Services provided by multifunctional agroecosystems : Questions, obstacles and solutions

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Review

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ABSTRACT

Agroecosystems are facing new challenges in the context of a growing and increasingly interconnected human population, and a paradigm shift is needed to successfully address the many complex questions that these challenges will generate. The transition to providing multiple services within an agroecosystem is a starting point for heightened multifunctionality, however, there is still hesitation among stakeholders about moving towards multi-service systems, largely because of the lack of knowledge linking productivity and multifunctionality. We reason that much of this reticence could be overcome through a better understanding of stakeholder requirements and innovative transdisciplinary research extended in the dimensions of time and space. We assembled experts in France to identify priority research questions for co-constructing projects with stakeholders. We identified 18 key questions, as well as the obstacles that hinder their resolution and propose potential solutions for tackling these obstacles. We illustrate that research into agroecosystem multifunctionality and service production must be a hugely collaborative effort and needs to integrate knowledge from different sectors and communities. Promoting dialogue, standardization and data-sharing would enhance transdisciplinary progress. Biodiversity is highlighted as a key factor to explore and incorporate into modelling approaches, but major advances must be made in the understanding of dynamic changes in the biodiversity-function-service nexus across landscapes. Resolving these research questions will allow us to translate knowledge into decision objectives, identify adaptation and tipping points in agroecosystems and develop social-ecological economic pathways that are adaptive over time.

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1. Introduction

An agroecosystem is defined as a spatially and functionally coherent unit that has been modified by people and is primarily dedicated to agricultural production (Martin-Clouaire, 2018), but is not restricted to the site of agricultural activity, as interactions with other ecosystems in the landscape (e.g. watershed and forest ecosystems) impact and are impacted by this activity. Ecosystem service (ES) demand and supply are highly interrelated with the environment, either positively or negatively (Zabala et al., 2021), and the main challenge for creating a sustainable agroecosystem is to achieve natural ecosystem-like characteristics while maintaining productivity and equitable social outcomes (Therond et al., 2017; Augstburger et al., 2018). This challenge is reinforced by the necessity to perpetuate climate-resilient social-ecological systems, whilst meeting increasing demands for energy, food and ecosystem services. Also, land resources are dwindling, resulting in a need to design and manage agroecosystems that can reliably provide multiple ES simultaneously (Manning et al., 2018). Agroecosystems are therefore facing new challenges in the context of a growing and increasingly interconnected human population, and a paradigm shift is needed to successfully address the many complex questions that these challenges will bring about. The transition to providing multiple ES within an agroecosystem has been a starting point leading towards heightened multifunctionality within a landscape, although ES and agroecosystem multifunctionality are different concepts that should be examined independently, before understanding the strong relationships that exist between them.

Ecosystem *function* multifunctionality is defined as the array of biological, geochemical and physical processes that occur within an ecosystem and that indicate the overall performance of an ecosystem. Ecosystem *service* multifunctionality is defined as the co-supply of multiple ES relative to their human demand, and is most relevant for applied research in which stakeholders have definable management objectives (Manning et al., 2018). Both types of multifunctionality are highly complex to quantify, and combined with the inherent interdependence among functions and services, render the production of guidelines for agroecosystem management extremely difficult. Nevertheless, recent advances in frameworks and the development of multifunctionality metrics (Manning et al., 2018) and long-term systemic evaluations (Wittwer et al., 2021), are paving the way for understanding how different farming systems provide specific bundles of ES, and how ecosystem processes mediate the construction of these bundles over time. Also, as biodiversity has been shown to enhance the multifunctionality of a managed ecosystem (Pasari et al., 2013; Mao et al., 2021; Wittwer et al., 2021), it is vital to consider this aspect when developing management scenarios for improving ES provision.

Since the seminal paper by Tilman (1999), more than 880 journal papers on agroecosystem services have been published (see Supplementary Material for bibliometric analyses). Services provided by agroecosystems range from food and fodder production to water pollution and erosion control. Conventional cropping systems deliver the highest production yields, but have reduced multifunctionality, whereas organic and conservation agriculture promote multifunctionality and simultaneously enhance regulating and supporting services, although yield can be reduced (Wittwer et al., 2021). The transition towards sustainable and multifunctional agroecosystems will undoubtedly result in a shift towards more agroecological farming methods. However, there is still considerable hesitation among stakeholders about moving towards less intensive agriculture, because of the lack of knowledge linking productivity and environmental protection (only 56 papers on agroecosystem multifunctionality have been published in the last 20 years: see Supplementary Material for bibliometric analyses). Nevertheless, some of this reticence could be overcome through an increase in ecological, biophysical and economic data, and an improved modelling of processes, leading ultimately to appropriate guidelines and a modification of practices. France is currently steering agricultural practices towards an agroecological transition and so is investing heavily into research on agroecosystems, and is the second largest producer of scientific papers in the world on the subject of agroecosystem services, after the USA (Fig. S1). Therefore, French research institutes and Universities have devised scientific strategies and programs aimed at developing productive, economically-robust agroecosystems that are the least damaging to the surrounding environment. Although our study focuses on opinions mostly from French researchers, we consider that information generated is generic enough to be applied to diverse regions, climates and ecosystems.

Research on agroecosystems and the services they provide has generally focused on the assessment of services or their quantification proxies (e.g., Dominati et al., 2014). Major knowledge gaps exist concerning the ecological processes that determine the responses of agroecosystems to future environmental change, the involvement of stakeholders and the consideration of the socio-cultural context (Balzan et al., 2020). Identifying critical questions will help scientists and stakeholders work together to provide the missing knowledge to fill those gaps, as well as contributing to the design of future research programs. Therefore, we asked experts (including agricultural consultants, young and senior scientists and policy-makers) working on agroecosystems to highlight the areas where research is most needed in the next decade. Participants were also asked to indicate the main obstacles that blocked advances in the field, and propose potential solutions, ranging from biophysical to ecological and economic issues. We then discuss how to address these research challenges through strategic institutional and collaborative opportunities.

1.1. Who should be reading this paper?

This paper is addressed to all those involved in the management of terrestrial agroecosystems and who wish specifically to develop approaches or methods to better understand the functioning of an agroecosystem, in order to enhance its multifunctionality. Although written largely with the western European social-ecological context in mind, the questions given in the paper are designed to encourage new research initiatives between practitioners and scientists around the world, and to identify knowledge gaps that need to be bridged.

In this paper, we have focused on agroecosystems beyond the limit of the urban environment. The research initiatives that we aim to identify can be applied to a broad range of units, from field to farm, within a landscape, but do not consider the presence of urban infrastructure within that environment (e.g. roads or railways that alter the ecological connectivity within an ecosystem). Similarly, specific aspects related to natural and anthropogenic hazards (e.g. floods, storms and fires) and the uncertain consequences of a changing climate have not been focused on. Services themselves are not necessarily defined, but the consideration of the ways in which they interact and can be manipulated is similar among different types of ecosystems, including for example forest and coastal ecosystems.

Throughout this paper, we have focused on the importance of multifunctional systems and bundles of services that a given ecosystem can provide. Managing bundles of services, their interactions and optimisation is a major challenge that practitioners face. Choosing appropriate management options for improving ES provision, requires holistic and specialised knowledge. Therefore, we have explored in detail how future modelling approaches could improve the optimisation of ES, as well provide a better understanding as to how ES interact.

2. Methods

This paper focuses on research questions and initiatives put forward mostly by scientists and agricultural consultants working in France and its overseas departments. The wide range of climates considered therefore include temperate, Mediterranean, subtropical and tropical, and so knowledge gaps can be extended to a broad range of countries. We followed a protocol similar to that used by Hays et al. (2016), of soliciting the views of experts on key questions in a selected area. Initially, a workshop was held in Paris, France (October 2019, https://seminaire.inrae.fr/ecoserv-bilan), to discuss recent and future research initiatives on services provided by agroecosystems in France and its overseas territories. The 85 participants were experts in the area of (agro)ecosystem services or (agro)ecosystem ecology and management and comprised mostly academics and agricultural/environmental consultants. With the aim of identifying priorities for scientific research and solutions for practitioners, the entire audience was asked to answer the following three questions:

1) *Context:* With regard to agroecosystem services in a managed environment, what do you think are the most important scientific questions that still need to be answered if we are to better link research to practice?
2) *Obstacles:* What are the main obstacles that would prevent these questions from being answered quickly?
3) *How to tackle the problem:* What is the best way to answer the scientific questions listed above?

Thirty-five participants of the workshop replied to the three questions and authors were contacted and asked to contribute to the writing of this paper (14 participants declined). Responses were compiled and 29 key questions were identified by the 21 co-authors of this paper. A face-to-face meeting was then held with the co-authors to discuss further and group similar key questions. Eleven key questions were the same or very similar and so were merged by the co-authors. A total list of 18 key questions (plus obstacles and how to tackle the problem) focusing on future research initiatives to link science and practice was derived and circulated to the group of co-authors, who agreed by consensus and compromise on the final list. The key questions were not prioritised into order of importance (Sutherland et al., 2006) but were grouped into three main categories: A) the understanding and potential management of multifunctionality in agroecosystems, followed by B) how the evaluation of agroecosystems and C) development of incentivization schemes would improve multifunctionality. Questions focused on: (i) understanding and identifying agroecological processes combined with ES, (ii) the management and implementation of agroecosystems taking into account ES; (iii) how to evaluate ES in agroecosystems and (iv) incentivization schemes for implementing ES in agroecosystem managment (Fig. 1).

3. Results

3.1. Understanding and managing multifunctional agroecosystems

Q1) How can we investigate the relationships between different levels of biological diversity, the processes and functions impacted by this diversity and the resulting ecosystem services?

Context: The current erosion of biodiversity calls for us to rapidly identify how the deconstruction of ecological networks impacts ecosystem functioning and the services provided (Cardinale et al., 2012). However, research often focuses on just one compartment of the ecosystem, or a limited number of taxa considered more or less representative of the functioning of the entire ecosystem. The biodiversity-function-ES nexus explores the complex linkages between biodiversity, functional traits (Damour et al., 2018), ecosystem functioning and ES (Garnier et al., 2016). Current challenges are how to: (i) integrate the diversity of different parts of the ecosystem, especially the above- and below- ground compartments (Bardgett and van der Putten, 2014); (ii) simultaneously investigate different levels of diversity, from genes to ecosystems, and the different facets of this diversity (i.e., taxonomic, functional and phylogenetic diversity); (iii) develop novel and dynamic frameworks in diversity-functioning research, considering explicitly climate change and (iv) understand how patterns of community organization impact ecosystem dynamics and functions. We need to establish the mechanisms that underlie the relationship between biological diversity and ecological stability, as well as determine the genericity of those mechanisms across a broad scope of ecosystems (Gross et al., 2014).

