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Upscaling of ecosystem service and biodiversity indicators from field to farm to inform agri-environmental decision- and policy-making

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ABSTRACT

Ecosystem services and biodiversity are frequently measured by field-scale indicators. Yet, many important agricultural and economic drivers as well as agri-environmental policies operate at larger sales, to which field-scale indicators first need to be upscaled. Therefore, this perspective is focussed on upscaling approaches from field to farm or to landscape scale. To understand how ecosystem services and biodiversity are affected by farm-scale drivers and to inform future decision- and policy-making while exploiting existing data sources, these need to be upscaled and analysed at farm scale. However, how this is done best for different types of indicators for ecosystems services and biodiversity received little attention so far.

In this work, we propose and discuss different options for upscaling ecosystem service and biodiversity indicators from field to farm scale. We base our novel conceptual work on a large body of literature and demonstrate that before deciding on an upscaling approach, different aspects of the indicators and the purpose of the assessment need to be considered. Our propositions start at the point where field-scale data is available for aggregation at farm scale. Such an aggregation needs to consider the relationship between ecosystem service supply and the benefit provided, i.e., the supply-benefit relationship, which describes how a change in supply affects the resulting benefit for farmers and/or society. We argue that this relationship can also be conceptualized for biodiversity, with benefit being the value of a field or farm for biodiversity conservation.

Because benefit does often not continuously increase with supply, but can exhibit breaking points defined by thresholds in supply, the shape of the supply-benefit relationship varies among different ecosystem services and biodiversity components. For example, for upscaling biodiversity indicators, a conservation value needs to consider that conservation benefit might non-linearly change with supply, i.e., habitat quality and quantity, and becomes marginal below certain thresholds. Only when such potential thresholds are considered, a suitable upscaling approach can be chosen from the approaches that we present in this work. While some indicators can be upscaled using a simple area-weighted total or average, for others, thresholds in supply are of great relevance for determining the best upscaling approach. We conclude that upscaling indicators to the farm scale holds untapped potential to inform agri-environmental assessments and future policies. By presenting and discussing suitable approaches for different types of indicators, we hope to facilitate upscaling as a tool to support agri-environmental decision-making in the future.

1. Introduction

Ecosystem services and biodiversity are deeply interwoven and closely connected to human wellbeing and planetary health, but at the same time they are heavily impacted by human activities, such as land-use intensification and land-use change (e.g., Dainese et al., 2019; Haines-Young and Potschin, 2018). Thus, in many countries, core policy targets are to secure ecosystem services and to protect biodiversity in agriculturally managed landscapes (e.g., Schipper et al., 2020; Zulian et al., 2013). Yet, the efficiency of many environmental and agricultural

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policies has been questioned (e.g., Pe'er et al., 2020). A key element in designing efficient policies is to dissolve spatial mismatches between underlying drivers and targeted environmental outcomes, here biodiversity and ecosystem services (Früh-Müller et al., 2019; Hein et al., 2006; Pelosi et al., 2010). While biodiversity and ecosystem services are often assessed via indicators at the *field scale* (Garland et al., 2021; Richter et al., 2021), key drivers such as agri-environmental policies and decision-making frequently operate at the *farm* or *landscape scale* (Belfrage et al., 2005; Huber et al., 2022; McDowell and Kaye-Blake, 2023). To overcome this spatial mismatch, field-scale indicators need to be upscaled to entire farms or landscapes to enable assessing and evaluating all relevant drivers (Fig. 1).

A spatial upscaling is, however, far from trivial as it needs to account for different methodological, agronomic and ecological aspects, as will be discussed in this work. Technically, upscaling consists of two steps. First, indicators are modelled for all fields of a farm or landscape, which is usually based on indicator measurements in a subset of fields (e.g., Le Clec'h et al., 2018; Maes et al., 2012; Neyret et al., 2023; Tasser et al., 2008; Willcock et al., 2023). This modelling step has received considerably more attention than the second step of the upscaling procedure, i.e., aggregating field data at the farm or landscape scale to inform about the respective "total" in ecosystem services or biodiversity. Therefore, our considerations start at the point where the data for all field-scale indicators is already available for aggregation. We defined a *field* as a contiguous area being homogeneously managed and without large variation in environmental factors and plant communities, which would otherwise require a stratified sampling according to, for instance, soil types.

In the following, we highlight the often-overlooked importance of the farm scale for ecosystem service and biodiversity assessments (section 2), discuss key considerations required for choosing an upscaling approach (section 3), and propose three such approaches to upscale different indicators to the farm scale (section 4). We base these novel conceptual propositions on a large body of literature and, finally, discuss policy implications (section 5) as well as remaining challenges and limitations (section 6) related to upscaling. The aim of our perspective is to facilitate upscaling of different types of indicators to larger spatial scales to support farm- and landscape-scale studies on drivers of biodiversity and ecosystem service change, and to improve the evidence base for future agri-environmental policy-making. While the focus of the work is upscaling indicators from the field to the farm scale, to enable whole-farm assessments, the approaches we propose can also be used for upscaling to the landscape scale. When referring to biodiversity (indicators), we include all components such as species and genetic diversity as well as habitats.

