Introduction

The concept of ‘ecosystem services’ provides a particular lens for framing social-ecological relationships. Since its emergence about 20 years ago it has provided valuable common ground for different disciplines to discuss broadly interdependent environmental and development goals. The concept does however, sometimes mask different and divergent ideologies and priorities with respect to ecosystem management and development planning. These differing principles are shaped by, and in turn shape, different epistemic communities, or ‘networks of professionals (e.g. in academia and policy) with recognised expertise and competence in a particular domain and an authoritative claim to policy relevant knowledge within that domain or issue’ (Haas, 1992).

The decade since the publication of the Millennium Ecosystem Assessment (MA) (2005) has seen a remarkable evolution in conceptual frameworks of ecosystem services for human wellbeing, reflecting the way different epistemic communities, including those in the science–policy interface, have embraced the approach. The relevant actors in research and policy development are a large, although not cohesive, group. They share debates about how to achieve ‘win-win’ solutions by managing ecosystems sustainably and reducing poverty with some common rationales, narratives and expected/desired outcomes. However, there are significant contrasts in values and positions. These differences are characterised by a variety of normative positions that reflect different mixes of concern, and interpretations about the linkages between ecosystem condition and economic development, creating a series of epistemic communities (Howe et al., 2018). These normative positions have evolved from earlier, narrower debates concerning poverty alleviation...
and biodiversity conservation (e.g. Adams et al., 2004), and have deepened as ecosystem services for human wellbeing frameworks emerged and developed. In turn, such changes in normative positions influence how the epistemic communities themselves connect to policy and so the frameworks co-evolve with epistemic communities, continually influencing one another. However, these interactions between epistemic communities and emerging concepts are seldom recognised in the literature, even while the ecosystem services concept has become a boundary object that enables integration across diverse bodies of knowledge (Abson et al., 2014).

Conceptual frameworks are tools by which complex systems can be clarified. They facilitate the deliberation of, and agreement on, essential components and interactions of the system being studied, as well as highlighting uncertainties and gaps in understanding (Tomich et al., 2010). The process of developing a conceptual framework is inherently value-laden, involving balance and contention among and between different epistemic communities and their underlying ideologies, principles and interests (Díaz et al., 2015). Where conceptual frameworks are used by a research programme – for example, the UK-led Ecosystem Services for Poverty Alleviation (ESPA) programme – they provide a basis for research design and fertile interactions among different epistemic communities involved (for example, researchers, policy makers and practitioners).

The MA (2005) was the first comprehensive attempt with global influence to unify thinking around ecosystem services. Its framing evolved from conceptualisations of natural capital (e.g. Daily, 1997; Jansson et al., 1994), and associated (weak vs. strong) sustainability schools of thought (Ekins et al., 2003), with an emphasis on the supply of ‘ecological goods and services’. The MA gave birth to many initiatives; from sub-global ecosystem assessments at both national, e.g. the United Kingdom National Ecosystem Assessment (UKNEA, 2011), and regional scale, e.g. the European Commission Mapping and Assessment of Ecosystem Services (MAES, 2013); economic assessments, e.g. The Economics of Ecosystems and Biodiversity (TEEB) (Kumar, 2010); and, perhaps the most politically important from a global standpoint, the UN-based Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) (Díaz et al., 2018).

The last decade has also seen the emergence of social-ecological systems (SES) thinking (e.g. Ostrom, 2009; Reyers and Selomane, this volume). This was pioneered by work undertaken by, for instance, the Programme on Ecosystem Change and Society of Future Earth, the Stockholm Resilience Centre, the Ecosystem Services Partnership and academic societies such as the International Society for Ecological Economics. These interdisciplinary research communities interact among themselves, and with the wider research and practice communities. They are, therefore, in an ideal position to broaden the boundary space and connect to other knowledge systems, including those of indigenous people, which generally do not match with the western scientific approach to systematising and generalising knowledge (Tengö et al., 2014). This thinking has coalesced within IPBES, where an evolution has occurred towards interpreting ecosystem services more broadly
as Nature’s Contributions to People (NCP) (Díaz et al., 2018). NCP emphasises the effects of nature as perceived and valued by individuals and social groups across different cultural, environmental and socio-economic contexts, allowing for both generalising (i.e. scientific) as well as more context-specific (e.g. indigenous and local knowledge) perspectives (Pascual et al., 2017a).

