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# Use of a seagrass residency index to apportion commercial fishery landing values and recreation fisheries expenditure to seagrass habitat service

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**Abstract:** *Where they dominate coastlines, seagrass beds are thought to have a fundamental role in maintaining populations of exploited species. Thus, Mediterranean seagrass beds are afforded protection, yet no attempt to determine the contribution of these areas to both commercial fisheries landings and recreational fisheries expenditure has been made. There is evidence that seagrass extent continues to decline, but there is little understanding of the potential impacts of this decline. We used a seagrass residency index, that was trait and evidence based, to estimate the proportion of Mediterranean commercial fishery landings values and recreation fisheries total expenditure that can be attributed to seagrass during different life stages. The index was calculated as a weighted sum of the averages of the estimated residence time in seagrass (compared with other habitats) at each life stage of the fishery species found in seagrass. Seagrass-associated species were estimated to contribute 30%–40% to the value of commercial fisheries landings and approximately 29% to recreational fisheries expenditure. These species predominantly rely on seagrass to survive juvenile stages. Seagrass beds had an estimated direct annual contribution during residency of €58–91 million (4% of commercial landing values) and €112 million (6% of recreation expenditure) to commercial and recreational fisheries, respectively, despite covering <2% of the area. These results suggest there is a clear cost of seagrass degradation associated with ineffective management of seagrass beds and that policy to manage both fisheries and seagrass beds should take into account the socioeconomic implications of seagrass loss to recreational and commercial fisheries.*

**Keywords:** commercial fisheries, ecosystem services, habitat valuation, seagrass, recreational fisheries

El Uso de un Índice de Residencia de Pastos Marinos para Distribuir Valores de Desembarco de Pesquerías Comerciales y Gastos de Pesquerías Recreativas para Servicios de Hábitat de Pastos Marinos

**Resumen:** *Donde los pastos marinos dominan las costas, estos tienen un papel fundamental en el mantenimiento de poblaciones de especies explotadas. Por esto, los pastos marinos en el Mediterráneo merecen protección, pero no se ha hecho intento alguno por determinar la contribución de estas áreas a los atraques de las pesquerías comerciales y los gastos de las pesquerías recreativas. Existe evidencia de que la distribución de los pastos marinos continúa declinando, pero hay poco entendimiento de los impactos potenciales de esta disminución. Usamos un índice de residencia de pastos marinos, con base en la evidencia del ciclo de vida, para estimar la proporción de valores de desembarco de pesquerías comerciales y de gastos totales de pesquerías recreativas que se le puede atribuir a los pastos marinos durante diferentes etapas de vida. El índice se calculó como la suma ponderada de los promedios de tiempos de residencia estimada en los pastos marinos (en comparación con otros hábitats) durante cada etapa de vida de las especies objeto de pesquerías que se encuentran en los pastos. Se estimó que las especies asociadas a los pastos contribuyen entre el 30*

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*y el 40% al valor de desembarco de las pesquerías comerciales y aproximadamente al 29% del gasto de las pesquerías recreativas. Estas especies dependen predominantemente de los pastos marinos para sobrevivir a las etapas juveniles. Durante la residencia, se estimó que los pastos marinos tuvieron una contribución anual directa de €58-91 millones (4% de los valores de desembarco comercial) y de €112 millones (6% del gasto recreativo) de las pesquerías comerciales y recreativas, respectivamente, a pesar de tener una cobertura menor al 2% del área. Estos resultados sugieren que existe un costo claro de la degradación de pastos marinos asociado con el manejo poco efectivo de los mismos y que la política de manejo tanto de las pesquerías como de los pastos marinos debe considerar las implicaciones socio-económicas de la pérdida de los pastos para las pesquerías comerciales y recreativas.*

**Palabras Clave:** pastos marinos, pesquerías comerciales, pesquerías recreativas, servicios ambientales, valoración de hábitat

## Introduction

Fisheries provide food security and socioeconomic and recreation benefits to humans (Pauly 2009). In the Mediterranean, commercial fisheries target a wide diversity of species (Rätz et al. 2010), and over 80% of fishing vessels registered in 2008 were artisanal fishing boats operating mostly inshore and targeting local markets for sale of their landings (Sacchi 2011). Total commercial first sale landings of species in the Mediterranean is approximately €1.6–2.2 billion/year (Hutniczac & Roth 2012). Recreational fishing activities also contribute to the economy (Toivonen et al. 2004; Pawson et al. 2007). In Europe alone, €25 billion is spent per year by anglers on equipment, transportation, and lodgings (Dillon 2004). Both recreational and commercial fishing activities depend on the physical resource and functioning of marine ecosystems (Bell 1997; Rees et al. 2010).

Seagrass meadows are considered to have a fundamental role in maintaining populations of commercially and recreationally exploited fisheries species by indirectly supporting coastal food webs (Vizzini et al. 2002) and directly providing one or more of the following: permanent habitat, allowing full life cycle completion; temporary nursery area for juvenile development; feeding area for various life-history stages; and refuge from predation (Jackson et al. 2001). These provisions have been called habitat services (TEEB 2010), and they improve fisheries and shellfisheries' quality and quantity and secure supply.

