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Quantifying welfare gains of coastal and estuarine ecosystem rehabilitation for recreational fisheries

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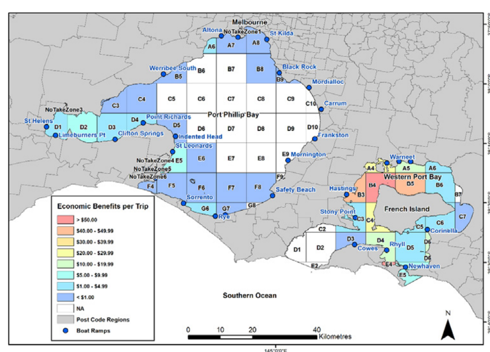
HIGHLIGHTS

- Welfare gain across locations varied due to heterogeneous coverage of seagrass.
- The benefits ranged from near-zero to AU $19.2 for 10% habitat increase scenario.
- The annual benefit could be up to AU $6.2 million for this 10% increase scenario.
- The benefits ranged from near-zero to AU $85.5 for 30% habitat increase scenario.
- The annual benefit could be up to AU $22 million for this 30% increase scenario.

GRAPHICAL ABSTRACT



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ABSTRACT

Coastal and estuarine ecosystems, such as mangroves, tidal marshes and seagrass meadows, provide a range of ecosystem services, but have seen extensive degradation and decline. Effective protection and rehabilitation of coastal ecosystems requires an understanding of how efforts may improve associated ecosystem services. In this study, we present a spatially-explicit angler catch function to predict boat-based recreational catch as a function of ecosystem and angler characteristics. We developed a choice model to investigate where recreational anglers launch their boats and fish in southeast Australia. By linking the recreational catch models with a choice model, we were able to quantify welfare gains of ecosystem rehabilitation. We found welfare gains across fishing locations varied widely due to heterogeneous coverage of seagrass. The welfare gains of different fishing locations ranged from near-zero in areas of low seagrass coverage, to AU $19.18 (10% increase in seagrass area) and to AU $85.55 (30% increase) per trip in location of high seagrass coverage. Given two million fishing trips occurring per year in Port Phillip Bay, and one million in Western Port, the aggregated welfare gain could scale up to AU $6.2 million with a 10% increase in seagrass coverage, and AU $22 million per annum with a 30% increase in seagrass. We also calculated the welfare loss associated with total loss of seagrass ecosystem in each fishing location to represent the current value, which varied significantly, ranging from near-zero in some locations to AU $87.47 per trip in other locations. Over the past several decades, the bay-wide seagrass ecosystem has dropped by 36.7% in Western Port, resulting in potential welfare loss of an estimated AU $ 86.7 million per annum. Our analyses provide insightful spatial policy implications for coastal and marine ecosystem rehabilitation in the region.

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1. Introduction

Coastal and estuarine ecosystems, such as mangroves, tidal marshes and seagrass meadows, are highly productive and provide people with critical ecosystem services; these include supporting fisheries, coastal protection, carbon sequestration and recreational benefit (Barbier et al., 2011; Boyer and Polasky, 2004; Brander et al., 2006), which is often measured in terms of society's willingness to pay (Loomis and White, 1996; Mendelsohn and Olmstead, 2009; Bockstael et al., 2000). Despite their ecological and economic importance, these systems are often threatened by ongoing loss and degradation (Valiela et al., 2001; Lotze et al., 2006; Duarte, 2002). Restoration can be costly, with median cost in developed countries estimated at $383,000 for seagrass, $52,000 for mangroves and $151,000 for tidal marsh US $ ha^{-1} (Bayraktarov et al., 2016). Given the high stakes and often limited budget available to achieve conservation objectives, the potential ecosystem services resulting from different rehabilitation efforts provides important information for resource managers.

Ecosystem services linked to fisheries are crucial services for developing and evaluating coastal rehabilitation projects, due to the contribution of coastal and marine fisheries to global and regional economies (Ernst and Young, 2015). There are two major approaches to economic evaluation of marine and coastal ecosystems. The first approach is bioeconomic modelling, which requires an understanding of underlying ecosystem dynamics, where ecosystem is integrated into mathematical models, for example by improving carrying capacity (Barbier and Strand, 1998), intrinsic growth rate (Kahn, 1987), recruitment (e.g. Sanchirico and Springborn, 2011), and fisheries biomass (e.g. Bell, 1997; Lynne et al., 1981); such approaches are typically applied to commercial fisheries.

An alternative approach to quantifying ecosystem value for recreational fisheries is through micro-econometric modelling of recreational behaviour using empirical data. Included in this approach are studies that integrate recreational demand model with population dynamics, for instance Massey et al. (2006); Newbold and Massey (2010) and Gao and Hailu (2018). Recreational fishing is a non-market activity, and its value is therefore not observable from market transactions. The value of recreational activity reflects the utility that anglers derive from the recreational opportunities and should be estimated with non-market evaluation techniques (Freeman, 1993). Welfare gain derived from these evaluation techniques is also termed "economic benefit" in economic literature such as Loomis and White (1996), Mendelsohn and Olmstead (2009), Bockstael et al. (2000). Here these two terms are used interchangeably. Multiple approaches have been used to estimate non-market values. The travel cost method uses the price of accessing recreational opportunities to estimate the consumptive value of some environmental goods. Travel cost is the cost of people traveling from their homes to a recreational location and the corresponding opportunity cost of time. It represents the "price" people incur from participating in that recreation activity (Phaneuf and Smith, 2005).

The random utility model (RUM) has been widely used to describe recreational demand (Phaneuf and Smith, 2005). Random utility models, introduced by McFadden (1974), are consistent with economic theories to resolve the discrete or continuous choice problem that agents face, for instance, the choice of health care provider (Borah, 2006) and transportation mode (Greene and Hensher, 2007) and supermarket consumer goods (Berry, 1994). For the application of recreational fisheries, RUMs aim to predict the demand of recreational services and describe the choice of recreational site as a function of angler's characteristics and site attributes. The estimated RUM can be used to calculate spatially explicit information with regard to welfare gains of improving site quality such as water quality (Bockstael et al., 1987; Kaoru, 1995; Massey et al., 2006) and ecosystem extent (Ahn et al., 2000; Knoche et al., 2015). For example, Ahn et al. (2000) developed a logit model to estimate the welfare cost resulting from loss of trout habitat, and Knoche et al. (2015) also used a similar model to estimate the ecosystem restoration benefits for pheasant hunters in Michigan. There are also contingent valuation (CV) studies that use choice experiments (stated preference) where survey participants are asked to provide their willingness-to-pay (WTP) for some environmental amenities.