2. Relevance of the farm scale for biodiversity and ecosystem services

Most agricultural and environmental policies specifically target one spatial scale, with the difficulty to comprehensively account for the main drivers of ecosystem services and biodiversity increasing from field over farm to landscape scale (Fig. 1). To date, studies on the effects of policies, land-use practices and environmental drivers on ecosystem services and biodiversity are frequently performed at field (e.g., Fischer et al., 2010; Allan et al., 2015; Wittwer et al., 2021) or landscape scale

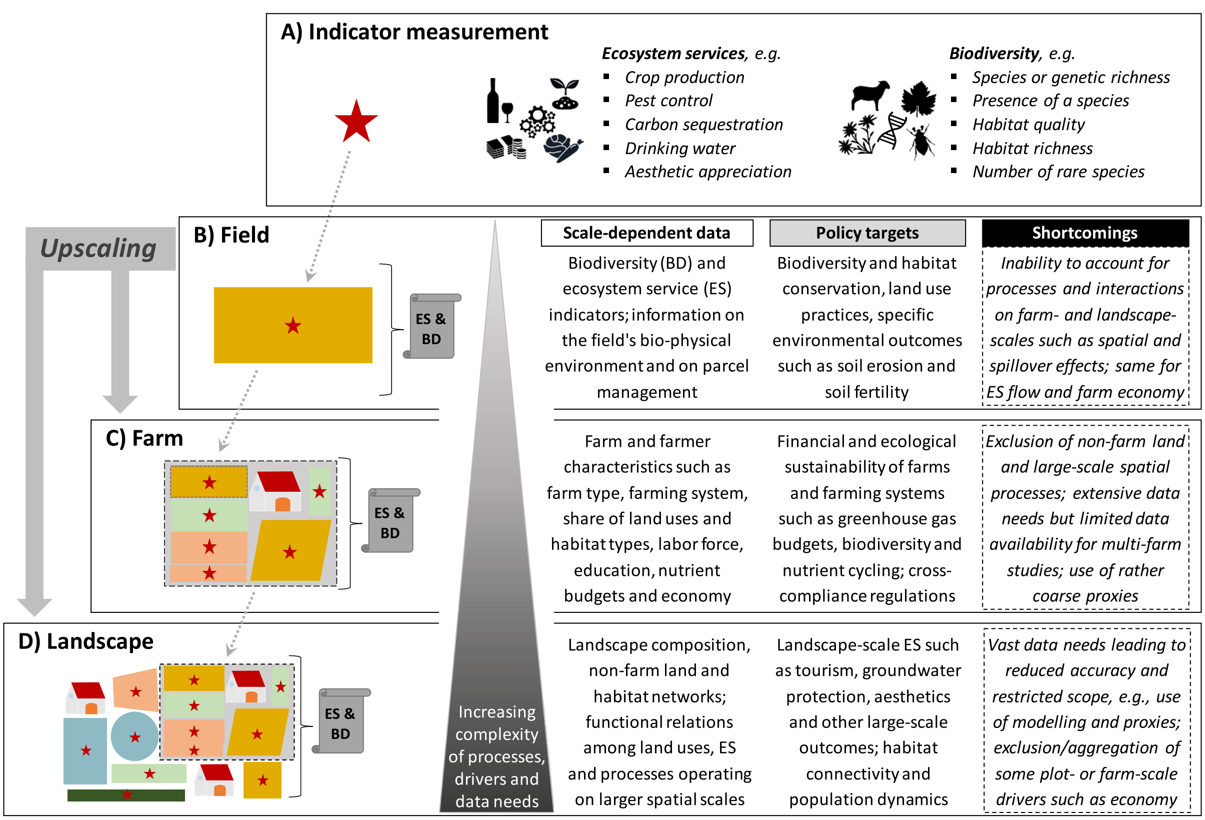

Fig. 1. Conceptual figure of indicators (A) for ecosystem services and biodiversity, which are evaluated across three different spatial scales, i.e., (B) the *field*, (C) the *farm*, and (D) the *landscape*. We provide examples of scale-dependent data, major policy targets and most relevant shortcomings of assessments at the respective spatial scale. For readability, further dimensions of biodiversity and ecosystem service assessments such as transects and off-site services are not displayed. In most cases, before data can be aggregated with the upscaling approaches presented in this work (section 4), modelling tools are needed to predict indicators for each parcel of the respective farm or landscape. Note that the fields of a farm, i.e., the spatial scale this work is focussed on, are usually not grouped together but dispersed across the landscape, increasing the relevance of landscape configuration and of neighbouring land use as co-drivers of biodiversity and ecosystem services. Icons: © Agroscope.

(e.g., Fahrig et al., 2011; Jeanneret et al., 2021). Yet, despite their mechanistic and ecological relevance, the field- and landscape scales fail to account for several drivers and policies that operate exclusively at the farm scale, such as regulations, cross compliances and direct payments (Fig. 1; Huber et al., 2022). The farm also depicts the key unit for understanding the effects of farm planning, structure, management, economic performance as well as of farming systems, land-use changes, and several social factors on ecosystem services and biodiversity (Heinze et al., 2022; Kuhn et al., 2022; McDowell and Kaye-Blake, 2023). Farm-scale studies are further needed to inform about environmental impacts of farm activities that degrade or restore natural capital stocks and modify their ability to produce ecosystem services (Byrne et al., 2007). Moreover, studies at the farm scale are required to assess benefits of whole-farm strategies such as crop diversification, agroecological measures and organic farming (Jeanneret et al., 2021; Schneider et al., 2014), while accounting for the (re-)distribution of management types and intensities across the farm. Thus, to comprehensively understand environmental changes and their relations to socio-economic drivers, it is crucial to include the farm scale in ecosystem service and biodiversity assessments.

