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Applying ecosystem services for pre-market environmental risk assessments of regulated stressors

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Abstract

Ecosystem services (ES) are the benefits that people obtain from ecosystems. Investigating the environment through an ES framework has gained wide acceptance in the international scientific community and is applied by policymakers to protect biodiversity and safeguard the sustainability of ecosystems. This approach can enhance the ecological and societal relevance of pre-market/prospective environmental risk assessments (ERAs) of regulated stressors by: (1) informing the derivation of operational protection goals; (2) enabling the integration of environmental and human health risk assessments; (3) facilitating horizontal integration of policies and regulations; (4) leading to more comprehensive and consistent environmental protection; (5) articulating the utility of, and trade-offs involved in, environmental decisions; and (6) enhancing the transparency of risk assessment results and the decisions based upon them. Realisation of these advantages will require challenges that impede acceptance of an ES approach to be overcome. Particularly, there is concern that, if biodiversity only matters to the extent that it benefits humans, the intrinsic value of nature is ignored. Moreover, our understanding of linkages among ecological components and the processes that ultimately deliver ES is incomplete, valuing ES is complex, and there is no standard ES lexicon and limited familiarity with the approach. To help overcome these challenges, we encourage: (1) further research to establish biodiversity–ES relationships; (2) the development of approaches that (i) quantitatively translate responses to chemical stressors by organisms and groups of organisms to ES delivery across different spatial and temporal scales, (ii) measure cultural ES and ease their integration into ES valuations, and (iii) appropriately value changes in ES delivery so that trade-offs among different management options can be assessed; (3) the establishment of a standard ES lexicon; and (4) building capacity in ES science and how to apply ES to ERAs. These development needs should not prevent movement towards implementation of an ES approach in ERAs, as the advantages we perceive of using this approach render it more than worthwhile to tackle those challenges. Society and the environment stand to benefit from this shift in how we conduct the ERA of regulated stressors.

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1. Introduction

On 1 January 2016, the 17 sustainable development goals of the 2030 Agenda for Sustainable Development officially came into force (United Nations, 2015). These sustainable development goals aim to end poverty, protect the planet, and ensure prosperity for all, requiring a healthy and productive environment. The 2030 Agenda is consistent with the Strategic Plan for Biodiversity 2011–2020 and its Aichi Biodiversity Targets adopted under the Convention on Biological Diversity in 2010 that set the global framework for priority actions on biodiversity (i.e. the variety of genes, species, or functional traits in an ecosystem).

The conservation and sustainable use of biodiversity, along with the safeguarding and restoration of ecosystems, feature prominently across many of the sustainable development goals and associated targets, as they contribute directly to human well-being and development priorities. The accelerating and unprecedented loss of biodiversity and changes to the functioning of ecosystems caused by anthropogenic drivers have prompted deep concern regarding adverse ecological effects and reduced delivery of ecosystem services (ES) (e.g. EEA, 2015; IPBES, 2019).

Ecosystem services can be defined as the suite of benefits or as the direct and indirect contributions that ecosystems provide to human well-being. They include 'goods' such as clean water, food and fibre (i.e. provisioning services) and process-based benefits such as climate regulation, pest and disease control and flood alleviation (i.e. regulating services). They also include cultural services such as recreational benefits, spiritual benefits and aesthetics. Although biodiversity is usually not explicitly mentioned as an ES, it underpins ES and plays an essential role in sustaining ecosystem functioning and the ability to provide benefits to humans (Potts et al., 2010, 2016; Cardinale et al., 2012; Mace et al., 2012).

Investigating the environment through an ES framework has gained broad interest in the international scientific community. Through the inclusion of ES, an ES approach ensures that the way in which nature benefits people is more clearly understood and explicitly included in environmental policy and management decisions (Beaumont et al., 2017; Costanza et al., 2017; Ainscough et al., 2019). In the European Union (EU), the 2020 Biodiversity Strategy aims to halt the loss of biodiversity and improve the state of Europe's species, habitats, ecosystems and the services they provide over the next decade.¹ Under Action 5 of the 2020 Biodiversity Strategy, EU Member States are called to map and assess the state of ecosystems and their services in their national territory, assess the economic value of such services, and promote the integration of these values into accounting and reporting systems at EU and national levels by 2020.² At the global level, the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) assesses the state of the planet's biodiversity, its ecosystems and the contributions they provide to society (Pascual et al., 2017a; Díaz et al., 2018; IPBES, 2019).

The environmental risk assessment (ERA) and risk management of regulated stressors connected to food and feed production (such as genetically modified organisms, plant protection products and feed additives) contribute to achieving environmental protection and thus global goals for sustainable development. Pre-market or prospective ERAs evaluate how likely it is that the environment, including biodiversity and ecosystems, may be impacted by the deployment of a regulated stressor, and to what degree. The legislative frameworks regulating stressors define general protection goals. These protection goals can vary among jurisdictions, but their overall aim is to reduce the harm to the environment, including biodiversity and ecosystems, caused by human activity.

An ES approach has the potential to enhance the ecological and societal relevance of the ERA of regulated stressors (Faber et al., 2019). It offers a practical framework that can be used to derive operational protection goals in a systematic, comprehensive and transparent manner, and facilitate communication about risks and benefits among stakeholders (Devos et al., 2015, 2016). While applying an ES approach to ERAs holds much promise for environmental decision-making (Munns et al., 2017), it can entail practical challenges. Here, we briefly describe some of the ES classifications, and subsequently address some of the opportunities, challenges and implications of applying an ES approach to ERAs, considering the experience gained in the context of the 'Chemicals: Assessment of Risks to Ecosystem Services' project. Suggestions on how to overcome these challenges, as well as areas of future work are discussed.

¹ http://ec.europa.eu/environment/nature/biodiversity/policy/index_en.htm

² Action 5 is implemented by the working group on Mapping and Assessment of Ecosystems and their Services. Further information is available at <http://biodiversity.europa.eu/maes>

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2. Examples of ES classifications

Any application of an ES-based approach generally starts with selecting the services to be assessed (and valued) from a list of services, that is, an ES classification system. Several ways of defining and classifying ES have been developed. It is thus important to describe the main classification systems to highlight their underlying concepts (Costanza et al., 2017; La Notte et al., 2017).

The Millennium Ecosystem Assessment (MEA)⁴ is the first to attempt to group ES into four categories: (1) provisioning services (e.g. food/feed, fibres, water, energy, genetic resources); (2) regulating services (benefits obtained from the regulation of ecosystem processes; for example, pollination, control of pests and diseases, purification of water and air); (3) cultural services (non-material benefits obtained from ecosystems through recreation and aesthetic experiences; they include ecotourism, cultural heritage, knowledge systems, and spiritual and religious values); and (4) supporting services (services that are necessary for the other ES to function, such as nutrient cycling, soil formation, oxygen production or habitat provision).

Haines-Young and Potschin (2010) and Potschin and Haines-Young (2011) developed the ES cascade framework that links ecosystem functions to ES, to benefits and then to value and present them as a production chain. In this framework, supporting services are considered a 'function' rather than a 'service'. Following the MEA, the Economics of Ecosystems and Biodiversity (TEEB)⁵ classification also explicitly refers to the cascade framework but refines the distinction between 'services' and 'benefits'. The idea of supporting services in TEEB is not further developed. Instead, 'habitat services' are introduced as an additional ES category, including 'maintenance of life cycles' and 'maintenance of genetic diversity'.