Recreational fishing is an important pastime in Australia with substantial economic value and contribution to local economies. An estimated 830,000 participants in 2013/2014 made 6.1 million fishing trips across the state of Victoria, with the Victorian industry alone estimated to support 16,257 direct jobs, generating AU $2.6 billion direct output and contributing to regional economic growth (Ernst and Young, 2015). Using a benefit transfer method based on several studies in Australia and the United States, one study used $444 per fishing trip to indicate the economic value of recreational fishing (Ernst and Young, 2015). Other research on Victorian recreational fisheries applied the travel-cost method using travel-cost and visitation information to estimate recreational demand and infer willingness-to-pay for access to fishing sites. For example, Hunt et al. (2017) used the travel cost method, and estimated recreationist's willingess-to-pay to be AU $84-AU $291 per person per day (excluding opportunity cost of time) in Lake Purrumbete in the Western District of Victoria. Recreational anglers were willing to pay AU $48 (equivalent to AU $60.7 in 2017) per trip in Victorian coastal areas (URS, 2007), with a total economic benefit for recreational fishing of AU $32.46 million across the two major urban bays in southeastern Australia, Port Phillip Bay and Western Port (Parks, 2015). These studies did not evaluate the value of natural ecosystems in supporting the fisheries, despite the significant relationship between fish stocks and ecosystems (i.e., mangrove, tidal marsh and seagrass ecosystems) (Meynecke et al., 2007; Saenger et al., 2013).

In this study, we used random utility maximization modelling to investigate the recreational fishers' site choice to evaluate welfare gain of ecosystem rehabilitation, taking advantage of a rich microeconomic dataset collected by the Victorian Fisheries Authority on recreational fishers in the greater Melbourne area in Victoria (angler surveys comprising of 3597 fishing trips). Specifically, we examined how coastal and estuarine ecosystem rehabilitation or loss could affect boat-based recreational fishing in Port Phillip Bay and Western Port in southeast Australia.. The area of each ecosystem was treated as an attribute associated with each fishing location and evaluated in a spatially explicit model of recreational catch, along with angler's characteristics including: rod count, angler count, hours fished, and whether the anglers targeted specific fish species. By linking this spatially explicit recreational catch model with a random utility model, we were able to estimate the choice models and investigate how locations would impact recreationists' welfare from scenarios of ecosystem rehabilitation or loss. Such information is valuable for environmental management because coastal and estuarine ecosystems (e.g. mangrove, tidal marsh and seagrass) have been subject to significant loss due to threats in this study area, including agricultural and urban development, increases in sediment loads and declining water quality over past the 100 years (Walker, 2011; Morris, 2013; Boon et al., 2015).

2. Material and methods

2.1. Data

We used data from the 2014–2016 Marine Survey of Recreational Fishing conducted by the Victorian Fisheries Authority (VFA) in Port Phillip Bay (PPB) and Western Port (WP) in southeast

Australia, comprising 3597 fishing trips. The recreational survey has a response rate of over 95%, and focuses only on boat-based recreational angling, excluding shore-based angling. It is only conducted at boat ramps on weekend days during peak fishing time from November to April each year due to limited budget, targeting the peak fishing time (Spring to Autumn) in these embayments. Sampling frequency per area/zone is set at a minimum of fortnightly over the 6-month survey period. The survey comprises a range of information including angler demographics, effort, gear, catch, and fishing locations, including attributes such as bottom type (Ryan and Conron, 2019, also see the survey in the appendix). The data we used to predict the catch rate includes the number of fishers per trip, hours spent fishing, total number of rods in a fishing trip, and target species information in a fishing trip (see Table 1). Each angler is required to purchase a fishing licence in Victoria and is provided with a recreational fishing guide by the Victoria Fishery Authority. This booklet includes detailed information such as locations and species open to fishing, and regulations including bag and size limits. Line fishing is the most common fishing method for anglers who can target multiple species. The top five species that recreational anglers catch in PPB and WP are southern calamari (*Sepioteuthis australis*), snapper (*Pagrus auratus*), sand and blue-spotted flathead (*Platycephalus bassensis* and *Cymbacephalus nematophthalmus*), King George whiting (*Sillaginodes punctatus*), and gummy shark (*Mustelus antarcticus*); see Appendix Table A1 for a comprehensive list of recreational species in PPB and WP.

The Victorian Fisheries Authority conducted the interviews at 20 active boat ramps in Port Phillip Bay and nine active boat ramps in Western Port, representing the launching points of the majority of fishing trips by recreational anglers residing in the greater Melbourne metropolitan area. To record spatially-explicit fishing effort in the surveys, the bays are delineated into 40 fishing blocks (based on a 5-min grid, ~9 km by 9 km) that vary in size due to overlap with the coastlines (Fig. 1; Morris and Ball, 2006). The delineated fishing block is directly defined as a fishing location.

Table 2 gives a brief description of the active boat ramps sampled and survey records. The five most popular boat ramps include Clifton Springs, Limeburners Point, Werribee, Carrum from Port Phillip Bay, and Hastings from Western Port. Due to the challenges in navigating the head of Port Phillip Bay, recreational anglers normally do not launch within one bay to access the other. Therefore, we assumed a recreational vessel launched from a boat ramp could only get access to any location within the same bay. One location originally coded E4 in Morris and Ball (2006) was listed as non-take zone. Therefore, there are a total of 1007 potential alternative ramp-location combinations for each angler to choose with 800 from within Port Phillip Bay and 207 within Western Port.

We estimated the proportion of each location cell occupied by different ecosystem types using existing habitat classifications from the Departments of Environment Land, Water and Planning classified using the Combined Biotope Classification Scheme (CBiCS) (Edmunds and Flynn 2015) at the ecosystem complex level (tier 3). CBiCS provides a hierarchical classification scheme of marine ecosystems, including mangrove, tidal marsh, seagrass, circalittoral rock, infralittoral rock, seaweed, reef, mud, sand, littoral and sublittoral sediment (Table 1). The proportions of each ecosystem type per location vary substantially between locations and bays, in particular with regard to mangrove, which has a very limited distribution in PPB.

Overall travel distance for each angler was calculated in ArcGIS 10.5 (ESRI 2017) using the Network Analyst extension. The travel distance was calculated across two separate networks: over land and over water. The land network was developed using the Vicmap Transport Road Network from the state of Victoria Department of Environment, Land, Water and Planning (DELWP). The start points in the road network were the nearest road to the centroid of each post-code polygon as a proxy for fisher's home location. The end points were the boat ramp points targeted for Marine Surveys. The water network for PPB and WP was created using georeferenced nautical charts of those areas and manually drawing polylines within navigable channels from each boat ramp to the centroid of each fishing location. The land and water networks were then run separately to develop two distance matrices: (1) minimum distance from each postal code centroid to each boat ramp; and (2) minimum distance from each boat ramp to each fishing location centroid in the respective bays. We assume income is representative of average values for postcode locations and a normalised travel speed of 72 km per hour on land and 15 knots on water. A lower speed and higher income will increase the estimated travel cost. To derive a proxy of each angler's income category, we used the median income level of the postcode area from the Australian Bureau of Statistics. The opportunity cost of travel time is approximated with one third of an angler's income divided by 2080 which reflects the hours a person works in a year (Parsons et al., 2000). We used Australian Consumer Price Index (http://www.abs.gov.au) to revise the estimates from Raguragavan et al. (2013) and Rolfe et al. (2011) for the per kilometre travel cost on land to be AUS $0.53/km and AUS $0.81/km over water. The total travel cost variable is then calculated as,

Table 1
Summary statistics of variables used in estimation.

