Seagrass Ecosystem Interactions with Social and Economic Systems

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Introduction

Seagrass meadows (in the UK dominated by one species, *Zostera marina*) are in the conservation spotlight: identified as Features of Conservation Importance (FOCI) for the proposed Marine Conservation Zones under the Marine and Coastal Access Act; as Biodiversity Action Plan Habitats (underpinned by the Convention of Conservation of Biological Diversity, Rio Earth Summit 1992); as a threatened and declining habitat under OSPAR and as a sub-feature of subtidal sandbanks for the designation of Special Areas of Conservation under the European Habitats Directive. There are multiple justifications for protecting these habitats from the numerous pressures on them. Seagrass beds are stated as providing a number of ecosystem services from provisioning, regulating and cultural categories (Rönnbäck et al. 2007). They function as important nursery and foraging habitat for fish, shellfish (Jackson et al. 2001a) and wildfowl (Ganter 2000b). They are also thought to oxygenate and stabilise sediments, providing shoreline stabilisation and protection from erosion (Koch et al. 2009a), and are natural hotspots for carbon sequestration and nutrient cycling (Kennedy et al. 2010). Finally, they are considered a foundation species, i.e. a species that provides habitat and enhances ecosystem biodiversity, is home to intrinsically valuable species such as the seahorse and is an important bio-indicator of system health.

The interactions between seagrass ecosystems and societal and economic systems are numerous and complex and activities on and around the seagrass put pressure on the health and survival of this habitat and compromise the provision of beneficial services. Recent reports on the status of seagrass habitats in the UK suggest that whilst activities which decrease water clarity or quality (for example eutrophication, aquaculture, coastal development, dredging and spoil disposal) negatively impact the health or productivity of seagrass, improvements in water quality through improved sewerage treatment and national regulations resulting from Urban Waste Water Treatment Directive and Water Framework Directive have started to negate these pressures. However, continued direct physical pressures (e.g. boat anchoring, propeller scarring, dredging and destructive fishing methods such as beam trawling) on seagrass beds are increasingly resulting in losses and fragmentation of many beds. Seagrass growth, distribution and function are regulated by a number of physical (light, hydrology, geology and temperature), chemical (salinity, oxygen, nutrients) and biological (competition, disease, anthropogenic) factors. Climate change and ocean acidification have the potential to influence each of these factors. Different pressures change different aspects of the seagrass plant, patch and landscape configuration, with knock on effects on the provision of different ecosystem services.

The aims of this scoping report are to present the first steps in an ecosystem interactions assessment for seagrass ecological, social and economic systems. The report presents our approaches and initial results and states any gaps and key research questions which could be addressed in a continuation of this work. The work is split into three objectives. In objective one we use the DPSWR framework to define and identify key interactions between seagrass ecosystems and socioeconomic systems and consider where these interactions are relevant to policy making and decisions. For the purposes of this scoping report we identify the main pathways of loss and degradation for seagrasses in the UK from drivers, through pressures to state change. We identify the types of state change resulting from these pressures. By identifying the ecosystem services provided by seagrasses (Objective 2) and the factors regulating ecosystem function and hence the provision of these services we will be able to identify the implications of different pressures for human welfare, and
examine whether standard monitoring techniques and information bases allow managers to account for these interactions. Finally, in objective 3 we explore how current institutional frameworks and policy are linked to the delivery of a range of ecosystem services by seagrass ecosystems using Rapid Policy Network Mapping (RPNM).

Figure 1 Complex socio-ecological systems with interactions between ecosystem, societal system and policy/institutional frameworks indicated (A-F) and are explained in the text below. Adapted from Kannan & Burkhard, 2009).

Figure 1 illustrates the interactions between the three Objectives. Objective 1 scopes linkages between the ecosystem and the human domain that involve the delivery of ecosystem services (A), and how ecosystem function can change with sectoral activities and their pressures (B). Also included is the policy frame which this socio-ecological system sits within (C and D). Objective 2 is concerned with the provision of ecosystem services (A) and their value to society and the synergies and tradeoffs (B, C and D) and how these are affected by changes in use and management (E and F). Objective 3 will map out the institutional and policy frameworks (C, D, E and F) and explore improvements to the delivery of ecosystem services. Objective 4 is focussed on evaluating the transferability of this information both to other systems (A, B, C) and other locations with different societal contexts (D, E and F).
Objective 1: Using the DPSWR framework define and identify the key interactions between seagrass ecosystems and socio-economic systems and consider where these interactions are relevant to policy making and decisions.

Emma Jackson and Olivia Langmead

Approach

The DPSWR (Driver-Pressure-State-Welfare-Response) model (see Figure 2) has proved a useful tool for clarifying complex social, economic and ecological interactions (Borja et al. 2006, Langmead et al. 2009, Atkins et al. 2011). Drivers are largely economic and socio-political (industrial or agricultural development, trade, regulations, subsidies, etc.) and can reflect the way benefits are derived from ecosystem goods and services; Pressures are the result of the way in which Drivers change the environment (agricultural runoff of nutrients, pollution discharges, physical disturbance from anchoring, introduction of alien species, etc.); State change is a measure (or proxy) of the consequences of Pressures on species or ecosystems; human Welfare changes are measures of the ‘costs’ to society as a result of State changes (including indirect effects such as the knowledge that a species is endangered) and this is through the change in the delivery of ecosystem services; and Responses are how society attempts to reduce the Welfare impact or compensate for it through policies. Responses can be directed toward the Driver (regulating sectoral activities), Pressure (e.g. through mitigation), state (restoration and protection) and/or human Welfare Impacts.

Figure 2 DPSWR diagram showing the relationships between social and ecological components of the system and the trade off between welfare impacts (the ‘costs’) and the benefits that society derives from exploiting ecosystem services. [Source: Mee, 2005]
Using the DPSWR framework, we built a conceptual model of the key interactions occurring within seagrass/human systems, focusing on D-P-S. Published literature dealing with seagrass habitat loss in the UK and abroad were reviewed. When evidence of D-P-S pathways was not available for the UK we looked at European studies and beyond, but focused purely on species of seagrass found within the UK. In addition to personal libraries, we used Web of Knowledge to identify papers containing set terms relating to the habitat, location and disturbance. Records were entered into a database given a spatial reference point and specific details were entered regarding dates and both measured variables and data relating to seagrass state change (split into structural and functional), pressures and drivers.

**Preliminary outputs**

A number of potential activities were identified as causing either physical, chemical or biological environmental change (Figure 3) which may result in a specific state change to the seagrass beds, whether that be a change in extent (manifested as a change in depth limit, fragmentation or cohesion) or by changing those factors which limit seagrass growth and health. Potentially negative and positive pathways of change were considered. Whilst this model is not exhaustive, it provides a framework of looking at the pathways of how different activities influence the seagrass via the pressures.

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*Figure 3 Driver – Pressure-State change conceptual model for seagrass in the UK*
Figure 4 Preliminary State- Welfare – Response model for UK seagrass

Figure 4 illustrates the relationships between some common state change variables in seagrasses and the ecosystem services provided. Some relationships maybe positive introducing some interesting questions regarding the trade-offs between the provisions of different services. There are also inter-linkages and dependencies between different intermediate services or processes. Figure 4 also illustrates some local scale responses directed at the pressures and state. Responses can also be at higher institutional levels and directed at Drivers as well as pressures (see Objective 3).

By populating these models with examples and evidence (via the database) it can be used as a basis for more detailed case specific pathway analyses, which can then be used to advise on the most appropriate site monitoring using fit for purpose state and pressure indicators. Under the DPSWR framework fit-for-purpose refers to those pressure indicators which show cause and effect relationships with specific aspects of seagrass state change, and that the indicators of state change can be related to ecosystem service provision (see Objective 3). These can then be matched to current monitoring standards both in the UK and abroad (e.g. Common standards monitoring for seagrass in SACs and angiosperm tools for monitoring under the WFD, see examples in Table 1 from the UK and Table 2 from the Mediterranean), or used to advise future monitoring, for example of new Marine Conservation Zones.
### Table 1 Examples of Indicators used in the conservation management of seagrass beds in the UK

<table>
<thead>
<tr>
<th>Indicator</th>
<th>Pros</th>
<th>Cons</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Extent</strong></td>
<td>Links to ecosystem services.</td>
<td>Extent does not indicate configuration</td>
</tr>
<tr>
<td><strong>Configuration</strong></td>
<td>Linked to ecosystem services and to mechanical pressures</td>
<td>Expert analysis required</td>
</tr>
<tr>
<td><strong>Depth limit</strong></td>
<td>Indicative of changes in water clarity/ quality</td>
<td>Natural variability is low</td>
</tr>
<tr>
<td></td>
<td>Easy to monitor regularly</td>
<td></td>
</tr>
<tr>
<td><strong>Density/ Cover</strong></td>
<td>Measurable by non specialist, Indicative of changes in water clarity/ quality</td>
<td>Natural variability is high</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Difficult/ time consuming to measure</td>
</tr>
<tr>
<td><strong>Epiphyte cover</strong></td>
<td>Fast response indicator of change in water clarity/ quality important in terms of a number of ecosystem services</td>
<td>Requires seasonal sampling</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Not all epiphytes bad (take a structural group approach)</td>
</tr>
<tr>
<td><strong>Wasting disease Index</strong></td>
<td>More indicative of disease than presence of Labyrinthula (the slime mold protists responsible for the disease)</td>
<td>Requires seasonal sampling and monitoring of water temperature</td>
</tr>
</tbody>
</table>

As outlined in Article 3 Clause 1a, the Marine Strategy Framework Directive focuses on the Economic Exclusive Zone beyond 1nm. Seagrass occurs within 1 nm of the coast and is included in the the macrophyte descriptor for the WFD, so the WFD, not MSFD, is the policy driver in this regard in 1nm. Article 10 of the MSFD stipulates that MS have to set a comprehensive set of environmental targets which “take into account the continuing application of relevant existing environmental targets laid down at national, Community or international level in respect of the same waters...”. Preamble 12 of the Marine Strategy Framework Directive (MSFD) refers to ‘coastal waters’ (as defined in Directive 2000/60/EC, the EU Water Framework Directive (WFD)). It states that these coastal waters, including their seabed and subsoil, are an integral part of the marine environment, and as such should also be covered by the MSFD, only *in so far as particular aspects of the environmental status of the marine environment are not already addressed through the WFD*, so as to ensure complementarity while avoiding unnecessary overlaps. This suggests that at least for those indicators/ quality elements that are common with WFD classification (like macrophytes), the MSFD targets should be compatible with the 'good ecological status' set as a part of the WFD implementation. However there is a need to examine whether these indicators appropriately cover the MSFD descriptors.
Table 2 Examples of Indicators used in the conservation management of seagrass beds in the Mediterranean

<table>
<thead>
<tr>
<th>Indicator/metric</th>
<th>Name</th>
<th>Details</th>
<th>Ref</th>
</tr>
</thead>
<tbody>
<tr>
<td>AQI</td>
<td>Angiosperm Quality Index</td>
<td>integrates two important directives: the Water Framework Directive (WFD) and the Habitats Directive (García et al. 2009)</td>
<td></td>
</tr>
<tr>
<td>BIPO</td>
<td>Biotic Index for Posidonia oceanica</td>
<td>WFD focus developed on the basis of all P. oceanica monitoring data available in the western Mediterranean and on a standard assessment of anthropogenic pressures good relationship with human pressures (Lopez y Royo et al. 2009, Royo et al. 2009, Lopez y Royo et al. 2010, Lopez y Royo et al. 2011)</td>
<td></td>
</tr>
<tr>
<td>PREI</td>
<td>Posidonia oceanica Rapid Easy Index</td>
<td>based on five metrics: shoot density, shoot leaf surface area, E/L ratio (epiphytic biomass/leaf biomass), depth of lower limit, and type of this lower limit. (Gobert et al. 2009)</td>
<td></td>
</tr>
<tr>
<td>CI</td>
<td>Conservation Index</td>
<td>Water Framework Directive (WFD) (Montefalcone 2009)</td>
<td></td>
</tr>
<tr>
<td>SI</td>
<td>Substitution Index</td>
<td>Water Framework Directive (WFD) (Montefalcone 2009)</td>
<td></td>
</tr>
<tr>
<td>PSI</td>
<td>Phase Shift Index</td>
<td>Water Framework Directive (WFD) (Montefalcone 2009)</td>
<td></td>
</tr>
</tbody>
</table>
Changes in state of seagrass is potentially relevant to a number of descriptors in the MSFD (for example 1, 2, 3, 5 and 6). More importantly, as the practical mechanism for implementing the *Ecosystem Approach* to Europe’s seas within the MSFD, healthy marine ecosystems will be a condition to realise the potential benefits resulting from the ecosystem services they provide. Indicators must be appropriately linked to ecosystem services and the drivers of change. Without this managers will lack the necessary evidence-based feedback to learn from, and improve upon previous management approaches (adaptive management). Fit-for-purpose social and ecological indicators need to be able to track the levels of biodiversity or flows in terms of ecosystem services, detect change before it becomes irrevocable damage and identify the causes of change. Potentially relevant states to be captured include:

- Loss of seagrass can occur at different spatial scales;
- Extent of seagrass may stay the same but the habitat may become increasing fragments or depth limits may change;
- Loss of seagrass but replacement by algae may allow some habitat function to persist; and
- Degradation without loss may reduce function without structural change.

