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LAND MANAGEMENT IN CENTRAL ASIA

4.- Land as a Natural and Ecological Resource

Reading

Ecological Role of Land Resources

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Land use and ecosystem services

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LAND USE AND ECOSYSTEM SERVICES

Julien Hardelin and Jussi Lankoski, OECD

This report assesses the crucial drivers of ecosystem services and proposes actions to develop a more effective policy mix. Several elements form the basis of this report. First, a literature review provides an overview of the state and trends of ecosystem services linked to agriculture, including issues related to land use. Secondly, results are presented from a quantitative model developed to illustrate the potential benefits of improving policy design as well as to investigate synergies and trade-offs among ecosystem services. This report also includes a review of experiences in an inventory of ecosystems in selected countries and policy initiatives that address ecosystem services linked to agriculture.

Keywords: Biodiversity, water quality, environmental tax, payment for ecosystem services, conservation auction, spatial targeting

JEL codes: Q15, Q18, Q57, Q58

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Executive Summary

Agriculture faces the unprecedented challenge of increasing production in order to satisfy a growing demand for food, whilst preserving natural ecosystems. Agriculture has a three-fold relationship with ecosystems. First, despite improvements over the past 20 years its *negative impact* on ecosystems continues to be significant in many OECD countries. Second, if sustainably managed, agriculture can contribute to the *provision* of many ecosystem services, such as water regulation, flood risk mitigation, biodiversity and carbon sequestration that benefit society. Third, while agriculture is a provider of ecosystem services, it is also *dependent* on ecosystems such as the biological control of pests and diseases, soil fertility, and pollination. Thus, degradation of ecosystems constitutes a serious threat to the long-term productivity growth of agriculture.

Several market failures are associated with ecosystem management in agriculture and can impact each of the relationships noted above. It is therefore critical to develop the right set of policies to address the complex interplay between multiple ecosystem services and agriculture. However, ecosystem services are often produced as bundles, and policies that target specific services may affect other services negatively or positively. For example, while a policy that increases commodity production per hectare increases the supply of *provisioning* services (food, fibre, fuel, medicinal and genetic resources), it can also reduce the supply of *ecosystem* services (such as water quality or biodiversity preservation) if it is associated with increased input use (such as fertilisers and pesticides). There are trade-offs between ecosystem service provision and the production of food, feed, and fibre, and the challenge for policy is to find ways and means to facilitate the enhancement of ecosystem services with increased agricultural production.

This report aims to clarify the complexity involved in designing policies by assessing the crucial drivers of ecosystem service provision and outlining the ways and means to achieve a better policy mix. The report builds on: i) a literature review on the state and trends of ecosystem services linked to agriculture, including land use; ii) a quantitative model specifically developed for this project to illustrate the potential benefits of improving policy design by investigating synergies and trade-offs among ecosystem services, analysing the importance of market and policy drivers in ecosystem service provision, and assessing the performance of current agricultural and agri-environmental policy packages in promoting a balanced set of ecosystem services; and iii) a review of selected country experiences in national ecosystem assessments and policy initiatives regarding ecosystem services linked to agriculture.

Most OECD countries have policies that address the limitations of markets (market failures) to provide ecosystem services in agriculture, although the effectiveness of policies can sometimes be improved. Moreover, policy drivers that can interfere with the provision of ecosystem services – in particular market price support and area-based crop specific payments – are still in place in many countries. The model developed for this report suggests that an increase in a crop area payment or a commodity price results in increased supply of commodity production (provisioning services) and a significant reduction of ecosystem services (biodiversity and water quality), so that social welfare is reduced relative to situations in the absence of the policy intervention.

The model results suggest that a policy package that combines decoupled area and agri-environmental payments, in conjunction with well-selected environmental compliance measures, performs much better than other forms of policy mix. This could be further improved by incorporating mechanisms that vary payments according to environmental benefits, for example by using an environmental benefit index to allocate payments. At this stage, the results of the model application are based on data from one country (Finland) and focus only on a subset of ecosystem services related to agriculture.

Enhancing the provision of ecosystem services and reducing pressure on ecosystems requires a structured approach. The decision tree proposed in this report focuses on identifying cost-efficient policy instruments through:

- assessing the agriculture-ecosystem relationships in terms of the pressure on ecosystems, the provision of, and the dependence on ecosystem services;
- identifying the most distorting signals to see how these can be eased, while taking into account synergies and trade-offs among different ecosystem services;
- setting policy priorities by taking into consideration the diversity of ecosystem services, their interactions in terms of synergies and trade-offs; and
- determining the optimal mix of policy instruments to achieve policy objectives regarding ecosystem service provision.

National ecosystem assessments are not yet fully integrated in the policy-making process in most countries. Moreover, structured and comprehensive evaluations of the impact of existing policy mixes on ecosystem services are often lacking. Such evaluations have the potential to foster concrete policy reforms in the short-run and be integrated in strategic plans in the long-term. The ecosystem assessments carried-out in the United Kingdom, the People’s Republic of China, and the European Union illustrate the benefits of creating a link between assessment and policy action with follow-up phases undertaken by, for example, specific institutions charged with providing independent evidence and advice to governments.

1. The relationship between agriculture and ecosystems

Key messages

- Ecosystem degradation constitutes a serious threat to the long-term productivity growth of agriculture. This calls for specific action to develop the necessary policy responses that will protect the ecosystems on which agriculture relies. It is therefore essential to better manage the relationships between agriculture and ecosystems to ensure sustainable productivity growth in agriculture.
- While there is a downward trend in ecosystem degradation from agriculture due to a combination of policy changes (environmental policies, reform of agricultural policies) in several OECD countries, the economic, environmental and health costs of ecosystem degradation due to agriculture is still significant, and pressure from agriculture on ecosystems continues to increase in several OECD and partner countries.
- There is significant potential for agriculture to provide more ecosystem services in most OECD countries. Beyond the necessary reductions of harmful impacts on ecosystems, agriculture could provide an economically significant bundle of ecosystem services – such as water regulation, flood risk mitigation or carbon sequestration – that would have clear benefits for society.
- There is evidence that several farm practices can deliver multiple ecosystem services, while maintaining or even increasing production. While such synergies between agricultural production and ecosystem services are possible, better evidence and dissemination of information are needed. The ability to take advantage of these synergies is important in order to ensure cost-efficiency and the acceptability of policies by farmers.
- However, there is a lack of policy investment in this area, which is also explained by the costs of implementation as well as the challenges in identifying and accounting for trade-offs and synergies among these services and the production of food, feed, fibre, and fuel.

Ecosystem services are “the contributions of ecosystems to benefits used in economic and other human activity”. The term “services” is considered to refer to both tangible (e.g. timber) and intangible services (e.g. scenic beauty of a landscape). Ecosystem services include various and diverse categories, such as *provisioning* services like food and timber; *regulating* services like carbon storage and flood control; and *cultural* services like tourism and landscape amenities. To understand the diversity of ecosystem services, several classification systems have been developed since the early 2000s, the most recent effort being the Common International Classification of Ecosystem Services (Annex 1.A1).

Economic sectors produce a set of outputs that use a combination of inputs (labour, capital, fertilisers, etc.) and ecosystem assets[1] (natural capital). Ecosystem assets are considered as stocks, and ecosystem services as flows. Figure 1 presents the economic framework linking ecosystem services, production and human well-being. The interactions between economic sectors and ecosystems are three-fold (see also the stylised relationship in Figure 2).

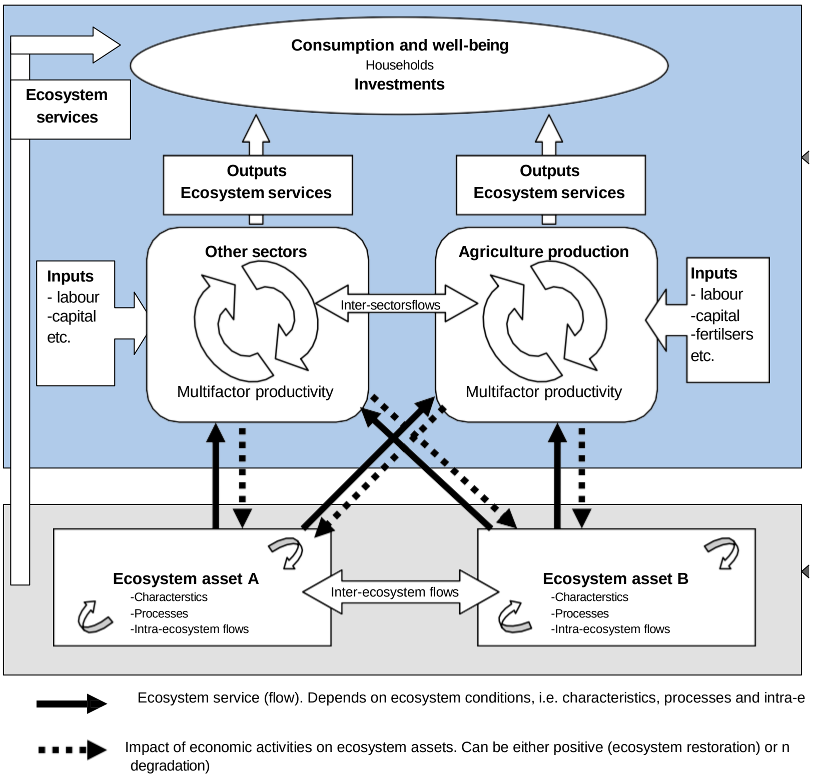
- *Pressure on ecosystems*: Economic sectors can be source of pressure on ecosystems, leading to the degradation and reduced capacity of the ecosystem to provide services (dashed arrows).

- *Dependence on ecosystem services*: Ecosystem services enter as inputs in the production process or have value for people (black arrows).

- *Provision of ecosystem services*: Economic sectors contribute to the provision of ecosystem services, including provisioning services (commodities, which are marketable), and regulating,maintenance and cultural services (many of which are not marketable).

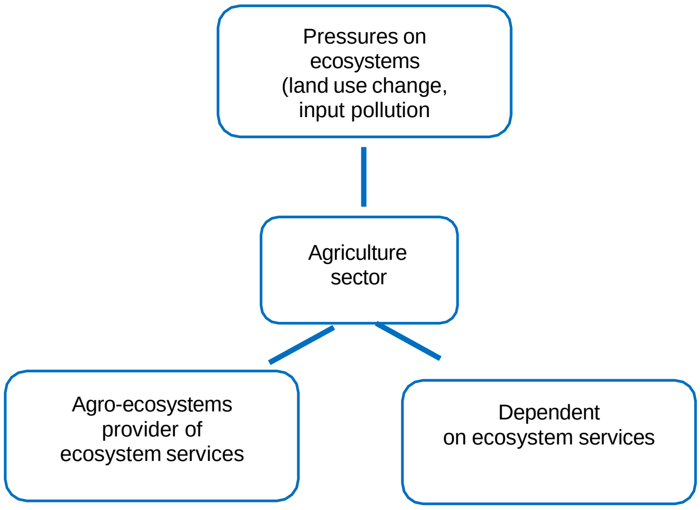
1. Ecosystem assets are defined in the SEEA-Experimental Ecosystem Accounting as “spatial areas comprising a combination of biotic and abiotic components and other elements which function together”. Ecosystem assets can be described by their *biochemical characteristics and processes* (i.e. changes or reactions, such as nutrient cycling and energy flows) and *functions* (i.e. potential of ecosystem to provide goods and services) (TEEB, 2009; de Groot et al., 2010; Haines-Young and Potschin, 2009). For instance, soil structure and fertility, nutrient cycling, water provision, and genetic biodiversity are important characteristics of ecosystem assets that matter for agricultural production (Power, 2010; Zhang et al., 2007). In t this report, the terms ecosystem assets and natural capital are considered as equivalent. This is a simplification, as natural capital also includes natural resources such as oil or water. See Annex 1.A2 for a brief history of the concept of ecosystem services.

Figure 1. Ecosystem assets contribute to productivity and human well-being via the ecosystem services they generate



Source: Adapted from United Nations, European Commission, Food and Agriculture Organisation, OECD and World Bank (2012).

Figure 2. The triptych of the relationship between ecosystem services and agriculture



Agriculture pressure on ecosystems is decreasing, but continues to have large economic and social costs in several regions

According to the Millennium Ecosystem Assessment and the European Environment Agency, there are five main pressures on terrestrial ecosystems, all economic sectors combined (Table 1): land use change; overexploitation of natural resources such as water; pollution due to input use (fertilisers, pesticides, industrial chemicals); invasive species; and climate change .

Pressures on ecosystems from agriculture include; water and air pollution (e.g. nitrogen deposition); habitat loss and fragmentation; and contribution to climate change. Since the 1950s, the factors that have tended to increase the pressure of agriculture on ecosystems include the intensification of farming practices (e.g. use of chemicals and synthetic fertilisers, irrigation, intensive grazing); changes in farmland uses, including a move towards specialisation and the uniformity of landscapes; and land conversion to and from agriculture.

On average, in OECD countries, agriculture accounts for 36% of land use, 44% of freshwater withdrawals, a high share of consumptive water use,[2] and it is a significant source of emissions of nitrogen, phosphorus, pesticides and ammonia (Table 2). According to the *OECD Environmental Outlook*, agriculture globally has been the largest driver of terrestrial biodiversity and ecosystem loss. In the European Union, the most reported pressures on natural habitats are agriculture and changes to water bodies.

Table 1. Main pressures causing ecosystem changes

Pressures	Description
Land use change	Land use change from natural habitat to alternative forms of land use can cause land fragmentation, soil sealing, soil erosion and soil degradation that can cause direct degradation of a habitat or its loss and replacement by another habitat type. For some areas, the abandonment of farmland leading to replacement by shrub or forest is also significant. For freshwater ecosystems, the main pressures are human modifications such as the creation of dams and diversion of rivers
Pollution due to input use	Pollution and nutrient enrichment occur when excessive harmful components such as excess fertilisers, pesticides and industrial chemicals are introduced into an ecosystem and exceed its capacity to maintain their natural balance and resulting in ending up in the soil, surface and ground water and seas, leading to ecosystem changes
Overexploitation (unsustainablewater use or management)	Pressures arise from the use of ecosystems for production of food, fuel and fibre, especially in case of intensive land management and overexploitation of natural resources, such as over-extraction of water
Invasive alien species	Invasive alien species can replace native species, occupying their habitats, reducing their survival and abundance and leading to loss of biodiversity
Climate change	Anthropogenic climate change causes fluctuations in the life cycles of plants and animals and extreme events such as floods, droughts and fires that change the health and characteristics of habitats and the species present

Source: European Environment Agency (2016a).

2. The share of agriculture in water consumption use is generally higher than its share in water withdrawals because for several uses (e.g. urban residential), a significant share of withdrawals may be put back into the original water source via return flows. In some countries, agriculture can be responsible for up to 80-90% of water consumption.

Table 2. The important role of agriculture regarding natural resources and the environment

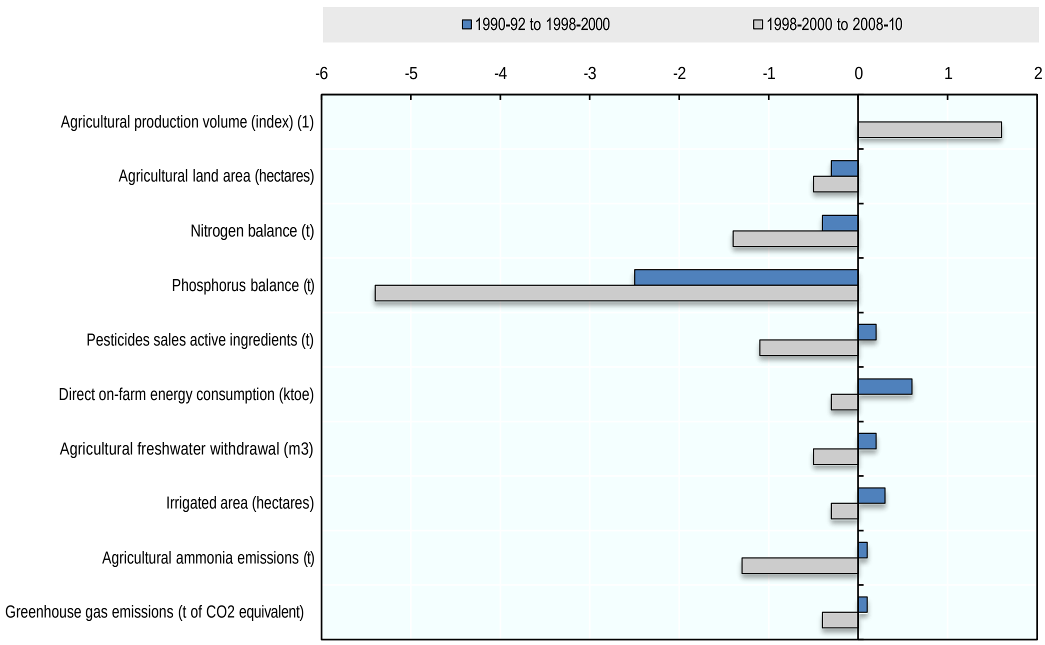
Share of primary agriculture (%)	OECD	World
GDP	2.6%	2%-60%
Land area	36%	30%-55%
Water withdrawals	44%	70%
Greenhouse gas emissions	8%	11%
Ammonia emissions	91%	80-99%

Note: GHG emissions do not include emissions due to land use and land use change, such as deforestation
Source: OECD (2013).

Pressures from agriculture on ecosystems exhibit large geographical variations, from low-pressure areas to "hot spot" areas, associated with agricultural intensification and irrigation. In France, for example, the average number of pesticides detected per surveillance point is equal to 16, but this figure varies between 6 (Rhone, Mediterranean, Corsica) and 29 (Artois-Picardie) depending on the catchment area considered , while nitrogen surpluses range from 16 kg to 69 kg of nitrogen per hectare of agricultural land. .

Overall, pressure from agriculture on ecosystems trended downward from 1990 to 2010 in most OECD countries. Figure 3 show the significant reduction in nutrient surplus, pesticide use, and emissions of ammonia and greenhouse gases by the agriculture sector in the decade since 2000, even though the volume of agricultural production continued to grow at about 1.5% per year. Pressure on ecosystems nevertheless remains high in several countries and regions, and some indicators – such as those of biodiversity (as indicated by the index of farmland birds) and water quality – are worsening (OECD, 2013).

Figure 3. Trends in agri-environmental performances in OECD countries



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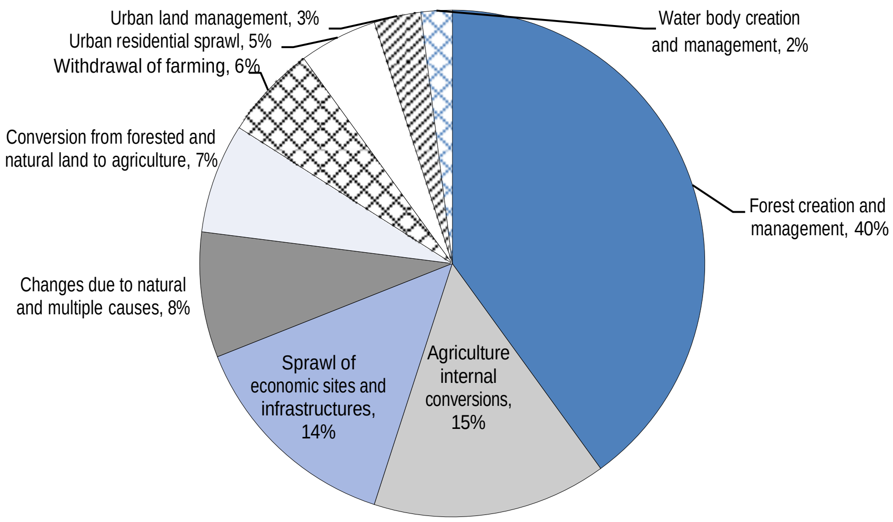
Source: OECD (2013).

Agricultural land area has decreased in most OECD countries since 1990, which suggests a fall in the pressure agriculture places on ecosystems in terms of land use change. In the 2000s, agricultural land area decreased by 0.5% per year in the OECD area (0.4% per year in EU15). The impact this has had on ecosystems depends on the type and extent of land use replacing agriculture. In the case of urban land, the impact may be negative on biodiversity and habitats. In the case of forestry, it is more likely to be positive, although this will depend on the type of forestry practice used. With regard to the connection between land-use change and ecosystem services, some sources of change (e.g. development) tend to be irreversible, which can have more significant impact for the potential to reallocate land use in the future to meet environmental goals.