The Common International Classification of Ecosystem Services (CICES)⁶ builds on the MEA and TEEB classifications and merges the four MEA categories into three categories: (1) provisioning services; (2) regulating and maintenance services; and (3) cultural services. In CICES, the supporting services, as proposed by MEA, are treated as part of the underlying structures, processes and functions that characterise ecosystems; they are pooled with the regulating services to tailor the classification to economic accounting.

Other classifications offer more economically focused definitions, and distinguish among intermediate services, final services, benefits, and well-being values. Final ES are those components of nature that are directly enjoyed, consumed or used to enhance human well-being, while others are referred to as intermediate services (Boyd and Banzhaf, 2007; Munns et al., 2015a). Intermediate ES contribute to ecosystem resilience and support the sustained provision of final ES (Lamothe and Sutherland, 2018). Values may be economic, health, or shared/social. In the latest version of the cascade framework that underpins CICES, ES are explicitly indicated as final services, while biophysical structure and function are considered to be supporting or intermediate services.

Two related classifications have been developed to standardise the accounting of ES across the spectrum of ecosystem types and the beneficiaries of those services. The National Ecosystem Services Classification System (NESCS; US EPA, 2015) is based on the existing hierarchical classification and accounting systems used for economic goods and services. It reflects both supply-side (i.e. production of ES) and demand-side (i.e. human uses and users of ES) components that 'produce' human well-being. It is designed to inform a range of policy impact analyses (e.g. cost–benefit analysis of environmental regulations) and decision contexts. The Final Ecosystem Goods and Services Classification System (FEGS-CS; Landers and Nahlik, 2013) shares many of the attributes of NESCS, focusing on independent components of ecosystem types (supply) and beneficiaries of the ES produced (demand). Both systems emphasise final ES to avoid double counting in benefit calculations.

More recently, IPBES developed the Nature's Contributions to People (NCP) framework. This framework refers to three broad groups: (1) regulating NCP; (2) material NCP; and (3) non-material NCP. NCP, which embodies ES, reflects a conceptual evolution based on more than a decade of interdisciplinary thinking, with increasing involvement from the social sciences and humanities. The classification considers that people's perception and experience of NCP are influenced by the cultural

³ All conference materials are available at <https://www.efsa.europa.eu/en/events/event/180918>

⁴ <http://www.millenniumassessment.org/en/index.html>

⁵ <http://www.teebweb.org/>

⁶ <http://cices.eu/>

context and emphasises the importance of sociocultural relations with nature. Hence, 'cultural ES' are not a separate category, and the role of culture is given more prominence (Díaz et al., 2018). The concept of NCP provides greater opportunities to incorporate non-monetary values than are available with the dominant use of an ES approach. The NCP framework is consistent with the IPBES approach to values, where diverse methods of valuation are recognised (Pascual et al., 2017a). Consequently, NCP can be beneficial or detrimental to people, depending on the cultural context.

3. Opportunities for ES-based ERAs

An ES approach can provide a comprehensive framework for considering nature's contributions to human well-being in ERAs and risk management of regulated stressors (e.g. Galic et al., 2012; Olander and Maltby, 2014; Devos et al., 2015; Munns et al., 2015b, 2017; SEP, 2015; EFSA, 2016; Mulder et al., 2016; Maltby et al., 2017a,b, 2018; Van den Brink et al., 2018; Faber et al., 2019). Incorporation of ES into ERAs can: (1) inform the derivation of operational protection goals; (2) enable the integration of environmental and human health risk assessments; (3) facilitate horizontal integration of policies and regulations; (4) lead to more comprehensive and consistent environmental protection; (5) articulate the utility of, and trade-offs involved in, environmental decisions; and (6) enhance the transparency of risk assessment results and decisions based on them (Munns et al., 2017).

3.1. Deriving operational protection goals

Environmental policy protection goals (e.g. protecting biodiversity and ecosystems) establish the context for ERAs by describing the components of the ecosystem that should be protected and thus considered during ERAs. Since such goals are generally broadly stated (Sanvido et al., 2012), refinement is required to make them operational; they need to be translated into specific, operational goals (also termed 'specific protection goals' or 'assessment endpoints') (Nienstedt et al., 2012; Devos et al., 2014, 2015; Garcia-Alonso and Raybould, 2014; Van den Brink et al., 2018).

Selck et al. (2017) recommended that protection goals be articulated on two levels: (1) universal protection goals that are broadly stated and generally desirable (e.g. 'maintaining biodiversity'); and (2) operational (or as they described it, 'tangible'), site-specific, region-specific, or context-specific protection goals that are relevant to the given risk management decision (e.g. population of bees as drivers for the 'pollination of crops' as an ES). They take the view that the translation of universal protection goals into operational protection goals be informed by science, societal values, and policy considerations. In their construct, ES are intended to focus protection goals at both levels, and to serve as the basis for both risk assessment and risk management (US EPA, 2016).

In the EU, the European Food Safety Authority (EFSA) has taken the lead in exploring the use of an ES approach for setting operational protection goals for several regulated stressors connected to food and feed production, such as genetically modified organisms, plant protection products and feed additives (EFSA, 2010a,b, 2016; Nienstedt et al., 2012; Devos et al., 2015). This framework has been shown to be potentially applicable to other stressors (Maltby et al., 2017a). EFSA's ES approach to defining operational protection goals follows three sequential steps: (1) identifying relevant ES potentially impacted by the use of regulated products; (2) identifying service-providing units – structural and functional components of biodiversity (Kontogianni et al., 2012) – that provide or support these ES; and (3) specifying the level of protection for these service-providing units. The level of protection is defined by: the ecological entity (e.g. a functional group) of the service-providing unit and its attributes, as well as the maximum magnitude and spatial and temporal scale of tolerable impacts (EFSA, 2016).

Instead of generating operational protection goals on a case-by-case basis, the US Environmental Protection Agency (US EPA) defined generic assessment endpoints that are valid for all regulated stressors, as this ensures consistency between regulated stressors when protecting the environment from harm (Suter, 2000; Suter et al., 2004). These generic assessment endpoints were subsequently expanded to encompass ES (Munns et al., 2009, 2015b). The application of ES-based generic assessment endpoints in ERAs can provide an improved means of communicating risks and informing management decisions because incremental changes in the endpoints directly or indirectly benefit humans (Selck et al., 2017).

The integration of ecological and societal objectives achieved by using ES to set operational protection goals can enhance the transparency of risk assessments and regulatory decision-making. Moreover, reaching agreement on operational protection goals and criteria from which to derive such goals will increase the value of ERAs by providing the information necessary for effective decision-making.

Collected data and their interpretation can then be directed towards evaluating the impact of any observed effect on what is desirable to protect.

3.2. Enabling integration of environmental and human health risk assessments

The risks of regulated stressors to humans and ecological systems are typically considered separately. Although the rationale for integrating environmental and human health risk assessments has been identified and approaches for doing so proposed (e.g. Munns et al., 2003a,b; Suter et al., 2003), risks to humans and the environment are generally not integrated systemically in decision-making (Munns et al., 2017). This is also true for the evaluations required by Registration, Evaluation, Authorisation and Restriction of Chemicals (REACH) in the EU and the Toxic Substances Control Act in the US, both of which address protection goals for human health and ecosystems separately.

Ecosystem services can be a 'common currency' that enables a holistic assessment of risks to human health and ecosystems (Munns et al., 2009, 2017). Changes in ES delivery that result from exposures to regulated stressors are known to affect human well-being, including many aspects of health (see for example Jackson et al., 2013). While much work remains to develop the relationships and techniques needed to evaluate systemic risk (risk to a broader system than just a single assessment endpoint), risk assessments that include ES can promote more holistic decision-making by representing both ecosystem and human health outcomes.

