D4.7 Implementation Guidance

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Contents

Executive Summary

Context:

The implementation sub-task is charged with developing guidance for the design and implementation of schemes that make use of the concepts of Natural Capital (NC) and Ecosystem Services (ES). Implementation schemes included under the remit of the sub-task include: Payment for Ecosystem Services (PES); Offsetting; Standards, Certificates, Reporting, Labelling and Procurement; Spatial Planning, Regulation and Development Control; and Nature Based Solutions. These represent important modes of implementation that operate under different logics in different application arenas (market creation and support, green business and finance, spatial planning), but which combine synergistically to support the development of Green Infrastructure (GI) and the implementation of GI strategies. They also give scope to interface with OPERAs exemplars by suggesting and testing improvements to schemes and to the informational, analytical and other tools and instruments they deploy.

There is a complementarity between the present deliverable (D4.7) on implementation and D 4.1. Schemes of implementation contribute to changes in how ecosystems are managed and exploited and, through this, contribute to attaining policy goals. The policy areas and goals and the specific Directives to which implementations can contribute have been explored in D4.1. D4.1 is concerned with the policy relevance of the NC/ES concepts. D4.7 addresses complementary concerns relating to the take-up of the concepts through schemes of implementation and aspects relating to the appropriate selection and design of schemes, which influence outcomes.

The implementation sub-task has studied past and on-going cases where the NC/ES concepts are operationalised in different applications contexts (geographies, scales, ecosystem types, ecosystem services) through schemes of implementation of different type.

In developing guidance for implementations, the sub-task has addressed three related concerns: the take-up of schemes and how this can be increased; the design of schemes and factors influencing scheme design and success; and the use of NC/ES tools and instruments in implementations.

- Depending on their status, the factors influencing implementation processes can act as drivers, triggers, enablers, or barriers to implementation, can be sources of potential strength or weakness, and can present threats or opportunities. Guidance must seek to clarify the determinants of implementation processes and their status and to identify opportunities to improve the take-up and use of the ES/NC concepts.

- To be successful, implementation schemes must be fit-for-purpose and -context. Guidance must seek to clarify the purposes that different schemes of implementation

can serve, identify criteria relevant to the appropriate selection, design and implementation of schemes, and clarify their implications for implementations.

- To achieve policy goals, the ES/NC concepts need to be integrated into tools and instruments, which in turn are operationalised through schemes of implementation. Guidance developed from an implementations perspective can support the development, selection and use of tools, instruments and capacities.

The favourability of conditions for implementation differs markedly across implementation contexts and schemes. Also, the challenge of operationalising ES/NC concepts through schemes of implementation is technically easier for some ES/NC than for others. ES/NC that can be more readily quantified and which are homogeneous, such as applies to carbon sequestration and storage, are more easily addressed than are habitat and biodiversity, which are heterogeneous and have qualitative, site-specific and non-equivalence aspects.

Progress in operationalising ES/NC concepts through schemes of implementation and market development is more advanced for the more easily addressed ES/NC. The sub-task has therefore placed special emphasis on the policy priorities of habitat and biodiversity conservation and enhancement and how ES/NC concepts may be used to achieve the policy goal of No-Net-Loss of biodiversity.

Methodological approach:

The methodological approach draws on recent developments in implementation science, which offers theory, frameworks, concepts, methods and tools for describing implementation endeavours, analysing determinants of implementation outcomes, evaluating implementations against their goals and objectives, and supporting new implementations.

Implementation science is a relatively new body of science that is used to help reduce the research-practice gap and to contribute to more effective implementations by developing, structuring and codifying understanding about determinants of successful implementations, implementation processes, and the relevance of contextual factors. It is a response to the specific challenges of evidence-based research. Evidence from existing implementations is used to develop insights to help design, execute and facilitate new implementations.

Our approach uses determinant frameworks and process models. Determinant frameworks are used to develop checklists of factors that are relevant for implementations as drivers, enablers, barriers and success factors. The advantage of determinant frameworks is that they provide ways to identify and structure factors that are relevant to implementation success. A limitation of determinant frameworks is that they can provide only limited information about the process aspects of implementation. We therefore complement the use of determinant frameworks with process models.

Process models are used to describe implementations as processes. They highlight: (i) any process steps/stages and the temporal sequence of these; (ii) any facilitation that is needed to support the implementation process; and (iii) contextual factors that are important for the process and/or outcomes. Process models can be used to describe existing implementations, but have the added advantage that they can be used also prescriptively in "planned-action" ("how-to-implement") mode, to guide the design and execution of new implementations.

The field of Ecosystem Service Science is still too young to be able to offer proven planned-action models, but it is important to start the process of developing such models and prototype guidance. Our approach involves review and meta-analysis of experience with individual implementation cases. We describe and evaluate these in terms enabling the development of prototype planned-action models and guidance. Importantly, insights can be drawn from both successful and less successful (or even failed) implementations.

The process of developing useful insights can be made more efficient by empirical case study work that combines an overview of a larger number of cases that are described and analysed at a lower level of detail as a basis for identifying a smaller number of case that are particularly interesting for in-depth study because they offer insight into specific drivers, enablers, barriers, success factors, fail factors or aspects of process or context. This approach also enables existing meta-analyses and reviews to be used to support original empirical work.

One of the challenges in implementation science when addressed to novel applications is that the long-term outcomes of recently-begun and ongoing implementations are intrinsically unknowable, so that implementations cannot be evaluated against their long-term goals and objectives and only against interim (and typically surrogate) indicators of success.

In the current context of often early development of new markets for ecosystem services and natural capital when longer-term outcomes of implementations are intrinsically un-knowable, continuity and smooth progress of the implementation process and continuing progress in building new markets for ecosystem services are useful interim surrogates for success alongside any early evidence of ecological effectiveness and ecological cost-effectiveness of schemes and their contributions to achieving policy goals. Continuity in these processes is necessary if they are to contribute to successful long-term ecological outcomes. This aspect can be monitored in real time. Equally, abrupt disruptions to erstwhile smooth progress of implementation processes and market infrastructure building offer opportunities to explore potential fail factors.

Factors influencing implementation:

Combining the use of determinant frameworks and process models, our approach identifies three broad groups of factors that influence the progress and outcomes of implementation schemes. These relate to aspects of the implementation context, process and design. An important element cutting across and influencing all these is governance.

Schemes of implementation have three main elements: a governance element, a process element and a content element. The governance element is concerned with who makes decisions about the implementation process and content. The process element is concerned with the process through which the intervention is designed. The content element concerns the intervention and what is needed to put it into effect.

In some cases, the NC/ES concepts can be integrated into existing governance arrangements. In other cases, a customised governance process for the implementation may be needed, involving the establishment of new governance institutions and arrangements involving actors from different sectors and interests coming together and engaging in some or all the processes of initiating, designing, executing, monitoring, evaluating and adaptively managing implementation frameworks, infrastructures and/or schemes.

On this basis, our approach has been to use a consistent descriptive and analytical framework to review implementation schemes and to explore factors relating to implementation context, process, design and governance. This was done across a wide range of different types of implementation scheme. The major products of this are: (i) descriptions, evaluations and analyses of existing implementations of different type across different scales and contexts and including both successful and less successful implementations; (ii) insights for policymakers and practitioners about important determinants of implementation outcomes; (iii) recommendations and suggestions for next action steps to take to encourage further implementations; and (iv) a prototype tool, CODIFIES, that offers a first-cut conceptual design for a Comprehensive Determinants Framework for Implementing Ecosystem Services, which can be developed further in the future to become an Implementation Design Framework.

CODIFIES offers a structuring framework for describing and analysing characteristics of implementation context, design and process (and cross-cutting governance aspects) as interrelated determinants of implementation outcomes. It offers diagnostics for selecting and designing schemes of implementation that are sensitive to the context and aims of the implementation. CODIFIES is a prototype tool, at an early stage of development. It was piloted within two OPERAs’ exemplars: Urban Dunes (Barcelona) and Seagrass (Mallorca).

Main messages:

The concepts of Ecosystem Services (ES) and Natural Capital (NC) hold a powerful potential to support more sustainable development, improve ecosystem restoration and conservation, provide nature-based solutions and improve the wellbeing of people.

Harnessing this potential in support of policy goals and human welfare ultimately depends on the concepts being integrated into information systems and used to support and facilitate decisions, actions and interventions that, directly and indirectly, influence how ecosystems are managed and which bundles of ecosystem services are preferred and delivered.

Different approaches and logics can be used to incentivise and drive changes in ecosystem management: market-based approaches (price-based, rights-based and information-based) are increasingly favoured over regulation or as complements to regulation. Changes in governance arrangements to include stakeholders and values that are often under-represented in decision making can also be used to reach more broadly-based decisions.

Policy interest lies in using the ES/NC concepts in support of policy goals and commitments, especially those relating to climate change mitigation and adaptation, habitat and biodiversity conservation, and the development of the 'green' economy. The ES/NC concepts have been used for some time already to support some areas of policy, for example climate policy. Their wider use in habitat/biodiversity conservation and enhancement is more recent and an important policy priority linked to the commitment, goal and principle of securing No-Net-Loss and/or Net-Positive-Gain of Biodiversity.

Scheme selection:

PES schemes are appropriate where the linkages between ecosystem management and service provision are well understood and where the need is to enhance or maintain a stream of ecosystem services when these are degraded, degrading, or threatened and this can be achieved by modifying, but not radically altering, the mode of ecosystem exploitation. Spatial-heterogeneity in costs and benefits of ecosystem service provision is a key determinant in the potential cost-effectiveness of PES schemes, but there are trade-offs between cost savings in securing benefits and higher transaction costs of more complex schemes. When benefits are public goods, these are bought by public institutions on behalf of society. When benefits are private or 'club' goods, the private beneficiaries pay. This offers scope for cost sharing when a mix of public and private benefits can be secured.

User charges are appropriate when costs are incurred to secure ecosystem service benefits, but benefits are excludable, opening the possibility to impose charges for access to benefits. Charges can be made for access to the amenity benefits of urban parks, national parks, wildlife reserves, privately owned forests and lakes, and tourists areas, especially when access can be controlled because there are only a limited number of transport routes or entry points. To avoid unfair distribution of the cost burden or perverse social impacts, charges may be made dependent on the characteristics of users, which benefits are accessed, and how benefits are accessed.

Offsetting schemes are appropriate when there is direct traceability between an event or action and resulting environmental damage, such that there is a clear line of responsibility between the responsible actor(s) and the damage and when all reasonable measures to avoid and mitigate the damage have been taken. Offsetting schemes are suited to situations where impacts are: immediate, direct, local and traceable; when they are measurable; and when they can be compensated for effectively through schemes of direct restoration or re-creation of equivalent habitat elsewhere; i.e. where the damage is clear and responsibility for it is clear-cut. Offsetting is only appropriate for non-critical ES/NC. Where to place the threshold between critical and non-critical ES/NC is therefore an important framing issue for implementing schemes of biodiversity offsetting. Were more stringent offsetting obligations to be introduced, such as no-net-loss requirements, this would generate a habitat and biodiversity ‘risk’ and actual costs on developers and investors. This holds implications for financial accounting, reporting, disclosure and rating.

Standards, certification and labelling schemes are appropriate in securing more sustainable ecosystem management and exploitation regimes. They prescribe and incentivise responsible practices by setting higher than legally-required minimum standards of protection. There are two main forms of implementation: business-to-consumer (B2C) implementations and business-to-business (B2B) implementations. Although there are important similarities, the forms have different drivers and (primarily) serve different stakeholder and purposes. They also operate on different logics and theories of change. B2C implementations have been dominant in the early development of standards, but there are limits to the environmental performance improvements that market-driven B2C implementations can deliver. There is a shift underway in favour of B2B schemes in supply chain management. An added value of voluntary standards is that they can extend the ‘reach’ of conservation policies beyond the legal jurisdiction of states. The cost burden of regulation and enforcement is also borne largely by private sector actors.

The concept of Green Infrastructure (GI), which refers both to networks of natural capital and to strategies for the development of these, provides a framework for understanding how different schemes of implementation can fit together synergistically. Although there is no ‘one-on-one’ correspondence, some schemes of implementation have higher relevance than others in conserving, enhancing and managing specific elements of GI (Table1)

Table 1: Implementation Schemes within a GI Framework

	Core Zones	Restoration Zones	Sustainable Use Zones	Urban Green/Blue Areas
PES Contracts	X	X	X	
User Charges	X	X		X
Offsetting	X	X	X	X
Standards			X	
Labels			X	
Reporting			X	
Procurement			X	
Nature Based Solutions		X		X
Planning & development regulations	X		X	

<u>Effectiveness and cost-effectiveness:</u>

Protecting existing core zones of high conservation value is the most effective way to sustain habitat and biodiversity. Spatial planning, regulation and development control are mainstays of implementation. To safeguard high value habits and biodiversity hot spots from damaging modes of exploitation conservation of core zones under private ownership can also be incentivised using PES contracts (largely publicly funded, but with scope for some private and/or hybrid financing arrangements) and user charges for access by private individuals to zones offering high amenity benefits. Use can be made of habitat banking and offsetting arrangements to cover some of the costs of maintaining and extending core zones.

The cost-effectiveness of restoring natural capital and ecosystem services is a function of their type and current status. Restoring degraded zones is never as effective as is protecting core zones, since full restoration to undamaged status is seldom possible. Restoration of degraded zones is nevertheless an important complement to protecting existing core zones and an essential part of overall GI strategy. Restoration can be incentivized using PES contracts, supplemented by user charges when possible, and funded by creating habitat banks and offsets. Some restoration projects can be implemented as nature-based-solutions, offering cost-effective alternatives to 'grey' infrastructure; e.g. in coastal protection by using restored salt marshes, dunes, or mangroves and in flood risk mitigation by restoring upland peat habitats and returning rivers to their natural courses.

For sustainable use zones, shifts to more sustainable modes of exploitation can be incentivized using PES contracts (e.g. in agriculture by replacing agricultural subsidies with stewardship schemes) and combinations of PES/Offsetting (e.g. in forestry by the production and sale of carbon offsets). The development and take-up of sustainable practice standards can be incentivised and driven by markets, but increasingly is driven also by businesses voluntarily shifting to more sustainable practices. Consumer-facing labelling schemes can drive uptake of standards and certification schemes, but inherent limits restrict what they can accomplish alone. Business-to-business schemes (supply chain management, private procurement policies) offer fuller scope and can be supplemented by public procurement policies. Business risk mitigation is increasingly important in driving the take-up of standards and certification schemes through pressure downstream businesses are able to exert on upstream suppliers, but also by supports offered to them. In turn, schemes for reporting and disclosure of information on business sustainability (which is salient for investors, shareholders, business partners and clients) re-enforces the drive toward sustainable supply chain management.

Urban GI is especially important for its multifunctionality and because its many benefits can be accessible to many people, including high priority groups, such as poorer people, younger people and the elderly.

Nature-based solutions are especially important in urban areas in contributing to urban GI, because they can provide effective multifunctional alternatives to grey infrastructure solutions and can contribute to a wide range of social and economic policy and development goals, as well as offer some habitat and biodiversity benefits. They are also important for disaster risk management as multifunctional alternatives to single-purpose 'grey' infrastructure that work with nature and that can offer more effective and more cost-effective solutions.

<u>Use of tools and instruments:</u>

Implementation schemes use different ES/NC tools and instruments in context- and purpose-specific combinations and sequences. There is no 'one-size-fits-all' schema. Understanding the context of the application is key to the design and implementation of schemes. Some tools and instruments are nevertheless important for all implementation schemes:

- Implementations are typically developed within higher-level policy frameworks and cascades. Higher level policy references are important drivers and enablers for implementations.
- Spatially-explicit assessment of ecosystem services (ES mapping) is fundamental for most schemes of implementation.
- ES and NC indicators are critical for making ecosystem assessments, setting targets and for designing, implementing, monitoring and evaluating schemes.
- Impact assessment has a very special role, since all implementation schemes depend on comparing factual with modelled counterfactual developments and this depends on the availability of ecosystem models and capacities to project and compare impacts.

- When management concerns common resources, stakeholder processes are important and can support a wide range of objectives. There is a risk, however, of undermining objectives such as securing buy-in and building trust if stakeholder processes are not sensitively managed.

<u>Implementation progress:</u>
The use of the ES/NC concepts in habitat and biodiversity conservation is much more challenging than in climate change or many other applications owing to the complexity of habitats and biodiversity, which arises from context specificity. The approach of habit/biodiversity offsetting is also more controversial and implies higher levels of judgment and subjectivity. Whereas units of carbon or carbon equivalence can be quantified and are the same everywhere, the significance and value of habitat and biodiversity are specific to their location and context and must be assessed in context, which involves qualitative judgement.

Increasingly, implementations of the ES/NC concepts are engaging with more complex challenges that require customised implementations, with context- and purpose- sensitive selection and design of implementation scheme. This holds implications for the need to develop the core infrastructure of locally-accessible competences and capacities to support implementation processes, for access to toolboxes with customisable tools and instruments, for guidance in designing and implementing schemes that are fit-for-context and fit-for-purpose and for exemplar cases that can inspire and guide concept mainstreaming.

Continuing environmental change, including continuing loss of ES/NC, and awareness that public funds and actions alone are insufficient for effective conservation are important factors shaping policy priorities. Important considerations for policy makers are the needs to *broaden responsibilities for ecosystem conservation* to include key actors and stakeholders beyond government and its agencies, to *integrate conservation into markets and other mechanisms for decision making*, and to *increase the flows of private investment into conservation* efforts.

Concerted progress in operationalising and implementing ES/NC concepts is being made. The range of different approaches to operationalising and implementing the concepts is extending and expanding across contexts, scales and applications arenas. This is important *because the ultimate viability of markets for ES/NC and the possibilities to harness and capture the significant potential of markets to contribute to preserving and enhancing ES/NC depends upon realising synergies across implementations and achieving critical mass*.

Schemes of implementation – such as Payment for Ecosystem Service (PES) schemes, Product Labelling Schemes or Offsetting schemes – have a dual function. They are the key mechanisms through which the ES/NC concepts are taken-up and used in practical applications to influence ecosystem management practices, but also through which key elements of this infrastructure for ES/NC markets and implementations are built.

The critical infrastructure for implementing ES/NC concepts is at different stages of development in different applications arenas and for different schemes of implementation.

Overall, however, many elements of critical infrastructure that are needed to support practical applications of ES/NC concepts are still in early establishment stages. This makes it important to distinguish between short-, medium- and longer-term ambitions for implementations. The ultimate goal is for practical implementations of the ES/NC concepts to support effective ES/NC conservation and contribute to sustainable development and use of ecosystems. The short- to medium- term objectives of implementations must include building the critical elements of infrastructure – data, information, concepts, tools, instruments, capacities and trust – to enable this ultimate goal to be achieved. For this reason, important interim indicators of 'successful' implementations at this stage in the process of moving the ES/NC concepts from science to practice include the contribution of implementations to building elements of critical infrastructure, maintaining progress in building and sustaining new markets, and creating new funding sources and streams for investing in ES/NC.

Many of the 'barriers' to implementation identified in earlier studies have begun to be addressed. In particular, the framework of policy references has been substantially strengthened at highest (EU) level. Policy references are being cascaded down the governance hierarchy into Member State and regional/local level policies and plans. There are still 'awareness' and 'informational' barriers to take-up, especially at local level and a need to continue to strengthen local access to NC/ES information, tools, competences and capacities. Transaction costs are high, but will fall as take-up grows. The need for custom-designed purpose- and context-sensitive implementations implies a need to continue to build local access to capacities and competences to support implementations. Lowering transaction costs also calls for more streamlined stakeholder engagement processes.

Take-up and confidence are still hampered by a lack of evidence of the ecological effectiveness of schemes of implementation. This is relevant for wider public acceptance of offsetting and the take-up and standards, certificates and labels.

More systematic monitoring and evaluation of ecological effectiveness is needed and, owing to the long lead times between interventions and their impacts, monitoring and evaluation need to be incorporated in implementation designs from the outset.

Adding to the available range of tools, instruments and capacities, improving access to these, lowering transaction costs, which are typically high in the early stages of developing markets but which are reducible, securing new financial flows, integrating ES/NC into ecosystem management decisions, and changing ecosystem management practices are all significant and operational interim indicators of implementation progress in the short- to medium- terms. Such 'process' indicators can help to identify success cases and support evidence-based policy making, especially when longer-term outcomes on ultimate targets, such as biodiversity conservation, may only be measurable over longer time-frames.

CODIFIES:

CODIFIES is a prototype tool, at an early stage of development. It was piloted within two exemplars: Urban Dunes (Barcelona) and Seagrass (Mallorca).

In the Urban Dune exemplar, it was suggested that the scope be extended from only dune reconstruction to embrace wider issues of coastal management, including to revisit the current hard-engineering approach to coastal defence. Experimenting with alternative approaches to coastal management using nature-based approaches is warranted because there are no proven and cost-effective solutions to the coastal management challenges presented by this case, but there are promising nature-based and hybrid interventions that involve working with (rather than against) nature. Also, the existing management approach imposes high and recurring annual costs and is demonstrably ineffective. New risks associated with climate change and sea-level rise increase the urgency to find new cost-effective solutions. It was proposed also to include a wider group of stakeholders in coastal management and specifically to engage those parties with interests and potential roles in delivering effective outcomes. It was further proposed that to finance nature-based experiments and to develop appropriate governance processes, some funds deployed currently in ineffective engineering approaches might be redeployed. This is a low or no risk option with potential to offer win-win outcomes, as successful experiments with low cost nature-based approaches to coastal management would deliver more comprehensive solutions and cost savings to the port authority that currently finances the annual, but ineffective, sand replenishment programme.

In the Mallorca seagrass (Posidonia) exemplar, it was suggested that an implementation scheme based on *user charges* would be most appropriate to mitigate stresses on seagrass arising from pressures linked to tourist activity. Seagrass beds sequester and store carbon, and are important in climate regulation, but this ecosystem service can only be secured and valorised *if* the long-term health of seagrasses is secured. Currently, the Mallorca seagrass beds suffer tourist-related stress from high levels of discharge of sewage effluent and from direct physical damage by the (illegal) dragging and dredging impact of pleasure boat anchors and chains. But the seagrasses are multifunctional. In addition to carbon regulation they provide other regulating ecosystem services: they remove sediments from sea water, protect beaches from storm damage, and provide critical habitat to support marine biodiversity. These are important in maintaining the high quality environmental features that attract tourists to Mallorca in the first place: sandy beaches and clean and clear bathing water.

Since the damage to seagrasses is linked directly to the additional stresses of tourists generally (effluent discharges during high season) and some recreational boat users specifically (illegal use of anchors over seagrass beds) and since these immediate causes of damage can both be addressed by known technical solutions (increases in sewage treatment capacity and the installation of permanent floating mooring buoys), an appropriate approach is to impose *user charges* for access to the tourist benefits of Mallorca’s marine and coastal ecosystem and to dedicate part of the revenues to cover investment and operating costs of the needed equipment.

Estimates (e.g. Aguilo *et al*) of own price elasticity of Balearic tourism demand from the major source countries, such as Germany (0.84), the UK (0.98) and the Netherlands (0.51), indicate that tourist demand is relatively inelastic to price and, therefore, that a tourist tax will increase total tourist revenue. Part of the receipts can be hypothecated to underwrite and amortise capital investment in enhancing sewage and waste water treatment capacities to enable these to address high season demands (heavily augmented by the seasonal tourist population) that are otherwise uneconomic and unaffordable to address by the (much lower) resident population. As Mallorca is an island and tourist access to ecosystem benefits depends on tourists entering through a limited number of airports and ports and their staying in hotels and other visitor accommodations, a general tourist tax is simple to administer by levying taxes on tourists based on length of stay. As the marginal environmental damage cost of tourism is a function of the overall number of tourists on the island at any given time and this varies across the year, some fine-tuning of any tax is warranted between high and low season.[1]

Since the direct physical damage to Posidonia is directly due to illegal mooring over Posidonia beds using anchors, the cost of installing mooring buoys and of policing/enforcing their use can be passed onto those using recreational boats. The problem is linked mostly to casual users of boats, rather than to experienced yachtsmen and could be addressed by providing information at boat hire stations to explain the rationale for the existing regulations that require using floating mooring buoys, by levying mooring charges as part of boat hiring fees, and by backing this with improved policing and enforcement. The user charges should reflect the actual cost of installing and maintaining a network of floating mooring points. Fines for illegal use of anchors over Posidonia beds should cover the costs of policing and enforcement. Work within the exemplar revealed sensitivities and differences of perspective among stakeholders over boating freedoms. Stakeholder processes could be established to run alongside trials aimed at raising standards using an evidence-based, adaptive management approach.

[1] During the OPERAs project, the Balearic government has approved a tourist tax. This has been implemented since July 2016. The tax level depends on accommodation type, ranging from €0.50 for campsites and hostels to €2 for five-star hotels. Children under the age of 16 are exempt. The tax is reduced to half rate from the eighth day on the island and during low season (November to April). A committee comprised of representatives of the tourist industry, environmental groups, the government, and trade unions has been established to decide how tourist tax revenues are used. The eligible fields include: the construction of new infrastructures for sustainable tourism; the protection and preservation of the environment; the conservation and restoration of historical and cultural heritage; and, research and technological innovation.

1. Introduction

1.1 Objectives of Task 4.4

The task contributes to achieving the overall goal of OPERAs, which is to explore the operational potential of the NC/ES concepts in improving ecosystem management and achieving policy goals.

Under this overarching goal the objectives of Task 4.4 are:

- To understand and improve the take-up and mainstreaming of ES/NC concepts through schemes of implementation in different arenas (and, with that, secure the take-up of NC/ES tools and information, including those developed in OPERAs)
- To identify drivers and opportunities for mainstreaming as well as barriers to uptake and ways of overcoming these
- To help ensure that implementations and operational designs for these meet stakeholder-defined performance criteria and are fit for purpose and context
- To help secure synergies between schemes.

1.2 Conceptualising ‘implementation’

Harnessing the NC/ES concepts in support of attaining policy goals and targets ultimately depends on the concepts being implemented in ways that stimulate and support more sustainable ecosystem management. NC/ES implementation involves actions and interventions, which are intended, directly and indirectly, to influence how ecosystems are managed. Implementations may also involve sets of actions and interventions in the form of multi-action and multi-instrument schemes of implementation. Implementation can therefore be conceptualised at the level of discrete actions and interventions, such as the integration of NC/ES knowledge into policy development processes in order to influence policy goals and to enrich the set of indicators used to measure progress. Equally, implementation can be conceptualised more comprehensively as involving sets of connected actions along the complete ‘data-knowledge-instruments-decisions-management’ chain or parts of the chain, such as integration of NC/ES concepts into data collection and into existing or new tools and instruments in order to develop richer NC/ES knowledge, the integration of this richer knowledge into decision processes, and, ultimately, into new frameworks for action and into ecosystem management practices. The ultimate objective of implementation of the concepts in either conceptualisation is to deliver healthier ecosystems and secure streams of ecosystem benefits by changing the ways ecosystems are managed.

Schemes of implementation typically involve consistent sets of actions and interventions that work together to influence how ecosystems are managed. Schemes of intervention also typically have a defining approach to change, which identifies a dominant change mechanism/instrument. Typically, comprehensive schemes of intervention are often named after these dominant mechanisms, such as PES schemes, offset schemes and certification schemes, even though these schemes also deploy many different tools and instruments. Schemes may be organised around principles of voluntarism and/or compliance. Schemes may be designed also to transform the framing conditions for change; for example, to raise awareness, build support for change, create and coordinate new networks of actors, and improve financing prospects. Schemes may be initiated by different lead actors from different sectors of society as well as by combinations of actors. The interests and perspectives of scheme initiators may therefore be reflected in scheme designs.

Schemes of implementation typically combine and use NC/ES instruments of different types (statutory/policy instruments, legal/regulatory instruments, financial/economic instruments, scientific/technical instruments, etc.) to help achieve intermediate objectives, such as creating markets, renewing governance, and securing new finance) and, through this, change ecosystem management (Figure 1.1).

Figure 1.1: From instruments to delivery mechanisms: some illustrative change mechanisms and instrument types that they might deploy

Typically, implementation schemes have an internal logic, such as ‘polluter pays’ or ‘beneficiary pays’ in respect of market creation and support, and they may adopt principles to motivate and drive change and to coordinate actors and actions across different levels and sectors, such as a ‘no-loss’ or a ‘no-net-loss’ principle. Different instruments play different roles in implementation schemes and, often, they play multiple roles. All schemes, such as those for corporate reporting, PES, or offsetting, offer design options; for example, choice of metric in respect to biodiversity/habitat offsetting or choice of a basis-for-payment in respect of PES schemes. Choices hold implications for scheme performance that need to be well understood if schemes are to be acceptable to stakeholders, efficient, and effective.

The application context is highly relevant for scheme design. Different contexts require different scheme designs. Contextual factors, such as who has property rights over ecosystems and whether benefits are public or private, are very important for implementation scheme development and design (e.g. in respect of deciding who are the main actors and stakeholders to engage with, in determining the potential costs and benefits of schemes, and in appreciating the potential distributional impacts of schemes); as, also, is the state of knowledge about the target ecosystem (i.e. how well understood it is). Some contextual factors, such as ecosystem scale and, in the case of voluntary markets, the scale of the markets into which ecosystem services are bought and sold, may also influence the prospects for successful implementations. There are important relationships, therefore, between implementation context, implementation design, and implementation performance.

Importantly, despite differences in the logics and principles that underpin schemes and differences in other aspects, such as the spatial scale at which schemes operate, different implementation schemes can work together synergistically. Change in how a particular ecosystem is managed may ultimately be made viable, operationally and financially, by the combined action of several different schemes of implementation. Identifying synergies among schemes and seeking deliberately to catalyse and capitalise on such synergy is therefore an important opportunity for taking up the potential of the NC/ES concepts.

In principle there is a potential for a positive feedback loop, or virtuous cycle, to develop, such that the more the NC/ES concepts are operationalised in different schemes of implementation and the more the different schemes work together synergistically, especially to change the incentives that natural resource owners and managers face, the greater the prospect for changing how ecosystems are managed and the greater the demand for NC/ES information, instruments, and support services.

1.3 Methodological approach

Against the backdrop of concern to understand and support the take-up of NC/ES tools, instruments, and knowledge, the overall aim of Task 4.4 is therefore to develop guidance for designing implementations that characterises the different phases, steps, and design options in implementation processes, the tool, instrument and information needs that these imply, and how these can be met.

This is achieved in Task 4.4 by exploring how NC/ES concepts are operationalised in practical schemes of implementation and by developing implementation design guidance aimed at supporting further implementation based upon lessons deriving from experience and evidence of implementations both outside and within the OPERAs project.

The guidance focuses on the relationships between implementation context, implementation design, and implementation performance as mediated through information and instrument needs and the use of specific NC/ES tools to meet needs.

Task 4.4 is designed to complement rather than to overlap with or duplicate the work of Task 4.1. The policy areas and goals to which implementations of the NC/ES concepts could contribute are explored in WP4 Task 4.1, which is concerned with the policy relevance and policy integration of the NC and ES concepts: i.e. with 'what' policy areas and objectives NC/ES concepts could serve; 'how' and 'how well' the concepts are integrated into policy processes, policies, programmes, and instruments; and, how the concepts might be more fully integrated into policies and programmes. Task 4.1 is concerned especially with the integration of the NC/ES concepts into high-level (EU) policies and instruments as direct policy implementations of the concepts and in creating policy frameworks for more decentralised implementation processes.

By contrast, Task 4.4 focuses on the design and delivery of decentralised implementation schemes, many of which are multi-level, multi-actor, and multi-instrument schemes. It focuses on scheme design, including on how different instruments are combined in specific schemes in particular application contexts. Equally, whereas Task 4.1 focuses especially on which policy areas and which policy goals might be supported by or could support the policy integration of NC/ES, Task 4.4 is focused more on how well an implementation performs across a wider range of criteria.

A challenge is that although there are many independently-developed descriptions and assessments of implementation schemes of different types reported in the literature, these are not developed according to any consistent approach and framework. Developing such a framework is therefore an important step in being able to develop and deliver guidance for scheme design.

A key aspect here is that individual implementations are context specific and, in order to perform well, individual implementations must be designed in relation to their purposes and application contexts. Developing guidance to support implementations that are 'fit-for-purpose' and 'fit-for-context' must therefore be based on an understanding and characterisation of contextual factors that are relevant to scheme design and scheme performance. The framework for describing, analysing, and comparing implementation schemes must therefore provide for the relationships between implementation context, design, and performance to be explored. For this, there is a need to develop a parameterising template for each of the three elements – context, design, and performance – that identifies potentially important aspects, so that the relationships among these can be explored systematically across many implementations.

In order to develop design guidance, there is therefore a need to parameterise and characterise *implementation contexts* as well as to parameterise and characterise *implementation design options,* and there is a need, also, to match these. In principle the concern is to design implementations able to perform well in the application context not only in relation to their intended purpose, but also on other aspects of performance that might be important for stakeholders, such as cost, or the contribution of the implementation to supporting wider goals of contributing to the green economy or to reducing business and/or financial risk. For this reason, the matching process must also take account of relevant *implementation performance criteria*, which must also be parameterised.

The steps in developing implementation guidance therefore include: parameterisation and characterisation of relevant features of implementation context, design, and performance (step one) and of tools and methods for assessing implementations (step two) as a basis for developing generic descriptions of broad types of implementation (step three) as well as describing and evaluating implementation experiences across a range of contexts for specific implementations within each broad implementation type (step four). Meta-analysis of schemes (step five) enables lessons to be learned from implementation experience about the importance and role of contextual factors in scheme design and performance, especially about what different features of context imply for implementation scheme selection and design, for information needs to support the implementation, and for tools/instruments that make use of the NC/ES concepts. Design guidance can then be drafted on the basis of lessons learned (step six). Guidance can be tested and refined in the OPERAs exemplars and beyond, providing also additional illustrative examples of evaluated implementations (step seven).

In terms of work organisation, steps 1 and 2 above are carried through in sub-tasks 4.4.1 and 4.4.2 respectively. All of the other steps are integral elements of each of the remaining sub-tasks, 4.4.3, 4.4.4, and 4.4.5. Each of these last three sub-tasks addresses a different broad type of implementation, describing each type in generic terms and through illustrative examples of specific implementations in a range of different applications contexts chosen to highlight relationships between contextual factors, design options, information and instrument needs, design choices and aspects of implementation performance. Guidance is developed from the experiences, which highlights information about these relationships and provides insights, also, into potential trade-offs between scheme design and performance and in which implementation contexts such trade-offs might arise.

1.4 Different frameworks for implementations

The approach to organising the empirical work recognises different arenas within which implementations can be developed and deployed, each addressing decision-making under regimes of power that are constituted differently and each contributing to the creation of different frameworks for designing and operationalising implementation schemes. These include the very different arenas of statutory power, business power, and financial power:

- The statutory arena is constituted through powers vested by statute in public governance institutions at different governance levels. It engages public policy and decision making from strategic to operational levels as this interfaces with ecosystem management through, inter alia, goal setting, policy, the management of public funds, spatial and physical planning, development control, etc. The underlying hypothesis is that there is a need for a renewal of the governance model and for an enhanced toolkit to support decision making for more sustainable ecosystem management in the statutory arena principally by taking account of a wider and/or differently weighted set of criteria in decision making, but that this is also an opportunity; e.g. for public/community engagement and for policy integration.

- The business arena is constituted through powers and influence that businesses hold as owners and/or orchestrators of the means of production, including through the impacts businesses have on ecosystems directly and indirectly through their uses of natural resources and ecosystem services, and that consumers have through effective demand and through changing the level and structure of demand. The underlying hypothesis is that business decision making and consumer decision making are both distorted by incomplete information and that richer information about NC/ES (e.g. via Environmental Management Systems (EMS), reporting systems, ratings systems, environmental product profiling, foot- printing, standards, certification, and labelling) would provide for better informed production/consumption decisions.

- The finance arena is constituted by powers and influence exerted through the creation and support of markets, of streams of finance, or of information needed for fully-considered and balanced investment and financial management decision making. The underlying hypothesis is there is underinvestment in NC/ES because of market failures (missing markets, missing information, etc.) and because information on the values of NC/ES (or the costs and risks of losses of NC/ES) does not enter equally into public- or private- sector decision processes and that corrections could come through public and private sector financial reform, market creation to internalise costs and benefits (PES, offsetting, user charging), procurement policies, etc.

Importantly, although they can be distinguished along the lines of how decision making powers are constituted, these three decision making arenas operate together interactively (as in Figure 1.2) rather than separately and discretely. This implies that implementations, as schemes for managing change, may operate in frameworks constituted by a single authority or within frameworks created by different authorities.

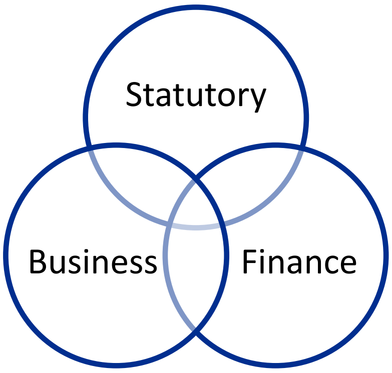

Figure 1.2: Three contextual arenas in interaction.

Table 1.1 below provides a summary of some implementation schemes (or key elements of these) in relation to the three arenas and matches implementation schemes and arenas to possible OPERAs exemplars.

Table 1.1: Implementation schemes represented in OPERAs exemplars

EXEMPLAR	STATUTORY ARENA	BUSINESS ARENA	FINANCE ARENA
	Public Sector Policy Renewal, Strategic Priorities, and Policy Integration, Spatial Planning, Development Control, Urban Governance Model & Toolkit	EMS, Reporting, Rating, Footprinting, Certification Labelling, Insurance	Public sector finance reform, Public Procurement, Private Purchasing, PES/User Charges, Credits, Offsetting/Habitat Banking, Crowdfunding Responsible Investment, PPP, Least cost investing
1. Greater Dublin	Social valuation, urban governance model & toolkit		Least cost investing
2. Urban Dunes	Social valuation, urban governance model & toolkit		Habitat Offsets, Crowdfunding, PES, User Charges
3. Montego (cork)	Valuation, strategic prioritisation, policy integration	Certification, Labelling	Pro-biodiversity business (PPP)
4. Balearic Islands	Valuation, strategic prioritisation		Biodiversity Offsets/Bio-Carbon Credits
5. Lower Danube	Valuation, strategic prioritisation, policy integration	Certification, Labelling, Footptinting, Reporting	Pro-biodiversity business (PPP), Public Procurement
6. Central Alps	Valuation, strategic prioritisation, policy integration	Certification, Labelling	Offsets, pro-biodiversity businesses, procurement
7. Wine	Valuation, strategic prioritisation	Certification, Labelling, Reporting, Rating	Offsets, pro-biodiversity businesses, procurement
8. Scotland	Valuation, strategic prioritisation	Certification, Labelling, Footptinting, Reporting	Pro-biodiversity businesses
9. Circum-Mediterranean	Valuation, strategic prioritisation, policy integration	Certification, Labelling	Offsets, pro-biodiversity businesses, procurement
10. Pan European	Valuation, strategic prioritisation, policy integration		Habitat banking
11. Global	Policy integration	Certification, Labelling	PES

Summary table of potential matches between implementation schemes (or elements of these) and exemplars. . Schematic scenarios of implementation possibilities could help decide which to explore.

Paul Weaver, University of Lund, 17.10.2013

1.5 Types of implementations explored

Against this backdrop, the broad types of implementation scheme considered in Task 4.4 are: Payment for Ecosystem Services (PES); Green Infrastructure; Offsets; and, Standards, Certification, Labelling, Procurement. These are selected because of their actual and potential importance in influencing how ecosystems are managed, for the scope that exists for developing synergies among them, and for the design issues they raise and which must be addressed through the development of guidance in order to support future deployment. A cross-cutting aspect is that Task 4.4 also looks at different funding sources and financing instruments/mechanisms in relation to these different broad types of implementation schemes.

2. Parameterising implementation contexts, schemes and performance

2.1 Templates for scheme characterisation

Owing to the diversity of implementation purposes and contexts there are no simple one-size-fits-all answers to questions concerning choosing a suitable type of implementation scheme or to addressing the details of implementation design. Rather, purpose and context matter for implementation design, so each implementation should involve processes to assess the application purpose and context and to design an implementation that is 'fit-for-purpose', 'fit-for-context', and that meets performance criteria relevant for stakeholders.

The aims and objectives of task 4.4.1 are to develop general templates for the parameterisation and characterisation of key elements of implementations – contexts, designs (process and content), and performance – as a basis for developing rich descriptions of implementations in subsequent sub-tasks (4.4.3, 4.4.4 and 4.4.5). The rich descriptions are used in these subsequent sub-tasks to develop understanding and insights into the relationships among these elements, such as contextual rationales for design choices or the influence of context and design choices on performance. In turn this provides for developing evidence-based guidance for the design of schemes, helping ensure future schemes are fit-for-context, fit-for-purpose, and acceptable to stakeholders.

Methodologically, the approach to developing templates is to draw on the consortium's own experience of past implementations and on literatures that describe others' experiences of past implementations in different contexts, using these to identify context, design, and performance parameters that recur across implementations and are identified by scheme developers and analysts as important in influencing scheme designs and outcomes. The work also draws on government, corporate and other documents that set out or give insight into aspects of scheme performance relevant to stakeholders.

Since the 'dimensionality' of the design, context, and performance parameters is unknowable in principle in advance, the templates must also evolve and develop through an iterative approach with initial templates being informed by existing knowledge but also being improved and extended through use in sub-tasks 4.4.3, 4.4.4, and 4.4.5. This means that exploring the dimensionality of contexts, designs, and performance – and adding further parameters – becomes an explicit objective of these sub-tasks, alongside developing rich descriptions of implementation contexts, designs and performance.

Some dimensions to include in the templates were clear early on. Others were added through empirical work during OPERAs. The templates (Tables 2.1, 2.2, and 2.3) can be considered as continuing works in progress, since they can continue to be developed.

Table 2.1: Parameterisation of the implementation context		
A: The ecosystem and its management		
Aspect	**Description**	**Case 1, 2, …n**
Which ecosystems and/or economic activities/sectors are targeted and why?	e.g., tropical forests, marine fisheries, agriculture, bio-fuel, mining…	
Are there threats to NC/ES and, if so, what is the spatial pattern of threats?	e.g., unsustainable current exploitation/management regime (deforestation, overstocking, overfishing), climate change, etc. To which extent are threats uniform across the ecosystem or more variable; are threats diffuse, cumulative or deferred (subject to lags and inertia) versus specific, direct and immediate; is damage (or threat of damage) directly traceable to specific sources, actions and actors or more diffuse	
Which NC/ES are targets of the implementation?	e.g., clean water, flood protection, carbon sequestration/storage, habitat, fish stocks	
What types of service are these?	e.g., provisioning, regulating, supporting, cultural	
What types of good are these?	e.g., public, private, club	
What is the scale of service provision?	e.g., global, international, national, catchment, local, etc.	
To which extent is it possible to delineate the limits and spatial extent of the ecosystem function and services?	e.g., precisely delineable (e.g. watershed) , fuzzy (e.g. pollination) or somewhere on the (easy-to-hard) definability continuum between these extremes	
Is there potential, in principle, to increase supply of target NC/ES?	e.g., are there potentially effective ecosystem management interventions; what do these entail (upfront action, continuous action, both); is it clear who could deliver management changes	

Table 2.1: Parameterisation of the implementation context

B. Economic and social aspects		
Aspect	**Description**	**Case 1, 2…n**
How is the structure of potential supply?	Specific or diffuse: single, few or many producers? Single or co-produced services?	
How is the structure of potential demand?	Specific or diffuse: single, few or many beneficiaries? Single services or service bundles?	
Is the involvement of particular producers or beneficiaries critical for the scheme?	Which? Why?	
What are the marginal costs of NC/ES supply?	What are the direct costs, the opportunity costs, the transaction costs?	
What is the spatial pattern of the costs of NC/ES supply?	How spatially uniform or variable are opportunity costs of supply?	
What is the potential value of marginal ES benefits?	Who are potential beneficiaries and what values do they attach to benefits?	
What are the spatial and social patterns of ES benefits and beneficiaries?	How spatially and socially uniform or variable are benefits? Are some potential beneficiaries in high priority social groups? Are benefits dependent on proximity or access to the ecosystem?	

Table 2.1: Parameterisation of the implementation context		
C: Scientific aspects		
Aspect	**Description**	**Case 1, 2…n**
How strong is the existing science and evidence base for scheme development and implementation?	How well understood is the ecosystem, its function, its dynamics, and the relationship between its management and NC/ES delivery? How well understood are threats to its function and to NC/ES? Do models of the ecosystem exist already? How well understood are relationships between management interventions and NC/ES? Are these relationships incorporated into any existing models of the ecosystem? If not, how easily could they be incorporated? How good is knowledge of the potential benefits and beneficiaries of different ES? How good is knowledge of the values ES represent to potential beneficiaries? How well understood are the opportunity costs of NC/ES supply? How accurate are available indicators or proxies for the targeted NC/ES? What measuring, monitoring and verification is needed? To which extent is measuring and monitoring of the NC/ES data-intensive, skills-intensive and/or work- intensive? What of the needed infrastructure and capacity exists currently. Is this in situ? Is there scope to measure or monitor by remote sensing or in a low cost way?	

Table 2.1: Parameterisation of the implementation context		
D: Governance aspects		
Aspect	**Description**	**Case 1, 2, …n**
In which political jurisdiction(s) is the ecosystem located?	e.g., EU and Member States, other states…	
What level(s) of governance are involved?	e.g., Global, EU, National, Regional, Local…	
What is the legal, social and political framing for the undertaking? Is ecosystem management framed significantly by regulations or policies? Which? At what levels of governance are these established? What is the nature of the influence?	e.g., agricultural policy, cohesion policy, biodiversity policy, water policy, fishery policy, maritime policy, nature and landscape protection policy; corporate policy; physical planning regimes and regulations...	
Are there any specific drivers and are there any triggers for the implementation? Is the undertaking related to a specific stage in an implementation or assessment cycle?	e.g. is the implementation policy-driven, opportunity-driven, risk-management driven, regulation-driven, etc.	
Who are the scheme proponents and what are their motivations?	e.g., government, planners, developers, trade associations, businesses, regulators, NGOs, etc.	
Who are the main actors, their roles and their interests/stakes?	e.g. who are the ecosystem managers, investors, service beneficiaries, intermediaries, knowledge providers etc. and what do they want from the scheme	
What types of actors are these?	e.g. state, private, civil society, etc.	
How clear are property and user rights over the ecosystem?	e.g., are rights clear or fuzzy, accepted or contested, etc.	
What is the degree of consensus/conflict over the goals and/or how to achieve them?	e.g., on the spectrum between widely accepted by actors and stakeholders versus contested. Is there broad agreement or	

Table 2.1: Parameterisation of the implementation context

	disagreement over what to achieve? Is there agreement/disagreement over how to achieve this?	
What are the most significant power relations between the actors, how are these established, how is power distributed in relation to stakes in the scheme?	e.g., equal or unequal power relations among stakeholders, strong or weak and positive or negative correlation between power and stake	
To which degree are citizens able to take part in shaping decisions that affect their lives and their opportunities?	e.g., what is the status and tradition of stakeholder participation, deliberative democracy and/or participatory budgeting	

Table 2.2: Parameterisation of schemes (generic types)

Aspect/Feature	Description
Definition	How is this type of implementation defined? Is the definition contested or accepted? What is/are the most-widely accepted definitions and the source(s) of these?
Rationale, goals and objectives	What is the central idea or concept that underpins this type of implementation? What ecosystem management and/or market deficiencies does it seek to address and how? What is the potential added-value of the approach? What policy goals and objectives can it serve?
Thematic (or other) variants	Are there variants of this type of implementation? How are these distinguished; e.g. by theme, unit of assessment? Do these need to be characterised individually?
Frameworks for the implementation/variant (policy and normative references, funding opportunities, etc.)	What are the main higher-level policy references that frame and/or facilitate this type of implementation? Which fiscal, regulatory, financial or other policies and initiatives are important? Does the implementation draw on higher-level normative references? Which? Which actors play roles in creating facilitating frameworks for this type of implementation?
Drivers	What are the trends in the use of this type of implementation and what drives uptake; e.g. government policies and commitments; requirements of industry regulators; scientific progress; awareness and capacity-building; regulatory compliance; search for efficiency gains; concerns for supply-chain security; taking up opportunities created by policy changes; etc.
General principles and underpinning implementation ‘logic’	Are schemes voluntary or mandatory; based on a beneficiary-pays or a polluter-pays principle; based on direct or indirect linking of service suppliers and beneficiaries; etc. Are such distinctions relevant in distinguishing variants of this type of implementation; e.g. are there voluntary and mandatory variants?
Similarities, differences, and complementarities with other schemes of implementation	How does this scheme of implementation compare with and relate to other approaches/schemes that influence how target ecosystems are managed? What are the similarities and differences in the

Table 2.2: Parameterisation of schemes (generic types)	
Aspect/Feature	**Description**
	implementation logics and principles of this type of implementation compared to other approaches? What is the added value of this type of implementation? Is there scope for synergy with other types and schemes of implementation? Which? How?
Implementation requirements	Are there any contextual conditions or other requirements that must be met for an implementation of this type to be possible in principle? What are these: e.g. existence of effective ecosystem management interventions; existence of win-win opportunities for both sellers and buyers; possibilities for sustainably-produced goods and services to command a price premium?
Barriers	Which are the main barriers to implementation?
Opportunities	Which are emerging opportunities for implementation?
Risks	Which are the main implementation risks?
Project initiation	Who or what can initiate or trigger an implementation project of this type?
Actors and roles	Which are the main actors and roles involved in designing and implementing schemes of this type? E.g. project protagonist, intermediary, knowledge provider, ES provider, ES beneficiary, industry regulator
Stakeholders	Which other stakeholders might affect the implementation or be affected by it?
Implementation design process: structure and sequencing	What are the main phases, steps and decisions in the *process* of designing the implementation? What are the inputs and outcomes for each phase/step? Is this process linear, iterative, circular, a combination of these?
Implementation design process: governance	What are the governance arrangements for the process? Who are the decision makers? How is this decided?
Implementation design elements	What are the generic design elements of this type of implementation; e.g. goals of the implementation, boundaries of the implementation, eligibility criteria, a monitoring protocol, etc.? What are the associated design choices/options? What are the associated information needs?
Implementation instruments: ecosystem management interventions	Does this type of implementation specify or prescribe specific ecosystem management interventions? Which?

