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Effects of farm type on food production, landscape openness, grassland biodiversity, and greenhouse gas emissions in mixed agricultural-forestry regions

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ABSTRACT

Context: The global demand for food is expected to continue increasing for decades, which may drive both agricultural expansion and intensification. The associated environmental impacts are potentially considerable but will depend on how the agricultural sector develops. Currently, there are contrasting regional developments in agriculture; expansion and/or intensification in some regions and abandonment in others, as well as changes in the type of farming. However, the environmental consequences of changes in farm type are not well understood.

Objective: We have evaluated the impacts of farm type on food production and three key environmental variables—landscape openness, grassland biodiversity and greenhouse gas (GHG) emissions—in three marginal agricultural regions in Sweden.

Methods: We do this by first dividing the population of farms in each region into types, based on their land-use and livestock holdings using an innovative clustering method. Thereafter we analysed changes in production activities for farm types over time and evaluated the environmental and food-production impacts, where landscape openness is quantified using a novel indicator.

Results and conclusion: Our results show that there is not one single farm type that would simultaneously maximize food production, grassland biodiversity, and landscape openness, whilst minimizing GHG emissions. However, there exists considerable potential to manage the trade-offs between food production and these environmental variables. For example, by reducing land use for dairying and instead increasing both cropping for food production and extensive livestock grazing to maintain landscape openness and biodiversity-rich semi-natural pastures, it would keep food production at similar levels.

Significance: Our farm typology allows us to assess the multifunctionality of farming, by relating contrasting production activities to multiple ecosystem services, grassland biodiversity and GHG emissions for informing policy towards more sustainable agriculture. We have demonstrated this with examples under Swedish conditions, but it should to a large extent also be applicable for other countries.

1. Introduction

Globally, agricultural ecosystems cover around 37% of the terrestrial surface (Food Agriculture Organization of the United Nations, 1997). The structure of agriculture shows contrasting regional developments with agricultural expansion into natural areas and/or intensification in some regions of the world, and agricultural abandonment in other regions (Levers et al., 2016; Levers et al., 2018). Agricultural expansion and abandonment are influenced by multiple socio-economic drivers, including: changes in the global demand for food (Tilman et al., 2011), loss of agricultural profitability in marginal areas (Ustaoglu and Collier, 2018), and loss of agricultural land because of, e.g., urbanization. These

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changes may have profound consequences for biodiversity and ecosystem services (Díaz et al., 2020), depending on the alternative to agricultural land-use. However, in a concomitant process the structure of agricultural landscapes also changes, often because of the same drivers that affect agricultural expansion and abandonment. In particular, ecological heterogeneity is lost at multiple spatial scales due to within-field intensification, loss of semi-natural habitat and increasing farm specialization (Emmerson et al., 2016). Moreover, these changes may have consequences for both biodiversity and ecosystem services (Benton et al., 2003). Hence, the consequences of agricultural expansion or abandonment cannot be understood without also accounting for concomitant structural changes in farming and landscapes. For instance, in some parts of the world, declining animal production could lead to the abandonment of semi-natural grasslands where extensive grazing has shaped highly valued species-rich communities of conservation concern (Auffret et al., 2018; Springmann et al., 2018) and which contribute to landscape multifunctionality (Bengtsson et al., 2019).

Although the specialization of farming and loss of farm-level heterogeneity are known to cause declines in biodiversity (Benton et al., 2003) and changes the mix of ecosystem services (Raudsepp-Hearne et al., 2010), surprisingly few studies have investigated such changes in an integrated analysis at the farm level, across farms with contrasting farming enterprises and practices. In this context, the specialization of farming allows grouping of farms into farm typologies based on their primary activity, which in turn makes it possible to link specific farming activities to responses in biodiversity, ecosystem services and emissions including greenhouse gas (GHG) emissions. Studies based on regionally aggregated data have demonstrated contrasting patterns in terms of environmental outcomes and production (e.g. Raudsepp-Hearne et al. (2010)). Spatially explicit farm-level data may more directly reveal the underlying drivers of land-use change and consequences for biodiversity and ecosystem services. In a European context, farmers' decisions are highly influenced by policy, particularly payments from the European Union's Common Agricultural Policy (CAP) (Leventon et al., 2017). A key challenge is thus to better understand the heterogeneity that exists among farms in terms of their farming activities, the patterns in how they use the land, and how land-use decisions affect ecosystem services (food, landscape openness), GHG emissions, and farmland biodiversity. This is in line with the goals of the European Union's *Farm 2 Fork* strategy presented as part of the *European Green Deal*, which is aiming to make European food systems (agriculture) environmental friendly, fair, and healthy (European Commission, 2020).

Over the past four decades, the number of farms in Europe have been decreasing and the average farm size increasing (De Roest et al., 2018). In parallel, farms have increasingly specialised their production to focus on dairying, arable cropping, or meat production (De Roest et al., 2018). In marginal farming areas, including mountainous, upland, and boreal-forest dominated regions, the continuation of extensive farming is important, not only for producing food, but also for preserving biodiversity and keeping the landscape open (Keenleyside et al., 2010), as well as sustaining regional economies and livelihoods. Recent studies on landscape-level ecosystem service provisioning have mostly focused on bundles of services for administrative regions or clustered areas (Queiroz et al., 2015; Mouchet et al., 2017; Quintas-Soriano et al., 2019), or single locations (Andersson et al., 2015; Nikodinoska et al., 2018). To our knowledge, there is a lack of studies that explicitly link changes in biodiversity and ecosystem services with increasing specialization of production, despite the latter being an important link between agricultural policy and environmental outcomes (Leventon et al., 2017).

The division of farms into types (typology) is one option for understanding farm diversity (Landais, 1998). For example, the European Union have for decades been using a farm typology when presenting and analysing agricultural statistics. Farm typologies are also used in research on agricultural systems for identifying clusters in objectives, resource allocation, and production (Bhattarai et al., 2017; Guiomar et al., 2018). The use of such a typology is also a practical way to analyse how farm specialization relates to biodiversity, ecosystem services and GHG emissions.

Here, we develop a method to divide farms into types based on their production activities using the Integrated Administration and Control System (IACS) database and data on livestock holdings. The method is based on a farm being a particular type if two thirds of the farm's income comes from a single production activity (Andersen et al., 2006), which if it includes livestock, can include the arable land used for fodder production as part of the livestock grouping (SCB, 2000). Our method allows the standard output (monetary value of agricultural output at the farm gate, an EU term) of both livestock and arable land production, either for fodder or for direct human consumption, to be combined, which gives a more accurate description of the actual farm specialization, which is of relevance in, for example, economic modelling.

We use the farm types to describe annual changes in the structure of agricultural production between 2008 and 2016 in three Swedish forestry-dominated regions. We also ranked how each of the farm types in 2012 relates to grassland biodiversity, landscape openness, food production, and GHG emissions. Only 7% of Sweden's land surface is currently used for agriculture, with forestry making up more than 70% of the balance, but geographical differences are large (The Royal Swedish Academy of Agriculture and Forestry, 2015; SCB, 2019). Swedish mixed agricultural-forestry regions, in particular, have experienced a substantial decrease in farm numbers (SCB, 2019; Swedish National Board of Agriculture, 2019), and the remaining farms are strongly dependent on Common Agricultural Policy (CAP) payments. Our farm typology approach sheds light on the role that different types of farming activities—and hence changes in production—have on shaping the landscape and providing ecosystem services in marginal agricultural regions, and how policy can affect these outcomes.

2. Methods

2.1. Study regions and data

We studied three Swedish mixed agricultural-forestry regions: Northern Scania, Jönköping County, and Västerbotten County (Fig. 1). We chose these regions, which we refer to as N. Skåne, Jönköping, and Västerbotten, to cover a representative range of farming and climatic conditions in forest-dominated areas of Sweden. The farm typology used spatially explicit agricultural information from the IACS database and from the national register of livestock holdings, retrieved from the Swedish Board of Agriculture. We calculated landscape openness by combining data from the European Digital Elevation Model from Copernicus (https://land.copernicus.eu/imagery-in-situ/eu-dem/eu-dem-v1.1), tree height data from Swedish Forest Agency, and road/trail data from The Swedish Land Survey Authority (Lantmäteriet). The tree height data is based on a laser scan of Sweden and includes trees in forests as well as along agricultural field borders and roads. We estimated grassland biodiversity data from the Swedish survey of semi-natural grassland habitats (TUVA) provided by the Swedish Board of Agriculture. This was supplemented with the High Nature Value (HNV) farmland data from European Environment Agency (EEA) (see below for details). TUVA and HNV were selected since they are the only datasets on farmland nature value that have close to full coverage of the study regions.

