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Invited feature-FLA

Quantitative estimate of commercial fish enhancement by seagrass habitat in southern Australia

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ABSTRACT

Seagrass provides many ecosystem services that are of considerable value to humans, including the provision of nursery habitat for commercial fish stock. Yet few studies have sought to quantify these benefits. As seagrass habitat continues to suffer a high rate of loss globally and with the growing emphasis on compensatory restoration, valuation of the ecosystem services associated with seagrass habitat is increasingly important. We undertook a meta-analysis of juvenile fish abundance at seagrass and control sites to derive a quantitative estimate of the enhancement of juvenile fish by seagrass habitats in southern Australia. Thirteen fish of commercial importance were identified as being recruitment enhanced in seagrass habitat, twelve of which were associated with sufficient life history data to allow for estimation of total biomass enhancement. We applied von Bertalanffy growth models and species-specific mortality rates to the determined values of juvenile enhancement to estimate the contribution of seagrass to commercial fish biomass. The identified species were enhanced in seagrass by 0.98 kg m^{-2} y^{-1}, equivalent to ~$A230,000 ha^{-1} y^{-1}. These values represent the stock enhancement where all fish species are present, as opposed to realized catches. Having accounted for the time lag between fish recruiting to a seagrass site and entering the fishery and for a 3% annual discount rate, we find that seagrass restoration efforts costing $A10,000 ha^{-1} have a potential payback time of less than five years, and that restoration costing $A629,000 ha^{-1} can be justified on the basis of enhanced commercial fish recruitment where these twelve fish species are present.

Editor's note

The Invited Feature Article in this issue highlights issues that are becoming increasingly important in conservation and management of coastal ecosystems. Many key coastal ecosystems are increasingly threatened: seagrass meadows are one such endangered habitat. The services furnished by such habitats need to be more widely appreciated by various sectors of coastal stakeholders. Further, we also could better assess the human-oriented benefits of certain services that the threatened ecosystems. While it is true that it is not always advantageous to convert environmental benefits into currency as not all benefits can be monetized, it does make sense to convey to stakeholders the socio-economic advantages of conservation of coastal environments. The Invited Feature Article in this issue by Blandon and zu Ermgassen includes contributions to all these aspects.

1. Introduction

Seagrass habitats are widely recognized to be important nursery grounds for fish, with juvenile fish routinely being found at higher densities in seagrass beds than in nearby unvegetated substrates (Heck et al., 2003). In addition to providing fish habitat, seagrasses play important roles in nutrient recycling, sediment stabilization, oxygenation of surrounding water, reduction of wave impacts and carbon sequestration (Short et al., 2011). Yet seagrass meadows are under increasing pressure from human development. An estimated third of seagrass meadows have already been lost globally, with losses occurring at a rate of 110 km^2 yr^{-1} since 1980 (Waycott et al., 2009). The decline in seagrass habitat can be attributed to numerous drivers, including destructive fishing practices, coastal engineering, cyclones, and anthropogenicaly driven water quality degradation (Orth et al., 2006). Although these pressures are being addressed in some locations (Greening and Janicki, 2006), the global rate of decline in seagrass is still believed to be accelerating (Waycott et al., 2009). The value of the associated ecosystem services is often a major impetus for the protection and restoration of threatened habitats, yet few attempts have been made to value the benefits derived from seagrass habitats (Barbier et al., 2011).

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Seagrasses in southern Australia are no exception to the global trend; six of the ten major areas of seagrass loss in Australia are located in the southern states (Kirkman, 1997), with near ubiquitous declines in the region (Waycott et al., 2009). These declines are increasingly being countered by stronger regulation and increased restoration efforts (Seddon, 2004; Western Australia Environmental Protection Authority, 2004). Following limited early success in restoration efforts, recent developments have resulted in improved site selection and more appropriate transplanting methodologies for the slow growing seagrass species that are under threat in southern Australia (Seddon, 2004). Restoration efforts are, however, extremely expensive, ranging in cost from \$A10,000 ha^{-1} for *Amphibolis* species, which may take from seed, to >\$A1,308,284 ha^{-1} for species that require transplanting of plugs (Ganassin and Gibbs, 2008; 1 \$A ≈ 0.9 \$US). Given the costs involved in restoring seagrass beds, valuation of the potential benefits arising from seagrass restoration efforts can play an important role in decision-making and in attracting necessary investment.

