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Thresholds of Environmental Sustainability
INTEGRATED PROJECT

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Sub-Priority 1.1.6.3 “Global Change and Ecosystems”

Stream 1 – D1.3
Review of Literature on Valuation of Coastal Ecosystem Services

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Lead authors for this deliverable: Tim Taylor, Anil Markandya and Harry Walton (UBATH)

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<th>Dissemination Level</th>
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<td>PP</td>
<td>Restricted to other programme participants (including the Commission Services)</td>
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<td>Confidential, only for members of the consortium (including the Commission Services)</td>
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Executive Summary

This paper provides a summary of the major literature on valuing coastal ecosystem services. We define coastal ecosystem services to include values associated with functions and goods, as well as explicit ecosystem services. Ecosystem functions amount to “the capacity of natural processes and components to provide goods and services that satisfy human needs, directly or indirectly” (de Groot, 1992, cited in de Groot et al, 2002). These include the following functions:

a) Regulation functions (e.g. bio-geochemical cycles);

b) Habitat functions (refuge and reproduction habitat);

c) Production functions (e.g. photosynthesis); and

d) Information functions (e.g. spiritual and recreational experiences).

Key goods and services examined in this literature review are:

- recreation;
- amenity;
- habitat (in terms of existence values of species);
- carbon fixing; and
- impacts on fisheries.

The major impact types for which little valuation has been done in the EU context are:

- jellyfish abundance;
- hedonic property valuation of coastal water quality; and
- coastal species existence values.

Of these, perhaps jellyfish abundance is most readily suited to a contingent valuation approach. Such a study has been conducted in the Chesapeake Bay area of the US. Jellyfish negatively affect recreation and have caused beach closure in Mallorca, one of the case study areas in the THRESHOLDS project. To conduct a hedonic study of property values and water quality, such as those reviewed in section 4 above, would be complex and would not necessarily focus on specific threshold-related impacts. There is a developing literature on existence values, but the three case study sites do not suit such a valuation exercise.

The existing literature on the valuation of algal blooms is developing, but to date little has been done by way of valuing different attributes of blooms in marine areas. This makes transfer of values difficult. One way of capturing the values would be to conduct a choice experiment based approach to assess the value of different attributes (such as impacts on swimming, aesthetics for recreation and tourist values for these impacts).

To conclude, there are two main gaps that could be addressed in the THRESHOLDS project:

- the valuation of different, spatially specific attributes of algal blooms; and
- the valuation of the negative impacts of jellyfish abundance.
1. Introduction

This paper provides a summary of the major literature on valuing coastal ecosystem services. It is structured as follows. Section 2 presents a definition of ecosystem services and highlights the main impacts to be examined under the FP6 Thresholds project. Other impacts may be identified later in the project and the literature review presented here will be expanded to consider these effects.

Section 3 presents a review of methods, focussing on techniques of valuation and critiques for these impacts. Section 4 presents a review of the literature as it applies to the valuation of ecosystem services in coastal areas. Section 5 presents some preliminary “gap” analysis – identifying gaps in the literature and suggesting possible valuation studies that could be conducted under the THRESHOLDS project or future work.

It should be noted that this is, by nature, an iterative report – an updated version will be delivered within the outputs of Work Stream 6 of this project. This review focuses on European literature where available, though given the nature of the services under consideration and the relative dearth of European studies on some impacts we also present US and other studies where these fill perceived gaps at European level.
2. Coastal Ecosystem Services

We define coastal ecosystem services to include values associated with functions and goods, as well as explicit ecosystem services. An integrated framework for the valuation of ecosystem functions, goods and services was presented by de Groot, Wilson and Boumans (2002). Figure 1 below presents their approach in graphical format. Ecosystem functions amount to “the capacity of natural processes and components to provide goods and services that satisfy human needs, directly or indirectly” (de Groot, 1992, cited in de Groot et al, 2002). These include the following functions:

- e) Regulation functions (e.g. bio-geochemical cycles);
- f) Habitat functions (refuge and reproduction habitat);
- g) Production functions (e.g. photosynthesis); and
- h) Information functions (e.g. spiritual and recreational experiences).

Ecosystem goods and services relate to such functions. For example, a service of flood prevention may result from the function of the ecosystem providing disturbance prevention through dampening of environmental disturbances. An example of this is given by the potential dampening effects of mangrove areas that reduced negative impacts of the Asian Tsunami in 2004.

Figure 2.1: Overview of valuation of ecosystem functions, goods and services

Key goods and services we will examine in this literature review are:

- recreation;
- amenity;
- habitat (in terms of existence values of species);
- carbon fixing; and
- impacts on fisheries.

The links between these and the ecosystem functions are described in Deliverable 1.1.2 of this project.
3. **Methods for valuation**

In valuing ecosystem services, it is often the case that market prices are unavailable, hence a range of techniques have been employed to estimate the benefits of these ecosystems. The following sections offer an overview of the most popular and accepted valuation techniques in the absence of complete markets, providing a brief description of each technique and pointing out the advantages and the limitations as well as the areas where their application is most suited.