The overall decline in agricultural land area in the 2000s most likely masks more complex patterns of land use changes. Data for the European Union show that cessation of farming accounted for 6% of total land use change, but that conversion from forested and natural land to agriculture represented 7% of total land use change (Figure 4). These data suggest more dynamic patterns of land use change than is apparent from aggregate numbers of land use. In addition, changes to the mix of crops cultivated accounted for 15% of total land use changes, which is the second highest form of land conversion after forest creation (40%). In the European Union, for example, the risks due to soil sealing (covering soils by impermeable material, such as buildings and road construction) in preventing natural soil functions and ecosystem services is increasingly recognised in policy strategies and research. In the context of the Soil Thematic Strategy, the European Commission points to the need to develop best practices to mitigate the negative effects of sealing on soil functions. In the Roadmap to a Resource Efficient Europe, the European Commission proposes that by 2020 EU policies take into account the impact of soil sealing on land use with the aim to achieve no net land taken through soil sealing by 2050.[3]

Figure 4. Drivers of land use change in the European Union, 2000-2006



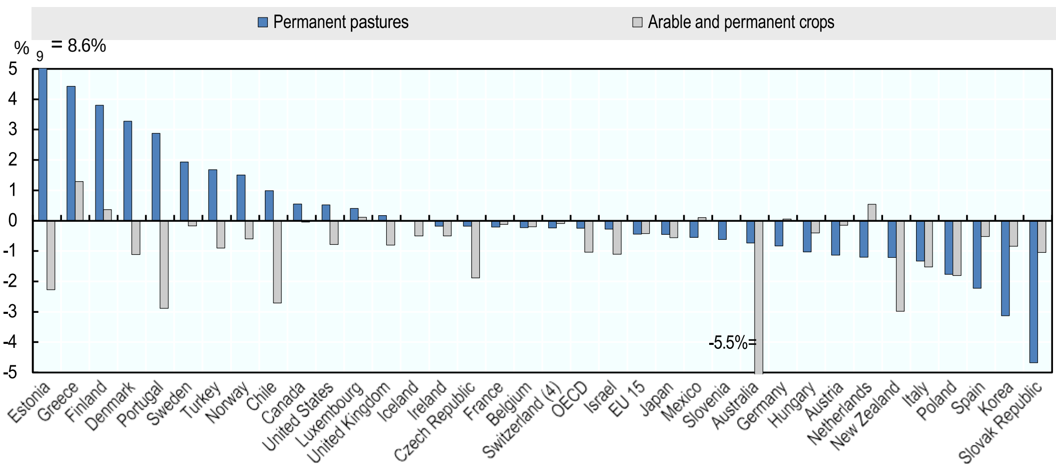
Note: Agricultural internal conversion relates to the change in crop structure on agricultural land areas.
Source: Based on European Environment Agency (2016).

3. The European Environment Agency provides specific assessment based on Corine Land cover data: http://www.eea.europa.eu/articles/urban-soil-sealing-in-europe.

The changes in the areas of crop and pasture lands, and thus the potential pressures they create, show marked differences across OECD countries since 1990 (Figure 5). This has likely affected the bundle of ecosystem services provided by agro-ecosystems. Pasture land is, in general, more likely to exhibit lower input intensity than croplands, and to provide more diversified bundles of ecosystem services, although caution must be exercised in making general statements.

Pressures on ecosystems from agriculture continue to represent significant costs for society. Many studies have used monetary valuation techniques to provide estimates of the costs from agricultural pollution, such as damage from nutrients (health and eutrophication costs) and pesticides (additional pesticide removal treatment costs). Recent examples include a French study, which estimated that social costs from nutrient and pesticide pollution range between EUR 0.9 to EUR 2.9 billion per year. These estimated costs included: water, air and soil pollution; greenhouse gas emissions; and damage to biodiversity. Moreover, reactive forms of nitrogen such as NOx, NH3, N_2O and nitrate also have negative health impacts.

Figure 5. Changes in permanent pasture, and arable and permanent crop land area in OECD countries, 2000-2010



Note: The statistical data for Israel are supplied by and under the responsibility of the relevant Israeli authorities. The use of such data by the OECD is without prejudice to the status of the Golan Heights, East Jerusalem and Israeli settlements in the West Bank under the terms of international law.
Source: OECD (2013).

The potential of agriculture to provide ecosystem services is significant, but remains underdeveloped

The economic value of potential services from agro-ecosystems could be significant

Agriculture can potentially provide a diversified bundle of ecosystem services from agro-ecosystems: *provisioning* services (food, wild food, feed, fibres, bioenergy, genetic, medicinal and ornamental resources), *regulating* services (protection from disturbance such as flood, erosion prevention, climate change mitigation, waste treatment, wildlife habitat, wild species diversity) and *cultural* services (aesthetic landscapes, tourism and leisure, religious and spiritual). In general, agro-ecosystems are, by the very nature of the agricultural activity, specialised in provisioning services such as food and feed, often at the expense of other ecosystem services. However, the provision of ecosystem services essentially depends on agricultural land cover (for example, cropland versus pasture) and management practices. For example, water quality and quantity may be either enhanced or degraded by agriculture.

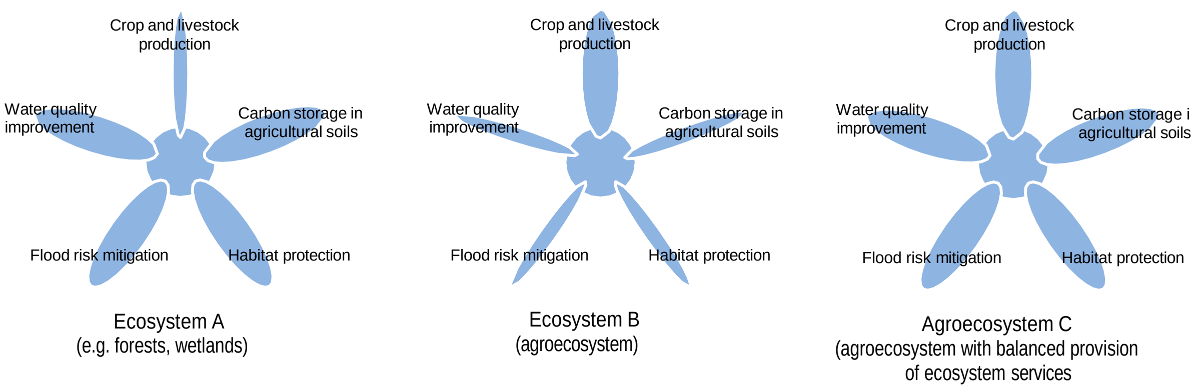
The term “bundle of ecosystem services” generally means that a set of services are associated in some way such that changes in the provision of one ecosystem service can affect the supply of others. This can be, for example, the set of services associated with a particular place or landscape. Alternatively, they may be functionally or causally linked. A simple representation of an ecosystem service bundle is the “flower-diagrams”, which can be used to depict the state and trends amongst a common set of ecosystem services.

Figure 6 provides an illustrative example of such a flower diagram. Natural or semi-natural ecosystems, such as forests and wetlands, typically provide regulating services such as habitat and biodiversity preservation, water quality regulation or carbon sequestration. By contrast, intensive cropland mainly offers crop-provisioning services. Converting land from natural ecosystems to cropland can thus significantly change the bundle of ecosystem services in favour of provisioning services. However, adoption of certain farm management practices on cropland can provide a more diversified bundle of ecosystem services including provisioning and regulating services.

Most ecosystem services linked to agriculture are non-marketed (or non-marketable) services, with the notable exception of food, feed and fibre production. For instance, the economic value of crop, livestock and timber production is given by their established market prices, which is not the case for many other services, such as wildlife habitat provision. Regulating services, such as carbon sequestration and water quality enhancement or degradation, fall somewhere between in that they do not have a clear market price, but their value could be estimated in terms of the opportunity cost, for example, of building more or fewer water treatment facilities.

Ecosystem services do matter for the economy: in some specific locations, the value of regulation services from agro-ecosystems and land uses may be large. For example, the Czech Republic and France have recently estimated the economic value of services provided by grasslands. Figure 7 shows the importance of regulating services in the Czech Republic for different types of grassland systems. Table 3 shows that services such as the regulation of water flows and erosion control represent a high economic value on French pastures (Table 3).

Figure 6. Agricultural systems and the provision of bundles of ecosystem services



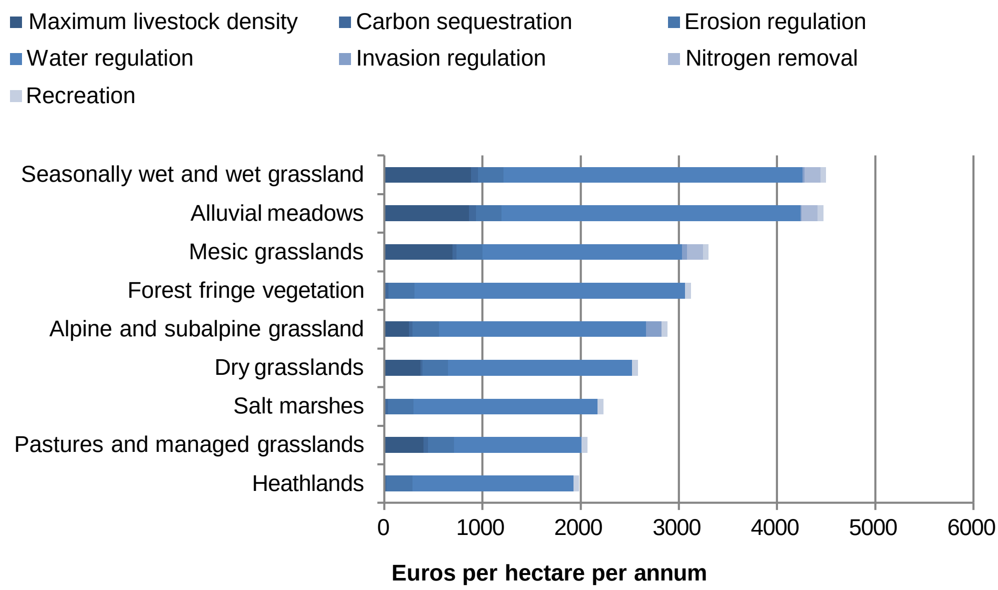
Note: Each flower represents the bundle of ecosystem services provided by agro-ecosystems. The size of each petal symbolises the quantity of ecosystem services supplied.
Source: Based on Foley (2005).

Table 3. Regulation and cultural services provide high social value on French pastures

Ecosystem service	Permanent pasture Value (EUR/ha/year))	Permanent pasture Method	Wet pastures Value (EUR/ha/year))	Wet pastures Method
Regulation services				
Carbon sequestration	20-50	Cost approach	Not assessed	-
Carbon storage	320	Cost approach	Not assessed	-
Other gas	Not assessed	-	Not assessed	-
Water availability	0	-	Not assessed	-
Water flow regulation	Not assessed	-	Not assessed	-
Water quality	90	Avoided costs	70-130	Cost of substitution
Soil protection (erosion, floods)	Not assessed	-	60-300	Replacement costs
Pollination	60-80	Through productivity losses	Not assessed	-
Biodiversity	Not assessed directly	-	Not assessed directly	-
Other regulation services (health, etc.)	Not assessed	-	Not assessed	-
Provisioning services				
Livestock products	Not assessed		260-1800	Market prices
Picking (excluding recreational activities)	Not assessed			
Cultural services				
Hunting	4-70	Revealed preferences		Market prices
Walking	Not assessed		290-1170	Stated preferences
Hiking	Not assessed			
Landscape amenities	60	Value transfer		
Educational and scientific values	Not assessed	-	10-15	Value transfer based on stated preferences
Non-use value of biodiversity	Not assessed	-	220-870	Stated preferences
TOTAL	600 (order of magnitude)		1 100-4 600 (order of magnitude)	

Source: CGDD (2012).

Figure 7. Pastures provide economically significant ecosystem services such as water regulation, carbon sequestration and nitrogen removal in the Czech Republic



Source: Vačkář D. et al. (2011).

Agro-ecosystems face significant challenges to fully realise their economic potential

While there is increasing evidence that agriculture could provide a more diversified bundle of ecosystems services, there is serious concern that such diversification would be at the expense of food, feed and fibre production. show in a meta-analysis that about two-thirds of articles on ecosystem services deal with such trade-offs, whereas one-third deal with synergies. Highly specialised agricultural systems may see their productivity reduced substantially if they are required to provide regulating services. In cases where the estimated benefits of diversifying the bundle may be higher than the costs in systems with a high degree of specialisation and economies of scale, one challenge is to prioritise policies that focus on reducing pressures placed on ecosystems by agriculture and to look for potential synergies between ecosystem services, so that the impact on productivity is minimised.

A second challenge to realising the full potential of ecosystem services is the difficulty to define and measure ecosystem services in practice. Measurement challenges include, in particular, the lack of data on physical flows of ecosystem services and the lack of knowledge on the influence of certain farm practices on the provision of the different ecosystem services. Information and measurement gaps concern not only physical indicators, but also the monetary valuation of ecosystem services (Box 1).

A third challenge to the provision of ecosystem services is spatial scale. At the landscape level (as compared to the farm level), the provision of ecosystem services depends on land composition (the distribution of different types of land use). Spatial scale is a key element of the provision of ecosystem services: different types of land use typically provide different bundles of ecosystem service per unit area, implying that land use composition affects the provision of ecosystem services. In addition, some ecosystem services, such as soil retention (riparian vegetation, floodplains) and water purification services are typically supplied at the landscape level and may exhibit economies or diseconomies of scale. For example, an additional area of floodplain in a landscape without floodplains is likely to provide higher marginal benefits in terms of flood risk prevention than in a landscape with an already significant floodplain area. Another important feature of several ecosystem services is their "spatial non-fungibility" - that their economic value is highly location-specific. This is the case for several non-market services such as wildlife habitat and recreational services.

Some ecosystem services not only depend on land *composition* but also on landscape *patterns* due to complex spatial interactions between units of lands (e.g. homogenous landscapes versus presence of patches). Landscape has a certain *composition* (e.g. proportion of non-arable land, diversity of land covers, distribution and richness of habitat patches) but also a *structure* (e.g. number, size, shape, diversity and connectivity of patches) which can affect ecosystem services provided at the landscape level. The potential for increasing the provision of ecosystem services through modifying landscape patterns (and not just composition)is potentially high for pollination, pest control, flood control, water purification, recreation, and biodiversity and habitat provision for mega fauna (Box 2).

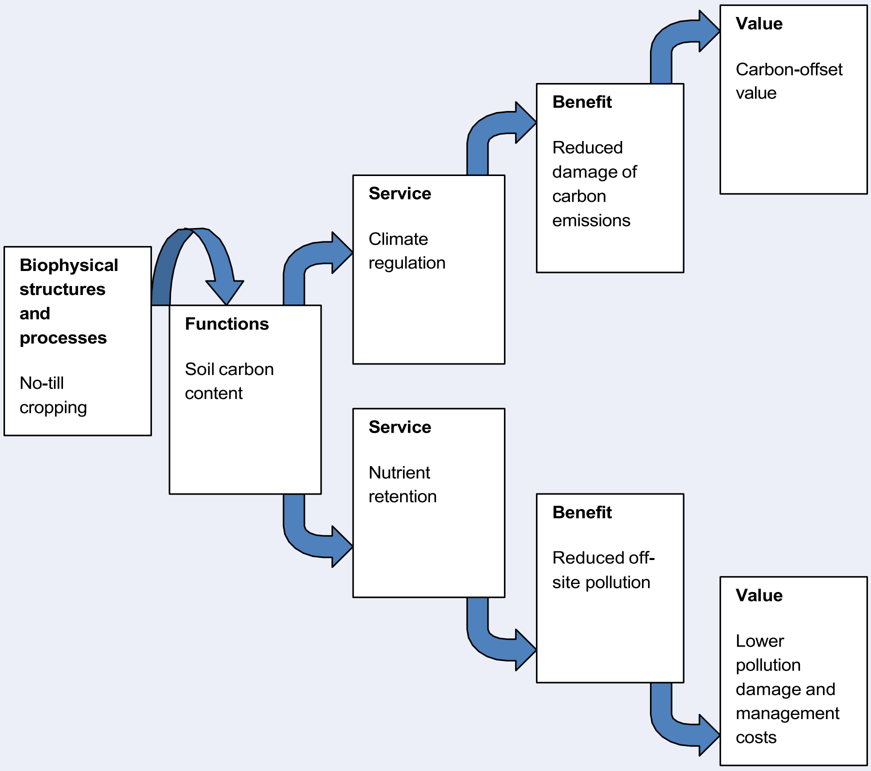
Ecosystem services also vary in time, and benefits from improving ecosystem services generally require investments that pay off far in the future and are uncertain. The relative prices of ecosystem services may change in time, due to shifts in preferences or to the evolving supply of some services (becoming rare, their relative value for society increases). These aspects are particularly challenging when building scenarios of ecosystem services for policy guidance, and how to use these scenarios in decision tools (modelling, projections). The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) recently reviewed approaches for building scenarios related to ecosystem services, and how to develop best practices to better help policy makers in this area.

Box 1. Ecosystem services can have large economic benefits, but how to measure and manage them in practice?

Measuring ecosystem services directly is often difficult, hence the need to rely on second-best evidence-based proxies. Various studies sometimes use different metrics to characterise similar ecosystem services. Direct measurement of these services is often costly and plagued by uncertainty. Good proxies for ecosystem services should account for spatial and temporal variations. For example, soil fertility or soil qualities are multidimensional concepts with a high degree of local specificity. Carbon is not always a good proxy: the relationship between certain characteristics of soils, such as carbon content, has some impact on crop productivity, but there is not a robust relationship between carbon content and crop yields.

Knowing how farm practices and ecosystem conditions affect ecosystem services is crucial to simple and efficient policy design but remains difficult in practice. Changes in farming practices can affect several ecosystem services simultaneously, with potential synergies or trade-offs. When available, the evidence base and quantification of these trade-offs and synergies is generally site-specific. Synergies imply that a single farm practice can have co-benefits whereas a trade-off implies there could be negative impacts on some services and positive impacts on others depending on the changes in farming practices. Figure 8 illustrates the mechanism behind how no-till cultivation can simultaneously improve two ecosystem services: climate regulation and nutrient retention.

Figure 8. One practice, two ecosystem services: Illustrative example from no-till



Monetary values have the theoretical advantage of being a common metric for all ecosystem services. But while information gaps regarding the monetary value of ecosystem services apply to all types of ecosystems, these are more challenging to establish for agro-ecosystems,

Despite this, solutions are increasingly being developed to better measure ecosystem services and new information and communication technologies will also help. For example, map bundles of ecosystem services in the heterogeneous landscape of the Norrström watershed in Sweden, using spatial data and geographical information systems. They assessed 16 terrestrial ecosystem services (six provisioning, five regulating, and five cultural) across 62 municipalities The indicators were standardised according to the municipal area and have the same data range so that the correlations across the set of municipalities could be determined.

Box 2. Complex landscape patterns matter for certain ecosystem services: Some examples

Biodiversity: Biodiversity conservation on cultivated land often depends on the amount of set-aside and the ability of habitats to facilitate the dispersal of species. In agro-ecosystems, structurally complex landscapes enhance local biological diversity, which may compensate for local high intensity management practices. Conversely, the potential to increase the provision of ecosystem services by modifying landscape patterns is low for soil fertility and soil structure, as well as for carbon sequestration, at least in the short term.

Natural pest control: The complexity of landscapes is generally associated with higher population of natural enemies and reduced pest pressure. Landscape patchiness and the presence of non-crop habitats, such as herbaceous (e.g. fallows, field margins) and wooded habitats, contribute to maintain pest control function.

Cultural services: A monetary valuation study in the Netherlands shows that tourists prefer rural landscapes of forest patches interlinked with hedgerows, rather than open landscape dedicated to agriculture, or of mostly forest landscape. Respondents also appreciated regions retaining landscape elements such as locally typical buildings, tree lines, lakes and rivers, forest, and wildlife viewing.

Agriculture depends on ecosystem services that contribute to productivity, but are often neglected in policy action

Agricultural production depends on key ecosystem assets and services: land, soil, water, biological pest control and animal pollination (Table 4). More generally, biodiversity at all levels has a complex but increasingly recognised and better understood role in agriculture production. In addition, genetic diversity can make agricultural production less sensitive to unfavourable meteorological conditions, or help to adapt to a changing climate.

Table 4. Major ecosystem services to agriculture, and the principal sources of supply at different scales

Ecosystem service to agriculture	Field	Farm	Landscape	Region/global
Soil fertility and formation, nutrient cycling	Microbes, invertebrate communities, legumes	Vegetation cover		
Soil retention	Cover crops	Cover crops	Riparian vegetation; floodplain	Vegetation cover in watershed
Pollination	Ground-nesting bees	Bees; other pollinating animals	Insects; other pollinating animals	
Pest control	Predators and parasitoids (e.g. spiders, wasps)	Predators and parasitoids (e.g. spiders, wasps, birds, bats)		
Water provision and purification		Vegetation around drainages and ponds	Vegetation cover in watershed	Vegetation cover in watershed
Genetic diversity	Crop diversity for pest and disease resistance			Wild varieties
Climate regulation	Vegetation influencing microclimate (e.g. agroforestry)	Vegetation influencing microclimate	Vegetation influencing stability of local climate; amount of precipitation; temperature	Vegetation and soils for carbon sequestration and storage

Source: Zhang et al. (2007).