While numerous researchers have explored the evidence that seagrass habitats benefit commercially important species and may increase fisheries yields (Coles et al., 1993; Jenkins et al., 1997; McArthur et al., 2003), quantitative assessment of the benefits is challenging and has rarely been attempted (Barbier et al., 2011). In southern Australia, only one previous study has sought to determine the value of commercial fisheries enhancement resulting from seagrass habitats. McArthur and Boland (2006) applied a model based on a predetermined index of seagrass residency (Scott et al., 2000), to catch per unit effort data of seven commercially important species in GARFIS fishing blocks with a known seagrass extent. The seagrass residency index used was based on expert opinion of the relative duration of each fish species' life history stage in seagrass (Scott et al., 2000). This methodology identified numerous species, in particular King George whiting (*Sillaginodes punctata* [Cuvier, 1829]), calamari (*Sepioteuthis australis* Quoy and Gaimard, 1832) and garfish (*Hyporhamphus melanochir* [Valenciennes, 1847]), whose abundance was strongly dependent on seagrass. They also found evidence of a short term decline in fish abundance corresponding to an episode of seagrass loss in one block (McArthur and Boland, 2006). The methodology was, however, limited as to the extent to which it was able to verify that seagrass declines were responsible for the fisheries declines (McArthur and Boland, 2006). The observed declines affected a region larger than the examined fishing block alone, suggesting that other external factors, for example related to hydrodynamic conditions, could have played a role in the observed fisheries declines (McArthur and Boland, 2006). Furthermore, as the seagrass residency index was developed on the basis of expert opinion as opposed to quantitative data, its application in a quantitative model should be viewed with caution. Nevertheless, this model represents the best current estimate of the commercial fisheries value of southern Australian seagrasses.

An alternative methodology, developed by Peterson et al. (2003) for estimating the fisheries value of oyster reef restoration in the south eastern United States, combines quantitative abundance data with established growth and mortality relationships to estimate the fish biomass enhancement for species that are enhanced at the juvenile stage by the presence of the habitat. A similar approach was used by Watson et al. (1993) to estimate the value of enhancement by seagrass to the penaeid shrimp fishery in northern Queensland and also by Powers et al. (2003) to estimate the value of artificial reefs in the US. The method is based on the assumption that, where nursery habitats have been severely reduced in extent, habitat can limit fish recruitment. In the current study, we apply this approach to seagrass habitat in southern Australia to derive an estimate of the value of enhancement for commercially important fish species per hectare of seagrass, as well as an estimate of the payback time of seagrass restoration efforts based on this ecosystem service.

2. Material and methods

2.1. Data collection

A review of the literature was undertaken in January 2012 using the Web of Knowledge Service with search terms "fish", "seagrass" and "Australia" to identify studies that fulfilled the following criteria: 1) conducted in southern Australia (defined as South Australia, New South Wales, Victoria and Western Australia below 24° latitude), 2) included data on individual fish species and their density in both seagrass and an unvegetated control and 3) used sampling techniques that are strongly biased towards the sampling of young of year fish (fine mesh seine nets and pop nets). Studies based on artificial beds created to mimic seagrass were excluded.

The search identified more than 400 articles, of which eleven fulfilled the required criteria (Table 1). Care was taken to ensure that studies included in the meta-analysis were independent of one another; while some locations are represented in multiple studies,

Table 1
Synopsis of the eleven studies from which data were extracted for inclusion in the meta-analysis.

Reference	State	Location	Seagrass species	Sampling method	Mesh size
Humphries et al., 1992	Western Australia	Wilson Inlet	*Ruppia megacarpa* Mason 1967	Seine	Wings: 6 m of 9 mm mesh, 4 m of 6 mm mesh Bunt: 6 mm mesh
Connolly 1994a	South Australia	Barker Inlet – Port River	*Zostera muelleri* Irmisch ex Ascherson 1867a	Seine	Large net: 6 mm Small net: 1.4 mm
Connolly 1994b	South Australia	Barker Inlet – Port River	*Z. muelleri*	Pop net	Not given
Edgar and Shaw 1995	Victoria	Western Port	*Heterozostera tasmanica* (Martens ex Ascherson) den Hartog 1970, *Z. muelleri*	Seine	1 mm
Gray et al., 1996	New South Wales	North East New South Wales	*Zostera capricorni* Ascherson 1867a	Seine	6 mm
Jenkins et al., 1997	Victoria	Port Phillip Bay & Corner Inlet	*H. tasmanica*, *Z. muelleri*	Seine	1 mm
Jenkins and Wheatley 1998	Victoria	Port Phillip Bay	*H. tasmanica*	Seine	1 mm
Hindell et al., 2000	Victoria	Port Phillip Bay	*H. tasmanica*	Seine	1 mm
Griffiths 2001	New South Wales	Shellharbour Lagoon	*Z. capricorni*	Seine	6 mm
Bloomfield and Gillanders 2005	South Australia	Barker Inlet – Port River	*Z. muelleri*	Seine and pop net	Seine: 1 mm Pop net: 1 mm
Smith et al., 2008	Victoria	Port Phillip Bay	*Heterozostera nigricaulis* J. Kuo 2005	Push net	1 mm

these studies did not include data sampled during the same year. Fig. 1 shows the geographical location of the studies included in the meta-analysis.