Research has identified many different farm-scale drivers to be potentially relevant for single or multiple ecosystem services as well as biodiversity (e.g., Birrer et al., 2014; Herzog et al., 2017; Kuhn et al., 2022). For example, Dalgaard et al. (2011) found the farm scale to be highly relevant to accurately predict and understand landscape-scale nitrogen losses because of considerable variability among different farming systems and non-linearities between the losses and their farm-scale drivers (e.g., livestock densities). Such a case cannot be fully understood with only a landscape-scale analysis. When analysing farmland biodiversity, Belfrage et al. (2005) found prominent farm-scale interactions between production system (organic versus conventional) and farm size. Moreover, studying non-provisioning ecosystem services of Swedish farmland, Karlsson et al. (2022) found farm type and size to significantly relate to the cultural services provided by the farms. In both studies, small farms were situated in more diverse landscapes with more diverse pedo-climatic conditions compared to large farms, highlighting the complex interactions between farm- and landscape-scale factors. Clough et al. (2020) identified ecological-economic trade-offs between farm economy and farmland biodiversity to be mediated by field size, a previously largely unknown dependency. These examples shown that farm-scale assessments are needed to inform policymakers, farm managers and researchers, but also the public, about farm performance and the agricultural, ecological and socio-economic drivers determining ecosystem services and biodiversity (Malinga et al., 2015; von Haaren et al., 2012). However, despite their great relevance, farm-scale studies and evaluations are still comparably rare, especially when more than a few farms and/or more than a few ecosystem services are to be included (Heinze et al., 2022; Vidaller and Dutoit, 2022).

While specific approaches for estimating farm-scale biodiversity and ecosystem services exist, these often rely on rather coarse proxies such as farming practices and mappings of habitats or land-use types (e.g., Birrer et al., 2014; Jeanneret et al., 2014; Lüscher et al., 2017; Quinn et al., 2013; von Haaren et al., 2012). Many such farm-scale approaches are not designed to include precise field-scale measurements such as of species' presence, species abundance, or local biophysical indicators (Kuhn et al., 2022; Marais et al., 2019; Tasser et al., 2019). The same consideration applies to the landscape-scale mapping of ecosystem services and biodiversity, which mostly relies on models using land cover data and similar spatially explicit information (Schipper et al., 2020; Zulian et al., 2013). Although data availability is still limiting comprehensive biodiversity and ecosystem services assessments (Robinson et al., 2023), significant data on field-scale indicators are already available from large-scale surveys. Yet, this data first needs to be upscaled to larger scales to serve comprehensive policy evaluations and farm comparisons. Information on field-scale indicators is, for example, gathered by national surveys and environmental monitoring programmes such as the LUCAS soil sampling of the European Union (Orgiazzi et al., 2018), national biodiversity monitoring schemes (e.g., Herzog and Franklin, 2016; Nielsen et al., 2012), citizen science projects (e.g., Sullivan et al., 2009), and long-term research initiatives (e.g., Fischer et al., 2010). To make use of these data sources for farm-scale studies, suitable upscaling approaches have to be employed.

3. Considering supply and benefit to choose a suitable upscaling approach

Upscaling indicators from field to farm scale faces the challenge to define an upscaling method that is informative and reliable according to the indicator(s) and the design of the assessment. A key consideration is how *benefit* is driven by *supply* (Fig. 2). At this, *supply* is the ecosystem service or biodiversity indicator of a field. The *benefit* of an ecosystem service is defined as its contribution to human wellbeing based on the societal demand for that service (Wolff et al., 2015). For biodiversity, benefit can be considered the net contribution of a piece of land to nature conservation such as providing suitable habitat for an organism or population.

For upscaling, information on field-scale supply is first used to calculate field-scale benefits, which are then aggregated at the farm scale (Fig. 2). A key aspect of upscaling biodiversity and ecosystem services is that non-linear, potentially abrupt changes in benefit can occur depending on supply. The *supply-benefit relationship* is a mathematical function that describes how much of a benefit is provided by a certain rate of supply (Manning et al., 2018). Benefit does often not continuously increase with supply, meaning that *one or more thresholds in supply* can strongly impact benefit (Fig. 2). For ecosystem services, the supply-benefit relationship can refer to, for example, how (a) farmers benefit from different crop yield qualities, (b) differences in soil quality determine the production potential of a fields, and (c) people value the aesthetic quality of different crops.

Two types of thresholds in supply-benefit relationships can be conceptualized, i.e., (i) thresholds in quality, and (ii) thresholds in quantity of ecosystem service supply. A *threshold in quality* refers to the condition of what is supplied, expressed as, for instance, chemical composition. For example, protein content depicts an indicator of crop yield quality and is subjected to thresholds regarding minimum protein content determined by quality standards, such as for feed wheat versus baking wheat. A threshold in *quantity* refers to how much is supplied and if an increase in the amount is constantly causing an increase in benefit. Quantity thresholds come into play when, for example, minimum supply is needed or demand is exceeded (Wolff et al., 2015).

We argue that such supply-benefit relationships can also be conceptualised for biodiversity regarding the conservation benefit of a piece of land such as for habitat quality or species diversity. Data on the quantity and quality of biodiversity indicators can therefore be combined to comprehensively assess biodiversity change, such as with the Biodiversity Change Index, in which *quantity* is the area of a specified habitat type and *quality* is the abundance of (indicator) species and/or other habitat quality parameters (Normander et al., 2012). At this, a threshold in the habitat quality of a field can occur, linked to a measure of suitability for a species to establish. Conservation benefit can thus be zero for a specific species if, for example, land-use intensity exceeds a critical level (Busch et al., 2019). A threshold in quantity relates especially to the size of a habitat, because the benefit of a field for nature conservation decreases non-linearly with its size and gets too small to support a viable population of a species below a certain threshold (Hylander and Ehrlén, 2013). As the vital needs of a taxon do not linearly relate to (a reduction in) area, below a certain size the field loses all its value for conserving this species. While the relevance of minimum population sizes for biodiversity conservation is long known (Shaffer, 1981), species-specific thresholds in habitat area are only available for few species of which most are large mammals or birds (Fahrig, 2001; Huggett, 2005; van der Hoek et al., 2015) an depend on, for example,

