SPLiCE Phase 1
Investigation of methods for comparing impacts and prototype framework for comparison
Output 4 for SPLiCE Phase 1
Executive summary

Purpose
1. The impacts (environmental, social and economic) of the diverse components of energy pathways are very different. This report examines how these impacts might be assessed and compared, through valuation and other approaches, and the strengths, limitations and potential linkages of different methodologies. In doing so it aims to provide support for policy decisions about the choices among the diverse range of energy supply and demand options.

Focus
2. The focus of the report is the assessment and comparison of the impacts on the environment in societally relevant ways. Social, health and economic impacts are also considered.

Approach
3. The approach has been to review different methodologies for impact assessment: ecosystem services assessment; monetary valuation and economic approaches; health impact and social impact; and social appraisal and to identify where they have or have not been applied to energy related issues. The key assessment approaches have been subjected to an evaluation through a Strengths, Weaknesses, Opportunities, Threats (SWOT) analysis.

Findings of the review

Ecosystem service assessment
4. Ecosystem service assessment approaches provide a framework to organise information about environmental attributes important to society. Their broad aim is to relate the environment to a policy and decision making context, and their anthropocentric perspective facilitates monetary and non-monetary valuation.

5. The steps required to apply ecosystem service assessment within impact assessment have been demonstrated, although rarely empirically applied in the context of energy options, and so issues of appropriate spatial and temporal scales must be considered in applying ecosystem service approaches in this context.

6. The National Ecosystem Assessment and Natural Capital Committee approaches have been developed for the UK context but are derivative from, and hence rather similar to, previous key frameworks and typologies, and so they are the most appropriate to use in assessing environmental impacts of different energy supply and demand options.

7. Ecosystem service classifications should be tailored and applied according to the environmental context in which they are being used. To assess impacts of different energy pathways on the environment, classifications should include abiotic services. The Common International Classification of Ecosystem Services (CICES) provides a useful starting point for considering the ecosystem services that should form part of the assessment.

8. In order to understand any trade-offs in environmental impacts it is essential to consider a comprehensive suite of ecosystem services (rather than selecting easily quantified elements).

Valuation of ecosystem service benefits
9. Economic valuation of natural resources seeks to better quantify the societal value of ecosystem goods and services and can also provide a common metric to allow the disparate services within an ecosystem to be better compared when evaluating costs and benefits.

10. There are some ecosystem services for which it is feasible and appropriate to use monetary valuation as a metric, assuming that the limitations of particular methods are taken into account in any assessment. Monetary valuation is particularly applicable to provisioning and carrier services (such as food, raw materials and transport), for which market prices are available. With the notable exception of recreation (usually valued through revealed preference methods) monetary valuation is not appropriate for most cultural services. It may also be possible to obtain monetary values for certain regulating services.

11. The TEEB ecosystem service Valuation Database, available from the Ecosystem Services Partnership website (http://www.fsd.nl/esp), holds information from a large number of empirical studies, the results of which could potentially be applied in a UK energy context. However, this database does not appear to have been updated since 2010.

Methods for assessment of diverse impacts
12. Health impact analysis is an established procedure that takes into account wider socio-economic and environmental considerations and considers vulnerable groups. Its inclusion in
the assessment of different energy options could help identify threats, but also opportunities and actions to improve health.

13. Social impacts are the direct or indirect effects an action, activity, programme, or policy has on a society, community, family or individual. Complementary quantitative and qualitative methods are available that can be usefully combined to capture the full range of possible social impacts.

14. While market and monetary values can approximate reasonably well for social impacts in some cases, in many cases it is just not possible to monetise an outcome, or capture its full value through monetary valuation in a meaningful way.

15. Macro-economic approaches (e.g. computable general equilibrium models and input-output analysis) can provide additional tools for assessment of impacts on the broader economy and specific sectors impacts, especially in terms of outputs, employment and GDP, due to changes in the use of different environmental resources.

**Evaluation of impacts**

16. By using a formal and established procedure, cost-benefit analyses and deliberative monetary valuation encourage the identification of economic benefits and costs in a way which aims to be systematic and objective in order to form an opinion of the social desirability of a proposal.

17. Whilst cost-benefit analyses provide a simple and potent approach to legitimate decisions, they lack scope for public and stakeholder engagement, and do not readily indicate the uncertainties and ambiguities that are particularly associated with environmental impact and ecosystem service valuation.

18. There are a number of different social appraisal methods that can be applied to compare and hence evaluate different impacts (environmental, social and economic) of a very diverse range of energy supply and demand options. Each may be implemented in a wide variety of different ways, subject to a range of different general and context-specific evaluative imperatives and hence each method comes with its own set of pros and cons.

19. The choice of an evaluative framework will depend on the focus of the issues being evaluated and the intended use of the evaluation. A cost-benefit analysis approach is more suited to supporting the justification of policy and decisions, with legitimation and acceptance potentially being driven by the most powerful policy actors supported by experts and key stakeholders.

20. A more inclusive approach (such as multi-criteria mapping, which considers different perspectives on policy options) would arguably be more suited to addressing divergent interests in society, and hence increasing robustness, accountability and legitimacy.

**Proposed evaluation framework**

21. A conceptual framework for the process of evaluating impacts arising from different energy options shows how the elements of the review interact and interlink (Figure 5.1, reproduced overleaf).

22. Changes in natural capital stocks and flows (ecosystem goods and services) can be assessed with various metrics (biophysical, monetary and non-monetary values, health and social metrics) as well as qualitative and narrative information. This range of information can, and should, all inform decisions related to energy pathways.

23. Cost-benefit analysis will not permit full evaluation of all environmental and social impacts as not all of them can be monetised.

24. Multi-criteria assessment methods, however, can accommodate information in a range of metrics, including the outputs of economic appraisals such as cost-benefit analysis, and thus forms basis of the evaluation framework proposed.

25. The choice of pathways within the evaluation framework is context dependant and will need to be tailored to the questions posed and the purpose for which the answer is sought, as well as to the resources available.

**Key recommendations**

26. The assessment process should utilise the Natural Capital Committee approach as an overarching framework to identify the natural capital and ecosystem services information that should be considered in comparing energy options.

27. Economic values should be used where available and appropriate, supported by additional relevant data (quantitative and qualitative) on environmental, social and health impacts.

28. The structured framework provided by multi-criteria assessment methods should be used to evaluate the resulting environmental, social and economic information, ideally using an inclusive approach that allows the perspectives of a range of stakeholders to be incorporated.
**Figure E1. Conceptual framework for the impact evaluation process**

**Natural capital assessment**

- **Natural Capital Stocks**
  - Geological, physical, chemical and biological components of major marine and land-use categories

- **Natural Capital Flows (Ecosystem goods & services)**
  - Intermediate/supporting and poorly understood regulating services
    - (e.g. biodiversity, primary production, larval and gamete supply, nutrient cycling, biological control)
  - Most cultural services
    - (heritage, cultural, aesthetic, symbolic, sacred/religious, existence, bequest)
  - Provisioning services
    - (nutrition, raw materials, energy)
  - Selected cultural services
    - (recreation, tourism)
  - Well understood regulating services
    - (carbon sequestration, control of flooding and erosion, water supply)

**Further assessment**

- **Health Impact Assessment**
- **Social Impact Assessment**
- **Macro-economic assessment**

**Assessment metrics**

- **Bio-physical metrics**
  - (e.g. quantities, rates, condition indices), which should relate where possible to national environmental monitoring programmes
- **Non-monetary values**
  - Qualitative and narrative information
- **Monetary values**

**Evaluation framework**

- **Social appraisal methods**
  - Multi-criteria Assessment
  - Multi-criteria mapping
  - Social multi-criteria evaluation
  - Qualitative participation deliberation
  - Q method

- **Cost Benefit Analysis**
- **Deliberative monetary valuation**

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1 Introduction

The impacts (environmental, social and economic) of the diverse components of energy pathways are very different. This report examines how these impacts might be assessed and compared, considering in particular the use of Ecosystem Services (ES) frameworks, where and how monetary valuation is appropriate in impact assessment, and the strengths, limitations and potential linkages of different methodologies. Thus, it aims to propose an approach that can provide support for policy decisions about the choices among the diverse range of energy supply and demand options. Sections in the report summarise the key approaches and issues, with further details provided in the Appendices.

We have first considered the different ES frameworks and classification systems and how these could be utilised as an overarching framework within SPLiCE to consider environmental impact in a societally relevant way. The number of conceptual frameworks and classifications for ES continues to increase, and this review focuses on those that are most relevant at the level of UK energy policy. Nationally and internationally accepted frameworks (such as the Natural Capital Committee approach, the UK National Ecosystem Assessment and The Economics of Ecosystems and Biodiversity (TEEB)) are therefore considered, but the review also provides examples of the application of frameworks in an energy context, and describes tools that have been developed to facilitate ES assessment. Gaps or limitations in the ES frameworks have been highlighted and recommendations have been generated on how to best apply ES frameworks for the purposes of SPLiCE, considering in particular their compatibility with the Natural Capital Committee objectives.

There are a variety of approaches to assess the economic, health and social impacts of energy options, which are also reviewed. Again, this component of the review is not intended to be exhaustive, but discusses key approaches that are widely employed. The review also includes methods that seek to address the known limitations of other approaches (for example, how deliberative valuation builds on more traditional cost benefit analysis, and how Input-Output models can be extended to include environmental accounts). This section also includes consideration of ‘social appraisal’ methods, which can accommodate data of various types and take account of wider preferences and values. A series of SWOT (Strength, Weakness, Opportunity, Threats) tables are presented providing an evaluation of the different methods for assessing ecosystem, economic and social impacts.

Expressing the impacts of energy developments in a common metric would facilitate the comparison of different policy options. Where this common metric is monetary value, the implications for natural resources can be considered at the same level as other economic costs and benefits. We identify the availability of existing values that may be amenable to benefits transfer, and consider how such valuations fit within the Natural Capital Committee framework. However, using monetary valuation is not straight-forward and we explore the issues related to this. We examine the controversy surrounding the methods used and identify where monetary valuation is particularly challenging or not applicable. From this assessment, we have considered if there exists a core of ES (for example those related to food provision, recreation and carbon sequestration) for which monetary values could be obtained that are applicable across different energy pathways. We conclude that monetary valuation of some ES, social, health and wellbeing impacts is not always desirable or possible, and so alternative metrics and methods must be utilised to permit assessment of the wider environmental, social and economic implications of different energy pathways.

The final section presents a prototype evaluation framework to be used to support decision making where multiple sources of evidence are provided of environmental, economic, social and health impacts of different scenarios of pathways to reduce carbon emissions. The framework proposes the use of multi-criteria assessment (MCA) methods used to weight impacts and assess trade-offs, and explores the role of participatory and deliberative approaches to this including Multi-criteria Mapping.

This review is not intended to present a final ready to use framework but propose a prototype that can then be brought to the wider Government analytical community for further development in SPLiCE Phase 2.

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2 Ecosystem Service Approaches

2.1 Introduction

Ecosystem services (ES) deliver human wellbeing, including health, happiness and economic success; these are the dominant factors in making the decisions that determine how we develop energy policy, illustrating the relevance of ES approaches within SPLiCE. Historically we have focused on those ES that are easy to quantify (e.g. the financial returns from provisioning services such as fisheries) but have encountered both unexpected outcomes and resistance as other factors we have not considered have come into play. By considering all ES and assessing the value placed on them by different groups we should be able to make more robust decisions. ES assessment approaches provide a framework to organise information about environmental attributes important to society. Their anthropocentric perspective facilitates monetary valuation, but ES assessments can also be used to support alternative methods for the evaluation and comparison of different policy options, and so should be considered separately to any valuation step.

The Millennium Assessment (MA; Millennium Assessment 2003, 2005) was the first global ES assessment and as such highlighted the global threats to ES and human well-being. The MA brought to wide attention the need to assess and improve management in relation to maintaining ES, and instigated international and national programmes and research to better assess ES, value biodiversity and apply approaches to the needs of decision makers. These programmes and associated research either addressed gaps and weaknesses in internationally recognised frameworks (such as the MA), further developed classification or typologies within frameworks, or addressed economic valuation and assessment of ES (to integrate ES concepts with existing national and global economic approaches) (Figure 2.1).

Key ecosystem services frameworks and typologies as well as examples of applications of the frameworks are reviewed in the Appendices. The review particularly identifies where the frameworks have or have not been applied to energy related issues. Appendix 1, the focus of the review, is on frameworks that have been developed or applied for the UK: the UK National Ecosystem Assessment (UKNEA) and its Follow-on (UKNEAFO), The Natural Capital Committee (NCC), EnergyScapes and TIDE, as well as the Hattam et al. (2014) study concerning indicators for marine ES. One European framework, the Common International Classification of Ecosystem Services (CICES) is included, as it provides a more comprehensive classification of ES than has been attempted in the UK frameworks.

Key international frameworks, which have influenced the development and practical application of UK approaches, are described in Appendix 2: the MA, The Economics of Ecosystems and Biodiversity (TEEB, and its Oceans and Coasts application), the System of Environmental–Economic Accounting (SEEA, and its fisheries application), and Mapping and Assessment of Ecosystem Services (MAES). Appendix 3 considers some important academic studies that have informed the frameworks and typologies: Fisher et al. (2008, 2009); Balmford et al.(2008, 2011); Atkins et al. (2011a,b).

Recent developments to frameworks and typologies have focused on clearly classifying ES in order to distinguish between intermediate and final services, as it is only the goods and benefits arising from the latter that should be valued (e.g. Fisher et al., 2008, 2009; UKNEA, 2011). However, as the work of Natural Capital Committee (2014) notes, it is insufficient to consider only these final goods and benefits. Their management requires understanding of the extent, status and resilience of the ecosystem responsible for their supply, such that thresholds and safe limits for sustainable use can be identified (NCC, 2014).

From this review, the NCC framework emerges as the most appropriate conceptual approach to ES assessment within SPLiCE, primarily because it incorporates the evaluation of stocks of natural capital as well as the ecosystem services that flow from them. The NCC framework is similar to, and builds on lessons learned from, other work such as the UKNEA. The typology and classification of ES proposed by CICES is also advocated because its level of detail supports practical application, although it is weak in the treatment of abiotic service (fundamental in energy assessments). Both the NCC and CICES have adapted or seek to support the development of the SEEA. This is also to their...
advantage, as this aspiration for ecosystem accounting systems and the development of international standards supports comparability between assessments. Although NCC and CICES are recommended as frameworks, they are likely to require some adaptation for the specific context of individual assessments – as yet, no single framework has been shown to address all the issue raised in practical field evaluation, particularly at local scales.

Figure 2.1 Development of ES frameworks and classifications / typologies since the Millennium Assessment, including the National Ecosystem Assessment (NEA), Common International Classification for Ecosystem Services (CICES), System of Environmental-Economic Accounting (SEEA) and The Economics of Ecosystems and Biodiversity (TEEB).

2.2 Spatial decision support tools for ecosystem services

The need to make decisions based on multiple competing land uses (or ecosystem services) requires spatially explicit information, this has led to the development of decision support systems, enabling users to account for the spatial configuration of Natural Capital, drivers of change (e.g. climate) and ecosystem services and to attempt to understand trade-offs and synergies. A review of user needs for Spatial Decision Support tools (Smart et al. 2013) identified these 3 criteria for functionality:

- The ability to evaluate trade-offs between information layers and competing decision solutions
- The flexibility to incorporate user-defined rules for evaluating those trade-offs
- The ability to include scenarios of future impacts of drivers such as land-use and climate change
This is a rapidly developing field and there are a number of different Spatial Decision Support tools available that vary in complexity and application. Tools can be fully functioning software applications or more commonly toolboxes and add-ons available within GIS software (e.g. LUCI, InVest). There are many issues and potential pitfalls associated with the use of tools in ecosystem service assessment e.g. simplification of the number of ecosystem services often based on ease of quantification and measurement, lack of finely resolved data, not accounting for thresholds and non-linear responses, (Scott et al., 2014).

Examples of these decision support tools (details of which are provided in Appendix 4) include:

- LUCI: Land Utilisation and Capability Indicator
- InVEST: Integrated valuation of Environmental services and Tradeoffs
- ARIES: Artificial Intelligence for Ecosystem Services
- SIAT: Sustainability Impact Assessment
- LEED: The Local Environment and Economic Development (LEED) Toolkit
- MIMES: Multiscale integrated Earth Systems model

2.3 SWOT summary for ES frameworks and tools

Analyses were undertaken of the key strengths, weaknesses, opportunities and threats (SWOT) of the different ES frameworks. Some of the SWOTs are common across many of the ES frameworks, as indicated in Table 2.1. SWOTs for individual frameworks and tools are presented in Appendix 5.

Table 2.1. Summary of key conclusions from a SWOT analysis of ecosystem service frameworks

<table>
<thead>
<tr>
<th>Strengths</th>
<th>ES classifications and economic approaches all aim to relate environment to a policy and decision making context (or transferable language).</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>For example, the NCC:</td>
</tr>
<tr>
<td></td>
<td>- Identifies stocks and flows, presenting environmental systems in a transferable language</td>
</tr>
<tr>
<td></td>
<td>- Utilises spatial units to aid assessment in planning decisions and impact assessment</td>
</tr>
<tr>
<td></td>
<td>- Incorporates existing natural resource management systems.</td>
</tr>
<tr>
<td>Weaknesses</td>
<td>Many frameworks and economic approaches are continually evolving, particularly in application to specific requirements for, or scale of, energy developments.</td>
</tr>
<tr>
<td></td>
<td>Areas of weakness are identifiable in individual frameworks, for example: CICES has only a provisional classification for abiotic materials in the current version (4.3), even though these were included fully in the earlier v3.</td>
</tr>
<tr>
<td>Opportunities</td>
<td>The steps required to apply ES assessment within impact assessment have been demonstrated through adaptation of frameworks, accounting systems, research to identify indicators that can be extrapolated across different sites and identification of research required to build a knowledge base on contribution of species and habitats and related functions to ES.</td>
</tr>
<tr>
<td></td>
<td>A need for more applied case studies/worked examples was identified for most ES assessment approaches.</td>
</tr>
<tr>
<td>Threats</td>
<td>Research/development is required for development/application of frameworks to the appropriate spatial and temporal scales and particular needs of impact assessments of energy developments.</td>
</tr>
<tr>
<td></td>
<td>Evidence gathering and research needs identified included better understanding of the contribution of species and habitats to ecological processes and ES services and identification of indicators that can be applied to IA scenarios.</td>
</tr>
<tr>
<td></td>
<td>Timescales to develop this research and knowledge base may inhibit application to the energy developments being considered in the near future.</td>
</tr>
<tr>
<td></td>
<td>Acceptance of the application of ES assessment within impact assessment would also be required for all approaches as regulators, consultancies and industries will be adapted/used to existing approaches.</td>
</tr>
</tbody>
</table>
2.4 Recommendations

- There is a diversity of approaches; although none are perfect, all have strengths inasmuch as they aim to relate changes in the environment to a policy and decision making context.
- ES, when applied properly, provide a powerful method to inform potentially controversial decisions; they need to be more widely publicised and appreciated.
- In order to assist in making practical decisions ES must be effectively registered in both time and space (i.e. assessed for the right location and at the right time).
- In order to understand any trade-offs in environmental impacts it is essential to consider a comprehensive suite of ES (rather than selecting easily quantified elements).
- Since the NEAFO and NCC approaches have been developed for UK habitats and are derivative from and hence rather similar to the other approaches, they are the most appropriate to use in assessing environmental impacts of different energy supply and demand options.
- ES typologies or classification should be tailored and applied according to the environmental context in which they are being used. The CICES ES typology and classification is more comprehensive than that of the NEA. Within each environmental context it is appropriate to use the CICES list to scope the relevant ES to that context.
- To assess impacts of different energy pathways on the environment, ES classification should include abiotic services, which are at present provisional within CICES.
- The use of spatially explicit models that address issues of application of ES approaches at different scales should be further explored and developed.
3 SWOT analysis of impact assessment methods

This chapter summarises the Strengths, Weaknesses, Opportunities and Threats (SWOT) of key impact assessment techniques. Sections 3.1 to 3.4 briefly describe economic assessment, health impact assessment, social impact assessment and social deliberation methods, before SWOT tables for the different methods are presented. Further details, of and issues related to, the methods considered in the SWOT are provided in the appendices.

3.1 Economic Assessment

3.1.1 Monetary valuation of environmental benefits

The benefits from ecosystem goods and services challenge neoclassical economic theory because they involve significant non-market values and often cannot be assigned property rights (Straton, 2006). The lack of private property rights prevents a market from becoming established, as commodities cannot be bought and sold without owners (Common and Stagl, 2005). There may also be externalities associated with traded goods and services if their price does not reflect the costs to the environment, and the associated effects on third parties. A polluter’s profits, for example, are often not affected by the impacts of his activity on others, and the value of a fish to a trawlerman does not take account of the implications for other fishermen (current and future) or to society at large (Lipsey, 1989). Nor are Governments necessarily better at pricing resources, as subsidies distort the wider economy and can encourage activity that further degrades natural capital (Markandya et al, 2002). Governments, and individuals, also tend to short-termism, while society as a whole operates in a longer time frame.

Economic valuation of natural resources seeks to better quantify the societal value of ecosystem goods and services and can be particularly useful if markets and common property regimes have failed to effectively reflect the social costs of environmental degradation (Howarth and Farber, 2002). A further objective of economic valuation is to provide a common metric to allow the disparate services within an ecosystem, and also the market- and non-market-based values, to be better compared when evaluating costs and benefits. However, ecosystem valuation is not about trying to place an absolute “dollar value” on facets of the environment, but instead seeks to evaluate how the change in ecosystem service provision is traded off against other things people value (Turner et al., 2003). Valuation is not a stand-alone solution, it is a tool for organising information to help guide decisions (Daily et al, 2000).

Ecosystem services can be classified according to the type of value they provide (Barbier, 1994). Such an approach consider the Total Economic Value (TEV) of the services, In addition to considering use values derived from direct utilisation, TEV includes the non-use values which are derived just from knowing a species or habitat exists (the existence value) and that resources will still be available for future generations (bequest value). The benefits derived from actually using ecosystems can be further divided according to whether the benefit is obtained from direct or indirect use, or from some as yet unknown future use (the option value).

Environmental values have been used to inform project decisions for more than 40 years, including for the construction of hydroelectric dams on Hells Canyon in USA in the late 1960s (Hanley, 1995). Ecosystem valuation has been shown to confirm the importance people place on their environment, and that their willingness to pay for conservation can vastly outweighs the costs involved (Howarth and Farber, 2002). The process of deriving quantitative measures of value is not straightforward, but even an imperfect expression of value can help to inform the decision-making process (Daily, 1997). Monetary valuation of environmental goods may be undertaken for a number of reasons, including attempts to determine a ‘snapshot’ of the total economic value at a given time, which may be used to highlight the relative value of different ecosystem types or different locations (Costanza et al. 1997; Martínez et al., 2007). Such snapshots could also be used to determine ‘green GDP’, so that natural...
assets can be reported in the same framework as market goods and services (Boyd and Banzhaf, 2007). Another use for valuation is in assessment of the costs and benefits of a specific project or policy intervention (Carpenter et al 2009; Fisher et al. 2009), which may involve consideration of the losses resulting from damage to a healthy ecosystem, or the costs to restore a degraded area (Turner et al., 2003). Valuation is also used to support the development of economic instruments such as habitat banking or payment for ecosystem services (PES) schemes (Ring et al., 2010; Fisher et al., 2009), or to determine the levels of compensation payable following oil spills or other accidents or incidents resulting in ecosystem damage (Dunford et al., 2004). Taxes on pollution or subsidies for better environmental practice have also been used in the development of environmental policy. The rate of ‘polluter pays’ taxation rarely has any direct link to the environmental costs (Palm and Larsson, 2007), and so valuation may also be useful in improving these economic instruments.

In the UK Policy context, documents such as the National Environment White Paper (HMG, 2011) make clear the importance placed by policy-makers on valuing natural capital and ecosystem services. More specifically, guidelines for appraisal and evaluation are explicit in the requirements for comprehensive assessment of the costs and benefits of policy options, and the use of monetary values is recommended wherever feasible (HM Treasury, 2013; BIS, 2013).

3.1.2 Availability of monetary values for ecosystem services

The growing interest in ecosystem services has led to the undertaking of a large number of empirical valuation studies, the results of which could potentially be applied in a UK energy context. A key source of information on these studies is the TEEB Valuation Database (van der Ploeg and de Groot, 2010). This is available from the Ecosystem Services Partnership website (http://www.fsd.nl/esp), where a commitment to continually update the database is made. However, there is no evidence that any updates have been undertaken as yet, as no studies after 2010 are currently included.

The TEEB database includes over 1,300 values across a range of global ecosystems. Only 58 of these are from the UK (mostly for marine and wetland habitats), but the benefits transfer process allows for geographical difference, and so there are potentially over 700 values that may be transferable to the context of energy policy assessment in the UK. These values relate to habitats including woods, agricultural land and urban green space as well as coastal and marine areas.

The studies within the TEEB database also illustrate where effort in empirical valuation has been focused. Nearly half (42%) of the values relate to food, raw materials and recreation. Regulating services including the prevention of storm, flood and erosion damage and also carbon sequestration and water-related services also feature strongly (combined, these equate to 20% of the values). This tendency for empirical valuations to consider services linked to market prices and revealed preferences indicates the level of practical challenges that remain for monetary valuation of ecosystem services. However, the valuations within the database also illustrate that progress is being made in understanding, and valuing, certain regulating services. A literature review for studies after 2010 would further demonstrate where such advances have been made.

3.1.3 Economic assessment techniques

The SWOT analysis evaluates four economic methods:

- Cost Benefit Analysis (CBA) compares the gains and losses in monetary terms of a particular decision, policy or intervention
- Deliberative monetary valuation seeks to address some of the challenges of CBA through a more participatory process
- Computable General Equilibrium (CGE) models are a macro-economic tool for evaluating the welfare and distributional implications of policies
- Input-Output (I-O) models are a second macro-economic method, which consider the relationships between different sectors of the economy and have the potential for extension to include ecosystem implications

These methods are described in more detail in Appendix 6, which also discusses micro-economic techniques.
3.2 Health Impact Assessment

Health Impact Assessment (HIA) is a combination of procedures, methods and tools by which a policy, program or project may be judged as to its potential effects on the health of a population, and the distribution of those effects within the population (WHO 1999). HIA studies are used to predict the consequences - negative as well as positive - on health of affected communities and provide information that can help decision makers implement prevention and control strategies throughout the project/policy cycle (IFC 2009). Its primary outcome is a set of evidence-based recommendations to ensure that projects or policies minimize potential negative health outcomes, maximize positive effects, and reduce any impact on health inequalities. Secondary aims of HIAs include raising awareness among policy-makers and in the local community (Mindell et al 2008). Additional detail is given in Appendix 7.

3.3 Social Impact Assessment

'Social impacts and consequences of policies for wellbeing have been acknowledged by the Prime Minister as essential to measuring how lives are improving, in addition to the established indicators of economic progress. This emphasis on quality of life - and its relation to how the economy grows is seen as central to the new independent measures of national wellbeing, that are being developed by the Office of National Statistics (ONS), and commensurate work, led by the Cabinet Office, to understand how government policies contribute to wellbeing..... Government needs a balanced framework of subjective and objective measures of wellbeing if policy is going to deliver positive social impacts". (Maxwell et al, 2011 p1)

The term social impact is used to mean the effects an action, activity, programme, or policy has on a society, community, family or individual directly or indirectly (IAIA, 2003). Impacts encompass marketed and non-marketed goods and services’ (DEFRA, 2011). Social impacts can be positive or negative and involve a range of different social variables affecting a community’s or person’s wellbeing (Plymouth Marine Laboratory and Marine Biological Association, 2013). The DEFRA (2011) framework on social impacts suggests that understanding social impacts of an intervention involves understanding (a) social capital stocks (alongside produced capital, human capital, and natural capital) and (b) flows of social goods and services (alongside market goods and services, private non-market goods and services, and environmental goods and services other flows) which emanate from the stocks and are then experienced or consumed. Social capital is understood as the stock of ‘social networks together with shared norms, values and understandings that facilitate cooperation within or among groups’ (Cote and Healy, 2001). The presence of social capital helps individuals to achieve things in collaboration with others because communities function with a greater degree of trust and understanding, among other things. Human capital is ‘the knowledge, attributes, competencies and skills embodied in individuals that facilitate the creation of personal, social and economic well–being.’ (OECD, 2001:18). It is owned by individuals and also consists of personal attributes, for example, strength and intelligence that contribute to earning potential (Halpern, 2005, in DEFRA, 2011).

The SWOT analysis considers:

- Social Impact Assessment (SIA), as an overarching framework for the evaluation of all impacts on people and their interactions with their surroundings;
- Social Accounting, for reporting the social and environmental impacts of organisations;
- Social Return on Investment, which measures non-market social, environmental and economic costs and benefits and compares them to resources invested;
- Outcome Mapping, a technique that focuses on outcomes in behaviours, relationships, activities, or actions of the people, groups, and organizations with whom a program directly works.

The specific purposes, key features, and uses of these methods are described in Appendix 8.

3.4 Specific ‘social appraisal’ methods

This section discusses a suite of decision support methods that can accommodate data of various types and take account of wider preferences and values.
An important single factor to bear in mind in considering these methods – and the review that follows – is that when it comes to evaluating them, one compared with another or compared to economic assessment methods, ‘the devil is in the detail’ (Mohr et al. 2014; Bussu et al. 2014). Each may be implemented in a wide variety of different ways, subject to a range of different general and context-specific evaluative imperatives (Dryzek 2002, Rowe & Frewer 2000; Bohmann 1996). Although there are general tendencies in each respect, these are more often a reflection of the associated disciplinary cultures, than they are of technical necessity. As a result, any process of evaluation must contend with a ‘fractal’ structure of pros and cons (Stirling 2011). There is no apparently positive characteristic in respect of any possible evaluation criterion that may not be reversed by some more detailed feature of the way in which a particular method is designed or implemented. The review that follows is therefore based around the general tendency in practice.

Summarizing the strengths, weaknesses opportunities and threats is obviously subject to the above qualification. It is equally obviously a matter of perspective. In a highly charged policy arena such as that in which these techniques are applied, the values under which the methods themselves are appraised are just as subject to political controversy as the issues to which they are applied (Stirling 2008). So what count as ‘strengths’ and ‘weaknesses’ will – to a large extent – be in the eye of the beholder. This review addresses this challenge by means of a SWOT table (as with the other sections in the report), with entries given explicitly in relation to particular political aims. This table includes as opportunities the issue of ‘linkages with other methods’. Under ‘threats’ it addresses the question of potential impacts on wider public value based controversies. By way of further crucial context, before this table, the intended purpose of each method is elaborated further below.

Overarching issues concerning applicability to different scales are included – where relevant – in the next section concerning purpose. But a crucial point to bear in mind is that all the specific methods addressed here (as well as the general approaches they represent) have been variously used at virtually every spatial level and temporal scale (Giampietro 2004). There is nothing intrinsic to the applicability of these different methods in this regard. Observable patterns of preference are more a reflection of cultural styles in the responsible disciplines, than any inherent features of the methods themselves.

Local/national trade-offs are just one instance of what can best be addressed in more general terms as the handling of issues of distribution and representation (O’Neill 2001). These are similar to the topic of ‘scale’ in the questions they raise. In short, there is nothing intrinsic about any of the focal methods that makes any of them inherently or self-evidently more favourable on this count.

Preferences in this regard will reflect the precise ways in which the methods are implemented, as well disciplinary affiliations and biases in how they are viewed. In principle, each approach can be designed in different ways, such as to address distributional issues like those concerning displaced local impacts (Rix et al. 2010; Dowding et al. 2004; Allan 2003).

Consideration of generally established patterns of usage across different policy areas, raises similar issues (and for the same reasons) to those discussed above in relation to scale, distribution and representation. In broad terms, every one of the selected methods has been used in some context and fashion to appraise strategic issues in energy policy in general, and low carbon transitions in particular. All those selected here are in principle highly applicable in this area. For instance, indicative examples of use in energy policy include for: MCA (Trutnevye 2013; Robinson 1992; Keeney et al. 1987; Mirasgedis & Diakoulaki 1997); SME (Kurka 2013; Saaty & Vargas 2006) and QPD (Dorfman et al. 2012; Chilvers & Longhurst 2012; Giurco et al. 2011; Einsiedel et al. 2013; Raven et al. 2009) and also for cost benefit analysis (CBA, see also Section 3.3.1) (ExternE 2005; Sundqvist & Soderholm 2002; Jensen & Al. 1994; Stirling 1997a; Pearce 2001); The references given in each respect also reflect some of this diversity.

Evaluation of the ‘utility’ of the different methods raises challenges very similar to those discussed above. Any unqualified expression of merit orders across these (or any other methods) would reflect the subjective values and assumptions on the part of the evaluator more than the inherent characteristics of the methods themselves. This is discussed especially in Section 5.2.3. Each mode of implementation reflects different fundamental notions of what might constitute the ‘utility’, usefulness, appropriateness or value of the technique itself and those with which it might be compared. So these central evaluative challenges must necessarily be addressed in a ‘plural and conditional’ fashion, rather than a matter of absolute or definitive objectivity (Stirling 2012). All of the reviewed techniques are susceptible to being used in some way with ecosystem service frameworks. The main question that might be posed in this regard is ‘why?’
It is arguable that many of the methods reviewed here actually constitute preferable means to appraise ecological values themselves (when compared with a narrow utilitarian ‘service’ framework) (Sagoff 2008). Some of the reasons for this are explicated in the course of discussing the rest of this review. Whether or not it is agreed with, this does mean that an exercise like the present project should be extremely careful about uncritically accepting an instrumental service framework as self-evidently appropriate even in relation to ecosystems themselves, let alone in relation to wider societal dimensions of energy strategies.

The methods addressed and reviewed for the SWOT analysis are:

- staged multi-criteria assessment, MCA (broadly based on principles of multiattribute utility theory, as operationalised in multi-criteria decision analysis);
- social multi-criteria evaluation, SME (based on a different approach to multi-criteria modelling and involving various interactive practices to augment them);
- qualitative participatory deliberation, QPD (a wide diversity of variously-designed approaches to inclusive public engagement, often based on citizens’ panels);
- multi-criteria mapping, MCM (qualitative / quantitative comparison of open-ended elicitations, that focuses not on aggregating, but mapping divergences);
- Q Method, QMA (a distinctive open-ended hybrid qualitative/quantitative approach to mapping out contrasting perspectives on any value-based issue).