Obstacles: We do not always know the origin of the functions/processes and their interactions with other factors that ultimately provide ES, thereby hindering the predictability of the response to management practices. It is also difficult to design and perform experiments that integrate adequately variations in current and future climates (Beier et al., 2012). As research focuses more on static than dynamic approaches, the lack of consideration of spatial and temporal dynamics limits our knowledge of ecological systems.

Although the biodiversity-function-ES nexus could be invoked to explore multifunctionality in agroecosystems (Altieri, 1999; Martin and Isaac, 2018), allowing managers to manipulate diversity, enhance and optimize trade-offs among functions (Gamfeldt et al., 2008; Finney and Kaye, 2017), an improved semantic terminology linking the concepts of biodiversity with ecosystem functions and ES (see Q13, Walls et al.,

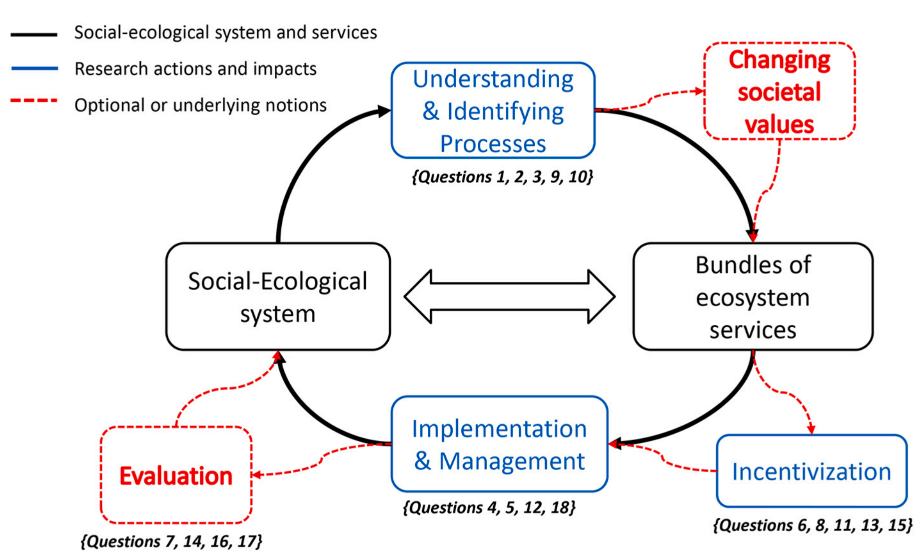

Fig. 1. We focused on 18 key research questions for co-constructing research projects with stakeholders. The main issues to resolve focused on: (i) understanding and identifying agroecological processes combined with ES, (ii) the management and implementation of agroecosystems taking into account ES; (iii) how to evaluate ES in agroecosystems and (iv) incentivization schemes for implementing ES in agroecosystem management.

2014; Senderov et al., 2018) is needed to enable scientists from the fields of biodiversity, ecology and agriculture to work together and envisage ecosystems as a whole.

How to tackle the problem in the future: We need to better characterize the concepts, levels and different facets of biodiversity. Data should be organised into semantically designed knowledge bases (see Q13). Then, a robust analysis of large datasets would highlight the general patterns of associations between taxons/biodiversity, functions and ES. Analysing multiple interactions together would permit a better understanding of the links between networks of biological interactions and ES. Despite the logistic difficulties, research over exended periods of time is vital to understand dynamics, and modelling approaches could be used to predict tipping points and 'failure' scenarios, that cannot be implemented experimentally (see Q9, 10). Understanding the determinants and drivers of biodiversity dynamics at different scales (intra and interspecific, from the individual to the ecosystem and landscape and from season to decade), is necessary for proposing new management scenarios.

Analysing the relationships between biodiversity and functioning will allow us to identify the dimensions of biodiversity that contribute to the provision of ES. For example, the complexity of food webs for functions such as carbon use efficiency, is tightly linked to ES provision (Bloor et al., 2021). Due to the correlations between network structure and ecosystem functions, it would be highly relevant to identify functional groups that reveal levels of vulnerability in biodiversity as agricultural practices occur. These functional groups could therefore act as sentinels, or bioindicators, of changes occurring within an agroecosystem and could be mobilized as a proxy for the ability of agroecosystems to maintain ES provision. Developing such transdisciplinary studies in collaboration with economic and societal actors will make it possible to co-design, evaluate and develop biodiversity-based solutions capable of making social-ecological agroecosystems more resilient within a changing climate. A conceptual challenge remains however: the necessity to consider explicitly the retroactive loops between mechanisms that are a result of human actions, from those that derive from the functioning of the ecosystem. Within a social-ecological agroecosystem, organisms are also acclimating and adapting to changes in the environment, as well as to the strategies of each other. Understanding these complex co-evolutionary dynamic relationships and the resulting emerging properties, requires us to take into account the interactions between different levels of organization, from the individual up to the landscape level.

Q2) How can the arrangements of planned and associated biodiversity and their interactions with cropping or rearing systems, affect pest regulation processes and services?

Context: While pesticide use continues to increase globally, and biodiversity in intensive agricultural environments is collapsing, most current crop protection strategies do not allow for a significant reduction in the use of pesticides. Generally, specific actions at the plot scale consist of enhancing crop robustness and mitigating damages to the crop (Toqué et al., 2015). These actions are only partially effective, as none of them can fully reduce the stock of pests, and should be complemented by other methods, such as strengthened regulation processes based on planned and associated biodiversity. The biodiversity managed by farmers within and around their fields, and the spatial organization of crops at the farm scale, are important drivers of biotic interactions in agroecosystems (Costanzo and Bàrberi, 2014). In terms of crop health, semi-natural habitats can play a major role at the landscape scale for promoting pest predators and parasitoids, but this is a level of organization at which an individual farmer is not the only manager. At the landscape level, other farmers and up- and down- stream actors of value chains also intervene to decide land use and management. Chaplin-Kramer and Kremen (2012), suggested that complexity at the local (farm and field) scale could replace landscape complexity, and that at the scale of a few plots within a farm, it is possible to organize space to promote biodiversity, e.g., via the planting of flower strips (Balzan et al., 2016).

Obstacles: The example of biological pest control illustrates the spatio-temporal issues related to the management of planned and associated biodiversity. The provision of trophic resources (e.g., nectar and pollen), via flower strips on a farm increases the overall presence of pest predators and parasitoids (Jonsson et al., 2008). However, most of these natural enemies do not complete their entire development cycle within a single field but move between fields and semi-natural habitats up to an area of a few hundred meters. The ability of natural enemies to disperse and colonize a given field is highly dependent on the spatial arrangement of surrounding sites (e.g. semi-natural habitats), where they are able to breed and overwinter. Also, pest control performed by many natural enemies often takes place after damage to the crop has occurred, as for univoltine coleopteran pests (Ulber et al., 2010). Strategies that promote this natural pest regulation can therefore be delayed in time and diminished in space. The management of biodiversity must therefore be based on territorial and multi-annual coordination and requires collaboration between the stakeholders who manage the areas concerned.

Data concerning the effects of diversity on services such as biological control and pest regulation are scanty. It is therefore necessary to identify agricultural techniques that enhance the diversity of plant and soil fauna/microorganisms in field crops, such as no-tillage and reduced phytosanitary treatment (Tamburini et al., 2020). A multi-criteria analysis (Craheix et al., 2016) indicated poorer pest regulation in the short term due to tillage suppression, but little is actually known about the biological regulations that are established in the longer term.

In addition to managed plant diversity, weeds also provide major trophic resources for numerous organisms e.g., seeds for farmland birds and pollen and nectar for pollinators and biocontrol agents (Storkey and Neve, 2018). In particular, in open landscapes of field crops, weeds are often the only flowering plants in May-June, when field edges have been mowed. Nevertheless, weeds are still considered as undesirable. Even if some farmers do not use herbicides, they usually try and eliminate weeds, that are rarely voluntarily left in a field or considered for their positive effects (but see Gunton, 2011). How to manage weeds and the ES they provide, rather than focusing on disservices to crop production, is a major question that remains to be answered in full.

How to tackle the problem in the future: To be effective, specific actions promoting pest regulation must be implemented simultaneously by all stakeholders in the same area. For this, stakeholders need to coordinate and co-construct a territorial crop health management project to achieve the desired objectives. It is also necessary to be able to monitor the temporal dynamics of regulations and have monitoring indicators to know if the approaches implemented make it possible to achieve the objectives targeted. When methods for measuring regulations are cumbersome, it would be useful to set up biological regulation indicators that are easy to deploy in a territory and relevant to the expected functions of biodiversity. For example, the use of sentinel prey, that involves monitoring the disappearance of prey items provided by scientists, enables the quantification of predation pressure exerted by natural enemies. Combined with camera trap technology, it is possible to document predator-prey interactions as well as intraguild predation over a wide range of field conditions (Kistner et al., 2017; Hemerik et al., 2018), offering promising opportunities to increase sampling across large spatiotemporal scales.

Q3) What are the ecological interactions between different ecosystems from the landscape level to a global scale and how do they affect the provision of ecosystem services?

Context: The ecological literature on agricultural landscapes provides significant evidence that several ES result from processes occurring at multiple space and time levels, and that they are often provided by multiple interacting ecosystems. For example, forests, meadows and streams provide different resources for mobile pollinators or biocontrol agents, that are beneficial to farmers (e.g. Alignier et al., 2014; Carrie et al., 2017; Vialatte et al., 2017; Raitif et al., 2018, 2019). The management of these mobile-agent-based ES (MABES, Kremen et al., 2007) requires consideration not only at the local scale, where services are delivered, but also the distribution of resources at the landscape scale, and the foraging ranges and dispersal movements of the mobile agents (Kremen et al., 2007). In particular, fluxes of matter, energy and information between habitats are the foundation of many ES, such as the dispersal of beneficial insects from forest to crops (Roume et al., 2011). The characteristics of interfaces between ecosystems e.g., field borders or forest edges, and their connectivity, are therefore of major importance for better understanding and managing these fluxes.

Beside ecological interactions, that need to be considered both above- and below- ground, there are also interactions resulting from anthropogenic activities and management practices. For example, the management of one given plot in a farm is often related to the global status of that farm and therefore the status of other plots in the farm. On a broader scale, there are also long-distance interactions, through fluxes of matter and energy, as well as social and economic interactions (Martin-Lopez et al., 2019), such as the link between animal feeding systems in Europe and their consequences on soya (*Glycine max* L.) production in other parts of the world. Ecosystems could then be interdependent with regard to certain ES, and these interactions could occur at larger spatial and temporal scales, up to the global scale. Understanding these ecological interactions between different ecosystems across space and time levels is therefore required for robust ES management at a broad scale.

