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Landscape-scale modelling of agroforestry ecosystems services: A methodological approach

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Abstract: The study explored the ecosystem services supply of agroforestry systems from a landscape perspective by developing a spatially explicit model. Focussed on provisioning and regulating ES, seven indicators (“biomass production”, “water recharge”, “nutrient retention”, “soil preservation”, “carbon sequestration”, “pollination” and “habitat diversity”) were assessed. The model was applied on a traditional agroforestry system (a cherry orchard) in Switzerland, as a case study. Eight landscape test sites of 1km x 1km, four with agroforestry and four with agriculture were mapped, and used as baseline for the model. We found that the provisioning ecosystem services, namely the biomass production, was higher in landscape test sites with agriculture; while the regulating ecosystem services were better represented in landscape test sites with agroforestry. The differences were found to be statistically significant for biomass yield, water recharge rate, nitrate leaching, carbon sequestration and proportion of semi-natural habitats.

Keywords: biodiversity, biomass production, carbon sequestration, erosion, groundwater recharge, nitrate leaching, pollination

1 Introduction

Agroforestry systems are traditional man-made agricultural land use practices, combining woody perennials with agricultural crops and / or animals to provide food, fodder and timber from the same field at the same time (Somarriba 1992). Agroforestry systems are widespread in Europe and currently cover around 15.4 million hectares, which is approximately 8.8 % of European agricultural land (den Herder et al. 2017). At the same time as providing a range of products, the systems can also offer many environmental benefits and help improve biodiversity (e.g. Nair 2012; Moreno et al. 2016).

In 2003 the Millennium Ecosystems Assessment (MEA) identified the extent and significance of the “benefits people obtain from ecosystems” in form of ecosystem services (ES) (MEA 2003). Since then, research investigations has aimed to define (e.g. Boyd & Banzhaf 2007; Braat & de Groot 2012), map (e.g. Burkhard et al. 2012; Clec’h et al. 2016) and assess these ES (e.g. Syswerda & Robertson 2014; Maes et al. 2016). Increasingly, understanding the spatial allocation and quantification of ES (Burkhard et al. 2013) and the effect of spatial pattern on bundles of ES (Raudsepp-Hearne et al. 2010) has become more significant.

Landscape patterns, especially in agriculture are formed by land cover, land management and spatial structures (Verburg et al. 2013) and are accountable for the function and supply of ES (Englund et al. 2017). Biodiversity rich areas can (but do not always) coincide with areas that are important for ES (Anderson et al. 2009). For this reason, Tscharntke et al. (2005) postulated that biodiversity assessments need to cover larger areas and require a landscape perspective.

The role of agroforestry systems in providing ES in temperate Europe has been investigated in several studies and these were summarized by Torralba et al. (2016). The key ES were (1) timber, food and biomass production, (2) soil fertility and nutrient cycling, (3) erosion control and (4) biodiversity provision. However, these investigations were mainly restricted to single services (Udawatta et al. 2008; Pumariño et al. 2015). A spatially explicit and systematic assessment of how heterogeneous agroforestry systems affect ES at a landscape scale, across time and space, is missing. However, computer simulation models can help to evaluate the long-term effects of different land use systems (Jose and Pallardy 2004).

The aim of this study was to assess and comprehensively quantify a bundle of ES of agroforestry systems with a semi-quantitative approach at the landscape scale through a combination of field investigations and modelling, using a tradition cherry orchard in Switzerland as case study region. In undertaking this evaluation, we first selected indicators that could be used to characterise the performance of agroforestry, agricultural, and forest systems. Then, algorithms for quantifying these indicators were identified, tested, adapted, and applied. Finally, we used the algorithms to compare ES provision from agroforestry (AF) and non-agroforestry (NAF) landscapes.

2 Data and Methods

2.1 Study area

The study was conducted in traditional high-stem cherry orchards in north-western Switzerland. The case study region covers 50 km^2 with seven municipalities and is characterised by an annual average temperature of 7.7 ^{o}C, and annual rainfalls of 800-1000 mm (Figure 1).

The evaluated system consists of around 80 cherry trees ha^{-1} of different ages on grassland, providing cherries, grass and timber.

<Figure 1>

2.2 Selection of Landscape Test Sites

Within our study, we subdivided the case study area into the broad land cover categories forestry, agroforestry and agriculture (mainly arable). Within the agroforestry (AF) and agricultural (non-agroforestry (NAF)) areas identified, we chose four landscape test sites (LTS) of 1 x 1 km, at random within each category, resulting in eight LTS altogether. In each LTS habitats and trees were mapped in the field (see Annex 1 for the habitat mapping protocol) during spring 2015 and 2016. For grassland, percentage cover of grass, clover and herbs was recorded. For woody perennials, the location, tree species, height and structure were recorded during field surveys. Single trees and AF trees were digitized from aerial photographs and classified by crown diameter as small (young), medium (middle age) and large (old) (Figure 2b).

<Figure 2>

The location of arable and other land was also identified and mapped and all information was combined in a habitat map and digitized using ArcGIS 10.4 (Figure 2a). It should be noted that it was not possible to find entire LTS under AF and NAF and therefore each LTS often included a mix of arable, grass, forestry, agroforestry and other (urban) land covers. However, agroforestry dominated (31 - 55 %) the land cover in the AF LTS and arable land dominated (19 - 72 %) the land cover in the NAF LTS (Table 4).

2.3 Indicator and model selection

A range of indicators were selected to compare ES delivery in the AF and NAF LTS. The selected indicators included those that (a) were listed as ES in the Common International Classification of Ecosystem Services (CICES) with focus on provisioning and regulating services (Haines-Young and Potschin 2013); (b) addressed differences of AF versus NAF (Alam et al. 2014; Torralba et al. 2016), and (c) could be evaluated by existing and available models adapted to agroforestry (Table 1).

<Table 1 >

2.4 Conceptual background

A spatially explicit ES evaluation model, shown in Figure 3, was developed, which comprised the selected indicators (Table 1) and also accounted for their interaction. In order to consider the spatial dependence location-dependent variables (habitat map, soil map, digital elevation model, and climate) were used to calculate each indicator. Model outcomes were ES maps (resolution 2 x 2 m) (Figure 4a), wherein each pixel contained the information for all indicators and specified the relationship to that specific location.

<Figure 3>

The indicator values (separately for each indicator) were then aggregated at a landscape scale for each LTS and quantified as mean per hectare values for the whole LTS area (including agricultural, forest and urban area).

2.4.1 Biomass production

Biomass production is the key indicator of use for assessing agricultural productivity and its economic value. It is spatially affected by regional growing conditions such as soil and climate and in turn, shapes the agricultural landscapes (Verburg et al. 2013; van der Zanden et al. 2016).

The total stock value at any one time (t DM ha^{-1}), and the annual use (t DM ha^{-1} yr^{-1}) of the biomass was calculated separately for agricultural, forestry and agroforestry systems (Table 2). No distinction was made regarding the type and quality of biomass.

The biomass yield for arable farming was based on Swiss agricultural statistics (BAFU 2013), grassland yield was from Baer (2009), and forest yields were from Swiss forest statistics (Brändli 2010, BAFU; BfS 2015a; BAFU; BfS 2015b).

The EcoYield-SAFE model (Palma et al. 2017) a daily time-step model developed from the YieldSAFE model (van der Werf et al. 2007), was calibrated to local monoculture yields of trees, crops, and grass, using the approach described by Graves et al. (2010), and then used to calculate the agroforestry biomass yield.

The calibration data for EcoYield-SAFE were derived from field measurements, the literature described above, local soils data, and weather data for a rotation of 60 years. The field data included tree and crown diameter, flowering time and fruit yield data. Information for grass yield, 6.0 t ha^{-1} for high-input grassland and 2.0 t ha^{-1} for low-input grassland, came from the literature and local farmers (Baer 2009). The fruit yields from the cherry trees were assumed to be 50 kg (16 % DM) per tree for small trees, 100 kg per tree for medium sized trees, and 150 kg per tree for large trees in years with good weather conditions (Windisch 1895). CliPick (Palma, 2017) provided daily data (on e.g. precipitation, temperature and solar radiation). Soil parameters were taken from the description of hydraulic properties of European soils (Wösten et al. 1999; Hiederer 2013a). For an agroforestry system of 80 cherry trees ha^{-1}, EcoYield-SAFE predicted 130 t of biomass in year 60, with a mean yield of 2.16 t ha^{-1} yr^{-1}. The mean fruit production was 16 t ha^{-1} yr^{-1} and grass production in between trees declined from approximately 6 t DM ha^{-1} in the first years to 2 t DM ha^{-1} in as the trees got older.

The habitat classification and tree diameter were then used to link the model outcomes to the LTS habitat map.

< Table 2>

2.4.2 Groundwater recharge

An intact water cycle is required to provide adequate water for biomass production and for human settlement. Sufficient availability of groundwater, and sustainable groundwater recharge, is needed for domestic and industrial purposes. These processes are directly linked to land cover, land management and landscape structure. The general water equation was given a:

$$P = E + R + \Delta S \quad \text{with} \quad \Delta S = (\Delta S_{Soil} + \Delta S_{Groundwater\ recharge}) \quad (\textit{Equations 1, 2})$$

Where P was the precipitation, E was the evapotranspiration, R was the surface runoff and ΔS was the storage change in soil (ΔS_{soil}) and groundwater ($\Delta S_{Groundwater\ recharge}$). Water flows were modelled by using FAO's CROPWAT 2.0 for crop performance indices (Allen et al. 1998) in combination with the spatial components of MODIFFUS 3.0 method (Hürdler et al. 2015). Our focus was on the amount of groundwater recharge and leachate as ES.