3.1. **Classification of Techniques**

A number of techniques for placing a value on non-marketed goods and services are available. These techniques have been classified in several ways. Mitchell and Carson (1989) offer a classification based on:

(i) Whether the data comes from observation of people acting in the market (revealed preferences) or from people’s responses to hypothetical questions on their willingness to pay for a change of the environmental services (stated preferences); and

(ii) Whether the methods yield monetary values directly or indirectly.

Munasinghe (1993) distinguishes three possible approaches: (i) conventional market approaches; (ii) implicit market approaches; and (iii) constructed market approaches. Dixon (1994) distinguishes the techniques able to measure the physical/technical relationship between the cause of change and the effect using techniques based on revealed or stated preferences and the observed behaviour of consumers.

OECD/EDI/ODA(1995) classifies the valuation methods according to their appropriateness in the measurement of the various types of impacts. Four categories of impact are identified for different sectors, namely: (i) productivity; (ii) health; (iii) amenity; and (iv) existence values. This classification is particularly useful in identifying the most appropriate technique to place a monetary value on the environmental impact according to the sector generating the impact.

3.2. **Description of techniques**

A simplified description of the different techniques that may be applied to value marine ecosystems is presented in Table 3.1 below. For a more detailed description see Markandya et al (2002). It is important to note that to obtain a measure of the Total Economic Value of an ecosystem, a range of techniques would need to be employed.

The purpose of this section is not to discuss the advantages or disadvantages of these techniques, but to give the uninitiated reader an understanding of the basis for economic valuation of ecosystems.
Table 3.1: Valuation techniques (simplified)

<table>
<thead>
<tr>
<th>Method</th>
<th>Description</th>
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<tbody>
<tr>
<td><strong>Revealed Preference</strong></td>
<td></td>
</tr>
<tr>
<td>Productivity Change Method</td>
<td>Changes in an environmental attribute lead to changes in the output of the marketed good other things equal. The value of the change in the environmental attribute is therefore estimated as the change in the market value of production, with adjustments where appropriate. In the case of marine ecosystems, this may involve bioeconomic modelling.</td>
</tr>
<tr>
<td>Opportunity Cost Approach</td>
<td>Estimates the value of unpriced goods and services by measuring the forgone benefits of using the same resource for other alternative objectives.</td>
</tr>
<tr>
<td>Defensive Expenditures</td>
<td>Estimates the value of impacts through the cost of measures taken to mitigate them.</td>
</tr>
<tr>
<td>Cost of Illness Approach</td>
<td>Used to value impacts on health.</td>
</tr>
<tr>
<td>Shadow Project Method</td>
<td>Uses the costs of providing an equal alternative good or service elsewhere to value the impacts.</td>
</tr>
<tr>
<td>Substitute Cost Approach</td>
<td>Draws on the cost of available substitutes for the particular non-priced service or good. The non-priced good can be either a consumer good or an input factor. In both cases, if the two substitutes provide an identical service, the value of the non-priced good is the saved cost of using the substitute.</td>
</tr>
<tr>
<td><strong>Indirect Proxy Methods</strong></td>
<td></td>
</tr>
<tr>
<td>Travel Cost Method</td>
<td>Based on the expenditures incurred by households or individuals to reach a site as a means of measuring willingness to pay for the recreational activity.</td>
</tr>
<tr>
<td>Hedonic Pricing Method</td>
<td>Estimates the differential premium on property value derived from proximity to some environmental attribute (including issues of quality).</td>
</tr>
<tr>
<td><strong>Stated Preference Methods</strong></td>
<td></td>
</tr>
<tr>
<td>Contingent Valuation Method</td>
<td>Involves directly asking people (usually via a questionnaire or by experimental techniques) what they are willing to pay for a benefit or what they are willing to receive as compensation for the damage caused.</td>
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4. Review of studies to date

This review breaks up studies into a number of distinct categories, and focuses exclusively on coastal zones. Table 4.1 presents an overview of the studies reviewed. It should be noted this is not an exhaustive list – although it intends to cover the main studies that exist in the EU and US on these topics. Where gaps exist in the EU literature, US studies are identified as a potential source for values for benefit-transfer or for methodological development purposes.

In the review, the impacts are classified by impact type (e.g., recreation) or by the cause of the impact (e.g., eutrophication). We start by examining the valuation of specific water-based environmental issues (eutrophication, jellyfish, seagrass decline), before examining the impacts on specific receptors (such as recreation, property price). All valuations have been converted to 2004 Euros.

4.1. Valuation of specific water-based environmental issues

4.1.1. Eutrophication

There has been concern over eutrophication of the Baltic Sea for several decades. In 1988, the countries around the Baltic Sea committed to reducing the emission of nutrients and other harmful substances by half, between 1987 and 1995. This was done in response to strong public reaction after a toxic algal bloom affected large areas of the sea, killing fishes, invertebrates and seaweed. Several studies, including Turner et al. (1995 and 1999) and Gren et al. (1995) have assessed the costs and benefits of alleviating eutrophication (as a result of nitrogen and phosphorus loading) in the Baltic Sea. Some of the valuation studies focused on similar valuation issues and employed elicitation methods, enabling a comparative assessment. Specifically, the same contingent valuation study was implemented in Sweden (see Soderqvist, 1995) and Poland (see Markowska and Zylicz, 1996). In both cases, the questionnaire included: questions on how people use the Baltic Sea, information on eutrophication and its effects, questions on peoples’ knowledge of these effects, a description of the valuation scenario and the WTP questions.