A seagrass fringe is found on almost all coasts of the Mediterranean (Green & Short 2003); estimated coverage is 2.5–5.5 million ha (Buia et al. 2000). The primary species of seagrass is the Mediterranean endemic *Posidonia oceanica*. These meadows are protected under the European Council (EC) Habitats Directive within designated protected areas (European Marine Sites), and EC Regulation 1626/94 excludes mobile fishing gear use within 1.5 nautical miles of the coast, where the seagrass occurs. Despite this protection, seagrass is in decline in the Mediterranean (Langmead et al. 2007) and many areas lack enforcement of protection measures. Principle 4 of the ecosystem approach (as defined by the CBD 2000) states that in order to recognize "potential gains from management, there is a need to understand and manage the ecosystem in an economic context." This requires an understanding of the economic associations of ecosystem services and markets through delineation of links between ecosystem state changes and impacts of these changes on human welfare. Identifying the services and underlying process and examining ways of valuing ecosystem services helps provide an economic argument for conservation and can inform decisions on the cost and benefits of development options.

Methods to provide evidence of the economic importance of seagrass ecosystem services in relation to commercial and recreational fisheries have developed over the last 15 years (Scott et al. 2000; Rönnbäck et al. 2007; Unsworth et al. 2010), but they remain largely biased by a lack of consideration of temporary seagrass use within a species' lifecycle, a consideration of only commercial fishery landings values, or an over reliance on expert opinion (but see McArthur et al. 2003; Blandon & zu Ermgassen 2014). A specific seagrass meadow's importance for habitat provision depends on pre- and postsettlement processes (Jackson et al. 2001), the availability and suitability of other areas, and the preferences, ontogenetic habitat shifts, and survival rates of different species. Often no data on survival and recruitment to adult stocks exist or are difficult to measure in the field, while age at maturity and maximum age information is more easily accessed and habitat use for different species is available for many areas.

To address the effect of ontogenic shifts in habitat association, Scott et al. (2000) formulated a seagrass residency index (SRI). They used residency as a surrogate for dependency and a weighted sum of the averages of the estimated residence time in seagrass (compared with other habitats) at each life stage. Scott et al. (2000) collated species life history information and habitat use from expert knowledge, which not only limited the number of species which could be incorporated in their analysis, but also limited the study to a small geographical region.

We sought to objectively assess the economic importance of seagrass as fish habitat. We used biological life history trait databases and published reports of habitat preference to identify potential residency in seagrass and to apportion seagrass contribution to the fisheries stock. In addition to apportioning the direct commercial fishery landings values and recreational fishery expenditure associated with seagrass, we considered the implication of seagrass loss for regional fisheries policy.

## Methods

### Identifying Commercial and Recreationally Important Species

Species were identified as being commercially and recreationally targeted if they were listed on the General Fisheries Commission for the Mediterranean (GFCM) fisheries capture statistics (1950–2010) (Food and Agriculture Organisation 2012). All species names were checked, updated and, where necessary, amalgamated, based on the World Register of Marine Species (WoRMS) (www.marinespecies.org).

### Calculating Seagrass Residency Indices

The extent of seagrass association varies among species. For example, black seabream (*Spondyliosoma cantharus*) is abundant on seagrass meadows, but it also occurs in a variety of other habitat types, whereas blackspot seabream (*Pagellus bogaraveo*) appears to be relatively ubiquitous. To differentiate habitat use, we calculated SRI (Scott et al. 2000) values for the relative amount of time a species spends in seagrass during different life stages relative to other habitats. If juvenile (highest mortality rate) survival is low, the species' contribution to fished stock will be low. Therefore, the amount of time spent in seagrass as a juvenile was also weighted relative to the adult stage. The final SRI was calculated as

$$\mathrm{SRI}_i = 1 - \exp\{-[\exp(-m_i(t_{ji} - t_{ai}))xi + y_i]\}, \quad (1)$$

where exp is the exponential; for each species $i$, $m$ is natural mortality; $t_{ji}$ is the time (years) spent as a juvenile; $t_{ai}$ is the time (years) spent as an adult; $x_i$ is the proportion of time spent in seagrass as a juvenile; and $y_i$ is the proportion of time spent in seagrass as an adult (Scott et al. 2000). The proportions of time spent in seagrass as an adult and as a juvenile were calculated as

$$x_i = \frac{(t_{ji}/t_i)}{(1/H_{ji})} \quad (2)$$

and

$$y_i = \frac{(t_{ai}/t_i)}{(1/H_{ai})}, \quad (3)$$

where $t$ is the average maximum age, $H_a$ is the number of habitats the species has been recorded as using (association) as an adult and $H_j$ is the number of habitats the species has been recorded as using (association) as a juvenile.

A systematic literature review was performed on habitat use (presence or absence) of commercially and recreationally important Mediterranean species. Where information was available, dominant or key habitats were noted, both for the species overall and for each life cycle stage. We used peer-reviewed literature, online databases, and factsheets (Fishbase, SeaLifebase, Larvalbase, Marine Species Information Portal, MarLIN, BIOTIC, Arkive, FAO factsheets, and FAO Adrimed). Key habitats were primarily derived from these widely acknowledged sources, unless specific mention was made of reliance on a particular habitat in the literature. Habitat association at each life stage for each species was ranked as 0 (no record found), 1 (rarely recorded or unusual), or 2 (consistently present in some abundance or many instances in the literature). This ranking was to illustrate confidence and was not used in the weighting calculations (Supporting Information).

