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Research Article

Valuation of ecosystem services: paradox or Pandora's box for decision-makers?

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Abstract

The valuation of ecosystem services (ES) employs a range of methods. Based on a literature review and selected empirical examples, we consider major opportunities and challenges in ecosystem services valuation. We analyse when different valuation methods are appropriate and most useful. We demonstrate that mechanisms to capture benefits and costs are needed; and that the use of valuation should be incorporated more widely in decision-making. However, we argue that ecosystems are complex systems: neither the ecosystems or the services that they provide are a sum, but are an interrelated system of components. If a component vanishes the whole system may collapse. Therefore, critical natural capital management, in particular, cannot rely on monetary values; whilst the maintanance of the whole system should be considered. Monetary valuation of biodiversity and landscapes is also problematic because of their uniqueness and distinctiveness, a shortage of robust primary valuations, and numerous complexities and uncertainties. We conclude that mixed method and deliberative discourse techniques, as well as proper integration of research tools, should be more widely applied to help decision-makers and the public to understand and assess changes in ES. The approaches developed and tested by us, as presented in this paper, can provide more complete, comprehensive and impartial insights into a range of benefits that humans derive from ecosystems.

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Keywords

Natural assets; public goods; forest; sustainability; non-monetary valuation; mixed methods; Scotland

1. Introduction

Ecosystem services (ES) contribute to the generation of income and wellbeing, and to the prevention of damages that inflict costs on society. The latter is characteristic of certain ES that provide insurance, regulation and resilience functions. The understanding of mechanisms to capture the values of services provided by nature, and the costs of their possible depletion and degradation (potential losses), is growing (Turner et al. 2010). All types of benefits (and costs) need to be accounted for in decision-making. In this paper, taking inspiration from ideas developed in recent assessments, we analyse ways and economic methods to value ES in order to help to address the United Nations (UN) Sustainable Development (SD) Goals' Agenda (United Nations 2015) in relation to the use of natural assets and provision of ecosystem services at a local level.

Recently published reports have highlighted the growing costs of ecosystem degradation (Kettunen and ten Brink 2006, O'Gorman and Bann 2008). The Scottish Government's aspiration to expand the area of wooded cover and increase the contribution of natural assets to the broad range of economic and social benefits is embedded in a number of policies. The Land Use Strategy (Scottish Government 2016b) and Scottish Forestry Strategy (Scottish Executive 2006) highlight the desirability of capacity development for ecosystems to address climate change and the production of more services, reinforced by the Rationale for Woodland Expansion (Scottish Government 2009). The Scottish Government's policies aim to stimulate investment in Scotland's natural capital and assets to deliver to public agendas of improving quality of life, tackling social exclusion, and promoting sustainable development, wealthy communities and healthy lifestyles.

The report on the "...valuation of the economic and social contribution of forestry for people in Scotland" (Edwards et al. 2009) highlights the importance of the provision of benefits to people in terms of health and well-being, learning, education, the ability to sustain a wealthy living and to contribute to the viability and vibrancy of the local communities. The United Nations 2015 set out an ambitious plan of action for transforming our world. It is underpinned by 17 goals and 169 targets, which will drive global efforts towards SD over the next 15 years. Scotland was amongst the first nations to commit to these (Scottish Government 2016a).

Defra's Guide (Department for Environment, Food and Rural Affairs 2007) and associated documents provides direction to the valuation of ES in the UK, including for policy appraisal purposes. The Economics of Ecosystems and Biodiversity (TEEB 2010) is a major international initiative which aims to promote a better understanding of the economic value of ES to help incorporate these values into decision-making at different levels. Also, the

IPBES initiatives seek to make a difference, and so address sustainability targets (Pascual et al. 2017), and for doing it, it's crucial to fisrtly understand what value categories are.

Overall, value concepts can be divided into ecological, socio-cultural and economic values (Farber et al. 2002, Chan et al. 2016). Ecological value is determined by the integrity of a forest system's provisioning, supporting and regulatory functions, and by its characteristics (i.e. indicators of ecological relevance, such as complexity, diversity, rarity, or naturalness, applied across spatial scales). The concept of socio-cultural values includes social values (e.g. equity) and end-user perceptions, including of their cultural and spiritual (or non-material) well-being. Indicator systems, including economic indicators (e.g. employment and income) have been used for assessing ecosystem services (Adamowicz 1995). However, in addition, the valuation of ES uses a number of concepts and methods developed specifically by social scientists, the most important and relevant of which are analysed in this paper.

The concept of total economic value, TEV, has become popular. TEV is equal to market value plus the consumer surplus, CS, i.e. the difference between what an individual is willing to pay for a good or service and what they actually pay. If a good has no market price, the consumer surplus represents the TEV. It is the total gain in well-being from a policy, which comprises use and non-use values of ES (O'Gorman and Bann 2008). The TEV is based on the economic concept of value (Nunes and van den Bergh 2001) which originates from neoclassical welfare economics. It is rooted in utilitarianism, and expresses the degree to which a service satisfies individual preferences. An estimate of TEV is usually considered as the sum of the direct, indirect and non-use benefits provided. The key components of TEV are presented in Fig. 1, taking the ES of trees as an example.

Despite recent advances in conceptualising ES valuation, and its importance in informing decision-making, the TEV concept has been criticised for using figures which are perceived as too abstract and indicative; while in reality, values are complex and dynamic (Porras 2012). In real life situations, values vary between different individuals and groups, and values change temporally and spatially. However, ES valuation is carried out using current knowledge, which is often incomplete. Therefore, (O'Gorman and Bann 2008) argue that future valuation work should focus on advancing the TEV conceptualization of cumulative value estimates, and on further development of marginal assessment of changes in services that can be provided by an ecosystem.

Economic trade prices concern relative values in exchange, set by marginal units sold (Spash 2008). Thus, marginal valuation is relevant as a measure of changes in ES, and marginal changes in values are important: for example, when one type of resource management is changed for another, and because economic analysis to inform policies usually concerns marginal changes (O'Gorman and Bann 2008).

Overall, the complexity of ES and their 'arrangements' (ecosystem condition, size, or connectivity) pose problems for ES valuation (Spash 2010). Economic values reflect the services of an ecosystem and not the economic value of that ecosystem. Insufficient understanding of ecological processes and numerous uncertainties (e.g. scientific uncertainties; those related to the context, cause and effect relationships being evaluated; and uncertainties related to the provision and use of ES) often lead to unreliable estimates. Valuation of supporting services is particularly difficult, concerning real integrated values (e.g. to avoid the double-counting).

