A farming systems approach to linking agricultural policies with biodiversity and ecosystem services

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Many countries are reshaping their agricultural policies to better enhance biodiversity and ecosystem services (BES) in farmlands, but measuring the effectiveness of policy instruments in BES delivery is challenging. Using the European Agricultural Policy as an example, we propose the application of a farming systems (FS) approach as a cost-effective tool for linking policy design and expected BES outcomes. On the basis of available data from subsidy payment agencies, such an approach can identify groups of farms that share similar management practices as well as the associations between FS and corresponding BES potential, and improve modeled outputs of farm management responses to policies and other drivers of change. We describe how this relatively unexplored source of information can help to support applied ecological research and relevant policy, and call for these data to be made available across Europe and elsewhere.

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A growing number of countries across the world are altering their agricultural policies to improve environmental performance in farmlands, primarily through the provision of financial payments to farms. These can be divided into three general approaches (Vojtech 2010). The dominant model in countries such as the US, Australia, and New Zealand focuses on payments for land retirement from production, as land is perceived to attain a higher environmental value when taken out of farming and returned to its natural state (Baylis *et al.* 2008). A second approach is based on granting funding to farmers to reduce their environmental impact while retaining intensive production systems. In contrast, the third approach, and the most common form across Europe and in several other regions of the world, is based on providing payments to incentivize farmers to implement more environmentally friendly farming practices, under the assumption that substantial environmental value is associated with existing farmland (Baylis *et al.* 2008; Vojtech 2010).

In a nutshell:

- We propose the use of a farming systems (FS) approach, based on farm-level spatially explicit agricultural data, to explore the links between policy design and biodiversity and ecosystem services (BES) outcomes
- The advantages of an FS approach include consistency in management among farms using similar systems, allowing improved predictions of farming impacts on BES; the type of FS selected by farmers will also depend on policy and other drivers
- FS approaches have potential for use in applied ecological research, and for more cost-effective policy design and evaluation involving lower administrative costs

Agricultural areas across Europe support high levels of biodiversity and ecosystem services (BES); indeed, approximately 30% of European farmland is considered to be of "high nature value" (HNV) (Lomba *et al.* 2020) that supports species and habitats of conservation concern. These areas also provide ecosystem services relevant to human society, including preservation of cultural landscapes, protection from natural hazards, and regulation of water quality (Lomba *et al.* 2020). Because most of this land is privately owned and managed by farmers, management decisions are typically driven by agricultural markets, policies, and socioeconomic conditions, as opposed to conservation-oriented goals.

Comprising ~40% of the total European Union (EU) budget, the Common Agricultural Policy (CAP) is one of the major drivers of agricultural management decisions in Europe (Pe'er *et al.* 2014). Since the 1990s, the focus of the CAP has been transitioning from food production, market regulation, and farmers' income support toward remunerating the provision of environmental public goods, following societal demands for improved sustainability and environmental performance. This shift calls for a reorientation of applied research aimed at supporting conservation policies, such that the interests of key actors in land management (ie farmers) and policy regimes are fully taken into account during the process of selecting appropriate analytical tools and approaches (Malawska *et al.* 2014; Pe'er *et al.* 2019). Here, we explore the use of a farming systems (FS) approach, where FS – defined

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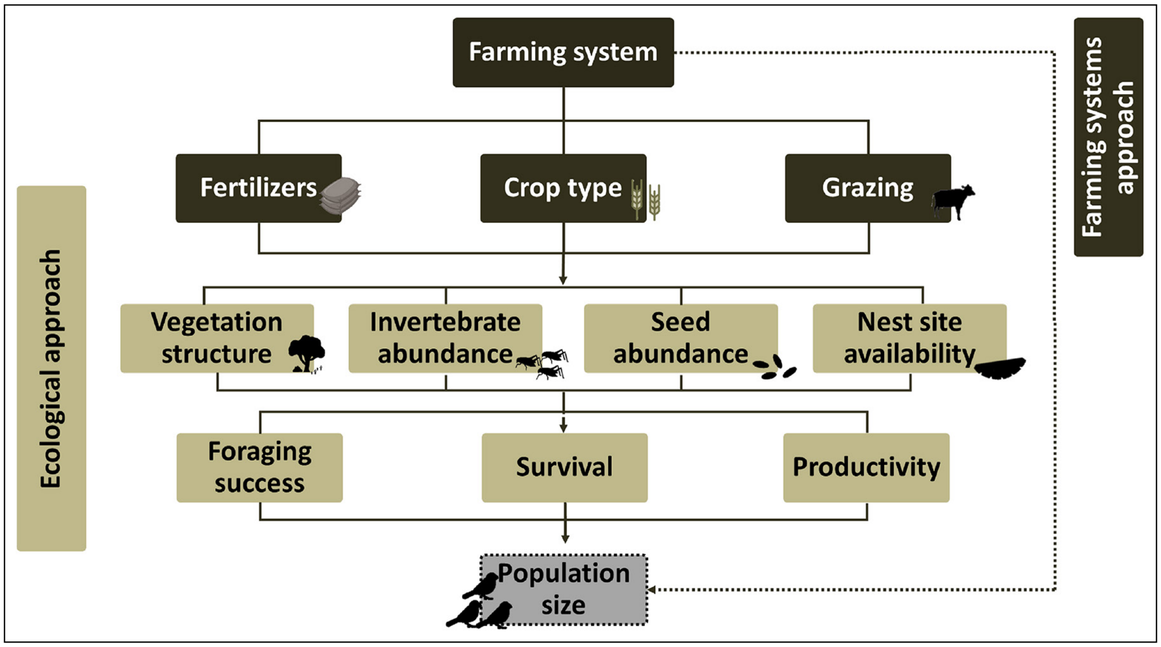