2.2. Synopsis of studies

Humphries et al. (1992) used a seine net to sample on and off seagrass in the lower region of the Wilson Inlet. The seine net swept an area of 116 m^2 for each sample, with sampling occurring monthly from March 1988 to February 1989. Connolly (1994a) used two seine nets to sample unvegetated areas and eelgrass areas in the Barker Inlet – Port River region and reported the data for both seine nets combined. The smaller net with mesh size 1.4 mm sampled an area of 84 m^2 and the larger net of mesh size 6 mm sampled an area of 347 m^2 for each sample. Sampling occurred in 5 periods from January 1990 to February 1991. Connolly (1994b) sampled in the same region in September 1991 using a 25 m^2 pop net to compare fish density on naturally vegetated and unvegetated sites. Bloomfield and Gillanders (2005) also sampled seagrass and unvegetated areas in the Barker Inlet – Port River region using a seine net and pop net. Both sampling methods swept an area of 9 m^2, with samples taken on four separate occasions in winter of 2002 and three times in summer of 2002. Jenkins et al. (1997) used a seine net to sample seagrass and unvegetated areas at four sites in Port Phillip Bay and three sites in Corner Inlet, sweeping an area of 150 m^2 in each sample. Sampling in Port Phillip Bay occurred monthly and in Corner Inlet it occurred bimonthly for a year over 1989 and 1990. Jenkins and Wheatley (1998) used a seine net to sample three sites in Port Phillip Bay, sweeping an area of 1250 m^2 in each sample. Samples were taken each month from October 1993 to March 1994. Hindell et al. (2000) used enclosure and exclusion cages to manipulate the abundance of predatory fish on and off seagrass in Port Phillip Bay. The control treatments for each experiment provided us with the data required for this study. The areas were sampled using a seine net sweeping an area of 16 m^2 for each sample. The samples were taken between November 1998 and January 1999. Smith et al. (2008) sampled on and off seagrass with a push net at three sites in Port Phillip Bay. The net swept an area of 5 m^2 for each sample. Sampling was undertaken between October 2005 and January 2006. Edgar and Shaw (1995) sampled unvegetated and vegetated areas in four sites in Western Port, sweeping an area of 77 m^2 for each sample. Sampling occurred every three months in each habitat type at each site over a year from 1989 to 1990. Griffiths (2001) sampled vegetated areas and areas of bare sand using a seine net in Shellharbour Lagoon, sampling an area of 25 m^2 in each sweep. Sampling occurred every 2 months from September 1997 to November 1998. Gray et al. (1996) sampled on and off seagrass using a seine net sweeping 25 m^2, in eight estuaries along the New South Wales coast. Sampling occurred in September and November 1994. A summary of the details of these studies can be found in Table 1.