Obstacles: Uncertainties related to the variability of ecological processes provided by multiple interacting ecosystems make management difficult. For example, while the influence of landscape structure on e.g., pollen flux and biological pest control is widely recognized, mobilizing habitat management within agroecosystems still remains highly context-dependant (Karp et al., 2018). Management practices at various scales, from field to landscape, strongly influence ES (Carrié et al., 2017; Ricci et al., 2019), but the available knowledge about ecological processes at local- and landscape- scales and their relation to ES via habitat management is still insufficient for implementation (Salliou et al., 2019).

How to tackle the problem in the future: No consensual rule has been defined for assessing interacting ES at the landscape level. Spatial gradients, as well as discontinuous zones in which ES are present, need further evaluation and consideration in long-term studies (Landis, 2017; Vialatte et al., 2019). Also, spatial modelling approaches, using, for example, meta-ecosystem models (Marleau et al., 2014), are important to understand the diverse interactions between ecosystems at multiple scales and to explore social-ecological scenarios of habitat management (Wu, 2013; Cong et al., 2014). As many interactions are context dependent and can vary strongly from one place to the other, generic management rules are difficult to define, thus locally adaptive methods would be more efficient for ES management.

Q4) How can we better investigate the relationships between management practices and ecosystem services in agroecosystems?

Context: There is a major gap in the linkage between management practices and agroecosystem services, largely because of the complexity of processes and concepts involved. The delivery of ES can be influenced by numerous biotic and abiotic factors and processes, including rearing or farming practices (e.g., cultivation method and stocking density of species, fertilization and tillage) (Alleway et al., 2018; Lazartigues et al., 2012; Palomo-Campesino et al., 2018). We need to better understand the description and provision of ES in agroecosystems that specifically produce livestock. To illustrate this point, we can consider ponds managed for the purpose of producing fish. Their functioning is based on the exploitation of natural biomass that develops from nutrients produced in the pond itself, provided by water from watersheds or supplied directly by fish farmers. The availability of nutrients influences the amount of fish produced and although not designed for the treatment of non-point sources of pollution, fish ponds can be an efficient way to trap pesticides carried in agricultural field runoff, thereby helping to reduce maximum pesticide concentrations and potential risks of adverse effects in downstream ecosystems (Gaillard et al., 2016). Low levels of pesticide residue occur in fish muscle, largely due to low bioaccumulation and/or low transfer to the muscle (Lazartigues et al., 2012) reducing the risk of pesticides accumulating in fish products.

Obstacles: Data on the relationships between agroecosystem practices and ES are still lacking. This is even more true for animal production systems, including aquaculture (Weitzman, 2019). Usually, only a limited number of ES are studied, such as food provisioning, eutrophication control and nitrogen sequestration. The partial assessment of the relationships between practices and ES makes it difficult to recommend changes in practices to promote the development of more sustainable methods. It also leads to a distorted view of these agroecosystems, leading to intense debates, as for instance in the case of dam ponds managed for fish production. For example, some stakeholders call for the removal of dam ponds in order to restore the ecological continuity of hydrosystems (Blayac et al., 2014), while fish farmers defend these agroecosystems, highlighting their roles for maintaining low water levels or improving water quality (Four et al., 2017; Gaillard et al., 2016).

How to tackle the problem in the future: To take into account the multifunctionality of agroecosystems and achieve a balanced management of trade-offs between ES, it is necessary to develop a holistic and interdisciplinary understanding of these systems (e.g., Weitzman, 2019). Examining agroecosystems within the common analytical framework of agricultural systems and the diversity of different available models, and their possible combinations at local, regional or global levels (Therond et al., 2017), would be a first useful step towards a shared vision of these systems. Subsequent steps could include the evaluation of and interactions between ES whilst taking into account all possible management practices and considering the development of specific indicators according to agricultural sectors. Furthermore, laboratory sites for the acquisition of multi-scale, multi-criteria and long-term data could usefully strengthen our knowledge of the interactions between agroecosystems and ES and thus contribute to more agro-ecological systems (Dumont et al., 2013). At the species level, the study of their functional traits is a relevant way to better integrate the metabolic functioning of agroecosystems and the ES they maintain and regulate (Alleway et al., 2018).

Q5) What are the effects of management practices on ecosystem services at different space and time scales?

Context: At a local level, farmers influence various habitats through the structure of cropping and rearing systems, as well as the nature, frequency and intensity of management practices (e.g. Boinot et al., 2019; Cerda et al., 2017; Winter et al., 2018). At a larger scale, farmers manage land use with regard to crop rotations, woodcutting and hedgerow maintenance or removal. These disturbances affect the quality of all the ecosystems that make up the landscape and the resulting ES provision at various space and time scales (Kim et al., 2017; Winter et al., 2018). For example, the intensity of pesticide use conditions the effects of landscape structure on biological pest control (Ricci et al., 2019), while agricultural practices in surrounding fields have a strong influence on pollinator communities in a given focal field (Carrie et al., 2017; Carrié et al., 2017; Vinatier et al., 2012). Identifying and quantifying the relative effects of management practices at different space and time scales is required for designing adapted ES management strategies; for example, when several farmers are interdependent on certain ES in a landscape, the coordination of their management practices would be beneficial (Barnaud et al., 2018). Also, different ES may need managing at various time scales, e.g. in vineyards, the regulation of pests and diseases are regulated by management practices within the grapevine cycle, whereas fruit production is largely determined by certain climatic conditions and management practices that occur in the year preceding the harvest (Guilpart et al., 2014, 2017).

Obstacles: Studies that deal with relationships between rearing or cropping system management and ES often focus on system structure and biodiversity, but lack consideration of the impacts of technical interventions on ES for a given structure. For example, the residues of cover crops, their degradation and mineralization, can vary depending on whether they are removed with herbicides or machinery (Ashford and Reeves, 2003), influencing also soil nitrogen and carbon fluxes (Coppens et al., 2006). Also, farmers often have very different ideas about a system's functioning and the consequences of their management practices (Salliou et al., 2019). This diversity is probably one of the main obstacles of ES management design at large spatial and time scales. In order to create a common understanding necessary for a collective action, (i) independent assessments of the effects of practices are required to avoid epistemic uncertainty and (ii) participation of farmers in decision-making processes is necessary (Etienne et al., 2011).

How to tackle the problem in the future: To quantify the relative effects of management practices and their interactions at multiple spatial and time scales, not only are adequate metrics and indicators needed (see Q6), but also long-term studies combined with experimental trials. Such studies require testing the effects of coordinated practices at large space and time scales and through a strong partnership with farmers. In addition, adaptive management should be considered as it enables the maintenance of a stable provision of a set of ES in changing conditions (Ripoche et al., 2011). Practically, modelling is a good tool to enhance the common understanding of farmers of a system, to clarify the effects of potential solutions and identify decision rules (see Q9 - Q11; Ripoche et al., 2011, Salliou et al., 2017, Voinov and Bousquet, 2010).

3.2. Evaluating multifunctional agroecosystems

Q6) Which metrics should be used to describe the diversification of cropping / rearing systems in complex multifunctional agroecosystems?

Context: To evaluate the ES (or associated functions) provided during diversification of cropping and rearing systems and compare them, it is necessary to first describe the structure of the complex agroecosystem with common metrics that take into account their various biodiversity levels.

Numerous diversity indexes can be found in the scientific literature, and many deal with taxonomic diversity, for example species richness or Shannon index (Garnier et al., 2016). It is now considered highly relevant to describe the functional diversity of an ecosystem based on knowledge of organism traits (Violle et al., 2007) when evaluating ES (Bello de et al., 2010; Díaz et al., 2007). Again, a variety of functional diversity indices has been proposed by authors (Petchey and Gaston, 2006), e.g. the functional richness, diversity or evenness (Villéger et al., 2008), functional dispersion (Laliberté and Legendre, 2010), Rao's quadratic diversity (Rao, 1982) or community-weighted mean (CWM) trait of a community (Garnier et al., 2004; Ricotta and Moretti, 2011). The CWM takes into account the traits of organisms comprising a community (e.g., a forest or a cropping system) and the relative abundance of each organism, following the mass-ratio hypothesis (Grime, 1998).

Obstacles: Although several metrics are used to describe the structure of a diverse community via a functional approach (Gaba et al., 2015; Garnier et al., 2016), these metrics lack consideration of spatial and temporal dynamics that are essential for ES provision (Kremen et al., 2007; Roume et al., 2011; Rusch et al., 2016). Additionally, the development of the functional approach has raised some questions about its transfer to cropping systems and domesticated species (Milla et al., 2014; Roucou et al., 2018; Tribouillois et al., 2015), and the dependence of ES on organism intraspecific trait variability, that remains to be further explored (Siefert et al., 2015; Violle et al., 2012; Garcia et al., 2020).

How to tackle the problem in the future: To develop appropriate metrics to describe the structure of diverse or complex agroecosystems, we need to determine appropriate spatial scales at which ES are provided (Paiola et al., 2020). For example, Rafflegeau et al., (2019) proposed the concept of *ecosystem service functional motif* to analyze the functioning of agroforestry systems, defined as the smallest representative spatial unit relevant to understand the provision of a set of targeted ES at a given time. This concept could be used along with functional diversity quantification so as to describe a wide range of complex cropping, forestry or rearing systems, and compare their ability to support ecosystem functions and deliver ES. We also need to develop metrics that can be scaled up from the field plot to landscape and larger scales (Boinot et al., 2019; Kremen et al., 2007; Rusch et al., 2016). As an example, He et al. (2019) proposed to calculate *ecosystem traits* for the functional description of ecosystems, that integrates a normalization per unit land area to enable such scaling, and could be developed for agroecosystems. Finally, ES temporal dynamics should be further investigated, as they represent a major concern for cropping systems, where crop rotations are predominant and the resulting biotic interactions change over time (Garcia et al., 2020; Schipanski et al., 2014).

Q7) Is it possible to link biophysical indicators to the provision of intermediate and final ecosystem services and measure an effective service value?

Context: An important challenge for ES assessment is to evaluate the relationships between the ecological indicators describing the functional characteristics of the ecosystem (or *ecological functions* or *ecosystem functions* or *intermediate services or supporting services*) and the final ES that are measured by their benefit to humans (Birkhofer et al., 2015). This challenge is particularly pertinent for regulating ES, as opposed to provisioning ES whose production functions and valuation methods are comparatively well established. For example, in the case of biological control, common ecological indicators are predation level or predator density, whereas the final ES is directly linked to crop yield. Working upstream with intermediate ES is necessary to account for the functional characteristics of ecological processes, including dynamics, feedbacks and uncertainties, and also for how management contributes to ES, and how it could be adapted under alternative and/or future conditions (Birkhofer et al., 2015).

