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Recreational fishing expenditure as an indicator of coastal wetland habitat value

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Valuing the ecosystem services provided by nature is essential for estuarine habitat conservation and restoration. Recreational fisheries rely on fish stocks that are dependent on productivity derived from the plants that comprise estuarine habitats, however the value of these habitats to recreational fishing is rarely considered. Here, we consider expenditure on recreational fishing activities as an indicator of coastal wetland habitat value, by synthesising data on routinely collected recreational effort, catch, and expenditure from telephone surveys alongside trophic subsidy models within a simple framework. The approach is demonstrated for the Clarence River and the Hunter River estuaries (New South Wales, Australia). Expenditure on recreational fishing activities was apportioned to mangrove and saltmarsh habitats *via* the 'trophic subsidy' (or nutrition) originating from primary producers in these habitats that fuels the biomass of important recreational species. The values estimated exceeded that of similarly apportioned commercial fisheries revenue, with the biggest difference observed for saltmarsh in the Clarence River (~$17 million AUD per annum [recreational expenditure] compared to ~$8 million AUD per annum [commercial fisheries total output]). When considered in an additive fashion and standardised by habitat extent, the values attributable to coastal wetland productivity were as high as $86 459 per hectare per annum for saltmarsh, and $20 611 per hectare per annum for mangroves. These values reflect the dependency of fisheries activities on the extent and condition of coastal wetland habitats, and the framework presented here is widely applicable for considering the economic value of these activities *i.e.*, fishing) as an indicator of habitat value.

Environmental significance

Estimating the economic value associated with coastal wetland ecosystem services is essential for ensuring the conservation and restoration of these habitats. However, research on this topic that deals with recreational fisheries is almost non-existent, despite this sector being a potentially substantial beneficiary. We present a feasible, generalised approach to quantify expenditure on recreational fishing activities as an indicator of coastal wetland habitat value, by synthesising data on routinely collected recreational effort, catch, and expenditure alongside trophic subsidy models within a simple framework. This innovative approach employs easy-to-obtain data and straightforward calculations that will be immensely useful for scientists and practitioners, and aid development of the economic rationale supporting habitat repair and fisheries co-benefit estimation for blue carbon restoration projects.

Introduction

Estuarine habitats support a range of ecosystem services which are often categorised as regulating, cultural, supporting and provisioning. Biological processes in coastal wetland habitats support ecosystem functions that underpin these services, thereby supporting extractive activities such as fisheries.[1] Fisheries are reliant on harvestable biomass, much of which has its physical origin in primary productivity of plants within coastal wetland habitats.[2] Energy and nutrients synthesised by primary producers are relayed through estuarine food webs to support consumer biomass,[3] and provide the nutrition responsible for fuelling broader population processes for exploited species such as growth (and survival), reproduction, and migration. This resulting fisheries productivity is one of the most widely considered benefits derived from coastal wetlands,[4] and decades of research have explored trophic and nursery linkages between emergent estuarine habitats such as saltmarsh, mangroves and seagrass[5–8] and commercially and recreationally exploited species.

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Appraising the economic value of ecosystem assets and the services they support is now an essential component of decision making in contemporary natural resource management.[9] This is particularly the case where ecosystem assets may be threatened, or an economic case is required to support conservation and restoration against other competing uses.[10] The advent of ocean accounting, and more generally environmental economic accounting and natural capital accounting,[11] have embedded a need to examine the benefits that flow from aquatic ecosystem assets (such as coastal wetlands), and consider the resultant economic impacts. The comparatively recent spotlight on 'blue carbon' restoration has further enhanced the need to link coastal wetlands with the various 'co-benefits' that arise from conservation and restoration activities (*e.g.*, see Hagger *et al.*[12] and Rogers *et al.*[13]), and estimate associated economic value. Quantifying the linkages between coastal wetlands and fisheries activities allows the revenue generated from these activities to be considered as a monetary indicator of the habitats value or importance for these activities. Following this, aggregation of monetary values associated with different ecosystem functions can reflect the monetary value of the ecosystem as a whole,[14] however consideration of value to different beneficiaries is essential to building this broader picture.

Linking the revenue generated from fisheries production with coastal wetlands in a quantitative way can be challenging.[15] While various approaches to achieve this have been developed, there is often a focus on commercial fisheries harvest,[16–18] likely because commercial fisheries generally provide accessible market-based measures of economic benefits (such as gross value of product, GVP). Recreational fisheries separately target the same stocks as commercial fisheries, and thus recreational fishers are also beneficiaries from the primary production originating in coastal wetland habitats that supports the biomass of captured animals. Due to the 'recreational' component, however, recreational fisheries are generally categorised as cultural services, as recreational fishing is motivated by both recreation and sustenance. Given the recreational component, the value of recreational fishing is sometimes approached through contingent valuation surveys,[19] but these surveys can be costly,[20] and are infrequently applied in the context of coastal wetlands and recreational fisheries. In contrast, off-site recreational fishing surveys (such as telephone recall or diary surveys[21]) provide a means to collect catch, effort, and economic data from comparatively large numbers of recreational fishers, and are conducted on an ongoing basis in many jurisdictions. The data from such surveys may be useful in appraising the value of recreational fishing activities that are supported by specific coastal wetland habitats, when framed within a novel method that models dependencies between such revenue generating activities and habitats within estuarine ecosystems. When attributed in this way, the revenue generated from recreational fishing can provide an indicator of the value or importance of particular habitats in monetary terms.

