



Purposes and degrees of commodification: Economic instruments for biodiversity and ecosystem services need not rely on markets or monetary valuation



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ABSTRACT

Commodification of nature refers to the expansion of market trade to previously non-marketed spheres. This is a contested issue both in the scientific literature and in policy deliberations. The aim of this paper is to analytically clarify and distinguish between different purposes and degrees of commodification and to focus attention to the safeguards: the detailed institutional design. We identify six degrees of commodification and find that all ecosystem services policies are associated with some degree of commodification but only the two highest degrees can properly be associated with neoliberalisation of nature. For example, most payments for ecosystem services (PES) are subsidy-like government compensations not based on monetary valuation of nature. Biodiversity offsets can be designed as market schemes or non-market regulations; the cost-effectiveness of markets cannot be assumed. To avoid the confusion around the concept 'market-based instrument' we suggest replacing it with 'economic instruments' since relying on the price signal is not the same thing as relying on the market. We provide a comprehensive framework emphasising the diversity in institutional design, valuation approaches and role of markets. This provides flexibility and options for policy integration of biodiversity and ecosystem services in different countries according to their political and cultural context.

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1. Introduction

Valuation of ecosystem services (ES) and the use of economic instruments are increasingly becoming part of the international discussions on scaling-up biodiversity financing. The Convention of Biological Diversity (CBD) states that "biodiversity values" should be integrated into development strategies, planning processes, national accounts, and reporting systems (Aichi Biodiversity Target 2) and calls for the elimination of harmful subsidies as well as the development of "positive incentives for the conservation and sustainable use of biodiversity" (Target 3).¹

The focus on biodiversity values and 'Innovative Financial

Mechanisms' (IFMs) has for some actors become extremely controversial within the CBD process, especially the use of economic instruments like payments for ecosystem services (PES) and biodiversity offsets.² Without appropriate institutional arrangements that safeguard (ensure) biodiversity and equity, there is a risk that economic instruments, as well as other types of policy instruments, will not contribute towards the three CBD objectives (Ituarte-Lima et al., 2014). These are (i) conservation of biological diversity, (ii) the sustainable use of its components, and (iii) the fair and equitable sharing of the benefits arising out of the utilisation of genetic resources. The CBD calls for a broader governance approach to valuation and financing so that the IFMs do not "undermine achievement of the Convention's three objectives" (CBD, 2010). This motivates a focus on safeguards, which we define as the specific factors in the institutional design and implementation

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¹ <http://www.cbd.int/sp/targets/>.

² E.g. the CBD COP-10 meeting in Nagoya, Japan, October 2010, failed to agree on Innovative Financial Mechanism (Biodiversity Financing Mechanisms) which motivated a special Dialogue Seminar in March 2012 to resolve these issues, see Farooqui and Schultz (2012).

procedures aimed to ensure good environmental and social outcomes.

The risks of using market-based instruments for financing biodiversity and ES range from ethical arguments about transforming human-nature relations by “commodity fetishism” (Kosoy and Corbera, 2010) and crowding-out moral obligations as motives for nature protection e.g. (Luck et al., 2012), to instrumentalist arguments focused on the efficiency and equity of the processes and outcomes of such schemes e.g. (Corbera et al., 2007). This is often associated to neoliberalism. Based on McAfee and Shapiro (2010), we define neoliberal ES policies as instruments designed on the premise that the market allocates scarce ecological resources more efficiently than ‘command-and-control’ regulations and treaties.

Normative framings of ES and commodification are important but sometimes become an obstacle to addressing the empirical instrumental question of how different economic incentive schemes actually perform (Gómez-Baggethun and Ruiz-Pérez, 2011: 622; Dempsey and Robertson, 2012). In this paper we emphasise an instrumentalist approach. The aim is to analytically clarify and distinguish between different purposes and degrees of commodification and to refocus attention on the detailed institutional design of policy instruments, in particular controversial economic instruments.

This paper interrogates the concepts of commodification, valuation, and markets in order to build a framework for policy integration of ES, i.e. addressing and integrating ES concerns in sector policy making (Nilsson and Persson, 2003). Based on this framework, we further analyse the foundations for payments for ecosystem services (PES) and biodiversity offsets, to explore a menu of options for tailoring these instruments to accommodate country-specific concerns. Since biodiversity and most ES are much more difficult to measure than carbon dioxide and other emissions, our framework of commodification is not directly transferable to pollution quotas, carbon markets or emission-trading systems.

2. Six degrees of commodification

Commodification of biodiversity and ES means, broadly speaking, the expansion of market trade to previously non-marketed areas of the environment (Luck et al., 2012). This is often described as a process related to idea of commensurability underlying monetary valuation (Aldred, 2002; Vatn, 2009). Kosoy and Corbera (2010) identify three necessary stages in commodification: first defining an ecosystem service, second assigning an exchange-value, and third create a market. Adding property rights, Gómez-Baggethun and Ruiz-Pérez (2011) identify four main stages of commodification: (i) economic utilitarian framing, (ii) monetary valuation, (iii) appropriating the value of ES through formalisation of property rights, and (iv) commercialisation, i.e. market trade by PES or offsets.

However, these stages need not be consecutive and the process is not necessarily unidirectional or irreversible (Gómez-Baggethun and Ruiz-Pérez, 2011), hence we use degrees rather than stages. Based on Muradian et al. (2010: 1206), we define the degree of commodification as the extent to which the value of biodiversity or an ecosystem service has become a tradable commodity.

We find it useful to analyse commodification in terms of policy integration and in this more empirical context at least two more degrees need to be added, as well as the zero degree (policy integration without commodification). The justification for these degrees is discussed below. Empirically, the degree of commodification is a matter of the institutional design of a particular policy instrument. The stated purpose of introducing this policy

instrument is tailored to the specific ideological orientation of the government and can be observed e.g. in national legislation and underlying government bills. Together, the degree (institutional design) and stated purpose (justification) influence the legitimacy of the instrument. When the detailed institutional arrangements are analysed we find that what are generally described as PES and biodiversity offsets may involve different degrees of commercialisation and hence commodification. Hence, we find it useful to distinguish between six degrees (plus the zero degree) of commodification:

0. “No commodification” (zero degree) includes intrinsic or relational appreciation of ecosystems, in which the rationale for protecting nature is nature itself. ‘Relational’ include indigenous cosmologies emphasising reciprocity and cyclical processes (MA, 2005a: 86–87) as well as interaction for reasons of spirituality and even subsistence farming (Turnhout et al., 2013). The policy options include long-established policy instruments with no commodification such as national parks and nature reserves as well as the more recent notion of ‘environmental flow’ in water governance³ when justified by deontological ethics (moral duties) or nature’s intrinsic values. This also includes approaches linked to the rights of nature or the inalienable rights of indigenous peoples to sustain their cultural and sacred practices (the viability of bio-cultural sites for cultural and sacred practices are often linked with protecting intrinsic values of biodiversity).