Final ES correspond to the intermediate step of the general cascade conceptual framework, positioned at the interface of biophysical and social systems (Haines-Young et al., 2012). Connecting these two systems requires interdisciplinary communication, and as such, faces the challenges inherent to multidisciplinary research, i.e., adapting methods, concepts and frameworks, scale and time frames. Bridging the gap between systems is all the more demanding when biophysical sciences are linked to social sciences, and a complete change of perspective is required by all involved. While biophysical disciplines lie on the supply side of ES, social and economic scientists also investigate the demand side, driven by the benefits that ES deliver in terms of increased well-being.

Obstacles: Measuring intermediate ES is possible, either in a direct form (e.g., pollination success of selected plants, or seed quantification in the case of biological control of weeds), or through indicators of ES provision (e.g., predator species richness in the case of biological control) and proxies that are indirectly linked to ES (e.g., proportion of semi-natural habitats in the vicinity of a given field) (Birkhofer et al., 2015). Linking these measurements to final ES, and ultimately the benefits for human well-being, must overcome a series of methodological obstacles:

(i) a common intermediate or several related intermediate ES can drive multiple final ES at the same time, resulting in final ES that co-vary, in a synergistic or antagonistic relationship. This type of interaction between multiple ES is indirect, as opposed to direct interactions that are mostly due to causal relationships between final ES (Bennett et al., 2009). For example, the proportion of semi-natural habitats surrounding crop fields increases both pollination and pest control in crop fields (Birkhofer et al., 2015). The consequence of the relationships between ES is the difficulty in assessing the marginal contribution of individual ES on a unique final ES (e.g., effect of pollination on crop yield, as crop yield also depends on the level of pest control).

(ii) Spatial mismatches between measurements of intermediate and final ES mean that in most cases, intermediate ES are measured at a fine spatial scale whereas final ES often need to be assessed at a more aggregate level. This mismatch is particularly true for direct ecological measurements related to an experimental plot (e.g., pollination success of selected plants), that may be especially difficult to scale up to a whole field or farm, due to biotic interactions and environmental conditions (Birkhofer et al., 2015), as well as management decisions.

(iii) The multifaceted evaluation of ES, increasingly used for decision making and planning (Gómez-Baggethun et al., 2016). Values of ES are usually grouped into three broad categories depending on the dimension of the definition they refer to. As a consequence, some categories are more suited for some types of ES, and each of them preferentially uses different types of evaluation units. The first category refers to ecological values, derived from biophysical assessments and relate to the status and condition of the ecosystem (Gómez-Baggethun et al., 2016): they strongly associate with supporting and regulating services. The second category consists of social and cultural values, and the third category of economic values; both categories are based on human principles and preferences. Social and cultural values relate to non-material benefits that people obtain from ecosystems and are heterogeneous in terms of approaches and methods. Economic values rely on market valuation through a variety of methods and are usually expressed in monetary terms (Cordier et al., 2014).

How to tackle the problem in the future: A general recommendation is to build ES standards across disciplines, which define terminology, acceptable data and methods, and reporting requirements in a particular context (Q13). Polasky et al. (2015) highlight that standards could help define which ES to include (Fisher et al., 2009; La Notte et al., 2017), the relevant geographic and temporal scales (Pagella and Sinclair, 2014; Geijzendorffer et al., 2015) and acceptable levels of uncertainty (Hamel and Bryant, 2017). To specifically overcome the obstacles in linking ES indicators along the cascade framework, Birkhofer et al. (2015) suggested choosing a small set of measurements that form joint, reliable indicators of an individual ES. For example, in the case of biological control, indicators could be chosen that cover aspects of service- and disservice- providing units (e.g., predator density, pest density, pest consumption rates and pest reduction), ecosystem management (e.g., pesticide use or tillage regime) and landscape modification (e.g., landscape patchiness or proportion of semi-natural habitats in the surrounding landscape). To tackle obstacle (i), it is necessary to better describe and understand relationships between multiple ES via a mechanistic perspective (Birkhofer et al., 2015). Experimental tests of ES relationships and statistical approaches using large and replicated datasets are complementary additional approaches.

The spatial issues developed in obstacle (ii) call for better representing the diversity of stakeholders involved in ES supply and demand, and for acknowledging that ES supply is determined not only by biophysical conditions, but also by demand and management. Additionally, it is necessary to integrate sensitivity to the spatial scale into analyses (Geijzendorffer et al., 2015). For obstacle (iii), it is vital to acknowledge the idea of added value, or of one ES having multiple values ('value pluralism'), that are not necessarily monetary values. Multicriteria analyses could then be performed that do not rely only on cost-benefit analyses (that compress all dimensions into a single value), and so are a way forward to better integrate the biophysical, social, cultural and economic components of ES evaluation (Gómez-Baggethun et al., 2016).

Q8) Can we develop operational indicators that make it possible for farmers to evaluate the services provided (or the associated functions)?

Context: Evaluating the ES, or associated functions, provided in agroecosystems highlights the need for robust ES indicators (Q7). Depending on the question to answer, or the decision to take, indicators can be defined using two principal approaches, (i) reductionist, using process-based indicators that describe specific properties of the system, or (ii) integrative, using indicators that describe results of the system's functioning and not the underlying processes (Kibblewhite et al., 2008). The use of ES indicators have led to different ES classification systems, in several attempts to develop a common framework for studying ES (e.g. Albert et al., 2016; Haines-Young and Potschin, 2010; Millennium Ecosystem Assessment (Program), 2005; Pascual et al., 2017; TEEB, 2010). However, ES still appear to be poorly quantified: in a review of 405 papers, Boerema et al. (2017) focused on 21 ES and found an average of 24 different measures per ES. ES indicators are either expressed as stocks or flows, with a minimal effort to normalize the ES indicators to time and/or area (Czúcz et al., 2018). It is therefore necessary to encourage the use of common units and normalization to evaluate the provision of ES in a diagnostic or management approach.

Obstacles: To date, most ES indicators have been designed through academic research with scientific credibility and precision as the main criteria, and are closer to analysis indicators (Wery et al., 2012) than to management or assessment indicators (van Oudenhoven et al., 2018), and finding a compromise that can be used by stakeholders and scientists at different levels is a challenge. Therefore, many ES indicators are not used for decision making (van Oudenhoven et al., 2018). In addition to credibility, ES indicators should match salience, legitimacy and feasibility to be used for decision making at farm and broader scales (Cash et al., 2003; van Oudenhoven et al., 2018).

Another important issue with the use of ES indicators for management or assessment purposes is the lack of monetary quantifications of ES: provisioning ES are better monetized than regulating or cultural services for which studies are scanty (Q7, Boerema et al., 2017; Czúcz et al., 2018). Although there are studies that have attempted ES monetization (e.g. Costanza et al., 1997; Porter et al., 2009), quantification of ES is often made for their ecological value, or social-economic value, but rarely for both, revealing an important gap for the use of ES indicators by decision makers (Boerema et al., 2017; Geijzendorffer et al., 2017; van Oudenhoven et al., 2018).

How to tackle the problem in the future: Although we acknowledge the necessity to develop biophysical ES indicators that integrate various spatial and temporal scales and comprise pluralism and sensitivity (Q7), it is also necessary to simultaneously develop operational indicators that correspond to stakeholders' demands, and that are easy to operate in the field with limited cost. For example, in plant production systems, soil function tools such as the Biofunctool® (Thoumazeau et al., 2019) and Greenback method (Calvaruso et al., 2020), have been developed to assess soil health by examining multiple indicators of carbon transformation, nutrient cycling and soil structure maintenance. An aggregated soil functioning score can then be calculated that is sensitive to land management and soil type. However, even the use of such simplified tools is not suitable for many farmers because of the sampling and measurements required. A more simple method may be appropriate, such as the Agroecosystem Service Capacity (ASC) approach, that aims at assessing the capacity of the land cover classes of an agroecosystem, and to provide one or several of a maximum of 20 different agroecosystem services. The approach provides results for each land cover class and is the basis for calculating an aggregate index for the whole agroecosystem (Augstburger et al., 2018).

While ES indicators have been much studied in terms of metric quantification (e.g., magnitude and quality), few data exist concerning how spatial and temporal variations affect their stability and vulnerability. Stability refers to the level of uncertainty of ES provision under a given condition, while vulnerability refers to the resistance and resilience of ES provision to perturbations (Oliver et al., 2015). For example, a boreal forest's ES of carbon sequestration can be very high in magnitude, but unstable because of its sensitivity to climatic conditions and vulnerability to catastrophic events (e.g., drought and wildfires). Uniquely relying on the magnitude and ignoring stability and vulnerability of an ES indicator can render decision-making subjective and not optimized, with serious consequences for agroecosystem functioning in the long-term. We propose that in the future, when choosing and characterizing an ES indicator, it is necessary to also characterise the magnitude, stability and vulnerability, featured as the 'inner attributes' of an ES indicator and validated by different communities focusing on 'outer attributes' such as salience, feasibility and legitimacy (Fig. 2).

Q9) How can we fully benefit from the potential of biophysical models for ecosystem service assessment in multifunctional agroecosystems?

Context: Biophysical modelling involves the explicit simulation of physical and biological processes in space and time. This approach has become popular in recent years because it is useful for highlighting interactions between ES, to understand the influence of human actions (e.g., agricultural practices) and also climatic change (e.g., the increase in the frequency of extreme events) on ES provision equilibrium and evolution (Lavorel et al., 2017). The biophysical approach is not the only one that can highlight synergies and antagonisms between ES (for example, statistical approaches can reach this goal in certain contexts), but only process-based models can allow robust scenario testing and predictions in evolutive contexts (Keane et al., 2015; Viglizzo et al., 2016). However, three major obstacles currently hinder the use of these biophysical and spatially-explicit models for ES assessment.

Obstacles: The first obstacle is the lack of knowledge on the functioning of certain biological communities. Functioning is directly associated with the level of diversity encountered in agroecosystems, with the typology of interactions between the biotic and abiotic components of the ecosystem (Rudi et al., 2020) and between communities (Schmitz, 2008). Noriega et al., (2018) also point out that the main reason why some communities are poorly studied is because they are considered as providers of disservices, not services, especially insects and weeds.

The second obstacle is the need to connect the scale of management with the scale of biophysical processes underlying ES, which could result in upscaling/downscaling issues. This connection between scales often imposes the necessity to work with fine resolution models, that calls for a detailed parameterization, for which data are only partially available (Duru et al., 2015; Lavorel et al., 2017). To achieve this challenge, a better conceptualisation of the landscape as a physical matrix, (i) conditioned by a given topography and pedology, (ii) where biotic and abiotic flows and processes occurred and (iii) that are modified by anthropogenic actions (e.g., agricultural practices), is needed (Poggi et al., 2018; Vinatier et al., 2016).

