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Perspective Article

Narrowing the ecological yield gap to sustain crop yields with less inputs

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ABSTRACT

Sustainable production of sufficient and healthy food requires efficient use of agricultural inputs. In many regions of the world with intensive agriculture and relatively small yield gaps, this calls for a reduction of external inputs (fertilizers and pesticides) while maintaining yields. Ecological intensification, defined as the use of practices that enhance on-farm ecosystem services to reduce external input requirements, has been proposed as a strategy to help achieve this. However, the effects of ecological intensification are context- and input-dependent, creating uncertainty on its effectiveness and feasibility. Here, we introduce the concept of an 'ecological yield gap' to provide a common analytical framework to strengthen collaboration between agronomists and ecologists in assessing the contribution of ecosystem services within the wider array of inputs, management practices, technologies, and biophysical limits that determine on-farm crop yields. We define the ecological yield gap as the yield increase that could be achieved in a given context (climate x soil x cropping system), and at a given input level, by increasing the delivery of ecosystem services via ecological intensification practices that support crop growth and substitute external inputs. We provide empirical examples of such practices, including crop diversification, service crops, and organic amendments that can increase the use efficiency of mineral fertilizers and suppress pests, weeds and diseases. The potential of these practices to narrow the ecological yield gap and their feasibility at farm level depend on how the ecosystem services they provide interact with other aspects of the farming system and requires analysis at farm level. This perspective paper aims to facilitate a shared research agenda among agronomists and ecologists to develop complementarity between ecosystem services and inputs at field and farm levels.

1. Introduction

Increasing input-use efficiency of synthetic fertilizers and pesticides in cropping systems around the world is necessary for producing sufficient and healthy food in a sustainable manner (Tilman et al., 2011). Input-use efficiency refers to the ratio between the amount of output (yields, harvested products) produced per unit of input applied. Increasing input-use efficiency helps to mitigate the trade-offs between food production, resource use, and environmental impacts (Van Noordwijk and Brussaard, 2014). Where input use is currently low, increasing input use and input-use efficiency can improve food security while suppressing economic and environmental costs, by allowing more food to be produced on current agricultural land (Van Ittersum et al., 2016). In countries where input use is high, the primary aim of increasing input-use efficiency is instead to reduce input use without compromising yields, thus reducing adverse environmental impacts

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while sustaining food production (Cui et al., 2018; Silva et al., 2021a).

The goal to reduce input use while sustaining crop production is clearly expressed in the European Union's (EU) Farm to Fork Strategy (European Commission, 2020) and its recent vision on agriculture and food (European Commission, 2025). In the EU, input use is currently high and relatively stable with an average nitrogen (N) surplus of over 50 kg N ha^{-1}, and exceeding 100 kg N ha^{-1} in some countries, leading to substantial losses that pollute air and water (De Vries et al., 2021; Ludemann et al., 2024). Global pesticide use has increased by nearly 50 percent since the 1990s from 1.2 to 1.8 kg ha^{-1}. In Europe, the use of herbicides, fungicides, and insecticides when expressed in kg active ingredient per ha is relatively stable (totaling ca. 1.6 kg ha^{-1} in 2020; FAO, 2022), but there are indications that pesticide use intensity, i.e., the number of applications per crop season, has increased (Hossard et al., 2017; Kniss, 2017). Frequent pesticide use is linked to resistance development (Hicks et al., 2018) and biodiversity decline, which in turn can negatively affect associated ecosystem services (Bommarco et al., 2011), and create public health problems (Jepson et al., 2020; Rani et al., 2021).

European farms, even from fairly restricted areas in terms of climate and soils, differ widely in input use for a given crop yield, indicating substantial opportunity to increase input-use efficiency (Fig. 1). For example, in data sets of arable farms from the DEPHY network in northern France (Fig. 1a) and of crop farms in the Netherlands (Fig. 1b),