3.3. Facilitating horizontal integration of policies and regulations

Environmental policies, regulations and regulatory programmes often focus on specific environmental compartments (e.g. water, air, or soil) and on single stressors because of legislated goals, institutionalised governance structures and regulatory frameworks. This situation gives rise to a kind of 'stove-piping' or 'silo-ing', in which decisions and actions concentrate largely on single environmental components without broader consideration of the effects that regulated stressors can have on other parts of the environment. Three problems can result from this: (1) inefficiencies are created as programmes independently collect data, perform analyses and take actions to manage different environmental risks; (2) there can be unanticipated and undesirable environmental and human well-being consequences because issues are not considered holistically; and (3) actions taken to support one regulation may be inconsistent with or even contradict actions taken to support other regulations (EFSA, 2015a; Munns et al., 2017).

Like other higher level assessment endpoints, ES can be affected by various stressors acting across several environmental components and over a range of spatial and temporal scales. The use of ES as additional operational protection goals can guide ERAs to consider the full social–environmental system, including cumulative effects arising from the exposure to different regulated stressors. ES approaches provide a common analytical framework and currency to integrate ERAs across multiple stressors, multiple scales and multiple environmental compartments offering the potential for more holistic environmental management. This framework facilitates the alignment of regulatory programmes and promotes synergies that help to minimise unanticipated consequences resulting from regulatory stove-piping (Munns et al., 2017; Rortais et al., 2017; Selck et al., 2017).

3.4. Leading to more comprehensive and consistent environmental protection

Ecosystem services are the products of nature whose delivery are quantitatively represented by ecological production functions, which can be defined as 'the types, quantities, and interactions of natural [biophysical] features required to generate ES' (Munns et al., 2015a,b) or more operationally as 'usable expressions (i.e. models) of the processes by which ecosystems produce ES, often including external influences on those processes' (Bruins et al., 2017). Regulated stressors can affect (positively or negatively) multiple parts of the network of biotic and abiotic features and their interactions that contribute to ES delivery. Operational protection goals that are based on ES, particularly those that directly benefit people (i.e. 'final ES'; Boyd and Banzhaf, 2007), orient ERAs toward portions of the ecological systems contributing to ES production (Munns et al., 2017). Direct risks to ES might also reflect risks to the underlying components of their ecological production functions because the latter are needed to produce those services. When combined with the conventional ecotoxicological endpoints commonly evaluated in ERAs, and the opportunities for horizontal integration of regulatory

programmes described above, operational protection goals defined by ES enable regulatory decision-making to encompass a more comprehensive set of ecological and societal objectives (Munns et al., 2015b). As a result, regulatory decisions will be better informed, comprehensive, and scientifically and societally defensible.

3.5. Articulating the utility of, and trade-offs involved in, environmental decisions

When based solely upon conventional ecotoxicological endpoints, ERAs per se provide little information regarding the potential benefits of those actions, because they focus on adverse effects on non-target organisms. Although regulatory decision makers consider management goals and objectives in addition to the mitigation of risks as part of the evaluation of various decision options, they typically lack the information needed to articulate the full range of benefits to society that result from the decision.

Ecosystems and biodiversity are not exclusively benign to humans, and regulated chemicals used to counter ecosystem disservices (e.g. pests, pathogens, disease vectors; ecosystem processes and functions that affect humans in 'negative' ways, causing damage and costs) may have positive and negative effects on human well-being (Maltby, 2013; Potts et al., 2016). This triggers a thorough evaluation of the potential risks to both humans and the environment. However, where risks to the environment cannot be excluded, the challenge facing decision makers is how to balance the well-being benefits provided by the use of plant protection products with the potential well-being costs via habitat degradation and loss of biodiversity. ES provide a common framework within which to compare well-being costs and benefits.

Quantitative measures of the changes in ES expected from decision options articulate the incremental benefits of those options in ways that policymakers and the public can understand and will care about. Alix et al. (2014) and Deacon et al. (2016) provided an early illustration of this. They quantified the changes in indicators of 10 ES (drinking water provision, air quality, water flow regulation, extreme event moderation, soil erosion prevention, cultural services, food provision, pollination, habitat services and aquatic species) in scenarios involving six treatment levels of a nematicide application (including no application) to open-field tomato cultivation in Italy. Because 'food provision' (i.e. tomato production) is a proxy for product efficacy, information related to it would likely be part of the regulatory deliberation in addition to data on the toxicity to non-target organisms. The remaining ES also rely to some extent on the impact to non-target organisms but translate these to a broader suite of outcomes that are readily appreciated by the public. One result of using approaches like those of Alix et al. (2014) and Deacon et al. (2016) is a clearer understanding by society of the benefits associated with the regulatory decision that has been made.

Decisions concerning regulated chemicals require consideration of the trade-offs that will occur by selecting one decision alternative (including not using chemicals) over another. Trade-offs can occur among product potency and efficacy, the economic costs of product application, undesirable environmental effects of product use, field worker and public safety, along with other objectives. For example, one plant protection product might be less expensive to use than another, although its efficacy in controlling crop pests, and therefore its benefits in terms of crop yield, might be lower. Similarly, the two products might have similar efficacies, but could affect non-target organisms differently. Furthermore, application rates of the preferred product (based on product efficacy and minimal effects on non-target organisms) affect both benefits and costs of use of that product. The multidimensionality of trade-off analyses can create challenges because the societal values associated with the various objectives can be difficult to weigh or compare.

ERAs based on ES can help to reduce the dimensionality of comparisons among decision alternatives because these endpoints aggregate societal benefits and costs more explicitly. Further, when changes in ES delivery and societal benefits are appropriately quantified, trade-offs among various decision options can be assessed using economic principles.

With a proper valuation of changes in ES delivery (see Section 4.5), a robust trade-off analysis would be possible that could include: the costs of application; field worker, public and environmental safety; and other objectives valued by society (Deacon et al., 2015; Holt et al., 2016). Monetisation of ES delivery would facilitate the aggregation of all benefits and costs into a single metric for comparison across decision options; economically based non-monetised benefit indicators could serve a similar purpose (e.g. Olander et al., 2018).

3.6. Enhancing the transparency of risk assessment results and decisions based on them

As currently practised, most ERAs and risk management decisions concerning regulated chemicals rely as a first step on standardised toxicity evaluations using a small number of model test species. Typically, toxicity tests measure organism-level responses (survival, reproduction or growth) of test species and extrapolation to the protection goal is achieved by applying safety factors. These conventional endpoints do not always resonate with or have meaning to non-scientists or the public. As a result, the linkages between assessment findings and the things people value are obscured, and the rationale used to select one alternative over another to manage risks can be difficult to communicate (Devos et al., 2015; Munns et al., 2015b, 2017; Lamothe and Sutherland, 2018).

Ecosystem services benefit people and therefore reflect societal values more directly than do conventional ecotoxicological endpoints. When used as assessment endpoints in ERAs informing decisions concerning regulated chemicals (Munns et al., 2015b), ES can clarify linkages between competing decision options or alternatives (e.g. plant protection product application rates) and human well-being. People should therefore better relate to the assessment approaches taken and to the meaning of the risk assessment results. This tighter integration of ecological and societal goals for environmental management enhances the articulation of chemical risks. Gains and losses in ES delivery can form the basis of clear communication of the rationale for decisions made to manage those risks. Because the relative utility of competing decision options can be explained in ways that the public can readily appreciate and care about, decision-making becomes more transparent, understandable, and societally defensible.

