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Environmental Valuation and Greening the National Accounts

Challenges and Initial Practical Steps

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Abstract

The national accounts are the single most important source of information about the economy, and are widely used in all countries to assess economic performance and for policy analysis. However, the national accounts have a number of well known shortcomings when it comes to treatment of the environment. For example, while the income from harvesting timber is recorded in national accounts, the simultaneous depletion of natural forest assets is not; perhaps more importantly, essential life-support services provided by forest ecosystems are not explicitly recognized at all. Environmental accounts—"greening the national accounts"—have been developed to address the shortcomings of the national accounts, but valuation of environmental services has been controversial.

Evolving progress in the application of environmental valuation methods as well as the increasing use of these methods in official policy appraisals indicates that a renewed and serious investigation of the scope for using environmental values to "green the national accounts" is both timely and important. The purpose of this paper is to discuss a number of arguably critical issues and challenges in any such reassessment. When pioneering work on greening the national accounts was being codified, non-market valuation approaches were in their relative infancy and, at least for some approaches, beset by controversy. These controversies have not gone away. What is new is that there has been twenty years of methodological and practical developments in the methods that—along with a growing empirical record—must now be brought into the reckoning. This record is, however, diverse and, as things stand, this is likely to hamper straightforward use particularly through value transfer exercises which arguably will be central to routine application of environmental valuation in green national accounts. Nevertheless, a number of categories of environmental damage or service are sufficiently well-understood that incorporation in green national accounts is close to being practical. Other categories such as the value of ecosystem services are arguably further away from being amenable to routine accounting. However, investigation here (as elsewhere) is rapidly evolving and, moreover, further consideration could usefully take advantage of current and on-going global and national assessments being carried out elsewhere.

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1. Introduction

The development of methods to value environmental (and related non-market) goods and services continues to evolve apace. In an increasing number of countries, the practical uptake of these methods has accelerated in numerous areas of public policy that have environmental consequences. One policy-related domain, however, where this uptake has been largely conspicuous by its absence is national accounting. In this paper, we hope to make the case that there is an opportunity to address this situation.

The national income accounts are the single most important source of information about the economy, and are widely used in all countries to assess economic performance and for policy analysis. However, the national accounts have a number of well known shortcomings when it comes to treatment of the environment. For example, while the income from harvesting timber is recorded in national accounts, the simultaneous depletion of natural forest assets is not; perhaps more importantly, essential life-support services provided by forest ecosystems are not explicitly recognized at all. This can result in quite misleading economic signals about economic growth.

To address this shortcoming of the national income accounts, environmental accounting was proposed. One of the primary motivations for the early environmental accounting efforts in the mid-1980s was concern that rapid economic growth in some countries was achieved through liquidation of natural capital—a temporary strategy that creates no basis for sustained advances in wealth and human well-being, unless this natural capital is converted efficiently into other forms of wealth. Under the aegis of the UN Statistical Commission, a handbook was drafted providing a comprehensive framework for environmental accounting (UN et al., 2003), the System of Environmental and Economic Accounts (SEEA). The SEEA is currently under revision and valuation of ecosystem services has been identified as an issue that needs to be addressed. While there has been a great deal of progress in valuing environmental services and damages, there are a number of significant challenges for introducing such values in the SEEA and the national accounting framework.

Primarily, it must be asked how methods which typically have been designed to inform policies or projects which involve small changes can be either generalised or 'scaled-up' in a defensible way for the accounts. In a related vein, there are important questions about geographical and topical scope of the empirical record. This may limit the readiness of incorporating environmental valuation within national accounts, even assuming methodological concerns can be resolved.

The remainder of this paper is organised as follows. Section 2 defines terms and describes some basic concepts that are critical for framing discussion about extending environmental valuation to national accounts. Section 3 discusses the policy context for this focus and initial obstacles within the accounting domain. Section 4 provides a (very) brief overview of environmental valuation methods and considers how such methods might inform the needs of green national accounts. Section 5 provides a preliminary assessment of the extent to which relevant categories of environmental damage or service are 'ready' for valuation in (experimental) green national accounts. This includes a more detailed discussion of progress in the valuation of human health and ecosystem services. Section 6 offers some concluding remarks.

2. Accounting for Environmental Values

'Greening the national accounts' is an expression that covers a variety of activities related to resource and environmental accounting. In this paper, we use this term to refer to a subset of these activities that involve putting a monetary value on those aspects of the environment to be included in extended accounts. Furthermore, our focus is exclusively on accounting for the value of *non-commercial* (or non-market) natural and environmental resources. In contrast to commercial natural resources (such as sub-soil assets), these resources have no 'easily' observable market prices to guide measurement. Nor, for that matter, are there (in many cases) straightforwardly obvious quantities of these goods or services to assign prices to.

This creates measurement challenges on two crucially important fronts. On-going international work to account for human capital, for example, shows the importance of this focus on physical quantities (in that case, educational attainment and so on) as well as (asset) prices (see, for example, Fraumeni, 2008). Boyd (2008), for example, argues that an analogous emphasis on both elements will be just as important for greening the accounts. The focal point of much of what follows, however, will be the latter issue of valuation. One reason for the prominence accorded here is that arguably it is this element—i.e. 'valuing the environment'—that, up to now, has been seen as the crucial obstruction to greening the accounts, as we have defined this.

In terms of relating such challenges to the national accounts, these are relevant to both product and wealth accounts. In respect of the former, this might entail measuring the current value of the environmental services (or goods) that people consume. This emphasis can be seen in the on-going debate about the importance of ecosystem *services*. Of course, much of this particular concern surrounds emerging evidence that these services are being lost because of the degradation and destruction of underlying ecological assets. Accounting, in this case, is also an issue for wealth accounts as we might wish to know more about the value of natural assets or stocks that give rise to services. Common to both product and wealth accounts is an interest in the value of *asset changes*: that is, the net accumulation of natural assets over an accounting period. In the case of asset accounting, the valuation challenge is broadened somewhat in that it requires possibly deep reflection on what the value of future services (giving rise to asset values) might be as well as debates about how to discount these services.

In valuing services or assets, a crucial element is the willingness to pay (WTP) of individuals for a marginal change (e.g. an increase) in the provision of an environmental good or service on the basis of use, possible future use or non- (or passive) use.[1] In environmental economics, the concept of total economic value (TEV) is used a convenient organising framework for thinking about these different sources of value.[2] Within this framework, some of these values plausibly may spill over to the residents of countries additionally to those who live within the same national boundary as some natural asset in question (see, for example, Atkinson and Hamilton, 2007).

This transboundary issue aside, sources of TEV will differ in terms of the extent that they already leave traces in the national accounts. For example, the environment clearly supports market activity in a number of important (but indirect) ways. In this sense, the environment is a non-market input to production and consumption that is captured in the national accounts as they currently stand. Examples might include the farmer whose crop productivity benefits from the services provided by insect pollinators or the self-employed person whose working productivity is impaired by illness arising from exposure to air pollution. Put another way, what are environmental externalities to one party are not necessarily external (currently) to the national accounts. The issue is that this impact is not attributed to its correct source (Nordhaus, 2006).

In many cases, non-market approaches to valuing the environment—or 'environmental valuation'—will make use of this fact that monetary values might be revealed by actual market transactions such as a related expenditure. In other words, even though no explicit market price may exist for an environmental service of interest, what is currently in the accounts may offer important clues as to the monetary value that people attach to receiving this service. This will be a question of degree. In many instances what we seek to measure could be something that lies purely outside of the national accounts as currently constituted. Generally speaking, as we move through sources of value based

[1] This is not the only way of looking at the way in which an individual might value a change in provision. That is, for an environmental improvement, the change in wellbeing that an individual enjoys can be measured by his or her WTP for, or his or her willingness to accept (WTA) compensation to forego, that improvement.

[2] It is usual to divide this notion of TEV into use and non-use (or passive use) values. Use values relate to actual use of the good in question (e.g. a visit to a national park), planned use (a visit planned in the future) or possible use. Actual and planned uses are fairly obvious concepts, but 'possible use' could also be important since people may be want to maintain a good in existence in order to keep open the option of using it in the future. Non-use value refers to the motive to maintain some good in existence even though there is no actual, planned or possible use (e.g. to preserve an ecosystem as a bequest for future generations or simply for its own sake). It is usual to divide this notion of TEV into use and non-use (or passive use) values. Use values relate to actual use of the good in question (e.g. a visit to a national park), planned use (a visit planned in the future) or possible use. Actual and planned uses are fairly obvious concepts, but 'possible use' could also be important since people may be want to maintain a good in existence in order to keep open the option of using it in the future. Non-use value refers to the motive to maintain some good in existence even though there is no actual, planned or possible use (e.g. to preserve an ecosystem as a bequest for future generations or simply for its own sake).

on actual use through to non-use this is more likely to be the case (albeit with interesting exceptions).[3]

3. 'Valuation Matters': The Policy Context and Initial Obstacles

A prominent theme in the appraisal of contemporary public policy has been the quantification—in monetary terms—of the impacts of policy actions. Within the domains of environmental policy, it is increasingly recognised that many of these impacts are intangible. This means that the value that the public places on these impacts cannot be observed simply with reference to market information such as price and quantities directly purchased. This has given rise to a proliferation of methods that have sought to uncover, in a variety of ways, the value of these 'unpriced' goods and services.