Figure 1. Comparing and contrasting the effectiveness of a traditional *ecological approach* and a *farming systems (FS) approach* as tools for linking agricultural management to biodiversity, using a grassland bird population (light gray box at bottom) as an example biodiversity target/indicator. While the ecological approach focuses on measuring the impacts of proximal ecological drivers, such as habitat and food resources (light brown boxes) on biodiversity outcomes, the FS approach explores the link (dotted line) between top-level management decisions (dark brown boxes), as drivers of the chain of top-down events (eg FS type will determine fertilizer use, which will determine vegetation structure, which in turn will affect bird foraging success and ultimately bird population size), and final biodiversity outcomes.

based on farmers' management choices – potentially acts as an indicator of BES delivery (Figure 1). Using an FS approach would enhance the links between alternative policy options and their respective BES outcomes.

The FS concept

Different FS concepts and approaches have been implemented since at least the 1960s, both for scientific purposes and policy support (Jones *et al.* 2017). In an environmental context, these include the identification of broad types of HNV farmlands delivering relevant BES (Andersen *et al.* 2003; Lomba *et al.* 2014, 2020), although details of the FS underpinning these arable, permanent crop and livestock-based HNV farmlands have yet to be fully assessed. The FS approach proposed here considers the farm as a system and unit of analysis (Reboul 1976). The farmer manages the farm according to her/his choices and aspirations, choosing preferred outcomes and the means by which they can be achieved. The system is open (as it has an environment that affects its state), dynamic (as changes can occur over time in one or more structural properties of the system), and goal-oriented or purposeful (Darnhofer *et al.* 2012). An FS is essentially a collection of farms that have similar characteristics, such as land type, labor, and means of production, as well as cropping and livestock subsystem combinations, with associated management decisions regarding crop types, fertilizer use, livestock rates, and so forth (Figures 2 and 3; Reboul 1976; Ferraton and Touzard 2009). Subsystems also relate to each other through, for instance, forage flows from the crop to the livestock subsystems or the manure flow in the opposite direction. Some systems may be composed exclusively of a single crop or livestock subsystem.

FS as a tool to link policies to environmental outcomes

The potential of an FS approach as a tool to explore the links between policy and BES is based on the four key aspects described in the following sections.

Aspect 1: Management coherence

Farms operating under a specific FS are managed in a goal-oriented way in which individual management decisions are best understood as a whole (ie a system) of strongly interrelated and context-responsive decisions. For example, use of a specific fertilizer or herbicide is often required to introduce a genetically improved, more productive variety of maize (*Zea mays*), and raising cows in a region with cold winters or dry summers requires harvesting hay, silage, or another form of conserved forage to store in preparation for the cold or dry season. Therefore, farm-level management practices with implications for BES are not independent of one another, but rather are linked together as a "bundle" of practices. One advantage of this interdependent nature of practices is that farm management details with important

BES impacts (eg harvest dates, use of agrochemicals, type of mechanical operations) can potentially be inferred from FS (Ribeiro *et al*. 2016a); such detailed information is typically unavailable from existing agricultural data sources and obtaining it often requires expensive farm surveys.

