Exploring the limits of monetisation in protecting ecosystem services

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Abstract
The monetary valuation of ecosystem services is gaining significant traction in policy and business communities. Several tools and decision making processes have been proposed including criteria such as scale, uniqueness and threat for when such monetary valuation should be used for the purpose of biodiversity conservation. This paper uses case studies of monetisation projects where the outcomes have been measured, at least to some extent, to explore the limitations and application of these criteria. It concludes that while there may be some aspects of monetisation that could be beneficial for biodiversity conservation there is currently limited evidence of the effectiveness of such schemes and indeed the majority are being applied in areas where the criteria would specifically exclude its use in isolation and require some quantitative minimum (or maximum) measurements to be applied through additional policy or governance measures.

Introduction

Traditionally tools to assess the effectiveness of biodiversity conservation policies focus on the ecological aspects of policy compliance, such as environmental impact assessment for habitat loss (Gontier et al., 2006) and biodiversity assessment tools (Hayek & Buzas, 2010). However, due to the increasing importance of sustainability principles in nature conservation, it is now widely recognised that any tool used in decision-making processes should also take into consideration aspects of economic development and social equity. While many argue that the use of valuation may alienate some stakeholders, is not able to fully reconcile cultural, economic, social and environmental justice (Matulis, 2014; Tallis & Lubchenco, 2014), and that any such approach is always going to be complex and involve conflicts (Pascual et al., 2017; Kenter, 2016), the role of ecosystem services as '*the benefits that people derive from ecosystems*' (Millennium Ecosystem Assessment, 2005), and how these can be valued, has become an integral part of decision making (Costanza et al, 2017; Costanza et al., 1997; Adams, 2014).

Tools for economic valuation of natural resources were first applied in the 1980s (Pagiola, 2008; Mitchell & Carsons, 1989) and several frameworks have been developed since then in order to conduct valuations of ecosystems (Bagstad et al., 2013, Costanza et al., 2017). A main benefit from the application of such techniques is that they improve and facilitate discussions on biodiversity conservation both at the scientific and policy level (Millennium Ecosystem Assessment, 2005; Kallis et al., 2013; Ruckelshaus et al., 2015). This is because they allow environmental contributions, and the consequences of their preservation, loss or restoration, to be expressed in terms that are comparable to other aspects relevant to land use choices (Randall, 2002; Hanley and Shogren, 2002).

Despite the wide development of valuation studies (Costanza et al., 2017) there are several concerns regarding their applicability and challenges have been identified (Adams, 2014). Interdependent ecosystems and ecosystem services are difficult to value in monetary terms (Gómez- Baggethun & Ruiz-Pérez, 2011; Vatn & Bromley, 1994) or to value separately due to their inherent complexity (Martín-López et al., 2008, Rodriguez et al., 2006). Equally, there is a high degree of uncertainty about the role and importance of specific parts of ecosystems and their functions (Farley, 2008). Here uncertainty characterizes the unknown probabilities of possible outcomes and is to be distinguished from risk which describes known probabilities of possible outcomes.

Valuations often rely on problematic economic assumptions implying normative judgments (Abson & Termanson, 2010). Markets often weight preferences by purchasing power (Farley et al., 2015) and are influenced by the power structures within and between the institutions in which they are made (Vatn & Bromley, 1994, Martinez-Alier & O'Connor, 2002; Røpke, 2005). Similarly, local scale studies are influenced by individual characteristics and perceptions such as the level of place attachment (Garcia-Llorente et al., 2012; Lopez-Mosquera & Sanchez, 2013) and personal environmental values (Spash et al., 2009; Lopez-Mosquera & Sanchez, 2012). In this context, some argue for a plural valuation approach, referring to the consideration of not only monetary values, but also ecological, social and cultural values (Kumar, 2010; de Groot et al., 2006; Farley, 2012; Kosoy & Corbera, 2010; Norgaard, 2010). However, there is some evidence that any attempt at valuation may in fact adversely impact pro-environmental behaviours through the reduction in the intrinsic value of nature (Goff et al., 2017).

Several authors have proposed certain conditions under which valuation studies can be useful especially on a policy level for nature conservation. Kallis et al. (2013) propose four criteria regarding the outcomes of monetary valuation as guidance for the decision on whether to value nature or not. These are summarised in Table 1. While additionality is an objective criterion, with the last three Kallis et al. (2013) apply also normative criteria.

Table 1: Four criteria for monetary valuation as proposed by Kallis et al. (2013).

Criteria	Definition
additionality	This refers to whether the valuation being implemented improves the ecological conditions of the ecosystem compared to the situation before and is additional to any other interventions that are in place
equality	More precisely, whether the valuation reduces inequality in the population who depend on the ecosystem
complexity-blinding	This implies that the valuation applied does not allow for other ways of valuation and other institutions to participate in the valuation
accumulation by dispossession	This refers to whether the valuation promotes expropriation of public goods.

