MONETARY VALUATION FOR ECOSYSTEM ACCOUNTING

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Executive Summary

Techniques for the monetary valuation of environmental goods and services are generally regarded as well established and robust and in recent years have increasingly been applied to ecosystem services. There now a vast case study literature applying these techniques to valuing ecosystem services, including at the national level with the recent UK National Ecosystem Assessment.

By monetising changes in the flows of benefit from an underlying stock of natural capital, valuation could also support a fuller inclusion of the value of ecosystem services and natural capital within wealth accounting approaches, thereby supporting an assessment of the sustainability of economic growth pathways.

Economic valuation can help disentangle the contribution of ecosystem services whose value is included at least implicitly in the System of National Accounts (SNA), for example the contribution of pollinators to agricultural production. It can also genuinely extend the accounting framework to encompass ecosystem services that lie outside of the SNA production boundary (e.g. carbon storage or flood protection from wetlands).

There remain however a number of technical challenges associated to applying economic valuation within an ecosystem accounting or wealth accounting context. In the first instance there are framing issues around the ecosystem service valuation per se, specifically:

- In the first instance, for consistency and to avoid double counting there is a need for an adequate accountancy framework. This needs to clearly distinguish between intermediate, final ecosystem services and goods/services valued by people and also

needs to link all of these to underlying, measurable stocks and quality of natural capital (e.g. hectares of a particular habitat).

- Secondly it is necessary to choose an appropriate methodological framework for valuation. Decisions need to be taken on whether to use demand-based approaches (looking at actual, surrogate or simulate markets), cost-based approaches (e.g., costs of replacement) or a pragmatic mix of both.

- Thirdly, issue arise in relation to non-marginal valuation. This is problematic where the relationship between marginal value and underlying size of the stock is non-linear and requires extrapolating values well outside the portion of the demand curve that analysts may be able to observe.

The further step of moving from estimates of the value of flows of ecosystem services to the underlying stocks of natural capital introduces further uncertainties. These include uncertainty about the level of the stocks of natural capital going forward and its degree of substitutability with other stocks of capital, choice of discount rate and future preference for ecosystem services.

Finally it is worth bearing in mind that valuing stocks does not tell us much about their underlying resilience and the risk of non-linearity and irreversibility thresholds. This is as a reminder of the importance of tracking key physical stocks as well as monetised quantities.

While overall these challenges points to the difficulties in applying valuation to ecosystem accounting it is worth noting that some of the challenges are in fact not unique to natural capital, and apply to all other assets on the national balance sheet. In these situations some of the difficult choices mentioned above are simply left to the market, where revenues and stock values implicitly reflect forward looking and uncertain assumptions.

However it is apparent that there may be some underlying tension between frameworks that revolve around the concept of total economic value of ecosystem services and aspects of SNA consistency. It has been argued by some commentators that approaches that aim to include ecosystem services in SNAs on the basis of measures of consumer surplus are difficult to reconcile with the monetary transactions recorded in the SNAs, and that approaches based on restoration costs or simulated revenues may be more consistent with the latter. On the other hand wealth accounting approaches explicitly revolve around a utilitarian framework concerned about maintaining a measure of intergenerational welfare.

Ruling out approaches that seek to measure change in utility as opposed to notional monetary flows (for all the additional complexities that this implies) would seem to us to be a restrictive approach. By contrast utilising frameworks that go beyond real or simulated market prices may support a richer (while more experimental) assessment of the value of the flows of ecosystem services and of the underlying stocks of natural capital.

Finally it is worth noting that different perspectives on valuation methodologies could be accommodated and applied on the basis of the same, underlying system of integrated, biophysical ecosystem accounts, clearly linking ecosystem goods and services to the underlying stocks of natural capital. In addition, such an integrated system of accounts could also support complementary approaches to wealth accounting , that is to say approaches that seek to embed stronger definitions of sustainability with reference to thresholds for critical natural assets.

1. Introduction

Environmental economics has long argued that the failure to recognise the full value of the environment lies at the root of most environmental problems, and that by contrast the solution to the latter could often involve monetising and internalising environmental value in the decisions facing economic agents. Efforts to develop valuation techniques therefore pre-date attempts to modify national accounts to measure sustainability, and go back instead to the desire to ensure a broader representation of environmental impacts in project and policy appraisal (Pearce, 2002).

This 'micro', policy-level context is also the context in which monetary valuation has more often been applied over the past few decades. For example monetary valuation of environmental goods and services has long featured in UK government guidance to project and policy appraisal as provided in the Treasury Green Book (HM Treasury, 2003) and significant effort has been undertaken over the last few years to provide practical support and guidance in a policy appraisal context[1].

In recent years (following the publication of the Millennium Ecosystem Assessment) monetary valuation of the natural environment has increasingly been related to the concept of ecosystem services, as for example in the recent TEEB study (TEEB, 2010). Arguably this allows analysts to capture the full range of environmental impacts more systematically, linking ecological effects to changes in human welfare. In turn this can help in informing a series of policy decisions that affect the management of ecosystem services and of the underlying stock of natural capital (e.g. strategic decisions about land use) as well as in wider communication on the value of the environment. These kinds of monetary estimates can also support determinations of liability for damage to the environment.

However it is not just at a 'micro' level that monetary valuation of ecosystem services can be useful. By monetising changes in the flows of benefit from an underlying stock of natural capital, valuation could also support a more explicit and complete inclusion of the value of natural capital within wealth accounting approaches, thereby supporting an assessment of the sustainability of economic growth pathways.

Currently some aspects of natural capital (such as for example the value of renewable and non renewable, subsoil resources) have been incorporated in empirical applications of wealth accounting (The World Bank, 2011), but to date these do not yet fully or explicitly take into account the value of ecosystem services. The links between ecosystem valuation and wealth accounting (within which ecosystems accounting is also a new area) are still to be made and this is at the heart of the agenda of the World Bank's WAVES project (Lange, 2011).

[1] http://www.defra.gov.uk/environment/natural/ecosystems-services/valuing-ecosystem-services/

In this issue paper we aim to provide a high level review of existing economic valuation frameworks and techniques, focussing specifically on the challenges that may arise in relation to the application of these techniques within wealth accounting approaches.
In Section 2 we provide a brief overview of existing monetary valuation techniques, including looking at their theoretical foundations and at their potential and limitations (from a conceptual as well as at empirical perspective). We draw a distinction between demand curve approaches (based on real or surrogate markets) and cost-based approaches. Among the former, we further distinguish between approaches that measure utility and approaches that measure exchange value. We refer to the complex issues around marginal vs. non marginal valuation. In Section 3 we then turn to consider the specific challenges of applying monetary valuation techniques within a wealth accounting context, looking in turn at the complexities associated with the forward looking nature of wealth accounting and at the complexities involved in moving from flows to stock valuation. The recent UK National Ecosystem Assessment (NEA, 2011) is arguably the most comprehensive study of the value of ecosystem services at a national level that has been undertaken so far. It is therefore used at various stages to illustrate the powers of monetary valuation techniques as well as the challenges of extending valuation from an ecosystem assessment to a wealth accounting context. Other experiences (such as the Spanish experience with the VANE project) are also referred to where relevant. Finally in Section 4 the paper we attempt to draw some conclusions and suggest some possible way forward as an initial contribution for discussion.

2. Overview of existing monetary valuation techniques

2.1 The Total Economic Value Framework and ecosystem services

Valuation is the last stage of an often detailed assessment of the impacts on ecosystem services arising from a policy change. The value of natural resources is often considered within the Total Economic Value (TEV) framework (Figure 1), and this can be used to value ecosystem services.