Animal pollination is key to global food production but is threatened by the significant decline in pollinator populations in most parts of the world. The role of pollinators in food production was recently assessed using best scientific available evidence by the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services. According to IPBES animal pollinators contribute to the pollination of three quarters of leading crops of the world, representing 35% of world crop production. The share of pollinator-dependent crops that are cultivated significantly has trended upward in recent decades, reinforcing the dependency of crop production on pollinators.

The decline in animal pollinators may significantly reduce not only crop yields, but also crop quality, with implications for food security and nutrition, although evaluations remain uncertain in that regard. Under an extreme scenario of *total pollinator losses*, estimate that crop production would be reduced between 1% to 40% for 45% of crops; between 40% to 90% for 28% of crops; and by more than 90% for the most pollinator-dependent crops. For partial loss of pollinators, the effects on yields are harder to quantify, although pollinator-dependent crops have already experienced stagnating yields, and generally display more volatile yields than crops which do not depend on pollinators. Lack of pollinators can sometimes limit output as has been established through a number of empirical studies, such as that of in their work on the commercial production of blueberries (*Vaccinium corymbosum*) in New Jersey. Recent evidence suggests that in many localities it is the abundance of a few common pollinator species that is the limiting factor for the ecosystem service, rather than species richness *per se* .

The causes of the pollinator decline globally remain unclear, which complicates the implementation of concrete measures to curb the phenomenon. While it has been found that pesticides have a lethal effect in experimental conditions (controlled experiments), their effect in real world conditions is more difficult to establish. There are probably multiple explanatory factors (fragmentation and habitat destruction; climate change; chemicals and toxins; invasive species, parasites, and pollinator pathogens; nutrition; forage quality; genetics, breeding and the biology of pollinators) that have additive or even interactive effects and which taken together can explain the declining trend of pollinators. In spite of these knowledge gaps, IPBES has identified strategies and concrete actions that could contribute to curb the decline in pollinators.

Biological pest control and biological regulation of diseases can also be a highly significant ecosystem service for crops and livestock. Regarding crops, estimate the value of natural control attributable to insects to be USD 4.5 billion annually globally. Wildlife can be a significant reservoir of disease vectors for livestock that also needs to be better understood and accounted for have shown the importance of landscape structure on the epidemiology of avian influenza viruses in wild populations and its impact on the agricultural sector and the wider population from a socio-ecological perspective, and have argued that impact is strongly influenced by landscape structure and context.

The issue of soil fertility has also been accorded stronger political importance in recent years, with implications for both agricultural productivity and ecosystem services. While the state of soils in terms of organic matter and nutrient balance is improving in a number of countries, especially in OECD countries (e.g. United States), the situation is more mixed globally. The International Year of Soils in 2015 was the opportunity to focus on these issues and develop a health check of the soil condition in the world, via the work of the Global Soil Partnership and the Technical Intergovernmental Panel on Soils. Their report *State of World's Soil Resources*, published in 2015, reviews the evidence-base on the major threats to soil function in the different regions of the world, i.e. soil erosion; increase in water soluble salts in soils (salinisation and sodification); organic carbon exchange; contamination; sealing; compaction; loss of soil biodiversity; soil acidification; nutrient imbalances nutrient (deficit or surplus); and waterlogging.

There is increasing evidence that soils are in poor and degrading conditions in many regions of the world, potentially threatening the capacity of ecosystems to provide ecosystem services in the medium to long run (Table 5). As the basic interface for key ecosystem processes and functions, soil degradation (erosion, acidification, etc.) is likely to negatively affect the capacity of ecosystems to provide services. For example, soil content in organic matter can change the water storage capacity of soils, and thus their ability to play a role in flood risk mitigation as well as resilience to droughts. Conversely, soil compaction can increase flood risks. Watershed management through natural rivers and irrigation infrastructure network, in particular with collective management approaches, provide an important ecosystem service as flood prevention and control can be significant to agriculture production and sustainability.

Table 5. Soil degradation is present worldwide, threatening agricultural productivity and the provision of ecosystem services in the medium to long run

	Very poor	Poor	Fair	Good	Very good
Soil erosion	NENA (-)	A (-) LAC (-) SSA (-)	E (+) NA (+) SP (+)		
Organic carbon change		A (~) E (~) LAC (-) NENA (-) SSA (-)	SP (~)		
Nutrient imbalance		A (-) E (~) LAC (-) SSA (-) NA (-)	SP (-)	NENA (~)	
Salinisation and sodification		A (~) E (-) LAC (-)	NENA (-) SSA (~)	NA (+) SP (~)	
Soil sealing and land take	NENA (-)	A (-) E (-)	LAC (~) NA (-)	SSA (=) SP (-)	
Loss of soil biodiversity		NENA (-) LAC (-)	A (~) E (-) SSA (-)	NA (~) SP (~)	
Contamination	NENA (-)	A (-) E (-)	LAC (~)	SSA (-) NA (+) SP (+)	
Acidification		A (-) E (~) SSA (+) NA (-)	LAC (~) SP (-)	NENA (~)	
Compaction		A (-) LAC (-) NENA (-)	E (~) NA (~) SP (~)	SSA (=)	
Waterlogging			A (-) E (~) LAC (=)	NENA (~) SSA (=) NA (~) SP (~)	

Note: A: Asia; E: Europa and Eurasia; LAC: Latin America and the Caribbean; NA; North America; NENA; Near East and North Africa; SSA: Africa and South of the Sahara; SP: Southwest Pacific. The list of countries included in each of these world regions is available in the full FAO report (FAO and ITPS, 2015b)
Source: FAO and ITPS (2015a, 2015b).

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Annex 1.A.
Common international classification of ecosystem services (CICES)

Annex Table 1.A.1. Ecosystems provide a wide array of provisioning, regulation, maintenance and cultural services

Section	Division	Group	Class
Provisioning	Nutrition	Biomass	Cultivated crops
			Reared animals and their outputs
			Wild plants, algae and their outputs
			Wild animals and their outputs
			Plants and algae from in-situ aquaculture
			Animals from in-situ aquaculture
		Water	Surface water for drinking
			Ground water for drinking
	Materials	Biomass	Fibres and other materials from plants, algae and animals for direct use or processing
			Materials from plants, algae and animals for agricultural use
			Genetic materials from all biota
		Water	Surface water for non-drinking purposes
			Ground water for non-drinking purposes
	Energy	Biomass-based energy sources	Plant-based resources
			Animal-based resources
		Mechanical energy	Animal-based energy
Regulation and maintenance	Mediation of waste, toxics and other nuisances	Mediation by biota	Bio-remediation by micro-organisms, algae, plants, and animals
			Filtration/sequestration/storage/accumulation by micro-organisms, algae, plants, and animals
		Mediation by ecosystems	Filtration/sequestration/storage/accumulation by ecosystems
			Dilution by atmosphere, freshwater and marine ecosystems
			Mediation of smell/noise/visual impacts
	Mediation of flow	Mass flows	Mass stabilisation and control of erosion rates
			Buffering and attenuation of mass flows
		Liquid flows	Hydrological cycle and water flow maintenance
			Flood protection
		Gaseous / air flows	Storm protection
			Ventilation and transpiration
	Maintenance of physical, chemical, biological conditions	Lifecycle maintenance, habitat and gene pool protection	Pollination and seed dispersal
			Maintaining nursery populations and habitats
		Pest and disease control	Pest control
			Disease control
		Soil formation and composition	Weathering processes
			Decomposition and fixing processes
		Water conditions	Chemical condition of freshwaters
			Chemical condition of salt waters
		Atmospheric composition and climate regulation	Global climate regulation by reduction of greenhouse gas concentrations
			Micro and regional climate regulation

Cultural	Physical and intellectual interactions with biota, ecosystems, and land-/seascapes (environmental settings)	Physical and experiential interactions	Experiential use of plants, animals and land-/seascapes in different environmental settings
			Physical use of land-/seascapes in different environmental settings
		Intellectual and representative interactions	Scientific
			Educational
			Heritage, cultural
			Entertainment
			Aesthetic
	Spiritual, symbolic and other interactions with biota, ecosystems, and land-/seascapes (environmental settings)	Spiritual and/or emblematic	Symbolic
			Sacred and/or religious
		Other cultural outputs	Existence
			Bequest

Source: Source: United Nations, European Commission, Food and Agriculture Organisation, OECD and World Bank (2012).

Annex 1.B.

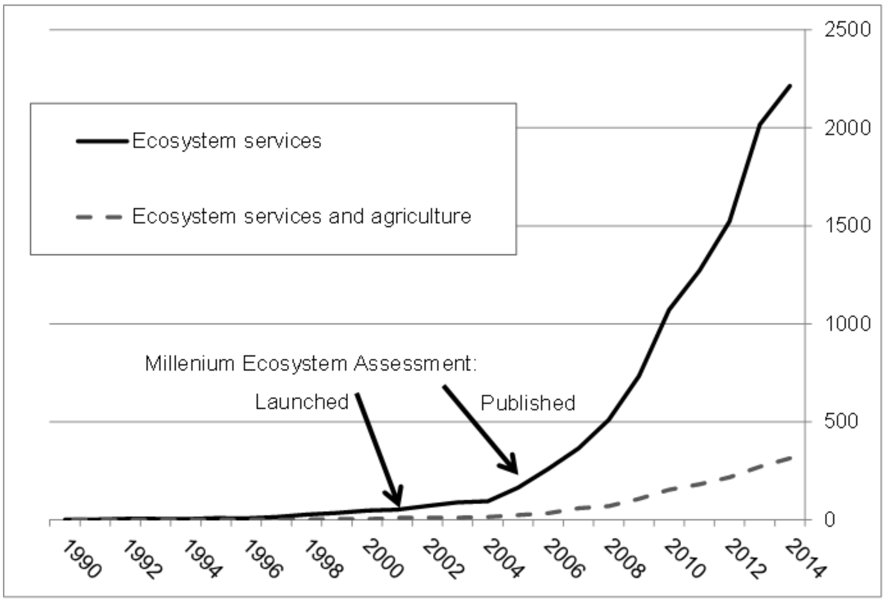
An historical account of ecosystem services and their implementation

The concept of ecosystem services was popularised in the early 2000s by the Millennium Ecosystem Assessment (MEA) at the initiative of the United Nations. The objective of the MEA was to "assess the consequences of ecosystem change for human well-being and the scientific basis for action needed to enhance the conservation and sustainable use of those systems and their contribution to human well-being." Since then, the concept of ecosystem services has increasingly been used in research and gained attention in policy spheres.

The concept of ecosystem services is now mobilised extensively by the research community in various disciplines (ecology, economics, water resources, political science); and in public policy circles. Hence, definitions still often vary, and have historically been themselves an issue of policy debate. Although early references to ecosystem services and their economic value may be traced back to the mid-1960s , the academic literature referring to the concept of ecosystem services was virtually non-existent until the mid-1990s, and has since grown exponentially, at an average annual growth rate of nearly 40% per year. Such an increase in the number of published papers is in itself a remarkable phenomenon, generally seen with major scientific advances. Figure 1.B1 shows the exponentially rising interest for ecosystem service was initially spurred by a policy initiative: the Millennium Ecosystem Assessment (MEA) launched by the United Nations in 2001 and released in 2005.

Annex Figure 1.B.1. Exponential growth of the academic literature on ecosystem services since the late 1990s

Number of documents published per year that include the term "ecosystem services", and "ecosystem services" jointly with "agriculture" in title, abstract or keywords found on Scopus



Source: OECD based on Scopus: www.scopus.com.

The focus on services provided by ecosystems and their impacts on human well-being has deeply reshaped the traditional policy approach to ecosystem conservation, by introducing a cost-benefit argument in the policy debate. The idea was that, in the face of strong policy and market drivers that potentially threaten ecosystems and biodiversity, cost-benefit analysis could provide policy makers an additional argument, on pure economic grounds, in favour of biodiversity conservation. This was expected to allow for better integration of ecosystem conservation in overall economic policy, on the basis of a common economic language. The works undertaken by the MEA have led to the development of the first frameworks linking ecosystem services, productivity and human well-being.

The environmental-economic framework of ecosystem services has thus gradually clarified and stabilised during the 2000s. An important step towards this standardisation was the publication of the Experimental Ecosystem Accounting in the framework of the System of Environmental-Economic Accounting (SEEA) 2012, under the auspices of the United Nations, the European Commission, the Food and Agriculture Organisation, the United Nations, the Organisation of Economic Co-operation and Development and the World Bank.

2. Policy packages to enhance ecosystem services linked to agriculture

Key messages

- This chapter investigates how different types of policy instruments or policy packages can be used to improve policy approaches to ecosystem services linked to agriculture, and which key policy design and implementation issues need to be considered for the success of a given policy instrument or policy package.
- Agricultural markets and policies can be important drivers of the agriculture-ecosystems relationship. To show the potential benefits of improving policy design, a quantitative model is developed for illustrating policy instrument choice and ecosystem service provision and to investigate synergies and trade-offs between ecosystem services, the importance of market and policy drivers in ecosystem service provision, and the performance of the current agricultural and agri-environmental policy package in promoting a balanced set of ecosystem services (that is, a balance between demand for and supply of different ecosystem services).
- The model developed in this report on the basis of Finnish data suggests that market and policy drivers, namely commodity prices and crop area payments, have strong impact on ecosystem service provision. Increase of crop area payment or commodity price result in increased supply of provisioning services (commodity production) and significant reduction of other ecosystem services (biodiversity and water quality) so that social welfare is reduced relative to situations without any policy intervention.
- The model based on Finnish data also suggests that a policy package combining decoupled area payments and agri-environmental payments performs much better and provides a reasonably balanced set of ecosystem services. However, such a policy mix is expensive in budgetary terms and has low budgetary cost-effectiveness due to lack of environmental targeting. By incorporating an environmental targeting mechanism (e.g. using an environmental benefit index, hereafter EBI) into current agri-environmental policy, the budgetary cost-effectiveness can be improved significantly, as the same environmental benefits can be achieved with a 42% lower budget in the case of Finland.
- Interactions between ecosystem services should be taken into account to properly select the "base" (e.g. input use, land use or tillage method) that policy instruments will target for addressing bundled ecosystem services. Ecosystem services are often produced as bundles and policies targeting specific services provision may affect other services negatively or positively. In the best case one single policy instrument (buffer strip payment) can promote a reasonably well-balanced set of services with small efficiency losses, while in the worst case another single instrument (fertiliser use constraint) results in strong imbalance and efficiency losses. Building on the model simulations, the report proposes a policy decision tree helping policy makers in their choice of policy packages to address the provision of ecosystem services linked to agriculture

The relationship between agriculture and ecosystems is subject to several types of market, policy and governance failures

Several market failures are associated with ecosystem management in agriculture, resulting in the potential degradation of ecosystem assets and lower provision of ecosystem services than would be socially optimal. Market failures can concern each of the three relations between agriculture and ecosystems: pressure on, provider of, and user of ecosystem services.

Pressures from agriculture on ecosystem assets may result in negative environmental externalities. The existence of negative environmental externalities calls for some form of government policy to reduce them. For example, excess nutrients in water due to agricultural nutrient surplus may raise costs of drinking water services, hamper productivity and competitiveness of the drinking water sector, and have negative health impacts. Other examples include greenhouse gas emissions, ammonia emissions, or pesticide pollution.

Ecosystem services provided by agriculture can also have positive environmental externalities, and thus be at risk of undersupply. For example, farm practices increasing carbon sequestration, or reducing flood risk in watersheds, provide benefits to other people and economic sectors, but are usually not paid for in market transactions, or exchanges. This type of market failure will more likely affect regulation services and cultural services.

Ecosystem assets can be common goods, and for this reason subject to risk of overexploitation and degradation. For example, water salinisation linked to irrigation practices can be analysed as a form of ecosystem degradation. Each irrigator has a private benefit from irrigating, but may not take into account the negative impact of their own irrigation practices on overall water salinity. Typically, without an appropriate public policy environment, private decisions may be unable to provide the socially optimal level of water quality, reducing both agricultural productivity and welfare. Another example is the maintenance of pollinator populations: if farmers adopt practices that negatively impact pollinators (e.g. excessive use of chemicals, monoculture), this will in turn reduce the associated pollination service, and may impede agricultural productivity growth.

These market failures can affect current, but also future generations. Some services may have no social value today, but have a positive value in the future due to a rising social demand for these services. In this context, the deterioration in the ability of ecosystems to provide these services goes unnoticed today because it has no social value at present or because its degradation is only noticed when it passes a particular threshold. However, reduced ecosystem capacities will cause loss of well-being for future generations. Markets may not properly account for such a long-run dimension.

Research, scientific knowledge and information production on ecosystem services can in certain cases also be considered as public goods. Managing ecosystem services is complex, and requires evidence-based approaches, involving knowledge and information that markets may fail to produce. For instance, there is a lot of scientific uncertainty about the causes of pollinator decline in several regions of the world; or about the role of soil carbon content in crop productivity. The government may have a role in helping correct market failures in information production and provision in this area.

These market failures taken together provide rationales for welfare-enhancing government policy with regard to those ecosystem services linked to agriculture in order to make private actors consider (internalise) their positive and negative effects (externalities) on ecosystem assets, and the social cost (benefit) of ecosystem degradation (improvement) in their decisions. These policies should ideally include current, but also potential future flows of ecosystem services. Policies should also support research, knowledge and information provision of ecosystem services and their impacts on productivity and well-being.

Existing agricultural policies can worsen the market failures associated with ecosystem services in agriculture, although the most distorting supports are trending downward in the OECD. Distorting forms of support include market price support and payments based on inputs or outputs. Distorting support can exacerbate pressures on ecosystems on agriculture by increasing fertiliser and pesticide use, or favouring monoculture. However, the most distorting forms of support significantly decreased in the last two decades in most OECD countries and several have been replaced by decoupled forms of support. This may have reduced distortions and associated pressures on ecosystems from agriculture.

With decreasing government support, agricultural markets become predominant drivers of farmers' production choices and influence of the agriculture-ecosystems relationship. If market failures associated with ecosystems (externalities, public goods) are not properly internalised, then markets can also increase pressure on ecosystems, basically in times of high output prices and/or low input prices. Market prices can also raise the cost of environmental policy, by increasing the opportunity cost of environmental regulations and environmental policy instruments for farmers. For example, increases in commodity prices can boost deforestation rates in a number of countries and increased commodity prices can also reduce participation in environmental conservation programmes in agriculture by increasing the opportunity cost of not cultivating lands .

Market failures can be aggravated by policy coherence problems, partly due to the scientific complexity and high local-specificity of the agriculture-ecosystems relationship. Policies are often designed separately and devoted to a subset of ecosystem services which may also affect the provision of other ecosystem services. The synergies and trade-offs between ecosystem services are not well known, or uncertain and rarely taken into account. This calls for more holistic and multidisciplinary approaches bringing ecological, agronomic, and economic and social sciences on board and the recognition of the role of the spatial dimensions, which can be challenging in practice.

Key policy design criteria to promote an adequate level of ecosystem services in agriculture

There are a wide array of policy instruments that do, or potentially could improve policies to manage ecosystem services linked to agriculture, including: spatial planning policies, environmental regulations; environmental cross-compliance linked to farm support, payments for ecosystem services (PES), auctions, and environmental credit and offset markets; but also: public investment in research and extension activities related to conservation practices, best management practices, or agricultural technologies that reduce input use or increase productivity per unit of output. It is a challenge for policy makers to choose the type of policy instruments and the combination of instruments, that would work best for different types of ecosystem services, taking into account the importance of land use and landscape patterns for most (but not all) ecosystem services related to agriculture.

The key criteria that separate efficient policies from others are: i) cost-benefit (net-benefit from policy intervention should be positive); ii) environmental effectiveness, or the capacity of the instrument to achieve stated goals or targets of practices; and iii) cost-effectiveness, or the costs of achieving society's objectives. When land owners are targeted by policies, the cost-effective policy instrument is the one that minimises their compliance costs while achieving the target. Two other important policy choice criteria contribute to the success and acceptability of policy instruments: implementation costs (policy-related transaction costs) and political feasibility (e.g.).