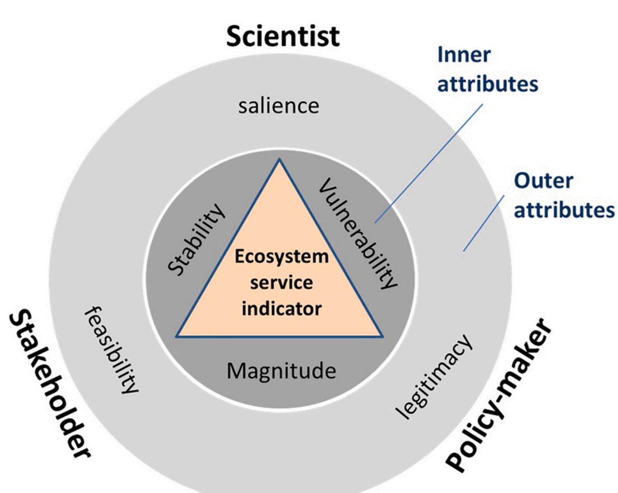

Fig. 2. When choosing and characterizing an indicator for an ecosystem service indicator, it is necessary to characterise the magnitude, stability and vulnerability, featured as the 'inner attributes' of the indicator and validated by different communities focusing on 'outer attributes' such as salience, feasibility and legitimacy (Q8, Q12).

The final obstacle is that integrated models for ES assessment represent a major number of ecosystem functions, and studying such several processes, on generally large spatial and temporal scales, make their simultaneous observation difficult. Furthermore, the search for genericity and exhaustivity for the representation of processes when coupling biophysical models can be accompanied by a complexification of the mathematical equations describing the biophysical processes (especially because simplified assumptions generally used for local studies are no longer valid).

How to tackle this problem in the future: We identify three main research needs associated with (i) data acquisition, (ii) partial validation of models at different scales and (iii) mitigation of the complexification of the mathematical formulations of coupled biophysical models. Large spatial and temporal-scale observations are required to calibrate and validate coupled models. National and international observatories, and specifically those designed to study long-term agronomic, ecological, and physical processes (e.g. Molénat et al., 2018), can be a guarantee for the continuity and consistency in the collected data. The need of detailed parameterization of the physical supports of the landscape, e.g. soil, is now being addressed by programs such as *Digital Soil Mapping* (Lagacherie, 2008), or e.g. land use, by the *Land Cover Map for France* (THEIA, 2023). These programs aim to provide users with high-resolution maps over large territories. Future research needs will require the integration of non-homogeneous data and development of new algorithms to map properties of interest (Lagacherie et al., 2018). Other promising programs for ES assessment are trait databases (e.g., Kattge et al., 2011; Kleyer et al., 2008), because trait-based approaches are a powerful tool to evaluate the functionality of diverse species found in ecosystems.

The issue of using biophysical models outside of the context in which they were developed is not new (Beven, 1989), but it is still not addressed fully. A proper assessment of ES based on a coupling of mathematical equations should focus more on clarifying the assumptions under which these equations have been developed, and should provide an individual validation of the models in the context of study, since the validation of the global coupling is rarely possible. Research should therefore focus on the development of methods to validate large-scaled coupled models.

Regarding the complexification of coupled biophysical models, three options could be explored: (i) research should continue on focusing on new methods for ensuring the reliability/velocity of numerical calculations (for example Crevoisier et al., 2009; Ross, 2003 for a numerical resolution of Richards' equations that represent water transfers in soils in unsaturated conditions), (ii) research should focus on developing semi-empirical laws that might be less correct from a physical perspective, but are more parsimonious (Antle and Capalbo, 2002), and less demanding in terms of calculation power (see for example the semi-empirical formulas of Dollinger et al., 2016 and Margoum et al., 2006 for an evaluation of the pesticides' immobilization function in agricultural ditches), (iii) research should propose methods for the hierarchization of the processes driving the system to simplify its translation into equations.

Q10) Is it possible to correctly model the multifunctionality of agroecosystems?

Context: In any social-ecological system, such as an agroecosystem, assessing its multifunctionality (represented by the richness and/or magnitude of a bundle of ES) is a complex task (Manning et al., 2018). Empirical or mechanistic ES models are useful tools for tackling such a challenge, especially within the context of interdisciplinary projects that have multisectorial cooperation (van der Plas et al., 2016). Each ES model usually targets one or several indicators representing a single ES and by coupling a list of ES models, a bundle of ES can be quantified and then assembled for the assessment of multifunctionality. This approach is now widely used, but it lacks the holistic treatment of a system, which is complex in terms of both spatial and temporal scales and component interaction (Mao et al., 2021).

Obstacles: Several drawbacks can be identified when coupling ES models for studying multifunctionality: (i) defining and determining multifunctionality of a system is a multifaceted challenge, spanning from ES identification and their hierarchical ordering, to indicator selection and ultimately to model availability, choice and calibration. Due to the differences in ES values and indicators, as well as the models used to predict ES, this modelling approach makes comparisons among systems complex, as well as the evaluation of trade-offs or synergies among functions difficult to perform. (ii) Candidate ES models to be coupled usually differ in mechanistic configuration and prediction fitness, making their outputs highly heterogeneous in terms of both scale and quality. Based on these juxtaposed outputs, diagnostics of multifunctionality can be subjective and biased, and to remove bias, modellers need to consider equally trade-offs and conflicts between services. (iii) The ES models to couple are usually discipline-dependent and fail to capture transdisciplinary interactions among components of a system and their resultant interdependence among the ES. Moreover, these models are usually independently operated by specialists and coupling these models demands a high level of coordination, cooperation and communication among different sectors, thereby containing a high failure risk from a management point of view.

How to tackle this problem in the future: While the coupling of models is valid because it is relatively easy to perform and produces quantitative solutions for decision-making, one alternative way to model multifunctionality is to holistically model a system as a whole and fully consider all possible interactions among the components. To do this, network-based discrete-event models are promising tools, as they can represent a system composed of components and processes, and simulate the system's dynamics and fates according a given initial state and defined scenario (Gaucherel and Pommereau, 2019). Network-based discrete-event models have recently been associated with the concept of ES to study multifunctionality (Mao et al., 2021). For example, in a case study of a complex mountain social-ecological system, Mao et al. (2021) modelled the system as an interaction network comprising 16 binary components and 51 processes using Petri Nets (Reisig, 2013). The presence/absence of certain components are associated with presence/absence of 22 ES. With this approach, the authors fully explored the fates of the system as well as their ES provision under different scenarios of climate change and local policy. Despite the discrete and qualitative nature in model configuration at the current stage, this approach allows a large number of components, processes and services to be combined simultaneously and then their effects and feedback to be studied in a comprehensive way (Mao et al., 2021).

Q11) Can the role of biodiversity be represented in models of ecosystem services to design the cropping agroecosystems of tomorrow?

Context: To design the cropping agroecosystems of tomorrow, we need multiple ES scenarios and models to simulate these scenarios and this need has been identified for all ecosystem types (IPBES, 2016). There is also an urgent need to address explicitly the role of biodiversity in ES modelling (IPBES, 2016; Lavorel et al., 2017). Process-based models will help fill this gap, as opposed to correlative or expertise-based models, that are more simple but too limited for the managing of complex situations.

Obstacles: When modelling plant production in agroecosystems, we commonly use crop, grassland or forest process-based models that describe a soil-plant-atmosphere system, and provide access to a number of ES related to the flows through that system, while simulating the effect of agricultural or forestry practices on these flows and thus on the resulting services (e.g. Demestihas et al., 2019; Kragt and Robertson, 2014; Temperli et al., 2012). This type of process-based model, however, is not a long-term solution in its current form. While biodiversity is recognized as a major driver of ES (Hooper et al., 2005), it is usually absent in most crop models, whether it is "associated" (i.e., biodiversity is colonizing the agroecosystem) or "planned" (e.g., by managers), very few crop models consider species mixtures (Gaudio et al., 2019). Grassland or forest models consider species mixtures but do not take into account other types of biodiversity.

The IPBES report (IPBES, 2016), which devotes a chapter to biodiversity models and another to ES models, suggests coupling both approaches but adds that this linkage is complex. The lack of consistent results on the apparent links between biodiversity and ES, linked to the variety of underlying dynamics and how they are understood (Ricketts et al., 2016) is a major obstacle.

How to tackle the problem in the future: We suggest five approaches to remove these barriers and better take into account the role of biodiversity in future models for predicting multiple ES in agroecosystems: i) We need to climb the "cascade of services" (Haines-Young and Potschin, 2010, see Q7), which extends from biodiversity to ES via ecosystem functions, then to benefits and values. Rather than extending biodiversity models to ES (top-down approach), it is more relevant to start by identifying the bundle of ES of interest (bottom-up approach).

ii) We must identify for each ES of interest, the multiple functions underpinning it. Several authors agree that understanding synergies and tradeoffs between ES requires working through the multiple underlying ecosystem functions and examining the effects of biodiversity on these functions (Bennett et al., 2009; Duncan et al., 2015).

iii) We should determine, for each ecosytem function, the components involved and their interactions. These components are both abiotic and biotic, and will emerge from planned and/or associated biodiversity and may correspond to populations, communities, functional groups and key species. Major ecological interactions take place between biotic and abiotic components and within biotic components (e. g. competition or intra-guild predation). These interactions must be translated into flows of matter or elements (e.g., biomass, carbon, nitrogen, nitrogen and water) or the energy flow relevant to the function.

iv) The integration of several functions (for one ES) and several ES requires a reconsideration of the components involved, their links, or their simultaneous contribution to several functions. Duncan et al. (2015) proposed grouping together ecosystem functions, according to their similarity in terms of biotic components that contribute positively or negatively to them. For the integration of ecosystem functions and ES, a network theory framework suggested by Dee et al. (2017) and supported by applications (Xiao et al., 2018) could be tested.

v) It is time to clarify the role of agricultural or forestry management in the production of ES, especially when stakeholders desire to improve biodiversity at key moments, that change the state of abiotic structures (for example, through irrigation and fertilization), or by acting directly on ecosystem functions, such as when pesticides are used that disrupt faunal population dynamics.

Q12) Is it possible to disentangle and optimise complex bundles of ecosystem services in multifunctional agroecosystems?

Context: In an agroecosystem, the multiple ES that are produced have positive or negative interdependencies and are either directly linked with each other or indirectly linked because they are affected by the same biotic and abiotic ecosystem components and processes. Over the last 15 years, studies have highlighted the importance of examining multiple ES or the multifunctionality of an ecosystem rather than one single ES (Bennett and Balvanera, 2007; Fagerholm et al., 2019) and many authors have demonstrated that trade-offs or synergetic patterns occur among multiple ES (Gonzalez-Ollauri and Mickovski, 2017; Turner et al., 2014). However, few studies have explored how to take into account such interdependency in the optimization of agroecosystem management.