Another potential advantage of an FS approach is the avoidance of drawbacks of policy-making approaches based on the setting of management requirements targeting specific BES as, for example, in agri-environmental regulations. These might result in prescription of management practices that are inconsistent with other practices and/or with those required under a specific FS. Consequently, although such combinations may appear to be ideal from a conservation perspective, such disjointed practices are often detrimental to farmers' interests (eg delaying cereal harvesting to protect the nests of ground-nesting birds in a system where early harvest to produce silage is required). There is evidence to suggest that farmers engaging in agri-environment schemes are more likely to adopt familiar practices that cause lower levels of disruption to their normal agricultural activities than complex management requirements (Van Herzele *et al*. 2013; Lastra-Bravo *et al*. 2015; Nilsson *et al*. 2019). Therefore, an approach based on allowing farmers to choose from the set of existing FS with better BES performance would likely be more readily acceptable to farmers than imposition of requirements to adopt ad hoc sets of practices.

Aspect 2: Links between FS, biodiversity, and ecosystem services

Different FS include specific field- and farm-level agricultural practices (eg crop selection, livestock management, maintenance of non-crop elements) to which biodiversity components respond. Several agricultural practices known to affect BES, such as harvest dates, stocking rates, or pesticide usage, are strongly dependent on the type of FS (Ribeiro *et al*. 2016a). For example, studies on HNV grasslands in southern Portugal improved understanding of the impacts of changing livestock management (Reino *et al*. 2010) and crop types (Delgado and Moreira 2002) on bird diversity in the region. Other studies have shown similar FS-related effects elsewhere and for other taxa (eg carabid beetles [Martel *et al*. 2019], pollinators [Le Féon *et al*. 2013], plants [Klimek *et al*. 2007]). Moreover, depending on the type of FS adopted, farms may retain an assortment of non-crop components like forest patches, scrublands, rough pastures, hedgerows for crop protection, and small dams to hold water for irrigation or drinking purposes. These elements create distinct landscape patterns across an FS (Ribeiro *et al*. 2016b) and likely deliver different BES (eg water quality, natural hazard prevention, cultural services) outcomes (Power 2010). In short, contrasting FS are expected to hold varying BES potential, especially if they are based on distinct crop types and grazing regimes.

An FS can also be characterized by three key dimensions that have differential impacts on BES: (1) production intensity, which can be measured as output per hectare of land (yields of specific crops, or total farm output in Euros per hectare), as per-hectare use of yield-raising inputs (eg fertilizers, pesticides, irrigation water), or as stocking rates; (2) specialization pattern, or how different activities are weighted on the farm as a whole, which can be measured through either shares of total area used by these activities (eg 25% of the farmland sown with wheat) or shares of total outputs (eg milk represents 80% of the total output of the farm, in Euros); and (3) dependency on human labor, which reflects the labor intensity of the FS (eg manual horticultural crops versus mechanized field crops or low-intensity livestock raising). The full use of these dimensions may be helpful in identifying the drivers of the observed impacts of agriculture on BES, and widens the scope of commonly used approaches that focus mostly on production intensity impacts (eg Green *et al*. 2005; Tscharntke *et al*. 2012).

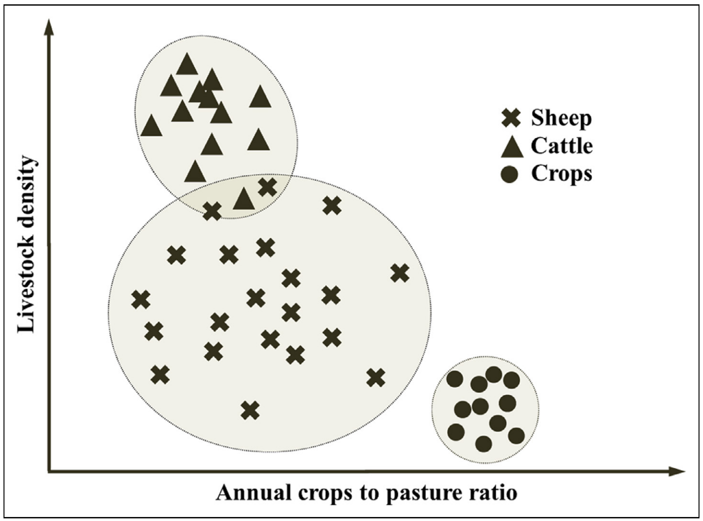

Figure 2. Conceptual representation of an FS. Each symbol corresponds to a farm that lies along two axes representing livestock density (livestock units per unit grazing area; y axis) and a pasture-to-cropland gradient, expressed as a ratio (area of cropland to area of pasture; x axis). Farms are clustered into three types of FS, consisting of crops, sheep, and cattle. Although some variability in management occurs among farms within a given FS along these axes, farms that adopted a particular FS will likely exhibit much greater similarity to one another than they will to farms that adopted a different FS.