TEV refers to the total gain in wellbeing from a policy measured by the net sum of willingness to pay (WTP) or willingness to accept (WTA). WTP/WTA refer to the monetary measure of the value of obtaining/forgoing environmental (or other) gain or avoiding/allowing a loss. These estimates therefore are the translation to a monetary metric of the change in welfare associated to an underlying change in the provision of an environmental good. For goods and services that are traded on markets, marginal WTP equates to the market equilibrium price and marginal WTP curves are essentially demand curves for the goods or services in question. For goods and services that are not traded on markets, demand curve and WTP/WTA have to be estimated using alternative economic valuation techniques (see Section 2.2).

TEV include use value and non use value. **Use value** includes direct use, indirect use and option value. Specifically:

- Direct use value arises where individuals make direct use of an ecosystem service, whether by extracting resources from the ecosystem (e.g. food, timber) or from non-consumptive use (e.g. recreation, landscape amenity).

- Indirect use value arises where individuals benefit from ecosystem services supported by a resource rather than directly using it. Global life-support functions (e.g. climate regulation) and local life support functions (e.g. water regulation; soil retention; nutrient cycling; pollination) can both generate this kind of value.

- Option value arises when people place value on having the option to use a resource in the future even if they are not current users. In the context of ecosystem services, option value describes the value placed on maintaining ecosystems and their component species and habitats for possible future uses, some of which may not yet be known.

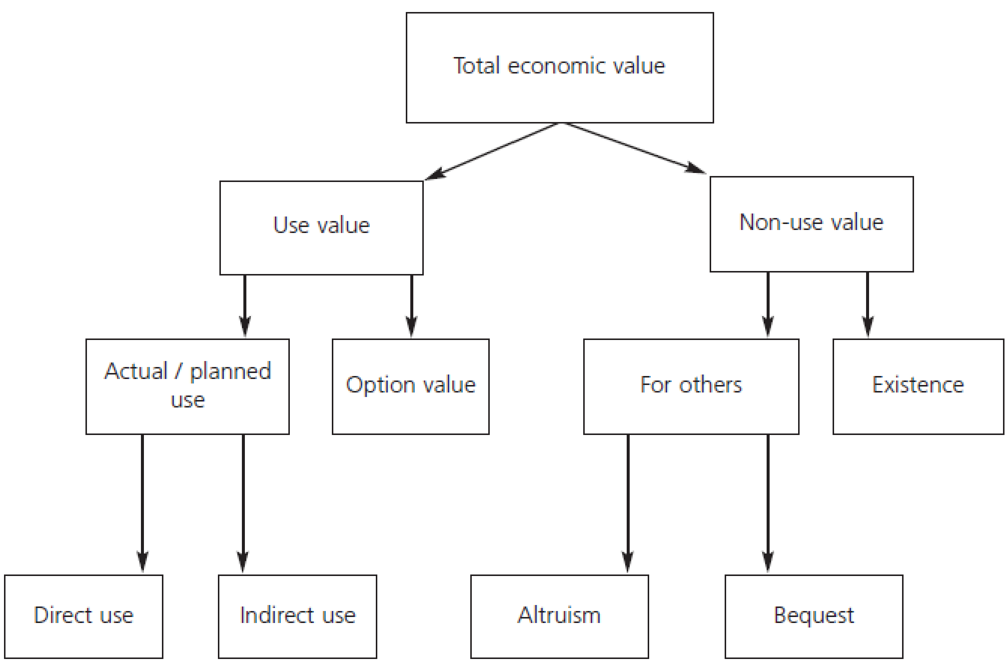
Non-use value is derived from the knowledge that the natural environment is maintained. There are three main components:

- Bequest value arises where individuals attach value from the fact that the ecosystem resource will be passed on to future generations.

- Altruistic value arises where individuals attach values to the availability of the ecosystem resource to others in the current generation.

- Existence value is derived from the existence of an ecosystem resource, even though an individual has no actual or planned use of it. For example, people are willing to pay for the preservation of whales, through donations, even if they know that they may never actually see a whale.

Figure 1: The Total Economic Value (TEV) framework



Source: Defra (2007)

The TEV framework and the Millennium Ecosystem Assessment framework for categorising ecosystem services can be seen as complementary (Table 1). The TEV framework is a useful tool for exploring what types of values for each ecosystem service we are trying to elicit. This helps in determining the valuation methods required to capture these values. What is perhaps worth emphasising here is that there are many dimensions of the value of ecosystem services are already included at least implicitly in the System of National Accounts (SNA), for example for provisioning services (e.g., timber) or when ecosystem services contribute as an input to the production of goods and services that are traded on the market (e.g., pollination services contributing to agricultural production). In many other cases the value of ecosystem services will not be included because they are flows outside of the SNA production boundary (e.g. carbon storage or flood protection from wetlands). In both cases explicitly measuring these flows can improve our ability to understand the links between the environment and the economy.

Table 1: Millennium Ecosystem Assessment (MA) framework and TEV framework

MA framework		TEV framework			
MA Group	Service	Direct Use	Indirect use	Option value	Non-use value
Provisioning	Includes: food; fibre and fuel; biochemicals, natural medicines, pharmaceuticals; fresh water supply	*		*	
Regulating	Includes: air-quality regulation; climate regulation; water regulation; natural hazard regulation etc.		*	*	
Cultural	Includes: cultural heritage; recreation and tourism; aesthetic values	*		*	*
Supporting	Includes: Primary production; nutrient cycling; soil formation	*Supporting services are valued through the other categories of ecosystem services*			

Source: Defra (2007)

2.2 Defining the object of ecosystem services valuation

It has been argued that environmental economics has been focussing for a long time on missing prices for non-market environmental good and services while not focussing sufficiently on developing a coherent and consistent definition of the quantity changes to which prices apply (Boyd et al., 2006). Recent valuation studies based on an ecosystem approach has tended to place more emphasis on a rigorous definition of the goods and services being valued. The UK NEA among other studies has emphasised the distinction between intermediate and final ecosystem services and between final ecosystem services and the goods and services that ultimately matter to people.

Supporting services can be defined as intermediate services and should be accounted for through impacts on other services and therefore not valued separately (Table 2a). Other ecosystem services will be intermediate or final services depending on their relationship with final goods and services that are valued by consumers (Table 2b). For example for angling water quality (an aspect of natural capital) is an intermediate service in the provision of fish, but so will be other capital inputs such as human capital (the skills of the fisherman and the time invested) and man-made capital (the fishing gear). By contrast for drinking water, water quality is a final service. One therefore needs to bear in mind the distinction between final and intermediate services when valuing ecosystem services to ensure that the good or service to be valued is clearly defined, that no double counting is introduced in valuing the flow of ecosystem services from a given habitat and finally to avoid overestimating the specific contribution of natural capital and ecosystem services to the production of the final good or services.