Variables	Mean	SD	Min	Max
Total catch (in a fishing trip)	5.55	9.06	0	96
Total Angler count (in a fishing trip)	2.10	0.85	1	7
Hour fished (in a fishing trip)	4.25	2.22	0	16
Total number of rod count (in a fishing trip)	3.17	1.37	1	9
Species targeted (binary)	0.15	0.36	–	–
Tidal marsh*	0.0035	0.02	0	0.26
Mangrove*	0.0056	0.025	0	0.31
Seagrass*	0.124	0.122	0	0.62
Rock*	0.03	0.085	0	0.64
Sediment*	0.22	0.33	0	0.98
Seaweed*	0.025	0.075	0	0.4
Reef*	0.0037	0.013	0	0.14
Mud*	0.3	0.34	0	1
Sand*	0.26	0.285	0	1

Target (whether angler targeted species or not) is a binary variable. Percentage cover of ecosystem equates to cover of ecosystem in a location divided by the size of that location.
*Indicators % area of ecosystems

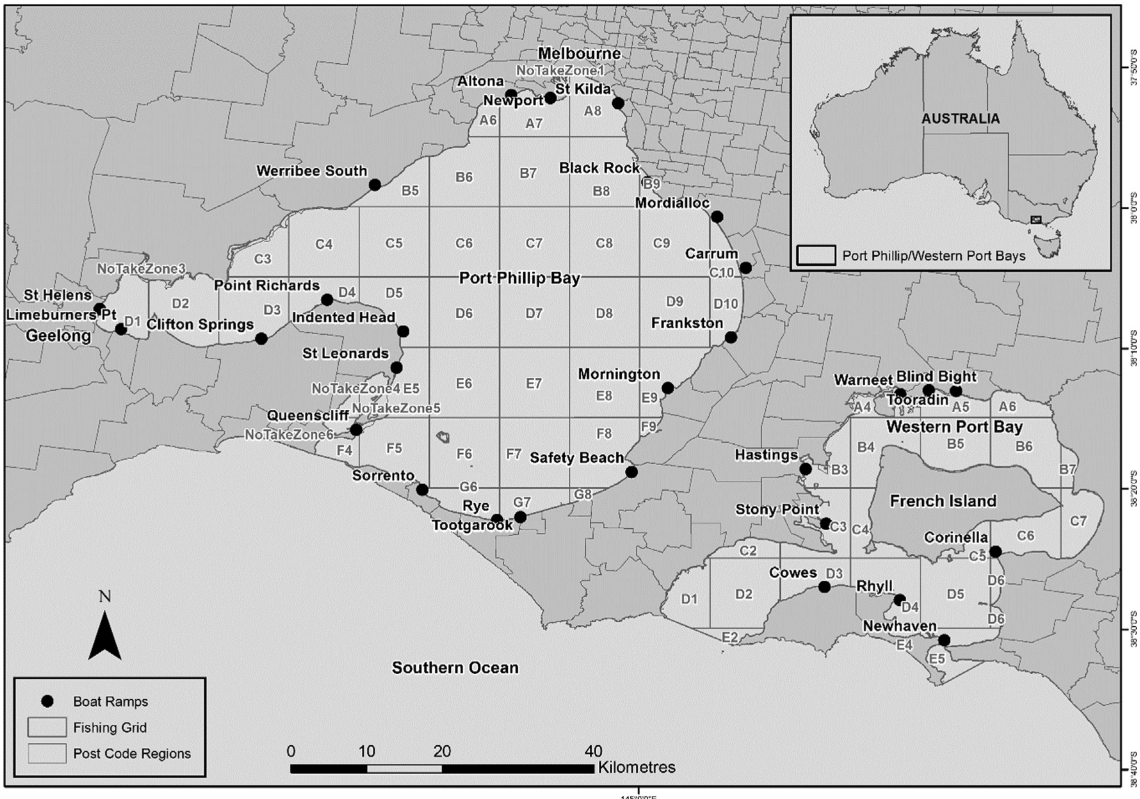

Fig. 1. Port Phillip and Western Port Bays maps. Location code are written within each fishing location.

Table 2
Active boat ramps and percentage of overall visits recorded in the survey visits (%).

ID	Boat ramp	Visit (%)	Bay	ID	Boat ramp	Visit (%)	Bay	ID	Boat ramp	Visit (%)	Bay
01	Clifton Springs	11.5	PPB	11	Newhaven	4.16	WP	21	Mornington	1.56	PPB
02	Limeburner Point	8	PPB	12	St. Leonards	3.25	PPB	22	Cowes	1.46	WP
03	Werribee	7.5	PPB	13	Rhyll	3.1	WP	23	Black Rock	1.06	PPB
04	Carrum	6.7	PPB	14	Point Richards	2.81	PPB	24	Safety Beach	1.06	PPB
05	Hastings	6.53	WP	15	Queenscliff	2.5	PPB	25	Indented Head	0.94	PPB
06	Stony Point	5.97	WP	16	Tooradin	2.36	WP	26	Mordialloc	0.81	PPB
07	Altona	5.82	PPB	17	Sorrento	2.25	PPB	27	Frankston	0.56	PPB
08	St. Helens	4.75	PPB	18	St. Kilda	1.94	PPB	28	Tootgarook	0.56	PPB
09	Warneet	4.56	WP	19	Rye	1.75	PPB	29	Blind Bight	0.51	WP
10	Corinella	4.2	WPB	20	Newport	1.75	PPB				

tc=\$0.53*km travelled on land

+\$0.81* km travelled on water
+ \$0.33* total hours travelled*annual income/2080

2.2. The catch model

We used zero-inflated negative binomial (ZINB) regression to model catch, y_i, reported by angler *i*. The ZINB model allows for the excessive zeroes and over-dispersal in the catch data. The ZINB model comprises two parts: the first part is where catch takes a value of zero (i.e. no catch); the second one takes nonnegative values (including zero), which is generated with a negative binomial distribution (Cameron and Trivedi, 2005):

$$p(Y(i) = y(i)|\mu(i), \alpha) = \frac{\Gamma(y(i) + \alpha^{-1})}{\Gamma(\alpha^{-1})\Gamma(y(i) + 1)} \left(\frac{1}{1 + \alpha\mu(i)}\right)^{\alpha^{-1}} \left(\frac{\alpha\mu(i)}{1 + \alpha\mu(i)}\right)^{y(i)} \tag{1}$$