Table 2.2: Parameterisation of schemes (generic types)

Aspect/Feature	**Description**
Implementation instruments: statutory/regulatory	Which statutory/regulatory instruments feature in this type of implementation?
Implementation instruments: technical	Which technical implementation instruments feature in this type of implementation: e.g. standards, certificates, legal contracts, strategic plans? What are the associated design choices/options? What are the associated information needs?
Implementation instruments: financial	Which financial implementation instruments feature in this type of implementation? What are the associated design choices/options? What are the associated information needs? What are the associated design choices/options? What are the associated information needs?
Implementation instruments: risk management	Which risk management instruments or measures feature in this type of implementation?
Implementation instruments: scientific	Which scientific/informational instruments and tools feature in this type of implementation?
Transaction costs	What is the scale and structure of transaction costs associated with this type of implementation? Who bears these?
Trade offs	What are the main trade-offs between design features that impact on scheme performance? While some design and performance criteria are compatible in some schemes of implementation and can be optimised independently, others mutually conflict. This places importance on highlighting trade-offs between performance criteria that can or will arise in designing schemes and in flagging up the actual or potential implications of making specific design choices.

Table 2.3: Parameterisation of performance	
Performance aspect	**Description**
Ecological effectiveness:	This concerns the extent to which intervention through the scheme achieves intended biodiversity and habitat conservation or ecosystem service goals and secures ‘additionality’; i.e. NC/ES outcomes better than those that could have been expected in the absence of the scheme . Several aspects of scheme design are relevant to ecological effectiveness and its determination. These include how the ecological objectives and the spatial and temporal frames within which these are to be achieved are specified, how the baseline status of the ecosystem is established, how the counterfactual or ‘control’ condition is established, the forms of alternative management practices available for delivering NC/ES improvements and what is known about their effectiveness, how progress on meeting the ecological objectives is monitored (what is monitored, ease and accuracy of monitoring), the enforceability of schemes, etc. The issue of measuring ecological effectiveness is made more complex in the cases of habitat and biodiversity by their intrinsic heterogeneity and context-specificity, which implies a need to invoke criteria of ‘ecological equivalence’ as a basis for setting goals and establishing whether these are met. How well an implementation achieves its ecological objectives may depend upon avoiding perverse incentives, which can provoke perverse responses, such as ecological pressure shifting from one place to another. It may depend also on the degree to which a scheme offers scope to extend influence over ecosystem management practices extra-territorially; for example through supply chain management measures or through trading standards.
Economic Efficiency:	This concerns the extent to which the ecological improvements are achieved at least cost and is a main argument for choosing schemes that use market-based instruments (MBI) to secure that ecological protection is delivered where and by who can deliver most protection benefits at lowest cost. The scope for economic efficiency is a function, inter alia, of differences in opportunity costs of delivering ecological improvements and in the case of market-based approaches depends also on market scale, market liquidity, and transaction costs.
Cost-effectiveness and cost-efficiency:	This concerns the relation between the overall benefits of schemes and their overall costs. A scheme that is ecologically effective and economically efficient may still not be absolutely or relatively cost-effective and therefore may not be worthwhile unless the benefits outweigh the costs and risks involved. Equally, an implementation that is effective might not be the most cost-efficient. This depends on whether there are alternative schemes that offer higher ratios of benefits to costs.
Opportunity costs:	This concerns the costs – and losses of benefits – associated with no longer using the natural resource in its former (next best alternative) use. Sometimes the political costs of changing the regime of ecosystem management might be high.

Table 2.3: Parameterisation of performance	
Performance aspect	**Description**
Equity and ‘fairness’:	This concerns the ‘logic’ of the scheme (beneficiary pays, polluter pays, government pays on behalf of society), the structure and distribution of costs, benefits, and risks of schemes, and consideration of the numbers of affected parties, their status and conditions. Special concern may apply to high priority groups, such as the poor and vulnerable, whose opportunities may be limited and who may depend more highly on the concerned NC/ES.
Transaction costs:	Transaction costs are the costs incurred in making an economic exchange or of participating in a market. There is a concern for schemes to have low transaction costs, which often translates in practice to a concern to lower the transaction costs of schemes. Average transaction costs tend to reduce as a market grows and matures, but are high in the early stages of developing markets because of the high up-front fixed cost component in establishing data bases and developing customised supports for relatively few transactions. There is a high information cost component to the development of markets for ecosystem services, for example, which, initially, is a barrier to market development. There may also be legal and enforcement costs. Such barriers can be addressed gradually by streamlining market support, which becomes increasingly efficient through the development of generalizable and transferable information, tools and instruments and their systematic use as markets establish. The development and use of technology is relevant here. Technology, for example for remote sensing of forest cover, forest growth and carbon sequestration levels in relation to PES contract obligations can lower transaction costs while improving monitoring effectiveness.
Administrative costs:	Independently of cost-effectiveness considerations and of equity and fairness considerations there is a concern on the part of public authorities and tax payers over the administrative costs of schemes and the burden of administrative costs that fall to government and public authorities. This concern is linked to pressure to reduce the scale and role of government. Administrative costs include costs of scheme establishment, operation, enforcement, and monitoring. Command-and-control style policies and schemes place the burden of these costs on government and its agencies. Once the framework for a market has been established, market-based schemes translate these administrative costs into transaction costs and place the burden of these costs on those involved in market transactions.
Subsidiarity:	The concern is to ensure that decision making is devolved to the lowest level at which effective decision making can take place and that the affected parties –

Table 2.3: Parameterisation of performance	
Performance aspect	**Description**
	and only these – are involved in decision making.
Financing:	It has been established that to halt and reverse the loss of habitat and biodiversity implies a level of investment in conservation that is far greater than is available to governments, which to date have been the major funders of conservation efforts. This establishes interest in schemes that are able to leverage public funding or to attract funding from sources in the private sector. The capacity of a scheme to attract private investment into habitat and biodiversity conservation efforts might therefore be a relevant performance criterion.
Policy integration:	This concerns the extent to which schemes in support of ecological goals are able to contribute simultaneously to achieving other policy goals. Relevant policy goal combinations alongside ecological protection include: poverty relief; economic development; building the 'green economy' (e.g. by contributing to green jobs, green products and services, green income, green exports, etc.); democratisation (shifting from representative to participatory democracy and from top-down to bottom-up modes of governance); and contributing to correcting market distortions and failures (internalising external costs, public expenditure reform, providing fuller information).
Risk:	There are several different categories of risk, which may apply to schemes including political risk (e.g. that there is a change of policy) and effectiveness risk (e.g. that benefits that should be delivered are not forthcoming). In turn these are components of investment risk that may deter investors from investing in schemes. Levels of risk may be reduced using supporting instruments, such as insurance.
Credibility:	Market-based schemes, especially, must be credible across the relevant stakeholders if they and their instruments are to be accepted and trusted. Broadly-based agreement is needed among stakeholders on the principles underpinning schemes, on the assured quality of the basic science, on the reliability of data, information and tools used in ecosystem assessments, etc. in order for schemes to be credible and for stakeholders to have confidence. Credibility and confidence are jeopardised and undermined if schemes are open to hi-jack, abuse or corruption or if there is a proliferation of schemes with little or no consistency between them. This is a danger, for example, with labelling schemes, where scheme proliferation can confuse users and undermine scheme effectiveness and credibility.

Measuring and monitoring prospective or actual performance of implementations against these criteria is an important task for impact assessment. Impact assessment plays several different roles in implementation processes. One role is to support the process of designing, evaluating, and monitoring schemes of implementation. Another role is in developing counterfactuals for purposes of ex ante or ex post assessment of schemes. These and other roles for impact assessment and the available impact assessment tools are considered in Task 4.4.2 as a basis for developing impact assessment guidance. The status of this work is reported in section 3.

3. The Assessment of Ecosystem Services

3.1 Introduction

The valuation and, sometimes, economic assessment of ecosystem services is not just important for the operationalisation of the NC/ES concepts, but is indispensable for policy and implementation appraisal. This is because every implementation seeks, directly or indirectly, to change how an ecosystem is managed and thereby to change the quantity, quality and sustainability of the flow of ecosystem services and ES benefits produced. At different stages in a policy or assessment cycle, the information needs and priorities of policy makers and other stakeholders and the kinds of assessment question asked change. Fundamentally, however, addressing assessment questions depends in every case on comparing different alternative ways in which ecosystems are being, could have been or might be used, involving their delivery of different streams of ecosystem service bundles.

Whether ecosystem services are assessed *ex ante* (e.g. as a basis for setting objectives, determining how best to achieve an objective, or exploring the implications of different objective functions), or *ex post*, (e.g. as a basis for determining how well objectives are being or have been met, how well an implementation is performing, what adaptive management measures might be needed), the need is to compare different sets of ecosystem services, the value of these, and the implications of delivering these in terms of costs, benefits and the structure and distribution of these. Valuation and, where possible, the economic assessment of ecosystem services is therefore an integral element of ecosystem assessment and of policy and implementation development, design and evaluation.

In view of the central role of economic assessment of ecosystem services in implementation processes, sub-task 4.4.2 seeks to:

- Inventory and characterise tools and methods for impact and cost assessment of implementations, such as ARIES and InVEST, applied to stakeholder-derived scenarios; review experiences with applications of these to date; include recent developments, such as TEEB and BESAFE results.

- Develop and apply criteria to evaluate the potential of different tools and methods for impact and cost assessment in different implementation contexts (e.g. whether tools are open-access, versatile, have a spatial dimension, are able to account for cumulative impacts arising from combinations of different instruments, etc.)

- Identify promising tools/combinations that match to implementation contexts, stakeholder information needs, etc.

- Propose implementation-specific modifications and improvements to promising tools

The overarching aim is to develop protocols for context-sensitive selection, modification and use of impact and cost assessment tools, methods and combinations.

In order to achieve this aim there is a need to reflect on which aspects of implementation contexts are relevant for tool selection. At one level, this covers different stages in the policy cycle, different levels of governance and different legal/social/political frameworks. At another level, we can look at the different types of issue in ecosystem management, such as invasive species, pollution control, and the designation of boundaries around protected areas. We can think also of different types of 'conflict' that might characterise the assessment context; in particular, is there broad agreement on what to achieve but uncertainty about how best to do it, or disagreement on what the objectives should be?

There is a need also to characterise available tools and methods. There are broad families of methods (e.g. ecosystem models, CBA, MCA, accounting frameworks, economic impact assessment) as well as specific suites (e.g. InVEST, ARIES, Our Ecosystem, and TIM). Equally, there are different approaches to valuation. Valuation is addressed in detail already in D3.2. Our concern here, therefore, lies in explaining how different tools/methods use, or do not use, valuation methods. In characterising the available tools and methods we highlight strengths and weaknesses and aspects that make a specific tool or method more or less suitable for particular purposes and contexts, thus highlighting criteria that might be used to develop guidance for tool/method selection.

3.2 Implementation contexts

Tools and methods cannot be understood outside the context in which they are to be used – the purposes to which they will be put, the requirements and abilities of the people who will use them, and so on. In short, there are no "one size fits all" solutions, but rather horses for courses. The challenge is therefore to match impact assessment tools to implementation contexts and purposes; i.e. to select and use the most appropriate tools; those that are most 'fit-for-purpose' in the application context.

This is not so easily achieved since, just as there is no "one-size-fits-all" impact assessment methodology, neither is there a "one-size-fits-all" approach for selecting impact assessment methods and tools. The process of selecting methods and tools – including what criteria are used and the weight given to different criteria – is, also, not fixed. It, too, is context dependent. The level of accuracy demanded from an assessment required as a rough guide to help thinking is very different from that required from a legal assessment of damages that must stand up in a court of law, for example. Not only would the assessment tools be different in these different contexts, but the procedures and criteria used to select a fit-for-purpose approach would also be context dependent.

The choice of tools and methods will depend in particular on the mode of governance and the stage in the policy or assessment cycle. Methods appropriate for exploratory, idea-generation and policy framing stages are different from those appropriate to policy implementation and monitoring, for example. The choice of tools and methods will depend in particular on the mode of governance and the stage in the policy cycle. Different methods are appropriate for exploratory, idea-generation and policy framing stages, from the methods appropriate to policy implementation and monitoring, for example.

Primmer et al. (2014) define policy processes as cyclic and iterative, first designed and then negotiated, developed, implemented, and evaluated. Different phases can be distinguished as presented in Figure 3.1

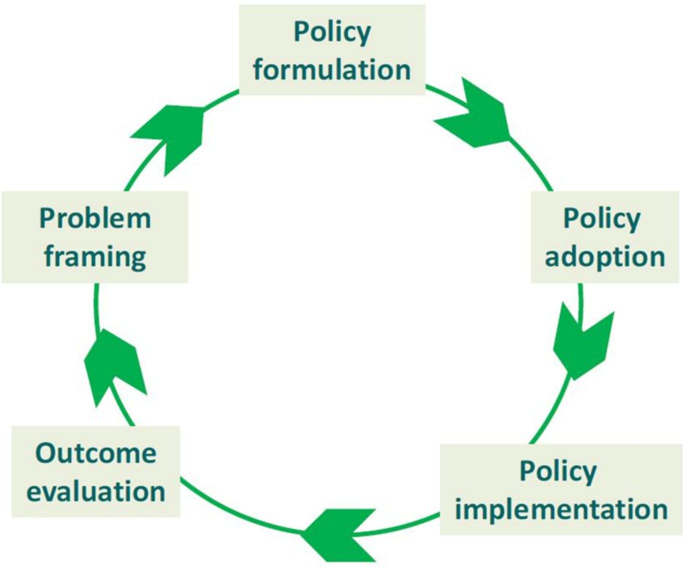

Figure 3.1: Phases in the policy cycle (Primmer et al., 2015)

Similarly, the social, economic and governance context will influence what will be appropriate. Primmer et al. (2015) distinguish four basic governance modes (Figure 3.2):

- Hierarchical governance: focus on agreed policies or decisions and their implementation; possibility of manipulation from powerful sectors /actors; need effectiveness of the policy and arguments to transfer ideas from high policy level to lower ones; obstacle: national/sector interests, technical argumentation.
- Scientific-technical governance: focus on scientific knowledge: focus on a specific conservation issue or on a specific sector; policy and scientific arguments are supposed to be operationalized and simplified, and then placed in the local context; issues for implementation can occur in case of arguments preceding the policy

- Adaptive collaborative governance: focus on learning, commitment and communication; knowledge accumulation; collective learning, sensitivity to changes; adaptive governance of social–ecological systems; do not take hierarchical governance as a starting point (bottom up governance); do not always assess the conservation outcomes (sustainable governance outcomes instead); include stakeholders; participatory process.

- Governing strategic behaviour: actors using ecosystems for economic purposes will be willing to secure their own interests; policies can be seen as mandates/guidance, or as barriers; policies influenced by actors.

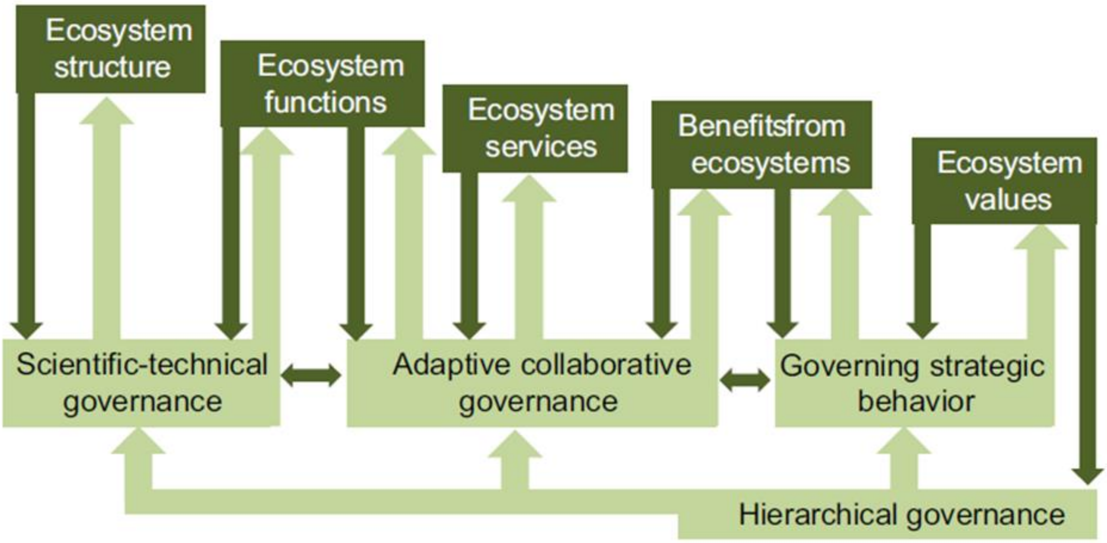

Figure 3.2: Framework for analysing governance of ecosystem services (Primmer et al., 2015)

An example of what it would mean to focus on each of these governance types can be given in the case of Natura 2000 network:

- Hierarchical governance: the nation states or regional/local administration are in charge of the implementation of the protected areas, with a focus on legal and administrative arguments.
- Scientific-technical governance: uses scientific arguments to justify and put in place the Natura 2000 network (based on data and knowledge).
- Adaptive collaborative governance: exchange of information between actors for the implementation and use of protected areas (based on different types of arguments).
- Governing strategic behaviour: focus on other interests and land-uses conflicting with the Natura 2000 protected areas (based on benefit/value related arguments).

From this example it is clear that governance types are not fixed or totally separate, but rather that governance type can vary over a policy cycle and, at different stages in the cycle, each might be present but to a different degree. In any case, therefore, it is a matter of the relative emphasis that needs to be taken into account in the matching process. Very often for an analysis of a policy instrument design, we start with a design at a central level, such as the state. Scientific-based tool and collaboration will be used for its implementation. Strategic behaviours are then taken into account.

3.2.1 Policy needs for operationalising ecosystem services

The dependence on implementation contexts underpins the policy needs and opportunities for operationalising the concept of ecosystem services.

Firstly, in respect to the policy context, OPERAs D4.1 has identified the main EU policy areas of interest:

- environmental policies (air, soil and water),
- policies related to the management of natural resources (agriculture and rural development, fisheries and marine areas and forest),
- policies having known impacts on nature and natural resources (regional development, climate, bioenergy and transport).

The contexts here are broadly similar, though a clear distinction can be made between policies that are clearly the remit of 'environmental' DGs (ENV and CLIMA) and ministries, and others that fall under the banner of "mainstreaming". At the EU level, particular policy priorities include the Europe 2020 strategy and overall emphasis on growth and employment, and policy needs are shaped by the European Treaties and in particular the principle of subsidiarity, whereas national priorities are more variable and subject to different historical features and relationships in the governance of particular areas, industries and systems.

Secondly, three main policy contexts have been targeted:

- information
- decision-support
- implementation.

These are closely related to the policy cycle stages, though the policy cycle framing highlights the cyclical nature of policy, with the monitoring and evaluation phases feeding back to new rounds of deliberation and policy formulation.

Finally, two degrees of integration have been highlighted: conceptual integration and integration into policy implementation, associated to four levels (comprehensive and explicit, explicit but not comprehensive, implicit and incomprehensive, no specific integration). The results about the current (inconsistent) level of integration of ecosystem services and natural capital into different EU policy sectors are presented in Table 3.1.

To improve this integration the three instruments listed above (information, decision support, and implementation) are very useful. For each policy sector a list of needs and opportunities and the explanation of how the needs can be met (through the policy instruments) have been explained. The results clearly show the different concerns for each sector and the different use a same instrument can provide according to the considered policy sectors. More details on this point can be found in D4.1.

Table 3.1: Level of integration of ecosystem services and natural capital into EU policy areas (sources: OPERAs, D4.1)

Policy sectors	***Conceptual integration***	***Integration into policy implementation***
Air	*Limited (implicit and incomprehensive*	*Limited (implicit and incomprehensive)*
Soil	*Explicit and comprehensive*	*Limited (implicit and incomprehensive)*
Water	*Explicit and comprehensive*	*Explicit and incomprehensive*
Agriculture and rural development	*Explicit and incomprehensive*	*Explicit and incomprehensive*
Fisheries	*Explicit and comprehensive*	*Explicit and incomprehensive*
Marine	*Explicit and comprehensive*	*Explicit and incomprehensive*
Forest	*Explicit and comprehensive*	*Limited (implicit and incomprehensive)*
Regional development	*Explicit and comprehensive*	*Explicit and incomprehensive*
Climate	*Explicit and incomprehensive*	*Explicit and incomprehensive*
Bioenergy	*Explicit and incomprehensive*	*Limited (implicit and incomprehensive)*
Transport	*Limited (no specific integration)*	*Limited (implicit and incomprehensive)*

3.3 Economic analysis

Economics is "the science which studies human behavior as a relationship between ends and scarce means which have alternative uses" (Robbins, 1935:15). This is often paraphrased as "scarcity implies choice": since it is not possible to achieve all objectives simultaneously, trade-off is inevitable, though it may be either implicit or explicit (Costanza *et al*., 2011). Where there are alternative uses of resources, economics seeks to evaluate the impact of resource allocation decisions on human populations, and to assess trade-offs in explicit terms (Farber et al., 2002). This is challenging both because resource allocation decisions impact on the welfare of the human population in multidimensional and complex ways, and because measuring and projecting these changes is often difficult.

Applied economics is an activity in which value judgements have to be made: any analysis has to choose "what matters?", "to whom?", "over what period?" and "how is this measured?" Economics embraces different analytical frameworks, each shaped by underlying value judgements that define the conceptual and spatial boundaries of an assessment and the appropriate indicators and measurements. We may consider impacts on incomes, employment, the enjoyment of the environment, the value of capital assets and so on. We can also describe the impact of resource allocation initiatives in terms of the consequences for particular groups of humans, including populations of identifiable geographical areas at different scales and/or specific 'communities' of interest. Similarly, assessments may focus more on present day and near future interests and impacts, or may take account of the long term impacts on future generations. The application of each analytical framework may take a partial equilibrium approach, focusing on just one market, or a more complex general-equilibrium approach that recognises inter-sectoral interaction. Analyses can be comparative-static, focused on equilibrium states, or dynamic, allowing for different rates of adjustment and feedbacks.

For economic evaluations to inform policy and debate in a useful way, the value judgements underpinning each analytical framework need to match the value judgements, boundaries and concerns of policy makers and stakeholders at different stages of the policy cycle. An appreciation of the different frameworks and associated value judgements is crucial to a better understanding of which measures can be logically compared or added together.

Trade-off is central both to ecosystem service valuation and to environmental management. Elmqvist et al. (2011) for example describe the relationship between provisioning and regulating ecosystem services, noting that in many situations provisioning services giving products directly traded in markets, such as food or timber, are prioritized compared to regulating services, such as soil quality, pollination and water quality. However, these regulating services underpin both provisioning services and human welfare at broader spatial and longer temporal scales. Often, a moderate increase of provisioning services production can be associated with a substantial decrease of regulating services, with long term consequences.

Effective management in primary production sectors often seeks to modify this relationship by selecting levels and modes of production of provisioning services that have less negative effects on other types of services. This can be understood as seeking trade-offs that recognise the value of regulating services. Burkhard et al (2013) argue that, while multiple indicators can be used for assessing trade-offs across different ecosystem services, the analysis can be easier if these are standardised to a common indicator of economic value. This is the basis of cost-benefit analysis and most valuation effort is expended in the context of policy and project appraisal.

Nevertheless, there can be limits on the applicability of trade-off, due to non-linear relationships between variables: beyond areas of smooth trade-off, management may need to respect constraints associated with critical natural capital, avoiding thresholds and hysteresis effects, and here conventional monetary valuation may be less directly useful. Economic valuation methods are mostly used to estimate marginal values for changes in service provision in a static and partial equilibrium setting. But this approach has some weaknesses, in particular regarding spatial interactions and cumulative impacts. These create the widely-recognised problem that independent valuation and summation of impacts of multiple projects can lead to a situation in which "too many proposals pass the benefit cost test" (Hoehn and Randall, 1989). With greater computing power, advancing understanding of human-environment interactions, and more environmental and economic data, there is increasing interest in integrating spatial interactions in analysis and appraisal, and in recognising indirect and induced economic effects.

In order to develop an economic analysis, we must first address and answer some fundamental questions about the intent of the analysis: what are we comparing, e.g. frameworks, notion of ecosystems and ecosystem services and their impact on the human system; in which context, e.g. scenario comparison; at which scale; using which tools and assumptions? Sections 3.3.1 – 3.3.4 aim at answering these questions.

3.3.1 Frameworks for understanding services

Applying valuation and mapping methods for policy support and appraisal requires some framework for classifying and measuring ecosystems and their services. Figures 3.3, 3.4 and 3.5 set out those developed and used by MEA, TEEB and IPBES respectively.

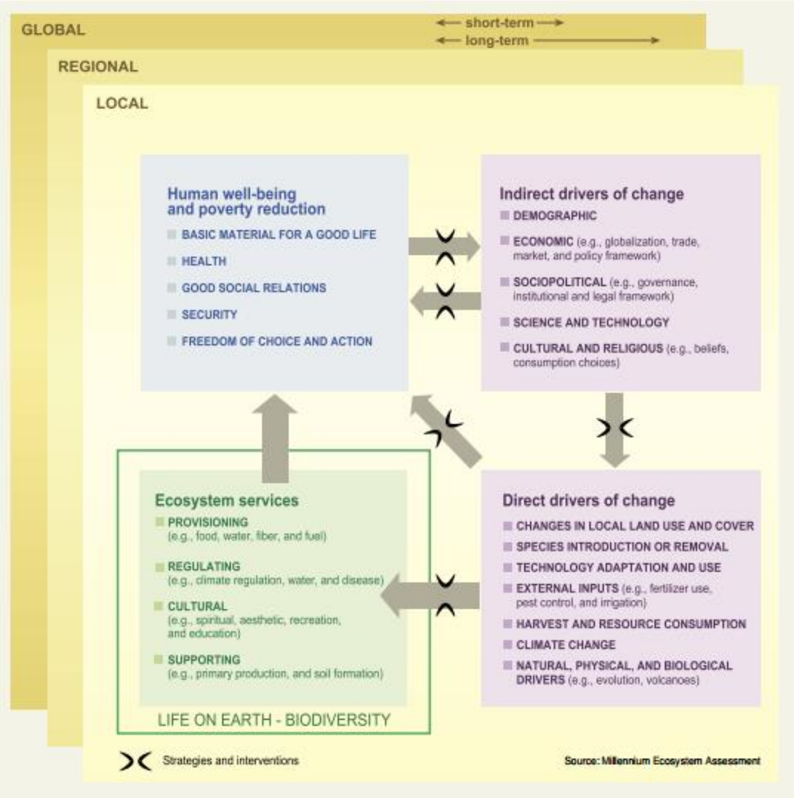

Figure 3.3: Millennium Ecosystem Assessment conceptual framework (Source: Duraiappah et al. (2005:iii))

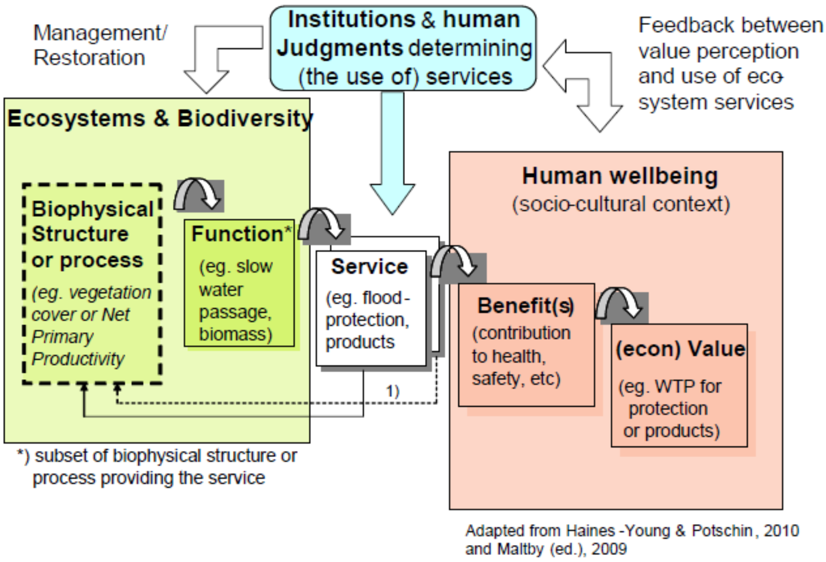

*Figure 3.4: TEEB classification (*Source: de Groot *et al*., 2010:11)

The Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) presents a broad conceptual framework that attempts to combine many different approaches to conceptualising the human-environment relationships.

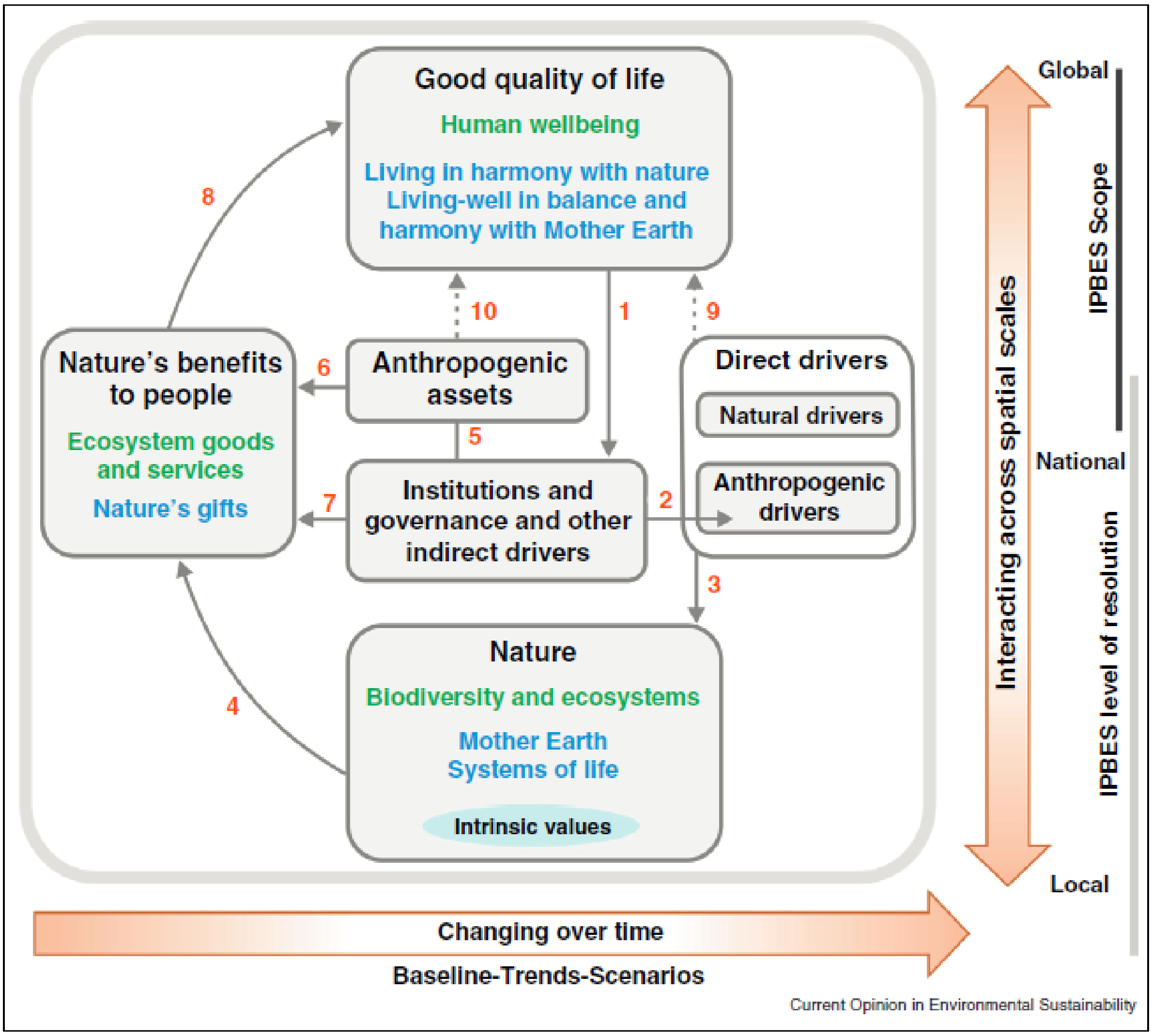

Figure 3.5: IPBES conceptual framework (Source: Diaz et al. (2015:3))

Practical frameworks for cost-benefit analysis and trade-off analysis more generally are more focused and reductionist. The MAES framework, for example, locates 'biodiversity' as the central feature of ecosystems, and then breaks it down further to a number of component parts which support ecosystem functioning and in some cases (including species and genetic diversity - the right wing of butterfly in Figure 3.6) also provide direct ecosystem services.

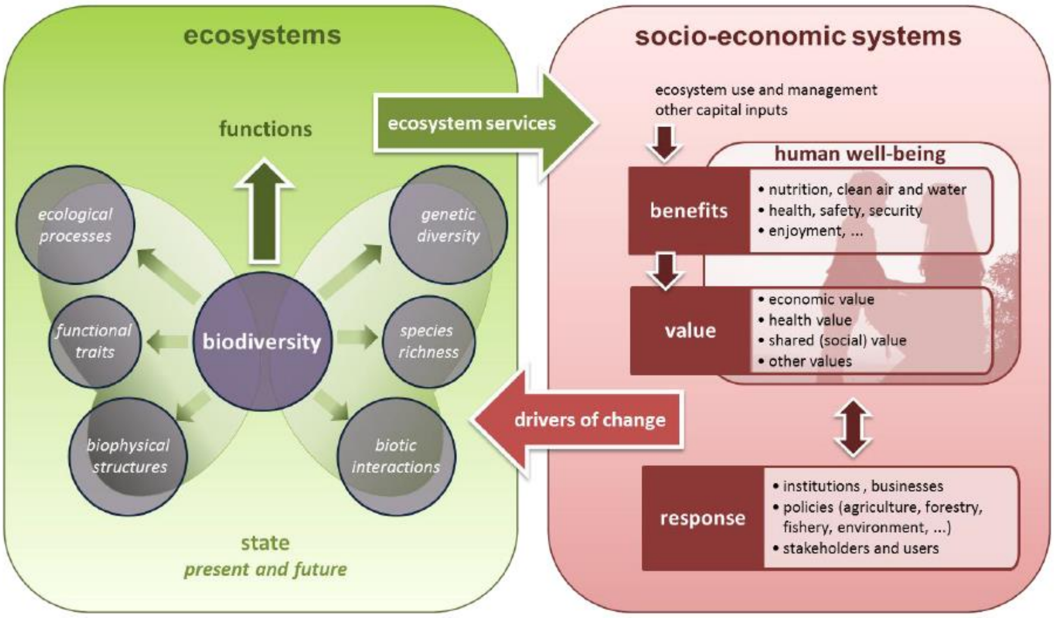

Figure 3.6: EU framework for ecosystem assessment (source: European Commission, 2013:17)

The Common International Classification of Ecosystem Services (CICES) goes further in targeting appraisal, by excluding biodiversity and supporting services[2]. The report (Haines-Young and Potschin, 2013) stresses a clear distinction between final ecosystem services and ecosystem goods or products:

- Human well-being arises from adequate access to the basic materials, freedom of choice and action, health, good social relations and security. This is partly dependent on access to ecosystem goods and benefits.
- Ecosystem goods and benefits are created or derived from final ecosystem services by humans. These products and experiences "are no longer functionally connected to the systems from which they were derived."
- Final ecosystem services, in contrast, retain a direct connection to the underlying ecosystem functions, processes and structures that generate them. They are 'final' as the outputs of ecosystems that most directly affect human well-being. CICES is a classification at this level; i.e. services, not benefits.
- Intermediate and supporting services are functions and processes that underpin the final services. They are not directly included in CICES because they are only indirectly consumed or used, and may simultaneously facilitate the output of many 'final outputs'.

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[2] See CICES V4.3; 17/3/15 (http://cices.eu/).

The exclusion of supporting services from CICES is not intended to suggest that they are unimportant. Rather, the rationale is directly connected to accounting: “if ecosystem and economic accounts are to be linked, then an essential step is to identify and describe the ‘final outputs’ from ecosystems that people use and value, so as to avoid the problem of double-counting” (Haines-Young and Potschin, 2013 p8). Though at the same time “there is no reason why fully developed environmental and economic accounts cannot also record changes in underlying ecological structures, processes and functions, and systems like CICES may well be extended to cover them” (ibid, p8) – but in physical terms, not monetary, to avoid double counting. As such, CICES is intended to provide a framework focused on final services within which information about supporting or intermediate services can be nested and referenced. Haines-Young and Potschin (ibid) argue that such treatment may be especially useful for mapping ecosystem services and propose that “CICES should be explored through the development of experimental accounts, especially in the context of using accounts to check the integrity of underlying ecological assets” (ibid, p8).

The US has worked in parallel to CICES on the FEGS-CS (Final Ecosystem Goods and Services Classification System). This likewise focuses on final goods and services, but with an emphasis on classifying both service and beneficiary together. Figure 3.7 illustrates how a six-digit classification code is built up.

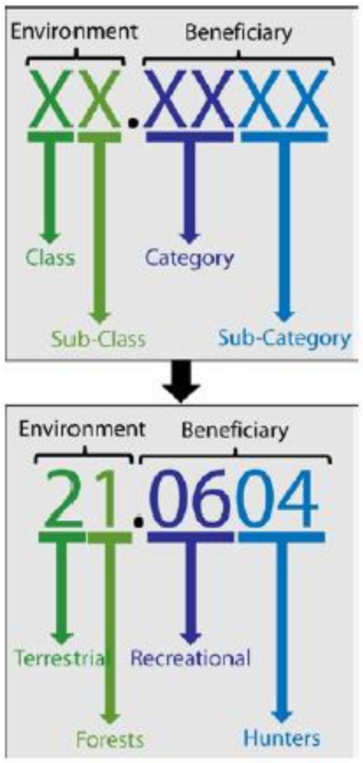

Figure 3.7: Final Ecosystem Goods and Services Classification System (FEGS) basic structure (source: Landers & Nahlik (2013:7))

3.3.2 Defining the baseline

Any form of economic appraisal involves the comparison of different “states of the world” – the state of the world under ‘baseline’ or ‘counterfactual’ conditions (i.e. without the change(s) that are the subject of appraisal), and one or more states of the world with the change(s) or intervention(s) that lead(s) to different outcomes. This can apply to comparisons of total values of whole sectors (though such comparisons do not usually correspond to realistic policy options) as well as to rather more realistic assessments of marginal changes in resource use and access.

When considering comparisons across different uses of resources, one essential issue is to compare like with like. Establishing a consistent and appropriate baseline against which comparisons can be made is therefore a key step in providing an accurate assessment of changes in resource allocation.

The baseline has a variety of slightly different definitions, depending on the context in which it is used. HM Treasury in the Magenta Book defines the baseline as “the situation before the policy” is implemented (HM Treasury, 2011). The European Commission (EC, 2009) states that the aim of the baseline is “to explain how the current situation would evolve without additional public intervention – it is the ‘no policy change’ scenario” (EC, 2009). Defra (2010) focuses on the “change in the provision of the policy good” which is the difference between the level of provision without the decision being appraised (the ‘baseline’) and the level of provision with the project or policy. This change can be measured as a quantity change (e.g. an increase in fish catch) and/or a quality change (e.g. an improvement in average fish size), and may be described qualitatively or measured quantitatively. Whilst the three baselines appear to be similar, the EC and Defra are dynamic baselines which suggest that the economic evaluation needs to predict what would happen in the absence of the policy initiative. Thus the economic practitioner might have to predict what the future would look like both with and without the policy. Since predicting the future is problematic, economic evaluations may develop a number of policy impact scenarios, such as optimistic and pessimistic outcomes. In some circumstances it may also be appropriate to also develop a number of dynamic baseline scenarios, and the simultaneous use of both baseline and policy impact scenarios can lead to quite complex evaluations.

A good baseline should have a strong factual basis and, as far as possible, be expressed in quantitative terms. It should also be set for an appropriate time horizon (neither too long to be practicable nor too short to cover all relevant impacts). The baseline projection has to provide a clear indication of how serious the problem is, or to what extent it would become more serious without immediate intervention, and whether there are irreversible consequences (EC, 2009).

However, the choice of counterfactual is not always clear-cut, and under some circumstances, prediction of the baseline scenario(s) can be as crucial and uncertain as the policy impact prediction(s). Changing conditions (such as social, economic, technological and climate changes) mean that the counterfactual is not simply a static 'status quo' scenario. Indeed, the choice of comparison case may depend on the specific question to be answered. It is not necessarily the 'most likely' alternative scenario in the absence of a specific policy intervention (though it often will be) and can in some cases be more of a 'baseline' than a realistic counterfactual. Many options can be identified: and some are easier to define and measure than others, and data requirements differ.

- "No activity": this may be appropriate for estimating the current total value of a sector. The counterfactual is a hypothetical situation where the entire sector instantaneously ceases to exist. Of course, this is almost always a quite unrealistic scenario, except at very local scales. In fact the question "*what is the total value of XXX?*" may be considered largely irrelevant for policy making. Economic evaluations are better able to inform current policy debates when they focus on relative or marginal changes rather than absolute values: the key indicator of value, price, reflects market equilibrium conditions, and can be very different, much higher and difficult to estimate for hypothetical very low levels of activity in a sector. However, because of the impressively large values often generated for, say, the number of jobs or profits currently supported by the sector, such figures are often used by stakeholders in advocating for more resources.

- "Status quo": this represents the most recent possible historical baseline, and one with substantial policy relevance, because policy options involve changes from current practices. Its main strength is that, in principle anyway, it can be directly measured. However it may be too static, ignoring technological and other changes and trends. Other historical baselines may be used (e.g. pre-industrial or pristine conditions) though these are generally used for environmental rather than economic concerns.

- "Business as usual": similar to "status quo", but a dynamic counterfactual, taking into account our best estimates of the likely evolution of activities in response to key drivers such as technological and climate change.

- "No active policy intervention": this is a baseline for policy analysis that considers hypothetical no-active-policy-intervention-from-now conditions. This does not imply "no activities", but rather an absence of active management. It can be hard to determine how activities would evolve in the absence of management interventions, though for unowned resources such as sea fisheries there is a good understanding of the consequences of completely open access.

In any particular case, additional considerations arise concerning the determination of system boundaries in space and time – essentially, all changes between the counterfactual and the scenario under analysis need to be taken into account, and we need guidelines to ensure the boundaries are set appropriately to allow for this.

This does not necessarily mean that the area under analysis should be extended to encompass all impacts. A local authority or tourist board, for example, may be entirely justified in limiting attention to impacts that occur within their areas of competence. A national authority might carry out assessment of the same change with different boundaries, leading to a different result. Again, the point is not that either approach is better than the other, but rather that different approaches are appropriate for different questions. Hence, when considering different analyses, it is important to check that like is being compared with like.

Similarly, the time horizon needs to be set in a manner appropriate to the questions at hand. Generally, decision makers are concerned with more than just the immediate impacts of decisions, and so assessment of resource allocation decisions with long-term or irreversible consequence call for long time horizons in the assessment. Questions relating to more easily reversible decisions, or comparisons of current flows of value, may reasonably focus on immediate or short-term impacts.

In economic appraisal, the use of discounting makes costs and benefits far in the future much less important than present costs and benefits. There is some debate concerning the appropriate use of discounting for ecosystem services, in particular for the far future; hyperbolic discounting (that is discounting, but at a declining rate) has been proposed. In the UK this is the official approach, with the discount rate dropping from 3.5% in years 1 to 30, to 3% in years 31 to 75 and 2.5% in years 76-125 (HM Treasury, 2003).

With a dynamic counterfactual, we need to account not only for current services and changes to them, but also future potential services and changes to them. For example, an area currently little-used for recreation may nonetheless have substantial future recreation value potential, if one or more of the following occur:

- Infrastructure is improved;
- Alternative recreation sites deteriorate;
- Site characteristics change;
- Local human population or population characteristics change, and
- Climate changes.

A study which (say) took into account the recreation improvements arising from the policy proposal, but failed to take into account possible recreation improvements in the baseline (that would happen anyway even without the policy), would risk overstating the benefits of the policy proposal. Thus a baseline should be defined in terms of how all the relevant factors are likely to be / change in the baseline, including but not limited to environmental, economic and social.

Unfortunately, some EIA do not explore baselines directly, but rather draw on estimates of expenditures without exploring the counterfactual. Where some part of the assumed changes would have occurred anyway, without the policy or intervention under consideration, this is termed “deadweight”, and failure to account for it can result in values being overstated. A particular issue for the fisheries sector is baseline assumptions regarding stock dynamics: if, in the baseline, it is simply assumed that harvesting is sustainable, values of the baseline may be overestimated if, in fact, stocks are being depleted. This would result in underestimation of the value of stock-conserving policies (because a policy intervention which lowered harvesting levels would be even more imperative than anticipated by the baseline). Full consideration of counterfactuals can help to account for these potentially complex dynamic effects.

3.3.3 Spatially-explicit assessments of ecosystem services

Mapping ecosystems and their services is one approach to improving the assessment of trade-offs and synergies among ecosystem services and between ecosystem services and biodiversity. It can be an efficient communication tool helping stakeholders and the general public to understand the ecosystems that exist in a given territory and how they affect human well-being. Mapping can make some contribution to the design and implementation of biodiversity protection policy, and can inform impact assessment and policy appraisal more generally, especially when combined with monetary valuation techniques.

Action 5 of the EU Biodiversity Strategy to 2020 requires that “Member States, with the assistance of the Commission, will map and assess the state of ecosystems and their services in their national territory by 2014, assess the economic value of such services, and promote the integration of these values into accounting and reporting systems at EU and national level by 2020”. To meet these requirements, the European Commission gathered a working group on Mapping and Assessment of Ecosystem Services (MAES). MAES also contributes to IPBES[3] regional assessments of ecosystems and ecosystem services, and to the development of natural capital accounts. The main challenges regarding the efficient development of MAES are about the operationalisation of ecosystem services (gather all available data, find the best way to use it, make it available for Member States, associate on-going European, national and sub-national assessments and future assessments), defining the link between ecosystems, ecosystem services and biodiversity and valuation (European Commission, 2013).

[3] http://www.ipbes.net/

There is growing academic interest in mapping services. Schägner et al (2013) analysed 79 separate case studies mapping ecosystem service values, dividing the mapping task into service estimation and service valuation steps. They report that most used simple one-dimensional proxies (generally LULC) relating habitat to service, though several used validated or unvalidated models of causal links to ecosystem service supply. A few economics-based studies modelled services only implicitly via value transfer functions. Costanza et al. (2014) explain that for valuation, there is a continuum of approaches to ecosystem service values (Kubiszewski et al (2013a, b) from basic value transfer, through expert-modified values, statistical transfer functions, and meta-analytical functions (Nelson and Kennedy 2009). Schägner et al (2013) found 78% of studies found in the literature used a basic unit value transfer approach. Features such as temporal variation and uncertainty are rarely considered (Pagella and Sinclair, 2014). Most studies focus on ecosystem service supply, though demand features are sometimes considered (Burkhard et al 2013; Pagella & Sinclair 2014). Spatially-explicit functional modelling represents the 'cutting edge' in environmental valuation (Costanza et al. 2014), but it is also the most resource intensive and technically demanding option.

Few studies assess ecosystem services at the global scale: Schägner et al (2013) reported five examples, while Pagella and Sinclair (2014) report only three examples of mapping at international scale. Detailed spatial modelling of values is beyond current techniques: existing global estimates, most (in)famously Costanza et al (1997) and most recently the update by Costanza et al (2014), adopt a form of benefit transfer that assumes a constant unit value per hectare of ecosystem type to arrive at aggregate totals. For the 2014 paper, they draw on de Groot et al. (2012) who estimated the monetary value of ecosystem services provided by 10 main biomes, via meta-analysis of 665 data points selected from the Ecosystem Services Value Database[4]. There is increasing interest in larger-scale mapping of ecosystem service values, with a wide variety of techniques (Crossman *et al*., 2013), and in analysing the macroeconomic implications of changes in ecosystem goods and services, including the UKNEA-FO scoping study (Anger *et al*., 2014).