International legal and policy options involving no commodification include new legal paradigms recognising rights of nature which have been characterised as “Earth Jurisprudence” (Burdon, 2011). As part of this, Higgins et al. (2013) question the belief that the market will provide effective and efficient remedies and have proposed “Ecocide” as part of international criminal law which would aim to pre-empt, prevent and prohibit mass damage, destruction or loss of ecosystems whether committed during or outside of war-time as well as impose an associated legal duty of care upon persons in positions of superior responsibility. Other international initiatives include the World People’s Conference on Climate Change and the Rights of Mother Earth,⁴ Global Alliance for the Rights of Nature, and the Community Environmental Legal Defence Fund (Daly, 2012). At the national level, examples of intrinsic values include the recent Constitutional recognition of the rights of nature in Ecuador (Burdon, 2011)⁵ and rights of Mother Earth and good living in Bolivia’s Law 071 and Law 300.⁶

1. The first degree of commodification arises under the instrumental (or even economic) framing of nature, though without explicit efforts at valuation. The separation of humans and nature and hence an instrumental view of nature can be found already in the works of Francis Bacon (1561–1626) (Merchant, 1980). The expansion of this instrumental framing to include ecosystem processes was popularised by e.g. Ehrlich and Mooney (1983) and Daily (1997). Since The Millennium Ecosystem Assessment (MA, 2005) ecosystem services and instrumental framing have become mainstream in environmental policy although this may not involve monetary (economic) valuation

³ Here understood as “how much water is needed by a river and when, in order to support the river’s basic ecological functions” (Groenfeldt and Schmidt, 2013: 2).

⁴ <http://pwccc.wordpress.com/2010/04/24/peoples-agreement/> Accessed 27 November 2014.

⁵ Ecuador became the first country to adopt a Constitutional provision endowing nature with inalienable rights. The Constitution recognises that “Nature or Pachamama, where life is reproduced and exists, has the right to exist, persist, maintain itself and regenerate its own vital cycles, structure, functions and its evolutionary processes” (Burdon, 2011).

⁶ See <http://www.ine.gob.bo/indicadoresddh/archivos/alimentacion/nal/Ley%20N%C2%BA%20071.Pdf> and <http://www.planificacion.gob.bo/sites/folders/marco-legal/Ley%20N%C2%B0%20300%20MARCO%20DE%20LA%20MADRE%20TIERRA.pdf>

Table 1
Degrees of commodification in terms of policy instruments for biodiversity and ecosystem services.

Degree of commodif.	Main category	Examples
0	Moral suasion and regulations justified by intrinsic value	<ul style="list-style-type: none"> Information appealing to moral responsibility. Recognising social equity and nature's intrinsic value, e.g. endangered species acts and nature reserves
1	Non-monetary regulations based on instrumental arguments	<ul style="list-style-type: none"> Nature reserves and other land-use plans focusing on nature's instrumental value to human wellbeing
2	Non-monetary regulations based on physical metrics (units of nature)	<ul style="list-style-type: none"> Ecological compensation with no role for price signals or market transactions
3	Non-monetary regulations designed to maximise economic efficiency	<ul style="list-style-type: none"> City park designed and managed to maximise calculated recreation values
4	Economic instruments (not traded)	<ul style="list-style-type: none"> Taxes and subsidies Subsidy-like PES paid by governments
5	Economic instruments (voluntary market trade)	<ul style="list-style-type: none"> Market-like PES Markets for ecosystem services (MES), e.g. biodiversity offsets trading conservation credits
6	Financial instruments	<ul style="list-style-type: none"> Forest bonds Biodiversity derivatives

(Dempsey and Robertson, 2012).

Even if instrumental framing has often been put forward as the first step of commodification (e.g. Kosoy and Corbera (2010) and Gómez-Baggethun and Ruiz-Pérez (2011)), there is little information on how this can be assessed. We suggest that the difference between an intrinsic (relational) and instrumental framing (zero vs. first degree) can be assessed by analysing the institutional design of the policy instrument, including the detailed management prescriptions (see Tables 1 and 2). There is no sharp demarcation line and many countries may use both framings for the same policy instrument.

- The second degree of commodification occurs when policy makers introduce “new” property rights and liabilities which involve measurements of biodiversity or ecosystem service units but without monetary valuation or price signals. An example of this is biodiversity offsets or ecological compensation and the idea is that a company that develops land and water resources for e.g. mining, housing, industries, infrastructure, etc. should compensate for the degradation of biodiversity and ES by investing in these values elsewhere (Conway et al., 2013). This could be voluntary or mandatory. Degradation (i.e. negative impacts) should, according to the mitigation hierarchy, first be avoided by choosing a site with less ecological values for the development project. Once a site has been approved for exploitation, degradation should be minimised. The third step of the mitigation hierarchy (sometimes included in the second step) is that the developer takes rehabilitation or restoration measures on the ecosystems impacted and the final step consists of offset measures outside the developed area to compensate for significant adverse residual impacts (Dickie et al., 2010).

Biodiversity offsets are often used synonymously with ecological compensation but while biodiversity offsets specifically seek measurable units to deliver no net loss (or net gain) of conservation outcomes, “ecological compensation is a broader term” (Conway et al., 2013: 2). In the scope of this paper, we focus on mandatory programs and let the use of market transactions define the difference between these instruments, so that biodiversity offsets use the market while ecological compensation does not (we will elaborate on the meaning of ‘markets’ later).

With this demarcation, ecological compensation belongs to the 2nd degree of commodification while biodiversity offsets belong to the 5th level. Ecological compensation is common in the European Union, e.g. development within Natura 2000 sites needs to be compensated for and this is done case-by-case by government agencies. On ordinary (non-protected) land, the

German compensation pools is an established policy instrument where the weighing of degradation and compensation as well as the decision of how much and which land should be used for compensation are done by municipal or private agencies, appointed by the state. Having a pool of compensation land to choose from facilitates the process of matching sites and enables the agency to consider habitat connectivity like green corridors. Even if the German compensation pools sometimes are referred to as “habitat banks” (e.g. Conway et al., 2013: 106), it is clear that there is very limited role for the market in determining price or quality (Conway et al., 2013: 113–115). The Western Cape Biodiversity Offset program in South Africa (Dickie et al., 2010: 51) would also be an ecological compensation program with our definition (Koh et al., 2014).