The water cycle was calculated in six steps: (I) P as mean annual precipitation in mm was derived from the CCM River and Catchment Database compiled by the European Commission and Joint Research Centre for the years 1975 to 1999 (Vogt et al. 2007). The data were interpolated on a 1 km grid. (II) E was estimated by multiplying effective rainfall and reference evapotranspiration coming from the FAO CROPWAT 2.0 model as monthly values and leading to a discrete crop evapotranspiration (ETC) using the Penman-Monteith (FAO-56 PM) method (Allen et al. 1998). (III) R was modelled using the MODIFFUS 3.0 (Hürdler et al. 2015), which incorporated slope as derived from the digital elevation model SwissALTI3D (swisstopo 2012), land use characteristics and water catchment areas (Vogt et al. 2007). (IV) ΔS was divided into ΔS_{Soil} and $\Delta S_{Groundwater\ recharge}$. ΔS_{Soil} was obtained combing data on the total available water content (TAWC) for topsoil in the European Soil Database (ESDB) (Panagos et al. 2012; Hiederer 2013a, b), storing and filtering capacity in Makó et al. (2017) and the available water content (AWC) in Ballabio et al. (2016). (V) The soil water balance was calculated to provide the $\Delta S_{Groundwater}$

$_{recharge.}$ (VI) The groundwater recharge rate (*GWRR*) represented the proportion of precipitation percolating to the groundwater (Equation 3) and was given as:

$$GWRR = \frac{\Delta S_{Groundwater\ recharge}}{P} * 100 \qquad \text{(Equation 3)}$$

2.4.3 Nutrient retention

The focus of this ES is on nitrogen and phosphorus losses. Nitrogen inputs are often used as proxy for management intensity in agriculture (e.g. Tieskens et al. 2017). Rising N application causes N losses to the environment (Sutton et al. 2011) and leads to pollution of groundwater and eutrophication of natural ecosystems (Velthof et al. 2014). Phosphorus losses affect surface waters through erosion and surface runoffs (Schoumans et al. 2014).

The nutrient loss assessment was based on MODIFFUS 3.0, an empirical model for nitrate and phosphorus losses in Switzerland (Hürdler et al. 2015). In this, leaching values for each land cover class were weighted by factors for soil characteristics, fertilizer application, grassland management, denitrifcation and drainage. N loss was then multiplied by total leachate, and P loss was multiplied by runoff water, both of which were calculated in the water cycle assessment.

2.4.4 Soil preservation

A major indicator of effective disturbance reguation is soil erosion. This indicator was assessed using the Revised Universal Soil Loss Equation (RUSLE) (Renard et al. 1997) (Equation 4), defined as:

$$A = R * K * L * S * C * P \qquad \textit{(Equation 4)}$$

Where: A was the estimated mean annual soil loss (t ha^{-1} yr^{-1}), R was a rainfall-runoff erosivity factor, K was a soil erodibility factor, LS was a slope-length factor, C was a cover-management factor and P was a practice-management factor.

The European digital map was used for retrieving the *R* (Panagos et al. 2016) and *K* factors (Panagos et al. 2014). The slope-length factor was calculated using the System for Automated Geoscientific Analyses (SAGA) (Olaya 2004; Conrad et al. 2015) with digital elevation data from SwissALTI3D, which has a spatial resolution of 2 m. The *C* factor was defined for each habitat according to Panagos et al. (2015) and Hürdler et al. (2015). The *P* factor was set to 1, as no special supporting practice was used (Panagos et al. 2015).

2.4.5 Carbon stock and sequestration

Retention of carbon (C) stocks and sequestration of atmospheric carbon are increasingly important in climate regulation (Lal 2004). According to the UNFCCC definition, carbon must be stored in long-lived pools to be effectively sequestered (UNFCCC 2007) and come from atmospheric CO_2 to contribute to climate regulation (Powlson et al. 2011). Our assessment of biomass carbon storage is based on the produced above and below ground biomass estimated in EcoYield-SAFE.

The estimation of root biomass and soil organic carbon (SOC) in agroforestry systems is challenging (Lorenz and Lal 2014). Many models, such as C-TOOL (Taghizadeh-Toosi et al. 2014) and RothC (Coleman and D.S. Jenkinson 1996) are limited to agricultural systems and could not be applied here. We used Yasso07 (Liski et al. 2005) which has been developed for evaluating tree and forest system, and assessed for afforestation, agroforestry, grassland, and coppice systems (Masera et al. 2003; Prada et al. 2016). The Yasso07 model was able to address the decomposition of biomass fractions, and their effects on soil carbon, and simulated the stock, annual change, and releases of carbon to the atmosphere based on site specific climate and stand information. The carbon content of trees was assumed to be 50% of tree biomass (Aalde et al. 2006) and carbon storage was divided into an annual sequestration rate and a permanent carbon stock. Carbon storage was divided into annual and permanent; respectively known as carbon sequestration rate (t C ha^{-1} yr^{-1}) and carbon stock (t C ha^{-1}). According to Keel et al. (2016) the carbon inputs in agricultural land range from 1.6 to 2.6 t C ha^{-1} yr^{-1}. Analysis results of Yasso07 for permanent grassland ranged from 10 to 20 t C ha^{-1} carbon stock and a sequestration rate of around 1 t C ha^{-1} yr^{-1}. An agroforestry system (80 trees, 60 years) produced 63.8 t C in biomass and 5.76 t C in soil, which corresponds to 1.3 t C ha^{-1} yr^{-1}.

2.4.6 Biodiversity

Pollination is essential in the regulation of the biotic environment. Pollinators require nesting and foraging resources in adequate quantity, quality and within reachability (Schulp et al. 2014). Flowering grasslands, insect-pollinated crops, and blossoming trees, provide nectar and pollen for pollinators (Schindler and Peters 2011; Pasquet et al. 2014). Here, the pollination indicator was developed using the Lonsdorf model (Lonsdorf et al. 2009; 2011), which estimates the habitat nesting suitability, the habitat flowering suitability and the reachability between these two.

For wild bees the nesting capacity was evaluated for both ground and cavity nesting. Ground nesting facilities were mapped in the field. Cavity nesting potential was assumed to be present in all habitats with woody elements such as hedgerows, agroforestry, and forest. The flowering potential was mapped, using the quantity of clover and herbs in grasslands, crops pollinated by insects (mainly rapeseed and horticultural crops) and blossoming trees.

The reachability between nesting and flowering sites depended on the pollinator species, their moving and foraging radius, and the ranges available between a distance of 100 m (e.g. *Megachile rotundata*) and 1500 m (e.g. *Apis mellifera)*. Nearby floral resources were preferred to distant foraging resources (Zurbuchen et al. 2010). To model the polinator index for range of pollinators, three moving corridors (100, 350, 500 m*)* were computed for the two nesting types.

The structural diversity of agroforestry systems contributes to the proportion and variety of semi-natural habitats for flora and fauna (Pointereau et al. 2007; Torralba et al. 2016) and several investigations have shown a relationship between landscape pattern and agricultural biodiversity (Bailey et al. 2007; Billeter et al. 2008).

Bailey et al. (2007) tested different metrics and proposed the Simpson Diversity Index (*SIDI*) (Equation 5) for fine scale and heterogeneously defined biological groups, which was defined as:

$$SIDI = 1 - \left(\frac{\sum n(n-1)}{N(N-1)}\right) \qquad \textit{(Equation 5)}$$

Where: n wass the number of habitat patches and N wass the total number of habitat types.

Billeter et al. (2008) reinforced this point and found that the share of semi-natural habitat (SoSNH) and habitat diversity (HD) correlated strongly with the species richness of several taxa. The share of semi-natural habitat was given by Equation 6:

$$SoSNH = \frac{SNH*100}{A} \qquad \textit{(Equation 6)}$$

Where: SNH wass the area in m^2 of semi-natural habitat types of the study site and A wass the size of the study site in m^2.

Habitat diversity (HD) (Equation 7) was the sum of the semi-natural habitat types (ToSNH) in the study site.

$$HD = \sum ToSNH \qquad \textit{(Equation 7)}$$

These indicators were computed using the habitat maps. They indicated relative levels of habitat and – potentially – species diversity in the case study region.

2.5 Spatial and statistical analysis

The statistical analyses were performed as an ANOVA with R (R Development Core Team 2013). The spatial analysis of the ES indicators was developed in QGIS 2.14.8 (QGIS Development Team 2015), SAGA System for Automated Geoscientific Analyses (Conrad et al. 2015) and ESRI ArcGIS10.4 (Environmental Systems Resource Institute 2016) using the datasets in Table 3. The ES indicator values were then compared in R (R Development Core Team 2013) using ANOVA to determine whether significant differences in ES delivery existed between the AF and NAF land covers.

<Table 3>

3 Results

3.1 LTS inventory

Altogether 23 different habitat types and 8,189 trees were recorded across the eight LTS. Figure 4a shows the results of the habitat mapping presenting all eight LTS and Table 4 summarizes the percentage of area occupied by forestry, agroforestry, agriculture and other land use types in each LTS. A detailed analysis of these data is presented below.

< Table 4 >, *< Figure 4 >*

3.2 Biomass production

The modelled biomass yields are shown in Figure 4b for the eight LTS. Across the LTS, mean annual biomass yields were found to be greater in NAF (6.5 t ha^{-1}) landscapes than in AF landscapes (4.6 t ha^{-1}). This effect was statistically significant ($p < 0.01$). However, in contrast, the biomass stock tended to be greater in AF LTS due to the tree biomass.

3.3 Groundwater recharge

The mean annual precipitation in the study area was 990 mm. In the AF LTS, 46.5 % of this was used by evapotranspiration, 2.7 % was remove from the area as surface runoff, and 49.5 % was used for groundwater recharge and soil storage. In NAF LTS the overall fate of precipitation was comparable, with an evapotranspiration taking 44 %, surface runoff taking 3.2 %, and groundwater recharge and soil storage taking 53 %. The groundwater recharge rate was 44.5 % in AF and 49 % in NAF result that was significantly different ($p<0.027$).

3.4 Nutrient retention

The assessment of nitrate leaching (Figure 4c) showed relatively high losses of nitrates associated with LTS with larger arable areas, such as NAF2 and NAF3 (>25 kg N ha^{-1} yr^{-1}). The overall nitrate loss was 13.8 kg N ha^{-1} yr^{-1} in NAF LTS, and significantly higher ($p<0.008$) than in AF LTS (7.6 kg N ha^{-1} yr^{-1}). The phosphorus loss in both AF and NAF LTS was below 1 kg P ha^{-1} yr^{-1} and is no longer accounted for.