But these are being more extensively reviewed (alongside the collation of information for the SRI) and compared to WFD indicators.
Objective 2: Investigate the ecosystem services (ES) provided by seagrass ecosystems, examine variability in the delivery of these services and review the interdependencies between different services. Determine to what extent this information is currently incorporated into policy and management decisions.

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Approach

Utilising current ecosystem literature we define all ecosystem services provided by seagrass meadows, identified in Objective 1 (Beaumont et al. 2008, Barbier et al. 2011) and utilising expert knowledge suggest appropriate indicators for these ecosystem services. Following this we reviewed approaches for the valuation of these services and carried out a preliminary valuation.

Seagrass ecosystem services

Seagrass beds are stated as providing a number of ecosystem services from provisioning, regulating and cultural categories (Barbier et al. 2011). They function as important nursery and foraging habitat for fish, shellfish (Jackson et al. 2001b, de la Torre-Castro et al. 2009, Warren et al. 2010) and wildfowl (Ganter 2000a). They are also thought to oxygenate and stabilise sediments, providing shoreline stabilisation and protection from erosion (Koch et al. 2009b), and are natural hotspots for carbon sequestration and nutrient cycling (Kennedy et al. 2010). They are considered a foundation species, i.e. a species that provides habitat and enhances ecosystem biodiversity and is home to intrinsically valuable species such as the seahorse (Garrick-Maidment et al., Curtis & Vincent 2005). They are also an important sentinels of system health, due to their sensitivity to both water quality and physical disturbances, and were developed as an indicator for the Water Framework Directive (Ward 1987, Foden & Brazier 2007). The landmark Costanza et al. (1997) paper ranked seagrass meadows, at US$3.8 trillion yr⁻¹, amongst the three most valuable ecosystems on earth on a per hectare basis, despite only the nutrient cycling function being considered. The fishery production value of seagrass in the gulf waters of South Australia alone has been estimated at $A114 million per year. (McArthur & Boland 2006, and see Fletcher et al. 2012 for other examples of valuations). In the UK The National Ecosystem Assessment highlight the other ecosystem services provided by seagrass, and their importance in terms of biodiversity, although no valuation was attempted (Norris et al. 2011), see Figure 5.
Although seagrass habitat’s ability to enhance local biodiversity has been examined and confirmed for the UK (Webster et al. 1998, Attrill et al. 2000, Bowden 2001), other functions have only been examined outside of the UK and, in some cases, outside of Europe, where conditions and species composition are significantly different. Even then, the link between seagrasses and the processes underpinning the delivery of these ecosystem services is still uncertain. What is certain is that all seagrass beds do not function in the same way or provide services to the same extent, yet they are currently all given equal status with regards to decisions of which landscapes to protect. In the following sections we review the evidence for different ecosystem service provision by seagrasses and examine the structural and functional characteristics of the habitat which underpin these services, and hence the appropriate indicators of change.

**Supporting services**

Seagrass beds have an important role in coastal primary production. Rates of productivity for the seagrass alone are large, the Mediterranean seagrass *Posidonia oceanica* can fix 550-1000 g C m$^{-2}$yr$^{-1}$, the middle-range of a tropical rainforest is 460-1600 g C m$^{-2}$yr$^{-1}$, but rates vary between species of seagrass and between different environments. *Zostera marina* annual primary production rates can be significantly lower 69 g C m$^{-2}$yr$^{-1}$ (Borum & Wium-Andersen 1980), or as much as 814 g C m$^{-2}$ (Wium-Andersen & Borum 1984). Up to 50% of...
the primary production in a seagrass meadow can be from the epiphytic algae (Mazzella & Alberte 1986).

The fate of seagrass primary production varies, primarily by location and species of seagrass (Duarte & Cebrian 1996). In 1997 Cebrián et al. examined the fate of leaf-blade production of four Mediterranean seagrass species including *Zostera marina* and *Zostera noltii*. *They found that Z.marina* transferred twice as much production to consumers as *Z. noltii* and *that most of the production was decomposed* by detritivores (Cebrián et al. 1997). Consumption of seagrass leaf production by herbivores was higher *Z.noltii* than for *Z. marina*. Excess production (not consumed nor decomposed) during the first year ranged from 9.2% for *Zostera marina* to only 1.5 % for *Z.noltii*. this difference was attributed to the faster-growing leaves of *Z. noltii* which lost a higher percentage of production to herbivores and recycling most of the residual detrital production, therefore storing relatively small pools of refractory detritus (Cebrián et al. 1997).

Seagrass detritus may form the basis of, or at least contribute to, coastal nutrient cycles and indirectly promote the health of a fishery. Wood *et al.* (1969) stated that seagrasses provide large quantities of detrital matter to coastal ecosystems. Bach *et al.* (1986) demonstrated that the export of *Zostera marina* detritus in a Beaufort (north Carolina, USA) estuary equaled, if not exceeded, that of *Spartina alterniflora*. Adams (1976a, 1976b) also suggested that the basis of the fish food chain in *Zostera* beds was detritus and its associated microbial community, whilst Brook (1977) bridged the gap between detritus and higher trophic level predators (including valuable commercial and sport fishes) by identifying a number of transient foragers.

**Provisioning services**

Historically, *Zostera marina* had a number of varied direct uses (raw material and food) across Northern Europe and North America (McRoy & Helfferich 1980). Cottam and Munro (1954) reported that eelgrass ash found at ancient village sites in Denmark may be due to the plants being burnt for salt, soda or just warmth. In Nova Scotia eelgrass was gathered and dried for commercial uses such as insulating buildings, upholstering furniture and sound proofing floors and walls of apartment buildings (Lewis 1932).

Seagrasses are still harvested in some counties for fertilizer (Hemminga & Duarte 2000, de la Torre-Castro *et al.* 2009) and in the Chesapeake Bay, USA, seagrass by-catch or beach-cast is used to keep crabs moist during transport. In parts of Africa seagrass is eaten and used to make jewelry and potions during rituals (de la Torre-Castro & Rönnbäck 2004, Fletcher *et al.* 2012). Currently in the UK there is no major direct use of seagrass, instead seagrass beds are thought to have a fundamental role in maintaining populations of commercially exploited fish and invertebrate species by providing one or more of the following: (1) a permanent habitat, allowing completion of the full life cycle, (2) a temporary nursery area for the successful development of the juvenile stages, (3) a feeding area for various life-history stages and (4) a refuge from predation (Jackson *et al.* 2001b). A number of reports have correlated diminishing seagrass cover to declining fish catches. Examples include the King George whiting (*Sillaginodes punctata*) in Westernport Bay Victoria, Australia (Kikuchi 1974, Bell & Pollard 1989) and soft-shell blue crab (*Callinectes sapidus*) in Chesapeake Bay, USA (Shabmann & Capps Jr 1985). Seagrass beds are highly complex diverse habitats in terms of both the landscape configurations and the microhabitats provided by the plant structure, supporting a diverse and productive fauna including a high
density of potential faunal prey items present (Adams 1976b, Webb 1991, Tupper & Boutilier 1995). Tupper and Boutilier (1995) hypothesized that the complexity of the seagrass (Z. marina) community meant that there was a greater range of prey items available to young-of-the-year cod, which resulted in better growth and better survival after leaving the seagrass bed. Similarly, Valle et al. (1999) suggested that the occurrence of juvenile barred sand bass almost exclusively within Z. marina was due to greater prey availability, enabling faster growth to a size that is less vulnerable to predation. In Limfjord (Denmark), hatchery-reared cod were released to seagrass (Zostera marina) beds to improve their initial survival (Støttrup et al. 1994).

More research is required that links juvenile abundance to adult populations and to the diversity of the landscape and associated biota. However there is sufficient knowledge to improve on current “one-size-fits-all” approaches to protecting seagrass irrespective of its potential to provide services (Boström et al. 2006, Jackson et al. 2006, Unsworth et al. 2007a, Unsworth et al. 2007b).

Regulating services

Coastal protection

It is estimated that coastal erosion will cause losses of up to £10 billion of economic assets over the coming decades and the Environment Agency allocated over £745 million of funding to reduce the risk of flood and coastal erosion in England and Wales for the year to March 2011 (Environment Agency 2011). Properly informed management of natural resources (including seagrass beds) to help reduce coastal erosion has clear socio-economic benefits. Hydrodynamic damping by seagrasses is well known (Fonseca & Cahalan 1992, Bradley & Houser 2009). Seagrass beds reduce current velocity and attenuate waves by extracting momentum from the moving water (Koch et al. 2006). Currents in seagrass beds can be 2 to 10 times slower than in adjacent unvegetated areas. Roots and rhizomes bind sediment and reduce resuspension and beach-casted debris is important for controlling onshore coastal erosion (Hemminga & Nieuwenhuize 1990). There are historical reports that in the Isles of Scilly, UK die back of seagrass in the 1930s following the wasting disease resulted in greater fractions of mud being transported into Crow Sound making it unsuitable as a trawling ground for flatfish.

Although these ecosystem functions of seagrass may result in coastal protection, it cannot be assumed that the presence of seagrass will lead to the full provision of this ecosystem service (Barbier et al. 2008, Koch et al. 2009a). Wave attenuation and reduction in current velocity is a function of density of the seagrass bed (Gambi et al. 1990, van Keulen & Borowitzka 2002), the hydrodynamic conditions of the area (Koch & Gust 1999a), as well as the depth of the water column compared to the height of the seagrass determines the impact seagrass has on the water movement. The balance between forces of sedimentation and accretion is shown to be important in Zostera bed (Koch, 1999). There is a temporal fluctuation in the sedimentation processes. The fine sediment that is trapped in the growing season can be released during the winter storms (Bos et al. 2005).
**Water purification**

Seagrass beds can ameliorate detrimental inputs to coastal waters via two processes: nutrient uptake and suspended particle deposition. Seagrasses remove nutrients from the sediments and both seagrass and their associated plant groups (e.g. epiphytes) remove nutrients from the water column (Short & Short 1983, Thomas et al. 2000). The nutrients incorporated into the tissue of seagrasses and algae may be released, slowly, back into the water column once the plants decompose or they may be buried within the seagrass or elsewhere and stored for a longer period (Romero et al. 2006). Seagrass beds can contribute to improved water column transparency through two main pathways. The attenuation of waves and baffling of currents by the leaf blades and associated epiphytic algae increases the sedimentation rate of (Koch & Gust 1999b) and suspension and deposit feeding animals actively remove rate of small particulate matter (Gacia et al. 1999).

Like many plants seagrasses are able to take up and concentrate heavy metals, organic compounds and substances such as Tributyltin (TBT), without any apparent adverse effects. Francois et al. (1989) studied the decomposition of TBT in the tissue of seagrasses and found that the plants acted as detoxifiers, releasing monobutyltin in to the surrounding water. Chemical oceanographers are appreciating that seagrasses represent biotic heavy metal reservoirs (McRoy & Helfferich 1980). Where the burial of seagrass detritus is high the seagrasses may act as heavy metal sinks, elsewhere however seagrass communities may remobilise and transport these elements to higher trophic levels.

**Carbon Sequestration**

Recent global initiatives such as “Blue Carbon” have highlighted the importance of seagrass beds for their potential long-term C sequestration capacity (Kennedy & Björk 2009, Nellemann et al. 2009). A global assessment of seagrass meadows as carbon sinks by Kennedy et al. (2010), showed that seagrass meadows were important repositories of carbon produced not just from the seagrass but elsewhere (e.g. terrestrial and plankton), at a ratio of 50:50. Kennedy et al. (2010) also noted that a large proportion of seagrass can be exported to adjacent beaches or even the deep sea, the latter constituting a site of long term storage.

Despite its recognised importance there have been few studies assessing the carbon burial rates, and knowledge of the sequestration capacity of Z. marina beds is rudimentary. One of the few published estimates of Z. marina carbon burial rates was made by Cala Jonquet on Spanish seagrass meadows at 0.52g C ha\(^{-1}\) yr\(^{-1}\)(Cebrián et al. 1997). Spatial and temporal variability in sequestration capacity may be linked to changes in the physical environment and vegetative traits of the seagrass (deeper seagrass beds may not sequester the same amount of carbon as shallow beds). For example accretion rates (and consequently burial rates) of carbon vary from site to site due to currents, growth rates and wave exposure. Also, whilst most data on carbon burial rates are obtained on a short term basis accretion varies over time due to mortality and erosion events (Kennedy et al. 2010). In addition some studies average over different species of seagrass or do not include net community production and the allochthonous carbon trapped in their seagrass sediments which accounts for almost half of the carbon buried (Kennedy et al., 2010).
Cultural services

Seagrasses are the primary food source for a number of culturally valuable species, for example manatees and dugongs (Lefebvre et al. 2000), green sea turtles (Lal et al. 2010), and critical habitat for thousands of other animals which are a non use value, for example seahorses (Curtis & Vincent 2005, Teske & Beheregaray 2009). The habitat value is attributed to lower predation risk (Choat 1982, Hindell et al. 2000), greater food availability (Edgar 1999), increased sediment stability, and refuge from hydrodynamic forces within seagrasses (Lewis & Stoner 1983).