A glance at the extent to which policy-makers have begun to rely on thinking explicitly about the monetary value of environmental benefits (and costs) appears to indicate that, for many governments, the 'political test' (for these methods) is beginning to be passed (see, for a review, Bureau and Glachant, 2006; Dale *et al*. 2009). Put another way, valuation methods have been judged by an increasing number of countries to be fit for the (policy) purpose that they primarily have been designed for: i.e. economic appraisal of the relative merits of actions that result in environmental improvements (or deterioration). Interestingly, this has led to a handful of emerging applications that seek to appraise large-scale policy actions such as, for example, the European Union's Water Framework Directive (WFD).[4] A study by Metcalfe *et al*. (2009), for example, discusses the findings of a study undertaken, on behalf on the UK's environment ministry (Defra)[5], to inform public policies to improve water quality in England and Wales. Both Dale (2009) and Delbeke *et al*. (2010) describe the European Commission's Clean Air for Europe (CAFÉ) programme as an area where health valuation has had a growing influence in guiding its air quality strategy for the European Union (EU).

Whether these same monetary approaches are useful in greening national accounts is an important item for meaningful dialogue between statistical offices *and* policy departments. Postponing such discussion carries the risk that environmental extensions to national accounts will lack relevance to policy thinking, at least for those countries that are beginning to use environmental valuation prominently. But there is also a need to anticipate the informational demands of the future for those countries where policy-

3 For example, some elements of 'non-use' are captured in the national accounts already through payments to conservation groups, government spending on protected areas and so on. The benefits arising are presumably imperfectly captured by these outlays but clearly do not lie beyond the market altogether.
4 The WFD requires Europes' waterways and waterbodies to have reached 'good ecological status' at least by 2015 and has led in a number of instances to significant efforts by EU member states' governments to understand the monetary value of the benefits of this regulatory initiative.
5 The Department for Environment, Food and Rural Affairs.

makers currently appear to place little emphasis on explicitly knowing the monetary value of environmental impacts.

Tellingly, Hecht (2005) claims that the arguments about the merits of, and progress in, non-market or environmental valuation have been heard and "have not convinced the accountants yet" (p203). The reasons cited for this in the literature on greening the national accounts are manifold. The concerns expressed are diverse and include the ethics of environmental valuation (e.g. Peskin and Lutz, 1993; de Haan and van de Ven, 2007) as well as whether incorporating environmental values within national accounts, based (in large part) on market data, is mixing 'like-with-like' (e.g. Harrison, 1993).

For example, one long-standing impasse in discussions about environmental valuation and the accounts is the alleged inconsistency between the conventional accounts, primarily based as they are on market prices, and approaches which generate values which lie outside the market (or perhaps are only indirectly reflected in market transactions). Harrison (1993), for example, expresses these concerns in the following: "Adjustments to be made to the SNA are […] a move toward a measure of welfare … This is not mixing like with like, however. If a true welfare measure were to be derived, then the basis for valuing all the other transactions recorded in the SNA at present would need to be examined and in many cases would need to be altered" (p72). There are probably at least two elements to this critique.

One facet of this concern arguably boils down to fundamental disagreements about the 'spirit' of the (existing and established) national accounts—i.e. to measure economic (or more specifically, market) activity—and the 'spirit' of valuing the environment and (non-market aspects of) environmental change—i.e. to measure (human) welfare or wellbeing.[6] On a strict view then of what the national accounts stand for, environmental policies seek to improve something (i.e. broader wellbeing) which, ultimately, must remain outside of these accounts. A critical issue is whether this apparent dividing line between what is market activity (the domain of the accounts) and what is the "non-market" is worth preserving *whatever the cost* in terms of losing an opportunity to building bridges to contemporaneous thinking about environmental policy.

A second prominent aspect of 'not mixing like with like' is the concern that non-market values are not at all like market prices which lie at the heart of valuation in the core accounts.[7] A specific example of this unease is the claim that these non-market values

[6] See, however, e.g. Hamilton and Clemens (1999), Weitzman (2003), Dasgupta and Maler, 2000, Hamilton and Withagen, (2007) and Dasgupta, (2009) for discussions about the links between economic and social welfare and the national accounts.

[7] Clearly, this is a little misleading. In almost all countries, a significant part of what is measured in the national accounts is government activity where output is not valued using market prices and, indeed, 'output' is not really measured at all.

measure consumer surplus which has no place in the national accounts.[8] While this concern has proved to be remarkably durable, contributions by Nordhaus (2006) and Abraham and Mackie (2006) conclude that its basis rests on whether non-valuation techniques are able to estimate something akin to a 'conventional' demand curve for a market good and, in doing so, describe (marginal) WTP at different levels of 'provision'. The crucial insight is that it is this (marginal) price which—for national accounting purposes—should be multiplied by the quantity (or quality) of some good or service consumed.

In principle, there is no inconsistency with valuation using market prices in that all of the major techniques of non-market valuation can be shown in theory to estimate marginal values (see, for example, Freeman, 2003). However, an interim comment is that it remains worthwhile exploring further how many studies in the literature in practice present data on non-market values in a way that would be immediately usable in national accounts. That is, the findings reported in these valuation studies more often than not are geared towards the informational needs of economic appraisals (of discrete changes in provision of environmental goods or services brought about by 'new' projects and policies). And, moreover, reported practical WTP estimates might be (re-)interpreted as marginal values (of the type needed for greening the accounts) but only under certain assumptions.

It also remains true that moving away from the principle of basing valuation on market prices inevitably invites general anxiety about accuracy and judgement that must be brought to the accounting problem. This is summed up recently by Stiglitz *et al*. (2009) in stating that: "In standard national accounting practice, the normative issue of defining preferences is generally avoided through the assumption that observed prices reveal the true preferences of people. No explicit normative choice is therefore to be made by the statistician. But as soon as we recognise that market prices cannot be trusted, alternative imputed prices must be computed, whose values will strongly depend upon normative choices." (p75).

Related to this are concerns about 'accuracy' of the resulting data based on such judgements. This is not restricted solely to the accounting domain (see, for example, OECD, 2004). Nonetheless, given its apparent focal role in framing discussions about environmental valuation in the accounts, what crucially needs to be spelled out in clear and explicit terms is exactly what the hurdle of accuracy is in this context. That is, how

[8] This consumer surplus, in effect, corresponds to the value that is estimated by measuring the entire area under a demand curve (for that good or service) between some reference point – such as the 'zero' level of provision – and the current level of provision (Nordhaus, 2006). Typically, cost-benefit appraisal – perhaps the primary use for non-market valuation – would be interested in small changes in the provision of a service (or a change in the price of the service). What is being measured as part of the benefit of that change is the change in consumer surplus between the existing and new provision of the service (or the current and new price). This is not an issue about non-market valuation as such. Consumer surplus (or, more specifically changes in it) is also the locus of a (cost-benefit) appraisal of policy actions that increase the provision (or change the price) of a market good.

‘problem-free’ do data need to be in order to sensibly inform policy decisions within national accounts (and, by this token, has existing data been subject to the same yardstick). Of course, simply to ignore environmental valuation will contribute next to nothing in terms of overcoming apparent difficulties. The decisive question is whether (current or future) policy needs indicate that these difficulties are worth confronting. If the answer to this question is positive then circumscribing these activities in terms of *experimental* or adjunct accounts would seem an obvious response. This would have the clear advantage of offering scope for proper and critical reflection on genuine issues of substance that will confront the use of environmental valuation in this domain.

Benefits/ Damage vs Restoration/ Maintenance Costs (Revisited)

Before we move on to discuss environmental valuation more specifically, within the accounting setting, a broad methodological choice is often presented between those approaches to valuing the environment which seek, for example, to value the *damage* (or loss of future services) that arises from the depletion or degradation of natural assets and those methods which seek to value the *costs of replacing* or *restoring* these same assets to some (pre-existing) level. These competing standpoints have been evident in early debates about accounting for the value of air and water pollution as well as, more recently, in deliberations about ecosystem accounting.

In the ecosystem context, a number of related papers from the European Environment Agency (EEA) have proposed that the degradation of ecosystems can be valued in national accounts with reference to the costs that would need to be incurred if all of this loss was to be restored. One defence of this approach has drawn a parallel with traditional practice in national accounting for produced capital where depreciation (‘consumption of fixed capital’) is valued as the amount that it would cost to maintain this stock used in production (Weber, 2009). By analogy, it is argued, what is good for the treatment of produced capital in the national accounts is also good for the treatment of the degradation (i.e. depreciation) of ecosystem assets in extended accounts.

It can be argued, however, the analogy does not work completely. Thus, United Nations (2008) in setting out how depreciation is treated in the current System of National Accounts (SNA) states that: “Consumption of fixed capital is […] determined by […] the benefits that institutional units expect to derive in the future from using the asset in question over the remainder of its service life.” (p124). And the justification for the focus on replacement or maintenance cost is then spelled out as that: “Conceptually, market forces should ensure that the purchaser’s price of a new fixed asset is equivalent to the present value of the future benefits that can be derived from it.” (pp124-5). For an ecosystem asset whose services are predominately non-market in nature, this relationship is unlikely to hold. At the very least it is important that those who propose cost-based methods are able to justify this advocacy by empirical correspondence of costs to those magnitudes which we are actually interested in: that is, benefit or damage valuation.[9]

These debates may have less significance in that policies might eventually be designed so as to establish explicit prices. These interventions might include instruments such as 'payments for environment services' or emissions permits which can be traded freely (see, for example, Engel *et al*. 2008; Hanemann, 2010; Dietz and Fankhauser, 2010). Such emerging markets are mostly in their infancy but, in due course, may be more clearly relevant to the issue of greening the national accounts. That is, for example, in the case of emissions trading schemes, markets in carbon (and other polluting substances or activities subject to overall limits and trading schemes) and associated prices (that emerge from the actions of buyers and sellers judging the costs and the benefits of additional trades) might help to blunt debates such as those described above.