The 'trophic subsidy' method partitions revenue arising from fisheries activities through trophic pathways to the different primary producers (plants) that form coastal wetland habitats.[17] The approach thus requires: (1) knowledge of the revenue generated through fisheries that exploit biomass that is dependent on coastal wetland habitats, quantified through fisheries catch metrics and/or associated economic data; and, (2) knowledge of the origin (*i.e.*, primary production by the plants that form coastal wetland habitats) and flow of nutrition that supports the animals captured by fisheries (*i.e.*, reflecting the 'trophic subsidy' from habitats that supports exploitable fish biomass), which is quantified through stable isotope mixing models.[17] The extent of the coastal wetland habitats can be used to standardise the resultant estimates to an areal unit (*i.e.*, per hectare of a particular habitat).

The trophic subsidy method has been applied to several estuaries in south-eastern Australia, to apportion commercial fisheries revenue as an indicator of seagrass and coastal wetland habitat value,[3,22–24] and is currently being incorporated in broader frameworks for considering the value of fisheries co-benefits arising from habitat repair.[25] The comparative simplicity of this method provides advantages for practitioners, particularly if appropriate data from the estuary under consideration are available to support calculations. Expenditure on recreational fishing activities is an indicator of the 'value' of recreational fishing, but this overall value is supported by the capture of a range of species that rely on primary production originating from different estuarine habitats to differing degrees. Apportioning this expenditure using the trophic subsidy method provides a pragmatic means to estimate the components of this overall value that are dependent on particular estuarine habitats. Here, we further develop and apply the trophic subsidy method in this fashion, and partition expenditure on recreational fishing activities (as an indicator of the value of recreational fishing) among coastal wetland habitats based on the nutritional contribution of primary productivity from plants within these habitats to the species that are captured. We use routinely collected recreational survey data to demonstrate this approach for coastal wetland habitats in two case study estuaries for which trophic linkage data are already available. Finally, we compare and combine these monetary values with those that reflect the value of commercial fishing in these estuaries, that have already been partitioned to wetland habitats using a similar approach.

Methods

Description of case study systems

The Hunter River estuary (−32.90, 151.78, hereafter referred to as the Hunter River) is a mature wave-dominated barrier estuary on the mid-north coast of New South Wales (NSW), Australia (Fig. 1), which drains substantial river systems and a large catchment (21 000 km^2). The estuary is adjacent to the City of Newcastle, a large urban centre and coal port, and supports a small but diverse commercial fishery. The estuary has also been the subject of substantial habitat repair activities over the past 20 years,[26] and has significant coverage of mangrove and saltmarsh habitat (but no seagrasses). The Clarence River estuary (−29.43, 153.37, hereafter referred to as the Clarence River, Fig. 1) is the largest estuary in NSW with a waterway area of 132 km^2 draining a catchment of 22 000 km^2, and supports

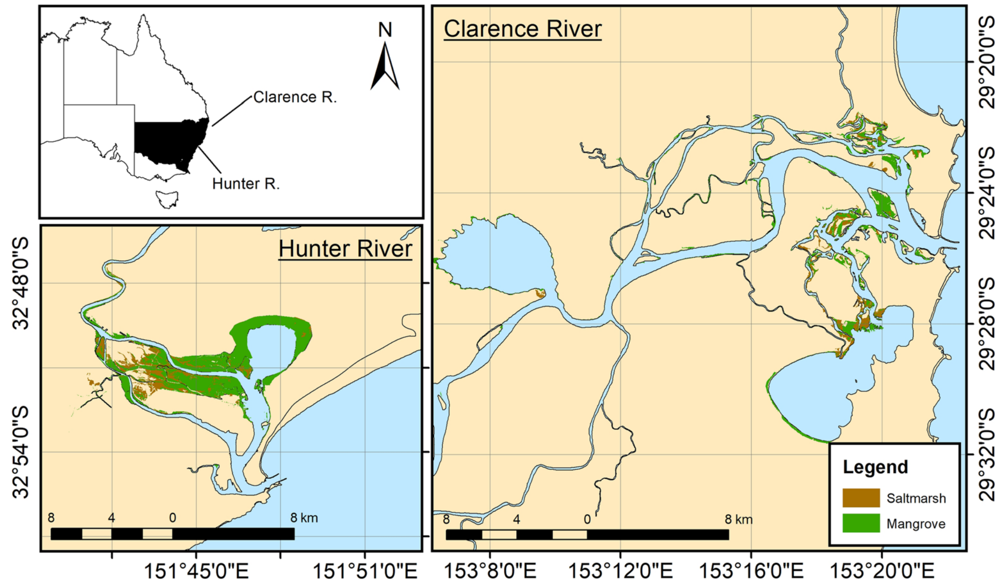

Fig. 1 Map of the two case study estuaries, Clarence River and Hunter River, showing the location of each on the eastern Australian seaboard, and distribution of mangrove and saltmarsh habitats through these estuaries.

the state's largest estuarine fishery. The Clarence River is also a mature wave-dominated barrier estuary,[27] so the two estuaries share some similar geomorphological attributes, and also support a similar species assemblage. There are substantial drained floodplains in the Clarence River catchment, but also considerable coverage of saltmarsh and mangrove.[28]

The Hunter River and Clarence River support a similar suite of commercially exploited species, including Yellowfin Bream (*Acanthopagrus australis*), Dusky Flathead (*Platycephalus fuscus*), Mulloway (*Argyrosomus japonicus*), Luderick (*Girella tricuspidata*), Sea Mullet (*Mugil cephalus*), Blue Swimmer Crab (*Portunus armatus*), Giant Mud Crab (*Scylla serrata*) and Eastern School Prawn (*Metapenaeus macleayi*), which are harvested by the Estuary General and Estuary Prawn Trawl fisheries. The Clarence River commercial fishery is much larger in volume than the Hunter River, and is important in supporting the local regional economy. Both estuaries also support recreational fisheries which heavily target a subset of these species (especially Yellowfin Bream and Dusky Flathead) through angling. Recreational harvest is managed under bag and size limits, whereas the main control on commercial fishing is limited entry and effort quota (within the Estuary General fishery). With the exception of Mulloway, which is currently listed as depleted, all the species listed above are assessed as sustainable under the current national status reporting framework (see **https://fish.gov.au/**).