- The third degree involves deliberate efforts to express or ‘demonstrate’ (TEEB, 2010: 11) the value of nature in monetary terms (monetary valuation). As mentioned already, this has been referred to as step towards commodification. Still, we are hesitant to include it as a degree since monetary valuation is ‘only’ an analytical calculation which is not necessary as decision support to justify economic instruments. We have chosen to include it because it introduces commensurability (expressing ES in monetary terms) (Aldred, 2002). Furthermore, Neuteleers and Engelen (2015) argue that monetary valuation contributes to real commodification and therefore “paves the way (discursively and sometimes technically) for commodification to happen” (Gómez-Baggethun and Ruiz-Pérez, 2011: 624). For example, a nature reserve targeted at biodiversity conservation has probably different management prescriptions compared to a nature reserve designed and managed to maximise recreation values based on cost-benefit calculus (Table 1). Furthermore, when governments demand a cost-benefit analysis of nature reserves, although in the short run institutional design may remain unaltered, in the long run alteration seems likely if nature reserves are provided by private and public landowners just like subsidy-like PES program. Sometimes ‘valuation’ is used as shorthand for ‘monetary valuation’ although ‘valuation’ is much broader and may involve anything between two extreme approaches; (i) a qualitative understanding and appreciation of the importance of the underlying ecological processes (Daily, 1997) and (ii) a monetary expression of the final ES “directly enjoyed” by people (Boyd and Banzhaf, 2007: 619). The first approach is central for ecological economics. It emphasises multi-functionality and the role of biodiversity (in genes, species, and ecosystems) to support and sustain livelihoods, especially the role of resilience. Such valuation is mainly expressed in non-monetary terms,

Table 2
Framework for ES valuation and policy integration

	Information in		
	Qualitative terms	Quantitative terms	Monetary terms
Stated purpose (as observed in national legislation)	Concern for non-measurable objectives like social equity, precautionary principle and safeguarding the insurance value of biodiversity.	Concern for reaching quantitative targets in cost-effective ways without expressing targets in monetary terms.	Concern for economic efficiency expressed in monetary terms and justified as a means to internalise externalities.
Methods for describing values. Decision-support.	SWOT analysis, identification, historical assessment, narratives, stakeholder consultation, Delphi methods, multicriteria analysis.	Technical/scientific mapping and assessment of trends e.g. water flows, pollination, and species abundance, multicriteria analysis.	Cost-benefit analysis is the frame, methods for monetary valuation include replacement cost, contingent valuation, and hedonic pricing.
Policy integration by non-monetary regulation and without market trade (DC 0, 1, 2, 3)	Land use planning, protected areas, and species acts targeted to intrinsic values (DC0) or instrumental values (DC1). N/A	Land use planning, protected areas, and species acts designed to reach measurable targets (DC0 or DC1 depending on institutional design). Liability for ecological compensation, using physical metrics, e.g. German compensation pools (CD2).	Land use planning and protected areas designed to maximise economic efficiency. (DC1 or DC3 depending on institutional design). N/A
Economic instruments (no market trade). (DC 4)	Subsidy-like PES paid by governments targeted at high biodiversity, multiple ES or poverty areas, e.g. EU agri-env. and PES in Ecuador and Costa Rica.	Subsidy-like PES paid by governments targeted at well-defined measurable units, e.g. PES for carnivores in Sweden.	PES when payment is informed by the stated value of the targeted ecosystem service, not by opportunity costs, e.g. Wimmera, Australia.
Economic instruments (market trade) (DC 5)	N/A	Biodiversity offsets trading conservation credits, e.g. US habitat banking, and market-like PES financed by users, e.g. Vittel watershed in France. Monetary value set by market actors.	US habitat banking, and market-like PES financed by users, e.g. Vittel watershed in France. Monetary value set by markets, e.g. biodiversity derivatives.
Financial instruments (DC 6)	N/A	N/A	Monetary value set by markets, e.g. biodiversity derivatives.

DC=Degree of Commodification. N/A=Not applicable
The examples provided are described in text.

focusing on understanding the link between ecosystem processes and human wellbeing. It belongs to the first degree of commodification. The second approach could be appropriate when the purpose of valuing ES is to integrate these values in national accounting and reporting systems (Fisher et al., 2009), e.g. in accordance to Aichi Target 2.

In between these extremes are two versions of monetary valuation that belong to the same box in Table 2 but still are ontologically different. The first version can be exemplified by replacement cost and avoided cost methods which are the most common methods for estimating the value of regulating services in monetary terms (Pascual et al., 2010: 206). These methods focus on the *consequences* of particular choices rather than preferences. The second version include methods analysing stated or revealed preferences. These are rooted in the neoclassical idea that the value of nature depends on human preferences. As a contrast, ecological economics is rooted in the idea that nature has value to human wellbeing regardless whether we understand it or not (Daily et al., 2000).

An important objection to using stated or revealed preferences to inform policy is that the “outcome of economic valuation is in this respect not more informed than the people whose values are being assessed” (Daily et al., 2000: 396). The ideal, to simulate markets, is also inappropriate because market prices do not only reflect present knowledge but also present institutions (Bromley, 1990; Vatn, 2010). Finding 4 of the Millennium Ecosystem Assessment states that the present ecosystem challenges can be met but this requires “significant changes in policies, institutions and practices that are not currently under way.”⁷ These significant changes would dramatically alter all relative prices in the economy and make “economic advice derived within the neoclassical paradigm... especially inappropriate” (Rammel and van den Bergh, 2003: 122).

In other words, it is questionable to use price tags, derived from

existing undesirable institutions, preferences, and wealth distribution, to inform us on what institutions would be ‘efficient.’ To do so would imply a strong bias in favour of status quo (Schmid, 1987: 213). This approach to valuation reflects the consumer tastes and institutional arrangements that together have undermined sustainability. We know from game theory that it is impossible to escape the suboptimal non-cooperative Nash equilibrium of the prisoners’ dilemma as long as the market behaviour and institutions, which results in the Nash equilibrium, are used as a norm for valuation and collective action.

- In the fourth degree taxes and subsidies are used to enhance ecosystem values. These monetary incentives employ the price signal in the Pigouvian sense to internalise externalities and evoke behavioural change but do not create markets: taxes and subsidies are not traded. The level of the tax or subsidy may or may not be informed by a monetary valuation, hence the third degree of commodification is not needed for the fourth degree. Often the government seeks to determine a level of taxation or subsidy that results in the desired effect. For example, the level of a subsidy/payment for farming organically is set to compensate a sufficient number of farmers for the forgone net benefits (opportunity costs) compared to conventional farming. In other words, it is the harvest decrease of wheat and potatoes that is commodified, not the expected increase in biodiversity and ES.

Indeed, the large public PES (agri-environment) programs in Europe are “Pigouvian-type PES” (Sattler and Matzdorf, 2013: 3) and hard to distinguish from more traditional subsidy programs. Sometimes this is called compensation for ecosystem services (CES) rather than PES (McAfee and Shapiro, 2010). Globally, CES or government-financed subsidy-like PES constitutes 97–99 percent of all biodiversity-PES, the lower figure for developing countries (Milder et al., 2010; Vatn, 2015), if we exclude biodiversity offsets and eco-certification like organic food in our definition of PES. Subsidy-like PES belong to the 4th degree of commodification whereas the more market-oriented

⁷ <http://www.maweb.org/documents/document.356.aspx.pdf>

Coasean-type PES (Sattler and Matzdorf, 2013), what Wunder et al. (2008) refer to as pure PES, belong to the 5th degree (Table 1). We explain more fully market and market-based below.