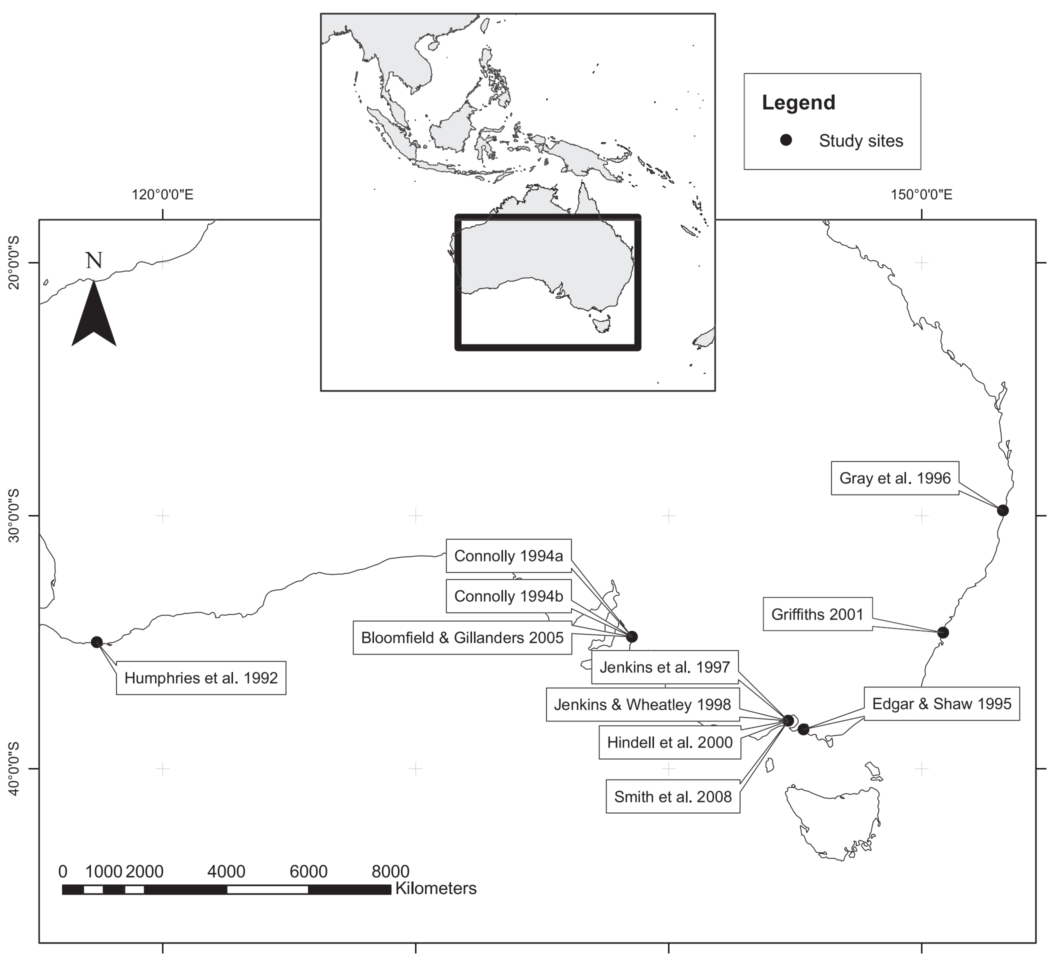

Fig. 1. Map showing the locations of each of the studies included in the meta-analysis.

2.3. Enhancement estimates

This methodology applies growth and mortality to a known age class of fish to derive an estimate of total biomass attributable to enhanced juvenile recruitment. The sampling methods included in the meta-analysis primarily target young of year fish, with the efficiency declining steeply for stronger swimming individuals (Bloomfield and Gillanders, 2005), and we therefore assume that all individuals caught are juveniles. Peterson et al. (2003) made the assumption that all yr 0 fish sampled would survive to their first birthday, however, a more robust and conservative logic is to assume that the yr 0 fish sampled throughout the year, were likely to be 6 months of age on average. We adopt this reasoning in our methodology.

Data were standardized to represent the mean number of individuals per m^2. For the vast majority of studies the densities were calculated using the total abundance for each species divided by the sampling frequency and area. The enhancement of the fish stock by the seagrass was then calculated for each species in each study using the equation: enhancement $= \rho_{\text{seagrass}} - \rho_{\text{unvegetated}}$, where ρ is the density of 0.5 yr old fish in individuals m^{-2}.

Studies included in the meta-analysis varied with regards to the number of independent sampling events representing the mean and therefore represented differing levels of confidence. In order to correct for the number of independent measurements represented by the means from each study, values were weighted by the number of independent samples representing the mean. We defined independent sampling events as samples that were collected in different seasons, or in different bays or estuaries. Samples taken in the winter season, defined as June, July and August, were not included as independent events. As such, only peak seasons of recruitment were given weighting, therefore better representing the reliance of juveniles on the habitat. Some of the studies sampled fewer control sites than unvegetated areas, in which case the smaller number was used to weight the value.

After applying this weighting, we calculated the mean enhancement for each species across the studies as follows: mean enhancement $= \sum_{\text{studies}}$ (enhancement $\times$ number of independent sampling events)/$\sum_{\text{studies}}$ (number of independent sampling events), so as to yield one enhancement value for each species, in individuals m^{-2}. Where a fish species was present in greater abundance in seagrass than the control, and was represented in two or more studies, that species was deemed to be recruitment enhanced by seagrass.

The efficiency of the fish sampling methods represented by the dataset are highly variable, with estimates ranging from 20 to 83% for beach seines depending on species (Jenkins et al., 1997). Previous studies have also illustrated high variability in catch efficiencies within species (Rozas and Minello, 1997). Given the large variability in catch efficiencies estimated we did not apply any correction factor to the data, but rather use the values as a conservative estimate of densities.

2.4. Species selection

As our aim was to determine the value of the seagrass habitat that is derived from recruitment enhancement, we focussed our study on economically important fish species. Fish species of importance were identified using studies by Edgar and Shaw (1995), Gray et al. (1996) and Griffiths (2001), as well as the New South Wales Status of Fisheries Resources 2008–09 (Rowling et al., 2010). The selection criteria for the species to be analysed were: a) economic importance, b) positive enhancement by seagrass and c) represented by two or more studies. Thirteen fish species met these criteria (Table 2).