Further information on these methods is provided in Appendix 9.
### 3.5 SWOT analysis

#### 3.5.1 Monetary Valuation Methods

<table>
<thead>
<tr>
<th>VALUATION METHODS</th>
<th>Cost benefit analysis</th>
<th>Deliberative monetary valuation</th>
<th>CGE models</th>
<th>Input-output models</th>
</tr>
</thead>
<tbody>
<tr>
<td>Strengths</td>
<td>• By using a formal and fairly established procedure, CBAs encourage the identification of economic benefits and costs in a way which aims at being systematic and objective in order to form an opinion of the social desirability of a proposal;</td>
<td>• The major strengths (where these aims apply) lie in the justificatory power for policy making, and associated capacity for narrowing down the scope of assessment and closing down associated political debates. The scope for deliberation, illumination of divergences and process learning are also additional strengths. And there is also the benefit of using, but somewhat diminished by the reduced credibility fostered by deliberation revealing the shortcomings of this metric.</td>
<td>• CGEs are built on the basis of a well-established economic theory explaining the behaviour of single economic agents, be it firms or consumers, on the basis of a set of behavioural assumptions. This is normally referred to as micro-foundation of CGEs (Böhringer et al 2003);</td>
<td>• IO analysis can provide information on economy-wide and sector specific impacts especially in terms of outputs, employment and GDP due to changes in the use of different environmental resources (and associated ecosystem services once the links have been developed).</td>
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<td></td>
<td>• Data requirements in CBAs can point out evidence gaps in the literature, unreliability of functional form used to estimate impacts and stimulate debate on the critical assessment of existing evidence in relation to the particular impacts of a proposal.</td>
<td></td>
<td>• CGEs can assess a wide range of policies, as testified by the proliferation of articles in the literature, and by the expansion of CGEs from core economic models to a number of other fields, including environmental and climate change policies. Policy simulation implies the change of a single or a set of parameters. Comparison between the counterfactual (without policy change) and the model results including the effect of the policy provides information on the policy impact (Allan et al 2007);</td>
<td>• It is based on economic theory with simplified assumptions that allow the internal workings of an economy to be revealed providing an alternative to “black-box” models.</td>
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<td>• Prices are endogenous to the model and are determined as part of the process leading to the equilibrium (market clearance) (Xie and Saltzman 2000). This can be particularly relevant when assessing environmental and energy policies having an impact on price either directly or indirectly through the introduction of quantity constraints.</td>
<td></td>
<td>• MRIO models allow environmental impacts associated with imports and exports attributed to countries to be tracked globally.</td>
</tr>
</tbody>
</table>
Weaknesses associated political debates. The tendency to narrow in appraisal and close down aggregation and sensitivities; lack of robustness associated with uncertainty; lack of transparency in reflecting key preferences (OECD 2006); although CBA assumes that those gaining from the proposal, through weights based on marginal utilities from income (lower for households with a higher disposable income) is open to debate and might not fully reflect social preferences (OECD 2006); although CBA assumes that those gaining from the policy will compensate those bearing the costs, whether such a compensation takes place is not part and cannot be enforced as part of a CBA. Persuasive power of ethical reasoning based only on potential compensation possibilities is unclear (Sen 2000). Some of those benefiting from the proposal could actually lack financial resources to pay required compensation; monetisation of some of the benefits and costs, especially those without a marketvalue, is both controversial (from an ethical standpoint) and problematic (from a methodological perspective); major weaknesses are the lack of scope for public and stakeholder engagement. If incorrectly carried, there is a risk of inadequate treatment of uncertainty and ambiguity; lack of transparency in reflecting key sensitivities; lack of robustness in relation to aggregation and – where these aims apply – a tendency to narrow in appraisal and close down associated political debates.

**Cost benefit analysis**
- Present value of benefits and costs in the future are very sensitive to the choice of discount rate. Present value of £100 20 years from now goes from £61 to £38 if rate increases from 2.5% to 5%. There are also doubts as to whether the discount rate should be constant across time, a function of some other socio-economic variable or reflect uncertainty about costs and benefits (Harrison 2010);
- The extent to which CBA takes into account distributional implications, i.e. allocation of the costs and benefits of the proposal, through weights based on marginal utilities from income (lower for households with a higher disposable income) is open to debate and might not fully reflect social preferences (OECD 2006);
- Although CBA assumes that those gaining from the policy will compensate those bearing the costs, whether such a compensation takes place is not part and cannot be enforced as part of a CBA.

**Deliberative monetary valuation**
- Although labelled as “deliberative”, the constraints of the monetary focus mean this is significantly less so in comparison with other methods (like SME, QPD and MCM);
- This can be seen as combining the worst of all worlds, in compromising on the justificatory power of CEA, whilst adding further complexity but failing to deliver the corresponding benefits of flexibility, transparency and robustness associated with multi-criteria or participatory deliberative methods.

**CGE models**
- Functional forms used in the model may be somewhat arbitrary especially in their assumptions of nested structure, i.e. with output or consumption determined through a series of pairwise substitution possibilities. Modellers are generally constrained to work with a limited number of functional form types to ensure that the model can be solved (Allan et al 2007);
- Behavioural assumptions, especially optimising behaviour and perfect information, are hardly representative of real-world economic system, implying that measures of welfare changes computed by the model might have only a limited relation to the real effect of the policies (Scrieciu 2007);
- Parameter values are taken from a variety of empirical studies in the literature, not always referring to the country or time period being modelled and necessarily consistent across them and not. Calibration of free parameters occurs on one year only with the non-testable assumption that the chosen year represents a benchmark equilibrium (Allan et al 2007);
- CGEs are sometimes criticised for being a “black-box” (where the inner working of the model is difficult to explain), making tracing the effect of a policy through the economy somewhat problematic (Böhringer et al 2003);
- Limitation in data availability make it generally difficult to build subnational CGEs, therefore severely impairing the utility of CGEs in the analysis of environmental problems with strongly localised impacts such as air pollution, water and land degradation (Scrieciu 2007). It is questionable whether equilibrium frameworks and models are appropriate for analysing the complex and uncertain environmental and social consequences of energy system changes.

**Input-output models**
- There are a number of assumptions which are used in the input-output analysis which include:
  - No supply-side constraints, i.e. if there is an increase in demand for particular goods then the supply-side of the economy will adjust to accommodate this increase. (However, Gosh’s approach to the calculation of the technical coefficient attempts to address this weakness.);
  - Fixed prices meaning that even though there might be an increase or decrease in production, prices will not change within the system;
  - Fixed production processes assume that there are no substitutes across industries;
  - CGEs to scale impose fixed proportions of inputs to outputs;
  - Homogeneity within production processes are assumed within a sector;
  - The input-output data is often published with a lag of up to five years. For example the most recent UK IO tables are for 2010.
### VALUATION METHODS

<table>
<thead>
<tr>
<th>Opportunities</th>
<th>Cost benefit analysis</th>
<th>Deliberative monetary valuation</th>
<th>CGE models</th>
<th>Input-output models</th>
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<tbody>
<tr>
<td>• CBA could incorporate insights from the different versions of the concept of sustainability, in particular strong and weak sustainability, by explicitly addressing the implicit assumptions of ability to trade different components of the costs and benefits caused by the proposal. This might be more easily applicable to a pool of proposals rather than a single one (OECD 2006);</td>
<td>• There is greater openness than in CEA for articulation with other techniques, because the deliberative aspect allows greater latitude than is the case in rigid calculative externalities assessment, for taking account of the qualitative implications of the pictures yielded by different appraisal methods. But, in the end, the monetary idiom raises similar difficulties.</td>
<td>• Relationship between the economy, the environment and the energy system can be modelled at different contact points, either on the consumption or production side of the economy;</td>
<td>• Integration of IO models with ecosystem services to provide economic impact analysis of alternative uses natural resources;</td>
<td></td>
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<tr>
<td>• Possibilities to combine CBA with other methodologies such as Multi-Criteria Analysis and deliberative methods might offer an interesting opportunity to merge the systematic assessment of CBAs with the more flexible and exploratory approaches of the other techniques.</td>
<td></td>
<td>• Environmental impacts related complex natural science background such as water stress or biodiversity loss are hardly represented in the literature. Pairing up CGE to specific bio-physical or socio-economic models could offer a more immediate approach than trying to add an environmental module to CGEs (Böhringer and Löschel 2006).</td>
<td>• IO tables can be combined with Life Cycle Analysis and footprinting to support macroeconomic assessments (UKNEAFO 2014).</td>
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<table>
<thead>
<tr>
<th>Threats</th>
<th>Cost benefit analysis</th>
<th>Deliberative monetary valuation</th>
<th>CGE models</th>
<th>Input-output models</th>
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<tbody>
<tr>
<td>• Strong views on the desirability of using CBA as a tool for public policy decisions may lead to partisan debates ignoring the contributions of the argument coming from the opposite point of view, as testified by the tone and language of a number of articles in the literature;</td>
<td>• If sensitivity analysis is used to convey the implications of divergent perspectives in the reporting of results, then this can be less threatening than conventional externalities assessment in respect of the issues raised for CEA.</td>
<td>• As users of CGEs need to understand their inner assumptions, the perceived “black box” nature of the models could impair adoption of the model by academics and practitioners. If results may are not perceived as transparent, they are unlikely to be trusted;</td>
<td>• The assumptions on which MIOTs are based and the lag in their production could reduce adoption of these models;</td>
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<tr>
<td>• If uncertainties and ambiguities are concealed in ostensibly precise and definitive monetary values, there is a risk of seriously suppressing the quality of policy debates and wider democratic discourse in the field.</td>
<td></td>
<td>• Other optimisation-based modelling approaches if one wants to optimize other, possibly non-economic, variables such as resource use and pollution</td>
<td>• The aggregation of sectors within the IO models and their lack of detailed microeconomic foundations might prompt researchers to use other more detailed microeconomic models.</td>
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### 3.5.2 Health Impact Assessment (HIA)

<table>
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<tr>
<th>Health Impact Assessment (HIA)</th>
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<tbody>
<tr>
<td><strong>Strengths</strong></td>
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<tr>
<td>• Established procedure able to take into account existing evidence in terms of dose-response, concentration-response and resource-response functions, as well as wider socio-economic and environmental considerations;</td>
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<tr>
<td>• Adequate consideration can be given to vulnerable groups. This is normally recommended in most guides of HIAs although there is disagreement about whether disadvantaged group should be identified at the beginning or during HIAs;</td>
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<tr>
<td><strong>Weaknesses</strong></td>
</tr>
<tr>
<td>• Limited scope as only health impacts can be assessed. HIAs are probably most usefully deployed as part of other wider tools such Environmental Impact Assessment and Strategic Environmental Assessment;</td>
</tr>
<tr>
<td>• Quantified methodologies can only be applied to the small number of risk factors for which there are well-defined dose-response, concentration-response and resource-response functions;</td>
</tr>
<tr>
<td><strong>Opportunities</strong></td>
</tr>
<tr>
<td>• Inclusion of HIA as part of the legal requirements of Strategic Environmental Assessment is an opportunity to influence developments in other sectors and a platform for cross sectoral dialogue at a very early stage of development. Health inclusive Strategic Environmental Assessment can help identify opportunities and adopt actions to improve health;</td>
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<tr>
<td>• Methodology being proposed to link HIA with ecosystem service assessment;</td>
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<tr>
<td><strong>Threats</strong></td>
</tr>
<tr>
<td>• Presenting single, summary metrics as part of HIA can obscure the complexities, uncertainties and qualitative judgement underlying the production of HIA studies. Policy makers can be tempted to ignore these complexities if only a summary measure is presented without discussion of assumptions and limitations and analysis of intensity of impacts across different health areas</td>
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### 3.5.3 Social impact assessment

<table>
<thead>
<tr>
<th>Strengths</th>
<th>Weaknesses</th>
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<tbody>
<tr>
<td><strong>Social impact assessment</strong></td>
<td>• Provides a useful complement to monetary approaches as can assess and value non market social (and other impacts) which are not readily addressed by CBA. • Aims to use non-monetary methods for predicting future costs and benefits. Where there are appropriate data sources (those which can be collected frequently) it allows for prediction of long term impacts but can also provide monitoring of short-term impacts which can provide a continual source of evaluation or check on the direction of forecasts made about social impacts. Also considers the second and higher order impacts and of cumulative impacts. (IAIA, 2003). Involves stakeholders in process.</td>
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<tr>
<td><strong>Social accounting</strong></td>
<td>• Provides a useful framework and common metrics for reporting organisational outcomes. Encourages stakeholder engagement to build accountability.</td>
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<td><strong>Social return on investment</strong></td>
<td>• Purports to be a more sophisticated version of CBA which offers a useful way of integrating assessment of market and non-market economic, social and environmental outcomes in a participatory manner. • Applies monetary values to as many social outcomes as possible to make them more visible to funders and decision makers. In principle this should make it easy to compare the social impacts of different organisations or interventions</td>
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<tr>
<td><strong>Outcome mapping</strong></td>
<td>• Focuses mainly on assessing outcomes relating to individual agency and organisational capacity. These issues are crucial for ensuring positive impacts from policies and interventions but are often neglected by conventional impact assessments. This will be a particularly important issue for decentralised energy options. • Methodology relevant for design stage, midterm or ex-post assessment</td>
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<tr>
<td>Social Impact Assessment Frameworks</td>
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<tr>
<td><strong>Social impact assessment</strong></td>
<td><strong>Social accounting</strong></td>
</tr>
<tr>
<td>Opportunities</td>
<td>There may be useful learning from some of the research methods and the participatory approaches</td>
</tr>
<tr>
<td>Threats</td>
<td>Even though the method combines monetary and monetary assessment there is still a risk that a greater value will be placed on social outcomes which have a monetary value rather than others such as participation, personal agency, organisational capacity etc.</td>
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### 3.5.4 ‘Social appraisal’ methods

<table>
<thead>
<tr>
<th>Staged Multi-criteria Assessment (MCA)</th>
<th>Social Multi-criteria Evaluation (SME)</th>
<th>Qualitative Participatory Deliberation (QPD)</th>
<th>Multi-criteria Mapping (MCM)</th>
<th>Q Method (QME)</th>
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<tbody>
<tr>
<td><strong>Strengths</strong></td>
<td><strong>Weaknesses</strong></td>
<td><strong>Strengths</strong></td>
<td><strong>Weaknesses</strong></td>
<td><strong>Strengths</strong></td>
</tr>
<tr>
<td>Major strengths (where these apply) are shared with CBA and DMV – in the capacity to close down and so justify particular decisions. That this is associated with a greater degree of flexibility and transparency can under some views enhance these benefits. The more fully the method is combined with participatory processes, the more this applies.</td>
<td>Similar to DMV, the compromise on different evaluative imperatives embodied in MCA mean that it can ‘fall between stools’. With respect to the aim of ‘opening up’, it lacks the flexibility breadth, transparency of MCM and the robustness and social learning associated with participatory approaches. But it also lacks the power to justify decisions, often associated with a monetary metric.</td>
<td>A diverse array of methods with different detailed strengths. If conducted in an open fashion, all tend to display high flexibility and robustness in relation to participants’ interests. Depending on how undertaken, can also promote strong learning. But if closed down in order to deliver legitimisation, justification or consensus, then these ‘strengths’ (where seen as such), can bring reduced legitimacy, transparency and learning.</td>
<td>Major strengths lie in the flexibility and broadening out of the scope of appraisal and the opening up of policy debates. Quantitative data is substantiated by rich qualitative material and a rigorous and transparent picture of uncertainties, ambiguities and divergent values. The process also fosters significant learning.</td>
<td>A very effective way to scope out underlying issues and illuminate how different perspectives and aspects relate to each other – possibly revealing associations and distinctions that are entirely unexpected. In this sense, Q method can be a powerful aid to opening up decisions. Q is also a relatively quick and easy to implement.</td>
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<tr>
<td>This can display essentially the same strengths as DMV, but with additional scope for stakeholder engagement, transparency and social learning. But it is less strong (where this aim applies) with regard to the use of the familiar and ostensibly objective metric of monetary value.</td>
<td>In many ways, this displays a similar set of compromises to MCA, but weighted somewhat more favourably on the side of opening up and learning and somewhat less so on the side of closing down and justification decision.</td>
<td>Under a viewpoint prioritising quantitative procedure as an end in itself, the purely qualitative form of these approaches is a serious disadvantage. Conversely, where value is attached to rigorous exploration of uncertainties and ambiguities and the opening up policy debates, then the frequent focus of these methods on closure and consensus can also be a serious disadvantage – in the extreme broadly comparable to CBA.</td>
<td>The main weakness of MCM (where this quality is seen as such), is the fact that a technique aimed at opening up policy debates, can have the effect of destabilising closure and the justification of particular decisions. The rigorous exploration of different aspects of appraisal required in this process can also be demanding for both analysts and participants.</td>
<td>Q is not primarily geared towards a focus on concrete policy actions, but rather at understanding the issues and perspectives that determine how these are viewed.</td>
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<tr>
<td>Staged Multi-criteria Assessment (MCA)</td>
<td>Social Multi-criteria Evaluation (SME)</td>
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<tr>
<td><strong>Opportunities</strong></td>
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<tr>
<td>• Due to the focus on closing down, there may under some views be a strong responsibility to link such approaches with other methods that open up wider perspectives to ensure more, democratically accountable policy making.</td>
<td>• The relative neutrality of these methods allow greater potential for articulation with other methods than above. But – to the extent that most remain utilitarian – these methods tend to exclude a full account of in-principle issues that are covered in QPD or MCM approaches.</td>
<td>• These techniques can be used as an overarching way to take account of the outputs of any other approach discussed here.</td>
<td>• The flexibility of MCM in addressing in a balanced way such a diversity of input, issues and perspectives, means it offers an especially strong tool for integrating the outputs of a range of other methods.</td>
<td>• Q can offer a powerful tool to inform the recruitment of participants in participatory processes and the framing of issues and options for deliberation and analysis alike.</td>
</tr>
<tr>
<td><strong>Threats</strong></td>
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<tr>
<td>• If sensitivity analysis is used to convey the implications of divergent perspectives in the reporting of results, then this can be less threatening than conventional externalities assessment in respect of the issues raised for CEA.</td>
<td>• The problem identified with respect to opportunities to the left presents a threat of a lesser degree but similar kind to that discussed above.</td>
<td>• Where the design, framing or implementation QPD are used to close down policy debates and justify particular decisions, then they can present essentially the same threats (albeit in different form) that are noted for CEA.</td>
<td>• There is no guarantee that MCM will be implemented in accordance with its own driving principles. Like other kinds of MCA and SME, it can be used in a narrow fashion to close down decisions.</td>
<td>• The complex and ambiguous configurations of perspectives that can be identified in Q, can be difficult to interpret or to relate in concrete ways to practical policy choices.</td>
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</table>
3.6 Recommendations

- An updated database of existing empirical valuation studies for ecosystem services would support cost-effective assessments.
- At a macro-economic level, the use of Input-Output (IO) analysis and Multi-Region IO models with ecosystem services to provide economy-wide and sector specific economic impact analysis of alternative uses of natural resources should be further explored. This could yield information in terms of outputs, employment and GDP.
- The developed IO tables should be combined with Life Cycle Analysis and footprinting to support macroeconomic assessments (UKNEAFO 2014).
- More systemised guidance will be required about (a) the likely social outcomes and impacts associated with future energy options (b) relevant indicators and data sources and (c) relevant tools to assess and value them;
- Complementary quantitative and qualitative valuation methods can be usefully combined to capture the full range of possible social impacts;
- A wide range of social indicators can be assessed in quantitative, rather than monetary terms (through percentages, numbers, subjective rankings). This obviates the need for monetary assessment to compare social outcomes, but means it is still difficult compare social and other types of outcomes;
- A number of wellbeing and sustainable development indices and data sources exist that combine multiple economic, social and environmental indicators;
- Strategies and outcomes related to individual behaviours, organisational capacity and relationships between actors need to be assessed as these will shape and mediate the impact of future energy options on energy use and carbon emissions particularly in relation to distributed energy scenarios where a range of new actors might be involved, but also more generally to maximise energy savings from technological interventions;
- The value of spending resources to attribute outcomes and impacts to one intervention or actor in a statistically valid way is questionable given that change is a result of interactions between multiple actors and factors.
- The choice of an evaluative framework will be very different depending the focal interests.
- An MCM-style approach would arguably be more suited to addressing divergent interests in society and democracy at large by trying to secure robustness, accountability and legitimacy, than a cost-benefit analysis approach.
4 Issues in Impact Assessment

4.1 Limitations of monetary valuation and its use in policy assessment

Using economic metrics to value environmental benefits attracts controversy, with opponents questioning the entire underlying philosophy of placing a monetary value on a natural resource, and also criticising the techniques involved, particularly those employing stated preference techniques and discounting (see Ackermann and Heinzerling, 2004, for an overview). Sandel (2012) summarises moral arguments against placing a monetary value on certain environmental goods: i) market valuation makes having or not having money matter more which is particularly damaging for social goods such as health; ii) certain moral and civic goods are diminished or corrupted if given a market value, or bought or sold; and iii) market valuation can crowd out or corrupt other ‘intrinsic’ pro environmental or social values people hold about the good or service.

In cases where market values approximate reasonably well for impacts, revealed and stated preference techniques may be appropriate. However, there are also goods and services where this assumption is not appropriate. In addition researchers have challenged the basic axioms underpinning the use of market prices as a proxy for value, by showing that ‘consumers’ do not always act in a self-interested way in the neoclassical economic sense, that they may also be motivated by altruistic values and that socio-psychological factors, such as social norms, may be important predictors of behaviour (e.g. Schwartz 1977; Cialdini 1990). Social Practice Theory has sought to show how individual’s decisions are not autonomous, but embedded in wider social practices involving shared meanings, know how, materials and rules and procedures (Shove, 2010). Moreover, single preferences do not generally take account of the interactions between different impacts: for example, people might feel more strongly negative about a project that imposes both environmental and social costs than would be estimated by adding separate valuations of the two effects. Further the use of discount factors to assess net benefit from flows of costs and benefits over time involves subjective prediction of the future, rather than objective measurement.

A specific limitation of the stated preferences approach is that it involves the collection of primary data which is susceptible to problems such as non-response bias, interviewer bias and information bias (HM Treasury 2013, 2011 b). However, extensive work by numerous psychologists spanning many decades (see, e.g. Diener et al. 1999; Kahneman et al. 1999), indicates the great progress made in the measurement of subjective self-reporting such that it has become recognised as an empirically adequate and valid approximation for individually experienced welfare.

Even where the collection of monetary values is appropriate, there are further issues related to the methods that apply them in decision making. Cost-benefit methodologies share a central common feature, in that the issues in relation to which all options are appraised, are typically addressed by means of the single (apparently simple) metric of monetary value. The reference to ‘externalities’ reflects the fact that many of the most important monetary values involved, are ‘external’ to existing markets (Ottinger 1993). They must therefore be elicited, inferred or modelled by means of various technical procedures (Hanley & Spash 1993). In cases where the issues under scrutiny are those related to provision of ecological services, then the resulting ‘ecosystem service frameworks’ constitute a member of this broad family of methods (Common & Stagl 2005).

Unfortunately, for the purpose of policy assessment, the limitations of applying economic metrics to certain ecosystem service benefits, or to other social and health impacts, presents a massive practical inconvenience because the various relevant impacts and measures are typically ‘incommensurable’ (Stirling 2003). In other words, they are ‘apples and oranges’ – intrinsically not subject to aggregation under a single metric. So, what cost-benefit analysis (CBA) apparently offers in this regard, is a way to reconcile this fundamental impossibility (O’Neill 1993). By assigning a single distribution of monetary values across all such issues, it renders alternative policy options not only roughly comparable in broad qualitative terms, nor just as subject to a neat approximate ordinal (i.e. relative) sequence; but as apparently amenable to an unambiguous ordering on a precise cardinal (i.e. quantitative) scale – expressing the exact ratios and intervals separating the overall merits of each policy option.
It has been demonstrated (in Nobel Prize winning analysis in the field of welfare economics and rational choice theory underlying these methods), that even the aspiration of a single definitive ordering of incommensurable issues is, in a plural society, not only impossible in practice to guarantee – but inherently meaningless even to contemplate (Arrow 1963) (Kelly 1978; MacKay 1980). The apparent policy utility therefore comes at the price of serious inaccuracy in relation to real-world complexities, uncertainties and subjectivities.

Nonetheless, there is the risk that CBA can still be used inappropriately where it is simply assumed that the aim is automatically to comply with the aims of clients (or wider incumbent interests in any given controversy), whether this be government, business or an NGO (Stirling 2008). In such cases, it has been argued that few policy rhetorics are more potent than one expressing unequivocal confidence over the aggregation of incommensurable issues and expressing these in the familiar and highly operational terms of monetary value, without acknowledging any uncertainty or ambiguity (Collingridge 1982). But, the argument continues, it follows from this same apparent strength under one view, that there exists a corresponding serious weakness under other views. These are, that CBA in all its forms, when used without proper acknowledgement of its scope and limitations, can serve effectively to suppress uncertainty, deny ambiguity and force one particular evaluative perspective at the expense of others – thus undermining both science (which it misrepresents) as well as democracy itself (Getzner et al. 2005).

Deliberative monetary valuation (DMV) also produces as an output the same kind of discrete arrays of monetary values that are typically produced in CBA. So it is in principle subject to the same kinds of concern. In this regard, a review by Stagl for DEFRA concludes “deliberative monetary valuation is most suitable for the appraisal of projects whose impacts are relatively well understood, where the impacts do not reach far into the future, and which do not affect complex ecosystem services such as biodiversity” (Stagl 2007).

In some cases it may not be possible to capture the full value through monetary valuation alone. As the HM Treasury (2013) notes that there are some social impacts such as “health, family and community stability, educational success, and environmental assets [which] cannot simply be inferred from market prices” (ibid 2013:57). It goes on to outline techniques for valuing non-market impacts, and some typical applications such as time-savings, health benefits, prevented fatality, design quality, and the environment. It points out that these approaches can be complex but are equally as important as market impacts. It, nevertheless, acknowledges that “there may always remain significant impacts that cannot sensibly be monetised”. In addition, there are stakeholders with principled objections to placing a monetary value on some social goods where their value may be considered intrinsic and incommensurable.

Where outcomes cannot be sensibly monetised – as discussed above - the Treasury Green Book (2013 p58) states that these can be quantified in non-monetary units or described in qualitative terms.... ‘Whatever the case, material costs and benefits that cannot be valued in monetary terms should clearly be taken into account in the presentation of any appraisal or evaluation’.

Beyond government a range of other non-monetary, subjective and/or participatory valuation methods have therefore been developed to assess and value social (or environmental) outcomes (Plymouth Marine Laboratory and Marine Biological Association, 2013). These often involve qualitative research methods such as, semi structured interviews, focus groups and structured workshop based methods, case studies, observation, participatory rural appraisal, mapping, needs assessment, tree diagrams among others.

4.1.1 Which ecosystem services are currently amenable to monetary valuation?

There are some ES for which it is feasible and appropriate to use monetary valuation, although as discussed in the previous section, the limitations of particular methods should be taken into account in any assessment. Monetary valuation is particularly applicable to provisioning and carrier services (such as food, raw materials and transport), for which market prices are available. Recreation is also amenable to monetary valuation (usually through revealed preference methods). It may also be possible to obtain monetary values for certain regulating services. For example, there has been much recent effort focused on determining the rates at which habitats sequester carbon, and in developing carbon prices. Flood and erosion prevention have also been the focus of empirical studies.

Monetary valuation is not appropriate for most cultural services (with the notable exception of recreation). Although valuation is technically possible through stated preference assessments, it is
questionable (and has long been debated in the literature) whether the outcomes are meaningful, as discussed in section 3.4 above, so their use, in an actual decision context, may be difficult to justify. Also, on a practical level, gathering monetary values that capture all aspects of cultural and non-use values is likely to be particularly expensive and time consuming, and, due to the context-specificity, the values obtained may not be amenable to benefits transfer. Alternative methods for the assessment of cultural services are therefore required.

Table 4.1 (below) illustrates whether particular categories of ecosystem services, as described by the CICES classification, are amenable to monetary valuation, and highlights that monetary valuation is only straightforward for limited types of ecosystem service. Many services are partially or potentially amenable to monetary valuation, and the table indicates where, at present, there is a lack of necessary data to consistently monetise all services within a particular class, or, in some contexts (notably the marine environment), incomplete understanding of how natural processes link to societal benefits. Alternative methods that can be used to assess social and economic impacts, and to consider them in policy decisions, are discussed in the following chapters.

It should also be noted that even where monetary valuation is appropriate, primary data is often lacking and may be costly to obtain. For this reason, the benefit transfer approach is often used, in which a value obtained in one location is transferred to another site. However, this approach is not without its limitations, and can be subject to significant error when the site for which a value exists is a poor match for that under consideration (Plummer, 2009). Errors due to sampling effects and extrapolation can be sufficiently large as to undermine any decisions that might be based on the outcome (Eigenbrod et al., 2010).

Table 4.1. Amenity of ecosystem services (as summarized from CICES) to monetary valuation

<table>
<thead>
<tr>
<th>Section Division</th>
<th>Class/examples</th>
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</thead>
<tbody>
<tr>
<td><strong>Provisioning</strong></td>
<td></td>
</tr>
<tr>
<td>Nutrition</td>
<td>Cultivated crops</td>
</tr>
<tr>
<td></td>
<td>Reared animals and their outputs</td>
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<tr>
<td></td>
<td>Wild plants, algae, animals and their outputs</td>
</tr>
<tr>
<td></td>
<td>Animals, plants and algae from in-situ aquaculture</td>
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<tr>
<td></td>
<td>Surface and ground water for drinking</td>
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<tr>
<td>Materials</td>
<td>Fibres and other materials from plants, algae and animals for direct use (including agriculture) or processing</td>
</tr>
<tr>
<td></td>
<td>Genetic materials from all biota</td>
</tr>
<tr>
<td></td>
<td>Surface and ground water for non-drinking purposes</td>
</tr>
<tr>
<td></td>
<td>Ground water for non-drinking purposes</td>
</tr>
<tr>
<td></td>
<td>Metallic and non-metallic minerals</td>
</tr>
<tr>
<td>Energy</td>
<td>Plant- and animal-based resources</td>
</tr>
<tr>
<td></td>
<td>Renewable abiotic energy sources</td>
</tr>
<tr>
<td></td>
<td>Non-renewable energy sources</td>
</tr>
<tr>
<td><strong>Regulation &amp; Maintenance</strong></td>
<td></td>
</tr>
<tr>
<td>Mediation of waste, toxics and other nuisances</td>
<td>Bio-remediation by micro-organisms, algae, plants, and animals</td>
</tr>
<tr>
<td></td>
<td>Filtration/sequestration/storage/accumulation by micro-organisms, algae, plants, and animals</td>
</tr>
<tr>
<td></td>
<td>Filtration/sequestration/storage/accumulation by ecosystems</td>
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<tr>
<td></td>
<td>Dilution by atmosphere, freshwater and marine ecosystems</td>
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<tr>
<td></td>
<td>Mediation of smell/noise/visual impacts</td>
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<tr>
<td></td>
<td>By natural chemical and physical processes</td>
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<tr>
<td>Mediation of flows</td>
<td>Mass stabilisation and control of erosion rates</td>
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<tr>
<td></td>
<td>Buffering and attenuation of mass flows</td>
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<tr>
<td></td>
<td>Hydrological cycle and water flow maintenance</td>
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<td></td>
<td>Flood protection</td>
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<td></td>
<td>Storm protection</td>
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<tr>
<td></td>
<td>Ventilation and transpiration</td>
</tr>
<tr>
<td></td>
<td>By solid (mass), liquid and gaseous (air)flows</td>
</tr>
<tr>
<td>Maintenance of physical, chemical, biological conditions</td>
<td>Pollination and seed dispersal</td>
</tr>
<tr>
<td></td>
<td>Maintaining nursery populations and habitats</td>
</tr>
<tr>
<td></td>
<td>Pest control</td>
</tr>
</tbody>
</table>
4.2 Equity and justice

The concept of justice is generally understood as consisting of the following elements (Preston et al., 2014)

(a) Distributive justice: which is concerned with how resources, benefits and burdens are allocated between or within countries or between generations

(b) Procedural justice: which is concerned with the fairness and transparency of the processes used to make decisions about societal goals i.e. ‘who decides’ and ‘who participates’ in decision making processes.

An important related concept is structural barriers: which relate to the different capabilities and socio-economic conditions that people face and hence their ability to participate in and benefit from policies and programmes in the first place (Bulkeley and Fuller, 2012).

In this context, i.e. understanding the multiple impacts of different energy options, all three elements are relevant. Different energy options will have differing distributions of benefits and costs depending on the nature of: the energy option, where it is situated, the policy framework and financial incentive structures, the delivery agents, and end users (ie whether vulnerable, disadvantaged, tenure etc).

Similarly, different groups of stakeholders are likely to be involved in choices about the energy options. Different types of end users will face differing sets of structural barriers to access and benefit from energy options.

In addition to moral arguments about the importance of equity, there is evidence that perceptions of fairness do influence how people perceive the legitimacy of a decision-making process, and that a fairer process and outcomes will increase acceptance of the outcome (Gross 2007).

To allow distributional justice concerns to be incorporated in decision-making, assessment frameworks need to be able to account for distributional impacts. Details of how this is best achieved depend on the scale of the decision, the nature or the intervention, who and what it affects, over what time period. So, for example, assessments of energy choices which have impacts over a very long time scale, e.g. nuclear energy, will need to pay particularly close attention to inter-generational equity and discounting issues.

Distributional justice is already factored in to UK government decision making in places (see Walker (2010) for more on the inclusion of distributional justice within environmental decision making). For household energy policy decisions, DECC is formally responsible for “fairness – making sure the costs and benefits of our policy are distributed fairly so that we protect the most vulnerable and fuel poor..."
SPLiCE Phase 1

Investigation of methods for comparing impacts and prototype framework for comparison

households” (DECC, 2014). However, this aim of distributional fairness has not been entirely met. An analysis of the distributional effects of government energy policy on households, showed that while the overall impact of policies on efficiency and renewables will be to lower household energy bills by 2020, the poorer will benefit less than other groups (Preston et al., 2013).

In some cases, policies which have an unjust distributional effect may be still be thought valuable. For example, Feed In Tariffs (subsidies) for solar PV are paid for by all energy bill payers, but primarily benefit wealthier households (Grover, 2013). However, this policy has retained government support, a central argument being that this policy will help reduce PV prices and deliver wider benefits as the technology becomes more affordable. Looking at the distributional impacts of one policy tool, or other decision, in isolation, or over a short period of time, is not the only relevant level of analysis.

Procedural justice has transparency and fairness of processes as its central concern. For both principled and practical reasons, it should be an important aspect in the design of any assessment process. All other things being equal, procedural justice would be more easily demonstrated by socially focussed methods of impact assessment, such as Multi Criteria Mapping, which tend to include a wider variety of stakeholders and which open up the decision being made.

4.3 Scale, trade-offs and uncertainty

The issue of scale is important within impact assessment: a broad generic evaluation of the impacts of a technology at a national scale is unlikely to identify site specific issues that are highly dependent on individual social and environmental contexts. These latter issues will, however, factor heavily during the planning and consents process as attempts are made to implement national policy. This suggests that, in practice, there is the need for a two stage process. A generic assessment will serve to highlight the key environmental, social and economic impacts of a technology, and will also show what evidence exists, and from where (geographically) it has been obtained. A subsequent area-based assessment (potentially requiring the commissioning of new research) will identify specific issues that could affect acceptance of those technologies in particular locations.

Obtaining an adequate understanding of context-specific issues requires using techniques that appropriately accommodate the different ways different individuals, organisations and communities value different services and impacts. The more open-ended and deliberative methods (such as participatory deliberation, the Q method, and multi-criteria mapping – see Section 3.4) seek to take account of diverse value constructs. Monetary methods generally assume a narrower framing, based on the aggregation of individual, self-regarding preferences to provide a measure of social welfare (Kenter et al., 2014), although deliberative approaches to valuation do exist, which seek to address some of these limitations.

Related to scale is the matter of trade-offs. A comprehensive impact assessment would identify different costs and benefits at individual, community and societal scales, and the issue then becomes how to trade these off against one another in reaching a final decision. Multicriteria analysis (MCA) provides a formal framework and an audit trail (DCLG, 2009) and so has advantages over informal approaches to making trade-offs. There is scope to include stakeholders in MCA, allowing a range of actors with different perspectives to weight the importance of the diverse impacts at different scales.

Even where best practice is followed at all stages, issues of scale are addressed and trade-offs made in an inclusive and transparent manner, uncertainty is inherent in any decision, due, not least, to incomplete knowledge and to the effect of external factors also acting on ecosystem goods and services. The cumulative effects of pressures on ecosystems and the services they provide are particularly poorly understood. To address this issue, the Green Book (HM Treasury, 2013) suggests methods for assessing uncertainty, including through sensitivity analysis, scenarios and Monte Carlo approaches.