The well-known Costa Rican PES program *Pago por Servicios Ambientales* (PSA) was launched in 1996 as a “neoliberal market mechanism” but should more properly be labelled a government “subsidy in disguise” (Fletcher and Breitling, 2012): 402). This is because it was largely financed through a carbon tax and water tariffs, enabled by a new Forest Law that banned land-use change but not sustainable use, and targeted high poverty areas and “biological corridors” (Matulis, 2013: 256). If the institutions of PSA would be consistent to the stated purpose, this program would fit in the 5th degree of commodification, not the 4th degree (Table 2). Another national PES program in Ecuador, Programa Socio Bosque (PSB) has avoided connotations to payments and market (and may therefore be called CES). Instead the stated purpose of the government in Ecuador has been combining ecosystem conservation with poverty alleviation (De Koning et al., 2011) by connecting it to the rights of Nature (Madre Tierra) as well as people’s rights to a good way of living (buen vivir).⁸ Such payments can be seen as rewards for good stewardship rather than economic incentive (Muradian et al., 2013). The degree of commodification is the same for the Costa Rican PES and the Ecuadorian PES/CES, but the governments used different stated purposes to adapt to different cultural or political contexts (Farooqui and Schultz, 2012).

Government-financed biodiversity-PES schemes typically target a commodity with fuzzy characteristics, based on assumptions about the relationship between a certain land use and the provision of desired but often non-explicit outcomes for biodiversity and ES (Wunder et al., 2008:839). For example, European agri-environment schemes pay farmers for grazing land and organic farming although it is not clear exactly which species or ES benefit from such land use. The same is true for Costa Rica where forest cover is regarded as a proxy for biodiversity conservation (Porrás et al., 2013). As a result, it is rather one hectare of forest or agricultural land under a specific land-use that is commodified, not the desired species or ES themselves (Muradian et al., 2010). Hence, unlike carbon-PES, government-financed subsidy-like biodiversity-PES should not be interpreted as price tags on specific units of nature although it constitutes a high degree (4) of commodification.

Wunder et al. (2008):843 identified only one PES program where providers are paid according to measured units actually delivered and that is payments by the Swedish government to indigenous Sami communities for documented presence and reproductions of wolves, wolverine and lynx on reindeer grazing land. A system where the monetary value is determined by a government auction, e.g. the Wimmera catchment pilot program for salinity control in Australia (Wunder et al., 2008 :838), can be regarded an ‘incomplete market with intermediary’ (Vatn, 2015 :227) where the government is acting on behalf of the real users (Table 2).

5. In environmental and ecological economics, distinctions are usually made between taxes, subsidies and cap-and-trade; these are considered as three separate economic instruments. These are founded in legislation but distinguished from conventional (non-monetary) regulations (Table 1). Economic instruments and regulations are in turn distinguished from policy instruments based on information or moral suasion (Common

and Stagl, 2005: 409).

The fifth degree includes market-traded biodiversity offsets and other markets for ecosystem services (MES) resembling cap-and-trade systems. Most are mandatory and the largest programs are in the USA: wetland mitigation, stream mitigation, and conservation banking (OECD, 2013). Conservation banking is legally mandated biodiversity offsets, modelled after wetland banking (McKenney and Kiesecker, 2010). The general idea, according to guidance by the US agency Fish and Wildlife Service (FWS, 2003), is that market actors (e.g. landowners willing to restore ecological values to compensate for other actors’ degradation) join a habitat bank and are issued conservation credits by a government agency and subsequently allowed to sell these credits to a developer (e.g. public transportation agency or private developer). In this way ownership of credits is transferred from the government to the bank, after which the credits can be sold and re-sold on market conditions. The government however controls this market by creating it and by determining the number of credits (the cap), the initial distribution and commands the demand for the credits, i.e. the rules for who needs to buy credits and how many they need to buy.

Non-mandatory biodiversity offsets with other buyers than governmental bodies also belong to this level, as well as non-mandatory PES. These user-financed PES schemes are more market-like since they are fully voluntary (not only on the provider side). Besides, they are usually focused on a single ecosystem service like the Vittel watershed protection in France (Wunder et al., 2008:838–839) which also increases the degree of commodification. Globally, these non-mandatory offsets/banking and PES programs embrace USD 10–17 million/year, which is only 0.5–1 percent of the payments within the mandatory programs (Milder et al., 2010).⁹

6. Financialisation is the sixth degree of ‘complete commodification’ and describes how the traded commodity is re-packaged and re-sold as financial instruments (e.g. bonds or derivatives). The financial flows of PES or MES schemes are the basis (underlying value) for this. Financialisation uses the ecosystems, which are commodified by PES or MES schemes, as collateral for investments and thereby significantly enhances the degree of commodification. Based on Sullivan (2012) and Dempsey and Robertson (2012) we define financialisation as a process in which financial actors invest in units of conserved nature and turn these investments into financial instruments which are traded on financial markets.

The forest bonds proposed by WWF and Global Canopy Programme offer an example of financialisation. They enable the forest owners, who issue forest bonds, “to raise large-scale finance now that will be repaid by existing and anticipated future income” (Cranford et al., 2011: 6). The issuer of the bonds “will need to convince investors that the cash flows they plan to pay the bond back with are sufficiently secure and predictable” (ibid.). A reliable global REDD+ program¹⁰ would provide such predictability but “support from the public sector through regulations or other commitments will be needed to ensure that these cash flows materialise” (ibid.). Goldman Sachs is one of the financial partners in this proposal and a potential investor in such bonds.

⁹ Mandatory offset programs embrace USD 380 million/year but the distribution of payments between public agencies and private developers is not known. Mandatory PES embrace USD 1450 million/year and are all paid by government bodies (Milder et al. 2010).

¹⁰ REDD+ stands for Reducing Emissions from Deforestation and Forest Degradation, conserving and sustainably managing forests and enhancing forest carbon stocks in Developing Countries.

⁸ Republic of Ecuador Constitution of 2008, <http://pdba.georgetown.edu/Constitutions/Ecuador/english08.html>, accessed 2 April 2014.

Biodiversity derivatives is another form of financialisation which has been discussed by Mandel et al. (2009). The government might buy a ten-year biodiversity derivative for a species of concern wherein a predefined amount of funds would be released by the seller if a species' population falls below a threshold. Basically, the seller (issuer or writer of the derivative) is making the bet that the species population will not fall below the threshold, or that it can take measures to prevent such an occurrence whose costs are within the range wherein they can make a profit given their proceeds from issuing the derivative.