In the UK, the predominantly *Zostera* spp. associated culturally important Long snouted Seahorse (*Hippocampus guttulatus*) (Garrick-Maidment et al. 2010). This species of seahorses is commonly associated with seagrass meadows (Lourie et al. 1999). In 2008, the two species of seahorse found in the UK, *H. guttulatus* and *H. hippocampus*, were given legal protection from disturbance under the Wildlife and Countryside Act (England only) due to their high cultural value. The identification of both species of UK seahorse in Studland Bay in Dorset has led to significant increase in the number of divers visiting the area.

Although some baseline data exist with respect to the occurrence of seahorse in seagrass in the UK, evidence of the functional importance of seagrass in providing bottom-up control of their abundance is limited. In 2005 Curtis & Vincent examined the effects of seagrass bed structural complexity on the habitat partitioning of the two species of seahorse found in the UK, *Hippocampus guttulatus* and *H. hippocampus* in the Ria Formosa lagoon in southern Portugal (Curtis & Vincent 2005). At a landscape scale, *H. guttulatus* abundance was positively correlated with an index of habitat complexity (the percentage of substrate covered by seagrass and sessile fauna). Conversely, *H. hippocampus* used more open and less specific habitats. At a microhabitat scale *H. guttulatus* grasped all types of structure with equal probability while *H. hippocampus* avoided fauna and flora that formed large colonies or tracts of dense vegetation. Such differences in habitat preference suggest that managing seagrass beds to conserve both of these species would be counterintuitive.

Intertidal *Zostera noltii* and shallow *Zostera marina* are also an important food source for a number of valued water birds in northern Europe, including swans, ducks, coots and geese. Of these the most significant in the UK are Brent Geese (*Branta bernicla*), as this species is more dependent on seagrass than other species (being exclusively herbivorous) (Cottam et al. 1944). The Brent goose is an Amber List species in the UK because of the important numbers found at just a few sites. The sudden decline of *Zostera* after the 1930s wasting disease (Den Hartog 1987) coincided with population crashes of brent geese on both sides of the Atlantic.

Methods for Valuing Seagrass Ecosystem Services

There is considerable benefit in the qualitative assessment of ecosystem services arising from a given ecosystem, particularly if this assessment can provide a before and after perspective of different development options. So, for example, in the case of seagrass meadows a simple qualitative description of each the ecosystem services and associated benefits will at least ensure that all the benefits, commercial and non-commercial, arising from the seagrass meadows are considered in the decision making process and there is clearly significant merit in this action alone.
However, to further enable the decision making process, and to clarify the costs and benefits of varying development options, it can be useful to quantify the ecosystem services. Valuation, both monetary and non-monetary, is useful to enable transparency in the extent of the trade-offs between different ecosystem services (and associated beneficiaries) if different development options are being considered. This clarity of exactly what is being gained and lost, and by whom, under different future scenarios is of significant importance in a decision making environment.

To value ecosystem services it is essential to address the benefits which they provide, rather than focussing on the services themselves (Fisher et al. 2009, NEA 2011). This enables the avoidance of double counting which can tend to be an issue in ecosystem service valuation (Fisher et al. 2009). A number of methods have been developed for environmental valuation that can be applied to ecosystem benefits provided by seagrass meadows. The 2011 NEA Economics chapter provides a valuable and comprehensive introduction to environmental valuation and methods available (Bateman et al. 2011 in NEA 2011)). These methods vary in the extent to which they depend on an understanding of the ecosystem. Some methods, such as production functions, input-output models, and replacement cost methods require an in depth understanding of the dynamics of the biogeochemistry of the environment. Other methods, such as contingent valuation and choice experiments (both stated preference approaches) require minimal understanding of the ecological processes underlying the ecosystem service provision. The selection of a valuation methodology will vary depending on the ecosystem service type, data availability, and time and resources available for application.

This section aims to detail the benefits provided by the ecosystem services, and discusses the methodologies which are available to value these benefits. The applicability of the methodologies, given current understanding and data availability, is also detailed.

**Benefits arising from provisioning services**

As described in the previous section there are known linkages between seagrass beds with the provision of refuge, nursery habitat and breeding grounds for commercial fish and invertebrate species, and as a source of food for all levels of the food chain. This service has previously been defined as a “habitat service” (TEEB 2010). A benefit of this service is the improved provision of fisheries and shellfisheries, in terms of quality, quantity and security of supply.

To value this benefit a production function method can be applied. Provided a quantitative relationship between the type and spatial extent of seagrass meadows and the fish and invertebrate productivity can be determined, then it becomes possible to estimate the contribution of seagrass beds to UK fisheries (McArthur and Boland, 2006). A value can then be attached to this benefit by combining market prices of UK landings, with seagrass productivity rates and areas, to value the seagrass beds in terms of their contribution commercial fisheries.

This methodology has been applied to Australian seagrass beds by McArthur and Boland (2006), and to UK saltmarsh by Fonseca (2011). There are no available studies on UK seagrass beds. Fonseca et al. (2011) determined a quantitative estimate of annual abundance of juvenile bass (up to two years old) per hectare of saltmarsh. This study then applied a range of average survival rates to an approximate length of 36cm (the minimum size for legal commercial capture of wild bass in the UK). This enabled an estimation of the
per ha contribution of saltmarsh to local inshore bass fisheries, in units of kg bass per ha saltmarsh. As the wild bass is sold at the market this contribution was then valued using market values. McArthur and Boland (2006) took a similar approach but investigated a wider number of fish species, estimating that seagrass habitats contribution to total secondary production is in the order of $A133.23/ha/yr (£83/ha/yr). This is equivalent to a contribution of $A114 million (£71 million) per year in terms of secondary production in the gulf waters of South Australia.

The methodology for the valuation of this service is well established, and the majority of the required natural and economic data is available, although the collation and analysis of this data would require considerable time. In terms of the natural science, exploited species supported by seagrass beds may include bream, bass, cuttlefish, pollack, wrasse and common prawn. Hence, to determine the true value of this service all the exploited species must be considered, including species collected via recreational fishing.

To enable the economic valuation UK fisheries data can be collected directly from the Marine Management Organisation (MMO) and with sufficient resources these data, covering both extent and value of UK fisheries, should be readily available and would enable a first estimate of value of this service. It is noteworthy however that to use the market value of the fisheries is an over simplification of the value of the ecosystem service. There are three key reasons why the market values do not equate directly to the ES value:

1. The market value of the fisheries will be a combination of both the ecosystem service value and the human and manufactured capital needed to extract the fish. To determine the ES component of the value the market value of the fisheries must be divided into two components: ecosystem service values and the cost of the human and manufactured capital. Attributing the entire value of landings to the value of the fisheries ecosystem service is a simplification and an over-estimation. To disaggregate the ecosystem service value from the non-ecosystem service values it is necessary to quantify all the human and manufactured costs used in fishing activities across different vessel and gear types. These will include fixed costs such as one off payments for fishing boats and gear; variable continuous costs such as fuel, wages, ice, boat and gear repair and maintenance, the purchase of quota and insurance; and compliance costs such as licences. Subsidies provided to the fishing industry would also have to be taken into consideration (e.g. exemption from fuel duty on marine diesel, support for vessel refits), as would capital investment and debt levels. Calculating the human and manufactured capital costs is difficult as the required data is not readily available. For example, there are data on number of fishermen, 39,380 in 1948, falling to 10,242 in 2008 (MMO data), but there are little data on salaries or income. Equally the MMO holds data on the number of vessels at sea and their capacity (gross tonnage) and engine power, but the temporal scope of these data is poor, and there is a paucity of data relating to the capital and maintenance costs of the vessels. The scarcity and irregularity of these data proves a real challenge to separating the ecosystem service value from the natural and manufactured capital costs, and this is an area which will require further research in the future.

2. A second complication of using market values to value the fisheries resource is that they tend to focus on the first sale price of fish, and do not consider the extensive range of secondary services which are supported by fisheries, both up and down the
supply chain. These vary from boat builders and repairers, and gear merchants to fish processors and food industries, including 480 fish processing sites that employ around 15,000 people (http://www.seafish.org). In 2005 31,633 people were employed in the catching, processing and aquaculture sector in the UK, representing 3.5% of the total employment in all 24 maritime industries in the UK (Pugh, 2008). In addition there are also cultural and health benefits associated with fisheries which should also be considered.

3. A final issue in this valuation is the need to consider the sustainability of the fishery. If the fishery is not sustainable the potential temporary nature of the supply should be included within the value of the ES. There are some studies on the sustainability of the UK fisheries, for example, Armstrong and Holmes (2011) report on the sustainability of UK fisheries. They found that the proportion of UK stocks being harvested sustainably has increased since the 1990s, but that the fishing mortality in most stocks remains high and above levels that will support the maximum sustainable long-term yields or economic returns under current environmental conditions. In addition a number of other commercially important species are not assessed due to inadequacies in the available data. Any environmental value estimate should include the sustainability of supply and more research is clearly needed to develop our understanding in this area.

In conclusion a ball park estimate of this service could be made utilising currently available information. However this estimate will have significant uncertainty associated with it, and there is considerable scope for improving this estimate given additional time and resource.

An alternative approach to valuing the contribution of seagrass to commercial fisheries was explored by Paulson (2007). This study investigated reductions in the seagrass habitat in Bohuslän and Halland counties along the Swedish west coast, and the associated commercial eel fishery losses. Professional eel fisherman based along the Swedish west coast were surveyed. The reduction in seagrass habitat was self-reported to result in an annual eel catch loss of 475 kg per fisherman, valued using the price of eel of 60 SEK/kg. The additional costs, including the cost of extra working hours, and higher fuel consumption are estimated to be 6,455 SEK per fisherman per year. The total economic loss in commercial eel fishing due to seagrass habitat loss was therefore calculated to be 6.4 million SEK for all eel fishermen on the west coast (2005 Swedish Kroner). The results of this study are not directly transferable to the UK, but the methodology may be, hence its inclusion here.

The role of UK seagrass meadows in the provision of raw materials is considered to be minimal and so is not valued here.

**Further recommended research:**

1. Improved understanding of the role of seagrass beds in the enhanced provision of UK fisheries, including shellfisheries and invertebrates.
2. Application of production function technique to determine the contributory value of UK seagrass beds to the provisioning ecosystem service.
3. Investigation of alternative techniques to value this ecosystem service.
4. Development of economic methodologies to derive the true ecosystem value of fisheries from the market values.
5. Further investigation into the value of non-commercial fisheries which are supported by UK seagrass meadows.
Benefits arising from regulating services: Disturbance prevention

Disturbance prevention is the dampening of physical environmental disturbances by biogenic structures (Beaumont et al. 2008). This service refers primarily to protection against floods, storms, waves and the control of coastal erosion. The primary benefit of this service is the protection of property, land and human safety. The extent to which seagrass meadows prevent disturbances such as floods and coastal erosion in the UK is not known. Without this understanding of the natural science it is not possible to attribute a value to this service at the current time. However it is of interest to consider the potential valuation methodologies.

There has been extensive conceptual research regarding the valuation of the disturbance prevention service (EFTEC 2010, Penning-Roswell 2010), and a variety of methods are available to value the resultant benefits, including coastal defence expenditure avoided (King & Lester 1995, NEA 2011), and damage costs avoided (Penning-Roswell et al. 2010).

One approach to valuing coastal defence is to utilise a replacement cost method, for example, to calculate the cost of replacing a habitat with a sea wall. King and Lester (1995) estimated that in sea defence terms, assuming an 80m width saltmarsh, UK saltmarshes could result in cost savings relative to building man-made structures of £0.47 million to £0.94 million per hectare in terms of capital costs, and £9400 per hectare in terms of annual maintenance costs (adjusted to 2010 prices). This approach was also applied in the NEA (2011). If a greater understanding of the role of seagrass meadows in sea defence was available this understanding could be coupling with cost savings data to estimate the cost of replacing UK sea grasses with man-made sea defences to provide an estimate of the value of this service.

There are however numerous problems with this replacement cost method. A primary source of inaccuracy is that values calculated by the replacement cost method do not capture the full economic value of the ecosystem service being valued; this is that the replacement of a natural structure with a man made alternative may result in the loss of other services which the ecosystem was providing, for example carbon storage and nutrient cycling. A second source of inaccuracy is that this approach does not consider the value of the land which is being protected. The value of this service will depend upon the type of land being protected, for example, high grade agricultural land or habitats of international conservation importance will have a greater value than low grade lands (Pye et al. 2007). A third inaccuracy is that this approach also does not take into account the risk of flooding, not least as the topography of the land behind the defence is not considered.

The damage cost avoided method, as promoted by Penning-Roswell et al. (2010) is an alternative to the replacement cost method. Damage costs avoided is more complex, and can include investigating the market prices of the property and land being protected, coupled with the extent of the protection afforded by the marine ecosystem. Penning-Roswell et al. (2010) present a series of methods for assessing the vulnerability of an area to flooding (from both rivers and the coast) and coastal erosion; calculating potential damage costs to land, property and recreational uses, and emergency costs (e.g. police, fire and ambulance; local authorities; and environment agency); estimating the probability of flooding based on
topographical data; and calculating the damage costs not avoided by the defence scheme (i.e. defence schemes are built that will not protect against all floods, some flooding may still therefore occur). This approach would provide a more accurate estimate of the value of this service, but again data availability is a restriction to this approach, and it also does not consider whether the two options are indeed perfect substitutes, or the willingness of individuals and society to replace a natural structure with a man-made alternative.