3.5 Soil preservation

The average soil loss is 1.88 t ha^{-1} yr^{-1} in AF and 1.46 t soil ha^{-1} yr^{-1} in NAF LTS. These results were not found to be statistically significant between the two land covers.

3.6 Carbon stock and sequestration

The mean carbon sequestration rate was 0.49 t C ha^{-1} yr^{-1} in NAF and 0.75 t C ha^{-1} yr^{-1} in AF LTS, which was significantly higher ($p < 0.01$). The maps in Figure 4d show that this effect was largely due to the high areas of arable land in NAF landscapes, such as found in NAF1, 2 and 3, whereas AF landscapes, such as AF1, 2 and 3, showed relatively high carbon sequestration rates associated with the agroforestry habitat. The mean carbon stock was also relatively high in AF LTS at 59.6 t C ha^{-1} (53.3 – 67.6 t C ha^{-1}) compared with 51 t C ha^{-1} (43.6 – 64.1 t C ha^{-1}) in NAF LTS but the differences were not found to be statistically significant.

3.7 Biodiversity

The AF LTS provided greater resources for pollinators. A mean area of 66.3 ha in the AF LTS was mapped as potential habitats for ground nesting pollinators, 44.8 ha for cavity nesting pollinators, and 21.8 ha provided flowering potential. In NAF LTS these figures were reduced, with 46.2 ha having ground nesting potential, 31.6 ha having cavity nesting potential, and 14.3 ha providing flowering potential.

Figure 5 shows the pollination service at different distances from nesting resources. Within a radius of 100 m of a nesting facility, results showed that a larger area of land could be reached by pollinators in AF LTS (97.5 % for cavity species, 98.8 % for ground nesting species) than in NAF LTS (84 % and 93 %, respectively). For cavity nesting species, these differences were significant ($p<0.1$), but no correlation was found for ground nesting species. At flying distances of 350 m and more, the total area could be accessed by both cavity and ground nesting species.

<Figure 5 >

The assessment of habitat and species richness was based on the landscape metrics SIDI, SoSNH and HD. In general, the AF LTS showed the biggest range of values for these indicators. The habitat diversity indicator SIDI ranged from 0.82 to 0.88 in AF and 0.85 to 0.89 in NAF and was similar across all the LTS. The share of semi-natural habitats, SoSNH*,* was much greater in AF LTS (27 - 49 %) than in NAF LTS (5 - 11 %). This difference was highly significant ($p > 0.001$). The number of semi-natural habitat types HD was between 35 to 84 in AF LTS and between 16 to 35 in NAF LTS, and was also found to be significantly different ($p<0.1$).

3.8 Summary of indicator values

Figure 6 provides a summary of the results using normalized indicators between -1 (for losses) and 1 (for gains). Statistically significant differences between the AF and NAF LTS, and p values are shown for each of the indicators.

<Figure 6>

4 Discussion

4.1 Modelling approach

The assessment of ES has taken place at various scales in previous research, ranging from broader assessments at national or international scales (e.g. Kienast et al. 2009; Mouchet et al. 2017), focusing on multiple ES and linking them to land-use types, or detailing field scale assessments modelling systems and processes (e.g. Tsonkova et al. (2014).

Our study fits in between these two perspectives. On the one hand, a reasonable level of detail is needed to address spatial effects of tree and crop interaction in agroforestry, while on the other hand we want to investigate effects of this particular land use system at the landscape scale. Therefore, we tried to balance the methodology between model complexity, data requirements and total error (see e.g. Schröter et al. 2014) with a spatial resolution of 1 x

1 km for a landscape test site (LTS). At this scale both aims, agroforestry assessment itself and their impact on landscape, could be evaluated and visualised.

4.2 Biomass production

Due to the broad product portfolio of agroforestry systems with timber, food and fodder production, we summarized biomass production into total values (see also Tsonkova et al. (2014); Fader et al. (2015) to make our assessment

The annual biomass yield was lower in LTS with AF than in NAF LTS, but the biomass stock in AF, mainly timber, was higher than in NAF. This was due to the difference rotation length between the annual crops and trees. When the AF and NAF LTS were compared over the rotation length of trees (60 to 80 years), greater total productivity was achieved for AF LTS compared with NAF LTS. Such results have been reported in previous research (e.g. Sereke et al. 2015), where growing trees and crops together can be more productive.

4.3 Groundwater recharge

Fine scale spatial variability in soil, climate, elevation and land use impede modelling approaches, in particular for groundwater recharge (Herrmann et al. 2009; Tetzlaff et al. 2015). Consequently, models contain a high degree of uncertainty. Therefore, we used the general water equation and subdivided it into tangible parts. Alexandris, Stricevic, & Petkovic (2008) found that the procedure outlined by the FAO (Allen et al. 1998) provides scientifically the most valid results. Soil maps with a spatial resolution of 25 to 1.000 m were used (Hiederer 2013a, b). This level of spatial resolution limits small scale analysis, yet it allows to generate trends for habitat types and to aggregate results on LTS level.

4.4 Nutrient retention

In the AF LTS, modelled nitrate losses were nearly half of those in NAF LTS pointing to a clear ES benefit in terms of reduced nitrate emissions to the environment. This reflects similar findings from e.g Nair et al. (2007) and Jose (2009), who showed that agroforestry systems can help reduce nutrient losses by 40 and 70%. López-Díaz et al. (2011) showed in greenhouse experiments that trees have a higher root density and a deeper root horizon, which led to a higher uptake of nitrate and a reduction of nitrate leaching of 38 to 85 %.

4.5 Soil preservation

Agroforestry systems are closely linked to soil conservation and management practices (Young 1997) and are often seen as a means of reducing soil erosion (Wezel et al. 2014). The RUSLE equation is based on empirical data, and has been tested in different regions at several scales (e.g. Panagos et al. 2015, Prasuhn et al. 2013). The soil loss calculated for our eight LTS was close to the mean value (2.46 t ha^{-1} yr^{-1}) calculated for Europe and long-term monitoring values (0.9 - 1.24 t ha^{-1} yr^{-1}) measured on agricultural land in the lowlands of Switzerland. No significant difference in soil erosion between AF and NAF LTS was found. This is different to former studies, where agroforestry systems have been shown to reduce soil erosion (Rodríguez-Ortega et al. 2014; Sánchez & McCollin 2015, Palma 2007). However, it is worth noting that in our LTS, topographical differences could mask the soil reduction benefits associated with agroforestry systems since the cherry systems occurred on steeper land than arable uses (20 % slope for AF LTS as compared to 9 % for NAF LTS).

4.6 Carbon stock and sequestration

Our results were similar to results reported by Cardinael et al. (2015) in agroforestry plots in France, who detected a below ground carbon sequestration of 0.09 to 0.46 t C ha^{-1} yr^{-1} and an above ground carbon sequestration of 0.004 to 1.85 t C ha^{-1} yr^{-1} in the tree biomass. Higher carbon sequestration rates have been reported for young plantations. Nabuurs and Schelhaas (2002) modelled the sequestration rate of 16 forest types and reported a mean sequestration rate of 2.98 t C ha^{-1} yr^{-1}, ranging from 1.15 to 4.1 t C ha^{-1} yr^{-1} for afforestation. For long established forest sites the sequestration rate reached a steady state at 0.8 t C ha^{-1} yr^{-1} with a range 0.3 to 1.4 t C ha^{-1} yr^{-1}. The carbon stock reported in these forests was between 52 and 196 t C ha^{-1}. In the eight LTS, carbon stocks were at the lower end of this range and similar for both AF and NAF LTS. The greatest carbon stocks in the LTS were found in the forests, which provided 175 t C ha^{-1}, well above that found in agroforestry (88 t C ha^{-1}) and agriculture (3 t C ha^{-1}) land uses. Whilst the differences between our AF and NAF landscapes were relatively small due to the composition of the LTS, both of which included substantial areas of forest, the results obtained from modelling the per hectare values suggest that the greatest sequestration benefit would be provided from forests, although use of agroforestry systems would provide some carbon sequestration benefit whilst allowing food production to continue.

4.7 Biodiversity

Assessing biodiversity is challenging because of its inherent complexity that cannot be represented completely by only a single or even multiple indicators (e.g. Herzog et al. 2013). Still, biodiversity plays a key role on all levels of ES assessment (Mace et al. 2012). Therefore, we opted for two types of indicators: “Pollination” stands for the multiple functions (or services) that wild species provide for agricultural production and the three indicators representing “habitat richness” stand for the conservation of species and habitat diversity.

Previous studies assessing pollination services (e.g. Kennedy et al. 2013, Schulp et al. 2014) highlighted the importance of woody elements in landscapes. Suitable nesting habitats and flowering resources were more present in AF LTS then in NAF LTS. However, the size of the LTS (1 x 1 km) limited any effect this had on the pollination service, because the moving corridors of the pollinators were larger than the LTS themselves. Three sizes of moving corridors were assessed, but differences between AF and NAF LTS only became significant at the 100 m level. This suggested that the living conditions for pollinators would be satisfied in both landscapes, but that fitness, resilience and resistance of each individual might be greater in the AF LTS.

The indicator group for “habitat richness” was derived from landscape metrics computed from the habitat maps. The SIDI was found to be statistically similar in both AF and NAF LTS, because the index is largely driven by the number of habitat types. In fact, it was slightly greater in NAF LTS, because different crop types were counted as different habitats. The SoSNH indicators was slightly higher in AF LTS because the traditional fruit orchards were assumed to qualify as semi-natural habitats in accordance with literature (Herzog et al. 2017) and with national regulations, where orchards are classed as ecological focus areas. The HD indicator was similar across all LTS, although a wider range of habitats (6 - 40) occurred in the AF LTS. Statistically significant differences between the AF and NAF LTS were found two of the four indicators for biodiversity, suggesting that biodiversity might be better supported in AF LTS than in NAF LTS. Studies by Birrer et al. (2007), and Bailey et al. (2010) support this conclusion since they have shown that fruit orchard landscapes in temperate Europe have relatively high species richness as well as specialised species such as orchard birds.