Where $\mu(i)$ and α are the mean and overdispersal parameters, respectively, and estimated with maximum likelihood. The mean (expected positive catch, $Y_g^E(i)$) is modelled with $X_g(i)$, the vector of regressors, including number of fishers within that trip, hour spent on fishing, total number of rods, and whether angler targeted any specific species (binary variable), percentage cover of varied ecosystem (tidal marsh, mangrove, seagrass, rock, sediment, seaweed, reef, mud and sand). We assume that the aggregated catch model can roughly approximate change in welfare estimation resulting from habitat restoration. The probability that angler *i*'s catch at site *g***,** partitioned into zero or a positive catch, is presented as,

$$p(Y_g(i) = n) = \begin{cases} \pi_g(i) + (1 - \pi_g(i))p(Y_g(i) = 0) ifn = 0 \\ (1 - \pi_g(i))p(Y_g(i)) ifn > 0 \end{cases}$$

where $\pi_g(i)$ is the probability of zero catch in fishing location *g*, represented by the logistic function, $\pi_g(i) = \frac{exp(\lambda X_g(i))}{1+exp(\lambda X_g(i))}$, where $X_g(i)$represents elements of effort, including number of anglers in the boat within that trip, hours fished, total number of rods, and whether the anglers targeted any specific species (binary variable), and hours spent fishing (Table 1). Therefore, we first estimate the probability of zero catch and the expected positive catch. Then, the expected catch rate$z_g(i)$is therefore:

$$E(z_g(i)) = (1 - \pi_g(i))Y_g^E(i)$$

The regression coefficients will be estimated using maximum likelihood estimation, and the log-likelihood function of the model is,

$$log(L) = \sum_{y_i=0} log(\pi_g(i) + (1 - \pi_g(i))p(Y_g(i) = 0)) + \sum_{y_i>0} log((1 - \pi_g(i))p(Y_g(i)))$$

2.3. A conditional logit model

We used the random utility model as a general framework to describe the observed angler's choice of location as a function of angler's characteristics, the characteristics of the location selected, and the characteristics of all potential fishing locations that the angler could have chosen but did not. We assume that when fishers decide where to fish, they first decide which bay and then what grid they would prefer to fish. This is a very reasonable assumption from the conversations with the local fishers. Therefore, a universal choice set would be considered when fishers decide where to fish from the onset of the fishing trip. Similar to Haab et al. (2008), we treated each ramp-location combination as a unique alternative and used a conditional logit model to investigate the choice of recreational angling location. We also ran a nested logit model with PPB and WP as branches and locations as the twigs in the choice tree structure, but found statistically insignificant with regard to estimated parameters and that the dissimilarity parameters were greater than one, violating the sufficient condition that, for the nested model to be consistent with utility maximization, the dissimilarity parameters should be greater than zero but smaller than one. When the dissimilarity coefficients are outside of the unit interval, Borsch-Supan (1990) derives conditions under which a nested logit model is consistent with utility maximization. Herriges and Kling (1996) corrects and extends Borsch-Supan (1990). In this paper here, we did not estimate the welfare gain using the nested logit model when dissimilarity parameters are greater than one. This is because the empirical implementation of these conditions for our analysis could be served as an independent research paper (interested readers may refer to Kling and Herriges (1995)). Instead, we reported the specification and results of our nested logit model in the appendix Table A2. Therefore we only discuss the coefficient estimates for conditional logit model and used them for welfare analysis.

Assuming that individual angler i chooses boat ramp (r) and goes to location (d) to fish, the indirect utility $U_{rd}(i)$which measures the satisfaction or benefit that an angler enjoys during a visit to angling route and site rd is presented as,

$$U_{rd}(i) = V_{rd}(i) + \varepsilon_{rd}(i)$$

where

$$V_{rg}(i) = \beta tc_{rg}(i) + \rho z_{rg}^E(i)$$

where $V_{id}(i)$ is the deterministic part of utility function, $\varepsilon_{rd}(i)$ is the unobserved error term, $tc_{rg}(i)$is the total travel cost, $z_{rg}^E(i)$is the predicted total catch in fishing location, and β, ρ are unknown parameters to be estimated.

Assuming that the error terms are independent and identically distributed with a Gumbel (type 1 extreme value) distribution, the probability that a recreational angler will choose a combination of boat ramp and location (rd) is therefore:

$$P(rg, i) = \frac{exp(V_{rg}(i))}{\sum_{j=1}^{N(i)} \sum_{k=1}^{M(i)} exp(V_{jk}(i))}$$

where *N(i)* and *M(i)* are the number of ramp and fishing location in angler *i*'s choice set. The random utility model allows us to calculate welfare gains associated with changes of angler's site attributes. The theoretical analysis and procedure have been developed by Small and Rosen (1981) and Hanemann (1999). First, we define initial condition in site attributes as h_0 and the altered condition that it will be changed into as h_1. Willingness-to-pay (WTP) is implicitly defined as,

$$V(y - tc, h_0) = V(y - tc - WTP, h_1)$$

WTP is regarded as the amount of income that compensates the individual for changing the attributes from initial condition h_0 to h_1. Note here income y is linear in the indirect utility function and will disappear in the welfare calculation (Small and Rosen, 1981). If the quality of site attribute improves, WTP will be positive, otherwise negative. We follow Small and Rosen (1981) and Hanemann (1999), the compensating variation (CV)-the maximum WTP for change in site attributes (CV) in relation to a change in angling site quality vector (h) - is,

$$CV = \frac{1}{\beta}\left\{ln\left(\sum_{rd \in S(i)} exp(V_{rd}(h_0))\right) - ln\left(\sum_{rd \in S(i)} exp(V_{rd}(h_1))\right)\right\}$$

β is the estimated marginal utility of income. $S(i)$ is the choice set available to angler i. Equation (9) will give us a per trip value for the change in site attribute.

2.4. Scenarios of ecosystem change

Over the past several decades, both Port Phillip Bay and Western Port (PPB and WP) have witnessed significant declines in seagrass cover (Ball et al., 2014; Blake and Ball, 2001). Depending on the location, the decrease of seagrass cover varies in Port Phillip Bay, with approximately 90% decreases at Blairgowrie and Bellarine Bank, and 60% at St Leonards between 1998 and 2010, with partial recovery in recent years (Ball et al. 2014). In Western Port Bay, seagrass cover suffered a major decline in the 1980s, with some minor recoveries since then (Blake and Ball, 2001). In an effort to reverse the declining trend, more than AU $3 millions of the Victorian recreational fishing licence trust funds has been invested in projects such as Victoria Fisher for Fish Habitat Program, in the past decade (https://www.ari.vic.gov.au/research/people-and-nature/victorian-fishers-for-fish-habitat-program). The success of seagrass restoration is variable (van Katwijk et al. 2015) and even in success, the timeframe to restore seagrass cover could vary. Here in our scenario analyses, we assume the restoration is successful and has reached functional equivalency in terms of fishing benefits. These hypothetical restoration scales are the starting point for our estimation.

Here we considered three ecosystem rehabilitation scenarios: a 10% seagrass cover expansion-corresponding to a modest plan; a 30% seagrass cover expansion, a relatively ambitious goal with spatial prioritisation; and 100% loss of the existing seagrass ecosystem. This is done by revising the ecosystem coverage in each fishing location correspondingly while maintaining ecosystem coverage in other locations as the base (unchanged) value. In the recreational literature with RUM, the welfare loss from site closure is "access value" of that site. We estimated the model with complete removal of seagrass ecosystem in a recreational fishing location and calculated the welfare loss, representing the current value of all seagrass ecosystem within a fishing location. For the purpose of economic evaluation of habitat, we did not estimate value in locations without seagrass ecosystems.