Demand for ecosystem services and cost-benefit criteria

The identification of market failures by itself is necessary but not sufficient to justify policy responses. From an economic point of view, the expected social benefits of policies should outweigh their expected costs. Environmental policies usually have a wide range of costs and benefits, and the legitimacy of a policy builds on the fact that the net benefits of the policy are positive.[4]

The concept of ecosystem services invites policymakers to be as exhaustive as possible on social benefits from ecosystems for better decision making. This consideration of the wide range of benefits is probably the most notable development of the ecosystem services field. Up to the late 1990s, cost-benefit analysis was looking at a narrower set of environmental costs and benefits, typically considered in isolation. Efforts undertaken in the 2000s through initiatives like MEA, TEEB, and United Nations made more consistent classification and accounts of ecosystem services available, as a basis for policy assessment and evaluation.

But this call for an exhaustive valuation of all ecosystem services is highly demanding in practice. For most ecosystem services (except for provision services in the short run), there is no market and thus no price. Hence the socially optimal bundle of ecosystem services can be identified but is difficult to quantify. Because it is difficult to measure ecosystem services and quantify their benefits, there is a risk that exhaustiveness may delay policy implementation or may increase transaction costs. Ecosystem services can have different types of values: direct use values, indirect use values and "non-use" (or existence) values () (Box 2.1). In addition, values derived from ecosystem services can affect different time horizons and human generations. For example, regulation services such as flood control, for which realisation is contingent on a flooding event, may be far-off in the future and uncertain (i.e. it has an option value[5]).

A recent OECD study underlined the role of national ecosystem assessment in influencing policy making, but also gaps in practical implementation in the case of United Kingdom, Japan, Spain, and Portugal, as well as other regional and international ecosystem assessments. According to the study, there is evidence that national ecosystem assessments contributed to informing policy decisions, especially by i) synthesising and communicating complex information; ii) demonstrating the economic benefits provided by ecosystem services, through the use of monetary valuation. Beyond this awareness and demonstrative impacts, integration of ecosystem services into practical decision making remains in its infancy.

4. This criterion, apparently simple, is difficult to implement in practice for several reasons analysed in great detail in numerous reference publications on the subject (Pearce et al., 2006): lack of information on costs and benefits, especially for non-marketable goods and services, choice of relative prices of environmental goods, choice of the discount rate, among others.

5. In cost-benefit analysis and social welfare economics, the term option value refers to the value that is placed on private willingness to pay for maintaining or preserving a public asset or service even if there is little or no likelihood of the individual actually ever using it.

Innovative responses are emerging to measure and map flows of ecosystem services to identify gaps between the supply and demand of ecosystem services. Notable efforts include the European Union Mapping and Assessment of Ecosystems and their Services (MAES) process[6] . Research also proposes innovative tools useful for policy guidance: for instance, regionalised mapping of the supply and demand of certain ecosystem services has been developed, such as pollination services at European level or flood control services. This provides policy makers maps of hot-spots relating to the supply and demand of ecosystem services. For example, showed that in 53% of European cropland the supply ofr pollination services is enough to meet the demand, but that 13% of European land areas are characterised by high demand-low supply of pollinator services, especially in central France, Germany, Hungary and northern Italy.

Box 3. Ecosystem services: An historical focus on monetary valuation

A large body of literature has focused on developing and applying methods in the monetary valuation of ecosystem services (ES). Methods depend on the types of ES and values, and include measures through market values (price or costs), revealed preference approaches (e.g. travel cost method, hedonic pricing), and stated preference approaches (e.g. contingent valuation, choice experiment). As part of the TEEB study, an Ecosystem Service Value Database includes more than 1 300 value-estimates of ecosystem services (http://www.fsd.nl/esp/80763/5/0/50). It shows that for provisioning ES such as food, raw material, genetic, medicinal and ornamental resources, value is generally directly assessed from market price. Water resources and quality are assessed from price and cost-based methods (i.e. avoided cost, replacement cost), as well as through hedonic pricing. With respect to regulating services (e.g. waste treatment, moderation of disturbances), valuation is generally conducted using cost-based methods. The flow of habitat services are generally assessed with contingent valuation (CV), while cultural services, particularly recreation, are assessed with CV and the travel cost method. An important methodological issue is how to transfer estimate of willingness-to-pay from one site and applying to another, adjusting to take into account differences, for example in the socio-economic characteristics of populations, or difference in the proposed change in provision.*

There is a need for the participation of all stakeholders in designing and implementing policies for ecosystem services in agriculture. Monetary valuation can be a useful contributor to decision making, but not a substitute: governance processes can play an important role in determining and legitimating the social demand for ecosystem services and ecosystem conservation. Several research studies have shown the value of participatory approaches to determine policy objectives for ecosystem management. Historically, the conservation objectives were based on the scientific expertise available. Several OECD countries have recognised the need for such governance approaches, but practical implementation is still at an early stage.

Environmental effectiveness and cost-efficiency

The cost-effectiveness of addressing ecosystem service supply is often spatially dependent and should be considered at the landscape level. This has to do with the allocation of service provision efforts across individual landowners within a region. Differences in both opportunity costs and site-productivity of service supply over space imply that the cost-effective achievement of landscape-scale ecosystem service targets will generally entail differential levels of effort across land owners.

A large body of literature demonstrates the existence of potentially large welfare gains from improving cost-efficiency of policy instruments regarding ecosystem services and agriculture. On the basis of these works, reviews and analyses different types of payments and auctions for biodiversity conservation in agriculture. Empirical analysis on the basis of

6. See: http://biodiversity.europa.eu/maes.

Finnish data shows that when targeted payment types are implemented, the cost-effectiveness gains from environmental targeting are very large and clearly exceed the increase in policy-related transaction costs (hereafter PRTCs). Similar findings are reported in OECD regarding spatially targeted agri-environmental payments and PES.

Other criteria: Transaction costs, budgetary constraints, and equity

Policy-related transaction costs (PRTCs) matter in the comparison of policy instruments addressing ecosystem services. Empirical literature shows that there is huge variation in PRTCs between different agricultural and agri-environmental policy instruments. Arable area payments have typically low PRTCs, ranging from 1% to 7% of payment transfer, while individually tailored agri-environmental management agreements have the highest PRTCs (25-66% of payment transfer) because high asset specificity and low frequency of transactions . Recent policy developments towards increased spatial targeting and tailoring of policies assume that there should be a trade-off between improving the precision of policies and increasing PRTCs.

The budgetary cost-effectiveness of policy instruments is also affected by management capacities of public agencies they require and by public sector transaction costs for design, implementation, monitoring and enforcement of policy instrument. Accounting for policy-related transaction costs (PRTCs) can be important for a comparison of alternative policy instruments and it can help the effective design and implementation of policy instruments to achieve policy objectives.

Different criteria may conflict each other, the most typical case being the trade-off between efficiency and equity. Policy instruments based on cost-efficiency criteria can redistribute income and wealth in a progressive or regressive way. Typical examples include the use of taxes versus subsidies. In the case of payments for ecosystem services, the particular design of the payment may favour some types of farmers depending on their characteristics such as farm size or opportunity costs (in case of a flat rate subsidy).

Policy instruments to address the bundle of ecosystem services in agriculture

This section reviews the available and potential policy instruments to address the provision of ecosystem services in agriculture. It starts with an argument that the cost-effectiveness of policy intervention to address the provision of ecosystem services requires spatial targeting and tailoring in order to address the heterogeneity of opportunity costs and site-productivity of ES provision. Because of this specificity, it is important to identify whether a given policy instrument can be targeted and tailored spatially or not, as this will affect its performance. The purpose of the review is to facilitate the logic and interpretation of the policy choice decision tree introduced at the end of the chapter.

Due to spatial variation in costs and benefits of ecosystem service supply, spatially differentiated landowner effort and tailored incentives are needed to ensure the cost-effectiveness of supply. Tailoring policy incentives within spatially targeted area involves, for example, fine-tuning of payment level according to landowner opportunity costs in order to reduce overcompensation, and thus improving the budgetary cost-effectiveness of intervention. As noted earlier, targeting and tailoring will increase the PRTCs of the policy, however the cost-effectiveness gains from targeting and tailoring can exceed increase in PRTCs and justify the targeting and tailoring effort.

Ecosystem services in governance and regulatory frameworks

Until recently, ecosystem services were not explicitly mentioned as such in the agriculture and environment regulatory frameworks. The relatively recent concept of ecosystem services originates from the world of academic research. Environmental policies developed since the 1990s did not refer to the concept, although they already provided some responses to ecosystem degradation.

In a growing number of OECD countries, the concept of ecosystem services is being integrated into general legal and regulatory frameworks. For instance, in the United States, the federal administration released a memorandum in October 2015 directing federal agencies to "factor the value of ecosystem services into Federal planning and decision-making". In the European Union, Action 5 of the EU Biodiversity Strategy commits member states to "map and assess the state of ecosystems and their services in their national territory to help achieve its target of 'maintaining and restoring ecosystems and their services".

Environmental regulation

Environmental regulations are compulsory and can take various forms such as bans, permit requirements, maximum rights or minimum obligations performance standards require reliable performance indicators and monitoring to be regulated (e.g. maximum nutrient surplus per ha of land) while input regulation can concern the amount, timing or method of input application. Environmental standards can be differentiated to account for heterogeneity of site-productivity of ecosystem service provision (for example proximity to surface water courses).

Environmental regulation remains a major part of policy packages in most OECD countries to reduce pressures from agriculture on ecosystems (Table 8). Typical examples include, in the European Union: the European Water Framework Directive, the Nitrate Directive, and the Birds and Habitat Directives. These regulations combines targets in terms of environmental outcomes (status of water quality, for instance) and requirements for action plans by member countries. In the United States, the US EPA regulates production practices of Concentrated Animal Feeding Operation (CAFO) under provisions of the 1972 Clean Water Act, including: permit requirements for pollution discharge; and requirement for Comprehensive Nutrient Management Plan (CNMP) that identifies site-specific practices to ensure agronomic use of nutrients.

Environmental regulations can also protect ecosystem services flowing to agriculture such as land resources, habitat conservation useful for biological pest control, or by banning the use of certain pesticides. For example, the European Commission restricted the use of three pesticides in 2013 based on an assessment of their impact on bees by the European Food Safety Authority (EFSA).

There is considerable variation and uncertainty about the costs and benefits of environmental regulations targeting agriculture. Regulations impose a burden on agricultural production (i.e. they impose a shadow cost) and provide a wide array of benefits for society. In most OECD countries, the necessary information to undertake proper cost-benefit assessment is lacking. For example, shows that the implementation of the Water Framework Directive is likely to impose substantial costs compared to business as usual. In the Netherlands, for example, such additional costs by a range between 5% and 30%, the main driver of cost magnitude being the gap between the objective and the reference scenario suggest farmers' costs of compliance to legislation on the environmental, animal health and welfare and food safety in the European Union are likely to represent less than 3-5% of total production costs for most agricultural products.

Table 6. Examples of regulations to limit pressures on ecosystems from agriculture

Environmental objective	Examples of regulations in place in several OECD countries
Reducing pollution due to input use	Time-limited approval of marketing authorisation for pesticides (all OECD countries) Rules on storage and application of chemical fertilisers and pesticides (prohibition of aerial spraying, or minimum distance to watercourses) Manure management: restrictions on the quantity of manure that can be spread; seasonal bans on manure application; manure storage requirements; and limitations on livestock densities and on the expansion of livestock units (European Union, New-Zealand) Permitting systems and discharge limits for large livestock operations (European Union, Japan, United States) Mandatory nutrient management plans for farmers Mandatory buffer strips and catch crops (Australia, Canada, Denmark, France, New Zealand, Sweden among other OECD countries)
Protecting habitats and biodiversity	Limitations on farm land use and practices (Birds and Habitat Directives of the European Union Mandatory conservation of habitats adjacent to agricultural lands: wetlands, hedgerows, bush and forests (Birds and Habitat Directives of the EU; Endangered Species Act in United States) Conservation areas (Natura 2000 network in the European Union) Banning marketing and use of certain pesticides
Limiting water demand by agriculture	Cap on water abstractions by agriculture Permitting requirements to develop irrigations Time restrictions to water abstraction by agriculture

Source: Based on Vojtech (2010).

Land use planning and regulation

Land use planning can achieve spatial targeting and may also be used to shape the landscape to obtain certain desired spatial patterns. This can be done through zoning and the related restrictions and obligations, e.g. avoidance of land conversion and maintenance of certain landscape features. Land use and spatial planning policy can take many different forms ranging from regulatory policies (e.g. zoning and direct land use controls) to economic instruments, including development impact fees, purchases of development rights and preferential property taxation provides a detailed typology of land use planning systems and policies.

Land use policy can be developed at the local, regional, and national levels. While governments in many countries monitor trends in land use conversions, most of the land use controls are executed by regional or local administrations. Some of these regulations have been quite effective, for example, in reducing the negative effects of urbanisation on ecosystems. In this context show that US state and local land use regulations have reduced land development on average 10% on the West Coast of the United States.

Many OECD countries have developed land use regulations since the 1970s to protect habitats and associated ecosystems. Examples in the European Union include the Natura 2000 Network and the sites designated for Birds Directive.

Environmental taxes

Environmental taxes can be used to reduce pressures on ecosystems from agriculture such as nutrient runoff and pesticide pollution. Taxes can be based on performance (based on indicator or proxy of ecosystem service outcome, such as nutrient surplus) or input-based (taxing inputs that are correlated with ecosystem service outcome, such as fertiliser or pesticide application); tax rates can be differentiated in order to account for heterogeneity in site-productivity of ecosystem service outcomes. In the case of input taxes, spatial differentiation may be difficult to implement in practice due to political reasons (distributional impacts) or because secondary markets may emerge (in which low tax rate farmers sell inputs to high tax rate farmers).

Environmental taxes on fertilisers and pesticides have been used in a few OECD countries with limited results. In the European Union, pesticide taxes have existed in Denmark, France, Norway, and Sweden;[7] and fertiliser tax is has been implemented by Sweden but is no longer in place[8]. Effects of these taxes on input use by farmers have been relatively limited, due to a combination of a low tax level set by regulators, combined with a low price-elasticity of demand for pesticides and fertilisers.

A higher level of unitary taxation may increase environmental efficiency, but also can reduce farm income substantially in the absence of revenue compensation systems. For example, in order to reduce pesticide use by 50% in France, a pesticide tax would lead to a 30% reduction in farmers' incomes. Compensating farmers through lump-sum payments based on agricultural land area can reduce this income fall to 5%.

Community-based approaches to ecosystem services

Community-based approaches to ecosystem services refer to situations in which groups of farmers and stakeholders take voluntary actions, by themselves to deal with environmental issues related to the provision of ecosystem services. Such approaches build on the principle that free negotiation among interested parties is one possible solution to the problem of externalities, i.e. the seminal idea of "coasian bargaining" in environmental economics, later formalised rigorously by game theory. Community-based natural resources management has also been studied in depth and advocated by Nobel Prize Elinor Ostrom.

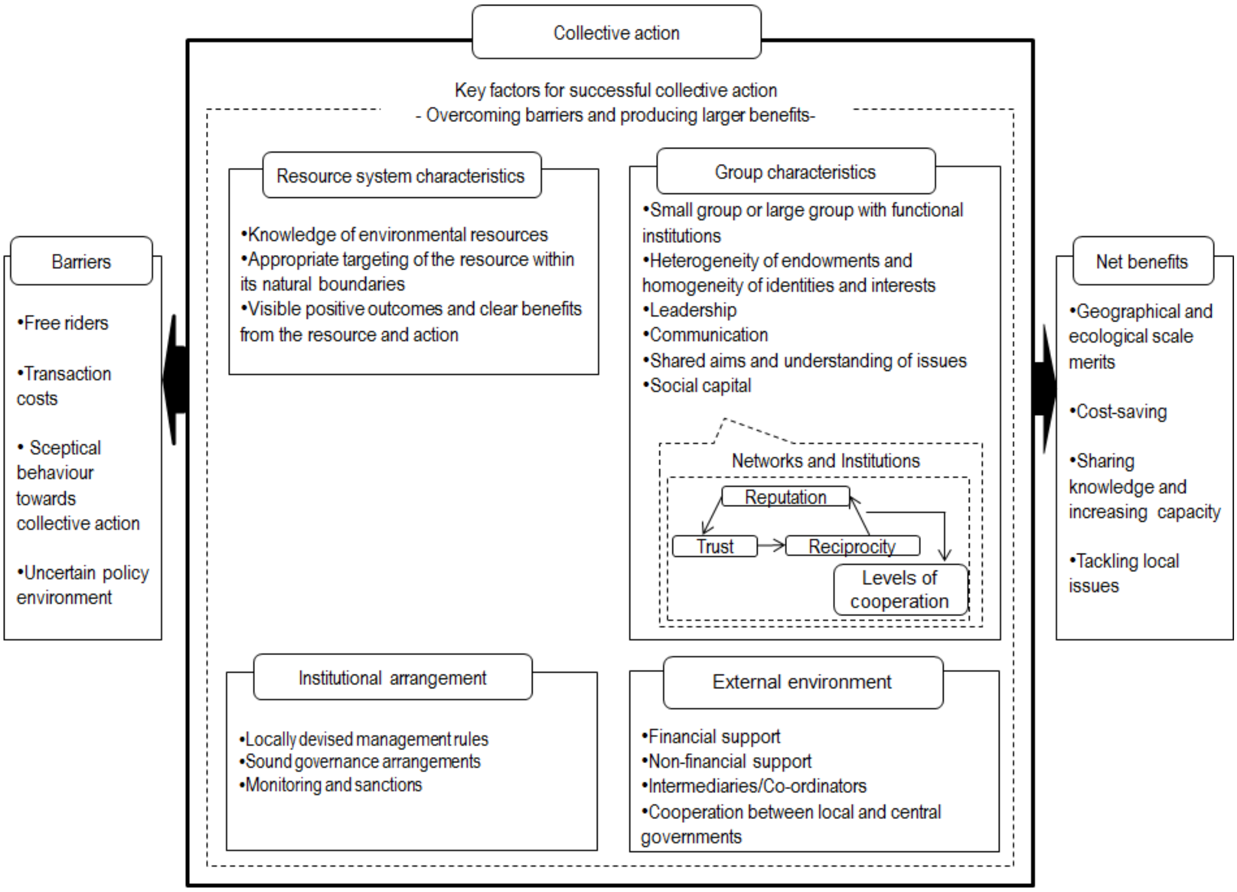
Many OECD countries use various forms of community-based approaches to ecosystem services in agriculture analysed thirteen OECD country case studies of collective actions for the provision of environmental public goods related to agriculture, to understand the advantages and disadvantages of such an approach, and provide policy recommendations. The report shows that there exist three major forms of collective action in the agri-environmental context: i) participants may create a organisations to interact following formal rules and governance principles; ii) external agencies (NGOs, governments, etc.) organise farmers to act collectively for a common purpose; iii) farmers collaborate with other farmers (and non-farmers), but do not form an independent organisation.

7. There was also a tax on pesticides Finland, but it no longer exists. The pesticides-marketing companies pay a fee for the approval of the pesticide selling permit.

8. In 1984 Sweden introduced an environment fee on nitrogen and phosphorous in commercial fertilisers. In 1995, it was replaced by a tax on nitrogen and cadmium in commercial fertilisers. The tax-level was 1.8 Swedish crowns/kg N for fertiliser with a percentage of nitrogen above 2%. For cadmium the level was 30 crowns/g Cd, for levels that exceeds 5 g Cd/tonne phosphorous in the fertiliser. The tax was withdrawn after 2009 and the main reasons concerned farm competitiveness and the fact that it was not possible to show that the tax had any obvious effects on the use of fertilisers. The fertiliser tax is an on-going topic for discussion in Sweden.

Community-based approaches can be effective to manage ecosystem services, but only in certain circumstances and under certain conditions. Such conditions have been underlined in the agri-environmental context by including: good knowledge of the natural resource, clear geographical boundaries, small number of participants with frequent interactions, confidence and social capital, and a favourable external environment provides a systematic review of these conditions in terms or resources characteristics, group characteristics, external environment and institutional arrangements (Figure 9).

Figure 9. Conditions for, benefits of and barriers to collective action in the agri-environmental context



Source: OECD (2013a).

When workable, there are potential gains from community-based approaches and government policy can encourage their development. Key potential benefits from community-based approaches include i) economies of scales, especially when ecosystem services provision makes sense at the landscape level, ii) better acceptability by farmers and stakeholders, and iii) diffusion of knowledge and increasing confidence. The main barriers, already underlined by are transaction costs, which increase with the size of the group, the complexity of ecosystem services (including the need to manage a bundle instead of a single ecosystem service), and the time needed for developing confidence and social capital also underlines the important role of policy incentives to encourage community-based approaches.