2.5. Production calculations

The total annual production of each fish species attributable to seagrass was determined following a revised methodology based on that developed by Peterson et al. (2003). The methodology was revised to remove the impact of fishing mortality (F) and hence to provide an estimate of the total potential catch, i.e. the value of all fish added to the system through the addition of seagrass habitat, without making assumptions about the response of the fishing community. A brief summary of the revised methodology is outlined below.

It is assumed that the distribution of age classes follows a stable age distribution, $y = e^{-Mi}$, where y is the proportion of the fish population surviving at age class i, and M is the species specific natural mortality. This equation is used to calculate the proportion of individuals in age class 0.5 surviving to age class i, therefore for each age class, the density enhancement was calculated using $N_i = N_{0.5} \times e^{(-M\times(i-0.5))}$, where N_i is the density enhancement for age class i. $N_{0.5}$ is given by the enhancement value calculated previously, using the equation: enhancement $= \rho_{\text{seagrass}} - \rho_{\text{unvegetated}}$.

For each age class, the length of an average fish was calculated using the von Bertalanffy growth equation and the average weight for each age class was then calculated from the length using the length–weight relationship. The total annual enhancement of a species (g m^{-2}) was calculated by summing the incremental increase in weight for an average fish in each year class, multiplied by the number (density) of fish (N_i) in each age class. The species-specific parameters used in these calculations are given in Table 2.

2.6. Life history parameters

We searched Fishbase (Froese and Pauly, 2012) for species specific growth parameters and estimates of natural mortality. Where species-specific a and b values for the length–weight relationship were not available we used known values from similar species as proxies. Suitable proxy species were either another species in the same Genus or Family for which length–weight relationship data were available, or a physiologically similar species as identified from the FAO literature. Where there were multiple similar species with length-weight data available, FAO literature was also used to identify the most appropriate related species to use as a proxy. A list of the species affected and the proxies used can be found in the Appendix. Where we were unable to determine the age of first harvest (r), we assumed r to be equal to the age of maturity as listed in Fishbase. Natural mortality for *Haletta semifasciata* (Valenciennes, 1840) was derived from the species' maximum age (Hoenig, 1983), as a direct value for M was unavailable. The parameters used to calculate annual production for each species are given in Table 2.

2.7. Economic valuation

The economic enhancement was calculated by multiplying the price of each species by the annual enhancement for age classes $\geq r$, thus providing an estimated value for the additional biomass of fish available to the fishery as a result of additional seagrass habitat. This value represents the enhancement of biomass in the system, not what is actually caught. Fish prices were sought from state authorities including Victoria Department of Primary Industries, Department of Fisheries Western Australia and the South Australian Research and Development Institute.

In calculating the discounted payback period for seagrass restoration efforts, the value of each fish species was accounted for from r years after restoration, thus accounting for the time lag between seagrass creation and fish entering the fishery as a result

Table 2
The thirteen fish identified as enhanced by seagrass with the parameters used for production calculation. All values are from Fishbase (Froese and Pauly, 2012), except for: New South Wales Status of Fisheries Resources 2008–09 (Rowling et al., 2010); $^{\alpha}$Hoenig 1983; †Gaughan et al., 1996; ‡Fowler et al., 2008; ☐Fowler et al. 2011.

Species	Common name	Max. age	M (natural mortality)	L_{∞} (asymptotic max. length)	K (Brody growth coefficient)	t_0 (age at 0 length)	a	b	r (age of first harvest)
Engraulis australis	Australian anchovy	7	0.82	12.1	0.39	−0.54	0.007615	3.04845	1*
Girella tricuspidata	Luderick	16	0.31	47.4	0.18	−0.83	0.016	3.021	5*
Haletta semifasciata	Blue weed whiting	9	0.49$^{\alpha}$	30.4	0.33	−0.5	0.0034	3.3685	2
Hyperlophus vittatus	Sandy sprat	3	1.5†	10.7	1.08	−0.19	0.0019	3.579	1*
Hyporhamphus melanochir	Southern sea garfish	5	0.88	41.4	0.61	−0.24	0.000005‡	2.99‡	1
Liza argentea	Flat-tail mullet	12	0.35	46.9	0.23	−0.65	0.0089	3.129	6*
Meuschenia freycineti	Six-spined leatherjacket	20	0.23	62.2	0.14	−1	0.0556	2.887	5
Meuschenia trachylepis	Filefish	14	0.38	41.7	0.2	−0.77	0.0556	2.887	4
Mugil cephalus	Flathead grey mullet	8	0.57	51	0.34	−0.42	0.0162	3.009	4*
Neosebastes scorpaenoides	Ruddy gurnard perch	–	0.4	41.7	–	–	0.0301	2.999667	–
Pelates sexlineatus	Six-lined trumpeter	5	1.37	15.9	0.63	−0.31	0.0134	2.958	1
Rhabdosargus sarba	Tarwhine	26	0.25	82.6	0.11	−1.19	0.0153	2.967	2*
Sillaginodes punctata	King George whiting	18☐	0.74	53.2	0.47	−0.3	0.003	3.2	3☐