The valuation scenario asked respondents to assume that:

- A large-scale international action plan against eutrophication has been proposed.
- The plan would be financed by an extra environmental tax for households, farmers, industry, etc. introduced in all countries around the Baltic.
- The plan would reduce eutrophication levels over 20 years to levels that the Baltic Sea can sustain.

A dichotomous choice format was employed, in which respondents were confronted with the following question: “If there were a referendum in Sweden (Poland) about whether to launch the action plan or not, would you vote for or against the action plan if your environmental tax would amount to SEK [X] per year during the 20 years?” Seven different amounts for X were randomly used in the question. The surveys measured changes in Total Economic Value – i.e. direct use plus indirect use plus non-use (Soderqvist, 1995).
<table>
<thead>
<tr>
<th>Category/Amenity</th>
<th>Activity/Issue</th>
<th>European Studies</th>
<th>North American/Other</th>
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<tbody>
<tr>
<td>Fisheries</td>
<td>Jellyfish Abundance</td>
<td>Knowler and Barbier (2000)</td>
<td></td>
</tr>
<tr>
<td>Fisheries</td>
<td>Anoxia</td>
<td>Smith and Crowder, 2005</td>
<td></td>
</tr>
<tr>
<td>Benthic vegetation</td>
<td>Climate Change  – Carbon capture</td>
<td>Social cost of carbon – see Defra (2005)</td>
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Both surveys were designed as mail questionnaires. The Swedish survey was sent to 600 randomly selected adults, of which about 60 percent responded. Likewise, the Polish survey was sent to 600 randomly selected adults yielding a 50 percent response rate.

The mean annual WTP of Swedish respondents were about €726.90 per adult (or €406.57 if non-respondents to the survey are assumed to have a zero WTP). (Gren et al (1995) suggest rounding the figures down to €677.61 and €369.61 per adult to reflect the fact that an open-ended WTP survey, undertaken as part of the same overall study, resulted in much lower estimates of mean WTP.) Mean annual WTP per Polish adult was €102.92 (or €52.69 if non-respondents to the survey are assumed to have a zero WTP).

The results from the Swedish valuation study were assumed to be typical of the market economies in the region (i.e. Finland, Germany, Denmark and Norway), and the results from the Polish study were assumed to be typical of the ‘Economies in Transition’ in the region (i.e. Estonia, Latvia, Lithuania and Russia). Mean WTP estimates from Sweden and Poland were transferred to each of the other countries after adjusting for differences in PPP GDP per capita. The adjusted values were subsequently multiplied by the adult population in each country to arrive at a measure of national total economic value. Aggregating over all nine countries, the present value of total basin-wide benefits of the action plan was estimated to be close to €171 billion (close to €93 billion if non-respondents to the survey are assumed to have a zero WTP) per year; using a 7 percent discount rate. This equates to an annualised benefit of close to €16 billion (close to €9 billion if non-respondents to the survey are assumed to have a zero WTP) per year.

In 1993 total annual nitrogen and phosphorus loads to the Baltic Sea were 1,061,642 t (1,022,754 t N and 38,888 t P). The 50 percent reduction in total nutrient loading under the action plan would abate 530,821 t nutrients per year. The constant marginal benefit of the action plan is thus around €30.18 (€16.63 if non-respondents to the survey are assumed to have a zero WTP) per kg of nutrient avoided.

In a separate exercise Sandstrom (1996) used a random utility maximization (RUM) model of Swedish seaside recreation to estimate the benefits (damages avoided) from reduced eutrophication of seas around Sweden. Sight depth data from around the Swedish coast was used as an indicator of water quality related to eutrophication. Sight depth is a good measure for at least two reasons. First, it is thought to be directly related to the recreationist's perception of water quality. Second, sight depth is highly correlated with nutrient load; an increase in the content of nutrients reduces the transparency of water.

By including the sight depth variable in the RUM model, consumer surplus becomes a function of, amongst other things, the level of eutrophication in Swedish coastal waters. In order to link the sight depth variable with nutrient loading, a relationship was estimated between sight depth and concentrations of total phosphorus (TP) and total nitrogen (TN). Sandstrom found that a 1 percent reduction in nitrogen content improved sight depth by 0.63 percent; a 1 percent reduction in phosphorus content increased sight depth by 0.18 percent. It is then possible to measure the benefits of policies to reduce TN and TP, in terms of increases in consumer surplus. The benefits of a 50 percent reduction in nutrient loading along the entire Swedish coastline was estimated to range between €29.6 million and €66.5 million, depending on the model specification used (between €17.25 and €38.81 per

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1 Unconverted figures are SEK 5500 from SEK 5900 and SEK 3300 to SEK 3000 respectively.
It should be noted that these damage estimates do not capture all the impacts related to eutrophication. Other forms of use value – e.g. commercial fishing – and non-use value are not included. Also, data on one-day trips were not used; since several Swedish cities are on the coast, the omission of day trips could possibly underestimate the values significantly.