5 Conclusion

Our study explored the ecosystem services supply from agroforestry systems from a landscape perspective by developing a spatially explicit model. Seven indicators (biomass production, water recharge, nutrient retention, soil preservation, carbon sequestration, pollination and habitat diversity) were chosen to represent these ES. The model was applied to traditional agroforestry system, cherry orchards, in Switzerland, as case study.

We found that the provisioning ES was higher in LTS without agroforestry while the regulating ES were higher in LTS with agroforestry. The differences were significant for annual biomass production, water recharge rate, nitrate leaching, carbon sequestration and the share of semi-natural habitats. These findings have implications for decision making, particularly given the many benefits that stakeholders increasingly seek from land.

To our knowledge, this is the first attempt to comprehensively quantify ecosystem services with a semi-quantitative approach at the **landscape scale** through a combination of field investigations and modelling. It thus goes beyond expert evaluations and modelling results. The resulting indicator values were plausible and within the range of values published in former studies. The approach is limited by the availability of spatial data (notably high-resolution soil maps) and by the state of the art of modelling, which reflects our current understanding of the relevant processes. However, this approach provides an example for spatially explicit quantification of provisioning and regulating ES and is suitable for comparing different land use scenario at a landscape scale.

Acknowledgements

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References

Aalde H, Gonzalez P, Gytarsky M, et al (2006) Chapter 4: Forest Land. In: 2006 IPCC Guidelines for National Greenhouse Gas Inventories.

Alam M, Olivier A, Paquette A, et al (2014) A general framework for the quantification and valuation of ecosystem services of tree-based intercropping systems. Agrofor Syst 88:679–691. doi: 10.1007/s10457-014-9681-x

Alexandris S, Stricevic R, Petkovic S (2008) Comparative analysis of reference evapotranspiration from the surface of rainfed grass in central Serbia , calculated by six empirical methods against the Penman-Monteith formula. Eur Water 21/22:17–28.

Allen RG, Pereira LS, Raes D, et al (1998) Crop evapotranspiration-Guidelines for computing crop water requirements-FAO Irrigation and drainage paper 56.

Anderson BJ, Armsworth PR, Eigenbrod F, et al (2009) Spatial covariance between biodiversity and other ecosystem service priorities. J Appl Ecol 46:888–896. doi: 10.1111/j.1365-2664.2009.01666.x

Baer U (2009) Erträge auf Wiesen und Weiden.

Bailey D, Billeter R, Aviron S, et al (2007) The influence of thematic resolution on metric selection for biodiversity monitoring in agricultural landscapes. Landsc Ecol 22:461–473. doi: 10.1007/s10980-006-9035-9

Bailey D, Schmidt-Entling MH, Eberhart P, et al (2010) Effects of habitat amount and isolation on biodiversity in fragmented traditional orchards. J Appl Ecol 47:1003–1013. doi: 10.1111/j.1365-2664.2010.01858.x

Ballabio C, Panagos P, Monatanarella L (2016) Mapping topsoil physical properties at European scale using the LUCAS database. Geoderma 261:110–123. doi: 10.1016/j.geoderma.2015.07.006

Billeter R, Liira J, Bailey D, et al (2008) Indicators for biodiversity in agricultural landscapes: A pan-European study. J Appl Ecol 45:141–150. doi: 10.1111/j.1365-2664.2007.01393.x

Birrer S, Spiess M, Herzog F, et al (2007) The Swiss agri-environment scheme promotes farmland birds: But only moderately. J Ornithol. doi: 10.1007/s10336-007-0237-y

Boyd J, Banzhaf S (2007) What are ecosystem services? The need for standardized environmental accounting units. Ecol Econ 63:616–626. doi: 10.1016/j.ecolecon.2007.01.002

Braat LC, de Groot R (2012) The ecosystem services agenda:bridging the worlds of natural science and economics, conservation and development, and public and private policy. Ecosyst Serv 1:4–15. doi: http://dx.doi.org/10.1016/j.ecoser.2012.07.011

Brändli U-B (2010) Schweizerisches Landesforstinventar. Ergebnisse der dritten Erhebung 2004–2006.itle.

Bundesamt für Umwelt; Bundesamt für Statistik SF (2015a) Holzernte nach Kantonen - 2004-2014.

Bundesamt für Umwelt; Bundesamt für Statistik SF (2015b) Waldfläche nach Kantonen - 2004-2014.

Bundesamt für Umwelt BAFU (2013) Nutzungsintensität der landwirtschaftlichen Fläche.

Burkhard B, Crossman N, Nedkov S, et al (2013) Mapping and modelling ecosystem services for science, policy and practice. Ecosyst. Serv. 4:1–3.

Burkhard B, Kroll F, Nedkov S, Müller F (2012) Mapping ecosystem service supply, demand and budgets. Ecol Indic 21:17–29. doi: 10.1016/j.ecolind.2011.06.019

Cardinael R, Chevallier T, Barthès BG, et al (2015) Impact of alley cropping agroforestry on stocks, forms and spatial distribution of soil organic carbon - A case study in a Mediterranean context. Geoderma 259–260:288–299. doi: 10.1016/j.geoderma.2015.06.015

Clec'h S Le, Oszwald J, Decaens T, et al (2016) Mapping multiple ecosystem services indicators: Toward an objective-oriented approach. Ecol Indic 69:508–521. doi: 10.1016/j.ecolind.2016.05.021

Coleman K, D.S. Jenkinson (1996) RothC-26.3 - A model for the turnover of carbon in soil. In: Powlson DS, Smith P, Smith JU (eds) Evaluation of Soil Organic Matter Models Using Existing Long-Term Datasets. Springer-Verlag, Heidelberg, pp 237–246

Conrad O, Bechtel B, Bock M, et al (2015) System for Automated Geoscientific Analyses (SAGA) v. 2.1.4. Geosci Model Dev 8:1991–2007. doi: 10.5194/gmd-8-1991-2015

den Herder M, Moreno G, Mosquera-Losada RM, et al (2017) Current extent and stratification of agroforestry in the European Union. Agric Ecosyst Environ 241:121–132. doi: 10.1016/j.agee.2017.03.005

Englund O, Berndes G, Cederberg C (2017) How to analyse ecosystem services in landscapes???A systematic review. Ecol. Indic. 73:492–504.

Fader M, Von Bloh W, Shi S, et al (2015) Modelling Mediterranean agro-ecosystems by including agricultural trees in the LPJmL model. Geosci Model Dev 8:3545–3561. doi: 10.5194/gmd-8-3545-2015

Graves A, Burgess PJ, Palma J, et al (2010) Implementation and calibration of the parameter-sparse Yield-SAFE model to predict production and land equivalent ratio in mixed tree and crop systems under two contrasting production situations in Europe. Ecol Modell 221:1744–1756. doi: 10.1016/j.ecolmodel.2010.03.008

Haines-Young R, Potschin M (2013) Common International Classification of Ecosystem Services (CICES): Consultation on Version 4, August-December 2012.

Herrmann F, Jahnke C, Jenn F, et al (2009) Groundwater recharge rates for regional groundwater modelling: A case study using GROWA in the Lower Rhine lignite mining area, Germany. Hydrogeol J 17:2049–2060. doi: 10.1007/s10040-009-0493-4

Herzog F, Jeanneret P, Ammari Y, et al (2013) Measuring farmland biodiversity. Solutions 4:52–58.

Herzog F, Lüscher G, Arndorfer M, et al (2017) European farm scale habitat descriptors for the evaluation of biodiversity. Ecol Indic 77:205–217. doi: 10.1016/j.ecolind.2017.01.010

Hiederer R (2013a) Mapping Soil Properties for Europe - Spatial Representation of Soil Database Attributes.

Hiederer R (2013b) Mapping Soil Typologies-Spatial Decision Support Applied to the European Soil Database.

Hürdler J, Prasuhn V, Spiess E (2015) Abschätzung diffuser Stickstoff- und Phosphoreinträge in die Gewässer der Schweiz.

Jose S (2009) Agroforestry for ecosystem services and environmental benefits: An overview. Agrofor Syst 76:1–10. doi: 10.1007/s10457-009-9229-7

Jose S, Pallardy SG (2004) Interspecific interactions in temperate agroforestry. Agrofor Syst 61:237–255.

Keel SG, Leifeld J, Taghizadeh-toosi A, Oleson JE (2016) Large uncertainty in soil carbon modelling related to carbon input calculation method. In: Geophysical Research Abstracts - EGU General Assembly 2016.

Kennedy CM, Lonsdorf E, Neel MC, et al (2013) A global quantitative synthesis of local and landscape effects on wild bee pollinators in agroecosystems. Ecol Lett 16:584–599. doi: 10.1111/ele.12082

Lal R (2004) Soil carbon sequestration to mitigate climate change. Geoderma 123:1–22. doi: 10.1016/j.geoderma.2004.01.032

Liski J, Palosuo T, Peltoniemi M, Sievänen R (2005) Carbon and decomposition model Yasso for forest soils. Ecol Modell 189:168–182. doi: 10.1016/j.ecolmodel.2005.03.005

Lonsdorf E, Kremen C, Ricketts T, et al (2009) Modelling pollination services across agricultural landscapes. Ann Bot 103:1589–1600. doi: 10.1093/aob/mcp069

Lonsdorf E, Ricketts T, Kremen C, et al (2011) Crop pollination services. Nat capital Theory Pract Mapp Ecosyst Serv Oxford Univ Press Oxford 168–187.