Other normative criteria that need to be taken into consideration include whether an ecosystem is publicly or privately managed (Lockie, 2013) and the source of funding for any ecosystem valuation (Mokhiber, 1999). Objective criteria that are discussed in the literature include the specific ecological conditions, the level of substitutability of the ecosystem (Turner et al., 2003; Ekins et al., 2003) and also the level of ecological resilience (Brand, 2009). There is also a debate regarding the scale that these measurements should take place with some authors being more in favour of local measurements (Ninan & Inoue, 2013; Turner et al., 2003) as they are more focused and adjustable to the conditions of the ecosystem in contrast to large-scale programs (Wunder et al., 2008). At the global scale, ecosystem assessments have shown that a number of important thresholds have already been passed (Rockstrom et al., 2009).In the context of this debate, Turner et al. (2003) have proposed that the ability of ecosystems to be valued depends significantly on the scale of change. Small-scale changes, for example the loss of a local forest, can be meaningfully valued whereas global, large-scale changes may entail too severe consequences for human life, or are not feasible or substitutable, to be meaningfully valued. This argument is depicted as a demand curve for the flow of ecosystem services. If the provision of ecosystem services falls below a critical threshold, then it is no longer possible to value it meaningfully as further losses of this service would have too severe consequences. Defining which change is marginal is in itself one of the main challenges of valuation. In small-scale valuation studies considering a comprehensive set of ecosystem functions and uses, Turner et al. (2003) demonstrate, that valuation can indeed show that conservation in comparison to conversion is economically more beneficial. Taking this argument a step further, Farley (2008) integrates the concept of critical natural capital (Ekins et al., 2003), which is not substitutable and essential to human life, into this framework. He proposes three regions for the state of critical natural capital or the provision of essential ecosystem services. Abundant and resilient stocks of critical natural capital are appropriate to be valued. Stocks of natural capital closer to a critical threshold become inappropriate to be valued because of increasing uncertainties and their growing marginal importance (Fig. 1).

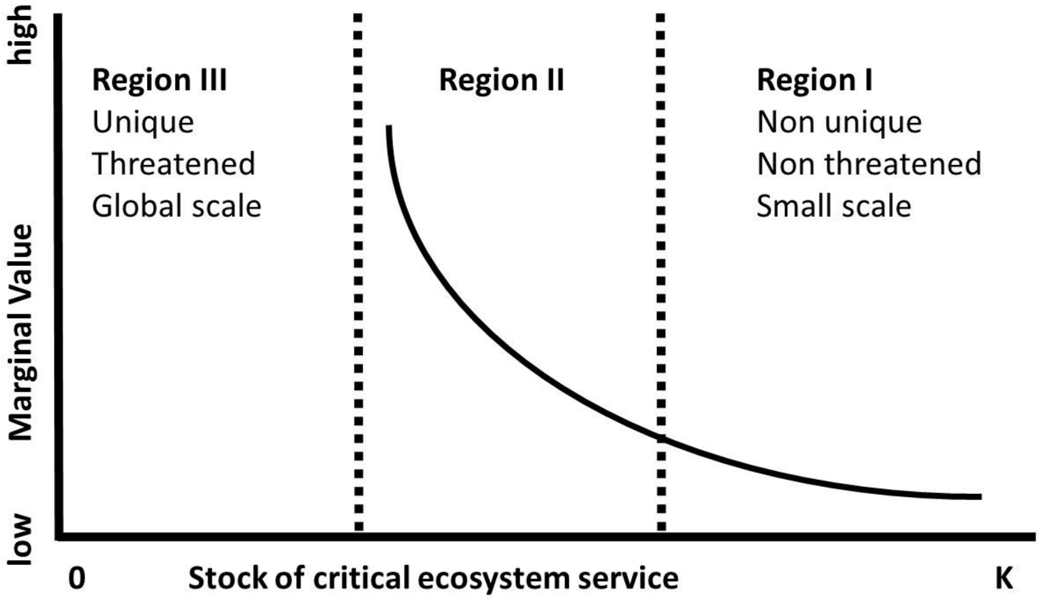

Figure 1: A conceptual framework for the valuation of natural capital stocks (adopted from Farley, 2008 and Turner et al., 2003)

Farley (2008) agrees with Turner et al. (2003) that valuation can, at least theoretically, support informing the decision to what degree ecosystems should be conserved but he mentions that in reality the market fails to allocate environmental goods. Especially at the state of the global ecosystem, which is not far from thresholds, valuation can no longer be applicable as a solution for the macro-allocation problem.

Daly (2007) suggests prices should adjust to the demand regulated by conservation, and not the other way around. When these minimum requirements are met, an eventual surplus may be converted to economic production. Here, valuation might actually be helpful to achieve these conservation requirements in the most cost-efficient way (Farley, 2008) and also to raise attention to the problem (Costanza et al., 1997).

As highlighted above, significant emphasis has been given to the design of valuation studies. However, the analysis reveals that *the conditions of the ecosystem* being valued are equally crucial in order to understand whether ecosystem valuation can be applied in an effective and meaningful way.

The aim of this paper, therefore, is to analyse what theory suggests as appropriate to be valued through monetisation and examine whether evidence can be found in the actual implementation of valuation studies in line with what theory suggests.

In the next section we outline the methodology behind the choice of case studies that have been examined in this paper. The case studies are then presented against three categories of monetisation. We then discuss the findings from the case studies and explore the criteria against which the applicability of monetisation for ecosystem conservation can be judged. Finally we draw some brief conclusions underlining new directions for research.

Methods

For this paper, a sample of case studies were chosen. Each case study needed to include a valuation method, and have ecosystems that had been directly affected or an impact on ecosystems was imminent. In addition, case studies needed to include evaluations of their environmental effectiveness. Evaluations were prioritised if they were carried out independently to the organisation involved in the ecosystem payments. In comparison to valuation studies, case studies on the actual outcomes from the implementation of ecosystem service valuations are rare (Laurans et al., 2013).