Le Goffe (1995) also investigated the costs of eutrophication, using a contingent valuation study to measure the benefits of reducing eutrophication in coastal waters near Brest, France. The survey, which was administered to over 600 Brest residents, elicited information for two goods:

- Local pollution problems from microbes – the risks to man from bathing and the consumption of wild shellfish. The respondents were asked their maximum WTP to be able, without risk, to bathe and to consume wild shellfish in the harbour bay area. This question defined the good “salubrity”.
- Local pollution from high concentrations of nutrients in the harbour and its consequences on the marine ecosystem. In this case the respondents were asked their maximum annual WTP to prevent the asphyxiation of the harbour waters from high concentrations of nutrients. This question defines the good “ecosystem”.

The average WTP was €38.90 and €28.95 per household per year, respectively, for the goods “salubrity” and “ecosystem”. The WTP figure for “salubrity” represents 10 percent of the annual water bill of residents (which was the payment vehicle used in the survey). No information was given on the physical benefit (e.g. improvement on water clarity) which was valued. Again, the welfare gains cannot be related to a specific change in nutrient loading.

Stoltel et al (2003) investigated how Harmful Organic Blooms (HABs) impact on tourism in Europe. The authors conducted a contingent valuation exercise in 4 locations: Riccione (Italy), Galway (Ireland), Hanko (Finland), and Hyéres, Les Pradet and Corquieranne (France). They sampled 780 individuals over all the locations during the months of June, July and August. In the four locations, they investigated tourists experience with problems caused by algal blooms and their willingness to pay to mitigate the associated problems. Using a one and half-bounded dichotomous choice format, they asked respondents if they would be willing to pay a tax to finance investment that would prevent algal blooms from forming.

When analysing each location separately, the authors found the HAB impact on tourism to be between 0.92 and 4.9 million Euro per year in Riccione (Italy), between 9.0 and 16.4 million Euro per year in Galway (Ireland), between 85 and 539 thousands Euros per year in Hanko (Finland), and between 4 thousands and 442 thousands Euros per year in Le Pradet, Hyéres and Corquieranne (France). However, the authors found that WTP is insensitive to previous experience and contact with algal blooms, though WTP is sensitive when all sites are analysed together. The authors point out that because the survey data is derived only from summer months, it is not necessarily a representative sample of the total visitor population, and therefore these results are not suitable for benefits transfer purposes. This is perhaps compounded by the low sample size relative to the number of locations studied.

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2 Eutrophication is one of the major pollution problems in Brest natural harbour. The principle sources of the problem (over enrichment by nitrate and phosphate), in decreasing order of importance are, animal wastes, fertilisers, and domestic waste water.

3 Respondents are asked if they would pay X, if yes they are asked if they would pay 50% more, if no they are asked if they would pay 50% less.
4.1.2. Food Web Disruption

Food web disruption may result in increases in jellyfish abundance. To our knowledge, these impacts have not been valued to any great extent using contingent valuation.

Lipton and Kirkley (1995) suggest the most important benefit from oyster populations may be reductions in jellyfish abundance and reductions in welfare losses to boaters, fishermen and bathers. Hicks, Haab and Lipton (2004) suggest that “although it has not been quantified these groups may have a significant willingness-to-pay to enjoy their [Chesapeake] Bay experience with reduced jellyfish populations”.

The Millenium Ecosystem Assessment highlighted the impacts that the introduction of Mnemiopsis leidyi (a jellyfish-like animal) had caused in the Black Sea in terms of the rapid loss of 26 major fisheries species and its links to continued growth of the oxygen-deprived “dead” zone (MEA, Ecosystems and Human Well Being – Biodiversity Synthesis, 2005). Knowler and Barbier (2000) estimated the losses to anchovy fisheries in the Black Sea of the introduction of the comb-jellyfish at €13.9 million annually using a bio-economic model.

4.1.3. Hypoxia

Hypoxia – a low level of dissolved oxygen in waters – has significant impacts on the ecosystem and on the environmental attributes of waters. These impacts include:

- biodiversity impacts resulting from changes in benthic communities; and
- increased nutrient concentrations in the water column resulting in phytoplankton production (leading to water clarity impacts and potential blooms).

Bioeconomic modelling has been used to value the impacts on fisheries of hypoxia in the Neuse River Estuary in North Carolina, USA (Smith and Crowder, 2005). They found a discounted net present value of fishery rent increase of €1.9 million for a 30% reduction in nitrogen loadings.

4.2. Valuation of specific goods and services

4.2.1. Water Quality Impacts on Recreation

Note that while the quality of bathing water impacts on recreational activity, it is also synonymous with health impacts of water quality.