The notion of scale and spatial assessment will be further developed below with modelling tools applied at the global scale (such as GUMBO or MIMES), or national/regional scales (such as INVEST, ARIES, SERVES, POLYSCAPE, TESSA, or TIM).

[4] http://www.fsd.nl/esp/80763/5/0/50

3.3.4 Tools for modelling ecosystem services values

3.3.4.1 Global scale

Non-spatial attempts to model ecosystem service values directly at global scale include the global unified metamodel of the biosphere (GUMBO) simulation meta-model (Boumans et al. 2002). This is a metamodel and thus includes several global unified meta-models of the biosphere. Several future scenarios based on different assumptions about future technological change or investment strategies have been defined. The economic component combines production of ecosystem goods and services with economic production based on stocks of social capital, knowledge, labour force and built capital, and split between personal consumption and savings rates for the main capital stocks; waste is modelled as a negative feedback. The marginal product of ecosystem services can be estimated in both the production and welfare functions, taking account of interdependencies and dynamics of economic and environmental systems at global scale. Ecosystem services have been valued under each scenario via their impact on economic production and human well-being. GUMBO estimates the value of ecosystem services as 4.5 times the value of Gross World Product in 2000 (Boumans et al. 2002).

MIMES (Multi-scale Integrated Model of Ecosystem Services, Boumans and Costanza, 2007) seeks to extend the GUMBO approach to allow spatially explicit modelling at various scales, but is highly complex and not applied at global scales. This model especially takes into account stakeholders' contributions and biophysical data to assess ecosystem services and help for decision-making. It has three main aims: improve our understanding of ecosystem functioning, ecosystem services and human well-being at different scales, develop new ecosystem services valuation techniques, and communicate on integrated models and their results. Within MIMES 16 ecosystem services are considered and classified according to their scales: carbon sequestration, carbon storage, existence of "nature", storm protection, waste treatment, pollination, water supply, flood protection, nutrient regulation, sediment regulation, rangeland of livestock, nitrogen mineralization for agricultural production, soil formation, raw material, non-timber forest products, aesthetic/recreation potential (Boumans & Costanza, 2007).

3.3.4.2 National, regional and local scales

For smaller scales, various modelling suites exist, in particular InVEST[5] (Integrated Valuation of Ecosystem Services and Trade-offs, Nelson et al., 2009a) which includes a number of modules related to specific ecosystem functions and services but demands detailed local modelling, including a full hydrological map. Different scenarios of land, water and marine uses are defined to observe and assess ecosystem services. Different kinds of outputs are produced which enables to describe the ecosystem services in terms of their role

[5] http://www.naturalcapitalproject.org/models/models.html

in biophysical and socio-economical processes and for human well-being. There are 17 models that can be used in the InVEST suite: blue carbon, carbon, coastal protection, coastal vulnerability, crop pollination, habitat quality, habitat risk assessment, managed timber production, marine fish aquaculture, marine water quality, offshore wind energy, recreation, reservoir hydropower production, scenic quality, sediment retention, water purification and wave energy.[6]

ARIES[7] (Artificial Intelligence for Ecosystem Services, Villa et al., 2014) seeks to make service evaluation less demanding via artificial intelligence (integrated models) and Bayesian networks. It also provides information and guidance for ecosystem service assessment and valuation to help the process of decision-making. The information provided by ARIES concerns location-specific environmental assets (description, explanation, quantification, valuation) and relies on people's needs and priorities. It provides maps of the sources, uses and sinks of ecosystem services. Agent-based flow algorithms are then used to map the ecosystem services flow to people. ARIES relies on a benefit-based approach to ecosystem services. This tool has been used for several purposes and scales: spatial mapping and quantification of ecosystem services (local and national), spatial economic valuation of ecosystem services (local and national), optimization of payment schemes for ecosystem services (regional), conservation planning (local and regional), spatial policy planning (local and regional), forecasting of change in ecosystem service provision (local and regional). ARIES has been developed and applied within 9 case studies[8].

The SERVES[9] toolkit (Simple and Effective Resource for Valuing Ecosystem Services) uses value transfer to provide high and low values by ecosystem service and land cover type. It is considered as a tool for natural capital financing purposes. A standard economic format is used to be able to include the natural capital within the economic planning. SERVES has been used for directing federal disaster recovery and mitigation funding, developing natural capital financing mechanisms, reforming accounting policy, calculating environmental damages for use in legal settlements, and providing a comprehensive overview of existing research and exposing critical knowledge gaps. However this tool is not freely available: the SERVES team offer contractual custom report design for CBA, EIA, and natural capital accounting.

POLYSCAPE[10] (Jackson et al. 2013b,) explores spatially explicit synergies and trade-offs amongst ecosystem services up to catchment scales, using MCA methods rather than economic values. It includes several land management options at local (field and farm) and larger landscape (especially for regulating services) scales and is considered as an

[6] Data requirements and outputs for each one of these models are specified at http://data.naturalcapitalproject.org/invest-releases/documentation/current_release/data_requirements.html
[7] http://www.ariesonline.org/
[8] http://www.ariesonline.org/case_studies.html
[9] http://esvaluation.org/values-reporting
[10] http://www.polyscape.org/

information/negotiation tool to stakeholders and land users for their management decisions and help implementing effective agri-environmental policies: it is a participative tool based on local knowledge from farmers (Pagella & Jackson, 2012). The LUCI framework has been developed as an extension of POLYSCAPE, based on a farmer-scientist partnership to define actions to improve economics and reduce environmental impacts. According to Jackson et al. (2013a), 7 services are modelled within LUCI (production, carbon, flooding, erosion, sediment delivery, water quality and habitat) as well as trade-offs and synergies. LUCI has for example been applied in the UK to highlight the best location for agri-environment measures to improve carbon, water flow and quality, biodiversity but in the meantime aiming at enhancing the productivity. It is not yet freely available though there are plans to release initial models for some areas in 2016.

TESSA[11] (Toolkit for Ecosystem Service Site-Based Assessment, Peh et al. 2013) aims at local non-specialist users, adapting accessible methods for identifying ecosystem services and evaluating benefits, comparing current and potential land-uses. It can be used as a guidance about highlighting significant services in a given site, the requested data for the assessment, the data sources, the provision of outcomes.

Co$ting nature focuses on water yield, carbon storage, nature-based tourism, and natural hazard mitigation services and assesses their values. The results are summarized in a "service index" for provision and beneficiary location.

A spatially explicit approach to modelling and optimising land use for ecosystem services is taken by Bateman et al (2014) in the UK NEA-FO 'TIM' (The Integrated Model; Figure 3.8). TIM is an "integrated, modular, optimizing approach" to multiple land uses. This approach represents a major advance on more traditional partial equilibrium analysis and standard appraisal methods, that look at alternative uses of resources at a specific place, but do not fully account for substitute sites and other spatial interactions. In TIM, biodiversity is represented by various models/indicators of biodiversity (including the farmland bird index, red list species, red/amber list species, woodland species, and total number of species) that are implemented as spatial constraints on land-use in an optimisation model; i.e. the indices must not decline within each grid cell in the model. This does not directly use any valuation. However, as a consequence of the analysis, it is possible to work out the (opportunity) cost of imposing the biodiversity constraint in the optimisation process; i.e. the possible gain in other services from marginally relaxing the constraint, and this could be interpreted and used as a 'value'. TIM observes the consequences of a desired change in multiple drivers. Both direct and indirect impacts are taken into account and they are assessed in quantitative terms and all except for biodiversity are measured in terms of economic values. The main advantage of TIM is the possibility to identify the optimal way in which to implement a policy change, through the analysis of every component modules simultaneously to examine the consequences of any specified change at any location and at any time over a specified

[11] http://tessa.tools/

period. The 'spatial targeting' approach to decision making is highlighted within TIM analysis, which allocates scarce resources to those locations which maximise the specified objective. This method thus does not specify pre-set end points through a conventional scenario analysis. TIM especially take into account the natural world into economic decision making.

Figure 3.8 Structure of TIM (source: UK NEA-FO)

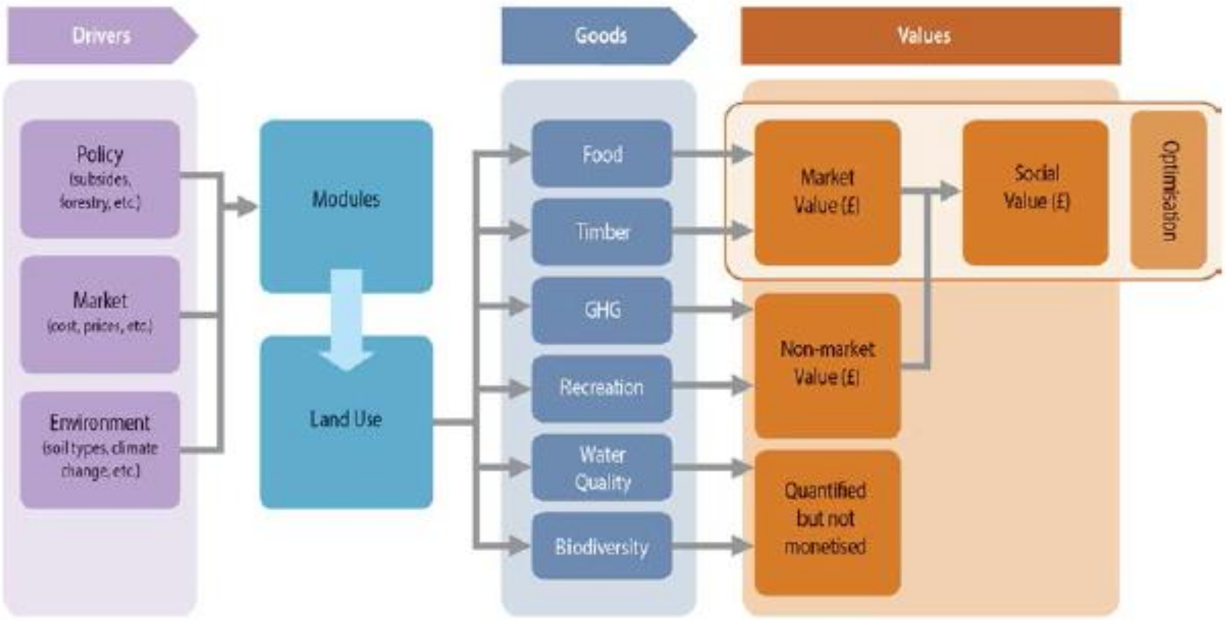

Table 3.2: Comparison of models (source: UKNEA-FO)						
Tool/methods	***Model***	***Timescale***	***Geographic scale***	***Costs/benefit economic valuation***	***Trade-offs***	***Reference***
GUMBO		*Historical calibrations: 1900-2000. Run until 2100.*	*Global*			*Boumans et al. 2002*
MIMES	*Detailed physics & integration of environmental, economic and social drivers*		*All scales: local/landscape, regional, national, global*	*Sometimes*	*Economic valuation of services and analysis of their interactions*	*Boumans & Costanza, 2007*
InVEST	*Detailed biophysical models and economic valuation*		*Landscape/regional/national*	*Yes*	*Biopysical and monetary units traded against each other*	*Nelson et al. 2009a*
ARIES	*Bayesian and agent based*		*Spatially specific/ local/landscape/regional/ national*	*No*	*Biophysical and analysis of service flow from provision to beneficiaries*	*Villa et al., 2014*

Table 3.2: Comparison of models (source: UKNEA-FO)						
Tool/methods	***Model***	***Timescale***	***Geographic scale***	***Costs/benefit economic valuation***	***Trade-offs***	***Reference***
SERVES			*All scales but spatially specific*			*Earth Economics*
POLYSCAPE LUCI	*Simplified biophysical models*		*Local/regional/national*	*No*	*Biophysical*	*Jackson et al. 2013b*
TESSA			*Local/landscape*			*Peh et al. 2013*
TIM	*Biophysical modules with robust economic valuation and formal optimisation*		*Medium catchment to national*	*Yes*	*Trade-offs analysed by explicit economic valuation of all services*	*Bateman et al (2014)*
Costing Nature	*Web-enabled model*		*Global coverage*	*No*	*Services categorised and flow to beneficiaries considered*	*Mulligan et al. (2010)*

3.4 Methods for economic analysis

3.4.1 Welfare-based methods

The Total Economic Value (TEV) framework is based on working out how individuals are willing to trade-off between resources. The estimation of economic value is usually based on ‘willingness to pay’ (WTP), a monetary expression of how individuals are willing to trade-off across different goods and services. In practical cases, it is also necessary to aggregate these values across individuals, to provide a monetary measure of society's preferences for alternative uses of its scarce resources. In effect, social preferences are taken as the aggregate of the individual preferences of members of society, to provide a monetary measure of society's preferences for alternative uses of its scarce resources. In effect, social preferences are taken as the aggregate of the individuals’ preferences. So this approach involves five main value judgements or assumptions:

- What “matters” are the preferences of individuals in society;
- Individual are the best judges of their own welfare and preferences;
- Individuals express preferences through rational economic choices via their ‘willingness to pay’ (WTP) or ‘willingness to accept compensation’ (WTA) for goods and services;
- Since WTP is constrained by ability to pay (wealth and income), the method in effect judges that the income distribution in society (under existing arrangements for redistribution via the tax and benefit system) is in some way ‘fair’ or acceptable; and
- It is valid to add up individuals’ expressions of WTP (or WTA) to reach societal values.

The concept of total economic value (TEV) (Figure 3.9) encompasses WTP for any item that enters an individual’s welfare function. The TEV framework is not inherently selfish, despite being based on preferences, since individuals often have altruistic preferences, and indeed express these through charitable donations and so on. Thus non-use values of biodiversity, including existence and bequest values, are covered by the framework. Nevertheless, the TEV framework is inherently anthropocentric and is focused on individual preferences not social goals. Other ethical systems can give rise to different evaluation frameworks and decision rules.

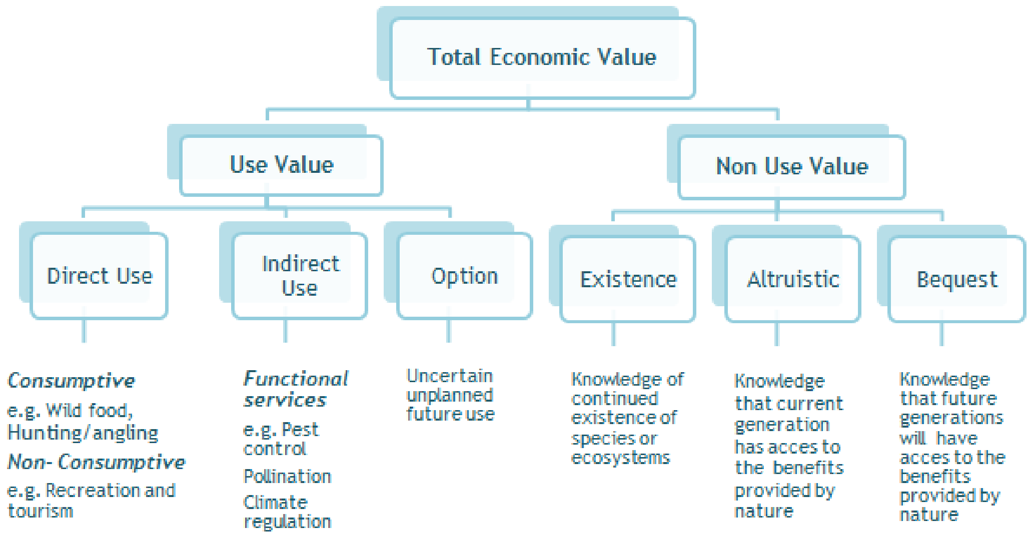

Figure 3.9: TEV Framework (adapted from Tinch and Mathieu 2011)

3.4.1.1 Extension to include natural insurance values

Pascual et al (2015) make space for 'natural insurance value' (NIV) as a component of 'total economic value' (Figure 3.10), with the more conventional components (use and non-use values) being classified as 'total output value' (TOV). McPhearson et al (2014) argue that insurance value reflects "the maintenance of ecosystem service benefits despite variability, disturbance and management uncertainty". Pascual et al further divide NIV into 'self-protection' (lowering the risk of a disturbance event) and 'self-insurance' (reducing the size of loss from an event). NIV is quite a specific concept relating to "the value of one very specific function of resilience: to reduce an ecosystem user's income risk from using ecosystem services under uncertainty" (Baumgärtner and Strunz, 2014:22).

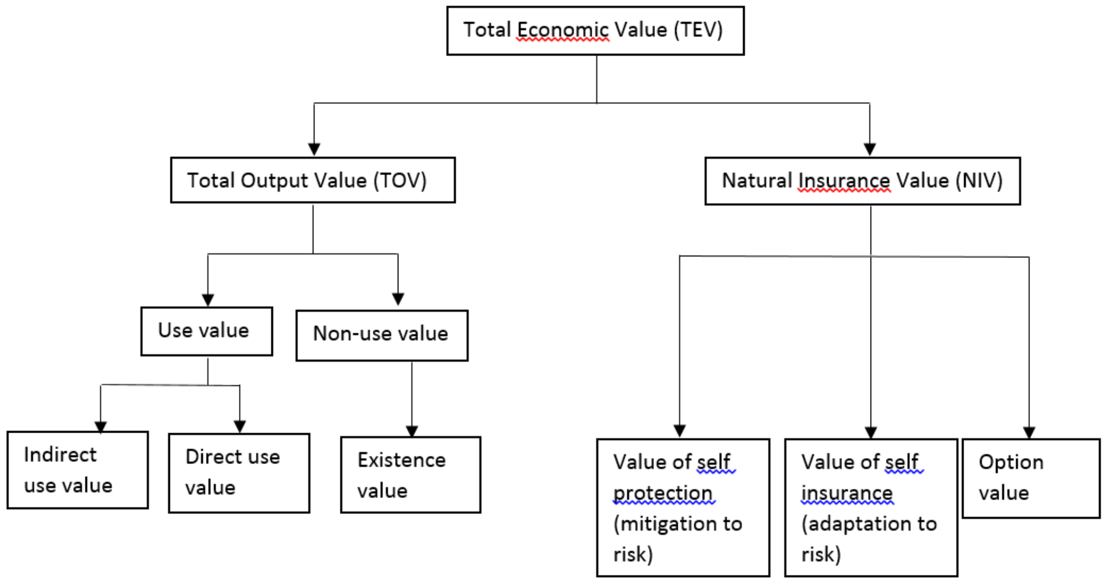

Figure 3.10 Definition of TEV by Pascual et al. (2015) (adapted)

The classification of “option value”, defined as the “importance that people give to the future availability of ecosystem services for personal benefit” (Pascual et al., 2010) is a source of debate. If the TEEB (The Economics of Ecosystem and Biodiversity) classified it under “use value”, we decided, based on information found in the literature, to classify it as an insurance value. Indeed, Plummer et al. (1992) concludes in their study on option value that this type of value was originally associated to a change of price under uncertainty which leads to a measurement issue different from the classic consumer surplus problem. “Option value has very little to do with the value of an option. Instead, if a price change is proposed under uncertainty, option value is a measure of the premium or discount a consumer is willing to pay or accept to purchase the price change by making a constant payment […] rather than a payment […] for each state of the world”.

3.4.1.2 Critique of methods relying on TEV and welfare

It is increasingly common to measure and interpret the ways in which ecosystems and their human uses and management underpin personal and societal well-being via an ecosystem services framework, often overlaid with assessment of economic value in total economic value (TEV) terms. The “classical” economic theory behind monetary valuation methods is grounded in individual utility and preference satisfaction (Wegner & Pascual, 2011). These standard economic methods are however controversial, particularly when they are extended outside areas traditionally managed through markets.

Treating aggregated TEV as an index of social welfare is problematic in two main ways: it assumes the inter-personal comparability of utility (without which, there is no obvious way to aggregate preferences at the societal level); and it assumes that the underlying income distribution is socially optimal, or at least an issue that is adequately dealt with via existing policies (notably taxation and benefits). This is generally overlooked (though in some cases income weights are used to adjust values). It can be argued that this is a reasonable approximation in the context of valuing current market exchanges, since our economic and political structures actually use these values, and tax/welfare policies act to redistribute incomes as a result of democratic processes. However, extending valuation outside the market (for environmental goods and services) is ethically contentious and could support policies that are regressive – for example, it appears more 'efficient' to cluster environmental 'bads' where people are poorer, because their willingness to pay (constrained by ability to pay) is lower than that of wealthier people.

Welfare-based tools are useful and remain appropriate in some, but not all, contexts. Several issues need to be kept in mind regarding these tools:

- basing a system of value on preferences assumes that individuals are the best judges of their own welfare
- accepting individual behaviour/statement as the indicator of preference assumes that individuals are capable of expressing values in this way, and that such preferences are stable
- values expressed through market behaviour are constrained by incomes/ability to pay
- data gaps
- optimism bias
- non-linearities, threshold effects, areas of highly inelastic demand / rapidly changing values
- partial-equilibrium focus of CBA or CEA (defined in sub-task 4.3.2.) make them too narrow and static for assessing and mainstreaming some policy option (e.g. climate policies)

The use of economic values for services such as biodiversity protection can therefore evoke strong responses from different perspectives, whereas this is not the case for ecosystem services traded in markets.

3.4.2 Economic Impact Assessment

The basis of Economic Impact Assessment (EIA) is an underlying judgement that "what matters" is the impact of policy on economic activity.[12] Economic impacts can be studied

[12] For further information on the system of national accounts, see http://unstats.un.org/unsd/nationalaccount/sna.asp For details in the UK context, see http://www.ons.gov.uk/ons/guide-method/method-quality/specific/economy/national-accounts/index.html For GVA at the regional level, see http://www.ons.gov.uk/ons/guide-method/method-quality/specific/economy/regional-accounts/index.html For EIA

from a monetary perspective, but also in real terms, notably in terms of employment, or sometimes in terms of resource and energy use. For any given case, 'economic impact' refers to changes in the level of economic activity that occur in an area as a result of a sector, project or policy. Changes in economic activity can be measured by a variety of indicators, including gross and net expenditure, value added (a measure of output), household incomes (aggregate wages and salaries), or employment (e.g. number of full-time equivalent jobs). These indicators are not independent, but represent different ways of looking at a change in activity - for example, both changes in employment and changes in income generated result from changes in expenditure on an activity, so they are not independent. In general, the primary focus of an EIA analysis is the consequence for:

- Income, usually measured by estimation of Gross Value Added; and/or
- Employment, usually measured in Full Time Job Equivalents.

An EIA has to be undertaken with reference to an identifiable constituency, and this is usually a geographical area. For many policy contexts the most relevant constituency is a whole country, but other constituencies are also common – assessments at the regional level, or for a specific administrative region or local community. It is important to remember that the results of an EIA analysis are sensitive to the constituency selected, so results from EIAs with different constituencies cannot be compared directly. For example, the creation of 200 jobs in Dumfries and Galloway (or Cornwall) does not necessarily imply there will be a 200 net increase in Scottish (or English) employment. The benefits calculated for the specific constituency under analysis may in part be losses ('displacement') from elsewhere.

EIA recognises that the various sectors which make up the economy are interdependent. Changes in activity in one sector will have impacts on the sectors that support it through providing inputs. At the same time, changes in incomes will lead to changes in expenditures, with consequences for other sectors. So EIA covers three different levels of impact:

- Direct impacts related to the specific sector under analysis: for example, jobs and incomes in sea fishing;
- Indirect impacts related to other sectors supporting it: for example, incomes and jobs in fish processing, marketing, vessel maintenance and other sectors supporting sea fishing; and,
- Induced impacts that result from those involved in the sector spending their income within an area in the community: for example, incomes and jobs supported by fishing families' expenditures on goods and services.

These impacts are often estimated using multipliers derived from Input-Output analysis, discussed below. For most applications, use of published I-O tables and associated

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methods generally, see http://www.ons.gov.uk/ons/rel/regional-analysis/measuring-the-economic-impact-of-an-intervention-or-investment/measuring-the-economic-impact-of-an-intervention-or-investment/economic-impact--paper-one.pdf multipliers is likely to be the most practical solution.[13] General equilibrium modelling (see below) is more suited to large-scale macroeconomic analysis and less likely to be a suitable choice for comparisons of sectors. The methods are applicable wherever the question (or one of the questions) of interest is the impact on economic activity and employment, but they do not reveal anything directly about impacts on human welfare overall.

3.4.2.1 GDP and GVA

Gross domestic product (GDP) is a key indicator of the state of the whole economy, measuring the totality of the national income. GDP is theoretically the amount that is currently being paid to households in the form of wages, profits, rents and interest. Other things being equal, a higher GDP is preferred to a lower level because this would enhance our potential level of consumption of goods and services. If it can be measured, the GDP of a particular sector would be the contribution of that sector to the national / household income. Arguably, GDP itself is only important because it measures our income and therefore a nation's ability to consume now (or in the future, should we choose to save some of that income).

Three main approaches can be used to estimate GDP and therefore the national income (all of which are used, in order to give a more robust overall picture of the economy):

- The production approach to estimating GDP looks at the contribution of each economic unit by estimating the difference between value of an output (goods or services) and the value of purchased inputs used in that output's production process. This difference is Gross Value Added and is an approximation of the amount distributed by each unit to households in wages, profits, rents and interest (though the government may appropriate parts of these flows before or after households actually receive these payments).
- The income approach to estimating GDP measures directly the incomes earned by individuals (e.g. wages, profits rents and interest).
- The expenditure approach to estimating GDP measures total expenditure on finished or final goods and services produced in the domestic economy. The expenditure on finished or final goods is equivalent to the sum Gross Value Added (GVA, i.e. household income) associated with the production of all goods (raw materials, intermediate and final goods)

There have been many critiques of GDP, including in recent years the Beyond GDP Conference (2007), EC Communication "GDP and beyond: Measuring progress in a changing world" (2009), Parliament Resolution (2011), and the Stiglitz/Sen/Fitoussi report (2009) on the measurement of economic performance and social progress. In the UK, ONS is developing well-being indicators.[14] Nevertheless, GDP and associated indicators remain

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[13] Guidance for the UK is provided by http://www.ons.gov.uk/ons/rel/regional-analysis/measuring-the-economic-impact-of-an-intervention-or-investment/measuring-the-economic-impact-of-an-intervention-or-investment/economic-impact--paper-two.pdf

[14] http://www.ons.gov.uk/ons/guide-method/user-guidance/well-being/index.html

the most widely used and recognised indicators of economic performance. Like EIA, they do not seek to represent social welfare or wellbeing, but limit attention to measures of national income. An alternative, welfare-focused approach to assessing changes in resource allocation is provided by the TEV framework, discussed further below.

In the UK, Input-Output Supply and Use Tables are used to reconcile these three different approaches (production, income and expenditure approaches) to measuring GDP, explaining any differences between the calculations by linking the inputs used, GVA and the outputs produced in a coherent overall framework (Akers & Clifton-Fearnside, 2008).

3.4.2.2 Input-Output analysis

Input-output (IO) analysis represents the interdependencies between industries in an economy.[15] It is a model of an economy where the transactions between each industry sector, the household sectors, and the economies outside the economy are summarised in a matrix (Ivanova and Rolfe, 2011). The coefficients in the matrix show the proportions of each industry's gross output that are attributable to inputs from other industries. The matrix provides a static and mechanistic overview of the relationships within an economy at a given point in time, giving insight into the value of economic transactions between different sectors in an economy, including outputs for exports, capital formation (investment) and final government and private consumption. Input-output tables can then be used to calculate the added value that each sector contributes to the final output of an economy.

The IO tables can therefore be used for industrial analyses and EIA consistent with the national accounts. This is generally carried out in terms of changes in GVA, or expenditure as a proxy for GVA, to assess the contribution to the economy of each industry or sector (ONS, 2015), in terms of direct impacts, direct and indirect impacts, or direct, indirect and induced impacts (see above).

The ONS regularly produces IO tables for the UK, and Eurostat produces tables for the EU. These tables are important in national accounting (see below) and can be used to carry out EIA. However, applying a national IO model would overstate the multiplier effects if the boundaries of the assessment are sub-national. This is because some part of the impacts of a change in the local economy will take place outside of the local economy. Input-output analysis can be modified for the regional level by simple adjustment of coefficients using local employment data, if these data are available.

Monetary IO tables can be 'extended' with environment-related information for each sector, such as its emissions, primary (natural) resource use, land use and other external effects per sector. These environmental externalities may be expressed in monetary terms as well (EU

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[15] For further details and information on IO methods, and the selection and use of multipliers, see Miller, R. E., & Blair, P. D. (2009). Input-output analysis: foundations and extensions. Cambridge University Press. For use and data sources in the UK context, see http://www.ons.gov.uk/ons/guide-method/method-quality/specific/economy/input-output/articles-and-analyses/index.html and in particular the downloadable glossary and bibliography.

and JRC, 2006). These methods are still under development (see for example http://creea.eu/ for details of the EXIOBASE model) along with a recent focus on extending the United Nations System of National Accounts to cover environmental and ecosystem accounting.

3.4.2.3 Multipliers for direct, indirect and induced impacts

Once direct impacts of (changes in) expenditure have been assessed, the indirect and induced impacts are generally estimated through use of multipliers derived from IO tables. The calculation of multipliers provides a useful tool in the form of an easy estimate of the wider economic impacts of a change (GHK, 2007). Multipliers can be derived for various indicators, including output, employment, and gross value added. Two main types are used:

- Type I multipliers cover direct and indirect effects only. This means they only estimate industrial impacts, thereby underestimating the total effect on the economy. But this avoids some unrealistic assumptions.
- Type II multipliers cover induced effects as well, thereby covering both industrial and consumption impacts. However, this requires the assumption that final consumers do not change their final consumption patterns in response to changes in income. ONS (2010) reports only Type I multipliers, partly because the assumption of no income effects is too unrealistic, and partly because suitable employment data are not available.

More generally, EIA based on IO models assumes that incomes and employment can increase as an outcome of expenditures without causing any wage or price increases. This is grounded in the assumption that previously unused or underused resources can be employed. This may be approximately correct for small changes in activities, or where resources are underused – in particular, where there is significant unemployment in the labour market. However the assumption is quite questionable for larger changes. An EIA will not give the full consequences of completely stopping an important activity in an area, for example, since this could lead to impacts that are not addressed in EIA – impacts on local wages and prices, for example, and people moving to other areas for work.

In practice, there are various reasons why multipliers based on IO models might give misleading results in any given case (Boardman et al, 2008, 124-5). Firstly, as the name suggests, multipliers estimate impacts as multiples of the direct impacts. Hence, any over-estimation of the direct impacts will result in over-estimation of the indirect and induced impacts.

Secondly, there are often transfers or displacement effects, where increased spending in one area simply displaces spending from another area. In effect this is a failure of the assumption about pools of unemployed resources – investment in tourism facilities, for example, will lead to greater tourist expenditure in an area, but only part of that will be truly additional, with the remainder being displaced from other tourist areas. This may or may not be relevant to a particular analysis, depending on the boundaries – specifically, are the areas from which expenditure is displaced inside or outside the boundaries of interest?

Thirdly, leakage effects may arise, whereby some part of the indirect and induced effects will occur outside the area of interest, with the goods and services required to support a sector being sourced from outside, and the incomes being spent outside, the boundaries. This also applies to enterprises that are owned largely or wholly by people outside the area, for example branches of large national or international industries that are listed on stock exchanges. Leakage is often expressed as a percentage of total impacts, based on averages from literature or on original research. Leakage does not mean that the impacts do not occur, just that they occur outside the area of interest. How important this is depends on the specific details of the analysis.

Finally, there can be second-order economic effects not represented in the fixed-price IO model that affect people living within the boundaries of the assessment. On the negative side, increased competition for space and goods can lead to increased prices and congestion. On the other hand, suppliers may be able to take advantage of economies of scale and reduce costs accordingly. So the net direction of these effects is unclear and depends on the specific case.

It should be noted that displacement and leakage both depend on the boundaries of assessment, in opposite ways. Thus if the area of assessment is small (a single tourist resort for example), there may be little concern about displacement (expenditure being displaced from other resorts is not an issue) but leakage may be very important (not only will resources be sourced from outside the town, but employees may live and spend their incomes in other areas). Conversely, for a regional or national level analysis, there may be relatively little concern about leakages (except for imports, but these are already identified in IO tables), but substantial concern about displacement of economic activity.

For these reasons, use of multipliers from IO models has been controversial. Some have argued that they should not be used at all - that '*economic impact studies based on multipliers are quite clearly an improper tool for legislative decision-making*' (Hunter, 1988, p.16) – but multiplier-based analysis is widespread, and can be useful if carried out and interpreted carefully. It is clear that multipliers based on IO models give a relatively quick and straightforward method for estimating direct, indirect and induced economic impacts. It is also clear that these measures rest on some questionable assumptions, as well as depending on the quality of data used to produce them, and so should not be interpreted as precise estimates. Furthermore, the results of any assessment are crucially dependent on the choice of the boundaries – which areas/populations are considered relevant, and which are excluded – and multipliers estimated for one scale cannot simply be transferred for use at another scale. And it should always be kept in mind that EIA focuses only on economic impacts and employment, so other facets that should enter decision processes –

environmental impacts, social justice and so on – might need to be considered alongside EIA results. Overall, results of EIA based on IO models should be interpreted with care, and should be seen as a means of simplifying complex information and relationships to support thinking about resource allocation decisions, and not as a form of hard-and-fast decision rule.

3.4.2.4 General equilibrium models

Partial equilibrium analysis of single markets makes the 'ceteris paribus' assumption that 'all else remains equal' – this means that it holds all prices/quantities other than for the specific good of interest constant, focusing on equilibrium in the specific market under analysis (hence, 'partial equilibrium') and ignoring knock-on changes in equilibria in all other markets (i.e. the 'general equilibrium' in the whole economy). Input-output models allow for some changes to ripple through other sectors, but hold prices and coefficients constant, in effect assuming that there are pools of unemployed resources that can be drawn on, or added to, with no impacts on prices or technological choices. For anything above marginal changes, these are very shaky assumptions.

One approach to relaxing the assumptions of fixed technological coefficients and perfectly elastic supply of factor inputs is to expand the IO model to include a supply-side, developing an econometric model allowing for price and quality adjustments and changes in output:input ratios and consumption patterns (Rey, 2000). This allows a combination of the sectoral detail of IO models with dynamic price forecasting in regional econometric models. However, Hunt et al (1996) and Hunt & Snell (1997) report little consistency across the results and methods of these models – the differences are explained by the structures of the models, the data sets and data treatment.

Computable General Equilibrium (CGE) models[16] also relax the assumptions of IO and allow for indirect/induced effects where changes in one market influence supply curves and alter the equilibrium for the whole economy. To do this, they replace the assumption of unemployed resources with one of perfect market clearing, i.e. there are never any unemployed resources. This too can be criticised, for being unrealistic in the other direction.

A CGE model is based around an I-O table augmented with elasticities that define how behaviour responds to price changes. Each transaction flow in the I-O table is disaggregated into two components, price and quantity, that both adjust in response to changes under analysis. Comparative-static CGE models allow for price adjustments but do not include a specific time dimension: this means that they capture quite complex behaviour regarding current resource allocation decisions, but ignore intertemporal choices about savings and investments. Recursive-dynamic CGE models add this temporal dimension, but require many additional assumptions, including definition of future steady-state conditions for the

[16] For further details of CGE models and their application, see Dixon, P. B., & Jorgenson, D. (Eds.). (2013). Handbook of Computable General Equilibrium Modeling SET, Vols. 1A and 1B. Newnes.

economic structures (since the models cannot be solved over infinite time). Dynamic stochastic CGE models allow for random shocks to the economy and include utility and production functions, in principle consistent with (neo-classical) microeconomic theory, but this represents another layer of assumptions. The models are highly complex, difficult to construct and solve, and generally involve less sectoral detail than IO models.

Overall, while CGE models add some realism regarding price and technology effects, this comes at the cost of much greater complexity, reducing the sectoral detail, and the introduction of additional assumptions and expert judgement of modellers. The results are sensitive to these choices, in a way that makes it difficult to compare the results from different assessments. The more complex dynamic and stochastic CGEs are primarily intended for macroeconomic forecasting and analysis of associated policy (e.g. changes in taxes and tariffs) rather than for comparison of specific sectors or resource uses.

3.5 Monetary valuation of ecosystem service impacts

3.5.1 Methods

One reason for the use of many different approaches is that there are also different purposes and uses for valuation evidence, including ecosystem accounting, policy and project appraisal, awareness raising, and so on. Each of these may call for different specific methods and coverage, and different requirements for accuracy and research expenditure commensurate with the context. A focus on trade-offs, comparisons of states of world and what may be lost or gained from decisions is more policy relevant than absolutes estimates, which make for catchy headlines but “have no specific decision-making context” (Costanza *et al*., 2014). Taking account of relationships and feedbacks at broad scales can help to defuse the objection that multiple projects change prices and substitute sets in ways that conventional appraisals overlook (Hoehn and Randall, 1989). Increasingly, attention is turning also to environmental and ecosystem accounting, calling for different types of value (exchange values rather than welfare/surplus values – see Ecosystem Accounting discussion paper), and many policy assessments consider economic impacts (contributions to gross value added and employment) as well as, or instead of, welfare-based estimates.

While monetary valuation has been controversial, this can be interpreted in the context of gradual progression in the framings of human-environment interactions. On a practical level, Mace (2014) recognises that most environmental decisions are made on the basis of economic arguments, arguing that refusing to engage with valuation risks further marginalisation of nature from decision-making: “If the benefits provided by nature are assigned no value, they are treated as having no value, and current trends in the decline and deterioration of natural systems will continue.” At the same time, strongly reductionist approaches to valuation are set in a ‘nature for people’ framing that is most likely to elicit rejection on principle. A softer ‘people and nature’ framing is more acceptable, and it is towards this that many initiatives (such as IPBES) are tending, but this represents a challenge for existing valuation methods.

The various groups of valuation methods are summarised in Table 3.3

Table 3.3: Valuation methods

Family and methods	Description	Suitability
Market-based techniques: • *Market prices* • *Production functions*	*Market prices are rarely equal to economic values. Market information may require substantial analysis to deliver usable values: for example correcting for taxes and subsidies, or estimating how values change with quantity.*	*Capture extent to which biodiversity supports marketed services, but not necessarily resilience. Very limited use for other biodiversity values, though price premiums on some “green” products (e.g. MSC fish) could reflect non-use values for conservation.*
Cost-based techniques • *Avoided costs* • *Replacement/ restoration costs*	*Proxies that do not assess economic value, but rather the costs that are avoided due to some ecosystem asset, or the costs that would be incurred to replace or restore the asset.*	*Widely applicable to restoration of ecosystems and potentially where targets for conservation and restoration exist. Risk of double-counting if these combined with values of services supported by the systems.*
Expenditure measures • *Expenditures* • *Gross value added* • *Employment*	*Measure expenditure, not economic value: the bases of estimating regional economic impacts through input-output modelling and multipliers.*	*Commonly used in the case of nature-based recreation and tourism, though generally this is valued as such, without splitting out a ‘biodiversity’ component. Not commensurate with TEV values but useful for other purposes.*
Revealed preference • *Travel cost* • *Hedonic pricing* • *Random utility model* • *Avertive behaviour*	*Methods based on values for environmental resources that are ‘revealed’ by behaviour in associated markets.*	*Applicable to use values for recreation and potentially aesthetic values, though again these are generally valued under those service categories without splitting out biodiversity.*
Stated preference • *Contingent valuation* • *Choice experiments*	*Methods based on surveys in which people give valuation responses in hypothetical situations.*	*Applicable to any good or service, including biodiversity, and capable of covering non-use values. However, double-counting is a risk, in particular due to embedding / part-whole bias.*
Value transfer • *Unit value transfer* • *Function transfer* • *Meta-analysis*	*Allow existing value evidence to be applied to new cases, with more or less sophisticated adjustment, without the need for primary valuation studies.*	*Applicable but dependent on availability of suitable source studies from one or more of the above categories.*

These methods are associated to the notion of value, explained in D3.2.

Economic valuation methods seek to determine individual’s preferences, whatever the individual’s tastes, motivations, status or knowledge - though in practice, most applications will use averages for a representative group of individuals, rather than identifying impacts for each individual. The strength of an individual's preference is measured in terms of the individual's WTP to secure some gain (or Willingness to Accept Compensation (WTA) for giving something up). The basic idea is that the more positively (or negatively) individuals are affected by a change, the more of their finite income and wealth they will be willing to trade-off in order to secure (or prevent) the change. This approach to assigning value is relatively straightforward and can be applied to a very diverse range of goods and services. And resulting monetary valuations can be compared and aggregated across individuals.

3.5.2 Critical natural capital and valuation

Non-linearity, threshold effects and areas of highly inelastic demand / rapidly changing values all have consequences for valuation, both within individual studies, and in particular for attempts to transfer values across studies, for grossing-up across spatial scales, or to construct meta-analysis functions. Where critical natural capital exists, valuation can be very difficult or impossible, suggesting a need for precautionary policies and setting limits to the applicability of cost-benefit methods where catastrophic changes are plausible.

Critical natural capital is usually defined as that part of the natural environment that performs important and unique functions, and therefore ought to be maintained in any circumstances for present and future generations. The idea of critical natural capital reflects the view that there is some level of natural capital that is 'essential' and provides important ecosystem services that cannot be substituted by other forms of capital, such as human or social capital (de Groot *et al*., 2003; Dietz and Neumayer, 2007). Depending on the scale, this could mean globally essential - for example, continuing human life on the planet - to locally essential – for example, a minimum level of accessible green space for psychological well-being - and anything in between. Typical examples include essential ecosystem services, such as freshwater resources, climate regulation and fertile soils (Ekins et al., 2003).

In economic terms, critical natural capital can be conceptualised as an area of perfectly inelastic demand below a certain level of provision; it is a natural extension to consider gradually increasing demand elasticity above an absolute threshold (Figure 3.11; Farley, 2008). There are limits to the use of economic methods where marginal values rise steeply, and the recognition that critical natural capital cannot be traded-off. Identifying critical natural capital is partly outside the remit of economics (a matter of biophysical science) but can also depend on ethical deliberation and how minimum thresholds of acceptable outcomes are defined. For example, it is possible to argue on cultural/ethical grounds that particular sacred sites should be accorded critical status, and excluded from trade-offs, though this has nothing to do with ecology or natural functions. This can go some way to addressing the concerns relating to incommensurability of values, by setting 'hands off' areas where trade-off is not permitted. However, it assumes that such critical capital will be protected by strong policy measures, so from the perspective of developing look-up values it is likely to be sufficient to assume that changes are of a non-critical nature.

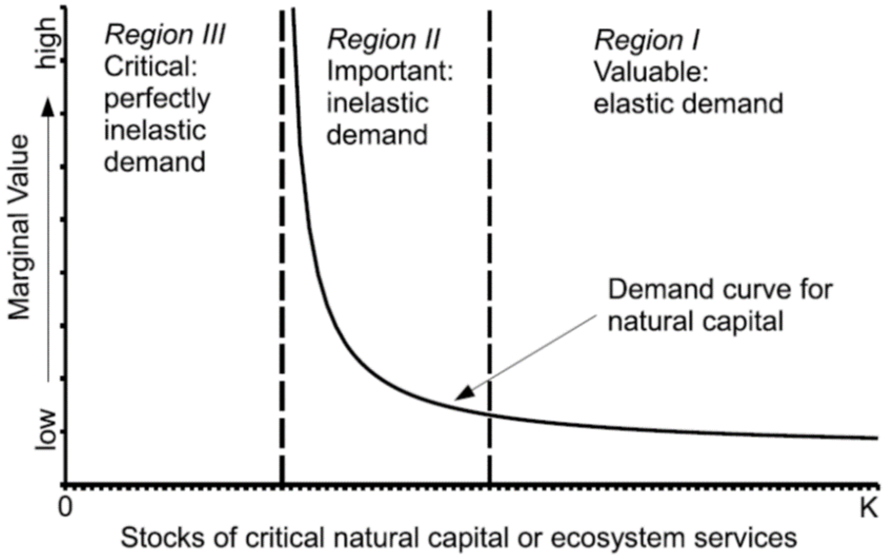

Figure 3.11: The demand curve for natural capital (Farley, 2008:3)

Assuming the system under investigation is in a state for which valuation is feasible and appropriate, ecosystem accounting principles imply that asset values would be measured as expected service flows, generally based on the current pattern of use (SEEA Para 2.40) unless there is strong evidence to think otherwise. Defra/ONS Principle 14.1 states that “Any departure from a constant service flow assumption would need to be justified and evidenced”. However, this appears to reverse the appropriate burden of proof in some cases (*cf.* the precautionary principle), in particular for exploitation of ecosystem goods and services that may not be sustainable. Where current flows are sustainable, the constant flow assumption is relatively unproblematic, although the capacity for enhanced future flows is ignored – but it is appropriate that any claim of increased future flows should be justified and evidenced.

However, current flows may not be sustainable (e.g. over-fishing, soil erosion) and in these situations the constant flow assumption could be dangerously wrong. It would be appropriate, therefore, were the burden of proof be to demonstrate that the sustainability of a constant flow is a reasonable assumption. It may be preferable to use dynamic models of ecosystem service provision to account for possible changes and risks. Even though the level of uncertainty in these models is likely to be significantly greater than the uncertainty in current flow measurement, it does not necessarily follow that the assumption of constant flows is less uncertain, or justified.

Table 3.4: Common degrees of inelasticity for ecosystem services

	Ecosystem service	Valuation methods	Sensitivity to location of humans	Sensitivity to substitutes	Sensitivity to complementary goods	Critical natural thresholds	Critical human demand thresholds
Provisioning services	*Food*	*Market price-based*	*Proximal*	*Crop scale: Substitutable* *General scale: not substitutable*	*Complementary with some regulating and supporting services such as nutrient cycling*		
	Raw material		*Proximal*	*Substitutable*	*Complementary with some regulating and supporting services such as nutrient cycling*		
	Fresh water		*Proximal but depends on the flow from upstream to downstream*	*Hardly substitutable*			
	Medicinal resources		*Proximal but depends on the flow from upstream to downstream*	*Substitutable (synthetic)*			

Table 3.4: Common degrees of inelasticity for ecosystem services

	Ecosystem service	Valuation methods	Sensitivity to location of humans	Sensitivity to substitutes	Sensitivity to complementary goods	Critical natural thresholds	Critical human demand thresholds
Regulating services	*Local climate and air quality*	*marginal damage cost*	*Non-proximal*	*Hardly substitutable (except for geoengineering but this is controversial)*			
	Carbon sequestration and storage	*avoided abatement costs* *marginal damage costs* *Preventive expenditure*	*Non-proximal*	*Substitutable (CSS)*			
	Moderation of extreme events		*Proximal*	*Hardly substitutable*			
	Waste-water treatment	*Replacement cost* *Preventive expenditure*	*Proximal*	*Substitutable*			
	Erosion prevention and maintenance of soil fertility	*Replacement cost* *Preventive expenditure*	*Proximal but depends on the flow from upstream to downstream*	*Substitutable? (Chemicals for soil fertility)*	*Complementary to provisioning services*		
	Pollination	*Replacement*	*Proximal*	*Hardly*	*Complementary to provisioning*		

Table 3.4: Common degrees of inelasticity for ecosystem services

	Ecosystem service	Valuation methods	Sensitivity to location of humans	Sensitivity to substitutes	Sensitivity to complementary goods	Critical natural thresholds	Critical human demand thresholds
		cost *Preventive expenditure*		*substitutable*	*services*		
	Biological control		*Proximal*	*Hardly substitutable*			
Supporting services	*Habitats for species*		*Non-proximal*	*Hardly substitutable*	*Complementary with provisioning services*		
	Maintenance of genetic diversity		*Non-proximal*	*Hardly substitutable*	*Complementary with provisioning services*		
	Nutrient cycling		*Proximal but depends on the flow from upstream to downstream*	*Substitutable*	*Complementary with some provisioning services such as primary production*		
Cultural services	*Recreation and mental and physical health*	*travel cost*	*Proximal*	*Substitutable*	*Complementary with other cultural services for example*		
	Tourism	*Stated preference* *Choice experiment*	*Proximal*	*Substitutable*	*Complementary with other cultural services for example*		
	Aesthetic	*Stated*	*Proximal*	*Hardly*	*Complementary*		

Table 3.4: Common degrees of inelasticity for ecosystem services

	Ecosystem service	Valuation methods	Sensitivity to location of humans	Sensitivity to substitutes	Sensitivity to complementary goods	Critical natural thresholds	Critical human demand thresholds
	appreciation and inspiration for culture, art and design	*preference* *Choice experiment*		*substitutable*	*with other cultural services for example*		
	Spiritual experience and sense of place	*Stated preference* *Choice experiment*	*Proximal*	*Hardly substitutable*	*Complementary with other cultural services for example*		

3.5.3 Elasticities

Answers to the key questions about marginal values of changes in resource allocations depend on elasticities. Elasticities measure the amount by which a variable changes in response to changes in another variable. If its value is higher than 1, the variable is elastic and varies more than proportionally to changes in other variables. If its value is lower than 1, the variable is inelastic. Different values of elasticity can be observed for the same variable at different starting points. Indeed, for example, if a producer provides a good for a low price, the quantity that will be supplied will rise much more than if the price is higher. Key elasticities in economic analysis include:

- Own price elasticity, measuring the extent to which demand decreases when price increases. In almost all cases own price elasticity is negative, i.e. demand falls as price rises. The price elasticity of demand corresponds to the change in the quantity requested for a considered good or service in case of a change in its price. In other words, if the price of the good/service changes by 1%, the price elasticity of demand shows the percentage change in the demanded quantity. Price elasticity of supply corresponds to the change in the quantity supplied for a considered good or service in case of a change in its price. In other words, if the price of the good/service changes by 1%, the price elasticity of supply shows the percentage change in the supplied quantity.

