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Priorities and opportunities in the application of the ecosystem services concept in risk assessment for chemicals in the environment

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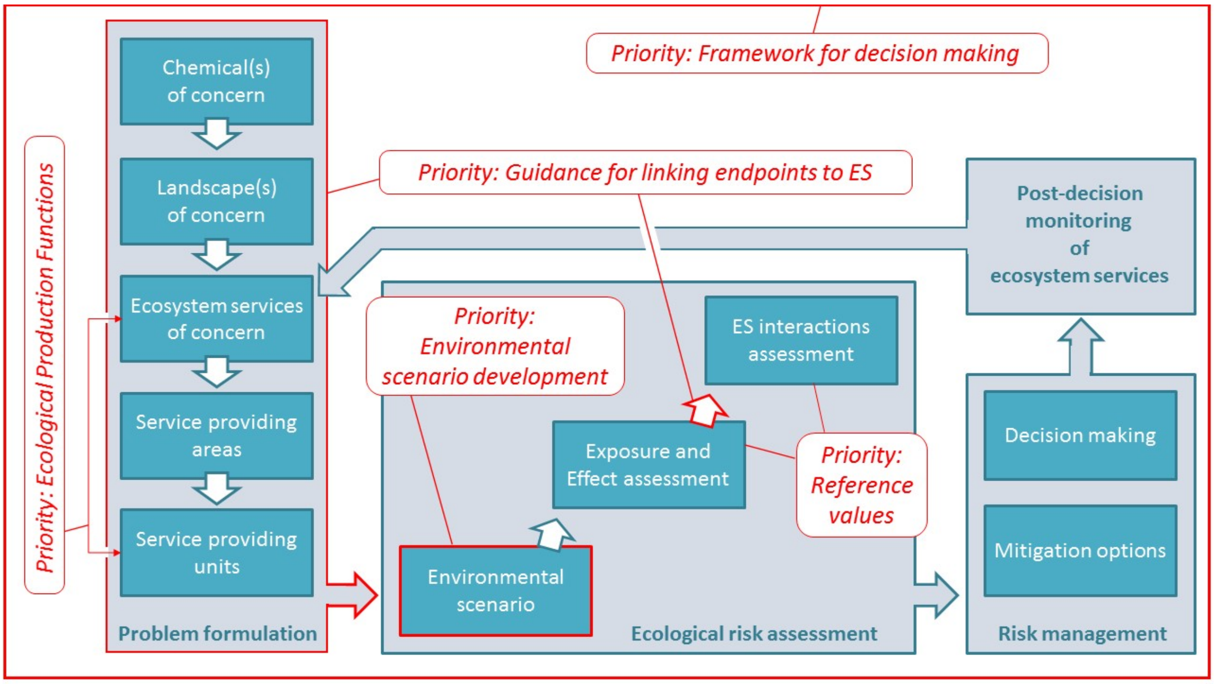
ABSTRACT

The ecosystem services approach has gained broad interest in regulatory and policy circles for use in ecological risk assessment. Whilst identifying several challenges, scientific experts from European regulatory authorities, the chemical industry and academia considered the approach applicable to all chemical sectors and potentially contributing to greater ecological relevance for setting and assessing environmental protection goals compared to current European regulatory frameworks for chemicals. These challenges were addressed in workshops to develop a common understanding across stakeholders on how the ecosystem services concept might be used in chemical risk assessment and what would need to be done to implement it. This paper describes the consensus outcome of those discussions. Knowledge gaps and research needs were identified and prioritised, exploring the use of novel approaches from ecology, ecotoxicology and ecological modelling. Where applicable, distinction is made between prospective and retrospective ecological risk assessment. For prospective risk assessment the development of environmental scenarios accounting for chemical exposure and ecological conditions was designated as a top priority. For retrospective risk assessment the top priority research need was development of reference conditions for key ecosystem services and guidance for their derivation. Both prospective and retrospective risk assessment would benefit from guidance on the taxa and measurement endpoints relevant to specific ecosystem services and from improved understanding of the relationships between measurement endpoints from standard toxicity tests and the ecosystem services of interest (i.e. assessment endpoints). The development of mechanistic models, which could serve as ecological production functions, was identified as a priority. A conceptual framework for future chemical risk assessment based on an ecosystem services approach is presented.

GRAPHICAL ABSTRACT



HIGHLIGHTS

1. The ecosystem services (ES) approach has potential to enhance ecological and societal relevance in ERA.
2. Stakeholders in EU regulation, industry, academia and NGOs agreed on priority research needs.
3. A framework for future chemical risk assessment based on an ES approach is presented.
4. Further development may benefit from recent progress in other disciplines.

Keywords: prospective risk assessment, retrospective risk assessment, landscape-scale risk assessment, research needs

1. Introduction

Ecosystem services are the direct and indirect contributions that ecosystems, and the biodiversity they support, make to human well-being (TEEB, 2010). 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. The concept of ecosystem services (ES) has gained broad interest in regulatory and policy groups for use in landscape management and risk assessment (Maltby 2013). It is presumed to provide a better basis for decision making because of the explicit connection between human well-being and ecosystem structures and processes (Nienstedt et al. 2012; Ågerstrand and Staveley 2015), although this presumption has not been tested robustly (van Wensem et al. 2017). In chemical ecological risk assessment (ERA), the European Food Safety Authority (EFSA) has taken the lead in exploring the use of an ES approach for setting specific protection goals for pesticides (EFSA 2010, 2016) and the framework developed by EFSA has been shown to be potentially applicable to other chemical sectors (Maltby et al. 2017a).