Kontogianni et al (2001) carried out a contingent valuation survey into WTP for improvements in a water treatment plant in Thermaikos Bay in Greece. The WTP for the water treatment plant as a whole is calculated as €17.93 based on a sample size of 480 residents and visitors. In the Kontogianni study, respondents were asked for the reasons they offered to pay, and the authors WTP estimates based on the subset of the sample that gave each reason. Though this allows them to examine a broad range of impacts, the WTP estimates only apply to the respective subset of the sample, and not to the sample as a whole. For example, the WTP for bathing is based on only 5.6% of the sample. Therefore there is significant sample selection bias, and the full-sample WTP for the water treatment plant as a whole is
more reliable. Furthermore, to examine the impact of water quality on specific activities we require studies that are specifically focused on these activities. Such studies are reviewed below.

Attention within the European Union has recently been focused on the costs and benefits of improving coastal water quality. This has come about both though moves to strengthen the existing Bathing Waters Directive (76/160), but also due to the continued failure of many waters to reach the standards set out in the current directive and the perceived high costs of meeting even these standards.

Using a combined revealed-stated preference method, Hanley et al (2001) surveyed beach visitors on the number of extra trips they would make if water quality on all beaches in South West Scotland were to increase to the standards required by the 1976 EU Water Quality Directive. Therefore, the welfare estimate applies to users only. Based on a sample size of 414, they estimated the annual welfare benefit as €11.40 per person annually. Hanley and Kristrom (2002) used contingent valuation methodology to assess the WTP of local residents for increasing water quality to the same standards in the coastal towns of Ayr and Irvine (South West Scotland). Hence, their study captures both user and non-user values, where non-users may derive amenity benefits or existence value from coastal water quality. Other studies that focus on amenity are discussed later. Using a payment card to elicit this range, Hanley and Kristrom asked respondents to indicate the values they definitely would or would not pay, and to leave ones they were unsure about blank. They posit that individuals are better placed to state a range of WTP rather than a single figure due to preference uncertainty. Doubts remain in the literature concerning whether payment cards lead to elicitation bias if individuals tend to give values in the middle of the options provided (Carson et al 2000). However, their welfare estimates are of the same order of that of Hanley et al (2001) for the area, suggesting that potential bias may be insignificant. Based on a sample size of 783, per person annual WTP is €19.44 for Ayr with a range of €13.40 - €24.84, and €11.84 for Irvine with a range of €8.31 - €14.63.

Two studies have focused on the EU Water Quality Directive in the East Anglia region of the United Kingdom. Georgiou et al (1998) used contingent valuation to assess the effects of increasing coastal water quality around the towns of Gt. Yarmouth and Lowestoft, disaggregating their results into both tourists and local residents. In both towns, they find that the differences in the valuations of the two groups are not statistically significant using a 95% confidence interval, though point estimates suggest that holiday-makers value water quality more than local residents. This is consistent with a study for all beaches in the Anglian region by Georgiou et al (2000). However, the 2000 study finds lower welfare estimates than the 1998 study, seemingly because the 1998 study did not remind respondents of substitute sites, leading to an embedding problem in the value estimates. This suggests that the 1998 results are less reliable than the 2000 study.

A study by Eftec (2002) used a multiple choice experiment to value water quality across the whole of the UK. A representative sample of 809 is used, and they elicit responses from both users and non-users of UK beaches. They calculate welfare change for a 1% decrease in the risk of suffering an episode of gastroenteritis. Given that this relates to a discreet and well-specified improvement in quality, this approach is superior to studies that ask respondents to value abstract changes in water quality following from the EU directive. However, given that they focus on all beach sites in the UK, perception of quality improvements may be lower than studies that focus on specific sites and in cases where respondents are interviewed on-site. Interestingly, Eftec also focus on the introduction of an

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4 Embedding refers to situations where valuations are inconsistent with changes in the size of the good valued – i.e. economic theory would predict a higher valuation for larger quantities of a good than for lower quantities.
‘advisory note system’ for all beaches that would indicate whether the water quality on a given beach is of EU statutory quality, based on the proposed revisions to the 1976 directive. The existing directive allows beaches to display a ‘blue flag’ if the water is of statutory quality, though does not advise against entering sea water if not present. While they find positive welfare estimated for all categories, respondents have a higher valuation for the introduction of an advisory note system than for a reduced risk of stomach illness itself. This indicates that beach users find it difficult to distinguish between different levels of water quality given that they value the introduction of more detailed information.

Using a contingent ranking instrument, Machado and Mourato (2002) studied the effects of a change in risk of suffering an episode of gastroenteritis on the Estoril Coast of Portugal. Per person WTP for avoiding an episode of gastroenteritis is €47.92. They also investigate the effects of a change in beach quality to ‘blue flag’ status. They find that the benefits of moving from a poor quality beach to a blue flag beach are €19.89, and €8.46 for moving from average to blue flag. The authors solicited values from local beach users only, as opposed to tourist users or other residents.

A study by Le Goffe (1995) looks at the effects of increasing water quality on the coast of Brest, France. Using contingent valuation methodology and a payment card format, he surveyed 607 individuals in 4 coastal sites and one non-coastal site in an attempt to elicit use and non-use values. Respondents were asked how much extra they would be prepared to pay in local taxes to fund investment in water quality in the region. The value is estimated as €38.96 per household per year. This was used as an alternative to an increase in water bills - the most natural and commonly used payment vehicle in studies into this issue – due to a sharp increase in water rates just before the survey was administered. He hypothesises that WTP, which is equivalent to 10% of the annual water bill of local residents, is quite high as a result. He further suggests that his results should have a lower weighting if included in a benefits transfer exercise.