TUVA is an inventory of the most valuable semi-natural grasslands, i.e., meadows and pastures, in Sweden, most of which were inventoried once between 2002 and 2004 (Swedish Board of Agriculture, 2005). We used information on the presence of 56 grassland indicator plant species (as did Auffret et al. (2018)), to calculate an indicator of plant biodiversity in the semi-natural grasslands in each region. The biodiversity measure was limited to a grassland diversity index and did not include hedges, trees, etc. across the landscape. The focus on grassland biodiversity is justified, however, both by the drastic loss of semi-natural

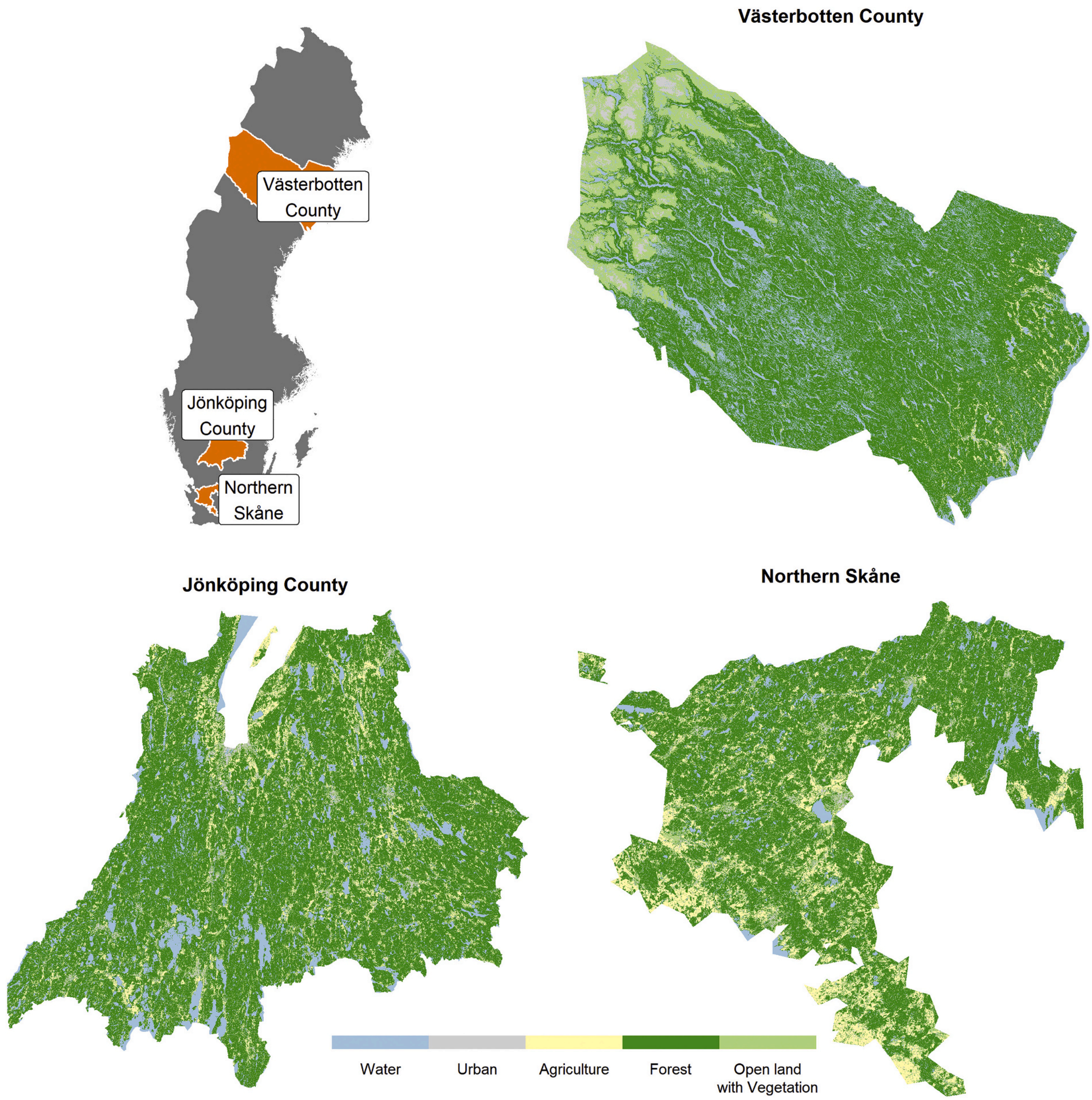

Fig. 1. Location of study regions within Sweden and their land cover. Land cover data from Swedish National Land Cover Database (Swedish Environmental Protection Agency, 2020).

grassland cover and density of associated species in the past 50–70 years (Auffret et al., 2018), and evidence of biodiversity declines in the more recent past (Tyler et al., 2018). The latest inventory date was used if the pasture had been inventoried more than once.

The HNV farmland dataset was developed to estimate the European distribution of farmland with high proportions of semi-natural vegetation, with a mosaic of low intensity agriculture and natural and structural elements, and/or areas supporting rare species or high proportions of the entire European or World populations of species using standardized methods across countries (Paracchini et al., 2008). The HNV data is binary and indicates whether a pixel (100 × 100 m) is considered to contain HNV farmland or not. In our study regions, HNV is based on CORINE land cover (CLC06 codes; 231-Pastures, 242-Complex cultivation patterns, 243-Land principally occupied by agriculture, 244-agroforestry areas, 231-natural grasslands, and 411&421-marshes) combined with TUVA, Important Bird Areas, and Natura 2000 areas (Paracchini et al., 2008). CORINE land cover has a minimum mapping unit of 25 ha (Büttner et al., 2004), and thus small farms might have been underrepresented by this biodiversity indicator. However, our analysis of the HNV dataset shows, for our three study regions, there are 5245 distinct HNV areas of which 55% are below 25 ha which shows its potential.

The complete analysis was done in the Swedish reference frame 1999 projection (SWEREF99TM) and the datasets used are described in detail in Table A.1.

2.2. Farm type

We assigned each farm in the study regions (Fig. 1) to a type based on its livestock holdings and use of agricultural land. The method used was based on a combination of the method presented by Andersen et al. (2006) and that used by the Swedish Board of Agriculture (SCB, 2000). Each head of livestock and hectare of land was converted to a standard output (SO) in Euros, using the regional SO coefficient from Eurostat (https://ec.europa.eu/eurostat/web/agriculture/so-coefficients). Initially we transferred the SO of intermediate activities (e.g., fodder production for own livestock) to reflect their contribution to the relevant final output, as done by the Swedish Board of Agriculture (SCB, 2000). This estimate was based on the method used in the Yearbook of Agricultural Statistics, described in detail in the year 2000 issue (SCB, 2000), see Table 1 for conversion factors. Accordingly, we transferred ley-SO from arable to dairy farming, based on the estimated ley area (i.e. sown grass for silage) used in milk production. We moved the SO for barley to pig production by allocating a maximum of 0.4 ha of barley per pig. If the maximum allocation of barley was not reached, we moved the SO of an additional 0.1 ha of oats to pig production, and hence a maximum SO corresponding to 0.4 ha of land was moved for each pig. We applied the same method of moving SO for fattening pigs, with 0.12 ha barley and 0.06 wheat, respectively, to reach a maximum of 0.12 ha (Table 1). Finally, the SO of heifers, calves, and bulls were moved into the dairy type with a maximum of one calf for each dairy cow and 40% of the number of dairy cows for heifers and bulls. Thereafter, we moved the remaining SO for calves, heifers, bulls, and suckler cows to the beef type.

In general, we considered a farm to represent a specific type if the SO of that type exceeded two thirds of the farm's total SO, following Andersen et al. (2006). However, a farm was deemed a small or hobby (i.e., non-commercial) type if the total SO was below 5800 € p.a., chosen to match the statistics of small farms presented in the Yearbook of Agricultural Statistics for Jönköping and Västerbotten Counties (N. Skåne omitted since it is not present in the statistical yearbook). The full set of farm types and typology rules are described in Table 2. The types Mixed, and Mixed Livestock were combined into a single Mixed type to reduce the number of types studied. The types Horticulture and Permanent Crop were very rare and therefore excluded from further analysis.

2.3. Development of farm types over time

We used the farm typology method described above on a time series of IACS and livestock data (2008–2016) to analyse how the farming systems in the three regions have changed over time. We used data from the median year 2012 to carry out the ensuing analysis.

2.4. Landscape openness

Landscape openness can be defined as the amount of open space that is visible to a viewer, the visible area. It can be an important

Table 1
Estimated arable land need for each livestock type per year used to move SO from the arable land group to livestock groups. The ley need is based on a dairy cow's need of 0.7 ha ley. The ley need for all other types is then scaled according to their livestock unit. The ley need is the land needed to satisfy the basic need, but to account for normal yield variation this is doubled (e.g., 1.4 ha for 1 dairy cow). Pastures can be included in the ley area to satisfy the basic need (e.g. 0.7 ha for 1 dairy cow).