ESPA was developed in response to the findings of the MA and has been significant in the evolution of conceptual framings, with an explicit emphasis on ecosystem management for poverty alleviation. It has contributed to ideas such as the co-production of ecosystem services (Lele et al., 2013; Reyers et al., 2013), ecosystem service trade-offs (e.g. Bennett et al., 2015; Howe et al., 2014) and multiple human values for ecosystem services (e.g. Pascual et al., 2017a) within governance contexts in the Global South (see Nunan et al., this volume). ESPA has been nourished by and branched out to tackle innovative views about social-ecological interdependencies (Ostrom, 2009), justice and equity (Sikor, 2013), disaggregated approaches to human wellbeing (Daw et al., 2011; Fisher et al., 2013; Coulthard et al., this volume) and tele-coupling, e.g. in environmental assessments (Pascual et al., 2017b).

This chapter explores the development of ecosystem services framings over the last decade. It also reflects on the potential direct implications for the design of ecosystem service-related policy instruments, such as payments for ecosystem services (PES), and takes a brief look at the future, by identifying emergent challenges associated with future framings.

**Historical background to ecosystem service frameworks**

*Evolution of core and satellite ecosystem service frameworks*

This section explores the emergence of ‘core’ and ‘satellite’ frameworks for connections between ecosystem services and human wellbeing, associated with different epistemic communities. We define ‘core’ frameworks as those which have become a fundamental part of the mainstream approach to addressing ecosystem services for human wellbeing. In contrast, ‘satellite’ frameworks have either significantly influenced the evolution of core frameworks, or have drawn on elements of ‘core’ frameworks to tackle more nuanced aspects like cultural perceptions and the social co-production of ecosystem services.

The MA framework is the first of several core frameworks born out of the integration of thinking by different influential epistemic communities (Reid and Mooney, 2016), including those working in ecology, economics (Jansson et al., 1994) and interdisciplinary scientists that embraced the idea of natural capital (Farley, 2012). Gretchen Daily (1997) defined ecosystem services as the ‘conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfil human life’, while Costanza et al. (1997) further promoted the concept through estimating the economic value of global ecosystem services.
These ideas developed into various global scientific programmes such as DIVERSITAS (the international programme on biodiversity science, established in 1991 by the International Council for Science Unions), with its focus on the impacts of biodiversity changes on ecosystem functioning and thereby the provision of ecological goods and services of relevance to human societies (Larigauderie et al., 2012). At the same time, they built on satellite frameworks such as that of the Drivers-Pressures-States-Impacts-Responses (DPSIR) (Smeets and Weterings, 1999) and the Sustainable Livelihoods Framework (Scoones, 1998), as well as the core MA (2005) framework. This in turn created opportunities for the development of other core frameworks such as those associated with global assessments, including TEEB (Kumar, 2010) and IPBES (Díaz et al., 2015), sub-global ecosystem assessments, including the UK NEA (UKNEA, 2011), as well as programme frameworks including ESPA (ESPA, 2013; also see Preface to this volume) and ecoSERVICES (Bennett et al., 2015) currently under the Future Earth Programme (previously under DIVERSITAS).

From the biophysical perspective, a significant difference between earlier and later conceptualisations is in the categorisation of ecosystem services. The MA defined four main categories: provisioning, regulating, supporting and cultural services. However, by 2009 efforts to formalise social-ecological systems for analysis had led to supporting services being incorporated as ecosystem functions and properties (Carpenter et al., 2009) now reflected in the IPBES conceptual framework (Díaz et al., 2015). In addition, in IPBES the NCP approach embraces the different perceptions and understandings of people about material, non-material and regulating NCP (Díaz et al., 2018), and thus the multiple ways that NCP values are conceptualised (Pascual et al., 2017a). The emphasis about the role of culture is one of the main contributions of the NCP approach, and this goes beyond the ways the ecosystem services approach embraces cultural services in the MA sense, including within ESPA projects.