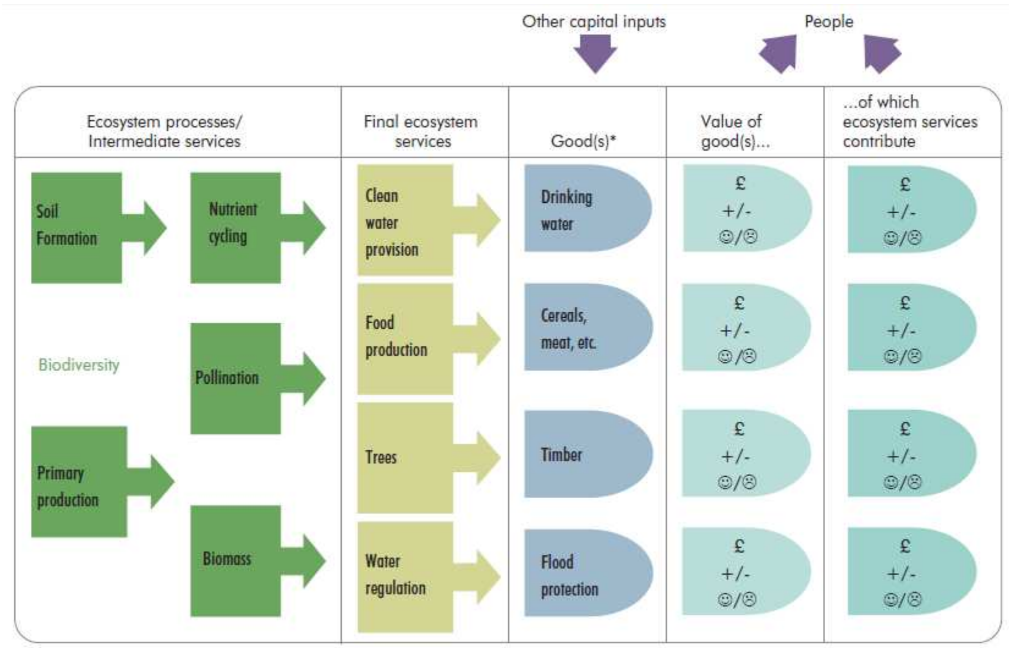
This is an issue which – while important in framing valuation studies whatever the application of the ensuing estimates – is particularly resonant in terms of ensuring consistency with SNAs. Indeed in the latter it is only end-products that are included in GDP, and not the value of intermediate services to the economy such as manufacturing processes. Therefore, developments in the methodology for valuing ecosystem services, which also focuses on valuing final goods arising from ecosystems, are helpful in terms of aligning valuation with national accounting approaches.

Table 2a: Ecosystem services in the UK NEA classified by type (provisioning, regulating, cultural and supporting) and whether or not they are final ecosystem services or intermediate services/processes

Ecosystem processes/intermediate services		Final ecosystem services	Example of goods
Supporting services		**Provisioning services**	
	• Primary production	• Crops, livestock, fish	Food
	• Soil formation	• Trees, standing vegetation, peat	Fibre, energy, carbon sequestration
	• Nutrient cycling	• Water supply	Domestic and industrial water
	• Water cycling	• Wild species diversity	Bioprospecting, medicinal plants
		Cultural services	
• Decomposition		• Wild species diversity	Recreation
• Weathering		• Environmental settings	Recreation, tourism, spiritual/religious
• Climate regulation		**Regulating services**	
• Pollination		• Climate regulation	Equable climate
• Disease and pest regulation		• Pollination	
• Ecological interactions		• Detoxification and purification in soils, air and water	Pollution control
• Evolutionary processes		• Hazard regulation	Erosion control, flood control
• Wild species diversity		• Noise regulation	Noise control
		• Disease and pest regulation	Disease and pest control

(*): the term 'good(s)' includes all use and non-use, material and non-material outputs from ecosystems that have value for people.

Table 2b: Links between ecosystem processes/intermediate services, final ecosystem services and goods and services that benefit people (subset of ecosystem services)



Source: UK NEA (2011)

2.3 Economic valuation techniques

Generally speaking the further one moves from direct use value towards indirect use value, option value and non-use value the less likely it becomes that the associated activities are traded on a market and the more challenging it becomes to measure values. At one end of the spectrum we are dealing with goods that are traded on a market (e.g. timber) and whose use value can be directly observed. Measuring indirect use values (e.g. water regulation) is often significantly more challenging as the associated value can only be inferred through surveys or by observing proxy markets. This applies to a greater extent to option values. Finally non-use values may only be measured through survey-based methods, though these raise even greater issues of reliability and robustness.

The variety of valuation techniques that can be applied to monetise ecosystem services (including their relative strengths and weaknesses) are briefly reviewed in the rest of this section, including through an overview of the techniques that were employed in the recent UK National Ecosystem Assessment. What is worth noting here from a wealth accounting perspective is that in several cases the value of ecosystem services (especially the value associated to direct and indirect use values) may be reflected to an extent in monetary transaction that are already captured in SNAs. For example with reference to the timber example above, the value of provisioning services from forested land is largely reflected in the forestry sector's turnover. Equally at least part of the recreation value of forestry may be reflected in the tourist and hospitality sector's turnover. However the value of these services is not be separately identified in SNAs and other ecosystem services from forestry (e.g., carbon storage, water and nutrient cycling, hazard regulation) are not included at all.

Market price-based approaches

Market prices can in some cases provide a direct measure of economic value of an ecosystem service. This may be the case for instance for the market price of provisioning services such as the market price for timber or fish (suitably adjusted to remove taxation or correct for subsidies).

But market prices can also provide indirect information about the value that economic agents place on certain types of ecosystem services. For example:

- Avertive expenditure approaches value ecosystem services by looking at the actual expenditure that is undertaken in other contexts to avoid environmental damage which is currently avoided as a result of some ecosystem services being provided. For example the value of pollution control and detoxification services can be assessed by looking at the costs being borne to avoid exposure to similar hazards in other contexts (e.g. costs of meeting health and safety regulations).

- Avoided damage cost approaches calculate the costs that are avoided by not allowing the ecosystem to degrade, e.g. the flood mitigation value of a wetland area can be estimated by looking at the increase in flood-related costs should that area be drained[2].

Approaches based on surrogate or hypothetical markets

The main types of economic valuation methods available for estimating public preferences for changes in ecosystem services are Revealed Preference (RP) and Stated Preference (SP) methods. Specifically:

- Revealed preference (RP) methods rely on data regarding individuals' preferences for a marketable good which includes environmental attributes. These techniques rely on actual markets. Included in this approach are: market prices, averting behaviour, hedonic pricing, travel cost method, and random utility modelling.

[2] It is worth noting that both avertive expenditure and avoided damage costs approaches typically require establishing a counterfactual and may well be used jointly depending on the assumptions in the counterfactual (e.g. what would happen to a piece of land after it has been drained, including for example mitigating flood risks by building canals but accepting a degree of increased residual risk).

- Stated preference (SP) methods use carefully structured questionnaires to elicit individuals' preferences for a given change in a natural resource or environmental attribute. In principle, SP methods can be applied in a wide range of contexts and are the only methods that can estimate non-use values which can be a significant component of overall TEV for some natural resources. The main options in this approach are contingent valuation and choice modelling.