of their enhanced recruitment to the habitat. While some authors support the use of low or zero discount rates, especially when analysing problems with long time horizons (Sumaila et al., 2000), we apply a 3% annual discount rate, the discount rate commonly used for habitat equivalency analysis for natural resource damage assessment in the US (Viehman et al., 2009). The time lag between seagrass creation and fish entering the fishery was also applied when calculating the maximum per ha expenditure for restoration that is justified on the basis of commercial fisheries alone.

3. Results

Thirteen commercially important fish species were found to be recruitment enhanced by seagrass habitat (Table 2). One species, the ruddy gurnard perch (*Neosebastes scorpaenoides* Guichenot, 1867), lacked sufficient information on its life history to carry out the calculations to quantify its enhancement beyond initial settlement and was therefore not included in the quantification of enhancement of biomass. Seagrass enhanced the twelve remaining fish species by a total of 980 g m^{-2} y^{-1}, such that each ha of seagrass restored in southern Australia may enhance commercial fish species by a total of 9.8 tonnes yr^{-1} (Table 3). Grey mullet (*Mugil cephalus* Linnaeus, 1758) was the single most significantly enhanced species, with an additional 9260 individuals ha^{-1} recruiting a year, resulting in an enhancement of 3500 kg ha^{-1} y^{-1} across all age classes. This was closely followed by the six-lined trumpeter (*Pelates sexlineatus* [Quoy and Gaimard, 1825]), for which recruitment was enhanced by an additional 8020 individuals ha^{-1} y^{-1} across all age classes. A full overview of recruitment enhancement by species can be found in Table 3.

Our estimate of the economic enhancement of fish in southern Australia by seagrass includes only the biomass of fish from the age at which they enter the fishery (r). The economic value of the enhancement is estimated at $A230,000 ha^{-1} yr^{-1} once all fish are fully recruited to the habitat (>26 yrs after restoration). The species specific economic enhancement and biomass resulting from recruitment enhancement in seagrass habitats are given in Table 3, and can be used to make alternative site specific estimates where species presence or absence is known. Tarwhine (*Rhabdosargus sarba* [Forsskål 1775]) accounts for the vast majority of the total value, with more than $A206,000 ha^{-1} yr^{-1} attributed to this one species, whereas the Southern sea garfish (*Hyporhamphus melanochir*) only contributes $A6.00 × 10^{-2} ha^{-1} (Table 3). Most other species are economically enhanced by values between $A800-7000 ha^{-1} (Table 3).

Our assessment of the cost effectiveness of restoration on the basis of commercial fish recruitment enhancement suggested that a low cost seeding approach to restoration, costing $A10,000 ha^{-1} would have a payback time of less than 5 years (at a 3% discount rate). The highest cited cost of $A1,308,284 ha^{-1}, however, had an infinite payback time if only these twelve fish species are taken into account. Our analysis found that a maximum cost of $A629,000 ha^{-1} can be justified on the basis of the recruitment enhancement of these twelve commercial fish species alone. However, since Tarwhine provides a large proportion of the economic enhancement, if it is removed from the calculation the

Table 3
Mean juvenile enhancement (individuals m^{-2}), total annual enhancement (across all age classes; g m^{-2}), and annual economic enhancement across all age classes > r ($A) for the twelve species included in the analysis. Prices found in a study by McArthur and Boland (2006)‡ were used for comparison between the studies only, the results of which are given in brackets.

Species	Common name	Mean enhancement (individuals m^{-2})	Total annual enhancement (g m^{-2})	Annual enhancement of year classes > r (g m^{-2})	Price ($A kg^{-1})	Economic enhancement ($A ha^{-1})
Engraulis australis	Australian anchovy	0.010	0.02	0.012	1.58	1.91
Girella tricuspidata	Luderick	0.052	15.54	4.333	2.00	866.59
Haletta semifasciata	Blue weed whiting	0.012	0.49	0.702	4.33	15.93
Hyperlophus vittatus	Sandy sprat	0.126	0.294	0.053	4.03	21.19
Hyporhamphus melanochir	Southern sea garfish	0.002	0.02 × 10^{-2}	0.008 × 10^{-2}	7.27 (5.76‡)	6.00 × 10^{-2} (3.93)
Liza argentea	Flat-tail mullet	0.122	33.06	4.026	3.33	1340.71
Meuschenia freycineti	Six-spined leatherjacket	0.014	20.76	8.674	5.74	4978.66
Meuschenia trachylepis	Yellowfin leatherjacket	0.095	38.86	11.152	5.74	6401.09
Mugil cephalus	Flathead grey mullet	0.926	350.16	34.979	1.78	6226.23
Pelates sexlineatus	Six-lined trumpeter	0.802	6.46	1.675	1.53	256.31
Rhabdosargus sarba	Tarwhine	0.618	514.36	416.912	4.95	206,371.22
Sillaginodes punctata	King George whiting	0.026	4.64	0.507	16.24 (12.21‡)	824.07 (6.01)