Aspect 3: Similar responses to policy and other drivers

A farmer's decision to adopt a specific type of FS is influenced by (1) the structural or biophysical characteristics of the farm (eg farm size, slope, soil quality, rainfall, availability of water for irrigation); (2) the particular attributes of the farmer and household (eg available family labor and their skills, investment capacity, attitude toward risk); (3) the socioeconomic characteristics of the region in which the farm is located (eg labor market, technical advice, access to input and output markets); and (4) the market and policy

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Figure 3. Variety of FS in a high nature value farmland area in southern Portugal (Ribeiro *et al.* 2014). Photos illustrate farms managed under (a) crops-, (b) cattle-, and (c) sheep-based systems.

environment, such as prices for different possible inputs and outputs, and available policies (eg whether policy income support is coupled or decoupled from production). Most of these factors act either as drivers or as constraints in the decision-making process leading to FS selection (Figure 4); farmers subjected to comparable drivers and contexts tend to choose the same types of FS, and farms operating within the same FS generally exhibit similar responses to biophysical, market, and policy drivers (Dixon *et al.* 2001). This enables the exploitation of such close relationships for modeling and prediction purposes (eg to predict shifts in FS based on policy change, as described in Ribeiro *et al.* [2018]; see aspect 4, below).

Aspect 4: Availability of EU-scale information about farm management

A basic requirement for the development of an FS typology is access to farm-level data. Ideally, such data should cover a large range of farm management aspects, which usually requires costly farm surveys. In Europe, an alternative to surveys that has attracted recent attention is the Integrated Administration and Control System (IACS) database (Beaufoy and Marsden 2013; Lomba *et al.* 2017). IACS data are collected on a yearly basis through farmers' declarations when applying for CAP payments, and include information on livestock and land use/cover at the farm-parcel level; moreover, it has the advantage of being spatially explicit when linked to the Land Parcel Identification System (LPIS). Although IACS/LPIS data are primarily collected for EU policy implementation purposes (eg management of CAP payments), there has been a recent trend toward making these data available to other stakeholders, which will boost research opportunities by providing access to a highly detailed agricultural database (at the parcel level) that is updated yearly and is potentially available at the scale of the EU (Tóth and Kučas 2016). Such data have recently been used in FS research (eg Ribeiro *et al.* 2014, 2016a; Lomba *et al.* 2017) and in the estimation of spatiotemporal choice models to predict FS choice under distinct policy scenarios (eg Ribeiro *et al.* 2018).

Strengths and weaknesses of an FS approach

The proposed FS approach linking agricultural policies to BES outcomes has several possible applications in two main areas: (1) applied research and (2) policy design and evaluation. These applications, as well as some of their strengths and weaknesses, are addressed in the following two sections. Although focused on the European context, the recent trend in the US from land retirement toward subsidies on working farmlands – a policy shift emphasized in the last two federal farm bills (Lichtenberg 2019) – suggests that these considerations may be relevant for the US context as well, and in other countries where policies relying on payments based on farming practices are implemented.

Applied ecological research

Additional research in different policy, socioeconomic, and ecological contexts is required to demonstrate the general usefulness of the proposed approach, and to identify some of its limitations. Three priorities are suggested: (1) modeling FS dynamics (in time and space) in relation to policy incentives and other drivers (eg Ribeiro *et al.* 2014); (2) evaluating the FS underlying HNV farmlands (Lomba *et al.* 2020); and (3) identifying the BES potential associated with different types of FS under various geographical contexts (see proposed methodological approach in WebPanel 1). The lattermost priority is especially important because, from an ecological perspective, FS act as relatively long-range indicators of the proximate drivers of BES (Figure 1); as such, they may have only weak relationships with intended BES outcomes, which are largely influenced by other drivers. Consequently, more detailed descriptions of proximate drivers, such as non-crop elements, habitat structure, and spatial configuration, may be required to understand the implications for BES.