Rationale

As noted earlier, the trophic subsidy method is principally a value attribution exercise which involves partitioning revenue or economic value derived from fishery, market or survey information, among primary producers within different estuarine habitats, with some incorporation of uncertainty (if this is desired, and suitable data are available). Application of this approach to apportion revenue from commercial fisheries was described in detail in Taylor *et al.*,[17] and involved partitioning realised monetary benefits derived from commercial harvest and sale of product (*i.e.*, gross value of product at first point of sale, with flow-on economic benefits [total economic output] also considered) among coastal wetland habitats, based on the outcomes of Bayesian mixing modelling of trophic connectivity using stable isotope data.[29] The fishery as a whole relies on multiple species, and the contribution of nutrition from different primary producers differs among species, so this modelling occurs at the species level, and the monetary benefits are summed across species to provide an estimate of the apportioned value (annualised) to each habitat that is considered. The adaptation of the trophic subsidy method to apportion expenditure on recreational fishing activities among estuarine habitats follows a similar logic to that used for commercial fisheries. The expenditure on recreational fishing activities at the estuary level is used as an indicator of the value of recreational fishing, and is partitioned among different habitats within that estuary based on the outcomes of stable isotope mixing models for the mix of species captured by recreational fishers, and the primary producers that support them. This modelling takes into account the approximate trophic level of each species to correct for trophic enrichment of stable isotopes that occurs during nutrient assimilation (see Post[30]). A broad conceptual summary of the approach is provided in Fig. 2.

Partitioning of value and supporting data sources

The main data sources employed to inform the parameters described below are summarised in Table 1. All economic values are expressed in per annum 2021 Australian dollars

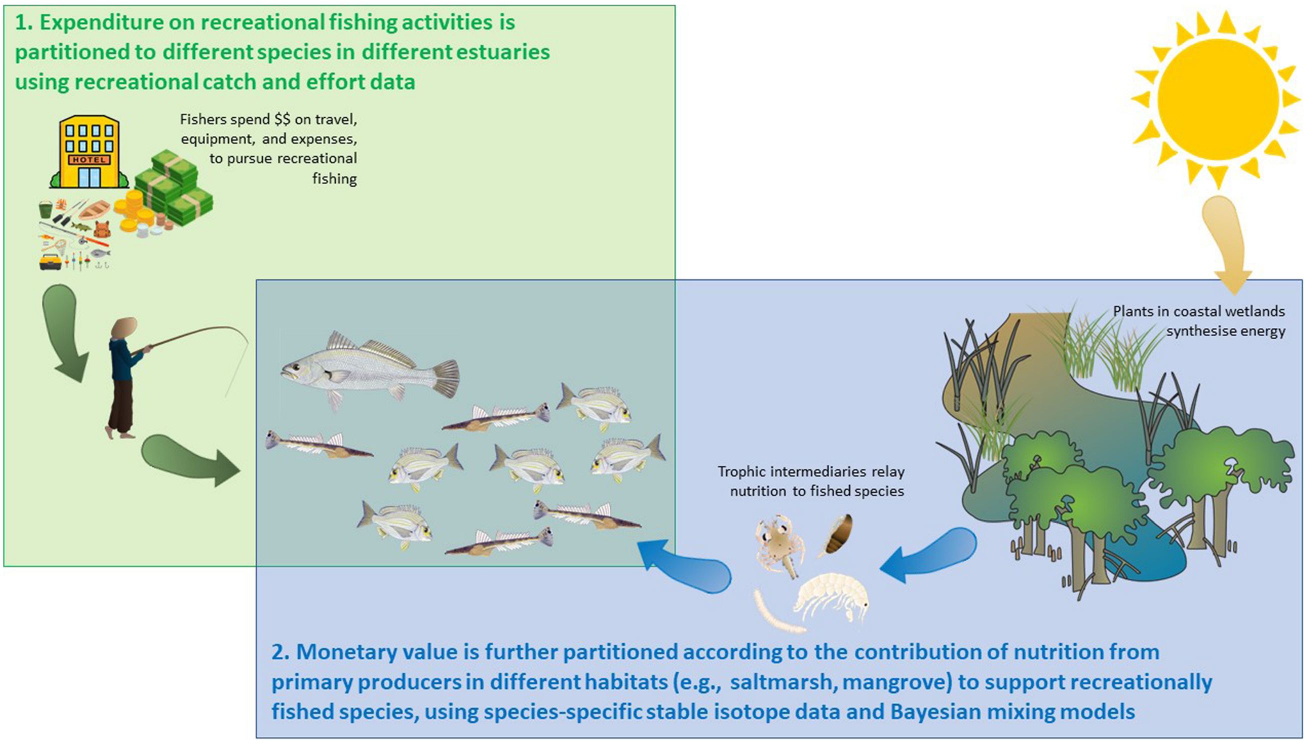

Fig. 2 Conceptual summary of the trophic subsidy approach as applied to partition expenditure on recreational fishing activities, as an indicator of the value of recreational fishing, back to the specific coastal wetland habitats that support these activities. Various data sources employed in the calculations presented here are outlined in Table 1. Symbols were obtained from Integration and Application Network Image Library (https://ian.umces.edu/media-library) and from NSW Department of Primary Industries Image Library.