There are several advantages of using an ES approach for ecological risk assessment (ERA) of chemicals. These advantages include: increased relevance by focussing protection goals on what stakeholders value; increased transparency, both in terms of the prioritization of ES and in describing trade-offs between them; increased integration of the risk assessment across multiple stressors, multiple scales and multiple environmental compartments; more effective communication by highlighting the direct and indirect benefits people get from nature and facilitating discussion on why it is important to protect ecosystems (Maltby et al. 2017b). However, there are still a number of scientific and technical challenges to overcome before it can be implemented effectively. Previously, we reported on research gaps and development needs as the outcome of a multi-stakeholder workshop between the major European chemical companies, policy makers, regulatory authorities and academics (CARES workshop). Key research needs that were identified include approaches to address heterogeneity in

ES delivery across landscapes; tools and test methods to assess ES-relevant endpoints; ecological production functions (EPFs) that link measurement endpoints to changes in ES delivery; tools and approaches for assessing ES trade-offs (Maltby et al. 2017b). The current paper expands on this work by presenting and discussing the outcome of two consecutive workshops where research gaps were prioritised and elaborated in consensus.

Several of the development needs identified by the first CARES workshop are not specific to the ERA of chemicals. Understanding landscape heterogeneity and its consequences for spatio-temporal variation in species distributions, functional traits and hence ES delivery, are key areas of research in landscape ecology and conservation biology (Tscharntke et al. 2012; Stein et al. 2014). The challenges of how to assess ES, the development of EPFs and the assessment of ES trade-offs are all areas of active research within the ecological, ecotoxicological and ecological modelling communities (de Groot et al. 2010; Harrison 2010; UNEP-WCMC 2011; Crossman et al. 2012; Haines-Young et al. 2012; Maes et al. 2013; Bruins 2017). The ES research literature has increased substantially over the last decade and covers a wide range of disciplines (McDonough et al. 2017). There are therefore opportunities to draw on these research developments to address the challenges of implementing and ES-based approach to chemical ERA.

1.4. Objective of this paper

Regulatory risk assessment of chemicals is an interaction between regulatory agencies and chemical industries that is underpinned by scientific research and understanding, much of which occurs in universities. To address scientific challenges and improve regulatory practice, it is important to bring these different communities together to agree research priorities and share knowledge and perspectives. Here we discuss the outputs of two further multi-stakeholder workshops that elaborated on the development needs as described earlier in Maltby et al. (2017b). The aims of these workshops were to: (1) reach consensus on the prioritisation of research needed to enable the implementation of an ES-based approach to chemical risk assessment; (2) evaluate opportunities for employing recent advances in ecology, ecotoxicology and ecological modelling to address the prioritized research needs. In this paper, we communicate the resulting consensus on research priorities and identify opportunities to capitalise on ideas and approaches from a range of areas of expertise to address them. We have focussed on the ecological aspects of linking ecotoxicological endpoints to ES assessment, and did not proceed to a next level of the economic aspects of valuing damage and costs of risk management measures. We use the workshop recommendations to develop a new comprehensive framework for ERA on the basis of using the ES approach. As such, this paper is a compilation of various discussions addressing different steps in ERA where research gaps were identified for. In addressing these, the narrative follows the virtual workflow in ERA through the consecutive steps of problem definition, risk assessment and risk management. But first, we briefly describe how a workshop approach was followed to identify and elaborate the research priorities.

2. Methods

Two 2-day multi-stakeholder workshops were organised under the auspices of the Society of Environmental Toxicology and Chemistry (SETAC) Europe. These workshops took place in May 2016 and November 2016 and were a follow-up on an initial workshop (May 2015) that discussed and evaluated the challenges associated with implementing an ES approach to chemical ERA (Maltby et al. 2017b). Workshops participants included 39 scientific experts from European and national regulatory authorities (31%), chemical industry (39%) and academia and non-governmental organisations (30%) and discussions took place in multi-sector breakout groups that focussed on either retrospective or prospective ERA.

One of the key challenges of implementing an ES approach to chemical ERA, is the lack of tools and approaches to assess the impact of chemicals on ES provision that take account of landscape heterogeneity in land use and ES provision and trade-offs (Maltby et al. 2017b). Workshop participants were therefore asked to consider: the suitability of current standardized approaches for assessing impacts on ES provision; the use of indicators to assess bundles of ES; the availability of mapping techniques and data for developing environmental scenarios; trade-offs between ES; upscaling of effects across biological, spatial and temporal scales. These discussions were used to highlight key knowledge gaps and identify research needs.

Research needs were discussed and collated in plenary after collecting individual participants' suggestions in smaller break-out sessions addressing different case studies (see below). Research needs were ranked based on participant voting, and separate rankings were generated for prospective and retrospective ERA. The top four research needs for retrospective ERA and the top four research needs for prospective ERA were prioritised for further discussion in a final workshop. The final workshop focussed on the opportunities provided by novel ecological, ecotoxicological and modelling approaches that can address the priority research needs.

Workshop break-out group discussions were facilitated by using case studies. The retrospective ERA case study explored how an ES-based approach might be used to inform a site-specific ERA for contaminated land. The case referred to an existing tiered ERA showing how risk assessment endpoints had been derived based on locally desired ES for a large scale contamination in a rural polder area ('Krimpenerwaard') in The Netherlands (Faber 2006). The prospective ERA case study explored how an ES approach might be used to inform an ERA for chemicals released in a river stretch. A hypothetical mixed-use catchment was considered in which exposure of aquatic habitats could occur via sewage treatment discharges, urban runoff, emissions from agricultural practises. The receiving habitats were highly varied in terms of typology and scale, potentially providing a wide range of ES. Food web information was based on Lombardo et al. (2015).

3. Prioritisation of research needs

Workshop participants identified several limitations in capability that constrained our ability to implement an ES-based approach to chemicals ERA. Limitations were identified for each of the three consecutive stages in the riskassessment process: problem formulation, risk assessment, risk management. A total of 11 research needs to address these limitations were identified, mostly associated with the risk assessment phase itself, but also linked to the initial phase of problem formulation or the later phase of risk management, or the entire ERA process (Table 1).

