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Identifying ecological production functions for use in ecosystem services-based environmental risk assessment of chemicals

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HIGHLIGHTS

- Ecosystem services (ES) approach can improve utility of ecological risk assessment.
- We reviewed quantitative relationships linking standardised test endpoints to ES.
- ES best covered: pollination, pest control, nutrient regulation, decomposition.
- ES assessment relevant standardised tests identified at wider taxonomic resolution.
- Ecological production functions may be used for defining specific protection goals.

GRAPHICAL ABSTRACT
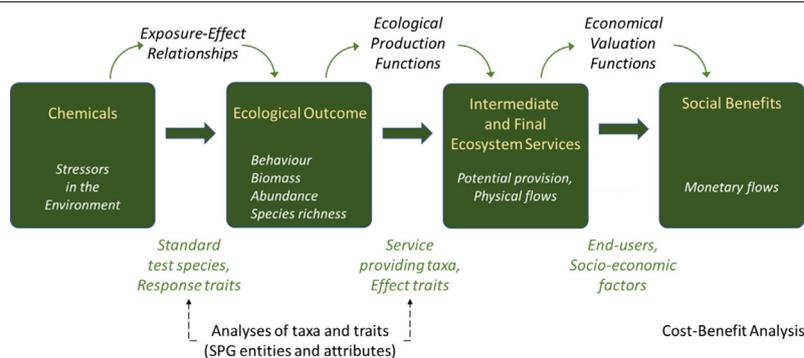
ARTICLE INFO

ABSTRACT

There is increasing research interest in the application of the ecosystem services (ES) concept in the environmental risk assessment of chemicals to support formulating and operationalising regulatory environmental protection goals and making environmental risk assessment more policy- and value-relevant. This requires connecting ecosystem structure and processes to ecosystem function and henceforth to provision of ecosystem goods and services and their economic valuation. Ecological production functions (EPFs) may help to quantify these connections in a transparent manner and to predict ES provision based on function-related descriptors for service providing species, communities, ecosystems or habitats. We review scientific literature for EPFs to evaluate availability across provisioning and regulation and maintenance services (CICES v5.1 classification). We found quantitative production functions for nearly all ES, often complemented with economic valuation of physical or monetary flows. We studied the service providing units in these EPFs to evaluate the potential for extrapolation of toxicity data for test species obtained from standardised testing to ES provision. A broad taxonomic representation of service providers was established, but quantitative models directly linking standard test species to ES provision were extremely scarce. A pragmatic way to deal with this data gap would be the use of proxies for related taxa and stepwise functional extrapolation to ES provision and valuation, which we conclude possible for most ES. We suggest that EPFs may be used in defining specific protection goals (SPGs), and illustrate, using pollination as an example, the availability of information for the ecological entity and attribute dimensions of SPGs. Twenty-five pollination EPFs were compiled from the literature for biological entities ranging from 'colony' to

'habitat', with 75% referring to 'functional group'. With about equal representation of the attributes 'function', 'abundance' and 'diversity', SPGs for pollination therefore would seem best substantiated by EPFs at the level of functional group.

Contents

1. Introduction

There is increasing research interest in how the concept of ecosystem services (ES) can be applied in the context of environmental risk assessment (ERA) of chemicals. Formulating regulatory environmental protection goals in terms of ES can be helpful in making general protection goals more clearly defined from a scientific perspective and hence more operational (Devos et al., 2015), while concurrently recognising that delivery of all services cannot be maximized at the same place and time (Forbes and Calow, 2013). ES assessment can also contribute to risk management by connecting ecosystem structure and process to what is valued, and analysing risk in this context is a way of making risk assessment more policy- and value-relevant (Forbes and Calow, 2013).

In a recent workshop, organised under the auspices of the Society of Environmental Toxicology and Chemistry Europe and supported by the European Chemical Industry Council, scientific experts from European regulatory authorities, the chemical industry and academia discussed case studies where an ES approach to both prospective and retrospective ERA of chemicals was implemented. The participants agreed that ES can inform prioritisation of risk, and aid risk communication and risk management in a clear and transparent way. The ES approach provides a 'integrative approach' in which chemical impacts in water and on land can be evaluated concurrently, enabling risk managers to evaluate trade-offs and synergies (Maltby et al., 2018; Faber et al., 2019).

Another driver for the incorporation of the ES approach in the chemical ERA is the desire to link up to ongoing developments in decision making for biodiversity and conservation management and policy. This desire has grown, since the publication of the Millennium Ecosystem Assessment (MA, 2005) and The Economics for Ecosystems and Biodiversity (TEEB, 2010), which have promoted the valuation of natural capital and ecosystem services in environmental policy making and planning in the EU and UK (EC, 2011; HM Treasury, 2018; Dasgupta, 2021). In ERA there is now a clear requirement to develop tools and approaches that identify what needs to be protected, where and when, and to link this to the sustainable use and management of natural capital and ecosystem services (Holt et al., 2016).

Conceptual approaches to promote the integration of ES in ecological and environmental assessment have been proposed for some time (Faber and van Wensem, 2012; Maltby, 2013; Geneletti, 2015; Maltby et al., 2017). Challenges to moving forward have previously been identified and prioritised (Maltby et al., 2018; Faber et al., 2019), and pilot studies and evaluations are being undertaken (Rosa and Sánchez, 2015, Brown et al. subm., Maltby et al. subm., Van den Brink et al. subm.).

An ES-based approach has been used by the European Food Safety Authority (EFSA) as a framework for the formulation of specific protection goals for the ERA of pesticides (Nienstedt et al., 2012; EFSA Scientific Comm, 2016a) and has been demonstrated to be potentially applicable to other chemicals (Maltby et al., 2017). In line with US-

EPA (2003, 2006) strategy, Munns and co-workers proposed that ERA would benefit by focusing ecological assessment endpoints on the ES that correspond most directly to restoration and damage compensation decisions, and they reemphasized development of generic ecosystem service assessment endpoints for application in hazardous site investigations (Munns Jr et al., 2009).

Retrospective ERA, as in site-specific ERA and environmental evaluations up-to-regional-scale, may perhaps be most readily associated with ES assessment through the use of endpoints that are directly or closely related to services required or desired by stakeholders in the chemical-contaminated area. For site-specific ERA of contaminated land, standardised approaches have been developed that aim to establish these stakeholder objectives and selection of relevant endpoints (NSI, 2010; ISO, 2017). In environmental monitoring, the EU Water Framework Directive (WFD) commits Member States to produce river basin management plans (EC, 2000), where locally defined 'objectives' often explicitly or implicitly reflect the delivery of a suite of ES desired by local stakeholders (Grizzetti et al., 2016), such as water discharge and storage, natural attenuation, natural conservation, and climate mitigation. The standard WFD biological and hydromorphological indices reflecting 'good ecological status', however, are applicable in ES assessment only to a limited extent (Vidal-Abarca et al., 2016), and the development of guidance for applying an ES approach in support of the implementation of the WFD has been recommended for some time (Martini et al., 2013; Blackstock et al., 2015). Practical implementation has been demonstrated using observed and expected taxonomic lists for WFD biological quality elements, in conjunction with generic data on functional traits relating taxa to ES delivery. However this approach is limited to quantifying potential, rather than actual (site-specific), ES delivery (Brown et al. subm.).

There are considerable challenges in developing the framework and tools required for an ES-based risk assessment of chemicals stressors. Whilst schemes have been developed to describe and evaluate ES, methods for assessing environmental impacts on ES are less developed. The effects of chemicals on biological entities are largely based on ecotoxicological data derived from simple, standardised, mostly, single species toxicity tests developed, for example, by OECD, ISO, ASTM. In principle, ES-based ERA could be done without using standard test species, but it makes sense to build on regulatory accepted tests and any existing data generated using them. These tests generally refer to individual species and do not measure population or community structure, and rarely measure ecosystem function (Maltby et al., 2018). Tests on assemblages of species populations are available, but have not been applied to the vast majority of chemicals emitted into the environment. Whilst the magnitude of provision of some ES can be derived from population-level biomass or abundance measures, many others represent functional processes that are not directly assessed in the standard toxicity tests conventionally used in chemical risk assessment.

One approach to overcoming the mismatch between the predominant types of conventional toxicity test endpoints and ES endpoints is to develop quantitative tools and models that enable extrapolation from what we generally measure to ES entities and their attributes. Ecological production functions, EPFs, quantify the contribution of service providing units (SPUs) to ES delivery; they describe direct relationships between characteristics of the SPU, e.g. abundance, biomass, and function and ES provisioning (NRC, 2005; Daily and Matson, 2008; Tallis and Polasky, 2009). An SPU is the collection of individuals from a given species and their characteristics necessary to deliver an ES at the desired level (Luck et al., 2009). This definition may be broadened to include more than a single species when referring to functions performed by a functional group of species with the relevant biological traits. Indeed, the use of EPFs in ERA has been called for repeatedly (Bruins et al., 2017; Maltby et al., 2017; Maltby et al., 2018). Although many studies have modelled effects on species that are important for ES delivery or for intermediate services, quantitative links to final ES are considered to be rare (Forbes and Galic, 2016).