For the analysis we allocate each of these case studies against three common types of monetisation in use: Payments for Ecosystem Services (PES), integrating ecosystem service valuation in cost-benefit analyses (CBA) and compensation payments for environmental damages. We define each as:

- Payments for Ecosystem Services involve monetary exchange between two or more parties in order to prevent or lessen any damage to a particular ecosystem.
- Cost-benefit analyses involves one party integrating a monetary value for a given ecosystem services into a decision making process to assess possible outcomes of different interventions or schemes. No monetary exchange takes place.
- Compensation payments are made between two or more parties following pollution damage of ecosystems.

When ecosystem service valuations are implemented in PES schemes, they are mostly approximated by the opportunity cost of providing the services rather than by their inherent value (Wunder et al., 2008). For CBA and compensation payments a variety of valuation methods are applied, including ‘benefit transfer’, which extrapolates valuation studies for other locations or times as approximation for the value of the ecosystem in question (Reid, 1999; Schmidt and Wittich, 2014; Turner et al., 2003), as well as ‘contingent valuation’ which examines the value people would pay to avoid the loss of a certain ecosystem service (i.e. Carson et al., 2003).

The case studies are analysed against the criteria judging the appropriateness to be valued of the ecosystem. These criteria are deduced from Turner et al. (2003), who suggest that ecosystems that are not substitutable, for moral reasons, which adds a normative dimension to the otherwise rather objective criteria used here, or because their degradation would entail severe consequences for human survival, are not appropriate to be valued. This includes any ecosystem valuations of global scale. In contrast, ecosystems on a local scale with marginal, small changes in their condition can be valued meaningfully as long as they are not close to a threshold beyond which the existence of the ecosystem is threatened. If an ecosystem is close to a threshold their state is highly uncertain making it inappropriate to be valued (Farley, 2008). Accordingly, we define the criteria determining the appropriateness to be valued as *scale of change*, *threat* and *uniqueness*, the latter indicating the substitutability of the ecosystem. The criticality of the ecosystems valued is approximated by these criteria and a judgement on whether in practice the ecosystems are appropriate to be valued on theoretical grounds can be made. If the ecosystem does not fulfil one of the previously defined criteria, it should only be valued under restrictions. Based on this scheme, we can assign the case studies to the regions outlined by Farley (2008) or that by Turner et al. (2003) and seen in Fig. 1. We are aware that criticality can be the result of other than ecological characteristics, but we concentrate on ecological criticality.

Results

Table 2 summarises the case studies that were assessed, the type of ecosystem involved and the source of data on the evaluation of environmental effectiveness. The results summarise the different types of ecosystem service valuation in practice and highlight whether these were deemed to have had positive environmental outcomes or not. The case study results are then used to compare the specific outcomes with the theoretical proposals for the conditions under which the use of such schemes are appropriate or not. The case studies span a large variety of geographic scales and are presented in chronological order starting with the oldest first.

Table 2: The case studies assessed and evaluated for this study.

Project	Country	Year	Ecosystem	Valued ESS	Finance source/ Valuation method	Review
Payments for Ecosystem Services (PES)						
The Environmentally Sensitive Areas (ESA)/Countryside Stewardship Scheme (CSS)	UK	1986/1991	historic landscapes	biodiversity, recreation	government-financed PES	This an early example of agri-environmental PES programmes (Dobbs and Pretty, 2008). In the following years, it was complemented by various other PES programmes, one of which was the CSS, targeting farmland in England. Both were intended to protect naturally and historically valuable landscapes and habitats. Financed by the European Union (EU) and the UK government to similar shares, farmers received compensation payments to reduce agricultural intensity and to restore historic landscapes and features, such as stone walls and farm buildings (DEFRA, 2004, p. 12). By 2003, the CSS covered 530,620 hectares and the ESA 640,000 hectares in England, 10 % of agricultural land. However, the programmes failed to integrate a large amount of high-intensive agricultural lands or those areas that required larger changes in land-use and were more successful in preserving landscape (Potter, 1998). Various studies valuing the protection of the ESA programme, with the contingent valuation method, found the benefits to be much larger than their cost (Stewart et al., 1997). The CSS showed some additionality, mostly at the margin, according to interviews with the landowners and field surveys (Carey et al., 2003). It is noted that demonstrating real additionality, to be sure that the benefits are caused by the programme alone, is very difficult (Hanley et al., 1999). After 2003, the programmes were replaced by a new policy package under EU's Common Agricultural Policy building on what was learned in ESA and related programmes (Dobbs and Pretty, 2008).
Communal Areas Management Programme for Indigenous Resources (CAMPFIRE)	Zimbabwe	1989	grassland	biodiversity, recreation	hybrid PES	The Communal Areas Management Programme for Indigenous Resources (CAMPFIRE) in Zimbabwe was started in 1989. It is financed by user fees of safari operators and external donations. The Rural District Councils (RDCs) manage the 4.3 million ha of communal land covered by the project and pay dividends to the farming communities as incentives to adopt land-uses less harmful to wildlife. The CAMPFIRE project was established as part of a process to promote commercial use of wildlife as an alternative for the communities to agricultural land use that