Similarly, prices in emerging markets in payments for ecosystem services might be used to guide valuation in that context. Nonetheless, as things stand, many of these transactions are currently subject to significant government intervention and/ or bear little relationship with benefits being provided (or opportunity costs being incurred) (Engel *et al*. 2008). Nor does it appear that is—as yet—any meta-study or data-base (in the public domain) of these transactions (although a variety of data are available through currently disparate sources).[10]

4. Valuing the Environment: A Brief Overview

Inevitably, extending the use of environmental valuation in national accounts will involve consideration of non-market valuation methods. It is the intention, therefore, of this section of the paper to provide a synopsis in this respect. We start by noting that the past two decades has seen a proliferation of methods, and applications of those methods, that have sought to uncover, in a variety of ways, the value of environmental impacts (and non-market impacts more generally).

Some of these approaches estimate original values by looking at *actual* behaviour: i.e. revealed preference (RP) methods. For example, in valuing recreational experiences, practitioners have utilized the fact that the (complementary) purchase of market goods (e.g. fuel, accommodation etc.) is typically required to access a recreational area (e.g. woodland, beaches etc.). Through so-called travel cost (TC) approaches, this insight can be used as a basis for valuation of these particular areas. Other approaches include those

[9] The unlikelihood that cost-based methods simply can substitute for estimates of benefits or (conversely) damages is unfortunate because the uncertainty around the former is likely to be relatively less in some environmental contexts (see, for example, Dietz and Fankhauser, 2010, for a discussion in the context of climate change).

[10] However, see, for example, the work of the Katoomba Group for practical discussion about what is known about markets and payments for ecosystem services (http://www.katoombagroup.org).

which focus on production decisions and, in doing so, look at the value of environmental services from the perspective of their affect on the costs and output of producers: that is, valuing the environment as a productive input (Hanley and Barbier, 2009).

Eliciting original values by looking at intended behaviour is the province of stated preference (SP) methods. This is an umbrella term for a range of survey-based methods that use constructed or hypothetical markets to elicit preferences for specified changes in provision of environmental services or natural assets. By far the most widely applied SP technique is the contingent valuation (CV) method.[11] However, in recent years, choice modelling has become increasingly popular in which In these variants, respondents are required to choose their most preferred out of a (possibly relatively large) set of alternative policy or provision options (see, for example, Hanley *et al*. 2001; Champ *et al*. 2003).[12]

4.1 Fit for Purpose and Beyond? Taking Stock of the Valuation Options

The case for (and against) environmental valuation methods has been well-rehearsed elsewhere in the environmental economics (and related) literature as well as policy documents (see, for example, Bateman *et al*. 2002, Champ *et al*. 2003, Haab and McConnell, 2002, Alberini and Kahn, 2006 and, in the context of ecosystem valuation, US EPA, 2009). A critical point made in all of these contributions is that developments of valuation methods have been accompanied by substantial critical reflection. Thus, increasing use of valuation has resulted in, on the one hand, ever greater sophistication in application and, on the other hand, ever-present scrutiny regarding validity and reliability.

Any legitimate critique must therefore engage with this evolving thinking (see, for example, Smith, 2006). Clearly, these debates are relevant to deciding on the efficacy of valuation methods that ultimately might be used to green the national accounts. If a particular category of valuation method is unsuitable (or at least problematic) for the purpose for which it was designed then it is unlikely that it will be any more fitting for broader use. At the same time, healthy scepticism with any approach must not be confounded with judging that method relative to a criterion of 'perfection' (Boyle, 2003).[13] Nor is it unrealistic to expect evolving debates to be resolved definitively before

[11] Carson's (forthcoming) bibliography of published and unpublished CV studies contains over 5,500 studies, undertaken in just under 100 countries.

[12] A number of studies combine RP and SP approaches in order to enhance the respective strengths of these data and minimising limitations (see, for example, Adamowicz *et al*. 1994 and Kling, 1997).

[13] Sugden (2005) notes two opposing perspectives on this debate about SP approaches. Arguably the position that is most prominent within environmental economics is that empirical findings largely support the validity and reliability of e.g. CV estimates – from well-executed studies – of the value of non-market goods. This emphasis on good studies is important and, in turn, has led increasingly to informal or perhaps even formal requirements for practitioners to follow, in some way, guidelines for best practice. A rather different perspective has sought to reconcile this with emerging evidence on preference anomalies from behavioural economics. List (2005), however, presents evidence that preference anomalies are

deciding which valuation methods are admissible as tools to green national accounts and which are not.

Notable reviews of this issue have proposed a hierarchy of valuation methods that might be used to green national accounts giving pre-eminence to market-based approaches and those approaches based on market behaviour. This view seems to be evident in the conclusions of, for example, in Nordhaus and Kokkelenberg (1999) as well as, more recently, Stiglitz *et al*. (2009). This gives primacy, it would appear, to approaches based on RP and so aspects of the environment that can be valued in these ways. By this token, it downgrades the role that might be played by SP approaches and those aspects of the environment which can only be valued using those methods. The downside of this is that "… revealed WTP is often an incomplete measure of a resource's total value" (Boyd and Krupnick, 2009, p38). This well may be a high price to pay where, for example, non-use is an important component of total value or where interest is in use values which have not been revealed by the market.

This emphasis on valuation methods based on market behaviour also raises the possibility of a paradox in that those methods which might be judged to be the most robust are those most likely to uncover values which are already in the accounts. Without conducting a detailed audit it is not really possible to say much more about the extent of this. However, RP methods are unlikely merely to just tease out values already in the accounts. But given the advance in SP methods, the conclusion of Nordhaus and Kokkelenberg (1999) seems even-handed in expressing a preference revealed preference data in greening the national accounts while not ruling out SP approaches given the work being devoted to this area.

4.2 The Inevitability (and Reliability) of Value Transfer

Advances in methods to generate primary data on the value of environmental goods and services have been a striking feature of modern benefit assessment. However, routine use of valuation in that context as well as in the national accounting domain arguably will rely heavily on using secondary data: so-called value or benefits transfer. In the policy appraisal context, this involves taking the findings of original studies of the value of environmental goods estimated in an original study and using these data (perhaps after some adjustment) to value benefits that arise when a new policy is to be implemented. In the national accounting context, this suggests a further challenge. The 'new policy'—to which unit values from original studies might be transferred—is, for example, the total (i.e. national) change in some natural asset.

likely to be a matter of degree and, in part, are determined by experience and familiarity which is often lacking when environmental goods are not traded directly. This might urge some caution to using judging the merits of SP approaches in greening national accounts.

This is a crucial issue as it is inconceivable that original valuation studies can be carried out for every service and every natural asset that would be needed for a comprehensive green national account. Even prioritising particular natural assets (perhaps on the basis of policy relevance and empirical significance) undoubtedly would represent too formidable a practical challenge. Value transfer thus represents a means of extending the empirical record in a pragmatic manner. Indeed, the holy grail of this approach is the consolidation of original data on non-market values in emerging transfer databases where values can be taken 'off the shelf' and applied to new policies and projects as needed (such as the web-based, Environmental Valuation Reference Inventory or, EVRI: see, www.evri.ca). The G8 initiated TEEB (The Economics of Ecosystems and Biodiversity) Review represents a more recent effort to assemble a database of (potentially) transferable values for a wide range of ecosystem services (www.teebweb.org).

Invariably, however, transferring values in this way introduces an additional dimension of uncertainty into any valuation exercise in that it entails further assumptions and judgements to those contained in original studies. As a result, the value transfer approach is the subject of a rapidly growing literature (see, for example, Boyle and Bergstrom, 1999, Desvousges *et al*. 1998, Navrud and Ready, 2007). Indeed, a number of contributions have sought to test the accuracy of transfer exercises. For example, Brouwer and Bateman (2005) investigate the temporal reliability of transfer values: that is, the question they ask is at what point 'older' vintages of original studies are likely to become too unreliable to transfer to more contemporary environmental changes. Distilling an overall message from these tests, however, is not straightforward. In some cases transfer error ranges are small while in other cases these ranges are extremely large indeed (see, for a review, Rosenberger and Loomis, 2003; Narvud and Ready, 2007).

Understanding when benefits transfer works and when it does not (as well as the size of possible errors it entails) is clearly crucial. That is, for example, are these on a par or in excess of errors likely to be associated with other physical elements of the green accounting problem? The challenges (related or distinct) do not end here. There may be difficulties in translating the findings of previous studies—which will often measure the value of discrete or distinct changes—into marginal prices which can be assigned to quantity or quality changes to be valued in a transfer exercise (Navrud and Ready, 2007). (This is clearly related to our earlier discussion of the consistency of practical valuation evidence with marginal values for accounting purposes.) Moreover, the bedrock of good practice remains whether there is an abundance of good quality original studies to facilitate a transfer exercise. Judging quality, in a rigorous way, is itself not without complications. For example, should what is deemed to be 'allowable' in the empirical record be restricted to peer-reviewed studies or is a broader sweep permissible that covers findings in the 'grey' (or unpublished) literature?