There exists a continuum between community-based approaches and other instruments, such as land planning and payments for ecosystem services. Community-based approaches cover various situations and degrees of formalities in terms of rules and governance principles, and groups engaging in collective approaches can agree on coercive rules, contracts, land management rules and payments for ecosystem services. The community-based approach remains voluntary as long as participants keep some freedom to leave the system, which raises the issue of the conditions of self-creation and stability of community-based initiatives.

Payment schemes

OECD countries are increasingly using payment schemes to reduce pressures on ecosystems from agriculture and to incite farmers to enhance the provision of ecosystem services from agro-ecosystems. Some well-known programmes have existed for a long time, such as the agri-environmental measures of the European Union initiated in 1985, or the Conservation Reserve Program (CRP) of the United States created in the 1950s, and substantially expanded after 1985. The development of payment schemes for agri-environmental purposes occurred mostly in the 1990s and the 2000s, in relation with increasing awareness of environmental degradation due to agricultural pressures and reforms of agricultural policies. These programmes aim at improving environmental outcomes beyond already existing legal and regulatory requirements. Beyond OECD countries, payments that may be considered beneficial for ecosystem services (PES) have expanded dramatically worldwide since the early 2000s. Already in 2007, several hundred PES initiatives were identified worldwide in several countries in Africa, Asia, and Latin America (OECD, 2010b). PES schemes often include agriculture, but also other types of land uses such as forest and urban land; and cover different types of environmental pressures and ecosystem services.

There exists two distinct terms for designating payment schemes related to ecosystem services: “payment for *ecosystem* services” on the one hand, and “payment for *environmental* services” on the other hand. These distinct terms are found in the academic literature and in the designation of real-world payment schemes developed by public authorities or private actors. The key difference between ecosystem service and environmental service in this context would be as follows. Ecosystem services are themselves the product of a combination of factors, including labour, capital, and natural capital (Figure 1). Ecosystem services therefore designate a final service, benefiting its end users. Environmental services then refer to a service provided by the land manager (e.g. the farmer) aiming at the protection of ecosystem assets or the increase in ecosystem services that benefits others. Environmental services thus generate positive externalities that benefit society as a whole, which justifies a payment. In reality, most payment schemes do not pay for the ecosystem service itself, but actually cover the costs of adopting certain practices by land managers which, combined with other factors of production, are likely to increase the provision of ecosystem services or the protection of ecosystem assets[9]. In the literature, the two terms (payment for ecosystem services and payment for environmental services) are often used interchangeably.

9. For example, France uses the term payments for environmental services to make the distinction when payments for services should be warranted (i.e. when changes in management practices result in additional services).

In this report, the term payment for ecosystem services is used as the generic term to designate a broad category of payment schemes to enhance the provision of environmental and ecosystem services. More recent definitions account for the diversity of payment schemes in the real-world and are thus more generic in nature than previous proposals in the literature, such as the one proposed by "transfers of resources between social actors, which aims to create incentives to align individual and/or collective land use decisions with the social interest in the management of natural resources". Such a generic definition allows payment schemes to differ in their characteristics, especially regarding:

- The public or private nature of the payment schemes.
- The voluntary versus mandatory nature of payment schemes.
- The local versus nation-wide, or global geographical scope of payment schemes.
- The presence or not of an intermediary in the transaction.

In addition to this fundamental distinction, payment mechanisms can indeed have very different, and sometimes opposite, characteristics. Originally thought of as a form of free market exchange, payments for ecosystem services have been defined as a voluntary transaction where a well-defined ecosystem service (or a land-use likely to secure that service) is being "bought" by an (minimum one) ES buyer from an (minimum one) ES provider, if and only if the ES provider secures ES provision (see Engel et al., 2008). In reality, most PES do not meet one or more of these criteria, especially government programmes which represent the vast majority of PES worldwide (Schomers and Matzdorf, 2013).

Most payment schemes observed in the world are based on land use or practices, rather than on the actual delivery of an ecosystem service, although there are some innovative schemes based on results that have been already implemented, for example in the European Union (Burton and Schwarz, 2013; European Commission, 2017). The relative cost-efficiency of practice-based versus performance-based payment schemes has been the subject of considerable analysis and research among economists. Performance-based schemes can outperform practice-based schemes because they directly target the desired environmental outcomes and give farmers more flexibility to achieve the environmental objective. On the other hand, performance-based schemes have additional implementation costs due to the difficulties in directly observing the level of ecosystem service or due to time lags between practice adoption and ecosystem service provision (Engel et al., 2008; OECD, 2010b). Moreover, performance-based schemes expose farmers to exogenous risks that, despite their efforts can affect the level of environmental performance, and may require the inclusion of a risk premium to make farmers accept taking these risks.

Well-targeted payments are generally considered more efficient than regulation with respect to cost effectiveness (Engel et al., 2008). However, targeted payments may suffer from low additionality (i.e. not leading to additional ecosystem benefits relative to the baseline) and be subject to leakage (i.e. when securing an ecosystem service in one location leads to increased pressure to convert or degrade ecosystem services in another) if the right safeguards and pay-off structures are not in place (Lankoski et al., 2015; OECD, 2010b).

There is increasing evidence that refining the design of payment schemes can lead to substantial gains in terms of cost-efficiency. Refinement of instrument design includes better tailoring and targeting, and integration of a spatial dimension.

Auctions and competitive bidding can substantially improve allocative efficiency and budgetary cost-effectiveness of payment schemes addressing ecosystem service supply in heterogeneous landscapes (Lankoski, 2016; OECD, 2010a). Development of sound ecosystem service indices to be employed with auctions is a prerequisite for maximising ecosystem service benefits. Targeting ecosystem services that are dependent on certain spatial pattern (e.g. connectivity of habitat patches) and scale requires that ecosystem service indices address co-operation among land-owners so that the ecosystem service index value of the given field parcel does not only reflect management practices adopted in that parcel but also those adopted in adjacent field parcels.

Bundling ecosystem services may also improve cost-efficiency of payment schemes. Bundling refers to a case where the adoption of a management practice receives a single payment for the provision of multiple ecosystem services. In the context of ecosystem service markets, bundling facilitates the trading of multiple ecosystem services as a single credit that reflects improvements in those ecosystem services. Payments and ecosystem service markets can be developed for bundled services. Bundling, however, requires the use of environmental indices to describe the effects of land use and management practices on various ecosystem services, or, at least a good proxy for these effects, such as an acre of wetland (Cooley and Olander, 2011; Marshall and Selman, 2011). One way to organise markets for bundled ecosystem services is to establish an aggregator institution that buys bundled credits from land owners and then unbundles and sells them to different individual ecosystem service markets (Binning et al., 2002; Marshall and Selman, 2011).

Additionality is a key criterion determining the efficiency of payments for ecosystem services and ecosystem services offset mechanisms. Additionality ensures that the adopted land use or management practice compensates (offsets) the adverse impact and thus ensures ecosystem service integrity of market mechanisms (Bennett, 2010; Cooley and Olander, 2011). As regards additionality, a practice adoption is eligible for a payment through ecosystem services markets only if the offsets generated come from practices that would not have occurred in the absence of the payment (Gillenwater, 2012). Thus, offsets must go beyond the current service provision. From this standpoint, selling offsets is not additional if these offsets are provided from management practices that are privately optimal for the land owner. For example, Claassen et al. (2014) show that in the context of US agricultural conservation programs additionality is high for some measures, particularly regarding establishment of conservation structures (e.g. terraces) and buffer strips, but not as high for conservation tillage. Precise determination of baseline service provision and increased supply from adopted practices at each individual field parcel can be very costly due to the heterogeneity of landscape. Thus, in practice less costly approaches to estimate baseline provision, based on observed technologies and practices, are typically used for determining a baseline.

Certain payment mechanisms can provide differentiated incentives adjusted to the supply of spatially dependent ecosystem services. Payments can be spatially targeted and tailored to address heterogeneous opportunity costs and site-productivity of ecosystem services supply (Lankoski, 2016). Obtaining desired spatial configuration (which may include co-ordinated land use among several farmers) however would be more challenging issue for payments. As habitat contiguity and collective management may enhance ecosystem services, an important question is how to stimulate co-ordination between farmers across farm boundaries (Goldman et al., 2007; Parkhurst and Shogren, 2007; Parkhurst et al., 2002; Reeson et al., 2011; Stallman, 2011).

Government may promote desired landscape patterns with incentives that directly reward cross-boundary action in the payoff structure of participants, for example by employing an agglomeration bonus payment. Lewis and Wu (2015) analyse policy mechanisms to co-ordinate landowners' decisions and internalise the input externalities that arise when ecosystem service production functions are spatially dependent. An input externality arises because the marginal benefit of conserving a particular parcel depends on whether neighbouring parcels also conserve or adopt certain management practices. Lewis and Wu, 2015 show that Pigouvian subsidy payments[10] will fail to generate a spatially optimal supply curve when there are multiple independent landowners who are subject to spatially dependent ecosystem service production functions and input externalities. As potential policy mechanisms to be used in the context of pattern-based spatial dependencies, they propose conservation auctions and agglomeration payments.

The gains from improving policies: Quantitative illustration of policy instrument choice and ecosystem service provision

In this section a quantitative illustration is developed to examine relationships between key policy variables, agricultural production, and ecosystem outputs, highlighting potential trade-offs and synergies. It is based on a representative farm setting (heterogeneous production units) with small grains[11] and hay as the primary commodities. This quantitative example is for illustrative purposes and naturally agricultural production and environmental conditions, agriculture-ecosystem linkages, and the suite of relevant policies may vary widely across country and regional settings.

Key policy questions to be addressed with a quantitative analysis

In this section, a quantitative model is developed and then employed for illustrating some of the key policy questions discussed in earlier sections of this report. The following questions are addressed.

- Are ES supply and demand reasonably well balanced under current policy settings?
- What are the main drivers of ES supply?
- How cost-effective is the current policy mix in the provision of ES?
- How to improve the economic and environmental performance of the current policy mix?

Model description

In order to respond to the key policy questions, a quantitative model (non-linear mathematical programming model) and analysis is developed on the basis of Finnish data (Box 2). A set of 100 different field parcels are modelled; these differ with respect to soil textural class, soil productivity, field parcel slope, and compass opening direction of field-margin semi-natural habitats (whether a field margin, a field-forest border, or a buffer strip). The field parcels are considered as independent production units and no explicit landscape configuration is assumed; that is, field parcels are not connected to each other in a specific fashion. In each field parcel, optimal input use (nitrogen fertiliser application and size of buffer strip) is solved first and then a field parcel is allocated to the crop with the highest profits or social welfare depending on whether profits or social welfare is maximised.

10. A Pigouvian subsidy is a subsidy provided to an activity on the grounds (i.e. with the explicit rationale) that the activity generates external benefits (i.e. positive externalities).

11 Cereals such as wheat, oats, barley, rye, rice having relatively small kernels or sometimes a relatively small plant

Box 2. Model description

Spring barley is selected as a representative cereal crop for this application as it is the most common cereal crop covering about half of the field parcels under crop cultivation in Finland. Alternative land use in the model is grassland (fodder hay production).

Per hectare crop yield is modelled as a function of nitrogen fertiliser application. Mitscherlich yield function is applied for spring barley to define the optimal fertiliser application and yield level.

$$f(l_i;q_i) = \varphi_i(1 - \sigma \exp(-\rho l_i)) \quad (1)$$

Where l_i is nitrogen application rate in a given field parcel i (i=1…100) and φ_i, σ and ρ are parameters. These parameters have been estimated by Bäckman et al. (1997) on the basis of Finnish field experiments. The yield function for grassland (fodder hay) is assumed to have the quadratic form

$$f(l_i;q_i) = a + b_i l_i + c l_i^2 \quad (2)$$

where a, b_i and c are parameters. These parameters are based on Lehtonen (2001).

The soil type under cultivation is assumed to be one of the three most common soil types in Finland; clay, sandy and organic soil. Clay and silt soils cover over half of the cropland area in Finland, organic about 14% and sandy soils 35% (Puustinen et al., 2010). The crop yields depend on soil type; clay > organic > sandy as well as soil productivity. Soil productivity differences are incorporated through maximum yield parameter, φ_i and slope parameter b_i. Production cost data (variable costs and labour costs) for barley cultivation and hay production are based on Tuottopehtori e-service (ProAgria Association of Rural Advisory Centres).

With regard to field slope distribution 60% of field parcels has a slope less than 1%, 33% of field parcels has a slope between 1% and 3%, and 7% of field parcels has a higher than 3% slope (Puustinen et al., 2010).

This model application focuses on provisioning ecosystem services and negative environmental externality in order to illustrate a balanced approach for policy instrument choice, design and implementation. Provisioning services include market goods (spring barley and dry hay) and non-market goods, namely semi-natural wildlife habitats provided by enlarged field margins, field-forest borders, buffer strips, and green fallowing. Nitrogen runoff from field parcels to watercourses represents negative externality.

Semi-natural wildlife habitat areas are linked to plant and butterfly species diversity by using concave species-area relationship, $S = \mu A^{\xi}$, where μ and ξ are parameters and A is the area of wildlife habitat. On the basis of Finnish data Ma et al. (2002) modified this relationship to describe the relative effect of the width and length of the wildlife habitat, $S = \psi \Lambda^{\varphi_\alpha} W^{\varphi_\beta}$, where φ_α (φ_B) is an estimate for the average change in species richness due to an increase in the length (width) of the area while keeping the width (length) of the area constant ($\psi = 1.6331$, $\varphi_\alpha = 0.0009$, $\varphi_\beta = 0.0977$).

In addition to the area of semi-natural wildlife habitat, plant and butterfly species richness is affected by soil type, field slope and compass opening direction of the semi-natural habitat. Soil type (soil textural class) and its chemical composition have large effect on plant species richness and through that on diversity of butterflies. On sandy soils plant and butterfly species, richness is usually higher than in other soil types (Kivinen et al., 2006). Generally, plant species richness increases when soil nutrient level decreases. As regards compass direction, for example, field-forest borders opening to the South have usually higher plant and insect species richness and abundance than forest borders opening to other compass directions (Bäckman et al., 2004). With regard to field slope species richness of rare meadow plant species is usually highest in steep slope meadows opening to the South (Luoto, 2000 and Pykälä et al., 2005). Also, butterfly species richness correlates with the slope of the field edge (Kuussaari et al., 2007). On the basis of this information biodiversity weights are derived for each differential field parcel that takes into account soil type, field slope and randomised compass opening direction of the semi-natural habitat in a given field parcel.

Following nitrogen runoff function based on Simmelsgaard (1991) and Simmelsgaard and Djurhuus (1998) is employed,

$$Z_l^i = (1 - m^{\eta})\phi_i \exp(\gamma_0 + \gamma l_i (1 - m)) \quad (3)$$

where Z_l^i = nitrogen runoff at fertiliser intensity level l_i, kg/ha, ϕ_i = nitrogen runoff at average nitrogen use, $\gamma_0 < 0$ and $\gamma > 0$ are constants and l_i = nitrogen fertilisation in relation to the normal fertiliser intensity for the crop, 0.5 ≤ l ≤ 1.5. This runoff function represents nitrogen runoff generated by a nitrogen application rate of l_i per hectare and the parameter ϕ_i reflects differences in crops, soil type, and field slope. The first right-hand side term of (3) describes nitrogen uptake by the buffer strips. It is calibrated to reflect Finnish experimental studies on grass buffer strips (Uusi-Kämppä and Yläranta, 1992, 1996, Uusi-Kämppä and Kilpinen 2000). The second right-hand side term represents nitrogen runoff generated by a nitrogen application rate of l_i per hectare when buffer strips take up a share *m* out of this field parcel.

In the absence of government policy, farmers' short-run restricted profits are defined as $\pi_i = pf(l_i;q_i) - cl_i - K$, where *p* and *c* represent the respective prices of crop and nitrogen fertiliser and *K* other cultivation costs per hectare.

In the case of government policy to enhance semi-natural habitat provision and to reduce nitrogen runoff from field parcels to watercourses let *m(q)* denote the share of field parcel of productivity *q* allocated to crop production that is retained as a semi-natural habitat (whether buffer strip, field-forest border biodiversity strip or enlarged field margin). Heterogeneous productivity of land implies that the establishment and management of semi-natural habitats, *m*, results in differential opportunity costs in different field parcels. Also, biodiversity benefits and nitrogen runoff reduction effectiveness differ due to differential site-productivity of environmental service provision (owing to differences in soil types, field slopes, etc.).

Depending on how much information government has regarding farmers' opportunity costs and site-productivity of environmental service provision either differentiated or uniform policy instruments can be designed and implemented. Simple policy instrument could be, for example, a uniform payment scheme in which government sets a fixed width for buffer strip or biodiversity strip, $\hat{m}$, (e.g. 5 or 10 metres) and provides uniform payment $\hat{b}$ for participating farmers. In this case farmers' profits are defined as

$$\pi_i = (1-\hat{m})\left[pf(l_i;q_i) - cl_i - K + \Omega\right] + (\hat{b} - h)\hat{m} \tag{4}$$

where *h* denotes the annualised establishment and management costs of buffer strip or biodiversity strip.

Note that if government also supports farm incomes by providing crop area payment, Ω, which is not paid for field parcel area under semi-natural habitats then this type of crop area payment will increase farmers' opportunity costs of establishing or maintaining semi-natural habitats as shown by equation (4). Similarly on the basis of equation (4) one can show that increased market demand and thus increased equilibrium price, *p*, for barley, or government price support that increases output price of barley will increase farmers' opportunity costs of establishing semi-natural habitats as farmers' income forgone increases. Government policy that lowers the cost of polluting input, that is, nitrogen fertiliser, would naturally result in policy failure that exacerbates environmental market failure (nitrogen runoff as negative externality) and it would also increase farmers' income forgone for establishment of semi-natural habitats. Hence, all these policy interventions increase farmers' opportunity costs and thus raise the level of ES payment required for adoption of the practice and thus establishment of semi-natural habitat.

Policy scenarios

Benchmark scenarios

Two benchmark scenarios are used to compare performance of alternative policy instruments and policy packages.

- *The private optimum*, obtained assuming farmers maximise profits from commodity production and do not consider positive or negative externalities (ES provision and nitrogen runoff, respectively)
- *The social optimum* incorporating social benefits from semi-natural habitat provision and social damage from nitrogen runoff.[12]

12. In order to solve and compute the social optimum, environmental valuation estimates have been used. Nitrogen runoff damage estimate is EUR 6.8/kg of N and is based on Gren (2001) and Gren and Folmer (2003). Biodiversity benefit estimate is EUR 54/ha of semi-natural habitat and derived from Yrjölä and Kola (2004).

The social optimum benchmark is used to analyse whether the ES supply corresponds to social demand. Another option to investigate the balance between ES supply and demand could be to use a government's explicit quantitative targets set for total nitrogen runoff and total area under different types of semi-natural habitats.

Market and policy drivers

To analyse and compare market and policy drivers of ecosystem services, two situations are compared with the above benchmarks:

- A scenario with a crop-specific area payment as the sole policy instrument (with EUR 50/ha), which represents traditional farm income support policy instrument, and is paid for arable land under barley cultivation;
- A scenario with an exogenous price increase for barley (9% increase) resulting from either an increased market demand or market price support (e.g. due to trade barriers). The level of price increase has been set to level in which it has the same direct impact on farmers' profits per hectare as a crop area payment of EUR 50/ha for barley.

Representation of the current policy mix

The current agricultural and agri-environmental policy mix in Finland represented in the model consists of several farm income support payments as well as several types of agri-environmental measures under "Agri-environment-climate payments". The aim here is not to provide a comprehensive representation of the current policy mix in Finland but rather to illustrate some key features by selecting a "stylised policy package" with strong influence on ES provision and nitrogen runoff. The following measures are selected for this stylised policy package:

- *Farm income support instruments*. They include the CAP basic payment (EUR 192/ha) (former Single Farm Payment) and the less-favoured area (LFA) payment (EUR 217/ha) and they are paid on agricultural land uses (including barley and hay but also environmental measures specified below);
- *Agri-environment-climate payments*. These include the following measures:
    - A farm-specific payment associated with nitrogen application constraints, which are differentiated according to soil type (sandy, clay and organic) (EUR 54/ha);
    - Three field-parcel specific payment measures: to support riparian buffer strips (also called buffer zones) (EUR 450-500/ha), biodiversity field (EUR 300/ha), and biodiversity strip (EUR 300/ha)[13].

Farmers' establishment and management costs for field parcel specific measures are derived mainly on the basis of Miettinen et al. (2014). The total annual establishment and management costs are EUR 187/ha for riparian buffer strips, and EUR 243/ha for biodiversity field and strips.

13. Biodiversity fields are modelled as field-forest border biodiversity strips since biodiversity fields can be implemented as a strips covering only part of field parcel.