payback period rises to 11 years for the same cost of $A10,000 ha^{-1} and the maximum justifiable cost falls to $A58,000.

4. Discussion

Our analysis confirms that seagrass habitat in southern Australia is of great importance as a nursery to a range of commercially important fish species and that seagrass restoration has the potential to yield valuable benefits to fisheries. While the role of seagrass in enhancing fish biomass has been quantified for a small subset of commercially important species (McArthur and Boland, 2006), our study provides the most complete current assessment of the value of seagrass as a nursery ground for commercially important species in southern Australia. While our determined values of biomass are suggestive of very substantial economic enhancement through seagrass restoration (with a potential total of $A230,000 ha^{-1} yr^{-1}), we believe these estimates to be conservative. Firstly, our criteria for species to be included in this analysis were rigorous and as a result a number of potential species were excluded from the economic analysis due to insufficient data. Secondly, seine nets, which were used in the vast majority of sampling events included in our meta-analysis (Table 1), have an efficiency range of 20–83% (Jenkins et al., 1997), indicating that our results can be viewed as conservative estimates of absolute abundance.

That many fish species are biomass enhanced by seagrass is unsurprising, as this conclusion is readily supported in the wider literature for a number of the species we identified. For example, King George whiting is consistently found to be at higher densities on seagrass beds compared to unvegetated areas (Hyndes et al., 1996; Jenkins and Sutherland, 1997; Morris and Ball, 2006), possibly due to the higher levels of food availability on seagrass for juvenile whitings (Jenkins and Sutherland, 1997). Similarly, Tarwhine, Luderick (*Girella tricuspidata* [Quoy and Gaimard, 1824]) and the filefish *Meuschenia trachylepis* (Günther 1870) have been found in higher abundances at seagrass than non-seagrass sites in southern Australia as a result of high recruitment to these sites each summer (McNeill et al., 1992; Middleton et al., 1984; cited by; Bell and Pollard, 1989). Furthermore, the family Monacanthidae to which the *Meuschenia* species belong, are often among the dominant families in seagrass habitat in the region (Bell and Pollard, 1989).

While there is general agreement between our study and earlier efforts to determine fish enhancement by seagrasses, our results differ as to the degree of enhancement suggested. For example, while we find Southern sea garfish to be only marginally enhanced by seagrass, with just ~20 more individuals ha^{-1} yr^{-1} recruiting to seagrass compared to unstructured bottoms, Scott et al. (2000) determined Southern sea garfish to have a seagrass residency index of 0.98 (with 1.0 indicating complete dependence on seagrass). We re-examined our data and confirmed that all of the studies we identified universally agreed that garfish was only weakly enhanced. This mismatch in the apparent dependency of this species on seagrass habitats should ideally be addressed through further research.

When our results are compared to those from another study in the region that sought to quantify the fisheries contribution of seagrass habitats (McArthur and Boland, 2006), a number of differences as to the enhancement value of certain species emerge. The methodologies in these two studies are, however, also strikingly different. Furthermore the studies had different aims as to the attribute of the fishery they sought to value. McArthur and Boland used catch per unit effort data and the seagrass residency index based on expert opinion developed in Scott et al. (2000), to estimate the impact of a seagrass loss event on actual catch value, whereas we have sought to determine the value of all fish that would be additionally recruited to the system in the presence of seagrass. Nevertheless, some of the differences between the results of these two studies are worthy of note. While 6 of the 7 species analysed in the McArthur and Boland study were present in the original data for our meta-analysis (the exception being the southern calamari, *Sepioteuthis australis* Quoy and Gaimard, 1832), only two fulfilled the criteria for inclusion in our analysis. The remaining 4 species were excluded because they were not represented in a sufficient number of studies. When the results for the two relevant species, the garfish and the King George whiting, were compared we found a very low enhancement value for garfish (0.02×10^{-5} kg ha^{-1} and $A0.06 ha^{-1}) relative to the McArthur and Boland study (0.68 kg ha^{-1} and $A3.9 ha^{-1}), and a high enhancement value for King George whiting (4.64×10^{-3} kg ha^{-1} and $A824 ha^{-1}) relative to the McArthur and Boland study (0.49 kg ha^{-1} and $A6 ha^{-1}). The prices represented in the McArthur and Boland study were used for these comparisons. The stark contrast in the value of the southern garfish enhancement can most likely be explained by the use of the high seagrass residency index derived for garfish by Scott et al. (2000) in the McArthur and Boland model. The discrepancy between the values determined for the King George whiting on the other hand, are likely due to numerous factors including differences in the geographical focus of the studies, in the seagrass species sampled, in sampling methodologies included, and differences in the focus of the two studies, with McArthur and Boland accounting only for fish caught. The explanation that likely accounts for much of the difference between the two studies is that seagrass in south-west Australia where the McArthur and Boland study was focussed, consists mainly of *Posidonia australis* (Hook.f.) and P. *sinuosa* (Cambridge and J. Kuo), which whiting rarely colonise (Hyndes et al., 1996), whereas our study included south-east Australia where large numbers of juvenile whiting are consistently found in *Zostera muelleri* and *Heterozostera tasmanica* (Hyndes et al., 1996).