To the author’s knowledge, there are no European based valuation studies for any other specific water-based coastal recreational activities. However, there is a wide literature from North America on some activities, which is reviewed below.

Park et al (2002) assessed the value of nature based tourism, such as snorkelling and glass bottom boat riding, in the Florida Keys region in the United States. In a joint Contingent Valuation – Travel Cost (CV-TC) study, they asked 460 summer and winter tourists about the number of trips made to the resort and the number that they would make if water quality was increased. Taking into account travel costs, this amount was equal to €353.72 per person per year. Similarly, Bhat (2003) examines the effects on tourism of improving the marine environment in Florida Keys. He finds that a 200% increase in fish stocks gives a change in per person consumer surplus of €2615.55, that a 100% increase in water quality gives a change of €3390.36, and a 100% increase in coral quality leads to a €3488.91 increase. The rather high values are perhaps due to the small sample size of 89. Roberts et al (1985) use contingent valuation to examine the WTP for recreational diving in the Gulf of Mexico. Using an iterative bid elicitation format, the survey approximately 900 respondents on how much they would pay for an annual pass to continue to use the site. They find that WTP is equal to €229.75, within a range of €191.70 - €267.81.

Another study by Thomas and Stratis (2002) examines the effects of an imposition of speed limits for boat trippers on the West Indian Manatee, Florida, suggested in order to protect endangered species. They find a compensating variation of €186.34 per year for a typical boater is applicable based on a sample of 188 users. Lipton (2003) sampled 1163 recreational boaters in Chesapeake Bay in the year
2000 using an open-ended CV survey instrument. He collected responses at 4 different points in the year to account for seasonal variability in WTP. He found that mean WTP for an improvement in water quality was €13.36 (Median WTP: €48.10). He also found that users who stored their boats in the water had a higher WTP than those who used a trailer, lending support to the validity of the survey instrument. Aggregated to state level, WTP was found to be €5.6 million or €111 million using a 5% discount rate.

Whitehead et al (2000) examine the effects of increasing fish stocks on the number of fishing trips. Based on 1012 responses, they find that the existing consumer surplus to be €52.54 per person per year, rising to €69.33 per person per year once fish catches have risen 60% and the opening of 25% more shellfish beds has occurred, a change of €16.79.

Rowe et al (1985) used a travel cost method to estimate the value of an increase in the angling catch rate by one fish per trip in Oregon, California and Washington on the Western Seaboard of the US. They find that this is equal to €1,016,603 for the entire population of users, based on a substantial sample of 36,802 users.

Bergland and Brown (1988) examined the recreational effects of ports in Oregon that are primarily used for catching salmon. Using a travel cost method, they find a compensating variation of €360.95 per person. Another study by Bell et al (1982) examined the WTP for salt water recreational fishing in the state of Florida. They elicit the choke price\(^5\) for respondents based on the need for an annual pass to enjoy access. They find that the WTP from tourists is €328.80 and €1507.10 from residents. This difference is possibly because tourists consider a wider range of other sites when deciding on trip location, and therefore are able to substitute towards sites where access is cheaper, while recreation by non-tourists is more likely to be spent locally.

Agnello and Han (1992) examine the consumer surplus that would be enjoyed by fishers at 20 Long Island fishing sites using a joint revealed/stated preference method. Given that the approach is based on several sites, the role of substitute sites is emphasised more to respondents than studies based on random utility models. Based on a sample of 580 users, the authors find that consumer surplus per trip from a 20% increase in catch rates is equal to €1.23 and €6.75 from an increase of 100% rise. Consumer surplus based on existing conditions was equal to €24.61.

4.2.2. General Recreational Impact

Bell et al (1990) attempt to value beach recreation for the entire state of Florida, and report a variety of estimates for different welfare types. They find that consumer surplus from beach recreation is €41.33, that the equivalent valuation is equal to €41.91, and the compensating variation to be €41.19. The sample is based on tourists only.

Four studies have used stated preference methods to assess the value of beach based recreation using proposed coastal sea defences.

\(^5\) The choke price is the maximum amount users would pay for a pass. This is elicited using an iterative bidding format, where the price offered is increased up to the point where respondents state they would not pay for a pass.
Bateman et al (2001) used contingent valuation to derive welfare benefits from the construction of coastal sea defences in Caister-on-Sea, United Kingdom, which would enhance the recreational qualities of the site. They also asked respondents to value a different scheme which offered the same level of protection, but impacted upon recreational activities. The difference in valuations represents the value of recreation benefits. Using a sample of 249 visitors and 203 local households, they found that holiday makers value sea defences more than the local residents, with values of €50.82 and €41.53 respectively. This is consistent with some studies of bathing water quality reviewed above.