(AUD_{2021}), unless otherwise indicated. Recreational fishing expenditure surveys have been conducted approximately every 10 years in NSW, and McIlgorm and Pepperell[31] report the most recent published expenditure data for recreational fishing within NSW at the time this manuscript was prepared, based on a survey conducted in 2012. In McIlgorm and Pepperell,[31] total expenditure was mapped to residents of Australian Bureau of Statistics (ABS) statistical division regions (Level 4 [SA4][32,33]): Sydney (Sydney statistical division region), north coast (Hunter, mid north coast and Richmond-Tweed statistical division regions), south coast (Illawarra and lower south coast statistical division and sub-division regions), and inland NSW (central west, far west, Murray/Murrumbidgee, ACT statistical sub-division regions). Expenditure data for interstate anglers (Victoria and Queensland) was also included. As the survey was based on data collected in 2012, expenditure was converted to AUD_{2021} using consumer price index (CPI) data collected and reported by the Reserve Bank of Australia (**https://www.rba.gov.au/calculator/**).

Total expenditure on saltwater recreational fishing activities by residents in the above regional groupings was partitioned to the case study estuaries using the results from the most recent published recreational fishing telephone diary survey for NSW, which was for the period November 2019 to October 2020.[34] The proportion of total effort expended by fishers residing within each statistical division region in each of the two case study estuaries, was used to partition the total expenditure that was applicable for each estuary e (E_e), using the formula (for i statistical regions): $E_e = \sum_{r=1}^{i} E_r P_{r,e}$, where E_r is the total expenditure on recreational fishing activities for residents within the statistical division region r, and $P_{r,e}$ is the proportion of total saltwater fishing effort (days) expended by fishers residing within the statistical division region r for fishing in estuary e.

Recreational fishing expenditure at the estuary level was further partitioned to species and the source of the nutrition, for a set of key recreational taxa (s) for which trophic linkage data were available, using the proportional catch data for each estuary and the proportional contribution of nutrition from primary producers within different coastal wetland habitats (determined from the outcomes of Bayesian mixing modelling published in Raoult *et al.*;[29] Table 1). This was estimated for each species in each estuary separately using the equation $E_{s,h} = E_e P_{e,s} C_{s,p}$, where $E_{s,h}$ is the expenditure associated with species s supported by nutrition derived from primary producers in habitat h, $P_{e,s}$ is the proportion-by-number of species s caught in estuary e, and $C_{s,p}$ is the proportional contribution of nutrition from primary producer p (in habitat h) for species s derived from stable isotope modelling, as described below. The numbers used to derive $P_{e,s}$ included caught and released fish, as both contribute to the recreational fishing experience and thus are both a motivation for expenditure. Proportion-by-number was employed (as opposed to weight), as objectives for recreational fishers in south-eastern Australia and fishing experience primarily relate to

Table 1 Summary of primary data sources employed in the calculations presented in this study

Source	Summary	Data used from this source	Spatial context of data	Parameters informed from source
Raoult *et al.*[29]	Uses stable isotope data from primary producers and fisheries species (consumers) within a Bayesian mixing model framework to estimate the origin of nutrition from dominant primary producers in costal wetland habitats	Proportion of nutrition that was derived from dominant primary producers in mangrove and saltmarsh habitats for Yellowfin Bream, Dusky Flathead, Mulloway, Giant Mud Crab, Blue Swimmer Crab, and Eastern School Prawn	Data was available from both Clarence River and Hunter River for most species	Proportional nutrition derived from dominant primary producers in costal wetland habitats within each estuary ($C_{s,p}$)
McIlgorm and Pepperell[31]	Presents both trip-associated and annual expenditure data for saltwater and freshwater fishing for residents within coarse statistical subdivisions, estimated from a telephone recall survey	Annual total expenditure data associated with saltwater fishing activity aggregated to statistical subdivision	Data were aggregates for residents within Sydney, north coast, south coast, inland statistical division region, and interstate residents	Total per-annum expenditure on saltwater recreational fishing for each statistical division region (E_r)
Murphy *et al.*[32]	Presents a summary of outcomes from a 12 months telephone diary survey collecting information on catch, effort, location, method and fishing platform, for long-term (1 year and 3 years) recreational fishing license holders and their households	Estuary-specific effort and catch (numbers of fish)	Effort and catch data were available for the entire jurisdiction (NSW), and separately for both Clarence River and Hunter River	Proportion of recreational fishing effort attributed to each estuary for residents within each statistical division region ($P_{r,e}$), and proportion-by-number of species caught in each estuary ($P_{e,s}$)

the number of fish caught,[35] and the off-site surveys do not collect information in the weight of fish captured. Embedded in this calculation is the assumption that the recreational expenditure associated with each primary species is reflected by the proportion of individuals of that species captured by recreational fishers. The numbers used to calculate proportions, however, excluded small-bodied invertebrate species such as prawns, which are caught in large quantities largely for direct consumption, and can distort estimates of proportion-by-number. These were dealt with as described below.

As noted above, the calculations in this study used the outcomes of Bayesian mixing modelling of stable isotope data collected in each of the case study estuaries, that was reported in Raoult *et al.*[29] (see Table 1). Raoult *et al.*[29] found that the saltmarsh grass *Sporobolous virginicus* was the major contributor (proportion of ∼0.5 or more) to the biomass of key species in both the Hunter River and Clarence River, and this was taken as the dominant primary producer supporting nutrition originating from saltmarsh habitats. In this dataset, however, the isotopic composition of mangroves was not significantly different from the epiphytic algae that grew on mangrove pneumatophores, and consequently these two producers were pooled in the analysis and both were considered to reflect the proportion of nutrition originating from mangrove habitats.