Policy design and evaluation

A relevant policy application for the proposed FS approach is in ex-ante evaluations of the environmental effects of policy reforms. In fact, changes in the policy and market price environment in which farmers make their FS choices, such

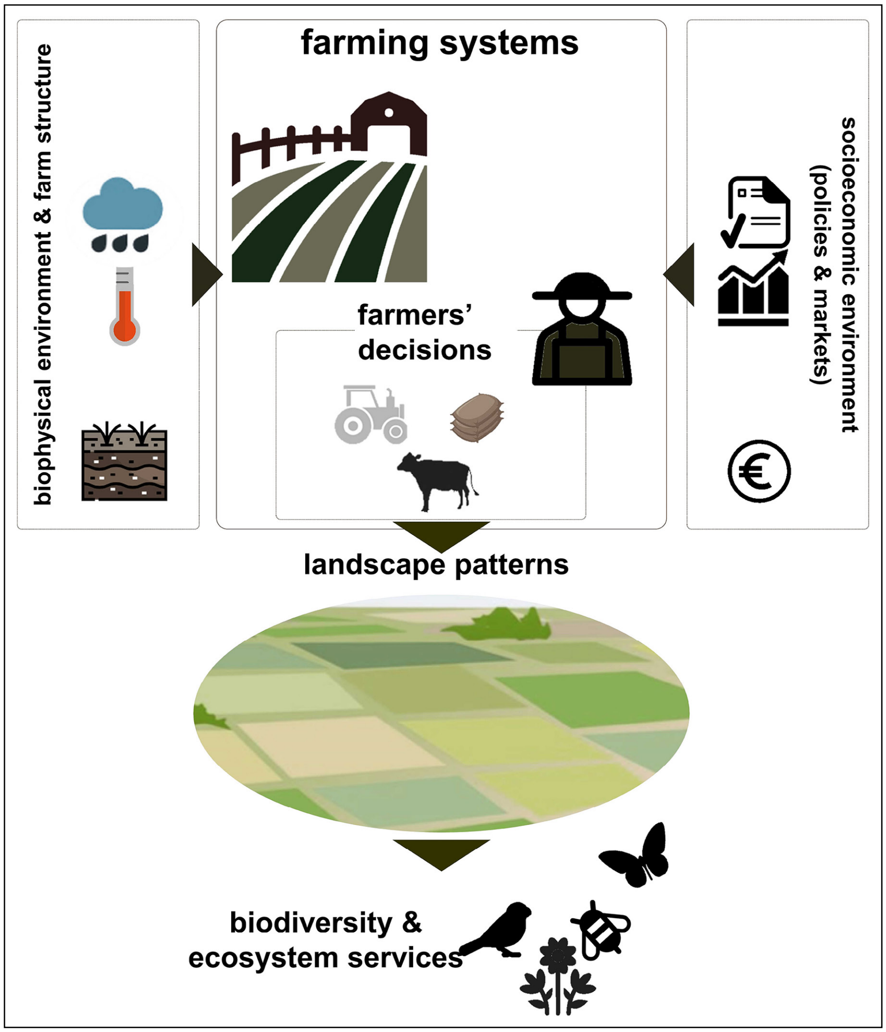

Figure 4. Conceptual framework of the FS approach linking agricultural policies to biodiversity and ecosystem services (BES). Drivers of FS selection include the biophysical environment and farm structure, as well as the socioeconomic environment. The choice of FS type will affect farming practices and landscape patterns, both of which are key drivers of BES.

as CAP reforms or international trade liberalization agreements, may lead to massive changes in FS at broad, supranational scales, with potential impacts on BES (Santos *et al.* 2016). In these cases, the proposed approach may be used to model the effects of policies as drivers of FS selection. The estimated choice models can then be used to predict how farmers would change (or keep) their FS under different alternative policy options and enable the estimation of spatiotemporal FS choice-models with economic data, which can be used to simulate outcomes from distinct scenarios of policy change (eg introduction of a policy in which farms operating a particular FS previously selected for its high level of BES delivery are paid a premium; Ribeiro *et al.* 2018). Assessing which FS have greater BES potential will provide the final step to delivering an ex-ante evaluation of these different policy options.

Another important application for the proposed approach is found in the context of the ongoing debate concerning how the European CAP should be reformed so that public funds are progressively directed to pay for environmental public goods demanded by society as a whole (Santos *et al.* 2016; Pe’er *et al.* 2019). In this context, two alternative paths have been advocated: (1) widening broad geographical scale policies (eg Pillar I greening measures under the last CAP reform, or Eco-schemes, their likely successor in the upcoming CAP reform); or (2) deepening targeted incentives promoting specific environmental public goods in particular areas (typically the focus of Pillar II agri-environment schemes). The former has the advantage of reducing transaction costs (expenses required to oversee compliance and manage payments) but results in lower conservation effectiveness, as conservation objectives and management prescriptions are often poorly specified. In contrast, the latter is tailored to meet biodiversity conservation objectives at local or regional levels but will incur higher associated administrative costs (Ribeiro *et al.* 2016a). Alternative approaches that strike the right balance between scheme precision and administrative costs (Vatn 2002), by retaining more focused management prescriptions while reducing transaction costs (Poláková *et al.* 2011), are therefore needed. The FS framework potentially represents a relatively simple and practical way to progress along these lines. For example, it could be applied to policy design within Pillar I of the future CAP (eg in the forthcoming Eco-schemes) as a convenient compromise between highly targeted agri-environment schemes and broad-brush horizontal policies. Panel 1 provides a step-by-step explanation of how this could be accomplished: after identifying the existing FS that perform best in terms of BES, policy support would be redirected toward farms that adopt those targeted FS as a premium support on top of the Pillar I base-payment level. This premium payment would be justified by the actual provision of public goods by a farmer under a particular FS (Cooper *et al.* 2009), which would be a major policy improvement vis a vis the current payment level that is based on the individual or regional level of historical support. Farmers would continue at this premium support level as long as their management actions remain within the range of existing variability in the targeted FS. This policy design approach would keep both private and public transaction costs relatively low, as it is grounded in the existing administrative framework for data collection from farmers and would not require further control measures to monitor farmers’ compliance with additional management commitments.