This contrasts to a study by Spurgeon and Brooke (1995), who find that local residents value sea defences more than both day-trippers and over night visitors in West Bay, Dorset, with values of €15.86, €0.93 and €3.58 respectively. This difference may be because residents in West Bay are more at risk from flooding than those in Caister-on-Sea. However, an additional explanation may lie in the fact that the authors were forced to vary the payment vehicle from council taxes for local residents to car-parking charges for visitors. As council taxes are much higher than parking costs, the proportionate change in council taxes is likely to be several times less than the proportionate change in parking fees. Therefore visitors may have been more reluctant to express a higher WTP.

Two studies have used stated preference methods to elicit WTP for an erosion control programme in Maine and New Hampshire, USA. Lindsay et al (1992), using contingent valuation, surveyed 1100 beach visitors and found that individuals were willing to pay €30.84 per year for the programme, though the type of protection and any potential effects were not specified to respondents. The same sites were analysed by Huang and Poor (2004) using a multiple choice experiment. 255 local households were asked to 8 different protection schemes. They find that the benefit of each mile of beach saved is €4.76 per individual. Unfortunately, differences between the studies in the presentation of benefits prohibit their comparison. However, it is notable that Huang and Poor did remind residents of the negative effects of each protection programme, and calculated the welfare loss for each effect: €2.91 for a 1/1000 increase in the risk of swimming injury, €2.77 for restricted beach and water access, €3.50 for disturbance to wildlife and €3.72 for a 10% reduction in salt water quality near the beach. Reminding respondents of the amenity costs of the protection schemes probably caused valuations to be lower.

4.2.3. Amenity: Hedonic Pricing Studies

Legget and Bockstael (2000) focus on water quality in Chesapeake Bay, based on variations in the faecal coliform count at 104 different points along the Arundel County Coast Line, analysing 1183 housing transactions. As with the bathing water quality studies, deteriorations in water quality are hypothesised to reduce recreational benefits, as well as costs to amenity in terms of smell and aesthetic qualities. They find that increases in the faecal coliform count have statistically significant negative effects on housing sales prices. A change of 100 faecal coliform counts per 150 mL leads on average to a 1.55% decrease in price.

Mendelsohn et al (1992) analysed the effects of PCB pollution in New Bedford Harbour, Massachusetts. They used repeat sales analysis, a version of hedonic pricing methodology which takes into account multiple sales of the same properties using panel methods. They analyse house prices both before and after pollution of the area, and find that the cost to houses is €8000 - €11000 per affected property.
Bourassa et al (2003) analyse the effects of an ocean view on property prices in Auckland, New Zealand. They analyse 4814 house price transactions in 1996, and find that properties with an ocean view are on average 59% more expensive than those without. They also find that this effect diminishes with distance from the coast line. The authors suggest that their findings should be treated with some caution as they only analyse one sales year. This was also the year in which the Auckland property market was at the peak of its cycle, and it is suggested that this may have exaggerated the impacts.

Benson et al (1998) looked at properties in Bellingham, Washington, USA, examining 5095 house price sales over a longer period of 1984-1994. They find that an ocean view adds 23.6% to the value of a property, which suggests that the results of Bourassa et al may well be inflated by property market fluctuations. Benson et al also disaggregate views into different qualities and different distances from the shore line. They find that properties located 0.1 miles from the shore line and with high quality views (those with little or no obstruction) are on average 58.9% more expensive, whilst those with low quality views (substantial obstruction) are 25.64% more expensive at the same distance. When the distance is changed to 1 mile, this falls to 44.72% for a high quality view and 12.45% for a low quality view.

Other studies have also focussed on shore line proximity as a determinant of with-in market house price variation using hedonic pricing methodology. Pompe and Rinehart (1995) examined the effects of beach proximity on house prices in 2 separate towns in South Carolina, USA using 355 house transactions and 169 vacant lot transactions. The advantage of studying the effects on vacant lots is that variation in price is not effected by differences in house structure quality but by location and size only. Using a dummy variables, they find that ocean front houses have a 26.62% higher value than those located elsewhere and vacant lots a premium of 34.18%. They find no significant effect for and ocean view, however. Note that the markets for vacant lots and for existing structures are likely to be highly differentiated, complicating their comparison. Pompe and Rinehart (1995) also examined the effects of beach width on residential property values. Beach width is hypothesised to represent beach quality in terms of recreational benefits and flood protection. They find that effect of an increase in beach width of 79 ft to 80 ft is to increase house prices by 0.59% for properties on the water front, falling to 0.27% for properties located ½ mile away. For vacant lots, the corresponding impacts are 1.84% and 0.40%.

Bin and Polasky (2002) examined 353 house sales in Eastern North Carolina, USA, with respect to coastal wetlands in the area. They find that property prices decrease at a rate of 0.18% per kilometre away from coastal wetlands. It is important both coastal and inland wetlands are abundant in the study area, numbering over 13,000, which may contribute to this comparatively low estimate. It may also be affected the large size of the study area, which is likely to include various different housing markets. Under these circumstances, impact estimates must be treated with some caution (Freeman, 2003).

4.2.4. Economic values of species in coastal environments

Some studies have attempted, by a variety of market and non-market valuation methods, to obtain values for species in coastal areas. A selection of the results of these studies is presented below. On the
whole, the coverage of these studies for coastal species is sparse and largely focuses on species that are either generally described, e.g. “birds” or on charismatic species.