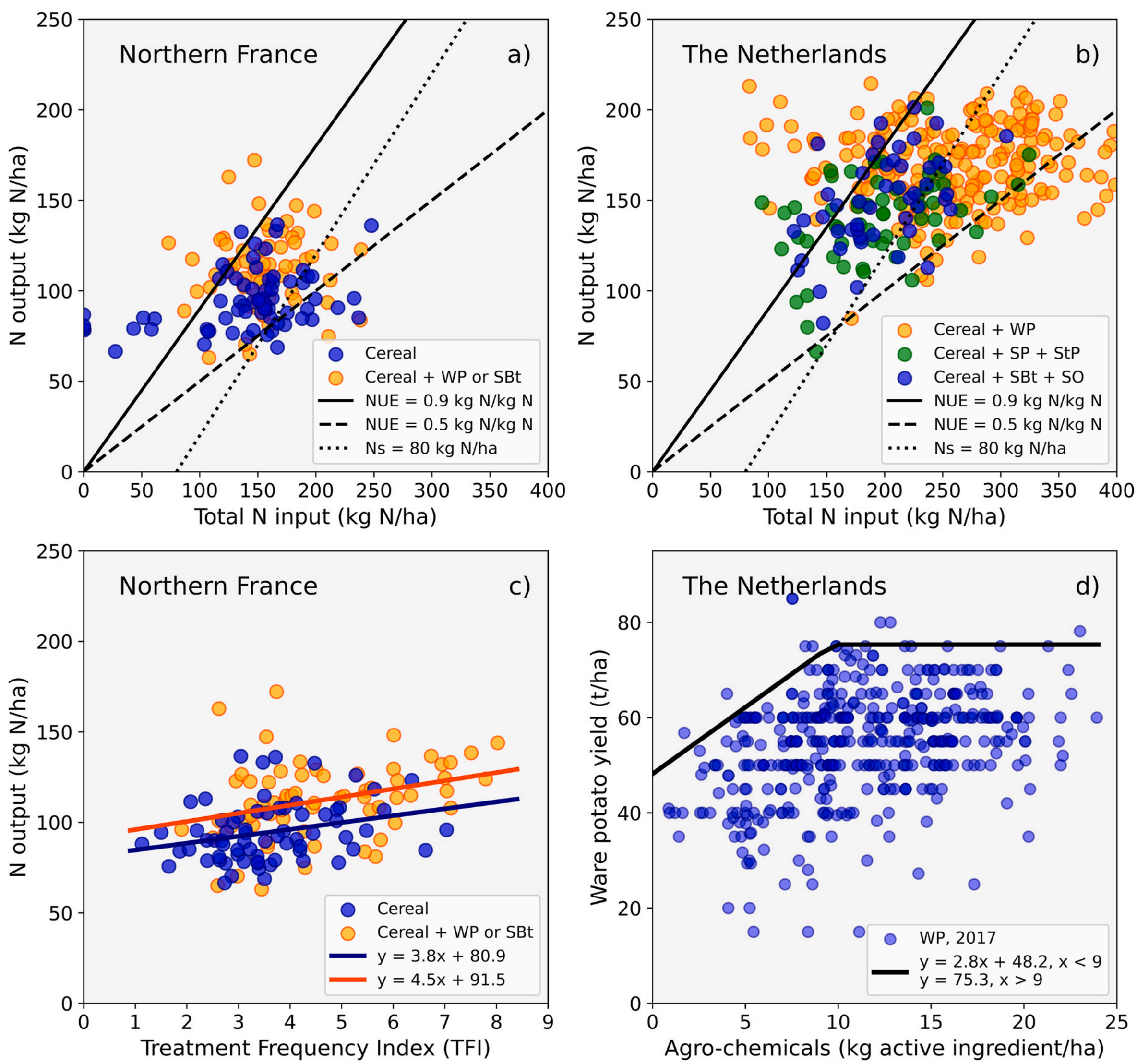


Fig. 1. Variability in input use and yield responses: examples from Northern France (left) and the Netherlands (right). In panels **(a)** and **(b),** N output versus total N input (i.e., including N deposition, mineral and organic N inputs from manure) following the approach of the EU Nitrogen Expert Panel (EUNEP, 2015) for different cropping systems on commercial farms in northern France (a) and in the Netherlands (b). Note that the legends do not show full crop rotations, rather crops that feature in the rotation. Dashed and solid lines in (a) and (b) mark the target range for NUE of 0.50–0.90 kg N kg/kg N respectively; the dotted line marks the N surplus (Ns) threshold value of 80 kg N/ha, all defined by the EU Nitrogen Expert Panel. In panel (a), each observation refers to the average of the last three years that a farm featured in the DEPHY network, with a distinction between farms that grew cereals only (blue) or cereals plus at least 10 % (by area) potato and/or sugar beet (orange). In panel (b), each observation refers to a farm x year combination (2015–2017); different colors of symbols refer to different key crops in rotation with cereals: ware potato (WP), seed potato (SP) and starch potato (StP), sugar beet (SBt) and spring onion (SO) in the Netherlands (figure reproduced from Silva et al., 2021b). In panels **(c)** and **(d)**, crop yield versus pesticide use is shown for the same farms as in (a) and (b), respectively. Panel (c) provides N output against pesticide use (expressed in Treatment Frequency Index – Brunet et al., 2008) for two types of crop rotations in northern France, which showed similar response to pesticide use. Panel (d) presents yields against pesticide use (expressed in kg active ingredients per ha) for ware potato (WP) in the year 2017 in the Netherlands (Guzman, 2018). The solid lines in (c) display linear regressions fitted to each cropping system (blue for the cereal rotation: R^2: 0.078, P-value: 0.019; orange for the cereal + WB or SBt rotation: R^2: 0.10, P-value: 0.0078), and the solid line in (d) depicts a linear-plateau boundary function fitted to the 95th quantile of the data.

there is a weak relationship between the amount of N fertilizer inputs and the amount of N harvested in crop yield (a proxy for both yield quantity and quality). Nitrogen use efficiency (N output/N input) in some farms is well outside the 0.5 and 0.9 kg kg^{-1} range defined as desirable by the EU Nitrogen Expert Panel (EUNEP, 2015), while N surplus (N input – N output) isoften above the proposed upper threshold of 80 kg N ha^{-1}. Similarly, variation in the use of pesticides at a given yield was high. In the DEPHY network, there was a weak relationship between Treatment Frequency Index – an indicator of farm's reliance on pesticides - and crop yields (Fig. 1c). In the Netherlands, the relationship between pesticide use and potato yield was mildly positive at low rates and flat at higher application rates (Fig. 1d).

Assuming no major effects of variation in climate and soil conditions (Silva et al., 2021b), the range of nutrient and pesticide amounts used to achieve a given output in Fig. 1 suggests that many farms could reduce nutrient inputs by at least 20 % and pesticide use by 50 % with little if any yield loss. Similar conclusions were reached in other studies (Scarlato et al., 2022; Ravensbergen et al., 2024). However, to achieve such reductions, we need to understand which factors explain the enormous horizontal and vertical variation in the input-output relationships and where and how external inputs can be reduced, substituted or used more efficiently.

One promising avenue to explore in pursuit of increased input-use efficiency is the role of ecosystem services in supporting crop yields. Practices enhancing on-farm ecosystem services, collectively termed ecological intensification (Cassman, 1999; Bommarco et al., 2013; Jacquet et al., 2022; Deguine et al., 2023), can increase crop yields and/or input-use efficiency (Rusch et al., 2016; Dainese et al., 2019; Tamburini et al., 2020; MacLaren et al., 2022). In other words, there is potential for ecological intensification to improve the input-output relationships such as those in Fig. 1. Indeed, some of the farms with more efficient input use in Fig. 1 may already be employing some degree of ecological intensification. To quantify its potential, the effects of ecological intensification need to be disentangled from other factors influencing resource-use efficiency, such as environmental conditions and other management aspects. Ecological intensification practices include crop rotation, intercropping, using available organic nutrient sources topped up with synthetic nutrients, reducing soil tillage intensity, prolonging soil cover throughout the year, synchronizing resources to the crop's needs via biotic regulation, and integrating habitat and plant diversity into farmed landscapes to support service providers such as pollinators and predators (e.g. Smith, 2015; Fontaine et al., 2024). By enhancing and relying on internal processes within agroecosystems, ecological intensification practices can make cropping systems less input-dependent, more autonomous, and more resilient in the face of pests and climatic variation (Bommarco et al., 2013; Tittonell, 2014).

Conceptual knowledge that ecological intensification *can* work does not easily translate into specific, applicable recommendations for actions that will reliably improve input-use efficiency for a specific farm or region (Kleijn et al., 2019). A complication is that the success of ecological intensification practices can be context-specific, and practices do not always work as intended (Karp et al., 2018). Faced with this uncertainty, farmers, advisors, and policymakers may understandably be reluctant to adopt or promote ecological intensification strategies, especially where they require substantial expenditure and/or farm or landscape redesign. A necessary next step in ecological intensification research is therefore a deeper investigation into how yield responses to underlying practices depend on the agronomic context, as a basis for more reliable recommendations, not only at crop level, but also at whole farm and even landscape level. For practices to be feasible and economically beneficial at farm level, resource constraints in relation to land, labour and capital must also be considered. In this paper, we argue that advances can be made by paying more attention to the agronomic functions of ecological intensification practices, i.e., how they affect the balance of resources and stresses that promote or limit crop growth. In particular, the functions of ecological intensification practices need to be considered in the context of other factors determining crop growth, such as input use, farm management, and climatic conditions and soils. A common analytical framework facilitating communication and collaboration between agronomists and ecologists would help to achieve this integration.

In this perspective paper, we propose to make use of the yield gap concept in agronomy (Silva et al., 2017; van Ittersum et al., 2013) and adapt that to capture the contribution of ecosystem services to crop input-output relationships, i.e., we delineate an ecological yield gap. Although the concept is valid and of interest to crop production systems across the globe, our perspective mostly draws on examples from relatively high-input systems in which the need to reduce external inputs and improve environmental performance is greatest. We first introduce the concept and show how it can be represented in production functions (Section 2), and then provide several empirical examples of ecological intensification practices and their influence on ecological yield gaps (Section 3). In Section 4 we indicate how research at process, crop and farm level by agronomists *and* ecologists can quantify ecological yield gaps.

2. The ecological yield gap

The yield gap is defined as the difference between the theoretical potential yield under perfect management conditions and the actual farm yield (van Ittersum et al., 2013). The theoretical potential yield refers to the maximum yield of an adapted crop variety that is possible to achieve in a given environment by avoiding yield limitations due to water (if irrigation is possible) and nutrients, as well as yield reductions due to weeds, pests and diseases (van Ittersum et al., 2013). The yield gap in a given farming system can be decomposed into efficiency, resource, and technology yield gaps to better understand and address the causes affecting yield variability in farmers' fields (Fig. 2a, Silva et al., 2017). The efficiency yield gap reflects the increase in yield that can be achieved, at a given resource supply and in a given context, by using more efficient practices, such as precision fertilizer application adjusted to the targeted yield level. The resource yield gap is the yield increase that could be achieved in the given context by providing more resources to the crop, for instance nutrients. Finally, the technology yield gap indicates the not-yet-realized yield difference between the potential yield and the yield achieved by highest yielding farms, hence reflecting the incapacity of current on-farm technologies to reach agronomic best practices. It is important to note that achieving the potential yield at farm level is generally not desirable from either an economic or an environmental point of view. Even intensive and highly productive farms often have some degree of yield gap, and yield gap closure is never an aim in itself. Yet, to assess whether and to what extent narrowing yield gaps is feasible, it is critical to understand the agronomic causes of the yield gaps in the farming context. On-farm practices that can contribute to the yield gap include sub-optimal timing of planting and management (Silva et al., 2022), lack of effective weed or pest management practices, or input levels used in highest-yielding fields being below the levels required to reach potential yield (e.g., van Dijk et al., 2020).