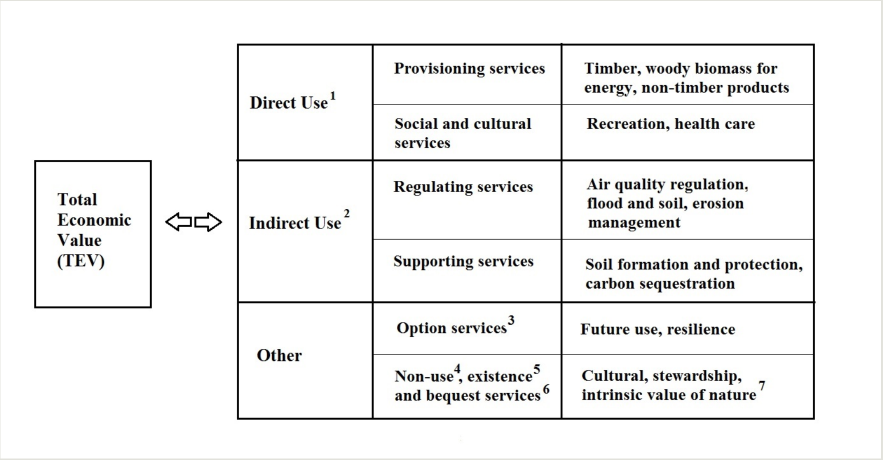

Figure 1.

Key components of the Total Economic Value concept. Figure adapted from Glaves et al. (2009)

As of the Department for Environment, Food and Rural Affairs 2007:

[1] Direct use value is where individuals make actual or planned use of an ES.

[2] Indirect use value is where individuals benefit from ES supported by a resource, rather than by using it directly.

[3] Option value is the value that people place on having the option to use a resource in the future.

[4] Non-use value is the value that is derived from the knowledge that the natural environment is maintained. This comprises bequest value, altruistic value and existence value.

[5] Existence is the value individuals derive from the knowledge that an ecosystem resource exists, even though they have no current or planned use for it;

[6] Bequest value (an example of non-use value) is the value individuals attach to the fact that the resource will be available for use by future generations.

[7] Intrinsic value is the worth of a good or service for its own sake.

Monetary values could be assessed using avoided-cost or replacement cost methods. However, effects of site-specific conditions and local scarcity means the value of ES generated at one locality can vary significantly from that at a different location. The value of services is also contingent on proximity to demand. An accessible landscape, for example, is worth more than the same landscape in a remote location with respect to use values. There is also an inherent variability in values across space, as a provided ES is spatially variable (e.g. the habitat of a rare species, or the potential for sequestering carbon). Also, ecosystem management could induce more ‘public bads’ (i.e. dis-benefits) in some areas compared to others. These arise e.g. from the visual intrusiveness of blocks of exotic conifers (Slee et al. 2008). Moreover, the potential use values of ES (i.e. option values) are challenging to assess. Ecosystems are judged on what they are known for and what they are now, rather than on their potential in the future. Therefore, option values (and those of existence and bequests, Fig. 1) are not incorporated in ES valuation.

It is particularly difficult to place values on ES when dealing with jointly produced services, delivered and utilised as bundles, as pricing individual components can be difficult. ES are inter-related and affect each other, and some types of services contribute to others, leading to potential double counting; supporting services may contribute to regulating services, or regulating services could contribute to cultural ES (de Groot et al. 2002). Double counting is an issue where multiple services are delivered and sold separately or included in schemes operated by different jurisdictions, for instance management authorities or private businesses (URS Scott Wilson Ltd. 2011). Such double-counting may arise when valuing primary ecological processes (e.g. soil formation, nutrient cycling) which support ecosystem functions (Bateman et al. 2010) leading to risks of overstating the TEV generated (Gren et al. 1994, Fisher et al. 2011).

Therefore, it is important to recognize the differences between valuation methods, including their strengths and limitations, and the range of their applicability at and across scales, to select the most appropriate approaches. In this paper, inspired by the UN SD Goals 2030 Agenda (United Nations 2015), and ideas developed in Scotland, and in the TEEB (TEEB 2010), MEA (Millennium Ecosystem Assessment 2005), UKNEA, Defra's Guide (Department for Environment, Food and Rural Affairs 2007) and related documents (HM Treasury 2011,Turner et al. 2010) we aim to contribute to analysing valuation methods and appropriate scales for primary valuation. We understand the necessity and opportunities for updating the evidence base. Our contribution in this paper is based on the analysis of concepts and methodological approaches to advance scientific knowledge of ES valuation and inform decision-makers on opportunities for wider and more efficient use of appropriate (case- and context-specific) valuation techniques in policy design and decision-making.

We firstly present an overview of conventional methods of ES valuation, paying attention to challenges of their application. Then, we introduce innovative, non-market evaluation techniques (in a convential type of research article, this section would be entitled 'Methods') and provide results from research on the use of the suggested techniques, at various scales of analysis. We conclude this article by discussing the applicability of valuation (vs. evaluation), and suggest ways forward for scientific research in the field and its practical 'on the ground' implementation.

2. Insights into conventional methods of ecosystem services valuation

There is a considerable variety of methods for valuing ES, with selected examples shown in Table 1. Some ecosystem services (e.g. provisioning services) take the form of 'economic goods'. They are derived from the use of a natural asset. In a well-functioning market, supply and demand determine the appropriate price level; and market valuation applies well. This valuation is largely 'objective' and is done either 1) directly, i.e. based on observed market transactions and actual prices, or through 2) indirect market valuation (Dixon et al. 1994).

Table 1.

Selected examples of methods used to valuate ecosystem services.

Examples of ES/goods	Valuation method	Value
Provisioning services		
Food, fibre, timber, woody biomass for energy	Market valuation	Market prices
Regulating services		
Carbon sequestration Climate regulation	Cost-effectiveness Market valuation	MAC (costs per tCO_2) Market prices (if CO_2 is traded)
Erosion alleviation Shelter belts	Replacement, relocation and avoided cost methods	Avoided losses in yields or cost of increased yields
Air quality	Avoided cost methods	Avoided losses
Flood regulation	Benefit transfer Relocation and avoided cost methods	BT estimates Avoided losses
Cultural services		
Recreation	SP, e.g. CVM RP, i.e. travel cost method Indirect market valuation	WTP values Travel cost estimates Market pricing
Landscape beauty, aesthetics	RP, hedonic pricing method SP, e.g. choice experiments	HP values WTP values
Health	Indirect market valuation	Changes-in-productivity Cost-of-illness estimates
Supporting services		
Oxygen	Replacement cost methods	Cost of oxygen
Soil formation and protection	Avoided cost method	Cost of purchasing top-soil from elsewhere
Species diversity	Indirect market valuation	Donations for conservation

Approaches using market values, but going beyond actual pricing, could be based on market prices of close substitutes, or shadow pricing. They could also be based on 'changes-in-productivity' or cost-of-illness considerations (a form of dose-response market analysis) (Kallis et al. 2013). Another approach is based on opportunity costs associated with (for example) changes in uses of ecosystems, or environmental and climatic impacts, which affect the provision of ES[*1].

Many ES enhance incomes: for instance, stand productivity improvements increase commercial timber produce, and therefore the profitability of the industry. Such ES can be valued using an indirect market valuation technique called the 'factor income method'. Natural assets can be treated as inputs to the production of other goods, based on resource linkages and market analysis (i.e. the 'production functions' technique), or through 'public pricing' (i.e. public investment, such as land purchase or monetary incentives, as a surrogate for market transactions) (Dixon et al. 1994).