It has been pointed out for some time that the linkage between ERA effects assessment endpoints and ES needs improvement (Munns Jr et al., 2009) and the science evaluating the connection between specific drivers and specific services is limited (Carpenter et al., 2009; Norgaard, 2010). For ES-based ERA existing work on stressor-driven changes in ecosystem structure and function need to be connected with ES provision of a landscape. Overarching requirements to do so and to develop and establish an effective accounting framework will require EPFs that quantitatively connect ecological processes to a complete range of ES and human benefits as well as ES valuation functions, that defensibly attach value to the damage costs per unit stressor and the costs of abatement, restoration or replacement (Compton et al., 2011).

The objective of this literature review was to compile quantitative information on ES provision by ecological receptors susceptible to environmental stressors, in particular chemicals. We focus on EPFs and other functional relationships that can be used to quantify the provisioning of an ES in response to associated service producing ecological entities. We address all provisioning and regulating and maintenance services, and discuss the use of cultural services in ERA. We synthesise the results of systematic and non-systematic literature reviews from ecological and ecotoxicological angles, and discuss future challenges and research needs and priorities for the development of EPFs for use in ERA. By compiling this information and providing a descriptive (but not comprehensive) synthesis we hope to ease and facilitate the application of the ES concept in ERA for setting specific protection goals, and in environmental decision making. To this extent we focus in detail on pollination as an example to illustrate how the available EPF literature may be used to assess pollination services at various levels of biological integration, extending from the individual pollinator to the landscape and habitat levels, each with appropriate attributes as quantified in the production functions, which may then be applied in a definition of specific protection goals. Thus, the paper is targeted at an audience of risk assessors and risk managers and policy makers dealing with environmental risks of chemicals, and may also serve in the operationalisation and target development for assessment of natural capital and sustainable development goals (Hák et al., 2016; Bebbington and Unerman, 2018).

2. Materials and methods

The literature review follows a combination of systematic and non-systematic approaches, comprising three searches. A first search identified studies that devised or used models incorporating species biomass, abundance, feeding rate, or other functional endpoints measured in prospective ERA. A second search identified studies where ecosystem services were either measured directly in the field (e.g. crop yield of insect pollinated fruits), or by using indicators of provision (e.g. density of pollinator insects), as usually applied for the purpose of retrospective ERA and environmental evaluations. A third search focussed on the environmental economics literature and identified studies that included a monetary valuation of those ecosystem services for which EPFs were obtained. As no studies were found that cover the whole range from standard ecotoxicological test to ES provision and economic valuation, we compiled studies that quantify part of the upscaling trajectory and would thus require limited extrapolation, or perhaps may be used in combination to cover the entire range.

2.1. ES classification

For the classification of ES we followed the Common International Classification of Ecosystem Services (CICES version 5.1) which is widely used for mapping, ecosystem assessment, and natural capital ecosystem accounting in Europe and beyond (Czúcz et al., 2018; Haines-Young and Potschin, 2018). The work in the EU on 'Mapping and Assessment of Ecosystems and their Services' for example, uses CICES as the framework for developing ecosystem service indicators (Maes et al., 2015). Besides, we consider CICES to provide maximum universal applicability

since the system has established broad equivalences to other classifications of ecosystem services, including the Millennium Ecosystem Assessment (MA, 2005), The Economics of Ecosystems and Biodiversity study (TEEB, 2010), the US-EPA Final Ecosystem Goods and Services Classification System (Landers and Nahlik, 2013), and the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) (Díaz et al., 2015; IPBES, 2017). Following common usage, CICES recognises the main categories of ecosystem outputs to be provisioning, regulating and cultural services. However, it does not cover the so-called 'supporting services' originally defined in the MA, as these are treated as part of the underlying structures, process and functions that characterise ecosystems. CICES is structured hierarchically, using a five-level hierarchical structure: Section (e.g. Provisioning), Division (e.g. Biomass), Group (e.g. Cultivated terrestrial plants for nutrition, materials or energy), Class (e.g. Cultivated terrestrial plants (including fungi, algae) grown for nutritional purposes), and Class type (e.g. Cereals; the ecological contribution to the growth of cultivated, land-based crops that can be harvested and used as raw material for the production of food). Each level is progressively more detailed and specific. The first four levels can be used for ecosystem accounting, for example, without reducing the utility of the classification for different users, such as those concerned with mapping who may need more detailed categories. We have focussed on the two most highly resolved levels, i.e. 'Class' and 'Class type', as only these are sufficiently specific to be able to be linked to ecotoxicological data. Thus, we have addressed a total of 45 biotic ES Class types, covering 24 provisioning and 22 regulation and maintenance services. Cultural services were covered by example only. In synthesizing the literature, we have aimed to provide the most relevant publications on each class of ecosystem service and for different service providing taxonomic groups, where relevant, but we did not aim for exhaustive and comprehensive collation at any taxonomical level.

ES driven by abiotic factors (CICES Sections 4, 5 and 6) were outside the scope of our study. Neither did we focus on ecological production functions for ES provision in response to abiotic stressors (e.g. for nitrogen as reviewed by Compton et al., 2011). We have included some of these EPFs however, when control levels of ES provision were presented across taxonomic diversity and geographical ranges where biomass or biological activities may vary implicitly (e.g. Thomas et al., 2010).

2.2. Search and selection of EPFs

We have searched for EPFs that describe the provision of (preferably final) ES, focussing on the CICES 5.1 divisions 'Provisioning services' and 'Regulation and maintenance services'. We searched for EPFs that are ready for use in the context of chemical ERA, requiring no change to the way that impacts on the effect end-points are currently measured, e.g. individual-level effects on survival, growth and reproduction, or population-level effects in terms of abundance and biomass, or community effects in terms of species composition or richness. These measures are an indication of a change in ecological endpoints and the service delivery from SPUs. Given this, we consider those EPFs appropriate for use in chemical ERA that are driven by such endpoints. For example, if the ecological endpoint is the abundance or feeding rate of some species, then the abundance and feeding activity of that species has to be central to the EPF, although that might only be one of the SPU's attributes that is included in the EPF model (Fig. 1). If an EPF connects a change in extent of land use/habitat to ecosystem service delivery, rather than considering individual- or population-level endpoints, or it does not indicate a change in quality of the habitat, then it is of limited use in the context of ERA for chemicals.

Ideal studies would be those that address the whole pathway between standard test endpoints and final ES delivery (cf. CICES), but studies that only addressed the link from SPUs to ES were also included. In screening EPFs for suitability to be applied in ERA, a total of nine attributes can be used as utility criteria (Bruins et al., 2017). Whilst we recognize the relevance of these "desirable attributes", we have not applied all during the screening as we anticipated a limited search outcome. For our study we have focussed the literature searches on EPFs that quantitatively estimate final ES outcomes, and that potentially respond to ecosystem condition and chemical stressor levels. Other attributes such as broad coverage of data, application history and wide user availability and user friendliness were not reviewed explicitly.

2.3. Systematic literature review

The Web of Science, Google Scholar and the EcoService Models Library hosted by the US Environmental Protection Agency (US-EPA, 2015) were searched for papers that might contain EPFs linking the effect of chemical stressors on biological characteristics of species that make up service providing units, and how this in turn affects the delivery of a particular ecosystem service. Existing ecosystem service toolkits not based on expert judgement were screened as some contain a suite of ecological models including EPFs (Bagstad et al., 2013):

- Corporate Ecosystem Services Review, ESR (Hanson et al., 2012), https://www.wri.org/publication/corporate-ecosystem-services-review
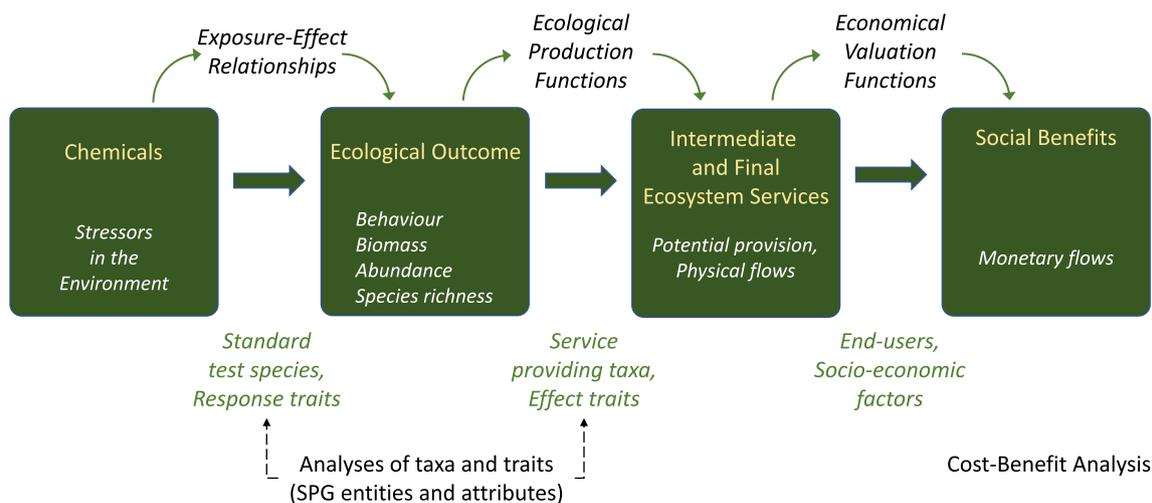
Fig. 1. The translation of effect data from standardised ecotoxicological testing for ES impact assessment using ecotoxicological exposure-effect relationships and ecological production functions based on response traits and functional traits (also called 'effect traits') respectively. EPFs quantify potential provision of intermediate or final services, and socio-economic factors determine actual flows of services 'physical flows' and benefits 'monetary flows'.