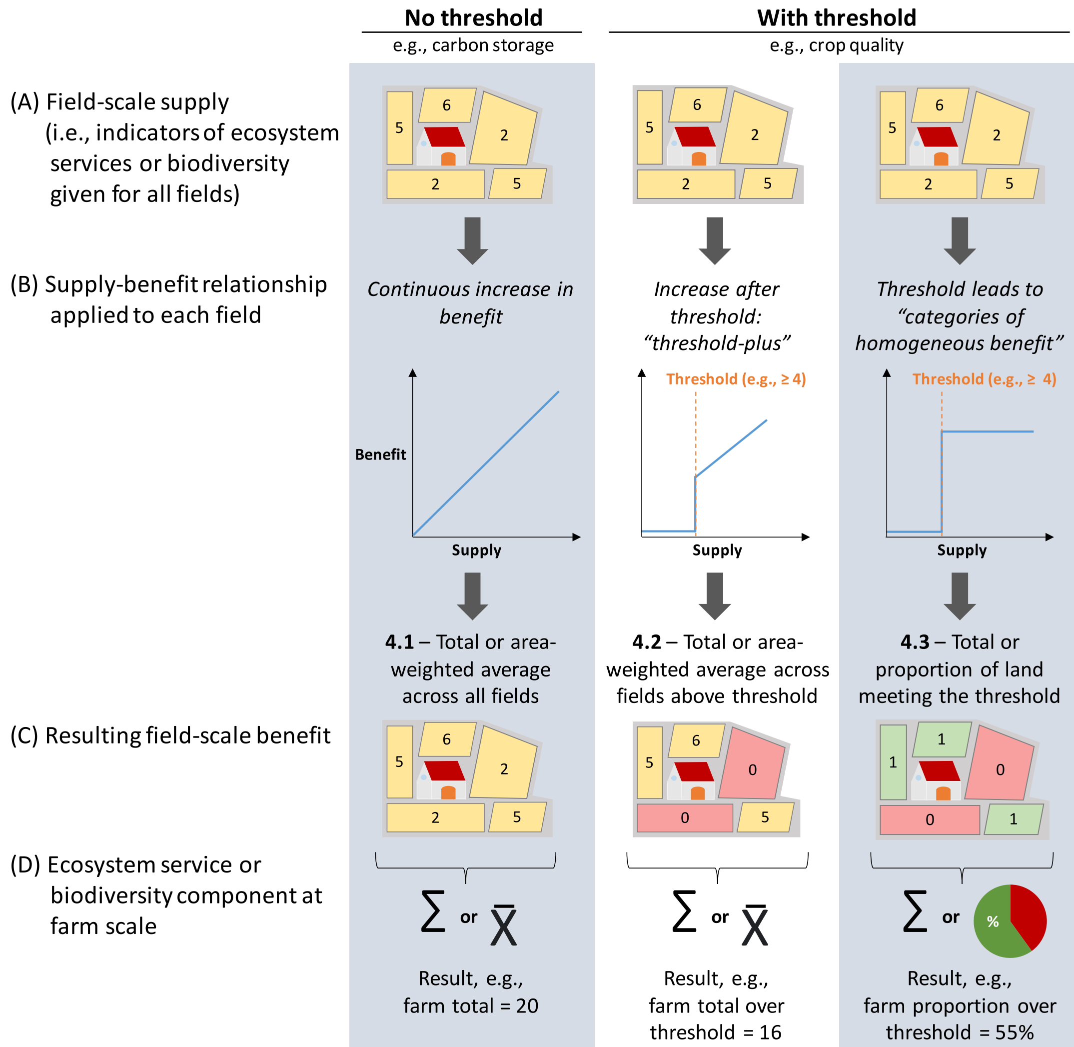

Fig. 2. Upscaling of indicators for ecosystem services and biodiversity. Depending on (A) field-scale supply, the (B) supply-benefit relationship and respective thresholds (section 3; following Manning et al., 2018) determine (C) field-scale benefit, which is used to calculate the (D) farm-scale ecosystem service or biodiversity component. Different approaches to calculate and present ecosystem services at farm scale are shown, with numbers 4.1 to 4.3 linking to the respective sections in the text. In these examples, the relationships shown for no-threshold (4.1) and increase after threshold (4.2) are linear, but they can have any other shape of a continuous increase (e.g., quadratic or logarithmic). Thresholds shown are for increasing benefit, but can also depict decreasing benefit, such as a concentration of a pollutant reducing water quality. Whether a total ($\sum$) or an average ($\overline{X}$) is to be calculated at farm scale relates to whether the outcome is meant to be affected by the size of the farm or if a per-area value is more informative. Note that an additional supply-benefit relationship can apply for the benefit at farm-scale (D), e.g., when farm-scale supply exceeds demand.

gap-crossing ability (Dale et al., 1994; Offerman et al., 1995). Thus, thresholds are likely highly context specific such as depending on the surrounding landscape (Kuussaari et al., 2009).