We paid particular attention to the habit loss over the past several decades in Western Port. Western Port has suffered extensive loss of approximately 70% coverage area and dropped to approximately 59 km^2 in 1983/1984 (Blake and Ball, 2001). From Blake and Ball (2001), we calculated that the seagrass and macroalgae area could be up to 196.7 km^2 in early 1970s. With 84% of area belonging to either seagrass or a mixture of seagrass and algae (Blake and Ball, 2001), the seagrass coverage in Western Port was approximately 165.2 km^2 in early 1970s. Our current estimates show seagrass coverage is ~100 km^2 suggesting that the bay-wide seagrass coverage in WP has decreased by approximately 36.7% from the 1970s. Due to the unknown ecosystem loss in each individual location in WP, in this particular case we estimated the welfare loss resulting from this 36.7% ecosystem loss in all fishing locations simultaneously.

Table 4
Random utility model coefficient estimates. Values in brackets are the standard error.

	Conditional logit model
Variable (N = 952,233)	
Total travel cost	−0.089 (−5 4) ***
Predicted total catch	0.038 (2.35) **
Model Statistics	
Log-likelihood	−7213
Wald Chi^2	3011

***p < 0.01, **p < 0.05

3. Results

3.1. Estimation result

We found that the number of anglers on the fishing trip (mean = 2.1, sd = 0.85), hours spent fishing (mean = 4.25, sd = 2.22), rod count (mean = 3.17, sd = 1.37), whether the angler is targeting specific fish species (mean = 0.15, sd = 0.36) all had significant and positive influences on the total boat-based recreational catch (mean = 5.55, sd = 9.06) (Table 3). As expected, the probability that the angler will have zero catch decreased as the hours spent fishing increases, all else being equal (estimated from the zero-inflation part of the ZINB).

Seagrass cover had a positive and statistically significant influence on recreational bay-wide catch, as did the areal extent of seaweed and infralittoral sediments. However, the coefficients associated with tidal marsh and mangrove ecosystems were not statistically significant for boat-based recreational catch. Therefore, we solely focused on the economic benefit of seagrass rehabilitation.

The coefficients of recreational location choice model estimates are presented in Table 4. The coefficients of total cost and catch rate were both statistically significant to fishing location choice, indicating that recreational anglers are more likely to choose closer locations and those that offer higher catch rate.

3.2. Valuing ecosystem change

We examined three scenarios of seagrass change in each fishing location: 10% expansion, 30% expansion and 100% loss. The mean proportion of seagrass coverage area per location is 0.05 (sd = 0.06) and 0.21 (sd = 0.15) for PPB and WP, respectively. Averaged across all locations that contain seagrass, welfare loss from removal of seagrass is AU $15.36 per trip. The welfare loss ranged from near-zero to AU $12.21 per trip for PPB with an average of AU $3.1 per trip, and near- zero to AU $87.47 per tip in WP, averaging AU $ 30.34 (Figs. 2–4, and for more value details see Appendix Table A3 and A4).

Table 3
Coefficient estimate of catch model, including the zero-catch and positive catch parts of the model.

VariablesPositive catch	**ZINB**	**Z-values**	**p-values**
Angler count	0.207(0.046)	4.44	0
Hour fished	0.097(0.021)	4.6	0
Rod count	0.046(0.026)	1.82	0
Target	0.59(0.093)	6.36	0
Tidal marsh*	6.22(9.14)	0.68	0.49
Mangrove*	−4.03(7.51)	−0.54	0.59
Seagrass*	2.86(0.58)	4.9	0
Rock*	0.35(0.6)	0.59	0.56
Sediment*	0.328(0.45)	0.74	0.46
Seaweed*	1.37(0.623)	2.2	0.028
Reef*	−2.36(2.67)	−0.88	0.37
Mud*	0.24(0.43)	0.56	0.58
Sand*	1.55(0.47)	3.34	0.01
Constant	−0.336(0.46)	−0.73	0.47
Zero inflated part			
Angler count	0.14(0.17)	0.82	0.41
Hour fished	−1.04(0.18)	−5.64	0
Rod count	0.06(0.127)	0.48	0.63
Target	−0.83(0.55)	−1.5	0.132
Constant	0.93(0.56)	1.67	0.09
Sample size (N = 1575)			

Standard errors in parenthesis. *Indicators % area of ecosystems

The economic benefit from seagrass rehabilitation also varied greatly across locations within and between PPB and WP, mainly due to the heterogeneity of the proportion of seagrass coverage within a fishing location (Figs. 2–4, Appendix Table A3 and A4). The values ranged from near-zero to AU $2.27 per trip and from near-zero to AU $7.35 per trip for 10% and 30% increase in seagrass cover in locations in PPB. In WP, the economic benefit per fishing trip ranged from near-zero to AU $19.18 and from near-zero to AU $85.55 for 10% and 30% increase in seagrass cover in fishing locations.

Averaged across all locations under 10% and 30% ecosystem rehabilitation scenarios, we found that the bay-wide economic benefits were AU $0.39 and AU $1.22 per trip in PPB, AU $5.49 and AU $19.57 per trip in WP. Although these per trip values may seem small, the aggregation value of all trips per annum for improving estuarine ecosystem can be quite substantial. We estimated two and one million fishing trips per annum taking place in PPB and WP respectively, based on Ryan et al. (2003) and Ernst and Young (2015). An estimate of two million fishing trips to PPB each year, suggests that the economic benefit for improving ecosystems by 10% and 30% is AU $0.78 million and AU $2.44 million per annum. In WP an estimate of one million fishing trips per year, suggests an economic benefit for improving ecosystem by 10% and 30% is AU $5.49 million and AU $19.57 million dollars for WP each year. The welfare loss resulting from 36.7% seagrass ecosystem in all fishing locations simultaneously in WP amounts to AU $86.75 per trip. This suggests that the annual welfare loss could be up to AU $86.75 million, assuming a million fishing trips to WP per annum.

We fitted the economic values with the proportion of seagrass coverage within a fishing location in each location for these three scenarios in both PPB and WP (Figs. 5–7). The fitted lines correspond to 10% and 30% seagrass rehabilitation scenarios in PPB (Fig. 5) and WP (Fig. 6), and 100% decrease in seagrass in fishing locations of these two bays (Fig. 7). There are several outliers in fishing locations B3, B4 and B5 in WP which might result from a combination of a relatively high proportion of seagrass cover (approximately 30%) and anglers' long travel distance to these locations. These figures indicated the economic benefit is a function of seagrass cover of a location under different rehabilitation and loss scenarios. The recreational fishing locations in WP saw a bigger variation in the proportion of seagrass coverage than those in PPB, creating bigger variation in economic value. Note that when we simulated the economic value of ecosystem rehabilitation or