4.4 Recommendations

- Monetary valuation can be used as a metric where feasible and appropriate, although the limitations of particular methods should be taken into account in any assessment.
• Monetary valuation is particularly applicable to provisioning and carrier ecosystem services (such as food, raw materials and transport), for which market prices are available. Recreation is also amenable to monetary valuation (usually through revealed preference methods),

• It may also be possible to obtain monetary values for certain regulating ecosystem services. For example, there has been much recent effort focused on determining the rates at which habitats sequester carbon, and in the development of carbon prices. Flood and erosion prevention have also been the focus of empirical studies.

• In many cases monetary valuation will not be valid or appropriate for ecosystem services or impact studies. Sometimes this will be due to lack of data. Other metrics may be used. Often it will be important to understand the context in which monetary valuation is not possible to support decision making.

• Monetary valuation should not be attempted for intermediate/supporting ecosystem services, as this will lead to double-counting. The value of these services can be captured instead as assets within the National Capital Committee framework.

• Monetary valuation is not appropriate for most cultural ecosystem services (with the notable exception of recreation). Although valuation is technically possible through stated preference assessments, it is questionable whether the outcomes are meaningful. Their use in an actual decision context may be difficult to justify. On a practical level, gathering monetary values that capture all aspects of cultural and non-use values is likely to be particularly expensive and time consuming, and, due to the context-specificity, the values obtained may not be amenable to benefits transfer.

• Alternative methods for the assessment of cultural services are therefore required. Monetary based metrics (such as monetary valuation) should be exploited where they are available but non-monetary measures should not be ignored, nor should attempts be made to force everything to have a monetary value where this would result in key costs or benefits not being assessed correctly; this can produce poor decisions that damage the environment or society but can be justified by the perpetrators.

• Investigation is needed of the variation in values set by different individuals and stakeholder groups. The strength and stability of values will determine the level and style of resistance to different energy proposals.

• Assessments of the impacts of different energy options should also include distributional analysis between and within generations.

• Procedures to assess the distributional impacts of some types of energy decisions already exist (e.g. for government household energy policy), but should be checked for completeness and extended to others.

• Procedural justice concerns should be incorporated into all impact assessment processes, to the extent that this is possible.

• Where procedural justice is a key concern in decision-making, social impact assessment frameworks are likely to be a good choice.
5 Prototype evaluation framework

5.1 Decision contexts and approaches

The appraisal process, in which decision makers (or their commissioned agents) search out and weigh up available knowledge is designed to formalise the provision of information for decision making (Russel et al., 2014). The institutions involved in decision making often have different priorities depending on their objectives. As such they have developed specific models for the utilisation of information, depending on whether they are seeking a strategic overview of drivers and impacts (EU and UK level), a balance between more specific development and environmental protection needs (subnational regional level), or must implement policy ‘on the ground’ with a potentially stronger influence of stakeholders (local level) (Jordan and Russel (2014), and references therein).

The scope of decision making includes different sectors and institutions working across different time frames and boundaries with different objectives, capacities, analytical processes and decision support tools (Russel et al., 2014). For example Impact Assessment (IA) is used at national policy level, with Strategic Environmental Assessment (SEA) applied regionally or sectorally and Environment Impact Assessment (EIA) a key tool in local planning. Different decision contexts may be more amenable than others to the inclusion of environmental, social, and ecosystem services information. IA primarily focuses on reducing regulatory burdens on particular sectors, and the economic aspects tend to crowd out social and environmental issues and cross-cutting initiatives like ecosystem services approaches (Russel et al., 2014). As has been clearly demonstrated in practice, the nature of SEA and EIA, however, is more accommodating to environmental and social information.

Of 249 IAs fitting sustainable development criteria, 80% treated economic information with medium or high rigour, but half treated social and environmental impacts with low rigour or not at all. This is not to say that there was evidence of significant environmental impacts being overlooked, but omissions of less significant impacts were common (Tinch et al., 2014). An analysis for the UK National Ecosystem Assessment Follow On showed further shortcomings in the use of environmental information in IAs for environmental policies or policies related to the environment (including agriculture, energy, and transport). Of 53 IAs from the period 2008 to 2012, 34% did not refer to any ecosystem or environmental knowledge on impacts, and a further 40% made at best a weak evaluation of environmental information (Russel et al., 2014). This compares to 51% of Strategic Environmental Assessments and 62% of Environmental Impact Assessments from the same period which were judged to have strong environment framing and evaluation (Russel et al., 2014).

It is also the case that while UK Government guidance highlights that environmental and social impacts for which monetary metrics cannot be obtained should remain a key component of any appraisal or evaluation (HM Treasury, 2013), this does not appear to be consistently applied in practice. There remains quite widespread agreement among practitioners themselves that failure to monetise environmental and social impacts risks their being underweighted in appraisal, and cultural services can tend to be overlooked (Tinch et al., 2014).

The amendments to the Treasury guidance are relatively recent, and the uptake of new approaches to appraisal may require cultural changes within institutions as well as modifications of individual behaviour (Russel et al., 2014). Also, policy drivers vary: the priority since 2008 has been reducing regulatory burden (as was also seen in early 1990s), while during the intervening period appraisal was framed more in the context of improving regulatory quality (Russel et al., 2014). Thus, the political context is also likely to influence the practice of policy appraisal.

Any such a framework thus enables fulfilment of the key objective of UK environmental policy to understand and value natural capital and ecosystem services (HMG, 2011).

5.2 Conceptual framework for the assessment process

The proposed assessment process has three key elements:
1. the process utilises the Natural Capital Committee approach as an overarching framework to identify the natural capital and ecosystem services information that should be considered;
2. macro-economic, social impact, health impact assessment methods can be used alongside to generate additional relevant data;
3. multi-criteria assessment methods should be used to evaluate the resulting environmental, social and economic information.

The purpose of the framework is to suggest a pathway and methods for evaluating environmental, social and economic impacts that have been reported in a range of metrics. The scope of the impact assessment (such as geographical boundaries, lifecycle stages, and whether the context is a generic assessment of particular technology or highly site-specific evaluation) is not considered within the framework, as this will be set at the preceding stage when the parameters for gathering the required evidence are assigned.

The ecosystem services framework proposed by the Natural Capital Committee (NCC) (Chapter 2, Appendix 1) has been recommended because it builds on the strengths and objectives of key national, regional and international work in this field. An additional strength of the NCC framework is that it includes stocks (the quality and quantity of habitats) alongside flows (the ecosystem goods and services generated) thus integrating existing natural resource management frameworks with developing approaches. The inclusion of stocks also provides a means, by inference, of acknowledging those ecosystem services that, at present, are rarely explicitly accounted for because the mechanisms of their delivery to people, and/or their relationship to the properties and functions of ecosystems are poorly understood. The NCC framework also allows for the inclusion of bio-physical and other non-monetary metrics. This is echoed in guidance for UK Government officials in appraising and evaluating policy, which encourages monetary valuation (and the associated cost-benefit analysis), but also recognises that this is not always appropriate or feasible (HM Treasury, 2013; BIS 2013). These conclusions are supported by the preceding chapters of this report.

Natural capital is one element of an impact assessment, and understanding the full implications of energy pathways and systems will also require consideration of wider health, social and economic impacts, possible methods for which are also discussed elsewhere in this report (Chapter 3, Appendices 6 to 8). There are established methods and best practice which should be applied when considering these different elements. For example, where cost benefit analysis is possible and valid it should be undertaken as a means of understanding key economic information, which can then be fed into a wider assessment.

The conceptual framework for the evaluation process (Figure 5.1) illustrates how the assessment of impacts on ecosystem services, health, society and the economy will produce outputs in different metrics, and shows the evaluation frameworks available for comparing impacts measured using these metrics. Changes in natural capital stocks and flows can, in some instances, be measured in monetary terms, although biophysical metrics, non-monetary values and qualitative and narrative information will feature heavily in any comprehensive assessment. Health and social impact assessments similarly produce outputs in mixed metrics; only economic assessments will be dominated by monetary information. Thus the use of cost benefit analysis is limited to a small fraction of the evidence that will be generated by a comprehensive impact assessment. Conversely, social appraisal methods (such as the multi-criteria, deliberative and Q method techniques described in Chapter 3 and Appendix 9) are able to accommodate information in any metric, including the outputs of cost benefit analysis. The outputs of any social appraisal can also be accommodated into existing impact assessment frameworks. For example the IA structure requires specification of key non-monetised costs and benefits, and a social appraisal process will permit these to be formally identified and potentially prioritised, weighted, ranked, or scored depending on the method used.

Entirely novel approaches have also been proposed that seek to ensure equal treatment of economic and non-monetary information and the accommodation of distributional issues (who gains and loses from environmental change and policy responses). For example, the Balance Sheet approach within the UK National Ecosystem Assessment Follow On (Turner et al., 2014) suggests three complementary assessments: i) a monetary cost benefit analysis which includes distributional impact analysis; ii) regional and local financial impacts and policy analysis (considering, for example, local jobs, community identity, related financial multiplier effects); and iii) non-monetary trade-off analysis to accommodate values across wider society (such as intrinsic value of biodiversity and cultural assets value). However, this approach remains a conceptual framework, and has not been developed into a practical methodology.
Multi-criteria assessment (MCA) is recommended for the purposes of SPLiCE because it is widely used and (when interpreted in its broadest sense, including modifications such as multi-criteria mapping, as is the intention here), MCA provides scope for assessment by individuals or groups and for different degrees of direct stakeholder involvement. The Better Regulation Framework Manual (BIS, 2013) and the Green Book (HM Treasury, 2013) note the potential use of MCA in evaluating non-monetary impacts, and supporting guidance is available (DCLG, 2009). A detailed evaluation of the use of MCA in the assessment and integration of social impacts into valuation and appraisal in the UK policy context was also recently commissioned by DEFRA (Maxwell et al., 2011).

The objective of MCA is to determine the relative importance of different elements within the decision context (Mendoza et al., 1999). MCA provides a systematic framework to synthesise a wide range of information and raise awareness of trade-offs, thus helping decision makers to make better use of all available information in evaluating, and choosing between, alternatives (Linkov et al., 2006). MCA can be used at the appraisal stage as well as to evaluate policies and projects retrospectively (DCLG, 2009), and MCA approaches also fit within an adaptive management framework (Linkov et al., 2006). A further advantage of MCA is that it does not require consensus, but instead allows a joint conclusion to be reached for which each individual involved in the process records his or her own judgements on the relative importance of the criteria or the rankings of options (Mendoza et al., 1999).

The structured framework provided by MCA has several advantages over informal judgement. In particular, it provides an open and explicit approach based on established techniques with a defined audit trail, and the objectives and criteria used in the decision can be analysed and amended as required (DCLG, 2009). MCA thus avoids the dangers of informal approaches, which include the increased risks of generating the wrong decision or of failing to gain public acceptance due to a lack of transparency in the decision process (Mendoza et al., 1999). In many decision contexts this requirement to explain how decisions were reached is crucial, and the ability of MCA to track the decision process and separate its elements makes it an ideal communication tool (Mendoza et al., 1999).

MCA is not without its limitations, not least the large cognitive burden associated with holistic appraisal of multiple complex options (Montibeller and Franco, 2010) and that, unlike cost benefit analysis, it cannot show whether there is a net gain in welfare (DCLG, 2009). There are also particular strengths and weaknesses of specific MCA methods, particularly related to their level of stakeholder involvement, some of which have already been highlighted in Chapter 3 and Appendix 9.

The choice of pathways within the proposed evaluation framework is context dependant and will need to be tailored to the questions posed and the purpose for which the answer is sought, as well as to the resources available. The remaining sections of this chapter discuss MCA techniques and their application under different circumstances; provide further details of how this framework might accommodate economic, social and environmental information (exploring integration with the Rapid Evidence Assessment process developed as part of SPLiCE Phase 1 (Smithers, 2015); and review possible approaches to the visualisation of outputs.

Engagement with decision-makers during development of the proposed framework included discussion with Government departments (DECC and DTI, as well as DEFRA) on the extent to which monetary and non-monetary appraisal methods are currently used. The proposed approach was also presented and discussed at a wider stakeholder workshop involving representatives from Government departments, statutory agencies, Non-Governmental Organisations and academia, at which a broad consensus on its suitability was obtained.
Figure 5.1. Conceptual framework for the impact evaluation process

**Natural capital assessment**

- **Natural Capital Stocks**
  - Geological, physical, chemical and biological components of major marine and land-use categories

- **Natural Capital Flows** (Ecosystem goods & services)
  - Intermediate/supporting and poorly understood regulating services
    - (e.g. biodiversity, primary production, larval and gamete supply, nutrient cycling, biological control)
  - Most cultural services
    - (heritage, cultural, aesthetic, symbolic, sacred/religious, existence, bequest)
  - Provisioning services
    - (nutrition, raw materials, energy)
  - Selected cultural services
    - (recreation, tourism)
  - Well understood regulating services
    - (carbon sequestration, control of flooding and erosion, water supply)

**Further assessment**

- **Health Impact Assessment**
- **Social Impact assessment**
- **Macro-economic assessment**

**Assessment metrics**

- **Bio-physical metrics**
  - (e.g. quantities, rates, condition indices), which should relate where possible to national environmental monitoring programmes
- **Non-monetary values**
  - Qualitative and narrative information
- **Monetary values**

**Evaluation framework**

- **Social appraisal methods**
  - Multi-criteria Assessment
  - Multi-criteria mapping
  - Social multi-criteria evaluation
  - Qualitative participation deliberation
  - Q method

**Cost Benefit Analysis**

- **Deliberative monetary valuation**

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5.3 Incorporating flexibility for different contexts

There is an extensive literature on MCA, which will not be reproduced here beyond an overview of the approach, the main steps of which are illustrated below (Figure 5.2), together with an indication of the interaction with a Rapid Evidence Assessment process. Key references for the policy context include the Green Book supplementary guidance on MCA (DCLG, 2009), which is designed to provide non-specialists with an understanding of MCA techniques and the resources required, and Maxwell et al. (2011), which focuses particularly on social impacts and wellbeing and also considers engagement and participation. There are many examples of MCA being carried out in the context of energy policy from which lessons for specific contexts can be learned. These include residential heating and domestic electricity consumption (Browne et al., 2010), alternative scenarios for the power generation sector (Diakoulaki & Karangelis, 2007), development of a national renewable energy strategy (Kowalski et al., 2009) and the strategic section of wind farms (Lee et al., 2009). Further reviews of the use of MCA in energy decision making are provided by Pohekar and Ramachandran (2004), Kowalski et al. (2009), Scott et al. (2012), and Kurka and Blackwood (2013).

Figure 5.2. The key stages in an MCA process (modified from DCLG, 2009)

Within an MCA framework, there is the possibility for different pathways depending on specific circumstances (in terms of policy scales, approaches, contexts, information needs and availability, timescales and resource levels). Central to MCA is the performance or impact matrix, which shows how each option being assessed performs for each of the criteria considered and will be discussed further in Section 5.3.1 below. At its simplest, an MCA process can involve informal evaluation of the performance matrix (i.e. the omission of steps 4a to 4c in Figure 5.2 above) as a means of...
differentiating at a high level between acceptable and unacceptable options or highlighting key issues. However (as described earlier in Section 5.2), such informal judgement is not recommended.

A more structured MCA process involves scoring the performance criteria such that all elements are on a consistent numerical scale (typically between 0 and 100) indicating the relative strength of preferences for each criterion (DCLG, 2009). These scores can simply be summed or averaged, or at the next level of complexity, a process of weighting is applied, to indicate the relative importance of each element in the decision (Linkov et al., 2006). The most straightforward approach to this weighting is through ranking or rating. In the former, a rank is allocated to each element to reflect its relative importance in the decision, while the similar process of rating involves assigning a score to each element such that the total for all elements adds up to 100 (Mendoza et al., 1999). Alternative methods for weighting include pairwise comparison techniques (such as the Analytic Hierarchy Process), which also have an increased analysis requirement. The most complex methods (such as multi-attribute models) further increase resource intensity. There are theoretical frameworks underlying these more advanced techniques that should be considered when deciding on an appropriate approach (Linkov et al., 2006). Commercial software packages (e.g. MCM, Definite, NAIADE) exist to facilitate MCA. These have a cost burden but their ease of use is intended to lead to time savings.

Best practice within MCA is to use a participatory process, as effective analysis requires accurate assessment of what changes are important and to whom, as well as the degree of relative change. As the Green Book (HM Treasury, 2013) notes the “weight to give to factors that are thought to be important by key players cannot be decided by ‘experts’”, and so the direct involvement of those key stakeholders is required. Ideally, these stakeholders should be involved in the framing of the MCA, and should have input into the choice of the criteria to be considered. Techniques such as the Q method or multi-criteria mapping can be used (in interview, focus group or workshop settings) to identify different priorities and concerns, which will illuminate different perspectives on why a policy might be considered positive or negative.

This idea of ‘opening up’ a strategy or policy area through the inclusion of stakeholder input is not far removed from existing approaches: consultation is a key element of both policy development and planning decisions. The deliberative and participatory techniques proposed provide a structured approach that has the same goal as consultation (understanding the response of stakeholders to a policy or action) but allows its outcomes to be better assessed and accommodated within the decision framework. A typical consultation process would identify a range of opinions and issues amongst different stakeholders. A formal deliberative process can document the relative strength of concerns, and map where these converge and diverge between different stakeholders, providing quantified outputs and broader insight.

The increasing involvement of stakeholders (particularly through a heuristic process that allows their input into the context and framing of the assessment as well as at the scoring and weighting stage) increases the transparency and robustness of the approach compared to informal or even structured assessment by an individual analyst (Figure 5.3a). Increasing the transparency and robustness of the outcome also increases the complexity of the task and the resource requirement. However, estimates from the University of Sussex suggest that participatory techniques such as MCM can be utilised for a cost in the range of £20,000 to £60,000. Within each approach, there is also the potential to select different methods which become progressively more resource intensive (Figure 5.3b). Advantages of these more complex methods include, in the case of multi-attribute models, improving the explicit treatment of uncertainty (Linkov et al., 2006).
5.3.1 Developing the performance matrix within the SPLiCE context

The criteria to be considered when assessing the impacts of an energy pathway or development will depend on the decision context. Ideally the criteria will be decided in a participatory process involving relevant experts and stakeholders and therefore it is not possible to design a universal matrix framework. However, the development of an extensive list of criteria, with appropriate indicators, from which context-specific performance matrices can be built would facilitate the impact assessment process.

There are a variety of sources of potential criteria and indicators. In the ecosystem services context, CICES (Haines-Young and Potschin, 2013a, Appendix 1) details specific ecosystem benefits that could form criteria, and, at the next level, more specific indicators for ecosystem services have been proposed in a number of studies (e.g. Hattam et al., 2014; van Oudenhoven et al., 2012).

Environmental indicators that link to national monitoring programmes (such as under the Water Framework Directive and Marine Strategy Framework Directive) are potentially particularly useful criteria, as they are likely to be supported by regular data collection. Sources of possible social welfare indicators include the Office of National Statistics (ONS, 2014) and, internationally, Organisation for Economic Co-operation and Development (OECD, 2013), both of which have developed wellbeing indicators that consider categories such as housing, income, jobs, community, education, health and...
safety. Further discussion of social welfare indicators is provided in Appendix 8. Again, linking with the ONS system suggests the potential availability of data, at least at a national level.

A range of criteria have been used in the performance matrices of existing energy-related MCA. Table 5.1 contains two examples, the first looks at forestry in the New Carbon Economy in Mexico for which indicators were developed by UK technical specialists and key informants in Mexico (Brown and Corbera, 2003). The second example seeks to compare the sustainability of different energy technologies and used a literature review as the primary means to identify indicators (Maxim, 2014). The indicators used in these studies do not overlap, illustrating how their development to date has been driven by the specific context, resources and approach of individual assessments. These examples also illustrate that existing MCA on energy themes has not been undertaken in the context of natural capital and ecosystem services, so appropriate typologies for performance matrices in this specific context require development.

### Table 5.1. Examples of indicators for economic, social and environmental criteria in relation to energy

<table>
<thead>
<tr>
<th>Forestry approaches in Mexico’s New Carbon Economy (Brown and Corbera, 2003)</th>
<th>Comparing the sustainability of energy technologies (Maxim, 2014)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Theme: Carbon</strong></td>
<td><strong>Theme: Economic</strong></td>
</tr>
<tr>
<td>• Net carbon sequestered</td>
<td>• Levelised cost of electricity</td>
</tr>
<tr>
<td>• Project internal rate of return</td>
<td><strong>Theme: Technical</strong></td>
</tr>
<tr>
<td>• Risk</td>
<td>• Ability to respond to demand</td>
</tr>
<tr>
<td>• Eligibility under CDM* criteria</td>
<td>• Efficiency</td>
</tr>
<tr>
<td><strong>Theme: Ecological</strong></td>
<td><strong>Capacity factor</strong></td>
</tr>
<tr>
<td>• Overall ecological value of the region</td>
<td><strong>Theme: Environmental</strong></td>
</tr>
<tr>
<td>• Species richness</td>
<td>• Land use</td>
</tr>
<tr>
<td>• Water availability</td>
<td>• External cost (environmental)</td>
</tr>
<tr>
<td>• Erosion</td>
<td><strong>Theme: Socio-political</strong></td>
</tr>
<tr>
<td>• Soil fertility</td>
<td>• External costs (human health)</td>
</tr>
<tr>
<td><strong>Theme: Social development</strong></td>
<td><strong>Theme: Social development</strong></td>
</tr>
<tr>
<td>• Income</td>
<td>• Job creation</td>
</tr>
<tr>
<td>• Property rights</td>
<td>• Social acceptability</td>
</tr>
<tr>
<td>• Access to forest resources</td>
<td>• External supply risk</td>
</tr>
<tr>
<td>• Involvement of community-based formal and non-formal organisations in project design, management and decision-making</td>
<td><strong>Participation by local people in project activities and perceived benefits</strong></td>
</tr>
<tr>
<td>• Participation by local people in project activities and perceived benefits</td>
<td><strong>Investment in education, health services and capacity building</strong></td>
</tr>
</tbody>
</table>

*CDM: Clean development mechanism

The examples in Table 5.1 are also relatively simple. As Table 5.2 shows, more detailed assessments can result in a large performance matrix with cells containing qualitative descriptions and a range of natural units (including price). An impact matrix incorporating economic, environmental and social information can rapidly become complex and unwieldy, suggesting a potential role for the development of indices to synthesise the large volume of information from individual indicators. Loomis et al. (2014) and Loomis and Paterson (2014) propose a hierarchical process in which individual indicators are rank ordered and aggregated in progressive steps to provide higher order indices which are more easily reported and interpreted. For example, a “water column” index comprises combined indicators for abiotic conditions, human health and eutrophication, with the latter itself an aggregation of indicators for phytoplankton and nutrients. For the ecosystem services component of an assessment, the hierarchical format of CICES already lends itself as a framework for progressively aggregating information. However, caution should be applied in the development of indices, as they can oversimplify complex issues and may be open to misinterpretation particularly if poorly constructed (Loomis and Paterson 2014).
Table 5.2. An example impact matrix, showing the range of criteria considered (and measurement metrics) in a multicriteria assessment of renewable energy scenarios (Kowalski et al., 2009)

<table>
<thead>
<tr>
<th>Criterion</th>
<th>Unit</th>
<th>Scenario A</th>
<th>Scenario B</th>
<th>Scenario C</th>
<th>Scenario D</th>
<th>Scenario E</th>
</tr>
</thead>
<tbody>
<tr>
<td>Climate change properties</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>- CO2 equivalents</td>
<td>t/TJ</td>
<td>18</td>
<td>16</td>
<td>17</td>
<td>21</td>
<td>18</td>
</tr>
<tr>
<td>Air quality</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>- SO2 equivalents</td>
<td>kg/TJ</td>
<td>276</td>
<td>236</td>
<td>179</td>
<td>289</td>
<td>265</td>
</tr>
<tr>
<td>TOPP</td>
<td>kg/TJ</td>
<td>359</td>
<td>312</td>
<td>240</td>
<td>399</td>
<td>353</td>
</tr>
<tr>
<td>Particulate matter</td>
<td>kg/TJ</td>
<td>94</td>
<td>78</td>
<td>69</td>
<td>124</td>
<td>72</td>
</tr>
<tr>
<td>Rational use of resources</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>- Cumulated energy input</td>
<td>GJ/TJ</td>
<td>2365</td>
<td>2099</td>
<td>1822</td>
<td>2444</td>
<td>2274</td>
</tr>
<tr>
<td>- Cumulated material input</td>
<td>kg/TJ</td>
<td>81,441</td>
<td>83,182</td>
<td>105,203</td>
<td>78,311</td>
<td>75,468</td>
</tr>
<tr>
<td>Water quality</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>- Phosphorus</td>
<td>mg/TJ</td>
<td>30</td>
<td>31</td>
<td>56</td>
<td>77</td>
<td>33</td>
</tr>
<tr>
<td>- Nitrogen</td>
<td>g/TJ</td>
<td>4</td>
<td>4</td>
<td>5</td>
<td>6</td>
<td>5</td>
</tr>
<tr>
<td>- AOX</td>
<td>mg/TJ</td>
<td>25</td>
<td>24</td>
<td>22</td>
<td>20</td>
<td>33</td>
</tr>
<tr>
<td>- CSB</td>
<td>kg/TJ</td>
<td>33</td>
<td>36</td>
<td>51</td>
<td>92</td>
<td>31</td>
</tr>
<tr>
<td>- BSB</td>
<td>g/TJ</td>
<td>967</td>
<td>1040</td>
<td>1467</td>
<td>2598</td>
<td>899</td>
</tr>
<tr>
<td>Costs</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>- Constant &amp; variable costs</td>
<td>€/TJ</td>
<td>43</td>
<td>41</td>
<td>51</td>
<td>40</td>
<td>46</td>
</tr>
<tr>
<td>Regional self-determinacy</td>
<td>Qualitative</td>
<td>Rather low</td>
<td>Low</td>
<td>Rather high</td>
<td>Medium</td>
<td>High</td>
</tr>
<tr>
<td>Social cohesion</td>
<td>Qualitative</td>
<td>Rather low</td>
<td>Low</td>
<td>Rather low</td>
<td>Medium</td>
<td>Medium</td>
</tr>
<tr>
<td>Diversity of technologies</td>
<td>Qualitative</td>
<td>Rather low</td>
<td>Medium</td>
<td>Medium</td>
<td>Low</td>
<td>Medium</td>
</tr>
<tr>
<td>Employment</td>
<td>Qualitative</td>
<td>Rather low</td>
<td>Medium</td>
<td>Medium</td>
<td>Rather low</td>
<td>Medium</td>
</tr>
<tr>
<td>Effect on public spending</td>
<td>Qualitative</td>
<td>Low</td>
<td>Rather low</td>
<td>Medium</td>
<td>Rather low</td>
<td>Medium</td>
</tr>
<tr>
<td>Import independency</td>
<td>Qualitative</td>
<td>Medium</td>
<td>Medium</td>
<td>Low</td>
<td>Medium</td>
<td>Medium</td>
</tr>
<tr>
<td>Quality of landscape</td>
<td>Qualitative</td>
<td>Low</td>
<td>High</td>
<td>High</td>
<td>Medium</td>
<td>High</td>
</tr>
<tr>
<td>Noise</td>
<td>Qualitative</td>
<td>Medium</td>
<td>Low</td>
<td>Low</td>
<td>High</td>
<td>Low</td>
</tr>
<tr>
<td>Social justice</td>
<td>Qualitative</td>
<td>Medium</td>
<td>Medium</td>
<td>Medium</td>
<td>Medium</td>
<td>Medium</td>
</tr>
<tr>
<td>Technological advantage</td>
<td>Qualitative</td>
<td>Low</td>
<td>Medium</td>
<td>High</td>
<td>Low</td>
<td>Low</td>
</tr>
<tr>
<td>Ecological justice</td>
<td>Qualitative</td>
<td>Low</td>
<td>Medium</td>
<td>Rather high</td>
<td>Low</td>
<td>Medium</td>
</tr>
<tr>
<td>Security of supply</td>
<td>Qualitative</td>
<td>Low</td>
<td>Medium</td>
<td>Medium</td>
<td>Medium</td>
<td>High</td>
</tr>
</tbody>
</table>

5.4 Utilising outputs from a Rapid Evidence Assessment

A Rapid Evidence Assessment (REA) methodology has been proposed within SPLiCE Phase 1 as a tool for defining the scope of the assessment, framing the evidence-gathering process and summarising its outputs (Smithers, 2015). The REA process provides a structured framework for gathering, summarising and determining the quality of information, and identifying data gaps, and so has a key role within an MCA. Any MCA would directly evaluate specific details of the evidence extracted during the REA process, and hence would replace the step of synthesising the information as described in the REA methodology (Smithers, 2015; Section 1.11.2, Table 4).

The REA guidance requires certain information to be recorded, as a minimum, for each piece of evidence considered, including author, year and title, location, scale, literature type and pedigree (Smithers, 2015). For the purpose of an MCA, additional key details must be systematically recorded. These are the:

- Subject detail (e.g. habitat type, economic sector)
- Outcome variable (e.g. species abundance, unemployment rate)
- Metric (e.g. GBP, individuals per m²)
- Outcome detail (the quantified impact, or qualitative description, as reported in the study)
The REA process will therefore demonstrate what evidence is available, and where impacts have been valued (in monetary or other terms), otherwise quantified or reported qualitatively. The evidence obtained will then form the performance matrix to be considered within the MCA.

The co-development of the MCA and REA will be an iterative process. The criteria identified for determining the impacts of each option (Stage 3 of the MCA) will guide development of the required search terms for REA, but progress of the REA may also identify additional criteria that should be considered.

5.5 Visualisation of outputs

As the amount of data and the access to it increases, and decisions are opened up to more stakeholder involvement, there is increasing need to find good ways of presenting data to meet different stakeholder needs throughout the decision-making process. There are a number of ways in which using visual methods of data representation can aid analysis, communication, engagement, debate and decision-making around energy. In particular, visualisation techniques can be important for capturing and addressing complexity (Lindquist, 2011).

Three different aspects of visualisation are briefly described and discussed:

- Visualisation of data
- Using interactive visual tools to explore values and priorities
- Beyond visualisation: Policy informatics

5.5.1 Visualisation of data

Data visualisation here refers to the visual representation of statistical and other types of numeric and non-numeric data through the use of pictures and graphics. It is used to transform (often) large quantities of data into images that can more clearly illustrate patterns, gaps, schemes, regularities and connections (Reuters Institute, 2015). Data visualisation is of particular value where assessment of the environmental, economic and social impacts of energy choices generates complex results, encompassing multiple, incommensurable criteria. It may be less relevant for frameworks which allow results to be expressed using a single metric, e.g. CBA.

A data visualisation project has been commissioned by the Department of Communities and Local Government, who are looking to understand the potential for visualisations to analyse and communicate data, and provide guidance for national and local analysts on making best use of visualisation (DataViz 2015). The project is carrying out a review of data visualisation - how data is presented and brought to life using charts, graphs, maps, timelines, animations etc. - for public sector decision-makers and researchers. There is already very helpful interim guidance on the website, as well as a large gallery of different types of visualisations. One of the project aims is to look for good examples of visualisation which have made an impact / helped communicate findings to decision-makers, and those which helped researchers understand the story behind the data.

One way of showing data about multiple impacts simultaneously is by using a spider chart (Figure 5.4), but of course there are many others. The DataViz site has a gallery of 300 different types of visual representation.

While high quality visualisation is an instinctively attractive idea, there appears to be relatively little in the way of research demonstrating its effect on energy-related communication and decision-making. There are literatures on how people prefer to receive information on their own energy use, energy label designs for appliances, and energy information on bills – but these are of limited relevance to larger scale decision-making.
5.5.2 Using visual tools to explore values and priorities

Visual tools can also be used to help explore values and priorities in decision-making. These can range from a relatively simple tool, to a model which allows a sophisticated exploration of future energy systems. Such tools can have similarities to those developed to allow investigation of different energy futures such as the DECC 2050 Calculator. The key difference is that values and priorities are made explicit – and this can help stakeholders understand others’ perspectives, and provide a basis for discussion and for understanding trade-offs.

One example of such a tool is the Energy – Multi Criteria Analysis (ENE-MCA) policy tool (IIASA 2015). Its central purpose is to aid decision makers in their assessments of future policy choices and how those choices impact the multiple dimensions of energy sustainability. The tool has been specifically designed for policy advice and the communication of scenario results, allowing users to visualise the complex (and not always obvious) synergies and trade-offs of specific policy choices and to better understand how varying the prioritization of the multiple energy sustainability objectives can lead to qualitatively different energy system futures. A less complex tool is the OECD Better life index. This index allows comparison of well-being across countries, based on 11 topics the OECD has identified as essential, in the areas of material living conditions and quality of life. Importantly, it also allows people to rate topics according to their importance, and then examine the data using that lens.

5.5.3 Beyond visualisation: Policy informatics

There is an emerging field of ‘policy informatics’ which seeks a much broader understanding of the role of information (and its presentation) in decision-making. The fundamental premise of policy informatics is that information can be efficiently and effectively mobilised to enable evidence-driven policy design, implementation and analysis. The long-term objective of this area of research has been described as “to create proven mechanisms that incorporate expertise regardless of discipline, institution or community into governance to provide effective support for social decision-making and collective action, while ensuring the legitimacy of widespread public participation” (Johnston, 2015). As this area of thought develops, it may offer valuable perspectives. Recent examples of this include i) the Office of National Statistics Well-Being Wheel of Measures (ONS, 2014b), available as both a static graphic and an interactive webpage, and ii) the Improved Impact Assessment developed by the New Economics Foundation on behalf of a consortium of Non-Governmental Organisations, which is due for release in early April 2015.
5.6 Recommendations

- Future evaluation of the impacts of energy systems and strategies needs to fit with existing structures, but there is clear scope to improve existing processes such that economic, social and environmental information is considered equally, and that non-monetary impacts are not marginalised (particularly in impact assessment).
- The assessment process should have the following three key elements: a) it should utilise the Natural Capital Committee approach as an overarching framework to identify the natural capital and ecosystem services information that should be considered; b) macro-economic, social impact, health impact assessment methods should be used alongside to generate additional relevant data; c) the structured framework provided by multi-criteria assessment methods should be used to evaluate the resulting environmental, social and economic information.
- The choice of pathways within the evaluation framework is context dependant and will need to be tailored to the questions posed and the purpose for which the answer is sought, as well as to the resources available.
- Best practice within multi-criteria analysis is to use a participatory process, as effective analysis requires accurate assessment of what changes are important and to whom, as well as the degree of relative change.
- The criteria to be considered when assessing the impacts of an energy pathway or development depend on the decision context, ideally the criteria will be decided in a participatory process involving relevant experts and stakeholders.
- The development of an extensive list of criteria, with appropriate indicators, from which context-specific performance matrices can be built would facilitate the impact assessment process.
- Outputs from REA(s), should inform the performance matrix developed within the multi-criteria analysis process.
- SPLICE 2 should use the outcomes of the DataViz project to help present research outputs according to best practice.
- Evidence should be gathered on the effectiveness of different types of visualisation throughout the decision-making process, to aid progress across government.
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Appendix 1: Key Ecosystem Service Frameworks for the UK Context

UK National Ecosystem Assessment (UK NEA)

The UK National Ecosystem Assessment (UK NEA) was initiated in 2009 following the results of the Millennium Assessment MA (see Appendix 2), which indicated the extent, globally that ES were being lost. Conducted between 2009 and 2011 the UK NEA undertook the first analysis of the UK’s natural environment in terms of the benefits it provides to society and continuing economic prosperity. The UK NEA assessment followed a similar framework to the MA to enable the identification and development of effective policy responses to ecosystem service degradation. The framework also incorporated post-MA advances, particularly for economic valuation methodologies to avoid double counting of ecosystem services. (Appendix 3 and also de Groot et al., 2010, Ring et al. 2010, Balmford et al. 2011, Fitter et al. 2010 Fisher and Turner 2008).