Regardless of the way in which the empirical record is defined, another problem that any transfer (and its conduct) faces is the lack of standardised approaches or ground-rules at the stage of conducting original valuation studies (despite emerging official 'guidelines' and 'manuals'). Even a relatively brief review of studies, in a limited number of environmental contexts, indicates the sheer diversity of the approaches that have been pitched at the valuation challenge. This heterogeneity is clearly a marked contrast to the national accounting domain. Of course, this could be said simply to be an indication of a (sub-) discipline that (relatively speaking) is still in its formative years. Whatever the reason, this situation does hamper the task of making sense of the information contained in the mass of studies that currently make up the empirical record. Increasingly, structured and statistical approaches—known as meta-studies or meta-analyses—have been brought to bear on this challenge.

Meta-Analysis: Making Sense of the Empirical Record

Stanley (2001) defines meta-analysis as a statistical technique where "… the dependent variable is a summary statistic, […], drawn from each study, while the independent variables may include characteristics of the method, design and data used in these studies" (pp131-2). For the most part, meta-studies—in the context of environmental valuation—use WTP values as these summary statistics (Rosenberger and Loomis, 2003). Researchers, in this context, seek to explain determinants of WTP across studies with reference to characteristics of the environmental asset in the location being studied and the households or individuals that benefit from provision (Navrud and Ready, 2007). In addition, it is possible to gain insights about the way in which differences in method (across original studies) influence WTP values.[14]

A growing number of meta-assessments have been carried out in respect of urban pollution, recreation, ecological services of wetlands, value of statistical life, noise and congestion. A strength of the approach, in general, is that these take advantage of the collective wisdom embodied within a wide range of studies. On the one hand, this simply offers a systematic way to understand study-to-study variation in valuation estimates for say wetland services. On the other hand, more robust transferable values may emerge from this analysis (although, see Nelson and Kennedy, 2009 for a discussion about best practice in using meta-approaches in this way).

Brander *et al*. (2008), in a study conducted for the European Environmental Agency, provides a meta-analysis to establish transferable values for wetland services.

[14] That is, some of these studies may have relied on revealed preference techniques to uncover WTP values. Others may have used stated preference methods and within this category the potential for variety is similarly large with some studies using choice experiments and other using some variant of contingent valuation. Furthermore, these values might have been elicited in different ways using, for example, distinct payment vehicles and time frames over which payments are made, elicitation formats (i.e. the way in which the WTP question was asked) and survey modes (i.e. in-person interviews or some alternative).

Interestingly, this study aims to scale-up these unit values so as to evaluate changes in wetland value at a relatively high level of aggregation (e.g. the national or regional level across Europe). The analysis carried out in that study itself makes use of a wide range of original research and, indeed, identifies from a database of almost 400 valuation estimates obtained from a little less than 170 studies. However, only those studies which sought to value types of wetland that occur in Europe were included in the analysis itself.

The first part of this study sought to provide a statistical insight into the determinants of wetland value. To facilitate this, a meta-regression was estimated which had, as its dependent variable, a measure of the value of (or WTP for) a hectare of wetland.[15] A number of explanatory variables were found to be significant WTP determinants including: whether the wetland was a peatbog (which seem to command lower values, other things being equal); whether the wetland provided flood control and storm buffering services (which command higher values); and, notably, the size of the wetland (in that larger wetlands appear to generate smaller—per hectare—values).

Understanding the sources of difference between values found across distinct original studies is clearly interesting. However, a second objective of this work was come up with defensible estimates that can be generalised across many such areas of wetland across Europe. The basis for these values was the statistical analysis as previously described and information about (50,533) individual wetland sites across Europe (and e.g. their vicinity to populations etc.). Some of these findings are presented in Table 1 which includes (in the upper part of the table) those countries with more than 250,000 hectares of wetland in total and (in the lower part of the table) a breakdown by wetland type (for Europe as a whole). So, for example, the findings indicate that peatbogs are the most abundant wetland type but command the lowest per hectare value.

Table 1: Proposed European Wetland Valuation Database

	Number of wetlands	*Wetland area (ha)*	*Mean value/ per hectare/ year*
Country			
Finland	14,140	1,971,961	224
France	1,419	358,163	5,693
Germany	1,391	418,945	4,353
Ireland	2,173	1,210,044	676
Netherlands	273	269,753	7,871
Romania	1,532	384,611	2,615

[15] In principle, this describes what the (affected) population of the country in which the wetland is located would pay for a unit change in area of that wetland) and standardised in a number of ways (e.g. currency, purchasing power and prices)

Sweden	20,242	2,729,131	263
United Kingdom	2,119	753,691	2,480
Wetland type (all Europe)			
Inland marshes	8,842	1,159,153	4,129
Peatbogs	38,644	6,712,309	214
Salt marshes	1,621	306,754	5,734
Intertidal mudflats	1,180	995,094	4,112
Salines	246	72,467	5,475

Source: adapted from Brander *et al*. (2008).

To what extent are these estimated values reliable? The authors themselves outline a number of caveats. These include the fact that (as currently estimated), these values are relevant to evaluating changes which involve say a decrease in wetland area perhaps through land conversion rather than more intermediate changes in wetland quality.

Other critical issues characterise any effort to establish values that might be generalised across wetland types in this way. Marginal values might differ across otherwise similar ecosystems. For example, more is significantly better when the reference point is a relatively small ecosystem area (Pascal *et al*. 2009).[16] This issue is not irrelevant to valuing goods currently included in the existing national accounts. But for those goods changes in marginal values should be reflected in changes in market prices. So the relevant question is whether non-market valuation studies, in e.g. the ecosystem context, are able to reflect this insight (or whether we must resort to simplifying assumptions such as assuming that unit values are constant). In the Brander *et al*. study, it is notable that this consideration is captured in the (aforementioned) finding that (unit) value varies with the size of a wetland (see, in addition, Barbier *et al*. 2008).[17]

Furthermore—ecologically speaking—'similar' areas could be in practice highly heterogeneous in terms of their ecological productivity (Barbier, 2007). In other words, geography matters in the sense that different locations across the land area covered by the ecosystem might provide different services of differing values. The issue then, in this case, is whether valuation findings do justice to this heterogeneity and what are the geographical limits beyond which otherwise transferable values should not be generalised.

[16] There are a number of reasons for this. Observed relationships, for example, based on distance-decay complicate matters in that some evidence exists for the proposition that WTP values are likely to decline the further away an individual (or a household) is from an ecosystem.

[17] This study examined the non-linear relationship between mangrove area and coastal protection (from storm and similar damage). Specifically, these authors find that the *marginal* benefit that additional mangrove conveys—in terms of attenuating waves—declines the larger the initial mangrove area is.

5. Towards Practical Implementation of Environmental Values in the National Accounts

Environmental valuation has been an active area of research and practice over a period of at least two decades. Given this expanding legacy, it is worth asking what is within the 'art of the possible' if we sought to use this evidence base to green the national accounts 'tomorrow'. Inevitably, this would be rudimentary and would pre-empt a great deal of methodological discussion. However, with such caveats not forgotten, Table 2 outlines a tentative hierarchy of the readiness—in terms of implementation in (experimental) national accounts—of a number of categories of either environmental degradation or services.

Before proceeding to a discussion of specific entries, a number of comments are worth making. First, the hierarchy presented is intended as an interim assessment only and so should be considered in that light. In addition, the identification of aspects of damage or services is not ideal and there is likely to be some overlap between categories. Secondly, the elaboration of categories is not exhaustive and, for example, in the case of ecosystems, in particular, these are both numerous and diverse and some will be better understood (in terms of valuation) than others. The table illustrates crudely the case of forests although within this category it does not make sense to speak of forests where there are different forest types (e.g. tropical, temperate and so on). However, what is indicated in the table is offered as a way to describe the broad issues about the readiness of ecosystem valuation for incorporation in the accounts.

In terms of this 'readiness', the categories of damage or service are themselves divided into three groups or columns within the table.

The first group consists of those items for which it can be stated with reasonable confidence that not only can valuation can be readily implemented but also it can be relatively widely applied. What this last point means is that current knowledge and valuation experience is substantial and has been undertaken in larger-scale settings (rather than exclusively in small 'micro-' studies) and across a relatively large range of countries. It also means that the requisite physical data are largely in place as well.

The second grouping refers to those categories which are not quite ready for wider implementation but where there is an institutional or policy context for wider scale valuation to be undertaken within a number of years or so. One example of this is the likely and increasing need to quantify, in economic terms, the value of impacts of various European Union environmental directives. This, in turn, should lead to a corresponding accumulation of experience in using valuation: that is, given the growing need to appraise policy actions that aid compliance. In such circumstances, further investigation of feasibility is warranted in future accounting work.