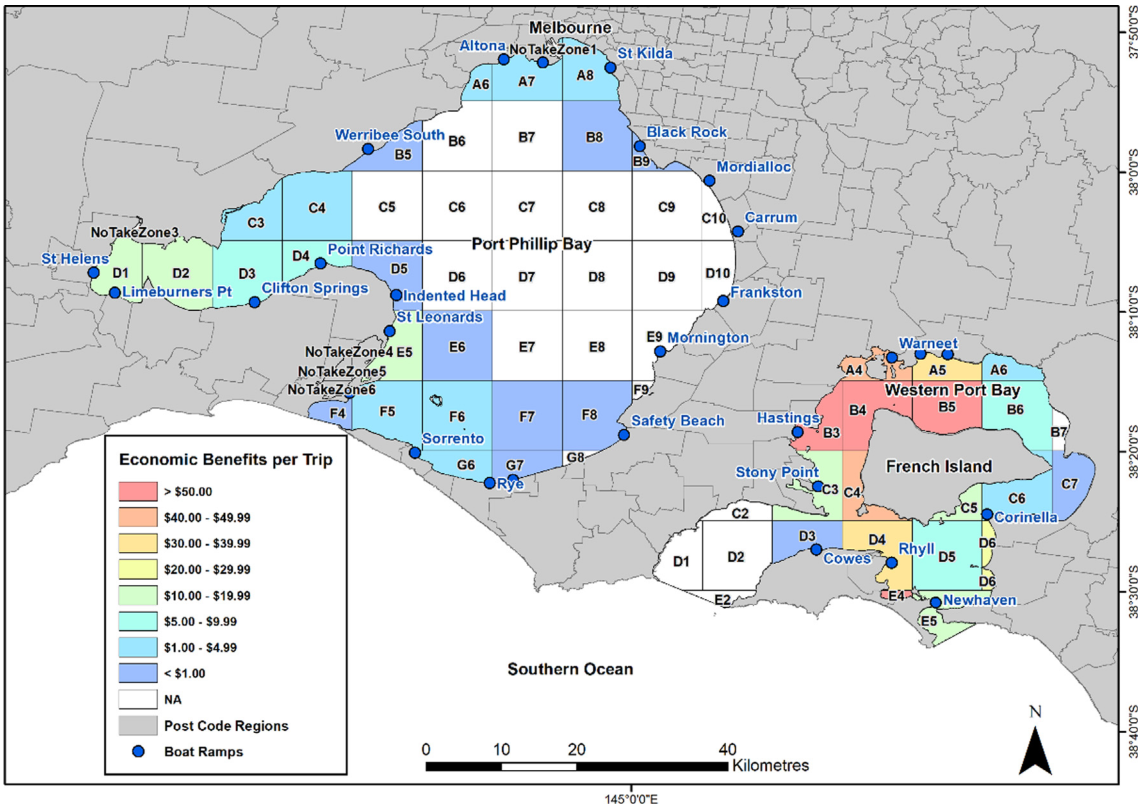

Fig. 2. Economic benefit per trip corresponding to current value of seagrass coverage.

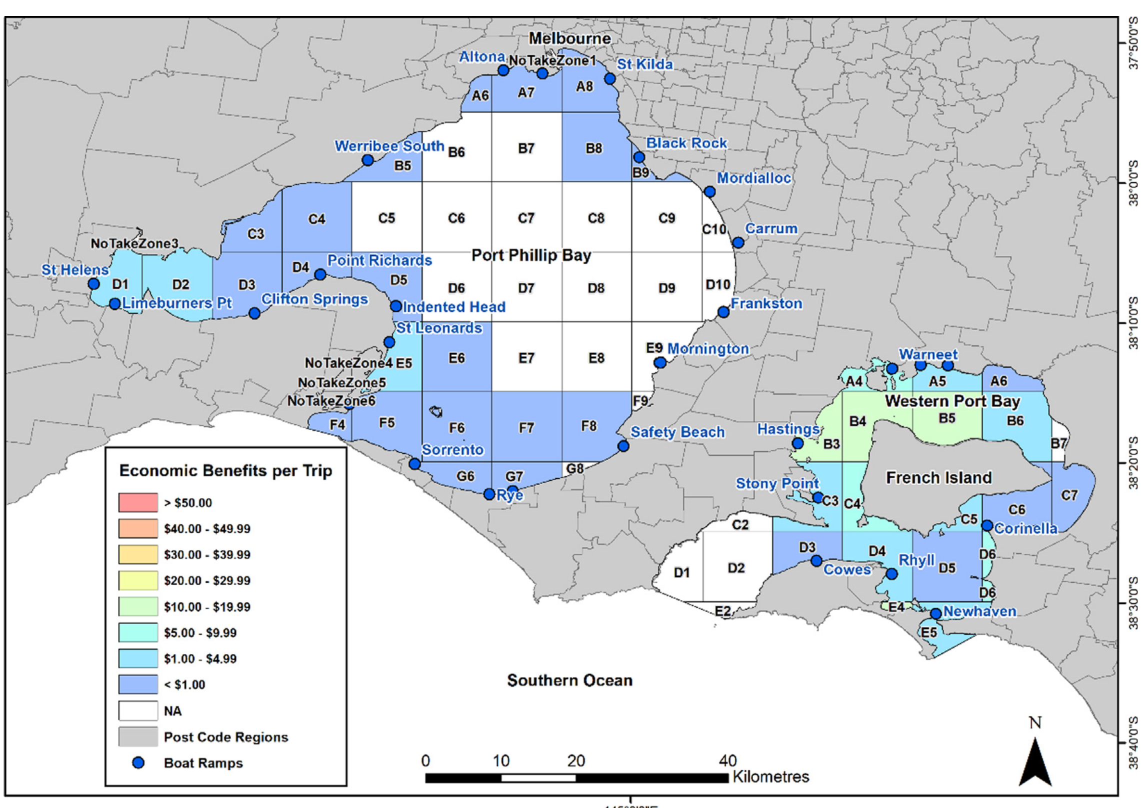

Fig. 3. Economic benefit per trip correspond to 10% increase in seagrass coverage.