4. Challenges for ES-based ERAs

Although an ES approach has potential to enhance the ecological and societal relevance of the ERA of regulated stressors, its application entails challenges. The anthropocentric and utilitarian nature of an ES approach has been criticised. For example, there is concern that if biodiversity only matters to the extent that it benefits humans, then the intrinsic value of nature is ignored (SEP, 2015). Moreover, our understanding of linkages among components and ecological processes that ultimately deliver ES is incomplete, valuing ES is complex, and a lack of generally accepted and comprehensive terminology and definitions hampers consistent communication. Some of the challenges of applying ES in ERAs are addressed below. Several recommendations emerged from the presentations and discussions during the breakout session of EFSA's third Scientific Conference that, if realised, would improve the assessment and management of regulated chemicals in the environment.

4.1. Using an anthropocentric and utilitarian approach

Ecosystems can deliver many goods and functions, but ecosystem functions (which can be defined as the biological, geochemical and physical processes and components that take place or occur within an ecosystem) can only be considered ES when they are associated with human beneficiaries (Fisher et al., 2009). ES are normative and their importance in terms of (desired) presence, level and rate are in the eye of the 'beholder' (e.g. farmers, society) who assigns their own values to them. Thus, society provides the frame of reference for determining the value of ES and changes therein, with environmental decisions being made with these values in mind (Munns and Rea, 2015).

A common criticism of ES-based approaches is their anthropocentric and utilitarian nature (Goulder and Kennedy, 2011; SEP, 2015; Rea and Munns, 2017; Maltby et al., 2018). In contrast to non-utilitarian (e.g. ecocentric) approaches to biodiversity conservation, some perspectives consider that an ES approach ignores the intrinsic value of nature (that is nature's value even if it does not directly or indirectly benefit humans), because biodiversity and ecosystems only matter in an ES approach to the extent that they benefit humans (McCauley, 2006; Reyers et al., 2012; Raymond et al., 2013; Delière and Neuteleers, 2014; Schröter et al., 2014; Silvertown, 2015). The concern is that by maximising only those aspects of ecosystems that provide benefits to humans, biodiversity may not necessarily be protected (Carrasco et al., 2014; SEP, 2015; Pascual et al., 2017b). However, others have argued that biodiversity is better conserved under the umbrella of ES (e.g. Skroch and López-Hoffman, 2009).

The different philosophies underlying intrinsic and instrumental values suggest that there is little common ground between utilitarian and non-utilitarian approaches to biodiversity protection (SEP, 2015; Maltby et al., 2018). However, as discussed by Loreau (2014), both approaches can be reconciled, as the

intrinsic value of nature is included under the category of 'cultural services', which is in line with the NCP framework developed in IPBES (Díaz et al., 2018). In addition, biodiversity may play a role as a regulator of ecosystem processes, or as a final ES (Mace et al., 2012; Laurila-Pant et al., 2015). Thus, an ES approach encompasses the idea of the intrinsic and economic values of biodiversity (Rea and Munns, 2017), incorporating them through the inclusion of cultural and provisioning services (Reyers et al., 2012; Schröter et al., 2014; Laurila-Pant et al., 2015). Hence, intrinsic and economic valuation need not be mutually exclusive; there is room for both perspectives (Costanza et al., 2017). Rea and Munns (2017) put forward the concept of a shared well-being that unites humans and nature through a common future. This places humans and nature on an equal footing and leads to a decision-making framework that explicitly considers both ecocentric and anthropocentric perspectives.

4.2. Linking biodiversity and ecosystem functions and services

Some researchers have argued that relying on an ES approach to protect biodiversity (and thus halt biodiversity loss) is misguided, as our understanding of the biological, geochemical and physical processes and components underpinning ES (including the stability and resilience of ecosystems) is incomplete, and complicated by the spatial and temporal scales over which ES operate and the interdependencies between ecosystem components and functions (Norgaard, 2010; Reyers et al., 2012; Bartkowski et al., 2015; Brock et al., 2018). A linear positive association between biodiversity and delivery of ES is not always manifest. Biodiversity–ES relationships have been found to take varying forms and shapes (e.g. non-linear, mixed) or be altogether non-existent (Munns et al., 2009; Duncan et al., 2015; Truchy et al., 2015). Consequently, forecasting changes in ES delivery due to biodiversity loss remains a complex and challenging task (Cardinale et al., 2012; Larigauderie et al., 2012; Duncan et al., 2015; Gascon et al., 2015).

The relationships between species richness and ecosystem processes need to be understood to predict the potential impacts of a stressor on ES delivery and human well-being (Sandifer et al., 2015). In some cases, this may be straightforward; yet, only in a few cases (e.g. Eisenhauer et al., 2010) is it clear to what extent the observed multifunctionality results from synergies or trade-offs or neutrality between processes (Manning et al., 2018). For example, a positive relationship between ecosystem processes and plant species richness and below- and above-ground organisms (e.g. taxonomic groups such as nematodes, mites or collembola and trophic groups such as herbivores or carnivores) was observed by Weisser et al. (2017). However, in nature, a few abundant species with high biomass often determine process rates. Consequently, a strong positive relationship between species richness and ecosystem processes is not always observed.

To understand the mechanisms behind the effects of an anthropogenic stressor on biodiversity, laboratory (microcosm) and field experiments are necessary and useful, but for realistic ERAs the greater complexity at the landscape scale and longer temporal scale must be addressed. Species can be lost under environmental pressures due to unfavourable response traits. This is reflected in a decrease of diversity and ecosystem multifunctionality at landscape scale, even if diversity at the scale of the sampling spot does not change due to local replacement from the landscape species pool (Mori et al., 2018).

Despite the diverse and complex relationships between biodiversity, ecosystem functions and ES, conclusions can be drawn from the relevant scientific evidence accumulated over the last two decades. For example, Cardinale et al. (2012) reviewed how biodiversity loss influences ecosystem functions, and the impacts that this can have on ES delivery. The authors concluded that 'there is now unequivocal evidence that biodiversity loss reduces the efficiency by which ecological communities capture biologically essential resources, produce biomass, decompose and recycle biologically essential nutrients'. Biodiversity effects seem to be remarkably consistent across different groups of organisms, among trophic levels and across the various ecosystems that have been studied (Cardinale et al., 2012).

Cardinale et al. (2012) also concluded that 'there is mounting evidence that biodiversity increases the stability of ecosystem functions through time'. Stability is likely to be higher if more than one species performs the same function (functional redundancy) because a decline in one species may be compensated for by stable or increasing numbers of another, especially if they respond differently to disturbances and environmental change (Brittain et al., 2013; Loreau and de Manzanourt, 2013; Mori et al., 2013, 2018; Winfree, 2013). Diversity within a species can also have a stabilising effect; high genetic diversity can make species more resilient against stressors and quicker to adapt to environmental change (Munns et al., 2009; SEP, 2015). Biodiversity in the form of life-history traits, such as the age at first reproduction, generation time or the timing of migration, may also be

important factors (SEP, 2015). There is also evidence that the reduction of biodiversity across several trophic levels is likely to have greater effects on ecosystem functioning than biodiversity loss within trophic levels (Munns et al., 2009; Cardinale et al., 2012; Winfree, 2013). For at least some taxa, functional diversity may be a better predictor of ecosystem processes than is species diversity (Munns et al., 2009).

Cardinale et al. (2012) emphasised that the importance of biodiversity for ecosystem functioning is often underestimated due to the short-term nature of studies. Many studies have suggested that the loss of a species has a lesser effect at high levels of biodiversity than at lower levels. However, Reich et al. (2012) demonstrated that, in the short term, the difference in biomass production between plots with medium and high plant diversity is negligible. Yet, this difference increased over time, with productivity becoming significantly higher in plots with high biodiversity. Consequently, the loss of species from even very biodiverse communities could impair ecosystem functioning (SEP, 2015).