- US-EPA EnviroAtlas (Pickard et al., 2015), https://www.epa.gov/enviroatlas
- GISCAME (Fürst et al., 2010a; Fürst et al., 2010b)
- Integrated Valuation of Ecosystem Services and Trade-offs, InVEST (McKenzie et al., 2012; Sharp et al., 2020), https://naturalcapitalproject.stanford.edu/software/invest
- Toolkit for Ecosystem Service Site-based Assessment, TESSA (Peh et al., 2013), http://www.birdlife.org/worldwide/science/assessing-ecosystem-services-tessa
- Land Utilisation and Capability Indicator, LUCI (Jackson et al., 2013), https://www.lucitools.org.

For each toolkit, the associated reference lists of review papers on how natural capital delivers ES were scanned, focusing on relevant fields and attributes (e.g. biodiversity - ecosystem function and functional trait literature, freshwater biology, agricultural science, ecotoxicology, functional processes, soils, plant roots, ecosystem services and modelling). Mapping tools as such were outside the scope of our study since their emphasis is on spatial areas rather than ecological units of service provision.

Search strings were comprised from three elements, i) <habitat>, ii) <SPU>, iii) <ecosystem services and processes>: Habitat *AND* SPU characteristic *AND* ecosystem service/process. Studies that also modelled the economic value of the ES were explicitly recorded. Grey literature was taken into consideration where known.

2.3.1. Habitat
Broad habitat categories were taken from the UK National Ecosystem Assessment habitats: coastal margins, farmland, grasslands, heathlands, rivers, urban, wetlands, woodlands (Mace et al., 2011).

2.3.2. SPU characteristics
Search terms ranged from broad to more specific terms that characterized specific measures of ecological endpoints, in alphabetical order:

abundance, arable crop, biodiversity, biomass, bioremediation, decomposer, ecological attribute, ecosystem service provider, feeding rate, fish, forests, functional character, functional composition, functional diversity, functional group, functional response, functional richness, functional trait, growth, invertebrates, macrophytes, microorganisms, organism providing service, plants, reproduction, richness, salmon, service providing unit, SPU, trees, vertebrates.

2.3.3. Ecosystem services and processes
Search terms were based on the CICES V5.1 classification of ES and were broadened to describe several processes that fit under the 'supporting services' or 'intermediate services' category. Terms also range from broad to more specific: ecosystem service, ecosystem process, ecosystem function, biological control, decomposition, food production, soil formation, erosion prevention, heat exchange, pollination, pest regulation, disease control, invasion resistance, oxygen regulation, climate regulation, carbon sequestration, carbon storage, surface water flow, run-off, nutrient/sediment retention, soil fertility, nutrient regulation/cycling, water regulation, water quality, water purification, waste treatment, flood regulation/ alleviation, flood protection, habitat provision, habitat for species, clean water provision, wild food production, timber, air quality regulation, air pollution removal, air purification, atmospheric regulation, erosion control, sediment retention, hazard prevention, bioremediation, genetic resources, genetic diversity, hydrological regulation, soil loss prevention, soil stability, cultural service, aesthetic value, cultural heritage, cultural value, ecological knowledge, environmental education, fishing, identity, inspiration, landscape beauty, recreation, scientific knowledge value, sense of place, spiritual, tourism, tranquillity.

Primary systematic searches were restricted to the years 2000–2020, as some of the broader search terms could return

thousands of papers. Also, searches were often refined further by adding the descriptor 'model'. The titles of the articles that the search revealed were scanned, and if deemed appropriate the abstract was read. The paper was accepted for full review if the abstract showed the paper addressed any of the links between stressors, SPUs and ES delivery.

2.4. Non-systematic search

Systematic literature searching may overlook publications that refer to a particular ES by its name (e.g. 'pollination') without quoting the words 'ecosystem services' per se in their abstract or keywords. These limitations have been previously identified (Prather et al., 2013). Also, the terms 'ecosystem service' and 'ecological production function' are relatively recent, and their use was not common prior to the 2000s, so some older publications addressing e.g. insect ES may not be detected. Therefore, we complemented the primary search with a non-systematic search by scanning reference lists of key relevant papers, and by a citation search for quoting publications.

2.5. Classification of EPFs

In order to classify EPFs we applied three approaches.

2.5.1. Classification by methodology in derivation
Selected publications were screened for quantitative description of SPU functioning in the provision of any ES, be it in mathematical terms (models), graphical representation (depicted correlations or regressions), or mere wording. Functional relationships were referenced that can be based on standardised tests species from chemical toxicity testing guidelines, or species that are taxonomically closely related to the service providing unit (SPU) species, and involve explicit measurement or estimation of a final service endpoint. The degree of quantification and the causality of the relationship were noted as: factorial (field observed difference between two levels of density of service provider and a control obtained by observation or experimentation), modelling prediction, wide range correlation, or experimentally obtained regressions. Relationships quantifying economic valuation of benefits or damage prevention were also noted. At all times the shape of the relationship was registered when specified.

2.5.2. Classification by ecological type
Production functions were distinguished by ecological type based on taxonomic identity of the SPU grouped by Class, and by functional mechanism in the provision of the ES. The classification is open, with no classes designated a priori, and EPFs were grouped or kept separate as deemed fit. For instance, six references on earthworms promoting plant growth were grouped to a single EPF, whilst a paper on termites and ants was distinguished as a single EPF, yet separate from earthworms, on the basis of burrow size and architecture, and effects on soil microbial activity in the borrow microhabitat. Likewise, EPFs for pollination were distinguished by crops based on e.g. flowering time, colour and habitat. EPFs have been associated to service providing taxa that provide the service in a comparable way (by a similar mechanism), so that functioning of all taxa in the SPU can be assessed using that EPF; if pollinators are specifically attracted to different flower colours, they are considered different SPUs for different crop species.

2.5.3. Classification by dimensions for specific protection goals
Protection goals outlined in legislation are often too general to be directly applicable for ERA. Therefore, they need to be translated into specific protection goals (SPGs) that delineate the environmental components to protect, the maximum impacts that can be predicted and, in the case of regulated products, tolerated, over what time period, and where. Recent guidance towards the development of SPGs for use in ecological risk assessment accounting for biodiversity and ecosystem services identified six dimensions to define SPGs: (1) the entities to be

protected, (2) the attributes and/or functions of those entities, (3) the magnitude, (4) the temporal and (5) spatial scale of the effects on these attributes and/or functions that can be tolerated without impacting the general protection goal, and (6) the required degree of certainty with which the protection goal defined should be achieved (Nienstedt et al., 2012; EFSA Scientific Comm, 2016a). The ecological entity refers to the level of biological organisation of the SPU (e.g. individuals, populations, etc.), complemented with the (bio)physical environment where organisms (or group of organisms) live or occur (habitat). These dimensions defining SPGs have been constructed as far as possible in a hierarchical way, so that the selection of an option at the left end of the dimension is protective for options that follow; the options for the ecological entity dimension are (EFSA Scientific Comm, 2016a):

individual<(meta)population<functional group<community

<ecosystem<habitat.

To each of these entities some optional attributes can be associated to detail the definition of a specific protection goal (Table 1).

The dimensions of entity and attribute represent statements about the nature of the endpoint to be assessed, and these are essential parts of EPFs as well. In the current paper we elaborate an illustration of this approach for one ES, pollination, which is well documented in the EPF literature. We classified pollinator SPUs and associated EPFs by the dimensions of 'ecological entity' and 'attribute' as represented in the production functions retrieved from the literature.

3. Results and discussion

Functional ecology has a long research history (Grime, 1973; Cummins, 1974; Diaz and Cabido, 1997; Tilman et al., 1997). However, after some five decades the functioning of species in the delivery of ES is not well-quantified. While the ES concept is just about equally aged ('natural services' (Holdren and Ehrlich, 1974), 'ecosystem services' (Ehrlich and Ehrlich, 1981)), it has only more recently gained wider acceptance (Costanza et al., 1997; MA, 2005), which may to some extent hamper a review of ES literature before mainstreaming of terminology

Table 1
Classification of ecological production functions by dimensions of ecological entity and optional attributes for the elaboration of specific protection goals in ecological risk assessment based on EFSA guidance (EFSA Scientific Comm, 2016a).

Ecological entity	Attribute options
Individual	1. Behaviour 2. Growth 3. Reproduction 4. Survival
Colony[a]	1. Behaviour 2. Abundance, biomass 3. Reproduction 4. Survival/growth
(Meta-) Population	1. Population dynamics in terms of abundance (e.g. numbers of individuals and their fitness) or biomass (EFSA PPR Panel, 2014) 2. Population growth
Functional group (FG)	1. Process (e.g. primary productivity, decomposition, nutrient cycling) 2. Abundance, biomass 3. Within- and between-species diversity
Community	1. Within- and between-species diversity 2. Biomass
Ecosystem	1. Processes determining resistance to change and resilience of ecosystem properties (stability) 2. Within and between species diversity
Habitat[b]	1. Landscape (land use, type) 2. Habitat structure (spatial distribution, connectedness)

[a] EFSA PPR Panel, 2012.
[b] EFSA Scientific Comm, 2016b.