For each species, we recorded the minimum, maximum, and mean duration of each life stage (juvenile and adult; years). These data were sourced from BIOTIC, Fishbase, and peer-reviewed literature. The juvenile stage included egg development time, larval phase duration, and juvenile stages up to sexual maturity or adulthood. For maximum adult age, the average maximum age was used where available, rather than the oldest on record, to prevent unrealistic skewing of results in fished populations. For sequential hermaphrodites the age of first becoming reproductively active was used; for gonochoristic species, with different male and female maturation ages, an average was taken (accounting for sex ratio, usually 1:1). We used Mediterranean data when possible and data from areas with similar water temperature otherwise. Where species groups (usually genus) were used, data were obtained to represent the most common species in the Mediterranean.

To calculate the weighting coefficient for the juvenile life stage, an estimate of natural mortality ($m$) was required. Some values for $m$ are available in the literature (e.g., Pauly 1980); however, because temperature affects $m$, all scores were re-estimated using the life history tool in Fishbase (Froese et al. 2005). This tool applies the maximum length and water temperature as follows:

$$m = 10^{(0.566-0.718*\log(Linf)} + 0.02 * T, \quad (4)$$

where $T$ is the mean annual water temperature; $L_{\mathrm{inf}}$ is the length that the fish within a population would reach if they were to grow indefinitely (also known as asymptotic length [Froese & Pauly 2000]). The mean Mediterranean Sea temperature (surface to 150 m) of 16.3 °C was taken from the annual average for 1960–2000 (Beuvier et al. 2010). Because many commercially valuable fish (particularly pelagic egg and larval stages) are found at shallower depths, where warmer temperatures allow for faster growth, this is likely to be a conservative estimate. Asymptotic length ($L_{\mathrm{inf}}$) was stated using accepted protocol for maximum length (i.e., fork length for scombroids [tuna and tuna-like fishes] and total length for all other fishes). Because we used length as a proxy for weight, natural mortality will be underestimated in eel-like fish and overestimated in sphere-shaped fish. Finally, age at first maturity (average age at which fish of a given population mature for the first time) was collated using the Fishbase life history tool or taken from peer reviewed publications (Supporting Information). Life history information and calculated SRI are shown in Table 1.

### Commercial Fisheries Landings Values

To determine Mediterranean commercial fisheries landings value (CFV), data were extracted from the Commercial Harvest Database (Hutniczac & Roth 2012); this database evaluates European fishing in monetary terms. To generate commercial fishing landings values in the Mediterranean, time-series data for landings in Mediterranean fishing zones (2006–2008) were derived from the GFCM database (Food and Agriculture Organisation 2012). Landings data are recorded as live weight equivalent. Data for the price per ton (ungutted fish) for landings were derived from the European Union statistics portal, as estimated by the national authorities when reporting to the European Commission (European Commission 2012). Mediterranean landings of commercially important fish from 2006 to 2008 achieved an annual average first sale estimated value of €1.9 billion (SD €294.8 million) (Hutniczac & Roth 2012).

### Recreational Fisheries Expenditure

Recreational fishing is defined here as "fishing activities exploiting marine living aquatic resources from which it is prohibited to sell or trade the catches obtained" (General Fisheries Commission for the Mediterranean 2010). For management purposes in the GFCM region, recreational fishing is divided into the following categories: leisure fishing, fishing practiced for pleasure; sport fishing, fishing contest practiced within an established institutional framework which sets rules, collects data on catches, and reveals to the public the outcomes of the event; charter fishing, recreational fishing practiced from a rented boat with a captain or fishing guide on board for leisure or sport purposes; and underwater fishing, fishing practiced as a sport or for leisure by snorkeling without the help of mechanical devices (e.g., a scooter).

*Pesca turismo,* or tourism fishing, is not included as a definition of recreational fishing because it is not solely geared toward recreational fishing (Gaudin & Young 2007; General Fisheries Commission for the Mediterranean 2012).

A variety of sources were used to determine Mediterranean recreational fishing expenditure. A literature search was conducted to identify data related to the recreational fishing activity of a Mediterranean country. Where primary studies were conducted, the study at the largest spatial scale (e.g., national level) was included for analysis. If a study included primary data on the number of recreational fishers (e.g., fishing licenses), then the broad expenditure per year for the number of fishers was derived from the regional economic expenditure for recreational fishers (demonstrated in Cisneros-Montemayor & Sumaila 2010). All recreational expenditure is expressed as expenditure per year by recreational fishers in a particular country. Types of expenditure included in each study varied, so expenditure could not be compared among countries unless reference was made to the original study. Annual expenditure of Mediterranean recreational fishers was therefore estimated at €2.5 billion. A full valuation table for recreation fishing in the Mediterranean is in the Supporting Information.

### Application of SRI to Landings and Expenditure

To apportion the commercial CFVs and recreational fisheries expenditure to seagrass habitat provision service in the Mediterranean (CFVseagrass), the Commercial Harvest Database Mediterranean section (Hutniczac & Roth 2012) was filtered to include those commercial fish species (icom in Eq. 5 associated with seagrass meadows [$n = 56$]). The average, minimum, and maximum annual landings values for the years 2006–2008 were calculated for each species. Commercial landings values related to pooled species (e.g., *Epinephelus* spp.) are included because the Commercial Harvest Database provides catch data on both individual species and broad species groups depending on the reporting requirements of the country where landings were made. We did not consider this double counting of commercial landings values because the fish is landed and categorized for reporting only once. If scores existed for broad species groups only, then we applied the SRI for the most abundant species in that group:

$$\text{CFV seagrass} = \sum \text{CFV Mediterranean}^{(\text{icom})} \times \text{SRI}^{(\text{icom})}. \tag{5}$$

To apply the SRI to recreation expenditure (RFVseagrass), it was necessary to determine those species favored by recreational fishers. To provide a measure of the proportion of recreation expenditure to assign to fish species targeted by recreational fisheries, the mean catch composition of recreational fishers in Spain, France, and Italy was used as a proxy for the wider Mediterranean recreational fisheries (Gordoa et al. 2004). This demonstrated that 27.2% of the recreational catch was species associated with seagrass ($n = 7$, irec in Eq. 6).