These prioritised research needs are presented in Table 1. Three topics were ranked in the top four for both prospective and retrospective ERA: (1) to develop mechanistic models, including EPFs, which link changes in ecosystem structure and processes to ES provision; (2) develop guidance to link measurement endpoints for environmental receptors to ES; (3) develop a framework for decision making for risk assessors and risk managers. For prospective ERA 81% of the workshop participants identified the development of commonly agreed environmental scenarios as the most urgent research need. However, this was considered much less relevant for retrospective ERA, where the specific study site is usually well-defined in terms of land use, exposure routes and ecological communities. Rather, for retrospective ERA the development of reference values or normal operating ranges (*sensu* Kowalchuk et al. 2003) for key indicators for service-providing species was prioritised, in order to be able to discriminate contaminant effects beyond ‘natural’ status or potential range of natural variation, respectively.

Table 1 here

4. Opportunities for an ES-based approach to ERA

The following sections address the prioritised research needs and evaluate opportunities for employing recent advances in ecology, ecotoxicology and ecological modelling. The discussion follows the consecutive steps in the ERA process; starting with problem formulation (section 4.1) and then considering how the boundaries for the ERA can be determined using environmental scenarios where appropriate (section 4.2). Next follows a section on the determination of data needs to assess potential impact on ES and the associated measurement endpoints. We discuss the need for guidance on selection of taxa and measurement endpoints relevant to ecosystem services (section 4.3). Section 4.4

addresses how to link measurement endpoints to ES using mechanistic models such as EPFs, and how EPFs should link between standard tests and final ES assessment. Next, section 4.5 briefly discusses the need for references in the assessment of ES impairment in comparison to conditions without chemical impact. We conclude by synthesising the whole process into an assessment framework that may guide an ES approach in ERA (section 4.6).

4.4. Problem formulation

The first step in the problem formulation for an ES-based ERA is to identify the contaminant(s) of concern, the landscapes potentially exposed and the ES of concern (Maltby et al. 2017a). The ES of concern are those that are potentially affected by chemical exposure. Ecosystem functions (*sensu* de Groot et al. 2002) only become ES when they are valued and demanded by beneficiaries. Therefore, stakeholder participation is an important element in ES identification and hence in the entire ERA process that follows. Once potentially exposed landscapes and ES of concern have been identified, the spatial units producing those ES are determined. These spatial units were termed service production areas by Fisher et al. (2009) and service-providing areas by Syrbe and Walz (2012). Service-providing areas can provide the basis for assessing and mapping a wide range of landscape classification units that may include aspects of both land use stakeholders as well as wildlife populations (Porter et al. 2009; Burkhard et al. 2012; Syrbe and Walz 2012). Service-providing units (SPUs, *sensu* Luck et al. 2003) are the ecological components important in delivering the ES within the service-providing areas. SPUs have a qualitative dimension, i.e. particular species or functional group(s) of species, or processes, as well as a quantitative dimension, i.e. what density, abundance or process rate is required to provide the service at the level required (by the stakeholder) (Luck et al. 2009; Kontogianni et al. 2010). Workshop participants considered the service-providing area and SPU concepts essential for addressing spatially defined protection goals, and for understanding the complex spatial and temporal dynamics of ES (Rieb et al. 2017). What to protect, and where, can be based on empirical analysis of landscape function or service provision, and landscape properties can be used in a spatial approach for indicator selection and quantification (de Groot et al. 2010). Factual knowledge of the location and amount of service supply (e.g. biodiversity observations, crop yield, level of aesthetics, etc.) is then linked to variables describing spatial landscape properties (e.g. Alessa et al. 2008; Willemen et al. 2008). Once SPUs have been determined the ERA can be scoped, the necessary assessment data generated and linked to the desired specific protection goals and ES, as discussed in the following sections. Crucial in the linking of SPUs to ES assessment is the availability of mechanistic models (e.g. EPFs), which are addressed in section 4.5.

4.5. Scenario development

Having established a problem definition, boundaries need to be determined for the ERA by narrowing down to the most realistic scenarios for exposure and ecological context. The term 'scenarios' may have different meanings, and can represent existing, historical, future, hypothetical, or typical or average situations, across different spatial scales (Alcamo and Henrichs 2008). Essentially, within the context of chemical ERA, scenarios define a set of environmental conditions that influence chemical exposure (exposure scenario) and ecological conditions that influence species occurrences and biological processes (ecological scenario). The combination of the exposure and ecological scenario is the overall environmental scenario (EFSA 2014; Rico et al. 2016; Franco et al. 2017). Scenarios take the heterogeneity of the landscape into account and enable, if needed, a more refined spatial and temporal exposure and effects assessment. To focus the ERA towards ES assessment, an environmental scenario should contain a description of the environmental characteristics of the service-providing areas (e.g. agricultural fields) and their distribution in the landscape, as well as a description of the identity and distribution of species present in the landscape and their traits. An assessment may then be made of ES that can be provided by the particular landscape, but may be affected by chemical exposure.

4.2.1. Assessment scale

The development of environmental scenarios for chemical ERA is in its infancy. For pesticide ERA, surface water exposure scenarios were developed almost two decades ago to account for spatial heterogeneity in European edge-of-field water bodies (FOCUS 2001). However, these exposure scenarios lack an ecological component, so cannot be used to link exposure with effects using an integrated modelling framework. Ecological scenarios are less well established within chemical ERA, but describe the range of species or traits potentially present in a given geographical context. An ecological scenario is defined by spatial and temporal scales, but what are the appropriate scales? ES are delivered at local, regional, global or multiple scales. For example, pest control operates at a local scale, forest albedo effects on climate operate at regional scales and carbon sequestration effects on climate operate at global scales (Kremen 2005). Species mediating ES may also operate across a range of scales; from wide-ranging mobile birds and mammals, to relatively immobile soil invertebrates, microbes and plants (Ekroos et al. 2016). In addition, metapopulation source-sink dynamics may result in chemical impacts in one location having effects on populations (and hence potentially ES delivery) at unexposed locations connected by the movement of individuals or propagules (i.e. action at a distance, Spromberg et al. 1998). The potential influence of 'action at a distance' on both the impact of, and recovery from, chemical exposures (Topping et al. 2014) led to the suggested inclusion of landscape-scale risk assessment for plant protection products (EFSA 2015). Workshop participants agreed that the scale of a scenario should be relevant to the ES of interest. They proposed that the scenario scale could be determined by the "home range" of the species or communities making up the SPUs, although they also noted that this can be a challenge given the huge differences in home range for some SPUs. They also proposed that the spatial scale should be sufficient to sustain the minimum population size of key species or functional groups required to provide an ES at the desired level.