Monetary (cost-assessing) approaches (for example, those used for comparing scenarios or management practices) are usually based on values of actual or potential expenditure, such as expenses in support of more sustained provision of ES. These approaches include cost-effectiveness; preventive expenditures (i.e. avoided cost method, AC[*2]; replacement costs[*3]; relocation costs (RC)[*4]; and shadow projects, i.e. a type of RC technique).

Where regulatory standards are set externally, the challenge may be the estimation of the least cost solution to meet regulatory needs (Yates 1999), rather than the estimation of non-market benefits. This generally requires cost-effectiveness (CE) analysis. It is widely recognised as a useful tool for considering the least cost of compliance with regulatory standards, such as those pertaining to good ecological condition of land when planning future ecosystem management.

The overall CE of delivery mechanisms of ecosystem management depends upon identifying which parts of the programme contribute most to effectiveness (i.e. outcome delivery), then assessing which programme components have the lowest cost (Phillips and Thompson 2009). The CE depends on the relationship between spending and outcome: spending is measured as money spent, and outcome can be evaluated either directly (using indicators, rather than definitively in monetary terms) or indirectly, through stakeholder evaluation (Nijnik and Mather 2008, Nijnik et al. 2016).

However, there is usually no optimal solution to the problems raised by the complexity of ES (Rittel and Webber 1973). Moreover, the fact that ES are complex systems, made up of the interactions of numerous ecological, economic, and social (and policy) factors, means that many ES decisions are also complex. Therefore, when there are no explicit markets, monetary estimates can be a poor approximation of value. Markets can work for those ES which have direct user values (e.g. commodities), but they tend to undermine the provision of other ES which favour public sector interventions (Farnworth et al. 1983).

A key issue with several ES is that they are non-excludable: recipients receive the service regardless of whether they pay for it, and non-payment does not lead to exclusion. Many ES are also non-rival: any number of people can use a resource without leaving less for others. Provisioning ES (e.g. food, timber and other commodities) are usually highly excludable. Complex property rights and market supply-chains have evolved to connect producers and/or managers of such ES to their end consumers. Similarly, via market intermediaries, access to clean water is typically excludable. Prospective users who would not pay for an ES could be excluded from using some recreational and cultural services, such as club goods (e.g. using fences and controlled access points). However, this is not practical for other ES (e.g. biodiversity, clean air). Consequently, it is difficult to charge recipients of such services. Direct market exchange between providers and recipients fails

(URS Scott Wilson Ltd. 2011[*5]). Economic valuation of non-rival and non-excludable ES, such as public goods, is highly controversial (Randall 1993, Scholte et al. 2015). Nevertheless, indirect marked grounded techniques can often provide a useful approximation.

Revealed preferences (RP) approaches are effective when dealing with 'use values' of ES (e.g. recreation), which are commonly assessed using travel cost (TC)[*6] estimates (Clawson and Knetsch 1966) and hedonic prices (HP) (Rosen 1974[*7]). However, RP cannot capture non-use values (European Environment Agency 2010), and the 'existence value' of ES (Table 1) remains overlooked.

Stated preference (SP) methods (Hill et al. 2003, Schläpfer et al. 2004[*8]), including such methods as bidding and trade-off games, take-it-or-leave it, Delphi techniques and others, have been used to overcome this limitation (Adamowicz 1995). First applied by Davis (Davis 1963) as a tool for valuing recreation, the contingent valuation method (CVM) of willingness to pay (WTP) or willingness to accept compensation for the loss of ES (WTA) (Hanley and Spash 1993, along with choice experiments, have become widely used (Brown et al. 1993, Willis et al. 2003)[*9].

However, RP and SP methods are essentially an extension of market valuation, which aims to assign monetary measures to the components of TEV. The biases of SP have been extensively discussed in the literature (Bateman et al. 2002, MacMillan et al. 2004). They fall into two broad categories: bias due to sampling error, and hypothetical bias (Bishop and Romano 1998). In addition to technical problems inherent in valuing non-marketed ES, there are concerns about gaps between hypothetical monetary values and reality (e.g. ES are complex; not all beneficiaries are willing to pay; the scientifically derived WTP may substantially exceed actual expenditure). Also, valuations reflect the current distribution of income, with those with higher abilities to pay better able to reflect their preferences by a higher WTP (Bateman et al. 2010).

Nevertheless, RP and SP can provide useful information to decision-makers. This is particularly true for SP and when a market is absent, for example where there are free public goods with zero prices (Jakobsson and Dragun 1996, Garrod and Willis 1997, Arrow et al. 2001, MacMillan et al. 2004). However, comprehensive analyses remain approximate and comparative analyses are rare (Nijnik and Mather 2008).

Recently, benefit transfer (BT)[*10] methods of ES valuation have received attention (Hanley et al. 2002)[*11]. BT has positive characteristics (Davis 1963) as this type of technique is relatively easy to understand and apply. Benefit transfer approaches have been used, for example, to estimate non-market benefits from recreation at Forestry Commission forests (Gelan et al. 2007). These values were compared with data on the costs of recreational provision. The analysis revealed large differences in non-market values across space, and that in only a minority of sites, normally those closer to built-up areas, was there a surplus of social benefit over the costs of provision. A peri-urban recreation site with modest parking facilities can generate in excess of 200,000 visits a year. In comparison, a significantly more visually and environmentally attractive site in a remote area may receive 10,000 visits a year (Department for Environment, Food and Rural Affairs 2010).

Results of BT application provide some insights into the values of ES. However, BT values are largely abstract and indicative, and often rely on the availability of data and classifications developed for other purposes and not necessarily at an appropriate scale. For example, conservation decisions are often carried out at a detailed level, such as a land management unit, with limited reference to occurrences at landscape or higher levels (Gelan et al. 2007).

Consideration of levels/scale (spatial and temporal, and the context of a valuation study) is important while valuing ES (Vermeulen and Koziell 2002 Consider, for example, provisioning services: food or timber have explicit (market) values at local, regional and global scales. However, this may not be true for medicinal plants with value for local people. Additionally, the value of some regulating services only exists at a regional scale, whereas values of carbon storage and sequestration by trees are evident at a global scale (Food and Agriculture Organization of the United Nations 2014).

For example, an assessment of the climate regulation service provided by a new woodland in Scotland is carried out at a catchment scale. Maximization of ES of new woodlands is determined by prioritising their creation and design in areas and ways that strengthen habitat networks, whilst avoiding prime agricultural land and thus not compromising food production. However, services of significant values at a local level (e.g. of soil erosion prevention) can be overlooked at larger levels of valuation (Castellazzi et al. 2010).

At regional to global scales, some ES can be approximated by simple links between ecosystem types and services, underpinned by general assumptions developed from information in the literature (Daily and Ellison 2002). The typology reported by Hermann et al. (2011) helps determine which service can be assessed at local, regional, or higher scales. A proposed match of scales of valuation to selected ES is available from the Nature Valuation Organization (Nature Valuation Organization 2005). However, we argue that the most appropriate scales for valuation are case- and context-specific and are not only dependent upon service type; additionally, trade-offs between different ES are also scale specific.