Policy analysis results

The results (from the analysis of one hundred differential soil type and land productivity combinations) are summarised through a set of synthetic outcomes (Table 7). Ecosystem services provision is reflected through three indicators: commodity production, nitrogen runoff and biodiversity benefits. Total commodity production (sum of barley and dry hay) depends on nitrogen application intensity, the size of buffer strips and land allocation between barley and hay. Total nitrogen runoff damage reflects nitrogen fertiliser intensity, the size (or lack of) buffer strips and land allocation between barley and dry hay (which has both lower nitrogen application intensity as well as lower propensity to nutrient runoff due to more permanent plant cover). Total biodiversity benefits mainly reflect the establishment of buffer strips and land allocation between barley and dry hay. A compound indicator is also calculated to estimate total social welfare (SW) as the sum of farmers' profits from commodity production (payment transfers excluded), nitrogen runoff damage and biodiversity benefits.

The current policy mix promotes a relatively good ecosystem management

In comparison to the private optimum, the current policy reduces nitrogen runoff damage, increases biodiversity benefits and improves social welfare. Relative to a situation without policies, the current policy mix reduces nitrogen application intensity and provides incentives for the establishment of buffer strips. Thus, nitrogen runoff damage is reduced while biodiversity benefits are increased. Although total commodity production is decreased, the overall social welfare is increased.

However, the current policy mix could do better in terms of ecosystem service provision and social welfare. In comparison to the social optimum, the current policy mix provides a lower bundle of ecosystem services. The current policy mix induces a similar fertiliser intensity compared to the social optimum, although the fertiliser use range is different from the social optimum. With relatively high payment levels, the current policy mix provides very strong incentives for the establishment of buffer strips and their size under the current policy mix clearly exceeds their socially optimal level. Compared to the social optimum, the buffer strips size is too high in locations where smaller size buffer strips would suffice (high productivity of land and less steep slope) and too small in locations where land productivity is low and field parcels are steeper. Moreover, when comparing total commodity production under the current policy mix with the social optimum one can see that extensive margin impact (land allocation between barley and hay) dominates intensive margin (nitrogen application intensity) in the sense that despite more stringent fertiliser application (range) and larger size of buffer strips total commodity production is higher under current policy due to land allocation shift from hay to barley. Despite smaller buffer strips, the biodiversity benefits under the social optimum are higher than under the current policy mix as more land is allocated to hay production under the social optimum.

These stylised results mask some inefficiencies produced by current policy. The level of buffer strip payment (EUR 450-500/ha) relative to other payments (EUR 300/ha) is so high that it is profitable for farmers to establish buffer strips in every field parcel. In fact under the actual Finnish policy in 2015, the area under buffer strips was also eligible for CAP basic payment (EUR 192/ha) and less-favoured area (LFA) payment (EUR 217/ha). Due to the high level of payments and the possibility to allocate a whole field parcel as a buffer strip, this measure became profitable for farmers to adopt. From the beginning of 2016, the Finnish Government has changed the rules so that the LFA payment is paid only to buffer strip areas that cover a maximum of 25% of farm's total arable land area (this constraint takes also into account other "fallowing" types of land uses within farm, such as biodiversity fields).

Crop-specific area payments are not environmentally neutral. Relative to the private optimum, the crop-specific area payment scenario does not affect optimal fertiliser intensity, but increases the total use of nitrogen through its impact on extensive margin, that is, more land is allocated to more fertiliser-intensive crop barley. Price support increases directly optimal fertiliser intensity and also increases total use of nitrogen fertiliser through its impact on the extensive margin. In comparison to the private optimum both the crop area payment scenario and the price support scenario increase nitrogen runoff damage. Hence, crop-specific area payments are not environmentally neutral due to the changes they induce in crop allocation. Price support is even more environmentally distortive due to its additional impact on the intensive margin (increases input use intensity). However, at similar levels, price support is less distortive on crop allocations (extensive margin) than crop-specific area payments. As a result, price support has slightly higher biodiversity benefits than crop-specific area payment due to more diversified land use. Environmental distortions induced by crop-specific area payments are also reflected in biodiversity benefits, since they are lower under the crop-specific area payment scenario than under the private optimum. The social welfare under the price support and crop-specific area payment scenarios is lower than under private optimum, despite increased commodity production. This result is mainly driven by a significant increase in nitrogen runoff damage.

Commodity price level has the strongest impact on ecosystem service provision in the current policy setting

Additional policy scenarios using the current policy mix as a basis are developed to identify which variables affect ecosystem services provision the most. To do so, variations are induced in some of the potential drivers of ecosystem services provisions: the price of commodities, the cost of fertiliser, the level of buffer strip payment, and the level of all support payments (Table 7).

Table 7. Policy scenarios and results

Policy scenario	Nitrogen application barley, kg/ha	Buffer strip size, % of field parcel	Total commodity production, kg	Total N-runoff damage, EUR	Total biodiversity benefits, EUR	Total SW, EUR	Land allocation, barley : hay
Private optimum	109 (95-121)	-	438 667	7773	1192	6260	90:10
Social optimum	94 (79-105)	3.8 (1.2-15.6)	369 012	5270	1539	7699	66:34
Current policy	94 (60-100)	10.0 (7.3-12.8)	435 276	6062	1401	6650	100:0
Crop-specific area payment	109 (95-121)	-	456 356	8271	1112	5557	100:0
Price support	114 (100-126)	-	458 307	8428	1128	5416	98:2

Commodity price level has the strongest impact on ecosystem service provision in the current policy setting

Additional policy scenarios using the current policy mix as a basis are developed to identify which variables affect ecosystem services provision the most. To do so, variations are induced in some of the potential drivers of ecosystem services provisions: the price of commodities, the cost of fertiliser, the level of buffer strip payment, and the level of all support payments (Table 8).

A commodity price boost increases commodity production significantly, but reduces ecosystem service provision and social welfare. A 20% price boost for barley naturally results in a strong increase of commodity production, support transfers, and profits while there are strong negative environmental effects in terms of increased nitrogen runoff damage and decreased biodiversity benefits. Hence, the total social welfare is clearly reduced relative to the current policy mix scenario with current prices.

A fertiliser price increase increases ecosystem service provision, while commodity production and profitability are reduced and as a result also social welfare. In the second scenario fertiliser price is assumed to increase by 20%. This fertiliser price increase favours hay relatively to barley, since hay is a less fertiliser intensive crop. Although this policy scenario performs reasonably well with regard to nitrogen runoff damage and biodiversity benefits, it reduces social welfare because the increased cost of fertiliser reduces farmer's profits from production.

A reduction of buffer strip payment level increases commodity production and diversifies land use, but also increases nitrogen runoff damage and reduces biodiversity benefits. In the third scenario the current high level of buffer strip payment is reduced by 40%, taking it to the same level as other environmental payments (that is, biodiversity field and biodiversity strip). This reduction of buffer strip payment affects land allocation so that hay production becomes more profitable in some locations and thus farmers shift from barley to hay. The total commodity production increases as established buffer strips are now much smaller, and the total payment transfers to the farmers decrease. Although more land is allocated to hay, the overall environmental performance decreases since smaller buffer strips results in higher nitrogen runoff damage and reduced biodiversity benefits. Hence, the total social welfare decreases.

A reduction of all support payments by 30% increases commodity production and diversifies land use, but increases nitrogen runoff damage and reduces biodiversity benefits. In the fourth scenario all government payments are reduced simultaneously by 30%. Again, this favours hay production relative to barley. The social welfare impact is limited compared to the current policy mix scenario since the higher total commodity production offsets increased nitrogen runoff damage and reduced by biodiversity benefits.

The introduction of a fertiliser tax along with environmental cross-compliance promotes ecosystem service provision and reduces payment transfers to farmers. The last scenario focuses on minimising the budgetary burden of ecosystem service provision. Here the idea is to implement a uniform 30% nitrogen tax together with CAP basic payment with environmental cross-compliance requirement of uniform 3-meter wide buffer strip. The fertiliser tax-revenue is used to finance an additional "top up" crop area payment to compensate environmental cross-compliance requirement. Relative to the current policy scenario, the commodity production is increased, payment transfers are reduced, and farmers' profits decreased. Nitrogen runoff damages increase as a uniform 3 meter wide buffer strip is not enough to reduce runoff in most of locations. However, the overall social welfare performance is not radically reduced relative to current policy.

The commodity price level appears to be the strongest driver of ecosystem services provision, followed by farm payments and fertiliser cost. When comparing the impact of a variation of the different drivers, the price of barley appears to have a stronger influence than the other drivers considered. A 20% increase for barley price induces more impact on nitrogen runoff and a similar impact on biodiversity benefits than a 30% or 40% variation of farm payments, and the difference is even more important when compared to the effect of an increase of fertiliser costs. In this simulation, farm payments influence ecosystem services provision more than the cost of fertiliser, since a 30% variation of farm payments induces twice the impact a 20% fertiliser price has on biodiversity benefits and three times the impact the latter has on nitrogen runoff.[14]

Table 8. Additional policy scenarios and sensitivity analysis

Policy scenario	Land allocation, barley: hay	Total commodity production, kg	Payment transfer, EUR	Profit, EUR	Total N-runoff damage, EUR	Total biodiversity benefits, EUR	Total SW, EUR
Private optimum	90:10	438667	-	11431	7773	1192	6260
Social optimum	66:34	369012	-	12989	5270	1539	7699
Current policy	100:0	435276	43572	54883	6062	1401	6650
Current policy + Barley price +20%	100:0	460680	53574	65738	8543	1266	4887
Current policy + fertiliser price increase 20%	92:8	406597	43593	52227	5558	1453	4529
		438649	41202	53939	7967	1262	6032
Current policy: all payments reduced by 30%	93:7	439517	29216	41762	7818	1285	6012
Simple policy: fertiliser tax 30% + basic crop-payment with ECC	100:0	431366	19200	31364	6947	1238	6454

Budgetary cost-effectiveness of existing policy mix could be improved

The current policy mix is expensive in budgetary terms and has low budgetary cost-effectiveness due to a lack of environmental targeting. Although the current policy mix performs well in terms of ES provision it is quite expensive in budgetary terms. The total budget for A-E measures (fertiliser use constraint and buffer strip subsidy) reaches

14. Since the study area is small, commodity prices are exogenous and thus there are no endogenous price feedbacks due to supply changes.

EUR 8 183 for 100 ha of land resulting in an average payment of about USD 82/ha. Modelling results show that farmers' income forgone from these two measures is EUR 2 493. Thus, under the current policy mix farmers' income forgone is overcompensated by EUR 5 690. This overcompensation is called an information rent and arises from asymmetric information between policy makers and farmers regarding the level of income forgone for adopting agri-environmental measures. The current uniform payment means that farmers whose income forgone is small will earn a large information rent and thus will be highly overcompensated. This is quite a remarkable overcompensation and has a quite dramatic effect on the budgetary cost-effectiveness of the current policy.

The following simulations analyse whether ES provision can be promoted with a smaller budget and more targeted measures.

Table 9 provides results from these simulations. In these simulations, the agri-environmental budget is set at EUR 2 000. Four different options are analysed: (i) the current uniform payment policy, (ii) the current uniform payment policy with environmental benefit index (EBI) targeting, (iii) the current uniform payment policy with EBI/cost targeting, and (iv) a discriminatory price auction with EBI targeting.

Better targeted payments provide large improvements in budgetary cost-effectiveness. The simulation results show that using instruments allowing more targeted payments would enable, with the same budget, to multiply the production of ES by 1.7 or even six times in the best case. The current policy performs poorly in this respect and thus with the defined budget only affects 22 field parcels. Adjusting the current policy with EBI or EBI/cost targeting reduces the average information rent by 50% and enables to accept more parcels in the programme. Environmental targeting (EBI) or EBI/cost targeting also greatly improves environmental benefits achieved with the same fixed budget. In fact total EBI points achieved with such environmental targeting are almost twice as large relative to the EBI achieved by current policy. This is also reflected in budgetary cost-effectiveness (EBI/EUR ratio) that improves greatly due to environmental or benefit/cost targeting. A discriminatory price auction reduces farmers' information rents even further relative to other measures. With such an auction system, the available budget can be stretched remarkably and almost all 100 field parcels can be accommodated into the programme. Due to the high rate of participation and the targeting allowed by this measure, the total of EBI points is high and the budgetary cost-effectiveness in terms of EUR/EBI is greatly improved.

The incorporation of policy-related transaction costs does not change the ranking of alternative payment approaches (last column of Table 9). This is consistent with the conclusions from Lankoski (2016) indicating that the benefit from targeting usually outweighs the additional policy-related transaction costs associated with such targeting.[15]

15. It is worth noting that complex agri-environmental programmes designed to improve cost-effectiveness may dissuade some land owners from applying, particularly when auction design minimises information rent (overcompensation of income forgone) or when there is a significant chance that landowners application (or bid) will be rejected (Palm-Forster et al., 2016).

Table 9. Cost-effectiveness gains from improved targeting

Policy scenario	N° of parcels accepted within the budget	Average information rent, EUR/ha	Budget	Total EBI points	EUR/EBI	EUR/EBI with policy-related transaction cost
Current uniform payment policy	22	73	1976	886	2.23	2.41
Current uniform payment policy with EBI targeting	24	36	1954	1510	1.29	1.35
Current uniform payment policy with EBI/cost targeting	25	38	1941	1541	1.26	1.32
Discriminatory price auction with EBI targeting	94	6	1985	5423	0.37	0.38

Could a single policy instrument be considered to address the provision of several bundled ecosystem services?

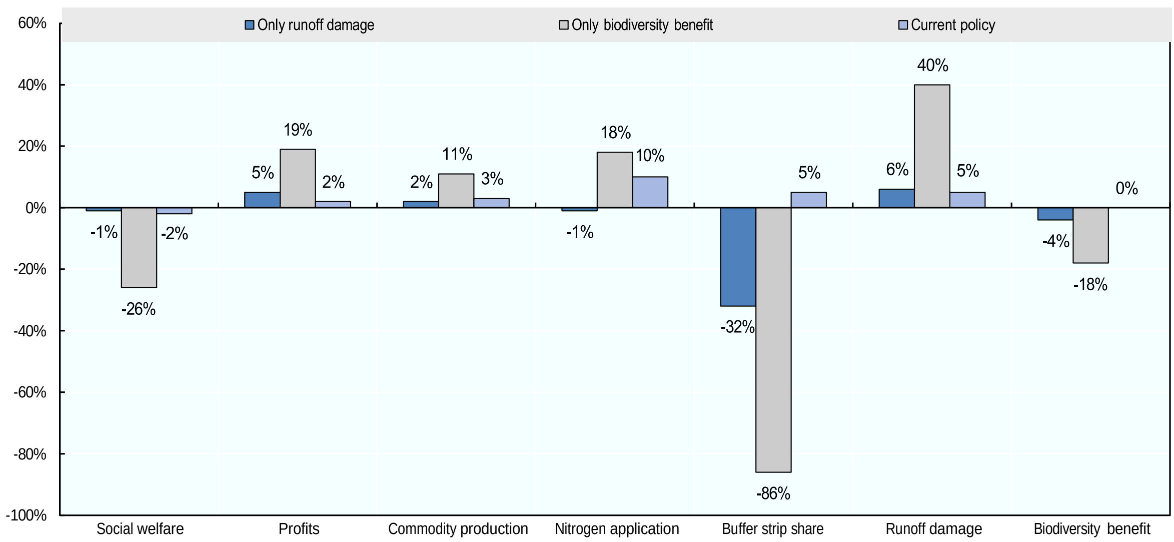
The fact that the provision of ecosystem services is often linked may seem to question the orthodox policy making view advising to use each policy instrument separately to address each market failure. In this section, this question is addressed through the simulations developed to illustrate whether single policy instruments are able to address the provision of multiple ecosystem services that may be linked.

Comprehensive versus partial consideration of ecosystem services

We first compare the current policy mix with the optimal provision of a single ecosystem service (biodiversity benefit or nutrient run-offs avoidance). Figure 10 compares the balance of ecosystem service provision under three different policy scenarios: the current policy mix described in the previous section, the socially optimal provision of nitrogen runoff avoidance, and the socially optimal provision of biodiversity benefit. The socially optimal provision of a given ES provision is solved by the model through the use of a combination of a tax on nitrogen fertiliser, and a buffer strip payment (the weight of the two instruments is different in each scenario). All scenarios are compared to the benchmark of the socially optimal provision of all ES services represented by the 0% line on the figure.

If a policy maker only considers the provision of one ecosystem service (either nitrogen runoff damage or biodiversity benefit from semi-natural habitat provision) then the optimal level of policy instruments (tax on nitrogen fertiliser and buffer strip payment) also changes as does the optimal provision of ecosystem services (Figure 10). If the policymaker focuses only on nitrogen runoff reduction then the divergence from socially optimal provision when both services are considered and internalised is not very large. This is reflected in the social welfare (SW) estimate which is only 1% lower in this case relative to social optimum where both ecosystem service effects are internalised. If the policy maker instead considers only biodiversity benefits from semi-natural habitat provision then the difference with the social optimum is much larger and the balance of ecosystem service provision is quite different. In this case the SW estimate is 26% lower. The main reasons for this are that the optimal buffer strip size is small when only biodiversity benefit is considered, that is, the opportunity cost of a buffer strip establishment stays same but the marginal social benefit of such establishment is decreased as its contribution to nitrogen runoff damage reduction is not considered. Moreover, nitrogen application is not addressed at all under this scenario, since nitrogen runoff damage is not considered as an objective. As a result of higher nitrogen application rate and smaller size of buffer strips, nitrogen runoff damage is increased by 40%. Commodity production increases by 11% but trade-offs related to runoff damage reduce the SW significantly.

Figure 9. Balance of ecosystem service provision under policy solutions relative to socially optimal provision of ecosystem services



The current policy performs relatively well in many respects with an overall balance of ecosystem service provision close to the socially optimal one. The SW estimate for the current policy scenario is only 2% lower than the social optimum. Nitrogen application is 10% higher than what would be socially optimal, but with respect to key indicators of ecosystem services: commodity production (+3%), nitrogen runoff (+5%), and biodiversity (+0.5%) it differs very little from social optimum. Hence, current policy does not involve major trade-offs regarding various ecosystem services, since both nitrogen application and buffer strip establishment are addressed and the level of both measures is close to socially optimal.

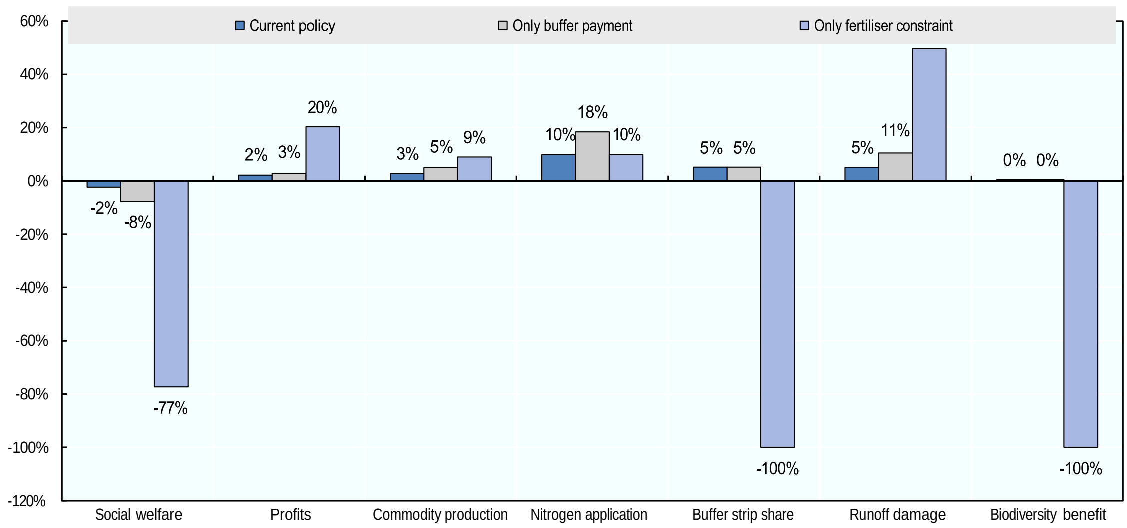
These results illustrate the linkages between ecosystem services and their provision and shows that focusing on a single service provision may result either in good solutions (when synergies occur) or socially non-satisfactory solutions (when synergies do not take place).

Complete versus partial toolbox of measures

The model is now used to illustrate the comparative performance of single environmental measures (either fertiliser use constraint or buffer strip) with that of their combination (implementation of both measures; buffer strip and fertiliser use constraint) regarding the balance of ecosystem services provision (Figure 11).

If only one of the measures is implemented then the SW is reduced relatively to the current policy scenario that implements both. The reduction of SW is large when only fertiliser use constraint is implemented (77%). In this case buffer strips are not established and thus biodiversity benefits are lost and nitrogen runoff damage increases by 50%. Commodity production and profits are increased (by 9% and 20%, respectively) but their increase does not offset increased environmental damage. Thus, fertiliser use constraint is a measure that involves clear trade-offs regarding various ecosystem services, and thus its performance as a single measure policy to address multiple ecosystem services is poor.