3. Framework for valuation and policy integration

In a broad economic sense valuation is about expressing a willingness to make sacrifices to gain or protect something. In the context of policy integration, the role of valuation is to raise awareness and provide decision-support expressed in qualitative, quantitative, and/or monetary terms (Table 2). The monetary valuation could be based on analytical calculations or the price from hypothetical or real markets. Table 2 provides a comprehensive and pluralistic framework of the relation between valuation and policy integration: there is no unique relation between the instrument chosen for policy integration (horizontal axis) and the political justification for these instruments or the way nature's values are described in the columns. Table 2 is an attempt to address both purposes and degrees of commodification.

Much of the confusion in the discussion of commodification can be attributed the misunderstanding that economic instruments assumes monetary valuation. Table 2 decouples the technical calculation of monetary valuation from policy instruments; monetary valuation is not needed for the lower degrees (1–2 and 4) of commodification and in the higher degrees (5–6) the price is a result of market trade. Table 2 also suggests that both proponents and critics to neoliberalism are making mistakes when they associate economic instruments with neoliberal frameworks, rather than assessing the actual institutions and performance (Dempsey and Robertson, 2012).

In reality, a PES scheme for grazing land can be justified by a qualitative analysis of vulnerable species depending on grazing land or targeted to forest communities as a tool for poverty alleviation. Reversely, a nature reserve can be justified by a cost-benefit analysis emphasising its extraordinary value for recreation or public health. Again, it is the institutional design of a particular policy instrument that determines the degree of commodification. However, public perceptions of commodification and the political legitimacy of the policy instrument will also be influenced by the stated purpose and the valuation framework used to justify the instrument.

We do not suggest that any method (column) of policy integration is better than the other. Qualitative information has many similarities to SWOT (Strength, Weakness, Opportunity, Threat) analysis in business (Hanson et al., 2008) and is common in business thinking on resilience (Hamel and Vlikangas, 2003). The right column represents the ideal in neoclassical economics, wherein all values are demonstrated in monetary terms in support of efficiency calculations.

4. 'Market-based instruments' is a confusing term

'Market-based instruments' are generally assumed to include taxes, subsidies and various cap-and-trade systems (e.g. Pearce (2013)) but often include a "wide a range of hybrid instruments" (Muradian and Gómez-Baggethun, 2013: 1118). Here we want to emphasise the confusion between the role of prices and the role of

markets. A tax definitely uses the price signal but it is not a market since the tax is imposed, not traded.

The price *mechanism* (or market mechanism) is something else: it can be described as the autonomous mechanism that determines the price in a market economy, as an equilibrium between supply and demand. The price *signal* is the result of the price mechanism in competitive markets and also in less competitive markets where a large producer or consumer can influence the price. However, for taxes and subsidies, the price signal is a result of a political decision to internalise externalities in the Pigouvian sense. 'Market-based instruments' could properly be used for the instruments under the 5th and 6th degrees of commodification but we prefer the more general term 'economic instruments' to describe all policy instruments relying on the price signal (monetary incentives), i.e. degree 4–6 (Tables 1 and 2).¹¹

Voluntary market-like Coasean-type PES and biodiversity offsets (mandatory or voluntary) trading conservation credits constitute the fifth degree of commodification because the price is the result of supply and demand of the market. The price signal is determined by the government in the 4th degree but by the market in the 5th and 6th degrees of commodification. Markets for ecosystem services (MES) indeed require more, not less, regulations than the previous degrees if MES are to contribute to environmental benefits (Glicksman and Kaime, 2013). Principles of international law and safeguards can contribute to designing and implementing MES including ensuring accountability in environmental governance (Ituarte-Lima et al., 2012).¹² Hence, 'markets' (reliance on market transactions) should not be understood ideologically as opposite to regulations even though they are often presented as such (Penca, 2013; Fletcher and Breiting, 2012). Moreover, the transaction costs associated with MES are often high compared to taxes, subsidies, and regulations, which suggests that *a priori* assumptions about cost-effectiveness should be avoided (Gómez-Baggethun and Muradian, 2015).

5. Discussion

5.1. The diversity of PES designs

Table 2 suggests there is no direct link between the methodological approach to valuation (non-monetary or monetary) and the choice of policy instrument. Concerning PES, Table 2 clarifies three issues. First, subsidy-like government-financed PES, also called Pigouvian-type PES or Compensation for ecosystem services (CES), represent a lower degree of commodification compared to the much smaller market-like Coasean-type PES.

Second, subsidy-like PES can be further differentiated based on stated purpose and use of valuation framework, as suggested by the three columns. If society wishes to use monetary incentives (or rewards) but avoid that monetary valuation reduces "dignities" (intrinsic values) to "commodities" (instrumental values), CES may be an option.

Third, the stated purpose may sometimes appear to be inconsistent to the actual institutional design of the program. The national PES program of Costa Rica and CES program of Ecuador are similar in terms of the degree of commodification, but are different in the particular design. In both countries, these differences have been exaggerated by the political justifications for the

¹¹ We thank Katia Karousakis at the OECD for suggesting the term "economic instruments" and pointing out that it is already used instead of 'market-based instruments' in many reports (e.g. OECD (2013)).

¹² See also p. 22 in Report of the ad hoc open-ended working group on review of implementation of the CBD on the work of its fourth meeting, UNEP/CBD/COP/11/4, 21 June 2012.

programs. Rather than criticising this, we suggest it illustrates how flexible the design of PES/CES programs can be and how they can be implemented and justified to accommodate country-specific concerns to gain legitimacy which is of key importance within the CBD (Farooqui and Schultz, 2012; Ogwal and Schultz, 2014).

Sometimes the definition of PES includes legally mandated private payments which are required to offset environmental impacts, e.g. wetland mitigation or habitat banking, as well as the price premium for eco-certified products that reaches the producers of ES (Scherr and McNeely, 2008; Milder et al., 2010). However, we believe it makes sense analytically i) to distinguish between PES and liabilities to compensate for damage (offsets or ecological compensation) regardless if they are mandatory or not; and ii) to distinguish between economic instruments on the one hand and eco-certification on the other. Although an eco-label often functions as a price premium for producers (monetary incentive), it can also be regarded, from a policy perspective, as an instrument based on information and moral suasion (Common and Stagl, 2005: 409).

5.2. Biodiversity offsets and ecological compensation

Ecological compensation schemes, including biodiversity offsets, are often controversial. Proponents emphasise that it is fair (analogous to the Polluter Pays Principle) that developers are required to pay for restoration activities so as to offset the degradation they cause and this may also steer development away from areas with high values for biodiversity and ES because these are expensive to compensate for (Conway et al., 2013: vii). Landowners who invest in biodiversity and ES can enter contracts and receive payments from the developers rather than from taxpayers. Interviews with private landowners in Germany and Australia have noted the economic attractiveness of entering a long-term maintenance contract with a compensation agency, as it ensured a stable source of income (Tucker et al., 2013:193; O'Connor, 2009:15).