It was concluded that, in general, the prospective environmental scenario should be 'as simple as possible, as complex as necessary'. When a scenario-based approach is adopted, the areas with the highest exposure should be identified and taken as a starting point for the scenario development (Maltby et al. 2017a). For example, for many chemicals in consumer and household products that are disposed to sewers (‘down the drain chemicals’) this will be the outlet of the waste water treatment plant, while for pesticides, drainage ditches or small streams may be the initial focal scenarios. For the down the drain example, one could start with a river basin, including all the habitats and typologies it runs through. If the initial assessment shows no or acceptable risk to the most exposed habitat then there is no need to go to next level.

4.2.2. Resilience and recovery

A chemical’s toxic mode of action will influence which ES are most vulnerable and hence prioritised. Vulnerability is a function of exposure, sensitivity and recovery potential (Ippolito et al. 2010; de Lange et al. 2009; van Straalen 1994). There is therefore a need to include sensitivity and recoverability analysis of ES into scenario development, focussing on potentially affected ES and the habitats and SPUs that provide them (e.g. de Lange et al. 2010, Rico and Van den Brink 2015). Vulnerability analyses that incorporate exposure, sensitivity and recovery, can be used to identify species, spatio-temporal scale and key habitat drivers for developing and populating ecological models used to assess impact (Chen et al. 2013). If, for instance, recovery is of interest, the spatial scale should be adjusted to the dispersal range of the SPU of interest. Large-scale scenarios may be most appropriate when it is possible to perform the assessment holistically, including multiple stressors, multiple land uses, etc. Small-scale scenarios may assess the effects of single chemical use on ES within a given land-use (e.g. agricultural field), while intermediate scale scenarios may evaluate risks of multiple chemicals within a given land use (e.g. at the farm-scale). Workshop participants identified an urgent need to establish environmental scenarios that are able to link ecological models to exposure models and thereby embed them into ERA (De Laender et al. 2015).

4.2.3. ES trade-offs

Ecosystems have the potential to provide multiple ES, but ES do not vary independently; they form positively (synergies) and negatively (trade-offs) interacting bundles (i.e. sets of ES that repeatedly appear together across space or time) (Raudsepp-Hearne et al. 2010). Therefore, managing ecosystems to increase the delivery of some ES may decrease the delivery of others (Smith et al. 2017) and the covariation between services may vary spatially (Emmett et al. 2016). For instance, soil tillage affects both plant growth and soil structure, the outcome being strongly related to soil type, and therefore promoting yields by increasing tillage intensity may lead to erosion and water logging (Morris et al. 2010). Workshop participants recommended that larger scale scenarios can be used to identify ES bundles and potentially conflicting protection goals. Large-scale scenarios should ideally consider all relevant ES and include ES trade-offs, i.e. one ecosystem service responding to factors resulting in a change in another (MEA 2005). Smaller scale scenarios are more likely to focus on a limited number of ES.

The outcomes of multiple ES assessments and their potential trade-offs can be communicated effectively using ‘flower’, ‘radar’, or ‘cobweb’ diagrams (e.g. Deacon et al. 2016; Mouchet et al. 2017; Williams and Hedlund 2014).

4.2.4. Tiered approach

Workshop participants considered how a tiered scenario approach could be linked to the current tiers of an ERA. The first tier could start with a few generic worst-case (exposure) scenarios and use the results of standard toxicity tests as an initial effect assessment. An initial first tier assessment should enable further work to be targeted on areas identified with the highest risks based on the initial scenario. Existing typologies (e.g. EFSA 2010; Van der Zanden et al. 2016) could be used as a starting point to develop more refined scenarios. Whether or not an ES should be prioritised or if all ES should be included in the risk assessment depends on the protection goals set by risk managers. For the refined ERA more tests may be required, which are more relevant to the SPUs delivering the specific ES of interest and the mode of action of the chemical.

4.2.5. Site-specific ERA

In site-specific ERA the environmental scenario follows from case-specific local circumstances, and will therefore be developed using specific, rather than generic, information. The comprehensiveness of local scenarios will depend on the availability of environmental data such as regional land use, desired ES, habitat type and characteristics, contaminants and other stressors in the defined area. Scenarios should represent the heterogeneity of habitats in the area of interest. A potentially useful typology for European agricultural landscapes is described in Van der Zanden et al. (2016), and the European Nature Information System (EUNIS) habitat classification provides a hierarchical typology for marine, freshwater and terrestrial habitats (Davies and Moss 1998; Davies et al. 2004). It is important that ES are defined for each site in consultation with stakeholders. For example, in the Krimpenerwaard case study (Faber 2006), an iterative stakeholder process was used to develop three scenarios and identify indicators that were relevant for the desired land use objectives and susceptible to the contaminants of concern. Such scenario definition as part of ERA has been protocolled under the Dutch standard NEN5737 (NEN 2010), and was recently published as an international standard (ISO 2017). When constructing a retrospective ERA scenario, not all potential ES from the range of habitats need to be included. Focus should be on the ES prioritised by the stakeholders in interaction with regulators and scientists. Limiting factors e.g. adjacent sites (mosaic situation, dependency) and budget restrictions for risk assessment and management should be taken into account. The level of resolution needed for scenario development depends on a number of factors including the specific conditions of the site, the specific protection goals as identified by the stakeholders and the ES of concern.