The quantification of a supply-benefit relationship is usually based on extensive data but can be derived from expert knowledge (Manning et al., 2018; Table 1). While the supply-benefit relationship is mainly used to transfer field-scale supply to benefit (Fig. 2A to C), a second supply-benefit relationship might need to be taken into account at the scale of the whole farm (Fig. 2D). The latter applies when, for instance, farm-scale supply exceeds demand or when farm-scale habitat area undercuts a minimum conservation threshold. Although we do not further discuss farm-scale supply-benefit relationships here, these operate basically in the same way as the field-scale relationship and its potential thresholds.

Thresholds in supply can be of ecological, economic or any other kind, potentially representing an ecological mechanism or a subjective stakeholder preference, which highlights the socio-economic dimension of ecosystem services (Cinelli et al., 2014). Ideally, all thresholds in the supply-benefit curve that determine the benefit of the ecosystem service according to its actual utilisation need to be known. Therefore, interviews with the actual beneficiaries of an ecosystem service might be needed to gather information about the (normative) thresholds in action, linked to stakeholder preferences (Neyret et al., 2023). This step also helps moving from a potential to a realised benefit as it links the actual demand and the flow of an ecosystem service to the actual beneficiaries (Jones et al., 2016). Yet, interviewing beneficiaries can be time-consuming and particularly challenging for benefits associated with the wider community and across larger spatial scales such as for climate services. Fuzzy logic can be used to statistically assess effects of ambiguous thresholds or group margins (Malczewski, 2006).

4. Upscaling approaches

The upscaling of field-scale ecosystem services and biodiversity indicators requires all fields of a farm to have values based on measurements or modelling approaches. Regarding modelling, several GIS-based

Table 1
Possible upscaling approaches for the most common (A) ecosystem services (examples follow the CICES framework; Haines-Young and Potschin, 2018) and (B) biodiversity components, using field-scale indicators to calculate a farm-scale outcome. The examples given are not exhaustive as for the supply-benefit relationship and the related upscaling approach different options might exist, which needs to be decided according to the aim of the assessment. Upscaling to the landscape scale is possible with the same approaches as for the farm scale.

(A)	**Ecosystem service** (examples from categories following CICES)	***Example of field-scale indicator*** *(unit)*	**From measurement to field**	**Supply-benefit relationship(s)** (different options may exist)	**Upscaling approach from field to farm** (different options may exist)
Provisioning	Drinking water	*Groundwater recharge (infiltration, mm per hour)*	Measurement applies to whole (homogeneous) field	Continuous	Area-weighted average
	Edible & medicinal plants	*Abundance of edible plants (% ground cover)*	Measurement applies to whole (homogeneous) field	Continuous or threshold	Area-weighted average (above threshold) or transformation into presence/absence above threshold
	Yield (quality)	*Protein concentration in crop grains (g per kg)*	Measurement applies to whole (homogeneous) field	Threshold plus or categories of homogeneous benefit	Average above quality threshold or transformation into quality classes
	Yield (quantity)	*Crop yield (weight per area)*	Multiplication with area	Continuous (threshold possible)	Farm total or average per ha
Regulation and Maintenance	Pollination	*Nectar provided by plants (weight per area)*	Multiplication with area	Continuous	Farm total or area-weighted average per ha
	Carbon storage	*Carbon stock in topsoil (per field)*	Multiplication with area	Continuous	Farm total or area-weighted average of all farmland
	Control of erosion	*Soil covered with vegetation (% ground cover)*	Measurement applies to whole (homogeneous) field	Continuous or threshold-plus or categories of homogeneous benefit	Several options such as % of farmland below/above threshold
	Pest control	*Abundance of weeds (% ground cover; reversed to approximate pest control)*	Measurement applies to whole (homogeneous) field	Continuous or threshold	Area-weighted average (above/below threshold) or % of farmland below/above threshold
	Regulation of soil fertility	*Available phosphate in topsoil (g per kg)*	Measurement applies to whole (homogeneous) field	Continuous or threshold-plus or categories of homogeneous benefit	Several options such as transformation into soil suitability classes
	Regulation of soil quality	*Soil suitability for arable farming (compound indicator; 1/0 or more levels)*	Measurement applies to whole (homogeneous) field	Presence-absence (or more categories)	% of farmland with presence of benefit (suitability)
Cultural	Aesthetics	*Rating of appreciation of crops by people (Likert-scale questionnaire data)*	Measurement applies to whole (homogeneous) field	Continuous or threshold	Area-weighted average or % of farmland below/above threshold
	Heritage, tradition and sense of place	*Signs of traditional management (e.g., historic hay barns or hay stacks)*	Measurement applies to whole field	Presence-absence (or more categories)	Total count per farmland, or average per ha, or % of parcels with presence
(B)	**Biodiversity component**	***Example of field-scale indicator***	**From measurement to field**	**Supply-benefit relationship(s)**	**Upscaling approach** (different options may exist)
Biodiversity	Genetic diversity	*Genetic variability in one species or a community*	Either no treatment (fixed sample size) or adjust to field area via accumulation curve	Continuous	Area-weighted average or, ideally, using all samples of farm for calculation of genetic diversity
	Habitat quality	*Suitability for a species (compound indicator; 0/1 or more levels)*	Same value for whole field	Presence-absence (or more categories)	Presence-absence at farm or % farmland with presence (suitable for species)
	Habitat presence or richness	*Number of specific structures like habitat trees (per area)*	Measurement applies to whole (homogeneous) field, alternatively presence-absence of structure or habitat	Continuous or threshold-plus or presence-absence	Several options such as total or area-weighted average per ha, or % farmland with presence
	Presence of species (or species group or community)	*Presence of red list or indicator species (e.g., transect counts)*	Same value for whole field	Presence-absence (or more categories)	Presence-absence at farm, such as a farm-scale species list, or % farmland with presence
	Species diversity index	*Shannon index of plant species (per area)*	Same value for whole field	Continuous or threshold-plus	Area-weighted average (within homogeneous habitat types)
	Species richness	*Number of plant species (per area)*	Adjust to field area via species accumulation curve	Continuous or threshold-plus or categories of homogeneous benefit	Several options such as transformation into classes and % of farmland of each homogeneous category