(Prather et al., 2013). Most studies have focussed on ecosystem functioning, often intermediate services or supporting services, and have not progressed to the next level of service use. Hence, our systematic search rendered only 111 papers for full review, from which we retrieved just 18 ecological production functions suitable for use in chemical ERA. The additional progressive search using ES names and terms from CICES, as well as screening of references and citations of key papers was more rewarding. We aimed for exemplary coverage for all 47 biotic ES as defined under CICES v5.1 with respect to provisioning and regulating services. From the search result hits we selected publications that provided functional relationships between service providing entity and ES to the highest degree of quantification and causality available. All in all, we have selected 235 papers describing a quantitative relationship of some sort between an SPU and an ES (Table SI-2, summarized in Table 2 and extended summary in Table SI-2).

A total of 121 different EPFs were compiled from the scientific literature that quantify and predict potential service provision by a specific SPU, and in addition we found 31 correlative functional relationships. Interestingly, we also found 57 economic valuations of ES provided by specific SPUs, of which 31 were combined with a quantification of the EPF (Table SI-1). For twenty services EPFs were found for multiple service providing taxonomic groups, on average 4.1 groups per service, five ES were provided by a single taxonomic group, and for three ES only correlative relationships were found (Table SI-2).

Twenty-two EPFs quantified provisioning services, and 99 related to regulating and maintenance services (Fig. 2, Table SI-2). As CICES v5.1 describes 47 final classes in these two ES divisions, this roughly equals on average three EPFs retrieved per ES. These EPFs were composed from 41 and 194 references respectively regarding provisioning services and regulating and maintenance services.

Our review of EPFs and standardised toxicological tests reveals that there is a distinct lack of EPFs that can facilitate extrapolation of chemical effects measured in these tests to ES delivery. Whilst ecological textbooks illustrate how functional responses, encompassing many cases of species interactions (e.g. predator-prey and herbivore-plant) relate to ecosystem processes, these relationships almost all refer to intermediate (supporting) ES while only a few include final ES (Luck et al., 2009; Jonsson et al., 2014; Bruins et al., 2017).

Standard toxicity tests are indicated in Table SI-1 when relevant to SPUs and EPFs compiled from the literature review. The assessment of relevant toxicity tests was limited to the methods published as technical guidelines by OECD, ISO or ASTM, totalling 107 different types of tests when similar tests between organisations were combined; four OECD guidance documents have also been included (Table SI-3). Although other organisations have developed and published test methods, the methods published by OECD, ISO or ATSM are generally used to support chemical assessments in Europe and North America and represent the great majority of regulatory accepted test methods. The relevance of production functions to existing standard tests is related to taxa and/or endpoint. Where there are similar methods available from OECD, ISO or ASTM, all are listed under broadly relevant taxonomic groupings. Tests with the highest level of relevance are listed, although for the many ES where tests with only distantly related taxa are available, these are included.

There was no relationship ($R^2 = 0.216$) between number of available tests and the volume of production functions encountered in the literature, the diversity of tests is not reflective of the functional diversity of the various taxonomic groups (Table 2). Summary of availability of ecological production functions in the scientific literature, and availability of organism relevant standardised tests for chemical toxicity testing under ISO, OECD, or ASTM guidelines (further detailed in Table SI-3). Similar or analogous standardised tests were grouped together as one. While a wide array of tests is available for groups such as microbes, insects and crustaceans, other groups seem largely underrepresented. Corals are not covered by OECD, ISO or ASTM standardised tests, whilst important in the protection against coastal flooding (Beck et al., 2018)
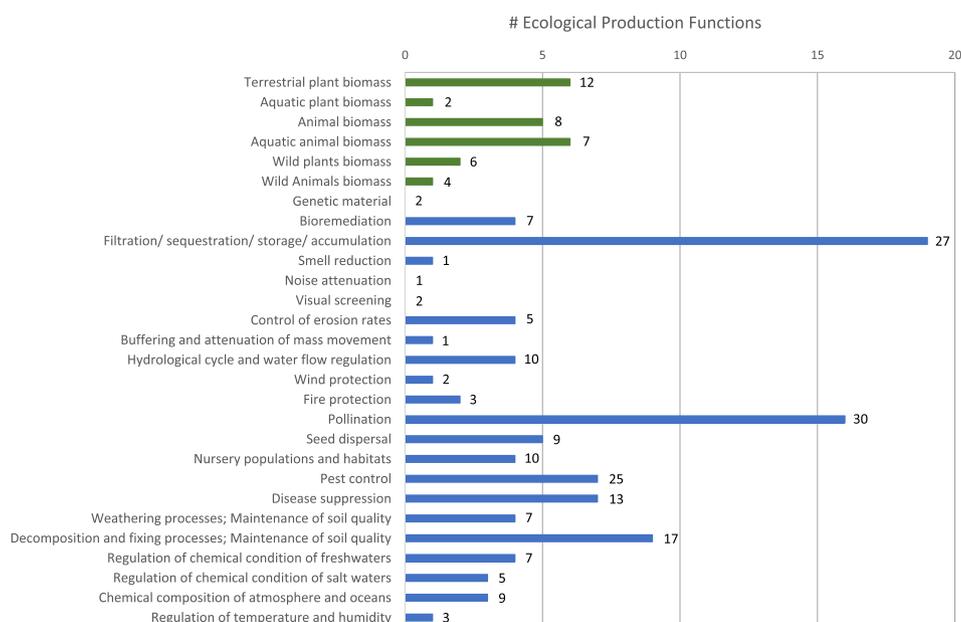
Fig. 2. Number of ecological production functions retrieved from the literature; relationships based on correlation were excluded. Green bars represent provisioning ecosystem services, blue bars are for regulating and maintenance services, as classified by CICES. The number of corresponding publications is given at the top of the bars. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

and erosion (Guannel et al., 2016). While corals cannot be sufficiently cultured in the laboratory for testing, sea urchins are often used as substitute taxa for testing, since the test involves reproduction endpoints looking at gametes released and survival and development of the embryos (which is a similar reproduction method in both taxa). Yet, while this practice may be acceptable as the best alternative for toxicity testing, ecological functioning is very different between the taxa, and the knowledge gap around the link between standardised testing and ES provision is relatively large here.

Table 2
Summary of availability of ecological production functions in the scientific literature, and availability of organism relevant standardised tests for chemical toxicity testing under ISO, OECD, or ASTM guidelines (further detailed in Table SI-3). Similar or analogous standardised tests were grouped together as one.

Organism group	Number of EPFs	Number of standardised tests	Comments
Microbes	20	26	Including enzyme assays; No test for Archaea
Algae	4	3	
Aquatic plants	10	3	
Terrestrial plants	31	5	
Rotifers	2	3	
Cnidaria	4	0	No test for corals
Nematodes	2	1	
Annelids	11	10	
Echinoderms	2	1	
Molluscs	5	7	
Insects (of which bees)	46 (10)	11 (5)	
Collembolans	1	2	
Arachnids	1	2	Two tests for mites
Other soil arthropods	1	0	EPF for soil invertebrate species combinations including millipedes and isopods
Crustaceans	5	13	
Fish	8	10	
Amphibians	0	5	
Reptiles	0	0	
Birds	12	2	
Mammals	11	3	

Archaea are another large group for which no test have been standardised. We have aggregated this group with bacteria, fungi and protists under 'Microbes', for reasons of functional homologies. Little is known however about the toxicological sensitivity of this separate kingdom that only fairly recently has gained awareness of system ecologists for their roles in nitrogen mineralisation, metal complexation and application potential for e.g. wastewater treatment (Cavicchioli, 2011).

Some groups seem underrepresented by EPFs, e.g. amphibians, reptiles and arachnids (particularly spiders and mites). Although literature is available to quote the contribution of such groups to a variety of ES (Valencia-Aguilar et al., 2013; Hocking and Babbitt, 2014), we have not found any references to describe quantification of the relationships with the exception of wildlife as food provision. Likewise, mites are well known as biological control agents and decomposers (Gerson et al., 2003; de Groot et al., 2016a), but EPFs have not yet been developed.

We were able to retrieve functional relationships for all provisioning and regulating services, but obviously the number of publications varied widely (Fig. 2). Likewise, the number of EPFs varied among ES, and increased proportionally with the number of publications selected ($R^2 = 0.847$).

3.1. Prominent ES

In terms of number of publications, the most prominent biotic ES types were (Fig. 2, Table SI-1): pollination (30), filtration/ sequestration/ storage/ accumulation (27), pest control (25), decomposition and fixing processes (17), disease suppression (13). The representation of EPFs for provisioning of food and fibres is exemplary and should be considered a large underestimation, as we did not seek a comprehensive overview for different plant and animal species considering that EPFs would be abundant in crop production science (Choudhary et al., 1996; Marcelis et al., 1998; Immerzeel et al., 2014), at the level of species or even cultivar. Our findings of these most prominent ES are only in part consistent with the most frequently studied CICES classes (or class clusters) of ecosystem services. Based on a review of 85 scientific papers from which 440 indicators were identified, the most frequently studied CICES classes (excluding cultural services) were global climate regulation and bio-remediation (Czúcz et al., 2018). This indicates that ecological production functions are not generally specified in ES studies,

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J.H. Faber, S. Marshall, A.R. Brown et al.   Science of the Total Environment xxx (xxxx) xxx

with exception of some ES. Pollination and pest control may represent services that have received relatively more attention from ecologists and risk assessors, but less from the socio-economic sciences as they can be seen as intermediate services. Monetisation is however very well possible (Table SI-1).