Critics of biodiversity offsets such as Forest Peoples Programme (2011) are concerned that ecosystems, as well as their functions for livelihood opportunities, are not fully replaceable. Local people in one region normally depend on the biodiversity and ES in that area for their livelihoods and other benefits. There is a risk that such schemes will offer incentives for developers and administrators “to downplay or ignore the requirement to first avoid and reduce their impacts under the false impression that any impact can be compensated for” (Quétier and Lavorel, 2011: 2991. This is sometimes referred to as the restoration myth or a ‘license to trash’ (Dickie et al., 2010:237). If biodiversity offset policies allow for the approval of land exploitation that would otherwise not have been accepted, biodiversity objectives as well as livelihood opportunities for local communities can be compromised.

This can be avoided only if the avoidance step of the mitigations hierarchy is respected, which means that ‘no-go’ areas are stipulated (Conway et al., 2013: viii) and “current rules to decide whether developments should go ahead do not change” (Dickie et al., 2010: 237). Hence, the proponents emphasise that introduction of a biodiversity offset scheme must not result in less stringent regulations.

Despite this agreement, the largest offset program, US Wetland mitigation, has focused too much on the compensation part and neglected the earlier stages of the mitigation hierarchy (Hough and Robertson, 2009). The result is poor performance (Kihlslinger, 2008; Turner et al., 2001). For example, an evaluation of 391 wetland offset projects in Massachusetts showed that 54% were not in compliance with the wetland regulations (Brown and Venneman, 2001) and Ambrose and Lee (2004) found that 46% of the 250 sites surveyed in California failed to replace key wetland ES

which had been determined by the state.

When the whole spectrum of institutional design from non-market liabilities (ecological compensation) to market-like trade (biodiversity offsets) is acknowledged, considerable flexibility is shown. This is promising since it allows for the emergence of country-specific programs with a cultural-political adaptation to gain legitimacy (Hahn, 2011). This is also a risk because the devil is in the detail; small changes in the institutional design may have large consequences for the performance. We give four examples:

First, weighing the development interest against the conservation interest at the site of proposed development must not be compromised by the (false) promise that the damage can and will be entirely compensated for elsewhere. A successful program should result in a *reduction* of permits obtained by developers because the liability to finance restoration at the compensation site increases costs, thereby deterring development projects with marginal profitability.

Second, and related to the first, even an ideally designed program may fail for ecological reasons, if restoration turns out to be much more complicated than anticipated (Hilderbrand et al., 2005; Maron et al., 2012).

Third, a successful program would align the interest of (city) planners with the interest of developers. Planners often try to steer development away from remaining green areas and developers would also avoid green land with high biological values since these would be expensive to compensate for.

Fourth, simple metrics (one acre equals one credit) and use of market trade might have lowered transaction costs of the large US programs but this cannot be regarded as cost-effective if the achieved effect, the actual performance of the program, is low. The evaluations referred to above suggest low performance. Briggs et al. (2009) warn that “[w]ithout careful regulation, habitat banks could offer low-cost compensation as a result of cutting corners on conservation, and the market would reward poorly managed banks and thus harm conservation efforts” (p. 117). The dichotomy government regulations vs. market is false (Vatn, 2015) and this is particularly obvious for ecological compensation/offsets since good performance seem to be a function of a comprehensive government regulation (Dickie et al., 2010: 237), i.e. safeguards.

6. Conclusion

The issue of monetary valuation of biodiversity and ES is a never-ending debate within ecological economics because it runs counter to one of the central beliefs within this community, namely that natural capital cannot be substituted for. However, there is an emerging scientific consensus that the price signals used in PES are in general set to compensate for opportunity costs and hence cannot be regarded as a monetary valuation of the desired ES. We also find that biodiversity offsets, designed with strong safeguards, would properly be named ‘ecological compensation’ and have very little to do with market trade or valuation of ES. Hence, these economic instruments should not by default be associated to neither monetary valuation nor neoliberalism.

Dempsey and Robertson (2012) argue that many social scientists “have largely positioned themselves as critical” to PES and other ES policies and associated these to commodification and “the neoliberalization of nature” (pp. 762, 772). Instead Dempsey and Robertson suggest “a useful solidarity with people engaged, through ES policy, in opposition to business-as-usual resource development” (p. 773). Our results support that ES policies in general can be useful to counter some of the market failures of business-as-usual and hence should not be understood as ‘market-based instruments’ or ‘neoliberalisation of nature.’ At the same time, we have used ‘commodification’ as an analytical, not

normative, term and found that all ES policies, by using an instrumental (human wellbeing) perspective, do involve at least low degrees of commodification.

The strength of markets, searching for lowest cost for provision, may result in low quality for complex 'products' such as conservation of biodiversity and ES, unless combined with strong safeguards including enforcement. Markets may entail low transaction costs and be cost-effective regarding simple products but this should not be assumed for complex products.

The institutional diversity and plurality in understanding and implementing various biodiversity financing mechanisms, especially economic instruments, gives flexibility for the 194 Parties of the CBD to adapt these instruments to their national political and cultural context. This flexibility in turn enhances prospects for reaching international agreement on biodiversity financing mechanisms and enhances policy options in each country.