practical tools are available to create ecosystems service maps such as with ESTIMAP, InVEST, ARIES, and Co$ting Nature (e.g., Maes et al., 2012; Zulian et al., 2013). In addition, prior upscaling, an indicator that is measured at one or more locations within a field might need to be adjusted to the area of the field (Table 1). If supply increases linearly with the field size, which is true especially for material ecosystem services such as carbon storage and crop yield, a measured indicator value can be easily adjusted to the area. For biodiversity, however, area often disproportionally affects supply. For example, a measured value of species or genetic richness might need to be adjusted to the area of a field using species accumulation or rarefaction curves (Gotelli and Colwell, 2001). If field-scale data is available, one of the following three sets of upscaling approaches can be applied: *area-weighted averaging or sum* (section 4.1), *threshold-plus* (section 4.2), and *categories of homogeneous benefit according to thresholds* (section 4.3).

4.1. Summing or averaging approach

A farm total or area-weighted average are the easiest approaches for upscaling indicators from the field to farm scale, which is acceptable when field size continuously increases benefit (though not necessarily linearly), when even low supply is valued (no minimum threshold) and demand is not exceeded (Fig. 2; Linders et al., 2021; Manning et al., 2018). If these conditions hold, the ecosystem service is delivered as a proportional mix of the total farm area, meaning a high performing area fully compensates for a low performing one (i.e., $0.5 + 0.5 = 0 + 1$). This situation can, for example, be assumed for soil carbon storage and crop yield of a homogeneous quality. Note that this supply-benefit relationship might have a linear shape but could also have a non-linear shape, such as logarithmic or quadratic (see Supplementary material 1A). If exceeding the demand leads to a stable upper limit (maximum) of benefit, a maximum value for benefit could be set to account for the point at which benefit does not further respond to increases in supply. These points highlight the need to first calculate benefit at the field scale and then average or sum these single-field benefits at the farm scale (Fig. 2)

We present an example of the calculation of benefits from supply using y_i and b_i to indicate the supply and benefits per field i, respectively. We consider y_i to be adjusted to the area of the field, e.g., by area weighting or rarefaction. Moreover, we give equation (1) for a linear supply-benefit relationships.

The field-scale benefit, b_i^{cont}, for a continuous increase with increasing supply is given by:

$$b_i^{cont} = y_i \rho \tag{1}$$

where ρ is some constant conversion coefficient. After field-scale benefit is calculated, single-field benefits can be summed or a per-hectare average per farm can be calculated to derive a farm-scale outcome. See Supplementary material 1B for formulas related to the upscaling approaches.

Some components of biodiversity can also be represented using this approach. For example, count data of single species and habitat structures can be used for a farm total or per-ha average value. Theoretically possible though often not recommendable, biodiversity data of areas of a fixed size such as for species or genetic richness or a diversity index can be averaged across the farm or a specific type of habitat (Table 1). However, when calculating area-weighted average richness (Tasser et al., 2008), high diversity areas are averaged with low diversity areas resulting in a suboptimal representation of the actual biodiversity benefit driven by the single fields (i.e., $0.5 + 0.5 \neq 0 + 1$). This consideration highlights the advantage of implementing thresholds for biodiversity data, which better accounts for actual single-field benefits.

4.2. Threshold-plus approach

A threshold-plus relationship is characterized by a break-even point with no or low benefit below but higher and continuously increasing benefit above the threshold (Manning et al., 2018). This relationship is a frequent case for environmental features subject to a minimum demand and regulatory thresholds such as for crop quality or when dynamic shifts in demand can occur such as for recreation or touristic value (Fig. 2). Similarly, the biodiversity conservation benefit of a piece of land can become zero if a given threshold in an indicator for habitat quality is undercut, becoming unsuitable for a certain species. Yet, after the threshold, habitat suitability and thus conservation benefit might further increase (Hylander and Ehrlén, 2013). Thus, from an ecological viewpoint, the threshold-plus approach can be applied to many biodiversity components that are affected by habitat quantity or quality. For all fields exceeding the threshold, a weighted farm-scale average or total can be calculated. Equation (2) shows an example of calculating field-scale benefits b_i^{tp}, for the threshold plus approach assuming a linear increase after a threshold x

$$b_i^{tp} = \begin{cases} y_i \rho, w_i \geq x \\ 0, w_i < x \end{cases} \tag{2}$$

If the threshold x is to apply to the area-weighted supply, then this consideration can use y_i instead of w_i. As mentioned before, the increase after the threshold can be linear but also of any other shape (see Supplementary material 1A below in this document), and the in- or decreasing part of the relationship can theoretically be before or after the threshold. Because the *threshold plus approach* does not inform about how often the threshold has been met given all separate fields, upscaling an ecosystem service with such a supply-benefit relationship might thus not only rely on the total or average over threshold but can also require presenting the proportion of land not meeting the threshold (Fig. 2).

