 1994. Comparison of catch and recreational anglers fishing on artificial reefs and natural seabed in Gulf St. Vincent, South Australia. Bull. Mar. Sci. 55 (2–3), 510–523.

Milon, J.W., 1989. Economic evaluation of artificial habitat for fisheries: progress and challenges. Bull. Mar. Sci. 44 (2), 831–843.

Pickering, H., Whitmarsh, D., 1997. Artificial reefs and fisheries exploitation: a review of the attraction versus production debate, the influence of design and its significance for policy. Fish. Res. 31 (1–2), 39–59.

Pollock, K.H., Jones, C.M., Brown, T.L., 1994. Angler Survey Methods and Their Applications in Fisheries Management. American Fisheries Society, Bethesda, Maryland, USA.

Redman, R.A., Szedlmayer, S.T., 2009. The effects of epibenthic communities on reef fishes in the northern Gulf of Mexico. Fish. Manage. Ecol. 16 (5), 360–367.

Santos, M.N., Monteiro, C.C., 1998. Comparison of the catch and fishing yield from an artificial reef system and neighbouring areas off Faro (Algarve, south Portugal). Fish. Res. 39 (1), 55–65.

Schenker, N., Gentleman, J.F., 2001. On judging the significance of differences by examining the overlap between confidence intervals. Am. Stat. 55 (3), 182–186.

Scott, M.E., Smith, J.A., Lowry, M.B., Taylor, M.D., Suthers, I.M., 2015. The influence of an offshore artificial reef on the abundance of fish in the surrounding pelagic environment. Mar. Freshw. Res. 66, 429–437.

Smallwood, C.B., Pollock, K.H., Wise, B.S., Hall, N.G., Gaughan, D.J., 2011. Quantifying recreational fishing catch and effort: a pilot study of shore-based fishers in the Perth Metropolitan area. Fisheries Research Report No. 216. Final NRM Report—Project No. 09040. Department of Fisheries, Western Australia.

Smallwood, C.B., Pollock, K.H., Wise, B.S., Hall, N.G., Gaughan, D.J., 2012. Expanding aerial–roving surveys to include counts of shore-based recreational fishers from remotely operated cameras: benefits, limitations, and cost effectiveness. North Am. J. Fish. Manage. 32 (6), 1265–1276.

Smith, J.A., Lowry, M.B., Suthers, I.M., 2015. Fish attraction to artificial reefs not always harmful: a simulation study. Ecol. Evol., 1–13.

Sokal, R.R., Rohlf, F.J., 1981. Biometry: the Principles and Practice of Statistics in Biological Research, 2nd ed. W. H Freeman and Co., New York.

Solonsky, A.C., 1985. Fish colonization and the effect of fishing activities on two artificial reefs in Monterey bay, California. Bull. Mar. Sci. 37 (1), 336–347.

Steffe, A.S., Chapman, D.J., 2003. A survey of daytime recreational fishing during the annual period, March 1999 to February 2000 in Lake Macquarie, New South Wales. In: New South Wales Fisheries Final Report Series No. 52. New South Wales Fisheries, Cronulla.

Steffe, A.S., Murphy, J.J., 2011. Recreational fishing surveys in the greater Sydney region. In: Fisheries Final Report Series No. 131. New South Wales Department of Primary Industries, Cronulla.

Steffe, A.S., Staines, J.F., Murphy, J.J., 1996. Recreational use of fisheries resources in Northern New South Wales. In: Project S7-Final Report. New South Wales Fisheries, Cronulla.

Steffe, A.S., Murphy, J.J., Chapman, D.J., Barrett, G.P., Gray, C.A., 2005a. An assessment of changes in the daytime, boat-based, recreational fishery of the Tuross Lake estuary following the establishment of a 'Recreational Fishing Haven'. In: Fisheries Final Report Series 81. New South Wales Department of Primary Industries, Cronulla.

Steffe, A.S., Murphy, J.J., Chapman, D.J., Gray, C.A., 2005b. An assessment of changes in the recreational fishery of Lake Macquarie following the establishment of a 'Recreational Fishing Haven'. In: Fisheries Final Report Series No. 79. New South Wales Department of Primary Industries, Cronulla.

Tinsman, J.C., Whitmore, W.H., 2006. Aerial flight methodology to estimate and monitor trends in fishing effort on Delaware artificial reef sites. Bull. Mar. Sci. 78 (1), 167–176.

Van Poorten, B.T., Carruthers, T.R., Ward, H.G.M., Varkey, D.A., 2015. Imputing recreational angling effort from time-lapse cameras using an hierarchical Bayesian model. Fish. Res. 172, 265–273.

Walsh, W.J., 1985. Reef fish community dynamics on small artificial reefs: the influence of isolation, habitat structure, and biogeography. Bull. Mar. Sci. 36 (2), 357–376.

Whitmarsh, D., Santos, M.N., Ramos, J., Monteiro, C.C., 2008. Marine habitat modification through artificial reefs off the Algarve (southern Portugal): an economic analysis of the fisheries and the prospects for management. Ocean Coast. Manage. 51 (6), 463–468.