Table 2: Preliminary Assessment of Readiness of Implementing Valuation in the Accounts

Damage/ Service	*Valuation can be readily and widely implemented given current knowledge and data availability*	*Valuation is feasible/ used but not ready for wider implementation. Policy context likely to need in next 5-10 years*	*Valuation largely experimental and widespread use is still a long way off*
Air pollution (Health)			
SO_2	X		
NO_X	X		
PM10	X		
PM2.5		X	
O_3 (ground-level)	X		
VOCs		X	
NH_3		X	
CO		X	
Lead		X	
Noise			
Annoyance		X	
Health			X
Water pollution			
Health		X	
Other impacts		X	
Climate change			
CO_2		X	
Other GHGs		X	
Land			
Soil erosion (on-site)		X	
Soil erosion (off-site)			X
Ecosystems (e.g. Forests)			
Provisioning: e.g.			
Food/ raw materials	X		
Water flow regulation			X
Medicinal			X
Carbon sequestration		X	
Erosion control			X
Habitat/ Supporting: e.g.			
Species/ biodiversity/ gene protection/ support			X
Nutrient cycling			X
Cultural: e.g.			
Recreation	X		
Aesthetic			X
Culture/ spiritual			X

Notes: SO_2 (Sulphur dioxide); NO_X (Nitrogen oxides); O_3 (Ozone); PM (Particulate matter, PM2.5 particles less than 2.5 microns in diameter; PM10, particles less than 10 microns in diameter); VOCs (Volatile organic compounds); NH_3 (Ammonia); CO (Carbon monoxide); CO_2 (Carbon dioxide); GHGs (Greenhouse gases).

The final grouping describes those elements of damage or service for which valuation is in all likelihood even further away in terms of readiness for implementation. In these cases, it is likely all that can be done is to maintain a watching brief on progress in future applications of valuation to these domains.

With regards to the (interim and tentative) demarcation in Table 2, it will be evident that a conservative approach has been taken. That is, only a handful of categories are indicated as being ready for widespread implementation. These largely correspond to what Hunt and Ferguson (2010), among others, refer to as the 'classical' air pollutants. The narrow range of items included in this grouping is, in large part, due to the conservative approach we deliberately have taken.

Retaining the air pollution example, in the case of 'less ready' pollutants, it might be that it is the physical evidence base that needs development. Examples here could include fine particles (PM2.5) where the health effects associated with exposure are now comparatively well understood (see, for example, WHO, 2006). However, what might be lacking is the monitoring of that particular categorisation of particulate. This is likely to be a question of degree in that some countries are likely to have well-advanced monitoring (of e.g. concentrations). Indeed, this is one area where advancements in knowledge and data can be relatively rapid (and the list of air pollutants which fall within the domain of *routine* economic assessment is increasing all of the time).

Valuation is also well-understood in the case of some ecosystem services notably for provisioning services such as food and raw materials as well as recreation arguably. In the case of the former, these goods are close to market and often may have then a commercial parallel on which to base valuation. In the case of the latter, valuation is long established (either using revealed or stated preference studies). The issue might be—respectively—the wider availability of physical data or the ability to scale-up or transfer the existing evidence base to all areas providing recreational opportunities.

In the case of carbon dioxide and other greenhouse gases (GHGs), national inventories of physical emissions are relatively common and there is improving information in many countries about (net) carbon sequestration and storage in certain types of biomass. Likewise, estimates of the social cost of carbon (SCC) have been available for many years (see, for example, the meta-study of Tol, 2005). Moreover, these values typically are expressed as damage per tonne of GHG and, on the face of it, 'easily' can be applied irrespective of location of emission (or sequestration) source. The issue here is, of course, the well-known uncertainties about the value that the SCC could take. Thus, the possible ranges (for these per tonne values) are large and are crucially dependent on assumptions about the future path of global GHG emissions (e.g. the SCC is higher if a 'business as usual' path is assumed rather than 'strong, early action' by all nations) (Stern, 2007). However, a number of countries are moving towards establishing (or have established) official values for the SCC (see, for example, DECC, 2009).[18]

[18] The UK approach outlined in DECC (2009) is based on an explicit judgement that this value should be consistent with that government's desired target for global GHG concentrations in the atmosphere. One element of this is to

A large number of valuation studies have looked at the issue of water valuation from the perspective of recreation, aesthetic values, health status and non-use. Thus, the empirical record is large. It is also diverse and, at the same time, clustered around particular geographical contexts (e.g. studies mostly undertaken in a handful of countries). For non-health categories of impact, the scope for value transfer across geographical boundaries may be small (that is, if the original context where values were estimated and the context where we would like to transfer are too different). In the case of health impacts of exposure to contaminated and polluted water, valuation of a number of crucial endpoints (notably gastro-intestinal illnesses) do exist (Hunt and Ferguson, 2010). Where data are available—on the link between exposure to polluted water and incidence of illnesses—relevant health values for (water-related) morbidity can be applied in a similar way to the air pollution case.

The water context is also one where policy developments could advance the wider application of valuation. One prominent example is the European Union Water Framework Directive (WFD). This legislation requires all water bodies achieve 'good ecological status' (by 2015) and has been accompanied by measurement effort directed towards extending inventories to assess the number of water bodies of particular types (within water management catchment areas) on basis of whether these have 'good', 'moderate', 'poor' or 'bad' status on ecological criteria. One UK study of the WFD looked at household willingness to pay to move to 'good ecological status' using a stated preference survey (Metcalfe *et al*. 2009). This WTP estimate itself can be broken down into the value of improvements of local and national water bodies (with the latter likely to be mostly based on 'non-use'). Practical issues include translating these values into what is happening in individual water catchment areas (there are 11 of these in England & Wales).

Another development, again in the European context, is the establishment of an evidence base on urban noise arising from the requirements of EU's Environmental Noise Directive. For example, in the UK, such maps have been (and are being) developed in 23 agglomerations around England. This noise is mapped within bounds (0-54.9dB; 55-59.9dB; 60-64.9dB; 65-69.9dB; 70-74.9dB; 75+dB). Exposure to noise of course also needs to be combined with maps of e.g. population densities and so on to assess numbers of people affected by noise within these bands. In principle, these estimates of exposed populations can be valued using available studies such as those

establish an official 'shadow price' based on an aspiration for a global target for concentrations (e.g. 450-550ppm) and then to estimate the marginal abatement costs for the UK associated with contributing to reaching that target. DECC (2009) indicates that this value is £52 a tonne of carbon dioxide equivalent in 2010. In fact, the UK approach is a little more complicated than this in that it distinguishes between non-traded and traded carbon. The latter refers to that carbon which is transacted via the European Emissions Trading Scheme. Valuation here is based around what is judged to be the 'likely' trading price. In 2010 this was reckoned to be £21/tCO_2e. In addition, low and high values for non-traded and are also given (and these are respectively £26/ £78 and £12/£27). Shadow prices for e.g. emissions from 2030 onwards are assumed to converge on the assumption that all sources of carbon will be subject to trading.

reviewed for the European context by Navrud (2004). UK transport appraisal guidelines, for example, set down official marginal values for 1dB changes in noise from different levels: e.g. Household WTP to reduce noise from 46dB to 45dB, 61dB to 60dB and 82dB to 81dB is set at £8.4, £48 and £98 respectively. These values themselves are derived from a study of the relationship between rail and road related noise and property prices by Day *et al*. (2007) and Nellthorp *et al*. (2005).[19]

With regards to the final column in Table 2, these categories of damage or service are those where the current preliminary assessment suggests are far from being available for widespread use. Typically, these involve elements where either few studies so far exist, the valuation issues are complex or both problems exist. Not surprisingly, many of these items in the final column are ecosystem services. However, this preliminary designation (in Table 2) carries with it an important caveat. The significant attention currently being directed towards understanding the value of ecosystem services indicates that there exists scope for substantial advance in the near future. In large part, a lot of recent and on-going discussion—about ecosystem services—has been geared understandably to scientific assessments. Increasingly, however, economic analysis has also become prominent (see, for example, Bateman *et al*. 2010). In particular, much can be learned in the very near future about the scope for incorporating ecosystem services within national accounts through the lessons learned in emerging large-scale assessments such as TEEB and the UK National Ecosystems Assessment.

In order to add some further detail to this brief and preliminary assessment of the state of the existing evidence base, we discuss further—in what remains of this paper—two prominent areas of valuation activity: human health and ecosystems. The reason for this dual focus is that arguably the most experience and understanding has been gathered in respect of the former. Thus, relatively speaking, health valuation is a mature area of the discipline. To the extent that practitioners know about where valuation works (and where problems exist) it is in this context. By contrast, progress in ecosystem valuation has been more recent although (as mentioned) significant effort is currently being directed to this topic. The indications are that developments in the empirical record are noteworthy but, equally, so are apparent methodological challenges.

5.1 Progress in Valuing Human Health

Environmental degradation affects human health in a number of ways. First, by increasing environmental risks to lives, it may result in premature mortality. Second, it may result in a morbidity cost arising from, for example, the harm caused to the health

[19] These values correspond to the annoyance that households endure as a result of exposure to road and rail related noise levels. Less is known about the implications of exposure to high levels of noise on health although as Hunt and Ferguson (2010) note there is likely to be some (implicit) degree to which values based on 'annoyance' capture some of this health aspect.

of those living with a disease such as a respiratory illness or increased incidence of minor symptoms among the relatively healthy.