Similar feedbacks were included from direct and indirect drivers of change in ecosystems and well-being as in the MA (Figure A1.1). However, the UK NEA framework adapted the MA classifications of ecosystems (within 8 broad habitat types occurring in the UK), ecosystem services, the processes driving change and their outcomes (on ecosystems, ecosystem services and human well-being) for the UK context.

Figure A1.1 The UK NEA framework incorporating drivers of change, future scenarios and social feedbacks.

UKNEA Follow-on Phase (UK NEAF0) 2011-2014

Conducted between 2011 and 2014 the NEAF0 builds on the framework of the existing NEA (Figure 2.2), to provide new information and tools, particularly aimed to help decision-makers across all sectors understand the wider value of ecosystems and the services they offer (UK NEAF0 2014). Advances have been built into the NEAF0 framework which are of relevance to assessing the impacts of energy generation, particularly due to the focus on embedding ES assessment in policy and planning decision making. Specifically, these advances have been in the following key areas: Natural capital, ESs and the macroeconomy, economic valuation of ESs, coastal and marine ESs, cultural ESs, shared and plural values, operationalising scenarios (scenarios were originally identified in the NEA), response options (to improve policy and practice for sustainable delivery of ESs), embedding ES framework into policy appraisal and the development of tools to aid decision makers to embed ES approach within policies and decisions (UK NEA FO 2014).
Examples of the key advances follow under broad topic areas:

- Natural capital’s relationship to regional and national economic performance
- The UK NEA FO incorporates assessments of natural capital and the relationship between natural capital and the wider economy.
- Relationships were mapped between ESs and sectors of the economy they support, such as agriculture or food manufacture.
- The Natural Capital Asset Check (NCAC) was developed to consider economic impacts arising from any changes in ESs by considering thresholds, trade-offs and the performance and resilience of UK ecosystems.
- Quantitative valuation of additional ecosystem services, relating ES change to land use and modelling marine and coastal ecosystems.
- Models were utilised to address ES in relation to changes in land use in terrestrial environments (e.g. different forest planting strategies, changes in agricultural outputs and farm incomes, net greenhouse gas emissions, recreational visits, water quality and biodiversity).
- Models were also identified that can be used to address the different components of the marine shelf ecosystem, and a number of options were proposed for linking land use change models to coastal waters in order to assess the consequences for coastal ecosystem services.
- The UK NEAFO has developed specific indicators, informed by a drivers, pressures, state changes, welfare impacts and policy responses pressures (DPSWR) scoping framework, for six marine related ecosystem services: fisheries and aquaculture, sea defence, prevention of erosion, carbon sequestration/storage, tourism and nature watching, and education.

- Advances in valuing cultural ecosystem services
- Cultural ecosystem services provide a range of material and non-material benefits to human well-being, but are frequently overlooked in decision-making.
- The UK NEAFO characterises the four key components of cultural ecosystem services as: environmental spaces; cultural values; cultural practices; and benefits.
- The UK NEAFO identifies (i) quantitative indicators and analysis of cultural ecosystem services, which draw on publically available datasets; and (ii) participatory and interpretative research techniques developed in the social sciences, and arts and humanities, which can be used to assess and understand cultural ecosystem services in location- and community-based contexts.
- The work highlights that group based valuation, done through a deliberative process (where group participants are allowed to exchange evidence and reflect on matters of mutual interest), tend to be different from the conventional aggregation of individual values.
- Mapping techniques within a group based participatory setting, (with members of the public identifying on physical maps the locations of particular importance to them), are also highlighted as a useful tool for identifying concentrations of cultural benefits and identifying associated management issues.
- Adaptive management principals to guide inclusion of ESs in policy and decision making

In summary the updates to the NEA framework are of specific use to the decision making process as they allow for:

- Understanding of the roles of governance and institutions in the decision-making process.
- Recognition of the importance of built, human and social capital in transforming natural capital and the flow of ecosystem services into goods and benefits for people (identified within the NCC’s work, NCC 2014, 2014a).
- Incorporation of adaptive management into the updated framework (to provide opportunity for flexible responses to inform decision making as knowledge grows).
- A Decision Support System (DSS) toolbox which offers a set of tools by which decisions regarding ecosystems and their services may be supported (The toolbox is available through an accessible web portal, the National Ecosystem Approach Toolkit (NEAT); http://neat.ecosystemsknowledge.net/)
- A balance Sheet Approach for interrogating and presenting evidence from appraisals.
Natural Capital Committee

The Natural Environment White Paper (HMG, 2011) provided the foundation for the establishment of the Natural Capital Committee (NCC), as an independent body to advise the Government on the state and sustainable use of England’s natural capital. In 2014, the NCC produced a working paper (NCC, 2014) in which it proposed a framework designed to facilitate consistent analysis of changes in natural capital.

The first component of this framework was a definition of natural capital: the elements of nature that directly and indirectly produce value or benefits to people, including ecosystems, species, freshwater, land, minerals, the air and oceans, as well as natural processes and functions. The NCC made clear that both biotic and abiotic elements were included, and that, in common with the UKNEA (2011), it considered natural capital assets to be a series of stocks, from which flows of ecosystem services were generated. Services (often combined with other capital inputs) provide goods, from which people obtain benefits, and these benefits can be valued in monetary and other terms (Figure A1.2).

The NCC aims to embed natural capital in the UK’s Environmental Accounts (NCC, 2013). National environmental accounting is an ambition shared by other countries, and so the NCC framework incorporates elements of the UN’s System of Environmental-Economic Accounting (SEEA) (United Nations Statistical Division, 2013). The NCC framework adopts the SEEA system of using major land-use categories as accounting units (spatial areas about which information is collected). In the NCC case, these units are the eight broad habitat types defined within the NEA (although potential disaggregation of these categories is being considered. These habitats were selected as they form the basis of existing monitoring schemes, are mutually exclusive, and cover the entire country (NCC, 2014a). The land-use categories serve to provide a linkage between assets and benefits. The supply of benefits results from complex interactions between natural capital assets, and the process of understanding these relationships is simplified if they are considered in terms of the characteristics of land-use types (NCC, 2014a). Also, there may be more data available at the level of the land-use category that for individual natural capital assets.

Figure A1.2 The conceptual framework proposed by the Natural Capital Committee (NCC, 2014a)

The three main elements of the NCC (2014) framework are i) natural capital stocks; ii) major land-use categories; iii) goods/benefits. The need to define metrics by which to measure temporal changes in these has been acknowledged. The NCC make some suggestions, proposing that metrics for natural capital stocks reflect the key features of the asset, such as species richness, abundance and...
distribution; habitat area and condition; and metrics applied for national monitoring of environmental quality, including the England Biodiversity Indicators (Defra, 2013). The metrics for the major land-use categories must relate to both the natural capital stocks and the benefits ultimately provided, and should summarise the quantity, quality and spatial configuration of the land-use category. For example, the larger the area of woodland (quantity), the more timber is likely to be available, provided that the structure and species composition (quality) of the area is properly maintained (NCC, 2014a). The recreational values of the area will also depend on where the woodland is in relation to population centres (spatial configuration) (NCC, 2014a). Metrics for benefits should be in terms of value, which will usually, but not necessarily, be monetary (NCC, 2014).

The NCC (2014) framework also explicitly considers the implications of changes in natural capital in a management context, by advocating that thresholds and safe limits for sustainable use are identified, taking account of the resilience of the system. These thresholds for natural capital assets are points at which a small change in a driver causes a large (and usually abrupt) decline in the status of the asset. Thresholds for benefits relate to the level at which the status of the natural asset has declined to the point that a sufficient benefit value is no longer generated. Safe limits apply a precautionary approach, providing a buffer before thresholds are reached. The NCC framework has only recently been published as a working paper and has not yet been applied. Additional guidance on the valuation of changes in natural capital will be forthcoming.

Towards a Common International Classification of Ecosystem Services (CICES)

The CICES programme (conducted by the European Environment Agency, EEA) represents the European contribution to a global UN initiative to provide standardisation between environmental accounting methods (Haines-Young and Potschin, 2013a). It was initiated in 2009 because, despite previous efforts, there had been no accepted definition or classification of ecosystem services. International standardisation of the way ES are described was needed to assist with the integration of ES assessment into decision making.

CICES has also highlighted that standardisation of the way ES are described is required beyond just the context of environmental accounting. Work on mapping and valuing ecosystem services and ecosystems assessments more generally have been identified as beneficiaries from more systematic approaches to naming and describing ecosystem services (CICES 2010). Since the CICES aim was to identify the ‘final products’ of ecosystems it deals only with provisioning, regulating and cultural outputs (Figure A1.3, Table A1.1).

Figure A1.3 CICES programme in relation to over-arching ES framework structure (reproduced from CICES, 2010)
CICES provides a framework for classifying final ecosystem services that are dependent on living processes (biodiversity). It is hierarchical in structure, with each level providing a more detailed description of the ecosystem service being considered, for instance cultivated crops (Figure A1.4). Within the over-arching ES categories (termed Sections in CICES) biological or material outputs and biophysical and cultural processes are subdivided through a series of stages (division, group, class and class type) to establish links between each output and process to easily identify service (benefit) sources.

Compared to other frameworks and classifications, CICES provides the greatest level of detail for the individual ES benefits. The hierarchical structure means that studies that are undertaken at different thematic and spatial resolutions can more easily compared (Haines-Young and Potschin 2013c).

Figure A1.4 The hierarchical structure of CICES v4.3 (Blue boxes) represented for a provisioning ES, cereal food crops (grey ellipses).

CICES was not intended to replace other classification systems (such as MA, TEEB and NEA) but to allow the easy translation between them. The CICES classification approach has been taken up by the European working group on Mapping and Assessment of Ecosystem Services (MAES, 2012).
Energyscapes

Energyscapes and Ecosystem Services was a Land Based Renewables (LBRs) pilot research study funded by NERC (2009-2011). It aimed to demonstrate how the inclusion of energy provision within an ecosystem service framework might help guide the deployment of LBRs in the UK. The concept was piloted over a 12-month period in a lowland arable area. The study involved energy system developers, local planners, regulators and the general public. The overall aim was to determine how an understanding of ecosystem services (ES) and the “energyscape” could help guide the deployment of LBR. The ultimate goal would be to help assist conflict resolution by reconciling the objectives of all parties involved in development. The project aims were:

1. To extend the ecosystem approach to the development of LBRs
2. To develop a methodology for describing an energyscape in terms of interacting resources and conflicting demands
3. To develop and demonstrate an integrated model linking an energyscape with models of key ecosystem services for a case study site, and
4. To identify how and where to deploy different LBRs in relation to specified objectives, and to recommend how the approach should be applied to other sites and situations.

Rather than a traditional scientific reductionist approach, a whole system approach was employed including the energy, ecosystem service and landscape perspectives that are necessary to support decision making. The project was applied in a case study site (Marston Vale in Bedfordshire).

An energyscape is a complete real landscape that supports people through the delivery of ecosystem services. It is distinguished from a normal landscape in that it is viewed from the perspective of the energy system that sits within it. The energy system itself is comprised of sources, infrastructure and demand that interact both with other parts of the energy system and the wider landscape, influencing the delivery of ecosystem services (Figure A1.5). An energyscape cannot be considered in isolation, but must recognise imports and exports from surrounding landscapes along with potentially inert throughput (e.g. a cable system linking power stations and population outside the area).

Figure A1.5 Energyscapes framework

Energyscapes used a modified form of de Groot’s classification in an attempt to cover all components of the environment (abiotic and biotic) that might influence how a decision is made or viewed. As ecosystems are neither easily spatially delimited nor comprehensive in coverage of landscapes, mapped habitats were used to spatially structure the landscape (Broad Habitats). Different individual stakeholders or groups were interviewed to determine their perception of the ecosystem services delivered by specific habitats in the specific area and their sensitivity to different potential development of the energy system (Table A1.2).
### Table A1.2 Questions about specific ecosystems services used to assess people’s values.

<table>
<thead>
<tr>
<th>Cultural services</th>
<th>Does the habitat make you think …</th>
</tr>
</thead>
<tbody>
<tr>
<td>Aesthetic</td>
<td>... it is beautiful?</td>
</tr>
<tr>
<td>Heritage</td>
<td>... about the past?</td>
</tr>
<tr>
<td>Jobs</td>
<td>... of opportunities for employment?</td>
</tr>
<tr>
<td>Recreation</td>
<td>... you want to spend more time here?</td>
</tr>
<tr>
<td>Scientific &amp; educational</td>
<td>... there is a chance to learn or observe something interesting?</td>
</tr>
<tr>
<td>Spiritual</td>
<td>... about the future?</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Habitat services</th>
<th>Is the habitat where you would expect to see….</th>
</tr>
</thead>
<tbody>
<tr>
<td>Flora</td>
<td>... wild plants</td>
</tr>
<tr>
<td>Fauna</td>
<td>... wild animals</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Provisioning services</th>
<th>What do you get from the habitat…</th>
</tr>
</thead>
<tbody>
<tr>
<td>Fibre</td>
<td>... fibre such as wood, flax or wool?</td>
</tr>
<tr>
<td>Food</td>
<td>... food for people or livestock?</td>
</tr>
<tr>
<td>Freshwater</td>
<td>... freshwater e.g. springs?</td>
</tr>
<tr>
<td>Fuel</td>
<td>... fuel e.g. firewood or biodiesel?</td>
</tr>
<tr>
<td>Genetic</td>
<td>... a genetic resource for the future?</td>
</tr>
<tr>
<td>Medicinal/oramental</td>
<td>... Medicinal or ornamental plants?</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Regulating services</th>
<th>Does the habitat help …</th>
</tr>
</thead>
<tbody>
<tr>
<td>Air quality</td>
<td>... improve the air we breathe e.g. dust, smells, ammonium?</td>
</tr>
<tr>
<td>Assimilation of carbon</td>
<td>... lock up carbon from the atmosphere in the soil or plants?</td>
</tr>
<tr>
<td>Buffer - chemicals</td>
<td>... e.g reduce pollution from acid rain, nutrients or pesticides?</td>
</tr>
<tr>
<td>Buffer - physical</td>
<td>... e.g. reduce erosion or flooding?</td>
</tr>
<tr>
<td>Buffer - economic</td>
<td>... e.g. “safe” jobs in times of recession?</td>
</tr>
<tr>
<td>Climate</td>
<td>... moderate the local (or global) climate?</td>
</tr>
<tr>
<td>Disease, pests &amp; natural hazards</td>
<td>... reduce the impact of pest and diseases, e.g., aphids, Lymes disease, etc.?</td>
</tr>
<tr>
<td>Erosion</td>
<td>... prevent erosion?</td>
</tr>
<tr>
<td>Fire</td>
<td>... prevent wildfires?</td>
</tr>
<tr>
<td>Pollination</td>
<td>... provide nectar resources for bees and other pollinators.</td>
</tr>
<tr>
<td>Water flow</td>
<td>... moderate water flows (quantity) e.g. floods and droughts?</td>
</tr>
<tr>
<td>Water quality</td>
<td>... improve water quality?</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Supporting services</th>
<th>Does the habitat help support other services by …</th>
</tr>
</thead>
<tbody>
<tr>
<td>Nutrient cycling</td>
<td>... reducing nitrogen and phosphorus losses?</td>
</tr>
<tr>
<td>Primary productivity</td>
<td>... growing vegetation?</td>
</tr>
<tr>
<td>Soil formation</td>
<td>... encouraging soil formation?</td>
</tr>
<tr>
<td>Hydrological cycling</td>
<td>... circulating water around the environment?</td>
</tr>
</tbody>
</table>
Hattam et al. 2014: Indicators of Ecosystem Services

Ecosystem service assessments have typically been focused at national and international scales. Consequently a large variety of ES classifications have also been developed at these scales, predominantly with a focus on terrestrial environments. Hattam et al. (2014) have produced a systematic assessment framework that follows the principles of TEEB, NEA and NEAFO to assess the interactions between marine ecosystems and human society. In particular, their framework aims to be transferable across, and adaptable to different sites and applications.

Hattam et al., propose a marine ecosystem service classification and identify distinct indicators for each service. The ecosystem service classification adapted the classification within TEEB, with modifications suggested by Henrichs et al., (2013). They applied this to a UK case study site, the Dogger Bank which is an offshore region of the North Sea that is of conservation interest (as a cSAC) and contains a leased area for offshore wind energy development (below). A distinction between indicators for ecosystem services (ES), ecosystem functions and ecosystem benefits is at the centre of the approach. The need for indicators was identified as a means of aiding management and planning decisions as well as supporting models of human interactions with the marine environment, and associated effects on human well-being. Their definitions for ES, ecosystem functions and ecosystem benefits are:

- Ecosystem services are the direct and indirect contributions of ecosystems to human well-being (TEEB, 2010).
- Ecosystem functions are the ecological processes that control the fluxes of energy, nutrients and organic matter through an environment (Cardinale et al. 2012).
- Ecosystem benefits are the things that people create or derive from ecosystem services. Benefits are turned into products or experiences that are no longer functionally connected to the systems from which they were derived (Haines-Young and Potschin 2013).

Indicators were identified that reflect the provision of an ES and how that ES changes over time. Indicators were required to be measurable and capable of detecting change in the ES considered, as well as meeting three further criteria:

- **Specificity**: can the indicator respond over time to changes in management as opposed to natural variability? Is this response predictable and does it have low variability?
- **Scalability**: can the indicator be aggregated or disaggregated to a different spatial scale and still retain its ability to indicate the change of interest?
- **Transferability**: is the indicator useful for other locations and hence studies?

**Case study: Dogger Bank, North Sea**

The ES within Hattam et al's (2014) tailored framework were examined in relation to the ability of the Dogger Bank (Figure A1.6) to produce (supply) those ES.

Indicators were similarly identified that were specific to the site. For instance where fish populations (and quality of the stock) were identified as an indicator for ES of food provision, populations of the fish species of relevance to the Dogger Bank were required (primarily flatfish species such as, sole and plaice).

The indicator selection process focused on ecosystem services rather than functions or benefits. However, a need for multiple indicators was identified across all categories, ecosystem service, function or benefit.
ES indicator identification for Dogger Bank.

- Provisioning service indicators.

Metrics such as biomass of living resources or quantity of a raw material were identified as indicators in this category. Although indicators were present for some ES such as food provision from wild capture seafood (biomass in tonnes km$^2$ of relevant fish species populations) specificity such as the separation of climate change effects from other effects was identified as problematic. Other provisioning services such as medicinal resources relied on the same indicators (biomass in tonnes km$^2$ of relevant species / resource), however there were insufficient spatially resolved information resources present, despite the high research interest in the Dogger Bank, to define indicators for the site (Hattam et al.2014).

- Cultural service indicators

Metrics identified in this category focused on species and habitat types related to services such as aesthetic experience, and used, for example, count data of species of individual interest (such as harbour porpoise, grey seal, seabirds, and fish species) or extent (km$^2$) of biotopes of key interest. It was challenging to identify indicators for cultural services such as aesthetic experience or inspiration for culture, art and design due to the difficulty of identifying the specific contributory role of an ecosystem to many cultural services. Indicators identified during expert workshops tended to be indicators of ecosystem benefits rather than of ecosystem services (Hattam et al 2014).

- Regulating and habitat service indicators

Generic indicators of regulating and habitat services were identified as being relatively straightforward to select for regulating services such as CO$_2$ regulation. Data were identified as achievable to collate through modelled or empirically determined data on air-sea and sediment-water fluxes of carbon and CO$_2$. Carbon storage indicators, included levels of carbon in different components of the marine ecosystem, for instance, modelled or empirically determined levels as biomass of carbon (g m$^2$). However, obtaining these data was again more challenging at a spatially resolved site specific level. Detailed data on community structure of biota were identified as being particularly important as these data provided relevant indicators for a number of regulating ES.

In relation to certain regulating services challenges were also identified due to a lack of knowledge on how ecosystem functions generate corresponding services. For example, in relation to air purification, indicators of air-sea flux of pollutants were lacking due to insufficient information.
Application: TIDE - Tidal River Development

"TIDE - Tidal River Development" was a project co-financed by the Interreg IVB North Sea Region Programme and implemented between 2010 and 2013. Focusing on North Sea region estuaries under a strong tidal influence, protected by European directives and serving as fairways to important seaports, TIDE gathered some of the leading European experts from universities, port authorities, waterways administrations and others to find multi-beneficial solutions for future sustainable estuary development.

Assessment of ES provision and spatial mapping of supply of key ecosystem services were conducted for the four case study estuaries as a tool to aid making integrated management and planning a reality. ES assessment provided the scientific methods and tools to allow for inter-estuarine comparisons. The project also addressed governance requirements for integrated management through regional working groups bringing together stakeholders to identify governance needs in each region. Mitigation and compensation measures and solutions were developed. The final element of the project addressed approaches to sharing information and raising awareness of the projects approaches. This stage aimed to increase understanding and acceptance of necessary changes within target groups, ranging from EU policy makers to estuary residents. The ES assessment stage of the project adapted the TEEB classification framework and aimed at providing a broad overview for inter-estuarine comparison and general conclusions on ES supply in NW European estuaries in general.

Professional estuarine experts were involved in obtaining a more general, but complete semi-qualitative overview of ES supply in NW European estuaries, rather than examine a small representation of individual estuarine ES and ecosystem functions in detail.

In summary the approach taken by the TIDE project;

- Focuses on comparing a broad bundle of services rather than on the detailed assessment of a few single services, well-known supply functions or case studies.
- Is based on experiential knowledge of people familiar with the system to provide a broad assessment in semi-quantitative units rather than quantitative data using diverse units and suffering from knowledge gaps.
- Involves professional experts from the TIDE regional working groups, conveying the importance and potential of ecosystem service based management to professionals active in estuarine management.

Methodology and Case study

ES assessment was applied to the four case study estuaries: the Elbe (DE), Weser (DE), Scheldt (BE/NL) and Humber (UK) estuaries. The TEEB classification was adopted to present an inventory or 'longlist' of ES provided by European estuaries, based on literature sources and consultation with estuarine experts (Jacobs et al., 2013). A multi-stage process was then followed:

1. Important ecosystem services for TIDE estuaries were distinguished from the “longlist” of estuarine services (using literature resources and expert assessment). The variation in demand (“societal importance”) was assessed along estuaries, salinity zones and for historical, present and future time scales. This utilised a questionnaire survey approach to gather expert opinion.

2. Ecosystem service supply levels for key ecosystem services were assessed. ES supply was mapped spatially along the estuaries in relation to basic underlying processes and structures.
   - An initial survey obtained input from estuarine experts on the importance of habitat types on the delivery of individual ES. This applied methods developed by Burkhard et al. (2012, 2010), essentially asking “how important is habitat x in delivery of ES y?.” Responses were entered into a 5 point scale from 1 (low, no importance) to 5 (high, essential importance).
   - The information was then further interpreted within salinity zones.
   - Statistical tests for variance in responses from estuarine experts were used to provide indications of confidence in the assessment.
   - Using the supply level score resulting delivery for each ES, was mapped spatially for each estuary in relation to underlying processes and structures (e.g. habitat type and salinity zone).
   - Finally, qualitative scorings on the state of the habitats to provide functions (that provide a number of services) and the trend over the most recent decade of habitat
extent and quality were assessed. A historical ES supply evolution through habitat change was estimated and an indicator for trade-off risk generated by differential supply of ES by habitats was discussed.

3. Expected effects of estuarine management measures on ES supply were estimated and the synergies in ES supplies (which ES supplies are increasing together) occurring from these measures were discussed.

4. Key questions were answered and recommendations for research, policy and practice presented.

Case study: ES supply level assessment for C sequestration and burial in European estuaries

Estuarine ecosystems are extremely productive biologically (Bianchi, 2007), with net primary production rates among the highest of the world. Consequently, these systems play a globally important role as carbon sinks (Chmura et al., 2003). Carbon sequestration and burial has both biological and hydrological drivers. Uptake of carbon in the estuarine food web (linked to productivity of algae and vegetation) and long-term sedimentation into deeper soil layers are the main drivers. Due to the link to biological productivity and sedimentation / burial of sediments, marshes and intertidal areas have the highest scores (Jacobs et al., 2013).

In the TIDE project long –term carbon storage, based on carbon removed over approximately 100 years (Crooks et al. 2010) was related to estuarine habitat types. This resulted in only sequestration within sediment being taken into account. Carbon sequestration capacity, determined by long term CO₂ equivalent fluxes (carbon burial versus greenhouse gases released by microbial mediation in sediments) was used.

A database was created of carbon sequestration values in the literature for estuarine habitats within salinity zones. The weakness was recognised that the effects of site-specific factors at original study sites could lead to inconsistencies in the data when pooling literature for each habitat type. The data base and surveys of expert opinion were used to develop supply level scores for each habitat type and salinity zone (Figure A1.7).

Figure A1.7 Importance of habitat within four salinity zones to delivery of the ES, carbon sequestration and burial.

<table>
<thead>
<tr>
<th>Importance of habitat to delivery of ES 'carbon sequestration and burial'</th>
<th>Supply level</th>
</tr>
</thead>
<tbody>
<tr>
<td>no important supply</td>
<td>1</td>
</tr>
<tr>
<td>less important supply</td>
<td>2</td>
</tr>
<tr>
<td>moderately important supply</td>
<td>3</td>
</tr>
<tr>
<td>important supply</td>
<td>4</td>
</tr>
<tr>
<td>essential supply</td>
<td>5</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Habitat</th>
<th>Importance of habitat to delivery of ES 'carbon sequestration and burial'</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Freshwater</td>
</tr>
<tr>
<td>Marsh habitat</td>
<td>4</td>
</tr>
<tr>
<td>Intertidal flat habitat</td>
<td>3</td>
</tr>
<tr>
<td>Intertidal steep habitat</td>
<td>2</td>
</tr>
<tr>
<td>Subtidal shallow habitat</td>
<td>3</td>
</tr>
<tr>
<td>Subtidal moderately deep habitat</td>
<td>2</td>
</tr>
<tr>
<td>Subtidal deep habitat</td>
<td>1</td>
</tr>
</tbody>
</table>

These supply level scores were then mapped for each case study estuary, the Weser (Germany), Scheldt (Netherlands), Humber (UK) and Elbe (Germany) (Figure A1.8).
Figure A1.8 Carbon sequestration and burial in the Weser, Scheldt, Humber and Elbe (clockwise from top left) estuaries based on habitat-specific supply scores per salinity zone, and involving local and site-specific scientific expertise, reproduced from Jacobs et al., (2013).
Appendix 2: International Ecosystem Service Frameworks

The Millennium Ecosystem Assessment (MA)

The Millennium Ecosystem Assessment (MA) was launched in June 2001 as a 4 year international work programme, and involved scientists from over 100 nations. The MA is introduced in this review as the first global ES assessment, and as such, the programme which highlighted the global threats to ES and human well-being. The MA programme focused on the links between ecosystem change and human well-being, particularly how humans have altered ecosystems and how changes in ecosystems have affected human well-being. The ES framework developed was designed to meet the needs of decision makers who require scientific information on the relationships between ecosystem change and human well-being. Ultimately the framework aimed to identify policy responses that could be adopted from local to global scales to improve ecosystem management. The MA framework allocated ES into four categories:

- **Provisioning:** Raw materials obtained from ecosystems: food, water, timber.
- **Regulating:** ES maintaining climate, water quality, flooding and regulation.
- **Cultural:** Non-material benefits to people from ecosystems such as, recreation, spiritual, aesthetic benefits.
- **Supporting:** Elements and functions on which all other ES depend such as; primary production, nutrient cycling and soil formation.

The MA linked the state and changes to these ES categories to human well-being, with a focus on poverty reduction. Human well-being was considered to rely on a range of factors, these included:

- Material minimum for a good life
- Health
- Good social relations
- Security
- Freedom and choice.

In addition, the MA examined how drivers of change in ES affected services and, thus, human well-being. Drivers of change were considered as indirect drivers such as:

- Demographic
- Economic
- Socio political
- Science and technology
- Cultural or religious.

Or direct drivers of change:

- Changes in local land use and land cover
- Species introductions or removals
- Technology adaptation and use
- External inputs (fertilizer, pest control)
- Harvest and resource consumption
- Climate change
- Natural physical and biological drivers (volcanoes, evolution).

The work of the MA not only demonstrated the importance of ES to human well-being, but also showed that at global scales, many key services are being degraded and lost. Actions (management responses) were considered that would be taken either to respond to negative changes or to enhance positive changes at all points in the interaction between drivers of change and ES categories and human well-being factors (Millennium Ecosystem Assessment 2005).
The Economics of Ecosystems and Biodiversity (TEEB)

The Economics of Ecosystems and Biodiversity (TEEB) program (Figure A2.1) was initiated in 2007 by the environment ministers of the G8 and five further countries. The program aimed to assess the global economic benefit of biological diversity, considering how the costs of the loss of biodiversity and the failure to take protective measures compared to the costs of effective conservation. Within TEEB ES are the direct and indirect contributions of ecosystems to human well-being (TEEB, 2010). TEEB modified ecosystem service classification slightly, adding Habitats to replace some of the Supporting Services. TEEB’s approach focuses on valuation frameworks and methodologies. The concept of delivery of benefits and values to human well-being, (derived from services which are provided by ecosystem functions and processes) coincides with the overarching MA framework (MA, 2005). TEEB focuses on valuation for supporting decisions (with reference to parallel ES valuation and accounting programmes such as CBA, MCDA and SEEA). TEEB reports addressed the needs of different actors, from international policy makers, through to, local policy makers/administrators, businesses and finally consumers and citizens (TEEB, 2010).

Figure A2.1. TEEB and related programme advances (blue boxes / light border) incorporated within the MA framework (yellow boxes / dark borders) (reproduced from TEEB 2010).

Three phases of the TEEB study were conducted. The first phase utilised the expertise and resources of various organisations to complete a study and report, which collated evidence and examples of valuation, identified elements of a biodiversity/ecosystem valuation framework, and considered long standing issues such as ethics in making choices regarding future values.

Three core principals guided TEEB’s approach to analysing and structuring valuation to achieve conservation and sustainable use, reflecting different situations in which ES valuation may be applied:
- Recognised value – Conservation and sustainable use can be achieved through value that is already recognised, such as a natural site being protected in its original state as it is regarded as a sacred spiritual site.
- Demonstrated value – Economic value is demonstrated to provide evidence for policy makers or business decisions that need to consider full costs and benefits of an ecosystem in addition to available market values (such as private goods).
- Capturing value – The value of ecosystems can be applied to provide direct reward or incentives to enhance conservation and sustainable use of an ecosystem, through measures such as:
  - Payments for ecosystem services
  - Reforming environmentally harmful subsidies
  - Introducing tax breaks for conservation

The second phase of TEEB led to further specific studies on economic valuation. These included reports on the fundamental concepts and state of the art valuation methodologies and an introduction to approaches and recommendations for mainstreaming the economics of nature into decision-making, as well as analysis and guidance on:
- How to value and internalize biodiversity and ecosystem values in policy decisions;
- Mainstreaming biodiversity and ecosystem values at regional and local levels;
- How business and enterprise can identify and manage their biodiversity and ecosystem risks and opportunities

Values provided by ecosystems and their services were then examined in respect to relevant economic sectors. Costs of biodiversity loss and ecosystem degradation were assessed in this phase with an ultimate focus on integrating findings from these studies into decision-making at all levels. The third phase of TEEB implemented the program at country level, aiding governments to build national, regional and local government capacity to produce tailored economic assessments of ecosystems and biodiversity. This phase aided implementation of the ES approaches into policy making.

TEEB is an ongoing initiative, and one of its current themes is to highlight the economic benefits of oceans and coasts, and to attempt to fill some of the knowledge gaps that hamper ES assessments within the marine environment.

Application of TEEB: TEEB Oceans and Coasts

The TEEB programme as a whole provided global, national and business applied approaches to assess the global economic benefit of biodiversity. The programme was initiated in response to concern over the rate of biodiversity loss. Marine and coastal ecosystems and ES were identified to continue to be degraded around the world. TEEB Oceans and Coasts developed tools, processes and information to enable the value of marine and coastal ES to be assessed. TEEB Oceans and Coasts aims to meet a growing demand for decision-makers to better understand and manage human dependence on healthy ocean ecosystems and biodiversity.

**Aims and objectives**

- Respond to the growing demand from policy makers to better manage human activities and their impact on ecosystems and their services.
- Identify and focus on the policy opportunities that are currently available in order to establish the value of an ES approach and to continue to build ES knowledge in decision-making.
- Bridge the gaps in knowledge on ocean ecosystem services and functions and support the mainstreaming of biodiversity and ecosystem considerations into both national policymaking and broader societal perspectives.

The key objectives of the project are stated as:
- To identify policies that would benefit from better information about the economic, social and cultural value of ocean ecosystems and biodiversity
- To observe and map the societal, cultural and biophysical values we place on the oceans, identifying the underlying drivers for change
- To connect stakeholders to the existing knowledge on oceans, acknowledging the global economic and environmental challenges and exploring potentials for another frame of economic thought
- To develop concept designs and prototype a variety of possible solutions and evolutionary ecosystem-based economic frames;
- To develop a research strategy that better leverages current knowledge, fills high priority gaps and enables improvement of knowledge over time
• To ensure that all stakeholders collectively implement the recommended solutions and policy options in order to develop a compelling argument to achieve the sustainable development of oceans, and a more sustainable future (TEEB 2012, 2013).

Methodology

TEEB Oceans approach is based around a 5 step process:

- Participatory design, collaborative project mapping and design will be used to secure the long-term engagement of relevant stakeholders and key partners in the policy, governance, industry and community/civil society.

- Knowledge Building upon TEEB. TEEB for Oceans and Coasts will develop a holistic ecosystem services framework in an effort to bridge the knowledge gap that exists between the demand for better information about the value of marine resources, and the understanding of how these values can be used to inform policy decisions.

- Prototyping A minimum of four ecosystem-level valuation exercises are planned (sites to be determined) as case studies to demonstrate how holistic valuation approaches can be adapted to respond to specific national policy questions.

- Policy integration The holistic ES framework will be presented in two workshops with regional decision makers. The workshops will promote the use of the tested framework in order to design cost-effective policy instruments and to ensure real policy implementation.

- Communication and outreach The project will make use of the media, print publications, in person communication, social media, and online tools, including a web-based knowledge portal to engage in targeted outreach and awareness-raising (TEEB 2012, 2013).

System of Environmental-Economic Accounting (SEEA)

Recommendations for member states to implement a system of integrated environmental and economic accounting to complement traditional national accounts were contained within Agenda 21 of the United Nations Conference on Environment and Development held in Rio in 1992. Development of the framework and guidance for its implementation has been ongoing since then, with the most recent revision published in 2014 (United Nations 1993; United Nations et al., 2003, 2014). This latest Central Framework for the System of Environmental-Economic Accounting (SEEA) has been adopted as the first international standard for environmental-economic accounting by the United Nations Statistical Commission, and as such aims to support the consistent presentation of accounting information and so facilitate comparison between countries. Additional guidance (European Commission et al., 2013, 2014) supports this central framework, although these latter documents are not endorsed statistical standards.

The SEEA Central Framework is designed to describe the stocks of environmental assets and the related flows of natural resources, and to increase understanding of the links between the economy and the environment (Figure A2.2). It applies the accounting concepts, structures, rules and principles of the System of National Accounts (SNA; European Commission et al., 2008), with the aim of allowing environmental information to be coherently integrated with standard measures of economic structure, wealth and activity. The SEEA is designed as full system of accounts, but the authors recognise that it can be implemented partially, as countries may focus on a selection (at least initially) depending on the most pressing environmental issues faced.