While there is a firm evidence base demonstrating the importance of biodiversity on ecosystem functioning, there is less research into whether biodiversity has the same pivotal role for ES. However, evidence suggests that higher biodiversity is needed for simultaneously sustaining multiple ecosystem functions and ES at high levels, a property known as multifunctionality in the long term and under environmental change (Gamfeldt et al., 2008; Cardinale et al., 2012; SEP, 2015).

Despite evidence of the importance of biodiversity for sustained provision of many ES, knowledge gaps remain about the ecological processes that link biodiversity, ecosystem functions and ES, the stability and resilience of ecosystems, as well as the interdependencies of multiple ES and biodiversity (Cardinale et al., 2012; Larigauderie et al., 2012; Duncan et al., 2015). We therefore encourage continued attention by the environmental and ecological research community to help establish such linkages, as it will contribute to a greater mechanistic understanding and predictability of biodiversity–ES relationships (Munns et al., 2009).

4.3. Linking measurement endpoints and ES delivery

Although ES are increasingly gaining attention in the ERA of regulated chemicals, there are wide gaps in knowledge of how the toxicity of regulated chemicals to test species translates into impacts on service-providing populations and on ES delivery. For example, data used to assess the ecological effects of plant protection products are most commonly based on measurements of survival, reproduction and growth of individual organisms in a few model species. Key challenges include extrapolating these organism-level responses to impacts at higher levels of biological organisation, as well as extrapolating responses in tested species to the many untested species that are also in need of protection. Extensive research has shown that responses of individuals, in terms of survival, growth, or reproduction, are not directly proportional to impacts on populations or groups of populations (Forbes et al., 2008; Galic et al., 2010). Thus, it is misleading to assume, as is done implicitly in current ERAs, that the toxicological sensitivity of individuals is a robust proxy for the vulnerability of populations or species to chemicals or other stressors. The reasons for this include the important role of life-history differences among species in determining the relationships between organism-level and population-level responses, as well as the role of key ecological variables (e.g. density dependence, trophic relationships) that modulate how populations respond to various impacts on the individuals of which they are composed (Forbes et al., 2008).

In recent years, there has been much progress in the development and application of ecological effects models that provide quantitative and mechanistic understanding of the relationships between toxicological responses of individual organisms and population-level impacts (Raimondo et al., 2018). However, at least two important challenges remain in order to strengthen the links between what is measured in ERAs and the delivery of ES. The first is that, just as individual toxic responses are not robust proxies of population-level impacts of chemicals, impacts of chemicals on population properties (e.g. population density, population growth rate, size structure) are not necessarily robust proxies for impacts on ES delivery. Ecological effects models can serve as effective tools to link key properties of ecological entities (such as populations) to ES through quantitative ecological production functions. For example, pollination is an ES that may be impacted by plant protection products and thus it is an important consideration for an ERA. In practice, pre-market/prospective ERAs of regulated chemicals compare the individual toxicological responses of model pollinator species with modelled or measured exposure estimates to assess risk. In rare cases, field or semi-field studies may be conducted for higher-tier risk assessments in which impacts of exposure on pollinator population properties might be measured. In neither case are impacts on the ES of pollination explicitly measured or modelled.

Progress is being made with the development of mechanistic models, such as BEEHAVE (Becher et al., 2014; EFSA, 2015b), that link the toxicological and other responses of individual bees to properties of bee colony dynamics. However, what remains to be done is to mechanistically and quantitatively link colony properties through an ecological production function to the service of pollination.

A second challenge in linking effects endpoints to ES delivery in an ERA is related to the growing trend towards using *in vitro* and high-throughput data, in combination with the adverse outcome pathway (AOP) framework (Ankley et al., 2010) for ecological effects assessments. Although the AOP framework provides a useful approach to organising effects data across levels of biological organisation, from molecules to populations, there remains much work to be done to underpin the proposed conceptual relationships with robust quantitative models that include relevant feedback and non-linearities (Forbes and Calow, 2013; Forbes and Galic, 2016). Whereas *in vitro* and high-throughput data can increasingly be generated in large quantities and at low cost, they are even further removed from ES delivery than more traditional organism-level effects endpoints. This added uncertainty remains to be addressed before such data can be used with confidence in the ERA of regulated stressors.

To address both above-mentioned challenges, efforts are underway to develop and test quantitative AOPs and extend these beyond population-level responses, via ecological production functions, to ES delivery. Using a case study approach, Murphy et al. (2018) explored how integrating dynamic energy budget modelling into an AOP framework could improve the predictability of cross-level linkages of stressor impacts from molecular initiating events to organismal (and potentially population) responses. In a parallel initiative, Forbes et al. (2017, 2019a) demonstrated how mechanistic effect models could be used to quantitatively link traditional effects endpoints in ERAs through population- and ecosystem-level responses to impacts on ES delivery. These analyses included an ES valuation step to show how trade-offs between ES could be compared in a way that could inform management decisions. These kinds of case studies are important for establishing proof of concept and for identifying specific areas in need of further research.

In the past, the use of mechanistic effects models in the ERA of regulated stressors was limited by a lack of guidance, a lack of well-tested models, and a lack of case studies to demonstrate the effectiveness of the models to assess risk. Substantial progress has been made on all these fronts in recent years. Detailed guidance now exists on model development, documentation, and evaluation (EFSA, 2014). A growing toolbox of models appropriate for use in the ERA of regulated stressors is available, together with case studies demonstrating their usefulness, not only in assessing risks, but also in informing management options, assessing recovery of impacted populations and exploring potential future scenarios under changing environmental conditions (Galic et al., 2010; Hommen et al., 2015; Forbes et al., 2016). Nevertheless, greater implementation of mechanistic effects models to link effects endpoints in ERAs to ES delivery would be facilitated by a more systematic approach to model development (Schmolke et al., 2017). This would not only improve the efficiency of the model development process, but would increase the consistency, transparency, and hence acceptance, of the generated models. One approach to achieving this, as recommended by Forbes et al. (2019b), could be through a multi-stakeholder collaboration to develop an agreed-upon suite of mechanistic effects models that balance the needs of standardisation and flexibility to appropriately address specific risk assessment questions.

There are concerns that adding new modelling approaches on top of current regulatory requirements would prove burdensome. Ideally, such models – once sufficiently validated and accepted by the risk assessment community – would replace current approaches to ERAs. However, until such a paradigm shift is achieved, it is likely that we will continue to try to assess ecological risks from toxicity tests performed with the same handful of model species. To the extent that effects assessments of regulated stressors continue to be based on toxicological responses of individuals or responses at even lower levels of biological organisation, protection of ES delivery is contingent on being able to confidently link what is measured to the services to be protected. While EFSA has made progress in developing a descriptive, conceptual framework for making these linkages (EFSA, 2010a,b, 2016), this has not measurably changed how ERAs are conducted or how effects data are interpreted in practice. Further progress towards this desirable outcome will require the descriptive framework to be underpinned with robust, mechanistic models that capture the feedback, complexities and non-linearities of ecological systems in a way that is efficient, consistent, and transparent.