- Cross-price elasticity, measuring the extent to which demand for a good changes when price of another good changes. If cross-price elasticity is positive, the goods are substitutes; if negative, they are complements.

- Income elasticity, measuring the extent to which demand for a good increases when income increases. It is the ratio of the percentage change in demand to the percentage change in income. A 'normal' good has income elasticity between 0 and 1, a 'luxury' good greater than 1, and an inferior good less than 0.

- Expenditure elasticity, measuring the extent to which expenditure on a good increases when expenditure overall increases (this is used as a proxy for income elasticities)

- Elasticity of substitution, measuring the extent to which one good or service or input can be replaced by another. The concept of the inverse of the elasticity of substitution (or elasticity of complementarity) has been introduced by John Hicks in 1932. The elasticity of substitution between two goods or services (buying G2 instead of G1 for example) is the ratio of the expenditure for G2 compared to the one for G1. Thus if the price of G2 increase, the total expenditure for G2 should increase. But, because the product is more expensive, the quantity requested (demand) can decrease, which reduce the total expenditure for G2. The magnitude of the elasticity of substitution will inform which of these two effects will dominate. If the elasticity if lower than 1, the demand decreases less than the increase of price, leading to an increase of the total expenditure for G2. G2 and G1 are complements. If the elasticity is higher than 1, the demand decreases more than the increase of price, leading to a decrease of the total expenditure for G2. G2 and G1 are substitutes.

Unfortunately, evidence on key elasticities is often lacking, and when it does exist, is specific to particular markets and times – although if it can be argued that situations, populations, tastes, incomes and economic structures are very similar, the elasticities are also likely to be similar. Large datasets and detailed statistical analysis are needed to produce robust estimates. Generally assessments are focused on specific markets, but one exception is a major 1996 international comparisons project (Seale et al 2003, Regmi & Seale 2010) that reports elasticities for various components of budgets (see: Figure 3.12 and Table 3.5).

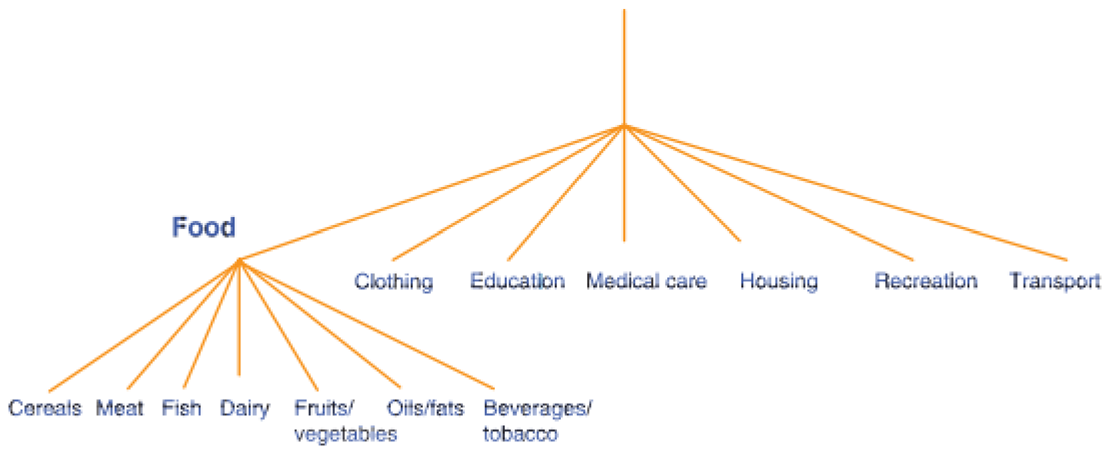

Figure 3.12 Two-stage budgeting model used by Seale et al (2003:6)

Table 3.5: UK elasticities from international comparison model (Seale et al 2003; Regmi & Seale 2010)		
Elasticity	***Recreation***	***Food***
Income	*1.298*	*0.330*
Expenditure	*1.270*	*0.322*
Own price (-)*	*0.959-1.050*	*0.249-0.316*
**Range from different methods (Frisch, Slutsky, Cournot, unconditional) of calculating elasticities.*		

These elasticities are based on old data (1996) but are broadly representative of elasticities estimated for high-income countries at that time. The key point to note is that in the UK (and more generally, in high income countries) recreation is clearly a luxury good, while food is a normal good. For a 1% income increase, expenditure on recreation increases by about 1.3% while expenditure on food increases by about 0.33%. Demand for recreation is quite sensitive to changes in recreation price, while demand for food is quite insensitive to changes in food prices. This is because food is a necessity. It is worth noting however that long-term elasticities for specific food products (e.g. beef) could be rather higher – people have consumption habits that may be quite inflexible in the short-term, especially for products that are a very small proportion of total budgets, but that respond more to price in the longer term as substitutes are increasingly used. One implication of these elasticities is that if people become wealthier on average (due to economic growth) they will spend proportionately more on recreation and proportionately less on food.

3.5.4 Costs and cost structures

3.5.4.1 Definitions

Cost:

Cost can be defined as “any item that makes someone worse-off, or reduces a person’s well-being” (Renda et al., 2013). A distinction also needs to be made between cost and price. The cost of a product is the opportunity cost of production, which focuses on the supply of the product and the alternative uses of the resources used in its production and supply, including “normal” profits as the return on investment. The price of this same product is determined by a market and thus combines information from the supply and demand sides: it is the interplay of supply and demand that give well-functioning markets the property of allocating resources efficiently among competing ends.

Environmental cost:

Environmental costs can have several definitions. The OECD defines them as “costs connected with the actual or potential deterioration of natural assets due to economic activities. Such costs can be viewed from two different perspectives, namely as (a) costs caused, that is, costs associated with economic units actually or potentially causing environmental deterioration by their own activities or as (b) costs borne, that is, costs incurred by economic units independently of whether they have actually caused the environmental impacts” . The United Nations explain that the lack of a standard definition of environmental cost and the fact that they are not traced and associated to specific processes or production, partly explains the difficulties about environmental management accounting. These costs take into account the disposal costs, investment costs and external costs.

Four categories of environmental costs have been used within the environmental cost scheme developed by the United Nations (concerning corporate environmental costs): waste and emission treatment (labour and maintenance materials, treatment, disposal, clean-up costs of waste and emissions); prevention and environmental management (annual costs for prevention of waste and emissions, including labour costs and external services for environmental management or extra expenditure for cleaner technologies); material purchase value of non-product output and production costs of non-product output (such as labour costs and energy cost) (United Nations, 2001; Jasch, 2003).

IFAC (2005) also defined several environmental cost categories:

- Materials costs of product outputs (purchase costs of natural resources used into the production process)
- Materials costs of non-product outputs (purchase or processing costs of energy, water and other materials that will become waste and emissions (non-product outputs)
- Waste and emission control costs (costs of handling, treatment and disposal of waste and emissions, remediation and compensation costs, and regulatory compliance costs)
- Prevention and other environmental management costs (such as costs of cleaner production projects or environmental planning and systems)
- Research and development costs related to environmental issues
- Less tangible costs (internal and external costs such as liability, company image, stakeholder relations or externalities).

3.5.4.2 Categories of cost of interest for economic analysis purposes

Opportunity cost:

The opportunity cost has been defined as the “the foregone economic benefits from alternative activities or uses of a resource on a particular site”. Sometimes, they are included in the financial transactions (e.g. as compensation payments, or land purchases). But most of the time, they are not taken into account. This type of cost can be categorized either in financial costs (real payment and expenditures such as compensatory payment, and which includes indirect payments such as transaction costs) or in the wider economic costs (uncompensated payments, such as losses from foregone development opportunities, income foregone because of limits about the extraction of natural resources, and losses of socio-economic opportunities) (Kaphengst et al., 2011).

Kaphengst et al. (2011) presents the share of opportunity cost within the total costs in EU biodiversity policy (Table 3.6). We can see that, according to these estimates, opportunity costs represent about 79.3% of the total costs.

Bateman et al. (2013) took into account the opportunity cost of species conservation (avoided decline) using regression model linking wild bird diversity to land use and location.

SEEA-CF also considers that, regarding the supply of resources, general operating costs and capital costs must be taken into account. Especially, the capital costs should include the consumption of fixed capital and the opportunity cost of the investment (rate of return) (United Nations, 2014).

Foster et al. (2001) explain the notion of “volunteering” as an opportunity cost (but also as a replacement cost for example). Indeed, this could be defined as a “donation in kind”, through providing some time. Giving time for an activity as a volunteer has a cost that is the same as the value of time that could have been spent in another activity. Thus, opportunity cost appears to be an essential type of cost to be assessed and the right overall concept for economics assessments.

Table 3.6: Annual costs and opportunity cost in EU biodiversity policy (source: Kaphengst et al., 2011:112)

Policy	**Estimated Annual Costs (€m)**	**Estimated Opportunity Costs (€m)**	**Share of opportunity costs over total (%)**
A. Natura 2000 Network	5,772	2,069	35.8
B. National Protected Areas	1,280	459	35.9
C. High Natural Value Farming	4,370	3,390	71.7
D. High Natural Value / Semi-natural Forestry	4,500+	4,500	n/a
E. Species Conservation	2,841	1,697	59.7
F. Marine Protected Areas	235	n/a	n/a
G. Biodiversity Research	648	n/a	n/a
H. Invasive Alien Species	193	Negative	n/a
I. Correction for Overlaps between above Estimates	-4722[1]	-3696	-
J. Total	**10,617**	**8,419**	**n/a**

Note: n/a = information not available

Shadow prices:

Often used to assess ecosystem services (welfare analysis), they can be distinguished from market prices in the case of inefficient markets which do not integrate all economic, social or physical constraints such as opportunity costs. They can be defined as the best approximation of the true opportunity cost or marginal valuation of a product or resource or service. Indeed market prices can be distorted for example because of taxes or a price change. Thus, the market prices are adjusted into shadow prices to present the opportunity cost of the goods and services used by the project. The shadow prices thus cannot be observable in the market. In the case of environmental issues and sustainability, shadow prices are useful because of the lack of efficient markets for ecosystem services and assets. Shadow prices are based on a welfare economic concept of value rather than an exchange value concept. They can be considered as marginal but are not similar to marginal prices related to a market mechanism. They instead express the change in welfare associated with a marginal change in the relevant good or service. (European Commission et al., 2013). The shadow price of capital can be defined as “a measure of the social value of a marginal unit of private investment”. This value can be assessed in 2 steps: 1) predicting additional consumption over time for the whole community if an extra unit of money (euro, dollar, pound…) is invested privately; 2) discounting to get a present value (Sugden & Williams, 1978).

Transaction cost:

Transaction costs are both the costs of contracting for the exchange of ownership rights, and the costs of creating these rights. In other words, they are the ex-ante costs of putting in place quasi-ownership rights, and the ex post costs of managing and improving these new rights (Krutilla & Krause, 2010). Transaction costs can sometimes be associated to opportunity costs as defined within the welfare economics. A link can be done between these costs and information costs because, even if they do not have the same meanings, a lack of information leads to expensive transactions. Thus, the more information is provided, the less important the transaction costs will be (Krutilla & Krause, 2010). Said differently, when transaction costs are low, the exchanges done within a market are considered as efficient. Hence the “Coase theorem” explaining that a regulation from the government should aim at decreasing the transaction costs to improve market transactions. Nowadays, this is not the only role of the government regarding exchanges in a market, but it is still considered that low transaction costs enable to enhance efficiency (Renda et al., 2013).

According to McCann et al. (2005), including transaction costs into policy analysis can highly improve the efficiency of this analysis because:

- They can be important: in the agriculture area, these costs can have a value up to 38% of total costs for a technical assistance program, which can have an impact on the choice and design of policy instruments.
- It would increase the credibility (improve the definitions and framework)
- It can provide a more precise comparison of different policy instruments and more accurate information to effectively design and implement policies
- They are part of the assessment of current policy to improve their effectiveness
- It provides a better analyse of the budgetary consequences of policies

Thus, the assessment of the opportunity cost cannot be realistic and complete if these transaction costs are not taken into account.

If the notion of opportunity cost seems to be the main concern for economic valuation, other types of costs can be requested according to the considered analysis.[17]

Other cost typologies:

We can also classify costs using other typologies (Table 3.7).

Table 3.7: Other types of cost

Type of cost	***Definitions***
One-off costs	Occur once; include investment and management costs
Recurring costs	Occur several times; include management planning costs and habitat management and monitoring costs
Compliance costs	Sum of capital costs (fixed costs), operating costs (variable costs) and financial costs (can be associated to either capital or operating costs)
Indirect costs	Secondary costs (substitution effect, uncertainties)
Externalised costs	An externality is the impact of a party's activity on another party without any compensation for it (positive or negative), which leads to the distinction private/social costs.
Total investment costs:	Made of fixed investments, start-up costs, changes in working capital[18].
Total operating costs:	All the disbursements foreseen for the purchase of goods and services, which are not of an investment nature since they are consumed within each accounting period. Direct production costs (consumption of materials and services, personnel, maintenance, general production costs), administrative and general expenditures, sales and distribution expenditures[19].
Financial return on investment costs:	The financial net present value of the project (FNPV), and the financial internal rate of return (FRR)[20].

3.5.4.3 Examples

Kaphengst et al. (2011) and Naumann et al. (2011) presented typologies of costs in the case of biodiversity action and green infrastructure respectively. They both classify costs in two categories: financial and opportunity costs (wider economic costs). The financial costs can be divided in one-off costs and ongoing costs (such as administrative, management, information). The opportunity costs could also be classified either in financial costs in the case of compensations (payment) or land purchase, or as a different category such as uncompensated opportunity costs (losses, e.g. of income, of socio-economic opportunities, output restriction on exploitation of natural resources).

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[17] For more information on the issues regarding transaction cost measurement, see http://www.researchgate.net/publication/222570616_Transaction_cost_measurement_for_evaluating_environmental_policies/links/0deec52dee468854b9000000.pdf

[18] http://ec.europa.eu/regional_policy/sources/docgener/guides/cost/guide2008_en.pdf

[19] http://ec.europa.eu/regional_policy/sources/docgener/guides/cost/guide2008_en.pdf

[20] http://ec.europa.eu/regional_policy/sources/docgener/guides/cost/guide2008_en.pdf

In the case of biodiversity action, Kaphengst et al. (2011) presented a typology of costs (Table 3.8):

Table 3.8: A cost typology for biodiversity action (source: Kaphengst et al., 2011:iii)		
Cost category	**Type of costs**	**Examples**
Financial costs	***Costs of resources expended:*** • *Costs of capital, labour, materials, energy;* • *Capital costs and recurrent management costs;* • *Administrative and transaction costs involving financial outlay*	• *Labour and materials for fences around nature reserves;* • *Salaries and equipment of biodiversity researchers;* • *Materials, labour and equipment for construction of visitor centres;* • *Costs of developing and administering species action plans*
	Costs that reflect opportunity costs: • *Payments to compensate for income foregone;* • *Compensation payments for foregone development/ exploitation rights;* • *Land purchase (reflecting income from land in alternative use)*	• *Agri-environment payments to compensate for loss of cereals output from leaving fallow land for nesting birds;* • *Compensation payments to fishermen for establishment of marine nature reserve;* • *Cost of purchase of farmland to establish new wetland reserve*
Wider Economic Costs	***Uncompensated opportunity costs:*** • *Lost income from foregone development;* • *Loss of socio-economic opportunities;* • *Output restrictions on exploitation of natural resources*	• *Loss of income from prevented commercial and industrial development;* • *Foregone opportunities for job creation and cohesion;* • *Loss of output of fisheries, wood, minerals, energy etc.*

In the case of green infrastructure, Naumann et al. (2011) presented a typology of costs (Table 3.9):

Table 3.9 Typology of costs of green infrastructure projects (source: Naumann et al., 2011:64)

Cost category	***Type of costs***		***Examples***
Financial costs	*One-Off Costs*	*Administrative, management and information costs*	*Establishing management bodies; Surveys; Research; Consultation; Management plans*
		Costs of green infrastructure provision	*Land purchase; One-off compensation payments; Creation of green infrastructure; Restoration of green infrastructure*
	Ongoing Costs	*Administrative, management and information costs*	*Running of administrative bodies; Monitoring; Ongoing management planning; Communications*
		Costs of green infrastructure provision	*Maintenance of green Infrastructure; Costs of management agreements; Costs of protective actions (e.g. ongoing planning controls, site wardens)*
Opportunity Costs		*Foregone development opportunities*	*Value of potential development foregone*
		Foregone resource use	*Loss of mineral extraction; Loss of water abstraction; Loss of land development rights*
		Foregone output from land management	*Foregone agricultural output; Foregone forestry output*
		Foregone socioeconomic opportunities	*Loss of regeneration opportunities; Loss of community uses of land*
		Reductions in land values	*Price of land*

In the specific case of illegal, unreported and unregulated fisheries, Tinch et al. (2008) presented different types of cost, or cost-structure, than the ones presented above (Table 3.10).

Table 3.10 A different typology of costs

Type of cost	***Details***
Environmental	• *Depleted stocks: impact of increase fish mortality on stock level* • *Size-related impacts: target on smaller or undersized fish* • *Ecological impacts: effects on the stock's prey, predators and competitors* • *Extinctions* • *Location- or time-specific environmental impacts*
Economic	• *Reduced profits: IUU fishing has medium- and long-term negative impacts for fishers and consumers* • *Data quality: IUU fishing is an unknown quantity/value which is an issue for fisheries data* • *Distorted markets: loss of market share for legal EU fisheries operations and trade distortions because of different cost structures in legal and IUU fisheries* • *Reduced access to fisheries markets* • *Tourism impacts: imbalances in ecological systems, which can have a negative impact on attractiveness of coastal waters; locally caught species can be a tourism "product" whose availability can decrease because of IUU fishing; reduction of other marine species that can attract tourists such as cetaceans; large species threatened* • *International negotiations: IUU fishing in the Member States decreases the Commission's assets in these negotiations*
Social	• *Reduced employment: stock depletion would reduce employment opportunities in fishing* • *Community impacts (unfair competition)*

Three groups of costs are presented: environmental, economic and social. Environmental costs are depleted stocks (impact of increase fish mortality on stock level), size-related impacts (target on smaller or undersized fish), ecological impacts (effects on the stock's prey, predators and competitors), extinctions, location- or time-specific environmental impacts. Economic costs are reduced profits (IUU fishing has medium- and long-term negative impacts for fishers and consumers), data quality (IUU fishing is an unknown quantity/value which is an issue for fisheries data), distorted markets (loss of market share for legal EU fisheries operations and trade distortions because of different cost structures in legal and IUU fisheries), reduced access to fisheries markets, tourism impacts (stock reductions because of IUU fisheries lead to imbalances in ecological systems, which can have a negative impact on attractiveness of coastal waters; locally caught species can be a tourism "product" whose availability can decrease because of IUU fishing; reduction of other marine species that can attract tourists such as cetaceans; large species threatened by overfishing and IUCC fishing are game fish), international negotiations (IUU fishing in the Member States decreases the Commission's assets in these negotiations). Social costs are a reduced employment (stock depletion would reduce employment opportunities in fishing), community impacts (unfair competition).

3.6 Selection of indicators and methods

The European Statistical System defines quality criteria for statistical data, but while relevant they are retrospective in application (i.e. relate to the performance of actual statistics[21]). We require a more prospective set of criteria, focused on selecting indicators for use in ecosystem service assessments, often forward-looking project/policy planning and assessment. For this, we draw on Heink & Kowarik (2010) who present a comprehensive list of criteria for biodiversity indicators (Table 3.11)[22].

Table 3.11 Criteria for the selection of indicators and their relevance (green = most relevant; red = least relevant; yellow = intermediate relevance)

Criterion	*Interpretation*	*Importance to OPERAs*
Feasibility		
• *Knowledge*	*How well is the category understood?*	*Essential that both researchers and stakeholder understand clearly what the indicators mean*
• *Portability*	*Wider relevance outside OPERAs framework*	*Desirable but not of primary interest*
• *Suitability for statistical analysis*	*Low random variation at relevant scales*	*Changes in the figures must have some interpretative validity: wide random fluctuations*
• *Existence of reference values*	*For comparison with base case*	*Desirable, but not essential, to be able to compare across scenarios.*
Efficiency of indicators		
• *Feasibility of data collection*	*Is the information available in models/ scenarios?*	*Essential to link indicators to modelling / stakeholder variables*
• *Universality*	*Widely applicable, i.e. relevance is not scenario-dependent*	*Indicators must be comparable across scenarios, and relevant to all.*
• *Parsimony*	*Particularly important for communicating results, i.e. ability to assess outcomes without too many indicators to present/graph/understand*	*Desirable, but also possible to use multiple or composite maps and indicators*
Relation between indicator and indicandum		

[21]http://ec.europa.eu/eurostat/web/social-protection/quality

[22] The criteria listed in the first column have been modified from Heink & Kowarik (2010) with additions and deletions appropriate to the changed context.

Table 3.11 Criteria for the selection of indicators and their relevance (green = most relevant; red = least relevant; yellow = intermediate relevance)

Criterion	***Interpretation***	***Importance to OPERAs***
• *Precision of correlation*	*For example if we want to measure "happiness" the Easterlin paradox would suggest that GDP is not a good choice*	*Desirable, but again multiple indicators can be used, and interpreted as appropriate.*
• *Validation*	*Can the relationship be tested/validated using available data?*	*Desirable, but could be acceptable to base indicators on theoretical justification.*
• *Construct validity*	*Is the indicator theoretically justified?*	*Need a clear justification for relating indicator to human wellbeing or other impacts of interest.*
• *Aggregation of a substantial amount of information*	*Single measure that is closely related to a wide range of features*	*Desirable, in particular in sense of aggregating impacts across multiple sources of threats/impacts (although this aggregation involves loss of information).*
Information to be provided by the indicator		
• *Relevance*	*In context of overall purpose, see also 'acceptance'*	*Indicators must be clearly relevant to policy interests*
• *Speed of response to change*	*Responsive to changes in the fundamental aspects of interest without long lags*	*Lags more an issue for real-time indicators. .*
• *Amplitude of response to change*	*Responds clearly to changes in the fundamental aspects of interest*	*Responsiveness important for comparing options.*
Perception of indicators		
• *Ethical grounding*	*Is the indicator justifiable on ethical/moral grounds?*	*Likely to be important for at least some indicators*
• *Acceptance*	*Do stakeholders 'like' the indicator?*	*Depends on role of stakeholders in assessment*
• *Comprehensibility*	*Does the indicator simplify complex information in an easily understandable way? (different from aggregation via focus on simplicity/understanding rather than combining information on several features).*	*Depends on audience and skillset.*
• *Economic importance*	*May be relevant if using results to motivate expenditures*	*Some indicators may be, but this is not a criterion for excluding others.*
Social characters/functions of the indicators		
• *Ability to invest responsibly*	*Usefulness as a guide to decisions*	*Depending on purpose important if focus on project expenditures*

Table 3.11 Criteria for the selection of indicators and their relevance (green = most relevant; red = least relevant; yellow = intermediate relevance)

Criterion	***Interpretation***	***Importance to OPERAs***
• *Ability to monitor and manage low probability outcomes*	*Indicators should operate under and be sensitive to high-end / extreme conditions*	*Depending on purpose – not an issue for forward-looking assessment. .*
• *Ductility in comparison with uncertainties and tipping-points*	*The indicator applies to all scenarios and does not 'break' if thresholds are reached*	*Depending on range / amplitude of uncertainties*
• *Familiarity of the indicator at the social level*	*For ready understanding without need for explanation/capacity building.*	*Depends on audience and skillset.*
• *Sustainability in the relationship between several social variables.*	*The interpretation of the indicator is not strongly context-dependent / dependent on other variables.*	*Important for comparison across scenarios and with base case.*

Selecting indicator methods in step with scenario

4. Payment for Ecosystem Services (PES)

4.1 Definition, rational, goals and objectives

PES schemes establish markets and prices for otherwise non-marketed and un-priced ecosystem services. They are schemes in which beneficiaries or users of ecosystem services provide payments to stewards or providers of ecosystems in return for *either* a guaranteed flow of ecosystem services at levels over and above those that would otherwise be provided *or* for land use or management actions that are expected to enhance the provision of targeted services.

How ecosystems are managed has a direct bearing on the state of the ecosystem and on the flow of ecosystem services it can supply. Many ecosystems are able to provide a variety of services simultaneously, but services are not generated independently of one-another and, usually, there are *trade-offs* among different services that could be provided or among sets of co-produced services. Delivering more of one service or set of services may mean delivering less of other services.

While products sold in an existing marketplace, like agricultural commodities or timber, have a clear financial value and automatically deliver a stream of revenue to their producers, other services that are also important for wellbeing but that are not sold in markets, such as providing clean water or regulating climate, have no attaching revenue stream. This leads to imbalance in decisions about how ecosystems are managed. There is relative over production of services that command a price in established markets and under production of those that do not.

One way to overcome this imbalance is for the value of all ecosystem services to be represented in decision-making and one way to achieve this is to develop actual or surrogate markets (using market-based instruments) for the services that hitherto have not been traded. Such newly-created markets provide for these values to be determined and reflected in price signals, which act then also as financial incentives.

The central idea of Payment for Ecosystem Services (PES) is to put a price on previously un-priced ecosystem services, so bringing these into economic decisions on level terms alongside conventionally priced services. The underlying logic is that those who provide ecosystem services should be paid for doing so, and this should be the case for the full range of services, not only for provisioning services for which there are already established markets. Equally, beneficiaries of services should pay for their provision. Payments to providers of ecosystem services are made either by beneficiaries of the targeted services or by organizations acting on behalf of beneficiaries, such as government or NGOs. PES schemes therefore constitute *market-based* instruments and schemes. They form a particular sub-group of instruments within this broader category.

4.2 Drivers of PES schemes and trends in use

PES is an increasingly important sub-group of market-based instruments and schemes. There has been a rapid growth in the development of PES schemes over the last 15 years and there are now several hundreds of schemes operating worldwide.

Main drivers for this expansion include:

- Government policies and commitments: There are policy commitments among some governments to promote PES scheme emergence: e.g. the UK Government (Natural Environment White Paper, The Natural Choice: securing the value of nature, 2011).
- Scientific progress in developing ways to value services and incorporate these into decisions using market-based instruments and schemes.
- Awareness- and capacity- building by intermediaries and brokers using novel approaches, such as beneficiary analysis, to identify service beneficiaries and their dependences on threatened or degrading services.
- Problems beginning to be experienced in supply of services, prompting service users to seek ways to augment supply, reduce supply risk, and secure access to vital and valued services.
- Beneficiaries seeking cost-saving ways to secure services.
- Changes in fiscal, regulatory, and public finance policies of government and (in the case of regulated industries and utilities) changes introduced by regulators.

4.3 Principles of PES schemes

PES schemes are characterised by a set of principles which, albeit not necessarily present in all schemes, are design foundations of many. They are:

- *Voluntary* schemes among the involved stakeholders
- Payment is made by the beneficiaries of services to those who provide services on a '*beneficiary-pays*' principle
- Payments are made for additional services (*additionality*); i.e. for services that are over and above business-as-usual levels or, in cases where services are under threat, for maintaining ecosystem service levels when otherwise these would decline

- Payments are *direct* to ecosystem service providers
- Payments are *conditional* on delivery of services or on undertaking management interventions intended to secure delivery of the contracted services

4.4 Relation of PES to other schemes

PES provides one means to affect how natural resources are managed and one among a set of approaches to combat ecosystem degradation. Others include: regulation, direct management of land by government, indirect management by spatial planning and development control authorities, voluntary efforts, and market-based approaches other than PES.

The relationship of PES to these other approaches is important. The management of land and natural resources may be subject to regulations to limit adverse impacts of practices on natural capital and on flows of ecosystem services, including spill-over impacts; e.g. eutrophication of rivers through excessive applications of fertilizer or too intensive livestock production and inadequate waste treatment. In some jurisdictions, *regulatory covenants* or *easements* can be used as regulatory/legal instruments for conservation purposes to restrict how land or other natural resources can be used or developed, effectively reducing the property rights of owners. Land and resource owners and managers may also undertake measures to protect NC and ES voluntarily as custodians and stewards of natural resources.

Any PES scheme is likely to be additional to any such existing stewardship regime, so its design should complement and build on existing arrangements. Equally the design of PES schemes might need to be sensitive to what is set out in existing strategic plans for the management of the areas concerned. Such plans are increasingly being developed for natural units (e.g. *catchment management plans*) and to support strategic approaches to conservation (e.g. *green grids*) or sustainable development (e.g. regional *green infrastructure plans*).

As with other schemes, PES schemes must be developed with concern for their environmental effectiveness, cost-effectiveness, fairness, etc., and with awareness of potential conflicts among these and the need for trade-offs.

PES schemes are related to other *beneficiary-pays* schemes, such as *certification* and *labelling*, in that the ultimate service beneficiary pays for the provided services. The difference with *certification* and *labelling* schemes is that payment in these schemes is indirect.

PES schemes differ from market-based schemes designed on the *‘polluter pays’ principle*, such as *offsetting*, in which those causing damage to natural capital (NC) and ecosystem services (ES) in one place make compensatory investment in NC and ES elsewhere.

4.5 Implementation requirements

Conceptually, PES involves: (i) a relationship between how an ecosystem is managed, the services it delivers, and resulting welfare, (ii) the idea that an ecosystem can be managed in different ways with each approach delivering different services and welfare outcomes, and (iii) the idea that service beneficiaries pay service producers to change how the ecosystem is managed in order for them to benefit from an enhanced flow of services.

Core components of PES schemes therefore include:

- Knowledge of what services an ecosystem produces and could alternatively produce (obtained through an ***ecosystem assessm****ent)*
- Knowledge of actual and potential beneficiaries and the value of each service to them (obtained through ***beneficiary assessment*** **and** ***valuation***)
- Knowledge of (or credible assumptions about) the relationships between changes in ecosystem management, changes in service delivery, and changes in welfare; i.e. *a causal model or 'theory-of-change' that links management actions to changes in service delivery*
- A ***baseline scenario****,* established using ***models*** and relevant *NC, ES* and welfare ***indicators****,* projecting the outlook for the ecosystem, its management, service delivery and welfare impact under the assumption there is no PES scheme, which can form the ***counterfactual*** basis for agreements and contracts between prospective buyers and sellers
- Alternative ***scenarios*** and ***prospective impact assessme****nts* of these exploring the likely outcome of different management interventions and changes in practices
- Boundaries for the scheme setting out its spatial and temporal scope, the NC/ES included in the scheme, eligibility criteria for scheme participation, etc.
- A mechanism for establishing levels of payments (the price of services) and the basis for payment
- PES contracts between buyers and sellers, setting out the management changes or service levels to be delivered, payment levels and arrangements, scheme lifetime, and any other terms
- Monitoring and reporting frameworks based upon ***NC and ES indicators*** **or** ***proxies***, enabling performance to be measured, providing scope for further learning and evidence gathering about causal relationships between management interventions and ES outcomes, and supporting adaptive management (i.e. adjustment and refinement of the scheme as necessary).

4.6 Barriers, opportunities and risks

Developing a PES scheme is in part a technical undertaking involving ‘product-related’ issues of scheme design, but it also involves a process of cultural change on the part of all the engaged actors, which is typically only slowly and incrementally achieved and depends on building trust among the parties.

The scale of the cultural challenge is related to scheme complexity. PES agreements may be reached relatively quickly for simple schemes, but may take several years for schemes that are complex, which face challenges of building trust among multiple parties and changing practices that may be long-established and deeply embedded.

This holds implications for the process of PES scheme development, especially for more complex schemes. Any process of building trust among multiple parties with the objective of identifying and implementing alternative practices and negotiating mutually-acceptable incentives is likely to be long-term, participatory and progressive.

Experience from the development of pioneering schemes suggests: the value of engaging an *intermediary* to liaise between potential buyers and sellers who should be an organization or persons already known to sellers and trusted by them; the need for clear communication of buyers’ motive to allay fears and suspicion; and the need, especially in the early stages of scheme development, to be alert to any concerns that emerge and to ensure these are addressed as a basis for confidence building.

The process of scheme development in its early stages is likely to be exploratory and, if there are significant uncertainties about aspects of scheme design or if actors need to have a high degree of confidence before being willing to engage fully with a scheme, scheme development may need to progress through pilot and demonstration projects before becoming accepted and able to be rolled out fully.

Lessons from experience to date with PES schemes suggest:

- The state of knowledge on causal relationships between ecosystem management and service outcomes is important in deciding whether to use the PES approach or whether to use alternative schemes. If causal links are unknown or the costs of establishing causal relationships is high, alternative policy instruments may be more appropriate or schemes must be built around assumptions about these relationships.
- PES schemes are particularly suited to situations where the linkage between ecosystem management and service provision are well understood.
- PES projects are useful in promoting enhancements in the delivery of existing ecosystem services; i.e. in situations where the need is to enhance or maintain a stream of ecosystem services when these are degraded, degrading, or threatened

and this can be achieved without radical alteration in the mode of ecosystem exploitation.

- PES provides explicit financial incentives to land and natural resource managers to provide public goods for which they are not currently paid; e.g. carbon sequestration, recreation.
- PES offers particular promise in relation to better targeting of payments to farmers and forest managers for the regulating and support services they provide:
  - Carbon sequestration: e.g. forests, woodlands, soil carbon, peat-land
  - Water quality and quantity; e.g. water resource supply, flood risk attenuation
  - Cultural services and species diversity; e.g. via user fees and visitor charges/payback schemes

The OECD considers that spatial-heterogeneity in costs and benefits of ecosystem service provision is a key determinant in the potential cost-effectiveness of PES schemes and that the degree to which ecosystem service payments are spatially targeted is a key element in taking-up this potential: "*the greater the spatial heterogeneity in costs and benefits of ecosystem service provision, the larger that gains that can be reaped by targeting and differentiating payments accordingly.*"

- The economic efficiency of PES can be enhanced by spatial targeting of payments to provide better value for money.
- PES may also be able to help target policy incentives to areas where they can optimize the supply of services in places where they are most needed, where they can be delivered most cost-effectively, and where they can function in harmony with other environmental objectives.

Taking up this potential may nevertheless be challenging. *Reverse auctions* to establish costs and spatial variation in these are suitable in contexts where there are many independent sellers, since this increases the competitive pressure on bidders and reduces scope for collusion and rent seeking behaviours, but can be less suitable if there are few potential suppliers.

The main barriers to PES project development are a widespread lack of awareness and understanding of PES on the part of potential service providers and service beneficiaries, lack of established markets, and lack of locally-available scientific support services. High set-up and high transition costs are barriers to establishing projects currently, but these will fall as experience with projects is translated into more effective and efficient project set-up and management.

The concept of PES is not widely understood and, because it is relatively new, PES is sometimes perceived as risky both by potential buyers and sellers. This is a constraint especially on the take-up of *user-financed schemes.*

- Potential buyers and sellers are often unaware of which ecosystem services could be provided or their value, including lack of appreciation by potential buyers of actual dependence on ecosystem services

- Many ecosystem services arise from complex processes, making it difficult to determine which actions affect their provision and precisely who are the providers and beneficiaries. Relevant here also is the diffuse nature of provision of some ecosystem services and time-lags between changes in management practices and changes in the mix and levels of ecosystem services.
- Establishing a robust counterfactual that can form the basis for agreements and contracts is also challenging. The systems involved are complex, uncertain, and contingent. A large and diverse range of factors needs to be taken into account in establishing the counterfactual, which adds to set-up and transaction costs (Jack et al. 2008).
- PES schemes typically have high *set-up costs* and high *transaction costs*.

PES remains a 'novel' approach. Most analysts acknowledge that PES projects are therefore also experiments and learning-by-doing opportunities.

The main risks for PES projects include that:

- Beneficiaries may not be sufficiently committed to long-term engagement with a PES scheme and that the costs of setting-up a scheme, which can be high, may not be warranted.
- Losses or disruption of flow of services for which payment has been made may arise if induced changes in land or natural resource management practices are subsequently reversed. This is a risk to the continuity and permanence of the gains made.
- Establishing a PES project in one area may deliver extra income for land owners, which could be used to develop other sites in unsustainable ways. A PES project may also have knock-on effects on prices for products whose supply is affected by the project, increasing incentives to farm adjacent land not included in the scheme more intensively. This risk is referred to as *scheme leakage*.
- Schemes reward – or are perceived to reward – land owners and managers with poor records of environmental stewardship. This is possible when payments target degraded or degrading ecosystems, which are most likely to deliver additional services, rather than land that is already delivering required services. This creates a risk that schemes will be perceived as *unfair*.
- A related risk is that schemes establish *perverse incentives* by setting a precedent that payments are established in response to poor environmental stewardship or by incentivising responses that increase delivery of a targeted service, but at expense of reducing delivery of valued services that are not included in the scheme; e.g. when carbon sequestration is most effectively achieved by focusing on a few rapidly growing species at expense of a natural mix that favours richer biodiversity.
- There are risks of schemes being undermined or exploited by land and natural resource managers whose participation in a scheme is crucial for its success and

so threaten to ‘*holdout’* from the scheme. The spatial patterns and relationships inherent in the delivery of some ecosystem services, such as the strategic importance of particular land parcels in connecting biodiversity corridors or in controlling critical sections of a river, give some resource owners or managers more influence than others over the overall viability of a PES scheme. This can arise also more generally when supply of a particular service requires suppliers to coordinate and cooperate in delivery. By threatening to undermine a scheme, specific resource managers may seek to exploit their situation opportunistically to obtain higher compensation. This is referred to as risk of ‘*holdout*’.

4.7 Project initiation

The process generally begins with a project protagonist. Early PES projects were often initiated by one or more buyers acting as scheme protagonists and approaching prospective sellers directly. This is possible if there is one major buyer of one or more services, a so-called *anchor* buyer on which the viability of the project rests. That buyer may contact other potential buyers and perhaps make attractive cost-sharing offers. The reverse situation is also possible where one or more prospective sellers approach prospective buyers directly. However, these represent exceptional circumstances, because this can happen only in contexts where a scheme holds a strong commercial interest for (usually) one prospective buyer or seller who is able to act spontaneously from self-interest which is self-evident.

In more usual situations, the self-interest of the potential parties is not so self-evident. Each individual actor has incomplete knowledge, so it requires strategic vision on the part of some protagonist to organise strategic cooperation among independent buyers and independent sellers. Strategic co-operation involves buyers organising to pool payments for services or sellers coordinating management interventions.

In these more general situations the intervention of a protagonist with a strategic perspective is necessary. Often this is in the form of a government organisation or an interest organisation, such as an NGO with conservation interests. Such interventions are often necessary to initiate a PES project development process.

In these cases, the government agency or NGO may act directly as scheme protagonists and also as overseers of project development. Alternatively, the protagonist may seek to establish a project steering group to oversee scheme development. The steering group is likely to include prospective scheme actors and stakeholders.

Support of independent scientific advisors (e.g. through appointment of specific advisors to those steering the project or establishing a formal scientific advisory board) can lend scientific weight to the process, increasing legitimacy and delivering scientific advice, support and resources.

4.8 Actors and stakeholders in PES projects

The main *actors* in PES schemes *and their roles* are:

- *Scheme protagonist*: This is a party interested to establish a scheme, which may be a lead buyer or seller, but may also be an agency of government or a conservation-NGO that sees policy and/or conservation benefits in getting a scheme in place.
- *Buyers*: These are the 'demand-side' beneficiaries and consumers of targeted services who might (actually or potentially) be willing-to-pay for services
- *Sellers*: These are the 'supply-side' land and natural resource owners or managers might (actually or potentially) be able to deliver additional services and be willing-to-accept payment for such delivery
- *Intermediaries*: These are agents who link potential buyers and sellers, help design appropriate schemes, and supply services important for implementing and running these. Intermediaries may act as the major proponents of PES schemes, often motivated by opportunities PES schemes offer for biodiversity and habitat conservation and enhancement.
- *Knowledge providers*: These are specialists in fields of knowledge or expertise relevant for scheme development; e.g. land and natural resource management experts, experts in valuation, spatial planning, regulation, law, etc.

Actor roles can be played by organizations or individuals. An actor may play more than one role; for example if the scheme protagonist is also a potential buyer or seller of services.

There are also wider *societal stakeholders*, including those who may be impacted by PES schemes albeit not being actively engaged in scheme development.

4.9 Phases and stages of implementation

Developing a PES project involves negotiating, implementing and managing a PES contract and, simultaneously, building trust among the involved actors and stakeholders. There are therefore two aspects of PES development and implementation: a process aspect, which is concerned with the negotiation processes through which the project is developed, and a content aspect, which is concerned with the technical design details; i.e. the specification of the PES contract, its terms, and how these are to be implemented and monitored. Both the process and the content elements are context-sensitive. Governance cuts across both.

The development of a PES project can be broken into phases, albeit these are not necessarily discrete and separate, but may involve iterations within and between phases to provide for modification and correction. The overall process is cyclical and may be a once-through cycle but is likely to involve a learning cycle such that all or parts of the cycle are repeated. This is more likely if there is uncertainty about the costs and benefits of delivering the contracted services, if sellers or buyers are risk-averse, or if the project starts from low levels of trust between the parties and trust has to be gradually built-up.

In such cases, there may need to be a pilot or demonstration project before a project can be rolled-out fully, although this will depend also on the significance of the scheme. Pilots and demonstration projects are more likely in the case of project that could have a large reach or where outcomes might influence the prospects for implementing comparable projects.

Broadly, PES project development may involve the following phases, although these may not necessarily follow in a set sequence:

- Prospecting; i.e. activities to establish that a PES project is feasible in principle. This involves checking that all required conditions are met. This is likely to be undertaken at a first scoping level by the potential scheme protagonist. The costs and benefits of alternative sets of ecosystem services will need to be assessed, at least at a provisional level (see Section 3).

- Project planning; i.e. activities to design the processes through which the project will be developed. This involves setting out the governance arrangements for the process, such as who will be involved, who will make decisions, and how decisions will be made; establishing the timeframe for the project planning phase; deciding on the actors to engage with, the terms of engagement, and the roles and responsibilities of actors in project development; deciding how to reach out to wider stakeholders; and, deciding whether to roll out the PES scheme fully or whether to develop it first through a pilot or demonstration project; etc.

- Establishing a counterfactual. This involves establishing how the ecosystem is likely to develop if there is no PES project. The counterfactual is needed as a basis for setting goals for the PES project, exploring how these might be reached, and developing monitoring protocols for project and progress evaluation and for assuring that the terms of the PES contract are being met.

- Risk assessment. This involves exploring the risks that any scheme might face, including the risk of buyers or sellers not committing (or being able to commit) to the project over the long term since this depends, also, on their own sustainability; the risk of projects being captured and held hostage by actors whose involvement is critical for project viability; the risk of creating perverse incentives; the risk of projects being perceived as unfair if they reward those most implicated in past loss of NC/ES; etc.

- Technical project design: This includes defining the goals and scope of the project; eligibility criteria for involvement in it; what safeguarding and risk management measures to take, the indicators to be used; the terms and conditions of payment; arrangements for measuring, monitoring, and performance verification; arrangements for handling contingencies, arrangements for adaptive management, etc.

- Contracting. This involves establishing a legal contract between sellers and buyers based on their negotiated agreement over the technical design of the project (content and structure) and agreements over implementation, monitoring, and dispute handling (processes).

- Implementation. This involves sellers undertaking ecosystem management interventions in accordance with the contract.

- Measuring, monitoring, verification. This involves processes to monitor that contractually-agreed ecosystem management interventions have been/are being undertaken or that contracted services have been/are being delivered.

- Assessment, lesson-learning, and adaptive management. This involves analysis of the effectiveness of the project and of project impacts, with lessons learned being used to correct the project design, enable full roll-out of the project, and/or provide transferable guidance for other PES projects.

Each of these phases is considered in further detail below, including which steps and activities each phase involves, what kinds of assessments (scientific tools and information) are needed to support the process at each step/phase, and what needs to emerge as output from each step/phase as an input to others.

4.9.1 Project prospecting

Conditions required for a PES scheme include:

- The existence of a potential to increase the supply of a target ecosystem service or services by undertaking specific land or resource management actions (interventions or management changes).

- Potential demand for the service must exist among one or more buyers and a willingness among them to pay a price that makes service provision financially viable.

- PES schemes operate on a principle of representing a win-win for both buyers and sellers: buyers must want the service and find that PES is the cheapest way to secure delivery; sellers must at least cover the costs of delivery including the opportunity cost; i.e. the value of any returns foregone as a result of implementing agreed interventions.

- For a PES scheme to emerge there must be a difference between the buyer(s)' willingness to pay and the seller(s)' willingness to accept. The actual price will be established between the minimum payment necessary to at least cover the sellers' opportunity cost and the maximum total (cumulative) value of all the benefits of the intervention to all beneficiaries less the transaction costs that buyers are likely to have to bear.

- There must be clarity over which land or resource managers can undertake actions to deliver the service(s).

An *ecosystem service assessment* involving an *economic assessment* of alternative ecosystem services (see Section 3) will be needed to support the prospecting phase, identifying which services might be provided, how, by whom, and at what cost. A *beneficiary analysis* may be needed to identify potential beneficiaries. *Valuation* of services may be needed to establish the value of benefits to beneficiaries.

4.9.2 Project planning and process governance

Project planning involves establishing the processes through which actors and stakeholders in a prospective PES project become engaged in the project development process and in negotiations leading to the prospective development and implementation of a PES contract.

The design of the PES project necessarily needs to involve and engage the prospective buyers and sellers, since the process involves both crafting and negotiating a mutually-beneficial scheme and, in the process, building mutual understanding and trust among buyers and sellers.

The fundamental task for the overseers of the project is to establish and implement a process for developing a viable PES project that commands confidence and support among both buyers and sellers.

A *PES project plan* may be developed that sets out the aims and objectives of the scheme, the rationale for the project, the arrangements for stakeholder engagement, communications actions to be taken, a timeframe and a timeline, the roles and responsibilities of the actors, etc. This will also specify governance arrangements for project and financial management.

The project plan should be aimed at delivering an agreed PES scheme design and an implementation plan within a specified time frame.

Issues to decide include how to make contact with prospective buyers and sellers and how to engage them in the project. Often *intermediaries* trusted by prospective parties to act as 'honest brokers' can play key roles in linking prospective *buyers* and *sellers*, enlisting support and contributions of *knowledge providers*, and arranging *participatory processes.* Depending on the envisaged geographical coverage of the scheme there may be a need for several intermediaries rather than only one.

Another important issue to decide in project planning is whether a project can be rolled out in full or whether there is first a need for a pilot or demonstration project. In situations where the scientific basis and evidence basis for scheme development is strong, it may be possible to plan for full scheme roll out from the start. When available information is insufficient to provide full confidence from the start, *pilot projects* offer scope to learn-by-doing, to fill information gaps, to support *adaptive management*, and to build stakeholder confidence and mutual trust.

4.9.3 Establishing a counterfactual

Establishing and agreeing a baseline position is a precondition for PES implementation. The counterfactual estimates the likely future provision of the targeted ecosystem services in the absence of any PES scheme and therefore forms the baseline against which the performance of any project is measured and against which payments are made.

Developing the counterfactual also delivers important information for PES scheme design, for example on context dynamics that could affect scheme performance over its lifetime in terms of cost-effectiveness, environmental effectiveness and equity (ScotWilson et al, 2011). By providing information on likely trends in ecosystem service provision into the future, counterfactuals also help establish:

- The magnitude of the challenge of meeting PES goals.
- The magnitude of the incentives needed (i.e. of payments required).
- Eligibility criteria for participation in the scheme and which are needed to assure *additionality.*
- A reference for monitoring scheme performance.
- A basis for making payments.
- A means to demonstrate the environmental effectiveness, cost-effectiveness and equity/fairness of the scheme.

Establishing a baseline against which actual performance can be compared is more involved than looking only at the level of ES at scheme start. This is because the level of ecosystem services is likely to change over the scheme duration owing to natural ecosystem changes and changes in policies, prices and climate. Such changes need to be taken into account by establishing a credible counterfactual that shows what will likely come about over the projected lifetime of a PES scheme if the scheme is not implemented. Counterfactuals are developed using *models* and *scenarios*.

4.9.4 Risk assessment

In designing and establishing a PES scheme there are important risk factors to explore including possible changes in the context for the scheme over its lifetime and the possibility that the project may have unintended and unwanted consequences.