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References

- Aldred, J., 2002. Cost-benefit analysis, incommensurability and rough equality. *Environ. Values* 11, 27–47.
- Ambrose, R., Lee, S., 2004. Guidance Document for Compensatory Mitigation Projects Permitted Under Clean Water Act Section 401 by the Los Angeles Regional Quality Control Board. California Environmental Protection Agency, Los Angeles.
- Boyd, J., Banzhaf, S., 2007. What are ecosystem services? The need for standardized environmental accounting units. *Ecol. Econ.* 63, 616–626.
- Briggs, B.D.J., Hill, D.A., Gillespie, R., 2009. Habitat banking—how it could work in the UK. *J. Nat. Conserv.* 17, 112–122.
- Bromley, D.W., 1990. The ideology of efficiency: Searching for a theory of policy analysis. *J. Environ. Econ. Manag.* 19, 86–107.
- Brown, S., Veneman, P., 2001. Effectiveness of compensatory wetland mitigation in Massachusetts, USA. *Wetlands* 21, 508–518.
- Burdon, P., 2011. Exploring Wild Law: the Philosophy of Earth Jurisprudence. Wakefield Press, Adelaide.
- CBD, 2010. Convention of Biological Diversity, point 8(c) of CBD COP10 Decision X/3, (www.cbd.int/decision/cop/default.shtml?id=12269) accessed 10.12.13.
- Common, M., Stagl, S., 2005. Ecological Economics: An Introduction. Cambridge University Press.
- Conway, M., Rayment, M., White, A., Berman, S., 2013. Exploring potential demand for and supply of habitat banking in the EU and appropriate design elements for a habitat banking scheme. Final Report submitted to DG Environment by ICF GHK Consulting Ltd in association with BIO Intelligence Service.
- Corbera, E., Brown, K., Adger, W.N., 2007. The equity and legitimacy of markets for ecosystem services. *Dev. chang.* 38, 587–613.
- Cranford, M., Henderson, I., Mitchell, A., Kidney, S., Kanak, D., 2011. Unlocking Forest Bonds – A High-Level Workshop on Innovative Finance for Tropical Forests. Workshop Report. WWF Forest & Climate Initiative, Global Canopy Programme and Climate Bonds Initiative, (www.theredddesk.org/fr/node/5627).
- Daily, G.C., 1997. Nature's services: societal dependence on natural ecosystems. Island Pr.
- Daily, G., Söderqvist, T., Aniyar, S., Arrow, K., Dasgupta, P., Ehrlich, P., Folke, C., Jansson, A., Jansson, B.O., Kautsky, N., Levin, S., Lubchenco, J., Mäler, K.G., Simpson, D., Starrett, D., Tilman, D., Walker, B., 2000. The value of nature and the nature of value. *Science* 289, 395–396.
- Daly, E., 2012. The Ecuadorian example: the first ever vindications of constitutional rights of nature. *Rev. Eur. Commun. Int. Environ. Law* 21, 63–66. <http://dx.doi.org/10.1111/j.1467-9388.2012.00744.x>.
- De Koning, F., Aguiñaga, M., Bravo, M., Chiu, M., Lascano, M., Lozada, T., Suarez, L., 2011. Bridging the gap between forest conservation and poverty alleviation: the Ecuadorian Socio Bosque program. *Environ. Sci. Policy* 14, 531–542.
- Dempsey, J., Robertson, M.M., 2012. Ecosystem services Tensions, impurities, and points of engagement within neoliberalism. *Prog. Hum. Geogr.* 36, 758–779.
- Dickie, I., Tucker, G., Ozdemiroglu, E., Cranford, M., Ohlenburg, H., Wende, W., Chapman, D., Donovan, C., Bishop, J., Brans, E., Reinders, R., Ekstrom, J., ten-Kate, K., Treweek, J., 2010. The use of market-based instruments for biodiversity protection: The case of habitat banking. Technical Report by eftec IEEP and others. (http://ec.europa.eu/environment/enveco/pdf/eftec_habitat_technical_report.pdf).
- Ehrlich, P.R., Mooney, H.A., 1983. Extinction, substitution, and ecosystem services. *BioScience* 33, 248–254.
- Farooqui, M.F., Schultz, M., 2012. Co-chairs' Summary of Dialogue Seminar on Scaling up Biodiversity Finance. Quito 6–9 March 2012, pp. 1–49. (www.cbd.int/doc/meetings/fin/ds-fb-01/official/ds-fb-01-02-en.pdf).
- Fisher, B., Turner, R.K., Morling, P., 2009. Defining and classifying ecosystem services for decision making. *Ecol. Econ.* 68, 643–653.
- Fletcher, R., Breitling, J., 2012. Market mechanism or subsidy in disguise? Governing payment for environmental services in Costa Rica. *Geoforum* 43 (3), 402–411.
- Forest Peoples Programme, 2011. Submission to the Convention on Biological Diversity relating to innovative financial mechanisms and the rights of indigenous peoples and local communities.
- FWS, 2003. Fish and Wildlife Service. Guidance for the Establishment, Use, and Operation of Conservation Banks. United States Department of the Interior.
- Glicksman, R.L., Kaime, T., 2013. A comparative analysis of accountability mechanisms for ecosystem services markets in the United States and the European Union. *Transnatl. Environ. Law* 2, 259–283.
- Gómez-Baggethun, E., Ruiz-Pérez, M., 2011. Economic valuation and the commodification of ecosystem services. *Prog. Phys. Geogr.* 35, 613–628.
- Gómez-Baggethun, E., Muradian, R., 2015. In markets we trust? Setting the boundaries of market-based instruments in ecosystem services governance. *Ecol. Econ.* 117, 217–224.
- Groenfeldt, D., Schmidt, J.J., 2013. Ethics and water governance. *Ecol. Soc.* 18. <http://dx.doi.org/10.5751/ES-04629-180114>.
- Hahn, T., 2011. Self-organized governance networks for ecosystem management: who is accountable? *Ecology and Society* 16 (2), 18 (<http://www.ecologyandsociety.org/vol16/iss2/art18/>).
- Hamel, G., Vlikaangas, L., 2003. The quest for resilience. *Harv. Bus. Rev.*, 52–63.
- Hanson, C., Ranganathan, J., Iceland, C., Finisdore, J., 2008. The corporate ecosystem services review, Guidelines for Identifying Business Risks and Opportunities Arising From Ecosystem Change. WRI, WBCSD and Meridian Institute.
- Higgins, P., Short, D., South, N., 2013. Protecting the planet: a proposal for a law of ecocide. *Crime, Law Soc. Chang.* 59, 251–266.
- Hilderbrand, R.H., Watts, A.C., Randle, A.M., 2005. The myths of restoration ecology. *Ecol. Soc.* 10, 19.
- Hough, P., Robertson, M., 2009. Mitigation under section 404 of the Clean Water Act: where it comes from, what it means. *Wetl. Ecol. Manag.* 17, 15–33.
- Ituarte-Lima, C., Schultz, M., Hahn, T. and Cornell, S., 2012. 'Safeguards in scaling-up biodiversity financing and possible guiding principles', Stockholm Resilience Centre at the Stockholm University, Information document for the 11th Conference of the Parties of the Convention on Biological Diversity, (UNEP/CBD/COP11/INF/7). (<http://www.cbd.int/cop11/doc/>) and Notification for Parties of CBD to submit comments (<http://www.cbd.int/doc/notifications/2013/ntf-2013-025-financial-en.pdf>).
- Ituarte-Lima, C., Schultz, M., Hahn, T., McDermott, C., and Cornell, S., 2014. Biodiversity financing and safeguards: lessons learned and proposed guidelines, Stockholm: SwedBio/Stockholm Resilience Centre at Stockholm University, Information Document UNEP/CBD/COP12/INF/27 for the 12th Conference of the Parties of the Convention on Biological Diversity in Pyeongchang Korea. (<http://www.cbd.int/doc/?meeting=cop-12>).
- Kihlslinger, R., 2008. Success of wetland mitigation projects. *Natl. Wetl. Newsl.* 30 (2), 14–16.
- Koh, N.S., Hahn, T., Ituarte-Lima, C., 2014. A comparative analysis of ecological compensation programs: The effect of program design on the social and ecological outcomes. Working Paper, Uppsala University. (<http://uu.diva-portal.org/smash/record.jsf?pid=diva2%3A772933&dsid=1454>).
- Kosoy, N., Corbera, E., 2010. Payments for ecosystem services as commodity fetishism. *Ecol. Econ.* 69, 1228–1236.
- Luck, G.W., Chan, K.M., Eser, U., Gómez-Baggethun, E., Matzdorf, B., Norton, B., Potschin, M.B., 2012. Ethical considerations in on-ground applications of the ecosystem services concept. *BioScience* 62, 1020–1029.
- MA, 2005a. Ecosystems and Human Well-being: Synthesis Report, The Millennium Ecosystem Assessment. Island Press.
- Mandel, J.T., Donlan, C.J., Armstrong, J., 2009. A derivative approach to endangered species conservation. *Front. Ecol. Environ.* 8, 44–49.
- Maron, M., Hobbs, R.J., Moilanen, A., Matthews, J.W., Christie, K., Gardner, T.A., Keith, D.A., Lindenmayer, D.B., McAlpine, C.A., 2012. Faustian bargains? Restoration realities in the context of biodiversity offset policies. *Biol. Conserv.* 155, 141–148.
- Matulis, B.S., 2013. The narrowing gap between vision and execution: neoliberalization of PES in Costa Rica. *Geoforum* 44, 253–260.
- McAfee, K., Shapiro, E.N., 2010. Payments for ecosystem services in Mexico: nature, neoliberalism, social movements, and the state. *Ann. Assoc. Am. Geogr.* 100, 579–599.
- McKenney, B.A., Kiesecker, J.M., 2010. Policy development for biodiversity offsets: a review of offset frameworks. *Environ. Manag.* 45, 165–176.
- Merchant, C., 1980. Death Nat.
- Milder, J.C., Scherr, S.J., Bracer, C., 2010. Trends and future potential of payment for ecosystem services to alleviate rural poverty in developing countries. *Ecol. Soc.* 15, 2.
- Muradian, R., Corbera, E., Pascual, U., Kosoy, N., May, P.H., 2010. Reconciling theory and practice: an alternative conceptual framework for understanding payments for environmental services. *Ecol. Econ.* 69, 1202–1208.
- Muradian, R., Arsel, M., Pellegrini, L., Adaman, F., Aguilar, B., Agarwal, B., Corbera, E., Ezine de Blas, D., Farley, J., Froger, G., 2013. Payments for ecosystem services