Figure 1.1 provides a timeline of the evolution of core frameworks, both assessment- and research programme-based. It also includes significant satellite frameworks that have been influential regarding the ideas behind the development of core frameworks. Conceptual framings have evolved from the earlier, more linear ‘natural capital as stock, ecosystem services as flow’ metaphor, to more system-based thinking which allows for feedbacks, and with governance (as the key indirect driver) having more centrality. Likewise, it shows an emerging emphasis on the multi-dimensionality of human wellbeing, with ESPA being a clear example of trying to capture both the biophysical dimension of ecosystem services and the multi-dimensionality of human wellbeing, and poverty specifically. This can be compared for instance with TEEB, which by being primarily interested in economic valuation of ecosystem services, has not addressed multi-dimensional wellbeing. Fisher et al. (2013) provides a useful summary of some of the frameworks discussed here, as well as several other satellite frameworks.

The Sustainable Livelihoods Framework (Scoones, 1998) was an early contribution to articulating the contributions of natural capital to livelihoods, especially
in the Global South. In a complementary approach to the weak vs strong sustainability debate in ecological economics, the development community brought natural capital to the fore as a fundamental asset that determines people’s livelihood options and outcomes. The framework was developed alongside more advanced analysis on, for example, the concept of ‘investment poverty’ whereby degradation of natural capital is seen as a key determinant of asset-based poverty status (Reardon and Vosti, 1995) which can ultimately lead to ‘poverty traps’ (Barrett et al., 2011).

The Drivers–Pressures–States–Impacts–Response (DPSIR) framework (Smeets and Weterings, 1999) was also articulated as a more linear framework that examines how changes in pressures via drivers affect environmental systems, but is weaker on how changes in the system then feed back to affect social drivers and thus pressures (Tomich et al., 2010). Early applications of the framework were as a tool in adaptive management, particularly in marine and coastal sectors (Vugteveen et al., 2015).

The MA introduced feedback loops and drivers of change as well as considering multiple spatial and temporal scales. Its central idea is that human wellbeing is tightly linked to the condition of ecosystems and the provision of different broad types of ecosystem services (MA, 2005). Therefore, it is assumed that their loss or degradation will have negative consequences for human wellbeing, offering the potential for ecosystem management and restoration to play a part in development strategies aimed at poverty alleviation. Another legacy of the MA has been the assumption of the trickle-down effect whereby improvements in ecosystem services should automatically increase wellbeing. This has led to consideration, especially within ESPA, of the differential impacts of changes in ecosystem service flows, and the trade-offs among different social groups, especially on those who depend most directly on ecosystem services (Daw et al., 2011; Fisher et al., 2013).

TEEB was launched as an economic complement to the MA. It took a more systematic approach to economic valuation as a contribution to biodiversity and ecosystems, giving them a more central role in decision making (Kumar, 2010) and helped to enrich economic conceptualisation of ecosystem service values (e.g. Pascual et al., 2010). Interestingly, while the TEEB approach has been heralded in many national and international policy fora, it has fallen short of being taken up in practice by governments or industry, partly because it did not fully connect to the realities of decision-making processes and also because the aggregate economic valuations it produced were difficult to put into practice (Wegner and Pascual, 2011).

More recently, the IPBES conceptual framework (Díaz et al., 2015) recognised strong feedbacks among ‘nature’ (i.e. biodiversity and ecosystems), NCP (including ecosystem services) and people’s ‘good quality of life’ (including wellbeing) via the centrality of institutions and governance systems. The IPBES framework aims to represent different epistemic communities and knowledge systems, including, prominently, those of indigenous peoples and local communities. This is also reflected in the diverse conceptualisations of values of NCP (Pascual et al., 2017a). It is
important to note that while the MA was not part of an intergovernmental process, its frameworks and findings were recognised by the Parties to the Convention on Biological Diversity (CBD). The IPBES framework, on the other hand, is officially endorsed by all 129 governments who are members of this intergovernmental body.