Furthermore, because many ES arise from complex processes, it is often difficult to determine which actions affect their provision and the identities of providers and beneficiaries (Glück 2002). The gap between providers and beneficiaries of ES is among the major challenges of designing and implementing payments for ecosystem services (PES) (URS Scott Wilson Ltd. 2011).

Also, there could be trade-offs in the valuation of ES, for example in favouring direct benefits such as employment, versus supporting or regulating services. At a national level, the financial benefits of logging were greater than those of conservation (Nature Valuation Organization 2005). However, the inclusion of carbon and biodiversity led to the conclusion that conservation benefits were greater. This accords with evidence from Nijnik et al.

(2012) indicating that it is only when non-use and existence values are taken into account, that forest ES values exceed the opportunity costs.

For example, the approximate value of timber and non-timber forest products is €125 ha^{-1} yr^{-1}; whereas the value of carbon, water and soil protection ES of forest exceeds €170 ha^{-1} yr^{-1} (Willis et al. 2003). This is consistent with the findings, also for Ukraine, when multifunctional afforestation was considered (Nijnik et al. 2012). Today, valuation evidence is generally based upon case studies, all of which are useful but difficult to upscale into a compelling narrative which can be used in different ways (Muriithi and Kenyon 2002). Thus, there is a challenge to create a proper framework to capture multiple values (Natural England 2010).

Furthermore, temporal scale is an important consideration (Department for Environment, Food and Rural Affairs 2007); particularly so in forestry, due to the time lag between tree planting and accrual of ES. By discounting and thereby converting all costs and benefits to present values, we can take into account the temporal distribution of the costs and benefits of ES provision (Hanley and Spash 1993. However, the next issue is to choose the most appropriate discount rate as this can have significant impacts on the final outcome of ES valuation (Nijnik et al. 2012b)[*12].

The overview of relevant literature sources, provided in this section, shows the importance of advancing research methods, and making them more relevant (e.g. case- and context-specific), accessible and effective in offering meaningful information to different audiences, such as through guiding public understanding of the consequences of ES changes, and aiding decision-making. Decision support systems for participatory planning and knowledge transfer in ways that are understandable to different types of end-users need to be co-constructed involving relevant stakeholders. Innovative social science approaches are needed to help realise the potential of policy analysis and sustainable ecosystem management.

3. Innovative approaches to ecosystem services valuation

Recent literature provides strong arguments that values for the social states of public goods can, and should, be determined through non-market-oriented stated preferences, or preferences that are revealed through mechanisms other than the market (Kant and Lee 2004, Department for Environment, Food and Rural Affairs 2011). Therefore, valuation methods that do not apply market analogies have been developed (URS Scott Wilson Ltd. 2011). Such approaches can, for example, enable researchers to develop 'conceptual content cognitive maps' to illustrate either individual or group values and preferences. Depending on valuation objectives, various surveys, focus groups, multi-criteria analysis (MCA) and multi-attribute utility analysis (MAUA) have been used.

Each valuation method is useful if appropriately used; however, each method has weaknesses and/or application challenges (Kearney et al. 1999). For example, focus groups can be unrepresentative of the population as a whole, and it is difficult to find methodological guidelines to develop a systematic understanding of value-relevant information. urveys can suffer from difficulties in design, administering questions, and interpreting results. MAUA is usually difficult for participants to understand; whilst ranking and MCA employ human subjectivity and do not provide aggregate estimates or generalizable results (Steelman and Maguire 1999, Nijnik and Mather 2008

Therefore, we considered the integration of analytical and participatory techniques for evaluation of ES that involved active participation of stakeholders. The inclusion of multiple actors with multiple objectives in the process improved its potential to become more inclusive and comprehensive (Steelman and Maguire 1999). Visualisation tools and scenarios of land use change have been tested and evaluated for facilitating participation (Miller et al. 2006). The focus was on the contribution of factors which could change the character of landscapes, and developing the capability of stakeholder involvement to aid in the assessment of options for sustainable management of natural assets.

Participatory approaches, based on mixed methods or the integration of methods, have been applied to achieve multiple objectives at different geographic or temporal scales. In addition, the use of one technique was validated by using a different technique for the same purpose, as explained in Nijnik et al. (2009). The involvement of stakeholders in the evaluation process has served as a means for mutual learning for stakeholders and researchers, and the co-development of decision-making capabilities, with the combination of valuation methods enabling values to be placed on ES.

Examples from other studies include approaches based on the market stall method (European Environment Agency 2010[*13]) and contingent behaviour model (Kenyon 2005 [*14]); and where there are competing social groups, the discourse-based valuation technique (Wilson and Howarth 2002). Group valuations, using deliberative processes, enable stakeholders to converge on a shared assessment of the values of ES (O'Neill 2001, Vermeulen and Koziell 2002, Miller et al. 2006). Group valuation approaches (Hein et al. 2006) often require people to go beyond their self-interest, and come to a more complete and socially just ES assessment, as a group. Public debate becomes part of the process to uncover existing values because ES valuation is not the aggregation of individual preferences, and the ES value is not just the total of its components.

If ES were not complex systems, and if all values were expressed in the same units (e.g. monetary) they could be aggregated. However, because of the complexity, and attempts to provide representative assessments of complex values (e.g. through group evaluation and deliberation), the individual values placed on ES can be presented side-by-side and compared (Christie et al. 2007). They can be compared using MCA, and stakeholders can be asked to assign relative weights to different sets of indicators (Nijkamp and Spronk 1979, Costanza and Folke 1997, Strijker et al. 2000, Balana et al. 2010). The decision-making process largely relies upon these types of stakeholder evaluation which incorporate the attitudinal diversity towards participatory decision-making of those who design and facilitate the process, and those who are involved in it.

In our research, the participation and visualisation tools (Miller et al. 2006, Scottish Executive 2007, Nijnik et al. 2011) have been combined with the Q-method (Nijnik and Mather 2008, Miller et al. 2009, Nijnik et al. 2013a, Nijnik et al. 2016), and in Nijnik et al. (2009) also with WTP and the method of aggregated ecological indexes (MAEI). The CVM indicated the individuals' WTP as an expression of public valuation, whilst MAEI was based upon expert knowledge. The WTP estimates formed expressions of intrinsic values, which people attach to inanimate components of a landscape (e.g. waterfall, lake, rock, mountain). The obtained estimates were used as relative values for a cross-comparison analysis.

The results obtained from using the combination of these techniques were compared to elicit public preferences, with the aim of providing advice for decision-making. The approach combined aspects of participatory methods with economic valuation. It added the knowledge of the study context, provided insights into the evaluation process, and in cases of a reasonable agreement between the obtained CVM and MAEI estimates, offered evidence in support of the validity of valuation.