The outcomes from isotope modelling were applied to a subset of the most common recreational species/species groups that are well represented in the telephone diary survey within the case study estuaries: Yellowfin Bream, Dusky Flathead, Mulloway, Giant Mud Crab, Blue Swimmer Crab and Eastern School Prawn. Bream (*Acanthopagrus* spp.) are reported as a complex of Yellowfin Bream, Black Bream (*Acanthopagrus butcheri*) and their hybrids in the telephone diary survey, however the estuaries examined in this case study primarily support Yellowfin Bream. Overall, this set of six species represented ∼70–90% of the total recreational harvest (by number) within these estuaries, however all species were not reported by survey diarists in both estuaries. As noted above, as a small-bodied invertebrate, Eastern School Prawn were not included in calculation of $P_{e,s}$, and were thus assigned a estimated nominal value of 0.05 for this parameter in both estuaries.

Calculations on this subset of key recreational species for which both species-specific isotope modelling and catch information was available were complemented by an extension of the analysis to incorporate other species for which isotope modelling was not available, to generate valuation estimates that more closely reflected all (100%) of the catch. This extension involved calculating the proportion of catch ($P_{e,s}$) not accounted for within the species set listed above, and using this alongside the average of $C_{s,p}$ values for that estuary, to calculate $E_{s,h}$ for these 'unaccounted species'. Embedded in this calculation is the assumption that the average producer trophic subsidy across the assemblage in the estuary for which stable isotope modelling was conducted, was representative of those less commonly caught species for which species-specific trophic linkage data was not available. Obviously, this would only be testable through the collection of species-specific stable isotope data for these taxa, and for this reason these 'extended' calculations of estimated value (EV, referenced as $EV_{Extended}$) are presented as an alternate set of standalone values for consideration (if such estimates prove suitable for a particular application of this approach).

Following the calculations for recreational fisheries outlined above, the value of commercial fishing supported by coastal wetland habitats within these estuaries (previously estimated in Taylor *et al.*[17]) were adjusted from AUD_{2018} to AUD_{2021} using a CPI-conversion, and both (1) compared alongside the recreational estimates for each habitat; and, (2) added to recreational estimates to provide an estimate of the aggregate value of fishing supported by the coastal wetland habitats in the case study estuaries being considered.

Results

Partitioning of expenditure and catch

Total annual statewide expenditure on saltwater recreational fishing,[31] when converted to AUD_{2021}, ranged between AUD 61 million for residents within the inland statistical division region, to AUD 994 million for residents within the Sydney region (Table 2). Expenditure by residents within the north coast region (where the two case study estuaries are situated) totalled AUD 316 million. Analysis of 2019/20 telephone diary data[34] indicated that the proportion of effort expended in the Clarence River was greater than the Hunter River across residents from all statistical division regions, with the exception of Sydney-based fishers who expended only a small proportion of effort in both estuaries. There was a comparatively high proportion of effort from both interstate and inland statistical division region residents in the Clarence River, greater than even residents within the north coast region. The breakdown of catch also varied between estuaries (Table 3). In both estuaries, Yellowfin Bream and Dusky Flathead dominated catch, but these two species comprised a much greater overall proportion of catch in the Hunter River (~0.85) than in the Clarence River (~0.42). Mulloway, a specialist recreational fishing species, represented a small proportion of catch in both estuaries (<0.02). The non-represented catch components were ~0.34 in the Clarence River, but only ~0.08 in the Hunter River.

Partitioning of recreational fishing expenditure to coastal wetland habitats

Estimates of the monetary values (based on recreational fishing expenditure) associated with primary production in coastal wetland habitats are presented in Table 4 for individual species, and summarised in Table 5 for each estuary. The value partitioned to saltmarsh productivity was greater than mangrove habitats across all species in both estuaries, with the exception of Yellowfin Bream in the Clarence River (Table 4), where the contribution of mangrove-derived nutrition in the diet exceeded that for saltmarsh (Table 3). Overall, the estimates for the Clarence River were much greater than the Hunter River, for both Recreational EV and Recreational $EV_{Extended}$, which largely reflected greater overall fishing effort in the Clarence River (Table 2).

When monetary values were aggregated across species, estimated values attributed to primary production originating from coastal wetland habitats were as high as ~AUD 16.6 million and ~AUD 2.2 million for saltmarsh in the Clarence River and Hunter River (respectively), and up to ~AUD 9.9 million and ~AUD 1.6 million for mangroves in these two estuaries (respectively). In all cases these values exceeded CPI-corrected estimates of value for commercial fishing that were similarly estimated previously, ranging from 1.6× to 2.7× the commercial TO estimate (Table 5). These sets of estimates were summed to reveal an estimate of the aggregate value of fishing supported by primary production from each of these habitats across both sectors (commercial + recreational), which ranged between ~AUD 2.3 million for mangroves in the Hunter River, to ~AUD 24 million for saltmarsh in the Clarence River. Expressing these on an alternative per-hectare-of-habitat basis revealed these values to be as high as ~AUD 87 000 per hectare per year (extended value estimates, for saltmarsh in the Clarence River).