4.4. Understanding how regulated stressors affect ES delivery across different spatial and temporal scales

Ecosystem services are delivered across a wide range of spatial and temporal scales, and these do not necessarily match the scales at which regulated stressors impact ES or the ecological production functions contributing to them. For example, when a plant protection product is applied to an agricultural field, pollinators living near to the field margin may be exposed. Typically, the degree of impact on the pollinators is expected to decrease with distance from the field margin. In contrast, the spatial scale at which the service of pollination is delivered will depend on the spatial distribution of relevant populations of pollinators, not just those living near the field margin. Impacts on the delivery of pollination will thus depend on the proportion of the service-providing unit (i.e. pollinators) impacted by the plant protection product and their spatial distribution in the landscape, which may differ from the scale at which exposure occurs. Likewise, temporal variability in exposure may not correspond to the temporal scale at which impacts on ES occur. For example, a runoff or spray drift event may result in a damaging, but brief, pulse of exposure to a nearby water body that dissipates quickly. However, constraints on recovery (e.g. from the timing of life-history events, physical barriers that prevent immigration from unexposed areas) could cause impacts on ES to persist over a much longer temporal scale than the exposure. Clearly, any attempts to integrate impacts on multiple services operating over different spatial and temporal scales will have to deal with these complexities. In this regard, temporally dynamic, spatially explicit effect models (e.g. Wang and Grimm, 2010; Rortais et al., 2017; Streissl et al., 2018) can be very helpful in assessing and visualising such complex scenarios.

4.5. Valuing ES

The outputs of ERAs should communicate and interpret the nature and magnitude (importance) of the risks expected under various decision alternatives to effectively inform decisions concerning regulated stressors. For strictly biophysical protection goals and assessment endpoints, risk can be quantified in terms of expected changes in biodiversity, non-target organism population size and distribution, or other ecological characteristics. The consideration of ES in ERAs, due to the anthropocentric nature of this perspective, brings an additional aspect to risk interpretation – namely, how society benefits or loses from expected changes in ES. A key challenge to applying ES in the ERA of regulated stressors is quantifying how people value changes in ES.

In the context of ES, valuation is the act or process of estimating the worth, merit, or desirability of environmental conditions in common units that can be aggregated and compared (Munns et al., 2015a). Two broad schools of thought exist for quantifying the value of ES: (1) environmental economics; and (2) ecological economics. Environmental economics approaches are based on a unifying framework grounded in the concept of individual utility and social welfare, and individual preferences. Environmental economics approaches consider social welfare to be the objective and assume ecosystems to be part of the economy (US EPA, 2016). An alternative paradigm and set of valuation approaches have been proposed by ecological economists who consider the economy to be one component of a broader environmental system (Adamowicz et al., 2008). This paradigm shifts the focus from humans to ecosystems and defines value in terms of biophysical stocks and flows instead of directly in terms of human welfare (Munns and Rea, 2015; US EPA, 2016). Various methods can be used for deducing value from this perspective, including those based on energy flow, and on comparisons of the ecological footprints required to support individuals and human communities. Despite the attractions that ecological economics offers to the issue of valuation, it has yet to converge on a central set of theories and core framework of analysis needed by environmental policy- and decision-making (US EPA, 2016).

Here, we focus primarily on economic valuation approaches because of their well-established theory and practice. 'Economic' in this context does not necessarily imply 'monetary' or 'monetisation'. According to standard definitions (e.g. that provided by Mansfield and Yohe, 2003), economics 'is concerned with the way in which resources are allocated among alternative uses to satisfy human wants'. Economics as a science is concerned with understanding why people make the choices that they do (US EPA, 2016). These choices need not be quantified in monetary terms, though there are advantages to doing so (Calow, 2015); money is a convenient common unit with which to quantify, aggregate, and compare societal values in the decision-making process. Further, money provides a common unit with which to compare the benefits of management actions with their costs. Still, approaches based on economic theory have been developed that do not monetise the benefits that

people receive from ES (e.g. benefit relevant indicators; Olander et al., 2018) and which can be used to inform environmental decisions.

4.5.1. Economic valuation methods

Table 1 describes the different kinds of benefits that people receive from ES and provides a convenient framework for describing economic valuation methods (as summarised by Munns and Rea, 2015). The value of ES that are bought and sold in markets, such as food and fibre, can be revealed by the money exchanging hands in these markets (though market price might not reflect the full benefits of a good). Revealed preference methods can also be used to quantify the value of certain ES not traded in markets (non-market benefits), such as those that affect market goods directly or indirectly (e.g. for environmental amenities that affect housing prices), or the aesthetic or recreational amenities provided by natural places (which can be inferred, for example, by the amount of time or money people have spent to visit those places).

Table 1: Types of benefits people receive from ecosystem services (Adapted from US EPA (2006) and reproduced from Munns and Rea (2015); © SETAC)

Benefit category		Explanation	Examples
Market		Generally relates to products that can be bought or sold	<ul style="list-style-type: none"> • Food and water sources: Commercial fish and livestock, game fish and wildlife, drinking water • Building materials: Timber • Fuel: Methane, wood • Clothing: Leather, fibres • Medicines: Nature-derived pharmaceuticals
Non-market	Direct use	Directly sought and used or enjoyed by society; includes both consumptive uses and non-consumptive uses	<ul style="list-style-type: none"> • Consumptive recreational: Fishing, hunting • Non-consumptive recreational: Boating, swimming, camping, sunbathing, hiking, bird watching, sightseeing, enjoyment of visual amenities
	Indirect use	Indirectly benefit society; may be valued because they support offsite ecological resources or maintain the biological and/or biochemical processes required for life support	<ul style="list-style-type: none"> • Maintenance of biodiversity • Maintenance and protection of habitat • Pollination of crops and natural vegetation • Dispersal of seeds • Protection of property from floods and storms • Water supply (e.g. groundwater recharge) • Water purification • Pest and pathogen control • Energy and nutrient exchange
	Non-use	Benefit does not depend on current use or indirect benefits; individuals might value the resource without ever intending to use it or might have a sense of environmental stewardship; includes bequest value, existence value, and cultural or historic value	<ul style="list-style-type: none"> • Perpetuation of endangered species • Wilderness areas set aside for future generations

In the absence of information describing values as revealed by people's past and current behaviours, stated preference methods can be used to evaluate the trade-offs people are willing to make to protect ecosystems (Munns and Rea, 2015). These methods depend on asking people how much they would pay to protect or enhance ecosystems and their services (willingness to pay), or to be compensated if such actions were taken (willingness to accept). Information is typically collected using survey instruments that offer choices among various environmental alternatives and the costs associated with each. Stated preference methods are useful for eliciting values of the direct-use, indirect-use, and non-use benefits described in Table 1.

A variety of other approaches, summarised by the SAB (2009) as social–psychological valuation methods, rely on the judgements of individuals and groups to inform environmental decision-making. Included are various methods (multicriteria decision analysis, Delphi methods, referenda, etc.) that seek to elicit the opinions and judgements that can help to uncover societal preferences and to rank the acceptability of alternative options under consideration. Although they do not lend themselves easily (if at all) to monetisation, such approaches provide information about the value people place on ES. Descriptions of a wide range of valuation concepts and methods are provided by Adamowicz et al. (2008) and SAB (2009). The valuation of cultural services is particularly challenging, as described next.

4.5.2. Cultural services

Cultural ES are a category within ES approaches for services that are intangible, subjective and to a large degree non-consumptive (Milcu et al., 2013). While there has been quite an extensive academic debate around the concept and its framing (Small et al., 2017), it is broadly understood as a notion for benefits that humans derive from species, ecosystems and ecosystem functions that depend very much on perceptions. Therefore, they differ between individuals and across sociocultural contexts, and are difficult to generalise and measure.