The EPFs found in the literature exhibit various degrees of quantification of the relationship between SPU and ES potential, ranging from simple calculation factors to detailed functional response curves over a wide range of the SPU explanatory variables, e.g. population density of functional group diversity (cf. Table 1). Also, the degree of causality varied from observational correlations to experimentally determined regressions. In our compilation of suitable references, we have aimed to select relationships with as best quantification and evidential weight as we could find, disregarding publications providing relationships less detailed if addressing a similar EPF. An overview of the classification of ecological production functions by degree of quantification and causality shows that sound EPFs have been established only for only a few ES: pest control, pollination, decomposition, and filtration/sequestration/storage/accumulation (Fig. 3). For some ES only factorial assessment factors are available based on correlation of service provider and service provision, such as for Genetic materials.

3.1.1. Pollination (CICES 2.2.2.1)

Partly motivated by the worldwide pollinators decline, pollination is among the best studied final ES and the scientific literature has been reviewed and synthesized extensively (e.g. IPBES, 2016). Based on our review, we identified 25 potentially useful quantifications of pollinator functioning in a substantial number of crops, in varying degrees of causality, quantification and economic valuation of potential service provision (Figs. 2, 3, Table SI-1).

From an ERA perspective, the study by Kleczkowski et al. (2017) is most interesting because it provides an EPF that links population dynamics (density) of wild and commercial bees that are closely-related

to the honey bee and bumble bee used in OECD standardised toxicity testing (see Table SI-4) to calculate pollination and crop yield, and then a profit function. The model was designed specifically for use in the context of pesticide use, to understand the trade-offs between using pesticides in strawberry production and the benefits that pollinators deliver in terms of yield. This EPF incorporated the whole causal pathway from stressor to final service, where the standard testing endpoints using honeybee (*Apis mellifera* L.) or bumblebees (*Bombus terrestris* L.) are adult or larval acute or chronic toxicity (mortality) under laboratory conditions after single or repeated doses (Table SI-4). Individual acute oral and dermal toxicity data for adult and larval bees can be used in population modelling to assess the effects of a chemical on population-level foraging activity as a proxy for pollination (Van den Brink et al. subm.).

In ES assessments however, many of the existing and widely used EPFs for crop pollination and yield quantity or quality are based on the spatial distribution of crops in a landscape and predictions of pollinator occurrence and movement. These estimates are based on deriving the probability of occurrence relative to the availability of nesting sites and floral resources, located using land cover maps. The pollinator service is then calculated taking account of crop location, potential pollinators and foraging distance. The EPF developed by Polce et al. (2013) is an adaptation of the Lonsdorf et al. (2009) model used in the InVEST toolkit (Sharp et al., 2020). It is based on data of presence or absence of pollinators rather than a quantitative estimate. This model may have the potential to be adapted further to include bee population dynamics (Sharp et al., 2020).

Other researchers have studied species diversity in functional groups and compared various taxa of wild pollinators to domesticated honeybees and their interaction in the production and quality of fruit and seed crops. Various studies have shown complementarity between species or traits in the pollinator community (Garibaldi et al., 2015; Garibaldi et al., 2018; Woodcock et al., 2019). Wild bees may be less
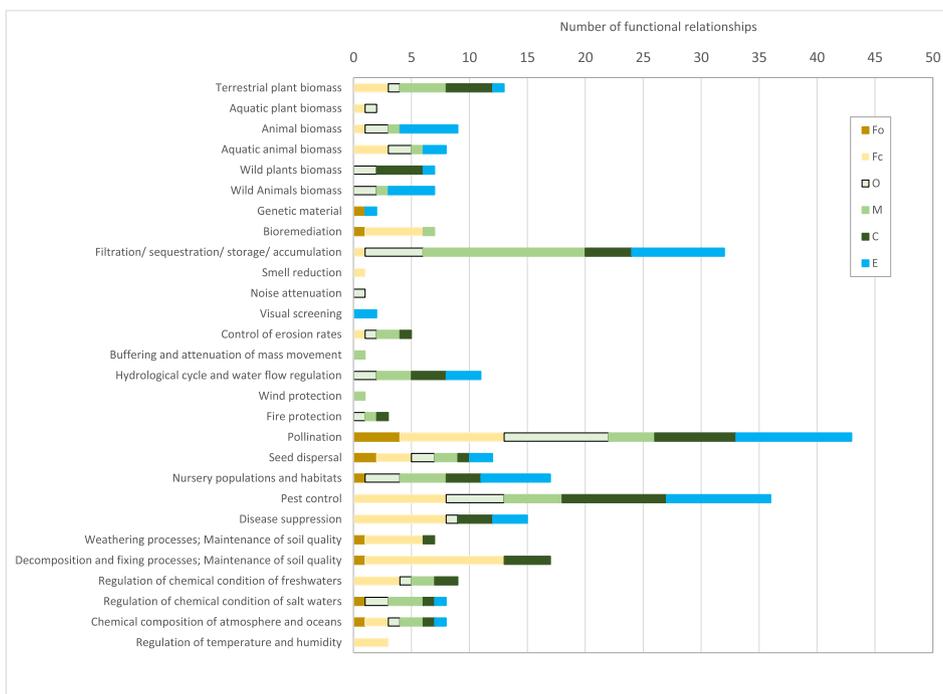
Fig. 3. Classification of ecological production functions and correlative functional relationships by degree of quantification and causality of the relationship. F, factorial (Field observed difference between two levels of density of service provider; subscript 'o' is for Observational (in brown), subscript 'c' is for Causal experimentally obtained (yellow)); M, Model prediction (light green); O, correlative (field Observed correlation between service and service provider based on more than two density levels (green)); C, Causal (field experimentally obtained relationship between more than two levels of service provider and service response (dark green)); E, includes Economic assessment of benefits or values (blue). Studies included in this presentation are "best available" in terms of describing a functional relationship as a function rather than factorial comparison of presence/absence, and with experimentally demonstrated causality rather than observed correlation: regression > correlation > factor, and field observation > lab observation. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

or equally effective in pollination than honey bees (Adler and Irwin, 2005; Rader et al., 2009), but particularly in comparison to pollination by commercial honey bees it is interesting to note that wild pollinators can further enhance economic value of harvests by increased yield biomass and quality (de Groot et al., 2015; de Groot et al., 2016b; Földesi et al., 2016; Horth and Campbell, 2018). Economic value of the crop may not associate with the most effective pollinator, therefore. Obviously, the extrapolation from ecological functioning to market value is best done with knowledge of both an EPF and an economic valuation function.

Many crops have specific pollinating taxa, varying from wild and cultivated bees, butterflies and moths, beetles, flies, to hummingbirds and bats (Westerkamp and Gottsberger, 2000; Thapa, 2006). EPFs may differ between pollinating taxa and, for a given pollinator, an EPF may vary between crops. Hence, we incorporated a range of studies covering a range of pollinator guilds and crops (Table SI-1). However, most studies do not quantify the flower-visiting and relative abundances of species involved, or at best do so indirectly by sampling the surrounding landscape. Pollination activity itself is poorly associated to SPU from a taxonomical point of view, as it is known that pollinating species and guilds can differ in pollen transfer efficiency (Jennersten and Morse, 1991; Miyake and Yahara, 1998, 1999; Larsson, 2005; Jauker et al., 2012; Macgregor et al., 2015) with subsequently differing impact on crop yield (Abrol, 1989; Garibaldi et al., 2013; Martins et al., 2015; Sutter and Albrecht, 2016; Hodgkiss et al., 2018). Species may even interact behaviourally or otherwise, affecting their contributions to service provision: functional trait complementarity, niche partitioning and synergism (Greenleaf and Kremen, 2006; Blüthgen and Klein, 2011; Brittain et al., 2013; Fründ et al., 2013; Martins et al., 2015; Blitzer et al., 2016; Garibaldi et al., 2018), or antagonism (Bronstein et al., 2003; Dainese et al., 2018) among species. In a modelling study using field data it was found that an increase in pollinator diversity was beneficial for the reproduction of some plants whereas it was harmful for others, the outcome depending on differences in effectiveness among pollinators. This suggests that in pollination systems the diversity–function relationship reflects the interaction between among-pollinator differences in effectiveness and frequency of interaction (Perfectti et al., 2009). Given the species-specific differences in pollination efficiency and contribution to crop yield, EPFs established for a particular pollinator taxon should be generalised for other taxa with some caution.

As described in Table SI-1, most publications of quantitative studies on pollination have been focused on insects, mostly Hymenoptera and some Diptera. Although contribution of birds and mammals to plant pollination has frequently been documented, involving relatively specialized feeding niches in some (sub)tropical fruits (pine apple, banana, durian, columnar cacti, *Agave tequilana*), flowers *(Protea)* and trees (Eucalyptus, date palm) and being relatively more important in some island communities (New Zealand, Hawaii)(reviewed by Whelan et al., 2015 and Gaston et al., 2018), we have only found quantifications of EPF pollination services by these taxa obtained by correlation, not experimentation (Table SI-1). Economic valuation of bat pollination services, however, is substantial, e.g. for durian fruits *(Durio)* US$ 120 million annually, and petai *(Parkia)* seeds and wood markets represent a value of US$15 million annually in peninsular Malaysia alone (Fleming et al., 2009).

3.1.1.1. Pollinator EPFs. As different biological processes may be involved in establishing seed set (pollination sensu *strictu*, affecting fruit biomass and morphology) and the chemical quality of seeds and fruits, we accounted for four basic types of EPFs by insect pollinators. On top of that social insects are thought to increase resource use through communication, thus rendering functional response different from non-social insects. Birds and bats were again considered to represent different EPFs, as they may cover longer distances between plants, increasing gene flow. EPFS were counted separately for different crops,

acknowledging differences but not classifying by the degree of dependency on animal pollinators of the crop (Klein et al., 2007). Thus, a total of 25 different EPFs were compiled from the literature (Fig. 2, Table SI-1).