Fish species distribution is often spatially variable by seagrass species and geographic location, this is especially likely the case in Australia, which is home to the most species rich and extensive seagrass habitat globally (Spalding et al., 2003). Such differences may be an important component in explaining fish community composition (MacArthur and Hyndes, 2001; Rotherham and West, 2002; Hyndes et al., 2003). For example, studies have consistently found large differences in fish community composition between *Zostera* and *Posidonia* seagrass beds (Bell and Westoby, 1986a; cited by; Bell and Pollard, 1989; Middleton et al., 1984; Young, 1981). While our results benefit from drawing on a large number of studies over a broad area to provide a general overview of fisheries benefits from seagrass habitat, the summary numbers do not account for such differences between sites. Our results can, however, be made more site specific by only including fish species that are present at a given site. For example, Tarwhine makes up the majority of the economic enhancement, contributing $A206,000 ha^{-1} of the total $A230,000 ha^{-1}. If this species is not present at the site of interest, our summary analysis would provide an overestimation of economic enhancement. Our species specific numbers (Table 3) can, however, be combined to provide more site specific estimates of fish enhancement, which can be used by decision makers to more accurately portray the relevant economic enhancement at a given site.

Numerous landscape factors not included in our study may further impact the abundance of fish species. For example, some studies suggest that the density of individuals found in smaller seagrass patches is higher than larger patches (McNeill and Fairweather, 1993) and that some species are naturally more abundant on the seagrass-sand boundary than in the interior of the habitat (Tanner, 2005). However, a review of the published literature suggests that seagrass patch size does not have any consistent impact on the resident fauna (Bell et al., 2001). We were unable to explore the impacts of patch size or relative habitat extent on fish abundance in this study as the studies on which our analysis is based did not all report such figures, however, understanding edge effects is an important next step in more widely applying ecosystem service valuations across sites and it should be acknowledged that our current estimates do not account for intra-habitat variability.

Seagrass meadows are under increasing pressure from a variety of sources: it has been estimated by Waycott et al. (2009) that the global rate of decline of seagrass area has been 7% yr^{-1} since 1990. Seagrass restoration has become an increasingly viable option to counter these losses as techniques improve (Wear, 2006). The cost of seagrass restoration efforts is, however, highly variable (spanning from \$A10,000-\$A1,308,284 ha^{-1}), depending in part on whether the species will take from seed, or whether plugs of seagrass have to be transplanted (Ganassin and Gibbs, 2008). Previous studies have suggested that seagrass restoration may be too expensive to be economically defensible (Wear, 2006; Paling et al., 2009), however, we show that some restoration programmes may in fact be economically justified with payback times as low as 5 years, given a 3% discount rate and based on recruitment enhancement of just a handful of commercially valuable fish species. Furthermore we illustrate that an expenditure of up to \$A629,000 ha^{-1} may be justified on the basis of commercial fish recruitment enhancement alone.

With the improvements in restoration techniques leading to lower costs and ongoing efforts to better parameterize the other ecosystem services associated with seagrasses (Wear, 2006), the outlook for seagrass restoration is improving. It is nevertheless important to keep in mind the influence of species identity in this calculation, as the presence or absence of a handful of fish species can have a dramatic impact on the potential value of seagrass habitat at a given site. Our results can be adapted by excluding species not present at a site, to provide a more accurate site specific estimate of potential fisheries benefits. As such our results have a broad application in assisting decision makers, practitioners and managers in determining the potential cost effectiveness of proposed restoration efforts.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at http://dx.doi.org/10.1016/j.ecss.2014.01.009

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