The three components of the yield gap can refer also to a lack of adoption of practices that could be considered part of ecological intensification, such as the use of organic resources, legume crops or sufficiently diverse crop rotations (e.g. Silva et al., 2017). However, the role of ecological intensification practices is not singled out, hiding their potential. Here, we introduce an 'ecological yield gap' that explicitly recognizes the role of ecosystem services in contributing to efficiency, resource, and technology yield gaps (Fig. 2a). We define the ecological yield gap as the yield increase that could be achieved in a given context (climate x soil x cropping system) and at a given input level by increasing the delivery of ecosystem services that support crop growth, such as nutrient cycling, weed control, and pest and disease regulation. Note that the ecological yield gap overlaps with the efficiency, resource

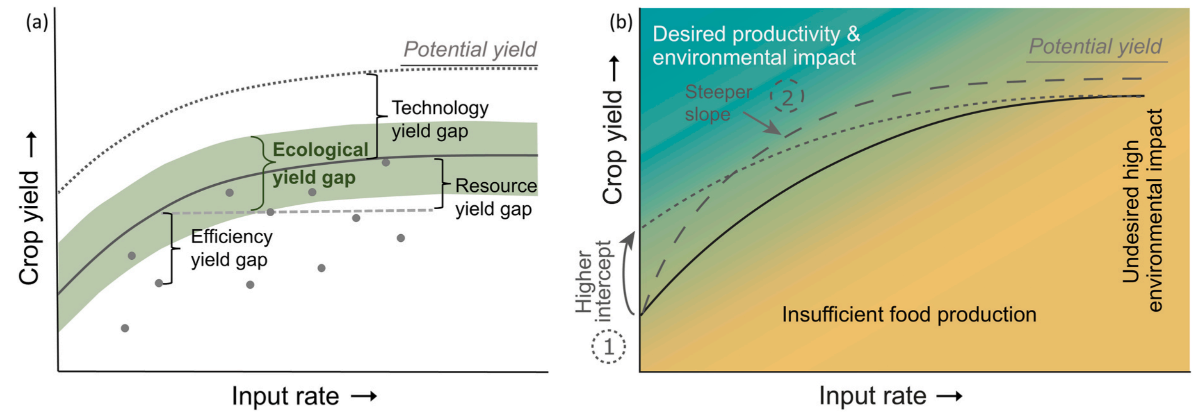


Fig. 2. **(a)** The yield gap concept and the decomposition of the yield gap into efficiency, resource, and technology yield gaps, following Silva et al. (2017), now completed with the ecological yield gap (green shading). Points represent actual on-farm data from fields with different use of yield-increasing inputs, i.e., nutrients or water (x-axis) and differences in farm practices affecting their crop yield. The maximum yields achieved by the most efficient fields (solid line) would be expected to increase further if they adopt new technologies (technology yield gap; dotted line). The ecological yield gap can overlap with the technology, resource, and efficiency yield gaps (green). Note that the x-axis provides the input rates (not accounting for e.g. soil nutrient resources). **(b)** Different changes in the response curve with ecological intensification practices point to different mechanisms: (1) an increase in intercept (i.e., higher yield at zero input level), but the same yield plateau, pointing at a substitutive effect of the practice; (2) an increase in slope and a higher yield plateau, pointing at a complementary effect of the practice, e.g., reduced biotic stress. The yellow shading in (b) indicates undesirable situations from either a food availability or a sustainability perspective, while the green area indicates desirable situations, i.e., increasing the amount of food produced with less external inputs. Note, that potential yields are the same in both panels.

and technology yield gaps, i.e., enhanced ecosystem services may increase efficiency, increase resource availability and/or reduce technology gaps. For example, using legumes in a crop rotation can deliver the ecosystem service of N provision, contributing to closing the resource yield gap, while the crop rotation itself could be considered a technology explaining part of the technology yield gap in farms not using it. It could be argued that many ecological intensification practices are part of 'good agronomy' or 'good agricultural practice'. We do not dispute this, rather, we see value in explicitly delineating the contribution of ecosystem services to crop yields to enhance collaboration between ecologists and agronomists around a *common analytical framework*. Recognizing an ecological yield gap would enable us to quantify the specific contributions of different ecological intensification practices to crop yields across different contexts, such as in the presence or absence of other inputs and technologies, or in relation to different potential yields. This would improve our ability to successfully integrate ecological intensification into varied farming systems, although the latter requires explicit upscaling of crop effects in time and space to the farming systems level (Sections 3 and 4).

The term ecological yield gap has been used before (Bonilla-Cedrez et al., 2021; Vanlauwe et al., 2023), but with the adjective 'ecological' used to contrast with the 'economic' yield gap. In that context, the ecological yield gap does not refer specifically to (agro-)ecological measures, but to the full set of agronomic practices explaining why farmers' yields are lower than the potential yield. Hence, in previous usages of the term, the ecological yield gap is equivalent to what is traditionally, and also in this paper, called yield gap (Van Ittersum and Rabbinge, 1997; Lobell et al., 2009; Van Ittersum et al., 2013). We therefore propose to use the term ecological yield gap to single out the contributions that ecosystem services can bring to narrowing the yield gap at given input levels.

To explore the size of the ecological yield gap and its dependence on the input, we can employ input-yield production functions describing how crop yields respond to an input depending on levels of other growth factors, including limiting resources (light, water, nutrients) as well as abiotic (temperature, pH) and biotic stresses (weeds, pests and diseases) (De Wit, 1992; Van Grinsven et al., 2022; Smith et al., 2023). Note, that the yield increasing input along the x-axis could also refer to water, although we do not elaborate on water in the present paper. If the goal is to reduce input use whilst sustaining yields, it is desirable that ecological intensification increases the yield achievable in the absence of external inputs or makes it possible to attain higher yields at lower inputs, with a higher maximum yield, or some combinations of these changes. We depict the two extreme cases in Fig. 2b: 1) increasing the intercept of the input production function, i.e., the yield in the absence of external input; 2) an increase in slope, i.e., an increase of the marginal use efficiency, at a given input and, potentially, also a shift in the maximum yield, i.e., of the asymptote. Intermediate cases are possible, i.e., a steeper slope but no or little change in yield plateau, and a larger intercept and steeper slope with a change in yield plateau.

Under the mechanistic point of view, to increase the intercept, ecological intensification practices would need to provide the same function as the input, for example substituting N fertilizer with manure or biologically fixed N. A pure substitutive effect would shift the curve to the left in the input-output space, maintaining the slope of the curve for a given output (Fig. 2b). To increase the slope of the input-output relationship at a given input, ecological intensification would need to provide complementary functions to the input that improve additional growth factors. It has long been established (De Wit, 1992; Supplementary Data), that resource-use efficiency is highest when crop growth factors are balanced. Growth factors include resources (light, water, nutrients) as well as no reductions from biotic stresses. The Laws of the Minimum (Liebig), Optimum (Liebscher) or Constant Activity (Mitscherlich) (Supplementary Data) differ in their predictions regarding the strength of the interactions, with evidence so far pointing toward either (or both) Liebscher or Mitscherlich being correct when properly accounting for synergy between different growth factors and inputs (De Wit, 1992; Van Grinsven et al., 2022). However, the general principle that balanced growth factors result in greater resource-use efficiency is undisputed (Nijland et al., 2008).