Minimising risks requires that risks are analysed beforehand and findings of risk analyses are fed into PES project design to reduce risks and to incorporate measures to manage residual risk. If the long-term commitment of beneficiaries is in doubt, for example, it might be wise to avoid high up-front costs for scheme set-up. If the long-term commitment of providers is in doubt, it might be best to contract interventions that cannot easily be reversed. Some schemes also use restrictive covenants to minimise reversal risk. Covenants impose permanent or long-term restrictions on the use of the natural resource by owners or managers. These may proscribe environmentally-damaging behaviour (e.g. overstocking) or prescribe positive action (e.g. planting cover crops after harvesting). More generally, it might be prudent to contract several different services, not just one, so that schemes are not dependent on one major service or on one major buyer or group of buyers.

4.9.5 Project design

Once it is established that a scheme is possible in principle, the single most important influence on PES scheme design is the nature of the targeted service(s). In turn, key differences in the nature of the targeted service(s) and in related contextual factors create a need for different designs of PES scheme. Key design dimensions include:

Which services are targeted; what are the goals

PES schemes mostly target five broad ecosystem services or types of service: carbon sequestration; biodiversity conservation; watershed protection; landscape aesthetics; and cultural/recreational services (e.g. via public access). These differ according to:

- The category of service they represent (provisioning, regulating, aesthetic; cultural; etc.), which has implications, especially for *valuation* of services.

- The scale of service provision, which has implications for the *geographical coverage* of a scheme.
- Whether *supply of the service is specific* (provided by one supplier whose contribution is clear) or *diffuse* (provided by many suppliers whose individual contributions may not necessarily be clear), which has implications for deciding the payment basis and whether schemes are targeted (geographically and on specific sellers) or open.
- Whether *demand for the service is specific* (relevant to identifiable individual buyers) or *diffuse* (where benefits are enjoyed by many or the public at large), which has implications for *scheme financing* (who pays for schemes).
- Whether the service is a pure public good (non-appropriable, non-excludable, non-rival) or a private good, which also has implications for *scheme financing*.
- What combinations of services can be co-produced from the same parcel of land or body of water, which has marketing implications; e.g. for whether and how services might be *packaged*.
- The level of knowledge over the *management interventions* available to enhance service delivery and their effectiveness (i.e. *cause-effect knowledge*), which has implications for the *type of approach to payment* (*input-based* or *output-based*) and for scheme *lifetime*.
- Spatial variation across land areas, water bodies and other natural resources in their capacities to deliver particular services, in *threats* to service provision, in *opportunity costs* of safeguarding or providing additional services, in the value that services represent to beneficiaries, and in the number of potential beneficiaries. These have implications for the *type of approach to payment* (*uniform* or *differentiated* payment).

Spatial scale:

Depending on the nature of the service, these may be produced and consumed locally (e.g. water catchment services within the catchment) or the providers and beneficiaries may be geographically separated (e.g. carbon sequestration for climate change regulation). This means PES schemes can be developed at a full range of spatial scales:

- *International:* In respect to global-scale or internationally-relevant services, such as climate regulation (e.g. the REDD and REDD+ schemes for Reducing Emissions from Deforestation and Degradation in developing countries).

- *National:* In respect to national scale programs, such as the UK's Environmental Stewardship environmentally-sensitive farming scheme that pays land owners and managers for services of public benefit.

- *Catchment:* In relation to management schemes for water catchments (e.g. as defined under the Water Framework Directive), such as where downstream water users make payments to upstream land managers to secure clean water at point of extraction.

- *Local:* In relation to local or neighbourhood services, such as maximising recreational and aesthetic values from public urban space.

Single service projects versus service bundles:

PES projects can focus on a single service or on a set or services depending on what combinations of services can be co-produced on the same land parcel or water body. It is possible that all co-produced services generated from a single parcel of land or body of water are sold, but it is also possible that only the main (umbrella) services are sold, in which case beneficiaries of the other services benefit as free-riders. The marketable services can be packaged and sold either by:

- *Bundling:* Where a buyer or buyers make payment for a full package of co-produced services arising from the same parcel of land or body of water, etc.
- *Layering (*also called *stacking):* Where different buyers pay for different services co-produced from the same parcel of land or body of water, etc.

Buyer-seller configurations:

There are different possible buyer-seller configurations of schemes depending on how many buyers interact with how many sellers. Different possible buyer-seller configurations require different scheme designs and have implications for scheme *transaction costs* through the relative ease/difficulty of establishing and operating schemes. Possible scheme designs are:

- *One-to-one:* Schemes that combine one buyer with one seller.
- *One-to-many:* Schemes that combine one buyer with a group of sellers
- *Many-to-one:* Schemes that combine many buyers with one seller
- *Many-to-many:* Schemes that combine many buyers with many sellers

Scheme financing:

PES schemes can be differentiated also on how they are funded. The three broad types are:

- *Private payment* (also known as *user-financed*) schemes: Payments are made to land or natural resource managers by one or more private beneficiaries for beneficiary-specific services.
- *Public payment* (also known as *government-financed*) schemes: Payments are made to land or natural resource managers by government to secure or enhance ecosystem services that benefit the public at large.

- Public-private payment (also known as *mixed-finance*) schemes: Payments to land or natural resource managers draw on both public and private funds to enhance ecosystem services that offer both public and private benefits.

Whether a scheme is financed privately (by users) or publicly (by government/NGOs) depends on whether production and/or consumption of the service are specific or diffuse and whether it constitutes a so-called 'pure public good'. This leads to the different payment arrangements

In the case of ecosystem services that are private goods (where benefits are appropriable and excludable), markets for services can arise spontaneously between potential buyers and sellers, so PES schemes for such services can be user-financed. In these cases, private users pay directly for defined and specific services (*private-payment* or *user-financed schemes*).

In the case of ecosystem services that are pure public goods (where benefits are non-rival, non-excludable and enjoyed by diffuse beneficiaries, e.g. biodiversity conservation, landscape beauty) individual beneficiaries have no incentive to pay for services, so markets for services are unlikely to arise other than through government intervention. PES schemes for such services therefore tend to be government or NGO financed (*public-payment* or *government-financed schemes*).

Intermediate cases are also possible. In the case of watershed protection, services may be provided diffusely by upstream managers in catchments, but many water-related services are so-called 'club-goods' in that only those located downstream in the watershed can benefit from them, so downstream beneficiaries may club together to contract directly with service providers upstream to secure benefits.

Mode of payment:

Payments and contracts can be established for targeted ecosystem services directly (output-based payments) or for proxies (input-based payment). If the relationships between changes in ecosystem management practices, ecosystem service provision and welfare are well understood, contracts and payment can be established around delivery of the target services. If relationships are less well understood, proxy schemes can be established, for example based around delivery of changes in ecosystem management. Schemes using proxies are based on the assumption that the contracted changes in management practices will enhance delivery of target services.

Input-based payment schemes are more usual than output-based payment schemes because often they are easier in practice to implement and monitor than output-based schemes, as well as potentially more acceptable to sellers because they entail greater certainty. However, an input-based payment scheme depends on confidence among buyers that the contracted changes in management practices will deliver the targeted services.

Open schemes versus targeted schemes:

A basic distinction can be drawn between PES schemes that are *open to all* land and natural resource managers within a geographical area (more appropriate in spatially-uniform contexts) and *spatially-targeted* PES schemes, which target payments toward specific areas (more appropriate in contexts of spatial variation). Targeting can be on the basis of spatial differences in the benefits provided, the costs of provision, their value, threats to service provision, or any combination of these.

The capacity to provide ecosystem services, the threats to continuing provision, the costs of additional provision, and the value of services may all vary between places. This spatial variation is an important factor in the scope for PES schemes to be *cost-effective*, since schemes may be designed to target payments toward areas that can deliver sets of high value services (*hotspots*) or to where these are most threatened, areas where highest additional service benefits can be gained at lowest cost, and areas where many beneficiaries can profit from the additional services (e.g. as might be the case for recreational benefits of green infrastructure, which will overall be greater in areas closer to population centres where many people might profit from the opportunities it provides).

Spatial variation is therefore an important factor informing PES scheme design. Information on spatial variation in the potentials of ecosystems to deliver services and on threats to service delivery (used to identify hotspots) may come from *ecosystem assessments* and information on the values of benefits from *valuation* studies. The cost to land and natural resource managers of providing additional ecosystem services includes both the direct costs of undertaking the management interventions needed to secure the additional services and the opportunity cost of using the land or natural resource in its best alternative use; e.g. the loss of revenue from foregone agricultural production. PES schemes can be designed to establish a system of differentiated payments to reflect differences in these costs. Information on costs and how these vary spatially may be developed using *competitive tendering* approaches, such as *reverse auctions*. These are bidding or tendering procedures through which prospective suppliers of services nominate both the services they are willing to deliver (or management interventions they are willing to make) and the financial compensation they are willing to accept in return. Available funding is allocated to bidders and contracts are established on the basis of selecting bidders in the order of providing services that add greatest value at least cost.

These different types of information, coming from different sources and obtained in different ways, can be presented in an integrated way through *ecosystem service mapping*.

Uniform payment versus differentiated payment:

Depending on spatial variation in the capacity to provide services, costs of provision, and the value of services, PES schemes may be based on *uniform payments* (i.e. the same payment per hectare) or *differentiated payments* (i.e. payment is different for each hectare or unit of area based on the capacity of the area to deliver wanted services). Schemes based on differentiated payments have higher transaction costs and this can become important for schemes operating over large geographic areas for which the trade-off between scheme complexity and transaction costs may be important in cost-effectiveness overall.

Payment terms:

The terms of payment include the nature, level, and timing of payments.

- Nature of payments: Different payment arrangements are possible. These include that payments are fully *buyer-funded* versus *co-funded* by the buyer and seller; fully *money-bas*ed versus a *blended payment mix* involving money and in-kind support (such as help in securing government grants); and that payments are made for service bundles versus for unbundled (separate) services. Choices among these are influenced mostly by how the costs and benefits of interventions are distributed across buyers and sellers. If interventions deliver some benefits to sellers as well as to buyers, sellers may co-finance the intervention. If the interventions deliver co-produced benefits and the different benefits are of interest to different buyers it might be possible to 'unbundle' the services and to sell each to different buyers rather than sell the 'bundle' of services. This depends on the possibility of estimating the relative contributions of different management interventions to delivery of the different ecosystem service benefits and whether the different services benefit different buyers. In most schemes, buyers make payments for the bundles of coproduced services delivered from a site, but in some (so-called layered) schemes different buyers may individually pay for a specific unbundled service with each buyer. In this latter case, the interventions that deliver the overall set of services are paid for by combining payments from different buyers for the different individual services that the interventions co-produce.

- Level of payments: Owing to complexity and spatial variation in the demand and supply of ecosystem services, prices for many services are not uniform and universal, but context-specific and must be determined by negotiation and through market forces. The key issue is whether the price paid for services more reflects the costs of the service to sellers or the benefits to buyers. The 'gap' between these costs and benefits represents the range within which negotiated prices can fall. Schemes also have transaction costs, such as the costs associated with establishing the baseline and monitoring the scheme, which need to be covered by buyers and sellers. This narrows the range within which prices can be established. Precisely what price level emerges within this rang is influenced by market conditions, such as the degree of competition in both demand and supply.

- Timing of payments: Payments may be made 'one-off' or through a series of payments spread over the scheme lifetime. Payment timing is related to the time-incidence of the costs of delivering services. If the intervention involves front-loaded costs (e.g. to cover the upfront costs of ecosystem restoration) one-off or front-loaded payment is appropriate. However, if the scheme depends on ongoing interventions and continuous management of the natural resource, a series of payments is more likely to be appropriate. The timing of payments (and, also, the preference of sellers for schemes that are input-based rather than output-based) can be related also to time-lags between management interventions and changes in service delivery levels. By being mostly input-based schemes, PES schemes differ from those preferred in most other domains where payment by result is the norm. This is because costs of interventions are often front-loaded and there can be long time lags between intervention and provision of the relevant services. Many PES schemes therefore operate on the (unusual) basis of upfront or front-loaded and input-based payments.

4.9.6 Contracting

The next phase involves formalising the negotiated scheme design in the form of a legal contract between the buyers and sellers.

The formal legal contract sets out the negotiated design features of the scheme (scheme start- and end-date; geographical scope or footprint of the scheme; the relevant management inputs and/or service outcomes being contracted; what constitutes additionality; measures to minimise leakage; measures to ensure that interventions endure; how results will be demonstrated; payment terms; who pays the transaction costs; assignation of responsibilities for various tasks, (e.g. for monitoring); how risk and burden of proof are apportioned; rules for modifying the contract or its terms, including provisions for adaptive management; and accepted reasons for voiding the contract).

4.9.7 Monitoring, evaluation and review

Monitoring is an integral part of any PES scheme, since payments are established for delivery of specific inputs and/or outputs and there is a need to verify delivery of these or progress toward delivery. Regular monitoring provides for trends in levels of provision of relevant services to be established compared with the baseline projection and with the additionality that is anticipated.

Monitoring is nevertheless multifaceted. It feeds into the sister processes of periodic evaluation and review. These allow any deviations from anticipated and targeted service levels to be identified and diagnosed. In turn, these support adjustment of the specific scheme (adaptive management). Monitoring and evaluation also play roles in building knowledge about the relationships between management interventions and service delivery, delivering information useful more generally for developing transferable lessons, guidance, and inspiration for other PES schemes.

Monitoring for *verification purposes* addresses several concerns, including that: contracted interventions or services are being delivered; management interventions are enhancing service delivery as anticipated (or prospects of service delivery); there are no unacceptable adverse impacts of the scheme on delivery of other ecosystem services (trade-offs) or leakage; and that statutory obligations and regulatory requirements are all being met. These concerns affect the scope of monitoring, which – to include trade-offs, spill-overs and other possible unintended impacts – may need to reach-out further than the geographical boundaries of the scheme itself.

The design of an appropriate monitoring regime involves balancing the costs of monitoring against the quantity and quality of the information monitoring delivers. The appropriate balance depends in each case on the information needs of the actors in the scheme, their willingness to pay for monitoring, and the characteristics of different possible monitoring schemes that could be used. The size of the market is also relevant. If the market is larger is might support higher overall monitoring costs if these translate to low average costs per actor.

Monitoring arrangements should be cost-effective, deliver required levels of accuracy and timeliness, and support confidence in the scheme.

If buyers or their stakeholders (e.g. regulators, auditors) need a high degree of assurance that the scheme is delivering as anticipated, independent third-party monitors and/or certification may be used.

Monitoring draws on the baseline already established. It makes use of relevant *indicators*. These can focus on:

- Levels of ecosystem services: to ensure the management changes are enhancing service provision (direct)
- Management measures: to assure that ecosystem managers are undertaking contracted management measures (indirect)
- Socio-economic impacts: to ensure that the welfare of participants is improved.

Whether or not the PES scheme is established as a direct or a proxy scheme, proxies may be used in monitoring. Direct monitoring of ecosystem service outputs may be difficult (e.g. because the systems producing ecosystem services are complex and because cause-effect linkages may involve time lags) or costly. Many PES schemes therefore rely on more easily-observable proxies, such as management actions, or intermediate outcomes, such as increase in forest cover.

Different monitoring and verification options include: direct measurement; the use of proxy indicators; and modelling. In the case of climate regulation, for example, monitoring and verification of a PES scheme aimed at carbon sequestration through afforestation could be monitored by directly measuring tree growth and carbon storage, by modelling fluxes of climate-relevant atmospheric gases, or indirectly by using tree planting as a surrogate for carbon sequestration. In the case of flood risk mitigation monitoring might involve directly measuring the water storage capacities of soils, modelling run-off and water flow rates; or using riparian tree planting as an indirect (surrogate) indicator.

Typically, costs are highest for monitoring regimes using direct measurement, since these involve establishing detailed measurement and full sampling programs. Costs are least for regimes using indirect (surrogate) indicators, since these are specifically chosen for availability of data or relative ease of data collection as well as the indicator having some known relationship to the contracted intervention or service. Modelling may, in some contexts, offer an intermediate option.

Direct monitoring is likely to be most appropriate when risks are high, buyers need a high degree of certainty that contracted interventions or services are being delivered, and buyers are therefore willing to pay for accuracy. It is appropriate also where monitoring costs can be shared over many buyers, where relevant data are being collected already as part of existing (and potentially adaptable) monitoring systems, and when an accurate overview can be obtained using data from relatively few sampling points and/or from infrequent surveys, for example as might apply when change is expected to be slow and homogeneous across the sampled area.

Indirect monitoring is likely to be most appropriate when other methods are constrained or precluded for financial or technical reasons (e.g. lack of data, models, skills available locally), when there are known and demonstrated relationships between proxies and the service changes of interest, and when risks and uncertainties are low and confidence can be maintained using proxy data and approximations. It is appropriate also when schemes and markets are small and there are few buyers to share costs.

Modelling offers an intermediate solution that may be suitable in cases where the required level of accuracy is greater than can be satisfied using proxies, but direct monitoring would be too costly. Modelling is most appropriate when suitable models of the system under management exist already and the proposed interventions can be integrated into the model or when a customised model can be built relatively easily, e.g. because the necessary data, biophysical knowledge, and modelling expertise are available.

In respect to some services, rapid advances in some monitoring technologies are simultaneously reducing monitoring costs and making possible more accurate and more frequent monitoring, including continuous real time monitoring of forest growth and carbon sequestration and storage rates using remote sensing and automated data analysis.

4.10 Lessons for PES project design

Table 4.1 develops and summarises lessons from experience to date with PES projects, including reflections on trade-offs between different aspects of scheme performance, which need to be taken into account in project design. The table comments on the relationships between aspects of context, project design, and performance.

Table 4.1 Lessons and guidance for PES project design

Aspect relevant for scheme design	Risks and opportunities	Relevant contextual diagnostics	Lessons	Trade-offs	Which outcomes are potentially affected	Design Implications
Structure of marginal benefits of the ES (constant versus variable ES benefits)	Simple schemes will not be environmentally effective if the marginal environmental benefits of the ES are not constant. More complex schemes are needed to avoid risks of schemes being environmentally ineffective.	Presence or absence of threshold effects in delivery of the ES.	The design of environmentally effective schemes and policies is more straightforward when marginal environmental benefits are constant across sources.	To take non-constant marginal benefits into account schemes can be designed based on ambient permits, differential taxes, or trading zones, etc., but these increase design complexity and cost.	Environmental effectiveness Cost of schemes and cost effectiveness	Degree of complexity of PES scheme design is linked to structure of marginal benefits. If marginal benefits are not constant, schemes will need to be more complex.
Capacity to measure benefits accurately	When direct benefits are costly or impossible to measure, schemes can be implemented using proxies. But environmental effectiveness depends on relationship between proxy and ES benefit.	Measurability of direct benefits; measurability of benefits indirectly using proxies; strengths of relationships between proxies and ES benefits; knowledge of these.	The design of environmentally-effective schemes is easier, and implementation costs are lower, when the relationships between management activities, ecosystem functions and ecosystem services are well understood and strong (e.g. between planting trees, carbon storage and climate stabilisation).	Depending on the certainty and strength of the relationships between the proxy and the desired ES, the use of proxies trades off scheme feasibility and cost against the risk of incentivising activities that may not be very effective environmentally.	Environmental effectiveness Cost of schemes and cost effectiveness	Advances in techniques for estimating ecosystem services from easily observable ecosystem properties are important for the long-run viability of PES schemes (Jack et al, 2008).

Table 4.1 Lessons and guidance for PES project design

Whether a PES scheme is warranted (over other approaches). Heterogeneity of costs of ES provision across potential suppliers.	Heterogeneity gives scope to achieve a given level of environmental service benefit at variable cost, depending on how provision of environmental benefits is arranged. Opportunity arises to save costs. The risk is one of failing to save avoidable costs.	Scope for cost saving depends on the extent of differences in ES providers' costs. Likely sources of differences in costs include: biophysical and locational features of the ecosystem associated with differences in opportunity costs; features of landholding (such as plot size); and features of landholders (such as education, risk aversion, etc.)	The potential for schemes to achieve significant cost savings for society is greater the greater is the heterogeneity in costs. Offering set payments for service provision or using a reverse auction (tendering) will attract suppliers into schemes whose opportunity costs are lowest, ensuring that society as a whole gains the same amount of ecosystem services at least cost.	Higher transaction costs of complex schemes versus the greater scope these offer over simpler schemes for saving costs in delivering environmental benefits.	Cost-effectiveness of schemes and level of savings achievable.	Under conditions of heterogeneity in costs, PES schemes are warranted relative to mandatory or uniform approaches. More complex PES schemes are more likely to be needed and warranted under conditions of heterogeneous costs. Payment levels can be set (and gradually increased if needed) to deliver target level of protection.
The scope for designing/delivering integrated policies	PES can offer opportunity for policy integration by coupling environmental protection with poverty alleviation policy.	The degree of coincidence between poverty, capacity to provide a high level of wanted ES, and relative opportunity costs of ES provision	PES policies are most likely to help alleviate poverty when the poorest hold land capable of delivering high value ES and also face the lowest opportunity costs of ES provision	PES schemes are likely to make a significant contribution to poverty relief only if they pay amounts substantially higher than opportunity costs. This implies a trade-off between the goals of cost-effectiveness in achieving environmental objectives and the scope for achieving poverty alleviation objectives.	Cost effectiveness Equity (policy integration objectives, income redistribution, poverty relief).	Scope to use PES for policy integration (i.e. for coupling environmental protection and poverty alleviation policies) depends on coincidence of poverty, a high ES potential, and low opportunity cost.

Table 4.1 Lessons and guidance for PES project design

Transaction costs for monitoring and enforcement	Monitoring & enforcement costs are higher for: area-based schemes; schemes involving a high number of individual, small-scale landowners; and schemes where collective activities alter the level of provision of a given ecosystem service. This could limit the situations when PES schemes can be used for coupling environmental and poverty alleviation agendas. Working through third party intermediaries may avoid some costs.	Resource ownership (number and scale of actors). Level of interdependence of actors in delivering ES. Existence and capacities of local community organisations and NGOs as potential intermediaries.	Scheme costs will increase as a function of number of agents and the degree to which ES provision is based on collective effort.	These cost structures imply a potential trade-off between cost-effectiveness and poverty alleviation objectives.	Cost effectiveness Equity (policy integration objectives, income redistribution, poverty relief).	When seeking to use PES schemes as mechanisms for implementing integrated policies involving poverty alleviation (which often implies working with large numbers of ES providers), transaction costs may be contained by working with a third-party local intermediary.
Structure of the funding mechanism. Voluntary and privately funded schemes versus mandatory or publicly funded schemes. Approaches to raising private/public funds:: market creation/support, taxation of beneficiaries, tradable development rights, etc.	New opportunities to provide private funding to back demand for public goods are created by mechanisms for transferable development rights where developers pay to 'set aside' land in one location in exchange for government allowing more intensive development elsewhere.	Whether ecosystem services are private (excludable goods) or public (non-excludable goods).	When services are linked to excludable goods (private), beneficiaries buy the service directly through the market. When services are non-excludable (public goods) either compulsory mechanisms for demand generation or government funds and payment schemes are needed.	How funds are raised has distributional and political economy consequences.	Cost effectiveness Equity, distributional consequences, impact on the political-economy.	Constraining factor on PES schemes to deliver public benefits is not lack of latent demand for ES, but lack of effective demand. Scope for using PES schemes to deliver public goods is likely to be linked to development and success of mandatory mechanisms for enlisting private funding.

Table 4.1 Lessons and guidance for PES project design

Political viability/feasibility	PES approaches are favoured by resource holders because they provide compensation for environmental improvements and participation is voluntary. However they risk opposition from those excluded from payments under the scope of the scheme and from those philosophically disinclined to PES on grounds of 'neo-colonialism', 'privatisation of nature', etc.	Viability is related to the preferences and powers of stakeholders, including providers of ecosystem services, beneficiaries, policymakers, financiers, community members and program administrators. Preferences may be influenced by more than just economic concerns; for example moral and philosophical perspectives concerning PES may be relevant. The distributions across stakeholders of costs and benefits of prospective schemes and of political influence/power.	Political feasibility depends on the political power of those who bear the costs and benefits of schemes.	Trade-offs may be implied in securing political support for a scheme. It may be necessary to widen the scope of potential beneficiaries under a scheme in order to enlist support for it, which may reduce cost-effectiveness.	Cost effectiveness Equity, distributional consequences, impact on the political economy.	The perspectives on scheme design of all stakeholders holding preferences and power needs to be taken into account in order to assure that schemes are viable. Alternative scheme designs should be compared on the basis of how costs and benefits are distributed under each alternative. To increase viability, PES scheme designs may need to be shaped to accommodate political considerations. Schemes providing for more widely distributed program benefits may be more viable politically. Nongovernmental entities may have better chances of implementing PES schemes because they are perceived as more likely to be neutral and because they may introduce external funds, relieving local communities and tax payers from the costs of environmental protection.

Table 4.1 Lessons and guidance for PES project design

Establishing need for a PES scheme. Avoiding conflicts with policies and instruments in place	The scope for interaction between PES schemes and existing policies and instruments offers opportunities for complementarity, but also risks of conflict; e.g. when existing policies increase opportunity costs of providing ES.	Presence of conflicting policy goals/instruments (especially subsidies) that undermine the effectiveness and cost effectiveness of the PES.	Removing existing perverse incentives may obviate the need for a PES scheme and, in the event a PES scheme is still needed, will increase the effectiveness of PES incentives.	There can be conflicts between policy goals established and pursued at different levels; such as between national economic development goals and international environmental protection goals.	Environmental effectiveness Cost effectiveness	Existing (perverse) policy incentives should be reviewed and eliminated before establishing the need for a PES scheme.
Design of incentives Avoid incentives that induce perverse behaviours and impacts, including displacement of environmental pressure and 'ransom' behaviours.	PES schemes that are successful in securing ES in one location may displace environmental stress to other locations, reducing the overall environmental effectiveness of the scheme. PES schemes may also induce participant behaviour aimed at securing higher compensation, such as threats of environmental destruction or actual acts thereof.	Comparability of ecosystems in locations within and without the scheme and connectivity of management regimes; e.g. through participants and through markets. The opportunity to displace pressure depends, inter alia, on: whether beneficiaries own other plots; whether they are credit-constrained and whether this constraint is lifted by the payments; and the scale of the PES program and its impact on regional agricultural prices.	If there is a fixed-factor of production or if schemes and impacts on regional agricultural prices are small, there is less likelihood of displacement effects. To avoid 'ransom behaviours' PES policies and schemes can be based on historical baseline data and on incentives linked to levels of activities (proxies) rather than on environmental changes.	Providing incentives for levels of activities (proxies) may create trade-offs between avoiding ransom behaviour and paying landowners for activities that might have occurred in the absence of the program (44).	Environmental effectiveness Cost effectiveness	Secondary effects, such as displacement of pressure, need to be taken into account in scheme design (to minimise the risk) and also when measuring scheme effectiveness (the environmental benefits gained overall).

Table 4.1 Lessons and guidance for PES project design

Design of incentives Building innovation into incentives	Depending on their design, PES schemes may (or may not) offer incentives for actors to develop or adopt innovative approaches to providing ES at lower cost. Opportunities and risks depend on whether this potential is taken up.		Innovation is most likely when rewards are tied to marginal improvements in actual environmental performance, when schemes offer flexibility over how ES are delivered (rather than when schemes prescribe activities), and when actors perceive that schemes and incentives are likely to be long-lived, making innovation worthwhile (16).	Most PES schemes and policies base rewards on proxy action rather than on production of final ecosystem services. The incentive to innovate is therefore not linked directly to provision of ES but to innovation in prescribed proxy activities. This trades-off cost of monitoring against scope to deliver wanted ES at lower cost.	Environmental effectiveness Cost effectiveness	
Design of incentives Building flexibility into incentives	Technology and price changes will affect the cost of delivering ES. The possibility to capitalise on opportunities for cost reduction or to be resilient in the face of cost increases (and, therefore, also the environmental effectiveness of schemes, their cost-effectiveness and the cost savings potential for society in the delivery of ES) all depend on building flexibility into schemes and incentives.	The need for flexibility is greater if technology or prices (or other contextual factors affecting costs of delivery of ES) are dynamic and if long-lived schemes are being designed.	Allowing a variety of ways to comply with an incentive-based approach (thus enabling participants to switch away from more expensive approaches in the face of price increases) will help ensure schemes are effective even if contexts change. Increasing the range of allowable proxies or directly rewarding the ultimate ES offers participants flexibility to adjust to price and technology dynamics by choosing the lowest cost ways to deliver ES.	Most PES schemes and policies base rewards on proxy action rather than on production of final ecosystem services. Using proxies potentially involves trade-offs between monitoring cost on the one hand and environmental effectiveness and cost effectiveness on the other hand. Increasing the range of allowable proxies also trades-off monitoring cost with the environmental effectiveness, cost effectiveness and cost savings potential of schemes.	Environmental effectiveness Cost effectiveness	Increasing the range of allowable proxies or directly rewarding the ultimate ES offers participants flexibility to adjust to price and technology changes by choosing the lowest cost ways to deliver ES.

5. Biodiversity offsetting

5.1 Definitions

Different definitions of biodiversity offsets are available. A simple definition is that biodiversity offsets are "*conservation activities designed to deliver biodiversity benefits in compensation for losses, in a measurable way*". A fuller definition is that biodiversity offsets are "*measurable conservation outcomes resulting from actions designed to compensate for significant residual adverse biodiversity impacts arising from development plans or projects after appropriate prevention and mitigation measures have been taken*". The goal of biodiversity offsets is "to achieve no net loss and preferably a net gain of biodiversity on the ground with respect to composition, structure, function and people's use and cultural values associated with biodiversity."

These definitions clarify that:

i. Biodiversity offsetting involves counterbalancing or compensation activities for losses of biodiversity due to development.
ii. Biodiversity offsetting is the last step in the mitigation hierarchy that developers must follow in the design and implementation of projects with impacts on habitats and biodiversity. It is to be used only after other appropriate actions have been taken to avoid adverse impacts on biodiversity and/or to reduce them, whether these are mitigation measures recommended following environmental impact assessment of projects (EIA) or conditions of planning consent.
iii. Offsets are distinguished from other forms of ecological compensation by their formal requirement for measurable outcomes and their explicit requirement for achievement of 'no-net-loss' to be demonstrated with respect to particular impacts.
iv. Achievement of 'no-net-loss' is a minimum requirement for offsets, but they are often associated with an aspirational goal of achieving a net benefit for biodiversity.

The concepts of 'biodiversity offsetting' and of 'no-net-loss' are based on assumptions about the substitutability of biodiversity. Offsetting implies that loss of biodiversity in one location can be substituted by gains in another location. Designing biodiversity offsetting schemes involves making decisions over the acceptable scope to substitution and its limits.

5.2 Relevance of ES/NC concepts for biodiversity offsetting

Offsetting has been applied successfully in other areas of conservation, especially water and air pollution (carbon). Interest, currently, is focussed on extending offsetting to habitats and biodiversity. Using offsets for habitat and biodiversity is more challenging than in these other areas of conservation because habitat and biodiversity are heterogeneous; i.e. whereas a unit of carbon is the same everywhere and a unit of carbon emitted at one location can be compensated for by a unit of carbon sequestered anywhere, the mix of habitat and wildlife is unique at any specific site. Operationalising biodiversity offsetting therefore depends on choosing quantifiable metrics or selecting surrogates/proxies to represent the habitat and biodiversity present at a site and using these to measure impacts and offsets that are considered to be 'ecologically equivalent' to each other, though not identical.

Metrics and currencies:

For straightforward and cost-efficient processes there is a need for 'workable' biodiversity currencies, but there are risks inherent in defining biodiversity too narrowly (over-simplistically) and failing to provide offsets for key biodiversity components or values. The assessment of biodiversity lost due to development or gained through an offset should ideally give consideration to both pattern (structure and composition) and process (functionality). Area-based metrics (hectares impacted or offset) cannot account sufficiently for the biodiversity supported by the land and its quality.

Using the ES/NC concepts offers scope to simplify the complexity by providing functionality-based metrics that enable impacts on one site and compensatory offsets at another site to be assessed in terms of ecosystem services and the values these represent to people. There is increasing interest therefore in how ecosystem services might be used as a currency as opposed to measures of biodiversity itself or simple area-based measures of losses and gains in habitat.

Biodiversity markets:

The ES/NC concepts also offer other potential advantages in operationalising biodiversity offsets.

Biodiversity offsetting arrangements involving relatively simple 'like-for-like' and 'local' compensations limit the potential to optimise biodiversity and habitat conservation benefits from offsetting. Ecological and economic effectiveness can both be improved, in principle, by establishing biodiversity markets that provide flexibility by offering a wider set of compensation options in terms of currencies, geography, timing and the scale of offsets.

Pooling financial resources from many small development projects can provide consolidated investment funds that might support larger-scale and more ambitious restoration and re-creation projects than can be supported by direct compensation. Bigger schemes are also easier to control, monitor and regulate and they give scope to secure more significant and more long-lasting biodiversity benefits.

The ES/NC concepts can be the basis also for habitat banking systems. These provide opportunity for land holders to offer high value potential sites for restoration or (re)creation as 'offsets' and to receive compensation for providing these in advance of impacts on habitat and biodiversity arising from development projects, so avoiding temporary ES/NC losses while waiting for offsets to become ecologically effective.

Spatial mapping of ecological value-adding potential is used to identify where investments in restoration, re-creation and the creation of green corridors and networks will deliver highest biodiversity and ES-functionality returns (see: section on Green Infrastructure).

Appropriate use of biodiversity offsets:

Offsetting is based on the concept of substitutability of natural capital and ecosystem services. There are limits to this substitutability. It is therefore necessary to distinguish between habitats that represents critical natural capital and ecosystem services, for which impacts cannot be offset, from habitat representing non-critical natural capital and ecosystem services, for which impacts can be offset. Offsetting is only appropriate for non-critical ES/NC. Where to place the threshold between critical and non-critical ES/NC is therefore an important framing issue for implementing schemes of biodiversity offsetting.

Offsetting is based on the polluter-pays-principle. The polluter-pays-principle is the basis also for eco-taxes. Offsetting is useful when impacts are: immediate, direct, local and traceable to a specific project; when they are measurable; and when they can be compensated for effectively through schemes of direct restoration or re-creation of equivalent habitat elsewhere; i.e. where the damage is clear and responsibility for it is clear-cut. This is the case for development projects, which makes schemes for direct compensation of damage (offsetting) more suitable than eco-taxes in these cases.

Biodiversity offsetting is more suitable for addressing local damage caused by discrete events while eco-taxes are more suitable for addressing contributions to aggregate and cumulative stress arising from continuous actions. An important consideration, therefore, is whether the damage to be compensated is associated with a one-time 'discrete' event or act, that brings irreversible, local structural change, or to continuous, on-going activities and processes. For example: agricultural activities exert continuing low-level cumulative stress (e.g. through fertilizer application) and are associated with impacts that are not just local. These are best addresses through eco-taxes. A development project that changes how land is used, by contrast, has immediate, clear, measurable and local impacts that are traceable specifically and only to the development project. The impacts of such projects can be addressed through biodiversity offsets.

Biodiversity offsetting is a more sure mechanism for achieving compensation compared with eco-taxes. Offsetting provides for impacts to be compensated directly by (re)creating or restoring habitats or indirectly by investing in projects of habitat (re)creation or restoration. Eco-taxes, however, are not necessarily reserved for making good the damage for which the tax is levied.

Biodiversity offsetting and the possibility to create markets for habitat and biodiversity offers important potential advantages:

- *Ecological effectiveness:* Biodiversity offsetting can help achieve overall conservation goals by implementing principles and standards defined by policy, such as no net loss (NNL). A No-Net-Loss principle is the basis for core overarching policy references concerning habitat and biodiversity conservation and is the objective of the International Convention on Biodiversity. Used strategically, compensation payments can help deliver greater than NNL outcomes; i.e. net positive ES/NC gains

NNL is a multifaceted concept. It specifies a threshold level for biodiversity that is not to be transgressed. This is established against the baseline of current levels of biodiversity. Biodiversity offsetting translates this macro-level threshold into a design and decision criterion for local-scale projects with impacts on habitat and biodiversity by requiring that losses are compensated for by gains elsewhere. Biodiversity offsetting can therefore help maintain the overall ecological functionality of the landscape in the face of cumulative development pressures and cumulative losses of habitat and biodiversity arising from many development projects, both large and small, which have impacts on habitat and biodiversity of any type. Biodiversity offsetting elevates the profile of biodiversity conservation of non-designated sites and features, responding to threats to the natural environment outside protected areas and extending compensation for impacts to a broader spectrum of biodiversity and habitats.

Biodiversity offsetting can be further extended to provide a mechanism for mitigating pressures arising from development for which planning permission is not required (e.g. when impacts considered in isolation appear insignificant, but which add to aggregate losses) and from failures to implement and monitor mitigation measures when these are recommended in environmental impact assessments of developments.

- *Economic efficiency*: Biodiversity offsetting offers an avenue for market creation for biodiversity, correcting a market failure. This can help make markets more efficient. By internalising the costs of conservation, biodiversity offsetting can help correct information failures. Through permitting procedures, ES/NC impacts and offsets can also be integrated indirectly into certification and labelling schemes; for example, to help developers with impacts, regulators and offset providers work together more effectively. The ES/NC concepts can also be incorporated into financial and investment decision making through changes in financial accounting; for example, by integrating the actual or likely cost of offsetting ES/NC impacts into the balance sheets of organisations and into risk analyses for assets or investments.

- *Fairness/Equity:* Biodiversity offsetting ensures that those having a significant residual impact on biodiversity make good this impact and bear the cost. It enables payments to be made to land holders in return for conservation management and outcomes, which increases fairness.

- *Reducing uncertainty*: Biodiversity offsetting could reduce uncertainty over outcomes of planning consent applications, helping to streamline planning processes by making decision making more consistent. Using a consistent approach can provide clarity to developers about the level of compensation required and how compensation can be provided. It can also increase the transparency of decision-making by allowing people to see how compensation requirements are calculated and how offset resources are used.

- *New investment streams*: Biodiversity offsetting can be achieved using novel types of financial mechanisms and instruments. These support the development of dedicated funding streams for investing in habitat and biodiversity. Biodiversity offsetting gives economic value to habitat restoration and re-creation, creating economic incentives for landowners to undertake conservation activities. A system of biodiversity offsets can also provide opportunity for developers and other interested organizations to buy additional conservation credits as part of their social responsibility commitments.

Developers of projects with impacts that cannot be mitigated can be encouraged or required to compensate for residual impacts. This ensures that the full costs of projects are borne by developers and, ultimately, by the beneficiaries of the development projects. Simultaneously, this secures a new dedicated stream of investment funds for habitat and biodiversity compensation projects. A requirement to compensate for impacts that cannot be mitigated also creates an incentive for developers to locate new development projects where they will do least harm to habitat and biodiversity, so addressing fundamental sources of pressure and stress on habitats and biodiversity.

Compensation costs are related to the costs faced by the offset provider in providing the offset, which include the opportunity cost of the offset site and the costs of interventions needed to re-create or restore habitat. They include, also, transaction costs of scheme design, implementation and verification.

Risks of biodiversity offsetting:

The fundamental assumption of offsetting is that (some) habitat and biodiversity is substitutable; i.e. that losses of habitat and wildlife in one area can be compensated for by (re)creation or restoration of habitat and biodiversity elsewhere. As well as offering important potential benefits, biodiversity offsetting therefore involves some risks, which have to be managed; for example: not all types of ecosystem and not all ecosystem functions can be compensated. The time taken for restoration or re-creation of different types of habitat or ecosystem function varies and can be very long. The core implementation issues involve recognising the limits to substitutability and responding effectively to the implied risks.

The main risk in biodiversity offsetting is that it becomes a ‘licence to destroy’; i.e. a mechanism enabling developers to circumvent existing habitat and biodiversity protections. A concern is that the possibility of biodiversity offsetting could lead to a relaxation of development controls, by providing for development projects to go ahead in cases where they might otherwise have been considered unacceptable due to the magnitude or severity of impacts. The downside risk is that poorly designed or poorly implemented schemes could increase rates of biodiversity and habitat loss, which would be counterproductive to the conservation intent. Protecting against that risk involves maintaining existing habitat and biodiversity protections and adhering to the principles of the mitigation hierarchy, with offsetting as the last step to be taken once avoidance and mitigation possibilities are exhausted.

Since not all habitat and biodiversity can be substituted, it is necessary to distinguish types of habitat and biodiversity that are not substitutable from types that are and to develop and apply rules and mechanisms for compensation appropriate for habitat/biodiversity of different types and in different situations presenting different kinds and levels of substitution (or 'delivery') risk.

Implementation governance, design and management are concerned with capturing and maximising the benefits of biodiversity offsetting while minimising misuse and delivery risk.

5.3 Policy references, drivers and triggers

The principle of NNL for all biodiversity is enshrined in international commitments and targets under the Convention on Biological Diversity. Biodiversity offsetting is already mandatory in the EU in respect of high value habitats and species designated and protected under the Nature Directives. At issue is the extension of mandatory offsetting to other contexts, such as routine development applications.

The main vehicle for the extension of biodiversity offsetting in the EU policy context is the EU No Net Loss Initiative (delayed). The goal of mitigation is (increasingly) to achieve 'no-net-loss' of biodiversity. Main options include: voluntarism based on stronger guidance or regulations mandating those with impacts on habitat or biodiversity to demonstrate NNL or better. The ultimate sectoral scope and reach of mandatory offsetting are still under discussion.

The trigger for mandatory biodiversity offsetting is at point of performing Environmental Impact Assessment and/or applying for development permission.

The current status of biodiversity offsetting

The EU has built requirements into the Birds and Habitats Directives in relation to the integrity of the Natura 2000 network. Biodiversity offsetting for wider applications than those currently mandated is under development. Overarching principles for implementing biodiversity offsetting and the evidence base for scheme design are being developed using pilot projects and evaluations of these.

Some EU Member States (e.g. Germany) have developed specific laws and regulations requiring offsets. In Germany the mitigation hierarchy is applied to impacts of projects on nature and landscape involving avoidance (preventive approach) and compensation for unavoidable impacts (corrective approach) under the Impact Mitigation Regulation (1976) to try to preserve the status quo. It has been noted that this has a very broad field of application to ecosystems, ecosystem capacities and natural scenery. Implementation begins with the identification and evaluation of the impacts of a project, plan or action on nature and the landscape in terms of their significance. The broad meaning and scope of nature, ecosystem and landscape and the comprehensive spatial approach imply that most actions subject to authorisation are obliged to carry out an EIA, but equally a great deal of importance attaches to what constitutes ‘a significant impact’ and requires an offset.

Wider international experience has involved different approaches to requiring and implementing NNL in respect to the sources of legislation within the governance hierarchy, the types and roles of different of legal instruments (conservation laws, planning laws and regulations, guidance), the biodiversity components addressed, the possibility of offsite mitigation and of involving third parties in mitigation, and approaches to monitoring and safeguarding. Some approaches require offsets under certain highly prescribed conditions. Others are more general in their approach and leave greater scope for interpretation in specific situations. Limited use has been made so far of mitigation banking as a mechanism to deliver offsets, but interest in it is increasing.

United States Federal Law requires both public and private sector developers to implement the mitigation hierarchy in respect to impacts of land development projects on certain types of habitat (wetlands, aquatic ecosystems, habitats of endangered species) as a precondition for obtaining permits. A key provision of the laws is the possibility for off-site mitigation by third parties where public authorities (state level) determine this is feasible and appropriate. Authorisation at state level has led to the widespread development of habitat or mitigation banks.

In Australia offsets are required under the Environmental Protection and Biodiversity Conservation Acts and under the conservation and planning laws of several states and territories. In the State of Victoria the mitigation hierarchy is invoked for impacts on native vegetation. Following avoidance and minimisation, permits and ‘native species offsets’ are required under planning law to clear native species. Losses and gains are calculated using a simple ‘habitat-hectares’ metric. The State has developed a set of support mechanisms including a computer-based system for matching offset requirements and for tracking and quality control of transactions (the Native Vegetation Credit Register).

5.4 Governance, actors and roles

Capturing potential benefits depends on taking a strategic perspective to the governance, planning, and design of biodiversity offsets.

In terms of governance, this involves a multilevel approach with schemes designed and managed at the local level as far as possible, but within a consistent Green Infrastructure strategy framework that can include: national priorities; a standard framework of principles, guidance and tools to provide levels of consistency for all those involved; and new national support institutions, such as habitat banks and one or more organisations to support long-term management of offsets and to verify offsets (see: section on Green Infrastructure). Important governance principles for designing and managing effective biodiversity offsetting schemes include subsidiarity, transparency, participatory approaches and management through partnerships that make sense spatially and ecologically, for example based on using natural areas and catchment boundaries.

In terms of strategic planning, there is an important role for local authorities. The importance and value of habitats needs to be integrated into local planning processes and recognised under planning policy. Important steps are to develop strategic plans for habitat and biodiversity development and to develop offsetting strategies and priorities. These should be agreed between local authorities, their conservation partners, developers and community members. They should clarify how offsetting is to be applied in the area, what is to be achieved, and what is expected of developers. Local development plans can also be assessed against the NNL principle to ascertain whether planned development within a territorial unit is consistent with the NNL constraint, whether local compensation opportunities are sufficient or whether there might be a need for offsets in locations beyond the boundaries of the local authority.

Maximising real biodiversity benefits involves designing schemes that provide additionality, for example, by: expanding and restoring habitat and not only protecting the condition and extent of what already exists; using offsets to contribute to enhancing ecological networks by creating more, bigger, better and joined up areas for biodiversity; and creating lasting habitat. These can be achieved by pooling compensation across a number of separate, small development projects into a larger 'habitat block' to create more effective and beneficial areas of habitat.

The task of achieving the NNL requirement, where mandated, falls largely to local governments and to spatial planning authorities, since this is the level at which spatial and developmental plans are developed and at which development project applications are received and evaluated through permitting (licencing) procedures.

5.5 Design of biodiversity offsetting schemes

A meta-level question concerns the level of ambition for biodiversity offsetting along the range from simple independent schemes of direct compensation to more sophisticated schemes involving indirect third-party compensation mediated through biodiversity markets. Market-based schemes involve the creation of market instruments, such as biodiversity credits and, potentially, the establishment of habitat banks. The design and development of individual schemes and the development of wider markets for biodiversity offsetting are inter-related, since each contributes to the other. As biodiversity offsetting becomes more mainstreamed, the more advantageous it will be for schemes to be developed within the overall framework of a biodiversity market, since this will make trades more efficient and effective. The design options available for individual schemes are therefore likely to expand as the market for biodiversity develops.

Key design questions for individual schemes concern the basis for compensation, the location and timing of offsets, and the measures to be undertaken to deliver the offset.

For habitats representing non-critical ES/NC, for which impacts can be offset, a key design question is: Which impacts are to be offset?

Different types of habitat need to be distinguished according to their distinctiveness and potential to be re-created. For habitats of high distinctiveness and/or low potential to be re-created offsets should be required to be of the same type ('within type'). For habitats of high distinctiveness, bespoke offsetting solutions need to be developed. For habitats of low distinctiveness, compensation can involve greater levels of trade-off between losses and gains of different type. A key design question, therefore, is: Which impacts are to be offset 'within type' and which can be offset 'out of type'?

Simple offsets use surrogate metrics, such as loss of area, to measure impact and to determine compensation. The ES/NC concepts provide for using more sophisticated metrics based on ecosystem functions or the value of these. Compensation on the basis of ecological functions can be 'like-for-like', based on judgements over 'ecological equivalence', or based on the value of ecosystem functions. A key design question therefore is: What metric should be used?

The principle for compensation is, at minimum, no net loss (NNL). Owing to technical and other risks of failing to meet the targeted compensation level, compensation can be set as a multiple of the loss to be compensated. A key design question therefore is: What multiplier should be used?

An approach built on ES/NC can provide for project-level, area-level, or community-level implementations. A key design question therefore is: What is the appropriate spatial or demographic boundary within which compensation should occur? Should NNL be achieved at project level or for an area or for a community of people? Where can the offset be located?

Offsets can be designed and monitored to achieve the targeted level of compensation over different time frames. A key design question therefore is: What is the time-span over which offsetting of impacts should be achieved? Over what time-span should attainment be monitored?

Several options for delivering offsets are possible: purchase of land by a developer on which compensation actions can be implemented (direct offsetting); payment of existing landowners by the developer to undertake offset activities on the developer's behalf; and paying a contribution to initiatives which achieve equivalent beneficial biodiversity outcomes but which may not be designed to offset a specific impact. Key design questions therefore are: What form(s) can compensation take?

In practical terms, providing biodiversity offsets involve undertaking actions to (re)create or restore habitat, to avert risks to and degradation of a vulnerable ecosystem or to create amenity value; etc. One possibility is to enter into contracts or covenants with land holders for them not to convert habitat.

Trade-offs in scheme design:

Biodiversity is multidimensional and varies depending on its spatial context. Since perfect substitution or replication is impossible, it is necessary to define what is acceptable in terms of the biodiversity delivered in the offset and its location. Decisions have to be taken over the extent to which offsets should be required to be '*like for li*ke' and where offsets can be located.

These decisions affect the scale, boundaries and liquidity of the market for biodiversity offsets that is being created. In turn, these are important factors that influence the contribution that the market can make to ecological effectiveness, economic efficiency and distributional outcomes. Design choices involve trade-offs between these different concerns.