Stemming from the MA and, more directly, the UK NEA, the ESPA programme framework aims at researching the general vision that ecosystems, when sustainably managed, can contribute to poverty alleviation as well as to inclusive and sustainable economic growth (ESPA, 2013). Various reviews highlighted key dimensions
for understanding and assessing the impact of ecosystem changes on human wellbeing (Fisher et al., 2013; Agarwala et al., 2014). These include, *inter alia*, the importance of multidisciplinary consideration of wellbeing (Suich, 2012), the need for frameworks that integrate subjective and objective aspects of wellbeing, the central importance of context and relational aspects of wellbeing, the constraints of access and of aggregate availability of ecosystem services, and the differences between ecosystem services, their pathways of co-production, distribution across social groups, and contributions to improving the livelihood of the poor and vulnerable people, most often in rural settings (Keane, 2016).

In general, the ecosystem services approach has not explicitly considered either the relationship between biotic (e.g. biodiversity and ecosystem service flows) and abiotic components (e.g. fossil fuels and minerals) associated with development pathways, or the relationships between different biotic elements and different constituents of wellbeing. Interestingly, the role of abiotic resources in development pathways has remained out of focus of ESPA, and this may perhaps have limited some of its potential policy uptake in relation to poverty alleviation. By contrast the focus on multi-dimensional aspects of human wellbeing, as well as how differentiated access to ecosystem services plays a role in poverty alleviation, has been a key contribution compared with other programmes.

Throughout the evolution of these frameworks, key concepts like the co-production of ecosystem services, systems thinking and the centrality of governance at the ecosystem service–wellbeing nexus, have come to the fore, drawn from emerging research. Each of the frameworks discussed has embraced these concepts to differing degrees depending on when they first appeared and the interests of associated epistemic communities. Figure 1.2 illustrates the emphasis that different frameworks place on biophysical, human wellbeing and governance concepts. We observe a gradual increase in depth of colour from left to right illustrating a greater inclusion of key concepts within newer ecosystem service frameworks.

Figure 1.3 illustrates how these concepts are connected to different epistemic communities (as grouped by Howe et al., 2018) and, in turn, influence core frameworks. The epistemic communities have evolved over time from integrated conservation and development projects (ICDPs), through a drive to address environmental conservation and poverty eradication (Adams et al., 2004), to attempts to achieve the holy grail of win–wins for both environmental conservation and poverty alleviation. Inevitably, the key concepts have been championed and/or embraced to differing extents. Together Figures 1.2 and 1.3 illustrate the interplay between epistemic communities and the various core frameworks, and provide an overview of how frameworks, emerging concepts and associated epistemic communities have co-evolved over the last decade.

**Biodiversity and poverty nexus in ecosystem service framings**

Generally, the frameworks consider biodiversity as an underlying driver, a regulator of the processes and flows providing ecosystem services, and/or as a final benefit
FIGURE 1.2 Predominance of key conceptual attributes within the ‘core’ and ‘satellite’ frameworks discussed in Figure 1.1. The intensity of the colour indicates whether a framework interacts lightly or heavily with the attribute. Green concepts relate to the biophysical, blue to human wellbeing and yellow to governance elements of the conceptual framings.
or good in itself (Mace et al., 2012). In earlier core and satellite frameworks, assumptions made about the relationships between biodiversity and ecosystem services were unclear and often confused by different uses of the terms (e.g. biodiversity, supporting ecosystem services, ecosystem benefits). In addition, in some later frameworks, such as the UK NEA (Bateman et al., 2011) and IPBES (Díaz et al., 2015), ecosystem services were specifically separated from benefits. The reasoning for this is that for robust valuation, the change in benefits perceived or realised by people needs to be attributable to changes in ecosystem contributions as well as non-ecosystem/anthropogenic contributions (Fisher and Turner, 2008).