						threatens wildlife populations. Environmental additionality could be achieved in terms of raised wildlife population and hunting revenues. However, a potential for 'leakage' exists and the success of the programme depends on the demand and market price for wildlife and landscape beauty, as well as on the relative abundance of wildlife in the respective area. (Frost and Bond, 2008).
Vittel watershed protection programme	France	1993	watershed/aquifer	water provision	user-financed PES	This is a successful example where single users, in this case water bottlers, have initiated PES schemes in order to protect their water spring area that would been otherwise contaminated by agricultural land-use. They pay farmers to secure water quality through environmental friendly land-use. Private water bottler Vittel offers cash payments and technical assistance to farmers in the 5100 hectare area surrounding the water spring. Payments are managed by an intermediary agency and are based on the individual opportunity cost of farmers undertaking the necessary changes. The water quality is frequently monitored and clearly improved thanks to the PES programme (Perrot-Maître, 2006).
PROFAFOR carbon-sequestration programme.	Ecuador	1993	forest	carbon sequestration	user-financed PES	This programme is organised by the PROFAFOR corporation which has paid landowners for afforestation and reforestation activities since 1993 (Wunder & Alban, 2008). PROFAFOR's financial source are Dutch electricity corporations that aim to offset their carbon emissions by buying carbon sequestration services from forests. The contracts are made with landowners of mostly highlands, but also in the coastal zone of Ecuador. 22,306 hectares of trees were planted so far. Most of the contracts have a duration of one cropping cycle after which the landowners receive financial incentives to replant the trees but are not obliged to do so. In comparison to surrounding areas, the tree cover planted under the control of PROFAFOR is regarded to be additional. However, the planted non-native species may have adverse effects on ecosystem services such as carbon sequestration.
Working for Water (WfW)	South Africa	1995	watershed	biodiversity, water provision	government-financed PES	In South Africa the invasion of alien vegetation had increased water scarcity. In water catchments in the Western Cape, the most affected area, 31 % of the total yearly water runoff are lost due to invasive species (Le Maitre et al., 2000). The programme Working for Water (WfW) was launched in 1995 with reducing unemployment rates as a primary objective. Previously

						unemployed are employed to clear invasive plants, especially highly water-absorbing species. The workers' salaries are conditional on their success. The programme is financed mainly by the government but also by water users, increasingly with water utilities that commission the programme to clear their catchment. Programme costs include not only the costs of clearing the invasive species but also social programmes such as health and training. The programme was successful in both objectives, reducing unemployment and securing native biodiversity and raised water provision (Turpie et al., 2008).
Pago por Servicios Ambientales (PSA)	Costa Rica	1997	forest	hydrological services, gas sequestration, biodiversity, recreation	hybrid PES	The oldest PES system in developing countries, it was established for watershed protection. By the end of 2005, it covered 10% of the country's forest area, in total 270,000 hectares, which were chosen according to biodiversity, poverty and water criteria. The programme was built on already existing infrastructure for payments for reforestation that facilitated the introduction of the PSA, including forest law protecting certain ecosystem services that can be attained from forests, such as water services, greenhouse gas sequestration, biodiversity and recreation. This is a hybrid programme, as it is financed by ecosystem service users through contracts with corporations, such as hydrological power generators, and through earmarked water and fossil fuel taxes. Additionally, the programme received funds from international NGOs. As in many PES schemes, an intermediary agency manages the cash payments to landowners for the maintenance and afforestation of tropical forests (Pagiola, 2008). Evaluation studies on the effectiveness of this type of PES implementation are widely divided as to whether the programmes contributed to actual net forestation in the country. This is because the effect of the programme is difficult to separate from other interventions and the studies often do not include high threat areas (Sanchez-Azofeifa et al., 2007; Arriagada et al., 2012, Muñoz-Piña et al., 2008).
Natural Forest Conservation Programme/ Grain to Green programme	China	1998/1999	forest, grassland	soil erosion control, carbon sequestration, biodiversity	government-financed PES	After the severe droughts and floods of 1997 and 1998, the Chinese government introduced several PES schemes for the protection and restoration of natural forests and grasslands. The NFCP includes a logging ban and compensation payments for corporate landowners as incentive to change their land-use from clearing the forest land to commercial forest management. Additionally, starting in 1999, the GTGP paid farmers cash and

						in-kind payments, as seeds, to incentivise them to convert steep cropland to grassland or forests (Liu et al., 2008). Having converted or afforested 21 million hectares by 2006, the GTGP is the largest PES globally. Both programmes have shown positive impacts on the ecosystem services they addressed through the retirement and reforesting of substantial areas, with outcomes including reductions in soil erosion and improvements in carbon sequestration capacities (Liu et al., 2008; Bennet, 2008). However, non-native species were planted, thus affecting native biodiversity (Liu et al., 2008). Studies have also suggested that both programmes may have the potential for 'leakage'. This refers to a situation where securing an ecosystem service in one location leads to the loss or degradation of ecosystem services elsewhere (OECD, 2010). For example, while Chinese timber production decreased after the programmes were established, timber imports increased in several years, potentially indicating 'leakage' of timber production from China to timber exporting countries, such as in the tropics (Liu et al., 2008).
Pimampiro municipal watershed-protection scheme	Ecuador	2000	watershed	water provision	government-financed PES	This was set up by the municipality of Pimampiro to protect the Palaurco River upper watershed, an area of 496 hectares that provides the inhabitants of the municipality with drinking water. A drought in 1999 and a newly built dam helped raise public awareness about the importance of the watershed. Included in the programme are the surrounding forests and grasslands of the right bank of the river, as they were increasingly converted to agricultural land. Landowners are paid for the maintenance or restoration of the original landscape and receive technical support as well as in-kind subsidies. The scheme is financed by a surcharge of water users, and interest from a water fund, as well as by the non-governmental organization Inter-American Foundation (Echavarría et al., 2004). The level of payments is designed to cover the opportunity costs of the landowners. Additionality in terms of water quality could not be well assessed, but in comparison with neighbouring areas, forest and grassland cover has increased strongly. However, agricultural profitability has also decreased during that time which may have contributed to lessening the impact of landuse (Wunder and Alban, 2008).