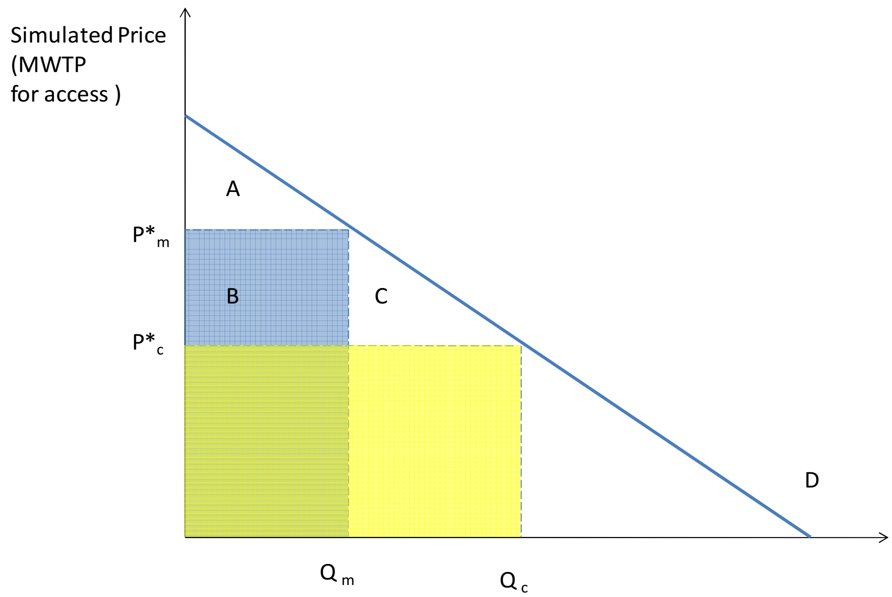
There is a huge case study literature on the application of these techniques which goes back several decades, although the techniques themselves have become more sophisticated over time both in terms of data gathering/survey administration and in terms of the econometric analysis applied to data in order to estimate WTP/WTA. More recently additional valuation approaches have been proposed specifically in order to assess the value of ecosystem services, specifically:

- Production function approaches (Barbier, 2007) consist of estimating the contribution of ecosystem services that enhance the productivity of production processes in terms of their contribution to the value of the final product being traded on the market (e.g. coastal wetland contribution to commercial and recreational fisheries through species maintenance and recharge). They therefore tend to focus on indirect use values in terms of the TEV framework mentioned above, and on disentangling the contribution of ecosystem services from monetary transactions already included in national accounts. Essentially the approach tries to determine the price that ecosystem services may command at the margin in a hypothetical market for inputs.

- The Simulated Exchange Value approach is an alternative approach to welfare-based valuation which has been proposed by a team of Spanish economists led by Pablo Campos and Alejandro Caparros (Caparros et al. 2003, Campos et al. 2006, Caparros 2010) in the specific context of green accounting in the forestry sector. The approach is explicitly not about measuring economic value, rather it aims to measure the income that would occur in a hypothetical market where ecosystem services were bought and sold. It involves estimating a demand and a supply curve for the ecosystem service in question and then making further assumptions on the price that would be charged by a profit-maximising resource manager under alternative market scenarios. It then takes the hypothetical revenue associated to this transaction (but not the associated consumer surplus) as a measure of value of the flow of ecosystem services (Figure 2).

It is worth noting that the Simulate Exchange Value approach still relies on SP techniques (or indeed other valuation approaches) to estimate demand curves for the ecosystem services it tries to value. However by estimating the value of ecosystem services in terms of revenue it can arguably represent a more consistent basis for including this value in national accounts alongside monetary transactions. A caveat is that economic valuation studies tend to adopt a partial equilibrium framework, so that even when they reflect directly or indirectly consumers' budget constraints (as in the case for example of RP studies or carefully crafted SP studies) the impacts on other markets is not being tracked, so some consistency issue also applies to Simulated Exchange Value approaches. On the other hand there are already example of simulated income flows in national accounts (e.g., notional income associated to owner-occupied properties) to which this small inconsistency also applies (Caparròs et al., 2003).

Figure 2: Economic value vs. simulated exchange value



The Simulated Market Price Approach uses demand and supply curve information for the ecosystem service in question to estimate a hypothetical monopoly price (P^*_m) and competition price (P^*_c). It then estimates the associated revenue under the demand curve by multiplying these prices for the associated, hypothetical quantities. What the approach does not do is to include in these calculations consumer surplus (areas A under monopoly or A+B+C under competition in the picture).

Finally as part of a growing policy interest in 'social wellbeing' in a number of countries innovative approaches have been put forward on measuring subjective wellbeing. In particular the so called 'Life Satisfaction Approach' could provide and alternative approach for economic valuation of ecosystem services as well as other non-market goods by estimating the life satisfaction they provide and converting this into a monetary figure by using estimates of the effect on income on life satisfaction (Fujiwara et al., 2011). These subjective wellbeing approaches to valuation have been applied in a few dozens of published studies dating back to the early 200s, including some looking at life satisfaction in relation to environmental quality, especially air quality. As the techniques mature they may become a useful addition to the tools available to the environmental economists to assess the value of ecosystem services, including perhaps those services (e.g., cultural services such as 'sense of place') where it may be difficult to identify surrogate markets or to develop surveys that place respondents in a simulated market context.

In practice approaches based on surrogate or hypothetical markets can also be used to support/complement market price-based approaches such as the avoided damage costs approaches described above, For example avoided damages from floods can include wider welfare impacts measured through RP or SP studies as opposed to purely financial costs. The key point to note is that the methods described in this and in the previous section are essentially demand curve methods, whether demand for the ecosystem services in question is directly observed on the market (adjusted market price), observed in other markets where similar goods or services are being purchased (avertive expenditure) or assumed by referring to willingness to pay to avoid costs estimated through RP or SP studies (damage cost approaches). These methods are therefore conceptually different from the cost-based approaches discussed below.

Cost-based approaches

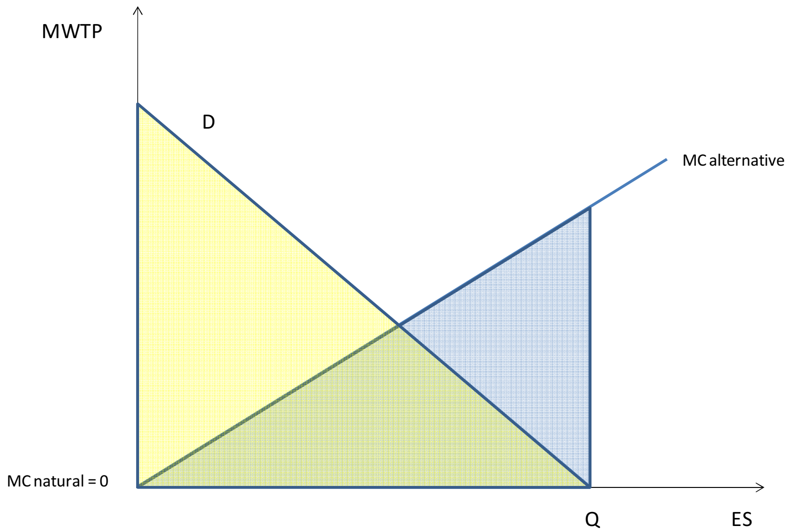
Cost-based approaches to valuing environmental goods and services consider the costs that arise in relation to the provision (or the restoration) of environmental goods and services, which may be directly observed from markets. Included under this heading are for example opportunity cost; cost of alternatives and replacement costs. Costs of habitat supply or restoration also fall in this category.

It should be noted however that as these methods are based on the supply curve for ecosystem services, they do not strictly measure utility. In other words these methods do not provide any information on the underlying demand curve for the relevant sets of ecosystem services methods and therefore do not necessarily convey any information about social welfare.

As an illustration, the cost of alternatives is an approach that considers the cost of providing a substitute good that would perform a similar function to an environmental good. For example, wetlands may be valued on the basis of the cost of building man-made flood defences of comparable effectiveness. Flood protection is one of many wetland services, but wetlands and man-made flood defences can be thought of as substitutes in the provision of a given level of flood risk which people will value. The value of the wetland can be then assumed to be at least as much as the cost of the man-made protection that would be required in the absence of the wetland.