Environmental assets are described within SEEA as “the naturally occurring living and non-living components of the Earth, together constituting the biophysical environment, which may provide benefits to humanity”. Within the central framework, environmental assets are considered only in terms of the provision of materials and space to economic activity (e.g. mineral resources, energy, timber, water, and land). The SEEA provides for these assets to be documented within supply and use tables, which show flows of natural inputs (e.g. timber), products (goods and services consumed within the economy) and residuals (e.g. waste, return flows of water), as well as in asset accounts, which record the stock of environmental assets at the beginning and end of each accounting period and the changes that occurred within it.
Only natural assets that have an economic value are included in the central framework in monetary terms. However, all natural resources within an economic territory are accounted for in physical terms. The SEEA advocates using observable market prices (following the principles of the SNA) to value all environmental assets, although it also proposes alternative estimated or modelled costs (still based on market prices) where there are no direct observable prices.

The SEEA central framework provides detailed guidance and examples for a range of environmental assets to illustrate completion of the accounts and tables, and also for monetary valuation. However, it was designed to provide a high-level classification and the authors recognise that further work is required on i) finer-level classifications (such as for land use); ii) standard definitions, concepts and structures; and iii) consistent valuation techniques.

The same accounting principles, structures and classifications described within the SEEA central framework are adopted within the complementary SEEA Experimental Ecosystem Accounting (European Commission et al., 2013). In common with the central framework, a spatial approach is taken, both monetary and physical metrics are considered, and the accounting seeks to record opening and closing stocks and the flows from and between environmental assets. The Experimental Ecosystem Accounting takes a broader perspective than the Central Framework as it looks beyond the role of environmental assets in economic activity and considers the flow of ecosystem services, defined as “the contributions of ecosystems to benefits used in economic and other human activity”. It is also intended to be implemented at sub-national as well as national scales. The approach is not currently an international standard, but is designed to be tested and developed as countries seek to implement ecosystem accounting.
The framework considers the relationship between stocks and flows. The stocks are represented by distinct spatial areas, which each comprise an ecosystem asset with a range of ecosystem characteristics (Figure A2.3). The flows within and between ecosystem assets are described, respectively, as the intra- and inter-ecosystem flows, with the latter recognising the interdependence of the different ecosystem assets. Flows are also considered in terms of ecosystem services, reflecting the interactions between people and the environment. The Millennium Ecosystem Assessment category of Supporting Services is not included within SEEA, but instead can be considered as synonymous with the intra- and inter-ecosystem flows described above. The benefits that arise from ecosystem services are divided into two categories: the goods and services that are part of the economic activity documented by the System of National Accounts (SNA benefits) and the non-SNA benefits, which are outside this system and generally are not traded on markets. The framework defines an ecosystem in broad terms – “a dynamic complex of plant, animal and microorganism communities and their non-living environment interacting as a functional unit” – allowing abiotic services, such as energy resources, to be considered.

The central concepts in accounting for the environmental stocks and flows are that the ecosystem assets should be measured in terms of i) the area covered by, and the condition of, an ecosystem (its extent and quality), and ii) the “basket” of ecosystem services generated by the asset at a particular point in time. The focus from an accounting perspective is to consider an expected basket of ecosystem services (generally based on current use patterns), but the framework can also be used to assess the trade-offs between different baskets of ecosystem services that would arise under different scenarios.

Three types of units for ecosystem accounting are proposed, reflecting the spatial basis of the framework: basic spatial units (BSU), land cover/ecosystem functional units (LCEU) and ecosystem accounting units (EAU), in order of increasing spatial scale. BSUs should be small (e.g. 1km²) cells that are grided across the relevant area and attributed initially with basic information such as land.
cover. Additional information on ecosystem characteristics (e.g. topography, species abundance, land use) and ecosystem services can be added depending on the purpose of the accounting. LCEUs can be considered the ecosystem assets, as they are intended to represent ecosystem-level delimitation, and cover areas satisfying a predetermined set of ecosystem characteristics. An LCEU can be formed from aggregating BCUs with the same core characteristics, or the LCEU can be overlaid with a grid to form BCUs. Sixteen provisional classes of LCEU have been defined (Table A2.1), based on the FAO Land Cover Classification System (FAO, 2009). Finally, EAUs represent the governance level, and should take account of administrative boundaries, designated areas, catchment boundaries and similar factors that reflect why changes in the area over time need to be understood and reported. EAUs may contain a range of LCEUs.

<table>
<thead>
<tr>
<th>Description of classes</th>
<th>Description of classes</th>
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<tbody>
<tr>
<td>Urban and associated developed areas</td>
<td>Sparsely vegetated areas</td>
</tr>
<tr>
<td>Medium to large fields rainfed herbaceous cropland</td>
<td>Natural vegetation associations and mosaics</td>
</tr>
<tr>
<td>Medium to large fields irrigated herbaceous cropland</td>
<td>Barren land</td>
</tr>
<tr>
<td>Permanent crops, agriculture plantations</td>
<td>Permanent snow and glaciers</td>
</tr>
<tr>
<td>Agriculture associations and mosaics</td>
<td>Open wetlands</td>
</tr>
<tr>
<td>Pastures and natural grassland</td>
<td>Inland water bodies</td>
</tr>
<tr>
<td>Forest tree cover</td>
<td>Coastal water bodies</td>
</tr>
</tbody>
</table>

In measuring the extent and quality of, and supply of ecosystem services from, an ecosystem asset, appropriate characteristics of the ecosystem must be defined, together with indicators that detect changes in those characteristics. CICES (see below) is proposed as an appropriate classification system for ecosystem services, although its limitations are noted, particularly that it does not include abiotic services, and also does not classify the ecosystem assets, processes and characteristics that form the other key aspects of the framework. The experimental nature of the framework is evidenced by the limited guidance for its implementation: minimal information on, for example, metrics is provided with no attempt made to address any known challenges such as the selection of indicators or to provide detailed examples or case studies. Other issues, such as scale and aggregation and even how accounting for ecosystem services and ecosystem degradation could be used to augment SNA economic accounts, are discussed in broad terms but no specific recommendations are provided. Needs for further research are identified, which encompass issues related to physical measurement, monetary valuation and communication of outputs.

### Application of SEEA: Integrated Environmental and Economic Accounting for Fisheries (SEEAF)

Fisheries are cross-sector natural assets, not subject to direct management such as that of farmed livestock, where growth of animals is enhanced and controlled by human activities. The existing SNA records production and changes in livestock. Therefore, for farmed livestock, consequences of changes such as stock depletion are fully accounted for. However, Under the SNA, over-exploited fish stocks only have income from recorded but not the corresponding depletion of the fish stocks.

The SEEAF was developed to address the dual impacts of excessive exploitation levels and habitat degradation which result in loss, or reduction, of the economic value of goods and services provided by aquatic ecosystems. The ecosystem wide approach adopted by the SEEAF aimed to address the cross-sectoral issues within fisheries management and cover all the important environmental-economic interactions to address threats to the health of fish habitat.

Threats such as changes in land use, pollution, forest cover, water flow, and other environmental components are included in the SEEAF assessment. The SEEA relationship to the SNA (as satellite accounts to SNA) links the SEEAF to the full range of economic activities. This provides a comprehensive classification for environmental resources, thereby, including in the SEEA information about all critical stocks and flows that may affect fisheries.

Methodological and practical guidelines for application of the SEEA standard approaches to fisheries were developed in 2004. The Integrated Environmental and Economic Accounting for Fisheries handbook aimed at providing a common framework for organizing economic and environmental information. The approaches and information provided were intended for data producers from national statistical offices, fisheries ministries or research institutes (UN FAO 2004).
Aims and methodology

The System of Economic Accounting for Fisheries (SEEAF) has several major objectives:

- Clarify the SNA and SEEA concepts and expand them for fisheries and related resources.
- Harmonise accounting practices for fisheries so that accounts are comparable across all countries.
- Promote accounting for the fisheries sector.
- Provide a guide and a training tool.

Stock and physical account calculation is outlined within the following approaches:

- Record captured fish as production
- Record the value of prices being sold at market
- Extraction from natural waters is attributed to the country where fish are caught whilst economic production is attributed to the country the vessel fishes from.
- Resource rent (value) for each species is estimated by subtracting the marginal exploitation costs (e.g., employee costs and costs of fixed capital) from the value of production (quality and price of catch).

Stock accounts for fish habitat

- Can be constructed for marine habitat types (such as mangroves, sea grass beds, coral reefs, lagoons) and also terrestrial ecosystem resources that affect fish habitat, such as forests.
- Physical accounts can be expressed in a combination of area (e.g., hectares) and qualitative classifications such as excellent, good, fair, bad.
- Degradation of aquatic ecosystems and habitats can be measured in quantitative (e.g., loss of area due to conversion or other factors) and qualitative terms or a combination thereof.
- Species diversity and changes over time can be expressed in numbers and proportions of the observed species (UN FAO, 2004).

In an SEEAF case study for the UK the Physical and Economic Accounts for UK Fisheries were produced in 2003 by the Office for National statistics (ONS 2003). The report tests out one of the number of methodologies described in the UN Fisheries Accounts Handbook to value five of the fish stocks captured by the UK fishing fleet (cod, haddock, whiting, plaice, sole).

Accounts were produced for:

1. Physical stock (opening and closing stocks of the whole stock regardless of which country’s fleet is harvesting it). Using survey data and international catch and effort data available through Cefas, collated through involvement with ICES working groups and the European Commission’s Scientific, Technical and Economic Committee for Fisheries (STECF) to estimate fish stocks (Figure 2.5).
2. Economic accounts, as a measure of the net present value of income (rent) that the UK’s share of the stock is expected to generate in the future (based on costs incurred and revenues generated by the UK fleet in the relevant year). Using data available through National accounts of landings, produced by ICES working groups and economic data for UK fishing industry collated by the industry group Seafish (Figure A2.4).

The Physical and Economic Accounts for UK Fisheries showed impact of over exploitation over a 10 year period (1991-2000) for the majority of species stocks. Overall, the stocks declined over the 20 year period (1981-2000) and levels of economic rent tended to be negative (ONS, 2003). Economic interpretation of the negative rents suggested that the fishery was being managed in an economically sub-optimal way. Useful extensions of the work that were suggested included calculation of rent based on the markets for fishing licenses in the UK, as this is a more accurate method of rent estimation. Analysis with a bio-economic model was also suggested as this could show how resource rent might change with the rate of exploitation of the fish stock. An ideal bio-economic model would attempt to measure the maximum potential value of a fish stock. Challenges to producing such a model include the nature of shared fisheries in the UK limiting the availability of data required to populate the model. Incorporation of further social costs of fishing (costs of completing stock assessments, costs of enforcement, costs of subsidies etc.) would be likely to show the negative resource rents calculated could still be underestimates (social costs of fishing in UK may far outweigh the benefits) (ONS, 2003).
Mapping & Assessment of Ecosystems and their Services (MAES)

The EU Biodiversity Strategy¹ to 2020 requires that the state and condition of ecosystems and their services are assessed to support their maintenance and restoration. A Working Group on Mapping and Assessment of Ecosystems and their Services (MAES) was set up under the Common Implementation Framework (CIF) to deliver the objective. The Working Group has developed a coherent analytical framework to be applied by the EU Member States. It started as part of the EU response to the Convention on Biological Diversity in Aichi in 2010, with the initial reporting in 2014 but will continue adding new national case studies through to 2020. The approach is strategic from the EU to deliver their obligations and devolved through national governments.

Ecosystems (i.e. biological communities of interacting organisms along with their inert physical environments) can be large and complex containing several habitats (environments that support organisms); their dynamic functional characteristics mean they are seen as providing services to man. MAES aims to map ecosystems to provide a direct link to ecosystem services mapping (ESS) and Ecosystem capital accounts (ECA) approaches. The project addresses both ecosystem structure (by mapping biophysical delineation and health) and functions (as predisposition to deliver ecosystem services). It covers marine and terrestrial systems.

The primary goal of MAES is to develop a consistent approach that can be applied across Europe by individual nations or their components that will allow ecosystem services and natural capital to be fairly assessed when devising or assessing policy for development. The focus is on conservation of biodiversity and habitats as set out at Aichi², so energy is not explicitly highlighted, but it is recognised to be an important driver of change. The initial timeframe is to meet targets set for 2020, but the approach is considered to be capable of continuing into the medium term and is designed so that it can be extended or updated as new data become available.

The approach has two strands, the first maps the extent and condition of habitats that it considers to equate to ecosystems; it is difficult to produce a map of habitats for Europe, ecosystems with their interactions, multiple communities and complex structures would be impossible. Initially extant data and typologies were used (including CORINE Land Cover, JRC forest, Soil Sealing, Open Street Map, Potential natural vegetation, soil, field capacity, soil water balance, environmental zone, DEM) which were qualified with field data describing vegetation classes.

The second strand addresses the sensitivity of different ecosystem services and natural capital to different drivers. This is achieved by the development of indicators of the “healthy state” of an ecosystem and its services through pilot studies, literature searches and expert panels. The outputs,

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² http://www.cbd.int/sp/targets/
or “MAES summary tables” should define the capacity of an ecosystem to deliver services in measurable quantitative units that will be a method of integrated ecosystem assessment. The impacts within the tables are qualified into four classes on two criteria the availability of data and the effectiveness of the information for policy.

MAES uses multiple frameworks from the overarching concept (Figure A2.5), to the tactical management of pilot studies.

**Figure A2.5 Conceptual framework for EU wide ecosystem assessment (MAES)**

The habitat mapping employs a slight modification of the top two levels of the EUNIS habitat classification with 12 main ecosystem (or habitat) types; this cross links to the Habitats Directive. As satellite derived maps are used (with a minimum mappable unit of approximately 25m x 25m) some of the habitats cannot be included (Pelagic water column; Woodland fringes and clearings and tall forb stands; Terrestrial underground caves, cave systems, passages and waterbodies)

Ecosystem services are described using the typology presented in CICES (version 4.3). MAES links and cross tabulates CICES to MA and TEEB categories.

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3 http://biodiversity.europa.eu/maes
4 http://eunis.eea.europa.eu/habitats.jsp
Appendix 3: Key studies that have adapted the frameworks and typologies and their applications

Fisher et al. (2008, 2009)

Fisher et al. (2008, 2009) seek to define and classify ecosystem services in such a way as to facilitate operational assessment and policy-relevant research. They first propose that ecosystem services “are the aspects of ecosystems utilized (actively or passively) to produce human well-being”, a definition that they see as highlighting the ecological nature of services and the possibility for indirect utilisation. They support this definition by proposing services are qualified by the terms intermediate and final to distinguish between indirect and direct consumption, although the definition of a service as intermediate or final is context dependent. Benefits describe the products that directly impact human welfare, and typically result from ecosystem services combined with human or built capital (Figure A3.1). Only the benefits generated by final services can be aggregated during accounting and valuation, as to add the value of services risks double counting.

Figure A3.1 The conceptual relationship between intermediate services, final services and benefits (Fisher et al., 2008)

<table>
<thead>
<tr>
<th>Intermediate Services</th>
<th>Final Services</th>
<th>Benefits</th>
</tr>
</thead>
<tbody>
<tr>
<td>Soil formation</td>
<td>Nutrient cycling</td>
<td>Clean-water</td>
</tr>
<tr>
<td></td>
<td>Pollination</td>
<td>Food productivity</td>
</tr>
<tr>
<td>Primary production</td>
<td>Water regulation</td>
<td>Water regulation</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Drinking water</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Fruit</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Flood protection</td>
</tr>
</tbody>
</table>

The authors highlight the importance of marginality in ecosystem service assessments. The value of a service is a function of small changes in its flow of that service. They also note that the scale of this marginal change has to be meaningful to the decision context. It is important to also understand the drivers and pressures on a system, and how it is changing or might change, with particular emphasis on safe minimum standards and tipping points. Spatial assessment is also advocated as a meaningful approach to support policy relevance. A classification system for ecosystem services is not proposed. Instead, it is suggested that multiple systems are appropriate in reflection of the dynamic complexity of the ecosystem, and the varying decision contexts, which can include education, cost-benefit analysis, landscape management, public policy and welfare equity, and meeting multiple objectives.

Examples of application

There is no doubt that this work has been highly influential within the academic community (as indicated by the 280 citations for the 2009 paper in the Web of Science database, which increases to 789 in Google Scholar) and so is an integral component of current thinking on ecosystem services assessment. Less easy to identify are examples of the direct application of the framework in a management context, although the concepts described have influenced, for example, the UK National Ecosystem Assessment.
An ecosystem services framework was proposed by Balmford et al. (2008) as part of the scoping process for TEEB, and was later refined in an academic paper (Balmford et al., 2011). Their conceptual approach was that the net economic consequences of policy actions to address biodiversity loss could be determined by quantifying and mapping individual benefits (such as fisheries, pollination, water provision) under different scenarios, and calculating the net changes (Figure A3.2).

Figure A3.2. Framework for assessing the economic consequences of losing biodiversity and ecosystems (Balmford et al., 2011)

The typology produced to support this concept was an example of two key developments in improving the operational usefulness of ES frameworks. Firstly, it was not constrained by the strict definition of ‘ecosystem’ (as being related to the biota) but instead explicitly also considered the abiotic components of the environment. The classification also aimed to clearly define the points at which human welfare is directly affected, facilitating monetary valuation. This latter development reflects criticism that the Millennium Ecosystem Assessment framework does not support economic assessment because its classification of services does not effectively distinguish between ecological processes and the beneficial products that result from them, and hence risks double counting (Fisher et al., 2008, Boyd and Banzhaf, 2007; Wallace, 2007).

In reflection of the lack of clarity in the definition of ecosystem ‘services’, Balmford et al.’s classification does not employ this term, but instead considers (Figure A3.3):

1. core ecosystem processes (the basic ecosystem functions);
2. beneficial ecosystem processes (that directly underpin benefits for mankind); and
3. benefits (the discrete products of ecosystem processes that directly impact human wellbeing and which can, in principle, be valued in monetary terms).
In producing the framework, Balmford et al. (2008) conducted 17 thematic reviews to consider the state of knowledge of the links between wild nature and certain high-level benefit categories (including, for example, pollination, fisheries, water, outdoor activities and non-use values). These reviews considered issues such as the relationship between habitat area and benefit provision, whether mapping the provision of, and changes to, benefits was possible, the main threats and the likelihood of abrupt changes. Although not assessed as a thematic review in itself, the framework did consider hydroelectric energy, in terms of its relationship to the beneficial ecosystem processes related of water regulation, water purification, water provisioning and global climate regulation. This exercise also resulted in the definition of a matrix to prioritise further assessment of these key themes according to their importance to human wellbeing and the feasibility of undertaking a detailed assessment within a twelve month period.
The work by Balmford et al. (2008) was incorporated within the TEEB framework, but has also been used directly in other contexts. The typology was adapted for the marine environment by Saunders et al. (2010), in work commissioned by the Crown Estate to examine valuation of the UK’s marine estate as a tool to support sustainable management. Their adapted classification included an additional category of carrier services (as had previously been proposed by e.g. De Groot, 2006), to describe the provision of space for infrastructure or transport. Valuation methods were also suggested, and some static, baseline monetary values obtained from secondary sources were provided for a limited set of benefits (primarily those resulting from provisioning services).

Atkins et al. (2011a,b), The Ecosystem Approach combined with DPSIR

The Atkins et al (2011a, b) paper proposes an Ecosystem Approach that combines Drivers-Pressures-State-Impact-Response (DPSIR) with ecosystem services and societal benefits (ES&SB). DPSIR was an approach proposed in the late 1990s by the European Environment Agency (Jensen 1999). Although it has been displaced to some extent in terrestrial research by global drivers of change and ecosystem services it remains a useful tool. There are criticisms of DPSIR (often from social scientists) suggesting that it introduces bias (see for example Svarstad et al., 2008 & Tscherning et al., 2011). It does seem to have retained a stronger following in marine system management (see for example Borja et al. 2006 – the most cited paper) and about 60% of publications describe marine systems. ES&SB have grown in popularity since the publication of the Millennium Ecosystem Assessment (MA) and the relationship between final ES and the SB is well presented in the text. The approach is not presented as an operational tool, but more focussing on the key issues in improving our understanding of management of marine management.

Aim and methodology

The aim is to provide a specific framework to support decision making for the marine environment. It seeks to link ecosystem services, social benefit and DPSIR in a consistent approach (Figure A.4.3.1). ES&SB is used to define the system boundary within which State Change and Impact of DPSIR are considered to occur: Drivers, Pressures and Responses may also occur within the domain, but are also likely to have major components beyond the boundaries. The approach requires scientific analytical and/or monetary enumeration of both fundamental and final ecosystem services. The social benefits may have market values, but it is recognised that these may not reflect the value of individual stakeholders. The paper calls for further analysis (market analysis, productivity gains and losses, production function analysis, hedonic pricing, the travel cost method, contingent valuation, the choice experiment method, damage costs avoided, defensive expenditures, relocation costs, replacement/substitution costs and restoration costs) and proposes that judgemental estimates should be valued higher than market prices.

Competing activities will each generate independent DPSIR that interact by different Pressures affecting the change in State (Figure A3.4). However there is usually incomplete evidence to make a full comparison, so the State Changes may not be readily quantifiable and the Impacts are often identified through circumstantial relationships.

Atkins et al (2011a) use the Convention on Biological Diversity (CBD, 2000) to define the Ecosystem Approach, then describe the development of ecosystem service classifications before opting for a modification of Beaumont et al. (2007). This is basically de Groot’s classification with Option use values added that will cover future potential.

Case study area

Two demonstrators are used: first all UK waters for marine aggregates extraction; and second Flamborough Head management for marine biodiversity. The former is seen as a more generic case as there is no full assessment of ecosystem services (fundamental, final or societal benefit) from the activity, but the stakeholders involved in Responses (who may benefit from the tool) are highlighted; i.e. those with responsibility for the areas (The Crown Estate) and regulators.

The second example is more sharply defined, yet reflects greater multifunctional complexity and is related to the existing Management Plan with multiple DPSIR cycles being postulated. Even here, where monitoring is underway, it is difficult to unequivocally relate State changes to Pressure.
Figure A3.4 Atkins et al’s (2011a, b) Ecosystem approach combined with DPSIR
Appendix 4: Spatial Decision Support Tools

LUCI: Land Utilisation and Capability Indicator

LUCI is the second generation extension and accompanying software implementation of the Polyscape GIS framework designed to explore spatially explicit synergies and tradeoffs amongst ecosystem services land management (Jackson et al 2013) and policy implementation across sectors (e.g. water, biodiversity, agriculture). Similarly to InVest (next section), it was designed to facilitate negotiation rather than to prescribe actions. Land owner preferences (captured through parameter, data and condition editing capabilities) have been used to ground-truth land cover data and engage local stakeholders. Polyscape was developed initially using a Welsh catchment as a case study and algorithms were created to explore the impacts of land cover change on:

- flood risk: topographical routing of water accounting for storage and infiltration capacity as function of soil and land use
- habitat: connectivity using Beal and quality using abiotic requirements for habitat
- erosion: slope, curvature, contributing area, land use, soil type
- carbon sequestration: based on soil and vegetation
- agricultural productivity: based on fertility, slope, drainage and aspect
- sediment delivery: erosion combined with detailed topographical routing
- water quality: export coefficients combined with water flow and sediment delivery models
- Tradeoffs/synergies: various layering options with categorised service maps; e.g. Boolean, conservative, weighted arithmetic.

Each of the models classifies elements within the landscape into one of five categories: very high existing value, high existing value, marginal value, opportunity for change or high opportunity for change. These classifications are visualised using a five-way colour system.

Algorithms allow identification of where change might be beneficial. As well as single criteria landscape valuations there is potential for identifying multiple service synergies and a traffic light coded impact map produced. Polyscape was developed to be a very spatially explicit tool at a resolution of 5m x 5m for the Digital Elevation Model and working with detailed catchment level data but with the potential to include more coarsely resolved data. It has been extended and re-packaged as LUCI and widely available national data incorporated.

Scenarios can be constructed by modifying input parameters and/or input land use data. The ES framework for LUCI has not been explicitly identified but it appears to be based on the UKNEA.

InVEST: Integrated valuation of Environmental services and Tradeoffs

InVEST is a tool for exploring how changes in ecosystems are likely to lead to changes in benefits that flow to people (Figure A4.1). Similarly to LUCI InVEST is a GIS based tool with add-on models for a range of ecosystem services, risk assessment and tradeoffs (Nelson et al. 2009, Tallis et al. 2011). It is part of a wider programme of work: the ‘Natural Capital project’ led by Stanford University, the Nature conservancy and the World Wildlife fund. The InVEST tool can operate as stand-alone software that can be incorporated into freeware such as the GIS package QGIS as well as ArcGIS.

Similarly to LUCI the overriding purpose of InVEST is to support decision making and to encourage negotiation by discussion of spatially explicit trade-offs. InVEST models are therefore spatially explicit, using maps as information sources and producing maps as outputs. InVEST returns results in either biophysical terms (e.g. tons of carbon sequestered) or economic terms (e.g. net present value of that sequestered carbon). InVEST includes tools to facilitate ES analyses and models for particular ES:

**Tools to facilitate ES analyses:**

- Overlap analysis model
- Coastal Exposure and vulnerability
Models included in InVEST

- Habitat Quality
- Habitat Risk Assessment
- Marine Water Quality
- Carbon Storage and Sequestration: Climate Regulation
- Water Yield: Reservoir Hydropower Production
- Nutrient Retention: Water Purification
- Sediment Retention: Avoided Dredging and Water Purification
- Pollinator Abundance: Crop Pollination
- Unobstructed Views: Scenic Quality Provision
- Visitation: Recreation and Tourism
- Wave Attenuation & Erosion Reduction: Coastal Protection (*only in ArcGIS version)
- Blue Carbon Storage and Sequestration: Climate Regulation (*coming soon)
- Managed Timber Production
- Wave Energy Production
- Offshore Wind Energy Production
- Marine Finfish Aquacultural Production
- Marine Fisheries Production (*coming soon)
- Managed Timber Production
- Managed Timber Production
- Managed Timber Production

In principle, InVEST can be applied at any scale, depending on data availability, although in practice there may be constraints for some of the models. Case studies have been international. Scenarios are developed externally with stakeholders and introduced as GIS layers or data tables.

InVEST uses a simple framework delineating “supply, service, and value.” “Supply” represents what benefits are potentially available from the ecosystem (i.e. what the ecosystem structure and function can provide). For example, the wave attenuation and subsequent reduction in erosion and flooding onshore provided by a particular location density of mangrove forest. “Service” incorporates demand and thus uses information about beneficiaries of that service (e.g., where people live, important cultural sites, infrastructure, etc.). “Value” includes social preference and allows for the calculation of economic and social metrics (e.g., avoided damages from erosion and flooding, numbers of people affected).

Figure A4.1 Schematic of the decision-making process in which InVEST is embedded. Stakeholders create scenarios that are assessed for environmental service value by biophysical and economic models that produce several types of outputs.
ARIES: Artificial Intelligence for Ecosystem Services

ARIES (Artificial Intelligence for Ecosystem Services) is a suite of applications delivered to the user via the Internet for assessment and valuation of Ecosystem Services (Bagstad et al., 2011). ARIES systematically uses Bayesian statistical approaches to map ecosystem services provision, use, and spatial dynamics. It also provides a modelling framework which can run external models in addition to its internal Bayesian probabilistic models. It incorporates a conceptual framework for mapping services comprising: source (ES), users (human beneficiaries), and sinks (depleted ES flows), and then maps flows of ES between source and use locations. Thus it aims to map potential ecosystem services, the beneficiaries, and the landscape routes which deliver those services. Valuation is currently lacking, in ARIES but is planned to be included. The structure allows users to supply data and knowledge at fine-scales to develop locally relevant case studies.

ARIES has been used for spatial mapping and modelling of ecosystem services, spatial planning and land management in a series of data rich case studies. ARIES internal probabilistic Bayesian statistical models can be based on sparse data, relatively simple models, and incorporate expert opinion. This means that it is possible to model ecosystem services in many different situations and where there are less quantitative measures e.g. cultural services. ARIES can also run external models, including ecological process-models employing a model-wrapping mechanism. ARIES can handle scenarios as separate data and information layers which are fed in as inputs. Certain modules have built-in scenario editors. The scenarios can be added as separate data and information layers which are fed in as inputs. Certain modules have built-in scenario editors.

ARIES does not have a framework as such but provides examples of source, sink, use, and flow characteristics for ARIES modules, such as that shown below:

<table>
<thead>
<tr>
<th>Service</th>
<th>Carbon sequestration &amp; storage</th>
<th>Open space proximity</th>
</tr>
</thead>
<tbody>
<tr>
<td>Benefit type</td>
<td>Provisioning</td>
<td>Provisioning</td>
</tr>
<tr>
<td>Medium/units</td>
<td>Tons CO2 absorbed/emitted</td>
<td>Open space (abstract units, 0-100)</td>
</tr>
<tr>
<td>Scale</td>
<td>Global</td>
<td>Walking distance</td>
</tr>
<tr>
<td>Movement</td>
<td>Atmospheric mixing</td>
<td>Walking simulation</td>
</tr>
<tr>
<td>Decay</td>
<td>None</td>
<td>Gaussian</td>
</tr>
<tr>
<td>Rival?</td>
<td>Rival</td>
<td>Nonrival</td>
</tr>
<tr>
<td>Source</td>
<td>Vegetation &amp; soil C sequestration</td>
<td>Open spaces esp. in urban areas</td>
</tr>
<tr>
<td>Sink</td>
<td>Stored C release (fire, land use change)</td>
<td>Obstructions (e.g., highways)</td>
</tr>
<tr>
<td>Use</td>
<td>CO2 emitters</td>
<td>Property/housing value</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Service</th>
<th>Aesthetic viewsheds</th>
<th>Flood regulation</th>
</tr>
</thead>
<tbody>
<tr>
<td>Benefit type</td>
<td>Provisioning</td>
<td>Preventive</td>
</tr>
<tr>
<td>Medium/units</td>
<td>Scenic beauty (abstract units, 0-100)</td>
<td>Water (runoff, mm/yr)</td>
</tr>
<tr>
<td>Scale</td>
<td>Viewsheet</td>
<td>Watershed</td>
</tr>
<tr>
<td>Movement</td>
<td>Line of sight</td>
<td>Hydrologic flow</td>
</tr>
<tr>
<td>Decay</td>
<td>Inverse square</td>
<td>None</td>
</tr>
<tr>
<td>Rival?</td>
<td>Nonrival</td>
<td>Rainfall &amp; snowmelt</td>
</tr>
<tr>
<td>Source</td>
<td>Mountains, water bodies, etc.</td>
<td>Water absorbed by soil &amp; vegetation</td>
</tr>
<tr>
<td>Sink</td>
<td>Visual blight</td>
<td>Economic assets in floodplains</td>
</tr>
<tr>
<td>Use</td>
<td>Property/housing value</td>
<td></td>
</tr>
</tbody>
</table>

SIAT: Sustainability Impact Assessment

SIAT (Sustainability Impact Assessment Tool) was the main product of SENSOR, an integrated EU project (Framework 6) with non-EU partners (China, Brazil, Argentina and Uruguay) designed to substantiate the impact assessment process and provide tools for environmental, social and economic effects of multifunctional land use in European Regions.

SIAT is a quantitative multi-modelling tool providing prospective scenario assessment across disciplines, sectors and sustainability dimensions. It was designed to support assessment of new policies on six land use sectors: agriculture, forestry, nature conservation, transport infrastructure, energy and tourism for the target year 2025 covering 574 regions in the EU (Helming et al. 2008). Each simulation computes 87 impact indicators and nine land use functions that illustrate the policy
impact on social, economic and environmental goods and services. The main focus within SIAT is on those indicators that highlight and measure the impacts of land use changes (and related policies).

**LEED: The Local Environment and Economic Development (LEED) Toolkit**

LEED was designed to help Local Enterprise Partnerships (LEPs) and Local Authorities (Las) meet their economic growth targets by fully realising the roles the environment can play. It provides an assessment of the opportunities and threats to relationships between the environment and economy to the LEPs plans for increasing local Gross Value Added (GVA), and has three levels with varying intensity and detail. It is being trialled as a working tool in East Anglia with expert advice in implementation by Natural England for LEPs and Las but is still in a developmental phase.

**MIMES: Multiscale integrated Earth Systems model**

The multiscale integrated Earth Systems model (MIMES; Boumans and Costanza 2007) is a multi-scale, integrated suite of models that aims to assess the true value of ecosystem services to enable ecosystem managers to quickly understand the dynamics of ecosystem services, how their services are linked to human welfare, and how their function and value might change under various management scenarios. The models quantify the effects of land and sea use change on ecosystem services and can be run at global, regional, and local levels.

The three major objectives are:

- A suite of dynamic ecological economic computer models specifically aimed at integrating our understanding of ecosystem functioning, ecosystem services and human well-being across a range of spatial scales.
- Development and application of new valuation techniques adapted to the public goods nature of most ecosystem services and integrated with the modelling work.
- Delivery of the integrated models and their results to a broad range of potential users.

MIMES simulates ecosystems and socio-economic systems in space by modelling systems and the interactions between them over time, and calculates specific values of ecosystem services through marginal cost pricing. The tool provides estimates of ecosystem service values for land use decision making and marine spatial planning through scenario analyses, and considers the production of an array of ecosystem services.
Appendix 5: Strengths, Weaknesses, Opportunities, Threats (SWOT) Analysis for Individual Ecosystem Service Frameworks and Tools

SWOT tables are displayed below in the following order:

1. National Frameworks (UK NEA, NEAFO, NCC)
2. International Frameworks (MA, TEEB, SEEA, CICES)
3. National Applications (Energy Scapes, TIDE)
4. International Applications (TEEB Oceans and Coasts, SEEA Fisheries, MAES)
5. Academic work to advance ES frameworks and assessment in relation to impact assessment (Fisher et al., Balmford et al., Hattam et al.)
6. Spatial assessment tools (LUCI, INvest, Aries, SIAT, LEED)
### Strengths

- Focus on methods and tools to embed ES assessment into decision making is of direct relevance to assessing the impact of energy development.
- Continues to build on existing national ES assessment frameworks which relate to internationally recognised approaches, supporting comparability of assessments.
- Assesses ESs, impacts and changes in relation to natural capital and regional and national economic performance, thus utilising a common, transferable economic language.
- Land use change models and indicators for marine environment ES health can potentially be applied to energy resource and energy development scenarios.

### Weaknesses

- Not all tools and methodologies identified have been applied, particularly in respect to energy development scenarios. (Although case studies are presented which are discussed in the opportunities section).

### Opportunities

- Worked examples, applying advances could be conducted for energy development scenarios (e.g. applying models, particularly those linking land use change to marine and coastal ecosystems for proposed and currently leased energy sites)

### Threats

- Without further worked examples, review and improvements, targeting specific policy and planning areas (such as energy generation) the framework, tools and methodologies may not be fully adopted in the decision making process.

### UK NEA and NEA FO

- Takes a more holistic approach to assessment – the framework does not just focus on the endpoints (benefits) and their value, but includes consideration of the status of the ecosystem as an integral component.
- Explicitly includes abiotic elements, and so allows consideration of all environmental features including energy.
- Seeks to build on existing national ES assessment frameworks and international standards, supporting comparability of assessments.
- Considers ecosystem status, uses, approaches and metrics (e.g. thresholds, safe limits, quality and quantity of habitats and species) that are well recognised in existing natural resource management strategies and, thus, the framework integrates existing and proposed assessment methods.
- The framework considers changes in spatial units over time, rather than static absolute values.