FONAG water fund	Ecuador	2000	forest, valleys	hydrological services	user-financed PES	This is the oldest example of PES schemes financing the protection of hydrological services of natural forests and valleys with the help of a trust fund. The fund is managed by an independent advisory board who chooses the areas to be restored within the 500,000 hectares area around Quito. The fund is financed by water users through the water utility company and used for restoration and training activities, often in the framework of community projects. The water fund appears to have successfully re-vegetated and reforested the area under control and stopped or reduced threatening agricultural practices such as cattle grazing while those continue in comparable areas (Buytaert et al., 2006).
Payments for Hydrological Environmental Services (PSAH)	Mexico	2003	forest	hydrological services	government-financed PES	The Payments for Hydrological Environmental Services (PSAH) was launched in Mexico in 2003. In order to decrease the growing water scarcity the state forest agency CONAFOR pays communities and landowners in national priority areas, in total 600,000 ha, for forest conservation. Cash payments are designed as to cover the opportunity costs of the farmers. However, similar to the PSA in Costa Rica, the programme appears to have low additionality as it targets low-threat areas because payments were decided to be distributed equally across the country. (Muñoz-Piña et al., 2008)
Los Negros Payments for ecosystem services	Bolivia	2003	watershed	water provision, biodiversity	user-financed PES	The declining forest cover and increasing water scarcity lead to conflict among water users in the watershed of the Los Negros Valley, which includes a cloud-forest, an important habitat for biodiversity, threatened by land conversion. The PES scheme covered 2774 hectares by August 2007 (Asquit et al., 2008). Upstream landowners receive in-kind-payments and technical assistance. The scheme contains six different payment types but the price is too low to engender real change (Wunder et al., 2008). In addition, mistrust between buyers and providers weakened the success of the programme. Non-threatened areas became protected under the programme, because farmers were able to select the areas to protect which they would not have cleared anyway (Asquit et al., 2008).
Northeim model project for agrobiodiversity	Germany	2004	grassland	biodiversity, recreation	hybrid PES, tendering	This is another example of where payments from a private foundation to farmers are set flexibly by forms of tendering. The project area covers 288 hectares that provides habitat for rich biodiversity, including threatened plant species. Environmental additionality may be high as the land-use intensity of

						participating farmers is reduced and 159 fields of grassland were generated (Bertke and Marggraf, 2004).
Wimmera catchment groundwater salinity control pilot programme	Australia	2005	watershed	hydrological services/ salinity control	government-financed PES, inverse auctions	This is an example of payments set flexibly by forms of tendering. The clearing of the natural landscape and use as grazing areas has distorted the salinity balance of the groundwater in the catchment area. Landowners in the upper Wimmera catchment, an area of 28,000 hectares, were paid by the Australian government to reduce the impact of their land-use. Payments were allocated by an inverse auction during which landowners offered a price for providing salinity reduction. The Catchment Management Authority (CMA) then chose participants who offered one expected unit of salt reduction for the lowest price. Although environmental additionality was designed to be high, there is no proof for the overall scheme's actual effectiveness (Shelton and Whitten, 2005).
European Union Emissions Tradion Scheme (EU ETS)	EU	2005	emissions	greenhouse gas sequestration	user-financed PES, auction	This is probably the most popular example of user-financed PES. In carbon markets, carbon sequestration services are assigned a market exchange value to make them comparable with emission reductions. The price depends on the cost for emission reductions. In comparison to other PES schemes, a carbon market is highly abstracted from ecosystem services (Kosoy and Corbera, 2010). The EU ETS covers 45% of all EU emissions. More than 11,000 power and industrial plants and airlines have to buy emission allowances for the amount they emitted in the previous year. During the current trading period, 2013-2020, 57% of all available allowances are allocated by an auction. The allowances are reduced every year (EC, 2016). More than 80 % of the revenues from the auction are planned to be used for climate related projects (EC, 2017). However, an excessive oversupply of emission allowances in the scheme makes the system ineffective as an incentive for companies to invest in strategic emissions saving. In order to control for this surplus, the market stability reserve is planned to reduce the available allowances from 2019 on (EC, 2016).
Slug it out	UK	2015	watershed/aquifer	water provision	user-financed PES	The programme was introduced to farmers in six small priority areas, over a total 7,500 hectares (Anglian Water, 2017), to avoid a certain pesticide (metaldehyde) and comply with the EU water directive. Farmers receive payments to cover the cost of substituting the pesticide with an alternative as well as bonuses for successfully reducing the contamination of the ground

						water. In some of the catchments, the contamination was successfully reduced below the threshold required by the directive. Additionally, some farmers decided to change practice on all their land, doubling the size of the impact area. However, in some areas, where multiple uses took place, the impact of the programme is less clear (Anglian Water, 2015).
Cost-benefit analysis integration						
Bala dam proposal	Bolivia	1999	watershed	biodiversity, habitat, greenhouse gas sequestration, recreation, existence value	CBA, benefit transfer	The construction of the power generating Bala dam was initially proposed in the late 1990s. The Conservation Strategy Fund (an international NGO) conducted a CBA including social and environmental impacts of the project, the latter being valued using benefit transfer. In the course of the construction work more than 200,000 hectares with globally important high biodiversity would have been flooded. Using a low- and high-cost scenario, the analysis came to the result that the project would cause a net financial loss of more than 400 million US dollars. The project was cancelled (Reid, 1999).
Thornton Creek Confluence Improvement project	USA	2014	watershed	biodiversity, habitat, water provision, flood control	CBA, benefit transfer	For this project, a CBA was conducted in order to identify the best management option of the Thornton Creek watershed in Seattle, USA, which is an area affected by frequent storm water flooding. The watershed provides water for a 12-square-mile region with relatively high biodiversity for an urban region, including a threatened salmon species. Seattle Public Utilities engaged economists from Earth Economics (a Non-Governmental Organisation) to conduct a CBA to work out whether to replace a water-absorbing metal pipe by a confluence floodplain. Using their Ecosystem Service Valuation Toolkit, the scientific team extrapolated primary valuations to the ecosystem services in the Thornton Creek watershed and examined the confluence floodplain to provide the most economic, social and environmental benefits. Building the confluence floodplain was agreed in 2014 (Schmidt and Wittich, 2014).
Belize Integrated Coastal Zone Management Plan	Belize	2015	coastal area	biodiversity, habitat, recreation	CBA, modelling ESS values	This coastal management plan was developed using a spatial model analysis. With a risk assessment examining threats the ecosystem services face and by modelling different management scenarios, a balanced plan emerged that combined protection of coastal ecosystems with safeguarding its economic benefits. The respective model scenario suggested that the revenues from lobster fishing and tourism would