	Livestock Unit	Ley need (ha)	Barley/Oats need (ha)	Barley/Wheat need (ha)
Dairy Cows	1	0.7		
Suckler Cows	0.8	0.56		
Heifers >2 yrs.	0.8	0.56		
Heifers 1–2 yrs.	0.7	0.49		
Bulls	1	0.7		
Beef	0.7	0.49		
Calves	0.4	0.28		
Sheep	0.1	0.07		
Horses	0.8	0.56		
Sows >50 kg	0.5		0.4/0.1	
Sows	0.3		0.4/0.1	
Fattening Pigs	0.3			0.12/0.06
Piglets <20 kg	0.3		0.4/0.1	

Table 2
Type definitions based on farm total standard output (SO) including all land and livestock.

Type	Short	Definition	Farms in 2012 (%)
Field Crop	FC	>2/3 SO from Arable & <2/3 from HC and/or PC	13.7
Horticulture[a]	HC	>2/3 SO from Horticulture	0.2
Permanent crop[a]	PC	>2/3 SO from Permanent crops	0.4
Dairy	D	>2/3 SO from Dairy	10.4
Grazing Livestock	GL	>2/3 SO from Sheep or >2/3 SO from Beef	26.8
Granivores	G	>2/3 SO from Pigs + Poultry	0.6
Horse	Horse	>2/3 SO from Horses	0.2
Small	Small	0 € < SO <5800 €	36.3
Passive[a]	Passive	SO = 0 €	0.2
Mixed Livestock	ML	None of above and >2/3 SO from (D + GL + G + Horse + Pasture)	0.4
Mixed	M	None of above.	4.4

[a] Only shown to indicate the rules but has not been used for further analysis in this paper.

characteristic of how the landscape is perceived, and how attractive it is to people (Tveit et al., 2006). Furthermore, structural heterogeneity in mosaic landscapes has been positively related to the aesthetic values of landscapes (Dramstad et al., 2006; Junge et al., 2011). These values need to be interpreted in a spatial context appropriate for the study (Dramstad et al., 2006). We have identified openness as a suitable indicator for the aesthetic values of our three study regions because a reduction in openness in these already forest-dominated study regions, corresponds to a reduction in structural heterogeneity. This is supported by Tahvanainen et al. (1996) who showed that an afforestation is in most cases associated with a loss of perceived scenic beauty in a Finnish rural landscape that is comparable to our study regions.

To calculate the consequences of a change in farm type, we quantified landscape openness by developing a novel visibility change analysis, which identifies how each farm type contributes to landscape openness. We created 100 random samples of 1000 points located on public roads (<7 m in width), walking paths or trails. We set the observation height to 1.8 m and used the earth curvature with a refractivity factor (atmospheric influence on visibility) of 0.13 (default value). To reduce computational demands, the maximum visible distance was set to 25 km in all directions.

We resampled the digital elevation model with the nearest neighbour to match the resolution of the tree height dataset. Since the digital elevation model defines the land surface elevation, we added the tree height data to generate a combined digital surface model. However, before adding the tree heights, we ensured that we did not add a tree to any of the pixels defined as an observer point. This could happen, for example, in the case of forest walking trails. In such cases, we set the pixel tree height to zero.

Thereafter, we performed a visibility analysis using the "viewshed" tool in ESRI ArcGIS 10.0 for each simulated observation to obtain a measure of the visible area. We then removed each farm type separately by replacing open farmland with forest and generating a new artificial digital surface model. The openness reduction was then calculated as the average reduced visible area perceived at an observation point where that farm type was visible. For example, only observers that had a view over dairy farmland were included in the calculation for the reduction in visible area when afforesting dairy farms. This was done to reduce the impact of the total size difference of the different farm types.

2.5. Grassland biodiversity and High Nature-Value farmland

We based our grassland biodiversity estimation on the contention that semi-natural grasslands contain much of the biodiversity of conservation value (Billeter et al., 2008; Cerezo et al., 2011). We carried out the grassland biodiversity and HNV farmland analyses by spatially connecting each farm to TUVA and HNV farmland data. Although these two datasets were not temporally aligned with the 2012 agricultural data used to determine the farm type, the total agricultural area did not change dramatically from year to year. Therefore, we assume that the effect of the temporal mismatch between data sources to be negligible when aggregated to the farm-type level. We chose these data sources because TUVA and HNV data cover a large spatial extent, which is necessary for connecting grassland biodiversity estimates to a large number of farms. We assigned the number of grassland indicator species for each farm by spatially matching its agricultural land-use parcels to the TUVA polygons. If a TUVA polygon overlapped with an agricultural land-use parcel in the IACS database, we assigned the number of grassland indicator species to that land-use parcel. If multiple TUVA objects overlapped with fields from the same farm, we calculated the mean number of grassland indicator species. To obtain an HNV indicator for each farm, we computed the fraction of the farm area identified as having high nature value for each farm, according to the HNV dataset.

2.6. Comparison between types of landscape openness, grassland biodiversity and High Nature-Value farmland

For each of the three response variables—proportion of HNV area, proportion of TUVA area, and the fraction of visible area remaining after replacing open farmland for a particular farm type with forest—we fitted GLMs for over/underdispersed binomial data (quasibinomial, logit link) with region and farm types as explanatory variables. We assessed whether the differences between farm types were significant with Log-likelihood-ratio tests, which, when significant ($P < 0.05$), were followed by pairwise post hoc tests with Tukey-adjusted P-values and Šidák-adjusted confidence levels.

2.7. Greenhouse gas emission and food produced

We related the farming activities to their carbon dioxide equivalent emissions per kg food produced following Leip et al. (2015). To this end, we converted the numbers of livestock and hectares of arable land to kg product. Regarding crop production we used standard yields for the major crop types from the Swedish Board of Agriculture. Concerning livestock products we used the slaughter age-factor and weight from an agricultural planning tool (Agriwise, 2016). We used these factors to convert meat production to annual equivalents for livestock with a lifespan of >12 months. We thereafter related carbon emissions to farm size and the number of calories produced (kcal) to allow a fair comparison in terms of both area used and food produced. To avoid over- or under-estimating the area of a farm, we recalculated farm area to only account for land that was used to produce food. This was done by using a dietary demand of 100 MJ per day per livestock unit (Spörndly and Glimskär, 2018). We assumed that the in-farm production was used as feed/pasture. If there was a demand for more feed, we added extra ley area to the total farm area. If the farm was overproducing feed (not directly consumable by humans) that area was omitted from the farms' total area. This was the case for most of the field crop farms, which we assume corresponds to the area used to feed horses, sold as feed, or kept as fallow.

The conversion of product from kg to kcal was done using factors from Shepon et al. (2016). We acquired conversion factors for sheep (lamb) from U.S. Department of Agriculture's database on food (FoodData Central NDB Number: 17002). We analysed carbon emissions for the major food-producing farm types dairy, field crop, grazing livestock, granivores and mixed. Small farms and horse farms were excluded since they are not considered to be food-production oriented farms. All factors used for the conversion to carbon emissions and kcal are presented in Appendix A.

2.8. Comparison of food, grassland biodiversity, contribution to HNV farmland, landscape openness and GHG emissions

We compared landscape openness, grassland biodiversity, GHG emissions and food produced for the farm types: dairy, field crop, granivores, grazing livestock and mixed. The comparison was based on how the average values of a farm type were related to the 5th and 95th percentile values of all the farms. The values are scaled relatively to the agricultural area or the food produced to allow comparison of farms independent of their size. Furthermore, to ensure a fair comparison in terms of food production and emissions, we only included the areas that are assumed to be related to food production of that farm type in this analysis. Therefore, by default, we do not include pastures and ley in the areas of the field crop farms since the human-consumable food production of that farm type is purely plant based. This also avoids double counting of areas, but it consequently reduces the grassland biodiversity estimates and the landscape openness contribution of the field crop type. The field crop farms had 17% HNV area, but after excluding pastures and ley this was reduced to 2%. This adjustment to the area of field crop farms was done only to produce Figs. 6 and 7.

3. Results

3.1. Development of farm types over time

We found that farm structural changes in all three study regions showed a similar pattern between 2008 and 2016 (Fig. 2). Our results show a small reduction in total farm area between 2008 and 2016 of −2.4%, −0.5%, and −5.6% for Jönköping, N. Skåne, and Västerbotten respectively (Fig. 2A). However, because of a reduction in the total number of farms, the average area of each farm increased by 6.5 ha for Jönköping, 6.7 ha for N. Skåne, and 9.8 ha for Västerbotten (Fig. 2B&C). The total area of semi natural pastures slowly decreased in all regions until 2014, after which the area increased slightly (Fig. 2D). Because of the almost identical trends in all three regions (Fig. 2), we combined the regions in all further analyses.