These health impacts relate to the national accounts in a number of important ways. There are a variety of impacts related to the product accounts. Among these are 'out-of-pocket' expenses when people are ill as a result of e.g. exposure to air pollution and arising from purchases of pharmaceutical treatments which address symptoms such as headaches, fever and other flu-like symptoms which some air pollutants are thought to cause. Further health care expenditures are not made by individuals alone, but by social administrators and ultimately the taxpayer.[20] To the extent that workdays are lost as a result of illness, there is lost market output. All of these items are indirectly included or are implicitly absent from the existing product accounts. More generally, there may be costs of illness over and above what is reflected in such magnitudes (that is, related to the psychological cost of being ill). But for many of these illnesses, these impacts relate to effects on current production of broader wellbeing over the accounting period. For mortality (or indeed some chronic exposure or illnesses), however, impacts are an issue for both the product and wealth accounts. That is, the impact is on future production or wellbeing. Thus, the impact is on human capital broadly construed to include both the market and non-market value of being healthy.

Valuing mortality and morbidity effects in monetary terms can provide extremely useful information for policy, not least in providing strong indications that the total burden of environmental disease is substantial in economic terms. Over the last 30 years, stated preference studies, together with revealed preference methods, have been used extensively to calculate both individual WTP to secure reductions in mortality risks of death arising from a policy and WTP to avoid particular health outcomes involving morbidity (see, for example, the pioneering work of Viscusi, 1992). Focusing, for the moment, on the valuation of mortality risks, these are normally expressed in terms of the value of statistical life (VOSL). This implies dividing the WTP for a given risk reduction by that risk reduction to obtain the VOSL (see, for example, Bolt *et al*. 2005; Krupnick, 2004). Various countries adopt single (or ranges of) values for the VOSL and then use them in policy appraisal. The US Environmental Protection Agency (EPA), for example, has used a VOSL range of $0.6 million to $13.5 million, with an average of $4.8 million (1990 US$) based on an assessment of the existing US literature (Robinson, 2007). However, whilst there is a very large body of research on health values for North America and Europe, there has been until very recently a dearth of evidence for developing countries.

This makes the gathering pace of recent efforts to expand the VOSL dataset beyond these countries (that traditionally have dominated the empirical record) all the more interesting. Recent hedonic wage studies have resulted in estimated VOSLs of $235,000

[20] This is the familiar distinction between methods which look at individual decisions – e.g. avertive expenditure approaches – and costs-of-illness approaches which look at the expenditures that arise from broader social decisions.

to $325,000 in Mexico (Hammitt and Ibarrarán, 2006), $375,000 in India (Madheswaran, 2007) and $790,000 to $2.41million in Poland (Giergiczny, 2008). A CV study based on reduced air pollution in Brazil produced a wide-ranging VOSL estimate of $770,000 to $6.1 million (Arigoni Ortiz *et al*. 2009), whilst a rural Thailand CV study produced a VOSL estimate of $250,000 (Gibson *et al*. 2007).

In a series of studies in China, a shared focus has been the potential aggregate benefits of reduced air pollution; primarily the avoidance of PM10 related deaths and associated diseases. Annual health damage costs equivalent to 6.5% of Beijing's GDP between 2000 and 2004 (Zhang *et al*. 2007) and damages of $29 billion in 2004 across 111 Chinese cities (Zhang *et al*. 2008) have been estimated. In addition, it has been calculated that potential corresponding health benefits from the implementation of low carbon energy scenarios could amount to $1.5 billion for Shanghai in 2010 (Chen *et al*. 2007). These studies contain no new VOSL data, instead relying on the transfer of the same 2001 Chinese CV study, which gave a figure of $44k, combined with internationally recognised dose-response functions. However, this value is at least in line with a recent Chinese study of the wage premium that workers in relatively risky occupations command. This estimated VOSL to be in the range $30,000 to $100,000 (Guo and Hammitt, 2009).

The basic building blocks for these aggregate calculations involve a combination of physical data on pollution concentrations and dose (or exposure) response relationships (see Table 3). In some countries, the concentration record is likely to be based on a mix of monitoring sites at various locations and mapping based on models of dispersion from emission sources. The reliability of these data will be a question of degree and notably will vary across pollutant (WHO, 2005).[21]

[21] Of course, there will be uncertainties in these data. For example, Defra (2007) reports on the uncertainties arising from that most basic unit of information: emissions inventories. These range, in this UK case, from relatively small uncertainties for pollutants such as sulphur dioxide and nitrogen oxides (±3% and ±8% respectively) to moderate in the case of e.g. PM10, carbon monoxide and lead (-20 to +50%) and large in the case of certain types of hydrocarbons (-70 to +200%).

Table 3: Incorporating Health Values in the Accounts

	Physical data	Impact-pathway	Valuation	Comments
Example: Air pollution	– National inventories on emissions, concentrations – Augmented where data lacking by international organisations modelling of latter (e.g. World Bank, WHO etc.)	– Dose-response functions eg. World Health Organisation (such as WHO, 2005 and related sources) – Some national information may be available for certain countries – E.g. US EPA, UK Air Quality Management Strategy) but most will tend to be drawn from 'international' evidence	– National studies, official values (where adopted) used for appraisal – Cross-country reviews of valuation studies for mortality and morbidity – E.g. OECD on-going work summarised in Lindhjem *et al*. (2010)/ Hunt and Ferguson (2010)	– Where national studies of values not available, international transfer might be an option bit clearly not ideal – Such judgements about transfers implicit, however, in many applications of dose-response relationships – Most comprehensive data for all elements (physical/ impact-pathway values) for PM, O_3, SO_2, and NO_2

In the case of the appraisal of a new proposal to tackle pollution, what is of interest is the change in pollution concentrations from some current level to the proposed level. For accounting purposes, however, what is arguably relevant is the damage that arises from the entire exposure to concentrations (although it might be appropriate to deduct from this any natural background levels of e.g. dust particles in measures of PM concentrations for which any 'policy' can do little to influence).[22] The dose-response function itself is based, in turn, on scientific evidence about the relationship between

[22] In addition, given that certain pollutants can travel large distances on wind currents, ambient concentrations may include pollutants from emission source which originate in other countries.

human exposure to a pollutant and its adverse consequences for health outcomes. In the case of PM, for example, exposure has been shown to be associated with decreased life expectancy, lower respiratory symptoms and decreased lung function in both children and adults as well as COPD (chronic obstructive pulmonary disorder) in the latter (WHO, 2005).

Examples of dose-response functions exist for a reasonably large range of health end-points particularly for sulphur dioxide, nitrogen dioxide, (ground-level) ozone and particulate matter (either fine particles such as PM2.5 or broader measures such as PM10). The range of these functions is, however, constantly evolving as 'novel' pollutants come within the ambit of this impact-pathway measurement and the specific role of individual pollutants becomes better understood (that is, for example, what is the contribution of nitrogen oxides to health outcomes over and above their role as precursors to ozone formation and nitrate particles in PM). Needless to say, at any point in time there are uncertainties in these functions. Indeed, Hunt and Ferguson (2010) cite evidence that uncertainties inherent in dose-response functions could be on a par with the uncertainty reckoned to be associated with monetary values commonly elicited in SP studies of health outcomes. This does not mean of course that this dose-response information is not valuable but it is worth acknowledging that valuation is not the only source of uncertainty in evaluating (as here) pollution impacts.

Valuing physical end-points—i.e. health outcomes—is the final piece in this accounting puzzle. A recent review by Hunt and Ferguson (2010) indicates that a large number of studies have been conducted into a correspondingly large number of morbidity-related health end-points (ranging from minor symptoms such as coughing through to chronic bronchitis in adults, hospital admissions, types of cancer and premature mortality). Some care must be taken about the potential for overlap between these health categories (and thereby double-counting). Moreover, the authors of that review indicate that a significant proportion of the likely costs of adverse health arising from exposure to air pollution might be captured by focusing on a handful of these end-points (particularly, premature mortality and hospital admissions for example). Interestingly, on Hunt and Ferguson's reckoning, these are also typically the health end-points where the quantity and quality of valuation information is relatively good.

At the research frontier, a number of questions have concerned practitioners. Recent research, with practical implications for accounting, has shown that the age of the respondent who is valuing the risk matters (see Krupnick, 2007, for a review). Environmental health is associated with both immediate and future risk. Strictly speaking, the damage caused by a general level of exposure to e.g. PM10 should be evaluated in terms of the (lower than immediate) valuations associated with younger people's valuations of future risks plus older persons' valuation of that risk as an immediate risk. An alternative approach that takes into account the age of persons saved by a particular policy, and that may be able to capture the shorter life expectancy phenomenon, is the value of a statistical life-year (VSLY) (see, for example, Hammitt, 2007).[23]

A rather distinct age-related issue is that some environmental risks fall disproportionately on the very young. One implication is that adults' (or parents') valuations of the risks on behalf of children need to be estimated with the finding in some studies so far of child/ adult WTP ratios of 2.8 in the UK and 1.6 in the Czech Republic (Bateman *et al*. 2009). A recent study carried out in Italy found tentative support for the notion that VOSLs were higher for children (Alberini *et al*. 2009). Efforts also have been made to yield valuations from actual behaviour by observing to what extent actions are taken to avoid potential (Mansfield *et al*. 2006) or mitigate existing health problems (Hanemann and Brandt, 2006).