A final option for the valuation of this service is value transfer (Eftec 2010). This approach transposes values estimated at one site (an original study site) to another (the site of interest). Eftec (2010) have applied this approach to three case study sites in the Humber Estuary. While this approach potentially considers some aspects of individual preferences, it also faces short-comings. For example, the extent to which it is realistic to transfer values from one site to another has been questioned and good studies may not be available from which to transfer values (benefit transfers can only be as good as the original studies used).

There are economic methodologies available to value the ecosystem service of disturbance regulation. However, whilst there is considerable documentation on the conceptualisation of these methodologies they require further refinement and validation through practical application before they can be applied with confidence.

**Recommended Future Research:**

- Quantitative estimation of the role of seagrass in coastal defence for specific locations;
- Methods to value the service of coastal defence: As discussed there are numerous methods of valuing this service, all of which have benefits and but all also have significant sources of inaccuracy. It would be valuable to develop a method specifically for application to seagrass meadows, but with the potential to be transferred to other ecosystems. This is currently a significant gap in the valuation research horizon. Seagrass meadows are particularly interesting as their role is less direct than the previously valued saltmarshes and sand dunes but may, on a per ha basis, be equally as significant. As such the economic methodologies will need to be specifically designed and applied.

**Benefits arising from regulating services: Bioremediation of waste**

The marine ecosystem has the capacity to process contaminants, alleviating pollution effects, through storage, burial and recycling (bioremediation of wastes). The immediate benefit of this service is that humans can utilise the marine ecosystem to process wastes, which otherwise would need to treated or stored in order to avoid health implications. As detailed in the previous section seagrass meadows play a significant role in the service of water purification. However the quantitative extent of this role is not currently known, and without this understanding of the natural science it is not possible to attribute a value to this service at the current time. However, as with coastal defence, it is worthwhile noting the valuation methodologies which would be available to value this service.

The value of this benefit can be determined in different ways. One method would be to assess the costs associated with treating the wastes if utilising the marine environment was not an option (a replacement cost method). This would entail investigating the amount of
wastes which enter the seagrass meadows and are then “processed”, through storage, burial and recycling into a non-harmful form. It would then be possible to calculate the cost of an anthropogenic technological alternative to process this waste. This method is not ideal as discussed in the previous section and tends to lead to inaccurate and substantial overestimates of this service (Costanza et al. 1997; Waycotta et al. 2009). A second method would be to determine the ecosystem health implications if human wastes entering the marine environment were not processed (a damage avoided cost). This would include impacts on human health, but also impacts on the wider ecosystem and processes, and in turn the ES provided. There are pros and cons to both these methods but care should be taken not to undertake both as this would result in double counting.

The valuation of this service is not well developed and there are few studies which have applied either of the above approaches. Greater development of both the theory and practical application of these approaches is much needed.

**Recommended Future Research**

- Quantitative estimation of the role of seagrass in water purification and bioremediation of wastes.

- Development of economic methodologies for the valuation of the ecosystem service of bioremediation of waste. Methods to value the service of water purification: There is minimal research, methodological and applied, on valuation methods for service of water purification. This service is regularly discussed in the literature, but there have been few attempts to attach values to this service. There are a variety of potential methods, but the development of these techniques would be valuable. The UK seagrass ecosystem would be an ideal ecosystem in which to develop and trial such methodologies.

**Benefits arising from regulating services: Climate regulation**

A specific benefit of the climate regulation service provided by seagrass meadows is the sequestration and storage of carbon. At present, about an estimated one third (~2 Gt C yr-1) of anthropogenic CO2 emissions are sequestered by the oceans (Orr et al. 2001, Takahashi et al 2002). Seagrass meadows have the potential to sequester greenhouse gases, including CO2 and as a result play a considerable role in local and global climate control. It is possible to determine a value for this service in terms of carbon sequestered.

The annual amount of carbon sequestered by UK seagrass meadows can be estimated by combining seagrass area estimates with long term carbon sequestration and burial rates (Equation 1). The seagrass area estimate for the UK is 22,066 ha (based on published data over the past 20 years and an estimate of seagrass area for point records) and there is only one published value for carbon burial in *Z. marina* which is 52gC m-2yr-1. However, this rate will vary depending on the circumstances, including depth and wave exposure to name just a couple.

Eq1: Estimated annual UK Sea grass carbon burial = 22,066 ha * 0.52tC/ha/yr

= 11,474 tC/UK /yr

There are a wide range of studies that can be applied to attributing a value to carbon sequestration, although it is notable that most values are calculated on a basis of CO2.
emitted, not CO\textsubscript{2} stored. To enable a meaningful valuation of this service it is useful to
explore these different studies. The release of greenhouse gases from land use
(predominantly nitrous oxide, methane and carbon dioxide) is typically expressed in terms of
a common global warming potential unit of carbon dioxide equivalent (CO\textsubscript{2}e). Tol (2005)
provides a valuable review of the marginal damage costs of carbon dioxide emissions; that is
the estimation of avoided costs of climate change, or the value of the welfare loss to society
of emitting an extra tonne of CO\textsubscript{2}, in the form of things such as health impacts,
environmental disasters etc. The Tol (2005) review examines 28 studies resulting in 103
estimates of marginal damage costs caused by a metric ton of CO\textsubscript{2} emissions in the near
future. The arithmetic mean of these estimates is $97/tC, with a standard deviation of
$203/tC. The majority of these high estimates are found in the grey literature and are based on relatively
unlikely scenarios of climate change, impact sensitivity and economic values. If the non-
reviewed literature is excluded a mean of $43/tC and an SD of $83/tC is found. Despite the
uncertainty associated with these estimates Tol (2005) concludes that they do provide a
useful benchmark to compare against costs of emission reduction policies.

A key reference in derivation of UK climate policy and carbon values was the Stern Review
(2006). This derived values for the social costs of carbon (SCC), defined as:

“\textit{a measurement of the full global cost today of an incremental unit of carbon (or equivalent
amount of other greenhouse gases) emitted now, summing the full global cost of the
damage it imposes over the whole of its time in the atmosphere. It measures the scale of the
externality which needs to be incorporated into decisions on policy and investment options in
government}” (Defra 2007).

The Stern review calculated the social cost of carbon under different emissions scenarios,
but notably determines the carbon value to be between £16/tCO\textsubscript{2}e and £53/tCO\textsubscript{2}e under an
emissions scenario that leads to stabilisation at 550ppm CO\textsubscript{2}e.

Until 2009 the UK Government was one of the few authorities to use a Shadow Price of
Carbon (SPC) approach to value carbon emissions, and thus mainstream climate change
mitigation into policy appraisal and evaluation. The SPC is based on estimates of the Social
Cost of Carbon (SCC) drawn from the Stern Review. However using the SPC allows the
SCC to be adjusted to reflect: i. estimates of the Marginal Abatement Costs required to take
the world onto the stabilisation goal; ii. other factors that may affect UK willingness to pay for
reductions in carbon emissions, for example political desire to show environmental
leadership (Defra 2007). The calculated SPC values are shown in Table 3.

| Table 3 Shadow Price of Carbon (SPC) under a 550ppm scenario, using 1990-2005 market exchange rate and GDP deflator (£/tCO\textsubscript{2}) (Defra 2007) |
In 2009 DECC/HMT changed their CO$_2$ value appraisal guidance and DECC now uses the costs of mitigation (DECC 2011), or the costs of reducing emissions. Both traded and non-traded values of carbon are provided by this guidance, but as environmentally sequestered carbon is not traded in the EU Emission Trading System, it is recommended to use the non-traded price to value this carbon. These non-traded values represent the maximum marginal abatement cost that will need to be incurred to ensure we can meet our emissions reductions targets in the non-traded sector, based on current emissions projections (Error! Reference source not found.). It is worth considering what this value means. In the case of environmental carbon sequestration the validity of this approach depends upon the implication that this is the additional cost that would have to be incurred elsewhere in the UK in order to meet our reductions targets if this carbon sequestration were not to occur in the environment.

A sample of the summary of non-traded carbon values and sensitivities over the 2008 – 2100, Real £2011 (DECC 2011)

<table>
<thead>
<tr>
<th>Year</th>
<th>Low</th>
<th>Central</th>
<th>High</th>
</tr>
</thead>
<tbody>
<tr>
<td>2008</td>
<td>27</td>
<td>53</td>
<td>80</td>
</tr>
<tr>
<td>2009</td>
<td>27</td>
<td>54</td>
<td>81</td>
</tr>
<tr>
<td>2099</td>
<td>73</td>
<td>287</td>
<td>501</td>
</tr>
<tr>
<td>2100</td>
<td>71</td>
<td>284</td>
<td>497</td>
</tr>
</tbody>
</table>

It is also notable that none of these estimates should be used to determine the total value of the carbon sequestration ecosystem service. This is because these carbon prices are all based on marginal impacts, it is therefore not appropriate to value very substantial quantities of carbon in this way.

Given the variability in the carbon value estimates it is recommended that a variety of values are applied. Two rates are used:

1. The DECC Shadow Price of Carbon as these values fall fairly centrally within the wider range detailed by Tol (1995), and therefore are presumed to be a reasonable representation of carbon value calculated using marginal damage costs.
2. The DECC Marginal Abatement Costs. Although there is some concern about the validity of this approach to environmental carbon sequestration it is useful to compare the results of the two approaches.

Carbon is converted to CO$_2$ by multiplying the ratios of molecular weights, that is 44/12 or 3.67. Thus, if the carbon burial rate is 0.52t/ha/yr the respective CO$_2$e sequestration rates will be 1.91t/ha/yr for UK seagrass meadows. Combining these tCO$_2$e/ha/yr figures with data on UK areas of seagrass meadows, estimates of the capacity of this habitat to sequester CO$_2$, and carbon values, an ecosystem service value can be derived (see Table 4).

Table 4 Annual value of the ecosystem service of C burial by UK seagrass in 2011

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>£/tonne CO2e (adjusted to real £2011)</td>
<td>£31.19/ tonne CO2e</td>
<td>£56/ tonne CO2e</td>
</tr>
<tr>
<td>CO2 burial</td>
<td>1.91 CO2e/ha/yr</td>
<td>1.91 CO2e/ha/yr</td>
</tr>
<tr>
<td>£/ha/yr</td>
<td>£59.58/ha/yr</td>
<td>£106.96/ha/yr</td>
</tr>
<tr>
<td>UK area of seagrass</td>
<td>22066</td>
<td>22066</td>
</tr>
<tr>
<td>£/UK/yr</td>
<td>£1,314,692/UK/yr</td>
<td>£2,360,179/UK/yr</td>
</tr>
</tbody>
</table>

It should be clarified that this is only a ball park estimate that does not include spatial and temporal variability in the carbon burial capacity of UK seagrass beds, and potential feedbacks. The permanence of this storage has also not been addressed.

It is notable that these values are per annum, so provided the seagrass area, and associated carbon burial remains the same, this value will be received every year into the future. It is possible to use the Net Present Value (NPV) function, coupled with the Defra and DECC carbon value tables, and a recommended 3.5% social discount rate (HM Treasury 2011), to determine the long term value of this service.

$$NPV = \sum_{t=0}^{N} \frac{R_t}{(1 + i)^t}$$

Where \( t \) is the year; \( i \) is the discount rate (in this case 3.5%); \( R_t \) is the value of the ecosystem service at year \( t \); and NPV is the Net Present Value.

This relatively straightforward calculation of NPV would provide an improved insight to the long term "true" value of the seagrass with regard to carbon sequestration. Whilst not without uncertainties the economic methodologies in this area are well developed, and it would be
both interesting and valuable to apply this approach to the seagrass ecosystem with greater rigour and detail.

**Recommended future research:**

1. Improve current estimate of area of UK seagrass.
2. Improved C burial estimates, including investigation of permanence of storage.
3. Exploration of different carbon values and their impact on the valuation of the carbon burial ES.
5. Future scenarios of changes in area of seagrass, and associated changes in the value of the ES of carbon burial.

**Benefits arising from cultural services**

Cultural ecosystem services (CES) are the non-material benefits people obtain from ecosystems through spiritual enrichment, cognitive development, reflection, recreation, and aesthetic experiences (Millennium Ecosystem Assessment 2005). They are non-consumptive and primarily intangible in nature. The UK National Ecosystem Assessment (UKNEA) defines CES as “the environmental settings that give rise to the cultural goods and benefits that people obtain from the environment” (NEA 2011). Environmental settings are considered to be the places at certain points in time that are valued because they satisfy the fundamental human need for social interaction with others and with nature. This suggests that the overall value assigned to ecosystem functions or benefits cannot be derived from simply adding together the value of biophysical ecosystem components.

Although the cultural values associated with seagrass beds are known to be extensive and varied, including supporting important bird species and seahorses and providing snorkelling and diving opportunities (see earlier section), there is little information on their quantitative value.