At the farm type level (Fig. 3), we found a strong positive trend in the average farm size for dairy (D, +4.9 ha/yr) and granivores farms (G, +8.2 ha/yr) (Fig. 3A). Concomitantly, these trends were accompanied with a large decrease in the number of farms over the study period (D: −35% and G: −58%, Fig. 3B). Field crop and grazing livestock farms were the only farm types that increased their total area (Fig. 3A). Field crop farms were the only farm type that increased in numbers between 2008 and 2016 in our study regions (+8.5%, Fig. 3B). The strong decrease in the number of small farms observed between 2009 and 2010 is mainly explained by a change in the minimum farm area required to receive the EU's Single Farm Payment, which was increased from 0.3 ha to 4 ha in 2010. The increment of semi-natural pastures for all regions after 2014 (Fig. 2D) was mainly driven by an increase in pasture area use by the grazing livestock farms of +1660 ha between 2014 and 2016 (Fig. 3D).

3.2. Landscape openness

Our analysis for year 2012 showed, on average across all regions and farm types, that a simulated abandonment of dairy farms had a slightly larger, significant, negative effect on landscape openness (reduced visibility) than GL, G, FC, and M, in particular in contrast to horse farms and small farms (Fig. 4), which are significantly smaller, on average, compared to the other farm types (Fig. 3C).

3.3. Grassland biodiversity and High Nature-Value farmland

Semi-natural pastures in the TUVA database, aggregated at the farm level, comprised 7% of the total farm area in the three study regions combined, and 8%, 10%, and 2% for Jönköping, N. Skåne, and

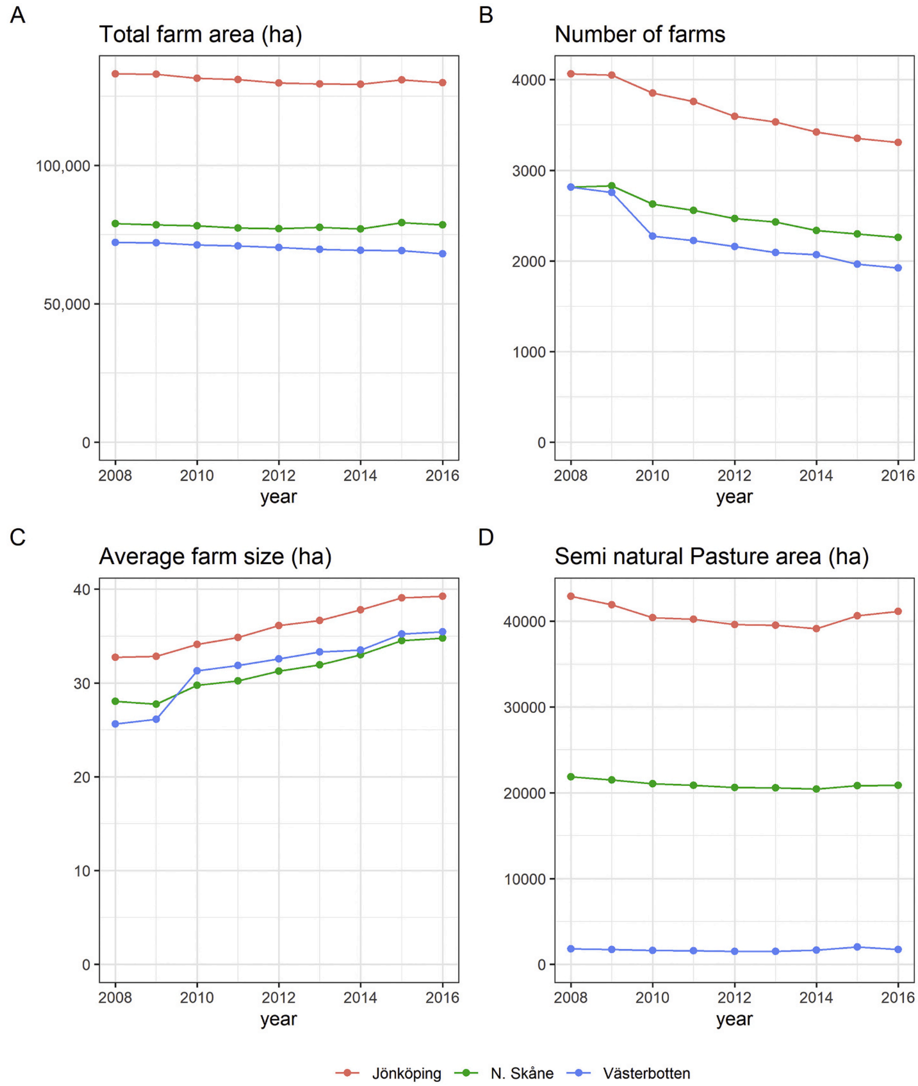

Fig. 2. Development of farms (A–C) and semi-natural pastures (D) over time for the three study regions Jönköping (red), N. Skåne (green), and Västerbotten (blue), A: Total farm area, B: Number of farms, C: Average farm size, D: Semi-natural Pasture area.

Västerbotten, respectively. After controlling for regional differences, we found that grazing livestock, mixed, and small farms had significantly larger proportions of TUVA pastures that contained high biodiversity values compared to dairy, field crop, and granivore farms (Fig. 5A). No clear differences were found for horse farms (Fig. 5A). The number of TUVA indicator species for the TUVA objects (Fig. 5B) did not differ significantly (at $p < 0.05$ level) between the types. Patterns in farm HNV cover were similar to those observed for the TUVA pasture cover in that grazing livestock, mixed, and small farms had significantly larger proportions of HNV cover compared to dairy, field crop, and granivore farms (Fig. 5C; models and test as for proportion TUVA pastures).

However, we found large differences in both HNV land area and indicator species among the regions. Semi-natural grasslands on farms in Jönköping contained 6.8 grassland indicator species on average, compared to 3.4 and 3.3 for N. Skåne and Västerbotten. Regarding HNV cover, the pattern was similar but more inflated (Jönköping 43%, N. Skåne 20%, and Västerbotten 16%). We found no relationship between the average size of the TUVA semi-natural grasslands in the three regions that could have influenced the result (Fig. A.1B). We also found a strong, positive, significant correlation (Pearson correlation $= 0.87$, $p < 0.01$) between the average semi-natural pasture (SNP) area and TUVA area per farm type and region (Fig. A.3). However, there was no significant correlation between the farms' TUVA area and average farm area (Pearson correlation $= -0.18$, $p = 0.41$). Thus, the type of farming activity was secondary in determining the HNV cover and the average number of indicator species, but instead the regional context was the main determinant of the occurrence of grassland indicator plants.

3.4. Greenhouse gas emission and food produced

Field crop farms produced more food (kcal) per hectare of land compared to the other farm types (Fig. 6A), while dairy farms were the biggest emitter of GHGs (kg CO_2e) per hectare of farm area (Fig. 6B). Field crop farms were also the farm type that produced the most food (kcal) per CO_2e emissions (Fig. 6C).

3.5. Combined analysis

We found the average dairy farm to be the farm with the highest contribution to maintaining landscape openness, but also with the highest GHG emissions per hectare (Fig. 7). The trade-offs between landscape openness, grassland biodiversity, HNV cover, and GHG emissions are also highest for dairy farms. Nevertheless, grazing livestock and mixed farming generate similar landscape openness, grassland biodiversity, HNV cover, compared to dairy farms, but with a much lower climate impact, measured per hectare. Hence, replacing dairy