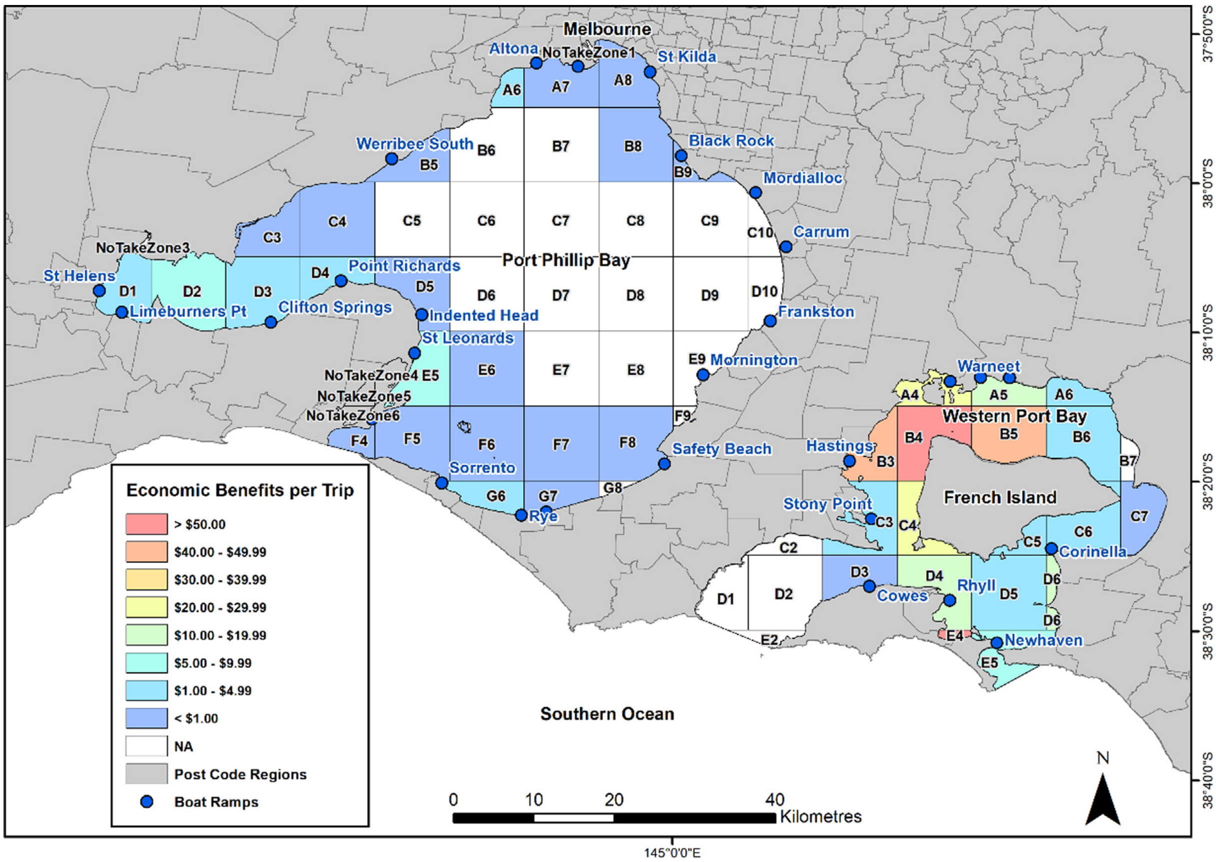

Fig. 4. Economic benefit per trip corresponding to 30% increase in seagrass coverage.

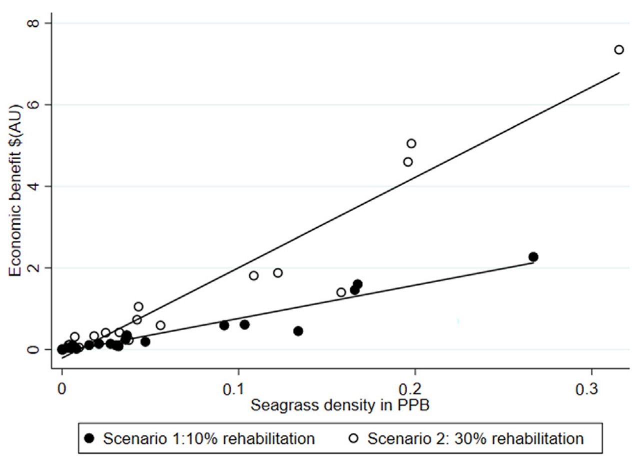

Fig. 5. Economic benefit as a function of seagrass cover rehabilitation in PPB.

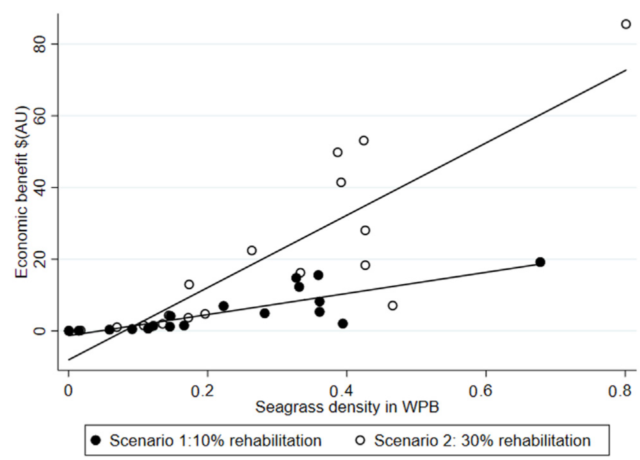

Fig. 6. Economic benefit as a function of seagrass cover rehabilitation in WP.

loss in each location, we maintained the ecosystem quality of all other locations as the base value. As shown in Figs. 5 and 6, the total economic benefit increased as the level of seagrass percentage cover of each recreational location increased. The greater slope of 30% rehabilitation as compared to that of 10% rehabilitation does not suggest that the marginal value – the value corresponding to an incremental change – is high when seagrass cover is high, rather that total economic benefit is greater in a location with a high proportion of seagrass cover than that with less seagrass. This is the scope effect instead, defined by Huang et al. (1997).

4. Discussion and conclusion

In this study, we evaluated welfare gains of estuarine and coastal ecosystem rehabilitations for recreational anglers in south-east Australia, by linking the catch model with the choice model to examine the attractiveness (in terms of travel cost and expected catch rate) of fishing location for recreational anglers. Admittedly, not all targeted species may directly rely upon seagrass. We found that seagrass was a significant predictor of catch, and its loss would substantially decrease the value of recreational opportunities. In contrast, a 10% increase in area across the two bays would see

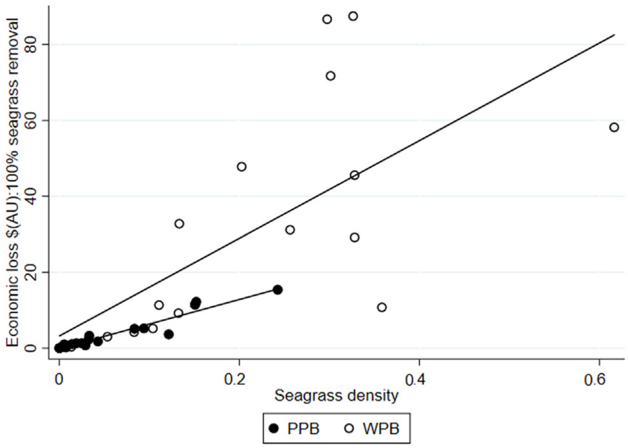

Fig. 7. Welfare loss as a function of seagrass cover removal in PPB and WP.

growth in economic benefit to recreational fishing of at least AU $6.2 million per year, while a 30% increase could add over AU $22 million each year.

We found that economic benefits for anglers varied widely across space due to the heterogeneous coverage of seagrass in fishing locations, ranging from near-zero to AU $19.18 corresponding to 10% increase in seagrass cover. In a previous study, Ahn et al. (2000) used a nested logit model and investigated the welfare cost resulting from varied trout habitat loss for anglers in North Carolina. They found that welfare loss from 8.3% trout habitat loss ranges from $0.0007-$18.97, and $0.002-$84.28 per trip for 43.3% trout habitat loss; our results are consistent with these estimates.