5.2 Progress in Valuing Ecosystem Services

Ecosystem services refer to the wide range of benefits that people derive from the multitude of resources and processes that are supplied by natural ecosystems (Daily, 1997). The next great challenge then for valuation practitioners is getting to a similar condition of understanding for ecosystems as arguably exists in the case of human health. This does not mean that little work has achieved in this respect to date.[24] On the contrary, a great many studies exist (see, for a comprehensive review, Pascal *et al*. 2009). Progress has been more limited in estimating the non-use values of forests and a lively debate surrounds the value of genetic material in forests for pharmaceutical research (see, for example, Simpson, 2007; Costello and Ward, 2006).

Table 4 illustrates some of the issues in a little more detail for an expanded number of ecosystem services. Using two of the reviews that have contributed to the TEEB assessment, (while not exhaustive) the table gives a rough assessment of the number of valuation studies conducted for different ecosystem services (for forests and wetlands) in the past twenty or so years. For some of these services, a relatively large number of studies exist: e.g. food, raw materials and recreation. For many other categories, very few studies exist at all. Moreover, for some of these categories (such as climate regulation), studies exist but the resulting values are based on cost-based methodologies (e.g. replacement costs) rather than conveying a signal about the value of the benefit provided by these services. The geographical distribution (as illustrated for the case of forest ecosystems) of existing studies is also an issue with some areas of the world having a paucity of studies.

[23] The VSLY is calculated by dividing the value of a statistical life by discounted remaining life expectancy, thereby converting VOSL estimates into a value per life-year saved. VSLY can then be multiplied by life-years saved, i.e. the remaining life expectancy, to value the statistical lives of persons of different ages.

[24] Fisher *et al*. (2009) present evidence, over the last 10 to 15 years, of a substantial (in the region of at least one order of magnitude) increase in the number of published papers each year which use the term 'ecosystem services' (or similar).

Table 4: Prevalence of Values for Ecosystems Services

		Ecosystem			
		Forests			**Wetlands**
Broad category/ Specific service		*Number of studies*	*Methods used*	*Geographical coverage*	*Number of studies*
Provisioning	Raw materials	28	Mostly market price or cost-based		7
	Food	26	Mostly RP, PF and market price	Mostly South America, some Asia	24
	Climate regulation	13	Mostly cost based		2
	Water	9	Mostly PF and cost-based		6
	Moderation extreme events	8	Mostly cost-based		6
	Medicine	6			1
	Soil formation	5			4
	Water purification	4			8
	Erosion prevention	4			3
	Air quality regulation	4			…
	Biological control	3			…
	Water flow regulation	2			5
	Genetic	1			…
	Pollination	1			…
Habitat/ Supporting	Gene pool/ species protection	10	Mix of SP and RP		3
	Biodiversity/ species support	4			2

	Nutrient cycling	…			2
Cultural	Recreation	59	Mix of SP and RP	Mostly North America & Europe	26
	Aesthetic	10	As above	As above	5
	Cultural	…			3
	Spiritual	8			1

Notes: Count of studies based on Pascal *et al*. (2010) for studies undertaken since 1990; Indications of geographical location of forest studies based on Kontoleon and Mullan (2008). These will not correspond directly to the more recent review in Pascal *et al*.
Source: Based on: Pascal *et al*. 2010; Kontoleon and Mullan 2008).

Before proceeding to a discussion of issues that might arise in trying to translate this work on ecosystem valuation to the accounting domain, an example of what is missed by 'ignoring' the (non-market) value of ecosystem services is provided in Table 5. Barbier (2007) estimates the area values for coastal mangroves in Thailand. The table indicates that those goods or services from mangroves which 'straightforwardly' command market prices (forest products and fisheries) account for less than 13 per cent of the total value of a unit of land in this use. The vast majority of mangrove value arises from the storm protection services provided by mangroves.[25]

Table 5: Land Use Values Per Hectare in Thailand, 1996-2004, US$

	(Net Present) Value per Hectare
Net income from collected forest products	$484 – $584
Fishery (via link to habitat extent)	$708 – $987
Storm protection service	$8,966 – $10,821
Total	$10,158 – $12,392

Source: Adapted from Barbier (2007).

The data in Table 5 are present values so, in this sense, represent the wealth of land under mangrove 'production' or the change in asset value that might arise when land is switched (permanently) from mangrove production to some other use. The use of 'land' as a unit of account here is both useful and convenient (see, in addition, Barbier, 2009, for a discussion of the conceptual merits of using ecological land area as the basic unit of account). Not all ecosystem valuation studies report values in this way (although an

[25] This is calculated as the reduction in risk of damage (and its severity) arising from mangroves (relative to their absence).

increasing number do). Nor is it necessarily straightforward to translate such services in readily understandable—and measurable—units such as land area.

A distinct, but no less important, focus is offered by Boyd (2008). He argues that the emphasis on ecosystem *valuation* in environmental economics has failed to distinguish adequately between price or (unit) value and the quantity of (some change in) the provision of e.g. ecosystem services. And while obscuring these two critical dimensions of total value matters relatively little for the specific policy uses that environmental economists typically have sought to address (i.e. valuing discrete policy changes), Boyd further asserts that the distinction is crucial in thinking about valuation in the setting of green national accounts. This brings what Boyd terms the measurement of 'ecological quantities' to the fore. Essentially, these quantities are those biophysical goods and services which are of interest as inputs to economic production and welfare. These inputs may contribute to economic welfare in combination with other (produced) inputs. There may also be issues to address as to the extent of the population that benefit from the outcomes that result from provision of these ecological inputs. In Boyd's (2008) schema these are matters to assign to what he terms "the valuation side of the ledger" (p8). Accounting for ecological quantities then would be reserved for a focus on establishing indices of basic biophysical inputs. These are not just envisaged to be land area but also inputs such as populations of species and so on.

Much of the broader debate about ecosystem accounting inevitably has centred on the pioneering classification of ecosystems services in the *Millennium Ecosystem Assessment* (MA) (MA, 2005) the focus of which was: (a) *provisioning services* such as freshwater, food and so on; (b) *regulating services* including various types of regulation of climate, water, pests etc.; (c) *cultural services* spanning relatively tangible services such as recreation through to aesthetic and spiritual values; and, (d) *supporting services* such as primary production and nutrient cycling.

Since the advent of the MA, a number of contributions have sought to evolve this schema. Haines-Young *et al*. (2009), however, provides an explicit link to ecosystem *accounting* in trying to build a bridge between the discussions of ecosystem services in the wider literature and in the SEEA 2003 (UN, 2003). Specifically, UN (2003) makes reference to *resource functions*—or "… natural resources drawn into the economy to be converted into goods and services …" (p5)—*sink functions*—that "… absorb the unwanted by-products of production and consumption …" (p5)—and *service functions*—that "… provide the habitat for all living things …" (p5). What Haines-Young *et al*. seek to achieve is a cross-tabulation of a list of ecosystem services (reflecting the evolution of thinking about classifications since the publication of the MA) with this representation in UN (2003) as well as other official statistical classifications of economic activity, products and consumption expenditures.

Table 6: Classifying Ecosystem Services

Abiotic inputs	*Intermediate services*	*Final services*	*Benefits*
Sunlight, rainfall, nutrients	Primary productivity	Water regulation	Drinking water
	Pollination	Primary productivity	Food

Source: Fisher and Turner (2008).

Within this literature, a common objective has been the imposition of more structure on the ecological accounting problem than was present in the MA classification. This has involved distinguishing between intermediate and final (ecosystem) services. One recent classification in this vein, by Fisher and Turner (2008), is illustrated in Table 6. The distinction here is three-fold between not just whether services are (i) final or (ii) intermediate but also whether (iii) what is the ultimate benefit following a distinction first drawn, for example, by Boyd and Banzhof (2007).

This is more than just semantics. Circumscribing terminology in this way is a means of ring-fencing the use of term 'benefits' for that thing that is closest to what enhances human well-being and is thus the thing that we want to value (e.g. food, drinking water, recreation, amenity etc.). Thus, as in the Table 6, food is a benefit which is derived from the final service of primary production (of ecological systems) which, in turn, depends critically on intermediate services such as pollination as well as abiotic inputs including sunlight. It is perfectly possible, however, for a service to be final in one sense (primary) production in the case of food production) and intermediate in another (primary production in the case of drinking water). A further complication is that some services might provide multiple benefits such as, for example, flood prevention as well as drinking water in the case of water regulation. Boyd and Krupnick (2009) call such instances 'dual commodities' while Fisher *et al*. (2009) refer to this as 'joint production'.

Boyd and Krupnick (2009) have sought to make all of this thinking in the abstract about ecosystem services more schematic in terms of what they coin as an 'ecological production theory'. Again the crucial distinction is on ecological inputs and endpoints where it is the latter that ultimately gives rise to wellbeing by providing things that people might place a value upon receiving. What links these endpoints to inputs is some form of (ecological) production function. Critically, this production (and transformation of some combination inputs) might well be a complex phenomenon. However, the end-point itself may be more straightforward to understand: for example, it might be clean drinkable water. From the perspective of valuation, finding out exactly how beneficiaries value this end-point seems an uncomplicated task. Use of the term 'uncomplicated' here is relative and refers to the fact that people could be likely to be more familiar with the output than they are with the ecological production process (or the inputs) that yielded this end-point. As 'familiarity' is in no small part central to ensuring robust valuation then this is, in essence, a reassuring conclusion.