### NCC

- Metrics for assessment are not fully defined, so the framework remains more conceptual than operational.
- Shortage of detailed worked examples also limits operational usefulness.
- The framework is designed for a broad-based assessment of natural assets at a national level, and may not be applicable at smaller scales.

### National: ES Frameworks and Classifications (Typologies)

<table>
<thead>
<tr>
<th>Strengths</th>
<th>Weaknesses</th>
<th>Opportunities</th>
<th>Threats</th>
</tr>
</thead>
<tbody>
<tr>
<td>• Focus on methods and tools to embed ES assessment into decision making is of direct relevance to assessing the impact of energy development.</td>
<td>• Not all tools and methodologies identified have been applied, particularly in respect to energy development scenarios. (Although case studies are presented which are discussed in the opportunities section).</td>
<td>• Worked examples, applying advances could be conducted for energy development scenarios (e.g. applying models, particularly those linking land use change to marine and coastal ecosystems for proposed and currently leased energy sites)</td>
<td>• Without further worked examples, review and improvements, targeting specific policy and planning areas (such as energy generation) the framework, tools and methodologies may not be fully adopted in the decision making process.</td>
</tr>
<tr>
<td>• Continues to build on existing national ES assessment frameworks which relate to internationally recognised approaches, supporting comparability of assessments.</td>
<td></td>
<td>• Links to national and international accounting frameworks</td>
<td>• The NCC was established for an initial period of three years, which comes to an end in 2015, suggesting a risk that the framework will remain conceptual.</td>
</tr>
<tr>
<td>• Assesses ESs, impacts and changes in relation to natural capital and regional and national economic performance, thus utilising a common, transferable economic language.</td>
<td>• Metrics for assessment are not fully defined, so the framework remains more conceptual than operational.</td>
<td></td>
<td>• The NCC was established for an initial period of three years, which comes to an end in 2015, suggesting a risk that the framework will remain conceptual.</td>
</tr>
<tr>
<td>• Land use change models and indicators for marine environment ES health can potentially be applied to energy resource and energy development scenarios.</td>
<td>• Shortage of detailed worked examples also limits operational usefulness.</td>
<td></td>
<td>• The NCC was established for an initial period of three years, which comes to an end in 2015, suggesting a risk that the framework will remain conceptual.</td>
</tr>
</tbody>
</table>
**INTERNATIONAL: ES Frameworks and Classifications (Typologies)**

<table>
<thead>
<tr>
<th>Strengths</th>
<th>TEEB</th>
<th>SEEA</th>
<th>CICES</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Provided global awareness of ES approach and categories.</strong></td>
<td><strong>Interprets the economic benefit of ES, thus, building a common economic language of costs and benefits.</strong></td>
<td><strong>Integrates ES into established national accounting systems.</strong></td>
<td><strong>Most recent, detailed and comprehensive systematic framework for classifying ES.</strong></td>
</tr>
<tr>
<td><strong>Provided the first global indication of the need to manage ES and maintain the environments and ecosystems that support them.</strong></td>
<td><strong>Work directed at oceans and coasts is of relevance to marine renewable energy and coastal energy development.</strong></td>
<td><strong>Seeks to develop further international standards for ecosystem accounting</strong></td>
<td><strong>Provides a detailed inventory of classes (final outputs / benefits)</strong></td>
</tr>
<tr>
<td><strong>Aimed at providing information on the state of ES for the needs of decision makers.</strong></td>
<td><strong>Considers integration of ES values into decision making aiding aim of SPLiCE to integrate ES into impact assessment.</strong></td>
<td><strong>Includes abiotic resources, including energy</strong></td>
<td><strong>Provides a means of comparing ES assessments in other regions and development sites.</strong></td>
</tr>
<tr>
<td><strong>Interprets the economic benefit of ES, thus, building a common economic language of costs and benefits.</strong></td>
<td><strong>Outlines how business and industry can manage ecosystem risks and opportunities.</strong></td>
<td><strong>Can be implemented at different spatial scales</strong></td>
<td><strong>Provides a common systematic framework for application within impact assessment of energy developments.</strong></td>
</tr>
</tbody>
</table>

**Weaknesses**

| **Broad scope** focused on wider human well-being (e.g. material minimum for a good life) rather than site specific considerations.** | **TEEB is ongoing and approaches, particularly for oceans and coasts are likely to alter as the study develops.** | **Guidance for the ecosystem services element is still experimental and lacks practical details, so does not advance the treatment of issues such as indicators, scale, and aggregation.** | **CICES is ongoing and the current version (v4.3) is likely to be further revised.** |
| **These broad approaches and aims are challenging to fit within IA frameworks (this is approached, however, by more recent ES studies which further develop and address limitations within the MA approach and framework).** | **Business approaches have focus on defining only risks and opportunities to the business rather than the risk and opportunities to ES required by SPLiCE.** | **Focus remains on measurement of marketed assets: the system for this is much more advanced than for the ecosystem services element.** | |
| **The separation of supporting services has presented challenges for economic accounting of ES in a region due to double counting, as supporting services are present in each of the other ES categories.** | | **The CICES classification framework is proposed, but this does not include abiotic services, which are an explicit part of SEEA.** | |
| | | **Classification frameworks for other components of the system (ecosystem assets, processes and characteristics) are not proposed.** | |
**INTERNATIONAL: ES Frameworks and Classifications (Typologies)**

<table>
<thead>
<tr>
<th>Opportunities</th>
<th>TEEB</th>
<th>SEEA</th>
<th>CICES</th>
</tr>
</thead>
<tbody>
<tr>
<td>- Recognition of drivers of change, as well as assessment of change and future scenarios within the MA provides a concept with which ES assessment can be integrated within impact assessment.</td>
<td>- Worked examples, applying advances could be conducted for energy development scenarios (e.g. applying models, particularly those linking land use change to marine and coastal ecosystems for proposed and currently leased energy sites).</td>
<td>- Potential to develop indicators for application within the SEEA framework. As indicators have not yet been proposed, there is the potential to learn from the development of indicators for statutory monitoring programmes (e.g. under WFD and MSFD) and to consider how these can be utilised in an ecosystem accounting framework.</td>
<td>- Worked examples, applying the CICES classification framework could be conducted for energy development scenarios.</td>
</tr>
<tr>
<td>- The concepts and general framework of the MA provide internationally recognised categories for ES, human well-being indicators, direct and indirect drivers of change.</td>
<td>- TEEB approach to integrate ES with policy and planning decisions has been adopted in national level frameworks (UK NEA).</td>
<td>- Methods to provide evidence of ES economic benefits have been adopted at a national level through the work of the NCC.</td>
<td>- Scope for identification of specific indicators for ES class types (related to energy development impact assessment aims), such as: the indicator species, role of habitat types, and the precise metrics in relation to ES.</td>
</tr>
<tr>
<td>- The MA framework, and the classification categories within, remain broad scale and need refinement to be applied to energy development scenarios. (This process has been undertaken by more recent ES studies).</td>
<td>- TEEB requires adaptation to the scale of energy developments as it focuses on international, national and business applications of ES valuation.</td>
<td>- Progress on further revisions of the Experimental Ecosystem Accounting may be slow: production of the Central Framework took 20 years.</td>
<td>- Detailed identification of classes and class types specific to impact assessment aims needed to aid application of CICES to energy generation impact assessment would benefit from</td>
</tr>
</tbody>
</table>

**Threats**

- The MA framework, and the classification categories within, remain broad scale and need refinement to be applied to energy development scenarios. (This process has been undertaken by more recent ES studies).
- TEEB requires adaptation to the scale of energy developments as it focuses on international, national and business applications of ES valuation.
- Progress on further revisions of the Experimental Ecosystem Accounting may be slow: production of the Central Framework took 20 years.
- Detailed identification of classes and class types specific to impact assessment aims needed to aid application of CICES to energy generation impact assessment would benefit from...
### NATIONAL: APPLICATIONS of ES Frameworks and Classifications (Typologies)

#### Energy Scapes

- Sets energy in a wider whole system, rather than seeing energy as the whole system
- Comprehensive coverage of ecosystem services and landscape
- Assesses perception (and values) of different people (stakeholders) and groups
- Recognises the variability, stability and robustness of people’s feelings towards toward energy development
- Examined the trade-offs between energy and other provision services in the case study
- Utilised existing (NERC) data and collected new data where appropriate
- Modelled changes in ecology from forecast changes in energy system
- Uses standard statistical methods to analyse results
- Doesn’t generate necessarily true values for any ecosystem service (but didn’t claim to).
- Can never be truly be comprehensive – may build expectations
- Only looked at one case study area, so despite claims has no proof of ability to transfer to other locations at different spatial scales
- No single simple metric or value was produced (although it was never intended)
- It takes time to interview or collect information from stakeholders and time to present results back to them (i.e. it needs commitment from all parties).
- Never fully developed plug and play for addition of new models and datasets

#### TIDE

- Applies ES classification to the habitats present within an environment of importance to energy development (in particular tidal energy).
- Uses an ES classification framework to identify key ES of importance at sites at the scale of energy developments rather than more general international and national scales.
- The broad scope and questionnaire approach to gather expert opinion does not provide detail on the role of fine scale habitat types, and in particular the role of species present and level of supply related to specific ecosystem functions.
- Application of the methodology and precision provided for impact assessment and regulatory decisions relies on the precision of the underlying habitat and species data available.

### Weaknesses

- Ecosystem services remain not widely understood and inconsistently defined with no unified approach (CICES is as close as we’ve got and that is changing); ES is becoming less comprehensive than it was (e.g. loss of abiotic, only final service or where human benefit is obvious).
- Whole systems approaches are not seen as scientific.
- Consortium has moved on (half of the original team – 8 out of 16 have moved to new institutions or have left the country).

### Opportunities

- Easy to test in other locations, at different scales
- Methodology is well worked out and peer-reviewed
- Approach is open to expansion with ‘new’ energy technologies (e.g. unconventional gas and hydraulic fracturing)
- Development is needed to translate into the marine environment or beyond Britain (Broad Habitats do cover both, but their effectiveness may need to be assessed)
- Plug and play – an interoperability approach that is designed to prevent obsolescence and allow audit is attractive.
- In field quantification of data relating to ES, such as carbon sequestration and burial (for instance; sampling and analysing carbon buried in sediment within habitats and CO² equivalent fluxes present) within habitats in each estuary would provide greater confidence in ES provision.

### Threats

- In field quantification of data relating to ES, such as carbon sequestration and burial (for instance; sampling and analysing carbon buried in sediment within habitats and CO² equivalent fluxes present) within habitats in each estuary would provide greater confidence in ES provision.

### Further development of evidence of habitat and species roles in providing ES, and role in ecosystem functions in estuaries is required to provide detail at level of efficient impact assessment.
## INTERNATIONAL: APPLICATIONS of ES Frameworks and Classifications (Typologies)

<table>
<thead>
<tr>
<th>Strengths</th>
<th>TEEB Oceans and Coasts</th>
<th>SEEA Fisheries</th>
<th>MAES</th>
</tr>
</thead>
<tbody>
<tr>
<td>Application of the internationally recognised TEEB programme to Ocean and Coast ecosystems.</td>
<td>Provides systematic approach that could be adapted under different marine (and terrestrial/coastal) energy development scenarios. Allowing stocks and physical accounts to be predicted.</td>
<td>Uses existing established frameworks and data</td>
<td></td>
</tr>
<tr>
<td>Directly applied to policy and decision makers needs</td>
<td></td>
<td>Seeks consistency across Europe whilst retaining national perspectives</td>
<td></td>
</tr>
<tr>
<td>Case study sites will apply methods and tools</td>
<td></td>
<td>Integrates remotely sensed information with field mapping and vegetation classification</td>
<td></td>
</tr>
<tr>
<td>Participatory design that will engage stakeholders</td>
<td></td>
<td>Provides indicators of condition of ecosystem services, qualified by the spatial co-registration of information</td>
<td></td>
</tr>
<tr>
<td>Identifies and approaches key evidence gaps</td>
<td></td>
<td>Aims to deliver numeric indicators</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Good publications, web presence and Ecosystem Services Partnership Visualisation tool and MAES digital atlas</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Well-structured approach</td>
<td></td>
</tr>
</tbody>
</table>

### Weaknesses

- Currently aimed at broader issues such as national level policy questions, rather than individual energy developments TEEB Oceans and Coasts is (if applied to energy development policy and environmental monitoring this could be interpreted as an opportunity)

- Scale: currently focuses on national level assessment, not necessarily the scale of planned energy developments.
  - Improvements are acknowledged by OMS, (2003) from incorporation of all available social costs to obtain more accurate resource rent values.

- Assumes habitats = ecosystems
  - Some ecosystem service condition indicators (especially cultural services) are not spatially harmonized so can only be interpreted at a broader aggregated level
  - Underpinning national data may be inconsistent or in places absent (e.g. Greece is absent from CORINE Land Cover)
  - Focus on numeric indicators may force the omission or undervaluing of some services
  - No major British involvement!
  - Not clear who will use it tactically
### INTERNATIONAL: APPLICATIONS of ES Frameworks and Classifications (Typologies)

<table>
<thead>
<tr>
<th>Opportunities</th>
<th>Threats</th>
</tr>
</thead>
</table>
| • **Opportunity for direct application to energy policy and specifically marine licensing and environmental monitoring.**  
  • If the data requirements for integration into marine environmental impact assessment and monitoring are identified as a key evidence gap by the TEEB Oceans and Coasts researchers, the resources and approach of TEEB could be applied to energy development IA as part of teh TEEB Oceans and Coasts study.  
  • **In early stages of development** | • **In early stages of development**  
  • Requires application to regional and development scale, to be applicable in impact assessments.  
  • Stock accounts require data from multiple countries presenting potential challenges to detailed assessment.  
  • Estimates only are available of illegal fishing catches and by catch (non-recorded / landed catch)  
  • Data limitations prevent full assessment of true catch and so exact physical stock and economic accounts |
| • Modelling and data resources exist that could be applied to the SEEA approach.  
  • A Bio-economic model to examine the physical and economic accounts for UK Fisheries would provide a useful tool for examining impact of MRE if if could be applied to development and regional scale applications.  
  • Assessment of fish habitat stocks and physical accounts could potentially be aided by existing environmental monitoring data collection.  
  • National scale assessment utilising the SEEAF approach would be applicable to large scale offshore wind and other MRE developments.  
  • Modelled fishing effort displacement from MRE developments could also be included in assessment. | • **In early stages of development**  
  • Requires application to regional and development scale, to be applicable in impact assessments.  
  • Stock accounts require data from multiple countries presenting potential challenges to detailed assessment.  
  • Estimates only are available of illegal fishing catches and by catch (non-recorded / landed catch)  
  • Data limitations prevent full assessment of true catch and so exact physical stock and economic accounts |
| • Case studies – Wales has been attempted, but England, Scotland, Northern Ireland and UK coastal waters should be explored (if they are not being already)  
  • Pilot studies - energy would make an excellent topic  
  • Extend CICES to include more than just biomass and mechanical energy as energy sources under Provisioning.  
  • Aligning projects to benefit from the resource and expertise. | • Changes in underlying data (e.g. CORINE) or structures (CICES) that are revolutionary rather than evolutionary.  
  • Scotland or UK leaves the EU. Even temporary renegotiation of status of any member state may make the information uneven or wholly.  
  • Member states make their own interpretation of the protocols (as happened in CORINE) or there is seen to be a geographic bias in approach (again with CORINE more Mediterranean than northern European).  
  • Alternative systems (maybe with goals other than biodiversity) emerge to deflect effort away from MAES |
### ACADEMIC DEVELOPMENT: ES Frameworks and Classifications (Typologies)

<table>
<thead>
<tr>
<th>Strengths</th>
<th>Fisher et al.</th>
<th>Hattam et al.</th>
</tr>
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<tbody>
<tr>
<td>• The typology (classification) was adapted in international frameworks (TEEB) and national level ES assessment (to examine valuation of the UK’s marine estate as a tool to support sustainable management (Saunders et al 2010)).</td>
<td>• Provided developments which have influenced current ES approaches such as UK NEA.</td>
<td>• Addresses the need to tailor generic classifications (e.g. TEEB 2010, EEA 2013) to a specific site.</td>
</tr>
<tr>
<td>• Explicitly includes abiotic elements, and so allows consideration of all environmental features including energy.</td>
<td>• Provides a clear definition of the point at which monetary valuation should be attempted.</td>
<td>• Provides an approach that suits impact assessment across different energy development sites.</td>
</tr>
<tr>
<td>• Facilitates monetary valuation as the end point is the quantification of benefits (as opposed to underlying processes/services).</td>
<td>• Allows for the utilisation of difference ecosystem service classification systems depending on context.</td>
<td>• Provides a systematic assessment framework to assess the interactions between marine ecosystems and human society which can be developed within impact assessments.</td>
</tr>
<tr>
<td>• Is applicable at any scale.</td>
<td>• Attempts to integrate existing natural resource management concepts such as safe minimum standards.</td>
<td>• Using the classification developed, indicators for the full suite of ecosystem services were derived as well as for associated functions and benefits.</td>
</tr>
<tr>
<td>• Metrics for assessment are not fully defined.</td>
<td>• Remains conceptual.</td>
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**Weaknesses**

- Metrics for assessment are not fully defined.
- Remains conceptual.
- Lack of evidence: Indicators selected will be sensitive to change over time. What change they will respond to, however, is unclear due to the current lack of evidence.
- Many of the indicators identified across ES categories are expected to lack specificity, as signals from climate change become confounded with other sources of change (e.g. fishing).
- There is a recognised absence of data to characterise ecosystem function indicators.
- Ecosystem service assessments require data that help explain the role of ecosystems in delivering ecosystem services, existing data resources are often inappropriate for achieving this.
- Where data resources exist sampling is often sporadic and not necessarily repeated in the same location.
- Greater evidence is required on how the indicators will react to human impact and natural environment changes.
**Opportunities**

• Proposes spatial assessment, and considers changes, and so has the potential to link to accounting frameworks.

• Currently opportunities to apply ecosystem service indicators to marine management are driven by policy goals, particularly given the focus on developing indicators provided by the EU Marine Strategy Framework Directive, the EU Biodiversity Strategy, and the Intergovernmental Platform on Biodiversity and Ecosystem Services.

• Confidence levels or levels of uncertainty for an indicator and associated data need to be established (Müller and Burkhard, 2012). This would aid confidence assessment for the ability for an indicator and associated data to suit impact assessment requirements.

• Scoring procedures could be employed to demonstrate how well indicators are supported by scientific evidence, to assess the quality of the indicators selected and their potential utility to management activities (Kershner et al., 2011).

• Indicators may highlight where to look when change occurs. If the indicator can show that a function or service is changing, the causes of this change and possible management actions can be explored.

• Greater understanding is needed of the components of the ecosystem that are responsible for ecosystem service provision, be they components of populations, species, guilds, food webs or even habitats (Luck et al., 2003; Kremen, 2005).

• Linking indicators to the ecosystem components and the functions those components carry out will improve the scientific basis behind the ecosystem services concept, and strengthen its political relevance and practical application (Seppelt et al., 2011).

• Opportunity for monitoring and evaluation programmes to collect data relevant to identifying which indicators can best describe ecosystem services, functions and benefits.

• Data scarcity for the marine environment results in many indicators being unquantifiable.

• Indicator specificity is a particular problem. Indicators of functions, services and benefits will likely respond to a number of different causes of change.

• Understanding how a specific location contributes to ES provision and the benefits those ES generate, and how they will respond to change remains a challenge.

• All indicators should be assessed in conjunction to obtain a more complete understanding of the implications of ecosystem change. Focusing on just ES or function or benefit indicators may misrepresent a situation and lead to counterproductive management interventions.

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**ACADEMIC DEVELOPMENT: ES Frameworks and Classifications (Typologies)**

Balmford et al.  
Fisher et al.  
Hattam et al.
## Ecosystem service SPATIAL ASSESSMENT TOOLS

<table>
<thead>
<tr>
<th>Strengths</th>
<th>LUCI</th>
<th>InVest</th>
<th>ARIES</th>
<th>SIAT</th>
<th>LEED</th>
</tr>
</thead>
<tbody>
<tr>
<td>States</td>
<td>• Modular framework so other models can be added. Designed to engage stakeholders, stakeholder involvement in development.</td>
<td>• Uses physical parameters developed based on scientific knowledge.</td>
<td>• Expert knowledge based modelling approach, can be based on sparse data and relatively simple models, and therefore can readily give estimates of ecosystem goods and services delivery in most situations.</td>
<td>• Large project with partners across EU and China, Brazil, Argentina and Uruguay.</td>
<td>• Identifies opportunities and threats of relationships between the environment and economy.</td>
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<td></td>
<td>• Values features and actions not just by modified area but by impacted area.</td>
<td>• Initiates discussions between stakeholders, encourages stakeholder participation, many successful examples of International case studies.</td>
<td>• Can also incorporate ecological process models and is capable of coping with differences in input data, units, modelling paradigms and applicable scales between component models.</td>
<td>• Ambitious project, a lot of work went into developing indicators, scenarios, linking up European scale models.</td>
<td>• Working tool provided with expert advice in implementation by Natural England for LEPs and LAs.</td>
</tr>
<tr>
<td></td>
<td>• Users can modify base data, parameters and define questions of interest using GIS toolboxes.</td>
<td>• Reasonable number of process based ES models developed in small scale case studies (i.e. data rich).</td>
<td>• Can be run remotely via web browsers and therefore does not need extensive computing power.</td>
<td>• Spatially explicit.</td>
<td>• LeED provides measures of uncertainty.</td>
</tr>
<tr>
<td>Weaknesses</td>
<td>• Software not currently widely available (version for testing is available), undergoing development, scaling up from small scale studies to national and including more variables.</td>
<td>• Models require parameterisation and in some cases modification for new regions. A global model is under developed but will operate at coarse resolution.</td>
<td>• May be less scientifically robust as based on opinions.</td>
<td>• Large project with partners across EU and China, Brazil, Argentina and Uruguay.</td>
<td>• LeED doesn’t appear to be working version of tool.</td>
</tr>
<tr>
<td></td>
<td>• Dependent upon resolution and availability of data.</td>
<td>• Constrained by resolution and availability of data.</td>
<td>• Every time a new factor needs to be considered (e.g. climate change, planning changes), unless these have been considered in the original expert knowledge elucidation, a follow-up has to be executed.</td>
<td>• Ambitious project, a lot of work went into developing indicators, scenarios, linking up European scale models.</td>
<td>• LeED methodology not completely clear, considers economic factors but uncertain how deals with environmental responses, may rely on expert advice rather than quantitative assessment.</td>
</tr>
<tr>
<td></td>
<td>• Doesn’t currently include all potential ES.</td>
<td>• Reasonable number of models but not all ES, biodiversity poorly covered.</td>
<td>• Complex tool requires expert guidance.</td>
<td>• Spatially explicit.</td>
<td>• LeED doesn’t appear to be working version of tool.</td>
</tr>
</tbody>
</table>
## Ecosystem service SPATIAL ASSESSMENT TOOLS

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<th><strong>SIAT</strong></th>
<th><strong>LEED</strong></th>
</tr>
</thead>
</table>
| **Opportunities** | - Currently being applied in Wales, wouldn't take much more work to extend to rest of UK as using UK scale datasets as spatial layers e.g. LCM, soils  
- Developer temporarily based in CEH Bangor, could liaise to develop particular models  
- InVest team keen to work with partners to apply models in different situations  
- Currently being applied in Wessex BESS project, can liaise for feedback  
- Use of Bayesian statistical methods make it more flexible for use where data is less available, e.g. cultural services  
- Many reports and papers arising from project provide useful information about development of indicators, scenarios and application of tool across Europe.  
- Being trialled in East Anglia  
- May be a practical tool that will be used by LEPs and Las, good to engage with them. | |
| **Threats** | - No one funding stream, developed under number of different projects  
- Software not yet released  
- May need to work with the Aries team to develop and parametrize models  
- Unsure of current funding for finalisation and production of working tool  
- Still in development phase. | |

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Appendix 6: Methods for monetary valuation and economic assessment

Non-market monetary valuation

There are two approaches to the economic valuation of non-market goods: demand-side valuation is based on the concept that value originates with individuals and is measured through its marginal value or price, while supply-side valuation considers cost-of-production and assumes that value originates in the things from which goods and services are made (Straton, 2006). Cost-of-production approaches tend to focus on the flows of matter and energy required to produce an environmental good (Patterson, 1998).

Cost-based approaches are more straightforward to apply as they use existing costs or prices, although they have disadvantages, particularly in that the costs do not necessarily equal the damages that would arise from environmental degradation or the benefits that would accrue from improvements (Markandya et al., 2002). Examples of cost-based approaches include replacement cost, which considers the cost of a manmade alternative to an ecosystem service such as waste-water treatment, and avoidance cost, which evaluates the extent to which the presence of the ecosystem service avoids the need for costly averting behaviours and mitigation (Lui et al., 2010). Avoidance costs provide only the lower bound of value, especially where there is no adequate substitute for the ecosystem service (Daily et al., 2000).

On the demand-side, the premise underlying non-market valuation is that a person may be willing to pay money to secure an environmental improvement, if this exchange of income for the environmental good leaves them indifferent (Hanley et al., 1997; Markandya et al., 2002). Money is used as a measure of value: it represents what the person could have purchased instead of paying to secure the improvement in question (Farber et al., 2002). A measure of monetary value can also be obtained by considering what the person would be willing to accept as compensation for forgoing a benefit or enduring a loss of the non-market good (Markandya et al., 2002).

Markets provide a means of capturing value that works relatively well for private goods which are excludable and rival in consumption (Farber et al., 2002). Most environmental services, as public goods, do not fall into this category. Where there is no explicit market for valuing environmental benefits or where market prices do not adequately capture the social value then more indirect means of assessing value are required.

One technique is to use production function methods. These consider the value of ecosystem services in terms of their contribution to the production of a marketed good or service (Barbier, 2007). A change in the provision of the environmental attribute may influence both the quantity of the marketed good and the production costs, and the implicit price of the environmental input is therefore the change in profit that results from the change in the input level (Markandya et al., 2002). The approach has the advantage of highlighting the essential role of ecosystem functions in the supply of environmental benefits, but it requires a thorough understanding of ecological processes, and this knowledge is typically incomplete (Barbier, 2007).

An alternative valuation method is revealed preference, which involves examining the prices people pay in real markets and relating these to the environmental commodity that influences preferences for the marketed good (Markandya et al., 2002). Hedonic pricing uses the price differential between two products that vary by a single environmental characteristic as a means of observing the monetary trade-offs individuals are prepared to make with respect to that characteristic (Taylor, 2003). Most environmental uses of hedonic pricing relate to house prices (Haab and McConnell, 2002). A second revealed preference technique that has been widely used is the travel cost method. As its name implies, the method considers the cost of travel to a site where the environmental good is provided (and also the frequency of trips) as an expression of the value places on this good, and is commonly utilised in valuing recreational uses of the environment (Parsons, 2003).

Revealed preference techniques are attractive because they are based on actual behaviour, but, as with the other valuation methods, they have drawbacks. Problems in determining the opportunity cost
of travel time, the treatment of substitute sites and visits with multiple purposes have not yet been adequately resolved for the travel cost method (Markandya et al., 2002; Martinez-Espineira and Amoako-Tuffour, 2009). Also, revealed preference methods can only measure use values. Determining non-use values requires stated preference techniques.

Stated preference methods involve directly asking respondents how much they would be willing to pay to increase the level of an environmental good, or how much they would be willing to accept in compensation if the level of the good were to decline. Obtaining this willingness to pay (WTP) or willingness to accept (WTA) requires careful construction of a hypothetical market; the drivers of the environmental change, its extent, timing and implications, the method by which payment would be collected and the timeframe for payment must all be described to respondents in a credible scenario (Boyle, 2003; Bishop et al., 1997). Care is also required because stated preference surveys may return biased values if the survey structure, wording or pool of respondents encourages a particular response (Adamowicz, 1995).

Finally, the Benefit transfer approach transfers information of values assigned to goods and services in one location and/or context to another. This method is often used when it is too expensive or there is too little time to carry out original valuation studies. (Fujiwara and Campbell 2011; Plymouth Marine Laboratory and Marine Biological Association, 2013).

**Microeconomic assessment**

Considering non-market values is essential for adequate assessment of environmental and social impacts. However, changes in energy systems will also have market implications at the microeconomic level.

Microeconomics helps explain the behaviour of households and firms in the market under certain assumptions in order to determine the price and quantity of goods that would produce the maximum total consumers and producers benefits. The bases of microeconomic analyses are the supply and demand curves which capture a number of explanatory variables of behaviour in their shape and slope. Demand curves are generically defined by downward sloping curves which capture: (i) the income effect i.e. when prices drop (rise) consumers can buy more (less) of the good/services; (ii) substitution effect i.e. lower priced goods will be preferred in general; (iii) diminishing marginal utility i.e. the more a good a person has the less they want of it. The slope of the demand curve represents the elasticity of demand which is the ratio of percentage change in quantity demanded given a percentage change in price. Shifts in the demand curve occur when there are changes to the non-price determinants.

Supply curves are predominately upward sloping due to: (i) ability of firms to increase profit by producing more; (ii) higher production incurring higher costs necessitating sale prices to increase; (iii) new entrants into the market due to higher prices. The slope of the supply curve, i.e. the elasticity of supply, is determined by the ability of firms to adapt to demands for new production of a commodity and change in costs to output. Shifts in the supply curve are also non-price determined and include changes in: costs of production, technology, taxes and subsidies, climatic conditions, prices of substitutes, number of producers in the market.

There are a number of tools and approaches used in the analysis of consumer and producer behaviour in the market including optimisation, comparative statistics, uncertainty, auctions, game theory, and asymmetric information. The first two approaches, optimisation and comparative statistics, are discussed below to highlight the usefulness of microeconomics to future energy pathways.

(i) **Optimisation**: used to solve economic decisions made by consumers and producers subject to varying conditions and constraints. This type of analysis considers the demand and supply curves separately, providing greater insight into the trade-offs each agent makes to maximise their utility or profit. Optimisation models have commonly been used in energy systems planning and a review by Zeng et al (2011) highlighted the range of studies and applications use to optimise greenhouse gas emission mitigation in the energy system including: design of energy systems with minimum cost, allocation of renewables among end-users, replacement of fossil-based energy with renewable energy systems, management of power distribution systems. Similarly, household and consumer studies are also used to identify the optimum use and behaviour change in energy demand. Hawasly et al. (2009) compared the individual and community use of electrical appliances within an eco-village highlighting the additional 30% savings that could be achieved if communities exploited their underlying social network.
(ii) Comparative statistics: compares two static equilibria to establish new prices and quantities after an intervention (e.g. taxes). It is assumed in economic analysis that the market is perfectly competitive and the derived price is reflective of economic efficiency. However, externalities, such as pollution, are often excluded from the market price and efforts to internalise these externalities often result in shifts of the demand and supply curves to new equilibria. Including environmental costs in the form of a Pigouvian tax, for example, would shift the supply curve upward and to the left (due to increased production costs) allowing the social optimum and “true” associated price to be supplied in the market. Similarly, internalising positive externalities to ensure a higher quantity of product which is seen to be socially beneficial can be accomplished by, for instance, subsidising the production of energy from renewable systems. This could be justifiable on the basis that society gains from the reduction of carbon dioxide emissions and therefore renewable energy producers should be supported. Subsidies essentially reduce cost of production for suppliers shifting the supply curve downwards and to the right. Other instruments, such as tradable permits are also available for internalising externalities (Parry 2002).

Some of these elements act as building blocks for other assessment tools at different spatial and temporal scales such as the cost-benefit analysis and general equilibrium models detailed below.

**Cost Benefit Analysis (CBA)**

Cost–benefit analysis (CBA) is an economic methodology which can be applied to any decision, policy or intervention producing negative and positive impacts. In a CBA all potential gains and losses from the proposal under scrutiny are identified, converted into monetary units, and compared on the basis of decision rules to assess the proposal’s social desirability (Nas 1996). CBA is designed and has been particularly relevant in relation to public sector investment projects. CBA corrects prices for market distortion when market prices are available, and relies on a number of evaluation methodologies when market prices are not available. CBA has become a very popular tool to assess policies in fields such as health, transportation, urban planning, agriculture, justice, defense, education, and the (OECD 2006). The potential appeal of CBA, like many other quantitative techniques, is that by monetizing the benefits and costs of the policy one can obtain a single metric aggregating many different categories of benefits and costs which can then be used in wider financial or budget analysis.

The theoretical justification for CBA is a potential Pareto improvement, as described in the Kaldor-Hicks rule, which justifies any reallocation of resources in an economy as long as it raises net social benefit (Nas 1996). A reallocation improving social welfare at the expenses of a limited part of society is accepted as a potential Pareto improvement if those who benefit could in theory compensate those who lose. This definition is helpful to appreciate the practical steps of CBA described below and related methodological considerations.

- **Identification of relevant costs and benefits.** This phase delivers a complete list of losses and benefits which should be incorporate in the CBA, identifies (abstractly and anonymously) the groups of individuals affected by the proposal and quantifies physical effects through specific frameworks related to the nature of the impact, e.g. dose-response functions for health-related effects or land-use models for polices affecting allocation of land across competing uses. Historical and economic costs and benefits should be distinguished. Historical costs are not relevant in a CBA, only the economic costs are, i.e. the value which could be produced elsewhere in the economy if the proposal did not go ahead. Similarly, benefits should be distinguished under before-after and with-without approaches. The former refers to the simple assessment of benefits before and after the proposal, while the latter rightly take into account benefit occurring from the project (with) in relation to benefits which would have occurred in a counterfactual scenario (without). Estimation of this counterfactual can be methodologically challenging and politically problematic as different parties may disagree on what the counterfactual might be (OCED 2006). Another distinction which needs to be taken into account is between real output value, i.e. having a clear effect on net social benefits, and pecuniary effect, i.e. a simply reallocation of costs and benefits across members of society but
not altering the welfare of society as a whole. The latter is not significant as far CBAs are concerned.

- **Measurement of costs and benefits.** This is a very demanding task and central to the final aim of CBA, i.e., comparison of costs and benefits. It is important to stress that goods and services entering CBAs which are transacted on the market are not necessarily measured at their market price as market prices are a real measure of scarcity only in the case of competitive market conditions and absence of any other market failure. In all other cases, market prices are adjusted into the so-called shadow prices which essentially measure the economic cost of a particular factor, i.e., the value which could be produced elsewhere in the economy. In the case of goods or services not transacted on the markets, evaluation is based on a number of direct and indirect approaches such as hedonic pricing, travel cost, averting behaviour and contingent evaluation, each of them coming with their particular list of (relative) advantages and disadvantages. It is important to consider that in this phase the costs of regulatory policies, e.g., environmental, have been systematically overestimated in the past before the introduction of the policy (OECD 2006). Although this might be due to several factors, e.g., lobbying power of affected industries, asymmetric information between the regulator and the regulated parties, or genuine underestimation of potential costs savings, it can be argued that this systematic bias should be included in CBA when assessing cost and benefits of potential policies.

- **Comparison of costs and benefits.** In this phase the present value of net benefits needs to be computed and eventually compared to the present value of the investment cost. As computation of present value implies discounting, i.e., using a discount rate to compute today’s value of future costs and benefits, the choice of discount rate is crucial in this phase. In the case of financial analyses, which also use discounting, the choice is relatively straightforward as the discount rate corresponding to the prevailing market interest rate would be used. This differs from social rate of time preferences which should be used in CBAs to reflect the way in which society is willing to trade-off today’s consumption with future consumption. This is particularly relevant in the case of environmental benefits and climate change policies where some of the costs and benefit tend to occur over a very long time horizon with a potentially skewed distribution of costs and benefits across time, i.e., costs occurring relatively early and therefore receiving higher weights compared to the benefits occurring later on. It is worth mentioning that adopting a low discount rate or no discount rate at all may imply excessive sacrifice on today’s welfare to preserve future generations’ welfare. This is both politically challenging and problematic as it paternalistically implies that the current generation knows what future generations may want.