Whether substitution or complementarity dominates, i.e., the shape of the ecological yield gap, is practice-specific. In Fig. 2a, we avoid suggesting that ecological yield gaps are more significant at either relatively low or high input levels, because it is unclear how interactions between inputs and practices would play out without being specific about the practice and input. On the one hand, it could be argued that high input levels often have the effect of suppressing ecosystem services, for example high mineral fertilizer input tends to limit microbial nutrient cycling (Grandy et al., 2022), N fertilizer reduces the contribution of biologically fixed N to subsequent cereals (Nilsson et al., 2023), and high pesticide use tends to suppress predators as well as pests (Geiger et al., 2010). This favors a hypothesis that ecological practices have a substitutive effect and are less effective at high resource input levels but could be useful to increase yields at low input levels. On the other hand, gains can be made by using ecological intensification practices with complementary effects at both low and high input levels, and the yield plateau can increase as a result of the practice. This is likely the case when ecological practices such as more diverse rotations or organic fertilizers enable the applied inputs to be used more efficiently because of better suppression of weeds, pests and diseases (rotations) or addition of other macro- or micro-nutrients and organic matter (from organic fertilizers) (MacLaren et al., 2022).

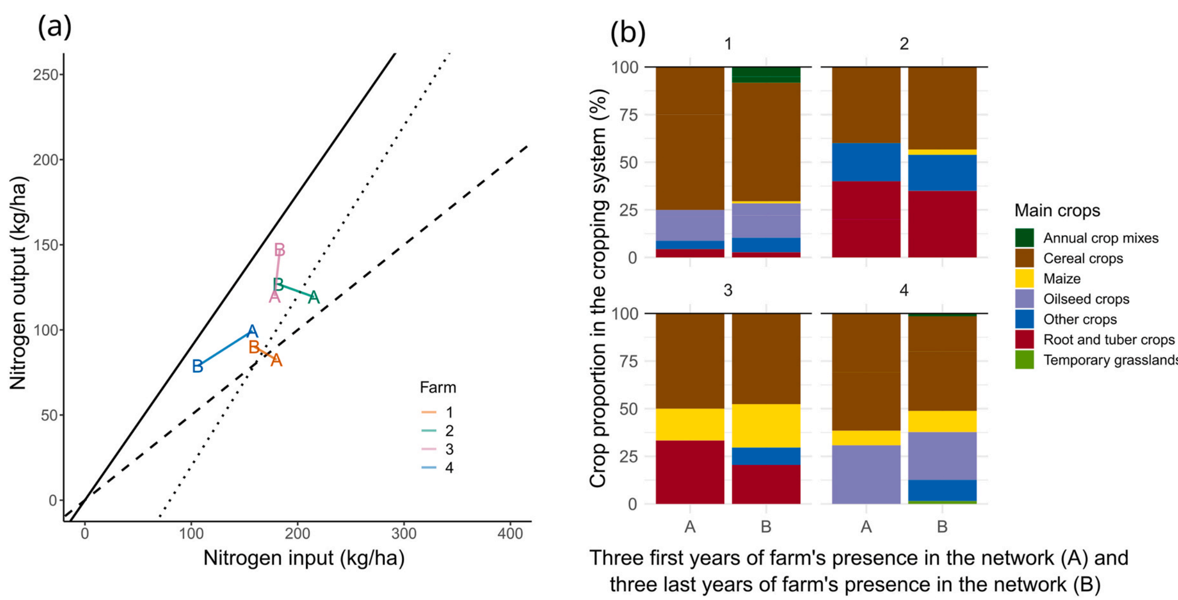
Using examples of crop diversification and organic amendments from experiments and on-farm observations, the next section explores how different ecological intensification practices might affect different growth factors, thus contributing to closing the ecological yield gap by increasing the intercept and/or the slope of input-yield production functions.

3. Examples of ecological yield gaps

3.1. Analyzing farm level developments

Fig. 1 revealed a large horizontal and vertical variation in input-output relationships of farms operating in similar soil-climate conditions. Investigating the reasons underlining such variations across farms and their contrasted trajectories of change over time can offer relevant initial insights and lead to hypotheses of what ecological intensification practices can mean in a real farming context. From the many farms of the French DEPHY network in Fig. 1a, we chose four examples and depicted their evolution in time (Fig. 3), i.e., the average N input and N output, cropping pattern, pesticide Treatment Frequency Index and economic gross margin of the three first and three last years of presence in the network during the 2010–2021 period. These four farms illustrate the diversity of changes in practices and performances associated with an improvement of their N output-input ratios. Farms 1 and 2 moved towards less N input and higher N output, Farm 3 moved to higher N output at similar N input, while farm 4 moved to less N input and less N output. All four farms show increased diversity in cropping pattern, lower reliance on pesticides, and improved gross margin. Over time, these farms optimized their crop management (better targeting of inputs to output levels), used ecological intensification practices related to diversification such as increasing the crop rotation length, adding functionally diverse crops and introducing more robust crops, and implemented other practices (e.g. growing other cultivars, using delayed sowing) (Nandillon et al., 2024).

The aim of this example is to demonstrate the potential value of on-farm data for the assessment of ecological intensification. Indeed, all four farms diversified their cropping systems and exhibited improved environmental and economic indicators. Revealing causality between e.g. diversification and performance is challenging in these four farms because of simultaneous other changes in the management that cannot be accounted for here. However, in a previous study by Nandillon et al. (2024), statistical analysis of a large number of farms from the DEPHY network and their changes in time allowed to test the hypothesis of causal links between diversification and for instance an improved N input-output relationship. Based on this, we can formulate hypotheses of successful ecological intensification (as well as optimization and of other practices) to move towards more favourable input-output relationships. Such hypotheses can then be tested in targeted experimental work of which we provide examples below.



(c)

Farm	Treatment Frequency Index (TFI) A	Treatment Frequency Index (TFI) B	Gross margin A (€/ha)	Gross margin B (€/ha)
1	6.5	2.5	-73	339
2	6.8	5.3	1439	1897
3	4.3	3.5	519	637
4	5.5	2.4	186	309

Fig. 3. Four example farms from the DEPHY farm network in Northern France (Fig. 1), showing how these evolved (from A to B) in terms of **(a)** N input-output performance, **(b)** diversification, **(c)** reliance on pesticide use (measured with the pesticide Treatment Frequency Index) and economic gross margin (excluding labour costs). A and B refer to three-year averages, respectively, when entering the network and the final years of presence in the network.

3.2. Experimental crop rotation diversification research

The existence of positive effects of crop rotations on crop yield and resource-use efficiency is a well-established fact in agronomy and ecology (see Bennett et al., 2012). For example, rotations with a high frequency of a single crop or a group of crops sensitive to the same soil-borne diseases increase the prevalence of soil-borne pathogens (Bollen et al., 1989; Scholte, 1992; Abawi and Widmer, 2000; Jalli et al., 2021). A high frequency of a crop or related crops in a rotation can also affect the composition and size of the weed seed bank (Doucet et al., 1999). Well-designed crop rotations can therefore reduce weed, pest and disease pressure, and limit the need to use chemical and other means of crop protection (Storkey et al., 2019). Inclusion of leys or legume crops in the rotation often lead to higher yields of indicator crops at lower N input rates. An illustration of this is provided in Fig. 4 using 48 years of data from a long-term experiment at three sites in Sweden (El Khosht et al., 2025). Here, the effect on winter wheat and spring oat yields is shown when including a two-year grass or mixed grass-legume ley compared with arable crops only for six-year crop rotations at four N input rates. Yield benefits of the leys, which were greater with inclusion of mixed grass-legume ley compared to grass only, disappeared at high N rates. The effects on wheat and oat yields grown the second and third year after terminating the ley were illustrative for other crops in the rotation (El Khosht et al., 2025). While yield benefits on indicator crops when including legumes in the rotation at relatively low N rates are well known, their economic benefits at farm level strongly depend on the context (Reckling et al., 2016; Van Loon et al., 2023). In conditions similar to the Swedish example (Fig. 4) it was estimated that inclusion of leys was beneficial for the farm economy at relatively low feedstock prices for anaerobic digestion or livestock (Tidåker et al., 2016), because leys had positive effects on yields or N input saving in all crops of the rotation, and the ley substituted the least profitable crops, such as second year wheat or spring barley. Leys also reduce environmental impacts (N_2O and CO_2 emissions, NO_3 leaching and erosion) (Nilsson et al., 2023) and may reduce the risk of herbicide resistance (Hicks et al., 2018).