4.3. Reference values for ES

Workshop participants prioritised the need to develop reference values for ES (Table 1). The assessment of ES impairment requires comparison to a benchmark or reference value and hence knowledge of the level of ES provision under control or unimpacted conditions, as well as normal operating ranges (*sensu* Kowalchuk et al. 2003) for key ES indicators. There is considerable focus on the development of ES indicators and their use for mapping ES delivery and determining ES reference values (e.g. Faber et al. 2013; Maes et al. 2014, 2016; Zulian et al. 2017). Recent work in this area includes the EU FP7 OpenNESS project (Smith et al. 2016) and the ongoing Working Group on Mapping and Assessment on Ecosystems and their Services (MAES), set up under the Common Implementation Framework to underpin the effective delivery of the EU Biodiversity Strategy to 2020 (Maes et al. 2014). Using CICES v4.3 as the baseline classification (CICES 2013), the MAES working group has produced an EU-wide matrix of ES, which was populated from a literature review and from assessing data and indicators available in the European data centres (European Commission 2014). Associated to MAES are mapping activities of ES and natural capital by individual EU member states. OpenNESS and the MAES approach have focussed on the development of methodologies for natural capital accounting, which includes mapping and assessing the state of ecosystems and their services by individual Member States, assessment of the economic value of such services, and integration of these values into accounting and reporting systems at EU and national level by 2020. Standardisation of ES indicators has therefore gone a relatively long way already, and it seems that in a near future, data will become available that may be used for setting ES reference values.

At a lower level of assessment, reference values are needed for ecological endpoints, especially in retrospective ERA. ERA for aquatic environments has seen more progress than the terrestrial counterpart. For example, the biological quality of rivers within the United Kingdom can be assessed using the RIVPACS (River InVertebrate Prediction And Classification System) reference database software package (Wright 2000), that offers site-specific predictions of the macroinvertebrate fauna to be expected in the absence of major environmental stress, using a small suite of environmental characteristics. The biological evaluation is then obtained by comparing the fauna observed at the site with the expected fauna. This could be developed as a bottom-up approach to deriving expected reference conditions for ES. Recent studies have explored how ES map on to the EU Water Framework Directive objectives (Vlachopoulou et al, 2014), how WFD indicators may provide information on ES (Vidal-Abarca et al, 2016) and how ES approaches inform WFD river basin management plans (Grizzetti et al, 2016). A recent study has concluded that achieving WFD water quality goals may not enhance recreational ES (Ziv et al, 2016) suggesting that an ES approach may provide added value.

4.4. Guidance on taxa and measurement endpoints relevant to ecosystem services

Well defined specific protection goals are required to determine the type and range of measurable endpoints needed to facilitate an ES-based ERA. EFSA has recently developed guidance on the derivation of specific protection goals, following three sequential steps: (1) the identification of relevant ES; (2) the identification of SPUs for these ES; and (3) the specification of options for parameters for and the level of protection of the SPUs (EFSA 2016). As proposed for plant protection products, specific protection goals are defined along several dimensions: ecological entity and attribute to protect, and the magnitude, temporal scale and spatial scale of the biologically relevant effects (impacting a specific protection goal). In addition, the level of tolerable change and the degree of certainty that the specified effect level will not be exceeded are defined (Nienstedt et al. 2012). Workshop participants considered EFSA guidance (EFSA 2010, 2016) to be suitably detailed, depending on the level of effect that can be accepted. To derive a suitable specific protection goal, all relevant SPUs need to be considered, addressing all relevant final ES –provisioning, regulating, or cultural-, although a prioritisation step may be required to ensure that the assessment is focused and pragmatic.

Standardised tests generally refer to individual species, do not measure community structure, and rarely measure ecosystem function (Maltby et al. 2017b). In addition, the development of complementary tests or additional measurement and assessment endpoints are required in the following areas:

- Redundancy, resilience and tipping points
- Indirect effects
- Ecological recovery rate and extent
- Cumulative effects, chemical mixture effects, multi-stressor effects
- Wider scale effects, including climate effects.

The large tool box of standardized tests is mostly related to biophysical structure and processes and to intermediate rather than final ES, e.g. enabling assessment of impacts on species or community structure and on selected, largely microbial-driven, functions (Maltby et al. 2017b). However, protection goals are likely to be described in terms of final ES. Guidance on when to use single or multiple tests and how to interpret the data (e.g. via a weight of evidence approach) needs to be developed. Such methods will need to enable assessment of functional endpoints in laboratory or semi-field tests, as well as assess resilience or recovery under (semi-)field conditions.

Selck et al. (2017) recommended an explicit division of protection goals into two levels: 1) universal protection goals (e.g., global assessment endpoints such as maintaining ecosystem services); and 2) workable, site-specific, region-specific, or problem-specific protection goals (i.e., site-specific, region-specific, or problem-specific assessment endpoints such as the specific ecosystem service of adequate water flow), where translation between the two levels is integrated (Linkov et al. 2014) and facilitated by input from risk assessors, risk managers, and communities of interest. Assessing specific protection goals may require tailor-made assessment endpoints of direct ecological relevance so that subsequent translation into ES assessment is straightforward. However, such endpoints often need development *de novo* and thus lack standardisation. They may be more costly and technically difficult to estimate than conventional (standardized) endpoints, and know-how and background data for comparison tends to be lacking. Hence, a trade-off exists between the use of tailor-made assessment endpoints and standardized tests, where the latter may be more difficult to link to specific protection goals and required ES. It seems that the solution to this dilemma must involve the development of relationships that enable standard tests to be linked to the necessary broad range of ecological structural and functional endpoints needed to assess specific protection goals.