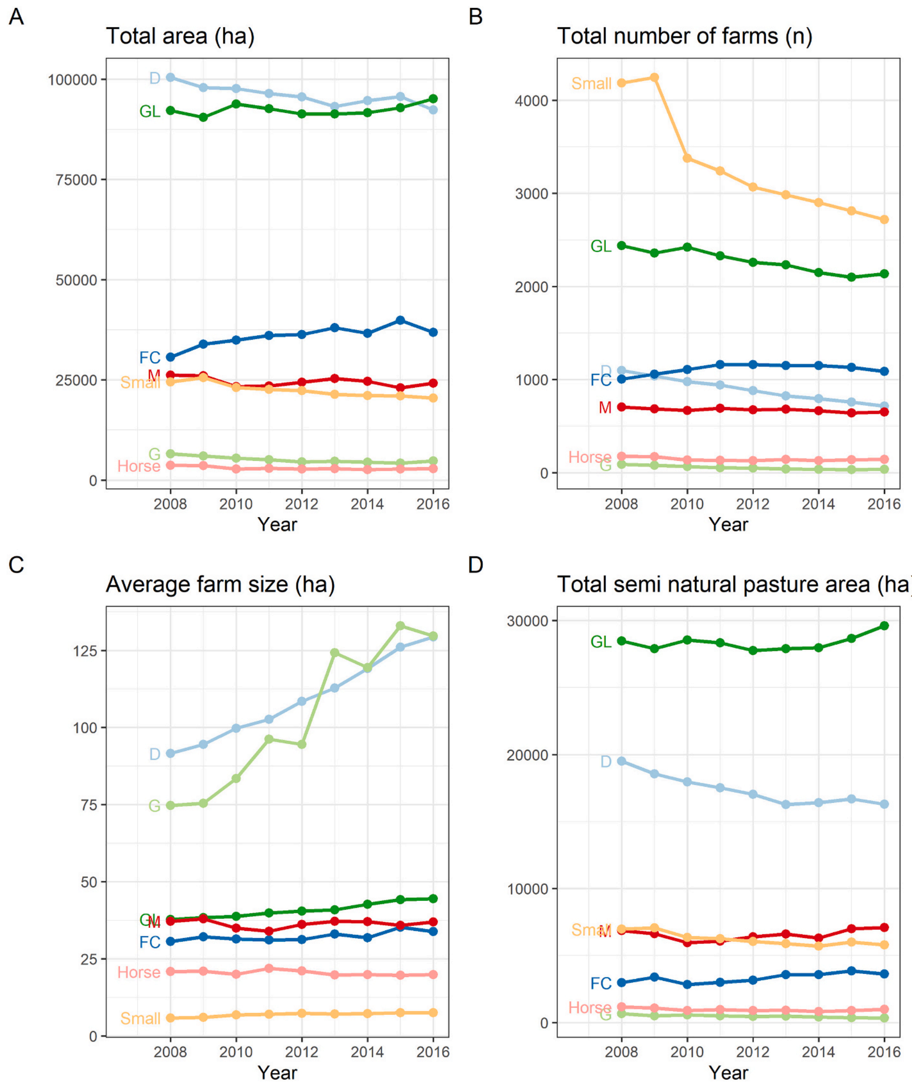

Fig. 3. Development of farm types (D: Dairy, GL: Grazing livestock, FC: Field Crop, M: Mixed, G: Granivores) over time for all the study regions combined. A: Total area of each farm type, B: Number of farms, C: Average farm size, D: Total area of semi natural pastures.

cows with suckler cows and sheep would maintain openness and grassland biodiversity values while simultaneously reducing GHG emissions per hectare. Field crop farms had the lowest GHG emissions per produced calorie and highest produced calories per hectare (Fig. 7). Our results indicate that a shift towards using the arable land for food production instead of feed to dairy cows would increase food production while lowering GHG emissions. However, field crop farms had the lowest HNV cover, plant indicator species and landscape openness values.

4. Discussion

We have developed a spatially explicit approach for mapping a large population of farms to farm types based on their major production activities, and related those production activities to their potential for providing food, landscape openness, and grassland biodiversity while limiting their GHG emissions. Our results show that there is not one single farm type, as described or presently configured, that would simultaneously maximize food production, grassland biodiversity, and landscape openness, whilst reducing GHG emissions. Our approach allowed us to 1) describe dynamic developments in farm types in the study regions, and 2) link production activity and/or type of farm to their impacts on GHG emissions and public goods such as grassland biodiversity and landscape openness to quantify trade-offs. Consequently, our approach enabled us to link production activity spatially to grassland biodiversity and landscape openness, which is not possible using regionally aggregated statistics. Our results are particularly relevant concerning the ongoing debate on the impact of different types of farming on societal welfare, considering that there are trade-offs between food production and other ecosystem services (Holt et al., 2016; Zabel et al., 2019). We show that farms in the study regions have decreased in number but increased in size while using a slightly smaller share of the total land area between 2008 and 2016 (Fig. 2), which mirrors the overall trend in Swedish agriculture (SCB, 2019). Based on area, dairy and grazing livestock farms are the two most common types of agriculture in the study regions (Fig. 3).

We identified major differences in food production and GHG emissions depending on farm type. Dairy farms produced most food in total, but also most GHG emissions per hectare. In comparison, field crop farms were, as expected, by far the most productive, producing most food as well as least GHG emissions per unit area, but were also associated with relatively low grassland biodiversity values (Fig. 7). The abandonment of small and horse farms had the least negative effect on landscape openness compared to the more production-oriented types (Fig. 4). The differences can originate from more trees on and around these farms compared to the fields belonging to more production-

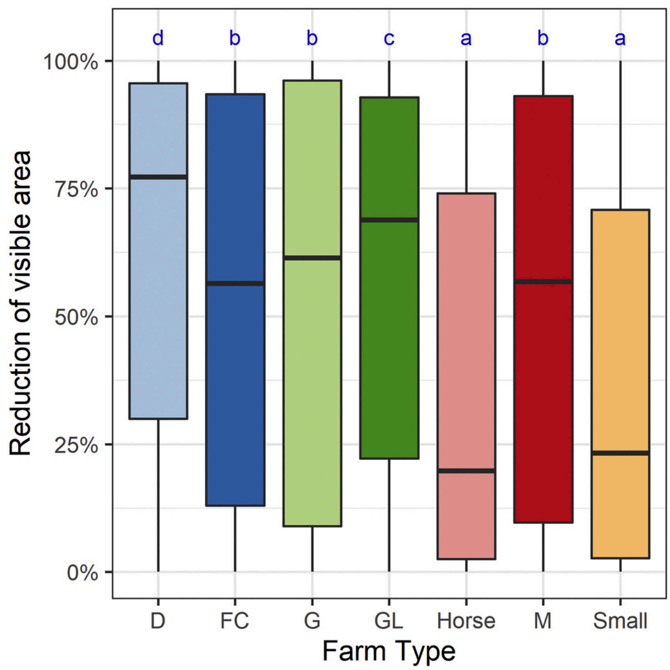

Fig. 4. The average reduction in visible area when afforesting fields belonging to a specific farm type. (D: Dairy, FC: Field Crop, G: Granivores, GL: Grazing livestock, M: Mixed). Different letters (blue above boxes) indicate significant differences according to the post-hoc tests (based on generalized linear models controlling for regional effects). The figure is based on data including all areas of all the farms.

oriented farms. There were no major differences in grassland biodiversity indicator values between farm types other than lower values observed for field crop farms, partly due to the exclusion of pastures and ley in the field crop type to only account for the food producing areas (in Figs. 6 and 7). Our results imply that structural change of agriculture, e.g., through agricultural and environmental policy, will be important for resolving agriculture's environmental challenges (GHG emissions, grassland biodiversity, and landscape openness).

The division of farms into types allowed us to perform a complete analysis of how the different production systems influence landscape openness, grassland biodiversity and GHG emissions in the Swedish mixed agriculture-forestry regions. However, our analysis is based on farm type and does not consider the potential for a single farm to, for example, reduce its GHG emissions without changing its activity. But the analysis still paves the way for evaluating the impacts of policy on farm structure (scale and type of farming) and the environment, both ex-post and ex-ante. For example, we observed a decrease in the number of small farms when the minimum area for receiving the CAP's Single Farm Payment was increased in 2009, and an increase in the area of semi-natural pastures for grazing livestock farms when the Voluntary Coupled Support to cattle was introduced in 2015 (Fig. 2). However, we cannot be sure if this increase is real or if it is due to more farmers enrolling their pastures in the support system because IACS data only includes areas that receive CAP support. This is a general issue with analyses of agricultural land-use change though, rather than a peculiarity of our approach. There are other challenges posed by the reliance on databases. In the combined analysis (Fig. 7) the values of the TUVA pastures were completely excluded from the field crop type since these were assumed not to be directly related to their food production. This assumption was motivated by the fact that this type had no (or minimal) livestock, but a high proportion of ley and pastures. There are multiple reasons for why farmers in the field crop type keep pastures and leys without livestock. For example, farmers may sell fodder obtained from their leys as horse feed and make their pastures available to a neighbour with cattle, which would allow the farmers to apply for policy payments for managing semi-natural pastures. This assumption may need to be relaxed, depending on the onward usage of the farm typology. The estimated food production and GHG emissions were calculated using a combination of coefficients based on heads of livestock or hectares of farmland which was a simplification of reality. However, we argue that it provides more pertinent information since we use the same method for all farms and only compare differences between farm types. Effects of technology choice and farm size on GHG efficiency could be incorporated in future applications.