- **Project selection.** This final phase delivers the assessment of the proposal based on a predefined criterion or a number of them such as benefit-costs ratios, the net present value and the internal rate of return. Choice of criteria is somewhat complicated by existing budget constraints although NPV is generally preferred to other criteria (OECD 2006). As one would expect, a proposal is deemed worthy of implementation if the benefits outweigh the costs. Based on the summary indicator produced by CBA, policy-makers are informed of the social desirability of the proposal, more precisely they can gather information on the direction (positive or negative) and the strength of social preferences (depending on the amount of net benefits). CBA also allows the determination of the socially optimal size of a program or project, i.e., the one that maximizes net benefits.

A report commissioned by the Natural Capital Committee has proposed ways to improve guidance for CBA undertaken in the context of projects with impacts on the natural environment (Maddision and Day, 2015). This guidance has not yet been adopted, but was commissioned to feed into the process of refreshing the Green Book.

**Deliberative Monetary Valuation (DMV)**

In this family of methodologies, various deliberative processes are used alongside conventional optimisation analysis, to assign a metric of monetary value to reflect the performance of a range of policy options across a set of relevant issues (Spash 1999). The main purpose is to address some of the acknowledged challenges summarised above in the case of more purely analytical forms of CBA (Aldred 1996).
Depending on how DMV is reported, the main general difference with CBA is that the process of assigning these monetary values can be more transparent in relation to diverse extant social and political perspectives (O’Neill & Spash 2000). And the particular values assigned are much more subject to the agency of those participants who are able to engage in the process. Processes of deliberation can offer a learning experience for those involved. And they open the possibility of subjecting the final results to some kind of sensitivity analysis to reflect the divergent views expressed during the process of deliberation. This is not usually undertaken, but might in principle be reconstructed by a third party in order to reveal some of the concealed ambiguities.

**Macro-economic assessment: Computable General Equilibrium Models**

Computable General Equilibrium models (CGEs) are economic models, with a non-linear specification, which are used as an empirical tool to assess aggregate welfare and distributional implications of policies such as taxes, subsidies, fiscal and monetary policies (Wing 2004). The main features of this approach are encapsulated in the name of the model:

- **Computable**: CGEs are numerical models based on a set of equations with a certain mathematical specification and parameters, sometimes taken from the econometric literature. Mathematical specifications vary according to the functional forms used in the model. The Cobb-Douglas, the Constant Elasticity of Substitution (CES), and the Leontief functions are the most common choices, although the Linear Expenditure System and the Almost Ideal Demand System are sometimes used (Femenia 2012). The value of free parameters, i.e. those not fixed based on econometric estimates from the literature or other sources, is determined as part of a process called calibration, rather than being estimated on the basis of statistical procedures (Böhringer et al 2003). Calibration consists of choosing the parameters so that the model is able to represent the configuration of the economy in a particular year.

- **General**: CGEs are general in the sense that they can represent economic systems with different levels of aggregation, going from the extreme of a single, specific country with very aggregated production sectors, such as services, industry, the public sector and the households sector, to multi-countries models with very disaggregated industrial and household sectors. The level of disaggregation depends on the availability of data, the resources available to develop the model and aim of assessment being implemented. The GTAP8 database, which is one of the options to source data for CGEs, provides data for as many as 129 countries and 57 sectors, giving an idea of the level of detail that one can possibly achieve.

- **Equilibrium**: CGEs build upon general equilibrium theory that combines behavioural assumptions of rationale (and in particular optimising) economic agents with the analysis of equilibrium conditions in the economy (Böhringer et al 2003). A general equilibrium is characterised by a set of price, input and output level so that supply equals demand across all markets of the economy. If the model is perturbed from its original condition of equilibrium, e.g. through a change in the policy or a parameter, prices and quantities vary until they adjust to another equilibrium (Wing 2004). However, this adjustment process occurs through logical time rather than any real temporal dimension. As the model assumes equilibrium in all good, services and production factors, one can only assess the difference between the equilibrium pre-existing the external shock and the one following it (Scricciu 2007).

CGEs are employed by several international organisations, e.g. the EU Commission, the IMF, the World Bank, the OECD and the Inter-American Bank for economic analysis and analysis of environmental policies related to climate change, agriculture, fisheries and forestry (Böhringer and Löschel 2006). The conceptual starting point of CGEs is the circular flow of commodities in the economy which occurs between three types of agents, i.e. the firms producing goods or producers, the consumers demanding the goods or households and the government\(^6\) (Wing 2004). These types of agents interact in a number of markets (Figure A6.1) i.e. the factor market where input for production processes is transacted, the capital market where savings are directed into productive investments and the product market where consumers can purchase the good they need (IDB 2014). In any open economy, economic agents from foreign countries can participate to any of these three markets. Interestingly for environmental applications, especially those related to resource efficiency, the equilibrium in the monetary flows results in the conservation of products being transacted and related

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\(^6\) This division is somewhat simplistic and introduced to help explanation as the difference between consumers and producers is not so clear-cut in the sense that firms produce and consume at the same time, as they need intermediate products to produce their own final output, with the final output of a firm being the intermediate input of another through the logic of input-output tables (see section 3.3.4).
materials and in the conservation of value which reflect accounting principles dictating that expenses must be balanced by income and that any expenses need to go toward the purchase of product (Wing 2004). Utility functions, used in the consumption-side of the economy, link consumption bundles to a certain level of satisfaction in the consumer and allow the derivation of demand function from the hypothesis of maximizing behaviour. A similar logic applies in the production side of the economy.

Core economic CGEs have been extended in a number of ways for environmental policy analysis (Xie and Saltzman 2000):

1. Through fixed pollution coefficients, i.e. computing coefficients describing the pollution per unit of sectoral or intermediate output. In this extension the environment is seen as an end-pipe addition to the core CGE model without any change in the conceptual structure of the model;
2. Through the introduction of environment-related inputs into production and consumption functions. Environmental constraints may affect production through required pollution control costs or consumption if satisfaction of consumers is affected by the amount of pollution;
• Through modelling explicitly the production function of pollution abatement technologies whose development and diffusion can be influenced by specific environmental policies.

Figure A6.1. The conceptual structure of a CGE model. ROW: Rest of the World Source: IADB (2014)

Macro-economic assessment: Input-output (IO) models

Deriving goods and services from services, including final ecosystem services, often requires the input of other capital (for example human, manufactured and financial) which results in economic benefits (UK NEA 2011). These economic benefits can be measured in terms of employment, gross domestic product (GDP) and trade balances of an economy (UK NEAFO 2014: WP2). One approach that can be used to measure these derived macroeconomic benefits is extended monetary input-output models. Monetary input-output tables (MIOTs) were produced in their crudest forms by Quesnay in 1758 and Walras in 1874, but were popularised by Wassily Leontief (1941) when he constructed matrices to undertake research on the US economy. They describe interdependencies of sectors by mapping in monetary terms how the inputs of one sector become outputs that can be used directly for final consumption or as intermediate goods into other sectors’ production processes (Figure A6.2).

Inverse Leontief coefficients are used in the IO models to calculate demand-side multipliers which measure the direct and indirect impacts on sectors, or backward linkages, due to a change in the final
demand of a sector's goods/services. These multipliers can be calculated to estimate the impacts on output, employment and GDP of an economy. Alternatively, supply-side multipliers can also be calculated and derived from a variation in the calculation of the technical coefficients within the IO tables. These supply-side multipliers are based on the work of Gosh (1958) who considered how the occurrence of supply-side constraints, such as resource scarcity, would impact an economy.

The estimation of economic impacts can also be carried out at a global or multi-regional scale. In this case national IO tables are connected to each other by trade flows (i.e. imports and exports) to produce a multi-regional input-output (MRIO) analysis. This allows impacts of the changes in one country to be mapped to changes in other countries (see Lenzen et al. 2012).

Economies are dependent on natural capital and ES to provide material inputs and waste assimilation services of the economic processes. IO tables have been extended to incorporate environmental accounts which help to translate the monetary flows within the IO tables into physical flows. These environmentally extended input-output (EEIO) models have been used to describe predominately; land use (Hubaceck and Sun, 2001), energy use (Papathanasopoulou, 2010), minerals (Konijn et al., 1997), fish (Seafish, 2007) and emissions (Li et al., 2015) associated with economies. EEIO models can be constructed by pre-multiplying standard inverted IO tables with vectors of environmental coefficients which can be constructed from the UK Environmental Accounts (ONS 2014).

The environmental accounts can also be included as extensions to the monetary IO tables by creating additional rows of physical inputs and outputs. This type of IO table is referred to as a Hybrid input-output table and is an intermediary step to producing physical input-output tables (PIOTs) which are IO tables constructed purely on physical flows between sectors. The construction of the Hybrid and PIOT tables require additional data and are therefore less common in IO analysis than MIOTs.

The extension of the environmental accounts to consider ecosystems is still very much in its infancy and the translation of these into ecosystem services have been primarily developed on a conceptual level. This can be seen in the literature on SEEA Experimental Ecosystem Accounting (UN 2012 a, b) and ecosystem classification systems such as the Common International Classification for Ecosystem Services (CICES) (Haines-Young 2012). CICES sets a framework to classify ecosystem services from ecosystems and the materials that can be derived from them. The knowledge and data required to populate these tables are limited (UK NEAFO 2014:5) however development of this area in the future provides the opportunity for macroeconomic tools such as input-output to be integrated with ES.

**Figure A6.2 Environmentally Extended Input-Output (EEIO) Source: UN (2012b:57)**

### Data in monetary terms

<table>
<thead>
<tr>
<th>Industries</th>
<th>Final demand</th>
<th>Total output</th>
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- **Industries:**
  - Total inputs: $q$
  - Value added: $\nu$

- **Final demand:**
  - Final consumption: $\gamma$
  - Gross capital formation: $c$
  - Exports: $f$
  - Imports: $m$

- **Total output:**
  - Total output: $q + m$
  - Exports: $e$

### Data in physical (non-monetary) terms

| Natural inputs/ residuals | |
|---------------------------|--|---|
| $r$                       | |

- **Natural inputs/ residuals:**
  - Natural inputs/ residuals: $r$

**Ref:** Ricardo-AEA/R/ED59332/OP4/Issue Number V1.1
Appendix 7: Health Impact Assessment

Although there is no universal procedure for HIAs, the different components of the methodology are generally accepted and include those listed below and shown in Figure A7.1 (IFC 2009). In the case of HIAs which are part of Environmental Impact Assessments (EIAs), it is not unusual to have the additional tasks of Risk communication and Risk management following the Assessment, in accordance to the procedural structure used in EIAs (Mindell et al 2008).

- Screening: this is a preliminary evaluation to assess whether any significant health issues are likely to be caused by a particular project or policy. It is generally assumed that that projects requiring environmental or social impact assessments are also likely to have potential health impacts;
- Scoping: this is a process for outlining the range and types of hazards and beneficial impacts, including the overall types and categories of questions that should be addressed. Input of stakeholders is particularly important at this stage to ensure that the range of concerns is adequately dealt with;
- Profiling: this provides a comprehensive representation of socio-economic and health-related factors in the population affected by the proposed policy or project. As it provides a baseline against which potential health impacts are measured, this task is also called Baseline data. Profiling provides a context for interpreting the results of the assessment based on manipulation of existing data and collection of primary data through stakeholders’ participation;
- Risk Assessment: This includes a set of activities to investigate, appraise, and (qualitatively or quantitatively) rank the impacts the project is likely to have on the health of the specified communities. The range of potential impacts is determined as well as their relative importance and at what level they are expected to occur;
- Health Action Plan: This considers the rankings developed in the risk assessment and develops a health action plan (HAP) containing proposed actions needed to mitigate identified impacts and promote health. Mitigation is a systematic process to avoid, reduce, remedy, or compensate for the negative impacts. Review and analysis by key stakeholders is critical at this stage;
- Implementation and Monitoring: This details how the mitigation actions will be implemented and monitored, and the related roles and responsibilities. This process ensures that mitigation progress is satisfactory and able to capture unanticipated effects and provide an early-warning system to alert of arising problems. Appropriate key performance indicators should also be defined;
- Evaluation and Verification: This is a system for determining that implementation has been accomplished and is achieving the intended results.
One can implement different types of HIAs, reflecting the severity and risks of health impacts and related levels of efforts and needs. At one end of the spectrum, comprehensive HIAs include all steps shown in Figure 4.1 with stakeholder consultation and communication taking place at each stage and often significant data collection in potentially affected communities. At the other end of the spectrum, rapid appraisal HIAs require less intensive efforts resulting in some cases in only a desktop HIA, i.e. a qualitative review of potential health impacts, and in some others resulting in a limited in-country HIA where field data collection is explicitly excluded. Sometimes these types of HIAs are incorporated as part Social Impact Assessment and Environmental Impact Assessment. Distributional implications are however an integral part of any HIA rather than a separate process, (WHO 1999) and studies are expected to consider both the aggregate and distributional aspects of health impacts. Most HIAs include consideration of inequalities, although to varying extents, and identify vulnerable groups with different degrees of specificity (Parry and Scully 2003).
In terms of disciplinary background, early HIAs tended to use biomedical models of health very much influenced by epidemiological and toxicological considerations and leaning towards quantitative assessments which mainly focused on projects rather than policies. Later HIAs tended to use a more diverse contextualization with a socio-economic or environmental model of health explicitly adopted (Mindell et al 2008). The advantages of this wider approach are related to its inclusive model of health, based on democratic values and community participation (Kemm 2000). In HIA studies, however, practitioners lean towards either model depending on the nature of the problem, availability of resources and importance of the socioeconomic environment. When adopting an explicit quantitative approach, quantitative methods as part of the phase of Risk Assessment can provide a measure of health risks hazards derived from dose–response, exposure–response or concentration–response functions. Comparison across a number of health impacts is allowed by computing their contribution to a summary indicator such as the disability-adjusted life year (DALY). Economic values can be applied to these summary measures to obtain monetary implications of different options in a way reminiscent of Costs-Benefits Analysis (O’Connell and Hurley 2009). While one can see the utility of these quantified and summary estimates, it is important to report the range of individual health impacts contributing to any summary measure to show the range and intensity of impacts and the extent to which they are taken into account by the summary metrics. It is also important to convey as part of the quantitative assessment, information related to the uncertainties associated with the quantities which are used in the assessment.

HIAs have acquired visibility within Strategic Environmental Assessment (SEA) as part of a robust legal and institutional framework. The EU SEA Directive (2001/42/EC) prescribed that certain plans and programmes prepared within the EU require an assessment of the likely significant effects on the environment, including human health while the United Nations Economic Commission for Europe (UNECE) Protocol on SEA, signed in 2003 by 35 European countries, prescribes full consideration of human health aspects. Whilst the SEA Directive is mainly concerned with plans and programmes, the SEA Protocol also aims at policies and legislation.

Recently efforts have been made to address the interaction between health impact assessment and ecosystem services in research that has developed the eDPSEEA (ecosystems-enriched Drivers-Pressure-State-Exposure-Effect-Action) methodology (Reis et al in press). This provides an interdisciplinary theoretical framework and stakeholder engagement tool which aims to join up the “siloed” public health and ecosystem health communities. The model recognizes convergence between the concept of ecosystems services which provides a human health and well-being slant to the value of ecosystems while equally emphasizing the health of the environment, and the growing calls for ‘ecological public health’ as a response to global environmental concerns now suffusing the discourse in public health.
Appendix 8: Social Impact Assessment

The International Association of Impact Analysis (IAIA) 2003 guidelines say that ‘social impact assessment includes the process of analysing, monitoring and managing the unintended social consequences, both positive and negative, of planned interventions (policies, programs, plans and projects) and any social change processes invoked by those interventions. Its primary purpose is to bring about a more sustainable and equitable biophysical and human environment ….’ (IAIA 2003, p3)

Social Impact Assessment (SIA) is generally seen as a complement to, rather than a substitute for, economic assessment frameworks. The IAIA states that ‘the objective of SIA is to ensure that development maximises its benefits and minimises its costs, especially those costs borne by people, including those in other places and in the future’. (IAIA, 2003 ). Some social costs and benefits may not be measurable or quantifiable and are often not adequately taken into account by decision makers, regulatory authorities and developers. By identifying impacts in advance: (1) better decisions can be made about which interventions should proceed and how they should proceed; and (2) mitigation measures can be implemented to minimise the harm and maximise the benefits from a specific planned intervention or related activity (IAIA, 2003).

SIA is best understood as an umbrella or overarching framework that embodies the evaluation of all impacts on humans and on all the ways in which people and communities interact with their socio-cultural, economic and biophysical surroundings. SIA thus has strong links with a wide range of specialist sub-fields involved in the assessment of areas. As such, comprehensive SIA cannot normally be undertaken by a single person, but requires a team approach.

SIA has a role in all the different stages of developing and implementing a policy or project. At the planning stage, SIA can be used to identify interested of affected peoples and facilitates their participation as stakeholders. It documents and analyses the local historical setting and cultural context of the planned intervention collecting baseline data (social profiling) to develop an understanding of local community values, scoping impacts and predicting stakeholder response and allows evaluation and audit of the impact assessment process and the planned intervention itself including cumulative impacts. The method also supports site selection and assists in evaluation of alternatives. . SIA describes potential conflicts between stakeholders and advises on resolution processes. It can be used in undertaking the valuation process and can also be employed to suggest possible mitigation and compensation (monetary and otherwise), and develop and propose coping strategies for impacts that cannot be avoided or mitigated. Finally, SIA can also be used in devising and implementing monitoring and management programmes.

In order to predict and analyse likely impacts social impact assessment specialists construct a matrix of social variables based on generic types of social impact (identified above) which is used to direct their investigation of potentially significant social impacts.

For each project/policy stage, the assessor identifies the magnitude and significance of potential impacts on key identified social variables. Relevant criteria for selecting significant impacts include the

- Probability of the event occurring;
- Number of people including indigenous populations that will be affected;
- Duration of impacts (long-term vs. short-term);
- Value of benefits and costs to impacted groups (intensity of impacts);
- Extent that the impact is reversible or can be mitigated;
- Likelihood of causing subsequent impacts;
- Relevance to present and future policy decisions;
- Uncertainty over possible effects; and
- Presence or absence of controversy over the issue.

The probable social impacts are formulated in terms of predicted conditions without the actions (baseline projection); predicted conditions with the actions; and predicted impacts which can be interpreted as the differences between the future with and without the proposed action. Methods of projecting the future lie at the heart of social assessment, and much of the process of analysis is tied up in this endeavour. Of the numerous methods available, the most commonly used include scenarios,
Social Accounting

Social accounting is a tool for reporting and communicating the social and environmental impacts of an organisation’s activities. A key difference from Cost Benefit Analysis is that it applies to the impact of an organisation as a whole rather than a specific policy or programme. It builds on the growth of environmental reporting and has now been followed by sustainability reporting.

Social Accounting helps an organisation to build on its existing financial monitoring and reporting systems to report on its social performance, and draw up an action plan to improve on that performance and build accountability by engaging with its key stakeholders. It therefore helps an organisation prove its value and improve its performance.

Social accounting involves clarifying what the organisation does, what it is trying to achieve and who it is working for and their expectations. Then, on the basis of this, it collects a mix of quantitative and qualitative information and data on a range of social and environmental issues which relates to its overall objectives (desired outcomes and impacts) and underlying values. This usually lasts one year and runs concurrent with the financial year. At the end of the social accounting year the organisation brings all the information together in the form of social accounts that are independently audited and after revisions the social accounts form a Social Report.

The Global Reporting Initiative (GRI) sets out sustainability reporting guidelines including the Principles and Standard Disclosures that organizations can use to report their economic, environmental, and social performance and impacts. In relation to social impacts the GRI covers labour practices, human rights (mainly relating to labour practices), society - relating to impacts on local communities, public policy and product responsibility.

Companies are currently expected to disclose information related to social accounting but there is no legal requirement to do so and there seems only a remote likelihood of a requirement for a full social accounting becoming law in the foreseeable future (Gray,2005),

Social Accounting is used by business as part of a corporate social responsibility (CSR) and is increasingly being used by NGOs, charities, and government agencies who are interested in quantifying social value. Gray (2005) reports that there has been a steady growth in reporting, especially in the last decade or so. It is a widespread practice in a number of large organisations in the United Kingdom. Royal Dutch Shell, BP, British Telecom, The Co-operative Bank, The Body Shop, and United Utilities all publish independently audited social and sustainability accounts. In many of these cases the reports are produced in (partial or full) compliance with the sustainability reporting guidelines set by the Global Reporting Initiative (GRI). The bulk of the increase in reporting has been of a voluntary nature and has, consequently it seems, been dominated by larger companies in the more obviously “developed” western nations.
Social Return on Investment (SROI)

SROI seeks to inform practical decision making about policies and programmes to make sure that social and environmental impacts are properly taken into account in resource allocation decisions. It was developed from both social accounting and CBA. SROI measures non-market social, environmental and economic costs and benefits and compares them to resources invested. It can be applied at different scales, to consider a single programme or an entire organisation. The method, which has been standardised by the SROI network (http://www.thesroinetwork.org/), seeks to measure change in ways that are relevant to the people or organisations that experience or contribute to it. SROI tells a story of how change is being created by measuring social, environmental and economic outcomes and uses monetary values to represent them. The assessment of value is based in part on the stakeholders’ views of impact.

The process can be carried out internally or can be led by an external researcher. There are two types of SROI:

1. Evaluative; which is conducted retrospectively and based on actual outcomes that have taken place.
2. Forecast; which predicts how much social value will be created if the activities meet their intended outcomes.

Rather than reporting on a range of indicators as in the Social Accounting method discussed above, SROI assigns monetary values to as many social outcomes as possible then adds up the outcome data and compares this to the investment made. This enables a ratio of benefits to costs to be calculated. For example, a ratio of 3:1 indicates that an investment of £1 delivers £3 of social value. However, SROI proponents say it is much more than just a number as it also includes case studies, and qualitative and quantitative data.

Carrying out an SROI analysis involves six stages:

1. Establishing scope and identifying key stakeholders. It is important to have clear boundaries about what the SROI analysis will cover, who will be involved in the process and how.
2. Mapping outcomes – engage stakeholders to develop an impact map, or theory of change, which shows the relationship between inputs, outputs and outcomes. (Guidance from AccountAbility recommends consideration of the views of stakeholders, societal norms, what their peers are doing, financial considerations, and organisational policies and objectives as criteria for judging materiality).
3. Evidencing outcomes and giving them a value – including both objective and subjective indicators This stage involves finding data to show whether outcomes have happened e.g. through interviews, record keeping, focus groups, workshops, questionnaires and then valuing them (e.g through market cost, willingness to pay, revealed preferences, travel cost method).
4. Establishing impact. Having collected evidence on outcomes and monetised them, eliminate those aspects of change that would have happened anyway or are a result of other factors from consideration (deadweight) and ask stakeholders to assess the interventions contribution to change
5. Calculating the SROI. This stage involves adding up all the benefits, subtracting any negatives and comparing the result to the investment using monetary proxies were possible. This is also where the sensitivity of the results can be tested.
6. Reporting, using and embedding. Easily forgotten, this vital last step involves sharing findings with stakeholders and responding to them, embedding good outcomes processes and verification of the report.

The Social Accounting and Audit (SAA) Network and the Social Return on Investment (SROI) Network have agreed on core principles for SROI. These principles are to involve stakeholders, understand what changes, value the things that matter, only include what is material, do not over-claim, be transparent, verify the result. A further tool that walks users through ten steps in developing an SROI analysis was developed in 2008 by Social Evaluator BV in the Netherlands created a tool. (http://www.socialevaluator.eu/)

SROI can be used by any entity to evaluate impact on stakeholders, identify ways to improve performance, and enhance the performance of investments by public or private sector organisations. Some European governments are seeking to make it become the industry norm for measuring social value in the third sector. In the UK the Office of the Third Sector (OTS) funded a three-year programme on measuring social value delivered by the SROI Network, the New Economics Foundation (NEF), Charities Evaluation Services, the National Council for Voluntary Organisations.
Outcome Mapping

“Outcome Mapping,” complements other impact assessments by characterizing and assessing the contributions made by projects, programs, or organizations to the achievement of particular development outcomes. The methodology is applicable during the design stage or during a midterm or ex-post assessment. It focuses on assessing certain type of outcomes rather than impacts. It recognizes the importance of impact as the ultimate goal toward which programs work but believes that (a) the complexity of the development process makes it extremely difficult to assess long term impact and attribution (b) the influence of interventions decreases along the impact chain (c) seeking to attribute impact to specific interventions risks missing the contributions of other actors and factors which are necessary for sustainable development (d) focusing assessment on long-term impacts does not necessarily provide the kind of information and feedback that interventions require to improve their performance. It sees itself as a supplement rather than a substitute for traditional forms of evaluation, which focus on changes in conditions or in the state of well-being and should still be measured. (Earl et al, 2001)

Outcome mapping therefore focuses on outcomes in behaviours, relationships, activities, or actions of the people, groups, and organizations with whom a program directly works. For example, the conventional method of evaluating the results of an energy programme might be to count the number of energy efficiency improvements to a house. But a focus on changes in behaviour acknowledges that energy savings won’t be maximised unless people know how to use the technologies. A program’s outcomes are therefore evaluated in terms of whether those responsible for installing and using the technologies have the appropriate tools, skills, and knowledge to do so. It therefore provides a method for interventions to plan for and assess the capacities that they are helping to build in the people, groups, and organizations who will ultimately be responsible for improving the well-being of people in their communities. (Earl et al, 2001)

Outcome Mapping recognizes that different partners operate within different logic and responsibility systems. It is not based on a cause–effect framework; rather, it recognizes that multiple, nonlinear events lead to change driven by multiple actors and factors. It does not attempt to attribute outcomes to any single intervention or series of interventions. Instead, it looks at the logical links between interventions and behavioural change and monitors and evaluates whether a program has contributed to changes in behaviours in a way that would be logically consistent with supporting development changes in the future. By doing this, Outcome Mapping assumes only that a contribution has been made, and never attempts attribution. In operational terms, this means that instead of attempting to monitor and evaluate all elements of the program with one set of tools, Outcome Mapping defines three distinct but highly interrelated sets of activities and changes, and offers tools to monitor each one. Thus, in addition to monitoring changes in partners, it also monitors the program’s strategies and organizational practices to enhance understanding of how the intervention has contributed to change. (Earl et al, 2001).

An assessment of behavioural outcomes is important for all policies and interventions and is particularly useful for small organisations who may only make relatively small contributions to impact and/or lack the capacity and resources for impact assessments. Although Outcome Mapping may be appropriate in various contexts, it has primarily been tested by development research organizations and programs working in Canada, Africa, Latin America, and Asia working with scientific organizations, government officials, policymakers, and NGOs. (Earl et al, 2001).

Metrics for assessing social impacts

Types of social outcome and impacts

Appraisal and evaluation techniques usually distinguish between outputs, outcomes and impacts which may be intended or unintended:

- Outputs are the direct products of an intervention
- Outcomes are the changes that come about because of the intervention: e.g. changes in energy related behaviour.
Impacts refer to the ultimate changes on people and the environment that an intervention delivers over time.

Outputs, outcomes and impacts are often presented in a logical impact chain (as in log frames) with activities leading to outputs leading to outcomes and then impacts, the relationship between them is not necessarily linear. For example in relation to energy interventions it was previously assumed that information would change attitudes which in turn would change energy related behaviours and hence reduce carbon emissions. In fact there are many factors that may intervene to prevent behaviours changing, there may be interactions and feedback loops between elements of the chain, and the direction of change may be reversed (EVALOC, 2012).

Moreover, social outcomes or impacts may be:

- Direct - as a direct result of the project;
- Indirect - i.e. secondary or ‘knock on’ effects of a project such as economic multiplier effects on the local and national economy due to increased spending from job creation;
- Cumulative impacts – impacts from the project that accumulate over time and space.

Various studies (TEP 2010, IAIA 2003) have looked at standardising the list of social outcomes and impacts to be used in social impact assessments but lists continue to vary according to the degree of detail. The social impact list referred to in the Department for Environment, Food and Rural Affairs (DEFRA, 2011) documents (Maxwell et al. 2011) is that of the IAIA (2003) which identifies social impacts as changes occurring in one of the following:

- People’s way of life – how people live, work, play and interact with one another on a day-to-day basis.
- Their culture – their shared beliefs, customs, values and language or dialect
- Their community – its cohesion, stability, character, services and facilities
- Their political systems – the extent to which people are able to participate in decisions that affect their lives, the level of democratisation that is taking place, and the resources provided for this purpose
- Their environment – the quality of the air and water people use; the availability and quality of the food they eat; the level of hazard or risk, dust and noise they are exposed to; the adequacy of sanitation, their physical safety, and their access to and control over resources
- Their health and wellbeing – health is a state of complete physical, mental, social and spiritual wellbeing and not merely the absence of disease or infirmity
- Their personal and property rights – particularly whether people are economically affected, or experience personal disadvantage which may include a violation of their civil liberties
- Their fears and aspirations – their perceptions about their safety, their fears about the future of their community, and their aspirations for their future and the future of their children.

In contrast, the TEP study (2010) for the European Commission suggests that the majority of social impacts can be summarised under a relatively limited list of impact types, namely:

1. Employment (including labour market standards and rights)
2. Income
3. Access to services (including education, social services, etc.)
4. Respect for fundamental rights (including equality)
5. Public health and safety

A metric is a system or standard of measurement, and an indicator is a specific piece of information that helps track changes in outputs, outcomes and impacts relating to an intervention, although it is now widely accepted that it is more important to measure outcomes and impacts than outputs. Value is considered as the overall benefit an individual or community gains from an activity both in monetary and non-monetary terms. (Plymouth Marine Laboratory and Marine Biological Association, 2013)

Metrics, indicators and value may be:

- Quantitative – expressed in monetary or numerical terms - mainly useful for tracking what has changed e.g. the changes associated with an intervention. Cost Benefit Analysis, for example, attempts to combine the value of a number of social outcomes/impacts as a single number;
- Qualitative – expressed through narrative description and analysis - which is useful for assessing outcomes that cannot be expressed in monetary or quantitative terms,
understanding why and how changes associated with an intervention have occurred and how people experience them, the detailed social impacts of particular policies, and their implications for wellbeing, for different groups of people and in different places.

- Combined methods - expressed through a mix of quantitative and qualitative data

Direct impacts are relatively straightforward to identify and measure, but the assessment of indirect and cumulative impacts is more complex and the determination of magnitude (size and extent of the impact) and significance (the importance for decision making) is difficult. Social impacts are often the most difficult to predict, due to the lack of a clear cause-effect relationship when working with human responses to change, meaningful baselines, etc.

Measuring and valuing social impacts is challenging due to the range of impacts, the fact that some are qualitative and do not have market or monetary value, and the “lack of appropriate methods, tools and/or data sources to assess social impacts” (TEP 2010). Nevertheless, a number of techniques, monetary and non-monetary, for measuring social impacts have been developed. In Appendix 8 these methods are described, their strengths and limitations assessed and an outline is provided of some combined social, environmental and economic indices.

Non-monetary Metrics for assessing social impacts

The strength of qualitative non-monetary approaches are well known. They can take into account people’s subjective views and experiences, capture aspects of wellbeing that are not taken into account by monetary methods and hence can provide unique insights into the potential impacts of future policies and interventions. The participatory and in depth nature of other approaches can help increase public understanding of new policy or interventions, as well as help researchers understand key concepts, improving the relevance and quality of research findings and empowering participants. They can be used to help improve the quality of survey-based quantitative evaluations by helping generate evaluation hypothesis; strengthening the design of survey questionnaires and expanding or clarifying quantitative evaluation findings. Some aspects of wellbeing – such as life satisfaction measures- can be presented as a single measure (OECD 2014; Maxwell et al 2011).

However, as noted above if monetary valuation is not applied to social or environmental impacts the implicit value that is attributed to them by government decision makers will often be zero. Moreover, qualitative research can be dismissed as subjective and extrapolation is limited because of the small and unrepresentative nature of their samples. Moreover, qualitative methods do not necessarily provide systematic ways of comparing and valuing outcomes/impacts across space and time, or the trade-offs between them, which is needed to assess potential government policy and interventions.

I. Participatory approaches

The Green Book emphasises the importance of involving stakeholders in appraisal (HMT, 2013) and points out that “the weight to give to factors that are thought to be important by key players cannot be decided by "experts", and should incorporate the judgments of stakeholders (HMT, 2013).

A number of participatory methods that can be used for assessing and valuing social impacts (Plymouth Marine Laboratory and Marine Biological Association, 2013), these include: participatory modelling (Videira et al. 2009), scenario workshops (Peterson et al. 2003, Caille et al. 2007), deliberative visioning (Kallis 2009), citizens’ jury (Aldred and Jacobs 2000). (These are discussed further in Section 5.

There are a number of advantages of participatory approaches: (a) there may be dimensions of collective social impacts and societal wellbeing that are not adequately reflected in the sum of individual impacts (b) including a wide range of perspectives and views, arguably means that decisions will be more informed (c) participatory decision-making processes can help stimulate wider civic engagement, offer participants a voice in the political process and help improve trust which are themselves important constituents of well-being (d) civic engagement, political voice, trust in institutions and government are themselves important constituents of wellbeing and participatory appraisal can also promote learning by policy makers (Maxwell et al, 2011).

II. Combined quantitative and qualitative metrics

Subjective well being

Some assessment methods such as valuation workshops (Álvarez-Farizo et al. 2009) or participatory multi criteria evaluation (Salgado et al. 2009) – use a mix of participatory and quantitative assessment methods (Plymouth Marine Laboratory and Marine Biological Association, 2013). These are assessed elsewhere. A relatively recent approach which is gaining currency with government to value...
non market social impacts, combines both qualitative and quantitative (non-monetary) approaches through the recent development in economics (and psychology) of subjective wellbeing research.

Subjective wellbeing approaches seeks to understand how respondents value, experience or interpret changes. This is done through self-reported (and hence subjective) surveys which ask people to rate their (a) life evaluation or satisfaction – a reflective assessment on a person’s life or some specific aspect of it (b) affect – a person’s feelings or emotional states such as anger, worry, happiness typically measured with reference to a particular point in time and (c) eudaimonic wellbeing - a sense of meaning and purposing in life, or good psychological functioning relating to feelings of competence, autonomy, meaning and purpose and (d) experience wellbeing. (OECD, 2014).

An application of subjective well-being is the Life Satisfaction Approach (LSA). This uses people’s (non-monetary) reported life satisfaction, measured in surveys such as the ONS’s Integrated Household Survey which began including questions on respondents' subjective well-being in April 2011. It then uses ‘objective’ econometric methods to estimate the life satisfaction provided by respondents’ access to non-market (including social) goods and services. The latter is then converted into a monetary figure by estimating the amount of income that would be required to hold life satisfaction constant following the gain or loss of the good or service. The approach therefore assesses the impact of policies on how people think and feel about their lives as a whole, instead of just assessing impact based on what people say they want and what they choose. (HM Treasury 2011a; Plymouth Marine Laboratory and Marine Biological Association, 2013). It can be used to value a wide range of different public goods and bads, negative and positive externalities and has been used to value climatic conditions, airport noise, urban regeneration schemes, droughts etc. (Frey, B et al 2009)

**Strengths and weaknesses of subjective well being**

The UK Government increasingly recognised the importance of subjective well-being assessments: ‘social welfare ultimately rests on individuals’ subjective assessment of their own wellbeing and therefore subjective measures of wellbeing measurement should complement ‘objective’ served impacts’ (HM Treasury 2013). As the OECD 2014 guidelines say a nation of materially wealthy and healthy but miserable citizens is not the kind of place most people would want to live (OECD, 2014, p4).