Obstacles: Diverse trajectories exist between ES and indicators; for example, compared with a simple pattern of one indicator per ES, some ES, such as soil fertility and biodiversity conservation, necessitate a joint use of multiple indicators (Figs. 2 and 3). A single indicator can therefore refer to several ES, or one ES can correspond to an indicator that is associated with another indicator of yet another ES (Fig. 3). All such complex cases in reality could therefore be derived from these simple patterns. We argue that differentiating and considering the diversity of

ES-indicator projection patterns is important in procedures that seek optimal management scenarios and for avoiding the effect of compensation or redundancy between ES (e.g., an indicator is counted twice during optimization).

How to tackle this problem in the future: Several multi-criteria optimization methods exist for ecosystem management, such as ES scoring or ranking with weighted or unweighted factors, correlations, principal component analyses and Pareto front analyses (Duncker et al., 2012; Temperli et al., 2012; Lafond et al., 2017; Andreotti et al., 2018). These methods differ in precision, efficiency, computation cost and genericity. Among them, Pareto front analyses, that are widely used in economic science, have been shown as promising for decision-making in agro-ecosystem management. For example, using Pareto front analyses, Lafond et al. (2017) sorted optimal scenarios from a bundle of candidate scenarios simulated by combining a set of factors for forestry management, and revealed that the scenarios that are currently used in practice are far from being optimal. In an agroforestry management context, Andreotti et al. (2018) demonstrated the interest of sorting and comparing the best and worst sites for ES provision following Pareto optimum conditions. However, how the patterns of projection between ES and indicators affect Pareto diagnoses are still unknown and remain a challenging research perspective.

Q13) Can we improve the supply of knowledge to models of ecosystem services in agroecosystems?

Context: One of the objectives of ES modellers is to develop a simulation framework in which existing agroecosystem situations can be represented and computational experiments performed, that allow stakeholders to explore management scenarios and analyse biological and physical behaviour of systems (Martin-Clouaire, 2018). As all ES models, regardless of type and utilisation, require knowledge or data as either input or to analyse output, it is necessary to create knowledge bases that comprise the information required to perform robust computational experiments. A knowledge base is a repository that stores complex structured and unstructured information that can then be shared, facilitating access by diverse softwares.

Obstacles: ES have been much quantified and mapped across diverse spatio-temporal scales, socio-political contexts and for different policy objectives, leading to an immense variety of approaches, methods, tools, modelling and mapping outputs (Drakou et al., 2019). Field observations, data, documents and visual supports, collected by different stakeholders such as biologists, foresters, farmers and breeders therefore exist, but the management and reuse of these data are made difficult by the multiplicity of media and formats used, and by the diversity of vocabularies used. Also, studies on agroecosystems require systemic approaches to understand, for example, how to better manage a site in response to climatic variables, pests or soil pollution, both in space and time (Conde Salazar et al., 2020). These approaches must also link closely to other fields of knowledge such as climatology, zoology, socio-politics and soil science.

How to tackle the problem in the future: In order to limit complexity and organize data into a knowledge base to be used by ES modellers for multifunctional agroecosystems, we propose the development of an

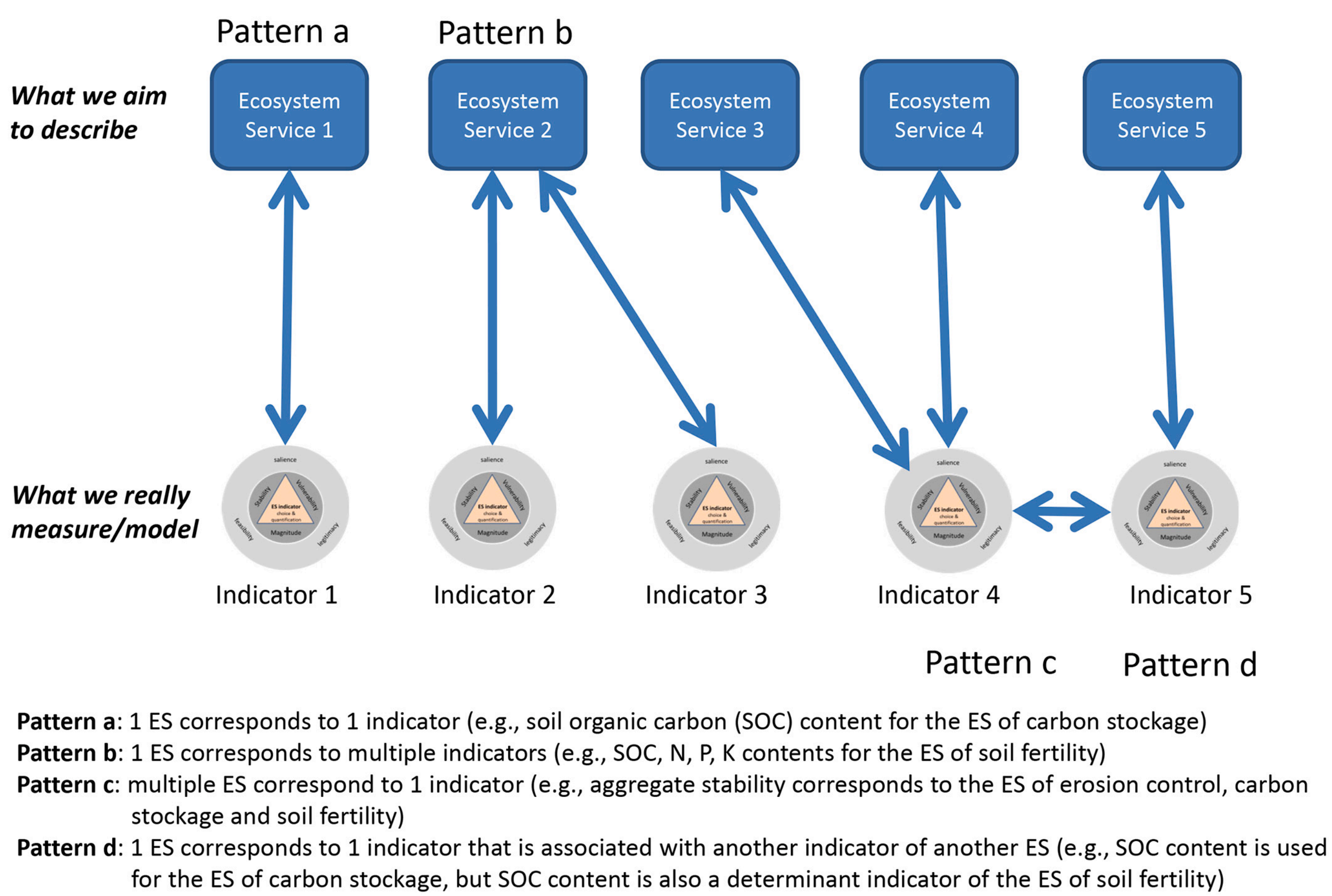

Fig. 3. Diverse relationships exist between ecosystem services (ES) and indicators. Compared with a simple pattern of one indicator per ES, some ES (such as soil fertility and biodiversity conservation in this example), necessitate a joint use of multiple indicators. A single indicator can therefore refer to several ES, or one ES can correspond to an indicator that is associated with another indicator of yet another ES. All such complex cases in reality could therefore be derived from these simple patterns. We argue that differentiating and considering the diversity of ES-indicator projection patterns is important in procedures that seek optimal management scenarios and for avoiding the effect of compensation or redundancy between ES (Q12).

ontology that will provide a global overview of existing data and will allow the integration of new data, regardless of format. An ontology explicitly, consensually and formally defines the terms used to describe and represent a field of knowledge, such as agricultural science (Uschold and Gruninger, 1996). Ontologies add a new concept to the description of knowledge, by making it possible to reason about the different types of knowledge in a field, and to collectively identify new knowledge. The foundation for an ontology of ES in the setting of agroecosystems exists but has not yet been created (Martin-Clouaire, 2018; Nimmagadda et al., 2019). Although an ontology for ES does exist (ESOnto, Drakou et al., 2019), it is generic across different scientific domains and is focused on the quantification and mapping of ES. Similarly, ontologies for agro-ecological knowledge management exist for the description and organization of knowledge related to cropping systems (Soulignac et al., 2019) and agroforests (Conde Salazar et al., 2020). Therefore, it is relatively easy to link these interoperable ontologies and create a dedicated knowledge base for ES modelling in multifunctional agro-ecosystems, considering the foundation laid down by Martin-Clouaire (2018).

3.3. Developing incentivization approaches to improve multifunctionality in agroecosystems

Q14) How can we assess uncertainty in analyses of ecosystem services and communicate better to decision makers?

Context: While most ES assessments claim to inform decision-making, only a minority includes uncertainty assessment (Hamel and Bryant, 2017). Yet, uncertainty directly affects how stakeholders perceive the quality of the analysis and the applicability of the results. Assessing to what extent conclusions are robust and identifying sources and ranges of uncertainty is critical to achieve credibility, guide implementation and enhance the effective use of evidence-based insights. Also, much ES research in managed ecosystems is oriented towards predicting the consequences of future management options on ES provision, in a context of climate change. As the effects of innovative management techniques in the mid- and long-term, and how they interact with the changing climate conditions, are mostly unknown, uncertainty assessment deserves particular attention (Birkhofer et al., 2015).

In the context of ES assessment, uncertainty can be defined as the unknown order or nature of things, a lack of confidence about possible outcomes and/or the inability to assign probabilities to these outcomes (Refsgaard et al., 2007; IPBES, 2020). A common typology identifies three dimensions of uncertainty: (i) source of uncertainty (e.g., inputs, drivers and model structure, parameters, model technical implementation, assessment tools), (ii) magnitude, and (iii) nature (lack of information or inherent variability) (Hamel and Bryant, 2017; IPBES, 2020; Walker et al., 2003; Grêt-Regamey et al., 2013; Hou et al., 2013).

Obstacles: Environmental and social changes are inevitably associated with uncertainty and complexity. The multidisciplinary nature and integrative position between human and environmental systems in ES assessments further increases uncertainty and complexity (Hou et al., 2013). Furthermore, different types of sources can contribute to uncertainty and new information may necessitate a significant update of the models (Birkhofer et al., 2015). The uncertainty that stems from strong nonlinearities, tipping points and stochasticity is even more difficult to evaluate (Grêt-Regamey et al., 2013).

Studies evaluating ES with monetary units are particularly prone to criticism in non-scientific audiences because the estimated values highly depend on the assessment tool used (e.g., cost-based methods including replacement cost, damage cost avoided and substitution cost, *versus* methods eliciting economic values such as the willingness to pay and to accept payment) and the aggregation method over space, time, and multiple ES (Hou et al., 2013, Song, 2018).

Hamel and Bryant (2017) identify four main challenges about the technical feasibility of uncertainty assessment: (i) there is too little guidance on uncertainty assessment, (ii) is time-consuming and complex,(iii) scarce and poorly characterized data create too much uncertainty to handle, and (iv) spatial data make it difficult to assess and communicate uncertainty. In decision-oriented studies, scientists are concerned about results from uncertainty assessment not improving decisions and have different perceptions of uncertainty among stakeholders and scientists, preventing dialogue (Hamel and Bryant, 2017).