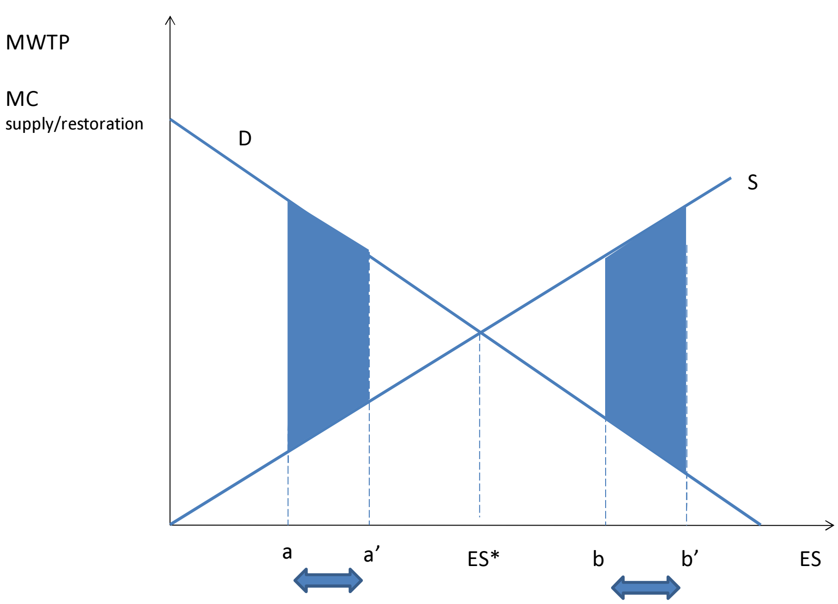
However, this approach is only a valid measure of value under a series of assumptions. First of all the man-made alternatives should be equivalent in quality and magnitude to the ecosystem functions they replace. Secondly the alternatives should be the least-cost alternative methods of performing the functions. Finally a group of individuals has to be willing to incur these costs to obtain the services. In general using costs as a proxy for value could lead to either an overestimate or an underestimate the latter (Figures 3a and 3b). Notwithstanding these limitations, cost-based approaches can be useful in validating the scale of values obtained from measurement of direct utility.

Figure 3a: Benefits of ecosystem services vs. costs of alternative



In this simple example we assume marginal costs of provision of an ecosystem service through natural capital are zero and quantity Q is demanded given a demand curve D. The associated benefits are the surplus area in yellow. This in general will be different from the costs of provision of the same level of service through a man-made alternative, which is the total cost area in blue.

Figure 3b: Marginal costs vs. marginal benefits of habitat supply/restoration



MWTP and marginal costs of habitat supply/restoration should be in theory equivalent in correspondence of the optimal level of provision of ecosystem services ES*. However they will generally be different and in any case using supply/restoration curve information will lead to underestimates of the value of the change (e.g. in moving from a to a') or overestimates (in moving from b to b'), depending on whether the initial level of provision is suboptimal or excessive.

Value transfer

It is worth mentioning at this stage that all the methods discussed so far are so-called 'primary valuation techniques', i.e. techniques that can be applied to produce monetary estimates of the value of ecosystem services in particular circumstances. In practice in many policy applications due to timing or budget constraints it may not possible, affordable or proportionate to undertake primary valuation studies. In particular producing state of the art Stated Preference or Revealed Preference studies can be an expensive exercise, involving multiple stages of analysis (including pilots to inform study design) and costing anywhere in the region of US$ 100,000 to several hundred thousand. In many situations therefore value transfer (also known as 'benefits transfer') can be a cost effective alternative, and arguably transfer of value derived from specific valuation case studies is likely to be the norm in attempts to include the value of ecosystem services within wealth accounting approaches.

Essentially value transfer techniques consist of applying estimates of the value of ecosystem services to a different geographical and policy context from the specific context in which they were develop, but a context that is nevertheless sufficiently similar for the transferring of (suitably adjusted) values to be meaningful. However while value transfer can be a swifter approach compared to undertaking primary valuation research it remains a technique that requires considerable judgement and expertise in order to produce acceptable results (Eftec, 2010).

An overview of various valuation methods in their application to ecosystem services is provided in Table 3 below from the UK NEA.

2.4 Valuing ecosystem services at the national level

While there is a vast literature on case-study applications of environmental valuation techniques to ecosystem services (Table 3), examples of comprehensive economic assessments of ecosystem services at the national level are still rare.

The UK National Ecosystem Assessment (NEA) (Bateman et al., 2010; UK NEA, 2011) arguably includes the most comprehensive and sophisticated of such assessments to date. The economic analysis for the NEA relied on a major literature review of previous literature on the value of ecosystem services as well as on new analyses undertaken as part of the NEA initiative. It also produced a forward-looking and high-level analysis of how the value of a subset of ecosystem service related goods may change over a plausible set of future land use scenarios to 2060. Overall, it made the case that the value of ecosystem services at the national scale is very significant and that fully reflecting the value of ecosystem services in policy decisions can support prospects for sustainable growth in the medium to long-term as well as underpin efforts to promote wellbeing.

The NEA applied most of the valuation techniques mentioned above, deriving a wealth of estimates on the value of the UK's ecosystem services (Table 4). In terms of choice of valuation techniques, the NEA appears to have adopted a valuation hierarchy whereby it relied on market prices where appropriate (e.g., for provisioning services), on surrogate markets where possible (e.g. in valuing amenity) and on stated preference techniques only where other approaches would not be viable (e.g., for non-use values of biodiversity, although the NEA raised a lot of questions about the appropriate use of stated preference approaches to such complex goods).

The NEA approach to value of carbon storage is worth a separate mention. The NEA used both estimates of the marginal damage cost of carbon emissions (from the Stern Review) and UK Government guidance on carbon values to come up with valuation ranges. The latter values changes in GHG emissions in terms of the opportunity cost of meeting climate policy targets, essentially using a 'target consistent' approach, based on estimates of the abatement costs that will need to be incurred to meet specific emissions reduction targets" (DECC, 2009). This is argued by DECC to be appropriate given the considerable uncertainty associated to available estimates of marginal damage costs of carbon emissions, and in terms of the typology described above is essentially a cost-based approach.

It is worth noting that the NEA authors were very aware of issues of scale and geography, which they tried to address where possible in some of the original analysis undertaken as part of the initiative. Quality of ecosystem services can vary between different locations and so can the level of demand for a specific service (e.g. potential visitors for a recreational site, reflecting population density in its area of attraction). To some extent these are typical issues that need to be addressed through appropriate value transfer techniques, but the issue of local vs. national demand in particular is also related to the issues discussed in the next section on the complexities and potential fallacies associated with moving from marginal to non-marginal valuation.

Project VANE (2008), the recent economic assessment of Spain's natural assets, is another recent attempt at estimating the value of flows of ecosystem services at a national level. VANE also applied a range of valuation techniques, though on balance appears to have relied predominantly on information derived from market prices and to have focussed on TEV components associated with direct use value (including non-consumptive direct use such as recreation), with some aspects of indirect value (water purification and carbon sequestration).

Table 3: Examples of application of different valuation methods to Ecosystem Services

Table 22.1 Various valuation methods applied to ecosystem services. Source: Bateman *et al.* (2011a).