An innovative integration of analytical approaches with participatory techniques to value ES has been the use of visualisation tools (Miller et al. 2006,Ball et al. 2007) for stakeholder evaluation of scenarios of land use change combined with social sciences techniques (e.g. mixed methods). A mobile Virtual Landscape Theatre (VLT) (www.hutton.ac.uk/learning/exhibits/vlt) was designed to facilitate stakeholder engagement and sharing of opinions, either individually or within audience groups. Its mobility enables its use in locations where audiences will feel comfortable, familiar and willing to attend and participate, such as at a community hall or school. Electronic voting tools enable the recording of opinions and subsequent analysis with respect to demographic factors of individuals or groups, and comparisons between different stakeholder groups. The interactivity of the facility includes movement through a 3D landscape, virtual reality, models of an area, switching between alternative representations of landscapes (e.g. under different land management practices), changes in land cover and landscape through time (for instance, a cycle of woodland management), changing environmental conditions (such as cloud and shadow), and the introduction, placement and relocation of features (e.g. trees and houses) at the direction of the audience. The functionality and mobility of the facility supports the end user evaluation of ES and end user evaluation of changes in the ES management.

Stakeholders were consulted to obtain their subjective values. We were interested in the perspectives of those who interacted directly with land-use systems, at a strategic or operational level, whether living or working remotely or locally. The 'people included' principle that identifies a creative management between the integrity of ecosystems and the livelihoods of people, living and working in the environment, was employed in this research. Details of the development and testing of the tools applied to a range of case studies at various scales of analysis are available in Miller et al. (2009) and Nijnik et al. (2010).

To quantitatively identify and analyse stakeholder attitudes and perspectives, a Q-method that originated from psychology was used. This method is explained in more detail in earlier studies (Vermeulen and Koziell 2002, Ball et al. 2007, Nijnik and Mather 2008, Turner et al.

2010, Nijnik et al. 2016). In summary, it is a quantitative means for examining human values. It enables the identification and assessment of subjective structures from the viewpoint of individuals being observed. Respondents are selected according to the research objectives and for each of our case studies. Q-method incorporates elements of behavioural studies into action research. It starts with consultation with stakeholders to identify research essentials, followed by interviews through either survey and/or focus groups.

The output data from the Q-surveys was assessed using the sequential application of correlation and factor analysis. It was followed by a discourse analysis to explain the results obtained. The final steps were the interpretation of the social discourses uncovered by the quantitative enquiry, contrasting the value outputs with the socio-economic background of respondents, and verification and communication of the results with/to respondents (Nijnik et al. 2010). This approach provided insights into respondents' value judgements and identified criteria of particular importance to individuals. These were then analysed with respect to respondents' socio-economic backgrounds (Ball et al. 2007, Nijnik and Mather 2008, Nijnik et al. 2013a). Valuation methods similar to the Q-method[*15] are not free of judgement (e.g. human subjectivity and assumptions). However, Q-method (Nijnik et al. 2013a) can structure 'wicked problems' that are characterized by much uncertainty and value-conflicts, because it enables the identification of patterns in stakeholder perspectives of the issues in question, thus reducing the complexity surrounding them.

Furthermore, the decision-making process concerning ES and their sustainable management relies on technical factors which include the incorporation of technological features in research tools and their effective use, and the incorporation of appropriate levels of information in the tools to communicate knowledge to those involved in the process (Ball et al. 2007, Turner et al. 2010). Among the tools which are increasingly used in the identification, interpretation and assessment of ES and their impacts on social-ecological systems, are those enabling different means of communication and understanding of location (e.g. Geographical Information Systems; GIS) and appearance (e.g. vizualization techniques). Findings and their interpretation has fundamentally changed with the provision of online data and web mapping services, including online perspective viewing and 3D models (for instance, Google Earth and Bing Maps).

The use of innovative tools in a socially-innovative, participatory environment (e.g. which the Horizon 2020 project "Social Innovation in Marginalised Rural Areas, SIMRA www.simra-h2020.eu is addressing) has enabled wider incorporation of inputs from relevant stakeholders into ES valuation, with added value of geographical data on factors which support the interpretation of ES context (e.g. the proximity of woodland to water), and changes through time (e.g. aerial imagery of an area at different dates through a life cycle of a woodland (Turner et al. 2010).

4. Results

The integration of analytical approaches, and participatory visualization techniques, in stakeholder evaluation for assisting in the sustainable use of natural assets was carried out at different spatial scales of analysis. Results, presented in this section, concern examples of application of the proposed in previous section innovative methods tested in our previous studies (Nijnik et al. 2011Nijnik et al. 2010Nijnik et al. 2013as well as in the current research.

Internationally, we considered ES associated with the socio-economic, ecological and visual aspects of land use and landscape changes in the Amazon region, along with the provision of ES in European landscape contexts (six countries, including Scotland). Each example involved stakeholders in ES evaluation and finding solutions of problems relating to the management and use of natural assets. The outcomes (Nijnik et al. 2013a) provided evidence that wider stakeholder involvement in decision-making has had a high level of participant satisfaction, and increased understanding of issues associated with ES. Comparisons of the similarities and differences between the studies provided a basis for the discussion of common and locally distinctive guidelines on good practices in ecosystem management (Miller et al. 2009).

At a *national level*, in Scotland, the Q method application (Nijnik et al. 2010, Nijnik et al. 2016) resulted in the identification of five attitudinal groups reflecting public evaluation of ES of woodlands (Fig. 2) The primary advocates of regulating and supporting ES belonged to groups 1 and 4. Those labelled as 'radical conservationists' (group 1) failed to value provisioning ES. Group 4 representatives named 'moderate conservationists' (according to their values) favoured multiple ES. Their support of nature conservation was accompanied by support for sustainable forest management. Respondents in attitudinal groups 3 and 5 allocated their highest values to provisioning services. Compared with group 3 ('radical productivists'), group 5 balanced timber production and provision of other ES. Group 2 respondents highly appreciated landscape beauty and outdoor recreation. They suggested that hunting and fishing were necessary to maintain the quality of ecosystems. This group ('recreants') positively ranked a range of ES, starting from the conservation of forests to a range of socio-economic benefits that forests provide for communities.

Findings indicate that biodiversity was valued by all attitudinal groups, except group 3. All groups, excluding group 4, considered cultural and social services as valuable. Provisioning services were valued by all, except the radical conservationists (group 1), while only the productivists (groups 3, 5) considered the importance of maintaining forest for timber above all else.

The results indicate that an increasing intensity of conservation measures may affect provisioning services of forest, and vice versa. At one end of a spectrum, ecological approaches emphasise environmental protection, and at the other end, climate change considerations promote carbon forestry. Despite the heterogeneity of public attitudes all groups identified support for the necessity of multiplying the wealth of local communities, concurrently putting the emphasis on ecosystems' resilience.

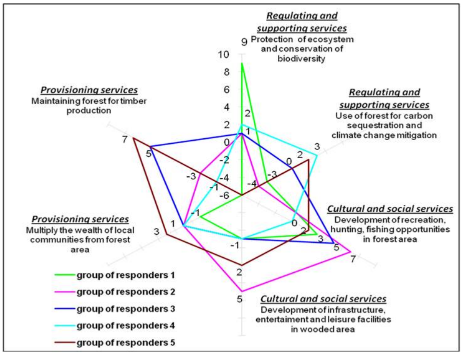

Figure 2.