Most commonly, only direct (often consumptive) uses of biodiversity and their positive links to limited dimensions of poverty (income, assets and food security) are studied (Roe et al., 2012). However, the realities concerning the linkages between biodiversity conservation and the provision of ecosystem services for the poor are significantly more complex. For example, many geographical areas of high biodiversity are located in countries that are relatively poor, and the favoured conservation approach of protected areas has often disadvantaged local residents and neighbours (Martin et al., 2013). At larger spatial scales and over much longer

FIGURE 1.3 Relationship between key emergent conceptual attributes, epistemic communities and core frameworks. Epistemic community principles (as grouped by Howe et al., 2018) are linked to key concepts and core frameworks, in order to illustrate the interaction within and between epistemic communities and examples of core framings for ecosystem services (MA, ESPA and IPBES). The thickness of the arrows represents the level of association.
time frames, loss of functional and phylogenetic biodiversity may pose more significant obstacles to ecosystem services and associated benefit flows (Mace et al., 2014). Over the short term, however, local loss of biodiversity may not always be an obstacle to poverty alleviation and is often associated with increased income.

Different frameworks deal with human wellbeing in different ways. The MA drew on the *Voices of the Poor* exercise for the five elements of wellbeing used (MA, 2005; Narayan et al., 2000), and while many more dimensions have been identified in the literature (Alkire, 2007), relatively few (easier to measure) dimensions (e.g. income, employment, health) have been the main focus of research. Generally, the MA framework did not make explicit the poverty aspects associated with wellbeing and took for granted that ecosystem services help the poor through an undefined trickle-down effect. ESPA has maintained a stronger focus on the wellbeing of the poor (as opposed to people more generally) and the impact of ecosystem services management on alleviating poverty, including via additional conceptualisations and complex pathways, e.g. capabilities, freedoms and security (Keane, 2016).

Ecosystem services are more likely to be associated with poverty prevention, rather than reduction, or with mitigation of some impacts of poverty (Dewees et al., 2010), as they act as a safety net rather than a route out of poverty (Fisher et al., 2013). However, many other aspects of the relationship between ecosystem services and poverty remain unresolved. This is in part due to the lack of empirical evidence on multi-causal pathways and time lags, as well as the complexity of conceptualising poverty in its different dimensions (Suich et al., 2015; Martin et al., this volume). The core conceptual frameworks do not fully address the direction of causality, i.e. whether poverty creates or is a result of environmental degradation (Sandker et al., 2012), whether ecosystem services can actually alleviate poverty (Suich et al., 2015) or whether a reliance on ecosystem services might in fact contribute to continuing poverty (Fisher et al., 2013).

Recent ecosystem service framings are moving towards acknowledging interconnected social-ecological systems (Reyers and Selomane; Dearing, both this volume). In both ESPA and IPBES, there has been a shift from an ecological framing of ecosystem services to a more balanced social-ecological construct between biophysical dynamics, people’s perceptions, needs and values for ecosystem services, and governance systems. This is clearly exemplified in Díaz et al. (2018), where the notion of NCP is highly dependent on the cultural lens and thus the perceptions of the relationships between ecosystem flows and human wellbeing, beyond a linear and somewhat deterministic stock-flow view. Alongside this shift in framing, there has been an increase in conceptual developments on e.g. plurality of values (Pascual et al., 2017a), co-production of ecosystem services (Reyers et al., 2013), power and justice (Sikor, 2013), social-ecological trade-offs (Bennett et al., 2015) and disaggregation of poverty analyses (Daw et al., 2011).
Emergent conceptual attributes from the evolution of ecosystem service frameworks

Several significant conceptual attributes have emerged over the last decade.