Ecological production functions for yield that are based on density of pollinators have been characterized as linear, curvilinear or log-linear relationships, incidentally a bell-shaped response curve was reported (Table SI-1). When based on diversity of pollinators, yield production functions were generally linear.

3.1.2. Specific protection goals for pollinators

If in ERA pollination were to be defined an SPG, what could be appropriate entities and attributes in pollinators, and how could these be supported by existing EPFs? Strictly in biological terms, the essential attribute would be individual or colony behaviour reflecting the number of pollen successfully transferred to flower styles, as fertilization will give rise to seeds and fruits. Obviously, in field experimentation aimed at quantifying the functional biodiversity-ES production relationship, this is not the preferred level of observation. Even the quantification of flower visits as a proxy for pollen deposition – in itself already a poor indicator for pollination efficiency (King et al., 2013) - is often replaced by quantifying local abundance of pollinating species. Thus, whilst individual behaviour in pollen deposition is ecologically the most relevant attribute in the assignment of pollinators as a specific protection goal, this is insufficiently supported by EPF literature.

Pollinator functioning has been expressed in terms of flower visits, amount of fertilizations, fruit biomass, or some measure of fruit/seed quality, and in response to mostly pollinator community (i.e. functional group) abundance or diversity (Table SI-1, summarized in Table 3). The service provider entities are most often described in terms of functional groups, either by function, abundance, or diversity of species or traits. In summary, two entity-attribute combinations in pollinators are represented in the established EPFs for various crops and regions in the world:

A. Added value of increased pollinator functional group *density* to crop yields; e.g. bee introductions will enhance crop yield;
B. Higher pollinator species *richness* and trait diversity will increase crop yield; wild bees and other pollinators improve yield biomass and quality more than honeybee by itself, indicating functional complementarity.

In conclusion: based on available quantitative production functions for pollination, the dimension of functional group at any attribute level would seem to be the best substantiated for developing quantitative specific protection goals.

3.2. Cultural services

Cultural services have been frequently reviewed, but the methodological aspects of these studies focussed on the socio-economic aspects of indicators rather than ecological (Hernández-Morcillo et al., 2013; Milcu et al., 2013; Pröbstl-Haider, 2015; Hegetschweiler et al., 2017; Cheng et al., 2019), and ecological production functions have not been identified. Except for spiritual and heritage values (e.g. totemic and symbolic species), there are few cases of specific services being represented by a single typical SPU species or group of species. Rather, culture and customs will vary locally, and a high dependence of indicators from data quality and availability has been outlined (La Rosa et al., 2016). The validity of a specific SPU may decline with the scale of an assessment where other species would substitute in the provision of the service. Cultural services with broader and more diverse SPUs will therefore require multiple EPFs. In site-specific ERA the assessment of cultural services may be more focussed, as the site of assessment may feature specific cultural services, with specific SPUs. For instance, particular game fish species, e.g. salmonids, can largely contribute to the

Table 3
Summary of entities for service provision and their attributes as expressed in ecological production functions in current literature, expressed by number of EPFs not including correlations.

SPU entity	SPU attribute			
	1	2	3	4
Individual	Behaviour	Growth	Reproduction	Survival
Colony	Behaviour	Abundance, biomass	Reproduction 1	Survival/growth 2
(Meta-) Population	Abundance, biomass 2	Population growth		
Functional group	Process 10	Abundance, biomass 6	Diversity 9	
Community	Diversity	Biomass		
Ecosystem	Resilience	Diversity		
Habitat	Land use, type 1	Habitat structure 2		

recreational values of angling (Knowler et al., 2003; Fulford et al., 2016), and observing nature is a benefit that can be established by lay people's aesthetic appreciation of meadows in relation to species richness (Lindemann-Matthies et al., 2010) or birdwatching (Gaston et al., 2018). The actual use of the service produced ('physical flow', Fig. 1) is to a large extent determined by local circumstances, and ecological production functions therefore seem difficult to generalise.

3.3. Indicators in ES assessment

Matching conventional endpoints in ERA with existing ES assessment frameworks meets with a few challenges. ES-based ERA can, if not should, make efficient use of indicators and their application in ES assessments, where available. This is particularly relevant when SPUs are used as indicators that have a close taxonomical or functional resemblance to species and endpoints in standardised testing. However, in ES assessment the choice of indicators tends to be at a level of high abstraction and biological integration, even more so if the scale of assessment is large. Maes et al. (2015) performed a trend analysis of ES provision across Europe in 2000–2010, using a total of 30 indicators (standardised across the EU, quantitative values at least at the national scale, and data available for at least two years). For some ecosystem services, no indicators could be found that comply with the criteria. Particularly, for most cultural ecosystem services specific data and indicators were lacking, and only three indicators were used. For regulating ecosystem services several gaps were evident as well. At the CICES group level indicators for mediation by biota, pest and disease control, and water conditions were not available, at the class level even more indicators were lacking. Typically, provisioning ecosystem services were currently better covered by indicators.

A review of 405 peer-reviewed ES research papers concluded that ES appear to be poorly quantified in many cases, as often only one side of the cascade is considered (either the ecological or socio-economic side) and oversimplified and variable indicators are often used (Boerema et al., 2017). Twenty-one ES analysed had on average 24 different measures, which may indicate the complex reality of ES and/or suggest a potential lack of consensus on what constitutes an ES. Uncertainty is often not included and validation mostly missing. When analysing which part(s) of the ES cascade each measure corresponded to, it was found that for regulating ES, ecosystem properties and functions (ecological aspects) are more commonly quantified (67% of measures). Conversely for provisioning ES, benefits and values (socio-economic aspects) are more commonly quantified (68%), and cultural ES were predominantly quantified using scores (35%).

Another complication for matching ERA with existing ES assessment frameworks is that ES indicators often represent provision by surface area, usually land cover/habitat type, and the assessment therefore is based on service providing areas rather than service providing units. For application in ERA such indicators can only be used if environmental quality in these areas can be quantified, i.e. where the ecotoxicological impact of the stressor to the service providing unit is known. This of course requires prior knowledge of environmental concentrations of the chemical, as well as exposure and effects in the SPU, or some related taxa. To get that far in establishing an environmental scenario, EPFs would be required to the level of supporting services at least, i.e. ecosystem functioning of relevant ecosystem structures and processes. At such a point, ES provision can be assessed either via enumeration of service providing areas, or more directly, via further extrapolation towards final ES provision. Which method to use is a case by case matter of quality of available data and remaining uncertainty.

For marine, coastal and estuarine habitats, the provision of several ES can be traced back to a small set of species groups, e.g. corals, seagrass, and mangroves. The lack of standard tests for many marine organisms, and relevant endpoint tests for these taxa in particular, hampers implementation of an ES-based approach for the marine environment as assessments will have to be based on surrogate freshwater or terrestrial plant and animal tests.

Various taxonomic groups, including most soil organisms, are not usually included in ES models, because many services provided by these groups are indirect. Moreover, those efforts that have incorporated such groups in ES models focused on temperate regions where the ecology, natural history, and functional roles of species are relatively well understood. For other parts of the world much remains to be learned about basic natural history, functional roles, distributions, and the community composition of such groups before their contributions to ecosystem services can be valued.

3.4. Diversity of species and traits

A relatively large proportion of papers retrieved by the search were about biodiversity relationships with ecosystem functioning and services. Functional relationships based on species richness or community trait diversity, however, cannot directly be associated with standardised toxicity testing, as no standard toxicity tests are available that focus on diversity of species or traits as such. However, intact communities may be used in cosm testing to study the effects of chemicals on species composition and functional groups. To this extent there is some guidance on testing for terrestrial soil core microcosms (ASTM, 2012), freshwater microcosms (ASTM, 2016), and simulated freshwater lentic microcosms and mesocosms field test (OECD, 2006). Alternatively, EPFs between ES and species diversity may be used in ERA in combination with species-sensitivity distributions (Posthuma et al., 2001), where the percentage of potentially affected species (Traas et al., 2002) at a given exposure concentration to a chemical may be evaluated as a proxy for community diversity effects. However, while analysis of single ecosystem processes in isolation generally reveals a positive but saturating relationship with increasing biodiversity (Tilman et al., 1997; Hector et al., 1999; Heemsbergen et al., 2004), analysis of ecosystem multifunctionality can reveal that different processes are not affected by exactly the same species. Moreover, because different species affect different processes, multifunctional ecosystems will require greater biodiversity than suggested by studies focusing on single ecosystem processes in isolation (Hector and Bagchi, 2007). The functional characteristics of the component species in any ecosystem are likely to be at least as important as the number of species for maintaining critical ecosystem processes and services. In particular, differences in functional group composition can have a larger effect on ecosystem processes than functional group richness alone (Hooper and Vitousek, 1997; Heemsbergen et al., 2004). In risk assessment and management for ecosystem services greater weight of evidence may thus be given to functional ecological aspects than to taxonomical richness, and to complementarity in functional traits (also called 'effect traits') and presumably complementary additive or synergistic species interactions,

rather than traits numbers. Furthermore, the buffering impact of species diversity on the resistance and recovery of ecosystem services as observed consistently in literature reviews and large scale analyses (Worm et al., 2006; Ives and Carpenter, 2007; Hooper et al., 2012) generates insurance value that should be incorporated into economic valuations and management decisions (Pascual et al., 2010; Pascual et al., 2015).