Valuation method	Value types	Overview of method	Common types of applications	Examples of ecosystem services valued	Example studies
Adjusted market prices	Use	Market prices adjusted for distortions such as taxes, subsidies and non-competitive practices.	Food; forest products; Research & Development benefits.	Crops; livestock; multi-purpose woodland.	Godoy *et al.* (1993); Bateman *et al.* (2003)
Production function methods	Use	Estimation of production functions to isolate the effect of ecosystem services as inputs to the production process.	Environmental impacts on economic activities and livelihoods, including damage costs avoided, due to ecological regulatory and habitat functions.	Maintenance of beneficial species; maintenance of arable land and agricultural productivity; support for aquaculture; prevention of damage from erosion and siltation; groundwater recharge; drainage and natural irrigation; storm protection; flood mitigation.	Ellis & Fisher (1987); Barbier (2007)
Damage cost avoided	Use	Calculates the costs which are avoided by not allowing ecosystem services to degrade.	Storm damage; supplies of clean water; climate change.	Drainage and natural irrigation; storm protection; flood mitigation.	Kim & Dixon (1986); Badola & Hussain (2005)
Averting behaviour	Use	Examination of expenditures to avoid damage.	Environmental impacts on human health.	Pollution control and detoxification.	Rosado *et al.* (2000).
Revealed preference methods	Use	Examines the expenditure made on ecosystem-related goods, e.g. travel costs for recreation; hedonic (typically property) prices in low noise areas.	Recreation; environmental impacts on residential property and human health.	Maintenance of beneficial species; productive ecosystems and biodiversity; storm protection; flood mitigation; air quality; peace and quiet; workplace risk.	See Bockstael & McConnell (2006) for the travel cost method and Day *et al.* (2007) for hedonic pricing.
Stated preference methods	Use and non-use	Uses surveys to ask individuals to make choices between different levels of environmental goods at different prices to reveal their willingness to pay for those goods.	Recreation; environmental quality; impacts on human health; conservation benefits.	Water quality; species conservation; flood prevention; air quality; peace and quiet.	See Carson *et al.* (2003) for contingent valuation and Adamowicz *et al.* (1994) for discrete choice experiment approach.

Source: UK NEA (2011)

Table 4: Ecosystem goods valued by the UK NEA

Good	Valuation method	Valuation metrics
Marine food production	Market prices	Annual value of UK fish landing and annual value of UK aquaculture (fish and shellfish). £ million p.a. However, there is insufficient data to isolate ecosystem contribution from manufactured capital inputs.
Pollination services	Production function method	£ million p.a.
Biodiversity non-use values	Stated Preferences	£ million p.a. for terrestrial biodiversity, inland wetlands, coastal wetlands, marine biodiversity.
Biodiversity non-use values	Revealed preferences (legacy values)	£ million p.a.
Timber production	Market prices	£ p.a. for total green tonnes £/ha p.a. hard wood £/ha p.a. soft wood No allowance made for manufactured capital inputs.
Carbon storage and GHG flux: Marine and Coastal Margins	DECC values	Marginal (and total) values for coastal margin carbon storage (sand dune and salt marsh). £/ha p.a. (marginal) and £ p.a. (total). Carbon storage in marine habitats non-quantified.
Water quality and quantity	Market prices, cost savings and stated preferences	Marginal and total water quality benefits of inland wetlands and coastal wetlands. £/ha p.a. (marginal) and £/p.a. (total). Potential benefits of improvements to river water quality. £ p.a. (total) and £/km p.a. (average) Impacts of losses due to climate change upon UK water availability. £ million p.a.
Flood protection: inland	Market priced cost savings	Climate change induced increases in flooding costs range up to £ billion p.a. Marginal value of flood defence from wetlands. £/ha p.a.
Flood protection: coastal Stated preference	Stated preference	Marginal and total value of flood defence from wetlands. £/ha p.a. (marginal) and £ p.a. (total).
Game and associated landscape values	Market prices	Woodland game revenues. £/ha p.a.
Amenity value of nature	Hedonic pricing, stated preference	Value of high environmental amenity £ p.a. /household. Marginal and total amenity value of inland wetlands. £ p.a./ha (marginal) and £/p.a. (total).
Education and environmental knowledge	Wage rate assessments, travel and time cost valuations	Environmental knowledge embodied in higher qualifications. £ billion p.a. . Value of school trips to just 50 nature reserves. £1.3 million p.a.
Health	Stated Preference	Value of Marginal increase in woodland within one km of a person's home. £/person p.a. Value of views of green space from the person's home. £/person p.a.

Source: UK NEA (2011)

2.5 Marginal vs. non marginal valuation

Typically valuation studies attempt to estimate the value of a marginal change in the underlying ecosystem services. Such estimates are then applied in assessing the impacts of marginal projects or policies, so that for the change being considered, prevailing market prices and/or available estimates of marginal WTP/WTA can be considered a good proxy of the associated change in utility. It is a well established principle of cost benefit analysis that where these changes are non-marginal it may no longer be possible to rely on current prices to value the change in utility, and it may instead be necessary to consider the shape of the underlying utility functions. When valuing nature this general problem is compounded by the potential presence of thresholds, non-linearities and irreversibilities when natural capital and the flow of ecosystem services that derives from it are depleted below certain levels. It is also compounded by the fact that determining where these critical thresholds for natural capital lie compared to the status quo is in itself an exercise which is subject to significant uncertainties.

The NEA focused on valuing the marginal value of ecosystem services, i.e. the value of changing a single unit of a stock. This is because for many goods and services, marginal values will change with the total size of the stock, even when the overall stock level is above sustainable levels (Figure 3). The NEA therefore emphasised that there are risks associated with extrapolating monetary values for ecosystem services beyond the realm of observation when relationships between marginal values and levels of the stock are non-linear, with the associated risk of either significantly overestimating the value of the resource (by integrating below an assumed demand curve) or significantly underestimating it (if integration is carried out by keeping marginal values constant at current levels of provision).

Notwithstanding the reservations about total valuation mentioned above, estimates of the total values of the flow of ecosystem services in terms of £ per year were also reported for illustrative purposes in the NEA. However the NEA did not attempt to undertake a temporal integration of monetary estimates of annual flows of ecosystem services to derive a monetary estimate of the value of the underlying stock.

Interestingly these concerns about total valuation would not seem to apply with equal strength to the Simulated Exchange Value method mention above, which makes it clear on the other hand that what is being valued is a hypothetical income corresponding to market equilibrium as opposed to a measure of value/economic welfare.

Project VANE also focussed on reporting values of annual flows (Euros/year) and average marginal values (Euros/ha/year) for a number of ecosystem services (including for example timber supplies, water supplies and carbon storage in soils). As with the NEA, VANE also did not attempt to estimate capitalised values for the underlying stocks of natural capital. The synthesis report did mention that mathematically that it would be possible to integrate the values of these flows over time to produce an estimate of the total value of the underlying stock in its current state, but this kind of calculations was not pursued as part of the project. The VANE synthesis report also emphasised the difference between stock values derived through temporal integration of flows of ecosystem services and costs of restoration of the stocks of natural capital that provide those services.