Table 2 Summary of per-annum expenditure data (AUD) associated with saltwater fishing from McIlgorm and Pepperell[31] by statistical division region (E_r), and estimates of proportional effort ($P_{r,e}$, rounded and expressed here as a percentage) by fishers residing within each of these regions, in each of the case study estuaries, derived from data collected by Murphy *et al.*[34]

Statistical division region	# interviews[a]	Total expenditure (saltwater fishing)		Proportional effort (%)	
		AUD_{2012}	AUD_{2021}	Clarence R. (%)	Hunter R. (%)
Sydney	366	840 749 912	993 878 329	0.1	0.1
North coast	407	267 431 019	316 139 068	5.1	1.1
South coast	117	99 200 573	117 268 284	—	—
Inland	230	51 715 617	61 134 744	6.0	0.7
Interstate	115	140 675 487	166 297 154	8.6	—

[a] Total number of completed interviews summed across sample frames for each statistical division region, as reported in McIlgorm and Pepperell.[31]

Table 3 Species-specific data including estimates of the proportion of recreational catch derived from the telephone diary survey for 2019/20 reported in Murphy *et al.*,[34] and proportion of nutrition derived from primary producers (and associated sample sizes [*n*]) in coastal wetland habitats from earlier work by Raoult *et al.*[29]

	Proportion recreational catch		Proportion nutrition[a]					
			Clarence R.[b]			Hunter R.		
Species	Clarence R.	Hunter R.	*n*	Saltmarsh[c]	Mangrove[d]	*n*	Saltmarsh[c]	Mangrove[d]
Yellowfin Bream	0.275	0.542	14	0.252	0.451	26	0.316	0.292
Dusky Flathead	0.140	0.314	14	0.531	0.297	10	0.627	0.166
Mulloway	0.012	0.016	—[e]	0.465[e]	0.175[e]	12	0.465	0.175
Giant Mud Crab	0.086	—	—[e]	0.456[e]	0.241[e]	47	0.456	0.241
Blue Swimmer Crab	0.102	—	—[e]	0.576[e]	0.183[e]	3	0.576	0.183
Eastern School Prawn	0.050	0.050	17	0.952	0.030	11	0.474	0.208
Non-represented catch	0.335	0.078	—	0.540	0.230	—	0.486	0.211

[a] Note that these proportions for saltmarsh and mangrove do not sum to 1 for each species within each estuary, because there are other sources of nutrition (other than plant species within the coastal wetland habitats considered here) that also support the biomass of these species (*e.g.*, fine benthic organic matter, see Raoult *et al.*[29]). [b] Note that data for Clarence River site C4 in Raoult *et al.*[29] was used here. [c] Parameter $C_{s,\text{saltmarsh}}$. [d] Parameter $C_{s,\text{mangrove}}$. [e] Contributions derived from isotope values from the Hunter River were used, as isotope modelling was not conducted for these species in the Clarence River.

Table 4 Per-annum recreational fishing expenditure (AUD_{2021}) attributed to primary production originating from coastal wetland habitats within the two case study estuaries, for species considered in this analysis

	Estimated value (per annum AUD_{2021})			
	Clarence R.		Hunter R.	
Species	Saltmarsh	Mangrove	Saltmarsh	Mangrove
Yellowfin Bream	2 422 284	4 335 118	868 715	802 736
Dusky Flathead	2 598 450	1 453 370	999 226	264 548
Mulloway	195 041	73 403	37 787	14 221
Giant Mud Crab	1 370 740	724 448	0	0
Blue Swimmer Crab	2 053 593	652 444	0	0
Eastern School Prawn	1 663 791	52 430	120 283	52 783
Non-represented catch	6 307 491	2 687 319	193 033	519 678

Discussion

The estimates presented here provide an indication of the value of the trophic subsidy that originates from coastal wetland habitats, based on the monetary expenditure on estuarine recreational fishing activities that are dependent on these habitats. The method of value partitioning follows simple logic, avoids complex modelling, and is highly adaptable to systems where suitable data are available from the system and/or geographic area under investigation. Also, the simplicity of the calculations involved means that estimates can be readily updated as new or improved data become available. Further application to different locations will provide an expanded set of estimates that better captures the full putative value of fisheries co-benefits that may arise through blue carbon restoration programs, and will further support the economic justification for investment in habitat conservation and restoration more broadly.

Factors influencing estimates of monetary value

The value of recreational fishing expenditure partitioned to the coastal wetland habitats greatly exceeded the estimates for commercial fishing, for both habitats in both estuaries, representing double or more the previous estimates for the value to commercial fishing in some instances. Combining estimates derived from recreational and commercial fishing activities revealed the magnitude of economic benefits that are supported by the nutrition synthesised by primary producers within coastal wetland habitats, equating to over AUD 20 million for the Clarence River (based on the data synthesised here). However, estimates were highly variable across the estuaries and habitats considered, with values ranging over an order of magnitude for mangroves in the Hunter River (which had the lowest valuation), compared to saltmarsh in the Clarence River (which had the highest valuation).

Inter-estuarine variation in partitioned recreational expenditure was influenced by a mix of estuary-specific attributes, such as the mix of species targeted in a particular estuary and their particular feeding habits, as well as the overall productivity of the estuary, and the magnitude of fishing effort that is expended within that estuary by both commercial and recreational fishers. For example, the lower Hunter River catchment, which includes substantial urban and industrial development,[26] is subject to contamination[36] and has substantial impoundment and extraction of freshwater from the major rivers that flow into the estuary. This is likely to influence recruitment processes for many of the species considered, as well as the recreational amenity and desirability of the estuary for fishers. The Clarence River, while subject to substantial floodplain management for agriculture,[28] is not impounded, is not impacted by industrial contamination, and generally receives strong recruitment for key exploited fishery species (making it the largest estuarine commercial fishery within NSW). These structural and functional factors that influence the estuarine

Table 5 Summary of per-annum recreational fishing expenditure attributed to primary production originating from coastal wetland habitats within the two case study estuaries. Both standard estimated value (referred to as Recreational EV) and extended EV (Recreational $EV_{Extended}$ [see text]) are reported. Previously reported estimates for value of commercial fishing that were similarly derived for these systems are also included, and expressed as gross value of product (Commercial GVP) and total economic output (Commercial TO). Recreational EV and $EV_{Extended}$ were summed with Commercial TO to yield Total EV and Total $EV_{Extended}$ estimates respectively (see footnote). These latter estimates reflect the cumulative monetary value of 'fishing activities' supported by each coastal wetland habitat in the case study estuaries