3.4.1. Species interactions and ecosystem coupling

Interspecific relations and species-environment interactions are numerous and often hidden (Valiente-Banuet et al., 2015), and a loss of ecological interaction may have far-reaching consequences for the functioning of ecosystems. In fact, it has been proposed that ecological interactions may disappear well before species do (Valiente-Banuet et al., 2015). The use of EPFs based on SPUs may overlook this as it is unclear to what extent they are sensitive for, and inclusive of, the degree of ecosystem coupling (Ochoa-Hueso, 2016). Studies that focus on ecosystem coupling are emerging, but have been focussed on ecosystem structure (Bascompte, 2009; Rzanny and Voigt, 2012; Weiner et al., 2014; Duan et al., 2016) and functioning (Sun et al., 2019; Ochoa-Hueso et al., 2020), and do not yet establish quantitative relationships with final ES provision. Nevertheless, more tightly coupled ecosystems may support a wider range of functions, which can be associated with a greater efficiency in the use of nutrient resources and the processing of organic matter (Morriën et al., 2017; Sobral et al., 2017; Risch et al., 2018). The approach has proven useful to identify key groups of ecosystem components in terms of biodiversity interactions synergistic to ecosystem functions and services, and to detect changes in the composition of such groups related to land use (Felipe-Lucia et al., 2020). Information of this kind may be useful in environmental risk assessment when accounting for interactions between species depending upon each other or having an effect on each other, and reducing uncertainty on the basis of the proportion of positive or negative links from all possible links in the network, weighted by the strength of the links, i.e. 'connectance' (Felipe-Lucia et al., 2020). Connectance quantifies the importance of synergies in the ecosystem in relation to trade-offs: high connectance indicates that many different trophic groups are important in driving functioning or service supply and that many different ecosystem functions are related to several services. In contrast, low connectance indicates a simpler system in which only a few trophic groups or functions are related to a function or service. It would also indicate the robustness of food webs when subjected to species loss (Gilbert, 2009), further informing risk management on the implications of biodiversity loss.

3.4.2. Functional redundancy

The concept of EPF was originally intended as a quantification of ES provision in response of the functioning of the SPU, where the SPU ideally would include all functionally relevant ((meta-)populations of a) species (Luck et al., 2003; Kremen, 2005). Theory and practice are not the same here. Quantification of ecological functioning is most often done for singular species, or small groups of species being closely related or belonging to the same functional group, but seldom for the full suite of service providing taxa (if that is even possible). For pollination, for instance, an increased service provision has been observed with increasing species richness in the functional group of insect pollinators (Greenleaf and Kremen, 2006; Jauker et al., 2012; Hodgkiss et al., 2018), and addition of species can further enhance yields (Horth and Campbell, 2018). A clear demonstration of functional redundancy within the functional group, i.e., species that overlap in their ecological roles (Lawton and Brown, 1993; Walker et al., 1999), has not been documented for pollination. Rather, fruit set can increase with species richness and more so in assemblages with high evenness, indicating that additional species of flower visitors contribute more to crop pollination when species abundances are similar (Garibaldi et al., 2015). Pollination services increased with wild bee abundance and richness, and

functional group diversity explained more variation in seed set than species richness (Hoehn et al., 2008; Martins et al., 2015; Blitzer et al., 2016), although correlative field scale studies cannot disentangle the effects of abundance and richness.

Functional redundancy has been suggested for mycorrhiza (Rineau and Courty, 2011; Gosling et al., 2016), but not for birds (Petchey et al., 2007) and soil decomposer communities (Heemsbergen et al., 2004; Setälä et al., 2005), showing a high degree of functional complementarity. Furthermore, high functional redundancy does not necessarily equate to high resilience if species within functional groups respond in a similar manner to disturbance, as shown for e.g. coral reefs (Bellwood et al., 2004; Nyström, 2006) where the disappearance of a single key species can impact the entire ecosystem (Lessios et al., 1984).

It seems that examples on functional redundancy can be found working both ways. For ERA this entails that, if no evidence for a specific service providing group exists, it cannot be excluded that ecotoxicological effects on particular species may be compensated for by other species taking over, and ES provision could be less hampered than the EPF predicts. The narrower an EPF is quantified for a potentially wide group of organisms composing a SPU, the greater the error in the assessment induced by ecosystem service resilience. An assessment of the degree of response diversity and functional redundancy that can be expected would be helpful additional information in ES-based risk assessment.

3.5. Spatial aspects of ES-based ERA

When using ecological production functions in ES-based ERA the aspect of spatial variability and validity of EPFs needs consideration, and two aspects in particular deserve attention. First is the uncertainty of generalisation: studies conducted in one climatic zone may have limited validity for another, and the same may be the case for translation between highly different landscapes. For instance, the regulating effect of trees and other green infrastructure on climate regulation and air filtering is not comparable for urban areas and their surrounding landscapes, and even needs fine tuning at smaller scales (Pincetl, 2010; Pataki et al., 2011). Risk assessment needs to take a pragmatic approach here and use available knowledge while accounting for uncertainty. We have indicated the geographical region of research in the description of the EPFs (Table SI-1).

Ecosystem service case studies have a strong tendency to be situated in countries with a very high human development index, bringing about a relatively narrow knowledge base in large parts of the world and a blind spot for low-developed countries in which societies depend much more on ES than in higher developed countries (Lautenbach et al., 2019).

A second important aspect in ES-based risk assessment of chemicals is the need to complement the quantification of the contribution of the SPU to the provision of the ES with an assessment of the area from which the service is derived (Seppelt et al., 2011), termed service production areas (Fisher et al., 2009) or surface providing areas (Syrbe and Walz, 2012). 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 can be determined. Essentially, the flow of ES in terms of use by local stakeholders or society at large is an in part spatial aspect that determines actual provision in comparison to potential provision (Hein et al., 2016), essential for determining the flow of revenues and their valuation. When protecting habitats where services are used, the supply of services reliant on mobile taxa moving between habitats should be considered. Thus, the use of combinations of habitats and taxa as SPUs have been recommended when informing ecosystem management and conservation (Culhane et al., 2018).

3.6. Economic valuation

In environmental risk management cost-benefit analysis, evaluating the costs of different risk management scenarios compared to their benefits, is necessarily informed by risk assessment considering the reduction in value of the ES benefits as a result of the impact of a chemical stressor, and the likelihood of expected outcomes of alternative scenarios. A great challenge for successful valuation of ES is to integrate studies of the ecological production function with studies of the economic valuation function (Tallis and Polasky, 2009). The main issue here is that the ecological production function needs to exist so the physical flow of the service can be quantified. Once quantified this can be valued, either from primary research or providing there are studies that exist from which a 'benefits transfer' can occur. This requires that the definitions of ecosystem goods and services match across studies, otherwise the results of ecological studies cannot be carried over into economic valuation studies due to risks of double counting or incompatibility of ecological and economical units of service provision and valuation. Attempts to value ES without this key link will either fail to have ecological underpinnings or fail to be relevant as valuation studies (NRC, 2005). The ES approach has the potential to facilitate evaluation of the costs and benefits of chemical use since it allows trade-offs to be made between different ES under different levels of chemical exposure. Trade-offs may be assessed directly for chemicals (e.g. pesticides) that are applied in a defined area of the landscape (e.g. in a field or orchard) to increase crop production, than for other chemicals (e.g. consumer chemicals or pharmaceuticals), which are used in domestic situations and subsequently released into the environment downstream (Maltby et al., 2017, Maltby et al. subm.).

Some scholars have argued that monetary value should not be the only metric of ecosystem services within a defensible framework, in part because we do not yet have approaches to give monetary value to all relevant services (Shackleton et al., 2017) and such a framework would thus be incomplete (Norgaard, 2010). We consider economic valuation useful because it is easily understandable by society and is a common unit that allows for simple summing of service bundles and providing a net value when comparing management options (Dodds et al., 2009; Birch et al., 2011), and is seen as essential for decision making (NRC, 2005).

Service provision may fluctuate in time and space depending not only on ecological fluctuations but also on economic and demand variability in the supply chain. Taking pollination as an example again, and disregarding ecological variation for sake of the argument (but see for a quantification e.g. Garibaldi et al. (2015)), EPFs quantify potential ES production and may help to assess ES physical flows as far as the farm gate. However, farm gate prices represent only a part of the final price paid by consumers and secondary consumers, and are in turn influenced by total market supply and demand. If pollination services were completely lost and fruit production would decline, farm gate prices may increase dramatically, reducing farmer losses, but causing a decrease in consumer welfare as increased costs would be passed on in the consumers chain.

Benefit assessments will be highly sensitive to shape (linear vs. nonlinear) of response functions in ecological outcome induced by an ecosystem stressor (Wainger and Mazzotta, 2011), and therefore can be expected to require EPFs that also have a well-defined shape. Simple factorial EPFs may thus introduce some error. Seven publications presented such factorial EPFs, out of the 31 predictive EPFs that we have retrieved from the literature search which come along with economic valuation of the provided ES. Another seven involved regression type relationships based on field experimentation with the service providing taxa and subsequent valuation of the service provided. Furthermore, we found 17 modelling studies quantifying an EPF and providing calculus for economic valuation of the associated ES (Table SI-1). Thus, predictive EPFs with associated economic valuation were compiled for nine ES, with multiple EPFs particularly found for air filtration of fine

dust particles and contaminants and carbon sequestration by urban trees (CICES 2.1.1.2), pollination by insects (2.2.1.1), coastal and riverine nursery populations and habitats for fish and crustaceans (2.2.2.3), and pest control by soil dwelling arthropods and birds (2.2.3.1).