Cultural ES are commonly divided into three subcategories: Learning and knowledge generation, contributing to learning processes that inspire people and allow them to acquire knowledge and develop skills; Physical and psychological experiences, including opportunities for recreation activities, such as hiking, skiing and berry picking, activities related to species, such as bird watching, as well as mental experiences, such as tranquillity and aesthetic enjoyment; and Supporting identities, referring to a sense of place, cultural heritage, spiritual experiences and also to human well-being emanating from knowledge about the existence of symbolic or iconic species and landscapes independent of their actual use. Maintenance of options, linked to values of inter-generational justice, can be added as a fourth subcategory, though it is also related to supporting and regulating ES (IPBES, 2018).

The values of cultural ES can to some extent be derived from quantitative measures, e.g. through market-oriented valuation, such as expenditure on ecotourism, hunting and fishing permits, and through contingent valuation for services not traded on a market. However, since many of the cultural ES are subjective and need to be contextualised, cultural and social valuation methods are often considered most appropriate. This is because these methods acknowledge the psychological, historical, cultural, social, ecological and political contexts and conditions (the broader social context), as well as underlying worldviews (IPBES, 2016). Cultural and social valuation methods are especially important for estimating relational values, i.e. values related to specific places or aspects of the environment, that reflect cultural identity, social cohesion, social responsibility and moral responsibility towards nature (Chan et al., 2016). Relational values are characteristic of how the way many cultural ES are valued. Some of the most common cultural and social valuation methods are:

- Ethnography: long-term living within a community, participant observation, daily note-taking and the writing of a descriptive monograph;
- Ethnoecological methods: participant observation, interviewing, cultural consensus analysis, cultural domain analysis and social network analysis;
- Geographical methods: participatory geographical information systems and human ecology mapping;
- Narrative valuation: stories, influence diagrams, visual and verbal summaries;
- Preference assessment: respondents are asked to rank or rate preferences in interviews or surveys.

When it comes to ERA, qualitative studies, carried out in social science and humanities research, are needed to value cultural ES, including sociocultural aspects and people's varying perceptions, recognising that different people and communities value differently. Since the values of the other ES will commonly be measured in quantitative terms, methods to integrate and bridge the various methods of valuation are needed (IPBES, 2016).

4.6. Establishing a standard lexicon

As currently practised, the ERA of regulated stressors is largely the purview of ecologists and toxicologists (as informed by protection goals). Full consideration of ES in decisions, however, requires participation of ecologists and economists, as well as the public affected and upon whose values the ES need to be judged to translate assessment findings into meaningful information that can be used to

evaluate trade-offs. Differences among stakeholders in definitions and terminology, and in the way each conceptualises and interprets ES (Munns et al., 2015a; La Notte et al., 2017; Lamothe and Sutherland, 2018), create additional challenges for employing an ES approach for the ERA of regulated stressors. Although the nuances of different ES definitions, interpretations and classifications can be valuable, there is a need to address the existing ambiguity to improve comparability among ES-based approaches. This will promote consistency, transparency and transferability of ERAs.

Munns et al. (2015a) summarised the historical development of ecological and economic concepts of 'ESs'. Largely, the ecological view broadly equates ES with ecosystem structure and processes. Conversely, the economic perspective generally considers ES to be the outputs of ecosystem functions as they provide utility (benefit) to people. In addition to leading to differences between the two groups in how ES are valued, the two perspectives create opportunity for misunderstanding and miscommunication in the syntheses of ERAs to decision makers.

To help overcome the above-mentioned challenges, Munns et al. (2015a) and others (La Notte et al., 2017; Lamothe and Sutherland, 2018) have promoted the development of a common set of definitions and terminology – a 'standard lexicon' – that, if generally accepted, would help to frame risk problems involving ES and interpret results more clearly to enhance the value of an ES approach to environmental decision-making.

5. Practical applicability of ES-based ERAs – Chemicals: Assessment of Risks to Ecosystem Services (CARES)

The CARES project brought together key stakeholders from the chemical industry, regulatory organisations and academia to develop a common understanding of the merits and feasibility of an ES approach to the ERA of chemical stressors and the implications for implementation (Maltby et al., 2018; Faber et al., 2019; Table 2). Clear advantages of using an ES approach were identified, including better informed risk management decisions and more relevant ERAs by focusing protection goals on what stakeholders value. The approach increases transparency both in terms of prioritisation of ES (what to protect and where) and in describing trade-offs – which ES will be enhanced, and which will be reduced by the management decision. The ability of an ES approach to integrate ERAs across multiple stressors, multiple scales and multiple environmental compartments was considered a major advantage, as it offers the potential for a more holistic environmental management. An ES approach highlights the direct and indirect benefits that people get from nature and therefore facilitates discussion on why it is important to protect ecosystems. Because stakeholder values are used to specify protection goals, the approach could improve communication between risk assessors and risk managers, and between scientists, regulators and the public.

Challenges to implementing an ES approach were also identified related to the challenges outlined in Section 4, including the fact that the approach is anthropocentric and utilitarian. Ecosystems have the potential to provide many ES and, although the added complexity of an ES approach increases ecological realism and can result in more targeted testing, it also requires greater ecological understanding, which was perceived as a challenge. The need to develop new tools that either measure ES directly or produce information (i.e. measurement endpoints) that can be robustly extrapolated to ES delivery (i.e. assessment endpoint) was identified. There was agreement that a tiered approach was necessary, and that ES-based ERAs should be based on the magnitude of impact rather than on toxicity exposure thresholds. The research needed to address these challenges was identified and prioritised. The top four priorities were: (1) the development of environmental scenarios (species or trait-based) accounting for chemical exposure and ecological conditions; (2) guidance on the taxa and measurement endpoints relevant to specific ES; (3) improved understanding of the relationships between measurement endpoints from standard toxicity tests and ES of interest (i.e. assessment endpoints); and (4) the development of mechanistic models, which could serve as ecological production functions (Maltby et al., 2017b).

Table 2: Advantages and challenges of applying an ecosystem services framework to pre-market/prospective and retrospective environmental risk assessments identified by workshop participants from business (B), government (G) and academia (A) (Reproduced from Maltby et al. (2018) which is an open access article under the CC BY-NC-ND license (<http://creativecommons.org/licenses/by-nc-nd/4.0/>))

Advantages	Challenges
Relevance: Focus risk assessment on what people want when defining protection goals (B, G, A)	Anthropocentric (B, G, A)
Transparency: Prioritisation and trade-offs made explicit (B, G, A)	Valuation – How to do it (B)
Integration: Integration across multiple stressors, habitats, scales and policies (B, G, A)	Complexity: Data hungry; spatio-temporal variation (B, G, A)
Communication: More effective communication (B, G, A)	Unfamiliar language (G)
Informed risk management decisions: Increased ecological realism; consider implications of different management in multifunctional landscapes; enable cost-benefit analysis of remedial action (B, G)	Cost – Needs resources (time, money) (B, A)
Enable combining ecosystem services with intelligent testing (B)	Tools: Convert conventional ecotoxicity testing to ecosystem services; lack of environmental risk assessment tools (B, G, A)

A decision-making framework that is integrated across the areas of risk assessment and risk management is essential to the successful implementation of an ES-based approach to chemical ERAs. A conceptual framework for future chemical risk assessments that elaborates on earlier conceptualisations is illustrated in Figure 1 and described in detail by Faber et al. (2019). The problem formulation is based on landscapes and ES of concern, defined as ES that are potentially impacted by the chemical of interest. The landscape units (i.e. service-providing areas; Syrbe and Walz, 2012) and ecological components (i.e. service-providing unit, in the meaning of Luck et al., 2003) that provide the ES of concern are then identified. Service-providing units may be species, functional groups of species or ecological processes, depending on the ES (Luck et al., 2009). CARES stakeholders considered the service-providing area and service-providing unit concepts essential for defining spatially defined protection goals and for focusing ERAs.