A plethora of new tests may not necessarily need to be developed if it is possible to develop models or relationships that provide quantifiable links, but a shift in focus is definitely needed. Functional tests may sometimes, but not always, be considered more relevant for the assessment of provisioning and regulating ES than structural tests, since mechanistic models link test measurements to ES based on functional or ecological processes. However, for cultural services such as angling, hunting, bird watching, and ecotourism for flora and fauna, structural endpoints may be more relevant where the presence and abundance, size or weight of particular species is the focus. To interpret structural endpoints more broadly, knowledge of structure-function relationships is needed. Semi-field tests may provide functional endpoints for ES assessment, but need validation to address the uncertainty in extrapolating to the field.

For retrospective ERA, linking measurement endpoints obtained in the laboratory or field to ES may be more straightforward and can aim to assess ES provision in situ on the basis of local data for specific and most relevant endpoints. Comparisons of field data, where prior understanding of impacts is available, helps identify endpoints associated with ES provision. For example, spatial and temporal mapping of chemical contamination can be compared to ES provision in exposed areas, and benchmarked against areas elsewhere, as shown in the Krimpenerwaard case study (Faber 2006). Biomonitoring data can be used to compare observed with expected species presence or abundance, but we should beware of confounding factors and compounding stress factors like excess nutrients or physical disturbance. Ecological models can also be used but the right level of complexity should be assessed as there may be a lack of mechanistic understanding of the relevant ecological processes.

4.5. Linking measurement endpoints to ecosystem services using mechanistic models

4.5.1. Population and foodweb modelling

Most standard toxicity tests measure effects on individual-level attributes (growth, survival, reproduction) in single species set-ups, or microbial-driven processes, but ES are driven by the abundance and functioning of populations and species assemblages (Maltby et al. 2017b). There is therefore a need to develop approaches for relating effects measured in standard tests (i.e. measurement endpoints) to potential effects on ES delivery. Mechanistic effects models, which include energy budget models, population models and food web models, provide one approach (Forbes and Galic 2016). Energy budgets and population models have been widely used in ecological studies to extrapolate changes in individual performance to effects on population structure and dynamics (Grimm and Railsback 2013; Nisbet et al. 2012). The modelling of species interactions and food webs is well developed (Rossberg 2013) and spatially-explicit ecological models have been developed that capture landscape heterogeneity and spatially-dependent biological processes (DeAngelis and Yurek 2017). The potential application of these modelling approaches to ERA was identified a number of years ago (e.g. Maltby et al. 2001; Pastorok et al. 2002) and although some of the models have been applied in ecotoxicological studies (Galic et al. 2010), their use in regulatory ERA has been extremely limited. There has been a concerted effort to develop mechanistic effect models that predict population-level effects from standard toxicity studies (e.g. Gabsi et al. 2014; Martin et al. 2013), but much less attention has been paid to developing mechanistic effect models that capture species interactions and the functioning of species assemblages (Lombardo et al. 2015; Park et al. 2008).

4.5.2. Ecological Production Functions

One of the major challenges in implementing an ES-based ERA is the limited understanding of how changes in the attributes of ecosystems influence their capacity to deliver ES (Maseyk et al. 2017). EPFs relate changes in the biophysical structure and ecological processes of ecosystems to changes in the ecological outputs (cf. ecosystem function *sensu* de Groot et al. 2002) that drive ES delivery (Munns et al. 2015). EPFs can therefore be used to characterise the relationships between ecosystem condition, management practices and ES delivery (Heal 2000, Naidoo and Ricketts 2006). In some cases, EPFs may describe simple statistical associations between measurement endpoints (e.g. SPU structure or function) and ES provision, and in other cases EPFs will have a more mechanistic basis (Bruins et al. 2017). Although our understanding of the relationship between land use, biodiversity and service provision is limited (Nicholson et al. 2009), some patterns are emerging. For example, a recent systematic review of 13 ES produced a typology of links between ES and natural capital (Smith et al. 2017). The five pathways identified were: amount of vegetation (related to air, soil and water regulation); provision of supporting habitat (related to pollination, pest regulation); presence of particular species, functional groups or traits (related to provisioning ES, species-based cultural services); biological and physical diversity (related to landscape-based cultural services); abiotic factors (related to water supply).

4.5.3. Do standard test species relate to EPFs?

EPFs can be made generic for application in a prospective tiered assessment scheme for some ES (e.g. pollination, natural enemies), but this may be more difficult for other services. It may not be easy to link specific species from standard tests to drivers for certain EPFs. The same species may be a key species for an EPF in one ecosystem but not in another, or of varying seasonal influence. Valid indicators for EPFs are needed to utilise the species that are already tested. Models need to be developed that allow extrapolation of the measurement endpoints of standard test species to characteristics of species (traits) that drive the EPF. An EPF is a function of species and their traits, especially effect traits or functional traits, which permit a quantitative assessment of the species' density or biomass affecting ecosystem processes (Lavorel and Garnier 2002). Also, diversity amongst functional traits is a driver for ecosystem functioning (Heemsbergen et al. 2004). Therefore, establishing traits is important for understanding the relationship between species and ES provision. Knowing species vulnerability, i.e. as defined by a series of ecological traits, can help to improve our understanding of what can happen to ES provision in different scenarios.

4.5.4. Do species-based EPFs relate to final ES?

EPFs or quantitative models incorporating EPFs are needed to perform ES-based ERA. Some conceptual or simple EPFs have been developed, e.g. for pollination (Blaauw and Isaacs 2014, Garratt et al. 2014), biological pest control (Jonsson et al. 2014; Östman et al. 2003), nitrogen cycling (Compton et al. 2011), carbon sequestration and water regulation (Tallis et al. 2011). The US EPA's EcoService Models Library is an online database of ecological models that may be used to quantify ES (www.epa.gov/eco-research/ecoservice-models-library). This is a very useful resource, however, the lack of validation is limiting the predictive capacity of EPFs and key services remain to be modelled and integrated into multi-service frameworks (Jonsson et al. 2014). Moreover, some EPFs relate to ecological processes or supporting services (e.g. nutrient retention, soil fertility) and therefore need to be translated into final services. Existing EPFs generally do not incorporate chemical dose-response relationships, and this omission must be addressed if EPFs are to be used in the ERA of chemicals.