A variety of contingent valuation studies have been undertaken in Australia to try and determine values for seagrass beds. Unfortunately none of these are in areas sufficiently similar to the UK allow benefit transfer of the results, but for interest they are reported here. Kragt and Bennett (2009) found marginal willingness to pay (WTP) to protect Tasmanian seagrass to be quite low, on average, equating to $A0.11 for a hectare increase in seagrass area. However two additional attributes were also valued: rare native animal and plant species, and native riverside vegetation. These attributes had WTP results of: $3.57 for a kilometre increase in native riverside vegetation and $8.42 for the protection of each rare native animal and plant species. Interestingly the presence of the seagrass will be integral to the provision of rare native animal and plant species, so although the WTP for the seagrass was quite low this is possibly due to a poor understanding of the supporting service which this habitat provides.

A second choice modelling study was undertaken by Rogers (2011) to estimate the benefits provided by Ngari Capes marine park in West Australia under different management regimes. Two distinct populations were surveyed: the general public of West Australia (public); and Australian marine scientists (experts). The study assessed similar
environmental attributes: seagrass, target fish stocks, abalone, and whales. The results for the seagrass are shown in Table 5, and they indicate that both the public and experts attach considerable value to increasing seagrass areas, although as may be expected the experts values were significantly greater.

### Table 5

<table>
<thead>
<tr>
<th>Increase in seagrass populations (sanctuary zone management; public)</th>
<th>5% increase in seagrass population</th>
<th>10% increase in seagrass population</th>
</tr>
</thead>
<tbody>
<tr>
<td>$49</td>
<td>$58</td>
<td></td>
</tr>
<tr>
<td>$52</td>
<td>$61</td>
<td></td>
</tr>
<tr>
<td>Increase in seagrass populations (government donation management; public)</td>
<td>$166</td>
<td>$287</td>
</tr>
<tr>
<td>Increase in seagrass populations (sanctuary zone management; expert)</td>
<td>$243</td>
<td>$261</td>
</tr>
</tbody>
</table>

A third and final study was undertaken by Rolf and Windle 2011. The objective of this choice modelling experiment study was to assess the values to protect the health of the Great Barrier Reef (GBR) at six different locations in Australia. The study investigated preferences for three different attributes of the GBR (coral reef, fish species and seagrass). The willingness-to-pay (WTP) estimates are detailed in Table 6, and again show a positive WTP to improve seagrass meadows, although this WTP is comparatively lower than for fish and coral reefs.

### Table 6

<table>
<thead>
<tr>
<th>City</th>
<th>Coral Reef</th>
<th>Seagrass</th>
<th>Fish</th>
<th>Aggregate Results</th>
</tr>
</thead>
<tbody>
<tr>
<td>Townsville</td>
<td>$15.58</td>
<td>$9.35</td>
<td>$13.60</td>
<td>$38.54</td>
</tr>
<tr>
<td>Brisbane</td>
<td>$12.45</td>
<td>$6.09</td>
<td>$7.99</td>
<td>$26.53</td>
</tr>
<tr>
<td>Sydney</td>
<td>$10.76</td>
<td>$4.93</td>
<td>$10.16</td>
<td>$25.85</td>
</tr>
<tr>
<td>Melbourne</td>
<td>$13.31</td>
<td>$4.45</td>
<td>$14.57</td>
<td>$32.32</td>
</tr>
<tr>
<td>Adelaide</td>
<td>$13.58</td>
<td>$6.58</td>
<td>$13.46</td>
<td>$33.62</td>
</tr>
<tr>
<td>Perth</td>
<td>$11.90</td>
<td>$5.88</td>
<td>$14.24</td>
<td>$32.01</td>
</tr>
</tbody>
</table>

It is unfortunate that these studies are not transferable to the UK, but they do show a positive WTP for seagrass meadows, and demonstrate that these methodologies have been trialled and established, and it is recommended that these approaches are now built upon in the UK. Given the absence of UK specific information it is recommended that a program of primary
surveys is undertaken to gain greater insights into the intangible values ascribed to ecosystems by groups of stakeholders. Results from such surveys can enhance the visibility of intangibles and enable their ranking in order of relative importance. Ideally a survey would be applied in several case study sites, and a questionnaire of this sort could be used to explore non-use benefits (bequest and existence values) and option use values by asking respondents how much they are willing to pay to conserve or improve seagrass meadows for their ecosystem services. The economic methodologies are available to develop value estimates for this service, and it is asserted that there would be great value in applying these in the future.

**Recommended Future Research**

- Fundamental understanding of cultural services, and associated valuation mechanisms: Despite the extensive and varied cultural benefits of UK seagrass meadows there is little information on the quantitative valuation of this service. This is a significant gap in the research. An investigation of potentially relevant cultural value studies, and a gap analysis of the research, would be a valuable addition; and
- Primary research valuing UK marine Ecosystem Services. The majority of valuation studies undertaken on the UK coast and seas use a Benefit Transfer approach, that is applying values from other locations. There is a fundamental absence of primary research in this area in the UK.

**Indicators**

As the practical mechanism for implementing the *Ecosystem Approach* to Europe’s seas, within the MSFD, healthy marine ecosystems will be a condition to realise the potential benefits resulting from the ecosystem services they provide. Indicators must be appropriately linked to ecosystem services and the drivers of change or managers will lack the necessary evidence-based feedback to learn from, and improve upon previous management approaches (adaptive management). Fit-for-purpose ecological and social indicators need to be able to track the levels of biodiversity or flows in terms of ecosystem services, detect change before it becomes irrevocable damage and identify the causes of change. Potentially relevant states to be captured include:

- Loss of seagrass can occur at different spatial scales;
- Loss can occur without root removal – so root functions are retained but above ground plant functions is lost;
- Replacement by invasive algae may allow some habitat function to persist; and
- Degradation without loss may reduce function without structural change.

For each ecosystem service, an understanding of the influence of different pressures on state change and a sound knowledge of how these state changes influence the provision of different service is a basic need for an Ecosystem based approach to management.
Table 7 provides an example of common state changes to seagrass resulting from the pressures discussed in section 1, and illustrates the likely consequences in terms of carbon sequestration.
Table 7 also indicates how well commonly used indicators of seagrass (under Common Standards Monitoring and developed for monitoring under the WFD, but now under consideration for the MSFD) capture these changes.
<table>
<thead>
<tr>
<th>State change (negative changes shown but positive change possible)</th>
<th>Pressures resulting in such state change</th>
<th>Influence on C-sequestration</th>
</tr>
</thead>
<tbody>
<tr>
<td>Loss of area of seagrass, rhizomes left intact</td>
<td>Physical disturbance</td>
<td>↓ Primary production severely reduced. Carbon sequestration halted. Erosion of rhizome mat increased as seagrass leaves no longer baffle currents. Area becomes a source of carbon (rate depends on the rate of breakdown of the mat)</td>
</tr>
<tr>
<td>Loss of area of seagrass, rhizome mat eroded/ perturbed Habitat shift to soft sediment.</td>
<td></td>
<td>↓ Primary production severely reduced. Carbon sequestration halted, although some carbon may still be locked up in sediments the area will become a source of carbon (rate depends on the rate of breakdown of the mat)</td>
</tr>
<tr>
<td>Replacement of seagrass with macroalgae (including invasive species)</td>
<td></td>
<td>↓ Rates of primary production reduced or maintained (dependent on algal productivity). However, loss of the root rhizome mat will result in severe reduction in carbon sequestration</td>
</tr>
<tr>
<td>Change in depth limit</td>
<td></td>
<td>↓ Rates of primary production reduced or maintained (dependent on cause of change in depth limit). The area of seagrass lost will release carbon however remaining bed will still lock up Carbon if limit is progressive or stable, but not if regressive, so an indication of limit type important.</td>
</tr>
<tr>
<td>Fragmentation</td>
<td></td>
<td>↓ Rates of primary production reduced or maintained (depending on overall loss in extent of seagrass). The area of seagrass lost will release carbon. Remaining bed will still lock up Carbon, but erosion may be increased (depending on wave energy on configuration of fragmentation), so may become a source of carbon.</td>
</tr>
<tr>
<td>Increased epiphytal load</td>
<td></td>
<td>↑ Rates of overall primary production will increase to a threshold where epiphyte growth compromises seagrass health and productivity. Enhanced wave baffling due to filamentous epiphytes may increase sedimentation and reduce erosion. Carbon sequestration increases unless combined with loss/degradation in seagrass.</td>
</tr>
<tr>
<td>Decrease in density</td>
<td></td>
<td>↓ Rates of primary production reduced. Wave baffling and erosion increased, however remaining bed will still lock up carbon (unless seagrass growing in a high energy environment)</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Indicators which capture this change in state</th>
</tr>
</thead>
<tbody>
<tr>
<td>Extent, configuration</td>
</tr>
<tr>
<td>Extent, configuration</td>
</tr>
<tr>
<td>% associated algae</td>
</tr>
<tr>
<td>Extent</td>
</tr>
<tr>
<td>Extent, configuration</td>
</tr>
<tr>
<td>Epiphyte community composition and % associated algae</td>
</tr>
<tr>
<td>% cover/density</td>
</tr>
</tbody>
</table>
Incorporating ecosystem service approaches and valuation into policy and management decisions

The ecosystem service approach clarifies the linkages between fundamental ecosystem processes and human welfare, enabling communication both between scientific disciplines and with the wider community. This approach enables policy makers to understand the broad ranging uses and socio-economic and ecological benefits arising from the marine environment, and hence maximises the potential of achieving sustainable use of marine resources and services. The ecosystem service approach is particularly valuable in highlighting the more intangible benefits that are obtained from the environment thus aiding the assessment of potential trade-offs when different development options are being considered (Daily 1997). As such regulators and stakeholders have increasingly been applying the ecosystem service approach to enable the development of more holistic and sustainable policies, for example Defra (Beaumont et al. 2007), Natural England (Fletcher et al. 2012), and the Crown Estate (Saunders et al. 2010).

The economic valuation of ecosystems and their services aims to define the value of ecosystem attributes in metrics (usually monetary values) which are directly comparable with competing commercialised options. The purpose of this is to raise awareness of the importance of the natural environment, but also to enable the direct comparison of ecosystems and their services with other anthropogenic activities. Ideally economic valuation should facilitate bringing the non-commercial benefits (such as bioremediation of waste, carbon sequestration and fundamental cultural well-being), onto the same decision making “table” as commercial uses (such as port development, aggregate extraction and commercial fisheries).

However, beyond raising awareness, at present there is little evidence of valuation being utilised in the policy environment. Very few valuations have had a demonstrable policy influence, and the reasons for this need to be addressed, ideally before further valuation work is undertaken. For example, it is advised that future valuation work is undertaken in response to clear policy questions and in close collaboration with the policy makers. The methodology used should be carefully chosen to address the policy question, and to provide results in a format that are readily applicable by the policy maker. Environmental valuation cannot provide results which are 100% accurate, but this level of accuracy is not essential provided the users understand the uncertainties and limitations associated with the data. Communicating this uncertainty, what the values actually mean, and how they should be applied, should be a key part of in the development of any valuation exercise and this is a key recommendation for future research.
Objective 3: Explore how current institutional frameworks and policy are linked to the delivery of a range of ecosystems services by seagrass ecosystems

Olivia Langmead, Tavis Potts, Caroline Hattam & Emma Jackson

Identifying stakeholders and institutional arrangements

The aim of this task is to build on objective 1 and to identify and analyse institutional arrangements at various scales related to seagrass beds, to assess which institutions are critical in the exploitation of ecosystem services provided by seagrass beds and in managing these ecosystems in terms of anthropogenic pressures.

Approach

**Rapid Policy Network Mapping**

The Rapid Policy Network Mapping (RPNM) approach is based on the technique developed by Bainbridge (2011). The method is based on the Ecosystem Approach, from the perspective that existing policy making institutions must be able to accommodate and adapt to a new multi-sectoral approach. Understanding how existing institutional structures function is an important first step towards this adaptation. RPNM generates a and visual output contributing to understanding of the relevant institutions in the policy development process by stakeholders and builds dialogue over institutional process and reform.

RPNM uses a structured approach to mapping relevant primary and secondary legislation and mapping key institutions and stakeholders. The approach begins by analysing the documents of a single organisation, and follows a chain of references from this point. This method adopts a snowball approach, where a policy actor or instrument linked to other relevant policy actors and instruments in a policy community and where the ‘centrality’ of the instrument or actor is a function of its importance within that network.

Policy instruments, reports, planning documents, organisation websites and policy statements pertaining to seagrass conservation and management in the two case studies were analysed and the relationships and dependencies of policy actors and instruments were simultaneously identified, categorised and recorded. The study began with a single policy actor and based on referrals from this source, information on linked, related or dependent policy actors and instruments was gathered, but only if referenced in the context of the policy under investigation. When referrals ceased to reveal new actors or instruments related to the relevant directive the process was terminated.