Boundaries for ERAs are established by identifying the most appropriate ecological and exposure scenarios. Exposure scenarios define the set of environmental conditions that influence chemical exposure and ecological scenarios define the set of ecological conditions that influence species occurrences and biological processes. Together they define the overall environmental scenario (Rico et al., 2016). Environmental scenarios should describe both the environmental characteristics and distribution of service-providing areas and the characteristics and distribution of service-providing units. Because environmental scenarios capture landscape heterogeneity, they enable a more spatially and temporally refined exposure and effects assessment. Exposure scenarios are well established for some regulated products in Europe (e.g. FOCUS scenarios for plant protection products), while ecological and hence environmental scenarios are less well-established and are an area of active research. CARES stakeholders concluded that environmental scenarios for prospective ERAs should be 'as simple as possible, as complex as necessary' and should initially focus on areas with the highest potential exposure (Maltby et al., 2017b).

Exposures and effects should be assessed against the most relevant environmental scenarios and any effects established using ES-relevant endpoints, scaled up to assess the impact on ES and associated ES trade-offs. A chemical's mode of action will influence which ES are most vulnerable and hence prioritised in ERAs. The assessment of ES can be further prioritised by considering the relative importance of the service-providing area for delivering specific ES of concern and initially focusing on vulnerable ES for which the relative importance of the service-providing area is high (Maltby et al., 2017b). A tiered approach to assessing the risk of chemicals to ES would start with a few generic worst-case (exposure) scenarios and would use the results of standard toxicity tests. Higher tiers could include the development of more refined environmental scenarios, additional tests that are more relevant to the service-providing units delivering the specific ES of interest and the mode of action of the chemical and mechanistic modelling to link measurement endpoints to ES (e.g. ecological production function).

Because landscapes provide multiple, non-independent ES, it is important that risk assessments provide risk managers with different options that not only consider the potential for effect and recovery, but also consider interactions between ES and possible effects on non-focal ES (i.e. ES trade-offs). The final step in the conceptual framework is post-decision monitoring of ES to validate ERAs and mitigation interventions and to evaluate their effectiveness in protecting the ES of interest.

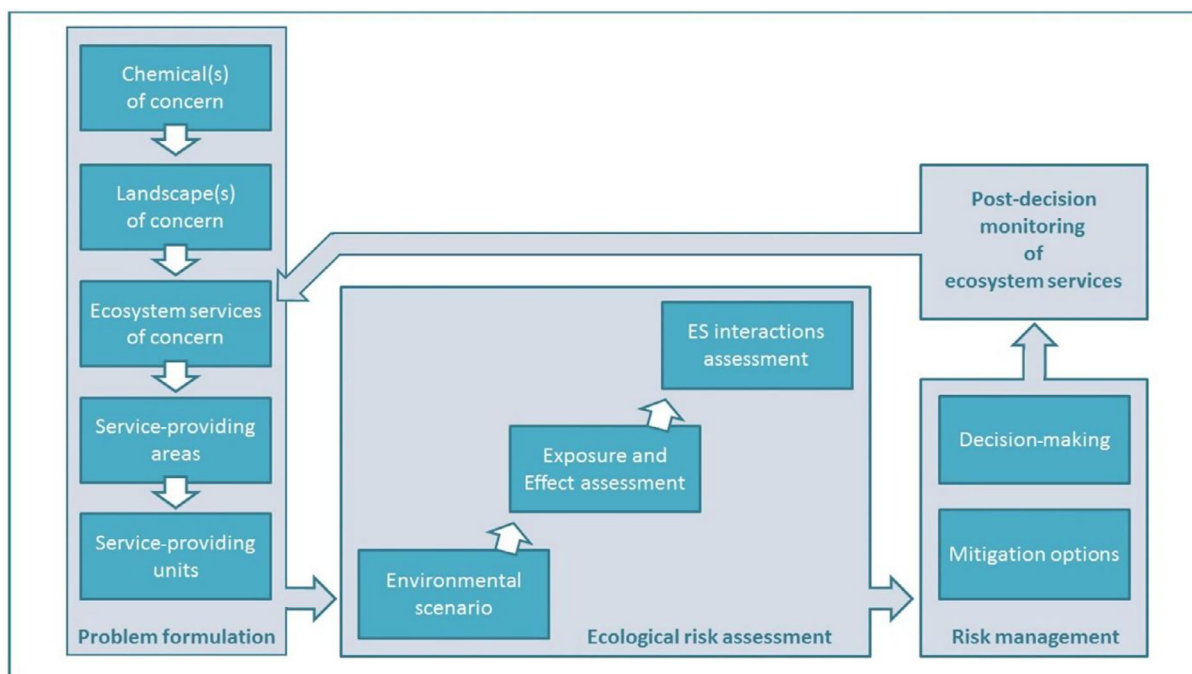


Figure 1: Conceptual framework for the environmental risk assessment of chemicals and decision-making based on an ecosystem services approach (Reproduced from Faber et al. (2019) which is an open access article under the CC BY-NC-ND license (<http://creativecommons.org/licenses/by-nc-nd/4.0/>))

6. Conclusions

An ES approach is presumed to provide a better basis for decision-making because of the explicit connection between human well-being and ecosystem structure and processes (Nienstedt et al., 2012; Munns et al., 2017). However, this presumption has not been tested robustly (Van Wensem et al., 2017). While progress has been made towards developing specific approaches for including ES in pre-market/prospective ERAs of regulated stressors and decision-making, actual applications remain relatively scarce. Moreover, ES-based ERAs will likely require greater amounts of data, modelling and basic ecological understanding than do current approaches, which may impede their acceptance by some stakeholders. Consequently, many of the assertions about the advantages of applying an ES approach to the ERA of regulated stressors remain largely unverified. Realisation of these benefits will require procedural constructs and guidance for including ES-based assessment endpoints in ERAs in a variety of regulatory contexts. In addition, improvements are needed in the ways that ES relevant to decisions are identified and quantified, and in how institutions adapt ES in policies and protection goals. Broadly based efforts to engage society (including an improved dialogue between risk assessors, risk managers and the public) about an ES approach and its value to environmental and public health protection can help in these regards. We specifically encourage: (1) further research to establish biodiversity–ES relationships; (2) the development of approaches that (i) quantitatively translate responses to chemical stressors by organisms and groups of organisms, via ecological production functions, to ES delivery across different spatial and temporal scales, (ii) measure cultural ES and ease their integration into ES valuations, and (iii) appropriately value changes in ES delivery so that trade-offs among different management options can be assessed; (3) the establishment of a standard ES lexicon; and (4) building capacity in ES science and how to apply ES to ERAs. Since an ES approach fosters a broader systems perspective of sustainability, our sense is that it will make an important

contribution to making science and ERAs more responsive to the needs of decision-makers. The issues and challenges we and others have identified will need to be overcome eventually, but the advantages we perceive of using this approach render it more than worthwhile to tackle those challenges. Society and the environment stand to benefit from this shift in how we conduct the ERA of regulated stressors.

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Abbreviations

AOP	adverse outcome pathway
CARES	Chemicals: Assessment of Risks to Ecosystem Services
CICES	Common International Classification of Ecosystem Services
ERA	environmental risk assessment
ES	Ecosystem services
FECS-CS	Final Ecosystem Goods and Services Classification System
IPBES	Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services
MEA	Millennium Ecosystem Assessment
NCP	Nature's Contributions to People
NESCS	National Ecosystem Services Classification System
REACH	Registration, Evaluation, Authorisation and Restriction of Chemicals
TEEB	the Economics of Ecosystems and Biodiversity
US EPA	US Environmental Protection Agency