4.3. Categories of homogeneous benefit according to threshold(s)

When continuous data is subjected to thresholds that are highly decisive for the benefit obtained, such as crop yield quality or water quality thresholds (e.g., McCullock et al., 2014; Zahedi et al., 2017), this can be used to form separate categories of quasi-homogeneous benefit below as well as above the threshold(s). Thus, it can be most informative to reduce a metric indicator to a categorical variable with two to many different categories separated by the threshold(s) (Fig. 2; Langhans et al., 2014). For the example of crop quality, a farm might be comprised of extensively managed grasslands yielding energy- and protein-poor hay and intensive grasslands yielding energy- and protein-rich forage. Thus, calculating an average crop quality would ignore the actual benefit of each separate type, as often yields of different qualities are not be mixed but used for different purposes, and thus sold at different prices (McCullock et al., 2014). In this case, calculating the share or total of the farm's land for each separate category is a suitable approach (Fig. 2), although it strongly reduces the original (metric) information provided by the indicator (Zhou et al., 2010), e.g., a measurement of crop quality, to an information on the occurrence of the respective categories of, e.g., distinct crop types, on the farm. However, when assessing smaller changes in the provision of a service, for example, quality corrected forage yields considering marginal changes of homogenous categories can also be straightforward

The field-scale benefits for the categories of *homogeneous benefit,* b_i^{hb}, for this approach can be calculated by:

$$b_i^{hb} = \begin{cases} a_i \rho, w_i \geq x \\ 0, w_i < x \end{cases} \tag{3}$$

This equation (3) considers one threshold, but following the same logic the equations can be extended to account for multiple thresholds. After evaluating the benefit of all fields, these outcomes can be presented by a total of land or a proportion of farm land that is above versus below threshold (Fig. 2).

The suggested approach of calculating categories of homogeneous benefit works especially well for thresholds that have a very strong impact on benefit, such as concentrations of pollutants for which legal thresholds exist, soil quality classes showing the suitability for growing different crops, and the ecological value of quasi-distinct habitat types such species-rich versus species-poor habitats (Quinn et al., 2013). When anyways deciding on the construction of categories, multi-criteria decisions to form these categories can be introduced to better characterise an ecosystem service or biodiversity component (Cinelli et al., 2014). As this better accounts for system complexity, multi-criteria assessments are likely to be more informative than the use of a single indicator, especially in a policy setting (Schils et al., 2022; Zahedi et al., 2017). For example, in the case of soil health with given thresholds in several chemical compounds such as heavy metals, the quality could be expressed in categories such as *excellent*, *good*, *medium*, or *poor quality* for each of the various compounds, and then be combined employing a suitable multi-criterion aggregation method (e.g., Langhans et al., 2014; Gan et al., 2017). While this approach depicts a clear simplification of the actual measurement data, it can be more informative when such multi-criteria thresholds are introduced for decision-making. Such multi-criteria assessments of ecosystem services involving a wide range of data types, potentially including qualitative data, require specific tools to be useful for decision-makers. Here, ordered weighted averaging with fuzzy quantifiers has been suggested as a GIS-based multicriteria evaluation approach (Malczewski, 2006).

5. Policy implications

The way ecosystem service and biodiversity indicators are scaled up has to align with the aim of the respective agri-environmental policy. At this, thresholds in the supply-benefit relationship are often particularly relevant to policy-making as thresholds can imply turning-points at which fundamentally different conclusions might be drawn (Braat and de Groot, 2012; Groffman et al., 2006). Thus, if an ecological or economic threshold is passed, a small change can have a large impact on the benefit derived, potentially translating into alternative management decisions, which can feed back on the future production of the ecosystem service. For example, farmers might “abruptly” change a field from intensive to extensive management (and vice versa) depending on whether policy thresholds can be achieved, such as a minimum number of indicator species required for a biodiversity-focussed payment scheme renumerating extensive field management (Elmiger et al., 2023). This highlights the importance to carefully set normative thresholds, especially since true ecological thresholds are often unknown or fuzzy (Huggett, 2005).

We further demonstrate the relevance of choosing an upscaling approach according to a policy target by giving two examples. First, experts are advocating for more results-based agri-environmental policies, which so far focus on field-scale outcomes like the number of occurring indicators species (Elmiger et al., 2023; O’Rourke and Finn, 2020). However, when policymakers want to compensate farmers for their overall ecological performance beyond the field scale, it is important how to scale up and aggregate respective indicators across the farm. As an illustration, policymakers may not intend to compensate two farmers equally although their average farm-scale “biodiversity score” is *medium* (score of 5), because one farmer’s fields might all score *medium* (5) while another farmer has 50 % of fields scoring *good* (10) but the remaining 50 % are *poor* (score 0). Given that ecological value and species communities supported will likely not be the same for both farms, payments based on a score derived by a farm average can potentially lead to unintended compensation. For instance, land-use intensity has often been used as an indicator or proxy for (mapping) ecosystem service supply and biodiversity at the field scale (e.g., Felix et al., 2022). However, it has been shown that for grassland farms, a combination of both intensive grassland (poor diversity but high yield) and extensive grasslands (high biodiversity but poor yield) was preferable to medium intensity management at the whole area, as this overproportionally reduces both yield and biodiversity (Nemecek et al., 2011). Therefore, using a farm-scale indicator derived from threshold or threshold-plus approaches is likely better integrating policymakers’ intentions and potentially reduces inefficiencies as compared to averaging. However, setting specific thresholds still remains up to the policymakers’ normative judgement (Hasund, 2011; Elmiger et al., 2023). The considerations of this first example apply not only to results-based payments but also to action-based payments and regulations, when field and farm management can be linked to the provision of ecosystem services and biodiversity at the farm scale.