**Table 1. Organism life history traits and proportion of time spent in seagrass at different life stages used to calculate the seagrass residency index (SRI) (for details see Supporting Information).[*]**

| *Common name* | *Scientific name* | *tMat (years)* | *tMax (years)* | *m* | *x* | *y* | *tj* | *ta* | *SRI value* |
|---|---|---|---|---|---|---|---|---|---|
| *Fish* | | | | | | | | | |
| axillary seabream | *Pagellus acarne* | 2 | 9 | 1.35 | 0.042 | 0.506 | −1.54 | −9.00 | 0.397 |
| black seabream | *Spondyliosoma cantharus* | 4 | 15 | 1.46 | 0.052 | 0.153 | −1.96 | −10.00 | 0.142 |
| blotched picarel | *Spicara maena* | 2.5 | 9.1 | 0.98 | 0.035 | 0.041 | −1.88 | −9.10 | 0.040 |
| bluefish | *Pomatomus saltatrix* | 2 | 9 | 1.66 | 0 | 0.086 | −0.65 | −6.00 | 0.083 |
| bogue | *Boops boops* | 1 | 6 | 1.33 | 0 | 0.083 | −1.37 | −11.00 | 0.080 |
| common dentex | *Dentex dentex* | 1 | 28 | 1.63 | 0.008 | 0.121 | −4.60 | −26.00 | 0.114 |
| common pandora | *Pagellus erythrinus* | 1 | 20 | 1.33 | 0.079 | 0.407 | −0.65 | −20.00 | 0.334 |
| dusky grouper | *Epinephelus marginatus* | 5 | 26 | 1.72 | 0 | 0.114 | −2.51 | −15.00 | 0.108 |
| european conger | *Conger conger* | 5 | 15 | 1.94 | 0.037 | 0.074 | −2.05 | −6.00 | 0.071 |
| european eel | *Anguilla anguilla* | 8.9 | 41 | 0.14 | 0.035 | 0.041 | −2.16 | −10.10 | 0.041 |
| European flounder | *Platichthys flesus* | 3 | 15 | 1.40 | 0.034 | 0.110 | −1.59 | −14.00 | 0.104 |
| European pilchard( = Sardine) | *Sardina pilchardus* | 2.5 | 10 | 1.19 | 0.030 | 0.151 | −0.15 | −4.00 | 0.140 |
| European seabass | *Dicentrarchus labrax* | 2 | 14 | 1.58 | 0.023 | 0.105 | −1.55 | −10.00 | 0.099 |
| garfish | *Belone belone* | 2 | 8.4 | 1.50 | 0.059 | 0.190 | −2.57 | −12.00 | 0.173 |
| gilthead seabream | *Sparus aurata* | 1 | 12 | 1.50 | 0.019 | 0.063 | −5.75 | −41.04 | 0.061 |
| grass goby | *Zosterisessor ophiocephalus* | 4.3 | 16.6 | 0.38 | 0.122 | 0.297 | -2.95 | -16.61 | 0.257 |
| pouting( = bib) | *Trisopterus luscus* | 1 | 5 | 1.44 | 0 | 0.200 | −0.67 | −5.00 | 0.181 |
| red mullet | *Mullus barbatus* | 2.7 | 10.1 | 1.23 | 0 | 0.037 | −2.51 | −6.01 | 0.036 |
| red porgy | *Pagrus pagrus* | 3 | 12 | 1.60 | 0.061 | 0.188 | −3.53 | −15.00 | 0.171 |
| saddled seabream | *Oblada melanura* | 2 | 11 | 1.37 | 0 | 0.091 | −0.35 | −5.00 | 0.087 |
| salema porgie | *Sarpa salpa* | 1.8 | 11 | 1.44 | 0.031 | 0.075 | −4.64 | −15.00 | 0.073 |
| sand steenbras | *Lithognathus mormyrus* | 2.5 | 12 | 1.39 | 0 | 0.096 | −1.54 | −11.00 | 0.092 |
| surmullet | *Mullus surmuletus* | 1 | 8 | 1.29 | 0.013 | 0.252 | −0.65 | −8.00 | 0.238 |
| white seabream | *Diplodus sargus* | 2 | 10 | 1.42 | 0.010 | 0.100 | −2.02 | −12.00 | 0.095 |
| whiting | *Merlangius merlangus* | 1.5 | 10 | 1.41 | 0.092 | 0.121 | −0.68 | −28.00 | 0.114 |
| Arthropoda | | | | | | | | | |
| caramote shrimp | *Melicertus kerathurus* | 0.29 | 2.5 | 0.94 | 0.0232 | 0.221 | −0.15 | −2.56 | 0.200 |
| common prawn | *Palaemon serratus* | 0.58 | 5 | 1.50 | 0 | 0.089 | −1.60 | −9.00 | 0.085 |
| spinous spider crab | *Maja squinado* | 2.5 | 6 | 1.50 | 0 | 0.073 | −0.67 | −2.07 | 0.070 |
| common shrimp | *Crangon crangon* | 1 | 2 | 1.50 | 0 | 0.071 | −0.67 | −12.00 | 0.069 |
| Mediterranean Green crab | *Carcinus aestuarii* | 0.5 | 7 | 1.50 | 0 | 0.028 | −0.30 | −7.00 | 0.027 |
| Molluscs | | | | | | | | | |
| common cuttlefish | *Sepia officinalis* | 1 | 2 | 1.30 | 0.125 | 0.250 | −1.57 | −8.40 | 0.221 |
| common octopus | *Octopus vulgaris* | 0.28 | 4 | 1.31 | 0 | 0.145 | −1.09 | −10.00 | 0.135 |
| Mediterranean mussel | *Mytilus galloprovincialis* | 1 | 50 | 1.50 | 0 | 0 | −2.56 | −9.00 | 0.001 |