Policy actor and instrument data was collated in Microsoft Excel (available on request). It was necessary to develop mapping templates for the actor and instrument policy communities to ensure consistency of reporting. A series of visual templates were created using CmapTools software (http://cmap.ihmc.us/). Templates and example of RPNM are also available from the Knowseas project website.¹

Both actor and instrument maps use the scale of international (treaties), European (Directives Communications, Policy); UK (Acts, Regs, Policy) UK Devolved and Regional (Regs, Policies) and local (Orders, Plans, Policy) to map out relevant issues and benchmarked against relevance to seagrass conservation and management. In the addition, the actor map uses the definitions in Table 8 to categorise key actors in the policy process.

<table>
<thead>
<tr>
<th>Influencer (I)</th>
<th>Organisation legally, morally or practically required, invited or involved in official policy development process. Influencers affect the outcome of the process using legitimate means based on opinions and views. E.g. Crown Estate, RSPB, NFU, WWF</th>
</tr>
</thead>
<tbody>
<tr>
<td>Owner Decision maker (ODM)</td>
<td>An organisation, entity or individual which has the authority to make a decision which can affect the outcome or which owns all, or parts, of the policy development process. Decisions may be made by Owner/Decision Maker’s following consultation and/or negotiation they have the ultimate authority to decide outcomes. E.g. Local authorities, central authorities e.g. MMO</td>
</tr>
<tr>
<td>Influencer / Deliverer (ID)</td>
<td>An organisation, entity or individual which is legally or practically required, invited or obliged to be involved in the policy process. They can affect the outcome of the process using legitimate channels based on opinions &amp; views. Are engaged in delivering a strategic action, process, or report which facilitates the transposition and/or implementation of the policy. e.g. IFGs, SEPA, Natural England</td>
</tr>
<tr>
<td>Deliverer (D)</td>
<td>Can affect the outcome of the policy process based on their delivery of operational actions, processes or reporting which facilitate the interpretation, transposition and/or implementation of the policy. They cannot, in principle, affect the outcome of the policy process based on their opinions and views. Focus on operations or support at ‘ground level’. e.g. NERC, MCCIP, SFPA</td>
</tr>
</tbody>
</table>

**High level policy**

The high level (international to national) instruments were identified according to the steps below:

1) Identifying conservation legislation relevant directly to the seagrass habitat or protects key species that use the habitat (thus giving the habitat de facto protection);
2) Identifying activities leading to pressures on seagrass habitats (using the outputs of Task 1.1);
3) Searching for legislation at all levels from international to national (acts -> regulations -> directions and guidance) that regulate the aspects of the activities that generate specific pressures identified (e.g. not all agriculture legislation is included - only aspects that control nutrient pollution and organic material, not agrochemicals since there is currently no evidence that these are a pressure on seagrass systems);
4) Where possible splitting out the different legislation for Scotland and Wales.
Case studies overview

The pattern of institutional arrangements at the local scale is very specific to the individual location, its conservation designations and the sectoral activities that occur there. Two contrasting case studies were undertaken to highlight differences in the use and management of seagrass habitats, both located in SW England: Tamar estuaries complex and Studland Bay.

**Tamar Estuaries complex**

The Tamar Estuaries has four main areas of eelgrass bed: Cellars Cove at the mouth of the Yealm estuary, Red Cove on both the north and south shores of the lower Yealm, to the north of Drake's Island and in Cawsand Bay to the west of Plymouth Sound, see Figure 6 (Irving 2010). There are further seasonal or transient beds off Jenny Cliff Beach, Firestone Bay and at the entrance to St John's Lake on the Tamar estuary. The seagrass habitats in the Tamar Estuaries fall within a European Marine Site which combines both the Plymouth Sound and Tamar Estuaries Special Area of Conservation (designated under the EC Habitats Directive 92/43/EEC) and a Special Protected Area (designated under the EC Bird Directive 79/409/EEC). The main reason for the designation of this area as an SAC is the presence of sandbanks that are slightly covered by seawater all the time (an Annex 1 feature as described in the 1992 EC Habitats Directive). Eel grass beds (*Zostera marina*) are one of the sub-features on these shallow sandbanks.

Other relevant designations in the area include a SSSI in the mouth of the Yealm, and the recommendation that the European Marine Site form part of an extended Marine Conservation Zone under the Marine and Coastal Access Act 2009.

![Figure 6 Location of the four main seagrass beds in the Tamar Estuaries complex (after Irving 2010).](image-url)
The European Marine Site is heavily used by people for commercial and recreational activities; it is bordered by the city of Plymouth, an active military base and dockyard, and a number of harbours and marinas. Many of the activities undertaken within the European marine Site therefore have the potential to affect the quality and status of the site, and the eelgrass beds. These activities include dredging, anchoring and mooring, bait collection, shore crab fishery and other fisheries, angling and coastal development (identified from the EMS risk assessment guidance Annex 3 Plymouth conducted by Natural England). In addition, activities further upstream, such as agriculture, may influence the quality of the site as they contribute to sedimentation and diffuse pollution within the waters entering the estuaries. Toxic contamination due to disturbance of contaminated sediments and invasive species are also of concern.

Management of these competing activities and interested within the European Marine Site is facilitated by the Tamar Estuaries Consultative Forum (TCEF). TCEF has no statutory power, but is responsible for the development of a management plan for the European Marine Site. It also agrees management actions among the different stakeholder groups and identifies who is responsible for each action.

**Studland Bay**

Studland Bay is shallow and well protected from the prevailing south-west winds, making it an ideal habitat for the seagrass, *Zostera marina* (Figure 7). It is also a popular anchorage for recreational vessels, due to its shelter and proximity to the port of Poole. The seagrass ecosystem in Studland Bay is also important due to its use as a nursery ground by the endangered undulate ray (*Raja undulata*) and as a breeding site by both British seahorse species; the spiny seahorse (*Hippocampus guttulatus*) and the short-snouted seahorse (*Hippocampus hippocampus*) (Dorset Wildlife Trust 2011).

Studland Bay has no statutory conservation designations that extend into the subtidal thus does not encompass the seagrass habitat. There are two SSSIs that cover adjacent coastal areas: 1) Studland and Godlinston Heaths (designated features include sand dunes and South Haven Peninsula) and 2) Studland Cliffs (designated for chalk cliffs, platforms and associated beaches). However, Studland Bay has numerous records of seahorses inhabiting the seagrass meadow. Both UK species of seahorses (*Hippocampus guttulatus* and *H. hippocampus*) were added to the list of protected species under the Wildlife and Countryside Act 1981. This means that both they and their place of shelter (the seagrass habitat) became legally protected. In addition in September 2011, Studland Bay was recommended to be designated as a Marine Conservation Zone for a variety of seabed and shoreline habitats including seagrass meadows and species of conservation importance: short-snouted seahorse, undulate ray and native oyster.
The main activities on going in Studland Bay are outlined in Table 3.3; these are centred around fisheries and recreational vessel mooring and anchorage.

**Preliminary results**

**High level policy matrix**

It is clear that there is a great deal of legislation that has the potential to manage human pressures on seagrass habitats. In terms of conservation policies, apart from the OSPAR and Berne Conventions (Habitat Action Plan, HAP, Tyler-Walters, 2007), the level of protection for seagrass habitat depends on the individual location having a conservation designation, either a RAMSAR site, a European Marine Site or in future, a Marine Conservation Zone (MCZ) in England or equivalent national marine protected area (Scotland, Wales). European Marine sites comprise Special Area of Conservation (SACs) under the Habitats Directive and Special Protection Area (SPAs) under the Birds Directive. Eelgrass communities are included in the following habitat types within SACs:

1110 Sandbanks which are slightly covered by seawater all the time;
1130 Estuaries;
1140 Mudflats and sandflats not covered by seawater at low tide;
1150 Coastal lagoons; and
1160 Large shallow inlets and bays.
In addition seagrass habitats gain protection in SPAs where another species that uses the habitat, such as brent geese). The individual designations for the two case study areas are given in Table 9 Statutory and possible future designations in each of the case study areas.

The main policies that regulate activities which have the potential to exert pressures on seagrass beds fall into several broad themes or sectors: fisheries and aquaculture, aggregates, ports and harbours, coastal defence, planning, agriculture, land use management, industry and household/urban (Annex 1, a-c). For most of these areas there are European policies and directives that are transposed into UK law (some with regional differences with the devolved authorities of Scotland and Wales producing their own acts, regulations, order and guidance, especially since 2010), while others are covered by wider planning legislation. The legislation at European level comprises both sectoral policies (e.g. Common Agricultural Policy, Common Fisheries Policy) and integrated environmental directives (Water Framework Directive, Marine Strategy Framework Directive).

**Case study characterisation**

The two case studies differ in terms of their conservation and other designations (Table 8) and activities that have the potential to cause pressures on seagrass habitats (Table 10).

<table>
<thead>
<tr>
<th>Designations</th>
<th>Studland</th>
<th>Tamar</th>
</tr>
</thead>
<tbody>
<tr>
<td>Conservation</td>
<td>SAC</td>
<td>✓</td>
</tr>
<tr>
<td></td>
<td>rMCZ</td>
<td>✓</td>
</tr>
<tr>
<td></td>
<td>RAMSAR</td>
<td>✓</td>
</tr>
<tr>
<td></td>
<td>SSSI</td>
<td>✓</td>
</tr>
<tr>
<td>Agriculture</td>
<td>NVZ</td>
<td>✓</td>
</tr>
<tr>
<td>Ports &amp; Harbours</td>
<td>Jurisdiction area</td>
<td>✓</td>
</tr>
</tbody>
</table>
Table 10 Activities that have potential to impact seagrass habitat ongoing in each case study area

<table>
<thead>
<tr>
<th>Category</th>
<th>Activity</th>
<th>Tamar Estuaries</th>
<th>Studland Bay</th>
</tr>
</thead>
<tbody>
<tr>
<td>Towed Gear</td>
<td>Beam trawling</td>
<td><img src="%E2%9C%93" alt=" " /></td>
<td><img src="%E2%9C%93" alt=" " /></td>
</tr>
<tr>
<td>Towed Gear</td>
<td>Otter Trawl¹</td>
<td><img src="%E2%9C%93" alt=" " /></td>
<td></td>
</tr>
<tr>
<td>Towed Gear</td>
<td>Scallop Dredging¹</td>
<td><img src="%E2%9C%93" alt=" " /></td>
<td></td>
</tr>
<tr>
<td>Manual Collection</td>
<td>Bait Digging / Collection</td>
<td><img src="%E2%9C%93" alt=" " /></td>
<td></td>
</tr>
<tr>
<td>Manual Collection</td>
<td>Shore Crab Fishery - Peeler crab collection in estuary</td>
<td><img src="%E2%9C%93" alt=" " /></td>
<td></td>
</tr>
<tr>
<td>Static gear</td>
<td>Crab/lobster pots²</td>
<td><img src="%E2%9C%93" alt=" " /></td>
<td></td>
</tr>
<tr>
<td>Shellfisheries</td>
<td>Cultivation of Mussels - Mussel farm (racks)</td>
<td><img src="%E2%9C%93" alt=" " /></td>
<td><img src="%E2%9C%93" alt=" " /></td>
</tr>
<tr>
<td>Shellfisheries</td>
<td>Cultivation of Scallops - hanging lantern nets (Small scale trial)</td>
<td><img src="%E2%9C%93" alt=" " /></td>
<td><img src="%E2%9C%93" alt=" " /></td>
</tr>
<tr>
<td>Toxic Contamination</td>
<td>Historic, persistent banned toxic synthetic compounds e.g. TBT &amp; PCBs.</td>
<td><img src="%E2%9C%93" alt=" " /></td>
<td><img src="%E2%9C%93" alt=" " /></td>
</tr>
<tr>
<td>Non-toxic Contamination</td>
<td>Discharges from Industry STWs &amp; Road run-off</td>
<td><img src="%E2%9C%93" alt=" " /></td>
<td><img src="%E2%9C%93" alt=" " /></td>
</tr>
<tr>
<td>Non-toxic &amp; Toxic Contamination - Diffuse Inputs</td>
<td>Agricultural Run-off</td>
<td><img src="%E2%9C%93" alt=" " /></td>
<td><img src="%E2%9C%93" alt=" " /></td>
</tr>
<tr>
<td>Shipping waste control</td>
<td>Control of sewage and garbage</td>
<td><img src="%E2%9C%93" alt=" " /></td>
<td><img src="%E2%9C%93" alt=" " /></td>
</tr>
<tr>
<td>Shipping waste control</td>
<td>Ballast Water Discharge (Alien Spp.)</td>
<td><img src="%E2%9C%93" alt=" " /></td>
<td><img src="%E2%9C%93" alt=" " /></td>
</tr>
<tr>
<td>Boating</td>
<td>Water Sports - sailing vessels (yachts, dinghies, wind surfing etc)</td>
<td><img src="%E2%9C%93" alt=" " /></td>
<td><img src="%E2%9C%93" alt=" " /></td>
</tr>
<tr>
<td>Licensed &amp; unlicensed dredging</td>
<td>Maintenance Dredging - inc extraction and agitation, water injection, plough dredging</td>
<td><img src="%E2%9C%93" alt=" " /></td>
<td><img src="%E2%9C%93" alt=" " /></td>
</tr>
<tr>
<td>Licensed dredging</td>
<td>Capital Dredging</td>
<td><img src="%E2%9C%93" alt=" " /></td>
<td><img src="%E2%9C%93" alt=" " /></td>
</tr>
<tr>
<td>Development Pressures / Land Claim</td>
<td>Major coastal development (industrial and residential) that would require EIA or AA.</td>
<td><img src="%E2%9C%93" alt=" " /></td>
<td><img src="%E2%9C%93" alt=" " /></td>
</tr>
<tr>
<td>Development Pressures / Land Claim</td>
<td>Small-scale coastal development (industrial and residential) i.e. cumulative impacts of.</td>
<td><img src="%E2%9C%93" alt=" " /></td>
<td><img src="%E2%9C%93" alt=" " /></td>
</tr>
</tbody>
</table>

¹ Trawling is only permitted outside of the breakwater, so would not affect most of the seagrass beds in the Tamar Estuaries complex, only the meadow at Cawsand and around the mouth of the Yealm.