The percentage cover of mangrove and tidal marsh is not statistically significant to boat-based recreational catch. Mangroves have a very limited distribution in PPB, while saltmarsh in both PPB and WP is only inundated very briefly during high tides, so there is very limited potential to catch fish in these two ecosystem types in these two Bays. This does not mean that tidal marsh and mangroves don't contribute to maintenance of fisheries and recreational fisheries. Indeed, these ecosystems may contribute to shore-based recreational fishing, where it is easier to fish within these ecosystems (Meynecke et al., 2008). Tidal marsh and mangroves may also contribute to fisheries as nursery grounds or through the diet of estuarine fisheries (Raoult et al., 2018), benefits not captured by our modelling approach.

Seagrass has declined in our study area over recent years; seagrass distribution in Western Port which has suffered loss estimated at ~37% over the past several decades, corresponding to an estimated loss of up to AU $86.75 million per annum. Our results suggest that the resulting economic benefit of ecosystem rehabilitation could be quite substantial if only a fraction of this is reversed. Seagrass rehabilitation incurs high cost. For instance, the cost for seagrass rehabilitation ranged from AU $10,000 ha^{-1} to AU $1,308,284 ha^{-1} in Australian New South Wales estuaries (Ganassin and Gibbs 2008). Given the high cost of seagrass rehabilitation, this spatially explicit valuation information could aid resource managers in effective targeting of ecosystem rehabilitation in high value locations.

Estuarine and coastal ecosystems are degrading worldwide. There is a need to recognise key ecosystem system services and the values they provide to drive conservation as well as the potential benefits of restoration efforts (Barbier et al., 2011). We demonstrate how survey information from recreational fishers and integrating ecological data and economic analyses can assist in quantifying the non-marketed values of ecosystems services of existing habitat and restoration scenarios. While many existing studies focus on valuing individual species, missing both ecological complementarity among species and the substation effect in economic models (Loomis and White, 1996), our emphasis based on multi-species (albeit in a simple aggregated form) and a habitat-based evaluation approach would be valuable to the management of coastal and estuarine ecosystems. Furthermore, ecosystems rehabilitation requires substantial financial support. With limited financial support, this would create huddles in habitat rehabilitation effort. Our study also indicates that anglers are willing to pay (sometimes significant amount) to improve estuarine ecosystems. Therefore, policymakers may be able to partially alleviate the financial burden by capturing some of the consumer surplus for instance with implementation of a well-structured and spatially explicit user fee.

Our estimates of the value of seagrass and the fishing locations may be an underestimate for several reasons. We did not consider the possibility that a higher catch rate resulting from seagrass improvement may lead some latent recreational anglers to become regular (or more active) recreational anglers, or anglers may substitute between other recreational activities and recreational fishing, indicating a lower bound for our estimates. This suggests further survey and study to include information of non-anglers. The Victorian state government's "Target One Million plan" aims to increase the number of recreational anglers to one million by 2020 (https://vfa.vic.gov.au), with PPB and WP as the two most popular angling locations, which could further drive demand. Our model does not account for any spill-over effects from neighbouring locations, or the uses by species of varied locations during their life cycle, potentially underestimating ecosystem value. Typically, spatially connected systems are not considered in recreational demand models (Newbold and Massey, 2010), with a few exceptions (Massey et al., 2006; Newbold and Massey, 2010; Gao and Hailu, 2018, but these studies did not investigate the effect of ecosystems on the recreational fishery). Coastal and estuarine ecosystems can be evaluated by integrating micro-econometric choice modelling with biophysical models that simulate population dynamics and ecosystems, or including a stock-recruitment relationship of some targeted species. Unfortunately there is a lack of this stock-recruitment relationship for major fishery species in PPB and WP (Jenkins, 2005; Hirst et al., 2014). By doing so, the bioeconomic modelling will account for the feedback connection of the coupled human-ecological systems and make it possible to investigate the long-term impact of ecosystem rehabilitation for varied species and recreational benefits.

In the modelling process, there are several limitations not captured in our approach. These include angler motivation and satisfaction in relation to no catch. Indeed, heterogeneity in the preference of anglers is an important factor to explain recreationist's site choice (Provencher and Bishop, 2004). A powerful and more attractive tool to tackle this unobserved preference heterogeneity is to use random parameters mixed logit model or mixture model. However, in the face of a large choice set, very limited evidence suggested these models' effectiveness and the computational limitation can make estimates difficult and intractable (von Haefen and Domanski, 2018). The conventional strategies for addressing these computational issues include aggregation and separatbility (Feather, 1994, Parsons and Hauber, 1998). Until very recently, Von Haefen and Homanski (2018) showed that the expectation maximization algorithm can be used along with McFadden's sample of alternative approach to consistently estimate latent-class mixed logit model. However, the size of alternative (569 in total) in von Haefen and Homanski (2018) is still much less than what we are faced with (1007 in total). Furthermore, it has been empirically shown that preference heterogeneity models such as the mixed logit and finite mixture model are not sufficiently preferred to the more traditional conditional and nested logit models, and the limitations of the traditional models do not seem to detract from its performance (Haab and Hicks, 1999). When we are faced with analyzing a model with a very large choice set resulting from combination of boat ramps and delineated grids within each bay, we must choose between a more accurate model (with mixed logit for instance, which could generate negative welfare measures and make it difficult to interpret) and a computationally feasible one (with a conditional logit and nested logit model). We chose condition logit model and have also estimated and tested the nested logit model.

Furthermore, this study has used a aggregated catch model. Conventionally, the paucity of data for some species at fishing site level lead to studies to aggregated across species (for instance, Bockstael et al., 1989; Green et al., 1997; Haab and Hicks, 1999; Kirkley et al., 1999). By doing that, these studies assume that the aggregated catch model can roughly approximate change in welfare estimation resulting from species-specific changes. In our study where approximately 25% of anglers are indifferent to what they caught, we have used the aggregated model of expected catch because of excessive zero catch. Our experience with the local fishers seems to suggest that the motivational factor for recreational fishing is multifaceted and could be moved beyond the fish they caught. The goal of the analysis here is to measure change in value due to change in habitat condition. In the absence of the knowledge of how varied species response to restoration of habitat, we therefore assume that the aggregated model can roughly approximate change in welfare estimation resulting from habitat restoration. To accurately assess angler values for marine fishing in a recreational demand setting, modelling of target species, the existence of substitutes and the human dimension of recreational fishing is important direction for further investigation.

The location-level economic benefit estimates illustrated in Figs. 2 and 3 provide vital information in relation to spatial policy implications of managing marine natural resources. When coastal managers have questions with regard to spatial effort allocation for ecosystem rehabilitation, information such as this could be instrumental for identifying rehabilitation hotspots of high economic benefit. This may be done, for instance, according to the principle of maximizing return on rehabilitation investment along with consideration of other ecosystem services that the ecosystem provides, such as carbon sequestration and coastal protection (Barbier et al., 2011; Boyer and Polasky, 2004; Brander et al., 2006). Other considerations in restoration optimisation would include cost, feasibility, likelihood of success (Joseph et al., 2009; Pannell and Gibson 2016), and of course other values such as biodiversity. All these issues described provide avenues for future analyses.

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.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2019.134680.

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