[a] *Abbreviations: tMat, age at maturity; tMax maximum age; m, mortality; x, proportion of time (years) spent in seagrass as a juvenile; y, the proportion of time (years) spent in seagrass as an adult.*

Therefore, 27.2% of RFVMediterranean was carried forward for analysis. Where recreation expenditure is associated with broad species groups only, then the application of the SRI for the most abundant species on that group was applied:

$$\text{RFV seagrass} = \sum \text{RFV Mediterranean}^{(irec)} \times \text{SRI}^{(irec)}. \quad (6)$$

## Results

The total annual landings value of commercial species that were, at some stage in their life history, associated with seagrass was approximately €681.2 million/year (35% of the total landings value of commercial fisheries in the Mediterranean [Table 2]). When the SRI scores were applied to determine the proportion of landings value directly attributable to the presence of seagrass, approximately €77.7 million (€58.3 million to €91.4 million; approximately 4% of the Mediterranean CFV) was directly associated with seagrass presence (Table 2). No Mediterranean fisheries' species was identified as being totally dependent on seagrass. Even the most strongly associated species, the axillary seabream (*Pagellus acarne*), showed some association with other habitats. From an ecological perspective, the commercial species that would be most affected by the loss of seagrass (highest SRI scores) are axillary seabream (*Pagellus acarne*), scorpionfishes and rockfishes (Scorpaenidae and Scorpaena), common pandora (*Pagellus erythrinus*), poor cod (*Trisopterus minutus*), caramote shrimp (*Melicertus kerathurus*), and great Atlantic scallop (*Pecten maximus*) (Table 1). From an economic perspective, the largest losses in landings if seagrass were to decline would be in commercial fisheries that target cuttlefish (Sepiidae, Sepiolidae), scorpionfishes (Scorpaenidae), octopuses (Octopodidae), European anchovy (*Engraulis encrasicolus*), and European pilchard (*Sardina pilchardus*). Despite having a minimal association with seagrass, the latter 2 species are so widely landed in in the Mediterranean that the economic impact on the fisheries could be substantial.

The total annual expenditure of recreational fishers pursuing catches of species that, at some stage in their life history, depend on the presence of seagrass was estimated at €723.3 million (approximately 28% of all Mediterranean recreational fishing activity; Table 2). When the SRI scores were applied to determine the proportion of that expenditure that could be directly attributed to the presence of seagrass, €112.6 million of that expenditure (approximately 6% of RFVMediterranean) could be attributed the presence of seagrass (Table 2). In terms of impacts on the recreation industry, from an ecological perspective the species that would be most affected by the loss of seagrass are scorpionfishes and rockfish (Scorpaenidae, *Scorpaena* spp.), porgies, seabreams (Sparidae), cuttlefish (Sepiidae), garfish (*Belone belone*), and pouting (*Trisopterus luscus*) (Table 1). From an economic perspective, the largest losses in associated expenditure if seagrass were to decline would be from the recreational fisheries that target seabass (*Dicentrarchus* spp.), cuttlefish (Sepiidae, Sepiolidae), and scorpionfishes (Scorpaenidae).

## Discussion

By providing an essential habitat, we found that seagrass meadows had an estimated direct contribution of 4% to the total value of landings of commercial fisheries and 6% to the total expenditure of recreational fisheries, equating to a total of approximately €78 million and €112 million per year, respectively. The total area of *Posidonia oceanica* was estimated at about 5.5 million ha (Buia et al. 2000), or 2% of the surface area of the Mediterranean Sea; thus, this habitat disproportionately contributed to these Mediterranean fisheries in terms of "habitat service" (TEEB 2010). We found that seagrass had higher direct market value (in terms of landings values and expenditure) to recreational fisheries than to commercial fisheries, due to the different target species of the 2 fishing sectors and the different markets under which each sector operates. This highlights that studies which consider only the value of commercial fisheries landings will greatly underestimate the contribution of seagrass (or other habitats) to wider regional economic activity (e.g., sport fishing). Additionally, we demonstrated that any loss or degradation of seagrass is likely to have a disproportionate negative economic impact on fishery sectors with smaller inshore fisheries.