² Dockyard Port Order prohibits this activity through much of the EMS.

**Tamar estuaries actor and instrument map**

The Tamar actor map reveals a complex array of actors at the local scale (Figure 8). This actor complexity relates directly to the diverse number of users, regulators and interest
groups in the Tamar and Plymouth Sound. Overall 36 actors are identified in influencing, owning the policy process or delivering policy outputs or services. Key users include the port authorities (e.g. Harbour Master and Ministry of Defence), UK government environmental regulators (Natural England and Environment Agency), several municipal authorities (Plymouth City Council, Devon County Council), and landscape or catchment groups. Identified issues include:

- Several conservation designations and associated management groups exist in the Tamar and Plymouth Sound region (e.g. SAC, SPA, SSSI and AONB). The presence of the SAC designation and the associated management group, the Tamar Estuaries Consultative Forum, allows for a degree of focus and centrality in decision making, including a means of managing the primary pressures upon seagrass ecosystem services. With a high number of users and stakeholders present in the policy cycle, investment is needed in mechanisms for coordination and delivery;
- While TECF provides a centre point and a coordination role, there are many actors with different roles in the policy process. This is part stems from the use of the area as a commercial port and the activity generated by a large local population. Not all flows of information, resources and access are equal, further exploration is required to identify the mechanisms that drive effective engagement;
- Five local authorities, 3 UK departments, and 3 English agencies are involved as owners or in an influencer / delivery role. The extent of coordination between local institutions and national institutions is presently unknown but potentially important for management and delivery of ecosystem services. The role of the TECF is therefore critical.
- Linking of actors and instruments (Annex 2) further up the catchment will improve management of pressures on seagrass. The emerging Tamar Catchment initiative should be integrated into existing groups, and the links between catchment activities e.g. nitrates and seagrass health elaborated.
- Operational authorities (e.g. harbour authorities, SW Water) are integrated into the policy system for seagrass management (i.e. TECF). Further investigation is needed to determine the extent that port authorities recognise and act upon the pressures on seagrass, and incorporate ecosystem services into existing regulatory functions.

**Studland Bay actor and instrument map**

The Studland Bay seagrass (and seahorse) issue is dominated by local and national ‘influencers’, i.e., predominantly community and non-government interest groups that do not deliver operational policy functions Figure 9. The Studland Bay management approach is more linear than the Tamar case study, and is underwritten by the decisions and actions of the Crown Estate and the Marine Management Organisation. No conservation designations exist on the site, but the site has been preliminary selected as an MCZ under the Marine and Coastal Access Act 2009 (Annex 3). In addition, the protection of seahorses and associated habitat under the Wildlife and Countryside Act 1981 and the listing of seagrass habitat under the UK Biodiversity Action Plan ensure there is a policy driver to initiate management action – in this case limiting recreational mooring and anchoring impacts upon seagrass habitat. Key features of the actor network include;
• Prevalence of national and local environmental and recreational stakeholder groups in the local policy cycle. Clearly the issue is of considerable importance to stakeholders at the local and UK scales. An international NGO (Project Seahorse) is also engaged. A policy solution (and potential MPA) will be required to engage and work with community interests across a number of sectors.

• The MMO is the primary decision maker in this case study. Key influencer / delivers are Natural England and the Crown Estate with the former responsible for wildlife management and protection and the later the owner of the sea floor and licensee for moorings. In comparison to the Tamar, Studland Bay has a linear and simpler policy environment for identifying a solution to manage seagrass beds (e.g. MCZ, voluntary agreement, new moorings regime). This will be linked to the management of activities under the potential designation under the Marine and Coastal Access Act 2009.

• Studland Bay is geographically bound, there is no identifiable catchment pressure and hence no link to WFD or catchment planning.
Figure 8 Tamar and Plymouth Sound Actor Map (RPNM).
Figure 9 Studland Bay Actor Map (RPNM)
Future work

This scoping report provides a foundation for the analysis of how current institutional frameworks and policy relate to the management of seagrass habitats. If the project were funded in its entirety, this work would be followed up by ground-truthing the relationships and policy flows by taking the matrices back to key parties in both case study areas (e.g. Tamar Estuaries Consultative Forum, Natural England in the Tamar Estuaries Complex and Marine Management Organisation and The Crown Estate for Studland Bay case study). This would ensure that no policy instruments or players have been missed and the linkages that we have mapped represent the current situation. Then, the validated maps would be elaborated to obtain a much more detailed understanding of the interactions between the governance system and the ecosystem.

The next step would examine how the objectives of different stakeholders interact in the contrasting case study areas, and how issues such as power balances affect these interactions. This will enable us to understand the opportunities presented by the existing frameworks, or combinations of frameworks, the overlaps between frameworks and any gaps in existing policy. Ultimately this would facilitate the delivery of a more integrated management approach to identify ways of improving the delivery of seagrass ecosystem services. Specific elements of future work for Objective 3 include;

- Refine the link between actors and their use of instruments at their disposal to protect and manage seagrass habitats based on specific identified drivers and pressures.
- Deliver a survey to each of the identified stakeholders from international to national scales that asks questions about connections, use of policy instruments, resources, and approaches to seagrass conservation and management of services.
- Based on survey and mapping results undertake a social network mapping exercise that identifies the key relationships, power structures and central players, but also indicates the existing barriers to improving current management practices.

A workshop would be held at each site to ground truth the actor / instrument / SNA maps and collaboratively develop and explore scenarios built around DPSIR criteria and policy solutions to deliver integrated management of seagrass beds.
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Appendix 1

ENVIRONMENTAL VALUATION: A general overview

TEV vs. Marginal
Stock vs. Flow
Use vs. Non-use
Methodology overview, inc. BT – how and why

The general finding of the literature searching seems to be that there has been very little work aiming at a quantitative analysis of seagrass. However there are indications in the literature of the types of analysis which might be expected to be useful in this context, and this paper sets out to review some of these approaches, and then to flag up some points for discussion regarding their possible application to seagrass beds.

There is a substantial body of literature within the neoclassical economics paradigm relating to the measurement of the value of non-marketed goods and services. Within this paradigm, free markets are seen as one possible way of achieving (Pareto) efficient production and distribution of goods and services, but with market failures of various sorts preventing the realisation of this efficiency. Externality, in which the non-marketed side-effects of some agent's activities impacts on some other agent's welfare, is one such failure. Solutions generally involve clarifying property rights (defining who is allowed to do what with certain resources) and may further involve an element of pricing, essentially creating a market where previously none existed. Informing this pricing process is one rationale for environmental valuation. Another is as a key input to cost-benefit analysis, the neoclassical project appraisal tool.

Price and value are not the same. In neoclassical economics, estimating the full value of a good requires an understanding of the whole demand curve, not simply a point measurement of price and quantity. This is because value is assumed to derive from the preferences of individuals. Although often, for practical purposes, households are used as the basic unit of analysis, the principle remains the same. The demand curve shows the locus of individuals' (or households') willingness to pay (WTP) values, and it is the aggregation of these, net of the costs of production, which shows the true value of the good or service.

In economic theory, WTP can exist for anything which enters an individual's utility function. For example, an individual might be willing to pay something simply to preserve some environmental resource or rare species, for example, independent of any possible use that person might make of the resource. Therefore value is not limited to direct use of a good or service, but can exist for a range of non-use categories, such as the value of preserving the option of future use, and values not associated with personal use (existence and bequest values). The composite of all values is referred to as "total economic value" (TEV).

The "ideal" goal of environmental valuation, within this framework, is to estimate the demand curve for environmental goods and services, for non-use as well as use, thereby allowing calculation of TEV. There are a number of methods of achieving this, although only stated preference techniques are able to capture non-use values. This problem is
particularly important for significant environmental goods for which non-use values could be substantial. There are also several methods which do not achieve demand curve estimation, but aim rather to develop proxy costs or prices for environmental goods. Although these are not values in the economic meaning of the word, the figures may nonetheless be of use to decision makers.

So a variety of techniques may be employed, some consistent with the theoretical underpinnings of welfare economics, and others on less “solid” foundations. But although lip service is sometimes paid to the philosophical problems inherent in interpersonal comparisons of utility, and beyond that to the importance of distribution or equity to most conceivable formulations of a social welfare function, as a rule, discounted monetary values are simply summed in CBA. It might therefore be argued that the theoretical justification for any particular valuation technique is of secondary importance, given that the subsequent use of the figures is itself divorced from the theory, to a clear understanding of the quantities which we are attempting to value, coupled with a robust method for obtaining approximate estimates of these quantities. Besides, all but the most gung-ho neoclassicists would agree that CBA should be seen as an aid to decision making, rather than as a means of generating a final decision.

I shall return to these points later, after reviewing the techniques available: the review is brief and does not go into great detail on many of the possible biases or statistical problems with the methods. The techniques consistent with welfare economics split into two main types: stated and revealed preference techniques. Stated preference involves the use of questionnaires to elicit valuations for non-marketed goods, while revealed preference involves the deduction of these values from actual behaviour in associated markets. Revealed preference techniques rely on observations of actual market behaviour to infer environmental valuations.

**Stated Preference**

Stated preference (SP) valuation techniques use questionnaires to discover individuals’ preferences for specific changes to the environment. SP techniques are widely applicable and, unlike revealed preference methods, are in theory capable of estimating non-use values of environmental resources.

There are two main forms of SP: contingent valuation (CV) and conjoint analysis (CA). CV focused on direct pricing of a specific change in quantity or quality of an environmental good, whereas CA looks at choices (usually pairwise) between different “bundles” of characteristics defining provision of an environmental good. By assessing the trade-offs between price and environmental aspects, the CA leads to estimates of environmental values.

The success of the survey instrument is jointly dependent on ability of researchers to communicate often complex issues to respondents and on the ability of respondents to process that information within an economic context. Thought CV is "simpler" to use, some authors have argued that CA is preferable because "choice" comes more naturally to respondents than "pricing": most people are not used to thinking about the financial
value they would place on environmental goods and services, but do regularly make choices between goods or courses of action which combine financial and environmental implications.

Stated preference techniques require large samples and careful testing of the survey questionnaire, making them expensive and time consuming. It is also essential that respondents understand fully the good or service they are being asked to value, and this makes direct application to complex effects, such as air pollution from road transport, rather difficult.

In such cases, dose-response relationships can be very useful, reducing a single complicated effect to several more tightly defined categories which are easier to value. Dose-response relationships linking different air pollutants to health effects, such as increased risk of pulmonary illness and cardiac arrest, can be combined with stated preference estimates of peoples’ willingness to pay to avoid each category of risk. Similarly, the effects of air pollution on the structure of buildings or on crops becomes relatively easier to quantify.

Contingent valuation

A CV survey aims to derive a monetary expression of individuals' preferences for changes in quantity or quality of a non-marketed good or service. This is achieved through a hypothetical (contingent) "market", defining the change in provision of the good or service, the means by which this change would come about, and the instrument by which the change would be financed.

The survey aims either to elicit willingness to pay (WTP) to secure a gain or avoid a loss, or willingness to accept (WTA) compensation for a loss or to forego a gain. A number of formats can be used, including:

- open-ended "how much would you pay?" questions
- selection from a list of possible values (with a number of possible presentation styles);
- a single yes-no question (are you WTP £X?), known in the literature as "dichotomous choice";
- two or more sequential yes-no questions.

Econometric analysis is applied to the survey results, using a range of information to explain the responses in a "bid function", which may include, for example, age, sex, income, experience of the good/service, membership of environmental groups and so on. From the results, it is possible to derive mean or median values for a given population, and thence to calculate aggregate values for a given change in environmental services for that population.

Conjoint analysis
CA is based on the theory that a good or service can be described by its component characteristics; the same idea underlies the hedonic pricing method of revealed preference. Rather than asking a pricing question, CA asks respondents to choose between or rank two or more options, each of which is described by different "levels" of a small number of "attributes" (using too many would result in overloading respondents). The attributes are chosen to reflect the relevant changes in provision of the good or service in terms of quality, quantity and price variables. A number of levels are selected for each attribute, representing a realistic range of possible outcomes, including the status quo option. Clearly the number of possible comparisons increases rapidly as more levels or attributes are added. Use of specific forms of design for the statistical estimation can dramatically reduce the number of comparisons actually required: at the simplest level, there is no need to compare options where one clearly dominates the other; beyond that it is possible to reduce much further by imposing assumptions on the types of interrelationships possible between the attributes in the underlying utility function, for example restricting the function to linearity (an "orthogonal main effects" design).