On co-production of ecosystem services

The benefits or contributions from ecosystems arise through a combination of ecosystem entities and anthropogenic assets (including human and social capital, financial and man-made capital) as well as human labour. For example, the production of food as provisioning service requires different inputs such as regulating soil services, farmers’ labour and knowledge. This process is understood as ecosystem service co-production (Reyers et al., 2013). It is important for understanding not only how ecosystem services are supplied, but also how service flows are distributed, since intermediary services and anthropogenic inputs are unequally distributed across ecosystems and accessible by different actors in society (Lele et al., 2013; Reyers et al., 2013). Palomo et al. (2016) differentiate two main types of co-production: the physical processes which affect final ecosystem service flows via ecosystem management using anthropogenic assets; and the cognitive processes, which are shaped, for example culturally by social norms, leading to changes in ways that ecosystem services are demanded and/or enjoyed.

A focus on ecosystem service co-production is key to understanding their delivery and distribution across social groups. Thus, we can understand the differentiated impacts on wellbeing among different people given different levels of access, control and use of anthropogenic assets and labour (Berbés-Blázquez et al., 2016). The MA acknowledged the role of different capital types in the co-production of ecosystem services, and the NCP approach of IPBES has (physical and cognitive) co-production as a central focus. However, the potential effects of different ecosystem service co-production pathways for poverty alleviation remain under-studied and under-conceptualised, and the nested nature of co-produced ecosystem service bundles via ecological and social factors is not yet well understood. For example, it is not the same to allocate young girls’ time vis-à-vis adults’ time to fuelwood collection, as this may impact on the young girls’ time for investing in human capital (e.g. formal education), which in turn may affect the co-production of other ecosystem services that may require certain levels of education and in particular, the increased education of girls/women.

On power and justice

Framing ecosystem services for poverty alleviation should help understand how social power relations shape ecosystem service governance systems that regulate entitlements to ecosystem resources. Focusing on power dynamics can help to better identify social trade-offs in terms of winners and losers (Berbés-Blázquez et al., 2016). The use of an ecosystem service framework, without acknowledging
the role of power relations, risks reifying the status quo at the expense of already disenfranchised social groups. In general, ecosystem assessments do not yet capture, nor question, the role that power relations play in the co-production of ecosystem services and the associated distribution of benefits and burdens across social groups.

Ecosystem service ‘cascades’ have been proposed to identify intermediary steps between biophysical functions of ecosystems and changes in human well-being (Haines-Young and Potschin, 2010). They have not yet been applied to identify social factors determining the operation of critical intermediate steps in the cascade. For example, this could involve identifying who may have power at any one of these stages in order to determine who has access to an ecosystem service, at the expense of others, thereby determining who wins and who loses. Likewise, the cascade approach could be adapted to help to identify whether, and to what extent, political and economic elites capture a disproportionate share of benefits from ecosystem services, as well as the systematic displacements of ecosystem service burdens onto disadvantaged actors as a result of power differentials (Berbés-Blázquez et al., 2016).

Including social power relations in ecosystem service frameworks is vital to understand the institutional context of differentiated social groups (Brown and Fortnam, this volume). For instance, land allocation and associated ecosystem service flows in many African societies respond less to productivity and more to cultural considerations via gender roles, allowing certain social elites to maintain control (Berbés-Blázquez et al., 2016). It follows that justice must become more central in ecosystem service framings, considering socially differentiated groups of people with respect to wealth, power, gender and identities (Sikor, 2013; Dawson et al., this volume).

On ecosystem service valuation

The MA, and subsequent core frameworks, emanate from understanding ecosystems or nature as a capital asset. In this context, the flow of ecosystem services can be viewed as the return or interest that society receives from natural capital, thus implying that the level of the interest (ecosystem service quality or quantity) changes as the level/quality of the asset changes (ecosystems are altered). The MA recognised that economic valuation was hard to achieve consistently and with confidence, as peoples’ preferences for ecosystem services are hard to express, and many, especially those associated with intangible benefits (e.g. cultural ecosystem services), do not easily lend themselves to economic valuation (Carpenter et al., 2009).