Second, policies that do not account for the farm scale and the way ecosystem services and biodiversity are scaled up can encourage farmers to decrease management intensity on one field to be eligible for payments there, while intensifying management on another field to keep overall productivity constant (e.g., Graveline and Mérel, 2014). This can be intended, such as to shift intensive land use away from particularly vulnerable or valuable areas but keep farm income and production constant, or it is possibly not. Thus, for appropriately rewarding an environmental outcome the policy intention for that outcome needs to be well understood. If farmers are not allowed to compensate the lower management intensity, which was reduced in one field to achieve a specific environmental outcome at that location, with increasing intensity in another, a minimum threshold can be applied. This could safeguard a baseline outcome across all fields of a farm to assure the agri-environmental regulation or payment scheme leads to overall gains in the desired outcome. These applied examples show that aligning the upscaling approaches with the actual policy targets is highly relevant for successful policy-making.

6. Remaining challenges and limitations

Although the farm scale is relevant to understand several drivers of ecosystem services and biodiversity, its strength is limited by not being fully able to account for those aspects and drivers of ecosystem services and biodiversity that operate at the landscape scale. These landscape scale aspects include spatial processes like the pollination of crops from different farms, habitat networks, and some other connections between ecosystem services, biodiversity and functional drivers (Fig. 1; e.g., Cong et al., 2014; Duarte et al., 2018; Le Provost et al., 2023; Jones et al., 2016; von Haaren et al., 2012). Yet, tools to analyse spatial connectivity are developing quickly in recent times (e.g., Field and Parrott, 2022; Metzger et al., 2021). Thus, they might soon enable farm-scale assessments to better integrate landscape-scale processes, making use of spatial information such as neighbouring habitat types and land uses, as well as ecological connectivity measures. For instance, simple connectivity measures can be used as a criterium for the habitat quality of a field (Moilanen and Nieminen, 2002), and can be upscaled along the lines presented in this work. In addition, our upscaling approaches are versatile in that they can also be used to upscale field-scale indicators to the landscape and to any other spatially distinct unit. Yet, the approaches are not designed to include spatial connectivity measures that affect the outcome at the farm scale (as a whole).

The need to account for interactions with the surrounding landscape also concerns the thresholds in habitat quality and quantity that shape biodiversity conservation benefits of single and multiple fields (Moilanen and Nieminen, 2002). Due to the high context dependency and variability among species (Huggett, 2005; Kuussaari et al., 2009), it is challenging to define thresholds for habitat quality and quantity of many taxa. These would, however, be needed to assess current and potentially also future field- and farm-scale contributions to overall biodiversity conservation. Thus, prospective research can contribute to this by exploring the shapes of supply-benefit relationships for the many taxa for which there is currently not sufficient evidence-based information on habitat (network) requirements.

Another fundamental restriction for upscaling indictors to policy-relevant scales is the availability of data. Especially if multiple ecosystem services are to be studied such as for multifunctionality assessments (Allan et al., 2015; Le Provost et al., 2023; Neyret et al., 2023), there is rarely sufficient data, even if available field- and farm-scale data are combined. This aspect highlights (i) the value of large-scale environmental mappings such as with remote sensing and consistent and long-term *in situ* monitoring programs (e.g., Fischer et al., 2010; Kao et al., 2012; Pereira et al., 2022), (ii) the need for comprehensive databases, and (iii) improved modelling of ecosystem services and biodiversity (Robinson et al., 2023). To further advance models for mapping and upscaling, future research needs to work towards (a) improved spatial linkage of the condition of nature to actual human wellbeing, (b) increased resolution of the drivers and components of land-use plus other key drivers of ecosystem services and biodiversity change, and (c) better integration of functional trait-level biodiversity in ecosystem services mappings (Dainese et al., 2019; Rosa et al., 2020).

Enhanced availability of farm data via the digitalization of farm management is a promising future development (e.g., Nezamova and Olentsova, 2022). Yet, disaggregation (i.e., "downscaling") of such farm and landscape-scale data to single fields can, on the other hand, also be difficult.

7. Conclusion

Upscaling of ecosystem service and biodiversity indicators from the field to the farm scale is required for advancing sustainable development by, for example, informing and evaluating agri-environmental policies and payments, comparing farming systems, and guiding changes in farm management. The framework of upscaling approaches and associated options and solutions as discussed here is key to facilitate respective farm-scale assessments and to move towards a standardised methodological basis for comparable studies. Based on our considerations, future decision- and policy-making should increasingly make use of the insight provided by assessments of whole-farm biodiversity and ecosystem service supply. We hope to stimulate more studies to include a high number of ecosystems services and/or biodiversity components, as only such joint analyses can assess trade-offs and synergies between all involved aspects across spatial scales. Future research can further contribute to this development by exploring the shapes of supply-benefit relationships and especially the ecological, agronomic and socio-economic thresholds associated with biodiversity and ecosystem service supply, which will lead to an improved accuracy of upscaling studies.

CRediT authorship contribution statement

Valentin H. Klaus: Writing – review & editing, Writing – original draft, Visualization, Methodology, Formal analysis, Conceptualization. **Sergei Schaub:** Writing – review & editing, Writing – original draft, Methodology, Formal analysis, Conceptualization. **Robin Séchaud:** Writing – review & editing, Writing – original draft, Conceptualization. **Yvonne Fabian:** Writing – review & editing, Funding acquisition. **Philippe Jeanneret:** Writing – review & editing, Funding acquisition. **Andreas Lüscher:** Writing – review & editing, Funding acquisition, Conceptualization. **Olivier Huguenin-Elie:** Writing – review & editing, Writing – original draft, Methodology, Funding acquisition, Conceptualization.

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.

Data availability

No data was used for the research described in the article.

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Appendix A. Supplementary material 1A and 1B

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