						increase. The plan is implemented by the Coastal Zone Management Authority and Institute (CZMAI, 2016; Arkema at el., 2015).
Compensation payments						
Exxon Valdez oil spill	USA	1989	coastal area	biodiversity, habitat, recreation	contingent valuation	The most famous oil spill cases is that of an Exxon Valdez tanker in 1989 that severely distorted the ecosystem on the coast of Alaska. It had been the largest oil spill in US history up to that point (Navrud and Pruckner, 1997). The state of Alaska conducted a contingent valuation study to assess the damage value in terms of what people would pay to avoid a similar catastrophe. The final sum of damage payments did not deviate much from the actual value proposed by the study. Exxon Valdez had to pay 1 billion US dollars for restitution and resource damage to the state of Alaska and the US government. In addition, Exxon Valdez spent more than 2 billion for the restoration of the area (Carson et al., 2003).
Prestige Oil spill	Spain	2002	coastal area	biodiversity, habitat, recreation	contingent valuation, passive use value	During this oil spill, over 1,500 km of shoreline and ecosystems were severely damaged, including in France and Portugal. It is recognised as one of the largest environmental catastrophes in Spanish history. With economic and contingent valuations, the total economic value of the loss through the oil spill was calculated to be 4.3 billion Euro. The amount also served as a reference point for courts to claim compensation payments (Loureiro et al., 2009). In 2016, after years of negotiations, the highest Spanish court agreed a much lower payment (1 billion Euro) but obliged the insurance company of the tanker that caused the oil spill to compensate for all damages. This was the first court decision in Spain considering passive use values (Von Bertrab et al., 2016). Passive use values derive from the non-use of an ecosystem such as the value associated with its existence.
Chevron Texaco pollution	Ecuador	2011	Amazonas Forest	biodiversity, habitat, recreation	variety of environmental damage valuation studies/ court decision for damage restoration	In the case of the pollution of the Amazonas forest by Chevron-Texaco in Ecuador, a variety of valuation studies were conducted to determine the amount of the penalty payment to be made. In 2011, the court ordered Chevron-Texaco to pay almost 9 billion US dollars which were used to restore the environmental damage and to build up cultural and health programmes for the affected communities (Kallis et al., 2013). The payment did not represent the full value of the ecosystem losses and deaths but could nevertheless be used to partly restore the damage as well

						as to serve as a deterrent (Martinez-Alier, 2011; Kallis et al., 2013).

Discussion

The case studies cover a variety of methods for the valuation of ecosystem services. Explicit valuation studies were undertaken for damage compensation payments, some even included passive use values which are measured by people's willingness to pay to prevent further environmental catastrophes (von Bertrab et al., 2016). Some studies used benefit transfer and extrapolated valuations of other ecosystems, such as the cost-benefit analyses (Schmidt and Wittich, 2014; Reid, 1999). The PES schemes valued ecosystem services either through flexible market interactions such as tendering where landowners offer the provision of certain ecosystem services for a certain price (Shelton and Whitten, 2005). Other PES set the payments for ecosystem service providers by approximating their opportunity costs and designing uniform payments based on this (Pagiola, 2008) or differentiating payment schemes (Asquit et al., 2008).

Here we analyse the case studies against the three criteria of *scale of change*, *threat* and *uniqueness*.

Scale of change

The case studies cover a wide range of spatial scales, from 288 hectares to 21 million hectares (Bertke & Marggrad, 2004; Liu et al., 2008). The median spatial scale is 28,000 hectares.

Those valuation studies covering only one region that is neither threatened by a threshold nor weakly substitutable are appropriate for ecosystem service valuation. To assess whether the ecosystems under consideration were appropriate to be valued, the case studies are categorized in terms of regions or ecosystems covered. Applied to PES schemes, undifferentiated payments are counted as large-scale valuations, whereas payment contracts made individually for every farm are regarded as local valuations. In the sample, some studies valued one certain ecosystem or similar ecosystems in one region, i.e. the region surrounding a watershed or catchment area (Schmidt & Wittich, 2014; Wunder & Alban, 2008; Asquit et al., 2008), while others valued different types of ecosystems simultaneously.