A similar procedure can be followed with ratings of different options (on a Likert scale or similar) or expressing strength of preference between two options. While this may seem to give more information than a simple binary statement of preference, it is difficult to interpret the additional information within the framework of welfare economics, and there is therefore a degree of subjectivity in using the results to derive economic valuation estimates.

To somewhat varying degrees, all forms of the stated preference techniques require large samples and careful testing of the survey questionnaire, making them expensive and time consuming. It is also essential that respondents understand fully the good or service they are being asked to value, and this makes direct application to complex effects, such as air pollution from road transport, rather difficult. In such cases, dose-response relationships can be very useful, reducing a single complicated effect to several more tightly defined categories which are easier to value. Dose-response relationships linking different air pollutants to health effects, such as increased risk of pulmonary illness and cardiac arrest, can be combined with stated preference estimates of peoples’ willingness to pay to avoid each category of risk. Similarly, the effects of air pollution on the structure of buildings or on crops becomes relatively easier to quantify.

**Revealed Preference**

Revealed preference (RP) techniques rely on actual market data to estimate values attached to some environmental (or other - e.g. safety) attributes of a marketed good which are not themselves directly tradable in markets. RP includes hedonic pricing, travel cost and averting behaviour techniques.

Because the techniques focus on actual market expenditures, only use values can be estimated. For example, while a consumer’s choice of house may reveal preferences about the cleanliness of the air around the consumer’s home, nothing can be revealed about any non-use values, for example altruistic or existence values for clean air, since
the choice of house does not impact on these values. Therefore RP estimates might be considered a lower bound on total economic value for a good or service. Again, this may be of lesser concern in the current context, where we are told that commercial effects are the primary interest.

**Hedonic pricing**

HP is based on the idea that a good or service may be described by its component characteristics, and that in effect it is these characteristics which are traded in the market. The most common applications have been to houses, cars, computers, and the labour market. A price function is estimated from observations of market transactions, which links prices to composite bundles of characteristics. In an environmental context, the housing market is the key application, with prices or rents modelled as a function of structural, accessibility, neighbourhood and environmental characteristics of properties.

This allows estimation of WTP values for environmental quality. By assuming that individuals (households) are equating marginal WTP for each housing characteristic with the marginal implicit price of that characteristic in the hedonic price function, it is possible to estimate the demand curve for the environmental characteristic, provided that the implicit (marginal) price varies for different levels of the characteristic.

Often, the implicit price function is interpreted as the demand curve, which assumes incomes and preferences are identical for all individuals. This is unrealistic and will lead to errors in estimating the value of significant changes. Ideally therefore it is necessary to go a stage further, estimating the demand curve by regressing the implicit prices on observations of socio-economic characteristics and levels of the environmental variable. This assumes a fixed supply of housing stock, which is rather more defensible, but still not ideal. But allowing for variations in housing stock would necessitate simultaneous estimation of demand and supply equations, a complication which may not be justified, given the other possible sources of error in the estimation process.

The hedonic method can only work to the extent that consumers in the market are aware of the environmental characteristic, and understand its true implications for their welfare. If, for example, people are not aware that noise can disturb their sleep without waking them, leading to tiredness and so on, that part of noise nuisance can not be included in the values estimated. Equally, the method is dependent on the existence of a range of choices. And the theoretical model underpinning the method assumes that it is possible to change house without cost – clearly this is not true, and this is a particular problem when trying to evaluate substantial changes in environmental conditions, since it may take the market a considerable time to adjust.

A further problem in looking at prices (rather than rental values) is that this does not measure WTP per unit time. Fortunately, in the current context, we might be most likely to look at rental values for commercial property, avoiding this problem.

**Travel cost method**
TC aims to derive values for recreation at significant sites such as national parks, forests, beaches and so on. The costs of travel to a site, including actual expenditure and an allowance for time, are interpreted as the cost of visiting the site. When coupled with information about the population numbers experiencing different travel cost levels, and actual visitation rates, this allows estimation of a demand curve for visits to the site. From this demand curve, the use value of the site can be estimated.

It is also possible, by observing changes in visit numbers when environmental quality changes (or by surveying how consumers say visits would change), to estimate the value associated with a change in environmental quality.

The TC method is highly dependent on the value assumed for travel time, and there is no real agreement on what values should be used. Commonly, some fraction of the wage rate is adopted, but evidence does not support a simple linear relationship. This problem has not been adequately resolved.

A problem for TC is the existence of alternative sites. Random utility modelling can be used to take account of several different sites with different characteristics, including travel cost. The probability that an individual will visit one particular site, rather than any other, is estimated as a function of these characteristics. This is similar to, and relies on the same theory as, conjoint analysis, but with observed rather than stated data.

**Production Function Approaches**

A number of techniques link environmental quality changes to changes in production relationships. Production functions can be defined for firms (producing goods and services) or for consumers (producing services that generate positive utility). In either case, environmental quality can be represented by one or more variables in the production function. The value of a change in the environmental variable can then be estimated via the relevant derivative of the production function.

*Dose-response approach*

Dose-response functions are one example of this approach, where the "production" of some end-point is described in a function containing the environmental variable of interest alongside population descriptors, weather variables and so on. The endpoint may be a marketed good (e.g. agricultural crops) or a non-marketed good which is easier to "value" than the raw environmental indicator (e.g. health endpoints). This can be done either using observations "in the wild" or via controlled experiments. The dose-response function produces a physical measure of damage, which must then be valued. The simplest approach is to use a current market price, but strictly speaking this is appropriate only for small changes in quality, since more substantial changes would lead to changes in price and resource use via the operation of the market. An alternative approach is to use an optimisation model, maximising profits or minimising costs subject to constraints including resource availability and the dose-response function. This is based on a theoretical understanding of what "should" happen under assumptions about
market performance and rational decision making. The "reverse" of this is to use econometric models based on actual historical data. This cannot be used where changes in technology or institutions are important, and data requirements are high. In all applications of dose-response, choice of functional form(s) is important.

Averting expenditure

The averting expenditure approach uses observations on individuals' (households') expenditures made to avoid disutility arising from a particular environmental effect. If environmental quality and averting expenditures are perfect substitutes, then the averting expenditures can be directly interpreted as a welfare measure. However in general this is not the case: both because households are not able to purchase any given desired level of environmental quality (for example, you can't fit "1.4" glazing, only single or double; and that won't reduce noise in your garden) and because there may be additional benefits or costs (double glazing also cuts heating bills; cycling masks cut pollution exposure, but are also unpleasant to wear) not counted in the actual expenditure. Similarly, the costs of time and effort in making averting expenditures, or in modifying behaviour in other ways to avoid costs, are generally overlooked. And, as in the hedonic case, the method can only work to the extent that consumers are aware of the effects, and of the options for abating them.

Cost-based techniques

A number of techniques focus on the costs of doing something as proxies for values. These do not deliver "values" in the neoclassical economic sense, and there is an inevitable circularity or tautology involved in using a cost as an estimate of value. But they may nonetheless be useful in practical applications.

Opportunity costs

Economic theory holds that the true cost of a resource is its value in its next best use, that is, what you have to give up in order to use the resource for a specific end. So the OC method looks at estimates of what must be given up in order to improve or maintain some non-marketed resource. For example, if considering pedestrianisation of a street, we might estimate the time and other costs to drivers no longer able to use the route.

However, as the name suggests, this method relates to the costs of providing a given (change in) service, and not directly to its value. While one might wish to argue that, because a certain change has been provided, it must be worth at least what was given up in order to achieve it, the argument is circular in so far as making use of this "value" for decision making purposes: "X is what must be given up to provide Y, therefore Y is worth at least X" is of no help whatsoever to the decision maker faced with the choice between X and Y.

On the other hand, "X is what was given up to provide Y, therefore Y is worth at least X" might be of use to a decision maker faced with the choice of providing another example of Y somewhere else - if it can be argued that the Ys are sufficiently similar, and that the
former decision was "correct" (or, at the very least, that consistency is important). The process of benefits transfer is discussed below.

Cost of alternatives

A similar method is the "cost of alternatives" approach, which looks at the cost of using some other resource to achieve the same end as the one in question. This suffers from the same kind of theoretical shortcoming as identified above. For example, the cost of a tunnel to take traffic below the city centre does not really provide a value for traffic nuisance in the city centre - rather, it reveals what that value would have to be for it to be worthwhile constructing the tunnel. Again, though, observing a decision to build a tunnel (or not) in one area might allow conclusions to be drawn about the value of traffic nuisance implicit in the decision to build the tunnel (or not), assuming that decision was "correct", thereby providing a lower (upper) bound on the damage of traffic nuisance. A similar stance is taken where observations of actual government payments for environmental improvements are interpreted as indicators of values.

Shadow project

This method looks at the costs of replacing an environmental resource with an equivalent alternative elsewhere. Again, this is not really a value estimate, but a cost. There may not be any particular problem provided the shadow project is actually carried out, and is really the equivalent of the resource destroyed (although this does beg the question of what has to be destroyed to allow the shadow project to take place). But when this is not the case the technique can be quite "dangerous". An extreme example, to illustrate the point, would be using an estimate of the cost of conserving genetic material in a seed bank (probably very low) as an estimate of the damage caused by making a plant species extinct in the wild (possibly very high), if the material is not actually stored in the seed bank.

Benefits Transfer

Benefits transfer is the term given to methods of applying valuation studies carried out for one place / time / population / good to another setting. This may be simply to get a handle on which types of environmental impact are likely to be significant, and therefore be candidates for further exploration. Or it may be for direct use in a cost-benefit framework. Because original studies are expensive, benefits transfer is attractive, in particular if dealing with a relatively small project where the potential costs and benefits are such that the costs of a full-scale valuation study would not be warranted.

Benefits transfers are at best only as accurate as the original valuation study/studies. So the quality of the data, and the appropriateness of the methodology, of the original studies must be assessed. Weighting procedures could be adapted, perhaps within a Bayesian framework, to take account of subjective judgements as to the "accuracy" of
individual studies (for example, giving more weight to stated preference studies with larger sample sizes).

Simply taking a value estimate for a "source" site and applying it, perhaps with adjustment, to the new site is the most common approach. It is simple and cheap, but unless it can be convincingly argued that the situation is "identical" to that in some other study, it is preferable to take account of differences among the goods, populations and the contexts within which valuations are carried out. Possible differences include:

- physical characteristics of the study and policy site
- the environmental change under consideration
- property rights (WTP or WTA context)
- market conditions such as the availability of substitutes
- socio-economic characteristics of the relevant populations;

Not all adjustment based on differences from a single study will be ad hoc: some changes can be based on theory, augmented with observations from other studies, or on other systematic grounds.

One change, commonly made, is the correction for income levels using an assumed income elasticity for environmental values, $\eta$.

$$\text{WTP}_N = \text{WTP}_S \left(\frac{Y_N}{Y_S}\right)^\eta$$

where $Y$ is income per capita, $N$ the new site and $S$ the source site. If transferring between different countries, income is generally converted not at market exchange rates but at purchasing power parity, which better reflects WTP as what must be sacrificed in real terms to secure environmental improvements.

A slightly more complex form of adjusting a single source study is to use a sub-sample of the original data, selected to be "representative" of the new site. The "cost" of this approach is the reduction in sample size. It also requires access to original data, which often is not published, whereas other approaches can use summary statistics which are more easily available.

A more sophisticated approach to the same objective is to transfer a benefit function estimated in the source study, if one is available. This involves replacing the variables in the benefit function $\text{WTP}_S = F(\ldots)$ with the values appropriate to the new site. But often the benefit function reported in a study will not be the most appropriate one for benefits transfer purposes, as it might contain variables which cannot easily be collected for the new site. If the original data is available, there is the option to re-estimate a benefit function involving only those variables available for both sites.

*Meta-analysis*

A better but more complex procedure is to carry out a meta-analysis, using regression analysis to explain the results of a number of studies as a function of the characteristics
of the environmental goods, relevant populations and study methodologies. This results in a function explaining how WTP varies with these characteristics, and allows benefits transfer using the characteristics of the new site, which must be measured. Differences in the reporting of characteristics of source studies can pose a substantial problem. There are a number of ongoing projects to build databases of environmental valuation studies, reporting characteristics in a standardised framework, and it is likely that these attempts will increase the feasibility and accuracy of meta-analyses as a means of benefits transfer.

**Non-monetary "valuation"**

Although some authors consider that it is possible, at least in theory, to derive monetary expressions of all values, others hold that some values are incommensurable. But even if the former position is accepted, the valuation techniques available have their limitations, and not all impacts can be monetised at present. It is appropriate at least to mention this in passing.

There are a wide range of scoring and weighting techniques, based on surveys or expert judgements, which may be applied to individual impacts. These do not in themselves contain information about the relative trade-off values among different impacts, but summarise the severity of impacts within each category.

Direct comparison across categories is not possible, since although the scoring scales may be the same, the categories are different, and because the application in any given study may be highly site or context specific. Comparisons therefore require subjective weightings to be determined. There are a number of approaches to multi-criteria assessment, within which the weightings may be based on expert judgement or survey / focus group methods. As a matter of best practice, the assumptions made in these subjective assessments must be clearly stated.