The choice of using TUVA and HNV as grassland biodiversity indicators was based on data availability, because these sources are the only ones that cover agricultural systems across entire landscapes and covering all three study regions. TUVA and HNV are not independent from each other since HNV is also including areas in the TUVA database. However, we found only moderate correlation (0.48) between HNV and TUVA coverage at the farm level and therefore argue that it adds additional value to include both data sources. Furthermore, we include HNV

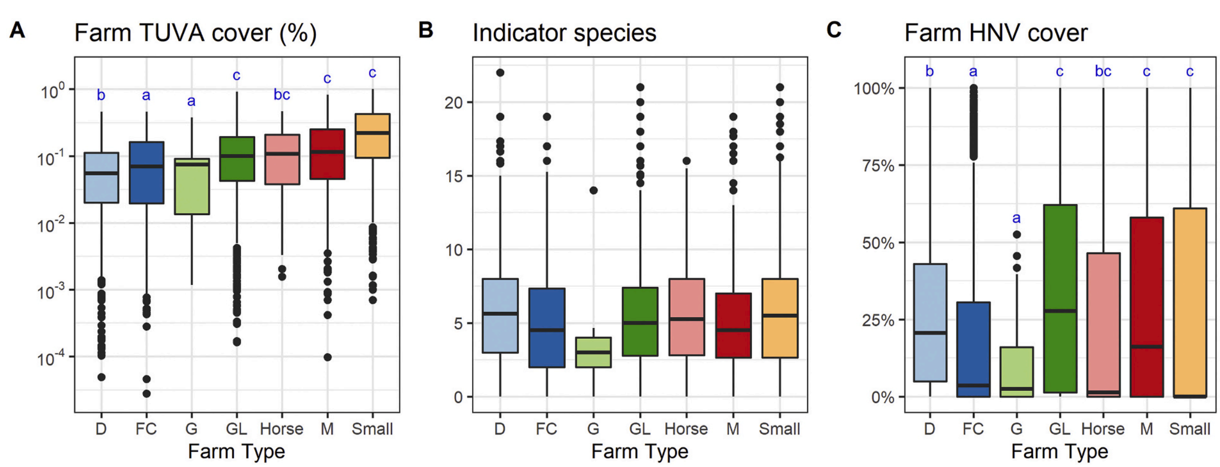

Fig. 5. A: The percentage of TUVA area for each farm type, note the logarithmic scale on the vertical axis B: The average number of indicator species in TUVA parcels per farm. C: The average farm area that had a high nature value in the HNV dataset. (D: Dairy, FC: Field Crop, G: Granivores, GL: Grazing livestock, M: Mixed). Different letters (blue above boxes) indicate significant differences according to the post-hoc tests (based on generalized linear models controlling for regional effects). Panel B shows no significant differences between farm types. The figure is based on data including all areas of all the farms. Note that the exclusion of TUVA areas in the field crop type is not done for this figure which is why its results differ from the combined analysis shown in Fig. 7. Also note that the minimum mapping unit of certain HNV areas might affect the result especially for the smaller farm types.

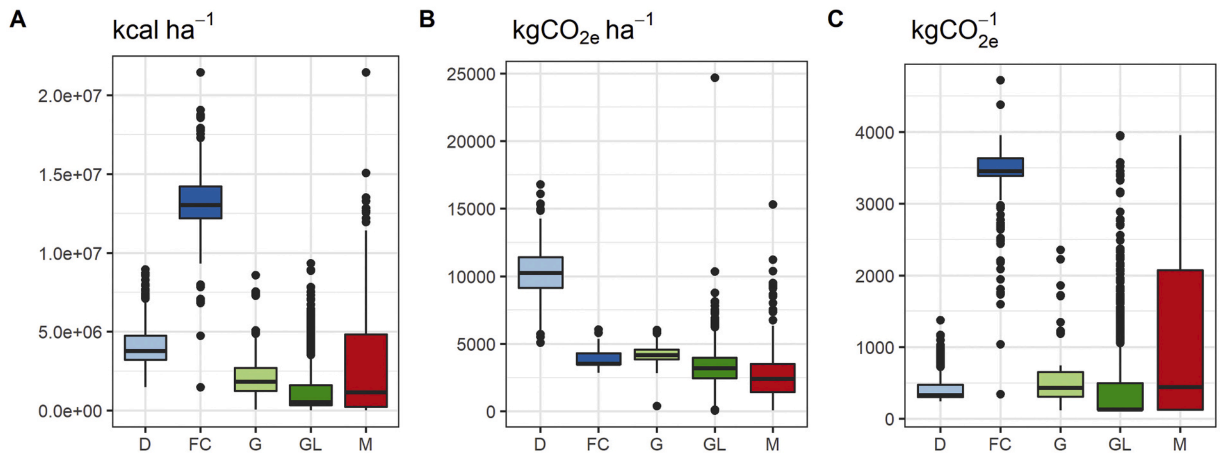

Fig. 6. A: Calories (kcal) per hectare (ha). B: Carbon emissions in carbon dioxide equivalents (kg CO_2e) per hectare (ha). C: Calories (kcal) per emission (kg CO_2e). (D: Dairy, FC: Field Crop, G: Granivores, GL: Grazing livestock, M: Mixed). Note that an increased emission per hectare is a disservice since an increase is representing an additional release of GHG.

as a land cover indicator while TUVA is included also as a quality indicator in terms of number of grassland indicator species of conservation concern. Another limitation of HNV is its minimum mapping unit of 25 ha, leading to an underrepresentation of HNV in farms of the small type (45% of the farms had HNV coverage compared to 70% for the other types, data not shown). However, the small type is not included in the full analysis (food production and GHG emissions) used to draw the main conclusions of this study on differences between farm types. We acknowledge a need for more independent large-scale landscape-scale biodiversity datasets and/or models that have a smaller minimum mapping unit in future studies.

Finally, we note that the use of landscape openness as an indicator for aesthetic values is tailored to the forest dominated regions that we studied (Tahvanainen et al., 1996). As such, landscape openness is not directly transferable to other regions, where aesthetic values could be perceived differently. More generally, we note that the relationships we found between environmental indicators for biodiversity and landscape openness are correlative. This does not preclude the use of the farm typology in applications such as policy assessment, but suggests that the validity of the linkages between farm types and environmental outcomes need to be critically re-assessed in the context of any particular application (Josefsson et al., 2020).The global growth in food demand (Tilman et al., 2011) is hypothesized to continue for decades (Godfray et al., 2010), thereby increasing pressure on agricultural land and exacerbating the current conflict between food production and environmental care. In Sweden, increasing food production to meet greater demand and to increase self-sufficiency is a declared political goal (Swedish government bill 2016/17:104, *A national food strategy for Sweden*). The environmental impacts of increasing agricultural production will, as our study indicates, largely depend on how the mix of production activities develops. Dietary change through reducing meat and dairy product consumption, could reduce the pressure on land from livestock since the efficiency of animals for converting plants to animal matter is around 10% (Godfray et al., 2010). In addition, reducing such consumption would help mitigate climate change since meat/livestock production comes at a cost of higher GHG emissions per kcal produced, as also shown by our results. Röös et al. (2017) find that a shift towards diets without dairy products or meat in the EU would reduce agricultural land use by around 50% and GHG emissions by two-thirds compared to an average diet containing meat. However, converting to a purely plant based diet (field crop farms only) would also reduce grassland biodiversity in Sweden due to a reduction in the area of semi-natural grasslands, which are species-rich habitats maintained by livestock grazing (Stenseke et al., 2016). Our results indicate that this conflict in policy goals could be balanced by shifting from dairying towards an increase in grazing livestock farms (e.g., extensive beef or lamb production), as suggested by Poux and Aubert (2018). This would however imply an additional need for agricultural land to produce the same amount of food, a reduction in landscape openness, and slightly higher GHG emissions (per kcal) compared to dairy farming; but overall generate a better balance between food production and environmental impacts (GHG emissions) when also considering grassland biodiversity values, which is also a declared political goal (*Swedish Environmental Objectives*). However, our analysis reflects the local impacts in terms of local production. Without a matching shift in consumption, both within Sweden and elsewhere, effects may be offset by increasing imports and increasing production of for example dairy products in other regions. Furthermore, our analysis does not analyse potential additional effects of changing land use. For example, a conversion from meadow to crops by ploughing would be associated with additional GHG-emissions (Vellinga et al., 2004).

Payments to farmers are important in marginal areas to prevent abandonment of agricultural land that is crucial for landscape openness and preserving grassland biodiversity, as in the study regions. The main policy instruments to prevent the abandonment of agricultural land in the EU are the direct payments included in CAP Pillar 1 as well as environmental payments and support to areas with natural constraints (known prior to the 2013 reform as "less-favoured areas") in Pillar 2. However, it has been shown that current support levels are too low to guarantee that truly marginal land is kept in production (Brady et al., 2017). Such effects are to some extent reflected in our results as the total agricultural area in the study regions has declined over the study period (Fig. 2). Estimates show that an increment in support in the order of 1000 SEK ha^{-1} $year^{-1}$ for pastures and 400 SEK ha^{-1} $year^{-1}$ for limestone pastures would have kept an additional 130,000 ha of pasture in production in Sweden between 2016 and 2020 (Jordbruksverket, 2017). This would likely have led to a higher usage of marginal land for food production, thereby also benefiting grassland biodiversity. However, it could at the same time lead to a net increase in GHG emissions depending on the type of farming, and compared to land abandonment that would eventually lead to natural forest re-growth or afforestation.