Figure 10. Balance of ecosystem service provision under current policy when both nitrogen fertiliser constraint and buffer strip



When the single measure policy is buffer strip payment then the SW is reduced relatively little (8%) and the "portfolio" of ecosystem service provision is relatively balanced. In comparison to the current policy scenario with both measures (fertiliser use constraint and buffer strips) it results in slightly higher commodity production, profits and nitrogen runoff damage while biodiversity benefits are the same. The SW is 5.6% lower than under Current policy. Hence, in a scenario with a single policy instrument, the buffer strip payment performs reasonably well in addressing several ecosystem services simultaneously.

Moreover, instrument base matters for the enforcement of the policy instrument, since buffer strips are relatively easy to monitor and control while field parcel specific fertiliser application is much more difficult to monitor and control without, for example, soil tests.

A key message from the analysis above is that ecosystem services are bundled through input use and land allocation and there are synergies and trade-offs between alternative ecosystem services. Policy instruments can contribute to both synergies and trade-offs by selection of instrument base (input use, land use, tillage practices). Intensity of synergies and trade-offs can be modified by smart choice of instrument base. In the examples above land use based measure buffer strips seem to perform well in terms of promoting synergies and reducing trade-offs while fertiliser use constraint clearly fails in this respect.

Towards a policy decision tree for the agriculture-ecosystem relationship

Before considering the design of new policy instruments to address ES provision, one needs to ensure that existing policies do not provide perverse incentives,[16] and if they do, these policies should be addressed first. As suggested by the quantitative modelling example, a key policy recommendation would be to reform policies that tend to increase agricultural commodity prices. Because of the bundled and spatial pattern-based characteristics of ecosystem service provision a policy mix may be required to achieve an efficient solution. A general recommendation as regards such policy mix is to be attentive to the coherence both among combined instruments and with other related policies. In the case of a policy that targets a particular ecosystem service in agriculture, this means taking into consideration the potential conflicts between this and other policies. In biodiversity conservation for example, removing environmentally harmful subsidies is considered a first step to improving policy consistency and coherence.

The decision tree focuses on cost-efficient economic instruments, but other types of instruments can be combined to strengthen performance, enhance enforcement or reduce policy-related transaction costs. For example, inherent information problems related to nonpoint source pollution may necessitate the use of multiple policy instruments (both regulation and economic instruments) to ensure both environmental effectiveness and cost-effectiveness. The role of different types of policy instruments in the policy mix is highly dependent on the given bundle of ecosystem services considered. For example, the role of direct regulation may be important in the case of ecosystem services that are close to thresholds leading to irreversible change (such as pollination services).

Overall, combined policies or policy packages should ensure effectiveness and cost-effectiveness of policy, with instruments aimed at safeguarding environmental performance (direct regulation) and others aimed at minimising costs (economic instruments).

The first step of the proposed decision tree (Figure 12) is a general assessment of the three dimensions of the agriculture-ecosystems services relationship: pressures from, provision of and dependence on. This assessment would help select the most important issues and services to focus on for these three dimensions. The following step is to identify the main drivers (policy and market) susceptible to influence these three dimensions. Policy drivers that provide perverse incentives (wrong or inconsistent signals) should be considered first and reformed. Moreover, ES policy instruments need to be consistent with the current policy mix. The effectiveness and cost-effectiveness of the ES instrument is highly dependent on how the existing policy mix affects land use and ecosystem services and how different policy instruments interact with each other. For example, the potential cost-effectiveness gains from spatially targeted and tailored policy instrument to address ES cannot be realised if policy instruments in the current mix conflict severely with the chosen policy instrument or limit their scope. In the case of conflicting signals and interactions one needs to consider reforming the existing policy mix first.

Then, on the basis of steps 1 and 2, the third step consists in setting policy priorities through investigation of potential synergies and trade-offs between provision of different ES and how different practices may modify these synergies and trade-offs. Steps 2 and 3 are

16. A perverse incentive is an incentive that has an unintended and undesirable result which is contrary to the interests of the incentive makers.

investigated in Figure 12 which goes more into detail on how to choose the right policy instrument for ecosystem services whose provision may be linked.

Figure 11. Approach for setting the bundle of policy priorities regarding the agriculture-ecosystem relationship

Step 1: Assessment of agriculture-ecosystem relationship

1.A Assessment of pressures on ecosystems from agriculture
Low, Medium, High
Ranking of pressures
Pressure 1
Pressure 2
…

1.B Assessment of ecosystem services provided by agriculture
Current supply
Demand
Gap=demand-supply
Ranking of ES provision gaps
ES 1
ES 2…
…

1.C Assessment of ecosystem services flowing to agriculture
Current supply
Demand
Gap=demand-supply
Ranking of ES dependencies
ESD 1
ESD 2
…

Step 2: Identify key market and policy drivers affecting pressures and ecosystem services
Market and policy drivers affect the opportunity cost of policy objectives regarding ecosystem services

Step 3: Setting of policy priorities
Considering synergies and trade-offs created by different management practices and policy measures:
3.1. For bundled ecosystem services, investigate how different instrument bases (input use, land use, tillage practices) create synergies and trade-offs regarding (see decision tree 2)
3.2. Investigate whether bundle linkages can be modified (see decision tree 2)
3.3. Based on available data, consider conducting *ex ante* cost-benefit analysis regarding alternative policy instrument choices, taking into consideration policy-related transaction costs.

Step 4: Determining the optimal mix of policy instruments to reach these objectives
See decision tree (Figure 13)

The first question for policymaking is to identify whether and how different ecosystem services provisions may be bundled (e.g. through input use, land use or tillage practice); and if so whether there are situations where the provision of one service (e.g. commodity production as a provisioning service) may increase or reduce the production of the others.

Four ecosystem services are used to illustrate the process: ecosystem services with positive effects (commodity production, semi-natural wildlife habitat provision, and soil carbon sequestration[17]); and one negative ecosystem services provision (nutrient runoff). To illustrate potential interactions among these four services, three different types of practices are considered: (i) nutrient precision application, (ii) no-till, (iii) buffer strips, enlarged field edges, and green fallowing.

17. Net-GHG emissions from agriculture, that is, GHG emissions minus soil carbon sequestration.

Potential synergies and trade-offs related to these practices are as follows:

> *Nutrient* precision *application* would positively affect net-GHG emissions, commodity production and nutrient runoff reduction and would be neutral regarding semi-natural habitat provision.
>
> *No-till practices* would positively affect net-GHG emissions and nutrient runoff (reduces nitrogen and particularly phosphorus runoff but may increase dissolved phosphorus runoff – overall effect positive), and biodiversity, especially regarding soil fauna. Commodity production may decrease in the short-run but after transition period (about five years) improvement in soil structure would support better crop yields.

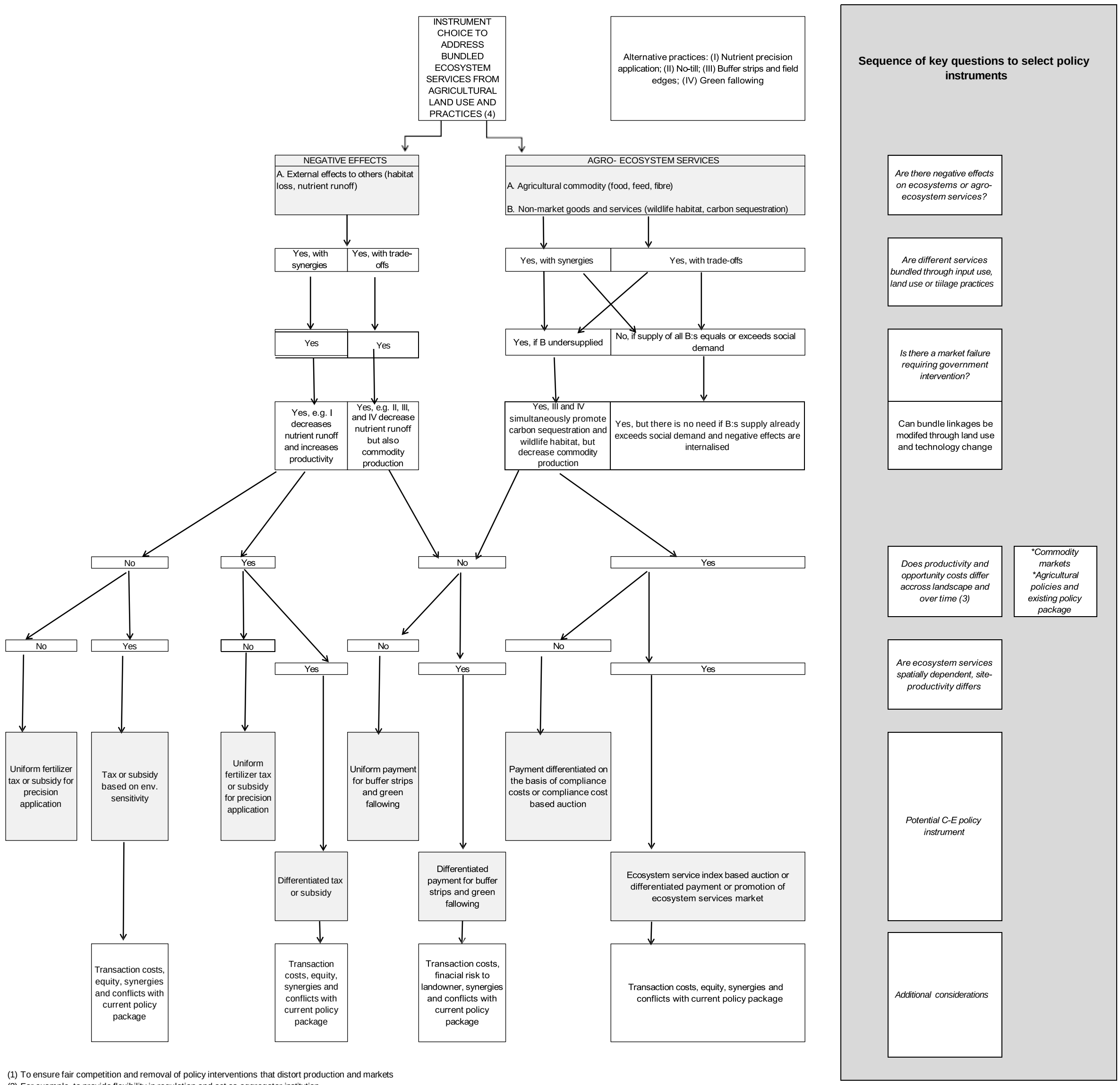
Buffer strips, enlarged field edges, and green fallowing would positively affect semi-natural wildlife habitat provision, net-GHG emissions, and nutrient runoff reduction but may negatively affect commodity production. If trade-offs and synergies exist between provision of the different services then the question is whether policies may be designed to strengthen synergies and minimise trade-offs. Here the key issue is the choice of instrument base (input use, land use, tillage practice). As the modelling example above illustrated, buffer strip payments can help address several ecosystem services, while a fertiliser use constraint failed in this regard. So, in the Finnish context a single instrument buffer strip payment policy seemed to work reasonably well in addressing multiple ecosystem services.

However, in many cases single instrument bases (targeting either input use, land use or tillage practice) do not suffice, and may intensify trade-offs. A combination of land use based measures (e.g. buffer strips, enlarged field edges and green fallowing) with input use based measures (e.g. constraints on fertiliser or pesticide application) would work better. Likewise, trade-offs and synergies related to tillage practice choice (no-till) can be modified with a combination of it with input use measures (constraint on fertiliser or herbicide application) and land use measures (buffer strip establishment to further reduce nutrient and herbicide runoff and promote soil carbon sequestration and wildlife habitat).

It is thus likely that situations where multiple services are at stake will require multiple instruments or at least multiple policy instrument bases. This is the case in the Finnish A-E programme in which the single policy instrument, that is a voluntary A-E payment, simultaneously controls input use and land use. Another option is to use an index-based policy instrument (conservation auction using environmental benefit index) combining several ecosystem service provisions in one tool. For each service then the next step is to identify whether there is heterogeneity of opportunity costs of supply, or site-productivity and whether there is spatial dependency of ecosystem services provision.

The presence of spatial heterogeneity regarding both agricultural productivity and environmental site-productivity (or sensitivity) will determine whether uniform or differentiated policy instruments should be preferred. Uniform tools work well when both productivity and environmental sensitivity are relatively homogenous, which is rarely the case. When opportunity costs vary but environmental quality is homogenous then differentiated payments on the basis of costs should be prioritised. When environmental site-productivity varies policy instruments employing environmental targeting mechanisms (auctions and differentiated payments) work better.

Figure 12. Policy instrument choice to address the provision of ecosystem services



(1) To ensure fair competition and removal of policy interventions that distort production and markets
(2) For example, to provide flexibility in regulation and act as aggregator institution
(3) For example, opportunity costs differ over time as commodity prices change or when climate change affects productivity
(4) Before considering new policy instrument one should address first the wrong signals sent by existing policy package an correct those.

The next step is to assess policy-related transaction costs of the considered policy instruments. Instrument base matters a lot for the enforcement of the policy instrument. Some instrument bases are relatively easy to monitor and control, such as buffer strips and green fallowing, while other instrument bases may be unobservable without, for example, soil tests (field parcel specific fertiliser, manure, and pesticide application constraints). Policy-related transaction costs need to be considered already when selecting the instrument base. While these transaction costs may be worth the added performance they allow, high transaction cost can sometimes impede practical implementation of the policies for budgetary or human resources reasons. In such cases, but also if a potentially most cost-effective policy instrument is loaded with huge information needs and is costly to implement (beyond the efficiency benefits it would provide) then one may need to consider alternative options that are potentially less cost-effective but also imply lower implementation costs.

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3. Country experiences in ecosystem services and agriculture

Key messages

- An increasing range of policy initiatives, at both national and international levels are undertaken to assess ecosystem and their services, including those related to agriculture. Examples of United Kingdom, the People's Republic of China (hereafter "China") and European Union show that these initiatives constitute a useful step to move towards evidence-based policies, combining a wide-range of data and methodological approaches and tools.
- These assessment efforts should be pursued in the future because they can contribute to reshaping and reform policies. These assessments also provide an analysis based more on scientific knowledge, and therefore contribute to better frame the public debate on the matter. To date, these assessments have not translated into significant enough policy reforms. They may however be useful to prepare the ground for future policy improvement.
- Ecosystem services flowing to agriculture, such as pollination or biological regulation of pests and diseases, remain poorly understood and under-considered in most assessments. At the same time, there is rising recognition they may be important to ensure agricultural productivity growth, sustainably.
- National ecosystem assessments should be fully integrated in the policy making process, so they can foster concrete policy reforms in short-run, and be integrated in long-term strategic plans The case of the United Kingdom, China and the European Union illustrates the benefits from having a close connection between assessment and policy action and from developing follow-up phases with specific institutional tools, such as an independent Natural Capital Committee advising governments.

Country experiences in the assessment of ecosystem services linked to agriculture

Since the mid-2000s, several OECD and partner countries have been conducting national ecosystem assessments (e.g. United Kingdom, Spain, Portugal, China), or are currently doing so (e.g France). These national ecosystem assessments have different characteristics (scope, governance structure, conceptual frameworks and budgets), but they all share the same general goal, which is to provide policy relevant, up-to-date evidence base on the state and trends of ecosystems and their services, the drivers of change, and possible responses to policy challenges.

In addition to these national initiatives, *international* assessments are also being conducted, in particular by intergovernmental and supranational organisations, notably:

- the Intergovernmental science-policy Platform on Biodiversity and Ecosystem Services (IPBES), created in 2012 and that includes 123 member countries
- the EU Working Group Mapping and Assessing Ecosystems and their Services, with the framework of EU Biodiversity Strategy to 2020

- The Economics of Ecosystems and Biodiversity (TEEB), an international initiative originally initiated by the German Federal Ministry for the Environment and the European Commission (EC).

Most national ecosystem assessments typically include agro-ecosystems in their scope, within a holistic perspective looking at all ecosystems and their interactions. Some assessments analyse agriculture and agro-ecosystems in more depth through targeted thematic studies. This is for example the case of the first report of IPBES, which provides a synthesis on pollinators, pollination and food production; and the on-going TEEB for Agriculture & Food initiative, which focus on eco-agro-food systems.

The following sections summarise the main outcomes from some of these ecosystem assessments in the case of agriculture and agro-ecosystems, including: i) the main issues for agro-ecosystems identified in these assessments; ii) the key drivers of change and possible solutions to overcome these problems; and (iii) the influence these assessment had on policy making.

The United Kingdom's National Ecosystem Assessment

The UK National Ecosystem Assessment is generally considered as one of the best references in terms of national ecosystem assessment, for its quality, depth of analysis, and its policy relevance. The UK NEA main technical report, published in 2011, provides an extensive, independent and peer-reviewed assessment of the state and trends of ecosystem services in the United Kingdom, the main drivers of ecosystem changes, and simulations of future possible scenarios to support decision making. Between 2007 and 2014, the UK NEA mobilised a large number of academics, policy makers, private sector and NGOs, and stakeholders.

A noticeable strength of the UK NEA has been its close involvement in the policy making process, leading to concrete policy follow-up. Initially, the UK NEA was commissioned by the government, following the recommendation of the House of Commons Environmental Audit in 2007. In 2011, the findings of the UK NEA have been directly used as evidence base for defining environmental policy priorities through the *Government's Natural Environment White Paper*. Later, between 2011 and 2014, the UK NEA follow-on phase provided additional recommendations for policy makers about how to practically include ecosystem services in the definition and design of policies.

The UK NEA provided policy makers a critical review of evidence base on ecosystem services provided and impacted by agro-ecosystems. Overall, agro-ecosystems essentially provide food provisioning and cultural services (landscapes) but it was also confirmed that they generate significant negative impacts on several other ecosystem services (Table 10).[18] Agro-ecosystems rank first in total land area, with 39.3% of UK total land area in 2007 (18.8% for arable and horticultural land, 20.5% for improved grassland), followed in second position by semi-natural grassland (16.4%). There exists robust evidence-base for the negative impacts of agriculture on ecosystem services in the United Kingdom, notably: emissions of greenhouse gas, water quality issues due to diffuse pollution, and biodiversity loss; although the magnitudes of these negative impacts vary a great deal across land-use types and locations (Table 3.1).

18. The overall trade-off between provisioning and landscape services from agro-ecosystems and other ecosystem services is similar in several OECD countries (OECD, 2013).

Table 10. Overview of ecosystem services provided and impacted by farmland in the United Kingdom and level of evidence base

Final ecosystem service	Importance of enclosed farmland for service	Impacts of enclosed farmland on service	Evidence base	Comments
Crops, plants, livestock, fish, etc. (wild and domesticated)	High	++	AA	Strong positive score: farmland is largely managed for crops and livestock production
Trees, standing vegetation and peat	Low	+	AB	Positive score, due to small but increasing areas of biomass crops
Climate regulation	High	--	AA	Strong negative score, due to GHG emissions and depletion of carbon in soils
Water quantity	High	+/-	AA	Important for catching water for ground and surface waters, though flood risk mitigation potential often compromised by management
Hazard regulation – vegetation and other habitats	High	--	AA	Negative impact on sediment loss to watercourses, increasing flood risk downstream
Waste breakdown and detoxification	High	--/+	AA	Negative score due to diffuse (mainly) pollution leaving farmland; positive score for ability to compost green waste/AD, and sewage disposal
Wild species diversity including microbes	High	--	AA	Negative impacts; status of microbes unknown
Purification	Low	--	AB	Negative impacts on water quality as a result of diffuse pollution
Environmental settings – meaningful places incl. green and blue spaces	Low	Zero	AB	Individual sites have less significance than spaces in cities or mountain tops
Environmental settings – socially valued landscapes and waterscapes	High	++	AB	Farming management is largely responsible for the landscapes that many people cherish

Crops, plants, livestock, fish, etc. (wild and domesticated)	High	++	AA	Strong positive score: farmland is largely managed for crops and livestock production
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Environmental settings – socially valued landscapes and waterscapes	High	++	AB	Farming management is largely responsible for the landscapes that many people cherish

Source: UK NEA (2011).

The UK NEA also provided policy makers a prospective analysis of different scenarios of land-use change and ecosystem services by 2060, using a spatial land-use model. Impacts on ecosystem services were quantified and given monetary values for agricultural output, carbon storage, greenhouse gas balances, and landscape amenities. Change in biodiversity was included as a constraint in the model, since the economic valuation of biodiversity was considered too controversial and uncertain. The model has a 2-km grid squares spatial resolution, allowing for a detailed study of spatial pattern of ecosystem services. Six policy scenarios for 2060 were considered, varying by their degree of consideration for environmental and food safety in public policy: World Markets, Nature at Work, Go with the Flow, Green and Pleasant Land Stewardship, Local and National Security.