López-Díaz ML, Rolo V, Moreno G (2011) Trees' role in nitrogen leaching after organic, mineral fertilization: a greenhouse experiment. J Environ Qual 40:853–9. doi: 10.2134/jeq2010.0165

Lorenz K, Lal R (2014) Soil organic carbon sequestration in agroforestry systems. A review. Agron Sustain Dev 34:443–454. doi: 10.1007/s13593-014-0212-y

Mace GM, Norris K, Fitter AH (2012) Biodiversity and ecosystem services: A multilayered relationship. Trends Ecol Evol 27:19–25. doi: 10.1016/j.tree.2011.08.006

Maes J, Liquete C, Teller A, et al (2016) An indicator framework for assessing ecosystem services in support of the EU Biodiversity Strategy to 2020. Ecosyst Serv 17:14–23. doi: 10.1016/j.ecoser.2015.10.023

Makó A, Kocsis M, Barna G, Tóth G (2017) Mapping the storing and filtering capacity of European soils.

Masera OR, Garza-Caligaris JF, Kanninen M, et al (2003) Modeling carbon sequestration in afforestation, agroforestry and forest management projects: The CO2FIX V.2 approach. Ecol Modell 164:177–199. doi: 10.1016/S0304-3800(02)00419-2

MEA (2003) Ecosystems and human well-being. Washington, DC: Island Press

Moreno G, Gonzalez-Bornay G, Pulido F, et al (2016) Exploring the causes of high biodiversity of Iberian dehesas: the importance of wood pastures and marginal habitats. Agrofor Syst 90:87–105. doi: 10.1007/s10457-015-9817-7

Nair PKR (2012) Carbon sequestration studies in agroforestry systems: A reality-check. Agrofor Syst 86:243–253. doi: 10.1007/s10457-011-9434-z

Nair VD, Nair PKR, Kalmbacher RS, Ezenwa I V (2007) Reducing nutrient loss from farms through silvopastoral practices in coarse-textured soils of Florida, (USA). Ecol Eng 29:192–199. doi: http://dx.doi.org/10.1016/j.ecoleng.2006.07.003

Olaya V (2004) Module Slope Length.

Palma J, Graves A, Crous-Duran J, et al (2017) EcoYield-SAFE model: Keeping a parameter-sparse concept in agroforestry modelling while integrating new ecosystem processes and dynamics. submittec

Palma JHN (2017) Clipick – climate change web picker. A tool bridging daily climate needs in process based modelling in forestry and agriculture. For Syst. doi: 10.5424/fs/2017261-10251

Panagos P, Ballabio C, Borrelli P, Meusburger K (2016) Spatio-temporal analysis of rainfall erosivity and erosivity density in Greece. Catena 137:161–172. doi: 10.1016/j.catena.2015.09.015

Panagos P, Borrelli P, Meusburger K, et al (2015a) Estimating the soil erosion cover-management factor at the European scale. Land use policy 48:38–50. doi: 10.1016/j.landusepol.2015.05.021

Panagos P, Borrelli P, Meusburger K, et al (2015b) Modelling the effect of support practices (P-factor) on the reduction of soil erosion by water at European scale. Environ Sci Policy 51:23–34. doi: 10.1016/j.envsci.2015.03.012

Panagos P, Borrelli P, Poesen J, et al (2015c) The new assessment of soil loss by water erosion in Europe. Environ Sci Policy 54:438–447. doi: 10.1016/j.envsci.2015.08.012

Panagos P, Meusburger K, Ballabio C, et al (2014) Soil erodibility in Europe: A high-resolution dataset based on LUCAS. Sci Total Environ 479–480:189–200. doi: 10.1016/j.scitotenv.2014.02.010

Panagos P, Van Liedekerke M, Jones A, Montanarella L (2012) European Soil Data Centre: Response to European policy support and public data requirements. Land use policy 29:329–338. doi: 10.1016/j.landusepol.2011.07.003

Pasquet M, Genries S, Lemoing M, et al (2014) Etude de l'offre alimentaire pour les pollinisateurs sur 33 exploitations agricoles de France metropolitaine. In: AFPP – DIXIÈME CONFERENCE INTERNATIONALE SUR LES RAVAGEURS EN AGRICULTURE MONTPELLIER – 22 ET 23 OCTOBRE 2014. p 9

Pointereau P, Paracchini ML, Terres J (2007) Identification of High Nature Value farmland in France through statistical information and farm practice surveys.

Powlson DS, Whitmore AP, Goulding KWT (2011) Soil carbon sequestration to mitigate climate change: A critical re-examination to identify the true and the false. Eur J Soil Sci 62:42–55. doi: 10.1111/j.1365-2389.2010.01342.x

Prada M, Bravo F, Berdasco L, et al (2016) Carbon sequestration for different management alternatives in sweet chestnut coppice in northern Spain. J Clean Prod 135:1161–1169. doi: 10.1016/j.jclepro.2016.07.041

Prasuhn V, Liniger H, Gisler S, et al (2013) A high-resolution soil erosion risk map of Switzerland as strategic policy support system. Land use policy 32:281–291. doi: 10.1016/j.landusepol.2012.11.006

Pumariño L, Sileshi GW, Gripenberg S, et al (2015) Effects of agroforestry on pest, disease and weed control: A meta-analysis. Basic Appl Ecol 16:573–582. doi: 10.1016/j.baae.2015.08.006

Raudsepp-Hearne C, Peterson GD, Bennett EM (2010) Ecosystem service bundles for analyzing tradeoffs in diverse landscapes. Proc Natl Acad Sci U S A 107:5242–7. doi: 10.1073/pnas.0907284107

Renard K, Foster G, Weesies G, et al (1997) Predicting soil erosion by water: a guide to conservation planning with the Revised Universal Soil Loss Equation (RUSLE).

Rodríguez-Ortega T, Oteros-Rozas E, Ripoll-Bosch R, et al (2014) Applying the ecosystem services framework to pasture-based livestock farming systems in Europe. Animal 1–12. doi: 10.1017/S1751731114000421

Sánchez IA, McCollin D (2015) A comparison of microclimate and environmental modification produced by hedgerows and dehesa in the Mediterranean region: A study in the Guadarrama region, Spain. Landsc Urban Plan. doi: 10.1016/j.landurbplan.2015.07.002

Schindler M, Peters B (2011) Mason Bees Osmia Bicornis and Osmia Cornuta as Suitable Orchard Pollinators? Erwerbs-Obstbau 52:111–116. doi: 10.1007/s10341-010-0118-z

Schoumans OF, Chardon WJ, Bechmann ME, et al (2014) Mitigation options to reduce phosphorus losses from the agricultural sector and improve surface water quality: A review. Sci Total Environ 468–469:1255–1266. doi: 10.1016/j.scitotenv.2013.08.061

Schröter M, Remme RP, Sumarga E, et al (2014) Lessons learned for spatial modelling of ecosystem services in support of ecosystem accounting. Ecosyst Serv 13:64–69. doi: 10.1016/j.ecoser.2014.07.003

Schulp CJE, Lautenbach S, Verburg PH (2014) Quantifying and mapping ecosystem services: Demand and supply of pollination in the European Union. Ecol Indic 36:131–141. doi: 10.1016/j.ecolind.2013.07.014

Sereke F, Graves AR, Dux D, et al (2015) Innovative agroecosystem goods and services: key profitability drivers in Swiss agroforestry. Agron Sustain Dev 35:759–770. doi: 10.1007/s13593-014-0261-2

Somarriba E (1992) Revisiting the past: an essay on agroforestry definition. Springer

Sutton MA, Howard CM, Erisman JW, et al (2011) The European nitrogen assessment: sources, effects and policy perspectives. Cambridge University Press

swisstopo (2012) swissALTI3D.

Syswerda SP, Robertson GP (2014) Ecosystem services along a management gradient in Michigan (USA) cropping systems. Agric Ecosyst Environ 189:28–35. doi: 10.1016/j.agee.2014.03.006

Szerencsits E, Bailey D, Junquera V, et al (2016) Habitat mapping protocol.

Taghizadeh-Toosi A, Christensen BT, Hutchings NJ, et al (2014) C-TOOL: A simple model for simulating whole-profile carbon storage in temperate agricultural soils. Ecol Modell 292:11–25. doi: 10.1016/j.ecolmodel.2014.08.016

Tetzlaff B, Andjelov M, Kuhr P, et al (2015) Model-based assessment of groundwater recharge in Slovenia. Environ Earth Sci 74:6177–6192. doi: 10.1007/s12665-015-4639-5

Tieskens KF, Schulp CJE, Levers C, et al (2017) Characterizing European cultural landscapes: Accounting for structure, management intensity and value of agricultural and forest landscapes. Land use policy 62:29–39. doi: http://dx.doi.org/10.1016/j.landusepol.2016.12.001

Torralba M, Fagerholm N, Burgess PJ, et al (2016) Do European agroforestry systems enhance biodiversity and ecosystem services? A meta-analysis. Agric Ecosyst Environ 230:150–161. doi: 10.1016/j.agee.2016.06.002

Tscharntke T, Klein AM, Kruess A, et al (2005) Landscape perspectives on agricultural intensification and biodiversity - Ecosystem service management. Ecol Lett 8:857–874. doi: 10.1111/j.1461-0248.2005.00782.x

Tsonkova P, Quinkenstein A, Böhm C, et al (2014) Ecosystem services assessment tool for agroforestry (ESAT-A): An approach to assess selected ecosystem services provided by alley cropping systems. Ecol Indic 45:285–299. doi: 10.1016/j.ecolind.2014.04.024

Udawatta RP, Kremer RJ, Adamson BW, Anderson SH (2008) Variations in soil aggregate stability and enzyme activities in a temperate agroforestry practice. Appl Soil Ecol 39:153–160. doi: 10.1016/j.apsoil.2007.12.002

UNFCCC (2007) Report of the conference of the parties on its thirteenth session, held in Bali from 3 to 15 December 2007. In: United Nations Framework Convention on Climate Change (UN FCCC).

van der Werf W, Keesman K, Burgess P, et al (2007) Yield-SAFE: A parameter-sparse, process-based dynamic model for predicting resource capture, growth, and production in agroforestry systems. Ecol Eng 29:419–433. doi: 10.1016/j.ecoleng.2006.09.017

van der Zanden EH, Levers C, Verburg PH, Kuemmerle T (2016) Representing composition, spatial structure and management intensity of European agricultural landscapes: A new typology. Landsc Urban Plan 150:36–49. doi: 10.1016/j.landurbplan.2016.02.005