From a socio-economic point of view, uncertainty stems from agents' behaviour. On the supply side, managers and farmers exhibit heterogeneous preferences with respect to the quantity and quality of ES they are willing to deliver. Thus, they respond differently to public policies and other incentives aiming at orienting ES supply. Human responses to ecosystem change are also partly unknown (Grêt-Regamey et al., 2013). On the demand side, ES consumers show varying preferences with respect to their willingness to support ES provision (see for example Johnson et al., 2012).

How to tackle the problem in the future: The evaluation of uncertainty is an important research challenge for improving assessments of ES (Birkhofer et al., 2015). Hamel and Bryant (2017) discuss qualitative approaches, such as uncertainty matrices, to characterize sources and levels of uncertainty, and also review a number of quantitative techniques. These techniques include very simple approaches like considering ranges and bounds derived from the literature or other data sources. When direct empirical data is missing, one can rely on expert elicitation to identify subjective beliefs about ranges of values. Data scarcity further highlights the importance of creating knowledge bases for ES assessment (Q 13).

Using alternate sources of raster inputs to approximate variability, or hypothetical landscapes to account for extreme assumptions, are intermediate options for uncertainty assessment in modelling (Hamel and Bryant, 2017). More advanced tools include sensitivity analyses, Monte Carlo analyses, Bayesian approaches (see Grêt-Regamey et al., 2013), development of scenarios for plausible futures and robust decision-making techniques. Hamel and Bryant (2017) also highlight the importance of knowledge co-production with stakeholders to build confidence in the results of ES assessments.

Developing standards for ES can help identify tolerable levels of uncertainty, harmonise methodology for uncertainty assessment, and finally contribute to the broad adoption of ES information (Polasky et al., 2015). In the economic field, an underexplored research area is how stakeholders and the general public react to different types of uncertainty and how these uncertainties are treated in ES analyses.

Questions 15-17 focus on policy options for improving the provision of ES. There are two broad types of policy interventions (Weitzman, 2019): command-and-control regulations (based on "quantity") such as norms and standards, and incentives (based on "price" mechanisms) such as taxes, subsidies and tradable permits. Depending on the context, each of these two types of interventions may be the most effective, but in practice command-and-control regulations are often preferred. In Q15-17, we will focus on recently popularized incentive mechanisms for improving ES, i.e., payments for ES (PES), that are typically used in addition to command-and-control interventions.

Q15) How to evaluate the effectiveness of Payments for Ecosystem Services (PES)?

Context: Potential policy solutions to improve ES are many, e.g., government interventions, which take the form of command-and-control regulation and incentive-based mechanisms, which include for example taxes, subsidies and tradable permits. The class of incentive- or market-based mechanisms for environmental policy includes Payments for Ecosystem Services (PES), that are voluntary contracts between a landowner and the conservation buyer (typically the government or an NGO), in which the landowner receives a payment in exchange for adopting 'environmentally friendly' practices. Offering PES contracts to small landowners has emerged as a potential strategy to achieve conservation goals in both tropical and temperate areas (Wunder et al., 2018). A question of primary importance is to determine to what extent PES can contribute to achieve the conservation goal for which it has been designed. Despite the incentives they offer, PES may not be effective in changing agricultural practices.

In PES schemes where farmers receive the same payment regardless of the costs of contractual compliance, farmers who face the lowest costs for adopting green practices are the most likely to enter the scheme – referred to as the selection effect (Ferraro, 2008; Chabé-Ferret and Subervie, 2012; Jack and Jayachandran, 2019). As a result, a PES program may end up paying some farmers for doing nothing differently from what they would have done in the absence of any payment. In this case, the additionality of the PES may be quite small – or even null, highlighting the need to assess the effectiveness of PES empirically.

Obstacles: The additional (or causal) effect of a PES program is the difference between the practices of participants after enrolment in the program and what their practices would have been had they remained outside the program (i.e. the counterfactual situation). A major problem then arises in that the counterfactual situation cannot be observed and thus has to be estimated using observational data.

A number of recent studies have tried to identify the causal effect of agroenvironmental programs, whether it has been to evaluate the effectiveness of payments for forest conservation (Simonet et al., 2018 for example), conditional payments offered by the EU's Common Agricultural Policy (Chabé-Ferret and Subervie, 2013; Kuhfuss and Subervie, 2018) or the US Environmental Protection Agency Brownfields Program (Haninger et al., 2017), to give some recent examples. However, this type of study uses identification strategies that are based on assumptions that are typically not directly testable (Imbens and Wooldridge, 2009; Imbens and Rubin, 2015; Athey, 2017).

How to tackle the problem in the future: It is possible to use more robust evaluation techniques to identify the causal effect of agroenvironmental programs. The gold-standard method of evaluating agroenvironmental policies (such as PES programs) is the systematic use of randomised-controlled trials (RCTs). RCTs involve randomly selecting two groups of individuals or regions and implementing a PES-based program only for one group, keeping the second as a control. The difference between the groups provides a direct measure of success. However, few agroenvironmental programs have been tested in this way. There are several reasons for this lack of action, both on the demand side (e.g. the type of evaluations requested or accepted by various stakeholders such as governments, international agencies and farmers) and on the supply side (e.g. the incentives available to the consultants and scientists operating in this area) (Behaghel et al., 2019). However, to our knowledge, there are at least two notable exceptions: Jack, (2013) and Jayachandran et al. (2017), who evaluate the effectiveness of PES-based forestry programs in Malawi and Uganda, respectively. These studies suggest that the lack of randomised experiments in the field of conservation policies is not insurmountable. A special effort should thus be made in the area of PES evaluation, to explain the limitations of current programs and provide a framework for designing more effective ones if needed. It is important to emphasize that the implementation of randomised experiments to assess the effectiveness of PESs will only make sense if it is possible to measure a change in ES. This requires that the latter has been correctly defined beforehand, through the use of multifunctionality assessment (Q10) together with standards (Q7).

Q16) Should the design of a Payments for Environmental Services scheme account for the economic and social constraints of stakeholders who provide the ecosystem services?

Context: Payments for Environmental Services (PES) are usually in the form of a flow of payments that the service provider receives if they perform a specific 'environmentally-friendly'activity. Economic constraints such as limited credit access (Jayachandran, 2013), or social constraints such as the governance regime (Ostrom, 1990) can affect the effectiveness of PES programs.

In the absence of credit constraints, a flow of payments can be as effective as an upfront payment in providing incentives for ES provision. However, a credit-constrained ES provider will not find a flow of payment as attractive as an upfront payment. One plausible reason why PES designs including upfront payments are not used in developing nations is that the PES supplier would have difficulties fining the ES provider if they cannot make the specified effort (Jayachandran, 2013), and low-income ES providers may simply not be able to pay the required fine. It is therefore not clear whether flow payments are more effective than PES that prioritise upfront payments. This uncertainty raises a number of questions such as: (i) when do upfront payments dominate flow payments in terms of costs and benefits? (ii) what is the optimal mix between upfront payments and flow payments? (iii) Do upfront payments increase participation to PES programs? (iv) Do upfront payments increase the cost of PES programs?

Interventions that clarify property rights, such as formalizing land rights, may increase the efficacy of PES, as PES contracts can be properly designed, but this raises a number of open research questions: (i) Do improved property rights, such as formalizing land rights, increase PES efficacy? (ii) Do PES work better when property rights are individual rather than collective? (iii) How to design effective PES when the property rights are collective?

Obstacles: The first main obstacle is the lack of theoretical predictions, since theoretical models of incentive theory including constraints on the ES provider's side have not yet been widely explored. An exception is Quérou and Soubeyran (2020) who make a first step in this direction, and built a model in which the ES provider has limited financial means and the PES contracts may include flow and/or upfront payments. Quérou and Soubeyran (2020) show that the optimal PES design is generally a combination of upfront and flow payments. There has not been much development of such models largely because they are difficult to solve analytically. The second main obstacle is that there is little evidence on the link between credit constraints and participation in PES programs. Using observational data, Jayachandran (2013) provides evidence of a negative link between credit constraints and participation in a flow payments PES program implemented in Uganda. However, to our knowledge, there is no evidence based on laboratory or field experiments that confirms this first result. As regards property rights, Wren-Lewis et al. (2020) show that formalizing land rights reduces deforestation (in a study in Benin), an effect that seems to be due to improved tenure security and community forest management. There is, to our knowledge, no evidence based on experiments studying the role of improved property rights on the efficacy of PES.

How to tackle the problem in the future: The first step is to build a model in which the ES provider is financially constrained and that accounts for various important dimensions such as risk aversion or time discounting and the type of property rights, in order to derive theoretical predictions. The second step is, through a collaboration between economists, sociologists and ecologists, to measure ES indices and implement random control trials with stakeholders to test the theoretical predictions.

Q17) Can differentiated payments improve the effectiveness of Payments for Environmental Services (PES)?

Context: One feature of PES is that there is no differentiation regarding the amount of money being awarded, as different farmers or landowners obtain the same payment for the same number of hectares managed. Yet it is increasingly acknowledged that the provision of ES is spatially heterogeneous, in particular when there is a mismatch between the scale of a natural resource and the scale of farmers or landowners' property rights (Broch et al., 2013).

A second important feature is that potential ES providers possess heterogeneous opportunity costs (Jack and Jayachandran, 2019). Taking the example of a program designed to avoid deforestation in lands where agriculture and cattle ranching constitute alternative activities, the opportunity costs of PES participation would correspond to the costs of conserving forests on plots devoted to cattle ranching and crops. This raises the issue of the best form of PES that would ensure the provision of a given level of service at least cost to the regulating agency in charge of the program. Since certain areas matter more for the provision of a service, it might be more cost-efficient to differentiate payments based on the geographical location of farmers' land, but this raises the following questions: (i) when is a differentiated structure of PES more appropriate than a uniform structure? (ii) If ES providers' opportunity costs are heterogeneous, what is the most appropriate PES contractual design? For (i), Gueye (2019) provides initial predictions regarding the most appropriate design of PES under heterogeneous neighbouring effects and Gueye et al. (2020) provide experimental evidence suggesting a notable effect of differentiated payments on participation decisions. Regarding (ii), the trade-off is far from obvious. While differentiated payment schemes might allow to better tailor incentives depending on individual features, one must address the problem of asymmetric information in contract design, as ES providers have better information about their individual opportunity costs than public institutions funding the programs. Since such information is private, contract design should make sure that ES providers have incentives to reveal itnformation: the theory of incentives (Laffont and Martimort, 2002) suggests that ES providers will require larger payments than what they would be willing to accept to provide the appropriate conservation efforts, were information not private. Another important and related dimension is to assess whether action-based or results-based PES are most appropriate. Prendergast (2002) provides related insights in the case of general incentive schemes, while Derissen and Quaas (2013) focus on the case of PES and highlight that this is an important question, as a mix may be appropriate under environmental uncertainty and information asymmetry.