- More stringent ecological equivalence requirements and more restricted geographical boundaries for trading limit the supply and demand for particular classes of biodiversity credit that can satisfy offset requirements. More stringent ecological-equivalence requirements reduce the scope for trading and increase the cost of offsets, but ensure that trades are specific in what they protect or conserve. Less stringent ecological-equivalence requirements and wider trading boundaries increase the scope for trading, reducing the cost of offsets. This offers greater scope for offsetting to be economically efficient, but at penalty of allowing some impacts on biodiversity to be offset through biodiversity gains that are different from the initial losses.

- A more limited spatial scope implies that the compensation will be made closer to the site of the development project where habitat, biodiversity and amenity losses have occurred, so the net effect on local stakeholders will be less that with a wider spatial scope. Wider geographical boundaries for the market offer greater market liquidity and greater potential efficiencies in offsetting costs, but hold a higher potential in principle for local stakeholders affected by biodiversity losses to lose out, since the compensating investment may take place at a greater distance from them and the habitat, biodiversity and amenity gains of the offset site may not be so accessible.

Ecological equivalence requirements and the geographical scope for trading are therefore important variables in designing biodiversity offsetting schemes and creating biodiversity markets.

Through more-restrictive or less-restrictive ecological equivalence requirements, biodiversity offset schemes can be designed to take account of differences in habitat and species conservation priorities; i.e. relatively stringent offset requirements can be established for priority habitats and species while more flexibility can be allowed over less critical habitats and species.

Thresholds can be established to confirm circumstances under which impacts can be offset (non-critical capital/services) or not offset (critical capital/services). For the former it is necessary to decide which impacts are to be offset. For the latter there is a need for procedures for deciding whether a project or development should go ahead nevertheless (e.g. on grounds of overriding public interest) and to specific compensatory conservation measures should it go ahead (i.e. compensations that are not offsetting and may be, for example, punitive).

Multipliers can be used to manage some of the delivery risks. The required compensation is then a multiple of the unmitigated damage, rather than only equal to it. Multipliers can be used to recognise that the new site may be of a different value to the ecological network than the original one, the habitat and biodiversity types might be different, there may be lags before damage is fully compensated, and the new site will be at some distance from the original site and less accessible to beneficiaries of the original site.

There are no hard-and-fast answers to these design questions. Rather designing implementations rests on exploring the implications of making different choices by developing and testing scenarios and undertaking sensitivity analyses and by drawing guidance developed from experience (e.g. Tables 5.1).

Table 5.1: Strategic and scheme level guidance for addressing biodiversity offset delivery risks		
Risk factor	Implementation guidance – strategic level	Implementation guidance – scheme design level
Some habitats cannot be recreated	Develop framing principles for scheme design: e.g. precautionary principle	Different types of habitat need to be distinguished according to their distinctiveness and potential to be re-created. For habitat types of high distinctiveness (that cannot easily be re-created) bespoke compensation solutions need to be developed. For habitat types of high distinctiveness, offsets should be required to be within type. For those of lesser distinctiveness, offsets that provide for losses of one type of habitat to be compensated by gains in another type may be acceptable.
Some habitats can take decades to (re)create or restore their biodiversity interest. Even where an offset has been started in advance, the time taken for habitat to mature will likely introduce time lags between impact (loss of benefits) and offset (full recovery of benefits).	Wherever possible, the created habitat should be in place before the original site is lost. Habitat 'banks' can be established for future projects.	In the early stages of developing biodiversity offsetting infrastructure, offsets are likely to be developed concurrently with the impact taking place. A multiplier can be applied to minimise and partly compensate for losses due to time lags.
Some habitat (re)creation or restoration schemes may be less successful than initially planned	Scientific work is needed to establish the effectiveness of different approaches to (re)creation or restoration and to establish appropriate multipliers to be applied to minimise delivery risk	Each individual offset scheme should aim to achieve a net gain for biodiversity and be designed using 'multipliers' to manage some of the delivery risk.
Benefits (compensation) may not last.	Establishment of organisations and mechanisms to support long-term management, assessment and verification. Establishment of insurance schemes to cover risk to permanence of benefits.	Each individual offset scheme should be accompanied by mechanisms to assure long-term management, assessment and verification. Requirement to insure the offset.

ES/NC knowledge and tools can be used to support informed decision making. Knowledge tools, their use within the design process and important decision considerations are set out in Table 5.2.

Table 5.2: Illustrative use of NC/ES knowledge and tools in offset design

Knowledge Tool	Use in the design process	Decision considerations
ES/NC metrics	To measure impacts (losses and gains) to establish compensation needs and sufficiency	There are different components of ES/NC to be considered and different ways to measure and value losses and gains.
Spatial and temporal ecosystem models and mapping tools	For making impact assessments; to project impacts of proposed development projects; to guide the location of offsets; to test restoration/re-creation scenarios; to know where and when ES/NC losses and gains arise; to establish whether planned developments are feasible within a territory under NNL constraints	The spatial and temporal distribution of losses and gains holds implications for the net impact and distributional consequences of offsetting. NNL is a multifunctional concept: a design principle, a decision rule and a developmental constraint.
Social and economic valuation tools/procedures	To know the distribution of losses and gains across stakeholders	The amenity value of habitat and biodiversity is a function of its accessibility to people.
Multi-Criteria-Decision-Analysis	To rank and compare diverse sets of ES/NC in the context of impacts and offsets	Some ES/NC is critical and cannot be substituted. Some types of habitat are highly distinctive and/or not easily (re)created; others are less distinctive. Not all impacts can be monetized. Different qualities of information need to be considered in offset decision making.
Scenarios and sensitivity analyses	To explore and compare alternative offsetting scenarios	Different scheme designs are possible; e.g. like-for-like and case-by-case versus aggregated compensation; area-based versus ecological functionality based compensation; with or without social valuation; effect of extending or reducing the geographical and/or temporal boundaries within which compensation is achieved.

In principle it is possible to simulate offsetting behaviour, but empirical information concerning the relative costs and benefits of alternative scheme designs depends on developing evidence from pilot schemes. From both public and private stakeholder perspectives, detailed *cost-benefit appraisal* of pilot schemes and design options is needed to inform policy on regulatory aspects (such as the need and scope for regulatory requirement for offsetting), to provide ‘when and how’ guidance for biodiversity offset scheme design and implementation, and to clarify the effectiveness, efficiency and equity implications of design options.

The work within OPERAs exemplars supports the development of the evidence base and the guidance needed to enable developers to determine whether and when a biodiversity offset is appropriate and required and what is the necessary nature, scale and location for any such offset.

6. Standards, certification, labelling, procurement and reporting

6.1 Background

This section provides a review and analysis of schemes that set and use voluntary standards to regulate and assure the quality of production processes and products, manage supply chains, and inform stakeholders.

The context for the review is important. There is at the same time declining use of statutory (mandatory) standards and increasing use of industry self-regulation based around voluntary standards. While the shift in roles and responsibilities offers some potential advantages, it also involves risks. The trend toward industry self-regulation is controversial.

This section seeks to address whether and how a regime of self-regulation can provide effective environmental and ecosystem protection. It asks:

- Are ES/NC concepts integrated into voluntary standards?
- Are standards sufficiently strict?
- Are there appropriate sanctions and are these sufficiently enforced?
- Is there evidence of the ecological effectiveness of self-regulation?

The purpose is both to evaluate the actual and potential effectiveness of self-regulation in ecosystem and habitat/biodiversity protection and to establish guidance for scheme design and improvement, especially through gap analysis and analysing factors in the strength or weakness of schemes.

The scope of the review includes standards and associated schemes of certification, labelling, reporting, disclosure and accreditation together with their related instruments as well as actual and potential drivers of scheme uptake, including private and public contracting, purchasing and procurement policies and green investment vehicles.

The focus of the analysis is on the role of standards in supporting the implementation and achievement of environmental protection and habitat/biodiversity conservation policies and goals by integrating ES/NC concepts into standards and driving the take-up of standards. Special emphasis is placed on the role of standards in implementing the mitigation hierarchy and supporting the EU policy commitment to No-Net-Loss of Biodiversity.

The methodological approach involves conceptual, empirical, and prospective elements to highlight the potential that standards hold for improving environmental performance, the actual extent to which this potential has been taken up, and how more of the potential might be captured.

- The conceptual element focuses on the theory of change that underpins the use of standards in harnessing market forces for positive environmental change and the potential for integrating ES/NC and habitat/biodiversity concepts into standards so

that market forces might be used to improve environmental performance and halt habitat/biodiversity loss.
- The empirical element involves a meta-level review and trend analysis of the market for standards, a meta-level review of currently-operating schemes, and more detailed structured description and empirical analysis of a set of front-runner schemes that includes how these schemes have integrated ES/NC concepts into requirements, how strict these are, and how rigorously they are enforced.
- The prospective element offers guidance for implementation design and governance and the roles key actors can play in relation to the core challenges.

6.2 Definition

Sustainability standards are adopted voluntarily by businesses or enterprises to demonstrate the performance of their organizations or products in relation to environmental, social, ethical, and/or health and safety concerns. Schemes establish a framework of good management principles, specify standards for the production, processing and handling of materials, and may also propose or prescribe actions to meet the standards. They can range in complexity from single-issue one-dimensional schemes to multi-issue and multi-dimensional schemes.

A standard is normally developed by a broad range of stakeholders and experts in a sector of productive activity. It comprises a set of environmental or ethical values and principles together with good practice requirements and criteria against which to assess the compliance or non-compliance of production processes and products. Standards are usually accompanied by a verification process - referred to as "certification" - to evaluate that an enterprise, its operations or its product complies with a standard, as well as a traceability process for certified products to be sold along the supply chain, often resulting in a consumer-facing product label. Certification programmes also focus on capacity building and working with partners and other organisations to support smallholders or disadvantaged producers to make the social and environmental improvements needed to meet the standard.

Standards, certificates and labelling schemes provide a market mechanism to adjust for environmental and social externalities. They prescribe and incentivise responsible practices by setting higher than legally-required minimum standards for environmental and social protection. Standards operate simultaneously as requirements and criteria for certifying units assessed to be compliant under the schemes.

Schemes directly concern provisioning services, but can be used to address all other aspects of ES/NC, including habitat/biodiversity. Of particular interest is the use made of tools, instruments and concepts that integrate ES/NC concepts including: Input-Output (I-O) and LCA-based tools in setting standards and determining the ‘environmental footprint’ of products; the use of sustainable harvesting concepts and models for harvesting wild species; the use of protected areas and High Conservation Value (HCV) concepts in regulating activities; and the integration of the mitigation hierarchy, including the use of set asides, environmental corridors and offsets within standards. Interest focuses also on aspects of scheme governance, including the role of stakeholders, the use of third-party verification procedures, monitoring and reporting.

6.3 Key elements and features of schemes

The main features of schemes by which they can be described and assessed are set out in Table 6.1

Table 6.1: Key descriptors of standards, certificates and labels

Managing organization: Who manages the standard/ecolabel?	Management organization Type of organization Year of establishment Funding Governance
Label details: What are the characteristics of the standard(s) related to this ecolabel?	Applicable life cycle and supply chain phases Social and environmental attributes Mutual recognition with other ecolabels Standard details, including standard document, review frequency
Label development: How were the standards for this label developed?	Standard development and management process Standard-setting norms followed for development of the ecolabel's standard Type of standard-setting process, external stakeholders, and funding sources
Conformity Assessment: How is compliance with this label's standard ensured?	Requirements to achieve certification (i.e. chain of custody, site visits, metrics) Audit / surveillance requirements Use of third-party assessors Duration of certification, time to achieve certification
Purpose and orientation: Who and what does the standard target?	Compliance type (pass/fail or tiered) Target audience (consumers / retailers / manufacturers)

- *Unit of assessment*: Standards can be developed for different units of assessment:
  - Production standards can be established for different kinds of production unit, such as individual farms, areas of forest, fisheries or mining sites in order to control exploitation (provisioning) practices.
  - Processing standards can be established to control operations and practices at material processing facilities, such as the use, handling and treatment of water, chemicals and wastes at a pulping facility or a palm oil facility.

- Operator standards can be established and applied to individual producers or companies in relation to their operations.

- Chain-of-custody (chain-of-responsibility) standards set out requirements for materials handling and storage as materials pass along the value adding chain. They include requirements for documenting transfers and the responsible holders and handlers of materials, which provide traceability.

- *Initiation:* Schemes can be established by different initiating organisations including: dedicated third-party certification organisations, governmental agencies, non-governmental organisations, industry organisations, trade associations, brand-owning businesses or mass-outlet retailers.

- *Higher references:* Standards are often developed in conformity with protocols established by higher level bodies and accreditation agencies.

  - The International Social and Environmental Accreditation and Labelling (ISEAL) Alliance has emerged as the authority on good practice for sustainability standards and its Codes of Good Practice represent the most widely recognised guidance on how standards should be set up and implemented to be effective. By complying with ISEAL codes and working with other certification initiatives, ISEAL members seek to demonstrate their credibility and work towards improving their positive impacts.
  - Standards often make use of existing internationally-agreed protocols in relation to specific tools (such as LCA) or management practices and procedures (such as EMS).
  - Legal compliance with relevant international, national and local regulations in force is a minimum requirement of schemes. Distinguishing legal production processes and products from those that are illegal is increasingly important in some sectors, such as fishing and forestry.

- *Orientations:* There are two main forms of implementation: business-to-consumer (B2C) implementations and business-to-business (B2B) implementations. They are important to distinguish, but the line of distinction between these is becoming increasingly blurred. B2C implementations are consumer orientated (consumer-facing). They provide information and assurances to consumers about the production standards of products. B2B implementations are orientated toward other businesses and are used by retailers and brand owners to exert control or influence over their suppliers directly by requiring supplier participation in third party schemes or compliance with standards of their own (business-facing).

  - There are important similarities between B2C and B2B implementations. They both harness the power represented by purchasing, market access and consumer spending in downstream sections of the value-adding chain to exert influence over the production and processing practices of upstream suppliers. Since value-adding chains are often international or global, schemes enable brand owners, retailers and consumers in richer countries to take greater responsibility for the environmental impacts of consumption. The most important environmental impacts arising from consumption in richer countries often occur upstream in the supply chain where materials embedded in final products are produced or processed. Schemes therefore enable the spending power of richer countries to be used to leverage influence over upstream producers and processors and their practices. B2C and B2B implementations also face common design and governance challenges. An important common challenge is to better integrate ES/NC and habitat/biodiversity requirements into standards.

    - B2C and B2B implementations have different drivers and (primarily) serve different stakeholders and purposes. They also operate on different logics and theories of change. In terms of the balance of implementations, there is also a shift toward B2B schemes. B2C implementations have been dominant in the early development of standards, certification, and labelling, but there are limits to the environmental performance improvements that market-driven B2C implementations can deliver. B2B implementations are part of a trend toward mainstreaming supply chain sustainability within business practices. Whereas B2C implementations depend on consumers to drive and facilitate upstream changes in production practices, retailers and brand owners use B2B implementations to influence upstream suppliers directly. B2B implementations can be used to determine whether producers and processors qualify as suppliers, but also to support and incentivise suppliers to improve their performance and to innovate.

    - In recent years, the B2B focus of sustainability standards has risen as it has become clear that consumer demand alone cannot drive the transformation of major sectors and industries. In commodities such as palm oil, soy, farmed seafood, and sugar, certification initiatives are targeting the mainstream adoption of better practices and pre-competitive industry collaboration. Major brands and retailers are also starting to make commitments to certification in their whole supply chain or product offering, rather than certifying a single product line or ingredient.

- *Legitimacy:* With increasing use of standards as a tool for making global production and trade more sustainable, it has become increasingly essential that there are ways to assess the legitimacy and performance of different initiatives. Company and government buyers, as well as NGOs and civil society groups committed to sustainable production, need clarity on which standards and ecolabels are delivering real social, environmental and economic results.

6.4 Trends in the use of standards, certification and labelling

Sustainability standards first emerged in the late 1980s and 1990s in areas where national and global legislation was weak (especially around social welfare issues) but where the consumer and NGO movements around the globe demanded action. This led first to the emergence of social welfare standards and labels, such as Fair Trade. The approach was adopted by leading brands seeking to demonstrate the environmental and health/safety merits of their products, leading to the emergence of several hundred sustainability and organic standards oriented to consumers or to other businesses.

In principle, standards, certification and labelling are voluntary schemes. B2C implementations offer a potential mechanism for upstream producers and processors of primary products and mass commodities to recover the costs of meeting higher environmental standards. Potentially, standards can reward producers for environmental leadership by delivering net gains in income. However, under B2B implementations the take up of standards by producers and processors can be made a condition for supply and for access to some markets or market segments.

In parallel with the mainstreaming of sustainability within routine business practices over the last 20 years, using standards, certification and labelling has become widespread in businesses in making purchasing decisions, managing supply, marketing, selling to B2B and B2C customers, guiding employees and responding to stakeholders and regulations.

There have been important changes over this same period also in the number and scope of certification and labelling schemes offered by scheme providers. The number of schemes offered has grown, but the scope of individual schemes has also broadened driven by more integrated approaches to sustainability in the supply-chain. Many schemes that originally had a specific focus on either social or environmental issues have expanded coverage to both kinds of issue. The coverage of different environmental issues has become more inclusive within individual schemes. The result of these developments is that schemes have become increasingly similar.

The continuing evolution of schemes is changing the nature of certification: i.e. what is being certified, by whom and for which purposes. Whereas retailers and brand owners have tended so far mostly to make use of third party certification schemes in B2C and B2B contexts, some are now developing their own certification schemes. Meanwhile, third-party owners of certification schemes and labels are beginning to switch their focus from certifying commodities and products to certifying retailers and brand-owning businesses on the basis of their processes.

From the perspective of using standard, certification and labelling to protect ES/NC and habitat/biodiversity two important considerations for developments going forward are that demand for certified products is increasing strongly and that ES/NC and habitat/biodiversity are complex concepts. Biodiversity, especially, is a very complex concept that is challenging to set out in clear, measurable terms within standards. Standards must often rely on establishing good management principles and processes and on requiring these to be integrated into the policies and practices of organizations.

The strong growth in consumer demand for certified products exerts a moderating pressure on the rates at which environmental performance standards can be raised and more complex concepts can be integrated into standards. In this market context, efforts are more likely to focus first on encouraging greater take-up of existing standards, raising standards progressively in respect of existing criteria and introducing only basic habitat/biodiversity criteria. Currently, habitat and biodiversity are not well represented in standards and, to the extent they are included, this is largely through indirect indicators of impact. The integration of habitat and biodiversity into standards using direct indicators is challenging and is likely to involve the development of sector- and context- specific criteria based upon in-context assessments of habitat, biodiversity and high conservation values. The general situation notwithstanding a few leading schemes and some newly-developed schemes are beginning to integrate direct habitat/biodiversity indicators into standards. Such pioneering schemes are important for the insights they offer into how habitat/biodiversity might be integrated into schemes more widely and the potential effectiveness of standards in conserving habitat and biodiversity.

6.5 Implementation issues and influencing factors

From the perspective of the role of standards in helping to achieve important environmental and biodiversity policy goals, important implementation issues involve:

- Ensuring that all sectors and supply chains with important ES/NC and habitat/biodiversity impacts are covered by schemes.
- Developing direct habitat/biodiversity goals and indicators and including these within schemes.
- Driving the uptake of schemes and improving their environmental outcomes.
- Demonstrating positive environmental outcomes.
- Stimulating demand for certified products.

Drivers: There are some common drivers of schemes and, also, some drivers that are more important for B2C implementations or for B2B implementations.

- Self-regulation is preferred by business to state regulation.
- There are first-mover advantages to be gained by being party to the process of establishing schemes, setting standards and working with other key stakeholders; including accreditation agencies, regulators, scheme operators, trade bodies, NGOs, etc.

- A common driver to date for all implementations has been corporate social responsibility commitment by larger companies, which encourages both B2C and B2B implementations.
- Corporate social responsibility is supported through opportunities for ISO 14001 certification, which requires that companies have Environmental Management Systems in place, and corporate reporting initiatives, which involve using reporting protocols to make the environmental policies, practices and performance of companies transparent to stakeholders. Individually and combined, these increase pressures on companies to act responsibly and improve environmental performance.
- Companies and their stakeholders seek to minimise reputational risk attaching to unsustainable production practices of upstream suppliers.
- Sustainability issues and specifically loss of habitat, biodiversity and ES/NC more generally are increasingly seen by business and investors as direct threats to business sustainability; i.e. both the severity and likelihood of risks arising to businesses from ES/NC losses are considered to be increasing. This has contributed to a shift in how companies perceive and manage sustainability from approaches based on sustainability as an external issue to sustainability as an important internal business issue.
- A main driver of B2C schemes has been growth in consumer awareness of environmental issues and growth in demand for information about the sustainability qualities of the production processes of products. Certification and labelling schemes in B2C implementations can offer business opportunities to grow market share, differentiate company products on markets and attract price premiums. These act as consumer-to-business drivers.
- The adoption of sustainable supply chain policies by private companies can drive take-up of certification schemes by upstream producers. Sustainably supply chain management can act as a strong business-to-business driver.
- Pressures from investors concerned for business survival and profitability, can act as a strong investor-to-business driver.
- Public procurement policies or regulations that exclude non-certified products from important markets can be a strong government-to-business driver of both B2C and B2B implementations; for example:
    - Products certified by the Roundtable on Sustainable Biomaterials (RSB) are considered compliant with the European Renewable Fuels Directive and qualify for entry into European markets; and,
    - The European Ecolabel is embedded into UK green public procurement policies and programmes.

- Some governments sponsor schemes of their own, endorse schemes or require businesses in some licensed sectors to join specific third-party schemes; for example:
    - Germany has its own organic farming standard; and,
    - Denmark requires the Danish fishing fleet to be MSC certified.

Theory of change: Underpinning the development of standards, certificates and labels and their use in environmental protection in B2C implementations is a theory of change, which sets out how externalised environmental costs might be internalised into costs and prices progressively through voluntary actions and environmental leadership.

When environmental costs are externalised producers showing leadership by voluntarily adopting higher environmental standards risk becoming uncompetitive or going out of business. This risk is higher when products produced to higher quality standards are not distinguishable on markets from products produced to lower environmental standards. Many products, especially mass commodity products of mining and extraction, agriculture, forestry and fishing appear homogeneous on markets irrespective of how they are produced; e.g. metals, aggregates, fibres, tea, coffee, timber and fish.

Standards, certificates and labels provide a possible solution because they provide for practices at production and processing sites to be certified and for products produced using materials from certified sites to be labelled to indicate conformity with the higher standards of a certification scheme. This signals their higher production quality to purchasers and consumers. Consumers can then distinguish and buy labelled products and pay a price premium for their higher environmental performance. Public and private sector procurement policies can also be used to establish eligibility criteria for market access based on certification.

By enhancing market access, increasing market share and enabling a price premium, certification and labelling schemes can generate extra revenues. Some or all the higher costs of more sustainable practices can be recovered *if* products attract a price premium and this is returned through the supply chain to the upstream producers and processors. If the returned premium is sufficient to deliver net income gains, labelling may also generate a new funding stream to support investment in further improving environmental practices.

Once a scheme is established and the key links in the causal chains are operating as hypothesised, further take-up can be driven *up to a natural market saturation level* by the market through a virtuous cycle where higher environmental performance leads to higher net incomes and further investment to improve environmental performance. This is hypothesised also to create incentives for additional producers and processors to join schemes.

Market-driven take-up naturally plateaus as more products become certified and there is less scope to differentiate products on their environmental performance or to charge a price premium. Driving uptake beyond the natural market saturation level then depends on regulation or shifting the balance of implementations from B2C to B2B.

At the level of the overall market for certification and labelling, building the overall market for certificates and labels is also an important element of building the green economy. Schemes can support and be supported by the green investment and green business agendas helped disclosure and reporting, by risk rating, and by public and private procurement policies. For example, standards, certification and labelling is synergistic with the development of green bonds and green procurement strategies as certificates and labels signal which companies and production facilities are the most sustainable, which supply chains are most sustainable, and which commodities are produced more sustainably than others.

Process of scheme development: Different phases in the design and implementation of certification/labelling schemes can be identified:

- A preparatory (pre-competitive) phase: during which the concerned actors from different sectors pool knowledge and negotiate the standards/criteria
- A market demand-driven phase: during which the market drives expansion of certification to address B2C and B2B demands and standards can also be raised, depending on feedback from experiences. This phase goes through launch, take-off and stabilisation stages. It reaches a natural limit when the market-share of certified products begins to level off.
- An intervention driven phase: during which interventions are needed to drive further uptake and expansion of certification to overcome market-share stabilisation and to mainstream standards into regular production.

Important here is that certification cannot achieve its full potential to support improvements in environmental performance using only the market to drive expansion. Interventions are needed to overcome market share stabilisation inertia.

Increasing the take-up of standards in B2C implementations during the market-driven phase involves strengthening the mechanisms and relationships at each link in the causal chain; i.e. ensuring that certified products attract a price premium, ensuring that a part of the extra revenues is transferred back through the supply chain to deliver higher net incomes to producers, ensuring that some of this extra income is re-invested in improving practices further, and ensuring that changes in practice deliver demonstrable environmental improvements. Increasing the take-up of standards beyond the market-driven phase may involve more emphasis on B2B implementations.

Relationship to policies and to other schemes: The implementation of standards, certification and labelling schemes can be addressed at the level of the overall market and at the level of individual schemes. At the meta-level, the development of the overall market for certificates and labels is integral to implementing the green business agenda. Standards, certification and labelling schemes are part of the infrastructure of a more complete market and are integral to developing the green economy, green markets, green products, green companies and green finance, enabling competition to take place within markets on the basis of the sustainability quality of production processes and products. At the level of individual schemes, the integration of ES/NC concepts into standards, raising standards progressive, and increasing scheme take-up are important for gradually internalising environmental costs.

The political context for addressing externalised environmental and social costs through fiscal reform is challenging, especially because of economic globalisation, but regulatory policy can also be difficult and costly for governments to implement and enforce. Certification and labelling schemes go some way toward the voluntary correction of market failures arising from unallocated property rights, externalised costs and missing information.

Standards, certification and labelling can be important private sector voluntary supports to delivering on public policy goals and commitments, especially in the arena transition toward more sustainable regimes of ecosystem exploitation. The no-net-loss principle can be implemented partly, by integrating the mitigation hierarchy into standards.

Certification can also be used in support of implementing PES schemes; for example, when payments under schemes are made for implementing sustainable forestry practices and certification is used as a verification mechanism.

An important added value of voluntary standards is that they can extend the 'reach' of policies beyond state jurisdictions, since supply chains for globally traded commodities are not limited by political boundaries. Much of the environmental damage of highest concern, especially for habitat and biodiversity loss, is associated to production practices in distant countries and in respect to exploitation of open-access resources, such as open ocean fisheries. Territorially, these are outside the jurisdictions of EU Member States. Multinational companies can deploy their geographical reach, market power and greater control and oversight of the supply chain to offer potentially more effective means to drive improvements in environmental performance than are open to government as a direct actor acting alone. This is increasingly important if government is to be able to deliver on more demanding policy goals and commitments, such as those under the Convention on Biological Diversity, which set No-Net-Loss of biodiversity as a principle and goal.

Potential scheme benefits: In principle, voluntary standards, certification and labelling schemes offer a set of important benefits:

- *Ecological effectiveness:* Schemes can be important drivers and facilitators for improving ecosystem management practices. They can support and drive environmental performance improvements and help achieve environmental protection and habitat/biodiversity conservation goals. Certification and labelling can drive improvements both directly, by prescribing better ecosystem management practices, and indirectly, by delivering net increase in producer income through increases in market access, market share and price premiums. Investment in improving productivity from existing production sites (yields and quality) can in principle help in reducing pressures to extend the cultivated area, reducing land conversion and incursion into protected areas.

- *Economic efficiency:* Schemes provide a market mechanism to adjust for social and environmental externalities. They adjust for some information failures of markets and help support better informed decision making.

- *Fairness:* In a successful B2C scheme, the costs of certification and labelling are paid by consumers via a premium for quality-assured products produced to the higher environmental standards of the certification schemes and for the service of enabling consumers to differentiate products produced more sustainably from those produced less sustainably. Schemes can help to reduce the financial risk of environmental stewardship. They enable responsible environmental behaviour to be recognised and, potentially, rewarded.

- *Transparency:* Schemes provide clarity over production standards and aspects of quality in the production process that are otherwise invisible. They make hidden costs clear for consumers. They enable apparently homogeneous products to be differentiated on the basis of production sustainability. They are an element in making business practices clear to stakeholders.

- *Policy support:* While schemes are part of the instrumental infrastructure of the emerging markets for environmental protection, habitat and biodiversity, they are also market creation instruments that can drive the development of the green economy, green business and green finance. Schemes are capable of extending the influence of government and companies over production practices using the incentive of access to consumer markets and higher prices to exercise extra-jurisdictional influence over producers and processors operating upstream in distant countries.

- *New financial streams*: To the extent that schemes are successful in delivering net income increases to producers and processors and/or in supporting the development of the market for green investment bonds, they can deliver new funding stream for investment in improving environmental performance and eco-efficiency. This is important because investment in value-adding chains in respect of agricultural commodities and biomass has historically been loaded toward the downstream section of the chains. There has been longstanding under-investment in the upstream sections of the supply chain where the incomes of raw material producers and processors have traditionally been low, as a function of their low relative economic and commercial power within the chains. Low incomes and poverty have been factors in producers' inability to invest in improving production. Certification can be linked to new investment instruments, such as green bonds.

Costs and their distribution: Schemes also have costs. The level and distribution of these are important implementation considerations.

- The costs of certification fall initially on producers and processors upstream in the chain. There are one-off costs in the transition to certified status associated with meeting standards and changing ecosystem management practices and recurrent (opportunity) costs. There are also transaction costs for subscribing to schemes and paying for inspection, assessment and verification services.

- These costs are potentially recoverable if certification and labelling delivers sustained increases in market access, market share and prices of certified products. Schemes will only be attractive to producers and processors and financially viable if there is a high likelihood that participation will deliver net increases in income. In this event, the costs of schemes are passed wholly or partially to end consumers as part of the price premium.
- The proportion of scheme costs borne by different parties will change as schemes and the markets for certified and labelled products mature..
- The shared interest of business and government in voluntary standards, certification, and labelling gives scope for implementations to be developed as private initiatives, public initiatives, and public-private partnership initiatives.
- Voluntary initiatives toward certification and labelling involve the private sector taking on some of the responsibility and costs of securing and assuring environmental protection and conserving habitat/biodiversity. This can relieve the public sector and help secure more effective outcomes.

Risks and their distribution: There are risks of different type, which need to be managed at different stages in the process of market development of schemes:

- *Financial risk:* Different stages in the market development of schemes can be distinguished. In the early stages of scheme development, the upstream producers and processors take the financial risk, as there is no guarantee that producers, processors or companies in the supply chain of certified products will increase their market share or receive a price premium. The level of (un)certainty about the impact on net income reduces as schemes mature.
- *Information risk:* There is a risk of ‘greenwashing’; i.e. businesses using schemes to give an appearance to regulators and consumers of taking environmental responsibility seriously while continuing with business-as-usual. There is a risk of ‘rogue’ certification and labelling schemes being introduced in attempts to appropriate price premiums attaching to labelled products and, also, of products from uncertified sources being marketed under counterfeit labels. Rogue standards and counterfeit goods and labels are risks to the credibility, market viability and ecological effectiveness of legitimate schemes. Legitimate schemes therefore need to be validated through accreditation by officially-recognised, high-level accreditation agencies.
- *Perverse outcomes:* There is a risk of schemes inducing perverse environmental outcomes if the criteria used are insufficiently comprehensive and the standard is ‘imbalanced’. This is important if there are important trade-offs that the scheme does not fully capture or if an important criterion is not included in the scheme. Which requirements are specified and how trade-offs between environmental performance criteria are handled in standard setting are important technical design issues to

address, especially in relation to how certificates and labels are used, for example, as conditions for market access or eligibility under procurement schemes.

- *Loss of trust and credibility:* There are different plausible ways in which increasing demand for certified products could be met, including by: increase in the number of third party standards; consolidation across schemes and the emergence of standards focussed on specific areas or aspects of sustainability; and retailers and brand owners setting their own standards to control sustainable supply and influence markets. Increase in the number of third-party standards holds risks of scheme proliferation, schemes becoming increasingly similar, or schemes emerging that offer low standards. Such developments risk the loss of trust and credibility. Consolidation across third-party schemes and the emergence of schemes focussed on specific areas or aspects of sustainability are possible responses. Increasingly, to address weaknesses retailers and brand-owners see in available third-party schemes they are setting their own standards to improve those aspects of sustainability in the supply chain that are particularly important for their business, brand and customers.

6.6 The roles and potentials of ES/NC in standards and labelling schemes

In principle, standards can be used to address the main drivers of environmental damage and habitat/biodiversity loss.

- The provisioning services of ecosystems and sustainability of supply chains are often undermined by poor land and ecosystem management practices in agriculture and by over-exploitation of natural biomass (e.g. fish, timber) through poor harvesting practices. This may involve requirements to maintain or increase soil depth and fertility on agricultural land or to maintain stocks of exploited fish populations. Actions to take to improve exploitation practices may be specified: for example, to restrict fish take to enable depleted fish stocks to recover to improve long-run sustainability of the fishery.
- Poor management practices in the exploitation of land and renewable resources and increases in demand for provisioning services both exert pressure to bring more land and natural resources under production, such as through land conversion. This risks loss of ES/NC of ecosystems not currently under intensive production. The ES/NC concepts can be integrated into standards to restrain land conversion/clearance and address deforestation by restricting certification to land already under production.
- The ES/NC of adjacent areas and ecosystems are also often undermined by spill-over impacts from production and processing sites. The ES/NC concept can be integrated into standards to require practices that reduce spill-over impacts. Standards can be used to specify requirements in relation to energy use and efficiency at production and processing units, to ban or require reductions in the use of chemical inputs in farming or to require reductions in water use and use of

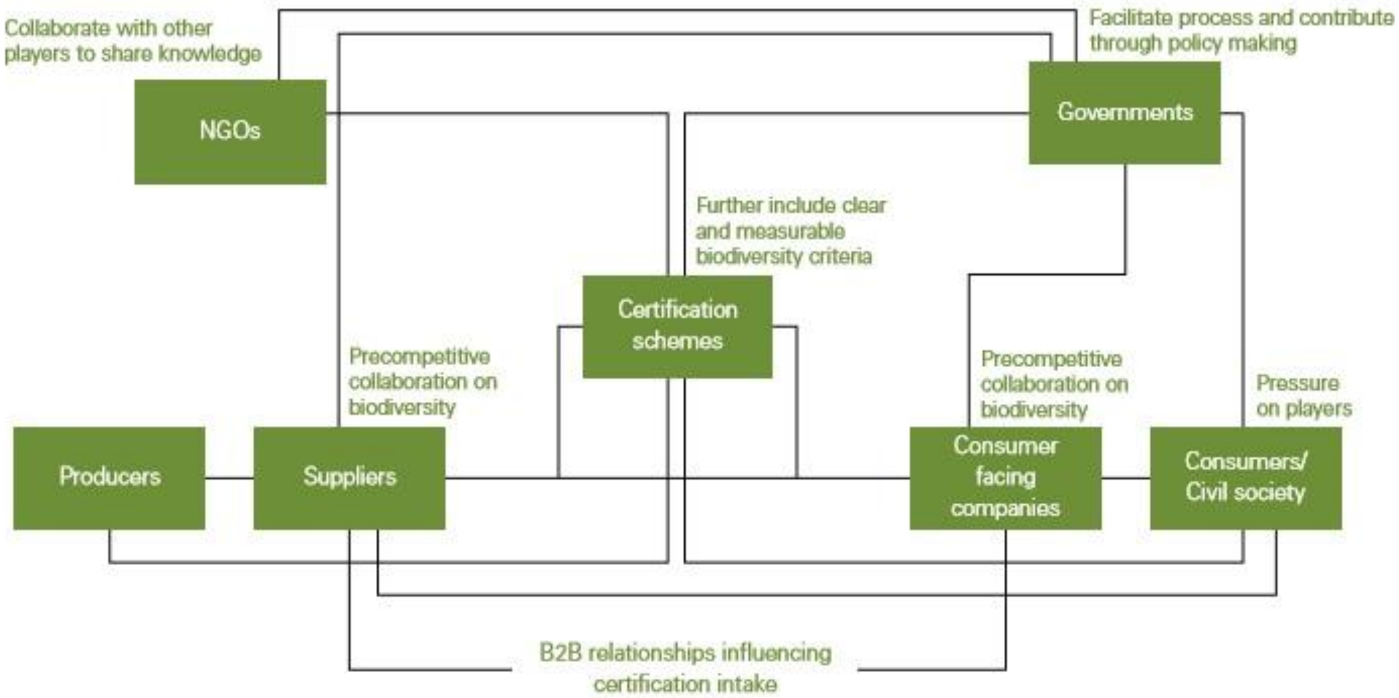
effective water recycling and waste water treatment systems at materials processing facilities. Standards relating to aggregates and to mining operations can be set to prohibit mining, ore beneficiation and dumping of mining waste in ecologically sensitive areas, to reduce water and energy use in mining and processing operations, and to reduce water contamination.

- Safeguards can be introduced to ban or restrict activities in designated conservation areas or if these pose threats to high conservation values. Use can be made of the mitigation hierarchy, including the use of set-asides on productive land and offsetting. ES/NC benchmarking and regular monitoring of habitat and biodiversity status on productive and adjacent land or in respect of harvested wild populations can be integrated as a requirement into schemes. Monitoring information can be used to evaluate the ecological effectiveness of the scheme, improve environmental management, and for reporting purposes.

ES/NC tools and concepts and their roles in schemes: Input-Output (I-O) and LCA-based tools are used in setting standards and determining the 'environmental footprint' of products. Sustainable harvesting concepts and models (e.g. based upon sustainable yields) are used in setting levels for harvesting wild species. More qualitative methods based on developing land management principles are also used in setting standards (e.g. based on integrating Protected Area (PA), High Conservation Value (HCV), and Mitigation Hierarchy (MH) concepts into standards. Mitigation hierarchy concepts include requirements for set asides, environmental corridors and offsets. Interest focuses also on tools and processes used in scheme design and governance: the use of multi-actor governance approaches, stakeholder processes, and third-party verification, monitoring and reporting procedures.

Actors and their roles: The main actors directly involved in certification/labelling include: certification scheme providers, government, businesses and NGOs. Consumers are important since they create demand that puts pressure on the direct actors. Investors, business lenders (banks) and insurers can also put pressure on businesses. The relationships between these actors on biodiversity and certification are set out in Figure 6.1

Figure 6.1: Interconnections between actors



(Source: KPMG 2012

Scheme design factors and considerations:

- At the meta-level, important considerations are the scope and coverage of schemes in terms of sectors, products and the geographical markets served.
- Design issues for individual schemes and instruments include: explicit integration of ES/NC criteria into standards; securing the market success of certification and labelling; ensuring that market benefits are transferred back up the supply chain to farmers and producers; and ensuring that increases in net income are re-invested in further improving environmental performance.
- Design options include: the choice of principles, standards and actions, especially what is included and excluded in support of environmental protection and habitat/biodiversity conservation goals; the choice of indicators; the choice of baseline dates from which standards should apply and any offsets should be calculated; whether different levels of compliance are to be allowed; the system of benchmarking, weighting and aggregation across criteria to be used to support certification and/or labelling; and the period of validity of certification.

Governance factors and considerations:

- Governance involves the framework of rules, practices and procedures through which the standards are established, approved and reviewed, compliance with the standard is assessed and verified, performance against objectives is monitored and reported and the standard is integrated into other areas, such as procurement.
- Governance issues include: how to secure scheme legitimacy and credibility; how to safeguard the scheme against outside threats; how to finance the scheme start-up; how to use the scheme to maximise positive ecological and economic impact; and what sanctions or penalties to impose for infractions.
- Governance options include: which actors and stakeholders (or their representatives) should be involved at different stages and what roles they should play; how to make best use of the information that certification provides to improve to drive compliance, improve environmental management and demonstrate scheme effectiveness.

Contextual factors and considerations:

- Context largely involves market considerations that impact on the take-up of certification and labelling and on the market effectiveness and ecological effectiveness of schemes.
- The main contextual issue concerns the stage of development of the market for certificates and labels.
- Options relate to identifying the main actors with interests in developing the market for certificates and labels, the roles they can play, and actions they can take.

Trade-offs in scheme design:

A key design challenge is to balance standard stringency with standard uptake. Standards that are too stringent are likely to discourage take-up, while standards that are insufficiently stringent will be ecologically ineffective.

6.7 Current status of schemes

Ecolabel Index (www.ecolabelindex.com) is the main, independent global directory of environmental certification schemes and ecolabels. It currently tracks 465 ecolabels, covering markets in 199 countries and 25 economic sectors.

Particularly of interest are schemes with a scope that includes sectors and commodities with high ES/NC or habitat/biodiversity impact potential. Such sectors include: mining and extraction, aggregates, construction, agriculture, forestry, fishing and aquaculture. Important commodity classes include foods, feedstocks, bio-materials and bio-fuels. Table 6.2 provides a listing of some of the major schemes and ecolabels that apply to sectors and commodities with high ES/NC or habitat/biodiversity impact potential and have a presence in European markets.

The main implementation challenges for certification and labelling lie in defining operational standards and criteria to assess habitats and biodiversity, determine base lines, and monitor and report the positive impacts of certification. These challenges are more pronounced in respect of habitat and biodiversity compared with other aspects of environmental protection because of their complexity and the complexity of the links between ecosystem management practices and habitat/biodiversity. Insights into how habitat and biodiversity standards can be integrated into schemes can be found in the approaches taken by leading schemes.

Schemes that are leading in integrating habitat and biodiversity into standards use entry points informed by drivers of loss of ES/NC and threats to habitats/biodiversity. Leading schemes:

- Deploy a combination of principles, standards and actions.
- Address the key direct drivers of habitat/biodiversity loss, e.g. land conversion and clearance from natural state to agricultural exploitation, over-harvesting of wild biomass, and dumping of mining waste, as well as addressing indirect drivers
- Make use of land and ecosystem classifications; for example distinguishing between protected areas (unavailable for production) and production areas.
- Set baselines for establishing the classification and status of land both in defining the standard and in developing benchmarks for measuring change.
- Use clear indicators so that criteria are operationally precise.

Table 6.2	Certification & labelling schemes addressing sectors and products with high ES/NC impact potential - illustrative										
	Aggregates & Construction Materials	Agriculture & Food	Livestock, Meat & Dairy	Viticulture & wine	Coffee, Cocoa or Tea	Cotton	Soy	Palm Oil	Biomass	Aquaculture, Fisheries & Fish	Forestry and Forest Products	Tourism
ASC												
Blue Angel												
BRE Certified												
Certified organic												
Coop Naturaline												
Cradle-to-Cradle												
Danish Ø–mark												
Demeter												
EPD												
EU Organic												
EU Ecolabel												
FSC												
Lagambiente Tourismo												
Global Organic												
LEAF-Marque												
MADE-BY												
MSC												
NATURTEXTIL												
OEKO-Tex												
Rainforest Alliance												
RSB												
RSPO												
RTRS												
SAOS												
UTZ Certified												

ASC (Aquaculture Stewardship Council); BRE (Building Research Establishment); EPD (Environmental Product Declaration); FSC (Forestry Stewardship Council); MSC (Marine Stewardship Council); RSB (Roundtable for SustainableBiomass); RSPO (Roundtable for Responsible Palm Oil); RTRS (Roundtable for Responsible Soy); SAOS (Soil Association Organic Standard); UTZ (originally Utz Kapeh)

A principle of the UTZ scheme is that "*the degradation and deforestation of primary forest is prohibited*". Its cocoa standard sets out that: "*production will not take place in protected areas, including officially proposed protected areas, and not in the immediate vicinity of those areas*". Immediate vicinity is defined as "*a distance of two kilometres*".

The Rainforest Alliance scheme specifies that in respect to the conversion of land on which crops are grown: "*no natural ecosystem shall have been destroyed after 2005. If any were destroyed between 1999 and 2005 compensation should be sought*". The guidance on action refers to the mitigation hierarchy and to offsetting.

The FSC, Rainforest Alliance and UTZ schemes all set requirements in relation to the expansion of farming activities. These specify the establishment of required habitat set-asides on farmland to mitigate negative impacts on biodiversity.

A search using the keyword 'biodiversity' to refine entries on the Ecolabel Index website produced 10 entries. These schemes and their key features are described in Table 6.3. The sectors covered by these schemes include: construction, tourism, farming and food processing, viticulture, aquaculture, fishing and consumer products. The units of assessment include companies and projects as well as sites and products. Requirements are based on either/both material-flow criteria and/or prohibited, prescribed or suggested practices. Standards may be based on an overall approach to production; for example standards in respect to farm management may be based on "organic", "mixed", "biologic" or "biodynamic" farming systems.

Table 6.3	Ecolabels with biodiversity content						
Label Name	**Market addressed**	**Label owner, legal status, accreditation**	**Unit of assessment**	**Basis of certification**	**Substance based I-O or LCA (quantitative) indicators**	**Action-, practice- or area- based indicators**	**Verification**
Best Aquaculture Practices	Global	Global Aquaculture Alliance, Non-profit	Seafood processing plants, farms, hatcheries and feed mills	Certification is based on standards for each unit of assessment and for different types of production (fin-fish, molluscs, crustaceans, etc.). These include standards for protection of ecologically-sensitive areas (for land-based operations) with damage to ESA since 1999 required to be offset by restoring an area 3x as large as that damaged; biodiversity protocols to prevent spread of disease to wild stocks; controls over use of therapeutic agents (antibiotics, antimicrobials), hormones not to be used as growth promoters; and protocols for wildlife interactions with predator species. Only passive deterrence methods allowed for species on IUCN Red List and for locally and nationally protected species. Non-lethal deterrence to be preferred.	✓	✓	
Denilat Bio Garantie	Europe (mainly Switzerland, Germany, Austria)	Denilat, For Profit	Vineyards (Viticultural and vinification aspects)	Requirements include: organic-certified grapes; biological management of the entire vineyard area; economical use of fertilizer in closed loop; maintenance of vineyard greenery throughout the year (except for dry areas); no synthetic pesticides or fertilizers; and the use of copper is limited to 4 kg per hectare and year. Biological management of the vineyard. Vineyard to be managed as an ecosystem. Assessment uses a step model. Scores across 100 evaluation criteria deliver one of three levels of fulfilment.	✓	✓	Independent (third party) verification following EN 45011

Table 6.3 (cont.)	**Ecolabels with biodiversity content**						
Label Name	**Market addressed**	**Label owner, legal status, accreditation**	**Unit of assessment**	**Basis of certification**	**Substance based I-O or LCA (quantitative) indicators**	**Action-, practice- or area- based indicators**	**Verification**
Demeter Biodynamic®	Worldwide	Demeter International, Non-profit Organization, IFOAM affiliated	Whole farms	Requirements for biodynamic farming include: avoiding all synthetic chemical pesticides, fertilizers, and transgenic material; use of farm-generated, living solutions to pest control and fertility instead; and setting aside a minimum of 10% of total farm area for biodiversity. The entire farm must be certified, not individual crops. Starting in mid-2017 Demeter certification will include soil testing for carbon sequestration.	✓	✓	Demeter approved and/or NOP-accredited inspectors
Effinature	France (PACA Region, Provence, Alps, Côte d'Azure)	NOVOCERT. Other	Construction sector projects	Effinature certification is designed to reverse trend of biodiversity deterioration in the construction sector and raise awareness of this issue among town planners. It supports the project from conception and design through realisation and operation. Certification uses over 100 biodiversity control points, which are applied across the different stages of a project. These involve: determining the value of the site and the ecological potential of the project; preserving the existing natural heritage and existing landscape; controlling the impacts of the project on biodiversity; unlocking the ecological potential of the project through a responsible and sustainable management of biodiversity; training those involved in the project and raising awareness.		✓	Conformity is assessed by Novocert itself (second party) through Ecocert and Prestaterre

Table 6.3 (cont.)	Ecolabels with biodiversity content						
Label Name	**Market addressed**	**Label owner, legal status, accreditation**	**Unit of assessment**	**Basis of certification**	**Substance based I-O or LCA (quantitative) indicators**	**Action-, practice- or area- based indicators**	**Verification**
EU Organics	EU and global trade with EU	European Commission; Government	Food products covering all categories of production and processing: plants and plant products; meat and dairy; wine; seaweed and aquaculture; processed products	Requires that at least 95% of agricultural ingredients must be "organic" as defined under EU legislation. EU rules on organic farming standards are set out in Regulation 834/2007. The standards prescribe acceptable agricultural, husbandry, aquacultural and viticultural practices. They prohibit use of mineral nitrogen fertilizers, synthetic pesticides and herbicides in agriculture and the use of growth promoters and synthetic amino-acids in animal husbandry. The standards promote multiannual crop rotations and, in organic livestock production, prescribe mixed farming and local production of organic feedstock. From 2012, the wine standard covers vinification as well as viticultural practices and sets maximum sulphite content of wines. Organic foods imported into the EU must be certified EU Organics compliant	✓	✓	Independent certifiers approved by the European Commission.
Green Tourism Business Scheme	UK, Ireland, Canada	Green Business UK Ltd.; Non-Profit	Businesses and organizations offering tourist services: providers of accommodation, attractions, activities; conferences and events; and travel/transport	Businesses are assessed against 145 criteria covering: energy carriers and efficiency; carbon footprint and efficiency; water efficiency; waste management; recycling; sustainable purchasing; nature and biodiversity. Businesses meeting the standard are awarded one of three levels (tiered).	✓	✓	Independent (third-party) verification by qualified auditing organization/auditor following ISO 19011 QMS and EMS auditing.