Framings for the economic valuation of ecosystem services have led to three important conceptual anchoring points: (i) ecosystem services are as dependent on ecological functions and processes as they are on social constructs connected to benefits and thus are experienced or perceived by people (Wegner and Pascual, 2011); (ii) separating intermediate (e.g. crop pollination as regulating service) from
final ecosystem services (e.g. crop production) and benefits to people (e.g. food contributing to health) is key to avoiding double counting in economic valuation of ecosystem services (Fisher and Turner, 2008); and (iii) an ecosystem property such as resilience ought to be seen as an intermediate ecosystem service in as much it provides stability to the flow of benefits via final services, and can be connected to the idea of the insurance value of biodiversity (Pascual et al., 2010).

Efforts to conceptualise ecosystem service values using an economic logic have resulted in frameworks downplaying non-instrumental values and the significance of ethical or other held-values/principles that people associate with nature (Jax et al., 2013). As a reaction to the dominant utilitarian approach of the ecosystem service framing, new ways of conceptualising ecosystem service values are being proposed, e.g. via the idea of ‘relational values’ (Chan et al., 2016). This is impacting the way IPBES, among others, is embracing the idea of wellbeing outcomes from ecosystem services beyond utilitarian benefits, and extending it to aspects of agency, place identity, empathy, caring for nature and other-regarding actions (Pascual et al., 2017a). Opening up the valuation space beyond a restricted utilitarian approach can create a more integrated social-ecological perspective where ecosystem service flows are associated with broader notions of sustainability and justice. It is important to recognise that valuation is itself a value-laden process and that power relations through which such different types of values are expressed ought to be a central component of valuation exercises and assessments.

Application and implications of ecosystem service framings for policy instruments

Ecosystem service–wellbeing framings have had some impact on the way certain policies are formulated and designed, as well as being influenced in their turn by policy–making processes.

Perhaps the key policy instrument that has been most greatly influenced by epistemic communities and their preferred ecosystem service framings, is centred on Payments for Ecosystem Services (PES). PES emerged in the early 2000s as a direct response to strict biodiversity conservation approaches based on ‘command and control’ or ‘fences and fines’ (mostly in the form of protected areas of high ecological and biodiversity value). These approaches were criticised for the lack of voluntary engagement of ecosystem service providers (Menton and Bennett; Porras and Asquith, both this volume), as well as a neglect of social concerns on the impacts on local communities (Adams et al., 2004). Different epistemic communities, each with their own value framings and interests, have influenced the various framings behind PES, understood as payments, compensations or rewards, and therefore their design (Menton and Bennett, this volume).

In areas with high poverty levels, the general logic behind pro-poor PES schemes is that providers of ecosystem services are poor landholders or disadvantaged communities, and that direct and conditional payments can contribute to poverty
alleviation of people voluntarily participating in the schemes, resulting in the alluring possibilities of ‘win-win’ outcomes (Muradian et al., 2013). Furthermore, for some scholars PES is thought to play a role in development in the Global South, as the cost-effectiveness of the schemes cannot be achieved without integrating complementary social objectives, especially with regards to social equity (e.g. Pascual et al., 2014). In practice, PES cannot be decoupled from political aspects of social legitimacy, and perceptions of social equity and fairness. Conceptual framings of the ecosystem service–poverty nexus, and experience in the implementation of PES schemes suggests that PES requires contextually customised design features, in terms of *inter alia* payment vehicles and contract flexibility, as well as understanding of impacts of payment differentiation, spatial targeting and enforced conditionality that prove highly sensitive to the social and poverty contexts (Engel, 2016).

The evolution of ecosystem service framings for human wellbeing, along with ideas of co-production, power relations, equity and justice, nourished by knowledge generated from programmes such as ESPA, is having a discernible impact on how PES is being reframed. There seems to be less emphasis on monetisation and a greater focus on the transfer of resources at the complex interplay between conservation and rural development, though still with a discernible conditionality component (Menton and Bennett; Porras and Asquith, both this volume).