The most prominent example where aggregated valuations were made at a uniform price is the EU ETS. Auctions under the EU ETS are generally uniform price auctions and allocate large amounts of allowances for the same market clearing price. For example, on September 14, 2017, a total 5,329,500 allowances were auctioned (European Energy Exchange AG, September 15, 2017). As already highlighted, the EU ETS has been ineffective in encouraging companies to make strategic investments for large scale emissions reductions. The damages evaluated in the course of claiming compensation payments in the cases analysed are also examples for aggregated valuation. The damage caused by the Prestige oil spill was evaluated by aggregating the losses of the whole area, including shorelines of three countries (Loureiro et al., 2009). In the framework of the PSA in Costa Rica the usage of uniform payments for participating landowners lead to one of the shortcomings of the programme (Pagiola, 2008). The payments were too low to induce owners of threatened areas to cease their land-use.

In contrast, the inverse auctions undertaken in the framework of the Wimmera catchment programme and the Northeim model project resulted in individual payment contracts (Shelton & Whitten, 2005; Bertke & Marggraf, 2004). PROFAFOR, as well as the payments under the PES in the Vittel catchment area, are agreed individually (Wunder & Alban, 2008; Perrot-Maître, 2006). The CBAs discussed evaluate ecosystem services at a rather local scale. For the spatial plan of the Belizean coastal zone benefits were modelled separately for each of the nine planning regions (Arkema et al., 2015).

Some PES schemes do not offer individually set payments but differentiated payments depending on the quality of the conserved area. Farmers receive higher payments for the preservation of original biodiversity or more important landscapes, such as in the PES schemes for certain watersheds (Wunder & Alban, 2008; Asquit et al., 2008) and in the schemes in China and the GTGP and NFCP in the UK (Liu et al., 2008; Dobbs & Pretty, 2008). In the above case studies there is some emergent evidence that supports the assertion that ecosystem service valuation should only be used at a local level if it is to be effective. For most case studies, the scale of change valued cannot be identified with certainty. This relates to the inherent problem of valuation which is the identification of marginal changes (Turner et al., 2003). However, most of the projects offer regionally differentiated valuations, but only few of them include individual valuations on the local level.

Threat

For all case studies justification can be found that the ecosystem monetised is somehow threatened, or that it comes close to its critical threshold (Farley, 2008), except in the case of the PES of PROFAFOR for which no information about the state of the ecosystem is given (Wunder & Alban, 2009). In the case of compensation payments, the evaluated damage degraded the ecosystems substantially and brought them close to this threshold or even beyond it. The discussed CBAs compare different management options with regard to several objectives, including the reduction of threats to which ecosystems are exposed.

However, it is possible to nuance the kind of threat to which respective ecosystems are exposed. In some cases this includes direct and recent disturbances, such as regular floods in the Thornton Creek watershed or China (Schmidt & Wittich, 2014; Liu et al., 2008). Additionally, endangered ecosystems or endangered species situated in the respective areas covered by the scheme would have been likely degraded without the implementation of certain conservation measures. Examples include the Belize Barrier Reef which was recently added to the list of World Heritage Sites in Danger; or the Chinook salmon in the Thornton Creek watershed (Schmidt & Wittich, 2014).

In cases such as in the UK, where historic landscapes have been increasingly changed and adjusted to modern agricultural land-use over the last decades (Dobbs & Pretty, 2008), there is no indication that this development urgently threatens their existence. In other cases, projects had been initiated to address threatened areas but the payments offered appeared to be too low for inducing landowners to participate and to stop their land-use, such as in the case of Los Negros in Bolivia (Pagiola, 2008; Muñoz-Piña et al., 2008).

With this nuanced perspective, our sample consists of 15 studies where the provision of the respective ecosystem service is threatened; and six where the ecosystem is in a state further from the threshold, but with a tendency of moving towards it before the start of its valuation. What can be seen is that more often than not, ecosystem service valuation is used to protect threatened species, although often these schemes also include other safeguards. In the case of compensation payments, the scheme is not used for prevention of damage and therefore should be treated separately within the context of ‘threat’.

Uniqueness

It is difficult to define the uniqueness of an ecosystem. However, it is possible to approximate the global role an ecosystem plays through arguments for its weak substitutability. Some of the ecosystems covered by the valuation projects are very rare, such as coral reefs at the coast of Belize, or serve as habitat for endangered species, such as the Chinook salmon in the Thornton Creek watershed (Arkema et al., 2015; Schmidt & Wittich, 2014).

In addition, some ecosystems are of global importance owing to their capacity of providing a certain, globally important service, such as the Amazon forest, which plays a key role for the global climate due to its enormous gas sequestration capacities (Flannery, 2005). The sample contains one valuation referring to the Amazon forest, referring to its pollution by Chevron Texaco (Martinez-Alier, 2011).

Assessing conditions for monetisation

Based on the categories of scale, threat and uniqueness, it is possible to assign the case studies to the three regions outlined by Farley (2008) or that by Turner et al. (2003) and seen in Fig. 1. If an ecosystem covers either a massive scale or is globally unique, both make it globally important. Equally, if it is very close to a threshold beyond which it can no longer sustain itself, the ecosystem is too critical to be assigned to a stable and meaningful value (Farley, 2008).

According to the framework presented in this paper, monetisation or valuation should only cover ecosystems that are in a noncritical state and are able to provide abundant ecosystem services. Ecosystems in this first region are not threatened, not unique and not of a globally important scale. If an ecosystem only meets one or two of these criteria, some kind of quantitative limit has to be applied in order to safeguard the sustainable minimum standard of the ecosystem or ecosystem service (Farley, 2008). These ecosystems represent the middle region (Region II). The cases that fulfil none of the criteria would be assigned to the region where valuation is impossible (Region III), as their value is infinite and their restoration or conservation is critical (Farley, 2008).