		Clarence R.		Hunter R.	
	Units	Saltmarsh	Mangrove	Saltmarsh	Mangrove
Habitat extent[a]	ha	280	664	509	1908
Recreational EV	AUD_{2021}	10 303 898	7 291 213	2 026 010	1 134 287
Recreational $EV_{Extended}$	AUD_{2021}	16 611 389	9 978 532	2 219 043	1 653 965
Commercial GVP[a]	AUD_{2018}	1 305 002	595 649	222 449	102 378
Commercial TO[a]	AUD_{2018}	7 207 619	3 517 005	1 312 494	604 012
Commercial GVP[b]	AUD_{2021}	1 375 550	627 849	234 474	107 912
Commercial TO[b]	AUD_{2021}	7 597 263	3 707 134	1 383 447	636 664
Total EV[c]	AUD_{2021}	17 901 161	10 998 347	3 409 457	1 770 951
Total $EV_{Extended}$[d]		24 208 652	13 685 666	3 602 490	2 290 629
Total EV[c]	AUD_{2021} ha^{-1}	63 933	16 564	6698	928
Total $EV_{Extended}$[d]	AUD_{2021} ha^{-1}	86 459	20 611	7078	1201

[a] From Taylor *et al.*[17] [b] From Taylor *et al.*[17] adjusted to 2021 Australian dollars. [c] Sum of Recreational EV and Commercial TO. [d] Sum of Recreational $EV_{Extended}$ and Commercial TO.

community are also important drivers of the fish populations (and their harvest levels) through which ecosystem services flow to beneficiaries.

Contextualising estimates

As noted in the Introduction, there is a comparative paucity of studies that consider coastal wetland value in the context of recreational fisheries, however some estimates are available. Bell[37] calculated that the capitalised value of saltmarsh to recreational fisheries in Florida ranged between \$981 and \$6471 $acre^{-1}$ (in 1984 US dollars). Carnell *et al.*[38] estimated that the value of recreational fisheries catch supported by saltmarsh and mangrove habitats in Victorian estuaries was only as high as \$15 ha^{-1} $year^{-1}$ and \$60 ha^{-1} $year^{-1}$ (respectively). Conversely, Sheld *et al.*[39] estimated the combined total benefit of living shorelines and saltmarsh to recreational fishing in Virginia to be ~\$1085 ha^{-1} $year^{-1}$. These previously published estimates range over orders of magnitude, which is likely due to the characteristics of the coastal wetlands and recreational fisheries evaluated, and the various valuation approaches employed. Notwithstanding differences in the age of the studies and the currency and units in which estimates are expressed, these previous estimates are mostly lower than those reported in the present study, possibly because they do not effectively capture the full scope of expenditure associated with recreational fishing.

In addition to support of fisheries, coastal wetland habitats provide substantial benefits to humans that are unrelated to extractive uses, and two of the most important include regulating and maintenance services. The overview of de Groot *et al.*[4] shows that the monetary values for regulating and maintenance services provided by coastal wetlands can range from \$65 ha^{-1} $year^{-1}$ for climate regulation, \$3929 ha^{-1} $year^{-1}$ for erosion prevention, to as much as \$162 125 for waste treatment (values are 2007 Geary–Khamis dollars[4]). In comparison, food provisioning services and cultural services for coastal wetlands were valued at \$1111 ha^{-1} $year^{-1}$ and \$2193 ha^{-1} $year^{-1}$ respectively.[4] Valuation of diverse ecosystem services inevitably incorporates varying approaches with variable levels of rigour and different assumptions, which mean that estimates may not always be directly comparable.[10] Consequently, when framing the monetary values associated with ecosystem services, it is essential to present in detail the quantitative basis for the estimates, the data sources used, and assumptions of the approach employed, as demonstrated here for recreational fisheries. This ensures that differences can be accounted for when comparing or aggregating values associated with different ecosystem functions, or using different transfer methods.

Assumptions, limitations, and caveats

Expenditure on recreational fishing activities provides a reasonable proxy for market-based value, although does not completely capture non-market benefits. Failure to account for these could mean that the estimates derived are conservative. Furthermore, when considering the estimates produced using the trophic subsidy approach, it is important to be aware of the fundamental assumptions of the method—(1) that modelling of stable isotopes effectively describes the trophic relationships, and these relationships are temporally stable; (2) that the fishery and/or market-based economic data reasonably reflect the bulk of the revenue derived from the activity that is supported by the ecosystem function; and (3) that the monetary values employed provide a good indicator of 'importance'. The utility of stable isotope data in this context has been discussed in recent manuscripts,[3,22,40] so is not repeated here. While the value of wetland habitat to the production of exploited species across a range of trophic levels was considered through trophic modelling, a given habitat extent inherently supports much larger lower trophic level biomass relative to higher trophic level biomass.[41] Future valuation exercises may wish to also consider the value of intermediate consumers in these food webs, especially in environments where they may have little other societal impact.

As with any model, the estimates derived will only be as good as the data incorporated, and bias may be introduced through the various data sets employed and the way in which they were collected. McIlgorm and Pepperell[31] used a defensible approach to the estimation of this data, drawing on a substantial (and statistically adequate) number of survey respondents. However, the recall survey method they used, where interviewees were asked to recall activities over the previous 6 months, can suffer from memory bias. This can be somewhat overcome by longitudinal telephone diary surveys with frequent call backs (monthly contact was used in Murphy *et al.*[34]), and an improved approach could involve incorporating expenditure-related questions into these telephone diary surveys where more frequent contact is made. This could probably be achieved with a comparatively modest increase in cost to these surveys, but would come with the added benefit of allowing expenditure to be directly mapped to spatial fishing habits on a fisher-by-fisher basis, prior to survey expansion. It is not clear if delving more deeply in this fashion would necessarily lead to better estimates, but we raise the issue here for future consideration by researchers or practitioners.