Selected results of the Q method application in Scotland to define the values of ES.

An exploratory study at *a local scale* was carried out in the Clashindarroch Forest in north-east Scotland. The design plan for Clashindarroch was due for review, thus providing a real case for testing how our deliberative-support techniques might contribute to the participatory decision-making process based on stakeholder evaluation of ES. Drawing on the attitudinal analysis, the designs for future management of the land were developed with respect to layout and distribution of woodland species.

This information was then used to develop representations of scenarios of proposed changes, specifically in relation to the introduction of native woodlands in areas of pasture and moorland. Visualisation tools were used to test public preferences for different scenarios of change. The scenarios developed for the Clashindarroch area were: (i) maximising the proportion of native woodland species (i.e. biodiversity and supporting ES), (ii) maximising timber woodland (i.e. provisioning ES), and (iii) diverse land cover of moorland, forestry and agriculture (i.e. multiple ES).

These scenarios were presented in the mobile Virtual Landscape Theatre (VLT). This was followed by a phase of knowledge transfer and raising public awareness of the issues associated with ES for each scenario. Scenarios of change were presented to several audience groups in the VLT, each group following the 'drive-through' of the area. To illustrate alternative land management scenarios, the content of landscapes were 'switched' between, and selected features (e.g. woodlands, recreation facilities) were introduced or re-located as directed by the audience. These functions supported tests of audience values for ES under the scenarios of landscape change. Fig. 3 shows that four distinctive scenarios were identified concerning landscape change and management decisions connected to the ES provision in the Clashindarroch area. The output showed heterogeneity in the values relating to scenarios of change, with selected results as follows.

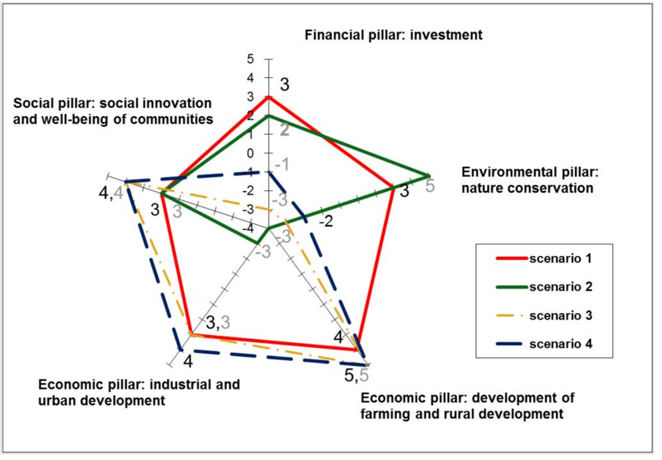

Figure 3.

Stakeholder evaluation of forestry changes in the Clashindarroch area.

The first ‘impartial’ scenario is, broadly, an equal distribution of peoples’ preconceptions of financial investment (+3); environmental pillar (+3); social pillar (+2) and economic pillar: farming (+4), and industrial/urban development (+2) in Clashindarroch. The second ‘environmental’ scenario reveals a strong environmental preference: with the environmental pillar (+5); social pillar (+2) and greater financial investment (+3). This scenario of change rejects the economic pillar, with (–4) for farming activity/rural development and (–3) for industrial/urban development. The third ‘economic’ scenario promotes the development of farming activity (+5) in combination with industrial/urban development (+4) in this landscape to meet the requirements of the social pillar (+4), whereas the importance of biodiversity conservation (–3) and financial investments (–3) is underestimated. The fourth ‘fair’ economic scenario is similar to the third one but with less pronounced carelessness to nature conservation and financial investment.

A second study area was a sub-catchment, the Tarland Basin in the River Dee catchment within north-east Scotland. This is an area with a current land use mix of agriculture (70%), woodland (21%), moorland (8%) and built (1%). Agricultural employment is 3%, 26% in tourism, 30% in the public sector, and 15% in financial services. Therefore, few local people have employment linked to provisioning ES of land/forest, but gain indirect benefits through landscapes managed for recreation, and residential quality of life. Objective scenarios of land use for 2050 were created using spatial modelling tools (Miller et al. 2006) and converted into visualizations of landscapes[*16]. They were presented in the VLT in venues remote from the area of study, i.e. in Birmingham, England (Fig. 4), and adjacent to the area - in Aboyne, Scotland (Fig. 5).

Figure 4.

VRT based stakeholder evaluation remotely from the study area (Birmingham, UK).

Figure 5.

VRT based stakeholder evaluation adjacent to the area (Aboyne, UK).

Stakeholder groups included policy-makers, land managers, foresters and farmers, the public, and young people under 20 years of age. Semi-structured group discussions were run with stakeholders voting on ES under each scenario, recording their values for different land use changes: woodlands, renewable energy, transport, housing, access, recreational facilities, and protection or enhancement of habitats, water quality and landscapes.

Findings from the Tarland Basin study indicated positive values for landscapes with a visible mix of land uses, sound stewardship, elements of perceived naturalness and visual diversity (consistent with the findings of Ode et al. 2009). From consultation events, commonality between audiences showed high values for amenity woodland adjacent to the village, quality recreation within the village, conservation interests, and recognition of risks to water quality with increased agricultural activity.

The local audiences were positive towards small-scale wind turbines associated with farming or communities. However, there were significant differences between stakeholder values with respect to medium-sized windfarms on hills north of the village. Those unfamiliar with the area (in Birmingham and Edinburgh engagement events) argued that renewable energy was a priority, highlighting open hilltops as opportunities for maximizing energy return. Those familiar with the area (in Ballater, Aboyne and Aberdeen engagement events) were conscious of the local significance of prominent hills and previous rejections of windfarm proposals.

Local preferences were more favourable towards amenity and broadleaf woodland and were unfavourable to coniferous woodland (associated with provisioning ES), compared to remote stakeholders. Participant groups from all areas favoured increases in mixed woodland (associated with the diversity of ES and landscape multi-functionality). When invited to identify benefits associated with land uses, participants local to the area recorded the negative and positive impacts of changes in individual woodlands due to felling and replanting. This reflected experience of actual uses of the woodland, loss of access, and long term regeneration, expressed in relation to changes in use through their lifetimes. For example, children in one family reported that they would not have access to a woodland, adjacent to their home, which had been used by two generations of the same family. The new felling plan covers the period for the remainder of their time at school. So the nature of the use of the woodland for their recreation would change through time as it is replanted and regrows. This was already changing the associations that different generations of the same family had with the woodland, and potentially between those of the current generations and those of the future.

We have found that subjective values vary between individuals and within cultural groups of people. This is in line with Christie et al. (2007) who found that distinct forest user groups (walkers, cyclists etc.) place different values on ES. Different people may have different perspectives on the values of different types of ES (Vermeulen and Koziell 2002). Thus, differentiating attitudinal 'types' can help improve the valuation evidence base for use in subsequent analyses.