4.5.5. EPFs in prospective and retrospective ERA

For prospective ERA, risk to ES or the ecological functions on which they depend, will be based primarily on effect data from standard toxicity tests, as discussed in section 4.4. Uncertainty in ERA will increase with the upscaling of effect data along the levels of biological organisation (i.e. up to populations and communities) and along spatial-temporal scales (e.g. to landscape and watershed scales and towards long-term time frames). The spatial scale of ES delivery and spatial co-occurrence of delivery and use varies between ES. An appropriate scale must therefore be chosen for model development, and this should be included in the ecological scenarios (section 4.2). For retrospective ERA, generic EPFs may be appropriate when assessing ES with high functional redundancy (e.g. ES driven by microbial processes) or where the ES is associated with a small group of species (e.g. water infiltration in soils associated with anecic earthworms) (Spurgeon et al. 2013). For other ES, it may be necessary to compare effects on ES indicators to regional or national reference values (section 4.3).

4.6. Development of an integrated decision making framework for risk assessors and managers

Whilst several research needs have been identified (Table 1), workshop participants agreed that this should not prevent movement towards implementation of an ES approach in ERA and risk management, as there are benefits that could be accrued now (Maltby et al. 2017b). However, they also agreed that a decision making framework that integrated across risk assessment and risk management was essential to the successful implementation of an ES-based approach to chemical ERA.

Elaborating on earlier conceptualisations (Faber and Van Wensem 2012; Munns et al. 2016; Paetzold et al. 2010) we developed a conceptual framework for chemical ERA (Figure 1). Essential to a focussed and effective ERA, the problem to be assessed needs to be defined *a priori*. The problem formulation (section 4.1) is based on landscapes and ES of concern, which determine relevant service-providing areas and SPUs that the risk assessment can be focussed on in terms of ecological and exposure scenarios. Exposures and effects can then be assessed against the most relevant environmental scenario (section 4.2), and any established effects using ES relevant endpoints (section 4.4) and ES reference values (section 4.5) are subsequently scaled up to assess impact on ES (section 4.5) and associated ES trade-offs. Because landscapes provide multiple, non-independent ES, workshop participants considered it important that risk assessments provide risk managers with different options that not only consider the potential for effect as well as recovery, but also consider interactions between ES and possible effects on non-focal ES. Undesirable trade-offs may exist between chemical risk mitigation or remediation and provisioning ES, as e.g. in plant protection products and crop yield in conventional intensive agriculture. Biodiversity and conservation values may not benefit –on a short term- from contaminated land clean-up sanitation. Whilst key ES remain to be modelled and integrated into multi-ES frameworks, explicit consideration and accounting of effects on multiple ES can potentially provide decision-makers with an integrated view of chemicals sources, damages and abatement costs.

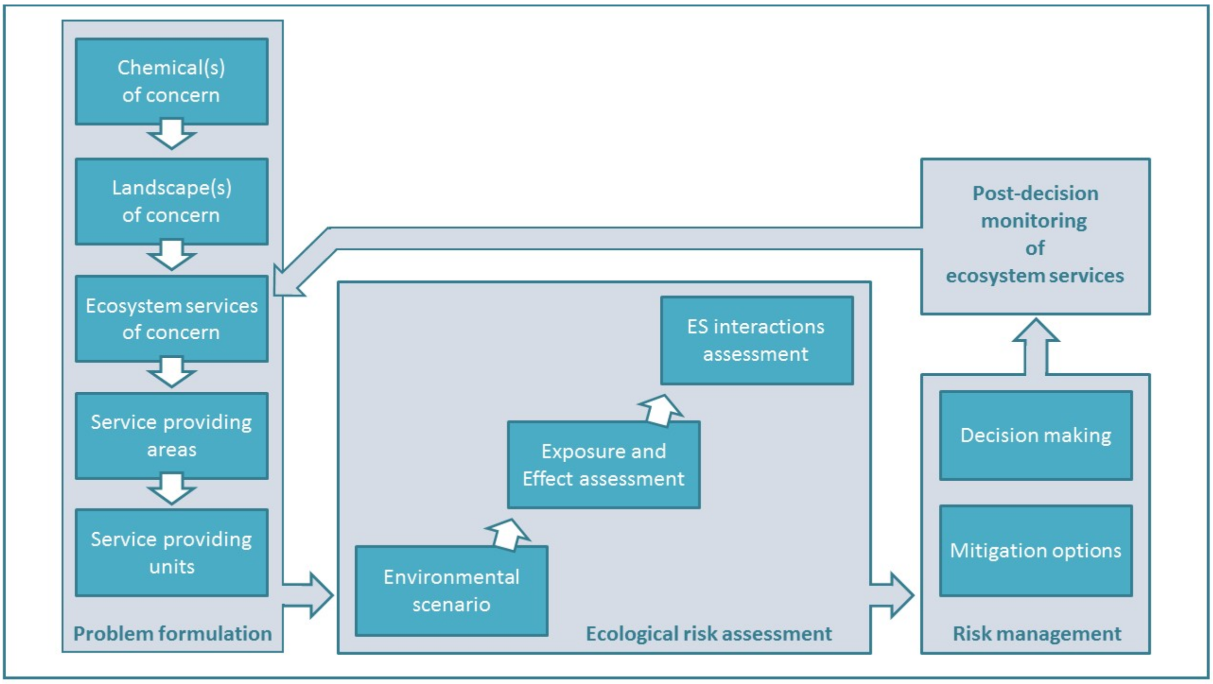


Figure 1. Conceptual framework for future chemical risk assessment and decision making based on an ecosystem services approach.