Table 6.3 (Cont.)	Ecolabels with biodiversity content						
Label Name	**Market addressed**	**Label owner, legal status, accreditation**	**Unit of assessment**	**Basis of certification**	**Substance based I-O or LCA (quantitative) indicators**	**Action-, practice- or area- based indicators**	**Verification**
Global Green Tag®	Global, but with strongest presence in USA, Australia, China, India, Malaysia, Singapore and South Africa	Global GreenTag Pty Ltd. Licenced operator of Global GreenTag Certified; For Profit; ISO	Products and packaging (broad range) including: building products, cleaning products, cosmetics and personal care products, forest products (paper, furniture), machinery and equipment, and textiles.	Global Green Tag® is a green product rating and certification system, underpinned by LCA. Method requires full disclosure of every product ingredient and process. Products are assessed and scored against others in the same functional purpose across six sustainability categories and 20+ criteria, based on I-O and LCA criteria, and biodiversity impacts. The Global Green Tag® ecolabel rating differentiates a product within the top end of the green product market by scoring, weighting and developing an EcoPOINT Score (-1 to + 1). The system provides metrics for sustainability that include net positive impacts of products for carbon and biodiversity. It awards certification to products on one of four levels: good, very good, excellent and world-leading. Deploys ISO 14040 and ISO 14044 for LCA; ISO 14067 for Greenhouse Gas calculation; ISO 14025 for Environmental Product Declarations (EPDs) and ISO 21930 and EN 15804 for specific need EPDs	✓		Independent (third-party) verification following ISO/IEC Guide 65 (Product Certification); Externally certified to ISO 9001 for Quality Management; Externally verified as compliant to: ISO 14024 for Type 1 (Third Party) Eco-labels; ISO 17065 for Conformance Assessment Bodies;.
Local Food Plus (LFP)	Canada	Local Food Plus; Non Profit	Farming and food processing operations	Requirements include: reduction or elimination of synthetic pesticides and fertilizer use; avoidance of hormones, antibiotics and genetic engineering; conservation of soil and water; protection and enhancement of wildlife habitat and biodiversity; and reductions of energy consumption and GHG emissions.	✓	✓	

Table 6.3 (Cont.)	Ecolabels with biodiversity content						
Label Name	**Market addressed**	**Label owner, legal status, accreditation**	**Unit of assessment**	**Basis of certification**	**Substance based I-O or LCA (quantitative) indicators**	**Action-, practice- or area- based indicators**	**Verification**
LIFE	International, but based in Brazil	LIFE Institute, Non-profit Organization	Organizations of any size or sector anywhere in the world that seek to undertake voluntary and effective actions toward biodiversity conservation.	Certification is based a methodology to assess an organization's environmental management and performance (including biodiversity impact) through a scoring system in order to propose minimum conservation actions that the organization should take to obtain certification. The approach is based on defining four main groups of biodiversity conservation actions in relation to: officially-implemented protected areas; non-officially protected areas; initiatives for species and ecosystem management and conservation; and, strategic, political or educational initiatives for biodiversity conservation.		✓	Independent (third-party) verification by LIFE Institute accredited agents following ISO 17011 (Accreditation), ISO 17021 (Management System) and 19011 (Auditor Qualification).
Salmon Safe	West Coast, USA	Non-profit Organization founded by a leading U.S. river and native fish conservation society	Certifies urban and agricultural operations, including: farms, vineyards, dairies, corporate campuses and other sites	Certification is based on the protection of water quality, riparian habitat and native biodiversity. Involves the elimination of chemical pesticides harmful to fish, reducing run-off into streams, restoring wetlands, and related conservation practices.	✓	✓	

Table 6.4	Delinat Bio Standards
Area	**Requirement**
Method of management	Whole plant must be organic. Biological / ecological management of all branches
Biodiversity, ecological balancing areas, promotion of small structures	Ecological compensatory area: at least 12% of the area under vines, 7% of which is within or directly adjacent to the vine. A further 5% must be within a distance of 1000m from the vineyards. In addition: fruit trees, wild shrubs and at least 5% flowering areas within the vineyards
Green vineyard	Vineyard to be managed as an ecosystem
Irrigation	Strict demands
Chemical plant protection products	Forbidden
Copper application	Stage 1: Maximum of 3.4 kg / ha / year
	Stage 2: Maximum of 2.9 kg / ha / year
	Stage 3: Maximum of 2.4 kg / ha / year
Artificial fertilizer	Forbidden
Genetically modified organisms (GMOs)	Forbidden
Vinification: processing	Technical procedures such as vacuum evaporators, reverse osmosis and cryoextraction are prohibited
Vinification: animal auxiliaries	From 2017 the Delinat directives exclude all animal-derived processing aids in wine production and prohibit fertilizers containing slaughter waste. From 2017 onwards all Delinat wines meet the requirements of the international Vegan definition and can be declared as vegan without exception.
Vinification: other processing aids	Strict requirements for sulphur, gelling agents and filtration aids
Incentive to the further development of the winemakers	Guidelines are based on a step model with three quality levels
Renewable energy	From 2021 onwards, each Delinat-certified company must generate a minimal share of renewable energy in the farm.
	Level 1: 30%
	Level 2: 60%
	Level 3: 100%
	By the time the 100% renewable energy target is achieved, each company will additionally demonstrate that at least three measures have been taken to implement energy efficiency; for example, energy-efficient cooling, heat recovery, insulation, solar thermal energy or solar and wind power generation.

The schemes highlighted in Table 6.3 address ES/NC and habitat/biodiversity conservation in different ways, which include requiring sites to be managed and maintained as an ecosystem complex (Denilat Bio Garantie, Effinature); requiring set-asides (Denilat Bio Garantie, Demeter Biodynamic); requiring offsets (Best Aquaculture Practices;); requiring policies and protocols concerning the management of wildlife interactions with predator species (Best Aquaculture Practices); and controlling risks to wild species arising from exposure to farmed species (Best Aquaculture Practices, Denilat Bio Garantie, Demeter Biodynamic, Local Food Plus, Salmon Safe) as well as through soil, water, energy and waste management. The reduction or elimination of synthetic inputs to production processes and/or releases of contaminants is a wide requirement across schemes and includes controls on a wide range of substances. While some of these substances are relevant across the whole farming sector, such as synthetic nitrogen fertilizer, others are specific to particular activities, such as use of hormones in the gender control of farmed fish. This highlights why standards need to be established on a case-by-case basis.

The Denilat Bio Standard for viticulture, vinification and wines (Table 6.3 and 6.4) has detailed requirements covering ecological balancing areas. The whole vineyard is to be managed as an ecosystem and is to be maintained year-round with green areas except in dry regions. Plants other than vines must be grown, including fruit trees, wild shrubs and wild flowers. An ecological compensation area at least 12% of the area under vines must be established, with 7% within or directly adjacent to the vines and 5% within 1000 metres of the vines. From 2021, vineyards that are Denilat-certified must generate a minimum share of renewable energy on site (three levels are specified: 30%, 60% and 100%) and implement energy efficiency measures.

An important approach illustrated by one of the schemes listed in Table 6.3, the Global Green Tag, involves establishing performance benchmarks for products within functional categories as a basis for assessing the relative performance of products. In this scheme, products are assessed on 20+ criteria across six sustainability dimensions based on I-O and LCA data and on biodiversity impact. Weighted scores are used to derive an EcoPOINT score with a possible range from -1 to +1. This is used to differentiate products from others serving the same function or purpose. Products within the top end of the green product market are then allocated to one of four levels: good, very good, excellent and world-leading.

An important emerging habitat and biodiversity concern is the development of the market for products derived from bio-products, including biofuels, bioenergy, bio-plastics, bio-packaging and bio-chemicals. as alternatives to synthetic organic products derived from fossil fuels. The Roundtable on Sustainable Biomaterials (RSB) is an international initiative, which develops and implements a global standard for sustainable production, conversion and use of biomass. The RSB Global Sustainability Standard seeks to demonstrate sustainable production of all derivatives from biomass. The scheme has been operating since July 2011. RSB is a Full Member of ISEAL. It uses independent (third-party) inspectors and auditors. The significance of RBS certification is that it is recognised by the EU as proof of compliance with the Renewable Energy Directive (2009/28/EC). Any RSB certified operator producing liquid biofuels can access the EU Market without further verifications.

The RSB has developed a GHG Calculation Methodology for the lifecycle GHG emissions of biofuels (RSB-STD-01-003-01). It sets a requirement for minimum greenhouse gas (GHG) emission reductions for biofuels replacing gasoline, diesel and jet kerosene compared with the applicable fossil fuel baselines (RSB-STD-01-003-02). The scheme is, however, more comprehensive than only requiring biofuels to mitigate climate change. It seeks to ensure: legal compliance, human and social rights, environmental conservation and protection, and effective management. These four themes are backed by 12 principles, each with several attaching criteria, as foundations for sustainable production. These include, for example, ensuring traditional land and water rights; ensuring rural and social development in regions of poverty and local food security; preserving conservation values, soil health, water quality and availability, climate change mitigation and the control of air pollution; and effective risk management and continuous improvement. The RBS principle on conservation (Principle 7) is concerned with avoidance of negative impacts on biodiversity, ecosystems and conservation values. Five criteria are defined to help make this principle operational for implementation purposes, including protection, restoration or creation of ecological corridors to minimise fragmentation of habitat and creation of set-aside and buffer zones to protect these.

There is, as yet, no standard listed on the Ecolabel Index that addresses mining. A mining standard is under development by the Initiative for Responsible Mining Assurance (IRMA), but has not yet been fully implemented. IRMA is a coalition of mining companies and trade unions, minerals and metals purchasing businesses, affected communities, and nongovernment organizations, which was formed in 2006 to establish a multi-stakeholder, independently-verified standard for responsible mining. The IRMA Standard has a broad scope. It is targeted at industrial-scale mines across all locations, commodities and mine types, with the exception of energy fuels. The scheme is not yet fully operational: IRMA is planning to beta test its certification system in 2017, but mine sites participating in the test will be eligible for certification.

The IRMA Standard sets out requirements for: business integrity; social responsibility; environmental responsibility; and positive legacy. The Environmental Responsibility element sets out an approach to conservation based upon a distinction between officially protected areas and areas outside officially protected areas. It also sets out criteria and verification requirements for water quality and quantity, air quality and GHG-emissions, mine waste management and the management of chemicals (cyanide, mercury) used in mining operations.

The Standard bans or restricts mining- related activities in or adjacent to different categories of formally protected areas. For this, the Standard distinguishes between three kinds of officially protected area: Highly Protected Areas (HPA) and two categories of other Protected Areas (Table 6.5). IRMA will not certify mines in Highly Protected Areas unless mining-operations pre-date HPA designation. Other protected areas are treated as special cases where conservation values are prioritized, but where mining-related activities may take place so long as such activities can be shown to be compatible with the maintenance of the values that the areas are designed to protect (Protected Areas I) or that the company can demonstrate a net positive impact on biodiversity (Protected Areas II).

Table 6.5 IRMA Standard – Mining Restrictions in Officially Protected Area

Highly Protected Areas (HPA)	Protected Areas (I)	Protected Areas (II)
No-Go Areas (unless areas were designated as HPA after mining-related activities were occurring already)	Mining allowed if company can demonstrate mining is compatible with maintenance of area’s special values	Mining allowed if company can demonstrate net positive impact on biodiversity
• World Heritage Sites; • Sites on a State Party’s official Tentative List for World Heritage Site inscription; • IUCN category I-III protected areas; • IUCN category I-V marine protected areas; • Core areas of UNESCO biosphere reserves; and • Areas where indigenous people live or where it is assumed that they might live in (voluntary) isolation	• IUCN category V-VI protected areas; • Natura 2000 sites; • Indigenous and Community Conserved Areas (ICCAs) in which free, prior and informed consent (FPIC) has been demonstrated; • Important Bird Areas (IBAs); • Official buffer zones of sites designated as Highly Protected Areas, and other areas outside the boundaries of Highly Protected Areas in which mining activities may affect the values for which the Highly Protected Area was designated for protection; and • Other officially designated protected areas.	• IUCN category IV protected areas; • Ramsar sites that are not IUCN category I- III protected areas; and • UNESCO Biosphere Reserves beyond the core areas

The management of biodiversity more generally, including its management outside areas that are formally protected is addressed using the High Conservation Values (HCVs) approach developed by the Forest Stewardship Council (FSC). This approach to mitigation and offsetting has been incorporated also into other standards as a generic, best-practice approach. The HCV approach (Box 6.1) establishes guidance for a three-step approach to identifying, managing and monitoring high conservation values represented by natural capital and ecosystem services, including significant habitat and biodiversity. Six different categories of HCV are defined for identification, management and monitoring. The approach draws on knowledge from experts and stakeholders, including the knowledge and values of local communities.

Box 6.1 The FSC High Conservation Values Approach
The FSC sets out common guidance for identifying, managing, and monitoring habitat/biodiversity of High Conservation Value (HCVs). It recognises six HCVs. • HCV1: Concentrations of biological diversity including endemic species, and rare, threatened or endangered species that have significance at global, regional or national levels. • HCV 2: Landscape-level ecosystems and mosaics. Intact forest landscapes and large landscape-level ecosystems and ecosystem mosaics that are significant at global, regional or national levels, and that contain viable populations of the great majority of the naturally-occurring species in natural patterns of distribution and abundance. • HCV 3: Rare, threatened, or endangered ecosystems, habitats or refuges. • HCV 4: Basic ecosystem services in critical situations, including protection of water catchments and control of erosion of vulnerable soils and slopes. • HCV 5: Sites and resources fundamental for satisfying the basic necessities of local communities or indigenous peoples for livelihoods, health, nutrition, water, etc., identified through engagement with these communities or indigenous peoples. • HCV 6: Sites, resources, habitats and landscapes of global or national cultural, archaeological or historical significance and/or of critical cultural, ecological, economic or religious/sacred importance for the traditional cultures of local communities or indigenous peoples, identified through engagement with these local communities or indigenous peoples.
The FSC describe a three-step process for identifying, managing and monitoring HCVs. • <u>Identification:</u> Identifying the presence or absence of HCVs involves interpreting what the six HCV definitions mean in the local or national context and deciding which HCVs are present in the area of interest (management unit, plantation, concession, etc.) or which HCVs in the wider landscape may be negatively impacted by project activities considering that impacts on water or wetland HCVs may occur well beyond the border of the management unit. The identification of HCVs involves an 'HCV assessment', which includes stakeholder consultations, analysis of existing information and the collection of additional information where necessary. HCV assessments should return a report on the presence or absence of HCVs, their location, status and condition. The report should provide information on areas of habitat, key resources, and critical areas that support HCVs as a basis for developing management recommendations to ensure that HCVs are maintained and/or enhanced.

Box 6.1 (cont): The FSC High Conservation Values Approach
• <u>Management:</u>Designing a management regime for HCVs should include investigation of existing and potential threats from proposed management activities, such as logging operations or plantation establishment, or from external activities such as hunting, illegal logging or construction of a new road or dam, and the establishment of management requirements. This can include delineating areas that need total protection and identifying areas that can be used for production provided that management is consistent with maintaining or enhancing HCVs subject to controls and policies; e.g. anti-poaching controls or fire management policies. For purposes of mapping and planning, the approach distinguishes between the locations of HCVs, which may be quite small and sometimes confidential and the sometimes much larger management areas where appropriate decisions and actions are needed. The issues will need to be discussed by the experts involved in the assessment using appropriate formats, such as discussion workshops. The outcomes should be documented, as these will be important to justify future decisions.
• <u>Monitoring:</u> A monitoring regime should be established to ensure that management practices effectively maintain and/or enhance the HCVs over time. The monitoring regime needs to translate the strategic objectives of the management regime into operational objectives. Appropriate indicators for these operational objectives must be chosen to assess the status of the HCVs, and thresholds for action to ensure that the HCVs are maintained or enhanced. Indicators and thresholds for action are likely to be site and/or country-specific. Strategic monitoring (i.e. monitoring of the HCVs) provides for corrective actions to be taken if any negative changes are identified. Operational monitoring (i.e. of the proposed management measures) is advisable also to ensure that management measures are actually being carried out as planned. Operational monitoring can help to identify potential problems before they actually become manifested and are detected through strategic monitoring.
The FSC advises that the three-step HCV approach should always be: • **Knowledge-based**, incorporating and using all relevant scientific data and local knowledge; • **Precautionary** in the face of gaps in existing information; • **Participatory and inclusive**, ensuring that relevant stakeholders are consulted and their views or the information they provide is incorporated into the process and that appropriate existing initiatives are engaged wherever possible; • **Open and transparent**, including public reporting of outcomes.

The IRMA Standard requires mining operators to carry out a Biodiversity Impact Assessment (BIA) covering both past and potential future impacts of its mining-related activities on biodiversity. This is to include direct, indirect and cumulative effects of proposed mining-related activities on biodiversity and actual and potential impacts associated with the project, both positive and negative, from the exploration phase onwards. The assessment is to consider past and potential future impacts on High Conservation Values 1 - 3 (HCV 1 - 3), including fish and wildlife, wetlands, and species listed as threatened or endangered; options to restore or offset past impacts from mining-related activities; and options to avoid, minimize, restore or offset potential future impacts. The BIA is to be carried out in consultation with stakeholders and must be made publicly available.

The BIA must be backed up by a Biodiversity Management Plan (BMP). The BMP is to follow the mitigation hierarchy of avoiding, minimizing, restoring and/or offsetting potential future impacts on biodiversity, prioritising the avoidance of existing protected areas, wetlands and areas containing or impacting on HCVs 1 – 3. It must describe the specific objectives, timelines, locations and activities that it shall implement to minimize, restore and/or offset any past or potential future negative impacts on biodiversity, demonstrate that impacted wetlands will be replaced on a “no net loss” basis; and demonstrate that the net impact of the operating company’s mining-related activities on biodiversity will be neutral or positive over the lifetime of the project. Biodiversity management planning is to be carried out and documented by competent professionals using best practice procedures to: identify key biodiversity indicators sufficient to monitor the impact of the operating company’s activities over time, and to demonstrate that the overall net impact is neutral or positive; conduct surveys or baseline studies to establish the status of the key biodiversity indicators prior to the commencement of site-disturbing operations; develop mitigation measures to be implemented to minimize negative impacts on biodiversity associated with specific operations or processes, and to enhance, protect or restore biodiversity; and develop a process for updating the plan if new information relating to biodiversity becomes available during the implementation of the mining project. The BMP is to be developed in consultation with stakeholders.

The operating company must also develop and implement a program to monitor the implementation of its BMP and the specified key biodiversity indicators over time. If monitoring shows that the biodiversity objectives are not being achieved as expected, the operating company must define and implement timely and effective corrective action in consultation with interested stakeholders. The Monitoring and Corrective Action Plan (MCAP) is to be developed and implemented in sufficient detail and regularity to evaluate the success in achieving the BMP objectives. The operating company is required to allocate sufficient personnel and other resources for full and effective implementation and monitoring of the BMP. The findings of the monitoring program are to be subject to professional review and must be made publicly available.

The IRMA Standard is therefore based on integrating mitigation and offsetting into mining operations using the High Conservation Value concept to achieve objectives that specify No-Net-Loss of Biodiversity at minimum. The Standard is to be implemented using professional experts involving consultation with stakeholders in the identification of HCVs and the development and implementation of the BIA, BMP and MCAP. The resourcing requirement is intended to ensure sufficient resources are earmarked for the process. Transparency is integrated through requirements to make all assessments publicly-available.

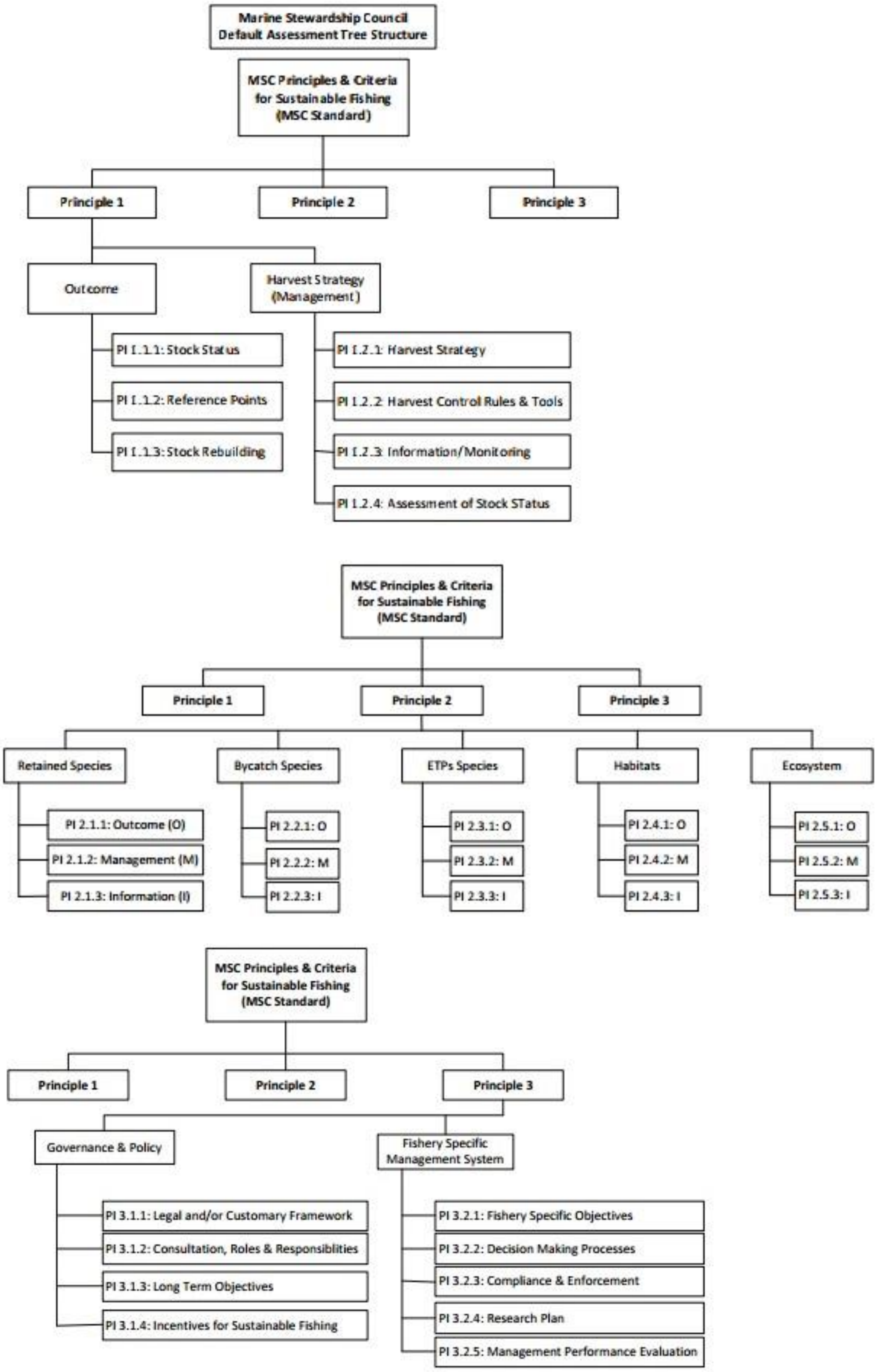
Other schemes of high habitat/biodiversity importance are those that address practices in sectors, such as fishing, where production involves the harvesting of wild species. The fishing industry has high ecological relevance not only because of harvesting of wild fish stocks for direct human consumption, but also because of the increasing absolute and relative importance of aquaculture in overall global fish consumption and, connected to this, growth in the harvesting and processing of low trophic level (LTL) species in the production of fishmeal and fish oil, which are important feedstocks in fish farming.

Different standards have been developed to address these different elements of the sector. The Marine Stewardship Council (MSC) is a program for wild fisheries and does not include aquaculture production. MSC is an independent non-profit organization. MSC was founded initially in 1996 by Unilever in association with the Worldwide Fund for Nature (WWF). It became independent of its founding partners in 1999. MSC has a related Chain of Responsibility Standard to ensure traceability of fish and enable fish to be labelled. The MSC also includes standards for capture of LTL species. The IFFO offers a standard covering fishmeal and fish oil production. Aquaculture is addressed by standards developed by the Aquaculture Stewardship Council (ASC) an independent non-profit organisation which sets a standard for sustainable aquaculture. ASC was founded in 2010 by WWF and the Dutch Sustainable Trade Initiative (IDH).

The MSC sustainable fishing standard was developed over two years through a consultative process involving more than 300 expert organizations and individuals. It is consistent with the ‘Guidelines for the Eco-labelling of Fish and Fishery Products from Marine Wild Capture Fisheries’ adopted by the UN Food and Agriculture Organization (FAO) in 2005. The MSC standard (Figure 6.2) is structured around three core principles and a set of performance indicators (PI). Principle 1 (sustainable fish stocks) states that fishing activity must be at a level which is sustainable for the targeted fish population and that any certified fishery must operate so that fishing can continue indefinitely and not overexploit the resources. Principle 2 requires that fishing operations are managed to maintain the structure, productivity, function and diversity of the ecosystem on which the fishery depends. Principle 3 is concerned with governance and management. The fishery must meet all local, national and international laws and have a management system in place to respond to changing circumstances and maintain sustainability.

The original version of the standard developed 31 performance indicators (PI) divided across the three principles (Figure 6.2). The current version has removed some redundancy and uses 28 PI. Fisheries are scored against the PI by an independent team of experts. Pass scores and related descriptors are established for each indicator. There are three different levels of pass scores reflecting successively higher probability that key elements of the ecosystem on which the fishery has an impact are not compromised. These bands range from the minimum pass score (60) to the highest pass score (100). To qualify for certification, the fishery must score a minimum 60 on every PI and have an average score of 80 across the PIs under each of the three principles. This is intended to ensure that every fishery certified against the MSC standard is operating at a very high level of precaution.

Figure 6.2: Structure of the MSC Standard



Since a certified fishery must achieve average scores of at least 80 across the three principles it is possible for a fishery to be certified with a score of between 60 and 80 for some PI. In these cases conditions are placed on the fishery, which it must fulfil within a set period, in order to remain certified. The fishery must introduce a plan of action that will raise its performance to at least 80 within a set period of time; e.g. by the start of the next certification period. To remain certified, fisheries also have to undertake an annual surveillance to check that they continue to meet the MSC standard. After 5 years, the fishery must be reassessed in full if it wants to continue to be certified.

In principle, the MSC system takes account of the impact of the fishery on the target stock, but also on other components of the wider ecosystem, such as habitat structure, productivity and biodiversity.

- The PI for Principle 1 (sustainable fish stocks) include: the stock status, which is to be at a level that maintains high productivity and has a low probability of recruitment overfishing; the development of limit and target reference points that are appropriate for the stock; and where the stock is depleted, evidence of an effective time-limit specified stock-rebuilding strategy. There must be: a robust and precautionary harvest strategy; well-developed and effective harvest control rules; and systems in place to collect information to support the harvest strategy and for assessing stock status. The standard draws on biological reference points as stock status indicators (e.g. B_{MSY} and F_{MS}) and for use in developing harvesting strategies, but also allows proxies based on fishing effort to be used. In that event, checks are required to be made on the effectiveness of these to make sure that biomass reference points are met. In relation to harvest control rules guidance refers to indices of exploitation rate (e.g., fishing mortality reference points in relation to benchmarks, such as F_{MAX} or F_{LIM}) to make sure that biomass reference points are met.
- The PI for Principle 2 (minimising environmental impact) require that the fishery does not pose a risk of serious or irreversible harm to the retained species and does not hinder recovery of depleted retained species. This must also be a management strategy and adequate information and monitoring to support the management strategy and its effectiveness. Equivalent requirements are specified for: bycatch species; endangered, threatened or protected (ETP) species listed under Annex 1 of CITES; habitats; and ecosystems. In relation to habitat, the standard requires that the fishery does not cause serious or irreversible harm to habitat structure, considered on a regional or bioregional basis, and function and that there should be a habitat management strategy and supporting information and monitoring in place. In relation to ecosystems, the standard requires that the fishery does not cause serious of irreversible harm to key elements of ecosystem structure and function, that there should be an ecosystem management strategy in place, and that information and monitoring should be adequate to support the strategy; e.g. through adequate knowledge of the impacts of the fishery on the ecosystem.

- The PI for Principle 3 (effective management) concerns the governance framework and fishery-specific management system operating within that framework. The management system must be consistent with and respect effective legal and/or customary frameworks, observe the rights of people dependent on the fishery for food or livelihood, and incorporate procedures for dispute resolution. Other indicators include the existence of effective stakeholder consultation processes, clear long-term objectives to guide decision making that are consistent with the MSC and a precautionary approach, and social and economic incentives for sustainable fishing. The fishery must not be supported by subsidies that contribute to unsustainable fishing. The fishery-specific management plan must have: clear and specific objectives to achieve outcomes expressed by the MSC; effective decision-making processes; monitoring, control and surveillance mechanisms to ensure compliance, including sanctions to deal with non-compliance; a research plan that addresses the information needs of managing the fishery; and a system for monitoring, evaluating and reviewing the performance of the management system against its objectives.

The MSC also operates standards for the exploitation of Low Trophic Level (LTL) species. Allowable harvesting rates are determined using a model of energy transfers between trophic levels within ecosystems.

6.8 Insights from the status review for market development

Different actors face different issues. Certification scheme providers face issues of scheme governance and design. Producers and processors face the issue of whether to apply for certification and how to choose the most appropriate label for their product. The issue facing consumers is how to compare and interpret different standards. Consumers also need to decide which labels and claims are credible, in which they can place faith and what price premium they are willing to pay for sustainability assurance of the production process. Greater transparency about standards and their effectiveness is needed for credibility and to support the design of effective schemes, the wider uptake of schemes, and informed choices by the concerned stakeholders. The various issues facing these different actors are interconnected.

From the perspective of environmental protection and habitat/biodiversity conservation goals, the market for certification and labels is still immature.

- The certification landscape is over complex. There is a proliferation of schemes, but not all market segments are covered. There are too many standards in some segments and too few in others. There are many instances of overlapping standards. The existing standards are not harmonised. This can create confusion for stakeholders, adding to difficulties for producers in deciding which schemes and standards to join and for consumers in interpreting labels, placing trust in them and paying price premiums for quality assurance.

- Different certification and labelling schemes are in different phases of development.
- The process of integrating ES/NC concepts into schemes is at an early stage generally. Habitat and biodiversity conservation are not yet well represented. Existing standards mostly address indirect drivers of habitat and biodiversity loss.
    - The use of quantitative, substance-flow based metrics and accounting tools, such as I-O, LCA, environmental profiling and footprinting is well established.
    - The use of sustainable yield metrics and models is established for fisheries, but there is uncertainty over the robustness of some models.
    - The integration of mitigation and offsetting into standards, certification and labelling in protecting habitat and biodiversity is at an early phase of development.
    - There is a lack of knowledge and evidence about the relation between management actions and biodiversity, which is a barrier to developing more fully specified standards.
- The potential of certification and labelling to contribute to habitat and biodiversity conservation is still largely to be developed, captured and demonstrated.
- The process of market development involves translating the still largely conceptual theory of change into a real and demonstrable mechanism for change. Until now, few schemes have been monitored at the needed level of detail to gather evidence of how well (or not) the mechanisms hypothesised at each causal link in the theory of change are working. This applies for each step in the theory of change.
- The lack of studies and evidence is partly because certification schemes have been operating only for relatively short periods and partly because few schemes include explicit and measurable habitat and biodiversity standards and criteria.
    - Increasing the take-up of certification and labelling schemes by all stakeholders depends, especially, on providing evidence of both market effectiveness and ecological effectiveness.
    - There is currently insufficient evidence of the impact of certification on market share; most certification schemes do not routinely track their market impact even though this is key management and marketing information for schemes, critical for incentivising scheme take-up.
    - There is uncertainty over the impact of certification on net farmer/producer income.
    - There is uncertainty over the effectiveness of certification and labelling in improving environmental performance and conserving habitat/biodiversity. Monitoring of ecological effectiveness is needed to underpin scheme legitimacy and credibility.
- The availability of skilled personnel able to undertake certification work is a constraint on the rate at which credible and trustworthy schemes can be developed.

- Performance improvement, demonstrating market/ecological effectiveness and market development are dynamic processes. In the earlier stages of market and scheme development, outside financial support to schemes and other forms of inducement are important for reducing financial risks to producers and processors. Underwriting financial risk encourages participation in schemes and is part of pump-priming for building markets for certification and labelling.

6.9 Lessons and guidance for scheme development and design

Lessons for scheme development and design can be learned from meta-analysis of existing schemes, front-runner schemes and their 'best practices', and analysis of market trends, including the changing purposes served by implementations and changes in technology.

Overall market development:

- There is a need for new standards to address under-represented areas of economic activity that have high habitat and biodiversity impacts, such as the mining and extractive industries.
- There is a need to rationalise and harmonise the overall market for certification across available schemes and standards.
- There is a need to increase the sustainability focus of each certification scheme.
- The roles of certification in the different relationships between actors should be developed: 'business to consumer', 'business to business' and 'business to investor'.
- Studies and evidence of the direct and indirect chain of impacts linking certification and labelling schemes with environmental protection and habitat/biodiversity conservation need to be further developed. Specifically, studies and evidence are lacking concerning:
    - The impact of specific labels on consumer choices, market shares and product prices and the stability of any changes in these.
    - The impact on incomes of upstream producers and processors, especially, how much of any price premium is returned to farmers and producers.
    - The impact of any increase in net producer income on improving commodity production and processing practices via extra investment in these.
    - The impact of practice improvements on other relevant variables where production takes place (increased yields, improvements in product quality, reduced water consumption, reduced chemical use, improved soil quality and depth, etc.).

- Impacts on improved production and processing practices on environmental protection, habitat conservation and biodiversity loss on and off production sites and areas.

Individual schemes:

- Individual schemes could make more explicit use of ES/NC and related concepts, such as set asides, protected areas, offsetting and the No-Net-Loss principle. The concepts could be integrated into scheme principles, standards and actions in respect to how and where biotic and abiotic materials should be produced, extracted, or harvested from ecosystems, how and where they should be processed, and how they should be handled throughout the value-adding chain.
- Individual schemes should be developed around clear, unambiguous and measurable technical standards that are straightforward to apply and to understand.
- An important issue in scheme design is to balance standard strictness against the cost burden they pose. If standards are perceived by producers and processors to be too costly to implement (set too high) they will deter potential participants from joining schemes. Standards should therefore be set as attainable targets and progressively reviewed and made more demanding.
- Standards should be set in relation to existing performance as well as in relation to environmental protection and habitat/biodiversity conservation goals. Incentives are needed to enable leading companies to continue to innovate and pull-up other producers through their example. At the same time, there is a need to incentivise and support relative performance improvements by all producers and processors. If all producers and processors are to be included in a single scheme, different levels or tiers of certification can be offered with each tier targeting producers with different current levels of ecological performance and facing different improvement challenges and possibilities. Otherwise, schemes risk that standards are pulled down in the attempt to increase scheme take-up.
- Individual schemes can be developed to recognise progress as well as absolute ecological performance attainment. Using different categories of certification, schemes can be designed to offer opportunities to recognise improving practices; i.e. practices that might not yet meet the highest standards set by the scheme but are in an active process of being improved.
- Granted certificates must be time-limited to support progressive improvements in ecological performance over time.
- Individual B2C schemes should be designed for *market success*. Market success is related to the certainty of increase of net income of upstream producers and processors. This depends on the ability of certification to *deliver and demonstrate market benefits* through market access, market share, price premiums and

economies of scale and on having *inbuilt mechanisms for transferring market benefits back to producers*.

- The economic and ecological effectiveness of individual schemes should be monitored routinely.

Governance:

- Inclusion of stakeholders across the supply chain and stakeholder interaction in scheme design and governance is important for defining technical standards, including how to integrate habitat and biodiversity conservation into schemes and for increasing awareness and gaining acceptance and buy-in to schemes.
- Inclusion of developing country producers is important. Their buy-in is essential for the ecological effectiveness of schemes. Integrating their representatives into scheme governance and their interests into scheme design is critical for scheme take-up and the prospects of schemes delivering better environmental and habitat/biodiversity outcomes.
- For reasons of transparency, legitimacy, and driving progressive improvements in environmental performance individual schemes should have explicit (documented) procedures for standard setting, review and approval.
- Individual schemes should use independent (third-party) inspectors for certification and verification to provide an additional source of reliability assurance.

Context:

The purposes that schemes are designed to serve are changing. Retailers and brand owners wanting to use certification to drive innovation and improvement among their suppliers may opt to develop their own standards and to use these alongside or instead of those provided by third-parties. This could offer a more outcome-focused approach, giving producers greater flexibility in the ways they achieve sustainable outcomes and providing incentives for producers to improve processes. It could also offer retailers and brand owners a way to focus on sustainability aspects important to their business, customers and stakeholders when these are not represented adequately in third-party standards. Unilever's Sustainable Agriculture Code is in this direction.

Developments in information technology are changing the implementation context for the design and use of certification and labelling:

- New ways of information sharing could limit the role of B2C implementations by offering customised ways to provide consumers with information they want using smart phones, apps and real-time enquiry-response facilities.
- Such developments would reinforce the shift in certification/labelling from predominantly B2C implementations to B2B implementations. Developments in supply chain visualisation and traceability software that will enable retailers and

brand owners to rapidly pinpoint the source of problems in the event of supply chain quality issues will also strengthen the control possibilities under B2B implementations.

- Social networking sites and tools for brand ranking will support shifts toward retailers and brand owners developing their own certification/labelling schemes and toward certification scheme owners re-orienting their activities toward the certification of companies and their practices.

Possible roles of the different direct actors in relation to core implementation issues are set out in Table 6.6

Table 6.6	Principle actors & roles in implementing certification/labelling and incorporating ES/NC/biodiversity into schemes		
Actors	**Issues**	**Actions/Roles**	**Inclusion of ES/NC/Habitat/Biodiversity**
Government	Design of standards	Identify key sustainability objectives and collaborate with other actors on these. Support knowledge and research institutions in developing clear, measurable indicators and technologies for measuring and monitoring impacts. Address weaknesses in market development and self-regulation; e.g. proliferation of standards, lack of transparency, lack of proven impact of standards on sustainability.	Work internationally to strengthen biodiversity goals in international frameworks. Work with other actors to develop voluntary agreements on required criteria and targets for ES/NC/biodiversity in certification schemes.
	Driving uptake and improving environmental outcomes	Support market development and self-regulation through collaboration with umbrella organisations, such as ISEAL. Help mainstream sustainability by identifying sectors which need extra stimulation, such as the extractive industries, and the construction sector. Identify and producers facing difficulties in joining schemes (e.g. poor, small-scale farmers) and support these through international development policies.	Set up benchmarking schemes to establish front-runners. Support front-runners. Use international development aid to support small farmers in developing countries to participate and address their ES/NC/biodiversity impacts.
	Demonstrating environmental outcomes	Support the development of clear and measurable environmental indicators. Fund and otherwise support research on scheme effectiveness.	Establish critical ES/NC/biodiversity targets. Support the development of a set of basic, clear, measurable indicators for ES/NC/biodiversity. Identify, prioritise and address evidence gaps of the impact of standards on ES/NC/biodiversity; e.g. by supporting work of knowledge and research institutes.
	Stimulating demand for certified products	Use market access, tendering policies and green public procurement policies to influence the market.Develop minimum statutory standards. Monitor statutory standards. Strengthen import controls, including traceability requirements and bans on illegally-sourced commodities.	Integrate ES/NC/biodiversity criteria into policies relating to market access, green public procurement, and tendering processes. Ensure coverage of the whole supply chain and all sectors with high ES/NC/biodiversity impact potential, including spatial planning, infrastructure and building design, materials choices, construction, food, catering, etc. Develop procurement instruments for performance on ES/NC/biodiversity criteria, such as CO2 emissions. Where scope allows, provide tax and subsidy incentives for certified commodities and products of schemes that have a clear ES/NC/biodiversity perspective.

Table 6.6	Principle actors & roles in implementing certification/labelling and incorporating ES/NC/biodiversity into schemes		
Actors	**Issues**	**Actions/Roles**	**Inclusion of ES/NC/Habitat/Biodiversity**
NGOs	Design of standards	Collaborate with all actors. Share knowledge and exert pressure to use knowledge in schemes	Exert pressure to include explicit and measurable ES/NC/biodiversity criteria in standards.
	Driving uptake and improving environmental outcomes	Act as a watchdog for certification schemes. Highlight good and bad practices within schemes and monitor actors and their performances so they can be held to account.	Exert pressure on scheme owners, government and businesses in relation to ES/NC/biodiversity. Demand accountable certification schemes and accountable sustainable supply chain models that include these aspects.
	Demonstrating environmental outcomes	Exert pressure on other actors to ensure transparency in respect to standard setting, impact disclosure and traceability. Communicate actively with civil society to raise awareness and maintain pressure on actors to demonstrate environmental effectiveness.	Communicate actively with civil society to require disclosure of environmental impact generally and ES/NC/biodiversity impact specifically.
	Stimulating demand for certified products	Communicate with civil society to raise awareness and maintain growth in demand for sustainable products.	Focus communication and awareness efforts on ES/NC/biodiversity loss as key issues, on priority products and sectors with high impacts in these areas, and practices that could be used in the relevant sectors to improve environmental performance.

Table 6.6	Principle actors & roles in implementing certification/labelling and incorporating ES/NC/biodiversity into schemes		
Actors	**Issues**	**Actions/Roles**	**Inclusion of ES/NC/Habitat/Biodiversity**
Scheme Providers	Design of standards	Collaborate with knowledge holders, research institutions and other actors to identify priority sectors for certification, to specify appropriate principles, standards and actions within certification schemes, and to define clear and measurable criteria. Contribute to research on business front-runners.	Collaborate with research institutions to further research on ES/NC/biodiversity and their incorporation within schemes. Organise and contribute to roundtables to develop ES/NC/biodiversity criteria.
	Driving uptake and improving environmental outcomes	Increase sectoral coverage to include high concern sectors for which there no certification schemes currently. Collaborate with research and knowledge institutes to improve monitoring systems and metrics for schemes; undertake own or commission research into the market effectiveness and environmental effectiveness of schemes; and analyse market trends relevant for future scheme evolution and effectiveness as a basis for modifying scheme designs and establishing a strong business case for scheme take-up. Collaborate with other certification schemes to share knowledge, develop training materials, identify best practices, etc. Collaborate with other schemes to simplify and rationalise schemes and to reduce the cost and administrative burden to producers of having to satisfy requirements of multiple schemes. Link collected information on indirect impacts to changes in ES/NC and biodiversity.	Improve coverage of high impact sectors, such as mining, mineral processing and construction that change land uses and have high ES/NC/biodiversity impact potential. Train and equip producers, inspectors, auditors and trainers with information and tools to better understand ES/NC/biodiversity and their conservation. Identify and address knowledge and capacity gaps in relation to ES/NC/biodiversity. Innovate with using new techniques and technologies for benchmarking and monitoring biodiversity. Collaboration among scheme providers to rationalise and simplify schemes would give scope to include additional ES/NC/biodiversity standards. Link collected information on indirect impacts to changes in ES/NC and biodiversity loss.
	Demonstrating environmental outcomes	Collaborate with research and knowledge institutes in monitoring the environmental effectiveness of schemes.	Collaborate with research and knowledge institutes in monitoring certification impacts on ES/NC/biodiversity, using controls to improve the quality of the evidence base.
	Stimulating demand for certified products	Develop schemes for certifying companies and their practices as well as (or as an alternative to) certifying materials, commodities and products.	Adapt schemes to enable mainstreaming and the more structural inclusion of biodiversity criteria in current and new schemes.

Table 6.6	Principle actors & roles in implementing certification/labelling and incorporating ES/NC/biodiversity into schemes		
Actors	**Issues**	**Actions/Roles**	**Inclusion of ES/NC/Habitat/Biodiversity**
Businesses	Design of standards	Collaborate with all actors in pre-competitive efforts. Where schemes provide insufficient ES/NC/biodiversity protection, develop own standards for sourcing and procurement.	Identify ways to integrate ES/NC & biodiversity into standards. Analyse the supply chain to identify high-impact (priority) commodities and how ES/NC/biodiversity impacts arise in order to identify ways to mitigate impacts.
	Driving uptake and improving environmental outcomes	Develop sourcing policies to meet demand for certified products. Incentivise the supply of sustainable products to create scope to include additional ES/NC/biodiversity criteria in standards. Collaborate with other actors and with knowledge institutes to identify actions that can be taken to mitigate environmental impacts.	Include ES/NC/biodiversity requirements in sourcing specifications. Collaborate with other actors and with research/knowledge institutes to identify best practices and prescribe actions to mitigate impacts on ES/NC/biodiversity.
	Demonstrating environmental outcomes	Collaborate with research and knowledge institutes in monitoring the environmental effectiveness of schemes.	Collaborate with research and knowledge institutions in monitoring environmental effectiveness and linking information on indirect impacts to change ES/NC/biodiversity.
	Stimulating demand for certified products	Set own sustainable procurement standards when scheme standards are insufficient	Include ES/NC/biodiversity requirements in procurement specifications

7. GREEN INFRASTRUCTURE

7.1 Definition and concept

The green infrastructure (GI) concept is complex and ambiguous. The term '*green infrastructure'* has developed in response to different needs in different contexts and has different meanings in different uses. A 2011 report noted that green infrastructure has been referred to as an interconnected network of natural areas and artificial features, an approach, and a conceptual framework for understanding the valuable services nature provides (IEEP, 2011).

To overcome potential confusion, attempts have been made to propose a standard definition, but there are strong arguments to suggest not only that this is problematic, but also that it would be counterproductive given that the concept is still evolving and can be used as a bridging concept at the boundary between domains. This capacity of the concept to be used in different domains and to support multiple perspectives on its use, such as between ecologists and investors, is a valuable aspect.

The EU Green Infrastructure Strategy therefore presents different definitions of the concept. There are references to green infrastructure as a network of ecosystem structures with other environmental features, which are designed and managed to deliver a wide range of ecosystem services. This definition stresses that green infrastructure is *a physical infrastructure* that is part of natural capital. That the physical infrastructure can be designed and managed from the perspective of the ecosystem services it can deliver leads to a second definition that refers to green infrastructure as *a strategy* to enhance natural capital. GI strategies are planned and implemented with the deliberate intent to maintain, enhance and manage the network of ecosystem structures and environmental features and, thereby, maintain and enhance the stream of ecosystem service benefits. A further definition is related to the instrumental use dimension of green infrastructure as a *tool* for providing ecological, economic and social benefits through 'nature-based' solutions, which can be designed either as complements to grey infrastructure and engineering solutions or as alternatives to these.

Important underling concerns relate to two core attributes of GI: connectivity and multifunctionality. Connectivity has both a static, structural component and a dynamic, functional component. Structural connectivity is a physical attribute of landscape, which refers to (and can be measured in terms of) habitat continuity. Functional connectivity is related to the ease of movement of species and the capacity of species to extend from their core areas to new areas. Measurement is related to tracking the numbers and spread of species across the landscape. Multifunctionality is directly related to the ES concept and the fact that GI can deliver multiple ecosystem services and benefits simultaneously from the same area.

Another important aspect of the GI concept involves use of the term 'infrastructure', which highlights that GI is a form of capital. This can help raise the profile and status of green infrastructure in decision making, raise awareness of the benefits it provides and the need for investment to maintain and enhance natural capital as a way to sustain and increase ecosystem service benefits that flow from it. It also opens the way to making comparison with 'grey' infrastructures, which typically have single-purpose designs, especially regarding the complementary flow of service benefits green infrastructure provides and the possibility that green infrastructure can offer 'nature-based' alternatives to conventional ways of providing ecological, economic and social benefits.

Nature-based solutions make use of the capacity of ecosystems, natural processes and green infrastructure to supply services that may be better and more cost-effective to deliver than similar services provided artificially using grey infrastructure. Nature-based solutions making use of ecosystem services for water cleansing and waste-water treatment are alternatives to grey infrastructure provision, for example.

Interest has increased recently in making use of the multifunctionality of GI to provide risk management benefits, especially in the context of climate change that has highlighted the limitations of engineering solutions to coastal protection and flood risk management. Initiatives to 'make space for water' and to address coastal squeeze through 'managed realignment' are in this direction. Nature-based coastal protection solutions that involve working with nature, rather than seeking to resist natural forces, include restoring salt marshes, coastal dune or mangrove ecosystems and creating oyster 'reefs'.

Interest has increased also in the use of urban GI in climate change mitigation and adaptation strategies, such as by channelling air flow through cities along green/blue corridors to provide natural cooling and by creating green roofs and facades to insulate city buildings against temperature extremes.