Previous studies assessing the economic importance of seagrass in relation to fishery species habitats focused on generic assessments which do not consider potential specificity or nonlinearity in service provision. For example, Rönnbäck et al. (2007) carried out a semiquantitative valuation of seagrass habitat secondary production by noting habitat association, market values, and landings of commercially fished species (see Rönnbäck et al. 2007). Although useful for assessing the potential implications of ecosystem state change on local markets, such an assessment does not consider the reliance of different life stages on different habitats, the effect of seagrass quality and density on fish production, or the indirect importance of species which, while not themselves commercially harvested, may be significant prey consumed by commercial species. The incorporation of seagrass quality and food web dynamics would greatly improve the accuracy of economic estimates, but this relies on the collation of extensive primary data. Such data on the reliance of different life stages on seagrass meadows are available in the literature, however, and provide a more realistic view of the contribution of seagrasses to commercial and recreationally important stocks. Without the application of the SRI (i.e., examining only habitat association of species [sensu Rönnbäck et al. 2007]), our data suggest

**Table 2. Commercial and recreational fisheries value for species associated with seagrass habitat in the Mediterranean, their calculated seagrass residency index (SRI), and the proportion of value directly attributable to the presence of seagrass.[a]**

| *Common name* | *Scientific name* | *CFV[b] Med[c] €* | *SD* | *RFV Med €* | *SRI* | *CFV Seagrass €* | *RFV Seagrass €* | *CFV Seagrass low €* | *CFV Seagrass high €* |
|---|---|---|---|---|---|---|---|---|---|
| *Chordata* | | | | | | | | | |
| axillary seabream | *Pagellus acarne* | 147,570 | 137,200 | | 0.397 | 58,643 | | 4,121 | 113,164 |
| black seabream | *Spondyliosoma cantharus* | 1,767,301 | 1,477,695 | | 0.142 | 250,433 | | 41,038 | 459,827 |
| blue whiting | *Micromesistius poutassou* | 9,841,546 | 2,780,560 | 7,396,395 | $0.40 \times 10^{-6}$ | 4 | 3 | 3 | 5 |
| Bluefish | *Pomatomus saltatrix* | 1,787,761 | 668,534 | 11,834,231 | 0.083 | 148,095 | 980,326 | 92,715 | 203,475 |
| Bogue | *Boops boops* | 14,385,930 | 877,107 | | 0.080 | 1,150,235 | | 1,080,106 | 1,220,365 |
| Brill | *Scophthalmus rhombus* | 388,178 | 46,240 | | $0.49 \times 10^{-5}$ | 19 | | 17 | 21 |
| chub mackerel | *Scomber japonicus* | 4,865,828 | 4,359,487 | | 0.000 | 193 | | 20 | 367 |
| common dentex | *Dentex dentex* | 5,416,204 | 4,708,978 | | 0.114 | 615,035 | | 80,309 | 1,149,761 |
| common pandora | *Pagellus erythrinus* | 5,265,870 | 2,516,492 | | 0.334 | 1,759,713 | | 918,769 | 2,600,657 |
| conger eel | *Conger conger* | 0 | 0 | 23,668,463 | 0.105 | 0 | 2,485,499 | 0 | 0 |
| dentex nei | *Dentex spp* | 360,844 | 123,360 | | 0.121 | 43,656 | | 28,731 | 58,580 |
| dusky grouper | *Epinephelus marginatus* | 2,912,992 | 1,168,600 | | 0.108 | 314,594 | | 188,389 | 440,799 |
| European anchovy | *Engraulis encrasicolus* | 157,778,239 | 30,900,822 | | 0.058 | 9,174,719 | | 7,377,853 | 10,971,585 |
| European conger | *Conger conger* | 2,652,773 | 1,423,742 | | 0.071 | 189,400 | | 87,749 | 291,051 |
| European eel | *Anguilla anguilla* | 615,515 | 343,020 | | 0.041 | 24,990 | | 11,063 | 38,916 |
| European flounder | *Platichthys flesus* | 597,160 | 558,014 | | 0.104 | 62,024 | | 4,066 | 119,982 |
| European pilchard | *Sardina pilchardus* | 62,134,439 | 2,634,731 | | 0.140 | 8,724,513 | | 8,354,561 | 9,094,465 |
| European plaice | *Pleuronectes platessa* | 4,123 | 3,007 | | Negligible | | | | |
| European seabass | *Dicentrarchus labrax* | 6,531,419 | 4,064,888 | 112,425,198 | 0.099 | 649,622 | 11,181,934 | 245,324 | 1,053,920 |
| Garfish | *Belone belone* | 769,285 | 80,782 | 14,792,789 | 0.173 | 133,421 | 2,565,581 | 119,410 | 147,431 |
| gilthead seabream | *Sparus aurata* | 12,177,915 | 6,505,716 | 124,259,429 | 0.061 | 742,647 | 7,577,725 | 345,908 | 1,139,386 |
| gobies nei | *Gobiidae* | 2,254,728 | 1,226,544 | | 0.257 | 580,293 | | 264,621 | 895,965 |
| gray gurnard | *Eutrigla gurnardus* | 6,429 | 6,114 | | 0 | 0 | | 0 | 0 |
| groupers, seabasses, nei | *Serranidae* | 1,088,132 | 675,840 | 5,917,116 | 0.099 | 108,227 | 588,523 | 41,007 | 175,447 |
| gurnards,searobins nei | *Triglidae* | 11,694,972 | 1,128,523 | | 0 | 0 | | 0 | 0 |
| Mediterranean shore crab | *Carcinus aestuarii* | 39,089 | 5,602 | | 0.027 | 1,068 | 0 | 915 | 1,221 |
| picarels nei | *Spicara spp* | 14,543,792 | 2,332,605 | 1,479,279 | 0.040 | 587,350 | 59,741 | 493,148 | 681,552 |
| poor cod | *Trisopterus minutus* | 4,015,989 | 324,080 | | 0 | 0 | | 0 | 0 |
| porgies, seabreams | *Sparidae* | 18,452 | 13,048 | 144,969,334 | 0.234 | 4,315 | 33,904,852 | 1,264 | 7,367 |
| Pouting | *Trisopterus luscus* | 121,999 | 70,174 | 1,479,279 | 0.181 | 22,115 | 268,148 | 9,394 | 34,835 |
| red mullet | *Mullus barbatus* | 61,239,560 | 6,182,432 | | 0.036 | 2,203,010 | | 1,980,606 | 2,425,415 |
| red porgy | *Pagrus pagrus* | 4,266,635 | 2,160,089 | | 0.171 | 729,470 | | 360,158 | 1,098,783 |
| round sardinella | *Sardinella aurita* | 1,757,707 | 1,522,274 | | 0 | 0 | | 0 | 0 |
| saddled seabream | *Oblada melanura* | 3,648,798 | 3,244,191 | | 0.087 | 317,078 | | 35,160 | 598,996 |
| Salema | *Sarpa salpa* | 1,734,583 | 1,616,959 | | 0.073 | 126,050 | | 8,548 | 243,553 |
| sand steenbras | *Lithognathus mormyrus* | 4,212,103 | 3,066,262 | | 0.092 | 387,042 | | 105,289 | 668,795 |
| Sargo breams nei | *Diplodus spp* | 7,332,731 | 6,398,666 | | 0.095 | 697,802 | | 88,888 | 1,306,716 |
| scorpionfishes nei | *Scorpaenidae* | 14,250,854 | 9,686,110 | 41,419,810 | 0.337 | 4,799,688 | 13,950,192 | 1,537,406 | 8,061,970 |