Continental and global datasets from long-term experiments confirm positive effects of diversifying the crop rotation on indicator crops (Marini et al., 2020; Bowles et al., 2020; MacLaren et al., 2022; Smith et al., 2023), as well as on total macronutrient production from the whole rotation (Costa, 2024). The benefit of crop diversification tended to be higher at low N input rates, in both mineral and organic forms, particularly where diversification included legumes and/or leys, although effects did not disappear at higher N input rates (MacLaren et al., 2022; Smith et al., 2023). These results imply an ecological yield gap that is larger at low than higher input levels, likely because the ecosystem services provided by diverse crop rotations have the same or a similar function to the inputs, i.e., there is at least a partial substitution effect. Indeed, the benefit of including legumes in a crop rotation is greater if the crops' N needs are not already met through N fertilizer (e.g., in organic farming, or where inputs are prohibitively expensive; cf. Fig. 3). But there are likely also complementary effects, as evidenced by the benefits remaining at high N inputs, possibly due to improved soil fertility and improved suppression of weeds, pests and diseases, in the more diverse rotations.

The contributions of the ecosystem services provided by crop diversification differ depending on which crops are included in a cropping system. Ideally, diversifying should use crop species that perform multiple functions and allow multiple inputs to be replaced simultaneously. For example, winter oilseed rape *(Brassica napus*) is an input-intensive crop grown on ca. 9 million hectares in Europe. In France the mean mineral N fertilizer application is 170 kg ha^{-1} and the pesticide Treatment Frequency Index is 5.6 (Verret et al., 2017). One option for input reduction in northwestern Europe is to intercrop oilseed rape with service crops that are grown for other purposes than directly providing a harvestable yield, such as annual legumes that freeze and desiccate over winter (Fig. 5). Such a cropping system can reduce N fertilizer application by 30–40 kg ha^{-1} and herbicide and insecticide use while maintaining yield (Verret et al., 2017).

The weed control effect in this example is due to service crop - weed competition, which likely continues in spring due to surface cover of the desiccating service crop (Verret et al., 2017; Ouattara et al., 2023). While annual non-legume service crops provide greater weed biomass reduction (−52 %) than legumes (−38 %), legumes compete less with oilseed rape (Verret et al., 2017) and provide biologically fixed N to the oilseed rape (Lorin et al., 2016). The reduction in N fertilizer use (in Fig. 5 ca. 40 kg N ha^{-1}) is possible due to mineralization in spring of N fixed by the service crop and likely additional mechanisms such as improved oilseed rape root exploitation of N with intercropping, increasing N use efficiency (Cadoux et al., 2015; Lorin et al., 2016). The pest control effect established thus far is mainly on cabbage stem flea beetle (*Psylliodes chrysocephala*), which is the most important insect pest of rapeseed in Europe (Emery et al., 2021). The service crop could also interfere with host plant location and decrease host crop attractiveness (Seimandi-Corda et al., 2023).

There are, however, challenges with this approach, including that the service crops do not fully substitute the need for herbicides, their effects are inconsistent over space and time, and their presence limits the possibilities of herbicide use in autumn. Similarly, soils that are rich in N, e.g., due to high N fertilizer application rates over time, could limit

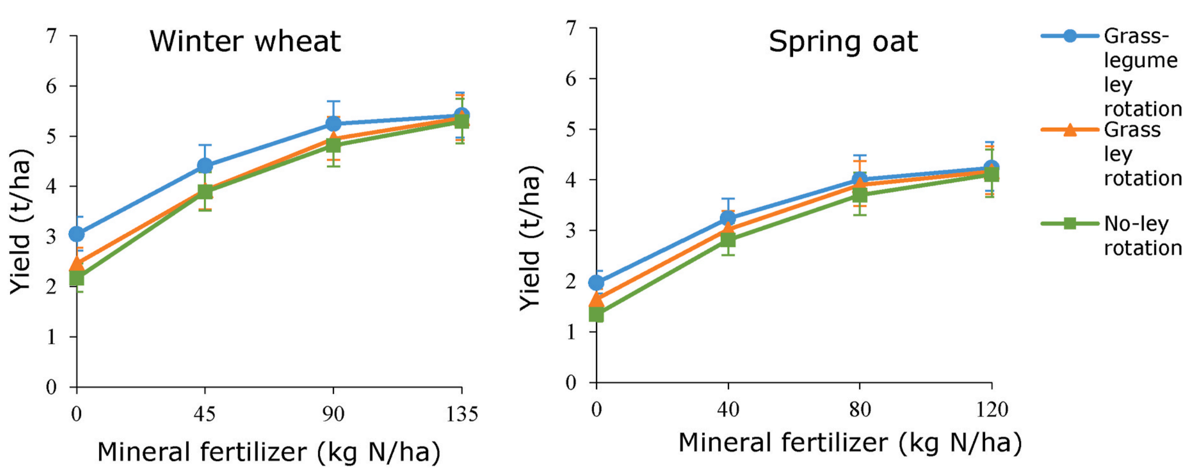


Fig. 4. Effect of N rate on dry matter yield of the winter wheat and spring oat indicator crops in three rotations of 6 years, without or with two-year grass or grass-legume leys. Data are averages from 48 years of a long-term experiment at three sites in Sweden. Figure redrawn from El Khosht et al. (2025).

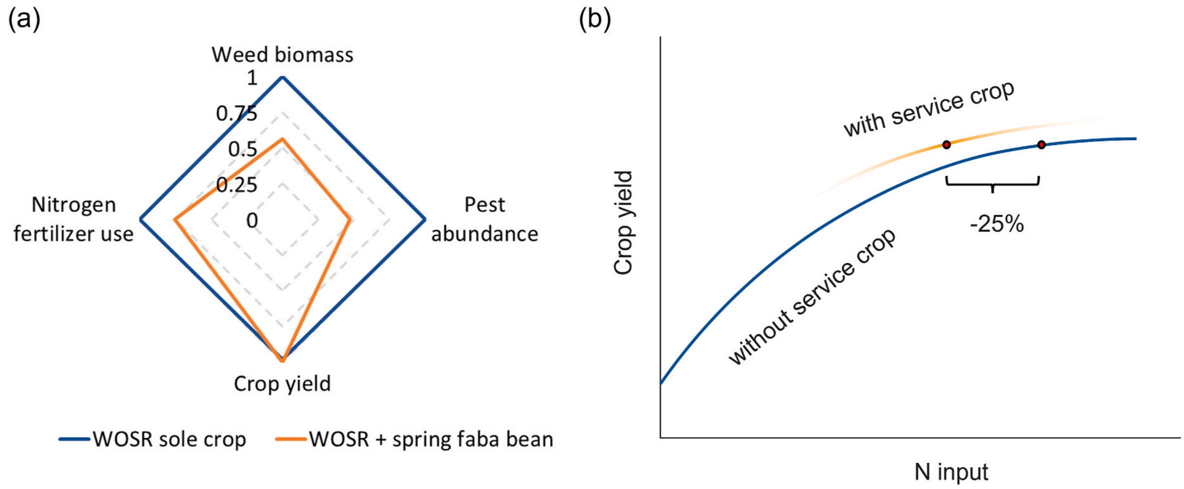


Fig. 5. **(a)** Intercropping winter oilseed rape (WOSR) with spring faba bean as a service crop in a field experiment without any pesticide use led to circa 50 % lower weed biomass and cabbage stem flea beetle abundance compared to the sole crop. Intercropping brought the pest abundance under the control threshold, reduced fertilizer use by 25 % and maintained yield (Emery et al., 2021). **(b)** shows two hypothetical production functions for the two cropping systems, illustrating that including a service crop could lower input use by 25 % while maintaining yield. The production function after implementing ecological intensification in the form of a service crop has been left fuzzy, reflecting that its intercept, shape and asymptote relative to the production function without a service crop is currently unknown.

the N fixation of the legumes and thereby their service delivery. This highlights that the success of ecological intensification practices depends upon their interactions with the local environment and cropping system, and these should be considered when promoting ecological intensification practices to sustainable agriculture.