Velthof GL, Lesschen JP, Webb J, et al (2014) The impact of the Nitrates Directive on nitrogen emissions from agriculture in the EU-27 during 2000-2008. Sci Total Environ 468–469:1225–1233. doi: 10.1016/j.scitotenv.2013.04.058

Verburg PH, van Asselen S, van der Zanden EH, Stehfest E (2013) The representation of landscapes in global scale assessments of environmental change. Landsc Ecol 28:1067–1080. doi: 10.1007/s10980-012-9745-0

Wezel A, Casagrande M, Celette F, et al (2014) Agroecological practices for sustainable agriculture. A review. Agron Sustain Dev 34:1–20. doi: 10.1007/s13593-013-0180-7

Windisch K (1895) Die Zusammensetzung des Kirschbranntweines. In: Die Zusammensetzung des Kirschbranntweines. Springer, pp 11–105

Wösten JHM, Lilly A, Nemes A, Le Bas C (1999) Development and use of a database of hydraulic properties of European soils. Geoderma 90:169–185. doi: 10.1016/S0016-7061(98)00132-3

Young A (1997) Agroforestry for soil management. CAB international

Zurbuchen A, Landert L, Klaiber J, et al (2010) Maximum foraging ranges in solitary bees: only few individuals have the capability to cover long foraging distances. Biol Conserv 143:669–676. doi: 10.1016/j.biocon.2009.12.003

Figures captions

Figure 1: Profile of the cherry orchard case study region, Switzerland (LU: livestock unit)

Figure 2: Habitat map (a) and AF trees (b) in the LTS plots [Tree types: small circles – young trees, medium circles – middle aged trees; large circles – old trees; Background Image: Swissimage © swisstopo (2015)]

Figure 3: Conceptual background of the model

Figure 4: Habitat maps (a), modelled biomass yield [t ha-1 yr-1] (b), nitrate leaching [kg N ha-1 yr-1] (c) and carbon sequestration [t C ha-1 yr-1] (d) of landscape test sites [LTS] grouped by land cover categories into agroforestry (AF) and non-agroforestry (NAF) sites

Figure 5: Pollination services in 100, 350 and 500 m range around nesting facilities in agroforestry (AF) and non-agroforestry (NAF) landscape test sites.

*Figure 6: Summary of the normalized indicators [-1,1] grouped into agroforestry (AF) and non-agroforestry (NAF) landscape test sites normalized to 1 for gains, and -1 for losses (N Loss and Soil Loss) [SIDI: Simpson's diversity index, SoSNH: share of semi-natural habitat, HD: Habitat Diversity; ***: p<0.0005 **: p<0.001, *: p< 0.01]*

Cherry Orchard, Switzerland	
Municipalities	Büren, Gempen, Hochwald, Lupsingen, Nuglar-St. Pantaleon, Seewen, Seltisberg
Area	49.89 km^2
Temperature Avg °C	7.7 °C
Precipitation	800 - 1000 mm
Elevation	430 - 670 m
Soil	fine
Land use	43 % Non-Agroforestry, 44 % Forestry, 5% Agroforestry, 8% others
Agriculture	1,972 ha farmland 83 farmers 1,522 LU (mostly cattle)
Livestock intensity	0.77 LU ha^{-1} farmland
Agroforestry system	80 cherry trees ha^{-1} + grassland
Products	Cherries for liquor, tinned food or direct consumption Grass as fodder for cattle (hay, silage or pasture) Timber

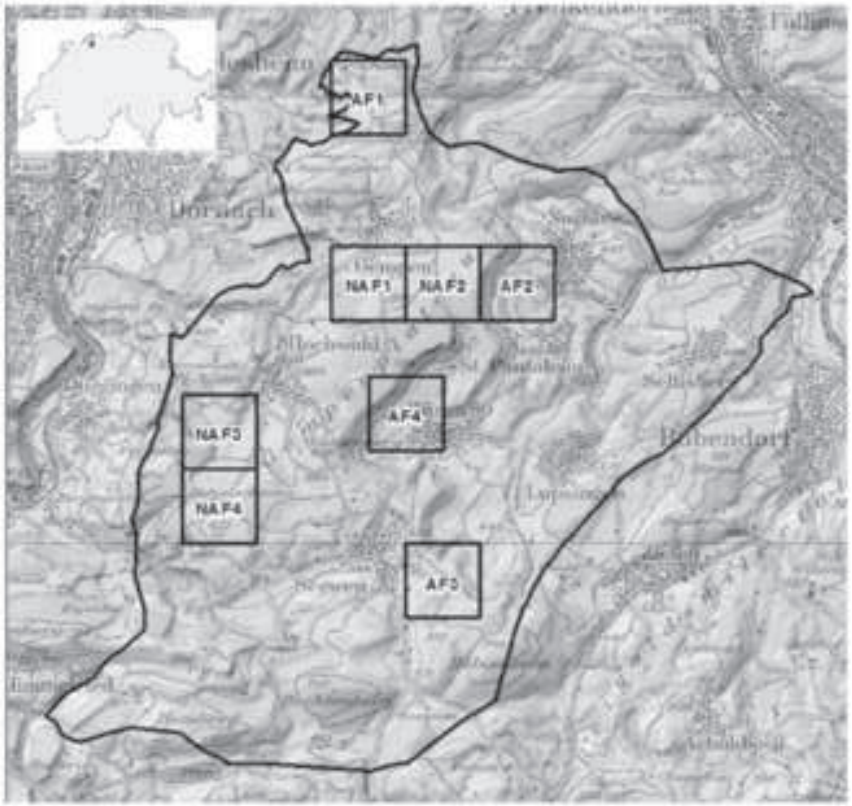


Fig_2

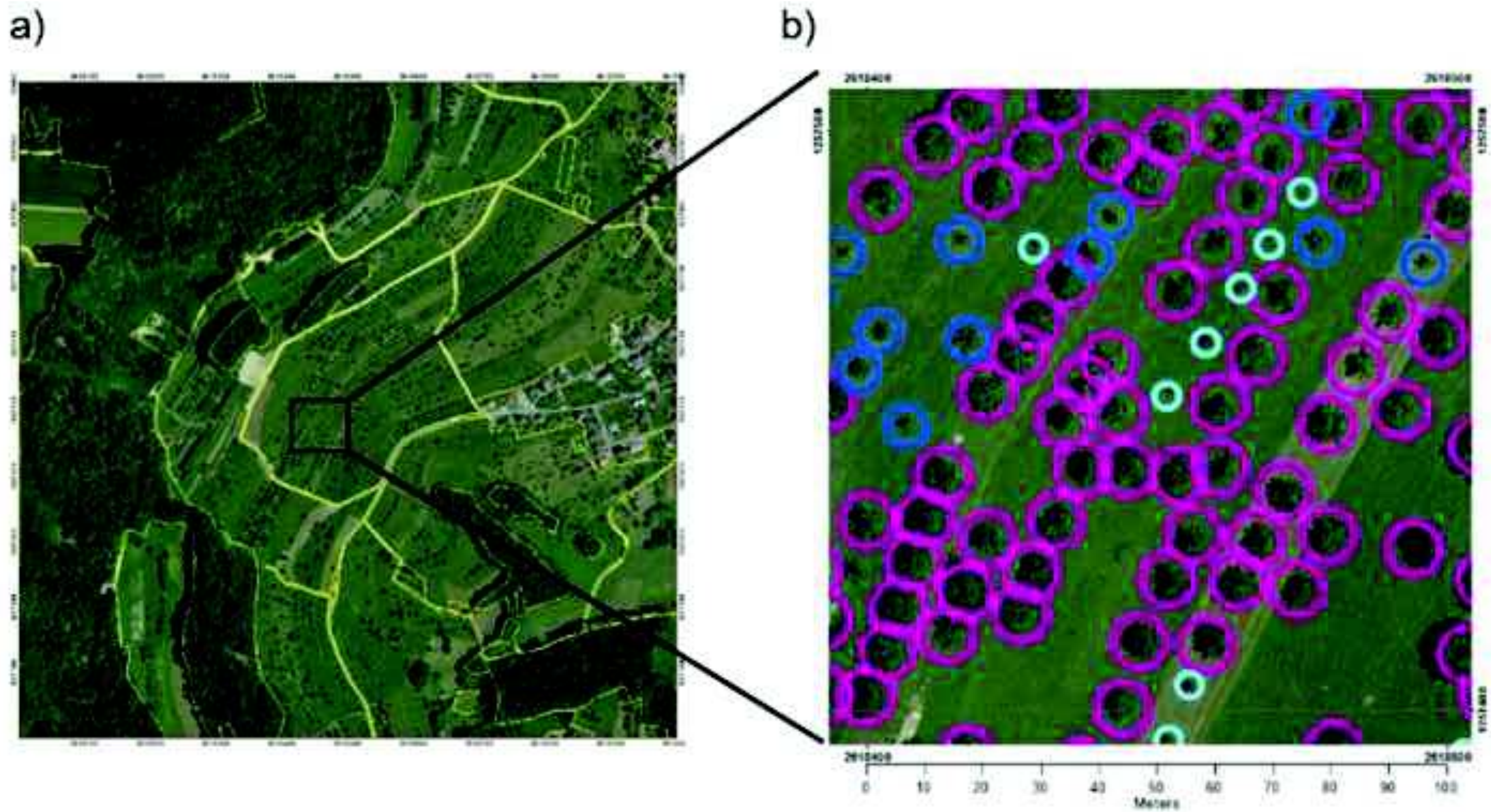
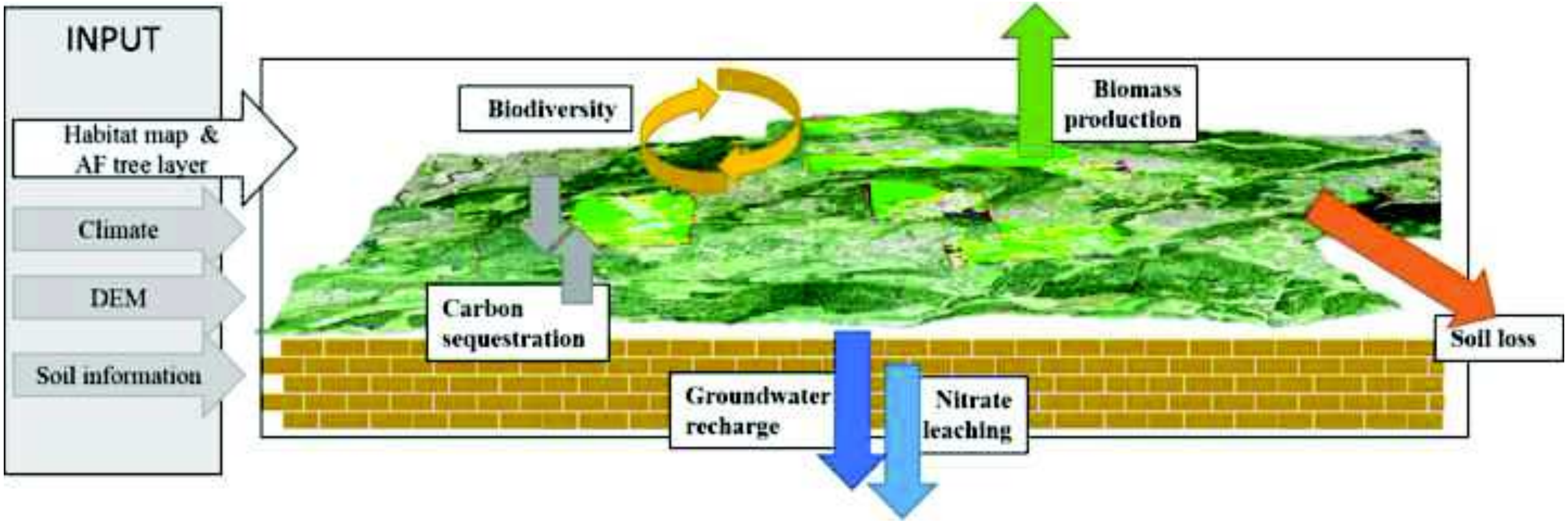