As expenditure is linked to effort, the application of these data therefore assumes that fishing effort levels have remained comparatively consistent since the expenditure survey was conducted (in 2012). Updated estimates of absolute fishing effort across the entire population would also be beneficial, however, the last survey of NSW fishers to use a sample frame encompassing the entire population was conducted in 2013.[42] Unfortunately, such surveys are becoming increasingly difficult as the use of landlines and conventional phone books declines,[43,44] and reliance on more modern digital platforms also comes with a new set of biases. Overall participation in recreational fishing is obviously an important factor that drives overall expenditure across the population, and this in turn directly impacts valuation using the methodology described here. Decreasing participation through time is expected in developed nations,[45] and it follows that this will impact the expenditure on recreational fishing activities, which in turn impacts the monetary value that may be attributed in the fashion presented here.

Finally, the trophic subsidy method deals exclusively with attribution of economic data through trophic flows, but in doing so may not completely capture the ‘nursery function’ conferred through occupation of coastal wetland habitats[46] and the value of this to either recreational or commercial fisheries. While there are other approaches that deal with recruitment subsidies accumulating from wetland nursery function (such as production enhancement modelling[47]), the trophic subsidy approach is likely to indirectly account for at least some of the benefits associated with this (*i.e.*, the provisioning functions of the nursery). Furthermore, fisheries productivity will also benefit from other ecosystem functions that improve environmental condition, such as regulation of water quality[48] and sediments,[49] which are not directly addressed in the trophic subsidy method.

Broader application of the approach

For application to other systems, some basal data requirements must be satisfied. Firstly, some assessment of trophic flows in the system are required for effective partitioning of expenditure (or other economic data). These data can be easily collected by replicating the methodology outlined in similar recent studies of estuarine food webs[3,40] in the estuary of interest. Alternately, if this is not possible, data and analysis from similar systems elsewhere, or meta-analyses (*e.g.*, Jänes *et al.*[24]) may be incorporated in their place. Recreational fishing survey and effort data is required, which is available in many jurisdictions, either at a state or national level. However, the spatial granularity to which survey data can be broken down depends on the nature of data collected, and increasingly finer spatial scales can often come at the cost of precision in estimates.

Recreational fishing expenditure data is most commonly collected and reported at broader spatial scales (*i.e.*, the jurisdictional or national level), but may be available at the estuary level in some cases. Availability and use of estuary-specific data would further simplify application of this approach. The principles underlying our method are flexible in their nature, and consequently are readily adapted to different types and forms of recreational fisheries expenditure surveys and economic data. This is important, because as noted earlier, the expenditure data used here do not incorporate non-market benefits, but economic data derived from other valuation techniques at a later date could be attributed to different habitat types using this approach. The ability to incorporate different types of data largely supports broader application to other systems, either to incorporate an estimate of the value of coastal wetland habitats to recreational fishers into existing valuations, or support estimates in places where commercial fishing does not occur (such as recreational fishing havens in NSW[50]).

Implications for the case study estuaries

The patterns derived for the case study estuaries are interesting to consider in a local context. Firstly, it is clear that tourism is a major driver of recreational fisheries expenditure in the Clarence River, with a larger proportion of interstate and inland effort (and expenditure) partitioned to this estuary. In contrast, Hunter River effort was dominated by north coast residents, which are probably local to the estuary. This is not surprising, since the Clarence River is a regional, sub-tropical location and is one of NSW most productive estuaries (in terms of fisheries production), and thus more likely to be a desirable destination for holiday makers intending to fish. The Hunter River estuary is adjacent to NSW's second-largest city (City of Newcastle), and as noted earlier is a highly impacted estuary surrounded by substantial industrial development, and subject to contamination. While Newcastle is a popular tourist destination, it is probably a much less popular destination for estuarine fishingrelated tourism, as is clearly reflected in the partitioning of the effort data.

The Clarence River (and current and former wetlands therein) is frequently identified as a high priority blue carbon site, and recent assessments have identified significant opportunities for wetland restoration and highlighted the potential benefits for fisheries in the estuary.[28,51] The most recent opportunity for habitat repair proposed in the Clarence River is Everlasting Swamp, a ~1300 hectare former estuarine wetland of substantial blue carbon potential.[52] It is unlikely that the value to commercial and/or recreational fishing will scale linearly and unbounded with habitat area for such a large site, as other factors may in turn limit fisheries productivity, such as juvenile recruitment. However, the size of this former wetland, considered alongside the estimated per-hectare value of habitats therein to fishers, does suggest that the value associated with fisheries-related co-benefits could be in the order of many millions of dollars per year, if the trophic subsidy and enhanced fish productivity lead to enhanced catch, effort, and tourism.

Conclusion

The trophic subsidy method provides an adaptable and feasible option for fisheries habitat managers and restoration practitioners to consider quantitative estimates of the value of coastal wetland habitats to recreational fishers, in the context of their conservation, repair or restoration efforts. Accounting for these values may support improved pricing of environmental offsets, improved valuation of enhanced fisheries productivity associated with habitat repair, derivation of more holistic 'habitat value' estimates (that include value to both commercial and recreational fishers) within environmental economic accounting activities, and premium pricing of blue carbon offsets that fully account for the value of fisheries co-benefits.

Data availability

Data used in this article are available from the sources outlined in Table 1.

Conflicts of interest

There are no conflicts to declare.

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