Obstacles: Little is known about appropriate contractual design in the specific setting of PES programs when information is asymmetric and there are several ES providers. Another important challenge relates to the implementation of random-control trials. This problem is particularly true in Europe, due to the difficulty in implementing this type of study into public policies, because of, among other reasons, pan-European law (see Behaghel et al. (2019) for a related discussion). Nevertheless, this implementation is not an impossible task, as several initiatives have already been employed (Jack 2013, Jayachandran et al., 2017).

How to tackle the problem in the future: The first step is to build a generic incentive theory model in which multiple ES providers interact, and where individual contribution decisions have heterogeneous effects on the other providers, or there is asymmetric information about ES providers' characteristics, in order to derive predictions. The second step is, through collaborative work between economists and ecologists, to implement laboratory experiments and random control trials to test the predictions. The appropriate design of geographically-differentiated payment schemes requires simultaneous knowledge of the ecological system and of the behavioral adjustment of individuals within the social system. Designing geographically-differentiated payment schemes requires an appropriate understanding of the ecological connections between land units, an assessment of the impact of management efforts on the level of ecosystem or environmental service, together with an assessment of farmers or landowners' willingness to receive payment in exchange for management efforts. Regarding the design of differentiated payments based on the heterogeneity in ES providers' characteristics, the assessment of opportunity costs in the providers' population also requires proper understanding of the functioning of the overall ecological system.

Q18) Can the design of Payments for Environmental Services (PES) schemes be improved by better considering the behaviour of providers of ecosystem service?

Context: Knowledge of ES providers' behaviour, i.e., how they how they make decisions and value their own benefits and costs as well as those of others, is crucial for the design of effective PES. It is now well acknowledged that economic agents' decisions often diverge from rational comparisons of financial benefits and costs (Ostrom, 1990; Kahneman, 2011). Using insights from behavioral economics has potential for improving the design of PES. For example, Ferraro (2008) show that non-monetary incentives based on both pro-social preferences (i.e., people give weight to the benefits of other people) and social comparisons (i.e., individuals judge their situation in comparison with the situation of other persons), may lead to greater conservation efforts. Outside the context of PES, several studies have analyzed participatory decisions in various programs and highlighted the important role of behavioral features, such as time inconsistency (i.e., an individual may disagree with the decisions they took in the past (Ashraf et al., 2006, Duflo et al., 2011, Le Cotty et al., 2019)), or intrinsic motivation signalling (i.e., people wish to signal their motivation for a cause (Bénabou and Tirole, 2006, Mellström and Johannesson, 2008)). Taking these behavioral dimensions into account may have a notable impact on the design of PES programs and on the associated costs and benefits. Important questions to be answered are: (i) how do social preferences, norms and other behavioural dimensions affect participation in PES programs? (ii) How should PES be designed to take these behavioural dimensions into account? (iii) Do these dimensions dominate PES schemes that do not already consider them in terms of costs and/or benefits?

Obstacles: Appropriate models are difficult to develop and lie at the frontier of two conceptually complex fields in economics: incentive theory and (theoretical) behavioral economics (Koszegi, 2014). In the context of PES contract design, Gueye (2019) and Soubeyran (2019) proposed an approach that focused on the design of cost effective PES programs when conservation efforts have to be coordinated, and they considered ES providers who exhibit social preferences (i.e., aversion to inequality or pro-social behaviour, respectively). Gueye et al. (2021) showed that the effect of aversion to inequality depended on the severity of the coordination problem and the type of inequality considered. When ES providers exhibit pro-social motivations, Soubeyran (2019) showed that such preferences lead to lower payments and increase inequality. These studies constitute a first step that may help derive predictions about the role of social preferences on the optimal PES design.

Evidence is lacking on how behavioural features affect the effectiveness of PES. Kuhfuss et al. (2016) implemented a choice experiment to study the effect of a collective bonus (conditional on collectively reaching a threshold level in terms of total acreage enrolled) in a scheme designed to induce less pesticide-intensive farming practices. These authors argued that a collective bonus can shift the pro environmental social norm and decrease the cost of the scheme. Gueye et al. (2020) provide experimental evidence on the role of social preferences in the adoption of a PES scheme with differentiated payments in the presence of neighboring effects. These authors show that subjects' decisions are more likely to be driven by pro-social motivations than by aversion to inequality.

How to tackle the problem in the future: The first step is to build a model using insights from incentive theory and behavioral economics in order to derive predictions about ES providers' participation decisions or about the optimal design of PES schemes. The second step is, as for Q17) to measure ES indices and implement random control trials either to test the predictions on ES providers' decisions or to test new PES designs based on the optimal PES scheme derived from the model.

4. Discussion

The 18 research questions, obstacles and solutions that we developed, illustrate that research into agroecosystem multifunctionality must be a hugely collaborative effort, combining knowledge and expertise from a broad range of disciplines, including agronomy, biology, ecology, economics, mathematics and social sciences. Within an agroecosystem, spatial aspects are determinant of many ecological processes, and also influence significantly the provision of supporting and regulating services. Knowledge gaps relating to the effects of spatial scales and the organisation of complex spatial units within a landscape were highlighted throughout the questions asked in this paper [Q 3, 5, 6, 7, 10]. To fully advance development in this field of research, we need to take an approach similar to that of 'One Health,' adopted by the United Nations in 2008 (AVMA (American Veterinary Medical Association), 2008). One Health is an integrated, unifying approach that aims to sustainably balance and optimize the health of people, animals and ecosystems. By designing and implementing programmes, policies, legislation and research, in which multiple disciplines, secteurs and communities work together, it contributes strongly to sustainable development. Applied to an agroecosystem and its diversity of spatial units, the One Health approach would be particularly relevant for the control of pests affecting crop health and zoonoses. Working closely together, scientists and practitioners with a range of expertise in different disciplines, would be able to better predict how spatial proximity affects the spread of vectors in agroecosystems, and how to design landscapes that limit the spread of pests and microbes, whilst providing multiple services.

To encourage communication and programs across sectors, disciplines and communities at varying levels of society, the concept of Living Labs (LL) could be used in combination with the One Health approach to implement the co-construction of projects and research initiatives [Q 2, 4, 5]. The European Network of LLs (ENoLL https://enoll.org/) defines Living Labs as user-centered, open innovation ecosystems based on systematic user co-creation approach, integrating research and innovation processes in real life communities and settings. The Living Lab approach therefore builds on multi-stakeholder requirements and extends them in the dimensions of time, geography (agroecosystem and landscape scales) and content (innovative transdisciplinary research). With the creation of an international Agroecosystems Living Laboratories (ALL) working group formed at the 2018 G20 Meeting of Agricultural Chief Scientists, a framework was developed to identify the defining characteristics of ALLs, and to foster dialogue, standardization, and the sharing of knowledge and data (McPhee et al., 2021). Promoting data-sharing through improved semantic terminology, open-access and interoperability is a necessity to advance any research field, especially when transdisciplinary issues are involved [Q 8, 13, 14]. Living Labs also encourage dialogue with stakeholders, so that issues such as how perception, preferences and incentivization drive management choices can be examined and understood better. Experimental studies in behavioural economics and random control trials performed in collaboration with stakeholders in Living Labs, would allow major advances in the understanding and adoption of incentivization schemes, that could then be used to create new policy mechanisms [Q 16, 17, 18].

Although the concept of ES has made it possible to unify efforts to conserve biodiversity and our ecosystems, it is criticized (Schröter et al., 2014), because the vision of ES is largely anthropocentric and reduced to a utilitarian vision. Biodiversity plays a role in defining ES and the effectiveness of ES depends on the local social and economic context. ES are also often indirect and in the long term, require ethical and political choices. As social-ecological systems are traditionally polarized into human and non-human components (Naess, 1973), the ES concept impacts the way we see and manage ecosystems, with humans as the target and the ecosystem as the source or provider. If we consider only a limited number of ES within an agroecosystem, such an oriented and utilitarian view can lead to a dangerous overexploitation of that agroecosystem. Therefore, along with an understanding of changing societal values (Fig. 1), the creation and management of multifunctional agroecosystems that also consider, incorporate and promote biophysical and ecological processes that improve ES with added or multiple values (even though economic value is reduced) must be reinforced [Q7]. Biodiversity is now the key factor to consider, and several studies have shown that an increase in biodiversity in a social-ecological system leads to improved ES provision, resulting in a win-win situation (Pasari et al., 2013; Mao et al., 2021; Wittwer et al., 2021). We ask how modelling approaches can better incorporate the role of biodiversity in multifunctional agroecosystems, taking into account bundles of ES and their interactions with complex biophysical and ecological processes in time and space [Q9, 10, 11]. Biodiversity indicators can also be used to reveal changes occurring within agroecosystems and their capacity to provide ES, but major conceptual and experimental advances must be made in the understanding of dynamic changes in the biodiversity-function-nexus across landscapes [Q1, 2, 3, 5]. Bundles of ES and the way they interact with each other and are influenced throughout space and time are particularly challenging to model, especially when big bundles occur in complex social-ecological systems [Q 10, 12]. New modelling approaches are now able to handle big bundles of ES within a dynmically changing system, and can identify cascades of ecological and socio-economic consequences under different management scenarios (Gaucherel et al., 2021; Mao et al., 2021). Applying these models to agroecosystems will go a long way to identifying diverse management options for optimising multifunctionality.

Global environmental change is leading to both long-term shifts in average conditions as well as potentially dramatic changes in environmental variation. Although we did not address directly the uncertain effects of climate change on agroecosystem multifunctionality and ES provision, many of the solutions identified would be affected by sudden and extreme variations in climate and we need to understand how agroecosystems will respond to such changes. Long-term plans and strategic decisions therefore need to be made based on uncertain information about future climate, social and economic situations [Q14]. Ignoring this uncertainty could mean that we limit our ability to adapt and so result in missed chances and opportunities. Compounded climatic events also cause a cascade of processes, and trade-offs and synergies between different sectors in agroecosystems and increase the complexity of strategic planning challenges. Ensuring that the decision context and processes that govern agroecosystems are considered throughout the whole process, from problem setting, multiple plausible climate scenarios, diverse socio-economic factors (e.g., prices, costs and demand), to decision-making, will allow stakeholders and policy-makers to (i) translate dynamic management and climate feedback into decision objectives and constraints, (ii) identify adaptation and tipping points and (iii) develop social ecological economic pathways that are adaptive over time and during deep uncertainty. We stress therefore, as indicated throughout the majority of our questions, barriers and solutions, that the co-construction of long-term projects and programs across sectors, disciplines and communities must be encouraged at varying levels of society.

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 data

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