This prospective analysis shows that despite the economic trade-off between agricultural production and other ecosystem services, better balancing the bundle of ecosystem services in land-use choice always improve social welfare. The additional benefits in monetary terms, of increased ecosystem services generally exceed the associated loss of agricultural output in value. The intensity of this trade-off varies across UK regions, depending on agricultural systems and the characteristics of the natural environment. In certain cases, the benefits from greenhouse gas mitigations in agriculture can be already sufficient alone to outweigh the losses of agricultural production (Table 11). In any case, there is likely significant welfare loss from basing land-use pattern on the sole consideration of agricultural market drivers.[19]

Table 11. Changes in values across the United Kingdom from 2010 to 2060 achieved by targeting policy options according to three rules

Decision component	Maximise agricultural value only	Maximise all monetary values	Maximise all monetary values with biodiversity constraint
Market agricultural value	971	-448	-455
Nonmarket GHG emissions	-109	1517	1 510
Nonmarket recreation	2 550	1 3854	12 685
Nonmarket urban green space	-2 520	4683	4 352
All monetary values	892	19 606	18 092

Source: Bateman et al. (2013).

Another important conclusion from this prospective analysis is that there are significant gains from spatially-targeted policies addressing ecosystem services. Such potential gains can be explained by the heterogeneity of locations in terms of ecosystem services supply and demand. In the UK context, it has been estimated that spatially-targeted policies, taking into account the bundle of ecosystem services rather than marketable goods only, could increase by 20% on average the social value of land in the United Kingdom, which may likely outweigh higher costs of implementing more refined, spatially-targeted policy instruments.

The UK NEA also reviews the evidence base on major drivers in change of ecosystem services linked to agriculture. Usual overarching drivers (e.g. demography, climate change)

19. The spatial model has several assumptions, including: i) it is based on a linear assumption between land-use change and the provision of ecosystem services; and ii) it does not include technological innovation, such as improved resource-use efficiency or new farming practices.

are recognised, but lack of precise knowledge about the proper impact of agricultural market and policy drivers on agro-ecosystems still impedes concrete policy guidance. Market prices are recognised to influence production and land-use change, but proper effects are rarely quantified. There exists a certain consensus that decoupling has limited excessive input use; and that well-designed agri-environmental measures have had some successes in several cases, but overall have not been able to reverse negative pressures from agriculture on ecosystems in general. A lack of evidence base also concerns the costs and benefits of environmental regulations affecting agriculture, such as the Habitat Directive, the Nitrate Directive, or the Water Framework Directive.

Finally, on this basis, the UK NEA suggests to policy makers a number of possible policy responses to better manage ecosystem services linked to agriculture. To do so, in its conclusions the UK NEA study suggests policy objectives should be reframed on these issues: increasing food and energy production per hectare of agricultural land; increasing the resource use-efficiency of food production; improving the delivery of ecosystem services lined to agriculture; and investigating potential synergies between agricultural production and ecosystem services through innovative agronomic approaches.

According to the UK NEA, despite the aforementioned potential benefits, concrete policy improvement regarding ecosystem services linked to agriculture is not without technical and political difficulties: the trade-offs between agricultural production and the provision of ecosystem services may be an issue for convincing farmers to adopt practices in favour of ecosystem services; different ecosystem services are valued differently by different stakeholders and evolve over time, complicating the acceptance and tenacity to pursue such policies; scientific knowledge on ecosystem services are improving but remain insufficient to change the game in terms of refinement of design of agri-environmental measures.

In terms of ecosystem services flowing to agriculture, the UK NEA insists on the significant knowledge gaps that remain a barrier to policy action in this domain. Among key concerns is the loss of biodiversity, including pollinators, although some of the agri-environmental schemes deployed in the last decade succeeded to stop biodiversity erosion in the country. Knowledge gaps in this area are not, however, specific to the United Kingdom.

The UK NEA suggests that the policy responses should rely into a dynamic approach to the problem, filling progressively technical, implementation and political gaps and barriers. Indeed, even under the existing set of techniques and policy instruments, the UK NEA shows that there is already room for improving social welfare by better considering the entire bundle of ecosystem services linked to agriculture. In the medium-run, increasing scientific knowledge and improved policy design could further increase the cost-efficiency of policy intervention, reducing the *intensity* of the trade-off between agricultural production and other ecosystem services[20]. In the longer-run, biological and agronomic innovation could even transform trade-offs into synergies, at least in certain cases and locations, as already suggested by an increasing number of cases studies.

The UK NEA made direct contributions to the policy-making process at the national level. First, a bundle of decision tools was developed to frame the policy-making process regarding ecosystem services during the follow-on phase of the UK NEA. Second, a Natural Capital Committee, an independent advisory body to government composed of experts from various disciplines, was established in 2013. The Natural Capital Committee

20. The fact that agricultural production has continued to rise while environmental pressures from agriculture have decreased suggests that the intensity of the trade-offs has decreased.

produces annual reports to the Government, with recommendations regarding biodiversity and ecosystem policies, on which the government provides answers through a public report. The Natural Capital Committee has been re-conducted for the duration of the current Parliament, and will especially focus his present and future efforts in advising the government to develop a 25-year environment plan (Box 4).

The UK NEA also had some influence on the international level, both in methodological terms - some assessments in some countries are based on the British approach – and inpolicy terms, for example through the creation of a coalition of business international, in partnership with the TEEB. The tools developed for the UK NEA were also used to develop additional research projects looking at, for example, the interaction between agricultural production, ecosystem services, and adaptation to climate change.

Box 4. United Kingdom: Towards a 25-year strategy for the natural environment

"Following the recommendations of the Natural Capital Committee, the United Kingdom will develop a 25-year plan for a healthy natural economy. The Plan will:

- Place us as a world leader in using data, tools and techniques to understand, map and monitor the condition and value of our land, water, air, sea and wildlife, the benefits they give us and how they are changing
- Give people more opportunity to use, enjoy and engage with the natural environment
- Help individuals and organisations at local, regional, national and international levels to understand the economic, social and cultural value of nature, the impact that their actions have on it, and to use this knowledge to make better decisions and facilitate the design of sustainable financing models.
- Identify our most important and threatened environmental assets, prioritising where investment in them will deliver the greatest benefits and identifying how that investment can best be secured
- Focus policies on delivering better environmental outcomes
- Deliver on the range of natural capital related commitments that government has made, including: putting in place a new Blue Belt' to protect precious marine habitats; spending £3 billion from the Common Agricultural Policy to enhance England's countryside over the next five years; planting an additional 11 million trees; launch an ambitious programme of pocket parks; tackling air and water pollution; and ensuring the value of Green Belts and AONB's, National Parks, SSSI's and other environmental designations are appropriately protected;
- The 25-year plan will initially look to address outstanding monitoring and data issues, so we can make better informed decisions about where strategic investments in natural capital are needed and what form these should take.

This plan will be an opportunity to improve how we manage our natural environment, aiding the delivery of multiple benefits, resilience in the face of pressure and change (e.g. population growth, global geo-political dynamics, climate change, health issues) and unleashing new and innovative opportunities for investment in our natural assets. Many of the benefits we get from nature are felt most strongly at a local level. Local businesses, land owners, and people all shape their landscapes, and this influences what local people get from their environment. We will equip local decision makers with the tools they need to assess the benefits that come from their land and water assets so they can use them most effectively, be it to support livelihoods, their local communities, the economy, the provision of public goods, or personal enjoyment."

Source: Extract from the Department for Environment, Food and Rural Affairs.

The Chinese Ecosystem Assessment

China recently conducted the Chinese Ecosystem Assessment (CEA),[21] the first national-scale ecosystem assessment ever undertaken. The project involved a multidisciplinary team of more than 3 000 scientists, and was conducted between 2012 and 2014 under the auspices of the Chinese Ministry of Environmental Protection and the China Academy of Science, with the support of Ministry of Finance of China.

The CEA is an *ex post* assessment of the state, trends and drivers of change of ecosystem services over the period 2000 to 2010 at the national level . Seven ecosystem services were included in the CEA: food production; carbon sequestration; soil retention; sandstorm prevention; flood mitigation; and habitat provision for biodiversity. The supply and importance (i.e. number of beneficiaries) of ecosystem services were not only quantified but also spatially mapped by combining remote sensing data (satellite images) and ground surveys of ecosystems under various estimation processes, showing the high spatial variability of ecosystem services bundles across Chinese provinces.

The CEA found that the nationwide supply of ecosystem services trended upward between 2000 and 2010, but decreased for habitat provision for biodiversity. The increase is particularly important for food production (+38.5%), carbon sequestration (+23.4%), soil retention (12.9%), and flood mitigation services (12.7%).

Despite such improvements, the bundles, levels and trends of Chinese ecosystem services vary considerably across locations. Some regions witnessed ecosystem services degradation between 2000 and 2010, notably in Northern and Western China, while other regions experienced increases of several ecosystem services. Some ecosystem services are highly localised, such as flood mitigation (provinces of Anhui, Jiangxi and Jiangsu) or sandstorms prevention (Inner Mongolia); while others relate to larger spaces, such as carbon sequestration and water retention, which cover much of Southeast China.

In terms of explanatory factors, the CEA highlights the significant role of land use change, including agriculture, and land conservation policies in the evolution of the provision of ecosystem services between 200 and 2010. Statistical analysis undertaken in the study indicates that: i) increased cropland is associated with reduced supply of soil retention and sandstorm prevention services; ii) increased livestock density is associated with reduced soil retention and water retentions services; iii) the proportion of grassland and shrub areas have significant impact on carbon sequestration; iv) soil conservation and reforestation policies, i.e. the Sloping Land Conversion Programme (SLCP) and the Reforestation Programme, have a significant and positive influence on ecosystem services, especially carbon sequestration and soil retention services.

Although covering a large bundle of ecosystem services, the CEA does not address all types of agricultural impacts on ecosystem services, as well as specific agricultural market and policy drivers. Several key environmental pressures from agriculture were beyond the scope of the assessment, including water scarcity, nutrient and chemical pollution from agriculture, and ammonia emissions from livestock. Such pressures can be important sources of trade-offs between agricultural production and maintenance of the ecosystem. The significant increase in food provisioning services observed in China may have increased those pressures. In terms of drivers of change, the CEA essentially focused on land-use change and did not include the influence of farming practices on ecosystem

21. Under the project title “Survey and Assessment of National Ecosystem Changes Between 2000 and 2010”.

services. The relative roles of intensive-margin and extensive-margin environmental impacts of agriculture were not studied, thus limiting the understanding of policy leeway for cost-efficient approaches in the agricultural sector.

However, pressures from agriculture on ecosystems have been analysed in more depth in other complementary works. The annual state of the environment provides data on the state and trends of environmental quality; a rising number of academic works have quantified, and in some cases spatialised, ecosystem services and environmental pressures in terms, for example, of nutrient pollution.

Agricultural production growth is closely linked to intensification, i.e. greater use of input such as fertilisers and pesticides per hectare of agriculture, in link with agricultural policy and market drivers. Intensive input use is a significant source of pressures on ecosystems, especially on water systems. The role of agricultural policy and market drivers on environmental impacts of agriculture is likely significant in China, although reliable quantitative estimates of the proper effects of these impacts are still lacking. It nevertheless remains that in China, agricultural domestic prices are on average 23% above world prices; potentially most distorting forms of support represent 81% of producers support estimate; and the share of PSE in farmers receipt has increased strongly since the mid-1990s to attain 20% in 2013-2015.

The CEA comes at a time when China is placing more emphasis on environmental policies and the protection of ecosystems. In recent years, in conjunction with increasing efforts to monitor and assess the state and trends of ecosystems, several policy initiatives have been developed to curb the negative pressures from agriculture related to land-use change and agricultural practices.

Another important challenge for China is the maintenance of ecosystem services flowing to agriculture, the degradation of which could impede agricultural productivity growth. Loss of pollinators is a worldwide policy challenge which also affects China. Water and soil pollution, from both agriculture and other sectors is a significant threat to agricultural production. In some Chinese regions, soil contamination by heavy metals prevents farming activities, and soil remediation could be costly.

On-going and future national ecosystem assessments

Other initiatives in terms national ecosystem assessments have been, or are being conducted in OECD and non-OECD countries. The Intergovernmental science-policy Platform on Biodiversity and Ecosystem Services (IPBES) provides policy makers an updated database of ecosystem assessment conducted all over the world. These initiatives can be of diverse nature, from TEEB approaches to more specific reviews in terms of themes, ecosystems, and sector addressed. Table 12 provides selected examples of these recent and on-going assessments .

Table 12. Recent and on-going assessments on ecosystem services in selected countries

Country	Ecosystem assessments
Netherlands	A national TEEB study has been started in 2012. It is focused on regional, city, and overseas cases, trade flow impacts on ecosystem services abroad, business dependencies on ecosystem services and the processing of new national policy cases (TEEB 2011).
Germany	A series of TEEB thematic reports on biodiversity and climate, rural areas, green infrastructure in cities, and innovative instruments are being carried out in Germany between 2012 and 2015 (TEEB 2011).
Brazil	A national study and several state level studies are being carried out in Brazil. A TEEB for Business initiative is also underway aimed at engaging the Brazilian business community in the sustainable use of biodiversity (TEEB 2011).
France	The French National Assessment of Ecosystems and Ecosystem Services, which was launched in 2012 and is planned to be completed in 2016.
Association of Southeast Asian Nations	Between 2011 and 2012, countries forming the Association of Southeast Asian Nations conducted policy dialogues and training workshops to raise awareness and build capacity, as part of a project called "Disseminating the Values of Ecosystems and Biodiversity to Enhance Climate Change and Biodiversity Strategies in Southeast Asia".
Finland	National Assessment of the Economics of Ecosystem Services in Finland (TEEB Finland) (Jäppinen and Heliölä, 2015)

Source: Extracted from Wilson et al. (2014).

Country experiences in designing policies to address the agriculture-ecosystem nexus

Enhancing the implementation of environmental regulations and greening the Common Agricultural Policy

Since the 1990s, several regulations and policies in European Union have addressed the provision of several environmental services related to agriculture, although without explicitly referring to the concept of ecosystem services. This is notably the case of agri-environmental policies, but also of a series of EU environmental regulations that particularly affect the agriculture sector: the Birds and Habitats directives, the Nitrate Directive, or the Water Framework Directive in particular. Paralleling this development of environmental regulation and agri-environmental programmes, the decoupling of farm support in the European Union has progressively reduced production incentives and thus has also likely contributed to reduce pressures from agriculture on ecosystems, although payments that encourage commodity production trended upward in recent years, and "vary greatly across sectors and countries".

The EU Biodiversity Strategy to 2020 was adopted in 2011 to curb trends in biodiversity loss and improve ecosystems in the European Union, following the failure to attain 2010 policy objectives regarding biodiversity. Among the new features was the incorporation of ecosystem services in the policy framework, and the proposed action to develop a mapping of ecosystems and their services in EU Member Countries by 2014.

The resulting Mapping and Assessment of Ecosystem Services (MAES) provides evidence that agriculture remains a significant source of pressure on ecosystems and their services, without significant progress and even, in certain cases, worsening trends (Table 13). The 2015 Mid-Term Review of the EU Biodiversity Strategy to 2020 insists on the fact that Target 3a to "increase the contribution of agriculture to maintaining and enhancing biodiversity" has not seen any significant progress towards the target. Other targets also related to the agriculture sector, such as the full implementation of the Birds and Habitat

Directive (Target 1) and the maintenance and restoration of ecosystem services (Target 2), have made "progress but at an insufficient rate".

Table 13. The European Union Biodiversity Strategy to 2020: Targets, actions, and mid-term review of implementation

Targets	Actions	Implementation in 2015
T1: Enhance implementation of nature legislation	Action 1 Complete the establishment of the Natura 2000 Network and ensure good management Action 2 Ensure adequate financing of Natura 2000 sites Action 3 Increase stakeholder awareness and involvement and improve enforcement Action 4 Improve and streamline monitoring and reporting	Progress but at unsufficient rate
T2: Restore ecosystems, establish green infrastructure	Action 5 Improve knowledge of ecosystems and their services in the EU Action 6 Set priorities to restore and promote the use of green infrastructure Action 7 Ensure no net loss of biodiversity and ecosystem services	Progress but at unsufficient rate
T3: Sustainable agriculture and forestry	Action 8 Enhance direct payments for environmental public goods in the EU Common Agricultural Policy Action 9 Better target Rural Development to biodiversity conservation Action 10 Conserve Europe's agricultural genetic diversity Action 11 Encourage forest holders to protect and enhance forest biodiversity Action 12 Integrate biodiversity measures in forest management plans	No significant progress towards the target
T4: Sustainable fisheries	Action 13 Improve the management of fished stocks Action 14 Eliminate adverse impacts on fish stocks, species, habitats and ecosystems	Progress but at unsufficient rate
T5: Combat alien invasive species	Action 15 Strengthen the EU Plant and Animal Health Regimes Action 16 Establish a dedicated legislative instrument on Invasive Alien Species	Currently on track with implementation
T6: Contribute to averting global biodiversity loss	Action 17 Reduce indirect drivers of biodiversity loss Action 18 Mobilise additional resources for globalbiodiversity conservation Action 19 'Biodiversity-proof' EU development co-operation Action 20 Regulate access to genetic resources and the fair and equitable sharing of benefits arising from their use	Progress but at insufficient rate

Source: European Commission (2015).

The Common Agricultural Policy 2014-2020 introduced several new features targeting environmental benefits, In particular, the introduction of green direct payments, i.e. 30% of Pillar I payments would be conditional on the adoption of one of the three types of farming practices:

- *Ecological focus areas*: Converting 5% minimum of farmland to Ecological Focus Areas or equivalent practices
- *Maintenance of grassland*: ensure that, at the national level, grassland area is not reduced beyond 5% maximum
- *Crop diversification*: farms beyond 10 ha should produce at least two crops (for farms within 10-30 ha) or three crops (for farms beyond 30 ha)

The green direct payment builds on the idea that there exists a sufficiently robust and general link between the concerned farming practices and a bundle of environmental benefits (in terms of reducing pressure on ecosystems or providing ecosystem services).

This approach builds on the existence of robust synergies across ecosystem services, and each practice is expected to provide a certain basket of environmental benefits to varying degrees under the various natural and socio-economic conditions of Member States' agricultural sectors (Table 14).

Table 14. Potential links between farming practices and environmental benefits

	Natura 2000	Semi-permanent pasture	Permanent pasture	Crop diversity	Soil cover	Ecological set-aside	Organic maintenance
Biodiversity	X	?	X	X	X	X	X
Water quality	?	X	X	?	X	X	X
Water quantity	?	X	?	?	X	X	X
Soil functionality	?	X	X	X	X	X	X
Climate mitigation	?	?	X?	X	X	X	X
Climate adaptation	?	?	?	?	?	X	?
Landscape	X	X	X	?	0	X	?
Resilience to fire	?	X	X	?	0	0	?
Resilience to flooding	?	X	X	?	0	0	?

Note: X' refers to likely positive impacts; '0' to unlikely to have a beneficial impact; and '?' to impact unclear – will be dependent on nature of requirements.
Source: Hart, Kaley and David Baldock (2011).

More generally, the Common Agricultural Policy has been redesigned in terms of environmental instruments, with three distinct and complementary levels of policies that address requirements and targeting. These range from the broad to the specific: compulsory cross-compliance, affecting the largest area and without additional financial support; compulsory green direct payments, affecting a large area; and rural development measures under Pillar II.

In light of the present analysis on policies that address ecosystem services, it would be interesting to see how the new CAP could combine the three major approaches into a single framework: compulsory cross-compliance that aims to ensure good agricultural and environmental conditions; green payments that are expected to increase ecosystem services in synergy, with potentially[22] low implementation and transaction costs; and Pillar II mobilised for more targeted and demanding actions, with higher implementation and transaction costs, but also higher marginal benefits in terms of ecosystem services provision.

CAP 2014-2020 will be assessed in the coming years, including in terms of its environmental impacts. These environmental impacts are likely to depend on several factors, including: i) the practical implementation of the greening measures by Member-States; ii) the articulation of greening payments with Pillar II measures and its practical implementation; and iii) the economic conditions borne by farmers, particularly in terms of

22. It is too early to have accurate estimates of the implementation and transaction costs associated with greening payments. Nevertheless, the *ex ante* analysis by Norell and Soderberg (2012) suggests that these costs may not be so low as compared to the environmental benefits of greening.

agricultural commodity prices. A recently conducted simulation using the CAPRI model points to positive but relatively modest impacts for the environment. The follow-up and subsequent evaluations will help answer these questions in more depth.[23]

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