Given this evolution in thinking about classifying ecosystem systems, it can also be asked how well do *existing* valuation studies of ecosystem services perform in terms of the compartmentalised thinking that these typologies recommend? Boyd and Krupnick (2009) argue that relatively few contributions appear to have anticipated these recent developments. Of course, what is just as interesting to ask is how this recent thinking can be used to guide the future conduct of studies.

Pascal *et al*. (2009) present a comprehensive overview of the state of valuation in ecosystem services. A striking aspect of that assessment of the literature is the extent to which basic frontier research issues tend to dominate concern about the evidence. As Barbier (2007) puts it, ecosystems "… give rise to particular measurement problems [and this] is especially the case for the benefits derived from the regulatory and habitat functions of natural ecosystems." (p182). Notably, many of these problems relate significantly to the extent of basic understanding of the physical and scientific dimension of ecosystems and ecosystem change. Nevertheless, these uncertainties have clear implications for caution in interpreting ecosystem values. For example, it is not clear that 'bottom up' approaches—whereby each type of service is valued separately and then the values are added or 'scaled up' to get some idea of the total economic value of the ecosystem—are capturing the 'whole' value of the ecosystem or ecosystems. Put another way, the value of the system as a whole may be more than the value of the sum of its parts perhaps because of complex ecological interactions (Arrow *et al*. 2000).

Pascal *et al*. (2009) distinguish between what they term the infrastructure (or primary) value and the output (or secondary) value of ecosystems. The latter is arguably what has been the focus of discussions about the 'value of ecosystem services' in, for example, Fisher and Turner (2008) and Boyd and Krupnick (2009) and practical ecosystems assessments such as Bateman *et al*. (2010) and the TEEB review (www.teebweb.org). The former might include the value of the stock of ecological assets themselves and is related to the ability of ecosystems to absorb (external) shocks and stresses—i.e. its *resilience*—and still provide services (which, in turn, may be a function of the diversity of the ecosystem) (see, for example, Perrings, 2006).[26]

It is fair to say that practitioners have made considerably more progress in understanding secondary rather than primary values. However, basic conceptual issues

[26] Another issue is the fear that, as ecosystem degradation continues, this bring us closer to ecological thresholds (see, for example, Walker and Meyers, 2004, on the different forms that such thresholds might take). At present, however, there is little of practical merit that can be said about this issue (although see Longo *et al*. 2007 for a study of the value that recreational users place on the approach of thresholds determining the extent of algal blooms in coastal zones of the north coast of Belgium).

are becoming better understood (Mäler *et al*. 2009; Farley, 2008). Recent contributions have likened ecosystem management to concepts of asset portfolio management which until recently enjoyed prominence in financial economics (see, for example, Heal, 2007). That is, having a more diverse portfolio is of greater value for maintaining resilience in the face of the risks that different possible states of the world might present. One interesting development that has emerged from theoretical work treats resilience as a stock (see, for example, Mäler, 2008). In other words, the ability of an ecosystem to withstand shocks has a distinct asset value which can be degraded (or enhanced) over time. Nonetheless, as things stand, practical valuation is far from any sort of comprehensive accounting for resilience. Walker *et al*. (2010) is an exception and provides an example for agriculture in South East Australia.

Moving beyond consideration only of (current) ecosystem services, in the context of valuing assets (in wealth accounts)—and changes in assets (in product as well as wealth accounts), inevitably decisions must about be made about how to discount flows of future services.[27] The debate that has ensued since the Stern Review (2007) illustrates the controversy that surrounds the choices about the magnitudes of any of the components of the *social* discount rate in the context of climate change. This includes the time preference (impatience and survival risks) the future productivity of the economy as well as the (utility) value to attach to the fruits of that productivity. A recent review by Gowdy *et al*. (2009) in the context of ecosystems and biodiversity illustrate that the issues there are likely to be no less controversial. Of course, official guidelines in countries will often set out specific advice on discounting practice. However, these guidelines often differ considerably between countries which may or may not raise issues about international comparability.

The valuation problem surrounding natural asset valuation raises distinct issues over and above how future values should be *discounted.* Horowitz (2002) notes that, even if the preferences held by present and future people are assumed to be identical (possibly a strong assumption), there are at least two important determinants of future values that might change over time. The first of these is *income*. If, for example, it is reckoned that future people will be richer than people now then marginal values should be adjusted to reflect the fact that future people will value the same change more highly. The basis for this would be an assumption that what is being valued is at least a normal good. A second factor is *environmental quality*. In this instance, if it is thought that environmental amenities will become more scarce in the future then it is plausible that the (marginal) value that will be placed on future losses of this amenity will be higher (than now).

[27] Discounting involves attaching a lower weight to a given unit (say $1) of future benefit (or cost) than to an equivalent present unit. Discounting implies that the weight, w_t, to be attached to a gain or loss in any future year, t, is less than 1. More specifically, the discounting formula is: $w_t = \frac{1}{(1+s)^t}$ where s is the (social) discount rate. This discount factor, w_t, there shrinks as t gets larger (i.e. as gains and losses become more distant).

This issue about how marginal values change as a result of changing environmental quality itself segues into an ever present concern about the extent to which the environmental values estimated using non-market methods truly reflect the relative importance of different assets. There are a number of aspects to this but arguably the most prominent surrounds substitutability between natural assets and other forms of wealth.[28] But while this debate is clearly critical, practical insight (rather than speculation) is itself rather scarce. One immediate problem in answering such a question, however, is that it typically involves working with a literature—and body of knowledge therein—which is at least one step removed from being straightforwardly operational. The practical import of such theoretical insights has been demonstrated more recently by Hoel and Sterner (2007) and Sterner and Persson (2008). In doing so, they demonstrate that not only will scarcity result in an increase in the relative price of a natural asset but that the magnitude of this increase will depend, among other things, on the substitution possibilities between the natural asset and other forms of wealth.

6. Concluding Comments

We began this paper by making the claim, in effect, that 'from-the-outside-looking-in' (and not withstanding important contributions made in other respects), on-going efforts to green the national accounts have been largely immune to the 'environmental valuation revolution' elsewhere. Given that an emphasis on environmental valuation is becoming ever more evident in policy thinking more generally, there is a real risk that what is being sacrificed — to preserve this immunity — is relevance. Of course, there is a corresponding risk of over-claim here as there are a number of reasons why we ought to be healthily sceptical about non-market valuation approaches. However, it is important to use this scepticism positively in order to ensure robust valuation only is used to green national accounts.

A number of challenges must be confronted in order to achieve this end:
Explicit guidance on the hurdle of accuracy that non-market valuation methods are expected to attain would be helpful (given that this is often cited as a focal reason to discount the role that environmental valuation might play in greening national accounts). In this way, a clearer assessment can be made about how far specific methods *or* particular categories of environmental impact are from satisfying any such requirements.

[28] This concern is typically characterized in terms of whether development should be weakly sustainable or strongly sustainable. While there is some debate about when exactly this terminology entered the literature, the main ideas can be found in Pearce *et al*. (1989). For *weak sustainability*, there is no special place for the environment as such. Put another way, it is the 'overall' portfolio of wealth bequeathed to the future that matters. As long as the real value of this portfolio is held constant it matters little that its constituent parts change over the development path. *Strong sustainability*, by contrast, requires that the environment is accorded explicit and special protection. There are a number of variants on this position. Most generally, it requires that 'natural wealth' should (in some way) be preserved intact through specific conservation rules.

This seems especially important if benefit or value transfer is to play a large role in greening national accounts. Valuation practitioners have become increasingly at ease with the idea that, in many cases, errors may not be sufficiently large so as to render policy appraisal based on transfer exercises of little practical worth. It may be that any such view is misplaced in the domain of green national accounting. At the very least, this is an issue that needs significantly more discussion. One issue is that the national accounting context possibly requires generalising the empirical record in a highly ambitious way.

Further work here can usefully profit from current and on-going national and international efforts to collate the empirical record in various areas subject to environmental valuation. In particular, for ecosystem services, such efforts (such as TEEB, UK National Ecosystem Assessment and so on) should reveal soon—in a systematic way—what critical data gaps exist and to what extent the existing record can be stretched to facilitate large-scale assessments. Formal national accounting arguably could do more to influence and shape these assessments. This would help ensure that what emerges will be more suitable for explicit use for national accounting purposes. The same comment could apply equally to influencing the future direction of environmental valuation more generally.

The challenges ahead are not restricted, by any means, to the above. We have focused primarily on valuation in this paper, the reason being that this is the area which has proved to be especially contentious in an accounting setting. However, at various points, we have touched on issues and perspectives which emphasise more prominently the physical (or quantity) side of the accounting problem. This raises important challenges too which need to be addressed. What should be borne in mind, however, is that this emphasis needs to be on physical 'end-points' that are capable—at some point—of being valued (i.e. that are meaningful in socioeconomic terms).

Along the way to a more meaningful engagement with environmental valuation, it is likely that there will be a number of issues which are genuinely at the valuation frontier and give considerable pause for thought. At the same time, neither should this wait for evolving research at the frontier be allowed to arrest development of 'simpler' elements of useful work in incorporating non-market or environmental values—within green national accounts—that currently are considerably better understood.

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