Fig_3


INPUT
Habitat map & AF tree layer
Climate
DEM
Soil information
Biodiversity
Biomass production
Carbon sequestration
Soil loss
Groundwater recharge
Nitrate leaching

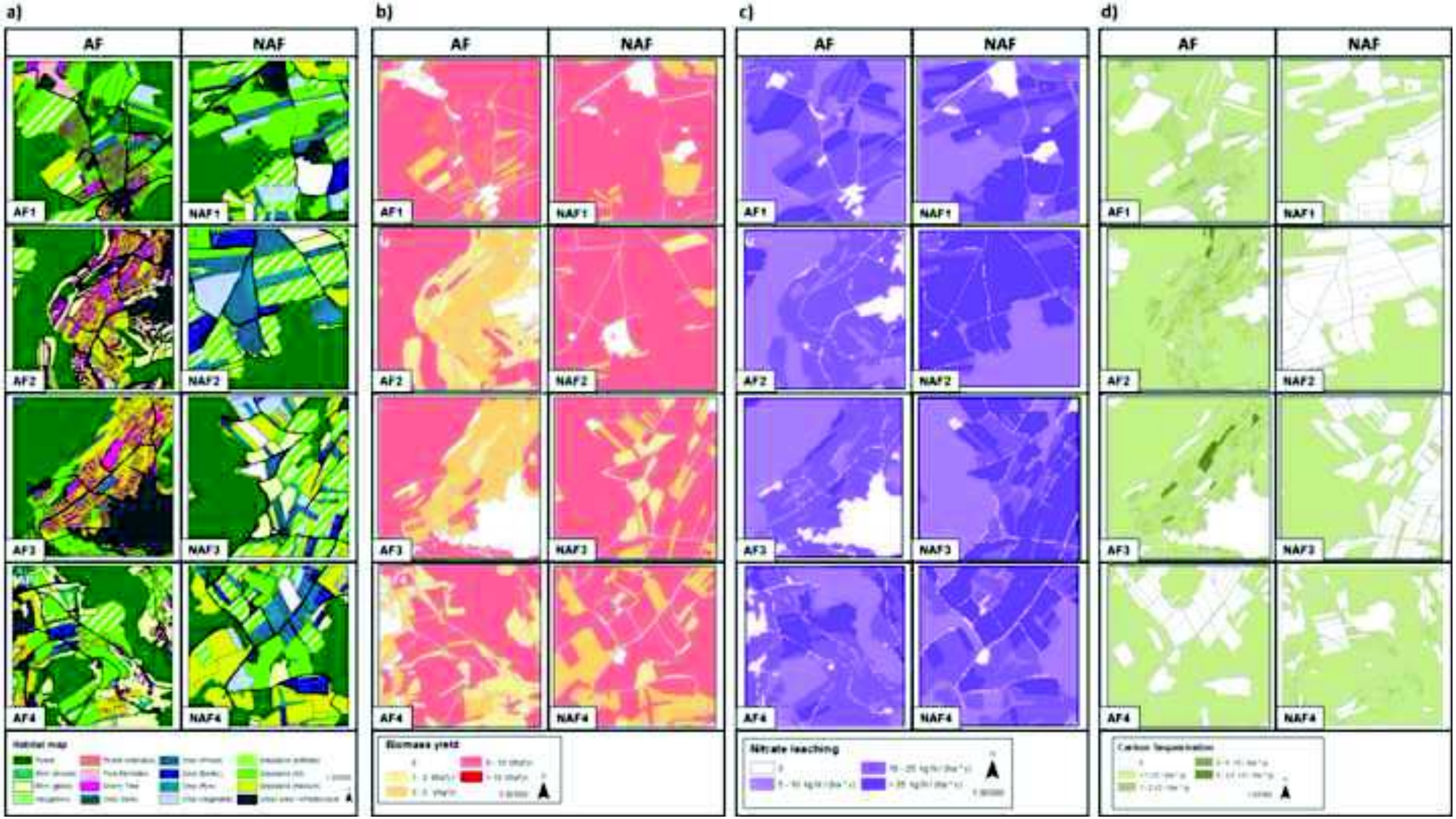
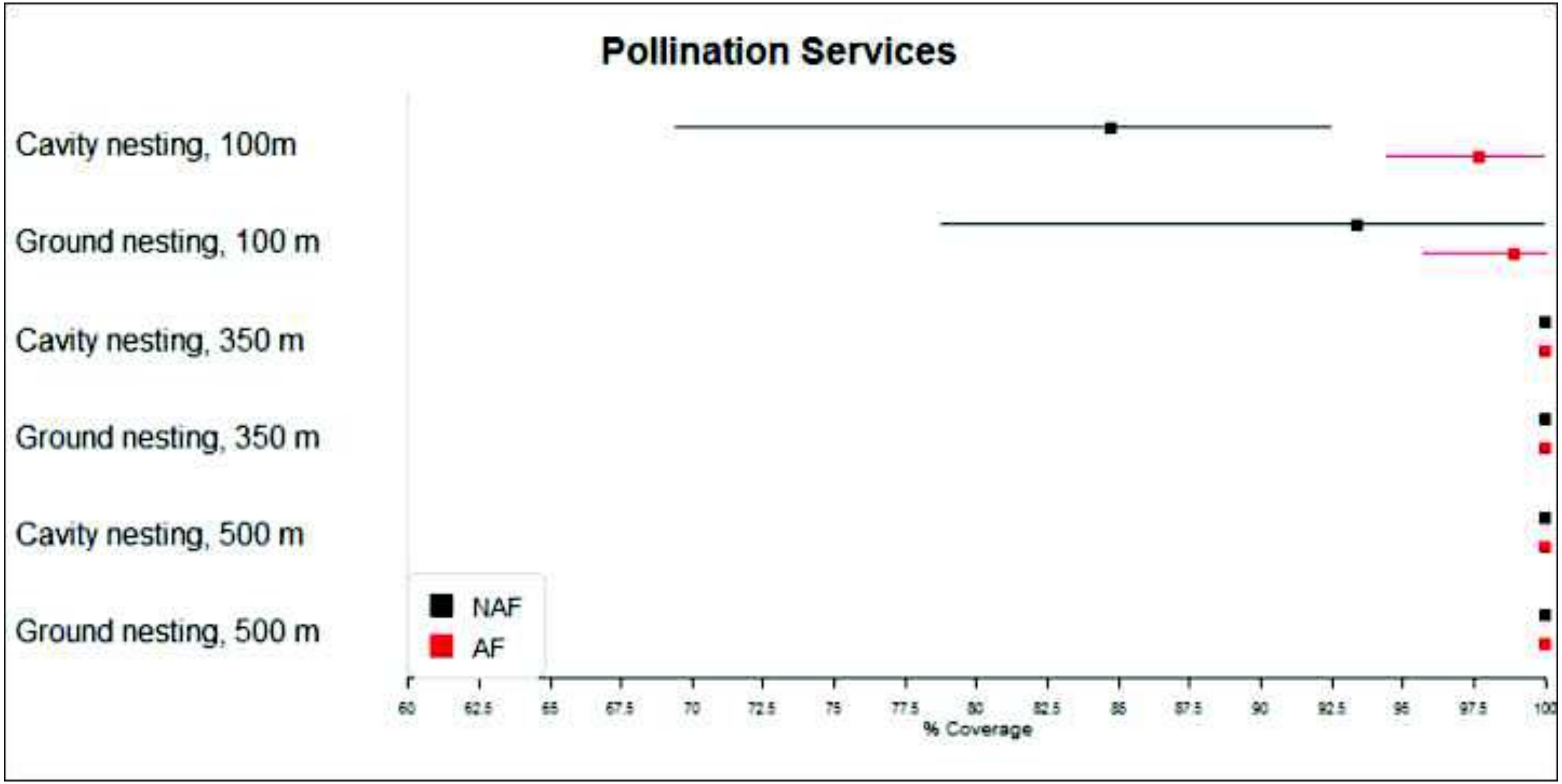

a)
b)
c)
d)
AF
NAF
AF1
NAF1
AF2
NAF2
AF3
NAF3
AF4
NAF4
Habitat map
Biomass yield
Nitrate leaching
Carbon Sequestration