The findings from the landscape study in the Tarland basin showed differences between those remote and local to the study area. This result points to the question of the relative importance of the values associated with different types of stakeholder, in terms of the governance and administration of an area, and the beneficiaries of its ES. Importantly, elected representatives, planners and the public reported positive views about using a combination of tools that we proposed during the workshops in planning adaptation to climate change (e.g. flood alleviation), public policy (e.g. increasing woodland, and managing existing woodlands), and testing public preferences for wooded landscape changes.

5. Discussion

Ecosystem valuation is generally considered to be useful in decision making, and when a good/service of ecosystems is excludable and rival, it is logical to value it economically. Society can make non-excludable and rival resources excludable, e.g. by setting aside some ES (Shabman and Scodari 2004). When, for example, privately owned land generates non-excludable and non-rival services, property rights can be limited to a total quota for excludable uses of land, allowing markets in uses that exceed that quota. Tradable development permits could cap total allowable development in an area whilst allowing landowners trade development rates so that the location of development (the value of ES in this location) is market determined (Stavins 2002).

However, the creation of artificial markets and use of off-setting (e.g. replacement) schemes are highly problematic (Spash 2010), especially when intrinsic value is considered (Bateman et al. 2010). Generally, off-setting schemes deal with 'items' (e.g. species), and sometimes with plant communities. However, ecosystems and their services are complex systems rather than the aggregate of components. Moreover, do woodlands created in remote areas have as much cultural value as those with easy access; and can the intrinsic value of natural habitats be wholly valued and offset?

Economic valuation is especially difficult in the field of biodiversity or landscapes, both as a result of their uniqueness and distinctiveness, and because of a shortage of robust primary valuations (Nature Valuation Organization 2005), and numerous uncertainties. However, it is possible to assign values to some public goods by using SP methods. This approximation is largely done to guide the decision-making towards sustainability, for instance, because outdoor recreation is expected to grow, as income increases, and as unique habitats become scarcer (Dixon et al. 1994, Glück 2002).

However, natural assets also have non-use values Fig. 1 that comprise human orientated (anthropocentric) intrinsic values, e.g. relating to cultural or spiritual benefits, the economic valuation of which is unlikely to be practical. Furthermore, it is impossible to capture the intrinsic value of biodiversity that exists irrespective of any value individuals might ascribe to it (O'Gorman and Bann 2008). Thus, whilst direct or indirect market instruments can provide effective tools in some cases, they do not work ubiquitously.

In some cases, economic values can be approximated and used to determine the level of taxes/fees to change behaviours that undermine conservation goals, or use of subsidies for activities that promote nature conservation (Baumol and Oates 1988). It may be possible to calculate values and then pay landowners for providing ES. Various PES schemes are becoming increasingly popular (Landell-Mills and Porras 2002, Pagiola et al. 2002, URS Scott Wilson Ltd. 2011). Creating market solutions may be less appropriate in considering ES as complex systems, under numerous uncertainties, and/or when conservation needs are site specific and conflict with existing property rights (Czech and Krausman 2001, Spash 2010). Market-driven decisions are particularly inappropriate when the ecosystems are of highest significance, e.g. those that contain endangered species.

Moreover, when there is an issue of critical natural capital, such as when ecosystems (or their components) are nearing critical thresholds (and 'tipping points'), valuing and managing ES cannot be driven by, and/or rely on economic variables. We agree with Daly (2007) that prices can respond to ecological constraints much more quickly than ecosystems can respond to economic variables. Therefore, the level of conservation should be price determining, not price determined. This particularly concerns non-marketed ES (i.e. public goods, having high intrinsic values).

Thus, because of the considerable complexity surrounding ES, and when it is unclear whether economic values represent a large share or a small fraction of the true TEV of unique and endangered ecosystems (Dixon et al. 1994) which are near thresholds, economic analysis alone will not be an appropriate solution (Costanza et al. 1997, Miller et al. 2009). The concept of the safe minimum standard[*17] and 'cautionary principle' should then be considered (Bishop and Romano 1998, Brondizio 2005). Given a range of uncertainties and potentially irreversible impacts of some decisions on certain types of ecosystems, and particularly on their intrinsic values, ethical and political choices should be made carefully.

The development of methods and tools to value ES for decision support has spurred scientists globally into interdisciplinary working, with concepts arising from areas of environmental assessments to improve public policy in addressing UN Agenda 2030 SD Goals (e.g. van Mansvelt 1997, Potschin and Haines-Young 2003, Münier et al. 2004, Daniel et al. 2012, Raymond et al. 2014). Valuation estimates aim to address benefits enjoyed by the global community, such as wildlife protection or carbon sequestration (and of option/non-use values, where possible) as well as of use values (to local communities and people on the ground). However, whether the aim is a more sustainable use of natural assets, knowledge exchange, or facilitating citizen actions, a framework which involves wider stakeholder participation needs to be adopted and adequately supported, enabling stakeholder and public awareness-raising, consistent with the UN Aarhus Convention (United Nations Economic Commission for Europe (UNECE) 1998) as well as the provision of information on ES values.

The decision-making process relies upon human and social factors, including: attitudes towards participatory decision making of those who design and facilitate the process; adequate resourcing of the process and capacity building to meet participatory objectives; the perception of the role of tools as participatory; and acceptability that participation will inform decisions. Technical factors include: the incorporation of relevant features in the visualization and valuation of ES at each stage and context of its use in the decision-making process; and the inclusion of appropriate levels of information content in the tools used to accurately communicate the information intended to those involved (Ball et al. 2007).

Technological advances continue to provide new approaches to representing landscapes of the past, present or future. However, existing approaches do not necessarily provide all the information required for ES valuation; interpretation of the consequences of environmental change, or how it should be tailored to different types of audiences. The rapid change in technology also means that gaps emerge in assessments of their effectiveness. This provides a challenge to further improvement of mixed methods to ensure that they are relevant, accessible and offer meaningful information for the ES valuation to aid the decision making processes.

Public policy, internationally, increasingly recognises and advocates more participatory, inter- transdisciplinary and holistic approaches for valuing ES (Kenter et al. 2011, Bunse et al. 2015, Kenter et al. 2015) to assist linking sustainable development goals with local level priorities and practices (Secco et al. 2017). Scientific research itself will benefit from inter- and transdisciplinary co-operation of scientific and stakeholder laboratories, and the range of requirements for bridging methodology, technology, and stakeholder engagement together.

The types of disciplines in such trans-disciplinary working typically are drawn from social and natural sciences, with prospective end-users (e.g. land and/or forest policy and management, and planning) from project outset and throughout the evaluation process. The geographic, cultural, institutional or demographic and other contexts might impair the application of certain approaches, or present challenges to achieving the objectives of the participation. So, questions arise as to how the methodologies used support and facilitate stakeholders to freely and effectively contribute to the advance of valuation methods for putting the use of ES on a more sustainable path (Eastwood et al. 2013[*18]).