3.3. Experimental research on organic amendments and mineral fertilizer N use efficiency

In this example, we explore the substitutive and complementary effects provided by organic amendments that could increase mineral fertilizer N use efficiency. Distinct functions and complementary effects are often assumed, given that N from organic amendments is released more slowly, over several years, while most of the N from mineral fertilizer becomes available during the year of application (Schröder, 2005). Organic amendments also contribute to soil organic matter, as well as many macro- and micro-nutrients, so organic amendments are thought to support long-term soil fertility while mineral N fertilizer supplies short-term N requirements (Palm et al., 2001). If this is true, the agronomic N use efficiency of mineral N fertilizer (additional kg yield per kg mineral N applied) should be improved by a prolonged use of organic amendments.

Evidence so far has been mixed. Vanlauwe et al. (2011) found a positive effect of organic amendments on the agronomic N use efficiency of mineral N fertilizer while Oelofse et al. (2015) and Schjønning et al. (2018) found no or even negative effects. One reason for diverging observations could be the difference in climate and/or soil organic matter contents between these studies, with potentially more benefits in tropical than temperate climate zones and soils, the former often being more depleted in organic matter and nutrients. Furthermore, some methods may conflate the substitutive effect of N supply with complementary effects arising from the distinct functions of organic amendments and mineral N fertilizer. Since the effect of organic amendment is considered to be on long-term soil fertility, its complementarity with synthetic input may become apparent only in the long run.

Three methods were compared here to assess the influence of organic amendments, such as manure, compost, and plant cuttings, on the agronomic N use efficiency of mineral fertilizer (i.e. the efficiency of applied N to increase yield). Assessing agronomic N use efficiency often relies on experimental set-ups with at least four treatments: one control plot with no N-fertilizer applied (only P and K added), one treatment with only mineral fertilizer applied, one treatment with only organic amendment applied, and one treatment with both mineral fertilizer N application and organic amendment applied. Ideally, these treatments are balanced for the input of available N. For a given mineral fertilizer N application, the yield increase due to that application is then assessed for the treatments with organic amendment application and compared to the yield increase in the treatments without organic amendments (method 1; Fig. 6a). However, in the treatments with organic amendments, the *total* soil N supply (and thus the crop yields) will be larger, and thus the use efficiency of the mineral fertilizer N might be lower due to diminishing returns at higher total N availability. In this case, the substitutive effect reduces the *apparent* benefit of organic amendments, obscuring any complementary effects. This could explain why no positive effects of organic amendments on mineral fertilizer N use efficiency are concluded (e.g. Oelofse et al., 2015; Schjønning et al., 2018).

A fairer assessment might be to assess the difference in agronomic N use efficiency at equal yields (method 2; Fig. 6c). However, yields with organic amendments will still be larger in the control plot, creating a bias towards the flatter part of the response curve to mineral fertilizer. Finally, one can also assess a difference in slope of the response curves, i.e., the marginal agronomic N use efficiency, at equal yields (method 3; Fig. 6e). This will most likely give the fairest comparison, but requires a more extensive experimental set-up with multiple N treatments, with and without organic amendments, to enable a fit of the entire input-output curves, as opposed to simply comparing pairs of input levels.

In a meta-analysis, data from 20 long-term experiments in Europe were collected (Hijbeek et al., 2017) and used to fit yield response curves following a modified asymptotic exponential equation (George, 1984). This allows the quantification of the effect of organic amendments on agronomic N use efficiency of mineral fertilizer using all three methods mentioned (Fig. 6b, d, f). If present, a positive effect of organic amendments on the agronomic use efficiency of mineral N fertilizer is most likely to be revealed using the third method comparing the slopes of response curves at equal yield. Indeed, the third method was the sole one showing a significant positive effect, but only for potato, i.e., organic amendments were found to increase the marginal agronomic N use efficiency of mineral fertilizer on average by 30 % ($P = 0.01$), while there was no effect observed for winter wheat suggesting a pure substitutive effect in this crop (Fig. 6f).

That organic amendments increase N use efficiency for potato, but not for wheat, suggests that the ecosystem services enhanced by organic amendments are more critical to potato than to wheat cultivation. Though we cannot rule out an effect of other macro- and micro-nutrients