Armed with information on ES effects, recovery potential and ES interactions, risk managers can evaluate the environmental and economic consequences of the different ERA options, consider potential measures for mitigating risk and make their decision. There is a variety of tools available to support the integration of ES into decision making, but only few studies clearly address a specific policy context (Grêt-Regamey et al. 2017). ES are most frequently addressed in policy sectors with a long tradition in the management of natural resources, such as agriculture, water and forestry, but also conservation and spatial planning. Recently developed ES tools aim at providing information for multiple policy sectors, supporting the implementation of ES tools in spatial planning (Grêt-Regamey et al. 2017). The final step in the framework is post-decision monitoring of ES. Workshop participants considered it important to monitor ES of interest post-decision to validate the ERA and mitigation interventions and to evaluate their effectiveness in protecting the ES of interest.

A future implementation roadmap for ERA would benefit from the development of a set of illustrative case examples that demonstrate the ES approach in both a prospective and retrospective ERA. These case studies should include a typology of the ecosystem of interest, e.g. the typology of waters used by the Water Framework Directive (European Commission 2000) or a typology of land use (e.g. Van der Zanden et al. 2016). This could be followed by the development of an overarching checklist of ES that are required for different land uses leading to a set of environmental scenarios that reflect different land uses.

5. In conclusion

We stated that current regulatory endpoints do not cover (most) ES, and therefore there is a need to develop guidance on what data to use and how to aggregate these for populations and landscapes at relevant spatiotemporal scales, as well as how to develop mechanistic models for extrapolation to ES. The development and implementation of such guidance is a new approach in ERA. As the aim of employing an ES approach in ERA and risk management is to facilitate decision making, the approach should help to reduce uncertainty, increase transparency, enable trade-offs between ES to be assessed, including the benefits and disadvantages of chemicals, and enable illustration of risk management options. The CARES workshops concluded that the ES approach is applicable to all chemical sectors and may contribute to greater ecological relevance for setting and assessing environmental protection goals compared to current European regulatory frameworks for chemicals. To this extent, the prioritisation and evaluation of opportunities to fill in major gaps may help to advance current ERA, and the conception of an ERA framework on the basis of an ES approach may roadmap some guidance.

Workshop participants considered that the approach may become quite complex, e.g. when attempting to breakdown and define ES provisioning, and in relation to environmental complexity in landscapes. In recognition of several research gaps, it was recommended to conduct a proof of concept study to elaborate notions in semi-realistic case studies in both prospective and retrospective settings in a stakeholder participatory approach.

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Table 1. Research needs for adopting an ecosystem services (ES) approach in prospective and retrospective ERA, expressed as percentage of Workshop 2 participant votes. Top 4 commonly identified research needs are marked in bold text and shaded cells.

Phase in ERA	Research need	Prospective ERA ranking (%)	Retrospective ERA ranking (%)
Problem formulation	Linking measurement endpoints to ES using mechanistic models Models such as ecological production functions can be used to link structural or functional endpoints of single or aggregations of species to provision of ES (i.e. service providing units (SPUs), *sensu* Luck et al. 2003). These models are needed because it will not be feasible to directly measure most ES endpoints, and therefore will serve as well to extrapolate effects in the risk assessment stage.	**2 (57%)**	**1 (57%)**
	Landscape mapping of ES Geo-referenced ecological, landscape and exposure data can be used to facilitate spatially referenced ERA, enabling environmental heterogeneity to be addressed. Geo-spatial mapping data are likely to be a key requirement for scenario development and, where sufficiently resolved, be of direct relevance to site-specific retrospective ERA.	6 (14%)	5 (14%)

Risk assessment	Development of, and agreement on, environmental scenarios Generalisation and "standardisation" of spatially resolved ecological and exposure scenarios (environmental scenarios) to assess or predict exposure and effects for ERA. These scenarios are needed to reduce environmental heterogeneity to a practical range of representative conditions for ERA.	**1 (81%)**	8 (10%)
	Guidance on taxa and measurement endpoints relevant to ES Guidance is needed to extend capability to link measured endpoints of current regulatory endpoints to ES. This may include extending the range of both structural and functional endpoints. This is needed because it will not be feasible to directly measure most ES endpoints.	**3 (33%)**	**2 (48%)**
	Calibration of a tiered approach and evaluation of conventional tests The tiered approach should be logically consistent (e.g. moving from conservative lower tier to more refined and predictive higher tier) and cost and resource efficient. Where feasible, extend use of standard tests using mechanistic models for extrapolation.	5 (24%)	11 (0%)

	Reference values for key ES Reference values are needed to provide quantification of representative ranges of ES across different environmental typologies. They also aid in discriminating contaminant effects from the likely natural variation within an 'unimpacted' ecological status, particularly in retrospective ERA.	10 (5%)	**3 (43%)**
	Measurement and prediction of ES resilience Assessment of ES sensitivity to, and recovery from, chemical exposure will be a key aspect for risk assessment and risk management.	11 (5%)	5 (14%)
	High-aggregation level modelling of populations and landscapes Modelling is needed to extend the use of EPFs for assessing ecological impacts on SPUs and associated ES on a relevant spatiotemporal scale. This is a key aspect of linking measurement endpoints to ES.	8 (10%)	10 (5%)

Risk management	Risk assessors to offer options to risk managers Risk assessors should indicate the range of potential impacts on ES depending on influences of different stressors and specific protection goals to the risk managers. Indicating potential trade-offs between benefits from chemical use and different ES within a defined landscape, whilst also considering interventions in other influences on ES provision, will aid decision making by risk managers.	9 (10%)	8 (10%)
Entire ERA process	Framework for decision making for risk assessors and risk managers A framework needs to include a consideration of ES interactions (synergies and trade-offs) as well as spatially defined protection goals and implications for landscape-scale risk assessment and risk management (e.g. multiple stressors). A framework helps to achieve consistency and transparency.	**4 (29%)**	**4 (38%)**
	Illustrative case studies Case studies can help to explain the ES-based approach and to demonstrate differences in methodologies and outcomes with current regulatory frameworks.	7 (14%)	5 (14%)