*Continued*

**Table 2. Continued**

| *Common name* | *Scientific name* | *CFV[b] Med[c] €* | *SD* | *RFV Med €* | *SRI* | *CFV Seagrass €* | *RFV Seagrass €* | *CFV Seagrass low €* | *CFV Seagrass high €* |
|---|---|---|---|---|---|---|---|---|---|
| *Chordata* | | | | | | | | | |
| scorpionfishes rockfishes, nei | *Scorpaena spp* | 187,896 | 5,327 | 17,751,347 | 0.337 | 63,284 | 5,978,654 | 61,489 | 65,078 |
| seabasses nei | *Dicentrarchus spp* | 3,677,300 | 1,186,949 | 112,425,198 | 0.099 | 365,748 | 11,181,933 | 247,693 | 483,803 |
| Anthropoda | | | | | | | | | |
| caramote prawn | *Melicertus kerathurus* | 32,103,164 | 10,760,179 | | 0.20 | 642,063 | | 426,860 | 857,267 |
| common spiny lobster | *Palinurus elephas* | 11,081,150 | 8,070,822 | | Negligible | | | | |
| common prawn | *Palaemon serratus* | 524,877 | 303,831 | | 0.085 | 44,607 | | 18,786 | 70,429 |
| common shrimp | *Crangon crangon* | 7,325,478 | 5,890,433 | | 0.069 | 504,998 | | 98,928 | 911,068 |
| spinous spider crab | *Maja squinado* | 40,785 | 15,972 | | 0.070 | 2,868 | | 1,745 | 3,991 |
| surmullet | *Mullus surmuletus* | 56,871,034 | 7,046,634 | | 0.238 | 13,509,937 | | 11,835,982 | 15,183,893 |
| surmullets nei | *Mullus spp* | 6,970,261 | 787,195 | 32,544,136 | 0.238 | 1,655,813 | 7,730,987 | 1,468,812 | 1,842,814 |
| white seabream | *Diplodus sargus* | 1,596,195 | 638,931 | | 0.095 | 151,898 | | 91,096 | 212,700 |
| whiting | *Merlangius merlangus* | 4,084,420 | 624,845 | | 0.114 | 464,883 | | 393,764 | 536,002 |
| wrasses,hogfishes nei | *Labridae* | 19,241 | 2,275 | 5,917,116 | 0.111 | 2,134 | 656,415 | 1,882 | 2,387 |
| Mollusca | | | | | | | | | |
| common cuttlefish | *Sepia officinalis* | 4,653,712 | 3,932,323 | | 0.221 | 1,029,397 | | 159,571 | 1,899,224 |
| common octopus | *Octopus vulgaris* | 20,831,324 | 18,341,194 | | 0.135 | 2,811,764 | | | |
| cuttlefish nei | *Sepiidae, Sepiolidae* | 85,116,576 | 8,449,894 | 54,733,320 | 0.221 | 18,827,720 | 12,106,967 | 16,958,610 | 20,696,830 |
| great Atlantic scallop | *Pecten maximus* | 5,938 | 0 | 0 | Negligible | | | | |
| octopuses nei | *Octopodidae* | 22,504,114 | 2,664,100 | 10,354,952 | 0.135 | 3,037,554 | 1,397,688 | 2,677,960 | 3,397,148 |
| scallops nei | *Pectinidae* | 982,259 | 974,137 | | 0 | 0 | | 0 | 0 |
| Total | | 681,203,166 | | 723,367,390 | | 77,720,132 | 112,615,167 | 58,349,734 | 91,467,006 |

[a]*Abbreviations: CFV, commercial fisheries value; RFV, recreational fisheries value; Med, Mediterranean; nei, not elsewhere identified in Food and Agriculture Organization landings.*
[b]*Averaged over 3 years (2006–2008) and the ranges of these and the corresponding CFV seagrass are presented.*
[c]*The 2006–2008 average value of landings.*

that the economic importance of seagrasses would be overestimated 10-fold. Targeted studies such as ours are not reductionist and do not detract from the broader socioecological importance of seagrass protection. Rather, they provide realistic parameters and a strong socioeconomic argument for fisheries managers to influence seagrass protection strategies.