The OECD (2014) states that subjective wellbeing can be used to support CBA and policy evaluation particularly when non-market outcomes are involved, to better understand the drivers of wellbeing and to help in identifying potential policy problems (OECD, 2014). Large scale subjective well-being data can provide an alternative way of deciding trade-offs between different policy options or interventions than listening to the arguments of relatively narrow interest and lobby groups. It can also offer a standard unit of comparison or metric between departments. It is relatively cost and time effective to collect and can draw on very large and representative samples, it can be applied to a whole variety of life events and circumstance, has fewer biases and less strategic behaviour on the part of respondents, and the data does not rest on assumptions about market structure. (OECD 2014)

It can also play an important role in challenging decision makers to think more carefully about the full range of impacts of their proposed policies and help them to question the values that they may otherwise place implicitly on these impacts. Subjective measurement may also give a better idea of the relative value of non-market goods, even if absolute values cannot yet be placed alongside market goods. All of this will help policy makers better ground their decisions in evidence (HM Treasury 2013). It can also be used to help value costs and benefits accruing to people with different incomes i.e. where people on higher incomes are willing to pay more for a non-market good or service than those on low incomes and which thus distorts the data (Layard et al, 2008 – see OECD 2014, p230) The OECD 2014 guidelines say that while this assumes that subjective well-being can be used as a measure of utility, which not everyone will agree with, it nonetheless provides a way of managing the impact of the marginal utility of income in an evidence based manner.

However, the Treasury report (2013) also states that while subjective wellbeing is recognised for the first time in the government Green Book discussion paper: inevitably, its methodology is still evolving.

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7 Studies of subjective wellbeing also include the collection of data on co-variates (that might predict the subjective wellbeing) including demographics, material conditions (income, expenditure and consumption, deprivation, housing quality), quality of life (employment status, health status, work-life balance, education and skills, social connections, civic engagement and governance, environmental quality, personal security) psychological measures (personality types, aspirations and expectations) (OECD, 2014)
and existing valuations are probably not robust enough yet for use in Social Cost Benefit Analysis. The Treasury report discusses the methodological challenges involved in life satisfaction surveys in detail. This includes:

1. difficulties in remembering past experiences;
2. context effects- people tend to base their judgements on information that is most accessible at the time (Treasure);
3. reporting of life satisfaction. – Individuals may adjust their life satisfaction scores to give more socially desirable responses.

Thus while measures of subjective wellbeing have become more accepted by government, they are not currently collected in a systematic and consistent way across OECD national statistical agencies. The two largest datasets containing comparable measures of subjective well-being are the Gallup World Poll and the World Values Survey. As well as a European Social Survey, there is also a British Household Panel Study that includes information on subjective wellbeing which has recently been integrated into the UK Household Longitudinal Study also known as ‘Understanding Society’ (OECD, 2014).

In relation to LSA specifically, the Treasury (2013) describes the Life Satisfaction approach as a 'potentially exciting development' since social welfare ultimately rests on individuals’ subjective assessment of their own wellbeing, and therefore subjective wellbeing measurement may soon provide a complement to the more traditional economic approaches. A strength of the LSA cited by the Treasury (2013) is that it correlates the degree of public goods or public bads with individuals’ reported subjective well-being and evaluates them directly in terms of life satisfaction, as well as relative to the effect of income on life satisfaction. This approach obviates some of the major difficulties inherent in both the revealed preference and stated preference methods. It is not based on observed behaviour, the underlying assumptions are less restrictive and non-use values can, to a certain extent, be measured. Moreover, individuals are not asked to value the social good directly, but to evaluate their general life satisfaction, which is considered a cognitively less demanding task, does not evoke answers considered desirable by the persons asked, and there is no reason to expect strategic behaviour (Treasury 2013).

III. Combined social, environmental and economic indices

In contrast to monetary valuation, which seeks to express net impacts of policies or interventions as a single number, there are also various attempts to capture the diverse (a) social impact indicators and (b) social, economic and environmental indicators into composite indices. These indices may include combinations of monetary and non-monetary indicators.

Indices including a composite of social indicators include the English index of Multiple Deprivation which is an overall measure of multiple deprivations and the English Government’s Social Justice Indicators. Indices including a mix of social, environmental and economic indicators are the English Sustainable Development Indicators and various Well Being Indices. These are outlined below. It is interesting that many of the metrics are quantitative – represented as percentages, rates or numbers - rather than monetary.

Composite measures of subjective wellbeing are difficult to express as one number and there is not yet a clear basis for determining the relative weights to assign to different dimensions of wellbeing. The OECD guidelines (2014) say that until further consideration has been given to how composites indexes could be developed the most sensible approach is to provide disaggregated measures so users can select those they need.

**English Index of Multiple Deprivations**

The Index of Multiple Deprivation (IMD) is an overall measure of multiple deprivation experienced by people living in an area and is calculated for every Lower layer Super Output Area (LSOA) in England (Communities and Local Government (2010 and 2011). It widely used in the UK as a socio-economic status impact indicator which can be used to track changes in communities over time.

The index uses 38 separate social impact indicators organised across seven distinct domains of deprivation (see below) which are combined, using appropriate weights, to calculate the area’s IMD.

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8 LSOA are geographical areas comprising 1000-3000 people, and 400-1200 households. LSOAs are nested within a hierarchy of local and regional geographic units, which include the European Union NUTS (Nomenclature of Territorial Units for Statistics) regions. NUTS levels 2 and 3 equate to UK Counties and Districts respectively. The IMD 2010 can be used to rank every LSOA in England according to their relative level of deprivation.
Domains include income deprivation, employment deprivation, health and disability deprivation, education, skills and training deprivation, barriers to housing and services deprivation, crime, and living environment deprivation. The detailed indicators are outlined in Table A8.1

Where possible the indicators are combined to produce a summary measure for each domain which is straightforward to interpret and expressed in meaningful units (e.g. proportion of people or of households experiencing that form of deprivation). Where this is not possible the indicators within the domain are standardised by ranking and transforming to a normal distribution. Factor analysis is used to generate the weights to combine the indicators into a final domain score. The domain weights are informed by theoretical considerations and responses to the consultation processes.

Table A8.1 Detailed indicators of social deprivation used in the Index of Multiple Deprivation (IMD)

<table>
<thead>
<tr>
<th>Deprivation domain</th>
<th>Indicators</th>
<th>Domain weight</th>
</tr>
</thead>
</table>
| Income deprivation          | • Adults and children in Income Support families  
• Adults and children in Income-Based Jobseeker's Allowance families  
• Adults and children in Pension Credit (Guarantee) families  
• Adults and children in Child Tax Credit families (who are not in receipt of Income Support, Income-Based Jobseeker’s Allowance or Pension Credit) whose equivalised income (excluding housing benefits) is below 60 per cent of the median before housing costs  
• Asylum seekers in England in receipt of subsistence support, accommodation support, or both | 22.5 %        |
| Employment deprivation      | • Claimants of Jobseeker’s Allowance (both Contributory and Income-Based) women aged 18-59 and men aged 18-64, averaged over 4 quarters  
• Claimants of Incapacity Benefit women aged 18-59 and men aged 18-64, averaged over 4 quarters  
• Claimants of Severe Disablement Allowance women aged 18-59 and men aged 18-64, averaged over 4 quarters  
• Claimants of Employment Support Allowance women aged 18-59 and men aged 18-644  
• Participants in New Deal for the 18-24s who are not in receipt of Jobseeker’s Allowance, averaged over 4 quarters  
• Participants in New Deal for 25+ who are not in receipt of Jobseeker’s Allowance, averaged over 4 quarters  
• Participants in New Deal for Lone Parents (after initial interview) aged over 18, averaged over 4 quarters. | 22.5%         |
| Health Deprivation and Disability deprivation | • Years of Potential Life Lost – an age and sex standardised measure of premature death  
• Comparative Illness and Disability Ratio – an age and sex standardised measure of morbidity and disability  
• Measures of acute morbidity – an age and sex standardised rate of emergency admissions to hospital  
• Proportion of adults under 60 suffering from mood or anxiety disorders – a modelled indicator for the proportion of adults suffering from mood and anxiety disorders. | 13.5%         |
### Deprivation domain: Education, Skills and Training deprivation

- Average points score of pupils taking English, Maths and Science Key Stage 2 exams
- Average points score of pupils taking English, Maths and Science Key Stage 3 exams
- Average capped points score of pupils taking Key Stage 4 (GCSE or equivalent) exams
- Proportion of young people not staying on in school or non-advanced education above age 16
- Secondary school absence rate – the proportion of authorised and unauthorised absences from secondary school
- Proportion of those aged under 21 not entering Higher Education.

### Deprivation domain: Barriers to Housing and Services deprivation

- Household overcrowding – the proportion of households within an LSOA which are judged to have insufficient space to meet the household’s needs
- Homelessness – the rate of acceptances for housing assistance under the homelessness provisions of the 1996 Housing Act (at local authority district level)
- Difficulty of access to owner-occupation (local authority district level) – proportion of households aged under 35 whose income means they are unable to afford to enter owner occupation.

### Deprivation domain: Crime

- Violence – number of reported violent crimes (19 reported crime types) per 1000 at risk population
- Burglary – number of reported burglaries (4 reported crime types) per 1000 at risk population
- Theft – number of reported thefts (5 reported crime types) per 1000 at risk population
- Criminal damage – number of reported crimes (11 reported crime types) per 1000 at risk population.

### Deprivation domain: Living Environment deprivation

- Social and private housing in poor condition
- Houses without central heating.

### Sources

Communities and Local Government (2011) The English Indices of Deprivation 2010

Communities and Local Government (2010), The English Indices of Deprivation 2010 Technical report

### Social Justice Indicators

The English government’s ‘Social Justice Framework’ is another example of an attempt to combine a range of social impacts into a single tool (HM Government, 2012). Here, the broad goal of social justice has been broken down into a number of themes and for each theme a small number of different indicators have been selected. The purpose of this is to inform politicians and civil servants about the state of social justice in England from year to year and whether or not their policies are helping certain goals to be met.
The government sets out very clearly how it expects to use this framework: "The Framework is divided into five areas that represent the five themes in [the report] 'Social Justice: transforming lives':

1. Supporting families
2. Keeping young people on track
3. The importance of work
4. Supporting the most disadvantaged adults

For each of these themes, we will pick one or two indicators of progress. These indicators are specific to Social Justice and represent the Government’s priority in each of these areas. However, the indicators cannot, and are not designed to tell the whole story. These five areas are inherently complex; there are many interrelated factors that contribute to, for example, family stability or long term worklessness. We will, therefore, continue to use information on a wide range of other contributory factors when developing Social Justice policy and when we report on progress..........

English Sustainable development indicators– integrated social, economic and environmental indicators

Sustainable development indicators are used at a range of levels: global, national, local, and corporate. They consist of collections of indicators arranged around key themes, the selection of which will depend on the nature and role of the organisation.

In February 2011 the government published its strategy for mainstreaming sustainability and committed to publishing a revised set of Sustainable Development Indicators (SDIs). This set was to replace the previous SDIs which had been maintained by Defra on behalf of government since 2001. The previous SDI set consisted of 68 indicators comprising 126 measures. The new set is formed of fewer indicators: 12 headline and 23 supplementary indicators, comprising 25 and 41 measures respectively. This follows the example of other international institutions in identifying a core set of headline indicators to highlight sustainable development priorities for users and government. It was also in part prompted by the need for alignment with the Office for National Statistics’ (ONS) development of national wellbeing measures, which are closely related to measures of sustainable development. Where appropriate the measures used in the indicator set also align with other indicator frameworks, such as those which measure progress against government departments’ business plans and the Department of Health’s Public Health Outcomes Framework.

Data is compared with data from 1990 or from the earliest year for which data is widely available after 1990. Some of the indicators are presented as an index, where a baseline year (for example 1990) is set as 100 and subsequent years are shown in relation to that value. This can mean that trends for measures with different units can be more easily compared. Where possible the indicators have been presented for England.

Each measure is assessed using a set of “traffic lights”. They do not show whether the measure has reached any published or implied targets; rather, they show whether changes in the trends are showing clear improvement or deterioration. The traffic lights are determined by identifying a period over which to assess change and comparing the value of the measure in the base or start year with the value in the end year. Where data are available, two assessment periods have been used:

1. Long-term – an assessment of change since the earliest date for which data are available (usually back to 1990). If the earliest data available is for after 2000, no long term assessment is made.
2. Short-term – an assessment of change for the latest five year period.

The traffic light assessments are as follows:

- Improving
- Little or no overall change
- Deteriorating
- Not yet assessed due to insufficient data

Where possible the traffic light assessment has been made by evaluating trends using statistical analysis techniques. The assessment may be made by Defra statisticians in collaboration with the data providers or undertaken by the data providers themselves. A green or red traffic light is only applied when there is sufficient confidence that the change is statistically significant and not simply a product of random fluctuations. The indicators (Table A8.2) are mainly for social impacts but have not
yet been assessed due to insufficient data. There is also a set of related supplementary measures specifically relating to energy supply and demand (Table A8.3.).

Table A8.2 English Sustainable development indicators

<table>
<thead>
<tr>
<th>Economy</th>
<th>GDP</th>
</tr>
</thead>
<tbody>
<tr>
<td>Economic prosperity</td>
<td>GDP</td>
</tr>
<tr>
<td></td>
<td>GDP per head</td>
</tr>
<tr>
<td></td>
<td>Median income</td>
</tr>
<tr>
<td>Long term unemployed</td>
<td>Proportion of adults unemployed over 12 months</td>
</tr>
<tr>
<td>Poverty</td>
<td>Proportion of children in relative low income households (before housing costs)</td>
</tr>
<tr>
<td></td>
<td>Proportion of children in absolute low income households (before housing costs)</td>
</tr>
<tr>
<td>Knowledge and skills *</td>
<td>Human capital (£) stock</td>
</tr>
<tr>
<td></td>
<td>Human capital per head</td>
</tr>
</tbody>
</table>

| Society                       | At birth: males              |
|                               | At birth: females            |
| Healthy life expectancy       |                              |
| Social capital *              | Proportion of people         |
|                               | - engaging in actions addressing issues of public concern |
|                               | - who have a spouse, family member or friend to rely on if they have a serious problem |
|                               | - engaging in any volunteering activity |
|                               | - agreeing that people in their neighbourhood can be trusted |
| Social mobility in adulthood  | Proportion of adults from less advantaged groups in managerial or professional positions |
| Housing provision             | Net additional dwellings     |

| Environment                   | UK GHG emissions             |
|                               | GHG emissions associated with UK consumption |
| Nature resource use           | Raw material consumption of non-construction materials |
|                               | Raw material consumption of construction materials |
| Wildlife:bird population indices | Farmland birds               |
|                               | Woodland birds               |
|                               | Sea birds                    |
|                               | Water and wetland birds      |
| Water use                     | Abstractions from non-tidal surface waters and ground waters |

Table A8.3 Supplementary measures for English Sustainable development indicators specifically relating to energy supply and demand

| Environment                   | Energy supply                |
|                               | Transport                    |
|                               | Business                     |
|                               | Residential                  |
|                               | Other                        |
| UK CO2 emissions by sector    | Propotion of gross energy consumption from renewable sources |
| Energy consumed in the UK from renewable sources | Mean SAP ratings of |
| Housing energy efficiency     | Existing housing             |
|                               | New housing                  |
Investigation of methods for comparing impacts and prototype framework for comparison

Source: DEFRA, 2013, New Sustainable Development Indicators, Gov.UK

Well Being Indices—integrated social, economic and environmental indicators

As discussed above, well-being indices are a relatively recent development that seeks to capture a range of social, economic and environmental impacts that go beyond people's income and material conditions. Wellbeing is a broad concept which requires that basic needs are met, that individuals have a sense of purpose, that they feel able to achieve important personal goals and participate in society. It is enhanced by conditions that include supportive personal relationships, strong and inclusive communities, good health, financial and personal security, rewarding employment, and a healthy and attractive environment. As numerous factors influence individual wellbeing it cannot be fully measured by a single indicator (Plymouth Marine Laboratory and Marine Biological Association 2013).

There are a number of different approaches to measuring well-being which may combine subjective and objective measures of wellbeing. Max Neef’s Human-Scale Development Matrix (H-SDM) is used in the UK National Ecosystem Assessment, The 3D concept of social wellbeing was applied by Britton et al. (2013) and the Human Well-being Index was noted in the Millennium Ecosystem Assessment (2005). Each of these approaches includes different weightings, ranges of groupings or domains and focuses on a variety of sociocultural and economic variables.

The UK Office for National Statistics has recently carried a national survey to measure well-being which was published in 2012 will be updated annually (ONS 2012). It includes both objective and subjective measures of well-being. There are 42 measures of national well-being in the ONS survey split across 10 domains - personal well-being, our relationships, health, what we do, where we live, personal finance, the economy, education and skills, governance. The social well-being measures are still being developed but the ‘objective’ measures include unemployment rate or number of crimes against the person as well as subjective data about people’s satisfaction about their lives (ONS 2014). The subjective measures for personal well-being include for examples self-ratings of satisfaction with their lives overall, how worthwhile the things they do are, their happiness yesterday, and their anxiety yesterday.

The OECD’s Better Life Initiative was launched in 2011 and aims to measure society’s progress across eleven domains of well-being including among others income, jobs, health, skills and housing, civil engagement and the environment. The measure of subjective wellbeing, i.e. how people think and about and experience their lives, is an important component of the framework (OECD 2013)

Max Neef’s H-SDM lists a number of fundamental needs such as: subsistence, protection, affection, understanding, participation, recreation, creation, identity and freedom which can occur in different types of existential settings such as: being (qualities), having (things), doing (actions) and interacting (settings). The table of fundamental needs running down the rows and the settings running across the columns makes a 36 cell matrix which is filled with different types of satisfiers to achieve different fundamental needs.

The 3D concept of Britton et al. (2013) considers three dimensions which contribute to overall well-being and include the: (i) material dimension which considers what people possess for example food, income, shelter, assets, employment and how it contributes to material wellbeing; (ii) relational dimension which includes interactions with others, relationships, institutions, rules and norms, access to resources for example and how it contributes to relational wellbeing; and (iii) cognitive or subjective dimension which considers the person’s own perception of how they see their lives and the values and beliefs that people hold in order to shape these perceptions.

The Human Well-being Index of the Millennium Ecosystem Assessment identifies the following social indicators to be considered within human well-being assessments:

- Security: personal safety, secure resource access, security from disasters
- Basic material for good life: adequate livelihoods, sufficient nutritious food, shelter, access to goods
- Health: strength, feeling well, access to clean air and water
- Good social relations: social cohesion, mutual respect, ability to help others
- Freedom of choice and action: opportunity to be able to achieve what an individual values doing and being.
Appendix 9: Summary of individual social appraisal methods

Staged Multi-criteria Analysis (MCA)

The label used above for this family of methods is the one chosen by Stagl in a useful review for DEFRA (Stagl 2007). This term is applied here, as it was by her, to address a diverse array of multi-criteria techniques (Treasury et al. 1996; Clemen 1991) – also reviewed elsewhere in detail for DEFRA (Dodgson et al. 2003). There exist many divisions within this field, some of which lead to methods that may derive contrasting findings when applied to the same kinds of policy challenge. But a common feature of all these methods, is a move away from the apparently unambiguous (but as we have seen, potentially highly misleading) metric of monetary value. The metrics used instead are more abstract measures of relative value, as variously produced by each method.

This said, all these methods share a basic overall purpose, which is to further address the challenges of incommensurability described in relation to CBA in Appendix 6. This is done by affording variously more sophisticated ways to explore the implications of divergent priorities and values – and sometimes uncertainties – across contrasting social perspectives. The ‘three stages’ sometimes referred to in the labelling of this methodology, is simply one means by which this can be achieved – as set out in a particularly relevant approach developed and widely applied in Germany (Renn et al. 1993; Renn & Webler 1998), including to energy issues.

In this form, MCA uses a ‘co-operative discourse’ approach, in which key uncertainties and ambiguities in appraisal are addressed in distinct ways at different stages of analysis. In short, the evaluation criteria are selected in advance by major stakeholders. The scoring of benefits and impacts under these criteria is undertaken entirely by experts. Here, there is some attention to uncertainties. But these are generally treated as if they were relatively tractable ‘risks’ (i.e.: amenable to the assigning of probabilities) (Stirling 1999). So – as is typically the case in CBA and deliberative monetary valuation (DMV) – uncertainties of a more challenging kind are correspondingly excluded from analysis (Stirling 2003).

Crucial to this kind of approach, is that the input of citizens is restricted to the exploration of alternative values. So, there is no opportunity for citizens to question the scope or structuring of issues as determined by major stakeholders. And the scoring of benefits and impacts by experts remains similarly inaccessible to interrogation. Since expression of uncertainties is also somewhat reduced (as discussed above), this leaves MCA to be rather circumscribed in its ability to explore a full range of alternative perspectives and possibilities (Stirling 2006).

In her earlier study for DEFRA, Stagl found that three-stage MCA is most appropriately used when the impacts of a policy, programme or project are reasonably well understood by experts but where there is a significant technical component (Stagl 2007). But this kind of application leaves unaddressed the issues of ambiguity and uncertainty mentioned above.

Social Multi-criteria Evaluation (SME)

Social multi-criteria evaluation differs from MCA in a number of ways that are important in methodological terms, but often less so in respect of the practical implications for policy and wider political debate. It was mentioned in discussing MCA (above), that the field of multi-criteria analysis is divided between many divergent approaches. Arguably the single most important such division is between a ‘Francophone’ tradition involving a procedure for pairwise comparison between options as compared under each criterion and an ‘Anglophone’ tradition more directly based on conventional utility theory and neoclassical ideas of rational choice (Salo et al. 2003).

Like MCA – and unlike CBA and DMV – SME makes use of an abstract value metric (Munda 2004). But the purpose in this case is to address more specifically than do any of these techniques, the challenges of complexity, ambiguity and uncertainty. SME does this by combining participatory techniques with a pairwise comparison approach to multi-criteria analysis. This affords greater agency
to participants of all kinds (including citizens) to frame the ways in which different policy, programme or project options are taken into account – and how conflicting interests are handled (Munda & Russi 2008).

A particular focus of SME lies in the provision of transparency – both to participants and third parties. It is intended that this help foster ‘social learning’ – so that the appraisal exercise itself is not simply about the outputs that are produced, but also about the process in which the different parties are engaged (Munda 2004).

In her earlier analysis for DEFRA, Stagl finds that “[t]his method is most suitable for the appraisal of policies, programmes or projects whose impacts are not well understood yet and therefore benefit from a multidisciplinary modelling of impacts” (Stagl 2007). In these terms, Stagl is referring more to a ‘transdisciplinary’ than a ‘multidisciplinary’ value, since the latter is more closely shared with the other techniques reviewed earlier here (Voss et al. 2006). After all, CBA and DMV are equally typically multidisciplinary (although the typical dominance of economics in the framing of the method means they are less interdisciplinary than MCA or SME). If it is used to involve citizens in more transparent, respectful and less circumscribed ways as aimed, then SME may also by this means claim some degree of ‘transdisciplinarity’ (Nowotny et al. 2003; Gibbons et al. 1994).

However, it remains the case that under principles of rigour shared by all rational choice approaches (including CBA, DMV and MCA), SME can be argued to display serious methodological deficiencies. In some circumstances, these can lead to artefacts in the ranking process, such as rank reversals that may confuse or undermine confidence (Salo & Punkka 2005). Despite the positive efforts that distinguish this approach over the others mentioned here, SME may also be challenged concerning the extent and depth in which it permits participants to explore the full range of ambiguities and uncertainties. So it may correspondingly prove limited in the degree to which it can deliver on the aims of social learning. But SME remains favourable in comparison with all techniques reviewed thus far, in relation to these particular aims.

Qualitative Participatory Deliberation (QPD)

Under this category of approach, are included an enormous diversity of different methods ranging variously through focus groups, citizen’s panels, stakeholder negotiation, interactive modelling, community visioning, do-it-yourself juries, open space, consensus conferences (Smith 2009; Fischer 2009; OECD 2001.; Grover-White et al. 1997; Blamey et al. 2000; Stirling et al 2007). Each particular method typically displays a variety of quite radical contrasting alternative ways in which it can be designed, commissioned, recruited, framed, bounded, overseen, focused, facilitated, staged, structured, reported, evaluated and articulated with other methods and with policy debates. Each of these attributes in the constituting of any particular instance of qualitative participatory deliberation is spelled out explicitly here, because each presents a dimension on which (as explained in Section 3.4) ‘the devil is in the detail’ in any attempt to draw general analytic or evaluative conclusions (Leach et al. 2010).

Compounding this complexity, is the fact that virtually any particular method of participatory deliberation (and any detail of the above kind in the implementation of each), can also be combined with any of the other methods reviewed in this survey. For instance, one particular approach to QPD reviewed by Stagl for DEFRA is ‘stakeholder decision analysis’ (SDA) (Stagl 2007; Burgess 2000). This initially employed a qualitative form of multi-criteria analysis. The method was later articulated with the quantitative procedure at the heart of MCM to form the synthetich approach called ‘deliberative mapping’ (DM) (Davies et al. 2003; Burgess et al. 2007). And DMV and SME inherently involve the use of some kind of participatory deliberation in association with their own quantitative procedures. Perhaps most flexible of all, ‘multi-criteria mapping’ (MCM) can be used as an adjunct to some variant of virtually any of the broad participatory approaches identified above (Section 5.2.4) (Stirling 1997b).

The task of generalising mentioned repeatedly here, is therefore especially difficult in respect of this category of approach. However, the bottom line response in relation to the key queries of interest here are as follows. The fundamentally qualitative nature of these processes means that the issue of metrics remains secondary. In short, it is possible to make use of any metric that might be considered relevant, but it is recognised that any comparison across different metrics will be subject to qualitative considerations – and that these properly form the central focus of appraisal. For those for whom adherence to a particular single metric is a matter of principle, then, all forms of QPD will tend to be seen as correspondingly deficient.
With regard to purpose, there arises another important point. In common with the real-world implementation of all other appraisal methods considered here, the underlying purpose of appraisal will typically differ as between different participants. Powerful incumbent interests will typically wish to use the exercise to justify some policy decision, such as to enhance the degree to which they are trusted, increase acceptance of their interests and reduce the risks attached to dissent and protest (Collingridge 1982; Collingridge 1980). This may, or may not, involve a desire to assert a particular pre-conceived referred decision (Stirling 2008).

Likewise, various stakeholder interests will wish to use appraisal as part of wider strategies to assert particular favoured policy outcomes – or to give a voice to perspectives that they have reason to believe are otherwise excluded. Practitioners of appraisal will typically experience great pressure to align with one or other of these powerful interests. But they may also hold strong interests of their own – for instance in broadening out the scope of appraisal and ‘opening up’ the picture given to policy concerning the implications of contrasting perspectives (Ely et al. 2013). For their part, ostensibly ‘disinterested’ participants like ordinary citizens will typically always have their own biases and enthusiasms – and will (like policy makers) often wish to gain a sense of satisfaction in contributing to a tangible policy outcome, sometimes even if this is at the expense of reducing complexity.

In general then, there is with participatory deliberation as with other approaches to appraisal, a need to be cautious about attributing any single clear-cut ‘purpose’ (Fiorino 1990; Fiorino 1999). Even individual perspectives may oscillate in complex ways between an instrumental purpose, aiming at some particular pre-conceived ‘right decision’, or a substantive purpose of finding in an open, balanced way what looks like the ‘the best decision’ under different views; and/or a normative purpose in ensuring that whatever method is used (and irrespective of the outcomes), the process itself is conducted appropriately (Stirling 2008). As with other methods, it is impossible to evaluate participatory deliberation in abstract terms, without being explicit as to the particular purpose.

**Multi-criteria Mapping (MCM)**

Multi-criteria mapping constitutes one attempt to address all the issues raised thus far in this review – spanning qualitative and quantitative approaches; giving balanced attention to an unconstrained array of issues and options; allowing participants ‘to be in the driving seat’ (without steering or constraining them in particular directions); but at the same time imposing clear principles of rigour in the ways that options and issues are appraised and the transparency with which this is conveyed to third parties for peer review (Stirling 1997b; Stirling et al. 2003). It is recommended in a DEFRA manual as an especially effective means to address these kinds of challenges (Dodgson et al. 2003).

At the same time MCM is distinct from all other methods reviewed here, in taking the fullest account of uncertainties and ambiguities and clearly expressing these in the final result – serving to help ‘open up’ the practical policy implications of divergent values, assumptions and contexts (Stirling & Mayer 2001). In short, MCM aims at the same time rigorously and accountably to deliver on all three kinds of purpose discussed in Section 5.2.3: instrumental (in allowing expression of particular interests); substantive (in illuminating the diverse concrete implications for decision making) and normative (in doing this in ways that are compliant equally with quantitative and qualitative principles of rigour) (Stirling 2006).

Like other methods, MCM can be implemented in various ways in order to meet different instantiations of these aims in contrasting contexts. It can be used purely as an interview technique, with deliberation carried out later on the basis of presentation of the qualitative and quantitative results. And – since the basic tool is an accessible form of web-based software – this may (depending on the aims) be undertaken either in person or remotely. With due caution, either approach can be combined and integrated with variously-staged group based deliberative processes. And as part of this, MCM can use as inputs (among others), the outputs of any of the other methods reviewed here – or scientific environmental assessment techniques. Likewise, it may itself be taken as an input to exercises in participatory deliberation reviewed in the previous section (Section 5.2.3) (Stirling & Coburn 2014).

On the positive side, MCM is relatively broad and flexible in scope, avoiding the imposition of constraints on the type of issue that can be taken into account or the way they can be defined. This contrasts with other multi-criteria techniques where appraisal is virtually always based exclusively on utilitarian trade-offs, where options and even criteria are sometimes prescribed in advance, where participants’ criteria are often aggregated on a single ‘value tree’, where scoring is usually performed by a narrow specialist group, leaving citizen or stakeholder input restricted to criteria definition and
weighting. These features allow MCM to faithfully reflect perspectives from a wide range of different participants without imposing undue constraints or engendering counterproductive tensions (Stirling et al. 2003).

On the negative side, MCM in itself and as it stands is still largely an individual interview-based tool. The interview process is quite demanding. Unless special additional arrangements are made, provision for effective group deliberation and citizen (rather than specialist) engagement can be limited. These deficiencies are readily addressed by incorporating MCM into a broader process providing both for citizen participation and intensive in-depth group deliberation involving both citizens and specialists. This more elaborate approach is termed ‘Deliberative Mapping’ (Davies et al. 2003).

It is important to recall, though, that under instrumental objectives prioritising the securing of justification, the distinctive degree of ‘broadening out’ and ‘opening up’ offered by MCM (Ely et al. 2013) can be viewed as a disadvantage. Although MCM can be used to illuminate a single ‘average’ picture of rankings across all perspectives, this is qualified by transparent acknowledgement of the degree to which this picture varies. Correspondingly, the lack of such transparency in other methods (like CBA, DMV, MCA and even consensus-oriented participatory approaches), can be seen as an advantage if the aim is simply to justify decisions (Collingridge 1983).

**Q Method (QME)**

Originating in social psychology (Stephenson 1953), Q Methodology is a powerful, mature and well-established approach, which unusually (like MCM) combines hybrid quantitative and qualitative dimensions (McKeown & Dan Thomas 1988; Stirling et al. 2003). It is particularly easy and cost-effective to implement. However, the style tends to be less well oriented to addressing the comparative performance of concrete policy options. Instead, the strengths of Q method lie in illuminating key distinct perspectives concerning the divergent reasons why different possible policies might be considered positive or negative. It is especially powerful as a way of identifying associations between contrasting ostensibly entirely separate enthusiasms and concerns. This can be useful, where the purpose is to understand better how different perspectives relate to each other (Cairns & Stirling 2014).

Q method is based on the compilation by the analyst of a large set of short clear statements on an issue in question, drawn from a rich diversity of sources and perspectives and covering a full envelop of the different evaluative dimensions associated with contrasting policy options. Engaged in relatively short individual interviews, representative individuals are recruited from a wide range of different perspectives to order these statements according to how much they agree or disagree with each. The results of these ‘Q sorts’ are processed statistically to reveal the degree to which positions on different dimensions associated and diverge from each other. As a result, Q can be very effective at identifying “similarities among individual attitudes, which may not have been known a priori” (Addams 2000) (p. 35, emphasis added).

The metric used in Q, such as it is, then, is an abstract measure of proximity and distance between perspectives. Like MCM, the purpose is as much to illuminate diversity and distinguish between contending reasons for different possible actions, rather than to focus single-mindedly on aggregated ‘best practice’ policies.

The main disadvantage of Q, in these terms, is that it is not primarily designed for application directly to alternative policy interventions. It is also not so much a directly interactive deliberative method – the kinds of learning that it can contribute to are more individual or collective on the basis of the results. There is typically no group interaction as part of the process itself. It is more effectively used to cast light on the divergent conditions and pros and cons associated with a set of policy options taken as a whole. However, by using Q to differentiate between contrasting perspectives and identify their principal priorities and concerns, it can be used as a powerful means to inform the recruitment of participants for other methods involving public engagement – like DMV, MCA, SME, QPD and MCM.
Appendix 10: References


Aldred, J., 1996. Are the Alternatives to the Contingent Valuation Method Any Improvement?


Bakker M, Verburg, P Scenario based forecasts of land use and land management at 1 km² grid and NUTS-X level, based on CLUE-S


DEFRA (2011) A framework for understanding the social impacts of policy and their effects on well being, A paper for the Social Impacts Task Force


Global Reporting Initiative, [https://www.globalreporting.org/reporting/reporting-framework-overview/Pages/default.aspx](https://www.globalreporting.org/reporting/reporting-framework-overview/Pages/default.aspx)


http://en.openei.org/wiki/Impact_Assessment_Toolkit#tab=Planning__E2_86_92


Lucas, K. and Mayne, R. 2013 A literature review for the EVALOC project: Social network theory and analysis EVALOC, Oxford, http://www.evaloc.org.uk/#!working-papers/c1x1h

LUCI: http://www.lucitools.org/


MAES (2012): An analytical framework for ecosystem assessments under action 5 of the EU biodiversity strategy to 2020. Discussion paper - draft version 9.6


Monitoring and Evaluation for Sustainable Communities (MESC) project, http://www.mesc-project.org/, School of Geography and the Environment, University of Oxford


National Accounting. Studies in Methods, Series F, No. 61. Sales No. E.93.XVII.12


Office for National Statistics, ONS, 2012. First annual ONS experimental subjective well-being results. ONS
Paper, S.W., Empowering Designs: towards more progressive appraisal of sustainability.


Sandel, M. 2012 ‘What Money can’t buy’, The Moral Limits of Markets, Allen Lane


Tools for integrated planning for land-use, biodiversity and ecosystem services. Defra project WC0794.


Smithers, R. 2015. SPLiCE Phase 1: A methodology for Rapid Evidence Assessments. Report prepared for DEFRA.

Social Auditing and Accounting http://www.socialauditnetwork.org.uk/

Social Return on Investment (SROI) http://www.neweconomics.org/projects/social-return-investment


TEEB (2012). Why Value the Oceans – A discussion paper. Edited by Yannick Beaudoin and Linwood Pendleton


UK National Ecosystem Assessment UK NEA FO 2014 *UK National Ecosystem Assessment: Synthesis of the Key Findings*. UNEP-WCMC, LVEC, UK.


United Nations Food and Agriculture Organization of the United Nations 2004