Fig_5

Click here to download Figure Fig_5.tif



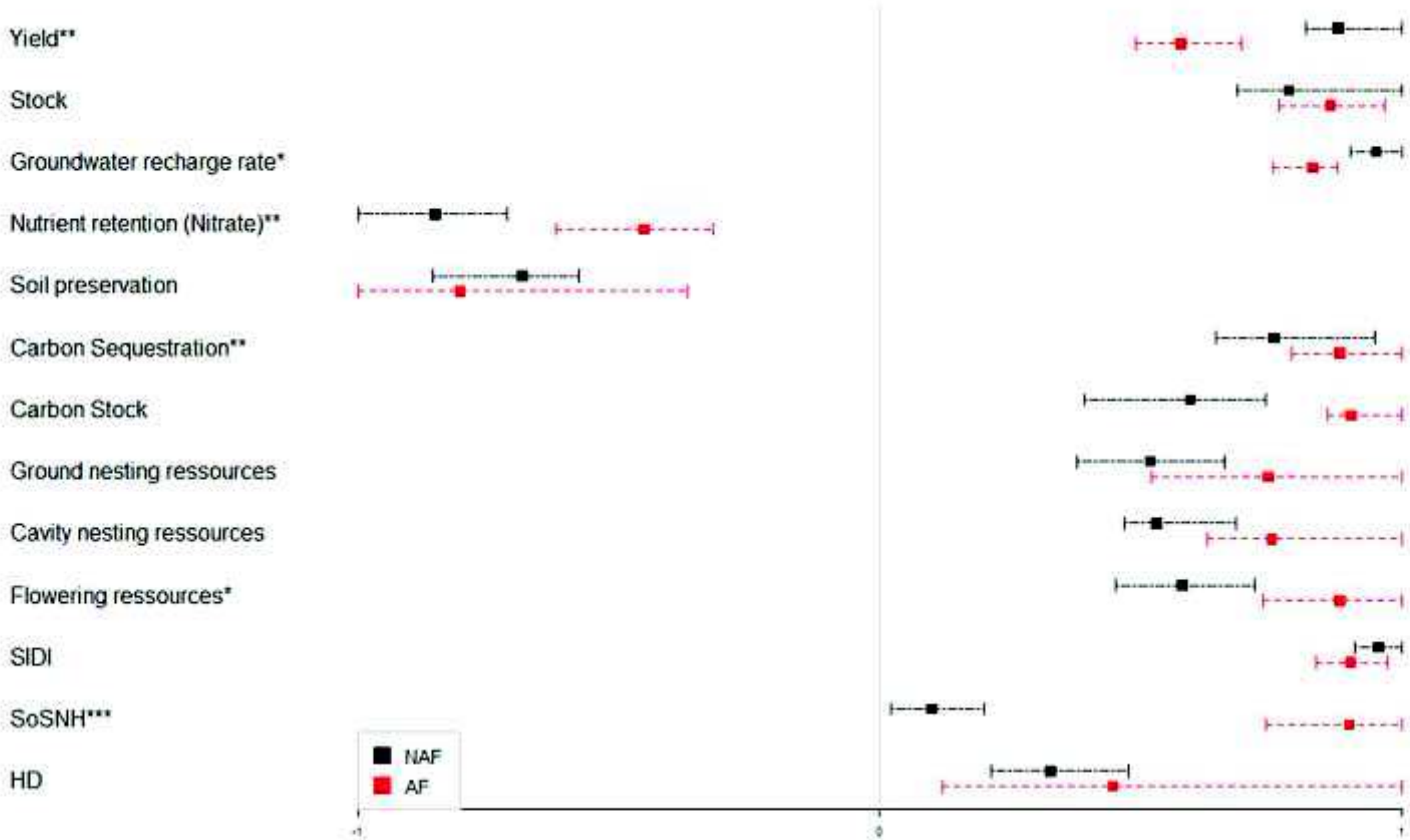

Yield**
Stock
Groundwater recharge rate*
Nutrient retention (Nitrate)**
Soil preservation
Carbon Sequestration**
Carbon Stock
Ground nesting ressources
Cavity nesting ressources
Flowering ressources*
SIDI
SoSNH***
HD
NAF
AF
-1
0
1

Table 1: List of provisioning and regulating ecosystem services (ES) according to CICES Classification (Haines-Young and Potschin 2013) *linked to indicators addressed by agroforestry literature and methodologies to assess these indicators [SIDI: Simpson's diversity index, SoSNH: share of semi-natural habitat, HD: Habitat Diversity]*

<table>
<tr><th colspan="2">CICES
Section - Division</th><th>ES indicators</th><th>Methods and models</th><th>References</th></tr>
<tr><td rowspan="4">Provisioning</td><td>Nutrition (Food / Feed)</td><td rowspan="3">Biomass production</td><td rowspan="3">EcoYield-SAFE</td><td rowspan="3">van der Werf et al. 2007; Palma et al. 2017</td></tr>
<tr><td>Material (Raw material, Genetic resources, Medicinal resources, Ornamental resources)</td></tr>
<tr><td>Energy</td></tr>
<tr><td>Water supply</td><td>Groundwater recharge</td><td>Water balance equation using CropWat2.0 & MODIFFUS 3.0</td><td>Allen et al. 1998; Hürdler et al. 2015</td></tr>
<tr><td rowspan="4">Regulating and Maintenance</td><td>Regulation of biophysical environment (Air purification, Waste treatment)</td><td>Nutrient retention</td><td>MODIFFUS 3.0</td><td>Hürdler et al. 2015</td></tr>
<tr><td>Flow regulation (Disturbance prevention, regulation of water flows, erosion prevention)</td><td>Soil preservation (Erosion)</td><td>RUSLE</td><td>Renard et al. 1997; Panagos et al. 2015</td></tr>
<tr><td>Regulation of physicochemical environment (Climate regulation, Maintaining soil fertility)</td><td>Carbon Sequestration</td><td>EcoYield-SAFE
Yasso07</td><td>Liski et al. 2005; Palma et al. 2017</td></tr>
<tr><td>Regulation of biotic environment (Gene pool protection, Lifecycle maintenance, Pollination, Biological control)</td><td>Biodiversity: Pollination, Habitat Richness</td><td>Pollination: Lonsdorf
Habitat Richness: SIDI, SoSNH, HD</td><td>Bailey et al. 2007; Billeter et al. 2008; Lonsdorf et al. 2009</td></tr>
</table>

Table 2: Average biomass production and biophysical yield in tons dry matter per land cover class [Conversion: 40 t Potatoes ha^{-1} (19 % DM) (BAFU 2013); 80 t sugar beet ha^{-1}(15 % DM)(BAFU 2013), 2 m^{3} / t atro wood (Riegger 2008)]

<table>
<tr><th>Category</th><th>Crop</th><th>Stock
(t DM ha^{-1})</th><th>Use
(t DM ha-1 yr-1)</th><th>Source</th></tr>
<tr><td rowspan="5">Annual crops</td><td>Cereals</td><td></td><td>10</td><td rowspan="5">BAFU 2013</td></tr>
<tr><td>Maize</td><td></td><td>17</td></tr>
<tr><td>Rape</td><td></td><td>3</td></tr>
<tr><td>Potatoes</td><td></td><td>7.6</td></tr>
<tr><td>Sugar beet</td><td></td><td>12</td></tr>
<tr><td>Grassland</td><td>Grassland</td><td>2</td><td>5</td><td>Baer 2009</td></tr>
<tr><td>Forest</td><td>Forest</td><td>175</td><td>3.25</td><td>Brändli 2010, BAFU; BfS 2015a; BAFU; BfS 2015b</td></tr>
<tr><td rowspan="2">Agroforestry</td><td>Trees
- young
-medium
-old</td><td>
5
20
30</td><td>
0 / 0.64
1 / 1.28
2 / 2.56</td><td rowspan="2">Results
EcoYield-SAFE</td></tr>
<tr><td>Grassland</td><td>2</td><td>3</td></tr>
</table>

Table 3: List of used datasets with title, provider and sources [ESDC: European Soil Data Centre, JRC: Joint research centre]

Dataset	**Title**	**Provider**	**References**	**Resolution**
Topography	Digital elevation model - SwissALTI3D	Swisstopo	(swisstopo 2012)	2 m
Climate	Gempen(CH) 1960-2000	CliPick	(Palma 2017)	
Soil	Total available water content (TAWC) for topsoil	ESDC, JRC	(Panagos et al. 2012; Hiederer 2013a, b)	1000 m
	Storing and filtering capacity		(Makó et al. 2017)	100 m
	Available water capacity		(Ballabio et al. 2016).	100 m
	Rainfall erosivity (R factor)		(Panagos et al. 2016)	1000 m
	Erodibility (K factor)		(Panagos et al. 2014)	500 m
	Groundcover (C factor)		(Panagos et al., 2015)	100 m
Water	CCM River and Catchment Database	ESDC, JRC	(Vogt et al. 2007)	100 m

Table 4: Landscape test sites – key figures (NH: Number of Habitats, F: forest, AF: agroforestry, A: agriculture, O: others)

LTS	**Name**	**Municipality**	**Class**	**NH**	**F**	**AF**	**A**	**O**
					%			
AF1	Schönmatt	Gempen	AF	125	35	37	27	2
NAF1	Ischlag	Gempen	NAF	119	28	0	64	7
AF2	Wacht	Gempen	AF	78	27	0	72	2
NAF2	Nuglar	Nuglar - St. Pantaleon	NAF	151	37	55	2	6
AF3	Blauenstein	Seewen	AF	165	28	39	28	6
NAF3	Rotenrain	Hochwald	NAF	142	39	0	58	3
AF4	Güggelhof	Seewen	AF	105	32	31	19	19
NAF4	Ziegelschüren	Hochwald	NAF	141	26	0	71	3

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