Beyond helping to explain past land-use changes, integrating our farm typology approach with policy evaluation models capable of simulating structural change such as AgriPoliS (Happe et al., 2006; Brady et al., 2012) would facilitate ex-ante analyses of policy proposals for balancing food production with environmental concerns, which is a pressing societal need (Alons, 2017; Pe'Er et al., 2019). Our mapping of farm types enables this by spatially explicit comparisons of food production, GHG emissions and grassland biodiversity outcomes following a policy change for any farm or type. Our farm typology also allows one

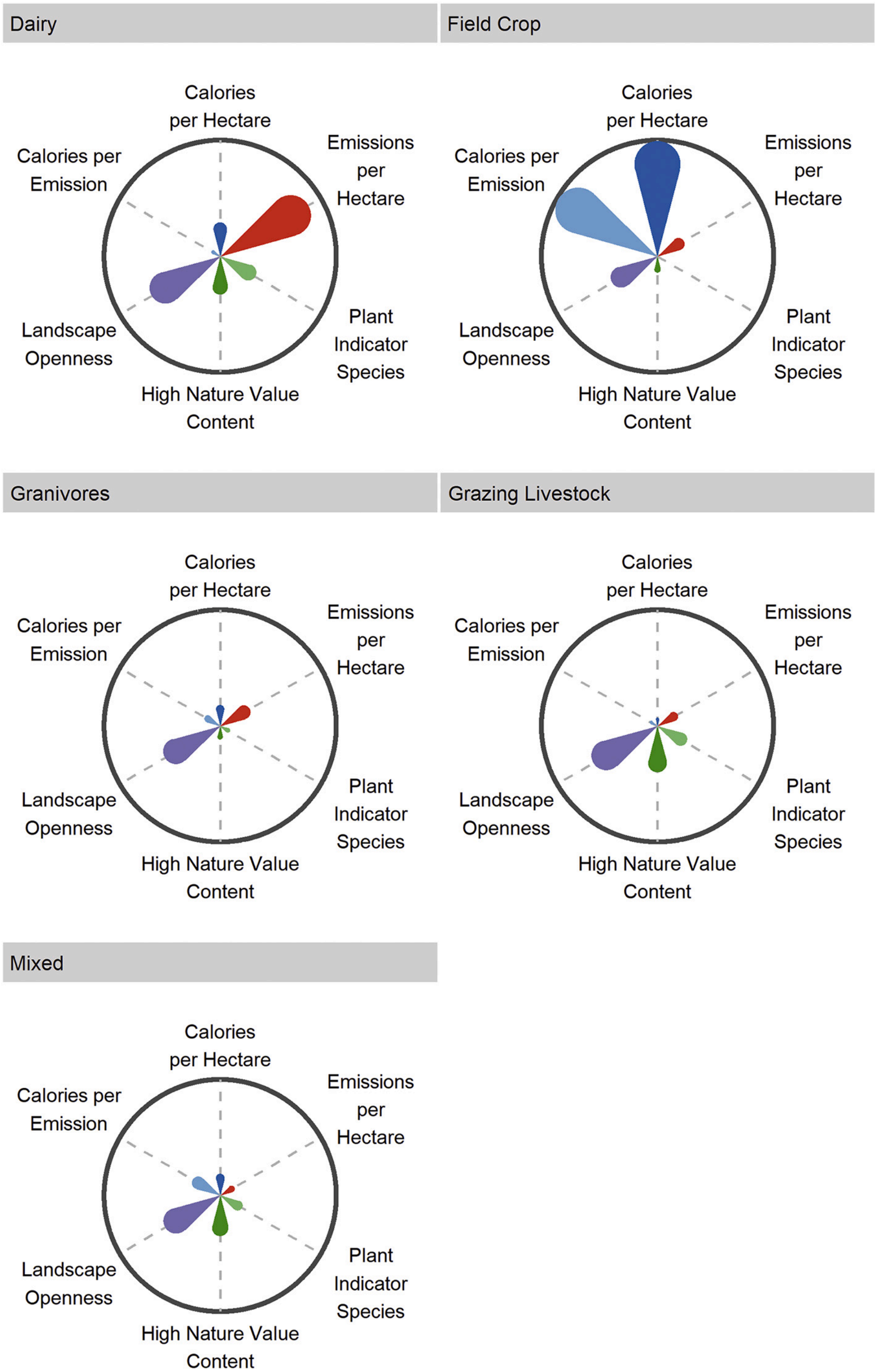

Fig. 7. Comparison of ecosystem services for Dairy, Field Crop, Granivores, Grazing Livestock, and Mixed farms. The circle centre (minimum) represents the 5th quantile and the outer circle (maximum) the 95th quantile of the service. We present unweighted values since the relative importance of different ecosystem services can depend on the stakeholder/reader point of view. Note that an increased emission per hectare is a disservice since an increase represents an additional release of GHG.

to assess the multifunctionality of farming, by relating contrasting production activities to multiple ecosystem services, grassland biodiversity and GHG emissions, as demonstrated here for Swedish forestry-dominated regions. The main policy implication of our results is that there are potentially large societal benefits of reducing land use for dairy farming which would greatly reduce GHG emissions, and instead increasing field crop area for food and grazing livestock to maintain landscape openness and grassland biodiversity. However, the suitability of this change needs also to be evaluated in relation to income forgone for the farmers to ensure that a potential policy change is efficient.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.agsy.2021.103071.

References

Agriwise, 2016. Data Book for Production Planning and Regional Enterprise Budgets (In Swedish). Department of Economics, Swedish University of Agricultural Sciences (SLU), Uppsala.

Alons, G., 2017. Environmental policy integration in the EU's common agricultural policy: greening or greenwashing? J. Europ. Public Policy 24, 1604–1622.

Andersen, E., Verhoog, A.D., Elbersen, B.S., Godeschalk, F.E., Koole, B., 2006. A multidimensional farming system typology. In: SEAMLESS Report No.12. SEAMLESS Integrated Project EU 6th Framework Programme Contract no. 010036-2, p. 30.

Andersson, E., Nykvist, B., Malinga, R., Jaramillo, F., Lindborg, R., 2015. A social–ecological analysis of ecosystem services in two different farming systems. Ambio 44, 102–112.

Auffret, A.G., Kimberley, A., Plue, J., Waldén, E., 2018. Super-regional land-use change and effects on the grassland specialist flora. Nat. Commun. 9, 3464.

Bengtsson, J., Bullock, J., Egoh, B., Everson, C., Everson, T., O'Connor, T., O'Farrell, P., Smith, H., Lindborg, R., 2019. Grasslands—more important for ecosystem services than you might think. Ecosphere 10, e02582.

Benton, T.G., Vickery, J.A., Wilson, J.D., 2003. Farmland biodiversity: is habitat heterogeneity the key? Trends Ecol. Evol. 18, 182–188.

Bhattarai, S., Alvarez, S., Gary, C., Rossing, W., Tittonell, P., Rapidel, B., 2017. Combining farm typology and yield gap analysis to identify major variables limiting yields in the highland coffee systems of Llano Bonito, Costa Rica. Agric. Ecosyst. Environ. 243, 132–142.

Billeter, R., Liira, J., Bailey, D., Bugter, R., Arens, P., Augenstein, I., Aviron, S., Baudry, J., Bukacek, R., Burel, F., 2008. Indicators for biodiversity in agricultural landscapes: a pan-European study. J. Appl. Ecol. 45, 141–150.

Brady, M., Sahrbacher, C., Kellermann, K., Happe, K., 2012. An agent-based approach to modeling impacts of agricultural policy on land use, biodiversity and ecosystem services. Landsc. Ecol. 27, 1363–1381.

Brady, M., Hristov, J., Höjgård, S., Jansson, T., Johansson, H., Larsson, C., Nordin, I., Rabinowicz, E., 2017. Impacts of Direct Payments – Lessons for CAP post-2020 from a Quantitative Analysis. Rapport 2017:2. AgriFood Economics Centre, Lund.

Büttner, G., Feranec, J., Jaffrain, G., Mari, L., Maucha, G., Soukup, T., 2004. The CORINE land cover 2000 project. EARSeL eProc. 3, 331–346.

Cerezo, A., Conde, M.C., Poggio, S.L., 2011. Pasture area and landscape heterogeneity are key determinants of bird diversity in intensively managed farmland. Biodivers. Conserv. 20, 2649.

De Roest, K., Ferrari, P., Knickel, K., 2018. Specialisation and economies of scale or diversification and economies of scope? Assessing different agricultural development pathways. J. Rural. Stud. 59, 222–231.

Díaz, S., Settele, J., Brondízio, E., Ngo, H., Guèze, M., Agard, J., Arneth, A., Balvanera, P., Brauman, K., Butchart, S., 2020. Summary for Policymakers of the Global Assessment Report on Biodiversity and Ecosystem Services of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services.

Dramstad, W.E., Tveit, M.S., Fjellstad, W., Fry, G.L., 2006. Relationships between visual landscape preferences and map-based indicators of landscape structure. Landsc. Urban Plan. 78, 465–474.

Emmerson, M., Morales, M., Oñate, J., Batary, P., Berendse, F., Liira, J., Aavik, T., Guerrero, I., Bommarco, R., Eggers, S., 2016. How agricultural intensification affects biodiversity and ecosystem services. Adv. Ecol. Res. 43–97. Elsevier.