The proposed FS approach should not be compared with locally targeted agri-environmental management commitments or result-based approaches (Herzon *et al.* 2018), but

Panel 1. Designing policies that support targeted FS to address BES issues: a step-by-step approach using IACS and LPIS data

Step 1. For a given administrative region, identify priority biodiversity and ecosystem services (BES) issues to be addressed by agricultural policies within the region. This can include threatened species conservation, fire hazard reduction, water quality improvements, or landscape conservation.

Step 2. Develop a farming systems (FS) typology for the region. See details in WebPanel 1.

Step 3. Assess the BES value of each FS in the region. Map the region's FS and evaluate, via published literature or dedicated field surveys (WebPanel 1), the spatial associations between FS and the indicators of priority BES issues in the region.

Step 4. Assess whether supporting targeted FS is an effective way to address priority BES in the region. FS that deliver higher levels of priority BES will be selected as potential candidates for policy-based support. Compare the average and variability of each priority BES indicator across FS (Figure 5). Effectively addressing BES delivery through supporting targeted FS requires that the differences between FS are substantial. If this is not the case, then alternative approaches (eg detailed agri-environmental commitments or result-based methods) are required to induce the desirable changes in BES delivery.

Step 5. Assess FS dynamics as a basis to decide whether to support targeted FS. Assess recent trends in FS in the region using Integrated Administration and Control System (IACS) time-series data. If the FS that were selected in Step 4 as candidates for support are stable (in terms of the number of farms and total area) to ensure the desired level of BES provision, do nothing; if these FS are declining, implement a policy payment to farms operating these FS. Payment must be high enough to encourage farmers to adopt the FS at a level adequate for meeting the desired degree of BES provision.

Step 6. Operational phase. In the operational phase, annually collected IACS data (derived from applications submitted by farmers for Common Agricultural Policy [CAP] payments) are used to reclassify farms under the existing FS typology for the region on a regular basis, following an automatic procedure within the IACS system. Those classified in the FS selected for support in Step 5 will receive the premium policy payment.

Step 7. Post-implementation phase. The FS typology should be updated every few years (eg during CAP reviews) to accommodate any changes in regional agricultural trends and practices, including new FS that improve BES performance.

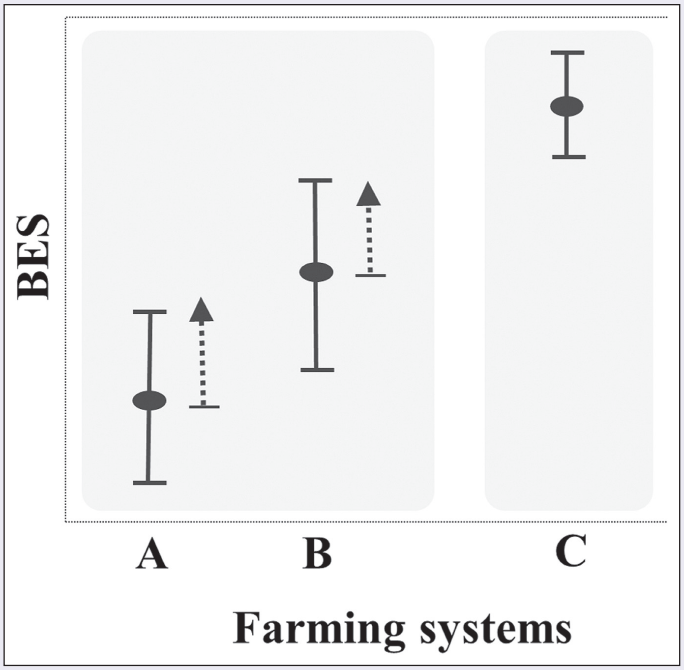

Figure 5. Potential delivery of BES (means and confidence intervals) across three FS types (A, B, and C). FS type C is clearly better than types A and B, and can be used for an FS approach. However, little difference exists between FS types A and B, and therefore locally targeted approaches such as agri-environment or results-based systems would likely be more effective in enhancing BES potential in these cases (dotted arrows).