### Table 4.2: Species valuation

<table>
<thead>
<tr>
<th>Species</th>
<th>Location</th>
<th>Method</th>
<th>Value</th>
<th>Reference</th>
<th>Notes</th>
</tr>
</thead>
<tbody>
<tr>
<td>Neotropic Cormorant</td>
<td>Venezuela</td>
<td>Productivity change method: Reduced time costs to fishermen, increased fish stocks due to secretion, consumption of fish stocks</td>
<td>€9.7 mn per year</td>
<td>de Weir et al (2004)</td>
<td></td>
</tr>
<tr>
<td>General bird abundance</td>
<td>Wadden Sea</td>
<td>CVM</td>
<td>Various</td>
<td>de Blaej et al (2005)</td>
<td></td>
</tr>
<tr>
<td>Birds from oil spill</td>
<td>USA</td>
<td>CVM</td>
<td>€55.37 annual per resident for protection of 50,000 birds</td>
<td>Green et al (1998)</td>
<td></td>
</tr>
<tr>
<td>Dolphins</td>
<td>USA</td>
<td>CVM</td>
<td>€11.91</td>
<td>Kahneman and Ritor (1994)</td>
<td></td>
</tr>
<tr>
<td>Birds (watching)</td>
<td>Sicily</td>
<td>CVM</td>
<td>€136,000 to €160,000 annual benefit</td>
<td>Signorello (1999)</td>
<td></td>
</tr>
<tr>
<td>Grey-blue whales</td>
<td>USA</td>
<td>CVM</td>
<td>€31.26 annual per household</td>
<td>Hageman (1985)</td>
<td>Valued reduction in hunting</td>
</tr>
<tr>
<td>Bottlenose dolphins</td>
<td>USA</td>
<td>CVM</td>
<td>€23.14 annual per household</td>
<td>Hageman (1985)</td>
<td>Valued reduction in hunting</td>
</tr>
<tr>
<td>Sea Otters</td>
<td>USA</td>
<td>CVM</td>
<td>€27.09 annual per household</td>
<td>Hageman (1985)</td>
<td>Valued reduction in hunting</td>
</tr>
<tr>
<td>Northern Elephant Seals</td>
<td>USA</td>
<td>CVM</td>
<td>€23.88 annual per household</td>
<td>Hageman (1985)</td>
<td>Valued reduction in hunting</td>
</tr>
</tbody>
</table>

### 4.2.5. Carbon fixing

A major benefit of a healthy benthic environment may be the fixing of carbon by benthic vegetation, such as sea grasses. Values for this service may be derived from the literature on the social cost of carbon. A recent review for the UK Department of Environment, Food and Rural Affairs (Defra) resulted in estimates for the shadow price of carbon as shown in Table 4.3 below.
<table>
<thead>
<tr>
<th>Year of emission</th>
<th>Central guidance* Lower central estimate</th>
<th>Upper central estimate</th>
<th>central lower bound</th>
<th>Upper bound</th>
</tr>
</thead>
<tbody>
<tr>
<td>2000</td>
<td>79.51</td>
<td>50.60</td>
<td>187.93</td>
<td>14.46</td>
</tr>
<tr>
<td>2010</td>
<td>93.97</td>
<td>57.83</td>
<td>231.30</td>
<td>17.35</td>
</tr>
<tr>
<td>2020</td>
<td>115.65</td>
<td>72.28</td>
<td>296.36</td>
<td>21.68</td>
</tr>
<tr>
<td>2030</td>
<td>144.57</td>
<td>93.97</td>
<td>375.87</td>
<td>28.91</td>
</tr>
<tr>
<td>2040</td>
<td>202.39</td>
<td>130.11</td>
<td>477.06</td>
<td>36.14</td>
</tr>
<tr>
<td>2050</td>
<td>303.59</td>
<td>187.93</td>
<td>607.17</td>
<td>43.37</td>
</tr>
</tbody>
</table>

Source: Defra (2005)
5. Preliminary Gap Analysis

The major impact types for which little valuation has been done in the EU context are:

- jellyfish abundance;
- hedonic property valuation of coastal water quality; and
- coastal species existence values.

Of these, perhaps jellyfish abundance is most readily suited to a contingent valuation approach. Such a study has been conducted in the Chesapeake Bay area of the US. Jellyfish negatively affect recreation and have caused beach closure in Mallorca, one of the case study areas in the THRESHOLDS project. To conduct a hedonic study of property values and water quality, such as those reviewed in section 4 above, would be complex and would not necessarily focus on specific threshold-related impacts. There is a developing literature on existence values, but the three case study sites do not suit such a valuation exercise.

The existing literature on the valuation of algal blooms is developing, but to date little has been done by way of valuing different attributes of blooms in marine areas. This makes transfer of values difficult. One way of capturing the values would be to conduct a choice experiment based approach to assess the value of different attributes (such as impacts on swimming, aesthetics for recreation and tourist values for these impacts).

To conclude, there are two main gaps that could be addressed in the THRESHOLDS project:

- the valuation of different, spatially specific attributes of algal blooms; and
- the valuation of the negative impacts of jellyfish abundance.
6. References