European Commission, 2020. Communication from the Commission to the European Parliament, the Council, The European Economic and Social Committee. A Farm to Fork Strategy for a Fair, Healthy and Environmentally-Friendly Food System.

Food Agriculture Organization of the United Nations, 1997. FAOSTAT Statistical Database. FAO, Rome (c1997-).

Godfray, H.C.J., Beddington, J.R., Crute, I.R., Haddad, L., Lawrence, D., Muir, J.F., Pretty, J., Robinson, S., Thomas, S.M., Toulmin, C., 2010. Food security: the challenge of feeding 9 billion people. Science 327, 812–818.

Guiomar, N., Godinho, S., Pinto-Correia, T., Almeida, M., Bartolini, F., Bezák, P., Biró, M., Bjørkhaug, H., Bojnec, Š., Brunori, G., 2018. Typology and distribution of small farms in Europe: towards a better picture. Land Use Policy 75, 784–798.

Happe, K., Kellermann, K., Balmann, A., 2006. Agent-based analysis of agricultural policies: an illustration of the agricultural policy simulator AgriPoliS, its adaptation and behavior. Ecol. Soc. 11, 49.

Holt, A.R., Alix, A., Thompson, A., Maltby, L., 2016. Food production, ecosystem services and biodiversity: we can't have it all everywhere. Sci. Total Environ. 573, 1422–1429.

Jordbruksverket, 2017. Effektivare kombination av jordbrukarstöden – för ökad miljönytta, lönsamma jordbruk och ökad samhällsekonomisk lönsamhet. Jönköping.

Josefsson, J., Hiron, M., Arlt, D., Auffret, A.G., Berg, Å., Chevalier, M., Glimskär, A., Hartman, G., Kačergytė, I., Klein, J., 2020. Improving scientific rigour in conservation evaluations and a plea deal for transparency on potential biases. Conserv. Lett. 13 (5), e12726.

Junge, X., Lindemann-Matthies, P., Hunziker, M., Schüpbach, B., 2011. Aesthetic preferences of non-farmers and farmers for different land-use types and proportions of ecological compensation areas in the Swiss lowlands. Biol. Conserv. 144, 1430–1440.

Keenleyside, C., Tucker, G., McConville, A., 2010. Farmland Abandonment in the EU: An Assessment of Trends and Prospects. Institute for European Environmental Policy, London.

Landais, E., 1998. Modelling farm diversity: new approaches to typology building in France. Agric. Syst. 58, 505–527.

Leip, A., Billen, G., Garnier, J., Grizzetti, B., Lassaletta, L., Reis, S., Simpson, D., Sutton, M.A., De Vries, W., Weiss, F., 2015. Impacts of European livestock production: nitrogen, sulphur, phosphorus and greenhouse gas emissions, land-use, water eutrophication and biodiversity. Environ. Res. Lett. 10, 115004.

Leventon, J., Schaal, T., Velten, S., Dänhardt, J., Fischer, J., Abson, D.J., Newig, J., 2017. Collaboration or fragmentation? Biodiversity management through the common agricultural policy. Land Use Policy 64, 1–12.

Levers, C., Butsic, V., Verburg, P.H., Müller, D., Kuemmerle, T., 2016. Drivers of changes in agricultural intensity in Europe. Land Use Policy 58.

Levers, C., Müller, D., Erb, K., Haberl, H., Jepsen, M.R., Metzger, M.J., Meyfroidt, P., Plieninger, T., Plutzar, C., Stürck, J., 2018. Archetypical patterns and trajectories of land systems in Europe. Reg. Environ. Chang. 18, 715–732.

Mouchet, M., Paracchini, M., Schulp, C., Stürck, J., Verkerk, P., Verburg, P., Lavorel, S., 2017. Bundles of ecosystem (dis) services and multifunctionality across European landscapes. Ecol. Indic. 73, 23–28.

Nikodinoska, N., Paletto, A., Pastorella, F., Granvik, M., Franzese, P.P., 2018. Assessing, valuing and mapping ecosystem services at city level: the case of Uppsala (Sweden). Ecol. Model. 368, 411–424.

Paracchini, M.L., Petersen, J.-E., Hoogeveen, Y., Bamps, C., Burfield, I., van Swaay, C., 2008. High Nature Value Farmland in Europe. An Estimate of the Distribution Patterns on the Basis of Land Cover and Biodiversity Data. EUR 23480.

Pe'Er, G., Zinngrebe, Y., Moreira, F., Sirami, C., Schindler, S., Müller, R., Bontzorlos, V., Clough, D., Bezák, P., Bonn, A., 2019. A greener path for the EU Common Agricultural Policy. Science 365, 449–451.

Poux, X., Aubert, P.-M., 2018. An Agroecological Europe in 2050: Multifunctional Agriculture for Healthy Eating. Findings from the Ten Years for Agroecology (TYFA) Modelling Exercise, Iddri-AScA, Study.

Queiroz, C., Meacham, M., Richter, K., Norström, A.V., Andersson, E., Norberg, J., Peterson, G., 2015. Mapping bundles of ecosystem services reveals distinct types of multifunctionality within a Swedish landscape. AMBIO 44, 89–101.

Quintas-Soriano, C., García-Llorente, M., Norström, A., Meacham, M., Peterson, G., Castro, A.J., 2019. Integrating supply and demand in ecosystem service bundles characterization across Mediterranean transformed landscapes. Landsc. Ecol. 34, 1619–1633.

Raudsepp-Hearne, C., Peterson, G.D., Bennett, E.M., 2010. Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. Proc. Natl. Acad. Sci. 107, 5242–5247.

Röös, E., Bajželj, B., Smith, P., Patel, M., Little, D., Garnett, T., 2017. Protein futures for Western Europe: potential land use and climate impacts in 2050. Reg. Environ. Chang. 17, 367–377.

SCB, 2000. Jordbrukstatistik sammanställning 2000 (Agricultural statistics 2000). Statistics Sweden, Örebro.

SCB, 2019. Jordbruksstatistisk årsbok 2019 (Yearbook of Agricultural Statistics 2019). Statistics Sweden, Stockholm.

Shepon, A., Eshel, G., Noor, E., Milo, R., 2016. Energy and protein feed-to-food conversion efficiencies in the US and potential food security gains from dietary changes. Environ. Res. Lett. 11, 105002.

Spörndly, E., Glimskär, A., 2018. Betesdjur och betestryck i naturbetesmarker. SLU, inst. för husdjurens utfodring och vård. Rapport 297.

Springmann, M., Clark, M., Mason-D'Croz, D., Wiebe, K., Bodirsky, B.L., Lassaletta, L., de Vries, W., Vermeulen, S.J., Herrero, M., Carlson, K.M., 2018. Options for keeping the food system within environmental limits. Nature 562, 519.

Stenseke, M., Lindborg, R., Jakobsson, S., Sandberg, M., 2016. An Exploration of Swedish Semi-Natural Grasslands.

Swedish Board of Agriculture, 2005. Ängs- och betesmarks- inventeringen 2002–2004. In: Agriculture. Jönköping. S.B.o. (Ed.).

Swedish Environmental Protection Agency, 2020. The National Land Cover Database.

Swedish National Board of Agriculture, 2019. Statistical Database.

Tahvanainen, L., Tyrväinen, L., Nousiainen, I., 1996. Effect of afforestation on the scenic value of rural landscape. Scand. J. For. Res. 11, 397–405.

The Royal Swedish Academy of Agriculture and Forestry, 2015. Forests and Forestry in Sweden, p. 24.

Tilman, D., Balzer, C., Hill, J., Befort, B.L., 2011. Global food demand and the sustainable intensification of agriculture. Proc. Natl. Acad. Sci. 108.

Tveit, M., Ode, Å., Fry, G., 2006. Key concepts in a framework for analysing visual landscape character. Landsc. Res. 31, 229–255.

Tyler, T., Herbertsson, L., Olsson, P.A., Fröberg, L., Olsson, K.A., Svensson, Å., Olsson, O., 2018. Climate warming and land-use changes drive broad-scale floristic changes in Southern Sweden. Glob. Chang. Biol. 24, 2607–2621.

Ustaoglu, E., Collier, M.J., 2018. Farmland abandonment in Europe: an overview of drivers, consequences, and assessment of the sustainability implications. Environ. Rev. 26, 396–416.

Vellinga, T.V., Van den Pol-van Dasselaar, A., Kuikman, P., 2004. The impact of grassland ploughing on CO 2 and N 2 O emissions in the Netherlands. Nutr. Cycl. Agroecosyst. 70, 33–45.

Zabel, F., Delzeit, R., Schneider, J.M., Seppelt, R., Mauser, W., Václavík, T., 2019. Global impacts of future cropland expansion and intensification on agricultural markets and biodiversity. Nat. Commun. 10, 1–10.