rather with those of the current "single payment" and "greening" schemes, both of which have been strongly criticized for their ineffectiveness not only by researchers (Pe'er *et al.* 2019) but also by the European Court of Auditors (ECA 2017). However, when BES delivery potential does not differ substantially across FS types, detailed agri-environmental commitments or result-based approaches will be required, which will involve higher transaction costs.

Another possible limitation of the proposed FS approach is that developing an FS typology from past data may hinder full consideration of the potential role of agricultural innovation in better addressing the BES issues at stake. For this reason, the FS typology will need to be routinely updated with new data (eg in the course of policy reviews).

Conclusions: a plea for data availability

We believe the FS approach represents an effective means of linking policy options with environmental outcomes. Contrasting with the often scarce information from ecological studies, a huge amount of information has been gathered by agricultural agencies across European countries for administrative and farmer payment purposes. Spatially explicit time-series of farm-level data are notoriously difficult to obtain in most EU countries, however, due to data confidentiality limitations (Andersen *et al.* 2003). In the US, a mechanism similar to the IACS database does not currently exist, although the US Department of Agriculture's initiative FSAfarm+, which allows producers to access information about such factors as field parcel sizes and boundaries, crops planted, and other elements (eg areas of the Conservation Reserve Program), suggests that a similar approach could be possible, provided current data accessibility constraints are relaxed. We therefore recommend that these data be made available for research purposes, as they would have great value for use in policy design and evaluation. This should include the identification of research priorities and the co-design of research questions together with agricultural agencies. However, even when farm-level information is not accessible, the FS principle can be extended to available regional-level statistics, at least for research purposes: for instance, to derive "farming landscape systems" across administrative regions and explore large-scale patterns of associated BES (Santos *et al.* 2016).

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References

Andersen E, Baldock D, Bennett H, and Beaufoy G. 2003. Developing a high nature value farming area indicator. Brussels, Belgium: Institute for European Environmental Policy.

Baylis K, Peplow S, Rausser G, and Simon L. 2008. Agri-environmental policies in the EU and United States: a comparison. *Ecol Econ* **65**: 753–64.

Beaufoy G and Marsden K. 2013. CAP reform 2013: last chance to stop the decline of Europe's high nature value farming? Gland, Switzerland: WWF.

Cooper T, Hart K, and Baldock D. 2009. Provision of public goods through agriculture in the European Union. London, UK: Institute for European Environmental Policy.

Darnhofer I, Gibbon D, and Dedieu B. 2012. Farming systems research: an approach to inquiry. In: Darnhofer I, Gibbon D, and Dedieu B (Eds). Farming systems research into the 21st century: the new dynamic. Dordrecht, The Netherlands: Springer.

Delgado A and Moreira F. 2002. Do wheat, barley and oats provide similar habitat and food resources for birds in cereal steppes? *Agr Ecosyst Environ* **93**: 441–46.

Dixon J, Gulliver A, Gibbon D, and Hall M. 2001. Farming systems and poverty: improving farmers' livelihoods in a changing world. Rome, Italy, and Washington, DC: Food and Agriculture Organization and World Bank.

ECA (European Court of Auditors). 2017. Greening: a more complex income support scheme, not yet environmentally effective. Luxembourg City, Luxembourg: ECA.

Ferraton N and Touzard I. 2009. Comprendre l'agriculture familiale. Diagnostic des systèmes de production. Versailles, France: Éditions Quae.

Green RE, Cornell SJ, Scharlemann JPW, and Balmford A. 2005. Farming and the fate of wild nature. *Science* **307**: 550–55.

Herzon I, Birge T, Allen B, *et al.* 2018. Time to look for evidence: results-based approach to biodiversity conservation on farmland in Europe. *Land Use Policy* **71**: 347–54.

Jones JW, Antle JM, Basso B, *et al.* 2017. Brief history of agricultural systems modeling. *Agr Syst* **155**: 240–54.

Klimek S, Richter gen Kemmermann A, Hofmann M, and Isselstein J. 2007. Plant species richness and composition in managed grasslands: the relative importance of field management and environmental factors. *Biol Conserv* **134**: 559–70.