7.2 Components of Green Infrastructure

Studies of GI initiatives (e.g. IEEP, 2011) identify different components of green infrastructure. These include:

- **Core areas**: are ecosystems and habitats that are healthy and functioning
- **Restoration zones**: are new areas of habitat for specific species or ecosystems to be restored to improve provision of ecosystem services
- **Sustainable use zones**: are zones used primarily for provisioning where ecological quality is to be maintained or improved through sustainable economic management regimes (also known as ecosystem service zones)
- **Urban and peri-urban green/blue areas**: are areas such as parks, gardens, churchyards, cemeteries and canals within towns and suburbs as well as features specifically designed to increase the green/blue area, such as green rooves and green facades
- **Natural connectivity features**: are linear features that provide connectivity naturally, such as riverside vegetation and hedgerows
- **Artificial connectivity features**: are features specifically introduced to assist species movement when otherwise this would be impaired by grey infrastructure,

such as green overpasses and eco-ducts in respect to roads and fish ladders in respect to dams.

The physical definition emphasises that GI is a network. The underlying idea is that interventions to conserve, restore, sustainably manage, extend and connect existing green/blue areas and features will help maintain ecosystem resilience in terms of their capacities to deliver a continuous stream of benefits to society. Deliberate strengthening of network coherence and connectivity is a way to mitigate pressures on ecosystems arising from socio-economic development, such as intensification of land use and fragmentation, and from climate change. GI strategy involves a combination of reducing the stress of socio-economic development by locating and designing development projects so that these do least damage to ecosystems and strengthening ecosystem resilience by improving network coherence and connectivity and by managing exploitation zones more sustainably.

GI strategy links ecosystem resilience and ecosystem benefit security to biodiversity. Healthy ecosystems provide a stream of benefits to society through the provisioning, regulating, support and cultural services they support. The quality and quantity of these is influenced by the ecosystem composition and dynamics, including their coherence and connectivity and the diversity and structure of their habitats, species and genetic resources.

Many strategic (landscape scale) GI initiatives are therefore aimed at increasing the resilience of ecosystems and their associated populations of species of conservation concern.

Core areas are sites of high ecological quality and conservation interest that act as refuges where species can thrive and from which they can disperse. Examples are protected and designated sites, such as Natura 2000 sites as designated under the Birds and Habitats Directives and sites designated under the Ramsar Convention on Wetlands of International Importance, but also other sites of national or regional significance for the habitats and species they represent and contain. While evidence is mixed, studies of protected status areas suggest that designation is beneficial in maintaining the habitat and biodiversity features of sites for which they were designated.

Restoration is aimed at degraded habitats. Ecological restoration appears in the EU Biodiversity Strategy to 2020. The headline target is to restore at least 15% of degraded ecosystems. Restoration can be passive, involving the cessation of damaging activities and natural regeneration, or active, involving targeted management interventions, such as planting. Restored habitats can form important components of GI, but restoration is unlikely to provide full recovery to pre-degraded status. The nature of interventions and knowledge of their effectiveness varies across habitat types. Interventions are easier and more effective for some habitat types (e.g. bog and fen) than for others (e.g. ancient woodland, complex wetland systems).

There are some ‘accepted’ approaches to restoration. In bog and fenland habitats an accepted approach involves blocking drainage channels installed to dry the land and removing invasive scrub. This enables natural re-colonisation of the area by peat-forming moss species. Effective grassland restoration on former agricultural land requires a more active approach. Taking the land out of agricultural production is seldom sufficient. Sowing grassland seed mixes is needed, but the effectiveness of seeding can be affected by often high residual soil fertility, so time may be needed to allow fertility levels to reduce. Canalised river and riparian systems can be rehabilitated by removing levees to re-establish a more natural flood plain and flooding regime, slowing the flow of water and re-creating a more natural regime of water regulation. Active restoration of degraded, deforested and monocultural woodland (i.e. by active replanting a mix of characteristic native tree and shrub species) is more effective than passive restoration, which is a slow and constrained process, but full restoration is rarely possible. GI restoration and creation programmes and projects: These can be used to restore GI elements and create connected networks aimed at securing biodiversity benefits, ecosystem services and coherent and resilient ecosystems.

7.3 Scales for GI implementation

The nature of these different GI building blocks highlights the wide range of scales relevant for GI implementation from green roofs, parks and gardens in urban areas or road overpasses/underpasses (i.e. sites), to international river corridors or transboundary mountain ranges (i.e. landscape). GI planning and implementation therefore operate across a range of scales: site, region, landscape.

GI initiatives can therefore be developed by a variety of actors (e.g. government, businesses and nature conservancy organisations) at a wide range of scales. The focus of initiatives differs across scales. Transnational, national and regional ecological network initiatives have an emphasis on biodiversity and the importance of connectivity in increasing ecosystem resilience. Urban green space initiatives have a greater focus on the multifunctionality of green/blue open space. Urban green/blue space provides high amenity benefits alongside a range of other ecosystem services that have high value in densely-populated urban areas, such as health benefits associated with reducing air pollution and providing opportunities for people to exercise and be active outdoors. The value of these ecosystem service benefits can be very high in urban areas because many citizens can benefit from them and the need for open areas is greater when people are living at high (and increasing) population densities. Urban GI also provides direct and indirect economic development benefits and opportunities.

Different initiatives at different scales are intended to be mutually supportive, so that individual initiatives can be part of strategic and integrated solutions with wider than local (*in situ*) impacts. Wider ‘systems-level’ impacts relate, especially, to habitat and biodiversity benefits and ecosystem resilience.

At European level, there is already a network of Natura 200 sites. Within the context of the EU GI Strategy, there is an explicit goal of improving connectivity between sites in the Natura 2000 network.

7.4 Green Infrastructure and Ecosystem Services

GI planning and implementation are based on identifying and assessing GI functions and benefits, how these are likely to be affected by socio-economic development pressure and what responses are needed. Table 7.1 sets out ecosystem services that GI can provide and that might be impacted by GI initiatives. In turn, these are related to different kinds of benefits through impacts on: food and food security; natural resources; water management; climate and climate change; recreation, health and wellbeing; education, culture and communities; employment and investment; land and property values; and protection and conservation of species and habitats.

Table 7.1 Ecosystem Services of Green Infrastructure

Provisioning services	Regulating services
• Food • Water • Raw materials (timber, fibre, rubber, etc.) • Genetic resources • Medicinal resources • Ornamental resources	• Air quality regulation • Climate regulation • Protection against extreme events and moderation of impacts of storms, floods, etc. • Regulation of water flows • Water purification and waste water treatment • Erosion prevention • Soil maintenance • Pollination • Biological control (e.g. control of pests and diseases) • Noise regulation
Cultural services	**Support services**
• Landscape and amenity • Recreation and tourism • Inspiration and focus for art, education, science	• Primary production • Nutrient cycling • Habitat provision • Maintenance of genetic diversity

7.5 Relation to other implementation initiatives

The GI concept refers both to networks of natural capital and to strategies for the development of these. It also offers a framework for understanding how different schemes of implementation can fit together synergistically. Although there is no ‘one-on-one’ correspondence, some schemes of implementation have higher relevance than others in conserving, restoring and managing specific elements of GI (Table 7.2)

Table 7.2 Implementation Schemes within a GI Framework

	Core Zones	Restoration Zones	Sustainable Use Zones	Urban Green/Blue Areas
PES Contracts	X	X	X	
User Charges	X	X		X
Offsetting	X	X	X	X
Standards			X	
Labels			X	
Reporting			X	
Procurement			X	
Nature Based Solutions		X		X

Protecting existing core zones of high conservation value is the most effective way to sustain habitat and biodiversity. Spatial planning, regulation and development control are mainstays of implementation. To safeguard high value habits and biodiversity hot spots from damaging modes of exploitation conservation of core zones under private ownership can also be incentivised using PES contracts (largely publicly funded, but with scope for some private and/or hybrid financing arrangements) and user charges for access by private individuals to zones offering high amenity benefits. Use can be made of habitat banking and offsetting arrangements to cover some of the costs of maintaining and extending core zones.

The cost-effectiveness of restoring natural capital and ecosystem services is a function of their type and current status. Restoring degraded zones is never as effective as is protecting core zones, since full restoration to undamaged status is seldom possible. Restoration of degraded zones is nevertheless an important complement to protecting existing core zones and an essential part of overall GI strategy. Restoration can be incentivized using PES contracts, supplemented by user charges when possible, and funded by creating habitat banks and offsets. Some restoration projects can be implemented as nature-based-solutions, offering cost-effective alternatives to ‘grey’ infrastructure; e.g. in coastal protection by using restored salt marshes, dunes, or mangroves and in flood risk mitigation by restoring upland peat habitats and returning rivers to their natural courses.

For sustainable use zones, shifts to more sustainable modes of exploitation can be incentivized using PES contracts (e.g. in agriculture by replacing agricultural subsidies with stewardship schemes) and combinations of PES/Offsetting (e.g. in forestry by the production and sale of carbon offsets). The development and take-up of sustainable practice standards can be incentivised and driven by markets, but increasingly is driven also by businesses voluntarily shifting to more sustainable practices. Consumer-facing labelling schemes can drive uptake of standards and certification schemes, but inherent limits restrict what they can accomplish alone. Business-to-business schemes (supply chain management, private procurement policies) offer fuller scope and can be supplemented by public procurement policies. Business risk mitigation is increasingly important in driving the take-up of standards and certification schemes through pressure downstream businesses are able to exert on upstream suppliers, but also by supports offered to them. In turn, schemes for reporting and disclosure of information on business sustainability (which is salient for investors, shareholders, business partners and clients) re-enforces the drive toward sustainable supply chain management.

Urban GI is especially important for its multifunctionality and because its many benefits can be accessible to many people, including high priority groups, such as poorer people, younger people and the elderly. Nature-based solutions are especially important in urban areas in contributing to urban GI, because they can provide effective multifunctional alternatives to grey infrastructure solutions and can contribute to a wide range of social and economic policy and development goals, as well as offer some habitat and biodiversity benefits.

7.6 Integration of GI into policy sectors

Significant progress has been made to integrate GI into relevant policy fields at EU level, including: climate, water, agriculture, nature conservation and regional policy. Thematic strategies (e.g. on the Urban Environment) and Directives (e.g. the Floods Directive) include recommendations or requirements to consider GI approaches and the EU GI Strategy supports this urging Member States to ensure that national planning policy gives regional and local authorities clear guidance and direction on how to plan and manage GI. CAP reform includes instruments to help implement GI strategies and cohesion policy co-finances investment in GI. GI policy and implementation intersects with a wide range of policy goals relating to sustainable ecosystem management, human health and wellbeing, the green economy and economic competitiveness. It intersects also with developments in governance, including shifts to more participatory forms of governance and decision making in urban areas, such as experiments with participatory budgeting.

An example of good practice guidance is that provided in the UK by the Town & Country Planning Association (TPA, 2012). This includes a set of (elaborated) planning principles for GI and biodiversity: GI needs to be strategically planned to provide a comprehensive and integrated network; GI requires wide partnership buy-in; GI needs to be planned using sound evidence; GI needs to demonstrate multifunctionality; GI creation and maintenance need to be properly resourced; GI needs to be central to development design and must reflect and enhance the locally distinctive character of the area; GI should contribute to biodiversity gain by safeguarding, enhancing, restoring and creating wildlife habitat and by integrating biodiversity into the built environment; GI should achieve physical and functional connectivity between sites at strategic and local levels; GI needs to include accessible spaces and facilitate physically active travel; and, GI needs to be integrated with other policy initiatives. The TPA guidance stresses the role of measurable standards for GI in policy development and gives example standards for the principle of accessible GI proposed by Natural England and the Woodland Trust respectively. These are based on maximum distances of people to green spaces or woodlands of specific minimum sizes and on the area of green space per 1000 people.

The TPA guidance stresses the need for funding in delivering and maintaining GI and that good practice involves identifying funding sources for creating, managing and monitoring GI within the Local Plan and any Community Infrastructure Levy (CIL) charging scheme. The guidance proposes that in new developments, new GI assets can be secured from the landowners' 'land value uplift' and as part of development agreements. The local planning authority is advised to include capital for GI purchase, design, planning, maintenance and management within its CIL schedule. To establish significant landscape features and corridors, other funding mechanisms are needed. Examples given include the Heritage Lottery Fund, Higher Level Stewardship and INTERREG European funding. To support management costs, the guidance suggests that some income can also be generated from GI, such as from woodlands managed for renewable energy resources and sustainable local food production.

7.7 Tools and instruments for GI implementation

A diverse range of tools and instruments of different type (statutory instruments, maps and information instruments, spatial planning instruments, scenario and assessment tools, economic instruments, financial instruments, governance instruments, communications instruments) can be relevant for GI planning and implementation.

As GI is a spatial concept, spatially-explicit assessment of ecosystem services (ES mapping) based on multifunctionality and connectivity is fundamental for the strategic identification and planning of GI networks and elements and is possible across the spectrum of spatial scales and from different entry points; e.g. from the entry point of delivering ES multifunctionality and identifying and differentiating areas with highest capacities to deliver wanted ES and/or from the entry point of delivering biodiversity conservation and functional connectivity and identifying and differentiating core habitats and wildlife corridors. Such assessments are used to select and distinguish between high performing core (conservation) areas and more moderately performing (restoration) areas.

Implementations are typically developed within higher-level policy frameworks and cascades. They make use of different instrument combinations depending on implementation context and goals.

- *High-level strategies and action plans:* High-level strategies express political commitment for the need to identify, preserve and invest in GI, establish objectives and targets, set out guidance and principles for taking GI into account in policy- and decision- making across policy areas, governance levels and in spatial planning, and establish priorities for action. These are generally translated into more detailed action plans outlining measures to be taken in support of GI in specific sectors and geographies.

- *Maps and GIS:* As an element of spatial planning at different geographic scales, maps and GIS are used to identify, locate and designate GI elements that are to be protected, restored or enhanced as well as where investment in newly-created GI and connectivity features will support network and ecosystem cohesion and connectivity and the delivery of ecosystem services.
- *Indicators:* Biodiversity and ecosystem service indicators are used to establish base line ecosystem health status (GI elements, quantity and quality), to set targets, to help in designing implementations (i.e. to select appropriate tools), to measure and monitor impacts of GI interventions and to improve understanding of linkages between GI interventions, the status of GI elements and ecosystem services.
- *Scenarios:* Scenarios can be used to explore different GI network configuration options.
- *Stakeholder processes:* Stakeholder processes are important for establishing how ecosystems are used and the value of benefits to beneficiaries. They are important also for empowering citizens and ensuring a better reflection of GI values in decision processes.
- *Valuation tools:* To integrate green infrastructure benefits into decision making, benefits need to be identified, quantified and valued to support the development of implementations. Valuation is used to establish the importance of GI, to justify interventions and help with their design, to justify investment, and to help identify appropriate investment sources and mechanisms (e.g. public investment, offsets, PES). Access to GI benefits by beneficiaries and the number and priority status of beneficiaries are important variables. Urban GI can be very important because it offers multiple benefits to potentially large numbers of people, many of whom can be in high priority target groups (poorer people, young people, older people, etc.).
- *Spatial planning instruments:* To secure delivery of biodiversity and ecosystem service benefits, the decisions of political authorities at different levels concerning preserving or enhancing GI need to be integrated into spatial plans and land use regulations used at each level to steer and control development. Vertical and horizontal coherence between spatial plans is needed for proper consideration of GI coherence and connectivity.
- *Impact assessment instruments:* Impact assessment instruments can be used to enable GI impacts of proposed developments (EIA) and plans or programmes (SEA) to be assessed prior to authorisation and as a basis for applying the mitigation hierarchy to proposals.
- *Legal standards and regulations:* These include: building standards and regulations that specify design requirements to be fulfilled at the scale of sites and buildings (e.g. requirement and specifications for green rooves).
- *Liability laws:* Liability laws are used to impose requirements on those whose actions inadvertently or unavoidably damage GI to make good or to offset the damage for which they are responsible. Offsetting requirements provide incentives to direct

development projects away from high-value GI and toward areas where development impacts on ecosystem services would be limited. The requirement to compensate for residual damage creates a source and mechanism to finance GI initiatives.

- Market instruments: Payment for Ecosystem Service (PES) approaches, whether in the form PES contracts, agri-environment schemes, or simpler arrangements, such as user charges, are used to incentivise and reward the delivery of a wider range of ecosystem services. Beneficiaries of ecosystem services (or public agencies on behalf of citizens in respect to public benefits) pay landowners or land managers to deliver wanted ecosystem services. Many GI benefits are public goods, but others are private goods, raising the possibility also of attracting private investment into GI initiatives and giving scope for joint (public-private) project financing.
- *Voluntary standards, certification and procurement:* Voluntary standards for land management (e.g. organic agriculture, sustainable forestry) are used to support sustainable exploitation regimes consistent with balanced delivery of provisioning services and other ecosystem services (regulating and support services). Ecosystem service indicators and requirements for set-asides, corridors and other elements that favour biodiversity can be integrated into standards. Private and public procurement policies can require that suppliers meet standards; i.e. that their production sites and production protocols comply with recognised standards.
- *Voluntary land management agreements:* Agreements negotiated between landowners and leaseholders are used to ensure that land is managed sustainably over the term of the lease. ES/NC concepts. Ecosystem service indicators can be specified to enable agreements to be monitored.
- *Investment sources and instruments:* Public investment can be used to secure public benefits of GI and supports most creation projects. Public finance reform and restructuring can make funds available for public investment in GI. Public-private financing mechanisms under development can help leverage the effectiveness of public investment. Offsetting schemes can deliver funds to cover some costs of GI conservation and enhancement.
- *Long-term scheme-financing mechanisms and instruments*: Long-term finance is needed to maintain GI. In addition to public funding and in-kind contributions of public authorities (such as employing GI managers), PES schemes offer ways for public and private beneficiaries of ecosystem services to contribute to maintenance of services; e.g. agro-environment schemes, user charges for parks and urban green spaces.
- *Governance arrangements:* Elements relating to governance include: creating new institutions or expanding the mission of existing institutions, especially to enable these to support and fund nature-based solutions as alternatives to grey infrastructure; involving stakeholders in decision-making processes with implications for green infrastructure to ensure that the full range of ecosystem service benefits, the value of benefits, and access to benefits are included in decisions; more direct stakeholder participation in project programming and funding (e.g., through

participative democracy and participatory budgeting schemes in urban settings); and implementing reporting requirements relating to natural capital, progress of GI initiatives and the contribution of initiatives to policy objectives.

- *Awareness and capacity building*: The range of measures relevant for stimulating the take up of GI initiatives include: information campaigns targeting policy makers and wider publics and aimed at raising awareness of GI approaches and the ecosystem service benefits and policy relevance of GI; guidance targeting the different actors and stakeholders (e.g. local and regional planners, land managers, potential investors in nature-based solutions); and capacity building measures to overcome perceptual and technical obstacles to promoting GI in delivering ecosystem services and supporting policy goals.

7.8 Marine Spatial Planning: an illustrative case

7.8.1 Background

Marine Spatial Planning is an emerging field of spatial planning. Its emergence offers opportunity for the NC/ES concepts to be integrated into marine spatial planning from the beginning across the full spectrum of marine spatial planning supports: data collection, tool development, information generation, decision support, plan development and plan implementation. We take the Marine Conservation Zone Project (MCZP) for England and Wales as an illustrative case and focus on the contribution to it of a regional-scale project, Finding Sanctuary (FS).

The key driver for MCZP is a set of policy references cascading from the Marine Strategy Framework Directive (MSFD). The concern of the MSFD is to establish and maintain "*good environmental status of marine and coastal ecosystems"*. This requires that decisions are taken concerning whether protection and conservation are needed, at what levels and in what forms. Good environmental status is defined as securing continuity in the supply of marine ecosystem services. Related responses in England and Wales include the Marine and Coastal Access Act (2009) and the Marine Bill (2012). Together, these provide a basis for establishing a network of Marine Protected Areas (MPA), which include Marine Conservation Zones (MCZ).

The Marine and Coastal Access Act (12 November 2009) created a new type of Marine Protected Area (MPA), called a Marine Conservation Zone (MCZ), to protect nationally-important marine wildlife, habitats, geology and geomorphology. MCZ are a new category of protected areas intended to help protect 'usual' ecosystem features rather than only exceptional features and to help provide connectivity by linking otherwise fragmented high-value protected areas. To maintain overall ecological viability of the system (resilience) in the face of global change and human pressure, sites were to be selected to protect not just what is rare and threatened but to protect the full range of marine wildlife and habitats.

To identify and recommend Marine Conservation Zones to Government, Defra, Natural England (NE) and the Joint Nature Conservation Committee (JNCC) established the Marine Conservation Zone Project (MCZP). The MCZP concerns the selection and delineation of boundaries of MCZ in English inshore waters and offshore waters next to England, Wales and Northern Ireland. MCZP was the umbrella project for four regional projects, each responsible for making recommendations for MCZ boundaries for a particular region of the UK. The four regional projects cover the South-West (Finding Sanctuary), Irish Sea (Irish Sea Conservation Zones), North Sea (Net Gain) and Eastern Channel (Balanced Seas).

MCZ, together with other types of MPA, were intended to deliver an ecologically-coherent MPA network; i.e. *“a collection of areas that work together to provide more benefits than an individual area could do on its own”*. MCZ were to be designated as part of an expanded network of marine protected areas (MPA). The existing MPA include other classes of conservation area; e.g. SSSI, RAMSAR sites and European Marine Sites. To ensure that the set of regional MCZ contribute to achieving an ecologically-coherent national MPA network, the regional projects were provided with national network design guidance.

There are key differences between the pre-existing conservation areas and the MCZ that were to be established through the MCZP:

- Whereas these other conservation areas have been defined to protect endangered species or to conserve valuable and threatened habitats, MCZ are intended to protect a broad range of regular habitats and species.
- Whereas the process that leads to the designation of Special Protection Areas and Special Conservation Areas under European legislation (the European Habitat Directive) is based solely on scientific criteria and has a largely ecological focus, the Marine Bill allows for decisions about MCZ boundaries to be based also on socio-economic considerations, as long as these do not undermine the creation of the network.
- Whereas other conservation areas are taken out of use entirely, MCZ can have a range of different management measures, not just ‘no-take’ zones. Management measures can include seasonal restrictions and can provide for differentiation across different uses.
- The management measures required within MCZ were to be decided on a site-by-site basis and were to depend on what the site has been designated for; i.e., not all sites would have the same management measures.
- There is provision under the Marine Act for the zone boundaries and management rules to change. This provides scope for future adaptive management.

MCZ were therefore intended as sustainable use zones within Marine and Coastal Ecosystems as part of the overall green/blue infrastructure. The provision to take socio-economic factors into account in their delineation was intended to “ensure that a network of sites can be achieved in a way that *minimises adverse impacts on sea users and maximises benefits for nature conservation*” (Joint Nature Conservation Committee: The Marine Conservation Zone Project, Version 2.1, January 2010).

Implementation of the MCZP followed an ‘ecosystems approach’, but with different levels in the governance hierarchy responsible for different component tasks based on a subsidiarity principle.

- National targets for overall area of sea to be designated as MCZ, ecosystem types to be included in the area and contiguity with existing MPA (natural capital to be conserved) were set by Joint Nature Conservation Committee (JNCC), Natural England (NE) and Defra at national level. The national level bodies were responsible also for final reconciliation of MCZ boundary recommendations from the four regional projects.
- Responsibility for making recommendations to JNCC and NE for MCZ boundaries was delegated to the four regional projects. This was to provide for boundary recommendations to emerge from *local/regional stakeholder processes* involving collection of data on the uses made of marine and coastal ecosystems (i.e. the ecosystem services provided) and dialog processes among stakeholders aimed at revealing the socio-economic costs of establishing and maintaining MCZ. Stakeholder dialog and negotiation was intended to provide for these costs to be taken into consideration in making boundary recommendations.
- The main issues for regional stakeholders to decide included: finding and agreeing on boundaries for MCZ that lower the economic and social costs of meeting conservation targets to stakeholders as a whole, negotiating and agreeing potential ‘loss sharing’ mechanisms, and pre-empting and proposing management measures to address possible behavioural adjustments to MCZ (e.g. sea users relocating their activities).

7.8.2 Choice of case study:

Finding Sanctuary (FS) was the regional MCZ project for the South-West region of the UK, covering the coastline of the south-west peninsula and inshore and offshore waters. Its mandate was to work with sea users and interest groups within the South-West region of the UK to make recommendations to government about sites, boundaries and management rules for MCZ within regional waters to be designated under the UK’s Marine and Coastal Access Act.

- Location (FS): Southwest England (coastal and marine ecosystems of Dorset, Somerset, Devon, Cornwall and the Isles of Scilly.
- Scale: Regional (92,000 km^2)
- Goals: Designating Marine Conservation Zone sites and boundaries that minimise socio-economic costs of conservation and that command support of stakeholders
- Project design phase: 2009-2011
- Project cost: ca £1 million

The MCZP and FS constitute important case studies because of the novel implementation design architecture they reflect (multilevel governance) and the central role played by stakeholder processes. The case is important from the perspective of determinant analysis (success factors, barriers, contextual determinants, etc.) and learning lessons about strengths and weaknesses of the design and implementation choices made.

The implementation design is influenced by features of the applications context. An important aspect of context was the lack of any existing comprehensive database of the uses made of the marine and coastal ecosystem. This had to be created within the project from stakeholder input by mapping stakeholders' uses of the marine and coastal ecosystem both in space (which elements of the ecosystem are used where and for which purposes) and time (use patterns across the annual cycle). This project-developed mapping was needed as a support for informed negotiation and decision-making about site, boundary and management rule recommendations.

Another important contextual factor is that marine ecosystems and activities are less easily monitored and policed than activities on land. A specific intent of involving stakeholders in delineating MCZ boundaries was to use this as an opportunity to raise stakeholders' awareness of the need for conservation, to raise awareness of each other's uses of the sea and coast, and to enlist stakeholders in efforts to monitor and enforce compliance. This last is important because stakeholders, as users of marine ecosystem services, have physical presence in and around protected areas and can oversee each other's compliance with regulations more cost-effectively than if enforcement is in the hands only of external authorities.

A wide range of stakeholder representatives were given a central role in designing the recommendations. During the formal phase of the project, 41 representatives of stakeholders from a wide range of marine sectors formed a Steering Group (SG). These included representatives of commercial, recreational and conservation interests. The role of the SG was to develop network recommendations, with support from the project team (PT).

The stakeholder process led to boundary recommendations being made from the regional projects to the national level, but to stakeholder disappointment and disillusion when these were not adopted in their entirety and integrity at national level. The process has been criticised on grounds that there were delays in the information flow from national to regional level and that the terms of reference were changed during the process, undermining the understandings and expectations of stakeholders about the purposes of MCZ, the stakeholder exercise and their roles. An important lesson is that when stakeholder buy-in is critical for scheme implementation, it is important to provide clear terms of reference and to maintain stakeholders' trust and faith in the process. The stakeholder process must retain integrity throughout. It risks otherwise becoming counterproductive.

7.8.3 Process and governance:

At the national level, background studies were commissioned by Defra to examine the case for conservation as a valid use of marine ecosystems. A study was undertaken of the potential benefits of establishing a national MPA-MCZ network based on scenarios concerning possible sites, boundaries and management rules, different assumptions about the rates and levels of recovery arising from conservation status, and the *in situ* and *non-site* benefits recovery might bring. The studies made a broad and provisional (first-cut) national level analysis of synergies and trade-offs among ecosystem service supply and made first-cut estimates of the net benefits that might come from different levels of successful conservation efforts. Additional studies were made of some specific high-value ecosystem services, such as the use of marine ecosystems by fin and non-fin fisheries. The fisheries studies generated spatially-distributed data of fishing values to provide a more spatially-explicit representation of fishing benefits; i.e. per square kilometre.

Findings from the background studies were used to establish broad, national-level criteria for the design of an MPA-MCZ network in terms of overall percentage of the marine ecosystem to protect, what kinds of habitat to protect, and requirements for ensuring habitat connectivity. These findings informed the national-level guidance given to regional projects, which the regional projects were to respect when making recommendations over the precise location and boundaries of the MCZ.

Implementation of the project was subject to procedures specified by Defra and the Joint Nature Conservation Council. These specify that:

- Each regional project should have a project manager with responsibilities for organising stakeholder engagement processes and providing technical support to stakeholders covering data collection, processing, representation and use
- Each regional project was to: gather data on the marine environment; gather data about what areas of sea are important to people and why; i.e. the uses (and patterns of use) made of the sea, the ecosystem services the sea provides; and create a stakeholder group that will use the collected data to recommend sites, boundaries and management rules for MCZ.
- In each region, sea users and interest groups were to be invited to apply to represent their sector on a stakeholder group.
- The stakeholder groups would have responsibility for developing recommendations on the location and proposed conservation objectives of the MCZ.
- Representatives from a UK-level stakeholder forum could be delegated to the regional stakeholder groups to ensure national and international interests are reflected in deliberations and decisions.
- The Regional Project team and an independent facilitator would ensure that all stakeholder group members have equal opportunity to voice their views during the process and that deadlines are met.
- The Regional Project team would be responsible for supporting stakeholder group representatives in communicating with their sectors and making sure individual stakeholders receive feedback on SG meetings and are kept up-to-date with project developments.
- Decisions (boundary recommendations) would be delivered through a process of professionally-facilitated stakeholder negotiation

- The Regional Projects would be provided with national network guidance for the MPA network. The MCZ recommendations that the stakeholder groups make would have to take this guidance into account to ensure the regional contribution to an ecologically-coherent MPA network.
- To help achieve this, stakeholder groups could submit draft recommendations to an independent Scientific Advisory Panel that would provide feedback on whether they meet the guidance.
- Natural England and the JNCC would advise Government on whether the network meets the national criteria and bring this advice to the stakeholder group during the planning stages.
- Each Regional Project was to prepare an ***Impact Assessment*** setting out the anticipated costs and benefits of the proposed network of sites and identifying the environmental, social and economic implications.
- The regional stakeholder groups were to submit final recommendations and their Impact Assessments to Natural England and JNCC. These would be compiled and combined into one document without changing them.
- NE and JNCC would formally submit to Defra:
    - A single document containing the unchanged recommendations of the four regional stakeholder groups.
    - Advice as to whether the recommendations are considered to be sufficient to meet the MCZ contribution toward an ecologically-coherent network of MPAs
    - Advice on any changes considered necessary
    - The advice of the Science Advisory Panel.
- Based on the recommendations and advice, Ministers were to decide what sites should be subject to a formal public consultation process.
- After considering public responses, MCZ were to be designated by Ministers.

7.8.4 Phases/stages in the implementation process:

The stakeholder processes for all four regional projects were iterative, with three rounds of stakeholder negotiation. At the end of each round, the regional projects submitted interim reports to the national Science Advisory Panel (SAP). The SAP provided scientific review of interim recommendations and feedback, which informed the next round.

7.8.5 Stakeholder participation

All major users of the marine and coastal ecosystem were represented: commercial fishing, shipping, military, heritage, leisure and tourism, sea-bed owners, companies with interests in offshore renewable energy, statutory bodies, scientists, etc. The project Steering Group (SG) charged with proposing boundary recommendations comprised over 40 stakeholders representing each of these major users.

In order to provide for an efficient process, two working (sub) groups of stakeholders were established: an inshore working group and an offshore working group. The role of the working subgroups was to make initial proposals to put to the SG for consideration. The inshore working group handled issues that are potentially more complex and contentious owing to the relatively small geographical area involved and intense competition for access among large numbers of uses and users. The offshore working sub-group concerned a smaller number of parties, but with mostly bigger and more commercial interests, including commercial fishing and shipping. In addition, smaller stakeholder groups were established at the local (County) level to develop initial site suggestions based upon local knowledge and usage patterns.

Stakeholders were provided with guidance (principles and criteria) their recommended Marine Conservation Zones (rMCZ) and overall networks of MPA should meet. Other than for reference areas, stakeholders were not given information on how activities (e.g. renewable energy development, submarine cables, aggregate extraction, dumping and disposal, coastal activities, port activities and fishing) would be restricted in areas they designated as MCZ. Facing this uncertainty, stakeholders in the FS project developed a set of working assumptions (***scenarios***) to underpin their planning and negotiating. These were described in a stakeholder narrative submitted alongside their recommendations. Specifically, the fundamental working assumptions were that: current activities within a MCZ would be allowed to continue unless they prevent the conservation objective from being achieved (perhaps subject to upper limits on the intensity of use), but that high levels of restrictions would be placed on ongoing activities in reference areas and that mobile bottom-towed fishing gear would not be permitted in any MCZ. Habitats were designated in relation to in-situ species but taking account of intermittent presence of important mobile species, such as cetaceans.

The final project report states that: the lack of unambiguous guidance about the consequences (restrictions on activities) that might follow MCZ designation "*posed the single most significant obstacle to constructive discussions throughout the project duration*" (Lieberknecht et al, 2011, p. 50). "*Most participants… found it very difficult to be faced with the task of designing a network when they did not know what restrictions would be put in place, and how sites would impact themselves and others*".

7.8.8 Development and use of tools

Data needed to make informed choices about geographical boundaries are not collected routinely, and had to be collected within the project. This includes data about the uses made of marine resources. Finding Sanctuary developed an extensive evidence base covering both the biophysical characteristics of the area and the intensity and patterns of use of sea areas by different stakeholder groups. This involved developing a spatially and temporally explicit ***mapping*** of the supply of and demand for ecosystem services. Data collection was achieved using liaison officers to interview stakeholders about their uses of the marine ecosystem and online data submission by stakeholders directly. Sea-users logging information numbered above 100,000.

Information from multiple sources was layered using a ***GIS tool***. This allowed for the identification of activity hotspots and conflict zones (areas where it would be potentially contentious to site a MCZ) and, also, areas where there are no activities, few activities or non-conflicting activities. It was possible also to identify where the same sea areas might be used by different stakeholders at different times and for different purposes in ways such that no conflicts are involved in the actual pattern of use. This spatially- and temporally- explicit data base of the ecosystem and its pattern of exploitation is an important project outcome as a potential instrument in helping build stronger social networks and consensus positions among stakeholders.

Methods and protocols for valuing fisheries (the Fishermap protocol) were developed in the project. The value of other benefits to other stakeholders were to be established through stakeholder dialog and negotiation processes and reflected in the boundary recommendations and supporting impact assessments of their costs and benefits.

7.8.9 Strengths and weaknesses:

The implementation had many strengths, but credibility in the scheme was undermined by government hesitancy in implementing the rMCZ.

Important strengths of the process included that:

- The geographical boundary of the analysis was chosen to reflect the regional scale of the ecosystem and its modes of exploitation and to be inclusive of the key stakeholders, actors and agents, so that decision making might adequately reflect the interests and knowledge of local and regional stakeholders.
- Key network design principles were developed at national level and cascaded down to the regional projects. The Natural Capital concept was operationalised through these principles, which identified what was to be conserved, and through a ***Vulnerability Assessment***, which determined the status of natural capital and, in relation to its vulnerability, established conservation objectives. The network design principles included:
    - *Representativeness:* The requirement to represent a list of species, habitats, geological features, and broad-scale habitats [These were based on EUNIS level 3 habitats. Broad-scale habitat targets were set for subtidal and intertidal habitats.]
    - *Adequacy:* The requirement to capture certain minimum amounts of broad-scale habitats within the network
    - *Connectivity*: The requirement to have sites with similar habitats spaced closely enough together to allow movement of animals (including larvae) between protected areas
    - *Viability:* The requirement that individual sites meet minimum size guidelines.

  - *Replication:* The requirement to represent listed species, habitats and broad-scale habitats in several different sites within the network [These were based on a list of specific features and habitats designated as FOCI – Features of Conservation Importance]

- The requirement to produce ***Impact Assessments*** to accompany each set of boundary recommendations and to follow an iterative procedure supported stakeholders in discovering the implications of their remit in terms of the key issues to decide, the need to establish and agree a robust process for decision-making, and the need for appropriate tools and data to inform the deliberation and decision-making processes.

- The design of the governance aspect of the implementation was intended to ensure that a high level of 'steer' was exerted over the regional projects from the national level so that national natural capital conservation targets would be met. Final decision authority concerning the precise boundaries of constituent MCZ and management rules applying to these was retained at the national level, but the process was intended to delegate influence over boundary delineation to regional and local stakeholders. In principle, this approach to subsidiarity provided for taking decisions at the appropriate spatial scale while recognising the cumulative impacts of decisions.

- The reflection of the project team was that the stakeholder process resulted in a set of recommendations underpinned by a sense of collective ownership by a group of representatives from across a diverse spectrum of interests.
  - The final FS report states that while "not all stakeholder representatives necessarily support all aspects of the project's final recommendations,… there is a general view that the recommendations, if implemented as recommended, constitute a set of sites that most stakeholders involved in the process could support, live with, or (as a minimum) accept as less bad than it might have been had we not been involved in the process" (FS, Final Report, 2011, p.58).
  - The fishing industry representatives stated clearly that they do not support reference areas.
  - Overall, stakeholder representatives expressed moderate levels of satisfaction with the recommended network.
  - Those less satisfied representatives expressed as their perceived weaknesses of the process:
    - Lack of clarity over management measures
    - Lack of opportunity to review outcomes of the Vulnerability Assessment
    - Uncertainty about the process beyond making recommendations; i.e. how the recommendations will be used.

  - The *more satisfied* representatives expressed as their perceived strengths of the process:
    - The recommendations were as good as could be achieved within the process and its constraints
    - Stakeholders genuinely had an influence on the recommendations
    - The outcome outstripped expectations.
- The final rMCZ network was evaluated for conformity with guidance targets for overall area to be protected and its make-up: broad-scale habitats and habitat of conservation importance each against principles of representativeness, adequacy, replication and viability; FOCI habitats against replication and viability targets); species of conservation importance against representativeness, replication and adequacy targets; and geological and geomorphological features of importance against a criterion of these being taken into consideration. Network statistics were calculated using ***ESRI ArcGIS version 9.3.1 in ETRS89/LAEA*** and generated by intersecting the broad-scale habitat, FOCI and geological data layers with the overall network shape*.* ***Pivot tables*** were created showing those habitats that were represented within existing MPAs and those within the MCZ network. ***Maps*** were used to provide visual representation of network performance against the connectivity criterion.

The main weakness has been seen to be the subsequent reluctance of the UK Government to implement the rMCZ boundaries as a coherent network and, rather, to have proceeded to a phased and partial introduction of MCZ. This has involved postponing decisions over the more sensitive reference areas, which would have involved restricting fishing activities in these areas. A phased and partial introduction is seen to have risked losing stakeholders' trust in the process, since this is inconsistent with the original intent to create a coherent network of protected areas.

8. Implementation in the exemplars

8.1 Background

Context is important for establishing how best to operationalise an ecosystems approach and what added values this approach can offer. To explore how contextual analysis can help operationalise an ecosystems approach and contribute to its design, implementation, monitoring and governance, suggestions based on the templates, insights and guidance developed in the work-task were offered to two of the OPERAs exemplars: urban dunes (Barcelona) and seagrass meadows (Mallorca). This work contributes toward developing a prospective tool, CODIFIES, as a comprehensive determinant framework for implementing an ecosystem services approach.

8.2 Barcelona coastal management

An exemplar was established at the start of the OPERAs project focused on the ecosystem services of dune ecosystems near Barcelona. The natural dune ecosystem has been degraded largely by anthropogenic disruption of the natural longshore drift sand transport processes caused by the expansion of the Port of Barcelona. Natural dune formation no longer takes place and attempts to construct artificial dunes have failed. The exemplar was therefore conceived initially as a Nature-Based Solution to test the possibility of creating hybrid dunes by replicating the generative processes of incipient (early-stage) dynamic dune formation and to explore ways of financing experiments and projects of hybrid dunes creation by selling dune ecosystem services, such as storm and flood protection.

Contextual analysis using CODIFIES involved exploring the present regime of management interventions, its basis and its cost-effectiveness. This led to the exemplar being re-framed and its scope broadened to cover wider issues of sand management along the full stretch of coast impacted by anthropogenic interference in natural sand dynamics. The reframed exemplar confirms warnings that current interventions are not solving the problem of beach and dune erosion and may be extending the problem westward from the main focus of current attention, El Prat de Llogrebat, to Ginesta.

The re-framed exemplar has made policy recommendations to widen the set of actors and stakeholders involved in beach (de facto coastal ecosystem) governance, change the current management goals and indicators, and experiment with Nature-Based interventions. It is proposed these are financed by cost savings made by down-scaling the current programme of artificial sand recycling and replenishment, which incurs high recurring annual costs, but is ineffective.

Historically, (pre-1970s) a natural, active dune ecosystem provided storm protection and contributed to an advancing coastline. Owing to anthropogenic interference since then, there has been no natural formation of active (dynamic) dunes. The dune and beach ecosystem near Barcelona is severely degraded and is no longer regenerating naturally. An advancing coastline (net sand accumulation) has become a receding coastline (net sand erosion) in many places.

This should be a source of growing concern as sea-level rise and expected increase in the frequency and severity of storms present increasing risks of storm and flood damage to high-value property and transport infrastructures along the coastline. However, there is poor awareness of the risk to ecosystem services that the degraded ecosystem status entails, even among the major beneficiaries of those services, such as property owners and beach users.

The problems of dune degradation and loss of capacity for natural dune regeneration are symptoms of a broader problem. This has at least three elements. One is anthropogenic interference in the natural sedimentation processes of the dynamic coastal environment. Another is the management response to these interferences which, rather than working with and helping restore natural processes, has focused on artificially recycling sand and artificially constructing the coastal features (dunes, beaches, lagoons) that natural processes would have produced. The third element concerns the governance of the issues. Currently, only a small number of actors and stakeholders are involved in decisions about sand management and these are not necessarily those with relevant expertise or interests in outcomes.

Anthropogenic disturbance began with the creation of a network of maritime footpaths to increase access to the littoral by locals. This led to some dunes being stabilised using artificial materials and by planting vegetation to fix and stabilise dunes. Much heavier use is now made of the beaches and dunes by locals for sunbathing during the summer months, altering sand moisture and sand exposure levels and preventing dune-forming winds from transporting sand across and up the beach and constructing incipient dynamic dunes. Longshore sediment supply was reduced by the development of Barcelona harbour, which involved constructing a 1.74 km dyke perpendicular to the coastline and relocating the mouth of the Llobregat estuary to the western side of the new port. These developments interfered with longshore drift, sediment replenishment and natural beach and dune building processes.

Artificial dunes have been constructed to compensate for the loss of natural dunes. Artificial beaches have also been created at El Prat de Llogrebat to compensate for beaches lost through the harbour development. An artificial lagoon was created alongside the realigned estuary. However, these artificial replicates are not products of natural forces and dynamic processes and they are not dynamically adaptive to the ever-changing conditions around them. They are static and rigid features vulnerable to sea and wind (and human) damage. This response therefore depends also on building defences to protect these artificial features, such as stone barriers that separate artificial beaches from the sea. This approach generates a vicious circle of interference. It also precludes that the artificial structures can provide the range of ecosystem services that the natural dune, beach, and lagoon ecosystems provided, such as storm protection. It also prevents interchange between the different elements, such as through channels between rain-fed lagoons and the sea, which would form naturally under natural conditions to create important brackish-water habitats and access to these.

The artificial coastal landscape also requires constant human intervention to maintain its structures. The main intervention has been to establish a continuous programme of sand replenishment in an attempt to compensate for reduced longshore drift. This involves dredging sand from the seabed in the vicinity of Ginesta and dumping it off El Prat de Logrebat. This incurs high levels of recurring expenditure (ca. 1 million € annually). However, this programme is not effective. Evidence-based analysis within the exemplar of long-term sediment, beach, and dune dynamics confirms that the replenishment program is not solving the problem of erosion in the area of Prad de Llogrebat and is changing the morphology of the beach in the area of Estany de la Roberta. The absence of recovery of the dredging ditches also increases the storm and flood risk the Port of Ginesta and the stability of its dyke.

The history of the current management regime shows that a narrow range of actors and stakeholders was involved in developing the present approach and that each has a specific role, responsibility and specialism. The current governance arrangement was established following an environmental impact assessment of the Port expansion and the re-alignment of the Llogrebat estuary. The Environmental Impact Declaration prescribed the creation of a Joint Committee of Monitoring and Environmental Control (CMSCA), with responsibilities to monitor the identified set of environmental issues. Members of the CMSCA are the Port Authority, the Coastal Administration of the Central Government, departments of the Catalan Government, and one Municipality (El Prat de Llogrebat), because this was the municipality most directly impacted by port expansion.

The port authority is responsible for financing sediment recycling as a response to interference in the natural sedimentation and longshore drift processes. However, the port authority is not a competent authority for coastal ecosystem management. Also the costs of the intervention programme are relatively small in the context of overall port operations. The responsibility of the post authority is limited to financing interventions and verifying these have taken place, but it does not extend to ensuring these are ecologically effective.

Our assessment of the current management regime highlights that problems have arisen by not looking holistically at the coastal ecosystem and instead taking a partial view, seeing the features of the landscape as separate from the dynamic natural forces that create them. Interventions have focused on addressing the most evident and immediate symptoms of interference and on locations where symptoms and impacts are manifest. Official monitoring of the coastal defence works does not cover the entire system, but wider evidence shows that interventions are not showing desired outcomes. However, those most affected by adverse impacts are not included in the CMSCA.

Our policy recommendations are therefore to: expand the group of actors and stakeholders involved in coastal defence governance; establish management criteria and indicators for the full stretch of impacted coastline; focus specifically on the emerged sand budget; reduce expenditure on the ineffective sand replenishment programme; divert some of the saved funds to finance experiments with Nature-Based solutions that work with natural processes to construct a hybrid and resilient beach ecosystem.

8.3 Mallorca Seagrass

In the Mallorca seagrass (Posidonia) exemplar, it was suggested that an implementation scheme based on *user charges* would be most appropriate to mitigate stresses on seagrass arising from pressures linked to tourist activity. Seagrass beds sequester and store carbon, and are important in climate regulation, but this ecosystem service can only be secured and valorised *if* the long-term health of seagrasses is secured. Currently, the Mallorca seagrass beds suffer tourist-related stress from high levels of discharge of sewage effluent and from direct physical damage by the (illegal) dragging and dredging impact of pleasure boat anchors and chains. But the seagrasses are multifunctional. In addition to carbon regulation they provide other regulating ecosystem services: they remove sediments from sea water, protect beaches from storm damage, and provide critical habitat to support marine biodiversity. These are important in maintaining the high quality environmental features that attract tourists to Mallorca in the first place: sandy beaches and clean and clear bathing water.

Since the damage to seagrasses is linked directly to the additional stresses of tourists generally (effluent discharges during high season) and some recreational boat users specifically (illegal use of anchors over seagrass beds) and since these immediate causes of damage can both be addressed by known technical solutions (increases in sewage treatment capacity and the installation of permanent floating mooring buoys), an appropriate approach is to impose *user charges* for access to the tourist benefits of Mallorca's marine and coastal ecosystem and to dedicate part of the revenues to cover investment and operating costs of the needed equipment.

Estimates (e.g. Aguilo *et al*) of own price elasticity of Balearic tourism demand from the major source countries, such as Germany (0.84), the UK (0.98) and the Netherlands (0.51), indicate that tourist demand is relatively inelastic to price and, therefore, that a tourist tax will increase total tourist revenue. Part of the receipts can be hypothecated to underwrite and amortise capital investment in enhancing sewage and waste water treatment capacities to enable these to address high season demands (heavily augmented by the seasonal tourist population) that are otherwise uneconomic and unaffordable to address by the (much lower) resident population. As Mallorca is an island and tourist access to ecosystem benefits depends on tourists entering through a limited number of airports and ports and their staying in hotels and other visitor accommodations, a general tourist tax is simple to administer by levying taxes on tourists based on length of stay. As the marginal environmental damage cost of tourism is a function of the overall number of tourists on the island at any given time and this varies across the year, some fine-tuning of any tax is warranted between high and low season.[23]

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[23] During the OPERAs project, the Balearic government has approved a tourist tax. This has been implemented since July 2016. The tax level depends on accommodation type, ranging from €0.50 for campsites and hostels to €2 for five-star hotels. Children under the age of 16 are exempt. The tax is reduced to half rate from the eighth day on the island and during low season (November to April). A committee comprised of representatives of the tourist industry, environmental groups, the government, and trade unions has been established to decide how tourist tax revenues are used. The eligible fields include: the construction of new infrastructures for sustainable tourism; the protection

Since the direct physical damage to Posidonia is directly due to illegal mooring over Posidonia beds using anchors, the cost of installing mooring buoys and of policing/enforcing their use can be passed onto those using recreational boats. The problem is linked mostly to casual users of boats, rather than to experienced yachtsmen and could be addressed by providing information at boat hire stations to explain the rationale for the existing regulations that require using floating mooring buoys, by levying mooring charges as part of boat hiring fees, and by backing this with improved policing and enforcement. The user charges should reflect the actual cost of installing and maintaining a network of floating mooring points. Fines for illegal use of anchors over Posidonia beds should cover the costs of policing and enforcement. Work within the exemplar revealed sensitivities and differences of perspective among stakeholders over boating freedoms. Stakeholder processes could be established to run alongside trials aimed at raising standards using an evidence-based, adaptive management approach.

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and preservation of the environment; the conservation and restoration of historical and cultural heritage; and, research and technological innovation.

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