- and the fatal attraction of win-win solutions. *conserv. Lett.* 6, 274–279.
- Muradian, R., Gómez-Baggethun, E., 2013. The institutional dimension of “market-based instruments” for governing ecosystem services: introduction to the special issue. *Soc. Nat. Resour.* 26, 1113–1121.
- Neuteleers, S., Engelen, B., 2015. Talking money: how market-based valuation can undermine environmental protection. *Ecol. Econ.* 117, 253–260.
- Nilsson, M., Persson, A., 2003. Framework for analysing environmental policy integration. *J. Environ. Policy Plan.* 5, 333–359.
- O'Connor, N.R.M., 2009. Evaluation after two years of operations. Maylands Department of Sustainability and Environment, Victoria. Retrieved from http://www.depi.vic.gov.au/_data/assets/pdf_file/0017/205019/BushBrokerEvaluationReportFinal2009.pdf.
- OECD, 2013. Scaling-up Finance Mechanisms for Biodiversity. OECD Publishing <http://dx.doi.org/10.1787/9789264193833-en>.
- Ogwal, S.F., Schultz, M., 2014. Co-Chairs' Summary of Second Dialogue Seminar on Scaling up Finance for Biodiversity, Quito. Secretariat of the Convention on Biological Diversity, Montreal 9–12 April 2014. (www.cbd.int/doc/meetings/fin/ds-fb-02/official/ds-fb-02-report-en.pdf).
- Pascual, U., Muradian, R., Brander, L., Gómez-Baggethun, E., Martín-López, B., Verma, M., et al., 2010. The Economics of Ecosystems and Biodiversity and Ecological and Economic Foundations. In: Kumar, Pushpam (Ed.), 2010. Earthscan, London and Washington, pp. 183–240.
- Pearce, D., 2013. *Blueprint 2: Greening the World Economy*. Routledge.
- Penca, J., 2013. Marketing the market: the ideology of market mechanisms for biodiversity conservation. *Transnatl. Environ. Law* 2, 235–257.
- Porras, I., Barton, D.N., Miranda, M., Chacón-Cascante, A., 2013. Learning from 20 Years of Payments for Ecosystem Services in Costa Rica. International Institute for Environment and Development, London.
- Quétier, F., Lavorel, S., 2011. Assessing ecological equivalence in biodiversity offset schemes: key issues and solutions. *Biol. Conserv.* 144, 2991–2999.
- Rammel, C., van den Bergh, J.C., 2003. Evolutionary policies for sustainable development: adaptive flexibility and risk minimising. *Ecol. Econ.* 47, 121–133.
- Sattler, C., Matzdorf, B., 2013. PES in a nutshell: From definitions and origins to PES in practice—approaches, design process and innovative aspects. *Ecosyst. Serv.* 6, 2–11.
- Scherr, S.J., McNeely, J.A., 2008. Biodiversity conservation and agricultural sustainability: towards a new paradigm of ‘ecoagriculture’ landscapes. *Philos. Trans. R. Soc. B* 363 (1491), 477–494.
- Schmid, A., 1987. *Property, Power, and Public Choice: An Inquiry Into Law and Economics*, Second edition. Praeger, New York.
- Sullivan, S., 2012. Banking nature? The spectacular financialisation of environmental conservation. *Antipode* 45, 198–217.
- TEEB, 2010. *The Economics of Ecosystems and Biodiversity: Mainstreaming the Economics of Nature: A synthesis of the approach, conclusions and recommendations of TEEB*.
- Tucker, G., Allen, B., Conway, M., Dickie, I., Hart, K., Rayment, M., et al., 2013. *Policy Options for an EU No Net Loss Initiative*. Institute for European Environmental Policy, London.
- Turner, E., Redmond, A., Zedler, J., 2001. Count it by acre or function – mitigation adds up to net loss of wetlands. *Natl. Wetl. Newsl.*, 5–16.
- Turnhout, E., Waterton, C., Neves, K., Buizer, M., 2013. Rethinking biodiversity: from goods and services to “living with”. *Conserv. Lett.* 6, 154–161.
- Vatn, A., 2009. An institutional analysis of methods for environmental appraisal. *Ecol. Econ.* 68, 2207–2215.
- Vatn, A., 2010. An institutional analysis of payments for environmental services. *Ecol. Econ.* 69, 1245–1252.
- Vatn, A., 2015. *Markets in environmental governance. From theory to practice*. *Ecol. Econ.* <http://dx.doi.org/10.1016/j.ecolecon.2014.07.017>
- Wunder, S., Engel, S., Pagiola, S., 2008. Taking stock: a comparative analysis of payments for environmental services programs in developed and developing countries. *Ecol. Econ.* 65, 834–852.