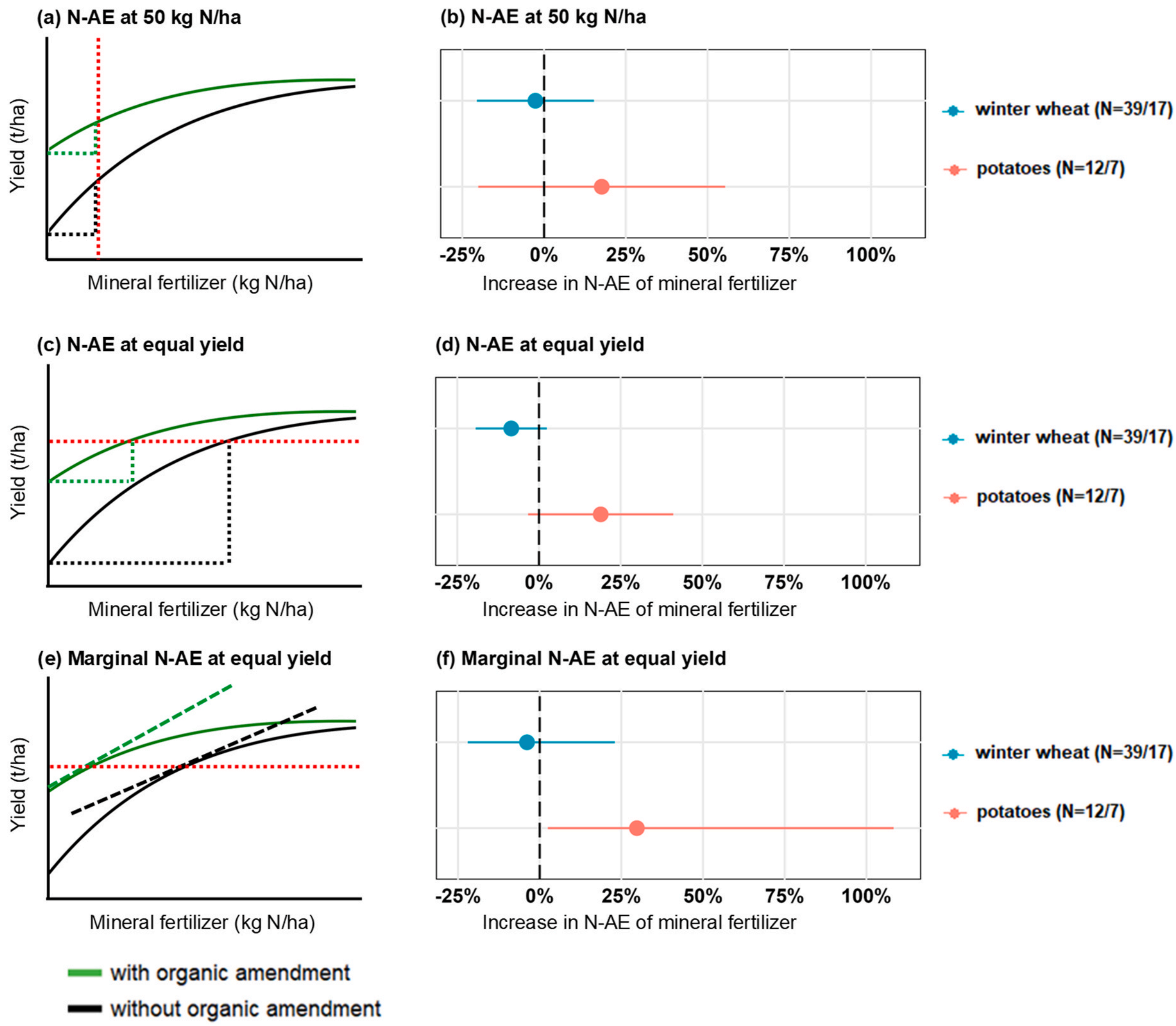


Fig. 6. Assessment and influence of organic amendments on the agronomic N use efficiency (N-AE) of mineral fertilizer. Panels a, c and e show the calculation methods 1, 2 and 3 used while panels b, d and f show the relative difference in agronomic N use efficiency of mineral fertilizer when an organic amendment is added. Because the focus is on the use efficiency of mineral fertilizer N, this is the input reported on the x-axis. **(a and b)** N-AE is determined at a fixed mineral fertilizer N application level (either with or without organic amendment) using a control plot (with or without organic amendment) as a reference. **(c and d)** N-AE is determined at an equal yield level in both response curves, using a control plot (with or without organic amendment) as a reference. **(e and f)** N-AE is based on the slope of a response curve to mineral fertilizer N application, with and without an organic amendment added, at equal yields in both curves. Results are based on 39 sets of response curves for winter wheat (17 experiments) and 12 sets of response curves for potato (7 experiments). Organic amendments include farmyard manure, slurry, straw, beet leaves, green manure or a combination of these. Data source: Hijbeek et al. (2017).

due to different depths of root systems between the two crops, we speculate this may arise at least partly from an improvement of soil structure and the role of organic amendments in supporting biotic mechanisms that contribute to crop protection, and the higher sensitivity of potato to soil-borne and soil-dwelling pests and diseases, which cannot or only partly be treated with crop protection agents. Organic amendments, be it as farmyard manure or crop residues, can suppress certain soil-borne diseases ((Scholte and Lootsma, 1998); Bailey and Lazarovits, 2003). They have been shown to enhance abundance and diversity of soil micro-, meso- and macro-biota (Scholte and Lootsma, 1998; Birkhofer et al., 2008; Viketoft et al., 2021; Heinen et al., 2023), generally leading to increases in beneficial organisms such as decomposers and predators, which in turn enhance suppression of crop pests (Birkhofer et al., 2008; Riggi and Bommarco, 2019; Aguilera et al., 2021). More research is required to assess how and when these different functions of organic amendments contribute to crop yields, and why some crops appear to benefit while others do not.

Further investigation is also required to assess whether the positive effects of organic amendments on potato N fertilizer use efficiency would lead to a net saving of N input to the cropping system. That would require an analysis of the total amount of N added to the system for all treatments, which was not possible for the dataset at hand. However, assuming that the organic amendments had already been produced elsewhere in the agricultural or food system, the benefit of recycling it and using it in combination with mineral fertilizers seems obvious for the systems level (Schröder et al., 2003).

4. Research for narrowing ecological yield gaps

4.1. From the field to the farm level

It is our assumption that the use of the ecological yield gap framework may help to advance the collaboration between ecologists and agronomists to build joint understanding and design of ecological intensification practices. The ultimate aim of this is to reduce external inputs, maintain or increase yields or input-use efficiency, such that it is beneficial in a whole farm context, also from an economic point of view. But how do ecological intensification practices affect the production, environmental and ultimately economic performance of the farming system? The latter requires the *upscaling* of the practices that narrow the ecological yield gap at field level to their assessment at entire crop rotations and farming systems. The obvious example to underpin the relevance of this is diversification of rotations: more diverse crop rotations may imply having to give up area shares of cash crops (e.g., vegetables, potato or onion) that are often much more profitable than cereals, leys, or grain legumes. The more so, if land is a scarce and very costly resource, and taking into account investments of farms in specialized machinery and buildings for the cash crops. Also, there may be market constraints and promotors at stake, i.e., markets for alternative crops, such as grain legumes in Europe, which are currently relatively small, while processing industries of vegetables and potato may request large volumes. Other examples of the importance of the farming system context refer to availability of organic manure and the feasibility of fodder crops in a diversified rotation, which both depend on the presence of relatively nearby livestock systems.

The *temporal* aspects of ecological intensification also require attention. Practices (e.g., integrated crop management practices to control pests and diseases) need to be reliable year after year. Moreover, some practices may become rewarding only when practiced for multiple years (e.g., use of organic resources, relying on improving soil structure and soil organic matter), deterring farmers who cannot or do not want to wait for the benefits. Compensation for lagged effects is thus relevant. Therefore, it is important to capture temporal variation and stability in effects of practices. This may be particularly important in the context of climate change adaptation. Effects of ecological intensification on stability of yields or input-use efficiencies remain unclear. On the one hand, ecological intensification practices could be expected to lead to lower stability because farmers have less control over how and when ecosystem services meet crop needs (e.g., nutrients becoming available through mineralization and presence of predators to manage pests or diseases) compared to meeting a crop's needs with (carefully timed) external inputs. On the other hand, relying on external inputs comes also with risks, both from the biophysical perspective (e.g., weeds, pests and diseases becoming resistant to pesticides) and from the societal domain (e.g., bans on certain pesticides). The fact that many ecological intensification practices are also multifunctional, i.e., can simultaneously supply nutrients, improve soil structure, and support beneficial microbes and arthropods, could contribute to improve yield stability and reduce risks. For example, complex crop rotations containing legumes, particularly perennial legumes, can supply N and enhance resilience to drought (Bowles et al., 2020, 2022; Grandy et al., 2022), whilst potentially also contributing to weed, pest and disease suppression (Davis et al., 2012; Storkey et al., 2019).

The uncertainty on temporal effects of practices brings us to an additional consideration in further research improving input-output relationships through ecological intensification, i.e., *risk attitude and decision making* of farmers. Attitudes towards variability and risk in system performance are likely important for the adoption of ecological intensification. In addition, such attitudes are likely to play a critical role in the observed huge diversity in input supply and poor relationship with yield (Fig. 1), and will thus be very important in the process of optimization and moving towards the frontier (Figs. 1 and 3). To take an economic example, price ratios of inputs and crop products are such that it can easily pay off to use, for instance, an extra dose of 50 kg N ha^{-1} mineral fertilizer (~50 € ha^{-1}), even if it only leads to a modest yield increase of 1 t ha^{-1} of potato tubers (~175 € ha^{-1}). Prices are indicative for the year 2023 in Western Europe. Note that such yield increases are equivalent to a mere 1–2 % of northwestern European yields. Clearly, from a purely economic perspective the extra fertilizer N might be regarded as a cheap insurance premium, which pays off even if there is a positive yield response in only one out of three years. On top of this, it must also be noted that the use of manure is a special case. This is particularly relevant in countries with large livestock populations, such as the Netherlands, where the manure price is often low or even negative because crop farmers get paid by livestock farmers to use slurry. Obviously, this is not an incentive to economize on the use of (organic) fertilizer (cf. Fig. 1b) and may explain why Silva et al. (2021b) found a negative (rather than positive) effect on N use efficiency if mineral fertilizers were complemented with animal manure. All this points at the importance of socio-economic perspectives on decision making in agricultural management, accounting for farmers' knowledge, attitude and cultural norms (Skevas et al., 2013; Daxini et al., 2018; Bakker et al., 2021; Dequiedt et al., 2023; Dietrich et al., 2025). This bears critical relevance for the enhanced adoption of ecological intensification practices and for the transition towards more favourable input-output relationships in general.