In the examples presented, stakeholder values, objectives and preferences have been incorporated into an analysis of options for the future of ES provision and use. The process of research, scientific networking and communications with local and remote stakeholders and end users has led to the identification of design features, and criteria for the development and use of mixed methods and tools.

Further research is required on how, and to what extent, stakeholder evaluation and the use of mixed methods can affect decision making on the ground. Amongst the issues to be addressed are the best approaches to incorporating stakeholder perspectives into new strategies and programmes addressing SD, and whether the increased social capital created through participation in research and consultation translates into more effective implementation of policies.

6. Conclusions

Ecosystem services' valuation seeks to provide estimates of how ES contribute to the generation of income and wellbeing. It assists in identifying beneficiaries and providing evidence on the scale of benefits. Valuation helps to inform policy and land management decisions regarding resource allocation, management practices and use. It helps with informing appropriate levels of PES and determining whether a PES scheme is worth implementation. When used in combination with cost estimates, and linked to demand for ES, valuation can help resolve potentially conflicting decisions and guide the prevention of damages that inflict costs on society. Is it then paradox or Pandora's box for decision-makers?

When markets are explicit, the direct economic valuation (based on prices) is largely applicable. Market instruments often provide effective tools, but they do not work everywhere. Ecosystems are complex systems, intimately linked to the services they provide. Insufficient understanding of processes, complexities and inter-connections within ecosystems and human-environment relationships could result in neglecting to recognize that the value of a system does not equate to the sum of the value of its components, and that a system may collapse (and its services may be lost) if a seemingly minor building block of the system is overlooked. The following conclusions can be derived from this research and preceding literature survey:

- Reliable economic valuation depends on the robustness of methods, their appropriateness (e.g. valuation objectives and types of ES) and the accuracy of quantifying relationships between service provision and human wellbeing. Inaccuracies multiplied by uncertainties lead to the unreliability of valuation.
- The way of assessing ES depends upon the nature of services, research objectives, and on temporal and spatial scales of analysis. Valuation is case specific, context sensitive and contingent to a social context. Values can be modified by social context, and social context can be modified by changes in the provision of ES. Values vary between individuals and groups. They change through time and space. Spatial arrangements pose challenges. Also, valuation is carried out using contemporary knowledge, which is usually incomplete, especially concerning future values.
- As multiple benefits of ecosystems are increasingly important (Nijnik et al. 2010), obtaining their values becomes a high priority for decision-makers. A combination of socio-economic valuation techniques, both monetary and non-monetary, offers an appropriate framework for identification and explanation of ES related values. Suitable approaches can combine different theoretical concepts, and integrate analytical and participatory techniques (Wilson and Howarth 2002,Nijnik and Mather 2008) participatory techniques, GIS and visualization tools (Miller et al. 2006); and CVM and MAEI (Nijnik et al. 2009). The suggested approaches can complement each other and, based on consultation with the public, can offer credible means of performing ES valuations, and enabling consideration of changes in ecosystems through time. Innovative integration of techniques, such as group valuation, deliberative discourses, MCA and others can provide a complete and socially just assessment of the benefits that ecosystems provide to humans. This can result in obtaining the values that are observed from different perspectives of analysis and through different (subjective) perspectives of end-users.
- Valuation and stakeholder evaluation should be incorporated more widely into decision-making processes. However, the economic valuation of non-use intrinsic values of nature is unlikely to be practical. The safe minimum standard and the ‘cautionary principle’ should be considered for ecosystems characterized by high intrinsic values (Ciriacy-Wantrup 1968). Where public good issues are concerned,

much will depend upon government intervention and on a range of proper incentives (both economic and non-economic) provided to end-users towards changing behaviours for more sustainable uses of ecosystems.

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Author contributions

In this research, the authors combined the use of participation and visualisation tools with the economic evaluation techniques, including the Q-methodology. Prof. Maria Nijnik, an ecological and natural resource economist, developed all sections of the paper, except its visualization and stakeholder engagement components, which were prepared by Prof. David Miller.

Conflicts of interest

There are no conflicting interests pertaining to this paper, and if required, its authors are ready to assign to the publisher the copyright to this contribution.

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Endnotes

*1 Other types of services can be valued by market valuation techniques. Supporting services (e.g. habitat functions) can also be valued through direct market pricing (e.g. donations for conservation).

*2 The AC method considers the costs that would have been incurred in the absence of services. Examples are flood control by maintenance of wooded areas, reducing risks to property damage, or loss of agricultural production.

*3 The replacement cost method considers the costs of service replacement (or off-setting) with an alternative (e.g. human-made). An example is natural flood protection which can be (partly) replaced with artificial systems.

*4 Relocation costs are the expenses necessary to displace or off-set, for example the relocation of a cultural monument or recreation site from land at risk of flooding or contamination, or replacing trees that have been lost to house building.

*5 Non-excludability (or low excludability) typically comes with ill-defined property rights. Rivalness is a property of the ES in question, unrelated to institutions: for example, climate stability or flood control (URS Scott Wilson Ltd. 2011).

*6 TC considers travel costs as a reflection of implied value of the service.

*7 The HP method (i.e. property and other land-value approaches) implies that ES demand is reflected in prices which people pay for associated goods. For instance, house prices near green spaces usually exceed prices of identical homes near less attractive sites.

*8 SP is based on the idea of creating hypothetical markets and examining implicit preferences (Bateman and Willis 1999, Bateman et al. 2002, Pearce et al. 2002).

*9 TEV of forest ES in Britain amounts to £1,023m, with recreation of £393m; biodiversity: £386m; landscape: £150m; and carbon sequestration: £94m (Brown et al. 1993).

*10 Environmental Valuation Reference Inventory (EVRI) coordinated by Environment Canada is a comprehensive value (benefits) transfer database of over 2,100 valuation studies, which is available at www.evri.ca. More information about EVRI is at the Defra website http://statistics.defra.gov.uk/esg/evri/evri/default.htm.

*11 BT uses estimates in one location to infer benefits elsewhere or over a wider area.

*12 Guidelines recommend a discount rate of 3.5% and the use of different declining discount rates over the longer term (Nijnik et al. 2012b).

*13 A group-based deliberative method combining the features of citizens' juries (Nijnik et al. 2011) with SP techniques.

*14 A combined RP–SP method.

[*15] This study (Nijnik and Mather 2008) was extended to several countries (Vermeulen and Koziell 2002, Nijnik et al. 2013a, Nijnik et al. 2016). Relevant findings show that attitudinal diversity and value trade-offs associated with forest ES are dependent upon socio-economic and political conditions, and cultural standards.

[*16] More information on scenario development is available at: www.hutton.ac.uk/research/themes/realising-lands-potential/scenarios-and-land-use-futures.

[*17] For more of our considerations concerning the valuation of the ES delivered by nature conservation are seen in at: http://jncc.defra.gov.uk/page-6580 and concerning PES with re to forestry in the URS Scott Wilson Report (URS Scott Wilson Ltd. 2011).

[*18] For more information on shared, plural and cultural values and integrated valuation see special issues (2016) of *Ecosystem Services*.