**Conclusions**

Since the twin milestones of the Sustainable Livelihoods Framework (Scoones, 1998) and the MA (2005), epistemic communities and their associated core and satellite ecosystem service frameworks have attempted to refine and improve ways to frame human–nature relations. There have been many routes for this, for instance by enlightening an economic dimension beyond commodification, stressing social-ecological co-production processes, linking institutions and governance systems with power relations and environmental justice frames as filters through which we can better understand the way management of ecosystems creates winners and losers, and opening the space of valuation of ecosystem services. The ESPA programme has influenced and has been influenced by this kaleidoscopic evolution of conceptual frameworks.

There is an ethical mandate and responsibility for the research and decision-making communities to broaden the Western utilitarian ecosystem service framing, derived from the legacy of a linear stock-flow type of human relationship with nature. It is necessary to re-politicise how, why, by whom and for whom ecosystems are managed and how concepts such as wellbeing and poverty are articulated (Fisher et al., 2013). The design and implementation of policy instruments do not occur in an intellectual vacuum. They are affected by, and in turn affect, the way ecosystem service–poverty frameworks are devised given the interplay of epistemic communities and communities of practice. Policy instruments, such as PES, appear to be preferred by policy makers to connect the conservation of ecosystem
service flows and developmental objectives in the Global South; however, as conceptual framings and epistemic communities continue to evolve, so will PES programmes.

Since the MA, investment in the scientific evidence base through programmes such as ESPA and other connected institutions and programmes (especially those connected to social-ecological system thinking) have influenced the lens through which the links between ecosystem services and wellbeing are seen in more inclusive ways. This evolution is also having an impact on the interpretation of human–nature relations at the science–policy interface, and can potentially influence the ways in which different epistemic communities work in the context of the Sustainable Development Goals. Allowing different knowledge systems to engage in dialogue and more inclusive (and culturally legitimate) co-construction of knowledge about the ecosystem service–wellbeing nexus is vital. But at the same time, we face accelerated global environmental change, including dramatic biodiversity loss and the ethical imperative to raise millions of people out of poverty, many of whom, directly or indirectly, rely on protecting and enhancing ecosystem service flows now and in the future. In this context, we point to two future challenges for research in ecosystem service framings.

First, we need to avoid thinking that a single framework, whether narrow or complex, can be the solution. While simple frameworks may help different epistemic communities to agree on common concepts and linkages, they may also promote the idea that win-win outcomes are easily attainable. Complex frameworks on the other hand, while recognising the intricacy of social-ecological systems, may still fall short of some key elements (such as power imbalances and associated political economy) and be too intricate for decision makers or researchers to use. At this stage, it may be that we have yet to find a way of describing the ecosystem service for human wellbeing relationships in a way that really speaks to all people, and this must continue to be one focal point of future efforts.

Second, given the exponential increase in the capability of information systems, there may be a strong temptation to develop ecosystem service approaches that are fixated in data-hungry technological developments, e.g. creating overly detailed typologies of ecosystem services, unduly enhancing the resolution of ecosystem service maps, etc. Such an approach can compete with alternative ways of mobilising funding and effort into more transdisciplinary collaborations among different epistemic communities in order to co-construct knowledge that can empower those often without a voice in debates and decision making, such as the landless and marginalised indigenous peoples. Of course, there is also an ideal position which implies devising and applying new technology that can foster data acquisition, processing and interpretation to help foster trans-disciplinary approaches with the active participation of stakeholders, including those who most need it in terms of poverty alleviation. Such involvement should also ideally help identify technological development that better favours more informed decision making across sectors, from poor households, development and conservation NGOs, business and governments.
We recommend that future frameworks reflect basic relationships between ecosystem services and human wellbeing but are adapted for the specific purpose they seek to address. In this manner, they can find a way of describing our relationship with nature that speaks to all people and thus can offer the potential to achieve social-ecological transformations required for environmental sustainability and poverty alleviation.

References

(ESPA outputs marked with ‘*’)