In short, adequate attention in research for upscaling, temporal aspects and farmers' risk attitude and decision making will enhance solid assessments of the feasibility at farm level of ecological intensification practices in narrowing ecological yield gaps. This will be essential in devising proper incentives and policies to enhance the uptake of ecological intensification practices.

4.2. Four types of research

The role that ecological intensification practices may serve in a production function context (Fig. 2) and the considerations to assess their feasibility in a farming context (Section 4.1) make us suggest that four types of research are needed. Firstly, incorporating ecosystem services into cropping systems requires further research into the *ecological mechanisms and relationships* at play in crop fields, and exploring how they can be managed. Recent research points at potential to enhance belowground ecosystem service provision via better knowledge and management of soil microbiology (Grandy et al., 2024), and toward the fact that pest suppression by wild predators may be greater than previously assumed (Frank, 2024; Romanowski et al., 2024). Having identified the potential of such mechanisms, the next step needed is to understand what actions can be taken by farmers to conserve and/or harness these ecosystem services.

We think that a particularly promising direction to identify rewarding ecological intensification practices would be to investigate interactions between crop nutrition and crop protection. These are two complementary functions that can either be provided by external inputs or, at least partly, by the multifunctional nature of ecological intensification practices. The validity of Liebscher's and perhaps Mitscherlich's law (Section 2 and Supplementary Data) points at the importance of balanced growth factors and strong and positive interactions between yield-increasing and yield-protecting inputs: nutrients (and also water) will be used more efficiently if the crop is well protected against weeds, pests and diseases. Yet, crop nutrition and crop protection are often studied by different scholars, which hinders the exploration of such interactions.

Another potentially rewarding interaction between crop nutrition and crop protection relates to interactions between weeds and their control, diversification, and fertilizer use. As mentioned in Section 3.2, lack of diversification in crop rotations has a selective effect on weed populations (Doucet et al., 1999). High N inputs further negatively affect weed species richness (Storkey et al., 2010, 2012), and shift the weed community towards more competitive species that are hard to control (Fried et al., 2009; MacLaren et al., 2020; Berquer et al., 2023). Resistance to herbicides and environmental legislation aggravate the challenge, making weed control very dependent on mechanical practices and/or scarce (hand) labor. It could therefore be easier to achieve both improved weed management and N use efficiency by tackling them together, rather than separately.

A second type of research refers to *on-station experimental research* to investigate effects of single or a few ecological intensification practices, including interactions. Preferably these are long-term experiments which can account for the dynamics, i.e., variability, evolution in time and stability of effects of practices. The long-term experiments on diversification and on the use of organic amendments (Section 3) are obvious examples. Experiments on organic agriculture or comparing mainstream and organic systems can be very instrumental. And, also in this type of research more focus on interactions between crop nutrition and crop protection is of interest, although it is not trivial to design such experiments.

In parallel to on-station experimental research, we thirdly also need *on-farm observations*, either using surveys or measurements (Section 3.1). One benefit of such research is that practices are already being used in a whole farm context and allow the assessment of economic impacts. We think that making use of detailed farm level data on crop management and yields, such as those employed in Figs. 1 and 3, will be very powerful for several purposes. It can be used for benchmarking, i.e., to learn from the variation among farms and to identify what farms at the frontier do differently from others and whether that can be used to improve performance of the others. Also, if farms are benchmarked over a longer period trajectories of change can be analyzed (Fig. 3). Obviously, attribution of effects to ecological intensification practices or to other changes in management (e.g., optimization of input use) requires sufficient detail in observations or measurements and relying on large datasets and statistical methods to reveal such effects. Observations and measurements should then include information about types of mineral and organic fertilizer inputs, timing and method of application, crop protection methods, crop rotations, use of service crops, buffer strips, and other ecological infrastructure and agro-biodiversity. Indeed, such studies may be the only way to quantify yield gap contributions made by ecological infrastructure implemented at the farm and/or landscape scales (Boinot et al., 2023; Storkey et al., 2024). While accounting for many confounding factors and incomplete management information will be challenging, at the very least on-farm observations can be used to formulate hypotheses to be tested further in experiments, on-station or on-farm.

That brings us to the fourth type of future research: *on-farm experimentation* with a few, relatively simple practice treatments, e.g., the substitution of (part of) the mineral fertilizers or pesticides by other inputs or practices. To make it feasible for farmers to (co-)manage such experiments, the number of replications per farm will have to be limited, but this could be made up by doing the experiments on a larger number of farms as is very common in, for instance, nutrient omission trials (cf. Aliyu et al., 2021).

5. Conclusions

In the context of growing societal and political pressure to reduce external input use and improve input-use efficiency in intensive agriculture, we introduce the notion of an ecological yield gap. We define it as the yield increase or input saving that could be achieved in a given context (climate x soil x cropping system) by increasing the delivery of ecosystem services through ecological intensification practices. These practices support nutrient cycling, pollination, and weed, disease and pest regulation, with substitutive and complementary roles vis-a-vis the external inputs including mineral fertilizers and pesticides. Our quantitative examples of ecological intensification practices refer to crop diversification in widened rotations, service crops, and the combined use of organic and mineral fertilizers. They show both substitutive and complementary roles, but we have not conclusively investigated whether one is more important than the other along the spectrum of input levels. Positive interactions between crop nutrition and crop protection against weeds, pests, and diseases further enhance the potential of ecological intensification. The concept of an ecological yield gap and employing production functions enriches the agronomic and ecological literature by offering a method to quantify the contribution of ecosystem services to crop yields alongside other yield constraints, and to understand how this contribution varies in relation to input use and other farm management practices and contextual factors. Due attention is needed for research and data beyond the individual field and crop level and for temporal dynamics to assess feasibility of practices at farm level. Such extended and novel research will allow the quantitative assessment of the potential of ecological intensification to save external inputs, improve resource-use efficiency and maintain yields. In this way, the concept of ecological yield gaps can enhance a much-needed communication and research collaboration between agronomists and ecologists to tackle some of the grand challenges related to sustainable food production.

CRediT authorship contribution statement

Martin K. van Ittersum: Writing – review & editing, Writing – original draft, Visualization, Supervision, Methodology, Funding acquisition, Conceptualization. **João Vasco Silva:** Writing – review & editing, Visualization, Methodology, Conceptualization. **Riccardo Bommarco:** Writing – review & editing, Methodology, Funding acquisition, Conceptualization. **Renske Hijbeek:** Writing – review & editing, Visualization, Investigation, Formal analysis, Data curation. **Ola Lundin:** Writing – review & editing, Visualization, Formal analysis, Data curation, Conceptualization. **Romain Nandillon:** Writing – review & editing, Visualization, Formal analysis, Data curation. **Göran Bergkvist:** Writing – review & editing, Funding acquisition, Conceptualization. **Alexander Menegat:** Writing – review & editing, Investigation, Conceptualization. **Ingrid Öborn:** Writing – review & editing, Project administration, Funding acquisition, Conceptualization. **Annika Söderholm-Emas:** Writing – review & editing, Conceptualization. **Frederick L. Stoddard:** Writing – review & editing, Conceptualization. **Giulia Vico:** Conceptualization, Visualization, Writing – review & editing. **Wytse J. Vonk:** Writing – review & editing, Data curation. **Christine A. Watson:** Writing – review & editing, Conceptualization. **Chloe MacLaren:** Writing – original draft, Visualization, Methodology, Conceptualization.

Declaration of competing interest

Statement about Conflict of interest related to manuscript *Narrowing the ecological yield gap to sustain crop yields with less inputs*.

The authors declare that they have no conflict of interest.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.gfs.2025.100857.

Data availability

Data will be made available on request.

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