From the presented case studies, a few do fall into the region which are regionally specific, non-unique and non-threatened and therefore a monetisation through ecosystem service valuation may be appropriate. For example, the GTGP and the NFCP in the UK can be assigned to the region where valuation is appropriate without restrictions, because the areas are not urgently threatened and the payments are at least differentiated depending on the country (Dobbs and Pretty, 2008).

However, monetisation has also been applied to threatened ecosystems, such as the coral reefs within the Coastal Zone of Belize (Arkema et al., 2015), as well as to the watersheds and catchment areas, or non-local ecosystems such as the EU ETS (EC, 2016). Therefore, their valuation should have been complemented with some other form of restrictions (Farley, 2008). Even though valuation without a minimum standard should not have been used in the middle region (Region II), in some, rather small scale projects, it appeared to have generated environmental additionality (Perrot-Maître, 2006; Wunder & Alban, 2008).

Ecosystem services valued for compensation payments would all fall into the left region (Region III) of Fig. 1 because of their enormous scale and the severe impacts of their degradation making them impossible to be valued meaningfully. However, as Kallis et al. (2013) note, compensation payments may achieve additionality, if they are used to restore environmental damages and if the risk of being charged a penalty fee deters companies from polluting the environment.

Although financial practice contradicts theory when a range of valuation schemes are applied to threatened ecosystems, some of them are embedded in a broader political strategy which indeed includes a kind of quantitative control element like a minimum standard, which Farley (2008) would suggest for the threatened ecosystem in the middle region (Region II). For example, both the PSAH in Mexico and the PSA in Costa Rica were implemented at the same time as a logging ban (Muñoz-Piña et al., 2008; Pagiola, 2008). Furthermore, the “Slug it out” project was introduced in order to comply with the drinking water directive that restricts the concentration of pesticides in water (Anglian Water, 2015).

To assess whether a programme was effective in terms of protecting the addressed ecosystem is not straightforward for two reasons. Firstly, defining a counterfactual is problematic. Most of the assessments did not define counterfactuals but compared the developments of the respective areas with neighbouring areas (Wunder & Alban, 2008). Linked to the need for a counterfactual, measuring effectiveness implies the need to measure additionality. Our case studies show that very few programmes have incorporated a detailed and systematic effort to formally quantify additionality, especially ex ante (Wunder & Alban, 2008). Secondly, the relatively recent implementation of these programmes makes it difficult to properly assess their long-term effectiveness, as this would require a longer time horizon (Dobbs & Pretty, 2008, Liu et al., 2008). With some contracts still ongoing, it is unclear what will happen when they eventually end and whether conservation or restoration will continue after payments stop (Wunder & Alban, 2008; Bennet, 2008).

Based on these findings we would like to propose certain directions for future research. New techniques need to be developed and incorporated in current valuation methodologies in order to assess their effectiveness. These techniques need to focus both on the short and long-term impacts on the ecosystem and the relevant wider socio-economic system taking into consideration critical issues such as additionality and equality (Kallis et al., 2013). Furthermore, in several of the case studies examined in this paper it was impossible to assess their effect separately from other factors which may have influenced the effectiveness, such as other political instruments implemented at the same time (Pagiola, 2008) or other market mechanisms (Wunder & Alban, 2008).

Conclusions

This paper explored the type and effectiveness of different ecosystem service valuations as applied in practice through the use of case studies. These were compared against the theoretical understanding as outlined by Farley (2008) and Turner et al. (2003). In particular the three conditions of scale, uniqueness and threat were identified as conditions which could indicate the appropriateness of application of monetisation towards ecosystem services. Monetisation should only be used as a standalone valuation method when ecosystems are at local scale, non-unique and are not threatened. If one or two of these conditions are not met, then additional measures to protect the ecosystem should be implemented such as minimum standards or further policy protection. If none of the conditions are met, ecosystem valuation is inappropriate.

In practice, the majority of the case studies include ecosystems that do not meet all three conditions. However, the case studies do also include additional measures to protect the underlying ecosystem. Therefore, determining the effectiveness of the particular implementation of monetisation or valuation is difficult as outcomes can be attributed to both the implementation as well as the additional policy measures. Indeed, in some cases the outcomes of the entire intervention are uncertain. The absence of ecosystem assessments prior to the launch of a valuation service or the absence of counterfactual information also makes it difficult to evaluate impact. However, evidence does exist that some practice has resulted in higher quality ecosystems, in particular in watershed management.

Besides the problem of measuring effectiveness, uncertainty with regard to how close an ecosystem is to its threshold, or even the existence of the threshold, makes the categorisation of when valuation is appropriate highly uncertain (Farley, 2012). Leopold (1993) argues that as we do not know with certainty which parts of the ecosystem are essential, it is straight forward to treat everything as critical.

We note that while the framework we propose in this paper rules out ecosystem valuation when the conditions mentioned above are not met, monetisation can inform policy-makers of values they would have not otherwise considered and thus making them more likely to protect ecosystem services. In some cases, when the costs of, for example, protecting an endangered species, clearly outweigh the benefits of this protection, but in which counterfactuals cannot be assessed, protection of the ecosystem services, regardless of monetary values, could be considered the actual counterfactual. However, in this case the decision to protect or not protect the ecosystem service should not be based purely on a valuation of the costs and benefits.

While some evidence exists that valuation can be effective in certain circumstances, this has rarely been the case in practice without additional policy measures to protect threatened ecosystems. Given this conclusion, we recommend that the current drive to rapidly expand valuation services and ecosystem evaluation, in particular through voluntary private sector measures, is highly risky and should be considered with much greater care. At the very least, full and thorough evaluations of their effectiveness need to be captured and shared so that lessons can be learnt and further refinements of such services can be made.

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