In addition to functioning as important nursery and foraging habitat for fish, shellfish (Jackson et al. 2001), and wildfowl (Ganter 2000), seagrasses are also thought to oxygenate and stabilize sediments, providing shoreline stabilization and erosion protection (Koch et al. 2009) and increasing potential economic value in terms of shoreline and property protection. Seagrasses are also considered a foundation species because they provide habitat and enhance ecosystem biodiversity and are carbon sequestration and nutrient cycling hubs (Kennedy et al. 2010). Habitat use represents only part of the secondary production importance of seagrasses; there is strong evidence that seagrass meadows have an important role in coastal primary production, for example. Rates of productivity for seagrass alone are large: *Posidonia oceanica* can fix 550–1000 g C $m^{-2} \cdot yr^{-1}$ (Mazzella et al. 1992). Cebrián et al. (1997) examined the fate of leaf and blade production of 4 Mediterranean seagrass species and found that >80% of seagrass production was consumed by herbivores or decomposed by detritivores, which may enhance food production for fishes.

The data we present here should be applied conservatively in terms of how commercial and recreational fisheries may be affected by habitat loss, most crucially because the study does not account for other nonlinearities in the provision of habitat for fishery species (i.e., species of seagrass, location, and the implications of degradation [Duarte 2000]). Variability in how commercial CFV and economic expenditure are spatially apportioned to seagrass meadows would not be surprising because biological resources and ecological services provided by seagrasses are based on plant physical structure and the large meadows they form (Boström et al. 2006).

The role of seagrass meadows as a nursery ground for many marine species, including those of commercial and recreational value, is well documented (Jackson et al. 2001; Heck et al. 2003), but evidence of this nursery function is often confined to studies which identify high juvenile density (Valle et al. 1999; Guidetti 2000). The question is whether seagrass meadows merely concentrate juveniles or whether the residents actually gain a selective advantage over individuals inhabiting other habitats (Beck et al. 2001). If the juveniles in seagrass meadows (or other habitats) fail to reach maturity, then these areas do not function as productive nurseries. Lacking habitat-specific survival data, we weighted habitats equally; a species found in many different habitats will have a lower SRI. Seagrasses may improve survival by providing shelter and food, however. They may also promote planktonic larval settlement and, for those species that do not have a pelagic larval phase, may act directly as spawning areas (e.g., cuttlefish [*Sepia officinalis*] [Ezzedine-Najai et al. 1997]). It is therefore likely that any decline in seagrass quality or extent will affect the entire CFV Mediterranean (€681million) and RFV Mediterranean (€723 million). Consequently, due to the nature of fisheries in the Mediterranean, where inshore areas support a large population of artisanal and subsistence fishers, any socioeconomic impacts resulting from a decline in seagrass could have a disproportionate effect on small to medium size fishing operations that are the cultural backbone of many Mediterranean coastal communities. It is imperative that artisanal and subsistence fishers be included and properly accounted for in seagrass protection strategies (e.g., Marine Protected Areas, MPAs) (Rodriguez-Rodriguez 2014).

While limits on fishing with mobile gears exist in the Mediterranean (Vlachopoulou et al. 2013), in some areas certain species of fish and shellfish are still directly harvested from seagrass (Gonzalez-Correa et al. 2005; Cardona et al. 2007), and we incorporated these species in our study. These extractions can have negative impacts on the seagrass itself (Ardizzone et al. 2000) and nontarget species, including juvenile fishery species for which seagrass is nursery habitat, and therefore compromise future value to fisheries (Rees et al. 2010). In some areas where conservation of the seagrass meadows has resulted in shifts in fishing practice, fishing the edges of the meadows with static gear has become increasingly productive (Savage et al. 2012). Knowledge of the links between seagrass meadows and commercial CFVs and the expenditure of recreational fishers may, however, not affect wider policies (e.g., on climate change), but this type of information may be useful for local fisheries managers where trawling in seagrasses remains a direct impact (Vlachopoulou et al. 2013). While initial ecosystem service valuations have been carried out for Mediterranean *Posidonia oceanica* meadows in terms of carbon sequestration (Luisetti et al. 2013), attempts to quantify other ecosystem services are still needed in order to estimate the full economic value of seagrass meadows. Such information has significant implications for management and policy implementation, particularly in regards to the EU Marine Strategy Framework Directive, which does not link descriptors of good environmental status that are relevant to seagrass and seagrass ecosystem services. For example, descriptor 1 (biological diversity) will examine the state of seagrass meadows, but seagrass meadows are not considered in descriptor 3, population of commercial fish and shellfish, or descriptor 4, elements of marine food webs.

The MSFD explicitly requires an economic and sociological analysis of the use of coastal and marine waters to determine the "cost of degradation" of the marine environment. We have demonstrated that there is a clear cost of degradation associated with ineffective management of seagrass meadows and that there is a potentially disproportionate impact of such management on small-scale artisanal fisheries that operate in inshore waters. European Union policy to manage both fisheries and seagrass meadows should align to take into account the socioeconomic implications of seagrass degradation on both recreational and commercial fisheries.

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## Supporting Information

Life history data collated for the calculation of the SRI, including source information (Appendix S1), and a full valuation table for recreational fishing in the Mediterranean, including source information (Appendix S2), are available online. The authors are solely responsible for their content and functionality of these materials. Queries (other than the absence of material) should be directed to the corresponding author.

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