Lastra-Bravo XB, Hubbard C, Garrod G, and Tolón-Becerra A. 2015. What drives farmers' participation in EU agri-environmental schemes? Results from a qualitative meta-analysis. *Environ Sci Policy* **54**: 1–9.

Le Féon V, Burel F, Chifflet R, *et al.* 2013. Solitary bee abundance and species richness in dynamic agricultural landscapes. *Agr Ecosyst Environ* **166**: 94–101.

Lichtenberg E. 2019. Conservation and the environment in US farm legislation. *EuroChoices* **18**: 49–55.

Lomba A, Guerra C, Alonso J, *et al.* 2014. Mapping and monitoring high nature value farmlands: challenges in European landscapes. *J Environ Manage* **143**: 140–50.

Lomba A, Moreira F, Klimek S, *et al.* 2020. Back to the future: rethinking socioecological systems underlying high nature value farmlands. *Front Ecol Environ* **18**: 36–42.

Lomba A, Strohbach M, Jerrentrup JS, *et al.* 2017. Making the best of both worlds: can high-resolution agricultural administrative data support the assessment of high nature value farmlands across Europe? *Ecol Indic* **72**: 118–30.

Malawska A, Topping CJ, and Nielsen HØ. 2014. Why do we need to integrate farmer decision making and wildlife models for policy evaluation? *Land Use Policy* **38**: 732–40.

Martel G, Aviron S, Joannon A, *et al.* 2019. Impact of farming systems on agricultural landscapes and biodiversity: from plot to farm and landscape scales. *Eur J Agron* **107**: 53–62.

Nilsson, L Clough Y, Smith HG, *et al.* 2019. A suboptimal array of options erodes the value of CAP ecological focus areas. *Land Use Policy* **85**: 407–18.

Pe'er G, Dicks L, Visconti P, *et al.* 2014. EU agricultural reform fails on biodiversity. *Science* **344**: 1090–92.

Pe'er G, Zinngrebe Y, Moreira F, *et al.* 2019. A greener path for the EU Common Agricultural Policy. *Science* **365**: 449–51.

Poláková J, Tucker G, Hart K, *et al.* 2011. Addressing biodiversity and habitat preservation through measures applied under the Common Agricultural Policy. London, UK: Institute for European Environmental Policy.

Power AG. 2010. Ecosystem services and agriculture: tradeoffs and synergies. *Philos T Roy Soc B* **365**: 2959–71.

Reboul C. 1976. Mode de production et systèmes de culture et d'élevage. *Économie Rural* **112**: 55–65.

Reino L, Porto M, Morgado R, *et al.* 2010. Effects of changed grazing regimes and habitat fragmentation on Mediterranean grassland birds. *Agr Ecosyst Environ* **138**: 27–34.

Ribeiro PF, Nunes LC, Beja P, *et al*. 2018. A spatially explicit choice model to assess the impact of conservation policy on high nature value farming systems. *Ecol Econ* **145**: 331–38.

Ribeiro PF, Santos JL, Bugalho MN, *et al*. 2014. Modelling farming system dynamics in high nature value farmland under policy change. *Agr Ecosyst Environ* **183**: 138–44.

Ribeiro PF, Santos JL, Santana J, *et al*. 2016a. An applied farming systems approach to infer conservation-relevant agricultural practices for agri-environment policy design. *Land Use Policy* **58**: 165–72.

Ribeiro PF, Santos JL, Santana J, *et al*. 2016b. Landscape makers and landscape takers: links between farming systems and landscape patterns along an intensification gradient. *Landscape Ecol* **31**: 791–803.

Santos JL, Madureira L, Ferreira AC, *et al*. 2016. Building an empirically-based framework to value multiple public goods of agriculture at broad supranational scales. *Land Use Policy* **53**: 56–70.

Tóth K and Kučas A. 2016. Spatial information in European agricultural data management. Requirements and interoperability supported by a domain model. *Land Use Policy* **57**: 64–79.

Tscharntke T, Clough Y, Wanger TC, *et al*. 2012. Global food security, biodiversity conservation and the future of agricultural intensification. *Biol Conserv* **151**: 53–59.

Van Herzele A, Gobin A, Van Gossum P, *et al*. 2013. Effort for money? Farmers' rationale for participation in agri-environment measures with different implementation complexity. *J Environ Manage* **131**: 110–20.

Vatn A. 2002. Multifunctional agriculture: some consequences for international trade regimes. *Eur Rev Agric Econ* **29**: 309–27.

Vojtech V. 2010. Policy measures addressing agri-environmental issues. Paris, France: Organisation for Economic Co-operation and Development.

Supporting Information

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