Figure 3: Marginal value of ecosystem services as a function of the underlying stocks

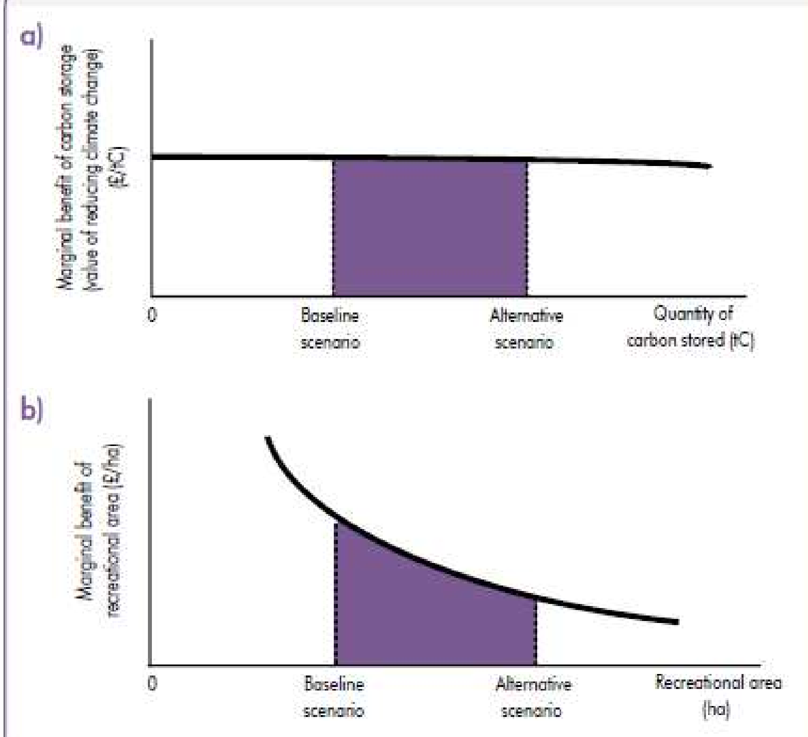


Figure 22.1 Marginal value curves for two goods: a) carbon storage (tonnes of carbon, tC) and b) recreational area (hectares, ha).

Source: UK NEA (2011)

3. Application of economic valuation of ecosystem services within a wealth accounting framework

3.1 Valuation in the theory of wealth accounting

There is an extensive body of economics literature on sustainability underpinning the theory of wealth accounting (Arrow et al. 2010; Dasgupta, 2008). This literature defines the social planner's problem as one of maximising an intergenerational social welfare function, defined as a discounted flow of utilities, which in turn depends on consumption and on capital stocks. Natural capital is one of the stocks of capital which is typically covered by this literature and which can contribute to utility by supporting production of goods and services or through direct enjoyment (e.g., wetlands providing flood defence services as well as recreation).

For example, in a standard four capitals framework man-made capital (K), human capital (H), social capital (S) and natural capital (N) are all input to the economy's production function. Therefore output Y at time t equals:

$$Y(t) = f(K_t, H_t, S_t, N_t)$$

An intergenerational social welfare function V is can then be defined at any time *t* as the temporal integral of social welfare *U* at each point in time for the foreseeable future (appropriately discounted). Specifically:

$$V(t) = \int_t^\infty U(C(z), N(z))e^{-\delta(z-t)}dz$$

Where at any time *t* consumption *C* equals production minus investments in the four stocks of capital and where natural resources *N* contribute directly to utility as well as one of the inputs into the economy's production function. A simple (weak) sustainable development rule can then be introduced to require that intergenerational welfare is non decreasing over time, i.e.:

$$dV(t)/dt \geq 0$$

<u>Accounting prices for stocks of capital</u>

The shadow price or accounting price p_t of different stocks of capital is the marginal contribution to intergenerational welfare associated to a unit change in the quantity (or quality) of these stocks. Formally:

$$p_t^n = \partial V(t)/\partial N(t)$$

As discussed by Arrow et al. (2010), these accounting prices are at any time a function of all stocks of capital, not just the particular asset they refer to. They are also a function not just of the economy today, but on the entire future of the economy. Therefore, they are a function of the degree to which various assets are substitutable for one another today and in the future. In other words, accounting prices for natural resources (and indeed for any other type of capital stocks) as defined by the theory of wealth accounting are heavily scenario dependent. Besides, the choice of utility discount rate also affects shadow prices (attenuating the potential impacts of future scarcities on today's prices). Therefore shadow prices embody the critical ethical choice around utility discounting that has been highlighted by the climate change debate in recent years (Dietz et al, 2007, Beckerman et al., 2007).

Moreover, accounting prices implicitly require assumptions about how changes in natural capital will be valued by future people as opposed by current people, or about how preference for nature and ecosystem services may evolve over time. This is linked to the discounting debate and particularly to whether consumption discounting (i.e. the element of discounting that reflects decreasing marginal utility of income under the expectation of increase in consumption per capita as opposed to mere 'impatience') should apply to all classes of ecosystem services. Atkinson (2009) identifies this as a significant issue, complicated by the fact that there are factors pointing to WTP for ecosystem services potentially changing in different directions over time. On the one hand richer individuals may attach relatively more value to ecosystem services, on the other hand behavioural economics (and in particular the importance of 'reference points') suggests that in some cases future people may not in fact attach a particular value to losses that occurred in the past.

Given a set of accounting prices, an economy's comprehensive wealth can be defined as the cumulative value of all its capital stocks at a particular point in time, i.e.:

$$W(t) = p_t^k K(t) + p_t^h H(t) + p_t^s S(t) + p_t^n N(t) \qquad (1)$$

While the value of natural capital in particular can be expressed as:

$$W^n(t) = p_t^n N(t)$$

The sustainability rule mentioned above can then be re-formulated to define a sustainable economy as one where (holding shadow prices constant) comprehensive wealth is non-decreasing over time, or (equivalently) where net or genuine savings (that is savings adjusted for resource depletion, environmental damage and net changes in other forms of capital) are non negative.

From a natural capital perspective these are of course simplistic rules in an environment characterised by highly non-linear natural processes, and there is recognition (for example in Dasgupta, 2008) that considerable adjustments are likely to be required to apply this theory in the real world. Specifically, the issues of non-linearity, irreversibility and uncertainty mentioned here in Section 2.5 point to more fundamental limitations of weak sustainability approaches and to the need for complementing monetary valuation and wealth accounting with assessments of critical stocks (e.g. Atkinson 2009, Price et al. 2010, UK NEA 2011, Howard et al, 2011, Turner, 2011). More generally, operationalising a four capitals framework such as the one illustrated here faces other significant challenges, including providing meaningful measures of social capital, though nevertheless has been proposed as a conceptual framework to guide policy (Harper et al., 2011).

Valuing flows and valuing stocks

With the important exception of those ecosystem services values that are derived from capitalised values of other assets (e.g., amenity values of wetlands derived from property values through hedonic price analysis), value estimates from environmental valuation studies tend to be changes in *current* marginal utility, i.e. they tend to be defined as:

$$\overline{p}_t^n = \partial U(t) / \partial N(t)$$

(Where in general $\overline{p}_t^n \leq p_t^n$)

The value of annual flows of ecosystem services can then be expressed as the product of marginal values (e.g., £/hectare of a given habitat) multiplied by physical stocks (e.g. hectares of a given habitat):

$$F(t) = \overline{p}_t^n N(t)$$

Temporal integration could then be applied in order to derive a monetary estimate of the underlying stock, but this would require some assumptions about physical stocks of capital N going forward. Formally, temporal integration can be expressed as:

$$W^n(t) = \int_t^\infty F(z) e^{-\delta(z-t)} dz = \int_t^\infty \overline{p}_z^n N(z) e^{-\delta(z-t)} dz$$

If N is assumed to be constant over time (a strong assumption), this expression becomes equivalent to the expression for the value of natural capital provided above, i.e.:

$$W^n(t) = N \int_t^\infty \overline{p}_z^n e^{-\delta(z-t)} dz = N \int_t^\infty \partial U(z) / \partial N(z) e^{-\delta(z-t)} dz = N\, \partial V(t) / \partial N(t) = N p_t^n$$

In other words, when the stock of natural capital is assumed to be constant the value of the stock of natural capital estimated using forward looking shadow prices applied to the current level of the stock is equal to the value estimated by time integrating the product of (constant) current marginal utility values and levels of physical stocks.