We can thus conclude that economic models necessary to link an ecological service flow to a change in ecosystem function are not common. The scarcity of clear methods for accomplishing the modelling of ecosystem effects and the consequent economic implications does however not imply that ecological impacts with no immediate consequences are of no economic concern. This is a major conceptual barrier in the quantification of economic benefits associated with nutrient cycling, water infiltration, biological diversity, and provision of habitat (Heninger, 2007).

From an economic perspective linking functions to service flow is tricky. Many valuations of benefits/final services have the processes and functions implicit in the valuation. But directly valuing the processes/functions and biodiversity is usually avoided.

Because the concept of ecosystem services is by definition anthropocentric, beneficiaries or users must be spatially connected to regions providing a service for that service to have value, with the exception of global services like carbon sequestration, or some non-use values. Research on spatial discounting has shown ecosystem service values to decline as distances between ecosystems and their beneficiaries increase (TEEB, 2010). Whereas most such analyses have used Euclidean distance to a resource, a more correct approach might be to spatially account ES values using service-specific flow paths. By not considering the location of beneficiaries relative to ecosystems, some ES values may be substantially overvalued (TEEB, 2010).

3.7. Criteria for EPFs

The utility of production functions has recently been discussed and a shortlist of nine "desirable attributes" was developed (Bruins et al., 2017). In our literature searches we have not strictly applied all of these attributes before including potential publications. Perhaps in contrast to the desired attribute that an EPF should estimate indicators of final ecosystem services, we also incorporated services in the CICES division of 'Regulation and maintenance', including ES which may be considered 'intermediate' by some and 'final' by others, such as pollination, and control of pests and diseases in crops. For ERA we found that good data from standardised testing is available for service providing species in the prominent ES (e.g. pollination and pest control), and that adequate valuation of associated final services (crop yields) is also often available (Table SI-1) to inform environmental decision making.

A second desirable attribute by Bruins et al. is quantification of the relationship between service and service provider. Although EPFs that yield qualitative outcomes can sometimes be useful for scoping and mapping, quantification is needed for the analysis of ES trade-offs. For our review we have only selected publications that report some quantification of the EPF, providing at least an assessment factor by comparing presence vs. absence or natural vs. elevated density of the SPU. In addition, as correlated observations do not necessarily imply causality, EPFs should preferably not be based on correlations but rather on suitable controls and confirmatory experiments. Although we included correlative relationships and assessment factors (O and F_o in Table SI-1), we consider that these have little predictive value in prospective ERA and did not count them as EPFs. Confirmation of causality by experimentation comes at the expense of intricate quantification of the EPF relationship between SPU and ES: most experiments have only a single treatment of manipulating the SPU (presence or absence). Multi-site field studies on the other hand may significantly increase the range of variation in SPU presence or activity, but generally fail to demonstrate mechanistic causality underlying the obtained relationship to ES provision by direct observation. Together, however, the two approaches may provide more robustness in quantity and quality of the evidence, e.g. as demonstrated in the assessment report of the Intergovernmental

Science-Policy Platform on Biodiversity and Ecosystem Services on pollinators, pollination and food production (IPBES, 2016).

3.8. Prospects and challenges for use in ERA

In prospective ERA, the limited availability of EPFs hampers a quantitative risk assessment of chemical impacts on most ES, trade-off analysis and economic valuation. Nevertheless, there is considerable scope to use the ES approach conceptually, in the first instance to explore how SPUs may be impacted by chemicals based on declining abundance or biomass, and how loss of capacity in functional traits is likely to affect ES delivery. In combination with the findings of this review, such conceptual studies would help to determine which ES are likely to be most vulnerable, and for which the development of EPFs are most needed. We have outlined some of the complexities in developing, applying and interpreting EPFs, which seems to be an overwhelming task. The pragmatic way through this, at least for use in prospective ERA, would be to adopt simplifying assumptions and landscape models (scenarios) that enable a generalised approach.

In retrospective ERA, the site-specific ES of concern are relatively easily established from management plans and stakeholder interviews, and relevant site-specific SPUs can be conceived to be transparently "translated" into ecotoxicological endpoints (Faber, 2006; Faber and van Wensem, 2012; Maltby et al., 2018), thus providing a meaningful and acceptable basis for assessment in the ERA. Currently, therefore the application of the ES concept on the basis of EPFs seems to have good potential for retrospective ERA and environmental quality assessment, as demonstrated in a case study extrapolating WFD endpoints monitoring data using functional trait data to link ecological status (based on ecological structure) to potential ecosystem service delivery (Brown et al. subm.). Good examples for population-based modelling of service provision on the basis of single-species test toxicity data are also forthcoming, e.g. for pollination (Van den Brink et al. subm.). These two case studies have been discussed in a multi-stakeholder workshop to evaluate the feasibility of adopting an ecosystem services approach to chemical risk assessment, and workshop participants concluded that there was added value in adopting an ecosystem services-based approach for regulatory decision making (Maltby et al. subm.). The quantitative linkage between data from standardised toxicity testing, ecological functioning of SPUs, and consequent provision of ES was recognised as a research priority to which our current paper aimed to contribute. Numerous knowledge gaps remain however, some of which are bulleted in Section 3.9.

The good news for ES-based ERA is that for all ES classes EPFs are available to cover at least some part of the trajectory to facilitate extrapolation, with varying degree of quantification and causality, and with varying degrees of taxonomical and ecological relevancy to available standardised tests. Quantification of this uncertainty obviously would be useful for risk managers, and the range of this uncertainty may be assessed in dedicated studies comparing the ES cascade range and array of taxonomical resemblance in the EPFs currently available. Already an EPF approach to assessing risk to ES delivery would seem more transparent than a black-box world of expert judgement.

Thus, there are three main challenges which need to be addressed to enable the implementation of ES approaches in ERA and environmental risk management: (i) The limited number of standardised test species and endpoints compared to the naturally occurring species assemblages, SPUs and ES. This includes an assessment of uncertainty from a pragmatic use of related taxa; (ii) Lack of EPFs linking SPU to final ecosystem service provision; (iii) Lack of quantitative evidence linking ecological production functions with economic valuation functions. Given the need felt by risk managers and policy makers for better evaluation and prioritisation of costly measures for environmental remediation and conservation (Wong et al., 2015), these limitations need to be addressed with urgency.

3.9. Knowledge gaps

- Important SPU taxa missing or underrepresented in standardised testing, whilst important for provision of ES: archaea, fungi, corals, and bats.
- EPFs covering the complete range from standard testing species up to ES valuation are unavailable (with the exception of honeybee).
- Generic extrapolation of production functions from site-specific ecosystem types and SPUs to other locations is uncertain and in need of validation.
- The potential for generic extrapolation of economic valuations from spatially defined studies seems limited, as ES flows and demand will differ spatially; relevancy seems linked to smaller scales, and extrapolation to higher scale needs uncertainty assessment.
- ERA and management assume recovery entails a return to baseline conditions which existed before chemical impact; however, specific responses may divert from generalised models in idiosyncratic ways. Illustrations come from fisheries and coastal eutrophication ecological monitoring (Nørring and Jørgensen, 2009). EPFs may not be consistent in deteriorating versus recovering ecosystems and should be used with some caution to predict system recovery after remedial actions.

4. Conclusions

A. Ecological production functions were retrieved for all CICES V5.1 classified biotic provisioning and regulating ES classes, but suitability for application in chemical ERA varied considerably. Direct linkage to standardised tests is limited, as for most taxa in EPFs no standardised tests exist. However, for most organism groups relevant tests may be selected for related taxa at higher taxonomic levels. Of the relevant literature, just a single study involving honeybees (*Apis mellifera*) extended over the whole range of extrapolation steps from standard test species to final ES delivery. This implies that for use in prospective ERA insufficient EPFs are available for straightforward application on the basis of existing toxicity data obtained from standardised testing. Nevertheless, for most CICES classes EPFs are available to cover at least some part of the pathway to facilitate extrapolation of ES provision, with varying degree of quantification and causality. The gap notwithstanding, we consider that EPFs can be used in these cases, if considering extrapolation uncertainty regarding taxonomic relatedness and variation in spatial and temporal variation in SPUs (and ecological interactions), as well as regarding the number of models or assumptions employed in sequence for the assessment using stepwise logic chains (Hayes et al., 2018). Extrapolation between taxa is already common practice with guidance for risk assessment in existence, and the spatial and temporal variability in ecological functioning of species is relevant to be considered in relation to ES provision. Here the risk scenario describing when and where ES are to be provided determines the need for specific EPF data, and thus the size of the knowledge gap; uncertainty will be smaller and more clearly definable the smaller the spatial scale of risk assessment and with more precise description of ES management objectives.

B. The extrapolation from ecological functioning to economic value is best done with knowledge of both the ecological production function and the economical valuation function. Contributing to service provision is not necessarily 1:1 equivalent to increasing economic value. The wild pollinators versus honeybees' examples clearly demonstrated ecological and economical differences in efficiency and effectiveness, where wild bees in various crops promoted yield quantity and quality beyond the capacity of domesticated bees.

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Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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