In practice as we have discussed above national-level valuation studies of ecosystem services such as the NEA and VANE have chosen not to pursue stock valuation approaches based on temporal integration of monetised flows of ecosystem services.

3.2 Challenges in applying economic valuation of ecosystem services within a wealth accounting approach

Operationally there are quite a few steps and issues that arise in developing estimates of the value of ecosystem services that are consistent with applications to wealth accounting and that can be the basis for valuing flows of ecosystem services and underlying stocks of natural capital.

The discussion so far has already highlighted that producing monetary measures for stocks of natural capital on the basis of estimates of values of ecosystem services carries a series of complexities. First of all, as discussed in Section 2, there are framing issues around the ecosystem service valuation per se, including:

- The choice of an adequate framework that clearly distinguishes between intermediate, final ecosystem services and goods/services valued by people in order to avoid double counting. Importantly within a wealth accounting approach, this framework also need to clearly establish the links between the ecosystem service pipeline and underlying, measurable stocks and quality of natural capital.

- The choice of an appropriate methodological framework for valuation, including a choice between demand-based approaches (looking at actual, surrogate or simulate markets), cost-based approaches (e.g., costs of replacement) or a pragmatic mix of both. The choices available at this stage could also include the adoption of a Simulated Exchange Value approach (focussing on simulated revenues associated to ecosystem services as opposed to welfare).

- Valuation of non marginal changes (e.g. total value of annual flows). This is problematic where the relationship between marginal value and underlying size of the stock is non-linear, for the reasons discussed in the NEA and summarised here in Section 2.4 (though it arguably applies to a lesser extent to the Simulate Exchange Value approach given its focus on simulated revenues).

Even if an acceptable solution for these complex issues can be found, projecting forward and time integrating estimates of total value of annual flows to derive estimates of the value of the underlying stocks of natural capital introduces further uncertainties. Specifically these uncertainties relate to:

- The level of the stocks going forward and its degree of substitutability with other stocks of capital. If this was to change significantly then the starting values would no longer apply at the margin, let alone being a good guide to cumulative utility. For example in a future where forested land (say) become significantly scarcer the marginal value of an extra hectare of forested land may be significantly higher (other things being equal) than current estimates.

- The choice of discount rates, including private vs. public, and which discount rate if public. This is acknowledged by most economists as having implicit ethical implications.

- Uncertainty about relative preferences for ecosystem services vs. other consumption goods and how they may change going forward. As discussed in Section 3.1 this issue is also linked to discount rate uncertainty and to the issue of whether consumption discounting should apply to all classes of ecosystem services.

Finally a, significant limitation of approaches for applying valuation of ecosystem services within a wealth accounting framework which goes deeper that some of the technical issues mentioned above is the fact that attempts at monetising stocks may not tell us much about their underlying resilience and the presence of irreversibility thresholds, This does not detract from the usefulness of including ecosystem services within wealth accounting approaches (not to mention of integrated ecosystem accounts that should ideally underpin these approaches) but perhaps acts as a reminder of the importance of tracking key physical stocks as well as monetised quantities (Price at al., Howard et al., Turner R. K.)

3.3 Putting the challenges in perspective: natural capital vs. other assets

While overall these challenges point to the difficulties in applying valuation to ecosystem accounting it is worth noting that some of the challenges are in fact not unique to natural capital, and apply to all other assets on the national balance sheet. For example, let's consider buildings (a fairly straightforward example of produced asset or man-made capital). These are typically valued for inclusion in the national balance sheet using prices actually observed in the market. In turn, based on conventional asset price theory, market prices for buildings should reflect the net present value of rents, discounted at market interest rates (in practice this is not always the case, for example the UK housing stock would appear significantly overvalued compared to its theoretical value thus derived).

It is therefore possible to draw a parallel between the value of a country's building stock and the value of its stock of natural capital in terms of some of the implicit assumptions that underpin value estimates. For example:

- The market value of the stock of building is an extrapolation of prices (or rent) set at the margin to the entire stock. It is not clear that the underlying relationship between rents and stock is linear and it would appear misleading to interpret the account value as either the economic cost required to build the stock from scratch or the economic costs associated to hypothetical obliteration;

- The market value of the stock of buildings reflects market expectations about future supply and demand and the attractiveness/risk of other classes of assets;

- The market value of the stock also reflects market expectations about prevailing interest rates (which are the private opportunity costs of holding a particular asset).

Therefore the market-observed value of the national stock of buildings can be considered to be in many ways as scenario dependent as an estimated value for its sock of natural capital. The main difference in this case is that the national accountant implicitly endorses the market views on future levels of demand and supply, as well as the market views on inter-temporal trade-offs.

4. Conclusions

Overall, developments in the application of economic valuation techniques for ecosystem services are encouraging in terms of the application of valuation within a wealth accounting framework.

There are a number of methodological issues that arise in developing estimates of the value of ecosystem services that can be used for valuing flows of ecosystem services and underlying stocks of natural capital. These issues that relate first of all to the choice of an appropriate framework for ecosystem accounting and valuation, issues about valuation of non marginal changes (e.g. total value of annual flows). When stock valuation is being pursue, these issues that relate to projecting values forward (including issues about future scarcity and substitutability, discounting and evolving preferences for ecosystem services).

On balance, it is possible to argue several of those issues are similar to issues that arise in relation to other assets on the national balance sheet. In general they could be addressed by proceeding on the basis of the best possible assumption and by making the same assumptions very transparent, although the problem of non-marginal valuation may need particular attention and may suggest placing greater confidence in estimates of the value of year on year (incremental) degradation or depletion as opposed to estimates of total annual flows or estimates of the value of stocks.

However it is apparent that there may be some underlying tension between frameworks that revolve around the concept of total economic value of ecosystem services and aspects of SNA consistency. It has been argued by some commentators that approaches that aim to include ecosystem services in SNAs on the basis of measures of consumer surplus are difficult to reconcile with the monetary transactions recorded in the SNAs, and that approaches based on restoration costs or simulated revenues may be more consistent with the latter. On the other hand wealth accounting approaches explicitly revolve around a utilitarian framework concerned about maintaining a measure of intergenerational welfare.

Ruling out approaches that seek to measure change in utility as opposed to notional monetary flows (for all the additional complexities that this implies) would seem to us to be a restrictive approach. By contrast utilising frameworks that go beyond real or simulated market prices may support a richer (while more experimental) assessment of the value of the flows of ecosystem services and of the underlying stocks of natural capital.

Finally it is worth noting that different perspectives on valuation methodologies could be accommodated and applied on the basis of the same, underlying system of integrated, biophysical ecosystem accounts, clearly linking ecosystem goods and services to the underlying stocks of natural capital. In addition, such an integrated system of accounts could also support complementary approaches to wealth accounting, that is to say approaches that seek to embed stronger definitions of sustainability with reference to thresholds for critical natural assets.

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